# Macrozoobenthos structure in relation to environmental changes in the Archipelago Sea, northern Baltic Sea

Jari Hänninen and Ilppo Vuorinen

Archipelago Research Institute, University of Turku, FIN-20014 Turku, Finland

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Since the 1960s, the major environmental change affecting the water quality of the Baltic Sea has been eutrophication. Several types of effects were attributed to increasing eutrophication in the benthic communities. In the present study, we describe the soft bottom benthic assemblages based on species number, abundance and biomass in the Airisto Inlet in 1994, and analyse the changes of community structure in relation to major environmental changes in the Archipelago Sea area since the 1950s. Special emphasis was put on alterations of Macoma balthica and Monoporeia affinis proportions. Our results provided evidence of a general increase of benthic macrofauna, especially in the middle and southern parts of the study area. The greatest relative increase seemed to occur to polychaetes and oligochaetes, whereas M. affinis showed the greatest absolute increase. However, the relative proportions of M. balthica and M. affinis generally remained unchanged. We conclude that, in the middle and southern parts, the changes observed in macrozoobenthos were due to general eutrophication in the Archipelago Sea. In the northern parts, the communities have been remarkably influenced by local pollution and dredging.

# Introduction

Since the 1960s, eutrophication has been the major environmental change affecting the water quality of the Baltic Sea both basin-wide and locally (e.g. Bonsdorff *et al.* 1991, Wulff *et al.* 1994). Several types of effects were previously attributed to increasing eutrophication in the benthic communities in the Baltic Sea, both in

the main basin and in shallow areas. Cederwall and Elmgren (1980) showed that macrofaunal abundance and biomass have increased significantly in the central Baltic proper since the 1920s. Leppäkoski (1975), Pearson and Rosenberg (1978), and later again Bonsdorff *et al.* (1991) have found that, depending on the earlier degree of disturbance in the seabed, caused by eutrophication, the alteration in the benthic macrofauna in shallow waters could be expressed as a functional (i.e. reduced complexity in terms of diversity and evenness) or a structural response (increased abundance and biomass of species). The principal mechanism for changes is considered to be the increased primary production, leading to higher organic content in sediments (improving effect), and at a certain level, resulting in temporal oxygen deficiency in near bottom waters (injurious effect). Moreover, another kind of effect has also been found. Norkko and Bonsdorff (1996) showed that increased algal cover on the bottom (i.e. accumulations of drifting algae), and induction of hypoxia through degradation of the algae, exhibited severe effects on the benthic community structure and a potential to accelerate local eutrophication.

The present study aims at describing the soft bottom macrobenthic assemblages based on species number, abundance and biomass of benthic communities in the Airisto Inlet in 1994, and analysing the changes of community structure in relation to major environmental changes in the Archipelago Sea area (eutrophication, pollution, dredging) since the 1950s. Before this work, the seminal study of Tulkki (1960) in the Airisto Inlet is the only one to cover both the inner and middle archipelago zones simultaneously. The principal idea has been to revisit sampling stations previously investigated by Tulkki in 1956 and to record overall benthic changes in time and space.

## Materials and methods

#### Study area

The study area, the Airisto Inlet, is situated mainly in the innermost archipelago (the southern parts verge on the middle archipelago), south-west of the city of Turku (Fig. 1). The central basin (area 246 km<sup>2</sup>) is mostly shallow (mean depth 20 m), but in the north-southerly direction, deeper fracture lines (50–60 m) form channels on the seabed. In the main basin, bottom sediments consist of Ancylus clay (Heino 1973). Some small rivers discharge into the sea, the largest being the Aurajoki (mean flow = 8.5 m<sup>3</sup> s<sup>-1</sup>; Pitkänen 1994). Average monthly

discharge of the rivers varies seasonally with peak loads in spring and autumn (Anonymous 1998). Salinity in the area varies between 3.5 and 7.0 PSU throughout the water column (e.g. Vuorinen and Ranta 1987, Hietaranta 1990, Viitasalo *et al.* 1990) following the surface water salinity in the northern Baltic Sea (Mälkki and Tamsalu 1985). The temperature of the sea surface ranges between 0–20 °C, the maximum being in August. Formation of permanent ice cover usually starts in December–January and the final disappearance of ice takes place in April (HELCOM 1993, 1996).

The great majority of nutrients, sediment and organic matter come as non-point-source loading with the river discharges (Pitkänen 1994, Hänninen et al. 1999). The main sources of excess nutrients have been industrial and municipal wastewater, forestry and agriculture (Jumppanen and Mattila 1994, Bonsdorff et al. 1997a, 1997b). During the last 20 years, fish farming has also exerted a remarkable influence on water quality in the middle and south areas (e.g. Bonsdorff et al. 1997a, Hänninen et al. 1999). The share of airborne nutrients and nutrients imported by currents from other parts of the Baltic Sea (Gulf of Finland, Gulf of Bothnia, Baltic Proper) have only recently been estimated (Jumppanen and Mattila 1994, Kirkkala et al. 1998, Helminen et al. 1998). However, the oxygen content of the water is usually high and only in the deepest areas does a deficiency of oxygen occur in some years (Jumppanen and Mattila 1994).

#### Field sampling

The sampling strategy was to revisit a number of sampling stations previously investigated by Tulkki (1960) in autumn 1956. In this earlier work, 65 stations were investigated for soft bottom benthic animals and some environmental factors all over the Airisto Inlet area. We used these stations as a material for cluster analysis (average linkage method with distance metric of Pearson correlation coefficient; SPSS 1997) to divide the study area into sub-areas on the basis of similarity in benthic community structure. This was done to decrease the unexplained vari-



**Fig. 1**. The study area in the Airisto Inlet (the Archipelago Sea, northern Baltic Sea) with sampling localities (1-30) and subareas (I-X) based on cluster analysis (see text for more accurate description).

ation in data in later analyses. Our criteria for division was that the maximum distance for similarity should exceed predestined value 0.80. With 10 sub-areas this value was 0.876. Because of limited resources, only three of the stations used by Tulkki were resampled at random in each sub-area (Fig. 1). Sampling was done in September–October 1994 according to the methods of Tulkki, concentrating on soft bottoms in the 1–50 meter depth zone. Five replicate grab samples per site were taken with a same Ekman–Birge type hand-operated box corer (sample area 231cm<sup>2</sup>) as Tulkki used. Samples were sieved on a 1.0 mm mesh size screen, and stored in buffered 4% seawater–formaline solution. All animals were determined to species level, counted under a dissecting microscope, and their wet weight was measured to the nearest 0.01 g in a laboratory.

#### Statistical analyses

In order to test for overall changes in temporal (between 1956 and 1994) and spatial (within sub-areas in 1956 and 1994) distribution of the zoobenthos (number of species, species abundance and biomass), a non-parametric Kruskall-

Wallis ANOVA was performed (SPSS 1997). In analysis we concentrated on only 'proper' species, which during their life span live chiefly in more or less soft deposits. Therefore e.g. species Prostoma obscurum and Gammarus spp. were excluded because those species inhabit mainly littoral or littoriprofundal zone usually occupied by loose red and brown algae hindering the quantitative sampling of the box corer. Because the numbers within species were in most cases insufficient for reliable tests, we pooled the species data sets to subcategories according to the taxonomic levels. Only numbers of the most abundant species, the bivalve Macoma balthica and the amphipod Monoporeia affinis, were sufficient for testing on species level. Spatial distribution of biomass was investigated only in 1994 because data exist only for that period. When needed, Bonferroni adjusted Mann-Whitney U-test was used in single comparisons of differences between sub-areas in both periods (SPSS 1997).

Logistic regression analysis (SAS 1995, 1997) was used for analysing the changes in dominant species proportion between 1956 and 1994. This analysis belongs to a 'family' of generalised linear models where, e.g. the familiar analysis of variance is a special case with an assumption of normal distribution. One of the greatest advantages of generalised linear models is that they are often useful with such non-normal data sets (e.g. Poisson or Binomial distribution), which are not possibly normalised correctly with transformations, because of e.g. several zero values. In the present study, we used binomial distribution and logit function as a link function. The response variable of interest was the sample proportion of the dominant species in the total benthic community. The dominant species were defined as the species, which usually have the largest biomass and are the most abundant in the area, i.e. Macoma balthica and Monoporeia affinis. The species were analysed separately. We used a 'case studies' model structure, which is

**Table 1**. The collected zoobenthos species or groups in 1956 and 1994. The asterisk indicates the species not involved in the analyses, i.e. not 'proper' soft bottom species (*see* text). The species names used here are according to current names.

1956	1994	1956	1994
Nemertini		Bivalvia	
-	Prostoma obscurum*	Cerastoderma glaucum	Cerastoderma glaucum
Nematoda		Mya arenaria	Mya arenaria
_	Nematoda*	Macoma balthica	Macoma balthica
Oligochaeta		Mytilus edulis	Mytilus edulis
Tubifex tubifex	Clitellio arenarius	Cirripedia	-
	Limnodrilus hoffmeisteri	Balanus improvisus*	Balanus improvisus*
	Potamothrix hammoniensis	Amphipoda	
	Peloscolex heterochaetus	Corophium volutator	Corophium volutator
	Stylaria lacustris	Monoporeia affinis	Monoporeia affinis
	Tubifex costatus	Gammarus sp.*	Pontoporeia femorata
Polychaeta			Gammarus salinus*
Harmothoe sarsi	Harmothoe sarsi	Isopoda	
Nereis diversicolor	Nereis diversicolor	Asellus aquaticus*	Saduria entomon
	Marenzelleria viridis	Idotea balthica*	
Priapulida		Saduria entomon	
Halicryptus spinulosus	Halicryptus spinulosus	Mysidacea	
Gastropoda		_	Mysis mixta*
Hydrobia ulvae*	Bithynia tentaculata*		Mysis relicta*
Theodoxus fluviatilis*	Hydrobia ulvae*		Neomysis integer*
	Hydrobia ventrosa*	Diptera	
	Potamopyrgus jenkinsi*	Chironomidae	Chironomus plumosus
			Chironomini spp.
			Tanypodidae

suitable for analysing the change in 'before–after' trials (SAS 1997). The benthic community structure in 1994 was treated as the response and the situation in 1956 as the baseline, which was used in analysis as a covariate. The sub-area was considered a fixed-effect factor and the station effect (nested under the sub-area) was introduced into the models as a random effect. The depth and the total abundance were also used as covariates. Before the actual analysis, the situation in 1956 was tested separately to clarify the species proportion at the baseline level. In this analysis, the model structure was identical to the actual analysis, apart from the baseline covariate. ESTI-MATE statements were used to determine the differences in species proportions between subareas (SAS 1997). Sattherthwite approximation for degrees of freedom was used. All the analyses were done with GLIMMIX; Generalised Linear Mixed Models macro in SAS (1997).

## Results

A list of collected benthic species in 1956 and 1994 is presented in Table 1. The overall complexity of the benthic community increased in the form of a significantly higher number of species in 1994 (Table 2:  $\chi^2 = 34.23$ , df = 1, p < 0.001). The species number among sub-areas

**Table 2**. Descriptive statistics and Kruskall-Wallis ANOVA results for comparisons of dominant groups: Species number and abundance (ind.  $m^{-2}$ ) between 1956 and 1994, species number and abundance (ind.  $m^{-2}$ ) within 1956 and 1994, and biomass (g wwt.  $m^{-2}$ ) within 1994.

	n	Mean	SD	$\chi^2$	df	p
1956 vs. 1994						
No. of species	60	5.0	2.6	34.23	1	< 0.001
Abundance (ind. m <sup>-2</sup> )						
Total	60	1874.7	2204.3	27.09	1	< 0.001
Polychaeta	60	44.7	78.6	22.87	1	< 0.001
Amphipoda	60	1401.5	2042.3	13.33	1	< 0.001
Chironomidae	60	10.5	21.6	0.98	1	0.324
Oligochaeta	60	47.7	144.9	4.32	1	0.038
M. balthica	60	289.2	336.9	34.13	1	< 0.001
M. affinis	60	1394.3	2037.5	12.19	1	< 0.001
Within 1956						
No. of species	30	3.1	1.2	17.68	9	0.039
Abundance (ind. m <sup>-2</sup> )						
Total	30	563.1	747.2	25.26	9	0.003
Polychaeta	30	7.8	21.9	19.60	9	0.021
Amphipoda	30	334.8	698.4	23.27	9	0.006
Chironomidae	30	7.5	17.4	8.23	9	0.512
Oligochaeta	30	8.4	20.8	28.56	9	0.001
M. balthica	30	91.8	90.6	20.89	9	0.013
M. affinis	30	331.9	699.1	25.23	9	0.003
Within 1994						
No. of species	30	7.0	2.2	16.49	9	0.057
Biomass (g wwt. m <sup>-2</sup> ) Abundance (ind. m <sup>-2</sup> )	30	109.2	79.9	16.79	9	0.052
Total	30	3186.3	2401.6	21.65	9	0.010
Polychaeta	30	81.6	96.2	10.36	9	0.322
Amphipoda	30	2468.2	2375.6	22.07	9	0.009
Chironomidae	30	13.5	24.9	16.41	9	0.059
Oligochaeta	30	87.1	197.6	19.78	9	0.019
M. balthica	30	486.6	376.9	21.26	9	0.012
M. affinis	30	2456.6	2371.1	22.04	9	0.009

14 12 Number of species 10 8 6 4 2 8000 (m<sup>2</sup>) 7000 ind. 6000 abundance 5000 4000 3000 **Total** 2000 1000 400 Fotal biomass (g wwt./m<sup>2</sup>) 350 300 250 200 150 100 50 0 п Ш VII VIII IX х IV v νı Sub-areas -o− Tulkki'56 Present study



differed significantly in 1956 ( $\chi^2 = 17.68$ , df = 9, p = 0.039), but only marginally significantly in 1994 ( $\chi^2 = 16.49$ , df = 9, p = 0.057). In both periods the highest numbers were generally found in northern Airisto (Fig. 2).

Correspondingly, the total abundance of the zoobenthos had increased significantly between the periods ( $\chi^2 = 27.09$ , df = 1, p < 0.001) and was roughly sixfold in 1994 (1956: mean = 563.1 ind. m<sup>-2</sup>, 1994: mean = 3186.3 ind. m<sup>-2</sup>). Although the increase was generally evident for almost all of the groups (as pooled the Chironomidea were the only exception, but some indication of higher proportions in the innermost areas were evident), the greatest relative increase



**Fig. 3.** Total, *Macoma balthica* and *Monoporeia affinis* abundances (ind.  $m^{-2}$ ) in 1956 and 1994. The boxes represent the interquartile ranges, which contain 50% of values. The whiskers extending from the boxes indicate the highest and lowest values. The line across the boxes show the medians.

seemed to occur for polychaetes and oligochaetes (~ tenfold for both groups), for latter especially in the innermost areas. To a great extent, the increase in polychaetes was due to appearing of the introduced polychaete, *Marenzelleria viridis*. For the most abundant species, *Macoma balthica* and *Monoporeia affinis*, the abundance was 5.3 and 7.4 times higher in 1994, respectively (Fig. 3). The increase of *M. balthica* and *M. affinis* seemed in general to be particularly intense in the middle and southern Airisto (Fig. 3). The highest increase of *M. balthica* occurred in sub-area X (1956: mean = 28.9 ind. m<sup>-2</sup>, 1994: mean = 1200.3 ind. m<sup>-2</sup>, Mann-Whitney U = 0.0, p = 0.050) and of *M. affinis* in sub-area V (1956: mean = 6.5 ind. m<sup>-2</sup>, 1994: mean = 4284.3 ind. m<sup>-2</sup>, Mann-Whitney U = 0.0, p =0.018).

The differences in total abundance within periods were significant in both study years (Table 2; 1956:  $\chi^2 = 25.26$ , df = 9, p = 0.003; 1994:  $\chi^2 = 21.65$ , df = 9, p = 0.010). However, there was only a marginally significant difference among sub-areas in total biomass in 1994 ( $\chi^2 = 16.79$ , df = 9, p = 0.052), indicating that the increase in abundance comprises mainly smaller species.

The results of the analysis of *M. balthica* and *M. affinis* proportions in 1956 and 1994 are shown in Tables 3–5. In 1956, there were significant differences in *M. balthica* and *M. affinis* proportions among sub-areas (Table 3; DDF = 11.60, F = 3.72, p = 0.020; DDF = 10.70, F = 8.00, p = 0.002, respectively). Moreover, it was obvious that *M. balthica* proportions were inversely influenced by depth (Table 4; DF = 10.7, t = -2.47, p = 0.032) and the total abundance of the community (DF = 9.5, t = -3.52, p = 0.006), i.e. when depth or community size increased, *M. balthica* proportions

tion decreased. In 1956, the highest parameter estimates, i.e. the highest significant proportions (calculated against sub-area X, which therefore obtain the value of *Intercept* estimate) for *M. balthica* were found in sub-areas V and II, and the lowest in VIII (Table 4). For *M. affinis*, the highest proportions were found in sub-areas IX and IV, and the lowest in VIII (Table 5).

When the 1994 situation was compared with the baseline level in 1956 (baseline-sub-area interaction) it became evident that no changes in M. balthica or M. affinis proportions had occurred between the periods (Table 3). Similarly, the baseline proportions had no effect on M. balthica or M. affinis proportions in 1994 (Tables 4 and 5). As before, significant differences in M. balthica and M. affinis proportions were found between sub-areas (Table 3; DDF = 14.10, F = 5.01, p = 0.004; DDF = 13.00, F =4.80, p = 0.007, respectively). Total community abundance had an inverse influence on M. balthica but a direct influence on M. affinis proportions (Tables 4 and 5). The highest M. balthica proportions in 1994 were found in sub-areas II and X, and the lowest in IV and VIII (Table 4.) M. affinis proportions were lowest in sub-area III (Table 5).

**Table 3**. Ill type *F*-tests for *Macoma balthica* and *Monoporeia affinis* proportions of the total benthic community in 1956 and comparison with 1994 situation when 1956 (Baseline'56) is used as a covariate. Other covariates are depth and total abundance of the community. Note that covariance estimates (variance components) are expressed as logit-scale.

Parameter		M. balthica			M. affinis	
	DDF	F	p	DDF	F	p
Sub-area 1956:	11.60	3.72	0.020	10.70	8.00	0.002
1994:	14.10	5.01	0.004	13.00	4.80	0.007
Depth (cov.) 1956:	10.80	6.08	0.032	11.80	1.20	0.295
. í 1994:	15.90	0.96	0.343	13.00	0.25	0.625
Total (cov.) 1956:	9.51	12.39	0.006	2.23	3.11	0.207
<b>1994</b> :	15.20	12.43	0.003	13.00	10.92	0.006
Baseline'56 (cov.)	15.70	0.64	0.436	13.00	0.20	0.661
Baseline'56 $\times$ Sub-area'94	4.10	0.28	0.955	6.00	0.21	0.978
Random effect			Covariance	e estimate		
St. (sub-area)'56		0.294			0.139	
St. (sub-area)'94		0.329			1.051	

Table 4. Parameteestimate is calculaESTIMATE statem	er estimate ted, empty ents (only :	s for the log space (cov: significant or	listic regr ariate) is r margine	ession n a comm ally signif	nodels for , ion effect fu ïcant resul	<i>Macoma L</i> or the peri ts are shov	<i>valthica</i> . The Sub-a ods. Dashed line s wn).	rea column d eparates the	escribes th parameter	e sub-are estimate:	ea for wl s calcula	nich the pa Ited differer	rameter ntly with
Parameter	Sub-area	Estimate	SE	Ъ	t	d	Parameter	Sub-area	Estimate	SE	Ч	t	d
M. balthica 1956							M. balthica 1994	-					
Intercep <i>t</i>		0.958	0.756	17.5	1.27	0.222	Intercept		1.810	0.902	15.4	2.01	0.062
Sub-area	_	-0.676	0.804	17.9	-0.84	0.411	Sub-area	_	-2.023	0.711	16.4	-2.84	0.012
Sub-area	=	1.231	0.773	17.7	1.59	0.129	Sub-area	=	0.521	0.745	17.0	0.70	0.494
Sub-area	≡	-0.149	0.682	16.3	-0.22	0.830	Sub-area	=	-1.244	0.668	16.3	-1.86	0.081
Sub-area	≥	-0.704	0.813	13.9	-0.87	0.401	Sub-area	≥	-2.451	0.685	16.0	-3.58	0.003
Sub-area	>	1.628	0.705	11.6	2.31	0.040	Sub-area	>	-1.119	0.602	10.3	-1.86	0.092
Sub-area	N	-0.076	0.671	17.8	-0.11	0.911	Sub-area	>	-1.354	0.533	10.9	-2.54	0.028
Sub-area	١١٨	1.202	0.831	10.4	1.45	0.178	Sub-area	١١٨	-1.732	0.640	12.8	-2.71	0.018
Sub-area	NIII	-1.805	0.915	13.2	-1.97	0.070	Sub-area	IIIV	-2.474	0.709	16.4	-3.49	0.003
Sub-area	×	-0.579	0.848	17.8	-0.68	0.504	Sub-area	×	-0.424	0.636	9.8	-0.67	0.521
Sub-area	×	0.000	I	I	I	I	Sub-area	×	0.000	I	I	I	I
Depth		-0.046	0.019	10.8	-2.47	0.032	Depth		-0.020	0.020	15.9	-0.98	0.342
Total		-0.010	0.003	9.5	-3.52	0.006	Total		-0.003	0.001	15.2	-3.53	0.003
							Baseline'56		-0.722	0.903	15.7	-0.80	0.436
	     		 5.043		 3.21	— — — — 0.007		     		5.715	- — — – 15.6	3.06	0.008
II vs. others		12.240	4.841	17.2	2.53	0.022	X vs. others		12.299	3.807	11.8	3.23	0.007
VII vs. others		11.946	6.680	12.1	1.79	0.099	IV vs. others		-12.207	5.026	16.9	-2.43	0.027
VIII vs. others		-18.123	5.851	10.1	-3.10	0.011	VIII vs. others		-12.436	5.452	16.9	-2.28	0.036

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estimate is calcul: effect for the peric results are shown	ated (note th ods. Dashed ).	at I and II a line separa	are not ind tes the p	cluded in arameter	the model estimates	s because calculatec	of no observations I differently with ES	s in either stu TIMATE state	dy year). Er ements (onl	mpty spac y significa	ce (cova ant or ma	riate) is a c arginally siç	ommon gnificant
Parameter	Sub-area	Estimate	SE	DF	t	٩	Parameter	Sub-area	Estimate	SE	Ч	t	d
M. affinis 1956		-					M. affinis 1994	-	-				
Intercept		-0.395	1.068	12.0	-0.37	0.718	Intercept		-2.006	1.356	13.0	-1.48	0.163
Sub-area	≡	-0.518	0.918	10.7	-0.56	0.585	Sub-area	≡	-1.690	1.016	13.0	-1.66	0.120
Sub-area	2	1.083	1.025	7.1	1.06	0.325	Sub-area	≥	2.102	0.998	13.0	2.11	0.055
Sub-area	>	-2.747	1.338	5.7	-2.05	0.088	Sub-area	>	1.546	1.203	13.0	1.29	0.221
Sub-area	N	-2.398	1.033	13.8	-2.32	0.036	Sub-area	N	0.938	1.260	13.0	0.74	0.470
Sub-area	١١٨	-2.112	1.442	6.4	-1.46	0.190	Sub-area	١١٨	2.567	1.257	13.0	2.04	0.062
Sub-area	NIII	-4.713	1.317	11.9	-3.58	0.004	Sub-area	IIIV	2.186	1.496	13.0	1.49	0.161
Sub-area	×	1.228	1.107	9.4	1.11	0.295	Sub-area	×	-0.058	1.099	13.0	-0.05	0.959
Sub-area	×	0.000	I	I	I	I	Sub-area	×	0.000	I	I	I	I
Depth		0.028	0.026	11.8	1.10	0.295	Depth		-0.009	0.019	13.0	-0.50	0.625
Total		0.006	0.003	2.2	1.76	0.207	Total		0.004	0.001	13.0	3.31	0.006
							Baseline'56		0.733	1.632	13.0	0.45	0.661
	     			5.1	3.70	— — — — 0.014		     	-21.112	5.607	13.0		0.002
IV vs. others		18.842	4.323	2.9	4.36	0.025							
VIII vs. others		-27.528	7.336	12.7	-3.75	0.003							

Table 5. Parameter estimates for the logistic regression models for Monoporeia attinis. The Sub-area column describes the sub-area for which the parameter

# Discussion

Our results provided evidence of a general increase of benthic macrofauna in the Airisto Inlet between 1956 and 1994. The increase was apparent in both parameters measured, i.e. in the species abundance and in the numbers of species. Although the abundance increase was general in nature, it seemed mainly to involve the middle and southern parts of the study area, and showed, together with biomass, an increasing trend towards the open sea. The number of the species did not present a corresponding pattern, but instead a rather constant level shift upwards, where the largest increase seemed to occur in the innermost sub-areas. When compared with the species list of Tulkki (1960), it can be seen that the most obvious input of indigenous soft bottom species occurred with oligochaetes. Altogether six new species appeared in 1994, and this accounts for the conspicuous increase of species number in the innermost stations near the polluted harbour. However, the increase is partly explained by the inability to identify all these species accurately in the 1950s, and probably this was also the reason for the appearance of the amphipode Pontoporeia femorata in the 1990s (Paavo Tulkki, personal communication). Thus, the polychaete Marenzelleria viridis was ecologically the only indisputable newcomer as this species was found for the first time in the Baltic in 1985 (Bick and Burckhardt 1989). Although M. balthica and M. affinis proportions of the total community remained unchanged when compared to the 1956 situation, some evidence of change in terms of higher proportions in the southern sub-areas was discerned, especially for M. balthica. M. affinis proportions seemed to become more even in the middle and south sub-areas, while a distinct decrease coincidentally occurred in the north.

The present results match other findings from the Åland Islands and the Archipelago Sea areas (e.g. Bonsdorff *et al.* 1991, Mattila 1994, Bonsdorff *et al.* 1997a, 1997b), all illustrating both structural and functional changes in the benthic system. Changes are mainly attributed to general eutrophication of the Baltic Sea as increased pelagic and benthic production and subsequent input of organic matter have been the basic reason for the alteration (Pearson and Rosenberg 1978). Although similar changes have also been recorded in the Baltic basin (Cederwall and Elmgren 1980), the effects of eutrophication are generally more pronounced in coastal areas (HELCOM 1991). The only exception in our results is the general increase in species numbers, which have shown an unchanged or decreasing pattern in the other studies. Although this increase was more or less evident in all groups, it covered notably oligochaetes and polychaetes, i.e., the 1st and 2nd order regressive benthic species according to classification of Leppäkoski (1975). For M. balthica, and also for M. affinis, the increase was more like extension to larger areas; former categorised as progressive and latter as regressive species by Leppäkoski (1975). We believe this discrepancy between earlier studies can be explained by a longer time gap between the years compared in our study. In the present study, the 'reference' is located in the mid-1950s when benthic communities evidently were closer to their natural state. The other studies have been done practically in the years when the eutrophication has already influenced the benthic assemblages, and therefore the numbers already reflect a higher starting level. The difference may also partly be explained by methodological differences (e.g. oligochaetes are better known nowadays).

In a similar study, Bonsdorff et al. (1991) made comparisons of hydrography and zoobenthos between 1973 and 1989 in the archipelago of the Åland Islands, and reported on altered number of species and increased abundance and biomass of benthos. They found that the spatial distribution of biomass showed an increasing trend towards the open sea and considered this to be caused by the general eutrophication process in the Baltic Sea imported to the local ecosystem. Moreover, a shift in the relative importance of species was found with the dominance of stress-tolerant species (oligochaetes and chironomid larvae) in the polluted innermost areas while, coincidentally, the dominating species in the system (Macoma balthica and Monoporeia affinis) were pushed towards the open sea. Our results are in general agreement with these conclusions, especially with the idea that, depending on the location, the changes simultaneously reflect both local disturbances and general eutrophication. In the Airisto Inlet, the innermost sub-areas cover wastewater-polluted harbour areas, which already in the 1960s were settled mainly by stress-tolerant oligochaetes (Tulkki 1960, 1964, Leppäkoski 1975). Moreover, the northern parts of the Airisto Inlet have since the early 80s been under intensive dumping of the dredged masses. This material, dredged mainly from Turku harbour and the Aurajoki river, was dumped during 1989-1995, on average at the rate of more than 130 000 m<sup>3</sup> per year (Kauppila and Wright 1997), and since then dumping has increased further because of the development and reconstruction of the harbour and shipyard. The dumping has contributed to the shift from suspension feeders (decline of crustaceans) to deposit feeders (increase of poly- and oligochaetes) of the benthic community. The reason for the changes is alterations in the seabed composition (increased sedimentation and siltation) and also accelerated local eutrophication. The middle and southern parts of the Airisto Inlet were influenced by the general eutrophication in the Archipelago Sea, and there the changes in benthos have followed the general pattern observed in the sea (see earlier refs.).

The main sources of excess nutrients in the Archipelago Sea have been industrial and municipal wastewater, forestry, agriculture (all chiefly via river runoff) and, during the last 20 years, fish farming (e.g. Bonsdorff et al. 1997a, 1997b). Only recently have the share of airborne nutrients and nutrients imported by currents from other parts of the Baltic Sea (Gulf of Finland, Gulf of Bothnia, Baltic Proper) been estimated (Jumppanen and Mattila 1994, Helminen et al. 1998, Kirkkala et al. 1998). Hänninen et al. (1999) made a comprehensive study of the eutrophication process in different zones of the Archipelago Sea. They found that nutrients coming with river runoffs mainly remain within inshore waters (involve the innermost and northern areas of the present study) and, therefore, the impact of eutrophication has been the most severe there. In the middle archipelago (the middle and southern parts of this study), the effect of fish farming could also be seen in water quality. After the early 1990s, the general rise in nutrient concentrations culminated, and subsequently levelled off or even fell because of general economic decline and decreased production in fish farm markets and coincident improvements in aquaculture techniques. In the outer zone, the influence of the background loading from the Baltic Sea mainly influences water quality, but this seems to depend on the location and is visible only occasionally, particularly in winter.

Along with human impacts, the relative importance of natural factors, probably causing similar changes in the benthic communities, should also be considered. Laine et al. (1997) made an analysis with long-term data (1965-1994) on macrozoobenthos in the Gotland Basin and the Gulf of Finland in relation to the hydrographical regime. They found that the fluctuation in salinity affected the community structure and distribution of zoobenthos. The result was very similar to that of zooplankton studies, i.e. more marine species were favoured by increased salinity, while less marine groups showed the opposite (e.g. Segerstråle 1969, Vuorinen and Ranta 1987, Viitasalo et al. 1994, Vuorinen et al. 1998). However, no direct effects on the quantitative amount of zoobenthos were detected in the study and, therefore, more discussion related to effects in the Airisto Inlet would be rather speculative.

One fundamental question in a work of this kind is how appropriate the method used is to record the changes in benthic community. This is partly a question of spatial scale and partly a question of a method's accuracy to observe the change. According to Bonsdorff et al. (1991), the local sources play an important role in the structuring of the aquatic ecosystem. Changes over time at one site, although significant, may reflect merely very local progress. Therefore, to get to grips with larger scale processes, the strategy should be to sample many stations over a large area with long time-intervals between sampling. Moreover, the sampling regime should involve a sufficient number of samples (Elliott 1993). Any overall changes will then reflect processes on a much larger scale (Cederwall and Elmgren 1980, Pearson et al. 1985, Heip 1995), although the detailed processes may remain partially unknown. Regardless of the large size and non-homogeneity of the present study area, we

believe that the experimental design used (30 similarly resampled stations and the entire area divided into more homogeneous sub-areas) allows an interpretation of our results at both local and regional levels. Thus, the changes recorded reflect not only an alteration among studied sub-areas but also between the whole Airisto Inlet and the entire coastal ecosystem of the Archipelago Sea.

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