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Impact of antifouling paint regulation on copper and zinc loads from leisure boats in Swedish waters

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ABSTRACT

The Baltic Sea, designated as a Particularly Sensitive Sea Area, hosts an extensive recreational boating sector, with Sweden contributing a substantial share due to its long coastline and high number of leisure boats. Antifouling coatings used on these vessels are sources of copper and zinc pollution, making it essential to identify where emissions occur and how biocide regulations affect environmental loads. To this end, this study mapped all boat berths along the Swedish coast to determine the number and spatial distribution of recreational vessels. Combined with release rate data for commercially available coatings, current inputs of copper and zinc to Swedish coastal waters from the nearly quarter-million leisure boats were estimated at 22.1 and 17.1 tons/yr, respectively. Copper load distributions varied geographically depending on boat density, permitted paint types and water salinity, with the more saline west coast accounting for the largest share (83%). Comparison with other coastal copper sources revealed antifouling coatings on leisure boats and ships to account for a significant proportion (32%) of the total yearly input. Scenario analysis further revealed that copper loads from current leisure boat products exceed nearly three-fold the maximum permissible level of the environmental risk assessment (8.8 tons/yr) that granted these products market approval (Scenario 1). However, enforcement of the updated European Union environmental risk assessment procedure (Scenario 2) could reduce copper loads substantially (to 8.0 tons/yr), if implemented correctly.

1. Introduction

There are numerous ways in which recreational boating can negatively affect the marine environment with modes of impact such as noise pollution, emission of sewage water, hydrocarbons and oil, anchor damage, littering, propagation of non-indigenous species and emission of toxic substances from antifouling paints (Byrnes and Dunn, 2020). For the semi-enclosed Baltic Sea, the management of recreational boating and its use of antifouling coatings are of particular environmental importance, given both the sensitivity of its unique ecosystem and the long water residence time of approximately 30 years (Snoeijs-Leijonmalm and Andrén, 2017).

The latest environmental status assessment by HELCOM found that only one of 24 assessed sea basins achieved Good Status in the evaluation of copper (HELCOM, 2023). Meanwhile, copper-based antifouling coatings on ships and recreational vessel hulls have recently been estimated to be a major source of copper to the Baltic Sea, accounting for nearly 40% of the annual input (Ytreberg et al., 2022). Even though most of this load was attributed to ships (518 out of 575 tons per year), the input from leisure boats should not be overlooked, as recreational boats in the Baltic Sea area are used during the most biologically active time periods of the year (spring/summer) and they are predominantly moored in sheltered, shallow bays such as nearshore areas of coastal water which constitute essential reproduction habitats for many species (Kraufvelin et al., 2018; Sagerman et al., 2020; Pursiainen et al., 2021). The use of copper-based antifouling coatings results not only in the release of copper into the marine environment but also of zinc, as zinc oxide is commonly added to paint formulations (Lindgren et al., 2018; Yebra et al., 2006). Lagerström et al. (2020a), measured seasonal dissolved copper and zinc concentrations in the waters of two Baltic Sea marinas (Sweden and Finland), showing exceedance of Environmental Quality Standards (EQS) during the boating season, with clear links to

antifouling paint leaching. Additionally, particles from antifouling paints also pose an environmental problem, as they can introduce concentrated sources of biocides and heavy metals into sediments and the surrounding water (Turner, 2021). Changes in water quality due to metal contamination have been widely shown to alter marine biological communities, including shifts in the abundance and composition of sessile and mobile organisms (Perrett et al., 2006; Neira et al., 2014; Miller et al., 2020), as well as substantial bioaccumulation of copper and zinc in exposed biota (Johnston et al., 2011; Neira et al., 2014; Bighiu et al., 2017; Watson et al., 2018).

Nine countries border the Baltic Sea, where Sweden holds the longest coastline and is also home to the largest number of recreational vessels (HELCOM, 2018). The latest national survey estimated the total number of seaworthy recreational vessels present in inland waters and along the coast to 864,200, equivalent to one vessel per every sixth Swedish household (Swedish Transport Agency, 2021). As many boaters coat their hulls with antifouling paint to prevent the attachment and growth of fouling organisms, recreational marinas in Sweden are well-known hotspots for metal pollution (Eklund et al., 2014; Lagerström et al., 2016, 2020a; Bighiu et al., 2017). All biocidal antifouling paint products currently approved for the Swedish recreational market are copper-based and nearly all also contain zinc oxide (Swedish Chemicals Agency, 2025). Upon immersion, the paints therefore typically leach both copper and zinc into the surrounding water, but the magnitude of release will depend on the formulation of the product as well as local physico-chemical variables such as salinity (Ytreberg et al., 2017; Lagerström et al., 2020b).

In the EU, the placement of antifouling coatings on the market is regulated by the Biocidal Products Regulation (BPR) requiring all biocide-containing products to be assessed for their efficacy and potential environmental risk (European Parliament and Council, 2012). Approval for market placement can be obtained at a national level or extended to several Member States if the company applies for so-called mutual recognition of their product (ECHA, 2025). In 2017, a new procedure for the environmental risk assessment (ERA) of antifouling paints for pleasure crafts was proposed to harmonize the ERA procedure between Member States and streamline applications for so-called "mutual recognition" between countries (ECHA, 2017). Previously, the ERA was either performed on a national level or not at all, resulting in large differences in deemed acceptable environmental risk between member states (Ytreberg et al., 2021). Such discrepancies could be remedied by the harmonized ERA, but approval of new or existing products under the updated procedure has so far been slow due to delays in its implementation by member states and the fact that many coatings struggle to meet the ERA requirements (ECHA, 2024). This is reflected on a national level by the approval status of the products currently on the Swedish market, where many have been granted multiple approval extensions pending updated decisions (Swedish Chemicals Agency, 2025). Meanwhile, several products are scheduled to be phased out from the market as their applications for approval were either denied or withdrawn by the applicant. New products will thus likely make their way onto the market to replace existing ones over the next few years given the newly implemented ERA.

Although national-scale estimations of copper loads from leisure boat antifouling coatings have been undertaken (e.g., Sweden: Johansson et al., 2020; Germany: Morling and Fuchs, 2021), no study has examined how changes in biocide regulation and its ERA could alter the resulting emission loads. For Sweden in particular, emissions on a regional scale are likely to be shaped not only by boat density but also by salinity, as its coastal waters span a pronounced salinity gradient from nearly freshwater in the northeast to almost fully marine conditions in the southwest. There is therefore a clear need for more spatially resolved load assessments that can also evaluate the regulatory sensitivity of emissions from the leisure boat sector.

The aims of the current study were to investigate (1) the impact of biocidal products regulations on the load of copper from antifouling paint on leisure boats to Swedish coastal waters and (2) the relative importance of leisure boats in these waters in comparison to other sources of copper and zinc. Both aims were examined across multiple geographical scales to identify potential local and regional differences. To achieve this, leisure boat berths were mapped along the Swedish coast to estimate the number and spatial distribution of boats. The coastal emissions of copper and zinc from leisure boats could then be estimated with a high spatial resolution for four investigated scenarios. The first scenario (Current) assessed emissions based on existing antifouling products on the market. Three additional scenarios focused on copper: Scenario 1 modeled emissions from coatings releasing the maximum allowable copper under the former national ERA, Scenario 2 applied the limits set by the newly EU-harmonized ERA and Scenario 3 estimated emissions from coatings releasing only the minimum effective amount of copper needed to prevent fouling, given that current products often exceed this threshold (Lagerström et al., 2020b). Finally, the load from leisure boats was compared to those of other sources, including commercial ships, atmospheric deposition, point sources and riverine input, and the regulatory measures by which to reduce these emissions discussed.

2. Materials & methods

2.1. Geographical distribution of leisure boats

2.1.1. GIS-mapping from orthophotos

Water body units are geographical management units delineated by each European Member State to assess water quality under the EU Water Framework Directive (WFD, European Parliament and Council, 2000). Surface water bodies designated as 'coastal water' include all coastal waters within 1 nautical mile from each country's baseline (i.e. the line that divides internal waters from the territorial sea). Here, the geographical distribution of leisure boats was determined by manually mapping the location of moorings, both in recreational marinas and at private docks, along the Swedish coast in all its 724 coastal water body units (CWB). Swedish recreational marinas are typically fully occupied with long waiting queues for berths, especially in urban areas. The number and location of boat moorings were thus used as a proxy for the counting and geographical distribution of leisure boats in this study. Only moorings intended for motorboats and yachts were considered and not smaller crafts such as rowing boats and dinghies.

The mapping was carried out in the Geographic Information System (GIS) software QGIS from orthophotos covering the Swedish coast during the months of the year when most boats would be launched in the water (April–September) to be able to identify actively used moorings. Coastal orthophotos (0.4 m resolution) from the most recently available years (2020−2022) were requested in December 2022 from the Swedish mapping, cadastral and land registration authority (Lantmäteriet). In QGIS, moorings were recorded by drawing either lines or points for all moorings identified on the orthophotos. The point layer was created to record single moorings (e.g. private docks), i.e. one point was equivalent to one mooring. When more than two side-by-side berths were identified (e.g. marina jetties), a single line along the jetty was drawn if boats were only moored on one side, while double lines were drawn if moorings were present along both sides (Fig. S1 of the Supporting information). Additionally, the orientation of the mooring, i.e. whether it was parallel (starboard or port side facing the dock) or perpendicular (bow or stern facing the dock), was recorded for each line to more accurately estimate the number of moorings based on the length of the line features, as described next.

2.1.2. Deriving the number of boats

For the point layer, one point was equal to one vessel. For the line layer, the relationship between the length of the lines and the number of moored boats needed to be determined. The size of recreational vessels may differ depending on location along the coastline. Mooring location

will also determine the antifouling products permitted to be applied on the hull. Hence, all lines and points on the Swedish coast were divided into three regions ('west', 'south-east' and 'north-east') with geographical boundaries reflecting those of the current national regulation of biocidal antifouling coatings (Fig. 1). The relationship between the length of the jetty and the number of moored boats was investigated and statistically validated within each of the three coastal regions and for the two types of mooring orientation (perpendicular or parallel) as outlined in the supporting information. The calculated conversion factors (i.e. meters of jetty per boat) were applied to the entire line layer. Together with the number of moorings from the point layer, a total boat count per CWB was derived. Mooring arrangement, i.e. whether the boat is docked at a private berth or in a marina was estimated. A simplified analysis was conducted whereby the jetties mapped in the line layer with an estimated ≥10 moored boats were assumed to be part of a marina.

2.2. Scenarios for load estimation of copper and zinc from leisure boats

2.2.1. Annual load calculation

All loads were firstly calculated on a local level per individual Swedish CWB. Leisure boats are only actively used during 10% of the boating season and typically navigate the water in close proximity of their home mooring (Swedish Transport Agency, 2010, 2015, 2021). 100% of the calculated release of metals from the painted hull of a leisure boat was therefore assumed to occur within the bounds of the

Fig. 1. Regional restrictions for the use of antifouling paints along the Swedish Coast. The use of biocidal paints is only allowed between the Norwegian border and Örskär. For the calculations, the Swedish coast was divided into three regions ('west', 'south-east' and 'north-east') with geographical boundaries reflecting those of the current national regulation of biocidal antifouling coatings. Crosses show the location of field studies where release rates of copper and zinc have been determined for various leisure boat coatings in 2015 (Lagerström et al., 2018), 2018 (Lagerström et al., 2020b) and 2023 (Lagerström et al., 2025).

mooring's CWB. The calculated loads per CWB could then be aggregated on different spatial scales to evaluate both the local (water body unit), regional (west, south-east and north-east coasts) and national perspectives. For all scenarios, the annual load of a metal (copper or zinc, in kg/year) per CWB was determined from the derived number of boats using the following equation:
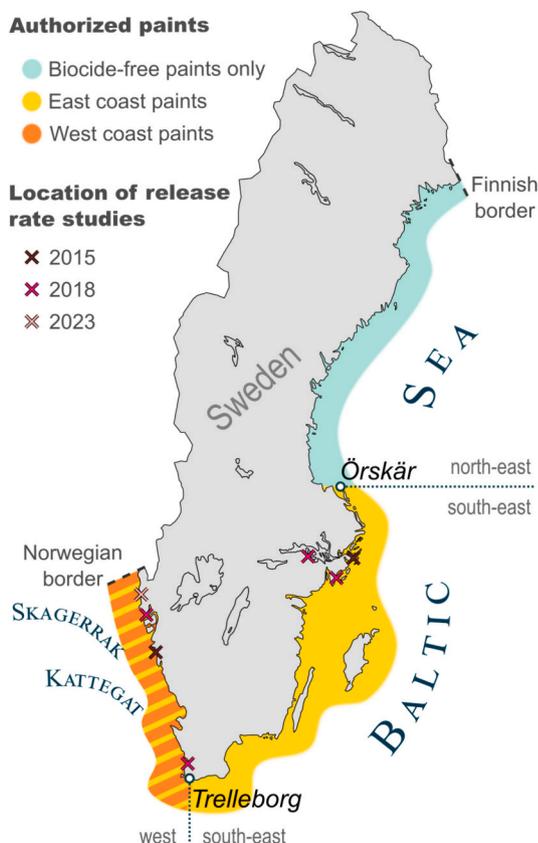
$$Annual\,load\,CWB_{Metal} = Number\,of\,boats \times WSA \times Application\,rate \times RR_{Metal} \times t$$
(1)

where *WSA* is the wetted hull surface area (in cm^2), the *Application rate* is the proportion of boats assumed to be coated with a copper-based antifouling paint (in %), RR_{metal} is the release rate of metal from the coating to the water (in $kg/cm^2/day$) and *t* is the mooring duration (in days). Estimates of the WSA, application rates and mooring duration were kept the same for all scenarios (Table 1, see Supplementary information for details on their derivation) while assumptions on paint selection and, consequently, the metal release rates differed.

2.2.2. Release rates in modeled scenarios

Four different scenarios were investigated. The current loads of copper and zinc resulting from the use of existing antifouling products on the market were first assessed (the 'current scenario'). Three alternative scenarios were also modeled specifically for copper (scenarios 1–3).

Currently, biocidal antifouling paints can be used on boats >200 kg with main mooring on the west and south-east coasts (Fig. 1). The Swedish paint market is divided into two categories: west coast paints and east coast paints. The latter are formulated with lower copper content and leaching rates, reflecting the higher level of environmental protection in place for the more sensitive south-east coast (Swedish Chemicals Agency, 2017). In the latest national boating survey, 90% of respondents stated that they use a paint approved for the waters of their home mooring (Swedish Transport Agency, 2021). Boaters were therefore assumed to comply with all restrictions and emissions were only calculated for boats moored in the CWBs on the west and south-east coasts (493 in total). Although east coast paints are permitted to be

Table 1
Parameters used in the load modelling from leisure boats for the 4 different scenarios, including the copper (Cu) and zinc (Zn) release rates (in $\mu g/cm^2/day$) used to derive loads depending on the coastal region. The release rates for scenarios 1 and 2 are the maximum allowable from a coating to pass the ERA, assuming it does not release any other active substance or substances of concern. The critical release rates for scenario 3 are the minimum necessary to inhibit settlement of macroscopic fouling.

Parameter	Applicable scenario	West coast	South-east coast
Wetted surface area (m^2)	All scenarios	17.69	17.95
Application rate (%)	All scenarios	62.2	41.5
Mooring duration (days)	All scenarios	148.2	148.2
Paint choice	All scenarios	West coast paints	East coast paints
	Current scenario (Cu)	Salinity-dependent model for west coast paints	Salinity-dependent model for east coast paints
	Current scenario (Zn)	5.07 (±1.31)	7.43 (±1.53)
Metal release rates ($\mu g/cm^2/d$)	Scenario 1: Former Swedish ERA (Cu)	5.38	0.97
	Scenario 2: New EU ERA (Cu)	3.44	2.39
	Scenario 3: Minimum effective (Cu)	7.0	2.0

used on the west coast, this practice was not deemed common. Hence, all boats moored on the west coast were assumed to be painted with west coast paints.

As sale statistics for specific antifouling products are not publicly available, representative average release rates from coatings used on each of the coasts had to be derived for the Current scenario. As water salinity has been found to influence the amount of copper released from any given coating (Ytreberg et al., 2017; Lagerström et al., 2018), it was also considered in the model. Release rate data for copper and zinc from coated panels statically immersed at 7 different locations in Swedish waters was compiled from three field studies conducted between 2015 and 2023 (Fig. 1). The dataset comprised of release rates from a total of 10 different leisure boat coatings, thus covering nearly half the products approved on the Swedish market at the time of the studies (Lagerström et al., 2018, 2020b, 2025). In the most recent of these studies, the release rates from statically exposed panels were shown to drop significantly after 4 months of immersion for most coatings. This was hypothesized to be due to the static water conditions not enabling renewal of the coating surface. Hence, to not underestimate the release from surfaces operating in more dynamic conditions, the average release rates from day 14 and 56 of exposure, as reported in all three studies, were used. A model for the average release rate, independent of paint choice and as a function of salinity was established from the compiled release rate data. Here, linear regressions between the compiled release rates and the salinities of the exposure locations were tested for statistical significance (Fig. 2). Significant relationships with salinity were ascertained for the release of copper from both west (termed WC1-WC6, $p < 0.0001$, $r^2 = 0.528$) and east (termed EC1-EC4, $p = 0.0002$, $r^2 = 0.538$) coast paints. The derived linear regression models for the two paint categories were therefore used to estimate the average release rate of copper as a function of each CWB's average salinity which were derived from the Baltic Sea salinity data set provided by the Institute Of Marine Research et al., 2012. For zinc, no significant relationship with salinity could be established ($p > 0.05$). The average zinc release rates of all coatings within the same category (west coast or east coast paint) were instead calculated and applied directly in Eq. (1). The 95% confidence interval (C.I.) of the linear regression models (copper) or the calculated averages (zinc) were propagated to the final calculated loads.
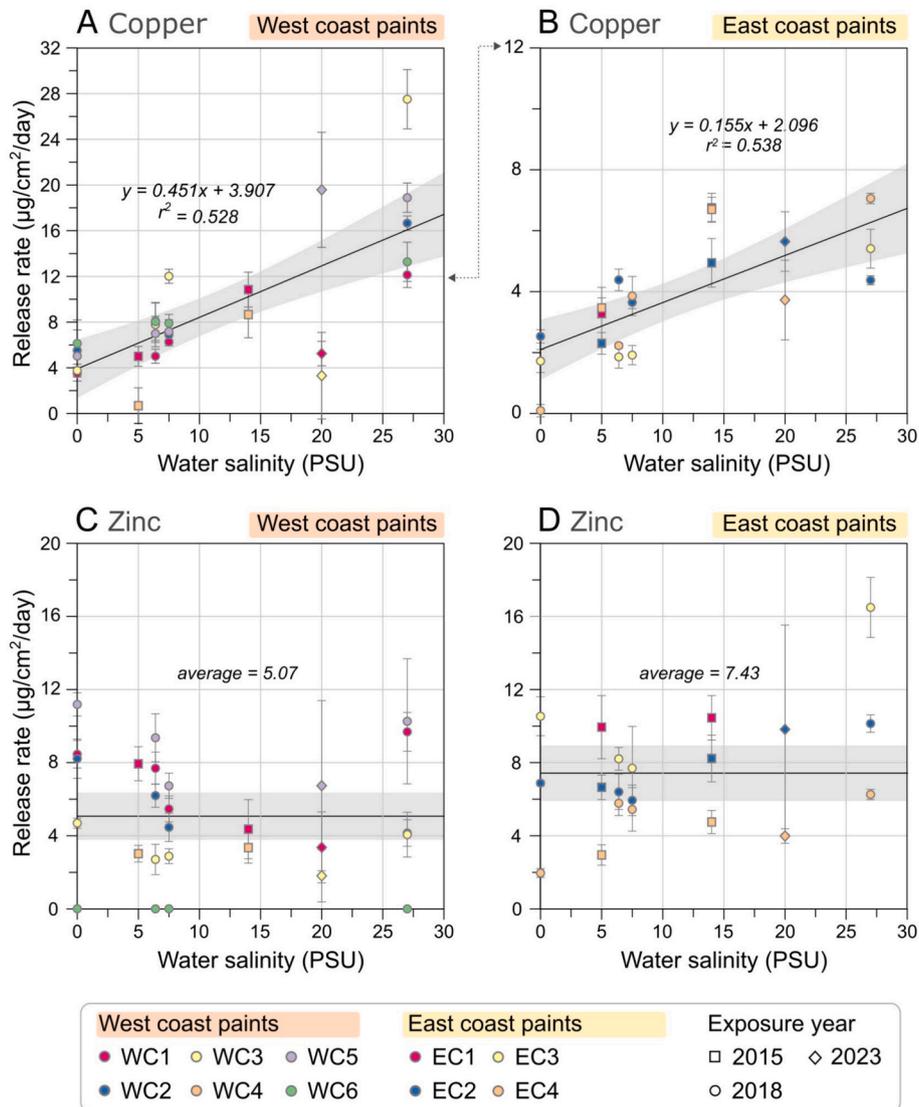
Fig. 2. Average release rates (day 14–56 of exposure) of copper and zinc from west (A, C) and east coast (B, D) paints from field studies in Swedish waters, as a function of salinity at the immersion sites in 2015 (Lagerström et al., 2018), 2018 (Lagerström et al., 2020b) and 2023 (Lagerström et al., 2025). Symbols show the average measured release rate and error bars show the standard deviation of replicates in the field experiment ($n = 4$ for all). Solid lines show either the linear regression when statistically significant (A and B), or the average across all salinities (C and D). The shaded grey areas illustrate the 95% confidence interval of the regression (A and B) or the calculated averages (C and D). Note the differences in scale for the y-axis (e.g. A and B).

Both in the previous (Swedish national) and current harmonized (EU) ERA, dissolved concentrations of active substances (e.g. copper) and substances of concern (e.g. Zn) are derived for modeled marinas using MAMPEC (Swedish Chemicals Agency, 2017; ECHA, 2017). A coating's expected release rate of e.g. copper is entered into the model to obtain its Predicted Environmental Concentration (PEC) in marina waters. To pass the ERA, i.e. be deemed to have an acceptable environmental risk, the derived PEC should not exceed that of the Predicted No Effect Concentration (PNEC). For both the national (scenario 1) and EU (scenario 2) assessments, maximum allowable release rates of copper to pass the ERA can be derived for the east and west coasts. These correspond to the release rates that should not be exceeded to ensure PEC < PNEC in the dissolved phase (Table 1). If a coating contains more than one active substance and/or substances of concern, a cumulative assessment is performed, reducing even further the allowed release of copper. The maximum allowable copper release rates are thus only applicable for products that do not release any other active substance or substance of concern. The loads derived using these release rates therefore represent theoretical worst-cases for both the previous (scenario 1) and new (scenario 2) ERA.

The final and third scenario was modeled to assess the potential redundancy of the current total copper load. Current market products often emit more copper than needed to prevent fouling (Lagerström et al., 2020b). For scenario 3, minimum copper release rates to completely prevent the settlement of macrofouling organisms, also known as *critical* release rates, were used as the in-going release rates in Eq. (1) for the load calculation (Table 1). Field studies coupling dose and efficacy have shown that releases of no more than 2 and 7 µg Cu/cm^2/ day are sufficient to deter fouling on the east and west coasts, respectively (Lagerström et al., 2020b, 2025).

2.3. Loads of copper and zinc from other sources

The emission of copper and zinc from shipping was evaluated using the Ship Traffic Emission Assessment Model (STEAM), version 4.3.0 (updated 16.03.2023). The model is comprehensively described in Jalkanen et al. (2021). In summary, STEAM employs AIS positions from all IMO-registered vessels and includes detailed information such as passenger capacity, main engine power, gross tonnage, vessel size, hull surface area, among others. This enables the production of activity and pressure maps (i.e. volumes of different waste streams) for individual vessels and for all shipping activities in Swedish coastal waters. Assessed waste streams included sewage, greywater, bilge water, open and closed loop scrubber water and antifouling paint (Table S3). To calculate ship emissions within each waterbody, an in-house script using Visual Studio Code (VSC) applying QGIS (v. 3.30's-Hertogenbosch) combined with OSGeo4W was applied. The loads of copper and zinc from point sources with outlets directly at the coast were compiled from the Swedish Agency for Marine and Water Management and their annual reports to HELCOM PLC and OSPAR RID, accessed via SMED (Svenska MiljöEmissionsData). This data covers the period from 2016 to 2018. Riverine input was derived from HELCOM PLC-Water (HELCOM, 2025) and OSPAR RID (OSPAR, 2025). The data includes annual loads from 40 Swedish rivers and was collected for the period from 2016 to 2022.

3. Results & discussion

3.1. Number and geographical distribution of leisure boats

Nearly a quarter of a million (232,963) boats are estimated to be moored in Swedish coastal water bodies from the mapped moorings (Table 2). According to the most recent national boating survey, there is an estimated 864,200 sea-worthy recreational vessels nation-wide (Swedish Transport Agency, 2021). When excluding all smaller crafts (e.g. kayaks, rowing boats, dinghies, jet skis) and only including crafts moored in coastal waters, the national boating survey yields 230,034

Table 2
Estimated number of boats moored in coastal water body units along the Swedish coast.

Coastal region	Number of boats (Percent of total count)
West	85,648 (37%)
South-east	122,269 (52%)
North-east	25,046 (11%)
Whole coast (total)	232,963 (100%)

leisure boats which is in very close keeping with the estimated number of boats here.

89% of the boats were found to be moored on the west (37%) and south-east (52%) coasts where the use of biocidal coatings is permitted. The geographical distribution of recreational boats in this area is illustrated in Fig. 3, where the number of boats per CWB ranges from 0 (typically water bodies located further from the coast) to 3807 boats, with an average of 422 boats. The three largest Swedish cities are Stockholm (south-east coast), Gothenburg (west coast) and Malmö (west coast) and they are all located on the coast. Not surprisingly, the highest number of leisure boats per CWB is found in proximity to these larger cities. In particular, the Stockholm archipelago area is a highly popular area for boating which is reflected by the number of boats: roughly 76,000 boats were found to be moored here, equivalent to 62% of all boats on the south-east coast (Table 2).

While the boats on the west coast are less numerous compared to those on the south-east coast, they may on the other hand be larger. The average length of jetty per mooring was found to differ significantly for parallel moorings, with longer mooring spaces on the west coast (10.5 m per boat) compared to the south-east (7.4 m) and north-east (6.2 m) coasts (Table S1). However, this difference was only partially reflected in the national boating survey, where the average WSA of boats was
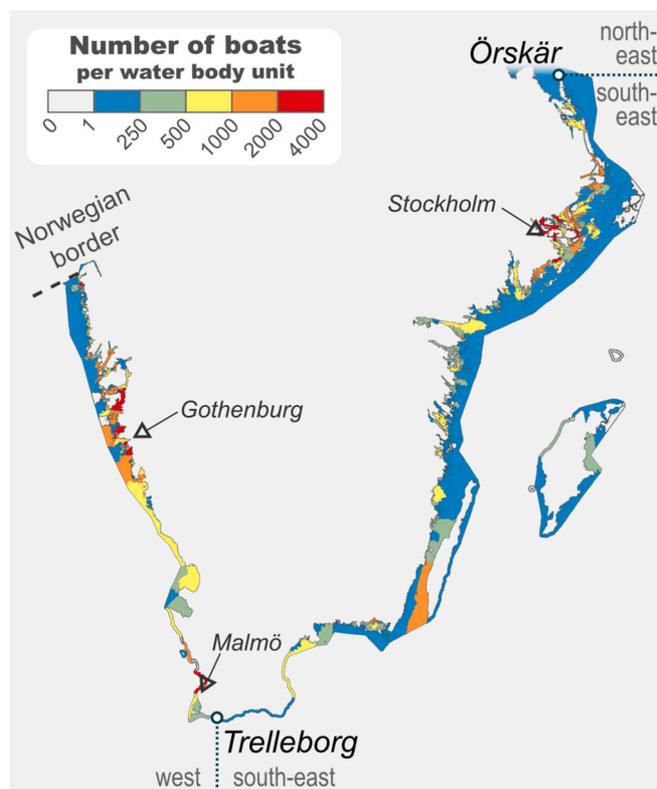
Fig. 3. Number of estimated leisure boats per coastal water body unit along the west and south-east Swedish coasts (i.e. from the Norwegian border to Örskär) where the use of biocidal antifouling paints is permitted.

17.69 m^2 on the west coast, 17.95 m^2 on the south-east coast, and smaller on the north-east coast at 12.77 m^2 (Table 1). Another difference between the three defined coastal regions lies in mooring arrangement, i.e. whether the boat is docked at a private berth or in a marina. Here a larger portion of boats were estimated to be moored in marinas on the west coast (78%) as compared to the south-east (60%) and north-east (55%) coasts. Marinas, typically sheltered environments with reduced water exchange, tend to be pollution hot-spots due to the congregation of boats painted with copper-containing coatings (Lagerström et al., 2020a). However, they may also play an important role in reducing the biocidal load to the environment by offering guidance and the possibility for more environmentally friendly hull maintenance facilities and services (Luoma et al., 2024). Emissions occurring in marinas are also easier for local authorities to monitor and regulate, especially if they are located on land owned by the municipality.

3.2. Loads from leisure boats to Swedish coastal waters

3.2.1. Current loads

Fig. 4A shows the number of boats derived for each CWB arranged from West to East for the part of the Swedish coastline where the use of biocide-based coatings is permitted. The boat counts were then used to determine the current emissions of copper and zinc from antifouling paint per CWB (Fig. 4B and C).

The average copper load per CWB on the west coast (121 kg/yr) was found to be 11 times higher compared to the south-east coast (11 kg/yr) (Fig. 4B). As the assumed mooring duration time was the same for both and the WSA also similar (Table 1), the coastal difference must be explained by one or more of the other three variables of the load calculation, i.e. number of boats, application rate and/or copper release rate (Eq. (1)). Even though the boat park is 30% smaller on the west coast (Table 2), the application rate is, on the other hand, 1.5 times higher, resulting in similar painted hull surface areas on the west (9421 km^2) and south-east (9104 km^2) coasts. The main differentiating variable is thus the copper release rate from the hull paint. The modeled release rate of copper was on average 5 times higher per CWB on the west coast (13.3 µg/cm^2/day) compared to the south-east coast (2.9 µg/cm^2/day). West coast paints have a higher average release rate of copper compared to east coast paints across the whole salinity range (Fig. 2). In addition, the difference in slope values of the release rate models (0.45 for west vs 0.15 for east coast paints), indicates that salinity increases copper release more strongly from west coast paints than east coast paints. This could perhaps be due to their higher content of seawater-soluble cuprous oxide pigments making them overall more soluble (Yebra et al., 2006). Hence, it is the combination of paint choice (west coast paint that contains a high weight percentage of copper) and the paint's interaction with local water parameters (salinity) that results in the higher copper loads observed here. If west coast boaters had instead
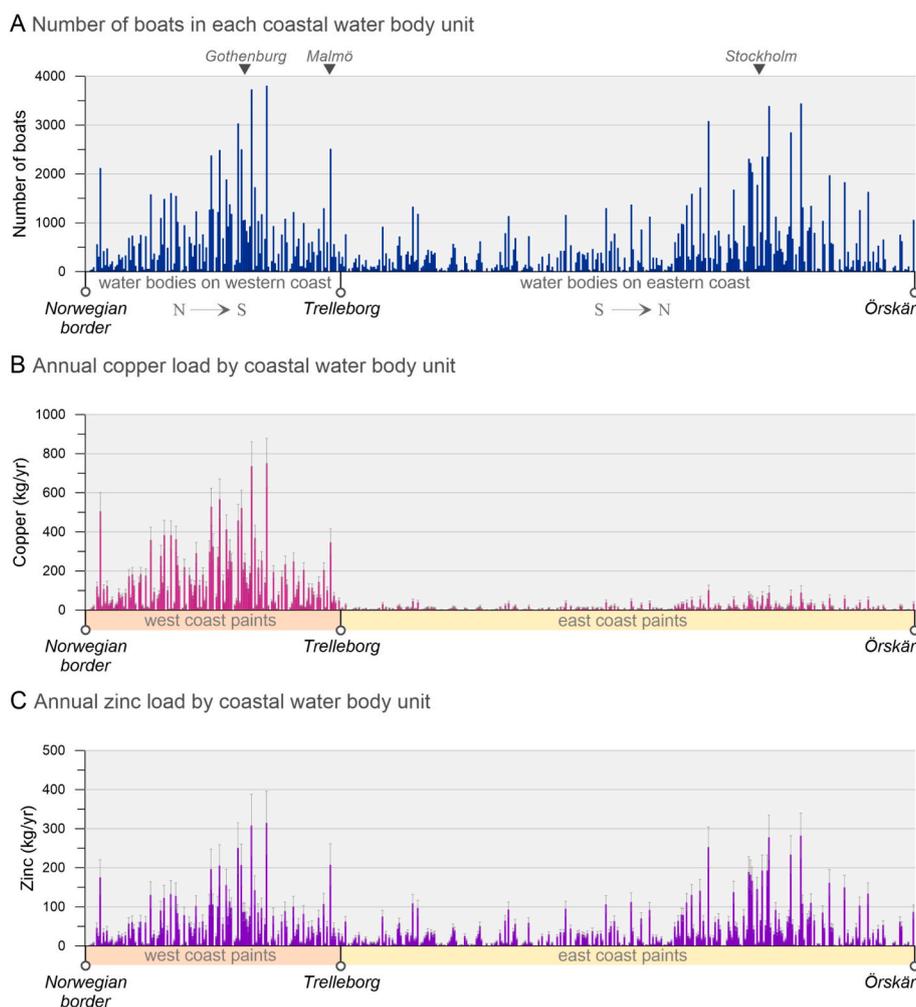
Fig. 4. Results per coastal water body units (each bar represents a CWB) where the use of biocidal antifouling paints is permitted, arranged from West to East along the Swedish coastline, showing (A) the estimated number of boats, (B) the annual load of copper (in kg/yr) and (C) the annual load of zinc (in kg/yr). The error bars represent the propagated 95% confidence interval of the release rate prediction model. The type of paint used to determine the loads of copper (A) and zinc (B) is displayed on the x-axis.

been assumed to use east coast paints, the average copper release rate per CWB would be substantially lower at 5.3 µg/cm^2/day compared to the current rate 13.3 µg/cm^2/day. For zinc, the difference between the two coasts is smaller (Fig. 4C). Given that the release rate of zinc was found to not be salinity-dependent and of similar average value for both types of paint (Fig. 2), the loads mirror the number of boats per CWB.

The differences between the west and south-east coasts are even more visible when the loads for these two regions are summed. Fig. 5 shows that 83% of the total yearly copper emission from antifouling paint on leisure boats (i.e. 18.3 tons), occurs on the Swedish west coast. For zinc, the load (total of 17.1 tons/yr) is more equally divided between the west (41%) and south-east (59%) coasts.

On a national scale, the average yearly estimated emissions of copper in this study (± 95% C.I.) sum up to 22.1 ± 4.4 tons/yr. This number compares well with national statistics of sold quantities of pesticides: an average of 28.1 tons/yr of copper were sold to households in the form of antifouling products over the time period (2015–2023) of the studies included in the release rate prediction model (Swedish Chemicals Agency, 2024). A lower estimate for the load of copper to coastal waters in comparison to sold quantities is to be expected as all the copper in the paint film will not necessarily be released to the water. According to EU's environmental risk assessment, for example, 90% of the copper in the paint film is assumed to be released over a coating's lifetime (ECHA, 2017; ISO, 2010). Other studies have predicted the total loads of copper from Swedish leisure boats to 15.9 tons/yr (Johansson et al., 2020) and 4.8 tons/yr (Comber et al., 2023). Hence, both are underestimations in comparison to both the current study and the national sales statistics, but the predicted load by Comber et al. (2023) is particularly low. In that study, estimates of all copper sources to European waters were derived, including leisure boats. Based on an assumed boating season length of 270 days, an estimated European (EU27) leisure boat park of 622,000 boats in total and an average copper release rate of 6 µg/cm^2/day, the load was calculated to 262 tons/yr from all European boats. National contributions to the total European boat park were then apportioned based on population. As a result, Sweden was estimated to host only 11,448 boats, i.e. 95% less than the number of boats mapped in this study, which explains the gross underestimation on national level. The total estimated 262 tons/yr from the EU27 leisure boat park may also be underestimated: given that the current study has found Sweden to alone host nearly a quarter of a million boats, the European boat park likely exceeds 622,000 boats. Additionally, an average release rate of 6 µg/cm^2/day appears low in comparison to those measured from commercial coatings (Fig. 2).

3.2.2. Loads of alternative scenarios

Assuming all mapped boats were coated with paints with release rates corresponding to the maximum permittable according to the previous (Swedish national, scenario 1) and current harmonized (EU, scenario 2) ERA, 'worst-case' loads of copper to Swedish coastal waters were derived on national and regional scales. The results show that a transition to the new ERA would mark a slight reduction in theoretical maximum loads from 8.8 to 8.1 tons/yr on a national level (Fig. 5). Notably, the current load of 22.1 tons/yr greatly exceeds both. The reason for the exceedance of the previous national ERA (scenario 1) under which the currently available products were placed on the market has been discussed in previous work and is due to underestimated copper release rates having been used for the ERA, resulting in erroneous product approvals (Lagerström et al., 2018). Consequently, the use of the products has led to exceedance of environmental quality standards in coastal marinas (Lagerström et al., 2020a). If new products are assessed henceforth based on copper release rates reflecting those of use, it would be possible to greatly reduce the input of copper to the marine environment. The maximum load with the new ERA was calculated based on the highest allowable copper release rates for products not emitting any other biocide(s) and/or substance(s) of concern. As zinc oxide is commonly included in most paint formulations which results in their release of the zinc, a substance of concern, there is potential for an even greater load reduction.

In comparison to the previous national ERA, the new harmonized EU approach restricts release rates of copper from coatings on the Swedish west coast (i.e. in the Baltic Transition), while allowing higher release rates on the south-east coast (i.e. in the Baltic) (Table 1). This is reflected in the loads on regional level, as the maximum allowable loads are lower for the west coast and slightly higher for the south-east coast. Implementing the new ERA could reduce the total copper load to the maximum allowable limit of 8.0 tons per year, resulting in a significant national reduction of 14.0 tons per year (or 64%). However, this reduction would not be uniformly distributed across the two regions concerned. The bulk of the decrease (96%) would occur on the west coast.
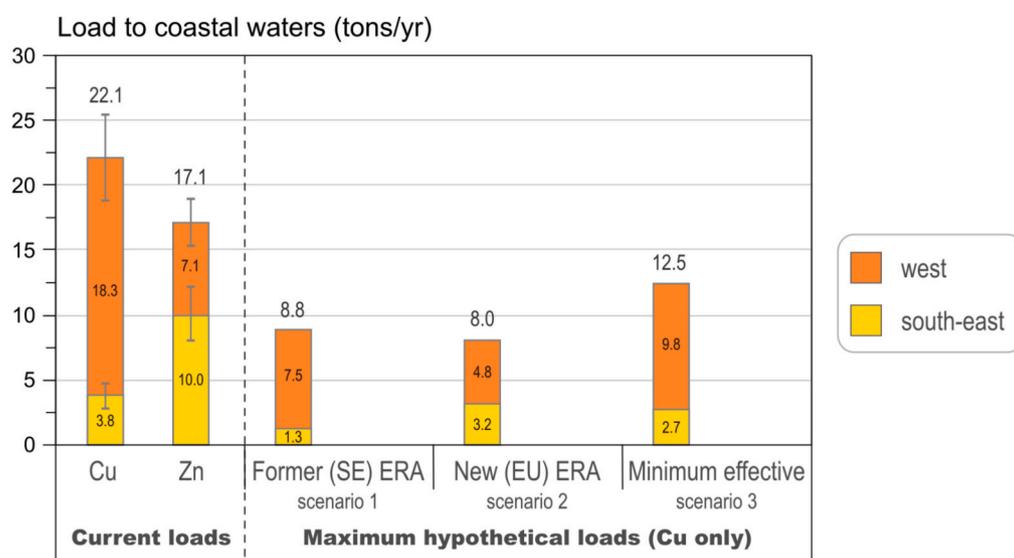

Fig. 5. Estimated average annual loads (in tons/year) of copper and zinc from antifouling paint on leisure boats to the Swedish coastal waters as a whole and per region (west or south-east coast). The 95% confidence interval from the release rate prediction model was propagated for the current loads and is represented by the error bars. For copper, both current and hypothetical loads (in tons/year) according to the evaluated scenarios are displayed. Note that the loads for scenarios 1 and 2 would be the maximum allowable only if the coatings do not release any biocide(s) other than copper or any substance(s) of concern.

The result from scenario 3 (Fig. 5) shows that if the copper paints used on Swedish leisure boats emitted only the minimum necessary dose of copper to prevent macrofouling, it would result in a total load of 12.5 tons/yr. Hence, nearly half (43% or 9.6 tons/yr) of the current amounts of copper may be emitted in excess. The redundancy is largest on the west coast, with emissions roughly twice the effective amount. Notably, even if the loads of copper from antifouling paints could be greatly reduced compared to current levels by a downward adjustment in dose to the minimum effective one, they would nonetheless exceed those of the maximum allowable to pass the ERA (scenario 2) on the west coast. This is because the minimum effective release rate exceeds that of the maximum permittable ones (Table 1). Hence, if implemented correctly, the new ERA would reduce the load of copper greatly to this stretch of the coast, but products may then not be able to completely deter all fouling. For west coast boaters, transitioning to physical prevention methods such as intermittent hull cleaning, using a hull cover or storing the boat out of water between uses could be alternatives. A viable option is also to switch to a biocide-free silicone-based coating, as these have been shown to be equally or more effective than copper-based antifouling coatings in long-term field tests at two locations on the Swedish west coast (Lagerström et al., 2022).

3.3. Importance relative other sources

The importance of copper and zinc emissions from antifouling paint on leisure boats to Swedish CWB relative other sources was assessed by compiling annual loads from ships, point sources, atmospheric deposition and rivers (compiled loads can be found in Table S4) (Fig. 6). The majority of the calculated copper and zinc loads from all vessels, both leisure boats and shipping, occur as emissions from antifouling coatings. As such, the compiled loads show that this single product category is a considerable source of copper and zinc to Swedish coastal waters.

For copper, leisure boating and shipping account for 32% (or 70 t/yr) of the total load on a national level, only surpassed by rivers (59% or 128 t/yr) (Fig. 6A). The bulk of the emission from ships and leisure boats occurs on the south-east and west coasts, while their contribution to the annual load of the north-east coast is smaller (only 6% of the total input). Due to the ban on the use of biocide-containing antifouling coatings on leisure boats in this region, their assumed load was set to zero. As for ships, icy winter conditions of the northern Baltic Sea promote the use of biocide-free hard coating over copper-based ones that are susceptible to ice damage (Finnish Environment Institute, 2003; Jalkanen et al., 2021), resulting in a load of just 5.8 t/yr. In the north-east, riverine input is instead the dominant source, with the majority of Swedish rivers discharging along this stretch of the coast.

While the emissions from leisure boats, ships and point sources are purely anthropogenic, the atmospheric and riverine inputs may be of either natural or anthropogenic origin. The chemical speciation of copper emissions may therefore differ depending on source nature. Riverine inputs of copper will mainly be bound to organic matter, whereas the releases of copper from antifouling coatings occur in ion form, which is the most toxic to marine organisms (Campbell, 1995). As a result, the fraction of dissolved bioavailable copper in marina waters can double during the boating season (Lagerström et al., 2020a).

For zinc, the total annual loads are higher, with riverine discharge constituting the single largest source on both national and regional levels (Fig. 6B). Similar to copper, the loads from ships and leisure boats are more prominent at the south-east and west coasts, but they account for only 16% and 11% of annual inputs, respectively. However, these figures are comparable to those from point sources which constitute 21 and 12% of the respective regional loads. This is noteworthy as emissions from point sources are more strictly regulated than those from antifouling coatings on leisure boats and ships, as discussed in the next section.
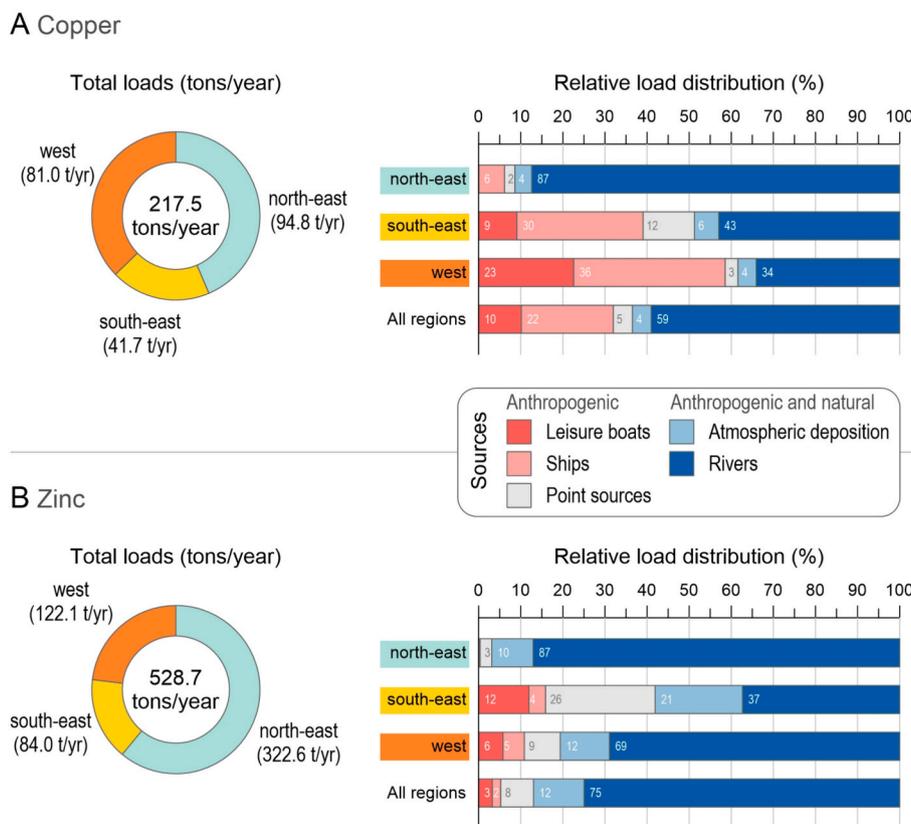

Fig. 6. Loads of copper (A) and zinc (B) to Swedish coastal water bodies. Pie charts (left) show the total annual loads (in tons/year) for the three respective regions while bar graphs (right) show the distribution of the loads (in %) between sources.

While the copper and zinc loads from ships are indeed greater than those from leisure boats on both regional and national level, it is worthwhile to consider that all emissions from boats occur during the boating season, which is only around 5–6 months long. Hence, leisure boats likely constitute a greater source of these metals in comparison to shipping during summer months (Johansson et al., 2020). The two sources can also be compared on a local level. In the two regions where the use of biocidal antifouling paint is permitted, annual inputs from leisure boats exceed those of ships in 45% of CWBs for copper) and 74% for zinc. Leisure boats are also more likely to moor and navigate in more pristine and shallower areas. The emissions from leisure boats are thus, at the current load levels, far from negligible, highlighting the importance of effective regulatory measures.

3.4. Regulatory challenges

All biocide-containing antifouling products must be approved at the national level or across multiple national levels before they can be placed on the market. If a product meets the efficacy and ERA criteria under the BPR, it fulfills the necessary requirements for sale (European Parliament and Council, 2012). Hence, ensuring that representative ERA are conducted for all biocidal antifouling coatings is important. It is therefore unfortunate that the effect of salinity on biocide release rates is not considered by the newly implemented EU ERA. Even though the salinity of marina scenarios of the new EU ERA (see Section 2.2.2) varies greatly between and within its four defined marine regions (Baltic: 6 ± 2.1; Baltic Transition: 17 ± 8.3; Atlantic: 31 ± 5.7; Mediterranean: 38 ± 0.9 PSU), a single release rate is entered into the ERA tool to model PECs for all marine waters, ignoring regional salinity differences that significantly affect copper leaching (Shan-I et al., 2013).

Improvements to the ERA are especially crucial to ensure that products approved under the BPR do not prevent the achievement of water quality targets on local and regional levels. Under the Water Framework Directive (2000/60/EC) and the Marine Strategy Framework Directive (2008/56/EC), member states are required to implement measures to improve the environmental status of water bodies. On a local scale, many municipalities in Sweden therefore mandate boat owners to collect and treat wastewater produced during pressure washing, an activity typically conducted in the fall on land near marina waters. While this measure may greatly reduce inputs of biocides to boatyard soils, it will not effectively decrease emissions into coastal waters (Ytreberg, 2012). Additionally, as it is not feasible to prohibit boaters from using legally approved biocidal products, local authorities may instead attempt to appeal to boat owners to make more sustainable choices, but the effectiveness of this strategy may be limited.

A possible means of restriction for leisure boats may be through the Baltic Sea Action Plan (BSAP), adopted by the HELCOM Contracting Parties in 2007 and updated in 2021. The BSAP is HELCOM's strategic program of measures and actions for achieving good environmental status of the sea, ultimately leading to a healthy Baltic Sea. Only one of the 24 assessed sea basins in the Baltic Sea achieved Good Status in the evaluation of copper (HELCOM, 2023). To significantly reduce copper loads in the Baltic region, stricter regulations on antifouling paints is necessary for all vessels. One action in the BSAP is HL30, where the contracting parties aim to minimize the release of biocides from antifouling products to the marine environment and, preferably by 2027, replace the use of biocidal antifouling products with biocide-free alternatives on structures, equipment, and recreational craft. How this action can be implemented without conflicting with the BPR remains unclear.

For commercial ships, the challenge to regulate biocide emissions is even greater. Even though international regulations exist under the International Maritime Organization (IMO), these only prohibit the use of certain biocides (organotin substances and cybutryne) and do not pose any limits on emission quantities (IMO, 2001). Within the EU, the BPR also requires coatings for vessels >25 m to pass an ERA, but the considered modelling scenario (OECD commercial harbour) and

endpoint (freely dissolved concentrations in the 'surrounding area' outside the harbour) makes it ineffective. A product with a theoretical release rate of 8000 µg/cm^2/day could currently pass the ERA (ECHA, 2017). Such an emission would entail a daily load of 8 tons of copper, i.e. 2920 tons per year, inside the ERA harbour. This exceeds the total estimated load of copper to all Swedish coastal waters by a factor of 10.

In the EU, emissions from various industries are regulated through Directive 2010/75/EU, which includes the establishment of best available techniques (BAT) as a key factor in reducing emissions. For instance, the Commission Implementing Decision (EU) 2016/902 sets BAT-associated emission levels (BAT-AEL) for wastewater in the chemical sector, specifying that copper emissions should be between 5 and 50 µg/L. The BAT-AEL is applicable if emissions exceed 5.0 kg/yr. Consequently, this regulation mandates monitoring and reporting, enabling regulators to assess overall emissions from this sector, in contrast to the shipping sector.

4. Conclusions

Leisure boats contribute significant copper (22 t/yr) and zinc (17 t/yr) loads to Swedish coastal waters, with 83% of emissions concentrated on the west coast due to salinity and paint type. Although the load modelling has limitations, such as assuming 100% of metal release occurs within the mooring's coastal water body and using representative average release rates due to lack of paint sales statistics, its result agreed well with national statistics of sold quantities of copper, supporting its robustness. Furthermore, even though the load compilation does not account for chemical fate or bioavailability, which are ultimately needed to assess ecological risk, the results can nonetheless be used to identify water bodies particularly at risk and guide targeted mitigation measures.

Current copper loads were found to exceed regulatory limits by nearly three-fold, indicating shortcomings in previous environmental risk assessments. While ships emit more copper overall, leisure boats dominate in many local water bodies, making them a critical source for management. Implementing the EU-harmonized ERA could reduce copper emissions from boats by over 60%, primarily on the west coast, but success depends on accurate release rate data and strict enforcement. The use of release rates reflecting those occurring during use is thus key to ensure effective regulatory enforcement and to achieve reductions in copper inputs to coastal ecosystems.

CRediT authorship contribution statement

Maria Lagerström: Writing – review & editing, Writing – original draft, Visualization, Supervision, Resources, Methodology, Investigation, Formal analysis, Conceptualization. **Ann-Kristin E. Wiklund:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization. **Anna Lunde Hermansson:** Writing – review & editing, Methodology, Data curation. **Jukka-Pekka Jalkanen:** Writing – review & editing, Resources. **Erik Ytreberg:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization.

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.

the bottom" (reference number 2021-02027).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2026.119312.

Data availability

All raw data can be found in the supplmentary material file. Authors are however not allowed to share the GIS map layers.

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