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Trend correlations for coastal eutrophication and its main local and whole-sea drivers – Application to the Baltic Sea

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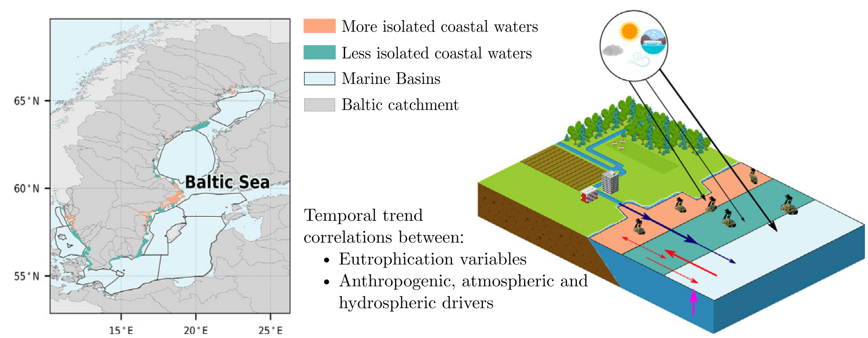
HIGHLIGHTS

- Eutrophication trends linked with anthropogenic, climatic and hydrospheric drivers
- Local water quality dynamics are distinguished between more and less isolated coasts.
- Coasts are melting pot of driver influences from local-land and open-sea conditions.
- Coastal eutrophication mitigation needs both local-catchment and whole-sea management.
- Results challenge unidirectional land source-to-sea paradigm for coastal water quality.

GRAPHICAL ABSTRACT



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ABSTRACT

Coastal eutrophication is a major environmental issue worldwide. In the Baltic Sea, eutrophication affects both the coastal waters and the open sea. Various policy frameworks aim to hinder its progress but eutrophication-relevant water quality variables, such as chlorophyll-a concentrations, still exhibit opposite temporal trends in various Baltic Sea marine and coastal waters. In this study, we investigate the temporal-trend linkages of measured water quality variables and their various anthropogenic, climatic and hydrospheric drivers over the period 1990–2020 with focus on the Swedish coastal waters and related marine basins in the Baltic Sea.

We find that it is necessary to distinguish more and less isolated coastal waters, based on their water exchanges with the open sea, to capture different coastal eutrophication dynamics. In less isolated coastal waters, eutrophication is primarily related to nitrogen concentrations, while it is more related to phosphorus concentrations in more isolated coastal waters. In the open sea, trends in eutrophication conditions correlate best with trends in climatic and hydrospheric drivers, like wind speed and water salinity, respectively. In the coastal waters, driver signals are more mixed, with considerable influences from anthropogenic land-based nutrient loads and sea-ice cover duration. Summer chlorophyll-a concentration in the open sea stands out as a main change driver of summer chlorophyll-a concentration in less isolated coastal waters. Overall, coastal waters are a melting pot of driver influences over various scales, from local land-based drivers to large-scale total catchment and open sea conditions. The latter in turn depend on long-term integration of pathway-dependent influences from the various coastal parts of the Baltic Sea and their land-based nutrient load drivers, combined with overarching climate

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conditions and internal feedback loops. As such, our results challenge any unidirectional local source-to-sea paradigm and emphasize a need for concerted local land-catchment and whole-sea measures for robust coastal eutrophication management.

1. Introduction

Coastal eutrophication affects more than 400 coastal waters worldwide, with only 13 of these waters in recovery (Selman et al., 2008). The effects can lead to ecological and economic damages through algae blooms, loss of biodiversity, oxygen depletion, fish deaths and dead zones (Altieri and Diaz, 2019). In the Baltic Sea, a semi-enclosed brackish sea situated in Northern Europe (shown in Fig. 1), several coastal waters are eutrophicated (Voss et al., 2011). Due to its large catchment area and limited water exchange with the North Sea, the open sea waters also suffer from eutrophication (Andersen et al., 2017), which further complicates the management of coastal waters (Vigouroux et al., 2019).

Eutrophication is generally driven by anthropogenic pressures, which increase nutrient inputs to the coastal waters (Nixon, 1995). This is also the case for the Baltic Sea, where most of the nitrogen and phosphorus loads are of anthropogenic nature, reflecting the importance of human activities in its catchment area (Gustafsson et al., 2012). About 90 million people inhabit its catchment area and around 26% of its area is used for agriculture and pasture (HELCOM, 2011). Activities in the catchment area have dramatically increased nutrient loads to the Baltic Sea in the past century, considered responsible for the onset of coastal and marine eutrophication (Voss et al., 2011; Meier et al., 2019). Several measures have been taken to reduce the nutrient loads. Improved wastewater treatment, for example, has resulted in a general decrease of both nitrogen and phosphorus loads since the 1990s (Reusch et al., 2018). However, further nutrient load reduction and water quality improvement is still needed. For this, various management frameworks have been implemented, both on regional scale through the Baltic Sea Action Plan (BSAP) of the Helsinki Commission (HELCOM), and on local inland and coastal scales through the Water Framework Directive (WFD).

The decrease in nutrient loads to the Baltic Sea since the 1990s (Reusch et al., 2018) has slowed down in the past decade, possibly due to increasing share of diffuse nutrient loads, representing cumulative consequences of anthropogenic activities that can be distant both in time and space (Le Moal et al., 2019). Moreover, despite achieved reductions in nutrient loads, chlorophyll-a concentrations during the summer months are still increasing in large parts of the Baltic Sea. Fig. 2 presents the evolution of summer chlorophyll-a concentrations (Summer Chl-a), a eutrophication indicator, for coastal and open sea waters with enough data availability during the period 1990–2020. Indeed, on the marine scale, Summer Chl-a has been increasing during the past three decades in most of the open sea waters (Fig. 2, panels

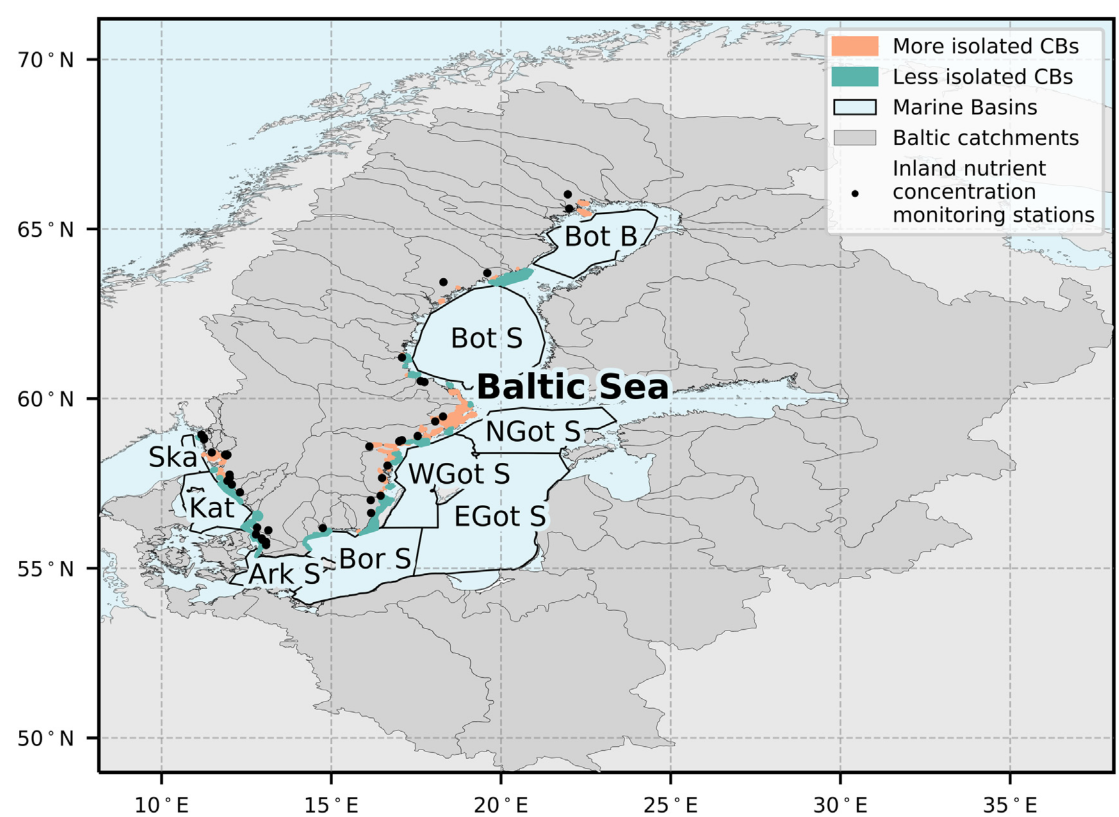
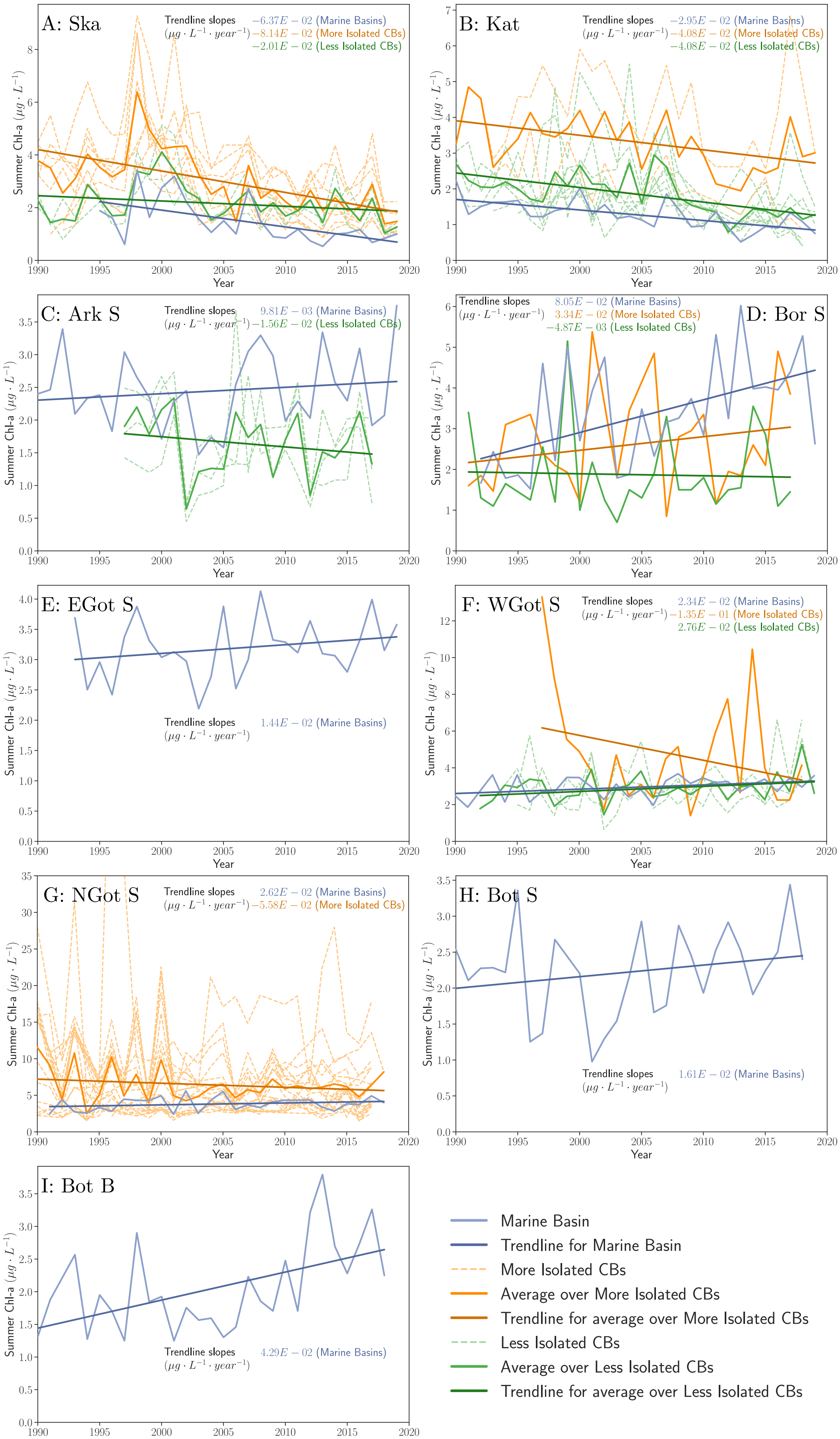


Fig. 1. Map of the Baltic Sea study area, situated in northern Europe. Coastal Basins (CBs) included in the study are divided into more isolated basins (in orange) and less isolated basins (in green). The classification is explained in Section 2.1. Baltic Sea Marine Basins (in light blue circled by black lines) 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 black dots show the locations of the monitoring stations that provide inland nutrient concentrations. The thin dark grey lines confine local catchment areas (in grey) that together build up and constitute the whole Baltic Sea catchment. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)


A: Ska
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$-6.37E-02$ (Marine Basins)
$-8.14E-02$ (More Isolated CBs)
$-2.01E-02$ (Less Isolated CBs)
B: Kat
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$-2.95E-02$ (Marine Basins)
$-4.08E-02$ (More Isolated CBs)
$-4.08E-02$ (Less Isolated CBs)
C: Ark S
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$9.81E-03$ (Marine Basins)
$-1.56E-02$ (Less Isolated CBs)
D: Bor S
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$8.05E-02$ (Marine Basins)
$3.34E-02$ (More Isolated CBs)
$-4.87E-03$ (Less Isolated CBs)
E: EGot S
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$1.44E-02$ (Marine Basins)
F: WGot S
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$2.34E-02$ (Marine Basins)
$-1.35E-01$ (More Isolated CBs)
$2.76E-02$ (Less Isolated CBs)
G: NGot S
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$2.62E-02$ (Marine Basins)
$-5.58E-02$ (More Isolated CBs)
H: Bot S
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$1.61E-02$ (Marine Basins)
I: Bot B
Trendline slopes ($\mu g \cdot L^{-1} \cdot year^{-1}$)
$4.29E-02$ (Marine Basins)
Summer Chl-a ($\mu g \cdot L^{-1}$)
Year
Marine Basin
Trendline for Marine Basin
More Isolated CBs
Average over More Isolated CBs
Trendline for average over More Isolated CBs
Less Isolated CBs
Average over Less Isolated CBs
Trendline for average over Less Isolated CBs

C–I). On the local coastal scale, trends in Summer Chl-a are more diverse, with both increasing and decreasing trends seen in even nearby situated coastal waters, for example in the coastal waters of the Northern Gotland Sea (Fig. 2, panel G). Thereby, other drivers than just nutrient loads may play important roles for the recent development of coastal eutrophication in the Baltic Sea. This has strong implications for management of coastal eutrophication, as implementation of effective management measures requires knowledge about the key drivers and their coastal effects.

Other anthropogenic drivers than just nutrient inputs can also affect eutrophication. For example, water cycling in the Baltic catchment area has been modified by flow regulation, hydropower and agricultural developments on land (Destouni et al., 2013), altering freshwater discharges and the nutrient loads they carry to the coast, as well as coastal hydrodynamics. In addition to land-based anthropogenic pressures, coastal and marine eutrophication conditions are also affected by atmospheric climate drivers (Meier et al., 2012; Savage et al., 2010). Climate-driven changes in precipitation and its partitioning between runoff and evapotranspiration also alter the freshwater discharge and the nutrient loads they carry to the coastal and open sea waters, in addition to such alterations driven by developments on land mentioned above. Climate drivers also influence the coastal and marine eutrophication conditions both by direct biogeochemical effects (Vigouroux et al., 2020) and by indirect effects on underlying hydrodynamics (Lips et al., 2017; Chen et al., 2019). As such, Vigouroux et al. (2020) found that high air-to-ocean net heat flux conditions complicate the effectiveness of management measures taken to reduce nutrient concentrations and eutrophication in the coastal case example of the Baltic Himmerfjärden bay, even under relatively low nutrient load conditions. Moreover, prolonged sea-ice cover can create nutrient-rich river plumes under ice, by reducing wind mixing, and lead to increased spring bloom biomass production in coastal areas (Kari et al., 2018).

Marine eutrophication depends on nutrient concentrations in the sea, physical water properties (e.g. water temperature) and internal processes governing the recycling of the nutrients, such as internal phosphorus loading under hypoxia (Meier et al., 2012). The Baltic Sea has a residence time of more than 30 years (Stigebrandt and Gustafsson, 2003), thereby changes in its eutrophication conditions depend on long term integration of both anthropogenically and hydroclimatically driven changes in catchment conditions, on direct climatic changes over coast and sea, and on the dynamics of coastal and sea responses to such changes. For example, salinity of coastal and marine surface water affects the stratification and thereby the growth of algae, and depends on both runoff from land and sea water exchange with the North Sea, with the latter also depending on variations in atmospheric pressure (Chen et al., 2019). Open coastal zones are affected by interactions between the open sea and the coastal waters as well as by freshwater and nutrient discharges from land (Almroth-Rosell et al., 2016). Moreover, marine eutrophication also influences coastal eutrophication, even in relatively isolated bays of the Baltic Sea (Vigouroux et al., 2020). Therefore, coastal eutrophication depends on a combination of variations and changes indirectly on land, and directly in the coastal areas as well as in the open sea.

Understanding the responses and feedbacks of coastal eutrophication to different parts of various driver combinations is necessary for developing robust and successful management strategies. The Swedish coastal waters of the Baltic Sea have experienced both increasing and decreasing eutrophication trends (Fig. 2). While modelling studies have examined the roles of various drivers in past and potential future evolution of eutrophication conditions in the Baltic Sea (e.g. using scenario simulations (Meier et al., 2012, 2019)), data-driven studies have mostly focused on drivers in the hydrosphere itself, i.e., within the coastal and marine waters, such as coastal hypoxia (Caballero-Alfonso et al., 2015) and coastal eutrophication in the Gulf of Bothnia (Lundberg et al., 2009), and in the anthroposphere and hydrosphere in the Gulf of Finland (Raateoja and Kauppila, 2019). No studies have systematically investigated observation based, data-given, interlinkages of eutrophication with combined anthropospheric, atmospheric, and hydrospheric drivers and variables. In this study, we aim to do this, with focus on data-given relationships emerging between dynamics of coastal Baltic eutrophication and its drivers on land and in the coastal and open sea parts of the coupled catchment-coast-sea system. With this aim, our research questions are: i) To what extent do temporal trends in various drivers explain observed trends in coastal eutrophication? ii) To what extent do identified driver-response relationships depend on influence locations and scales? To ensure consistency of data availability for different investigated coastal zones, we focus our study on the Swedish coast of the Baltic Sea, which encompasses a large variety of coasts, from the Bothnian Bay to Kattegat, spanning the whole hydroclimatic gradient experienced by the Baltic Sea system, for the recent period 1990–2020. The investigated drivers span the anthroposphere, the atmosphere, and the hydrosphere, and the local land catchment and coast scales along with the large open sea scale of the Baltic Sea and its whole catchment area.

2. Methods

2.1. Definition and classification of Coastal and Marine Basins

In this study, we have investigated dynamics and potential drivers of coastal eutrophication during medium to long-term periods (10 to 30 years). We focus on the Swedish coastal waters, which extend from the westernmost to the northernmost waters of the Baltic Sea varying from archipelagoes to open bays. The Swedish coastal waters have been partitioned into Coastal Basins (CBs). This partitioning has been made according to the basins used in the Swedish implementation of the WFD (Naturvårdsverket, 2006). The open Baltic Sea has been partitioned into Marine Basins (MBs) that are used both in the analysis and as drivers of coastal conditions and follow the spatial definition from HELCOM (HELCOM, 2017). For the analysis of coastal water quality variables and drivers, the CBs have been further classified into more isolated CBs and less isolated CBs, following the existing classification defined by the Swedish Environmental Protection Agency (2000). The Swedish Environmental Protection Agency (2000) have classified the CBs into three water exchange classes (water exchange rate of 0–9 days, 10–39 days, and ≥40 days), based on their morphologies (representing their link to the next CB or to the open sea) and basin sequences (representing the number of CBs between a given CB and the open sea), and has been verified by modelling the exchange rates for 139 of the 560 Swedish CBs. The water exchange class with a rate of 0–9 days corresponds to the less isolated CBs in this study (open coasts and open bays directly linked to the open sea or to an open coast that is linked to the open sea). Together, the water exchange classes with rates of 10–39 days and ≥40 days corresponds to the more isolated CBs (CBs in the basin sequence 1 with a narrow straight or a shelf in the seaward margin, or in an archipelago, and in higher basin sequences). CBs that have been changed or added CBs between the initial classification by the Swedish Environmental Protection Agency (2000) and their current

Fig. 2. Summer chlorophyll-a concentrations (Chl-a) based on yearly averaged data for the Marine Basins (in blue) and their associated Coastal Basins (CBs) from 1990 to 2020 that have at least a 20-year long time series for Summer Chl-a. The data sources and aggregation are presented in Sections 2.2.1 and 2.3.1, respectively. More isolated CBs are shown in orange and less isolated CBs in green, and their average in the respective color in bold. Panel A: Skagerrak, panel B: Kattegat, panel C: Arkona Sea, panel D: Bornholm Sea, panel E: Eastern Gotland Sea, panel F: Western Gotland Sea, panel G: Northern Gotland Sea, panel H: Bothnian Sea, and panel I: Bothnian Bay. The sole more isolated CB (orange) in the Western Gotland Sea (shown in Supplementary materials Fig. S1) represents an outlier point in the analysis of relationships between water quality variables, with strong variations in its Summer Chl-a. Note the varying vertical scales between the panels. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

definition (Naturvårdsverket, 2006) have been reclassified using the same criteria. Fig. 1 shows the CBs included in the analysis (green for less isolated and orange for more isolated) and their associated MBs (light blue). CBs and MBs were selected based on their water quality data availability, as described in the Section 2.3.1 and the number of CBs associated with each MB is presented in Supplementary materials (SM) Table S1. Morphological properties of the CBs are given in SM Table S2 per CB class and associated MB. Both the drivers and the water quality variables were aggregated for these selected basin classes (All CBs, more isolated CBs, less isolated CBs, MBs), as described in the Section 2.3.1.

2.2. Data sources

This section presents and justifies the choice of water quality and driver data used in the analysis of the relationships between temporal trends in eutrophication-relevant water quality variables, and between temporal trends in water quality variables and drivers. The study focuses on the recent period 1990–2020, which has experienced the greatest range of hydroclimatic changes in terms of water temperature and sea-ice cover (The BACC II Author Team, 2015), as well as decreasing nutrient loads to the whole Baltic Sea. Table 1 summarizes the datasets used in the study as well as their sources. Data processing is described in the Section 2.3.

2.2.1. Water quality variables

In this study, we mainly focus on eutrophication-relevant water quality indicators, defined by and related to the Swedish implementation of the WFD (Havs- och vattenmyndigheten, 2019), which also provide a consistent framework for data access and analysis. For the variable analysis, summer has been defined as the months of June–August or July–August and winter as the months of November/December–February/March, depending on the CB or MB location (given per MB in SM Table S3), following the Swedish implementation of the WFD (Havs- och vattenmyndigheten, 2019). Definitions of summer and winter months depend on the water quality dynamics in the different coastal and marine locations. Winter months correspond to the period following the growth season and preceding the spring bloom (when dissolved nutrient levels are highest), and summer months cover the growth season after the spring bloom (HELCOM, 2017). For all the variables, surface measurements have been used, corresponding to measurements within the 0–10 m depth zone.

Hydrospheric variables that are part of the ecological status definition for coastal waters of the Swedish implementation in the WFD (Havs- och vattenmyndigheten, 2019) are defined below. Surface concentration of chlorophyll-a (Chl-a) during summer (Summer Chl-a) is a commonly used proxy for eutrophication, indicating summer algae and cyanobacteria blooms. Thus, Summer Chl-a was chosen as the main variable for this study. Nutrient concentrations were also investigated to

Table 1
Dataset information of variables and drivers considered in the present study. Data processing is detailed in Supplementary materials S1.2 Coastal Basin (CB); Marine Basin (MB).

Type	Name	Short description	Period with data	Data time resolution	Data source
Hydrosphere variables within CBs and MBs	Water quality variables concentrations in CBs and MBs (DIN, DIP, TN, TP, Chl-a)	Concentration of water quality variable in the surface waters of the CBs and MBs	Varying (longest 1990–2019)	Yearly	SMHI SHARK[a], ICES[b] (HELCOM stations)
Large-scale land-catchment drivers associated with MBs	Q to MB	Freshwater discharge (Q) into the MB	1994–2010	Yearly	HELCOM PLC 5.5[c]
	Nutrient concentrations in the Q to MB (DIN, DIP, TN, TP)	Nutrient concentrations in the freshwater flowing into the MB	1994–2010	Yearly	HELCOM PLC 5.5[c]
	Nutrient loads into MB (DIN, DIP, TN, TP)	Nutrient loads entering the MB	1995–2010	Yearly	HELCOM PLC 5.5[c]
Multi-scale land-catchment drivers associated with CBs	Large-scale Q to CB-related MB	Freshwater discharge into the specific MB associated with the CB	1994–2010	Yearly	HELCOM PLC 5.5[c]
	Local Q to CB	Freshwater discharge from local catchment into the CB	1990–2019	Monthly	SMHI HYPE (version 5.9)[d]
	Nutrient concentrations in local Q to CB (DIN, DIP, TN, TP)	Nutrient concentrations in the local freshwater flow into the CB	Varying (longest 1990–2019)	Varying	MVM[e]
	Nutrient loads by local Q to CB (DIN, DIP, TN, TP)	Local nutrient loads by the freshwater discharge entering the CB	Varying (longest 1990–2019)	Monthly	MVM[e], SMHI HYPE (version 5.9)[d]
Hydroclimate conditions over multi-scale land-catchments associated with CBs and MBs	Precipitation, Temperature	Precipitation/air temperature over the catchment of the MB/CB	1990–2018	Monthly	CRU TS 4.03[f]
Multi-scale climate drivers in/over CBs and MBs	Last day with ice	Last day with 10% or more ice cover over the MB/CB	1993–2006	Yearly	FMI[g]
	Duration 10% ice cover	Duration of the period with 10% or more ice cover over the MB/CB	1994–2006	Yearly	FMI[g]
	Wind speed	Wind speed at 10 m over the MB/CB	1990–2019	Monthly	ERA5[h]
	Net shortwave radiation, total cloud cover	Net shortwave radiation and total cloud cover over the MB/CB	1990–2019	Monthly	ERA5[h]
Multi-scale hydrosphere drivers within CBs and MBs	Water temperature and salinity	Average temperature and salinity over the surface layer of the MB/CB	Varying (longest 1990–2019)	Monthly	SMHI SHARK[a]
Large-scale hydrosphere variable in CB-related MB	Variable concentration in CB-related MB (Chl-a, Ratio Winter TN:TP)	Concentration of Chl-a and Ratio Winter TN:TP in the specific MB associated with the CB	Varying (longest 1990–2019)	Yearly	SMHI SHARK[a], ICES[b] (HELCOM stations)

[a] SMHI SHARK (The Swedish Ocean Archive) database.
[b] ICES (International Council for the Exploration of the Sea) Dataset on Ocean Hydrography (keeping only the HELCOM monitoring stations).
[c] Helsinki Commission (HELCOM) Pollution Load Compilation (PLC) 5.5 (HELCOM, 2013).
[d] Swedish Meteorological and Hydrological Institute (SMHI) Hydrological Predictions for the Environment (HYPE) (version 5.9) (Lindström et al., 2010).
[e] Soil, Water, Environment (MVM) database from SLU university (https://miljodata.slu.se/mvm/Default.aspx).
[f] Climate Research Unit Time Series version 4.03 (CRU TS 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).

understand the dynamics of the system and their relationships with Summer Chl-a. The investigated nutrient variables include surface winter and summer Total Nitrogen (TN) concentrations (Winter TN and Summer TN) and Total Phosphorus (TP) concentrations (Winter TP and Summer TP), and surface winter Dissolved Inorganic Nitrogen and Phosphorus (DIN and DIP, respectively) concentrations (Winter DIN and Winter DIP, respectively).

Moreover, we included surface summer DIN and DIP concentrations (Summer DIN and Summer DIP, respectively), Winter limiting Dissolved Inorganic (DI) nutrient, and the ratio between Winter TN and Winter TP (Winter TN:TP) into the analysis. Summer DIN and DIP are related to the algal growth and can indicate nutrient excess during the growth period. The Winter limiting DI nutrient was calculated using the Redfield ratio (in mass) as the minimum between Winter DIN divided by the mass Redfield ratio between nitrogen and phosphorus in algae (7.2) and Winter DIP (Redfield, 1958). This gives the nutrient concentration (in DIP equivalent concentration) that limits the algal growth according to the stoichiometric nutrient ratio contained in algae. Winter TN:TP describes the relative evolution of winter TN and TP. Winter ammonium, nitrite, and nitrate surface concentrations, which summed together form the Winter DIN variable, were also considered individually as potentially important variables (SM Section S6). The summer bottom dissolved oxygen concentration (Summer bottom O2) during the months of August and September was also considered as an additional potentially important variable (SM Section S6).

2.2.2. Drivers

Different types of drivers were considered in order to investigate how their variations and changes relate to those in studied water quality variables (Table 1). The drivers were divided into three main categories, based on their locations and scales within the overall land-coast-sea system: i) land-catchment drivers, ii) Coastal or Marine Basin drivers (i.e., drivers acting within or over the Coastal or Marine Basin in question), and iii) open sea drivers (acting over the larger scale of the whole Baltic Sea and/or its total land catchment).

First, land-catchment drivers represent anthropogenic, land-use and hydroclimatic changes in the local coast catchment or the larger Baltic catchment that can influence the local coastal water or the open sea water conditions, respectively. Freshwater discharges and their TN, TP, DIN and DIP concentrations and loads to coastal and open sea waters were considered, as they represent some of the main anthropogenic eutrophication drivers (Nixon, 1995). We also considered precipitation and air temperature conditions over the catchment areas associated with Coastal and Marine Basins in the analysis.

Second, local coastal and larger-scale marine drivers represent variations and changes in the hydrospheric local coastal and larger-scale marine water subsystems and their atmospheric forcing, respectively. Sea-ice, in addition to its direct biogeochemical and physical effects (Kari et al., 2018), is also an indicator of winter climatic conditions and thus, we included the length of ice season and the last winter/spring day with ice cover as related drivers. Seawater salinity and temperature conditions were also considered. Moreover, wind conditions over the CBs and MBs can be expected to affect the horizontal mixing between basins (Chen et al., 2019; Lehmann et al., 2002) and the vertical mixing and stratification within a basin (Lips et al., 2017). Net shortwave heat flux at the CB and MB water surface and related cloud cover conditions can indicate energy balance and light available for photosynthesis, as well as affect both hydrodynamic and water quality conditions through water temperature and stratification (Omstedt et al., 2014).

Third, considered open sea drivers represent variations and changes in water quality conditions of the open sea, which can further influence coastal water quality conditions through the water exchanges between coastal and open sea waters.

2.3. Data analysis

This section summarizes the temporal, and CB and MB scale data aggregation methods, which are described in further detail in SM Section S1.2, and explains the temporal trends and correlation analysis.

2.3.1. Spatial and temporal aggregation

In order to analyze the effects of the identified drivers on water quality variables, various datasets were retrieved and aggregated onto each CB and MB. Table 1 summarizes, references and abbreviates the datasets used for the investigated variables and drivers. Sets of directly measured data were preferred, but reanalysis data were also used for drivers for which direct measurements were unavailable or too challenging to interpolate. To calculate temporal trends, all variables were resampled to a yearly resolution, as well as to seasonal resolutions if possible.

The different hydrospheric variables of Summer Chl-a, Summer and Winter DIN, DIP, TN and TP, the winter limiting DI nutrient, and Winter TN:TP were aggregated onto CBs and MBs, following the Swedish implementation of the WFD (Havs- och vattenmyndigheten, 2019) for the CBs and the HELCOM BSAP calculations for the MBs. Data requirements imposed by these management frameworks were slightly relaxed for the winter variables to have enough data from the CBs and MBs for the statistical analysis. For each variable, surface concentration measurements (0–10 m) from monitoring stations within a CB were averaged to obtain yearly mean concentration values. For the Summer Chl-a variable, both measurements at a given depth within 0–10 m and depth-integrated measurements within 0–10 m were used. For the nutrient variables, only measurements at a given depth within 0–10 m were used. CBs and MBs with at least 50% data availability for every 6-year period within at least a 10-year long time series were included. Every CB is also associated either with the MB that has the highest water exchange with it, or with the nearest MB with sufficient data availability. These pairs of CB and MB were used to calculate the open sea influence on the coastal waters for the investigated hydrospheric variables. The selected CBs and MBs with at least a 20-year long time series for Summer Chl-a are those with data variations and trends shown in Fig. 2.

The land-catchment drivers of freshwater discharges (abbreviated as Q in tables and figures) and associated nutrient loads of TN and TP into the MBs were obtained from the HELCOM PLC5.5 dataset. Flux average riverine concentrations of TN and TP were further calculated as the ratio of reported nutrient load divided by the corresponding reported freshwater discharge to a MB (according to SM Eq. (2)). For the CBs, data from concentration monitoring stations in the local coastal catchments of CBs were obtained from the SLU MVM database (Table 1, with source reference). The official Swedish data reported for freshwater discharges in that database represent validated simulation results by the hydrological HYPE model of the Swedish Meteorological and Hydrological Institute (SMHI) for the mouths of rivers with corresponding concentration measurement stations for DIN, DIP, TN and TP in the SLU MVM database (Table 1). Associated loads were further calculated by multiplying discharge and concentration for each given station and adding the loads for all stations associated with each investigated CB. Total flux averaged concentrations for DIN, DIP, TN and TP were further calculated by dividing the total load by the total freshwater inflow for each CB (according to SM Eq. (1)). Precipitation and temperature conditions were averaged over the land-catchments of different scales of each CB and MB (local catchment for CB, larger-scale catchment for the CB-related MB).

Coastal and Marine Basin drivers were aggregated over the corresponding CB or MB. Sea water temperature and salinity were calculated in a similar way to the water quality variables, using surface measurements of monitoring stations within the CB or MB. Sea-ice concentration data were obtained from the Finnish Meteorological Institute (FMI, Table 1) and averaged over each CB or MB area to calculate the last winter/spring day of the year with more than 10% sea-ice concentration and the number of days with more than 10% concentration of sea-ice cover over the CB or MB. Wind speed, net surface shortwave radiation and total cloud cover were obtained from the ERA5 reanalysis dataset (Table 1) and averaged over each CB or MB area.

2.3.2. Trend and correlation analysis

Trend correlation analysis was carried out to investigate the trend relationship between two variables or between a variable and a driver. Temporal trend slopes of each variable and driver (if applicable) were calculated for each variable-variable or variable-driver pair during their intersecting period with available data for each CB and MB. These trends were further compared with and trend correlations with corresponding variable or driver trends were calculated across all basins in each class (more or less isolated CBs, all CBs, all MBs; Fig. 1) using the Pearson correlation. Resulting coefficients of determination and *p*-value ranges, showing the significance (used here strictly in its statistical meaning) of the correlation, for each and all variable-variable and variable-driver pairs are displayed in form of heatmaps in the following Section 3. The number of CBs and MBs included in the correlation analysis, as well as the average length and standard deviation of the periods used are presented in SM Figs. S6–S11.

Due to the varying temporal availability of both variable and driver data, the time series for temporal trend correlation analysis were optimised to ensure a sufficient amount of data. For variable-driver comparison the minimum time-series length was 10 years, due to shorter time-series lengths of some drivers, such as the sea-ice and the MB nutrient load and concentration drivers. For the variable-variable comparison, as well as for the comparison between coastal variable and open sea drivers, the minimum time-series length was set to 20 years. This was chosen as both time series allowed for up to 50% missing data and for possible higher measurement uncertainties in nutrient and Chl-a concentration measurements since measurement depths and stations could vary for a given CB or MB, and such measurements can also be subject to horizontal heterogeneity (Scheinin et al., 2020).

This analysis allows for investigation of linear relationships between temporal trends in variable-variable or variable-driver pairs. Results indicate the relative part of the temporal trend variation of a studied variable that can be explained by the variation in temporal trend of another possible explanatory variable or driver. However, the analysis cannot show relationships for variable-driver pairs without temporal trends. In such cases, the interannual variations may be greater than the overall temporal trend. Drivers without any evident such trend are thus not considered in this analysis, even if their interannual variability influence may be strong. Moreover, drivers may also lead to nonlinear or delayed variable responses, which would necessitate further and other types of analysis, outside the scope of the present study.

Outlier data points may further affect trend correlations in ways that do not represent characteristic relationships for most of the CBs or MBs. Spearman's rank correlations, assessing the monotonicity of a relationship without testing its linearity, have then also been carried out in this study, in a similar way to the aforementioned method, but less susceptible to outliers. However, Pearson correlations are used in the main analysis because they give information about linearity of a relationship and thereby indicate how dominant a driver is in explaining variable temporal trends. Spearman's rank correlations are used as a complement, to verify that the Pearson correlations are not driven by outliers, and are presented in SM Figs. S3–5. Finally, for the variable-variable analysis, the northern CBs are underrepresented as no northern CB has available data for a 20-year period for Summer Chl-a, and only two have such availability for the nutrient variables. Thereby, emerging statistical relationships between hydrospheric variables may not be representative for the Bothnian Sea and Bothnian Bay.

3. Results and discussion

3.1. Relations between hydrospheric variables

Fig. 3 presents trend correlations between the water quality variables for the investigated CBs in different coastal classes (all CBs, more isolated CBs and less isolated CBs). This figure shows to which extent trend variations in a local coastal water quality variable are related to trend variations in another local coastal variable, thereby indicating medium to long term local internal dynamics and feedbacks. The analysis focuses on Summer Chl-a and its relationship with other variables.

For the more isolated CBs (panel B in Fig. 3, shown in orange in Fig. 1), trends in Summer Chl-a correlate strongly and significantly with trends in Summer TN, moderately and nonsignificantly with trends in Winter nitrite and nitrate (negative, coefficient of determination (r^2) of 0.3, SM Fig. S13) as well as moderately and negatively with trends in the Winter TN:TP (significant, r^2 of 0.4). Thus, in more isolated CBs, an increase in this ratio is associated with a decrease in Summer Chl-a. Trends in the ratio are further significantly correlated with trends in Winter TN and TP (small positive and moderate negative, respectively) and very strongly with Winter DIP (negative, r^2 of 0.8). This shows that Winter TN:TP is more influenced by phosphorus than by nitrogen for the more isolated CBs. Given the correlation between trends in Summer Chl-a and in Winter TN:TP, Summer Chl-a trends could be expected to be more affected by winter phosphorus trends than by nitrogen trends. This influence does not emerge from the analysis of all more isolated CBs (panel B), but is indicated by the moderate correlation between trends in Summer Chl-a and Winter DIP (r^2 of 0.3, nonsignificant) when removing the outlying CB of Inre Oskarshamområdet (panel A in Fig. 3, location shown in SM Fig. S1, and Summer Chl-a concentration shown in Fig. 2, panel F).

For the less isolated CBs (panel C in Fig. 3, shown in green in Fig. 1), trends in Summer Chl-a also correlate strongly and significantly with trends in Summer TN, as well as with trends in Winter DIN and TN (r^2 of 0.6 and 0.5, respectively), and with their component trends in Winter nitrate and ammonium (r^2 of 0.4, SM Fig. S13). However, trends in Summer Chl-a do not correlate with trends in the phosphorus variables. Summer Chl-a trends are also moderately but significantly correlated with trends in Winter TN:TP (r^2 of 0.4). Thus, for the less isolated CBs, an increase in the ratio is associated with an increase in Summer Chl-a. Trends in the Ratio Winter TN:TP are also moderately and significantly correlated with Winter TN trends, while the correlation with Winter TP trends is significant but small. Trends in Winter TN:TP also correlate strongly and significantly with trends Summer bottom O2 (r^2 of 0.6, SM Fig. S13).

The less isolated CBs exhibit similar water quality dynamics to the MBs (panel E in Fig. 3), for which the trends in Summer Chl-a are best explained by those in the nitrogen variables (Winter TN and DIN). Moreover, for both the less isolated CBs and the MBs, trends between Summer Chl-a and Winter TN:TP are similarly correlated, suggesting a nitrogen limitation in both cases. Indeed, anoxic conditions in large areas of the open sea can cause internal phosphorus release from the sediments (supported by strong, negative and significant correlations of Summer bottom O2 with Winter TP and DIP in SM Fig. S13 panel D) and enhance the denitrification process (supported only for the less isolated CBs in SM Fig. S13 panel C) (Savchuk, 2018). Both of these processes contribute to nitrogen-limited conditions in the open sea, especially in the Baltic Proper, favourable to cyanobacteria blooms (Vahtera et al., 2007; Meier et al., 2019). The similarity in water quality dynamics between the less isolated CBs and the MBs is consistent with the classification of the less isolated CBs as those with a strong water exchange with the open sea (residence time lower than 10 days). The trend correlation results support that water quality variables and processes in the less isolated CBs are relatively homogenized with the open sea. Thereby, in both the less isolated CBs and the MBs, the results indicate that medium to long term changes in Summer Chl-a conditions

Fig. 3. Coefficient of determination (r^2) for correlations between trends in coastal water quality variables for A: the more isolated Coastal Basins (CBs) except the outlier point from the Inre Oskashamnsområdet CB (shown in Supplementary materials Fig. S1), B: all the more isolated CBs, C: the less isolated CBs, D: all the studied CBs, and E: the Marine Basins (MBs). Excluding the outlier point influenced most the Winter DIP concentration (circled in A and B in black). The main variables in this study, Summer Chl-a and Ratio Winter TN:TP, shown in the lowest row and in the leftmost column, respectively, are written in red. r^2 is given by the number in the cell. - sign indicates a negative correlation. *: $p<0.05$; **: $p<0.01$; ***: $p<0.001$. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

are mostly linked to changes in the nitrogen variables (Summer and Winter TN, and Winter DIN).

For all CBs (panel D in Fig. 3), trends in Summer Chl-a only appear as strongly and significantly correlated to Summer TN (r^2 of 0.5), and to Summer DIN (r^2 of 0.3). However, these relationships bring limited additional information on system dynamics as phytoplankton are accounted for in both the Summer Chl-a and the Summer TN variables, and as Summer DIN is directly related to algal growth. Moreover, the absence of other significant correlations between trends in Summer Chl-a and those in other key variables indicates that the relationships distinguished for more and less isolated CBs, are masked when mixing all CBs in the analysis. Indeed, for the more isolated CBs, separated from the open sea through channels or straits, the significant negative correlation seen between trends in Summer Chl-a and those in Winter TN:TP supports that Summer Chl-a is more strongly limited by phosphorus, in contrast to the less isolated CBs (Walve et al., 2021). This is consistent with freshwater systems being generally phosphorus limited and marine systems being generally nitrogen limited (Paerl, 2009). The more isolated CBs act as a buffer between freshwater and open sea waters, suggesting greater variability in their water quality dynamics. This would explain the lower correlations seen between trends in Summer Chl-a and those in the other local coastal variables for the more isolated CBs. Winter TN:TP, which correlates strongly with Summer Chl-a for both more and less isolated CBs, is a standard indicator of nutrient limitation, and captures the relative evolution of winter nutrient concentrations, is therefore also used in the following driver analysis (comparison with other ratios used in Baltic Sea studies in SM Section S5). It should be noted that the northernmost CBs (in the Bothnian Sea and Bay) are not included in the variable analysis due to lack of 20-year long time series for these. The Bothnian Sea is reported to have transitioned from a phosphorus and nitrogen limited system to a nitrogen limited one over the past 20 years, while the Bothnian Bay still experiences general phosphorus limitation (Rolff and Elfwing, 2015; Tamminen and Andersen, 2007).

3.2. Relations between drivers and key hydrospheric variables

Local land-catchment drivers in Fig. 4, feeding directly into the local coastal zone, represent integrated anthropogenic and hydroclimatic changes in these catchments, influencing their local river discharges, nutrient concentrations in these, and associated nutrient loads to the coast (blue arrows in Fig. 5). For the larger-scale MBs, increasing precipitation trends are significantly correlated with decreasing Summer Chl-a trends, in consistency with the (weaker) relationship seen also between trends in freshwater discharge and Summer Chl-a in the MBs. The stronger correlation of catchment precipitation than freshwater discharge trends with those in Summer Chl-a may indicate precipitation effects due to its covariation with the saltwater inflow as well as with the freshwater discharges to the Baltic Sea (Chen et al., 2019).

For the MBs, trends in Summer Chl-a are further moderately correlated to those in TN and TP concentrations (r^2 of 0.3), but not to those in TN and TP loads. These results may differ from expectations of climate-driven increase in precipitation increasing nutrient loads and leading to increased eutrophication (Bring et al., 2015; Meier et al., 2012). However, the residence time of phosphorus in the Baltic Sea is longer (28 to 46 years; Eilola et al., 2011) than the 15-year trend periods investigated here for the MBs, which may be influenced by climate variability. The long phosphorus residence times affect the Baltic Sea internal dynamics (Stigebrandt, 2018) and may delay MB responses to changes in nutrient loads (Gustafsson et al., 2012).

For the coastal waters, in contrast to the open sea, trends in Summer Chl-a correlate with those in nutrient loads, rather than with nutrient concentrations in the freshwater discharges to the coast, in consistency with previous results for coastal waters (Voss et al., 2011). Trends in DIN load are significantly and positively correlated with those in Summer Chl-a (r^2 of 0.4 and 0.5 for the more and less isolated CBs, respectively). Trends in DIP load correlate with those in Summer Chl-a only for the less isolated CBs (r^2 of 0.6), while correlation of Summer Chl-a trends with those in TN and TP loads is weak for all coastal classes (r^2 of 0.1–0.2).

From the coastal variable analysis in the previous section, increasing DIN loads could be expected to increase Winter TN:TP for the more isolated CBs, while increasing DIP loads could be expected to decrease it for the less isolated ones. While the driver analysis does not show this (Fig. 4), the complementary Spearman's rank correlation results (SM Fig. S4) do show a strong and significant negative influence of DIP load on Winter TN:TP for the more isolated CBs, and the opposite for the less isolated ones. Overall, freshwater DIN loads exhibit a robust strong influence on Summer Chl-a for both the more and the less isolated CBs. Surprisingly, the local nutrient loads appear to have slightly stronger influence on the less than the more isolated CBs. However, for the less isolated CBs, only 8 CBs are considered in the analysis, compared to 26 more isolated CBs (SM Fig. S8). This sample size imbalance implies higher uncertainty in the less isolated CB results, which may underlie such small counterintuitive result differences compared with the more isolated CB results.

The direct CB and MB drivers in Fig. 4 represent direct climate-related conditions over and in the water bodies themselves. Sea-ice conditions, and especially the last day of the year with ice coverage, show the strongest and most robust correlations with Summer Chl-a, with coefficient of determination of 0.4 (significant), 0.3 (significant) and 0.2 (non-significant), for all, the most isolated, and the less isolated CBs, respectively (black arrows in Fig. 5). Summer Chl-a thus tends to increase with earlier thawing of sea-ice. This could be due to freshwater plumes that commonly develop under the sea-ice in coastal waters with freshwater inflow (Kari et al., 2018; Merkouriadi and Leppäranta, 2014), which have been observed in the Himmerfjärden Bay (Kari et al.,

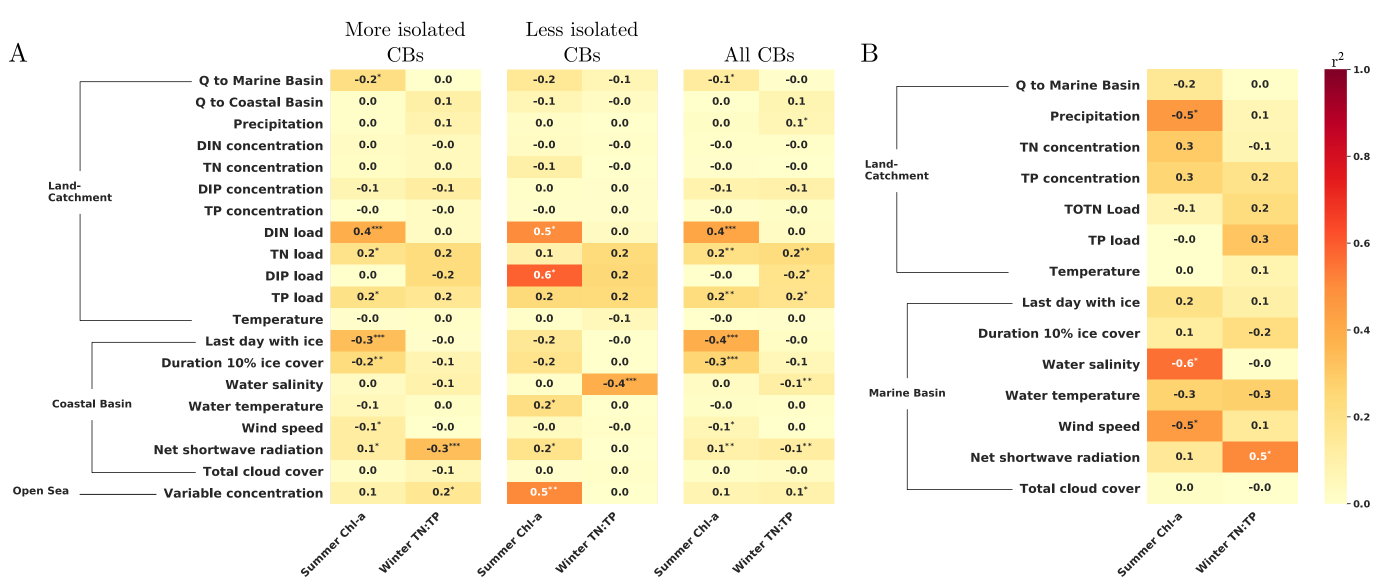


Fig. 4. Coefficient of determination (r^2) for correlations between trends in water quality variables and in drivers for A: the investigated Coastal Basins (CBs) classes and B: the Marine Basins. r^2 is given by the number in the cell. - sign indicates a negative correlation. *: $p<0.05$; **: $p<0.01$; ***: $p<0.001$.

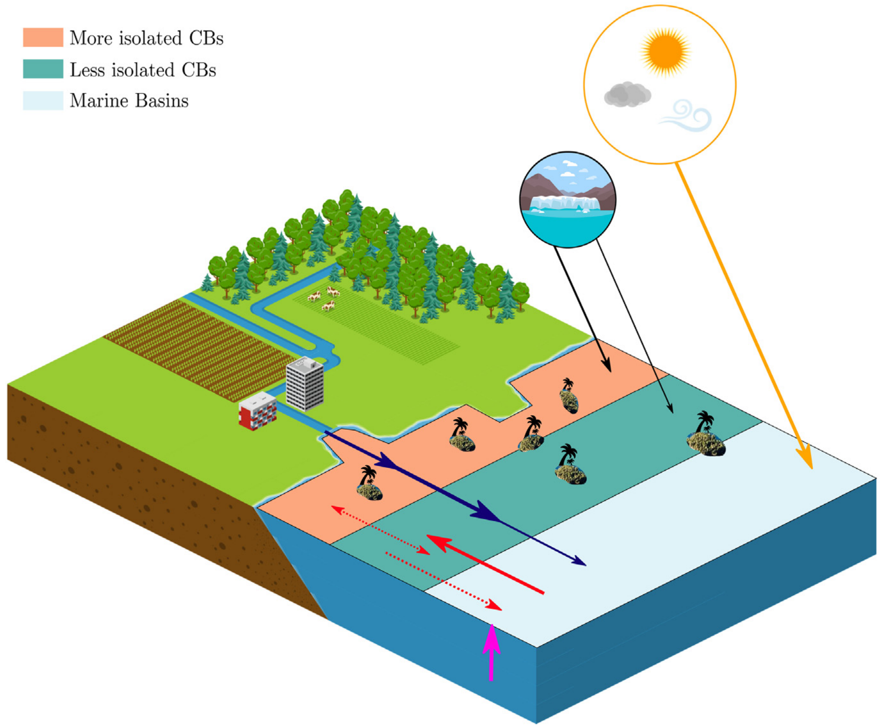


Fig. 5. Schematic illustration of the main drivers (identified in Fig. 4) and their dominant influences on the coastal and marine scales. The magenta arrow represent hydrospheric drivers within the Coastal Basin (CB) or Marine Basin (MB). The red arrows represent hydrospheric influences between coastal and marine scales (solid line: investigated in the study, dotted lines: possible influences not investigated here). The blue arrows represent land-catchment drivers, the black arrows represent sea-ice drivers and the orange arrow represents hydroclimatic drivers. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

2018). Under-ice plumes have been found to indicate a more intense growth period during the spring bloom, despite delaying its onset, and to modify the phytoplankton species composition (Kari et al., 2018), which can further influence the summer productivity.

For the MBs, trends in Summer Chl-a are also strongly and significantly correlated with those surface water salinity (r^2 of 0.6), with MBs that experience decreasing trends in the latter more likely to have increasing trends of Summer Chl-a (magenta arrow in Fig. 5). This indicates increased summer stratification, favouring summer cyanobacteria blooms (Loick-Wilde et al., 2019), which is also consistent with the strong negative correlation between trends in Summer Chl-a and those in wind speed (r^2 of 0.5, orange arrow in Fig. 5), and further supported by the frequency of cyanobacteria blooms being negatively correlated to strong westerly winds (Kahru et al., 2007). Higher wind speeds can also increase the mixed layer depth, leading to less phytoplankton, and reduced cyanobacteria blooms, associated hypoxia, and subsequent internal phosphorus loading (Meier et al., 2019). These effects are less evident when considering the CBs, possibly due to varying CB responses depending on depth, stratification conditions and wind exposure (Raateoja and Kauppila, 2019). Moreover, the monthly resolution of the wind dataset may not be able to capture eutrophication-relevant coastal wind conditions.

For all CB classes, seasonal trends in spring net shortwave radiation, and in winter and spring total cloud cover show moderate correlations (r^2 between 0.2 and 0.4) with trends in both Summer Chl-a and Winter TN:TP (SM Fig. S2), in contrast to trends in corresponding annual averages. This, together with the relationship between sea-ice cover and Summer Chl-a trends, indicates that sea-ice conditions could be a proxy for relevant seasonal aggregation of winter and spring hydroclimatic conditions that influence Summer Chl-a possibly through the spring stratification. For the MBs, summer net surface radiation shows significant and strong (r^2 of 0.8) positive correlation with Summer Chl-a. Moreover, the negative correlations between trends in Summer Chl-a and those in yearly precipitation, wind speed and water salinity are mainly determined by their summer component for the MBs (SM Fig. S2). Thereby, for the MBs, summer hydroclimatic conditions show a dominant effect on Summer Chl-a, and conditions associated with increased summer stratification and light availability are also associated with increased Summer Chl-a (Meier et al., 2019; Loick-Wilde et al., 2019).

In general, correlations between open sea drivers and local coastal variables may represent influences of the open sea on the CBs and/or of other drivers on both the coastal and marine scales (red arrows in Fig. 5). For the less isolated CBs, correlations are weak between drivers of the open sea conditions and the local coastal variables, indicating direct influences of open sea conditions on these coasts (SM Fig. S16). Correlations are similar (r^2 of 0.5) for both the less and the moderately isolated CBs (subset of the more isolated CBs defined in SM Fig. S1), indicating commonly strong open sea influences on Summer Chl-a conditions for most of the coastal waters except the most isolated ones, as well as for Winter DIP across all coastal classes (SM Fig. S17). Indeed, Raateoja and Kauppila (2019) has shown that sea-based conditions dominate after the outer brinks of estuaries in the Gulf of Finland and Ménesguen et al. (2018) has shown that even non eutrophic open sea conditions may account for a large share of the inorganic nutrients present in coastal zones. Mitigation of local coastal eutrophication thus requires large-scale management of open sea eutrophication conditions and not just nutrient load reduction in the local coastal land catchment (Almroth-Rosell et al., 2016; Vigouroux et al., 2020). However, local coastal eutrophication management cannot passively wait for measures to be taken in other coastal land catchments for improving the open sea conditions, to in turn reduce eutrophication in their own local coastal waters. Nutrient load mitigation is required over the total land catchment of the whole Baltic Sea and all its coastal waters in order to improve the open sea conditions, such that they can then reinforce the local efforts. Moreover, coastal ecosystem management that increases nutrient retention in the coastal zone can help to trap nutrients from the open sea (Carstensen et al., 2020).

Each such analysis is specific for its study period with available data (1990–2020 in the present study), beyond which uncovered relationships may not hold. Additional studies of longer data time series with use of expanded statistical methods are called for to also capture delays and nonlinearities (study limitations mentioned in Section 2.3.2). Moreover, variations of morphological properties within a CB class (SM Table 2), along with topographical variations, can influence CB susceptibility to hypoxia (Virtanen et al., 2019). This can explain the lack of correlation between Summer Chl-a and Summer bottom O2 (SM Fig. 13), and also influence coastal eutrophication responses to other drivers. Finer coastal classification, for example based on more detailed morphological and topographical characterizations, can add further insights, but requires also greater data availability. Other ecological conditions not considered in this work may also influence coastal eutrophication, such as fish communities (Bergström et al., 2019), and macrophyte habitats (Berthold et al., 2018; Donadi et al., 2018). However, such ecological conditions are also influenced by eutrophication, and thus their effects on eutrophication cannot be distinguished via trend correlation analysis. Nevertheless, the relatively long time period with data investigated in this study, covering three decades, has revealed some key temporal-trend relationships of recent coastal eutrophication developments for and across conditions prevailing along the long Swedish Baltic Sea coast.

4. Conclusions

In this study, we have analyzed data-given relationships of temporal trends in water quality variables and those in their combined anthropogenic, atmospheric and hydrospheric drivers. The analysis has focused on Swedish coastal waters, partitioned into more and less isolated Coastal Basin classes, and their associated open sea Marine Basins for the period 1990–2020. Key conclusions from this analysis are summarized as follows:

- Local internal dynamics of coastal water quality differ between more and less isolated coastal waters. In general, phosphorus plays a dominant role for Summer chlorophyll-a concentrations and eutrophication in more isolated coastal waters, while nitrogen is dominant for less isolated ones. The latter coastal waters exhibit similar relationships between eutrophication related water quality variables as in open sea waters.
- On the coastal scale, trends in sea-ice conditions, possibly representing a eutrophication-relevant aggregation of winter and spring hydroclimatic conditions, and in local nutrient loads emerge as the main drivers influencing trends in Summer chlorophyll-a concentrations. On the larger open sea scale, trends in Summer chlorophyll-a concentrations are primarily influenced by trend in wind and salinity, likely relating to stratification, and by precipitation and nutrient concentrations in freshwater discharges. Overall, climate conditions emerge as the strongest influences on water quality in the open sea over the past 30 years.
- For less isolated coastal waters, trends in Summer chlorophyll *a* concentrations correlate best with those in corresponding open sea concentrations, emphasizing a key role of open sea conditions for these coastal waters. More isolated coastal waters exhibit more mixed influence signals of combined, overlapping and competing, land, climate, coast and open sea drivers. At any rate, coastal classification based on simple estimation of water exchange with the open sea can provide valuable information for coastal eutrophication management by identifying different dynamics and driver responses of more and less isolated coastal waters.
- The dominant roles of either or both nitrogen and phosphorus in different coastal and open sea parts of the Baltic system emphasize the importance of dual nutrient management strategies for reducing both coastal and marine eutrophication.

Overall, our results challenge a simplified unidirectional source-to-sea consideration of the coastal waters as recipients of mainly land-based nutrient loads, without also accounting for the possible even stronger influences of the open sea on the coastal conditions. Such oversimplification may be exemplified by the WFD, which focuses on improving water quality in inland and coastal waters, with conditions in the latter viewed as relatively straightforward reflections of nutrient and pollutant management measures in the local land catchment of each coast. In contrast, our results indicate that improvement of water quality in any local coast requires relevant measures to be taken over the whole large-scale Baltic Sea catchment with associated open sea improvements then further feeding into and reinforcing the local catchment and coastal efforts. For such combined cross-scale efforts, analysis of trend correlations between water quality and eutrophication variables and their drivers on different scales can provide essential information on likely management responses and feedbacks.

CRediT authorship contribution statement

Guillaume Vigouroux: Conceptualization, Methodology, Software, Formal analysis, Investigation, Writing – original draft, Data curation, Visualization. **Elina Kari:** Methodology, Investigation, Writing – review & editing, Visualization. **José M. Beltrán-Abaunza:** Data curation, Writing – review & editing, Visualization. **Petteri Uotila:** Validation, Writing – review & editing. **Dekui Yuan:** Validation, Writing – review & editing. **Georgia Destouni:** Conceptualization, Methodology, Supervision, Writing – review & editing, Visualization, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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- SMHI kindly provided river discharge and nutrient loading datasets.
- The freshwater nutrient concentrations were extracted from the Miljödata MVM database maintained by the Department of Aquatic resources and the Department of Aquatic Sciences and Assessment (SLU).
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Appendix A. Supplementary materials

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