

# COASTAL FISH COMMUNITY INDICATORS IN SWEDEN - VARIATION ALONG ENVIRONMENTAL GRADIENTS



**Lena Bergström & Jens Olsson**

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# COASTAL FISH COMMUNITY INDICATORS IN SWEDEN - VARIATION ALONG ENVIRONMENTAL GRADIENTS

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## WATERS partners:



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WATERS is a five-year research programme that started in spring 2011. The programme's objective is to develop and improve the assessment criteria used to classify the status of Swedish coastal and inland waters in accordance with the EC Water Framework Directive (WFD). WATERS research focuses on the biological quality elements used in WFD water quality assessments: i.e. macrophytes, benthic invertebrates, phytoplankton and fish; in streams, benthic diatoms are also considered. The research programme will also refine the criteria used for integrated assessments of ecological water status.

This report is a deliverable of one of the scientific sub-projects of WATERS and evaluates two methods used in environmental monitoring of coastal fish communities with respect to how they are likely to perform in an indicator-based assessment of environmental status.

WATERS is funded by the Swedish Environmental Protection Agency and coordinated by the Swedish Institute for the Marine Environment. WATERS stands for 'Waterbody Assessment Tools for Ecological Reference Conditions and Status in Sweden'.

Programme details can be found at: <http://www.waters.gu.se>



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## Summary

Coastal fish communities have a central role in both environmental and fisheries management. The following report summarizes the current state (2014) of indicator-based approaches in Sweden, to assess the status of coastal fish communities in relation to internationally agreed directives.

Coastal fish is not included as a biological quality element in the European Water Framework Directive (WFD), with the exception of transitional waters, but they are included in the European Marine Strategy Framework Directive (MSFD). The report is particularly focused on potential connection points between the MSFD and WFD, in order to facilitate the harmonisation of assessments of different ecosystem components and geographical areas. One important aspect would be to develop geographically based assessment methods, to make better use of data from inventory studies.

Key aspects for this development are explored in an example case study, which is based on data with wide geographical coverage. The study addresses general patterns in the distribution of species and indicators among geographical areas in the Baltic Sea, and explores the relationship between indicators and environmental variables. Changes in the indicators were to a large extent attributed to gradients in natural environmental variables, such as temperature, salinity and wave exposure. The results indicate that all these variables should be included in a geographically based assessment. Variables attributed to eutrophication were important for five of the eight studied indicators. This was mainly coupled to a gradient in water transparency. Variables attributed to the mortality of fish were less influential. Possibly, the indicators assessed were not sensitive enough, or the studied gradient was not strong enough for evaluating this pressure. Potentially, also, the explanatory variables that were used were not quantified in an adequate way. A need was seen to update information on the geographical distribution of recreational fisheries and top predators (cormorants, seals), in order to support the assessment of pressure-state relationship, and identify connection points to management measures. All these aspects need to be considered further in the continued indicator development.

The environmental variables explained a reasonable part of the observed variation in the data set, although a relatively large part of the variation was left unexplained. The unexplained variability may potentially be reduced by more refined quantitative analyses, which can also explain variation at different geographical scales. The study was also limited by available environmental data. In terms of additional explanatory variables, habitat quality is often expected to have high influence on species abundances, and hence on indicators. However, this variable could not be included, due to a lack of data with sufficient geographical coverage.

## Svensk sammanfattning

Kustnära fisksamhällen har en central roll i både miljö- och fiskeriförvaltning. Den här rapporten sammanfattar det aktuella läget (2014) för indikatorbaserade metoder för att bedöma miljöstatus hos kustfisksamhällen i Sverige, i enlighet med internationellt överenskomna direktiv.

Kustfisk ingår inte som en biologisk kvalitetsfaktor i ramdirektivet för vatten (WFD), med undantag för vatten i övergångszon. Kustfisk ingår som en del i det marina direktivet (MSFD). Rapporten är särskilt inriktad på gemensamma beröringspunkter för de båda direktiven, i syfte att underlätta en harmonisering av bedömningar mellan olika geografiska områden, och mellan ekosystemkomponenter som bedöms enligt olika direktiv.

Bedömningen av status inom kustfiskövervakningen baseras på förändringar i trender över tid. En annan viktig aspekt är att utveckla geografiskt baserade analyser, för att öka antalet geografiska områden som kan bedömas. Ett ökat antal områden kan övervakas till exempel genom mätkampanjer. Aspekter som är viktiga för denna utveckling utvärderas genom en fallstudie av data med stor geografisk spridning. I studien används data från provfiske i Östersjön, från norra Bottenviken till Blekinge. Generella mönster i fördelningen av arter och indikatorer undersöks, och därtill studeras sambandet mellan indikatorerna och miljövariabler.

Förändringar i indikatorerna kunde till stor del kopplas till förändringar i naturliga miljövariabler, såsom temperatur, salthalt och vågexponering. Resultaten antyder att alla dessa variabler bör ingå i en geografiskt baserad bedömning. Därtill var övergödningsrelaterade variabler viktiga för fem av de åtta studerade indikatorerna. Detta kopplades framför allt till en effekt av siktdjup. Variabler som relaterade till fiskdödlighet (fiske, predation från skarv) hade svagt eller inget samband med indikatorerna. Tänkbara förklaringar till det resultatet är att indikatorernas känslighet behöver förbättras, eller att den studerade gradienten inte var tillräckligt tydlig. Det är även sannolikt att kvaliteten på de variabler som användes spelade roll. Det vore bra att kunna inkludera bättre information om den rumsliga fördelningen av fritidsfiske och toppredatorer (skarv, säl). Detta skulle underlätta utvärdering av förhållandet mellan påverkansfaktorer och indikatorer, men även utvärdering av potentiella förvaltningsåtgärder.

De studerade variablerna förklarade en skälig del av den observerade variationen i indikatorerna, även om en relativt stor del av variationen inte kunde förklaras. Den oförklarade variationen kan eventuellt minskas med mer förfinade kvantitativa analyser som också tar hänsyn till variationen på den geografiska skalan. Studien begränsades också av tillgängliga miljödata. När det gäller förklarande variabler kan kvaliteten på fiskens livsmiljöer, till exempel rekryteringsområden, förväntas ha stor påverkan. Detta kunde inte tas med i den aktuella studien, på grund av brist på data med tillräcklig geografisk täckning.



# 1 Introduction

Coastal fish are here defined as fish species that spend a large part of their life cycle in shallow coastal areas. Most coastal fish species in Swedish waters are coastal residents, and live in coastal habitats almost all of their life cycle. However, migrating species are also common. These use coastal habitats during specific parts of their life cycle, for example for spawning or as nursery areas, or are seasonal migrant species, which make regular seasonal visits to coastal habitats (Elliott and Dewailly, 1995, Pihl and Wennhage, 2002). Most of the migrating species in coastal areas have a marine origin, both in the Baltic Sea and on the Swedish west coast. Some species are also anadromous, and migrate into freshwater rivers for spawning. Coastal residents are represented by species of both marine and freshwater origin, although freshwater species clearly dominate in the Baltic Sea (Karlsson et al., 2012).

Several coastal fish species are commercially important in Swedish waters, and even more species are of interest for recreational and household fisheries (SwAM, 2012a). In the Swedish part of the Baltic Sea, the socio-economically most important species (important within both these categories) are pikeperch, perch, pike, whitefish and flatfishes, such as flounder and turbot. Evidence is also accumulating for the role of coastal fish in supporting and regulating food web process, by both bottom-up and top-down mechanisms. For example, coastal fish species are important food sources for piscivorous fish, birds and marine mammals. By eating smaller fish species and invertebrates, and thereby controlling their abundance, coastal predatory fish species may also have a regulating function by controlling the amount of prey species in an area, thereby inducing top down processes (Eriksson et al. 2011). Due to their combined importance for both fisheries and environmental aspects, fish communities have a central role in coastal and marine management, and there are several motives for maintaining sustainable coastal fish communities and identifying appropriate management measures in order to achieve this.

With respect to the assessment of ecological status, the jurisdiction of the European Water Framework Directive (WFD) extends to the seaward limit of one nautical mile from the baseline of territorial waters (EC, 2000). However, coastal fish is not included as a biological quality element (BQE) within the WFD. Fish is only included for inland waters and coastal transitional waters<sup>1</sup>(EC, 2000). Geographically, the European Marine Strategy Framework Directive (MSFD) covers sea areas until the border of the WFD, and thematically, it covers ecosystem components that are not included in the WFD (EC, 2008). This implies that ecosystem components and geographical areas that are not

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<sup>1</sup> The transitional waters are defined as bodies of surface water in the vicinity of river mouths which are partly saline but substantially influenced by freshwater flows (EC 2000).

included in the WFD should be included (or considered for inclusion) in status assessment in relation to the MSFD. Within the MSFD, biodiversity related aspects of coastal fish are reported under Descriptor 1 (Biodiversity), and some aspects of fish community status may also be reported under Descriptor 4 (Food webs). Commercial fish species are reported under MSFD descriptor 3 (Commercial fish and shellfish; eg HELCOM (HELCOM, 2013b)). Also, some ecosystem components may be reported under the WFD in coastal areas and under the MSFD in other sea areas. Hence, there is a general need to harmonize the assessment routines among the WFD and the MSFD, in order to facilitate an integrated assessment of ecosystem status in coastal areas, which may include ecosystem components assessed under both directives. Also, there is need to develop methods for a “seamless assessment” between the MSFD and the WFD in relation to the geographical areas where they meet (Borja et al., 2010). One purpose of Work Package 3.4 within WATERS is to provide background information in order to support this, when it comes to method for assessing coastal fish community status.

This report gives a summary of the current state of indicator-based approaches for assessing the status of coastal fish in Sweden, which is currently in a strong development phase relating to status assessments for the MSFD (EU 2008) and the Baltic Sea Action Plan (HELCOM, 2007). The summary is based on information from various projects at national and international level that have not been put together in one report before. The report additionally explores indicator variability along natural environmental gradients at a large scale, in order to highlight and support ongoing questions relating to the development of management targets and the identification of biologically relevant assessment units.

## 2. Background

This section summarizes and compares general key aspects of the assessment approaches of the Water Framework directive (WFD) and the Marine Strategy Framework Directive (MSFD), to the extent that is needed in order to understand the following sections.

### 2.1 Good status - What does it look like?

Within the Water Framework directive (WFD), ecological status of a biological quality element is defined by assessing the deviation between its status in the observed condition and the undisturbed condition, where the undisturbed condition is unaffected by human activities (reference condition):

*“The values of the biological quality elements for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion.”* (EC, 2000).

A key aspect is that deterioration and improvement of “ecological status” is defined by the response of the biota, rather than by changes in environmental parameters (Birk et al., 2012). Since the ecological landscape is altered by anthropogenic influence since long time, it is generally not possible to identify areas representing truly undisturbed conditions, and reference conditions are primarily defined as areas with very minor ecological effects of human pressures (EC, 2003a).

Good environmental status (GES) according to the Marine Strategy Framework Directive (MSFD) refers to a condition associated with sustainable use.

*“... the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable...”* (EC 2008).

The principal steps of an environmental status assessment within the MSFD may be described as in figure 1. A key concept of the MSFD is that the assessment of environmental status is integrated among different ecosystem components, and combined with an analysis of societal and economic aspects (SwAM, 2012b). Further, approaches for assessing environmental status should ensure that the definition of good environmental status can be regularly re-assessed in order to take account of continuous broader changes in the marine environment (EC, 2010).

In practice, the definitions of good status according to the MSFD and the WFD, respectively, can be seen to meet at the good/moderate boundary of the WFD and the good environmental status (GES)/sub-GES boundary of the MSFD (EC, 2010). According to both directives, a status below this boundary calls for management measures in order to improve the status (Borja et al., 2010).

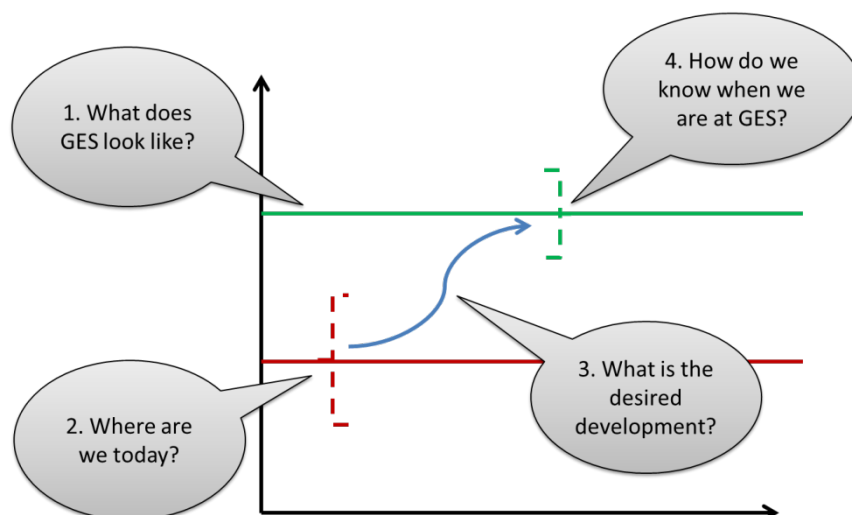


FIGURE 1. Principal steps of a status assessment within the MSFD. Central elements are the adaptive process and the definition of good environmental status (GES) as a long term sustainable ecosystem condition.

## 2.2 Approaches for defining management targets

Both the WFD and the MSFD requires that member states provide quantitative definitions of the management targets. Within the WFD, the target is defined in relation to the reference condition. The reference condition is not the target, but is used as information about the desired trajectory for improving ecological status (EC, 2003a). Typically, management targets are defined using spatial empirical approaches, where impacted areas are compared with areas representing minimally impacted conditions (with respect to the pressure in focus). Other potential approaches are using historical data or predictive modelling. In cases where data deficiency is evident, expert judgment may also be referred to (Borja et al., 2010, EC, 2003b).

The MSDF is less prescriptive on how to define the quantitative targets (EC, 2010). However, temporal approaches may be viewed as indispensable in order to monitor and evaluate progress, as required within an adaptive management framework (Levin et al., 2009). They are also important in order to identify potential problems of shifting baselines in relation to target setting (Airolidi and Beck, 2007, Cardinale et al., 2011). However, naturally, temporal approaches are only possible in cases where long term monitoring data

is available. Spatial approaches may be preferred in order to make use of information from recently initiated monitoring programs, and data from inventories in order to improve the geographical coverage of the assessment.

## 2.3 Assessment units

Coastal areas of Sweden are subdivided into 662 coastal water bodies, based on local topographical and hydrological conditions ([www.smhi.se](http://www.smhi.se)). The water bodies are the basic reporting unit of the WFD. The MSFD does not prescribe particular assessment units by its definition, but requires that the geographical scale at which each indicator is reported reflects biologically meaningful properties of the indicator (EC, 2008).

When defining a quantitative target for any reporting unit, it is naturally assumed that the target is biologically relevant within the unit. The importance of biogeography is obvious for assessments that rely on species-specific indicators, or on indicators that build on the appearance and disappearance of sensitive versus tolerant species. However, local environmental conditions may also influence the performance of indicators that are not species-specific, via their underlying species structures. The same species may also respond differently to the same anthropogenic pressure in different areas, depending on other properties of the external environment.

The approach of applying typologies and type-specific reference conditions is designed to take account of these aspects. Water bodies within the same typology are expected to hold similar external preconditions, and thereby potentially support similar biological communities. From this is also inferred that the biological relevance of different status indicators and their responses to anthropogenic pressures is also similar. Alternatively, species and indicator responses along environmental gradients may be studied on a continuous scale, using statistical modelling.

An evaluation of species and indicator changes along natural biological gradients is particularly motivated for ecosystems in which species show a high level of dispersal and connectivity, such as marine and coastal ecosystems. In these situations, different species within the same unit may be governed by different environmental factors, acting at various spatial scales (EC, 2003a, EC, 2008). This is also true for coastal fish. Fish are highly mobile, and are thereby likely to pick up environmental signals from various parts of the coastal area during their life time, and be subject to environmental drivers at various spatial scales (Olsson et al., 2012a). On the other hand, the genetic population structure of coastal fish species may be highly local (Olsson et al., 2011, Saulamo and Neuman, 2002, Laikre et al., 2005a, Laikre et al., 2005b, Olsson et al., 2012b) and recruitment success may be strongly connected to local conditions (Sundblad et al., 2014). Hence, knowledge on the distribution patterns of species in relation to natural environmental variables are important for identifying the geographical scale at which status assessment are best targeted.



### 3 Objective

The following report summarizes the development so far with respect to indicators for status assessment of coastal fish in Sweden. Approaches for assessing the environmental status of ecosystem components in coastal and marine areas are currently under ongoing development, driven by the Marine Strategy Framework Directive (SwAM, 2012c, EC, 2008). Simultaneously, existing approaches for assessing ecological status in relation to the Water Framework Directive are being reviewed and revised (EC, 2012). In order to facilitate the integration of fish community status assessments with status assessments of other parts of the coastal ecosystems, the report is particularly focused on potential connection points between the MSFD and the WFD.

Further, the natural variability in coastal fish communities and indicators is assessed by a large scale case study from the Baltic Sea. The analyses support the development of spatially based approaches for status assessment, which is defined as key aspect for harmonization with the WFD. Similar analyses were also applied in the initial indicator selection process within HELCOM (HELCOM, 2012, Olsson et al., in prep.). However, recent data collection on both response data (fish) and environmental data have made it possible to include a slightly larger data material. Patterns in species composition and indicators/potential indicators are explored in relation to changes in natural environmental variables that may be associated with the latitudinal gradient, or with coastal topography. In addition, the relative importance of natural and anthropogenic pressure factors is assessed.

## 4. Summary of status assessments of coastal fish in Sweden

### 4.1 The development of an indicator-based assessment

In the Baltic Sea, initiatives towards an indicator-based assessment of coastal fish communities were first taken by HELCOM with the initiation of a network of experts on coastal fish in 2003, the HELCOM fish project. The project established a connection among experts from all states around the Baltic Sea, and enabled a coordinated monitoring and assessment of coastal fish in the region (HELCOM, 2006, Adjers et al., 2006).

The efforts received increased importance with the introduction of the Baltic Sea Action Plan (BSAP) where coastal fish was included as one central element (HELCOM, 2007). The HELCOM fish expert network later suggested a set of 10 status indicators reflecting aspects of biodiversity, species composition, size composition and trophic structure in status assessments of coastal fish (HELCOM, 2012). The indicators were combined in a multivariate assessment to explore changes over time, and to identify key features of observed changes by identifying the indicators that contributed most to the observed patterns. The indicators were selected based on general criteria for indicator selection (Rice and Rochet, 2005, Rochet and Trenkel, 2003), and also included aspects of policy relevance and conceptual relationship to pressures.

With the implementation of MSFD, HELCOM promoted the use of CORE indicators, which are set to represent key aspects of various biotic ecosystem components and pressures commonly agreed among member states (HELCOM, 2013b). The purpose of the CORE indicators is to support international coordination of assessments in relation to the MSFD by the member states at regional (Baltic Sea) scale. For coastal fish, CORE indicators have been derived from the set of indicators proposed in 2012 (HELCOM 2012), and represents *the abundance of key species* and *the abundance of key functional groups*. In addition to the CORE indicators, a set of candidate indicators are under evaluation, and may potentially be included at a later stage in the MSFD process. The national reporting is not restricted to the regional CORE indicators. The initial Swedish MSFD report (SwAM, 2012c) included five potential coastal fish indicators (Table 1).

The corresponding international coordination in the OSPAR region, to which the Swedish West coast areas of Skagerrak and Kattegat belong, has not included coastal fish. A series of indicators was suggested for Swedish coastal waters in the OSPAR region (Wennhage

et al., 2012, SwAM, 2012c). The indicators were selected in order to be coordinated at a national level with corresponding assessments in the Baltic Sea. The performance of the Swedish West coast coastal fish indicators were evaluated with respect to precision by Bergström et al (2013).

Since 2012, the environmental status of coastal fish communities is also included in the Swedish national environmental legislation (“Miljökvalitetsnormer”, Table 1) (HVMS, 2012).

TABLE 1. Overview of current indicators for status assessment of coastal fish. CORE indicators according to HELCOM are denoted “Core” (HELCOM, 2013), Swedish national MSFD indicators are denoted “Swe” (SwAM, 2012c). Indicators included in the national legislation are denoted “L” (HVMS, 2012). The numbers in column one denote the criteria to which the indicator is referred to in the MSFD (EC, 2008). “Primary anthropogenic pressure” indicates the pressure that the indicator is most likely responsive to in relation to short term management measures (HELCOM, 2013). “Mortality” refers to fishing pressure or natural predation.

	<b>Indicator</b>	<b>Ref</b>	<b>Primary anthropogenic pressure</b>	<b>Desired direction</b>
1.2	Abundance of key species	Core, Swe, L	Habitat quality, Mortality	Should not be too low
1.3	Size structure of key species	Swe, L	Mortality	Should not hold too few large species
1.6	Abundance of important functional groups; Piscivores; Cyprinids	Core, Swe, L	Habitat quality, Mortality; Eutrophication	Should not be too low; Should not be too high or too low
1.6	Size structure in the fish community	Swe, L	Mortality	Should not be too low
1.7	Trophic level in the fish community	Swe	Mortality, Eutrophication	Should not be too high or too low (the indicator is a ratio)

## 4.2. Geographical units for the assessment

For coastal fish within the MSFD, Sweden has defined the basic unit for national reporting as the scale of the 25 Swedish coastal water areas (SWaM, 2012c). At international scale within the Baltic Sea region, reporting of coastal fish is planned at the level of HELCOM Assessment units level 3 (National coastal waters within sub-basins; Figure 2; HELCOM, 2013b). Currently, Swedish long term environmental monitoring of coastal fish communities covers 14 coastal water areas at least partly (Fredriksson 2014, Figure 2). The monitoring takes place in sites that are considered not at all or only marginally affected by direct anthropogenic disturbance. Changes that are observed over time areas are considered to reflect changes in large scale environmental factors, and synchronous changes in different areas are interpreted as supporting the presence of large

scale environmental drivers. Four of these areas are located at the Swedish west coast including the strait of Öresund, and nine in the Baltic Sea (Fredriksson, 2014). Hence, 11 coastal water areas are not covered by coastal fish monitoring at all. In relation to the assessment units of the WFD, the monitoring covers 107 of 662 (16%) coastal water bodies.

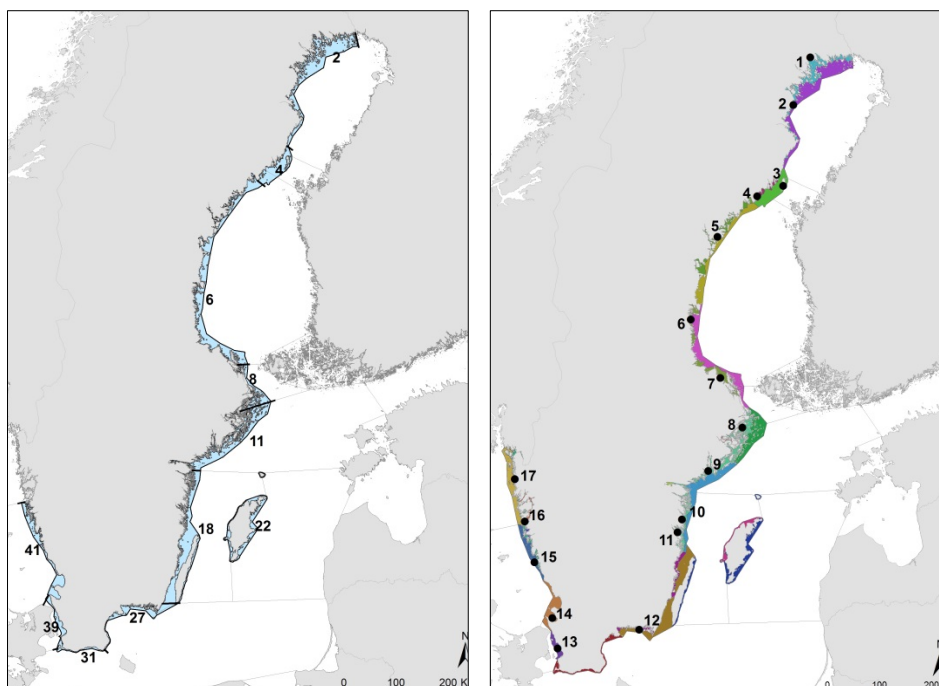


FIGURE 2. Map to the left: Swedish coastal waters divided into the HELCOM Assessment units level 3 (Coding: 2 = Bothnian Bay, 4 = The Quark, 6 = Bothnian Sea, 8 = Åland Sea, 11 = Northern Baltic Proper, 22 = Eastern Gotland Basin, 27 = Bornholm basin, 31 = Arkona Basin, 39 = The Sound, 41 = Kattegat). Map to the right: Delineation of Swedish national coastal areas. Points give the position of areas where long term monitoring of fish communities takes place. (1 = Råneå, 2 = Kinnbäcksfjärden, 3 = Holmön, 4 = Norrbyn, 5 = Gaviksfjärden, 6 = Långvindsfjärden, 7 = Forsmark, 8 = Lagnö, 9 = Asköfjärden, 10 = Kvädöfjärden, 11 = Vinö, 12 = Torhamn, 13 = Barsebäck, 14 = Kullen, 15 = Vendelsö, 16 = Älgöfjorden, 17 = Fjällbacka).

### 4.3 The assessment protocol

Approaches for assessing good environmental status for coastal fish are currently under establishment. The indicators are to be operational at Swedish national level in 2016 and implemented in the MSFD reporting of the second cycle in 2018.

The suggested assessment protocol, as summarized below, is agreed on at Baltic Sea regional level. It is based on the development over time in established areas for long term monitoring. Indicator values during the assessment period are compared with values during a baseline period, and environmental status is assessed based on the level of

deviation from the baseline. The baseline is obtained from the same monitoring data set, and holds a minimum of 10 consecutive years. The baseline represents the situation within the data set during a time period with known environmental status and without ongoing trend within the baseline. The baseline condition can represent a situation with either good environmental status (GES) or not good environmental status (sub-GES). In the case where the baseline represents good environmental status, the boundary level for the indicator is to remain within the confidence limits of the baseline condition. In the case where the baseline represents not good environmental status, the boundary level for the indicator is to leave the confidence limits of the baseline condition into the desired direction (Table 1, last column). The boundary for the baseline conditions are set based on observed values of the indicator during the baseline years. A bootstrapping procedure is applied in order to identify the indicator boundary value at a defined percentile. The percentile is different for different indicators, depending on the desired direction of the indicator (Olsson et al., in prep.). The indicator value during the assessment period is defined as the median value of five consecutive years.

In cases where only shorter time series data is available, a trend based assessment is used instead. In these cases, status is determined by comparing the slope of the indicator value over a time with the desired direction of the indicator (eg (HELCOM, 2013a).

The temporally based approaches are suitable for status assessment in areas that are covered by long term monitoring. These areas provide a valuable source of information on long term changes in the environment and an anchoring point for the identification of a “sustainable condition” in relation to climatological changes and other natural long term variation, as required by the MSFD (EC, 2008). A remaining point of development is to identify suitable approaches for aggregating of assessment results into relevant reporting units (see next section). Also, there is a need to develop a spatially based approach in order to assess status in areas that are not covered by long term monitoring. This would be required for the harmonization with the WFD and is also needed in order to identify priority areas for improved management measures in relation to the MSFD and national environmental objectives. A key issue in order to address these questions is to identify the response of the indicators in relation to natural environmental gradients, and to quantify the indicators’ relationship to anthropogenic pressures. This knowledge is a basis for identifying natural geographical patterns in the fish community and for identifying boundary values for areas where there is not enough long term monitoring data available for the temporally based approaches.

## 5. Methods for the case study

### 5.1 Fish data

Fish data was obtained from ongoing environmental monitoring programs in the Baltic Sea ([www.slu.se/faktablad-kustfisk](http://www.slu.se/faktablad-kustfisk)) and other sampling campaigns performed using the same methodology. The added data included information obtained during a fish monitoring campaign in 2011 (Söderberg and Mattsson, 2011), within the WATERS gradient study in 2013 (Pihl et al., in prep) and from other inventory and dedicated monitoring studies. The total data set thereby also included areas of increased anthropogenic impact, not only undisturbed reference areas for environmental monitoring. The variation in anthropogenic impact was mainly related to differences in nutrient status, but also to diffuse impacts. Some of the areas were fished close to areas with high human population density, industrial areas, or commercial harbors. These areas were potentially subject to enhanced fishing mortality (by being close to densely populated areas) or enhanced contaminant levels (close to industries and harbors). Also, data from some areas closed to fishing was included. The data set was not analyzed with respect to all of these variables. The focus of the case study was on changes in the fish community in relation to natural environmental variables (relating to salinity, temperature and wave exposure), as contrasted to gradients in eutrophication and assumed levels of fish mortality (through fishing and predation). These aspects are further outlined below. In all, the data set represented a latitudinal difference of ten degrees (from 56 to 66°N), and a salinity gradient from 2.1-7.7 PSU along (Table 2, Figure 3).

TABLE 2. Areas included in the study. Columns give the years included, their latitude and mean annual salinity (salinity according to SMHI, see text). The column "Type" indicates the original purpose of the sampling: EM= annual environmental monitoring, OM= Other yearly repeated survey, C= Campaign conducted only one year, for example inventories. "Stations" gives the total number of stations sampled at 0-20 m depth (the study included data from 0-10 m depth).

Area	Abbrev.	Years	Latitude	Salinity	Stations	Type
Asköfjärden	ASKÖ	2005-2012	58° 50	6.2	48	EM
Askviken	ASKV	2009-2013	59° 06	5.8	45	OM
Bråviken	BRÅV	2009	58° 33	6.5	40	C
Finbo, Åland	FINB	2002-2013	60° 17	Nd	45	EM

<i>Table 2, continued</i>	<i>Abbrev.</i>	<i>Years</i>	<i>Latitude</i>	<i>Salinity</i>	<i>Stations</i>	<i>Type</i>
Forsmark	FORS	2002-2013	60° 26	5.3	45	OM
Gaviksfjärden	GAVI	2007-2013	62° 52	4.8	45	EM
Gräsö östra skärgård	GRÄS	2012	60° 22	5.3	44	C
Holmön	HOLM	2002-2013	63° 40	4.2	30	EM
Inre Bråviken	INBR	2013	58° 37	5.3	29	C
Inre Slätbaken	SLÄT	2011-2013	58° 27	3.7	75	C
Kaggebofjärden	KAGG	2013	58° 00	6.2	30	C
Karlshamn	KAHA	2010	56° 08	7.2	33	C
Karlskrona	KARL	2009	56° 05	7.2	50	C
Torhamn	TORH	2002-2013	56° 09	7.2	44	EM
Kärrfjärden	KÄRR	2006; 2013	58° 21	6.5	70	C
Kinnbäcksfjärden	KINN	2004-2013	65° 02	2.5	45	EM
Klacksten	KLAC	2007	58° 24	6.5	27	C
Kumlinge, Åland	KUML	2002-2013	60° 13	Nd	47	EM
Kvädöfjärden	KVÄD	2002-2013	58° 00	6.3	45	EM
Lagnö	LAGN	2002-2013	59° 33	5.3	45	EM
Långvindsfjärden	LÄNG	2002-2013	61° 27	5	45	EM
Lännäkersviken	LÄNN	2009-2013	59° 06	5.9	45	OM
Licknevarpefjärden	LICK	2013	58° 03	6.5	36	C
Lilla Värtan	LILL	2011	59° 20	2.3	43	C
Möllefjorden	MÖLL	2009	56° 00	7.7	35	C
Norrbyn	NORR	2002-2013	63° 31	4.1	45	EM
Nynäshamn	NYNÄ	2010	58° 52	5.9	39	C
Örnsköldsviksfjärden	ÖRNS	2011	63° 15	4.1	45	C
Ost Löno	OST	2013	58° 35	6	18	C
Östhammarsfjärden	ÖSTH	2011	60° 15	5.1	45	C
Pukaviksbukten	PUKA	2006; 2009	56° 08	7.1	70	C
Råneå	RÅNE	2002-2013	65° 50	2.2	45	EM
Skelleftebukten	SKEL	2011	64° 39	2.6	45	C
Svensksundsviken	SVEN	2006	58° 37	5.1	30	C
Tjäröfjärden	TJÄR	2008	56° 09	7.3	26	C
Torsås	TORS	2011	56° 24	6.9	45	C
Trännöfjärden	TRÄN	2013	58° 24	6.1	30	C
Tromtöfjärden	TROM	2009	56° 09	7.1	30	C
Västra Sjön	VÄST	2007	56° 38	6.8	30	C
Vissvassfjärden	VISS	2004-2013	59° 10	5.3	40	C
Yttre Fjärden Bråviken	YTTR	2011	60° 42	3.9	45	C



Figure 3. Position of the areas included in the case study. The names of the areas are abbreviated, and the full names are given in table 2. In addition to the Swedish areas, part of the analyses included data from the Åland Islands (FINB, KUML), which are also reported to the database that was used.



## 5.2 Environmental data

The environmental data used are presented in table 3. The data included natural environmental variables, which cannot be affected by management measures but which are likely to have an effect on species composition, species abundances and indicators. The natural environmental variables included were temperature, salinity and wave exposure.

The study also included a set of anthropogenic pressure variables, focusing on the aspects of eutrophication and fishing mortality. The common feature of these variables was that they are potentially manageable by human activity. In the category of fishing mortality, one variable describing natural predation was also included (cormorant density), as this also has an effect on fish mortality, and is potentially manageable by regulating the number of top predators in the system.

The environmental data was obtained from different sources. Data for the variables “Temperature at fishing” and “Water transparency” was sampled in connection to the fishing. Temperature was measured at each station when lifting the nets, at the fished depth. Water transparency was measured in a central part of the fished area each day of fishing. Estimates for one area and year were typically based on 3-5 daily measurements, as each fishing trip typically lasts 3-5 days. In the analyses, the variable water transparency was multiplied by a factor -1, so that its expected effect would have the same direction as the other anthropogenic pressure variables (higher values implying increasing pressure).

Information on commercial fish landings, human population density, cormorant density and wave exposure was obtained from GIS layers. Data for the variable “Commercial landings of coastal fish” was compiled from data on Swedish and Finnish national catch statistics. This is reported on the level of ICES statistical rectangles, which are 55 by 55-60 km. The values that were used represented interpolated average catches per km<sup>2</sup> water area during the years 2004-2008 and include all species landed except herring, sprat and vendace. The selection of years was based on data availability, and it was assumed that geographical differences captured by the data were also representative for earlier and later years. The variable “Human population density” was used as an index to approximate the level of recreational fishing pressure. According to Thörnqvist (2009), half of the recreational fishing trips in Sweden are carried out within 30 km from the fisherman’s home. Hence, assuming that catches in recreational fisheries are proportional to the number of persons living in an area, the distribution of recreational fishermen was estimated based on the number of inhabitants within a 30 km radius, based on data on human population density (Sweitzer et al 1996). “Cormorant density” was estimated based on Swedish count data on nesting great cormorants (*Phalacrocorax carbo sinensis*) in 2006 (Staav 2007). A kernel density function with a 20 km radius was applied to interpolate the catches over the areas within feeding flight distance from the colonies, resulting in estimates of kg fish caught per km<sup>2</sup> water area. Data for the variable “Wave exposure” was obtained using the “Wave Impact” software (Isæus, 2004), which combines fetch

calculations with wind conditions and also accounts for refraction and diffraction effects. Data for the variables “Mean annual temperature”, “Salinity” and “Chlorophyll-a” were obtained from hydrographical model data (<http://vattenweb.smhi.se/>).

TABLE 3. Environmental data used in the analyses, their abbreviations as used in the resulting figures, their ranges and units.

Variable name	Abbreviation	Range	Unit
<b>Natural environmental variables</b>			
Temperature at fishing	Temp_F	14.5-21.6	°C
Salinity	Salinity	2.2-7.7	PSU
Mean annual temperature	Temp_Ann	5.6-10	°C
Wave exposure	WaveExp	4.6-64 000	(Index)
<b>Anthropogenic pressure variables - Eutrophication</b>			
Chlorophyll-a	Chl-a	0.4-5.4	mg/L
Water transparency	WaterTR(-1)	0.6-7.5	M
<b>Anthropogenic pressure variables - Fishing mortality</b>			
Commercial landings of coastal fish	CommFish	0-1000	kg/km <sup>2</sup> and year
Human population density	People	0-507	n/km <sup>2</sup>
Cormorant density	Cormorants	0-610	n/km <sup>2</sup>

### 5.3 Data preparation

The basic unit for the analyses was the fishing area. Hence, data from all stations within each fishing area and year were combined to a mean value, and the focus of the analyses was on differences among areas and years, not on variation among sites within each area.

The fish community data set was analyzed in two forms. One data set included the catches of all fish species, and one other data set included a set of indicators. A description on how the indicators were computed is presented in table 3. The indicator data set included the regionally coordinated indicators suggested by HELCOM (2013); “Abundance of key species”, “Abundance of important functional groups; cyprinids and piscivores” and two Swedish national potential indicators (SwAM, 2012c); “Size structure of key species” and “Trophic level in the fish community”. In addition, metrics relating to species diversity, the size structure of piscivores, and the total size structure of the fish community were included for evaluation purpose, in order to represent biodiversity and food web aspects.

The analyses were limited to stations fished within 0-10 m depth, in order to minimize the effect of sampled depth on the results. Hence, measures were based on data from 17-50 stations per area, typically 30-40 stations. Only fish with a body length at least 12 cm were included, since smaller individuals were not fully captured by the gear (HELCOM 2012).

The length limit of 11 cm was verified by observations of length frequency diagrams computed based on the assessed data set.

TABLE 4. Computation of the indicators analyzed in the case study. Full names refer to the name of the indicator in its original reference (see text). Name in brackets is the name used in this report, specifying also how it was computed in this report. CPUE (catch per unit effort) corresponds to the number of fish per net and night. Piscivores are defined as species with a trophic level of at least 4 ([www.fishbase.com](http://www.fishbase.com)). Cyprinids are fish within the taxonomic family of *Cyprinidae*.

Indicator	Computation
Abundance of key species	CPUE of perch
Size structure of key species	CPUE of perch $\geq$ 25 cm
Abundance of important functional groups 1 (Abundance of piscivores)	CPUE of piscivores
Abundance of important functional groups 2 (Abundance of Cyprinids)	CPUE of Cyprinids
Trophic level in the fish community (Proportion of piscivores)	CPUE piscivores/Total CPUE
Size structure in the fish community (Proportion of large fish)	CPUE fish $\geq$ 30 cm /Total CPUE
Size structure of piscivores (Proportion of large piscivores)	CPUE piscivores $\geq$ 30 cm/CPUE piscivores
Species richness in the fish community (Species richness)	Number of species per 100 individuals

## 5.4 Analyses

### Variation among areas in species composition and indicators

The geographical variability in species composition and indicator values among areas and years was explored using principal coordinates analysis (PCO). This is a type of unconstrained multivariate analysis. Samples are arranged along a number of axes in order to capture the total observed variability among data points and maximize the share of the variability captured by the first axes (Anderson et al., 2008). The different areas and years were compared with each other with respect to similarity in species composition and indicator values.

Similarity according to the species data set was assessed by the Bray-Curtis similarity index. The analysis was based on square-root transformed data in order to down-weight the importance of dominant species. Similarity according to the indicator data set was assessed by Euclidian distances based on normalized data. Indicators that were defined based on CPUE levels were square root-transformed prior to the analyses, in order to improve linearity.

The analyses were focused on exploring the relative importance of temporal (within area, among year) and spatial (among areas) variability. Also, the species and indicators that mainly characterized the observed differences were identified by examining their vector scores on PCO axis 1. Supporting information was obtained based on the corresponding vector scores on PCO axis 2.

### Identification of main environmental gradients

The total environmental data set included natural environmental variables and anthropogenic pressure variables. The data set was explored by PCO in the same way as the indicator matrix, in order to identify main environmental gradients. However, no information on variation among years was included in this analysis. Several of the variables were originally represented by only one value per area, without inter-annual variation, and the remaining variables were averaged into mean values for all years in order to focus on spatial aspect. Similarities among areas were quantified based on Euclidian distances and normalized data. Prior to the analyses, data on wave exposure, commercial landings, people density and cormorant density were log+1- transformed in order to improve linearity.

### Relationship between indicators and environmental data

The relationship between indicators (Table 4) and potential explanatory variables, including natural environmental variables and anthropogenic pressure variables (Table 3), was explored using distance-based linear modelling (DistLM). This method is conceptually similar to a multiple linear regression, but the response variable is quantified by the distance between data points according to a resemblance matrix. By this, the method is flexible towards assumptions on data distributions, and it is useful for estimating the relative importance of different explanatory variables on a response data set (Anderson et al., 2008). Since the analysis is based on the output of a resemblance matrix, it is possible to combine several response variables in the analysis. However, only one indicator at a time was assessed here, in order to evaluate their individual relationship to the potential explanatory variables. Similarities were quantified using the Bray-Curtis similarity index.

The analyses were made on area level, with the purpose of achieving a balanced design of response variables and potential explanatory variables. Data from areas sampled during more than one year were combined into one average value for all years prior to the analyses. The merging of environmental data was in line with the procedure for the unconstrained analyses (see above). The merging of indicator data was justified by the results of the unconstrained analyses of species and indicator data sets (see above and results section), which showed that differences among areas were systematically higher than differences among years. Data from the Åland islands (Finbo, Kumlinge) was not included, as there was no corresponding environmental data to compare with for these areas.

In order to facilitate the interpretation of the results, redundant explanatory variables were identified prior to the analyses, based on analysis of variance inflation factors (VIF). In cases where high VIF values were observed, indicating a strong correlation among two variables, the variable with the highest VIF value was removed sequentially until only values below 5 were present (Zuur et al., 2007). This procedure resulted in the exclusion of one variable “Mean annual temperature”, which was strongly correlated with “Salinity”. The linear correlation coefficient between these two variables was 0.80.

The explanatory variables that were most strongly related to variation in the indicator were identified for each indicator at a time, using the “best” option available in DistLM. The best obtainable combination of explanatory variables was identified based on minimizing the criterion AICc, which compensates for the number of variables included. The variables included in the best obtainable model, and in all alternative models that had AICs values within the range of 2 units from the best model, were identified. Together, these are referred to as the “alternative best models” in the results section. The maximum number of alternative models for each indicator was set to 10. The explanatory level of the alternative best models was quantified by their  $R^2$  values

In addition, the distribution of selected species along the salinity gradient was visualized in scatterplots. The purpose of this was to give an overview on the geographical distribution of species that have a potentially high influence on the indicator values. Hence, the natural geographical distribution of these species in relation to salinity could potentially have an effect on reference values for the indicators along a salinity gradient.

All multivariate ordination analyses were performed using the PRIMER 6.0 and Permanova+ software (Anderson et al., 2008) and the VIF analyses were performed using Brodgar 2.6.6 ([www.brodgar.com](http://www.brodgar.com)).

## 6. Results

### 6.1 Geographical variability

#### Variation among areas and years in species composition

According to the PCO, the areas were clearly separated from each other in terms of species composition. There was also some variability among years for areas sampled many years, but this variation was consistently smaller than the variation among areas (Figure 4). The species contributing most to the observed pattern among areas were roach, ruffe, white bream, herring, whitefish, smelt and bream. The species showing strongest correlation with the second PCO axis were pike, pikeperch and perch (vectors not shown in the figure).

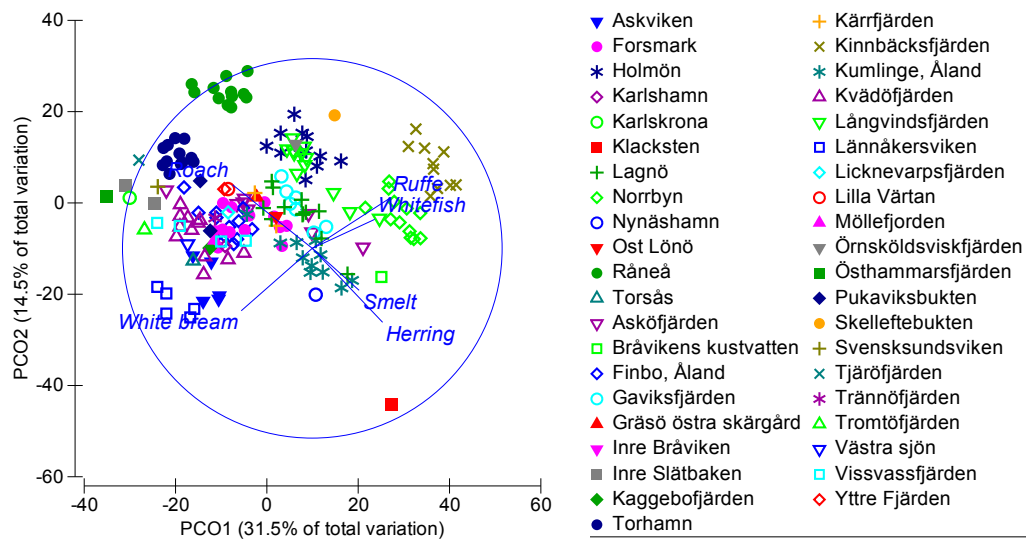


FIGURE 4. Output of the PCO based on the species matrix. The graph shows the position of data points along the first two PCO axes. Data points located close to each other are more similar in species composition than data points located far from each other. Data points are coded with a unique symbol for each area. Similar symbols represent sampling in the same area during different years. The vectors (blue lines) indicate which species contributed most to differences among areas. For example, roach and white bream were relatively more common in areas located in the left part of the ordination, whereas ruffe, whitefish, smelt and herring were more common in areas located in the right part. The vectors point in the direction of areas with relatively high abundances of the species that it represents. A longer line indicates a stronger relationship. Together, the first two axes comprised 46.0% of the total variability in the data set (31.5% on PCO1 and 14.5% on PCO2), as measured by the Bray Curtis similarity index.

### Variation among areas and years in indicator values

The different areas showed considerably less variation in indicator values than in species composition. When comparing the PCO based on species composition (Figure 4) and the PCO based on indicators (Figure 5), the data points were clearly more aggregated in the latter case. However, a few areas deviated. These were characterized by low values in indicators typically influenced by the abundance of perch (areas located towards the right end of the plot).

Several indicators had similar influence on the overall pattern. The indicators “Abundance of key species” and “Abundance of piscivores” were closely connected, as were “Proportion of piscivores” and “Size structure of key species” (vectors pointing in similar direction, see figure 5). The PCO also indicated a potential redundancy between the indicators “Proportion of large fish”, “Proportion of piscivores” and “Species richness”.

The indicators contributing most to differences among areas were “Abundance of key species”, “Abundance of piscivores”, and “Species richness” (these indicators had the strongest correlation with the first PCO axis). The other indicators showed the strongest correlation with the second axis, with the exception of the indicator “Abundance of Cyprinids”, which had weak correlation with both the first two axes. This indicator showed a relatively high correlation with the third axis (not shown, PCO3 was attributed to 16.5% of the total variation).

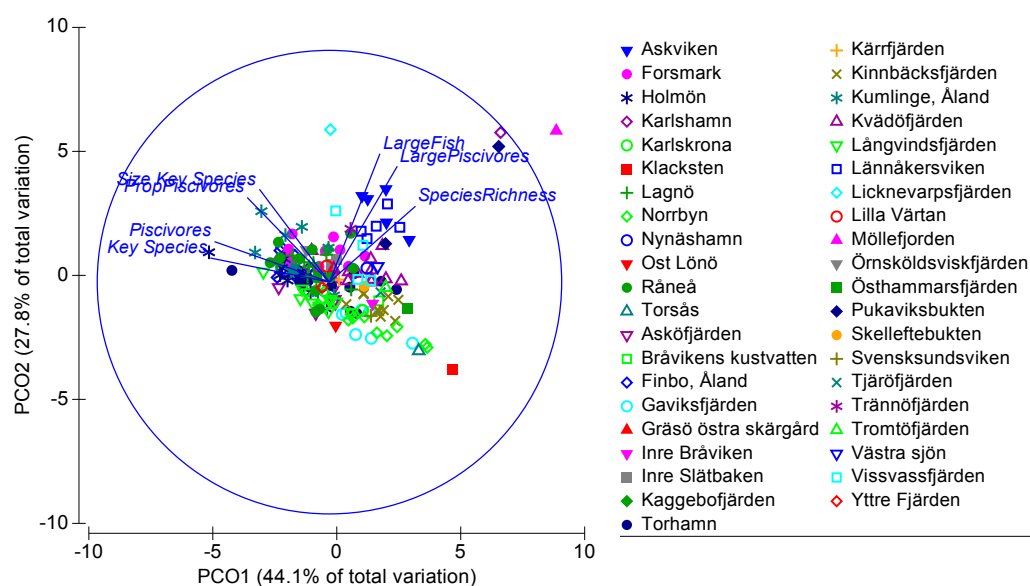


FIGURE 5. Output of the PCO based on indicators. The graph shows the position of data points along the first two axes. Points located close to each other have more similar indicator values than data points located far from each other. Similar symbols represent sampling in the same area during different years. The vectors (blue lines) indicate which indicators contributed most to differences among areas, pointing in the direction of increasing values of that indicator. A longer line indicates a stronger relationship. Together, the first two axes comprised 71.9% of the total variability in the data set (44.1% on PCO1, 27.8% on PCO2). For a more detailed description of the indicators, see table 3.

### Environmental gradients

The main gradients in the environmental data set, when exploring natural environmental and anthropogenic variables together, are shown in figure 6. Several variables contributed to differences among areas. The main differences could be associated with changes in salinity and mean annual temperature. These were correlated with each other, both increasing towards lower latitudes. Other important gradients were changes in water transparency, wave exposure and chlorophyll-a. Water transparency and wave exposure were related to each other, so that water transparency was generally lower in areas with low wave exposure. The variables associated with fish mortality had small influence on the



overall pattern. Of these, the strongest influence was from “Cormorant density”, which was mainly associated with the first axis (not shown in the figure, due to short vector).

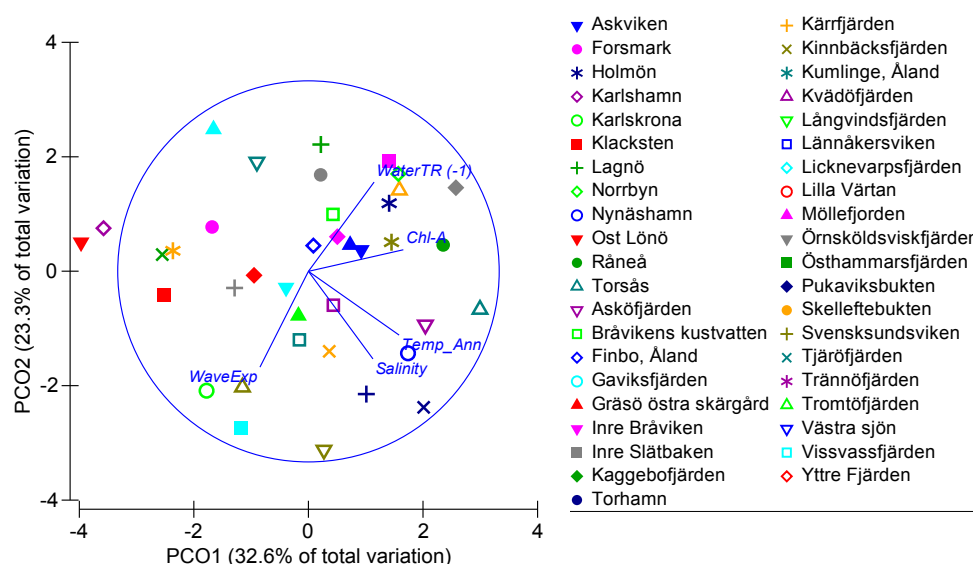


FIGURE 6. Output of the PCO based on natural environmental variables and anthropogenic pressure variables. The graph shows the position of areas along the first two PCO axes. Areas located close to each other have similar environmental characteristics. The vectors (blue lines) show the variables which contributed most to differences among areas, pointing in the direction of increasing values of that variable. A longer line indicates a stronger relationship. Together, the first two axes comprised 55.9% of the total variability in the data set (32.6% on PCO1 and 23.3% on PCO2). For variable abbreviations, see table 3.

## 6.2 Relationship between indicators and environmental data

For most indicators, the ten best models selected by the DistLM analyses were in the same range of AICc values (within 2 units). A more limited number of best alternative models were identified for the indicators “Abundance of key species“ (6), “Abundance of piscivores“ (9), and “Abundance of Cyprinids” (5, Table 5). The identified best models included between 1 and 3 variables for all indicators except for the indicator “Species Richness”. Here, the models included between 3 and 5 variables.

The explanatory level of the best alternative models was smallest for the indicators “Proportion of piscivores”, “Proportion of large fish”, and “Size structure of Key species”, as measured by their average  $R^2$ -values (between 0.08 and 0.10 for each of these indicators). The explanatory level was highest for “Species Richness” (average  $R^2 = 0.49$ ), which also included the highest mean number of variables. For the remaining indicators, the average  $R^2$ -values were between 0.21 and 0.35,

The variables selected by the best alternative models are presented in table 5. All three natural environmental variables were among the four most frequently selected variables. The most frequently selected variable was “Temperature at fishing”, followed by “Salinity”, “Water transparency” and “Wave Exposure”. The variables “Human population density” and “Cormorants” were not among the two most frequently selected variables for any of the indicators.

For two of the indicators, the best alternative models favored a selection of natural environmental variables. This was the “Abundance of piscivores”, which was mainly related to changes in salinity and wave exposure, and “Proportion of large fish”, which was mainly related to all of the three natural environmental variables.

The indicator “Size structure of key species” (Large perch) was related to changes in wave exposure, together with the variable Commercial fishing, which was attribute to fishing mortality. Further inspection of this pair of data revealed that there was a positive relationship, so that commercial catches were higher in areas where the abundances of large perch were also relatively high according to the environmental monitoring.

The other indicators were mainly related to variables attributed to eutrophication, in addition to the natural environmental variables. “Abundance of key species” was mainly related to changes in salinity and chlorophyll-a, whereas “Abundance of Cyprinids”, “Proportion of piscivores”, “Proportion of large piscivores”, and “Species richness”, were related to changes in temperature at fishing as well as water transparency.

The variable salinity was frequently included in the best alternative models, but was the most frequently included variable only for two indicators; “Abundance of Key Species” and “Abundance of piscivores”. It was the third most frequent variable for the indicator “Species richness”. As the variable “mean annual temperature” was omitted prior to the analyses due to its correlation with salinity, the results for salinity may be interpreted as reflecting changes in mean annual temperature, as well. The distribution of single species potentially underlying the observed changes in indicator values over time is presented in figures 7-9.

TABLE 5. Summary of the results of the DistLM analyses. The analysis identifies the explanatory variables that were most strongly related to changes in the indicator values (separated into natural environmental variables and anthropogenic pressure variables relating to eutrophication or to the mortality of fish). The indicators were assessed one at a time. Values give the frequency at which the explanatory variable was selected for inclusion in the best alternative models. The two most frequently selected variables are highlighted for each indicator. Column “N” gives the number of alternative models given by the analyses output (the maximum of models explored was 10).

Indicator	N	Natural environmental		Eutrophication			Mortality of fish		
		Temp- at fishing	Salinity	Wave Exposure	Chlorophyll-a	Water transparency	Commercial Fishing	Human pop. density	Cormorants
Abundance of key species	6	0	<b>100</b>	17	33	17	17	0	0
Size structure of key species	10	20	0	40	0	10	60	0	30
Abundance of piscivores	9	11	<b>89</b>	67	11	11	11	11	11
Abundance of Cyprinids	5	60	20	0	20	100	0	20	0
Proportion of piscivores	10	70	10	20	10	60	30	0	0
Proportion of large fish	10	50	30	30	10	10	10	0	0
Proportion of large piscivores	10	50	30	20	40	70	0	0	10
Species richness	10	100	<b>70</b>	20	0	100	40	10	50

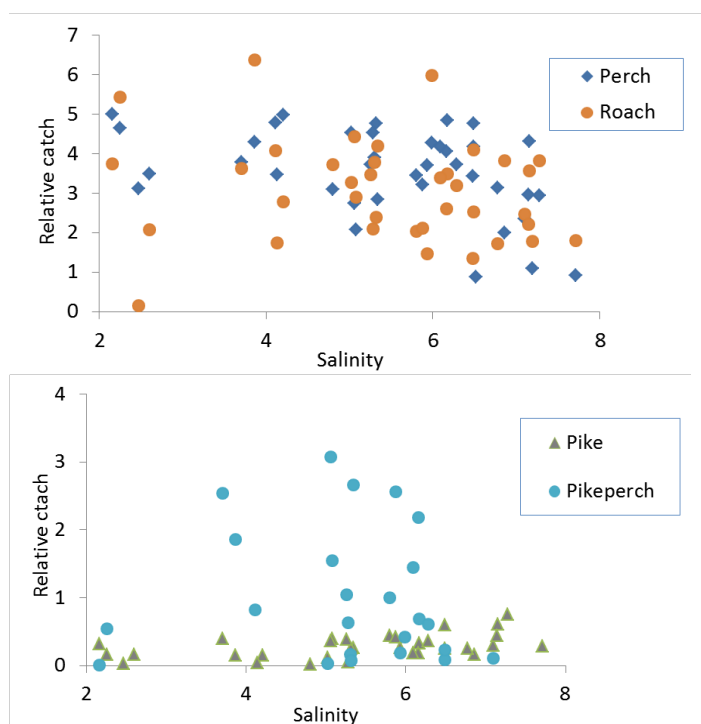


Figure 7. Raw abundances of perch (*Perca fluviatilis*), roach (*Rutilus rutilus*), pike (*Esox lucius*) and pikeperch (*Sander lucioperca*) in relation to salinity. Relative catch is given as average catch per net and fishing night per station. All of these species are included in the computation of indicators. With the exception of roach they are also important target species for commercial and household fisheries in Sweden. The species mainly influence indicators relating to key species (perch), piscivores and large fish (perch, pike, pikeperch), as well as cyprinids (roach). All species are potentially influential on the indicator Species richness and on indicators including proportion estimates.

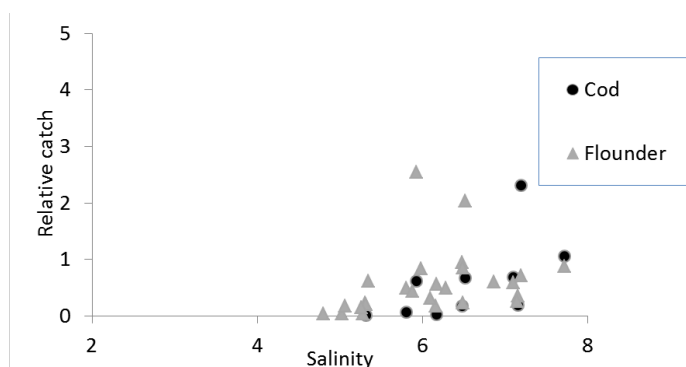


FIGURE 8. Raw abundances of cod (*Gadus morhua*) and flounder (*Platichys flesus*) in relation to salinity. Relative catch is given as average catch per net and fishing night per station. Both species are important target species for fisheries in Sweden and potentially influential on indicators attributed to the abundance or proportion of large fish. Cod also influences on indicators attributed to the abundance or proportion of piscivores. All species are potentially influential on the indicator Species richness and on indicators including proportion estimates.

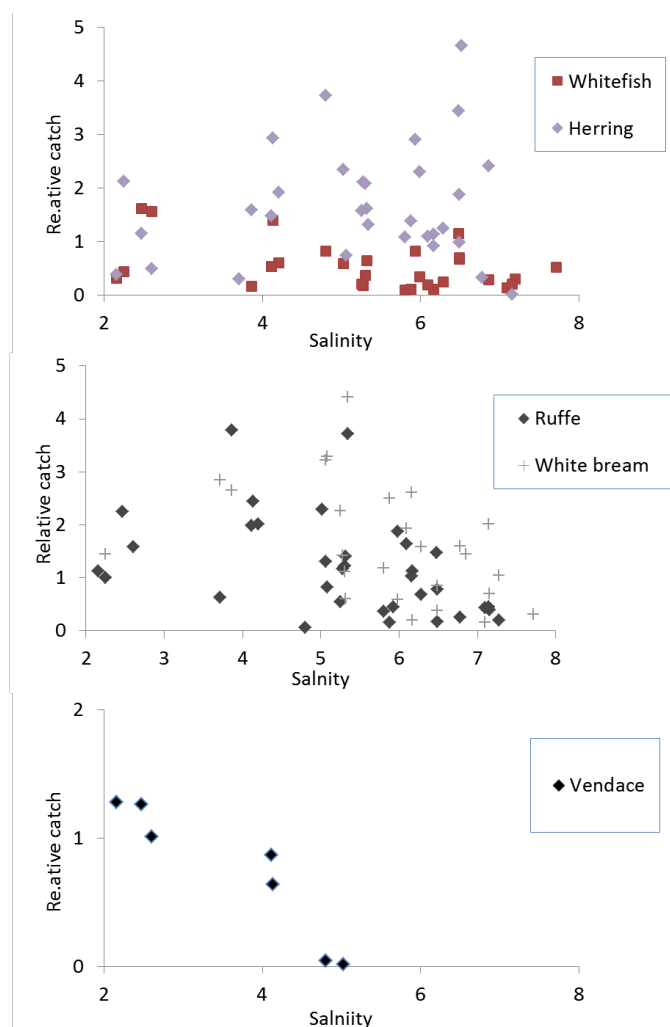


FIGURE 9. Raw abundances of whitefish (*Coregonus maraena*), herring (*Clupea harengus*), ruffe (*Gymnocephalus cernuus*), white bream (*Blicca bjoerkna*) and vendace (*Coregonus albula*) in relation to salinity. Relative catch is given as average catch per net and fishing night per station. Whitefish, herring and vendace are target species for coastal fisheries in Sweden, and mainly influence the indicator "Species richness". White bream is included in the computation of the indicator "Abundance of Cyprinids" and "Species Richness". All species are potentially influential on the indicator Species richness and on indicators including proportion estimates.

## 7. Discussion

The indicator-based assessment of coastal fish communities that is described in this report is in a phase of ongoing development. The present report documents the state of the process at the time of writing (2014), and suggests aspects to focus on in the continued work. In order to support the continued work, the report further investigates the preconditions for developing geographically based assessment, relating indicators to natural environmental gradients, and assessing pressure-state relationships. Studies of large scale geographical variation are essential for widening the range of geographical areas that can be included in an indicator-based assessment. Such an approach is also foreseen to benefit the harmonisation between assessments within the WFD and MSFD, with respect to reporting units. An understanding of the relationship between indicators and environmental variables (natural and anthropogenic) is important for defining ecologically relevant boundaries for good environmental status within these units.

The currently suggested protocol for status assessment is based on analyzing changes in specified monitoring series over time. Monitoring of long term trends is crucial in order to distinguish changes caused by large scale environmental processes from those that can be attributed to human activities at a shorter management scale. It is also important for keeping track of potential shifting baselines. The temporally based assessment is also convenient for identifying ecologically realistic boundary values for good environmental status, as the boundary values are identified empirically within the same monitoring data set. However, assessing environmental status solely based on time series data conveys high costs for monitoring if the density of monitoring areas is to be enhanced to a biologically relevant level. Coastal fish communities in Sweden are currently monitored at least partly only in 14 out of 25 coastal water areas (Figure 2). In addition, the currently running coastal fish monitoring does by definition not include areas subject to direct anthropogenic disturbance. The monitoring is thereby not designed for identifying areas in need of improved management action. The assessment protocol could possibly be extended into additional areas, and make use of additional data sets from inventory studies, if it can be calibrated against the long term monitoring data series.

As also shown in the case study of this report, there are strong differences among areas in species composition. This difference is also seen in the indicators, although to a lesser extent. A key aspect for the continued development of the indicator would be to define indicators in a way that minimizes their sensitivity to changes in species composition, but rather represents structural changes. By achieving this, the geographical relevance of each indicator can be enhanced and variability among areas potentially reduced.

The case study in this report was based on data from the Baltic Sea, because it was possible to obtain the largest set of comparable data with wide geographical coverage from this region. All monitoring initiated later than 2002 follow the same method (Söderberg, 2008), and this method is also widely used in inventories and research studies. Environmental monitoring programs initiated in the Baltic Sea earlier than 2002 (Olsson et al., 2012a, Adjers et al., 2006) were not included, nor data from the Swedish west coast, which is also sampled using other methodology (Bergström et al., 2013). A natural next step for indicator development is to assess similar aspects as highlighted here also for coastal fish communities at the Swedish west coast, with respect to both the temporal and geographical perspective.

The analyses provide an overview over general patterns, and the statistical relationships at the level of individual indicators are to be further defined. Changes in the indicators were to a large extent attributed to gradients in natural environmental variables, such as temperature, salinity and wave exposure. In the analyses, temperature was represented by the variable “temperature at fishing”. The variable “mean annual temperature” was omitted prior to the analyses due to its correlation with salinity. Hence, the results for salinity should be interpreted as reflecting changes in mean annual temperature, as well. The results indicate that all the studied natural environmental variables should be included in a geographically based status assessment.

Variables attributed to anthropogenic pressures were among the top two explanatory variables for six of the eight studied indicators. Five of the cases were related to variables attributed to eutrophication, mainly water transparency. A relationship between fish abundance and water transparency has also been seen in other studies, with respect to roach (Adjers et al., 2006) and pikeperch (Bergstrom et al., 2013). These studies support the observations made here, showing that the indicators “Abundance of cyprinids” (including roach) as well as “Proportion of piscivores” and “Proportion of large piscivores” (including pikeperch) were related to water transparency. In addition, the variable “water transparency” was related to the indicator “Species richness”.

Variables attributed to the mortality of fish were less influential. The strongest relationship was seen between the indicator “Size structure of key species” and commercial fisheries. The direction of the relationship indicated that commercial fishing mainly takes place in areas with high abundance of large perch. Hence, the result was not in line with the expected pressure-state relationship (Table 1), which assumes a negative relationship. The contribution of the other two variables attributed to fish mortality was even lower. From the perspective of indicator development, the results may be interpreted either so that i) the indicators that were assessed were not sensitive to the pressures attributed to fishing mortality, ii) the studied gradient was not strong enough for an evaluating this pressure, or iii) the pressure variables were not quantified in an adequate way. All these aspects need to be considered further in the continued indicator development.

The environmental variables explained a reasonable part of the observed variation in the data set, although a relatively large part of the variation was left unexplained. The unexplained variability may potentially be reduced by using more refined quantitative

analyses, which can also explain variation at different geographical scales. The study was also limited by available environmental data. This was particularly true for data on fishing mortality. A need was seen to update information on the geographical distribution of recreational fisheries and top predators (cormorants, seals), in order to support the assessment of pressure-state relationship, and the potential connection to management measures. In terms of additional potential explanatory variables, habitat quality is often expected to have high influence on species abundances, and hence on indicators (Sundblad and Bergström, 2014, Sundblad et al., 2014). This variable could not be included in the current study, due to a lack of data with sufficient geographical coverage.



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## COASTAL FISH COMMUNITY INDICATORS IN SWEDEN - VARIATION ALONG ENVIRONMENTAL GRADIENTS

Coastal fish communities have a central role in both environmental and fisheries management. This report summarizes the current state of indicator-based approaches for status assessment of coastal fish in Sweden, in relation to international directives. The report is particularly focused on potential connection points between the MSFD and WFD, in order to facilitate the harmonisation of assessments of different ecosystem components and geographical areas. One important aspect would be to develop geographically based assessment methods, to make better use of data from inventory studies. Key aspects for this development are explored in a case study based on data with wide geographical coverage in the Baltic Sea. The study addresses general patterns in the distribution of species and indicators among geographical areas, and explores the relationship between indicators and environmental variables.