Conditions that influence Good Environmental Status (GES) in the Baltic Sea

Contents

Preface

The main aim of WP5 was to review the current knowledge and analyze how natural conditions influence the recovery of the Baltic Sea and how the projected future change in climate will affect the measures taken to improve the Baltic Sea. In this report, the outcomes of several activities in the HELCOM ACTION project WP5 are presented. It focuses on natural conditions that influence achieving of GES in regard to hazardous substances (Section 3) and biodiversity (Section 4). The eutrophication topic is described in a separate report. Task 5a outcomes have been presented at several HELCOM meetings, and this report summarizes the findings (Chapter 2). An evaluation of the impacts of projected climate change on achieving GES due to secondary inputs and/or internal processes are also presented (Section 5). References are listed at the end of each Sub-Section. Conclusions are given in the form of summarizing tables in Section 6. This report was prepared by Natalja Kolesova, Germo Väli and Urmas Lips (Tallinn University of Technology) with support of other WP5 partners.

1. Introduction

The Baltic Sea is a unique brackish water ecosystem. The Baltic Sea coastal countries have committed to protect it from all sources of pollution from land, air and sea and to take measures on conserving its habitats and biological diversity. A major joint programme for this is the HELCOM Baltic Sea Action Plan (BSAP) adopted in 2007. It has an ambitious goal to restore the good ecological status of the Baltic marine environment by 2021. However, several natural conditions are known to slow down the recovery of the Baltic Sea environmental status despite the implemented measures. These include the long residence time of the water and accumulation of nutrients and contaminants in the sediments. In addition, the consequences of the projected climate change in the region could increasingly influence the status of the Baltic Sea in the future. Thus, the natural conditions that influence the recovery of the Baltic Sea have to be considered when analysing the sufficiency of measures.

The HELCOM countries being EU Member States are implementing the EU Marine Strategy Framework Directive (MSFD) with the aim to achieve or maintain good environmental status (GES) of marine waters. When the countries established their respective Programmes of Measures (POM) in 2016, many of them also reported exceptions of not reaching GES by 2020. Most of such exceptions were justified according to Article 14(1)(e) of MSFD by natural conditions which do not allow timely improvement in the status of the marine waters concerned. The analysis of the reported exceptions could contribute to the coherence in approaches to exceptions under MSFD Article 14 by EU Member States in the future, e.g., by picking up the best practices regarding their justifications.

The aim of WP5 in the HELCOM ACTION project is to summarize the current knowledge on natural conditions that influence the recovery of the Baltic Sea to GES. We also aim to describe how the projected future changes in climate could impact the effectiveness of measures taken to improve the Baltic Sea environmental status. The results will be used in the HELCOM ACTION analysis of sufficiency of measures (SOM analysis). The findings of this activity and the report will be communicated to the relevant HELCOM Working Groups (GEAR and STATE & CONSERVATION).

The first step in the analysis was collecting information on reasons, analyses and results which were used to justify exceptions reported by the HELCOM countries being EU Member States under MSFD POM in 2016. We listed the reported exceptions by the Baltic Sea countries and tried to identify common features and aspects most widely perceived as causing the failure to meet GES. This analysis enabled us to identify examples of best practices to justify exceptions and provided suggestions on how to elaborate them further. The results were presented to the HELCOM STATE & CONSERVATION and GEAR groups.

The second step in the analysis was the review of scientific literature and recent project outcomes, including scenario simulations, to identify gaps or delays in achieving GES due to natural conditions. We focused on some selected topics under the following GES Descriptors: D1 – Biodiversity, D5 – Eutrophication, and D8 –

Contaminants. The activity involved also an evaluation of the impacts of projected climate change on the effectiveness of measures taken to improve the Baltic Sea.

Below are the examples of criteria, features, and processes that were considered for the analyses.

Eutrophication

Decades with high nutrient loadings have changed the Baltic Sea. Particularly with respect to nutrient pools. Large amounts of nitrogen and phosphorus is now bound in the sediments and present as dissolved organic matter in the water column. Despite input reductions these internal pools, particularly in anoxic areas (likely exacerbated by climate change), release inorganic nutrients (internal loading) and prevent GES from being reached. Furthermore, observed and predicted temperature increase could counteract the nutrient reduction by intensifying cyanobacterial blooms and consequent nitrogen fixation (acting as additional nitrogen source). Coastal zone acts as a filter between land-based nutrient loadings and the open sea, transforming inorganic nutrients into organic bound nutrients, which implies that the effectiveness of load reductions for open sea GES varies dependent on the types of inputs (e.g. inorganic, organic bound, river born, atmospheric deposition). The recovery of the Baltic Sea and the time delay that can be expected between load reduction and signs of improved GES were analysed in relation to the following MSFD criteria: D5C1 (nutrient concentrations), D5C2 (chlorophyll a), D5C4 (water transparency), D5C3 (harmful algal blooms), and D5C5 (dissolved oxygen). The connectivity between sub-basins and with the coastal zone was also addressed and comparisons between GES for the coastal zone within the WFD and GES for the open Baltic Sea were made.

Hazardous substances

The 'State of the Baltic Sea' report (HELCOM, 2018) shows that despite measures, hazardous substances remain a concern. The semi-enclosed character of the Baltic Sea, persistent historical contamination and re-release of historic sediment-deposited contaminants are all potential factors. When estimating by when GES could be achieved, climate change related factors, such as inputs due to elevated run-off, altered bio-geochemical processes or increased potential for bioaccumulation need to be considered. The factors mainly linked to MSFD criterion D8C1 (concentration of hazardous substances) were analysed.

The report is arranged as follows. First, an overview of the exceptions reported by the EU Member States in 2016 is presented. In the following sections, literature reviews regarding natural conditions that influence achieving GES are provided. These cover the selected topics under GES Descriptors D1 – Biodiversity, D5 –

Eutrophication and D8 – Contaminants. A further review deals with the effects of the projected future change in climate on achieving GES. The final section summarizes the analyses results.

Biodiversity

The 'State of the Baltic Sea' report (HELCOM, 2018) has concluded that not achieving GES for biodiversity is partly caused by natural conditions and changes in climatic conditions. For instance, the ringed seal status is critical in its southern distribution range, partly because its breeding is restricted by suitable sea ice conditions. Poor oxygen conditions in deep waters of the Baltic Sea limit benthic fauna distribution and can alter food web productivity, a factor that to some extent is explained by the restricted/sporadic water exchange with the North Sea. The apparent inadequate nutritional condition of grey seals may be explained by the seal population approaching its ecological carrying capacity (natural population plateau), which beside human impacts (e.g., overfishing) could reflect natural ecosystem processes. While Baltic Sea salinity and temperature regimes may limit the spread and establishment of some non-indigenous species, the predicted changes in climate could weaken such barriers. These natural conditions and possible future changes were analysed to reveal their impacts on achieving GES regarding the following MSFD criteria: D1C2 (species abundance), D1C4 (species distributional range), D2C1 (newly-introduced non-indigenous species), D2C2 (abundance and spatial distribution of established non-indigenous species), and D6C3 (spatial extent of habitats).

2. Exceptions reported in 2016

While reporting on the Programme of Measures (POM) in 2016, all Member States except Germany reported exceptions of not achieving GES by 2020 (Table 2.1). The highest number of countries indicated the exception regarding D5 – eutrophication. Also, most of countries marked D8 – contaminants (and/or D9 – contaminants in seafood) as an exception of not achieving GES. The GES Descriptors related to species and habitats, as D1 – biodiversity, D2 – non-indigenous species, D3 – commercial fish, D4 – food webs, and D6 – seabed integrity, were mentioned as well. No exceptions were reported on GES Descriptors D7 – changes in hydrographic conditions, D10 – litter, and D11 – underwater noise, most probably since the GES thresholds were not defined yet for those descriptors.

Table 2.1. Summary of reported exceptions by the Member States when reporting on POM in 2016. Summary is given by MSFD Descriptors.

Country	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	D11
Denmark					X						
Estonia	X			X	X			X			
Finland			X		X				X		
Germany											
Latvia					X						
Lithuania	X	X		X	X			X	X		
Poland	X	X		X	X	X		X			
Sweden					X			X			

Among main reasons for not achieving GES regarding D5 – eutrophication were the natural specificities of the Baltic Sea – very closed marine area, limited water exchange with the North Sea and accumulation of nutrients in the seabed sediments over the past decades. Also, the time-lags due to retention of nutrients in the drainage area were mentioned. However, the major reason to consider was the internal load of nutrients. The historically-enriched sediments may continue to be a net source of nutrients for decades after nutrient loads have been sufficiently reduced. Climate change could expand the extent of oxygen-deficient areas in the deeper basins of the Baltic Sea, possibly leading to the release of nutrients from sediments and increased levels of dissolved nutrients in the water column.

Regarding D8,9 – contaminants, the countries reported that persistent pollutants are still found in high concentrations in sediments and biota. It could be linked to the past pollution events and the specific natural conditions of the Baltic Sea as it is a very closed basin and has limited water exchange with the ocean. The contaminants are retained in the drainage area and accumulated in seabed sediments. Some contaminants are persistent and take a long time to break down. Also, inputs of contaminants from atmospheric deposition have to be considered since these are transboundary in nature and not controlled by regional actions only. The time-lags between the measures and achieving GES depend on the substance, but the estimates of time-lags were not provided. The substances mentioned were dioxins, polychlorinated biphenyls and heavy metals (e.g., Hg).

Exceptions of achieving GES regarding the GES Descriptors related to species and habitats are reported by some countries. These were justified by climatic conditions that define the suitable habitats, e.g., the extent of ice cover is crucial factor determining the distributional range and abundance of ringed seals. Since the slow growth rate of benthic organisms and low natural oxygen levels, the recovery of benthic habitats takes time. Also, there will be a time-lag between the measures and the achievement of sustainable stock status of fish species, such as sea trout, pike, perch and other migratory whitefish species.

These examples of exceptions were taken into account when selecting the topics and GES criteria to be analyzed in this activity. As a rule, the time-lags were not given when the countries reported and justified the exceptions. In this report, we try to give some time-lag estimates for selected topics/substances based on scientific literature. Also, the future climate change and its impact was not considered by most of the countries in the exception justifications. However, it is a factor which could impact the time-lags, especially in regard to eutrophication and biodiversity. This topic will be discussed in a section below.

3. Hazardous substances

We focused on selected substances or groups of substances that have the major concern (see HELCOM assessment results – HELCOM, 2018; Fig. 3.1) and could potentially experience a noticeable time-lag between the measures taken and status improvement. The substances are heavy metals (Hg, Cd, and Pb), TBT and organotins, pharmaceuticals, PFOS and other perfluorinated alkyl substances (PFASs). Additional information on the sources, pathways, retention and time lags for these substance as well as dioxins, PCBs and PBDEs was obtained from the materials collected by the topic team on hazardous substances of the HELCOM platform on Sufficiency of Measures (SOM platform).

The analysis is based on a literature review. Shortly, the sources of contaminants and their pathways in the marine environment are listed and characterised. We consider the accumulation and/or persistence of substances and identify what the activities or processes leading to secondary inputs are. In all steps, the natural conditions of the Baltic Sea that influence the persistence of substances and could cause secondary inputs are described. Finally, the estimates of time-lags are given.

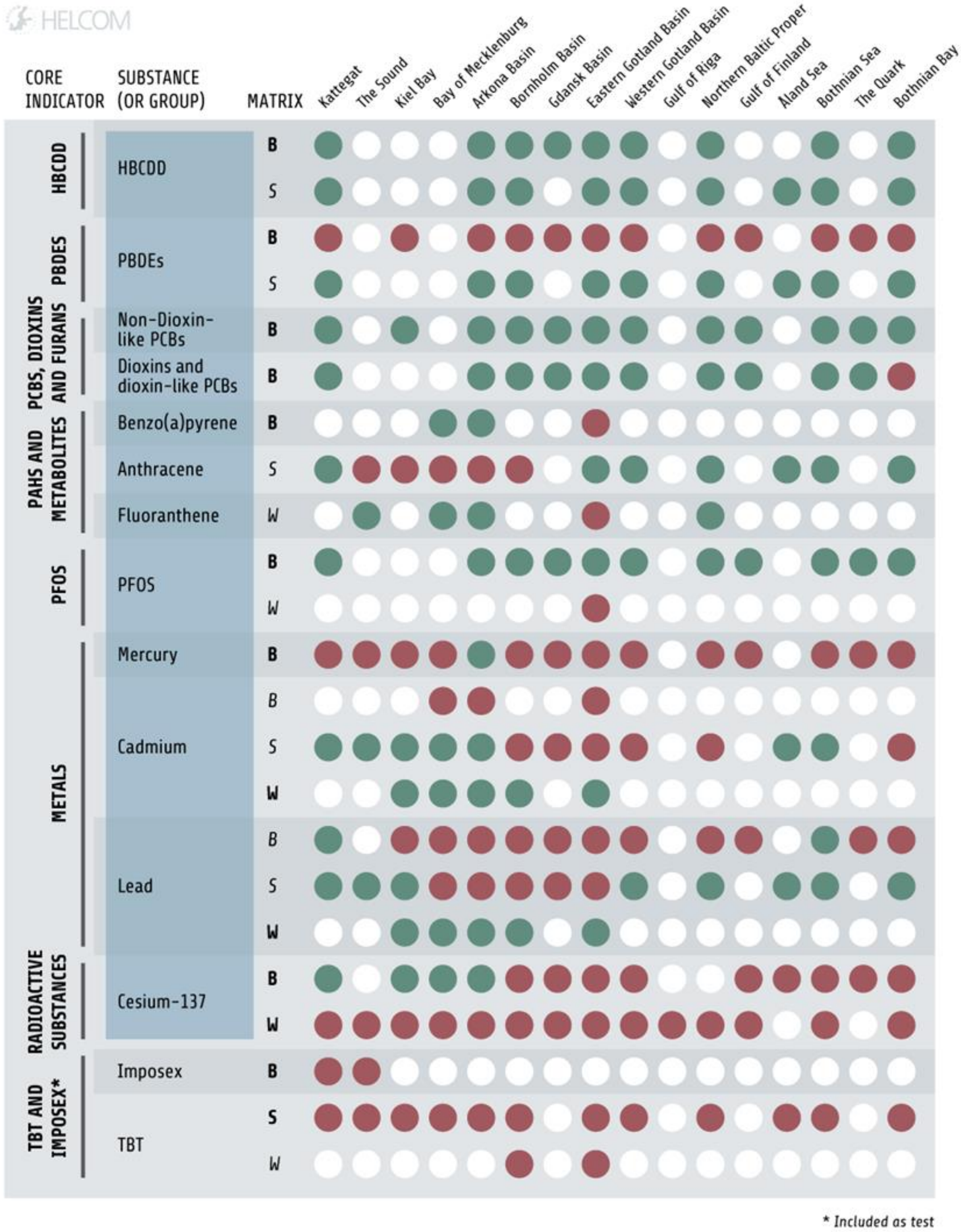

Figure 3.1. Detailed results for the HELCOM hazardous substances assessment in the open sea assessment units, by core indicators and substances (HELCOM; 2018).

3.1. Heavy metals

Mercury (Hg), lead (Pb) and cadmium (Cd) are very toxic even at relatively low concentrations (Komárek et al., 2008; HELCOM, 2010; Zaborska, 2014; Carolin et al., 2017). Metals are bioaccumulated by marine organisms, and Cd and Hg are known to biomagnify, i.e. the concentration levels increase upwards through the food chain (HELCOM, 2018).

The Baltic Sea is vulnerable to the accumulation of pollutants, as it is an inland sea, where water exchange with the North Sea is very limited, resulting in a mean seawater residence time of 20–30 years. The Baltic Sea is also characterised by the low water depth and a large inflow of fresh water from the surrounding highly urbanised areas with large populations (Schneider et al., 2000; Wängberg et al., 2001; HELCOM, 2010; Jędruch et al., 2017). The large catchment area (the surface of the Baltic Sea is four times smaller than the surface of its drainage area) of the Baltic Sea leads to the contamination of the reservoir by hazardous substances (HELCOM, 2010; Jędruch et al., 2017; Saniewska, 2019a).

The activities as sources of hazardous substances are traffic, shipping, energy production, incineration of wastes, and even small-scale household combustion (HELCOM, 2010). A major current source of heavy metals (including mercury, cadmium and lead) is the burning of fossil fuels, leading to atmospheric deposition (HELCOM, 2018). Thus, the metals in the environment come mainly from the production (mining and metal smelting), use of coal-fired power plants and combustion of fossil fuels (coal, former use of leaded gasoline), the fertilizer industry, recycling, use of products (batteries, plastics, ceramics), disposal, sewage sludge application, etc. (Rieuwerts et al., 1999; Ahlberg et al., 2006; Komárek et al., 2008; HELCOM, 2010; Zaborska, 2014; Carolin et al., 2017; Consani et al., 2019).

The main pathways of mercury, cadmium and lead from highly industrialized and densely populated areas to the Baltic Sea are the atmospheric deposition and riverine input (Schneider et al., 2000; Knuuttila, 2009; Zaborska, 2014; Bełdowska et al., 2015; Senze et al., 2015; Jędruch et al., 2017; HELCOM, 2010, 2018; Remeikaitė-Nikienėa et al., 2018).

The total annual input of mercury to the Baltic Sea during the period 2012-2014 has been estimated in a range of 4.8 – 5.6 tonnes, taking into account atmospheric deposition, riverine input and direct point sources (HELCOM, 2018). The annual mercury deposition from atmosphere in the period 2012-2014 constituted 3.1 – 3.7 tonnes per year (HELCOM, 2018). In 2016, the contribution was estimated at about 3.0 tonnes (Gusev, 2018a). It made around 70% of total Hg input to the Baltic Sea (HELCOM, 2018). It was estimated that in 2016 anthropogenic sources of HELCOM countries contributed to the mercury deposition over the Baltic Sea about 13% that constituted 0.38 tonnes per year.

According to Baltic Sea Environment Fact Sheet (Gusev, 2018b), annual atmospheric emission of mercury from anthropogenic sources of HELCOM countries to the atmosphere have decreased by 45% during the period 1990 – 2016. The most significant decrease of mercury emissions was in Denmark and Sweden, 90% and 73%, respectively. Higher emission for mercury in 2016 comparing to 1990 was estimated for Russia by 12%.

The total annual input of Cd and Pb to the Baltic Sea during the period 2012-2014 has been estimated in ranges of 23 – 45 and 443 - 565 tonnes, respectively. It is reported by HELCOM that the riverine input of Pb and Cd makes up of the total inputs around 64% and 79%, respectively (HELCOM, 2018). In 2015, HELCOM countries emissions of lead and cadmium contributed about 29% and 36% to the total annual lead and cadmium deposition over the Baltic Sea (Bartnicki et al., 2017). According to Baltic Sea Environment Fact Sheet (Gusev, 2018b), annual atmospheric emission of cadmium from anthropogenic sources of HELCOM countries to the atmosphere have decreased by 37% during the period 1990 – 2016. During the same period the annual deposition of cadmium from atmosphere to the Baltic Sea has decreased by 61%. The most significant decrease of cadmium emissions was in Finland and Estonia, 85% and 82%, respectively.

Substantial amounts of heavy metals are discharged from the catchment area to the Baltic Sea through rivers. The processes occurring in the mixing area (estuary, coastal zone) have an important role on the behaviour and fate of pollutants (Elbaz-Poulichet et al., 2001; Dang et al., 2015; Oursel et al., 2014). Although the riverine input of Hg to the Baltic Sea in the period 2012-2014 made around 30% of the total input (HELCOM, 2018), the mercury transport from the river systems can represent the highest share of the total mercury input to the coastal waters (Gworek et al., 2016; Bełdowska et al., 2015; Saniewska, 2019a). For instance, in the Gulf of Gdansk the Hg load through atmospheric deposition can be seven times smaller than the input through rivers (Saniewska, 2019a).

Riverine inputs of Cd and Pb to the Baltic Sea exceeded the atmospheric deposition in the period 2012-2014, and were in range 15-39 and 264-386 tonnes, respectively (HELCOM, 2018). Loads of metals from direct point sources were negligible.

The Hg load transported through rivers to the sea are influenced by the amount of precipitation (Bełdowska et al., 2014; Saniewska et al., 2014c, 2018; Saniewska, 2019a), snow melting (Gębka et al., 2019) river flow (Lawson et al., 2001; Saniewska et al., 2014a, 2018; Saniewska, 2019a; Bełdowska et al., 2014) and retention capacity of the catchment area (Lawson et al., 2001; Babiarz et al. 2012; Bełdowska et al., 2014; Saniewska et al., 2014a, 2018; Saniewska, 2019a).

From atmospheric wet and dry deposition Hg retains in the surface soil (Obrist et al., 2011; Babiarz et al. 2012; Bełdowska et al., 2014; Saniewska et al., 2019b, Gębka et al., 2019). Saniewska et al. (2019b) described that content of highly mobile and toxic water-soluble mercury in soil is small in relation to total mercury, and only a small Hg fraction can be washed out to the aquatic environment from natural soil. Saniewska et al. (2019b) concluded that importance of Hg remobilization from soil as a source of mercury in the coastal zone is increasing as anthropogenic emissions become limited.

Pb isotopic studies proved that increased Pb concentrations in soil can originate not only from anthropogenic sources, but from natural processes as well (Komárek et al., 2008).

Thus, river catchments are reservoirs of heavy metals accumulated there over a long period, and soils of the catchment area are a significant source of them in the aquatic environment. (Driscoll, 1998; Komárek et al., 2007; Komárek et al., 2008; Eckley and Branfireun, 2008; Babiarz et al., 2012; Mason et al., 2012; Saniewska et al., 2018; 2019b; Gębka et al., 2019).

Replacement of the natural surfaces by impervious surfaces due to urbanization causes the disturbance of natural water circulation and leads to noticeable faster surface runoff in the urban catchment areas compared to the natural area (during precipitation) (Nawrot et al., 2019). Increasing intensity and duration of precipitation, storm and flooding events may significantly increase input of heavy metals to streams and rivers (Nicolau et al., 2012; Oursel et al., 2014; Nawrot et al., 2019). During flood events, significant amounts of trace metals are rapidly brought to surface aquatic system through runoff processes (Nicolau et al., 2012; Oursel et al., 2014).

Nawrot et al. (2019) observed the increase of the concentration of metals in sediments deposited in retention tanks along Oliwa Stream in Gdańsk, Poland after flood event for lead by 1.5 to 7.2 times and for cadmium by 1.4 to 11.4 times.

Some authors described that in undisturbed areas with natural soil profile, only 10-30% atmospheric Hg deposited to the catchment outflow from the soil to the river and 70-90% can be retained in the catchment area (Bełdowska et al., 2014; Saniewska et al., 2014a; Saniewska, 2019a). However, up to 65% of mercury can be transported by catchment outflow in areas with anthropogenic disturbance (Mason et al., 1997; Eckley & Branfireun, 2008; Babiarz, 2012; Saniewska, 2019a; Saniewska, 2019b).

Lawson et al. (2001) estimated a high variability in retention of Hg in the catchment with different land use of the most of six investigated rivers in US. In the areas where anthropogenic disturbance of the natural soil profile takes place (change in the land use or coverage of the natural surface with impermeable surfaces), Hg retention can be significantly reduced causing greater amounts of the element to be transported to the sea (Mason et al., 1997; Driscoll et al., 1998; Eckley & Branfireun, 2008; Babiarz et al. 2012; Bełdowska, 2015; Saniewska et. al, 2014a, 2018, 2019b; Saniewska, 2019a).

Saniewska et al. (2019b) suggested that rational management of river catchments and riparian zone can to be an important method of limiting the inflow of Hg to the sea from watershed, however more research is needed. In the urbanized and agricultural areas of the Gulf of Gdansk, river water can be two to three times more polluted with mercury than in the forests (Saniewska et al., 2014a; Saniewska, 2019a). Natural areas covered by forests elevate mercury retention that reaches them through atmospheric deposition (Saniewska et. al, 2018). Wetlands and forested watershed composition provide suitable conditions for production of methylmercury (MeHg) (Driscoll et al., 1998; Grigal, 2002; Babiarz et al. 2012).

Skyllberg et al. (2009) found an increase of methylmercury concentrations in draining streams 0-4 years after the forest clear-cut. 1/6 of increased methylmercury was remobilized from soil, while 5/6 was a result of new methylation after harvest. New methylation was caused by an enhanced availability of electron donors for methylating bacteria.

An increase in total annual precipitation as well as an increase in the intensity and frequency of extreme phenomena i.e. storms and floods can lead to the increasing of metals transport to the sea by washing out of Hg from both the atmosphere (wet deposition) and bound with particles from land (river runoff and coastal erosion) (Bełdowska et al., 2016, Saniewska et al., 2014a, 2014b, 2018; Saniewska, 2019a, Kwasigroch et al., 2018).

Gębka et al. (2020) concluded that due to climate change the higher bioavailable mercury flux can be transported to the seas. During high flow, the Hg daily outflow from the catchment could increase over tenfold than during the period with an average flow rate (Saniewska et al., 2014a, 2014b, 2018; Saniewska, 2019a; Bełdowska et al., 2014).

Grigal (2002) described, based on previous studies (Babiarz et al., 1998; Scherbatskoy et al., 1998), that a single summer storm event in an agricultural watershed in Wisconsin transported 20 times more Hg than normal flow and in hardwood watershed in New York, U.S.A., Hg transport from the watershed during high flow events made 60 to 75% of the annual Hg export by particles.

During the Vistula flood event in 2010, 1197 kg of mercury were transported into the Gulf of Gdansk within a month, that made 75% of annual Hg load through the Vistula river. Around 314 kg of dissolved mercury reached the sea (Saniewska et al., 2014b; Saniewska, 2019a).

The importance of the coastal erosion as a source of mercury in the aquatic environment is usually underestimated (Bełdowska et al., 2016; Jędruch,et al., 2017; Kwasigroch et al., 2018; Saniewska, 2019a). It seems to be the third most important source of mercury to the Baltic Sea after atmospheric deposition and riverine input (Bełdowska et al., 2016; Kwasigroch et al., 2018).

Bełdowska et al. (2016) estimated that coastal erosion of cliffs increased the mercury inflow into the Gulf of Gdańsk over 5% and was responsible for about 14.3 kg of mercury annually transported to the sea. The calculated load of mercury into the gulf caused by erosion constituted around 80% (about 18 kg) of the wet deposition (with the precipitation) and was around 50 % (about 9.5 kg) higher than the amount of mercury introduced with dry deposition. Authors suggested that coastal erosion should be included in the Hg cycle in the environment and taken into account while calculating Hg loads and budget.

Kwasigroch et al. (2018) estimated the input of bioavailable mercury originating from coastal erosion that enters the Gdansk Basin, as 10 kg. According to the authors, the load of labile Hg can be enhanced by episodic abrasion events during heavy storms and rains up to 50%.

Metals entering the Baltic Sea through the river discharge are either adsorbed onto suspended particles or in dissolved forms (Yurkovskis and Poikāne, 2008; Remeikaitė-Nikienėa et al., 2018). Once entered the marine system, trace metals are removed from the surface water body by internal fluxes like sedimentation on biogenic or terrigenic particles, by diffusive exchange of dissolved species across interfaces or by advective vertical transport (Pohl et al., 2006).

Zaborska et al. (2019) found in the seawater the highest concentrations of Pb and Cd in the inner part of Puck Bay and close to the Vistula outflow. Results of the latter study revealed that concentrations of Cd and Pb measured recently are higher or not lower than those measured in the 1980s. Authors suggested that concentrations of Pb in seawater are not decreased, despite of the reduced discharge to the environment, because of the importance of secondary Pb sources, e.g., surface run-off and/or sediment resuspension.

In water systems, metals tend to accumulate in sediments in association with organic matter, fine-grained sediments, sulphides and iron-manganese hydroxides and they may be released with changing conditions in sediments, such as changes in pH, dissolved oxygen or temperature (Leivuori et al., 2000; Rigaud et al., 2013; Dang et al., 2015; Remeikaitė-Nikienėa et al., 2018; Consani et al., 2019).

Yurkovskis and Poikāne (2008) concluded that suspended clay minerals appear to be dominant inorganic conveyers for trace metals in the mixing zone. Flocculation of clay colloids, Fe and Mn hydroxyoxides, and also precipitation of Ca carbonates seems to provide supplemental inorganic sorption surfaces.

Particles settling to sediments seabed turn this compartment into an important sink for trace metals but also a potential source of pollution for the surrounding ecosystem through diffusive processes or sediment resuspension events (Rigaud et al., 2013; Oursel et al., 2014; Superville et al., 2014).

Zaborska (2014) measured the highest Pb concentrations in sediments deposited in to Gulf of Gdańsk between 1960s and 1970s. The author studied that in the period of highest Pb contamination, the anthropogenic Pb fraction reached up to 93%. Isotopic composition of sedimentary Pb confirmed that the discharge of Pb originating from leaded gasoline was smaller than the Pb of coal-burning origin.

Studies in the Kiel Bay showed that the highest contamination of sediments by Hg, Cd and Pb occurred in the period 1917-1970. The Pb isotopic analysis showed that Pb in the sediment layers with highest contamination originates mainly from coal burning (Orani et al., 2020).

From the river system, mostly in particle-bound form (Lawson et al., 2001; Grigal, 2002; Skyllberg et al., 2009; Saniewska et al., 2014a; Jędruch et al., 2017; Saniewska et al., 2019b), mercury is transported to the coastal waters and then deposited to the sediment (Gworek et al., 2016; Damrat et al., 2017). A large proportion of Hg introduced into the basin accumulates close to the shore – near the industrialized shore as well as in the area of the river mouths (Jędruch et al., 2017; Saniewska, 2019a).

Since sediments are important sink of metals in the sea (Bełdowski et al., 2009; Rigaud et al., 2013; Saniewska, 2019a), sediments can act as secondary source of pollutants. (Mustajärvi et al., 2017, 2019; Molamohyeddin et al., 2017). Hazardous substances can be released to the water from sediment in several ways: resuspension of sediment particles caused by physical disturbance (Bełdowski et al., 2009), molecular diffusion (Koelmans et al., 2010; Benoit et al., 2009; Gworek et al., 2016), gas ebullition (Yuan et al., 2007), bioturbation/bioirrigation (Benoit et al., 2009; Mustajärvi et al., 2017, 2019).

Bełdowski et al. (2009) estimated annual load of mercury to the studied sediment about in range 3-6 ng/cm2, depending on studied area. Authors calculated that in the Gdańsk Deep and Arkona Deep the return flux of mercury, may constitute for up to 50% of the mercury load deposited yearly in the muddy sediments.

Field and laboratory studies have revealed that activity of benthic organisms (bioturbation) can significantly increase the release of hazardous substances from sediments (Point et al., 2007; Granberg et al., 2008; Hedman et al., 2009; Josefsson et al., 2010; Koelmans & Jonker, 2011; Mustajärvi et al., 2017, 2019; Benoit et al., 2009). The effect of bioturbation depends on both benthic species composition (Mustajärvi et. al, 2019) and density (Benoit et. al, 2009; Mustajärvi et. al, 2019).

Benoit et. al (2009) have observed a significant increase in total methylmercury (MeHg) flux from the sediment compared to molecular diffusion in bioirrigated sediments in Boston Harbor. Results demonstrated that total sediment flux of methylmercury can be up to ten times higher compared to diffusive flux. This study reveals the importance of measuring total fluxes in bioturbated sediments. The same study showed the strong linear relationship between MeHg fluxes and burrow densities that indicates more efficient export of MeHg in sediments with dense infaunal populations than in sediments with lower infaunal densities. The results demonstrate that MeHg flux from the subsurface can be stimulated by macroinfauna, which have a 2-fold effect on MeHg cycling (Benoit et. al, 2009).

Time lag and potential effects of climate change

Secondary contaminant inputs are related to the legacy pollution in the soil of the catchment and sediments. Input from the catchment could stay high for decades due to accumulation in soils and discharges could locally or occasionally increase due to changes in land use, replacement of the natural surfaces by impervious surfaces due to urbanization, increasing intensity and duration of precipitation, storm and flooding events.

Cossa and Tabard (2020) showed that concentrations of mercury in marine mussels in the St. Lawrence Estuary did not change between 1977-1979 and 2016-2019. Regional Hg distribution patterns indicated that rivers constitute a significant Hg source in the estuarine systems. While Hg concentrations in the Atlantic Ocean are known to have decreased in the last decades (e.g., Cossa et al., 2019), the changes in the coastal environments do not seem to be rapid, probably because of the long residence time of Hg in catchment soils (Cossa and Tabard, 2020; Burgess et al., 2013).

Jonsson et al. (2017) demonstrated that increased terrestrial organic matter input to the pelagic zone can enhance the MeHg bioaccumulation in zooplankton by a factor of 2 to 7. They predict that if runoff would increase 15-30% in the future, the MeHg concentration in zooplankton could increase by a factor of 3 to 6 in coastal areas.

The average residence time of mercury in oceanic waters is 20–30 years (Gworek et al. 2016). The Baltic Sea is an inland sea with a water residence time also about 20–30 years. Since hazardous substances can be released to the water from sediment in several ways (resuspension of sediment particles caused by physical disturbance, molecular diffusion, gas ebullition, bioturbation/bioirrigation), natural conditions that influence these processes could cause changes in secondary input of contaminants. Possible changes in the wave climate towards higher wave heights in most of the Baltic Sea regions (Groll et al., 2017) could increase natural resuspension and release of metals (e.g. Hg). Changes in sediment conditions as changes in pH, dissolved oxygen or temperature, either caused by anthropogenic of natural factors, could also influence the release of metals from the sediments.

Concentrations of Hg in the studied Swedish subpopulation of white-tailed eagle (body feathers were analysed) showed a clear decline of 70% from 1967 to 2011, however, the concentrations still remained well above natural background concentrations (Sun et al., 2019).

3.2. PFOS and other PFASs

Per- and polyfluoroalkyl substances (PFASs) are a group of more than 4700 anthropogenic chemicals (OECD/UNEP) that have been widely used since the 1950s in a broad range of consumer products and industrial applications (Prevedouros et al., 2006; Buck et al., 2011; KEMI, 2015; Joerss et al., 2019). PFASs have both hydrophobic and hydrophilic properties and are used as surfactants in various branches of manufacturing, such as in the metal, textile, paper and electrical industries. (Paul et al., 2009; Ahrens, 2011; Buck et al., 2011; KEMI, 2015; Ahrens et al., 2015; Junttila et al., 2019). In addition, firefighting foams are a considerable application of PFASs and source of them in the environment (Ahrens, 2011; KEMI, 2015; Norström, et al., 2015; Xiao, 2017). Relatively high concentrations of PFASs (e.g. PFOA) are found in personal care products as sun creams, body lotions etc. (Fujii et al. 2013; KEMI, 2015).
As a consequence of the widespread use of PFASs and their resulting emissions, a wide range of these substances have been globally found in the environment, wildlife, and humans. (Giesy & Kannan, 2001; Prevedouros et al., 2006; Buck et al., 2011; Scheringer et al., 2014; Heydebreck et al., 2015; Blum et al., 2015; Nguyen et al., 2017; Junttila et al., 2019; Gao et al., 2020).

PFASs are emitted to the environment through industrial processes, product use, military, firefighting operations (KEMI, 2015; Ahrens, 2011; Blum et al., 2015; OECD, 2015; Norström, et al., 2015; UNEP, 2017). PFASs (e. g. PFOS and PFOA) can move from consumer and industrial products into air (Shoeib et al., 2011; Ericson Jogsten et al., 2012; Harrad et al., 2019) and dust (Shoeib et al., 2011; Ericson Jogsten et al., 2012; de la Torre et al., 2019; Harrad et al., 2019), food (Trier et al. 2011; de la Torre et al., 2019), soil (Strynar et al. 2012), ground and surface water and migrate to drinking water (Eschauzier et al. 2012; Blum et al., 2015; Crone et al., 2019). According to Xiao (2017), once released to the natural environment they are not readily decomposable by physical, chemical, and biological mechanisms because of the strong carbon fluorine bond.

In the last decade, legacy long-chain PFAS (PFOS, PFOA, etc.) and fluorinated alternatives (replacement compounds) were found in the rivers (Junttila et al., 2019), coastal waters (Nguyen et al., 2017; Joerss et al., 2019), sediments (Zacs & Bartkevics, 2016; Joerss et al., 2019) and boita (Koponen et al., 2015; Zacs & Bartkevics, 2016; Junttila et al., 2019; Sonne et al., 2019) of the Baltic Sea.

Attention has been drawn to the role of long-chain PFASs, especially perfluoroalkyl carboxylic acids (PFCAs, CnF2n+1COOH, $n \geq 7$, include PFOA), perfluoroalkane sulfonic acids (PFSAs, CnF2n+1SO3H, $n \geq 6$) and their corresponding anions as global contaminants, which have been shown to be more bioaccumulative (Conder et al., 2008; Olsen et al. 2009; Heydebreck et al., 2015), persistent and toxic than their short-chain analogues (Buck et al., 2011; Scheringer et al., 2014; Heydebreck et al., 2015; Joerss et al., 2019). The most closely observed substances are and have been PFOS and PFOA (Buck et al., 2011; Ahrens, 2011; KEMI, 2015). PFOS and PFOA are the two ‘‘long-chain’’ perfluoroalkyl acids most often reported and discussed in the scientific literature (Buck et al., 2011).

Regulations

In 2000, the major manufacturer 3M announced a global phase-out plan to be carried out by 2002 for products derived from perfluorooctane sulfonyl fluoride (PFOSF), including the C6 and C10 homologues (3M, 2000; Buck et al., 2011; Joerss et al., 2019). The phase-out was completed in 2002 for PFOS and in 2008 for PFOA (Danish EPA, 2015). In 2009, PFOS (perfluorooctane sulfonic acid), its salts and PFOSF (perfluorooctane sulfonyl fluoride) were added to Annex B of the Stockholm Convention on Persistent Organic Pollutants (POPs), that restricts use and production of these substances in Europe. (Buck et al., 2011; Ahrens, 2011; SC Secretariat, 2017; Joerss et al., 2019; Nguyen et al., 2017; Junttila et al., 2019). Perfluorooctanoic acid (PFOA), its salts and PFOA-related compounds were added to part I of Annex A to the Stockholm Convention of POPs (persistent organic pollutants) in 2019, implementing elimination on their production and use (SC Secretariat, 2019).

As a result of the phase-out of long-chain PFASs and their precursors, an industrial replacement by shorter-chained analogues (short-chained PFASs) and fluorinated alternatives has been taking place (Scheringer et al., 2014; Junttila et al., 2019; Cousins et al., 2019; Joerss et al., 2019; Gao et al., 2020).

Short-chain PFASs are less toxic and bioaccumulative, but they are still resistant to environmental degradation (Heydebreck et al., 2015). Furthermore, a higher amount of short-chain PFASs is necessary to achieve a comparative level of water and oil repellency (Heydebreck et al., 2015). In addition, these regulations have caused the shift of the production of long-chain PFASs toward less regulated countries in Asia, especially China (Wang et al., 2014; Heydebreck et al., 2015; Joerss et al., 2019). Despite the regulations, PFOS and PFOA have been still detected in the WWTP effluents (Loos et al., 2013; Kibambe et al., 2020) that indicates releasing of these substances from PFAS-containing products (Loos et al., 2013).

Sources

Sources of PFASs in the aquatic environment can be classified as point and nonpoint sources. The surface ocean and atmosphere are reservoirs, while the deep ocean, deep soil and sediment are potential sinks of PFASs (Ahrens, 2011).

WWTP effluents represent important point sources of different PFASs in the marine environment due to insufficient removal of substances (Becker et al., 2008; Ahrens, 2011; Xiao et al., 2012; Wang et al., 2013; Loos et al., 2013; Castiglioni et al., 2015; Dauchy et al., 2017; Gallen et al., 2019; Kibambe et al., 2020; Ji et al., 2020), especially for short-chain PFASs (Kibambe et al., 2020) and when they receive industrial wastes (Xiao et al., 2012; Castiglioni et al., 2015).

According to Loos et al. (2013) the discharge of municipal wastewaters is one of the principal routes of entry of perfluoroalkyl substances such as PFOA and PFOS into the aquatic environment. Several studies report even increasing of some PFAS concentrations during wastewater treatment processes due to the degradation of precursors (Becker et al., 2008; Guo et al., 2010; Xiao et al., 2012; Castiglioni et al., 2015; Eriksson et al., 2017; Gallen et al., 2018; Johansson & Undeman, 2020). Dauchy et al. (2017) concluded that the activated sludge process in WWTP leads to a dramatic increase in mass flows of PFASs and generates unidentified PFASs.

PFOA from WWTP is generally fully discharged into receiving rivers, while PFOS is partially retained in the sewage sludge (Becker et al., 2008; Guo et al., 2010; Castiglioni et al., 2015). However, Kibambe et al. (2020) showed that depending on WWT process design, removal efficiency can achieve 94% for PFOS and 70% for PFOA, when some other short-chain PFASs such PFBA, L-PFHxS, PFDA and PFPeA stayed mainly unremoved. Castiglioni et al. (2015) also explained different removal efficiency of PFAS in different WWTPs by the different secondary treatment. In Sweden, Eriksson et al. (2017) investigated a net mass increase of 83%, 28%, 37% and 58% for PFHxA, PFOA, PFHxS and PFOS, respectively after waste water treatment due to degradation of precursor compounds.

Some studies reported that concentrations of PFASs in surface water, sediments and biota are higher in densely populated (urbanized) areas (Junttila et al., 2019; Castiglioni et al., 2015; Nguyen et al., 2017). The sewage sludge, when it is used as fertilizer in fields and can become a source of contamination in soils and groundwater (Castiglioni et al., 2015; Johansson & Undeman, 2020). Important nonpoint (diffusive) sources are dry or wet atmospheric deposition (Filipovic et al. 2013, 2015), and soil or street surface runoff (Ahrens, 2011; Filipovic et al. 2015).

Areas, where firefighting foams were applied are an important source of PFASs in the environment causing contamination of soil and groundwater (Ahrens, 2011; KEMI, 2015; Xiao, 2017), that may last for many years (Xiao, 2017).

Rivers are important input pathways of PFAS into the marine environment (Ahrens, 2011; Castiglioni et al., 2015; Junttila et al., 2019). Generally, concentrations of PFASs decrease gradually with the increasing distance from WWTP effluents (to the open-water or along the river) (Ahrens, 2011; Wang et al., 2013).

Due to the replacement of long-chain PFOS and PFAS by short-chain alternatives there is a shift in contamination present (Joerss et al., 2019; Junttila et al., 2019) and increase of the load of short-chain PFAS is expected (Junttila et al., 2019; Joerss et al., 2019). According to Xiao et al. (2017), studies published between 2009 and 2017 have discovered 455 new PFASs (including nine fully and 446 partially fluorinated compounds), that have been found worldwide in the aquatic environment. The stockpiles of PFOS containing products as firefighting foams can be important source of the substance in developing countries (UNEP, 2017).

The Baltic Sea

Despite the regulations, PFOS and PFOA are still present in the biota, coastal waters and sediments of the Baltic Sea (Gebbink et al., 2016; Junttila et al., 2019; Joerss et al., 2019). However, there is scarce data about PFASs in the Baltic Sea and further investigations are needed (Junttila et al., 2019; Johansson & Undeman, 2020).

Junttila et al. (2019) showed a wide dispersion of PFASs in the Finnish aquatic environment. They estimated PFOS input into the Baltic Sea via Finnish rivers 10 kg yr^{-1} and the total annual loading of PFASs from the Finnish mainland 80 kg. Scientists assumed that the products containing PFOS are the main source of the substance to WWTPs and from there find their way into aquatic environment. They suggested that in the future the load of PFAS via WWTP will decrease due to replacement of products containing PFOS. Authors concluded that contaminated land areas, such as areas with firefighting activities or where WWTP sludge has been applied, are possibly major contributors to riverine PFAS loading, but further investigations are needed. It was also found that the load of PFBA, which is an alternative of long-chain PFAS, from Finnish rivers to the Baltic Sea was relatively high, 17 kg yr^{-1} in total, contributing 6–48% (on average, 26%) to the total PFAS loading. Vieno (2013) estimated the load of PFOS from WWTPs to surface waters in Finland as 12 kg yr^{-1} in 2013.

Filipovic et al. 2013 calculated a mass balance for some PFASs (PFHxA, PFOS, PFDA, PFOA) in the Baltic Sea. The study showed that WWTPs have a minor contribution to the total PFAA input to the Baltic Sea, instead, atmospheric deposition is a dominant source of PFAAs to the Baltic Sea. They calculated that WWTP effluents made only a moderate contribution to riverine discharge (21% for PFOA, 6% for PFOS), while atmospheric deposition to the watershed was 1−2 orders of magnitude greater than WWTP discharges. Scientists concluded that input of observed PFAAs to the Baltic Sea exceeds the output, possibly due to retention and delayed release of PFAAs from atmospheric deposition in the soils and groundwater of the watershed. Johansson & Undeman (2020) summarized this study in the HELCOM report. According to the report, observed PFAAs are mostly stored in the water column, which was estimated to contain 78%, 96%, 91% and 46% of the Baltic Sea inventory of PFOS, PFHxA, PFOA and PFDA, respectively. The main output of studied PFAAs takes place through the Danish Straits.

PFOS and PFOA alternatives

HFPO-DA is known as alternative for PFOA (Heydebreck et al., 2015; ECHA, 2017; Gao et al., 2020), with an annual production volume in Europe 10-100 t (Heydebreck et al., 2015; ECHA, 2017). In July 2019, ECHA added HFPO-DA (2,3,3,3-tetrafluoro-2-

(heptafluoropropoxy) propionic acid) to Candidate List of Substances of very high concern for Authorisation due to its probable serious effects to human health and environment (ECHA, 2019).

Recently, many studies identified HFPO-DA as a dominant pollutant of the marine environment in the different areas around the globe (Heydebreck et al., 2015; Joerss et al., 2019). According to Joerss et al. (2019), the shift to the replacement compound HFPO-DA as one of the dominant PFASs in surface water from the German Bight shows changes in pollution levels as a consequence of action taken by regulatory authorities and industry aiming to restrict long-chain PFASs.

6:2 Cl-PFESA and 8:2 Cl- PFESA have been used as alternatives of PFOS, with an annual production capacity of 20–30 t/year in China (Wang et al., 2013; Gao et al., 2020). Myriad PFASs are still used in industry, but the ecotoxicity and behaviour in environment of most of them are poorly known (Junttila et al., 2019).

Time lag

No direct estimates of the time lag between the stop of manufacturing of PFOS and decline in concentrations in the environment can be found yet. An analysis of PFOS concentrations in white-tailed eagle (Haliaeetus albicilla) body feathers (1968-2015) showed that the decreasing trends occurred within 10 years after voluntary phase-out of PFOS manufacturing in two studied populations (outside the Baltic Sea area), except the central Baltic population (Sun et al., 2019).

3.3. TBT, organotins

The input of organotins to the sea originates mainly from antifouling paints (Viglino et al., 2004; Radke et al., 2008; Jokšas et al., 2019). TBT-based antifouling paints were used since the 1960s to protect ship hulls from the attachment of algae and invertebrates (Omae, 2003; Fent, 2006; Lagerström, 2017; Michaud nad Pelletier, 2006; Antizar-Ladislao, B., 2008; Radke et al., 2008; Filipkowska et al., 2014). The application of antifouling systems based on organotins allowed the shipping industry to reduce the maintenance cost of fuel and to protect the hull effectively (Antizar-Ladislao, B., 2008; Radke et al., 2008).

World production of organotin compounds was estimated at about 40,000 ton/year in 1985, increasing to 50,000 ton/year in 1996 (Antizar-Ladislao, B., 2008). In addition to use organotin compounds (OTC) in antifouling paints, they also were widely used as biocides in agriculture, plastic (PCV) stabilizers, in wood preservation etc. (Hoch, 2001; Fent, 2006; Radke et al., 2008; Antizar-Ladislao, 2008).

TBT is highly toxic compound for aquatic organisms even at very low concentrations (Lee et al., 2006; Antizar-Ladislao, 2008; Jacobson et al., 2011; HELCOM, 2017; Horie et al., 2018). TBT is bioaccumulated by marine organisms causing harmful effects that mainly depend on the level of its final concentration in the tissues (HELCOM, 2017). TBT may act as an endocrine disruptor for aquatic invertebrates (Omae, 2003; Lee et al., 2006; Nakanishi, 2007; Sunday, 2012;).

But other organotins, like triphenyltin (TPT), also show toxic effects (Horiguchi et al., 1997; Hoch, 2001; Lee et al., 2006; Nakanishi, 2007). DBT is less toxic and its toxicity action is by blocking the absorption of oxygen in the mitochondria (Antizar-Ladislao, B., 2008).

Accumulation and persistence in environment

In the aquatic environment, TBT is quickly removed from the water column and adheres to sediments because TBT has a high specific gravity (near 1.2 kg l-1 at 20 °C) and low solubility (less than 10 mg l-1 at 20 °C and pH 7.0) (Antizar-Ladislao, 2008).

Organotin compounds in the aquatic environment, due to their low water solubility (hydrophobic character) tend to adsorb onto suspended particulate matter (SPM) and accumulate in sediments (Dowson et al., 1996; Hoch, 2001; Filipkowska et al., 2011; Briant et al., 2016). TBT can be rapidly degraded in the water column by biological activity and photomediated reactions to form less-toxic species such as monobutyltin (MBT) or dibutyltin (DBT) (Rodriguez-Gonzalez et al. 2013; Rodriguez-Cea, 2016).

Accumulation in sediments of organotins depends on the fine fraction and organic carbon content of sediments (Hoch, 2001). Many published papers have demonstrated that butyltins are adsorbed onto sediments preferably by binding with organic material (Berg et al., 2001; Burton et al., 2004) and there exists a significant correlation between the TOC and butyltins concentration (Filipkowska et al., 2011; Ruczyńska et al., 2016). But, contradictory results can be found (Furdek et al., 2016). Some authors found a weak (Viglino et al., 2004) or no correlation (de Oliveira et al., 2010) between butyltins and organic carbon content in sediments.

Organotin compounds in sediments can persist in the range of years (Mora and Pelletier, 1997; Omae, 2003; Takeuchi et al., 2004; Viglino et al., 2004; Jokšas et al., 2019) rather than days or weeks in the water (Hoch, 2001; Omae, 2003; Rodriguez-Gonzalez et al. 2013; Rodriguez-Cea, 2016; Lee et al., 2006). The half-life obtained for TBT to form DBT by Rodriguez-Gonzalez et al. (2013) was lower than 18 days under light conditions in seawater, whereas those for DBT to form MBT were lower than 5 days. Rodriguez-Cea (2016) observed full degradation of TBT and DBT in river water under light/darkness cycle at room temperature after 2 weeks. Furdek et al. (2016) found half-lives of TBT and DBT in porewaters 9.2 days and 2.9 ± 0.1 under oxic conditions, respectively. Half-life for DBT under anoxic conditions in porewaters was 9.1 ± 0.9 days (Furdek et al., 2016).

De Mora and Pelletier (1997) summarized half-life of TBT in sediments ranging between 1.8 and 3.8 years, Omae (2003) reported half-lives as about 1 to 9 years. Cornelissen et al. (2008) estimated the half-life of TBT in the sediment in the Oslo harbor area >10–20 years. Half-life of TBT in the deep-sea sediment (Saguenay fjord, Western Canada) has been estimated to be approximately 8±5 years for the surface oxic layer and 87±17 years for the deep anoxic layer (Viglino et al., 2004). Thus, sediments might be acting as both a major reservoir and secondary pollution sources of organotins at previously contaminated sites (Lee et al., 2006; Filipkowska et al., 2014; Briant et al., 2016; Jokšas et al., 2019).

The adsorbed organotins can be released from sediments into water column by resuspension or diffusion into the water column (Fent, 2006; Lee et al., 2006; Filipkowska et al., 2014)

Furdek et al. (2016) found that in contaminated sediments rich in organic matter, less butyltins will be desorbed, and a higher persistence of toxic TBT can be expected than in sediments with lower organic matter content. The results of their study strongly suggest that a limiting step in organotin compounds degradation in marine sediments is their desorption into porewaters because their degradation in porewaters occurs notably fast.

Degradation rates of organotin compounds (OTCs) depends on environmental conditions (Omae, 2003; Viglino et al., 2004; Filipkowska et al., 2014). Degradation of organotin compounds is mainly caused by UV photolysis and microbial activity (Gadd, 2000; Omae, 2003; Fent, 2006; Raudonyte-Svirbutaviciene et al., 2018; Rodriguez-Cea et al., 2016) and occurs via progressive loss of the organic substituents (Gadd, 2000; Omae, 2003; Furdek et al., 2016).

Filipkowska et al. (2014) have described that in the marine environment, in particular on the sea bed, biological cleavage is the most important one. There is evidence that some microorganisms, like bacteria (e.g., Pseudomonads, Alcaligenes faecalis, Shewanella putrefaciens) and phytoplankton (e.g., Skeletonema costatum, Chlorella vulgaris, Scenedesmus dimorphus), have the ability to degrade organotin compounds (Hoch 2001; Lee et al. 2012; Sampath et al. 2012; Tam et al. 2002; Bridou et al., 2018). As a consequence, the environmental conditions determining the growth of these microorganisms, such as pH, temperature, oxygen, turbidity, and light, are also factors recognized as responsible for the degradation of these contaminants (Filipkowska et al., 2014).

The persistence of butyltin compounds in sediments (the Southern Baltic Sea) can be enhanced by the following factors: high salinity, low temperature, high water column depth, high organic matter content, and high percentage of fine grain-size fraction (Filipkowska et al., 2014).

The rate of OT degradation also depends on the sediment type, chemical species of the OTs (e.g., hydroxides, chlorides, oxides), even on the OT concentration itself (Hoch, 2001; Omae, 2003; Filipkowska et al., 2014).

The high stability of bytiltins in the sediment can be explained by low water temperatures (2-10 °C) and anoxic conditions (Takahashi et al., 1999; Fent, 2006; Negri and Marshall, 2009; Viglino et al., 2004; Filipkowska et al., 2014) that inhibit TBT degradation. Debutylation of TBT to DBT and MBT occurs mainly by aerobic biological processes and the absence of oxygen will result in a much slower degradation in deep-sea sediment (Viglino et al., 2004).

Some studies have shown that aerobic biodegradation is faster than the anaerobic one (Antizar- Ladislao, 2008; Furdek et al., 2016).

Sources of organotins

The use of TBT-based antifouling paints was regulated in most European countries (Champ, 2000) as early as in 1989 (Jokšas et al., 2019). In 1989 regulations banned the use of TBT-containing antifouling paints on vessels up to 25 m long (Council Directive 89/677/EEC). (Albanis et al., 2002; Radke et al., 2008; Eklund et al., 2016). The Helsinki Convention published on April 9, 1992 also introduced such ban over the Baltic Sea area (Radke et al., 2008).

Subsequently, the use of TBT in new antifouling system has been restricted in 2003 with the Regulation 782/2003/EEC (EU, 2003) followed by international ban on using harmful organotins in antifouling paints in 2008 by the International Maritime Organization (IMO 2001) (Radke et al., 2008; Jokšas et al., 2019; Filipkowska et al., 2014; Egardt et al., 2017).

Despite the restrictions, high levels of organotin compounds were detected in coastal sediments over the last decade around the world (Furdek et al., 2016; Batista-Andrade et al., 2018; Lofrano et al., 2016; Paz- Villarraga et al., 2015).

Jokšas et al. (2019) have shown significant decreasing of tributyltin pressure after prohibition of TBT in the south-eastern Baltic Sea.

The problem of organotins in the environment has not been totally solved after total ban in 2008. The level of TBT and its degradation products in sediments continues to be high particularly in sediments of the ports area and small boat harbors around the Baltic Sea (Filipkowska et al., 2014; Suzdalev et al., 2015; Eklund et al., 2016; Ruczyńska et al., 2016; Jokšas et al., 2019), and TBT remains the matter of concern in this area. Ruczyńska et al. (2016) has detected signifcant decreasing of butyltin compiunds in areas far from potential sources of pollutants.

The main threat of the TBT pollution is related to the historic contamination but the fresh input of TBT is still present in hot spots (areas associated with ports, pleasure boats harbors and shipyards) around the Baltic Sea (Roots & Roose, 2013; Filipkowska et al., 2014; Suzdalev et al., 2015; Ruczyńska, 2016; Egardt et al., 2017; Eklund et al., 2018; Jokšas et al., 2019) and in the ohter regions in the world (Paz-Villarraga et al., 2015; Lofrano et al., 2016; Batista-Andrade et al., 2018). TBT and its degradation products also are found in natural harbors in the Baltic Sea (Swedish coast) (Eklund et al., 2016; Egardt et al., 2017).

Semi-enclosed bays with a limited water exchange and TBT pollution sourses as ship-building, cleaning and repairing companies have perfect conditions for accumulation of the pollutants (Jokšas et al., 2019).

According to Filipkowska et al. (2014), after the total ban, organotins contaminated sediments rather than vessels became a significant source of TBT and TPhT in the Sothern Baltic coastal zone (Poland). In this area the high concentration and fresh input of TBT into sediments were found close to anchorage and dumping sites. The heavy traffic of small vessels and high amount of fishing nets did not cause particularly high contamination of this basin. TBT concentration in sediment samples, collected in the 2008-2009, was in the range 0.2 – 115.2 ±11.4 ng Sn g^{-1} d.w.

In the south-eastern Baltic Sea (Klaipeda Strait, Lithuania) maritime traffic also had only a minor impact on sediment contamination (Jokšas et al., 2019). Highly organotins contaminated sediments with fresh TBT input in this area were related to the ship repairing facilities and dockyards operating in port, where the primary source of TBT is probably the antifouling agent, leaching from the ships during ship repairing and cleaning activities (Suzdalev et al., 2015; Jokšas et al., 2019). In the south-eastern Baltic Sea (Klaipeda Strait, Lithuania) TBT concentration in sediments is very much location-related and varies from 1 to 5,200 ng Sn g^{-1} d.w. Samples were collected in 2010–2012. (Suzdalev et al., 2015). In the samples collected in 2013 TBT concentration was in range <1 – 737 ng Sn g^{-1}. (Jokšas et al., 2019).

The periodical dredging of the port channels and shipping routes, and disposal of contaminated sediments are the reason of butyltins remobilization from sediment to the water column. These activities pose a great risk to the sensitive marine organisms around the dumping sites (Filipkowska et al., 2014; Jokšas et al., 2019).

Roots & Roose, 2013 found high contents of organotin compounds in the water samples taken from the effluents of Baltic Ship Repair Company and Company Baltic Premator and which flow into Tallinn Bay (MBT – 614 ng l-1; DBT – 7058 ng l-1;TBT – 9090 ng l-1; monophenyltin – 51 ng l-1; and diphenyltin – 25 ng l-1) and in the the sewage sludge of Baltic Premator Company (MBT – 152 lgkg-1 DM, DBT – 150 lgkg-1 DM,TBT – 22.5 lgkg-1 DM, tetrabutyltin – 27 lgkg-1 DM, monophenyltin –59 lgkg-1 DM, diphenyltin – 34 lgkg-1 DM and triphenyltin – 15 lgkg-1 DM).

One important sourse of TBT in the Baltic Sea environment are the leasure boat hulls with old coating. High levels of TBT in surface sediment are found in marinas on the Swedish coast (Eklund et al., 2016; Egardt et al., 2017). Despite on ban in 1989 that prohibited use of TBT-based paints on vessels smaller than 25 m, TBT still release from older paint of small boats that has been covered with newer coats (Eklund et al., 2008, 2010, 2014, 2018; Eklund and Eklund 2014; Lagerström et al., 2017). Lagerström et al. (2017) suggests that many boat owners decided not to remove the organotin paint or that they were not successfully sealed. However, illegal paints still remain on sale in some areas (Turner & Glegg, 2014) and probably were being in use causing leaching of TBT into environment (Egardt et al., 2017).

Lagerström et al. (2017) have found presence of TBT on all of 23 sampled leisure boat hulls from Sweden, Germany and Finland. A high portion of the samples also contained TPhT, suggesting that this biocide may be more commonly occurring on leisure boats than previously thought.

The study, performed around the Baltic Sea in the period April 2015 to February 2017, showed that 23–42% of the in total 377 measured boats in Denmark, Finland, and Germany still have amount of tin (organotin compounds) on their hulls. Mean values of tin >400 µg/cm2 was found on 10% of the Danish boats and in 16% of the Swedish boats, 5% of Finland and 1% of Germany boats. Apparently, the enforcement of the TBT regulation has differed in the investigated countries. This is a source to leakage to the environment and is one cause to elevate organotin concentrations in the sediments along the Baltic shores and a reason why good status according to the Marine Strategy Framework Directive (2008/56/EC) is not achieved (Eklund and Watermann, 2018).

The reason of high inputs of TBT into small boat harbors can be both the release from old paint layers in connection with high pressure washing applying on the boat hulls (Eklund et al., 2008, 2010; Eklund and Watermann, 2018) and TBT run-off from adjacent boat yards, which often are heavily polluted by organotin compounds (Eklund and Eklund 2014; Eklund et al., 2014; Eklund and Watermann, 2018).

High-pressure washing is common practice in Germany, Finland, Sweden and Denmark albeit antifouling paints except hard coatings are designed to erode or polish and cannot stand hp-washing without damage or removal of upper paint layers (Eklund and Watermann, 2018).

TBT pollution in the boatyards soils probably originates from older layers of boat hulls paint that come off during maintenance work and accumulate in the ground. From soils TBT can be released by washing out by rain water into the adjacent waters (Eklund and Eklund 2014; Eklund et al., 2014).

According to Eklund and Watermann (2018) the national laws around the Baltic Sea are quite differing. In Germany, according to the Federal Water Act, it is illegal to clean boat hulls with tap or high-pressure water outside wash-down areas or tarps as collection systems and is only allowed on wash-down areas with collection and filter systems. In Sweden, there is a national recommendation not to clean boats outside areas with filtering and collection systems. The national authority leaves it to the municipality to make the final decision about the regulation in their respective community. A similar approach is found in Denmark where the local harbors can decide on regulations for the boats in their harbor (personal communication H. Anker, Professor at Copenhagen University).

The reason for still finding organotin compounds in environment may be due to a deficiency of regulations and lack of enforcement of existing legislation (Eklund and Eklund 2014; Egardt et al., 2017).

Eklund and Watermann (2018) suggested the following options to be enforced by environmental authorities, harbor operators, boat clubs, etc. to reduce organotins concentration in marinas: 1) Certificate for boats built before 2008 on the existing paint layers. In case of existing paints with illegal compounds like organotin, diuron, or irgarol, these boats should be strictly excluded from hp (high preassure)-washing outside wash-down areas. 2) Cleaning with a sponge should be performed with a tarp to protect the ground and collection of the wash water. Or obligation to remove all paint layers down to the primer in case no certificate can be delivered. 3)Prohibition of hp-washing for boats from other harbors, resp. guest boats unless presenting a certificate. 4)In harbors where the erection of a wash-down area with collection and filter systems is too expensive, as maintenance practice should be recommended to clean the hulls after lifting in autumn with a soft sponge and sand them softly, if necessary, before launching in the next spring. 5) These maintenance works should include protection measurements of the ground.

Egardt et al. (2017) suggests, in combination with enforcement of regulations, to carry out an information campaign directed at the boating community to change an attitude regarding acceptable antifouling paint use.

Time lag

Organotins are accumulated in sediments, especially in the vicinity of harbours and shipping routes. The degradation rate of organotin compounds depends on various physical, chemical and biological factors, and TBT retention in the marine sediments might last from months to tens of years (Jokšas et al., 2019; Takeuchi et al., 2004; Viglino et al., 2004).

The differentiation between historical and continuing input of organotins can be done by calculating indices based on the ratio of TBT and its less toxic degradation derivatives DBT and MBT (Michel et al., 2001). A high TBT to DBT ratio (Üveges et al., 2007) as well as the ratio of MBT + DBT to TBT (BT degradation index-BDI) below 1.0 (Díez et al., 2002) relate to lower degradation of TBT, indicating fresh contamination.

3.4. Pharmaceuticals

More than 3,000 active pharmaceutical substances are being authorized worldwide in prescription medicines, over-the-counter therapeutic drugs, and veterinary drugs and reach production volumes of up to 100,000 tons per year by pharmaceutical companies in both the industrialized and the developing countries (Umweltbundesamt, 2016; Lindim et al., 2017; Kötke et al., 2019). A significant growing trend for the consumption of pharmaceuticals has been predicted due to increasing investments in the health-care sector, advances in research and development, pervasive global market availability, and demographic changes such as growing and aging population (aus der Beek et al., 2016; Kötke et al., 2019; UNESCO and HELCOM, 2017; Arnold et al., 2014; IFPMA, 2017).

Pharmaceuticals are considered as groups of emerging environmental contaminants that have received increasing attention in recent years due to their widespread occurrence (aus der Beek et al., 2016; Björlenius et al., 2018; Wolecki et al., 2019), potential biological activity (Borecka et al., 2015; Szymczycha et al., 2020) and impact on the environment (Klatte et al., 2017; Kötke et al., 2019). Due to their continuous discharge into the environment through different entry paths, they are regarded as a class of pseudo-persistent contaminants (Li, 2014; Kötke et al., 2019; Wolecki et al., 2019; Szymczycha et al., 2020).

Björlenius et al. (2018) reported that in surface waters, concentrations of pharmaceuticals depend on human population density in the drainage area, volume of the receiving water body and technologies used in WWTPs. Main sources of pharmaceuticals are human (used by human patients) and veterinary (lifestock, aquaculture and pets) medicine (Lonappan et al., 2016; aus der Beek et al., 2016; Klatte et al., 2017; Szymczycha et al., 2020). The leakage from a pharmaceutical industry is considered as a minor source (Lonappan et al., 2016).

The release of pharmaceuticals, their metabolites and transformation products into aquatic environments occurs through multiply pathways, including wastewater treatment plant (WWTP) effluents as primary source, and also domestic wastewaters, hospital discharges, improper manufacturer disposal, water treatment plant (WTP) effluents and run-off of veterinary medicines (Liu and Wong, 2013; Yang et al., 2017; UNESCO and HELCOM, 2017; Desbiolles et al., 2018; Kötke et al., 2019). Many other studies also confirm that the predominant load of pharmaceuticals to the aquatic environment occurs through waste water treatment plants (WWTPs) (Fatta-Kassinos, 2011; Liu & Wong, 2013; Arnold et al., 2014; aus der Beek et al., 2016; Klatte et al., 2017; Desbiolles et al., 2018; White et al., 2019; Palma et al., 2020)

Pollution hotspots by pharmaceuticals were found mostly in enclosed or semi-enclosed water bodies that are impacted by direct and/or undirect sewage discharges (Borecka et al., 2015; Biel-Maeso et al., 2018). Due to higher dilution in marine waters, the concentrations of pharmaceuticals are expected to be low compared to freshwater settings and wastewater (Borecka et al., 2015; Biel-Maeso et al., 2018).

The presence of different pharmaceuticals has been found in the water and sediment of the Baltic Sea (Borecka et al., 2013, 2015; Siedlewicz et al., 2014, 2016, 2018; Nödler et al., 2014; Fisch et al., 2017; UNESCO and HELCOM, 2017; Björlenius et al., 2018; Szymczycha et al., 2020). In the Baltic Sea, wastewater treatment plants (WWTPs) are also considered as the major pathway of pharmaceuticals to the environment (Turja et al., 2015; UNESCO and HELCOM, 2017; Fisch et al., 2017; Bolmann et al., 2019).

Additional sources of pharmaceutical residues to the Baltic Sea may include emissions from scattered dwellings not connected to centralized sewage systems, runoff/leaching from land where manure or sewage sludge has been applied, and landfill leachate in cases where medical waste has been incorrectly disposed of via solid waste (Szymczycha et al., 2020).

UNESCO and HELCOM (2017) identified that the most frequently detected substances in seawater in the Baltic Sea were carbamazepine (135 out of 218 samples; 60%), diclofenac (79 out of 322 samples; 25 %), ibuprofen (38 out of 260 samples; 15 %), and sulfamethoxazole (12 out of 140 samples; 9 %).

After oral administration, some PhACs are metabolized, while others remain intact before being excreted (Biel-Maeso et al., 2018). Human pharmaceuticals are generally excreted and emitted into the sewage system following use and reach wastewater treatment plants (WWTPs) (Fatta-Kassinos, 2011; Lolić et al., 2015; Biel-Maeso et al., 2018).

According to Carvalho and Santos (2016), municipal, agricultural, and industrial wastewater are the major entrance sources and pathways of antibiotics and their by-products, in the environment, since substantial amounts (30% to up 90%) of antibiotics administered to humans and animals are excreted into waste stream via urine and faeces, largely unmetabolized. Overall, antibiotics comprise approximately half of pharmaceuticals and personal care products contamination (Yang et al., 2017).

WWTPs includes also pharmaceuticals from incorrect disposals of unused drugs and hospital sewage (Bagheri et al., 2016; UNESCO and HELCOM, 2017; Kötke et al., 2019). From wastewater treatment plants (WWTPs) pharmaceuticals release (i) to surface waters via effluents or (ii) to terrestrial systems when sewage effluent is used for irrigation or where sewage sludge is applied as a fertilizer to agricultural land (Lindim et al., 2017; Biel-Maeso et al., 2018).

Many authors reported about inefficient removal of many pharmaceuticals during the wastewater treatment process (Verlicchi et al., 2012; Liu & Wong, 2013; Lolić et al., 2015; Carvalho and Santos, 2016; Lonappan et al., 2016; Fernández-López et al., 2016; Klatte et al., 2017; Yang et al., 2017; Biel-Maeso et al., 2018; Siedlewicz et al., 2018; Björlenius et al., 2018; Kötke et al., 2019). Turja et al. (2015) demonstrated the presence of WWTP originated contamination as a large number of different pharmaceuticals detected in the passive samplers attached to the mussel cages.

WWTPs have been mainly intended to remove organic matter and suspended solids and their effect on the removal of micropollutants may be, in some cases, negligible (Lolić et al. 2015; Bagheri et al., 2016; Yang et al., 2017). In fact, the removal efficiency of pharmaceuticals in WWTPs have been studied and a very wide range of values is observed, from compounds which pass these plants almost intact and others presenting a removal efficiency close to 100% (Verlicchi et al., 2012; Lolić et al., 2015; Fernández-López et al., 2016; Yang et al., 2017; Biel-Maeso et al., 2018). The removal efficiency of pharmaceuticals depends on technology used in WWTP (Zupanc et al., 2013; Bagheri et al., 2016; Kowalska et al., 2020).

Another important source of pharmaceuticals in the environment is animal husbandry (Liu & Wong, 2013; Arnold et al., 2014; aus der Beek et al., 2016; Klatte et al., 2017). A contamination of soils with pharmaceuticals may result if manure or sewage sludge are used as fertilizers. If manure or sewage sludge are used to fertilize land, pharmaceuticals accumulate in soil. Then they can relocate to water bodies and groundwater after precipitation events (aus der Beek et al., 2016; Klatte et al., 2017).

Apart from human pharmaceuticals, veterinary pharmaceuticals carry a significant environmental risk, as the active agents and their degradation products enter the environment directly, never passing through a waste water treatment plant system (Arnold et al., 2014; Klatte et al., 2016; Siedlewicz et al., 2018).

Aquaculture activities may contribute locally to pharmaceutical (antibiotics) contamination in coastal waters (Zheng et al., 2012; Liu & Wong, 2013).

Tourism and port activities can also be responsible for higher pharmaceuticals concentrations in coastal waters (Lolić et al., 2015; Siedlewicz et al., 2018). Lolić et al., 2015 that higher concentrations of pharmaceuticals along bathing season occurred in August and September, which can be due to an increment of population in the study areas during the holidays among north Portuguese coast.

In the riverine system concentration of pharmaceuticals attenuated exponentially during their transport process towards the open sea (Fisch et al., 2017; Biel-Maeso et al., 2018). Fisch et al. (2017) reported decreasing of pharmaceuticals concentrations in studied rivers towards the Baltic Sea. Turja et al. (2015) found a clear gradient of pharmaceuticals from point of WWTP efflux. Shi et al. (2014) reported about higher concentrations of pharmaceuticals around river discharges and WWTP effluent.

According to Szymczycha et al. (2020) the role of groundwater systems in the cycling of pharmaceuticals is underestimated and play important role. In coastal areas groundwater can enter the sea directly via submarine groundwater discharge (SGD).

Diclofenac

Diclofenac is HELCOM pre-core indicator. According to HELCOM (2018), currently the distribution, role and fate of diclofenac in the Baltic Sea marine environment is not clearly understood, with limited information from few monitoring and screening studies available.

Diclofenac mainly enters aquatic environments via inputs from WWTP due to the ineffective conventional treatment process in WWTP (Verlicchi et al., 2012; Vieno and Sillanpää, 2014; Lonappan et al., 2016; Yang et al., 2017; Bonnefille et al., 2018).

The extent of diclofenac degradation in WWTP depends on the wastewater treatment technology used (Verlicchi et al., 2012; Vieno and Sillanpää, 2014; Bonnefille et al.,2018). The low diclofenac biodegradability often results in low elimination rates during biological wastewater treatment, and only a minor portion is adsorbed to sludge (Vieno and Sillanpää, 2014). Verlicchi et al. (2012) reported an average removal efficiency rate of 3% to 65% for during the different wastewater treatment process. Vieno and Sillanpää (2014) reported average diclofenac elimination rates 36%-48% in WWTPs with different biological treatment processes.

Direct photolysis is the predominant removal process for diclofenac in freshwater (Tixier et al., 2003; Kunkel and Radke, 2012), exhibiting a half-life of 8 days (Tixier et al., 2003). Removal rate of ibuprofen and carbamazepine in freshwater estimated 32 days and 63 days, respectively (Tixier et al., 2003). However, other studies suggest that elimination by photolysis is not so dominant for most substances in rivers. Even under (near) optimal conditions (small stream depth, sparse bank vegetation) photolysis contributes only 50% to the total elimination for a highly photolabile substance like diclofenac. Therefore, in larger (and deeper) rivers, photolysis of organic micropollutants is supposedly to be even of less importance (Kunkel and Radke, 2012).

Volatilization from water surfaces is not expected to be an important fate process. Little or no biodegradation using a freshwater inoculum suggests that biodegradation is not an important environmental fate process in water for diclofenac (PubChem Database, 03.05.2020).

Further investigations

Even though the occurrence of pharmaceuticals in the water environment is thought to be a potential problem for human health and aquatic organisms, many authors reported about the insufficient level of knowledge of their sources, pathways, fate in wastes and the environment, metabolism and excretion (pharmacokinetics) in wildlife and presence in the marine ecosystem and need of further investigations (Kunkel and Radke, 2012; Arnold et al., 2014; Vieno, N., Sillanpää, M., 2014; Shi et al., 2014; Borecka et al., 2015; Siedlewicz et al., 2018; Fang et al., 2019; Kötke et al., 2019; Szymczycha et al., 2020; Palma et al., 2020; Gros et al., 2020).

Some authors declared the importance of further research dealing with the development and optimization of new treatment processes to more efficiently remove pharmaceuticals in WWTPs (Meribout et al., 2016; Desbiolles et al., 2018; Biel-Maeso et al., 2018; Kowalska et al., 2020)

Generally, surveys on the occurrence of pharmaceuticals in marine waters are scarce compared to studies in other surface waters, especially rivers as well as WWTPs. Most studies for marine waters focus only on a few selected compounds such as carbamazepine, diclofenac, or sulfamethoxazole (Kötke et al., 2019).

Reduction

Many authors reported about the importance of effective wastewater treatment and improvement of removal efficiency of pharmaceuticals in STPs/WWTPs in order to reduce concentration of pharmaceuticals discharged to the environment and thus eliminate the potential effects to the natural (Li, 2014; Vieno & Sillanpää, 2014; Yang et al., 2017; Klatte et al., 2017; Nurmi et al., 2019; Rizzo et al., 2019).

Klatte et al. (2017) summarized short-, medium- and long-term measures for the reduction of the pharmaceuticals in the environment.

4. Biodiversity

Natural conditions and possible future changes were analyzed to reveal their impacts on achieving GES regarding the following MSFD criteria:

- D1C2 (species abundance), D1C4 (species distributional range) – ringed seals, grey seals, white-tailed sea eagle, and cod were considered.
- D2C1 (newly-introduced non-indigenous species), D2C2 (abundance and spatial distribution of established non-indigenous species) – for the latter, round goby was considered.
- D6C3 (spatial extent of habitats) – benthic habitats

4.1. Ringed seals

The four areas, where the Baltic ringed seals (*Pusa hispida botnica*) occur for breeding and moulting, are the Bothnian Bay, Archipelago Sea, Western Estonia (Gulf of Riga and Estonian coastal waters), and the Gulf of Finland (HELCOM, 2018a). The main reasons, which caused the decline of the ringed seal abundances in the 20th century, were hunting (Harding and Härkonen, 1999) and high concentrations of environmental pollutants (see references in Kauhala et al., 2019). Hunting was stopped in the 1970-1980s, but is allowed again in a few northern countries in low numbers in recent years. Still, it is not considered as an important human-induced pressure anymore. The numbers of incidental by-catch are not available for the Baltic ringed seal, but gillnets are considered as the main risk for the ringed seal population in the Lake Saima, where after introduction of certain restrictions, 6-13 specimen are incidentally caught annually (Jounela et al., 2019).

According to HELCOM (2018b), the ringed seals in the Bothnian Bay management unit have reached good status for population size but not for growth rate. One of the reasons for slow population growth rate is mild winters with poor ice conditions (Sundqvist et al. 2012; HELCOM, 2018b). The state of distribution of ringed seals is not good since the area of occupancy is currently more restricted compared to pristine conditions >100 years ago (HELCOM, 2018a).

Pollution

Pollution (e.g., Kovacs et al. 2012; Troisi et al., 2020) and poor food resources (Kauhala et al., 2019) are considered as human-induced factors having negative population-level effects on ringed and grey seals. High levels of environmental pollutants, especially PCBs and DDT, are suggested as the main factor causing reproductive disturbances of both ringed and gray seals (*Halichoerus grypus*) in the 1950-1970s (Helle et al., 1976; Nyman et al., 2003). After hunting of ringed seals was prohibited and concentrations of organochlorines declined in the Baltic Sea, the ringed seal population started to grow slowly in the 1990s. However, the annual growth rate of the population is about 5%, while in a fast-growing seal population, it could be up to 10–12% (Harding et al. 2007).

Due to their persistent nature, PCBs have been the greatest contribution to total organic pollutant burdens in seals and other apex predators for many years after their ban. Troisi et al. (2020) found 4-5 times higher PCB concentrations in ringed seal blood sampled from the Baltic than those sampled at Svalbard in 1997-2002.

Food availability and quality

Kauhala et al. (2019) suggested that one of the reasons behind the low growth rate of the ringed seal population in the Bothnian Bay may be the relatively low birth rate caused, at least partly, by the declining nutritional status of adult females. The quality and quantity of food resources may affect individual growth and nutritional status of seals which, in turn, may have an impact on their mortality and reproductive rates – it has been shown for gray seals, but could also be applicable for ringed seals (Kauhala et al., 2017).

Ice conditions

Climate change has direct negative impact on ringed seals (Meier et al., 2004; Sundqvist et al. 2012; Kovacs et al. 2012; Auttila et al., 2014; Ferguson et al., 2017), whose breeding success depends on suitable ice coverage and snow (Laidre et al., 2008; Auttila et al., 2014; Ferguson et al., 2017). Availability of suitable breeding ice is very important factor for pup survival (Sundqvist et al. 2012).

According to Ferguson et al. (2017), the long open water period and late ice formation may be the reason of high stress level, low ovulation rate, low pregnancy rate, and sick seals.

Sundqvist et al. (2012) have shown that the area of breeding ice as such imposes a strong regulating factor for the Baltic ringed seals. They predicted reduced growth rates in all sub-populations of Baltic ringed seals during this century. The projected mean number of Baltic ringed seals in 2100 is 30730 individuals. Since the only fairly good winter sea-ice habitat could be confined to the Bay of Bothnia (Meier et al., 2004), the southern sub-populations, especially in the Gulf of Riga, have a high risk of extinction.

4.2. Grey seals

The main reasons, causing the decline of the grey seal abundances in the 20th century, were hunting (Harding and Härkonen, 1999) and high concentrations of environmental pollutants. An analysis by Vanhatalo et al. (2014) showed that the total yearly by-catch of grey seals by trap and gillnets in Finland, Sweden and Estonia is in a range of 1550-1880 specimen. Based on data from 2011-2016, grey seals are evaluated as having achieved good status with regard to the area of occupancy (HELCOM, 2018a). They have also achieved the threshold value regarding the breeding sites except for the Southwestern Baltic (HELCOM, 2018a).

Grey seal reproduction is not in good status with regard to reproductive rate in the entire Baltic when evaluated as one single population (HELCOM, 2018c). Nutritional status of seals is considered good when the subcutaneous blubber thickness is above the defined threshold value. Based on data from 2011-2016, the grey seal failed to achieve the threshold value of nutritional status, and the population is not in good status for the whole Baltic Sea (HELCOM, 2018d).

Pollution

High levels of environmental pollutants, especially PCBs and DDT, are suggested as the main factor causing reproductive disturbances of both ringed and gray seals (*Halichoerus grypus*) in the 1950-1970s (Helle et al., 1976; Nyman et al., 2003). Schmidt et al. (2020) have analyzed liver histopathology of Baltic grey seals from 1980-2010 and reported a clear decreasing trend of both PCB and DDT concentrations in adipose tissue (blubber) and the prevalence of portal mononuclear cell infiltration. Since the 1990s, the PCB concentrations did not exceed the blubber threshold concentrations for PCBs suggested by Sonne et al. (2020), except in one sample. Thus, the impact of legacy contaminants has been phasing out during 10-20 years after the regulations entered into force.

Food availability and quality

Kauhala et al. (2017) found that a herring catch size (an index of herring abundance) is negatively correlated with blubber thickness of grey seals. They suggested that herring quality is important for the nutritional status of Baltic gray seals. Furthermore, Kauhala and Kurkilahti (2019) showed that body condition (blubber thickness) of male pups of Baltic grey seal correlated positively with prey fish quality (herring and sprat weight). Adult males reached the asymptotic length at the age of 10.3 years, and their body length was positively related to herring and sprat weight in their birth year. Thus, the prey fish quality could have the delayed effects on grey seals, and continuous incremental food limitation can be detrimental to population size (Silva et al., 2020).

Ice conditions and winter temperatures

Baltic grey seals alternate between land and ice breeding, depending on ice conditions. Jüssi et al. (2008) showed that indices of life-time net reproductive rate (pup survival) and pup quality (weaning weight and health) were more auspicious on ice as compared with land. Thus, diminishing ice fields will lower the fitness of Baltic grey seal females and substantially increase the risk for quasi-extinction (Jüssi et al., 2008). Although the impact on grey seal abundances is not so severe than ringed seals, changes in ice conditions affect their status.

According to Kauhala and Kurkilahti (2019), birth rate of females (age 7–24 years) is negatively related to winter temperature in their birth year. They conclude that changes in climate may affect body condition of pups and later on the reproductive rate of Baltic grey seals.

4.3. Benthic habitats

Demersal trawling, dredging, shipping and nutrient inputs, cause the majority of impacts on benthic habitats in the Baltic Sea (Korpinen et al., 2013). Oxygen deficiency is a widespread threat to coastal and estuarine communities (Baden et al., 1990; Villnäs et al., 2012). Van Denderen et al. (2020) showed that 6% of the Baltic Sea region (about 30 000 km^2) is currently impacted by both bottom trawling and oxygen deficiency.

Fishing (bottom trawling)

Bottom disturbance during trawling may pose serious risks to the seabed and benthic habitats (Collie et al., 2000; Rijnsdorp et al., 2016; Rijnsdorp et al., 2018; Kaiser et al., 2018; Hiddink et al., 2017, 2019). Depending on gear type, fishing disturbance is able to remove about 50% of benthic individuals (Collie et al., 2000). Hiddink et al. (2017) showed that depletion of biota and trawl penetration into the seabed are highly correlated. Otter trawls caused the least depletion, removing 6% of biota per pass and penetrating the seabed on average down to 2.4 cm.

As a result of trawling, a shift from communities dominated by relatively high biomass species towards dominance by high abundances of small-sized organisms can take place (Collie et al., 2000). In the Kattegat area, the baseline state of the benthic communities is far from the theoretical carrying capacity due to a historical high trawling intensity (Bastardie et al., 2020).

Recovery rates after trawling depend on recruitment of new individuals, growth of surviving biota, and active immigration from adjacent habitat (Hiddink et al., 2017). Kaiser et al. (2018) showed that the recovery rates of biota depend on life-history factors, such as larval longevity and dispersal potential. Collie et al. (2000) found that recovery of benthic habitats was slower if the spatial scale of impact was larger, as it would be on heavily fished grounds.

Several studies revealed that recovery is quicker for short-lived fauna, than for long-lived fauna. (Kaiser et al., 2018; Bastardie et al., 2020). Simulations performed by Bastardie et al. (2020) showed that in the Baltic Sea, the dominance of short-lived benthic species indicates that little significant evidence is found for improvements in the benthic community from the displacement of fisheries activity away from peripheral areas. The increase in the relative benthic state is insignificant across the central Baltic region, even if reducing the spatial fishing extent significantly. Scientists anticipate a better benthic status in the Kattegat from concentrating the fishing effort in a smaller area. This region-specific difference arises from the different types of benthic communities that are affected by both environmental gradients and fishing pressure as well as the initial benthic status in the two areas.

Hiddink et al. (2017) estimated median recovery times post trawling (from 50 to 95% of unimpacted biomass) between 1.9 and 6.4 years depending on fisheries and environmental context. The effect of bottom trawling in comparative studies increased with longevity, with a 2-3 times larger effect on biota living >10 years than on biota living 1–3 years (Hiddink et al., 2019). Authors attributed this difference to the slower recovery rates of the long-lived biota (Hiddink et al., 2019).

Study performed in Lyme Bay, UK, revealed the recovery time for species with a high dispersal potential and less habitat specific requirements <3 years and up to 20 years for some longer-lived species with low dispersal potential and specific habitat requirements (Kaiser et al., 2018).

Eutrophication. Hypoxia.

Oxygen (normoxia, hypoxia, anoxia) is a strong predictor for species diversity and occurrence in the zoobenthic assemblage (Rousi et al., 2019). Villnäs et al. (2012) showed that increasing duration of hypoxia gradually deteriorate the benthic community and caused concurrent changes in ecosystem function (sediment oxygen and nutrient fluxes). As oxygen is a limiting factor for the biota at the seabed, the benthic fauna in the Gulf of Finland is in a state of constant change largely driven by the fast changes in oxygen conditions (Rousi et al., 2019).

Rousi et al. (2019) showed that in order to maintain healthy marine communities, it is essential to counteract excess nutrient inputs and their indirect effects on sufficient O2 conditions for the benthic habitats. Authors revealed the increase of amount of species in space and time with increasing oxygen concentrations.

5. Climate change

Potential impacts of climate change on achieving or not achieving GES (e.g., delays of reaching targets) are analysed in regard of the same topics as in the analysis of time lags– eutrophication, hazardous substances and biodiversity. The following meteorological and hydrological/hydrographic parameters and their impact taking into account the projected changes are considered:

- Change in meteorological conditions (air temperature, humidity and precipitation, wind, runoff, etc.)
- Change in temperature and salinity
- Change in stratification
- Change in sea level
- Change in sea ice conditions

5.1. Eutrophication

Model simulations indicate that the Baltic Sea suffers from eutrophication due to the increased anthropogenic load of nutrients, mainly via rivers, but also atmosphere and direct inputs (Meier et al., 2019). However, changes in meteorological and hydrological conditions during the last 150 years also contributed, for instance, to the spreading of hypoxia at the Baltic Sea bottoms. Below, the potential effects of projected changes in selected parameters on eutrophication are described (Table 5.1). Eutrophication criteria considered are nutrients, chlorophyll a and oxygen conditions.

Table 5.1. Potential impacts of projected changes in meteorological and hydrological/ hydrographic parameters on eutrophication criteria.

Parameter	Projected changes	Potential impact
Air temperature	Air temperatures are projected to increase, especially in winter. The increase of winter air temperatures could be as high as 3-8°C in the north, and 2-4°C in the south (BACC II, 2015).	Higher air temperatures could also cause higher sea surface temperature, lower salinity (increased precipitation) and potentially stronger stratification. It all leads to higher DIP and DIN concentrations, more intensive cyanobacterial blooms, and reduced oxygen conditions, both in coastal and deep basins (Meier, 2019b). Thus, if all measures are implemented to achieve BSAP nutrient load reduction targets, the time lag of achieving GES would be longer than the suggested 30-40

		years (due to accumulated nutrients and natural conditions of the Baltic Sea).
Humidity and precipitation	Precipitation is projected to increase in the future, especially in winter, but in the north also in summer (Christensen & Kjellström, 2018). Projected warming could increase atmospheric moisture content and potentially the frequency of heavy rain events (Rajczak & Schär, 2017).	An increase in precipitation in winter will cause an increase in runoff and nutrient loads. Also, more frequent floods could wash out the nutrients from the soils on land, and thus, result in an increase in nutrient loads. The time lag of achieving GES would be longer than the suggested 30-40 years (due to accumulated nutrients in the catchment).
Runoff	The runoff to the Baltic Sea is projected to increase, but the estimates vary a lot depending on the scenario, season and region. It is predicted that runoff will increase during winter but probably decrease during summer (Saraiva et al., 2019) The spring floods are projected to decrease in the north, but large floods could be more frequent in the south (Roudier et al., 2016).	An increase in runoff and more frequent floods could wash out the nutrients from the soils on land, and thus, result in an increase in nutrient loads. The time lag of achieving GES in regard of nutrient concentrations, chlorophyll a content and oxygen conditions would be longer than the suggested 30-40 years (due to accumulated nutrients in the catchment).
Wind	Wind and wave climate projections are uncertain, but a slight increase in wind speed in autumn is predicted (Ruosteenoja et al., 2019). Significant wave heights could increase (Groll et al., 2017) taking also into account a decrease in ice cover.	No clear impact on eutrophication, although more intense vertical mixing could improve near-bottom oxygen conditions in shallow areas. It also could increase nutrient fluxes from the deep areas to the surface layer.
Sea temperature	Sea surface temperatures have been increasing since 1850s and it is projected that the increase will continue. Depending on climate change scenario, the increase until the end of this century would be between 1.2°C and 4.7°C (Gröger et al., 2019; Meier and Saraiva, 2020).	Higher sea surface could cause stronger stratification and it leads to higher DIP and DIN concentrations, more intensive cyanobacterial blooms, and reduced oxygen conditions, both in coastal and deep basins (Meier et al., 2019b). Warming may stimulate nitrogen-fixing cyanobacteria blooms, increase sedimentation and enhance oxygen depletion (Meier et al., 2019a).
Salinity	No clear projections exist predicting an increase or decrease in salinity. An	No clear impact on eutrophication. However, an increase in salinity in the deep layer of the Baltic Proper

	increase in river runoff could lead to the salinity decrease while a projected sea level rise would result in a salinity increase (Saraiva et al., 2019).	means also ventilation of these layers by oxygen-rich waters.
Stratification	Vertical stratification is projected to strengthen in summer due to the surface layer warming (Gröger et al., 2019). Potential changes in haline stratification are uncertain (Saraiva et al., 2019).	Stronger stratification and it leads to higher DIP and DIN concentrations, more intensive cyanobacterial blooms, and reduced oxygen conditions, both in coastal and deep basins.
Sea level	The Baltic Sea mean sea level is projected to rise although land uplift will continue in the north. At the end of the century, the sea level would be higher by 35-60 cm depending on the region, if the mid-range scenario of climate change is considered (BACC II, 2015).	No direct impact on eutrophication.
Ice conditions	The maximum sea ice extent is projected to decrease by 6400-10900 km^2 per decade, depending on the climate change scenario (Luomaranta et al., 2014). Also, the extent of the ice season is decreasing.	No direct impact on eutrophication.

5.2. Hazardous substances

Persistent hazardous substances from human activities have been accumulated in the environment, including the soils in the catchment area of the Baltic Sea and seabed sediments. Secondary pollution load can be caused by both human interventions and natural processes. Below, the potential effects of projected changes in selected meteorological and hydrological parameters on spreading of hazardous substances are described (Table 5.2). Substances under considerations were heavy metals (Hg, Cd, Pb), PFASs, and TBT, but also other persistent contaminants.

Table 5.2. Potential impacts of projected changes in meteorological and hydrological/ hydrographic parameters on spreading of hazardous substances.

Parameter	Projected changes	Potential impact
Air temperature	Air temperatures are projected to increase, especially in winter. The increase of winter air temperatures could be as high as 3-8°C in the north, and 2-4°C in the south (BACC II, 2015).	Higher air temperatures could increase the risk of forest fires. It could increase the input of some hazardous substances (e.g. metals) to the aquatic environments, as it has been observed in Australia and Mediterranean countries (e.g. Silva et al., 2016).
Humidity and precipitation	Precipitation is projected to increase in the future, especially in winter, but in the north also in summer (Christensen & Kjellström, 2018). Projected warming could increase atmospheric moisture content and potentially the frequency of heavy rain events (Rajczak & Schär, 2017).	Intense rains and flooding could occasionally increase riverine input of contaminants (accumulated in the soils of the catchment) to the coastal zone (Saniewska et al., 2018).
Runoff	The runoff to the Baltic Sea is projected to increase, but the estimates vary a lot depending on the scenario, season and region. It is predicted that runoff will increase during winter but probably decrease during summer (Saraiva et al., 2019) The spring floods are projected to decrease in the north, but large floods could be more frequent in the south (Roudier et al., 2016).	Frequent floods could increase riverine input of contaminants to the coastal zone (Saniewska et al., 2018). Leaking of contaminants accumulated in soils of the catchment will delay the effects of measures.

Wind and waves	Wind and wave climate projections are uncertain, but a slight increase in wind speed in autumn is predicted (Ruosteenoja et al., 2019). Significant wave heights could increase (Groll et al., 2017) taking also into account a decrease in ice cover.	Stronger winds and higher waves could cause resuspension of sediment particles and contaminants accumulated in the coastal environment.
Sea temperature	Sea surface temperatures have been increasing since 1850s and it is projected that the increase will continue. Depending on climate change scenario, the increase until the end of this century would be between 1.2°C and 4.7°C (Gröger et al., 2019; Meier and Saraiva, 2020).	Changes in sediment conditions, including temperature, could also influence the release of metals from the sediments. However, higher temperatures could favor faster TBT degradation in sediments.
Salinity	No clear projections exist predicting an increase or decrease in salinity. An increase in river runoff could lead to the salinity decrease while a projected sea level rise would result in a salinity increase (Saraiva et al., 2019).	No direct impacts.
Stratification	Vertical stratification is projected to strengthen in summer due to the surface layer warming (Gröger et al., 2019). Potential changes in haline stratification are uncertain (Saraiva et al., 2019).	Stronger stratification could lead to coastal hypoxia and anoxia in deep basins. Half-life of TBT in the deep anoxic layer could be as long as 87±17 years for the (Viglino et al., 2004).
Sea level	The Baltic Sea mean sea level is projected to rise although land uplift will continue in the north. At the end of the century, the sea level would be higher by 35-60 cm depending on the region, if the mid-range scenario of climate change is considered (BACC II, 2015).	Rise of the sea level could increase coastal erosion and related flux of accumulated contaminants (e.g. Hg) into the coastal environment (Beldowska et al., 2016).
Ice conditions	The maximum sea ice extent is projected to decrease by 6400-10900 km² per decade, depending on the climate change scenario (Luomaranta et al., 2014). Also, the extent of the ice season is decreasing.	No direct impact.

5.3. Biodiversity

Status of habitats and abundance and distribution of species are influenced by human-induced pressures and natural conditions. Since climate change alters environmental conditions it also impacts the distributional range of species and habitats. We focused on the following criteria: D1C2 – species abundance, D1C4 – species distributional range, D6C3 – spatial extent of habitats. Below, the potential effects of projected changes in selected species and habitats are described (Table 5.3).

Table 5.3. Potential impacts of projected changes in meteorological and hydrological/ hydrographic parameters on biodiversity (selected species and habitats).

Parameter	Projected changes	Potential impact
Air temperature	Air temperatures are projected to increase, especially in winter. The increase of winter air temperatures could be as high as 3-8°C in the north, and 2-4°C in the south (BACC II, 2015).	Higher air temperatures could also cause higher sea surface temperatures and indirectly affect the habitats and species distribution. It could offer more suitable conditions for some already arrived non-indigenous species, e.g. round goby (Nurkse et al., 2018) and introduction of new species.
Humidity and precipitation	Precipitation is projected to increase in the future, especially in winter, but in the north also in summer (Christensen & Kjellström, 2018). Projected warming could increase atmospheric moisture content and potentially the frequency of heavy rain events (Rajczak & Schär, 2017).	The effect is not direct if precipitation would not cause a decrease in salinity. Indirectly, if reduction in nutrient loads is slower then eutrophication would impact the habitats for longer period.
Runoff	The runoff to the Baltic Sea is projected to increase, but the estimates vary a lot depending on the scenario, season and region. It is predicted that runoff will increase during winter but probably decrease during summer (Saraiva et al., 2019) The spring floods are projected to decrease in the north, but large floods could be more frequent in the south (Roudier et al., 2016).	The effect is not direct if increased runoff would not cause a decrease in salinity. A potential decrease in salinity would be compensated by an increase due to a global sea level rise. Indirectly, if reduction in nutrient loads is slower then eutrophication would impact the habitats for longer period.
Wind and waves	Wind and wave climate projections are uncertain, but a slight increase in wind speed in autumn is predicted (Ruosteenoja et al., 2019). Significant wave heights could increase (Groll et al., 2017) taking also into account a decrease in ice cover.	Probably, no major impacts with influences in opposite directions. More mixing would improve near-bottom oxygen condition but increase sediment resuspension and water turbidity.

Sea temperature	Sea surface temperatures have been increasing since 1850s and it is projected that the increase will continue. Depending on climate change scenario, the increase until the end of this century would be between 1.2°C and 4.7°C (Gröger et al., 2019; Meier and Saraiva, 2020).	Higher sea water temperatures could mean more suitable conditions for some non-indigenous species, e.g. round goby (Nurkse et al., 2018). Increasing water temperatures would create better spawning conditions for some fish species. At the same time, it could worsen near-bottom oxygen conditions.
Salinity	No clear projections exist predicting an increase or decrease in salinity. An increase in river runoff could lead to the salinity decrease while a projected sea level rise would result in a salinity increase (Saraiva et al., 2019).	Since no clear trend is projected, the potential impacts are also unclear. However, MBIs are important, for instance, for the reproduction of cod.
Stratification	Vertical stratification is projected to strengthen in summer due to the surface layer warming (Gröger et al., 2019). Potential changes in haline stratification are uncertain (Saraiva et al., 2019).	Vertical stratification and related oxygenation/deoxygenation influence the reproduction of cod – due to oxygen deficiency and sedimentation related mortality Eastern Gotland Basin has deceased as a spawning area for cod (Hinrichsen et al., 2017). Strengthened stratification and related oxygen deficiency affect benthic habitats and their recovery after eutrophication-related pressures are reduced.
Sea level	The Baltic Sea mean sea level is projected to rise although land uplift will continue in the north. At the end of the century, the sea level would be higher by 35-60 cm depending on the region, if the mid-range scenario of climate change is considered (BACC II, 2015).	Higher sea level could lead to improved oxygen conditions that would positively influence the recovery of habitats (Meier et al., 2019b).
Ice conditions	The maximum sea ice extent is projected to decrease by 6400-10900 km^2 per decade, depending on the climate change scenario (Luomaranta et al., 2014). Also, the extent of the ice season is decreasing.	Change in sea ice conditions has significant impact on seals abundance and distributional range. If the extent of ice cover and duration of ice season continue to decrease, the southern sub-populations, especially in the Gulf of Riga, have a high risk of extinction.

6. Conclusions

Baltic Sea environment deviates from the good status due to the human-induced pressures and not natural conditions. The main pressures have historically been input of nutrients and contaminants, fishing and hunting, but also introduction of non-indigenous species, maritime activities, input of litter and underwater noise.

Baltic Sea states have agreed to implement the Baltic Sea Action Plan to reduce the pressures and improve the state of the marine environment. However, despite of actions implemented regionally and nationally, the latest assessment of Baltic Sea environmental status indicates the existence of many problems. Slow recovery of the marine environment could partly be related to the environmental conditions of the Baltic Sea (or marine environment in general). Furthermore, the changes and fluctuations in climatic conditions have to be taken into account when planning and implementing the measures and assessing their effectiveness.

Under the contaminants Metals – mercury (Hg), lead (Pb), and cadmium (Cd), TBT (tributyl-tin), PFOS (PFASs), Pharmaceuticals were analyzed. The review of scientific literature identified the causes of long-lasting or secondary inputs and estimated time lags of achieving GES presented in the following table.

Table 6.1. Summary table of natural condition influencing achieving GES regarding levels of contaminants.

Substances	Natural conditions influencing achieving of GES	Estimated time lag
Metals (Hg, Pb, Cd)	Soils of the catchment areas and seabed sediments are reservoirs of Hg, Cd and Pb accumulated there over a long period. Secondary inputs from the catchment could increase due to changes in land use, replacement of the natural surfaces due to urbanization, increasing intensity and duration of precipitation, storm and flooding events. Metals may be released from the sediments by physical disturbance, molecular diffusion, gas ebullition, bioturbation/bioirrigation. The flux is depending on changing conditions, such as changes in pH, dissolved oxygen or temperature.	The time lag of achieving GES after the measures are implemented is about 30 years –water residence time of the Baltic Sea is 20–30 years and the average residence time of mercury in oceanic waters is 20–30 years.

TBT	TBT has a high degradation rate in the water column, but it is low in sediments. The persistence of butyltin compounds in sediments can be enhanced by high salinity, low temperature, high water column depth, high organic matter content, and a high percentage of the fine grain-size fraction. The half-life of TBT in the deep-sea sediment has been estimated at 8±5 years for the surface oxic layer and 87±17 years for the deep anoxic layer. Periodical dredging of the port channels and shipping routes and disposal of contaminated sediments are the reasons for butyltin remobilization from sediment to the water column.	Estimates are in the range of 8-80 years.
Pharmaceuticals	Direct photolysis is the predominant removal process for diclofenac, exhibiting a half-life of 8 days. The removal rate of ibuprofen and carbamazepine in freshwater estimated at 32 days and 63 days, respectively. Due to their continuous discharge into the environment through different entry paths, pharmaceuticals are regarded as a class of pseudo-persistent contaminants.	No evidence of long time lags.
PFOS (PFASs)	Highly persistent, especially long-chain PFASs (e.g. PFOS and PFOA). Most PFAAs are not buried in sediments to a substantial degree. Runoff from background soil and atmospheric deposition of PFAAs associated with sea spray are the main sources of secondary pollution. No clear temporal trends observed for PFOS since the late 1990s; it means environmental PFOS concentrations have not yet declined as a response to reduced emissions.	Not estimated, but highly persistent.

The analysis under the Biodiversity topic focused on certain species (e.g., ringed seals and grey seals and their abundance and distributional range) and benthic habitats. Both human impacts and natural ecosystem processes that may influence the achieving of GES were indicated. Examples of the analysis results are shown in the table below.

Table 6.2. Summary table of natural condition influencing achieving GES regarding some selected species and habitats.

Species/ habitat	Main reasons of non-GES status	Conditions influencing achieving of GES
Ringed seals	The main reasons for the decline of the ringed seal abundances in the 20th century were hunting and high concentrations of environmental pollutants. Incidental by-catch is considered as one of the risks. The ringed seals in the Bothnian Bay management unit have reached good status for population size but not for growth rate. The state of distribution of ringed seals is not good since the area of occupancy is currently more restricted compared to pristine conditions >100 years ago.	One of the reasons for the slow population growth rate is mild winters with poor ice conditions. The low growth rate of the ringed seal population in the Bothnian Bay may be the relatively low birth rate caused, at least partly, by the declining nutritional status of adult females. The area of breeding ice is a strong regulating factor for the Baltic ringed seals. The projected mean number of Baltic ringed seals in 2100 is 30730 individuals. Southern sub-populations, especially in the Gulf of Riga, are at high risk (due to ice conditions).
Grey seals	The main reasons for the decline of the grey seal abundances in the 20th century were hunting and high concentrations of environmental pollutants. Also, by-catch is considered as one of the risks. Grey seal reproduction is not in good status, and they failed to achieve the threshold value of nutritional status.	Changes in ice conditions affect the status of Baltic grey seals. Food availability and quality (e.g., herring catch size) are important for their nutritional status. The prey fish quality could have delayed effects on grey seals, and continuous food limitation can influence population size.
Benthic habitats	Demersal trawling, dredging, shipping and nutrient inputs cause the majority of impacts on benthic habitats in the Baltic Sea. Oxygen deficiency, caused by both eutrophication and certain hydrographic conditions, is a widespread threat to coastal and estuarine communities. As a result of trawling, a shift from communities dominated by relatively high biomass species towards dominance by high abundances of small-sized organisms can occur.	Recovery time post trawling depend on species dispersal potential, longevity and habitat-specific requirements. The recovery time for species with a high dispersal potential and less habitat-specific requirements is <3 years, for longer-lived species with low dispersal potential and specific habitat requirements – up to 20 years. The status improvement of deep benthic habitats is linked to the time lag of recovery from eutrophication (30-40 years). Changes in climatic conditions could have a significant impact.

The climate change effects on achieving GES regarding the above-listed Descriptors, substances, and species were analyzed by reviewing the published scientific literature results. The factors analyzed were: changes in meteorological conditions (air temperature, humidity and precipitation, wind, runoff), seawater temperature and salinity, sea level, ice conditions, and stratification.

References

References – heavy metals

Ahlberg, G., Gustafsson, O., Wedel, P., 2006. Leaching of metals from sewage sludge during one year and their relationship to particle size. Environ Pollut 144, 545–53.

Babiarz, C., Hoffmann, S., Wieben, A., Hurley, J., Andren, A., Shafer, M., & Armstrong, D., 2012. Watershed and dis- charge influences on the phase distribution and tributary loading of total mercury and methylmercury into Lake Superior. Environmental Pollution 161, 299–310.

Babiarz, C.L., Hurley, J.P., Benoit, J.M., Shafer, M.M., Andren, A.W., and Webb, D.A., 1998. Seasonal influences on partitioning and transport of total and methylmercury in rivers from contrasting watersheds. Biogeochemistry 41, 237–257.

Bartnicki, J., A. Gusev, W. Aas, H. Fagerli & S. Valiyaveetil (2008): Atmospheric supply of nitrogen, lead, cadmium, mercury and dioxins/furans to the Baltic Sea in 2006. EMEP report to HELCOM. Available at: http://www.helcom.fi /environment2/ hazsubs/EMEP/en_GB/emep2006/

Bartnicki, J., A. Gusev, W. Aas, Gauss, M., & Jonson, J. E., 2017: Atmospheric supply of nitrogen, cadmium, mercury, lead, and PCDD/Fs to the Baltic Sea in 2015. EMEP report to HELCOM. https://www.emep.int/publ/helcom/2017/index.html

Bełdowska, M., Saniewska, D., Falkowska, L., 2014. Factors influencing variability of mercury input to the southern Baltic Sea. Marine Pollution Bulletin. 86, 283-290.

Bełdowska, M., Jędruch, A., Słupkowska, J., Saniewska, D., Saniewski, M., 2015. Macrophyta as a vector of contemporary and historical mercury from the marine environment to the trophic web. Environ Sci Pollut Res 22, 5228–5240. DOI 10.1007/s11356-014-4003-4

Bełdowska, M., Jędruch, A., Łęczyński, L., Saniewska, D., Kwasigroch, U., 2016. Coastal erosion as a source of mercury into the marine environment along the polish Baltic shore. Environmental Science and Pollution Research 23(16), 16372-16382.

Bełdowski, J., Miotk, M., Pempkowiak, J., 2009. Mercury fluxes through the sediment water interface and bioavailability of mercury in southern Baltic Sea sediments. Oceanologia. 51, 263-285

Benoit, J.M., Shull, D.H., Harvey, R.M., Beal, S.A., 2009. Effect of bioirrigation on sediment–water exchange of methylmercury in Boston Harbor, Massachusetts. Environ Sci Technol 43, 3669–3674.

Bieser, J. & Schrum, C., 2016. Impact of marine mercury cycling on coastal atmospheric mercury concentrations in the North- and Baltic Sea region. Elementa: Science of the Anthropocene 4, 000111. doi: 10.12952/journal.elementa.000111

Boening, D. W., 2000. Ecological effects, transport, and fate of mercury: A general review. Chemosphere 40, 1335-1351.

Burgess, N.M.; Bond, A.L.; Hebert, C.E.; Neugebauer, E.; Champoux, L. Mercury trends in herring gull (Larus argentatus) eggs from Atlantic Canada, 1972–2008: Temporal change or dietary shift? Environ. Pollut.
2013, 172, 216–222.

Carolin, C.F., Kumar, P.S., Saravanan, A., Joshiba, G.J., Naushad, M., 2017. Efficient techniques for the removal of toxic heavy metals from aquatic environment: a review. J. Environ. Chem. Eng. 5, 2782–2799. https://doi.org/10.1016/j.jece.2017.05.029

Consani, S., Ianni, M., C., Cutroneo, L., Dinellic, E., Carbonea, C., Capelloa, M., 2019. Environmental implications of metal mobility in marine sediments receiving input from a torrent affected by mine discharge. Marine Pollution Bulletin 139, 221-230.

Cossa, D.; Knoery, J.; Boye, M.; Marusczak, N.; Thomas, B.; Courau, P.; Sprovieri, F. Oceanic mercury concentrations on both sides of the Strait of Gibraltar decreased between 1989 and 2012. Anthropocene 2019,
29, 100230.
Damrat, M., Zaborska, A., & Zajączkowski, M., 2013. Sedimentation from suspension and sediment accumulation rate in the River Vistula prodelta, Gulf of Gdańsk (Baltic Sea). Oceanologia, 55(4), 937–950

Dang, D., H., Lenoble, V., Durrieu, G., Omanovic´, D., Mullot, J.-U., Mounier, S., Garnier, C., 2015. Seasonal variations of coastal sedimentary trace metals cycling: Insight on the effect of manganese and iron (oxy)hydroxides, sulphide and organic matter. Marine Pollution Bulletin 92, 113–124.

Driscoll, C. T., Holsapple, J., Schofield, C.L., Munson, R., 1998. The chemistry and transport of mercury in a small wetland in the Adirondack region of New York, USA. Biogeochemistry 40, 137-146.

Eckley, C. S., Branfireun, B., 2008. Mercury mobilization in urban stormwater runoff. Science of the Total Environment 403, 164-177.

Elbaz-Poulichet, F., Morley, N.H., Beckers, J.-M., Nomerange, P., 2001. Metal fluxes through the Strait of Gibraltar: the influence of the Tinto and Odiel rivers (SW Spain). Mar. Chem. 73, 193-213.

Gębka, K., Bełdowska, M., Saniewska, D., Korejwo, E., Saniewski, M., 2020. Meteorological phenomenon as a key factor controlling variability of labile particulate mercury in rivers and its inflow into coastal zone of the sea. Environmental Research 184, 109355. https://doi.org/10.1016/j.envres.2020.109355

Gębka, K., Bełdowska, M., Szymczak, E., Saniewska, D. 2019. Temporal changes in the content of labile and stabile mercury forms in soil and their inflow to the southern Baltic Sea. Ecotoxicology and Environmental Safety, 182, 109434

Granberg, M.E., Gunnarsson, J.S., Hedman, J.E., Rosenberg, R., Jonsson, P., 2008. Bioturbation-driven release of organic contaminants from Baltic Sea sediments mediated by the invading polychaete Marenzelleria neglecta. Environ. Sci. Technol. 42, 1058-1065.

Grigal,D. F., 2002. Inputs and outputs of mercury from terrestrial watersheds: a review. Environmental Reviews 10, 1–39.

Groll N., Grabemann I., Hünicke B., Meese M. 2017. Baltic Sea wave conditions under climate change scenarios. Boreal Env. Res. 22: 1–12.

Gusev, A., EMEP MSC-E, 2018a. Atmospheric deposition of heavy metals on the Baltic Sea. HELCOM Baltic Sea Environment Fact Sheets. Online. [15.07.2020] https://helcom.fi/baltic-sea-trends/environment-fact-sheets/

Gusev, A., EMEP MSC-E, 2018b. Atmospheric emissions of heavy metals in the Baltic Sea. HELCOM Baltic Sea Environment Fact Sheets. Online. [15.07.2020] https://helcom.fi/baltic-sea-trends/environment-fact-sheets/

Gworek, B., Bemowska-Kałabun, O., Kijeńska, M., Wrzosek-Jakubowska, J., 2016. Mercury in Marine and OceanicWaters—a Review. Water Air Soil Pollut 2016, 227-371

Hławiczka, S., 1998. Assessment of heavy metals emission to the air in Poland. Part II Emission in years 1980–1995. Arch Environ. Pollut. 24 (4), 91–108 (in polish).

Hedman, J.E., Tocca, J.S., Gunnarsson, J.S., 2009. Remobilization of polychlorinated biphenyl from Baltic Sea sediment: comparing the roles of bioturbation and physical resuspension. Environ. Toxicol. Chem. 28, 2241-2249.

HELCOM, 2010. Hazardous Substances in the Baltic Sea. An Integrated Thematic Assessment of Hazardous Substances in the Baltic Sea. Baltic Sea Environment Proceedings No. 120B.

HELCOM, 2018. Inputs of hazardous substances to the Baltic Sea. Baltic Sea Environment Proceedings No. 161. https://helcom.fi/media/publications/BSEP162.pdf

Jędruch, A., Kwasigroch, U., Bełdowska, M., Kuliński, K., 2017. Mercury in suspended matter of the Gulf of Gdańsk: Origin, distribution and transport at the land– sea interface. Marine Pollution Bulletin. 118(1-2), 354-367.

Josefsson, S., Leonardsson, K., Gunnarsson, J.S., Wiberg, K., 2010. Bioturbation- driven release of buried PCBs and PBDEs from different depths in contaminated sediments. Environ. Sci. Technol. 44, 7456-7464.

Knuuttila S., 2009. Waterborne inputs of heavy metals to the Baltic Sea. HELCOM Indicator Fact Sheet.

Koelmans, A.A., Jonker, M.T.O., 2011. Effects of black carbon on bioturbation-induced benthic fluxes of polychlorinated biphenyls. Chemosphere 84, 1150-1157.

Koelmans, A.A., Poot, A., Lange, H.J.D., Velzeboer, I., Harmsen, J., van Noort P.C.M., 2010. Estimation of in situ sediment-to-water fluxes of polycyclic aro- matic hydrocarbons, polychlorobiphenyls and polybrominated diphenylethers. Environ Sci Technol 44, 3014–3020

Komárek, M., Chrastný, V., Štíchová, J., 2007. Metal/metalloid contamination and isotopic composition of lead in edible mushrooms and forest soils originating from a smelting area. Environ Int 33, 677–84.

Komárek, M., Ettler, V., Chrastný , V., Mihaljevič, M., 2008. Lead isotopes in environmental sciences: a review. Environ. Int. 34, 562–577.

Kwasigroch, U., Bełdowska, M., Jędruch, A., Saniewska, D., 2018. Coastal erosion—A “new” land-based source of labile mercury to the marine environment. Environmental Science and Pollution Research. 25(28), 28682-28694.

Lawson, N. M., Mason, R.P., Laporte, J.M. (2001). The fate and transport of mercury, methylmercury, and other trace metals in Chesapeake Bay tributaries. Water Res 35, 501–515. https://doi.org/10.1016/S0043-1354(00)00267-0

Layglon, N., Misson, B., Durieu, G., Coclet, C., D'Onofrio, S., Dang, D.H., François, D., Mullot, J.-U., Mounier, S., Lenoble, V., Omanović, D., Garnier, C. 2020. Long-term monitoring emphasizes impacts of the dredging on dissolved Cu and Pb contamination along with ultraplankton distribution and structure in Toulon Bay (NW Mediterranean Sea, France). Mar. Pollution. Bull., 10.1016/j.marpolbul.2020.111196.

Leivuori, M., Jokòas, K., Petersell, V., Larsen, B., Floderus, S., 2000. Distribution of heavy metals in sediments of the Gulf of Riga, Baltic Sea. Boreal Environment Research 5, 165-185.

Mason, R. P., Choi, A. L., Fitzgerald, W. F., Hammerschmidt, C. R., Lamborg, C. H., Soerensen, A. L., & Sunderland, E. M., 2012. Mercury biogeochemical cycling in the ocean and policy implications. Environmental Research 119, 101–117

Mason, R.P., Lawson, N.M., Sullivan, K. A., 1997. The concentration, speciation and sources of mercury in Chesapeake Bay precipitation. Atmospheric Environment 31, 3541-3550. https://doi.org/10.1016/S1352-2310(97)00206-9

Molamohyeddin, N., Ghafourian, H., Sadatipour, S., M., 2017. Contamination assessment of mercury, lead, cadmium and arsenic in surface sediments of Chabahar Bay. Marine Pollution Bulletin 124, 521-525.

Munthe, J., Xiao, Z.F., Lindqvist, O. 1991. The aqueous reduction of divalent mercury by sulfite. Water Air Soil Pollut. 56, 621–630.

Munthe, J. 1992. The aqueous oxidation of elemental mercury by ozone. Atmos. Environ. 26A, 1461–1468.

Mustajärvi, L., Eek, E., Cornelissen, G, Eriksson-Wiklund, A.-K., Undeman, E., Sobek, A., 2017. In situ benthic flow-through chambers to determine sediment-to- water fluxes of legacy hydrophobic organic contaminants. Env. Pol. 231, 854-862.

Mustajärvi, L., Nybom, I., Eriksson-Wiklund, A.-K., Eek, E., Cornelissen, G., Sobek, A., 2019. How Important is Bioturbation for Sediment-to-Water Flux of Polycyclic Aromatic Hydrocarbons in the Baltic Sea? Environ. Toxicol. Chem. 38 (8), 1803-1810.

Nawrot, N., Matej-Łukowicz, K., Wojciechowska, E., 2019. Change in Heavy Metals Concentrations in Sediments Deposited in Retention Tanks in a Stream after a Flood. Pol. J. Environ. Stud. 28 (1), 9-14.

Nicolau, R., Lucas, Y., Merdy, P., Raynaud, M., 2012. Base flow and stormwater net fluxes of carbon and trace metals to the Mediterranean Sea by an urbanized small river. Water Res. 46 (20), 6625-6637.

Obrist, D., Johnson, D.W., Lindberg, S.E., Luo, Y., Hararuk, O., Bracho, R., Battles, J.J., Dail, D.B., Edmonds, R.L., Monson, R.K., Ollinger, S.V., Pallardy, S.G., Pregitzer, K.S., Todd, D.E., 2011. Mercury distribution across 14 U.S. forests. Part I: spatial patterns of concentrations in biomass, litter, and soils. Environ. Sci. Technol. 45, 3974–3981. https://doi.org/10.1021/es104384m

Orani, A. M., Vassileva, E., Schmidt, S., Berail. S., Barre, J. P. G., 2020. Temporal variation of trace elements, rare earth elements and Pb isotope ratios in sediment core from Kiel western Baltic Sea. Environmental Chemistry. Online. https://doi.org/10.1071/EN20078.

Oursel, B., Garnier, C., Pairaud, I., Omanovic´, D., Durrieu, G., Syakti, A.D., Le Poupon, C., Thouvenin, B., Lucas, Y., 2014. Behaviour and fate of urban particles in coastal waters: settling rate, size distribution and metals contamination characterization. Estuar. Coast. Shelf Sci. 138, 14–26.

Pacyna, J.M., Pacyna, E.G., 2000. Atmospheric emissions of anthropogenic lead in Europe: improvements, updates, histrorical data and projections, GKSS report nr 2000/31, Geesthacht, Germany.

Pirrone, N., Cinnirella, S., Feng, X., Finkelman, R. B., Friedli, H. R., Leaner, J.,Mason, R., Mukherjee, A. B., Stracher, G. B., Streets, D. G., & Telmer, K., 2010. Global mercury emissions to the atmosphere from anthropogenic and natural sources. Atmos. Chem. Phys. 10, 5951–5964.

Pohl, C., Löffler, A., Schmidt, M., Seifert, T., 2006. A trace metal (Pb, Cd, Zn, Cu) balance for surface waters in the eastern Gotland Basin, Baltic Sea. Journal of Marine Systems 60, 381–395.

Point, D.; Monperrus, M.; Tessier, E.; Amouroux, D.; Chauvaud, L.;Thouzeau,G.; Jean, F.;Amice,E.; Grall, J.; Leynaert, A.; Clavier, J.; Donard, O. F. X., 2007. Biological control of trace metal and organometal benthic fluxes in a eutrophic lagoon (Thau Lagoon, Mediterranean Sea, France). Estuarine Coastal Shelf. Sci. 72, 457–471.

Remeikaitė-Nikienėa, N., Garnaga-Budrėa, G., Lujanienėb, G., Jokšas, K., Stankevičius, A., Malejevas, V., Barisevičiūtė, R., 2018. Distribution of metals and extent of contamination in sediments from the south-eastern Baltic Sea (Lithuanian zone). Oceanologia 60(2), 193-206.

Rieuwerts, J., Farago, M., Cikrt, M., Bencko, V., 1999. Heavy metal concentrations in and around households near a secondary lead smelter. Environ Monit Assess 58, 317–35.

Rigaud, S., Radakovitch, O., Couture, R.-M., Deflandre, B., Cossa, D., Garnier, C., Garnier, J.-M., 2013. Mobility and fluxes of trace elements and nutrients at the sediment–water interface of a lagoon under contrasting water column oxygenation conditions. Appl. Geochem. 31, 35–51.

Saniewska, D., Bełdowska, M., Bełdowski, J., Saniewski, M., Szubska, M., Romanowski, A., Falkowska, L., 2014a. The impact of land use and season on the riverine transport of mercury into the marine coastal zone. Environmental Monitoring and Assessment 186(11), 7593-7604.

Saniewska, D., Bełdowska, M., Bełdowski, J., Jędruch, A., Saniewski, M., Falkowska, L., 2014b. Mercury loads into the sea associated with extreme flooding. Environmental Pollution. 191, 93-100.

Saniewska, D., Bełdowska, M., Bełdowski, J., Falkowska, L., 2014c. Mercury in precipitation at an urbanised coastal zone of the Baltic Sea (Poland). Ambio 43(7), 871-877.

Saniewska D, Bełdowska M, Bełdowski J, Saniewski M, Gębka K, Szubska M, Wochna, A., 2018. Impact of intense rains and flooding on mercury riverine input to the coastal zone. Marine Pollution Bulletin. 127, 593-602.

Saniewska, D., 2019a. Mercury Cycling in the Gulf of Gdańsk (Southern Baltic Sea) [Online First], IntechOpen, DOI: 10.5772/intechopen.86159.

Saniewska, D., Gębka, K., Bełdowska, M., Siedlewicz, G., Bełdowski J., 2019b. Impact of hydrotechnical works on outflow of mercury from the riparian zone to a river and input to the sea. Marine Pollution Bulletin. 142, 361-376

Scherbatskoy, T., Shanley, J.B., and Keeler, G.J., 1998. Factors controlling mercury transport in an upland forested catchment. Water Air Soil Pollut. 105, 427–438.

Schneider, B., Ceburnis, D., Marks, R., Munthe, J., Petersen, G., Sofiev, M., 2000. Atmospheric Pb and Cd input into the Baltic Sea: a new estimate based on measurements. Mar. Chem. 71, 297–307.

Schroeder, W.H., and Munthe, J. 1998. Atmospheric mercury — an overview. Atmos. Environ. 32, 809–822.

Semeena, V. S., Bartnicki, J., Gusev, A., Aas, W., 2014. Atmospheric supply of nitrogen, lead, cadmium, mercury and dioxins/furans to the Baltic Sea in 2014. EMEP Centers Joint Report for HELCOM EMEP/MSC-W Technical Report MSC-W 2/2014, Oslo.

Senze, M., Kowalska-Góralska, M., Pokorny, P., Dobicki, W., Polechoński, R., 2015. Accumulation of Heavy Metals in Bottom Sediments of Baltic Sea Catchment Rivers Affected by Operations of Petroleum and Natural Gas Mines in Western Pomerania, Poland. Pol. J. Environ. Stud. 24 (5), 2167-2175. doi: 10.15244/pjoes/40273

Skyllberg, U., Westin, M.B., Meili, M., Bjorn, E., 2009. Elevated concentrations of methyl mercury in streams after forest clear-cut: a consequence of mobilization from soil or new methylation? Environ. Sci. Technol. 43 (22), 8535–8541.

Superville, P.-J., Prygiel, E., Magnier, A., Lesven, L., Gao, Y., Baeyens, W., Ouddane, B., Dumoulin, D., Billon, G., 2014. Daily variations of Zn and Pb concentrations in the Deûle River in relation to the resuspension of heavily polluted sediments. Science of the Total Environment 470-471, 600-607.

Wängberg, I., Schmolke, S., Schager, P., Munthe, J., Ebinghaus, R., Iverfeldt, Å., 2001. Estimates of air-sea Exchange of Mercury in the Baltic Sea. Atmospheric Environment 35, 5477–5484.

Yuan, Q., Valsaraj, K.T., Reible, D.D., Willson, C.S., 2007. A laboratory study of sediment and contaminant release during gas ebullition. J. Air Waste Manag. Assoc. 57, 1103-1111.

Yurkovskis, A., Poikāne, R., 2008. Biogeochemical, physical and anthropogenic transformations in the Daugava River estuary and plume, and the open Gulf of Riga (Baltic Sea) indicated by major and trace elements. Journal of Marine Systems 70, 77–96.

Zaborska, A., 2014. Anthropogenic lead concentrations and sources in Baltic Sea sediments based on lead isotopic composition. Marine Pollution Bulletin 85, 99-113.

Zaborska, A., Siedlewicz, G., Szymczycha, B., Dzierzbicka-Głowacka, L., Pazdro, K., 2019. Legacy and emerging pollutants in the Gulf of Gdańsk (southern Baltic Sea) – loads and distribution revisited. Marine Pollution Bulletin 139, 238–255.

References – PFOS and other PFASs

3M, 2000. Letter to US EPA, Re: Phase-out Plan for POSF-Based Products (226-0600). US EPA Administrative Record. vol. 226 pp. 1–11.

Ahrens, L., 2011. Polyfluoroalkyl compounds in the aquatic environment: A review of their occurrence and fate. J. Environ. Monit. 13, 20–31.

Ahrens, L., Norström, K., Viktor, T., Cousins, A.P., Josefsson, S., 2015. Stockholm Arlanda Airport as a source of per- and polyfluoroalkyl substances to water, sediment and fish. Chemosphere 129, 33-38.

Becker, A.M., Gerstmann, S., Frank, H., 2008. Perfluorooctane surfactants in wastewaters, the major source of river pollution, Chemosphere 72, 115–121.

Blum, A., Balan, S.A., Scheringer, M., Trier, X., Goldenman, G., Cousins, I.T., Diamond, M., Fletcher, T., Higgins, C., Lindeman, A.E., Peaslee, G., de Voogt, P., Wang, Z., Weber, R., 2015. The Madrid statement on poly- and perfluoroalkyl substances (PFASs). Environ. Health Perspect. 123, 107–111.

Buck, R. C., Franklin, J., Berger, U., Conder, J. M., Cousins, I.T., de Voogt, P., Jensen, A. A., Kannan, K., Mabury, S. A. and van Leeuwen, S. P., 2011. Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification, and origins, Integr. Environ. Assess. Manage. 7, 513–541.

Castiglioni, S., Valsecchi, S., Polesello, S., Rusconi, M., Melis, M., Palmiotto, M., Manenti, A., Davoli, E., Zuccato, E., 2015. Sources and fate of perfluorinated compounds in the aqueous environment and in drinking water of a highly urbanized and industrialized area in Italy. Journal of Hazardous Materials 282, 51–60. http://dx.doi.org/10.1016/j.jhazmat.2014.06.007

Conder, J.M., Hoke, R.A., de Wolf, W., Russell, M.H., Buck., R.C., 2008. Are PFCAs bioaccumulative? A critical review and comparison with regulatory criteria and persistent lipophilic compounds. Environ Sci Technol 42, 995–1003.

Cousins, I.T., Goldenman, G., Herzke, D., Lohmann, R., Miller, M., Ng, C.A., Patton, S., Scheringer, M., Trier, X., Vierke, L., Wang, Z., DeWitt, J.C., 2019. The concept of essential use for determining when uses of PFASs can be phased out. Environ. Sci. Process. Impacts 21, 1803–1815.

Crone, B. C., Speth, T. F., Wahman, D. G., Smith, S. J., Abulikemu, G., Kleiner, E. J., Pressman, J. G., 2019. Occurrence of per- and polyfluoroalkyl substances (PFAS) in source water and their treatment in drinking water. Critical reviews in environmental science and technology 49(24), 2359–2396.

Danish EPA, 2015. Short Chain Polyfluoralkyl Substances (PFAS). A literature review of information on human health effects and environmental fate and effect aspects of short-chain PFAS. Environmental Project No. 1707; Danish EPA: København, Denmark.

Dauchy, X., Boiteux, V., Bach, C., Colin, A., Hemard, J., Rosin, C., Munoz, J.-F., 2017. Mass flows and fate of per- and polyfluoroalkyl substances (PFASs) in the wastewater treatment plant of a fluorochemical manufacturing facility. Science of the Total Environment 576, 549–558.

ECHA European Chemicals Agency, 2017. Justification for the selection of a substance for CoRAP inclusion. Ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)propanoate. (15 June, 2020)

ECHA European Chemicals Agency, 2019. Inclusion of substances of very high concern in the Candidate List for eventual inclusion in Annex XIV. https://echa.europa.eu/documents/10162/13638/ED_71_2019_12828_en.pdf/a661cbe6-c805-b9c7-1287-53dee5cd1df5 (15 June, 2020)

Ericson Jogsten, I., M. Nadal, van Bavel, B., Lindström, G., Domingo, J., L., 2012. Per- and polyfluorinated compounds (PFCs) in house dust and indoor air in Catalonia, Spain: Implications for human exposure. Environment International 39, 172–180.

Eschauzier, C., Beerendonk, E., Scholte-Veenendaal, P., De Voogt, P., 2012. Impact of treatment processes on the removal of perfluoroalkyl acids from the drinking water production chain. Environ Sci Technol 46(3), 1708–1715. doi:10.1021/es201662b.

Feng, X., Ye, M., Li, Y., Zhou, J., Sun, B., Zhu, Y., Zhu, L., 2020. Potential sources and sediment-pore water partitioning behaviors of emerging per/polyfluoroalkyl substances in the South Yellow Sea. Journal of Hazardous Materials 389, 122124. https://doi.org/10.1016/j.jhazmat.2020.122124

Filipovic, M., Berger, U., McLachlan, M.S., 2013. Mass balance of perfluoroalkyl acids in the Baltic Sea. Environ. Sci. Technol. 47, 4088–4095. https://doi.org/10.1021/es400174y

Filipovic, M., Laudon, H., McLachlan, M.S., Berger, U., 2015. Mass balance of perfluorinated alkyl acids in a pristine boreal catchment. Environ. Sci. Technol. 49, 12127–12135. https://doi.org/10.1021/acs.est.5b03403

Fujii, Y., Harada, K.H., Koizumi, A. (2013). Occurrence of perfluorinated carboxylic acids (PFCAs) in personal care products and compounding agents. Chemosphere 93, 538-544.

Gallen, C., Eaglesham, G., Drage, D., Hue Nguyen, T., Mueller, J. F., 2018. A mass estimate of perfluoroalkyl substance (PFAS) release from Australian wastewater treatment plants. Chemosphere 208, 975-983. https://doi.org/10.1016/j.chemosphere.2018.06.024

Gao, L., Liu, J., Bao, K., Chen, N., Meng, B., 2020. Multicompartment occurrence and partitioning of alternative and legacy per- and polyfluoroalkyl substances in an impacted river in China. Science of the Total Environment 729, 138753. https://doi.org/10.1016/j.scitotenv.2020.138753 0048-9697

Gebbink, W. A., Bignert, A. & Berger, U., 2016. Perfluoroalkyl acids (PFAAs) and selected precursors in the Baltic Sea environment: Do precursors play a role in food web accumulation of PFAAs? Environ. Sci. Technol. 2016, 50, 6354−6362. https://doi.org/10.1021/acs.est.6b01197

Giesy, J.P. & Kannan, K., 2001. Global distribution of perfluorooctane sulfonate in wildlife. Environ Sci Technol 35, 1339–1342.

Guo, R., Sim, W.-J., Lee, E.-S., Lee, J.-H., Oh, J.-E., 2010. Evaluation of the fate of perfluoroalkyl compounds in wastewater treatment plants. Water Res. 44, 3476-3486.

Harrad, S., Wemken, N., Drage, D. S., Abdallah, M., A.-E., Coggins, A.-M., 2019. Perfluoroalkyl substances in drinking water, indoor air and dust from Ireland: implications for human exposure. Environ. Sci. Technol. 53, 13449−13457. https://doi.org/10.1021/acs.est.9b04604

Heydebreck, F., Tang, J.H., Xie, Z.Y., Ebinghaus, R., 2015. Alternative and legacy perfluoroalkyl substances: differences between European and Chinese River/Estuary Systems. Environ. Sci. Technol. 49, 14742–14743

Ji, B., Kang, P., Wei, T., Zhao, Y., 2020. Challenges of aqueous per- and polyfluoroalkyl substances (PFASs) and their foreseeable removal strategies. Chemosphere 250, 126316. https://doi.org/10.1016/j.chemosphere.2020.126316

Joerss, H., Apel, C., Ebinghaus, R., 2019. Emerging per- and polyfluoroalkyl substances (PFASs) in surface water and sediment of the North and Baltic Seas. Science of the Total Environment 686, 360–369. https://doi.org/10.1016/j.scitotenv.2019.05.363

Johansson, J. and Undeman, E. 2020. Perfluorooctane sulfonate (PFOS) and other perfluorinated alkyl substances (PFASs) in the Baltic Sea – Sources, transport routes and trends. Helcom Baltic Sea Environment Proceedings n°173. https://helcom.fi/wp-content/uploads/2020/06/Helcom_173_PFOS_PFAS.pdf

Junttila, V., Vähä, E., Perkola, N., Räike, A., Siimes, K., Mehtonen, J., Kankaanpää, H., Mannio, J., 2019. PFASs in Finnish rivers and fish and the loading of PFASs to the Baltic Sea. Water, 11, 870. https://doi.org/10.3390/w11040870.

KEMI, 2015. The Swedish Chemicals Agency (KEMI). Occurrence and use of highly fluorinated substances and alternatives. Report from a government assignment. https://www.kemi.se/en/global/rapporter/2015/report-7-15-occurrence-and-use-of-highly-fluorinated-substances-and-alternatives.pdf

Kibambe, M. G., Momba, M. N. B., Daso, A. P., Coetzee, M. A. A., 2020. Evaluation of the efficiency of selected wastewater treatment processes in removing selected perfluoroalkyl substances (PFASs). Journal of Environmental Management 255, 109945. https://doi.org/10.1016/j.jenvman.2019.109945

Koponen, J., Airaksinen, R., Hallikainen, A, Vuorinen, P. J., Mannio, J., Kiviranta, H., 2015. Perfluoroalkyl acids in various edible Baltic, freshwater, and farmed fish in Finland. Chemosphere 129, 186–191. https://doi.org/10.1016/j.chemosphere.2014.08.077

Loos, R., Carvalho, R., Antonio, D.C., Comero, S., Locoro, G., Tavazzi, S., Paracchini, B., Ghiani, M., Lettieri, T., Blaha, L., Jarosova, B., Voorspoels, S., Servaes, K., Haglund, P., Fick, J., Lindberg, R. H., Schwesig, D., Gawlik, B. M., 2013. EU-wide monitoring survey on emerging polar organic contaminants in wastewater treatment plant effluents. Water Res. 47, 6475–6487.

Nguyen, M.A., Wiberg, K., Ribeli, E., Josefsson, S., Futter, M., Gustavsson, J., Ahrens, L., 2017. Spatial distribution and source tracing of per- and polyfluoroalkyl substances (PFASs) in surface water in Northern Europe. Environ. Pollut. 220, 1438–1446. https://doi.org/10.1016/j.envpol.2016.10.089Get rights and content

Norström, K., Viktor, T., Cousins, A. P. & Rahmberg, M., 2015. Risks and Effects of the dispersion of PFAS on Aquatic, Terrestrial and Human populations in the vicinity of International Airports. IVL Report B2232.

Olsen, G. W., Chang, S.-C., Noker, P. E., Gorman, G. S., Ehresman, D. J., Lieder, P. H., Butenhoff, J. L., 2009. A comparison of the pharmacokinetics of perfluorobutanesulfonate (PFBS) in rats, monkeys, and humans. Toxicology 256, 65–74.

Organisation for Economic Cooperation and Development (OECD), 2015. OECD environment, health and safety publications series on risk management no. 30: working towards a global emission inventory of PFASs: focus on PFCAs - status quo and theway forward. Heal. Saf. Publ. Ser. Risk Manag. 30, 33.
SC Secretariat, 2019. (8.07.2020) http://chm.pops.int/Implementation/Alternatives/AlternativestoPOPs/ChemicalslistedinAnnexB/Perfluorooctanesulfonicacidandperfluorooctane/tabid/5869/Default.aspx

Paul, A. G., Jones, K. C. & Sweetman, A. J., 2009. A first global production, emission, and environmental inventory for perfluorooctane sulfonate. Environ. Sci. Technol. 43, 386–392.

Prevedouros, K., Cousins, I.T., Buck, R.C., Korzeniowski, S.H., 2006. Sources, fate and transport of perfluorocarboxylates. Environ. Sci. Technol. 40 (1), 32–44. https://doi.org/ 10.1021/es0512475

SC Secretariat, 2017. Stockholm Convention Secretariat. The 16 new POPs. http://chm.pops.int/Portals/0/download.aspx?d=UNEP-POPS- PUB-Brochure-16NewPOPs-201706.English.pdf, Accessed date: 18 June 2020.

SC Secretariat, 2019. Stockholm Convention Secretariat. SC-9/12: Listing of perfluorooctanoic acid (PFOA), its salts and PFOA-related compounds. Accessed date: 18 June 2020.

Scheringer, M., Trier, X., Cousins, I.T., de Voogt, P., Fletcher, T., Wang, Z., Webster, T.F., 2014. Helsingør statement on poly- and perfluorinated alkyl substances (PFASs). Chemosphere 114, 337–339. https://doi.org/10.1016/j.chemosphere.2014.05.044.

Shoeib, M., Harner, T., Webster, G. M., Lee, S. C., 2011. Indoor sources of poly- and perfluorinated compounds (PFCS) in Vancouver, Canada: implications for human exposure. Environ Sci Technol 45(19), 7999–8005. doi:10.1021/es103562v.

Sonne, C., Vorkamp, K., Galatius. A., Kyhn, L., Teilmann, J., Bossi, R., Søndergaard, J., Eulaers, I., Desforges, J.-P., Siebert, U., Dietz, R., 2019. Human exposure to PFOS and mercury through meat from baltic harbour seals (Phoca vitulina). Environmental Research 175, 376-383. https://doi.org/10.1016/j.envres.2019.05.026

Strynar, M. J, Lindström, A. B., Nakayama, S. F., Egeghy, P. P., Helfant, L. J., 2012. Pilot scale application of a method for the analysis of perfluorinated compounds in surface soils. Chemosphere 86(3), 252–257. doi:10.1016/j.chemosphere.2011.09.036

Sun J., Bossi, R., Bustnes, J.O., Helander, B., Boertmann, D., Dietz, R., Herzke, D., Jaspers, V.L.B., Labansen, A.L., Lepoint, G., Schulz, R., Sonne, C., Thorup, K., Tøttrup, A.P., Zubrod, J.P., Eens, M., Eulaers, I., 2019. White-tailed eagle (Haliaeetus albicilla) body feathers document spatiotemporal trends of perfluoroalkyl substances in the northern environment. Environ. Sci. Technol., 53, 12744−12753.

de la Torre, A., Navarro, I., Sanz, P., de los Ángeles Mártinez, M., 2019. Occurrence and human exposure assessment of perfluorinated substances in house dust from three European countries. Science of the Total Environment 685, 308–314.

Trier, X., Granby, K., Christensen, J. H., 2011. Polyfluorinated surfactants (PFS) in paper and board coatings for food packaging. Environ Sci Pollut Res Int 18(7), 1108–1120. doi:10.1007/ s11356-010-0439-3

United Nations Environment Programme (UNEP), 2017. Draft. Guidance for the Inventory of Perfluorooctane Sulfonic Acid (PFOS) and Related Chemicals Listed Under Stockholm Convention on Persistent Organic Pollutants. (8.07.20)

Vieno, N., 2014 Hazardous Substances at Wastewater Treatment Project Report (In Finnish); Vesilaitosyhdistyksen monistesarja nro 34; Finnish Water Utilities Association: Helsinki, Finland.

Wang, Z., Cousins, I.T., Scheringer,M., Buck, R.C., Hungerbühler, K., 2014. Global emission inventories for C4–C14 perfluoroalkyl carboxylic acid (PFCA) homologues from1951 to 2030, part I: production and emissions fromquantifiable sources. Environ. Int. 70, 62–75. https://doi.org/10.1016/j.envint.2014.04.013.

Wang, S.W., Huang, J., Yang, Y., Hui, Y.M., Ge, Y.X., Larssen, T., Yu, G., Doeng, S., Wang, B., Harman, C., 2013. First report of a chinese PFOS alternative overlooked for 30 years: its toxicity, persistence, and presence in the environment. Environ. Sci. Technol. 47, 10163–10170

Xiao, F., et al., 2012. Input characterization of perfluoroalkyl substances in wastewater treatment plants: source discrimination by exploratory data analysis. Water Res. 46 (9), 3101-3109.

Xiao, F., 2017. Emerging poly- and perfluoroalkyl substances in the aquatic environment: a review of current literature. Water Res. 124, 482-495. https://doi.org/10.1016/j.watres.2017.07.024

Zacs, D. & Bartkevics, V., 2016. Trace determination of perfluorooctane sulfonate and perfluorooctanoic acid in environmental samples (surface water, wastewater, biota, sediments, and sewage sludge) using liquidchromatography – Orbitrap mass spectrometry. Journal of Chromatography A 1473, 109–121. https://doi.org/10.1016/j.chroma.2016.10.060

References – TBT, organotins

Antizar-Ladislao, B., 2008. Environmental levels, toxicity and human exposure to tributyltin (TBT)-contaminated marine environment. A review. Environ Int 34, 292–308.

Batista-Andrade, J., A., Caldas, S., S., Batista, R., M., Castro, I., B, Fillmann, G., Primel, E., G., 2018. From TBT to booster biocides: Levels and impacts of antifouling along coastal areas of Panama. Environmental Pollution 234, 243-252.

Berg, M., Arnold, C., G., Müller, R., S., Mühlemann, J., Schwarzenbach, R., P. (2001). Sorption and desorption behavior of organotin compounds in sediment-pore water systems. Environ. Sci. Technol. 35, 3151–3157.

Briant, N., Bancon-Montigny, C., Freydier, R., Delpoux, S., Elbaz-Poulichet, F., 2016. Behaviour of butyltin compounds in the sediment pore waters of a contaminated marina (Port Camargue, South of France). Chemosphere 150, 123-129.

Burton, E., D., Phillips, I., R., Hawker, D., W., 2004. Sorption and desorption behavior of tributyltin with natural sediments, Environ. Sci. Technol. 38, 6694–6700.

Cornelissen, G., Pettersen, A., Nesse, E., Eek, E., Helland, A., Breedveld, G., D., 2008. The contribution of urban runoff to organic contaminant levels in harbour sediments near two Norwegian cities. Mar Pollut Bull 56, 565–573.

De Mora, S.J., Pelletier, E., 1997. Environmental tributyltin research: past, present, future. Environ. Sci. Technol. 18, 1169-1177.

de Oliveira, C., R., dos Santos, D., dos Santos Madureira L., A., de Marchi, M., R., R., 2010. Speciation of butyltin derivatives in surface sediments of three southern Brazilian harbors. J. Hazard. Mater. 181, 851–856.
IMO (2001) International convention on the control of harmful antifouling systems on ships. International Maritime Organization. http://www.imo.org

Dowson, P.,H., Bubb, J., M., Lester, J.,N., 1996. Persistence and degradation pathways of tributyltin in freshwater and estuarine sediments. Estuar Coast Shelf Sci 42, 551–562.

Egardt, J., Nilsson, P., Dahllöf, I., 2017. Sediments indicate the continued use of banned antifouling compounds. Marine Pollution Bulletin 125, 282–288.

Eklund, B., Elfström, M., Gallego, I., Bengtsson, B.-E., Breitholtz, M., 2010. Biological and chemical characterization of harbour sediments from the Stockholm area. Soil Sed Pollut 10, 127–141

Eklund, B. & Watermann. B., 2018. Persistence of TBT and copper in excess on leisure boat hulls around the Baltic Sea. Environmental Science and Pollution Research 25, 14595–14605. https://doi.org/10.1007/s11356-018-1614-1

Eklund, B. & Eklund, D., 2014. Pleasure boat yard soils are often highly contaminated. Environ Manag 53, 930–946.
http://www.springerlink.com/openurl.asp?genre=article&id=doi:10.1007/s00267-014-0249-3

Eklund, B., Hansson, T., Bengtsson, H., Eriksson Wiklund, A.-K., 2016. Pollutant concentrations and toxic effects on the red alga Ceramium tenuicorne of sediments from natural harbors and small boat harbors on the west coast of Sweden. Arch Environ Contam Toxicol 70, 583–594.

Eklund, B., Elfström, M., Borg, H., 2008. TBT originates from pleasure boats in Sweden in spite of firm restrictions. Open Environ Sci 2, 124–132.

Eklund, B., Johansson, L., Ytreberg, E., 2014. Contamination of a boatyard for maintenance of pleasure boats. JSS. doi:10.1007/ s11368-013-0828-6

EU (2003). Regulation (EC) No 782/2003 ofthe European Parliament and of the Council of 14 April 2003 on the prohibition of organotin compounds on ships. Official Journal ofthe European Union, L 115/ 1, 09.05.2003

Fent, K., 2006. Worldwide occurrence of organotins from antifouling paints and effects in the aquatic environment. Handb Environ Chem 5, 71–100.

Filipkowska, A., Kowalewska, G., Pavoni, B., Leczynski, L., 2011. Organotin compounds in surface sediments from seaports on the Gulf of Gdansk (southern Baltic coast), Environ. Monit. Assess. 182, 455–466.

Furdek, M., Mikac, N., Bueno, M., Tessier, E., Cavalheiro, J., Monperrus, M., 2016. Organotin persistence in contaminated marine sediments and porewaters: in situ degradation study using species-specific stable isotopic tracers. J. Hazard Mater. 307, 263–273.

Gadd, G., M., 2000. Microbial interactions with tributyltin compounds: detoxification, accumulation, and environmental fate. The Science of the Total Environment 258, 119-127.

HELCOM, 2017. TBT and imposex. HELCOM core indicator report. July 2017. ISSN 2343-2543

Hoch, M., 2001. Organotin compounds in the environment - an overview. Applied Geochemistry 16, 719-743.
Lee, C.-C., Hsieh, C.-Y., Tien, C.-J., 2006. Factors influencing organotin distribution in diferent marine environmental compartments, and their potential health risk. Chemosphere 65, 547–559.

Horie, Y., Yamagishi, T., Shintaku, Y., Iguchi, T., Tatarazako, N., 2018. Effects of tributyltin on early life-stage, reproduction, and gonadal sex differentiation in Japanese medaka (Oryzias latipes). Chemosphere 203, 418–425.

Jacobson, T., Sundelin, B., Yang, G., Ford, A., T., 2011. Low doose TBT exposure decreases amphipod immunocompetence and reproductive fitness. Aquatic Toxicology 101, 72-77.

Jokšas, K., Stakėnienė, R., Raudonytė-Svirbutavičienė, E., 2019. On the effectiveness of tributyltin ban: Distribution and changes in butyltin concentrations over a 9-year period in Klaipėda Port, Lithuania. Ecotoxicology and Environmental Safety 183, 109515.

Lagerström, M., Strand, J.,Eklund, B.,Ytreberg, E., 2017. Total tin and organotin speciation in historic layers of antifouling paint on leisure boat hulls. Environmental Pollution 220, 1333-1341.

Lofrano, G., Libralato, G., Alfieri, A., Carotenuto, M., 2016. Metals and tributyltin sediment contamination along the Southeastern Tyrrhenian Sea coast. Chemosphere 144, 399–407.

Michaud, M.-H., Pelletier, E., 2006. Sources and fate of butyltins in the St. Lawrence Estuary ecosystem. Chemosphere 64, 1074–1082.

Nakanishi T., 2007. Potential toxicity of organotin compound via nuclear receptor signalling in mammals. J Health Sci 53, 1–9.

Negri, A., Marshall, P., 2009. TBT contamination of remote marine environments: Ship groundings and ice-breakers as sources of organotins in the Great Barrier Reef and Antarctica. J Environ Manage 90, S31–S40.

Omae, I., 2003. Organotin antifouling paints and their alternatives, Appl. Organomet. Chem. 17, 81–105.

Paz-Villarraga, C., A., Castro, I., B., Miloslavich, P., Fillmann, G., 2015. Venezuelan Caribbean Sea under the threat of TBT. Chemosphere 119, 704–710

Radke, B.,Łęczyński, L., Wasik, A., Namieśnik, J., Bolalek, J., 2008. The content of butyl- and phenyltin derivatives in the sediment from the Port of Gdansk. Chemosphere 73, 407–414.

Raudonyte-Svirbutaviciene, E., Neagu, A., Vickackaite, V., Jasulaitiene, V., Zarkov, A., Tai, C.-W., Katelnikovas, A., 2018. Two-step photochemical inorganic approach to the synthesis of Ag-CeO2 nanoheterostructures and their photocatalytic activity on tributyltin degradation. J. Photochem. Photobiol., A 351, 29–41.

Rodriguez-Cea, A., Rodriguez-Gonzalez, P., Garcia Alonso, J.I., 2016. Study of the degradation of butyltin compounds in surface water samples under different storage conditions using multiple isotope tracers and GC-MS/MS. Environ. Sci. Pollut. Res. Int. 23 (5), 4876–4885.

Rodriguez-Gonzalez, P., Bouchet, S., Monperrus, M., Tessier, E., Amouroux, D., 2013. In situ experiments for element species-specific environmental reactivity of tin and mercury compounds using isotopic tracers and multiple linear regression. Environ Sci Pollut Res 20, 1269–1280. doi:10.1007/ s11356-012-1019-5

Roots, O. & Roose, A., 2013. Hazardous substances in the aquatic environment of Estonia. Chemosphere 93, 196–200.

Ruczyńska, W., Szlinder-Richert, J. and Drgas, A., 2016. The occurrence of endocrine disrupting compounds in off-shore sediments from the southern Baltic Sea. Environ. Sci.: Processes Impacts, 18, 1193.

Suzdalev, S., Gulbinskas, S., Blažauskas, N., 2015. Distribution of tributyltin in surface sediments from transitional marine-lagoon system of the south-eastern Baltic Sea, Lithuania. Environ Sci Pollut Res 22, 2634–2642.

Sunday, A.O., Alafara, B.A., Oladele, O.G., 2012. Toxicity and speciation analysis of organotin compounds. Chem. Speciat. Bioavailab. 24 (4), 216-226.

Takahashi, S., Tanabe, S., Takeuchi, I., Miyazaki, N., 1999. Distribution and specific bioaccumulation of butyltin compounds in a marine ecosystem. Arch. Environ. Contam. Toxicol. 37, 50–61.

Takeuchi, I., Takahashi, S., Tanabe, S., Miyazaki, N., 2004. Butyltin concentrations along the Japanese coast from 1997 to 1999 monitored by Caprella spp. (Crustacea: Amphipoda). Mar Environ Res 57, 397–414.

Viglino, L., Pelletier, E., St-Louis, R., 2004. Highly persistent butyltins in northern marine sediments: a long-
term threat for the Saguenay Fjord (Canada). Environ. Toxicol. Chem. 23 (11), 2673–2681.

Turner, A., Glegg, G. (2014). TBT-based antifouling paints remain on sale. Mar Pollut Bull 88, 398–400.

References – Pharmaceuticals

Arnold, K., E., Brown, A., R., Ankley, G., T., Sumpter, J., P., 2014. Medicating the environment: assessing risks of pharmaceuticals to wildlife and ecosystems. Phil. Trans. R. Soc. B 369: 20130569. http://dx.doi.org/10.1098/rstb.2013.0569

Aus der Beek, T., Weber, F.-A., Bergmann, A., Hickmann, S., Ebert, I., Hein, A., 2016. Pharmaceuticals in the environment – global occurrences and perspectives. Environmental Toxicology and Chemistry 35 (4), 823–835.

Bagheri, H., Afkhami, A. and Noroozi. A., 2016. Removal of pharmaceutical compounds from hospital wastewater using nanomaterials: a review. Anal. Bioanal. Chem. Res. 3 (1), 1-18.

Biel-Maeso, M., Baena-Nogueras, R., M., Corada-Fernández, C., Lara-Martín, P., A.,2018. Occurrence, distribution and environmental risk of pharmaceutically active compounds (PhACs) in coastal and ocean waters from the Gulf of Cadiz (SW Spain). Science of the Total Environment 612, 649–659.

Björlenius, B., Ripszám, M., Haglund, P., Lindberg, R., H., Tysklind, M., Fick, J., 2018. Pharmaceutical residues are widespread in Baltic Sea coastal and offshore waters – Screening for pharmaceuticals and modelling of environmental concentrations of carbamazepine. Science of the Total Environment 633, 1496–1509.

Bollmann, U., E., Simon, M., Vollertsen, J., Bestera, K., 2019. Assessment of input of organic micropollutants and microplastics into the Baltic Sea by urban waters. Marine Pollution Bulletin 148, 149–155.

Bonnefille, B., Gomez, E., Courant, F., Escande, A., Fenet, H., 2018. Diclofenac in the marine environment: A review of its occurrence and effects. Marine Pollution Bulletin 131, 496–506.

Borecka, M., Białk-Bielińska, A., Siedlewicz, G., Kornowska, K., Kumirska, J., Stepnowski, P., Pazdro, K., 2013. A new approach for the estimation of expanded uncertainty of results of an analytical method developed for determining antibiotics in seawater using solid-phase extraction disks and liquid chromatography coupled with tandem mass spectrometry technique. J. Chromatogr. A 1304, 138–146.

Borecka, M., Siedlewicz, G., Haliński, L., P., Sikora, K., Pazdro, K., Stepnowski, P., Białk-Bielińska, A., 2015. Contamination of the southern Baltic Sea waters by the residues of selected pharmaceuticals: Method development and field studies. Marine Pollution Bulletin 94, 62–71.

Carvalho, I.T., Santos, L., 2016. Antibiotics in the aquatic environments: a review of the European scenario. Environ. Int. 94, 736–757. http://dx.doi.org/10.1016/j.envint. 2016.06.025

Desbiolles, F., Malleret, L., Tiliacos, C., Wong-Wah-Chung, P., Laffont-Schwob, I., 2018. Occurrence and ecotoxicological assessment of pharmaceuticals: is there a risk for the Mediterranean aquatic environment? Sci. Total Environ. https://doi.org/10.1016/j. scitotenv.2018.04.351.

Fang, W., Peng, Y., Muir, D., Lind, J., Zhang, X., 2019. A critical review of synthetic chemicals in surface waters of the US, the EU and China. Environment International 131, 104994. https://doi.org/10.1016/j.envint.2019.104994

Fatta-Kassinos, D., Meric, S., Nikolaou, A., 2011. Pharmaceutical residues in environmental waters and wastewater: current state of knowledge and future research. Anal. Bioanal. Chem. 399, 251-275.

Fernández-López, C., Guillén-Navarro, J.M., Padilla, J.J., Parsons, J.R., 2016. Comparison of the removal efficiencies of selected pharmaceuticals in wastewater treatment plants in the region of Murcia, Spain. Ecol. Eng. 95, 811–816.

Fisch, K., Waniek, J., J., Schulz-Bull, D., E., 2017. Occurrence of pharmaceuticals and UV-filters in riverine run-offs and waters of the German Baltic Sea. Marine Pollution Bulletin 124, 388-399.

Gros, M., Ahrens, L., Levén, L., Koch, A., Dalahmeh, S., Ljung, E., Lundin, G., Jönsson, H., Eveborn, D., Wiberg, K., 2020. Pharmaceuticals in source separated sanitation systems: Fecal sludge and blackwater treatment. Science of the Total Environment 703, 135530. https://doi.org/10.1016/j.scitotenv.2019.135530

HELCOM, 2018. Diclofenac. HELCOM pre-core indicator report. Online. [28.04.20], https://helcom.fi/wp-content/uploads/2019/08/Diclofenac-HELCOM-pre-core-indicator-2018.pdf

IFPMA (International Federation of Pharmaceutical Manufactures & Associations), 2017. The Pharmaceutical Industry and Global Health: Facts and Figures 2017. ISBN: 978-2-940498-50-5.

Klatte, S., Schaefer, H.-Ch., Hempel, M., 2017. Pharmaceuticals in the environment — a short review on options to minimize the exposure of humans, animals and ecosystems. Sustain. Chem. Pharm. http://dx.doi.org/10.1016/j.scp.2016.07.001.

Kowalska, K., Maniakova, G., Carotenuto, M., Sacco, O., Vaiano, V., Lofrano, G., Rizzo, L., 2020. Removal of carbamazepine, diclofenac and trimethoprim by solar driven advanced oxidation processes in a compound triangular collector-based reactor: A comparison between homogeneous and heterogeneous processes. Chemosphere 238, 124665. https://doi.org/10.1016/j.chemosphere.2019.124665

Kunkel, U. & Radke, M., 2012. Fate of pharmaceuticals in rivers: Deriving a benchmark dataset at favorable attenuation conditions. Water Research 46, 5551-5565.

Kötke, D., Gandrass, J., Xie, Z., Ebinghaus, R., 2019. Prioritised pharmaceuticals in German estuaries and coastal waters: Occurrence and environmental risk assessment. Environmental Pollution 255, 113161.

Li, W., C., 2014. Occurrence, sources, and fate of pharmaceuticals in aquatic environment and soil. Environ. Pollut. 187, 193–201. https://doi.org/10.1016/j.envpol. 2014.01.015.

Lindim, C., van Gils, J., Cousins, I., T., Kühne, R., Georgieva, D., Kutsarova, S., Mekenyan, O., 2017. Model-predicted occurrence of multiple pharmaceuticals in Swedish surface waters and their flushing to the Baltic Sea. Environmental Pollution 223, 595-604.

Liu, J.-L., Wong, M.-H., 2013. Pharmaceuticals and personal care products (PPCPs): a review on environmental contamination in China. Environ. Int. 59, 208–224. https:// doi.org/10.1016/j.envint.2013.06.012.

Lolić, A., Paíga, P., Santos, L. H.M.L.M., Ramos, S., Correia, M., Delerue-Matos, C., 2015. Assessment of non-steroidal anti-inflammatory and analgesic pharmaceuticals in seawaters of North of Portugal: Occurrence and environmental risk. Science of the Total Environment 508, 240–250.

Lonappan, L., Brar, S.K., Das, R.K., Verma, M. & Surampalli, R.Y., 2016. Diclofenac and its transformation products: Environmental occurrence and toxicity – A review. Environment International 96, 127-138.

Meribout, R., Zuo, Y., Khodja, A.A., Piram, A., Lebarillier, S., Cheng, J., Wang, C., Wong-Wah- Chung, P., 2016. Photocatalytic degradation ofantiepileptic drug carbamazepine with bismuth oxychlorides (BiOCl and BiOCl/AgCl composite) in water: efficiency evaluation and elucidation degradation pathways. J. Photochem. Photobiol. A Chem. 328, 105–113. https://doi.org/10.1016/j.jphotochem.2016.04.024

Nurmi, T.M.A., Kiljunen, T.K., Knuutinen, J.S., 2019. A fugacity model assessment of ibuprofen, diclofenac, carbamazepine, and their transformation product concentrations in an aquatic environment. Environmental Science and Pollution Research 26, 328–341. https://doi.org/10.1007/s11356-018-3485-x

Nödler, K., Voutsa, D., Licha, T., 2014. Polar organic micropollutants in the coastal environment of different marine systems. Mar. Pollut. Bull. 85, 50–59. http://dx.doi. org/10.1016/j.marpolbul.2014.06.024.

Palma, P., Fialho, S., Lima, A., Novais, M., H., Costa, M., J., Montemurro, N., Pérez, S., de Alda, M., L., 2020. Pharmaceuticals in a Mediterranean Basin: The influence of temporal and hydrological patterns in environmental risk assessment. Science of the Total Environment 709, 136205.

PubChem Database. National Center for Biotechnology Information. Diclofenac, CID=3033. Available online: https://pubchem.ncbi.nlm.nih.gov/compound/Diclofenac (accessed on 3 May 2020).

Rizzo, L., Malato, S., Antakyali, D., Beretsou, V.G., Đolić, M.B., Gernjak, W., Heath, E., Ivancev-Tumbas, I., Karaolia, P., Lado Ribeiro, A.R., Mascolo, G., McArdell, C.S., Schaar, H., Silva, A.M.T., Fatta-Kassinos, D., 2019. Consolidated vs new advanced treatment methods for the removal of contaminants of emerging concern from urban wastewater. Sci. Total Environ. 655, 986-1008.

Shi, H., Yang, Y., Liu, M., Yan, C.X., Yue, H.Y., Zhou, J.L., 2014. Occurrence and distribution of antibiotics in the surface sediments of the Yangtze estuary and nearby coastal areas. Mar. Pollut. Bull. 83 (1), 317–323.

Siedlewicz, G., Białk-Bielińska, A., Borecka, M., Winogradow, A., Stepnowski, P., Pazdro, K., 2018. Presence, concentrations and risk assessment of selected antibiotic residues in sediments and near-bottom waters collected from the Polish coastal zone in the southern Baltic Sea — Summary of 3 years of studies. Marine Pollution Bulletin 129, 787–801.

Siedlewicz, G., Borecka, M., Białk-Bielińska, A., Sikora, K., Stepnowski, P., Pazdro, K., 2016. Determination of antibiotic residues in southern Baltic Sea sediments using a tandem solid-phase extraction and liquid chromatography coupled with tandem mass spectrometry. Oceanologia 58 (3), 221–234. http://dx.doi.org/10.1016/j.oceano. 2016.04.005.

Siedlewicz, G., Pazdro, K., Borecka, M., Kornowska, K., Białk-Bielińska, A., Stepnowski, P., 2014. Determination of tetracyclines residues in the Gulf of Gdansk (southern Baltic Sea) sediments using a tandem solid-phase extraction with liquid chromatography coupled with tandem mass spectrometry. In: Zielinski, T. et al. (Eds.), GeoPlanet: Earth and Planetary Sciences, Insights on Environmental Changes, Where the World is Heading. Springer International Publishing, Switzerland, pp. 33–48.

Szymczycha, B., Borecka, M., Białk-Bielińska, A., Siedlewicz, G., Pazdro, K., 2020. Submarine groundwater discharge as a source of pharmaceutical and caffeine residues in coastal ecosystem: Bay of Puck, southern Baltic Sea case study. Science of the Total Environment 713, 136522.

Ternes, T.A., Stumpf, M., Mueller, J., Haberer, K., Wilken, R., Servos, M., 1999. Behavior and occurrence of estrogens in municipal sewage treatment plants - I. Investigations in Germany, Canada and Brazil. Sci. Total Environ. 225, 81–90.

Tixier, C., Singer, H., P., Oellers, S., Müller, S., R., 2003. Occurrence and fate of carbamazepine, clofibric acid, diclofenac, ibuprofen, ketoprofen, and naproxen in surface waters. Environmental Science & Technology 37 (6), 1061-1068.

Turja, R., Lehtonen, K. K., Meierjohann, A., Brozinski, J. M., Vahtera, et al., 2015. The mussel caging approach in assessing biological effects of wastewater treatment plant discharges in the Gulf of Finland (Baltic Sea). Marine pollution bulletin 97(1), 135-149.

Umweltbundesamt (German Environmental Protection Agency), 2016. Pharmaceuticals in the environment: Global occurrence and potential cooperative action under the Strategic Approach to International Chemicals Management (SAICM). TEXTE 67/2016. https://www.umweltbundesamt.de/

UNESCO, HELCOM, 2017. Pharmaceuticals in the Aquatic Environment of the Baltic Sea Region – A Status Report. UNESCO Emerging Pollutants in Water Series – No. 1. UNESCO Publishing, Paris.

Verlicchi, P., Al Aukidy, M., Zambello, E., 2012. Occurrence of pharmaceutical compounds in urban wastewater: removal, mass load and environmental risk after a secondary treatment—a review. Sci. Total Environ. 429, 123–155. https://doi.org/10.1016/j.scitotenv.2012.04.028

Vieno, N., Sillanpää, M., 2014. Fate of diclofenac in municipal wastewater treatment plant — a review. Environ. Int. 69, 28–39.

White, D., Lapworth, D.J., Civil, W., Williams, P., 2019. Tracking changes in the occurrence and source of pharmaceuticals within the River Thames, UK; from source to sea. Environ. Pollut. 249, 257–266.

Wolecki, D., Caban, M., Pazdro, K., Mulkiewicz, E., Stepnowski, P., Kumirska, J., 2019. Simultaneous determination of non-steroidal anti-inflammatory drugs and natural estrogens in the mussels Mytilus edulis trossulus. Talanta 200, 316–323.

Yang, Y., Ok, Y.S., Kim, K.-H., Kwon, E.E., Tsang, Y.F., 2017. Occurrences and removal of pharmaceuticals and personal care products (PPCPs) in drinking water and water/ sewage treatment plants: a review. Sci. Total Environ. 596:303–320. https://doi.org/ 10.1016/j.scitotenv.2017.04.102.

Zheng, Q., Zhang, R.J., Wang, Y. H., Pan, X. H., Tang, J. H., Zhang, G., 2012. Occurrence and distribution of antibiotics in the Beibu Gulf, China: impacts of river discharge and aquaculture activities. Mar Environ Res 78, 26–33.

Zupanc, M., Kosjek, T., Petkovsek, M., Dular, M., Kompare, B., Sirok, B., Blažeka, Ž., Health, E., 2013. Removal of pharmaceuticals from wastewater by biological processes, hydrodynamic cavitation and UV treatment. Ultrason Sonochem 20(4), 1104–1112.

References – Ringed seals

Auttila, M., Niemi, M., Skrzypczak, T., Viljanen, M., Kunnasranta, M., 2014. Estimating and mitigating perinatal mortality in the endangered Saimaa ringed seal (Phoca hispida saimensis) in a changing climate. Ann. Zool. Fennici 51, 526–534.

Ferguson, S. H. et al. 2017. Demographic, ecological, and physiological responses of ringed seals to an abrupt decline in sea ice availability. PeerJ 5: e2957.

Harding, K.C., and Härkonen, T. 1999. Development in the Baltic grey seal (Halichoerus grypus) and ringed seal (Phoca hispida) populations during the 20th century. AMBIO 28: 619–627.

HELCOM, 2018a. Distribution of seals. HELCOM core indicator report. Visited 04.11.2020. Online: https://helcom.fi/wp-content/uploads/2019/08/Distribution-of-Baltic-seals-HELCOM-core-indicator-2018.pdf.

HELCOM, 2018b. Population trends and abundance of seals. HELCOM core indicator report. Visited 04.11.2020. Online: https://helcom.fi/media/core%20indicators/Population-trends-and-abundance-of-seals-HELCOM-core-indicator-2018.pdf

Helle, E., Olsson, M., Jensen, S. 1976. PCB levels correlated with pathological changes in seal uteri. Ambio 5:261–262.

Jounela, P., Sipilä, T., Koskela, J., Tiilikainen, R., Auttila, M., Niemi, N., Kunnasranta, M. 2019. Incidental bycatch mortality and fishing restrictions: impacts on juvenile survival in the Endangered Saimaa ringed seal Pusa hispida saimensis. Endangered Species Research, 38, 91-99.

Kauhala, K., Bergenius, M., Isomursu, M., Raitaniemi, J., 2019. Reproductive rate and nutritional status of Baltic ringed seals Kaarina. Mammal Research 64, 109–120.

Kauhala, K., Bäcklin, B. M, Raitaniemi, J., Harding, K. C., 2017. The effect of prey quality and ice conditions on the nutritional status of Baltic gray seals of different age groups. Mammal Res 62, 351–362

Kovacs, K. M., Aguilar, A., Aurioles, D., Burkanov, V., Campagne, C., Gales, N., Gelatt, T., Goldsworthy, S. D., Goodman, S. J., Hofmeyr, G. J. G., Härkönen, T., Lowry, L., Lydersen, C., Schipper, J., Sipilä, T., Southwell, C., Stuart, S., Thompson, D. & Trillmich, F. 2012: Global threats to pinnipeds. — Marine Mammal Science 28, 414–436.

Laidre, K. L., Stirling, I., Lowry, L. F., Wiig, O., Heide- Jorgensen, M. P. & Ferguson, S. H. 2008. Quantifying the sensitivity of Arctic marine mammals to climate- induced habitat change. — Ecological Applications 18, S97–S125.

Meier, H. E. M., R. Döscher, R., Halkka, A., 2004. Simulated distributions of Baltic Sea-ice in the warming climate and consequences for the winter habitat of the Baltic Ringed Seal. AMBIO 33, 249–256.

Nyman, M., Bergknut, M., Fant, M.L., Raunio, H., Jestoi, M., Bengs, C., Murk, A., Koistinen, J., Backman, C., Pelkonen, O., Tysklind, M., Hirvi, T., Helle, E. 2003. Contaminant exposure and effects in Baltic ringed and grey seals as assessed by biomarkers. Mar. Environ. Res. 55, 73–99.

Sundqvist, L., Harkonen, T., Svensson, C. J. & Harding, K. C. 2012. Linking climate trends to population dynamics in the Baltic ringed seal: impacts of historical and future winter temperatures. Ambio 41, 865–872.

Troisi, G. M., Barton, S. J., Liori, O., Nyman, M., 2020. Polychlorinated Biphenyls (PCBs) and Sex Hormone Concentrations in Ringed and Grey Seals: A Possible Link to Endocrine Disruption? Archives of Environmental Contamination and Toxicology 78, 513–524. https://doi.org/10.1007/s00244-020-00716-z

References – Grey seals

Harding, K.C., and Härkonen, T. 1999. Development in the Baltic grey seal (Halichoerus grypus) and ringed seal (Phoca hispida) populations during the 20th century. AMBIO 28: 619–627.

HELCOM, 2018a. Distribution of seals. HELCOM core indicator report. Visited 04.11.2020. Online: https://helcom.fi/wp-content/uploads/2019/08/Distribution-of-Baltic-seals-HELCOM-core-indicator-2018.pdf.

HELCOM, 2018b. Population trends and abundance of seals. HELCOM core indicator report. Visited 04.11.2020. Online: https://helcom.fi/media/core%20indicators/Population-trends-and-abundance-of-seals-HELCOM-core-indicator-2018.pdf.

HELCOM, 2018c. Reproductive status of seals. HELCOM core indicator report. https://helcom.fi/wp-content/uploads/2019/08/Reproductive-status-of-seals-HELCOM-core-indicator-2018.pdf.

HELCOM, 2018d. Nutritional status of seals. HELCOM core indicator report. https://helcom.fi/wp-content/uploads/2019/08/Nutritional-status-of-seals-HELCOM-core-indicator-2018.pdf.

Jüssi, M., Harkonen, T., Helle, E., Jüssi, I. 2008. Decreasing ice coverage will reduce the breeding sucess of Baltic Grey seal (Halichoerus grypus) females. Ambio, 37, 80-85.

Kauhala, K., Bäcklin, B. M, Raitaniemi, J., Harding, K. C., 2017. The effect of prey quality and ice conditions on the nutritional status of Baltic gray seals of different age groups. Mammal Res 62, 351–362.

Kauhala, K., Bäcklin, B.M., Harding, K., Raitaniemi, J. 2017. The effect of prey quality and ice conditions on the nutritional status of Baltic gray seals of different age groups. Mammal Res 62:351–362.

Schmidt, B., Sonne, C., Nachtsheim, D., Wohlsein, P., Persson, S., Dietz, R., Siebert, U. 2020. Liver histopathology of Baltic grey seals (Halichoerus grypus) over three decades. Environment International, 145, 106110, doi: 10.1016/j.envint.2020.106110.

Silva, W.T.A.F, Harding, K.C., Marques, G.M., Bäcklin, B.M., Sonne, C., Dietz, R., Kauhala, K., Desforges, J.-P. 2020. Life cycle bioenergetics of the gray seal (Halichoerus grypus) in the Baltic Sea:

Population response to environmental stress. Environment International, 145, 106145, doi: 10.1016/j.envint.2020.106145.

Sonne, C., Siebert, U., Gonnsen, K., Desforges, J.P., Eulaers, I., Persson, S., Roos, A., Bäcklin, B.M., Kauhala, K., Olsen, M.T., Hårding, K., Treu, G., Galatius, A., Andersen-Ranberg, E., Gross, S., Lakemeyer, J., Lehnert, K., Lam, S.S., Peng, W., Dietz, R., 2020. Health effects from contaminant exposure in Baltic Sea birds and marine mammals: a review. Environ. Int. 139, 105725.

Vanhatalo, J., Vetemaa, M., Herrero, A., Aho, T., Tiilikainen, R. 2014. By-Catch of Grey Seals (Halichoerus grypus) in Baltic Fisheries—A Bayesian Analysis of Interview Survey. PLoS ONE, 9(11): e113836. doi:10.1371/journal.pone.0113836.

References – Benthic habitats

Bastardie, F., Danto, J., Rufener, M.-C., van Denderen, D., Eigaard, O. R., Dinesen, G. E., Nielsen, J. R., 2020. Reducing fisheries impacts on the seafloor: A bio-economic evaluation of policy strategies for improving sustainability in the Baltic Sea. Fisheries Research 230, 105681. https://doi.org/10.1016/j.fishres.2020.105681

Collie, J. S., S. J. Hall, M. J. Kaiser, and I. R. Poiner. 2000. A quanti- tative analysis of fishing impacts on shelf-sea benthos. Journal of Animal Ecology 69:785–798.

van Denderen, P. D., Bolam, S. G., Friedland, R., Hiddink, J. G., Norén, K., Rijnsdorp, A. D., Sköld, M., Törnroos, A., Virtanen, E. A., Valanko, S., 2020. Evaluating impacts of bottom trawling and hypoxia on benthic communities at the local, habitat, and regional scale using a modelling approach.

Hiddink, J.G., Jennings, S., Sciberras, M., Bolam, S.G., Cambiè, G., McConnaughey, R.A., et al., 2019. Assessing bottom trawling impacts based on the longevity of benthic invertebrates. J. Appl. Ecol. 56 (5), 1075–1084.

Hiddink, J.G., Jennings, S., Sciberras, M., Szostek, C.L., Hughes, K.M., Ellis, N., Rijnsdorp, A.D., McConnaughey, R.A., Mazor, T., Hilborn, R., Collie, J.S., Pitcher, C.R., Amoroso, R.O., Parma, A.M., Suuronen, P., Kaiser, M.J., 2017. Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. Proc. Natl. Acad. Sci. 114, 8301–8306.

Kaiser, M. J., Hormbrey, S., Booth, J. R., Hinz, H., and Hiddink, J. G. 2018. Recovery linked to life history of sessile epifauna following exclusion of towed mobile fishing gear. Journal of Applied Ecology, 55: 1060–1070.

Korpinen, S., Meidinger, M., Laamanen, M., 2013. Cumulative impacts on seabed habitats: An indicator for assessments of good environmental status. Marine Pollution Bulletin 74, 311–319. http://dx.doi.org/10.1016/j.marpolbul.2013.06.036

Rijnsdorp, A. D., Bolam, S. G., Garcia, C., Hiddink, J. G., Hintzen, N. T., van Denderen, D. P., and Van Kooten, T. 2018. Estimating sensitivity of seabed habitats to disturbance by bottom trawling based on the longevity of benthic fauna. Ecological Applications, 28: 1302–1312.

Rijnsdorp, A. D., Bastardie, F., Bolam, S. G., Buhl-Mortensen, L., Eigaard, O. R., Hamon, K. G., Hiddink, J. G. et al. 2016. Towards a framework for the quantitative assessment of trawling impact on the seabed and benthic ecosystem. ICES Journal of Marine Science 73, i127–138.

Rousi, H., Korpinen, S. and Bonsdorff, E., 2019. Brackish-water benthic fauna under fluctuating environmental conditions: the role of eutrophication, hypoxia, and global change. Frontiers in Marine Science 6, 464. doi: 10.3389/fmars.2019.00464

Villnäs, A., Norkko, J., Lukkari, K., Hewitt, J., and Norkko, A. 2012. Consequences of increasing hypoxic disturbance on benthic communities and ecosystem functioning. PLoS One, 7: e44920.

References – Climate change

BACC II Author Team, 2015. Second assessment of climate change for the Baltic Sea Basin. Regional Climate Studies. Cham: Springer.

Bełdowska, M., Jędruch, A., Łęczyński, L., Saniewska, D., Kwasigroch, U., 2016. Coastal erosion as a source of mercury into the marine environment along the polish Baltic shore. Environmental Science and Pollution Research 23(16), 16372-16382.

Christensen, O.B., Kjellström, E., 2018. Projections for Temperature, Precipitation, Wind, and Snow in the Baltic Sea Region until 2100. Oxford Research Encyclopedia for climate science. Oxford University Press. https://doi.org/10.1093/acrefore/9780190228620.013.695

Groll, N., Grabemann, I., Hünicke, B., Meese, M., 2017. Baltic Sea wave conditions under climate change scenarios. Boreal Env. Res. 22, 1–12.

Gröger, M., Arneborg, L., Dieterich, C., Höglund, A., Meier, H.E.M., 2019. Hydrographic changes in the North Sea and Baltic Sea projected in an ensemble of climate scenarios downscaled with a coupled regional ocean-sea ice-atmosphere model. Climate Dynamics, https://doi.org/10.1007/s00382-019-04908-9

Hinrichsen, H.H., von Dewitz, B., Lehmann, A., Bergström, U., Hüssy, K., 2017. Spatio-temporal dynamics of cod nursery areas in the Baltic Sea. Progress in Oceanography, 155, 28-40.

Luomaranta, A., Ruosteenoja, K., Jylhä, K., Gregow, H., Haapala, J., Laaksonen, A., 2014. Multimodel estimates of the changes in the Baltic Sea ice cover during the present century, Tellus A: Dynamic Meteorology and Oceanography, 66:1, 22617, DOI: 10.3402/tellusa.v66.22617.

Meier, H.E.M., Dieterich, C., Eilola, K., Gröger, M., Höglund, A., Radtke, H., Saraiva, S., Wåhlström, I., 2019a. Future projections of record-breaking sea surface temperature and cyanobacteria bloom events in the Baltic Sea. AMBIO, 48, 1362–1376, https://doi.org/10.1007/s13280-019-01235-5

Meier, H.E.M., Eilola, K., Almroth-Rosell, E., Schimanke, S., Kniebusch, M., Höglund, A., Pemberton, P., Liu, Y., Väli, G., Saraiva, S., 2019b. Disentangling the impact of nutrient load and climate changes on Baltic Sea hypoxia and eutrophication since 1850. Climate Dynamics 53, 1145–1166. https://doi.org/10.1007/s00382-018-4296-y

Meier, H.E.M., Saraiva, S., 2020. Projected oceanographical changes in the Baltic Sea until 2100. In: Oxford Research Encyclopedia, Climate Science. Oxford: Oxford University Press, doi:10.1093/acrefore/9780190228620.013.699

Nurkse, K., Kotta, J., Rätsep, M., Kotta, I., Kreitsberg, R, 2018. Experimental evaluation of the effects of the novel predators, round goby and mud crab on benthic invertebrates in the Gulf of Riga, the Baltic Sea. Journal of the Marine Biological Association of the United Kingdom, 98 (1), 25−31. doi: 10.1017/S0025315417001965

Rajczak, J., Schär, C., 2017. Projections of future precipitation extremes over Europe: A multimodel assessment of climate simulations. Journal of Geophysical Research: Atmospheres, 122, 10773–10800. https://doi.org/10.1002/2017JD027176

Roudier, P., Jafet, C., Andersson, J.C.M., Donnelly, C., Feyen, L., Greuell, W., Ludwig, F., 2016. Projections of future floods and hydrological droughts in Europe under a +2°C global warming. Climatic Change 135: 341–355. http://dx.doi.org/10.1007/s10584-015-1570-4.

Ruosteenoja, K., Vihma, T., Venäläinen, A., 2019. Projected changes in European and North Atlantic seasonal wind climate derived from CMIP5 simulations. Journal of Climate, 32(19), 6467-6490.

Saniewska D, Bełdowska M, Bełdowski J, Saniewski M, Gębka K, Szubska M, Wochna, A., 2018. Impact of intense rains and flooding on mercury riverine input to the coastal zone. Marine Pollution Bulletin. 127, 593-602.

Saraiva, S., Meier, H. E. M., Andersson, H. C., Höglund, A., Dieterich, C., Gröger, M., Hordoir, R., Eilola, K., 2019. Uncertainties in projections of the Baltic Sea ecosystem driven by an ensemble of global climate models. Frontiers in Earth Science, 6:244, https://doi.org/10.3389/feart.2018.00244.

Silva, V., Abrantes, N., Costac, R., Keizer, J.J., Goncalves, F., Pereira J.L., 2016. Effects of ash-loaded post-fire runoff on the freshwater clam Corbicula fluminea. Ecological Engineering, 90, 180-189, doi: 10.1016/j.ecoleng.2016.01.043.

Viglino, L., Pelletier, E., St-Louis, R., 2004. Highly persistent butyltins in northern marine sediments: a long-
term threat for the Saguenay Fjord (Canada). Environ. Toxicol. Chem. 23 (11), 2673–2681.