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Zoobenthos and problems with monitoring; an example from the Åland Area

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Abstract

In 1986 and 1987 a transect off SW Åland, N Baltic Sea was sampled for macrozoobenthos at 14 stations ranging from 1 to 226 m depth. In connection to this survey comparative samples were taken at a "standard depth" of 19–20 m at 2 additional stations to estimate variability in space and time. The data were analyzed for primary community parameters (i.e. no. of species, abundance and biomass) including spatial and temporal differences, and the information is evaluated in relation to topography and hydrography. A more detailed analysis using the dominant components of the zoobenthic communities illustrated some of the difficulties in monitoring of the marine zoobenthos. There were significant differences in abundances and biomasses even over short distances. The differences get even more pronounced when trying to relocate an exact station with another boat and using different grab types on the same station. This is exemplified by samples from a 19-meter station using a modified Olausen box corer and an Ekman-Birge type grab sampling on different days from different vessels; the primary community data obtained with both grabs was 10 *vs.* 10 species (with *Pontoporeia affinis vs. Macoma balthica* dominating), 1945 $\pm$ 529 *vs.* 2168 $\pm$ 385 ind/m^2, and 21.6 $\pm$ 9.5 *vs.* 209.0 $\pm$ 45.3 g/m^2, respectively. Similarily, samplings at stations less than 1 n. mile apart at 20 m depth using the Ekman-Birge grab yielded 2168 $\pm$ 385 and 7000 $\pm$ 607 ind/m^2; a highly significant difference. The no. of species (10/13) and biomasses (209.0 $\pm$ 45.3/184 $\pm$ 89.0 g/m^2) showed no significant differences. The data also showed some depth dependency, but there was no correlation (r=0.22) between abundance and biomass for the entire material pooled, illustrating the importance of always measuring both. When using key-species for monitoring purposes, their natural distribution should also be known, as the dominance-patterns shows clear gradients with depth and sediment type.

Introduction

Monitoring the state of the Baltic Sea is attracting ever more attention at national and international levels. Primarily these monitoring programmes are concentrated to the open sea, and when located near shore, are often directly connected with specific sources of pollution or other forms of impact on the marine environment. Much of the information obtained is presented as status reports and/or summarized by e.g. HELCOM (see e.g. ROSENBERG et al. 1984, and the "Baltic Sea Environment Proceedings No. 19" 1986 for current references).

For the Swedish coastal and near shore waters an extensive survey was conducted in the early 1980's (ROSENBERG 1984), but no such analysis exists from the vast archipelago areas of the Finnish coastal waters. During the 1980's repeated blooms of potentially toxic cyanobacteria (PERSSON et al. 1984, RINNE et al. 1986) not only in the open sea, but also the coastline have actualized the question of eutrophication in this area, however. Eutrophication of large sea areas is in many ways a very delicate problem (LARSSON et al. 1985, PEARSON et al. 1985, SCHULZ and KAISER 1986), as no single point source can be found – in fact much of the nutrient load to the Baltic is airborne (NEHRING and WILDE 1982). This makes it very difficult to measure eutrophication in biological terms (chemically the most common measure is the nutrient level of the water or sediment: FONSELIUS 1978, NEHRING et al. 1984, LINDAHL and WALLSTRÖM 1985, NIEMISTÖ 1986, PITKÄNEN and KANGAS 1986, YURKOVSKIS 1986), but measurements of pelagic primary production (HÄLLFORS and NIEMI 1986, SCHULZ 1986, WULFF et al. 1986, MELVASALO and VILJAMAA 1987), phytobenthos (PLINSKI and FLORCZYK 1984, KAUTSKY et al. 1986, v. WACHENFELDT et al. 1986) and zoobenthos (CEDERWALL and ELMGREN 1980, JÄRVEKÜLG and SEIRE 1985, PEARSON et al. 1985, ANDERSIN 1986) have also been used for this purpose.

Some problems connected with monitoring zoobenthos in coastal waters are based on the lack of quantitative historical data from non-affected areas in Finland. It has proven hard to convince the authorities to finance monitoring programmes designed simply to study the long-term changes of the environment based on arguments like an "overall change", as it has seemingly not been possible to point at the effects and/or the sources. Some of the problems with studying e.g. eutrophication are connected with the wide amplitude of the measurements of nutrient levels and primary production. To overcome that, zoobenthos is utilized partly because of the relative longevity of the animals, and partly because of the relative ease of obtaining a satisfactory number of replicate samples at any chosen site (WESTERBERG 1978).

The aims of this paper are to illustrate some of the difficulties in biological monitoring using macrozoobenthos along a transect off southwestern Åland into the deep of the Åland Sea (Fig. 1). The transect covers a depth range from 1 to 226 m, and a variety of sediment types (Table 1). Furthermore a horizontal (temporal and spatial) transect was included to illustrate the variability of the natural ecosystem. The comparison also includes two different methods of sampling. All data presented were collected in 1985–87, and the area is free from any major source of pollution or municipal effluent, and can be classified as "clean" in an overall Baltic context. Biological monitoring of the coastal waters around Åland has previously been summarized by LEPPÄKOSKI et al. (1986).

Materials and methods

The outer part (stations S-1 to S-9; 30–226 m) of the transect described (Fig. 1, Table 1) was sampled in June 1986 and June 1987 using a modified Olausen box-corer (20 x 20 cm surface area; about 30 kg, ANDERSIN and SANDLER 1986) for the benthic samples and for describing the sediment properties. All these samples were washed on a set of double sieves, with 1.0 x 1.0 mm and 0.5 x 0.5 mm mesh sizes. The animals were picked from the 1.0 mm mesh and stored in 2.0 % buffered formaldehyde solution. For the 0.5 mm sieve the entire residuals were preserved. The basic hydrographical parameters (temperature and salinity) were measured using a submersible probe with electrodes connected to a computer aboard.

The inner, more shallow stations (S-9a to S-14, SU, and ÅL-E; 1–20 m) were sampled by hand using an Ekman-Birge box grab (17 x 17 cm; about 5.5 kg) for sediment description and zoobenthos. These samples were washed on a 0.5 x 0.5 mm screen, and the entire residue was preserved in 4 % buffered formaldehyde. Samples for the hydrographical analysis were taken using a hand-operated Ruttner-sampler. Temperature was read immediately, and salinity was measured with a conductometer in the laboratory In all cases samples for oxygen content were taken from the bottom near water, and analyzed by the Winkler-method. As the oxygen values obtained were never limiting for the fauna ($>$ 7 mg O_2/l even at 226 m depth), these results will not be reported here.

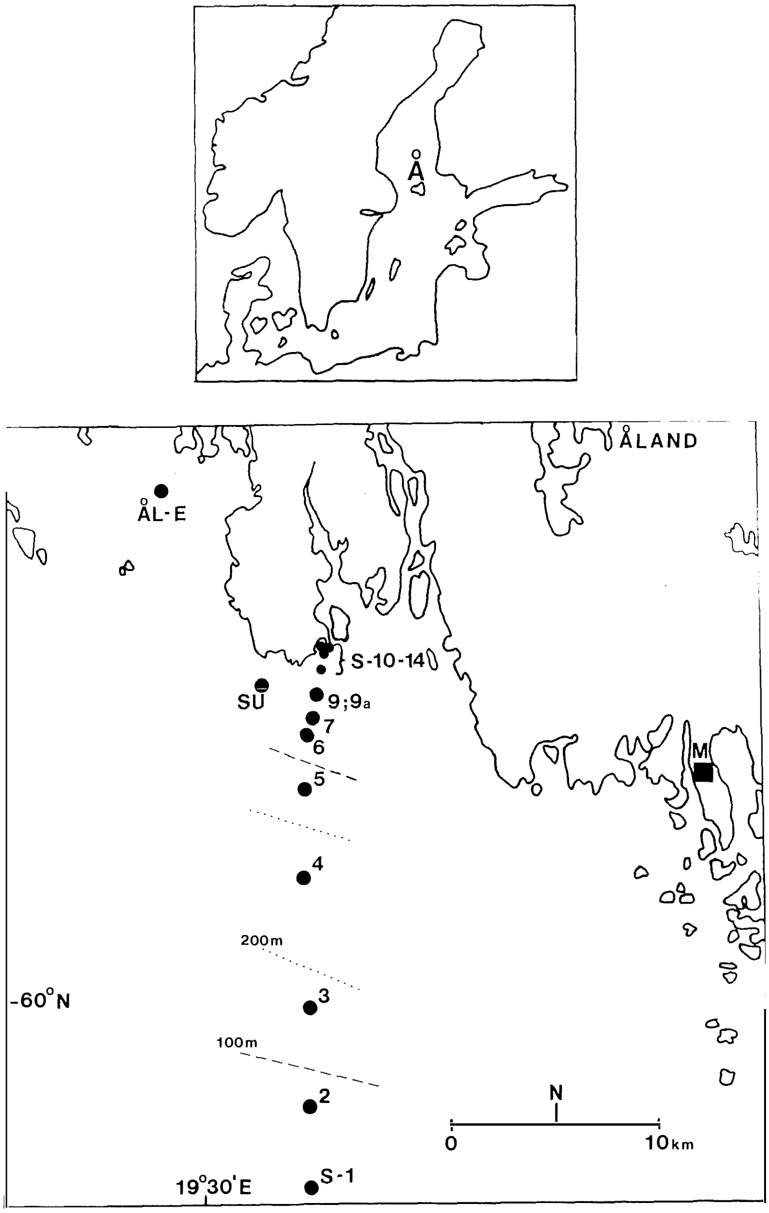

Figure 1

The investigated area (Å = the Åland Islands; M = Mariehamn) in the central, northern Baltic Sea. The major depth regions (100 and 200 m) are shown within the vertical transect from south (open sea) to north (stations S-1 to S-14). Stations ÅL-E, SU and S-9, S-9a constitute a horizontal transect at 19-20 m depth.

The basic hydrographical conditions (T°C, $S^0/_{00}$) in the bottom near water along the transect ranging from 2.5 to 226 m depth (June 1986) are shown in Fig. 2. The salinity is fairly stable throughout the transect, but the temperature naturally increases with

Table 1

Some basic station data for the localities sampled in 1986/87. The sediment quality is described based on the grab samples

Station	Depth (m)	Sediment type	Grab used
S-1	62	sandy mud, concretions	Olausen box corer ($0.04\ m^2$)
S-2	77	sandy mud, concretions	– „ –
S-3	151	mud, clay, concretions	– „ –
S-4	226	soft mud, clay, concretion nuggets	– „ –
S-5	99	mud, clay	– „ –
S-6	54	clay, coarse sand	– „ –
S-7	30	coarse sand, gravel, mud	– „ –
S-9	19	coarse sand, mud, clay	– „ –
S-9a	19	coarse sand, mud, clay	Ekman-Birge grab ($0.0289\ m^2$)
S-10	12	sand, mud	– „ –
S-11	9.5	fine sand	– „ –
S-12	5	fine sand	– „ –
S-13	2.5	fine sand	– „ –
S-14	1	very fine sand	– „ –
SU	20	coarse sand, mud, clay	– „ –
ÅL-E	20	sand, some gravel, mud, clay	– „ –

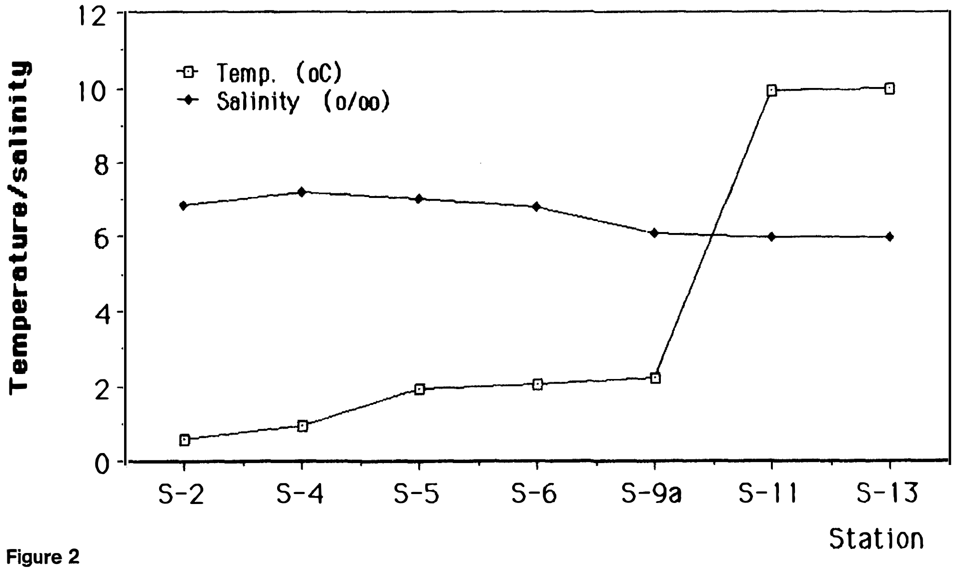

Figure 2

Hydrographical conditions (temperature: T°C and salinity: $S^0/_{00}$) in June 1986 along the transect from the open sea towards the coast.

decreasing depth. For the innermost part of the transect a thorough analysis of hydrography, sediment properties, fauna and flora was presented by BLOMQVIST and BONSDORFF (1986).

The zoobenthic samples were all sorted under a stereomicroscope, and all individuals (with the exception of the oligochaetes) were determined to species. Biomass (formaline wet weight) was determined on 0.1 mg accuracy. All data are presented on a per square metre basis.

The data is mainly presented as primary community data, i.e. number of species, abundance and biomass. The dominance (% of abundance) of key-species is also illustrated along the transect. The methods largely follow those recommended for the Baltic Sea by DYBERN et al. (1976).

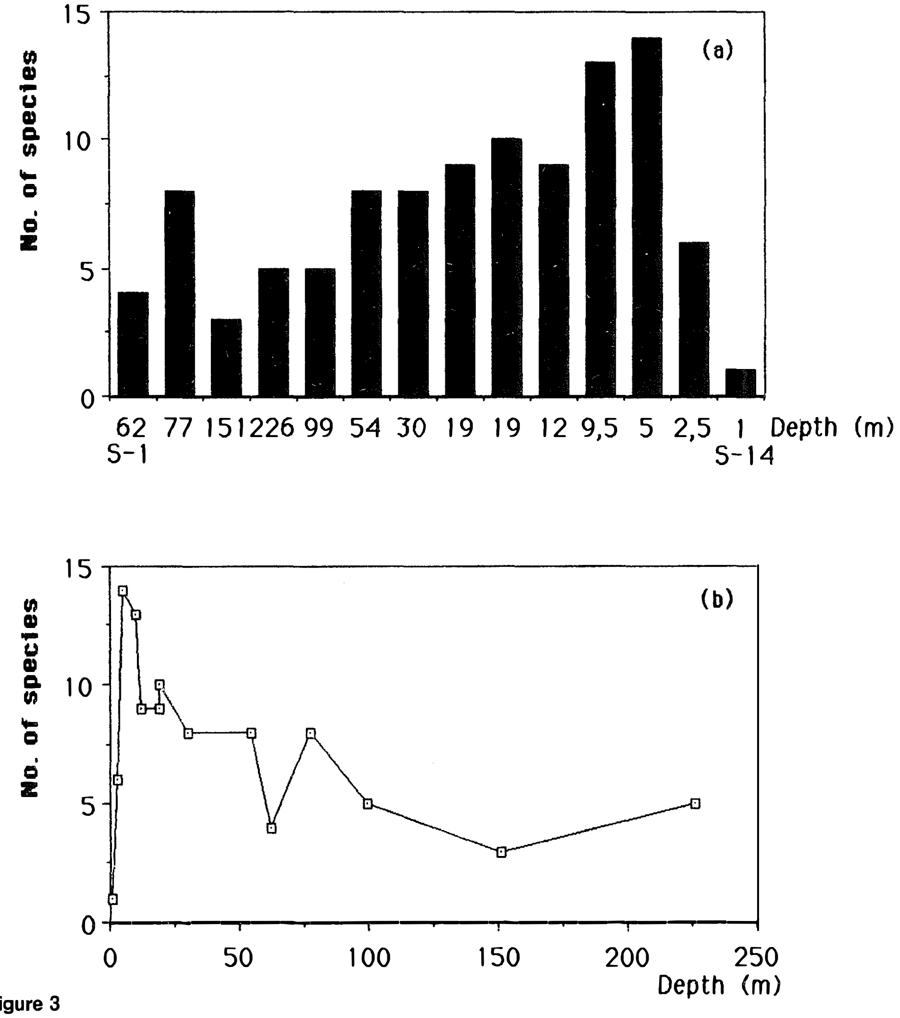

Figure 3

The zoobenthic community diversity expressed as number species (a) along the vertical transect studied (stations S-1 to S-14), and (b) in relation to depth in June 1986.

Results

The numer of species, total abundance and biomass-values along the transect (June 1986; no significant differences 1986/1987) from the southern end (station S-1; 62 m) towards land (station S-14; 1 m) are presented in Figs. 3–5. The number of species at the shallow water station between 5 and 19 m depth is comparably high mainly due to molluscs (*Mytilus edulis, Cardium glaucum, Hydrobia* spp., *Potamopyrgus jenkinsi* etc.), crustaceans (*Bathyporeia pilosa, Gammarus* spp., *Corophium volutator, Idotea* spp., *laera albifrons* coll.) and polychaetes (*Nereis diversicolor, Pygospio elegans, Manayunkia aestuarina*) only found in this region. The other common species within the above groups (such as *Macoma balthica, Pontoporeia affinis, P. femorata, Mesidotea entomon* and *Harmothoe sarsi*) occur more or less along the entire transect. A complete list of species found in the area is presented in BLOMQVIST and BONSDORFF (1986), but it is worth

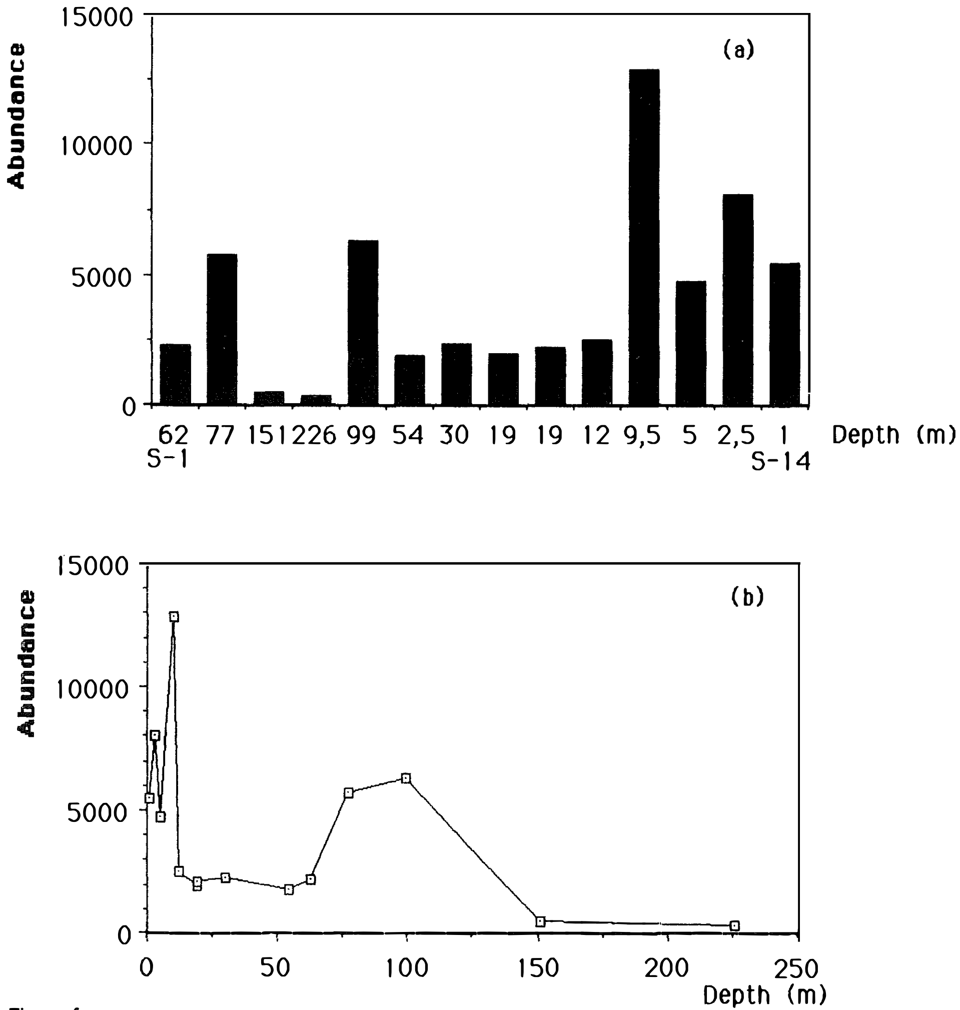

Figure 4

The total community abundance (ind/m^2) at (a) stations S-1 to S-14, and (b) in relation to depth in June 1986.

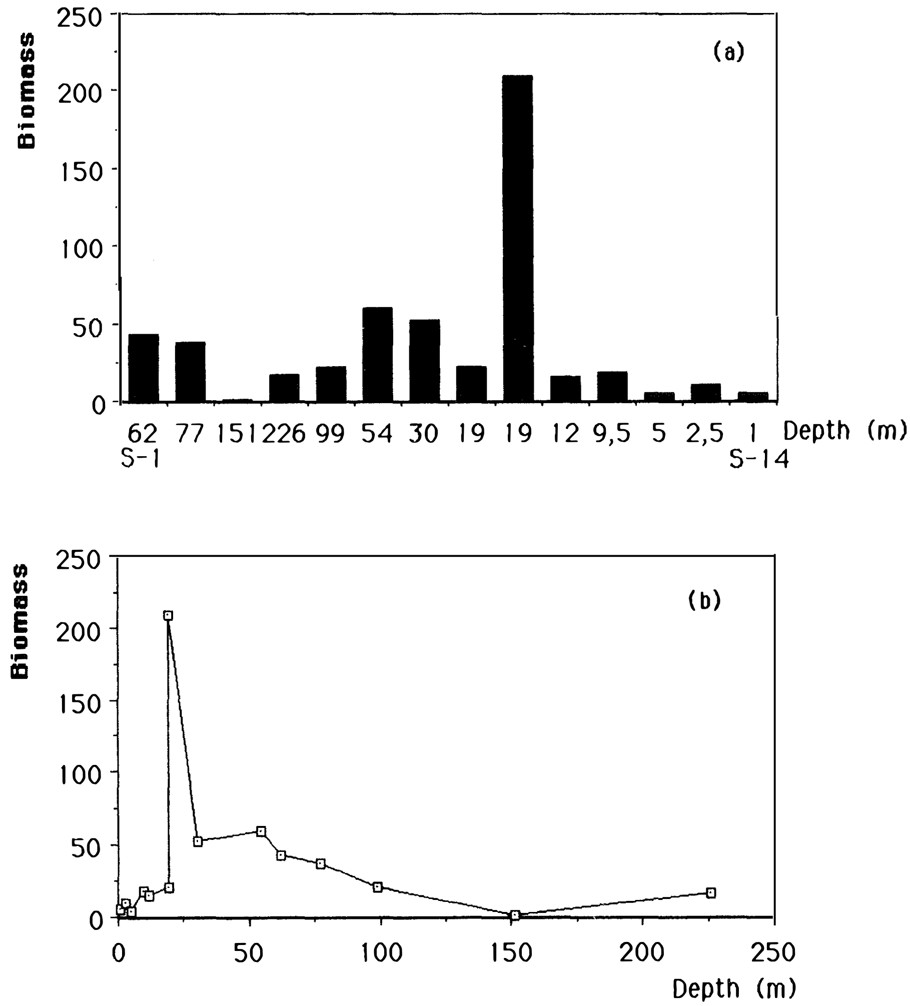

Figure 5

The total community biomass (g wwt/m^2) at (a) stations S–1 to S–14, and (b) in relation to depth in June 1986.

noting that virtually all taxa are of marine or brackish origin, with no peaks in occurrence of species classified as pollution-tolerant (according to the classification for the Baltic Sea presented by LEPPÄKOSKI 1975).

Generally the deepest (151 and 226 m) and the most shallow (1 and 2.5 m) stations have low species numbers, although they have no species in common. The peaks in abundance at 2.5–9.5 m and again at 77 and 99 m are due to amphipods (*B. pilosa* and *P. affinis / P. femorata* respectively), whereas the elevated biomasses closer to the coast line are due to the bivalves *M. balthica* and *M. edulis* (Figs. 3–5). Although there is a clear negative trend with increasing depth from the sublittoral stations downwards for all the parameters described, there is no significant correlation (r=0.22; n.s.) between abundance and biomass (Fig. 6), since amphipods and polychaetes (mainly *P. elegans*) with

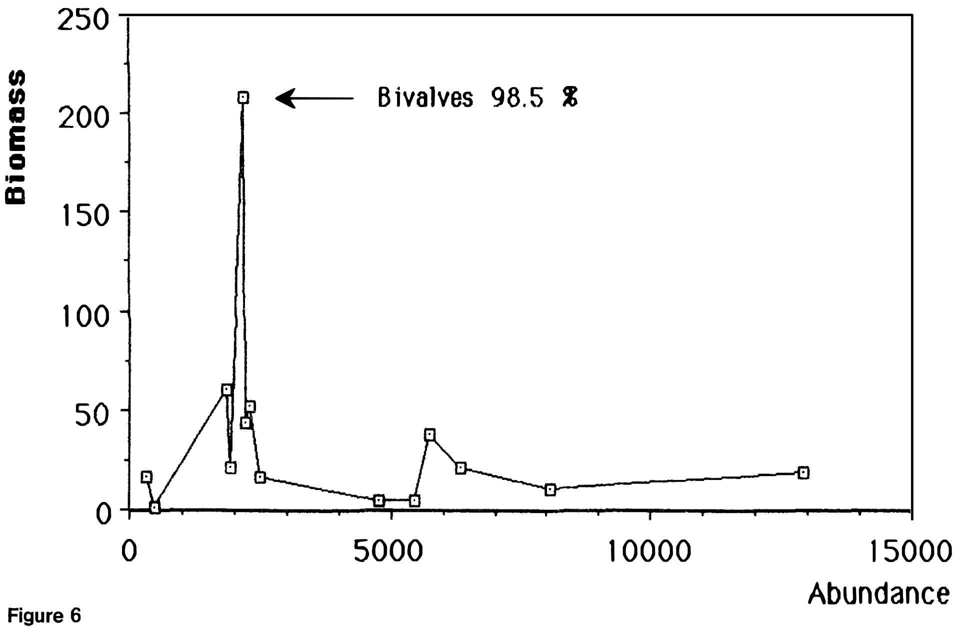

Figure 6

The relationship between total abundance (ind/m^2) and total biomass (g wwt/m^2) for stations S–1 to S–14 (r = 0.22; n.s.).

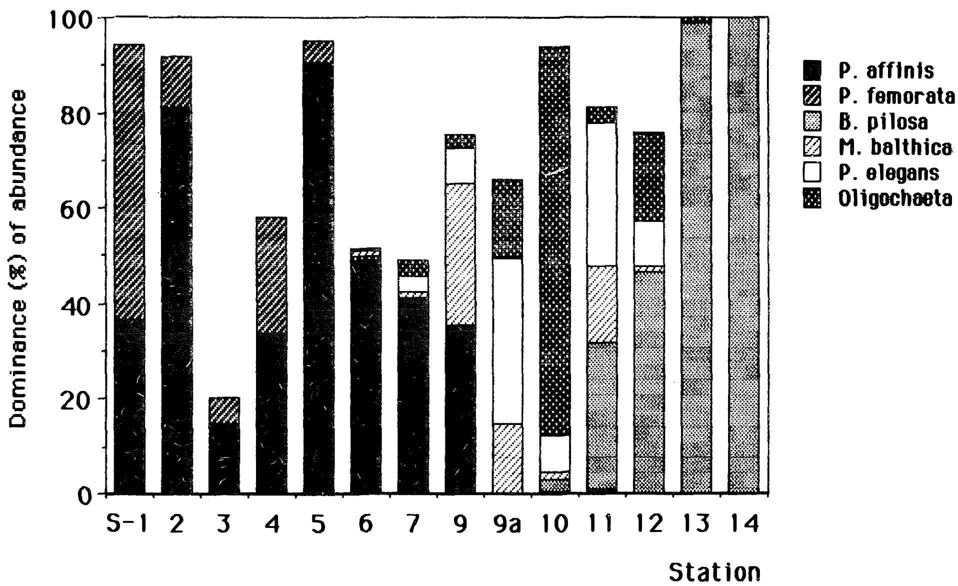

Figure 7

The shift in relative importance (% of total community abundance) of some dominant species along the transect S–1 to S–14, illustrating the spatial variability of the zoobenthic assemblage.

low individual weights constitute the bulk of the community (Fig. 7). The shift in species dominance along the transect shown in Fig. 7, illustrates the need for sampling many depth zones and substrates when monitoring the zoobenthos, as there may be a highly significant difference in community composition between stations, although the number of species is similar (cf. Figs. 8, 9).

The differences in sampling methods and/or problems with location of stations is exemplified by the fact that samples taken at the same station (S-9 and S-9a) only a few days apart (June 1986), but with the different grabs mentioned, yielded some major differences in the community data. The species number (10 *vs.* 10) and total abundance (1945 $\pm$ 529 ind/m^2 *vs.* 2168 $\pm$ 385 ind/m^2) show no differences, but the biomass values (21.6 $\pm$ 9.5 g wwt/m^2 *vs.* 209.0 $\pm$ 45.3 g wwt/m^2) differ significantly ($p < 0.001$; t-test). Not only the biomasses differ; the community structure was also different, with *P. affinis* and *M. balthica* being the most abundant species (64.8 % of the total abundance) at station S-9, and *M. edulis* and Oligochaeta (43.1 %) at station S-9a.

Thus on the same station (located by landmarks) less than one week apart the two methods gave significantly different results. Comparing station S-9a June 1986 with S-9a June 1987 (using the Ekman-Birge grab) gave no significant differences, however, although the biomass in 1987 was higher than in 1986 (333.4 $\pm$ 83.7 g wwt/m^2; Fig. 8). Comparing two stations (S-9a and SU; Fig. 1) of similar depth (19–20 m) and sediment type (basically coarse sand/mund and clay; see Table 1) sampled during June-July 1986 no significant differences in species number (10 *vs.* 13) or biomasses (209.0 $\pm$ 45.3 and 184.0 $\pm$ 89,0 g wwt/m^2, respectively) were recorded. In total abundances (2168 $\pm$ 385 *vs.* 7000 $\pm$ 607), however, a highly significant difference ($p < 0.001$; t-test) was shown. The main reason for this difference is the dense population of *P. affinis* recorded at station SU (44.1 % of the entire community). In both cases the biomass values were completely

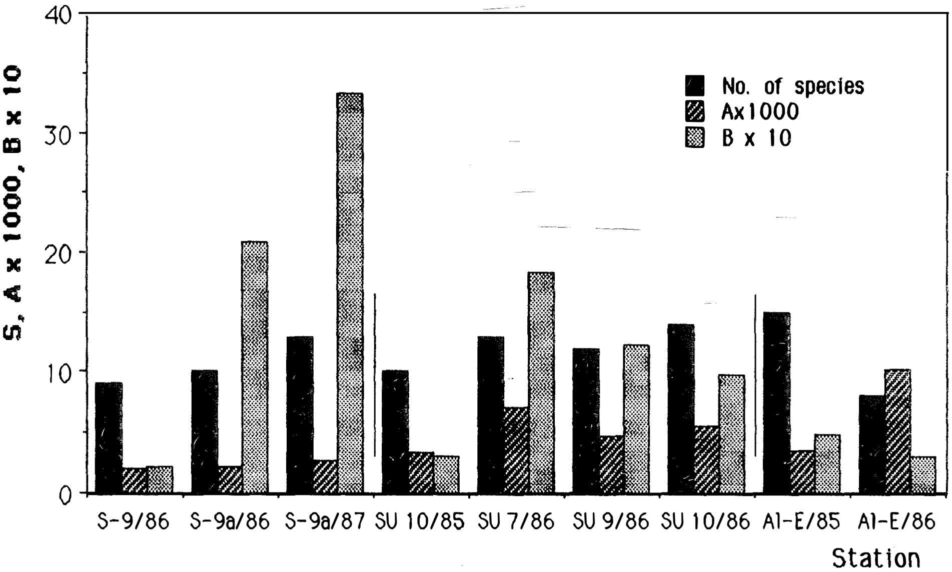

Figure 8

Temporal and spatial variability of primary community parameters (number of species; S: total abundance (ind/m^2); A: and total biomass (g wwt/m^2); B: at stations of similar depth (19–20 m) and sediment type.

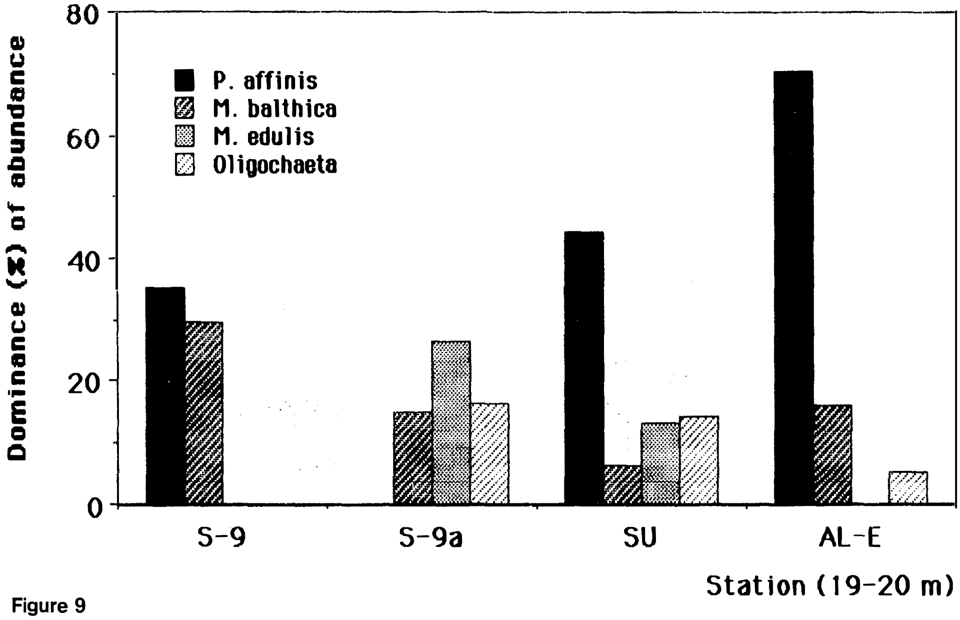

Figure 9

The relative importance (% of total community abundance) of some dominant taxa at stations of similar depth (19–20) and sediment type.

dominated by the bivalves *M. balthica* and *M. edulis* (98.5 % at station S-9a, and 85.3 % at station SU), and the overall species compositions were quite similar.

If temporal changes and more stations (S-9, S-9a, SU, and ÅL-E about 10 km away; Fig. 1) are added to the picture, the analysis gets more complicated (Figs. 8, 9). Then the differences can be summarized as "within stations" and "between stations" on a seasonal and interannual basis in a situation where all samples have been treated similarily. The "within station" temporal variability shows that abundance and biomass may vary significantly (Fig. 8; at station SU the community biomass varies significantly at $p < 0.05$ within one annual cycle, and at station ÅL-E community abundance during a year ranged from 3515 ± 281 to 10259 ± 489 ind/m^2; $p < 0.001$). The "between stations"-variation indicates that any measurable community parameter may be significantly different (Fig. 9 illustrates this difference in the from of dominant species) in spite of short distances, similar types of sediment and similar hydrographical conditions. The results at community level (number of species, abundance, biomass) thus displays significant variability i& space and between seasons, but relative stability between years at single stations (Fig. 8).

Discussion

The choice of methods naturally influences the results obtained, and in this case the smaller, lighter grab, in combination with a possible error in relocating the exact sampling site a few days later, yielded a quite different picture of the community. These results differ from those of ANDERSIN and SANDLER (1986), who got similar results from deeper bottoms using both the Van Veen grab and the modified Olausen box corer. It is equally important to use standard sieves, as small differences in mesh sizes (e.g. 0.5/0.6 mm) can show highly significant differences (BONSDORFF, unpubl. data).

The temporal variability and the spatial heterogenity described seem to be common features on a wider scale (ANDERSIN et al. 1978, ELMGREN 1978, KARJALA and LASSIG 1985), and is in good accordance with e.g. PERSSON (1982), who described zoobenthos population – and community dynamics from different sediment types and depth zones in the southern Baltic Sea.

An aspect frequently studied when monitoring zoobenthos e.g. along pollution gradients is the population dynamics of dominant species, such as *M. balthica* (MÖLSÄ et al. 1986). This measure may also vary considerably in natural populations basically unaffected by pollution or eutrophication largely due to biotic interactions with other species (GLASSER 1979, COMMITO and AMBROSE 1985, BONSDORFF et al. 1986, REISE 1987) and to ranges of occurrence in relation to depth or sediment type (see Fig. 7). Along the transect described *M. balthica* occurs from station S-6 (54 m) inwards to station S-13 (2.5 m), with a clear peak at intermediate stations (S-9; 19 m to S-11; 9.5 m; maximum density 2083 $\pm$ 276 ind/m^2). Thus the analysis of the population structure will largely depend on wether the samples were taken within the optimal limits of occurrence or not. At the deeper stations all individuals of *M. balthica* found were adults ($\triangleright$ 11.5 mm), whereas the sizes recorded at the shallowest stations ranged from 0.5–5.0 mm. At the intermediate localities all size classes were recorded, although with a clear dominance of juveniles at 9.5 m and of adults at 19 m. This is illustrated by the mean individual weight, which at station S-9a was $>$ 300 mg wwt/ind (including the shell) and at station S-11 only about 3–5 mg/ind. Between the stations of similar (19–20 m) depth there is also a significant ($p < 0.05$) difference in the mean individual weights, with values ranging from below 10 to above 300 mg/ind, depending on the sediment type and density (OLAFSSON 1986). Thus basically any population- or community parameter chosen shows large variations within the investigated area.

The results presented in this paper, coupled to the data of WESTERBERG (1978), BLOMQVIST and BONSDORFF (1986) and LEPPÄKOSKI et al. (1986) from the coastal waters of Åland, demonstrate the need for caution when setting up monitoring programmes designed to trace slowly occuring changes, such as eutrophication in coastal waters. On the other hand, the usefulness of off-shore monitoring in the northern Baltic Sea has previously been demonstrated and discussed by e.g. ANDERSIN et al. (1978). ELMGREN (1978, 1984) and ANDERSIN (1986), and the couplings to eutrophication by CEDERWALL and ELMGREN (1980). For Finnish coastal waters only few data sets from non-polluted waters exist, that could be used as a baseline for monitoring the transition zone between the archipelago and the open sea (KARJALA and LASSIG 1985, SARVALA 1985 continuing the works in the same area by SEGERSTRÅLE 1933, 1960), thus further emphazising the need for a national network of coastal monitoring localities.

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