Ann. Zool. Fennici 25:277–292. 1988

Meiobenthos in relation to macrobenthic communities in a low saline, partly acidified estuary, Bothnian Bay, Finland

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Meriläinen J. J. 1988: Meiobenthos in relation to macrobenthic communities in a low saline, partly acidified estuary, Bothnian Bay, Finland. — Ann. Zool. Fennici 25:277–292.

The meiobenthos of the Kyrönjoki estuary (63°N, 22°E) in the northernmost basin of the Baltic Sea was studied in June and August at 10 sampling stations distributed over the inner, middle and outer estuary. Regional variation of the mean abundance of meiobenthos was very high, ranging from 0.04×10^6 ind./m^2 (16 mg OC/m^2) at the river mouth to 2.4×10^6 ind./m^2 (512 mg OC/m^2) in the shallow area of the outer estuary. The meiofauna sampling sites represented seven zoobenthic communities, which were discriminated by DCA-ordination of the previously published, more comprehensive data on macrobenthos. Estuarine communities were controlled by two major environmental factors, the river water and sediment quality. Due to decreasing salinity, the shallow water communities of the middle estuary were mainly composed of freshwater species. As indicated in previous studies, several benthic freshwater species found in the middle estuary were absent from the inner estuary. This paper discusses the possible reasons leading to this discontinuity in the distribution of freshwater species in the inner estuary. It was concluded that the periodical acidity of the river waters and interrelated factors, especially the high concentrations of labile aluminium (200–320 µg/l, in places even 2300 µg/l), were mainly responsible for the deteriorated fauna of the inner estuary. Species of the acid sensitive groups, such as Gastropoda, Lamellibranchiata, Hirudinea and Ephemeroptera were totally absent from the inner estuary. The absence of the meiobenthic Turbellaria and Ostracoda from this area may indicate that the freshwater species of both these groups are sensitive to acid, Al-contaminated waters.

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1. Introduction

The regional distribution of the meiobenthos was studied in an oligohaline estuary in the Bothnian Bay, the northernmost basin of the Baltic Sea. The animal material was sampled from the inner, middle and outer estuary. There have been few studies on the estuarine fauna in the Bothnian Bay, especially in relation to the meiobenthos. The fauna in the River Luleå archipelago in the northwestern part of the basin was examined by Kautsky et al. (1981). Valtonen et al. (1984) studied the deep bottom in the northeastern Bothnian Bay. Elmgren et al. (1984) have shown the main differences in the quantity and structure of the meiobenthos between the two large basins of the Gulf of Bothnia, the Bothnian Sea and the Bothnian Bay.

This study, together with the previous studies on the estuarine macrobenthos (Meriläinen 1984, Bagge & Meriläinen 1985) and data on the outflow and dilution of the acidic waters in the estuary (Meriläinen 1985), including recent observations on the concentrations of labile aluminium, also made it possible to examine the effects of the acid, aluminium contaminated river waters on the quantitative and qualitative distribution of the benthic invertebrates in the estuary. The periodical acidity is a striking occurrence in the nutrient rich waters of the River Kyrönjoki.

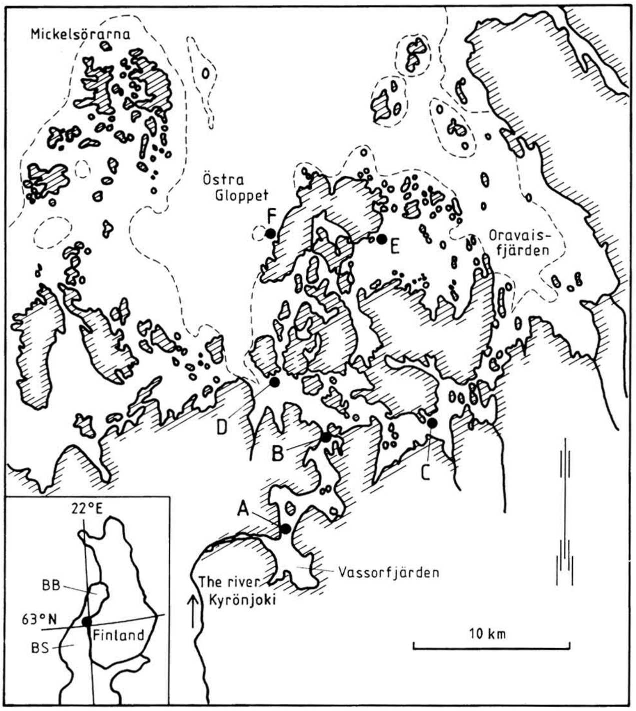

Fig. 1. The Kyrönjoki estuary and location of the study areas A–F. The dotted line shows the isobath of 10 m. BB = The Bothnian Bay, BS = Bothnian Sea.

The acidification of the waters in the lower reaches of the river originates from the very acid runoff from the sulphureous soils, the ancient seabed of the Littorina Sea. According to Alasaarela (1982), the acidification of the river waters is correlated with the sulphate concentration in the runoff waters entering the river from the Littorina basin and with the degree of dilution occurring in the river itself. The drainage of these lands for agricultural use has increased the acidity problem since the 1960s. During the most acidic periods the river waters entering the estuary have had a pH of 3.9, while values below five are very common during the spring and autumn floods.

The acidification of the water in the inner estuary (effects of strong acidity and mobilized aluminium) closely resembles the airborne acidification which is threatening poorly buffered lakes and streams (e.g. Drabløs & Tollan 1980, Kämäri 1988, Wright 1988). The results obtained from this extreme environment probably provide some insight into the main effects of strong acidification on the freshwater benthic communities in the brown-coloured waters that are common in Finland.

2. Study area

The River Kyrönjoki (MQ = 45 m^3/s) flows into the southern Bothnian Bay initially as a shallow and narrow inlet extending about 10 km up to the Nabben area (B in Fig. 1), where it bifurcates. The main outflow is directed through the deeper western branch into the open sea, where the salinity in the water column varies from 3.5 to about 4.0‰.

The river is heavily loaded by the runoff from intensive agriculture and it also receives treated waste waters from population centres. The annual mean of the total phosphorus in the river water is 100 μg/l, corresponding to an average load of 154 tons P/a, 32 $kg/km^2/a$ (Meriläinen 1986). The brown river water has an average colour value of about 200 Pt mg/l and is rich in organic matter, the mean concentration of TOC being 21 mg/l (National Board of Waters and Environment). The annual outflow of TOC averages about 29 000 tons, of which 14% is particulate organic carbon (Meriläinen 1985). The oxygen saturation in the river water is fairly high throughout the year, averaging about 84%, with an observed minimum of 66% (National Board of Waters and Environment). Few oxygen measurements on the estuarine waters have been made. According to the observations made in area B (Fig. 1), the oxygen saturation of the near bottom waters was rather high throughout the year, ranging from 66 to 90% (Meriläinen 1984). The uppermost sediment layers were aerobic at all sampling stations shown in Table 1 (Meriläinen 1984). Heikkilä (1986) showed

Table 1. River water percentage (average and min–max) and minimum pH value observed in water (Meriläinen 1985) and sediment quality at the stations studied (the number of the station code indicates the depth in metres). Content of organic matter in % (Org) and water (W) from Meriläinen 1984. The meiobenthos was studied at the stations marked with an asterisk.

Station	River water	Min. pH	Sediment Org	W	Type
A1	99 (36–100)	4.6	24	85	loose mud
A2	99 (36–100)	4.6	9.7	86	loose mixed mud
A3*	99 (36–100)	4.6	6.5	70	loose mixed mud
B1	85 (21–100)	4.6	7.9	70	loose mixed mud
B2	54 (21–100)	4.7	14	86	loose mud
B3*	35 (21–90)	5.5	31	95	loose mud
B5*	10 (6–30)	6.7	23	95	loose mud
C1	63 (27–91)	5.2	5.8	59	mixed mud
C2	57 (27–90)	5.2	15	85	loose mud
C3*	46 (27–90)	5.8	22	90	loose mud
D1	60 (8–97)	5.1	1.2	34	silt
D2	43 (8–94)	5.2	2.4	43	mixed silt
D3*	30 (8–41)	6.7	3.1	58	mixed silt
D5	13 (7–23)	7.0	10	74	loose mud
D10*	<5 (0–10)	7.4	13	81	loose mud
E1	10 (0–24)	7.6	0.3	24	packed fine sand
E2	4 (0–15)	7.6	1.0	30	packed very fine sand
E3*	4 (0–15)	7.6	0.8	29	packed very fine sand
E5	3 (0–15)	7.6	3.5	59	mixed silt
E10*	<2 (..)	7.9	11	82	loose mud
F1	13 (0–38)	6.5	0.4	22	packed medium sand
F2	7 (0–38)	6.9	0.4	22	packed fine sand
F3*	5 (0–23)	7.3	0.1	21	packed fine sand
F5	3 (0–20)	7.4	1.2	30	packed fine sand
F20*	<2 (..)	8.0	5.7	70	loose mixed mud

that the sediments in the inner estuary (area from the river mouth to B in Fig. 1) were reduced to some extent. The Fe/Mn ratio in these sediments was high (44–170) compared to the sediments in the middle and outer estuary, where the ratio ranged from 2 to 29. Heikkilä stated that the reducing conditions were caused by the acidic waters and low bioturbation.

During the investigation period, 7.3.1983–16.5.1984, the average river water content in the surface water (0–3 m) of the inner and middle estuary (areas A, B, C and D) had its lowest value at D3 (30%) and its highest value in the river mouth (99%), the corresponding maximum values ranging from 41 to 100% (Table 1). In the outer estuary the quantity of river water only occasionally exceeded 15%. Soft organogenic sediments prevail in the upper estuary and at deeper stations. In the shallow outer regions (areas E and D) clearly minerogenic sediments are common.

Table 2. Occurrence of acidic floods with pH below 5.0 in the River Kyrönjoki in the period 1970–1984 (from Meriläinen 1985).

Year	Duration in weeks (period)	Variation in pH values	MQ (m^3/s) for the period
1970	7 (30.4.–15.6.)	4.4–4.7	110
1971	5 (13.5.–15.6.)	4.6–4.9	67
1972	9 (21.4–21.6.)	3.9–4.8	140
1973	4 (16.4.–17.5.)	4.4–4.8	100
1974	<1		
1975	*		
1976	6 (26.4–7.6.)	4.5–4.8	68
1977	7 (15.4–16.6)	4.5–4.9	168
1978	<1		
1979	<1		
1980	2 (5.5–19.5)	4.8–4.9	51
1981	<1		
1982	*		
1983	4 (17.4.–15.5.)	4.6–4.9	81
1984	2 (8.5.–20.5)	4.6–4.9	24

* recorded pH values ≥ 5.0

The most acidic periods generally occur when the high spring discharge is decreasing. In the course of the 15 years from 1970 to 1984 very acidic periods have been recorded during nine spring floods (Table 2). The pH measurements made in the estuary showed that about 10% sea water dilution was required to raise the pH from 4.6 to about 5 and about 20% was needed to raise it above 6 (Meriläinen 1985). This means that the acid zone, with a pH of around 5, may sometimes extend far down the estuary, including the whole of the middle estuary. At such times the vertical extent of the acid waters is limited to the uppermost two metres in the outer part of this region, as in area D (Table 1). Acid periods have been found to occur in the autumn as well. However, it is difficult to estimate the duration of these periods, since there only a few pH measurements are taken each month. The acidic autumn periods do not have as far reaching effects on the estuary as the spring periods because the discharge is usually much lower. An unusually acid autumn period is known from 1972. From 8 November to 15 December 1972 the pH lay between 4.3 and 4.9 (7 observations), and the discharge averaged 54 m^3/s. The recent monitoring in the Nabben area (B in Fig. 1) revealed a prolonged acid period extending from 1.10.1986 to 7.1.1987. During this period the pH at 1 m varied from 4.5 to 4.9 ($n = 25$).

According to the water quality monitoring data compiled by the National Board of Waters and Environment, the highest Al concentrations in the river water are usually found during acid periods. In 1979–1986 the mean concentration of total Al in spring (2700 µg/l) was two or three times higher than in other seasons, the annual mean of total Al being 1670 µg/l. The first measurements made on labile Al (monomeric inorganic Al) in the waters of the Kyrönjoki river basin revealed high concentrations during the pH depression in May and August 1988. On 16 May the concentrations at the river mouth and in the Nabben area (area B) varied from 200 to 210 µg/l at pH 4.8–5.0, the total soluble Al being 1100–1400 µg/l. In the Vassorfjärden bay (the southernmost bay near the river mouth) the labile Al was 690 µg/l at pH 4.5 (total Al 2800 mg/l). During the spring period (19.4–12.5.1988) the mean (±*SD*) concentration of labile Al in the upper reaches of the river Kyrönjoki was 53 (±21) µg/l, the total soluble Al being 600 (±154) µg/l (*n* = 20).

In August (24.8.1988) the concentrations of labile Al in the inner estuary were even higher: 320 µg/l (pH 4.9) at the river mouth, 300 µg/l (pH 4.9) in the Nabben area, and as high as 2300 µg/l (pH 4.2) in the Vassorfjärden bay. The corresponding values for the total soluble Al were 1 800, 1 800 and 2 800 µg/l.

The worst sources of Al contaminated water are the runoff waters from the Littorina soils. In order to improve the cultivation conditions of the low-lying fields in these areas, the groundwater level is regulated by pumping. At two of these pumping stations the concentrations of labile Al were (16.5.1988) 12 000 and 14 000 µg/l. The River Lehmäjoki, a tributary of the River Kyrönjoki strongly affected by the runoff from the Littorina soils, had concentrations of 2 900 µg/l (labile Al) and 8 300 µg/l (total Al). The highest value for labile Al, viz. 24 000 µg/l, found so far in these waters was measured on 24 August 1988 in the water discharged from the pump station situated at the southern end of the Vassorfjärden Bay.

3. Material and methods

Estuarine meiofauna was sampled at a total of 10 stations in 6 different areas (Fig. 1). During the same period of investigation these stations were also included in a more comprehensive study on the estuarine macrobenthos, in which the quantitative material was collected from all the 25 stations shown in Table 1 (Meriläinen 1984).

In both June (4–17.6.) and August (19–26.8.) 1980 three quantitative replicate meiobenthos samples were taken from each station with a single 15.2 cm^2 corer (Hakala 1971). For all areas, the stations sampled were located at 3 m and at the maximum depth for the area. In two areas, A and C, the maximum depth is only 3 m, so that only one station was sampled at these two areas.

The uppermost 5–6 cm of each core sample was preserved in a 4% formaldehyde solution buffered with hexamine. The samples were sieved at the laboratory through a set of sieves with mesh sizes of 0.6 mm (to remove macrofauna), 0.2, 0.1 and 0.04 mm. The 0.1-mm fraction was subsampled 1:4 and the 0.04-mm fraction 1:8 in an Askö sample splitter (Dybern et al. 1976). Before sorting, the animals were stained with Rose Bengal (approx. 1 g Rose Bengal in 100 ml of water, plus 5 ml concentrated aqueous solution of phenol). Animal groups (with length measurements) were sorted and oligochaetes were identified as far as species or genus. Calanoids, which were abundant in some samples (up to 8000 ind./m^2), were excluded since they are planktonic animals.

Biomass was estimated as organic carbon (OC). The carbon content of the animals, which were preserved in buffered, unstained formaldehyde, was determined with the Unicarb carbon analyser (Salonen 1979). Before the carbon determination the animals were kept for a few minutes in a series of two baths of deionized water to remove any preservative on their surfaces.

The carbon values used were obtained from the length-carbon regression calculated for each animal group (Nematoda, Tubificidae, Naididae, Ostracoda, Cyclopoida, Harpacticoida, Chironomidae), or from the mean values determined for different size classes of some animal groups. Since the variation of the carbon content was great in certain groups (e.g. Turbellaria) the mean values are only rough estimates. Other sources of error in determining OC-biomass also exist. Preservation for 5 to 7 months before carbon determination probably results in a decrease in the carbon content. This loss in carbon content during the formaldehyde preservation may be high, ranging from about 30% to 50% (Salonen & Sarvala 1980, 1985).

In order to facilitate comparison with other studies the wet and dry mass estimates were made using the average individual biomasses for the different animal groups in the Bothnian Bay as presented by Kautsky et al. (1981). The dry mass was calculated using the conversion factors reported by Ankar & Elmgren (1976). I also used the following wet mass values per individual (calculated from the carbon values):

10 µg for *Amphichaeta sannio* in the 0.2-mm fraction
0.1 µg for Aeolosomatidae in the 0.04-mm fraction
10–100 µg for the different size classes of midge larvae in the 0.2-mm fraction.

To examine possible patterns in the zoobenthic assemblage, the quantitative abundance data of macrobenthos collected from 25 sites (Meriläinen 1984) were subjected to DCA-ordination (Detrended correspondence analysis, Hill 1979). The more comprehensive species-level macrofauna material was more useful in community analysis than the meiofauna material, which was identified mainly to group level. The correlations of the environmental data available for the sampling stations (Table 1) were calculated on the ordination axes produced by DCA. The data for the river water outflow and pH dilution were collected from a different period (1983–1984) to those on the fauna. However, the data for the average river water percentage gave an appropriate idea of the river water influence at different stations, since the yearly differences in the average outflow cannot be high. The minimum pH varies from year to year, as does the duration of the acid period as shown in Table 2, but the observed differences in the minimum pH of the river water between the spring periods of 1980 and 1983 were small.

Two methods were used to determine the specific forms of aluminium in the river waters, the Barnes-Driscoll method (Barnes 1975, LaZerte 1984) and the photometric PCV-method (Seip et al. 1984). In both methods a procedure for the fractionation of aluminium, given by Driscoll (1984), was used. All the other analytical methods for the water quality were those in standard use at the National Board of Waters and Environment.

4. Results

4.1. Differences in meiofauna quantities

Regional variation in the total abundance of the meiobenthos (Tables 3–8) was very high, ranging from 0.038 × 10^6 (the mean value at the river mouth, Table 3) to 2.4 × 10^6 ind./m^2 at 3 m in the outer estuary

Table 3. Mean individual number (10^3 ind./m^2) of the meiobenthos at station A3 and differences in the abundance between the months (one-way ANOVA, statistical significance: *** P<0.001, ** P<0.01, * P<0.05).

	Mean ± *SD*	June	Aug.	
Nematoda	16.2 ± 16.9	1.8	30.5	**
Tubificidae	3.3 ± 2.8	2.0	4.6	NS
Aeolosomatidae	5.9 ± 8.0	–	11.8	NS
Naididae	7.2 ± 8.2	–	14.5	**
Cyclopoida	2.7 ± 1.1	2.9	2.6	NS
Harpacticoida	1.4 ± 2.6	0.2	2.6	NS
Others	1.6 ± ..	0.3	2.9	..
Total	38.3 ± 36.3	7.2	69.5	**

Table 5. Station C3 as in Table 3 (10^3 ind./m^2).

	Mean ± *SD*	June	Aug.	
Turbellaria	4.4 ± 4.9	–	8.8	*
Nematoda	138.4 ± 66.1	113.0	163.9	NS
Tubificidae	1.3 ± 1.5	0.2	2.4	NS
Aeolosomatidae	15.1 ± 24.6	–	30.3	NS
Naididae	30.3 ± 30.0	4.4	56.1	*
Cladocera	2.0 ± 1.9	0.9	3.1	NS
Ostracoda	1.2 ± 1.8	1.5	0.9	NS
Cyclopoida	3.6 ± 4.4	7.1	–	*
Harpacticoida	46.5 ± 51.7	68.2	24.8	NS
Copep. nauplii	2.2 ± 4.2	4.4	–	NS
Chironomidae	7.6 ± 4.8	3.3	11.8	*
Others	0.4 ± ..	0.9	–	..
Total	253.0 ± 124.7	203.9	302.0	NS

(Table 7). The latter abundance from the sheltered station was much higher than that at the exposed station in the outer estuary (F3), 0.96×10^6 ind/m^2 (Table 8). The total numbers at the stations of the middle estuary varied from 0.10×10^6 to 0.76×10^6 ind./m^2. The proportion of small animals steadily grew seawards. The mean percentage of the individuals in the 0.04–0.1 mm fraction remained well below 10% of the total abundance in the inner and middle estuary, whereas the proportion in the outer areas averaged 35% and was highest (46%) at the deepest station, F20.

The mean abundance was usually somewhat lower at the maximum depth than at 3 m. The differences were greatest in area B (Table 4) and E, the individual number at 3 m being more than twice as high as that at the maximum depth in the same area.

The difference in the abundance between the early summer (June) and late summer (August) was great-

Table 4. Stations B3 and B5 as in Table 3 (10^3 ind./m^2).

	Mean ± *SD*	June	Aug.	
B3				
Nematoda	16.3 ± 10.7	12.5	20.2	NS
Tubificidae	1.3 ± 1.5	–	2.6	**
Cladocera	3.1 ± 3.8	0.2	5.9	*
Ostracoda	10.6 ± 10.8	0.9	20.4	***
Cyclopoida	20.4 ± 5.5	20.8	20.0	NS
Harpacticoida	166.4 ± 49.0	129.9	202.9	*
Copep. nauplii	4.4 ± 7.6	8.8	–	NS
Chironomidae	0.8 ± 1.0	0.4	1.1	NS
Others	0.5 ± ..	–	1.1	..
Total	223.8 ± 64.5	173.5	274.2	*
B5				
Nematoda	10.6 ± 20.2	0.9	20.4	***
Tubificidae	0.5 ± 1.1	1.1	–	NS
Naididae	2.5 ± 2.2	1.5	3.5	NS
Cladocera	2.1 ± 3.4	0.2	3.9	NS
Ostracoda	1.0 ± 0.9	0.2	1.8	***
Cyclopoida	41.0 ± 44.8	9.2	72.8	*
Harpacticoida	40.1 ± 32.6	16.2	64.0	*
Chironomidae	1.3 ± 1.6	1.5	1.1	NS
Others	0.8 ± ..	1.0	0.5	..
Total	99.9 ± 84.4	31.8	168.0	***

Table 6. Stations D3 and D10 as in Table 3 (10^3 ind./m^2).

	Mean ± *SD*	June	Aug.	
D3				
Turbellaria	29.8 ± 18.0	16.4	43.2	*
Nematoda	440.0 ± 243.3	219.8	660.2	***
Tubificidae	2.4 ± 2.8	0.2	4.6	*
Naididae	79.9 ± 30.9	106.6	53.3	**
Cladocera	16.6 ± 19.1	–	33.1	**
Ostracoda	57.4 ± 16.7	48.0	66.7	NS
Cyclopoida	5.6 ± 6.8	9.7	1.5	NS
Harpacticoida	112.8 ± 115.3	9.7	216.0	***
Copep. nauplii	9.6 ± 17.4	18.3	0.9	NS
Chironomidae	9.7 ± 8.7	2.9	16.5	*
Total	763.8 ± 366.7	431.6	1096.0	***
D10				
Turbellaria	14.0 ± 87.3	19.7	8.3	*
Nematoda	472.8 ± 187.0	331.2	614.4	*
Tubificidae	2.3 ± 3.1	3.7	0.9	NS
Naididae	29.0 ± 17.7	15.1	42.8	*
Cladocera	0.5 ± 1.0	0.8	0.2	NS
Ostracoda	1.4 ± 1.4	0.2	2.6	**
Cyclopoida	2.6 ± 2.8	4.8	0.4	VI*
Harpacticoida	67.1 ± 63.8	21.7	112.5	***
Copep. nauplii	7.2 ± 9.9	13.6	0.9	NS
Chironomidae	0.1 ± 0.3	–	0.2	NS
Others	0.8 ± ..	1.5	–	..
Total	597.8 ± 253.0	412.3	783.2	NS

Table 7. Stations E3 and E10 as in Table 3 (10^3 ind/m²).

	Mean ± *SD*	June	Aug.	
E3				
Turbellaria	33.8 ± 12.3	41.9	25.7	NS
Nematoda	2094.0 ± 458.2	2192.0	1995.9	NS
Tubificidae	1.5 ± 0.8	1.1	2.0	NS
Naididae	155.1 ± 76.8	220.6	89.5	**
Cladocera	18.4 ± 21.8	0.4	36.4	*
Ostracoda	38.7 ± 33.3	23.2	54.2	NS
Cyclopoida	3.4 ± 3.7	–	6.8	***
Harpacticoida	23.3 ± 13.2	17.3	29.2	NS
Copep. nauplii	4.4 ± 10.7	8.8	–	NS
Chironomidae	24.5 ± 26.1	7.5	43.2	NS
Others	0.8 ± ..	0.3	1.3	..
Total	2398.8 ± 431.6	2513.1	2284.6	NS
E10				
Turbellaria	21.7 ± 12.0	29.6	13.8	NS
Nematoda	979.8 ± 321.9	1262.9	696.8	**
Tubificidae	0.4 ± 0.5	0.8	–	**
Naididae	21.2 ± 17.7	5.7	36.6	**
Cladocera	0.5 ± 1.3	–	1.1	NS
Ostracoda	3.9 ± 3.8	0.7	7.2	**
Cyclopoida	1.4 ± 1.7	0.2	2.6	NS
Harpacticoida	55.7 ± 45.9	20.4	91.0	*
Copep. nauplii	7.5 ± 8.1	14.0	0.9	*
Chironomidae	0.7 ± 1.3	–	1.3	NS
Others	1.2 ± ..	0.1	2.3	..
Total	1094.0 ± 272.5	1334.4	853.6	**

Table 8. Stations F3 and F20 as in Table 3 (10^3 ind/m²).

	Mean ± *SD*	June	Aug.	
F3				
Turbellaria	16.8 ± 16.5	7.5	26.1	NS
Nematoda	454.2 ± 226.6	282.5	626.0	*
Tubificidae	2.1 ± 2.3	–	4.2	**
Aeolosomatidae	353.1 ± 120.2	361.5	344.8	NS
Naididae	26.2 ± 25.7	3.5	48.9	**
Enchytraeidae	1.2 ± 2.3	2.4	–	NS
Cladocera	8.1 ± 17.7	–	16.2	NS
Ostracoda	11.3 ± 13.3	3.7	18.9	NS
Harpacticoida	68.8 ± 46.5	39.0	98.5	*
Copep. nauplii	9.8 ± 15.9	19.5	–	NS
Chironomidae	9.7 ± 11.3	2.0	17.3	NS
Others	1.6 ± ..	–	3.2	..
Total	962.9 ± 289.3	721.6	1204.1	*
F20				
Turbellaria	8.5 ± 4.8	7.7	9.2	NS
Nematoda	737.2 ± 155.1	646.2	828.2	NS
Tubificidae	1.1 ± 1.3	2.2	–	*
Naididae	0.5 ± 0.9	–	1.1	NS
Enchytraeidae	0.3 ± 0.4	0.2	0.4	NS
Cladocera	0.1 ± 0.3	–	0.2	NS
Ostracoda	1.5 ± 1.4	0.8	2.2	NS
Harpacticoida	26.4 ± 14.6	13.4	39.5	*
Others	0.4 ± ..	0.7	–	..
Total	776.0 ± 169.7	671.2	880.8	NS

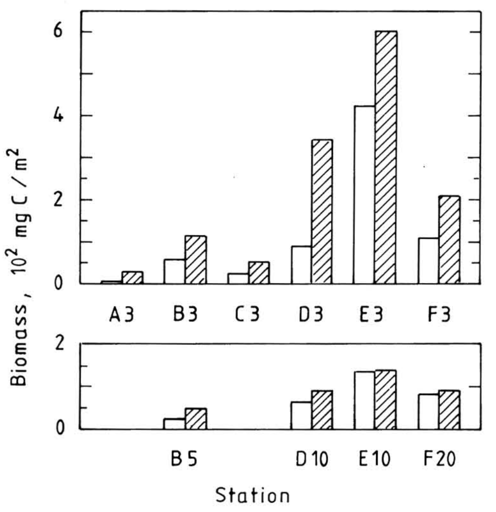

Fig. 2. Biomass of the meiobenthos, mg OC/m² in June (open bars) and in August (hatched bars) at the sampling stations shown in Fig. 1. The number in the station code indicates the depth in metres.

est at the river mouth (A3), where the mean value in August was nearly 10 times as high as in June. The seasonal difference is also reflected in the biomass values (Fig. 2). The biomass at 3 m in all areas and at B5 and D10 was higher in August, whereas the differences were negligible at the greater depths in the outer areas.

In June the biomass at 3 m ranged from a low of 3 mg (A3) to 423 mg OC/m², and in August from 30 to 600 mg OC/m². The variation was not so high at the greater depths, the corresponding values being from 23 to 137 in June and from 50 to 150 mg OC/m² in August.

The mean wet and dry biomasses for the stations are given in Table 9. The correlation coefficient for the regression of dry mass on carbon values was not very good ($r = 0.90$***, $n = 20$). In this material the dry mass values given for Ostracoda were usually too high. The correlation was somewhat better when the regression was calculated leaving out the Ostracoda: $y = 64.5 + 2.14\,X$, $r = 0.96$***.

Table 9. The mean biomass of the meiobenthos given as carbon values (mg OC/m²), dry mass (g/m²) and wet mass (g/m²). Stations shown in Fig. 1.

Station:	A3	B3	B5	C3	D3	D10	E3	E10	F3	F20
Organic carbon	16	88	37	38	216	82	512	143	159	90
Dry mass	0.06	0.4	0.2	0.2	1.4	0.2	1.6	0.4	0.6	0.2
Wet mass	0.3	1.7	0.9	0.9	4.9	1.2	6.3	1.7	3.0	0.9

Table 10. Occurrence of Oligochaeta taxa in the meiobenthos at the different stations (shown in Fig. 1) in the Kyrönjoki estuary. (o = <1 000, + = 1 000...9 999, ++ = 10 000...99 000 and +++ = >100 000 ind./m²)

Stations	A3	B3	B5	C3	D3	D10	E3	E10	F3	F20
Lumbriculidae	–	–	o	–	–	–	o	–	o	–
Limnodrilus spp.	+	o	o	–	o	–	o	o	o	–
Potamothrix hammoniensis (Mich.)	o	+	o	o	+	+	–	+	–	+
Psammoryctides barbatus (Grube)	–	–	–	–	–	o	+	–	+	–
Chaetogaster spp.	+	–	+	++	–	o	–	–	–	–
Amphichaeta sannio Kallst.	–	–	–	+	++	++	+++	++	++	–
Paranais litoralis (Müll.)	–	–	–	–	o	–	o	o	+	–
Uncinais uncinata (Oerst.)	–	–	o	–	o	–	–	–	–	–
Nais communis Pig.	–	–	–	o	–	–	–	–	–	–
N. elinguis Müll.	–	–	o	–	–	o	+	+	o	o
N. simplex Pig.	–	–	–	o	–	–	–	–	–	–
Stylaria lacustris L.	–	–	–	–	o	–	–	–	–	–
Vejdovskyella comata (Vej.)	o	o	–	–	–	–	–	–	–	–
Pristina spp.	o	–	–	–	–	–	–	–	–	–
Enchytraeidae	–	–	o	–	–	–	o	–	+	o
Aeolosomatidae	+	o	–	++	–	–	o	–	+++	–

4.2. Meiofauna composition

The Nematoda were the most abundant group in the estuarine meiobenthos. Abundances were exceptionally low in the innermost areas, A and B (Tables 3, 4). In area B they accounted for only 8% of the total numbers. The highest densities were found at 3 m on very fine sand in the outer estuary (E3, Table 7), but their proportional values were highest at greater depths, e.g. at F20 nematodes made up 95% of the individuals and 76% of the biomass (Table 8).

Harpacticoids were especially numerous in area B, at 3 m on loose mud (166 × 10³ ind./m²). Their proportion of the total was as high as 74% at 3 m and 40% at 5 m, whereas the highest proportional value at other stations was 18% (C3, Table 5). Their proportion of the biomass was also greatest at B3 (48%) and at B5 (32%), but in other areas it fell between 1 (E3) and 15% (D10).

The third major group was the Naididae, of which *Amphichaeta sannio* was by far the most abundant species. This brackish water species was absent from areas A and B, but was relatively numerous at D3 (107 × 10³ ind./m² in June), and had its highest densities on very fine sand at E3 in June (219 × 10³ ind./m²). *Chaetogaster* spp. were quite abundant in the upper estuary and *Nais elinguis* and *Paranais litoralis* in the brackish water area; all other naidids occurred in quite low densities (Table 10). The abundant member of the Aeolosomatidae at F3 (fine sand) was a different species to the larger aeolosomatids which occurred in mud bottoms of the inner estuary.

Some groups were less numerous, but their proportion of the biomass was high in some areas. At the river mouth (A3) tubificids (mainly *Limnodrilus*) made up 60% of the meiobenthic biomass. Ostracods, which were totally absent from the river mouth and quite sparse at C3, accounted for a high proportion (usuallly about 14%) of the biomass at 3 m in the outer areas, but at D3 on mixed silt their biomass, 80 mg OC/m², was as high as 37% of the total. Turbellarians, absent from areas A and B, accounted for about 10% of the biomass at the greater depths in the outer estuary and somewhat less at 3 m (from 4 to 10%). Midge larvae were common at C3 (19% of the total biomass) and in the shallow parts of the outer estuary (5–18%). Benthic cladocerans, almost entirely absent from A3, were rather common in other shallow areas (usually about 5% of the biomass). The most common benthic cladoceran species were *Alona quadrangularis* O.F.M. and *A. affinis* Leydig, the latter being found even at the river mouth. *Ilyocryptus acutifrons* G.O. Sars was common in the upper parts of the estuary and a third *Alona* species (*A. rectangula* G.O. Sars) inhabited the sand areas at F3.

Echinoderes levanderi Karling (Kinorhyncha), a common species in more saline areas like the Bothnian Sea and the Gulf of Finland (e.g. Keynäs & Keynäs 1978, Elmgren et al. 1984), was found only at D10. Water mites (Halacaridae and Hydrachnellae), juvenile molluscs, and juveniles of the amphipods *Pontoporeia affinis* and *Corophium volutator* were occasionally found.

The only group more abundant in June than in August was the Copepoda in the nauplius stage. Most other groups, such as the Nematoda (except at E10), Cladocera, Ostracoda, Harpacticoida and Chironomidae, were far more numerous in August.

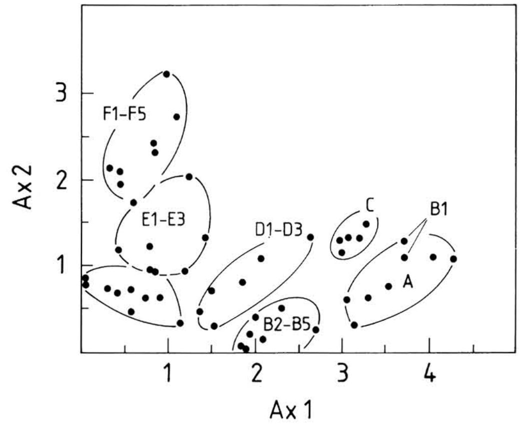

Fig. 3. DCA ordination for the 25 sampling stations in the Kyrönjoki estuary (see Fig. 1). The analysis is based on the quantitative data of the macrobenthic species published in Meriläinen 1984. Axis 1 represents the increasing influence of the river waters. The variation in axis 2 is best explained by the sediment quality: low values represent soft organogenic sediments with high water content, and high values represent tightly packed minerogenic sediments. For example, the group with the symbol F1–F5 represents the rather uniform animal community found in the exposed littoral zone of the outer estuary (including the depths 1, 2, 3 and 5 m). The group without symbols includes all the deeper sites in the middle and outer estuary.

4.3. Zoobenthic communities

DCA-ordination of the quantitative data of macrobenthos grouped the 25 stations into seven patterns (Fig. 3). According to the correlation analyses made, there are two strong abiotic factors controlling the benthic life in the Kyrönjoki estuary. One is the river water, the other the sediment quality. The correlation of the average river water percentage on axis 1 was 0.91***. The river water influence also includes the effects of the periodical acidity; the correlation of the minimum pH was 0.88***. When the sediments were classified into eight groups (Table 1) ranging from soft organogenic (loose mud) to minerogenic sediments (medium sand), the Spearman rank correlation coefficient of sediment type on axis 2 was 0.67***. On the other hand "sediment type" is dependent on many factors, e.g. exposure, currents, wave action, deposition rate, depth, etc., many of which are interrelated. The zoobenthic patterns obtained are considered here to represent (macrobenthic) communities, i.e. groups of species found in a habitat ecologically more closely related to each other than they are to other groups.

The following habitats of communities could be separated. The first four habitats are clearly influenced by the river waters, whereas the others are brackish water habitats in the outer estuary with a nearly constant salinity of 3.5–4‰, 1) Mud bottoms in the river mouth, 2) Mud bottoms in area B with highly varying but strong river water influence, 3) Shallow mud bottoms in the sheltered bay of Kvimofjärden, area C, with a distinct fresh water influence, 4) Shallow silt bottoms of the middle estuary (area D) with a periodically strong river influence, 5) Shallow minerogenic beds down to 3 m (very fine sand or fine sand) in a rather sheltered area E in the outer estuary, 6) Minerogenic beds down to 5 m (fine or medium sand) in a more exposed area F, and finally 7) Deeper, mainly soft mud bottoms at depths of 5–20 m in the middle and outer parts of the estuary. The main features of the communities are given in Tables 11 and 12.

Table 11. The main features of the zoobenthic communities in the inner and middle parts of the Kyrönjoki estuary. The biomasses are given as organic dry mass for macrobenthos and organic carbon for meiobenthos. The average values of Shannon diversity are based on the ln- values of abundances. The total number of taxa includes all macrobenthic data, as well as meiobenthic oligochaetes. Location of the stations as shown in Fig. 1. (References: Meriläinen 1984, Bagge & Meriläinen 1985 and this study.)

Stations:	River mouth A1–A3	Inner archipelago B1–B5	C1–C3	D1–D3
Macrobenthos				
Biomass, ODM g/m²	0.12	0.27	0.29	0.90
Individuals/m²	400	700	2000	3900
Diversity	1.0	1.1	1.4	2.0
Species with highest biomass	*Limnodrilus hoffmeisteri*	*Potamothrix hammoniensis*	*Tanytarsus* spp. *Endochironomus albipennis*	*Potamothrix hammoniensis* *Tanytarsus* spp. *Pontoporeia affinis*
Number of brackish water species	–	–	1	4
Number of taxa				
Mollusca	–	4	6	6
Ephemeroptera	–	2	3	2
Crustacea	1	4	3	8
Hydracarina	3	18	21	18
Diptera	17	18	24	30
Oligochaeta	6	3	4	10
Meiobenthos (at 3 m)				
Biomass, OC mg/m²	16	88	38	216
10³ individuals/m²	38	224	253	764
Main groups (% of tot. biomass)	Oligochaeta (68) Nematoda (10)	Harpacticoida (48) Cyclopoida (14)	Nematoda (36) Chironomidae (19)	Ostracoda (37) Nematoda (16)
Total number of taxa	43	64	83	90
River water influence	drastic (acidity)	strong (esp. in shallow areas)	distinct (partly from other rivers)	distinct (low salinity)

Table 12. The main features of the zoobenthic communities in the outer estuary (exposed and sheltered littoral) and in deeper bottoms in the middle and outer estuary. (Explanations in Table 11.)

	Exposed littoral F1–F5	Sheltered littoral E1–E3	Deeper bottoms D5–D10, E5–E10, F20
Macrobenthos			
Biomass, ODM g/m²	3.3	2.1	1.7
Individuals/m²	1400	6300	4900
Diversity	2.0	2.0	0.9
Species with highest biomass	*Macoma baltica* *Mesidotea entomon*	*Potamopyrgus jenkinsi* *Pontoporeia*	*Pontoporeia affinis*
Number of brackish water species	12	9	6
Meiobenthos	(3 m)	(3 m)	(Sts. D10, E10, F20)
Biomass, OC mg/m²	160	512	105
10³ individuals/m²	963	2399	823
Main groups (% of tot. biomass)	Nematoda (23) Aeolosomatidae (16) Ostracoda (14)	Nematoda (41) Chironomidae (18) Ostracoda (12)	Nematoda (62) Turbellaria (10) Harpacticoida (9)

5. Discussion

5.1. Comparison with other areas

This study revealed that seasonal differences (between early and late summer) in the abundance and composition of the meiofauna may be high as they were in the innermost parts of the estuary, where the variation in the water quality is high. These differences should be borne in mind when comparing the fauna of different studies. The comparisons made below refer to the average values obtained during the investigation period.

A comparison with the few results available from the Bothnian Bay suggests that the proportion of the smallest fraction of the meiofauna (0.04–0.1 mm) is lower in the shallow coastal areas than in the open Bothnian Bay. The average value of 66% of the total abundance is reported for the Gulf of Bothnia (Elmgren et al. 1984). In the archipelago of the Luleå river, in the northwestern part of the Bothnian Bay, the mean value at depths of 0–6 m was 35% (Kautsky et al. 1981), and in the outer parts in the Kyrönjoki estuary it varied from 30 to 46%. It is interesting to note that Sarvala (1986) found a decrease in the mean size of benthic harpacticoids and omnivorous cyclopoids with increasing depth in an oligotrophic Finnish lake.

The hypothesis of decreasing size of benthic organisms with increasing depth in oceans was presented by Thiel (1975). In spite of the basic differences in the overall size spectra between marine and lacustrine zoobenthos (see details e.g. Warwick 1984, Strayer 1986), one can ask whether the hypothesis could not be put forward in a more embracing way to include inland seas and larger lakes.

According to Elmgren et al. (1984) the mean abundance for the whole Gulf of Bothnia at depths greater than 5 m was 1.2×10^6 ind./m^2, and the wet biomass 1.8 g/m^2. These values are slightly higher than the average values (0.8×10^6 ind. and 1.3 g/m^2) in deeper areas ($\geq$10 m) of the Kyrönjoki estuary but much lower than the mean at 3 m in the outer estuary (see Table 9).

In the Luleå archipelago, the weighted mean abundance was 0.95×10^6 ind. and dry biomass 0.3 g/m^2 in the 0–6 m depth zone (Kautsky et al. 1981). There the biomass maximum was attained on muddy bottoms at 0.5–2 m, but it amounted to only 0.61 g/m^2 (dry mass). These biomass values are clearly lower than those in the outer parts of the Kyrönjoki estuary. The mean dry biomass (0.3 g/m^2) is very near to those in the middle estuary (see dry mass values in Table 9), although the individual numbers are lower here than in the Luleå area. The difference arises from the fact that larger meiobenthic animals, like Ostracoda, are less numerous in the Luleå area.

Very low abundances were reported by Valtonen et al. (1984) from the northeastern Bothnian Bay. The values averaged from about 0.1×10^6 in sand and clay areas to 0.3×10^6 ind./m^2 in muddy bottoms. These values were obtained from depths of 22–95 m and they indicate the extreme poverty of the deeper bottoms in the northern Bothnian Bay.

5.2. Zoobenthic communities in the Kyrönjoki estuary

Zoobenthic animals of the estuary can be considered to form a community continuum from a fresh water ecosystem to a brackish water ecosystem. In this continuum it is possible to distinguish several more or less distinct communities controlled by various combinations of environmental factors. Most of the 13 macrobenthic brackish-marine species found in the outer estuary were not able to live in the shallow water communities of the middle estuary (Meriläinen 1984). The decreasing salinity from the outer estuary to the middle estuary thus constitutes a boundary for the brackish-marine species. The shallow water communities in the middle estuary (section 5.2.2) can also be considered as freshwater communities influenced by brackish water, since the fauna is composed mainly of freshwater species (Table 11).

5.2.1. Communities of the outer estuary

In addition to salinity, sediment quality, in a broad sense, is one of the most decisive factors controlling the benthic life of the outer estuary (Fig. 3), where the influence of the river waters remain low. Due to methodological difficulties, quantitative data on the animals of the rocky and stony shores common in the outermost parts of the estuary are lacking.

Distinct differences were found between the communities of the sheltered and more open littoral sites (Table 12 and Meriläinen 1984). Owing to the high abundances of the large-sized *Macoma baltica* and *Mesidotea entomon*, the macrobenthic biomass was higher in the exposed area than in the sheltered littoral. In the latter area many small insect larvae (e.g. *Polypedilum nubeculosum, Tanytarsus* spp. and *Cladotanytarsus* spp.) were abundant. Contrary to the case with the macrofauna biomass, the meiofauna biomass was higher in the sheltered area. The Aeolosomatidae abundant (353×10^3 ind./m^2) on fine sand at the exposed site (F3) were sparse on very fine sand at the sheltered site (E3) and they were not found in deeper areas (Table 10).

All the deeper bottoms harboured a rather distinct community, where *Pontoporeia affinis* was by far the most abundant macrobenthic species, nematodes being the most numerous in the meiofauna fraction. *Macoma baltica* was very sparse in deeper areas of the middle estuary, but fairly abundant in outer deep areas (Meriläinen 1984). Differences in the composition of the meiofauna at group level were rather small, Nematoda, Turbellaria and Harpacticoida being the most abundant groups in all deeper areas.

5.2.2. Communities influenced by the river waters

The river water influence is pronounced even in the outermost shallow water community of the middle estuary (D1–D3 in Fig. 3). There, only four macrobenthic brackish water species occur, viz. *Paranais litoralis, Nais elinguis, Neomysis integer*, and *Gammarus zaddachi*. Equivalent facts about the meiofauna cannot be presented, but two of the six meiobenthic oligochaete species found at this site were brackish-marine species (Table 10). It is possible that species in some other meiobenthic groups are more eurytopic, e.g. most of the 17 harpacticoid taxa reported by Kaytsky et al. (1981) from the low-saline Luleå archipelago (salinity mainly below 3‰) were eurytopic brackish-marine species. The biomass of macrobenthic animals remains rather low (Table 11), because the large brackish water species, such as *Macoma* and *Mesidotea*, are absent from this community. The meiobenthic biomass is higher than in the exposed site (F3) but lower than in the sheltered site (E3) in the outer estuary (cf. Table 12).

Area C is a rather sheltered, shallow basin, where the benthic animals are almost wholly of freshwater origin, mainly chironomids (Table 11). The area is highly productive, since the small midge species usually have two generations during the growing season. Water mites, which are mainly carnivorous, are very abundant in this area. Bagge & Meriläinen (1985) found the average catch of 24 hours to be as high as 98 individuals/trap, whereas in other parts of the estuary the catch ranged from 0.5 (at the river mouth) to 29 (in area D). Both the macro- and meiobenthic biomass remained well below those in area D (Table 11).

The river water influence is strong in area B but the effects, discussed in detail in the next section, are not so severe as in the river mouth area. For example, station B1 is grouped into "group A" in Fig. 3 (because of the high proportion of *Limnodrilus hoffmeisteri*) but there are no great differences in animal composition between B1 and the other depths in area B. Some freshwater molluscs were also able to live at B1 but not in the river mouth (Table 11 and Meriläinen 1984).

Both the macrobenthic and meiobenthic fauna is poor in the inner estuary (area A in Table 11). Meriläinen (1984) showed that several freshwater species found in the low-saline areas of the middle estuary, and even the outer estuary, were totally absent from the inner estuary. The most evident reason for the deteriorated fauna is the periodical acidity of the river waters.

5.3. Effects of acid river waters

Certain animal groups known to be sensitive to acidity, such as Gastropoda, Lamellibranchiata, Ephemeroptera and Hirudinea (Wiederholm & Eriksson 1977, Mossberg & Nyberg 1979, Økland & Kuiper 1980, Engblom & Lingdell 1984, Raddum & Fjellheim 1984, Leuven et al. 1986, Økland & Økland 1986) were totally absent from the inner estuary. According to Meriläinen & Hynynen (1988), *Lymnaea peregra* is the most acid tolerant snail of Finnish lakes, but it was absent from lakes with a minimum pH of below 5.3 and lakes with a mean pH of below 5.7. The innermost finds of this species in the estuary were from area B, about 10 km seawards from the river mouth (Meriläinen 1984). This was the innermost limit also for *Gyraulus albus* and *Bithynia tentaculata*, but the first occurrences of the other freshwater snails (*Valvata piscinalis* and *V. pulchella*) were in the middle estuary about 15 km seawards from the river mouth. The innermost finds of freshwater mussels were those of *Pisidium casertanum* from the area about 6 km seawards from the river mouth. *P. casertanum* was the most acid tolerant species of the 12 species of small mussels found in acid and acid sensitive lakes (Meriläinen & Hynynen 1988). It was absent from lakes of minimum pH below 4.5 and lakes of mean pH below 5.0. The corresponding pH-limits in lakes for *Caenis horaria*, the only common mayfly species in the middle estuary, were 5.3 (min. pH) and 6.0 (average pH) and for leeches 4.8 and 5.3. These pH-limits cannot be regarded as absolute values, but they give a general idea of the sensitivity of these species to acidic waters.

Meiobenthic Turbellaria and Ostracoda were also absent from the river mouth. This may indicate that the freshwater species of both these groups are sensitive to acid waters. The meiofauna material from the inner estuary was, however, scanty compared to that of macrofauna sampled with several different methods from different depths in the area (see Meriläinen 1984, Bagge & Meriläinen 1985).

The extent of the change in the distribution of freshwater species is underscored by the fact that the species-rich groups, like Chironomidae and Hydracarina, had only a few representatives in the inner estuary (Table 11). Many chironomid species are tolerant to an acidity of about pH 4–4.5 and usually they contribute the major part of the macrobenthic abundance in acidified waters, especially in lakes (Wiederholm & Eriksson 1977, Mossberg 1979, Mossberg & Nyberg 1979, Buskens et al. 1986, Bradt & Berg 1987). However, the number of midge taxa decrease with increasing acidity in lakes (Leuven et al. 1986, Meriläinen & Hynynen 1988). Observations on water mites in the estuary suggest that they are not an acid tolerant group, even though some species were able to live in the inner estuary. Unidentified water mites have been found at pH 4.2 in Scotland (Harriman & Morrison 1982).

Many oligochaetes tolerate the conditions in the upper estuary. *Limnodrilus hoffmeisteri* was the most abundant animal in both the macro- and meiofauna of the innermost zone of the estuary. However, the species (including immature specimens of the genus) was not found in lakes with an observed pH minimum of below 4.7 or in lakes with an average pH of below 5.3 (Meriläinen & Hynynen 1988). This shows that it can survive in areas with periodical acidity but it is not able to tolerate prolonged acidity.

5.4. Acidity and aluminium

The effects of acid waters on animals, like the effects of any single factor, is usually a complicated phenomenon in nature (see e.g. Økland & Økland 1986). The effects can be divided simply into direct and indirect effects. The acidity can directly affect the physiology of the species. Indirectly acidity may increase the toxic effects of other substances, or it can increase the concentration of the toxic fraction of substances. Some studies suggest that acid water affects the food supply, which in turn affects the number of animals (see e.g. Sutcliffe & Carric 1973, Ziemann 1975, Townsend et al. 1983). As stated by Townsend et al. (1983), it should be noted that the various hypotheses are not mutually exclusive, but that they can all play some role in combination. Many investigators have concluded that mobilized aluminium, in addition to low pH, is a major determinant of the abundance and distribution of aquatic invertebrates in acidified waters (Hall et al. 1980, Hermann & Baron 1980, Harriman & Morrison 1982, Havas & Hutchinson 1982, Burton & Allan 1986, Wicham et al. 1987, Hall et al. 1987).

Hall et al. (1987) found in their experimental field study that insect drift increased in streams at pH 5.5 with a monomeric inorganic Al concentration of 200 µg/l but not at pH values ranging from 5.25 to 5.5 at lower Al concentrations (≤12 µg/l). They concluded that mayflies and some midges were sensitive to dissolved Al mobilised during short-term pH pulses that by themselves were not stressful.

Monomeric inorganic Al is the most toxic form of Al in waters. Driscoll et al. (1980) and Baker & Schofield (1982) showed in laboratory studies on fish that the most toxic fractions were AlOH complexes, AlF complexes were moderately toxic, and organically complexed Al was non-toxic. The experiments made by Clark & Hall (1985) suggest that AlF complexes were toxic to amphibians.

Hildén & Hirvi (1987) found the critical limit for perch survival to be about pH 5 in the test water sampled from the Kyrönjoki estuary. According to them, the most likely explanation for the high critical pH value in the estuarine waters was the high aluminium concentration present in test water. They concluded that the lethal effect may have been caused by elevated concentration of monomeric inorganic aluminium, since the measured concentration of soluble and total aluminium was high, 0.88–2.11 mg/l and 1.39–3.47 mg/l, respectively. The first observations made on labile inorganic Al in the river waters support this view. During the pH depressions in 1988 the concentrations of labile Al ranged from 200 to 320 µg/l in the inner estuary, and they were even higher in the innermost bay near the river mouth (690–2300 µg/l). It is evident that these concentrations are high enough to cause additional stress and enhance the effects of acid waters in the inner estuary. Experimental studies have also shown that several invertebrates (Hall et al. 1980, 1987) and amphibian embryos (Clark & Hall 1985) exposed to low pH and high concentrations of monomeric inorganic Al can die during a short time period.

Table 13. Mean metal concentrations (± *SD*) in the river water at Skatila found during high acidity periods, in the Vassorfjärden bay (the bay south from the area A in Fig. 1) and in the acid runoff from the Littorina soil. (Unpublished data of Mr. Matti Verta, National Board of Waters and Environment.)

	River water	Vassorfjärden bay	Acid runoff
Fe (mg/l)	2.27 ± 0.80	1.90 ± 0.33	21.55 ± 27.51
Mn (mg/l)	0.40 ± 0.20	0.39 ± 0.04	19.30 ± 25.03
Cd (μg/l)	37 ± 22	..	..
Ni (μg/l)	16 ± 8	202 ± 319	581 ± 429
Zn (μg/l)	7 ± 4	189 ± 306	330 ± 276
Cu (μg/l)	3 ± 1	69 ± 114	46 ± 28

It has been shown that acid runoff contains high concentrations of some other metals, but the concentrations in the river water during the pH depression are not very high (Table 13). These concentrations may have some importance in areas where the influence of the acid runoff is greatest, as e.g. in the Vassorfjärden bay. According to Heikkilä (1986) the concentration of heavy metals in sediments of the inner Kyrönjoki estuary was equal (for Hg) or somewhat lower than those in the middle and outer estuary. The concentrations were not high and they remained well below those values usually reported from metal polluted areas. The following concentrations (mean ±*SD*, μg/g dry sediment) were found in the inner estuary: Pb, 25 ±10; Cu, 23 ±6; Zn, 128 ±32; Cd, 1.9 ±0.33 and Hg, 0.085 ±0.027.

What are the reasons for the low biomass of macro- and meiobenthic invertebrates in the inner estuary? It has been shown in many works that benthic biomass has no significant correlation with water pH (Wiederholm & Eriksson 1977, Mossberg & Nyberg 1979, Dermott 1985, Dermott et al. 1986, Buskens et al. 1986, Bradt & Berg 1987, Meriläinen & Hynynen 1988). Opposite findings also exist (see Økland & Økland 1986) and at least four theories have been put forward to explain this disagreement. One of these theories suggests that the acid sensitive species are to some extent compensated by an increase in acid tolerant large insect species when the fish predation (population) has decreased in acidified waters. The importance of fish predation in structuring limnetic animal communities is stressed in many studies (Grahn et al. 1974, Mossberg & Nyberg 1979, Eriksson et al. 1980). Bendell & McNicol (1987) even concluded that fish predation is the most immediate factor structuring aquatic insect assemblages and is responsible for their change coincident with lake acidification.

It is noteworthy that the lowered (or nearly lacking) fish predation in the inner estuary (Hudd et al. 1984) has not led to an increase either in the benthic biomass or in the number of acid tolerant species. I assume that the high concentrations of toxic Al complexes during acidic periods are mainly responsible for the low benthic biomass in the inner estuary. One additional stress factor is the continually changing chemical environment, as the widely varying pH (from about 4 to 7) may prevent the development of a more stable community composed of acid tolerant species.

5.5. Acidity, aluminium and humus

Recent studies have shown that interactive effects between pH, Al and organic matter are very important in determining the sensitivity of animals to acidic waters. Burton & Allan (1986) found in their experimental field study that the toxicity of Al appeared to be influenced by the organic matter of water through its influence on Al speciation. Toxic monomeric Al only existed as a high percentage of the total Al in water with a low organic content. Humus rich water may diminish the toxic effects of other metals occurring in high concentrations in runoff from the sulphureous soil (Table 13), e.g. Paulauskis & Winner (1988) demonstrated that humic acid decreases the cronic and acute toxicity of Zn on *Daphnia magna*. Hargeby & Petersen (1988) found that added humus decreased the mortality of *Gammarus pulex* in test water with a pH of 6.0. Humus tended to benefit food conversion and it was suggested that a low concentration of humus can be directly beneficial to organisms in acidified water in ways other than by complexing toxic metals.

It is highly evident that the high concentration of organic matter in the water of the River Kyrönjoki diminishes the toxic effects of Al. During normal pH conditions (pH above 6) the mobilized Al exists mainly as organically complexed, nontoxic Al. During the pH depressions the concentrations of toxic Al fractions increase in the waters of the river and its estuary. The highest concentrations of toxic Al complexes probably occur during the worst pH depressions, when the organic matter of the brown-coloured river water is precipitated and the water turns almost transparent (see Sevola 1979).

All the studies concerning the distribution of the invertebrates in the Kyrönjoki estuary were made during a period when the acid floods were relatively restricted (Table 2). It can be concluded that during the worse periods, as for example in the early 1970s, all the shallow water communities in the middle and inner estuary have suffered from the acid outflow. The innermost zone, characterized by a very impoverished fauna, has been wider than it was in 1980–1981, when the zone extended 7–8 km from the river mouth and covered an area of about 11–15 km^2 (cf. Meriläinen 1984).

Acknowledgements: I should like to express my thanks to my "Nabben" colleague, Esa Koskenniemi for assistance in the field work, to Matti Verta for permission to use his unpublished data and to Pauli Bagge, Ronald Dermott, Jouko Sarvala and Jukka Särkkä for critical reading of the manuscript. Lasse Hakkari identified most of the cladocerans. Leigh Plester kindly revised the language.

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Received 3.I.1988, revised 3.XI.1988
Printed 30.XII.1988