Baltic Sea Environment Proceedings No. 129B

Development of a set of core indicators: Interim report of the HELCOM CORESET project

PART B: Descriptions of the indicators

Helsinki Commission

Baltic Marine Environment Protection Commission

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PART B: Descriptions of the indicators

Helsinki Commission
Baltic Marine Environment Protection Commission

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For bibliographic purposes this document should be cited as:
HELCOM, 2012. Development of a set of core indicators: Interim report of the HELCOM CORESET project. PART B: Descriptions of the indicators. Balt. Sea Environ. Proc. No. 129 B.

Design and layout: Bitdesign, Vantaa, Finland

Photo credits: front cover Metsähallitus/Essi Keskinen, back cover Samuli Korpinen, page 5 Samuli Korpinen, page 7 Jannica Haldin, page 19 Christof Herrmann, page 35 Jannica Haldin, page 71 Metsähallitus, page 107 Christof Herrmann, page 156 Samuli Korpinen, page 201 HELCOM photo gallery.

Number of pages: 219

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1. Introduction

This report is Part B of the report "Development of a set of core indicators: Interim report of the HELCOM CORESET project". Part A of the report describes the process of selecting core indicators (HELCOM 2012 [5]). The Part B of the report contains interim descriptions of the proposed core indicators, candidate indicators and supplementary indicators by the CORESET project.

Part A of the report focuses on the description of the selection process of the core indicators and gives only short narrative descriptions of the proposed core indicators whereas Part B aims to give more detailed

5 HELCOM, 2012. Development of a set of core indicators: Interim report of the HELCOM CORESET project. PART A. Description of the selection process. Balt. Sea Environ. Proc. No. 129 A

descriptions of the indicators. The final objective is to use these descriptions as background material for the final core indicator reports that will be placed on the HELCOM website. This report should be seen as a milestone report that presents interim products of the HELCOM CORESET project and provides a more or less broad framework for the core indicators. In some cases essential details of indicator methods or indicator computations are still lacking.

All core indicators were selected on the basis of the HELCOM common principles for core indicators and targets (endorsed by HELCOM HOD 35/2011), the EC decision document 477/2010/EU [6] and additional selection criteria that were defined prior to the process. The main criteria were that the core indicators must be Baltic-wide (including coastal and transitional waters), include a target showing good environmental status, be scientifically sound, reflect or directly measure an anthropogenic pressure, measure a key component of the ecosystem (biodiversity) or a substance with high PBT properties and worrying levels or trends in the sea (hazardous substances). All the selection criteria are given in the Part A of the report. The development of the indicators is still on-going and their classification into core and candidate indicators is still provisional.

The report is structured so that the proposed core indicators for biodiversity are presented in Chapter 2, Chapter 3 provides the proposed core indicators for hazardous substances, Chapter 4 introduces candidate indicators suggested for further development during the project, and in Chapter 5 supplementary indicators identified as supportive indicators for environmental assessments are described. The biodiversity core indicators are, firstly, summarized in a cover page and then more detailed background documentation is presented, giving scientific support for the selection as well as functionality and GES boundary of the core indicator. The core indicators for hazardous substances and their effects have a documentation presenting the selection criteria for each of the indicators and the proposed GES boundary. The documentations also include information of the monitoring and analyses of the indicators.

HELCOM CORESET was not able to finalize the validation of all the proposed indicators that had been identified as potentially important for the assessment of the environmental status of the Baltic Sea by the autumn 2011. These indicators were hence labelled as candidate indicators. Some of the candidate indicators lack GES boundaries, some have methodological challenges and others need compilation of data and further testing. The CORESET expert groups stated that it is important to continue developing them towards operational core indicators.

The report also presents some supplementary indicators for environmental assessments. These are quantitative indicators, which did not fulfil the criteria of core indicators, but are a rich source of supporting information. Some of them measure fluctuations caused by climatic variability. Others can have a linkage to anthropogenic pressures, but they cannot be used as core indicators because they are either not applicable to the entire Baltic Sea or they do not have measurable thresholds required for core indicators. As the HELCOM CORESET project did not focus on developing supplementary indicators, only a few of them have been included in this report.

6 Commission decision of 1 September 2010 on criteria and methodological standards on good environmental status of marine waters (2010/477/EU). OJ L 232/14, 2.9.2010.

2. Proposed core indicators for biodiversity

Core indicators are those indicators discussed and developed in the HELCOM CORESET project that fulfil HELCOM common principles, endorsed by HELCOM HOD 35/2011, and address the ecological objectives of the HELCOM Baltic Sea Action Plan and the qualitative descriptors and criteria of the EU Marine Strategy Framework Directive and the EC decision document 477/2010/EU.

This chapter presents the proposed core indicators for biodiversity including background documentation.

The CORESET expert group on biodiversity has come up with 15 core indicators that will be further developed during the project (**Table 2.1**). Each of these indicators is described below. The proposed core indicators are also discussed in Part A of this report.

Table 2.1. **Proposed core indicators for biodiversity.**
1 Blubber thickness of marine mammals
2 Pregnancy rates of marine mammals
3 Population growth rate of marine mammals
4 White-tailed eagle productivity
5 Abundance of wintering populations of seabirds
6 Distribution of wintering populations of seabirds
7 Fish population abundance
8 Metric mean length of key fish species
9 Fish community diversity
10 Proportion of large fish in the community
11 Abundance of fish key trophic groups
12 Fish community trophic index
13 Multimetric macrozoobenthic indices
14 Lower depth distribution limit of macrophyte species
15 Trends in arrival of new non-indigenous species

2.1. Blubber thickness of marine mammals

2.2. Pregnancy rate of marine mammals

<table>
<tr><td colspan="2">1. Working team: Marine mammal team
Authors: Britt-Marie Bäcklin, Charlotta Moraeus, Mervi Kunnasranta and Marja Isomursu
Aknowledged persons: Stefan Bräger, Anders Galatius, Britta Knefelkamp, Anna Roos, Ursula Siebert and Stefanie Werner as well as Members of the HELCOM SEAL EG.</td></tr>
<tr><td>2. Name of core indicator
Health status measured by blubber thickness and pregnancy rate</td><td>3. Unit of the core indicator
- millimetre blubber layer, and
-% pregnant mature animals</td></tr>
<tr><td colspan="2">4. Description of proposed indicator
Blubber thickness is a commonly used method to describe the nutritional state of marine mammals. The sternum blubber thickness in Baltic seals has been measured, mainly in by-caught seals, since the 1970s. Pregnancy rate in sexually mature female seals during the pregnancy period have been noted since the 1970s in by-caught, and later also in hunted Baltic seals. Blubber thickness and pregnancy rate have also been noted in harbour porpoises.</td></tr>
<tr><td colspan="2">5. Functional group or habitat type
Toothed whales and seals</td></tr>
<tr><td colspan="2">6. Policy relevance
MSFD GES criteria 1.3 (Population condition) of 2010/447/EU.
HELCOM recommendation on Conservation of seals in the Baltic area 27-28/2 2006-07-08. HELCOM-Baltic Sea Action Plan: partially addressed in “Indicators and targets for monitoring and evaluation of implementation of the Baltic Sea Action Plan (BSAP)” under Hazardous Substances. Listed in Annexes II and V of the Habitats Directive.</td></tr>
<tr><td colspan="2">7. Use of the indicator in previous assessments
Harbour porpoises and Seal health monitoring programs, HELCOM indicator fact sheets, ASCOBANS, e.g. in the Jastarnia Plan (2002 & 2009)</td></tr>
</table>

8. Link to anthropogenic pressures Fishing causing changes in the food web, ecosystem changes (food web, introduction of pathogens and non-indigenous species), contaminants, climate change, pressure from other anthropogenic activity e.g. underwater noise.
9. Pressure(s) that the indicator reflect Hazardous substances, Biological disturbance, Ecosystem changes, Fishing, Climate change
10. Spatial considerations Assessment should be carried out in accordance to management units defined in HELCOM RECOMMENDATION 27-28/2 i.e. 1) harbour seals in the Kalmarsund region (Sweden);2) Southwestern Baltic harbour seals (Denmark, Germany, Poland, Sweden); 3) Gulf of Bothnia ringed seals (Finland, Sweden); 4) Southwestern Archipelago Sea, Gulf of Finland and Gulf of Riga ringed seals (Finland, Estonia, Latvia, Russia); 5) Baltic Sea grey seals (all Contracting Parties to the Helsinki Convention).
11. Temporal considerations In addition to periodic assessments, the indicators should be followed as time trends
12. Current monitoring Harbour porpoise and seal health monitoring programs, HELCOM indicator fact sheets
13. Proposed or perceived target setting approach with a short justification. **Blubber layer in seals:** Geometric mean blubber thickness above a reference level (=GES) measured in the species specific pregnancy period (seals) in juvenile and adult (< 21 years) Baltic seals. The GES limit will be different between juveniles and adult male and females and between species. The years 1999-2004 is used as a reference state for a healthy population. Blubber thickness is also recorded in harbour porpoises, ringed seals and harbour seals; although a low number of mature animals are investigated. More knowledge is needed about the season for measure in harbour porpoises and normal blubber thickness in harbour porpoises, ringed and harbour seals before proposing them as core indicators. **Pregnancy rate in seals**: per cent females with the presence of a foetus after the delayed implantation period in sexually mature seals (4-20 years old). The pregnancy rate in grey seals in 2008-2009 is used as a reference state for a healthy population. The levels that can be considered to represent GES for ringed and harbour seals and harbour porpoises remain to be compiled or investigated.

Introduction

Several health parameters in marine mammals are investigated by the HELCOM Contracting Parties (CPs). The significance or cause of some pathological findings remains to be looked into. Furthermore, Baltic countries have different possibilities and access to conduct marine mammal necropsies. Therefore pregnancy rate and blubber thickness have been prioritised to roughly reflect the health in marine mammals since they are routinely measured in several CPs (see section on monitoring). Agents that are lethal to foetuses or endocrine disrupting causing a decreased pregnancy rate, and also agents or starvation causing a thin blubber layer, could seriously affect the survival of the population. The species considered here are grey seals (*Halichoerus grypus*), ringed seals (*Pusa hispida botnica*), harbour seals (*Phoca vitulina*) and harbour porpoises (*Phocoena phocoena*).

Blubber thickness

The thickness of the blubber layer is important for the individual survival in marine mammals and in females also for the survival of their offspring. A seasonal difference in blubber thickness with a decrease during the reproduction, lactating and molting periods in the spring and an increasing blubber thickness towards the autumn has been described for adult seals in many studies (Nilssen et al. 1997, Sparling et al. 2006, Hauksson 2007). The mean autumn/winter blubber thickness has decreased significantly in Baltic grey seals since the beginning of 2000s, especially in 1-4 year-old seals from by-catch and hunt (Bäcklin et al. 2010). This decreasing trend has also been observed in young Baltic ringed seals (Kunnasranta et al. 2010). There are also data of blubber thickness in harbour seals and maybe in harbour porpoises that needs to be compiled. There could be several reasons for a thin blubber layer in the autumn/winter season e.g., disease, contaminants, decreased fish stocks and change in diet, or a change in the quality of the diet. The reason for the decreasing trend in blubber thickness in seals is unknown but so far no correlations to disease have been found.

The blubber thicknesses and the pregnancy rates of marine mammals can be obtained from institutional necropsies or hunters. By sampling the female reproductive organs (reproductive status), the lower jaw (age determination) and measuring the sternum blubber thickness and reporting the date of death, and sending it to an institute, it should be possible to collect more data than at present.

Pregnancy rates in Baltic grey seals and ringed seals

During the 1970s and the first half of the 1980s, uterine obstructions causing sterility were commonly found in necropsied Baltic grey seals and Baltic ringed seals (Helle et al. 1976a,b; Olsson 1977; Bergman & Olsson 1985). Since then, the pregnancy rate in the examined Baltic grey seals has increased from 9% (1977-1986) to 60% (1987-1996) (Bergman 1999). In 2008-2009 the pregnancy rate was 88% in 4-20 year old grey seal females. The most recent case of uterine obstruction in grey seals was found in 1993 in Sweden. In the 2000s, about 20% of the examined Baltic ringed seals still suffer from uterine obstructions and the pregnancy rate of 68% in ringed seals in 2001-2009 is therefore probably lower than 'normal' (Helle et al., 2005, Kunnasranta et al. 2010). The low gynaecological health among the Baltic seals is most probably explained by high concentrations of polychlorinated biphenyls (PCBs) (Helle et al. 1976a, b; Bredhult et al. 2008). No observations of uterine obstructions in Baltic harbour seals or harbour porpoises have been reported, but there are data on pregnancy rates in these species that needs to be compiled and evaluated.

Policy relevance

The policy relevance for the indicator "health in marine mammals" is described under criterion 1.3 (Population condition) of 2010/447/EU, the HELCOM recommendation on Conservation of seals in the Baltic area 27-28/2 2006-07-08 with the long-term objective of attaining a health status that secures the continued existence of the populations. In the HELCOM Baltic Sea Action Plan, mammals are partially addressed in the Other Documents -section under Hazardous Substances. The three seal species that are present in the Baltic Sea and harbour porpoise are listed in the Habitats Directive Annexes II and V.

Methods to evaluate the core indicators

Pregnancy rate

Pregnancy rate is measured as the presence or absence of an embryo or foetus during the pregnancy period in mature females (presence of an ovarian C*orpus luteum*). It is expressed as the percentage of pregnant females in all mature females (age 4-20 years in seals) and it is investigated during the relevant season(s).

Grey seals. It is estimated that age-specific birth rates increase steeply from the age of four to six (Hamill & Gosselin 1995). The birth rates for the six-year old females in the Northwest Atlantic, British, Norwegian and Baltic populations ranged from 60-91%. In a sample of 526 female grey seals from the Northwest Atlantic, pregnancy rates were estimated from the presence/absence of a foetus. The pregnancy rate for the Northwest Atlantic population was relatively stable at about 90% after the age of six (Hamill & Gosselin 1995; Harding et al. 2007). In the Baltic grey seal population, the pregnancy rate was 88% in 4-20-year old females in 2008-2009 (**Figure 2.1**). This pregnancy rate seems to be normal in the 4-20-year old Baltic grey seals and indicates that the population is healthy (**Figures 2.1 and 2.2**). This rate is also close to the pregnancy rate of Northwest Atlantic grey seals older than five years.

The pregnancy rate for the 4-5-year old individuals is 65% and for the 6-20-year old individuals is 95.5% among grey seals caught via hunting and as by-catch in 2002-2009 in Sweden (**Figure 2.2**).

Annually, Sweden does not receive more than 4-13 grey seal females between 4 and 20 years of age in the pregnancy period.

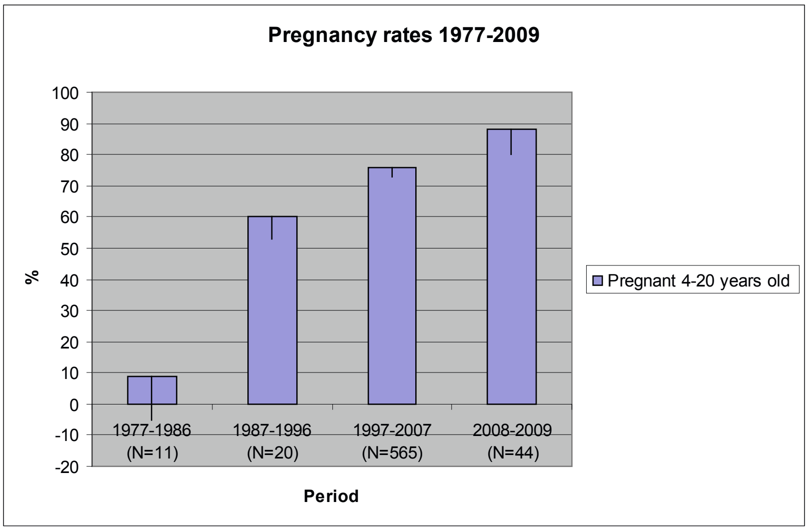

Figure 2.1. *Pregnancy rates, mean values and one-sided 95% confidence intervals for a proportion, in 4-20-year old female Baltic grey seals (August to reproductive season). Finnish data is included in the period 1997-2007.*

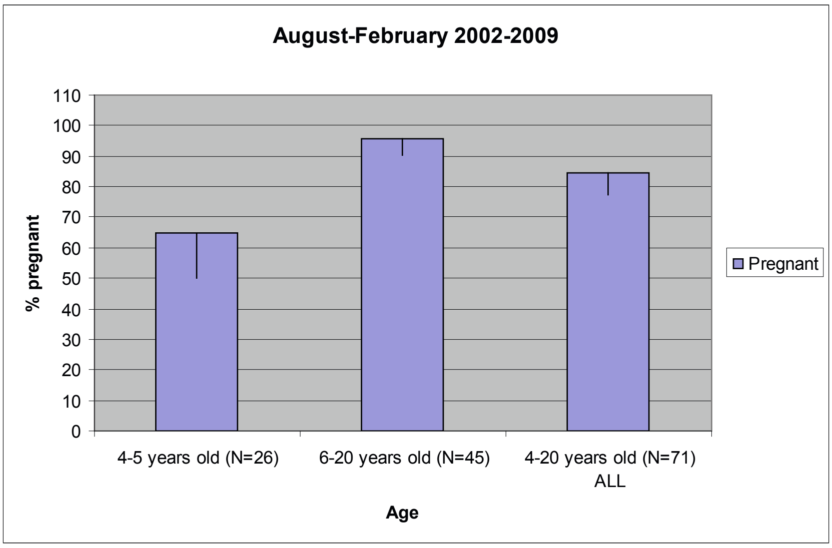

Figure 2.2. *Pregnancy rates, mean values and one-sided 95% confidence intervals for a proportion, in examined 4-20-year old grey seal females from by-catch or hunt in Sweden 2002-2009, divided into 4-5-year old and 6-20-year old seals.*

Ringed seals. The number of 4-20-year old Baltic female ringed seals and Baltic harbour seals (Kalmarsund population) that are investigated annually during the pregnancy period is very small. In **Figure 2.3**, pregnancy rate of a total number of 19 ringed seals examined 1981-2009 is shown. The pregnancy rate in ringed seals was 68% in 2001-2009 (limited sample size) compared to 84.5% in grey seals. Also, the ringed seals are still suffering from uterine occlusions. Data from harbour seal and harbour porpoise investigations remains to be compiled and evaluated.

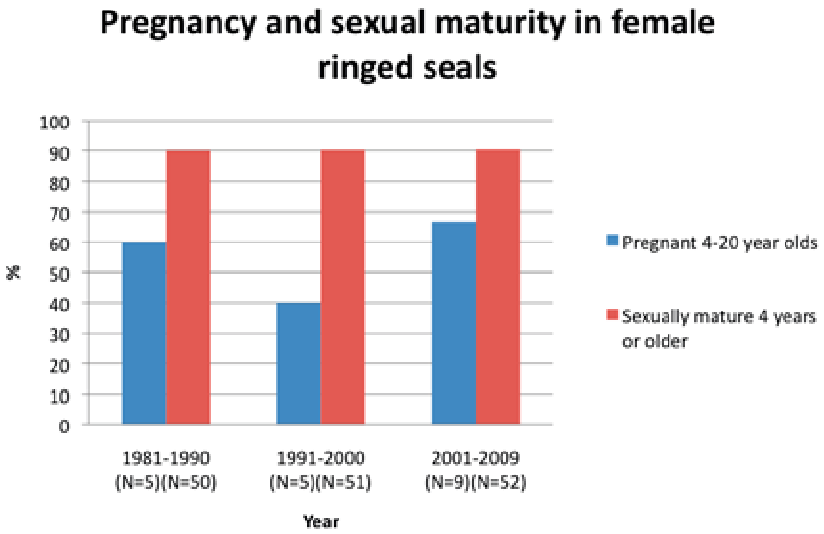

Figure 2.3. *The prevalence of pregnant, and sexually mature (presence of Corpus luteum) females in examined Baltic ringed seals (4 years or older) in Finland and Sweden. Data of prevalence of pregnancies was limited to samples acquired in the implantation period between August to February (Kunnasranta et al. 2010).*

Blubber thickness

In 1977-2002, blubber thickness in seals necropsied at the Swedish Museum of Natural History (SMNH) was only measured ventrally at three sites (either sternum, belly and hips, or neck, sternum and hips) between the muscle layer and the skin. Therefore, at SMNH, only the sternum blubber thickness of seals measured today is comparable with earlier data. Two questions have been addressed when evaluating blubber thickness as core indicator:

– Does the sternum blubber thickness reflect the nutritional status/body condition of the animals?
– What blubber thickness could be considered to be normal?

Investigations of blubber thickness in ringed and grey seals conducted at SMNH and a survey of published data are summarised below.

LMD-index. Ryg et al. (1990) tested a method to estimate the total blubber content of a seal as a percentage of the body weight (LMD-index) in five seal species (phocids). The investigation was performed on shot or by-caught seals and the blubber of 132 ringed seals, 8 bearded seals, 38 grey seals, 20 harp seals and 3 harbour seals was measured and weighed. The results showed that % blubber of the body weight was equal to $4.44 + 5693 \sqrt{L/M} \times d$, and SE = 3%, where L is body length in meters (nose to tail), M is the body mass in kg, and d is the xiphosternal (a site located dorsally at 60% of the body length from nose) blubber thickness in meters.

At SMNH the % blubber of the body weight has been tested using the mathematical model from Ryg et al. (1990). The results were compared with the "real" weight of the blubber as % of the body weight in two ringed seals and one grey seal. For these three seals, the calculated LMD-index was similar to the weighed % blubber of the body weight (**Table 2.2**). The modest experiment also showed that the LMD-index is a good method for calculating % blubber in both ringed and grey seals, if body length, body weight and the xiphosternal blubber thickness are known.

Table 2.2. Calculated % blubber (LMD-index) and respective factors used for calculations (from Ryg et al. 1990).						
Seal	**Length m**	**Body weight kg**	**Blubber m**	**Blubber weight kg**	**% Blubber of body weight**	**Calculated % blubber (LMD)**
Ringed	1,25	66,3	0,055	30,7	46	47
Ringed	1,08	23,4	0,009	3,5	15	15
Grey	0,98	21,9	0,013	4,2	19	20

LMD-index and sternum blubber thickness. At SMNH, the relation between the sternum blubber thickness and the LMD index (calculated with the xiphosternal blubber thickness) has also been investigated in Baltic ringed and grey seals. The measured sternum blubber thickness was positively correlated with the calculated LMD-index (**Figures 2.4 and 2.5**). Thus, the results indicate that the sternum blubber thickness is a good indicator for the nutritional status/body condition in ringed and grey seals.

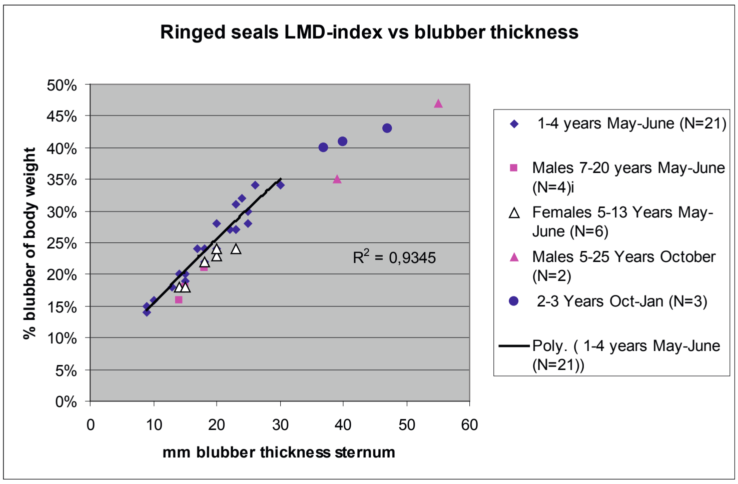

Figure 2.4. *Sternum blubber thickness (mm) in Baltic ringed seals from hunt in relation to percentage blubber of the body weight (LMD-index). –1-4-year olds include both males and females. Most of the animals were shot in the spring (thinnest season). N= number of investigated ringed seals.*

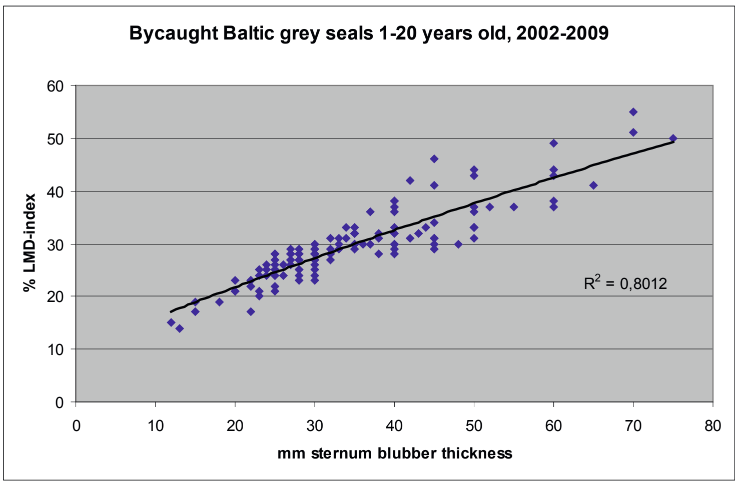

Figure 2.5. *Sternum blubber thickness (mm) in by-caught Baltic grey seals in relation to percentage blubber of the body weight (LMD-index). Trend line is polynomial.*

Seasonal changes in blubber thickness. In order to avoid measuring seals that have starved due to natural causes (e.g. poor teeth due to old age or poor survivors due to young age), it is suggested that only seals that are 1-20 years old are included in the assessments of blubber thickness. The blubber layer in the mature ringed and grey seals fluctuates with season and is low after the reproductive season. The intention is to measure how well seals have managed to gain blubber after the reproductive season, and hence the measuring period is suggested to be the autumn/winter season. In order to investigate in which month the blubber thickness starts to increase, a mean value was calculated for each month, sex and age class in grey seals from hunt[7] . It seems that the blubber layer is thickest between the pregnancy period (August-February) (**Figure 2.6, Table 2.3**). The data presented in **Figure 2.6** represent measurements done by the hunters, who were provided with instructions, and the sternum blubber thickness has thereby been measured by different people using different instruments.

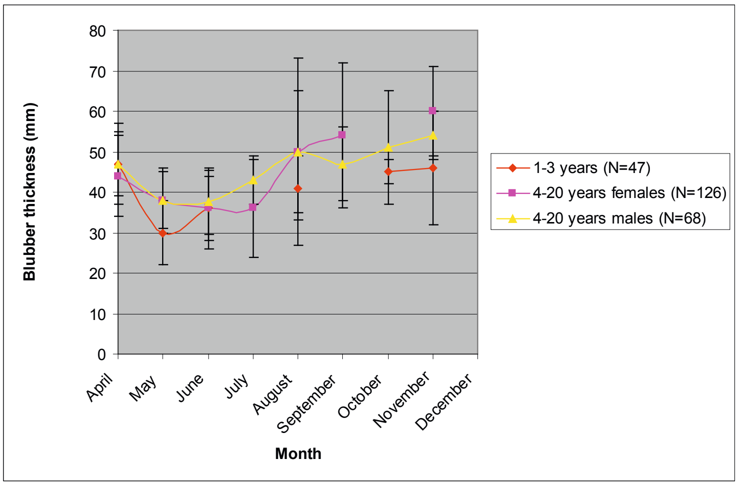

Figure 2.6. *Mean blubber thickness (mm) ± SD of at least 3 individuals per month in Baltic grey seals from hunt, 2002-2006. N= total number of animals measured.*

Table 2.3. Number of measured animals each month in Figure 2.6.

Age years/sex	April	May	June	July	Aug	Sept	Oct	Nov
1-3	3	15	10	-	10	-	5	4
4-20 /females	3	65	32	7	11	4	-	4
4-20 /males	9	11	8	3	5	8	14	10

Normal blubber thickness in grey seals. Beside Swedish data, data of sternum blubber thickness in grey seals was kindly provided by the UK (P.D. Jepson) and Norway (K.T. Nilssen) and comparisons were made in the pregnancy period[8] of animals examined in and before 2004 (**Table 2.4**). It should be noted that the available data include animals with different causes of death (stranded, shot or by-caught).

Assuming grey seals from hunt represent a fairly random sample from the population; geometric mean[9] blubber thicknesses with confidence intervals were calculated to represent reference levels from Norwegian and Swedish grey seals from hunt 1999-2004 (Table 3).

Pregnant grey seals, Farne Islands. Boyd (1984) made sternum blubber thickness measurements on female grey seals around the time of implantation. The mean ± SEM in females with implantation in progress was 36 ±3.5 mm. For females with a fully implanted embryo it was 46 ± 2.5 mm. These results were based on dissections of 72 shot adult grey seal females; however the number of investigated females was not given for the means.

7 Since 2001, Swedish hunters have sent the inner organs, lower jaws, a piece of blubber with skin, and data on body length, sternum blubber thickness and date of death from between 80-110 Baltic grey seals per year.
8 In UK and Norway; March-September and in the Baltic, August-February
9 Data is not normally distributed

Table 2.4. Summary of the geometric mean blubber thicknesses and the 95% confidence interval in grey seals from Norway, the UK and Sweden, during the pregnancy period. GM= geometric mean, CI= 95% confidence interval and N= number of grey seals.									
Country	**1-3 years old**			**5-20 years old or > 170 cm males**			**5-20 years old or > 170 cm females**		
	mm GM	mm CI	N	mm GM	mmCI	N	mm GM	mm CI	N
Norway hunt (1999-2004	24	21-26	24	34	29-39	25	36	30-44	18
Sweden hunt (2002-2004)	42	34-51	13	52	46-60	16	57	49-68	11
Sweden & Norway hunt (1999-2004)	29	26-32	37	40	36-45	41	43	37-50	29
Sweden by-catch (2002-2004)	34	32-36	22	41	35-48	13	*)		
UK stranded (1990-2004)				43	36-50	8	49	35-64	8

*) no available by-caught 5-20 years old females in 2002-2004 during the pregnancy period.

Ringed seals. The number of investigated ringed seals in the autumn/winter season is rather small but there are some data (**Table 2.5**) available in Kunnasranta et al. (2010). The period showing the thickest blubber layer was 1991-2000, but there are only 7 seals from that period.

The blubber thickness in harbour seals and harbour porpoises remains to be compiled and evaluated.

Table 2.5. Mean blubber thickness (cm, sternum) in examined 1-3 yr old and 4-20 yr old Baltic ringed seals during spring (January-June) and fall (July-December) in Finland and Sweden (Kunnasranta et al., 2010).												
	1-3 years old						**4-20 years old**					
Period	**Spring**	**SD**	**N**	**Fall**	**SD**	**N**	**Spring**	**SD**	**N**	**Fall**	**SD**	**N**
1981-1990	2,54	0,6	5	4,00	-	1	3,43	1,3	19	5,30	2,3	5
1991-2000	3,96	0,9	12	4,70	0,4	2	3,44	0,9	61	5,94	2,4	5
2001-2009	2,41	0,8	34	3,94	0,5	7	3,20	1,3	63	5,73	0,9	16

Age determination

Age determination in seals is performed by examination of the annual growth pattern (GLGs) in cementum zones in tooth sections (*Hewer 1964*). The method is modified for harbour seals (Dietz et al. 1991) and is also used when examining ringed seals and harbour porpoises, however in harbour porposes the annual growth pattern is examined in the dentine.

Approach for defining GES

Pregnancy rate suggestions

Pregnancy rate is measured as presence or absence of a foetus in the pregnancy period in 4-20-year old seals. GES data proposed to be assessed every third year (pooling the data for each 3-year period) for 4-20 years old, and every sixth year pooling the data for each 6-year period, separately for young and adult females. Today's figures suggest that GES in 4-20 years old could be set at the lower limit of the 95% confidence interval i.e at about 80%, referring to the period 2008-2009 which is proposed to be defined as representative of a healthy population in **Figure 2.1**. Data should also be presented as trends.

Whether or not similar GES limit for pregnancy rate can be suggested for harbour seals, ringed seals and harbour porpoises remain to be investigated.

Blubber thickness suggestions

Blubber thickness is measured at the sternum between the muscle layer and the skin during the season of pregnancy (August-February for grey and ringed seals). Suggested reference levels for GES are the lower limit of the 95% confidence interval for the geometric mean. These have been calculated for 1-3 years old, 5-20 years old males, and 5-20 years old females in the Norwegian and Swedish grey seals from hunt in 1999-2004 (**Table 2.4**). The reason for basing the proposed GES boundary to data from before 2005 is that since this year the available data indicates a trend of decreasing blubber thickness.

Suggestion GES boundaries for grey seals during the season of pregnancy from stranded, by-caught or hunted animals (from **Table 2.4**).

Age class	Sex	GM – CI = GES boundary
1-3 years	females and males	≥26
5-20 years	males	≥36
5-20 years	females	≥37

Data should also be presented as trends in blubber thickness (separated into by-catch, hunt and age classes) with geometrical means and 95% confidence intervals (**Figure 2.7**). Whether the upper or lower limit of the confidence interval or the geometrical mean should be below the GES boundary for assessment of sub GES is to be discussed. Data could probably be presented every third year (i.e. pooling the data for each 3-year period) for grey seals.

In the Baltic, the causes of death have been shown to influence the result of the blubber measurements. Stranded seals often show a thin blubber layer (starvation due to disease or old age) and by-caught seals are often thinner than seals received from hunt (Bäcklin et al. 2010, 2011). Therefore, these groups are suggested to be presented separately (**Figure 2.7**) since their proportions will influence the GES determination. However, the comparisons of data from stranded (exceeding 25 mm), shot or by-caught grey seals from different countries in **Table 2.4**, did not reveal big differences (no data from 1-3 year old animals).

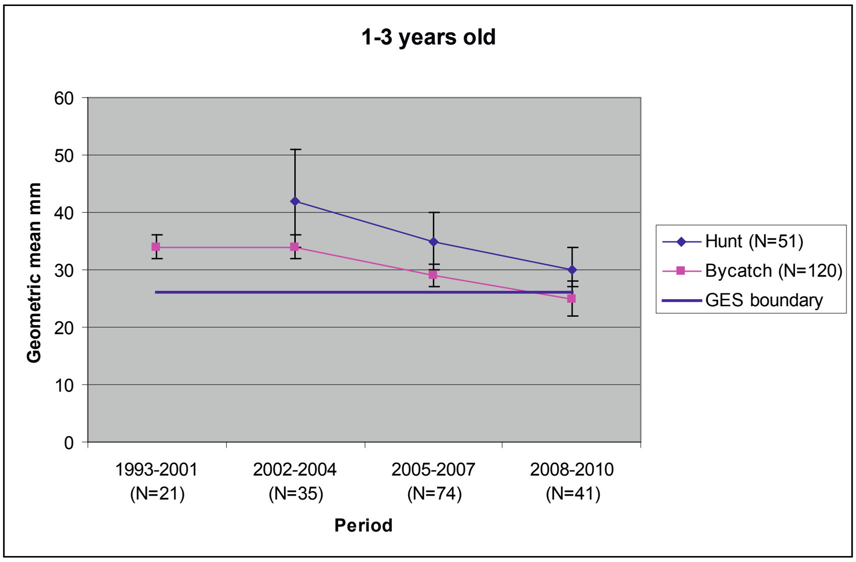

Figure 2.7. *The geometric mean blubber thicknesses with 95% confidence interval in four periods in examined 1-3 years old by-caught (1993-2010) and hunted (2002-2010) grey seals in Sweden. All were by-caught or shot between August and February. The proposed GES reference level (26 mm) is shown in blue. N is the number of investigated animals*

Using proposed GES boundaries for geometric mean blubber thickness, assessment of the period 2008-2010 (**Figure 2.7**) reveals that the environmental status is good for the hunted 1-3 years old. The by-caught 1-3 years old geometrical mean is sub-GES.

GES limits for blubber thickness in ringed seals and harbour seals are still to be considered or investigated as well as for harbour porpoises.

Existing monitoring data

Health of the Baltic marine mammals is investigated in Finland, Lithuania, Poland, Germany, Denmark and Sweden.

Table 2.6. Monitoring of the proposed indicators in the Baltic Sea. Information from several countries is missing.

Country	Area	Coastline	Species	Month	Interval	Type of carcass	Start of data series
Germany	Western Baltic Sea	Hiddensee Westküste	Harbour porpoise			stranded	
Germany	Mecklenburg-Western Pomerania	Bay of Meck-lenburg & Pomeranian Bay, internal lagoons	Harbour porpoise	All	always	stranded and by-caught	1990
Lithuania	Southeastern Baltic sea	Lithuania coastline					
Sweden	whole Baltic Sea	Swedish	Grey seal	All	always	by-caught, stranded, hunt	1977
Sweden	Baltic proper	Swedish	Harbour seal	All	always	by-caught, stranded	1977
Sweden	Western Baltic Sea	Swedish	Harbour seal	All	always	by-caught, stranded, hunt	1977
Sweden	whole Baltic Sea	Swedish	Ringed seal	All	always	by-caught, stranded	1977
Sweden	Western Baltic Sea and Baltic proper	Swedish	Harbour porpoise	All	always	by-caught, stranded	1977
Finland	Baltic Sea	Finnish	Grey seal	16.April-December	always	hunted	1998
Finland	Baltic Sea	Finnish	Ringed seal	16.April-December	always	hunted	2010
Finland	Baltic Sea	Finnish	Grey seal	All	always	by-caught	1999
Finland	Baltic Sea	Finnish	Ringed seal	All	always	by-caugh	1999
Finland	Baltic Sea	Finnish	Grey seal	All	sporadic	stranded	2010
Finland	Baltic Sea	Finnish	Ringed seal	All	sporadic	stranded	2010

Weaknesses/gaps

Monitoring of the Baltic marine mammals started in the 1970s when the health of the seal populations was seriously threatened by contaminants, especially organochlorine. The populations have slowly recovered but new threats have arisen (e.g. other contaminants). Therefore, it could be said that the knowledge of normal pregnancy rate and blubber thickness is limited in Baltic marine mammals. The “point of no return” for blubber thickness has not been reached according to any report. There is some evidence that historically the blubber layers in the Baltic grey and ringed seals were thicker and the pregnancy rates were lower. If this is the case, it would be appropriate to use older data (before and early 2000s) for normal blubber thickness and more recent data for normal pregnancy rate.

Data from outside the Baltic could be used to determine normal limits but the ecosystem outside the Baltic Sea is different with different opportunities to forage. In the Baltic, grey seals also have a smaller body size than in the northeast Atlantic (UK and Norway) which in turn are smaller than in the northwest Atlantic (McLaren 1993). The proposed GES boundaries for blubber thickness is partly based on data measured by different Swedish hunters compared to data from by-caught grey seals that have been measured by the SMNH. In order to investigate the accuracy of the blubber thickness measurements made by hunters, an additional measurement on 37 blubber samples was made at the SMNH in 2005, if skin; blubber and muscle layer was visible in the sample. The means of the measurements did not differ significantly (42, 4 ± 9, 6 vs. 42, 2 ± 10, 4) between the hunters and SMNH. This indicates that the mean measurements of blubber thicknesses were comparable (Bäcklin et al. 2011).

There is a lack of data, especially for ringed seals. Data from investigations on the western population of harbour seals could probably serve as normal data also for determine GES in the Kalmarsund harbour seal population.

It is important to combine population and distribution investigations for the evaluation of the significance of decreased pregnancy rate or mean blubber thickness.

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2.3. Population growth rate of marine mammals

1. Working team: Marine mammal team *Author*: Tero Härkönen *Aknowledged persons:* Britt-Marie Bäcklin, Stefan Bräger, Anders Galatius, Kaarina Kauhala, Britta Knefelkamp, Charlotta Moraeus, Anna Roos, Ursula Siebert and Stefanie Werner as well as Members of the HELCOM SEAL EG.	
2. Name of core indicator Population growth rate of marine mammals	*3. Unit of the core indicator* Population growth rate (% per year)
4. Description of proposed indicator Marine mammals are top predators of the marine ecosystem and good indicators for the state of the food webs, hazardous substances and direct human disturbance, such as hunting and habitats loss. Deviations from the maximum rate of population growth during the phase of exponential increase are indicative of that the population is reaching its carrying capacity or is affected by human impacts in form of excessive mortality or impaired fertility. Near or in the carrying capacity, the population fluctuates but a continuous decline indicates that the population is not in GES.	
5. Functional group or habitat type Seals and toothed whales	
6. Policy relevance Descriptor 1, criterion 1.2 Population size Descriptor 4, criterion 4.1 Productivity of key species or trophic groups Descriptor 8, criterion 8.2 Effects of contaminants Marine Strategy Framework Directive, and a number of IGO resolutions (e.g., HELCOM, OSPAR, CMS, ASCOBANS etc.) The Baltic Sea Action Plan provides the following target: ""By 2015, improved conservation status of species included in the HELCOM lists of threatened and/or declining species and habitats of the Baltic Sea area, with the final target to reach and ensure favourable conservation status of all species""	
7. Use of the indicator in previous assessments Seal monitoring programs, ASCOBANS, e.g. in the Jastarnia Plan (2002 & 2009). ICES MME: Development of Ecological Quality Objectives (EcoQO:s)	
8. Link to anthropogenic pressures The growth rate of the seals and marine mammals has a clear linkage to anthropogenic pressures.	
9. Pressure(s) that the indicator reflect Hunting, by-catches of fisheries, environmental pollution	
10. Spatial considerations The indicator is expected to vary spatially and among species. The assessment units for the indicator depend on species (HELCOM Recommendation 27-28/2): for grey seal it is the entire Baltic Sea, for ringed seal three areas, for harbour seal two areas and for harbour porpoise there are two populations which should be assessed separately.	
11. Temporal considerations Indicators can show long-term trends. Annual counts are required.	
12. Current monitoring Monitoring coordinated among Baltic Sea countries	
13. Proposed or perceived target setting approach with a short justification. Long-term objectives of the 2006 HELCOM seal recommendation: Natural abundance and distribution. Population growth rate should be positive until hampered by natural limitations. Proposal for GES boundary is given below.	

Introduction

Several international initiatives have suggested means to measure the environmental quality of marine ecosystems. The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) has been ratified by all North Sea countries. This convention lists a number of Ecological Quality Objectives (EcoQOs) for the North Sea, which were developed in collaboration with the International Council for the Exploration of the Sea (ICES) and aim to define a desirable state for the North Sea. EcoQOs have been developed for some components of the ecosystem, e.g. commercial fish species, threatened and declining species, and marine mammals. An EcoQO is a measure of real environmental quality in relation to a reference level where anthropogenic influence is minimal. The ecological quality elements "population trends" and "utilization of breeding sites", which have been suggested for marine mammal populations, may serve as suitable tools for evaluating current population status. The term "population trend" is defined for this purpose as a change in abundance of a population, increasing or decreasing within a specified area over a certain number of years. The EU Water Framework Directive (WFD) includes status categories for coastal waters as well as environmental and ecological objectives, whereas the EU Habitats Directive (European Commission 1992) specifically states that long-term management objectives should not be influenced by socio-economic considerations, although they may be considered during the implementation of management programmes provided the long-term objectives are not compromised. In line with both the OSPAR Convention and the Marine Strategy Framework Directive, the Helsinki Commission (HELCOM) in its HELCOM CORESET project is developing a framework using indicators for the Baltic ecosystem. All seals in Europe are also listed under the EU Habitats Directive Annex II (European Commission 1992), and member countries are obliged to monitor the status of seal populations. Consequently, the Coreset core indicator "Population trend" is similar to the EcoQ element with the same name in the ICES and OSPAR frameworks, with the distinction that two latter EcoQ:s include "No decline in population size or pup production exceeding 10% over a period up to 10 years" for populations "minimally affected by anthropogenic impacts". We suggest this condition to be appropriate also for the Coreset indicator "Population trend" when seal populations are close to natural abundances.

The OSPAR and ICES frameworks provide some guidance also for populations far below "natural" or "pristine" abundances. Applying the term "anthropogenic influence is minimal" would imply that a population should grow close to its intrinsic rate of increase when not affected by human activities. The theoretical base for this measure is outlined below and compared with empirical data from seal populations.

Approach for defining GES for populations below carrying capacity

Long term maximum growth rates in seals

The maximum rate of population growth is limited by several factors in grey seals and ringed seals. Females have at most one pup a year, and first parturition occurs at about 5.5 years of age. It is also evident that not all adult females bear a pup each year, especially not young females (Pomeroy et al. 1999, Bäcklin 2011). An additional limitation for the population growth rate is given by the survival of adults. In most seal species the highest measures of adult survival are about 0.95-0.96, and for grey seals the best estimate available is 0.935 (Harwood and Prime 1978). An additional constraint is the observation that pup and subadult survival is always found to be lower and more variable compared to adult survival in all studied species of seals (Boulva and McLaren 1979, Boyd et al. 1995, Härkönen et al. 2002).

These biological constraints impose an upper ceiling of possible rates of long-term population growth for any seal species which can be found by manipulations of the life history matrix. In **Figure 2.8** we illustrate how fertility and mortality rates known for grey and ringed seals can combine to produce different long-term population growth rates. It is found that growth rates exceeding 10% (λ= 1.10) per year are unlikely in healthy grey seal populations (top stipled line in Fig. 1). Reported values exceeding 10% should be treated sceptically since they imply unrealistic fecundity and longevity rates. Such high growth rates can only occur temporally, and can be caused by e.g. transient age structure effects (Härkönen et al. 1999, Caswell 2000), but are also to be expected in populations influenced by considerable immigration.

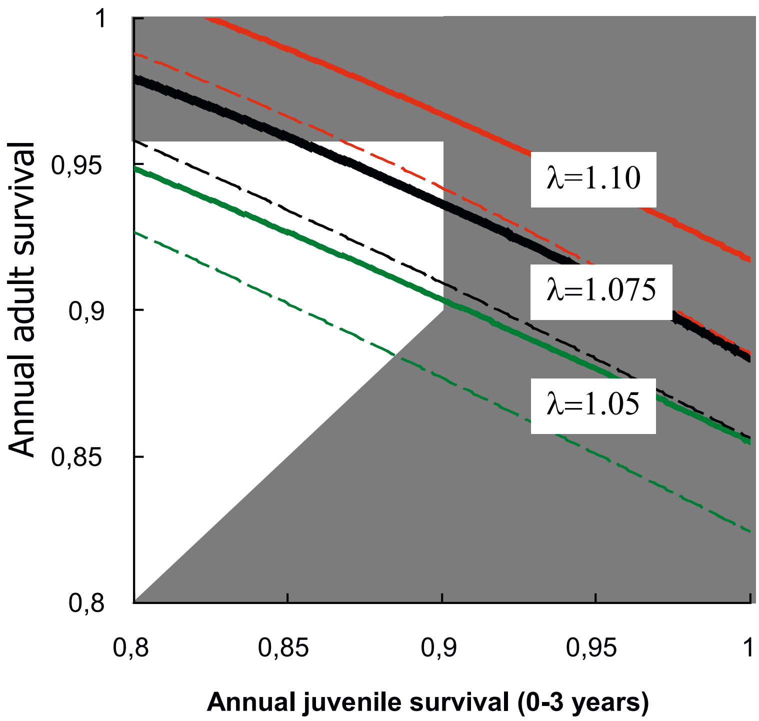

Figure 2.8. *Biological constraints delimit the maximum possible rate of increase in populations of grey and ringed seals. The shaded area denotes unlikely combinations of adult and juvenile survival rates. Any given point along the 6 lines shows a combination of adult survival and juvenile survival that produces a given growth rate (λ). The two uppermost lines are for λ = 1.10, the two lines in the middle for λ = 1.075, and the lowest two lines show combinations that result in λ = 1.05. The stippled lines show combinations of adult and juvenile survival rates given that the mean annual pupping rate is 0.95. The bold full lines show the possible combinations given that the pupping rate is 0.75.*

The upper limit of individual reproductive rate is reflected at the population level, and gives an upper theoretical limit for the population rate of increase (**Figure 2.8**). The mean values of fecundity and mortality will always be lower than the theoretical maximum rate of increase, also for populations which live under favourable conditions. Chance events such as failed fertilisation or early abortions reduce annual pregnancy rates, and in samples of reasonable sizes, mean pregnancy rates rarely reach 0.96 (Boulva and McLaren 1979, Bigg 1969, Härkönen and Heide-Jørgensen 1990). Another factor that will decrease mean pregnancy rates is senescence (Härkönen and Heide-Jørgensen 1990). Further, environmental factors will reduce fecundity and survival rates. The impact from extrinsic factors may occur with different frequency and amplitude. Environmental pollution and high burdens of parasites can decrease population-specific long-term averages of fecundity and survival (Bergman 1999), while epizootic outbreaks and excessive hunting have the capacity to drastically reduce population numbers on a more short-term basis (Dietz et al. 1989, Harding and Härkönen 1999, Härkönen et al. 2006). The type of variation in fecundity and survival rates will determine the structure of a population. In a population with a constant rate of increase (thus no temporal variability), the age- and sex-structure quickly reaches a stable distribution, where the frequencies of individuals at each age class are constant. Populations with low juvenile survival typically have steeper age distributions compared to populations with higher juvenile survival rates (Caswell 2001). We have shown the full span of theoretically possible combinations of vital rates at different population growth rates (**Figure 2.8**). It turns out that population growth rate of grey seals can only reach 10% if fertility rates are high (0.95).

Harbour seals mature about one year earlier than grey seals and ringed seals, which is why maximum rate of increase in this species is 12-13% per year (Härkönen et al. 2002).

Long term maximum growth rates in whales

Work carried out under the umbrella of the International Whaling Commission (IWC) have shown that that an appropriate dafault value for the realized annual maximum rate of increase for most whales is about 4% (Best 1992). Similar values have also been estimated for harbour porpoises (Woodley and Read 1991).

Empirical evidence

With few exceptions, most populations of seals have been severely depleated by hunting during the 20th century. Detailed historical hunting records for other pinnipeds are available for the Saimaa ringed seal Baltic ringed seal Baltic grey seal and the harbour seal in the Wadden Sea, Kattegat and the Skagerrak. Analyses of these hunting records documented collapses in all populations, which were depleted to about 5-10% of pristine abundances before protective measures were taken. After hunting was banned and protected areas were designated most populations started to increase exponentially.

Harbour seal populations outside the Baltic increased by about 12% per year between epizootics in 1988 and 2002, whereas all seal species in the Baltic showed lower increase compared with oceanic populations (**Table 2.7**).

Regression analyses of time series of abundance data can thus be used to test (ANOVA) if the observed rate of increase in exponentially growing populations deviates significantly from expected values.

Table 2.7. Rates of increase in seal populations depleted by hunting. Grey seals from the UK, Norway, and Iceland are not included here since they have been consistently hunted over the years. Canadian grey seals have life history data similar to harbour seals.

Species	Area	Annual growth rate	Period	Reference
Harbour seal	Skagerrak	+12%	1978-1987	Heide-Jorgensen & Härkönen (1988)
Harbour seal	Skagerrak	+12%	1989-2001	Härkönen et al. 2002
Harbour seal	Kattegat	+12%	1978-1987	Heide-Jorgensen & Härkönen (1988)
Harbour seal	Kattegat	+12%	1989-2001	Härkönen et al. 2002
Harbour seal	Baltic	+ 9%	1972-2010	Härkönen & Isakson 2011
Harbour seal	Wadden Sea	+12%	1980-1988	Reijnders et al. 1994
Harbour seal	Wadden Sea	+12%	1989-2001	Wadden Sea Portal
Grey seal	Baltic	+8.5%	1990-2002	Karlsson et al. 2009
Grey seal	Canada	+ 13%		Bowen et al 2005
Ringed seal	Baltic (BB)	+4.5%	1988-2011	Härkönen unpublished

Proposed GES boundaries

The proposed core indicator "Population trend" is appropriate for marine mammals when used in the OSPAR and ICES contexts. It is feasible in two scenarios of population growth: exponential rate of increase and when the population is close to carrying capacity. A depleted population can evaluated as obtaining GES, when its observed rate of increase doesn't deviate significantly from its intrinsic rate of increase (harbour porpoises 4%, grey and ringed seals 10%, and harbour seals 12%). When populations are close to their carrying capacities, populations obtain GES if the rate of decrease is less than 10% over a period of 10 years as stated in the OSPAR convention. Variances for these maximum estimates are available for all management units, and the statistical analyses can be performed using e.g. ANOVA tests. There is currently not a clear agreement whether the Baltic grey seal population has reached the carrying capacity or not.

Existing monitoring data

Information derived from national reports to HELCOM CORESET (*note that not all countries have reported their monitoring*)

Table 2.8. Monitoring of marine mammal abundance.

Country	Area/Basin	Species	Method	Noted parameters
Germany	Kiel Bay & Little Belt, Bay of Mecklenburg	subpopula-tion *Phocoena phocoena* western Baltic	line transect sampling	n individuals, n pups
Germany	Kiel Bay, Bay of Meck-lenburg, Southern Baltic Proper	subpopula-tion *Phocoena phocoena* western Baltic	POD (Porpoise detectors = self-contained submersible data logger for cetacean echolocation clicks)	See method
Germany	Bay of Mecklenburg & Pomeranian Bay, inter-nal lagoons	Harbour Seal	observation of potential haul-out sites; collection of accidental sightings	n individuals
Germany	Bay of Mecklenburg & Pomeranian Bay, inter-nal lagoons	Grey Seal	observation of actual and potential haul-out or resting sites; collection of accidental sightings	n individuals
Lithuania	Southern Baltic proper			n individuals?
Sweden	Baltic Proper, Gulf of Bothnia	Grey seal	Aerial, boat or land of grey seal haulouts	n individuals
Finland	Gulf of Bothnia, Kvarken, Åland Sea, Archipelago Sea, Gulf of Finland	Grey seal	Aerial surveys of grey seal haulouts during the molting season in spring	n individuals
Finland	Archipelago Sea	Grey seal	Aerial surveys of grey seal pupping islands during the breeding season in early spring	n individuals, n pups
Finland	Archipelago Sea, Gulf of Bothnia	Baltic ringed seal	Aerial surveys of ringed seals during the molting season in spring	n individuals
Finland	Gulf of Bothnia, and the Quark	Baltic ringed seal	Aerial surveys of ringed seals during the molting season in spring	n individuals
Sweden	Gulf of Bothnia and the Quark	Baltic ringed seal	Aerial surveys of ringed seals during the molting season in spring	n individuals
Sweden	Kalmarsund	Baltic harbour seal	Aerial surveys during moult	n individuals
Sweden	Kalmarsund	Baltic harbour seal	Landbased pup counts in June and July	n pups
Sweden/ Denmark	Southern Baltic and the Kattegat	Harbour seal	Aerial surveys during moult	n individuals

Sampling

Monitoring of marine mammal abundance require methods tailored for the different species. Whales and porpoises have usually been surveyed using ship based line transect methodology, where a certain proportion of the sea surface is covered during favourable weather conditions. Large-scale surveys such as SCANS have monitored the abundance of whales in the entire North Sea and adjacent waters (Hammond et al. 2002). This method is appropriate in areas where whale abundance is relatively high, but gives very wide confidence limits in low abundance areas such as the Baltic. The cost in man hours is also very high which is why such surveys only have been repeated about once a decade.

Alternative methods in low density areas include submerged hydrophonic devices that record sounds produced by whales. Such devices have been used in the Southern Baltic and an on-going project is deploying sonic equipment elsewhere in the Baltic. This method provides information on the distribution of porpoises but still needs to be evaluated for abundance estimates.

Ringed seals are monitored annually in the Bothnian Bay using strip sensus methodology (Härkönen et al. 1998), where more than 13% of the sea ice is covered during peak moulting season in the end of April (20th of April to the 1st of May) each year. Such surveys have been conducted since 1988 in the Bothnian Bay, whereas the southern populations in the Archipelago Sea, the Gulf of Finland and the Gulf of Riga and Estonian coastal waters only can be surveyed with this method when ice cover is permitting. Land based surveys of hauled out ringed seals provide complementary information from the Gulf of Finland.

Harbour seals in the Kalmarsund, southern Baltic and the Kattegat are surveyed during the peak moulting season in the latter half of August each year. All seal sites are photographed and seals are later counted on the photos. All seal sites are surveyed three times each season, and the mean number hauled out in the two highest counts are used for abundance estimates and trend analyses (Teilmann et al. 2010). Surveys are coordinated between Sweden and Denmark.

Grey seals are surveyed in a similar way as harbour seals, where all haul-out sites of seals are photographed and where seals are counted on the photos retrospectively. Surveys are conducted during peak haul-out season in the last week of May and the first week of June. Flights are coordinated among teams from Estonia, Finland and Sweden.

Methodology of data analyses

All methods except for the sonic method used for harbour porpoises give data on relative abundance since some seals always are submerged. However, since the surveys are standardized, a similar proportion of the seals can be expected to haul out during surveys among years. Consequently, estimates of relative abundance can be used for trend analyses, and the growth rate of populations can be estimated with good precision (Teilmann et al. 2010). Using capture/recapture methodology photo-id studies or branded or tagged animals can be used to estimate total abundance.

Sub populations are treated separately in the analyses where abundance and trend estimates are given for the following management units:

- Ringed seal: The Bothnian Bay including the North Quark, the Archipelago Sea, the Gulf of Finland, Estonian coastal waters including the Gulf of Riga.
- Harbour seal: Kalmarsund, the Southern Baltic, and the Kattegat.
- Grey seal: The entire Baltic.

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2.4. Productivity of white-tailed eagles

1. Working team: Sea birds *Authors:* Björn Helander, Christof Herrmann and Anders Bignert	
2. Name of core indicator Productivity of white-tailed eagles	*3. Unit of the core indicator* The mean number of nestlings of at least three weeks of age, out of all occupied nests.
4. Description of proposed indicator The productivity of white-tailed eagle in the coastal zone (15 km zone landwards) of the Baltic Sea is an indicator describing not only biomagnification of contaminants but also persecution, disturbance of nest sites, food availability and availability of suitable nesting sites. Thus, it describes in reproductive terms the growth rate and condition of the population and indirectly indicates the potential for increased abundance.	
5. Functional group or habitat type Top predatory birds	
6. Policy relevance Descriptor 1, criterion 1.3 Population condition Descriptor 4, criterion 4.1 Productivity of key species or trophic groups	
7. Use of the indicator in previous assessments Used in the HELCOM "Predatory bird health"-indicator. For more detailed descriptions see the "Predatory bird health"-indicator fact sheet (2009)	
8. Link to anthropogenic pressures Scientifically established links to hazardous substances.	
9. Pressure(s) that the indicator reflect Directly linked to inputs of synthetic and non-synthetic compounds, disturbance and habitat loss	
10. Spatial considerations The parameters should be sampled only from the 15 km coastal zone. Assessment units can be sub-basin wide coastal strips (per country), where an average productivity is calculated.	
11. Temporal considerations Frequency: can be updated annually	
12. Current monitoring Monitored in Sweden, Germany and in Finland. Data on breeding attempts, breeding success and brood size are collected at as many nests as possible. Early season air surveys are made to find breeding attempts in Sweden. These are later followed up by nest visits to check success and number of young.	
13. Proposed or perceived target setting approach with a short justification. Historical data exists that has been used for setting the GES boundary. GES boundary exists for Sweden and can tentatively be used in other areas.	

Introduction

The white-tailed sea eagle is a species that faced strong persecution in the 19th and early 20th century causing the population to crash by early 20th century. Protection measures increased the population, but in 1950s' the population crashed again because of organic pollutants, mainly DDT, which caused egg-shell changes (including thinning) and, hence, wide-spread failure in reproduction.

The white-tailed sea eagle was the first species that signaled the deleterious effects from environmental pollutants in the Baltic Sea. If white-tailed sea eagle reproduction had been monitored earlier during the 20th century, the negative impact of DDT could have been signaled as early as in the 1950s in the Baltic Sea. The sea eagle is the ultimate top predator of the Baltic ecosystem, feeding on fish, sea birds as well as on seals, and is thus strongly exposed to persistent chemicals that magnify in the food web.

Reproduction in the Baltic eagle population in the 1970s was reduced to 1/5 of the pre-1950 background level. Following bans of DDT and PCB during the 1970s around the Baltic, eagle productivity began to recover in the 1980s and since the mid-1990s is largely back to pre-1950 levels. The population on the Swedish Baltic coast has increased at 7.8% per year since 1990.

The improvement in reproduction of the Baltic white-tailed sea eagle populations came no earlier than 10 years after most countries around the Baltic had implemented bans of DDT and PCB. This is a clear reminder of the potentially long-term effects from persistent pollutants. The subsequent recovery, from an 80% reduction in reproductive ability in the 1970s, is nevertheless an important evidence of successful management actions.

The core indicator measures the productivity of white-tailed eagle in the coastal zone of different parts of the Baltic Sea thereby describing not only biomagnification of contaminants but also persecution, disturbance of nest sites, food availability and availability of suitable nesting sites. Thus, it describes in reproductive terms the condition of the population and indirectly indicates the potential for increased abundance and distribution.

Policy relevance

The maintenance of viable populations of species is one of the biodiversity objectives of the HELCOM Baltic Sea Action Plan. EU Birds Directive (79/409/EEC) lists the white-tailed sea eagle in Annex I, binding member states to undertake measures to secure reproduction and survival of the species. The species is listed in the following international conventions: Bern Convention Annex II (strictly protected species), Bonn Convention Annex I and II (conservation of migratory species), Washington Convention (CITES) Annex I (regulating trade). As a top predator in the marine ecosystem, white-tailed sea eagle is also a suitable indicator for the implementation EU Marine Strategy Framework Directive (2008/56/EU), which requires good environmental status (GES) of marine ecosystems by 2020. Particularly the following GES criteria apply to this core indicator:

– Species distribution (D1.1),
– Population size (D 1.2),
– Population condition (D 1.3),
– Productivity of key species or trophic groups (D 4.1).

Monitoring of sea eagle population health as environmental indicator, as well as monitoring of contaminants in eagles and their prey, is recommended in an international Species Action Plan, adopted under the Bern Convention in 2002 (Helander & Stjernberg 2003).

Approach for defining GES boundaries

Definition of GES for the core indicator (productivity) and for the supporting parameters, brood size and breeding success, are based on a Swedish data set during 1850s'-1954. The reference condition was an average of the parameter values over that time period. The GES boundary *sensu* EU Marine Strategy Framework Directive was set to the lower 95% confidence limit of the observations during the reference period. The GES boundary is for breeding success 60%, for brood size 1.64 nestlings and for productivity >1.0 nestlings. The observations should be measured as average of the last 5 years. These thresholds are based on data on the 15 km zone of the Swedish Baltic coast (Helander 2003a). 15 km has been widely observed to be the range for foraging among white-tailed sea eagles. The applicability of the proposed GES boundaries to other parts of the Baltic Sea should be validated. The GES boundaries can be tentatively used in Germany (C. Hermann, pers. comm.).

Temporal development and current state

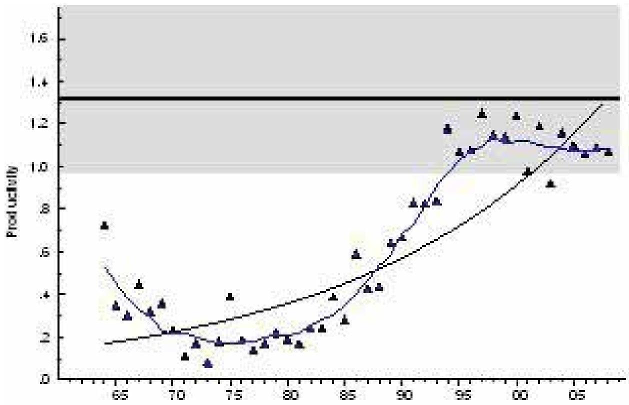
White-tailed sea eagle reproductive ability is monitored annually by assessing the frequency distribution of occupied eagle nests containing 0, 1, 2 or 3 nestlings (3 being the maximum in this species). Survey techniques and sampling methods are presented in (Helander 1985, 2003a, Helander et al. 2008). Three indicators of reproductive ability are calculated from these data: productivity, breeding success and nestling brood size. In addition, nutritional condition of nestlings is assessed. The productivity of the white-tailed sea eagle population was chosen as the core indicator to assess the status of the species.

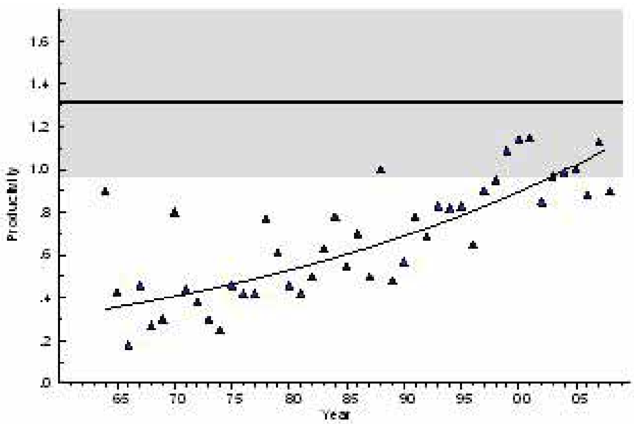
Productivity

The mean number of nestlings of at least three weeks of age, out of all occupied nests ([n1] + [n2x2] + [n3x3] / [n0] + [n1] + [n2] + [n3]).

This indicator combines the breeding success and brood size into a single indicator and assesses the reproductive output of the population. It is a useful indicator in studies on relationships between reproduction and anthropogenic pressures, such as contaminants, persecution and disturbance. It is also a vital parameter in assessments of population status in management perspectives.

The productivity has reached GES in Swedish coasts of the Gulf of Bothnia and Central Baltic Proper and in Mecklenburg-Vorpommern in Germany (**Figure 2.9**).





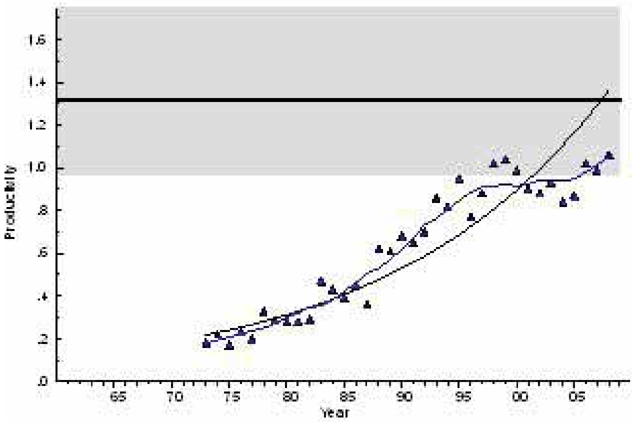
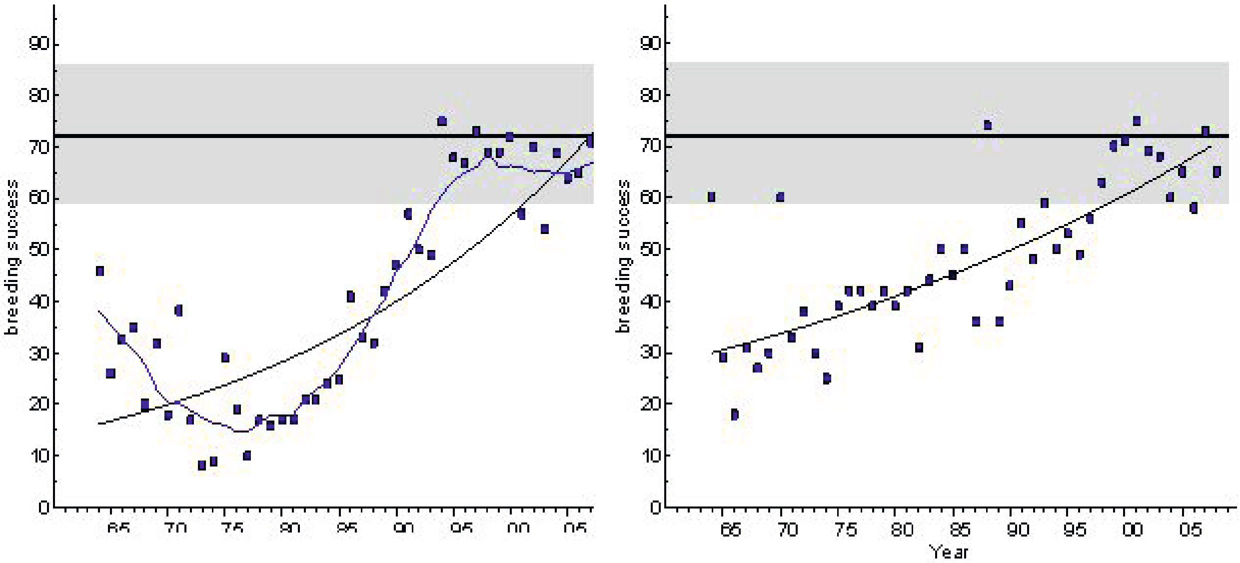


Figure 2.9. *Mean annual productivity of white-tailed sea eagle on the Swedish coast of the Baltic Proper (upper left)and the Gulf of Bothnia (Bothnian Sea and Bothnian Bay), 1964–2008, and in Mecklenburg-Western Pomerania, Germany, 1973-2008 (lower). The data set from Germany includes nests that were inspected only from the ground. Reference level given with range based on confidence limits for breeding success and brood size according to Helander (2003a). Whether the reference level, estimated from data from the Swedish Baltic coast, is fully relevant for the German eagle population has not been validated.*

Breeding success

The proportion of nests containing at least one nestling of at least three weeks of age, out of all occupied nests ([n1] + [n2] + [n3] / [n0] + [n1] + [n2] + [n3]).

Trends in breeding success of sea eagles on the northern, central and southern Baltic coast over time are presented in **Figure 2.10**. As the population has grown over the study period, the number of annually checked pairs has increased: in the Baltic Proper from 20–30 pairs before 1975 to 176 pairs in 2006, and in the Gulf of Bothnia from around 10 pairs before 1975 (all in the Bothnian Sea) to 89 pairs in 2006 (incl. also the Bothnian Bay, when it was repopulated). Similarly, the number of annually checked pairs in the sample from Mecklenburg-Western Pomerania, Germany, increased from around 75 to 219 between 1973 and 2008. Retrospective studies have shown that the breeding success on the whole Swedish Baltic coast decreased from on average about 72% before the 1950s to 47% in 1954-1963 and 22% in 1964-1982 (Helander 1985, 1994a). Breeding success increased significantly in the Baltic Proper as well as the Gulf of Bothnia from the early 1980s (Figure 3). By the middle to late 1990s, breeding success in both areas was no longer significantly different from the background level. The development in the southern Baltic (Germany) is similar to that in the central Baltic (Sweden, Baltic Proper, see Figure 3), but the breeding success seems to have stabilized at a lower level in Germany. The difference between the German sample and the two Swedish samples, respectively, is statistically significant. Impacts of intraspecific competition in areas with high density of breeding pairs have been discussed as a possible reason for the lower breeding success in Mecklenburg-Western Pomerania (Hauff 2009).



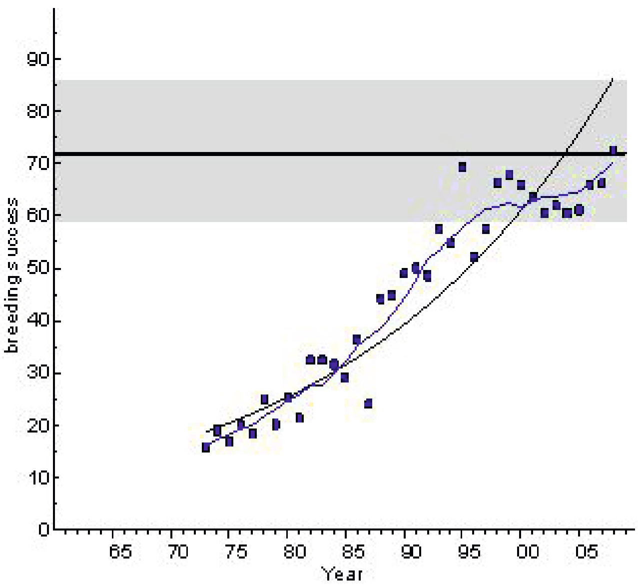
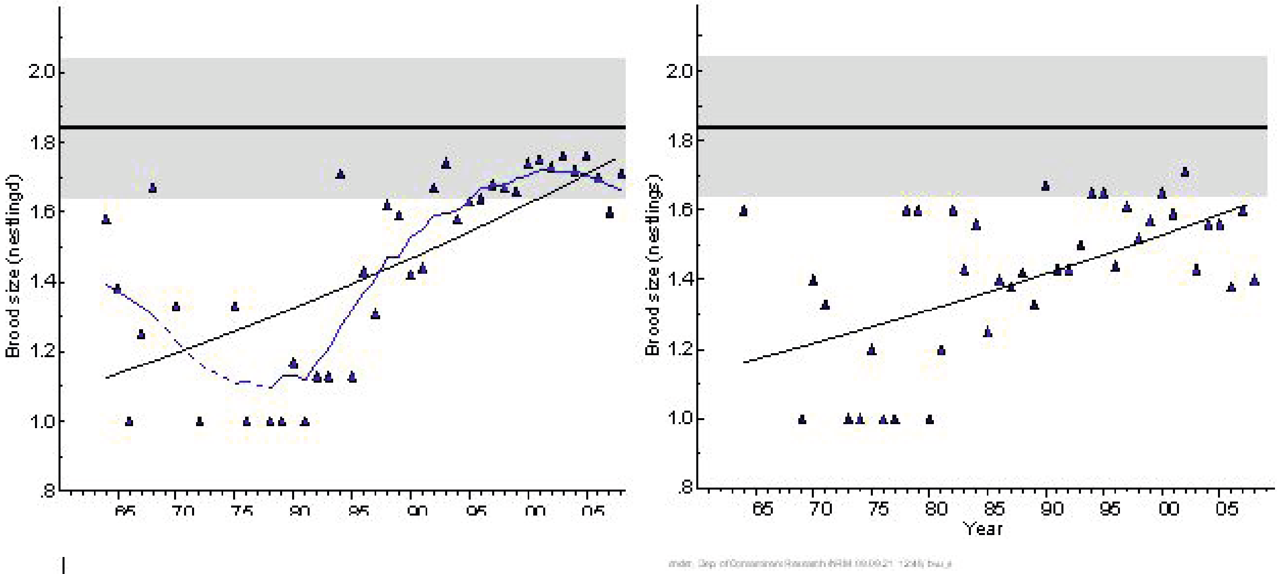


Figure 2.10. *Breeding success (%) of white-tailed sea eagle on the Swedish coast of the Baltic Proper (upper left) and the Gulf of Bothnia (Bothnian Sea and Bothnian Bay), 1964–2008, and in Mecklenburg-Western Pomerania, Germany, 1973-2008 (lower). The blue line included in the set of breeding success data represents a LOESS smoother that explained significantly more than the linear regression line. Reference level with 95% confidence limits is given according to (Helander 2003a). Whether the reference level, estimated from data from the Swedish Baltic coast, is fully relevant for the German eagle population has not been validated.*

Nestling brood size

The mean number of nestlings of at least three weeks of age in nests containing young ([n1] + [n2x2] + [n3x3] / ([n1] + [n2] + [n3].



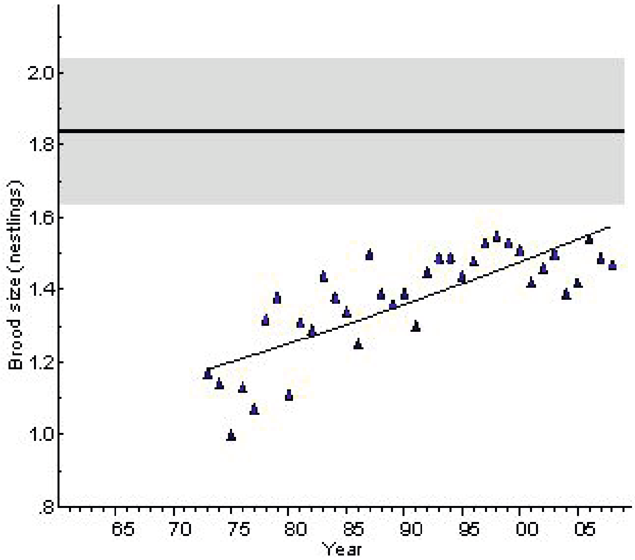


Figure 2.11. *Mean nestling brood size of white-tailed sea eagle on the Swedish coast of the Baltic Proper (upper left) and the Gulf of Bothnia (Bothnian Sea and Bothnian Bay), 1964–2008, and in Mecklenburg-Western Pomerania, Germany, 1973-2008 (lower). The data set from Germany includes nests that were inspected only from the ground. Reference level with 95% confidence limits is given according to (Helander 2003a). Whether the reference level, estimated from data from the Swedish Baltic coast, is fully relevant for the German eagle population has not been validated.*

Based on data from nests inspected by climbing the nest tree, and excluding nests checked only from the ground, nestling brood size is a precise standard. Nestling brood size began to increase in both areas from the 1980s, roughly in synchrony with the increase in breeding success (**Figure 2.11**). This is inherent with an improvement in the hatching success of the eggs, affecting both these indicators in parallel. Brood size reached back to the pre-1950 reference level in the Baltic Proper in the late 1990s. In the Gulf of Bothnia, however, brood size is still significantly below this reference level. This is mainly due to smaller broods in the southern part of the Bothnian Sea, as illustrated in **Figure 2.11**.

The current brood size in Germany is lower than in Sweden (**Figure 2.11**). During the period 1996-2004, 1.48 nestlings/nest have been recorded in Mecklenburg-Western Pomerania. It should be mentioned that this sample includes data from nests only checked from the ground, which results in a certain error due to nestlings not visible from this position. However, this bias does not explain the full difference from the data obtained for Sweden. Data received from ground observations in Germany indicates an underestimation of the real number of nestlings by 11% (Hauff & Wölfel 2002). Using this correction factor for the nests not climbed (about 50% of the total German sample), the corrected brood size for Mecklenburg-Western Pomerania is 1.56, which is still clearly on the low side compared to most coastal records from Sweden (**Figures 2.11 and 2.12**).

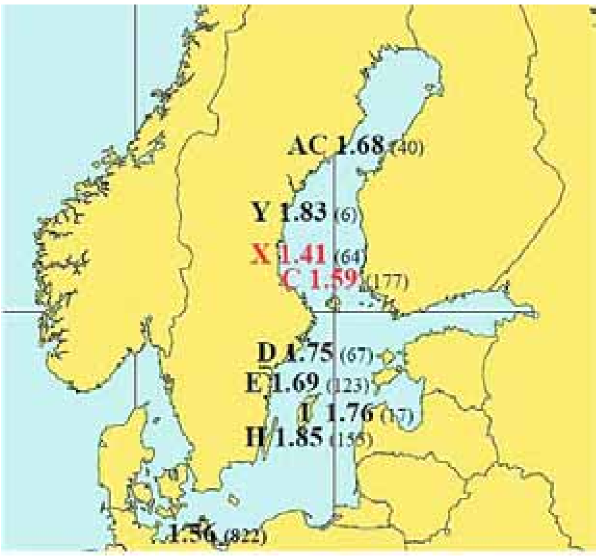


Figure 2.12. *Mean nestling brood size 1996-2004 on the Swedish Baltic coastline (15 km zone), counties indicated by letters, and in Mecklenburg-Western Pomerania, (German data corrected for nests checked from the ground). Sample sizes given in brackets. The reference level up to 1950 based on data from the Swedish coast was 1.84, with 95% confidence limits 1.64 - 2.04.*

Factors affecting the white-tailed sea eagle reproductive success

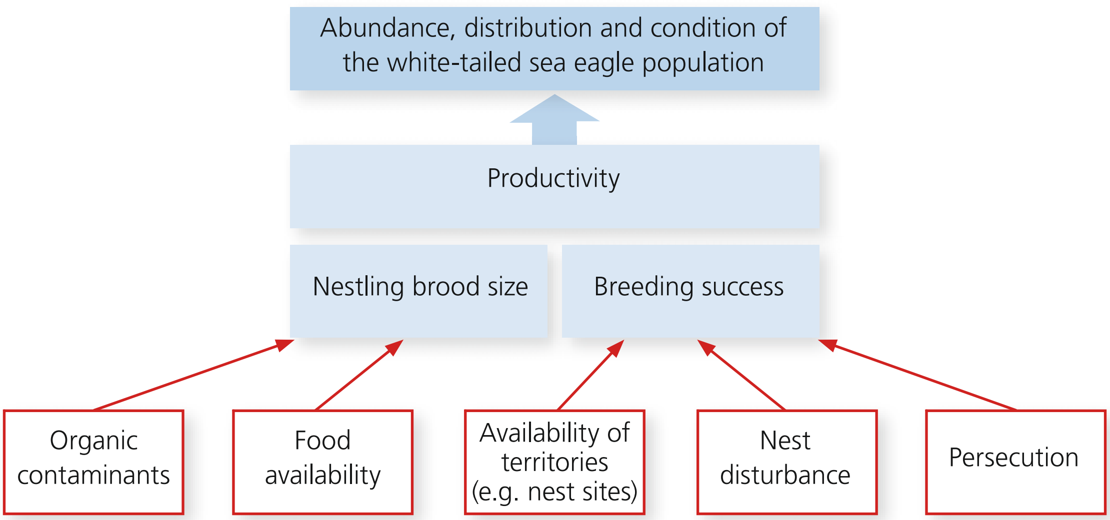


Figure 2.13. *The productivity of the white-tailed sea eagle is affected by several anthropogenic pressures acting through the nestling brood size (number of nestlings) and the breeding success (success in raising one nestling per pair).*

Monitoring

In Sweden, surveys of breeding populations and reproduction, sampling, sample preparation, storage in specimen bank and evaluation of results are carried out by the Department of Contaminant Research at the Swedish Museum of Natural History, Stockholm. Surveys of breeding populations and reproduction of reference freshwater populations are carried out by the Swedish Society for Nature Conservation (Project Sea Eagle), Stockholm. Chemical Analysis is carried out at the Institute of Applied Environmental Research at Stockholm University.

In Western Pomerania, Germany, data are collected by voluntary ornithologists, co-ordinated by the "Project group for large bird species" under the auspices of the Agency for Environment, Nature Conservation and Geology. The country-wide white-tailed sea eagle data are compiled by Peter Hauff, who submits the annual reports to the mentioned governmental agency.

Eagles are presently breeding along the coasts of the whole Baltic Sea, and are monitored in a network of national projects with harmonized methodology. Monitoring of nests is done in all coastal areas of the Baltic Sea with circa 300 nests in Sweden, 300 in Finland and 230 in Germany. There are no large gaps in the monitoring, but the compilation of data has not been done yet, except from Finland, Germany and Sweden.

Monitoring of sea eagle reproduction in Sweden is included in the National Environment Monitoring Programme since 1989 as indicator of effects from chemical pollutants. In Finland, the monitoring is done by WWF working group.

The first years of the data sets are as follows: in Sweden 1964, in Germany 1973, in Finland 1970.

Method and frequency of data collection

As nests are climbed for assessment of the reproductive parameters, nestlings are also measured (wing chord for estimation of age in days, tarsus width and depth for estimation of sex, see Helander 1981, Helander et al. 2007), and weighed (for nutritional status), sampled (feather and blood), and ringed within an international colour ringing programme, for identifications in the field (Helander 2003b). Dead eggs and shell pieces are collected for measurements, investigation of contents and chemical analyses, for studies on relationships with reproduction. Also shed feathers from adults are collected at all sites and archived. These materials are used in the assessment of other parameters/indicators. Brood size records from nests inspected only from the ground in Germany were corrected by multiplying by a factor 1.11 (Hauff & Wölfel 2002).

Methodology of data analyses

Simple log-linear regression analysis has been carried out to investigate average changes over time. To check for significant nonlinear trend components, a LOESS smoother was applied and an analysis of variance was used to check whether the smoother explained significantly more than the regression line. Statistical power analyses were used to estimate the minimum annual trend likely to be detected at a statistical power of 80% during a monitoring period of 10 years. To investigate the possible effect of a future reduced sampling scheme, repeated random sampling (5000 times) from 1991 to 2006 in the current database was carried out, simulating a maximum of 50, 25, 20, 15, and 10 records each year. Contingency analysis, using the G-test with Williams correction, a log-likelihood ratio test, was applied for comparisons between geographical regions and time periods. For references see (Helander et al. 2008).

Gaps and weaknesses

Minimum detectable yearly trend (%) for a 10-year monitoring period at a statistical power of 80% has been estimated for Swedish data for different sample sizes, based on random sampling from data collected during 1991 – 2006 (Helander et al. 2008). Minimum detectable trends based on the raw data set between 1991–2006 (with a varying annual number of observations) was 1.3% for brood size (Baltic Proper), 2.0% for breeding success (Gulf of Bothnia) and 3.0% for productivity (Gulf of Bothnia). The national survey methods are very similar with the only differences being whether to climb to the nest or survey it from the ground (applying the conversion factor).

The reliability of the core indicator can be increased by continuing to develop the GES boundary levels and further studying their linkage to anthropogenic pressures, such as disturbance in the vicinity of nests, wind farms and contaminants.

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2.5. Abundance of wintering populations of seabirds

1. Working team: Sea birds *Authors:* Henrik Skov, Martin Green, Susanne Ranft, Martti Hario	
2. Name of core indicator Abundance of wintering populations of seabirds	*3. Unit of the core indicator* A summary index, based on population sizes of selected seabird species
4. Description of proposed indicator Seabirds are important predators in the marine ecosystem. In the wintertime, seabirds aggregate in certain feeding grounds where their abundances can be monitored. The indicator follows the abundance of seabirds in the winter. It follows the birds on three levels: species, functional groups and total abundance. Integration at the group level: Species have been assigned to functional groups where the species abundances are weighted based on population size of respective species. Integration of the all species regardless of their functional groups describes the total abundance of wintering seabirds in an area. With repeated collection of data trends can be calculated over time.	
5. Functional group or habitat type Coastal pelagic fish feeder, offshore pelagic fish feeder, Subtidal offshore benthic feeder, Subtidal coastal benthic feeder, Subtidal herbivorous benthic feeder	
6. Policy relevance Descriptor 1, criterion 1.2 Population size Descriptor 4, criterion 4.2 Abundance/distribution of key trophic groups and species (Descriptors 5 & 6: indirectly)	
7. Use of the indicator in previous assessments None	
8. Link to anthropogenic pressures The seabird abundance in the winter is directly impacted by oil spills, by-catch, hunting, displacement by offshore constructions and shipping traffic. It is indirectly impacted by eutrophication and physical disturbance of bottom sediments (through changes in food supplies).	
9. Pressure(s) that the indicator reflect Selective extraction of species, introduction of synthetic compounds (oil spills), input of fertilisers and organic matter, abrasion and selective extraction, changes in siltation and thermal regime, other physical disturbance.	
10. Spatial considerations Winter concentrations of key species do not occur abundantly in the northern part of the Baltic but are confined to areas in the Baltic proper and southwards.	
11. Temporal considerations Monitoring frequency: ideally as often as possible, but at maximum circa every fifth year.	
12. Current monitoring Part of international waterfowl monitoring in several member states. Some kind of annual data collection (mainly coastal) is currently being made. Offshore monitoring is being conducted with longer intervals but plans for more regular monitoring (every 3-5 years) exist in at least some member states.	
13. Proposed or perceived target setting approach with a short justification. The GES is tentatively defined as a 50% deviation from mean of the reference period of 1992-1993.	

Introduction

Water birds are an important part of the marine ecosystem, being predators of fish and benthic fauna and herbivores in coastal areas. Their abundance is supported by the ecosystem productivity, but they also have top-down impacts on their prey species. In the Baltic Sea, majority of waterbird species overwinter in the marine area, aggregating in suitable feeding habitats. Hence, the abundance wintering and breeding populations respond to different pressures and they should be assessed separately.

Policy relevance

The seabirds have been recognized as a part of the marine ecosystem, and should be included MSFD assessments (Annex III, Table 1). They also fit to the BSAP biodiversity policy goal. In the MSFD, indicator "Abundance of wintering populations of seabirds" would fit best under the GES descriptor 1 (biodiversity) and descriptor 4 (food web). There is relatively little monitoring of the abundance of wintering seabirds in the offshore areas, but three studies are of key importance: (1) Inventory of Coastal and Marine Important Bird Areas in the Baltic Sea (Skov et al. 2000), (2) A quantitative method for evaluating the importance of marine areas for conservation of birds (Skov et al. 2007) and (3) the SOWBAS project's final report (Skov et al. 2011).

Method

The indicator is proposed to be based on key seabird species, which have functional significance in the marine ecosystem. They are listed below and sorted under the functional groups of bird species that have been identified in the CORESET project (**Table 2.9**).

Table 2.9. Species provisionally selected for the indicator and categorized by their functional groups.

Species (winter populations)	Functional group
Black-throated diver *Gavia* arctica	Coastal pelagic fish feeder
Red-throated diver *Gavia stellata*	Coastal pelagic fish feeder
Great crested grebe *Podiceps cristatus*	Coastal pelagic fish feeder
Goosander *Mergus merganser*	Coastal pelagic fish feeder
Red-breasted merganser *Mergus serrator*	Coastal pelagic fish feeder
Razorbill *Alca torda*	Offshore pelagic fish feeder
Common guillemot *Uria aalge*	Offshore pelagic fish feeder
Black guillemot *Cepphus grille*	Offshore pelagic fish feeder
Velvet scoter *Melanitta fusca*	Subtidal offshore benthic feeder
Common scoter *Melanitta nigra*	Subtidal offshore benthic feeder
Long-tailed duck *Clangula hyemalis*	Subtidal offshore benthic feeder
Eider *Somateria mollissima*	Subtidal offshore benthic feeder
Tufted duck *Aythua fuligula*	Subtidal coastal benthic feeder
Greater scaup *Aythua marila*	Subtidal coastal benthic feeder
Goldeneye *Bucephala clangula*	Subtidal coastal benthic feeder
Mute swan *Cygnus olor*	Subtidal herbivorous benthic feeder
Mallard *Anas platyrhynchos*	Subtidal herbivorous benthic feeder
Coot *Fulica atra*	Subtidal herbivorous benthic feeder

All the selected seabird populations are affected by the eutrophication state (**Table 2.10**). In the oligotrophic end of the eutrophication state, the bird populations are limited by the availability of food sources, whereas towards eutrophic conditions plant and zoobenthos biomass increases which first benefit seabird populations, but in the extreme end cause decrease in food availability.

Oil pollution affects most of the seabirds, oiling feathers and causing hypothermia. Although the number of oil slicks has significantly decreased in the Baltic Sea, oily surface waters still are a significant anthropogenic pressure for seabirds. Estimates of the number of birds oiled are uncertain.

By-catch of seabirds in fishing activities is a problem for all fish feeders and benthic divers. Estimates of the number of birds drowned in fishing gear are uncertain.

Hunting of seabirds is a significant pressure for some of the selected key species. Particularly, bags of eider and goldeneyes are heavy.

Table 2.10. Pressures affecting the waterbird populations.	
Species (wintering population)	**Anthropogenic pressure**
Black-throated diver	eutrophication, oil, by-catch
Red-throated diver	eutrophication, oil, by-catch
Great crested grebe	eutrophication, oil, by-catch
Goosander	eutrophication, oil, by-catch
Red-breasted merganser	eutrophication, oil, by-catch
Razorbill	eutrophication, oil, by-catch
Common guillemot	eutrophication, oil, by-catch
Black guillemot	eutrophication, oil, by-catch
Velvet scoter	eutrophication, oil, by-catch
Common scoter	eutrophication, oil, by-catch
Long-tailed duck	eutrophication, oil, by-catch
Eider	eutrophication, oil, by-catch
Tufted duck	eutrophication, by-catch
Greater scaup	eutrophication, by-catch
Goldeneye	eutrophication, by-catch
Mute swan	eutrophication
Mallard	eutrophication
Coot	eutrophication

Because the pressures affecting the selected key seabirds in the winter populations are similar, it is possible to make an index indicator where assessment can be first made on the species level and then functional groups are assessed separately. Finally, an integration of all the species can be made to describe abundance of all wintering seabirds. How this integration will be made is not yet clear.

Because the species and functional groups may have different significances in the ecosystem, weighting factors should be considered. They could be based on the conservation value of the Baltic population in the European context or the proportion of the species in the wintering seabird abundance.

Approach for defining GES boundaries

All the species or functional groups respond to anthropogenic pressures slightly differently, although the pressures behind the change are similar. Therefore the targets, which show the boundary of GES, must be set for each species separately. It is proposed that the GES boundaries are set on the basis of (1) time series data and (2) relation to other indicators (e.g. nutrients, chlorophyll, zoobenthos, plant abundance, fish stocks).

The GES boundaries should be given separately for all the sub-basins of the Baltic Sea (see Figure). As defining GES for all of the sub-basins may take time, the first step should be to define GES for those sub-basins which have highest abundance of seabirds in the winter.

Time series data: it is obvious that there are gaps in the time series datasets of wintering seabirds. The available data sets should be used as far as possible, using also best estimates. Temporal trends should be checked, because they show changes in the environment. Available sources of information are Skov et al. (2000, 2007 and 2011).

Relations to other indicators: When assessing GES of the Baltic Sea, there should not be any mismatch between GES of different indicators. For example, the nutrient concentration targets in the Baltic Sea have been agreed in the HELCOM BSAP. Therefore, the seabird GES boundaries should not be set on levels which cannot be reached when nutrient targets have been reached. In addition, competitive interactions between fish feeding birds and large fish affect the target setting. With the current long-term management plan of cod, the cod stocks will increase, which likely affects the food availability for birds. The GES boundaries for birds should not be set too high in such conditions.

The policy decisions under different frameworks have possibly conflicting objectives. The Favourable Conservation Status under the Birds Directive may be difficult to reach, if the environment changes to more oligotrophic direction. However, decrease of by-catch, oil pollution and hunting would allow higher bird populations and may mitigate this conflict.

Provisional GES boundaries: Until modeling studies have confirmed possible GES boundaries for the selected bird species, it is proposed that provisional GES boundaries are used. The GES is tentatively defined as a 50% deviation from mean of the reference period of 1992-1993 (based on available temporal trends in Skov et al. 2011).

Ultimately, the GES could be set by modelling the population size based on the GES for eutrophication related core indicators, because the abundance of seabirds depends on the trophic state of the ecosystem and the objective of decreasing the eutrophication will affect the seabird populations.

Assessment units for the seabird indicators

The abundance of wintering seabird populations differs among Baltic sub-basins. Therefore the indicator should be assessed per sub-basin. That means that also the targets must be set for each of the sub-basins.

Figure 2.14. *Sub-basin assessment units in the Baltic Sea (blue lines).*

Sampling and data analyses

See Skov et al. 2007 and 2011

Gaps and weaknesses

The indicator has currently a couple of weaknesses which must be addressed in near future:
– not all wintering grounds are covered,
– monitoring methods differ between the offshore monitoring and national monitoring practices,
– GES boundary is tentative, because of the uncertainty of interlinkages with other GES boundaries.

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2.6. Distribution of wintering seabirds

1. Working team: Seabirds *Authors:* Henrik Skov, Susanne Ranft, Martti Hario	
2. Name of core indicator Distribution of wintering seabirds	*3. Unit of the core indicator* Quantitative changes in the main distribution area of seabird species determined from density surface models developed on the basis of line transect survey data in consideration of species specific habitat suitability and sensitivity towards anthropogenic pressures
4. Description of proposed indicator Changes in the main distribution area determined from density surface models like GAMs or GLMs (higher end of the distribution e.g. the 75th or 90th percentile of all the sampled densities during the reference situation) are analysed. The indicator consists of well-known species of high numerical and environmental importance only, for which sufficient coverage by line transect data is available, such as Common Eider, Velvet Scoter, Common Scoter and Long-tailed Duck. Time series data can provide information on changes over time and reveal reoccurring spatio-temporal patterns. In combination with data on anthropogenic pressures, naturally driven patterns can then be distinguished from pressure based changes. Pressure based changes in distribution may occur due to changes in resource quality and availability, habitat loss and disturbances or barriers.	
5. Functional group or habitat type For the time being the indicator is species specific.	
6. Policy relevance Descriptor 1 GES criteria 1.1.1 distributional range and 1.1.2 distributional pattern of species Biodiversity segment of the BSAP: Objective of natural landscapes and seascapes.	
7. Use of the indicator in previous assessments None	
8. Link to anthropogenic pressures Directly impacted by: resource depletion (selective extraction of species), habitat loss or deterioration (destructive fishing techniques, construction of artificial structures), temporal or permanent displacement by constructions and ship traffic. Indirectly impacted by: changes in species composition and consequently e.g. resource competition with other bird or non-bird species, changes in resource quality and availability due to contamination by hazardous substances and eutrophication.	
9. Pressure(s) that the indicator reflect selective extraction of species, shipping and other physical disturbances, physical damage through dredging and sand/gravel/boulder extraction, nutrient and organic matter enrichment	
10. Spatial considerations Areas of suitable habitat to be examined, need to be defined / modelled for each species separately, thereby keeping in mind the naturally occurring spatio-temporal variation, areas at times of sea ice coverage and oxygen depletion should be avoided	
11. Temporal considerations Times of highest occurrence, high naturally occurring spatio-temporal variation has to be considered, combination of several observations at varying weather and hydrographical conditions. For waterbirds this would typically mean mid winter.	

12. Current monitoring SOWBAS, national monitoring for the identification of SPAs and IBAs, to be included in the HELCOM Waterbird Monitoring
13. Proposed or perceived target setting approach with a short justification. Deviation from favourable reference value (higher distribution) or deviation from reference state at the beginning of the assessment (the target would be resettlement of areas where anthropogenic pressures lead to avoidance behaviour and no further loss of habitat). Maybe a combination of both approaches.

Introduction

Seabirds are found across the world's oceans in many different habitats with aggregations occurring over a wide range of spatial and temporal scales (O'Driscoll 1998). The structure and occurrence of aggregations is thereby related to a number of environmental factors, including water temperature and salinity, fluid dynamics, meteorological conditions, food availability and the presence of other marine predators (see e.g. Lewis et al. 2001, Markones et al. 2008). The species-specific effects of these parameters precipitate explicit environmental requirements and dependencies which in combination with the distinct behaviour and biology of the diverse seabird species shape their distributional range and pattern. Thereby, the different species reveal varying distribution patterns ranging from very wide spread to locally concentrated and unpredictable short-term to regular long-term aggregations (BfN 2006, Garthe 2003, Markones et al. 2010, Sonntag et al. 2007). Especially for species with regular long-term aggregations in habituated and predefined localities of specific environmental conditions frequented for breeding, resting or feeding, the availability of suitable and undisturbed habitats is a key factor for the well-being of their population.

Anthropogenic pressures, however, decrease the extent of suitable seabird habitats or effect seabird distribution in various other direct or indirect ways (see e.g. Garthe 2003). Human activities such as dredging, sand or gravel extraction as well as the use of destructive fishing techniques or the construction of artificial structures destroy habitats or reduce habitat quality. The latter, in line with shipping, marine wind farms, infrastructure and other physical disturbances, also lead to habitat displacement (Bellebaum et al. 2006, Garthe & Hüppop 2004, Garthe et al. 2004, 2008, Pettersson 2005, Petersen et al. 2006, Sonntag et al. 2007). Marine infrastructures may also act as barriers altering the distributional patterns of bird species at sea (Mendel & Garthe 2010). Fishing or other selective extraction of species influence seabird distribution as resources are being depleted or the species compositions altered leading for example to resource competition with other marine predators (Garthe 1997, Siebert et al., 2009). Furthermore, contamination by hazardous substances and nutrient and organic matter may modify resource availability and quality, therefore indirectly influencing seabird distribution (BfN 2006).

With its distinct oceanographic and geographical characteristics the Baltic Sea holds important and unique habitats for breeding and wintering seabird species. The importance of the Baltic as a wintering ground for example can be underlined by the fact that during mild winters over 90% of west-palearctic Long-tailed Ducks and Velvet Scoters overwinter here (Mendel et al. 2008). Changes in distributional patterns of species tied to specific habitats analysed with respect to habitat suitability and anthropogenic pressures should provide information on the environmental status of the species as well as on pressures causing a decline or change in distributional pattern. The authors of the document at hand believe changes in distribution of seabirds to be a suitable indicator for the state of the Baltic Sea marine ecosystem in general and Baltic Sea seabirds in particular.

Description of the indicator

The indicator is defined as quantitative changes in the main distribution or concentration area of selected species. The main distribution area thereby refers to the higher end of the species specific distribution that is the 75th percentile. Calculations should be based on time series of line transect data (e.g. 6 yrs) to reduce sample bias which is likely to occur due to e.g. weather conditions. Area of main distribution is determined by application of the 75th percentile on surface density models (e.g. GAMs and GLMs) based on the transect data. Species should be selected based on data availability and accessible background information. Knowledge on their biology, behaviour and seasonal rhythm as well as habitat dependency should be available. Changes in species distribution have to be analysed with respect to habitat suitability and anthropogenic pressures taking into account species-specific density thresholds, species-specific pressure sensitivity as well as within- and between-species competition. Time series data may provide information on reoccurring spatio-temporal patterns allowing for a differentiation between naturally driven variation and pressure based changes.

Because of its tight linkage to waterbird abundance, this indicator should be used together with the indicator Abundance of wintering populations of waterbirds.

For the time being focus shall be put on wintering seabirds. This pays tribute to the important role of the Baltic Sea as wintering area and goes along with the proposed core indicator on wintering seabird abundances. Being among the most abundant and ecologically dominating seabirds wintering in the Baltic Sea the following four species are proposed for the appliance of the presented indicator: Common Eider, Velvet Scoter, Common Scoter and Long-tailed Duck. Other species may be added. For the time being all species should be analysed separately interpreting indicator results on the species level only. In the future, as knowledge improves, the quantitative changes in distribution of several species may be integrated on the level of functional groups.

Policy relevance

The proposed indicator applies to the MSFD GES criteria 1.1 "species distribution" and the parameters "distributional range" (1.1.1) and "distributional pattern" (1.1.2) listed under descriptor 1.

The EC Birds Directive requires special conservation measures for seabird species to ensure their survival and reproduction in their distribution areas. Measures specifically include classifying the most suitable territories as Special Protection Areas. Consequently assessments are required to provide information on the distribution of species for the designation of SPAs as well as ongoing monitoring schemes to detect changes in distribution to adapt management and secure conservation of the target species.

The ecological objectives of the Baltic Sea Action Plan (BSAP) currently do not comprise targets and indicators for wintering waterbirds or distribution of waterbirds in general. However, the biodiversity segment of the BSAP includes the objective of natural landscapes and seascapes and the parameter "percentage of important migration and wintering areas for birds within the Baltic Sea covered by the BSPAs, Natura 2000 and Emerald sites". This implies knowledge on the distribution of seabird species.

Testing of the core indicator and examples from the literature

The proposed indicator has not been used as such in the past nor have there been - to the knowledge of the authors - any attempts. However, seabird distribution and density measures have been used for the identification of Important Bird Areas and Special Protected Areas and are integral elements of environmental impact assessments. The underlying studies have detected relations between seabird distribution and anthropogenic activities.

Demonstrations of how the 75 percentile of density measures can provide a robust indication of changes in main distribution areas of waterbirds can be found in the recently finished report of the SOWBAS (Status of wintering waterbird populations in the Baltic Sea) project (Skov et al. 2011). SOWBAS was launched in 2006 and carried out co-ordinated surveys of waterbirds in all Baltic waters during 2007-2009. The project attempted to fill gaps in knowledge of the status and recent trends in the populations of wintering waterbirds in the Baltic Sea and provides a follow up of the first Baltic wide survey on seabird distribution in 1992-1993 (Durinck et al. 1994). Compared to the report covering the results from the first census the results from the new SOWBAS report have been achieved through the application of spatial modelling. **Figure 2.15** demonstrates how percentile distributions (here the 75 percentile) can be used as a means to delineate the higher end of distributions of waterbirds in a region. The examples show comparisons of densities of Long-tailed Ducks wintering in the Bornholm Basin and Pomeranian Bay between the Baltic -wide surveys in 1992-1993 and 2007-2009. Despite big changes in absolute densities between the two periods the area of high habitat suitability marked by the 75 percentile provides a robust indication of changes in the main distribution pattern within the region between the two periods.

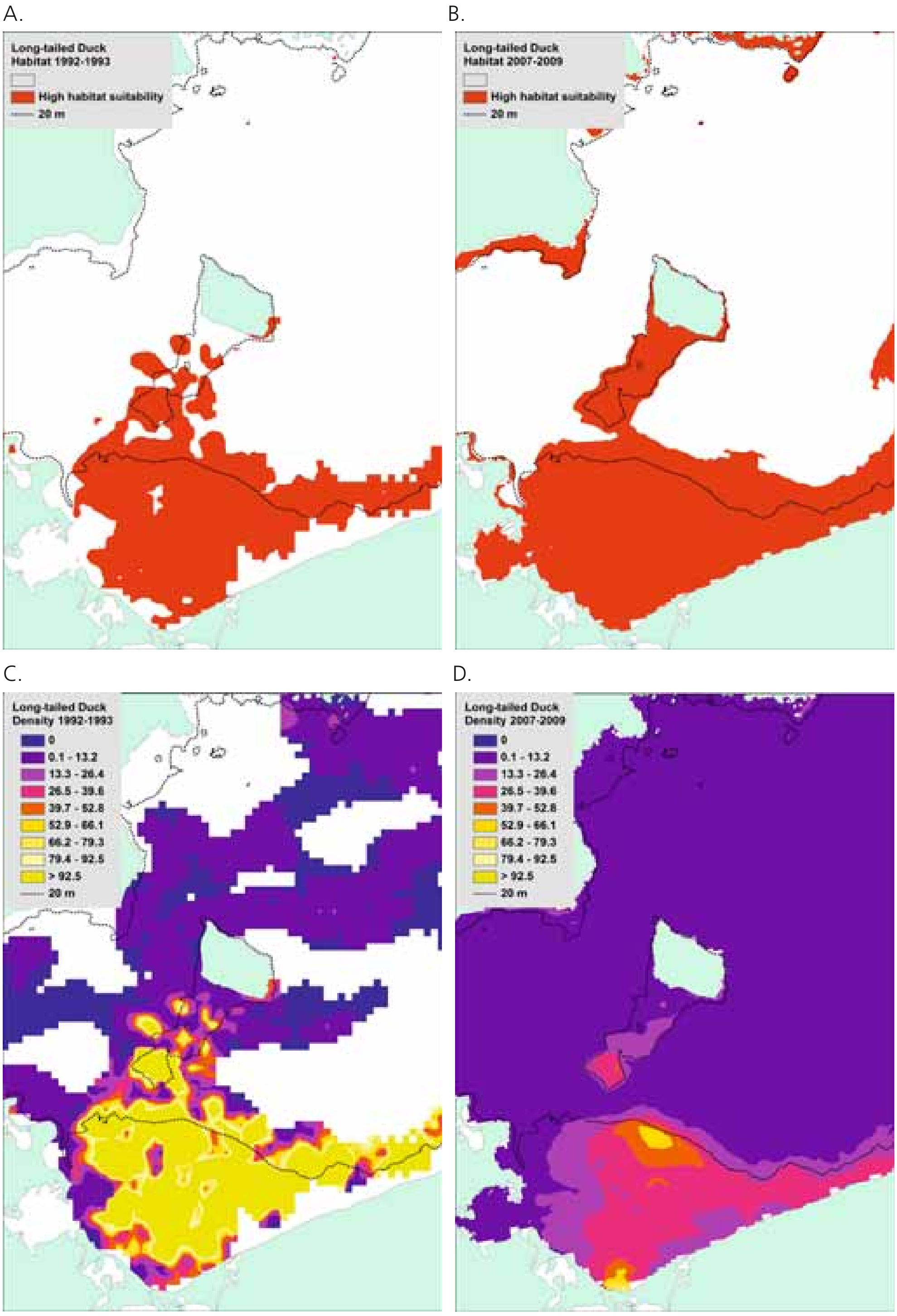


***Figure 2.15**. Comparisons of habitat suitability (panels Aand B) and densities (panels C and D) of Long-tailed Ducks wintering in the Bornholm Basin and Pomeranian Bay between Baltic -wide surveys in 1992-1993 (panels A and C) and 2007-2009 (panels B and D). Source: Skov et al. 2011 by courtesy of H. Skov.*

Approach for defining GES

All species respond differently to anthropogenic pressures. Therefore the targets, which show the boundary of GES must be set for each species separately. Based on data availability two different ways of determining GES are proposed.

The GES boundary can be set as an acceptable deviation from a favourable reference value that is the potential distribution of a species defined by habitat suitability modelling. A different approach is to set a threshold value for acceptable/ required deviations from a reference state, e.g. as defined at the beginning of the assessment. The target would be resettlement of areas where anthropogenic pressures lead to avoidance behaviour or no further loss of habitat.

The high naturally occurring spatio-temporal variations in species distribution (Garthe et al. 2008, Markones & Garthe 2009, Sonntag et al. 2010) have to be considered at all times and detailed spatial statistical analyses have to be developed.

Existing monitoring data

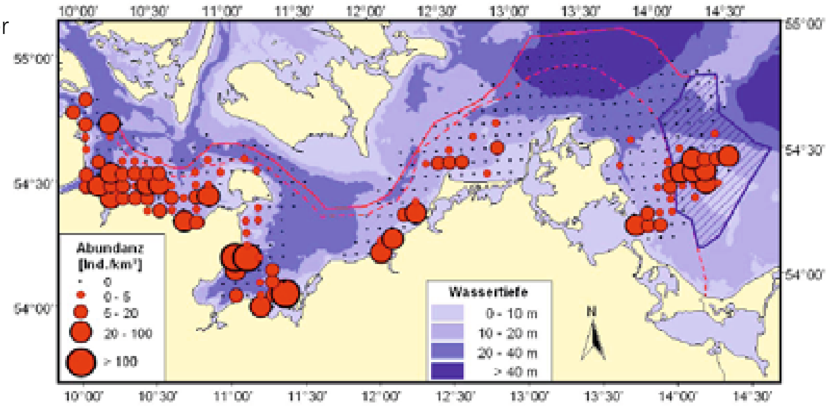
Currently there is almost a complete lack of internationally co-ordinated monitoring data on waterbirds, especially in offshore areas. Until today there have been only two Baltic wide studies on seabird distribution. In 1992, the first survey (ship transects) covering all major offshore areas was carried out by Ornis Consult (Durinck et al. 1994). This was followed up by international surveys from both aeroplane and ships in 1993. In 2006 the SOWBAS (Status of wintering Waterbird populations in the Baltic Sea) project was launched and carried out co-ordinated surveys of waterbirds in all Baltic waters during 2007-2009.

Other counts of wintering waterbirds in the Baltic Sea stem from the midwinter counts of Wetlands International. These counts generally cover birds of the coastal zone and lagoons, while offshore areas are surveyed only infrequently.

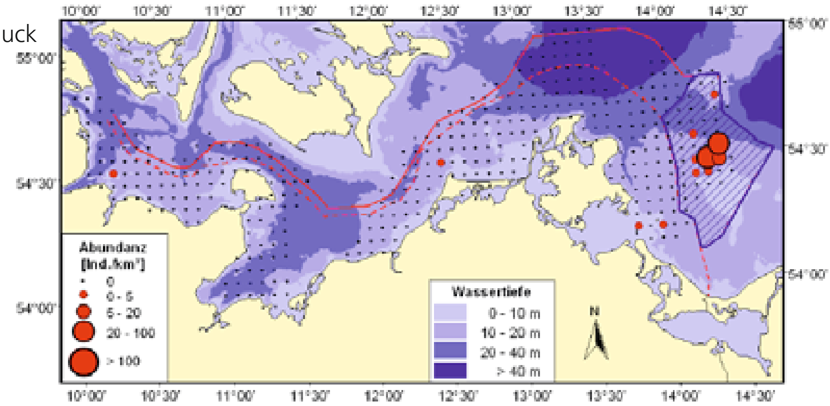
In addition there are several national monitoring projects for the identification of IBAs and SPAs as well as regional projects including spatial-assessment of seabirds at sea such as the Baltic LIFE project.

Germany, for example, conducts as part of the "Seabirds at Sea" program since the year 2000 ship- and airplane-based transect surveys assessing the distribution and abundance of seabirds within the German Baltic Sea area (**Figure 2.16**). Moreover, the German Federal Agency for Nature Conservation (BfN) commissions since 2009 a seabird monitoring within the framework of Natura 2000 with special focus on the German Exclusive Economic Zone. Additional research includes for example airplane-based transect counts in deep waters of the Baltic Sea (Markones et al. 2010). In depth information on the occurrence and distribution of seabirds in German waters can be found in Garthe (2003), BfN (2006), Mendel et al. (2008), Dries and Garthe (2009), Markones and Garthe (2009) and Sonntag et al. (2006, 2007, 2010). Below examples of the distribution of Common Eiders, Long-tailed Ducks, Velvet Scoters and Common Scoters in the German Baltic Sea as of January/ February 2009 are displayed (Markones and Garthe 2009).

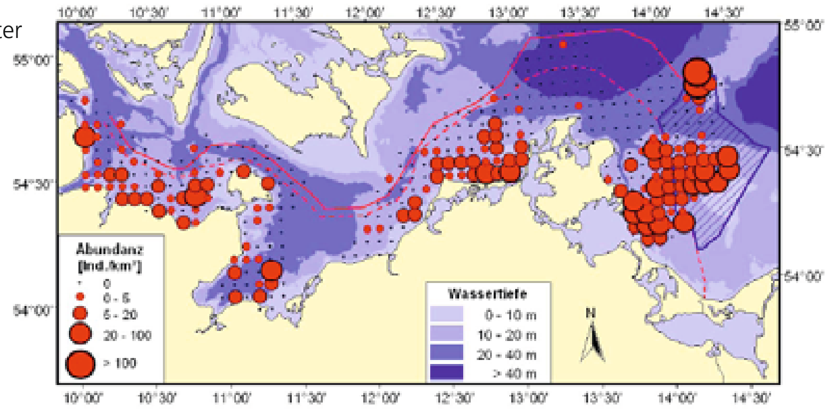
(a) Common Eider



(b) Long-tailed Duck



(c) Common Scoter



(d) Velvet Scoter

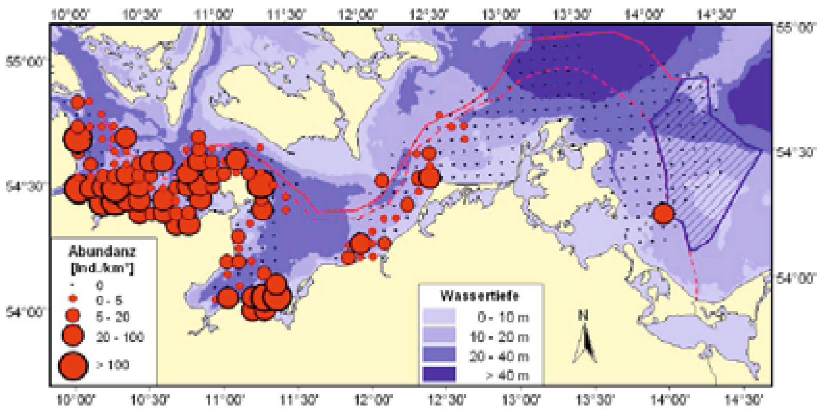


***Figure 2.16**. Distribution of (a) Common Eiders, (b) Long-tailed Ducks, (c) Velvet Scoters and (d) Common Scoters in the German Baltic Sea January/ February 2009 (airplane-based surveys). Source: Markones & Garthe 2009.*

Sampling

It is suggested to include surveys of wintering seabirds with special emphasis on shallow offshore areas into the HELCOM Waterbird Monitoring which should fall under the COMBINE regulations.

As a minimum requirement for a Baltic-wide monitoring programme for wintering waterbirds key habitats which may be regarded as holding significant proportions of the European wintering populations should be assessed. Monitoring should take place at times of highest occurrence (mid-winter) following standard procedures.

In additional to surveying (principally transect surveying) individual animal tracking could be applied to explore distribution. While the first provides large scale information the latter may provide important additional background information for the interpretation of detected changes in distribution. Habitat association can be inferred by comparing the animal's locations with available habitat within the bird's potential range or directly by use of data loggers providing environmental data on prevailing oceanographic conditions. Moreover, animal tracking delivers high quality information on the individual, such as its activity and status which are important variables aiding in the interpretation of distribution and habitat association.

Weaknesses/gaps

As pointed out above, besides the naturally occurring spatio-temporal variation, also multiple pressures can be identified as playing an important (either negative or positive) role in the distributions of most species of waterbirds. Teasing out the relative influence of natural determinants and each pressure on the distribution and conservation status of every species requires detailed statistical analyses, which could not be developed within the scope of this report. The proposed core indicator requires further work, but can be used tentatively in environmental assessments. In addition, the paper at hand aims at promoting a Baltic wide spatially explicit monitoring of seabirds especially in the offshore areas.

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2.7. Fish population abundance

2.8. Mean metric length of key fish species

2.9. Fish community diversity

2.10. Proportion of large fish individuals in the community

2.11. Abundance of fish key trophic groups

2.12. Fish community trophic index

Authors and acknowledged persons of the coastal fish indicators: Magnus Appelberg, Jens Olsson, Håkan Wennhage, Antti Lappalainen, Kaj Ådjers, Markus Vetemaa, Outi Heikinheimo, Adam Leijk, Atis Minde, Iwona Psuty and all Members of the HELCOM FISH-PRO project.

Introduction

This description of core indicators focuses on the work done in the HELCOM FISH-PRO project (HELCOM 2011). The core indicators identified in the HELCOM CORESET project are meant to apply to a wider set of species and methodologies, developed elsewhere (e.g. ICES working groups, LIFE+ project MARMONI, etc) or in the second phase of the CORESET project in 2012. Some elements of that work can be seen in Chapter 4 of this report, where candidate indicators are described.

Description of coastal fish indicators

Indeces for coastal fish were developed for the following biodiversity levels within D1.

Table 2.11. Proposed core indicators and their links to the MSFD GES criteria.

GES criteria	Coastal fish Indeces	Proposed core indicators
1.2	Species Abundance Index	Fish population abundance
1.3	Species Demographic Index	Mean metric length of key fish species
1.6	Community Diversity Index, Community Size Index	Fish community diversity, Proportion of large fish individuals in the community
1.6	Community Abundance Index	Abundance of fish key trophic groups
1.7	Community Trophic Index	Fish communitytrophic index

Indices for coastal fish community status estimate changes over time within one monitoring area. The indices can potentially also be used for estimating differences among geographic areas, provided that the same monitoring method is applied in all areas to be compared (HELCOM 2011).

Coastal Fish - Species Abundance Index (D1.2.1)

The index estimates the abundance of a key fish species in Baltic Sea coastal areas, such as Perch (*Perca fluviatilis*). Perch is a freshwater species that commonly dominates quantitatively coastal fish communities in the Baltic Sea (Ådjers et al. 2006). As such, areas of good ecological status generally have strong populations of Perch (Eriksson et al. 2011). The species is a piscivore in the adult stage and an appreciated target for both small scale commercial fisheries and recreational fishing (Swedish Board of Fisheries 2011). The index reflects the integrated effects of recruitment and mortality. Recruitment success is expected to be mainly influenced by climate and quality of recruitment habitats. Mortality is influenced by fishing, but potentially also by predation from apex predators, such as seals, sea birds and fish.

Policy relevance: The index will show if the abundance and productivity of Perch is at appropriate level for supporting coastal ecosystem function, focusing on food provision but also reflecting the trophic state. In areas where the index does not signal good environmental status, GES may be achieved by restoration of recruitment habitats, improving water quality or regulating fishing pressure.

Coastal Fish - Species Demographic Index (D1.3.1)

The index reflects the size structure of a key fish species in Baltic Sea coastal areas, such as Perch (*Perca fluviatilis*), which was described in the section above. The index is based on the metric *mean length of Perch*, but the metric *abundance of large Perch* could be used as additional information or complement.

The estimate *mean length of Perch* is expected to reflect changes in recruitment success as well as in mortality. Low levels may signal the appearance of a strong year class of recruits, decreased top down control in the ecosystem, or high fishing mortality, but potentially also density-dependent growth. High levels in the indicator may signal a high trophic state, but potentially also decreased recruitment success. Because of this, the indicator should be interpreted together with the Species Abundance Index.

The estimate *abundance of large Perch* is calculated as the catch per unit effort of Perch larger than 25 cm. The estimate is expected to provide a more direct measure of fishing mortality on the actual species, as well as of ecosystem health. The index will, however, not be indicative of changes in the younger year-classes of the population.

Policy relevance: The index will show if the size structure of Perch is at an appropriate level for supporting coastal ecosystem function, including food provision with a focus on trophic state. In areas where the index does not signal good environmental status, GES may be achieved mainly by regulating fishing pressure.

Coastal Fish - Community Diversity Index (D1.6.1)

The index reflects biological diversity at the community level and is based on the Shannon Index. High values reflect high species richness and low dominance of single species, whereas low values reflect the opposite.

The index has both an upper and a lower boundary since very high levels of the index potentially also may reflect a decrease in the abundance of a naturally dominating species.

Policy relevance: The index will indicate whether the biodiversity structure of coastal fish communities is at an appropriate level for supporting coastal ecosystem function, including ecosystem resilience, or not. The index reflects the general state of the fish community. In areas with sub-GES conditions, actions to achieve good ecological status should target the species level.

Coastal Fish - Community Size Index (D1.6.1)

The index reflects the general size structure at the community level and is based on estimates of the abundance of large fish (measured as catch per unit effort). Depending on the maximum mesh size of the gear, large fish are defined as individuals larger than 30 (Net series) or 40 (Coastal survey nets, Nordic coastal multi-mesh gillnets) cm.

Generally, large fish are abundant in coastal communities indicative of good ecological status in the Baltic Sea (Pauly et al. 1998). The index is expected to mainly reflect changes in fishing mortality at the community level, where low values reflect increased fishing mortality. However, the value of the index may to some extent also be influenced by environmental conditions such as temperature and nutrient status.

Policy relevance: The index will indicate if the size structure of coastal fish communities is at appropriate level for supporting coastal ecosystem function, including food provision and ecosystem resilience. In areas where the index does not signal good environmental status, GES may be achieved by mainly regulating fishing pressure.

Coastal Fish – Community Abundance Index (D1.6.2)

The index is based on estimates of the abundance of two different functional groups: *Abundance of Cyprinids* and *Abundance of Piscivores*, and reflects the integrated effects of recruitment and mortality of the species included in each functional group. Recruitment success is expected to mainly be influenced by the quality and availability of recruitment habitats, climate and eutrophication. Mortality is influenced by fishing, but predation from other animals, such as seals, sea birds and fish are also perceivable. The two metrics included in the index are expected to differ in their responses to anthropogenic pressure factors, in that *Abundance of Cyprinids* is expected to show the strongest link to eutrophication and *Abundance of Piscivores* the strongest relationship to fishing pressure.

Policy relevance: The index will indicate whether or not the abundance and productivity of coastal fish communities is at appropriate level for supporting coastal ecosystem function and resilience, including food provision for man and other marine organisms. In areas where the index does not signal good environmental status, GES may be achieved by restoration of recruitment habitats for piscivores, reduction of nutrient loads, and by regulating mortality of piscivores by reducing fishing pressure and predation from apex predators.

Coastal Fish - Community Trophic Index (D1.7.1)

The index reflects the general trophic structure at the community level and is based on estimates of the proportion of fish at different trophic levels. Alternatively, estimates of the proportion of piscivores in the fish community may be used.

The index provides an integrated measure of changes in the trophic state of the fish community. Typically, very low values of the index may reflect high fishing pressure on piscivores (Pauly et al. 1998) and/or domination of species favoured by eutrophic conditions. Since high levels of the index also may reflect a decreased abundance of some naturally dominating non-piscivore species the index has both an upper and a lower boundary.

Policy relevance: The indicator will show if the trophic structure of the coastal fish community is at appropriate level for supporting coastal ecosystem function, including ecosystem resilience. The indicator reflects the general state of the fish community, and is likely to be closely linked to fishing pressure as well as eutrophication. In areas with sub-GES conditions, actions should target the species level.

Method(s) used to test the indicators

The metrics on which each index is based have previously been used for assessing the state of coastal fish communities in Sweden, Finland, Estonia, Latvia and Lithuania based on data coastal fish monitoring programs (as analyzed within the HELCOM Fish PRO group; HELCOM 2011).

The suggested metrics were identified after evaluation of a range of parameters potentially reflecting coastal fish community status. The selection of metrics was based on multivariate analyses (PCA) of monitoring data where the signal strength, biological relevance and redundancy of individual parameters as suggested by Rice and Rochet (2005) were assessed (as analyzed within the HELCOM Fish PRO group; HELCOM 2011). In addition, the biological relevance of the metrics was mainly evaluated based on empirical observations.

Relationship to anthropogenic pressures

The relationship between metrics and anthropogenic pressures was assessed using a data set covering the Bothnian Bay, Bothnian Sea, Gulf of Finland and the Central Baltic Sea. Samples covered areas with different levels of natural and anthropogenic environmental pressures, and the relationship was analysed using distance-based linear modeling (DISTLM as implemented in PERMANOVA+ of PRIMER v6). The results in combination with expert judgments provided a basis for a preliminary concept of the relationship between the indicators and the pressures Fishing pressure and Eutrophication (see **Figure 2.17**). The relationships will, however, be evaluated further.

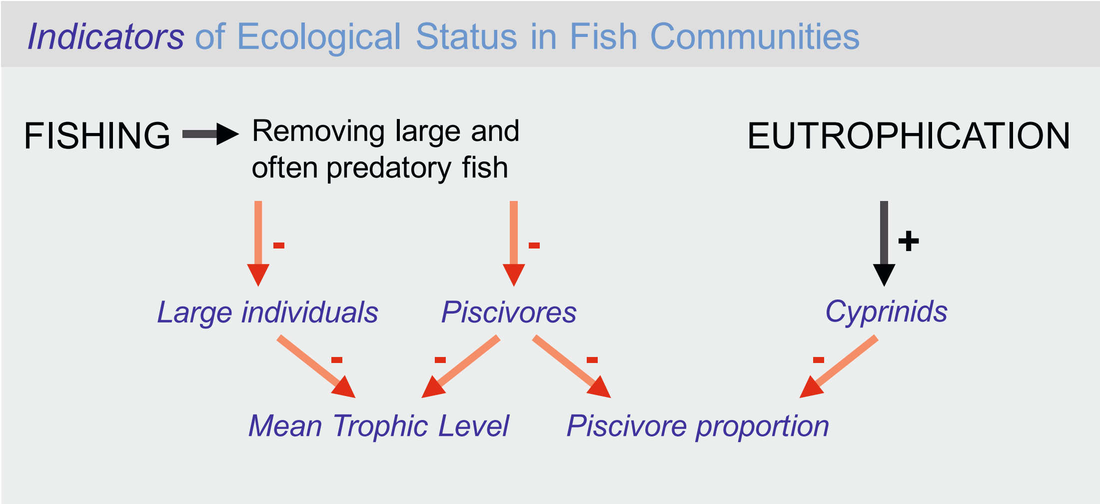


Figure 2.17. *Relations of indicators and pressures.*

Other considerations

In addition to the anthropogenic factors included above, the indices might potentially also be influenced by changes in ambient environmental conditions, such as temperature and salinity levels (Olsson et al., submitted). As such, these effects should be considered in the assessment of coastal fish community status (i.e. Carstenssen 2007). The suggested approach for achieving this is presented in subsection 8.

Approach for defining GES

Analyses of the indices with respect to their temporal and spatial variation in the Baltic Sea showed that the expected values (in the absolute sense) are typically site specific, depending on local properties of the ecosystem such as topography and geographical position (as analyzed within the HELCOM Fish PRO group; HELCOM 2011). Values are also highly dependent on the monitoring method used. This implies that defining a common GES boundary for a larger region should not be attempted, and that the GES boundary rather should be identified separately for each data set. In the suggested approach for achieving this, as presented below, the assessment of the ecological status of coastal fish communities is based on time series data where the state of the assessment period is contrasted against a reference data set. Because the data set does not reach long in the past, the reference data set does not represent necessarily "pristine conditions".

Definition of the reference data set

For analyses of time series, the reference data set is defined by the following criteria

– The minimum number of years to be included is at least two times the generation time of the species mainly influencing the index. The total number of years included is maximized based on available data, but does not include the years of the assessment period.
– No significant trends of the reference data set are present.
– The reference data set is judged by expert opinion in light of prevailing physiographic, geographic and climatic conditions and as either representing GES or sub-GES conditions, based on concurrent trends in the other related indicators or historical information.

When a reference data set cannot be defined

If a reference data set cannot be identified based on the above criteria, trends in the available data are used to assess the environmental status. Prior to the assessment, the available data (not including the assessment period) in combination with other relevant information, is judged by expert opinion as either representing GES or sub-GES conditions.

Criteria for GES within the assessment period

In case of reference data set representing GES conditions

In cases when a reference data set representing GES is used, for GES the median value of the indicator during the assessment period must be above the 5th (or within the 5th and the 95th, depending on the index) percentile of the median distribution of the reference data set. The trend within the assessment period is given as supplementary information.

If a reference data set cannot be identified, the long term trend in the whole monitoring data set is analyzed. For GES, the slope must not be significantly negative (or positive depending on the indicator).

In case of reference data set representing Sub-GES conditions

In cases when a reference data set for sub-GES is used, for GES the median value of the indicator during the assessment period must be above the 99th percentile of the median distribution of the reference data set.

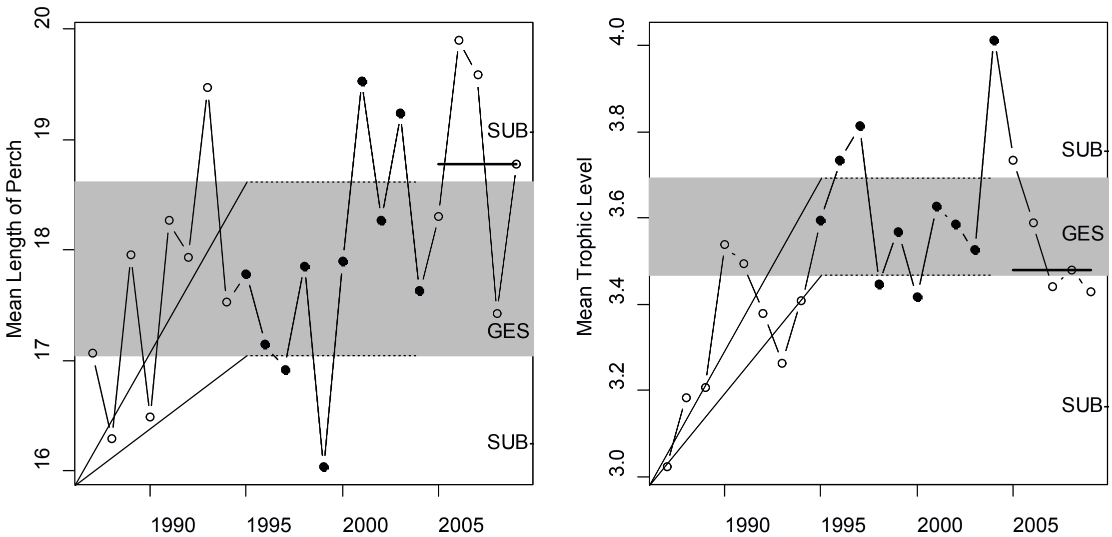
If a reference data set cannot be identified, the long-term trend in the whole monitoring data set is analyzed. For GES, the slope must be significantly positive (or negative depending on the indicator).

Proposed GES/subGES boundaries

As described in section 2, the anticipated GES/subGES boundaries are data dependent and will be different for different monitoring areas and methods. The suggested approach for defining the GES boundary was described above. In **Figure 2.18**, examples for GES determination in one area are provided for the indicators described earlier in this document.

Kvädöfjärden

Species Demographic Index *Community Trophic Index*



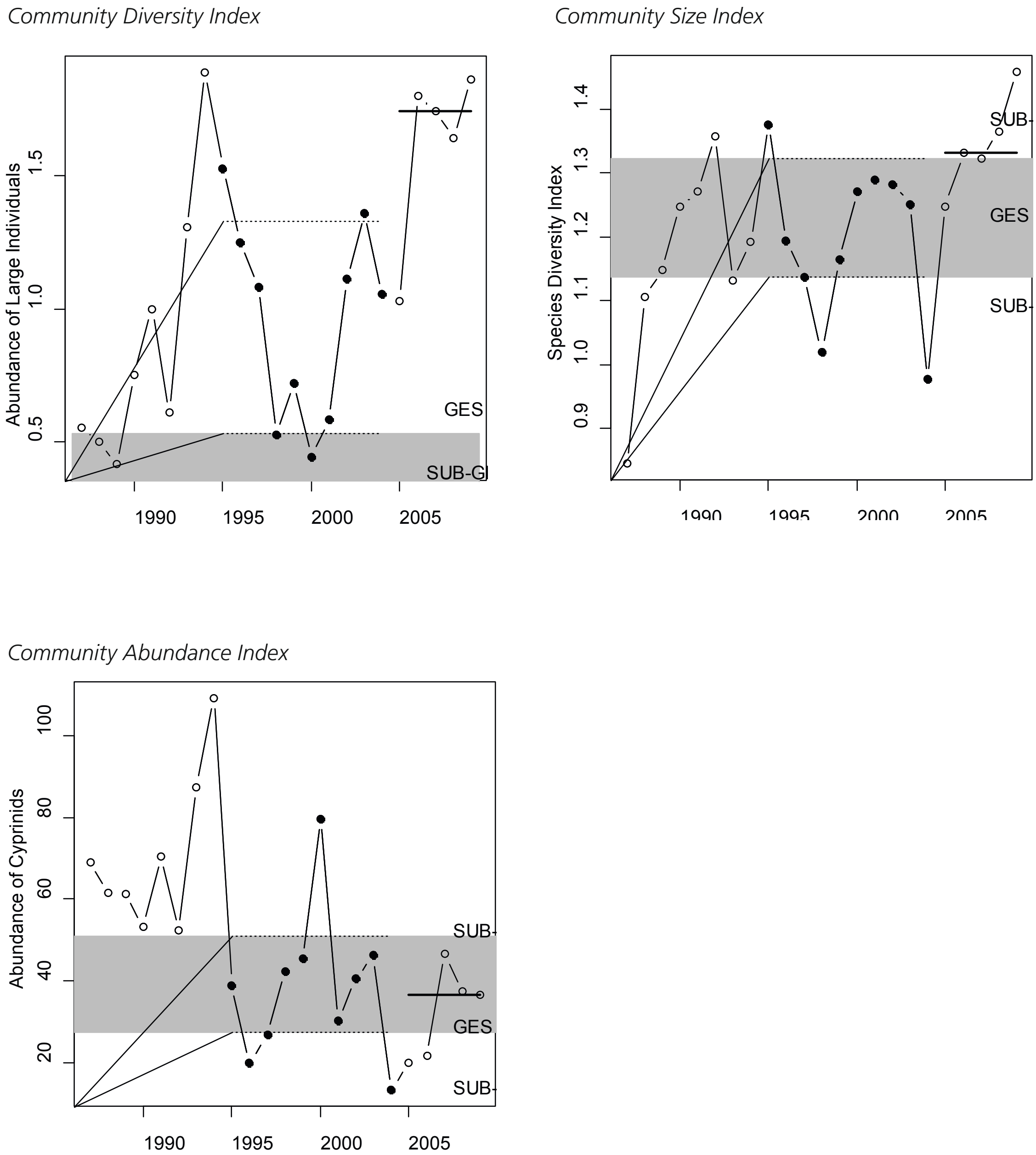


Figure 2.18. *Results of the six core indicators in Kvädöfjärden, Sweden. The range of GES has been marked by grey.*

Out of the five indices outlined above, two indicate GES (Community Trophic Index and Community Size Index), whereas three indicate subGES in that the diversity (Community Diversity Index) and demographic characteristics (Species Demographic Index) is above the expected distribution of the reference data set, and the abundance of piscivores (Community Abundance Index) is below the expected distribution (**Figure 2.18**). Further testing of the suitability of the boundaries for GES will be elaborated in the nearest future.

Existing monitoring data

Coastal fish monitoring is performed annually all over the Baltic Sea (**Table 2.12)**. The HELCOM Fish PRO Project includes data from monitoring areas in Finland, Estonia, Latvia, Lithuania and Sweden (HELCOM 2011). Coastal fish communities in the Baltic Sea areas of Denmark, Poland and Russia are monitored as well, but were not included in the present evaluation of metrics used for assessing coastal fish community status (as analyzed within the HELCOM Fish PRO group). The longest time series is 22 years, but several were initiated as late as in the 2000s.

Table 2.12. Existing monitoring areas of Coastal Fish in the Baltic Sea. Years in parentheses indicates last year of monitoring.

Area	Country	Basin	Nordic coastal multi-mesh net	Coastal survey nets	Net series
Råneå	Sweden	Gulf of Bothnia	2002	1994 (2004)	
Kinnbäcksfjärden	Sweden	Gulf of Bothnia	2004		
Holmön	Sweden	Gulf of Bothnia	2002	1989	
Norrbyn	Sweden	Gulf of Bothnia	2002		
Gaviksfjärden	Sweden	Gulf of Bothnia	2004		
Långvindsfjärden	Sweden	Gulf of Bothnia	2002		
Haapasaaret	Finland	Gulf of Finland	2003 (2008)		
Kaitvesi	Finland	Gulf of Bothnia	2005		
Helsinki	Finland	Gulf of Finland	2005		
Forsmark	Sweden	Gulf of Bothnia	2002	1987	
Finbo	Finland	Gulf of Bothnia	2002	1991 (2008)	
Kumlinge	Finland	Gulf of Bothnia	2003		
Brunskär	Finland	Gulf of Bothnia	2002	1991 (2004)	
Tvärminne	Finland	Gulf of Finland	2005		
Lagnö	Sweden	Baltic Proper	2002		
Hiiumaa	Estonia	Baltic Proper			1991
Asköfjärden	Sweden	Baltic Proper	2005		
Kvädöfjärden	Sweden	Baltic Proper	2001		1987
Vinö	Sweden	Baltic Proper			1995
Daugavgriva	Latvia	Gulf of Riga			1995
Jūrkalne	Latvia	Baltic Proper			1999
Torhamn	Sweden	Baltic Proper	2002		
Curonian lagoon	Lithuania	Baltic Proper			1994

Sampling

The coastal fish monitoring takes place in August and reflects trends in species that occur in coastal areas during the warm season of the year. Fishing is performed using survey nets that mainly target demersal and benthopelagic species, but some pelagic species are also captured (HELCOM 2008).

Three different monitoring methods are used around the Baltic Sea (HELCOM 2008). In the Baltic Proper, the longest time series data are from monitoring using Net series, which consists of four 30 m long and 1.8 m deep nets. Each net is made up of a single mesh size, 17, 21.5, 25 and 30 (in Latvia also 38) mm (knot to knot), respectively. In the Bothnian Sea, Coastal survey nets are used. These nets are 35 m long, 3 m deep and are composed of five 7 m long panels with mesh sizes 17, 21, 25, 33 and 50mm (knot to knot). In both methods, fishing is repeated three nights at each position.

Sampling method using Nordic coastal multi-mesh nets was introduced in 2001 (Appelberg et al. 2003). This gear is 45 m long, 1.8 m deep and is composed of nine mesh sizes (10, 12, 15, 19, 24, 30, 38, 48 and 60 mm, knot to knot). A random depth-stratified sampling design is applied. Typically, 45 positions are distributed over four different depth intervals; 0-3 m, 3-6 m, 6-10 m and 10-20 m. Each position is fished with one net for one night. A minimum of 10 stations are fished in each depth interval down to 10 meters depth, and a minimum of 5 stations in the deepest depth interval. The method is currently used only in Finland and Sweden, where it is routinely used in all areas established after 2001.

Monitoring by all methods is performed at fixed stations. The gears are set between 14.00 and 16.00 and lifted the next day between 07.00 and 10.00. Catches at each station are registered as numbers per species and length group (2,5 or 1 cm), separately for each mesh size (or net). Additionally, wind strength and direction, water temperature, and water transparency measured using a Secchi disc, are routinely monitored during the fishing period.

Additional data sources such as commercial catch statistics based on EU-data collection system, coastal echo sounding, etc. might serve as a complement if the data proves to be of enough quality to assess the biodiversity of coastal fish communities.

Methodology of data analyses

Index computation

Species Abundance Index. The indicator is estimated as the catch per unit effort of the key species (i.e. Perch) within the target size range of the gear used.

Species Demographic Index. The indicator is estimated as the mean length of all individuals of the key species (i.e. Perch) within the target size range of the gear used and/or the catch per unit effort of all *Perca fluviatilis* equal to or above 20 cm length.

Community Diversity Index. The indicator is the Shannon index calculated based on catch per unit effort of all species targeted by the gear used and within its target size range.

Community Size Index. The indicator is the catch per unit effort of individuals equal to or larger than 30 (Net series) or 40 (Coastal survey nets and Nordic coastal multi-mesh nets) cm of all species targeted by the gear used, within its target size range.

Community Abundance Index. The index includes measures of catch per unit effort of the groups of Cyprinids and Piscivores, respectively. The group Cyprinids includes all species within the Cyprinidae, and the group piscivores includes all species with a trophic level equal to or higher than 4.0 according to Fish Base (www.fishbase.org).

Community Trophic Index. The index represents the mean trophic level of the community and is based on catch per unit effort of all species targeted by the gear used, within its target size range. The trophic level of each species is based on values from Fish Base (www.fishbase.org), and the index is calculated as Σ (Trophic level x Relative Abundance) species i. As a complement, the proportion of piscivores is used, defining piscivores as species with a mean trophic equal to or above 4, according to Fish Base.

Target species and size range

In order to only include species and size-groups suited for quantitative sampling by the method, individuals smaller than 12 cm (Nordic Coastal multimesh nets) or 14 cm (Net series, Coastal survey nets), and all small-bodied species (gobies, sticklebacks, butterfish), and species with eel-like body forms (taeniform, anguilliform or filiform shapes) are excluded (HELCOM 2011).

Weighting

For Nordic Costal multimesh nets, the analyses are based on weighted means of data from all depth strata, in order to account for the depth-stratified sampling design.

Corrections

For all data sets, the relationship between of the indicators to local temperature and salinity was checked prior to further analyses. If the correlation was significant, further analyses were performed based on the remaining variation (regression residuals).

Weaknesses/gaps

The current monitoring program does not provide full spatial coverage, and is mainly targeting areas with relatively low levels of direct anthropogenic influence. Currently, methods for extrapolating the results to areas without monitoring are under development. Given the current state of knowledge, it is suggested that the status classifications achieved for monitored areas are extrapolated to areas without monitoring by direct interpolation in combination with expert opinion. Alternatively, additional data sources such as commercial catch statistics based on EU-data collection system, coastal echo sounding, etc. might serve as a complement if the data proves to be of enough quality to assess the biodiversity of coastal fish communities. Extended data collection with a focus on polluted areas is being performed in 2011 and will bring further knowledge on natural environmental factors driving spatial variation in indicator values and the relationship of indicators to anthropogenic pressure factors, such as eutrophication and fishing pressure.

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2.13. Multimetric macrozoobenthic indices

<table>
<tr><td colspan="2">1. Working team
Benthic habitats and associated communities</td></tr>
<tr><td>2. Name of core indicator
Multimetric diversity indices of benthic macrofauna communities (B, BBI, BQI, DKI, MarBIT, ZKI) or species richness</td><td>3. Unit of the core indicator
Unitless (but based on abundance, ind./m2, data)</td></tr>
<tr><td colspan="2">4. Description of proposed indicator
The indicator shows the diversity of benthic macroinvertebrate communities. The indicator can be calculated by various indices, which all have their own GES boundaries (e.g. relation of sensitive, tolerant species, abundance and taxonomic composition).
The indicator describes the condition of the biological component/habitat.
The species are sensitive to general contamination of the sediment, physical disturbance and to hypoxic events.
It is suggested that the offshore areas of the Baltic Sea are measured by sub-basin wide species richness based on the work by Vilnäs & Norkko (2011), until the applicability of the benthic quality index has been tested. This species richness indicator has also been used as an eutrophication core indicator.</td></tr>
<tr><td colspan="2">5. Functional group or habitat type
all soft sediment habitats (hydrolittoral, infralittoral, circalittoral, below the halocline)</td></tr>
<tr><td colspan="2">6. Policy relevance
Descriptor 1, criterion 1.6 Habitat condition
Descriptor 6, criterion 6.2 Condition of the benthic communities
BSAP: Ecological objectives “Natural distribution and occurrence of plants and animals” (Eutrophication) and “Thriving communities of plants and animals” (Nature conservation)</td></tr>
<tr><td colspan="2">7. Use of the indicator in previous assessments
In the WFD assessments, HELCOM thematic assessments of eutrophication and biodiversity.</td></tr>
<tr><td colspan="2">8. Link to anthropogenic pressures
Faunal communities are adversely affected by the eutrophication, changes in water and sediment quality and hydrographic conditions such as salinity, temperature. Therefore coastal construction works or water outflows affecting hydrography and general decrease in water and sediment quality due to increased eutrophication, turbidity, silt content and input of hazardous substances decrease the species richness of macrozoobenthos at sites.
Slight eutrophication improves species richness and diversity, but severe eutrophication reduces diversity, making it therefore difficult to use this indicator for an assessment – direct/indirect impact
Physical disturbance (due to abrasion, smothering, changes in siltation) reduces the species richness and diversity – direct impact
Physical loss (due to sealing or selective extraction) reduces the species richness and diversity – direct impact
Introduction of synthetic compounds (due to ship accidents or harbours) reduces the species richness and diversity – direct impact
Changes in the hydrological conditions (due to changes in salinity and/or temperature) reduces the species richness and diversity – direct impact</td></tr>
<tr><td colspan="2">9. Pressure(s) that the indicator reflect
Input of fertilizers and organic matter, Introduction of synthetic compounds, Introduction of non-synthetic substances and compounds, Introduction of radio-nuclides, Changes in siltation, Abrasion, Smothering, Sealing, Selective extraction of seabed resources, Changes in salinity regime, Changes in thermal regime.</td></tr>
<tr><td colspan="2">10. Spatial considerations
Baltic wide but with sub-regional references.</td></tr>
<tr><td colspan="2">11. Temporal considerations
Annual frequency.</td></tr>
</table>

12. Current monitoring Monitored by all Contracting States.
13. Proposed or perceived target setting approach with a short justification. Targets (used here for GES boundaries) for the indices in coastal waters have been set by the Contracting Parties under the EU WFD (Baltic GIG). Targets in the offshore have been proposed in Vilnäs & Norkko (2011).

Introduction

The multimetric macrozoobenthic indices have been developed under the EU Water Framework Directive in the EU Member States. The limitation of these indices is their applicability to national territorial waters only. To fill this gap, HELCOM thematic assessment of eutrophication (HELCOM 2009) presented an offshore indicator for macrozoobenthic invertebrates, which was developed by Villnäs and Norkko (2011).

In this brief summary of the macrozoobenthic indices, only references to national work has been given and the second phase of the CORESET project will fully apply these indeces when the outcome of the Baltic wide intercalibration work under the Eu Water Framework Directive has been published.

Offshore indicator: sub-basin wide species richness

The offshore areas of the Baltic Sea were agreed in the CORESET Biodiversity Expert Group to be provisionally assessed by the indicator, which calculates a sub-basin wide species richness, i.e. gamma diversity. The indicator and its GES boundaries have been described by Villnäs & Norkko (2011) and in the HELCOM core indicator report for eutrophication. The indicator was also used in the HELCOM thematic assessment of eutrophication (HELCOM 2009).

The HELCOM TARGREV project has further developed the species richness indicator and linked it to environmental conditions, such as hypoxia.

The GES boundaries for the indicator in the sub-basins were set on the basis of historical data and standard deviations from those (Villnäs & Norkko, 2011). Reference values and acceptable deviations for the indicator were based on long-term monitoring data at >200 monitoring stations during 1964. Data from ~1800 sampling occasions was used. Generally only stations with a depth >40 m were included and anoxic and/or hypoxic periods (<2 mL O2/L) were excluded from the data. The reference value for each sub-area was identified as the average of the 10% highest annual average regional diversity values during the monitoring period.

Acceptable deviation from reference conditions determines the Goodle deviation from reference conditi critical border between an acceptable and non-acceptable condition of benthic diversity (cf. the EU Water Framework Directive) and it should incorporate natural variation and decadal time-scale fluctuations of species numbers in an area. Based on the long-term data used for identifying reference conditions, the acceptable deviation was defined as the relative standard deviation of average regional diversity in a sub-area per year. An average acceptable deviation for each sub-area was based on data from several years. The highest acceptable deviation allowed was set to 40%.

The indicator responds mainly to the anthropogenic eutrophication, which causes hypoxia and anoxia in bottom waters (Pearson and Rosenberg 1978, Hyland et al. 2005, Norkko et al. 2006). The indicator reflects the increase in nutrient levels only indirectly and therefore the quantitative relationship to nutrient levels is difficult to ascertain. The relationship has however been found in the Bothnian Bay, where the background concentrations of nutrients are relatively low (HELCOM 2009). Increasing amounts of nutrients were seen to result in a surplus of organic material, leading to large fluctuations in benthic diversity as sensitive, large-sized and long-lived species did not tolerate the altered consitions. At more advanced stages of organic enrichment, the diversity starts to decline. The single strongest factor influencing the benthic diversity is, however, hypoxia.

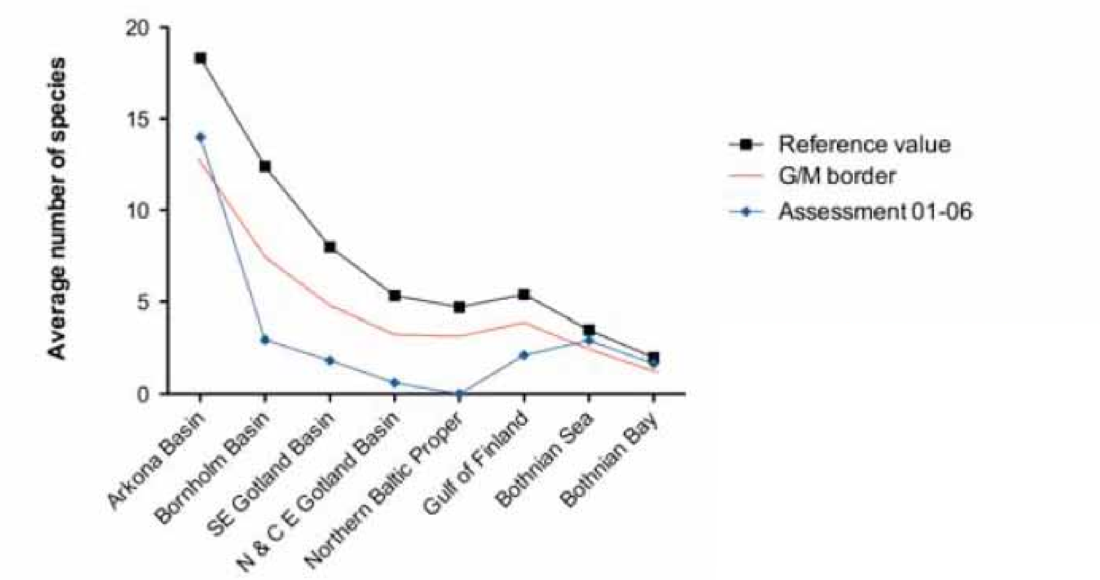

Figure 2.19. *The reference values, good-moderate border and assessment 2001-2006 of the offshore macrozoobenthos indicator. Source: HELCOM 2009.*

The national indices and the used sieve size for sampling

– Finland: Brackish Water Benthic Index BBI (BQI and shannon), 5 pooled Ekman grabs, 0.5 mm sieve.
– Reference: Perus et al. (2007).
– Estonia: ZKI (biomass based), 0.25 mm sieve (Anon. 2007)
– Latvia: BQI, Van Veen, 0.5 mm sieve
– Lithuania: BQI (and number of species per sample), Van Veen, 0.5 mm sieve
– Poland: B, 1 mm sieve(?)
– Germany: MarBit, vanVeen, 1 mm sieve, Kautsky frame, 1 mm sieve
– Denmark: DKI v2 (salinity corrected), Van Veen and Haps, 1 mm sieve
  – Reference: Josefson et al. 2009
– Sweden: Benthic Quality Index BQI, Van Veen, 1 mm sieve
  – Reference: Blomqvist et al. (2007)

Intercalibration of the coastal indices in the Baltic GIG

An overview of the intercalibration process is described in Carletti and Heiskanen (2009). The report describes all the national methods, gives background information of their responses to anthropogenic pressures, compares them to each other and describes the target setting procedures and the targets themselves.

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2.14. Lower depth distribution limit of macrophyte species

<table>
<tr><td colspan="2">1. Working team
Benthic habitats and associated communities</td></tr>
<tr><td>2. Name of core indicator
Lower depth distribution limit of macrophyte species</td><td>3. Unit of the core indicator
m (valid for last specimen or minimum coverage value of 10%)</td></tr>
<tr><td colspan="2">4. Description of proposed indicator
This indicator follows the depth distribution limits (last specimen or minimum coverage value of 10%) of specific sensitive species, which are important for habitat structure (key-species) and perennials with a persistent biomass for a certain time scale adapted to stable conditions (K-Strategy).
The indicator describes the condition and abundance of a habitat-forming macrophyte species and indirectly the condition for all the associated flora and fauna. The deeper the macrophyte extends, the larger is the volume of the habitat, thus, supporting more viable populations, having more diverse species assemblages, offering feeding grounds for predators and ensuring enough spawning grounds for pelagic fish.
As habitat builders, these macrophyte species are the representative indicators for all species (macrophytes, epifauna, small fish) associated to this kind of habitat/phytal community.
The indicator has been established in the coastal waters of the EU Member States, whereas it does not yet contain the offshore waters (reefs) where the GES boundary needs to be estimated.</td></tr>
<tr><td colspan="2">5. Functional group or habitat type
Infralittoral hard substrata
Infralittoral sediments</td></tr>
<tr><td colspan="2">6. Policy relevance
Descriptor 1, Criteria 1.5 Habitat extent and 1.6 Habitat condition
Descriptor 5, Criterion 5.1 Direct effects of nutrient enrichment
Descriptor 6, Criterion 6.1 Kind and size of relevant biogenic substrata
BSAP: Ecological objectives "Natural distribution and occurrence of plants and animals" (Eutrophication) and "Thriving communities of plants and animals" (Nature conservation)</td></tr>
<tr><td colspan="2">7. Use of the indicator in previous assessments
WFD assessment in Denmark (eelgrass), Estonia (Fucus), Finland (Fucus), Germany (Fucus, eelgrass, charophytes), Lithuania (Furcellaria– for coastal and transitional waters along the Baltic coast; Potameids – for transitional waters (the Curonian lagoon) and Sweden (several species). The intercalibration of the national indicators was not finalized.
Has been used in the HELCOM thematic assessments of eutrophication and biodiversity, but the geographical distribution of the indicator was limited.</td></tr>
<tr><td colspan="2">8. Link to anthropogenic pressures
Reduced water transparency (due to input of fertilizers, organic matter and physical disturbance-increased turbidity) causes a reduced light availability and a reduction in depth distribution of macrophytes – indirect impact.
Increased nutrient availability increases the abundance of opportunistic algae growing also on perennial algae and overshadowing them.
Physical disturbance (due to abrasion, smothering, changes in siltation) reduces the photosynthetic rates of macrophytes resulting in a loss at the lower depth limit (reduction in depth distribution) – direct impact.</td></tr>
<tr><td colspan="2">9. Pressure(s) that the indicator reflect
Input of fertilizers, Input of organic matter, Changes in siltation , Abrasion, Smothering</td></tr>
</table>

10. Spatial considerations *Fucus vesiculosus* – Baltic-wide (except inner Gulf of Finland and the Bothnian Bay). Charophytes: Baltic-wide in sheltered habitats with reduced salinities (lagoons). *Furcellaria*: Baltic-wide (except inner Gulf of Finland and Northern part of Bothnian Bay). Eelgrass: from Kattegat to Archipelago Sea and mid-Gulf of Finland (to mean surface salinity of 6–8 psu), missing from the Polish, Lithuanian and Latvian open coasts. Other angiosperms (*Ruppia, Potamogeton*): Baltic-wide in sheltered habitats with reduced salinities (lagoons).
11. Temporal considerations Annual data should be aimed at, but less frequent data may be also adequate as the changes in the depth distribution may take years. Harmonization of the sampling method may be required.
12. Current monitoring Part of WFD monitoring in several EU Member States (see 7).
13. Proposed or perceived target setting approach with a short justification. Reference values and classification can be achieved by light models and/or historical data but the direct correlation of the indicator to the pressure can be interfered by biological competition for space. Inter-calibration of the targets (here used as GES boundaries) was initiated but not finalized in the EU expert group for Water Framework Directive 'Baltic GIG'. They are based on old data sets (Reference conditions + Acceptable deviations) and expert judgment. Targets for some species have been estimated on the basis of wide datasets in varying environmental conditions. A slow-reacting indicator due to long recovery time of perennials.

Introduction

The core indicator for macrophytes is an indicator for coastal waters and shallow offshore waters (i.e. off-shore reefs). Various macrophyte indicators have been developed under the EU Water Framework Directive, which apply to national territorial waters. The national indicators or indexes cover not only depth distribution parameters but some also include parameters for coverage, species proportions, species richness or densities. In order to have a joint parameter to the set of core indicators, the HELCOM CORESET project decided first to focus on the depth distribution core indicator. This is measured currently by at least 7 of the 9 countries. Because extensive reports are available for the national indicators and methodologies, this short description of the indicator is only referring to them. When the outcome of the Baltic intercalibration work under the EU WFD is published, the national approaches will be presented in detail and differences and similarities in methodologies will be discussed more thoroughly.

National approaches to assess macrophytes

Danish and German intercalibration of angiosperms is presented in Carletti & Heiskanen 2009.

Denmark:
– Eelgrass depth limit, the depth at the 5% cover (Krause-Jensen et al. 2005)
– Total erect macroalgae cover
– Number of perennial macroalgae species
– Ratio of opportunistic macroalgae (% cover)

Estonia:
– Estonian Phytobenthos Index EPI follows the maximum depth limit of the attached macrovegetation, including bladderwrack.
– Proportion of perennial species.
–

Finland:
– Bladderwrack lower depth limit

Germany:
– Lower depth limit of stands of >10% cover (Schories et al. 2006)
– BALCOSIS is a multimetric index, which was developed for outer coastal waters (type B3), and includes a parameter for depth distribution (>10% density) (for further description see http://www.marilim.de/informationen-wrrl/balcosis.php).
– ELBO is a composition indicator, which was developed for inner coastal waters, and includes a parameter for depth distribution (Steinhardt et al. 2009).

Poland:
– a biomass x cover indicator for 18 species (MQAI) (Osowiecki et al. 2010).
– Latvia: no method

Lithuania:
– Furcellaria depth limit in the open coast
– Potamogeid depth limit in the Curonian lagoon

Sweden:
– multispecies maximum depth index (maximum depth for 3-9 species)

References

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Schories D, Selig U, Schubert H. 2006. Testung des Klassifizierungsansatzes Mecklenburg-Vorpommern (innere Küstengewässer) unter den Bedingungen Schleswig-Holsteins und Ausdehnung des Ansatzes auf die Außenküste. Küstengewässer-Klassifizierung deutsche Ostsee nach EU-WRRL. Teil A: Äußere Küstengewässer.

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2.15. Trend in the arrival of new non-indigenous species

1. Working team: Non-indigenous species *Authors:* Manfred Rolke, Maiju Lehtiniemi, Malin Werner, Alexader Antsulevich, Monika Michalek-Pogorzelska. *Acknowledged persons:* Paulina Brzeska, Elena Gorokhova, Bożenna Kaczmaruk and Solvita Strake	
2. Name of core indicator Trend in the arrival of new non-indigenous species	*3. Unit of the candidate indicator* number of new arrivals against a baseline per six-years assessment period
4. Description of proposed indicator The indicator follows numbers of non-indigenous species found in an assessment area within an assessment period of six years. The indicator requires a baseline study, identifying the number of already arrived non-indigenous species. Every new non-indigenous species (NIS) arriving after the baseline year is counted as a new species. New NIS comprises not only established organisms but all new identified species even if they will not establish (because species which cannot establish stable populations, as also short living introductions, will be regarded as a failed management). New NIS that have already been counted will not be added to the baseline and not be added to the counts of future periods.	
5. Functional group or habitat type From phytoplankton to vertebrates. Data sampled from harbours, main shipping lanes and anchoring places and nature conservation areas	
6. Policy relevance Descriptor 2 BSAP ecological objective: no new introductions of non-indigenous species.	
7. Use of the indicator in previous assessments None	
8. Link to anthropogenic pressures Directly impacted by: Ballast water exchange, introductions for mariculture or aquaria, biofouling, unintentional introductions by pleasure boats.	
9. Pressure(s) that the indicator reflect The indicator reflects mainly the impact of shipping, but it is difficult to distinguish that from other vectors of NIS.	
10. Spatial considerations The information should ideally cover the assessment units (national coastal and offshore waters divided to sub-basins) but at least the sub basins separately and include harbours, main shipping lanes, anchoring places and nature conservation areas	
11. Temporal considerations The indicator should be assessed every six years, but data should be collected continuously.	
12. Current monitoring The COMBINE biological monitoring can be used as the basis. Gaps should be covered by strengthening the focus on new species identification and by extending the areal coverage as needed, particularly on areas near (or in) harbors, shipping lanes and anchoring sites.	
13. Proposed or perceived target setting approach with a short justification. GES boundary refers to the BSAP objective "No new introductions" and is based on the implementation of the IMO Ballast Water Convention. GES is reached when no new introductions are found per assessment period.	

Introduction

Draft Example for German, Swedish and Polish waters
- Data must be updated and other countries included

The introduction of invasive species into oceanic waters and especially coastal waters is among the four highest risks for our marine environment and can cause extremely severe environmental, economic and public health impacts.

These non-indigenous invaders can induce considerable changes in the structure and dynamics of marine ecosystems. They may also hamper the economic use of the sea or even represent a risk for human health. Environmental impacts comprise changes of marine communities changing e.g. the structure of the food web by outcompeting original inhabitants. Economic impacts range from financial losses in fisheries to expenses for cleaning intake or outflow pipes and structures from fouling. Public health impacts may arise from the introduction of microbes or toxic algae.

Different vectors are made responsible for human introduced non-indigenous species (NIS). In some cases, NIS have been deliberately introduced for fishing or aquaculture, but most have been brought by ships, which can rapidly transport aquatic animals, plants and algae across the world in their ballast waters and attached to their hulls.

A problem for the non-indigenous species issue is that, once a marine organism has been introduced to its new environment, it is nearly impossible to eradicate the unwanted organism, if it has established to the area. The consequence is that assessing a status of an area as "bad" means that the area will stay in the bad status without the possibility to return to the past condition. Therefore the status of the NIS in the Baltic Sea is described by a trend of the number of new arriving NIS. The number of new species in each assessment unit during a six-year assessment cycle is first of all an indication of the pressure the NIS cause on the native ecosystem, but it is also an indication of management success with the IMO Ballast Water Convention and ballast water treatment and it encourages to take measures against new invaders instead of futile effort or fatalistic inactiveness in view of effects of established ones.

The indicator uses primarily species data from the conventional biodiversity monitoring programmes, but also additional information about new arrivals must be gathered, for example, harbours or in the vicinity of main shipping lanes where ships exchange their ballast water or in the vicinity of areas used for aquaculture. The impact on native communities, another indicator requested by MSFD, will be automatically covered by monitoring for other descriptors or for WFD assessment of environmental status.

Policy relevance

Since the early 90s when the Marine Protection Committee (MEPC) of the International Maritime Organisation (IMO) put the NIS issue on the agenda, the problem got more and more weight in marine environmental protection. In 2004, the Ballast Water Convention was adopted by the IMO. The convention asks for ballast water management procedures to minimize the proliferation of non-indigenous organism with ballast water. Once entered into force every ship has to follow ballast water management procedures.

In order to minimize adverse effects of introductions and transfers of marine organisms for aquaculture ICES drafted the "ICES Code of Practice on the Introductions and Transfers of Marine Organisms". The Code of Practice summarizes measures and procedures to be taken into account when planning the introduction of non-indigenous species for aquaculture purposes. On the European level, the EC Council Regulation No 708/2007 concerning the use of NIS and locally absent species in aquaculture is based on the ICES Code of Practice.

With the maritime activities segment of the Baltic Sea Action Plan HELCOM expresses the strategic goal to have maritime activities carried out in an environmental friendly way and that one of the management objectives is to reach “No introductions of alien species from ships”. In order to prepare the implementation of the Ballast Water Convention a road map was established with the ultimate to ratify the BWM Convention by the HELCOM Contracting States preferably by 2010, but in all cases not later than 2013.

In the Baltic Sea Action Plan (in the Roadmap towards harmonised implementation and ratification of the 2004 International Convention for Control and Management of Ships’ Ballast Water and Sediments), the CPs agreed to adjust/extend by 2010 the HELCOM monitoring programmes to obtain reliable data on non-indigenous species in the Baltic Sea, including port areas, in order to gather the necessary data to conduct and/or evaluate and consult risk assessments according to the relevant IMO Guidelines. As a first step, species that pose the major ecological harm and those that can be easily identified and monitored should be covered. The evaluation of any adverse ecological impacts caused by non-indigenous species should form an inherent and mandatory part of the HELCOM monitoring system.

The good environmental status (GES) according to the EU Marine Strategy Framework Directive is to be determined on the basis of eleven qualitative descriptors. One of the qualitative descriptors concerns non-indigenous species and describes the GES for this descriptor as “Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystem”.

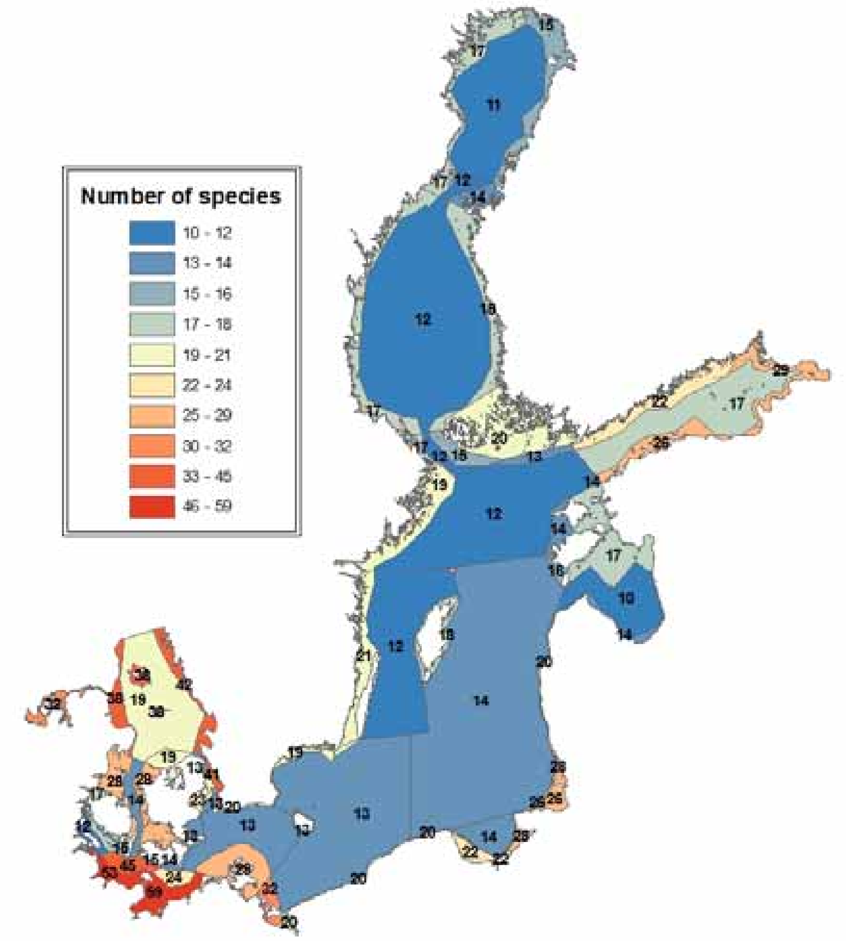

Figure 2.20. *Number of non-indigenous species in the assessment units in 2011. These values are used as baselines for the indicator; all new introductions during the assessment period are counted in this indicator. The species present in the assessment units have been reported by the experts of HELCOM HABITAT and HELCOM MONAS and HELCOM MARITIME in several occasions during 2008-2011. The data spreadsheet is a living document, open for revision. The data is shown in a table in the section “Dataset on trends in arrival of non-indigenous species”.*

Temporal trends in arrival of non-indigenous species

Figure 2.20 shows the number of species in each assessment unit in 2011. This is the baseline for the assessment period of 2012-2017. **Figure 2.21** presents the rate of introductions of NIS to the Baltic Sea starting from the beginning of the 19th century. Altogether 120 non-indigenous species have been found in the Baltic Sea marine environment (excluding terrestrial mammals and waterbirds). In the sections below, rates of introductions in the past and during the assessment period are presented for each country separately.

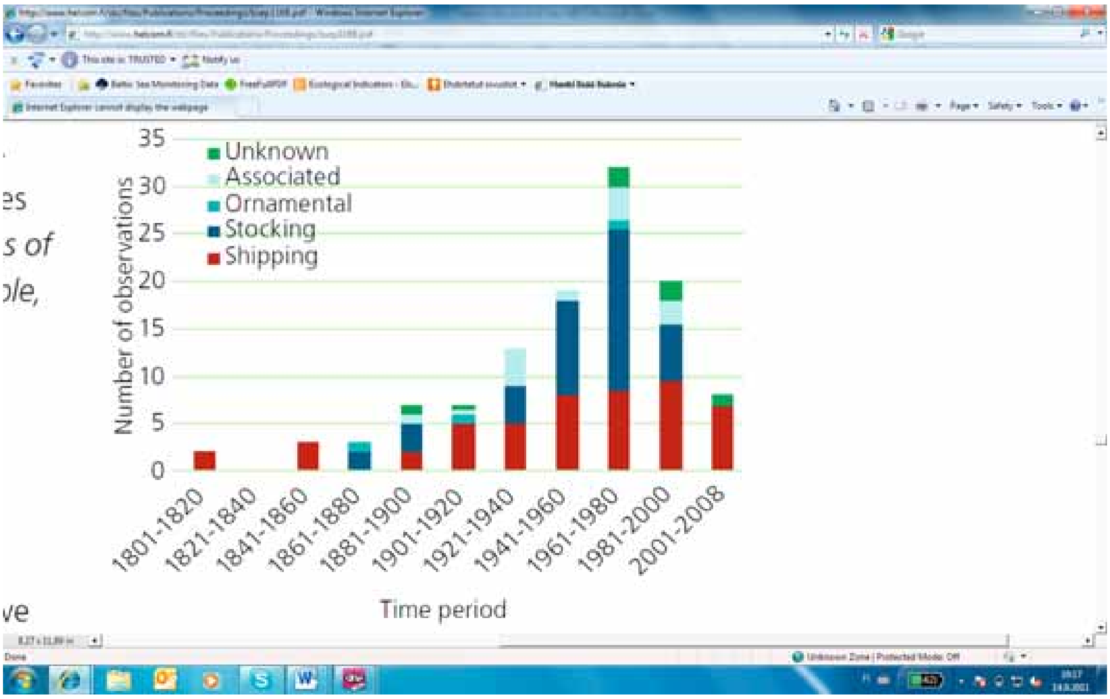

Figure 2.21. *Introductions of non-indigenous species to the Baltic Sea. Source: HELCOM 2009.*

Germany

Altogether 23 organisms are known to be introduced in the German Baltic Sea excluding inland waters (**Figure 2.22**). Assuming that the influence of man before the industrial revolution (< 1850) can be regarded as negligible, the natural rate of introductions for this area is one and represents a percentage of around 4% of the present total amount of introductions. In the following time until the 1960s of the last century the number of recognized introductions increased only slightly with an average of two (representing 9% of the present total amount). Beginning of the 1970s an appreciable rising of new introductions can be recognized with a maximum of 7 recognized organisms (representing 30% of all introductions).

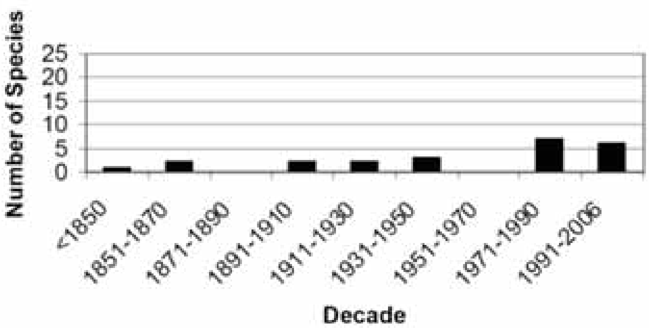

Figure 2.22. *Rates of detected non-indigenous species in the German Baltic Sea for 20-year intervals between 1850 and 2006 (Gollasch & Nehring 2006)*

Sweden

Figure 2.23 shows the rate of introductions of NIS to Sweden, excluding the west coast (Kattegat and Skagerrak). The introductions have increased greatly during the last four decades. Altogether 39 NIS have been found from the Swedish waters in the Baltic Sea.

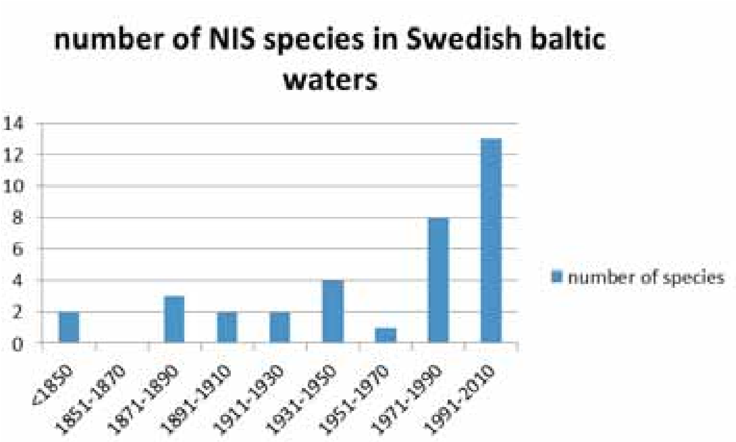

Figure 2.23. *Rates of detected non-indigenous species in the Swedish Baltic Sea for 20-year intervals between 1850 and 2010 (http://www.frammandearter.se/index.html). Altogether 39 NIS have been found from the Swedish waters on the Baltic Sea side, excluding Kattegat.*

Poland

Figure 2.24 shows the rate of introductions of NIS to the Polish marine waters. Altogether 36 NIS have been found from the Polish marine waters. No consistent increase or decrease in the introductions can be seen during the studied time period unless the recent increase during the last four decades can be seen as an indication of a new wave of introductions.

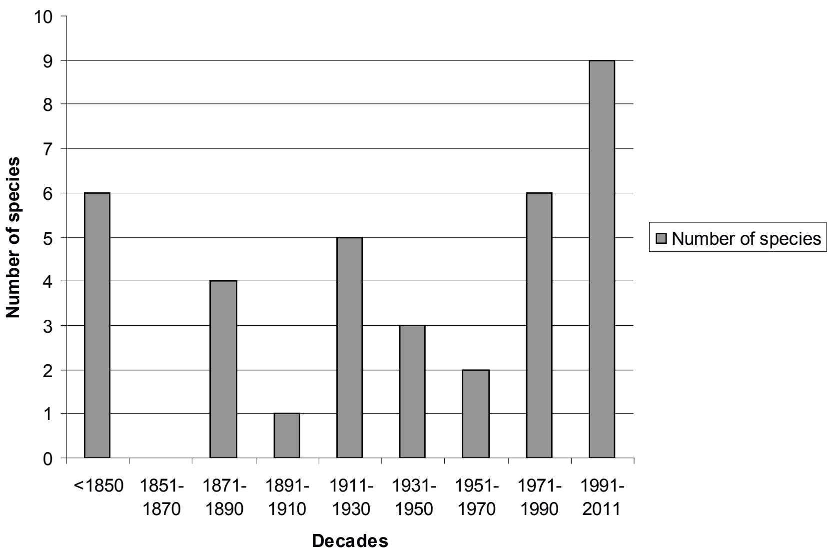

Figure 2.24. *Rates of detected non-indigenous species in the Polish waters for 20-year intervals between 1850 and 2011 (http://www.iop.krakow.pl/ias/Baza.aspx, Binpas). Altogether 36 NIS have been found.*

GES and classification method

The ultimate goal is to minimize man made introductions of non-indigenous organisms to zero. The boundary between GES and sub-GES is "no new introductions of NIS per assessment unit during a six year assessment period".

The indicator requires an estimation of the already existing NIS in each area and counts of new introductions. Hence, it is important to distinguish between naturally spreading and anthropogenically introduced species. In reality in some cases it is impossible to distinguish between man-made and natural introductions and therefore all species are first treated as NIS and only species which can be shown to be naturally spreading will be removed from the indicator. The BSAP Roadmap towards harmonised implementation and ratification of the 2004 International Convention for Control and Management of Ships' Ballast Water and Sediments (HELCOM 2007, p. 99) presents some advice on this matter (see Introduction above).

Systematic studies on NIS introductions are very scarce in the past, especially in the marine area. Therefore, for the purpose of this indicator, reviews and national databases are taken as a basis for an estimation of the baseline (Germany: Gollasch and Nehring 2006, Poland: http://www.iop.krakow.pl/ias/Baza.aspx, Binpas data, Sweden: http://www.frammandearter.se/index.html).

Existing monitoring

Data sources are the HELCOM monitoring programme fed by the national monitoring programmes. Proven information provided by other sources such as research institutes can also be used.

Sampling and data analyses

Number of species. All neobiota independent of their state of establishment have to be taken into account.

The indicator assesses the entire Baltic Sea: national coastal and offshore waters divided to sub-basins.

Currently, the monitoring of coastal and estuarine biodiversity is not structured enough to reliably show the distribution and abundance of several non-indigenous species. As a result of this robustness, assessments must be made on a sub-basin scale. The indicator is assessed by a six year assessment period. Yearly data based on at least two sampling exercises per year for an assessment period of six years.

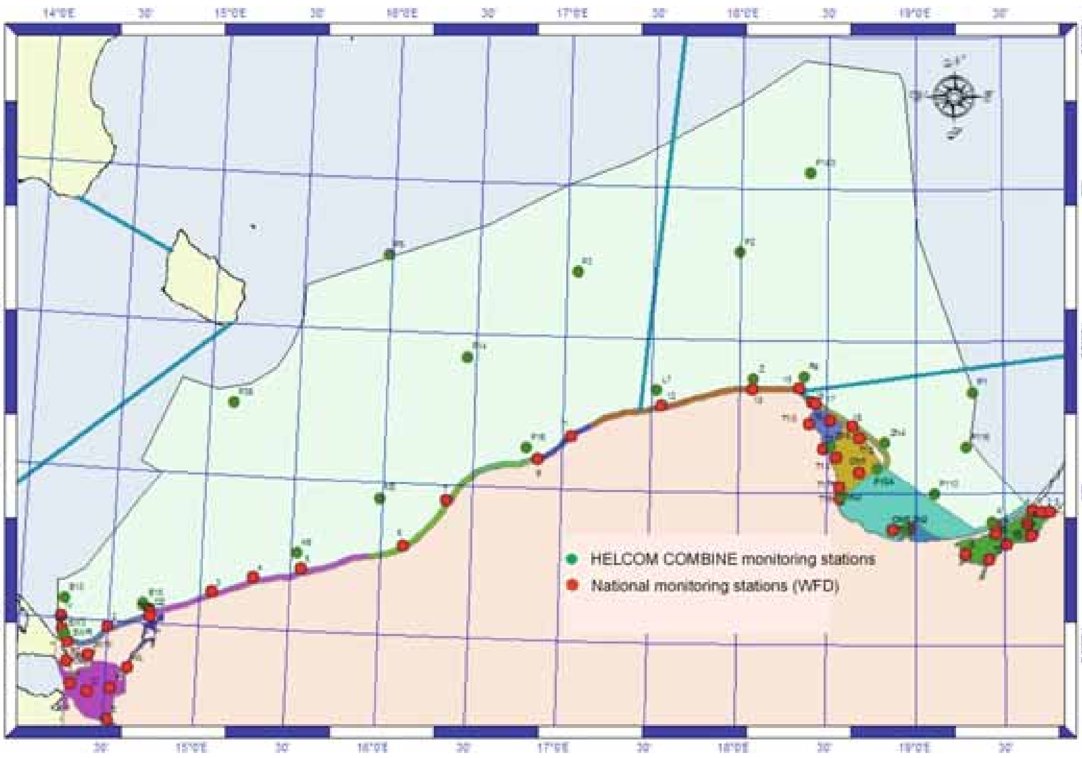

Figure 2.25. *Map of Polish monitoring stations in the Baltic Sea.*

Strengths and weaknesses of data

Strengths: The approach shows promise to give an indication of the success of measurements to minimize the man-made introduction of non-indigenous species. It has harmonized targets in the Baltic Sea. It is a simple measurement.

Weaknesses: differences in national data sets, quality problems of old data, geographical and temporal gaps in sampling.

Existing data are usually from times where non-indigenous species did not play an important role in the monitoring. Intensified and structured monitoring might identify more non-indigenous species already present in some areas.

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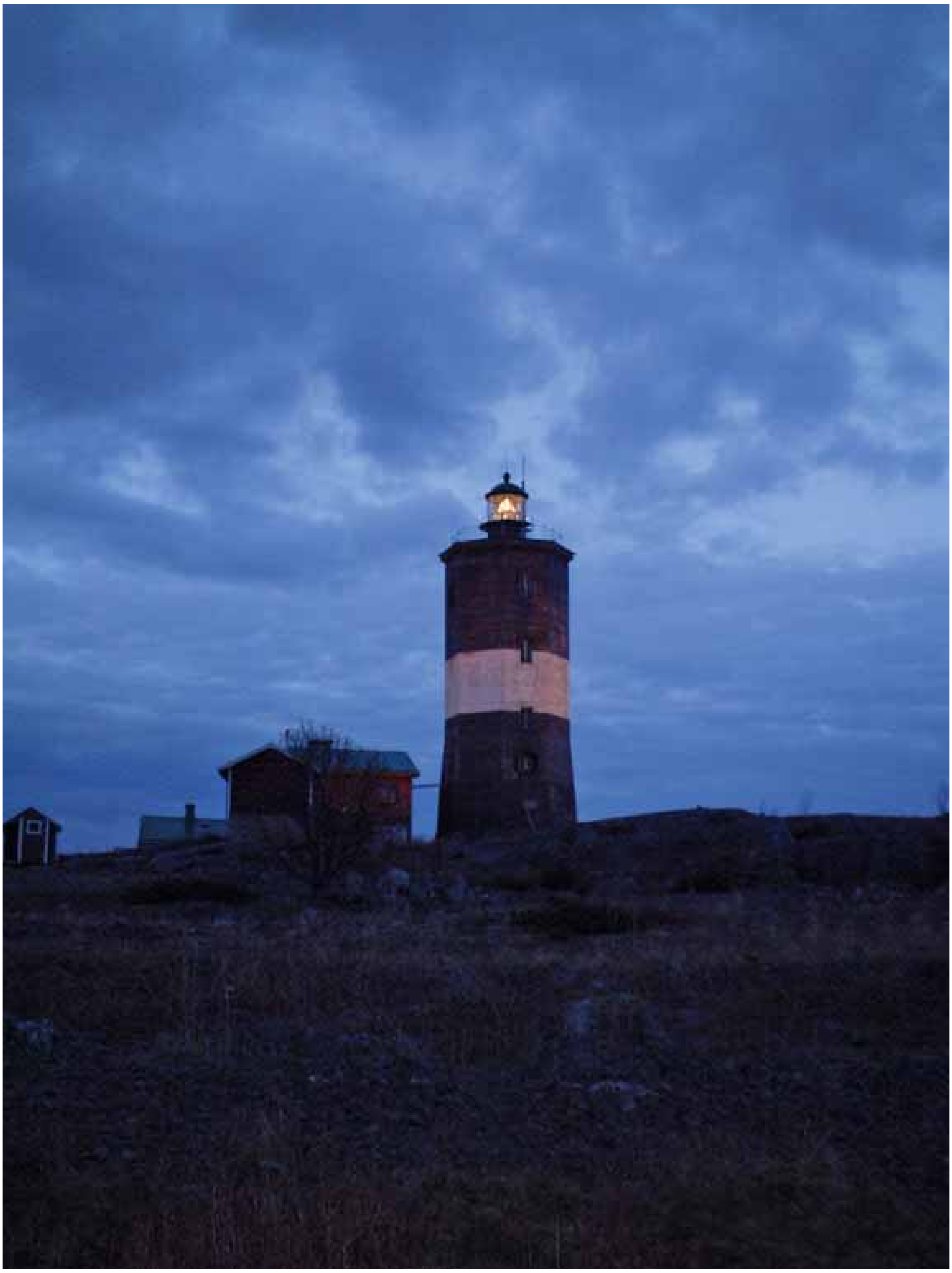
EC, 2008: Directive 2008/56/EC of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, 22pp

Gollasch, S. and S. Nehring, 2006: National checklist for aquatic alien species in Germany. Aquatic Invasions 1 (http://www.reabic.net), 245-269

ICES, 2005: ICES CODE of Practice on the Introductions and Tranfers of Marine Organisms 2005, 30pp

IMO, 2004: International Convention for the Control and Management of Ships' Ballastwater and Sediments, 2004. IMO document BMW/Conf/36, 38pp

3. Proposed core indicators for hazardous substances and their effects



This chapter presents the proposed core indicators for hazardous substances and their effects on biota. The CORESET expert group for hazardous substances has come up with 13 core indicators, which will be proposed to be finalized as core indicator reports during the project.

All the proposed core indicators have targets that show the boundary for good environmental status (GES). The GES boundaries were primarily selected among the EU environmental quality standards (EQS), but it is good to acknowledge that many of the EQS have not been finalized and accepted in the revision process of EQS and the EU Priority Substances [10]. The presented EQS for biota and sediment are, hence, provisional and

10 The EQS are developed in the WFD working group E. Draft version are available in the respective folder in CIRCA.

need to be revisited in light of progress of EQS development in the EU and experience gained in their environmental application.

All the proposed core indicators are listed in the summary table below and the following sections will provide descriptions of the proposed core indicators, including background documentation. Each core indicator has been given a separate section, following the numbering of the summary table.

Table 3.1. Proposed core indicators for hazardous substances and their effects.

Proposed core indicators	**Parameters**
Polybrominated biphenyl ethers	Congeners 28, 47, 99, 100, 153,154.
Hexabromocyclododacene	Hexabromocyclododacene (HBCD)
Perfluorooctane sulphonate	Perfluorooctane sulphonate (PFOS)
Polychlorinated biphenyls and dioxins and furans	CB congeners 28, 52, 101, 118, 138, 153 and 180. WHO-TEQ of dioxins and furans and dioxin-like PCBs.
Polyaromatic hydrocarbons and their metabolites	The US EPA 16 PAHs in bivalves and sediment and selected metabolites in fish.
Metals	Cadmium (Cd), mercury (Hg) and lead (Pb)
Radioactive substances	Caesium-137
Tributyltin compounds / imposex index	Tributyltin and/or imposex index
Pharmaceuticals	Diclofenac and 17-alpha-ethinylestradiol
General stress indicator	Lysosomal membrane stability
General stress indicator for fish	Fish disease index
Genotoxicity indicator	Micronucleus induction
Reproductive disorders	Malformed embryos of eelpout and amphipods

3.1. Poly Brominated Diphenyl ethers (PBDE)

Author: Jaakko Mannio

Acknowledged persons:
Anders Bignert, Elin Boalt, Anna Brzozowska, Galina Garnaga, Michael Haarich, Jenny Hedman, Ulrike Kamman, Thomas Lang, Martin M. Larsen, Kari Lehtonen, Rita Poikane, Rolf Schneider, Doris Schiedek, Jakob Strand, Joanna Szlinder-Richert, Tamara Zalewska

General information [11]

General properties (e.g. herbicide, lipophilic, bioaccumulating, persistence, volatile)

The polybrominated diphenyl ethers (polyBDE) are diphenyl ethers with degrees of bromination varying from 2 to 10. Among these polyBDEs which are mainly used as flame retardants, three are available commercially: "pentaBDE commercial product", "octaBDE commercial product" and "decaBDE commercial product". However, these products are all a mixtures of diphenyl ethers as shown in the table hereunder. Therefore, to differentiate "pentaBDE commercial product" and "octaBDE commercial product" from "pentaBDE substance" and "octaBDE substance", respectively, these commercial products will be referred to as "c-pentaBDE" and "c-octaBDE" in the present fact sheet.

All polyBDE's are hydrophobic or very hydrophobic substances, are very likely to adsorb on particulate matter and not likely to volatilize from water phase. PBDEs have the potential to photodegrade in the environment.

11 Following mostly the EU/CIRCA WFD /WGE draft dossier 22 Dec 2010

Overall, measured BCF values for BDE congeners range from very low values for highly brominated congeners (<5) to very high values for lower brominated congeners (up to 35 100 (measured value) for tetraBDE).

Main impacts on the environment and human health

Endocrine disrupter (Category 2) for humans, meaning "*potential for endocrine disruption. In vitro data is indicating potential for endocrine disruption in intact organisms. Also includes effects in-vivo that may, or may not, be ED-mediated. May include structural analyses and metabolic considerations*".

PBDEs have also been shown to have hormone-disrupting effects, in particular, on estrogen and thyroid hormones. Has been shown to disturb development of the nervous system.

Status of a compound on international priority lists and other policy relevance

The substances "pentaBDE" and "octaBDE" have been prioritised through 2 consecutive prioritisation procedures under the WFD: pentaBDE was prioritised following COMMPS (COmbined Monitoring-based and Modelling-based Priority Setting scheme) procedure in 2001, while octaBDE has recently been prioritised in the context of the second European Commission proposal for a new list of priority substances, for the reason that it is a PBT (Persistent, Bioaccumulative and Toxic) and a vPvB (very Persistent and very Bioaccumulative) substance. Following this latter prioritisation and the fact that pentaBDE EQS needed to be revised, it was decided in the WFD procedure to produce a unique fact sheet reporting a common EQS for all BDE congeners linked to c-pentaBDE and c-octaBDE, that is to say tetra- to nona-BDE congeners (draft 22.12.2010 in CIRCA).

PBDEs are on the HELCOM BSAP priority list and in the Stockholm Convention Annex A (Elimination).

Status of restrictions, bans or use

PolyBDE, including pentaBDE and octaBDE, are mainly used as flame retardants. The use of pentaBDE and octaBDE is however restricted within the EU since August 15, 2004 (Commission regulation (EC) No 552/2009). PentaBDE and OctaBDE are not allowed to be placed on the market as substances, in mixtures or in articles in higher concentration than 0.1% by weight.

Furthermore, use of polyBDEs in electrical and electronic products (E&Es) was restricted even earlier by the Directive 2002/95/EC (RoHS). From June 30, 2008, this directive covers also DecaBDE. This implies that the only permitted use of PBDEs in Europe is now the application of decaBDE in products other than E&Es. As a result of this new regulation, the majority of the previous use of decaBDE in the EU is now prohibited (corresponding to ca 80 percent of the total EU use in 2001). It is, however, still possible for industries to apply for exemptions for certain applications under the procedure laid out in article 5 of the RoHS Directive.

PentaBDE is now included in Annex A (Elimination) to the Stockholm Convention and should no longer be on the EU market as well as hexaBDE and heptaBDE contained in c-octaBDE.

GES boundaries and matrix

Existing quantitative targets

As for aquatic ecotoxicity, results are available mainly for the commercial mixtures of c-pentaBDE and c-octaBDE. The information on individual congener is insufficient to determine the contribution of each compound to the overall toxicity of the mixtures. The lowest NOEC= 0.4 mg kg^{-1} food is derived from a test in which pregnant rats administered a single dosage of c-pentaBDE exhibited reduced sperm production in male offspring at adulthood.

Considering the data set available, it is proposed to use this worst case value for the determination of the $QS_{biota,\ sec.\ pois}$. The choice of this value based on a mixture of low brominated compounds can be justified by the growing indications that polyBDE have the tendency to debrominate either in organisms or environmental compartment (pers. comm., POP Review Committee, 2010).

Table 3.2. Existing quantitative targets for PBDEs (WFD WG E draft proposal for Environmental Quality Standards (dossier 22 Dec 2010)).

Protection objective	Unit	Value
Pelagic community (freshwater)	[µg l^{-1}]	0.049
Pelagic community (marine water)	[µg l^{-1}]	0.0049
Benthic community (freshwater)	[µg kg^{-1} $_{dw}$]	22.9
	[µg l^{-1}]	7.7 10^{-6} – 0.167
Benthic community (marine)	**[µg kg^{-1} $_{dw}$]**	**4.5**
	[µg l^{-1}]	1.5 10^{-6} – 0.033
Predators (secondary poisoning)	[µg $kg^{-1}_{biota\ ww}$]	**4 µg $kg^{-1}_{biota\ ww}$**
	[µg l^{-1}]	1.3 10^{-5} (freshwater) 6.33 10^{-7} (marine waters)
Human health via consumption of fishery products	**[µg $kg^{-1}_{biota\ ww}$]**	**0.016** **(Critical QS)**
	[µg l^{-1}]	4.5 10^{-8} (freshwater) 2.3 10^{-9} (marine waters)
Human health via consumption of water	[µg l^{-1}]	9 10^{-4}

QS for protection of human health is the "critical QS" for derivation of an Environmental Quality Standard for the sum of polybrominated diphenyl ethers (**Table 3.2**).

The Scientific Committee on Health and Environmental Risks (SCHER) agreed that the sum of all PBDEs is the relevant approach and that the WS for human health is the most critical EQS (SCHER 2011).

Kinetic properties of PBDEs indicate that these compounds have dioxin-like, bioaccumulating properties in mammals (de Winter-Sorkina et al., 2006). The methodology for derivation of the threshold level was the same as has been applied to the dioxin 2,3,7,8-TCDD, as proposed by JECFA (Joint expert committee on food additives; JECFA (2002) as quoted in de Winter-Sorkina et al., 2006). The threshold value is derived on the basis of kinetic calculations combined with extrapolation factors. The TLhh of 0.26 ng kg^{-1}bw d-1 should be used for the derivation of the $QS_{biota,\ hh}$.

Preferred matrix

Biota (e.g. fish and mussels) is the primary matrix and sediment is the secondary matrix.

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Table 3.3. Existing monitoring of PBDEs in the Baltic Sea.

PBDE	Sediment			Biota				Water	
	profile	surface	frequency (y)	species	organ	stations	Frequency (y)	Stations	Frequency (y)
Denmark				eelpout	liver	12	1		
				flounder	liver	5	1		
Germany				herring	liver	1	1		
				cod	liver	1	1		
				dab	liver	1	1		
Poland					(muscle)	(3)	project		
Russia									
Sweden	SGU?	16?	?	herring	muscle	6+2	1		
				cod	liver	2	1		
				guillemot	eggs	1	1		
Finland	4		>5	herring	muscle	4	1		
		(2 screen)	>5	perch	muscle	2	1		
Estonia				(herring + perch)	(muscle)	(5)	project	(5 project)	
Lithuania		(9 project)	3	(herring + flounder)	(muscle)	(2)	project	1	1
Latvia				(herring + perch)	(muscle)	(2)	project	(2 project)	

Gaps in the monitoring of the compound

Sweden, Denmark, Finland and Germany have permanent monitoring presently on fish. Germany prepares for monitoring in sediment. Estonia, Latvia, Lithuania and Poland have screening or research data. No information from Russia.

Present status assessments

Known temporal trends (also from sediment core profiles)

A Swedish study found significant increasing trend from the end of 1960s until the end of 1990s followed by a decreasing trend during the last 10 years for BDE47, BDE99 and BDE100 level in the eggs of the guillemots nesting on Stora Karlsö Island west of Gotland. The spatial analysis of BDE levels in herring muscle during 1999–2004 do not show any firm geographical differences, except that the levels of BDE47, 99, 100, 153 and 154 in the Southern Baltic Proper seem to be higher than in the other six sites from Skagerrak to Bothnian Bay. In general, BDEs seem to be more evenly distributed in the Swedish marine environment compared to, e.g. PCBs (Bignert et al. 2006).

Spatial gradients (incl. sources)

The HELCOM thematic assessment of hazardous substances in the Baltic Sea showed that concentrations of the congener BDE-47 (a tetraBDE) in fish exceeded the selected threshold (5 µg kg^{-1} lw) in several parts of the sea area (HELCOM 2010).

PentaBDE concentrations in biota (e.g. in herring and seal blubber) are higher in the Baltic Sea compared to the west coast of Sweden in the 1980s. Furthermore, concentrations increased with the age of the fish and were higher in seals than in fish in the Baltic Sea, indicating bioaccumulation and biomagnifications (EU-RAR 2000).

In Denmark, the highest contamination with BDEs was found in sediment and mussels close to populated urban areas. The congener BDE47 is both bioconcentrated and biomagnified to a higher degree than any other congeners, whereas the amount of BDE99 decrease at higher trophic levels.

The Swedish sediment monitoring programme, covering 16 stations in the coastal and offshore areas of the Baltic Sea, showed that concentrations of BDE47, BDE99 and BDE100 were clearly the highest in the Kattegat, 0,4-0,6 µg/kg dw (BDE209 not reported).

The occurrence of BDEs is widespread in the Baltic marine environment. It is probable that current legislative measures (penta- and octaBDE banned in the EU since 2004) have already decreased penta- and octaBDE levels in the Baltic Sea. While PentaBDE and octaBDE do not seem to pose a risk to the marine environment in the Western Baltic Sea, the situation may be different in the eastern part of Baltic Sea. Information on the occurrence of penta-, octa- and decaBDE in the eastern Baltic Sea (e.g. in biota) and in discharges (e.g. WWTPs) and emissions, especially from eastern HELCOM contracting Parties, is thus greatly needed. More information on the occurrence of penta-, octa- and decaBDE discharges from e.g., landfills and waste sorting sites is needed from the whole Baltic Sea area.

Recommendation

DecaBDE is the dominant congener from sources (e.g. WWTPs) and in the Baltic Sea sediments; it can also be found in Baltic Sea fish, although tetraBDE is the most dominant congener in biota. Levels of decaBDE may be increasing because its use has not been restricted. However, because of the environmental problems of decaBDE and anticipating regulatory measures, the European industry has taken voluntary action to reduce releases of decaBDE. This would be expected to lead – over time – to decreasing concentrations. PBDEs with smaller molecules are more toxic and bioaccumulative. The biotic and abiotic debromination of highly brominated BDEs, such as decaBDE, to these smaller forms is a possibility and justifies that monitoring is based on a broad set of congeners.

PBDE are recommended to be included as a core indicator.

References

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3.2. Hexabromocyclododecane (HBCD)

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General information [12]

General properties (e.g. herbicide, lipophilic, bioaccumulating, persistence, volatile)

The commercially available brominated flame retardant hexabromocyclododecane (HBCD or HBCDD) is lipophilic, has a high affinity to particulate matter and low water solubility. Depending on the manufacturer and the production method used, technical HBCD consists of 70-95% γ-HBCD and 3-30% of α- and β-HBCD.

HBCD has a strong potential to bioaccumulate and biomagnify. Available studies demonstrate that HBCD is well absorbed from the rodent gastro-intestinal tract. Of the three diastereoisomers constituting HBCD, the α-form is much more bioaccumulative than the other forms. HBCD is persistent in air and is subject to long-range transport. HBCD is found to be widespread also in remote regions such as in the Arctic, where concentrations in the atmosphere are elevated.

The low volatility of HBCD has been predicted to result in significant sorption to atmospheric particulates, with the potential for subsequent removal by wet and dry deposition. The transport potential of HBCD was considered to be dependent on the long-range transport behaviour of the atmospheric particles to which it sorbs.

Main impacts on the environment and human health

HBCD is very toxic to aquatic organisms. In mammals, studies have shown reproductive, developmental and behavioral effects with some of the effects being trans-generational and detectable even in unexposed offspring. Besides these effects, data from laboratory studies with Japanese quail and American kestrels indicate that HBCD at environmentally relevant doses could cause eggshell thinning, reduced egg production, reduced egg quality and reduced fitness of hatchlings. Recent advances in the knowledge of HBCD induced toxicity includes a better understanding of the potential of HBCD to interfere with the hypothalamic-pituitary-thyroid (HPT) axis, its potential ability to disrupt normal development, to affect the central nervous system, and to induce reproductive and developmental effects.

HBCD has been found in human blood, plasma and adipose tissue. The main sources of exposure presently known are contaminated food and dust. For breast feeding children, mothers' milk is the main exposure route but HBCD exposure also occurs at early developmental stages as it is transferred across the placenta to the foetus. Human breast milk data from the 1970s to 2000 show that HBCD levels have increased since HBCD was commercially introduced as a brominated flame retardant in the 1980s. Though information on the human toxicity of HBCD is to a great extent lacking, and tissue concentrations found in humans are seemingly low, embryos and infants are vulnerable groups that could be at risk, particularly to the observed neuroendocrine and developmental toxicity of HBCD.

The HELCOM thematic assessment of hazardous substances in the Baltic Sea showed that HBCD exceeds threshold values in several parts of the Baltic Sea and increasing trends have been found in the eggs of common guillemot (HELCOM 2010).

12 Following mostly the EU/CIRCA WFD /WGE draft dossier 22 Dec 2010

Status of a compound on international priority lists and other policy relevance

HBCD has attracted attention as a contaminant of concern in several regions, by international environmental forums and academia. In the EU, HBCD has been identified as a Substance of Very High Concern (SVHC), meeting the criteria of a PBT (persistent, bioaccumulative and toxic) substance pursuant to Article 57(d) in the REACH regulation. In December 2009, HBCD was considered by the Executive Body (EB) of the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP) to meet the criteria for POPs, set out in EB decision 1998/2. It is a substance (group) of specific concern to the Baltic Sea and candidate for revised WFD Priority Substance list.

HBCDD is on the HELCOM BSAP priority list.

Status of restrictions, bans or use

In the EU, HBCD has been identified as a Substance of Very High Concern (SVHC), meeting the criteria of a PBT (persistent, bioaccumulative and toxic) substance pursuant to Article 57(d) in the REACH regulation. In May 2009, HBCD was included in the European Chemicals Agency (ECHA) recommendation list of priority substances to be subject to Authorisation under REACH, based on its hazardous properties, the volumes used and the likelihood of exposure to humans or the environment. A proposal on classification and labeling of HBCD as a possible reprotoxic substance is currently under discussion within the EU (Proposal for Harmonised Classification and Labelling, Based on the CLP Regulation (EC) No 1272/2008, Annex VI, Part 2 Substance Name: Hexabromocyclododecane Version 2, Sep. 2009) .

GES boundaries and matrix

Existing quantitative targets

Table 3.4. Existing quantitative targets for HBCD (WFD WG E draft proposal for Environmental Quality Standards (dossier 19 Jan 2011)).

Protection objective	**Unit**	**Value**
Pelagic community (freshwater)	[µg.l^{-1}]	0.31
Pelagic community (marine waters)	[µg.l^{-1}]	0.031
Benthic community (freshwater)	[µg.kg^{-1} $_{dw}$]	860
	[µg.l^{-1}]	
Benthic community (marine)	**[µg.kg^{-1} $_{dw}$]**	**170**
	[µg.l^{-1}]	
Predators (secondary poisoning)	**[µg.kg^{-1}$_{biota\ ww}$]**	**167 (Critical QS)**
	[µg.l^{-1}]	0.0016 (freshwaters) 0.00080 (marine waters)
Human health via consumption of fishery products	**[µg.kg^{-1}$_{biota\ ww}$]**	**6100**
	[µg.l^{-1}]	0.058 (fresh and marine waters)
Human health via consumption of water	[µg.l^{-1}]	

QS for secondary poisoning is the "critical QS" for derivation of an Environmental Quality Standard (**Table 3.4**).

The Scientific Committee on Health and Environmental Risks (SCHER) agreed that the QS for secondary poisoning of top predators is the most critical EQS (SCHER 2011).

Preferred matrix

Sediment and biota are preferred. The measured data on HBCD concentration in Baltic Sea water is very scarce and the detection limit has been too high to draw any conclusions (HELCOM 2010).

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Table 3.5. Monitoring of HBCD in the Baltic Sea.

HBCD	Sediment			Biota				Water	
	profile	surface	frequency (y)	species	organ	stations	frequency (y)	stations	frequency (y)
Denmark		screening		screening					
Germany									
Poland				3 fish in project	(muscle)	(2)		(2 proj)	
Russia									
Sweden	SGU?			herring	muscle	6+2	1		
				cod	liver	2	1		
				guillemot	egg	1	1		
Finland			>5	(pike)	(muscle)	(3)	project		
		(2 screen)	>5						
Estonia				(herring + perch)	(muscle)	(5)	project	(5 proj)	
Lithuania				(herring + flounder)	(muscle)	(2)	project	1	1
Latvia				(herring + perch)	(muscle)	(2)	project	(2 proj)	

Gaps in the monitoring of the compound

Only Sweden has permanent monitoring presently. Denamrk includes HBCD in 2011 in monitoring. Germany does not monitor HBCD in biota, but water monitoring in under development and sediment monitoring in a planning phase. Finland, Estonia, Latvia, Lithuania, Poland and Denmark have screening data. No information from Russia.

Present status assessments

Known temporal trends (also from sediment core profiles)

HBCD is found to be widespread in the global environment, with elevated levels in top predators in the Arctic. In biota, HBCD has been found to bioconcentrate, bioaccumulate and to biomagnify at higher trophic levels.

Swedish trend studies show an increase of HBCD in the guillemot eggs until recent years. Its increased presence in the environment is likely attributable to the increased global demand. The general trend is to higher environmental HBCD levels near point sources and urban areas (waste disposal sites including those whose processes include either recycling, landfilling or incineration).

The Swedish results show that HBCD levels in Baltic Sea fish are generally low and always lower than the estimated PNEC level. (Also the levels in the sediments of the Swedish coastal area are very low compared to the estimated PNEC level (170 µg/kg dw)).

A temporal analysis (EU-RAR 2006) showed that HBCD levels in seals in the Baltic Sea have increased. The median levels in the 1980s ranged between 16 and 35 µg/kg lw with a median concentration of 28 µg/kg lw (n=7). In the 1990s, the levels ranged between 34 and 177 g/kg lw with a median of 73 µg/kg lw (n=12). From 2000, data from only one seal are available and has a HBCD concentration of 64 µg/kg lw. However, another study found that the HBCD level in the blubber of 30 grey seals during 2000–2002 ranged from 31–554 µg/kg lw with a mean of 101 µg/kg lw. The results indicate that the HBCD levels in seals have not decreased.

Spatial gradients (incl. sources)

The spatial analysis of HBCD levels in herring muscle during 1999–2004 does not show any firm geographical differences, except that the level in the Southern Baltic Proper seems to be higher than another six sites from Skagerrak to Bothnian Bay. In general, HBCD seems to be more evenly distributed in the Swedish marine environment compared to, e.g. PCBs (Bignert et al. 2006).

Recommendation

HBCD can commonly be found in fish from the Swedish coastal area of the Baltic Sea. However, the situation may be different in other parts of the Baltic Sea. Thus, information on the occurrence of HBCD in the Baltic Sea (e.g. in biota) and in discharges (e.g. WWTPs, landfills and waste sorting sites) and emissions in the HELCOM countries is greatly needed. The need to include in CORESET is to some extent contradictory:

– PBT and HPV substance with limited restrictions in use, high priority or candidate in many lists
– but found levels (scarce data) relatively low compared to presented effect levels. HBCD is recommended to be included as a core indicator.

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3.3. Perfluorooctane sulphonate (PFOS)

Based on the HELCOM Thematic assessment of hazardous substances, chapter on PFOS by Urs Berger and Anders Bignert

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General information

General properties

Perfluorooctane sulfonic acid (PFOS), perfluoro octanoic acid (PFOA) and other perfluorinated compounds are considered as global environmental contaminants. PFOS and PFOA are chemically and biologically inert and very stable (Poulsen et al. 2005). PFOS meets the P (Persistent) and vP (very Persistent) criteria due to slow degradation. PFOS is also bioaccumulative (B) and toxic (T) (RPA & BRE 2004, OSPAR 2005). PFOA is considered as very persistent (vP) and toxic (T), but not bioaccumulative (Van der Putte et al. 2010). It has a capacity to undergo long-range transport.

PFOS related substances and PFOA are members of the larger family of perfluoroalkylated substances (PFAS). Perfluorooctanyl sulfonate compounds are all derivatives of PFOS and can degrade to PFOS, also called as PFOS-related compounds. The abbreviation PFOA is used as a group name for perfluorooctanoic acid and its salts. Some 100–200 PFOS-related compounds have been identified (KEMI 2006). PFOS binds to blood proteins and bioaccumulates in the liver and gall bladder unlike most POP compounds, which accumulate into fat (Renner 2001). Some indicative compounds related to use are presented e.g. in HELCOM report on selected hazardous substances (HELCOM 2009).

Main impacts on the environment and human health

PFOS has been shown to disturb immune system, development and reproduction (encrine disruption) of organisms and influence the lipid metabolism, to reduce weight gain and food consumption. It is also suspected to induce liver necrosis.

Status of a compound on international priority lists and other policy relevance

PFOS is included in the Stockholm Convention list of POPs, Annex B, which requires the parties to the convention to restrict the production and use of the substance.

The HELCOM Baltic Sea Action Plan has the objective of “Hazardous substances close to natural levels” and PFOS and PFOA are chosen as priority substances under the BSAP.

PFOS is included on the revision list of the EU Priority Substances. The EU Marine Strategy Framework Directive requires that “contaminants are at levels that do not give rise to pollution effects” (GES Descriptor 8).

Status of restrictions, bans or use

The production and use of perfluorooctane sulfonate (PFOS), one of the major PFA representatives, have been regulated in some countries (e.g., Canada and the EU), but large-scale PFOS production continues in other parts of the world.

Pathways of PFOS to the Baltic ecosystem

Some PFAS have been manufactured for more than five decades. They are applied in industrial processes (e.g., production of fluoropolymers) and in commercial products such as water- and stain-proofing agents and fire-fighting foams, electric and electronic parts, photo imaging, hydraulic fluids and textiles.

PFOS is both intentionally produced and an unintended degradation product of related anthropogenic chemicals. PFOS is still produced in several countries.

GES boundaries and matrix

Existing quantitative targets

The threshold values originate from the report of the EU expert group on the review of priority substances. The threshold values are final drafts and should be changed if the final values will change.

The threshold value for biota is the Quality Standard for human health via fishery products (9.1 µg kg^{-1} ww), because it is stricter than the QS for secondary poisoning of predators (33 µg kg^{-1} ww). The Scientific Committee for Health and Environmental Risks supported this choice of the WFD WG E (SCHER 20101. The QS for marine waters is 0.23 µg l^{-1}. Currently, there is no QS for benthic organisms (sediment).

Preferred matrix

Fish and sediment are appropriate matrixes to be used in the monitoring of PFOS in the Baltic Sea. Instead, water is more appropriate matrix to be used in the monitoring of PFOA. Sediment data is currently not appropriate for the core indicator, because of the lack of a proper threshold value.

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Only Sweden has PFOS in the national monitoring programme. Danish monitoring programme will include PFOS in 2011.

The data in this assessment has been sampled between 2003 and 2009.

Swedish National Monitoring Programme for biota (years 2005-2007).

HELCOM SCREEN project, data from 2008 (Lilja et al. 2009).

Project 202 22 213 of the Federal Environmental Agency, Germany. Data from 2003-2005 (Theobald et al. 2007).

Strand, et al. (2007). DMU (NERI) Rapport 608. Data from 2003.

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Gaps in the monitoring of the compound

There is a need to include PFOS in several CPs' monitoring programme.

Present status assessments

Concentrations and temporal trends of PFOS in the Baltic Sea

Exponentially increasing concentrations of some PFAs in wildlife have been reported during the 1990s (Holmström et al. 2005). According to the HELCOM thematic assessment of hazardous substances in the Baltic Sea (HELCOM 2010), PFOS concentrations are generally below threshold levels, but frequently exceed them in many parts of the Baltic Sea. The PFOS and PFOA levels in fish and water seem to be similar in different parts of the Baltic Sea. There is not enough data to assess temporal changes in PFOS concentrations in fish. However, the data covers several fish species and blue mussel and can therefore be considered quite exhaustive, whereas the lack of time series inhibits a making of a comprehensive assessment.

Assessment of temporal trends of PFOS

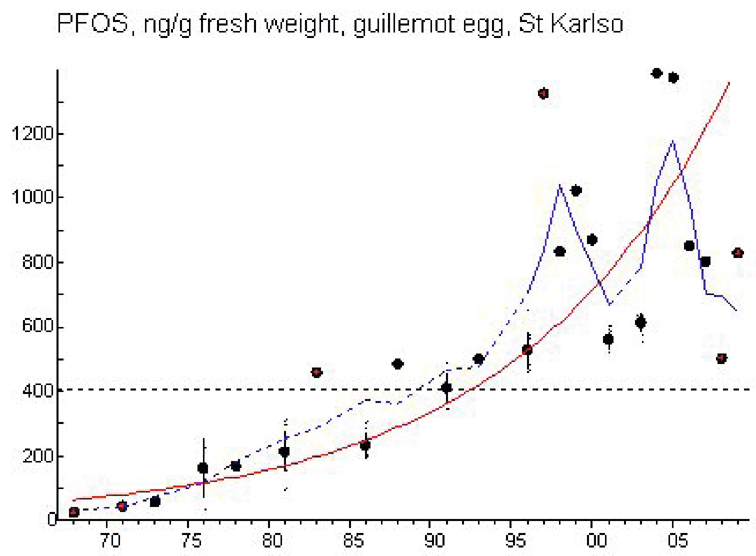

Figure 3.1. *Concentration of PFOS in eggs of common guillemot (Uria aalge) from 1968 to 2007. The mean annual PFOS value shown as red dot in the figure of the time series is based on pooled samples or mean values of individual samples. Source: HELCOM 2010.*

According to Swedish data on common guillemot eggs, a significant increasing trend is observed for PFOS in guillemot eggs with 7-10% per year (**Figure 3.1**), which is equal to an increase to 25-30 times higher levels in the early 2000s as compared to the late 1960ties (Bignert et al. 2009).

Concentrations in fish exceed the threshold level

PFOS levels in fish liver (e.g. herring, perch, pike, eelpout, flounder, eel and cod) exceeded the threshold level for the protection of predators via secondary poisoning (9.1 µg/kg wet weight in prey tissue; see section on targets below) in several areas of the Baltic Sea (HELCOM 2010). Thus, PFOS may cause adverse effects for top predators.

One of the highest concentrations was found from the liver of pike (*Esox lucius*) from the Gulf of Finland (close to Helsinki and Espoo) containing 200 to 550 µg PFOS kg^{-1} ww, as well as up to 140 µg kg^{-1} ww of PFOSA, a non-persistent precursor compound of PFOS (Nordic Council of Ministers 2004). Additional hot spots seem to be the mouth of the river Oder in the Bornholm Basin (coast of Poland; Lilja et al. 2009), the Hanö Bight (Bornholm Basin, coast of Sweden; SEPA 2006) and the Kattegat (Nordic Council of Ministers 2004). In all these regions, fish liver values of around 60 µg PFOS kg^{-1} ww have been observed (perch, cod and eelpout, respectively).

In regions less affected by anthropogenic pollution, typical PFOS levels in fish liver were in the range 1–20 μg kg^{-1} ww. However, for wildlife or general human consumption, whole body or muscle concentrations would be more relevant as food matrices, which so far have not been found to exceed the PNEC value for PFOS. Compared to liver, PFOS concentrations in muscle were lower, typically in the range <1 to 5 μg kg^{-1} ww (Theobald et al. 2007, Berger et al. 2009b). In blue mussels from the Kattegat, Great Belt and the Sound, PFOS was below the detection limit of 0.2 μg kg^{-1} ww (NERI 2007).

The distribution of PFOS in herring liver was found to be quite homogeneous throughout the Baltic Sea (around 10 μg kg^{-1} ww), which probably is a result of the extraordinary persistence of the compound and its use for more than three decades. A somewhat higher level of 26 μg kg^{-1} ww was found along the Swedish coast of the Northern Baltic Proper, reflecting the proximity of the city of Stockholm.

High concentrations in mammals and birds

Marine mammals are considerably higher contaminated with PFOS than marine and freshwater fish, and were found to be the most contaminated of all Nordic biota studied (HELCOM 2010). Several hundreds to one thousand μg kg^{-1} ww of PFOS have been found in the livers of grey seals (Southern Baltic Proper and Bothnian Sea; Nordic Council of Ministers 2004), harbour seals (Great Belt and the Sound; Nordic Council of Ministers, 2004) as well as ringed seals (Bothnian Bay; Kannan et al. 2002). In the eggs of common guillemots (Western Gotland Basin), PFOS concentrations were greater than 1000 μg kg^{-1} ww (Holmström et al. 2005). OSPAR risk assessment (OSPAR 2005) on marine environment concluded that the major area of concern for PFOS is the secondary poisoning of top predators, such as seals and predatory birds.

Concentrations in surface waters

Only a few measurements of PFAs in the Baltic Sea surface water exist (Nordic Council of Ministers 2004, Theobald et al. 2007, Lilja et al. 2009). They were mostly performed in potentially affected coastal areas. Perfluoro octanoid acid (PFOA) and PFOS dominated the water samples. Concentrations of PFOA were determined in the range 0.57-0.68 ng l^{-1} (Little Belt, Kiel Bight, Mecklenburg Bight, Arkona Basin) up to 4–7 ng l^{-1} (Little Belt, the Sound, coast of Poland, Gulf of Finland). PFOS was found at levels 0.34-0.90 ng l^{-1} for all locations mentioned, with the exception of single measurements of 2.9 ng l^{-1} (coast of Poland) and 22 ng l^{-1} close to Helsinki (Gulf of Finland). Further away from the coast in the Arkona Basin, PFOA and PFOS levels were 0.35-0.40 ng l^{-1}.

Concentrations in surface sediments

Limited data exist for PFA concentrations in Baltic Sea sediments (Nordic Council of Ministers 2004, SEPA 2006, NERI 2007, Theobald et al. 2007). PFOS and/or PFOA were occasionally detected, but consistently at levels below 1 μg kg^{-1} dw or ww. The highest levels reported so far have been from the Gulf of Finland close to Helsinki (PFOS 0.9 μg kg^{-1} ww), close to Stockholm (PFOS 0.6 μg kg^{-1} ww) and along the coast of Poland (PFOS and PFOA both around 0.6 μg kg^{-1} dw). In the German Baltic coast, concentrations of PFOS in sediments were on the order of 0.02-0.67 μg/kg dw, those of PFOA 0.09-0.68 μg/kg dw (Theobald et al. 2007).

Recommendation

PFOS should be included as a core indicator in the Baltic Sea. The substance has high policy relevance, it shows adverse effects in the environment.

The core indicator for PFOS requires better geographical coverage in national monitoring programmes and time series data to assess temporal trends.

Common HELCOM sampling and analysis procedures should be agreed on.

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3.4. Polychlorinated biphenyls and dioxins and furans

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General information

General properties

Polychlorinated biphenyls (PCBs) and PCDD/Fs (dioxins) are persistent organic pollutants (POPs) that can cause severe, long-term impacts on wildlife, ecosystems and human health. The substance groups are characterized by low water solubility and low vapor pressure. Due to their persistent and hydrophobic properties, the substances accumulate in sediments and organisms in the aquatic environment. In the environment, dioxins can undergo photolysis, however, they are generally very resistant to chemical and biological degradation.

Polychlorinated biphenyls (PCBs) consist of two linked benzene rings with chlorine atoms substituted for one or more hydrogen atoms. Theoretically, 209 congeners are possible, but only around 130 are found in commercial mixtures. Some PCBs are called dioxin-like (dl-PCBs) because they have a structure very similar to that of dioxins and have dioxin-like effects (i.e. four non-ortho substituted PCBs: CB-77, CB-81, CB-126, CB-169, IUPAC and eight mono-ortho substituted: CB-105, CB-118, CB-156, CB-157, CB-167, CB-114, CB-123, CB-189, IUPAC) (Burreau et al. 2006). The name "dioxin" refers to polychlorinated dibenzo-*p*-dioxin (PCDD) and dibenzofuran (PCDF) compounds, i.e two benzene rings with one (furans) or two (dioxins) oxygen bridges and substituted with 1-8 chlorine atoms. Of the 210 possible congeners, the 17 compounds (10 furans, 7 dioxins) substituted in positions 2,3,7,8 are considered to be of toxicological importance.

PCBs are synthetic chemicals and do not occur naturally in the environment. Due to their properties, PCBs have been used in a wide variety of manufacturing processes, especially as plasticizers, insulators and flame-retardants. They are widely distributed in the environment through, for example, inappropriate handling of waste material or leakage from transformers, condensers and hydraulic systems. According to some estimates, the total global production of PCBs from 1930 to the ban in most countries by the 1980s has been in the order of 1.5 million tons. PCDD/Fs were never produced intentionally, but they are minor impurities in several chlorinated chemicals (e.g., PCBs, chlorophenols, hexachlorophene, etc.), and are formed in several industrial processes and from most combustion processes, such as municipal waste incineration and small-scale burning under poorly controlled conditions. Formerly, pulp bleaching using chlorine gas was an important source of PCDD/Fs.

The PCBs included in this Indicator Information Sheet are the 7 PCB congeners that have been monitored since the beginning of the HELCOM and OSPARCOM monitoring programmes, carefully selected mainly by ICES working groups due to their relatively uncomplicated identification and quantification in gas chromatograms and as they usually contribute a very high proportion of the total PCB content in environmental samples. These are the "ICES 7": CB-28, CB-52, CB-101, CB-118, CB-138, CB-153 and CB-180.

Main impacts on the environment and human health

PCBs and PCDD/Fs can cause a variety of biological and toxicological effects in animals and humans. Most toxic effects are explained by the binding of PCBs and PCDD/Fs to the aryl hydrocarbon (Ah) receptor, however the specific mechanism is not fully understood. Long-term effects of PCBs from human and laboratory mammal studies include increased risk of cancer, infections, reduced cognitive function accompanied by adverse behavioral effects, as well as giving birth to infants of lower than normal birth weight (Carpenter 1998, Carpenter 2006). There are also indications that PCBs are involved in causing reproductive disorders in marine top predators, e.g. seals and bald eagles. PCBs are also assumed, together with p,p'-DDE, to cause eggshell thinning and reduced number of offspring in white-tailed sea eagle and uterine leioymas in grey seal (Helander et al 2002, Bäcklin et al. 2010).

The most relevant toxic effects of PCDD/Fs are developmental toxicity, carcinogentiy and immunotoxicity. The sensitivity of various species to the toxic effects of PCDD/Fs vary significantly. 2,3,7,8-TCDD is the most toxic and well-studied congener and is used as a reference for all other related chemicals. Each of the 17 relevant congeners is assigned a toxic equivalency factor (TEF), where 2,3,7,8-TCDD equals 1 (Van den Berg et al., 1998; Van den Berg et al., 2006). Dioxin concentrations are commonly reported as toxic or TCDD equivalents (TEQ), which is the sum of the individual congener concentrations multiplied with its specific TEF.

The HELCOM thematic assessment of hazardous substances showed that the concentrations of dioxins exceed the thresholds (4 ng kg^{-1} ww TEQ dioxins, 8 ng kg^{-1} ww TEQ dioxins + dl-PCBs) in several areas of the Baltic Sea (HELCOM 2010). In the southern parts, it was mainly the dl-PCBs which were on alarming levels. Non-dioxin-like PCBs also exceeded the threshold (0.08 mg kg^{-1} lipid weight for CB-153) both in the southern and northern sub-basins of the Baltic Sea.

Status of a compound on international priority lists and other policy relevance

The ICES 7 PCBs are listed as mandatory contaminants that should be analysed and reported within both the OSPARCOM and the HELCOM conventions and are classed as priority POPs under the Stockholm Convention. In the proposed revised guidelines for OSPARCOM (1996) the congeners CB-105 and CB-156 are added to this list. PCBs are not included on the Water Framework Directive (WFD) priority substance lists, but they are in the Marine Strategy Framework Directive (MSFD).

Dioxins are included in several international agreements, of which the Stockholm Convention and the Convention on Long Range Transboundary Air are among the most important for the control and reduction of sources to the environment. WHO and FAO have jointly established a maximum tolerable human intake level of dioxins via food, and within the EU there are limit values for dioxins in food and feed stuff (EC 2006). Several other EU legislations regulate dioxins, e.g. the plan for integrated pollution prevention and control (IPPC) and directives on waste incineration (EC, 2000, 2008). The EU has also adopted a Community Strategy for dioxins, furans and PCBs (EC 2001).

PCDD/Fs are currently not included in the Water Framework Directive but are on the list of substances to be revised for adoption in the near future. HELCOM has listed PCDD/Fs and dl-PCBs as prioritized hazardous substances of specific concern for the Baltic Sea (HELCOM 2010), like OSPAR on the List of Chemicals for Priority Action (OSPAR 2010b).

Status of restrictions, bans or use

The Helsinki Convention (1974, 1992) has recommended special bans and restrictions on transport, trade, handling, use and disposal of PCBs. The Ministerial Declaration from 1998, within HELCOM and the 1995 Declaration of the Fourth international conference of the protection of the North Sea called for measures against toxic, persistent, bioaccumulating substances like PCBs to cease their inputs to the environment completely by the year 2020.

Under the Stockholm Convention, releases of unintentionally produced by-products listed in Annex C4, including dioxins and dl-PCBs, are subject to continuous minimization with the ultimate goal of elimination where feasible. The main tool for this is a National Action Plan which should cover the source inventories and release estimates as well as plans for release reductions. At the EU level, a Strategy for dioxins and PCBs was adopted in 2001. The Strategy includes actions in the area of feed and food contamination and actions related to the environment, including release reduction. Over the past decade, important legislation has been adopted to reduce the emissions of PCDD/Fs, in particular in the areas of waste incineration and integrated pollution prevention and control. Releases of POPs, including dioxins, from industrial installations are mainly regulated by the IPPC Directive and the Waste Incineration Directive, the former requiring Member States to establish permit conditions based on the Best Available Techniques (BAT) for a wide variety of industry sectors, and the latter setting maximum permissible limit values for PCDD/F emissions to air and water from waste incineration. The proper and timely implementation and enforcement of the IPPC Directive remain a key priority in order to ensure the necessary reduction of emissions from major industrial sources. However, at present or in the near future, nonindustrial sources are likely to exceed those from industrial sources (Quass et al. 2004).

GES boundaries and matrix

Existing quantitative targets

Table 3.6. Existing quantitative targets for PCBs.

PCBs	
Source	**Value and description**
OSPAR	
EAC sediment	CBs µg/kg dw 2.5% TOC. CB-28: 1.7, CB-52: 2.7, CB-101: 3.0, CB-118:0.6, CB-138: 7.9, CB-153: 40, CB-180: 12.
EAC $_{passive}$ fish	CBs µg/kg lw. CB-28: 64, CB-52: 108, CB-101: 120, CB-118:24, CB-138: 316, CB-153: 1600, CB-180: 480
EAC mussel	CBs µg/kg dw. CB-28: 3.2, CB-52: 5.4, CB-101: 6.0, CB-118:1.2, CB-138: 15.8, CB-153: 80, CB-180: 24
BAC sediment	CBs µg/kg dw 2.5% TOC. CB-28:0.22, CB-52: 0.12, CB-101: 0.14, CB-118:0.17, CB-138: 0.15, CB-153: 0.19, CB-180: 0.1
BAC fish	CBs µg/kg ww from CEMP 2008/2009. CB-28: 0.1, CB-52: 0.08, CB-101: 0.08, CB-118:0.1, CB-138: 0.09, CB-153: 0.1, CB-180: 0.11. ∑ICES7CBs from ASMO 09/7/3: 1.2.
BAC mussel	CBs µg/kg dw. CB-28: 0.75, CB-52: 0.75, CB-101: 0.7, CB-118:0.6, CB-138: 0.6, CB-153:0.6, CB-180: 0.6
EC	
Proposed MAQ-EQS (marine waters)	3.2 10^{-5} µg/l
Proposed AA-EQS biota (marine waters)	0.003 µg/kg ww
Food stuff directive	Sum of CB-28, CB-52, CB-101, CB-138, CB-153 and CB-180 (ICES – 6) 75 ng/g ww (with exeptions)
Effect Range -Low	
ERL sediment	CBs µg/kg dw 2.5% TOC. Total CB: 23 ∑ICES7CBs: 11.5
PCBs Recommended GES boundary	Water: non-applicable
	Sediment: OSPAR EAC for CB 118 and 153
	Biota: OSPAR EAC for CB 118 and 153
	Seafood: ∑6PCBs (28, 52, 101, 138, 153, 180): 75 µg/kg ww.

Table 3.7. Existing quantitative targets for dioxins/furans and dioxin-like PCBs.	
PCDD/Fs and DL-PCBs	
Source	**Value and description**
OSPAR	
(OSPAR, 2010a)	No target levels for PCDD/Fs
EAC sediment	PCB-118 (dl-PCB): 0.6 ug/kg dw
EAC fish	PCB-118 (dl-PCB): 24 ug/kg lw
EAC mussel	PCB-118 (dl-PCB): 1.2 ug/kg dw
BAC/BC sediment	PCB-118 (dl-PCB): 0.17 ug/kg dw (BAC)
BAC/BC fish	PCB-118 (dl-PCB): 0.6 ug/kg ww (BAC)
BAC/BC mussel	PCB-118 (dl-PCB): 0.1 ug/kg dw (BAC)
LC sediment	PCB-118 (dl-PCB): 0.05 ug/kg dw
EC	
EQS water	-
Draft EQS water	-
Draft EQS biota	4.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs) 8.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs+dl-PCBs) Human health is the critical endpoint EQS_{Biota}, Predators (secondary poisoning): 0.23 ng / kg ww (Σ PCDDs+PCDFs)
Draft EQS sediment	ng WHO_{98}-TEQ / kg dw (Σ PCDDs+PCDFs+dl-PCBs) Benthic community: 0.85 ng / kg dw (Σ PCDDs+PCDFs)
Draft EQS biota	
Food stuff directive	(EC, 2006), 881/2006/EC* Muscle meat of fish and fishery products and products thereof with th exception of eel: 4.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs) 8.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs+dl-PCBs) Muscle meet of eel (*Anguilla nguilla*) and products thereof: 4.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs) 12.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs+dl-PCBs) Fish liver: 25.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs+dl-PCBs)
Canadian Sediment Quality Guidelines PEL sediment	Probable Effect Level: 21.5 ng TEQ/ kg dw (PCDD/Fs)
Canadian Sediment Quality Guidelines TEL sediment	Threshold Effect Level: 0.85 ng TEQ/ kg dw (PCDD/Fs)

PCDD/Fs Recommended GES boundary	Water: non-applicable Sediment: 0.85 ng / kg dw (Σ PCDDs+PCDFs) Biota: 0.23 ng / kg ww (Σ PCDDs+PCDFs) Seafood: Fish muscle: 4.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs), 8.0 ng WHO_{98}-TEQ / kg ww (Σ PCDDs+PCDFs+dl-PCBs)

* NOTE: This regulation is under amendment and new target levels should be decided during 2011. The new levels will be based on the WHO-2005 TEF-values (current levels are based on the WHO-1998 TEFs).

The CORESET expert group decided that, due to uncertainties in the target setting on the OSPAR and EU working groups, the PCBs should be assessed by two congeners only: CB-118 (dioxin like) and 153 (non-dioxin like). Tentatively the OSPAR EACs for these two congeners are suggested to be used.

The Scientific Committee on Health and Environmental Risks (SCHER) criticized the derivation of sediment and biota QSs for dioxins (SCHER 2011). Therefore, the GES boundaries for dioxins are seen as tentative until WFD WG E proposes new QSs.

Preferred matrix

Due to the hydrophobic properties of PCBs and PCDD/Fs, monitoring of very low concentrations in the water column is not appropriate. PCBs and PCDD/Fs accumulate in sediments and biota in the aquatic environment, which are thus preferred matrices for monitoring. E.g. coastal surface sediment and muscle tissue of fat fish (e.g. herring, salmon). For guidance on monitoring strategies, see e.g. the Guidance Document for chemical monitoring under the Water Framework Directive (EC 2009).

Dl-PCBs elicit toxic effects through the Ah receptor and thereby contribute to the TEQ of a sample. Thus, dl-PCBs should be included in the quantitative target level if it is based on total TEQ.

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Present status of monitoring network in the Baltic Sea is presented in **Table 3.8** below.

Table 3.8. Monitoring of PCBs and dioxins/furans in the Baltic Sea.

Nation	Sediments		Biota		Water	
	PCBs (ICES 7)	PCDD/Fs	PCBs (ICES 7)	PCDD/Fs	PCBs (ICES 7)	PCDD/ Fs
Denmark			Yearly in shellfish (13) and fish (17)	Yearly (7 mussels, 17 fish)		
Germany	Yearly		Yearly		Twice a year	
Poland	3—5 years	No regular monitoring, screening	Yearly	Yearly (randomly through-out the year), herring, sprat, salmon (3 sites)		
Russia	Yearly					
Sweden	Yearly		Yearly	Yearly (autumn), herring (3 sites) and guillemot eggs (1 site)		-
Finland	6-10 years		Yearly	Yearly (autumn), herring 4 sites. Surveys, several species, 7 sites (2002-2003, 2009-2010)		
Estonia			Yearly			
Lithuania	Yearly (starts 2008)		Yearly in fish (State Food and Veterinary Service)	Yearly in fish (State Food and Veterinary Service)	yearly (starts 2008)	
Latvia			2002			

Gaps in the monitoring of the compound

There are no big gaps.

Present status assessments

Known temporal trends (also from sediment core profiles)

As a result of measures taken to reduce discharges of PCBs to the environment, concentrations of PCB, including CB-153 and CB-180, show significant declining trends for herring, perch and blue mussels in several regions surrounding the Baltic Sea. Noteworthy is that few of the presently available data sets have time series long enough (to draw statistical conclusions regarding time trends with an annual change of 5%. In herring, perch, mussel and cod, time series between 14 to 22 years are required to detect such changes (Bignert et al. 2004).

Decreasing trends for other PCB congeners, as well as for content of the sum of seven PCBs are also reported for some locations along the Baltic Sea (Bignert et al. 2008, GIOŚ 2007). It is estimated that estimated levels of sPCB along the Swedish coast in fish and mussels are decreasing with approximately 5-10% per year since the end of the seventies (Bignert et al. 2008). The analysis of dated slices of laminated sediment cores, however, revealed big regional differences in temporal trends (Schneider & Leipe 2007).

For dioxins, there are few historical sediment data (profiles) from the Baltic Sea and some data are from the late 1980s and thus unable to reveal very recent trends. All the cores, however, show a decline in surface PCDD/F concentrations compared with deeper sediments, with the highest concentrations generally dated back to the 1970s or 1960s in the northern basins, the Baltic Proper and the Kattegat - Danish straits.

There is not much information about past or recent trends in PCDD/F concentrations in different fish species and generally the data do not cover past decades. The Swedish Museum of Natural History reported dioxin concentrations in the muscle of small herring collected from 1990 to 2008 at three stations on the Swedish coast that showed no indications of change during that period, but the guillemot egg data showed a major and significant decrease since 1970 (Bignert et al. 2010). Similarly, no decreasing trend of PCDD/Fs or dl-PCBs was observed in fish from the southern Baltic Sea during 2002–2006 (Szlinder-Richert et al. 2009). Recently, Karl et al (2010) repeated a study on PCDD/F and dl-PCB concentrations i herring from the south and western Baltic Sea (Karl & Ruoff 2007) and concluded that the TEQ concentrations had not changed between 1999 and 2006. Thus, there seems to have been a levelling off of the concentrations in fish from many areas in the Baltic Sea during the last decades.

Spatial gradients (incl. sources)

The levels of PCBs are about five times higher in the Baltic Sea compared to the North Sea (Mehtonen 2009). Concentrations more than three times above threshold levels in the Baltic Sea were found in the Little Belt, southern parts of the Kattegat, the Sound, the Szczecin Lagoon, southern parts of the Bothnian Sea, and in the Bothnian Bay (HELCOM 2010). In contrast, the concentrations of CB-180 were not found to exceed the threshold level (EAC, 0.480 mg kg^{-1} lipid weight) (OSPAR 2009) in any part of the Baltic Sea. The highest concentrations of CB-180 in this assessment were found in the Pomeranian Bay, where concentrations were between 0.100 and 0.200 mg kg^{-1} lipid weight.

For dioxins, sediment surveys have revealed some major sediment contamination with dioxins in the River Kymijoki estuary, Finland (Isosaari et al. 2002; Verta et al. 2007) and a more local contamination on the Swedish coast of the Gulf of Bothnia (Sundqvist et al., 2009) originating from local industrial sources. Major data gaps are currently for the southeastern and eastern coastal regions of the Baltic Proper and the southern Gulf of Finland.

Numerous recent papers have shown differences in PCDD/F and dl-PCB concentrations in Baltic herring, sprat and salmon between the Baltic Sea basins (e.g., Bignert et al. 2010; Karl et al. 2010). Higher concentrations have been detected in the northern basins where dioxin and dl-PCB levels in herring exceed established maximum limit concentrations for human consumption. Regional variation within a sub-basin has been found in the Swedish coastal region of the Bothnian Sea (Bignert et al. 2007). Since the atmospheric deposition pattern (lowest in the north) is different from concentrations in fish (generally highest in the north), other factors or sources are thus likely to be involved in determining concentrations in fish. The reasons remain unclear, but higher historical PCDD/F discharges from point sources in the northern basins have been suggested. In general, the contribution from the dl-PCBs to the TEQ is substantial and seems to increase the further south in the Baltic region the samples are collected.

Recommendation

PCBs and PCDD/Fs should be included as a core indicator in the Baltic Sea. The substances have adverse impacts on the environment, they are monitored throughout the entire Baltic Sea area and there are several quantitative targets (i.e. GES boundaries) available.

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3.5. Polyaromatic hydrocarbons (PAH)

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General information

General properties

There are 16 PAHs that are recommended as priority pollutants by the U.S. EPA, WFD and MSFD. These PAH compounds include two-ring compounds (naphthalene); three-ring compounds (acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene); four-ring compounds (fluoranthene, pyrene, benzo(a)anthracene, chrysene); five-ring compounds (benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h) anthracene); six-ring compounds (indeno(1,2,3-c,d)pyrene, benzo(g,h,i)perylene). Due to their low water solubility and hydrophobic nature, PAHs tend to associate with particulate material. The deposition of these particles in rivers and coastal waters can lead to an accumulation of PAHs in the sediment. PAHs are persistent, especially in anaerobic sediments, with the higher molecular weight PAHs being more persistent than the lower molecular weight compounds (Kennish 1997; Webster et al. 2003). Bioaccumulation of PAHs from sediments by marine organisms depends, thermodynamically, on the ratio between adsorption capacity of the sediment and absorption capacity of the organism. Different profiles of contaminants have been observed in organisms of different trophic levels. These differences were attributed to a partial biotransformation of the contaminants in the organisms of higher trophic levels (Baumard et al. 1998b).

Main impacts on the environment and human health

Polycyclic aromatic hydrocarbons (PAHs) are of concern due to their persistence and potential to accumulate in aquatic organisms, particularly invertebrates, such as bivalves and crustaceans. In most vertebrates, PAHs are fairly rapidly metabolized, but they and their toxic intermediates emerging during metabolic degradation can cause deleterious effects in fish.

PAH are important environmental contaminants which may lead to increased levels of neoplastic aberrations or tumors in fish liver. Therefore monitoring of PAH and their effects are part of several international environmental programmes.

Epoxides originating from the oxidation of PAH by cytochrome P4501A1 may be further oxidised to carcinogenic diolepoxides. These dioleepoxides are known to bind to DNA and/or cause mutations which may lead to cancer. Increased levels of neoplastic aberrations or tumors were found in fish which have been exposed to PAH contaminated sediments. For this reasons PAH contamination in marine ecosystems is a cause for concern. To assess the PAH exposure of fish, concentrations of the main metabolites such as 1-hydroxypyrene, 1-hydroxyphenanthrene and 3-hydroxybenzo(a)pyrene can be determined in bile by HPLC with fluorescence detection (HPLC-F), by synchronous fluorescence scanning, gas chromatography with mass selective detection (GC/MS) and also by UPLC/MS/MS (Bayer et al. 2010; Ariese et al. 2005). PAH metabolites in fish bile reflect the exposure of the fish to PAHs via sediment and food usually during the last days – depending on the feeding activity.

Status of a compound on international priority lists and other policy relevance

Anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, indeno(1,2,3-cd), pyrene, fluoranthene and naphthalene are identified as priority substances by European Commission (Directive 2008/105/EC). PAHs are included in the OSPAR list of substances of priority action.

For US EPA it is the list of 16 priority compounds quoted under 1.1.

Relevance of the indicator for describing the developments in the environment

Low-molecular-weight PAH compounds, containing two or three rings, are acutely toxic to a broad spectrum of marine organisms. Examples of low-molecular-weight PAHs that tend to be toxic are anthracene, fluorene, naphthalene and phenanthrene. The high-molecular-weight PAH compounds, containing four, five, and six rings, are less toxic but have greater carcinogenic potential, e.g. benzo(a)pyrene, dibenz(a,h) anthracene, benzo(b)fluoranthene (Kennish 1997). While PAHs can be weakly carcinogenic or non-carcinogenic, they can modify the carcinogenic activity of other PAHs in complex mixtures (Marston et al. 2001). Therefore, synergistic effects of PAHs can be larger than the total levels of PAHs would indicate. Higher concentrations of PAHs are also harmful to reproduction of fish and can damage cellular membrane structures (Knutzen 1995). When PAHs are exposed to sunlight, the mechanism known as phototoxicity is involved, producing reactive and toxic photomodification products (HELCOM 2010).

Anthropogenic PAH sources in the marine environment include the release of crude oil products (petrogenic source) and all types of incomplete combustion of fossil fuels—coal, oil and gas or wood and waste incineration (pyrolytic sources) (Neff 2004). Some PAHs are formed naturally, but the majority of PAHs in the marine environment come from anthropogenic activity. Each source generates a characteristic PAH pattern, enabling distinction of the sources in a sample; concentration relationships of individual PAH compounds can be used to reveal the sources of the PAH compounds (Baumard et al. 1998, Sicre et al. 1987, Yunker et al. 2002). Molecular indices calculated from both sediment and biota showed that pyrolytic sources predominate in the Baltic Sea PAH contamination. However, in the Gulf of Finland and some areas in the western Baltic Sea (Sound, Belt Sea and Kattegat), molecular indices indicated a significant contribution of petrogenic PAHs. This may indicate that atmospheric deposition combined with shipping activities is the main source of PAHs in these areas. The dominance of pyrolytic sources could be surprising in view of the heavy maritime traffic and illegal oil discharges. On the other hand, no reliable information is available on the airborne deposition of PAHs onto Baltic Sea surface waters (Pikkarainen 2004).

Status of restrictions, bans or use

The maximum levels of benzo(a(pyrene and also a sum of benzo(a)pyrene, benz(a)anthracene, benzo(b) fluoranthene and chrysene are regulated in food stuff according to the Commission Regulation (EC) No 835/2011. There are no other regulations for the production or use of PAHs.

GES boundaries and matrix

Existing GES boundaries

Note: targets based on EQS are subject to changes

Table 3.9. Existing quantitative targets for PAHs.

Substance	GES boundary for biota	GES boundary for sediment	GES boundary for water
Dibenz[ah]anthracene		ER-L 63.4 µg kg^{-1} dw	
Fluoranthene	EAC mussel 110 µg/kg dw (OSPAR 2009)	EAC/ERL 600 µg kg^{-1} dw (OSPAR 2009)	EQS: 0.1 µg L^{-1}
Anthracene	EAC Mussel 290 µg/kg dw (OSPAR 2009)	EAC/ERL 85 µg kg^{-1} dw (OSPAR 2009)	EQS: 0.1 µg L^{-1}
Naphthalene	EAC mussel: 340 µg kg^{-1} dw (OSPAR 2009)	EAC/ERL 160 µg kg^{-1} dw (OSPAR 2009)	EQS: 1.2 mg L^{-1}
Benzo[ghi]perylene	EAC Mussels 110 µg/kg dw (OSPAR 2009)	EAC/ERL 85 µg kg^{-1} dw (OSPAR 2009)	BghiP+I123cdP EQS: 0.002 µg L^{-1}
Benzo[a]pyrene	EAC Mussels 600 µg kg^{-1} dw (OSPAR 2009)	EAC/ERL 430 µg kg^{-1} dw (OSPAR 2009)	BaP EQS: 0.05 µg L^{-1}
Benzo[k]fluoranthene	EAC Mussels 260 µg kg^{-1} dw (OSPAR 2009 back-ground doc)		BbF+BkF EQS: 0.03µg L^{-1}
Benzo[b]fluoranthene			BbF+BkF EQS: 0.03µg L^{-1}
Pyrene	EAC Mussels 100 µg kg^{-1} dw (OSPAR 2009)	EAC/ERL 665 µg kg^{-1} dw (OSPAR 2009)	
Fluorene			
Benz[a]anthracene	EAC Mussels 80 µg/kg dw (OSPAR 2009)	EAC/ERL 261 µg kg^{-1} dw (OSPAR 2009)	
indeno[1,2,3-cd] pyrene	BAC mussel: 2.4 µg/kg dw (OSPAR 2009)	EAC/ERL 240 µg kg^{-1} dw (OSPAR 2009)	BghiP+I123cdP EQS: 0.002 µg L^{-1}
chrysene	BAC mussel: 8.1 µg/kg dw (OSPAR 2009)	EAC/ERL 384 µg kg^{-1} dw (OSPAR 2009)	
phenanthrene	EAC Mussels: 1700 µg kg^{-1} dw (OSPAR 2009)	EAC/ERL 240 µg kg^{-1} dw (OSPAR 2009)	
acenapthylene		ER-L 44 µg kg^{-1} dw	
acenapthene		ER-L 16 µg kg^{-1} dw	
1-hydroxypyrene	483/909 ng/g[GC-MS] for cod/turbot		
1-hydroxyphenan-threne	518/1832 ng/g [GC-MS] for cod/turbot		

Background response and Assessment Criteria for metabolites

EAC and BAC values depend on the metabolite, the fish species as well as on the analytical method used. Both values can be used for the North Sea as well as the Baltic Sea if species fits.

The recommended way to calculate BACs is to use the 90th percentile of reference site data. Possible reference sites are Iceland and Barents Sea. Data from additional reference sites may improve the quality of the BAC in future. BACs have been calculated for dab, cod and haddock. In Fig. 3.1 monitoring results are presented in two colours representing the proportions of the fish above and below the BAC. No EAC values were exceeded. Looking at the values regional differences seems to be much more important than species differences. If BAC should be developed for fish species which prefere more polluted habitats (coast), it could be helpful to use an species independent BAC for fish instead. SGIMC 2010 proposed a joint BAC for three fish species cod dab and haddock. A BAC value applicable for more marine fish species would be helpful regarding inter-species evaluation of monitoring data.

Even if PAH metabolites are only a marker of exposure, high levels of metabolites can be linked to deleterious effects in fish. EACs have been be identified using results from toxicological experiments linking oil exposure and PAH metabolites in fish with DNA adducts and fitness data (Morton et al. 2010; Skadsheim et al. 2004; Skadsheim et al. 2009), where the latter serves as the effect quantity for the calculation of the EAC presented in **Table 3.10**. EAC are available for some fish species only at the moment. More investigations are needed to calculate proper EAC values for Baltic fish species. EACs cannot be easily transferred to other species because they may differ in sensitivity to PAH effects.

Table 3.10. Background Assessment Criteria (BAC) and Environmental Assessment Criteria (EAC) for two PAH metabolites, different fish species and methods. Data partly taken from WKIMC 2009 and SGIMC 2010.

Biological Effect	**Fish species**	**BAC [ng/ml] HPLC-F**	**EAC [ng/g] GC/MS**
Bile metabolite 1-hydroxypyrene	dab	16	
	cod	21	483
	flounder	16 [4)]	
	haddock	13	
	dab, cod, haddock	17	
	turbot		909
	halibut		745
Bile metabolite 1-hydroxyphenanthrene	dab	3.7	
	cod	2.7	518
	flounder	3.7 [4)]	
	haddock	0.8	
	dab, cod, haddock	2.4	
	turbot		1832
	halibut		262

Biological Effect	**Fish species**	**BAC [µg/ml] Synchronuos Fluor. 341/383**	**EAC [µg/ml] Fixed Fluor. 341/383**
Bile metabolites of pyrene-type	dab	0.15	22 [1)]
	cod	1.1	35
	flounder	1.3	29 [2)]
	haddock	1.9	35 [3)]
	turbot		29
	halibut		22
	herring/sprat		16

AC based on [1)]halibut, [2)]turbot, [3)]cod and [4)]dab

GES boundary for food: Benz(a)Pyrene (fish): 2 µg/kg ww; Benz(a)Pyrene (crustaceans): 2 µg/kg ww; Benz(a)Pyrene (bivalves): 2 µg/kg ww.

Quality Assurance of PAH metabolite targets

In ICES Times No. 39 there are several methods described and equally recommended. Due to different principles not all of the methods can be compared.

BONUS+ project BEAST conducted an intercalibration for PAH metabolites in fish bile in 2010. The results showed a good comparability between the participating labs as well as between the analytical methods GC-MS, HPLC-F and synchronous fluorescence scanning. A factor was used to compare synchronous fluorescence to the other two methods mentioned above.

Confounding factors

The season and the feeding status (freshly filled gall bladder or not) seem to be confounding factors for fish. Normalisation of metabolite concentration to bile pigments (expressed as absorption at 380nm or biliverdin concentration) can help to reduce variation in some data sets (Kammann 2007, Ariese et al. 1997). In other data sets normalisation leads to no advantage. Male and female fish tend to have different levels of PAH metabolites (Vuorinen et al. 2006).

Monitoring of the PAHs

Status of monitoring network (geographical and temporal coverage)

PAHs are monitored mostly in sediments (or in water), but not in mussels (except Denmark and Sweden). In Finland the monitoring of PAHs in sediments it is not coherent regarding sites & frequency, not nearly all sites visited yet. Lithuania analyses only 8 individual PAHs: naphthalene, anthracene, flouranthene, benzo(b) fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, benzo(g,h,i)perylene and indeno(1,2,3-cd)pyrene.

PAH metabolites:
Baltic countries with regular monitoring of PAH metabolites in fish bile: Germany and Denmark.
Baltic countries with pre-monitoring stage of PAH metabolites investigations: Finland, Poland

Temporal coverage

Temporal trends of PAH concentrations in biota and surface sediments cannot be assessed in the majority of the Baltic Sea area due to temporally and spatially fragmented data sets. In Denmark sediments will be taken according to a rotating sampling scheme, so no temporal trends will be available for sediments.

Present status assessments

Temporal development of PAHs concentrations in sediments and biota

Assessment

Taking into account data from 1999 to 2008, temporal trends for individual PAHs have been determined using Danish national monitoring data. Benzo(a)pyrene concentrations in mussels from Århus Bight and Sound were characterized by statistically significant decrease, while mussels from Great Belt show temporally relatively constant concentrations. However, it is difficult to detect and interpret temporal variation without long time series and case studies, including examination of environmental condition (HELCOM 2010).

The highest levels of PAHs are observed in lagoon areas (e.g. Szczecin lagoon), in the vicinity of harbours (e.g. port of Copenhagen) or in the accumulation areas (e.g. Arkona Deep or Gdańsk Deep). In general, the concentrations of light molecular weight PAHs like fluoranthene and phenathrene in the Baltic biota and sediments do not exceed the OSPAR toxicity threshold values (OSPAR, 2009) in any of the sub-regions. The heavy molecular weight compound benzo(a)pyrene, which has been shown to be highly toxic, carcinogenic and mutagenic, is below the threshold values both in sediment and bivalves in the whole the sea area. Benzo(g,h,i)perylene is present in high concentrations in the Baltic Sea sediments, often exceeding the threshold values. In bivalves and sediments, it was found to exceed the threshold value in the southern and south-western sea areas. Benzo(b)fluoranthene was found to exceed the threshold values in sediments in all the other basins except Bothnian Sea and Bothnian Bay. However, the threshold value for benzo(b) fluoranthene is not normalized to sediment carbon and therefore the spatial comparison may be misleading due to different sea bed characteristics (HELCOM 2010).

Detectable concentrations of anthracene have been found in fish from Swedish background stations. It has been measured in sediment from the Stockholm area (with concentrations falling inversely with distance from central Stockholm) and homogeneous coastal samples, indicating small local impact. It has also been measured in detectable concentrations in water areas sampled with the use of passive sampling devices. Fluoranthene is frequently present in fish from Swedish background stations, and also found in sediment and sludge. It has been found in all water samples from Sweden taken by means of passive sampling devices, and it is detectable in groundwater samples (Swedish EPA 2009).

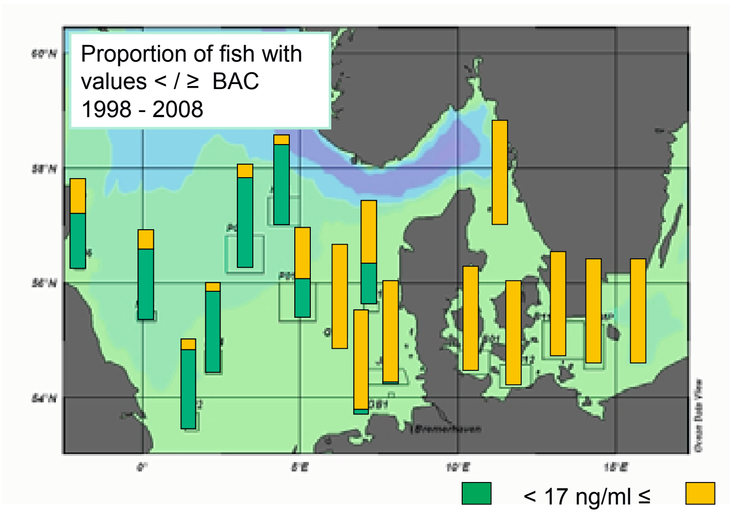

***Figure 3.3**. 1-Hydroxypyrene in bile fluids of dab, flounder and cod caught between 1998 and 2007 categorized by the species overarching BAC of 17 ng/ml. Proportion of single fish per station are categorized in relation to BAC (SGIMC 2010)*

Clear differences in PAH metabolite concentration in fish bile between the lower contaminated central North Sea and the higher contaminated western Baltic Sea have been detected (**Figure 3.3**). In distance of point sources there are no temporal trends detectable in dab and flounder from the North Sea and the western Baltic Sea caught during 1997 and 2004 (Kammann 2007). Lower values than in North Sea (dab, cod, flounder, haddock) and Baltic Sea (flounder, cod, herring, Vuorinen et al. 2006; eelpout) have been detected in Barents Sea (cod) and near Iceland (dab). Higher concentrations are present in fish caught in harbour regions or in coastal areas (eelpout, Kammann and Gercken 2010).

Strengths and weaknesses of data

For tracing sources of alkylated versions of PAHs would be preferred over the parent PAHs. E.g. Sweden will include alkylated versions in 2011 in monitoring in order to evaluate whether they should be included in the yearly monitoring program.

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3.6. Lead, Cadmium and Mercury in fish and shellfish

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General information

General properties

Metals are naturally occurring substances that have been used by humans since the iron age. The metals Cd, Pb and Hg are the most toxic and regulated metals today, and have no biological function. Mercury is bioaccumulated, mainly in its organic form (Methyl-mercury) and due to high evaporation pressure can be transported from soil to the Baltic, and concentrate in the arctic.

Main impacts on the environment and human health

Lead and mercury have been connected to impaired learning curves for children, even at small dosage. Lead can cause increased blood preasure and cardio-vaskular problems in adults. Acute metal poisoning generally results in vomiting. Long term exposures of high levels of lead and mercury can affect the neurological system. Mercury can lead to birth defects as seen in Minimatta bay among fishermen in a mercury polluted area, and also after ingestion of methylmercury treated corn in Iran. Cadmium is concentrated in the kidney, and can result in impaired kidney function, and cadmium can exchange for calcium in bones and produce bone fractures (Itai-Itai disease).

The HELCOM thematic assessment of hazardous substances (HELCOM 2010) showed high concentrations of mercury and cadmium in biota and sediment all over the Baltic Sea. Lead was not assessed.

Status of the indicator on international priority lists and other policy relevance

Mercury and cadmium are included in the HELCOM Baltic Sea Action Plan. All the three metals are included in the EU WFD (Pb and Cd in water, Hg in biota) and EU shellfish directive in shellfish. Part of EU food directives, limits set in a range of fish species, shellfish and other seafood. In the OSPAR CEMP to be measured on a mandatory basis in fish, shellfish and sediment (OSPAR 2010).

Status of restrictions, bans or use

All three metals have been used for centuries, but in the last decades have been banned for most uses. Today, cadmiums and mercurys main use is in rechargeable batteries, and for mercury low energy light sources. Main source of all three metals are burning of fossile fuels. The air deposition is mainly long range transport from outside the Baltic Sea catchment area (60-84%). Sources of mercury was use in amalgams for dentist, reduced by installing mercury traps in sinks and generally reducing amalgams in dental works, as electrodes in paper bleaching, in thermometers and mercury switches and a range of other products that have been faced out. Current legal use include batteries and low energy light sources. For lead, the main source was leaded fuels until their ban in Europe in the 1990ies. Both cadmium and lead have hot-spots in connection with metal processing facilities, and cadmium is coexisting with all zinc ores, and typically present at levels of 0,5- 2% in the final products. Weathering of outdoor zinc-products thus leads to cadmium pollution. Current legal use of cadmium includes rechargeable Ni-Cd batteries and for lead car batteries.

GES boundaries and matrices

Existing quantitative targets e.g. EACs, BACs and food safety standards

Metals have been assessed since the beginning of OSPAR and HELCOM conventions, and the OSPAR assessment criteria are available for cadmium, lead and mercury.

Table 3.11. OSPARs assessment criteria for metals, as used in QSR 2010 (OSPAR 2010). Note that the OSPAR criteria currently are under review, with the target to update them by 2012.

	Sediment (µg/kg dry weight)			*Mussels* (µg/kg dry weight)			*Fish* (µg/kg wet weight)	
	BC	*BAC*	*ERL*	*BC*	*BAC*	*EC*	*BAC*	*ECfood*
Cd	*200*	*310*	*1200*	*600*	*960*	*5000*	*26*	*1000$*
Hg	*50*	*70*	*150*	*50*	*90*	*2500*	*35*	*500*
Pb	*2500*	*3800*	*47000*	*800*	*1300*	*7500*	*26*	*1500$*

$ bivalve tissue.

Table 3.12. *EUs assessment criteria for metals in WFD (2008/105).*

	Water (µg/l)		*Biota* (µg/kg wet weight)		*Sediment* (µg/kg wet weight)	
	AA-EQS	*Mac-EQS*	*AA-EQS*	*Mac-EQS*	*AA-EQS*	*Mac-EQS*
Cd	*0.08-0.25/ 0.2*	*0.45 – 1.5*	*n.a.*	*n.a.*	*n.a.*	*n.a.*
Hg	*0.05$*	*0.07*	*20*	*n.a.*	*n.a.*	*n.a.*
Pb	*7.2*	*n.a.*	*n.a.*	*n.a.*	*n.a.*	*n.a.*

$: If memberstates don't use biota AA-EQS, the water AA-EQS should have the same protection power as the biota AA-EQS.

n.a.: Not applicable

Table 3.13*. EUs assessment criteria for metals in marine food sources 466/2001.*

	Fish (mg/kg wet weight)		*Shellfish* (mg/kg wet weight)		
	General	*specific*	*Crab*	*Bivalves*	*Octopus*
Cd	*0.05*	*0.1*	*0.5*	*1.0*	*1.0*
Hg	*0.5*	*1.0*			
Pb	*0,2*	*0,4*	*0.5*	*1.0*	*1.0*

Shellfish are available in the whole of the HELCOM area, but assessment criteria are based on *Mytilus edulis*, available in the more saline parts. The GES boundary values for other species should be verified against *Mytilus edulis*, and it should also be noted that different Mytilus species exists in the Baltic Sea (*Mytilus trossulus*), that is more adapted to low salinity waters. Other species used for monitoring is *Macoma baltica*.

Preferred matrix

Shellfish or sediment for local surveys.

Fish for regional surveys

Monitoring the parameter

Status of monitoring network (geographical and temporal coverage)

Mussels have been used world wide as a monitoring organism for metals. In the Baltic Sea, the latest HELCOM assessment indicated that there is a fairly dense grid of monitoring stations for mussels at the shoreline, but very few stations in the open Baltic Sea.

HELCOM and OSPAR guidelines for measuring metals in biota and sediment exist. Guideline for monitoring in water is being developed. Quality assurance in form of international workshops and intercalibrations has been organized yearly by QUASIMEME since 1993, with two rounds each year for water, sediment and biota.

Gaps in the monitoring of the compound

Further studies are needed for establishing the interspecies correlations for mussels. Existing data of sediment cores from the different regions of the Baltic Sea should be evaluated, and used to establish regional background concentrations. Normalization of sediment data to same level of TOC/clay-silt content should be tested.

Present status assessments

Known temporal trends (also from sediment core profiles)

The temporal trends for metals in biota are inconclusive, as there are both areas with increasing trends of mercury and cadmium, and areas with decreasing trends (HELCOM 2010). There are most decreasing trends, though.

Spatial gradients (incl. sources)

The spatial gradients are mainly linked to different species or sample type (liver/muscle in fish) for biota. In sediments, the highest levels are found in the Bothnian Bay, eastern Gulf of Finland, off the southeast part of Sweden and in the sound. Cadmium patterns is similar, except for low concentrations in The Sound and high Northern Baltic Proper, Western and Eastern Gotland Basins and the Pomeranian Bay.

Recommendation

Mercury, Cadmium and Lead have high policy relevance, they are well monitored and they have adverse effects on environment and therefore they should become core indicators.

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3.7. Cesium-137

Authors: Jürgen Herrmann, Günter Kanisch and Sven P. Nielsen
Acknowledged persons:
Members of the HELCOM MORS-PRO project

As the indicator already exists as a HELCOM Indicator Fact Sheet, it is presented in that format.

What are the concentrations of the artificial radionuclide caesium-137 in Baltic Sea herring?

Key message

- Overall, the Cs-137 activity concentrations in herring in the Baltic Sea basins are approaching pre-Chernobyl levels.
- Radioactive fallout over the Baltic Sea from the Fukushima accident in Japan in March 2011 is very small and may not be detectable in seawater and fish. The corresponding radiological risks are estimated to be negligible.

Policy relevance

The development and use of nuclear power for military and peaceful purposes have resulted in the production of a number of man-made radioactive substances. Explosions of nuclear weapons in the atmosphere distribute radioactive substances in the environment, while underground nuclear explosions release little or no radioactivity into the environment. The routine operations of nuclear power plants give rise to small controlled discharges of radioactive substances, but accidents at nuclear power plants can cause releases of considerable amounts of radioactivity into the environment. Man-made radionuclides of particular concern to man and the environment are 90Sr and 137Cs, which are both formed by nuclear fission.

A study on worldwide marine radioactivity enables a comparison of levels of anthropogenic radionuclides in Baltic seawater against those in other marine areas of the world. The Baltic Sea has the highest average 137Cs levels in surface water (IAEA 2005). Radioactive fallout from the Chernobyl accident in 1986 is the dominating source for 137Cs in the Baltic Sea. The levels of 137Cs in the Baltic Sea, both in water and biota, have shown declining trends since the early nineties.

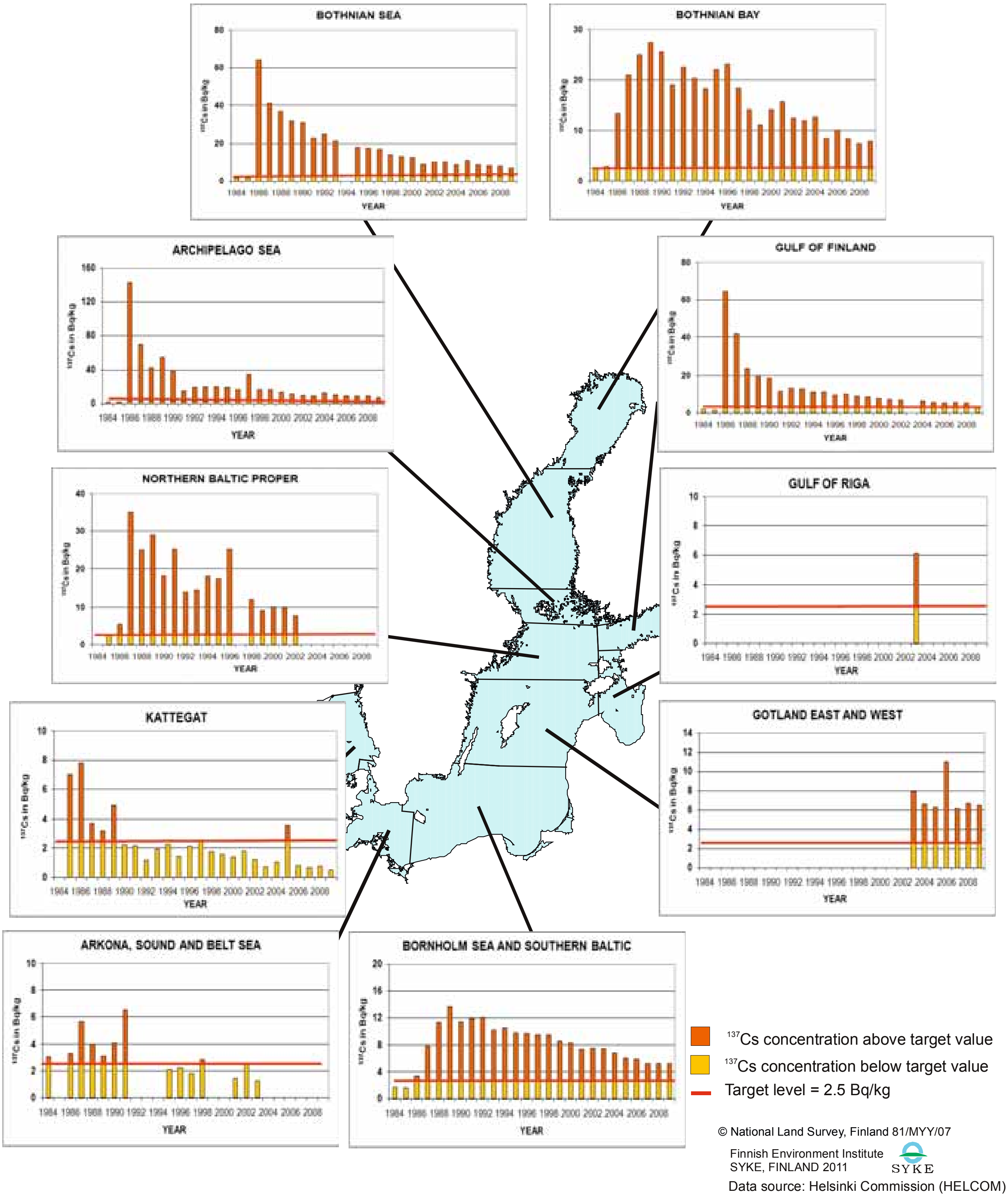

Figure 3.4. *137Cs concentrations (in Bq/kg) in herring muscle in 1984-2009, as annual mean values by basin. GES boundary values have been calculated as averages of pre-Chernobyl (1984-1985) concentrations. (Note: variable scales in the graphs)*

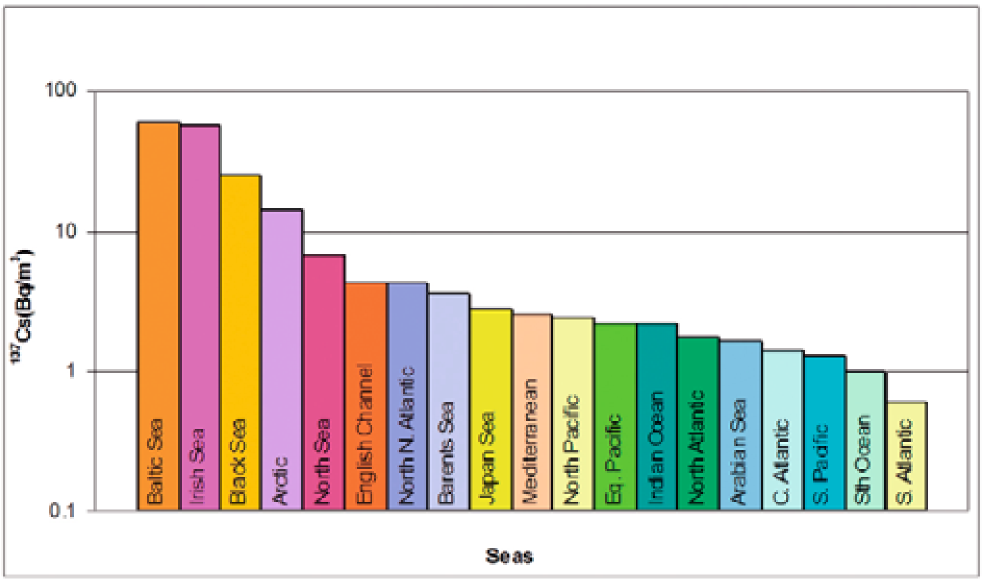

Figure 3.5. *Average surface levels of 137Cs in the world's oceans and seas (estimates for 01.01.2000).*

Ingestion of 137Cs in fish is the dominating exposure pathway of humans from man-made radioactivity in the Baltic Sea. Therefore, 137Cs concentrations in herring are well suited as indicators for man-made radio-activity in the Baltic Sea.

Internationally recommended maximum permitted concentrations of 137Cs in foodstuff are in the range 500-1250 Bq/kg depending on origin of pollution.

Reaching one of the ecological objectives given by the BSAP, i.e. "radioactivity at pre-Chernobyl level" defined by associated GES boundary values, will help to assure healthy wildlife and all fish being safe to eat, both with respect to radiation exposure. Concentrations of radioactivity in marine wildlife in the Baltic Sea have always been low causing negligible risks to wildlife from radiation exposure and to humans from consumption of seafood.

HELCOM Monitoring of Radioactive Substances (MORS) projects, and now the HELCOM MORS Expert Group, have been working to implement the Helsinki Convention on matters related to the monitoring and assessment of radioactive substances in the Baltic Sea. This work is based on HELCOM Recommendation 26/3 and supports the work of the HELCOM Monitoring and Assessment Group (HELCOM MONAS), by assessing the progress towards the ecological objective *Radioactivity at pre-Chernobyl level* which was defined in the HELCOM Baltic Sea Action Plan (BSAP).

This indicator supports the implementation of the EU Marine Strategy Directive (MSFD) Descriptor 9 on *contaminants in fish and seafood for human consumption.*

The work of HELCOM MORS also supports the implementation of the Euratom Treaty, of which all EU Member States are signatories, which requires actions in relation to monitoring and effects of discharges on neighbouring states.

Authors

HELCOM Monitoring of Radioactive Substances Expert Group

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Temporal trends in concentrations of the artificial radionuclide caesium-137 in plaice and flounder muscle as well as sea water in the Baltic Sea basins

Key message

- Overall, the Cs-137 activity concentrations in plaice and flounder muscle, as well as of surface waters, in the Baltic Sea basins are approaching pre-Chernobyl levels.
- Cs-137 is continuously transported from the Baltic Sea to the North Sea via Kattegat. Routine discharges of radioactivity from nuclear power plants into the Baltic Sea area are small and only detectable locally.

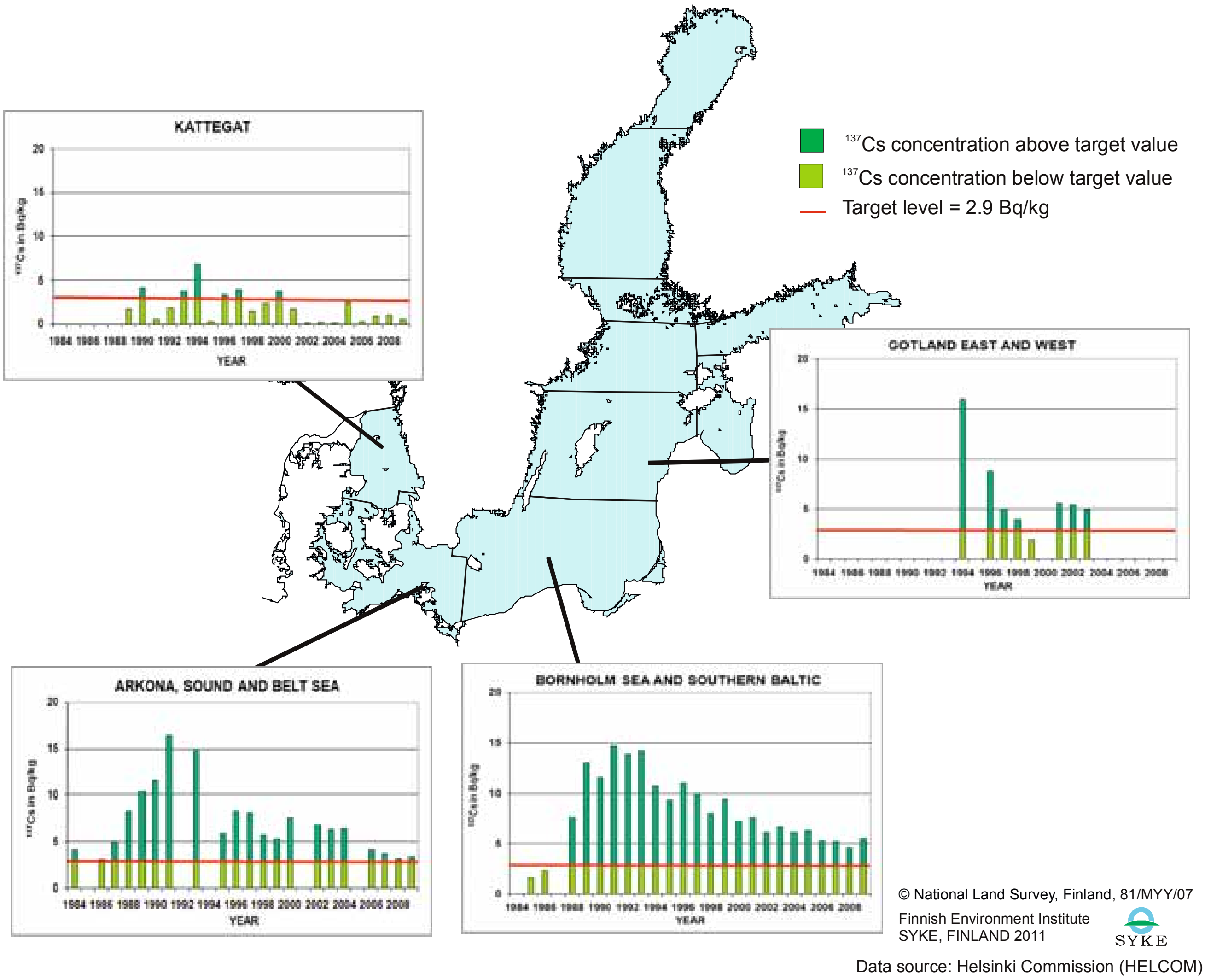

Figure 3.6. *137Cs concentrations (in Bq/kg) in plaice and flounder muscle in 1984-2009, as annual mean values by basin. GES boundary values have been calculated as average of pre-Chernobyl (1984-1985) concentrations.*

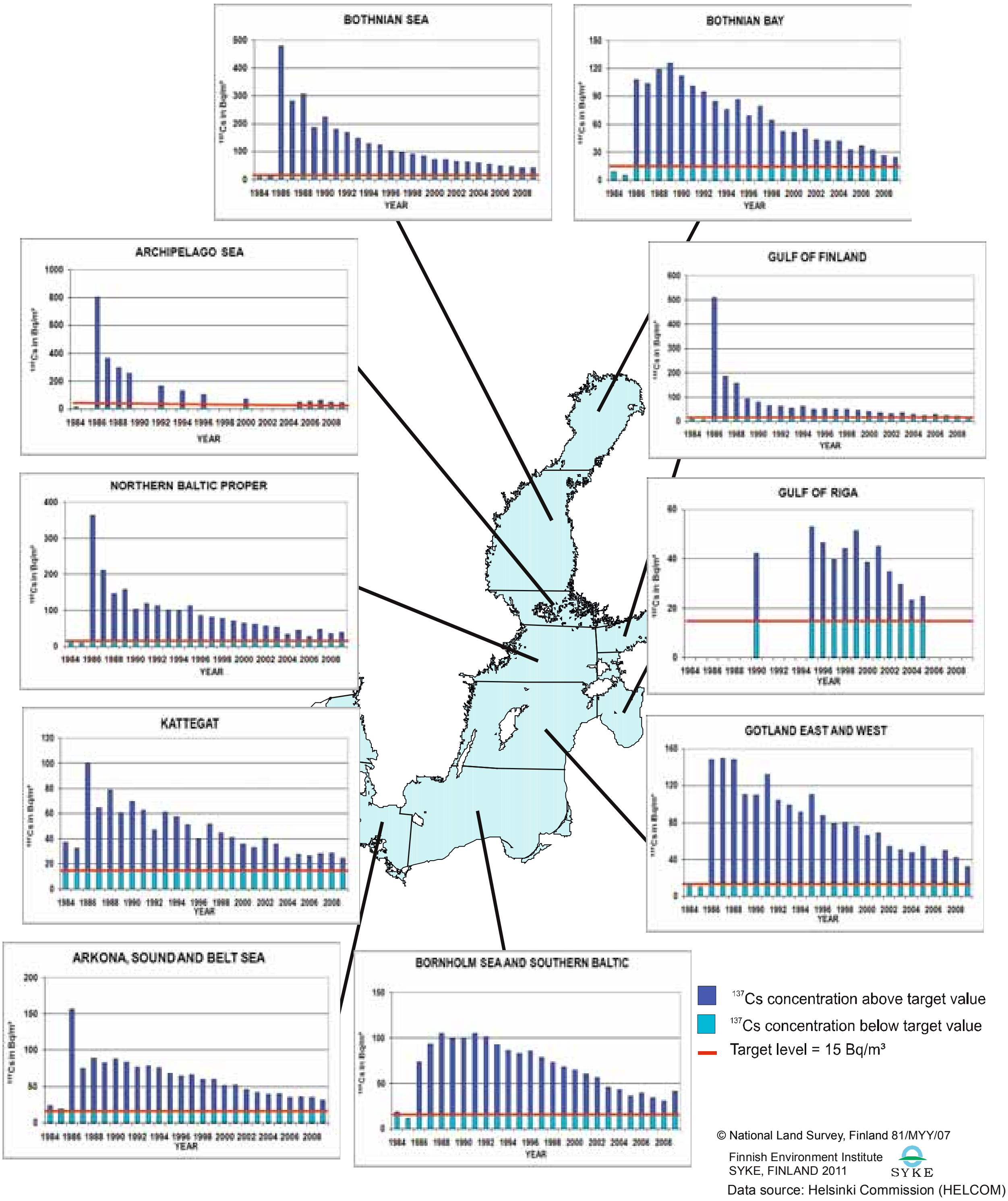

Figure 3.7. *137Cs concentrations (in Bq/m3) in surface water (sampling depth <=10m) in 1984-2009, as annual mean values by basin. GES boundary values have been calculated as average of pre-Chernobyl (1984-1985) concentrations. (Note: variable scales in the graphs)*

Background

The most significant source of artificial radioactivity in the Baltic Sea is fallout from the Chernobyl accident. The direct total input of ^{137}Cs from Chernobyl to the Baltic Sea was estimated at 4700 TBq. Secondary riverine input from Chernobyl fallout added further 300 TBq of ^{137}Cs.

Other important sources are global fallout from atmospheric nuclear weapons tests performed during the late 1950s and early 1960s and discharges from the nuclear reprocessing plants Sellafield and La Hague, located at the Irish Sea and the English Channel. The latter sources have become of minor radiological importance, due to significant reduction of ^{137}Cs discharges from Sellafield in the past two decades.

The Chernobyl accident resulted in the very uneven ^{137}Cs deposition in the Baltic Sea region with the Bothnian Sea and the Gulf of Finland having been the two most contaminated sea areas. Since 1986, the spatial and vertical distribution of Chernobyl-derived ^{137}Cs has changed as a consequence of river discharges, the mixing of water masses, sea currents, and sedimentation processes (Ilus 2007). In the early phase after Chernobyl, ^{137}Cs concentrations decreased rapidly in the Gulf of Finland and in the Bothnian Sea, while at the same time increasing in the Baltic Proper (**Figure 3.7**).

Assessment

During the period 1999-2009 concentrations of ^{137}Cs have continued to decrease in all regions of the Baltic Sea (**Figure 3.7**).

The effective half-life of a radioactive contaminant is the time required for its concentrations to decrease by 50% as a result of physical, chemical and biological processes. Half-lives are specific to each radionuclide and each environment where they may occur. Effective half-lives have been calculated for ^{137}Cs in various parts of the Baltic Sea. Currently, the effective half-lives of ^{137}Cs in surface water vary from 9 years in the Bothnian Bay to 15 years in the Baltic Proper. The longer residence time of ^{137}Cs in the Baltic Proper is most likely due to inflows of more contaminated water from the northern part of the Baltic Sea. In the time period following Chernobyl, 1986-1988, the effective half-lives of ^{137}Cs were much shorter in most contaminated regions: 0.8 years in the Gulf of Finland and 2.5 years in the Bothnian Sea. The shorter effective half-life of ^{137}Cs in Gulf of Finland as compared to the Bothnian Sea during 1986-1988 was probably due to different water exchange and sedimentation processes in these two regions (Ilus et al. 1993).

Based on the inventory estimates, the effective half-life of ^{137}Cs in Baltic seawater during the period 1993-2006 has been 9.6 years. With this decay rate, the ^{137}Cs inventory in the Baltic Sea would reach pre-Chernobyl levels (250 TBq) by the year 2020, presuming that the effective half-life will stay constant, and no substantial remobilization of ^{137}Cs from sediments will occur.

Levels of radionuclides in marine biota are linked to the corresponding levels in seawater and sediments, via accumulation through food chains. The complexity of food chains increases with the trophic level of the species considered. Fish, the biota type in the Baltic Sea most important for human consumption, accumulate most of their radionuclides from food, not from water.

The biota of the Baltic Sea received the most significant contribution to their radionuclide levels following the Chernobyl accident in 1986, predominantly in the form of ^{137}Cs and 134Cs. As shown in **Figure 3.5**, concentrations of ^{137}Cs are continuing to show generally slowly decreasing trends in herring muscle. In the western parts of the Baltic Sea, i.e. the Kattegat, the Sound, the Belt Sea and the Arkona Sea, the values already show levels slightly below the GES boundary of 2.5 Bq kg^{-1} wet weight. In the remaining Baltic Sea basins, the GES boundary is still exceeded, in the Bothnian Bay and in the Gotland area, by a factor of up to 5.

Figure 3.6 shows the ^{137}Cs time series for the flat fish group, consisting of flounder (*Platichthys flesus*), plaice (*Pleuronectes platessa*) and dab (*Limanda limanda*), in the western and southern Baltic Sea areas. Samples of fillets/flesh were used for these measurements. At the end of the assessment period, the values were below about 8 Bq kg^{-1} wet weight.

The ratio 134Cs/^{137}Cs in Baltic biota agree very well with that of the Chernobyl fallout. High trophic level species, including predators such as cod and pike, have shown the highest ^{137}Cs levels, but there was some delay in reaching their maximum values after 1986, when compared to trends in seawater. In the long-term, ^{137}Cs time trends in biota closely follow the trends in seawater.

Marine biota concentration factors (CF) show clearly that for marine fish species the ^{137}Cs CF values increase from western Baltic Sea areas to eastern/northern areas, which is explained by the corresponding increase of freshwater contributions to the seawater (HELCOM 2009).

Doses

The total collective radiation dose from ^{137}Cs in the Baltic Sea is estimated at 2600 manSv of which about two thirds (1700 manSv) originate from Chernobyl fallout, about one quarter (650 manSv) from fallout from nuclear weapons testing, about 8% (200 manSv) from European reprocessing facilities, and about 0.04% (1 manSv) from nuclear installations bordering the Baltic Sea area.

Dose rates and doses from natural radioactivity dominate except for the year 1986 where the individual dose rates from Chernobyl fallout in some regions of the Baltic Sea approached those from natural radioactivity.

The maximum annual dose since 1950 to individuals from any critical group in the Baltic Sea area due to ^{137}Cs is estimated at 0.2 mSv y^{-1}, which is below the dose limit of 1 mSv y^{-1} for the exposure of members the public set out in the EU Basic Safety Standards, 1996. It is unlikely that any individual has been exposed from marine pathways at a level above this dose limit considering the uncertainties involved in the assessment. Doses to man due to liquid discharges from nuclear power plants in the Baltic Sea area are estimated at or below the levels mentioned in the Basic Safety Standards to be of no regulatory concern (individual dose rate of 10 μSv y^{-1} and collective dose of 1 manSv). It should be noted that the assumptions made throughout the assessment were chosen to be realistic and not conservative. Consequently, this also applies to the estimated radiation doses to man.

References

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Relevance of the indicator for describing the developments in the environment

The occurrence of man-made radioactive substances in the Baltic Sea has four main causes:

1. During 1950-1980 the United States and the Soviet Union carried out atmospheric nuclear weapons tests, which peaked in the 1960s, causing radioactive fallout throughout the northern hemisphere. This pollution is still noticeable in the seas and on land (UNSCEAR 2000).
2. The accident at the Chernobyl nuclear power plant in 1986 caused heavy pollution in the vicinity of the power plant, and also considerable fallout over the Baltic Sea.
3. The two European facilities for reprocessing of spent nuclear fuel, at Sellafield in the UK and La Hague in France, have both discharged radioactive substances into the sea. Some of this radioactivity has been transported by sea currents to the North Sea, from where a small proportion has entered the Baltic Sea.
4. Authorised discharges of radioactivity into the sea occurring during the routine operation of nuclear installations in the Baltic Sea region (nuclear power plants and nuclear research reactors) have also contributed.

Radioactive substances enter the marine environment either as direct fallout from the atmosphere or indirectly as runoff from rivers. Radionuclides may also be discharged directly into the ocean as liquid waste or from dumped solid wastes. Some radionuclides will behave conservatively and stay in the water in soluble form, whereas others will be insoluble or adhere to particles and thus, sooner or later, be transferred to marine sediments and marine biota.

Conceptual model of the sources, transport and impacts of caesium-137 in the Baltic ecosystem

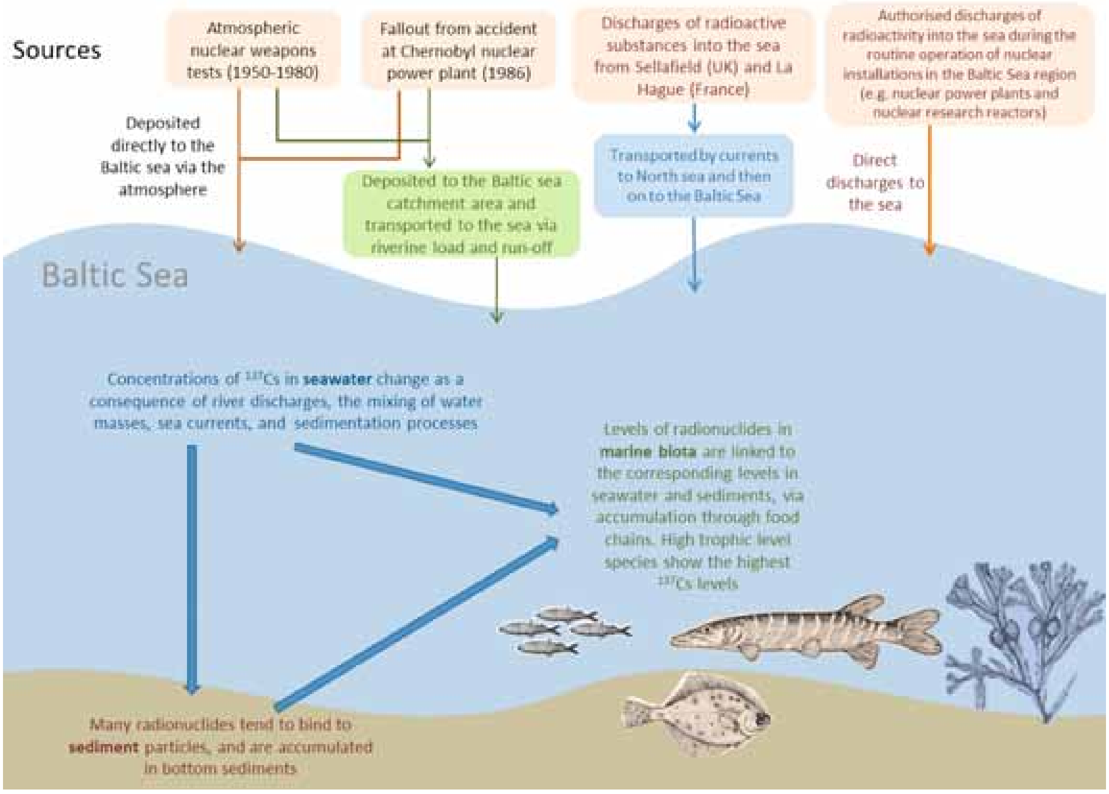

Figure 3.8. *Conceptual model illustrating the sources and pathways of ^{137}Cs in the Baltic Sea.*

Levels of radionuclides in marine biota are linked to the corresponding levels in seawater and sediments, via accumulation through food chains. The complexity of food chains increases with the trophic level of the species considered. Fish, the biota type in the Baltic Sea most important for human consumption, accumulate most of their radionuclides from food, not from water.

References

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Technical data on assessing concentrations of artificial radionuclide ^{137}Cs in biota

Data source

Data have been collected by the Contracting Parties of HELCOM and submitted to the MORS (Monitoring of Radioactive Substances) database.

Description of data

The data are based on ^{137}Cs concentrations of a) herring (*Clupea harengus L.*), b) flounder (*Platichthys flesus L.*) and plaice (*Pleuronectes platessa L.*) and c) surface seawater (samples 0-10 m). Analyses have been made either from round fish (without head and entrails) or filets (herring), and for plaice and flounder from filets, only. Concentrations (Bq/kg) have been calculated from wet weight of the samples.

Seawater concentrations (Bq/m^3) have been analyzed from surface water samples 0-10 m.

Data of each media (herring, plaice and flounder and sea water) have been averaged by basin and by year.

Spatial and temporal coverage

Herring data covers all the areas except Gulf of Riga and the area of Gotland East and West, as there are data only of 2003 and 2003-2009, respectively. There are no data on the Northern Baltic Proper in recent years (2003-2009). In the Sound, Belt and Arkona Sea several years of data are missing (1992-1994, 1999-2000, 2004-2009).

Plaice and flounder data are very scarce both temporarily and spatially covering only four sea areas and several years missing. Sampling on plaice and flounder takes place only in some of the countries.

Sea water data coverage is almost complete, except the missing years in the Gulf of Riga and in the Archipelago Sea.

Methodology and frequency of data collection

The average number of biota samples collected annually by the MORS-PRO group through the sampling period 1999-2009 was about 110. Over the whole period the numbers of samples collected were 871 for fish, 170 for *aquatic plants*, and 131 for benthic animals.

More detailed information on national monitoring activities are available for Finland, Germany, Lithuania **and** Poland.

Methodology of data analyses

More than ten laboratories from the nine countries bordering Baltic Sea have contributed to the monitoring programmes of Baltic Sea by analyzing radionuclides from marine samples. The various analytical methods used in the different laboratories are summarized the HELCOM thematic assessment: Radioactivity in the Baltic Sea, 1999-2006 (HELCOM 2009).

Strengths and weaknesses of data

Quality assurance is a fundamental part of radioanalytical work, needed to confirm the precision and the long-term repeatability of analyses. The radiochemical procedures and counting techniques used by laboratories are well tested, up-to-date, and similar to those used by laboratories worldwide.

Eight intercomparisons were organised during the HELCOM MORS-PRO project period (1999-2006) for seawater and sediment samples, and their results are also presented in the HELCOM thematic assessment (HELCOM 2009). The intercomparisons confirm that the data produced by the MORS group is of very good quality and can be considered comparable. Less than five percent of the results were considered outliers

GES boundary values and classification method

The GES boundaries for ^{137}Cs concentrations in sea water, sediments and biota have been set at pre-Chernobyl levels.

Average concentrations of ^{137}Cs prior the Chernobyl accident have been used as GES boundary values. These are for herring (2.5 Bq/kg), flounder and plaice (2.9 Bq/kg) and seawater (15 Bq/m^3).

Further work required

The reported uncertainties in data vary considerably between laboratories. Each laboratory calculates uncertainties in its own particular way, and the harmonization of uncertainty calculations would improve the comparability of the data.

References

HELCOM (2009): Radioactivity in the Baltic Sea, 1999-2006 HELCOM thematic assessment. Balt. Sea Environ. Proc. No. 117: 60 pp.

3.8 Tributyltin (TBT) and the imposex index

Authors: Rita Poikane, Jakob Strand and Martin M. Larsen
Acknowledged persons:
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General information

General properties

Tributyltin compounds (TBT-ion CAS No. 688-73-3 (36643-28-4)) belong to organometallic compound (OTC) class. Usually triorgancompounds used as biocide – antifoulant paints, agricultural pesticides, molluscicides and wood preservative. TBT compounds are hydrophobic and associate strongly to particles in natural waters and ultimately are deposited in the sediments. Adsorption to organic rich particles (soils and sediments) is stronger than to particles (soils and sediments) of mineral origin. Degradation (photodegradation or biodegradation) of TBTs in environment occurs due to dealkylation and depends on aerobic condition. Degradation of TBT under anaerobic condition may last long time. Half-life of TBTs in natural waters may range from a few days to several weeks but in soils and sediments one to few years. TBTs accumulate in individual organisms – often stronger in benthic organisms than in fish. TBT and triphenyltin accumulate in the food web, but a large variance in accumulation potential has been found between species, even within the same trophic level. This is probably due to different abilities to degradate TBT or triphenyltins between the species.

Main impacts on the environment and human health

TBT compounds are very toxic to aquatic organisms especially to benthic organisms. As a consequence of toxic effects is shell deformation, endocrine disruption and impaired larval recruitment as well as immunosuppression. TBTs cause endocrine disruption and different types of malformation of the genital system for certain marine and freshwater bivalve and gastropods species at very low concentrations. The process is known as „imposex" and „intersex". For human health high levels of TBT are potential risk to cause endocrine disruption. Several OTCs have negative toxic effects: immumosuppresive, neurotoxicity, hepatoxicity, renal and dermal toxicity, teratogenic and carcinogenic effects.

The ecological relevance of imposex and intersex development in marine snails is high because of the links to reproductive disorders. In severe stages, reproductive failure in female snails occurs as they are getting sterile. For instance, sterile female of the red whelk Neptunea antiqua has been found in the Inner Danish waters (Strand 2009).

Effects of sterile females on population structures have been shown in other studies of TBT contaminated areas.

The HELCOM thematic assessment of hazardous substances (HELCOM 2010) showed that the thresholds for fish (15 µg kg^{-1} ww), mussels (30 µg kg^{-1} dw) and sediment (2 µg kg^{-1} dw) were exceeded all over the Baltic Sea.

Status of a compound on international priority lists and other policy relevance

The recent environmental issues surrounding tributyltin have been increasing environmental pressures on all butyltin compounds and other OTCs in general.

TBT is a substance, which is identified on the priority list of the HELCOM Baltic Sea Action Plan.

Tributyltin compounds (TBT-ion) are classified as Priority Hazardous substances under the Daughter Directive (2008/105/EC) of the EC Water Framework Directive (2000/60/EC). The EU Marine Strategy Framework Directive (2008/56/EC) refers to the priority substances of the WFD. Progress towards good environmental status will depend on whether pollution is progressively being phased out, i.e. the presence of contaminants in the marine environment and their biological effects are kept within acceptable limits, so as to ensure that there are no significant impacts on or risk to the marine environment.

OSPAR supports implementation of EU legislation and measurements of organic tin compounds are included as a part of The Coordinated Environmental Monitoring Programme (CEMP) on a mandatory basis (OSPAR 2010). Imposex, caused by TBT is also part of OSPAR CEMP and therefore are to be measured on a mandatory basis in the North Atlantic region (OSPAR 2010).

Status of restrictions, bans or use

In accordance with point 20 of Annex XVII to Regulation (EC) No 1907/2006 of the European Parliament and of the Council of 18 December 2006 concerning the Registration, Evaluation, Authorization and Restriction of Chemicals (REACH), establishing a European Chemicals Agency, amending Directive 1999/45/EC and repealing Council Regulation (EEC) No 793/93 and Commission Regulation (EC) No 1488/94 as well as Council Directive 76/769/EEC and Commission Directives 91/155/EEC, 93/67/EEC, 93/105/EC and 2000/21/EC pursuant to Commission Regulation (EC) No 552/2009 of 22 June 2009 amending Regulation (EC) No 1907/2006 of the European Parliament and of the Council on the Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) as regards Annex XVII.

Organostannic compounds shall not be placed on the market, or used, as substances or in mixtures:

1. where the substance or mixture is acting as biocide in free association paint.
2. where the substance or mixture acts as biocide to prevent the fouling by micro-organisms, plants or animals of:
   a) all craft irrespective of their length intended for use in marine, coastal, estuarine and inland waterways and lakes;
   b) cages, floats, nets and any other appliances or equipment used for fish or shellfish farming;
   c) any totally or partly submerged appliance or equipment.
3. where the substance or mixture is intended for use in the treatment of industrial waters.

On the global level, the use of TBT in antifouling paints has been banned by the 2001 International Convention on the Control of Harmful Anti-fouling Systems on Ships (AFS convention), which entered fully into force in 2008. From 1 January 2008, ships bearing an active TBT coating on their hulls will no longer be allowed in Community ports (782/2003/EC). All Baltic Sea countries except Russian Federation have ratified the Convention.

GES boundaries and matrix

Existing quantitative targets e.g. EACs, BACs and food safety standards

Table 3.14. Existing quantitative targets for TBT.	
Source	**Value and description**
OSPAR	
EAC sediment	0.01 ug TBT/kg DW
EAC water	0.1 ng TBT/L
EAC mussel	12 ug TBT/kg DW
BAC/BC sediment	Zero for man-made compound
BAC/BC water	Zero for man-made compound
BAC/BC mussel	Zero for man-made compound
LC/BC mussel	1 ug TBT/kg DW
EC	
EQS water	0.2 ng TBT/L (AA) 1.5 ng TBT/L (MAC)
Draft EQS water	
Draft EQS biota	
Unofficial EQS sediment	0.02 ug TBT/kg DW
Unofficial EQS biota	230 ug TBT/kg WW (predators second poisoning)
Food stuff directive	15.2 ug TBT/kg WW for seafood
Effect Range -Low	
ERL sediment	
Other	
PNEC	AA 0.2 ng/L; MAC 1.5 ng/L TBT **/** 1.0 ng/L TPhT
WHO (uptake by food)	0.25 ug per kilo of human weight per day

Recommended GES boundaries	Water: 0.2 ng/L Sediment: 0.02 ug/kg Biota: 12 ug/kg DW (for mussels) Seafood: 15.2 ug/kg WW for seafood

Preferred matrix

Water – Optional. (low solubility in water, high level of attachment to particles)

Sediments – Optional. (Could be a co-factor in decreasing biodiversity in areas with low biodiversity)

Biota (mussels or gastropoda snails): Preferred. (mussels are sensitive to TBT). Fish (liver) can be used as alternative if molluscs are not present.

Biological effects assessment classes for ECOQO on imposex/intersex

Table 3.15. Imposex assessment classes for N. lapillus and other selected gastropods					
Assessment class	***N. lapillus***	***L. littorea***	***N. reticulatus***	***B. undatum***	***N. antiqua***
Criterion	**VDSI**	**ISI**	**VDSI**	**PCI**	**VDSI**
A Level of imposex is close to zero	<0.3	<0.3	<0.3	<0.3	<0.3
B Level of imposex (~30-~100% of the females have imposex) indicates exposure to TBT concentrations below the EAC derived for TBT	0.3 - <2.0	<0.3	<0.3	<0.3	0.3 - <2.0
C Level of imposex indicates exposure to TBT concentrations higher than the EAC derived for TBT	2.0 - <4.0	<0.3 - <0.7	0.3 <4.0	0.3 - 4.0	2.0 - 4.0
D Reproductive capacity in the gastropod populations is affected as a result of the presence of sterile females, but some reproductively capable females remain	4.0 – 5.0	0.7 - <2.0	May occur beyond 4.0	May occur beyond 4.0	May occur beyond 4.0
E Populations are unable to reproduce. The majority, if not all females within the population have been sterilised	5.0 – 6.0	>2.0			
F Populations are absent/expired	-				

Assessment class	*Nucella*	*Nassarius*	*Buccinum*	*Neptunea*	*Littorina*	*TBT Water*	*TBT mussel*	*TBT sediment*
	VDSI	VDSI	PCI	VDSI	ISI	(ng TBT/l)	(µg TBT /kg dw)	(µg TBT/ kg dw)
A	< 0.3	< 0.3[1]	< 0.3[1]	< 0.3	< 0.3[2]	<0.025	< 3	n.d.
B	0.3 - <2.0			0.3 - <2.0		0.025-0.25	3-30	< 2
C	2.0 - < 4.0	0.3 - <2.0	0.3 - <2.0	2.0 - 4.0[3]		0.25-5	30 - <600	2 - <50
D	4.0 - 5.0	2.0 – 3.5	2.0 - 3.5		0.3 - < 0.5	5-7.5	600 - < 900	50-<200
E	>5.0[4]	> 3.5[4]	>3.5[4]		0.5 - 1.2	7.5-37.5	900 - 4200	200 -500
F					> 1.2	>37.5	>4200	>500

Figure 3.9. *Links of assessment classes between imposex and concentrations of TBT in water, mussels and sediment. Source: draft 2011 OSPAR background document for TBT.*

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Table 3.16. Monitoring of TBT concentrations and imposex in the Baltic Sea.

Country	Water	Sediments	Biota
Denmark		40 samples per year (rotating sampling scheme)	Standard mussel program – ~66 samples, shellfish directive – ~13 samples, flounder – ~12 stations per year.
Estonia			Screening studies (2008/2009) within HELCOM BSAP project in muscle of *Perca fluviatilis* and *Clupea harrengus*.
Finland		Since 2009 monitoring of organotin compounds is planned and station net is not decided yet. Sampling frequency >5yr.	Since 2009 yearly organotin compounds measurements in perches and herrings from 7 – 9 sampling areas.
Germany	Since 2010 6 station in coastal area 8 times per year.	Since 1996 10 stations: 5 coastal and 5 open area 2 times per year, since 1998 47 stations all in coastal area every 2-3 years.	Biota since 1998 2 coastal areas 2 times per year.
Latvia	Screening of 4 rivers estuaries (2009/2010): 3 largest rivers of the Gulf of Riga, 1 river of the Baltic Proper.	Screening of 4 rivers estuaries (2009/2010): 3 largest rivers of the Gulf of Riga, 1 river of the Baltic Proper.	Screening studies (2008/2009) within HELCOM BSAP project in muscle of *Perca fluviatilis* and *Clupea harrengus*.
Lithuania	Since 2010 national monitoring program: yearly sampling at 4 stations in the coastal area, 2 station in the harbour area and 2 stations in the Curonian Lagoon.	Planned in national monitoring program from 2011: yearly sampling at 4 stations in the coastal area, 2 station in the harbour area and 3 stations in the Curonian Lagoon. Since 2009 yearly monitoring of TBT in Klaipeda harbour area (several stations) performed by authorities of harbour.	Screening studies (2008/2009) within HELCOM BSAP project in muscle of *Perca fluviatilis* and *Clupea harrengus*.
Poland		Data from publications.	Data from publications.
Russia			
Sweden		Monitored yearly in coastal areas at 13 reference (no local discharges known) stations.	Yearly in *Hydrobia ulvae* at 12 stations (5 geographical regions). Coastal areas.

Gaps in the monitoring of the compound

Table 3.17. Gaps in the monitoring of the compound of TBT and imposex.	
Country	**Gaps**
Denmark	Little geographic coverage around Bornholm. Time trend station at Bornholm will be established from 2011 onwards. Sediments will be taken according to a rotating sampling scheme, so no temporal trends will be available for sediments.
Estonia	
Finland	Sediment profile sites not decided yet. TBT measurements are performed in fish not in mussels.
Germany	
Latvia	No data about TBT in sediments and in mussels. TBT is not included in monitoring program plan till 2012.
Lithuania	Monitoring program does not include TBT measurements in biota.
Poland	
Russia	
Sweden	

Present status assessments

Known temporal trends (also from sediment core profiles)

Since the EU ban on TBT, temporal trends showed that TBT concentration in sediments decrease as well as in benthic biota and fishes. In some regions TBT concentration remains high but decreasing trend can be expected in a coming years.

Spatial gradients (incl. sources)

Evaluation of spatial gradient in all the Baltic Sea is rather difficult due to the gaps in geographical coverage. Clear spatial gradients have been established in relation to areas with high ship densities, which are in line with that ship traffic is regarded as the main source of TBT in marine environments.

Quality Assurance of imposex

OSPAR has developed international monitoring guidelines for imposex and intersex in five species of marine snails (OSPAR 2008). An ICES guideline also exists for monitoring intersex in periwinkle (Oehlmann 2004). A detailed method description for imposex in the mud snail Hydrobia can be found in Schulte-Oehlmann et al. (1997)

Quality Assurance in form of international workshops and intercalibrations has been organized almost yearly by QUASIMEME since 1998. National workshops have also been organized three times in relation to the Danish monitoring program NOVANA.

Recommendation

TBT measurements in benthic biota should be included as core indicator for the next 8 – 10 years till next indicator revision. Measurements in sediment and water can support or supplement biota monitoring in areas with little or no target organisms.

Imposex and intersex should become core indicators, because they reflect the effects of TBT in sensitive organisms in the marine environment very well. They can be regarded as counterpart of TBT measurements e.g. in water, sediments and biota.

A scheme for assessing imposex/intersex and chemical measurements have been described by OSPAR and Strand et al (2006), and can be used to give an overall assessment of the status of TBT contamination.

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3.9. Pharmaceuticals: Diclofenac and 17-alpha-ethinylestradiol

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General information

General properties

Diclofenac is an active pharmaceutical ingredient belonging to a group called nonsteroidal anti-inflammatory drugs (NSAIDs). It works by reducing hormones that cause inflammation and pain in the body.

Diclofenac is sufficiently persistent to pass sewage water treatment plants and reach surface waters (Tixier, C. et al. *Environ. Sci. Technol.*, 2003, 37(6):1061–1068 **DOI:** 10.1021/es025834r). In many cases the reduction of diclofenac waste water treatment plants is close to zero, i.e. the same concentration of diclofenac is found in the treated effluent water as in incoming waters.

In the aquatic environment, pharmaceuticals have been widely found and Diclofenac is a drug that is detected at high frequency. Zang et al. *Chemosphere* 2008 73(8):1151-1161

Diclofenac is bioaccumulating in fish exposed to treated sewage waters (Brown et al. *Environmental Toxicology and Pharmacology* 24(3)2007:267-274)

17α-ethinylestradiol (EE2) is an active pharmaceutical ingredient used as a component of combined contraceptives. EE2 is an endocrine disrupter of great concern, with fish feminization induced for concentrations around 1 ng per litre or less.

Main impacts on the environment and human health

Diclofenac is toxic to the kidneys in fish. NOEC 1 micro gram /liter water (e.g. Schwaiger et al. Aquatic Toxicology 2004, 68:141-150) Diclofenac in the marine environment is not likely to cause *acute* toxic effects at environmental concentrations. However chronic effects need cautious consideration.

Kidney failure caused by residues of the analgesic and anti-inflammatory drug diclofenac, is considered responsible for a decline by >95% in the population of oriental white-backed vulture, one of the (previously) most common raptors in India and Pakistan (Oaks et al, 2004, Shultz et al, 2004; Reddy et al, 2006; Swan et al, 2006; Cuthbert et al, 2006, 2007).

Extensive evidence points to a causal link between exposure to **ethinylestradiol** and feminization of fish in the environment. These mainly include the following observations:

Ethinylestradiol at sub ng/L levels cause both vitellogenin induction (Purdom et al, 1994; Thorpe et al, 2003; Jobling et al, 2003) and intersex/sex-change in fish (Örn et al, 2003; Parrot and Blunt, 2005).

1. Ethinylestradiol up to a few ng/L is found in effluents from waste water treatment plants and water recipients.
2. Fish exposed to waste water treatment plant effluents can bioconcentrate estrogens, including ethinylestradiol, very efficiently as demonstrated by extremely high levels of conjugated metabolites in their bile (Larsson et al. 1999).

3. Frogs have been shown to approximately as sensitive as fish to EE2 exposures; 1.7 ng/L resulted in skewed sex ratios of adult frogs and malformations of their gonadal duct system (Pettersson and Berg. Environmental Toxicology and Chemistry 2007. 26 (5) 1005-1009

Status of a compound on international priority lists and other policy relevance

Both are on the draft revision list for EU Priority Substances

Status of restrictions, bans or use

The use of Diclofenac as a human drug is not restricted. India is phasing out diclofenac because of its impacts on the vulture population. EE2 is not restricted.

GES boundaries and matrix

Existing quantitative targets

Table 3.18. Existing quantitative targets for diclofenac and EE2.

Source	**Diclofenac**	**EE2**
EC		
Draft QS water	0.1 µg L^{-1}	0.0035 ng L^{-1}
Draft QS biota	1 µg kg^{-1} ww	0.067 µg kg^{-1} ww
Draft QS sediment		
NOEC	(open literature) = 1 µg/L	LOEC = approximately 1 ng/L
Recommended GES boundaries	Water: 0.1 µg L^{-1} Sediment: Biota: 1 µg kg^{-1} ww Seafood:	Water: 0.0035 ng L^{-1} Sediment: Biota: 0.067 µg kg^{-1} ww Seafood:

According to SCHER (2011), there is uncertainty in the background material of the EQS and therefore the GES boundary for diclofenac is only tentative.

The SCHER opinion on the EE2 supports the WFD WG E proposal (SCHER 2011), but the EQSs are still considered tentative.

Preferred matrix

Diclofenac can readily be analysed in water and in fish plasma.

EE2 can be analyzed in water and in fish plasma.

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Diclofenac and EE2 is not monitored anywhere in the Baltic Sea, but it is included at least in the Swedish EPA's screening program.

Gaps in the monitoring of the compound

There are gaps.

Recommendation

Diclofenac and EE2 are recommended as core indicators, because of their policy relevance and known adverse effects in environment.

3.10. Lysosomal membrane stability

Author: Katja Broeg, Doris Schiedek and Kari Lehtonen
ICES SGEH Biological Effects methods Background Documents for the Baltic Sea region (ICES/OSPAR document from the ICES SGIMC Report 2010, complemented and modified by SGEH 2011 with information relevant for application in the Baltic Sea region)

Description of the indicator

Lysosomal functional integrity is a generic common target for environmental stressors in all eukaryotic organisms from yeast and protozoans to humans (Cuervo 2004), that is evolutionarily highly conserved. The stability of lysosomal membranes is a good diagnostic biomarker of individual health status (Allen and Moore 2004; Broeg et al. 2005; Köhler et al. 1992, Lowe et al. 2006). Dysfunction of lysosomal processes has been mechanistically linked with many aspects of pathology associated with toxicity and degenerative diseases (Cuervo 2004; Köhler 2004; Köhler et al. 2002; Moore et al. 2006a, b, Broeg 2010). Lysosomes are known to accumulate many metals and organic xenobiotics. Metals such as copper, cadmium and mercury are known to induce lysosomal destabilisation in mussels (Viarengo et al. 1981, 1985a, b). LMS is strongly correlated with the concentration of PAHs and PCBs in mussel tissue (Cajaraville et al. 2000; Krishnakumar et al. 1994; Moore 1990; Moore et al. 2006a, b; Viarengo et al. 1992, Strand et al. 2009), as well as organochlorines and PCB congeners in the liver of fish (Köhler et al. 2002, Broeg et al. 1992). LMS of various species of mussel and fish from different climate zones clearly reflect gradients of complex mixtures of chemicals in water and sediments (Da Ros et al. 2002; Pisoni et al, 2004; Schiedek et al. 2006, Barsiene et al. 2006; Sturve et al. 2005), point sources of pollution, single pollution events and accidents (Garmendia et al. 2011; Einsporn et al. 2005; Broeg et al. 2002, Broeg et al. 2008, Nicholson and Lam, 2005) and also serves for the discovery of new "Hot Spots" of pollution (Bressling 2006; Moore et. al. 1998; 2004a).

LMS can also be used as a prognostic tool, able to predict liver damage and tumour progression in the liver of various fish species (Broeg et al. 1999; Diamant et al. 1999; Köhler et al. 2002; Köhler 2004, Broeg 2010). Also hepatopancreatic degeneration in molluscs, coelomocyte damage in earthworms, enhanced protein turnover as a result of radical attack on proteins, and energetic status an indicator of fitness of individuals within a population can be predicted (Allen & Moore 2004; Kirchin et al. 1992; Köhler et al. 2002; Moore et al. 2004a, 2006a; Nicholson & Lam 2005; Svendsen & Weeks 1995; Svendsen et al. 2004). Recently it is tested for its prognostic potential with respect to reproductive disorders in amphipods in the Baltic Sea. For eelpout, this prognostic potential could already been demonstrated. Low membrane stabilities coincided with distinct reproductive disorders that indicated adverse effects at the population level (Broeg and Lehtonen 2006).

Thus, LMS has been adopted by UNEP as part of the first tier of techniques for assessing harmful impact in the Mediterranean Pollution programme (MEDPOL Phase IV) and is also recommended as biomarker to be included into the OSPAR Coordinated Environmental Monitoring Programme (pre-CEMP). LMS of blue mussel from the Inner Danish waters and the Danish Belt Sea is part of the Danish monitoring programme NOVANA since 2003 (Strand et al. 2009). It is also under consideration for the Swedish monitoring programme (Granmo, pers. comm.). Methods applied to assess LMS are the Neutral Red Retention test (NRR) on living cells like mussel haemocytes, and the cytochemical test on serial cryostat sections performed from snap-frozen tissue. These methods are described in detail by Moore et al. (2004b). Currently a new method is developed for the assessment of LMS in single tissue sections of small indicator species like amphipods (Broeg and Schatz, in prep.).

Beside LMS, adverse lysosomal reactions to xenobiotic pollutants include swelling, lipidosis (pathological accumulation of lipid), and lipofuscinosis (pathological accumulation of age/stress pigment) in molluscs but not fish (Köhler et al. 2002; Moore 1988; Moore et al. 2006a, b; Viarengo et al. 1985a). LMS in blue mussels is correlated with oxygen and nitrogen radical scavenging capacity (TOSC), protein synthesis, scope for growth and larval viability and inversely correlated with DNA damage (incidence of micronuclei), lysosomal swelling, lipidosis and lipofuscinosis (Dailianis et al. 2003; Kalpaxis et al. 2004; Krishnakumar et al. 1994; Moore et al. 2004a, b, 2006a; Regoli 2000; Ringwood et al. 2004). In fish liver, LMS is strongly correlated with a suppression of the activity of macrophage aggregates, and lipidosis (Broeg et al. 2005).

A conceptual mechanistic model has been developed linking lysosomal damage and autophagic dysfunction with injury to cells, tissues and the whole animal; and the complementary use of cell-based bioenergetic computational model of molluscan hepatopancreatic cells that simulates lysosomal and cellular reactions to pollutants has also been demonstrated (Allen & McVeigh 2004; Lowe 1988; Moore et al. 2006a, b, c). Various biomarker indices and decision support systems have been developed based on LMS as "guiding" parameter to interpret the results of other biomarkers (**Figure 3.10**) which show "bell-shaped" responses since it reflects deleterious effects of various classes of contaminants in an integrative linear manner (Dagnino et al. 2007, Broeg et al. 2005, Broeg & Lehtonen 2006).

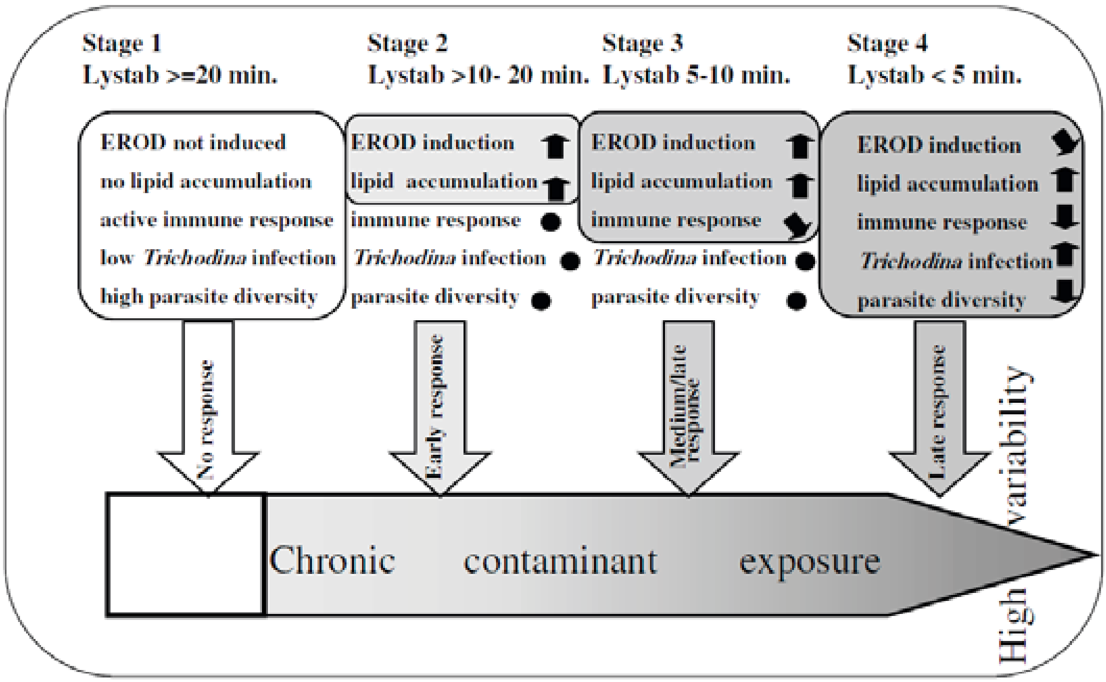

Figure 3.10. *Progression of biological effects detected on the basis of LMS in individual flounder of the German Bight (Broeg et al. 2005).*

Confounding factors

LMS is an integrative indicator of individual health status and will be affected also by non-contaminant factors such as severe nutritional deprivation, severe hyperthermia, prolonged hypoxia, and liver infections associated with high densities of macrophage aggregates (Moore et al. 1980; Moore et al. 2007, Broeg 2010). Processing for neutral red retention (NRR) in samples of molluscs adapted to low salinity environments should use either physiological saline adjusted to the equivalent ionic strength or else use ambient filtered seawater. The major confounding factor in respect of biomonitoring is the adverse effect of the final stage of gametogenesis and spawning in mussel, which is a naturally stressful process (Bayne et al. 1978). In general, this period should be avoided anyway for sampling purposes, as most physiological processes and related biomarkers are adversely affected (Moore et al. 2004b).

However, for fish, spawning has only a minimal effect on LMS and does not mask harmful chemical induced damage to LMS (Köhler 1990, 1991). Salinity changes didn't provoke significant effects on LMS in flounder (Broeg, unpublished results).

Using the cytochemical approach, temperature stress during the tissue incubation at 37°C has to be considered when working with animals from subpolar and polar regions. For these animals, temperature stress leads to a significant decrease of LMS. In this case, temperature during incubation should not be higher than 20°C above the ambient temperature of the sampling location to avoid effects of too severe hyperthermia.

Ecological relevance

Lysosomal integrity is directly correlated with physiological scope for growth (SFG) and is also mechanistically linked in terms of the processes of protein turnover (Allen and Moore 2004; Moore et al. 2006a). Ringwood et al. (2004) have also shown that LMS in parent oysters is directly correlated with larval viability. It is also inversely correlated with reproductive disorders in eelpout (Broeg & Lehtonen 2006). Finally, LMS is directly correlated with diversity of macrobenthic organisms in an investigation in Langesund Fjord in Norway (Moore et al. 2006b), and with parasite species diversity in flounder from the German Bight (Broeg et al. 1999).

Quality Assurance

Intercalibration exercises for LMS techniques have been carried out in the ICES/UNESCO-IOC-GEEP Bremerhaven Research Workshop, the UNEP-MEDPOL programme, in the framework of the EU-project BEEP and the BONUS+ project BEAST as well as for the neutral red retention method in the GEF Black Sea Environmental Programme (Köhler et al. 1992; Lowe et al. 1992; Moore et al. 1998; Viarengo et al. 2000, BEEP 2004). The results from these operations indicated that both techniques could be used in the participating laboratories in an effective manner with insignificant inter-laboratory variability.

Comparisons of the cytochemical and the neutral red retention techniques have been performed in fish liver (ICES-IOC Bremerhaven Workshop, 1990) and in mussels experimentally exposed to PAHs (Lowe et al. 1995). An AWI/Imare international workshop on "Histochemistry of lysosomal disorders as biomarkers in environmental monitoring" in Bremerhaven, 2008, demonstrated good correspondence of results obtained by the participants by applying various different assessments by computer assisted image analysis and light microscopy. In 2010, an ICES\OSPAR Workshop on Lysosomal Stability Data Quality and Interpretation (WKLYS) has been held in Alessandria, Italy (ICES 2010). This workshop concentrated on the NRR.

Guidelines for LMS procedures are published as ICES Times Series (Moore et al. 2004b), and in the UNEP/Ramoge biomarker manual (UNEP 1999).

Assessment Criteria

Health status thresholds for NRR and cytochemical methods for LMS have been determined from data based on numerous studies (Cajaraville et al. 2000; Moore et al. 2006a, Broeg et al. 2005, Broeg & Lehtonen 2006).

LMS is a biophysical property of the bounding membrane of lysosomes and appears to be largely independent of taxa. In all organisms tested to date, which includes protozoans, annelids (terrestrial and marine), molluscs (freshwater and marine), crustaceans (terrestrial and aquatic), echinoderms and fish, the absolute values for measurement of LMS (NRR and cytochemical method) are directly comparable. Furthermore, measurements of this biomarker in animals from climatically and physically diverse terrestrial and aquatic ecosystems also indicate that it is potentially a universal indicator of health status. For the cytochemical method animals are considered to be healthy if the LMS is ≥20 minutes; stressed but compensating if <20 but ≥10 minutes and severely stressed and probably exhibiting pathology if <10 minutes (Moore et al. 2006a, Broeg et al. 2005, Broeg & Lehtonen 2006). Similarly for the NRR method, animals are considered to be healthy if NRR is ≥120 minutes; stressed but compensating if <120 but ≥50 minutes and severely stressed and probably exhibiting pathology if <50 minutes (Moore et al. 2006a).

The use of different fish species as indicators at identical locations in the Baltic Sea showed species differences with respect to their liver LMS in the following order:
Herring < Eelpout < Dab < Flounder.

At locations which are higher affected by anthropogenic impact, differences are pronounced. Potential causes are higher fishing stress and high frequencies and intensities of parasite infections in almost all livers of herring and eelpout as confounding factors. Thus, for these species the assessment criteria for the cytochemical test are defined as follows: animals are considered to have no toxically-induced stress if the LMS is ≥15 minutes; are stressed but compensating if <15 but ≥8 minutes and are severely stressed and probably exhibiting irreversible toxicopathic alterations if <8 minutes (Broeg et al., in prep.) (see **Table 3.19**).

The following species have been tested as indicator species for LMS in the different regions of the Baltic Sea:

Fish	Herring (*Clupea harengus*), flounder (*Platichthys flesus*), eelpout (*Zoarces viviparus*), dab (*Limanda limanda*).
Bivalves	Blue mussel (*Mytilus edulis, Mytilus trossulus*)
Amphipods	Gammarids, *Monoporeia affinis*

Table 3.19. Quantitative targets for Lysosomal membrane stability.

Biological effect method (unit, other information)	**Target species/ tissue/endpoint**	**BAC**	**EAC**
Lysosomal membrane stability (LMS) (*minutes*)			
a. Cytochemical method	Herring and eelpout liver	15	8
	Perch and flounder liver	20	10
	All other species studied liver, digestive gland	20	10
Method modification: Acridine Orange	Amphipods	under development in BEAST	under development in BEAST
b. In vivo method (Neutral Red Retention test)	*Mytilus* spp. haemocytes	120	50

Acknowledgements

Research on the application of LMS as a biomarker for the monitoring and assessment of general toxic effects in different Baltic Sea regions and indicator organisms has been funded by the EU Project BEEP Contract n° ECK3-CT2000-00025, and the BONUS+ Project BEAST.

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3.11. Fish Disease Index:

Externally visible fish diseases, macroscopic liver neoplasms and liver histopathology

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ICES SGEH Biological Effects methods Background Documents for the Baltic Sea region (ICES/OSPAR document from the ICES SGIMC Report 2011, complemented and modified by SGEH 2011 with information relevant for application in the Baltic Sea region)

Description of the indicator

Diseases of wild marine fish have been studied on a regular basis by many ICES Member Countries for more than two decades. Disease surveys are often integrated with other types of biological and chemical investigations as part of national monitoring programmes aiming at an assessment of the health of the marine environment, in particular in relation to the impact of human activities (Lang 2002).

On an international level, fish disease data have been used for environmental assessments in the framework of the North Sea Task Force and its Quality Status Report (North Sea Task Force 1993), the OSPAR Quality Status Reports 2000 and 2010 (OSPAR Commission 2000, 2010a) and in the 3rd and 4th HELCOM assessments (HELCOM 1996, 2002). Studies on externally visible diseases, macroscopic liver neoplasms and liver histopathology are on the list of techniques for general and contaminant-specific biological effects monitoring as part of the OSPAR pre-CEMP (OSPAR 2010b).

In the Baltic Sea, fish diseases have been monitored on a more or less regular basis since the beginning of the 1980s (Lang 2002). Baltic Sea countries currently carrying out fish disease surveys in the Baltic Sea on an annual basis are Germany (vTI Institute of Fisheries Ecology, Cuxhaven), Poland (Sea Fisheries Institute, Gdynia) and Russia (AtlantNIRO, Kaliningrad) (ICES 2011). While Polish and Russian studies are restricted to national EEZs, the German programme covers larger areas of the southern Baltic Sea, including sampling sites in ICES Subdivision 22, 24, 25 and 26. Other Baltic Sea countries not mentioned have some experience in fish disease monitoring from studies carried out in the 1980s and 1990s, but have stopped regular activities.

Most of the regular disease surveys are so far focussed on fish species sampled in offshore areas, with the main target species flounder (*Platichthys flesus*), cod (*Gadus morhua*) and, to a lesser extent, herring (*Clupea harengus*). In the western Baltic Sea, the common dab (*Limanda limanda*) is another target species. Other common species have been examined on a more irregular basis. A wide and species-dependent range of diseases (incl. some parasite species) is being monitored, with an emphasis on externally visible lesions and parasites. Only in flounder have regular studies on liver pathology (largely related to neoplastic lesions) been included partly (Lang et al. 2006). The methodologies applied largely follow ICES guidelines (Bucke et al. 1996, Feist et al. 2004) which can easily be adapted for other species relevant for fish monitoring in the Baltic Sea. Methodologies and diagnostic criteria involved in the monitoring of contaminant-specific liver neoplasms and liver histopathology have largely been developed based on studies with flatfish species, in Europe mainly dab and flounder, but can also be adapted to other flatfish species (e.g. plaice (*Pleuronectes platessa*) and also to bottom-dwelling roundfish species, such as viviparous blenny (*Zoarces viviparus*).

New disease trends in Baltic Sea fish species have been reviewed regularly by the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and relevant information has partly been incorporated in the HELCOM Periodic Assessments (HELCOM 1996, 2002). However, compared to the North Sea, fish disease monitoring and assessment in the Baltic Sea is less developed and, so far, only few fish disease data from the Baltic Sea have been submitted to the ICES Environmental Databank. In the light of present developments in ICES and in HELCOM, the ICES WGPDMO recommended at its 2007 meeting that

Baltic Sea countries running fish disease monitoring programmes in the Baltic Sea make attempts to submit their disease data to the ICES Environmental Databank in order to make them available for integrated assessments, such as those carried out by the ICES/HELCOM Working Group on Integrated Assessment of the Baltic Sea (WGIAB) and as part of the periodic HELCOM assessments (ICES 2007).

In 2005, the ICES Workshop on Fish Disease Monitoring in the Baltic Sea (WKFDM) started to develop an integrative tool for the analysis and assessment of the health status of fish which was later termed 'Fish Disease Index (FDI)' (ICES 2006a,b). In contrast to previous attempts, largely focusing on the analysis and assessment of changes in prevalence of single diseases, the FDI approach was developed with the primary aim to analyse and assess changes in spatial and temporal patterns in the overall disease status of fish, by summarising information on the prevalence of a variety of common diseases affecting the fish species as well as their severity grades and effects on the host into a robust numerical value calculated for individual fish and, as mean values, for representative samples from a population. The common dab (*Limanda limanda*) from the North Sea was selected as a model species for the construction of the FDI approach because most existing data are from fish disease surveys with the dab as primary target species. However, the FDI approach is constructed in a way that it can easily be adapted to other fish species for which disease data are available. The development of an analogous FDI approach for Baltic Sea fish species is on the agenda of the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and will be finalised during 2011 (ICES 2011). In first instance, the efforts will focus on flounder and cod, species for which most data are available from national fish disease monitoring in the Baltic Sea.

Fish disease surveys and associated FDI data analyses and assessments according to the OSPAR and ICES requirements address four categories of diseases. These categories also are the basis for fish disease monitoring/assessment in the Baltic Sea (see **Table 3.20**). Other target species and diseases may be added when more experience and data are available.

Table 3.20. Categories and diseases/lesions for disease monitoring and assessment with Baltic Sea fish species (ICES 2011; modified)

Disease Category	**Diseases/Lesions**	
	Flounder (*P. flesus*)	**Cod (*G. morhua*)**
Externally visible diseases	Lymphocystis Acute/healing skin ulcerations Acute/healing fin rot/erosion Epidermal hyperplasia/ papilloma *Cryptocotyle sp.* *Lepeophtheirus pectoralis*	Acute/healing skin ulcerations Acute/healing fin rot/erosion Skeletal deformities Pseudobranchial swelling Epidermal hyperplasia/papilloma *Cryptocotyle lingua* *Lernaeocera branchialis*
Macroscopic liver neoplasms	Benign and malignant liver tumours > 2 mm in diameter	Benign and malignant liver tumours > 2 mm in diameter
Non-specific liver histopathology	Non-specific degenerative/regenerative change Inflammatory lesions Parasites	Non-specific degenerative/regenerative change Inflammatory lesions Parasites
Contaminant-specific liver histopathology	Early toxicopathic non-neoplastic lesions Foci of cellular alteration Benign neoplasms Malignant neoplasms	Early toxicopathic non-neoplastic lesions Foci of cellular alteration Benign neoplasms Malignant neoplasms

Confounding factors

The multifactorial aetiology of diseases, in this context in particular of externally visible diseases, is generally accepted. Therefore, externally visible diseases have correctly been placed into the general biological effect component of the OSPAR CEMP (OSPAR 2010b). Most wild fish diseases monitored in past decades are caused by pathogens (viruses, bacteria). However, other endogenous or exogenous factors may be required before the disease develops. One of these factors can be environmental pollution, which may either affect the immune system of the fish in a way that increases its susceptibility to disease, or may alter the number and virulence of pathogens. In addition, contaminants may also cause specific and/or non-specific changes at various levels of biological organisation (molecule, sub-cellular units, cells, tissues, organs) leading to disease without involving pathogens.

The occurrence of significant changes in the prevalence of externally visible fish diseases can be considered a non-specific and more general indicator of chronic rather than acute (environmental) stress, and it has been speculated that they might, therefore, be an integrative indicator of the complex changes typically occurring under field conditions rather than a specific marker of effects of single factors. Because of the multifactorial causes of externally visible diseases, the identification of single factors responsible for observed changes in disease prevalence is difficult, and scientific proof of a link between contaminants and externally visible fish diseases is hard to achieve. Nevertheless, there is a consensus that fish disease surveys should continue to be part of national and international environmental monitoring programmes since they can provide valuable information on changes in ecosystem health and may act as an "alarm bell", potentially initiating further more specific studies on cause and effect relationships.

A thorough statistical analysis of ICES data on externally visible diseases (lymphocystis, epidermal hyperplasia/papilloma, acute/healing skin ulceration) of dab from different North Sea regions, confirmed the multifactorial aetiology of the diseases under study since a number of natural and anthropogenic factors (stock composition, water temperature, salinity, nutrients, contaminants in water, sediments and biota) were found to be significantly related to the long-term temporal changes in disease prevalence recorded. (Lang and Wosniok 2000; Wosniok et al. 2000).

The presence of macroscopic liver neoplasms and of certain types of histopathological liver lesions is a more direct indicator of contaminant effect and has been used for many years in environmental monitoring programmes around the world. Liver neoplasms (either detected macroscopically or by histopathological analysis) are likely to be associated to exposure to carcinogenic contaminants, including PAHs, and are therefore considered appropriate indicators for contaminant-specific biological effects monitoring. The study of liver histopathology (comprises the detection of more lesion categories (non-specific, neoplastic and non-neoplastic toxicopathic lesions), reflecting responses to a wider range of contaminants (including PAHs) but also to other environmental stressors and is, therefore, considered an appropriate indicator for both general and contaminant-specific biological effects monitoring.

The liver is the main organ involved in the detoxification of xenobiotics and several categories of hepatocellular pathology are now regarded as reliable biomarkers of toxic injury and representative of biological endpoints of contaminant exposure (Myers et al. 1987, 1992, 1998; Stein et al. 1990; Vethaak & Wester 1996; Stentiford et al. 2003; Feist et al. 2004). The majority of lesions observed in field collected animals have also been induced experimentally in a variety of fish species exposed to carcinogenic compounds, PAHs in particular, providing strong supporting evidence that wild fish exhibiting these lesions could have been exposed to such environmental contaminants.

Ecological relevance

Fish diseases are considered as ecosystem health indicators, reflecting ecologically relevant effects of environmental stressors at the individual and population levels. As such, they differ from other types of indicators that reflect changes at lower levels of biological organisation (e. g. molecules, cells) and the ecological relevance of which is considered as low or unclear (e. g. biomarkers of exposure to contaminants.) (ICES 2009b)

Fish diseases may act at the individual level by adversely affecting behaviour, growth, reproduction, and survival of affected specimens. Individual effects may lead to ecologically relevant population effects (especially in epidemic situations) and ultimately to biodiversity effects at the community level. Diseases in wild fish may affect aquaculture due to transmission of pathogens. A high prevalence of a conspicuous fish disease may affect fishery profit because fish with prominent disease signs cannot be marketed. Although direct human health effects of diseases affecting wild fish are unlikely (except for a few cases), diseased fish may act as carriers of pathogens that pose a risk to human consumers.

Quality Assurance

Since the early 1980s, ICES has played a leading role in the initiation and coordination of fish disease surveys and has contributed considerably to the development of standardised methodologies. Through the work of the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), its offspring, the Sub-Group/Study Group on Statistical Analysis of Fish Disease Data in Marine Stocks (SGFDDS) (1992–1994) and the ICES Secretariat, quality assurance procedures have been implemented at all stages, from sampling of fish to submission of data to the ICES Data Centre and to data assessment.

A number of practical ICES sea-going workshops on board research vessels were organised by WGPDMO in 1984 (southern North Sea), 1988 (Kattegat), 1994 (Baltic Sea, co-sponsored by the Baltic Marine Biologists, BMB) and 2005 (Baltic Sea) in order to intercalibrate and standardise methodologies for fish disease surveys (Dethlefsen et al. 1986; ICES 1989, 2006a; Lang & Mellergaard 1999) and to prepare guidelines. Whilst first guidelines were focused on externally visible diseases and parasites, WGPDMO developed guidelines for macroscopic and microscopic inspection of flatfish livers for the occurrence of neoplastic lesions at a later stage. Further intercalibration and standardision of methodologies used for studies on liver pathology of flatfish were a major issue of the 1996 ICES Special Meeting on the Use of Liver Pathology of Flatfish for Monitoring Biological Effects of Contaminants (ICES 1997). This formed the basis from which the quality assurance programme Biological Effects Quality Assurance in Monitoring (BEQUALM) (www.bequalm.org) developed for the application of liver pathology in biological effects monitoring (see below) (Feist et al. 2004).

A fish disease database has been established within the ICES Data Centre, consisting of disease prevalence data of key fish species and accompanying information, submitted by ICES Member Countries. Submission of fish disease data to the ICES Data Centre has been formalised by the introduction of the ICES Environmental Reporting Format designed specifically for the purpose. This is used for fish disease, contaminant and biological effects data. The programme includes internal screening procedures for the validation of the data submitted providing further quality assurance.

The ICES fish disease database is extended on an annual basis to include data from other species and areas within the OSPAR and HELCOM area as well as data on studies into other types of diseases, e.g. macroscopic liver neoplasms and liver histopathology. To date, the data comprise mainly information from studies on the occurrence of externally visible diseases and macroscopic liver lesions in the common dab (*Limanda limanda*) and the European flounder (*Platichthys flesus*) from the North Sea and adjacent areas, including the Baltic Sea, Irish Sea, and the English Channel. In addition, reference data are available from pristine areas, such as waters around Iceland. In total, data on length, sex, and health status of more than 700 000 individual specimens, some from as early as 1981, have been submitted to ICES, as well as information on sampling characteristics (Wosniok et al. 1999, Lang and Wosniok 2008).

Current ICES WGPDMO activities have focussed on the development and application of statistical techniques for an assessment of disease data with regard to the presence of spatial and temporal trends in the North Sea and western Baltic Sea (Wosniok et al. 1999, Lang and Wosniok 2008). In a more holistic approach, pilot analyses have been carried out combining the disease data with oceanographic, nutrient, contaminant and fishery data extracted from the ICES Data Centre in order to improve the knowledge about the complex cause-effect relationships between environmental factors and fish diseases (Lang and Wosniok 2000; Wosniok et al. 2000). These analyses constituted one of the first attempts to combine and analyses ICES data from various sources and can, therefore, be considered as a step towards a more comprehensive integrated assessment.

Quality assurance is in place for externally visible diseases, macroscopic liver neoplasms and liver histopathology via the ongoing BEQUALM programme. Regular intercalibration and ring-test exercises are conducted. The basis for QA procedures are provided in two key publications in the ICES TIMES series (Bucke et al. 1996, Feist et al. 2004) and a BEQUALM CD ROM of protocols and diagnostic criteria and reporting requirements for submission of data to ICES. Guidelines on fish disease monitoring in the Baltic Sea have been prepared by ICES (2006a).

Assessment Criteria

The development of assessment tools for externally visible diseases, macroscopic neoplasms and liver histopathology has been addressed by the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) (ICES 2006b, 2007, 2008, 2009a, 2011). Further additions were proposed at the 2009 ICES/OSPAR Workshop on Assessment Criteria for Biological Effects Measurements (WKIMC) (ICES 2009b) (see further below).

For the analysis and assessment of fish disease data, the ICES WGPDMO developed a Fish Disease Index (FDI), using data on diseases of the common dab (*Limanda limanda*) as a model. The aim of this tool is to summarise information on the disease status of individual fish into one robust and easy-to-understand and easy-to-communicate numeric figure. By applying defined assessment criteria and appropriate statistics, the FDI can be used to assess the level and temporal changes in the health status of fish populations and can, thus, serve as a tool for the assessment of the ecosystem health of the marine environment, e.g. related to the effects of anthropogenic and natural stressors. Its design principle allows the FDI to be applied to other species with other sets of diseases. Therefore, the FDI approach is applicable for wider geographical areas, e.g. as part of a convention-wide HELCOM monitoring and assessment programme.

For the calculation of the FDI, the following components are required:
– information on the presence or absence of a range of diseases monitored on a regular basis, categorised as externally visible diseases, macroscopic liver neoplasms as well as non-specific and contaminant-specific liver histopathology (see **Table 3.11**);
– for most diseases, data on three severity grades (reflecting a light, medium or severe disease status) are included;
– disease-specific weighting factors, reflecting the impact of the diseases on the host (assigned based on expert judgements);
– adjustment factors for effects of size and sex of the fish as well as for season effects.

The result of the calculation is a FDI value for individual fish which is scaled in a way that values can range from 0 to 100, with low values representing healthy and high values representing diseased fish. The maximum value of 100 can only be reached in the (purely theoretical and unrealistic) case that a fish is affected by all diseases at their highest severity grades. From the individual FDIs, mean FDIs for a sample from a fish population in a given sampling area can be calculated. Usually a sample in the present sense consists of the data collected in an ICES statistical rectangle during one cruise. All assessment is based on mean FDI values calculated from these samples. Depending on the data available, FDIs can be calculated either for single disease categories or for combinations thereof.

The assessment of the mean FDI data considers (a) long-term FDI level changes, (b) FDI trends in the recent five years time window and (c) comparing each FDI to its Background Assessment Criterion (BAC) and Environmental Assessment Criterion (EAC) where these are defined. While assessments (a) and (b) are done on a region-wise basis, global BAC and EAC are used by assessment (c). The assessment approaches (a) and (b) do not apply any global background or reference values or assessment criteria as is often done for chemical contaminants or for biochemical biomarkers. Instead, these assessment approaches use the development of the mean FDI within the geographical units (usually ICES rectangles) over a given period of time, based on which region-specific assessment criteria are defined. The reason for choosing this approach is the known natural regional variability of the disease prevalence (even in areas considered to be pristine), making it implausible to define generally applicable background/reference values that can uniformly be used for all geographical units to be assessed. This approach is based on the availability of disease data over a longer period of time (ideally 10 observations, e.g. in the case of biannual monitoring over a period of five years) for every geographical area to be assessed. The assessment approach (c) ignores the known regional differences and involves globally defined Assessment Criteria (BAC, EAC; see above) with the consequence that within-region variation might be dominated by general differences in regional levels. However, by applying globally defined Assessment Criteria, the FDI can also be used for exploratory monitoring in areas not studied before or for newly installed fish disease monitoring programmes after some modification.

The final products of the assessment procedure are:
- graphs showing the temporal changes in mean FDI values in a geographical unit over the entire observation period; and
- maps in which the geographical units assessed are marked with green, yellow or red smiley faces, indicating long-term changes (e.g. comparing the past five years to the preceding five-years period) in health status of the fish population (green: improvement of the health status; yellow: indifferent variation; red: worsening of the health status, reason for concern and motivation for further research on causes),
- maps in which the geographical units assessed are marked with green, yellow or red smiley faces, indicating trends in health status of the fish population during the past five years (green: improvement of the health status; yellow: indifferent variation; red: worsening of the health status, reason for concern and motivation for further research on causes),
- maps in which the geographical units assessed are marked with green, yellow or red smiley faces, indicating the level of the FDI observed at a defined point in time (green: below the BAC; yellow: between BAC and EAC; red: above the EAC, reason for concern and motivation for further research on causes).

The ICES WGPDMO applied the FDI approach and the assessment for the common dab from the North Sea using ICES fish disease data extracted from the ICES Environmental Data Centre twice in 2008 and, using an extended dataset, in 2009 (ICES 2008, 2009a). The results have been included in the OSPAR QSR 2010 as a case study (OSPAR 2010).

At the 2009 ICES/OSPAR Workshop on Assessment Criteria for Biological Effects Measurements (WKIMC) and the 2011 meeting of the ICES WGPDMO, Background Assessment Criteria (BAC) and Environmental Assessment Criteria (EAC) to be used for externally visible diseases, non-specific liver histopathology, macroscopic liver neoplasms and contaminant-specific liver histopathology in North Sea dab were proposed (ICES 2009b, 2011). A common strategy was developed for externally visible fish diseases (EVD) and non-specific liver histopathology (NLH), and a modified strategy was developed for macroscopic liver neoplasm (MLN) and contaminant-specific liver histopathology (SLH). Two strategies are needed because the first two categories require an external harm entity that is to be controlled by the EAC, while the last two categories themselves already constitute measures of harm. The approach leading to a BAC for EVD and NLH is guided by the following considerations:
- No "pristine" reference area is available from which a BC (background concentration) or a BAC could be obtained and transferred to the ICES area.
- A certain number of diseases in a population seems inevitable as the vast majority of disease rates from fish disease monitoring samples is larger than zero, i.e. has FDI > 0. This suggests using a lower bound for the mean FDI as BAC (each mean FDI is calculated from data from one cruise, one ICES rectangle).
- Using the smallest historical positive FDI value produces an unstable BAC estimate.
- Preferably a small percentile of the FDI distribution should serve as BAC. The FDI value below which only a defined small proportion (e.g. 10%) of all values lies would be used as BAC.
- A BAC should be derived in this way separately for each species and sex (and the disease category).
- The BACs obtained are considered valid for the whole area from which the basic data originated.

An EAC is the threshold beyond which "unacceptable effects" must be expected. The "effect" considered for EVD and NLH is the loss in condition factor (CF) that is associated with increasing FDI. Loss in CF is defined as the difference between the mean CF for FDI = 0 and the mean CF for an FDI > 0, expressed as percentage of the mean CF for FDI = 0. The EAC is then defined as that FDI value above which the loss in CF exceeds the acceptable amount (e.g. 10%). The essential point in this approach is that a link was established between a biomarker (fish diseases) and a relevant effect, in this case the loss in condition. Therefore an EAC could be based on loss in condition. With BAC and EAC available, the FDI results can be represented in the usual three-colour scheme, also on a map (see above).

Deriving BAC and EAC for macroscopic liver neoplasm (MLN) and contaminant-specific liver histopathology (SLH) follows slightly different lines. As macroscopic liver neoplasms are themselves unacceptable effects, there is no need to employ a further effect for determining an EAC. Also, there is no point in defining a BAC, as each effect in the MLN and SLH category is unacceptable.

The suitability of these BACs/EACs for fish disease monitoring and assessment in the Baltic Sea will be evaluated in the course of 2011 (prior to the 2012 meeting of ICES WGPDMO) and modifications will be done as required.

Acknowledgements

Research on the application of the **Fish Disease Index (FDI)** as indicators for the monitoring and assessment of environmental stressors (incl. contaminants) in the Baltic Sea has partly been funded by the EU Project BEEP Contract n° ECK3-CT2000-00025, and the BONUS+ Project BEAST.

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3.12. Micronucleus test

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ICES SGEH Biological Effects methods Background Documents for the Baltic Sea region (ICES/OSPAR document from the ICES SGIMC Report 2010, complemented and modified by SGEH 2011 with information relevant for application in the Baltic Sea region)

Description of the indicator

In environmental genotoxicity indication system, the micronucleus (MN) test has served as an index of cytogenetic damage for over 30 years. MN consists of acentric fragments of chromosomes or whole chromosomes which are not incorporated into daughter nuclei at anaphase. These small nuclei can be formed as a consequence of the lagging of a whole chromosome (aneugenic event) or acentric chromosome fragments (clastogenic event) (Heddle 1973; Schmid 1975). A MN arises in cell divisions due to spindle apparatus malfunction, the lack or damage of centromere or chromosomal aberrations (Fenech, 2000).

Clastogens induce MN by breaking the double helix of DNA, thereby forming acentric fragments that are unable to adhere to the spindle fibres and integrate in the daughter nuclei, and are thus left out during mitosis. Aneuploidogenic agents are chemicals that prevent the formation of the spindle apparatus during mitosis which can generate not only whole chromatids that are left out of the nuclei, thus forming MN, but also can form multinucleated cells in which each nucleus would contain a different number of chromosomes (Serrano-García & Montero-Montoya 2001). Furthermore, there are indications that MN additionally may be formed via a nuclear budding mechanism in the interphase. The formation of such type MN reflects in an unequal capacity of the organisms to expel damaged, amplified DNA, failed replicated or improperly condensed DNA, chromosome fragments without telomeres and centromeres from the nucleus (Lindberg et al. 2007).

The MN test involves the scoring of the cells which contain one or more MN in the cytoplasm (Schmid 1975). The assay was first developed as a routine in vivo mutagenicity assay for detecting chromosomal mutations in mammalian studies (Boller and Schmid 1970; Heddle 1973). Hooftman and de Raat (1982) were the first, who successfully apply the assay to aquatic species. Since these initial experiments, other studies have validated the detection of MN as a suitable biomarker of genotoxicity in a wide range of both vertebrate and invertebrate species (for review see Chaudhary et al. 2006; Udroiu et al. 2006; Bolognesi and Hayashi 2011). In fish most studies have applied circulating erythrocytes (blood) cells but can also be sampled from a number of tissues, such as liver, kidney, gill or fin epithelium (Archipchuk & Garanko 2005; Baršienė et al. 2006a; Rybakovas et al. 2009). The frequency of the observed MN may be considered as a suitable index of accumulated genetic damage during the cell lifespan providing a time integrated response of an organism's exposure to contaminant mixtures. Depending on the life span of each cell type and on their mitotic rate in a particular tissue, the frequency of MN may provide early warning signs of cumulative stress (Bolognesi and Hayashi 2011).

As an early warning indicator MN induction was successfully used in studies of environmental genotoxicity in gas (Gorbi et al. 2008) and oil platform zones (Hyland et al. 2008; Rybakovas et al. 2009, Brooks et al. 2011), also after the oil spills (Baršienė et al. 2006b, 2006c; Santos et al. 2010). Environmental genotoxicity levels in organisms from Baltic Sea, North Sea, Mediterranean and Northern Atlantic have been described in indigenous fish and mussel species inhabiting reference and contaminated sites (Wrisberg et al. 1992; Bresler et al. 1999; Baršienė et al. 2004, 2005a, 2006b, 2006c 2008a, 2008b, 2010a; Bagni et al. 2005; Bolognesi et al. 2006a; Magni et al. 2006). The MN test was validated in laboratory with different species after exposure to a large number of various chemical agents (Fenech et al. 2003; Bolognesi et al. 2006b; Baršienė & Andreikėnaitė 2007; Andreikėnaitė 2010; Bolognesi & Hayashi 2011).

The majority of studies to date have used haemolymph and gill cells of molluscs and peripheral blood cells of fish for the MN analysis (Bolognesi & Hayashi 2011). There are other studies (albeit limited) available describing the use of other haemopoetic tissues, such as liver, kidney, gills, and also fins (Archipchuk & Garanko 2005; Baršienė et al. 2006a; Rybakovas et al. 2009). The application of the MN assay to blood samples of fish is particularly attractive as the method is non-destructive, easy to undertake and results in an easy quantifiable number of cells present on the blood smears for microscopic analysis. However, studies must be undertaken to assess the suitability of any species or cell type analyzed.

The detected MN frequency in fish erythrocytes is approximately 6-10 times lower than in mussels and clams. The large inter individual variability associated to the low baseline frequency for this biomarker confirming the need for the scoring of a consistent number of cells in an adequate number of animals for each study point. Sampling size in most of studies conducted with mollusc species have been scoring 1000-2000 cells per animal (Izquierdo et al. 2003; Hagger et al. 2005; Bolognesi et al. 1996, 2004, 2006a; Magni et al. 2006; Baršienė et al. 2006a, 2006b, 2008b, 2010; Kopecka et al. 2006; Nigro et al. 2006; Schiedek et al. 2006; Francioni et al. 2007; Siu et al. 2008; Koukouzika & Dimitriadis 2005, 2008) and previous reviews have suggested that when using fish erythrocytes at least 2000-4000 cells should be scored per animal (Udroiu er al. 2006; Bolognesi et al. 2006a). Previously scorings of 5000-10000 fish erythrocytes where used for a MN analysis (Baršienė et al. 2004). Since 2009-2010, the frequency of MN in fish from the Baltic seas was mostly scored in 4000 cells. In stressful heavily polluted zones, the scoring of 5000-10000 cells in fish is still recommended.

Mussel sampling size in MN assays range from 5 to 20 mussels per site as reported in the literature (Baršienė et al. 2004, 2006c, 2008b; Francioni et al. 2007; Siu et al. 2008). Evidence suggests that a sample size of 10 specimens per site is enough for the assessment of environmental genotoxicity levels and evaluation of the existence of genetic risk zones. In heavily polluted sites, MN analysis in 15-20 specimens is recommended, due to higher individual variation of the MN frequency. MN analysis in more than 20 mussel or fish specimens shows only a minor change of the MN means (Fig. 1 in Fang et al. 2009; Baršienė et al., unpublished results).

Most of the studies have been performed using diagnostic criteria for MN identification developed by several authors (Heddle et al. 1973, 1991; Carrasco et al. 1990; Al-Sabti & Metcalfe 1995; Fenech 2000; Fenech et al. 2003):
- the size of MN is smaller than 1/3 of the main nucleus;
- MN are round- or ovoid-shaped, non-refractive chromatin bodies located in the cytoplasm of the cell and can therefore be distinguished from artifacts such as staining particles;
- MN are not connected to the main nuclei and the micronuclear boundary should be distinguishable from the nuclear boundary.

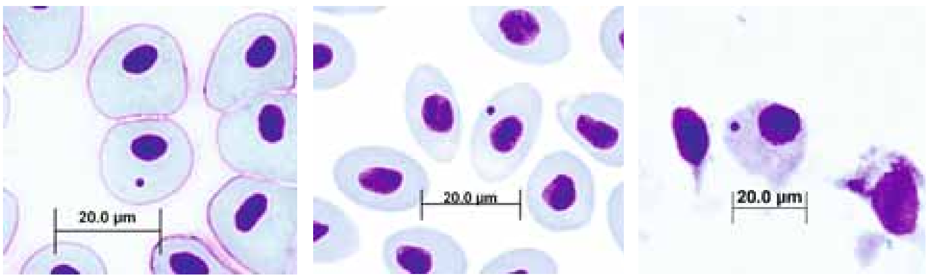

Figure 3.12. *Micronuclei in blood erythrocytes of (a) herring (Clupea harrengus),(b) flounder (Platichthys flesus), and (c) in a gill cell of the mussel Mytilus edulis.*

After sampling and cell smears preparation, slides should be coded. To minimize technical variation, the blind scoring of MN should be performed without knowledge of the origin of the samples. Only cells with intact cellular and nuclear membrane can be scored. Particles with color intensity higher than that of the main nuclei were not counted as MN. The area to be scored should first be examined under low magnification to select the part of the slide showing the highest quality (good staining, non overlapping cells). Scoring of MN should then be undertaken at 1000x magnification.

Confounding factors

Earlier studies on MN formation in mussels have disclosed a significant influence of environmental and physiological factors (Dixon et al. 2002). Therefore, the role of the confounding factors should be considered prior to the application of MN assay in biomonitoring programs, as well as in description of genetic risk zones, or ecosystem health assessments.

Water temperature**.** MN induction is a cell cycle-related process and depends on water temperature, which is a confounding factor for the mitotic activity in poikilotherm animals. Several studies have demonstrated that baseline frequencies of MN in mussels are related to water temperature (Brunetti et al. 1988, 1992; Kopecka et al. 2006). Baseline frequencies of MN are regarded as the incidence of MN observed in the absence of environmental risk or before exposure to genotoxins (Fenech 1993). In fish MN frequencies showed also seasonal differences in relation to water temperature with lower MN levels in winter than in autumn (Rybakovas et al. 2009). This was assumed to be an effect of higher mitotic activity and MN formation due to high water temperatures in the autumn (Brunetti et al. 1988). Additionally, it has been reported that increases in water temperature (4 - 37°C) can increase the ability of genotoxic compounds to damage DNA (Buschini et al. 2003).

Cell type. MN may be seen in any type of cell, both somatic and germinal and thus the micronucleus test can be carried out in any active tissue. Nevertheless there are some limitations using different types of cells, for example, agranular and granular haemocytes in mussels. There are also differences between MN induction level in mussel haemolymph and gill cells, mainly because gills are primary targets for the action of contaminants. The anatomical architecture of the spleen in fish does not allow erythrocytes removal in the spleen (Udroiu et al. 2006), though, in mammals this process go.

Salinity. The influence of salinity on the formation of MN was observed in mussels from the Danish coast located in the transitional zone between the Baltic and North Sea. No relationship between salinity and MN frequencies in mussels could be found for mussels from the Wismar Bay and Lithuanian coast. Similar results were found for *Macoma balthica* from the Baltic Sea in the Gulfs of Bothnia, Finland, Riga and in the Lithuanian EEZ (Baršienė *et al.*, unpublished data).

Individual size. Since the linear regression analysis of animal's length and induction of MN shows that the size could be a confounding factor, sampling of organisms with similar sizes should take place (Baršienė et al., unpublished data). It should also be noted that size is not always indicative of age and therefore age could also potentially affect the response of genotoxicity in the fish.

Diet. Results have shown that MN formation was not influenced in mussels who were maintained under simple laboratory conditions without feeding (Baršienė et al. 2006d).

Ecological relevance

Markers of genotoxic effects reflect damage to genetic material of organisms and thus get a lot of attention (Moore et al. 2004). Different methods have been developed for the detection of both double- and single-strand breaks of DNA, DNA-adducts, MN formation and chromosome aberrations. The assessment of chemical induced genetic damage has been widely utilized to predict the genotoxic, mutagenic and carcinogenic potency of a range of substances, however these investigations have mainly been restricted to humans or mammals (Siu et al. 2004). MN formation indicates chromosomal breaks, known to result in teratogenesis (effects on offspring) in mammals. There is however limited knowledge of relationships between MN formation and effects on offspring in aquatic organisms. With a growing concern over the presence of genotoxins in the aquatic media, the application of cytogenetic assays on ecologically relevant species offers the chance to perform early tests on health in relation to exposure to contaminants.

Quality Assurance

The MN test showed to be a useful in vivo assay for genotoxicity testing. However, many aspects of its protocol need to be refined, knowledge of confounding factors should be improved and inter-species differences need further investigation. In 2009 an inter-laboratory comparison exercise was organised within the framework of the MED POL programme using the mussel *M. galloprovincialis* as test species. The results are expected by mid 2011.

Intercalibration of MN analysis in fish was done between experts from NRC and Caspian Akvamiljo laboratories, as well as between NRC experts and the University of Aveiro, Portugal (Santos et al. 2010). It is recommended that these relatively simple interlaboratory collaborations are expanded to include material from all the commonly used indicator species in 2011/12.

Assessment Criteria

Baseline or background frequency of MN can be defined as incidence of MN observed in the absence of environmental risk or before exposure to genotoxins (Fenech, 1993). In fish, MN frequencies lower than 0.05‰ has been suggested by Rybakovas et al. (2009) as a reference level in the peripheral blood erythrocytes of the flatfish flounder (*Platichthys flesus*) and dab (*Limanda limanda*) and also cod (*Gadus morhua*) after analyzing 479 specimens from 12 offshore sites in the Baltic Sea. The frequencies of MN in marine species sampled from the Baltic Sea reference sites are summarized in **Table 3.21**.

Table 3.21. The reference levels of micronuclei (MN/1000 cells) in Baltic Sea species *in situ*.

Species	Tissue	Response MN/1000 cells	Reference
Mytilus edulis	Gills	0.37 ± 0.09	Baršienė et al. 2006b
Mytilus trossulus	Gills	2.07 ± 0.32	Baršienė et al. 2006b; Kopecka et al. 2006
Macoma baltica	Gills	0.53 - 1.28	Baršienė et al. 2008b, unpublished data (NRC, Lithuania)
Platichthys flesus	Blood erythrocytes	0.15 ± 0.03	Baršienė et al. 2004
Platichthys flesus	Blood erythrocytes	0.0 ± 0.0	Kohler, Ellesat, 2008
Platichthys flesus	Blood erythrocytes	0.08 ± 0.02	Napierska et al. 2009
Zoarces viviparus	Blood erythrocytes	0.02 ± 0.02	Baršienė et al. unpublished data (NRC, Lithuania)
Gadus morhua	Blood, kidney erythrocytes	0.03 ± 0.02	Rybakovas et al. 2009
Clupea harengus	Blood erythrocytes	0.03 ± 0.03	Baršienė et al. unpublished data (NRC, Lithuania)
Scophthalmus maximus	Blood erythrocytes	0.10 ± 0.04	Baršienė et al. unpublished data (NRC, Lithuania)
Perca fluviatilis	Blood erythrocytes	0.06 ± 0.02	Baršienė et al. 2005a; Baršienė et al. unpublished data (NRC, Lithuania)

Assessment Criteria (AC) have been established by using data available from studies of molluscs and fish in the Baltic Sea (NRC database). The background/threshold level of MN incidences is calculated as the empirical 90% percentile (P90). Until more data becomes available, values should be interpreted from existing national data sets. Note: the values given here are provisional and require further validation when new data becomes available.

The 90% percentile (P90) separates the upper 10% of all values in the group from the lower 90%. The rationale for this decision was that elevated MN frequency would lie above the P90 percentile, whereas the majority of values below P90 belong to unexposed, weakly-medium exposed or non-responding adapted individuals. P90 values were calculated for those stations/areas which were considered being reference stations (*i.e.* no known local sources of contamination or those areas which were not considered unequivocally as reference sites but as those less influenced from human and industrial activity).

ACs in bivalves *Mytilus edulis, Mytilus trossulus, Macoma balthica* (data from MN analysis in 2370 specimens), in fish *Limanda limanda, Zoarces viviparus, Platichthys flesus, Gadus morhua* and *Clupea harengus* (data from MN analysis in 3239 specimens) from Baltic Sea have been calculated using NRC (Lithuania) databases (**Table 3.22**).

Table 3.22. *Assessment criteria of MN frequency levels in bivalve mollusc and fish. BR =Background response; ER = Elevated response; n = number of specimens analysed.*

Species	Size (cm)	T (°C)	Region	Tissue	BR	ER	*n*
Mytilus edulis	1.5-3	8-18	Baltic Sea	Gills	<2.50	>2.50	1810
Mytilus trossulus	2-3	3-15	Baltic Sea	Gills	<4.50	> 4.50	230
Macoma balthica	1-3	13-18	Baltic Sea	Gills	<2.90	> 2.90	330
Zoarces viviparus	15-32	7-17	Baltic Sea	Erythrocytes	<0.38	>0.38	824
Limanda limanda	18-25	8-17	Baltic Sea	Erythrocytes	<0.49	>0.49	117
Platichthys flesus	17-39	10-17	Baltic Sea coastal	Erythrocytes	<0.29	>0.29	828
Platichthys flesus	18-40	6-18	Baltic Sea offshore	Erythrocytes	<0.23	>0.23	970
Gadus morhua	20-48	13-15	Baltic Sea	Erythrocytes	<0.38	>0.38	50
Clupea harengus	16-29	6-18	Baltic Sea	Erythrocytes	<0.39	>0.39	450

Distribution of indicator species in Baltic Sea subregions

MN test has generally been applied to organisms where other biological-effects techniques and contaminant levels are well documented. That is the case for mussels and for certain demersal fish species (as European flounder, dab or Atlantic cod), which are routinely used in biomonitoring programs and assess contamination along western European marine. However, the MN assay may be adapted for alternative sentinel species using site-specific monitoring criteria.

When selecting an indicator fish species, consideration must be given to its karyotype as many teleosts are characterised by an elevated number of small chromosomes (Udroiu *et al*. 2006). Thus, in certain cases MN formed after exposure to clastogenic contaminants will be very small and hard to detect by light microscopy. This can be addressed to a certain extent by using fluorescent staining. After selecting target/suitable species, researchers should also ensure that other factors including age, sex, temperature and diet are similar between the sample groups. If conducting transplantation studies, consideration needs to be given to the cellular turnover rate of the tissue being examined to ensure sufficient cells have gone through cell division. For example, if using blood the regularities of erythropoiesis should be known prior to sampling.

Large-scale and long-term studies took place from 2001 to 2010 at the Nature Research Centre (NRC, Lithuania) on MN and other abnormal nuclear formations in different fish and bivalve species inhabiting various sites of the Baltic Sea. These studies revealed the relevance of environmental genotoxicity levels in ecosystem assessments. NRC established a large database on MN and other nuclear abnormalities in 8 fish species and in mussels and clams from the Baltic Sea. Fish and bivalve species were collected from 117 coastal and offshore sites. The following organisms have been tested as the target species for MN test in the different regions of the Baltic Sea:

Fish	flounder (*Platichthys flesus*),
	dab (*Limanda limanda*,
	herring (*Clupea harengus*),
	eelpout (*Zoarces viviparus*)
	plaice (*Pleuronectes platessa)*
	Atlantic cod (*Gadus morhua)*
	perch (*Perca fluviatilis*)
	turbot (*Scophthalmus maximus, Psetta maxima)*
Bivalves	blue mussels (*Mytilus edulis, Mytilus trossulus*
	Baltic clam (*Macoma baltica*)
Amphipods	Gammarids

Acknowledgements

Research on the application of the MN test as a biomarker for the monitoring and assessment of environmental genotoxicity in different Baltic Sea regions and indicator organisms has been funded by the EU Project BEEP, Contract EVK3-CT-2000-00025, BONUS+ Project BEAST and Lithuanian Science Council GENCITOX project MIP-10123.

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3.13 A. Reproductive disorders in fish and amphipods: Reproductive success in eelpout

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ICES SGEH Biological Effects methods Background Documents for the Baltic Sea region (ICES/OSPAR document from the ICES SGIMC Report 2010, complemented and modified by SGEH 2011 with information relevant for application in the Baltic Sea region)

Description of the indicator

Eelpout (*Zoarces viviparus*), also called viviparous blenny, is a benthic fish species that is widely used in ecotoxicological studies and as a bioindicator of local pollution due to its stationary behaviour in coastal marine environment.

Eelpout is a recommended fish indicator species for assessing the environmental conditions in the Baltic Sea and eelpout is included in the environmental monitoring programmes of several Baltic States, both in relation to biological effects, contaminants and integrated fish monitoring. For instance, Sweden and Germany have routinely measured contaminant concentrations in eelpout for >15 years, and additional samples are archived in environmental specimen banks allowing retrospective studies on chemical burdens. Similar long time series is also available for some other biomarker studies.

The eelpout is viviparous, i.e. there is an internal fertilization of the eggs and the female fish gives birth to fully developed larvae. The eelpout's mode of reproduction (vivipari) enables the study of "reproductive success" on an individual level, and larval developmental disorders can be directly associated with e.g. the health status of the female or body burdens of toxic substances during pregnancy.

Different types of abnormal larvae development can be distinguished and be characterised into different groups, i.e. early dead embryos, late dead larvae, growth retarded larvae and deformed larvae, which can be divided into different subgroups of severe gross malformations.

Reproductive success in eelpout as a biological effects method is regarded as a general, i.e. non-specific, integrative indicator for impaired fish reproduction, and a significant biological endpoint for assessing potential population-relevant effects. Because the reproductive success in fish is a generic "stress" indicator; causal agents may, however, only be identified through a combination of chemical analyses of fish tissue and also other biological effects measurements. Several types of hazardous substances such as organochlorines, pesticides, PAH, heavy metals and organometals are known to have the potential to affect embryo and larval development in fish, generally (Bodammer 1993). Several of these substances, which may induce developmental, morphological and/or skeletal anomalies, have also been identified as endocrine disrupting substances (Davis 1997).

Reproductive success in eelpout is included in the list of parameters for biological effect monitoring for supporting programme in the HELCOM COMBINE manual for marine monitoring in the coastal zone. Part D. Programme for monitoring of contaminants and their effects (HELCOM 2006).

The method is also part of OSPAR pre-CEMP and JAMP guideline for general biological effects monitoring (OSPAR 2010).

According to the monitoring guideline, the sample size should consist of examinations of 40 – 50 individuals of pregnant females of eelpout per station, which should be sampled in the period between October 15 and December 1.

There is in some countries restrictions on eelpout fishery that depends on national legislations, e.g. fishing dispensation for catching pregnant females in the autumn is needed in Denmark. National concerns for impact of fishery on local populations and ethical guidelines for humane handling and fish killing should also be considered.

In the Baltic Sea region, reproductive success in eelpout is or has been used for monitoring or pre-monitoring investigations at least by labs from Denmark, Sweden, Germany and Poland and as part of integrated fish monitoring, often in combination with contaminant, biomarker studies and/or population studies.

Studies in the Baltic Sea has shown that that spatial differences occur with elevated levels of adverse developmental effects of embryo and larvae in eelpout broods have been found in populations living in contaminated areas with effluents from cities and industry. In comparison, only low levels of such effects generally occur in populations living in areas regarded as reference sites (e.g. Vetemaa et al. 1997, Ådjers et al. 2001, Sjölin et al. 2003, Strand et al. 2004, Kalmarweb 2005, Gercken et al. 2006) as shown in **Figure 3.13**.

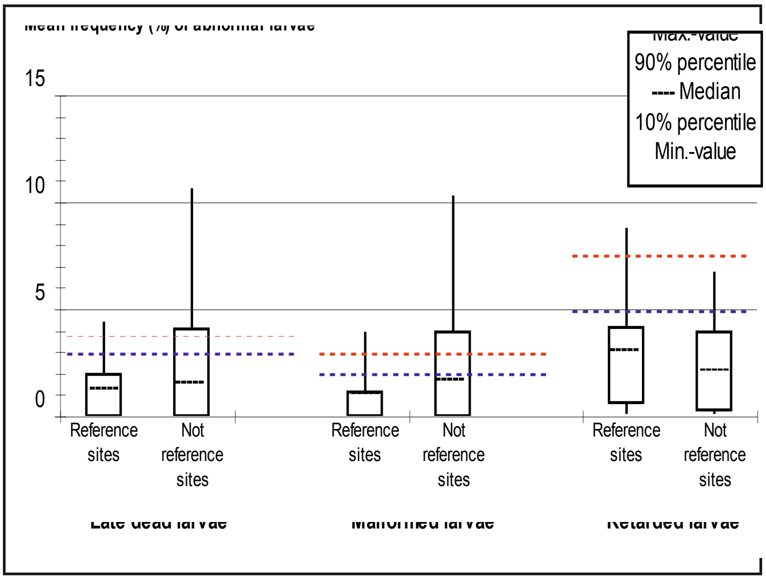

Figure 3.13. *Comparison of data distribution of compiled monitoring data on mean frequencies of late dead, malformed and growth retarded larvae in eelpout broods from reference sites and area not regarded as reference sites. The blue dotted line refers to the 90% percentile of data from the reference sites. The red dotted line refers to significantly elevated levels compared to the 90% percentile of the reference sites.*

Clear temporal developments with up- or down-going trends for the presence of different types of abnormal larvae development have not been established in monitored baseline areas with longer time series. However, some year-to-year variations can occur.

Studies of point sources have shown that acute larval mortality also been observed in eelpout exposed to pulp mill effluents (Jacobsson et al. 1986). Skewed sex ratios with significant more males in the eelpout broods have also been found nearby pulp mill effluents indicating effects of endocrine disruption substances (Larsson & Förlin 2002).

Confounding factors

Other environmental stressors like increased temperature and severe oxygen depletion events may however also affect eelpout reproduction (Veetema 1999, Fagerholm 2002, Strand et al. 2004). There have been some indications that some specific types of abnormal larvae development like early dead embryos and late dead larvae can be induced by severe oxygen events. However, deformed larvae with severe gross malformations, which can be distinguished from the other types, seem to be more related to contaminant effects (Strand et al. 2004, Gercken et al. 2006).

Ecological relevance

The ecological relevance of reproductive success of fish is high, because of the links to reproductive disorders. Population modelling support that elevated levels of abnormal larvae developments also can be important for population sizes.

Quality Assurance

The methodology for reproductive success is well defined for studies in coastal waters and national guideline exists (Jacobsson et al. 1986; Neuman et al. 1999, Strand & Dahllöf, 2005). An international guideline is in preparation and to be published in the ICES TIMES series.

As method quality assurance, some international and national workshops have been held in relation to the monitoring programmes (e.g. BEQUALM 2000). A Baltic workshop has been held in 2009 as part of BONUS+-projects BALCOFISH and BEAST. National workshops in relation to NOVANA monitoring activities have also been held in Denmark (Strand 2005a).

Backround response and Assessment Criteria

Assessment Criteria for reproductive success in eelpout based on below and above the background response has been proposed by ICES/OSPAR SGIMC 2010.

The derivation of assessment criteria have been based on data for either late dead larvae or deformed larvae from the Swedish and Danish monitoring programmes from several areas regarded as less polluted reference sites in the Baltic Sea, the Kattegat and the Skagerrak studies, where only low frequencies of abnormal larvae have mainly been found in areas, which were considered as reference sites, if any.

Background response values as baseline is based on 90% percentiles have been found to be <1% deformed larvae, <2% late dead larvae and <4% growth retarded larvae, respectively. Alternatively, the background response can also be based on the frequency of broods with >5% abnormal larvae development (**Table 3.23**).

Table 3.23. Background response for the presence of 3 types of abnormal larvae developments in eelpout, i.e. deformed larvae, late dead larvae and growth retarded larvae per station (ICES/OSPAR SGIMC 2010)

Type of abnormal larvae development	Background response, based on mean frequencies per station	Background response, based on frequency of broods with >5% abnormal larvae development
Deformed larvae	< 1% of all larvae	<5% of broods
Late dead larvae	<2% of all larvae	<5% of broods
Growth retarded larvae	<4% of all larvae	-
	Background response determined as the upper limit is the 90% percentile of response at so-called reference sites.	

Environment Assessment Criteria reflecting the GES boundary are under development. However, these assessment criteria will be revised and evaluated in 2011-2012 within the BONUS+ project BALCOFISH, so that the derivation of the background response levels are performed according to the principles recommended by the OSPAR/ICES working group SGIMC.

Distribution of indicator species in Baltic Sea subregions

The eelpout inhabits coastal waters and is widely distributed and common in almost all subregions of the Baltic Sea. Eelpout occurs also from the White Sea to the southern North Sea.

Acknowledgements

The BONUS+ project BALCOFISH shall be acknowledged for the progress on the linkage between contaminants, early warning biomarkers and reproductive disorders and also on the developments of assessment criteria and evaluation of the ecological relevance using population modelling.

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***Table 3.25**. Summary table for reproductive success in eelpout.*

Evaluation Criteria	**Rating**	**Description**
Recommended indicator species	-	Eelpout (*Zoarces viviparus*)
Recommended matrix	-	Pregnant females
Recommended sample size per station	-	40 - 50 fish
Monitoring guideline (SOP) in place	Yes	Swedish and Danish monitoring guidelines, ICES guideline in prep.
ACs in place, i.e. background response (BaR) and/or EAC	Yes	BaR <1% for deformed larvae, <2% for dead larvae. EAC under development.
QA in place, i.e. ongoing intercalibrations or workshops	Yes	BALCOFISH/BEAST workshop in 2009, National workshops
Ecological relevance of effects for populations	High	Links to impaired reproduction
Persistent damage, not repairable effects	Yes	Irreversible effects
Contaminant sensitive response (Elevated effect levels in open waters, coastal waters or only point sources)	High	Coastal waters, point sources
Contaminant-specific cause-effects response	No	See below
General effect, response to several contaminant groups	Yes	Can respond to several contaminant groups like metals, OCs, PAH, EDS
Stable for confounding factors	Medium	Depending of the type of abnormal larvae development. The presence of deformed seems most related to contaminants.
Applicable indicator species in 1, 2-3 or >3 Baltic Sea subregions	Yes	Eelpout occurs in all Baltic Sea subregions
Already used in monitoring in 1, 2-3 or >3 countries	Yes	Sweden, Denmark and Germany
Available data, spatial coverage in the Baltic Sea - number of countries and stations (<10, 10-20, >20 stations)	Medium	S: 6-10 stations, DK: 10-15 st. D: 3 st.
Available data, length of time series (<5, 5-10, >10 years)	Good	S: 1994-2010, DK: 2002-2010, D: 2003-2009
Costs of analyses per station, all required individuals (<500, 500-1500, >1500EUR)	High	~2000 EUR per station

3.13 B. Reproductive disorders in fish and amphipods: Reproductive success in amphipods

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ICES SGEH Biological Effects methods Background Documents for the Baltic Sea region (ICES/OSPAR document from the ICES SGIMC Report 2010, complemented and modified by SGEH 2011 with information relevant for application in the Baltic Sea region)

Description of the indicator

Crustacean amphipods are regularly used in bioassays and laboratory exposure experiments for effects of contaminants. They carry their brood in an egg chamber until hatching and by analyzing the reproduction success we can score the effects of contaminant load in sediment and water. Twenty years of ecotoxicological studies in soft-bottom microcosms and studies of field populations collected in contaminated industrial areas have demonstrated toxicant-sensitive variables on the embryonic development of the Baltic amphipod species *Monoporeia affinis* and *Pontoporeia femorata* (Sundelin 1983, 1984, 1988, 1989, 1998, Eriksson et al. 1996, Eriksson et al. 2005) and other amphipod species (Ford et al. 2003a, Sundelin et al. 2008, Bach et al. 2010).

When exposed to heavy metals, chlorinated organic compounds, pulp mill effluents or contaminated sediments in bioassays as well as in field studies, the frequency of malformed embryos has been demonstrated to be significantly higher when compared to control microcosms and reference areas (Elmgren et al. 1983, Sundelin 1983, 1984, 1988, 1989, 1991, Eriksson et al. 1996, Sundelin & Eriksson 1998, Eriksson et al. 2005) suggesting the variable to be a general bioindicator of contaminant effects. Organic contaminants are often associated to lipids. During oogenesis large quantities of lipids are deposited into the developing oocytes (Herring 1974, Harrison 1990, Harrison 1997, Wouters et al. 2001, Rosa & Nunes 2003). These lipids, which consist mostly of monounsaturated fatty acids, are utilized and consumed during embryo development (Morais et al. 2002, Rosa et al. 2003, 2005), potentially leading to toxic effects of lipophilic contaminants increasing during embryogenesis. These effects also arise in low concentrations that do not demonstrably affect the sexual maturation, fertilization rate, fecundity (eggs/female) and rate of embryo development (time to hatching), indicating embryogenesis to be even more sensitive than other variables of the reproduction cycle.

All amphipod species show a similar direct embryo development despite differences in sexual behaviour before mating and in duration of embryogenesis that differs, mainly due to ambient temperature (Bregazzi 1973, Lalitha et al. 1991, McCahon & Pascoe 1988). Therefore, this similar development allows for a consistent method of staging embryogenesis amongst all amphipod species and any resultant aberrations, which makes them particularly good for biomonitoring reproduction effects *in situ.* Other embryo aberrations respond to oxygen deficiency, scarcity of food quality and quantity and temperature stress (Eriksson et al. 2001, 2004, Sundelin et al. 2008). Multiple stressors act in concert in the environment and by analysing different types of aberrant embryo development we can discriminate between some of them. The method gives information about health status of the amphipod populations since diseases, parasite infection, sexual maturation in terms of oogenesis in females and sexual development in males and fecundity are scored.

Confounding factors

Malformed embryos seem to be comparatively insensitive to other environmental stressors but contaminant exposure. However a seven year field study showed a correlation between organic content in the sediment and malformation rate (Eriksson et al. 2004). This could likely depend on higher concentrations of contaminants in sediments with higher load of organic content. The same study didn't show any relationship between oxygen concentrations in bottom waters, temperature and malformation rate. A negative correlation was found between females carrying a dead brood and the oxygen concentration of the bottom water. Fecundity was positively correlated with the carbon content of the sediment but negatively correlated with the temperature of the bottom water. These results confirm the findings of previous laboratory experiments (Eriksson et al. 2001). Undeveloped eggs (undifferentiated eggs) are not correlated to contaminant exposure but seem to occur due to low food resources and possibly to increased temperatures but there are no clear correlations so far (Sundelin et al. 2008). To meet the possible confounding factors additional variables i.e. organic content and oxygen in sediment and bottom waters should be measured on sampling stations.

Ecological relevance

Crustaceans are one of the most abundant invertebrate groups and it is relevant to include them in monitoring activities. Amphipods are regarded as particularly sensitive to contaminant exposure (Conlan 1994). Furthermore amphipods lack pelagic larvae and thus they are comparatively stationary facilitating the linkage between effects and environmental conditions. The deposit-feeding amphipod *Monoporeia affinis*, and the marine species *Pontoporeia femorata* are important benthic key stone species in the Swedish fresh and brackish water environment and are efficient bioturbators and important for the oxygenation of the sediment. They are significant food source for several fish species and other macrofauna species (Arrhenius & Hansson 1993, Aneer 1975). By analyzing reproduction variables as malformed embryos and other aberrant embryo development we combine the supposedly higher sensitivity of low-organization level biomarkers with the higher relevance attributed to variables giving more direct information on next-generation and population level effects (Sundelin 1983, Tarkpea et al. 1999, Cold & Forbes 2004, Heuvel- Greve et al. 2007, Hutchinson 2007).

Quality Assurance

Guidelines are available in ICES Techniques in Marine Environmental Sciences (TIMES) no 41. Quality assurance declaration is updated regularly at website at Swedish EPA http://www.naturvardsverket.se/upload/02_tillstandet_i_miljon/Miljoovervakning/programomraden/kust_och_hav/kvalitetsdeklaration_embryonal_vitmarla.pdf. Quality assurance has been practiced during training courses and workshops when different persons analyzing the embryos checked the accordance by examining the same brood. The accordance was between 90 to 97%. Since *M. affinis* is a glacial relict of fresh water origin occurring in inland waters below the highest coastline and the method has been used and evaluated also in Swedish greater lakes as Lake Vänern and Lake Vättern (Sundelin et al 2008). For method description see Sundelin et al. 2008 (http://ices.dk/pubs/times/times41/TIMES41.pdf).

Assessment Criteria

The reproduction success of the freshwater amphipod *Monoporeia affinis* and the marine species *Pontoporeia femorata* in terms of various embryo aberrations have been measured in Baltic proper and Bothnian Sea since an international evaluation in 1993 priorotized the method as one of the most useful for effect monitoring of contaminants in the Baltic.

The method has also been used for other amphipod species in coastal waters outside Great Britain, Gulf of Riga, Gulf of Gdansk and in the Belt Sea. The method is used in the Bonus Beast programme as a core biomarker in all areas of the beast programme.

All species of amphipods could be analyzed for embryo aberrations and health status. The same protocol and method could be used for all of them (See TIMES 41). Field studies using different species of amphipods inhabiting the same area show a similar background level of malformation rate. However assessment criteria have only recently been developed for *Monoporeia affinis* where there exist a long-term trend series since 1994 in the Bothnian Sea and Baltic proper.

Table 3.14. Assessment criteria for malformed embryos of Monoporeia affinis in the Baltic. Seventeen years data were used for the calculation of background response (<5.7% malformed embryos) and assessment criteria. The limit between good and moderate status was put at a value where all (> 99%) variation in the reference dataset is included. The yearly mean and the the 99th percentile in the reference dataset were estimated by bootstrapping. Three stations with at least 10 gravid females were put as minimum for classifying the status of the area.

Status class	Malformed embryos%
High	< 0,029
Good	0,029 < 0,057
Moderate	0,057 < 0,086
Poor	0,086 < 0,114
Bad	> 0,114

Distribution of indicator species in Baltic Sea subregions

M. affinis occurs in the whole Baltic from northern part of the Bothnian Bay to the Gulf of Gdansk and Gulf of Riga. In the Belt Sea, where salinity is too high for *M. affinis* only the related species *Pontoporeia occurs*. The marine related species *P. affinis* occurs up to the northern Bothnian Sea where it disappears due to lower salinity. The deposit-feeding amphipod *Corophium sp.*occurs in the whole Baltic. Various gammarid species occur in the whole Baltic but most species are restricted to more shallow areas than *M.affinis* and *P. femorata* and live in the seaweed belt.

Acknowledgements

We acknowledge Anders Bignert at The Swedish Museum of Natural History and Kjell Leonardsson at Umeå University for important support during the calculation of assessment criteria. We also thank the Bonus project BEAST for opportunity to validate the methods for other amphipod species and the dep of Applied Environmental Science (ITM) for facilitating the progress of the work.

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4. Candidate indicators for biodiversity and hazardous substances

The HELCOM CORESET project identified several indicators which fulfilled the basic criteria to be proposed as core indicators, but that lacked some of the selection criteria, for example, validation of the scientific basis, links to pressures or the GES boundary. These indicators were labelled tentatively as candidate indicators and they will be revisited during the CORESET project. If finalized, the candidate indicators would fill significant gaps in environmental assessments, covering functional groups, anthropogenic pressures or hazardous substances that are not addressed by any of the proposed core indicators. Although many of the candidate indicators require more work, some of them are close to being finalized. This chapter presents the candidate indicators, which were identified in the HELCOM CORESET project.

The candidate indictors are listed in the summary table below and then described following the same numbering in separate sections. **Table 4.1** also includes expected approaches to define good environmental status for the indicators.

Table 4.1. Candidate indicators for biodiversity, hazardous substances and effects of hazardous substances.

Candidate indicators	Approach to define GES boundary
Distribution of harbour porpoise	Based on historic reference distribution.
By-catch of marine mammals, seabirds and non-target fish	By-catch close to zero. Deviation from zero can be higher for non-target fish species which have healthy stocks. Mammals and birds should have stricter targets.
Impacts of anthropogenic underwater noise on marine mammals	Modeling of noise intensity and mapping human activities causing noise.
Fatty-acid composition of seals as measure of food composition	Basic scientific research on the fatty-acid composition, describing a balanced species composition of prey species.
Abundance of breeding populations of seabirds	Defining sub-basin wise reference levels (mean of a selected time window) and a deviation from that. See OSPAR EcoQO.
Proportion of oiled seabirds	A decreasing trend (see OSPAR EcoQO).
Incidentally and non-incidentally killed white-tailed eagles	A decreasing trend.
Salmon smolt production capacity	GES boundary is based on a%-level which allows sustainable exploitation of the stock. The HELCOM proposal for the salmon potential smolt production capacity (PSPC) is 80%.
Sea trout parr density	GES boundary should be set on a%-level which supports a viable sea trout population in a river, taking into account also the exploitation pressure on the stock. Such target does not currently exist and therefore an interim target may need to be set in order to make initial assessments. The quality of the spawning habitat needs to be assessed on a class scale.
Large fish individuals from fishery-based data sources	Reference conditions from fishery independent data sets.
Abundance of cyprinids in archipelago areas	Expert judgment
Ratio of opportunistic and perennial macroalgae	Site-specific ratios. Partly established under WFD.
Cladophora length	Based on experimental data. Under development.
Size distribution of benthic long-living species	Based on low-impacted conditions (reference areas/periods).
Blue mussel cover	Long-term means + expert judgement
Cumulative impacts on benthic habitats	Based on the favourable conservation status of the EU Habitats Directive.
Biomass of copepods	Based on time periods of good nutritional status of herring.
Biomass of microphageous zooplankton	Based on time periods of good water transparency
Zooplankton species diversity	Based on species lists.
Mean zooplankton size	Long-term means + expert judgement.

Zooplankton-phytoplankton biomass ratio	Long-term means + expert judgement.
Phytoplankton diversity	Long-term means + expert judgement.
Seasonal succession of functional phytoplankton group	Long-term means + expert judgement.
Alkylphenols	Environmental Quality Standards for nonyl- and octylphenol
Vitellogenin induction	A level not showing feminization of male fish
AChE inhibition	BAC established and EAC estimated.
EROD/CYP1A induction	BAC is under development.

4.1. Distribution of harbour porpoise

1. Working team: Marine Mammals *Author*: Stefan Braeger	
2. Name of candidate indicator Geographical distribution of the critically endangered Baltic Proper harbour porpoise	*3. Unit of the candidate indicator* Presence as indicated by the frequency of registrations per area in a year (e.g., >10 registrations/1000km^2)
4. Description of proposed indicator The current population size of the Baltic harbour porpoise is extremely small and due to its low abundance no longer reliably quantifiable. Therefore, it appears impracticable to propose the abundance of harbour porpoise as a state indicator or a quantitative target for abundance as a conservation goal. At extremely low densities, such target would be almost impossible and very costly to monitor. The Baltic porpoise population has not only dwindled in numbers to less than 250 reproducing adults (IUCN 2008) but also evacuated large parts of its historic range throughout the Baltic Proper. Therefore, the extent of the distribution range appears to be a suitable proxy for population size assuming that an increasing population would also be likely to expand its range. Annecdotal information on (pre-industrial) porpoise distribution indicates a probably continuous distribution throughout the Baltic Proper, possibly also covering the entire Gulf of Bothnia as well. Therefore, a regular basin-wide presence could serve as proxy for successful population recovery.	
5. Functional group or habitat type Harbour Porpoise (a piscivorous top predator)	
6. Policy relevance Habitats Directive, Marine Strategy Framework Directive, and national obligations under a number of IGO resolutions (e.g., HELCOM, OSPAR, CMS, ASCOBANS etc.). MSFD Descriptor 1, criterion 1.1 Species distribution. The Baltic Sea Action Plan calls for "Abundance, trends, and distribution of Baltic harbour porpoise" as preliminary indicator for nature conservation and biodiversity. To achieve a viable population of this species, it provides the following targets: "By 2012 spatial/temporal and permanent closures of fisheries of sufficient size/duration are established thorough the Baltic Sea area" and "By 2015 by-catch of harbour porpoise, seals, water birds and non-target fish species has been significantly reduced with the aim to reach by-catch rates close to zero"	
7. Use of the indicator in previous assessments ASCOBANS, e.g. in the Jastarnia Plan (2002 & 2009).	
8. Link to anthropogenic pressures Directly linked to gillnet fisheries, persistant organic pollutants, underwater noise (e.g., from pile-driving, underwater explosions, seismic surveys, military sonar), disturbance from shipping, and habitat destruction (e.g. from gravel extraction, offshore structures, coastal development) among others.	

9. Pressure(s) that the indicator reflect Selective extraction of species, including incidental non-target catches, as well as others mentioned and point 8.
10. Spatial considerations Since harbour porpoises are highly migratory mammals, the spatial considerations would be determined by the desired species distribution, e.g. Baltic Proper.
11. Temporal considerations Harbour porpoises live in the Baltic Sea year-round, but have to avoid complete ice cover. The re-colonisation of the entire Baltic Proper cannot rely on immigration from other populations and will thus depend on intrinsic population growth from a very low abundance. Therefore, it is likely to take several decades at best.
12. Current monitoring Two aerial surveys of the southwestern part of the Baltic Proper (between southern Sweden and the coast from Darss Ridge to Gdansk) in 1995 and 2002 resulted in best estimates of 599 and 93 porpoises, respectively. Currently, porpoise densities are regarded as too low to make visual surveys any longer viable. Therefore, an ongoing international research project ("SAMBAH") uses static acoustic monitoring in 300 locations in the Baltic Proper in water depth between 5 and 80 metres and first results regarding the geographical distribution are expected to become available in the year 2014 (www.sambah.org).
13. Proposed or perceived target setting approach with a short justification. To measure the success of conservation measures that results in an increase of porpoise distribution range (and by analogy in porpoise numbers), a SAMBAH-like survey should be periodically repeated e.g. every ten years. Additionally, the number of sighted and locally stranded porpoises may provide a useful indicator for the regular presence of porpoises as well as insights into population health. Such information could be based on promotion of new or already existing voluntary reporting schemes such as provided in Poland (http://www.morswin.pl/index_base.php?Screen_Option=1&Page_ID=72), Germany (http://www.meeresmuseum.de/de/wissenschaft/sichtungen.html), Sweden (http://www.nrm.se/sv/meny/forskningochsamlingar/enheter/miljogiftsforskning/rapporteringavdjur.445.html), and Finland (http://www.environment.fi/default.asp?contentid=190711&lan=EN). Ultimately their entire historical range throughout the Baltic Proper should be recolonised by Baltic harbour porpoises. By then porpoises densities should have recovered sufficiently to allow reliable abundance estimation and the setting of alternative conservation targets. The Baltic Sea Action Plan, for example, also recommends pregnancy rate, fecundity rate, and the occurrence of pathological findings as indicators and targets for the ecological objective.

4.2. By-catch of marine mammals and birds

1. Working team: Marine Mammals
Author: Stefasn Braeger

2. Name of candidate indicator	*3. Unit of the candidate indicator*
Bycatch of marine mammals and birds	Numbers of individuals bycaught in fishing gear

4. Description of proposed indicator

Bycatch in fishing gear is known to be the most important threat to biodiversity and potential disruption of the food web as far as marine mammals in the wider Baltic Sea are concerned. Bycatch is also regarded as one of the main direct anthropogenic pressures on marine diving bird species. It can be measured as number of bycaught porpoises or seals either by a near-complete coverage with on-board observers/ CCTV-recording or by examining beached individuals. All net-setting vessels should be monitored since monitoring only a subset of vessels would lead to an estimate with considerable variance. Beached individuals with clear signs of by-catch injuries confirmed by pathological investigations can give indications that by-catch occurred in an area, but it is very difficult to use such data for obtaining total numbers of animals being affected and hence impacts on populations.

For healthy mammal populations (with an abundance ≥80% of a population at carrying capacity) a tolerable bycatch rate may amount to 1.0% (plus another 0.7% anthropogenic take due to other impacts such as pollution, noise etc.) of the local population. For depleted populations such as the "critically endangered" (according to Hammond et al. 2008) porpoise population of the Baltic Proper and the rapidly decreasing (according to Teilmann et al. 2011) porpiose population of the Belt Sea, bycatch was recommended to be reduced to near zero immedeately (cf. ASCOBANS 2002). For birds it has been shown that present bycatch rates are close to, or for some species even exceeding, levels that can be sustained by the populations.

5. Functional group or habitat type

Toothed whales, seals and birds (i.e., the piscivorous top predators)

6. Policy relevance

Habitats Directive, Marine Strategy Framework Directive, and national obligations under a number of IGO resolutions (e.g., HELCOM, OSPAR, CMS, ASCOBANS, AEWA etc.)

The Baltic Sea Action Plan provides the following targets: "By 2012 spatial/temporal and permanent closures of fisheries of sufficient size/duration are established thorough the Baltic Sea area" and "By 2015 by-catch of harbour porpoise, seals, water birds and non-target fish species has been significantly reduced with the aim to reach by-catch rates close to zero"

7. Use of the indicator in previous assessments

ASCOBANS (2002 & 2009) in the Jastarnia Plan, none for seals and birds.

8. Link to anthropogenic pressures

Directly linked to commercial fishery and recreational fishing with gillnets, and to a lesser degree and indirectly (through prey depletion) with other commercial fishery such as pelagic and bottom trawling as well as other coastal stationary gear.

9. Pressure(s) that the indicator reflect

Selective extraction of species, including incidental non-target catches

10. Spatial considerations

Since harbour porpoises are highly migratory mammals, the spatial considerations would be determined by the desired species distribution, e.g. Baltic-wide. Bycatch of birds is higher in areas where large bird concentrations occur. Such areas, however, are distributed throughout the Baltic.

11. Temporal considerations Harbour porpoises live in many parts of the Baltic Sea year-round, but have to avoid complete ice cover. Therefore, considerations could be less stringent in northern parts of the Baltic Sea in winter during ice-cover. Bird bycatch rates are higher in winter (Oct-April) due to several bird species then occurring in the marine environment in large concentrations. Bycatch, however, does occur all year around, although it affects different species at different times of the year. Areas covered completely by ice do not hold any diving birds and hence have no bycatch in winter.
12. Current monitoring In some rather small parts of the western Baltic Sea, porpoise monitoring is being attempted under EC Regulation 812/2004, however, with insufficient coverage. Furthermore, it is also required under the Habitats Directive. There is no monitoring for bird bycatch so far.
13. Proposed or perceived target setting approach with a short justification. In the Baltic Proper and in the Belt Sea, porpoise bycatch needs to be close to zero to allow recovery of the populations. The anthropogenic removal (including by pollution, ship strike, noise etc.) of more than one porpoise from this entire population has been modeled to thwart recovery from the brink of extinction (Berggren et al. 2002). The target of zero bycatch should be linked to a policy decision, e.g. to close the area for net-setting fisheries for the remainder of the year as soon as more than one harbour porpoise has been caught (cf. NOAA 2010, NZ Minister of Fisheries 2006). For the vulnerable Belt Sea population, an anthropogenic removal of 1% of the total population per year in Danish, Swedish and German waters combined would not allow recovery of the population from a 60% reduction between 1994 and 2005 and jeopardise its continued survival (Teilmann et al. 2011). For birds, levels of of tolerable total anthropogenic removal have been calculated for a few species based on known population sizes and demographic parameters. Such levels can be used to calculate the levels of bycatch that can be sustained by the populations (if knowing other sources of anthropogenic mortality as well). GES boundaries could then be set at levels below these to guarantee continued survival of the populations.

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4.3. Impacts of anthropogenic underwater noise on marine mammals

1. Working team: Marine Mammals *Authors*: Stefan Braeger & Stefanie Werner	
2. Name of candidate indicator Impacts of anthropogenic underwater noise on marine mammals	*3. Unit of the candidate indicator* Single and cumulative impacts on marine life from high-amplitude, low and mid-frequency impulsive sounds and low frequency continuous sound emitted per area and time
4. Description of proposed indicator There is a broad range of physiological or behavioral reaction to noise. Noise presents at least following threats: diversion of attention and disruption of behavior, habituation, masking of important signals, temporary and permanent effects of hearing and injury to other organs, sometimes leading to death. Seals are thought to be more sensitive to certain low frequency sound sources than harbor porpoises, but there are evident gaps in knowledge about their hearing capacities. Noise mapping should be conducted to analyze noise budgets of regional sea areas. Especially high-frequency and mid-frequency SONAR as well as percussive pile-driving during offshore construction and the use of airguns during seismic explorations and explosions during clearing of old ammunition are known to impact marine mammals behaviour severely or cause morphological damage. Shipping is known to be the dominant continuous sound source leading to ever increasing ambient noise levels. Acoustic indicator need to represent all noise components that have impact on marine life. Based on this overall allowable noise budgets can be set up. The indicator should aim on measuring loud, low and mid frequency impulsive sound as well as continuous low frequency sound sources and predict for their single and cumulative effects on marine life. Models need to be applied that allow predicting sources and frequency specific sound levels and sound propagation. Three-dimensional noise (propagation) maps should be produced showing the single and cumulative impact of different noise sources on a species by species basis. Knowing the potential habitat of marine life of concern and the potential impact of noise, it is possible to describe the potential impact on marine life. The species dependent impacts need to be estimated and weighted. Metrics for some of the sound induced effects on marine life are or will most likely become available over the next years. It is already possible to determinate impairment of significant life functions. Non permanent auditory injury (Temporal Threshold Shift - TTS) as well as the limited acoustical availability of habitat (masking) can already be expressed in metrics. The question above which needs to be answered is: when do these adaptive responses to an environmental stress, which are within an animals capacity to respond, lead to negative consequences for vital rates and populations.	
5. Functional group or habitat type Harbour porpoises, Grey, Harbour and Ringed seals	
6. Policy relevance Marine Strategy Framework Directive, Habitats Directive and a number of IGO resolutions (e.g., HELCOM, OSPAR, CMS, ASCOBANS etc.) The Baltic Sea Action Plan provides the following target: ""By 2015, improved conservation status of species included in the HELCOM lists of threatened and/or declining species and habitats of the Baltic Sea area, with the final target to reach and ensure favourable conservation status of all species""	
7. Use of the indicator in previous assessments ASCOBANS, e.g. in the Jastarnia Plan (2002 & 2009)	

8. Link to anthropogenic pressures Impulsive sound inputs in the Baltic are directly linked with offshore construction of wind farms, use of different types of Sonar, Depth Sounder and Fish Finder, explosions mainly for clearing of old ammunition and the use of Acoustic harassment devises for fishing and scaring of marine mammals during construction periods of wind turbines. Shipping as well as dredging of sand and gravel and the operation of offshore wind farms are related to the introduction of continuous sounds introduction.
9. Pressure(s) that the indicator reflect Anthropogenic underwater noise in the baltic environment.
10. Spatial considerations Sound travels in water about five time faster than in air and absorption is less compared to air. Due to t relatively good transmission underwater, sound acts at considerable spatial scales leading to vast impacted areas (e.g. >1000km² during pile-driving of a wind park) avoided by porpoises for a longer time. Noise of different sources can add up and may lead to synergies. Transmission varies with frequency: low frequency signals typically travel further whereas higher frequencies attenuate more rapidly, therefore fewer individuals might be exposed. Due to the low salinity of the Baltic a considerably lower absorption can be observed compared to other sea areas, which means that the radiated acoustic power travels over broader distances, especially at higher frequencies.
11. Temporal considerations Although each single sound may attenuate rather quickly under water, the total amount appears to increase constantly. Persistence of sound is very variable – ships on passage generate continuous sound whereas explosions are very short-term. From late spring to early autumn, harbour porpoises give birth and tend to very young calves that depend completely on the guidance of their mothers. This relationship appears to be particularly sensitive to disturbance by underwater noise. On days with pile driving activities in Nysted 20-60 percent less grey and harbour seals seeked their resting places than on usual days. A negative correlation was shown between main shipping lanes in the Baltic and harbour porpoise abundance.
12. Current monitoring So far, no monitoring of underwater noise appears to be in place in the Baltic Sea beside of the obligations for construction and operation of offshore wind farms.
13. Proposed or perceived target setting approach with a short justification. The introduction of impulsive and continuous sounds should be measured and modelled in order to predict for the cumulative impacts on marine life. Data from aerial and ship-based abundance surveys of marine mammals need to be used for habitat modelling. The sound sources that exceed thresholds that are likely to entail significant impact on marine animals have to be defined and these levels need to be defined in a precautionary way. Maximum sound exposure levels at a certain distance from the sound source for pile driving activities are already in force in Germany, for example, and could become compulsory in all EU waters. Overall allowable noise budgets per area and time need to be defined. The total amount of energy introduced into any area over a standardized time should be generally limited and permitted on a case by case basis with binding mitigation measures in place. To establish the species specific impact as a function of the distribution of noise over time and space the above mentioned steps can be used to create a (threshold) factor as an indicator for the impact of noise. For species of concern, it is therefore necessary to develop a three-dimensional (propagation) model that takes account of the duration of noise events. The result of this modeling process would be a map with a grid of related impacts based on the Sound Exposure Level (SEL). The benefit of this approach is the possibility of defining acceptable levels based on scientific estimates. High-frequency sounds, e.g. from depth sounders, fish finders and other SONAR should be limited, especially in shallow coastal areas, to the minimum. The level of underwater noise below 300 Hz is dominated by noise inputs from ships. This noise is broadband and has a maximum level of 50 Hz. It can definitely be correlated with the propeller of a ship. Ship quietening measures need to be identified with the view on how to influence different operational parameters of the ship (e.g. fuel consumption).

4.4. Fatty-acid composition of seals as measure of food composition

This candidate indicator is a measure of food composition and, thus, follows changes in the food web. There is very little data available of the fatty-acid composition in the seals. The data set should be widened and used to determine the fatty-acid composition which indicates good environmental status of food webs. It should also be ensured that the fatty-acid composition of studied individuals is not caused by other factors such as illnesses or hazardous substances.

This indicator requires basic scientific research until it can be used in environmental assessments.

4.5. Abundance of breeding populations of seabirds

<table>
<tr><td colspan="2">1. Working team:
Seabirds</td></tr>
<tr><td>2. Name of candidate indicator
Abundance of breeding populations of seabirds</td><td>3. Unit of the candidate indicator
Species-level: individuals
Integrated level: proportion of species above the target</td></tr>
<tr><td colspan="2">4. Description of proposed indicator
Seabirds are significant predators and herbivores of the marine ecosystem. The indicator follows population sizes of pre-selected seabird species, belonging to different functional groups. The indicator follows the OSPAR EcoQO for breeding birds, where each species has a separate reference level and target level but the indicator measures the proportion of species reaching their targets.
The indicator should be set separately to different sub-basins of the Baltic Sea, as species abundances and even species composition varies geographically in the Baltic Sea.</td></tr>
<tr><td colspan="2">5. Functional group or habitat type
Coastal herbivores, coastal benthic feeders, coastal pelagic fish feeders, offshore pelagic fish feeders</td></tr>
<tr><td colspan="2">6. Policy relevance
Descriptor 1, criterion 1.2 Population size
Descriptor 4, criterion 4.2 Abundance/distribution of key trophic groups and species
(Descriptor 5 & 6: indirectly)</td></tr>
<tr><td colspan="2">7. Use of the indicator in previous assessments
OSPAR EcoQO</td></tr>
<tr><td colspan="2">8. Link to anthropogenic pressures
The breeding seabirds are directly impacted by habitat loss, oil spills, by-catch of fisheries, hunting, displacement by offshore constructions and shipping traffic.
Indirect impacts include eutrophication and physical disturbance of bottom sediments (through changes in food supplies).
Although eutrophication affects the population only indirectly, it is the most significant factor affecting the abundance of breeding seabirds.</td></tr>
<tr><td colspan="2">9. Pressure(s) that the indicator reflect
Habitat loss, selective extraction of species, introduction of synthetic compounds, input of fertilisers and organic matter, abrasion and selective extraction, changes in siltation and thermal regime, other physical disturbance.</td></tr>
<tr><td colspan="2">10. Spatial considerations
Seabirds breed in all the sub-basins of the Baltic Sea, but the species composition and abundance varies geographically. Therefore, each sub-basin should have an own list of selected indicator species with own reference levels and target levels.
The assessments should be made on the sub-basin level.</td></tr>
</table>

11. Temporal considerations Frequency: ideally as often as possible, but realistically perhaps every fifth year
12. Current monitoring Monitoring done in all the Contracting States. Measured parameters must be compared and the indicator species agreed and included in the monitoring.
13. Proposed or perceived target setting approach with a short justification. The reference level for each seabird population should be set at a population size that is considered desirable for each individual species within each geographical area. This should be set for each species based on expert judgement of when population levels were subject to low impact by human activities. In the OSPAR EcoQO the target level was set to the level of standard deviation from the reference level. The lower target depended, however, partly on the species; species laying one egg were given stricter target (being more sensitive) than species laying more eggs. The proportion of species reaching their target is the "integrated indicator". The target level could be 75% as in the OSPAR EcoQO.

Additional indicators for the breeding seabird populations

The abundance of breeding seabirds is a good indicator for following long-term changes, whereas it is not feasible for detecting short-term changes. Short-term responses to environmental changes can be better followed by **reproductive parameters**, such as breeding success or brood size. The OSPAR EcoQO (under development) for the breeding success of the kittiwake is an example of such an indicator. It may be used as an indicator for the abundance of sandeel that also serves as a major food source for many other bird, fish and marine mammal species. Sandeel availability may be low because of natural reasons, or through industrial fishing that competes with seabirds. When the breeding success of kittiwake is low over a three-year period, this is likely to be triggered by low abundance of sandeel in coastal areas. The indicator thus shows the status of sandeel and kittiwake populations and the need to reduce industrial fishing.

In the Baltic Sea, the availability of data on brood size and breeding success needs to be clarified.

The national monitoring programmes to monitor the breeding populations of sea birds can be also used to develop a **distribution indicator**. The indicator would require sub-basin wise species selection and setting of GES boundaries. In addition, it should be decided which density threshold is used to determine the margin of distribution.

4.6. Proportion of seabirds being oiled

1. Working team Sea birds	
2. Name of candidate indicator Proportion of seabirds being oiled	*3. Unit of the candidate indicator* Number of oiled seabirds per species
4. Description of proposed indicator Proportion of oiled seabirds found in the winter-time population counts are being used for the estimate of this indicator. In addition, proportion of oiled seabirds among those found dead or dying on beaches can be used as an additional information.	
5. Functional group or habitat type Mainly offshore benthic feeders and offshore pelagic and surface feeding birds	
6. Policy relevance Descriptor 1 Descriptor 4	

7. Use of the indicator in previous assessments None
8. Link to anthropogenic pressures Directly linked to oil slicks/oil spills (shipping)
9. Pressure(s) that the indicator reflect oil slicks/oil spills (shipping)
10. Spatial considerations
11. Temporal considerations Frequency: should be updated annually
12. Current monitoring None
13. Proposed or perceived target setting approach with a short justification. A baseline could be set based on the current situation (baseline monitoring needed!) and then the approach adopted by OSPAR could be used. They suggest the following GES boundary "The average proportion of oiled birds in all winter months (November to April) should be 20% or less by 2020 and 10% or less by 2030 of the total found dead or dying in each of 15 areas of the North Sea over a period of at least 5 years"

4.7. Incidentally and non-incidentally killed white-tailed eagles

1. Working team Sea bird team based on the HELCOM Indicator Fact Sheet on health status of white-tailed eagle.	
2. Name of candidate indicator Incidentally and non-incidentally killed white-tailed eagles	*3. Unit of the candidate indicator* Proportion of dead eagles caused by anthropogenic causes
4. Description of proposed indicator Death causes of found dead eagles are estimated for birds handed in to the authorities.	
5. Functional group or habitat type Top predatory birds	
6. Policy relevance Descriptor 1 Descriptor 4 (Descriptor 8)	
7. Use of the indicator in previous assessments Used in the HELCOM "Population Development of Baltic Bird Species: White-tailed Sea Eagle (*Haliaeetus albicilla*)"-indicator. For more detailed descriptions see the indicator fact sheet (2009)	
8. Link to anthropogenic pressures Directly linked to hunting, persecution, synthetic and non-synthetic compounds, electrocution, collisions with wind turbines.	
9. Pressure(s) that the indicator reflect Hunting, persecution, synthetic and non-synthetic compounds, electrocution, collisions with wind turbines.	
10. Spatial considerations	
11. Temporal considerations Frequency: can be updated annually	
12. Current monitoring Monitored in Finland, Sweden, Germany and Denmark.	
13. Proposed or perceived target setting approach with a short justification. Historical data exists that can be used as reference data.	

4.8. Status of salmon smolt production, smolt survival and number of spawning rivers

<table>
<tr><td colspan="2">1. Working team
Author: Atso Romakkaniemi on the basis of work in ICES WGBAST</td></tr>
<tr><td>2. Name of candidate indicator
Status of salmon smolt production, smolt survival and number of spawning rivers</td><td>3. Unit of the candidate indicator
Three parameters:
smolt survival (%)
smolt production (individuals)
increase/decrease in number of rivers with natural spawning of salmon (% change from xxxx(2010?) situation)</td></tr>
<tr><td colspan="2">4. Description of proposed indicator
Salmon (Salmo salar) is a big predatory species in the Baltic Sea marine ecosystem. Its abundance is affected not only by commercial fishing but also by the condition of the spawning rivers and the marine ecosystem. When smolts enter the sea, they must have enough suitable food items along their migration paths and they must be able to avoid predation and by-catching in order to survive over the first, critical year. Many of the spawning rivers have been dammed to produce hydroelectricity, and the spawning grounds have in many rivers degraded due to increased siltation and eutrophication (forestry, agriculture). There are also former salmon rivers with few or no migration obstacles; natural reproduction could be re-established especially in these rivers.

This indicator is a combination of three parameters, which can be assessed separately and then combined (all or part of them) to give a single measure of the status of salmon.
survival of smolts in the sea, (can be also a separate indicator the conditions in marine ecosystem)
number of smolt produced annually in a river, and
number of salmon rivers (trend in the number of rivers). (The latter two reflect the reproduction of salmon and has a linkage to the marine ecosystem)</td></tr>
<tr><td colspan="2">5. Functional group or habitat type
Anadromous fish</td></tr>
<tr><td colspan="2">6. Policy relevance
Descriptor 1: Criterion 1.1 – Species distribution
Criterion 1.2 – Species abundance
Criterion 1.5 – Habitat extent
BSAP Ecological objective “Thriving communities of plants and animals” (Nature conservation)</td></tr>
<tr><td colspan="2">7. Use of the indicator in previous assessments
Derived from the ICES salmon assessments. Can be supplemented by the results of MS’s monitoring for Habitats Directive, which has not been included in ICES work which focuses only on original stocks.</td></tr>
<tr><td colspan="2">8. Link to anthropogenic pressures
Smolt survival decreases as a result of lack or mismatch of suitable food (invertebrates and prey fish like young herring/sprat/stickleback the abundance of which are affected by human) and commercial fishing (by-catch).
The smolt production depends on the number of spawners (affected by fishing) and the quality of the river (damming and degradation of suitable habitats).
The number of salmon rivers decreases as a result of damming and degradation of suitable habitats and increases as a result of restoration efforts (improving passage through migration obstacles, measures to improve water quality, flow regimes, physical quality of spawning and nursery habitats).</td></tr>
<tr><td colspan="2">9. Pressure(s) that the indicator reflect
The indicator reflects fishing (commercial and recreational), state of the sea ecosystem (e.g., availability of food for young salmon), river connectivity, siltation (forestry, agriculture) and inputs of nutrients and organic matter (agriculture, waste waters, animal husbandry).</td></tr>
</table>

10. Spatial considerations Baltic-wide, all rivers of wild or mixed salmon populations. Salmon migrates widely and hence is a part of the Baltic food web all over the basin.
11. Temporal considerations Smolt production and survival may vary annually depending on fishing effort and annual conditions at sea. Indicators describing a specific year become available by the end of May 2012.
12. Current monitoring Estimates produced frequently by national research institutes and compiled by ICES working group WGBAST. Smolt survival and smolt production are basically ready for operational use. WGBAST may further develop the number of salmon rivers as an indicators, as well as can consider ways of combining the second and the third indicator; the outcome of this work is expected to be operational by 2013.
13. Proposed or perceived target setting approach with a short justification. GES boundary is based on a %-level which allows sustainable exploitation of the stock. The HELCOM proposal for the salmon potential smolt production capacity (PSPC) is 80%. The GES boundary for this indicator could be based on this approach, but requires further development.

4.9. Sea trout parr densities of sea trout rivers vs. their theoretical potential densities, and the quality of the spawning habitats

1. Working team Author: Atso Romakkaniemi on the basis of work in ICES WGBAST	
2. Name of candidate indicator Trout parr densities of sea trout rivers vs. their theoretical potential densities, and the quality of the spawning habitats	*3. Unit of the candidate indicator* % (proportion of parr density reached) and an index for classification of spawning habitats
4. Description of proposed indicator Sea trout (*Salmo trutta*) is a big predatory species in the Baltic Sea marine ecosystem. Its abundance is affected by fishing and the condition of the spawning rivers. Many of the spawning rivers have been dammed to produce hydroelectricity and the spawning grounds have in many rivers degraded due to migration obstacles and increased siltation and eutrophication (forestry, agriculture). This indicator is a combination of two parameters. The first one follows the realized parr densities of trout in rivers as a percentage of the estimated, theoretical maximum densities. This parameter serves as an overall response to the quality of the spawning grounds and the adjacent sea and the fishing pressures. The second parameter estimates the quality of the spawning habitat in the spawning rivers. This latter parameter is still under the development. The data for the indicators is compiled by national research institutes and the indicators would be calculated in the ICES WGBAST. WGBAST is further developing both indicators. The first indicator it is expected to be operational by 2012/2013 and the latter one by 2013/2014.	
5. Functional group or habitat type Anadromous fish	
6. Policy relevance Descriptor 1: Criterion 1.1 – Species distribution Criterion 1.2 – Species abundance Criterion 1.5 – Habitat extent BSAP Ecological objective “Thriving communities of plants and animals” (Nature conservation)	

7. Use of the indicator in previous assessments In ICES assessments
8. Link to anthropogenic pressures The parr densities in sea trout rivers decrease as a result of decreased quality of the spawning habitat or adjacent sea or due to increasing fishing pressure.
9. Pressure(s) that the indicator reflect The indicator reflects fishing pressure, siltation (forestry, agriculture), inputs of nutrients and organic matter (agriculture, waste waters, animal husbandry), and river connectivity.
10. Spatial considerations Baltic-wide. Sea trout does not migrate as widely as salmon and therefore the quality of the spawning rivers has a more direct connection to the adjacent sub-basin.
11. Temporal considerations The indicator responds to the river condition and therefore may respond slowly to management measures. Among the smallest rivers, annual variation in river flow is a driving force in annual variation of parr densities.
12. Current monitoring Estimates produced frequently by national research institutes and complied by ICES WGBAST.
13. Proposed or perceived target setting approach with a short justification. GES boundary should be set on a%-level which supports a viable sea trout population in a river, taking into account also the exploitation pressure on the stock. Such a target does not currently exist and therefore an interim target may need to be set in order to make initial assessments. The quality of the spawning habitat needs to be assessed on a class scale. The criteria for the classification are not yet ready.

4.10. Large fish individuals from fishery-based data sources

1. Working team MARMONI Life –project (Antti Lappalainen et al.)	
2. Name of candidate indicator Large fish individuals (complement)	*3. Unit of the candidate indicator* Proportion of large fish individuals in fish populations. Mean size of sexual maturation.
4. Description of proposed indicator The original core indicator "Large fish individuals" is monitored in HELCOM FISH project by gill nets. Perch and some Cyprinids typically form the bulk of the catch. Other data sources, such as coastal trawl surveys, can be used in order to get population level data from a wider group of species The fish catch data collected under the EU Data Collection Regulation is also a source which can be used for this indicator. A part of this data, e.g sampling of predatory species from herring traps, covers all size classes and can be treated here as fishery independent data. These data can be used e.g. to calculate: Proportion of fish larger than the mean size of first sexual maturation mean size at first sexual maturation These issues (1 and 2) are closely linked to effects of fishing pressure on fish populations and on population level biodiversity. In the Marmoni –project, Finland (FGFRI) will shortly evaluate the usability of the fishery data (EU Data Collection regulation) and national trawl survey data for this kind of indicators. This work will be focused on pike-perch. In Estonia (EMI), there are long time series of trawling survey data, which might be used to assess the effects of fishery on pike-perch stocks. The usability of coastal commercial catch data to monitor the state of coastal fish stocks will be evaluated out, too. Latvia has long time series of trawling survey data, too, and the data will be evaluated in the Marmoni –project.	
10. Spatial considerations Valid in the entire Baltic Sea.	
11. Temporal considerations	
12. Current monitoring Fishery dependent data (catch samples from commercial fishery) is collected by all countries under the EU Data Collection Regulation. Trawling surveys are also carried out in several countries around the Baltic Sea.	
13. Proposed or perceived target setting approach with a short justification. GES boundaries not set yet. Reference conditions could be based on fishery independent monitoring results from earlier time period(s). There is also a general principle that every fish individual should have a chance to spawn at least once before they are targets for effective fishery	

4.11. Abundance of Cyprinids in archipelago areas

1. Working team Finnish Game and Fisheries Research Institute (Lappalainen, Lilja and Heikinheimo)	
2. Name of candidate indicator Abundance of Cyprinids in archipelago areas	*3. Unit of the candidate indicator* biomass (kg/ha) or catch per unit effort
4. Description of proposed indicator Cyprinids, especially bream (*Abramis brama*), roach (*Rutilus rutilus*) and white bream (*Blicca bjoerkna*) have become increasingly abundant in large archipelago areas of the northern Baltic Sea (e.g. in the Gulf of Finland, Archipelago Sea and the Åland archipelago). The main reason for the increased abundance is the coastal eutrophication favouring the reproduction of cyprinids in the inner archipelago, but the warm springs and low salinity may have contributed. Similar development has widely been reported from eutrophied lakes. The present high abundance of Cyprinids likely tends to maintain the eutrophic conditions in the archipelago areas thorough the effects on food webs and by increasing the leaking of nutrients from soft bottoms. The high amount of Cyprinids in the gill-net and trap net catch in the Finnish coastal areas has also been regarded as a problem for coastal commercial fishery and a bulk of the catches have been discarded. A high abundance of cyprinids also indicates a biased fish community structure, which may indicate "less than good environmental status" under the GES Descriptor 1. The relative abundance of Cyprinids has so far been monitored by multi-mesh gill-nets at certain study areas especially in Sweden, but also in some study areas in Finland, Estonia and Lithuania. In both of the Finnish study areas, located in the Gulf of Finland, Cyprinids have formed over half of the total catches. There is relatively high temporal and spatial variation in the gill-net monitoring catches and it seems that this survey method underestimates the abundance of bream and white bream, which are the species mostly benefiting of coastal eutrophication. Gill-net monitoring could be seen here as one tool for 'an overall screening' and more thorough assessments should be considered in areas where the abundance of Cyprinids is evidently high. There are approaches and survey methods which could be used for more detailed assessments of Cyprinid abundance. Horizontal echo-sounding (with trawl or seine samples of the fish) is a promising method to survey large shallow areas and produce even species-specific biomass estimates for the most common species. This method is under development and testing at the moment. Commercial fishery on Cyprinids with trap nets could offer a possibility to use the log-book data (and regular sampling of the catch) to calculate catches per unit of effort (CPUEs) which could be reliable indexes of Cyprinid abundance. The results of these sampling methods would not be directly comparable with the existing gill net monitoring by HELCOM FISH. However, these methods could be used to get a finer assessment of Cyprinid populations in specific coastal areas, where effects of human pressures, e.g. eutrophication, on coastal fish communities are evident. Proper data on historical reference conditions does not exist. GES boundaries could be set as expert judgements, e.g. 30-40 % reduction in Cyprinid biomass in coastal regions where Cyprinids are at present very abundant.	
5. Functional group or habitat type Cyprinids	
6. Policy relevance Descriptor 1: Criteria 1.2 and 1.3– Population size, population condition Descriptor 1: Criterion 1.6 – Habitat condition (Condition of the typical species and communities, Relative abundance and/or biomass) Descriptor 5: Criterion 5.3 – Indirect effects of nutrient enrichment BSAP: Ecological objectives "Natural distribution and occurrence of plants and animals"	
10. Spatial considerations Valid in archipelago areas and lagoons. Applicability of the indicator in other kinds of areas should be validated.	

11. Temporal considerations Sampling should be made outside spawning season. Timing of sampling/survey is important as Cyprinids have some feeding and spawning migration (aggregation) patterns. Timing should be adjusted locally/regionally.
12. Current monitoring No current monitoring. The methods are under development.
13. Proposed or perceived target setting approach with a short justification. GES boundaries are not set yet. GES boundary can be set by expert judgment.

4.12. Ratio of opportunistic and perennial macroalgae

1. Working team: Benthic habitats and associated communities Author: Karin Furhaupter	
2. Name of candidate indicator Biomass relation perennials/opportunists	*3. Unit of the candidate indicator* % (based on g DW m^{-2} values)
4. Description of proposed indicator The candidate core indicator for the Biomass relation of perennial and opportunist macrophytes is an indicator which is used by Germany, Denmark, Poland and Estonia in the WFD assessment. The indicator has not been further developed in the CORESET project and it has not been tested in the northern Baltic Sea, e.g. in Finland and Sweden. Moreover, there seem to be differences in the monitoring procedures among the countries using it. The indicator basically follows the biomass relation between K- and r-species. K-species are habitat-structuring, perennial species (usually key-species) with a persistent biomass for a certain time scale adapted to stable conditions. R-species are fast growing small, tiny species adapted to instable conditions and opportunistic responses to, e.g. nutrient availability. This is indicator for the relation of most important functional groups and for the habitat condition.	
5. Functional group or habitat type Hydrolittoral and infralittoral hard substrata and/or sediments	
6. Policy relevance Descriptor 1, criterion 1.6 Habitat condition Descriptor 5, criterion 5.2 Direct effect of nutrient enrichment Descriptor 6, criterion 6.2 Condition of benthic communities BSAP: Ecological objectives "Natural distribution and occurrence of plants and animals" (Eutrophication) and "Thriving communities of plants and animals" (Nature conservation)	
7. Use of the indicator in previous assessments WFD assessment in Estonia and Germany (not in the hydrolittoral zone in Germany) and Poland (only for infralittoral sediments)	
8. Link to anthropogenic pressures The indicator reflects a shift from perennial species (macroalgae and angiosperms) towards opportunistic species as a consequence of increased nutrients causing rapid growth of opportunistic species, which overgrow and displace perennial vegetation – direct impact increased nutrients and increased amounts of organic matter and silt preventing the attachment of perennial species – direct/indirect impact increased physical disturbance (due to abrasion, extraction, dredging) favouring opportunistic species, which are adopted to "instable" conditions resulting in a displacement of perennials – direct impact changes in salinity and thermal regime favouring growth of opportunistic species resulting in a displacement of perennials – direct impact	

9. Pressure(s) that the indicator reflect Input of fertilizers, changes in siltation, Abrasion, Smothering, Changes in salinity regime, Changes in thermal regime.
10. Spatial considerations Baltic-wide but eventually with sub-basin specific and even site-specific GES boundaries
11. Temporal considerations Timing of measurement is important (adjusted to local conditions). Harmonization of the sampling method may be required (time of sampling, depth, species lists, variables measured, etc.). Frequency of monitoring not necessarily high as change in perennial macroalgae is slow.
12. Current monitoring Part of WFD monitoring in several member states (see 7.) and in several national monitoring programs.
13. Proposed or perceived target setting approach with a short justification. Difficult definition of reference values, as historical data are more or less only qualitatively available. Use of current WFD values, expert judgement and ecological models. Modelling approaches should be used, based on abundance ratios along a water quality gradient. The GES boundaries will be site-specific. A fast reacting indicator due to high productivity of opportunists.

4.13. Cladophora length

1. Working team: Benthic habitats and associated communities Author: Ari Ruuskanen	
2. Name of candidate indicator Length of *Cladophora glomerata*	*3. Unit of the candidate indicator* mm
4. Description of proposed indicator Several macroalgal species of the ephemeral life cycle are efficient in assimilating nutrients from the surrounding sea water. They use the nutrients in their growth, which can be measured in the length or biomass of the filaments. Especially nitrate has been found to be the limiting nutrient for the ephemeral macroalgae. The length/biomass of ephemeral macroalgae is an indicator for long term nutrient availability. Nutrient concentrations fluctuate widely in the sea water, which affects their reliability as an indicator. As the algae assimilate nutrients over the whole growing season, they indicate the total availability of nutrients over a longer time period. The use of this indicator relies on prior work for length/biomass –nitrate correlation in laboratory conditions as well as in marine environment. Such experimental work is going on in Finland in the summer 2011. As a result, response curves for the use of the indicator will be published. The indicator will first apply to *Cladophora glomerata*, a dominant species all over the Baltic Sea, but other species can be included at later stages. There are scientific studies of the use of color as an indicator of the nitrate concentration in *C. glomerata*. The use of colour as a supporting or alternative parameter need to be studied. The indicator will not require costly monitoring as the data can be collected without diving from shores, navigational buoys and other man-made structures. The indicator will be developed also in the LIFE+ MARMONI project.	
5. Functional group or habitat type Hydrolittoral and infralittoral hard substrata and/or sediments (large stones, other structures).	
6. Policy relevance Descriptor 1, criterion 1.6 Habitat condition Descriptor 5, criterion 5.2 Direct effect of nutrient enrichment Descriptor 6, criterion 6.2 Condition of the benthic community BSAP: Ecological objectives “Natural distribution and occurrence of plants and animals” (Eutrophication) and “Thriving communities of plants and animals” (Nature conservation)	

7. Use of the indicator in previous assessments Not
8. Link to anthropogenic pressures The indicator reflects the response of ephemeral (opportunistic) macroalgae to increased nitrate availability. Nutrients cause rapid growth of opportunistic species, which can also overgrow and displace perennial vegetation.
9. Pressure(s) that the indicator reflect Input of fertilizers (Waterborne discharges of nitrogen, atmospheric deposition of nitrogen; aquaculture) and ship traffic.
10. Spatial considerations Baltic-wide but eventually with region-specific GES boundary values
11. Temporal considerations Monitoring in the end of growing season. Timing of measurement is important.
12. Current monitoring Not, but could be included in the current field monitoring with low extra expenses.
13. Proposed or perceived target setting approach with a short justification. Does not require historical reference conditions. GES boundary will be set on the basis of the response of the alga length/biomass to the nitrate concentrations. Thus, the target for nitrate concentration in water determines the boundary between GES and sub-GES. This target has been already established for coastal water types under the EU WFD and for open sea areas in HELCOM EUTRO PRO. Classification requires the use of a response curve, which relates the length or biomass measurement to nitrate levels (and status classes) and wave action of study site. GES boundaries (response curves) require regional validation (at least on main basin level).

4.14. Size distribution of benthic long-lived species

1. Working team: Benthic habitats and associated communities Author: Karin Furhaupter	
2. Name of candidate indicator Size distribution of benthic long-lived species (e.g. *Cerastoderma*, *Macoma* or *Saduria*)	*3. Unit of the candidate indicator* Ind./size class
4. Description of proposed indicator The population structure (abundance per size class) of specific, long-lived species like bivalves or *Saduria* or decapod crustaceans, which are key-species of different communities or target species of other ecosystem components (e.g. birds). The indicator describes the condition (functionality) and abundance of the biological component. Population structure of food web and community key species is an indicator for the whole habitat condition.	
5. Functional group or habitat type Infralittoral, circalittoral infauna communities (eventually also sediments below the halocline)	
6. Policy relevance Descriptor 1, criterion 1.6 Habitat condition Descriptor 4, criterion 4.3 Abundance/distribution of key trophic groups and species Descriptor 6, criterion 6.2 Condition of the benthic communities BSAP: Viable populations of species	
7. Use of the indicator in previous assessments Not.	

8. Link to anthropogenic pressures Population structure of bivalves is affected by changes in the food web due to selective extraction of target species, including incidental non-target species – direct/indirect impact Population structure of bivalves is affected by eutrophication. The proportion of small individuals increases under strong predation by cyprinid fish, which have increased in abundance due to eutrophication of Baltic Sea. Population structure of bivalves is affected by physical damage/disturbance as adult species got lost and recruitment success is reduced Population structure of bivalves is affected by changes in hydrography as reproduction and recruitment is dependent of specific T and S values
9. Pressure(s) that the indicator reflect Selective extraction of species, Input of fertilizers, Changes in siltation, Abrasion, Smothering, Changes in salinity regime, Changes in thermal regime
10. Spatial considerations *Cerastoderma glaucum* – Baltic-wide but eventually with region-specific reference values; *C. edule* only in subregions (Western Baltic) *Mya arenaria:* from Kattegat to Bottnian Sea (to mean bottom salinity of 6 psu) *Macoma balthica:* from Kattegat to Nordic Baltic Proper/Gulf of Finnland (to mean bottom salinity of 8 psu) *Saduria*: from Kattegat to Bothnian Sea (to mean bottom salinity of 6 psu)
11. Temporal considerations Frequency: every 2 years sufficient due to slow recruitment Harmonization of the sampling method may be required.
12. Current monitoring Only monitored in specific scientific surveys? Or: can the national benthos monitoring be used to derive the size distribution?
13. Proposed or perceived target setting approach with a short justification. GES boundaries should be set on the basis of known size distribution in low-impact conditions (reference sites, reference times).

4.15. Blue mussel cover

1. Working team: Benthic habitats and associated communities *Author*: Karin Furhaupter	
2. Name of candidate indicator Blue mussel cover	*3. Unit of the candidate indicator* % cover
4. Description of proposed indicator Coverage of blue mussels on hard and soft bottom along the depth gradient. The indicator describes the abundance and extent of the biological component/community with blue mussels as the habitat forming key-species.	
5. Functional group or habitat type Infralittoral, circalittoral hard substrata Infralittoral, circalittoral sediments	
6. Policy relevance Descriptor 1, criterion 1.5 Habitat extent Descriptor 6, criterion 6.1 Habitat size of biogenic substrata BSAP: Nature conservation and biodiversity – "Thriving and balanced communities of plants and animals" and "Natural distribution and occurrence of plants and animals"	

7. Use of the indicator in previous assessments –
8. Link to anthropogenic pressures Introduction of synthetic compounds (due to oils spill, ship accidents) causes a habitat loss – direct impact Sealing (due to harbours, coastal defence structures, bridges and coastal dams, wind farms) causes a loss of habitat area (but blue mussels accept artificial solid substrates quite well!) Selective extraction (due to dredging and sand/gravel/boulder extraction) causes habitat loss – direct impact Physical damage (due to abrasion, smothering or change in siltation) causes habitat loss - direct impact
9. Pressure(s) that the indicator reflect Introduction of synthetic compounds, Sealing, Smothering, Abrasion, Selective extraction
10. Spatial considerations from Kattegat to Bothnian Sea (to mean bottom salinity of 6 psu)
11. Temporal considerations The cover should be measured at the time of yearly maximum (if applicable). Regular monitoring, but not necessarily annually.
12. Current monitoring Currently not monitored, but datasets are available.
13. Proposed or perceived target setting approach with a short justification. Difficult definition of reference values, as historical data are more or less only partly available. GES boundary could be based on long-term means and expert judgement. Also trend-based GES boundary could be used. High natural variability especially at shallow sites due to varying reproduction success and high predator pressure (ducks, sea stars)

4.16. Cumulative impact on benthic habitats

1. Working team: *Authors*: Manuel Meidinger and Samuli Korpinen *Acknowledged persons*: David Connor, Maria Laamanen, Hans Nilsson and Johnny Reker	
2. Name of candidate indicator: Cumulative impact on benthic habitats	*3. Unit of the candidate indicator* Percentage
4. Description of proposed indicator The indicator measures the proportion of a benthic habitat being significantly impacted by a cumulative impact of anthropogenic disturbances. The indicator gives a result for each habitat type. The habitat data is based on the recent benthic habitat model of the EUSeaMap project and data sets on anthropogenic impacts are from the HELCOM Initial Holistic Assessment. The indicator relies on reliable habitat and pressure data, but also on estimates of the weights of each pressure in the cumulative impact score, definition of a significant cumulative impact, and acceptable proportion of a habitat type being significantly disturbed.	
5. Functional group or habitat type: All predominant benthic habitats, defined by depth zones and substrate type	
6. Policy relevance MSFD Descriptor 1 (Biodiversity), Criterion 1.6 Habitat condition MSFD Descriptor 6 (Seafloor integrity), Criterion 6.1 Extent of habitats being impacted. BSAP Ecological Objective: Natural marine landscapes.	

7. Use of the indicator in previous assessments None
8. Link to anthropogenic pressures Directly impacted by physical disturbance pressures on the seabed.
9. Pressure(s) that the indicator reflect Directly impacted by bottom trawling, dredging, extraction of sand and gravel (etc), disposal of dredged material (etc), shipping in shallow water, installation of wind farms, piers, platforms, bridges, dams, cables and pipelines, coastal erosion defence structures and replenishment of beaches. Also large salinity and temperature changes caused by waste water treatment plants and nuclear power plants (and other industry) disturb the seabed habitats.
10. Spatial considerations The index is Baltic wide, but can be calculated to smaller areas, e.g. sub-basins. The tool has no spatial limits, but the limitations arise from the data reliability.
11. Temporal considerations Human activities being slowly changing in larger spatial scales, the indicator can be updated every three years.
12. Current monitoring The data on pressures is compiled from different sources.
13. Proposed or perceived target setting approach with a short justification. Under the EU Habitats Directive a threshold of 25% is being used to classify a habitat type to 'Bad conservation status'. Using this threshold as a basis, a threshold of 15% or less is being proposed for this indicator.

Policy relevance of the indicator

The HELCOM Baltic Sea Action Plan (BSAP) and the EU Marine Strategy Framework Directive (MSFD) both require an assessment of the state of the habitats and the intensity and distribution of anthropogenic pressures impacting them. Particularly, the MSFD calls for an improved understanding and management of pressures and impacts arising from human activities and ultimately aims at reducing them. This should lead to improved state of the ecosystem and hence enhanced resilience of marine ecosystems to counteract natural and human induced changes whilst ensuring the sustainable use of ecosystem goods and services.

The MSFD requires Member States to put in place measures to achieve or maintain 'Good Environmental Status' (GES) in the marine environment by 2020 at the latest. To reach this overall goal Member States must develop and implement marine strategies. *The marine strategies must include an assessment of pressures and impacts and* develop environmental targets and associated indicators. These environmental targets and associated indicators should help to guide the progress towards achieving and maintaining GES.

The BSAP ecological objectives, relevant for benthic habitats, state that the distribution and extent of marine landscapes must be "natural" and species communities must be "thriving" and "balanced". Although the BSAP does not mention habitats or biotopes separately, the landscape and community-level objectives together set the standard for GES of marine habitats.

In the MSFD, the GES of the benthic habitats is particularly assessed with the qualitative descriptor 6 "*Seafloor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected*". The objective of the descriptor is that human pressures on the seabed do not hinder the ecosystem components to retain their natural diversity, productivity and dynamic ecological processes, having regard to ecosystem resilience.

The descriptor and particularly its criterion 6.1 require an assessment of the extent of significant impacts on benthic habitats.

A proposal for an approach to assess anthropogenic impacts on benthic habitats

Selection of data

The benthic habitats used in this indicator were based on the modeling work of the EUSeaMap project, which modeled over 200 biotopes (EUNIS level 4) for the Baltic Sea, based on substrate type, depth zone, salinity and bottom energy (**Table 4.2**). Because the use of over 200 biotopes is cumbersome, the first testing of the indicator was done with 18 biotopes describing substrate type and depth zone (and combining the other parameters). More detailed habitat types can also be used, for example, in national or sub-regional assessments. The 18 benthic biotopes are described below.

Table 4.2. Categories of broad-scale habitats from the EUSeaMap project.

Infralittoral	**Circalittoral**	**Deep sea**
mud & sandy mud	mud & sandy mud	mud & sandy mud
sand & muddy sand	sand & muddy sand	sand & muddy sand
coarse sediment	coarse sediment	coarse sediment
mixed sediment	mixed sediment	mixed sediment
till	till	till
bedrock & boulders	bedrock & boulders	bedrock & boulders

The project selected twelve physical pressures, which pose direct disturbance on seabed. The selected pressures were shipping in shallow water (<15 m), dredging and sand extraction, disposal of dredged matter, waste water treatment plants, bridges and dams, oil rigs, coastal defense structures, coastal nuclear power plants, cables and pipelines, wind farms, bathing sites and bottom-trawling fishery. They were selected from the HELCOM Data and Map Service (http://maps.helcom.fi), where all the pressure data is visible and downloadable. The pressures are also visible in the HELCOM background report for the Baltic Sea Impact Index (HELCOM 2010a, Korpinen et al. 2012). The coarseness of some of the pressure data sets was estimated to cause bias in the outcome of the indicator, but they were included in order to finalize the testing of the approach.

Most of the anthropogenic pressures had quantitative intensities based on the underlying human activities; some used areal coverage of the activity or amount of material being moved, while some were limited to the presence-absence scale. As there are no or very few data sets available where the actual pressure has been measured, the intensities were proxies for pressures. More reliable proxies or more direct measurements are needed in future assessments. The intensity scales were log-transformed and normalized to 0-1 scale to have similar scales for all the twelve pressures.

Estimating impacts

Each pressure has individual impacts on the different habitats. These impacts are difficult to objectively measure (at least for all pressures) and therefore expert judgment was used to estimate the magnitude of the impact. These *weighting scores* are presented in the background report to the HELCOM Baltic Sea Impact Index. The normalized pressures were multiplied by the weighting scores and then summed up for every benthic habitat type. This cumulative impact was used as a measure for the total human impact on each of the habitats. Cumulative impacts were calculated for 276 m x 276 m squares over the entire Baltic Sea region.

The weighting scores naturally limit the number of habitats being included in the assessment. For every new habitat type, twelve more weighting scores must be developed.

Benchmarking of individual or total impacts to unacceptable or acceptable levels is a task that remains to be done in the Baltic Sea. Until that has been accomplished, the HELCOM CORESET project used the mean cumulative impact of the entire Baltic Sea as the level to represent "significant impacts". In practice, the *mean impact* means that there are either *several low impacts* present in the site or *one heavy impact*. Such an approximation was seen rough but still applicable as a first step towards more detailed assessments of cumulative human impacts.

Classification of Impacts

The candidate indicator uses four different impact categories (not impacted, low impact, medium impact and high impact) to describe the cumulative impacts using mean and standard deviation to define class boundaries (**Table 4.3**). The border between low and medium was defined as "significant impact".

Technical details of the data handling and the GIS procedures are given in the section "Technical data" below (some information still missing).

Table 4.3. A tentative classification of impacts in the benthic cumulative impact index.

Category	Cumulative impact score
Not impacted	0
Low impact	0 < à Mean
Medium impact	Mean à Mean + standard deviation
High impact	Mean + stdev à max. cumulative impact score

Approaches to set the GES boundary for the condition of benthic habitats

The HELCOM core indicators for BSAP and MSFD assessments have quantitative targets which define the boundary between GES and sub-GES. The MSFD GES criterion 6.1 of the EC Decision 477/2010/EU calls for an indicator to measure "*Extent of the seabed significantly affected by human activities for the different substrate types*". This means that two thresholds must be defined:
– the level of significant impact (a precondition for the next definition) and
– the level for GES, i.e. the proportion of benthic habitats which is not significantly affected.
The first definition requires a thorough investigation of the impacts of different pressures on benthic habitats (which depend on the habitat type). If aiming at perfection, one should also estimate the significance of synergistic impacts. At present, this information is not available and the CORESET project suggests using *mean cumulative impact* to denote "significant impact" (see also text above).

The second definition may be hard to set by ecological means. All anthropogenic impacts degrade the habitat quality and it may be difficult to find sudden drops in the habitat quality along the pressure-state response curves. Another approach would be so-called "assessment of the ecological coherence" which measures the size, number and connectivity of areas and which has been previously used for assessments of MPA networks (see HELCOM 2010b and publications of the EU BALANCE project [13]). A practical solution is to select a basis which is also used in the EU Habitats Directive. Under the directive a habitat which has 25% of area significantly impacted is classified to 'Unfavourable - Bad status'. In this indicator we can assume that 15% of the habitat area is allowed to be significantly impacted and still be in good environmental status, allowing some human use of the marine environment.

13 http://www.balance-eu.org/, particularly http://balance-eu.org/xpdf/balance-interim-report-no-25.pdf

Technical data

Metadata

This is an indicator to estimate the cumulative impact of anthropogenic pressures on benthic habitats. It measures the amount of each habitat type being significantly impacted by cumulative impacts. The indicator is targeted to be used in the HELCOM biodiversity assessments and to assess the EU MSFD GES criteria 1.6 and 6.1.

Description of data

Metadata and the maps of each of the pressure data layers are available in the HELCOM Data and Map Service (http://maps.helcom.fi).

The benthic habitats are from the EUSeaMap project: http://jncc.defra.gov.uk/page-5020.

Data source

EUSeaMap (raster, cell size/resolution 276m)

Relevant anthropogenic (physical) pressure data from prior projects (HELCOM HOLAS) were selected:

Dredging, Dumping, Windfarms, Oil rigs, cables and pipelines, coastal defense structures, bridges and dams, bathing sites, coastal shipping <15m water depth, nuclear power plants, bottom trawling, municipal waste water treatment plants MWWTPs

Geographical coverage

All regions of the Baltic Sea, whereas the data quality varies among data regions and data layers.

Temporal coverage

The pressure data is from period 2003-2008, biased towards 2006-2007. Exact years are available in pressure metadata.

Methodology and frequency of data collection

See HELCOM Data and Map Service.

Methodology of data analyses

– The main tool used for data preparation and analyses are version ArcGIS 9.3.1 with Spatial Analyst extension.
– In case the original pressure layers had point or polyline feature characteristics they were converted to polygon shapefiles using an individual buffer for each pressure layer:
  – shipping in shallow water (<15 m) – average monthly shipping density in 2008
  – dredging and sand extraction, disposal of dredged matter – both with buffers of 2000m
  – waste water treatment plants with buffer of 100m
  – bridges and dams with buffer of 100m
  – coastal defense structures with buffer of 100m
  – coastal nuclear power plants with buffer of 1000m
  – cables and pipelines with buffer of 50m
  – wind farms and oil rigs – original polygon shapefile
  – bathing sites with buffer of 500m
  – bottom-trawling fishery – ICES rectangles
– Then the individual pressure layers were converted to raster format with a cell size of 276m.
– Each individual pressure layer was log-transformed, normalized to 0-1 scale, multiplied by the corresponding expert judgment weighting score for each benthic habitat and summed up. The weighting scores are from the HELCOM background report 25.
– Six cumulative impact score rasters have been produced for each biozone (infralittoral, circalittoral, deep sea) and were then clipped with the corresponding habitat shapefile and sub-basin shapefile.

Weaknesses of the indicator

– coarseness of pressure data sets
– reliability of the habitat model
– artefacts in the map
– weighting of pressures
– benchmarking the impacts

Examples of the use of the indicator

The indicator was tested in the HELCOM Secretariat by using 12 pressure data layers from the HELCOM HOLAS project, 18 benthic habitats from the EUSeaMap project and tentative GES boundary of 15% of acceptable proportion being impacted.

Figure 4.1 shows the cumulative impacts on six infralittoral benthic habitats on a continuous scale. In **Figure 4.2**, the cumulative impacts are divided to four status classes (not impacted, low, intermediate and high) by using the mean cumulative impact as the boundary between 'low' and 'intermediate' and standard deviations to define the other two boundaries. **Figure 4.3** shows the same result as proportions of the total habitat area.

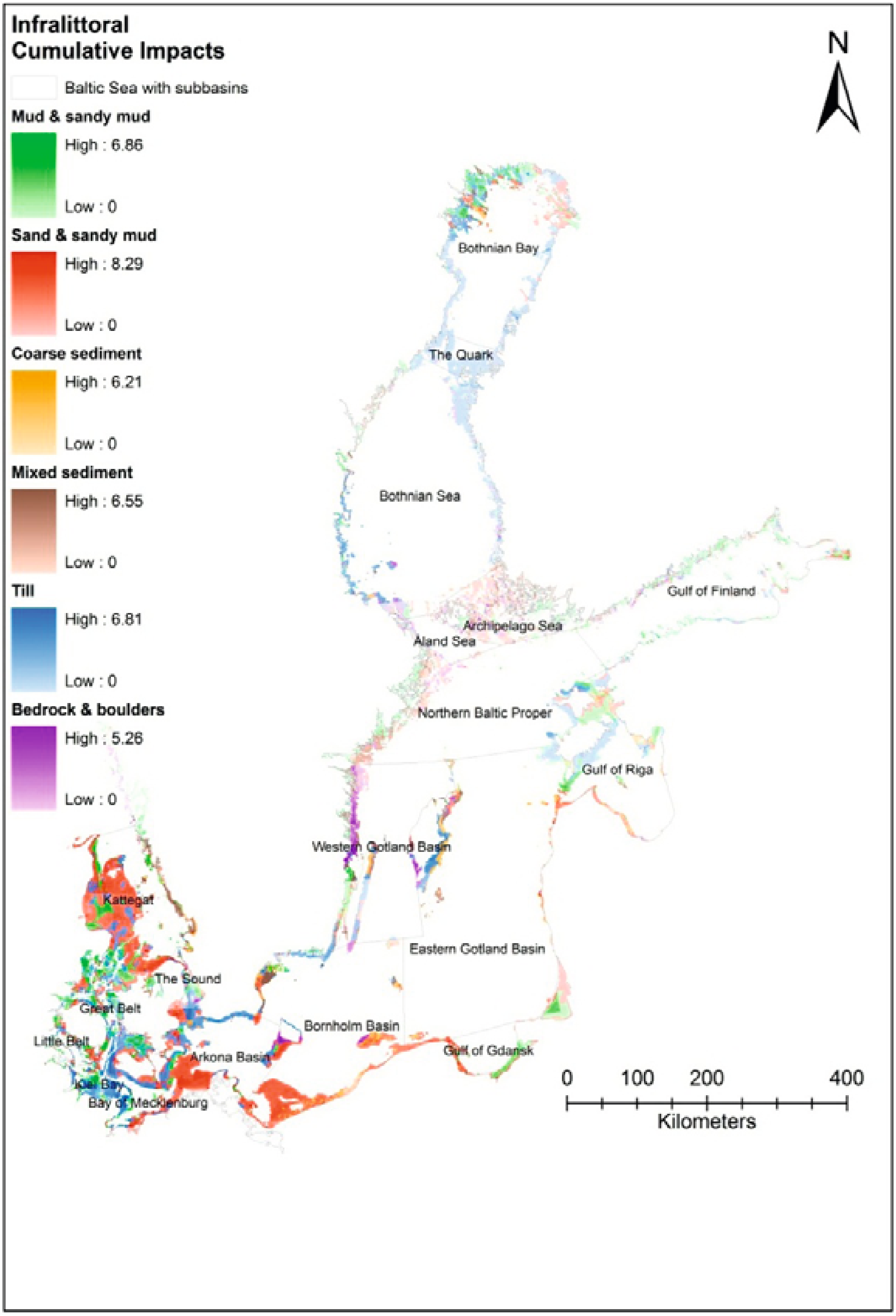

Figure 4.1. *Map showing the cumulative anthropogenic impacts on six selected infralittoral habitats on a Baltic Sea wide scale.*

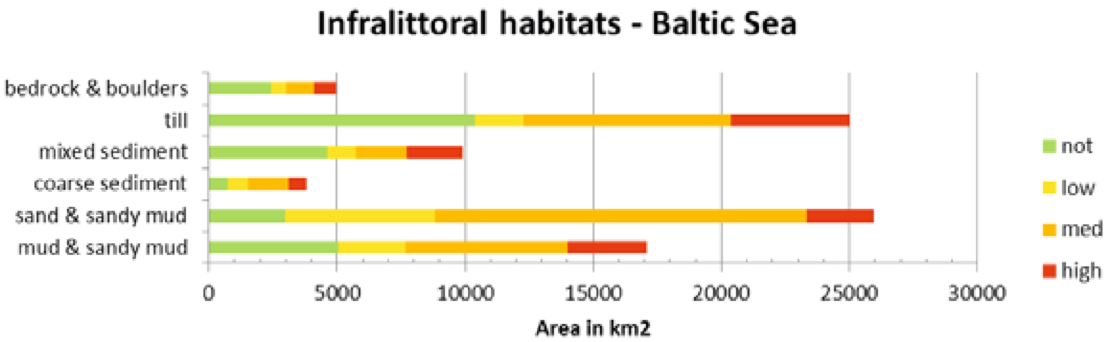

Figure 4.2. *Extent of infralittoral habitats with classified impact categories on a Baltic Sea wide scale.*

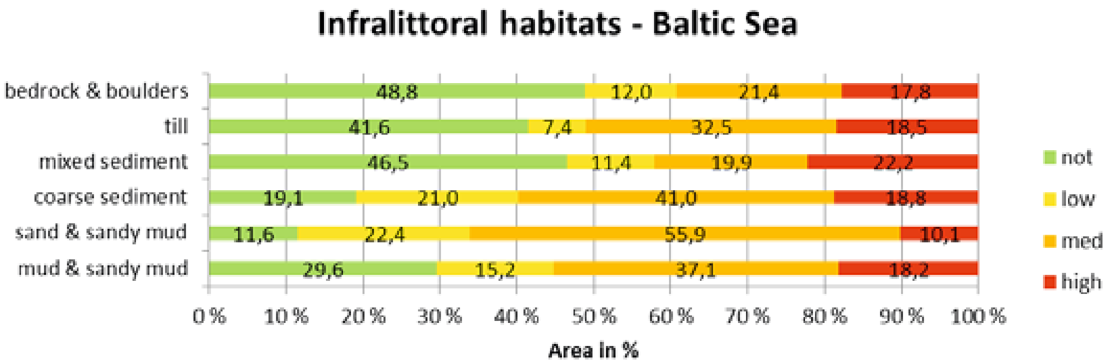

Figure 4.3. *Percentage wise proportion of impact classes for infralittoral habitats on a Baltic Sea wide scale.*

References

HELCOM (2010a) Towards a tool for quantifying anthropogenic pressures and potential impacts on the Baltic Sea marine environment: A background document on the method, data and testing of the Baltic Sea Pressure and Impact Indices. Balt. Sea Environ. Proc. No. 125. Available at: http://www.helcom.fi/stc/files/Publications/Proceedings/bsep125.pdf

HELCOM (2010b) Towards an ecologically coherent network of well-managed Marine Protected Areas – Implementation report on the status and ecological coherence of the HELCOM BSPA network. Balt. Sea Environ. Proc. No. 124B. Available at: http://www.helcom.fi/stc/files/Publications/Proceedings/bsep124B.pdf

Korpinen S, Meski L, Andersen JH & Laamanen M (2012) Human pressures and their potential impact on the Baltic Sea ecosystem. Ecological Indicators 15: 105-114.

4.17. Biomass of copepods

1. Working team: Zooplankton (ZEN) *Authors: Elena Gorokhova and Maiju Lehtiniemi* *Acknowledged persons: Lutz Postel*	
2. Name of candidate indicator: Biomass of copepods	*3. Unit of the candidate indicator* mg/m^3
4. Description of proposed indicator Copepod biomass is calculated using abundance and individual weights. Alternatively (or in addition), contribution of copepod biomass to total mesozooplankton biomass may be used. The indicator reflects composition of zooplankton community and food availability for zooplanktivorous fish. This is a state indicator.	
5. Functional group or habitat type: Zooplankton/plankton	
6. Policy relevance MSFD Descriptor 1 (Biodiversity), Criteria 1.6.2. Relative abundance and or biomass. BSAP Ecological Objective: Viable populalation of species, Target: By 2021 all elements of the marine food webs, to the extent that they are known, occur at natural and robust abundance and diversity.	
7. Use of the indicator in previous assessments Used as preliminary indicators in offshore BEAT cases	
8. Link to anthropogenic pressures Directly impacted by climatic changes, altered predation, introduction of synthetic compounds and invasive species (via predation). Indirectly impacted by eutrophication.	
9. Pressure(s) that the indicator reflect Negative impacts are expected with increased predation pressure and changes in food web structure. Both positive and negative responses can result from changes in thermal regime and salinity.	
10. Spatial considerations The index is strictly area-specific, possibly limited to the open sea areas in the Baltic proper, western Gulf of Finland and southern Baltic. Consistent sampling stations and methods, species identification and biomass calculation methods should be used for estimating and interpreting trends.	
11. Temporal considerations Averaged over growth season. In areas where seasonal monitoring data are available, the assessment could be done on a seasonal basis.	
12. Current monitoring National monitoring programmes, HELCOM (reported variable)	
13. Proposed or perceived target setting approach with a short justification. The applicability and targets should be tested and validated for specific areas. The long term data or data from areas not affected by changes in fish community structure and abundance must be provided by national labs to serve for target setting. A discussion regarding development of common target setting approach and analogous indices for areas where copepods are not dominant species is needed.	

4.18. Biomass of microphageous mesozooplankton

<table>
<tr><td colspan="2">1. Working team: Zooplankton (ZEN)
Author: Elena Gorokhova and Lutz Postel
Acknowledged persons: Maiju Lehtiniemi</td></tr>
<tr><td colspan="2">1. Working team: Zooplankton (ZEN)</td></tr>
<tr><td>2. Name of candidate indicator:
Biomass of microphagous mesozooplankton</td><td>3. Unit of the candidate indicator
mg/m^3</td></tr>
<tr><td colspan="2">4. Description of proposed indicator
Biomass is calculated using abundance of microphagous feeders present in mesozooplankton community and their individual weights. Alternatively (or in addition), contribution of microphagous biomass to total mesozooplankton biomass can be used.
The indicator reflects composition of zooplankton community and availability of small-sized phytoplankton and bacterioplankton, the increase in the latter is commonly observed with increasing eutrophication. It also negatively related to food availability for zooplanktivorous fish.</td></tr>
<tr><td colspan="2">5. Functional group or habitat type:
Zooplankton/plankton</td></tr>
<tr><td colspan="2">6. Policy relevance
MSFD Descriptor 1 (Biodiversity), Criteria 1.6.2. Relative abundance and or biomass.
BSAP Ecological Objective: Viable population of species, Target: By 2021 all
elements of the marine food webs, to the extent that they are known, occur at natural and robust abundance and diversity.</td></tr>
<tr><td colspan="2">7. Use of the indicator in previous assessments
None</td></tr>
<tr><td colspan="2">8. Link to anthropogenic pressures
Directly impacted by (1) fisheries (through predation), (2) changes in thermal regime and salinity, (3) introduction of synthetic compounds and (4) invasive species (via predation)
Indirectly impacted by (1) eutrophication (through changes in food abundance and size spectra), and (2) commercial fisheries (through changes in pelagic food webs),</td></tr>
<tr><td colspan="2">9. Pressure(s) that the indicator reflect
Eutrophication increases abundance and productivity of small-sized phytoplankton and bacterioplankton, which stimulates production and standing stocks of microphagous species. In addition, increased biomass of microphagous species implies decreased food quality and availability for fish.</td></tr>
<tr><td colspan="2">10. Spatial considerations
The index is strictly area-specific, possibly limited to the areas generally dominated by crustacean zooplankton, such as open sea areas in the Baltic proper, western Gulf of Finland and southern Baltic.</td></tr>
<tr><td colspan="2">11. Temporal considerations
Averaged over growth season. In areas where seasonal monitoring data are available, the assessment could be done on a seasonal basis.</td></tr>
<tr><td colspan="2">12. Current monitoring
National monitoring programmes, HELCOM (reported variable).</td></tr>
<tr><td colspan="2">13. Proposed or perceived target setting approach with a short justification.
The applicability and targets should be tested and validated for specific areas. The long term data or data from relatively pristine areas must be provided by national labs to serve for target setting. A discussion regarding development of common target setting approach is needed.</td></tr>
</table>

4.19. Zooplankton species diversity

1. Working team: Zooplankton (ZEN) *Author: Elena Gorokhova* *Acknowledged persons: Maiju Lehtiniemi, Lutz Postel*	
1. Working team: Zooplankton (ZEN)	
2. Name of candidate indicator: Zooplankton species diversity	*3. Unit of the candidate indicator* unitless
4. Description of proposed indicator Using long term species lists for different subregions of the Baltic Sea, area-specific species diversity index is calculated as a ratio between the number of species actually observed in the area and species number registered in the area. The indicator reflects changes in taxonomic diversity in the area and therefore could be used as a general indicator of zooplankton species diversity. Potentially could be also used for indication of bioinvasions, both in planktonic and benthic communities if meroplankton is included in zooplankton community analysis.	
5. Functional group or habitat type: Zooplankton/plankton	
6. Policy relevance MSFD Descriptor 1 (Biodiversity), Criterion 1.6 Habitat condition BSAP Ecological Objective: Viable population of species, Target: By 2021 all elements of the marine food webs, to the extent that they are known, occur at natural and robust abundance and diversity.	
7. Use of the indicator in previous assessments None	
8. Link to anthropogenic pressures Directly impacted by: climatic changes, introduction of synthetic compounds and invasive species.	
9. Pressure(s) that the indicator reflect Directly impacted by: introduction of invasive species, both positively by adding new species and negatively by eliminating native species via predation/competition (very rare cases). Other pressures are (1) changes in thermal regime, pH and salinity (can cause both expansion of species from neighbouring areas and disappearance of existing species from the area, and (2) introduction of synthetic compounds, with possible local effects in areas situated close to municipal and industrial effluents.	
10. Spatial considerations The index is strictly area-specific. Consistent sampling stations and methods, and species identification should be used for estimating and interpreting trends.	
Annual assessment. In areas where seasonal monitoring data are available, the assessment could be done on a seasonal basis.	
12. Current monitoring National monitoring programmes, HELCOM	
13. Proposed or perceived target setting approach with a short justification. The applicability should be tested and validated for specific areas. The GES boundary is 1, i.e. no change in biodiversity. Consequently, ratios <1 correspond to decreased diversity, =1 is no change, >1 means either invasion or colonization from neighboring areas. The long term species list must be provided by national labs.	

4.20. Mean zooplankton size

The indicator was meant to follow long-term changes in the zooplankton size as a response to changes in food web (predation pressure) and eutrophication (hypoxia, altered phytoplankton species composition). Larger zooplankton size indicates a better state of the environment.

No description of the indicator is available, but see the indicator for copepod biomass for some discussion (below).

4.21. Zooplankton-phytoplankton biomass ratio

The indicator was meant to follow long-term changes in the biomass ratio of zooplankton and phytoplankton as a response to changes in food web (predation pressure) and eutrophication (hypoxia, nutrient availability). Bias to zooplankton indicates stronger top-down control and hence a better functioning food web (piscivorous fish controlling planktivorous fish, releasing zooplankton from high predation.

No description of the indicator is available but see the indicator for copepod biomass for some discussion (below).

Description and testing of zooplankton indicators: Introduction

In aquatic ecosystems, changes in species composition and abundance of small, rapidly reproducing organisms, such as plankton, have been considered among the earliest and sensitive ecosystem responses to anthropogenic stress (Schindler 1987). Zooplankton are integral to aquatic productivity, serving as primary consumers of nutrient-driven primary producers, and as prey for fish. Despite their potential as indicators of environmental changes and their fundamental role in the energy transfer and nutrient cycling in aquatic ecosystems, zooplankton assemblages have not been widely used as indicators of ecosystem condition (Stemberger and Lazorchak 1994), and zooplankton is not included as a relevant quality element for the assessment of ecological status within Water Framework Directive. Nevertheless, changes in primary productivity and physical conditions due to eutrophication and warming and the consequent reorganization of zooplankton communities have been documented worldwide, albeit, more often in freshwater than in marine systems. In the Baltic Sea, alterations in fish stocks and regime shifts received a particular attention as driving forces behind changes in zooplankton (Casini et al. 2009).

Here, we consider a possibility of using zooplankton as indicators for eutrophication and fish feeding conditions. With respect to eutrophication, it has been suggested that with increasing nutrient enrichment of water bodies, total zooplankton biomass increases (Hanson and Peters 1984), mean size decreases (Pace 1986), and relative abundance of calanoids generally decrease, while small-bodied cyclopoids, cladocerans, rotifers, copepod nauplii, and ciliates increase (Brook 1969; Pace and Orcutt 1981). With respect to fish feeding conditions, the following properties of zooplankton assemblages are usually associated with good food availability: high absolute or relative abundance of large-bodied copepods and low contribution of small zooplankters.

Description of proposed indicators

Copepod biomass (CB; mg/m^3) –zooplankton-as-food indicator; calculated using abundance and individual weights. Alternatively (or in addition), contribution of copepod biomass to total mesozooplankton biomass (CB%) may be used. This is a state indicator; reflects composition of zooplankton community and food availability for zooplanktivorous fish. In the Baltic Sea, copepods contribute substantially to the diet on zooplanktivorous fish, such as sprat and young herring, and fish body condition and weight-at-age (WAA) have been reported to correlate positively to abundance/biomass of copepods (Cardinale et al. 2002, Rönkkönen et al. 2004). Copepods included here are mostly herbivores, therefore, this indicator would be indirectly impacted by eutrophication (via changes in primary productivity and phytoplankton composition), whereas direct impacts are expected from climatic changes, predation, introduction of synthetic compounds (at point sources) and invasive species (via predation). Both positive and negative responses can result from changes in thermal regime and salinity.

Microphagous mesozooplankton biomass (MMB; mg/m^3) – eutrophication indicator; calculated using abundance of microphagous feeders present in mesozooplankton community and their individual weights. Alternatively (or in addition), contribution of microphagous biomass to total mesozooplankton biomass (MMB%) can be used. The indicator reflects composition of zooplankton community and availability of small-sized phytoplankton and bacterioplankton. The increase in the latter is commonly observed with increasing eutrophication that favors growth of smallest primary producers. This provides a high food supply for grazers capable of efficient filtration of small particles; these grazers are usually also small. MMB and MMB% are also negatively related to food availability for zooplanktivorous fish as increased proportion of microphagous species, at least to some extent, implies a reciprocal trend in large copepods that have low efficiency for filtration of small algae. This indicator is directly impacted by eutrophication (through changes in food abundance and size spectra), changes in thermal regime and salinity, introduction of synthetic compounds (at point sources) and invasive species (via predation). Impacted by commercial fisheries, both indirectly (through changes in pelagic food webs) and directly (via predation by fish larvae).

Policy relevance for zooplankton indicators

MSFD Descriptor 1 (Biodiversity):
– Criterion 1.2 Population size; Indicator 1.2.1 Population abundance and/or biomass, as appropriate;
– Criterion 1.6 Habitat condition; Indicator 1.6.2. Relative abundance and/or biomass;
– Criterion 1.7 Ecosystem structure; Indicator 1.7.1 Composition and relative proportions of ecosystem components (habitats and species)

MSFD Descriptor 4 (Food webs):
– Criterion 4.3 Abundance/distribution of key trophic groups/species; Indicator 4.3.1 Abundance trends of functionally important selected groups/species;

BSAP Ecological Objective: Viable population of species.

Methods used to test the indicators

See provisional guidelines for a specific description of zooplankton indicator testing procedure. The approach is based on examining zooplankton time series in relation to:
– the time series on zooplanktivorous fish growth (WAA, or condition) and fish stocks from relevant ICES subdivision(s) to identify time periods when fish growth and fish stocks were relatively high (e.g., Rahikainen and Stephenson 2004) **when setting GES boundary for CB (CB%)**;
– the existing GES values for water transparency or chlorophyll values (e.g., figs. 2.13 and 2.20 in HELCOM, 2009) **when setting GES boundary for MMB (MMB%)**.

In the example below, the testing was done using coastal zooplankton data from the northern Baltic proper (ICES subdivision 29); the data are from Swedish national monitoring conducted by Stockholm University collected and analyzed according to HELCOM guidelines (HELCOM 1988).

Approach for defining GES

Data analysis and definition of GES boundary

To establish baselines and acceptable variability for specific indicators, trends and reference periods were evaluated using indicator Control Chart (Manley 2001; Guthrie et al. 2005). For CB% and MMB%, mean values and 95% confidence intervals (CI) were used. As these variables are non-normally distributed, standard deviation (SD) was calculated using transformed values and then back-transformed to arrive to the upper and lower 95% limits of CI. In the context of GES for CB (CB%), upper limit is not relevant, whereas lower limit is not relevant for MMB (MMB%). Thus, GES boundary for CB (CB%) is the lower 95% CI limit, and for MMB (MMB%), it is the upper 95% CI limit. These values are set as threshold values which mark the boundary between acceptable and unacceptable conditions, i.e. CB (CB%) should not decline below its GES boundary, whereas MMB (MMB%) should not increase above its GES boundary.

GES boundary for CB%

For the data set analyzed, the most easy interpretable results were obtained with CB%. The GES boundary was estimated to be 73% based on the reference period 1983-1990, when herring and sprat stocks were relatively stable and had high WAA and body condition. This reference period was selected based on the data presented by Rahikainen & Stephenson (2004) and Casini et al. (2006). See *Guidelines* below for details.

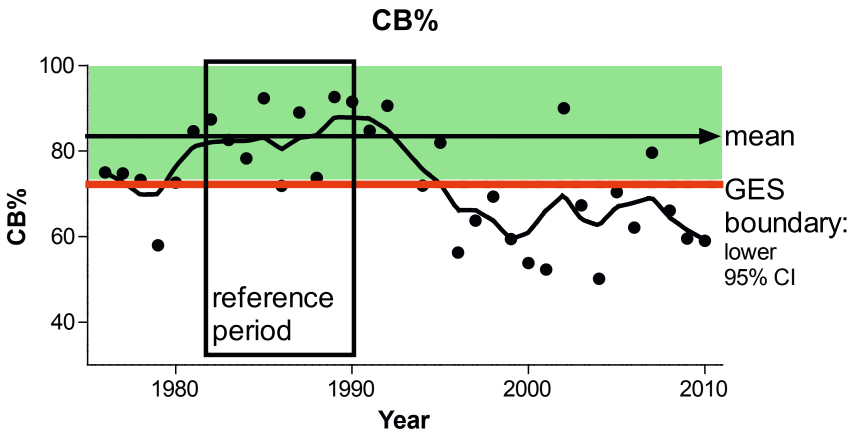

Figure 4.4. *Long-term changes in copepod contribution to the total mesozooplankton biomass (CB%) in the Askö area, northern Baltic proper (1976-2010; average summer values, n=6). According to the trend observed, CB% >73% indicate good feeding conditions for zooplanktivorous fish (green area).*

GES boundary for MMB%

For the data set analyzed, the most easy interpretable results were obtained with MMB%. The GES boundary estimated was 22% based on the reference period 1976-1990, when water transparency complied with GES levels (various HELCOM documents; this period might need refining).

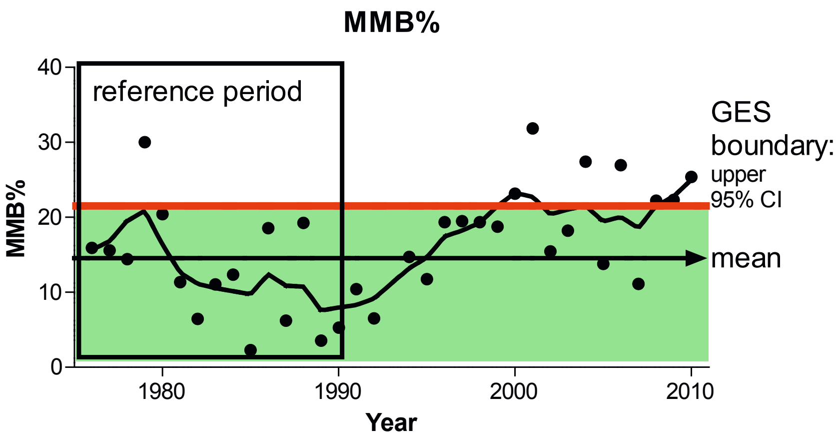

Figure 4.5. *Long-term changes in microphagous zooplankton contribution to the total mesozooplankton biomass (MMB%) in the Askö area, northern Baltic proper (1976-2010; average summer values, n=6). According to the trend observed, MMB% <22% correspond to water quality in GES (green area).*

Notes for the GES boundary

The copepod biomass is an indicator of fish feeding conditions and, accordingly, the reference period is selected on the basis of the fish growth status. As we know, moderate eutrophication is actually beneficial for fish nutrition, and therefore, it is not surprising that highest growth of zooplanktivorous fish coincides with some (but not the highest!) eutrophication (expressed as Chl a or Secchi depth). It would be against the theoretical expectations to observe the best feeding conditions in an oligotrophic system.

The example that is used in this example is for a relatively pristine area (Askö) and in 1980-ties the Secchi and Chl values were in a better status than in the next 20 years. Therefore, the highest copepod biomass did not coincide with the heavy eutrophication. One has to remember that all reference periods should be area-specific.

Existing monitoring data

Zooplankton data required for this analysis are collected on a regular basis within national and HELCOM monitoring programs. Laboratories that follow HELCOM methodology for sampling and sample analysis, should possess all data necessary for indicator development and use. Depending on the sampling frequency, a specific period for zooplankton stocks should be considered and used consistently; this period may vary between different areas/countries/laboratories, because sampling frequency is not uniform. Unfortunately, in some areas, sampling coverage is low and not all sea areas are equally well represented (see also *Weaknesses and Concerns*).

Data interpretation

A dialogue with experts (and data holders) responsible for establishing GES values for eutrophication and fish stocks would greatly facilitate selection of the reference periods for zooplankton indicators and estimating GES boundaries.

Although the empirical relationships between zooplankton abundance and eutrophication status are common in scientific literature, the underlying mechanisms are not well understood. For zooplankton indices to have relevance to management, it is necessary to postulate and test (through research and/or data analysis) hypotheses that explain the response of zooplankton to water quality. Similarly, it is necessary to identify fish species individually or feeding groups for which the zooplankton-as-food indicator is relevant. It is also necessary to include other aspects of habitat quality, particularly for coastal fish, and the zooplankton-as-food indicator can serve as one element in a more comprehensive index of habitat condition.

Zooplankton communities include herbivores, predators and omnivores, i.e. organisms with different trophic roles in the food web, but from an ecological viewpoint, all of them are intermediate players, i.e., subject to bottom-up pressures as well as top-down demand. Therefore, zooplankton information is most useful within the framework of a broader, multi-trophic-level monitoring providing indicators of ecosystem functioning (i.e., MSFD Descriptor 4).

Weaknesses and concerns

Presently, we do not have a good overview on south-north and east-west variability in zooplankton community structure, population stocks and seasonal fluctuations in the Baltic Sea. This complicates development of indicators applicable in different areas and may further hamper between-area comparisons;

Due to difference in zooplankton compositions between the Baltic Sea areas, some modifications may be required for taxa composition in each specific indicator;

Data are generally noisy, owing to multiple factors affecting zooplankton growth and mortality;

Different sampling/analysis methods might have been used in different labs in previous years resulting in either over- or underestimations of specific zooplankton groups/species. If these methodological difference are not corrected for, the long-term data might be of little use;

Poor spatial coverage, and, in some areas, poor frequency of sampling;

Biomass calculation is not based on actual measurements of individual weight/size, but a fixed value, whereas seasonal and spatial variability of individual body size is well recognized in all zooplankton groups;

Strong confounding effects of salinity and temperature that may obscure relationships between the indicators and relevant pressures.

Provisional guidelines for testing zooplankton-based indicators proposed within CORESET Biodiversity group

The indicators proposed were:

Zooplankton Species Diversity (ZSD);
Copepod Biomass (CB, CB%);
Microphagous mesozooplankton biomass (MMB, MMB%);

Additional indicators that appeared as promising during testing using datasets from a coastal area in the northern Baltic proper and discussions at Coreset Biodiversity meeting (15-17/6, Riga) were:

Mean zooplankter size (MeanSize) calculated as a ratio between total mesozooplankton biomass and abundance. Note: only species/groups that are included consistently in zooplankton assessment should be used. The rationale is that as eutrophication favors small-bodied forms that are more competitive in feeding on nano- and picoplankton.

Zooplankton/phytoplankton ratio; this is a food web indicator of trophic transfer efficiency. The rationale is that higher grazing efficiency implies less losses in the food webs, less energy and nutrients going via microbial loops, and, consequently, more energy transferred to the higher trophic levels, i.e. fish. Again, only species/groups that are included consistently in phyto- and zooplankton assessment should be used.

To start on the evaluating procedure, obtain the following **background information**:

Step 1: Relevant for your area (=your ICES subdivision) **time series on zooplanktivorous fish growth** (weight-at-age, commonly abbreviated as WAA, or condition) and fish stocks. This is needed to identify a time period when fish growth was high and fish stocks were relatively high (for example, Rahikainen M, Stephenson RL 2004. *Consequences of growth variation in northern Baltic herring for assessment and management*. ICES Journal of Marine Science 61: 338-350). Consult your fish-colleagues for data location and interpretation if necessary. The time period should span at least 5-7 consecutive years;

Step 2: Relevant for your area **time series for water transparency or chlorophyll values** (see, for example, figs. 2.13 and 2.20 in HELCOM, 2009. *Eutrophication in the Baltic Sea – An integrated thematic assessment of the effects of nutrient enrichment and eutrophication in the Baltic Sea region*. Balt. Sea Environ. Proc. No. 115B, available at www.helcom.fi for download; and/or other relevant data or assessments). This is needed to identify a reference period when eutrophication levels were below or at currently accepted targets. Consult your colleagues involved in Water Framework Directive activities for data location and interpretation if necessary. The time period should span at least 5-7 consecutive years.

Step 3: Prepare **a list of species/groups routinely identified in zooplankton samples** (if unavoidable, use groups that are pooled in a consistent manner) to calculate ZSD values as a ratio between the number of species observed in a given year (or season if the assessment is done on a seasonal basis) and the total number of species ever observed in routine zooplankton monitoring in this area. This is a proxy for species richness.

Step 4: Prepare available **time series** or data published in various reports for each of the main indicators (ZSD, CB, CB%, MMB and MMB%) and additional indicators (MeanSize, Zooplankton/Phytoplankton ratio):

a) Pay particular attention to the data availability for the periods of the high fish growth conditions and acceptable eutrophication levels indentified from the time series for these background data;
b) Note that both absolute biomass (mg/m3; CB and MMB) and relative (proportion or percentage of the indicator group in the total mesozooplankton biomass; CB% and MMB%) will be needed;
c) When calculating CB and CB%, include all copepodites stages of copepods;
d) When calculating MMB and MMB%, include only holoplankton (i.e., no benthic larvae); most relevant being the following species/groups:
   i rotifers,
   ii appendicularians,
   iii small (<20 mm) ctenophores,
   iv small cladocerans feeding mostly on nanoplankton (e.g., *Bosmina maritima*, *Penilia avirostris*),
   v pelagic harpacticoids,
   vi Nauplii of all copepod species and tintinnids can be included if assessment of their abundance follows the same methodological guidelines for the entire dataset.

Step 5: To establish baselines and acceptable variability for specific indicators, evaluate trends using **indicator Control Chart** (see presentation at Coreset BD meeting by Johan Wikner, *Status Assessment Control Diagram* attached; consult also various statistical books, e.g., Manley, F.J.M. 2001. Statistics for environmental science and management, 1th ed. Chapman and Hall/CRC, New York: McGraw-Hill Book Company). For CB% and MMB%, we used mean and 95% confidence interval (CI). Please, remember that percentages are not normally distributed by definition and use an appropriate transformation to calculate CI.

Example to illustrate the workflow: Testing **CB% as an indicator of fish feeding conditions** in a coastal area of the northern Baltic proper
a) Based on Fig. 2 in Rahikainen & Stephenson (2004) and Casini et al. (2006), **the reference period** (=high growth in zooplanktivorous fish) in this area (ICES subdivision 29) occurred in 1983-1990;
b) **Time trend for CB%**, with calculated mean for the reference period and target value set at lower 95% CI limit (Fig. 1). Observe that for values expressed as percentage, SD is calculated using arcsin sqrt transformed values and then back-transformed to represent upper and lower CI limits;
c) **The GES boundary value for CB%** it is set as a threshold value, which mark the boundary between acceptable and unacceptable conditions, i.e. CB (CB%) should not decline below its GES boundary value.
d) **Evaluation of the indicator trend:** Since 1995, CB% is generally below its target (i.e. GES boundary) value, indicating poor nutritional conditions for zooplanktivorous fish. The decline from the baseline (mean value) started in early 1990-ties and particularly low values were observed in the late 1990-ties - early 2000-ties, which correspond well with fish condition index in this area (Casini et al. 2006). Thus, we conclude that indicator performs reasonably well.

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4.22. Phytoplankton diversity

1. Working team: Pelagic (phytoplankton) *Authors*: Vivi Fleming-Lehtinen, Laura Uusitalo and Heidi Hällfors *Acknowledged persons*: Seija Hällfors, Andres Jaanus and Lauri London	
2. Name of candidate indicator Phytoplankton diversity	*3. Unit of the candidate indicator* unitless diversity index
4. Description of proposed indicator The indicator describes phytoplankton species diversity by an applied Shannon diversity index. It shows changes in the phytoplankton species composition as a result of the eutrophication of the marine environment. Initial testing has been conducted concentrating on dominant phytoplankton species and their diversity by an applied Shannon's index based on the most abundant species that together make up >95% of the total biomass. Since the occurrence of rare species in the samples is random and uncertain, excluding these species from the indicator enhances its robustness. The preliminary testing of the indicators show promise but needs further elaboration. SYKE continues to develop the indicator in the LIFE+ MARMONI project. Indicator depends on monitoring on the number and abundance of phytoplankton species on species level or as close to species level as possible.	
5. Functional group or habitat type Pelagic habitats	
6. Policy relevance Descriptor 1, criterion 1.6 Habitat condition.	
7. Use of the indicator in previous assessments Not	
8. Link to anthropogenic pressures The linkage to eutrophication has been established. Eutrophication alters the abiotic condition of the pelagic habitat (bottom-up driven changes).	
9. Pressure(s) that the indicator reflect Inputs of nitrogen and phosphorus (all sources and forms).	
10. Spatial considerations The indicator can be used in the sub-basin level, following changes in large scale.	
11. Temporal considerations Requires weekly/biweekly monitoring and is therefore restricted to ship-of-opportunity samples.	
12. Current monitoring Monitored along a few shipping lanes in the Baltic Sea.	
13. Proposed or perceived target setting approach with a short justification. The GES boundary should be set on the basis of historical data and diversity results in low-impacted areas.	

4.23. Seasonal succession of phytoplankton groups

1. Working team: Pelagic (phytoplankton) *Author*: Andres Jaanus	
2. Name of candidate indicator Seasonal succession of phytoplankton groups	*3. Unit of the candidate indicator* unitless index
4. Description of proposed indicator The seasonal succession of phytoplankton functional groups is an index which has previously been presented by Devlin et al. (2007) for UK waters. The method is based on firstly defining reference growth envelopes for functional groups of interest using long-term data or un-impacted sites. Present state is assessed by comparing the present seasonal distribution of each functional group (Phytoplankton counts are averaged over months, and monthly mean and standard deviations calculated for each functional group) to the reference by use of a normalized score (Z score). Monthly Z score establishes comparable seasonal distributions for each functional group for a sampling year. A positive Z score indicates that the observation is greater than the mean and a negative score indicates the observation is less than the mean. The index has been tested for diatoms and dinoflagellates in the Baltic Sea area. Further testing is needed to evaluate the appropriateness of the indicator.	
5. Functional group or habitat type Pelagic habitats	
6. Policy relevance Descriptor 1, criterion 1.6 Habitat condition.	
7. Use of the indicator in previous assessments Not	
8. Link to anthropogenic pressures The linkage to eutrophication has been established. Eutrophication alters the abiotic condition of the pelagic habitat (bottom-up driven changes).	
9. Pressure(s) that the indicator reflect Inputs of nitrogen and phosphorus (all sources and forms).	
10. Spatial considerations The indicator can be used in the sub-basin level, following changes in large scale.	
11. Temporal considerations Requires weekly/biweekly monitoring and is therefore restricted to ship-of-opportunity samples.	
12. Current monitoring Monitored along a few shipping lanes in the Baltic Sea.	
13. Proposed or perceived target setting approach with a short justification. Reference growth envelopes should be established from long-term data or data from low-impacted sites.	

4.24. Alkylphenols

1. Name of candidate indicator Alkylphenols: nonylphenol and octyphenol	*2. Preferred matrix* Herring and perch muscle, cod liver (lipid + fresh + LW%). Bivalve soft tissue (dry + fresh + DW%).
3. Description of proposed indicator Nonylphenol and octylphenol are toxic and possibly bioaccumulating in mussels. According to the thematic assessment of hazardous substances (HELCOM 2010), nonylphenol exceeded EQS only in the southern parts of the Baltic Sea and only in sediment samples. Octylphenol exceeded the EQS also in the Northern Baltic Proper, but only in the sediments. The CORESET expert group for hazardous substances indicators noticed the lack of high concentrations in biota and noticed that more information should be compiled before alkylphenols could be proposed as core indicators.	

4. Policy relevance Nonylphenol and octylphenol are both listed under the HELCOM BSAP and under the EU Priority Substances. Descriptor 8, criterion 8.1 of the MSFD. BSAP, ecological objective *Concentrations of hazardous substances close to natural levels.*
5. Use of the indicator in previous assessments Thematic assessment of hazardous substances (HELCOM 2010).
6. Current monitoring Monitored by Denmark, Germany and Sweden.
7. Proposed or perceived target setting approach with a short justification. Environmental Quality Standards.

4.25. Intersex or vitellogenin induction in male fish

The candidate indicator for measuring the estrogenic activity and its effects is developed by the BONUS+ project BEAST. This report lacks a proper documentation for the indicator.

Table 4.4. Overview of the use, monitoring, cause-effect relationships, assessment criteria and available guidance.

Used by OSPAR CEMP or pre-CEMP MEDPOL	1) Used in national monitoring programmes in Baltic Sea countries 2) Research monitoring data available	Studied in large Baltic Sea research projects	Biological Indication (BI) Cause/effect (C/E) relationship	Availability of AC specific for the Baltic Sea (for detailes see Table 1	Monitoring Guidelines, other important info QA Method costs (High-Medium-Low)
	2) DE, SE, DK	BEEP BEAST BALCOFISH	BI – exposure to estrogenic contaminants	BAC, EAC under development (BEAST)	ICES TIMES 31 QA–NO Low costs

Abbreviations

- BAC – Background Assessment Criteria
- EAC – Environmental Assessment Criteria (concentrations above which there is concern that negative effects might be observed in marine organisms. EAC describes the direct linkage to adverse health effects of the individuals)
- AC – Numerical Assessment Criteria
- QA – Quality Assurance measures
- CEMP – guidelines, quality assurance and assessment tools are in place - monitoring of the component is mandatory for OSPAR contracting parties
- pre-CEMP – agreed to be included as components of the CEMP, guidelines, QA tools and/or assessment tools are currently not all in place. Monitoring of the components is voluntary

Country codes: DK-Denmark, EE-Estonia, FI-Finland, DE-Germany, LV-Latvia, LT-Lithuania, PL-Poland, RU-Russia, SE-Sweden

4.26. Acetylcholinesterase (AChE)

General information

General properties

The analysis of acetylcholinesterase (AChE; EC 3.1.1.7) activity and its inhibition in marine organisms has been shown to be a highly suitable method for assessing exposure to neurotoxic contaminants in aquatic environments. AChE activity method is applicable to a wide range of species and has the advantage of detecting and quantifying exposure to neurotoxic substances without a detailed knowledge of the contaminants present. AChE activity is a typical biomarker that can be used in *in vitro* bioassays and field applications.

Main impacts on the environment and human health

AChE has traditionally been used as a specific biomarker of exposure to organophosphate and carbamate pesticides. More recently, its responsiveness has been demonstrated to various other groups of chemicals present in the marine environment including heavy metals, detergents and hydrocarbons. Its usefulness as a general indicator of pollution stress in mussels from the Baltic Sea has been shown within the EU-BEEP project. AChE inhibition mostly agreed well with the studied pollution gradients, especially in mussels. Seasonal differences in activity were notable in flounder, eelpout and mussels, possibly resulting from variations in the occurrence of affecting substances (e.g., pesticides from river input or run-off from agricultural sources) during the year. Additional field studies and laboratory experiments showed that AChE in Baltic mussels is influenced by temperature and salinity, while also salinity has an effect on the uptake (and therefore on toxicity) of substances (Lehtonen et al. 2006).

Status of a compound on international priority lists and other policy relevance

OSPAR pre-CEMP, MED POL Phase IV (2° Tier).

ICES SGIMC has recommended AChE in molluscs as biomarker to be included into the OSPAR Coordinated Environmental Monitoring Programme (pre-CEMP).

AChE has been adopted by UNEP as part of the second tier of techniques for assessing harmful impact in the Mediterranean Pollution programme (MEDPOL Phase IV).

GES boundaries and matrix

The existence of extremely low thresholds for induction of inhibitory effects on AChE suggests that detection is possible after exposure to low concentrations of insecticides (0.1 to 1 µgl-1; ICES 2010).

Standardisation of the sampling strategy and regular intercalibration exercises on specific organisms are still necessary before using AChE in routine pollution monitoring.

No formal quality assurance programmes are currently run within the BEQUALM programme but one major intercalibration exercise was carried out during the BEEP project.

Baseline levels of AChE in different marine species have been estimated from results derived from in the Atlantic Ocean, the Mediterranean and the Baltic Sea (**Table 4.6**; ICES 2010) and ongoing studies within Bonus+ BEAST.

Generally it has been accepted that 20% reduction in AChE activity in fish and invertebrates indicates exposure to neurotoxic compounds. Depression in AChE activity more than 20% up to 50% indicates sublethal impact. In the field, several species have baseline AChE activities within the same order of magnitude among different studies/measurements (**Table 4.6**; ICES 2010,). However, differences between sea areas and seasons are obvious, with values activity values in *Mytilus* spp. varying from 25 to 54 nmol min mg protein. (ICES 2010). Therefore Baltic Sea specific data have been collected which are presently assessed as part of Bonus+ BEAST activities.

Preferred matrix

Fish muscle, bivalve gills

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

Suitability of AChE in bivlaves and fish has been tested during the EU-BEEP project; AChE is also a core biomarker within Bonus+ BEAST field activities in different Baltic Sea sub regions.

Gaps in the monitoring of the compound

Spatial gaps for the Baltic Sea have been identified which are presently filled in within the Bonus+ BEAST project.

Present status assessments

Known temporal trends

Under investigation in Bonus+ BEAST, results will be available during 2011.

Spatial gradients

Under investigation in Bonus+ BEAST, results will be available during 2011.

Recommendation

AChE should become a CORE indicator to be used in a future integrated chemical and biological effects monitoring and assessment programme in the HELCOM region. AChE is a good indicator for direct neurotoxic effects caused, e.g., by organophosphate and carbamate pesticides (specific exposure) also in brackish-water systems. It seems also to indicate general toxicity, often following a similar pattern to LMS (Lehtonen et al., 2006). When applying AChE activity as a biomarker of contaminant effects in the Baltic Sea, local abiotic factors and seasonal differences have to be considered.

Results obtained within Bonus+ BEAST will further support suitability of AChE and also provide data needed to develop appropriate assessment criteria for the Baltic Sea.

AChE inhibition can be used for integrated biomarker approach.

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Table 4.5. Overview of the use, monitoring, cause-effect relationships, assessment criteria and available guidance.					
Used by OSPAR, CEMP or pre-CEMP or MEDPOL	1) Used in national monitoring programmes in Baltic Sea countries 2) Research monitoring data available	Studied in large Baltic Sea research projects	Biological Indication (BI) Cause/effect (C/E) relationship	Availability of AC specific for the Baltic Sea (for details see *Table 1*	Monitoring Guidelines, other important info Method costs (High-Medium-Low)
MEDPOL (2° Tier)	2) FI, LV, DE, SE	BEEP BEAST	BI – neurotoxic effects, early warning C/E – YES	BAC and EAC	ICES SGIMC 2010 QA –YES Low costs

Table 4.6. Numerical Assessment Criteria (AC) for Baltic Sea organisms to assess biological effects. Values for Background Assessment Criteria (BAC) and Environmental Assessment Criteria (EAC) are given where available or relevant. Full details of AC and how they have been derived can be found in the ICES SGIMC 2010, SGIMC 2011 and WKIMC 2009 reports on the ICES website and in the SGEH Background Documents for the individual biological effects methods. NOTE: All missing information will soon be available from the output of the ongoing projects mentioned.			
Biological effect method (unit, other information)	*Target species/ tissue/endpoint*	BAC	EAC
Acetylcholinesterase activity (AChE) *[nmol/min/mg protein]*	*Mytilus* spp. gill tissue	under development (Finnish and and German seasonal data;, cf. OSPAR temperature corrected values	under development (Finnish and German seasonal data;, cf. OSPAR temperature corrected values)
	Macoma balthica muscle tissue (foot)	under development (Finnish seasonal data)	0.7 x BAC
	Herring muscle tissue	under development (BEAST data) Seasonal BAC values	0.7 x BAC (cf. OSPAR AC) Seasonal BAC values
	Flounder muscle tissue	under development (BEEP, BEAST and Polish data) Seasonal BAC values	0.7 x BAC (cf. OSPAR AC) Seasonal BAC values
	Eelpout muscle tissue	under development (BEEP & BEAST data)	0.7 x BAC (cf. OSPAR AC) Seasonal BAC values
	Perch muscle tissue	under development (BEEP data) Seasonal BAC values	0.7 x BAC (cf. OSPAR AC) Seasonal BAC values

4.27. EROD/CYP1A induction

The candidate indicator for measuring the Ethoxyresorufin-O-deethylase (EROD) activity is developed by the BONUS+ project BEAST (particularly Henryka Dabrowska). This report lacks a proper documentation for the indicator, but information provided by the BEAST project has been compiled.

Table 4.7. Overview of the use, monitoring, cause-effect relationships, assessment criteria and available guidance. Country codes: DK-Denmark, EE-Estonia, FI-Finland, DE-Germany, LV-Latvia, LT-Lithuania, PL-Poland, RU-Russia, SE-Sweden.

Used by OSPAR, CEMP or pre-CEMP or MEDPOL	**1) Used in national monitoring programmes in Baltic Sea countries** **2) Research monitoring data available**	**Studied in large Baltic Sea research projects**	**Biological Indication (BI)** **Cause/ effect (C/E) relationship**	**Availability of AC specific for the Baltic Sea (for detailes see *Table 2***	**Monitoring Guidelines, other important info** **Method costs (High-Medium-Low)**
pre-CEMP	1) DK, SE, 2) PL, FI, Pl, EE	BEEP BEAST	BI – to Ah-receptor active chemicals such as PAH, planar PCB, and dioxins	BAC, under development (BEAST) EAC –not yet available	ICES TIMES 13 and 23 QA-YES Low costs

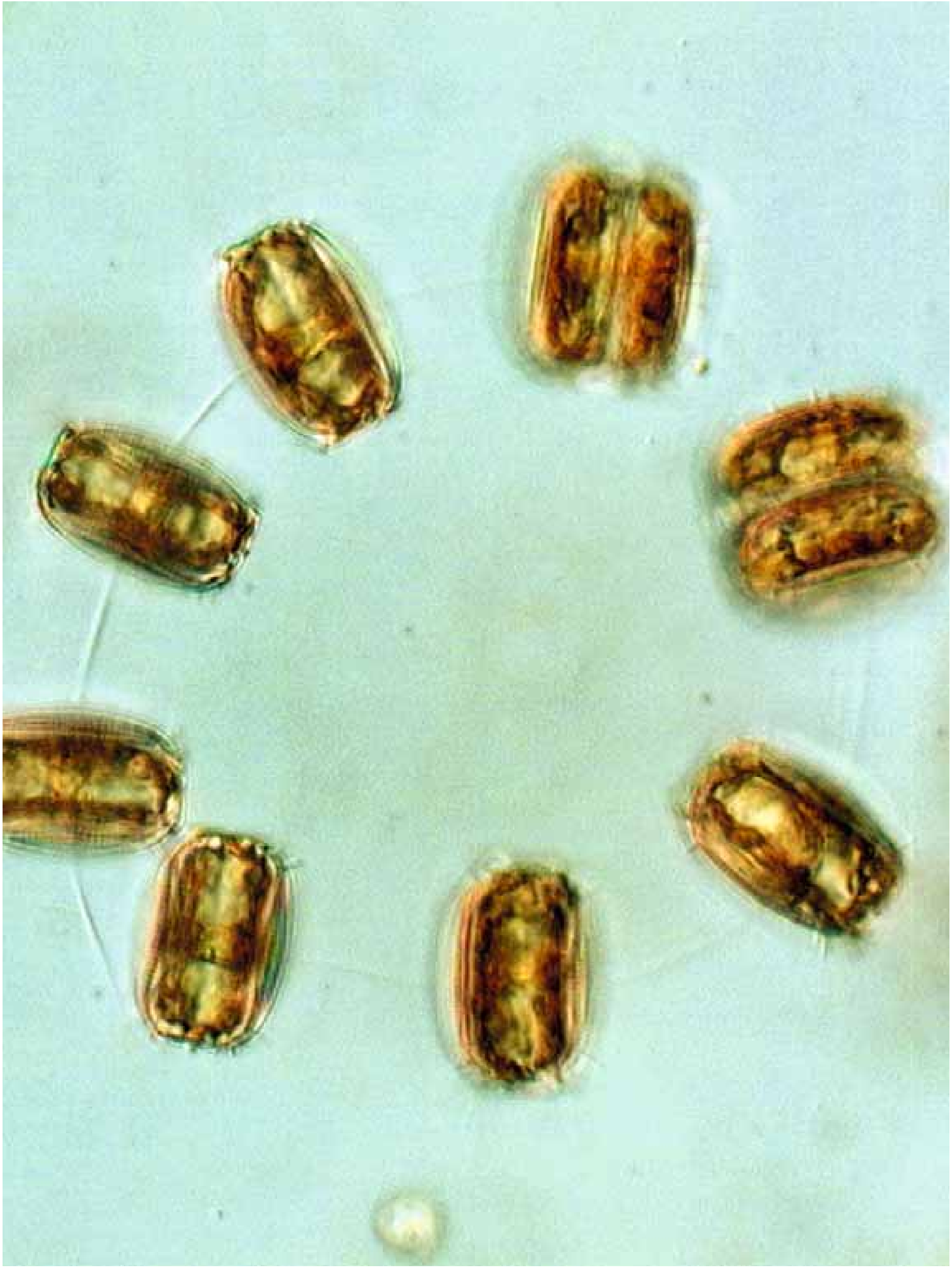
Abbreviations

– BAC – Background Assessment Criteria
– EAC – Environmental Assessment Criteria (concentrations above which there is concern that negative effects might be observed in marine organisms. EAC describes the direct linkage to adverse health effects of the individuals)
– AC – Numerical Assessment Criteria
– QA – Quality Assurance measures
– CEMP – guidelines, quality assurance and assessment tools are in place - monitoring of the component is mandatory for OSPAR contracting parties
– pre-CEMP – agreed to be included as components of the CEMP, guidelines, QA tools and/or assessment tools are currently not all in place. Monitoring of the components is voluntary

Table 4.8. Numerical Assessment Criteria (AC) for Baltic Sea organisms to assess biological effects. Values for Background Assessment Criteria (BAC) and Environmental Assessment Criteria (EAC) are given where available or relevant. Full details of AC and how they have been derived can be found in the ICES SGIMC 2010, SGIMC 2011 and WKIMC 2009 reports on the ICES website and in the SGEH Background Documents for the individual biological effects methods. NOTE: All missing information will soon be available from the output of the ongoing projects mentioned.

Biological effect method (unit, other information)	Target species/ tissue/endpoint	BAC	EAC
Ethoxyresorufin-*O*-deethylase activity (EROD) *[pmol/min/mg protein]* pmol/min/ mg S9 protein *pmol/min/ mg microsomal protein	Flounder (male)	24	not yet available
	Eelpout	10 (Belt Sea data)	not yet available
	Herring	under development BEAST	not yet available

5. Supplementary indicators for environmental assessments



The HELCOM CORESET project identified a number of supplementary indicators, which were not included in the core set, because they either did not fulfil the HELCOM principles for core indicators or they were considered as redundant. Some supplementary indicators can fulfil some of the strict requirements of the core indicators, including the requirement for a quantitative GES boundary, while others can be more descriptive indicators, for example, describing temporal development of parameters that are not linked or weakly linked to anthropogenic pressures. Nevertheless, the supplementary indicators provide valuable information for environmental assessments. Their role is to support the core indicators and provide information of causative factors behind status assessments.

5.1. Summary of all supplementary indicators

The supplementary indicators are listed in the summary table below with a short description or a web link to HELCOM Indicator Fact Sheets. Indicators without a web link are further described in this chapter below.

Table 5.1. Supplementary indicators. Web links to HELCOM Indicator Facts Sheets have been provided, if available.

Supplementary biodiversity indicators	**Objective of the indicator**
Population Development of Sandwich Tern	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/SandwichTern/
Population Development of Great Cormorant	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/Cormorant/
Population Development of White-tailed Sea Eagle	http://www.helcom.fi/BSAP_assessment/ifs/ifs2009/en_GB/White-tailedSeaEagle/
Decline of the harbour porpoise (*Phocoena phocoena*) in the southwestern Baltic Sea	http://www.helcom.fi/BSAP_assessment/ifs/ifs2009/en_GB/HarbourPorpoise/
The abundance of comb jellies in the northern Baltic Sea	http://www.helcom.fi/BSAP_assessment/ifs/ifs2009/en_GB/CombJellies/
Ecosystem regime state in the Baltic Proper, Gulf of Riga, Gulf of Finland, and the Bothnian Sea	http://www.helcom.fi/BSAP_assessment/ifs/archive/ifs2007/en_GB/ecoregime/
Ratio of diatoms and dinoflagellates	Describes the change in taxonomic group composition, presumably caused by eutrophication. Not applicable for the entire sea area. *See text below.*
Ratio of autotrophic and heterotrophic organisms	Describes a change in functional group composition and energy flow in the food web. Not properly tested. *See text below.*
Intensity and areal coverage of cyanobacterial blooms	Describes effects of phosphorus inputs and internal loading (http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/Cyanobacterial_blooms/, http://www.helcom.fi/BSAP_assessment/ifs/archive/ifs2008/en_GB/CyanobacteriaBloomIndex/)
Abundance and distribution of non-indigenous invasive species	Presents the distribution, abundance and temporal trends of selected invasive non-indigenous species in the assessment units.
Biopollution index	Minor impact from newly arrived species. *See text below.*

Table 5.1. (continues)	
Supplementary indicators for the status of hazardous substances	**Objective of the indicator**
Organochlorine pesticides	Includes DDTs, HCB and HCHs (incl. lindane). **Separate description below.**
Copper in biota, sediment and water	Concentrations of Cu have increased in some areas of the Baltic Sea (see HELCOM 2010). Monitoring is done by several HELCOM Contracting States.
Zinc in biota, sediment and water	Concentrations of Zn have increased in some areas of the Baltic Sea (see HELCOM 2010). Monitoring is done by several HELCOM Contracting States.

Table 5.1. (continues)	
Supplementary environment indicators	**Objective of the indicator**
Surface water salinity	Describes environmental conditions caused by climatic variability.
Near bottom oxygen conditions	Describes condition of the near-bottom habitats caused by climatic variability and nutrient inputs.
Sea water acidification	Describes a temporal change in sea water pH. To be published as Indicator Fact Sheet in 2011.
The ice season 2009-2010	Describes the extent of sea ice and reflects also a potential threat for ice-breeding seals: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/iceseason/
Total and regional Runoff to the Baltic Sea	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/Runoff/
Water Exchange between the Baltic Sea and the North Sea, and conditions in the Deep Basins	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/WaterExchange/
Hydrography and Oxygen in the Deep Basins	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/HydrographyOxygenDeepBasins/
Development of Sea Surface Temperature in the Baltic Sea in 2009	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/SeaSurfaceTemperature/
Wave climate in the Baltic Sea 2009	http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/waveclimate2009/
Bacterioplankton growth	http://www.helcom.fi/BSAP_assessment/ifs/ifs2011/en_GB/bacterioplankton/

Table 5.1. (continues)	
Supplementary pressure indicators	**Objective of the indicator**
Nitrogen emissions to the air in the Baltic Sea area	Describe anthropogenic pressures for all biota in the form of eutrophication: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/NitrogenEmissionsAir/
Emissions from the Baltic Sea shipping in 2009	Describe anthropogenic pressures for all biota in the form of contamination and eutrophication: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/ShipEmissions/
Atmospheric nitrogen depositions to the Baltic Sea during 1995-2008	Describe anthropogenic pressures for all biota in the form of eutrophication: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/n_deposition/
Spatial distribution of the winter nutrient pool	Describe anthropogenic pressures for all biota in the form of eutrophication.http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/WinterNutrientPool/
Waterborne inputs of heavy metals to the Baltic Sea	Describe anthropogenic pressures for all biota in the form of contamination: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/waterborne_hm/
Atmospheric deposition of heavy metals on the Baltic Sea	Describe anthropogenic pressures for all biota in the form of contamination: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/hm_deposition/
Atmospheric deposition of PCDD/Fs on the Baltic Sea	Describe anthropogenic pressures for all biota in the form of contamination:http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/pcddf_deposition/
Shipping	Describes the extent of underwater noise, disturbance for sea birds, vector for non-indigenous species, off-shore wastewater and, in shallow areas, seabed disturbance.
Illegal discharges of oil in the Baltic Sea during 2009	Describe anthropogenic pressures for all biota, particularly seabirds, in the form of contamination: http://www.helcom.fi/BSAP_assessment/ifs/ifs2010/en_GB/illegaldischarges/

5.2. Ratio of diatoms and dinoflagellates

It has been widely suggested that the ratio of diatoms and dinoflagellates indicates environmental change in marine ecosystems. Most often changes in that ratio have been contributed to eutrophication or climatic changes. The same factors have been suggested to have caused the increase of dinoflagellates in the Baltic Sea spring bloom over the last decades, both in the northern Baltic Sea (Jaanus et al. 2006, Kremp et al. 2008, Olli & Trunov 2010), as well as in the central and southern parts (Wasmund & Uhlig 2003, Alheit et al. 2005). Klais et al. (2011) however concluded that on the time scale of previous four decades there has been no visible Baltic Sea wide trend in the ratio, but that the role of dinoflagellates has increased in the Gulf of Finland and the Gulf of Bothnia, where also eutrophication process has taken place at the same time.

At the moment, the indicator for the ratio of diatoms and dinoflagellates is seen by the CORESET project as a supplementary indicator and its applicability to indicate anthropogenic eutrophication in different areas of the region should be tested.

References

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Olli K, Trunov K (2010) Abundance and distribution of vernal bloom dinoflagellate cysts in the Gulf of Finland and Gulf of Riga (the Baltic Sea). Deep-Sea Research II 57: 235–242.

Wasmund N, Uhlig S (2003) Phytoplankton trends in the Baltic Sea. ICES Journal of Marine Science 60: 177–186.

5.3. Ratio of aututrophic and heterotrophic organisms

This indicator is based on the same rationale than the previous one: Increasing eutrophication reduces the ratio of autotrophic to heterotrophic processes. This indicator requires more consideration.

5.4. Abundance and distribution of non-indigenous invasive species

The supplementary indicators for the distribution and abundance of selected invasive non-indigenous species are under development. The indicators will show the distribution map on the basis of the best available information, abundance map and graphs of the temporal changes of species.

No text has been finalized for this report. The first NIS species selected to be worked on are the zebra mussel (*Dreissena polymorpha*) and the fish hook water flea *Cercopagis pengoi*.

5.5. Biopollution level index

1. Working team: Non-indigenous species Authors: Maiju Lehtiniemi, Manfred Rolke, Malin Werner and Alexander Antsulevich	
2. Name of the indicator Biopollution level index	*3. Unit of the indicator* An index based on the number, abundance, distribution and impacts of non-indigenous species on native communities, habitats and ecosystem functioning
4. Description of proposed indicator Biological pollution (magnitude of bioinvasion impacts) is defined as the impact of non-indigenous species (NIS) on ecological quality and includes (but is not confined to) the genetic alteration within populations, the deterioration or modification of habitats, the spreading of pathogens and parasites, competition with and replacement of native species. The BPL method (Olenin *et al.*, 2007) takes into account the abundance and distribution range (ADR) of NIS in relation to native biota and aggregates data on the magnitude of the impacts in three categories: 1) impacts on native communities, 2) habitats and, 3) ecosystem functioning. ADR varies within five classes, ranking a NIS from low abundance in a few localities (A) to occurrence in high numbers in all localities (E). After ADR is established, three categories of impacts are considered, whose magnitude is ranked on five levels ranging from no impact (0) to massive impact (4) based on qualitative changes in an invaded ecosystem. The theoretical justification uses several well established ecological concepts, e.g. "key species", "type specific communities", "habitat alteration, fragmentation and loss", "functional groups", "food web shift", etc. BPL aggregates the results of the assessment into five categories: "No bioinvasion impact", "Weak", "Moderate", "Strong" and "Massive".	
5. Functional group or habitat type All non-indigenous species	
6. Policy relevance Descriptor 2 (Descriptor 1 & 4: indirectly)	
7. Use of the indicator in previous assessments None	
8. Link to anthropogenic pressures Directly impacted by: Introductions of new species e.g. from shipping, intentional stocking to aquaculture purposes or aquaria.	
9. Pressure(s) that the indicator reflect Shipping activities (untreated ballast water, hull fouling)	
10. Spatial considerations The indicator should be estimated for specific areas (e.g. sub-basins or assessment units consisting of national coastal and offshore waters) but can be generalized for larger areas as well.	
11. Temporal considerations The indicator will be assessed every six years, but data collection is continuous.	
12. Current monitoring The monitoring of NISs is rare in the Baltic Sea as such. However, in other biological monitoring programs (in the open sea and in coastal areas) e.g. in COMBINE program, NIS are often observed and counted, which gives a certain level of data for the indicator. Nevertheless it would be important to increase monitoring in specific less monitored habitats. The most problematic is to get information on the impacts of NIS, which is not currently monitored. The data on impacts have to be searched from the literature.	
13. Proposed or perceived target setting approach with a short justification. The goal is to minimize human mediated introductions of non-indigenous species. GES boundary for non-indigenous species should be 'no new introductions'. For the indicator in question the GES boundary should be 'No new non-indigenous species with known impacts'. This means that when an assessment is made only the species, which have been introduced after the previous assessment will be taken into account.	

Introduction

Introductions of non-indigenous species (NIS) in the Baltic Sea are increasing. In order to prevent new harmful introductions all human mediated introductions should be prevented. Not all introduced species are harmful, thus biopollution level index, which takes into account impacts of introduced species, gives a good overall view on the situation in the area in question concerning non-indigenous species.

Biopollution level index needs to be calculated if new introductions have happened meaning that 'trend in arrival indicator' have shown an increase during the six year assessment period. Furthermore, non-indigenenous species with a BPL<1 should be re-evaluated in every assessment period.

The impacts of non-indigenous species

The introduction of non-indigenous species is among the four largest threats to marine environment. They can have severe ecological, economic and public health impacts.

NIS can induce considerable changes in the structure and dynamics of marine ecosystems. They may also hamper the economic use of the sea or even represent a risk for human health. Ecological impacts include changes in habitats and communities and alterations in food web functioning, even losses of native species can occur in extreme cases. Economic impacts range from financial losses in fisheries to expenses for cleaning intake or outflow pipes of industries and structures from fouling. Public health impacts may arise from the introduction of microbes or toxic algae.

Due to the fact that only a minority of non-indigenous species (NIS) are invasive i.e. have a potential to cause negative impacts on the environment, the plain high numbers of introduced species do not provide sufficient basis for the assessment of biopollution i.e. effects non-indigenous species have on the ecosystem. Hence, the NIS need to be analysed and classified according to the magnitude of their impacts on the environment and biodiversity. In this regard, those NIS which cause most harm on the environment and/or humans are most important, and not only in terms of assessing the current and changing status of the ecosystems (requirements from the WFD and MSFD), but also in terms of the marine management perspective in order to facilitate strong move towards implementation of the ecosystem based approach.

One method for this is the Biopollution Level Index (BPL, Olenin et al. 2007). The proposed method has species-approach and considers each NIS separately, based on its impact on community, habitat or ecosystem levels. According to this method, the biopollution calculation is based on abundance and distribution range of the non-indigenous species under consideration and the magnitude of their impacts on native species, communities, habitats and ecosystem functioning. Index may get values from 0 (no biopollution) to 4 (massive biopollution). The method has been tested in the Baltic Sea with all known non-indigenous species recorded (Zaiko et al. 2011) as well as for a specific case of the dinoflagellate (*Prorocentrum minimum*) (Olenina et al. 2010).

Policy relevance

Since the early 1990's when the Marine Protection Committee (MEPC) of the International Maritime Organisation (IMO) put the issue on the agenda, the problem became more and more important for marine environmental protection. In 2004 the Ballast water Convention was adopted by the IMO. The convention requires ballast water management procedures to minimize the proliferation of non-indigenous species via ballast water and sediment. Once entered into force every ship has to treat its ballast waters unless exemption is given based on risk analysis.

In order to minimize adverse effects of introductions and transfers of marine organisms for aquaculture ICES drafted the 'ICES Code of Practice on the Introductions and Transfers of Marine Organisms'. The Code of practice summarizes measures and procedures to be taken into account when planning the introduction of

NIS for aquaculture purposes. On the European level the EC Council Regulation No 708/2007 concerning the use of non-indigenous and locally absent species in aquaculture is based on the ICES Code of Practice.

With the maritime activities segment of the Baltic Sea Action Plan HELCOM expresses the strategic goal to have maritime activities carried out in an environmental friendly way and that one of the management objectives is to reach "No introductions of alien species from ships". In order to prepare the implementation of the Ballast Water Convention a road map has been established with the ultimate to ratify the BWM Convention by the HELCOM Contracting States preferably by 2010, but in all cases not later than 2013.

The EU Marine Strategy Framework Directive in order to maintain or achieve good environmental status in the marine environment established a framework for Member states. The good environmental status shall be determined on the basis of qualitative descriptors. One of the qualitative descriptors concerns non-indigenous species stating 'Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystem.'

Biopollution Level index has been reviewed by the EU MSFD good environmental status task group 2 on non-indigenous species and is seen as a straight-forward and practical way of assessing impacts of the NIS and their potential to become invasive.

BPL index is also viewed as a practical solution for the Ballast Water Management efforts to distinguish between species which pose a threat on regional voyages and which should therefore be taken into account in Risk Assessments done for IMO BWM Convention when considering if a certain ship route may get an exemption from ballast water treatment procedures.

Biopollution Level assessment has presently been performed only once. Thus there are no temporal trends yet available concerning this indicator.

Assessment

This assessment is a baseline assessment, since the target for next assessments is to estimate impacts of new introduced species. Good Environmental Status is met when further impacts from NIS are minimized, with the ultimate goal of no adverse alterations to the ecosystems during the assessment period.The baseline assessment was performed for nine Baltic sub-regions. It revealed that documented ecological impact is only known for 43 NIS, which is less than 50% of the species registered in the sea. The highest biopollution (BPL = 3, strong impact) occurs in coastal lagoons, inlets and gulfs, and the moderate biopollution (BPL = 2) in the open sea areas (**Figure 5.1**). None of the Baltic sub-regions got low impact classifications (BPL = 0 or 1) indicating that invasive species with recognized impacts are established in all areas.

The most widely distributed species, observed nearly all around the Baltic Sea were *Marenzelleria* spp., *Potamopyrgus antipodarum*, *Eriocheir sinensis*, *Cercopagis pengoi*, *Mya arenaria* and *Balanus improvisus.* However, their magnitude of impact (BPL) differed between different sub-regions of the sea (**Figure 5.2**). For seven species (*Neogobius melanostomus, Obesogammarus crassus, Pontogammarus robustoides, Dreissena polymorpha, Gammarus tigrinus, Balanus improvisus and Cercopagis pengoi*) high biopollution level (BPL = 3) was defined in one or more analyzed subregions. None of the analyzed species got the maximum biopollution level (BPL = 4).

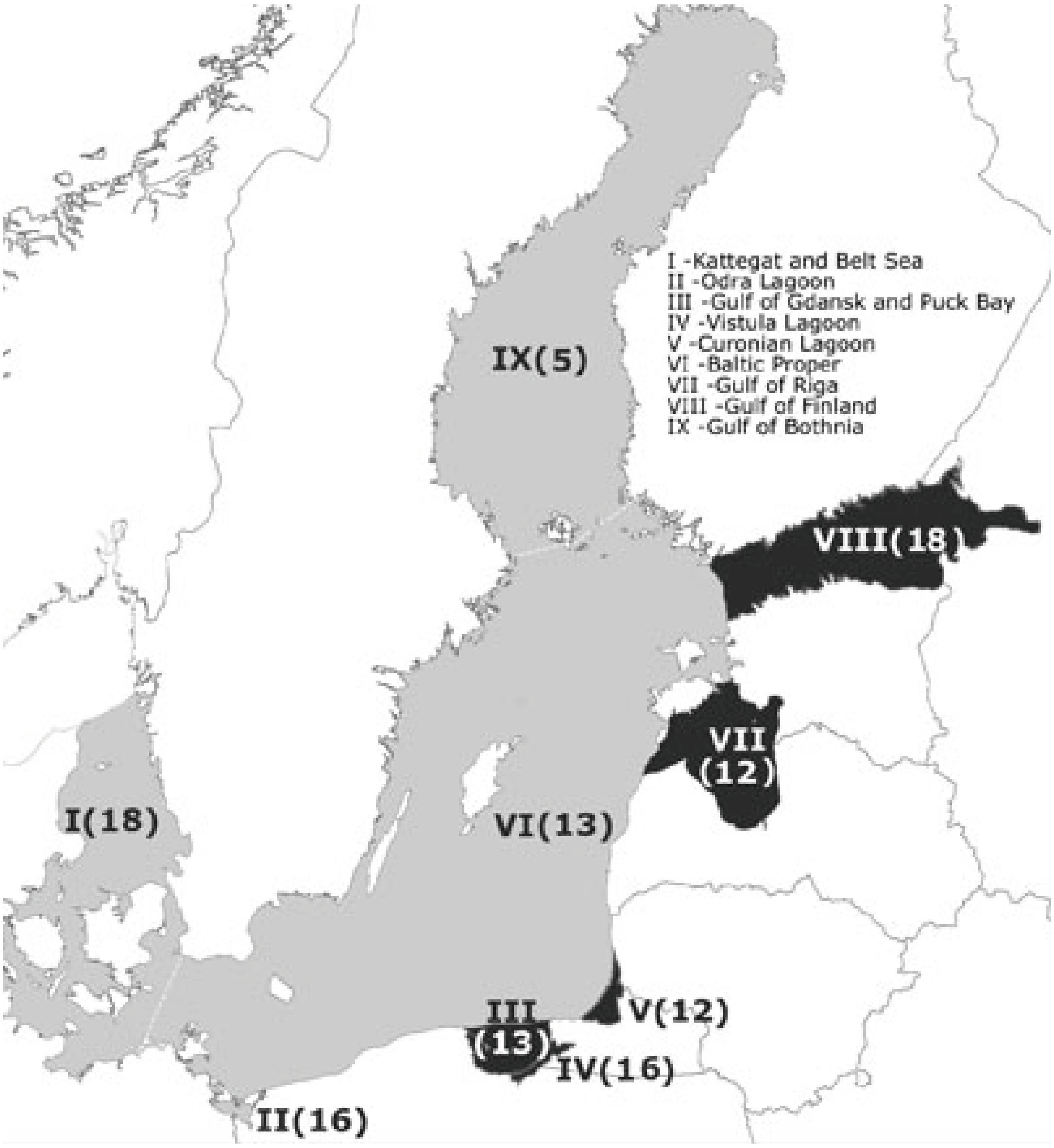

Figure 5.1. *Biopollution level in the assessed sub-regions in the Baltic Sea (lighter regions have BPL = 2 moderate biopollution, darker BPL = 3 strong biopollution). Numbers in parentheses indicate the number of impacting non-indigenous species in an assessment unit (with BPL>0). BPL may get values from 0 to 4.*

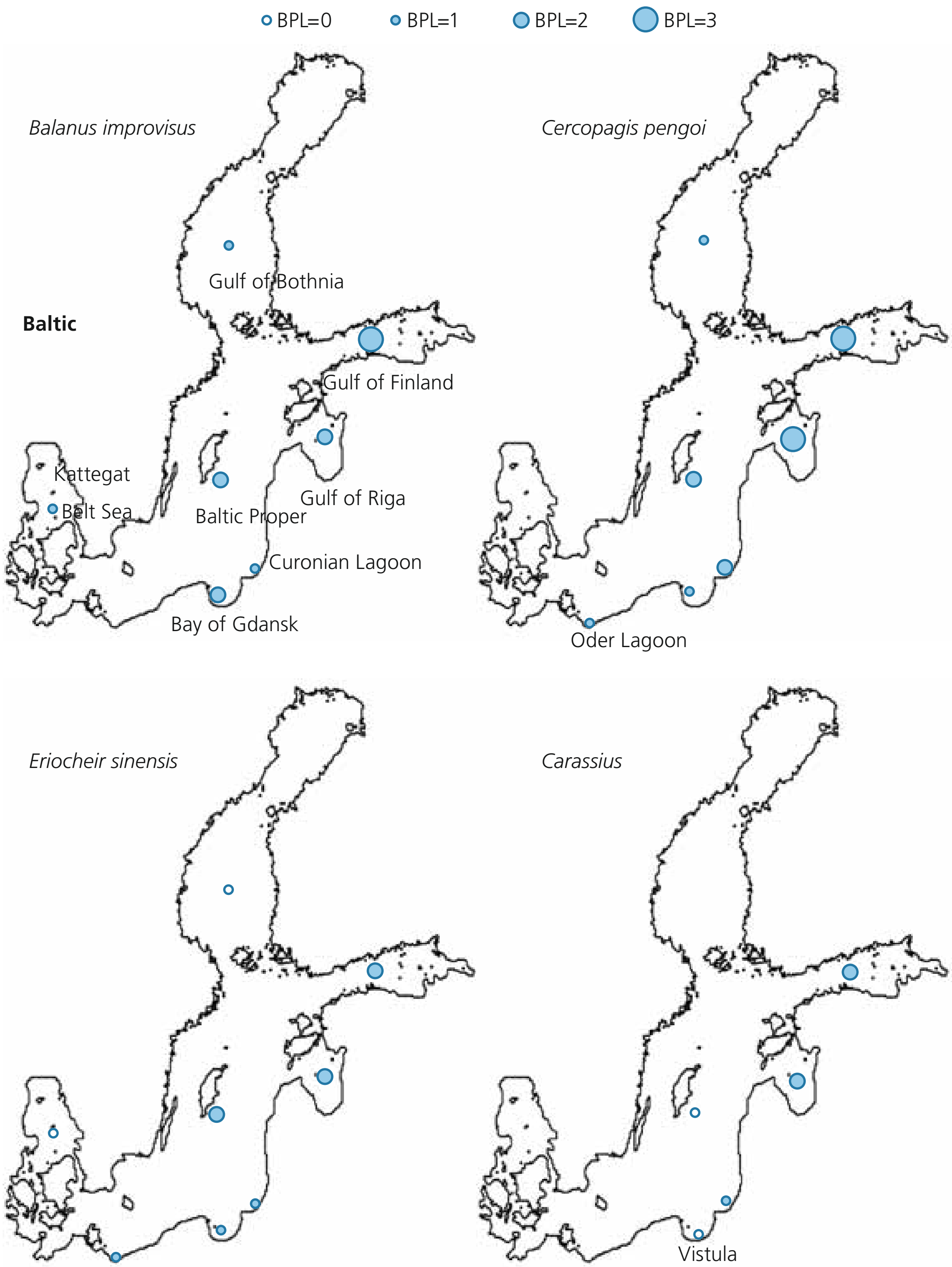

Figure 5.2. *Biopollution level for some of the assessed non-indigenous species, which are either widely spread in the assessment area, the Baltic Sea or have strong impacts in some of the sub-regions of the sea.*

GES and classification method

A problem related to non-indigenous species is that once a marine organism has been introduced and established in a new environment it is nearly impossible to eradicate it. The consequence is that assessing a status of an area as bad depending on the presence of invasive species means that the area will stay in a bad status without a possibility of improvement.

The goal is to minimize human mediated introductions of non-indigenous species. GES boundary for non-indigenous species should be 'no new introductions'. For the indicator in question the GES boundary should be 'No new non-indigenous species with known impacts'. This means that when an assessment is made only the species, which have been introduced after the previous assessment will be taken into account. GES is met when further impacts from NIS are minimized, with the ultimate goal of no adverse alterations to the ecosystems.

A contrasting point against the assessment of new arrivals only is that some NIS are known not to cause any impacts for a long time and when environmental conditions change they suddenly become invasive. Hence at least the species that were shown to have BPL≤1 should be reassessed. Furthermore, 6 years does not even leave enough time for research on impacts in situ.

Increasing biopollution indicates additional stress for the ecosystem. It is a signal of failed management concerning introductions of NIS.

Methodology for indicator calculation

The assessment is performed on 4 levels;

1. abundance and distribution (ADR)
2. impacts on communities
3. impacts on habitats
4. impacts on ecosystem functioning

and should be delivered on a defined aquatic area (e.g. a coastal lagoon, offshore sand bank, or even entire sub-basins) and for a defined period of time. After ADR is estimated, it is related to the magnitude of bioinvasion impacts, in order to reach the biopollution level index ranging from 0 to 4:

(0) no impact
(1) weak impact
(2) moderate impact
(3) strong impact
(4) massive impact.

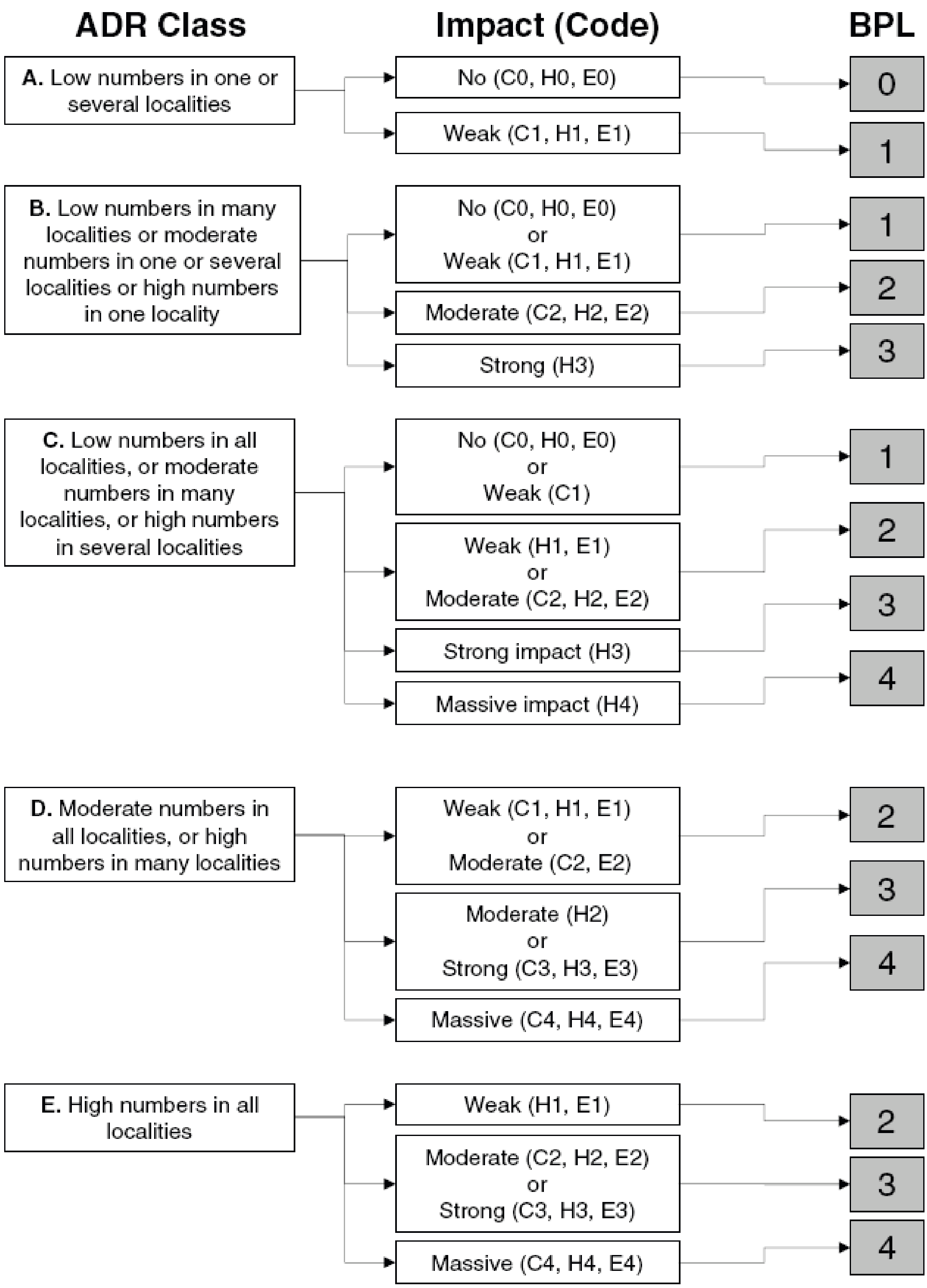

Figure 5.3. *The scheme for assessment of biopollution level (BPL) index where ADR is abundance, distribution and range of NIS and impact codes (range from 0 = no impact to 4 = massive impact) are C on communities, H on habitats, E on ecosystem functioning. Source: Olenin et al. 2007.*

Existing monitoring

There is no monitoring on NIS in the Baltic Sea but data is available on species abundance and distribution as well as on impacts of the most harmful species, which is stored e.g. in BINPASE system and Baltic Sea Alien Species database as well as in published literature. Proven information provided by other sources such as research institutes maybe taken into account.

Data can be viewed and downloaded from the internet on BINPAS-pages (http://www.corpi.ku.lt/databases/index.php/binpas/) where BPL assessments can be done.

Description of data

All established non-indigenous species in an assessment area are taken into account. For indicator calculation, see above.

Geographic coverage

Available data covers the whole Baltic Sea. Gaps in needed data depend on the size of the assessment units. If the assessment is done for a small area, it is possible that some data is lacking concerning both abundances and impacts.

Strengths and weaknesses of data

strengths: harmonized targets, calculation is made easy through the online BINPAS system.
weaknesses: differences in national data sets, gaps in knowledge on impacts of certain species, reliance on expert judgement

Further work required

The descriptions of different abundance-distribution-range classes as well as impact classes should be made more clear and developed further in order to minimize differences between index calculations made by different experts.

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5.6. Organochlorine compounds

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General information

General properties

Hexachlorobenzene (HCB) , Hexachlorohexane isomers (HCHs) and 1,1,1-trichloro-2,2-di(chlorophenyl) ethane (DDT) and its metabolites DDD (1-chloro-4-[2,2-dichloro-1-(chlorophenyl)ethyl]benzene) and DDE (1,1-*bis*-(chlorophenyl)-2,2-dichloroethene) are toxic organic pollutants, which have been used mainly as insecticides (HCHs, DDTs) or fungicides (HCB) in the Baltic region partly up to the mid-1990s. HCHs and DDTs have been applied as technical mixtures, the DDTs dominated by the para-phenyl-chlorinated isomers, the HCHs by alpha and gamma-HCH in the past, later in Western Europe only more or less pure gamma-HCH was used. They are less volatile and water soluble, in the marine environment they are mainly attached to particles of suspended matter (SPM) and sediments and accumulate in the lipids of organisms. Due to their properties these substances are belonging to the so-called PBT-compounds (persistent, bio-accumulating and toxic).

Main impacts on the environment and human health

Due to their persistence HCB, HCHs and DDTs are still remaining in the marine environment even decades after the use having been banned or phased out. They are taken up by organisms, bio-accumulated and bio-magnified in the food chain up to the top-predators, the marine mammals, seabirds and humans. The prevailing toxic effects described for these substances are negative impacts on behavior, growth, enzyme and hormone status and reproduction, histo-pathological findings, carcinogenic, increased mortality, acute intoxication and decrease of populations. The best example for the latter was the threat to the sea eagle at the Baltic Sea. The ecotoxicological classification for fish is ranging from high to very high toxic (only HCB is ranked as moderate), and for molluscs, crustaceans and plankton differing and depending on substance and isomer from slight (DDE in zooplankton) to very high toxic for DDD in crustaceans or DDT in phyto-plankton.

The HELCOM thematic assessment showed that the concentrations of DDE exceeded in many sites the thresholds (HELCOM 2010), whereas HCHs and HCB were generally below the thresholds.

Status of a compound on international priority lists and other policy relevance

In **Table 5.2** the most important conventions, directives and related monitoring programmes are listed, which are including HCB, HCHs and DDTs as indicator, parameter or hazardous substance of concern.

Table 5.2. Occurrence of HCB, HCHs and DDTs on international priority lists.				
Convention/ Programme/ List	**Monitoring**	**Compound/-group**		
		Hexachlorobenzene (HCB)	hexachloro-cyclohexane (HCH)-isomers	DDT and Metabolites DDD (TDE) and DDE
HELCOM Combine	Biota	herring/perch muscle (core/coastal core programme)	herring /perch muscle (core/coastal core programme)	herring/perch muscle (core/coastal core programme)
		cod liver (main programme)	cod liver (main programme)	cod liver (main programme)
			Macoma baltica soft tissue (main programme)	Macoma baltica soft tissue (main programme)
			Blue mussel soft tissue (supporting programme)	Blue mussel soft tissue (supporting programme)
	Sediment	*Not part of the regular COMBINE programme*	*Not part of the regular COMBINE programme*	*Not part of the regular COMBINE programme*
	Water	In total sea water	In total sea water	In total sea water
HELCOM List of potential substances of concern (Recommendation 19/5; Attachment Appendix 2)		Substance for immediate priority action	Substance for immediate priority action	Substance for immediate priority action
HELCOM BSAP		*No indicator of the BSAP list of Substances or substance groups of specific concern to the Baltic Sea*		
OSPAR CEMP		*No indicator of CEMP-monitoring*		
OSPAR MON Assessment				
OSPAR List of Chemicals for Priority Action (Update 2007)		**Background Document** Reference number 2004-12	*Not part of the list and no background document*	*Not part of the list and no background document*
TMAP	Biota	fish, mussel, bird eggs (mandatory)	fish, mussel, bird eggs (mandatory)	fish, mussel, bird eggs (mandatory)
	Sediment	mandatory	mandatory	mandatory
EU Directive 2000/60/ EC WFD		Appendix X, replaced by of EU Directive 2008/105/EC, Annex II	Appendix X, replaced by of EU Directive 2008/105/EC, Annex II	

EU Directive 2008/105/ EC Daughter Directive to WFD		Annex I, part A: EQS for inland and other surface waters	Annex I, part A: EQS for inland and other surface waters	Annex I, part A: EQS for inland and other surface waters
Stockholm POP Convention (2009 decisions SC-4/10 to SC-4/18)		Annex A, C	Annex A	Annex B
Medpol Phase III	Different matrices	Halogenated Hydrocarbons (recommended)		
Black Sea Conven- tion: Integrated Monitoring and Assessment Pro- gramme (BSIMAP)	Biota		Mandatory (turbot, bivalves)	Mandatory (turbot, bivalves)

Status of restrictions, bans or use

DDT, lindane and HCB are no longer used in the Baltic Sea region. DDT was banned already by the 1974 Helsinki Convention. The Stockholm Convention prohibits the use, import and export of HCB. WHO identifies lindane (gamma HCH) as moderately hazardous and its international trade is restricted and regulated by the Rotterdam Convention. In 2009 the production and agricultural use of lindane was banned under the Stockholm Convention.

GES boundaries and matrix

Existing quantitative targets

In OSPAR, no revision of EACs had been made for the QSR 2010 for HCB, HCHs and DDTs, as these substances are not part of the CEMP and have not been assessed. Therefore, the OSPAR values listed in the following table are reflecting the status in 2004 (ASMO 08/6/5 Add.4-E (L).

Table 5.3. Existing quatitative targets for HCB, HCHs and DDTs.	
Source	**Value and description**
OSPAR	
EAC sediment	HCH (gamma): 1.1 µg kg^{-1} dry wt (normalised to 2.5% carbon) DDE: 1.6 µg kg^{-1} dry wt (provisional)
EAC fish	HCH (gamma): 1.1 µg kg^{-1} wet wt
EAC mussel	HCH (gamma): 0.29 µg kg^{-1} wet wt
EAC water	HCH (gamma): 0,002 µg/l DDE: 0,000001 µg/l
EC	
EQS water for "other surface water" (Directive 2008/105/EC)	HCB: AA: annual average = 0,01 µg/l HCB: MAC: maximum allowable concentration = 0,05 µg/l HCH (combined): AA = 0,002 µg/l, MAC = 0,02 µg/l DDT (total): AA = 0,025 MAC = not applicable DDT (p,p'): AA = 0,01 MAC = not applicable
EQS biota	HCB: 10µg/kg prey tissue (wet weight)
QS biota	HCH (gamma): QS Biota, secondary poisoning(WFD datasheet): 33 µg/kg (tissue of prey, wet wt)
QS sediment	HCB: QS Sediment (WFD datasheet): 3.7 µg/kg wet wt (≈16.9 µg/kg (dry wt) HCH: QS Sediment (WFD datasheet): 0.24 µg/kg wet wt (≈ 1.1 µg/kg dry wt)
QS Human health	HCH (gamma): 61 µg/kg fishery product (wet wt)
Effect Range -Low	
ERL sediment	DDT (total): 1,6 µg/kg dry sediment p,p'-DDE: 1,2 µg/kg dry sediment

Preferred matrix

DDTs: Herring and perch muscle, cod liver, bivalve soft tissue. (DDD sediment).
HCHs: Herring and perch muscle, cod liver, bivalve soft tissue. Water.
Herring and perch muscle, cod liver. Bivalve soft tissue.

Monitoring the compound

Status of monitoring network (geographical and temporal coverage)

The following table presents an overview of results about HCB, HCHs and DDTs as being available from public accessible sources like reports from the HELCOM website and ICES EcoSystemData maps. Compared with the national commitments it is obvious, that all member states are monitoring the substances at least in biota (information from Russia is missing). However, no statement on completeness of geographical and temporal coverage can be made on this basis, as the reports are reflecting only a certain choice of data, and the data extraction from the ICES Data base DOME into the maps may not visualize the same status as possibly exist in the national data centres or at the originators.

Table 5.4. Monitoring of HCB, HCHs and DDTs in the Baltic Sea.				
		Compound/-group		
Member State	**Compartment**	**Hexachlorobenzene (HCB)**	**hexachloro-cyclohexane (HCH)-isomers**	**DDT and Metabolites DDD (TDE) and DDE**
DK	Biota	u	u	u
	Sediment	u	k	u k
	Seawater			
DE	Biota	u	u	u
	Sediment	j	j k	k
	Seawater	u	u k	u
PL	Biota	u	u	u k
	Sediment		k	k
	Seawater		k	
LIT	Biota			k
	Sediment		u k	u k
	Seawater	j	u k	u
LAT	Biota	u	u	u k
	Sediment			
	Seawater	j	k	u
EST	Biota	u	u	u k
	Sediment			
	Seawater			
RUS	Biota	No information	No information	No information
	Sediment			
	Seawater			
FIN	Biota	u	u x	u k
	Sediment		k	k
	Seawater		k	
SWE	Biota		x	k
	Sediment		k	k
	Seawater		k	
– j indicated in ICES EcoSystemData maps, only 1 time within the period 2004-2009				
– u indicated in ICES EcoSystemData maps, more than 1 time within the period 2004-2009				
– k indicated in maps of report "Hazardous Substances in the Baltic Sea", HELCOM BSEP 120B				
– part of National Monitoring Programme as committed for Combine to HELCOM (Manual D.13)				
– x published in HELCOM Indicator Fact Sheets				

Gaps in the monitoring of the compound

This paragraph needs further investigation, as no actual overview is available. There has been no compliance check been performed by HELCOM MONAS the last years, to which degree contracting parties have fulfilled their commitments.

Present status assessments

Known temporal trends (also from sediment core profiles)

With some exceptions particularly from coastal sites, HCB, HCHs (alpha and gamma) and DDTs show distinct decreasing trends in biota within the last two decades (chapter 2.2.5 BSEP 120 B). In the last time, concentrations have reached a relatively low level, and a small resulting trend may be masked by the within- and between-year variability of the data.

No sufficient information on sediment time series found! Water is only analysed officially by two member states.

Spatial gradients (incl. sources)

For DDE in 1995 a spatial gradient from the northern to the southern Baltic Sea is obvious with increasing concentrations by a factor of about seven, in principle sill remaining also 2005, but the factor has decreased to only half (3.6) of that ten years before. For gamma-HCH, in 1995 also an increasing gradient from north to south with a ratio of 3 can be observed, but in 2005 concentrations in all areas are at an comparable level and no gradient occurs any longer.

Table 5.5. Spatial and temporal trends of DDE and gamma-HCH (lindane).

	DDE 1995	DDE 2005	Gamma-HCH 1995	Gamma-HCH 2005
Bothnian Bay:	*80 µg/kg lw*	*40 µg/kg lw*	*10 µg/kg lw*	*< 5 µg/kg lw*
Bothnian Sea	*150 µg/kg lw*	*70 µg/kg lw*	*12 µg/kg lw*	*< 5 µg/kg lw*
Gulf of Finland	*4 µg/kg ww*	*1 µg/kg ww*		
North. Baltic Proper:			*22µg/kg lw*	*5 µg/kg lw*
Arkona Basin	*570 µg/kg lw*	*143 µg/kg lw*	*29 µg/kg lw*	*6.3 µg/kg lw* *4.1 µg/kg lw (in 2009)*

The data in BSEP 120 B in fig. 2.19 and 2.20 are displayed on a different basis. In 1995 (and following years) the small herring samples from the Arkona Sea had extractable lipids in muscle between 3-4% . Calculating on lipid basis and compared with the other areas would result in the estimate sshown in this table.

Recommendation

Recommended matrices (in order of priority):

1. Biota: HCH (alpha-, beta- and gamma-) and <u>4,4'-DDE, 4,4-DDD and 4,4'-DDT, (HCB only where of local relevance)</u>
2. Water: HCH (alpha-, beta- and gamma-), 4,4'-DDE, 4,4-DDD and 4,4'-DDT
3. Sediment: 4,4'-DDE, 4,4-DDD and 4,4'-DDT

For HCHs and DDTs the relative pattern within each group is indicating local differences as well as changes due to recent inputs or remobilization. HCB is relatively uniform distributed, so that it may be less suitable for geographical differentiation.

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