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Defining the coastal water quality in Estonia based on benthic invertebrate communities

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Abstract. The European Union Water Framework Directive requires that all member states assess the status of their coastal areas and develop or use existing classification systems to support future monitoring. To set the quality assessment system for the Estonian coastal sea, the composition of modern zoobenthic communities was compared to the communities from the 1950s–1960s. Sensitivity values of benthic taxa were determined, and the macrozoobenthic community index ZKI and boundaries for the classification system were developed. The ZKI index was further validated against nutrient loads and the spatial location relative to pressure, represented by the Baltic Sea Pressure Index (BSPI). High variability in Ecological Quality Ratio assessment results was found for both ZKI and brackish water benthic index (BBI) based assessments. However, in the study area, ZKI assessments fluctuated to a smaller extent and displayed a better correlation with the BSPI than the BBI assessments.

Key words: Baltic Sea, benthic indicators, soft-bottom benthic macroinvertebrates, coastal waters, nutrients, Baltic Sea Pressure Index BSPI, ZKI, BBI.

INTRODUCTION

The European Union Water Framework Directive aims at protecting, enhancing, and restoring all bodies of surface water with the ultimate aim of achieving good surface water status by 2015. In order to implement the directive, all member states have to assess the status of their coastal areas and develop or use existing classification systems to support future monitoring. Consequently, classification will be a key part of the implementation of the Directive (European Commission, 2000).

It is known that aquatic ecosystems are complex mixtures of plants and animals. Aquatic systems may respond to variations in their physical, chemical, and biological environments in many very different ways because these assemblages typically include organisms with a wide range of physiological tolerances, feeding modes, and trophic interactions (e.g. Bonsdorff & Pearson, 1999; Kotta et al., 2008). This is also the reason why plant and animal assemblages are rarely similar between sites, and their interactions with prevailing physical, chemical, and biological environments determine their responses to human-induced stresses

(Kotta et al., 2007; Veber et al., 2009; Lauringson et al., 2012). Classification systems seek to describe all these interactions and artificially divide the observed continua into discrete classes using statistical manipulations. While classification systems have considerable value as management concepts, it has to be remembered that they are at best an approximation of actual ecological quality (e.g. Southworth et al., 2004; Bolliger & Mladenoff, 2005).

In the northeastern Baltic Sea, harsh environmental conditions result in a low number of benthic species (Bonsdorff & Pearson, 1999); nevertheless, these species can be considered very tolerant to various disturbances including anthropogenic stresses (Kotta et al., 2007, 2009). Thus, it becomes of utmost challenge to separate natural variability from human-induced changes that have occurred since the so-called pre-eutrophication era. In fact, benthic studies in the pre-eutrophication era are rare and often hampered by the lack of quantitative estimates (see also Kotta & Kotta, 1995; Eriksson et al., 1998; Kovtun et al., 2009). The extent to which the benthic life has deteriorated compared with the pre-eutrophication era is difficult to assess given the lack of comprehensive data sets. In this respect, the earlier documentation by A. Järvekülg in the Central Databases of the Estonian Marine Institute provides a unique opportunity to compare the benthic macroinvertebrate communities over the last 50 years, and these data can be used to record the sensitivity values of zoobenthic taxa as well as to define the high quality status for zoobenthic communities.

Biological water quality indices developed for the brackish water conditions raise the issue of the Estuarine Quality Paradox, as estuaries are naturally highly stressed environments and inhabited by stress-tolerant biota (Dauvin, 2007; Elliott & Quintino, 2007) that has to cope with both high natural loads of organic matter and decreased salinity. The most widely used biotic indices such as Marine Biotic Index (AMBI) and biological quality index (BQI) were developed for marine areas, and their use in brackish waterbodies has been found problematic in several cases (Borja et al., 2009; Munari & Mistri, 2010). The salinity in the Baltic Sea ranges from over 25 in the entrance to less than 1 in the innermost ends and represents the main large-scale structuring factor for benthic communities (Voipio, 1981). The salinity gradient has caused problems in using the Shannon diversity index (H′), AZTI's AMBI, and BQI in the more saline southern part of the Baltic Sea (Zettler et al., 2007), and problems were also encountered in using the AMBI in the less saline, very species-poor part of the sea (e.g. Perus et al., 2007).

In the Baltic Sea, several modified approaches have been tested for the ecological quality assessment in recent years (Perus et al., 2007; Fleischer & Zettler, 2009; Josefson et al., 2009; Leonardsson et al., 2009). The spatially nearest ready index solution is the Finnish brackish water benthic index (BBI), which is used at the northern side of the Gulf of Finland (Perus et al., 2007). Although the BBI is not based on historical data, it resembles the zoobenthic community index (ZKI) in that area-specific sensitivity lists of zoobenthic species are used. The BBI index uses the sensitivity list of Swedish BQI, as the BBI incorporates the BQI formula. Contrary to the ZKI, relative abundances instead of biomasses are used in BBI calculations. However, the Finnish coastal region differs from the Estonian coastal sea in climate, salinity, and/or hydromorphological features like the complexity of coastline and bottom topography. It is yet unknown whether functional responses of zoobenthic communities to eutrophication also differ between these areas.

In this study we give an overview of long-term changes of benthic invertebrate communities in the Estonian coastal range and seek whether the changes can be attributed to regional eutrophication. In order to do so, the composition of the modern zoobenthic communities was compared to the communities from the 1950s–1960s. Based on the observed changes, each taxon was given a sensitivity value. The macrozoobenthos community index ZKI was built and boundaries for the classification system were developed. The variability of the ZKI index in relation to different pressure indicators was described. Finally, the ZKI and BBI index assessments were compared in relation to the Baltic Sea Pressure Index (BSPI) gradient in the southern Gulf of Finland.

METHODS

The study area is located in the northeastern Baltic Sea and consists of the coastal part of Estonian territorial waters. This is a large ecosystem with strong seasonality in temperature, oxygen, and light conditions. There exists also a spatial salinity gradient from west to east. According to the hydrological regime, the study area is divided into 16 waterbodies sensu the European Union Water Framework Directive (WFD), and the assessment of the ecosystem state is made by these basic WFD management units (Fig. 1).

Two data sets were used in this study. The historical data set was collected by Arvi Järvekülg in 1959–1967, and the modern data set was collected by the Department of Marine Biology at the Estonian Marine Institute in 1996–2010. Historical data cover all Estonian waterbodies sensu WFD except for 3 and 15. These two waterbodies are very small compared to other waterbodies and it is most likely that information from the adjacent sea is adequate for the description of reference conditions of these waterbodies. Modern data cover all the waterbodies.

Altogether 526 benthos samples were collected in the 1950s–1960s and 2353 samples in the 1990s–2000s. The depth at the study stations was 5–30 m and the sediments were soft, including mixed sands. Historical studies took place before the explosive increase in agricultural activities in the 1970s and 1980s in the Eastern European countries. At that time, no symptoms of system-wide man-induced acceleration of eutrophication were detectable in the northeastern Baltic Sea (Kotta et al., 2004, 2008). Elevated nutrient loads from a few larger point sources and rivers resulted in an increased biomass of benthic invertebrates only locally (Kotta & Kotta, 1995; Kotta et al., 2008). For this reason, Pärnu, Tallinn, and Narva bays (corresponding to water bodies 13, 6, and 1, respectively) from the vicinity of large towns or mouths of rivers were excluded, and adjacent waterbodies with similar salinity, temperature, and exposure regimes were used as reference areas.

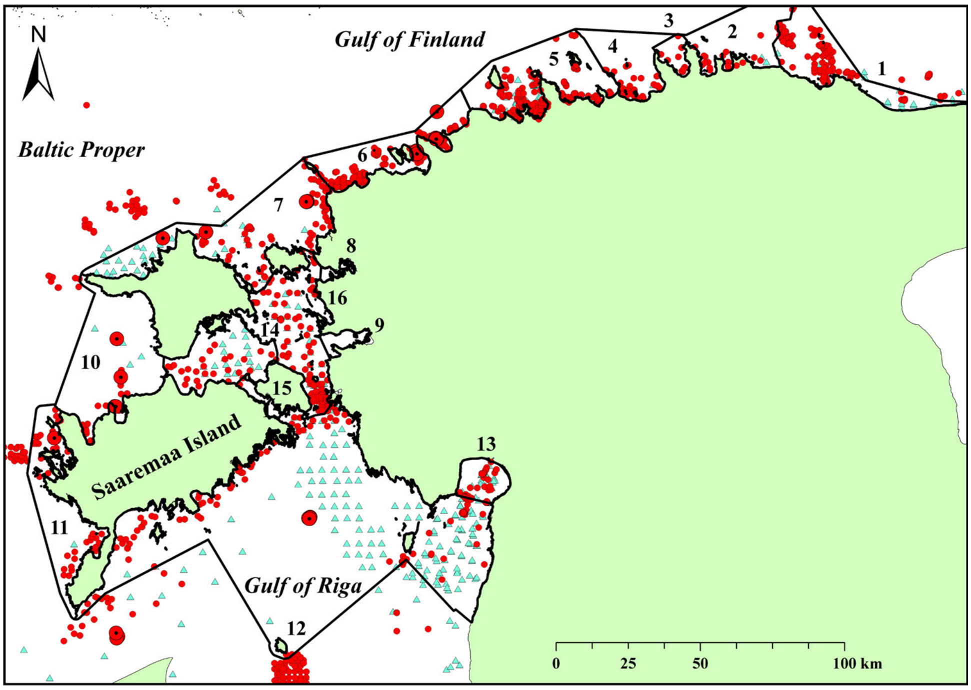


Fig. 1. Waterbodies of the Estonian coastal sea. Triangles indicate the location of sampling sites in the 1950s–1960s and circles in the 1990s–2000s. Larger circles indicate sites used to derive the G/M boundary.

The historical methodology matched the modern methodology in terms of grab types, mesh size, and laboratory procedures. Ekman (sampling area 0.02 m^2) or Van Veen (sampling area 0.1 m^2) types of grabs were used, and samples were sieved on the mesh size of 0.25 mm. An earlier pilot study showed that the used grabs catch benthic invertebrates at similar efficiencies on the studied soft bottoms resulting in no statistical differences in invertebrate species biomasses among grabs. After collection, the samples were frozen until analysis. In laboratory, animals were determined to species level, except Oligochaeta, Ostracoda, *Pisidium* and *Sphaerium* clams, leeches, insect larvae, and juvenile gammarids. Animals were counted with binocular microscope, dried at 60 °C for two weeks, and then the dry weight was measured. In order to avoid a bias towards the status of biomass dominants (i.e. bivalves such as *Cerastoderma glaucum*, *Dreissena polymorpha*, *Macoma balthica*, *Mya arenaria*, and *Mytilus trossulus*), the dry shell-free biomass of bivalves was used in the calculations. If such information was not provided by monitoring, we obtained the ratios of shell-free to total dry mass using the central database of the Estonian Marine Institute, which has an extensive data set of morphometrical measurements of bivalves (i.e. the coefficients are based on more than 20 000 measurements on shell length, shell mass, and shell-free mass). The derived coefficients were as follows: *Cerastoderma glaucum* = 0.45, *Dreissena polymorpha* = 0.07, *Macoma balthica* = 0.38, *Mya arenaria* = 0.48, and *Mytilus trossulus* = 0.16.

The sensitivity values of invertebrate taxa were calculated using ANOSIM and SIMPER procedures of the PRIMER version 6.1.5 (Clarke & Gorley, 2006). Invertebrate biomass data were not transformed. Similarities between each pair of samples were calculated using a zero-adjusted Bray–Curtis coefficient. The statistical differences in the dominance structure of benthic invertebrate communities among the historical and modern data sets were assessed using ANOSIM analysis. SIMPER analysis was used to describe the changes in the biomasses of benthic invertebrate taxa among the study periods. Non-metric multidimensional scaling analysis (MDS) on benthic invertebrate biomasses was used to visualize dissimilarities in the community composition of benthic invertebrates in the last 50 years. Based on the literature (e.g. Järvekülg, 1970; Kotta & Kotta, 1995; Leonardsson et al., 2009) and this comparison, invertebrate species were divided into three groups reflecting their sensitivity to eutrophication. Groups 2 and 3 were defined based on the difference between past and present conditions. The species whose relative biomass was smaller in modern samples compared to the historical data set were considered as sensitive to human disturbance, and these species form group 3. Group 2 consisted of species whose relative biomass in similar habitats has either remained the same or was higher nowadays compared to the historical data set. However, group 2 species cannot stand heavily disturbed conditions (e.g. elevated loads of organic matter, frequent occurrence of hypoxia, presence of drift algae, etc.). Group 1 was formed of opportunistic taxa that are able to form single-species associations or highly dominate communities under heavily disturbed conditions. Information on such taxa was obtained from the literature (Leonardsson et al., 2009). The indicative values of invasive species (those established later than the 1960s) were obtained either from the literature and/or functional relationships between nutrient load and species biomass data collected in the 1990s and 2000s.

The Pearson–Rosenberg model (Pearson & Rosenberg, 1978) of the community succession at a gradient of organic enrichment was used as the theoretical basis for the ZKI. According to the model, a progressively greater carbon loading results in increased productivity, loss of species diversity, dominance of opportunistic species, but a too high level of carbon loading leads to the disappearance of benthic invertebrates due to anoxia.

The ZKI was then validated against a pre-known pressure gradient. To indicate the intensity of local human pressures, the Baltic Sea Pressure Index (BSPI) (HELCOM, 2010) and annual nutrient loads to respective waterbodies were used (Table 1). The BSPI index is assigned to every 5 km × 5 km spatial unit, and it combines several pressure metrics, including atmospheric deposition of nitrogen over the years 2005–2007, waterborne inputs of nitrogen and phosphorus based on the data from the year 2000, riverine input of organic matter during 2003–2006, dredging activities during 2003–2007, disposal of dredged material from

Table 1. Average nutrient loads (t a^{-1}) ± SD and median BSPI values in the waterbodies used for validation. n – number of years with biological data

Waterbody	Point N	Riverine N	Point P	Riverine P	n	BSPI
1	462.3 ± 267.7	11 882 ± 3 727	13.4 ± 5.4	704 ± 231	12	75
2	0.6 ± 0.4	1 273 ± 375	0.1 ± 0.1	14 ± 3	3	66
3	1.3 ± 0	600 ± 12	0.2 ± 0	10 ± 3	2	77
4	0.1 ± 0.1	136 ± 47	0.0 ± 0	7 ± 2	2	65
5	789.8 ± 170.2	2 257 ± 771	57.4 ± 13	40 ± 13	13	78
6	6.8 ± 2.1	2 780 ± 1 039	1.6 ± 1.2	59 ± 20	13	45
7	4.4 ± 5.1	851 ± 237	1.3 ± 1.9	25 ± 7	12	44
10	0	893	0	28	1	45
11	0	463	0	12	1	46
12	31.3 ± 14	1 138 ± 414	5.0 ± 2.7	42 ± 12	13	54
13	17.9 ± 11.6	4 922 ± 1 521	3.1 ± 2.3	124 ± 40	11	57
14	No information on nutrient loads				1	44
15	0.1	165	0.1	4	1	44
16	2.6 ± 0.4	212 ± 85	0.2 ± 0.1	8 ± 2	12	46

2005 to 2007, heavy metal and radionuclide input data from 2003 to 2006, data on harbour cargo volumes from the year 2008, distance to harbours, boating and shipping estimates, and accounts for human population density. The combination of several metrics increases the robustness of the BSPI, and the BSPI was considered a satisfactorily adequate measure to be used as an indicator of spatial pressure gradient in the study area. Data on the annual point source and riverine loads of total N and total P to the different waterbodies of the Estonian coastal sea in 1996–2010 were obtained from the Estonian Ministry of Environment. These data represent a proxy of the inter-annual variability in the amount of nutrients arriving at the study stations.

We applied data from 1996 to 2010 in the validation process. In addition, we applied separately the values of the ZKI from the year 2007 to match the temporal range of data that were used to calculate the BSPI. The spring and early summer benthic community data were chosen for the validation procedure in order to exclude the seasonal variability in the ZKI. The ZKI was related to waterbody-level annual nutrient loads in the whole study area and also separately in two subregions. Spearman correlations of the ZKI with the BSPI and nutrient loads were calculated using the statistical package Statistica (StatSoft, 2011).

The principle of the whole water quality assessment procedure is to measure deviation from the reference conditions. According to the normative definition of the WFD, reference conditions represent a status with no or only minor anthropogenic impact. Ecological status assessments shall permit classification of waterbodies into five classes – poor, bad, moderate, good, and high (European Commission, 2000; Torn & Martin, 2011). The boundary values for the zoobenthos quality element in the Estonian coastal sea were set following a mixed protocol. High/good (H/G) and good/moderate (G/M) boundaries were derived from the variability of metrics at historical reference sites or modern least-impacted sites as described below. Moderate/poor (M/P) and poor/bad (P/B) boundaries were set by an equidistant division of the remaining ZKI gradient.

As natural variability in the ZKI is high (Lauringson et al., 2012), the H/G boundary is based on the lower tail (20th percentile) of the natural variability in the years 1960–1965 in historical near-reference communities from sites far away from local pollution sources and assumedly with a low diffuse pollution loading. The historical data set was bootstrapped for 10 000 times, 20 stations were picked randomly with replacement every time, using the statistical package R (R Development Core Team, 2011). The mean values of these bootstrapped data sets (each with 20 values) were calculated. The lower 20th percentile of the distribution of these mean values was set as the H/G boundary.

The G/M boundary is based on the lower 20th percentile of the variability in the present-day communities from sites at some distance from local pollution sources (see Fig. 1). Such sites are supposedly subject to both natural forces and diffuse human impacts, which generally cannot be resolved by environmental policy measures of just a single country. For the G/M boudary, 75 data points were picked from the years 1997–2010 from areas with no direct local human impact. A similar bootstrapping procedure was used as for the H/G boundary.

The results did not vary at bootstrapping sample sizes from 15 to 30. The bootstrapping sample size 20 was selected to match the number of samples in the later assessment procedure. It must be mentioned that in the later assessment procedure of EQRs, similar sample sizes should be used as when setting the boundaries (15 to 30 benthic samples per waterbody).

RESULTS

Comparison of the past and present benthic invertebrate communities showed that regardless of some overlap, the structure of benthic invertebrate communities significantly differed between the studied periods (ANOSIM $p < 0.001$, Fig. 2). As seen from the graph, the present communities are more uniform (shown by their lower between station dissimilarities/distances) compared to the past communities. Besides, the total invertebrate biomass systematically increased in all waterbodies. The majority of polychaetes and molluscs (both bivalves and gastropods), as well as some crustacean species, significantly increased their biomass and biomass proportion within invertebrate communities along the human-induced stress gradient including eutrophication. The largest differences were due to the bivalves and gastropods *Mytilus trossulus*, *Macoma balthica*, *Mya arenaria*, *Cerastoderma glaucum*, *Theodoxus fluviatilis*, and *Dreissena polymorpha*. While *M. trossulus* increased its biomass in western waterbodies (more saline, characterized as frontal areas), other species increased their biomasses in more sheltered waterbodies. The increase in the biomass of *M. trossulus* exceeded manifold that of the other invertebrate species (SIMPER analysis).

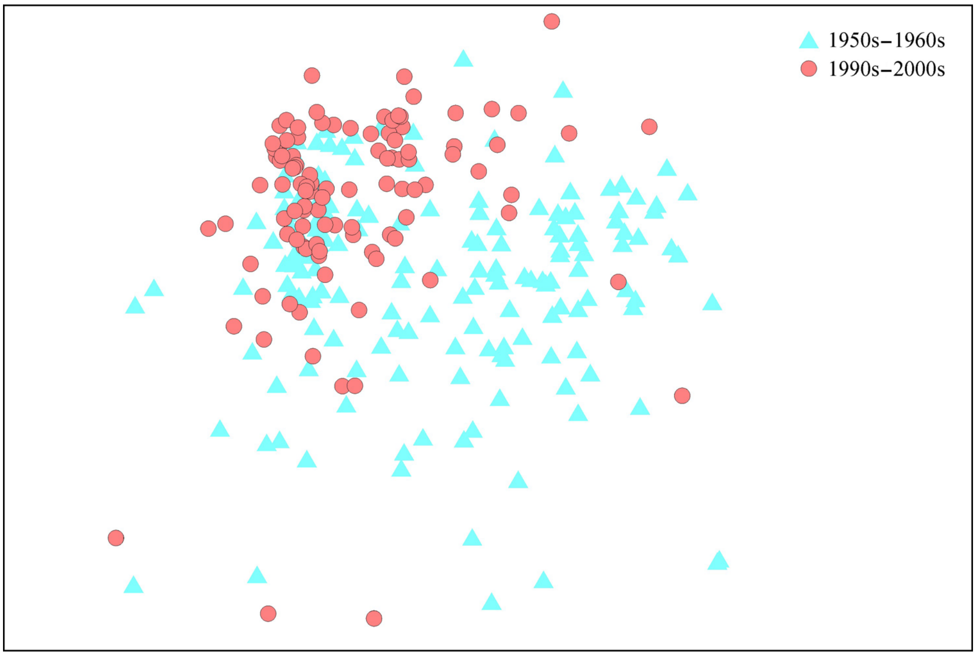


Fig. 2. MDS ordination of biomasses of benthic invertebrates in the past and present conditions. Each data point represents a community sample from one year and station.

Mainly crustacean species decreased in biomass in the course of the study period. Among the studied species, only *Monoporeia affinis* systematically declined its biomass in all the studied waterbodies. Nevertheless, the observed decline was minor compared to the increasing trends of the species mentioned above (SIMPER analysis).

No species had disappeared in the study area from the 1950s onwards. A few invasive species had established in the Estonian coastal sea since the 1970s. Most important invasive species were *Marenzelleria neglecta*, *Gammarus tigrinus*, *Chelicorophium curvispinum*, and *Pontogammarus robustoides* (SIMPER analysis). As the distribution area of those species often coincides with highly eutrophicated sites, those species are considered to gain from eutrophication. The results of all these analyses are summarized in Table 2, which lists all benthic invertebrate taxa observed in the Estonian coastal range and their sensitivity to human-induced eutrophication and organic enrichment.

The Estonian water quality classification system for surface waters is based on type-specific reference conditions. Thus, the index incorporates the waterbody-specific information on species composition. The diversity term takes into account the number of species at the station and compensates this diversity term for salinity gradients. The compensation term S_{max} is based on waterbody-specific maximum values for the number of species calculated from the entire content of the

Table 2. Sensitivity of benthic invertebrate species that inhabit the Estonian coastal sea. Species belonging to class 3 inhabit pristine conditions, species belonging to class 2 remain indifferent or gain biomass under moderate eutrophication, and species belonging to class 1 may be dominant over other species in heavily eutrophicated conditions

Taxon	Sensitivity	Taxon	Sensitivity
Alderia modesta	3	*Jaera albifrons*	3
Ancylus fluviatilis	3	*Laomedea flexuosa*	2
Argulus spp.	3	Lepidoptera	3
Asellus aquaticus	2	*Leptocheirus pilosus*	2
Astarte borealis	3	*Limapontia capitata*	3
Balanus improvisus	2	*Lymnaea peregra*	2
Bathyporeia pilosa	3	*Lymnaea stagnalis*	2
Bithynia tentaculata	3	*Macoma balthica*	2
Boccardia redeki	2	*Manayunkia aestuarina*	3
Bylgides sarsi	3	*Marenzelleria neglecta*	2
Calliopius laeviusculus	3	*Melita palmata*	3
Cerastoderma glaucum	2	*Monoporeia affinis*	3
Ceratopogonidae	2	*Mya arenaria*	2
Chelicorophium curvispinum	2	*Mysis mixta*	3
Chironomidae	1	*Mysis relicta*	3
Coleoptera	3	*Mytilus trossulus*	2
Collembola	3	*Neomysis integer*	2
Cordylophora caspia	2	Neuroptera	3
Corixa spp.	2	Odonata	3
Corophium volutator	2	Oligochaeta	1
Crangon crangon	3	*Orchestia cavimana*	3
Cyanophthalma obscura	3	Ostracoda	3
Diastylis rathkei	3	*Palaemon adspersus*	2
Diptera	3	*Palaemon elegans*	2
Dreissena polymorpha	2	*Paramysis intermedia*	2
Echinogammarus stoerensis	3	*Physa fontinalis*	3
Electra crustulenta	2	*Piscicola geometra*	3
Ephemeroptera	3	*Pisidium* spp.	3
Eurydice pulchra	3	Planorbidae	3
Gammarus duebeni	3	Plecoptera	3
Gammarus lacustris	3	*Pontogammarus robustoides*	2
Gammarus locusta	3	*Pontoporeia femorata*	3
Gammarus oceanicus	3	*Potamopyrgus antipodarum*	3
Gammarus pulex	3	*Praunus flexuosus*	3
Gammarus salinus	3	*Praunus inermis*	3
Gammarus zaddachi	3	*Pygospio elegans*	2
Gammarus tigrinus	2	*Saduria entomon*	3
Gonothyraea loveni	3	*Scoloplos armiger*	3
Halicryptus spinulosus	3	*Sphaerium* spp.	3
Hediste diversicolor	2	*Stagnicola palustris*	3
Hemiptera	3	*Tenellia adspersa*	3
Hirudinea	3	*Terebellides stroemi*	3
Hydrachnellae	3	*Theodoxus fluviatilis*	2
Hydrobia ulvae	2	Trichoptera	3
Hydrobia ventrosa	2	*Valvata macrostoma*	2
Idotea balthica	2	*Valvata piscinalis*	3
Idotea chelipes	3	*Viviparus viviparus*	2

national database (Table 3). The macrozoobenthos community index was calculated by the following equation:

$$\mathrm{ZKI} = [0.5 \times (\mathrm{Class}\ 1 + 2 \times \mathrm{Class}\ 2 + 3 \times \mathrm{Class}\ 3) - 0.5] \times \left[\frac{S}{S_{\max}} \right],$$

where Class i is the ratio of the sum of the dry weights of the species belonging to Class i to total invertebrate biomass at the station; S is the number of species/taxa per grab; $S_{\max}$ is the waterbody-specific value of the maximum number of species per grab.

Class 1 designates opportunistic taxa that are able to form single-species associations or highly dominate communities under heavily disturbed conditions. Class 2 includes taxa that are indifferent to or favoured by moderate eutrophication, but cannot tolerate heavily disturbed conditions. For taxa of Class 3, eutrophication is unfavourable.

The values of the ZKI vary between 0 and 1 and the index increases with the health of communities.

In all regions, the ZKI correlated with N and P loads from point sources (Table 4, Fig. 3) and also with P loads from the riverine sources in all areas except the Gulf of Finland (Spearman R, $p < 0.05$). In addition, the ZKI correlated with the BSPI separately in the Gulf of Finland and in the rest of the study area (Spearman R, $p < 0.05$, Fig. 4). When samples from the Gulf of Finland and the rest

Table 3. Maximum values of species number per grab ($S_{\max}$) by waterbodies

Water-body	Estonian name	English name	$S_{\max}$
1	Narva laht	Narva Bay	13
2	Käsmu-Kunda	Käsmu-Kunda	9
3	Hara laht	Hara Bay	7
4	Kolga laht	Kolga Bay	12
5	Tallinna piirkond	Tallinn sea area	19
6	Soome lahe lääneosa	Western Gulf of Finland	14
7	Läänesaarte põhjaosa	North of West Estonian Archipelago	16
8	Haapsalu laht	Haapsalu Bay	8
9	Matsalu laht	Matsalu Bay	11
10	Soela väin	Soela Strait	11
11	Saaremaa läänerannik	West of Saaremaa Island	14
12	Liivi laht	Gulf of Riga	19
13	Pärnu laht	Pärnu Bay	13
14	Kassari laht	Kassari Bay	20
15	Väike väin	Väike Strait	11
16	Väinameri	West Estonian Archipelago Sea	18

Table 4. Significant correlations between the ZKI and anthropogenic impact indicators

Region and type of human impact	Spearman *R*
Whole study area	
Point N	−0.17
Point P	−0.16
Gulf of Finland (waterbodies 1–5)	
Point N	−0.23
Point P	−0.26
BSPI	−0.18
Other areas except Gulf of Finland (waterbodies 6–16)	
Point N	−0.30
Point P	−0.26
Riverine P	−0.14
BSPI	−0.13

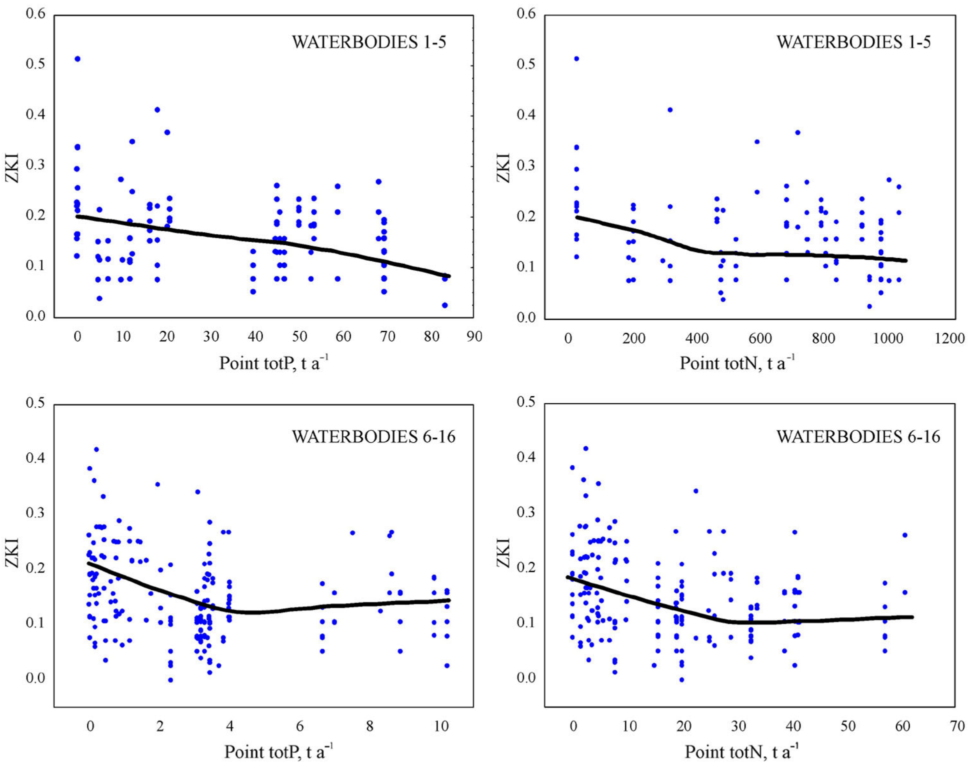


Fig. 3. Relationships between annual nutrient loads and the ZKI in the Gulf of Finland (waterbodies 1–5) and all other regions (waterbodies 6–16). Local polynomial regression line constructed using function loess in R (R Development Core Team, 2011) was added to the figures for visualization purposes.

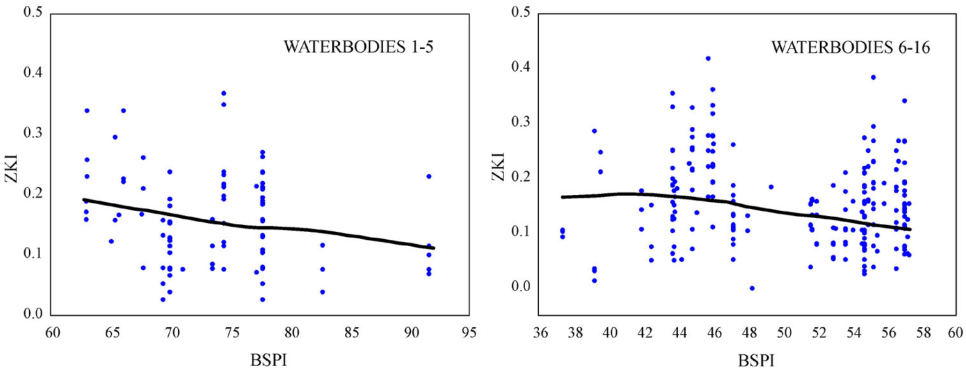


Fig. 4. Relationship between the BSPI and ZKI in the Gulf of Finland (waterbodies 1–5) and all other regions (waterbodies 6–16). Local polynomial regression line constructed using function loess in R (R Development Core Team, 2011) was added to the figures for visualization purposes.

of the study area were pooled, this effect disappeared. The large scatter is due to the interannual variation of benthic invertebrate communities. Although the BSPI is the best available human-induced pressure metric in the northern Baltic Sea region, it takes into account the spatial variability but not the temporal trends in pressures.

For the calculations of the Ecological Quality Ratio (EQR) of a waterbody, the ZKI values for all the replicates from the waterbody were calculated separately. Then these ZKI values were averaged, and the mean value was divided by the reference value 0.74. The reference value is the maximum ZKI value from a single data point in the historical data set.

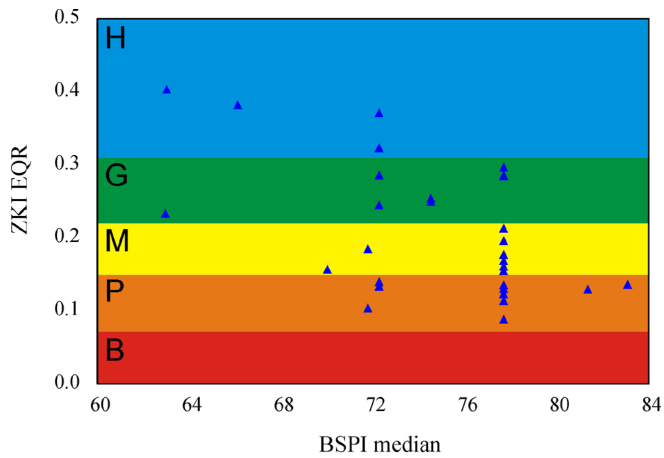
The quality status of macrozoobenthos of a waterbody was determined by comparing the EQR of macrozoobenthos quality element to the boundary values. The boundary values for the Estonian coastal sea were set as described in the Methods section and are as follows: 0.31 – borderline between the high and good water quality classes, 0.22 – good/moderate borderline, 0.15 – moderate/poor borderline, and 0.08 – poor/bad borderline.

There are certain criteria that need to be fulfilled for a correct macrozoobenthos quality assessment:

(1) The EQR calculation should be based on at least three grab sampling stations replicated three times.

(2) Stations should be representative of each waterbody.

The obtained ZKI EQR ranks correlated with the median BSPI values in the Gulf of Finland waterbodies 1 to 5 (Spearman $R = -0.4$, $p < 0.05$; Fig. 5). The BBI EQR ranks in these waterbodies did not correlate with the BSPI, and the BBI EQR values showed higher variability at the BSPI gradient than the ZKI EQR values (Spearman R, $p > 0.05$; Fig. 5).



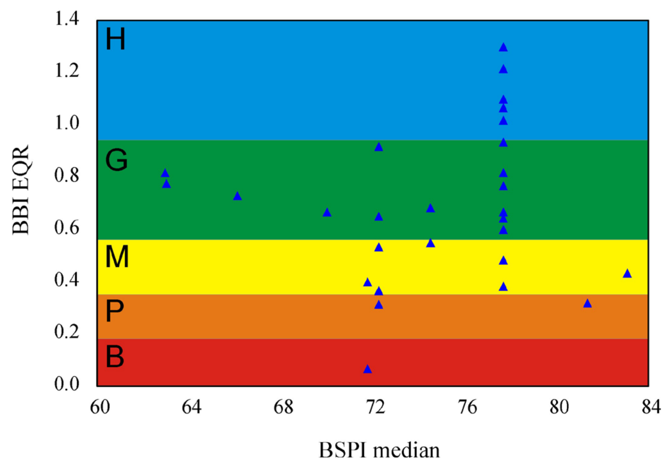


Fig. 5. ZKI-based and BBI-based Ecological Quality Ratio values (ZKI EQR and BBI EQR, respectively) from 1997 to 2010 at the Baltic Sea Pressure Index (BSPI) gradient in the southern Gulf of Finland, waterbodies 1–5 (*see* Fig. 1 for the location of the waterbodies). EQR classes are as follows: H – high, G – good, M – moderate, P – poor, B – bad.

DISCUSSION

In this study we presented and validated a new index and quality assessment system based on benthic invertebrates for the Estonian coastal sea according to the requirements of the WFD. The classification process was used to assign the status class of water quality to each surface waterbody. Such class value represents an estimate of the degree to which the benthic invertebrate communities have been altered by all the different pressures to which that body is subject. Thus, the current classification scheme reflects the impacts of a much wider range of pressures on the water environment than for example eutrophication.

Salinity gradient is widely known as a factor of utmost importance for the biota in the waterbodies of the Estonian coastal sea, and its impact on the performance of benthic indices has been already corroborated in the Baltic Sea scale (Zettler et al., 2007). In addition, previous studies have demonstrated that benthic invertebrates inhabiting frontal areas exhibit different relationships with nutrient loads than the communities inhabiting waterbodies with a longer residence time (Kotta et al., 2007). The inclusion of the waterbody-specific reference value into the ZKI equation accounts for such variability in hydrographical conditions and allows a comparison of the health of benthic invertebrate communities across different waterbodies.

Our study demonstrated the value of the long-term data set that the Estonian Marine Institute sustains, and such continuity provides a basis for understanding the spatio-temporal patterns of benthic invertebrate communities as well as developing and implementing water quality indices. Recent studies have shown that historically benthic invertebrate communities strongly responded to changes in abiotic variables such as salinity. Since the 1970s, however, the increasing levels of nutrients in seawater resulted in a large scatter of data on the salinity–biota plots (Kotta et al., 2004, 2007; Orav-Kotta et al., 2004). Moreover, in the most eutrophicated regions benthic invertebrates did not respond to the change in salinity any more, instead the species biomasses were a function of nutrient loading (Kotta et al., 2007, 2008). As benthic invertebrates represent an intermediate trophic level, the observed patterns are due to changed habitat quality as well as feeding conditions (e.g. Kotta & Møhlenberg, 2002; Orav-Kotta & Kotta, 2004). In general, increasing nutrient loads lead to algal blooms and intensified sedimentation of organic material (Paalme et al., 2002). Improved food conditions promote higher invertebrate biomasses (Kotta & Ólafsson, 2003, Lauringson & Kotta, 2006). Too high a nutrient loading, however, results in hypoxia (Conley et al., 2011) and regime shifts in communities (Karlson et al., 2002).

It is interesting to note though that even at the highest loads of nutrients benthic invertebate communities in the Estonian coastal sea remained largely unchanged in terms of species composition, and there were almost no species that could be ranked as sensitive to disturbance. It is widely acknowledged that the coastal ecosystem of the Baltic Sea is very dynamic and characterized by high physical disturbances (Bonsdorff et al., 1996; Kotta et al., 2008). Therefore, it is likely that stress tolerant species of the Baltic Sea can easily cope with various disturbances including eutrophication. Although increasing nutrient loads resulted in elevated biomasses of mussels, our study revealed no clear shifts in communities (i.e. the disappearance of most species and the dominance of chironomid larvae), as observed along the coasts of Finland and Sweden (e.g. Rosenberg, 1985; Bonsdorff et al., 1997a, 1997b). Thus, the loading of organic matter poses no clear threat to benthic invertebrate communities unless hypoxia or anoxia is developed. Although the coastal zone of the Baltic Sea displays a widespread unprecedented occurrence of hypoxia and an alarming trend with hypoxia steadily increasing with time since the 1950s, the Estonian coastal sea seems exceptional in terms of oxygen dynamics (Conley et al., 2011). Plausibly, the strong ventilation by bottom currents keeps sediments in a healthy state.

In the 1990s and 2000s increasing intensity of the establishment of non-indigenous species was observed in the Estonian coastal sea (Ojaveer et al., 2011). All these invasive species displayed either stable or abrupt increases in biomass over time. Some of the invasive species led to prominent structural changes in invaded communities and a considerable reduction or even local disappearance of native species (Kotta et al., 2001, 2006, 2010; Kotta & Ólafsson, 2003). Considering the ephemeral character of most invasive species, the observed increasing trend in the establishment of non-indigenous species poses a serious threat to the health and stability of benthic invertebrate communities. Unfortunately, this risk cannot be reduced using local measures such as counteracting the effect of eutrophication or trying to eradicate already established invasive aquatic species.

The boundary values set using the lower 20th percentile of the distribution might arguably mean that we draw the H/G and G/M boundaries (and thus also the other two) too low. On the other hand, for example our H/G boundary is such that in one occasion out of five we are likely to classify the waterbody into a lower category than it actually belongs to. It would, of course, be desirable to be able to estimate the probability of failing to classify a waterbody into one of the three lower categories when it in fact belongs there, but due to the nature of the data such an estimation cannot be made as we do not know which waterbodies actually belong to these classes.

At the highest BSPI values, the EQR showed results ranging from poor to good quality in our data set. This can raise a justified question about the accuracy of the water quality estimation provided by the ZKI. However, EQR values based on another index used in the northeastern Baltic Sea (BBI) show even a wider range of variability (Fig. 5). We think that no zoobenthic index for the northeastern Baltic Sea could be considered a comparably sensitive tool to any zoobenthic index developed for and used in more saline or freshwater areas.

A serious reduction in the number of species can be observed while moving eastward along the salinity gradient in the Baltic Sea. Besides, the species that survive in the eastern Baltic Sea do not represent a random subset but are the most eurytopic and tolerant selection of taxa. Such selection cannot be assumed to display a similar sensitivity towards anthropogenic disturbances as substantially more species-rich associations from euhaline areas.

Another problem is that zoobenthic water quality indices have been shown to be sensitive to natural disturbances even at fully marine, species-rich, and thereby supposedly much more sensitive systems than the northeastern Baltic Sea. For example, index assessment results have been shown to fluctuate over a range of 3 (out of 5) quality classes in a relatively unimpacted site in the southern North Sea in relation to climatic variability (Kröncke & Reiss, 2010). Thus, we definitely cannot assume the communities in the present study area, which are composed of a selection of the most eurytopic taxa, to be driven solely by anthropogenic factors.

Indeed, benthic communities in the northeastern Baltic Sea are shown to be related to both the variability in nutrient loads and in climatic factors (Veber et al., 2009; Lauringson et al., 2012). The community composition may also depend on difficult-to-predict factors such as stochastic settlement events, drifting detached annual algal accumulations, and patchy predation by vertebrate (fish or waterfowl) predators.

In addition, most of the present study area has a very good water exchange, even at locally polluted sites, which further impairs the linkage between anthropogenic drivers and benthic community composition. Also, the spatial location relative to pressure, described by the BSPI, represents a disturbance indicator temporally integrated over several years, while anthropogenic factors display yearly fluctuations, adding to the noise in the present analysis. However, despite the climatic forces, the fluctuations in anthropogenic pressures and the variability from other/unknown sources, a relationship between the BSPI and the community composition relative to sensitivity groups still occurred in the study area. The high variability in the ZKI values was also evident in the historical data set used as a reference in the present study. This hints at the necessity to take into account the natural variability of communities while deriving or modelling alternative reference conditions, especially for dynamic and physiologically challenging environments like the Baltic Sea.

The ZKI performed better in the study area than the BBI. The different performance of these two indices may result from differences between Finnish and Estonian environmental conditions, including both salinity levels and coastline complexity. A similar level of pollution may for example result in an increase in benthic biomass due to the high biomass of the bivalve *Macoma balthica* at flat, shallow, and oxygen-rich sea areas characteristic of the present study area (Kotta & Kotta, 1995) but lead to the formation of azoic sediments or a small remaining biomass of chironomid larvae in bottom cavities of enclosed bays at the highly mosaic coastline of the Finnish Archipelago area (Leppäkoski, 1975; Bonsdorff et al., 1991). As the ZKI is based on invertebrate biomasses not abundances, it is possible that the lack of truly sensitive indicator species is slightly compensated for by the increase in the biomass (but not necessarily the abundance) of pollution-tolerant species like *M. balthica* in the middle range of the pollution gradient common in the study area. Faunal responses to pollution in Finnish and Estonian coastal areas with similar types of coastline may differ due to different salinity levels and also depending on the origin of the taxa (e.g. Leppäkoski, 1975; Fleischer & Zettler, 2009). Salinity differences may become important also technically, as the waterbody-specific species richness is taken into account in the ZKI formula, while national type area-specific species richness is taken into account in the calculation of the BBI EQR values. A spatially large national type area may, in turn, cover an important spatial salinity gradient. Different natural stressors have led to the understanding that the consequences of eutrophication can take different pathways in different parts of the Baltic Sea, meaning that the Baltic Sea cannot be regarded as a uniform waterbody in modelling the consequences of human impacts (Rönnberg & Bonsdorff, 2004).

Although the BSPI seems to describe satisfactorily the state of the environment within the Gulf of Finland and within the rest of the Estonian coastal sea separately, comparison of biological data between these two regions shows an apparent mismatch in the BSPI values. Indeed, in the coastal area of the Gulf of Finland, the status of benthic communities is good and even high at BSPI values above 60. Such pressure values are hardly met in the coastal sea areas outside the Gulf of Finland. Therefore, we suggest that the values of the BSPI at the southern side of the Gulf of Finland may be unjustifiably high, which will cause problems if we wish to translate these human pressure values to the human impact on biological communities. As a difference from an assumption of the BSPI, the loads from the Neva River are less important at the southern than at the northern side of the gulf, or alternatively, the loads may be mostly accumulating in the deeper area and affect less the coastal biota at both sides of the gulf. Here we suggest that BSPI values within and outside the Gulf of Finland in the Estonian coastal sea form two different groups, which should be treated separately when comparing the BSPI with biological data, including the validation of biological water quality indicators.

Our study demonstrates that benthic invertebrate communities (in terms of EQR values) correlate with the integrated anthropogenic pressure metrics BSPI and nutrient loads in the study area. This suggests that the ZKI captures the variability of the environmental health. Considering the large variability of the abiotic environment in the Baltic Sea, however, it is acknowledged that the ZKI responds besides human-induced pressures to other stressors such as climate change (Lauringson et al., 2012). The relative magnitude in the human- and naturally-induced variability needs to be resolved in future studies as assessments are likely to become misinterpreted if knowledge about the natural variability is lacking.

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REFERENCES

Bolliger, J. & Mladenoff, D. J. 2005. Quantifying spatial classification uncertainties of the historical Wisconsin landscape (USA). *Ecography*, **28**, 141–156.

Bonsdorff, E. & Pearson, T. H. 1999. Variation in the sublittoral macrozoobenthos of the Baltic Sea along environmental gradients; a functional-group approach. *Aust. J. Ecol.*, **24**, 312–326.

Bonsdorff, E., Aarnio, K. & Sandberg, E. 1991. Temporal and spatial variability of zoobenthic communities in archipelago waters of the Northern Baltic Sea – consequences of eutrophication? *Int. Rev. Hydrobiol.*, **76**, 433–449.

Bonsdorff, E., Diaz, R. J., Norkko, A. & Cutter, G. R. 1996. Characterization of soft-bottom benthic habitats of the Åland Islands, northern Baltic Sea. *Mar. Ecol. Prog. Ser.*, **142**, 235–245.

Bonsdorff, E., Blomqvist, E. M., Mattila, J. & Norkko, A. 1997a. Coastal eutrophication: causes, consequences and perspectives in the archipelago areas of the northern Baltic Sea. *Estuar. Coast. Shelf Sci.*, **44** (Suppl. A), 63–72.

Bonsdorff, E., Blomqvist, E. M., Mattila, J. & Norkko, A. 1997b. Long-term changes and coastal eutrophication. Examples from the Åland Islands and the Archipelago Sea, northern Baltic Sea. *Oceanol. Acta*, **20**, 319–328.

Borja, A., Miles, A., Occhipinti-Ambrogi, A. & Berg, T. 2009. Current status of macroinvertebrate methods used for assessing the quality of European marine waters: implementing the Water Framework Directive. *Hydrobiologia*, **633**, 181–196.

Clarke, K. R. & Gorley, R. N. 2006. *Primer v6. User Manual/Tutorial*. Primer-E, Plymouth.

Conley, D. J., Carstensen, J., Aigars, J., Axe, P., Bonsdorff, E., Eremina, T., Haahti, B.-M., Humborg, C., Jonsson, P., Kotta, J., Lännegren, C., Larsson, U., Maximov, A., Rodriguez Medina, M., Lysiak-Pastuszak, E., Remeikaité-Nikiené, N., Walve, J., Wilhelms, S. & Zillén, L. 2011. Hypoxia is increasing in the coastal zone of the Baltic Sea. *Environ. Sci. Technol.*, **45**, 6777–6783.

Dauvin, J.-C. 2007. Paradox of estuarine quality: benthic indicators and indices, consensus or debate for the future. *Mar. Pollut. Bull.*, **55**, 271–281.

Elliott, M. & Quintino, V. 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. *Mar. Pollut. Bull.*, **54**, 640–645.

Eriksson, B. K., Johansson, G. & Snoeijs, P. 1998. Long-term changes in the sublittoral zonation of brown algae in the southern Bothnian Sea. *Eur. J. Phycol.*, **33**(3), 241–249.

European Commission. 2000. Directive2000/60/EC of the European Parliament and the Council of 23 October 2000 establishing a framework for community action in the field of water policy. *OJL*, **327**, 1–72.

Fleischer, D. & Zettler, M. L. 2009. An adjustment of benthic ecological quality assessment to effects of salinity. *Mar. Pollut. Bull.*, **58**, 351–357.

HELCOM. 2010. 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.*, **125**, 1–72.

Järvekülg, A. 1970. Bottom fauna as an indicator of pollution of the marine benthal in the vicinity of Tallinn. In *Estonian Contributions to the International Biological Programme*, vol. 1, 158–193. Academy of Sciences of the Estonian S.S.R., Tartu.

Josefson, A. B., Blomqvist, M., Hansen, J. L. S., Rosenberg, R. & Rygg, B. 2009. Assessment of marine benthic quality change in gradients of disturbance: comparison of different Scandinavian multi-metric indices. *Mar. Pollut. Bull.*, **58**, 1263–1277.

Karlson, K., Rosenberg, R. & Bonsdorff, E. 2002. Temporal and spatial large-scale effects of eutrophication and oxygen deficiency on benthic fauna in Scandinavian and Baltic waters – a review. *Oceanogr. Mar. Biol. Ann. Rev.*, **40**, 427–489.

Kotta, J. & Kotta, I. 1995. The state of macrozoobenthos of Pärnu Bay in 1991 as compared to 1959–1960. *Proc. Estonian Acad. Sci., Ecol.*, **5**, 26–37.

Kotta, J. & Møhlenberg, F. 2002. Grazing impact of *Mytilus edulis* and *Dreissena polymorpha* (Pallas) in the Gulf of Riga, Baltic Sea estimated from biodeposition rates of algal pigments. *Ann. Zool. Fenn.*, **39**, 151–160.

Kotta, J. & Ólafsson, E. 2003. Competition for food between the introduced exotic polychaete *Marenzelleria viridis* and the resident native amphipod *Monoporeia affinis* in the Baltic Sea. *J. Sea Res.*, **342**, 27–35.

Kotta, J., Orav, H. & Sandberg-Kilpi, E. 2001. Ecological consequence of the introduction of the polychaete *Marenzelleria viridis* into a shallow water biotope of the northern Baltic Sea. *J. Sea Res.*, **46**, 273–280.

Kotta, J., Simm, M., Kotta, I., Kanošina, I., Kallaste, K. & Raid, T. 2004. Factors controlling long-term changes of the eutrophicated ecosystem of Pärnu Bay, Gulf of Riga. *Hydrobiologia*, **514**, 259–268.

Kotta, J., Kotta, I., Simm, M., Lankov, A., Lauringson, V., Põllumäe, A. & Ojaveer, H. 2006. Ecological consequences of biological invasions: three invertebrate case studies in the north-eastern Baltic Sea. *Helgol. Mar. Res.*, **60**, 106–112.

Kotta, J., Lauringson, V. & Kotta, I. 2007. Response of zoobenthic communities to changing eutrophication in the northern Baltic Sea. *Hydrobiologia*, **580**, 97–108.

Kotta, J., Lauringson, V., Martin, G., Simm, M., Kotta, I., Herkül, K. & Ojaveer, H. 2008. Gulf of Riga and Pärnu Bay. In *Ecology of Baltic Coastal Waters* (Schiewer, U., ed.), pp. 217–243. Ecol. Stud., 197. Springer.

Kotta, J., Herkül, K., Kotta, I., Orav-Kotta, H. & Aps, R. 2009. Response of benthic invertebrate communities to the large-scale dredging of Muuga Port. *Estonian J. Ecol.*, **58**, 286–296.

Kotta, J., Orav-Kotta, H. & Herkül, K. 2010. Separate and combined effects of habitat specific fish predation on the survival of invasive and native gammarids. *J. Sea Res.*, **64**, 369–372.

Kovtun, A., Torn, K. & Kotta, J. 2009. Long-term changes in a northern Baltic macrophyte community. *Estonian J. Ecol.*, **58**, 270–285.

Kröncke, I. & Reiss, H. 2010. Influence of macrofauna long-term natural variability on benthic indices used in ecological quality assessment. *Mar. Pollut. Bull.*, **60**, 58–68.

Lauringson, V. & Kotta, J. 2006. Influence of the thin drift algal mats on the distribution of macrozoobenthos in Kõiguste Bay, NE Baltic Sea. *Hydrobiologia*, **554**, 97–105.

Lauringson, V., Kotta, J., Kersen, P., Leisk, Ü., Orav-Kotta, H. & Kotta, I. 2012. Use case of biomass-based benthic invertebrate index for brackish waters in connection to climate and eutrophication. *Ecol. Indic.*, **12**, 123–132.

Leonardsson, K., Blomqvist, M. & Rosenberg, R. 2009. Theoretical and practical aspects on benthic quality assessment according to the EU Water Framework Directive – examples from Swedish waters. *Mar. Pollut. Bull.*, **58**, 1286–1296.

Leppäkoski, E. 1975. Assessment of degree of pollution on the basis of macrozoobenthos in marine and brackish-water environments. *Acta Acad. Aboensis, Ser. B Math. Phys.*, **35**, 1–90.

Munari, C. & Mistri, M. 2010. Towards the application of the Water Framework Directive in Italy: assessing the potential of benthic tools in Adriatic coastal transitional ecosystems. *Mar. Pollut. Bull.*, **60**, 1040–1050.

Ojaveer, H., Kotta, J., Põllumäe, A., Põllupüü, M., Jaanus, A. & Vetemaa, M. 2011. Alien species in a brackish water temperate ecosystem: annual-scale dynamics in response to environmental variability. *Environ. Res.*, **111**, 933–942.

Orav-Kotta, H. & Kotta, J. 2004. Food and habitat choice of the isopod *Idotea baltica* in the north-eastern Baltic Sea. *Hydrobiologia*, **514**, 79–85.

Orav-Kotta, H., Kotta, J. & Kotta, I. 2004. Comparison of macrozoobenthic communities between the 1960s and the 1990s–2000s in the Väinameri, NE Baltic Sea. *Proc. Estonian Acad. Sci. Biol. Ecol.*, **53**, 283–291.

Paalme, T., Kukk, H., Kotta, J. & Orav, H. 2002. "In vitro" and "in situ" decomposition of nuisance macroalgae *Cladophora glomerata* and *Pilayella littoralis*. *Hydrobiologia*, **475/476**, 469–476.

Pearson, T. H. & Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. Mar. Biol. Ann. Rev.*, **16**, 229–311.

Perus, J., Bonsdorff, E., Bäck, S., Lax, H. G., Villnäs, A. & Westberg, V. 2007. Zoobenthos as indicators of ecological status in coastal brackish waters: a comparative study from the Baltic Sea. *Ambio*, **36**, 250–256.

R Development Core Team. 2011. *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria (http://www.R-project.org/) (visited 15.09.2011).

Rönnberg, C. & Bonsdorff, E. 2004. Baltic Sea eutrophication: area-specific ecological consequences. *Hydrobiologia*, **514**, 227–241.

Rosenberg, R. 1985. Eutrophication – the future marine coastal nuisance? *Mar. Pollut. Bull.*, **16**, 227–231.
Southworth, J., Munroe, D. & Nagendra, H. 2004. Land cover change and landscape fragmentation – comparing the utility of continuous and discrete analyses for a western Honduras region. *Agric. Ecosyst. Environ.*, **101**, 185–205.
StatSoft, Inc. 2011. *Electronic Statistics Textbook*. StatSoft, Tulsa, OK (http://www.statsoft.com/textbook/) (visited 15.09.2011).
Torn, K. & Martin, G. 2011. Assessment method for the ecological status of Estonian coastal waters based on submerged aquatic vegetation. *WIT Trans. Ecol. Environ.*, **150**, 443–452.
Veber, T., Kotta, J., Lauringson, V. & Kotta, I. 2009. Influence of the local abiotic environment, weather and regional nutrient loading on macrobenthic invertebrate feeding groups in a shallow brackish water ecosystem. *Oceanologia*, **51**, 541–559.
Voipio, A. (ed.). 1981. *The Baltic Sea*. Elsevier Oceanography Series, **30**. Elsevier, Burlington, MA.
Zettler, M. L., Schiedek, D. & Bobertz, B. 2007. Benthic biodiversity indices versus salinity gradient in the southern Baltic Sea. *Mar. Pollut. Bull.*, **55**, 258–270.

Eesti rannikumere vee kvaliteedi klassipiiride määratlemine suurselgrootute alusel

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EL Veepoliitika Raamdirektiivi kohaselt tuleb liikmesriikidel hinnata oma veekogude seisundit ja luua seire jaoks klassifikatsioonisüsteem. Eesti rannikumere seisundi hindamissüsteemi loomiseks võrreldi tänapäevaseid põhjaloomastiku kooslusi ajalooliste, 1950.–1960. aastate kooslustega. Suurselgrootute tundlikkus määrati taksonite kaupa, töötati välja põhjaloomastiku koosluse indeks ZKI ja seati hindamissüsteemi aluseks olevad klassipiirid. ZKI sobivust hinnati seoses lämmastiku ja fosfori koormuste ning piirkondliku häiringuga Läänemere koormusindeksi BSPI (Baltic Sea Pressure Index) põhjal. Võrreldi ZKI indeksil ja Soome BBI indeksil põhinevaid ökoloogilise kvaliteediseisundi (ÖKS) hinnanguid Soome lahes seoses piirkondliku häiringu tugevusega vastavalt BSPI näitajale. ZKI ÖKS-i hinnangud kõikusid uurimisalal vähem ja olid piirkondliku häiringuga tugevamalt seotud kui BBI ÖKS-i hinnangud.