FISH PRO II

Status of coastal fish communities in the Baltic Sea during 2011–2016 – the third thematic assessment

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Executive summary

Coastal fish, the fish assemblages in relatively near-shore and shallow (< 20 m depth) coastal areas, in the Baltic Sea harbour a mixture of species with marine and freshwater origins. Typical species are perch, pikeperch, pike, roach and breams, which are of freshwater origin and confined to coastal areas, as well as marine species like flounder, cod and herring, which often have seasonal migrations between nearshore and outer sea areas. Coastal fish populations and communities provide important ecosystem services, contributing to ecosystem functioning and high socio-economic and cultural values. They represent key elements for assessing environmental status in relation to environmental objectives within the Baltic Sea Action Plan (BSAP) and the Marine Strategy Framework Directive (MSFD).

The main aim of this report is to provide a comprehensive assessment of the status of coastal fish in the Baltic Sea during 2011-2016, in support of the second holistic assessment of HELCOM (HOLAS II). The results are included in summary in the State of the Baltic Sea report.

The report contains the latest status assessment of coastal fish in the Baltic Sea using two Baltic-wide CORE indicators and an agreed assessment approach (Chapter 3). The report also includes a review of major pressures acting on coastal fish communities (Chapter 2), and of potential measures to restore and support coastal fish communities in the Baltic Sea (Chapter 4). The report ends with future recommendations for continued work on coastal fish monitoring and assessment in the Baltic Sea in relation to the BSAP and MSFD (Chapter 5).

The status assessments presented in this report includes data from Finland, Estonia, Latvia, Lithuania, Poland, Denmark and Sweden. Experts from Russia and Germany have also participated in the HELCOM work on coastal fish during recent years, but data from these countries are not included in the current assessment. Status is assessed based on the two HECLOM core indicators 'Abundance of key species' and 'Abundance of key functional groups', of which the latter is composed of a component assessing the abundance of piscivores and one on the abundance of Cyprinids/mesopredators.

The results show that the status of the coastal fish communities varies between areas, regions and indicators, but the general status based on the two core indicators appears weak. Only approximately half of the assessed areas and assessment units are classified as being in good status. The more northern areas, where perch is used as the key species, are more often assessed to be in good status than more southern areas, where flounder is recognised as the key species. For the two functional group indicators, the status of piscivores follows a similar pattern of the key species indicator, with relatively better status in more northern areas. The other important functional group, cyprinids, shows insufficient status also in more north-eastern areas of the Baltic Sea as a result of too high abundances.

The current monitoring network for coastal fish is rather extensive, but does still only support an assessment in half or less than half of the 42 listed assessment units for coastal fish in the Baltic Sea. The confidence of the assessment is moderate to low, depending on area and indicator, mainly as a result of short time-series, poor spatial representation, and data quality issues.

The review of pressures acting on coastal fish communities show that a multitude of natural and human-induced pressures potentially affect coastal fish communities simultaneously. These include for example fishing, habitat exploitation, climate change, eutrophication and natural interactions within the coastal food-web. Whereas a few pressures are strong and often explain a large proportion of the variation in fish abundance and distribution, the effects of others can only be observed locally or under certain conditions and vary across areas and among communities. The potential for generalizations across areas is hence limited, and evaluations of which pressure is of key importance should be undertaken for each individual case separately considering the ambient local conditions in each area together with the range of potential human induced pressures.

As a result, measures to restore and protect coastal fish communities should also be developed with a local perspective, and different measures might be relevant in different geographic areas. In general, however, scientific evidence on the effectiveness of different measures is poor, as only few thorough evaluations of implemented measures have been undertaken in the Baltic Sea. The few measures that have been scientifically evaluated with proven effects includes actions aimed at reducing the mortality of fish (e.g. no take areas). There is also partial support for temporary fishing closures and gear and catch restrictions. Among measures that aim at improving the production of fish, habitat protection and restoration have proven to be effective. For many other potential measures there is a general lack of scientific support, including biomanipulation, nutrient and substance abatement, as well as stocking of hatchery-reared fish.

The assessment results and reviews presented in this report show that coastal fish assessments and monitoring in the Baltic Sea have taken noteworthy steps forward since the last thematic assessment. For example, Baltic-wide CORE indicators on coastal fish have been agreed on, as well as a regionally agreed concept for assessing the status of coastal fish communities. Knowledge has improved on the key pressures impacting on coastal fish communities, and on measures that are potentially most effective for supporting and restoring coastal fish communities. In spite of this, there are still several knowledge gaps and development needs, which should be considered in the future. There is a need to safeguard existing monitoring programs and to initiate new monitoring programs for coastal fish in geographic areas that are currently poorly covered or not possible to assess at all. There is also a need to continue harmonizing assessment approaches to enable comparisons across monitoring programs and data sources, and to further develop common indicators and assessment methods. This concerns the further evaluation and development of current indicators, as well as the development of new generic indicators, focusing on aspects of size-structure in the assessed fish communities. Finally, it is clear that initiatives to strengthen the evaluation measures must be undertaken. It is recommended that the results presented in this report should be used as the basis for following up on the objectives of the BSAP and MSFD, as well as for the development of national management plans and coastal fish assessments in the Baltic Sea.

1. Background

1.1. Coastal fish in the Baltic Sea

The Baltic Sea exhibits strong environmental variability, mainly characterised by a pronounced salinity gradient with decreasing salinity towards the inner areas. As a result, coastal fish communities, here referred to as the fish assemblages in relatively near-shore and shallow (< 20 m depth) coastal areas in the region, often harbour a mixture of species of marine and freshwater origin (Ojaveer *et al.*, 2010; Olsson *et al.* 2012). Typical freshwater species are perch (*Perca fluviatilis*), ruffe (*Gymnocephalus cernuus*), pikeperch (*Sander lucioperca*), whitefish (*Coregonus maraena*) and fishes from the carp family (Cyprinidae), whereas the common marine species found in coastal areas are herring (*Clupea harengus*), flounder (*Platichtys flesus*) and cod (*Gadus morhua*; HELCOM 2012; Olsson *et al.* 2012; Bergström *et al.* 2016a). In the eastern and northern parts of the Baltic Sea with lower salinity, coastal fish communities are dominated by species of freshwater origin (HELCOM 2012), whereas in the more saline southern and western parts an increased segment of marine species are commonly found. In addition, in more exposed coastal areas marine species and those preferring lower water temperatures (whitefish, smelt (*Osmerus eperlanus*) and sculpins) are more common.

There is also seasonal variation in the species composition. During the warmer parts of the year and in more sheltered parts of the archipelago, fish communities in coastal areas are dominated by freshwater species and those preferring by higher water temperatures (Olsson *et al.* 2012). As a contrast, the fish communities during the colder parts of the year are dominated by species of a marine origin and those preferring lower water temperatures.

Due to the influence from the environmental gradients and seasonality on species composition, the predominating species in the coastal fish community varies. Generally, perch is recognised as a key species in the northern and eastern parts of the Baltic Sea, during the warmer period of the year and in more sheltered parts of the coastline (HELCOM 2017a). Cod and flounder are commonly recognised as key species in the more saline western and southern parts, and in more exposed archipelago areas (HELCOM 2017a).

Another key feature of coastal fish communities is the relatively restricted migration and hence local population structure in many species as compared to offshore and more marine species (Laikre *et al.* 2005; Östman *et al.* 2017a). Freshwater species, such as perch, whitefish and pikeperch, exhibit rather strong genetic population subdivision on a small scale and restricted migration across coastal areas (Laikre *et al.* 2005; Olsson *et al.* 2011, 2012; Östman *et al.* 2017a). Populations of species like cod, flounder and herring populations, on the other hand, are typically characterized by substantial gene flow and migration across areas, and relatively weak population sub-structuring (Nielsen *et al.* 2003; Jorgenssen *et al.* 2007; Florin and Höglund 2008). A recent study on flounder, however, challenged this view by describing speciation of flounder in the Baltic Sea (Momigliano *et al.* 2017).

As a result, coastal fish communities are also local in how they may respond to environmental conditions and pressures (Bergström *et al.* 2016b; Östman *et al.* 2017b). Taken together, the local population structure and variability in species composition implies that assessments of coastal fish communities need to be considered on a local geographic scale, preferably in relation to of the migration distance of the most common species in the communities (Bergström *et al.* 2016b; Östman *et al.* 2017a, b). This also implies that management measures to restore and/or strengthen coastal fish communities should have a local approach.

1.2. Ecological role and societal relevance of coastal fish

Coastal fish are important both for the Baltic Sea ecosystems and for humans with respect to socio-economic and cultural values. Fish are found in the central part of the food-web and hence have a key role in linking different processes. As such, the status of coastal fish conveys information on the general status of coastal ecosystems in the Baltic Sea (HELCOM 2006, 2012). Evidence is accumulating on the central role of coastal fish in maintaining ecosystem structure and functioning (Östman *et al.* 2016). Piscivorous fish are, for example, observed to regulate the abundance of lower trophic level species, and may also influence the occurrence of eutrophication symptoms, such as blooms of ephemeral algae (Eriksson *et al.* 2011). Healthy populations of coastal piscivorous fish might, via trophic cascading mechanisms, repress eutrophication symptoms via trophic cascades at a magnitude comparable with nutrient abatement measures in coastal areas (Östman *et al.* 2016).

Last but not least, coastal fish communities are important for human consumption and for recreational values. Many coastal fish species are targeted by small-scaled coastal commercial fishery, but they are to a large extent also important for the recreational fisheries sector (HELCOM 2015a). The outtake of coastal fish species in recreational fisheries often greatly outnumbers that of the commercial fishery in many Baltic countries (HELCOM 2015a).

The ecosystem services provided by costal fish communities can be described and assessed according to a variety of approaches and frameworks (Hattam et al, 2015), which as a rule should be tailored to be context specific. How coastal fish may contribute to ecosystem services in a Baltic Sea context is exemplified in Tables 1 and 2 (modified from Holmlund & Hammer, 1999). The many connection points between costal fish and ecosystem services illustrate their important role for achieving environmental objectives, as described by the Baltic Sea Action Plan (HELCOM 2007), the Marine Strategy Framework Directive (MSFD, European Commission 2008), and the Common Fisheries Policy (European Commission 2002, 2013), for example.

Table 1.
Examples of ecosystem services that coastal fish provide for ecosystem functioning (modified from Holmlund & Hammer, 1999).

Contribution to ecosystem functioning	
Regulating services	**Linking services**
— Regulation of food web dynamics — Recycling of nutrients — Regulation of ecosystem resilience — Redistribution of bottom substrates — Regulation of carbon fluxes from water to atmosphere — Maintenance of sediment processes — Maintenance of genetic, species, and ecosystem biodiversity	— Linkage within aquatic ecosystems — Linkage between aquatic and terrestrial ecosystems — Transport of nutrients, carbon and minerals — Transport of energy — Acting as ecological memory

Table 2.
Examples of ecosystem services by which coastal fish contribute to human well-being (modified from Holmlund & Hammer, 1999).

Contribution to human use	
Cultural services	**Information services**
— Production of food — Production of medicine — Control of hazardous diseases — Control of algae and macrophytes — Reduction of waste — Supply of aesthetic values — Supply of recreational activities	— Assessment of ecosystem stress — Assessment of ecosystem resilience — Revealing evolutionary tracks — Provision of historical information — Provision of scientific and educational information

1.3. HELCOM FISH PRO and earlier coastal fish assessments

Coastal fish monitoring has a long tradition in the Baltic Sea, with monitoring in some areas dating back to the early 1970's (Olsson *et al.* 2012). HELCOM has carried out coordinated coastal fish monitoring and assessments in the Baltic Sea since 2003, under different constellations (HELCOM FISH, HELCOM FISH PRO, HELCOM FISH PRO II). The currently ongoing project HELCOM FISH PRO II is active during 2012-2018 and includes experts from all nine HELCOM Contracting Parties.

Two thematic assessments describing the status of coastal fish communities in the Baltic Sea based on indicators have previously been produced (HELCOM 2006, 2012). Further, the Baltic Sea Action Plan (HELCOM 2007) and the implementation of the EU Marine Strategy Framework Directive (European Commission 2008) has led to an increased focus on regionally harmonized indicator development, assessment methods and coordinated monitoring programs also for coastal fish (HELCOM 2013).

In comparison to the previous thematic assessments (HELCOM 2006, 2012), this report includes data from a higher number of countries and includes monitoring results also from Poland and Denmark. Further, the status assessment is based on regionally agreed indicators, assessment approaches, and threshold values (HELCOM 2017a,b). With these improvements, the assessments give an update on the status of coastal fish communities in the Baltic Sea, as called for by the on-going change in the Baltic Sea environment and coastal fish communities (Olsson *et al.* 2012; Bergström *et al.* 2016a).

1.4. Monitoring of coastal fish in the Baltic Sea

To date, coastal fish monitoring is undertaken in some form by all Contracting Parties except Russia (HELCOM 2015b; Figure 1), and data was available for assessments from all countries except Germany when producing this report. Monitoring is coordinated by HELCOM, but carried out with somewhat different methods in different countries due to different traditions in monitoring practices and different local settings. Fisheries-independent monitoring dominates and is mainly carried out using passive gears, such as gill nets, fyke nets or trap nets. Active gear types are used in some areas, for example bottom trawls (see the HELCOM monitoring guideline, HELCOM 2015b, for details). In Finland and Denmark, fisheries-independent monitoring programs are not developed. The status assessment for Finnish areas is

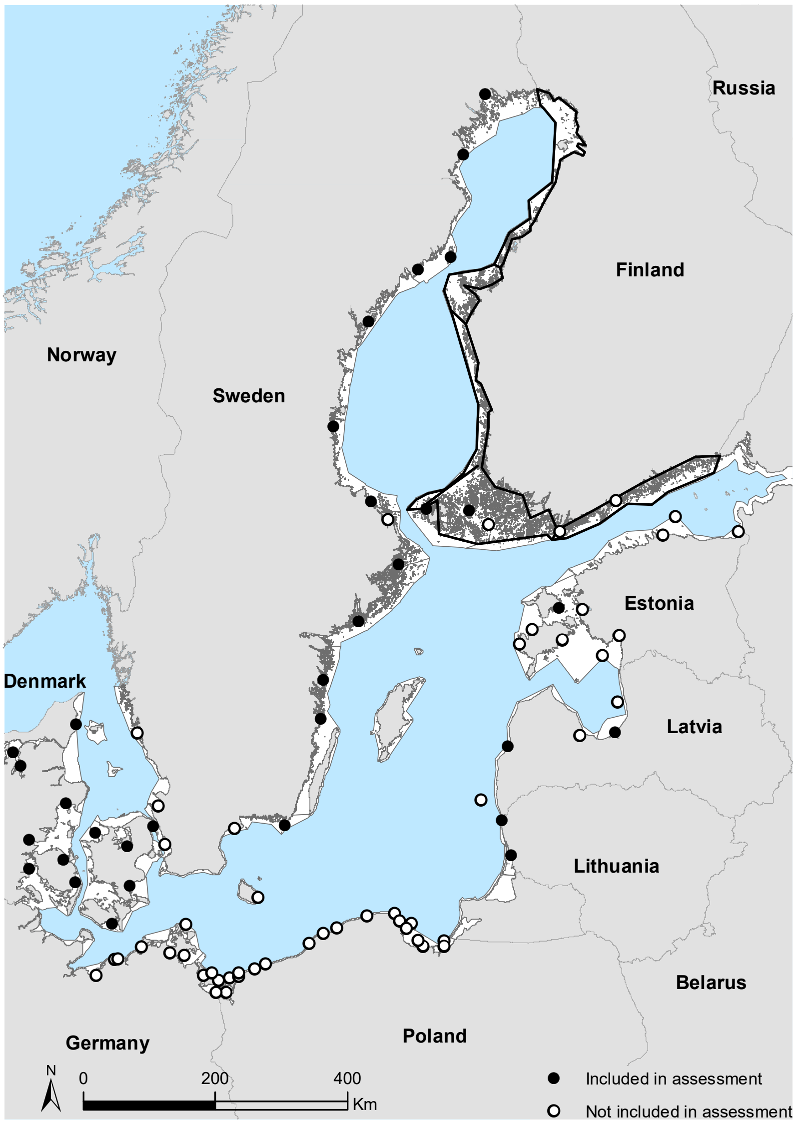


Figure 1.
Areas for coastal fish monitoring in the Baltic Sea, as by 2017. Dots denotes areas included (black) and not included (white) in the assessment as presented in this report. The white areas represent the assessment units applied in the status assessment (see table 3 for names of the assessment units). Note that in Finnish coastal areas, status is assessed based on catch statistics from the small-scaled coastal commercial fishery, which is obtained at sub-basin scale in the Bothnian Sea, The Quark, Bothnian Sea, Åland Sea and Gulf of Finland (marked in black frames).

based on data obtained from small-scale coastal commercial fishery (HELCOM 2015c) and in Denmark based on recreational catch registration (Denmark, Pedersen *et al.* 2005; Sparrevohn *et al.* 2009; Støttrup *et al.* 2012; Kristensen *et al.* 2014). The indicators used in this assessment are, however, generic and based on time-series, and were therefore applied to all monitoring methods in a similar way (HELCOM 2015b).

The focal species of the monitoring are in most cases local species with a freshwater origin, but marine and migratory species are included depending on season and geographic location, as described above. Monitoring of coastal fish is designed to primarily detect changes in the fish communities in relation to large-scale changes in the environment. For this reason, several monitoring areas are located in so-called reference areas, where the level of direct human impact is comparably low.

1.5. Objectives of the report

In this report, we aim to:

- review the major pressures acting for coastal fish and their impacts on coastal fish communities (Chapter 2),
- assess the current status across monitoring areas of the two Baltic wide CORE indicators for coastal fish (Chapter 3),
- review and propose potential measures for coastal fish communities in the Baltic Sea (Chapter 4), and finally
- provide future recommendations for further indicator development and assessment of coastal fish in the Baltic Sea (Chapter 5).

2. Factors influencing coastal fish communities

Coastal areas are amongst the most productive environments worldwide, but also heavily impacted by human activities (Lindeboom 2002; Airoldi and Beck 2007). Several human induced pressures may impact coastal ecosystems, including fishing, habitat exploitation, climate change, eutrophication and exposure to hazardous substances (Collie *et al.* 2008; Brown *et al.* 2018). Due to their central position in the food web, fish are also influenced by species interactions and internal population processes (Persson *et al.* 2000; Shelton & Mangel 2011). Hence, coastal fish communities may be subject to a plethora of potentially impacting pressures, which are likely to differ among areas and seasons, due to the local population structure of most coastal fish species and due to different combinations of pressures in space and time (Olsson *et al.* 2012; Östman *et al.* 2017b). In this chapter we list potential pressures on coastal fish communities and examples of their impacts. The aim of this chapter is to review the potential drivers for change in coastal fish communities in the Baltic Sea, and to increase our understanding on which drivers in the environment measures for coastal fish should be focused on.

2.1. Temperature

Temperature regulates the productivity of coastal ecosystems, hence indirectly influencing food and energy availability for coastal fish. Temperature also directly affects ectothermic organisms, such as fish, where increasing temperatures have a positive effect on both activity and growth. However, in shallow coastal systems, temperatures can often exceed the physiological limits of cold-adapted species, periodically excluding them from particular habitats.

Several freshwater species (e.g. percids and cyprinids) in coastal fish communities prefer warmer waters (Böhling *et al.* 1991; Karås and Thoresson 1992; Karås 1996). In contrast, species of a marine origin (e.g. herring and cod), and even some of the freshwater species living in coastal areas of the Baltic Sea (e.g. salmonids and sculpins) prefer cooler waters (Karås and Thoresson 1992). Summer temperatures largely determine the year-class strength of perch and pikeperch in turn causing strong fluctuations in the abundance and catches of the species (Böhling *et al.* 1991; Lappalainen *et al.* 1996; Kjellman *et al.* 2001; Heikinheimo *et al.* 2014). There is also a match between long-term increases of freshwater species as percids and cyprinids and increasing water temperatures in coastal areas of the Baltic Sea and a subsequent decrease of cold-water adapted species (Olsson *et al.* 2012; Östman *et al.* 2017b). Moreover, juvenile flatfish species utilising coastal growth areas can be impacted negatively during periods of higher water temperatures, especially as temperature maximums in late summer coincide with these groups' coastal habitat use (Vinagre *et al.* 2013; Lavergne *et al.* 2015). The abundance of adult flounder is, however, somewhat favoured by increasing water temperatures (Florin *et al.* 2013; HELCOM 2017a).

Short-term changes in water temperature, caused by weather conditions and currents, also impacts the activity of coastal fish, hence directly affecting their catchability in passive gears such as gillnets and fyke nets. Given that the water temperature is not manageable in the shorter time perspective, it is important to consider the effect of temperature during sampling when assessing the status of coastal fish communities (Bergström *et al.* 2016b; Östman *et al.* 2017b).

2.2. Salinity

The salinity in a coastal area can impact the survival of eggs, larvae and juveniles as well prevent adults from utilizing potential habitats. This has been shown for both commercially important species, such as herring (Illing *et al.* 2016), and ecologically important species, such as sticklebacks (De Faveri and Merilä 2014) and sand goby (Lehtonen and Kvarnemo 2015) alike.

Being a brackish water system, the salinity in the Baltic Sea has a substantial impact on the distribution patterns of organisms (Johanneson and Andre 2006; Wennerströn *et al.* 2013). The variability of salinity observed in parts of the Baltic Sea creates overlap in the distribution of different fish species, and in many coastal areas a co-occurrence of marine species, like cod, and freshwater species, as perch or roach, can be observed. Generally, however, the segment of marine species decreases with increasing latitude in the Baltic Sea (HELCOM 2012). The abundances of species of freshwater origin drastically decrease in the more southern and western parts as the salinity exceeds 10 psu.

Salinity might also act as a barrier to gene flow. The presence of divergent populations of the same species are exemplified by the differential salinity tolerances of allopatric cod stocks (Kijewska *et al.* 2016), and different reproductive strategies of sympatric flounder (Nissling and Dahlman 2010, see also Momigliano *et al.*, 2017).

Salinity has affected the long-term development and structure of coastal fish communities in parallel with temperature. During the last decades, the salinity of the Baltic Sea has decreased and influenced a shift in coastal fish communities in favour of freshwater species and a subsequent decrease in species of a marine origin (Olsson *et al.* 2012; Olsson *et al.* 2015). With a continuing freshening of the Baltic Sea, the segment of freshwater species such as percids and cyprinids are expected to increase, whereas the abundances of marine species like herring, cod and flounder are expected to decrease. Similarly as for temperature, salinity is not a factor that is manageable in the shorter time-perspective and should hence be considered when assessing the status of coastal fish communities (Bergström *et al.* 2016b; Östman *et al.* 2017b).

2.3. Eutrophication

The trophic conditions regulate the productivity of the coastal ecosystem when temperature is not limiting, and hence ultimately affects the energy intake, growth and reproduction of fish. Eutrophication, for example, influences the balance between lower trophic groups of organisms, which in turn ultimately affect the food type and quality for fish. Excessive eutrophication might also lead to oxygen-deficiency, reduced habitat quality and lowered visibility in the water, ultimately affecting particular species' behaviour and physiology in different ways.

The Baltic Sea is subject to long lasting input of nutrients, which makes it one of the world's most eutrophied seas (HELCOM 2010; Fleming-Lehtinen *et al.* 2016), and eutrophication has a substantial impact on the distribution and occurrence of its organisms. The structure and function of coastal fish communities are also impacted by eutrophication (Lappalainen 2002; Bergström *et al.* 2016b; Östman *et al.* 2017b). A common observation is an increased abundance of cyprinid species with increasing nutrient levels (Bonsdorff *et al.* 1997; Lappalainen 2002; Ådjers *et al.* 2006; Härmä *et al.* 2008; Snickars *et al.* 2015; Bergström *et al.* 2016b). Other species are to some extent favoured by moderate levels of eutrophication. For example, the abundance of adult flounder is higher under moderate eutrophication in areas with low fishing pressure (Olsson *et al.* 2012; Florin *et al.* 2013). However, in more shallow coastal areas, increased presence of ephemeral macroalgae, as typically seen with eutrophication, reduces the suitability of nursery habitats for flounder and a variety of other species (Wennhage & Pihl1994; Carl *et al.* 2008; Jokinen *et al.* 2015; Kraufvelin *et al.* 2018), as might also be the effect of lowered water transparency (Bergström *et al.* 2013).

Eutrophication may, however, interact with coastal fish communities in that the structure of the fish communities via trophic cascades might affect the degree of eutrophication symptoms in coastal areas (Eriksson *et al.* 2011; Sieben *et al.* 2011; Baden *et al.* 2012; Östman *et al.* 2016). Piscivorous fish species generally have a structuring role in the ecosystem, and in areas with weak populations of piscivorous fish species, the subsequent lack of top-down control might result in mass-occurences of mesopredatory fish species (mid trophic level fish), also resulting in blooms of ephemeral algae (Eriksson *et al.* 2011; Sieben *et al.* 2011; Baden *et al.* 2012). The effects on the production of ephemeral algae might be as strong as the effects of nutrient additions, and the most pronounced effects are seen in already heavily eutrophied systems (Östman *et al.* 2016). Fisheries management should hence be considered in eutrophication management since viable populations of piscivorous fish species are generally supported in systems with few eutrophication symptoms and balanced food-webs (Eriksson *et al.* 2011).

2.4. Fishing

Fishing can cause different types of pressures on fish communities. The first distinction comes between the direct effects of extracting species, and the indirect effects resulting from trophic cascades and physical habitat disturbance (Airoldi & Beck 2007). With respect to the direct effects, some fish may be targeted for exploitation, whilst others are affected as incidental catch. Generally, however, the catches of unwanted species in the small-scaled coastal commercial fishery is minor compared to the large-scaled commercial trawl fishery. The incidental catch of vulnerable species as for example sea trout (*Salmo trutta*) in coastal fisheries might, however, pose a threat to the species in some areas.

Both small-scale commercial fisheries and recreational fisheries are targeting coastal fish populations, but patterns differ between regions. For obligate coastal species as perch, pike-perch, pike and whitefish, which are mainly targeted in the eastern and northern parts of the Baltic Sea, the outtake from the recreational fisheries sector greatly outnumber that of the small-scale commercial fishery in many countries (Karlsson *et al.* 2014; HELCOM 2015a). In the south-western Baltic Sea, proportionally large recreational catches in coastal areas are also seen for species like cod, flounder and eel (Ferter *et al.* 2013; Eero *et al.* 2015; Sparrevohn & Storr-Paulsen 2012).

Whereas the commercial fishery is regulated and obliged to report catches and efforts to the authorities, the recreational fishery sector lacks general legislation in many countries and today reporting of the catches and outtakes in some form occurs only in a few Baltic countries (Karlsson *et al.* 2014; HELCOM 2015a). Due to the presumably high outtake from the recreational fisheries in combination with poor reporting and regulation of the sector, the effect of recreational fishing on coastal fish communities is likely underestimated.

Fishing has potentially strong effects on recipient populations and communities. This is mainly manifested as direct mortality reducing the abundance and mean size of targeted species (Edgren 2005; Florin *et al.* 2013; Bergström *et al.* 2016c). In addition , the indirect effects of fishing are diverse and vary from changes in individual species life-history traits caused by fisheries induced selection (Cardinale *et al.* 2009; Kokkonen *et al.* 2015), changes in trophic regulation leading to trophic cascades within and across systems (Österblom *et al.* 2007; Eriksson *et al.* 2011; Baden *et al.* 2012; Casini *et al.* 2012), and physical destruction of habitats by fishing gear (Hiddink *et al.* 2006).

2.5. Habitat availability and quality

Habitat availability can become limiting for coastal fish populations when activities such as coastal development, resource extraction, dredging or filling of sand to combat erosion (so called beach nourishment) take place on large scales (Kraufvelin *et al.* 2018). Coastal development includes for example building of marinas, ports or coastal residences. The activities can physically cause displacement of fish by drastically altering the bathymetry, hydrography and seafloor type (Dafforn *et al.* 2015; Kraufvelin *et al.* 2018)expanding resource sectors, increasing population, regulation to river flow, and on-going land change and degradation. While protection of natural coastal habitat is recommended, balancing conservation with human services is now the challenge for managers. Marine infrastructure such as seawalls, marinas and offshore platforms is increasingly used to support and provide services, but has primarily been designed for engineering purposes without consideration of the ecological consequences. Increasingly developments are seeking alternatives to hard engineering and a range of ecological solutions has begun to replace or be incorporated into marine and coastal infrastructure. But too often, hard engineering remains the primary strategy because the tools for managers to implement ecological solutions are either lacking or not supported by policy and stakeholders. Here we outline critical research needs for marine urban development and emerging strategies that seek to mitigate the impacts of marine infrastructure. We present case studies to highlight the strategic direction necessary to support management decisions internationally.

To date, few studies have demonstrated the role of habitat availability and quality for coastal fish in the Baltic Sea (reviewed in Kraufvelin *et al.* 2018), but evidence is accumulating. Although the effect may be very local in each individual case, the cumulative effects of coastal development have been shown to reduce the total available habitat for important life-history stages (Sundblad and Bergström, 2014), which in turn might limit the overall productivity of the stock (Sundblad *et al.* 2014).

Other forms of resource extraction in the coastal zone can also reduce available habitats by removing structure and complexity. This has been documented in the loss of coastal boulder reefs (Støttrup *et al.* 2014)Denmark, has now been re-established with the aim of restoring the reef\u2019s historical structure and function. The effects of the restoration on the local fish community are reported here. Fishing surveys using gillnets and fyke nets were conducted before the restoration (2007. There is some evidence that coordinated extraction and core-building of sand banks in previously eroded areas can lead to increased diversity and fish biomass (De Jong *et al.* 2014) approximately 220 million m3 sand was extracted between 2009 and 2013. In order to decrease the surface area of direct impact, the authorities permitted deep sand extraction, down to 20m below the seabed. Biological and physical impacts of large-scale and deep sand extraction are still being investigated and largely unknown. For this reason, we investigated the colonization of demersal fish in a deep sand extraction site. Two sandbars were artificially created by selective dredging, copying naturally occurring meso-scale bedforms to increase habitat heterogeneity and increasing post-dredging benthic and demersal fish species richness and biomass. Significant differences in demersal fish species assemblages in the sand extraction site were associated with variables such as water depth, median grain size, fraction of very fine sand, biomass of white furrow shell (Abra alba. On the other hand, practices such as beach nourishment and indiscriminate extraction of material lead to a loss of habitat especially for fish with obligate coastal life-history stages (Foden *et al.* 2009).

2.6. Offshore processes

Besides the direct effects from pressures acting in the coastal zone, it must also be considered that coastal habitats are contiguous with characteristically different offshore areas. Both physical processes underway in these offshore habitats and biological linkages via migratory species influence coastal fish populations.

Large-scale physical phenomena can have profound impacts on the water quality of coastal habitats. In the Baltic Sea, these can be best exemplified by the regional climate-driven inflow and mixing of saline North Sea water with the Baltic brackish water (Bendtsen *et al.* 2009). This mixing of water masses of different densities affects the dispersal patterns of passively drifting eggs and larvae from deepwater spawning species to coastal juvenile growth areas, and may also influence on the survival of larvae (Hinrichsen *et al.* 2012, Petereit *et al.* 2014). Under the current nutrient regime, exchange of water masses between coastal and offshore areas in the Baltic Sea also has a large influence on nutrient concentrations in the coastal areas, and hence their eutrophication status (Bryhn *et al.* 2017).

Connectivity due to species migrations between offshore and coastal habitats can also introduce substantial influence on coastal fish assemblages. In the northern and eastern parts of the Baltic, increased seasonal migrations of the mesopredatory three-spined stickleback (Gasterosteus aculeatus) to the coast might have negative effects on coastal piscivorous fish as perch and pike (Bergström *et al.* 2015; Byström *et al.* 2015). The effects come from interactions between the species, which includes predation on coastal fish eggs or high predation pressure of mesopredators on grazers leading to increased algal growth and reduced water quality (Eriksson *et al.* 2011). Similarly, feeding migrations of piscivorous fish into coastal areas can cause changes to food-web interactions and ultimately whole coastal fish assemblages (Casini *et al.* 2012; Lindegren *et al.* 2014).

2.7. Other important factors

Many other natural and human induced pressures also potentially influence coastal fish. A non-exhaustive list of natural factors acting more on the local scale include wind/wave exposure, the bathymetry and morphology of the coastal area, sunlight intensity (peak and cumulative), as well as predation pressure from apex predators (eg. seabirds, mammals or higher trophic level fish) and other interactions within the food web. The local abiotic settings of a coastal area sets the limits for the current production rate of the fish (HELCOM 2012; Bergström *et al.* 2016b). The role of food web processes as internal dynamics and predation are, however, likely different between areas and communities (Vetemaa *et al.* 2010; Lehikoinen *et al.* 2011; Heikinheimo *et al.* 2016; Östman *et al.* 2012, 2016).

On larger scales, the frequency of and pattern shifts in periodic pressures like saltwater inflow, run-off from land and ice coverage followed by the overarching patterns of multi-annual and multi-decadal weather patterns such as the North Atlantic Oscillation also affect the distribution of coastal fish (Olsson *et al.* 2012). Similar to the effects of salinity and water temperature, the variables listed above to some extent represent ambient conditions of the system that are not easily managed in a shorter time-perspective.

In terms of human-induced pressures, which could potentially be managed, a non-exhaustive list includes additionally invasive species, hazardous substances, input of organic matter, aquaculture and maritime transport (commercial and recreational). Only little evidence is however available today on the direct negative effects of invasive species in a Baltic context (Ustups *et al.* 2015), hazardous substances (Hanson *et al.* 2009), and maritime transport (Sandström *et al.* 2005). In addition, efforts to restore degraded habitats can in some cases affect coastal communities in different ways, so that the efforts are beneficial to some species/populations whilst having negative impacts on others (Wortley *et al.* 2013), meaning that trade-offs must be considered.

2.8. Conclusions

A multitude of natural and human-induced pressures potentially affect coastal fish communities simultaneously. A few, strong pressures often explain a large proportion of the variation in fish abundance and distributions, whilst the effects of others can only be observed locally or under certain conditions. However, the extent to which different pressures affect coastal fish varies substantially across coastal areas and among communities. The potential for generalizations across areas is limited and for each case an individual evaluation should be advocated. To that end, in order to address the extent of impacts from human activities on coastal fish populations, one must take a full set of potential human induced pressures into initial account and assess them within the context of the natural pressures and ambient environmental conditions of the specific area.

3. Status assessment

Box 1.
Indicators used

Coastal fish may inform on the general status of the coastal ecosystem, as they are influenced by processes in different parts of the food web as well as general environmental conditions (HELCOM 2006; HELCOM 2012). Further, the status of key coastal fish species and functional groups can have substantial influence on the structure and function of coastal communities and ecosystems (Eriksson *et al.* 2009, 2011; Östman *et al.* 2016). These aspects are reflected in the indicators Abundance of coastal fish key species and Abundance of coastal fish key functional groups.

The indicator *Abundance of coastal fish key species* is represented by the biomass/abundance of key species, which are perch (*Perca fluviatilis*), flounder (Platichtys flesus) or cod (*Gadus morhua*), depending on the sub-basin. Perch is generally the key species in the less saline eastern and northern Baltic Sea (coasts of Sweden, Finland, Estonia and Latvia), and in more sheltered coastal areas of Lithuania, Poland and Germany. In more exposed coastal parts of the central Baltic Sea and in the western Baltic Sea, perch is less abundant and flounder is used as key species. Cod is the representative key species in the western Baltic Sea where salinity is the highest.

The indicator *Abundance of coastal fish key functional groups* is represented by two functional groups: piscivorous fish species and cyprinids/mesopredatory fish. Piscivorous fish are typically represented by perch, pike (*Esox lucius*), pikeperch (*Sander lucioperca*) and burbot (*Lota lota*) in the eastern and northern Baltic Sea (coasts of Sweden, Finland, Estonia, Latvia and Lithuania) and in sheltered coastal areas of Poland and Germany. In the more exposed coastal parts of the central Baltic Sea and in its western parts, piscivores are typically represented by cod and turbot (Scophthalmus maximus). Common species within the cyprinid family (Cyprinidae) are for example roach (*Rutilus rutilus*) and breams (mainly Abramis brama and Blicca bjoerkna). Cyprinids are most abundant in the eastern and northern Baltic Sea. In areas where cyprinids do not occur naturally, the group "mesopredatory fish" is assessed. This group includes coastal fish species representing lower trophic levels, such as wrasses (Labridae), sticklebacks (Gasterosteidae), flatfishes, clupeids and gobies (Gobiidae), and is assessed in more exposed parts of the central and western Baltic Sea coasts.

As highlighted in chapter 2, several pressures may influence on the status of coastal fish communities, and likely have variable and cumulative effects. In addition, coastal fish are dependent on ambient natural conditions, such as temperature, salinity, and coastal morphology, as well as on other species in the coastal community. In terms of manageable pressures, the indicator Abundance of coastal fish key species and the piscivores group of the indicator Abundance of coastal fish key functional groups are considered to be mainly influenced by fishing, habitat availability and quality, as well as to some extent eutrophication. The group cyprinids/mesopredators of the indicator Abundance of coastal fish key functional groups represent mid-trophic level fish. These are to a larger extent considered to be influenced by eutrophication, habitat availability and quality, and to a lesser extent by fishing.

3.1. Methods for status assessment

The assessments presented in this report are based on the status of the two currently operational CORE indicators for coastal fish (Box 1). The first indicator is *Abundance of coastal fish key species*, which describes the status of the key fish species (perch, flounder or cod) in the coastal areas. The second indicator is *Abundance of coastal fish key functional groups*, which describes the state of important functional groups in the coastal fish communities; piscivores, and cyprinids or mesopredators. The indicators estimate the relative abundance and/or biomass of key coastal fish species or species groups, as defined by each indicator, related to a site-specific threshold value or trend. The estimates are obtained from fishery independent monitoring, recreational catch registration and/or commercial catch statistics, as described further below. For more information on these indicators see below and HELCOM (2017a,b).

There are some general features of the assessment of coastal fish that should be noted:

- **First,** the indicators are evaluated in relation to conditions corresponding to sustainable use within prevailing environmental (climate and hydrography) conditions (European Commission 2008).
- **Second,** the approach for indicator-based assessments depends on the length of the time-series:
    - A threshold value (baseline approach) is used when the time-series covers more than 15 years (Figures 2a and 3a).
    - A trend-based approach is used when the time series covers less than 15 years (Figures 2b and 3b).

Importantly, threshold values for the status assessments are identified based on site-specific time-series data for each indicator. Site specific values are used, as coastal fish generally have local population structures, limited migration, and show local responses to environmental change (see references in previous sections of the report). Furthermore, as the data supporting the indicators are derived from different types of monitoring programs, catch registration and data collection, the threshold values are not directly comparable across monitoring areas and data sources.

The following principles for assessing indicators status are used:

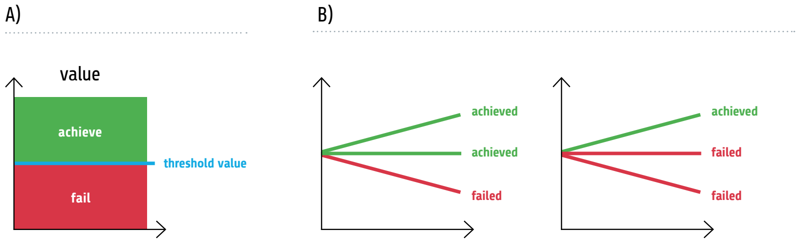


Figure 2.
Assessment approach for the indicators *Abundance of coastal fish key species* and *Abundance of coastal fish key functional groups* (piscivores). In the baseline approach (A), a threshold value is used to define if good status is achieved or not (fail). This approach can be applied when time-series data spans over at least 15 years. In the trend-based approach (B), the status is defined based on the direction of the trend compared to the desired direction of the indicator over time (left when the beginning of the time-series represents good status, right when the beginning of the time-series represents poor status). This trend-based approach is used when the baseline approach cannot be applied.

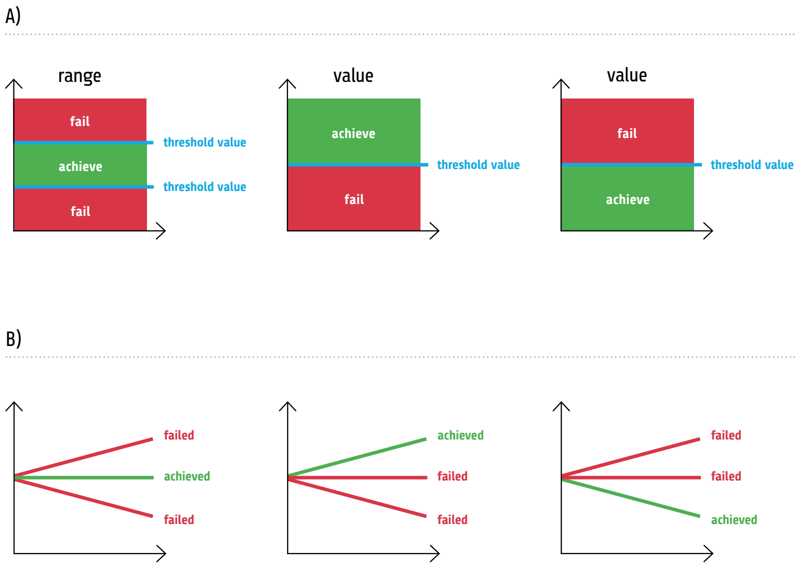


Figure 3.
Assessment approach for the indicator *Abundance of coastal fish key functional groups* (cyprinids/mesopredators). In the baseline approach (A), two threshold values together define the range of indicator values representing good status. This approach can be applied when data spans over at least 15 years. The above figure denotes cases when the baseline represents good status, lower left when the baseline represents poor status due to too low values of the indicator, and lower right when the baseline represents poor status due to too high values of the indicator. The trend-based approach (B) is used when the baseline approach is not applicable. Status is defined based on the direction of the trend compared to the desired direction of the indicator over time. The above figure denotes cases when the beginning of the time-series represents good status, lower left if the beginning of the time-series represents poor status due to too low values of the indicator, and lower right if the beginning of the time-series represents poor status due to too high values of the indicator.

- *Abundance of coastal fish key species* indicator value should be above the site-specific threshold value, which is also specific for each species assessed (Figure 2).
- *Abundance of coastal fish key functional groups* indicator value regarding the group of piscivores should be above the threshold value (Figure 2).
- *Abundance of coastal fish key functional groups* indicator value regarding the group of Cyprinids/mesopredators, should be within an acceptable range (Figure 3).

3.1.1 Assessment protocol

Baseline approach

By the baseline approach, status is assessed in relation to a quantitative threshold value. This is derived based on information from previous years of the time-series that is assessed, which form a baseline.

The following points should be considered when applying the baseline approach (see also Figure 4):

- For the currently addressed coastal fish indicators, data series covering at least 10 years are needed to establish the threshold value, in order to extend over more than twice the generation time of the typical species being represented and cater for natural variation, such as strong and weak year classes.
- The baseline period must represent a stable time period with respect to external conditions. Most substantially, shifts in the Baltic Sea food web structure were apparent in the late 1980s in the open sea (Möllmann *et al.* 2009), and in coastal fish communities in the late 1980s and early/mid 1990s (Olsson *et al.* 2012). In some areas, there have also been minor temporal changes in fish community structure later (Bergström *et al.* 2016a). Stability is addressed by verifying that there is no linear trend ($p<0.1$) in the indicator values, and hence no development towards a change in status during the years representing the baseline.
- The baseline period should be possible to characterize as representing either good or not good status. This can be done either by comparing the baseline data with data dating further back in time, by using additional information on external conditions/indicator values, or by expert judgment. For example, if data from a time period preceding the baseline have much higher indicator values in comparison, the baseline might represent not good status for an indicator where higher values are indicative of a good status.

Based on data from the years representing the baseline, provided that the three criteria above are fulfilled, the threshold value is defined as the value of the indicator at the Xth percentile of the median distribution during the baseline period. The median distribution is computed by resampling (with replacement) from the dataset representing the baseline. In each repetition, the number of samples equals the number of years in the assessment period. In order to improve precision, a smoothing parameter is added in each repetition. The smoothing parameter is computed as the normal standard deviation of the re-sampled dataset divided by the number of samples.

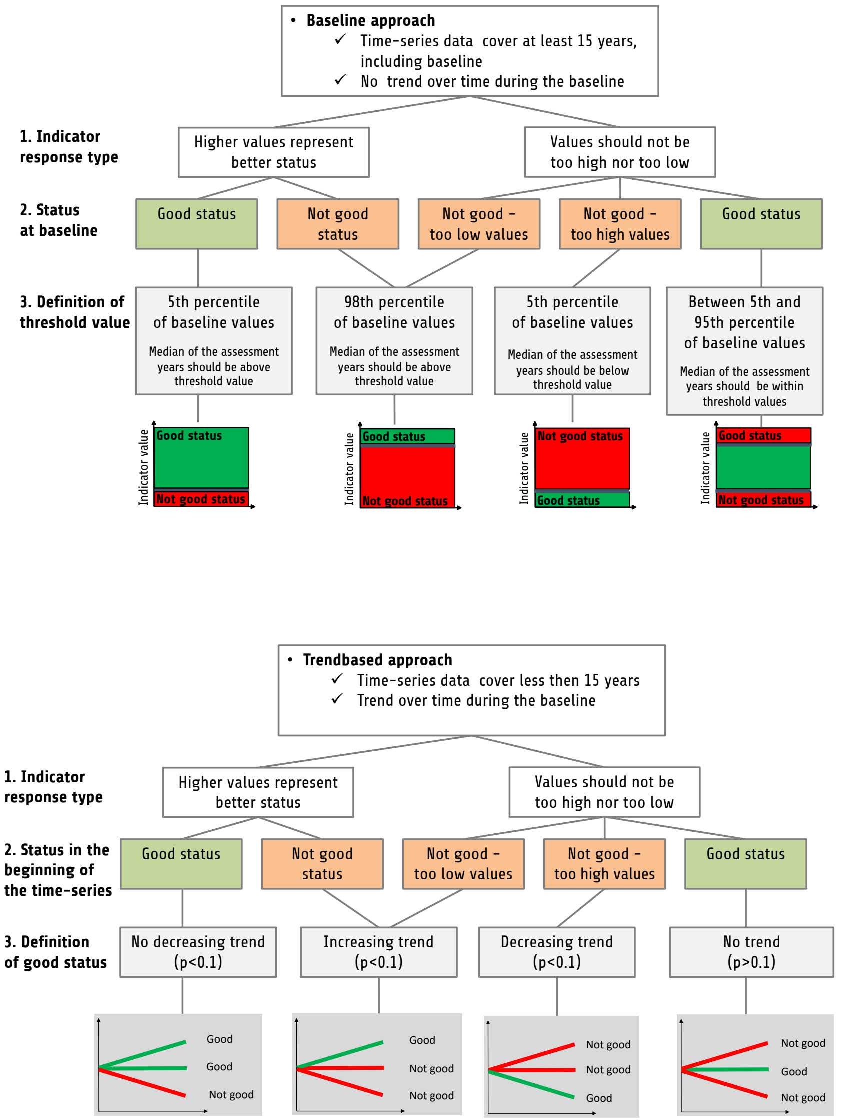


Figure 4.
Decision tree for assessment of coastal fish community indicators. The indicators are abbreviated as follows: abundance of key fish species as 'key species', abundance of piscivores as 'piscivores' and abundance of cyprinids as 'cyprinids'. Baseline refers to the period 1998–2010. $M_{ass\ period}$ refers to the assessment value (2011–2016), perc = percentile, $M_{distr\ baseline}$ refers to the bootstrapped median distribution of the baseline period, and K refers to the slope of the linear regression line over the whole time period at p < 0.1. From HELCOM 2017a.

The assessment period, again, should cover at least five years to cater for natural variability in the indicator value. The assessment value is the median of all indicator values during the assessment period, and is compared with the threshold value, as was defined above. Hence, the assessment in all requires that at least 15 years of data for the assessed time-series is available. In the current assessment, the status of coastal fish communities addresses during the period 2011-2016, and the principal time period for the baseline was the years 1998-2010.

At the indicator level, threshold values are defined as follows for indicators where higher values reflect better status (*Abundance of coastal fish key species* and *Abundance of coastal fish key functional groups* - piscivores):

- If the baseline is defined as representing good status, the threshold value is at the 5th percentile of the median distribution of the dataset during the baseline (Figure 2A and 4).
- If the baseline is defined as representing not good status, the threshold value is at the 98th percentile of the median distribution during the baseline (Figure 2A and 4).

For the indicator *Abundance of coastal fish key functional groups* (cyprinids/mesopredators), both too high and too low values can signal not good status, and threshold values are defined as follows:

- If the baseline is defined as representing good status, two threshold values are used. The lower threshold value is at the 5th percentile and the upper value is at the 98th percentile of the median distribution during the baseline (Figure 3A and 4).
- If the baseline state is defined as representing not good status, only one threshold value is used but it is defined differently depending on the desired direction of the indicator (Figure 3A). If the indicator values during the baseline are too high, the indicator should decrease in order to reflect improved status and the 5th percentile of the median distribution is used as a threshold value (Figure 4). If the indicator values during the baseline are too low, the 98th percentile of the median distribution of the dataset during the baseline is used as the threshold value (Figure 4).

Trend-based approach (time series < 15 years)

When using the trend-based approach the assessment evaluates the predominating trend over time based on the available time-series. The trend-based approach is used if the requirements for a baseline approach are not met (due to too short time series, or the presence of a linear development during the proposed baseline period, for example; Figure 4). As a minimum requirement for how long time-series can be used, time-series with available data dating back to at least the mid-2000s

(2008) were included in the current assessment.

In the trend-based approach, status is defined based on the observed direction of the indicator trend compared to the desired direction over time (Figure 2B and 3B). The desired direction is evaluated as for the baseline approach. That is, the situation in the beginning of the time-series is evaluated by inspecting preceding data, if available, by using additional information on external conditions/indicator values, or by expert judgment.

For the indicators *Abundance of coastal fish key species* and *Abundance of coastal fish key functional groups* (piscivores), where higher values reflect better status, the following evaluations apply (see Figure 4):

— If the beginning of the time series represents good status, the trend of the indicator over time should not be negative in order to represent good status (Figure 2B).
— If the beginning of the time series represents not good status, the trend in the indicator should be positive in order to represent good status (Figure 2B).

For the *Abundance of coastal fish key functional groups* (cyprinids/mesopredators), indicator values should neither be too high nor low, and the following evaluation applies (see Figure 4):

— If the beginning of the time series represents good status, there should be no directional trend in the indicator over time in order to represent good status (Figure 3B).
— If the beginning of the time series represents not good status, the trend of the indicator over time must be negative if indicator values in the beginning of the time-series are too high (Figure 3B). If, on the other hand, values of the indicator are too low in the beginning of the time-series, the direction of the trend must be positive in order to represent good status (Figure 3B).

Data should be ln-transformed to enhance linearity. The presence of a trend is assessed at the level of significance $p < 0.1$.

3.1.2 Assessment units and aggregation

The assessments have been reported at scale 3 of HELCOM assessments units; 'Open sub-basin and coastal waters'. However, the indicators are not applicable in the open sea sub-basins.

Since the status evaluations of coastal fish communities are representative at rather small geographical scales (see Chapter 1), the assessments are first carried out at the scale of each monitoring area. Assessment results at the scale of sub-basin coastal waters (unit 3) have been obtained by merging the assessment results for each monitoring area, using conditional rules.

The overall status in each assessment unit (Sub-basin coastal water) is determined as the status of the majority of the monitoring areas evaluated within that unit. If there is an equal number of monitoring areas with good and not good status, the status of the assessments unit is determined as not good.

3.1.3 Data used in the assessment

The evaluations are based on data from fishery independent monitoring, recreational catch registration and/or commercial fisheries catch statistics. For detailed information on the data and areas included in the assessment, see Appendix 1.

Fishery independent monitoring

The evaluations are based on catch per unit effort (CPUE) data given as abundance (number of individuals of the species included in the indicator) per unit gear type. The data represents annual averages of all monitored stations in each area. To only include fish of sizes that are sampled representatively in the gear (hence being suited for quantitative evaluation of mean abundances) individuals smaller than 12 cm (in the case of Nordic Coastal multimesh nets) or 14 cm (other net types) are excluded. Abundance is calculated as the number of individuals of the species included in the indicator per unit effort (CPUE).

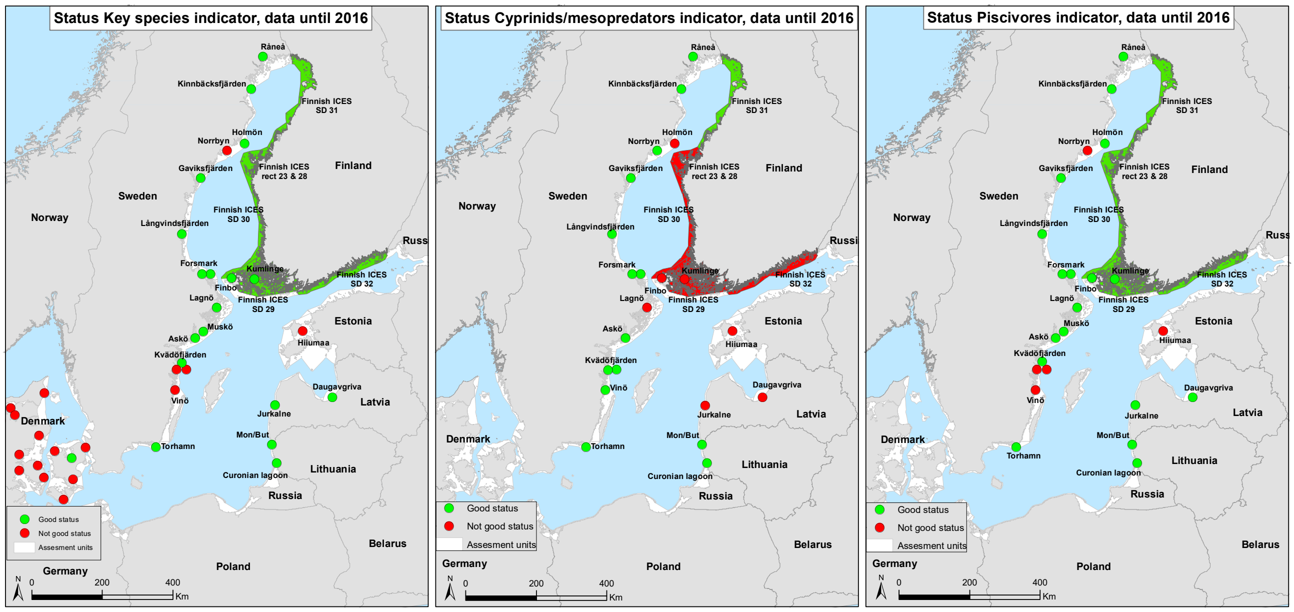
Figure 5.
Position of the gillnet (left) and fykenet (right) data for all the years that the recreational catch registration has taken place. Some of these positions may not have been fished each year during the assessment period, and some may also have changed with the recruitment of new fishermen.

Commercial catch data

The evaluations are based on catch per unit effort data (CPUE) given as biomass (kg/gillnet day), and each data point represents the total annual catches per area. The gillnets used in the fishery have mesh sizes between 36-60 mm (bar length) and hence target a somewhat different aspect of the fish community in the area, compared to the fisheries independent monitoring data. Further, the fishing is not performed at fixed stations, nor with a constant effort across years. As a result, the estimates from the gillnet monitoring programmes and commercial catch data are not directly comparable, and only relative changes in status across data sources should be compared.

Recreational catch registration

The evaluations are based on catch per unit effort data (CPUE) provided by a citizen science monitoring programme (Støttrup *et al.*, in review), and are given as abundance (number of individuals of the species used in the indicator) caught in a uniform gear and standardised to a twelve hour fishing period. The gear utilised are gillnets (monofilament, mesh size: 65mm, mesh depth: 8.5kn, knot 120 length: 2400 kn, floatline nr. 1.25, sinkline nr 1.5, mounted length: 39 m) or fyke nets (80/7 with 8 m net between the two traps). Both gear types were fished at fixed positions up to three times in the month of August, to coincide with the other monitoring methods, however, some of these stations moved between years. Only fish >14 cm were included in the analyses. The number of stations sampled each year did to some extent differ within and between assessment units (Figure 5). For the purpose of the assessment in this report, data from individual stations (fishermen) was aggregated according to the areas presented in figure 6 and table 3 below.



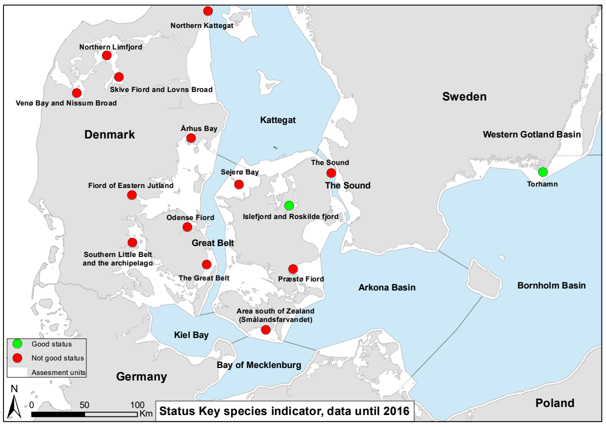
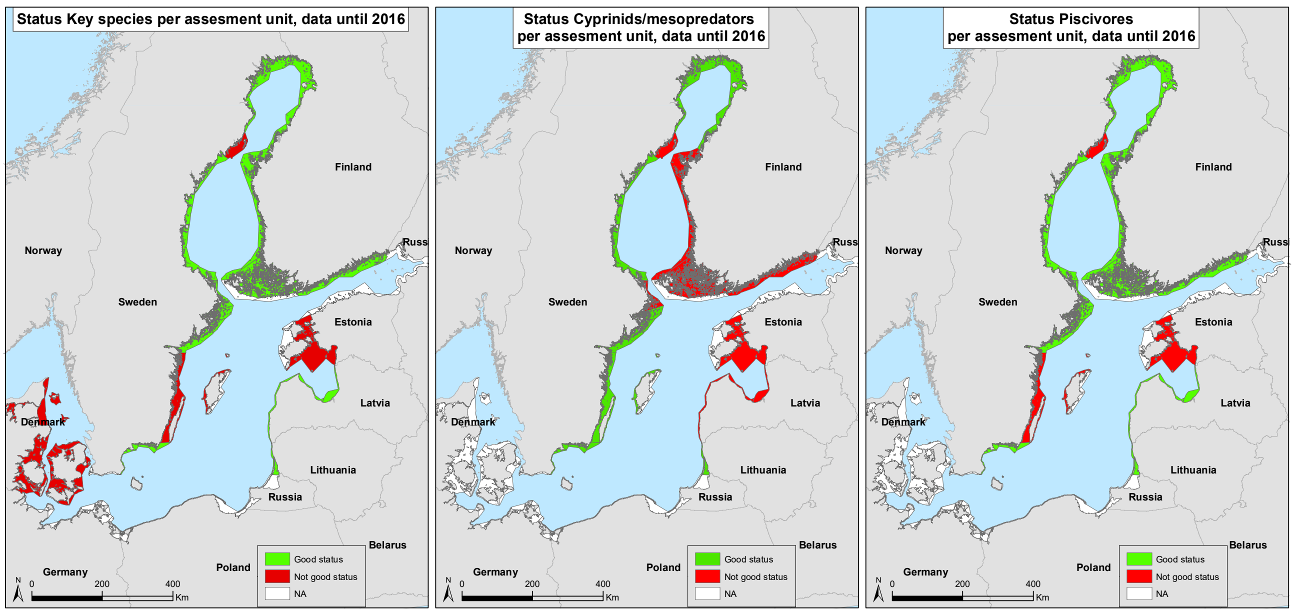


Figure 6.
Status of coastal fish during 2011–2016 for the three indicators assessed. The top panel shows status per monitoring area, representing the level at which status is originally evaluated. The middle panel a zoomed in map of the status per monitoring area in Danish water for the Key species indicator. The lower panel shows aggregated status per assessment unit, which is determined based on a conditional approach in the case were assessment results from more than one monitoring area are available.



3.2. Assessment results

3.2.1 Status per sub-basin

Bothnian Bay

In the Bothnian Bay, three data sets (one Finnish and two Swedish) were evaluated; the Finnish ICES SD31 (Finland), Råneå (Sweden) and Kinnbäcksfjärden (Sweden). All three data sets suggest good status for all three indicators (Figure 6 and 7; Tables 3-5). The indicators have been stable over time in all three areas.

The Quark

Four data sets, two Finnish (ICES rectangle 28 and 23) and two Swedish (Norrbyn and Holmön, were evaluated for the Quark (Tables 3-5). For the indicator Key species (perch), both Finnish data sets indicated good status, but only one of the Swedish data sets (Holmön; Figure 8). Hence, the status for Key species in the assessment unit 'Quark Finnish coastal waters' is good, and for the assessment unit 'Quark Swedish coastal waters' the status is not good (Table 3). For the indicator Cyprinids, both data sets in the Finnish assessment unit show not good status due to high abundances of Cyprinids. Along the Swedish coast, the Norrbyn data set show stable abundances of Cyprinids whereas Holmön show increasing abundance, rendering the overall status of the assessment unit as not good (Figure 8; Table 4). The results for the indicator Piscivores are similar to those of the Key species indicator; good status in the Finnish assessment unit, and not good status in Norrbyn and good status in Holmön in the Swedish assessment unit (Figure 8; Table 5).

Bothnian Sea

In the Bothnian Sea five data sets were evaluated; one Finnish (ICES SD 30) and four Swedish (Gaviksfjärden, Långvindsfjärden, Forsmark and Forsmark long time-series). Hence, in the Swedish assessment unit two different time-series were available for the area Forsmark, with different length and monitoring method (Figure 9). All data sets show good status for the indicators Key species and Piscivores (Figure 9), yielding overall good status for these in both the Finnish and Swedish assessment units (Tables 3 and 5). In the Finnish ICES SD 30 there is a positive trend for the Piscivore indicator during 2008-2016, and for Cyprinids a high abundance, rendering not good status of the assessment unit, whereas all four Swedish data sets show no directional development and hence an overall good status of the assessment unit (Figure 9: Table 4).

Åland Sea

Only one data set from Sweden was available for evaluation of the Åland Sea sub-basin; Lagnö (Sweden). The indicators Key species and Piscivores indicate good status, whereas the indicator Cyprinids indicate not good status due to increasing abundances of the ecosystem component (Figure 10; Tables 3-5).

Archipelago Sea

Three Finnish data sets were evaluated for the Archipelago Sea sub-basin; Finnish ICES SD29, Finbo and Kumlinge. The indicators Key species and Piscivores show good status for all areas, reflecting stable or increasing abundances (Figure 11; Tables 3 and 5), whereas the indicator Cyprinids show not good status in all areas as a result of increasing or too high abundances (Figure 11; Table 4).

Northern Baltic Sea

In this sub-basin, only data from two Swedish areas: Askö and Muskö, were available for assessments. The monitoring in Muskö is carried out during autumn, and flounder is used as the Key species, and the indicator Cyprinids is not assessed for this data set due to low representation of cyprinids in the fish monitoring during autumn. For all three indicators, the status is good in both areas, due to stable abundances over the years, yielding an overall good status in the assessment unit (Figure 12; Tables 3-5).

Gulf of Finland

Only data from the Finnish coast was available for assessments in the Gulf of Finland; ICES SD 32. For the indicators Key species and Piscivores there has been abundances over time, yielding good status in the assessment unit (Figure 13; Tables 3 and 5). For the Cyprinid indicator, the status of the assessment unit is not good due to increasing abundances during the past eight years (Table 4).

Gulf of Riga

Two data sets, one from the northern parts; Hiiumaa (Estonia), and one from the more southern parts; Daugavgriva (Latvia), were included for the assessment of the Gulf of Riga. Due to lack of funding, monitoring in Daugavgriva was not undertaken in 2014, and monitoring was carried out using a different gear in 2016. Comparable data for this area is only available until 2015. In the Estonian assessment unit, the status is not good for all three indicators due to low abundances during recent years (Figure 14; Tables 3-5). In the Latvian assessment unit, the status is good for the Key species and piscivore indicator, but not good for the indicator Cyprinids due to increasing abundances (Figure 14; Tables 3-5). In 2015, however, the abundance of cyprinids was very low, which is likely as a result of extraordinary low water temperatures during monitoring in the area.

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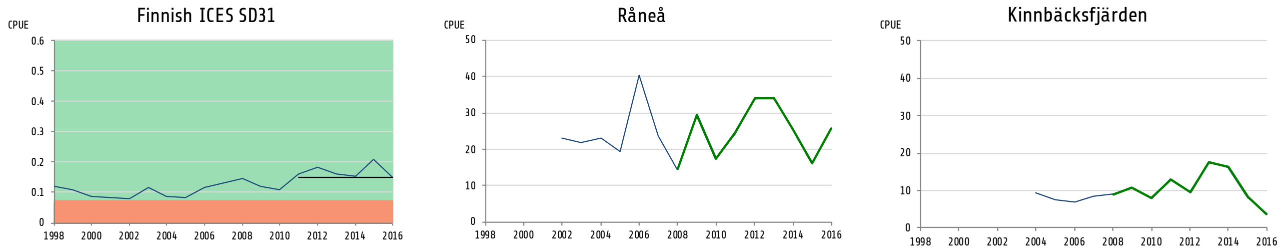
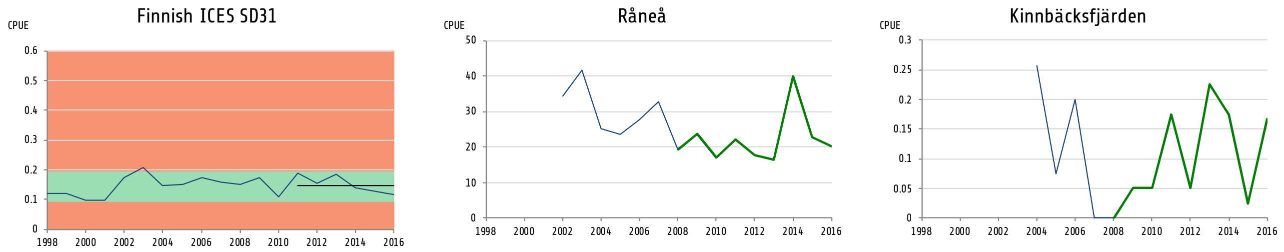
Key species



Cyprinids



Piscivores

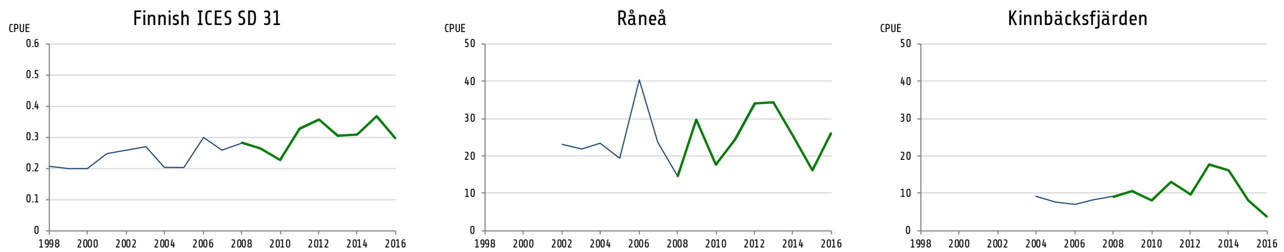
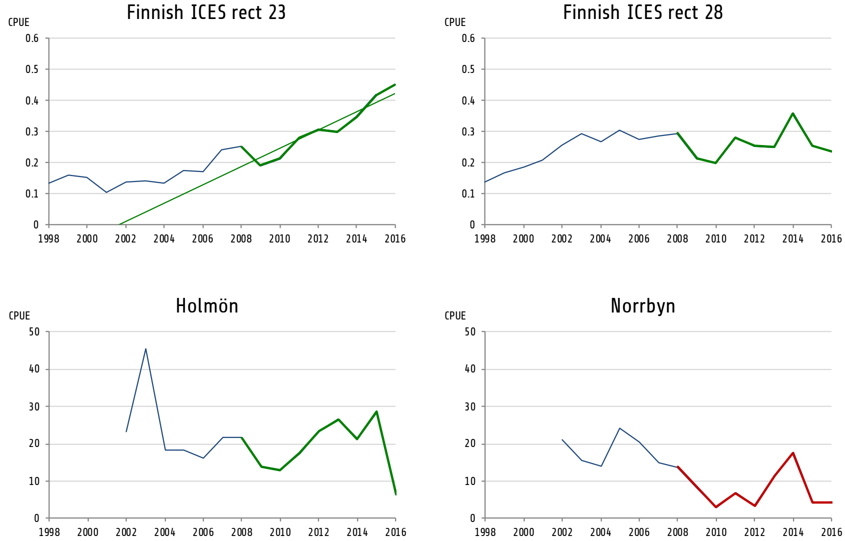
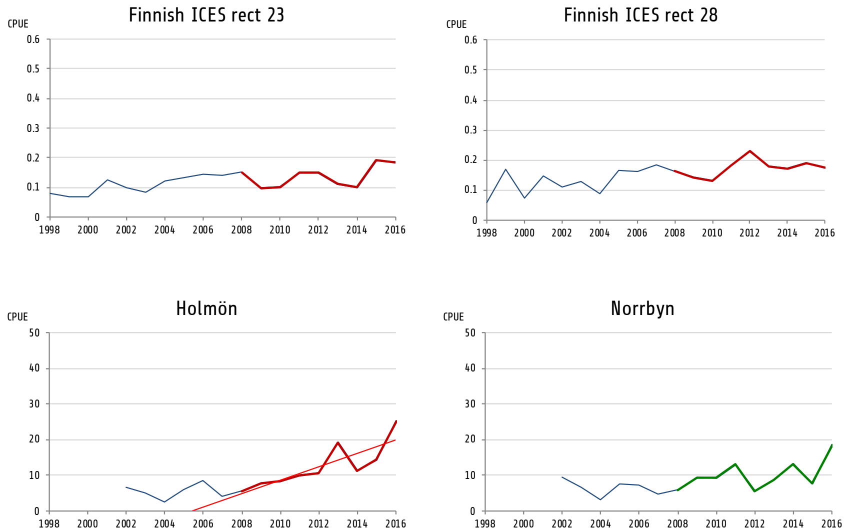


Figure 7: Bothnian Bay
Temporal development of the indicators Key species (top), Cyprinids (middle) and Piscivores (bottom) for the three evaluated areas in the Bothnian Bay. Status is evaluated using the baseline approach for the Key species and Cyprinids indicators in Finnish ICES SD31, and using the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011-2016. Here good status is achieved in both cases. In the trend-based assessments (all other areas), the indicators meets the conditions for good status, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

Key species



Cyprinids



Piscivores

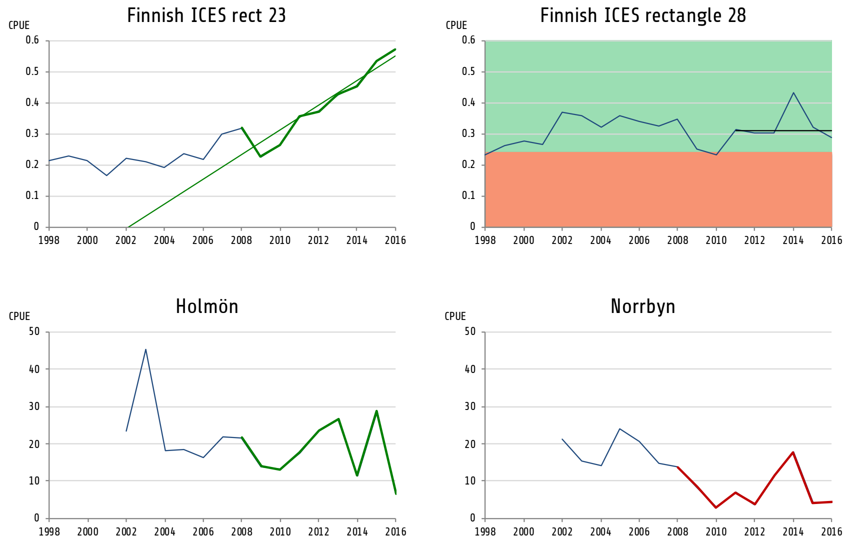
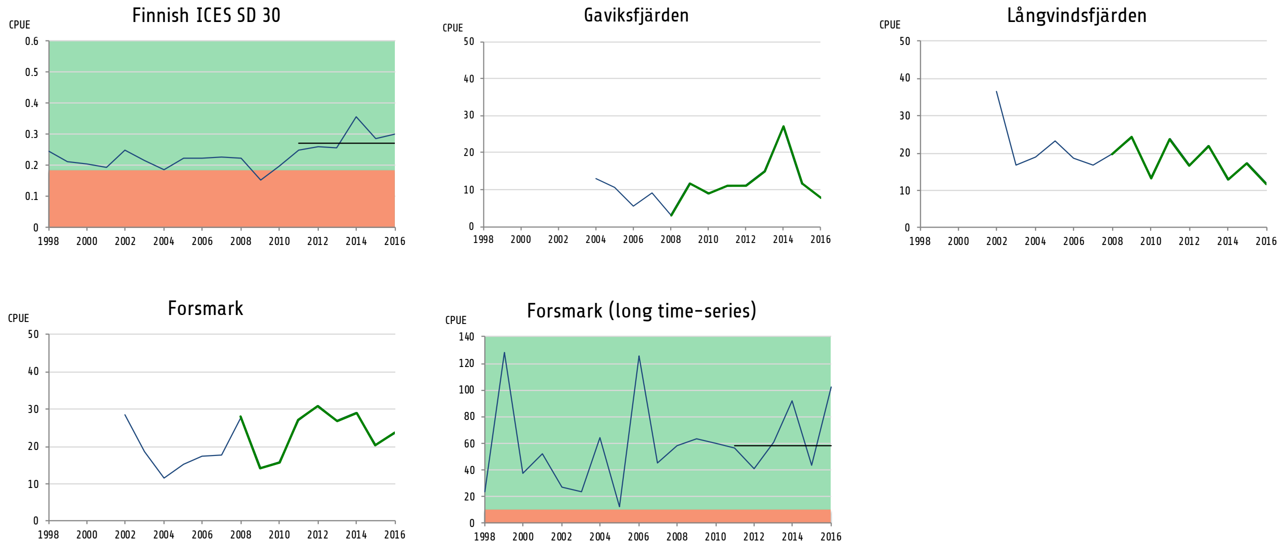
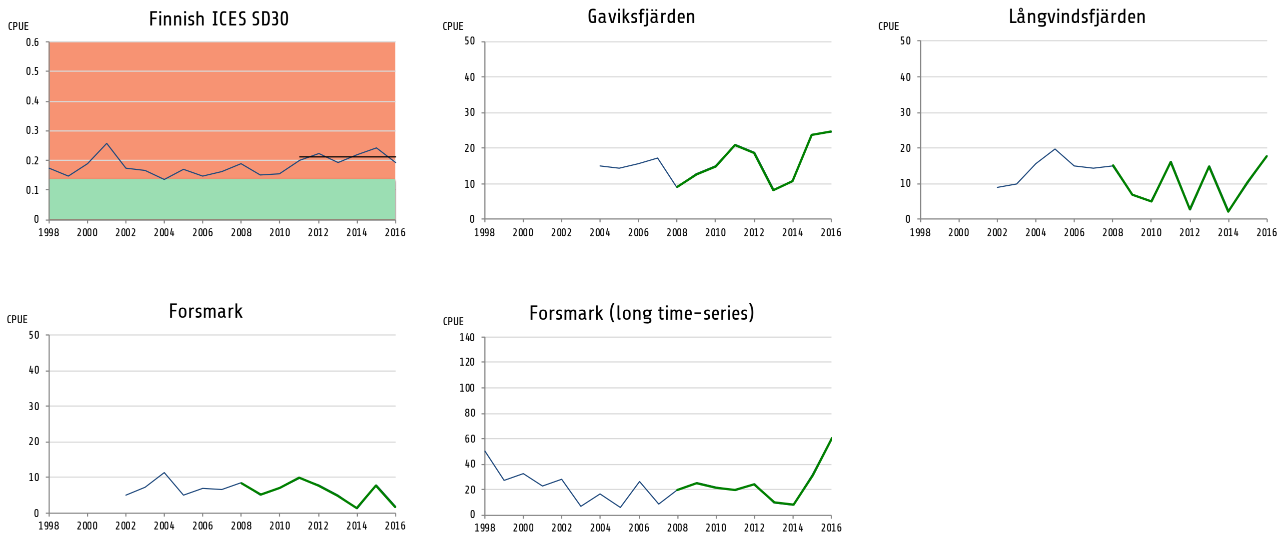


Figure 8: The Quark
Temporal development of the indicators Key species (top), Cyprinids (middle) and Piscivores (bottom) for the four evaluated areas in the Quark. Status is evaluated using the baseline approach for the Piscivores indicator in Finnish ICES rectangle 28, and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values are not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011-2016. Here good status is achieved in the data set. In the trend-based assessments (all other areas), the thin trend-line is shown when there is a significant linear trend between 2008-2016 at p<0.1. The indicators meets the conditions for good status in six out of 11 of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

Key species



Cyprinids



Piscivores

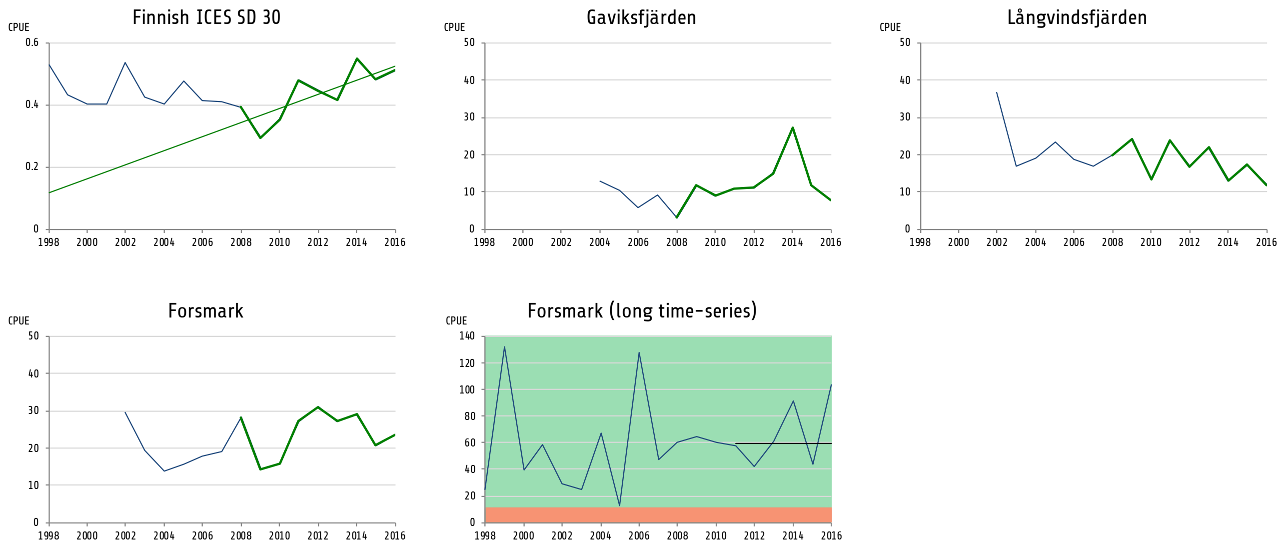


Figure 9: Bothnian Sea

Temporal development of the indicators Key species (top), Cyprinids (middle) and Piscivores (bottom) for the five evaluated data sets in the Bothnian Bay. Status is evaluated using the baseline approach for the Key species (Finnish ICES SD 30, Forsmark long-time-series), Cyprinids (Finnish ICES SD 30) and Piscivores (Forsmark long-time-series) indicators, and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011-2016. Here good status is achieved in all cases except for Cyprinids in the Finnish ICES SD 30. In the trend-based assessments (all other areas), the thin trend-line is shown when there is a significant linear trend between 2008-2016 at p<0.1. The indicators meets the conditions for good status in all trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

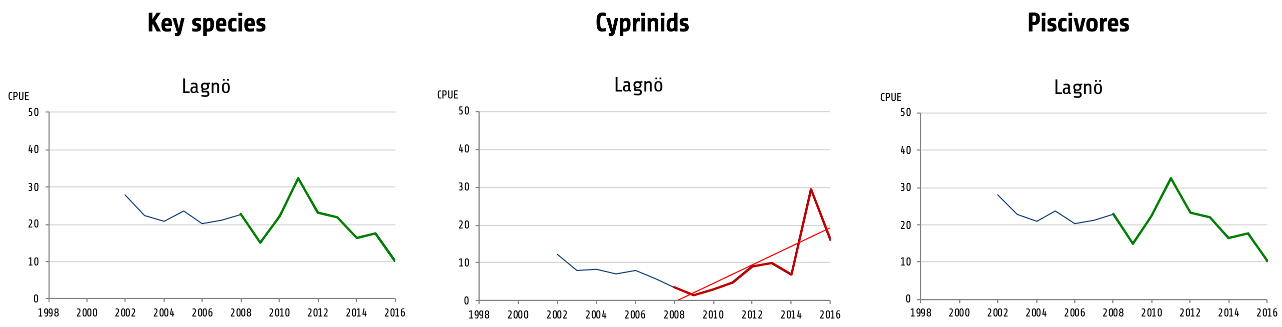
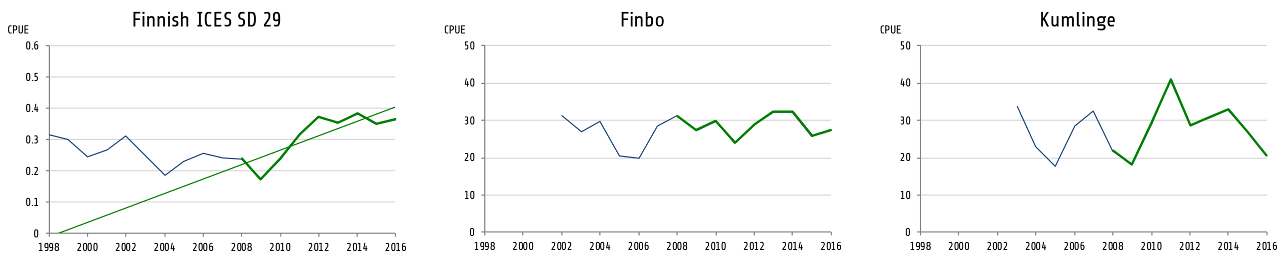
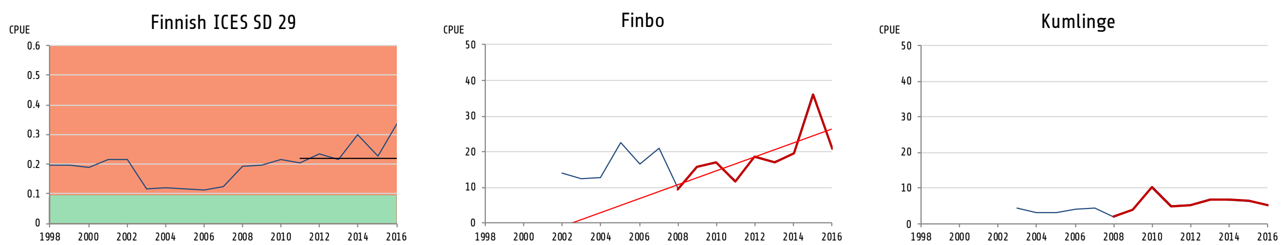


Figure 10: Åland Sea
Temporal development of the indicators Key species (left), Cyprinids (middle) and Piscivores (right) for the evaluated area in the Åland Sea. Status is evaluated using the trend-based approach for all indicators. A thin trend-line is shown when there is a significant linear trend between 2008-2016 at p<0.1. The indicators meets the conditions for good status in two out of three of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

Key species



Cyprinids



Piscivores

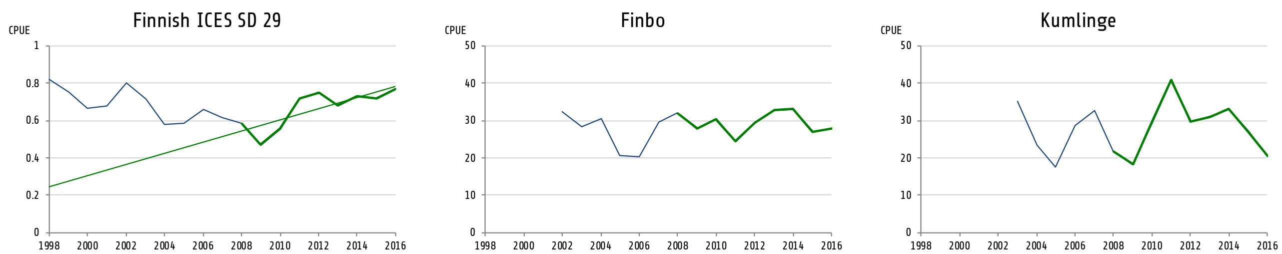
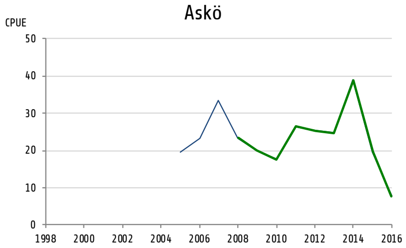
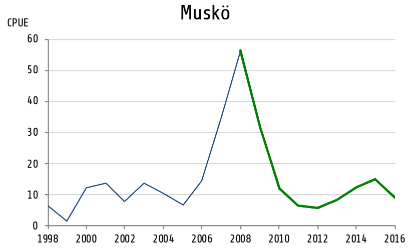


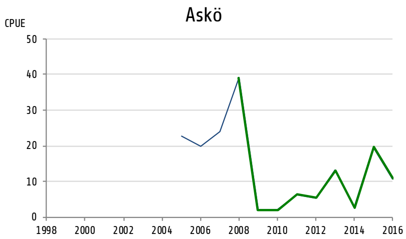
Figure 11: Archipelago Sea
Temporal development of the indicators Key species (top), Cyprinids (middle) and Piscivores (bottom) for the three evaluated areas in the Archipelago Sea. Status is evaluated using the baseline approach for the Cyprinids Finnish ICES SD 29, and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011-2016, indicating that good status is not achieved for Cyprinids in the Finnish ICES SD 29 area. In the trend-based assessments (all other areas), the thin trend-line is shown when there is a significant linear trend between 2008-2016 at p<0.1. The indicators meets the conditions for good status in six out of eight of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

Key species

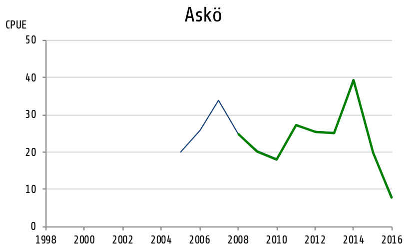




Cyprinids



Piscivores



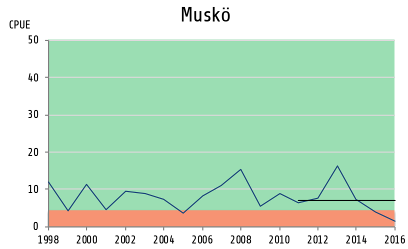
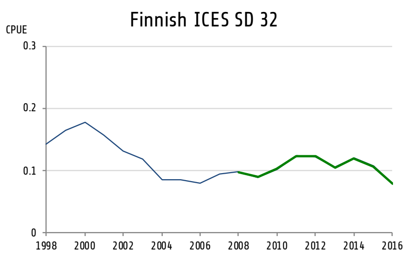
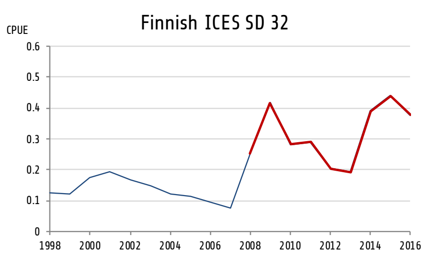


Figure 12: Northern Baltic Sea
Temporal development of the indicators Key species (top), Cyprinids (bottom left) and Piscivores (bottom and middle right) for the two evaluated areas in the Northern Baltic Sea. Status is evaluated using the baseline approach for Piscivores in Muskö, and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011–2016. Here good status is achieved in Muskö. The indicators meet the conditions for good status in all trend-based assessment cases (all other areas) in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3–5 below.

Key species



Cyprinids



Piscivores

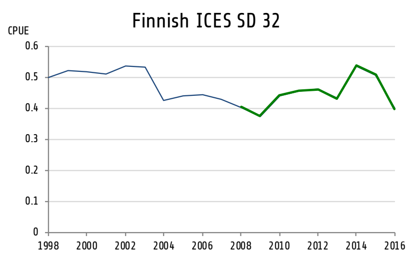
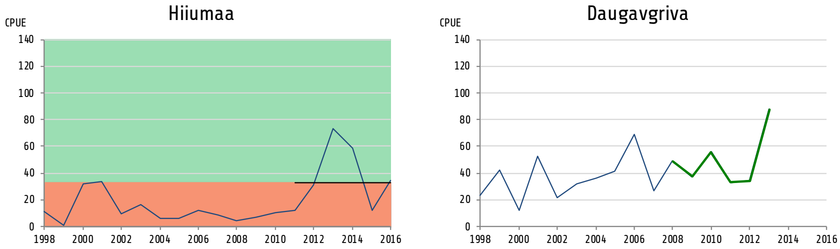
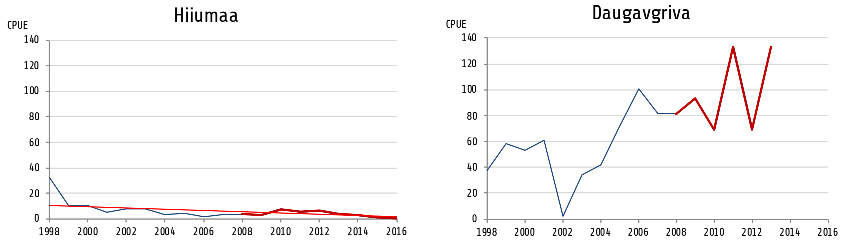


Figure 13: Gulf of Finland
Temporal development of the indicators Key species (left), Cyprinids (middle) and Piscivores (right) for the evaluated area in the Gulf of Finland. Status is evaluated using the trend-based approach for all indicators. The indicator meets the conditions for good status in two out of three of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3–5 below.

Key species



Cyprinids



Piscivores

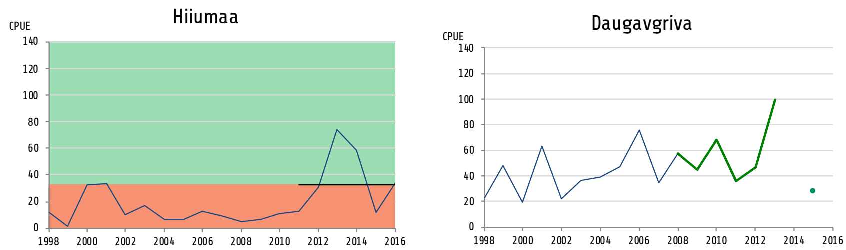


Figure 14: Gulf of Riga
Temporal development of the indicators Key species (top), Cyprinids (bottom left) and Piscivores (bottom and middle right) for the two evaluated areas in the Gulf of Riga. Status is evaluated using the baseline approach for Key species (both areas) and Piscivores (Hiiumaa), and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011–2016. Here, good status is only achieved in Daugavgriva for Key species. In the trend-based assessments (all other areas), the thin trend-line is shown when there is a significant linear trend between 2008–2016 at p<0.1. The indicators meets the conditions for good status in one out of three of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3–5 below.

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Western Gotland basin

In the Western Gotland basin, data from two areas in Sweden; Kvädöfjärden and Vinö were available for assessment. In the Kvädöfjärden area, monitoring is undertaken using two parallel gears during summer (data sets named Kvädöfjärden and Kvädöfjärden long time-series), and also in the autumn (data set named Kvädöfjärden autumn). Hence, four data sets are available to assess the status of Piscivores and Key species. The assessment of Key species is based on perch in the summer data sets and on flounder in the Kvädöfjärden autumn data set. Three data sets are available to assess Cyprinids, since this indicator is not supported by the autumn sampling. The Key species indicator showed not good status in all but the Kvädöfjärden long time-series data set, (Figure 15; Table 3). This renders an overall not good status to the assessment unit for the indicator. In contrast, all three evaluations of the Cyprinid indicator indicated good status, giving an overall status of good status for Cyprinids in the assessment unit (Figure 15; Table 4). For the Piscivore indicator, the data set representing Kvädöfjärden long time-series showed good status, whereas the other three data sets showed not good status (Figure 15; Table 5). The overall status for the Piscivores indicator was hence not good status in the assessment unit.

Eastern Gotland basin

For this sub-basin, data from one Latvian (Jurkalne) and two Lithuanian (Monciskes/Butinge and Curonian Lagoon) areas were available for assessment. The indicator Key species in Jurkalne and Monciskes/Butinge are represented by flounder, and in Curonian Lagoon by perch. Further, in Monciskes/Butinge the status of Mesopredators is assessed instead of Cyprinids due to low natural occurrence of cyprinids. Monitoring was not undertaken in Jurkalne in 2015 due to lack of funding, and in 2016 another monitoring gear was used in the area. Hence comparable data for assessments is only available until 2014. For the two Lithiuanian areas, funding for monitoring ended in 2012 and the assessments of Monciskes/Butinge and Curonian Lagoon areas are therefore based on data until 2012.

In all three areas, the indicators Key species and Piscivores indicated good status, reflecting rather high and stable abundances (Figure 16; Tables 3 and 5), and the overall status in the both the Latvian and Lithuanian assessment units of the sub-basin was assessed as good. For the Cyprinids/Mesopredator indicator, the status was not good in the Latvian assessment unit (due to too low abundances of cyprinids), and GES in both Lithuanian data set, indicating good status in the assessment unit ‘Lithuanian coastal waters” (Figure 16; Table 4).

Gdansk basin

No monitoring data was available to support an assessment of this sub-basin

Bornholm basin

In the Bornholm basin, only data from one Swedish area, Torhamn, was available for assessment. All three indicators indicate good status, reflecting increasing abundances of the Key species (perch) and of Piscivores, and stable abundances of Cyprinids (Figure 17; Tables 3-5).

Arkona basin

Only data to assess the indicator Key species was available for this sub-area, from one Danish area; Præstø fjord, where flounder is used as Key species. Data was only available for five years (Figure 18), which lowered the confidence of the assessment (See also section 3.2.2). However, the available data suggest low catches of flounder compared to what would be expected, yielding not good status for the assessment unit (Figure 18; Table 3).

Mecklenburg bight

Only data to assess the Key species indicator (flounder) was available, from one Danish area; Smålandsfarvandet. Data was only available for seven years, resulting in a lowered confidence (see also section 3.2.2). The existing data suggest low abundances of flounder and not good status for the assessment unit (Figure 19; Table 3).

Kiel bight

No monitoring data was available to support an assessment of this sub-basin

Belt Sea

In this sub-basin, data for six Danish areas are available for assessment, however the information is confined to assessing the status of Key species (flounder). Despite higher numbers of flounder in some areas during more recent years, the overall catches are low and the status is not good in all areas (Figure 20; Table 3).

The Sound

Data from one Danish area; The Sound, were available for assessment in the Sound, confined to supporting the indicator Key species (flounder). Status was assessed as not good due to very low abundances of flounder during the last years of the assessment period (Figure 21; Table 3).

Kattegat

Data from five Danish areas; Islefjord and Roskilde fiord, Northern Kattegat, Northern Limfiord, Skive Fiord and Lovns Broad and Venø Bay and Nissum Broad, were available for assessments of the Kattegat sub-basin. Only data supporting the Key species indicator (flounder) was however available. The catches of flounder have been low or

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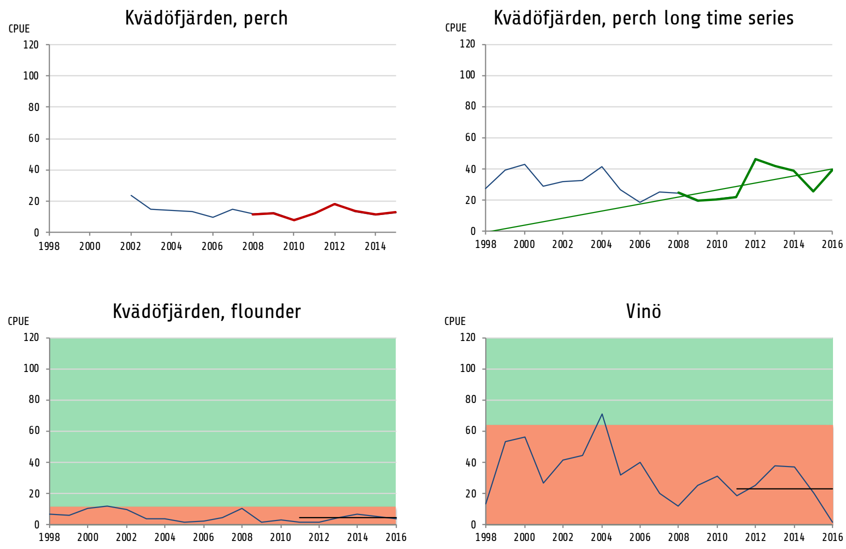
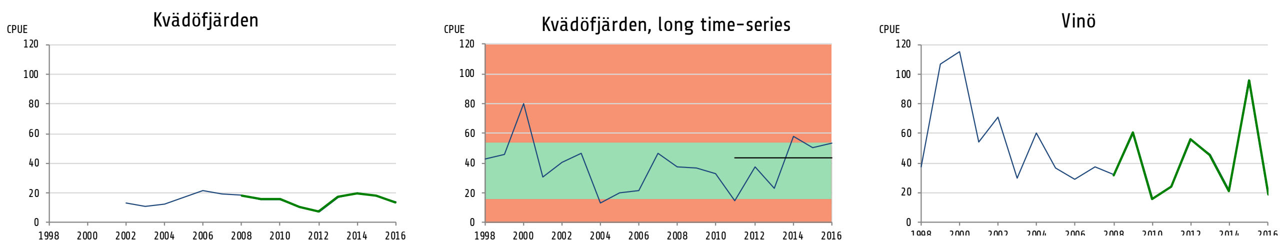
Key species



Cyprinids



Piscivores

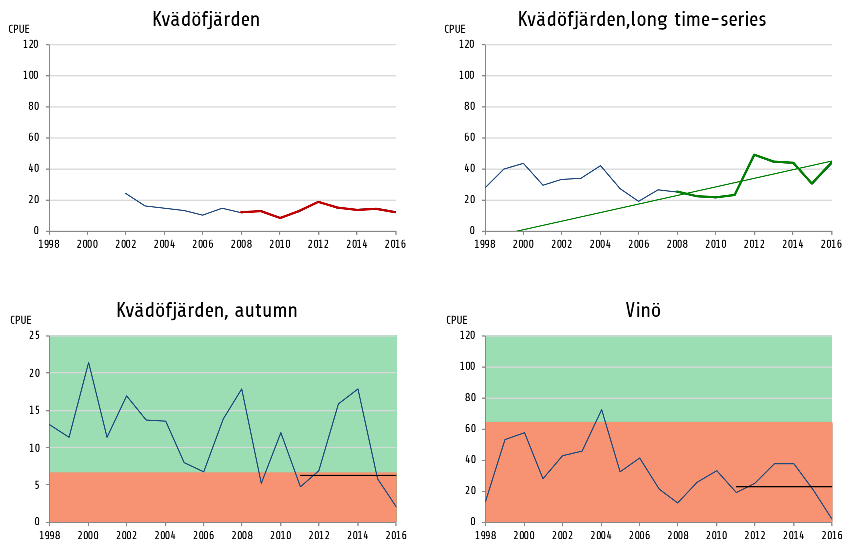
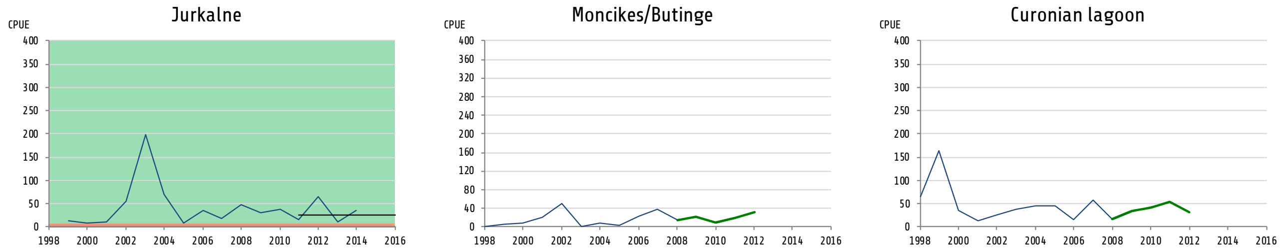
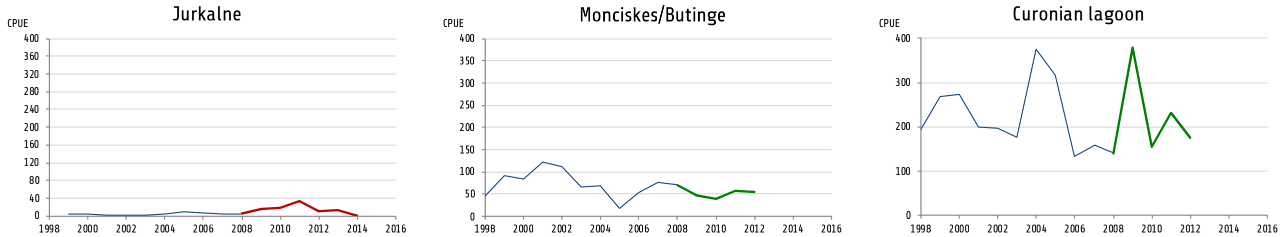


Figure 15: Western Gotland Basin
Temporal development of the indicators Key species (top), Cyprinids (middle) and Piscivores (bottom) for the four evaluated data sets in the Western Gotland Basin. Status is evaluated using the baseline approach for Key species (Kvädöfjärden, autumn and Vinö), Cyprinids (Kvädöfjärden, long times-series) and Piscivores (Kvädöfjärden, autumn and Vinö), and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011–2016. In the trend-based assessments (all other areas), the thin trend-line is shown when there is a significant linear trend between 2008–2016 at p<0.1. For more details on threshold values, assessment values and evaluated status, see tables 3–5 below.

Key species



Cyprinids/Mesopredators



Piscivores

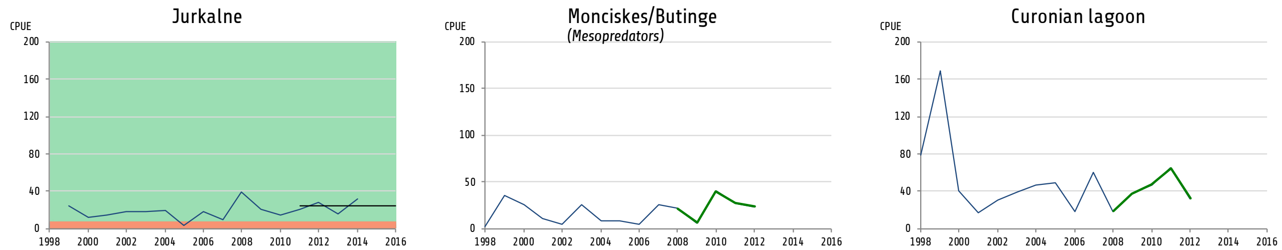


Figure 16: Eastern Gotland Basin
Temporal development of the indicators Key species (top), Cyprinids (middle) and Piscivores (bottom) for the three evaluated areas in the Eastern Gotland Basin. Status is evaluated using the baseline approach for Key species and Piscivores in Jurkalne, and the trend-based approach in all other cases. For the baseline approach, the red fields denote areas within which the threshold values is not achieved and green fields denote areas where the threshold value is achieved, suggesting good status. The black horizontal line shows the assessment value, which is the median value for years 2011-2016. Here, good status is achieved for Key species and Piscivores in Jurkalne. The indicators meets the conditions for good status in six out of seven of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

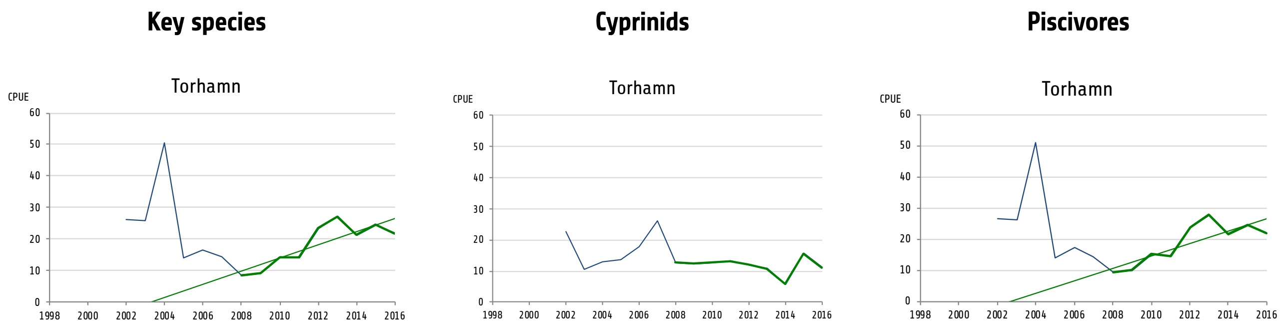


Figure 17: Bornholm Basin
Temporal development of the indicators Key species (left), Cyprinids (middle) and Piscivores (right) for the evaluated area in the Bornholm basin. Status is evaluated using the trend-based approach for all indicators. A thin trend-line is shown when there is a significant linear trend between 2008-2016 at p<0.1. The indicators meet the conditions for good status in all of the trend-based assessment cases in the sub-basin, as denoted by the thick green line. For more details on threshold values, assessment values and evaluated status, see tables 3-5 below.

Key species

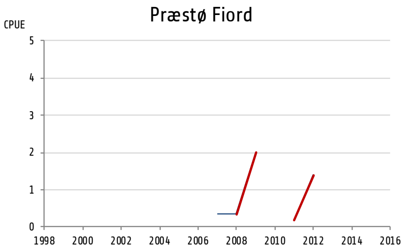


Figure 18: Arkona Basin
Temporal development of the indicator Key species for the evaluated area in the Arkona basin. Status is evaluated using the trend-based approach. The indicator does not meet the conditions for good status in the area, as denoted by the thick red line. For more details on threshold values, assessment values and evaluated status, see table 3 below.

Key species

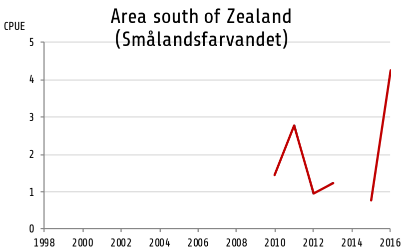


Figure 19: Mecklenburg bight
Temporal development of the indicator Key species for the evaluated area in the Mecklenburg bight. Status is evaluated using the trend-based approach. The indicator does not meet the conditions for good status in the area, as denoted by the thick red line. For more details on threshold values, assessment values and evaluated status, see table 3 below.

Key species

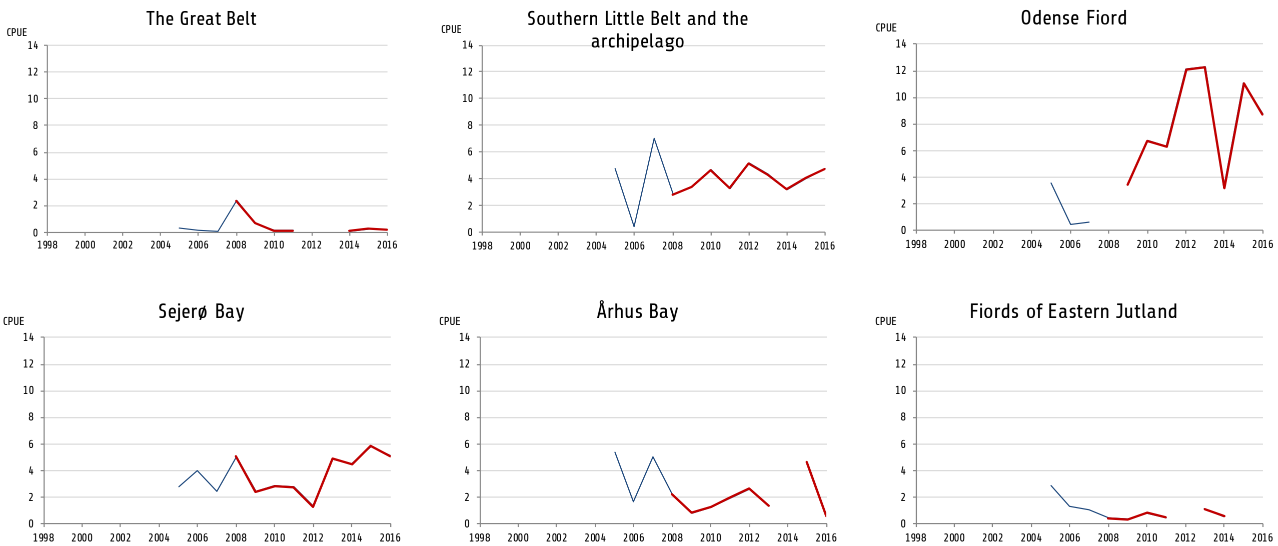


Figure 20: Belt Sea
Temporal development of the indicator Key species for the six evaluated areas in the Belt Sea. Status is evaluated using the trend-based approach in all areas. The indicator does not meet the conditions for good status in any of the areas, as denoted by the thick red line. For more details on threshold values, assessment values and evaluated status, see table 3 below.

Key species

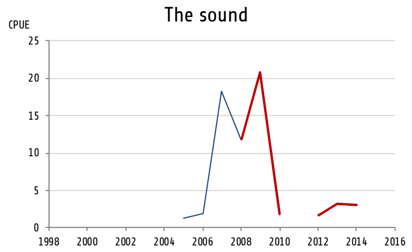


Figure 21: The Sound
Temporal development of the indicator Key species for the evaluated area in The Sound. Status is evaluated using the trend-based approach. The indicator does not meet the conditions for good status in the area, as denoted by the thick red line. For more details on threshold values, assessment values and evaluated status, see table 3 below.

Key species

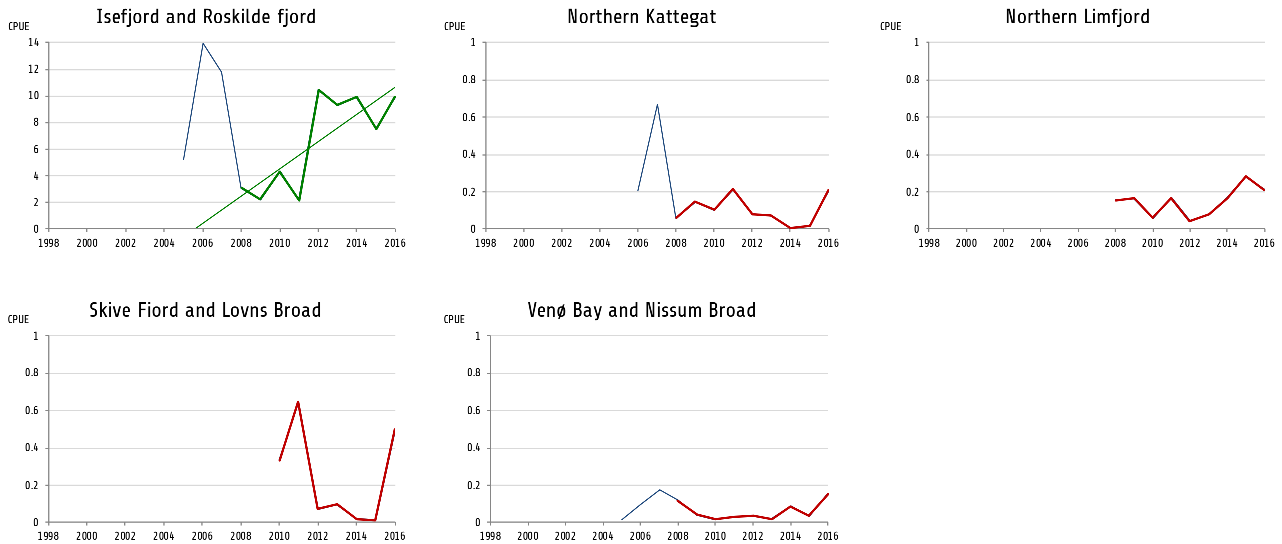


Figure 22: Kattegat
Temporal development of the indicator Key species for the five evaluated areas in Kattegat. Status is evaluated using the trend-based approach in all areas. A thin trend-line is shown when there is a significant linear trend between 2008-2016 at p<0.1. The indicator does not meet the conditions for good status in four out of five of the areas, as denoted by the thick red line. For more details on threshold values, assessment values and evaluated status, see table 3 below.

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decreasing in four of the five areas during the recent years, indicating not good status (Figure 22; Table 3). In one of the areas (Islefjord and Roskilde fiord), the catches of flounder have increased, suggesting good status in this area. The overall status of the Danish assessment unit is assessed as not good (Table 3).

Summary of status

Considerable differences are apparent among monitoring areas in the status assessment, as also expected due to the low dispersal distances of several Baltic Sea coastal fish species. Nonetheless, there are some overarching patterns in the assessed indicators that can be seen at Baltic scale during the time-period considered. In general, the overall status of coastal fish, based on the three assessed indicators, is considered as rather poor. Approximately half of the assessment units obtain good status. According to the assessment, the status is better in the more northern areas of the Baltic Sea and in areas where perch is used as the Key species.

The Key species indicator was evaluated in 43 areas, covering 21 out of 42 coastal assessment units, and 16 out of 18 sub-basins (Figure 6). Where perch is used as the Key species (more northern areas), the status is most often assessed as good, and good status is achieved in 21 of 25 areas assessed (Table 3). The overall status is poorer in more southern sub-basins, where flounder is used as key species. Good status is only achieved in four out of 18 areas assessed (Figure 6, Table 3). Combining information for the two species, the indicator Key species achieved good status in 25 out of 43 areas, and in 13 out of 21 evaluated assessment units assessed.

For the indicator Cyprinids/Mesopredators, the spatial coverage of data to support an evaluation is poorer. The indicator is evaluated in 27 areas, covering 16 out of 42 assessment units and 11

Table 3.
Overview of the status assessment based on the indicator Key species by monitoring areas/data sets and coastal assessment units. The columns give information on the key species used (perch or flounder), area/data set specific assessment approach (baseline or trend), initial status to determine the desired direction (Ref period status), threshold value for good status (when using a trend-based approach the + or − sign indicate the desired direction of the trend), current indicator value, status of the monitoring area, and status of the assessment unit. GS denotes good status and nGS not good status.

Sub-basin	Coastal area name (assessment unit)	Monitoring area/data set	Identity of key species	Assessment method	Ref. period status	Threshold value	Current value	Status monitoring location	Status coastal area
Bothnian Bay	Bothnian Bay Finnish Coastal waters	Finnish ICES SD 31	Perch	Baseline	GS	0.07	0.15	GS	GS
	Bothnian Bay Swedish Coastal waters	Råneå	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.55	GS	
		Kinnbäcksfjärden	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.53	GS	GS
The Quark	The Quark Finnish Coastal waters	Finnish ICES rect 23	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.0003 (+)	GS	
		Finnish ICES rect 28	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.66	GS	GS
	The Quark Swedish Coastal waters	Holmön	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.77	GS	
		Norrbyn	Perch	Trend	nGS	Slope p<0.1	P slope = 0.64	nGS	nGS
Bothnian Sea	Bothnian Sea Finnish Coastal waters	Finnish ICES SD 30	Perch	Baseline	GS	0.18	0.27	GS	GS
	Bothnian Sea Swedish Coastal waters	Gaviksfjärden	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.19	GS	
		Långvindsfjärden	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.12	GS	
		Forsmark	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.49	GS	
		Forsmark, long time-series	Perch	Baseline	GS	10.34	58.67	GS	GS
Åland Sea	Åland Sea Finnish Coastal waters	Lagnö	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.16	GS	GS
Archipelago Sea	Archipelago Sea Coastal waters	Finbo	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.78	GS	
		Kumlinge	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.68	GS	
		Finnish ICES SD 29	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.005 (+)	GS	GS
Northern Baltic Sea	Northern Baltic Proper Swedish Coastal waters	Askö	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.44	GS	
		Muskö	Flounder	Trend	GS	Slope p>0.1 (+)	P slope = 0.1	GS	GS
Gulf of Finland	Gulf of Finland Finnish Coastal waters	Finnish ICES SD 32	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.99	GS	GS
Gulf of Riga	Gulf of Riga Estonian Coastal waters	Hiiumaa	Perch	Baseline	nGS	33.17	32.78	nGS	nGS
	Gulf of Riga Latvian Coastal waters	Daugavgriva	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.67	GS	GS
Western Gotland Basin	Western Gotland Basin Swedish Coastal waters	Kvädöfjärden, perch	Perch	Trend	nGS	Slope p<0.1 (+)	P slope = 0.70	nGS	
		Kvädöfjärden, perch long time-series	Perch	Trend	nGS	Slope p<0.1 (+)	P slope = 0.07 (+)	GS	
		Kvädöfjärden, autumn	Flounder	Baseline	nGS	11.74	4.24	nGS	
		Vinö	Perch	Baseline	nGS	63.85	22.65	nGS	nGS
Estern Gotland Basin	Eastern Gotland Basin Latvian Coastal waters	Jurkalne	Flounder	Baseline	GS	6.22	25.95	GS	GS
	Eastern Gotland Basin Lithuanian Coastal waters	Mon/But	Flounder	Trend	GS	Slope p>0.1 (+)	P slope = 0.43	GS	
		Curonian lagoon	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.27	GS	GS
Bornholm Basin	Eastern Gotland Basin Swedish Coastal waters	Torhamn	Perch	Trend	GS	Slope p>0.1 (+)	P slope = 0.002 (+)	GS	GS
Arkona Basin	Arkona Basin Danish Coastal waters	Præstø Fiord	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.39	nGS	nGS
Mecklenburg Bight	Mecklenburg Bight Danish Coastal waters	Area south of Zealand (Smålandsfarvandet)	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.33	nGS	nGS
Belt Sea	Belts Danish Coastal waters	The Great Belt	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.13	nGS	
		Southern Little Belt and the archipelago	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.22	nGS	
		Odense Fiord	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.35	nGS	
		Sejerø Bay	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.27	nGS	
		Århus Bay	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.85	nGS	
		Fiords of Eastern Jutland	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.29	nGS	nGS
The Sound	The Sound Danish Coastal waters	The Sound	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.14	nGS	nGS
Kattegat	Kattegat Danish Coastal waters, including Limfjorden	Islefjord and Roskilde fjord	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.01 (+)	GS	
		Northern Kattegat	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.81	nGS	
		Northern Limfjord	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.27	nGS	
		Skive Fiord and Lovns Broad	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.48	nGS	
		Venø Bay and Nissum Broad	Flounder	Trend	nGS	Slope p<0.1 (+)	P slope = 0.53	nGS	nGS

out of 18 sub-basins (Figure 6, Table 4). When the indicator is represented by cyprinids, good status is obtained in 14 of the 27 areas assessed (Table 4). The areas represented by not good status are found in the Swedish part of the Quark, Finnish parts of the Bothnian Sea, the Åland Sea, Archipelago Sea, Gulf of Finland, Gulf of Riga and Eastern Baltic Sea (Figure 6; Table 4). When good status is not achieved, this is mostly due to high abundances of cyprinids. The indicator was assessed based on the species group of mesopredators is one area (Monciskes/Butinge in Lithuania), and here it indicates good status. In total, the indicator shows good status in 15 out of 27 assessed areas, and in seven out of 16 assessment units covered.

The Piscivore indicator is assessed with similar spatial coverage as the Cyprinids/Mesopredator indicator. It was evaluated in 29 areas, covering 16 out of 42 assessment units and 11 out of 18 sub-basins (Figure 6; Table 5). The indicator shows good status in all but four of the 29 assessed areas. Status according to the indicator Piscivores appears to be somewhat better in the northern compared to the southern Baltic Sea (Figure 6). Since perch is the most common piscivore in many of the data sets used, the results are the same as for the indicator Key species when this is based on perch. Scaling up the results to assessment unit level, good status is obtained in 13 out of 16 assessments units assessed.

Coastal fish monitoring in the northern Baltic Sea. © Jens Olsson

Table 4.

Overview of the status assessment based on the indicator Cyprinids/Mesopredators by monitoring areas/data sets and coastal assessment units. The columns give information on the identity of the indicator used (Cyprinids or Mesopredators), monitoring method, area specific assessment approach (baseline or trend), initial status to determine the desired direction (Ref period status), threshold value for good status (when using a trend-based approach the + or − sign indicate the desired direction of the trend), current indicator value, status of the monitoring area, and status of the assessment unit. GS denotes good status and nGS not good status.

Sub-basin	Coastal area name (assessment unit)	Monitoring area/data set	Identity of indicator	Monitoring method	Assessment method	Ref. period status	Threshold value	Current value	Status monitoring location	Status coastal area
Bothnian Bay	Bothnian Bay Finnish Coastal waters	Finnish ICES SD 31	Cyprinids	Commercial stats	Baseline	GS	0.092; 0.19	0.15	GS	GS
	Bothnian Bay Swedish Coastal waters	Råneå	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.52	GS	GS
		Kinnbäcksfjärden	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.17	GS	
The Quark	The Quark Finnish Coastal waters	Finnish ICES rect 23	Cyprinids	Commercial stats	Trend	nGS	Slope p <0.1 (-)	P slope = 0.20	nGS	nGS
		Finnish ICES rect 28	Cyprinids	Commercial stats	Trend	nGS	Slope p <0.1 (-)	P slope = 0.23	nGS	
	The Quark Swedish Coastal waters	Holmön	Cyprinids	Gill net	Trend	nGS	Slope p <0.1 (-)	P slope = 0.0008 (+)	nGS	nGS
		Norrbyn	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.16	GS	
Bothnian Sea	Bothnian Sea Finnish Coastal waters	Finnish ICES SD 30	Cyprinids	Commercial stats	Baseline	nGS	0.14	0.21	nGS	nGS
	Bothnian Sea Swedish Coastal waters	Gaviksfjärden	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.19	GS	GS
		Långvindsfjärden	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.95	GS	
		Forsmark	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.11	GS	
		Forsmark, long time-series	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.61	GS	
Åland Sea	Åland Sea Swedish Coastal waters	Lagnö	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.0014 (+)	nGS	nGS
Archipelago Sea	Archipelago Sea Coastal waters	Finbo	Cyprinids	Gill net	Trend	nGS	Slope p <0.1 (-)	P slope = 0.016 (+)	nGS	nGS
		Kumlinge	Cyprinids	Gill net	Trend	nGS	Slope p <0.1 (-)	P slope = 0.22	nGS	
		Finnish ICES SD 29	Cyprinids	Commercial stats	Baseline	nGS	0.10	0.22	nGS	
Northern Baltic Sea	Northern Baltic Proper Swedish Coastal waters	Askö	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.90	GS	GS
Gulf of Finland	Gulf of Finland Finnish Coastal waters	Finnish ICES SD 32	Cyprinids	Commercial stats	Trend	nGS	Slope p <0.1 (-)	P slope = 0.37	nGS	nGS
Gulf of Riga	Gulf of Riga Estonian Coastal waters	Hiiumaa	Cyprinids	Gill net	Trend	nGS	Slope p <0.1 (+)	P slope = 0.04 (-)	nGS	nGS
	Gulf of Riga Latvian Coastal waters	Daugavgriva	Cyprinids	Gill net	Trend	nGS	Slope p <0.1 (-)	P slope = 0.18	nGS	nGS
Western Gotland Basin	Western Gotland Basin Swedish Coastal waters	Kvädöfjärden	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.93	GS	GS
		Kvädöfjärden, long time-series	Cyprinids	Gill net	Baseline	GS	15.3; 53.67	43.60	GS	
		Vinö	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.91	GS	
Estern Gotland Basin	Eastern Gotland Basin Latvian Coastal waters	Jurkalne	Cyprinids	Gill net	Trend	nGS	Slope p <0.1 (-)	P slope = 0.51	nGS	nGS
	Eastern Gotland Basin Lithuanian Coastal waters	Mon/But	Mesopredators	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.75	GS	
		Curonian lagoon	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.98	GS	GS
Bornholm Basin	Bornholm Basin Swedish Coastal waters	Torhamn	Cyprinids	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.50	GS	GS

Table 5.

Overview of the status assessment based on the indicator Piscivores by monitoring areas/data sets and coastal assessment units. The columns give information on the species included in the indicator, monitoring method, area specific assessment approach (baseline or trend), initial status to determine the desired direction (Ref period status), threshold value for good status (when using a trend-based approach the + or − sign indicate the desired direction of the trend), current indicator value, status of the monitoring area, and status of the assessment unit. GS denotes good status and nGS not good status.

Sub-basin	Coastal area name (assessment unit)	Monitoring area/data set	Monitoring method	Assessment method	Ref. period status	Threshold value	Current value	Status monitoring location	Status coastal area
Bothnian Bay	Bothnian Bay Finnish Coastal waters	Finnish ICES SD 31	Commercial stats	Trend	GS	Slope p >0.1 (+)	P slope = 0.14	GS	GS
	Bothnian Bay Swedish Coastal waters	Råneå	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.56	GS	GS
		Kinnbäcksfjärden	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.53	GS	
The Quark	The Quark Finnish Coastal waters	Finnish ICES rect 23	Commercial stats	Trend	GS	Slope p >0.1 (+)	P slope = 0.0001 (+)	GS	GS
		Finnish ICES rect 28	Commercial stats	Baseline	GS	0.24	0.31	GS	
	The Quark Swedish Coastal waters	Holmön	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.56	GS	nGS
		Norrbyn	Gill net	Trend	nGS	Slope p <0.1 (+)	P slope = 0.63	nGS	
Bothnian Sea	Bothnian Sea Finnish Coastal waters	Finnish ICES SD 30	Commercial stats	Trend	nGS	Slope p <0.1 (+)	P slope = 0.01 (+)	GS	GS
	Bothnian Sea Swedish Coastal waters	Gaviksfjärden	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.19	GS	GS
		Långvindsfjärden	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.12	GS	
		Forsmark	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.49	GS	
		Forsmark, long time-series	Gill net	Baseline	GS	11.63	59.23	GS	
Åland Sea	Åland Sea Swedish Coastal waters	Lagnö	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.15	GS	GS
Archipelago Sea	Archipelago Sea Coastal waters	Finbo	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.83	GS	GS
		Kumlinge	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.68	GS	
		Finnish ICES SD 29	Commercial stats	Trend	nGS	Slope p <0.1 (+)	P slope = 0.01 (+)	GS	
Northern Baltic Sea	Northern Baltic Proper Swedish Coastal waters	Askö	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.39	GS	GS
		Muskö	Gill net	Baseline	GS	4.39	6.89	GS	
Gulf of Finland	Gulf of Finland Finnish Coastal waters	Finnish ICES SD 32	Commercial stats	Trend	nGS	Slope p >0.1 (+)	P slope = 0.2	GS	GS
Gulf of Riga	Gulf of Riga Estonian Coastal waters	Hiiumaa	Gill net	Baseline	nGS	33.37	32.92	nGS	nGS
	Gulf of Riga Latvian Coastal waters	Daugavgriva	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.54	GS	GS
Western Gotland Basin	Western Gotland Basin Swedish Coastal waters	Kvädöfjärden	Gill net	Trend	nGS	Slope p >0.1 (+)	P slope = 0.39	nGS	nGS
		Kvädöfjärden, long time-series	Gill net	Trend	nGS	Slope p >0.1 (+)	P slope = 0.03 (+)	GS	
		Kvädöfjärden, autumn	Gill net	Baseline	GS	6.74	6.31	nGS	
		Vinö	Gill net	Baseline	nGS	64.98	22.65	nGS	
Estern Gotland Basin	Eastern Gotland Basin Latvian Coastal waters	Jurkalne	Gill net	Baseline	GS	7.48	24.86	GS	GS
	Eastern Gotland Basin Lithuanian Coastal waters	Mon/But	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.54	GS	GS
		Curonian lagoon	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.34	GS	
Bornholm Basin	Bornholm Basin Swedish Coastal waters	Torhamn	Gill net	Trend	GS	Slope p >0.1 (+)	P slope = 0.003 (+)	GS	GS

3.2.2 Confidence in the assessment

Confidence in the status assessment for each assessment unit is based on evaluation of the length of the time-series, congruence in the observed status, and data precision (Table 6). The confidence is higher when status is represented by long time-series, when different status assessments within the same assessment unit indicate the same results and when data precision is good. In addition, the spatial representation of monitoring within an assessment unit is provided by the number of fishing stations (fisheries independent monitoring) or fishermen (recreational catch registration and commercial catch statistics) per assessment unit size. Since this information is difficult to standardize across monitoring data and methods hence to classify as low, moderate or high, the information is merely provided as descriptive information.

Confidence is assessed by evaluating each of these four criteria and assigning them scores at three levels (low, moderate and high; Table 6). The final confidence is the highest level that is met by all three criteria.

This approach is somewhat more detailed than the confidence assessment applied in the HELCOM HOLAS II assessment (HELCOM 2017c), in order to reflect the monitoring of coastal fish in a more nuanced way. Whereas both of the confidence scoring approaches use criteria related to temporal coverage, spatial representability and methodological confidence/data precision, the coastal fish approach, as presented in Table 6, also considers congruence in the observed status in different monitoring areas within the same assessment unit. The two approaches give overall similar confidence assessments, but the coastal fish approach presented here is slightly stricter.

According to the confidence assessment scoring table as outlined in table 6, the overall confidence of the different assessment units generally ranges from low to moderate for the different indicators and assessment units (Tables 7-9). The overall confidence score is considered as high in only one of the assessment units, Gulf of Riga Estonian Coastal waters. Here the time-series are longer than 15 years and the method used is fisheries independent gill-net monitoring. Since there is only one monitoring station in the area (Hiiumaa) the congruence criteria is also scores as high, but the spatial representation estimate (number of stations per assessment unit area) is very low (Tables 7-9; Figure 1). For the other assessment units, the Finnish data from the commercial fishery gave the highest scores with respect to time-series length and congruence in status assessment. In general, this data source also had the best spatial representation. Since the data are fisheries dependent with no quality routine control system, however, the scoring of the precision of data criterion can never be higher than moderate (for cyprinids even low since the species group is incidental catch in the fishery). As a contrast, data from national gill-net programs from fisheries independent surveys always scores high on the precision of data criterion, but instead often has poorer spatial representation, and lower scores on time-series length and often also for the congruence criteria.

Table 6.
Criteria for assigning confidence in the coastal fish assessment as presented in this report. For each criterion, confidence is assigned as low, moderate or high based on the descriptions presented in the cells. Overall confidence is determined as the highest level which is met by all four criteria.

Confidence level	Time series length	Congruence in status	Precision of data
Low	Below 10 years	High degree of departing status across areas (50% same status)	Poor (i.e. cyprinids in commercial catch data, recreational fishermen data)
Medium	10-15 years	Lower degree of departing status across areas (75% same status)	Medium (i.e. key species and piscivores in commercial catch data)
High	15 years or more	Similar status across areas (85% same status)	Good (i.e. fisheries independent data)

3.2.3 Other results

The Polish coastal fish monitoring program has been in place since 2011. In 2012, however, no monitoring was undertaken due to lack of funding. In general, the sampling stations were located in transitional waters and coastal areas along the Polish coast which are divided into 18 water bodies (nine transitional and nine coastal, Figure 23). All stations were fished simultaneously only in 2011. In the other years, monitoring covered from three to seven different water bodies.

The monitoring is conducted in the summer (July-August). In 2011, additional surveys were carried out in the autumn (October-November). Three net types were used, the Polish coastal survey net, the Polish coastal multi-mesh net, and the Nordic coastal multi-mesh gillnets (see HELCOM, 2015b for a detailed description). Polish coastal survey net and Polish coastal multi-mesh net were used until 2013, and Nordic coastal multi-mesh gillnets from 2014 an onward.

At each station a set of two, four or six nets were set for approximately 12 night hours. At each station and season, samples were collected two times per season (one night per fishing event). Due to harsh environmental conditions in the river mouths of the Vistula, Dziwna and Świna rivers, an

Table 7.
Scoring of confidence in the status assessment over the different assessment units covered for the Key species indicator. The confidence scoring is based on the criteria listed in table 5 above, and an overall summary of the scores across all three criteria (Summary) is given. L = low, M = moderate and H = high confidence scoring. NA = assessment unit not assessed. Given is also an estimation of the spatial representation of the monitoring within an assessment unit. "No stations/fisherman" represents how many stations (fisheries independent data) or fishermen (commercial catch data and recreational catch registration) there are within an assessment unit during the years 2011–2016. In the Archipelago Sea, there is a combination of fisheries independent data and commercial catch data. "Area" represents the area of the assessment unit in km2. In Danish areas, the number of fishermen has varied between years and hence a range is presented.

		Assessment criteria				
Coastal area name (assessment unit)	**Coastal area code**	**Time series length**	**Congruence**	**Precision**	**Summary**	**Spatial representation** (No stations or fishermen/area)
Bothnian Bay Finnish Coastal waters	1	H	H	M	M	0.053 (295/5551)
Bothnian Bay Swedish Coastal waters	2	M	H	H	M	0.17 (90/5352)
The Quark Finnish Coastal waters	3	H	H	M	M	0.49 (172/3482)
The Quark Swedish Coastal waters	4	M	L	H	L	0.40 (75/1849)
Bothnian Sea Finnish Coastal waters	5	H	H	M	M	0.11 (473/4269)
Bothnian Sea Swedish Coastal waters	6	M	H	H	M	0.023 (143/6254)
Åland Sea Finnish Coastal waters	7	NA	NA	NA	NA	NA
Åland Sea Swedish Coastal waters	8	M	H	H	M	0.027 (45/1692)
Archipelago Sea Coastal waters	9	H	H	M	M	0.020 (219/10686)
Northern Baltic Proper Finnish Coastal waters	10	NA	NA	NA	NA	NA
Northern Baltic Proper Swedish Coastal waters	11	M	H	H	M	0.011 (53/4892)
Northern Baltic Proper Estonian Coastal waters	12	NA	NA	NA	NA	NA
Gulf of Finland Finnish Coastal waters	13	H	H	M	M	0.030 (174/5767)
Gulf of Finland Estonian Coastal waters	14	NA	NA	NA	NA	NA
Gulf of Finland Russian Coastal waters	15	NA	NA	NA	NA	NA
Gulf of Riga Estonian Coastal waters	16	H	H	H	H	0.0014 (12/8366)
Gulf of Riga Latvian Coastal waters	17	M	H	H	M	0.0034 (6/1744)
Western Gotland Basin Swedish Coastal waters	18	H	M	H	M	0.014 (81/5756)
Eastern Gotland Basin Estonian Coastal waters	19	NA	NA	NA	NA	NA
Eastern Gotland Basin Latvian Coastal waters	20	M	H	H	M	0.012 (6/550)
Eastern Gotland Basin Lithuanian Coastal waters	21	M	H	H	M	0.0064 (4/550)
Eastern Gotland Basin Swedish Coastal waters	22	NA	NA	NA	NA	NA
Eastern Gotland Basin Russian Coastal waters	23	NA	NA	NA	NA	NA
Eastern Gotland Basin Polish Coastal waters	24	NA	NA	NA	NA	NA
Gdansk Basin Russian Coastal waters	25	NA	NA	NA	NA	NA
Gdansk Basin Polish Coastal waters	26	NA	NA	NA	NA	NA
Bornholm Basin Swedish Coastal waters	27	M	H	H	M	0.028 (45/1614)
Bornholm Basin Polish Coastal waters	28	NA	NA	NA	NA	NA
Bornholm Basin Danish Coastal waters	29	NA	NA	NA	NA	NA
Bornholm Basin German Coastal waters	30	NA	NA	NA	NA	NA
Arkona Basin Swedish Coastal waters	31	NA	NA	NA	NA	NA
Arkona Basin Danish Coastal waters	32	L	H	M	L	0.00058 (1/1741)
Arkona Basin German Coastal waters	33	NA	NA	NA	NA	NA
Mecklenburg Bight German Coastal waters	34	NA	NA	NA	NA	NA
Mecklenburg Bight Danish Coastal waters	35	L	H	M	L	0.0099 (1 to 3/302)
Kiel Bight Danish Coastal waters	36	NA	NA	NA	NA	NA
Kiel Bight German Coastal waters	37	NA	NA	NA	NA	NA
Belts Danish Coastal waters	38	L	H	M	L	0.0035 (22 to 31/8816)
The Sound Swedish Coastal waters	39	NA	NA	NA	NA	NA
The Sound Danish Coastal waters	40	L	H	M	L	0.0060 (1 to 2/332)
Kattegat Swedish Coastal waters	41	NA	NA	NA	NA	NA
Kattegat Danish Coastal waters, including Limfjorden	42	L	M	M	L	0.0050 (15 to 29/5797)

Table 8.
Scoring of confidence in the status assessment over the different assessment units covered for the Cyprinids/mesopredator indicator. The confidence scoring is based on the criteria listed in table 5 above, and an overall summary of the scores across all three criteria (Summary) is given. L = low, M = moderate and H = high confidence scoring. NA = assessment unit not assessed. Given is also an estimation of the spatial representation of the monitoring within an assessment unit. "No stations/fisherman" represents how many stations (fisheries independent data) or fishermen (commercial catch data) there are within an assessment unit during the years 2011–2016. In the Archipelago Sea, there is a combination of fisheries independent data and commercial catch data. "Area" represents the area of the assessment unit in km2.

		Assessment criteria				
Coastal area name (assessment unit)	**Coastal area code**	**Time series length**	**Congruence**	**Precision**	**Summary**	**Spatial representation** (No stations or fishermen/area)
Bothnian Bay Finnish Coastal waters	1	H	H	L	L	0.053 (295/5551)
Bothnian Bay Swedish Coastal waters	2	M	H	H	M	0.17 (90/5352)
The Quark Finnish Coastal waters	3	H	H	L	L	0.49 (172/3482)
The Quark Swedish Coastal waters	4	M	L	H	L	0.40 (75/1849)
Bothnian Sea Finnish Coastal waters	5	H	H	L	L	0.11 (473/4269)
Bothnian Sea Swedish Coastal waters	6	M	H	H	M	0.023 (143/6254)
Åland Sea Finnish Coastal waters	7	NA	NA	NA	NA	NA
Åland Sea Swedish Coastal waters	8	M	H	H	L	0.027 (45/1692)
Archipelago Sea Coastal waters	9	H	H	M	M	0.020 (219/10686)
Northern Baltic Proper Finnish Coastal waters	10	NA	NA	NA	NA	NA
Northern Baltic Proper Swedish Coastal waters	11	M	H	H	L	0.009 (45/4892)
Northern Baltic Proper Estonian Coastal waters	12	NA	NA	NA	NA	NA
Gulf of Finland Finnish Coastal waters	13	H	H	L	L	0.030 (174/5767)
Gulf of Finland Estonian Coastal waters	14	NA	NA	NA	NA	NA
Gulf of Finland Russian Coastal waters	15	NA	NA	NA	NA	NA
Gulf of Riga Estonian Coastal waters	16	H	H	H	L	0.0014 (12/8366)
Gulf of Riga Latvian Coastal waters	17	M	H	H	L	0.0034 (6/1744)
Western Gotland Basin Swedish Coastal waters	18	H	H	H	M	0.012 (69/5756)
Eastern Gotland Basin Estonian Coastal waters	19	NA	NA	NA	NA	NA
Eastern Gotland Basin Latvian Coastal waters	20	M	H	H	L	0.012 (6/550)
Eastern Gotland Basin Lithuanian Coastal waters	21	M	H	H	M	0.0064 (4/550)
Eastern Gotland Basin Swedish Coastal waters	22	NA	NA	NA	NA	NA
Eastern Gotland Basin Russian Coastal waters	23	NA	NA	NA	NA	NA
Eastern Gotland Basin Polish Coastal waters	24	NA	NA	NA	NA	NA
Gdansk Basin Russian Coastal waters	25	NA	NA	NA	NA	NA
Gdansk Basin Polish Coastal waters	26	NA	NA	NA	NA	NA
Bornholm Basin Swedish Coastal waters	27	M	H	H	L	0.028 (45/1614)
Bornholm Basin Polish Coastal waters	28	NA	NA	NA	NA	NA
Bornholm Basin Danish Coastal waters	29	NA	NA	NA	NA	NA
Bornholm Basin German Coastal waters	30	NA	NA	NA	NA	NA
Arkona Basin Swedish Coastal waters	31	NA	NA	NA	NA	NA
Arkona Basin Danish Coastal waters	32	NA	NA	NA	NA	NA
Arkona Basin German Coastal waters	33	NA	NA	NA	NA	NA
Mecklenburg Bight German Coastal waters	34	NA	NA	NA	NA	NA
Mecklenburg Bight Danish Coastal waters	35	NA	NA	NA	NA	NA
Kiel Bight Danish Coastal waters	36	NA	NA	NA	NA	NA
Kiel Bight German Coastal waters	37	NA	NA	NA	NA	NA
Belts Danish Coastal waters	38	NA	NA	NA	NA	NA
The Sound Swedish Coastal waters	39	NA	NA	NA	NA	NA
The Sound Danish Coastal waters	40	NA	NA	NA	NA	NA
Kattegat Swedish Coastal waters	41	NA	NA	NA	NA	NA
Kattegat Danish Coastal waters, including Limfjorden	42	NA	NA	NA	NA	NA

Table 9.
Scoring of confidence in the status assessment over the different assessment units covered for the Piscivore indicator. The confidence scoring is based on the criteria listed in table 5 above, and an overall summary of the scores across all three criteria (Summary) is given. L = low, M = moderate and H = high confidence scoring. NA = assessment unit not assessed. Given is also an estimation of the spatial representation of the monitoring within an assessment unit. "No stations/fisherman" represents how many stations (fisheries independent data) or fishermen (commercial catch data) there are within an assessment unit during the years 2011-2016. In the Archipelago Sea, there is a combination of fisheries independent data and commercial catch data. "Area" represents the area of the assessment unit in km2.

		Assessment criteria				
Coastal area name (assessment unit)	**Coastal area code**	**Time series length**	**Congruence**	**Precision**	**Summary**	**Spatial representation** (No stations or fishermen/area)
Bothnian Bay Finnish Coastal waters	1	H	H	M	M	0.053 (295/5551)
Bothnian Bay Swedish Coastal waters	2	M	H	H	M	0.17 (90/5352)
The Quark Finnish Coastal waters	3	H	H	M	M	0.49 (172/3482)
The Quark Swedish Coastal waters	4	M	L	H	L	0.40 (75/1849)
Bothnian Sea Finnish Coastal waters	5	H	H	M	M	0.11 (473/4269)
Bothnian Sea Swedish Coastal waters	6	M	H	H	M	0.023 (143/6254)
Åland Sea Finnish Coastal waters	7	NA	NA	NA	NA	NA
Åland Sea Swedish Coastal waters	8	M	H	H	L	0.027 (45/1692)
Archipelago Sea Coastal waters	9	H	H	M	M	0.020 (219/10686)
Northern Baltic Proper Finnish Coastal waters	10	NA	NA	NA	NA	NA
Northern Baltic Proper Swedish Coastal waters	11	M	H	H	M	0.011 (53/4892)
Northern Baltic Proper Estonian Coastal waters	12	NA	NA	NA	NA	NA
Gulf of Finland Finnish Coastal waters	13	H	H	M	M	0.030 (174/5767)
Gulf of Finland Estonian Coastal waters	14	NA	NA	NA	NA	NA
Gulf of Finland Russian Coastal waters	15	NA	NA	NA	NA	NA
Gulf of Riga Estonian Coastal waters	16	H	H	H	L	0.0014 (12/8366)
Gulf of Riga Latvian Coastal waters	17	M	H	H	L	0.0034 (6/1744)
Western Gotland Basin Swedish Coastal waters	18	H	M	H	L	0.014 (81/5756)
Eastern Gotland Basin Estonian Coastal waters	19	NA	NA	NA	NA	NA
Eastern Gotland Basin Latvian Coastal waters	20	M	H	H	L	0.012 (6/550)
Eastern Gotland Basin Lithuanian Coastal waters	21	M	H	H	M	0.0064 (4/550)
Eastern Gotland Basin Swedish Coastal waters	22	NA	NA	NA	NA	NA
Eastern Gotland Basin Russian Coastal waters	23	NA	NA	NA	NA	NA
Eastern Gotland Basin Polish Coastal waters	24	NA	NA	NA	NA	NA
Gdansk Basin Russian Coastal waters	25	NA	NA	NA	NA	NA
Gdansk Basin Polish Coastal waters	26	NA	NA	NA	NA	NA
Bornholm Basin Swedish Coastal waters	27	M	H	H	L	0.028 (45/1614)
Bornholm Basin Polish Coastal waters	28	NA	NA	NA	NA	NA
Bornholm Basin Danish Coastal waters	29	NA	NA	NA	NA	NA
Bornholm Basin German Coastal waters	30	M	H	H	M	NA
Arkona Basin Swedish Coastal waters	31	NA	NA	NA	NA	NA
Arkona Basin Danish Coastal waters	32	NA	NA	NA	NA	NA
Arkona Basin German Coastal waters	33	NA	NA	NA	NA	NA
Mecklenburg Bight German Coastal waters	34	NA	NA	NA	NA	NA
Mecklenburg Bight Danish Coastal waters	35	NA	NA	NA	NA	NA
Kiel Bight Danish Coastal waters	36	NA	NA	NA	NA	NA
Kiel Bight German Coastal waters	37	NA	NA	NA	NA	NA
Belts Danish Coastal waters	38	NA	NA	NA	NA	NA
The Sound Swedish Coastal waters	39	NA	NA	NA	NA	NA
The Sound Danish Coastal waters	40	NA	NA	NA	NA	NA
Kattegat Swedish Coastal waters	41	NA	NA	NA	NA	NA
Kattegat Danish Coastal waters, including Limfjorden	42	NA	NA	NA	NA	NA

additional survey was carried out using a bottom trawl with a standardized 10 mm mesh in the cod end and trawling duration from 15 to 30 minutes with at approximately 3.0 knots of haul speed.

All fish caught were identified to species, counted and measured for their total length (nearest cm). For three specimens for each centimeter length-class, individual body weight, sex, maturity stage and age were also recorded. Catch data were expressed as the catch per night and net, and catch per one hour of trawling, respectively. The monitoring also included the following environmental parameters: position, bottom type, any signs of disturbance (e.g. debris, physical damage), water depth, water temperature, wind direction, salinity and Secchi depth.

The fish community composition differed between the different water bodies (Figure 24). In general, however, there have been no major changes over the years studied with the respect to fish community structure and composition within the water bodies monitored. Flounder and herring were dominating the catches in the stations located at the open Polish coast. These stations include the water bodies Hel Peninsula, Outer Puck Bay, Vistula Spit, Władysławowo-Jastrzębia Góra and Eastern and Western Rowy-Jarosławiec (Figure 23, 24). Perch and pikeperch were also common in certain areas, and perch dominated the catches in the water bodies, Świna Mouth, Szczecin Lagoon, Dziwna-Sarbinowo, Dziwna-Świna (Figure 23, 24). Round goby mainly dominated the stations located in Puck Lagoon (Figure 23, 24) but also occurred in the catches along the open Polish coast (Smoliński and Całkiewicz, 2015). For a full list of species in the monitoring see Appendix 2.

With regards to the common coastal fish indicators as described above, the following results are apparent for the Polish coast. For the Key species

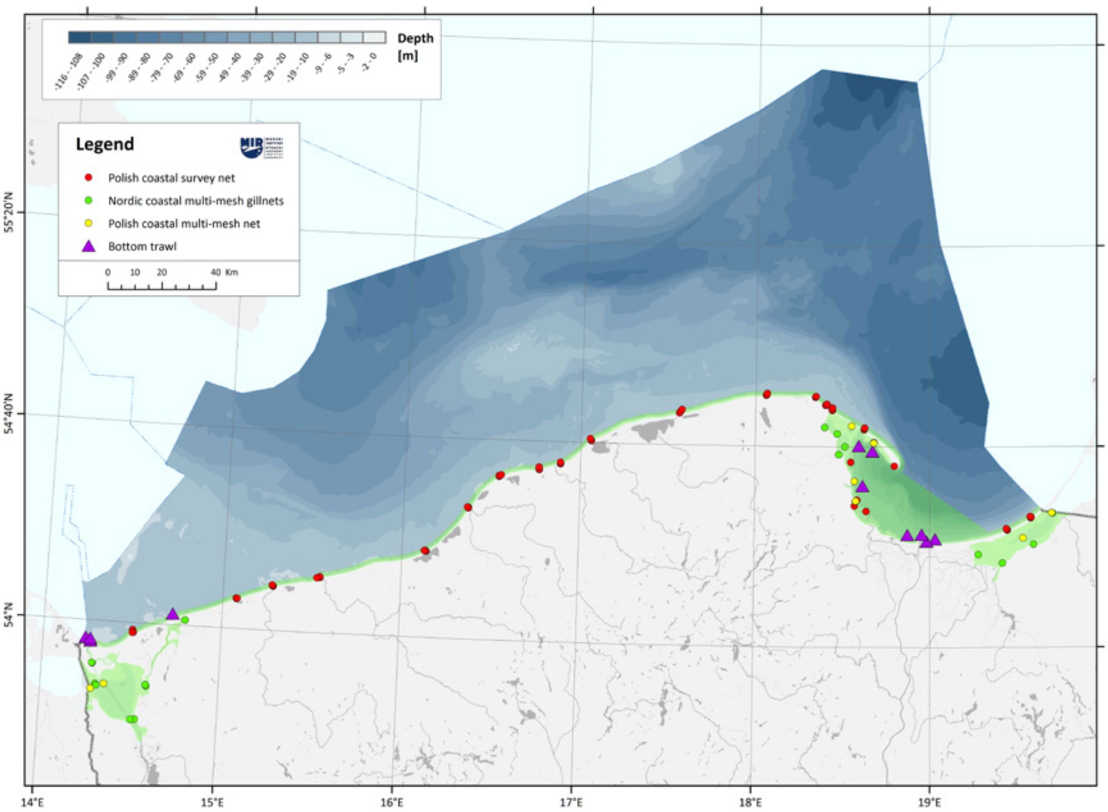

Figure 23.
Map of locations of Polish coastal fish monitoring stations (years 2011–2016). The water bodies are numbered according to the following: 1: Szczecin Lagoon; 2: Swina Mouth; 3: Dziwna – Swina; 4: Dziwna Mouth; 5: Kamienski Lagoon; 6: Dziwna – Sarbinowo; 7: Jaroslawiec – Sarbinowo; 8: Rowy – Jaroslawiec W; 9: Rowy – Jaroslawiec E; 10: Jastrzebia Gora – Rowy; 11: Wladyslawowo – Jastrzebia Gora; 12: Puck Lagoon; 13: Outer Puck Bay; 14: Hel Peninsula; 15: Inner Gulf of Gdansk; 16: Vistula Mouth; 17: Vistula Spit; 18: Vistula Lagoon.

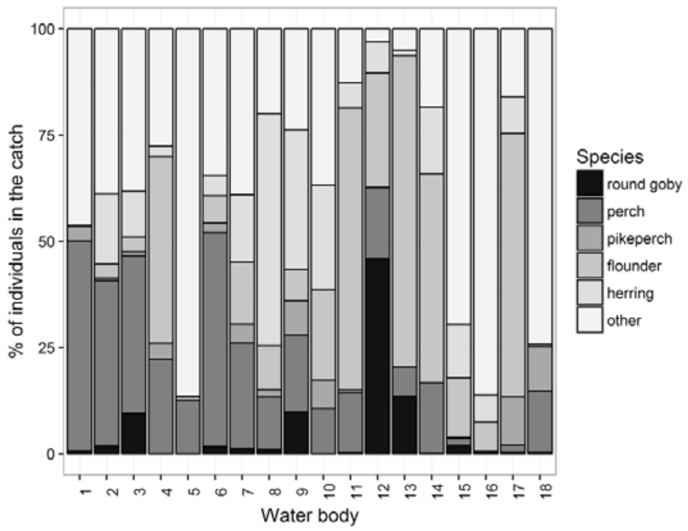

Figure 24.
Share (%) of the five most abundant fish species (round goby, perch, pikeperch, flounder and herring) in Polish monitoring catches during the summer season of 2011–2016 in particular water bodies. List of species classified as "other" is given in the Appendix 2. The water bodies are numbered according to the following: 1: Szczecin Lagoon; 2: Swina Mouth; 3: Dziwna – Swina; 4: Dziwna Mouth; 5: Kamienski Lagoon; 6: Dziwna – Sarbinowo; 7: Jaroslawiec – Sarbinowo; 8: Rowy – Jaroslawiec W; 9: Rowy – Jaroslawiec E; 10: Jastrzebia Gora – Rowy; 11: Wladyslawowo – Jastrzebia Gora; 12: Puck Lagoon; 13: Outer Puck Bay; 14: Hel Peninsula; 15: Inner Gulf of Gdansk; 16: Vistula Mouth; 17: Vistula Spit; 18: Vistula Lagoon.

indicator represented by perch, the species is distributed along the whole Polish coastline. A concentration is observed in the western parts of the Polish marine waters (Szczecin Lagoon, Dziwna-Świna, Kamieński Lagoon and Dziwna-Sarbinowo), as well as in the eastern parts (Hel Peninsula, Outer Puck Bay, Gulf of Gdańsk and Vistula Lagoon; Figure 25). For the Key species indicator represented by flounder, the highest abundance is found in the eastern parts of Polish coast, especially in Władysławowo-Jastrzębia Góra, Hel Peninsula, Outer Puck Bay, Gulf of Gdańsk and Vistula Spit (Figure 25). The functional group indicator Piscivores (dominated by perch, pikeperch and cod) occurred in higher abundances in the Szczecin Lagoon, but also in Hel Peninsula, Władysławowo-Jastrzębia Góra, Gulf of Gdańsk, Outer Puck Lagoon, and Sarbinowo-Dziwna and Dziwna-Świna water bodies (Figure 25). The highest abundances of the functional group indicator Cyprinids are concentrated in the Kamieński Lagoon, Szczecin Lagoon and Vistula Lagoon (Figure 25). A likely explanation for these findings is elevated nutrient levels and a lower salinity in the water bodies, conditions that favor cyprinid fish (Bergström *et al.* 2016b; Smoliński and Całkiewicz 2015). To date, no assessment of mesopredatory fish has been carried out in the Polish coastal waters.

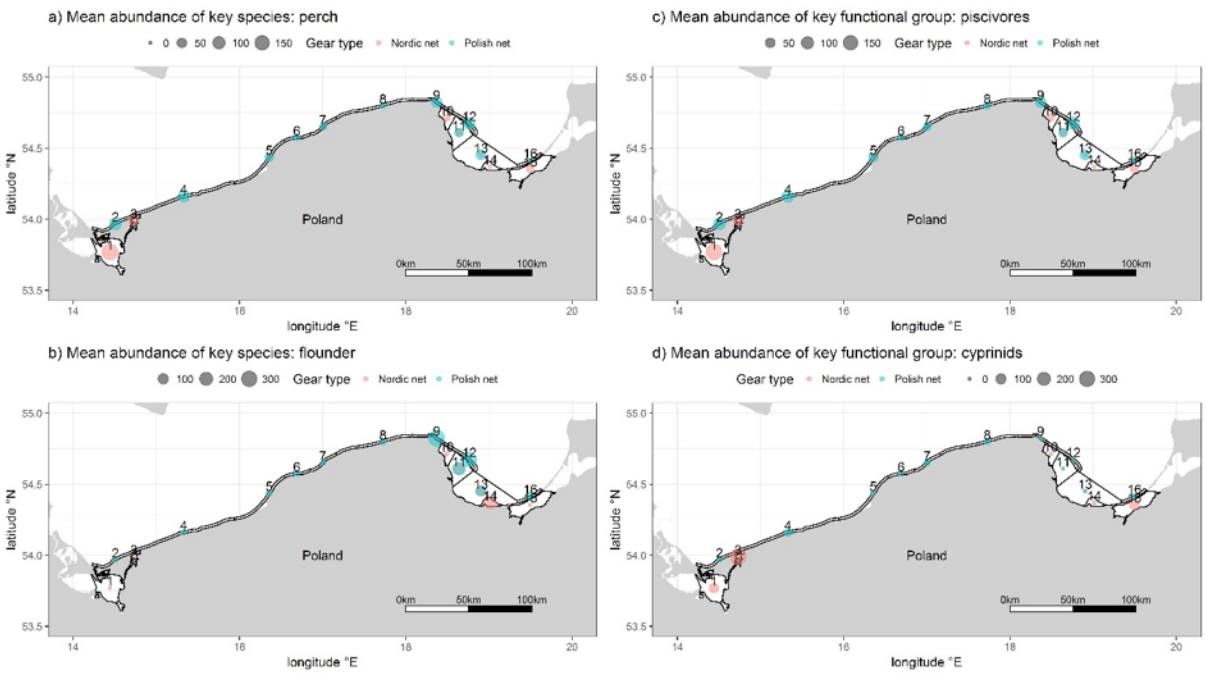

Figure 25.
Mean abundance over monitored years per coastal water body in Poland for the commonly agreed coastal fish indicators. The water bodies are numbered according to the following: 1: Szczecin Lagoon; 2: Dziwna – Swina; 3: Kamienski Lagoon; 4: Dziwna – Sarbinowo; 5: Jaroslawiec – Sarbinowo; 6: Rowy – Jaroslawiec W; 7: Rowy – Jaroslawiec E; 8: Jastrzebia Gora – Rowy; 9: Wladyslawowo – Jastrzebia Gora; 10: Puck Lagoon; 11: Outer Puck Bay; 12: Hel Peninsula; 13: Zatoka_Gda□ska_Wewn□trzna; 14: Vistula Mouth; 15: Vistula Lagoon; 16: Vistula Spit.

4. Measures for coastal fish

Given that there is a multitude of potentially impacting factors regulating coastal fish community development (chapter 2), it is not possible to identify a generic measure for restoring coastal fish communities in the Baltic Sea. Rather, the recommended recipe likely differs from case to case and should be identified accounting for the specific environmental setting and structure of the fish community in focus.

In table 10 we list potential measures, their links to pressures and the scientific support for the effectiveness of the measure for fish in the Baltic Sea. Based on this review, we present in detail the measures that are potentially suitable for restoring/protecting coastal fish communities. This part contains only measures that have been observed by scientific evaluation to have positive effects within current management structures in the Baltic Sea. For example, since climate change is not manageable in the shorter time frame in the Baltic Sea, measures to combat climate change are not included. Also, supporting and regulatory properties of the ecosystem, such as the regulation of species by natural predation, are not included as they are considered as a natural part of the ecosystem. Although not included as direct measures, however, both mitigation of climate change and, for example, natural predation, are expected to influence the status of coastal fish communities, and should be considered from an ecosystem perspective when identifying suitable measure for restoring fish communities.

The measures are subdivided into those aiming at reducing the mortality of the fish and those supporting the production of the fish.

Table 10.
Table showing potential measures for coastal fish in the Baltic Sea divided by the major aim of the measure (reducing mortality or supporting productivity). Provided is the name of the measure, which pressure the measure are targeting and if there are scientific support for the effectiveness of the measure in the Baltic Sea, X = no and Y = yes. For the measures with a Y, the scientific support is described further down in the text.

Aim of measure	Measure name	Link to major pressures	Scientific support for effectiveness for fish in the Baltic Sea
Reducing mortality	Permanent fisheries closures (no-take areas)	Fishing	Yes (see below)
	Partial fisheries closures	Fishing	Yes (see below)
	Regulation of fishing gears and catch	Fishing	Yes (see below)
Supporting productivity	Stocking of young fish	Fishing	No
	Nutrient reduction	Eutrophication	No
	Habitat protection	Physical exploitation	Yes (see below)
	Habitat restoration	Physical exploitation, Eutrophication	Yes (see below)
	Reduction of hazardous substances	Input of hazardous substances	No
	Biomanipultion (extraction of for example Cyprinid fish)	Fishing, Eutrophication	No

4.1. Measures reducing mortality

This section includes measures aimed at regulating the mortality of fish. There are two sources of mortality for natural fish stocks, mortality as a result of fishing and natural mortality from for example apex or top predators in the system. As discussed above, we only present measures targeting fisheries and fishing here. Moreover, since the fishery on typically coastal fish species in the Baltic Sea like perch, pikeperch, pike, whitefish and cyprinid species is currently not regulated by catch quotas, other options for measures needs to be considered. The majority of fishing methods, targeting coastal as well as off-shore fish communities and stocks, aim at large size individuals and species in the top of the food-web, hence leading to changes the fish size composition and fish community function (Pauly *et al.* 1998). As such, the part of the coastal fish community described by the indicators Key species and Piscivores are in focus for the measures regulating fishing mortality as presented in this report. Cyprinid and mesopredatory fish as represented by the Cyprinids/Mesopredator indicator for coastal fish are commonly not targeted directly by fisheries in the majority of the Baltic countries and are hence not directly impacted by fisheries related measures. An indirect effect on lower trophic level fish as cyprinids and mesopredators from fisheries regulations is, however, likely as a result of cascading effects in the food web (Casini *et al.* 2008, 2012; Eriksson *et al.* 2011; Östman *et al.* 2016). Measures aiming at regulating fishing mortality with scientifically documented effectiveness for coastal fish in the Baltic Sea, includes permanent fisheries closures, partial fisheries closures, as well as gear and catch regulations. Below we describe these measures in detail with respect to expected effects of the measure as well as general and Baltic specific evidence for the effectiveness of the measure.

4.1.1 Permanent fisheries closures (no take areas)

No-take marine reserves, where no harvesting is allowed, have been recommended as a general tool for an ecosystem approach to fisheries management (Halpern 2003; Halpern *et al.* 2010). Here, fishing mortality is regulated by permanent cessation of fishing activity in a particular area. By preventing fishing, this measure can potentially result in a more balanced size-structure of the fish community and higher prevalence of larger individuals and larger species. In other words, fish populations and communities within the boundaries of the closed areas will get an opportunity to recover from fisheries exploitation with respect to their abundance and size structure.

Indirect effects of a fishing closure might also include spill-over effects of adult fish, pelagic eggs and larvae to adjacent areas and systems (Abesamis and Russ 2005, Halpern *et al.* 2010), and also general and positive ecosystem effects on other parts of the food-web besides the targeted fish populations (Thrush och Dayton 2010, Baskett och Barnett, 2015; Bergström *et al.* 2016c). These effects might, however, often be slow since fish populations in marine reserve can have slower growth rates as a result of increased density dependence (Gårdmark *et al.* 2006).

There is evidence for a positive effect of no take areas in marine ecosystems, regardless of their size (Halpern 2003). No take areas might lead to increases in biomass, density, individual size, and diversity in all functional groups of the targeted fish community (Halpern 2003; Halpern *et al.* 2010). European marine reserves have been shown to promote key biological functions and variables as species richness, biomass, density, and body size of targeted populations (Fenberg *et al.* 2012).

There are only a few examples of effects of no take areas for fisheries currently in the Baltic Sea (Edgren 2005; Bergström *et al.* 2016c; Florin *et al.* 2013). Available studies on the effects of these in Swedish coastal waters suggest a higher density and older individuals in a reserve targeting flounder and turbot (Bergström *et al.* 2007, 2016c; Florin *et al.* 2013), increased abundance and individual size of pike and perch in a no take area compared to a reference area (Edgren 2005; Bergström *et al.* 2007, 2016c), and increased abundance of whitefish in no take area compared to a reference area (Bergström *et al.* 2016c).

4.1.2 Partial fisheries closures

This measure concerns closing of an area from fishing during a specific time or season in order to reduce the mortality of exploited species and stocks. The closing time usually target vulnerable life stages as the reproduction season and/or sensitive juvenile stages of the targeted population. The key objective of this measure is to increase the egg and larvae production, to protect juveniles from overexploitation and to reduce the risk of potential genetic selective effects of fishing. To that end, the main objective of this measure is similar to that of no take areas with the only difference that partial closures might be easier to advocate for fisheries managers.

Seasonal closures have been considered as beneficial mostly for restoring commercial shellfish (e.g., shrimp, lobster fisheries; reviewed by Everson 1986). However, recent studies have also demonstrated positive effects of partial closures on fish populations (Gwinn & Allen 2010; Samy-Kamal *et al.* 2015). In the Baltic Sea, there is not much evidence for positive effects of temporal closures for coastal fish, but for open sea populations, a spawning time closure targeting the western Baltic cod stock have proven to be successful (ICES 2017).

4.1.3 Regulations on fishing gears and catch

These types of measures aim at reducing the mortality of targeted fish populations and communities by limitation of the number and type of gears and vessels in the fishery, as well as restrictions in fishing licences and total allowable catch. Besides this, the measures in this section might also include mesh size restrictions of the gears used and minimum and/or maximum size limits of the catchable size of the fish in that

only a sub-section of the exploited populations and communities are targeted.

A reduction in the effort (number of gears and vessels allowed, and licences permitted) of a fishery can have a positive effect on targeted stocks and species by a reduction in mortality (e.g. Roberts and Polunin 1991). This might result in long-term sustainable out-take from the fishery and maintain the spawning stock biomass of targeted populations at an appropriate level. The type of gear used typically impact both target and non-target species. Overharvest of large and piscivorous fish might result in un-wanted alternations of the size structure and species composition in the food web (Pauly *et al.* 1998), but might also result in over-harvest of immature individuals that have not yet spawned. Non-target species are mainly regarded as incidental catch and is often discarded back to the sea, which in turn can affect the trophic structure of the recipient ecosystem (e.g., increased abundance of scavengers; Gislason 2002). A fishery might in addition negatively impact non-targeted species and populations if the incidental catch is substantial. By altering the size- and species selectivity of the gears used in the fishery, the negative effects on targeted and non-targeted fish populations and communities might be reduced.

Several measures of the types discussed in this section are already in place for coastal fish in the Baltic Sea (see HELCOM 2015a), but to date there are to the best of our knowledge no studies showing the effects of the measures on recipient fish populations and communities in the Baltic Sea. According to a bio-economic simulation model by Heikinheimo *et al.* 2006, however, mesh size regulations was suggested to have a positive effect on the biological sustainability of the pikeperch fishery in the Archipelago Sea, Finland. The model indicated that a larger mesh size would double the spawning stock biomass of pikeperch, which in turn would benefit the fishery in the long term (Heikinheimo *et al.* 2006).

4.2. Measures supporting productivity

Whereas the measures listed above mainly are targeting the adult life stage of the populations and communities, the ones listed in this section are generally focused on safeguarding or boosting the production of early life-stages of fish. Recent studies in the Baltic Sea have suggested that the availability and quality of essential habitats are of substantial importance for coastal fish (Sundblad *et al.* 2014; Bergström *et al.* 2016c; Kraufvelin *et al.* 2018). As a contrast to the measures aimed at regulating mortality of coastal fish above, the ones presented here target all three indicators for coastal fish as included in this report.

4.2.1 Habitat protection

The first and most important measure in this category focus on protection of already functional and essential habitats for coastal fish. In this respect it should be noted that it is always more cost-effective to protect and minimize impacts than to restore an essential habitat (Kraufvelin *et al.* 2018). The idea behind this measure is to prevent habitat degradation that negatively impact recruitment and production of juvenile fish. By safeguarding recruitment and production of juvenile fish, yields of adult populations of fish might be sustained (Sundblad *et al.* 2014; Kraufvelin *et al.* 2018). To maximise the effect of this type of measure, it should be combined with fisheries regulations as discussed above (Bergström *et al.* 2016c).

The measure includes the protection of habitats from various impacts as physical exploitation via coastal constructions and infrastructure as boating traffic, eutrophication, dredging and destructive fishing methods. It could also include protection from dam constructions in river mouths and up-stream brooks and rivers.

Although there is no direct evidence from the Baltic Sea of positive effects on coastal fish from habitat protection, substantial indirect evidence for the support of the measure is available (Kraufvelin *et al.* 2018). Sundblad *et al.* (2014) showed that habitat limitation in early life stages of perch and pikeperch may restrict the abundance of later adult stage fish. In addition, from Sweden there is evidence of long-term negative effects of coastal development on fish reproduction habitats (Sundblad and Bergström 2014), and of negative impacts on the habitat and hence production of juvenile fish from recreational boating traffic (Sandström *et al.* 2005). Moreover, in Denmark the extraction of large boulders (i.e. “stone-fishing”) from coastal reefs for construction of harbours and coastal protection in Kattegat have destroyed many cavernous reefs and modified macroalgal coverage in the area, which in turn have led to degradation of the habitat for local fish populations (Støttrup *et al.* 2014; Kristensen *et al.* 2015).

4.2.2 Habitat restoration

An alternative and often complementary measure to that of habitat protection is to restore already impacted and partly destroyed habitats for fish. The main objective of this measure is to restore degraded habitats affected by physical interferences to a state where they can support biodiversity and productivity of fish populations.

Habitat restoration can either be undertaken by re-creating the physical structure of the habitats, or by compensatory efforts by constructing new and artificial habitats (Loughlin & Clarke 2014). Some examples of habitat restoration along the Baltic Sea coast includes construction of artificial stone reefs (Støttrup *et al.* 2014; Kristensen *et al.* 2015; Stenberg *et al.* 2015), restoration of eelgrass meadows (Moksnes *et al.* 2016), and restoring wetlands and tributaries as reproduction habitats for coastal anadromous fish species as pike, ide and turbot (Nilsson *et al.* 2014).

In Denmark, building of artificial stone reefs and mussel beds has attracted fish species with a preference for rocky habitats, increased biodiversity and the abundance of larger specimens of certain species of fish (Støttrup *et al.* 2014; Kristensen *et al.* 2015; Stenberg *et al.* 2015). Biogenic reefs of mussels might also increase the structural complexity and biodiversity of the habitat and associated fauna, something that in turn might lead to an increase in fish growth and diversity. Whether or not the above-mentioned observations is the result of pure attraction effects of the fish or effects also at the population abundance level, is to date not established.

Eelgrass meadows are of substantial importance for the production of juvenile fish in marine habitats (Lilley *et al.* 2014; Cole and Moksnes 2016), but to date a substantial proportion of these important habitats has disappeared along the Baltic coasts

(Baden *et al.* 2003; Frederiksen *et al.* 2004). Despite the uncertain success of eelgrass meadow restoration attempts and the resulting effects on fish production to date, eelgrass meadow restoration might be an important measure to consider in the future when more evidence is accumulated.

Many coastal fish species of a freshwater origin in the coastal zones of the Baltic Sea undertake spawning migrations to coastal tributaries and wetlands (Engstedt *et al.* 2010; Nilsson *et al.* 2014; Rohtla *et al.* 2012, 2014, 2015). The quality of these habitats has undergone substantial deterioration during past decennia in many regions of the Baltic Sea (Engstedt *et al.* 2010; Nilsson *et al.* 2014). Recent efforts to restore these wetlands as reproduction areas for foremost pike have proven to result in a drastic increase in the production of juvenile pike as a result of optimal spawning conditions, predation refuge and food production (Nilsson *et al.* 2014). The resulting effects on the adult populations of pike are, however, not yet well established (but see Fredriksson *et al.* 2013).

4.3. Summary

Based on the above literature review, the strongest scientific evidence appears to be for the permanent fishing closures (no take areas). Concerning the other measures listed, there is indirect evidence or weak direct evidence of their effectiveness in supporting and restoring coastal fish populations and communities in the Baltic Sea. To gain stronger support for these measures and for those not yet suggested in this report, it is of outmost importance that on-going and past measures for coastal fish is scientifically evaluated, something that unfortunately is seldom undertaken.

Designed in a proper manner and applied for a specific coastal area, the measures listed in this report likely have positive effects on targeted populations and communities by directly reducing mortality and supporting reproduction. This, in turn, might enhance species diversity and mediate a more balanced size-structure of the targeted population and community. If designed properly, a measure taken is likely also beneficial for the whole ecosystem since fish are key elements with regulatory roles in marine food webs.

In order to further develop appropriate measures for coastal fish, a more in-depth and detailed meta-analysis over the existing literature is needed. Such an effort would also facilitate estimations of the expected effect size of the different measures. Despite that scientific evaluations of most measures undertaken in the Baltic Sea are generally weak or lacking, it is promising that many countries nowadays consider fisheries and environmental management of coastal ecosystems jointly (Kraufvelin *et al.* 2018). The best effects on recipient systems likely comes from a combination of a measures targeting the fishery and protecting the essential habitats of the fish.

5. Conclusions and recommendations

This report provides information on the current state of HELCOM regional collaboration on Baltic Sea coastal fish communities, their monitoring and assessments, current knowledge on important pressures impacting coastal fish communities, as well as on measures to restore and sustain the status of coastal fish in the Baltic Sea. A significant part of the report includes an assessment of the current status (2011-2016) of coastal fish communities evaluated based on HELCOM core indicators. Below, we summarize the key findings of the report and outline potential next steps to take this work further.

The status of the coastal fish communities varies to some extent between areas, regions and indicators. In general, however, the status appears to be rather poor and approximately only half of the assessed areas and covered assessment units obtain good status. There is a geographical pattern with a somewhat better status in more northern areas where perch represents the Key species indicator in comparison with more southern area where flounder is recognised as key species. One central reason for the Functional group indicator not achieving good status in many areas, is too high abundances of the cyprinid species group. Despite a rather extensive network of coastal fish monitoring stations/areas, the available data is only allows assessing 21 of the total 42 assessment units for the Key species indicator. For the Functional group indicator, the coverage is somewhat poorer, enabling evaluation of 16 assessment units for both the cyprinid species group and the piscivores species group. Mainly reflecting the underlying data situation, the confidence of the assessment is moderate to low depending on area and indicator, mainly due to short time-series, poor spatial representation in the monitoring and data quality issues. In order to improve the confidence in future assessments, further development, continuing the already existing monitoring programs and establishment of new monitoring areas in the Baltic Sea is crucial.

A multitude of natural and anthropogenic pressures simultaneously and potentially also synergistically impact on the status of coastal fish communities and their development over time. Whereas a some pressures may have a strong effect and explain a large proportion of the variation in fish abundance, several other pressures have relatively smaller effects or may be apparent only under certain environmental conditions. In general, coastal fish communities are highly influenced by prevailing natural environmental conditions in their area of occurrence, such as hydrography, water depth, predation regime and climatic conditions. Among human induced pressures, the most noticeable are fishing, habitat degradation and eutrophication.

Reflecting this variation, potential measures to restore and support coastal fish populations and communities should have a local perspective, in order to identify the most important pressures and potential remedies in a specific area. However, there is a strong scarcity of evaluations of measures to restore coastal fish in the Baltic Sea. Only a few measures have been scientifically assessed and provided support for their effectiveness. Measures aiming at reducing the mortality of the fish have proven effective, mainly via instating no take areas but to some extent also temporary fishing closures, gear and catch restrictions, as have also measures to improve the reproduction of fish via habitat protection or restoration. There is no general scientific support for measures related to biomanipulation, nutrient and substance abatement, and stocking of hatchery-reared fish.

Coastal fish assessments and monitoring in the Baltic Sea has taken noteworthy steps forward during recent years. This includes for example the development of regionally agreed CORE indicators for coastal fish, the development of a generic concept and method for assessing the status of coastal fish communities across monitoring programs, improved knowledge on the key pressures impacting coastal fish communities and on measures to support and restore coastal fish communities. The work has been driven by increased cooperation between HELCOM Contracting Parties as undertaken within the FISH PRO II, CORESET and the HOLAS II projects, as well by nationally funded development work and research projects.

To that end, the work as presented in this report contributes to the follow-up of the objectives of the Baltic Sea Action plan, the regional coordination of reporting in relation to the Marine Strategy Framework Directive in the Baltic Sea, and national management and assessments of coastal fish.

In spite of recent advances, several knowledge gaps and development needs for coastal fish are evident and should be considered in the future, to enable regionally adequate assessment results with sufficient spatial coverage and confidence. This need also includes the further development and coordination of relevant management actions to support coastal fish communities and their recovery where needed. Important future aims/activities include (responsible body for implementation in italics):

— **Maintaining the current level of monitoring as a minimum and initiate, if possible, new monitoring programs and relevant data collection.** This is essential for increasing the confidence future status assessments, as the current assessment only covers about half of the assessment units in The Baltic Sea. *— Contracting parties of HELCOM*
— **Continued development of the present set of indicators.** During recent years there have been substantial advancement in the use of regionally agreed indicators and assessment approaches among Contracting Parties. Despite this, further refinement of the indicators used in this assessment, with emphasis on data quality and confidence in threshold values is still needed. *— HELCOM FISH PRO III*
— **Harmonization and development of assessment methods.** This includes developing assessment methods that does not require long time-series to enable inclusion of assessment results from additional monitoring programmes. *— HELCOM FISH PRO III*
— **Developing generic size based indicators.** The size-structure of fish populations is of key significance for ecosystem functioning and usually responds strongly to fishing. Currently, however, there is no generic indicator on coastal fish size structure applicable throughout the whole geographical range of the Baltic Sea. *— HELCOM FISH PRO III*
— **Expand the use of coastal fish data.** This could for example include using the existing monitoring network for coastal fish to further follow the distribution, expansion and effects of the round goby. The network of coastal fish monitoring stations offers a unique possibility to study these effects as data before and after the establishment of the species exist in many areas. *— Contracting parties of HELCOM and revelant HELCOM groups with the support of HELCOM FISH PRO III.*
— **Evaluation of measures to restore and support coastal fish communities.** A wide range of measures has been implemented for fish in the Baltic Sea, but there is generally a lack of scientific evaluations and evidence on the effects of many of the measures. This significantly limits the work with restoring and supporting coastal fish communities and stocks. *— Contracting parties of HELCOM and relevant HELCOM groups with the support of HELCOM FISH PRO III*

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