

**PENTTI KANGAS AND MIKAELA FORSSKÅHL (eds.)**

**PROCEEDINGS OF THE THIRD FINNISH-SWEDISH  
SEMINAR ON THE GULF OF BOTHNIA**

**Pori, Finland, August 20–21, 1984**

**VESI- JA YMPÄRISTÖHALLITUS —  
NATIONAL BOARD OF WATERS AND ENVIRONMENT, FINLAND  
Helsinki 1986**

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ISBN 951-47-0881-4  
ISSN 0355-0982

Helsinki 1987. Valtion painatuskeskus

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## PREFACE

The primary aims of the Finnish-Swedish co-operation agreement on the Gulf of Bothnia is to study the state of pollution of the Gulf of Bothnia, to detect possible long-term changes in the sea and to collect the information necessary for decisions on the utilization and protection of the marine environment. To acquire information on current scientific research, the Committee for the Gulf of Bothnia has organized seminars. The first seminar was held in Vaasa, Finland, in 1978 (Finnish Marine Research No. 244) and the second in Luleå, Sweden, in 1981 (Naturvårdsverket SNV PM No. 1618).

The third seminar on the Gulf of Bothnia was held on 20—21 August 1984 in Pori, Finland. It was attended by about 120 representatives from

various authorities, research institutes and universities in Sweden and Finland. The greater part of scientific results was presented in posters; the 41 scientific articles published in this volume are based on 47 posters and 10 lectures. Thus most of the texts are very brief. The articles have been grouped according to the fields of the four working groups of the Committee for the Gulf of Bothnia. In each of them the papers are given in alphabetical order after the concluding presentation of the respective group.

The manuscripts have been reviewed by the experts of the working groups of the Committee. As the technical standard of the figures varied greatly, they have been redrawn by Mrs. Paula Ullakko according to the editors' concepts. The English of the whole volume has been revised by Mrs. Anna A. Damström.

Helsinki, June 1986

Pentti Kangas and Mikaela Forsskåhl

The editors of the proceedings

## OPENING ADDRESS

Honoured seminar participants

Concern about trends in the condition of the Baltic Sea is nothing new. Scientists were already discussing this subject at the end of last century and heated controversy was caused in the late 1960s by the widespread oxygen deficiency observed in the basin water. So far, scientists have not been able to determine definitely what part is played in these changes by natural processes and what part by man. Irrespective of the total extent of human influence, however, the administrative policy should be to ensure that the burden imposed upon the marine environment is not excessive. Future generations must inherit a healthy living sea.

The Gulf of Bothnia is the most important separate basin system in the Baltic Sea. In 1972 an agreement was reached on the coordination of research aimed at environmental protection in the Gulf. The basis of the agreement was the initiative taken by the Nordic Council in 1970. According to the agreement, the coordination of research between Finland and Sweden is in charge of the Committee of the Gulf of Bothnia, whose members in Finland are the Board of Waters and the Institute of Marine Research and in Sweden the Nature Conservation Department.

When the convention for the protection of the marine environment of the Baltic came into being in 1974, it was natural that the Committee of the Gulf of Bothnia should be made responsible for the research duties included in the convention, as had already been provided in the convention text. The resultant collaboration has been consistently fruitful.

The bilateral collaboration between Finland and Sweden can be considered a good example of how effective such work can be. It has also provided a forum where new matters can be tested and initiatives developed further for treatment by the Baltic Commission. The main duty of the Committee for the Gulf of Bothnia is to collect a common body of information as a basis for decisions regarding water conservation. The Committee has also taken steps to direct research in accordance with the need for water conservation measures and mapped problem areas in research work. From 1978 onwards, summaries have been made regarding the loading of the Gulf of Bothnia. Important help in the elucidation of the condition of the Gulf has been received from the seminars arranged from time to time. The first was held in Vaasa in 1978, the second in Luleå in 1981 and this seminar in Pori will be the third. The abundance of

results reported in all three has been extremely satisfactory; for example, besides the general lectures and introduction to discussions, the present seminar will comprise over 40 special reports. I believe that this will constitute a firm basis for an assessment of the situation in the Gulf of Bothnia during the last few years, of the measures called for by the situation and of the recommendations that should be made for the research programme for the coming years.

One of the most important duties of the Ministry of the Environment, which started work last October, is the protection of the marine environment. The condition of the sea area around Finland is strongly influenced by measures taken in the interior, along rivers, in built-up areas and in business and industry. The work of the Ministry involves decisions which will have far-reaching and long-lasting effects, decisions that influence the economy and are often closely concerned with economic and regional policy. In making such decisions, it is of the utmost importance to have access to the best and most objective information available on the effect of the different alternatives on the environment. This necessitates comprehensive and continuous research. The collection of data and assessment of their reliability is greatly facilitated if a system is always at hand for this purpose. For this reason the Ministry attaches great importance to the findings of this seminar. Both Sweden and Finland have sent representatives from the institutes and university departments undertaking research on the Gulf of Bothnia. The range of the research is also remarkably wide; extending from biological studies to the effect of technical measures, it embraces loading of terrestrial and atmospheric origin and the processes taking place in the sea. I believe that the corresponding authorities in Sweden share my opinion of the importance of the findings of this seminar.

Environmental research, including research on the marine environment, is one of the points of focus of the work of the Ministry of the Environment. It is the aim of the Ministry to achieve a strong increase in the resources available for research. For example, research on increasing acidity is just beginning and the Government has decided to invest 37 million marks in this project during the next five years. As I have already observed, the decisions taken in environmental management should be based on scientific research that is as competent and reliable as possible. This requirement is strengthened by the great, sometimes even critical, economic importance of the decisions. Evidence of this can be found no more than a few kilometres from here, at the Vuorikemia

factory of Kemira Ltd. — I hope that research — also within the framework of the Committee for the Gulf of Bothnia — will make it possible to follow the condition of the Gulf at these latitudes as well and lead to solutions allowing an improvement of both the water quality and the treatment of the factory effluent.

On behalf of the Ministry of the Environment, I bid all the participants in the seminar most heartily welcome to Pori. I hope that your discussions will be fruitful and provide the results that are needed by us in the administration. I have pleasure in declaring the seminar open.

Pori, 20 August 1984

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## THE HYDROGRAPHY OF THE GULF OF BOTHNIA. REPORT FROM THE HYDROGRAPHY GROUP

Stig Fonselius<sup>1)</sup>

FONSELIUS, S.H. 1986. The hydrography of the Gulf of Bothnia. Report from the hydrography group. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

A short description is given of the general hydrography of the Gulf of Bothnia. Diagrams showing annual mean values of salinity at three representative hydrographic stations reveal a long-term salinity increase in the deep water since the beginning of the century. The salinity trends since the 1960s in the surface and the deep water are discussed and diagrams showing salinities, regression lines, correlation coefficients and confidence levels at stations F 64, SR 5 and F 9 are given. Similar diagrams showing increasing trends for total phosphorus and nitrate at these stations are discussed. The nutrient relations in the surface water of the Baltic Sea are figured and differences in the N:P ratio are demonstrated between the Bothnian Bay and the Bothnian Sea. Finally, comparisons are made between the calculated and measured nutrient concentrations in the surface water and a table showing the calculations is given.

Index words: Salinity, phosphate, total phosphorus, nitrate, long-term trends, nutrient relations, input values, volumes, calculated nutrient values, measured nutrient values, maximum depths, sill depths, hydrography

### 1. GENERAL HYDROGRAPHY

The Gulf of Bothnia forms a hydrographically isolated part of the Baltic Sea and therefore has distinctive properties. Geographically, it comprises the Åland Sea, the Archipelago Sea, the Southern Quark, the Bothnian Sea, the Quark and the Bothnian Bay.

The border between the Gulf of Bothnia and the Baltic Proper is generally drawn along the ridge running from the Swedish side to the lighthouse Svenska Björn, then on to Utö on the Finnish side and to the Hangö point. This line is also the southern border of the Archipelago Sea and the Åland Sea. The sill depth is generally reported to be 40 m, but during recent times a 70 m deep

channel has been found, which cuts through the sill and allows small amounts of deep water to flow into the Gulf of Bothnia. The Åland Sea is divided into the Southern Åland Sea and the Northern Åland Sea. These are separated by a 65 m deep sill between Söderarm and Lågskär. The maximum depth in the Southern Åland Sea is 219 m and in the Northern Åland Sea it is 301 m.

An approximately 100 m deep channel goes from the Northern Åland Sea to the deep area of the Bothnian Sea through the Southern Quark. The deep area of the Bothnian Sea extends northward, east of the centrally situated Finngrundsbanks and then turns westward to the deepest spot in the Bothnian Sea, the Ulvö Deep located close to the Swedish coast SSE of Ulvön. On Swedish nautical

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charts the depth is given as 293 m. Winterhalter (1981) reports 249 m and this depth is also found on the Navy relief chart.

According to all published hydrographic measurements, both Finnish and Swedish, depths of more than 212 m have never been recorded at the station Ulvödjupet (F 24) since the beginning of the century.

The border of the Bothnian Bay is located in the Northern Quark and is delimited by a sill with a depth of 25 m. In the Bothnian Bay the maximum depth is 147 m (Tulkki 1977). The deepest hydrographic series are from 130 m.

## 2. WATER BALANCE

The water balance is positive and is regulated completely by the large discharge of river water. This gives the Gulf of Bothnia a very low salinity, the value fluctuating between 1.5 and 4.5 units in the Bothnian Bay and 3.5 and 8.5 in the Bothnian Sea. In both basins the river discharge is around  $97 \text{ km}^3 \text{ a}^{-1}$ . Evaporation and precipitation are difficult to estimate. Brogmus (1952) reports that the evaporation is around  $32 \text{ km}^3 \text{ a}^{-1}$  in the Bothnian Sea and around  $12 \text{ km}^3 \text{ a}^{-1}$  in the Bothnian Bay, and the precipitation 35.5 and  $15 \text{ km}^3 \text{ a}^{-1}$ , respectively.

Dahlin (1976a) has calculated the water exchange between the Bothnian Bay and the Bothnian Sea, and between the Bothnian Sea and the Baltic Proper. The outflow and inflow values for the Bothnian Bay were  $365 \text{ km}^3 \text{ a}^{-1}$  and  $246 \text{ km}^3 \text{ a}^{-1}$ , respectively; the corresponding values for the Bothnian Sea were  $1570 \text{ km}^3 \text{ a}^{-1}$  and  $1380 \text{ km}^3 \text{ a}^{-1}$ . An outgoing current dominates in the surface and an ingoing current in the deep water. In both basins, the annual mean circulation of the surface water is counterclockwise, probably due to the Coriolis force. This means that we find the lowest salinities along the Swedish coast, in an average southerly current. The water is weakly stratified, with a weakly developed halocline. In the Bothnian Sea this is situated at 50–70 m depth and in the Bothnian Bay at 20–40 m. The deep water in the Bothnian Sea is formed by cold surface water from the northern Baltic Proper, which during the winter sinks down over the Åland Sea sill and flows along the bottom into the Bothnian Sea, after first filling the Åland Sea up to the depth of the sill. In the Bothnian Bay the deep water is formed in the same manner by cold winter water sinking down

over the sill in the Northern Quark.

Since only insignificant amounts of the oxygen-deficit deep water of the Baltic Proper can penetrate into the deeper parts of the Bothnian Sea, the oxygen conditions there are good and no lack of oxygen is found. The oxygen conditions are still better in the Bothnian Bay. In both areas the vertical winter convection extends to all parts, with the exception of the deepest bottom areas.

## 3. MONITORING ACTIVITIES

From the late 1890s up to the end of World War I (1918), the main part of the hydrographic measurements with research vessels in the Gulf of Bothnia were carried out by Finland. Only sporadic measurements at single stations were performed by Sweden. Swedish (1880–1969) and Finnish (1904–1964) light-vessels, however, carried out long series of measurements of temperature, salinity and currents. Unfortunately all light-vessels have now been replaced by permanent unmanned light-houses. Finland operates a number of permanent coastal stations, which have worked almost without interruption since the beginning of the century. On the Swedish side observations were started some 10 years ago with funds from the Environment Protection Board. The sampling is carried out by SMHI (the Swedish Meteorological and Hydrological Institute) with the help of coastguard vessels.

As the Gulf of Bothnia is normally covered by ice during the winter, only summer and autumn measurements were carried out earlier. The light-vessels were also withdrawn during the ice period, when the conditions became too severe. During the 1960s Professor Palosuo, from the Institute of Marine Research in Helsinki, worked on Finnish Icebreakers and carried out measurements of temperature and salinity. On the Swedish side SMHI began to carry out temperature measurements of the surface water with the help of the personnel on icebreakers, in order to make ice prognoses. In connection with the beginning of Finnish-Swedish cooperation, the Environment Protection Board granted money for hydrographic-chemical measurements on icebreakers in 1974. This marked the beginning in the Gulf of Bothnia of reliable winter records of salinity, temperature, oxygen, alkalinity, pH, phosphate, total phosphorus, silicate, nitrite, nitrate, ammonia and total nitrogen. The positive attitude of the Swedish

icebreaker management greatly contributed to the success of this work. During the summer and autumn, expeditions have been carried out by the R/V "Argos" and data covering practically the whole year have thus been obtained from the Bothnian Bay. During the extremely mild winter 1974–75 icebreakers were needed only in the Bothnian Bay and therefore the only data obtained from the Bothnian Sea were from the journey of the icebreakers down to their summer port. The work on the Swedish icebreakers has now ceased, due to lack of funds.

On the Finnish side the R/V "Aranda", which is strengthened for work in ice, has partly been used for winter measurements and sampling has also been done with the aid of helicopters.

The monitoring program which has been carried out in the Gulf of Bothnia since 1979 is part of the larger monitoring program of the Helsinki Convention for the whole Baltic Sea, but it has been enlarged within the frame of the Finnish-Swedish cooperation for the protection of the Gulf of Bothnia. At present the cooperation in the Gulf of Bothnia is so arranged that Finland carries out measurements in January–February and in August–September, and Sweden makes an expedition in May–June, and another in November–December. In this manner all the seasons are covered by four complete expeditions.

#### 4. RESULTS OF THE MONITORING WORK

The results so far obtained have yielded interesting information. In the literature it is generally stated that the vertical convection in the Gulf of Bothnia extends down to the deepest parts during the winter, renewing the bottom water. This statement has been found to be at least partly wrong. In the Bothnian Sea bottom water with a higher salinity seems to persist all the year round, flowing from the Åland Sea through the Southern Quark counterclockwise around the Finngrundsbanks to the Ulvö Deep. This water originates from the surface water of the northern Baltic Proper, which sinks down during the winter and flows into the deep parts of the Åland sea over the sill at Svenska Björn. In addition, small amounts of deep water from the Baltic Proper are brought into the Åland Sea through the 70 m deep channel which goes through the sill on the Finnish side. This is evident from the high salinities in the deepest parts of the

Bothnian Sea. There is also an indication of decreasing oxygen values close to the bottom in the Ulvö Deep.

In the Bothnian Bay also, deep water could be traced throughout the year, at least during the winters 1973/74 and 1974/75. This water originates south of the Northern Quark and flows over the sill to the deepest parts of the Bothnian Bay.

Another remarkable result is that the phosphate values are extremely low all the year round in the Bothnian Bay. We have found that the enhanced phosphate values recorded in the neighbourhood of Rönnskärsverken are caused by the presence of arsenic in the water. Arsenic reacts in the same way as phosphate with molybdenum reagents and is therefore included in the phosphate values. It is probable that a part of the phosphate reported in the Bothnian Bay is actually arsenic. This matter is being investigated.

The nitrate concentrations in the Bothnian Bay are, however, unusually high. It seems that they have increased from  $4 \mu\text{g mol l}^{-1}$  in 1968 to  $8\text{--}9 \mu\text{g mol l}^{-1}$ . A satisfying explanation for this has not been found.

It has been possible to make rough estimates of the water and material transport with the data obtained until now. In this manner we have been able to achieve a relatively good understanding of the water movements in the two deep basins of the Gulf of Bothnia. For mathematical modelling work, we need longer and more extensive monitoring of hydrographic and chemical parameters during the whole year (Ambjörn 1985). This makes the Finnish-Swedish cooperation especially important.

#### 5. THE SECULAR SALINITY TREND IN THE DEEP WATER

Long hydrographic data series thus exist only from the ice-free period and mainly from the summer. The salinity of the Gulf of Bothnia has increased during the present century (Fonselius 1969, Fonselius et al. 1984). During the 1930s the salinity was unusually low, but after World War II it increased to values higher than in the beginning of the century. This has also been noted by other scientists (Launiainen and Koljonen 1984). From permanent Finnish stations unbroken series exist. Jouko Launiainen, University of Helsinki, has studied the surface salinities at those stations.

The annual means of salinity at stations F64,

SR5, F24 and F9 (Fig. 1) during this century are shown in Fig. 2. Most values for the years before 1960 are based on single observations for a year, but treated as annual mean values. At station F64 (Fig. 2a) at 200 m depth the salinity was relatively high in the beginning of the century and decreased until World War I, when the measurements ceased. After the war the salinities were at about the same level, but decreased again during the 1930s. The last value recorded before World War II, when the measurements again ceased, shows a sudden increase. The same increase has also been observed in the Baltic Proper. After the war the salinities were at a considerably higher level. The change was about one salinity unit. The salinity increased slowly, but minor periodic variations occurred. The maximum was reached in 1977 and since then the salinity has decreased.

Fig. 2b shows the salinity changes in the Ulvö Deep in the Bothnian Sea at 175 m depth. Here we find the same trends as in the Åland Sea, but the variations are somewhat displaced, e.g. the maximum in the Ulvö Deep is obtained in 1980, about three years later than in the Åland Sea. It seems that the bottom water probably needs three years to move along the bottom from the Åland basin to the Ulvö Deep.

Fig. 2c shows the long-term variations of salinity in the Bothnian Bay at station F9. Bottom values have been used. Again we find partly the same trends, with minima and maxima. Here, too, the salinity maximum is reached in 1980. It must, however, be pointed out that the deep water in the Bothnian Bay originates from the surface water of the northern Bothnian Sea. The spreading thus goes via the surface water.

## 6. TRENDS OF SALINITY AND NUTRIENTS AFTER WORLD WAR II

A working group appointed by the Helsinki Commission, the Ad hoc Group of Experts on the Assessment of the State of the Marine Pollution of the Baltic Sea (GEA), has studied the long-term variation in the Gulf of Bothnia after World War II. I will here show some of the results for which we have calculated the regression, the correlation and the probability. Fig. 3a and 3b shows the salinity variations at the surface and at 200 m in the Åland Sea at F64. Fig. 3c—3f shows the variations at the surface and at 100 m in the Bothnian Sea at SR 5, and in the Bothnian Bay at

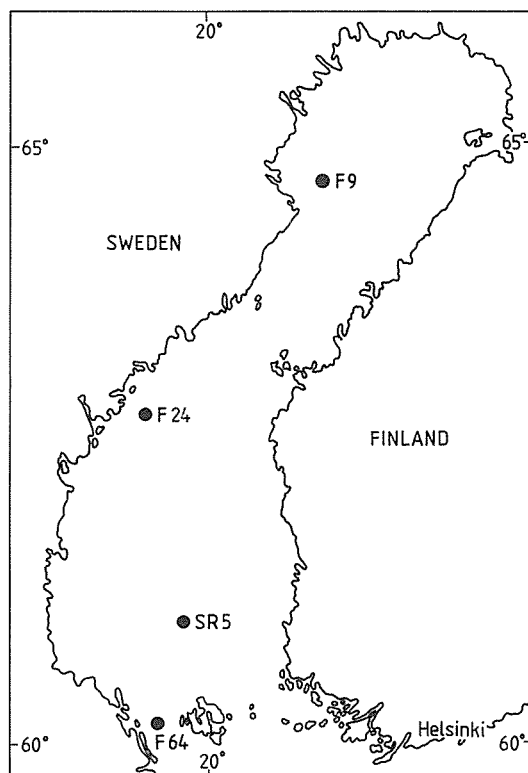


Fig. 1. The Gulf of Bothnia and the sampling stations.

F9. In all these figures salinity shows an increasing trend, with a probability of 99.9 % at F 64 and SR 5, and 95 % at the surface and 99 % at 100 m at F 9. The most interesting result is that after 1977—1980 a decreasing trend can be found. It seems that at least in the deep water the salinity has begun to decrease. Whether this decrease will continue is impossible to predict; it may be a fortuitous fluctuation.

Series of phosphate measurement data are available from the beginning of the 1960s and nitrate and total phosphorus measurement data from the beginning of the 1970s. The difficulty in studying these substances is that they are biologically active, that is they are taken up by developing planktonic organisms and therefore decrease to values close to zero in the surface water during the summer.

From the winter period there are only a few measurements and most of them are from the most recent years. Therefore it is impossible to find reliable trends in the winter series from the Gulf of Bothnia and we have been forced to use all available data, including both summer and winter values. For phosphate it was found that the

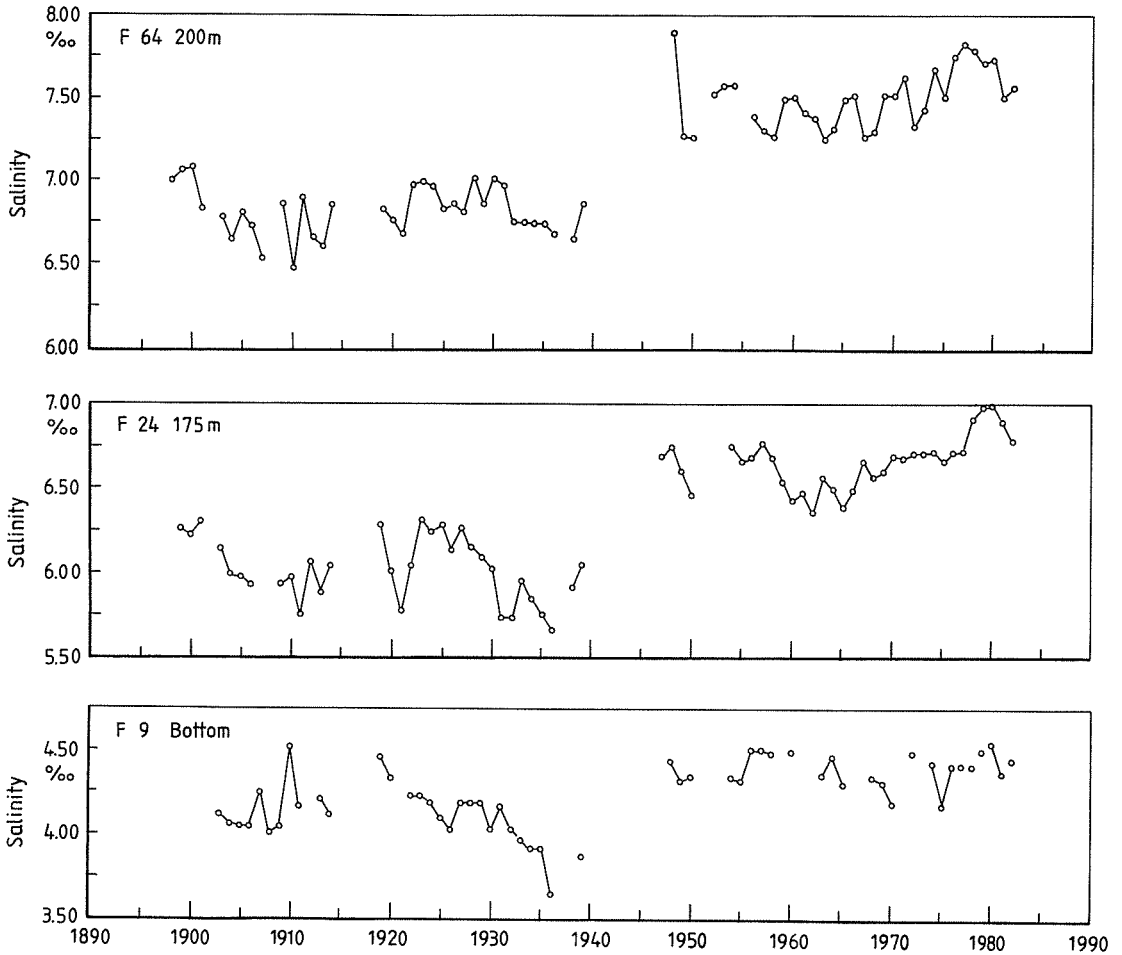


Fig. 2. Annual mean values of bottom salinity in the Åland Sea (station F64) during the period 1898–1982; in the Bothnian Sea (F24) in 1898–1982; and in the Bothnian Bay (F9) in 1902–1982.

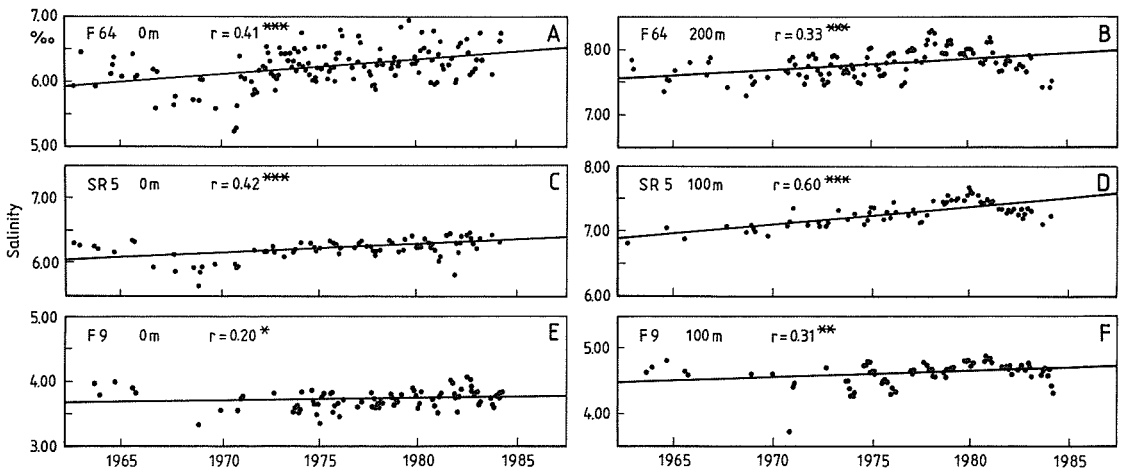


Fig. 3. Salinity of surface (left) and bottom (right) water in the Gulf of Bothnia at the stations F64 in years 1962–1983, SR5 (1962–1984) and F9 (1963–1984). Confidence levels of the correlation coefficient (r): \*, 96 %; \*\*, 99 %; and \*\*\*, 99.9 %.

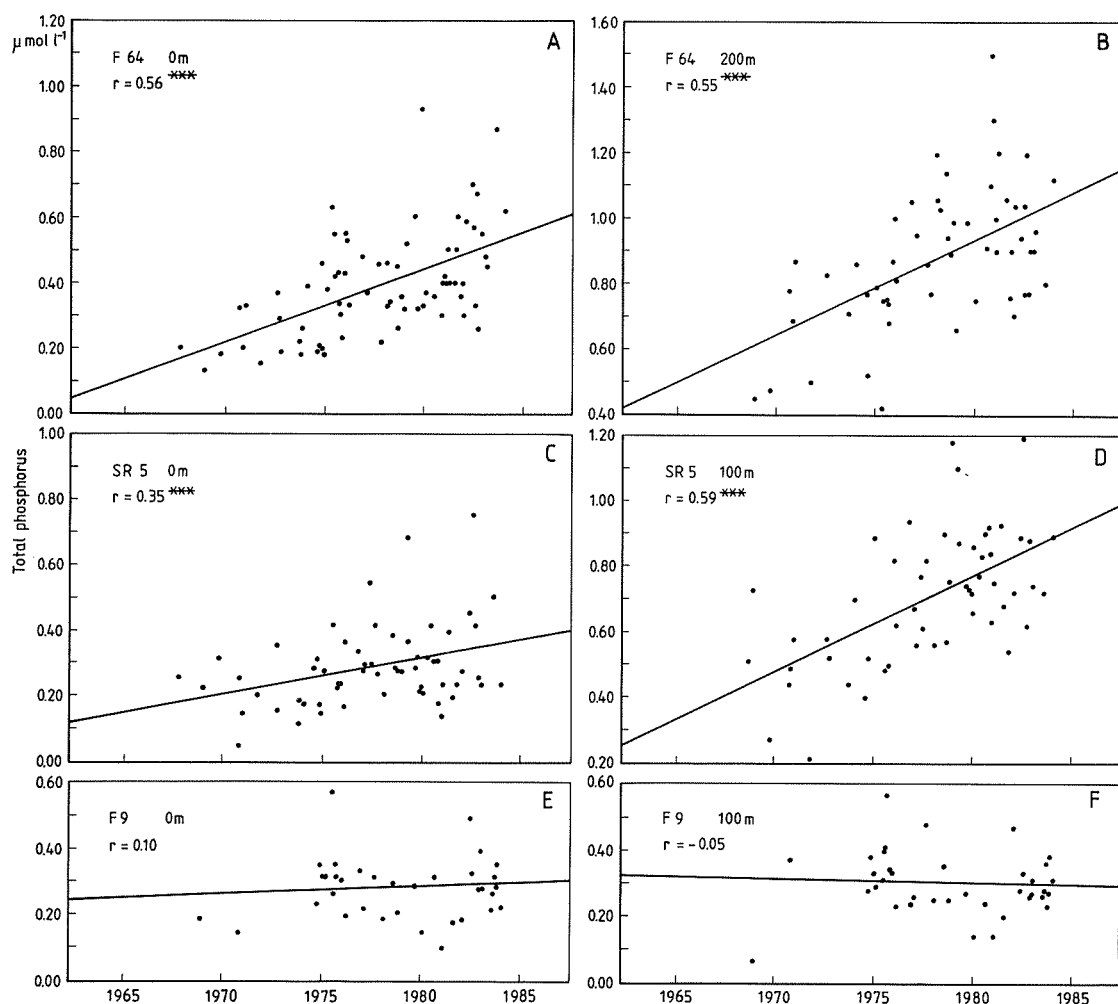


Fig. 4. Total phosphorus variation ( $\mu\text{mol l}^{-1}$ ) in the Gulf of Bothnia (surface and bottom) at the stations F64 (1967—1984), SR5 (1967—1984) and F9 (1969—1984). Confidence levels as in Fig. 3.

concentrations in the Gulf of Bothnia were so low that it was difficult to distinguish significant trends. Total phosphorus and nitrate gave considerably better results and I will limit myself to examples of trends in these two parameters.

Fig. 4a—4g shows the trends for total phosphorus at the same stations and depths as the figures for salinity. We obtain a clear trend with 99.9 % probability for both the surface and deep water in the Åland Sea and the Bothnian Sea. At F9 in the Bothnian Bay we could not find any significant trend. Fig. 5a—5f shows the trends for nitrate. A very strong trend with a probability of 99.9 % can be distinguished in all three basins in

the deep water. A weak trend is evident in the surface water, but it is disturbed by the low summer values.

## 7. NUTRIENT RELATIONS IN THE SURFACE WATER

It has been found that the nutrient salts in the deep water of the oceans are accumulated in constant proportions, due to the fact that the plankton

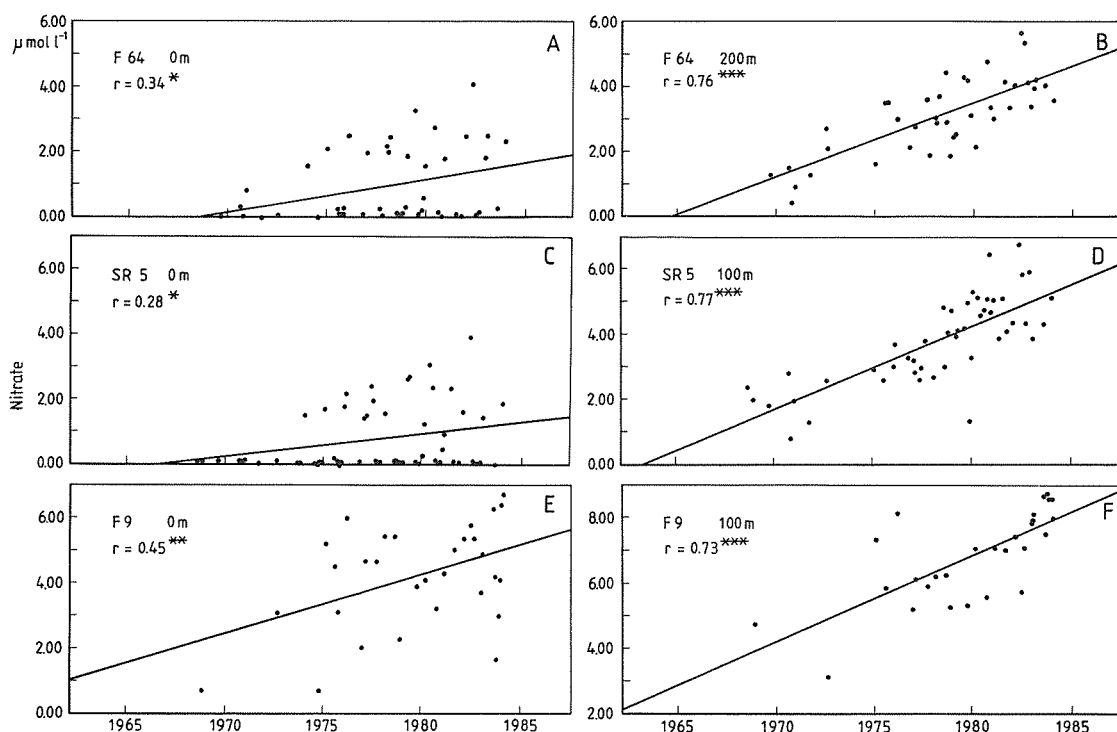


Fig. 5. Nitrate variation ( $\mu\text{mol l}^{-1}$ ) in the Gulf of Bothnia (surface, left and bottom, right) at stations F64, SR5 and F9 in the period 1969–1984. Confidence levels as in Fig. 3.

organisms in the sea have a relatively constant chemical composition. This has been demonstrated by, for example Redfield et al. (1963) and Richards (1965). According to these scientists, the relation between nitrogen and phosphorus in plankton averages 16:1, expressed in atoms or moles. This means that for every phosphorus atom in plankton organisms, there are 16 nitrogen atoms and that growing plankton should utilize these atoms in the ratio 16:1. This is difficult to prove, because phosphorus and nitrogen have different turnover times in the sea. It may, however, be of interest to look at these relations in the Baltic Sea and, especially for us, in the Gulf of Bothnia, in order to see in what concentrations and proportions these two substances are available in the surface water.

In Fig. 6 these substances have been plotted against each other. Phosphorus is expressed as phosphate ( $\mu\text{mol l}^{-1}$ ) and nitrogen in the same unit as the sum of nitrate, nitrite and ammonia. Only values above the thermocline have been used. The theoretical line for the 16:1 ratio has also been drawn. It can be seen that all values for the Bothnian Bay are found in the upper left corner of

the figure. This indicates a large excess of nitrogen (Fonselius 1978, 1981). The values for the Bothnian Sea are found close to or along the line 16:1. This indicates that both phosphorus and nitrogen will be exhausted simultaneously and thus both may be limiting for plankton growth.

## 8. DISCUSSION OF NUTRIENT INPUTS

The annual input of phosphorus and nitrogen to the Gulf of Bothnia has been calculated by a Finnish-Swedish hydrographical group. The Committee for the Gulf of Bothnia has included the river discharge, the discharge from communities, industries and the atmospheric input in precipitation. It was found that the Bothnian Bay annually received 3 400 tons of total phosphorus and 60 000 tons of total nitrogen (Larsson et al. 1985) (Table 1 (3)). The volume of the Bothnian Bay is  $1\,500\text{ km}^3$

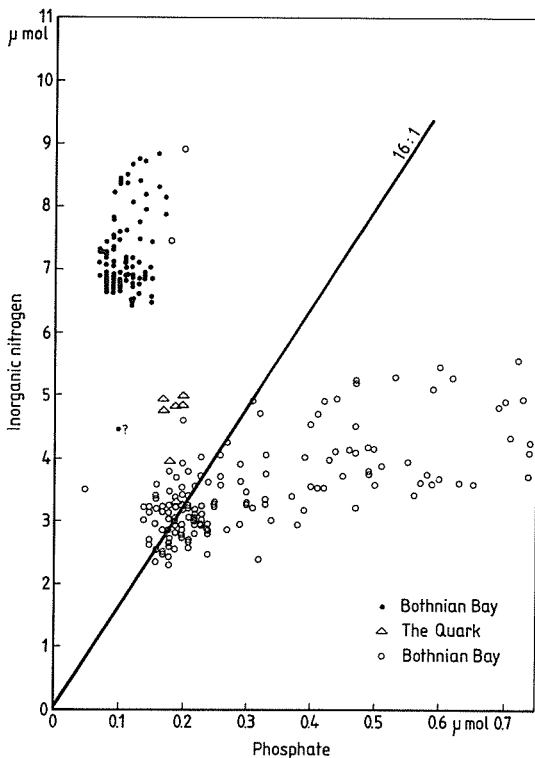


Fig. 6. The relation between total inorganic nitrogen and phosphate in the surface water in different parts of the Gulf of Bothnia in November–December 1975.

(Table 1 (2)). This gives an annual addition of  $0.073 \mu\text{mol l}^{-1}$  phosphorus and  $2.86 \mu\text{mol l}^{-1}$  nitrogen (Table 1 (4)). The N:P ratio in atoms in the surface water is then 35:1, which is about double the ratio in plankton. If we assume that all phosphorus is changed into inorganic form and is used up in primary production by plankton,  $16 \times 0.073 \mu\text{mol}$  nitrogen will be used in this production (Table 1 (5)). The difference between input and the amount used will then be  $1.7 \mu\text{mol l}^{-1}$  (Table 1 (6)). If the water in the Bothnian Bay has a residence time of 4 years (Dahlin 1976b) (Table 1 (7)), we obtain approximate accumulation of  $4 \cdot 1.7 = 6.8 \mu\text{mol l}^{-1}$  of inorganic nitrogen in the water (Table 1 (8)). The observed value for the inorganic nitrogen concentration in the water is between 5 and  $7 \mu\text{mol l}^{-1}$  (Table 1 (9)). If we carry out a direct calculation of the accumulation (4 times the input value), we obtain  $0.29 \mu\text{mol l}^{-1}$  total phosphorus and  $11.4 \mu\text{mol l}^{-1}$  total nitrogen (Table 1 (10)). Table 1 also shows similar calculations for the Bothnian Sea. The volume of the Bothnian Sea is  $4\,700 \text{ km}^3$  and the annual inputs of nitrogen and phosphorus have been estimated at 85 000 tons of total

nitrogen and 38 000 tons of total phosphorus (Larsson et al. 1985). This gives an annual addition of  $0.026 \mu\text{mol l}^{-1}$  of total phosphorus and  $1.29 \mu\text{mol l}^{-1}$  of total nitrogen. The residence time of the water in the Bothnian Sea has been estimated by Dahlin (1976b) at around 3 years. The N:P ratio in the surface water is 17.7:1, thus fairly close to the theoretical value 16:1. If we multiply the phosphorus value by 16, we obtain the amount of nitrogen used by phytoplankton during the annual primary production =  $0.42 \mu\text{mol l}^{-1}$ . The difference between added and utilized nitrogen is  $0.77 \mu\text{mol l}^{-1}$  and by multiplying this value by 3 we obtain the amount accumulated in the water,

Table 1. Calculated and observed concentrations of nutrients in Gulf of Bothnia (see text).

| (1)  |                                                                           | Bothnian Sea        | Bothnian Bay    |
|------|---------------------------------------------------------------------------|---------------------|-----------------|
| (2)  | Volume, $\text{km}^3$                                                     | 4 700               | 1 500           |
| (3)  | Discharge, $\text{t a}^{-1}$                                              | N 85 000<br>P 3 800 | 60 000<br>3 400 |
| (4)  | Input, $\mu\text{mol l}^{-1}$                                             | N 1.29<br>P 0.026   | 2.86<br>0.073   |
| (5)  | Calculation using $16 \times \text{P}$ , $\mu\text{mol l}^{-1}$           | N 0.42              | 1.17            |
| (6)  | Remaining conc. in water, $\mu\text{mol l}^{-1}$                          | N 0.77              | 1.69            |
| (7)  | Residence time, years                                                     | 3                   | 4               |
| (8)  | Calc. accumulation (6) $\times$ (7), $\mu\text{mol l}^{-1}$               | N 2.31              | 6.76            |
| (9)  | Obs. concentration in water, $\mu\text{mol l}^{-1}$                       | N 2.10              | 5.80            |
| (10) | Direct calculation Tot-N and tot-P, $\mu\text{mol l}^{-1}$                | N 3.9<br>P 0.08     | 11.4<br>0.29    |
| (11) | Conc. in the water outside the sill, $\mu\text{mol l}^{-1}$               | N 15<br>P 0.60      | 16<br>0.28      |
| (12) | Inflow over the sill $\text{km}^2 \text{ a}^{-1}$                         | 1 380               | 246             |
| (13) | Added amount (11) $\times$ (12) : (2), $\mu\text{mol l}^{-1}$             | N 4.40<br>P 0.176   | 2.62<br>0.046   |
| (14) | Sedimentation, $\mu\text{mol l}^{-1}$                                     | N 0.11<br>P 0.03    | 0.33<br>0.10    |
| (15) | Remaining amount [(4) + (13) - (14)] $\times$ (7), $\mu\text{mol l}^{-1}$ | N 16.7<br>P 0.52    | 20.6<br>0.08    |
| (16) | Observed conc. in the water, $\mu\text{mol l}^{-1}$                       | N 16<br>P 0.28      | 19<br>0.30      |

which will be  $2.31 \mu\text{mol l}^{-1}$ . The observed concentration in the surface water is  $2.1 \mu\text{mol l}^{-1}$ . Direct accumulation calculations give  $3.9 \mu\text{mol l}^{-1}$  for total nitrogen and  $0.08 \mu\text{mol l}^{-1}$  for total phosphorus.

We have to take into account, however, that nutrients are added to the Bothnian Sea from the Baltic Proper and to the Bothnian Bay from the Bothnian Sea. If we know the annual water transports, we can calculate this addition of nutrients. With the help of the existing calculations (Dahlin 1976a) (Table 1 (12)), I obtain the following input of nutrients: the Bothnian Bay  $2.62 \mu\text{mol l}^{-1}$  of N and  $0.046 \mu\text{mol l}^{-1}$  of P; the Bothnian Sea  $4.40 \mu\text{mol l}^{-1}$  of N and  $0.176 \mu\text{mol l}^{-1}$  of P (Table 1 (13)) (the concentrations in Table 1 (11) are from the hydrographical data, National Board of Fisheries). But we also have to consider the sedimentation, which will decrease the values. I have here used the sedimentation estimates of Dahlin (1981) from the Second Seminar on the Gulf of Bothnia, (Luleå, Sweden, June 16–17, 1981). I have estimated roughly that 50 % settles in the Bothnian Bay and 50 % in the Bothnian Sea (Niemistö et al. 1981). When all the input values are added together (Table 1 (4) + (13)) and the sedimentation values are subtracted (14), we obtain the total annual input. The total concentration in the water can be obtained by multiplying the annual input by the residence time (7). When this value (15) is compared with the observed value (16), we can see that we obtain surprisingly good agreement for total nitrogen in both basins, while the total phosphorus shows poor agreement. The reason for this may be that the low phosphorus concentrations in the Gulf of Bothnia reduce the analytical accuracy. We also have to remember that the estimates used here are rough and that many factors are poorly known. With better data it would certainly be possible to obtain more satisfactory results.

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# THE WATER EXCHANGE BETWEEN THE GULF OF BOTHNIA AND THE BALTIC PROPER AS SIMULATED BY THE METEOROLOGICAL CONDITIONS IN A STATISTICAL MODEL

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AMBJÖRN, C. 1986. The water exchange between the Gulf of Bothnia and the Baltic proper as simulated by the meteorological conditions in a statistical model. Publications of the Water Research Institute. National Board of Waters, Finland. No. 68.

Current measurements were performed in the sill area of the northern Åland Sea during approximately six months in summer/autumn/winter 1974/75 and summer 1977. A pilot investigation was made in order to study the statistical relationships between water transport and winds, air pressure and water level. The current data were transformed into empirical orthogonal functions. The linear correlation coefficients between the values predicted by the model and the transport in the upper and lower layers were both 0.60, which is not entirely satisfactory. If the transport during each month is considered instead, the correlation between the measured and predicted values is better. The relationships with meteorological factors can be used to calculate the water exchange in the Gulf of Bothnia during other periods. The next step will probably be to try numerical models in these simulations.

Index words: The Gulf of Bothnia, water transport, statistical model

## 1. INTRODUCTION

Water and nutrient transports between the Gulf of Bothnia and the Baltic Proper have been calculated (Ambjörn and Gidhagen 1979). The calculations are based upon current measurements in the Åland Sea, Fig. 1, and Archipelago Sea, and on the volume variations in the Gulf of Bothnia and the fresh water supply.

Statistical relationships were studied between meteorological factors and the water transport. These could make it possible to calculate the water transport during periods without current measurements, which would increase the knowledge of the water balance in the Gulf of Bothnia.

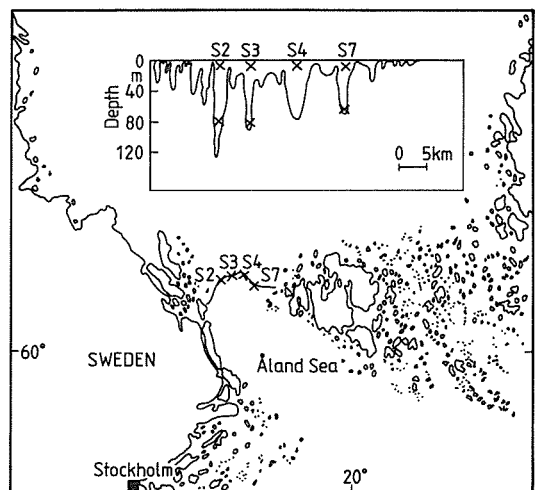


Fig. 1. Location of current meters.

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## 2. THEORETICAL CONSIDERATIONS

The currents in the Åland Sea normally have a southward direction in the surface layer and a northward direction below the halocline. A scaling estimation is made to find the importance of the different terms, in the Navier Stokes' equations. This estimation shows that typical values for: the inclination of the water surface, the air pressure difference and the wind stress give about 70 per cent of the acceleration terms for the upper layer. The horizontal density gradient is important, but daily measurements are not performed. Thus the factors used in the correlation calculations are the following: local wind components: a) north-south and b) west-east, c) water level differences and d) air pressure differences across the Quark. As the wind velocities in the central part of the basins on each side did not give any information about the horizontal density differences, they were omitted.

## 3. MODEL CALCULATIONS

The current information from every 10th minute is transformed into empirical orthogonal functions (Holmström and Stokes 1978). This gives seven new time series with decreasing energy and the same information as in the original series. The first five of these new series cover 98 per cent of the measurements, and therefore the other series can be omitted. Model selection by cross validation is applied to the data (Hjort and Holmström 1981). This gives the optimum variance-cover by the different meteorological series.

The first current mode series has a 0.63 correlation coefficient with the meteorological series. These two curves are plotted for every 6th hour in Fig. 2. Now the current information is transformed back to the original form. A comparison between the currents at each measured point and the simulated current is shown in Table 1. This shows roughly the size of the deviations at each point, which of course is also connected with the respective velocities. The correlation coefficient between the measured transport and simulated values in both the upper and the lower layer is 0.60. The monthly transports in the upper and lower layers give better correspondence between the measured and simulated values. The study period is 2.5 months, which is short for comparing results on a monthly basis.

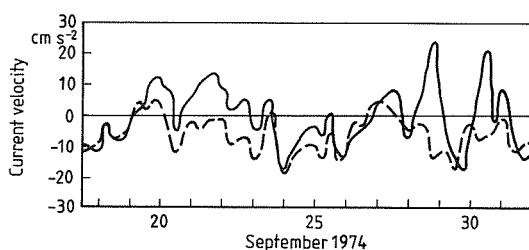


Fig. 2. Current velocity in the first mode of the empirical orthogonal functions. Summation of water level and air pressure differences in a north-south direction across the sill, broken line.

Table 1. The differences between the simulated and the measured current velocities. The figures are given as percentages of the total amount of observations at each station. The values are 6-hour means.

| Stations | $< 10 \text{ m s}^{-1}$ | $10\text{--}30 \text{ m s}^{-1}$ | $> 30 \text{ m s}^{-1}$ |
|----------|-------------------------|----------------------------------|-------------------------|
| S2—8 m   | 37 %                    | 51 %                             | 12 %                    |
| S2—80 m  | 99 %                    | 1 %                              | 0 %                     |
| S3—8 m   | 46 %                    | 41 %                             | 13 %                    |
| S3—80 m  | 57 %                    | 39 %                             | 4 %                     |
| S4—8 m   | 59 %                    | 38 %                             | 3 %                     |
| S7—8 m   | 100 %                   | 0 %                              | 0 %                     |
| S7—70 m  | 62 %                    | 35 %                             | 3 %                     |

This pilot study was performed in order to find out whether it is fruitful to continue this statistical approach or if other ways should be used, e.g. numerical models. The results show that the method is not useful for short time scales, but can be applied to monthly values. But even here it is not so good as could have been wished. It would probably be best to make simulations by other methods.

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## THE USE OF SATELLITE PICTURES FOR MAPPING THE ICE CONDITIONS AND SURFACE TEMPERATURES IN THE GULF OF BOTHNIA

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GRÖNVALL H. & KALLIOSAARI S. 1986. The use of satellite pictures for mapping the ice conditions and surface temperatures in the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The Institute of Marine Research has received NOAA-6, NOAA-7 and NOAA-8 satellite pictures since March 1981. So far, the pictures have mainly been used in the routine ice service. Daytime pictures have been analysed using the wavelength channels near the frequency of visible light and night-time pictures in the infra-red wavelength channel only. Although clouds sometimes hinder ice observations, the results from winters 1981–82 and 1982–83 have been very good. The infra-red channel is also being used for mapping the relative distribution of the surface temperature in the Baltic Sea.

Index words: Satellite, ice, temperature

### 1. TIROS/NOAA RECEIVING STATION

Since 1981, the Institute of Marine Research has received NOAA-6 and NOAA-7 pictures with a satellite picture receiver constructed by the Technical Research Centre of Finland. The satellite pictures have been used for mapping the ice conditions and the surface temperature of the water in the Baltic Sea.

The construction of the TIROS/NOAA HRPT satellite picture receiver is described by Santamäki (1981). It consists of a parabolic antenna (ø 3 m) (Fig. 1), a mini-computer for data input and calculation of the output (Commodore 4016), a receiver and the data-processing unit, a data recording with a tape-recorder (Kennedy 9100), a processing of the picture (Muirhead K 560) (Fig. 2).

The tracking of the satellite is done by "the program tracking" system, which calculates the azimuth and elevation angle of the satellite at 10-

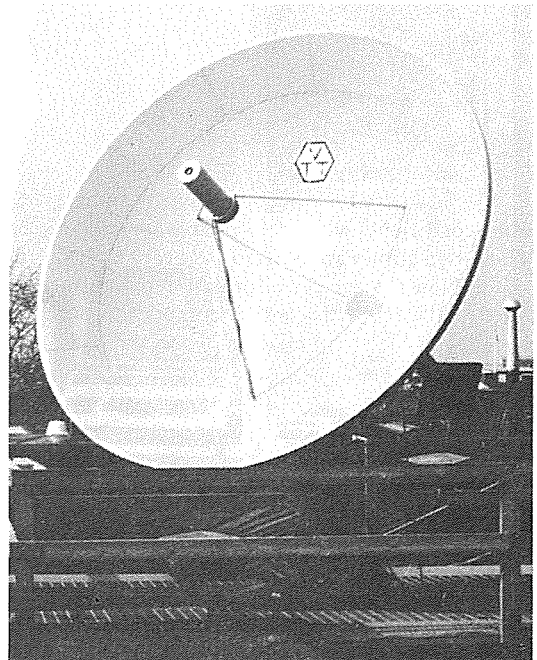


Fig. 1. Satellite antenna.

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second intervals. For this purpose the following orbit data input parameters are needed:

- inclination angle
- equator crossing longitude
- satellite altitude
- the flight time around the earth

This information is received daily from the USA by the telex route of WMO. The altitude of the satellite is about 850 km and the flight time around the earth is 1 hour 40 minutes. The route of the satellite moves 25 degrees westwards between two successive passes.

The satellites transmit information in 5 different wavelength channels:

- channel 1, 0.55—0.68  $\mu\text{m}$
- channel 2, 0.725—1.10  $\mu\text{m}$
- channel 3, 3.55—3.93  $\mu\text{m}$
- channel 4, 10.5—11.5  $\mu\text{m}$
- channel 5, 11.5—12.5  $\mu\text{m}$

The data are transmitted in digital form simultaneously in all the channels and are collected for further processing in the tape-recorder of the receiving station. Simultaneously with the data, it

is possible to supply a picture from one channel. The resolution of the pictures is 1.1 km. The picture is processed in different shades of grey, max 32 shades of grey/picture. The shades of grey are obtained by adjusting the white and black intensities. It is possible to enlarge the pictures to eight times their original size. Geometric corrections are not performed for the pictures.

## 2. USE OF THE PICTURES IN ICE SERVICE

Satellite pictures are used daily in the Finnish ice service. Because of clouds, it is very seldom possible to see the ice conditions in the whole Baltic Sea. At the beginning of the winter, when it is dark, and at night, the pictures are analysed in the infra-red channel.

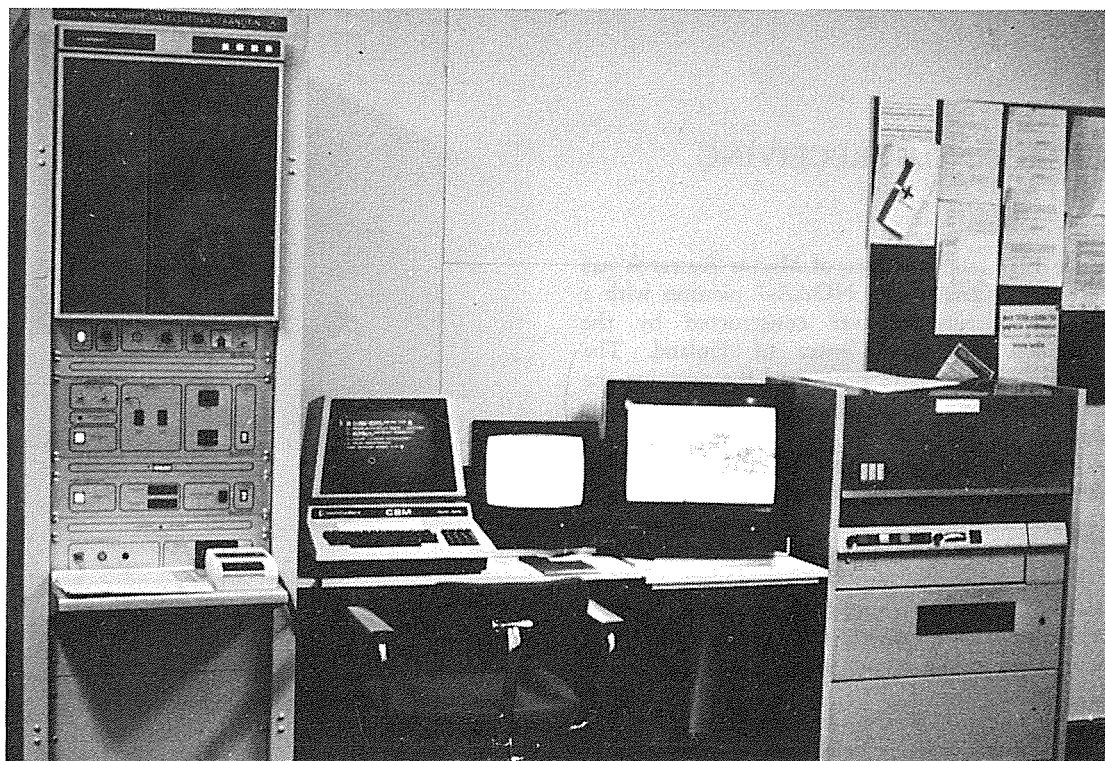


Fig. 2. Satellite receiving station.

In midwinter and spring, the daytime pictures are analysed in channel 2, which is near the frequency of visible light. In the pictures from 12—15 April 1983 (Figs. 3 and 4) the ice conditions in the Bay of Bothnia can be clearly seen. Along the Finnish coast, seaward of the fast ice, a wide lead runs from the northwestern parts of the Bothnian Bay to the latitude of Kokkola. The ice in the middle of the sea is slowly drifting westward. It is easy to follow changes in the ice situation from day to day.

temperature curves on the basis of infra-red satellite pictures supplemented with other temperature records. When the temperature difference in some sea areas is great, the best accuracy is obtained by using different grey scales in analysing the pictures. The picture in Fig. 5 gives the surface temperatures on 9.5.1983. The dark areas indicate warm and the light areas cold water. The clouds cause difficulties only in the areas south of Gotland. The water is warmest in the vicinity of the coast. In the middle areas of the sea the temperature is 4—7°C.

### 3. USE OF THE PICTURES IN MAPPING THE SURFACE TEMPERATURE

The infra-red channel is also used to map the relative distribution of the surface temperature. In some cases it has been possible to draw accurate

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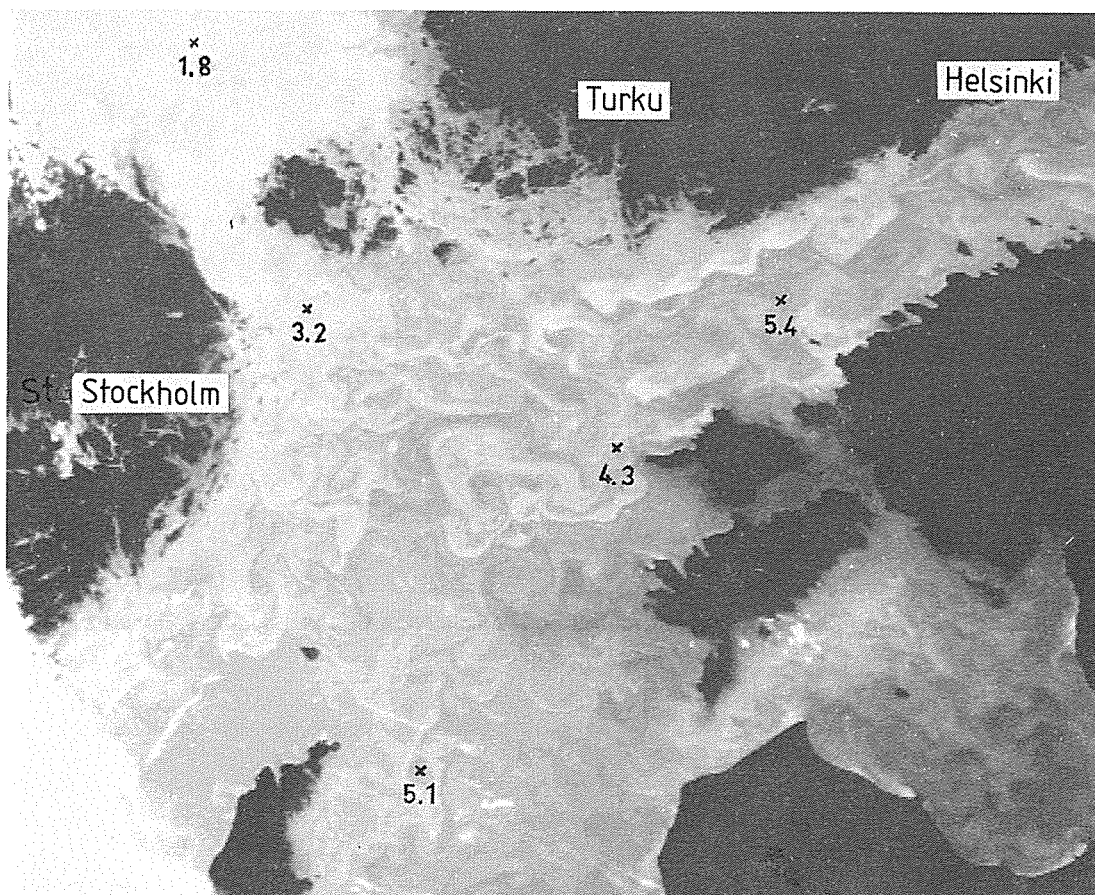


Fig. 5. The satellite picture from the surface temperature of the water in the Baltic Sea. NOAA-6 9.5.1983 klo 9.29, channel 4.

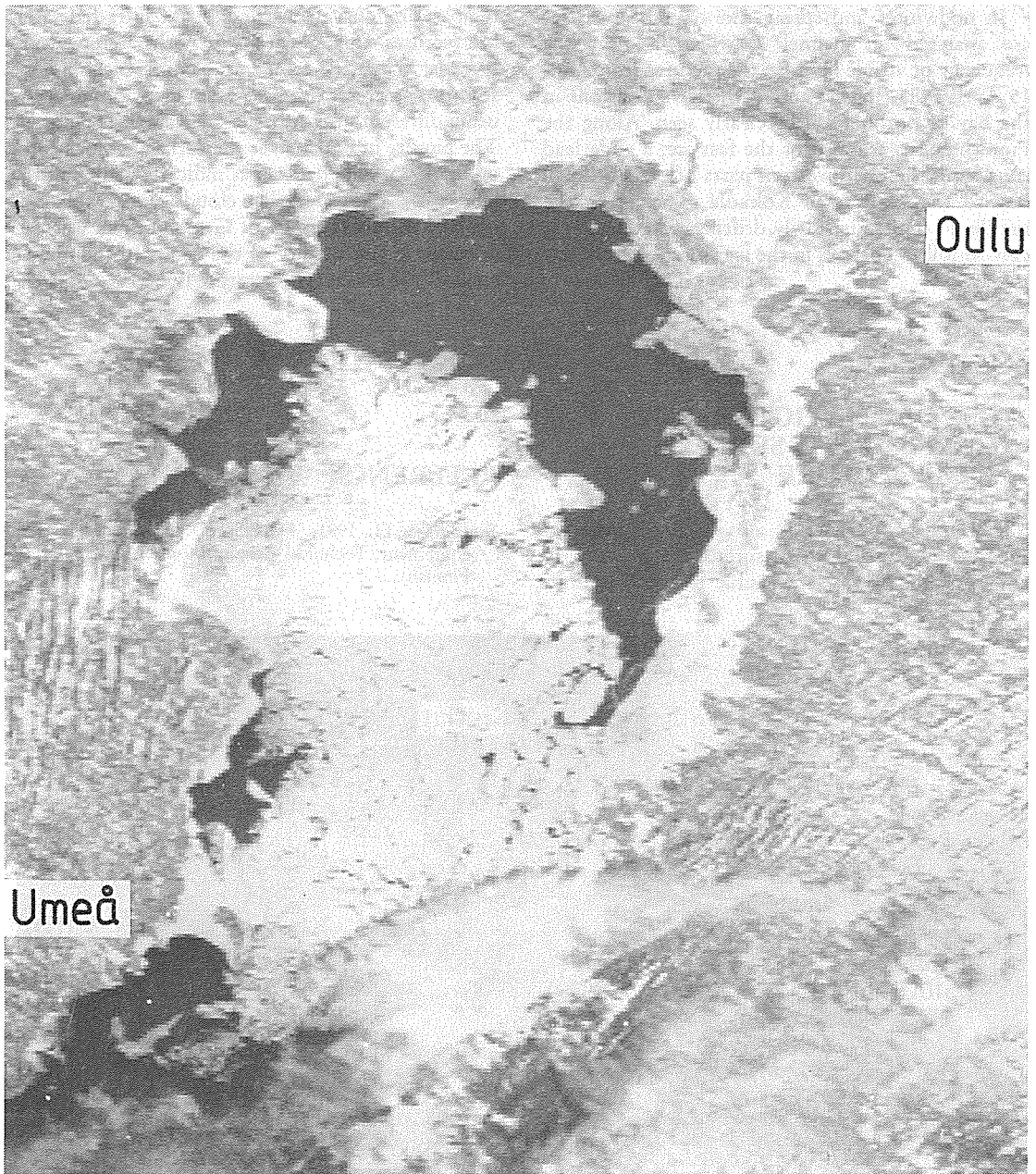


Fig. 3. The satellite picture from the ice conditions in the Bay of Bothnia. NOAA-6 12.4.83 klo 10.09, channel 2.

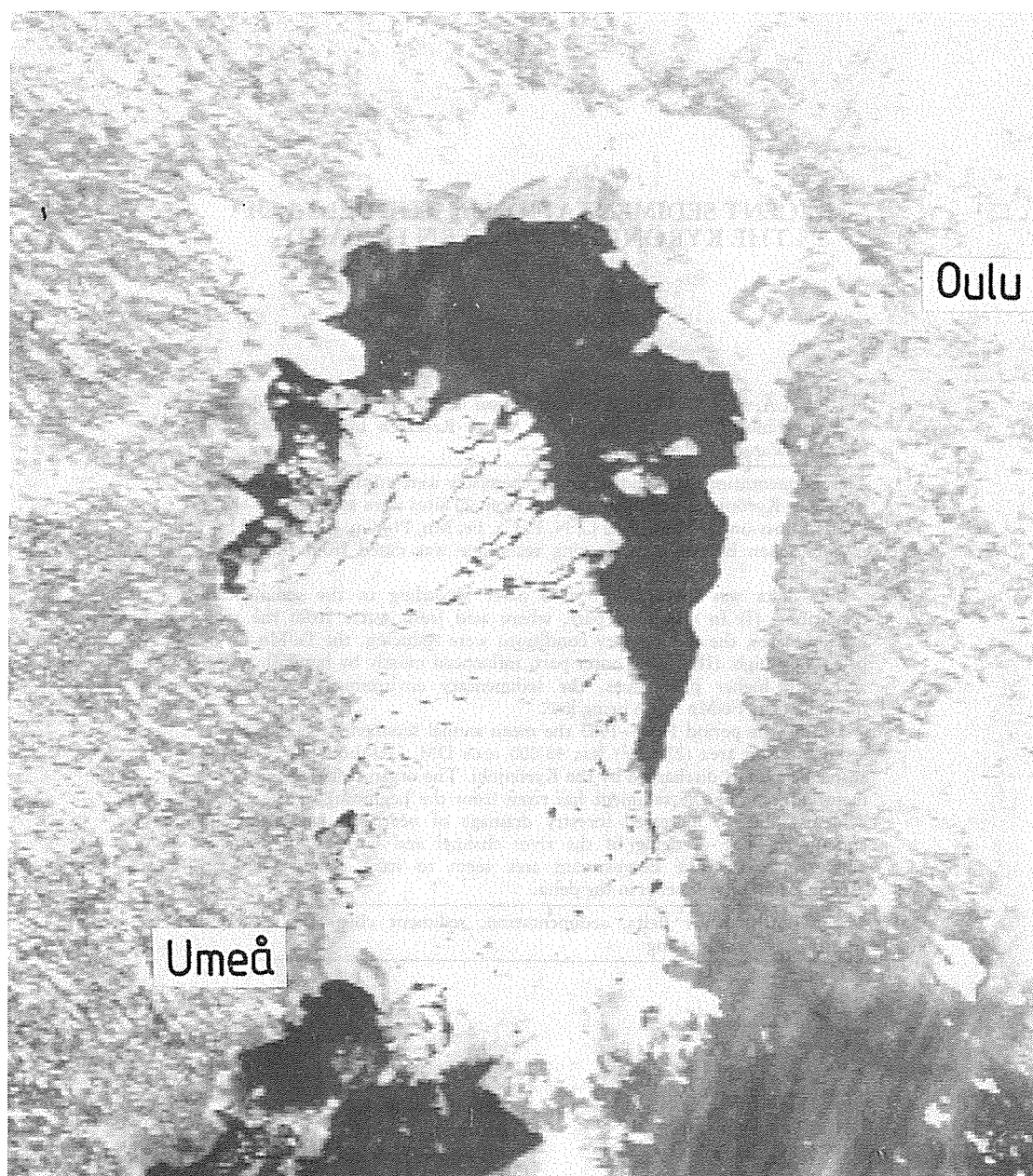


Fig. 4. The satellite picture from the ice conditions in the Bay of Bothnia. NOAA-6 15.4.83 klo 10.38, channel 2.

## RECENT SEDIMENTATION IN THE DELTA OF THE KYRÖNJOKI, WESTERN FINLAND

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HEIKKILÄ, R. 1986. Recent sedimentation in the delta of the Kyrönjoki, Western Finland. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The sedimentation rates and sediment chemistry were studied in the delta of the River Kyrönjoki. Sediment samples from 20 sites were analysed for water content, loss-on-ignition, total C, N, P, Ca, Fe, Mn, Pb, Cu, Zn, Cd and Hg. A core taken by the core freezing technique was dated from the annual lamination.

The delta was divided into two parts according to the sedimentary conditions. (I) In the inner part, where acid fresh water from the river predominates, the sedimentary conditions were reducing, the Fe/Mn ratios being very high. (II) In the outer part, influenced mostly by brackish water, and with higher pH values, the sedimentary environment was mainly oxidizing, the Fe/Mn ratios being low.

During the period 1958—1983 the mean annual sedimentation rate in the sedimentation area (27 km<sup>2</sup>) was 48 000 tons DM, i.e. about 70 % of the suspended solids discharged by the Kyrönjoki. The organic matter and heavy metal content of the sediment has risen from the beginning of the 1960s, evidently due to increased forestry drainage of peatlands, and increased agriculture. The dredging of the river channel and building of artificial embankments in the downstream area seem to have caused increased accumulation of sediments in the delta.

Index words: River delta, sedimentation, sediment chemistry, peatland forestry, river engineering

### 1. INTRODUCTION

The hydrology and sediment load of rivers in Western Finland have changed greatly in recent decades, due to human activities. In the drainage basin of the River Kyrönjoki, the factors having the greatest influence on sedimentation have been agriculture, forestry drainage of peatlands, peat harvesting and watercourse works. The aim of this study was to assess the influence of watercourse engineering on the delta of the River Kyrönjoki.

The sediments and sedimentation of river deltas have not been studied very extensively in Finland in recent years. Rosberg (1895) studied the morphology of the deltas of the rivers flowing into the Gulf of Bothnia, and he described the delta of the Kyrönjoki as modified, with wide alluvial plains. Today the delta is almost entirely submarine, mostly due to artificial embankments in the downstream area and estuary. The delta of the Kyrönjoki, especially its zoobenthos, has recently been studied by Meriläinen (1984, 1986).

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## 2. STUDY AREA

The Kyrönjoki delta is situated at 63°10'N 22°00'E, some 30 km NE of Vaasa, a city on the west coast of Finland. The drainage area of the Kyrönjoki is 4920 km<sup>2</sup>, of which 47 % is forests, 26 % peatlands, 24 % arable land, 2 % built-up areas and 1 % lakes. In the river valleys, the soils are chiefly clay and silt. In the watershed region till and peat cover wide areas. Sand and gravel exist only in limited areas near the watershed. Most of the fields and built-up areas lie along the river. The land upheaval in the delta is 8 mm a<sup>-1</sup> (Kääriäinen 1975).

The discharge of the Kyrönjoki fluctuates over a wide range (NQ : MQ : HQ = 1 : 43 : 528 m<sup>3</sup>s<sup>-1</sup>). Spring, summer and autumn floods are frequent. The river channel has been cleared many times since the 1600s to prevent floods and to allow agriculture on the adjoining land. Since 1963 4 reservoirs and 4 power stations (with daily regulation) have been built.

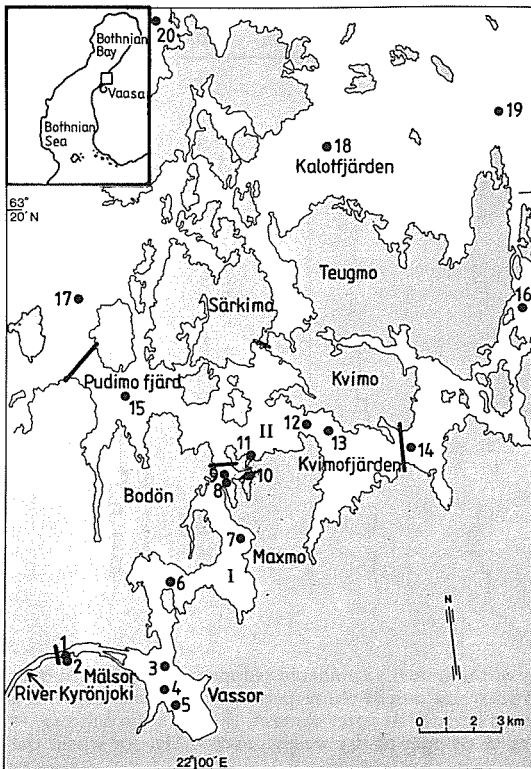


Fig. 1. Map of the study area and sampling sites. The delta is delimited with thick lines. I = the inner part of the delta, II = the outer part of the delta (see text). The sediment core is taken at st. 10 in Nabbviken bay.

The Kyrönjoki water has a high humus content due to drainage from extensive peatland areas. Furthermore the water is often very acid due to the sulphide-bearing Litorina clays along its course (Alasaarela 1982).

## 3. MATERIAL AND METHODS

Sediment samples were taken from 20 sites in the delta (Fig. 1). Some echo-sounding was carried out to locate the areas of accumulation and erosion. The samples were taken from accumulation areas, mainly with a core freezer (Renberg 1981), but also with a piston corer and a Kajak sampler.

The samples were divided into subsamples of 0.6–3.5 cm, according to the visible layers. The subsamples were dried to a constant weight at +50°C and then ignited at +550°C for 3 hours. The organic content was determined as loss-on-ignition (IG). The ignited samples were extracted using 37 % HCl. Total Fe, Mn, Pb, Cu, Zn, Cd, Hg and Ca were analysed by AAS. Total P was determined spectrophotometrically by the molybdenum blue method. C and N were determined with an HP 185B CHN analyser.

The Nabbviken sediment (Fig. 1, site 10) was dated using the lamination, which was found to be annual on the basis of the seasonal succession of diatoms (for method see Simola 1977).

## 4. RESULTS

### 4.1. Sedimentation rates

The Nabbviken sediment is composed of regularly alternating light and black layers, which form varves, 0.6–1.8 cm in thickness. The deeper layers are thinner than the uppermost ones, mostly due to sediment compaction. The accumulation rates fluctuated over the period 1958–1983, from 1140 to 3000 g DM m<sup>-2</sup> a<sup>-1</sup> (Fig. 2).

If the average accumulation rate at Nabbviken is used for the total accumulation area (27 km<sup>2</sup>), the mean annual sedimentation rate in the delta of the Kyrönjoki was 48 000 tons DM during the period 1958–1983. This is about 70 % of the suspended solids discharged by the Kyrönjoki (see Meriläinen 1986).

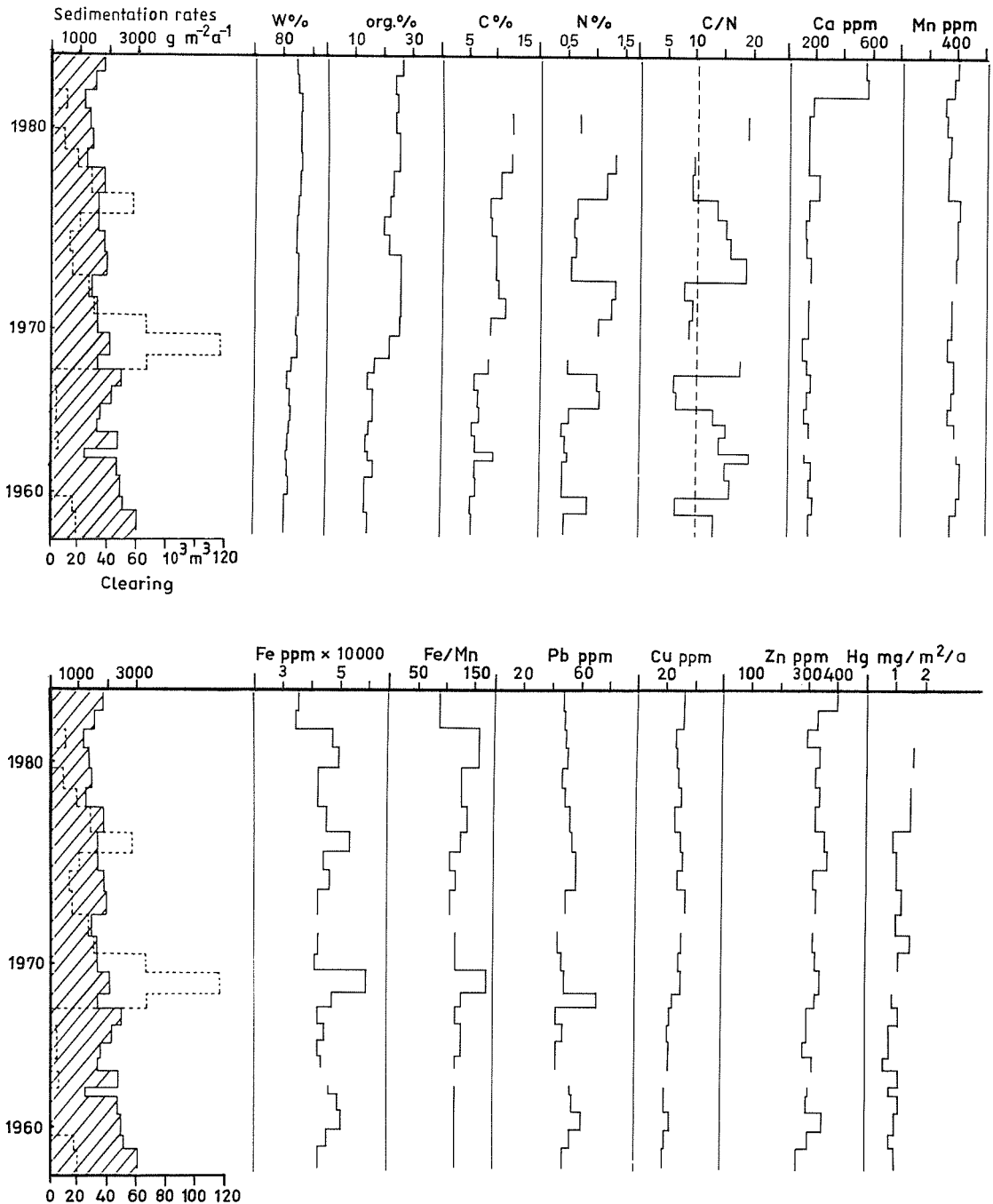


Fig. 2. Sedimentation rates (diagonal hatching) and stratigraphy of the annually laminated sediment in Nabbviken over the period 1958–1983. Clearing of the river channel in the downstream area in the respective years is marked with a dashed line (to the left in the both diagrams). The core was 33 cm long.  $W$  = water content of fresh sediment, org. = organic content per unit dry weight, element contents given as % or ppm of dry weight, except Hg, for which the annual precipitation is given.

## 4.2 Sediment chemistry

The organic content of the sediment clearly increased after 1958 (Fig. 2). The increase is evidently due to agriculture, and to drainage of peatlands for forestry. The sediment samples studied microscopically contained abundant remains of *Sphagnum* spp.

The carbon and nitrogen content also show a recent increase. All the carbon in the sediment is organic, because there are only small amounts of carbonates in the bedrock of the Kyrönjoki basin. Moreover, due to the acidity at Nabbviken carbonates are soluble and are not precipitated. Carbon had a strong correlation with IG ( $r = 0.93^{***}$ ). The ratio of IG to total carbon was 2.3 : 1 (cf. Håkanson and Jansson 1983). The correlation between IG and nitrogen was not very clear ( $r = 0.51^{***}$ ). This was due to the large amount of allocthonous material with a high C/N ratio, mostly over 10. According to the C/N ratio, the sediment is mostly allocthonous (Hansen 1961, Fig. 2).

The Fe content of the sediment is high, being highest in the outer part (II) of the delta, where brackish water predominates. In the inner part (I), with acid freshwater from the river, and in the sea area, it is somewhat lower (for the subdivision of the delta see Fig. 1). The Mn content is low in the inner part of the delta, but in the outer part and in the sea area it is considerably higher.

The Fe/Mn ratio is an indicator of the redox conditions of the environment of sedimentation (e.g. Simola 1983). The delta was divided into two parts according to the Fe/Mn ratio in the sediment surface (0–2 cm) (Fig. 1). In the inner part (I), the ratio is over 50, and the sedimentary conditions are accordingly reducing. In the outer part (II), the ratio is below 20, and the sedimentary conditions are mostly oxidizing.

The content of some heavy metals (Cu, Zn, Hg) in the sediment has risen slightly during recent decades (Fig. 2). This might be due to the increased amounts of organic matter in the sediment (Förstner 1976). The heavy metal content was clearly lower than in areas polluted by industrial works (cf. Förstner 1976, 1982, Cato 1977). The heavy metal content in the surface sediments is 2–10 times as high as that determined from postglacial clays lying below the fluvial sediments.

The Ca content of the sediment is lower in the inner than in the outer part of the delta. This is due to the preponderance of acid river water in the inner part; Ca is soluble in the acid water and is not precipitated until it reaches the neutral

brackish water. The Ca content in the sediment surface does not differ from that of the postglacial clays below the fluvial sediments.

## 5. DISCUSSION

The increase of the organic matter and heavy metal content in the sediment over recent decades is evidently due to the increased intensity of agriculture, forestry and peat harvesting in the drainage area. Drainage of peatlands in particular has increased the erosion and the organic sediment load of the river.

Waste-water from towns and villages has caused an increase in the amount of phosphorus during recent decades, but, due to the movement of phosphorus in the interstitial water of the sediment (Callender 1982), the influence of waste-water is not clearly visible in the sediment stratigraphy. After 1979, the phosphorus content of the water in the Kyrönjoki decreased, but the decrease is not yet evident in the sediment strata.

An increase in erosion is caused during water-course engineering, when the work is carried out in the river channel. The daily regulation of the power stations causes continuous erosion, especially downstream from the stations. On the other hand, the reservoirs collect most of the coarse sediment running off their drainage areas.

The clearing of the river channel and building of artificial embankments in the downstream area of the river (in the delta, and 10 km upstream) seem to have had an effect on the sedimentation rates in the delta (Fig. 2). It should be pointed out that the discharge was fairly low in the spring and low in the summer and autumn in 1958, 1960, 1969, 1973, 1976 and 1978, but high in the spring, summer and autumn in 1961, 1966, 1967, 1974 and 1977.

## ACKNOWLEDGEMENTS

I am very grateful to the following persons: Assoc.Prof. Matti Seppälä for his invaluable advice and encouragement, Pertti Sevola from the Vaasa District of the National Board of Waters for his useful advice and practical help, Assoc.Prof.

Kimmo Tolonen for advice, especially on sediment chemistry, and for determining the diatoms, Prof. Toive Aartolahti and Assoc.Prof. Rauno Ruuhijärvi for providing excellent working facilities, Hanna Heikkilä, Sinikka Jaurila and Esa Tulisalo for most of the laboratory analyses, Kirsti Lehto for technical assistance, Anna A. Damström and Tony Meadows for language revision, and many others. The National Board of Waters initiated the project and gave financial support.

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# THE CONCENTRATION OF IRON IN THE WATERS OF THE GULF OF BOTHNIA DURING 1973—1984, WITH A NOTE ON OTHER HEAVY METALS IN BALTIC WATERS

Folke Koroleff<sup>1)</sup> & Vappu Tervo<sup>1)</sup>

KOROLEFF, F. & TERVO, V. 1986. The concentration of iron in the waters of the Gulf of Bothnia during 1973—1984, with a note on other heavy metals in Baltic waters. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The iron concentrations in the water of the Baltic, mainly in the Gulf of Bothnia, have been measured regularly during the period 1973—1984.

In the Bothnian Sea, the iron concentrations in the water layer 10—20 m above the bottom are considerably higher than in the other Baltic Sea areas. The trend calculations show that the iron content in the Bothnian Sea has decreased slightly since 1973 both in and above the near-bottom layer.

Data are also presented on the concentrations of other heavy metals (Mn, Zn, Cu, Pb, Cd, Co, Ni, Cr) in different Baltic Sea areas in 1977—1982.

Index words: Iron, heavy metals, sea water, Gulf of Finland, Gulf of Bothnia, Bothnian Sea, Baltic Proper

## 1. INTRODUCTION

Iron has been measured in Finnish coastal waters by the National Board of Waters twice a year from 1966 to 1970 (Kohonen 1973) and in the winter season in the Gulf of Bothnia from 1966 to 1977 (Pitkänen 1979). In 1968 Koroleff (1968) determined soluble and total iron in all Baltic Sea areas.

In order to detect possible changes in the iron concentration in the Bothnian Sea, the metal has been measured during yearly cruises with R/V Aranda since 1973.

A considerable amount of data is now available on other trace metals (Magnusson and Westerlund 1980; Brüggmann 1981). The values presented here have been obtained by a co-precipitation technique and by AAS (Koroleff 1984).

## 2. MATERIAL AND METHODS

The samples were taken with 5-litre TPN samplers (Hydrobios, Kiel, FRG) during cruises with R.V. Aranda in 1973—1984. The main iron-sampling stations are shown in Fig. 1. Samples for other trace metals were taken at the stations shown in Fig. 2. Total iron was measured on board after acid-persulphate oxidation using TPTZ (2, 4, 6-tripyridyl-s-triazine)reagent (Koroleff 1983).

In 1984 some samples for iron were filtrated in all-glass filtration equipment with 0.4  $\mu$ m Nucleopore filters. Using  $Mg(OH)_2$  as collector, the heavy metals were precipitated by adding NaOH to unfiltered sea water in carefully cleaned PVC bottles. The precipitate was dissolved in  $HNO_3$  and analysed ashore with flameless AAS, Zn by

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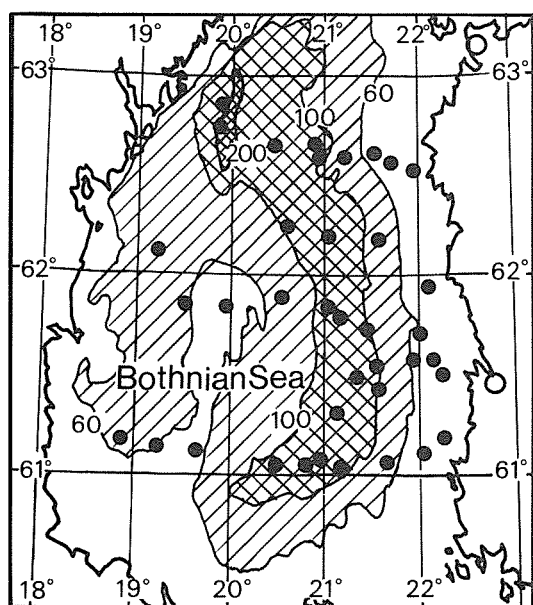


Fig. 1. Sampling stations for iron and depth conditions in the Bothnian Sea.

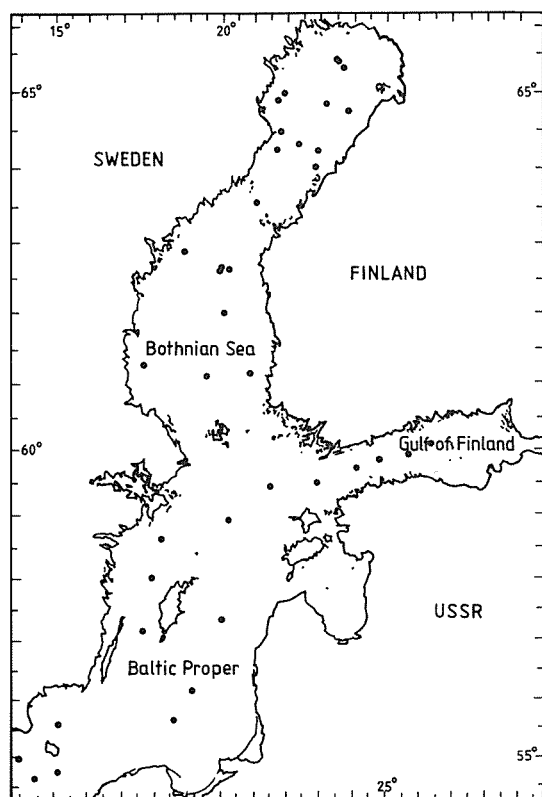


Fig. 2. Sampling stations for trace metals.

flame (Perkin-Elmer 300 with HGA 72, since 1981 P-E 4000 with HGA 400 and AS40).

### 3. RESULTS AND DISCUSSION

#### 3.1 Iron

In general, the total iron concentrations in the surface layers of different areas of the Baltic Sea vary from some  $\mu\text{g dm}^{-3}$  to about  $30 \mu\text{g dm}^{-3}$ . Occasionally in summer, after a rich production period, the concentration has sunk below the detection limit of  $1 \mu\text{g dm}^{-3}$ .

In the Bothnian Sea the iron concentrations in the near bottom water layers, 20 m from the bottom, could vary at the same sampling time from 100 to  $500 \mu\text{g dm}^{-3}$ . In the Bothnian Bay and in the Gulf of Finland iron values above  $100 \mu\text{g dm}^{-3}$  have been measured occasionally in the water close to the bottom. Results from the open area of the Bothnian Sea are presented in Figs. 3 and 4. Both trend calculations are based on about 400 measurements during the years 1973–1982. The results for the water layer 20 m from the bottom were taken for the trend calculations because the greatest concentrations of iron occur near bottom. The observations were divided into two groups, those made at stations deeper than 79 m and those from stations with a depth less than 79 m. Slight but statistically significant diminishing trends are evident from 1973 to 1982. However, the results should be treated cautiously because the iron concentrations can vary greatly at all sampling depths, since iron hydroxide sinks slowly and moves continuously with the currents.

According to Koroleff (1968), the total iron concentrations measured in 1968 in the surface layers of the Bothnian Sea were less than  $10 \mu\text{g dm}^{-3}$ , and in the deeper layers,  $> 50 \text{ m}$ , they were around  $40 \mu\text{g dm}^{-3}$ , being higher than the values recorded in the other sea areas.

The iron measurements made in winter 1966–1977 showed increasing trends in the Bothnian Sea (Pitkänen 1979). According to the latest measurements from May and August 1984, the total iron concentrations in the Bothnian Sea were  $5\text{--}10 \mu\text{g dm}^{-3}$  in the surface layer and  $50\text{--}250 \mu\text{g dm}^{-3}$  in the near-bottom layer. At the deepest stations in the Bothnian Sea, the mean obtained for total iron below 100 m was  $199 \mu\text{g dm}^{-3}$  in 1982, and  $139 \mu\text{g dm}^{-3}$  in 1984.

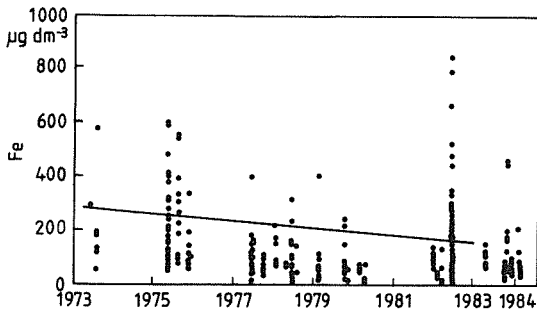


Fig. 3. Total iron concentrations in near-bottom waters in the Bothnian Sea at shallow stations, depth less than 79 m.

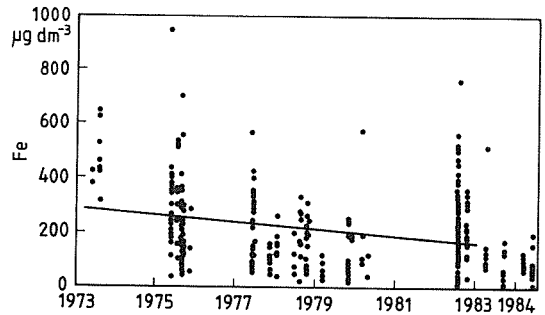


Fig. 4. Total iron concentration in near-bottom waters in the Bothnian Sea at stations deeper than 79 m.

In the filtrated samples analysed in 1984, the total soluble iron in the Bothnian Sea was 1–4  $\mu\text{g dm}^{-3}$  in both the surface and near-bottom waters.

### 3.2 Other trace metals

The concentrations of Cu, Zn, Pb, Cd, Mn, Cr, Co and Ni vary greatly at all stations. The results are presented in Fig. 5. The mean values are well within the ranges given by other authors, e.g.

Brügmann (1981). No clear differences in the metal concentrations could be found between different sea areas or between the sampling depths at any of the stations with the exception of manganese, which can occur in great concentrations in deep waters in the Baltic Proper and in the Gulf of Finland. Periods of stagnation in the deep waters are characteristic of the Baltic Proper. All oxygen is consumed and hydrogen sulphide is formed. As both iron and manganese form fairly soluble sulphides, the metals are released from the surface of the sediments and reach high concentrations, several hundreds of micrograms per litre.

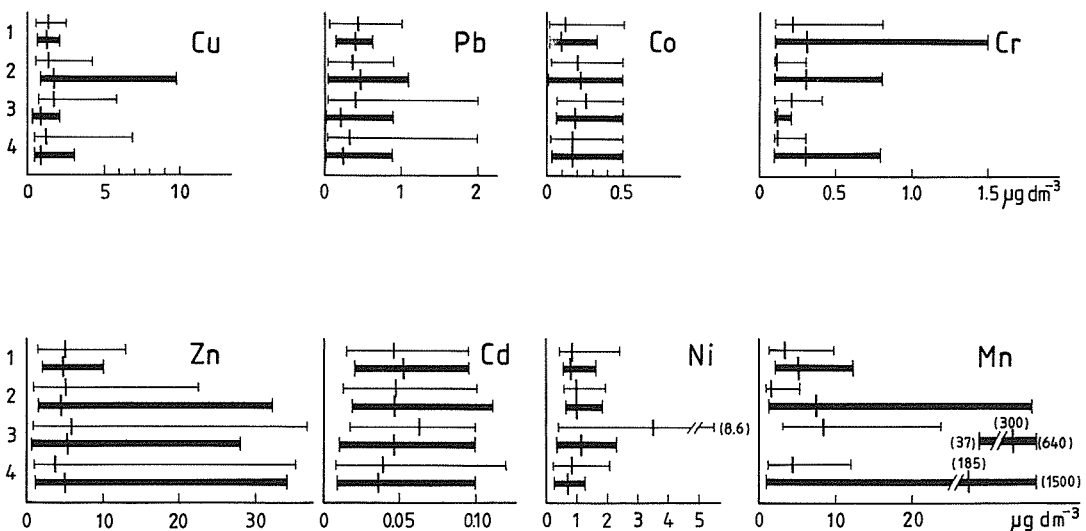


Fig. 5. Concentration ranges and means for heavy metals in the water layers above (the upper line) and below 40 m, (the thick line) in the Bothnian Bay (1), Bothnian Sea (2), Gulf of Finland (3) and Baltic Proper (4) during 1977–1982.

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## STATISTICAL FEATURES OF THE HYDROGRAPHY IN THE GULF OF BOTHNIA

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LEPPÄRANTA, M. & ALENIOUS, P. 1986. Statistical features of the hydrography in the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The basic statistical nature of the water masses of the Gulf of Bothnia is characterized. The data consist of CTD profiles and observations from fixed oceanographic stations. The mean vertical structure and the range of variation of temperature and salinity are presented, and the mean monthly depths of the thermocline and halocline are given. Average sections across the Bothnian Sea are constructed for October.

Index words: Hydrography, temperature, salinity, sea

### 1. INTRODUCTION

Statistical analysis of hydrographic data gives useful information on the climatology of sea water basins. The temperature-salinity (TS) characteristics of the water are described better by statistical than (as is often done) by selected profiles. The depths of the thermocline and halocline can be studied from statistical profiles and need not be determined for individual profiles. Furthermore, statistical features of the hydrography can serve as the basic data for climatological investigations of the heat and mass budgets of the sea. This method is becoming more popular in descriptive physical oceanography (cf. Pickard and Emery 1982) and has been used for the Baltic Sea by Alenius and Leppäranta (1982).

This paper is a progress report of our climatological investigations and modelling of the Baltic Sea. We shall study selected CTD (Conductivity-Temperature-Depth) samples from the Gulf of Bothnia collected on cruises with the R/V Aranda during 1977–1982. The main deficiency in the CTD data is that only a few observations are

available from the period December–March. In order to obtain annual statistical characteristics, we have used the data of the Finnish fixed oceanographic stations.

### 2. RESULTS AND DISCUSSION

#### 2.1 Fixed oceanographic stations

The data are records of the temperature and salinity made at 5–10 m depth intervals every 10 days. At present there are eight stations operating along the Finnish coast and in the Archipelago Sea. Here we present some statistical results for the station Märket, Åland Sea, based on the data in Granqvist (1938).

Consider the one-dimensional diffusion equation for a hydrographic property  $f = f(z, t)$  where  $z$  stands for depth and  $t$  for time:

$$\frac{\partial f}{\partial t} = \frac{\partial}{\partial z} K \frac{\partial f}{\partial z} \quad (1)$$

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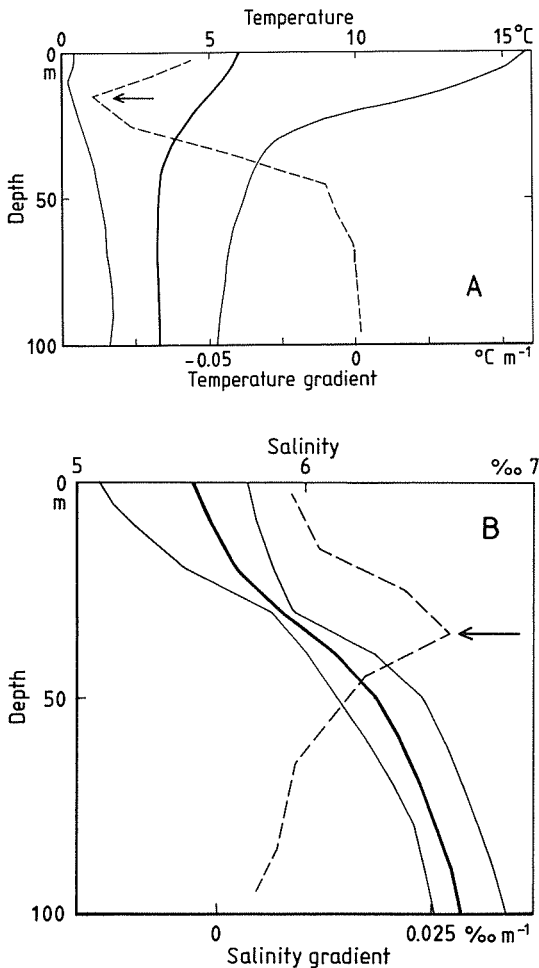


Fig. 1. Annual mean (thick solid line), range (thin solid lines) and gradient (dashed line) of (A) temperature and (B) salinity profiles at Märket. The predominant depth of the thermocline and halocline are indicated with arrows.

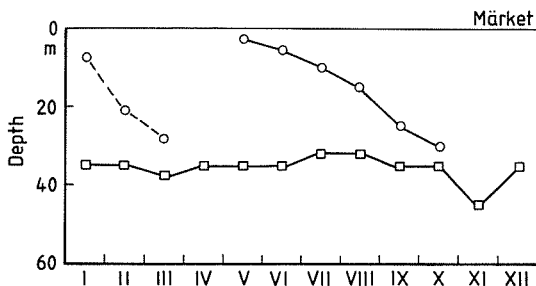


Fig. 2. The annual course of the thermocline and halocline depths at Märket:  
○—○ summer thermocline, ○— — ○ winter thermocline, □—□ halocline.

Here  $K$  is the eddy diffusivity coefficient. Averaging Eq. (1) over a time interval ( $T_1$ ,  $T_2$ ) gives

$$f = \frac{\partial}{\partial z} \left\langle K \frac{\partial f}{\partial z} \right\rangle \quad (2)$$

where  $\langle \rangle$  is the time averaging operator. In particular, if  $K$  is constant and the left-hand side of Eq. (2) is zero, the profile is linear. The left-hand side can be made zero by operating through the annual cycle.

The annual mean temperature at Märket decreases linearly with depth in the upper layer and then becomes nearly constant (Fig. 1a). This form is probably due to the eddy diffusivity being much smaller in the lower layer. The structure of the salinity distribution is somewhat different (Fig. 1b). It is seen that the slope is highest at 30–40 m depth and decreases towards the surface and the bottom to about the same value. Advective effects are important to the average salinity profile.

The gradient of a time-averaged temperature or salinity profile can be considered to be the vertical distribution of the gradients of the individual profiles. Hence the modes correspond to the predominant locations of the thermocline and halocline (Fig. 1). The temperature and salinity gradient profiles are both unimodal with the modes at 15 and 35 m depth, respectively. Fig. 2 presents the typical depths of the main thermocline and halocline for each month. Not all the monthly gradient profiles are unimodal and the depths taken correspond to the greatest mode.

The greatest mode was not well defined for all the months in the temperature records. The summer thermocline deepens 5–10 m per month and disappears in October as it reaches the halocline. The halocline depth is 30–40 m, except in November, when it deepens to 45 m.

## 2.2. CTD data from R/V Aranda cruises

In an earlier study (Alenius and Leppäranta 1982), some representative stations were chosen in different parts of the Baltic Sea to describe the TS structure of the basins. They showed that the water masses of the Bothnian Bay and Bothnian Sea form two separate groups. In the Bothnian Bay the salinity is 3–4 ‰ and in the Bothnian Sea 5–7 ‰. In the Bothnian Sea the cold winter water layer could be identified above the more saline deep water.

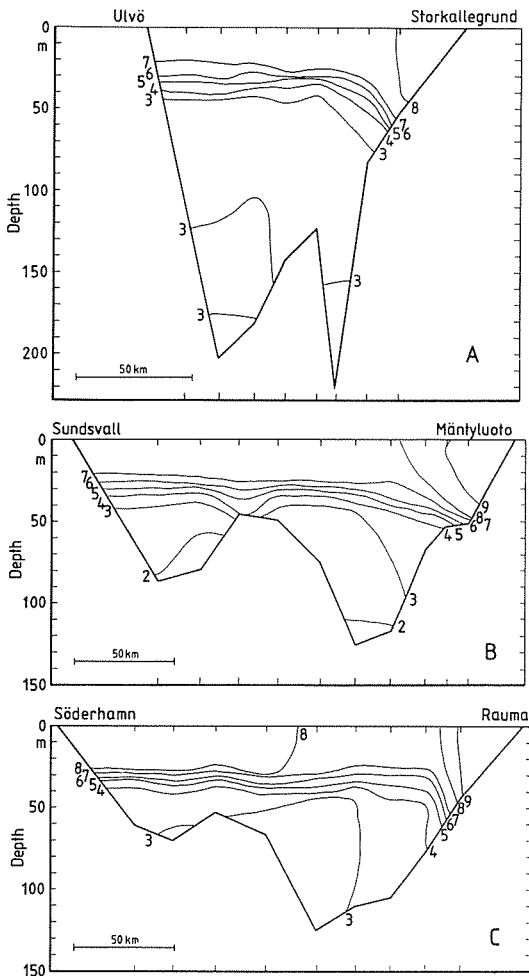


Fig. 3. Mean temperature ( $^{\circ}\text{C}$ ) sections during October in (A) the northern Bothnian Sea, (B) the central Bothnian Sea, and (C) the southern Bothnian Sea. The tick marks on the horizontal axes show the locations of the CTD stations.

The mean annual profiles were constructed and the range of variation of each quantity was determined by taking the minimum and maximum values separately for each depth. The Bothnian Bay waters were characterized by an almost linear increase in salinity from  $3.5\text{‰}$  at the surface to about  $4\text{‰}$  near the bottom. No clear halocline could be identified except possibly near the Quark. The mean temperature profile showed that the mixing in autumn and winter reaches the bottom, so that no deep water layer exists. The typical depth of the summer thermocline is  $13 \pm 2\text{ m}$ . In the Bothnian Sea, the surface salinity is about  $5.5\text{‰}$ . The salinity gradient profile is rather flat with a wide maximum from 25 to 60 m. At depths greater than 100 m the salinity is about  $7\text{‰}$ . A

clear winter water layer can be identified in the mean temperature profiles at about 55 m depth. The typical summer thermocline depths is  $14 \pm 2\text{ m}$ .

The CTD data are insufficient for constructing annual mean sections. However, some representative monthly mean sections can be computed. In terms of the amount of data, October and November are the best months and they represent the situation during autumn cooling. As an example we can take the "mean October" in the Bothnian Sea. The mean temperature sections (Fig. 3) show that the temperature of the surface layer in general increases eastward, indicating advection from the south along the eastern side of the basin. The isotherms are almost horizontal from the Swedish coast eastward to 2/3 of the basin width, but further east they incline steeply downwards towards the Finnish coast. The salinity is also higher at the Finnish than at the Swedish side of the basin. These features persist through the mean sections for November, although the temperatures in the upper layer are about  $3^{\circ}\text{C}$  lower. The thermocline is typically at about 30 m depth during October and at about 50 m depth during November.

The mean temperature sections support the assumption that during autumn there is a climatological coastal current flowing northward in the eastern side of the basin.

The width of this current appears to be about 50 km. This is in accordance with the yearly very slight mean anticlockwise circulation suggested by Palmén (1930). The temperature variation from the southern to the northern Bothnian Sea is only  $1^{\circ}\text{C}$  near the coast and even less in the central part of the basin.

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## VARIATIONS IN DRY MATTER, PHOSPHORUS AND ORGANIC CARBON IN TWO BOTHNIAN BAY SEDIMENT CORES IN RELATION TO HYDRAULIC ENGINEERING WORKS

Kalervo Mäkelä<sup>1)</sup>

MÄKELÄ, K. 1986. Variations in dry matter, phosphorus and organic carbon in two Bothnian Bay sediment cores in relation to hydraulic engineering works. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Since the 1950s, hydraulic engineering works have been carried out on a large scale in Ostrobothnia, western Finland. This paper deals with the possible effects of this activity on the offshore sediments. A dating method based on sulphate gradients revealed good agreement between the periods of river clearing and the variation of the dry matter content in the sediments. In the central part of the Bothnian Bay, the concentrations of sedimentary phosphorus appeared to reflect the intensity of river engineering, although they were possibly also elevated by floods. The organic carbon in the sediment increased at the time of the first extensive dredging operations.

Index words: Water engineering, Ostrobothnia, sedimentation, dry matter, phosphorus, organic matter

### 1. INTRODUCTION

Extensive hydraulic engineering works have been carried out in Ostrobothnia in order to prevent floods in coastal areas (Fig. 1). Since 1950, one million cubic metres of land mass have been moved; the corresponding masses in the drainage of farm land amount to  $75 \times 10^6 \text{ m}^3$  and in forest drainage to  $100 \times 10^6 \text{ m}^3$  (Vesihallitus 1984).

Dredging of rivers has adverse effects on the waters to which they discharge. Increased concentrations of suspended solids, iron, phosphorus, nitrogen and heavy metals are observed at least 30 km downstream from the dredging area (Vesihallitus 1983). During the spring floods, river water can be identified as far as 20 km from the coast, but it is uncertain how far the effects of rivers are recognizable in the sediments, and what role is played by construction work in the transport of suspended matter to the Bothnian Bay.

### 2. MATERIAL AND METHODS

Two sediment samples were taken with a gravity corer for analysis of dry matter, organic carbon, phosphorus and interstitial sulphate. The characteristics of the stations are given in Table 1.

Two methods were used to estimate the sedimentation rates,  $^{210}\text{Pb}$ -dating and study of the sulphate gradient. At station F 15 additional information was provided by the thickness of the postglacial sediment bed, which indicated a long-term average of  $4\text{--}6 \text{ mm a}^{-1}$ . The method of  $^{210}\text{Pb}$ -dating gave about  $4 \text{ mm a}^{-1}$  for the uppermost part of the sediment at station F 15 and  $1.9 \text{ mm a}^{-1}$  at station BO 3. The sulphate gradient was assumed to parallel the rate of sedimentation (Berner 1978). Fig. 2 shows the relationship between the  $^{210}\text{Pb}$ -datings and the sulphate gradients in five cores from the Baltic Sea. Because of their good agreement with the lead dating, the

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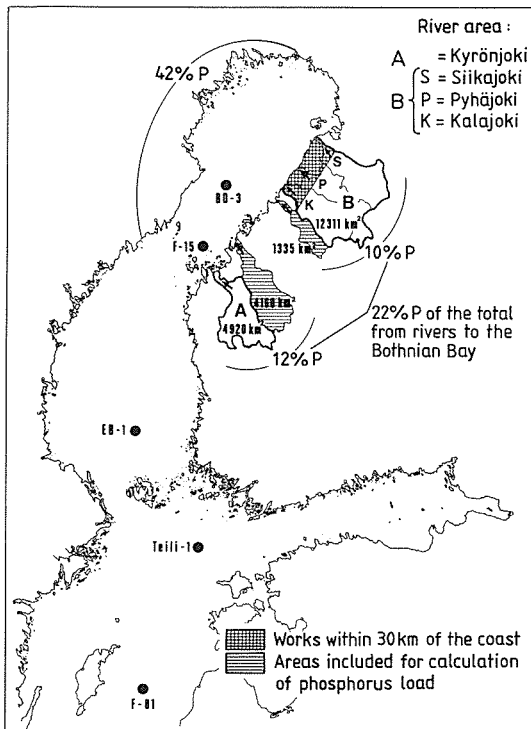


Fig. 1. The study area and sampling stations. (Values for phosphorus are from Wartiovaara (1978) and from Kommittén för Bottniska viken 1983).

Table 1. Station characteristics.

| Station      | Location           | Distance from the coast<br>km | Depth<br>m | Thickness of oxidized layer<br>cm | Sedimentation rate                          |                                         |
|--------------|--------------------|-------------------------------|------------|-----------------------------------|---------------------------------------------|-----------------------------------------|
|              |                    |                               |            |                                   | SO <sub>4</sub> -grad<br>mm a <sup>-1</sup> | <sup>210</sup> Pb<br>mm a <sup>-1</sup> |
| BO 3<br>1981 | 64°18'N<br>22°21'E | 80                            | 108        | 7                                 | 1.84                                        | 1.9                                     |
| F 15<br>1982 | 63°31'N<br>21°31'E | 40                            | 49         | 2                                 | 4.16                                        | 4.0                                     |

values of the sulphate gradients were used to construct time scales for both cores.

### 3. RESULTS AND DISCUSSION

Fig. 3 presents the dry matter content, organic carbon and interstitial sulphate in the sediment core at station F 15. Several peaks can be distinguished in the dry matter content. Some coincide with the clearing periods in the downstream areas of the river Kyrönjoki. The greatest peak is centred around the period 1929–35, when one million cubic metres of land mass were excavated in the lower reaches of the river (Bilaledtin 1983). The recent clearing and embankment works at the river mouth are also reflected as positive anomalies in the dry matter content. A clear transition from grey to black sulphidic clay was observed at a depth of 24 cm. This coincides with the time of the first intensive clearing in the 1920s. Organic carbon has an increasing trend in the younger layers, whereas sulphate curve has a concave up appearance around the dry matter maximum. This indicates a change in the rate of sedimentation (cf. Gieskes 1975).

In the central part of the Bothnian Bay the sedimentation conditions are more stable (sulphate gradients), but minor variations can be observed in the dry matter (station BO 3, Fig. 4). At this station the positive anomalies coincide with extensive works in the watercourses of the rivers Siikajoki, Pyhäjoki and Kalajoki (Fig. 1, B), documented by Turunen (1983). The concentrations of sedimentary phosphorus reflect the drainage and clearing activities fairly clearly. They are possibly further increased by floods, but floods are not solely responsible for the increase of phosphorus and solids is the river discharge, as

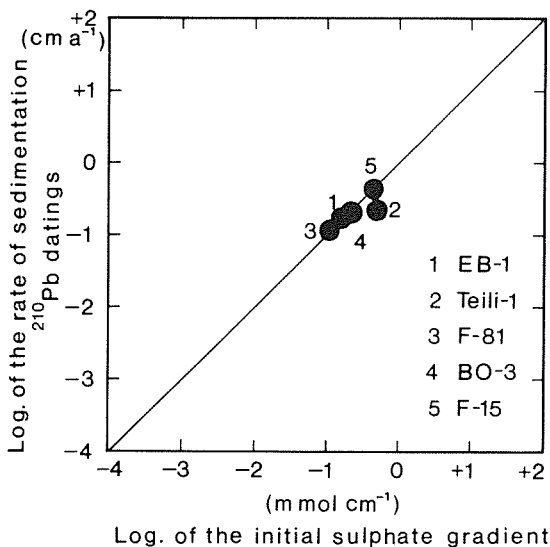


Fig. 2. The method of estimating sedimentation rates from sulphate gradients according to Berner (1978). The line was determined by Berner on the basis of 15 different sedimentary environments around the world. The small scatter shows the applicability of the general theory to special basins such as the Baltic Sea.

indicated by the studies of Kauppi (1979) and Seuna (1983). The rapid increase of phosphorus in the river water when the construction works are started is supported by other observations (Maataloushallitus 1969). Of particular significance is the decrease of phosphorus during wartime, during the post-war economic depression and after the reduction in the intensity of construction works in this area in 1974.

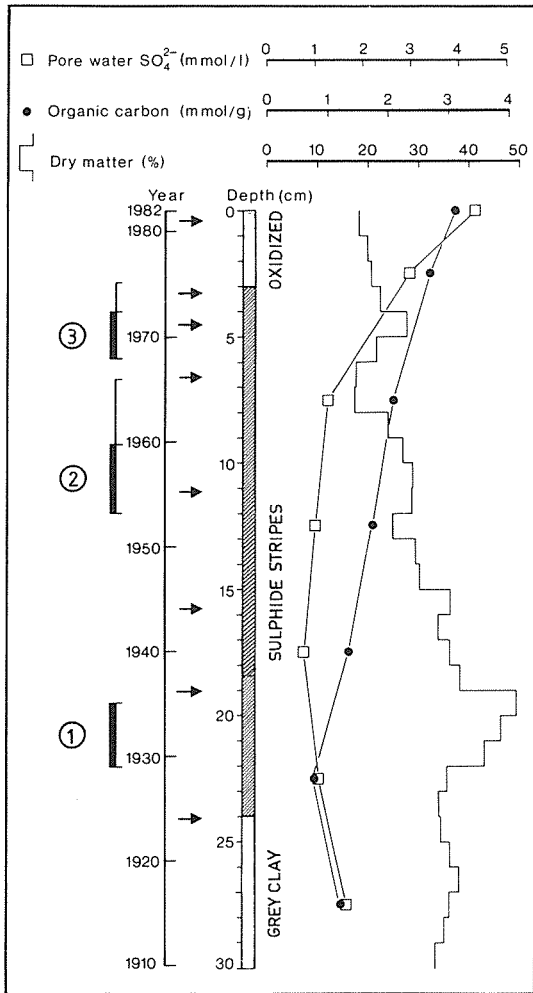


Fig. 3. Sediment properties in a core sample 40 km from the river mouth (St F 15, 1982). Note the peaks in dry matter content, which coincide with the construction works in the river Kyrönjoki area A, as shown by the segments and black bars (intensified dredging periods). The arrows indicate abnormally high floods and are included in the figure because solids and runoff have a positive correlation (Hyyvärinen 1982, Seuna 1983).

(1) Dredging in the downstream area in 1929–35 (One million cubic metres moved), (2) Embankment of the Bay of Vassor in the river mouth in 1953–66, (3) Embankments in the lower reaches of the river in 1968–75 (1.1 million cubic metres of land masses moved since 1950).

## 4. CONCLUSIONS

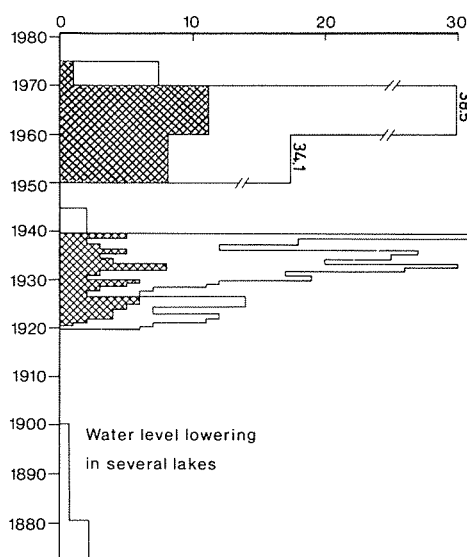
The dating method based on sulphate gradients yielded a time scale in good agreement with dry matter sedimentation and the construction works in the study area. The conclusions concerning phosphorus may be simplistic because the role of iron is not considered. Phosphorus mimics the iron distribution and iron can migrate in reduced sediments. However, iron showed marked variation in these sediments (unpublished), and this element also largely originates from rivers. Increasing trends have been shown by iron in many rivers in Ostrobothnia, and in the Bothnian Bay as well (Pitkänen 1978, Laaksonen and Malin 1980). It is possible that hydraulic engineering works are partly responsible for this development.

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# DRAINING, CLEARING AND OTHER WORKS IN NORTHERN OSTROBOTHNIA (B)

Number of water construction works in  
Kalajoki (K), Pyhäjoki (P) and Siikajoki (S) area



# SEDIMENT PROPERTIES IN A CORE SAMPLE 80 KM FROM THE COAST (BO-3, 1981)

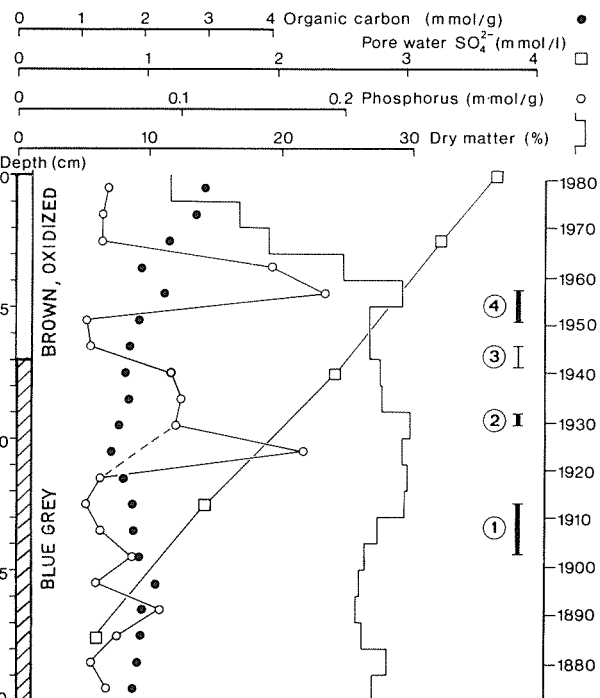


Fig. 4. Water construction works in northern Ostrobothnia (left, Turunen 1983) and sediment properties (right) at station BO 3, 80 km from the coast, in 1981.

The number of all the water construction works in the area B (Fig. 1) is presented as open bars (note overscaling). Works within 30 km of the coast are cross-shaded. They correspond better with the variation in phosphorus than do all the works together. In the right-hand part of the figure attention is focused on the variation of the dry matter content of the sediment. Positive anomalies in dry matter content are emphasized as dark bars. They coincide with the known water construction works listed below: (1) Regulation of the river Kalajoki, 1903—13 (increase in dry matter content), (2) Clearing of the river Pyhäjoki basin, 1929—32 (increase in dry matter), (3) Slowing down of construction works (War time, decrease in phosphorus), (4) Flood prevention in the river Pyhäjoki, 1952—57 (increase in dry matter and phosphorus). Note also the increase in organic carbon since the nineteen-twenties. Variation in phosphorus content is possibly amplified by floods (dotted line). The occurrence of floods is indicated by the arrows in the middle of the figure.

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## SURFACE DISTRIBUTION OF TOTAL PHOSPHORUS AND TOTAL NITROGEN IN THE FINNISH COASTAL WATERS IN 1979—1983

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Matti Perttilä<sup>3)</sup>

PITKÄNEN, H., KANGAS, P., EKHOLM, P. & PERTTILÄ, M. Surface distribution of total phosphorus and total nitrogen in the Finnish coastal waters in 1979—1983. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

A general picture is presented of the total phosphorus and total nitrogen levels in the surface layer of the Finnish coastal waters. The material consists of data obtained from about 350 sampling stations, during monitoring of the coastal waters and the open sea in 1979—1983. Isopleths are given for salinity, total phosphorus and total nitrogen in winter and late summer. The distribution of nutrients is largely explained by the distribution of terrestrial loading and by the mixing of river and waste waters with water from the open sea. In the sheltered inner coastal waters the distribution is evidently modified to some extent by sedimentation. The total terrestrial input to the Finnish coastal waters is estimated at 5100 t a<sup>-1</sup> of phosphorus and 80 000 t a<sup>-1</sup> of nitrogen.

Index words: total phosphorus, total nitrogen, salinity, loading, water quality, coastal waters, Gulf of Bothnia, Gulf of Finland, Baltic Sea, monitoring

### 1. INTRODUCTION

The levels of inorganic nutrients in the open sea areas around Finland were studied as early as 1932 by Buch. During recent years, a number of papers have dealt with the concentrations of inorganic or total phosphorus and nitrogen (e.g. Voipio 1973, Kohonen 1974, Pietikäinen et al. 1978, Perttilä et al. 1980 and Pitkänen and Malin 1980).

The nutrient levels in the coastal waters have been examined in numerous studies, the most intensively investigated areas being the NE Bothnian Bay (Alasaarela 1978, 1979a, b, c) and the Tvärminne—Pojo Bay area on the Finnish SW coast (Halme 1944, Niemi 1973, 1975, 1978, Häll-

fors et al. 1983). The spatial distribution of nutrients has also been studied, e.g. in the Northern Quark (Sevola 1978), the inner Archipelago Sea (e.g. Jumppanen and Kolehmainen 1984) off Kopparnäs (Bruun et al. 1980), off Helsinki and Espoo (e.g. Pesonen and Rinne 1985), off Skjöldvik (Penttinen 1980) and in the easternmost Finnish waters in the Gulf of Finland (e.g. Kettunen and Lempinen 1983).

The aim of this study is to present and discuss the spatial distribution of total phosphorus and total nitrogen in the surface layer of the Finnish coastal waters using all available material collected in winter and late summer in the period 1979—1983.

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## 2. MATERIAL AND METHODS

The bulk of the material originated from the monitoring performed by the National Board of Waters (90 sampling stations, Fig. 1). Additional data were obtained from local pollution surveillance projects, supervised by the Board (240 stations) and from open sea monitoring carried out by the Finnish Institute of Marine Research (26 stations). The chemical analyses were performed by the standard methods used by the Board and the Institute (Koroleff 1979, National Board of Waters 1981 a).

The winter samples were taken during February—March, when most of the study area is usually covered by ice (SMHI and Finnish Institute of Marine Research 1982). Samples taken during July—September were considered to represent the late summer. Only surface data (0—2 m) were used for the coastal waters, but the uppermost 10 metres were included for the open sea. For most of the stations the number of observations made per season during the study period was 5—10.

The spatial distribution of salinity, phosphorus and nitrogen is shown with the isopleths of the mean values for summer and winter. In areas where data were missing or the station network was sparse, a broken line has been used to indicate the approximate location of the isopleths. They are thus unreliable in certain unpolluted waters and also in parts of the outer coastal waters. On the other hand, the water quality in these areas is relatively stable and the errors are unlikely to be large, except in certain enclosed parts of the inner archipelago.

## 3. RESULTS

### 3.1 Salinity

As in earlier studies, the highest values for the open sea were recorded in the outer Archipelago Sea and the entrance to the Gulf of Finland (6.8—6.9 ‰ in winter and 6.5—6.8 ‰ in summer, Figs. 2 and 3). The salinity decreased with the increasing proportion of fresh water towards the NE Bothnian Bay (< 3 ‰) and the easternmost Finnish part of the Gulf of Finland (4—5 ‰).

The salinity gradient from the open sea towards the river mouths was especially pronounced at the surface during winter. At that time almost pure fresh water lying just below the ice extended over large areas, particularly in the NE Bothnian Bay

and off the estuaries of the Rivers Kokemäenjoki and Kymijoki. The waters in which the surface salinity was less than 1.0 ‰ had a total area of about 2 500 km<sup>2</sup> and extended along about 25 % of the coastline. Lower salinities than in the open sea were observed nearly all around the coast. During summer only areas off the estuaries of the largest rivers and the innermost parts of the archipelagoes were directly affected by river waters.

### 3.2 Total phosphorus

In wintertime the concentration of total phosphorus in the open sea averaged 6—8 mg m<sup>-3</sup> in the Bothnian Bay, 11—13 mg m<sup>-3</sup> in the Bothnian Sea, 22—23 mg m<sup>-3</sup> in the Archipelago Sea and 25—53 mg m<sup>-3</sup> in the Gulf of Finland (Fig. 4).

In the coastal waters the concentration usually increased towards the coastline. The mean concentrations were as high as 30—40 mg m<sup>-3</sup> over extensive areas. In the strongly loaded inner bays by Helsinki, Porvoo, Salo and Uusikaupunki, mean concentrations of over 100 mg m<sup>-3</sup> were recorded.

Extensive areas with elevated concentrations were found in the NE Bothnian Bay, the archipelago of the Quark, the waters off the Kokemäenjoki estuary and the inner Archipelago Sea. Off the mouth of the River Kymijoki, an area of about 1000 km<sup>2</sup> was affected by phosphorus-poor river water, and the concentration decreased towards the coast. In general the relative difference between the concentrations in the coastal waters and the open sea was smaller in the Gulf of Finland than in the Gulf of Bothnia, due to the high concentrations in the open Gulf of Finland.

In summer the open sea concentrations were 6—7 mg m<sup>-3</sup> in the Bothnian Bay, 8—9 mg m<sup>-3</sup> in the Bothnian Sea, 12—13 mg m<sup>-3</sup> in the Archipelago Sea and 13—17 mg m<sup>-3</sup> in the Gulf of Finland (Fig. 5). The values were thus clearly lower than the winter level, except in the Bothnian Bay, where no marked difference was observed.

In the coastal waters the summertime mean concentrations usually ranged from 10 mg m<sup>-3</sup> to 40 mg m<sup>-3</sup>. The values were generally lower than in winter. Near the estuaries of the Rivers Kymijoki, Karjaanjoki, Kokemäenjoki and Oulujoki, however, higher mean concentrations were observed than in winter. The areas whose concentrations were extensively elevated compared with the levels in the open sea were the same as in winter, but also included the coastal waters off Helsinki—Espoo and Kymijoki—Kotka—Hamina on the southern coast.

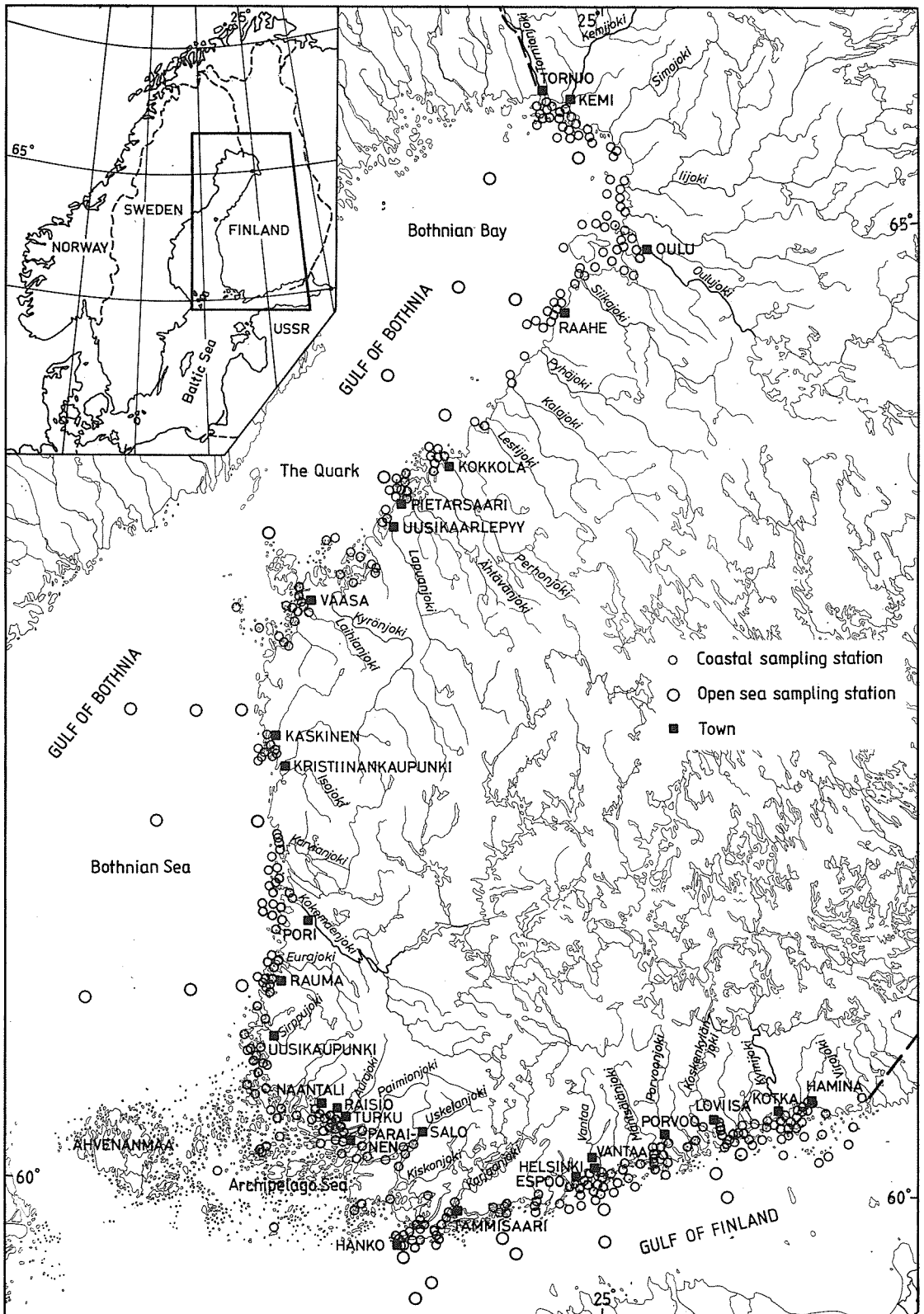


Fig. 1. The study area and sampling stations.

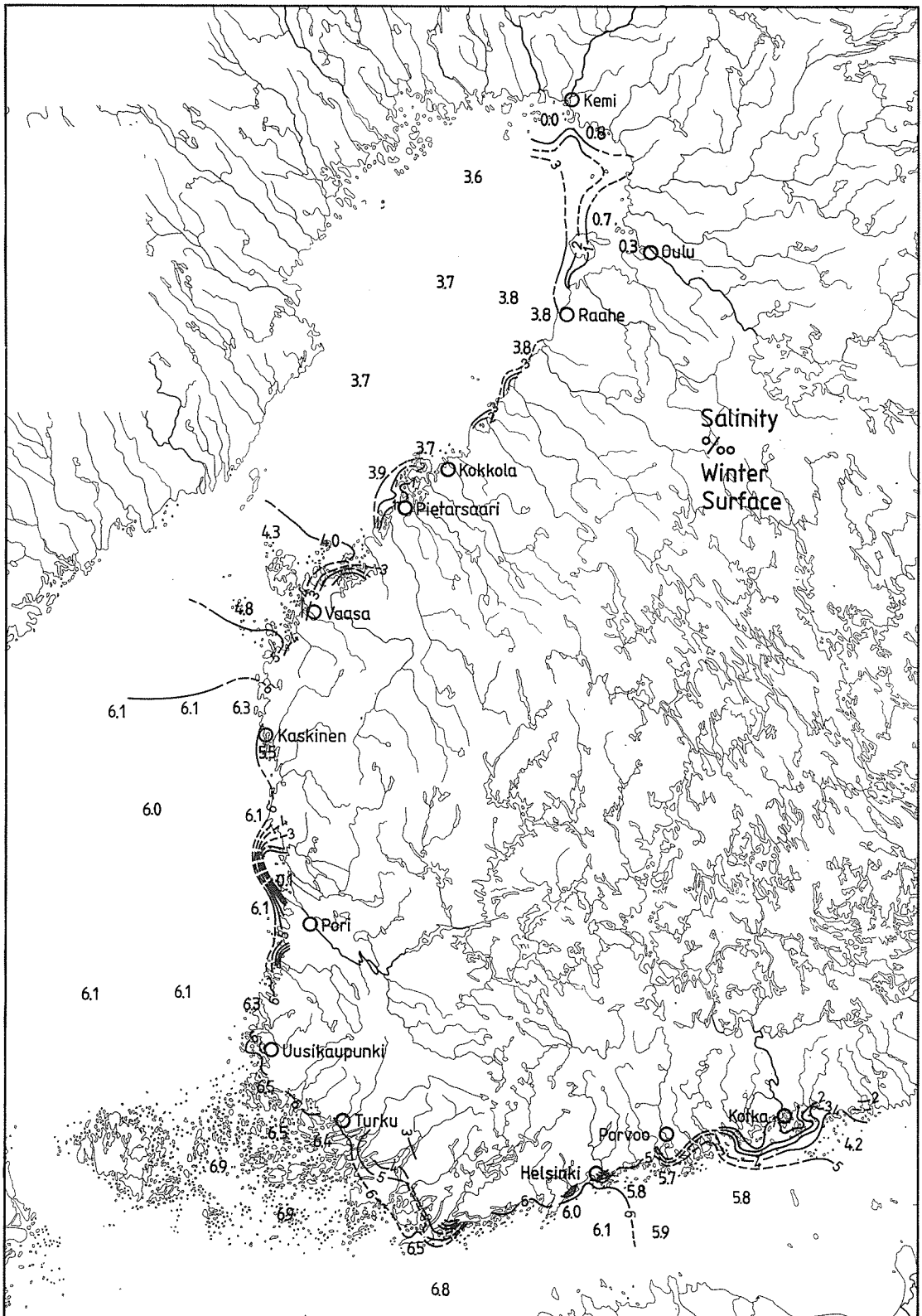


Fig. 2. Mean surface values for salinity in winters 1979–1983. Some of the salinity data for the inner coastal waters are calculated from electric conductivity values according to the regression equation of Jumppanen (1981).

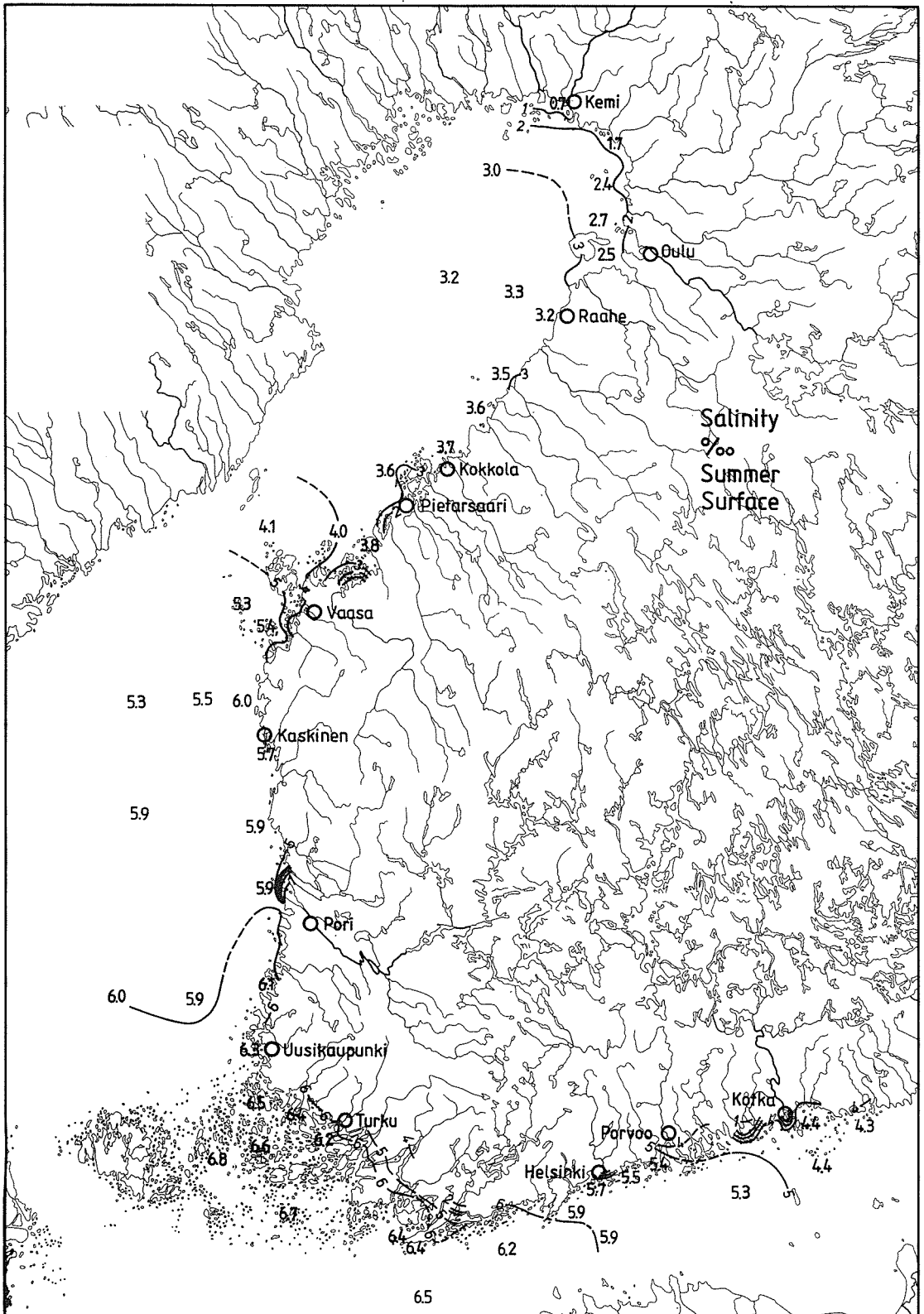


Fig. 3. Mean surface values for salinity in late summers 1979–1983. Explanations as in Fig. 2.

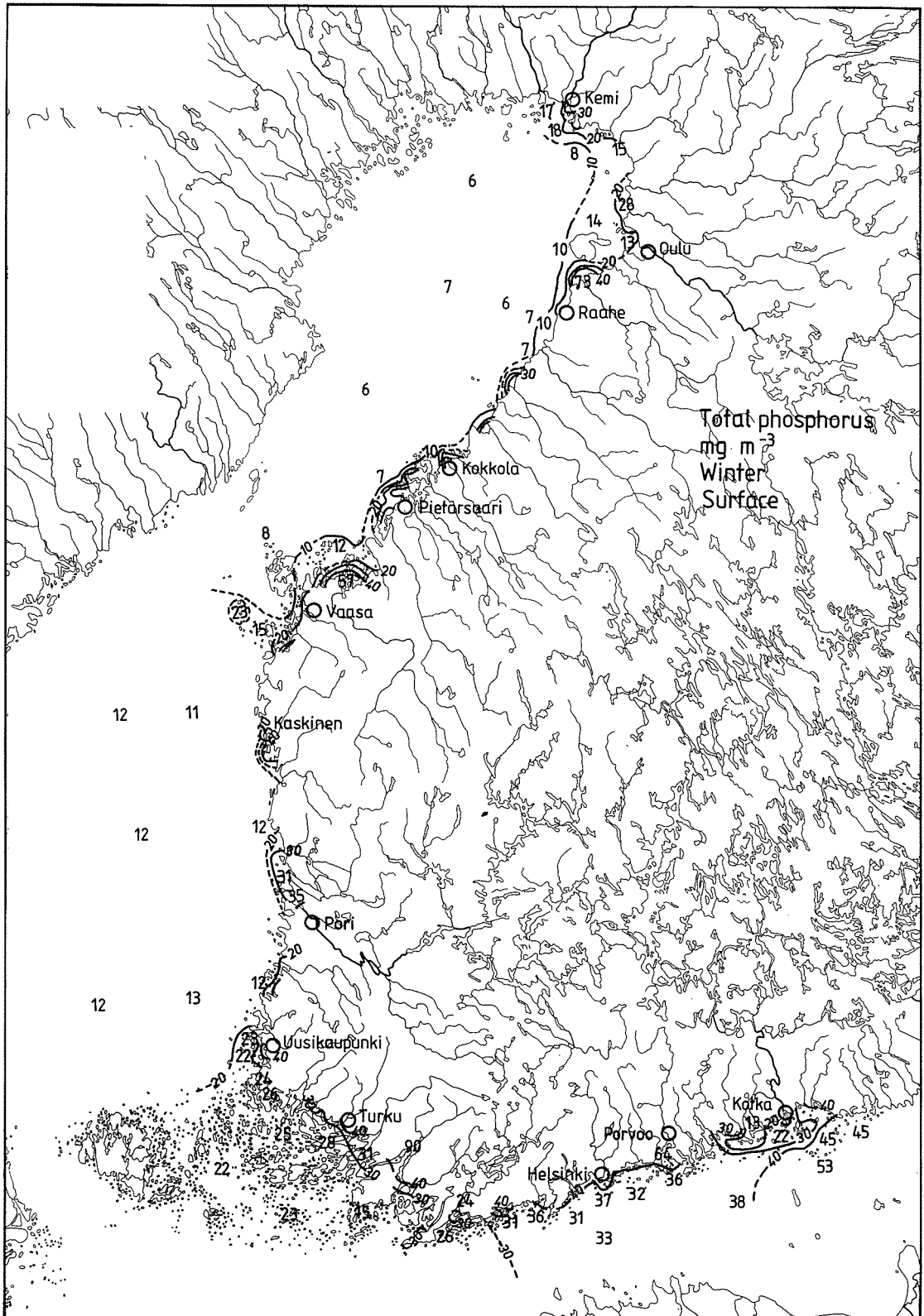


Fig. 4. Mean surface values for total phosphorus in winters 1979–1983.

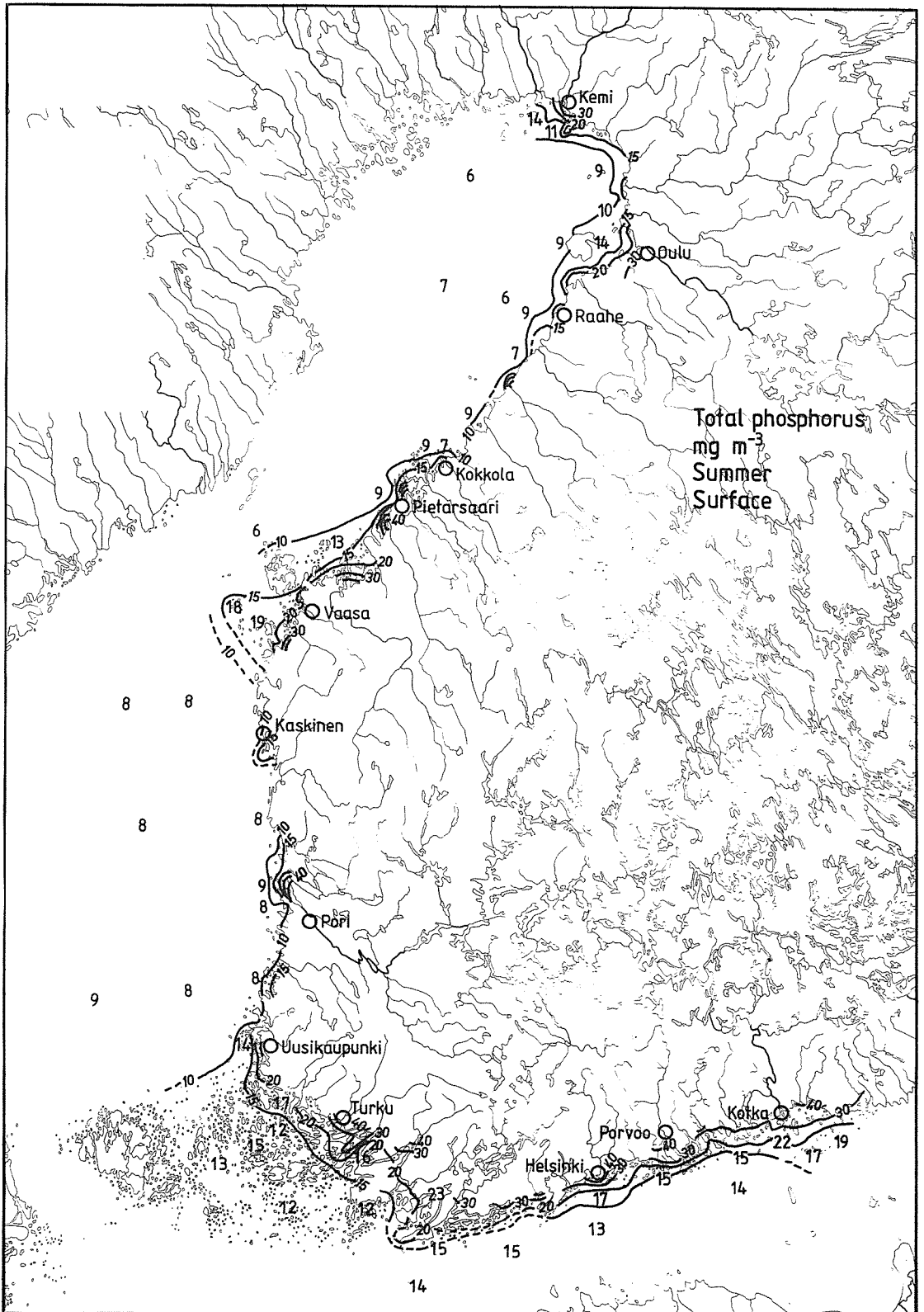


Fig. 5. Mean surface values for total phosphorus in late summers 1979–1983.

### 3.3. Total nitrogen

The wintertime level in the open sea was 260–300 mg m<sup>-3</sup> in the Bothnian Bay and the Bothnian Sea, 310–330 mg m<sup>-3</sup> in the Archipelago Sea and 350–450 mg m<sup>-3</sup> in the Gulf of Finland (Fig. 6).

In the coastal waters the mean concentrations generally ranged from 300 to 600 mg m<sup>-3</sup>. Greater values were observed in the estuaries of most of the coastal rivers and the River Kokemäenjoki (Pitkänen 1986), where mean values of 800–2 000 mg m<sup>-3</sup> were obtained. The areas with extensively elevated concentrations were the same as in the case of wintertime phosphorus. Off the mouth of the River Kymijoki the concentration increased only slightly towards the coast.

The late summer level was 200–270 mg m<sup>-3</sup> in the Bothnian Bay, 230–250 mg m<sup>-3</sup> in the Bothnian Sea, around 300 mg m<sup>-3</sup> in the Archipelago Sea and 290–320 mg m<sup>-3</sup> in the Gulf of Finland (Fig. 7). The values were lower than in winter, especially in the Archipelago Sea and the Gulf of Finland.

The coastal water areas with elevated concentrations in summer were much more restricted than in winter. The mean summertime level varied between 350 and 500 mg m<sup>-3</sup> in the NE Bothnian Bay, off Kokkola–Pietarsaari, off the mouths of the Kyrönjoki and Kokemäenjoki, in the inner Archipelago Sea and in most parts of the coastal waters of the Gulf of Finland. The mean concentration reached 1 000 mg m<sup>-3</sup> in the estuaries of the Kyrönjoki, Kokemäenjoki, Uskelanjoki, Vantaa and Porvoonjoki.

The relative difference between the concentrations in the open sea and coastal waters was generally much smaller in summer than in winter. The situation was thus the opposite to that of the total phosphorus concentrations.

## 4. DISCUSSION

The spatial distribution of total phosphorus and total nitrogen in the surface layer of the open sea is mainly in good accordance with earlier studies (Voipio 1973, Kohonen 1974, Pietikäinen et al. 1978, Perttilä et al. 1980, Pitkänen and Malin 1980), but a general increase has taken place in the concentrations since the 1960s, especially in the Gulf of Finland and the outer Archipelago Sea. This is most probably due to intensification of the vertical exchange of nutrients in the Baltic Proper,

together with increased anthropogenic influences (Larsson et al. 1983, Nehring 1985).

The distribution of nutrients in the open sea is largely determined by the hydrographical conditions in the Baltic Sea (Buch 1932, Fonselius 1969, Grasshoff and Voipio 1981, Nehring 1981). The same mainly applies to the outer coastal waters, although the proximity of the land generally affects the water quality to some extent, at least during late winter and spring (e.g. Niemi 1973, 1975, Alasaarela 1979a, c, Laakkonen et al. 1981, Hällfors et al. 1983, Pitkänen 1985). In the inner coastal waters the concentration depends closely on the quantity and quality of terrestrial inputs, the mixing conditions and the various processes occurring in the water and between the water and the sediment (e.g. Niemi 1973, 1978, Alasaarela 1978, 1979a, b, c, Rinne et al. 1981, Rekolainen 1982, Ekholm 1986).

Although both spatial and temporal variation exists in the relation between the levels of total nutrients and the biologically available nutrients, the correspondence between total nutrients and the variables describing primary productivity in the different parts of the coastal water is fairly good (e.g. Niemi 1973, 1975, 1978, Alasaarela 1978, 1979b, c, Lassig et al. 1978, Bruun et al. 1980, Penttinen 1980, Hällfors et al. 1983, Kettunen and Lempinen 1983, Jumppanen and Kolehmainen 1984, Pesonen and Rinne 1985, Pitkänen et al. 1985). This applies in particular to total phosphorus, but largely also to total nitrogen.

The total terrestrial input to the Finnish coastal waters is on an average 5100 t a<sup>-1</sup> of total phosphorus and 80 000 t a<sup>-1</sup> of total nitrogen (Tables 1 and 2). Rivers and direct loading are responsible for 86–88 % and 12–14 % of the loading, respectively. During both winter and late summer the proportions are roughly 80 % for rivers and 20 % for direct sources in the case of nitrogen and 75 % for rivers and 25 % for direct sources in the case of phosphorus. The inputs from lake-rich inland drainage basins are relatively high during winter and late summer, while the inputs from small coastal drainage basins are usually small, due to the weaker stream flow during these seasons (Table 2, Wartiovaara 1978, Pitkänen 1986). The nutrients which accumulate in the surface layer of coastal and open sea waters during winter and early spring are largely fixed and sink to deeper layers and the bottom during the spring bloom of phytoplankton (e.g. Niemi 1973, 1975, Hällfors et al. 1983, Kuparinen et al. 1984). Their effect on the pelagic primary production is thus essentially restricted to the vernal stage. The production level of the late summer is primarily affected by inputs

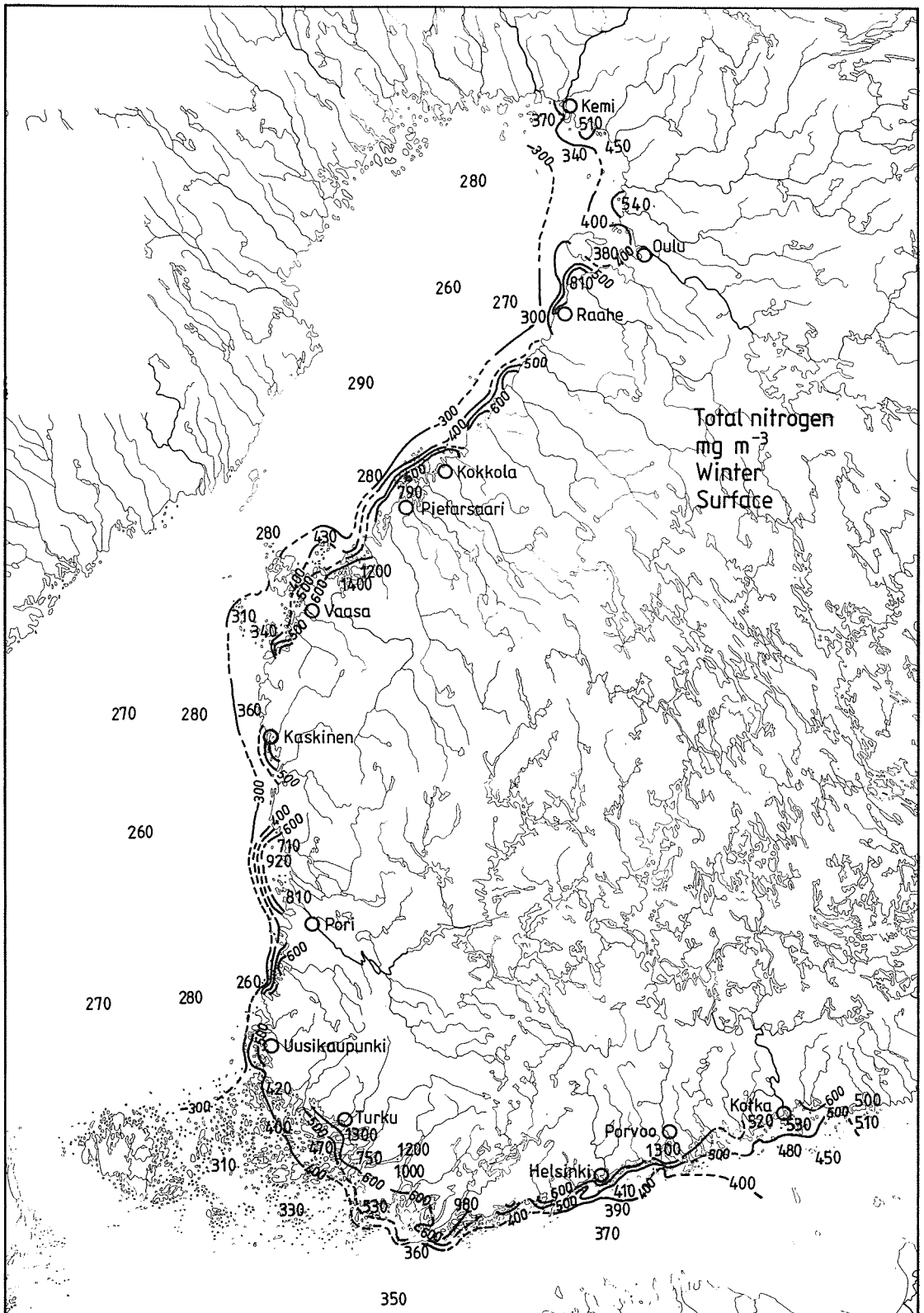


Fig. 6. Mean surface values for total nitrogen in winters 1979–1983.

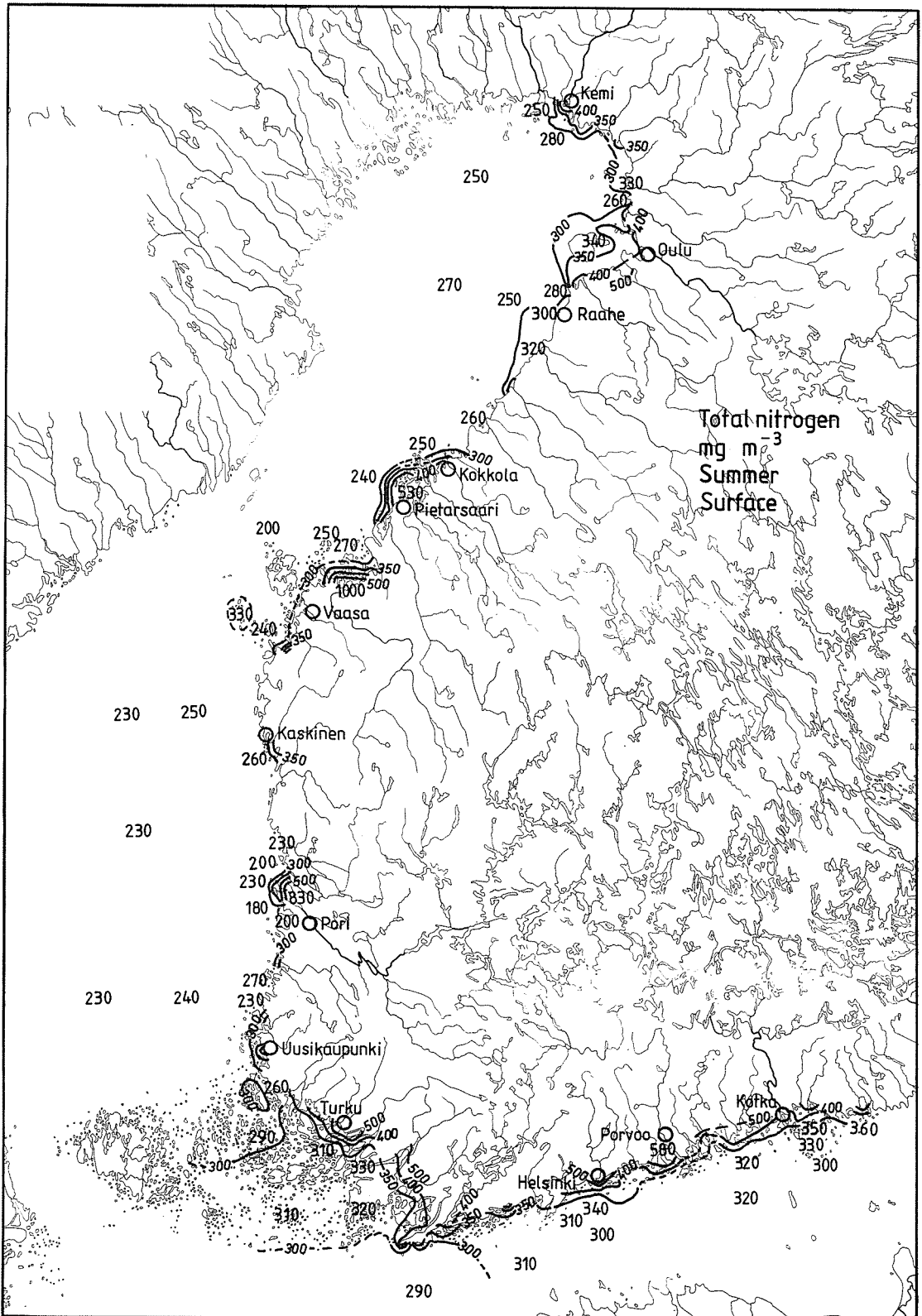


Fig. 7. Mean surface values for total nitrogen in late summers 1979–1983.

Table 1. The discharges of phosphorus and nitrogen from municipal and industrial sources to Finnish coastal waters during 1979–1983 (National Board of Waters 1980, 1981b, 1983, 1984a, b, Data Bank of Industrial Water Pollution Control)<sup>1)</sup>. Small sources of loading not presented individually but included in the total values.

| Coastal region                    | Discharges of phosphorus (t a <sup>-1</sup> ) |                  |                  | Discharges of nitrogen (t a <sup>-1</sup> ) |            |       |
|-----------------------------------|-----------------------------------------------|------------------|------------------|---------------------------------------------|------------|-------|
|                                   | Municipal                                     | Industrial       | Sum              | Municipal                                   | Industrial | Sum   |
| Tornio                            | 5.9                                           | 0.2              | 6.1              | 51                                          | 77         | 130   |
| Kemi                              | 19                                            | 54               | 73               | 110                                         | 280        | 380   |
| Oulu                              | 25                                            | 28               | 53               | 510                                         | 150        | 660   |
| Raahe                             | 3.5                                           | 2.0              | 5.5              | 69                                          | 28         | 97    |
| Kokkola                           | 2.1                                           | 7.6              | 9.7              | 110                                         | 1 100      | 1 200 |
| Pietarsaari <sup>2)</sup>         | 8.1                                           | 32               | 40               | 110                                         | 140        | 240   |
| Bothnian Bay                      | 67                                            | 130              | 200              | 1 000                                       | 1 700      | 2 700 |
| Vaasa                             | 18                                            | 2.5              | 20               | 190                                         | 13         | 200   |
| Kaskinen <sup>3)</sup>            | 5.0                                           | 47               | 52               | 25                                          | 160        | 190   |
| Pori                              | 12                                            | 13               | 25               | 300                                         | 56         | 360   |
| Rauma                             | 4.7                                           | 25               | 30               | 75                                          | 91         | 170   |
| Uusikaupunki                      | 1.2                                           | 10 <sup>4)</sup> | 11 <sup>4)</sup> | 59                                          | 29         | 88    |
| Bothnian Sea                      | 42                                            | 97               | 140              | 650                                         | 350        | 1 000 |
| Turku                             |                                               |                  |                  |                                             |            |       |
| and neighbouring towns            | 29                                            | 2.2              | 32               | 900                                         | 110        | 1 000 |
| Salo                              | 14                                            | 0.2              | 14               | 92                                          | 6          | 98    |
| Hanko                             | 0.0                                           | 0.0              | 0.0              | 0                                           | 31         | 31    |
| Archipelago Sea <sup>5), 6)</sup> | 51                                            | 5.0              | 56               | 1 100                                       | 160        | 1 200 |
| Hanko                             | 9.9                                           | 0.0              | 9.9              | 66                                          | 0          | 66    |
| Tammisaari                        | 4.1                                           | 1.2              | 5.3              | 52                                          | 11         | 63    |
| Helsinki—Espoo                    | 120                                           | 0.0              | 120              | 3 100                                       | 1          | 3 100 |
| Porvoo                            | 6.2                                           | 5.6              | 12               | 120                                         | 210        | 330   |
| Loviisa                           | 1.8                                           | 0.0              | 1.8              | 46                                          | 2          | 48    |
| Kotka—Hamina                      | 35                                            | 31               | 66               | 260                                         | 220        | 480   |
| Gulf of Finland                   | 180                                           | 40               | 220              | 3 700                                       | 460        | 4 200 |
| Total direct load from the coast  | 340                                           | 270              | 610              | 6 500                                       | 2 700      | 9 200 |

<sup>1)</sup> the discharges from fish farming are excluded due to insufficient data. In 1982 the load was 54 t of phosphorus and 530 t of nitrogen (Halonen 1985)

<sup>2)</sup> includes loading from the town of Uusikaarlepyy

<sup>3)</sup> includes loading from the town of Kristiinankaupunki

<sup>4)</sup> in addition large amounts of phosphorus are leached to the sea from a waste heap by the fertilizer factory. The loading was 50 t in 1982 and 20 t in 1983 (Turku Water District 1984)

<sup>5)</sup> the additional load from fish farming in 1982 was 30 t of P and 270 t of N (Halonen 1985)

<sup>6)</sup> the Province of Åland is excluded

during the summer months, at least near sources of loading.

If only the late summer season is examined (Tables 1 and 2), rivers are generally found to contribute the main part of the nutrient input to the coastal waters of the Bothnian Bay, to the middle part of the Bothnian Sea, to the Archipelago Sea and to the Gulf of Finland, except the Helsinki—Espoo area. The nutrient loading of the coastal waters by rivers is equalled or exceeded by

the discharges from the industries and municipalities of Raahe, Kokkola and Pietarsaari in the Bothnian Bay, from Vaasa, Kaskinen, Rauma and Uusikaupunki (mainly for phosphorus) in the Bothnian Sea, from Turku and Salo in the Archipelago Sea and from Hanko and Helsinki—Espoo in the Gulf of Finland. Substantial amounts of nutrients are released from industries and municipalities of Kemi and Oulu in the NE Bothnian Bay and Kotka—Hamina in the eastern Gulf of Fin-

Table 2. River discharges of phosphorus and nitrogen to the Finnish coastal waters in 1979–1983 and percentages for winter (January–March) and late summer (July–September) during this period (material from Pitkänen 1986 and the National Board of Waters, Finland). Brackets indicate uncertainty due to missing data.

| River                                                   | Coastal region          | Discharges of phosphorus    |                  |                  | Discharges of nitrogen      |                  |                  |
|---------------------------------------------------------|-------------------------|-----------------------------|------------------|------------------|-----------------------------|------------------|------------------|
|                                                         |                         | annual<br>t a <sup>-1</sup> | winter<br>%      | late summer<br>% | annual<br>t a <sup>-1</sup> | winter<br>%      | late summer<br>% |
| Tornionjoki                                             | Tornio                  | 300                         | 3                | 20               | 4 300                       | 6                | 23               |
| Kemijoki                                                | Kemi                    | 470                         | 10 <sup>1)</sup> | 13 <sup>1)</sup> | 7 600                       | 16 <sup>1)</sup> | 13 <sup>1)</sup> |
| Simojoki                                                | NE Bothnian Bay         | 41                          | 6 <sup>2)</sup>  | 13 <sup>2)</sup> | 710                         | 7 <sup>2)</sup>  | 10 <sup>2)</sup> |
| Iijoki                                                  | "                       | 180                         | ..               | ..               | 2 400                       | ..               | ..               |
| Kiiminkijoki                                            | "                       | 67                          | 6                | 9                | 1 000                       | 6                | 11               |
| Oulujoki                                                | Oulu                    | 190                         | 22               | 19               | 3 400                       | 28               | 18               |
| Siikajoki                                               | NE Bothnian Bay         | 130                         | 11               | 9                | 1 500                       | 10               | 9                |
| Pyhäjoki                                                | central Bothnian Bay    | 82                          | ..               | ..               | 1 100                       | ..               | ..               |
| Kalajoki                                                | "                       | 160                         | 10               | 14               | 2 100                       | 11               | 12               |
| Lestijoki                                               | "                       | 31                          | 16               | 27               | 490                         | 18               | 20               |
| Perhonjoki                                              | Kokkola                 | 73                          | 16 <sup>1)</sup> | 4 <sup>1)</sup>  | 980                         | 13 <sup>1)</sup> | 5 <sup>1)</sup>  |
| Ähtävänjoki                                             | Pietarsaari             | 21                          | 17               | 26               | 390                         | 27               | 18               |
| Lapuanjoki                                              | Uusikaarlepyy           | 120                         | 11               | 14               | 2 200                       | 17               | 10               |
| Kyrönjoki                                               | The Quark               | 160                         | 10               | 16               | 3 000                       | 9                | 12               |
| Rivers discharging to the Bothnian Bay <sup>3)</sup>    |                         | 2 300                       | (10)             | (14)             | 36 000                      | (14)             | (13)             |
| Laihianjoki                                             | Vaasa                   | 12                          | ..               | ..               | 380                         | ..               | ..               |
| Isojoki                                                 | Kristiinankaupunki      | 40                          | 7                | 7                | 540                         | 7                | 6                |
| Karvianjoki                                             | central Bothnian Sea    | 120                         | ..               | ..               | 1 400                       | ..               | ..               |
| Kokemäenjoki                                            | Pori                    | 570                         | 25               | 19               | 9 000                       | 25               | 15               |
| Eurajoki                                                | central Bothnian Sea    | 20                          | ..               | ..               | 490                         | ..               | ..               |
| Sirppujoki                                              | Uusikaupunki            | 14                          | ..               | ..               | 450                         | ..               | ..               |
| Rivers discharging to the Bothnian Sea <sup>3)</sup>    |                         | 940                         | ..               | ..               | 15 000                      | ..               | ..               |
| Aurajoki                                                | Turku                   | 48                          | 4                | 13               | 590                         | 4                | 10               |
| Paimionjoki                                             | Archipelago Sea         | 49                          | ..               | ..               | 540                         | ..               | ..               |
| Uskelanjoki                                             | Salo                    | 28                          | ..               | ..               | 310                         | ..               | ..               |
| Kiskonjoki                                              | Archipelago Sea         | 19                          | ..               | ..               | 280                         | ..               | ..               |
| Rivers discharging to the Archipelago Sea <sup>3)</sup> |                         | 360                         | ..               | ..               | 4 300                       | ..               | ..               |
| Karjaanjoki                                             | Tammisaari              | 21                          | 26               | 18               | 550                         | 32               | 15               |
| Vantaa                                                  | Helsinki                | 78                          | 13               | 12               | 1 400                       | 17               | 10               |
| Mäntsälänjoki                                           | Porvoo                  | 29                          | ..               | ..               | 420                         | ..               | ..               |
| Porvoonjoki                                             | Porvoo                  | 73                          | 12               | 14               | 1 400                       | 15               | 15               |
| Koskenkylänjoki                                         | central Gulf of Finland | 27                          | 15               | 14               | 420                         | 15               | 10               |
| Kymijoki                                                | Kotka                   | 280                         | 20               | 26               | 6 300                       | 25               | 20               |
| Virojoki                                                | E Gulf of Finland       | 11                          | ..               | ..               | 170                         | ..               | ..               |
| Rivers discharging to the Gulf of Finland <sup>3)</sup> |                         | 750                         | (15)             | (17)             | 15 000                      | (19)             | (14)             |
| Total load from rivers                                  |                         | 4 400                       | (13)             | (14)             | 70 000                      | (15)             | (13)             |

<sup>1)</sup> data only for 1982

<sup>2)</sup> data only for 1983

<sup>3)</sup> extrapolated values for the whole catchment area

land. These discharges are, however, clearly exceeded by discharges from local rivers.

Fish farming is a notable source of nutrients, particularly in the Archipelago Sea (Hälonen 1985). During the summer of 1982, this input accounted for about 40 % of the phosphorus and 20 % of the nitrogen discharged to the Archipelago Sea (Turku Water District 1984).

The relationship between the average concentrations and salinities suggests that dilution is the

main factor causing the observed nutrient distributions in the coastal waters during both winter and late summer (Fig. 8). Off the city of Helsinki, for example, the loading seem to spread and become diluted with brackish water, without marked net sedimentation. Sedimentation seems to be promoted by complexity of the topography of the coastal region and high concentrations in the river discharge, as in the estuary of the River Uskelanjoki. It is also possible that the sediments at times

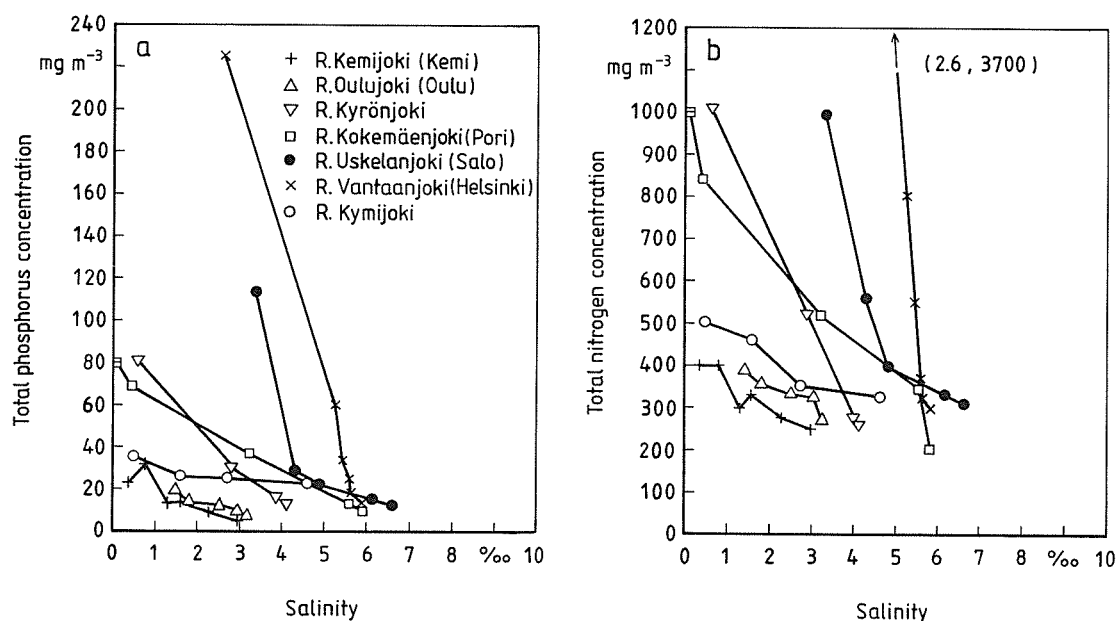


Fig. 8. Dependence of mean phosphorus and mean nitrogen concentrations on mean salinity in the surface layers of some estuaries and adjacent waters, a, phosphorus in late summer, b, nitrogen in late summer.

act as a net source of nutrients, especially in eutrophicated coastal waters (cf. Liss 1976, Aston 1980, Rekolainen 1982, Håkanson et al. 1984, Heikkilä 1986, Ekholm 1986, Pitkänen et al. 1986).

Retention of soluble fractions of nutrients near the coast is probably more pronounced during summer than winter, due to intense primary production during summer and to extensive spreading of river and waste waters in a separate layer below the ice during winter (Alasaarela 1979a, Pitkänen et al. 1986). On the other hand, the conditions below the ice cover during winter favour sedimentation of particulate matter, which evidently affects the concentrations of total phosphorus.

The net behaviour of total nitrogen seems to be more conservative than that of total phosphorus, due to differences in the state and physico-chemical properties of these substances and due to differences in the biogeochemical processes they undergo during the mixing of river and saline water (e.g. Liss 1976, Aston 1980).

## 5. CONCLUSIONS

From the spatial patterns presented above and the available loading data, it can be concluded that

- The present material provides a general picture of the spatial distribution of the potential nutrient resources of the coastal surface waters. Although the relation between the total amounts of nutrients and the biologically available amounts varies both spatially and temporally, fairly good correspondence has been found in various studies between the total concentrations of nutrients and the variables related to the production level.
- During winter and late summer, the total concentrations of phosphorus and nitrogen in most parts of the coastal surface waters are higher than in the open sea. The areas where this coastal effect is most extensive are the NE Bothnian Bay, the archipelago of the Northern Quark, the inner Archipelago Sea, the waters off the estuaries of the Rivers Kokemäenjoki and Kymijoki (during summer), and the waters off Helsinki—Espoo.
- The total terrestrial loads of phosphorus and nitrogen imposed on the Finnish coastal waters are estimated at 5 100 t a<sup>-1</sup> and 80 000 t a<sup>-1</sup>, respectively. The natural and anthropogenic nutrients discharged by rivers account for 86—88 % of the load. During winter and late summer, the average contributions of the rivers are 75—80 %. During the late summer the nutrient loading of the coastal waters by rivers was equalled or exceeded by the loading from

direct discharges of waste waters in the regions of Raahe, Kokkola and Pietarsaari in the Bothnian Bay, Vaasa, Kaskinen, Rauma and Uusikaupunki in the Bothnian Sea, Turku and Salo in the Archipelago Sea, and Hanko and Helsinki—Espoo in the Gulf of Finland.

- The main factors regulating the distribution of total nutrients in the surface layer of the coastal waters are the quantity and quality of the inputs, the mixing conditions (coastal morphology, season) and the concentrations in the open sea. Pronounced net sedimentation seems to take place primarily in sheltered inner parts of the archipelagoes.

## ACKNOWLEDGEMENTS

We would like to thank Mr Martin Forsius for help in processing the data. We are greatly indebted to Professor Åke Niemi, Docent Paavo Tulkki and Dr Guy Hällfors for valuable comments on the manuscript.

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## THE POLLUTANT LOAD IMPOSED ON THE GULF OF BOTHNIA — A SURVEY

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ENCKELL-SARKOLA, E. 1986. The pollutant load imposed on the Gulf of Bothnia — a survey. Publications of the Water Research Institute, National Board of Waters, Finland. No 68.

The annual load of nutrient substances imposed on the Gulf of Bothnia averages 8 000 tons of phosphorus and 150 000 tons of nitrogen. Roughly half of this load is transported by rivers, while about 25 % of the phosphorus and 40 % of the nitrogen is deposited from the atmosphere. The major part of the organic matter comes from industry and rivers. Together with the municipal load it corresponds to 460 000 tons of BOD<sub>7</sub> per year. The quantity of other pollutants is also surveyed. Gaps in load measurements and the lack of information on the effects of the pollutants are pointed out. Trends in the pollution load and their reasons are discussed.

Index words: Gulf of Bothnia, pollutants, discharge, municipal sewage, industrial effluents, river load, atmospheric deposition

### 1. INTRODUCTION

The load of polluting substances imposed on the marine environment can be divided into five sectors, according to the source: municipal sewage, industrial effluents, river load, atmospheric deposition and discharges from ships. One of the aims of the collaboration around the Gulf of Bothnia is to elucidate the influence of the first four sectors on the Gulf and to identify the critical factors most urgently requiring attention.

The chief subject chosen for the 1983 Annual Report of the Committee on the Gulf of Bothnia was the pollutant load imposed on the Gulf. A survey was made of the development during the 1970s and up to 1982. This subject is dealt with more fully in several of the presentations at the seminar.

### 2. THE TOTAL LOAD IMPOSED ON THE GULF OF BOTHNIA

According to the survey presented in the Annual Report of the Committee on the Gulf of Bothnia (Enckell-Sarkola et al. 1984), the total load of phosphorus received by the Gulf of Bothnia averages 8 000 tons a year. About a quarter of this load is received from the atmosphere. Of the load originating from the land (6 000 t a<sup>-1</sup>), about 80 % is carried into the Gulf by the rivers and 20 % comes with municipal and industrial effluents discharged directly to the sea (Fig. 1).

The total nitrogen load is about 150 000 tons a year, of which 40 % comes from the atmosphere and 60 % from the land. Of the land-based load (89 000 t a<sup>-1</sup>), 90 % enters the Gulf in the rivers and 10 % is discharged directly to the sea in

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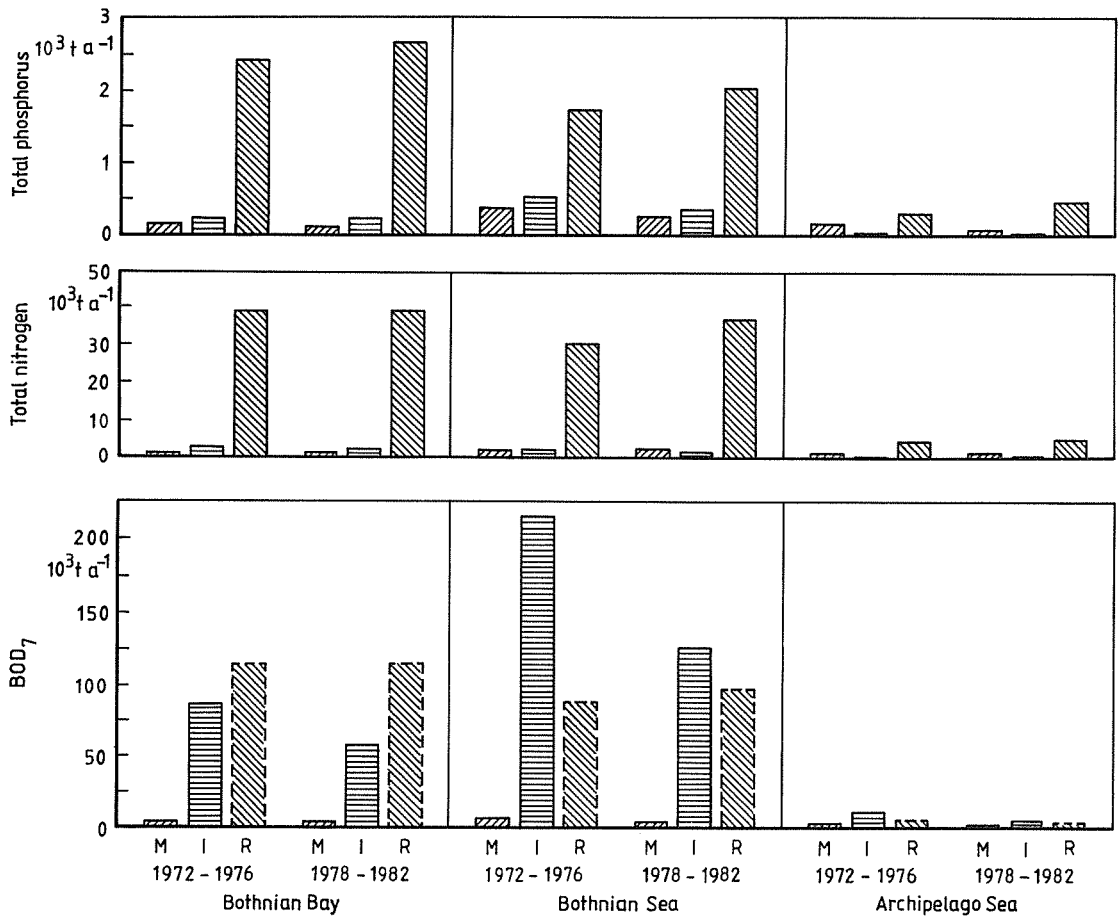


Fig. 1. The average loads of phosphorus, nitrogen and organic matter (BOD<sub>7</sub>) received by the Gulf of Bothnia from municipal wastewater (M), industrial effluents (I) and rivers (R).

municipal and industrial effluents (Fig. 1).

The total load of organic matter from the land has been estimated to correspond to 460 000 tons of BOD<sub>7</sub> per year. The air-borne fraction has not been estimated, but it is clearly a smaller proportion of the load than in the case of phosphorus and nitrogen (Järvinen and Haapala 1980). The contribution from the rivers was almost as great as that from industry, whereas municipal sewage contributed only 2 % of the land-based load.

A comparison of the loads of heavy metals is hampered by the fact that data on the contributions from the rivers are available from both countries only for copper, zinc and iron. Nor is information available on the municipal metal discharges. The discharges of several heavy metals from industry decreased during the 1970s and are now smaller than the corresponding river-borne loads. However, the industrial discharges of arsenic to the

Bothnian Bay and of iron to the Bothnian Sea are still of the same size order as the loads discharged by the rivers. The discharges of chromium, cadmium and lead are also fairly great in certain regions, though they are clearly smaller than the combined riverborne loads. The studies on the atmospheric deposition of cadmium and lead show that this is of the same size order as the estimate of the combined loading from rivers and point sources (Table 1) (Enckell-Sarkola et al. 1984).

The oil discharged by industry to the Gulf of Bothnia in 1982 was ca. 130 tons (Enckell-Sarkola et al. 1984). The load from population centres has been estimated to be of the order of 100–600 tons a year, according to a national report delivered to the Working Group on Criteria and Standards (WGS) of the Helsinki Commission. It should be borne in mind, however, that the effect of the different pollutant sources is not directly proportional to the quantity of the load. This applies not

Table 1. Metal loads ( $\text{t a}^{-1}$ ) received by the Gulf of Bothnia

| Source             | Hg  | Cd  | As | Cr | Pb  | Cu  | Zn    | Fe      |
|--------------------|-----|-----|----|----|-----|-----|-------|---------|
| Industry (1982)    | 0.5 | 0.7 | 60 | 20 | 10  | 30  | 300   | 25 000  |
| Population centres | ..  | ..  | .. | .. | ..  | ..  | ..    | *       |
| Rivers             | ..  | 1)  | 2) | 3) | 4)  | 520 | 1 200 | 110 000 |
| Atmosphere         | ..  | 7   | .. | .. | 180 | ..  | ..    | *       |

1) Finnish rivers  $2.8 \text{ t a}^{-1}$ 2) Finnish rivers  $52 \text{ t a}^{-1}$ 3) Finnish rivers  $97 \text{ t a}^{-1}$ 4) Finnish rivers  $45 \text{ t a}^{-1}$ 

\* small in comparison with the loads from other sources

only to oil, but to all discharges. For example, the heavy metals transported by the rivers are mainly in particulate form, where as industrial effluents often contain metals in dissolved form, which are more dangerous for aquatic organisms. It should also be remembered that the loads listed here do not cover the entire field of polluting factors. Great gaps in our knowledge still exist. Besides those already mentioned, the most important are perhaps the lack of information on the toxicity of the effluents from the pulp industry and on the discharges of other organic environmental toxicants (Enckell-Sarkola et al. 1984).

significance of sufficient hydraulic capacity and a sewer network that is impervious to rainwater. He also emphasized the importance of reliable overflow monitoring.

Widell (1984) had compiled in his poster statistics on the 76 municipal wastewater treatment plants on the Swedish coast of the Gulf of Bothnia. He pointed out that the discharge of phosphorus will be reduced further by increased use of chemical precipitation at biological treatment plants.

### 3. THE DISCHARGE FROM MUNICIPAL SEWER SYSTEMS

At the beginning of the 1970s, the load imposed on the Gulf of Bothnia by the municipal sewer systems still showed an increasing trend on account of new connections and enlargement of the systems. The construction of new and improved treatment plants, however, led to subsequent decreases in the BOD loading and in the discharges of phosphorus up till 1980, the reduction being particularly marked in the Bothnian and the Archipelago Seas (Enckell-Sarkola et al. 1984).

To illustrate the methods of measuring the municipal discharges and the variation in the loads, operational statistics were presented at the seminar, for a Swedish and a Finnish treatment plant. The discharge monitoring is fairly similar in the two countries, with the exception of nitrogen, which normally is not measured in Sweden (Enckell-Sarkola et al. 1984, Eriksson 1984, Luntamo 1984). Eriksson showed clearly the

### 4. THE DISCHARGE FROM INDUSTRY

The discharge of industrial effluents to the Gulf of Bothnia was greatest during the 1960s. During the first half of the 1970s a sharp decrease occurred in the BOD loading and the discharges of toxic substances. In Sweden, the production of sulphite pulp, including NSSC pulp, was discontinued at 11 factories during the years 1974–1983 and the declining trend in the effluents from the pulp industry has continued into the 1980s. In Finland the production of sulphite pulp was terminated at four factories during the years 1974–1978. In addition, production ceased at four sulphate pulp factories in Sweden, while the average capacity of more modern factories increased. Two new sulphate factories came into use, one in Sweden in 1975 and one in Finland in 1977. Of the 22 factories on the Swedish coast, five have biological wastewater treatment at present. Of the nine pulp and paper mills on the Finnish coast, three have biological wastewater treatment (Enckell-Sarkola et al. 1984).

In spite of the structural rationalization and water protection measures, the greatest threat to

the Gulf of Bothnia is still posed by the effluents from the woodprocessing industry together with discharges of iron and other metals, such as arsenic, chromium, lead and zinc (Enckell-Sarkola et al. 1984). Part of the polluting substances transported by the rivers also originated from industry. Målstedt (1984) in his poster showed that the drainage water from mines and their slag-heaps can result in considerable discharges to the water-courses.

## 5. THE LOAD TRANSPORTED BY THE RIVERS

During the period 1970—1982, the load carried by the rivers draining to the Gulf of Bothnia increased by about 13 % and the phosphorus, nitrogen and organic matter transported to the Gulf increased by 9, 24 and 12 %, respectively. The reason for the increasing trend in the load from the rivers was chiefly the increase in river flow. The increased nitrogen load was also due to the increased discharge of sewage, the increased wash out of organic matter and atmospheric deposition (Enckell-Sarkola et al. 1984, Pitkänen 1986).

Towards the end of the period 1973—1983, the nutrient salts carried to the sea by the River Aura increased more than the river flow. Jumppanen and Kolehmainen (1986) consider that the most probable reason is changes in agricultural practices. The annual variation in the river is great and about half the transport of nutrient salts occurs during April and May. Similar annual variation is evident in the River Kyrö, but towards the end of the 1970s the phosphorus load transported by the river decreased, due to more effective treatment of the municipal sewage (Meriläinen 1986).

Regulation of the rivers evens out the amounts of substances carried by them during the year, as was shown in the poster by Ahl (1984). Ahl also pointed out that studies are needed on the effect of flow regulation on the ecology of the coast and sea.

A study was made in 1983 of the material balances in a river mouth in the Gulf of Finland (Pitkänen et al. 1986). Of the solid substances and phosphorus arriving in the area in the course of a year, about one quarter evidently remained in the river mouth, but no net retention of nitrogen occurred. During the time of the ice cover, the substances in solution and fine-grained material were carried further out to sea in the river water than when the sea was icefree.

## 6. ATMOSPHERIC DEPOSITION

The loads received by the Gulf of Bothnia from the atmosphere were estimated on the basis of Finnish and Swedish measurements. The figures show that the methods of measuring are rather uncertain. One of the reasons for this is that the measurements are performed at various coastal stations and not out on the open sea (Enckell-Sarkola et al. 1984). The Swedish measurements are presented by Granat (1986), who notes that the deposition of nitrate no longer shows the consistent increase that was recorded earlier.

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## WET DEPOSITION OF INORGANIC NITROGEN COMPOUNDS ( $\text{NH}_4$ and $\text{NO}_3$ ) OVER THE GULF OF BOTHNIA ESTIMATED FROM MEASUREMENTS IN SWEDEN

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GRANAT, L. 1986. Wet deposition of inorganic nitrogen compounds ( $\text{NH}_4$  and  $\text{NO}_3$ ) over the Gulf of Bothnia estimated from measurements in Sweden. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

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On the basis of measurement made over land in Sweden, the wet deposition of ammonium and nitrate during 1983 was estimated at 12 000 ton N ( $43 \mu\text{mol l}^{-1}$ ) over the Bothnian Bay and 25 000 ton N ( $47 \mu\text{mol l}^{-1}$ ) over the Bothnian Sea.

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Index words: wet deposition, precipitation chemistry, Gulf of Bothnia, ammonium, nitrate

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### 1. INTRODUCTION

The wet deposition of ammonium and nitrate, together with several other major and minor constituents is measured within the national Swedish precipitation chemistry network (the IMI/PMK network, see Monitor 1983). In the absence of extensive direct measurements of wet deposition in the Baltic, the results from the stations in Sweden are used here to estimate the wet deposition over the Bothnian Bay and the Bothnian Sea. These estimates are to be compared with estimates based on measurements from other sampling sites around the Baltic and with other inputs to the Baltic.

### 2. MATERIAL AND METHODS

The number of IMI/PMK sampling sites was increased to about 30 in 1983. Typically, two bulk collectors are run at each site, at ten of the sites there is also a wet-only collector.

The sampling is continuous, each sample covering one month. The wet-only collector is described by Bäcklin et al. (1978). The bulk collector is of a funnel and bottle type. During winter this is replaced by a 1.2 m long polyethylene bag collector with evaporation shield. Collected samples are sent to our laboratory and stored at  $+4^\circ\text{C}$  until they are analysed. Ammonium is determined colorimetrically as indophenol blue.

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Nitrate is reduced to nitrite with cadmium and reacted with sulphanilamid and NEA to give an azo dye, which is measured colorimetrically. Both methods are used in a continuous flow application.

### 3. RESULTS AND DISCUSSION

The samples from wet-only and bulk collectors were compared at six sites for one year. The average difference (wet only-bulk) was  $-0.01 \text{ mg NO}_3\text{-N l}^{-1}$  and  $0.04 \text{ mg NH}_4\text{-N l}^{-1}$ , as compared with a mean concentration at the sites of  $0.37$  and  $0.40 \text{ mg N l}^{-1}$ , respectively.

The additional deposition in the bulk collectors is most likely due to organic dead and living material originating from the immediate surroundings of the collector. Thus, when the values from bulk collectors are extrapolated to the open water bodies, the nitrogen deposition may be overestimated by some 5 %. No correction is attempted, since this possible error is rather small compared with errors due to the extrapolation procedure, possible presence of organic nitrogen components and dry deposition.

The weighted mean concentration of ammonium and nitrate for each sampling site in 1983 is given in Fig. 1. The concentration of  $\text{NH}_4$  is roughly equal to that of nitrate. The concentration of nitrate + ammonium decreases from about  $40 \text{ } \mu\text{mol l}^{-1}$  to  $10 \text{ } \mu\text{mol l}^{-1}$  near the Norwegian border. The gradient in N—S direction is much less pronounced. This is most likely due to the location of the sources in northern Europe, from which most of the pollutants are carried with winds from the S—W to E sector, while winds from the west can bring abundant precipitation but little ammonium or nitrate.

The reason why concentration, and not deposition, is evaluated in Fig. 1 is the following. In order to obtain a concentration or deposition field from point measurements, it is desirable to know how spatial variation in the (annual) precipitation affects the concentrations measured, other factors being constant. Not much is known about this relation, but we have some measurements from the west coast of Sweden. Within some tens of kilometres, the annual precipitation can here vary with a factor of 1.5, while the mean concentration changes by only about 10 %. Thus, although the concentration (weighted annual mean value) is to some degree dependent on spatial variation in the precipitation, a better approximation is obtained by using constant concentration — rather than constant deposition.

Table 1 gives the results obtained when the data from land are extrapolated to the basins of the Gulf of Bothnia.

Precipitation chemistry measurements have been carried out since the mid 1950's in south-central Sweden and they may give an indication of the conditions over the Gulf of Bothnia. From the mid 1950's up to the early 1970's the concentration of  $\text{NH}_4$  and  $\text{NO}_3$  increased by a factor of 1.5—2. Since then no significant change has been observed, but it must be pointed out that there are large fluctuations from year to year and that the deposition/concentration during a single year can deviate from the long-term average by as much as 30 % (Monitor, 1983).

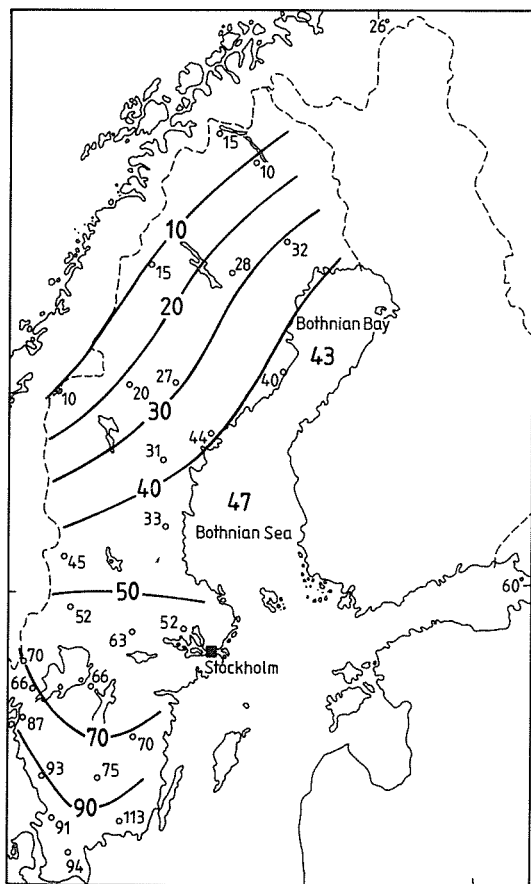


Fig. 1. Concentration of nitrate and ammonium in 1983. The numbers are the sum of yearly mean weighted concentration for ammonium and nitrate expressed in micro moles/l. Estimated average concentration for the Bothnian Bay and Bothnian Sea are also given.

Table 1. Annual input of inorganic nitrogen compounds ( $\text{NO}_3$  and  $\text{NH}_4$ ) to the Gulf of Bothnia in 1983.

| Sea area        | Area          | Precip.<br>amount  | Average<br>conc.         | Average<br>deposition               | Total                |
|-----------------|---------------|--------------------|--------------------------|-------------------------------------|----------------------|
|                 | $\text{km}^2$ | $\text{mm a}^{-1}$ | $\mu\text{mol N l}^{-1}$ | $\text{mmol N m}^{-2}\text{a}^{-1}$ | $\text{kg N a}^{-1}$ |
| Bothnian<br>Bay | 36 000        | 550                | 43                       | 24                                  | 12 000               |
| Bothnian<br>Sea | 70 000        | 550                | 47                       | 26                                  | 25 000               |

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## NUTRIENT DISCHARGE BY THE RIVER AURAJOKI TO THE ARCHIPELAGO SEA

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JUMPPANEN, K. & KOLEHMAINEN, O. 1986. Nutrient discharge by the River Aurajoki to the Archipelago Sea. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The amounts of phosphorus and nitrogen discharged by the River Aurajoki to the Archipelago Sea, and the non-point source loading in the river basin, were calculated for the years 1973–1983. The calculations of the material transport were based on daily stream flow observations and on water samples taken 16–45 times a year; in the spring sampling frequency was 4–8 samples per month, depending on the stream flow.

The transport of phosphorus to the sea area varied annually from 16 tons to 68 tons and that of nitrogen from 310 t to 770 t. The diffuse nutrient load was 53 kg (19–90) phosphorus and 785 kg (410–1050) nitrogen  $\text{km}^{-2} \text{a}^{-1}$ . The concentrations of phosphorus in the river water increased significantly during the observation period. This is due to changes in agriculture, e.g. increased fertilization. The effect of sampling frequency and calculation methods on the accuracy of nutrient transport values is discussed.

Index words: River monitoring, nutrient discharge, diffuse loading

### 1. INTRODUCTION

The amounts of nutrients and other material discharged by rivers to the Archipelago Sea greatly affect the quality of sea water and phytoplanktonic primary production. In 1979 the total annual nutrient load of the Archipelago Sea was estimated at 430 tons phosphorus and 6000 tons nitrogen; diffuse loading accounted for 330 tons, or 77 % of the phosphorus and 4700 tons (78 %) of the nitrogen (Jumppanen 1981).

The transport of nutrients by the River Aurajoki to the Archipelago Sea has been studied intensively for the years 1973–1983, with the aim of assessing the runoff of nutrients and diffuse loading, and their annual and seasonal variation. It was hoped to reveal possible trends in diffuse loading and in the quality of the river water.

### 2. MATERIAL AND METHODS

The calculations of nutrient transport were based on daily stream flow observations at the station at Halistenkoski (drainage area 730  $\text{km}^2$ , mean discharge  $7.1 \text{ m}^3 \text{ s}^{-1}$ , and on water samples taken 16–45 times a year during 1973–83 (Fig. 1). The sampling frequency was generally 4–8 samples per month in the spring and 2–4 samples per month in the autumn, depending on the stream flow. This frequency and timing of sampling gives reliable data concerning water quality and loading in such rivers as the Aurajoki (Kohonen 1982, Pekkarinen 1979).

The samples were analysed by the laboratories of the Turku water works, the Turku Water District and the Water Protection Association of South-West Finland, using the same standard methods

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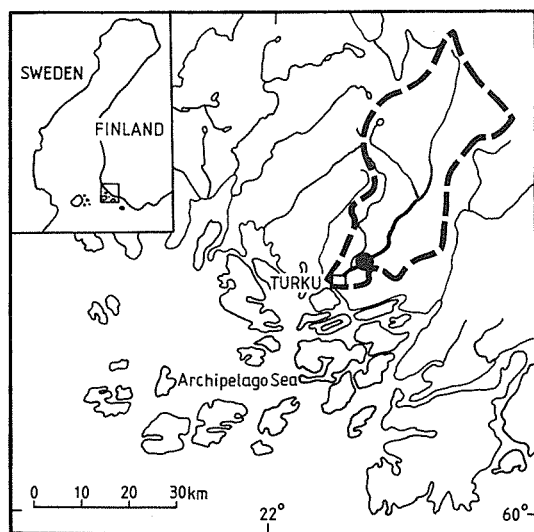


Fig. 1. The study area. River Aurajoki and its drainage area ( $F = 860 \text{ km}^2$ ,  $L = 0,0 \%$ ). ● = station Halistenkoski.

(Erkoma et al. 1977). The parameters determined were  $\text{BOD}_7$ , total phosphorus, soluble phosphorus (total phosphorus after centrifugation), total nitrogen, ammonium nitrogen, nitrate nitrogen and suspended solids.

The volume of nutrients discharged by the river was calculated for hydrological periods of one week to one month by multiplying the mean daily stream flow and nutrient concentration within each period. These values were summed to obtain the annual material transport. For the purpose of comparison, the nutrient loads were also calculated by multiplying the stream flow values and the annual mean concentrations (cf. Wartiovaara 1978, Kohonen 1982).

### 3. RESULTS AND DISCUSSION

#### 3.1 Discharge of nutrients

During 1973–83 the transport of phosphorus to the sea varied annually from 16 tons to 68 tons and that of nitrogen from 310 to 770 tons. The linear correlation between phosphorus transport and mean stream flow is statistically highly significant (Fig. 2). The proportion of the variance of the phosphorus discharge explained by the variation in the stream flow is 90 %.

In rivers like the Aurajoki, the variations in nutrient discharges are rapid and considerable. On average, 38 % of the total annual phosphorus load and 37 % of the nitrogen were discharged in April–May and 44 % of the phosphorus in the autumn. The nutrient discharge in the summer (June–August) was generally only 1–10 % of the annual total (Table 1).

During 1979–83 the discharge of soluble phosphorus was 26.4 tons or 54 % of total phosphorus, and inorganic nitrogen accounted for 64 % of total nitrogen.

When known point sources were excluded, the nutrient load was 19–90 kg ( $\bar{x} = 53$ ) phosphorus and 410–1050 kg nitrogen ( $\bar{x} = 785$ )  $\text{km}^{-1} \text{ a}^{-1}$ . This load is clearly greater than the corresponding values for small drainage basins (Kauppi 1979). In 1979 the diffuse load composed 82 % of the total annual phosphorus and 70 % of the nitrogen load.

The annual nutrient load in the Aurajoki seems to have increased towards the end of the observa-

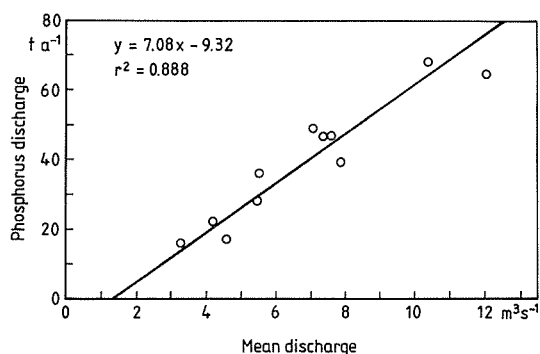


Fig. 2. Dependence of phosphorus transport of stream flow (MQ) in the River Aurajoki in 1973–1983.

Table 1. Seasonal variation of nutrient discharges in the River Aurajoki in 1974–1983 (% of mean annual discharge).

|        | Phosphorus % |       | Nitrogen % |       |
|--------|--------------|-------|------------|-------|
|        | Mean         | Range | Mean       | Range |
| Winter | 11           | 1–52  | 13         | 1–57  |
| Spring | 38           | 13–82 | 37         | 12–72 |
| Summer | 7            | 1–20  | 6          | 2–16  |
| Autumn | 44           | 7–81  | 44         | 8–63  |
| Sum    | 100          |       | 100        |       |

tion period, partly due to the increase in nutrient concentrations (Fig. 3). The annual mean concentrations of phosphorus weighted by daily stream flow increased highly significantly ( $r = 0.86^{***}$ ,  $n = 11$ ) during 1973–1983, while the nitrogen concentrations decreased ( $r = -0.78^{**}$ ).

The increasing concentrations of phosphorus in the river water are explained by changes in the use of agricultural land. Since 1960 the area of pasture land has decreased by one third; furthermore, fertilization increased greatly up to the mid 1970's

and the soluble phosphorus content of cultivated land therefore increased from  $6.7 \text{ mg l}^{-1}$  to  $14.9 \text{ mg l}^{-1}$  ( $r = 0.96^{**}$ , Fig. 4). Swine raising has been extended greatly since 1960 and is concentrated in big swineries. This has also contributed to the increasing trend of the phosphorus discharge to the sea.

The stream flow too seems to affect the concentrations of phosphorus in the river water (Fig. 3). The highest mean concentrations have been observed in the years 1979–1983, when — with the exception of 1983 — the stream flow has been at a relatively high level. Typical of the years 1979–1982 was also exceptionally abundant rainfall in the late autumn, when most of the fields are without vegetation.

### 3.2 Sampling frequency and estimates of diffuse loading

The sampling frequency considerably affects the accuracy of nutrient discharge records, especially in small drainage areas (e.g. Kohonen 1982). In order to observe the effect of the records on the figures for phosphorus transport, different methods were compared in dealing with the data for the years 1979 and 1982 (41–45 phosphorus samples, Table 2).

The phosphorus transport figures calculated from the mean values are 21 % lower than the figures calculated from the same data by summing the phosphorus discharges of short hydrological periods. Four samples a year (observations in March, May, August and October in running waters) gave figures that were 39–27 % lower than the calculation method used in this study and 12 samples a year figures that were 28–32 % lower. The observations closest to the middle of the month were chosen to represent the monthly concentration. The sampling frequency and computational approach therefore significantly affect material discharge values and estimates of diffuse loading. Due to rapid fluctuations in stream flow and the concentrations of phosphorus, frequent sampling is needed to give proportional representation in the material to major stream flow events, which often decisively affect material transport and means. For example the phosphorus discharge on 21 November 1980 was 4 200 kg, or 9 % of the total phosphorus discharge in 1980 (49 100 kg). As sampling frequency decreases, the mean concentrations and loads will be underestimated, as shown in Table 2, since extensive

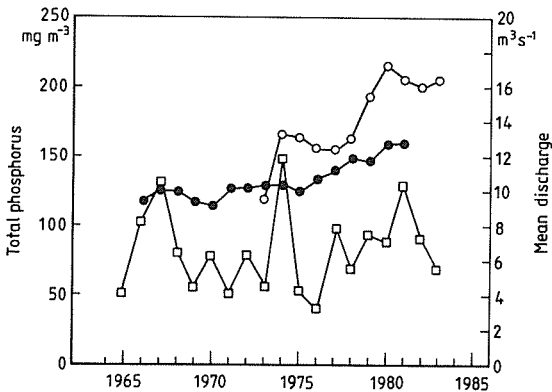


Fig. 3. Concentrations of phosphorus and stream flow in the River Aurajoki in 1966–1983.

○—○—○ mean value weighted by stream flow in 1973–1983 (minimum 15 samples a year)  
●—●—● moving average of three years (4 samples a year)  
□—□—□ mean discharge (MQ)

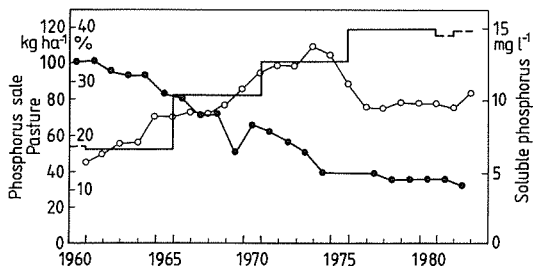


Fig. 4. Trends in agriculture in Finland Proper in 1960–1983.

○—○—○ sale of fertilizer phosphorus,  $\text{kg P}_2\text{O}_5$  per hectare (information given by Kemira Oy)  
— mean soluble phosphorus content of cultivated land (Kurki 1980, 1982)  
●—●—● proportion of pasture land in total cultivated land (Suomen virallinen tilasto 1960–82)

Table 2. Phosphorus transport ( $\text{kg a}^{-1}$ ) and the average phosphorus concentration in the River Aurajoki calculated in different ways from data in 1979 (mean discharge  $7.6 \text{ m}^3 \text{ s}^{-1}$ , 41 samples) and in 1982 (mean discharge  $7.4 \text{ m}^3 \text{ s}^{-1}$ , 45 samples).

|                                                                                         | Mean concentration<br>$\text{mg P m}^{-3}$ (SD) |         | Transport<br>$\text{kg P a}^{-1}$ |        |
|-----------------------------------------------------------------------------------------|-------------------------------------------------|---------|-----------------------------------|--------|
|                                                                                         | 1979                                            | 1982    | 1979                              | 1982   |
| 1. Summing of phosphorus discharges in short hydrological periods                       | 195                                             | 201     | 46 600                            | 46 900 |
| 2. Monthly sampling: concentration $\times$ monthly stream flow (12 samples)            | 151                                             | 149     | 36 100                            | 34 500 |
| 3. Annual arithmetical mean concentration $\times$ mean stream flow (41 and 45 samples) | 153(67)                                         | 160(63) | 36 700                            | 37 300 |
| 4. — " — (12 samples)                                                                   | 140(29)                                         | 136(21) | 33 500                            | 31 700 |
| 5. — " — (4 samples)                                                                    | 119(26)                                         | 147(17) | 28 500                            | 34 400 |

discharges in the spring and autumn are easily lost. The summing of material discharges in short hydrological periods probably gives highly representative values for material discharges and diffuse loading.

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## LOADS OF NUTRIENTS, ORGANIC MATTER AND SUSPENDED SOLIDS DISCHARGED BY THE RIVER KYRÖNJOKI INTO THE BOTHNIAN BAY

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MERILÄINEN, J.J. 1986. Loads of nutrients, organic matter and suspended solids discharged by the River Kyrönjoki into the Bothnian Bay. — Publications of the Water Research Institute, National Board of Waters, Finland. No. 68.

The loads discharged by the River Kyrönjoki were evaluated on the basis of the water quality and daily stream-flow data recorded at a station near the river mouth by the National Board of Waters. In the years 1979—1983 the average annual load of total phosphorus was 154 tons ( $32 \text{ kg km}^{-2} \text{ a}^{-1}$ ), 78 tons of which was phosphate phosphorus. Earlier (1972—1979), before the start-up of the sewage purification plant in the town of Seinäjoki, the annual phosphorus load was as high as 205 tons. The difference in phosphorus concentrations between these observation periods was significant, whereas the difference between the corresponding values for total nitrogen was not significant. The annual load of nitrogen was 2 680 tons. During the period 1971—1983 the annual load of organic carbon was 29 000 tons. A linear relationship generally existed between the stream flow and the concentration of suspended solids, but an exponential relationship was observed at times of rapidly increasing flood discharge. In normal years, with a mean discharge of  $40\text{--}50 \text{ m}^3 \text{ s}^{-1}$ , the annual load of suspended solids was between 39 600 and 66 400 tons ( $8.2\text{--}13.8 \text{ tons km}^{-2}$ ).

Index words: River monitoring, loading, nutrients, organic matter, suspended solids, Bothnian Bay

### 1. INTRODUCTION

The River Kyrönjoki, loaded by municipal waste water and the intensive agriculture practiced along the course of the river, drains into the southern Bothnian Bay ( $63^{\circ}20'N$ ,  $22^{\circ}00'E$ ). The amounts of nutrients and organic matter discharged by the river have been examined by Wartiovaara (1975, 1978) on the basis of fairly short observation periods.

In this study the loads discharged by the river were evaluated for nutrients (phosphorus, phosphate phosphorus, total nitrogen), organic matter

(COD, organic carbon) and suspended solids. The calculations were based on fairly comprehensive data on the water quality, recorded during the last 10—15 years by the National Board of Waters. It was also interesting to examine the effects on the nutrient load of the chemical treatment of the municipal effluents. The purification of the municipal waste water was notably increased during the last years of the 1970's, especially in 1979, when a water purification plant with chemical treatment started to operate in the town of Seinäjoki, the greatest population center in the river basin. Due to the low percentage of lakes (one

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percent of the drainage basin of about 4 900 km<sup>2</sup>), the variations in discharge are high, from below five to about 490 m<sup>3</sup>s<sup>-1</sup>. A positive relationship is known to exist between stream flow and the concentration of suspended solids. In this study it was possible to examine this relationship separately during the flood period, when the discharge increases rapidly.

## 2. MATERIAL AND METHODS

The water quality parameters studied (total phosphorus, phosphate phosphorus, total nitrogen, total organic carbon (TOC), chemical oxygen demand (COD<sub>Mn</sub>) and suspended solids) were determined on samples taken at the station at Skatila (63°08'N, 21°51'E) and analyzed at the laboratory of the Vaasa District Office of the National Board of Waters. The analytical methods were those in standard use at the National Board of Waters.

The concentrations of nutrients and organic matter do not correlate significantly with the stream flow in the River Kyrönjoki (Wartiovaara 1978). The average annual loads of phosphorus and nitrogen and the COD could therefore be estimated on the basis of the mean monthly values for each parameter calculated for the observation periods in question (Table 1). The load of TOC was estimated from the regression on the COD. The mean monthly discharges were obtained from the daily stream-flow data recorded at Skatila for the period 1971–1981 (excluding 1975, for which the data were incomplete). The mean discharge for this period was 45 m<sup>3</sup>s<sup>-1</sup>, while the mean value for the period 1931–1960 was 43 m<sup>3</sup>s<sup>-1</sup> (cf. Fig. 1).

The amounts of suspended solids discharged by the river were estimated separately for each year during the period 1977–1981 and also for 1972, in

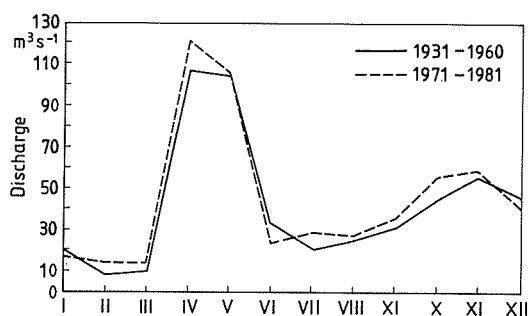


Fig. 1. The mean monthly discharge of the River Kyrönjoki at Skatila station in the years 1971–1981 and 1931–1960.

which year the observations on suspended solids were exceptionally numerous. The annual load was calculated from the mean monthly concentrations (on average two observations per month) and the discharge. During the flood period, however, when the discharge was increasing rapidly, use was made of regression values obtained from the equation calculated between the mean daily discharge and the observed concentration of suspended solids. The rapidly increasing flood discharge ( $Q_t$ ) for the River Kyrönjoki was defined here as follows:

$$Q_t = Q_r \text{ if } 1. \quad Q_t \geq 50 \text{ m}^3\text{s}^{-1}$$

$$2. \quad Q_t > Q_{t-1}$$

$$3. \quad Q_t \geq \frac{(Q_{t-1} + Q_{t-2} + Q_{t-3})}{3} + 10$$

$Q_t$  = mean discharge (MQ) for the observation day  
 $Q_{t-1}$ ,  $Q_{t-2}$ ,  $Q_{t-3}$  = MQ for the three preceding days

If the flood was two-peaked, only the first peak was examined.

## 3. RESULTS AND DISCUSSION

### 3.1 Nutrients

During the period 1972–1979 the mean concentration of total phosphorus was 149  $\mu\text{g l}^{-1}$ , but in 1979–1983 it was only 98  $\mu\text{g l}^{-1}$ . The difference was statistically significant (t-test,  $P \leq 0.001$ ). The decrease in the phosphorus concentrations was obviously caused by the increased treatment of the

Table 1. Water quality parameters studied.

|                                | Number of samples | Observation period |
|--------------------------------|-------------------|--------------------|
| Total phosphorus               | 257               | 1972–1983          |
| Phosphate phosphorus           | 120               | 1972–1983          |
| Total nitrogen                 | 254               | 1972–1983          |
| COD (KMnO <sub>4</sub> method) | 241               | 1971–1983          |
| Organic carbon (TOC)           | 88                | 1971–1983          |
| Suspended solids               | 234               | 1967–1981          |

municipal effluents. In 1978 only 41 percent of the effluents were treated chemically, while two years later the percentage was 96. During the flood time (April–May), when the diffuse load reaches its maximum, the difference in concentrations between the periods was not significant, whereas at other times, the decrease in concentrations was considerable (Fig. 2). In the years 1972–1979 the phosphorus concentration reached its maximum (about  $190 \mu\text{g l}^{-1}$ ) during February–April and in September; in 1979–1983 only the mean value of April was above  $130 \mu\text{g l}^{-1}$ . The average annual load for the period 1979–1983 was 154 tons. Half of it was discharged during the flood time (April–May) and 32 % in April alone (Fig. 3) Earlier (1972–1979) the load was 205 tons, 40 % of which was discharged during the flood time. Thus the decrease in the average load was as high as 25 %.

Phosphate phosphorus averaged 52 % of the total phosphorus, the proportion being at its maximum during January–March (63 %), and having its minimum in June (40 %). The average load of phosphate phosphorus in the years 1979–1983 was 78 tons  $\text{a}^{-1}$ .

The mean concentration of total nitrogen was  $1.81 \text{ mg l}^{-1}$ . The maximum values were recorded somewhat later in the year than for phosphorus, viz. during March–April and during November–December (Fig. 4, cf. Fig. 2). The difference in nitrogen concentrations between the periods 1972–1979 and 1979–1983 was not significant. The increase in the treatment of municipal effluents during the late 1970's had no discernible effect on the nitrogen load discharged by the river, the average load being 2 680 tons  $\text{a}^{-1}$ . Nearly half

(45 %) of the annual load was discharged during the flood time (Fig. 3).

The average discharge of total phosphorus is still high,  $32 \text{ kg km}^{-2}\text{a}^{-1}$ , though reduced from the average of  $43 \text{ kg}$  for 1972–1979. Phosphorus discharge values of  $5\text{--}8 \text{ kg km}^{-2}\text{a}^{-1}$  have been observed in forest and bog areas in natural state (e.g. Kauppi 1979, Kenttämies 1981). According to the loading values reported by Wartiovaara (1978),

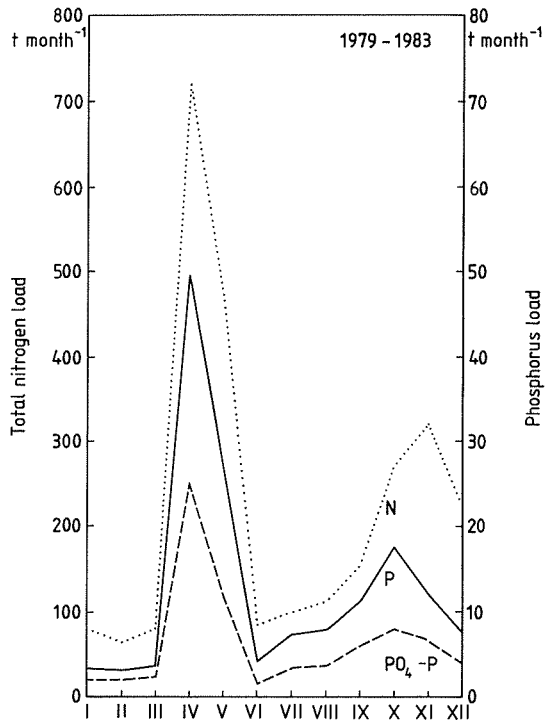


Fig. 3. The seasonal distribution of the nutrient load (tons per month) discharged by the Kyrönjoki.

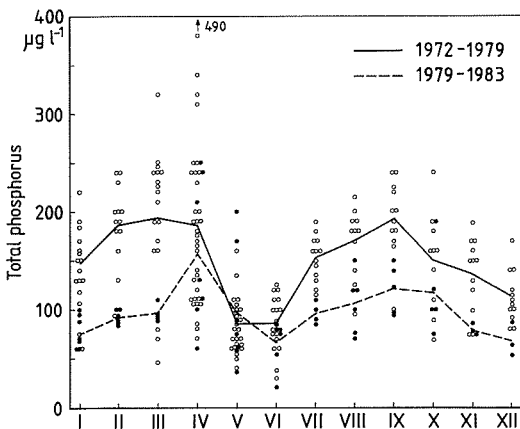


Fig. 2. The concentration of total phosphorus recorded in the years 1972–1979 ○ (open circles) and 1979–1983 ● (closed circles).

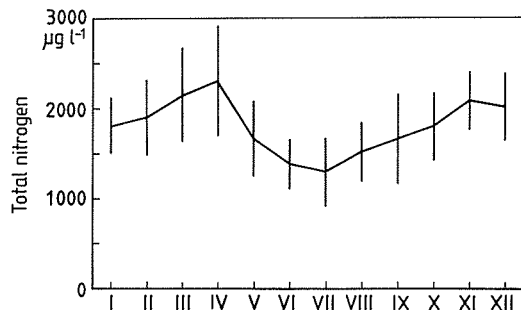


Fig. 4. The mean monthly concentration of total nitrogen ( $\pm$  SD = vertical bars) recorded between the years 1972–1983.

the average phosphorus discharge for the other large Finnish rivers (including the Tornionjoki) draining into the Bothnian Bay is about  $16 \text{ kg km}^{-2}\text{a}^{-1}$ , i.e. half of that observed here for the River Kyrönjoki.

### 3.2 Organic matter

The mean value for COD was  $27 \text{ mg O}_2 \text{ l}^{-1}$ , the monthly variation clearly being high (Fig. 5). The lowest values were observed during the flood period in April ( $22 \text{ mg O}_2 \text{ l}^{-1}$ ), while the maximum values ( $30\text{--}33$ ) occurred during July–November (Fig. 5). The regression of TOC on COD was significant (Fig. 6). The average annual COD was 37 000 tons and the average load of TOC 29 000 tons. The earlier calculations, based on rather limited material, are not far from these values (Table 2).

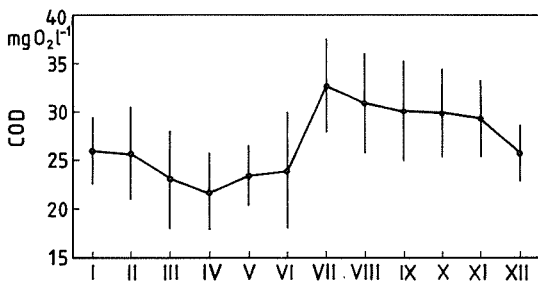


Fig. 5. The mean monthly COD values ( $\pm$  SD = vertical bars) in the years 1971–1983.

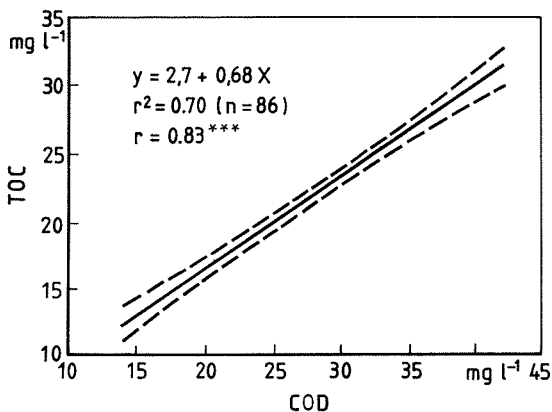


Fig. 6. The regression of TOC on COD in the Kyrönjoki. (Dashed lines indicate the 95 % confidence limits of the regression.)

Table 2. Average annual loads (tons  $\text{a}^{-1}$ ) of nutrients, and averages for COD and TOC during the periods 1970–1972 (Wartiovaara 1975) and 1974–1976 (Wartiovaara 1978) and the present estimates (see text).

|                      | Average annual loads (tons) |           |         |
|----------------------|-----------------------------|-----------|---------|
|                      | 1970–1972                   | 1974–1976 | present |
| Total phosphorus     | 180                         | 207       | 154     |
| Phosphate phosphorus |                             |           | 78      |
| Total nitrogen       | 2 100                       | 2 700     | 2 680   |
| COD                  | 32 000                      | 44 000    | 37 000  |
| TOC                  | 26 000                      | 36 800    | 29 000  |

### 3.3 Suspended solids

The equation  $y = 25.7 \cdot 1.0065^x$  expressed the relationship between the rapidly increasing flood discharge ( $Q_f$ ) and the concentration of suspended solids. At other times of the year the relationship was linear (Fig. 7). A strong relationship was clearly demonstrated during the flood in 1972 (Fig. 8). As the discharge rose, the concentration of suspended solids increased rapidly and as soon as the flood peak was reached, the concentration

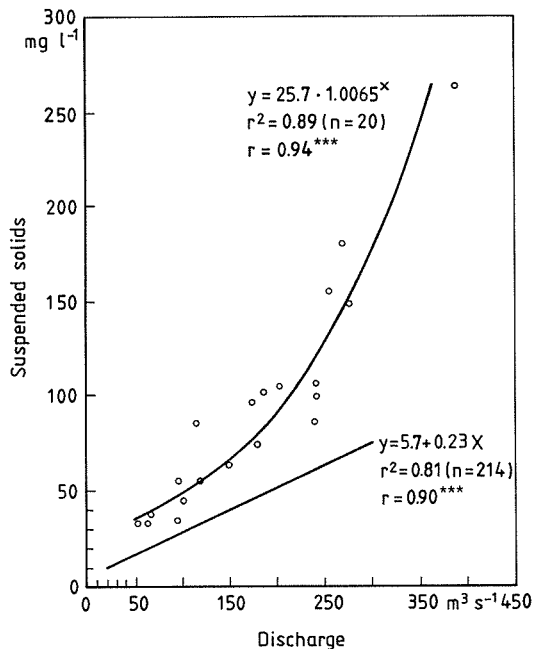


Fig. 7. The regression of the concentration of suspended solids on discharge in periods of rapidly increasing flood discharge (upper curve, observations marked with dots) and at other times (lower curve).

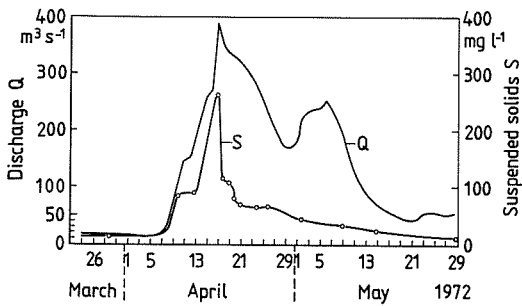


Fig. 8. The daily mean discharge (Q) and the concentration of suspended solids (s) recorded in the period 24.3.—29.5.1972 at Skatila station.

Table 3. The loads of suspended solids (tons  $a^{-1}$ ) during normal (1), dry (2) and wet years (3). The loads for 1974 and 1976 calculated from runoff values given by Wartiovaara (1978).

| Year    | Mean discharge | Annual load | Portion discharged during April—May (%) |
|---------|----------------|-------------|-----------------------------------------|
| 1. 1972 | 49             | 66 400      | 75                                      |
| 1979    | 46             | 39 600      | 38                                      |
| 2. 1976 | 25             | 18 300      | ..                                      |
| 1978    | 27             | 16 700      | 48                                      |
| 1980    | 31             | 30 700      | 65                                      |
| 3. 1974 | 83             | 89 000      | ..                                      |
| 1977    | 53             | 68 300      | 80                                      |
| 1981    | 65             | 83 900      | 64                                      |

began to decrease. It is also notable that during the second rise in the discharge (early May, Fig. 8) there was no increase in the concentrations of suspended solids. In other words, the river bed had been washed out by the first peak of the flood.

If the years examined here are classified according to their mean discharge, the following groups can be distinguished. 1) Normal years ( $MQ = 40\text{--}50 \text{ m}^3\text{s}^{-1}$ ), 2) dry years ( $MQ < 40$ ) and 3) wet years ( $MQ > 50$ ). The annual load of

suspended solids was 39 600—66 400 tons in normal years, 16 700—30 700 tons in dry and 68 000—89 000 tons in wet years (Table 3). The load discharged during the flood time in April—May was often more than half of the total, once being even 80 % of the yearly total. In 1979 the flood was exceptionally low, and the load discharged during April—May of that year (38 % of the total) can be considered a very low value for the flood time.

## ACKNOWLEDGEMENTS

I am grateful to Miss Eeva Ranta and Mr. Pertti Sevola, who provided valuable assistance during the work.

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## DISCHARGES OF NUTRIENTS AND ORGANIC MATTER TO THE GULF OF BOTHNIA BY FINNISH RIVERS IN 1968—1983

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PITKÄNEN, H. 1986. Discharges of nutrients and organic matter to the Gulf of Bothnia by Finnish rivers in 1968—1983. Publications of the Water Research Institute, National Board of Waters, Finland. No. 68.

Values for the transport of phosphorus, nitrogen and organic matter (COD) by the Finnish rivers discharging to the Gulf of Bothnia were calculated for the years 1968—1983 and the factors affecting the transport were evaluated. The total mean transport of phosphorus was  $3\,300\text{ t a}^{-1}$  ranging from  $2\,000\text{ t a}^{-1}$  to  $5\,300\text{ t a}^{-1}$ . The corresponding value for nitrogen was  $49\,000\text{ t a}^{-1}$  ( $34\,000\text{—}73\,000\text{ t a}^{-1}$ ) and for COD  $960\,000\text{ t a}^{-1}$  ( $610\,000\text{—}1\,600\,000\text{ t a}^{-1}$ ). The river load was concentrated in the northeastern Bothnian Bay and the central Bothnian Sea. The nutrient input to the Archipelago Sea was high in relation to the small drainage area of this sea area due to the high field percentage and low lake percentage in the drainage basins. Increasing trends prevailed in the transport values during the study period. The trends were mostly due to the increasing trends in the stream flow. In many of the coastal rivers, however, nutrient transport seemed to increase more than could be expected from the changes in stream flow. This was assumed to be caused by increased diffuse loading. The effect of the decrease in the waste water loading was clearly observable in the transport of materials by the Rivers Kokemäenjoki and Oulujoki during the late 1960s and the 1970s.

Index words: river transport, nutrients, COD, organic matter, diffuse loading, physiographic factors, Finnish rivers, Gulf of Bothnia, Baltic Sea

### 1. INTRODUCTION

Rivers are responsible for the main part of the nutrient and organic matter input in the Gulf of Bothnia (Ahl et al. 1977, Enckell-Sarkola et al. 1984). Their contributions of natural and anthropogenic substances have not only direct effects on the sea areas in the vicinity of their estuaries, but also indirect effects on the ecological balance of the outer sea.

The transport of materials by Finnish rivers has been evaluated regularly since 1970. This subject was studied, however, as early as the 1910s and 1920s (Holmberg 1935). The materials transported by the rivers draining to the Gulf of Bothnia have been studied, for example, by Wartiovaara (1975, 1978), Ahl et al. (1977), Siira (1983, 1984), Ahl (1984), Enckell-Sarkola et al. (1984), Alasaarela and Heinonen (1984), Ekholm (1986), Jumppanen and Kolehmainen (1986) and Meriläinen (1986).

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The aim of this study was to examine the geographic distributions and temporal trends of the inputs of phosphorus, nitrogen and organic matter by the Finnish rivers discharging to the Gulf of Bothnia.

## 2. MATERIALS AND METHODS

The data on the 24 rivers studied here (Fig. 1) originated from the water quality and hydrological monitoring carried out by the National Board of Waters, Finland. The sampling and analyses were performed by the Water District Offices, using standard methods (Erkomaa 1977, National Board of Waters 1982a, b). Organic matter was analyzed as chemical oxygen demand,  $\text{COD}_{\text{Mn}}$  (SFS 3036, 1981). The improvement of the chemical procedures may have a decreasing effect on the time series of nutrient concentrations when the level is low.

The frequency of sampling varied rather widely during 1968–1983. In general, after 1973–1974 the larger rivers were monitored at least once a month and the smaller rivers 4 to 12 times a year. In the late 1960s and early 1970s the frequencies were lower, usually 4 to 8 times a year. The stream flow was measured daily.

The annual transport of materials was obtained by summing the monthly values, which were calculated by multiplying the mean monthly concentration by the mean monthly flow. Missing monthly concentrations were replaced with the mean values of the corresponding year. The calculated transport values were extrapolated to correspond to the whole drainage area of the river. Extrapolation was also used in order to obtain transport estimates for the whole Finnish catchment area of the Gulf of Bothnia and the subareas of the Gulf (the Bothnian Bay, the Bothnian Sea and the Archipelago Sea).

The calculation method may cause substantial errors in cases in which the sampling frequency was low (e.g. Walling and Webb 1985, Jumppanen and Kolehmainen 1986). Thus, in many of the rivers studied the calculated transport values for individual years must be considered merely indicative. Extrapolation causes errors in those rivers where the station for monitoring water quality lies well above the estuary (River Karvianjoki and River Aurajoki during 1972–1983). For the Archipelago

Sea the extrapolated estimates of total river transport are probably quite rough, because the rivers studied cover only 40 % of the total drainage area; for the Bothnian Sea and the Bothnian Bay the values are 87 % and 94 %, respectively.

## 3. RESULTS

### 3.1 Stream flow

The mean yearly discharge of the Finnish rivers to the Gulf of Bothnia was  $2\,200\text{ m}^3\text{ s}^{-1}$ , ranging from  $1\,600\text{ m}^3\text{ s}^{-1}$  in 1976 to  $3\,100\text{ m}^3\text{ s}^{-1}$  in 1981 (Tables 1 and 2). About  $1\,800\text{ m}^3\text{ s}^{-1}$  (81 %) was discharged to the Bothnian Bay. The values for the Bothnian Sea and the Archipelago Sea were  $340\text{ m}^3\text{ s}^{-1}$  (15 %) and  $78\text{ m}^3\text{ s}^{-1}$  (4 %) respectively. The four large inland rivers (Oulujoki, Iijoki, Kemijoki and Tornionjoki) discharging to the NE Bothnian Bay were alone responsible for 61 % ( $1\,300\text{ m}^3\text{ s}^{-1}$ ) of the whole stream flow.

### 3.2 Input of phosphorus

The mean total input was  $3\,300\text{ t a}^{-1}$  (min.  $2\,000\text{ t a}^{-1}$ , max.  $5\,200\text{ t a}^{-1}$ , Tables 1 and 2). The shares of the three sea areas, from north to south, were  $2\,100\text{ t a}^{-1}$  (64 %),  $830\text{ t a}^{-1}$  (26 %) and  $320\text{ t a}^{-1}$  (10 %). Thus the relative inputs to the Bothnian Sea and the Archipelago Sea were much higher and the relative input to the Bothnian Bay much smaller than in the case of stream flow.

In the catchment area of the Bothnian Bay, the inputs from the four large inland rivers and those of the coastal rivers were almost equal,  $1\,100\text{ t a}^{-1}$  and  $970\text{ t a}^{-1}$ , respectively, although the stream flows were 3:1. The inputs from the Rivers Kyrönjoki ( $150\text{ t a}^{-1}$ ) and Kalajoki ( $130\text{ t a}^{-1}$ ) were especially large in relation to their stream flow and drainage area.

In the Bothnian Sea the catchment area of the River Kokemäenjoki contributed  $530\text{ t a}^{-1}$ , which corresponded to 17 % of the total river input. The load carried by the rivers to the Archipelago Sea ( $320\text{ t a}^{-1}$ , 10 %) was very high in relation to the stream flow, due to the high phosphorus concentrations of the river waters in SW Finland.

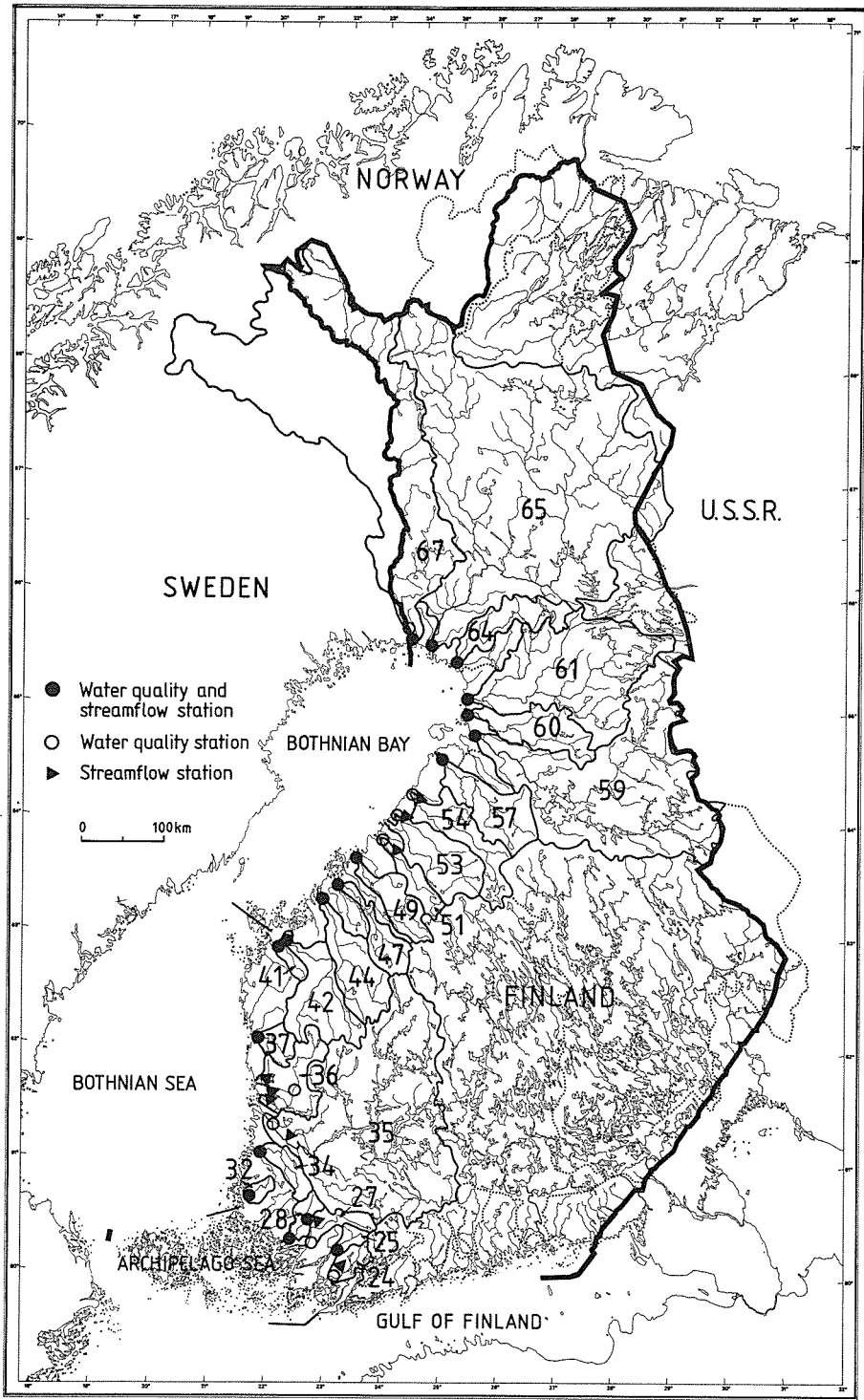


Fig. 1. The study area and sampling stations. Names and drainage areas of the rivers are presented in Table 1.

Table 1. Mean stream flow (MQ) and mean transport of phosphorus (P), nitrogen (N) and organic matter (COD) by the studied rivers in 1968—1983. Mean values in brackets indicate uncertainty due to missing data. Rivers 32 and 47 discharge to the sea via a coastal freshwater basin.

| River basin     | Drainage area<br>km <sup>2</sup> | MQ<br>m <sup>3</sup> s <sup>-1</sup> | Transport of materials |                        |                                                         | N:P  |
|-----------------|----------------------------------|--------------------------------------|------------------------|------------------------|---------------------------------------------------------|------|
|                 |                                  |                                      | P<br>t a <sup>-1</sup> | N<br>t a <sup>-1</sup> | COD<br>10 <sup>3</sup> t O <sub>2</sub> a <sup>-1</sup> |      |
| 24 Kiskonjoki   | 1 000                            | 9.2                                  | 14                     | 270                    | 3.1                                                     | 19   |
| 25 Uskelanjoki  | 600                              | 5.2                                  | 27                     | 340                    | 2.0                                                     | 13   |
| 27 Paimionjoki  | 1 100                            | 8.8                                  | 45                     | 600                    | 3.6                                                     | 13   |
| 28 Aurajoki     | 890                              | 7.9                                  | 40                     | 580                    | 4.2                                                     | 15   |
| 32 Sirppujoki   | 430                              | 4.7                                  | 9.3                    | 340                    | 1.9                                                     | 37   |
| 34 Eurajoki     | 1 300                            | 9.6                                  | 19                     | 400                    | 3.1                                                     | 21   |
| 35 Kokemäenjoki | 27 100                           | 222                                  | 530                    | 7 900                  | 130                                                     | 15   |
| 36 Karvianjoki  | 3 400                            | 34.1                                 | 87                     | 1 100                  | 19                                                      | 13   |
| 37 Isojoki      | 1 100                            | 15.1                                 | 38                     | 510                    | 11                                                      | 13   |
| 41 Laihianjoki  | 480                              | (3.7)                                | (9.6)                  | (320)                  | (2.4)                                                   | (32) |
| 42 Kyrönjoki    | 4 900                            | 42.7                                 | 150                    | 2 400                  | 35                                                      | 16   |
| 44 Lapuanjoki   | 4 100                            | 36.0                                 | 98                     | 1 700                  | 28                                                      | 17   |
| 47 Ähtävänjoki  | 2 000                            | 14.8                                 | 16                     | 300                    | 6.8                                                     | 19   |
| 49 Perhonjoki   | 2 700                            | 22.6                                 | 46                     | 650                    | 16                                                      | 14   |
| 51 Lestijoki    | 1 300                            | 12.2                                 | 24                     | 350                    | 9.5                                                     | 15   |
| 53 Kalajoki     | 4 200                            | 40.4                                 | 130                    | 1 600                  | 35                                                      | 12   |
| 54 Pyhäjoki     | 3 700                            | 32.7                                 | 62                     | 860                    | 24                                                      | 14   |
| 57 Siikajoki    | 4 400                            | 43.1                                 | 120                    | 1 100                  | 34                                                      | 9.2  |
| 59 Oulujoki     | 22 900                           | 258                                  | 220 <sup>2)</sup>      | 3 900 <sup>2)</sup>    | 96 <sup>2)</sup>                                        | 18   |
| 60 Kiiminkijoki | 3 900                            | 42.5                                 | 58 <sup>2)</sup>       | 840 <sup>2)</sup>      | 25 <sup>2)</sup>                                        | 14   |
| 61 Iijoki       | 14 400                           | 169                                  | 170                    | 2 200                  | 71                                                      | 13   |
| 64 Simojoki     | 3 200                            | 35.8                                 | 34                     | 590                    | 17                                                      | 17   |
| 65 Kemijoki     | 51 400                           | 531                                  | 430                    | (6 900)                | 170                                                     | 16   |
| 67 Tornionjoki  | 35 000 <sup>1)</sup>             | 372                                  | 280                    | (4 100)                | 91                                                      | 15   |

<sup>1)</sup> the value is not exact because of bifurcation

<sup>2)</sup> mean values for 1970—1983

Table 2. Mean discharge of water (Q), phosphorus (P), nitrogen (N) and organic matter (COD) by the Finnish rivers to the different sea areas of the Gulf of Bothnia in 1968—1983.

| Sea area                    | Transport of materials              |                        |                        |                                         |
|-----------------------------|-------------------------------------|------------------------|------------------------|-----------------------------------------|
|                             | Q<br>m <sup>3</sup> s <sup>-1</sup> | P<br>t a <sup>-1</sup> | N<br>t a <sup>-1</sup> | COD<br>t O <sub>2</sub> a <sup>-1</sup> |
| Archipelago Sea             | 78                                  | 320                    | 4 500                  | 32 000                                  |
| Bothnian Sea                | 340                                 | 820                    | 13 000                 | 190 000                                 |
| R. Kokemäenjoki             | 220                                 | 530                    | 7 900                  | 130 000                                 |
| coastal rivers              | 120                                 | 290                    | 5 100                  | 60 000                                  |
| Bothnian Bay                | 1 750                               | 2 100                  | 32 000                 | 730 000                                 |
| inland rivers <sup>1)</sup> | 1 330                               | 1 100                  | 17 000                 | 430 000                                 |
| coastal rivers              | 420                                 | 970                    | 15 000                 | 300 000                                 |
| Gulf of Bothnia             | 2 170                               | 3 300                  | 49 000                 | 960 000                                 |

<sup>1)</sup> R. Oulujoki, R. Iijoki, R. Kemijoki and R. Tornionjoki

### 3.3 Input of nitrogen

The mean total input was  $49\,000\text{ t a}^{-1}$  (min.  $34\,000\text{ t a}^{-1}$ , max.  $73\,000\text{ t a}^{-1}$ , Tables 1 and 2) and the shares of the three sea areas, from north to south, were  $32\,000\text{ t a}^{-1}$  (65 %),  $13\,000\text{ t a}^{-1}$  (26 %) and  $4\,500\text{ t a}^{-1}$  (9 %).

The geographic distribution of the nitrogen inputs was close to that of phosphorus. The mean ratio of the nitrogen to phosphorus input for the rivers discharging to the Bothnian Bay, Bothnian Sea and Archipelago Sea was 15, 16 and 14, respectively (Table 1). Only in some small coastal rivers did the ratio deviate markedly from these values.

### 3.4 Input of organic matter

The mean total river load of organic matter measured as COD was  $960\,000\text{ t a}^{-1}$  (min.  $610\,000\text{ t a}^{-1}$ , max.  $1\,600\,000\text{ t a}^{-1}$ , Tables 1 and 2). The geographic distribution differed rather widely from that of the nutrients. On an average,  $730\,000\text{ t a}^{-1}$  (77 %) was discharged to the Bothnian Bay. The Bothnian Sea and the Archipelago Sea received  $190\,000\text{ t a}^{-1}$  (20 %) and  $32\,000\text{ t a}^{-1}$  (3 %), respectively.

The four large rivers discharging to the NE Bothnian Bay contributed  $420\,000\text{ t a}^{-1}$ , which corresponded to 44 % of the whole input from the Finnish rivers. This proportion is higher than in the case of nutrients, but still clearly lower than in the case of stream flow.

The proportion from the coastal rivers discharging to the Bothnian Bay ( $300\,000\text{ t a}^{-1}$ , 31 %) was almost equal to the corresponding nutrient input.

In the drainage area of the Bothnian Sea, the proportion contributed by the River Kokemäenjoki ( $130\,000\text{ t a}^{-1}$ , 14 %) was slightly smaller than in the case of nutrients. The rivers discharging to the Archipelago Sea had low COD values and their relative contribution of COD was only one third of the corresponding value for nutrients.

### 3.5 Trends in 1968–1983

In general, both stream flow and transport of nutrients and organic matter increased during 1968–1983 (Figs. 2–5).

The increase in the transport of materials was pronounced in most of the coastal rivers dis-

charging to the Bothnian Bay, especially as regards nitrogen. Among the inland rivers, the average transport values increased only in the Kemijoki and Tornionjoki. No trends were observed in the Iijoki and the Oulujoki showed a strong decreasing trend.

Despite the increasing trend in the stream flow of the Kokemäenjoki, the discharges of phosphorus and organic matter clearly decreased till the late 1970s. In the case of nitrogen, however, an increasing trend was evident.

The stream flow to the Archipelago Sea increased only slightly, as did also the transport of nitrogen and organic matter. The transport of phosphorus, however, increased very clearly. This trend was especially pronounced in the River Kiskonjoki, but an increase was recorded in other rivers as well.

## 4. DISCUSSION

### 4.1 Dependence of nutrient and COD transport on stream flow

The transport of phosphorus, nitrogen and COD to the Gulf of Bothnian obtained in this study is in general somewhat higher than that calculated in earlier studies with yearly mean values (Wartiovaara 1975, 1978, Enckell-Sarkola et al. 1984). The difference is greatest in the rivers discharging to the Bothnian Bay, where the relative changes in concentrations between high-flow and low-flow periods are also greatest. The calculated values of the present study probably still underestimate real transport due to sparse sampling, especially during high-flow periods (e.g. Walling and Webb 1985).

A strong positive dependence of the transport and concentration of materials on stream flow has been shown by Wartiovaara (1975, 1978) for many of the Finnish rivers. In fact, in some cases the transport of materials can be satisfactorily predicted from stream flow alone. The dependence varies, however, both seasonally and annually due to changes in meteorological and physiographic factors in the drainage basins (Alasaarela and Heinonen 1984, Ekholm 1986, Jumppanen and Kolehmainen 1986, Meriläinen 1986).

River discharge is generally largely concentrated in April–May. In the coastal rivers (drainage area  $< 5\,000\text{ km}^2$ ) roughly 50 % of the yearly stream flow takes place at that time (National Board of

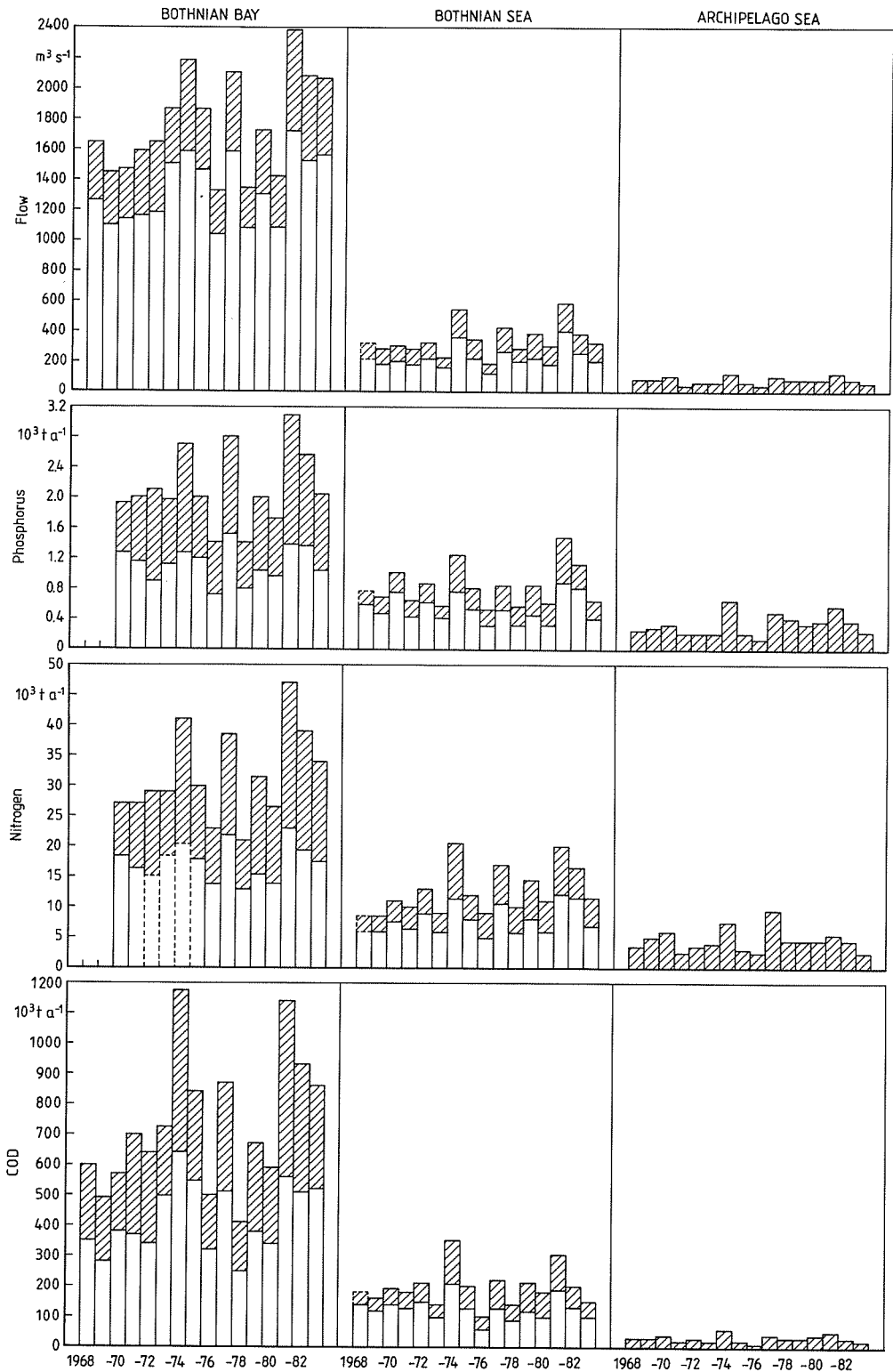


Fig. 2. The discharge of water, phosphorus, nitrogen and COD to the different sea areas in 1968–1983. Shaded and open bars indicate coastal and inland rivers, respectively. Dotted bars indicate uncertainty due to missing data.

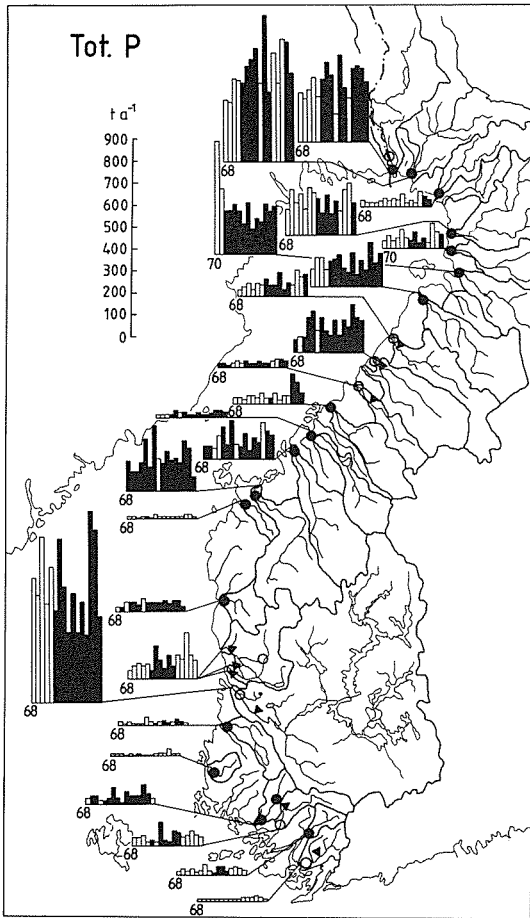


Fig. 3. The transport of phosphorus by the studied rivers in 1986–1983. Black bars indicate  $\geq 10$  yearly samples, open bars indicate  $< 10$  yearly samples.

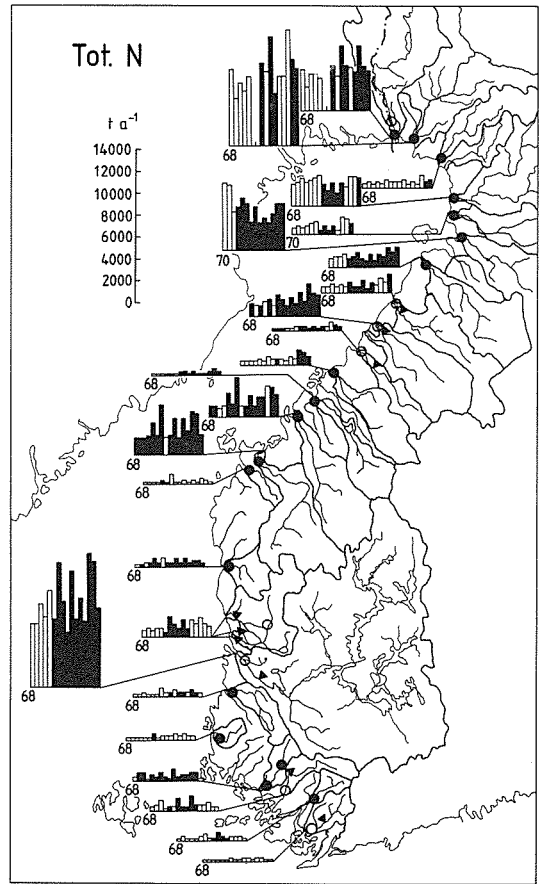


Fig. 4. The transport of nitrogen by the studied rivers in 1968–1983. Black bars indicate  $\geq 10$  yearly samples, open bars indicate  $< 10$  yearly samples.

Waters 1983). As regards nutrients and COD, the corresponding percentage is usually even higher, due to the positive correlation between concentration and stream flow (cf. Siira 1983, 1984, Ekholm 1986, Jumppanen and Kolehmainen 1986, Meriläinen 1986). In the rivers Tornionjoki, Kemijoki and Iijoki, discharging to the NE Bothnian Bay, the transport of materials also has a pronounced maximum in spring. In the Oulujoki and Kokemäenjoki, on the other hand, the yearly distribution is fairly even, due to the high proportion of lakes in the drainage area and regulation of stream flow (cf. Alasaarela and Heinonen 1984).

#### 4.2 Dependence of nutrient and COD transport on physiographic factors

In addition to the stream flow, the proportions of lakes and fields in the drainage basins and the waste water input clearly affect the transport of materials in Finnish watercourses (Laaksonen 1970, Wartiovaara 1978).

In the 17 coastal rivers with drainage areas between 500 and 5 000 km<sup>2</sup> the transport of nutrients and organic matter clearly decreases with increasing lake percentage. The lake percentage explains 76 % and 57 % of the variation in the

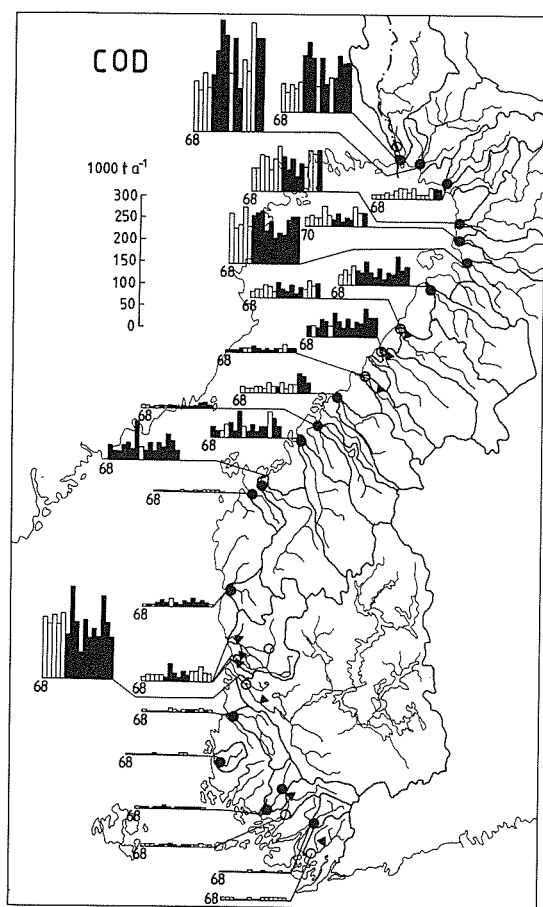


Fig. 5. The transport of COD by the studied rivers in 1968–1983. Black bars indicate  $\geq 10$  yearly samples, open bars indicate  $< 10$  yearly samples.

transport of phosphorus and nitrogen, respectively, when loglog-transformation is applied (Fig. 6). COD transport seems to decrease almost linearly with increasing lake percentage, when the drainage basins with high ( $\geq 30\%$ ) and low ( $\leq 15\%$ ) percentages of peatland are considered separately.

The correlation between nutrient transport and the percentage of fields is clearly positive in the different lake percentage groups (Fig. 7). An increase of 5% in the field area seems to increase the transport of total phosphorus by about 3 kg km<sup>-2</sup>a<sup>-1</sup> and the transport of total nitrogen by about 60 kg km<sup>-2</sup>a<sup>-1</sup>. The values are on the same

level or even higher than those obtained by Kauppi (1984) in small experimental drainage basins. Ahl (1977) has obtained similar values in the catchment area of Lake Vänern. The dependence of COD transport on the field percentage is weak because the COD values are more strongly affected by the percentages of lake area and peatland.

In the large inland rivers discharging to the NE Bothnian Bay, the average transport of phosphorus, nitrogen and COD is 8–12 kg P km<sup>-2</sup>a<sup>-1</sup>, 120–170 kg N km<sup>-2</sup>a<sup>-1</sup> and 2.6–4.9 t O<sub>2</sub> km<sup>-2</sup>a<sup>-1</sup>, respectively. For the Kokemäenjoki the corresponding values are 20 kg P km<sup>-2</sup>a<sup>-1</sup>, 290 kg N km<sup>-2</sup>a<sup>-1</sup> and 4.7 t O<sub>2</sub> km<sup>-2</sup>a<sup>-1</sup>. The dependence on the lake percentage is weaker than in the case of small coastal rivers.

The positive correlation between nutrient transport and the field percentage is clear. Except in the drainage basin of the Kokemäenjoki, however, the field percentage is very low ( $\leq 2\%$ ). In the Kokemäenjoki and Oulujoki the mean transport values are clearly elevated due to the heavy waste water loading during the late 1960s and early 1970s. In the Kokemäenjoki the high field percentage and low lake percentage in the lower part of the drainage basin also increase the transport values.

#### 4.3 Dependence of nutrient and COD transport on diffuse loading

The increasing trends in the transport values of most of the rivers studied during 1968–1983 are very clearly connected with simultaneous increasing trends in stream flow. Increasing trends are also observed in rivers where the sampling frequency has remained almost unchanged (see materials and methods). A general increase has been observed in the nutrient concentrations of the coastal rivers as well (Laaksonen and Malin 1984).

An increase in the sampling frequency evidently has an increasing effect on the time series of yearly transport values (Walling and Webb 1985, Jumpanen and Kolehmainen 1986). In those cases where the sparse sampling earlier gave too much weight to the high-flow period in spring, i.e. mainly in the rivers discharging to the NE Bothnian Bay, the effect can also be the opposite.

It seems that the increasing flow and the changes in the sampling frequency cannot explain the whole increase in the transport of phosphorus by the rivers discharging to the Archipelago Sea and possibly also by the coastal rivers discharging to the Bothnian Sea (Fig. 8). It is possible that this

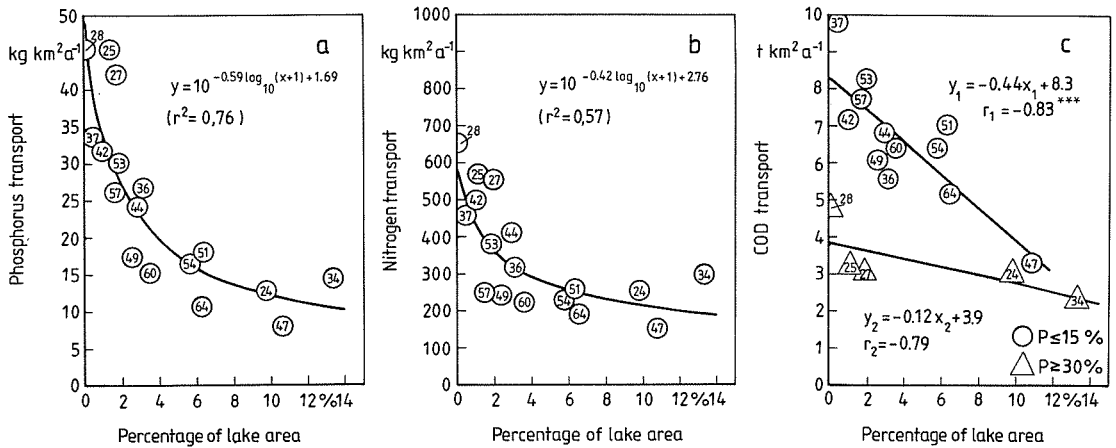


Fig. 6. The dependence of annual transport of materials by the coastal rivers on the percentage of lake area in their drainage basins, a, phosphorus, b, nitrogen, c, COD. Confidence levels: \*95%, \*\*99%, \*\*\*99.9%.  
 △ percentage of peatland ( $P \leq 15$  (in c))  
 ○ percentage of peatland ( $P \geq 30$  (in c))

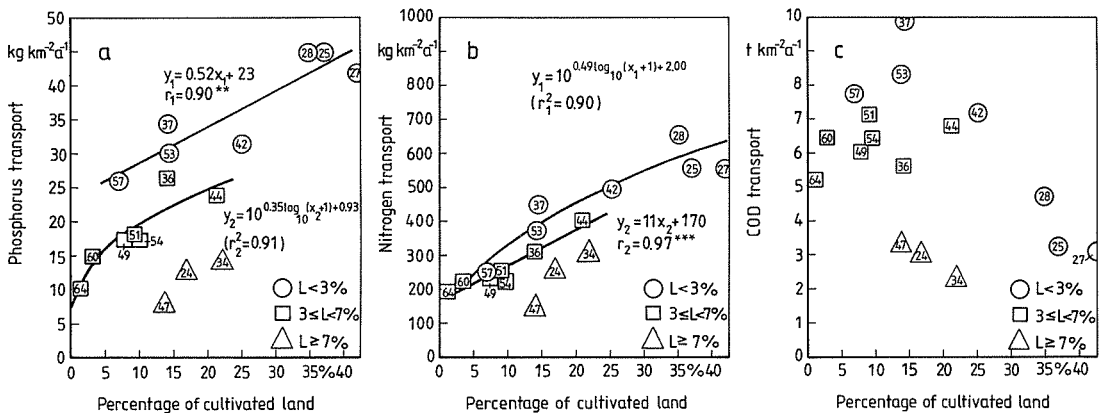


Fig. 7. The dependence of annual material transport by the coastal rivers on the percentage of field area in their drainage basins, a, phosphorus, b, nitrogen, c, COD. Confidence levels: \*95%, \*\*99%, \*\*\*99.9%.  
 ○ percentage of lake area ( $L < 3$ )  
 □ percentage of lake area ( $3 \leq L < 7$ )  
 △ percentage of lake area ( $L \geq 7$ )

change is also due to increased diffuse loading from agriculture (Mansikkaniemi 1982, Jumppanen and Kolehmainen 1986).

The direct loading of the coastal rivers is in general relatively low and the changes in it cannot explain the increase in the concentrations. In the Kokemäenjoki and Oulujoki, on the other hand, the decreasing trends in the phosphorus transport are clearly connected with decreased direct loading of these rivers during the 1970s (Oravainen 1982, Laaksonen and Malin 1984, Myllymaa and Riihimäki 1984).

Transport of nitrogen by the coastal rivers seems to have increased in relation to stream flow in the coastal catchment area of the Bothnian Bay and also to some extent in the coastal catchment area of the Bothnian Sea (Fig. 9b, c). According to Kauppi (1984), the diffuse loading of nitrogen has increased in the small agricultural research basins during 1968–1975 in parallel with the increased use of fertilizers.

The increase in the runoff-related transport of nitrogen by the coastal rivers discharging to the Bothnian Bay seems to have been roughly 40 kg

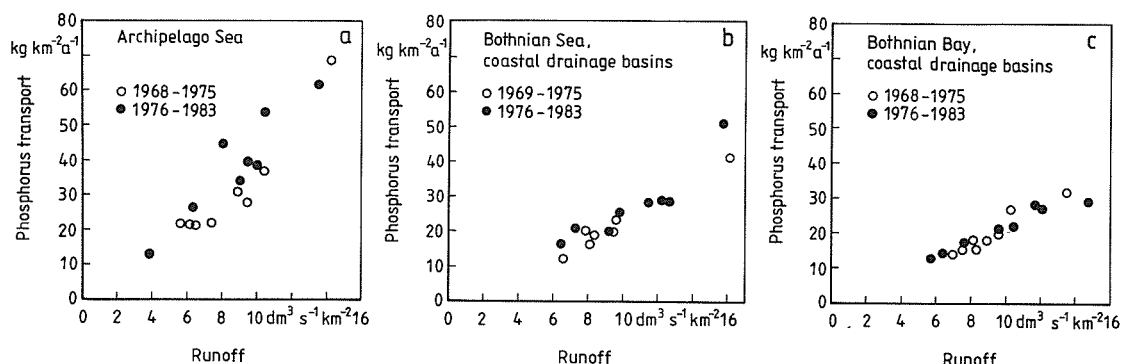


Fig. 8. The dependence of annual phosphorus transport on runoff in the coastal rivers in 1968–1975 and 1976–1983; a, drainage area of the Archipelago Sea, b, coastal drainage area of the Bothnian Sea, c, coastal drainage area of the Bothnian Bay.

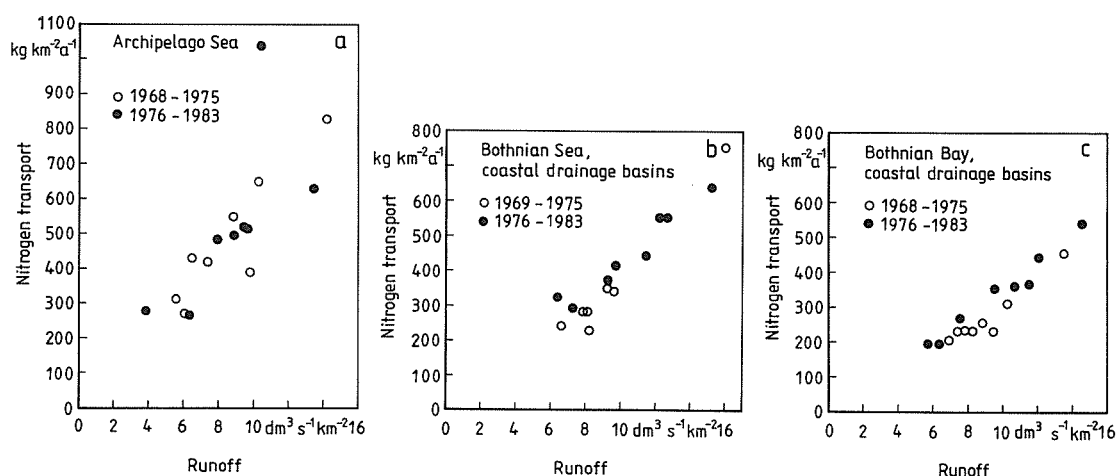


Fig. 9. The dependence of annual nitrogen transport on runoff in the coastal rivers in 1968–1975 and 1976–1983; a, drainage area of the Archipelago Sea, b, coastal drainage area of the Bothnian Sea, c, coastal drainage area of the Bothnian Bay.

$\text{km}^{-2}\text{a}^{-1}$  between the periods 1968–1975 and 1976–1983 (Fig. 9c). This increase can partly be explained by the increased agricultural use of nitrogen fertilizers. The average increase in the use of nitrogen has been about  $1\,500\text{ kg field}\cdot\text{km}^{-2}\text{a}^{-1}$  between 1968–1975 and 1976–1983 (Rikkihappo Oy 1969, 1970, Kemira Oy 1971–1984). If about 10 % of fertilizer is leached from the fields (Kauppi 1984), an average increase of  $20\text{ kg N km}^{-2}\text{a}^{-1}$  is obtained for the coastal drainage area of the Bothnian Bay when the average field percentage of 12 is used.

In the drainage area of the Archipelago Sea the level of nitrogen transport has not increased consistently between the two periods (Fig. 9a).

This accords with the fact that the use of fertilizers was highest during 1972–1975, subsequently decreasing somewhat.

The atmospheric deposition of nitrogen in southern and central Finland has increased by about  $200\text{ kg km}^{-2}\text{a}^{-1}$  during the 1970s (Järvinen 1986). The total increase in the deposition is thus close to that in the use of nitrogen fertilizers in the coastal catchment area of the Bothnian Bay. This may partly explain the increased river transport of nitrogen there and also in the rivers discharging to the Bothnian Sea, in whose drainage basins the use of fertilizers has been fairly steady during the 1970s. In the drainage area of the Archipelago Sea, the role of agriculture is so dominant that the

changes in the atmospheric load are probably not clearly reflected in the present data.

In the larger rivers the changes in nitrogen transport are satisfactorily explained by changes in stream flow or in direct loading. In the strongly loaded Kokemäenjoki, however, the effects of the different factors are difficult to separate from each other.

The changes in the yearly COD transport are mainly explained by the changes in the stream flow. As in the case of nutrients, the variation in the chemical sampling affects the calculations. The decreasing trends in the transport of COD by the Oulujoki and the Kokemäenjoki are the result of decreased direct loading of these rivers in the 1970s. Comparison of the results from the 1920s and 1970s shows that the yearly transport of COD by the nonloaded rivers has remained fairly constant (Alasaarela and Heinonen 1984).

## 5. SUMMARY

Values for the transport of phosphorus, nitrogen and organic matter (COD) by the Finnish rivers discharging to the Gulf of Bothnia were calculated for the years 1968–1983 and the factors affecting the transport were evaluated. The data originated from the water quality and hydrological monitoring carried out by the National Board of Waters, Finland.

The total mean transport of phosphorus was  $3\,300\text{ t a}^{-1}$  ranging from  $2\,000\text{ t a}^{-1}$  to  $5\,300\text{ t a}^{-1}$ . The corresponding value for nitrogen was  $49\,000\text{ t a}^{-1}$  ( $34\,000$ – $73\,000\text{ t a}^{-1}$ ) and for COD  $960\,000\text{ t a}^{-1}$  ( $610\,000$ – $1\,600\,000\text{ t a}^{-1}$ ). About 35 % of the input of nutrients and 45 % of the input of organic matter was concentrated in the NE Bothnian Bay. The share of the River Kokemäenjoki, discharging to the middle part of the Bothnian Sea, was about 15 % as regards both nutrients and COD. River input relative to the area of the drainage basins was greatest in the rivers discharging to the Archipelago Sea as regards nutrients and in the rivers discharging to the southern Bothnian Bay as regards organic matter. The smallest values were observed in the inland rivers discharging to the NE Bothnian Bay.

The greater the percentage of lakes in the coastal drainage basins, the smaller was the transport of nutrients and organic matter by the corresponding rivers. In contrast, the greater the

percentage of fields, the greater was the transport of nutrients. The transport of organic matter increased with the percentage of peatland in the drainage basin.

The strong variation between the yearly transport values was mainly due to the variation in the stream flow. Increasing trends observed in the material transport were chiefly due to large discharges during the latter half of the study period. Increased diffuse loading caused by intensified agriculture and increased atmospheric loading probably affected the river transport of nutrients in the coastal drainage basins.

Transport of phosphorus, nitrogen and COD by the River Oulujoki and transport of phosphorus and COD by the River Kokemäenjoki decreased during the late 1960s and the 1970s due to the decreased direct waste water loading of these rivers.

## ACKNOWLEDGEMENTS

I would like to thank Väinö Malin, M.Sc. and Petri Ekholm, M.Sc. for their help in processing the data. Further I would like to thank Professor Reino Laaksonen, Dr Jyrki Wartiovaara and Dr Lea Kauppi for valuable comments of the manuscript.

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## PHOSPHORUS AND NITROGEN BALANCE OF A COASTAL BAY

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PITKÄNEN, H., NIEMI, J., SARKKULA, J. & SANDMAN, O. 1986. Phosphorus and nitrogen balance of a coastal bay. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Annual material balances of phosphorus, nitrogen and suspended solids were estimated in a shallow enclosed bay of 53 km<sup>2</sup> situated in the coast of the Gulf of Finland. The data obtained included water quality data, discharge and current measurements, echo-soundings and chemical sediment analyses. The material balance of a substance was calculated from the equation  $\Delta m/\Delta t = \Sigma \text{ input} - \Sigma \text{ output} - \text{retention}$ . The results showed that the phosphorus retention of the bay was about 40 t in 1983, which corresponded to 26 % of the annual input from rivers. The budget of suspended solids was also positive, but retention of nitrogen did not appear to occur. Echo-soundings and chemical analyses of sediments indicated that accumulation of sediments occurred in the central part of the bay and in the vicinity of the river mouths. In addition to the rivers, the outer sea seemed also to be a substantial source of nutrients, especially in summer and autumn.

Index words: Nutrient balance, phosphorus, nitrogen, suspended solids, estuary, coastal dynamics, sedimentation

### 1. INTRODUCTION

A substantial part of the nutrients entering the Baltic Sea is discharged by rivers (Pawlak 1980). The calculations of nutrient transport in different parts of the Baltic Sea are based on discharge and concentration figures for the lower reaches of rivers. However, the quantitative and qualitative changes of river borne loads in coastal waters have been studied in much less detail.

Outside the Baltic Sea, the balances and behaviour of nutrients in estuaries have been actively investigated (e.g. Burton and Liss 1976, Morris et al. 1978, Olausson and Cato 1980). The dynamics and transport of nutrients have been studied in the estuaries and neighbouring waters of the northern Baltic Sea, e.g. by Halme (1944),

Niemi (1973), Karlgren and Ljungström (1975), Wartiovaara (1978), Alasaarela (1980) and Heikkilä (1986).

The objective of this work was to estimate the annual nutrient budget of a coastal bay with river input and the amount of nutrients leaving to the outer sea.

The study area, the Bay of Ahvenkoskenlahti and the estuaries of the Rivers Kymijoki and Taasianjoki, is situated on the coast of the Gulf of Finland (Fig. 1). This kind of shallow, semi-enclosed area is typical of the Finnish coast of the Baltic Sea. The area of the bay is 53 km<sup>2</sup>, its mean depth is 4.5 m and maximum depth 16 m. The bay receives fresh water from the Rivers Kymijoki (MQ 140 m<sup>3</sup>s<sup>-1</sup>) and Taasianjoki (MQ 5 m<sup>3</sup>s<sup>-1</sup>). No waste water is discharged directly to the bay.

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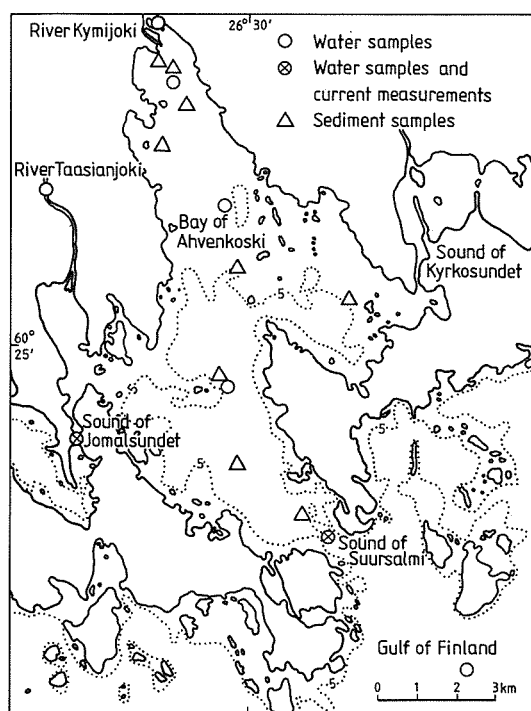


Fig. 1. The study area and sampling stations.

## 2. MATERIALS AND METHODS

The material consisted of water quality data, current measurements, bottom sediment analyses and echo-soundings gathered in 1983 (Fig. 1). Daily discharge measurements were available from the River Kymijoki. Discharge of the River Taasianjoki was estimated on the basis of data from a neighbouring river.

Water samples were taken with a Ruttner sampler once or twice a month. Sediments were sampled with Mackereth and Ekman samplers. Chemical analyses of water and sediment were carried out with standard methods used by the National Board of Waters, Finland (Erkomaa et al. 1977). The oxygen consumption of sediment was determined by the methods of Sandman and Simola (1983). Current measurements were performed in 1983 in both icecovered (3.2.—25.3.) and open water (11.8.—29.9.) periods with recording current meters (Aanderaa RCM4). Echo-soundings were made with an Atlas Echograph 58 with a sounding frequency of 39 kHz. The echograms were interpreted as described by Haavisto and Kukkonen (1975).

Mass balance was calculated from the equation  $\Delta m/\Delta t = \Sigma \text{input} - \Sigma \text{output} - S$ , where  $m$  = amount of a substance in the water body,  $t$  = time and  $S$  = net exchange of the substance between sediment and water (water and atmosphere).

Material inputs from rivers were calculated by multiplying the mean monthly discharge by the mean monthly concentration of a substance. In addition, nutrient input was calculated on the basis of the relationship between water quality and discharge, when this was statistically significant. The differences in the results obtained by these two methods were negligible.

Material flows in the sounds were calculated separately for spring, (April—May), summer (June—September), autumn (October—November) and winter (December and January—March). Current measurements carried out in the winter could be applied directly to the whole ice-covered period. Statistical relationships were calculated between current speed, wind components and sea level variation for the estimation of currents during the open-water period. The thicknesses of inflowing and outflowing water layers in the sounds were estimated on the basis of salinity and conductivity measurements. For each season the mean concentrations at depths of 1 m and 5—6 m were used as the concentrations of the inflowing and outflowing waters, respectively.

The seasonally averaged amount of a substance in the water phase of the bay ( $m$ ) was calculated from volume-stressed substance concentrations and deposition from air according to the values presented by Järvinen and Haapala (1980).

## 3. RESULTS

### 3.1 Currents

The results showed that the bay was strongly stratified during the ice-covered season. In the Sound of Suursalmi a fresh water layer, about 2 m thick, flowed to the sea below the ice cover. Another layer, just below the first one and also 2 m thick flowed to the sea and contained about 20 % river water and 80 % sea water. A corresponding amount on sea water entered the bay below a depth of 4 m. Variations in the sea level caused only slight, transient changes in this general pattern.

During the open-water season the water discharged from the bay contained 10—80 % sea

water depending on the winds, sea level and river discharges. These factors explained 61 % and 42 % of the current variation in the upper and lower layers, respectively (Fig. 2). The thickness of the upper layer varied from 3 to 6 m, being greatest in spring.

In the Sound of Jomalsundet the net discharge was towards the sea both in winter and in the open-water season. During the open-water period the equation explained 82 % of the variation in currents. The net discharge was, however, only a small percentage of that in the Sound of Suursalmi. In the Sound of Kyrkosundet the effective cross-section of the sound is so small that the net discharge through it was considered to be negligible.

### 3.2 Water quality

In the winter the river water flowed under the ice in a strictly separate layer. Concentrations of total phosphorus and suspended solids ( $> 1 \mu\text{m}$ ) in this layer varied in the range of  $15\text{--}20 \text{ mg m}^{-3}$  and

$1.3\text{--}3 \text{ g m}^{-3}$ , respectively, while in the underlying sea water, with a salinity of  $4\text{‰}$ , they were  $35\text{--}40 \text{ mg m}^{-3}$  and  $0.5\text{--}4 \text{ g m}^{-3}$ , respectively (Figs. 3 and 4). The concentrations of total nitrogen in the upper fresh water layer and in the lower sea water layer were approximately  $600 \text{ mg m}^{-3}$  and  $500 \text{ mg m}^{-3}$ , respectively.

The ice cover of the bay melted in the beginning of April and simultaneously the material input from rivers, increased considerably. The surface water of the bay still consisted almost entirely of fresh water. Between the fresh and sea water layers was a boundary layer where effective mixing occurred. In the fresh water layer the concentrations of phosphorus ( $25\text{--}50 \text{ mg m}^{-3}$ ), nitrogen ( $400\text{--}800 \text{ mg m}^{-3}$ ) and suspended solids ( $4\text{--}10 \text{ g m}^{-3}$ ) were higher than those in the sea water layer ( $20\text{--}45 \text{ mg m}^{-3}$  for P,  $400\text{--}600 \text{ mg m}^{-3}$  for N and  $3\text{--}6 \text{ g m}^{-3}$  for suspended solids).

In the summer, discharges of fresh water decreased and mixing of the two water masses occurred in the surface layer of the whole bay. The salinity of the outflowing water was about  $2\text{‰}$  and that of the inflowing sea water about  $5\text{‰}$ . In early September, when the fresh water discharge

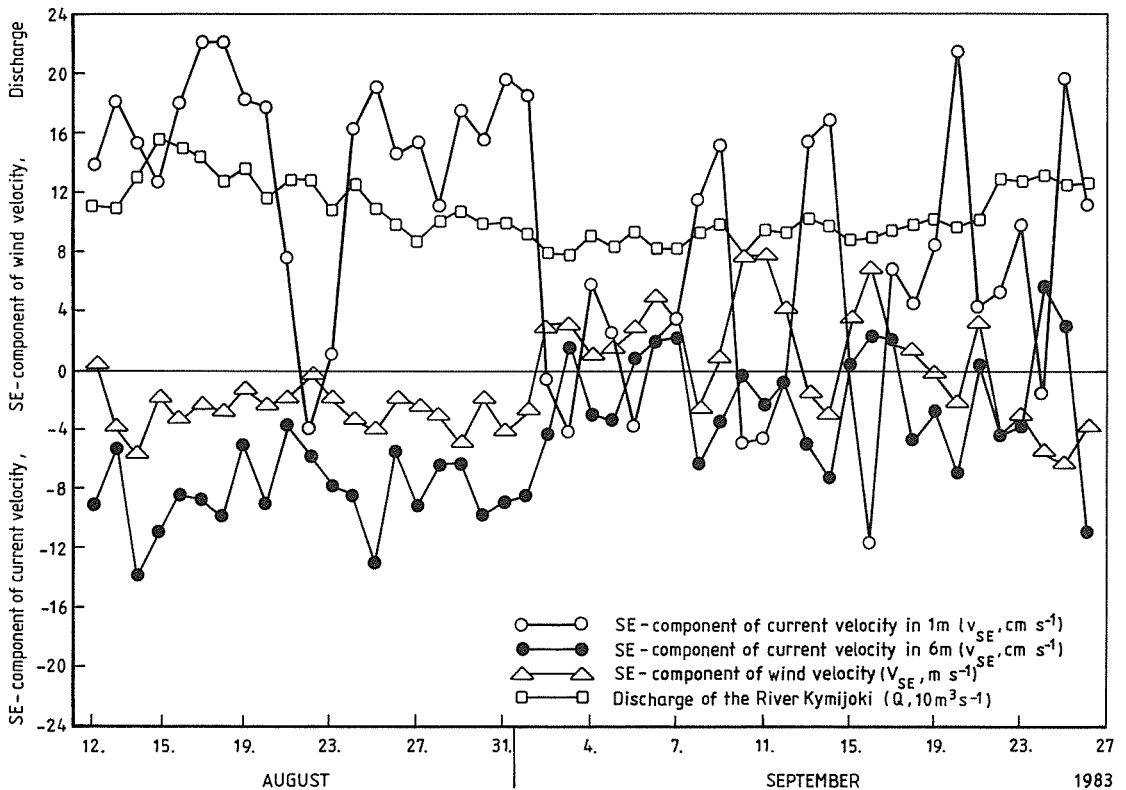


Fig. 2. Currents in the Sound of Suursalmi.

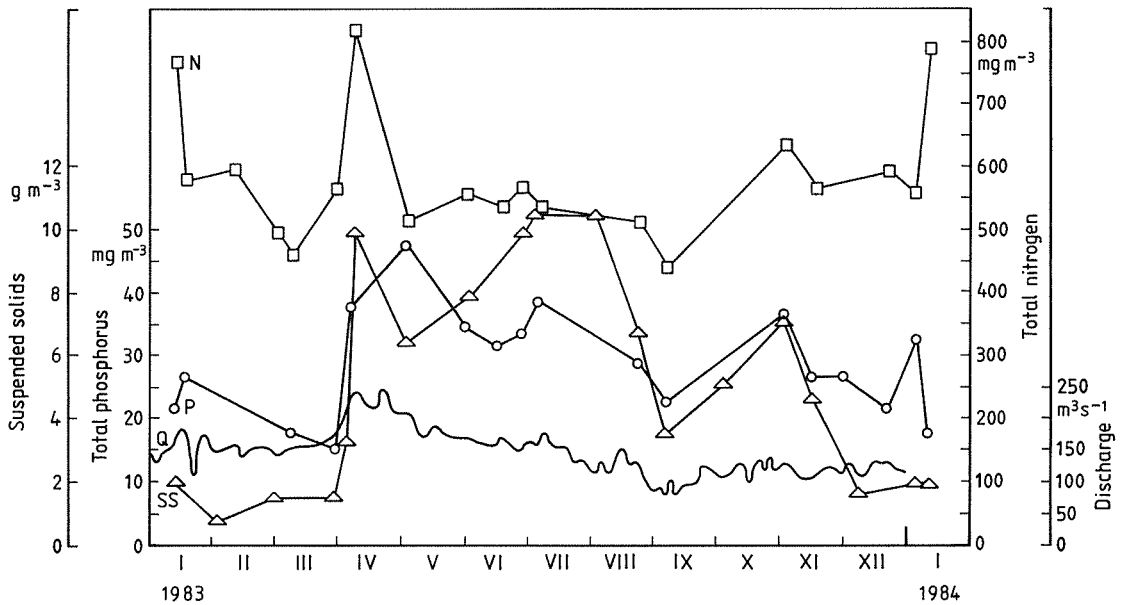


Fig. 3. Discharge (Q) and concentrations of total phosphorus (P), total nitrogen (N) and suspended solids (SS) in the mouth of the River Kymijoki in 1983.

reached its minimum, mixing was almost complete in the bay. During the summer months the halocline and thermocline were situated at the same depth, which stabilized the stratification.

Phosphorus and nitrogen concentrations in the River Kymijoki decreased during the summer with the decreasing discharge (Fig. 3). However, the concentration of suspended solids increased until the end of July and decreased sharply thereafter. In both fresh and sea water layers of the bay a clear concentration gradient existed towards the river mouth. Concentrations in the surface layer varied from 15 to 40 mg m<sup>-3</sup> for phosphorus, from 350 to 600 mg m<sup>-3</sup> for nitrogen and from 3 to 11 g m<sup>-3</sup> for suspended solids, while those in the sea water layer varied from 20 to 45 mg m<sup>-3</sup> for phosphorus, from 300 to 500 mg m<sup>-3</sup> for nitrogen and from 2 to 14 g m<sup>-3</sup> for suspended solids.

In the autumn the discharges and currents were about the same as in the late summer. The water mass was almost isothermic and mixing of the two water layers was more effective than in summer, occurring practically from surface to bottom. The salinities of the outflowing and inflowing waters were 3 ‰ and 5 ‰, respectively. The concentrations of the measured substances had increased both in the river water and in the outflowing surface water compared with the concentrations in late summer. In the sea water layer, only the concentration of phosphorus was clearly higher than the level observed in summer.

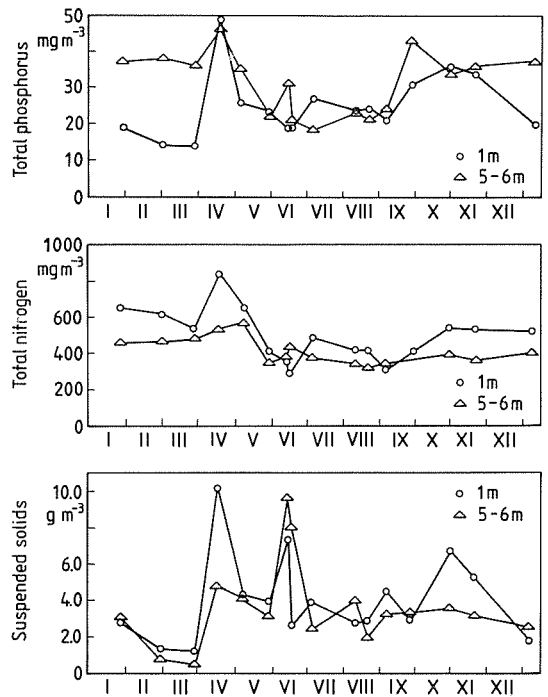


Fig. 4. Concentrations of total phosphorus, total nitrogen and suspended solids in the Sound of Suursalmi in 1983.

### 3.3 Sediments

The bottom sediments in the deep part ( $> 10$  m) of the bay were found to be homogeneous reduced muddy clay containing sulphides and gas. In the two uppermost centimetres the percentage of dry matter was about 18 % and the loss-on-ignition from dry matter about 14 %. The concentration of total phosphorus in the surface sediment was about  $3 \text{ mg g}^{-1}$  dry weight. The bottoms of the shallower areas were found to be both sedimentation and erosion bottoms with varying particle sizes. The phosphorus concentration of the surface sediment varied from 1 to  $6 \text{ mg g}^{-1}$  dry weight.

The nature of the sediments did not allow exact dating by  $^{210}\text{Pb}$  analysis. It was assumed that the clearest changes in the sediment variables in the sediment depth of 10–20 cm coincided with the first half of the 20th century. This leads to the conclusion that the sedimentation rate of the deep ( $> 10$  m) part has been ca.  $5 \text{ mm a}^{-1}$ . On the basis of these figures it was calculated that about 2000 t of dry matter, about 300 t of organic matter and about 6 t of phosphorus have settled yearly in the deep part of the bay, whose area is about  $3 \text{ km}^2$ . Naturally, these estimates are approximate.

The nature of the sediments in the shallow parts of the bay varied both locally and temporally. The currents caused erosion and redeposition of sediments. Erosion bottoms were common. Echo-soundings indicated that retention of the substances discharged by the rivers occurred in the areas in the vicinity of the river mouths and in certain central areas of the bay where the water depth is 5–10 m. These sediments are generally of non-depositional type and difficult to date because there are discontinuities in sedimentation succession.

### 3.4 Material budget

Calculations based on water quality, current and discharge measurements indicated that the total input of phosphorus was 40 t greater than the output in 1983 (Table 1). The rate of retention was greatest in spring and smallest in autumn.

The annual water phase input and output of nitrogen seemed to be in balance, while no net retention seemed to occur (Table 2).

The input of suspended solids was 7 300 t greater than the output (Table 3). Net retention occurred in spring and summer, while in autumn and winter output prevailed over input. Calculated from the river input, the retention of phosphorus

and suspended solids was 26 % and 23 %, respectively. The true figure for suspended material is probably even greater because autochthonous production could not be included in the calculations. When the net input from the sea was taken into account as a separate factor, the percentage of the retained part of the total input was 16 % for phosphorus and 17 % for suspended solids. The nutrient input from the sea was most substantial in autumn.

## 4. DISCUSSION

Because of similarities in the annual variation of the concentrations of phosphorus and suspended solids, it is reasonable to assume that the phosphorus entering via rivers and settling more or less permanently in the bay is mostly associated with suspended matter. Nitrogen seems to act differently in this respect and according to the results of 1983 no net sedimentation occurred.

The main factors affecting net sedimentation of phosphorus and suspended solids in the spring and summer seasons are probably increased river input in spring, substantial input from the sea in summer and increased biological activity and relatively calm wind conditions during both seasons.

In autumn, wind-induced currents become stronger and the output of suspended solids exceeds the input. Due to increased input from the sea, the input of phosphorus is somewhat greater than the output. Limited mixing under the ice cover in winter, low concentration of substances in river water and low biological activity generate circumstances in which nitrogen seems to flow through the bay unaffected while part of the phosphorus is retained.

According to calculations of concentrations, currents and discharges, the deepest area of the bay retains only a small part (ca. 15 %) of the sedimented phosphorus. This is due to the small area suitable for permanent sedimentation and the rather low content of phosphorus in the sediment. The river mouths and certain relatively shallow parts (5–10 m) of the bay are probably the main areas for sedimentation. Because of great annual variations in sedimentation conditions, the average net sedimentation may differ substantially from the value calculated from the present data.

The results cannot be applied directly to other Finnish estuaries because of great local variations. Heikkilä (1986) has estimated on the basis of

Table 1. The budget of total phosphorus (t) in 1983 (retention =  $\Sigma$  input —  $\Sigma$  output  $\mp \Delta m$ ).

|                | Winter | Spring | Summer | Autumn | Whole year |
|----------------|--------|--------|--------|--------|------------|
| Input; rivers  | + 35   | + 51   | + 46   | + 24   | + 156      |
| sounds         | + 10   | + 20   | + 29   | + 31   | + 90       |
| air            | 0      | + 1    | + 1    | + 0    | + 2        |
| $\Sigma$       | + 45   | + 72   | + 76   | + 55   | + 248      |
| Output; sounds | — 36   | — 58   | — 62   | — 52   | — 208      |
| $\Delta m$     | + 2    | — 1    | + 1    | — 2    | 0          |
| Retention      | + 11   | + 13   | + 15   | + 1    | + 40       |

Table 2. The budget of total nitrogen (t) in 1983 (retention =  $\Sigma$  input —  $\Sigma$  output  $\mp \Delta m$ ).

|                | Winter  | Spring  | Summer  | Autumn | Whole year |
|----------------|---------|---------|---------|--------|------------|
| Input; rivers  | + 920   | + 770   | + 740   | + 450  | + 2 880    |
| sounds         | + 120   | + 280   | + 420   | + 320  | + 1 140    |
| air            | 0       | + 30    | + 20    | + 10   | + 60       |
| $\Sigma$       | + 1 040 | + 1 080 | + 1 180 | + 780  | + 4 080    |
| Output; sounds | — 1 100 | — 1 200 | — 1 100 | — 770  | — 4 170    |
| $\Delta m$     | — 20    | — 10    | + 40    | — 10   | 0          |
| Retention      | — 80    | — 130   | + 120   | 0      | — 90       |

Table 3. The budget of suspended solids (t) in 1983 (retention =  $\Sigma$  input —  $\Sigma$  output  $\mp \Delta m$ ).

|                | Winter  | Spring   | Summer   | Autumn  | Whole year |
|----------------|---------|----------|----------|---------|------------|
| Input; rivers  | + 2 700 | + 12 000 | + 12 000 | + 4 900 | + 31 600   |
| sounds         | + 580   | + 2 200  | + 5 500  | + 2 900 | + 11 180   |
| $\Sigma$       | + 3 280 | + 14 200 | + 17 500 | + 7 800 | + 42 780   |
| Output; sounds | — 4 200 | — 12 000 | — 11 000 | — 8 300 | — 35 500   |
| $\Delta m$     | + 670   | — 850    | + 440    | — 260   | 0          |
| Retention      | — 250   | + 1 350  | + 6 940  | — 760   | + 7 280    |

sedimentological studies that 60–70 % of the suspended solids and greater flow variations than Kyrönjoki are retained in the estuarine area. This is considerably more than the figure (ca. 25 %) obtained in this study. The River Kyrönjoki has a more sheltered estuary, a higher concentration of suspended solids and greater flow variation than the River Kymijoki, which may explain this difference. Furthermore both of these studies are based on rather limited material, which can generate large errors in calculations. For example, the role of autochthonous material in the sedimentation of suspended matter may be substantial (Kuparinen et al. 1984).

Morphometrical features suggest that the estuaries in the Finnish coasts of the Gulf of Finland, the Archipelago Sea, the Southern Bothnian Sea and the Northern Quark may have relatively good sediment retention capacities, whereas in the more open coasts of the Northern Bothnian Sea and the Bothnian Bay the capacity is probably poorer. However, in addition to morphometry several other factors affect sedimentation (e.g. Morris et al. 1978, Postma 1980). The capacity of sediments to retain nutrients is of importance and depends mainly on redox potential, pH and temperature (e.g. Aston 1980, Rekolainen 1982).

Due to the limited amount of data available, the

calculations may contain substantial errors and these results should therefore be considered tentative. Because of great variations between estuarine ecosystems, intensive measurements of water quality, currents and sedimentation are needed for more accurate estimates of nutrient budgets.

## ACKNOWLEDGEMENTS

We are grateful to the personnel of the Water District Office of Kymi, especially to Ilppo Kettunen, limnologist, for help in taking and analysing water samples, to the personnel of the Research Laboratory of the National Board of Waters, Finland, and the Water District Laboratory of Helsinki for analysing the water samples and to the personnel of the Naval Guard Station of Boistö for assistance in sampling.

Heikki Pitkänen planned the work, calculated the material budgets and wrote the manuscript. Jorma Niemi participated in the development and realization of the program and the interpretation of the results. Juha Sarkkula was responsible for the current measurements and computation of the results and Olavi Sandman for taking and analysing sediment samples and carrying out echo-soundings and calculation of the results.

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## LONG-LIVED RADIONUCLIDES IN THE SEDIMENTS OF THE GULF OF BOTHNIA

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TUOMAINEN, K., ILUS, E. & TAIPALE, T.K. 1986. Long-lived radionuclides in the sediments of the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Varying amounts of the long-lived radionuclides  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  have entered the aquatic environment since the nuclear weapon tests of the 1960s. A considerable part of the deposited caesium and plutonium has been accumulated in the bottom sediments, whereas strontium occurs mainly in soluble forms and stays in the aqueous phase.

Since the 1970s, sediments from the Gulf of Bothnia have been analysed for  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$ . The activities of these nuclides in the sediments of the open sea basins varied from 55 to 87  $\text{Bq m}^{-2}$ , from 305 to 3100  $\text{Bq m}^{-2}$  and from 41 to 135  $\text{Bq m}^{-2}$ , respectively. The Olkiluoto nuclear power station has not increased the burden of  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  in the Gulf of Bothnia.

Index words: Gulf of Bothnia, sediment, long-lived radionuclides, radioecology

## 1. INTRODUCTION

Varying amounts of the long-lived radionuclides  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  have entered the aquatic environment since the nuclear weapon tests of the 1960s. A considerable part of the deposited caesium and plutonium has been accumulated in the bottom sediments, whereas strontium occurs mainly in soluble forms and stays in the aqueous phase.

Since 1976, sediment samples from the Gulf of Bothnia have been analysed at the Finnish Centre for Radiation and Nuclear Safety. The activities of  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  have been determined in order to study the background levels and behaviour of the nuclides in the sediments, to monitor the

impact of the Olkiluoto nuclear power station and to date the sediment layers.

## 2. MATERIAL AND METHODS

The sediment sampling locations are shown in Fig. 1. Open sea sampling was carried out in 1976—1982 by the R/V Aranda of the Institute of Marine Research.

The samples were taken with a box corer or a gravity corer (Niemistö 1974) and cut into sections of 5 or 1 cm to a depth of 10—30 cm.

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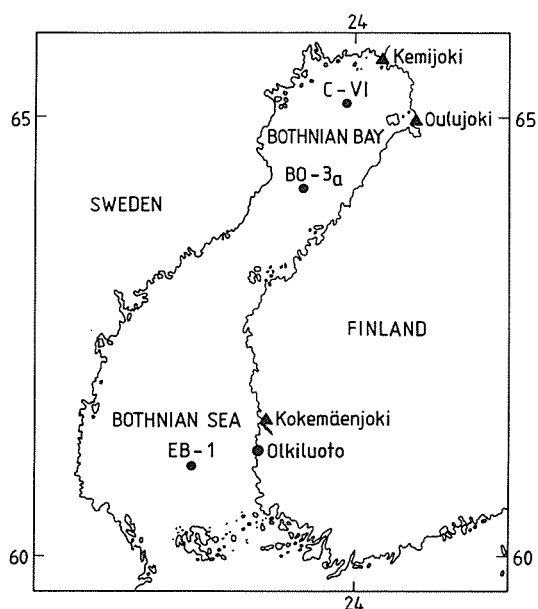


Fig. 1. Sediment sampling locations in the Gulf of Bothnia. ● = open sea station, ▲ = river mouth area.

The radiochemical analyses for  $^{90}\text{Sr}$  and  $^{239,240}\text{Pu}$  were carried out by methods described elsewhere (Insitute of Radiation Protection 1979).  $^{137}\text{Cs}$  was measured with Ge (Li) gamma spectroscopy.

### 3. RESULTS AND DISCUSSION

#### 3.1 Activity per unit area

The ranges of the activities of  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  per unit area ( $\text{Bq m}^{-2}$ ) in sediments from the Gulf of Bothnia are presented in Table 1. At Olkiluoto these activities varied widely. The highest activities were found in soft sedimentation bottoms with high organic contents.

In the open sea sedimentation basins, the activities of accumulated nuclides were rather low in comparison with the highest Olkiluoto values. However, the accumulated  $^{137}\text{Cs}$  activities were fairly high at EB-1. This may be due to the great proportion of small particles ( $< 0.002 \text{ mm}$ ) in that sediment material (see Table 2). The ready absorption of  $^{137}\text{Cs}$  into clay particles is widely known.

#### 3.2 Vertical distributions

Figs. 2 and 3 show some examples of the vertical distributions of  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  in the sediment cores.

In all the cores the nuclide concentrations usually decrease sharply to zero below the depth of maximum concentration. The concentration peaks reflect the period of maximum fall-out in 1962–65.

This information was applied in evaluating the sedimentation rates. It was estimated that the year 1963 is represented by the depth at which the contents of  $^{137}\text{Cs}$  or  $^{239,240}\text{Pu}$  clearly differ from the preceding value. With this method the sedimentation rates were evaluated at about  $10 \text{ mm a}^{-1}$  at Olkiluoto (Station 2),  $2\text{--}4 \text{ mm a}^{-1}$  at EB-1,

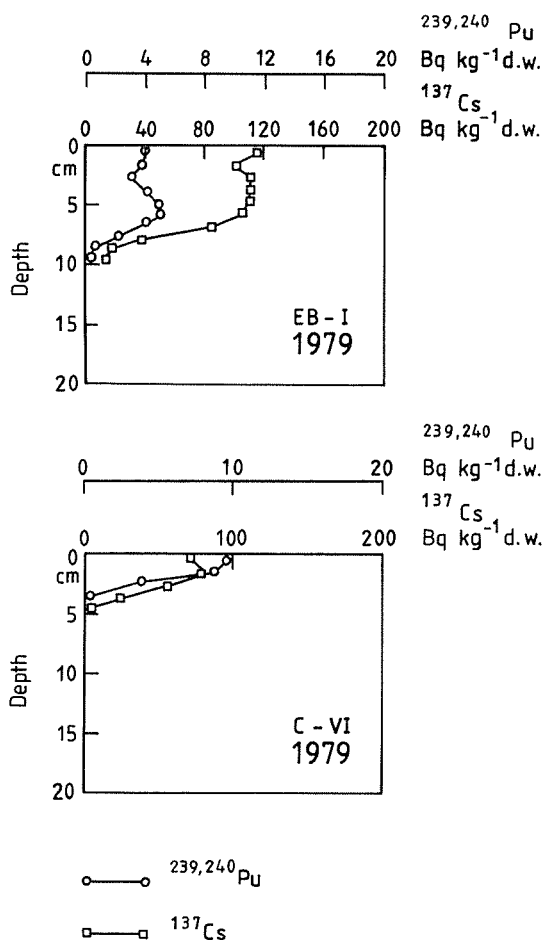


Fig. 2. The vertical distributions of  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  concentrations in the sediment cores from stations EB-1 and C-VI (1 cm sections).

c.  $2.5 \text{ mm a}^{-1}$  at BO-3a and  $1.5\text{--}2 \text{ mm a}^{-1}$  at C-VI. Using the  $^{210}\text{Pb}$ — $^{210}\text{Po}$  method, the Institute of Marine Research (IMR) has determined the sedimentation rates as  $1.0\text{--}3.0 \text{ mm a}^{-1}$  at EB-1 (Niemistö and Voipio 1981) and  $1.3 \text{ mm a}^{-1}$  at C-VI (IMR unpubl. results). The results agree fairly well with each other.

### 3.3 Total activities

The areas of soft and hard bottoms in the Gulf of Bothnia were calculated from the maps provided by Winterhalter et al. (1981). It has been estimated (Salo et al. 1986) that the accumulated activities of  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  per unit area on the hard bottoms

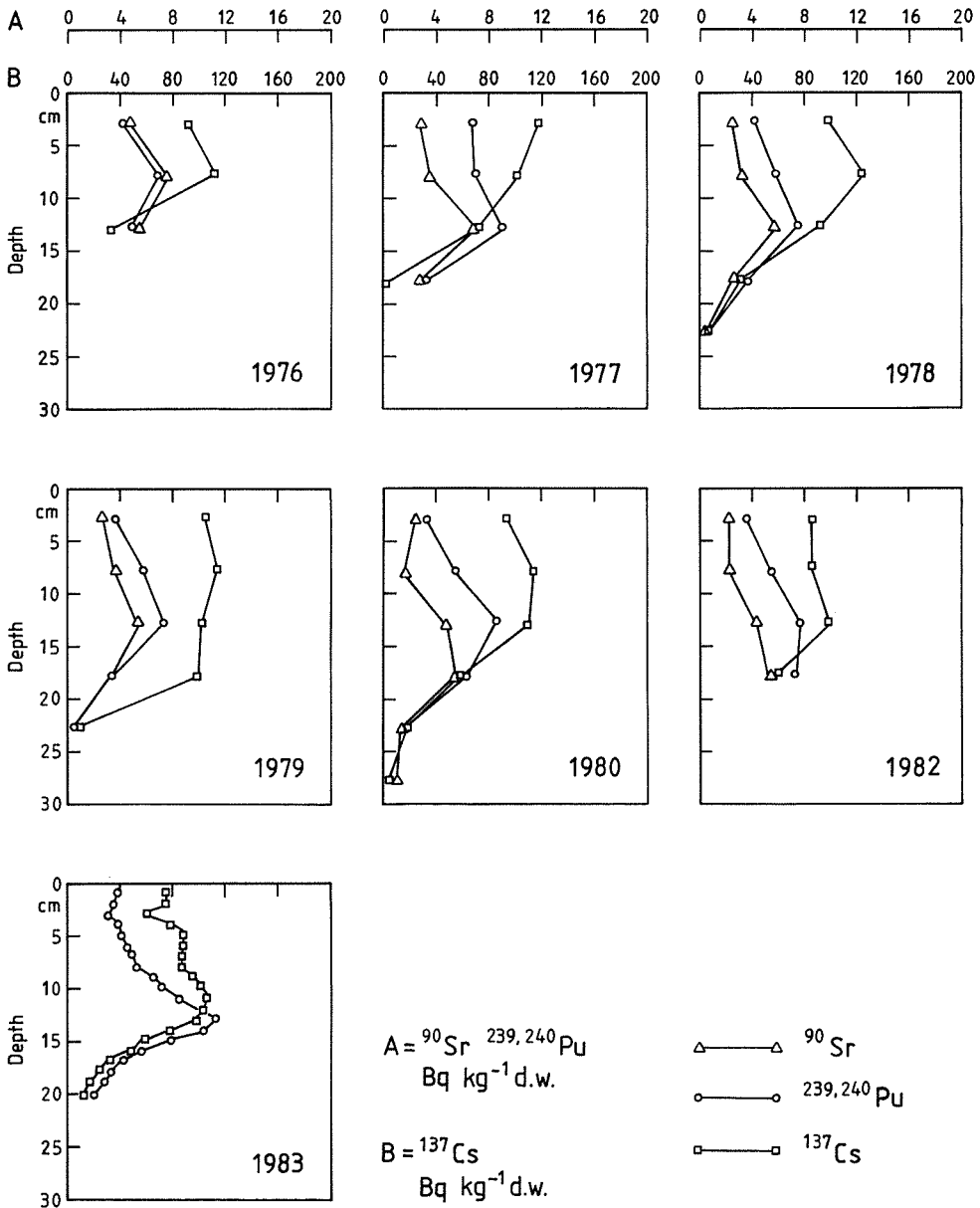


Fig. 3. The vertical distributions of  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{239,240}\text{Pu}$  concentrations in the sediments from Olkiluoto 2 (5 cm sections in 1976–82, 1 cm sections in 1983).

Table 1. The ranges of the total activities (Bq m<sup>-2</sup>) of <sup>90</sup>Sr, <sup>137</sup>Cs and <sup>239,240</sup>Pu in sediments from the different sampling locations in the Gulf of Bothnia.

| Location          | Depth<br>m | Range of total activities, Bq m <sup>-2</sup> |                   |                       |
|-------------------|------------|-----------------------------------------------|-------------------|-----------------------|
|                   |            | <sup>90</sup> Sr                              | <sup>137</sup> Cs | <sup>239,240</sup> Pu |
| Open sea stations |            |                                               |                   |                       |
| C-VI              | 52         | 77—79                                         | 478—890           | 41—117                |
| BO-3a             | 108        | ..                                            | 305               | ..                    |
| EB-1              | 126        | 55—87                                         | 1 390—3 100       | 52—135                |
| Coastal stations  |            |                                               |                   |                       |
| Olkiluoto area    | 5—16.5     | 16—192                                        | 316—4 260         | 16—252                |
| River mouths      |            |                                               |                   |                       |
| Kemijoki          | 15—26      | 81—140                                        | 1 800—2 000       | 115—231               |
| Oulujoki          | 17—38      | 82—120                                        | 2 200—2 700       | 119—135               |
| Kokemäenjoki      | 3.2—19.2   | 9.3—210                                       | 560—7 000         | 7.3—55                |

Table 2. Organic carbon, ignition loss and clay content in bottom sediment samples from the Gulf of Bothnia in 1982.

| Location    | Depth<br>cm | Organic carbon <sup>a)</sup> | Ignition loss | Clay (< 0.002 mm) |
|-------------|-------------|------------------------------|---------------|-------------------|
|             |             | %                            | %             | %                 |
| C-VI        | 0—5         | 2.6                          | 8.5           | 17                |
|             | 5—10        | 1.8                          | 6.8           | 17                |
|             | 10—12.5     | 1.7                          | 5.8           | 30                |
| EB-1        | 0—5         | 2.7                          | 9.8           | 48                |
|             | 5—10        | 2.5                          | 7.7           | 64                |
|             | 10—15       | 2.5                          | 6.6           | 74                |
|             | 15—18       | 1.8                          | 7.0           | 74                |
| Olkiluoto 2 | 0—5         | 5.1                          | 15.4          | ..                |
|             | 5—10        | 4.8                          | 13.7          | 24                |
|             | 10—15       | 4.7                          | 13.2          | 25                |
|             | 15—20       | 4.2                          | 11.6          | ..                |

<sup>a)</sup> samples taken in 1979.

Table 3. The total amounts of <sup>90</sup>Sr, <sup>137</sup>Cs and <sup>239,240</sup>Pu found in the sediment and sea water from the Gulf of Bothnia (Institute of Radiation Protection\* 1979, 1980, 1982 a, b and 1983, Salo et al. 1986 and Miettinen et al. 1982).

|                                        | Total amount, TBq |                   |                       |
|----------------------------------------|-------------------|-------------------|-----------------------|
|                                        | <sup>90</sup> Sr  | <sup>137</sup> Cs | <sup>239,240</sup> Pu |
| Sediment                               |                   |                   |                       |
| Bothnian Bay (37 000 km <sup>2</sup> ) | 1.9               | 27                | 2.25                  |
| Bothnian Sea (79 000 km <sup>2</sup> ) | 3.6               | 113               | 4.3                   |
| Sum                                    | 5.5               | 140               | 6.5                   |
| Sea water, 1981                        |                   |                   |                       |
| Bothnian Bay (1 500 km <sup>3</sup> )  | 30                | 21                | 0.023                 |
| Bothnian Sea (5 200 km <sup>3</sup> )  | 135               | 88                | 0.042                 |
| Sum                                    | 165               | 109               | 0.065                 |
| Sea water + sediment                   | 170               | 249               | 6.6                   |

\* Present name: Finnish Centre for Radiation and Nuclear Safety

are one fifth of the activities accumulated on soft bottoms and that the corresponding value for  $^{239,240}\text{Pu}$  is one tenth. These figures were used to estimate the total activities of nuclides in the Gulf of Bothnia and the results showed that roughly 3 % of the  $^{90}\text{Sr}$ , 56 % of the  $^{137}\text{Cs}$  and 99 % of the  $^{239,240}\text{Pu}$  in the Gulf of Bothnia are found in the bottom sediments (Table 3).

The results indicate that the Olkiluoto nuclear power station has not increased the burden of  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  or  $^{239,240}\text{Pu}$  in the Gulf of Bothnia.

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## REVIEW OF THE BIOLOGY OF THE GULF OF BOTHNIA

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ANDERSIN, A.-B., GRIMÅS, U., KANGAS, P., LEHTONEN, H. & LEPPÄNEN, J.-M. 1986. Review of the biology of the Gulf of Bothnia. Publications of the Water Research Institute, Finland, No. 68.

The paper reviews the latest biological information published on the state of the Gulf of Bothnia. Signs of oceanization and some increase of macrobenthic biomass are seen in the records from the open sea. The state of the coastal waters varies with the local conditions and the waste-water load. The fish fauna has deteriorated due to destruction of the spawning and nursery areas.

Index words: Biological variables, water quality, Gulf of Bothnia, Archipelago Sea

### 1. INTRODUCTION

The preceding review of changes in the state of the Gulf of Bothnia was made for the second seminar in 1981 (Lassig and Grimås 1983). Since then the understanding of the status of the Gulf of Bothnia and the trends in these waters has been improved by results of the bilateral and national environmental monitoring programmes, programmes for the surveillance of waste water discharge areas and ongoing research projects. This article will review the latest information on the open sea and coastal areas of the Gulf of Bothnia.

### 2. OPEN SEA

Most of the biological data have been collected from the coastal waters and only a few reports have been published on the pelagic and benthic biology

of the open waters of the Gulf of Bothnia.

The phytoplankton and chlorophyll data collected during the first stage of the Baltic Monitoring Programme in 1979—1983 (Huttunen et al. 1986) do not show significant changes during this period or when compared with the chlorophyll values from 1969—1975.

Long-term studies on the mesozooplankton in the open area of the Bothnian Sea in 1969—1983 reveal a slight increasing trend in the mesozooplankton abundance (Tyystjärvi-Muuronen 1984). This is caused by the increased number of copepods, the abundance of the other groups having remained at the same level. It is pointed out that the observed increase in the number of individuals is connected with the increased salinity of the area and does not necessarily mean increased zooplankton production.

A review by Kankaala et al. (1984) summarises the results of phyto- and zooplankton production studies carried out in the northeastern and central Bothnian Bay in 1968—1978. Differences in the pelagic production level are shown between the

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undisturbed open sea and polluted coastal areas. The high ratio of zooplankton production to primary production in the Bothnian Bay as compared with the other areas of the Baltic Sea is discussed. The explanations suggested are efficient utilization of the phytoplankton production by the zooplankton and a large proportion of allochthonous organic matter in the pelagic food web.

Despite great annual variation, the macrozoobenthos data compiled from the standard stations in the Gulf of Bothnia for the period 1965–1983 show signs of an increase in the biomass of the dominant species in both the Bothnian Sea and Bothnian Bay (Andersin 1986). Analyses of the species composition indicate slight oceanization of the bottom fauna in the Bothnian Sea. At a deep locality in the southern Bothnian Bay a rapid increase was followed by an almost total collapse of the benthic fauna.

The Programme for Environmental Monitoring (PMK) in Sweden includes a project for revisiting the open sea zoobenthos stations sampled in the 1920s (Hessle 1924) and 1950s (Sellerberg 1961) in order to study eutrophication in the Åland and Bothnian Seas. According to the report given by Cederwall and Leonardsson (1984), the macrobenthic biomass values in 1983 were 4.1 times as high as in the 1920s and 3.6 times as high as in the 1950s.

The decline of the autumn trawl catches of the Baltic herring in the late 1970s and early 1980s in the sea area off Pori (Parmanne et al. 1983) prompted extensive investigation of the possible effects of the acid-iron wastes discharged by a titanium dioxide plant (Tulkki et al. 1983). Data collected on the phytoplankton (Kononen 1983), mesozooplankton (Tyystjärvi-Muuronen 1983) and macrozoobenthos (Andersin 1983) in the open waters of the Bothnian Sea did not point to any changes in the ecological conditions in this area. The laboratory tests showed that one plausible reason for the low catches is that the fish avoid the acid-iron effluents, even when highly diluted (Hernejärvi 1983).

In a study on the food of Baltic herring the crustacean species of the genus *Mysis* were found to be one of the chief food items (Isaksson 1983). Routine methods for studying the zooplankton and -benthos proved to be insufficient for an investigation of the biology of the *Mysis* species and a joint project was started in 1984 to study this subject.

The data on fish populations collected during the 1970s and 1980s show that the total abundance of Baltic herring has not changed significantly in the open Gulf of Bothnia (Anonymous 1984).

Recruitment of cod from the southern Baltic Sea to the Gulf of Bothnia increased rapidly during the second half of the 1970s but in recent years it has diminished sharply and the size of the population is decreasing (Aro and Sjöblom 1983).

### 3. COASTAL WATERS

Active monitoring and research are conducted in the coastal waters as a result of the national programmes, of the local programmes for monitoring waste-water discharge areas and special studies carried out by the universities. The heterogeneous data and varied coastal topography make it difficult to compile a comprehensive synopsis of the state of the coastal waters on the basis of the biological variables studied. The recently published information is reviewed here to provide an overall picture of the different biological processes and their surveillance in the Gulf of Bothnia and the Archipelago Sea.

The littoral communities in the Archipelago Sea have undergone marked changes during the late 1970s and early 1980s (Haahtela and Lehto 1982, Haahtela 1984, Rönnerberg 1984, Mäkinen et al. 1984, Hällfors et al. 1984). Over wide areas *Fucus vesiculosus* has declined and alterations occurred in the algal and faunal species composition and animal abundance. The basic reason for this series of changes seems to be upwelling of nutrients from the Baltic deeps rather than eutrophication caused by man (Kangas et al. 1982, Haahtela 1984). In a few sea areas far from the direct influence of the open sea, e.g. off Turku, nutrients from the land have probably played a major role in the decline of *Fucus*.

Chemical water analyses often fail to show huge discharges of nutrients, e.g. from fish farms. The ongoing development of a method for measuring periphyton production (Leskinen et al. 1986) will improve monitoring of eutrophication. Determination of the nutrient contents in the tissues of the green alga *Cladophora* will also increase information on the availability of nutrients in coastal waters (Mäkinen and Aulio 1986).

The paper by Kippo-Edlund and Niemi (1986) on the phytoplankton in the sea area off Turku and the Archipelago Sea is based on coastal monitoring data. During the investigation period 1966–1982, the phytoplankton species composition has become more marine and the biomass increased in the inner archipelago, reflecting the

observed oceanization of the Baltic Sea (see Fonselius 1986).

A series of reports has been published on the heterotrophic processes in the area of the Bothnian Sea receiving pulp mill effluents (Tamminen and Kuparinen 1983, Kuparinen et al. 1983). Most of the reports, however, are methodological in nature. The knowledge of the factors controlling production has improved in the northern Bothnian Sea (Larsson and Hagström 1982, Hagström et al. 1983). Experiments with addition of nutrients have shown that during the growing period, the primary production in the area can be stimulated by either nitrogen or phosphorus. There is also evidence that the addition of molybdenum increases the rate of nitrogen fixation and production of algae (Åke Hagström, pers. comm.).

Extensive unpublished material exists on the pelagic system, which can be expected to provide information on trends in the water quality. The zooplankton has been observed by the Finnish Institute of Marine Research every tenth day from three stations in the Gulf of Bothnia. Frequent measurements of the primary production and chlorophyll have been started at a total of five stations within the national monitoring programmes in Finland and Sweden. Considerable heterogeneity exists in the biological variables studied in the programmes, depending on the local needs and the quality of the discharges. Several programmes do not include any biological variables. The programmes for monitoring the waste-water discharge areas are intended to show the local impact of the effluents on the ecosystem, and it is thus difficult or impossible to draw conclusions regarding trends in wider areas or to make comparisons between different receiving waters.

Some results indicate undesirable developments, but in most areas no changes have been reported since 1981. Off Turku, no major change has occurred in the state of the bottom fauna since 1979 (Häkkilä and Puhakka 1984). In the outer area signs of slight eutrophication can be seen. At the same time there has been an increase of the nutrients discharged to the sea by rivers (Jumppanen and Kolehmainen 1984, Pitkänen 1986) and the primary production capacity has risen (Jumppanen and Kolehmainen 1983). Fish farming has been observed to cause local eutrophication, e.g. by Häkkilä (1982) and Leskinen et al. (1986). Man-made embankments cause a permanent change in the environment and it has been shown that it takes years for organisms to fully adapt to the new conditions (Bonsdorff et al. 1984).

In some minor areas the water quality has

improved, as judged by the biological parameters. In the Nagu Sea area in the Archipelago Sea a noticeable recovery of the bottom fauna has been observed after the closing of the fish industry (Leppäkoski et al. 1983). Signs of improvement can also be seen in Halikko Bay and the sea area off Rauma (Jumppanen 1982, 1983, 1984, Häkkilä 1983). In the coastal waters around Pori planktonic and benthic data show improvement since the 1970s (Oravainen 1980, 1982, Mankki and Kosonen 1982 and Häkkilä 1983). This is mainly due to the decrease of municipal and pulp mill waste waters.

In Ljusne Bay, on the Swedish side of the central Bothnian Sea the biomass and total abundance of the macrofauna have increased after a reduction in the load of nutrients and organic matter (Cederwall and Persson 1986). Some species sensitive to pollution, mainly *Pontoporeia affinis*, are still under-represented in the fauna. More biological results from the Swedish side are presented by Cederwall (1986).

In many areas the composition of the fish fauna has changed because of the industrial effluents. Off Rönnskärverket the catches of autumn- and winter-spawning species have diminished (Hansson and Thoreson 1978). The effect of the heated effluents from the Forsmark nuclear power plant have been studied both in the open receiving waters, Öregrundsgrepen in the Bothnian Sea, and in the enclosed Biotest Basin. The reactions of the fish communities have been similar to those reported at other outfalls of thermal discharges in Sweden (Neuman 1983). For instance, herring have been attracted to the warm water in spring, with early spawning as a result, and warm-water species, such as roach and perch, have increased in abundance in the Basin, while the cool-water species have almost disappeared (Sandström 1985). In the sea area around Pori the populations of coast-spawning species have diminished because of spawning failures. Eyeless Baltic herring are found regularly, mainly in the Bothnian Sea. The connection between the acid-iron effluents from the titanium dioxide plant (Järvinen 1983) and myxobacterial infection (Bylund 1983) is still under discussion.

The distribution and abundance of pike suffering from a tumour disease (sarcomatosis) were studied in coastal areas off southwestern Finland. This disease occurs in the area south of Rauma and is most common in the Archipelago Sea and Åland, where up to 10–20 % of the pike may be contaminated (Nyström 1982). The disease is probably caused by a virus (Thompson 1982).

The environmental changes have been greatest near the shore in shallow waters. The fish stocks

which spawn there, have diminished in many places and fishing has therefore become increasingly directed at a few species, such as Baltic herring, salmon, whitefish and, in the Bothnian Bay, vendace (*Coregonus albula* L.) (Hildén et al. 1982). Thus the fishing pressure on these fish species has increased, which may finally result in overfishing. The salmon stocks of the Gulf of Bothnia are already too heavily fished (Toivonen 1983). The fishing mortality of the vendace does not appear to have increased when calculated by the yield per recruit method (Lehtonen 1981, 1983, Hildén et al. 1982), but the fishing mortality of the sea-spawning whitefish is close to the allowable maximum and should not be greatly increased (Lehtonen 1981). The present fishing mortality of the Baltic herring is still below the allowable maximum and it has been possible to raise the mortality in Finnish waters without endangering the stock (Parmanne and Sjöblom 1983). In the Bothnian Bay, however, the fishing mortality is already near the maximum (Anonymous 1985). In the northern part of the Archipelago Sea the fishing of pike-perch, pike, bream and perch is also near the optimum, whereas the catches of roach could still be increased (Lehtonen et al. 1983). At the estuary of the river Kyrönjoki the fishing mortalities of perch, burbot, bream, pike and smelt are near the optimum (Hudd et al. 1986).

Most fish species, especially freshwater species, spawn in shallow waters in rivers and streams and their estuaries. Failures of reproduction mostly coincide with poor water quality and especially with years when anthropogenic acidification occurs. The dependence of fish on estuaries and rivers increases northwards in the Gulf of Bothnia, as the environment becomes harsher. The estuaries offer areas rich in nutrients and with warm water in spring when the larvae hatch (Eriksson and Müller 1981, Hudd et al. 1984).

#### 4. SUMMARY AND CONCLUSIONS

According to the recently published data on the plankton and benthos, no decisive changes have occurred in the state of the open Gulf of Bothnia. It is difficult to make a trend analysis using the present data because of great year-to-year and seasonal fluctuations. Compilation of the results reveals, however, that the recent biomass values are higher than the values measured at or before the

beginning of the 1970s. A slight oceanization effect can be seen in the species composition in the Bothnian Sea. The changes in species composition and biomass values can be attributed to the increased salinity and related increase of nutrients. As pointed out by Fonselius (1986), however, the oceanization cannot completely explain the increase of the nutrient level. Some part may thus be played by such factors as anthropogenic eutrophication and changes in the seasonal distribution of fresh water caused by regulation of river stream flow.

The changes in the state of coastal waters depend on the local conditions and the variations in the waste-water load. Undesirable developments have been observed in several areas, but recoveries of pelagic and benthic communities have also been reported in areas where the waste-water load has decreased or totally ceased. In future special attention must be paid to obtaining comparable results from different coastal areas.

The composition of the fish stocks has changed considerably due to environmental changes caused by man, e.g. destruction of spawning areas, and to intensified fishing. Besides this industrial pollution has had an injurious effect on fishing in the open sea. Fish diseases and deformities are also more usual in polluted areas than elsewhere. The observation that coastal fish populations are recruited from rivers and streams is of special interest. It seems that the quality of these important nurseries must be considered particularly important for the coastal fish populations of the Gulf of Bothnia.

This survey of data on the biota of the Gulf of Bothnia shows clearly that reliable estimates of the water quality and the trends in it require long-term series of observations made at suitably short intervals. Deficiencies still exist in the design and coordination of the studies (e.g. choice of variables and sampling strategy). The natural fluctuations in the biota and the controlling factors have to be known for correct interpretation of the results. Therefore accurate knowledge of the function of the ecosystem is needed. With its low species diversity and short food chains, the Gulf of Bothnia offers an excellent opportunity for ecosystem studies. The level of primary and secondary production and their main seasonal patterns are already known, and some investigations have been made on the ecological processes and the factors controlling them. A proper understanding of the effects of different environmental factors will require further investigations and effective co-operation between Finland and Sweden.

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# THE QUESTION OF EUTROPHICATION IN THE BALTIC SEA — RESULTS FROM A LONG-TERM STUDY OF THE MACROZOOBENTHOS IN THE GULF OF BOTHNIA

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ANDERSIN, A.-B. 1986. The question of eutrophication in the Baltic Sea — results from a long-term study of the macrozoobenthos in the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Fluctuations in the biomass of numerically dominant macrobenthic species were studied at seven stations in the Gulf of Bothnia and the Åland Sea during the period 1965—1983.

At four stations, the fluctuations were fairly synchronous, with three maxima and two minima evident during the investigation period. At these stations a significant increasing trend in the biomasses could be noted, the implications of which are discussed.

At one station in the southern Bothnian Bay, a strong increase in 1974—1977 was followed by an almost total collapse of the zoobenthos community in 1980, a development for which no adequate explanation can be given at present.

At the two stations in the Åland Sea, particularly strong fluctuations were noted. These can be explained mainly by patchy distribution of the fauna and by the influence of the oxygen conditions in the Baltic Proper.

Index words: Macrozoobenthos, long-term studies, eutrophication, Gulf of Bothnia

## 1. INTRODUCTION

A very urgent question today is to what extent the high load of nutrients imposed on the Baltic Sea during the twentieth century has influenced the water quality in the Baltic as a whole.

Although local effects of the increased nutrient load are well documented along the coasts of the Baltic Sea, there is very little definite information about the response to the reported nutrient increase in the open sea areas (Pietikäinen et al. 1978, Dybern and Fonselius 1981, Koljonen et al. 1983, Nehring 1984, Perttilä et al. 1984, Voipio 1984).

The results of zoobenthos studies in the deep parts of the Baltic Proper show that the area in which the macrofauna is lacking or strongly reduced because of critical oxygen conditions has increased considerably during this century, especially in the 1960s. The main reason for this phenomenon is believed by many scientists to be the increased load of nutrients (Zmudzinski 1976, 1983, Ehlin et al. 1982, Cederwall and Elmgren 1980). In well-oxygenated open sea areas much higher macrobenthic biomasses have been observed in the 1980s than in the 1920s, a result which has been explained both by overall eutrophication (Cederwall and Elmgren 1980) and by overfishing

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of flatfish in the 1930s (Persson 1981).

Indications of increasing macrobenthic biomass have been reported from the Gulf of Bothnia as well (Andersin et al. 1978, 1981, Andersin 1983, Cederwall and Leonardsson 1984).

At the Finnish Institute of Marine Research (FIMR) the first zoobenthos samples from the Gulf of Bothnia were taken in 1954, and regular sampling started in 1961. Part of the material has been treated previously (Andersin et al. 1977, 1978, 1981, 1984, Andersin 1983). The present study examines the question of eutrophication in the Gulf of Bothnia using selected results from this long-term study.

## 2. MATERIAL AND METHODS

The following stations were selected: F 69 and F 64 in the Åland Sea, SR 5 and US 6b in the Bothnian Sea and F 15, BO 3 and C VI in the Bothnian Bay (Fig. 1). The study covers the period 1965–1983. Only material from May–July were used, the two main sampling periods being May–early June, and July.

Only the fluctuations in the numerically dominant species are treated, e.g. *Pontoporeia affinis* at

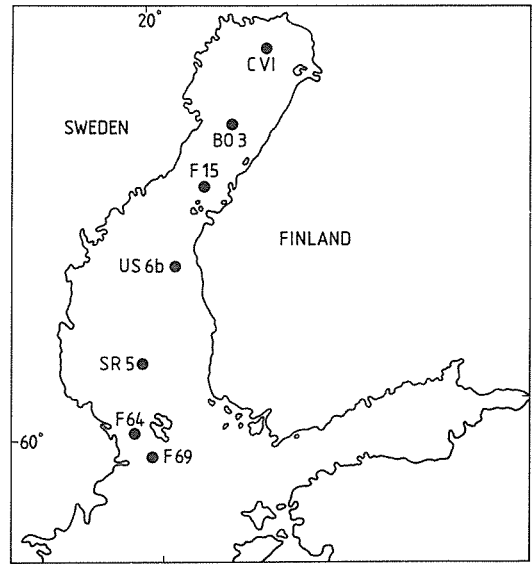


Fig. 1. Sampling localities.

stations C VI, US 6b and SR 5, *Macoma balthica* at F 15, *P. femorata* at F 69, and *P. femorata* and *P. affinis* at F 64 (cf. Table 1). The only other species of importance for the total biomass is the isopod *Mesidotea entomon*. The abundance of this species is, however, very low, its distribution uneven and

Table 1. The 1975 level of macrobenthic abundance (ind m<sup>-2</sup>) and biomass (ash-free dry weight, g m<sup>-2</sup>) in the Gulf of Bothnia. For location of the stations see Fig. 1.

| Station       | C VI       |         | BO 3       |         | F 15       |         | US 6b      |         |
|---------------|------------|---------|------------|---------|------------|---------|------------|---------|
|               | P. affinis |         | P. affinis |         | M. baltica |         | P. affinis |         |
|               | Abundance  | Biomass | Abundance  | Biomass | Abundance  | Biomass | Abundance  | Biomass |
| MAY           |            |         |            |         |            |         |            |         |
| Dominant spp. | 908        | 0.16    | 447        | 0.08    | 344        | 2.10    | 2 283      | 1.28    |
| Total         | 922        | 0.17    | 447        | 0.08    | 351        | 2.10    | 2 294      | 2.90    |
| JULY          |            |         |            |         |            |         |            |         |
| Dominant spp. | 775        | 0.19    | 398        | 0.09    | 281        | 2.47    | 2 619      | 3.28    |
| Total         | 793        | 0.20    | 398        | 0.09    | 287        | 2.47    | 2 633      | 4.42    |

| Station       | SR 5       |         | F 64                     |         | F 69        |         |
|---------------|------------|---------|--------------------------|---------|-------------|---------|
|               | P. affinis |         | P. affinis + P. femorata |         | P. femorata |         |
|               | Abundance  | Biomass | Abundance                | Biomass | Abundance   | Biomass |
| MAY           |            |         |                          |         |             |         |
| Dominant spp. | 2 888      | 1.39    | 367                      | 0.25    | 81          | 0.08    |
| Total         | 2 972      | 1.44    | 397                      | 2.25    | 97          | 0.56    |
| JULY          |            |         |                          |         |             |         |
| Dominant spp. | 2 462      | 2.96    | 491                      | 0.41    | 339         | 0.41    |
| Total         | 2 601      | 3.06    | 521                      | 1.91    | 381         | 0.71    |

its size very variable, all facts which make it difficult to study fluctuations in the populations by the present sampling methods.

The ash-free dry weight was chosen as a less subjective measurement of biomass than the wet weight. The ash-free dry weight has been used as a standard parameter since 1972 and was determined for the older samples in this study. In connection with this the wet weights were redetermined. (For detailed information on the methods see e.g. Ankar et al. 1979).

In order to eliminate the effect of differing sampling periods as far as possible, and to calculate the means of the different stations, the biomass values for each year are expressed as the relative difference from the values of 1975 (Table 1), a year when sampling was carried out during both main sampling periods.

In order to detect possible signs of increasing biomass, linear regressions were calculated, with time on the x-axis and the mean relative changes of the sampled stations on the y-axis (Fig. 3). The results showed a significant rising trend in the biomass values during the investigation period.

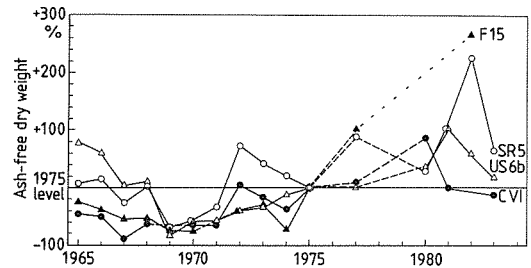


Fig. 2. Fluctuations in the ash-free dry weight of numerically dominant species at four stations in the Gulf of Bothnia in 1965–1983, expressed as the relative difference from the values of 1975.

### 3. RESULTS

The stations can be roughly divided into three groups on the basis of the relative changes in the biomass of their numerically dominant species.

Stations SR 5, US 6b, F 15 and C VI show a clear alternation between periods with high values and periods with low values (Fig. 2). The distribution in time of these periods is roughly similar at the four stations. Two periods with low biomass values and three with high values can be observed. After the last period with high values, in 1981–1982, a decrease can be discerned at all the stations visited. According to preliminary results from 1984, this trend is continuing.

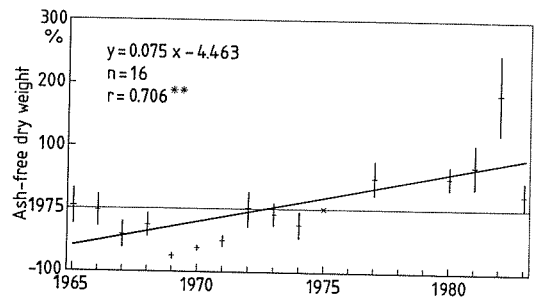


Fig. 3. Trend of ash-free dry weight of numerically dominant species, calculated from the mean values for stations C VI, F 15, US 6b and SR 5. The vertical lines indicate the standard error of the means, \*\* = 1 % significance level.

Table 2. A. The relation between the maximum biomass values (ash-free dry weight) of the numerically dominant species for different periods. B. The relation between maximum and subsequent minimum biomass values in the 1960s. For location of the stations see Fig. 1.

|               | SR 5 | US 6b | F 15 | C VI | Mean $\pm$ SE (SR 5-C VI) | F 69 | F 64 |
|---------------|------|-------|------|------|---------------------------|------|------|
| A. max. 1970s |      |       |      |      |                           |      |      |
| max. 1960s    | 1.6  | 0.6   | 0.9  | 1.9  | $1.2 \pm 0.24$            | 0.6  | 0.3  |
| max. 1980s    |      |       |      |      |                           |      |      |
| max. 1970s    | 1.9  | 2.0   | 5.2  | 1.8  | $2.7 \pm 0.82$            | 10.9 | 3.2  |
| max. 1980s    |      |       |      |      |                           |      |      |
| max. 1960s    | 2.8  | 1.1   | 4.8  | 3.3  | $3.0 \pm 0.76$            | 1.8  | 2.0  |
| B. max. 1960s |      |       |      |      |                           |      |      |
| min. 1960s    | 3.7  | 9.8   | 2.9  | 4.3  | $5.2 \pm 3.14$            | 60.0 | 4.7  |

The trend is significant at each station except US 6b. It is also obvious that the rising trend is chiefly caused by the high values in the 1980s (Table 2).

At station BO 3 in the southern part of the Bothnian Bay the fluctuation pattern is quite different (Fig. 4). In 1973 an increase was noted and the values reached a very high level in 1977. In 1980 the macrozoobenthos community had almost collapsed, consisting of only a few individuals of *P. affinis*, *Prostoma obscurum* and *Mesidotea entomon*. According to preliminary results the recovery seen in 1983 has continued in 1984.

At the two stations in the Åland Sea (Fig. 5) the differences between the years are much greater than at the other stations. Although these differences are largely due to uneven distribution of the fauna in this area, some agreement with the main fluctuation pattern can be observed. Though roughly similar at the two stations, however, the distribution in time of high and low biomass periods is not quite the same as at the stations in the Gulf of Bothnia. As in the Gulf of Bothnia the highest values were noted in the 1980s (Table 2). In the Lågskär Basin (F 69) the extremely high values in 1980 were followed by an almost total collapse of the fauna in 1981.

#### 4. DISCUSSION

Although the stations are few, their fauna shows clearly the well-known quantitative differences between the different parts of the Gulf of Bothnia (cf. Andersin et al. 1977, Elmgren et al. 1984).

Among the three different fluctuation patterns recorded, that of the four stations, SR 5, US 6b, F 15, C VI, from different parts of the Gulf of Bothnia is the most interesting. The similarity of the distribution in time indicates that some connection may exist with the fluctuations of some large-scale environmental factors, e.g. climatological factors. This question has been discussed by Josefsson and Smith (1984) in their examination of zoobenthos fluctuations in the Skagerak area. As regards the increase in biomass from the 1960s to the 1980s it is necessary to examine the results critically. The reweighing of the old material has certainly caused a loss in biomass, and thus the maximum values in the 1960s are too low and the real increase may be assumed to be smaller than shown by the figures. The fact remains, however, that the values around

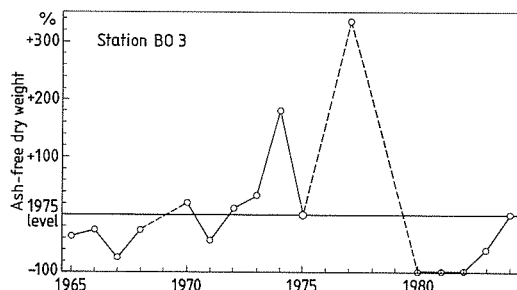


Fig. 4. Fluctuations in the ash-free dry weight of *Pontoporeia affinis* at station BO 3 in the Bothnian Bay in 1965–1984, expressed as the relative difference from the values of 1975.

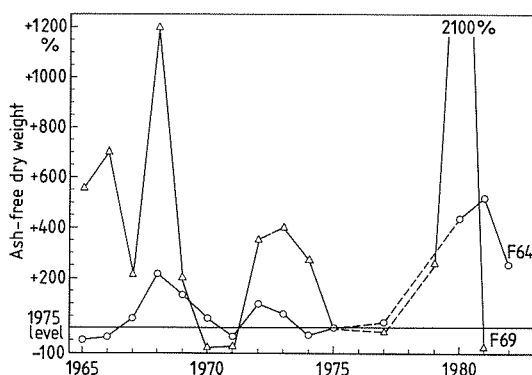


Fig. 5. Fluctuations in the ash-free dry weight of the numerically dominant species at two stations in the Åland Sea in 1965–1983, expressed as the relative difference from the values of 1975.

1980 are higher than even observed before, at all stations except US 6b.

At present no adequate explanation can be given for the strong impoverishment of the fauna at station BO 3. The pattern is classical for eutrophicated shallow areas (cf. e.g. Leppäkoski 1975), with increased values followed by collapse when the decomposition of the organic material has consumed all the available oxygen. No drastic change in the oxygen content of the bottom water has been observed at this station, however, (unpublished results of the FIMR), and thus this possibility must be excluded. One reason could be uneven distribution of the fauna. Preliminary results from a patchiness study in this area indicated, however, that the fauna was equally poor over a larger area in 1982. A plausible explanation may be a disease or parasite infection. This theory is supported by the gradual recovery observed in 1983–1984.

At the stations in the Åland Sea, the fluctuations in the oxygen content of the deep water in

the Baltic Proper seem to be an important reason for the fluctuations in the benthic fauna, especially in the Långskär Basin (cf. Andersin et al. 1983), where an almost total collapse of the fauna was observed in 1981. In view of the very great range of the fluctuations, they must also, at least partly, be attributed to uneven distribution of the animals, since both stations are situated in unstable areas, F 69 on a slope and F 64 in an area with bottom currents.

Although the material of this study is restricted, some general conclusions can be made. The results confirm the necessity of a good knowledge of natural fluctuations in demonstrating the consequences of human activity for the ecosystem. More information is also needed on special features of the ecosystem in this area, especially the influence of environmental fluctuations on the processes within the ecosystem. The obvious need of a representative number of stations and parameters for elucidation of the connection between environmental and zoobenthic fluctuations makes it desirable to utilize all the material available.

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## THE BENTHIC MACROFAUNA OF THE HOLMÖ AREA (NORTHERN BOTHNIAN SEA)

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ANKAR, S. & LEONARDSSON, K. 1986. The benthic macrofauna of the Holmö area (northern Bothnian Sea). Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

In August 1981, the benthic macrofauna was sampled in the marine nature sanctuary area of Holmöarna, situated in the Quark. The area was poor in species with a mean of 3.5 taxa  $0.1 \text{ m}^{-2}$ . The species composition was typical of the northern Bothnian Sea and the dominants being *Mesidothea entomon*, *Pontoporeia affinis* and *Macoma balthica*. These species made up nearly 100 % of the macrofaunal abundance, biomass and production, which amounted to  $3\,310 \pm 1\,020$  ind  $\text{m}^{-2}$ ,  $43.9 \pm 9.9 \text{ g m}^{-2}$  ( $\pm$  standard error of mean) and  $90 \text{ kJ m}^{-2} \text{ a}^{-1}$ , respectively. The estimated production is half that calculated for the northern Baltic proper, but three times that of the Bothnian Bay. *P. affinis* showed a 2-year life cycle, shorter than expected for such a northern latitude.

Index words: Bothnian Sea, benthic macrofauna, abundance, biomass, production

### 1. INTRODUCTION

A marine nature sanctuary has been established by the National Environment Protection Board of Sweden on Holmöarna (Fig. 1), islands situated in the Quark between the Bothnian Sea and Bothnian Bay ( $63^{\circ}34' - 63^{\circ}51' \text{N}$ ,  $20^{\circ}43' - 21^{\circ}03' \text{E}$ ). The purpose of this study was to make an inventory of the benthic macrofauna around the islands.

The sea bottom area in the sanctuary is  $113 \text{ km}^2$ . The water is shallow with a mean depth of 11 m and has a salinity of around 3.5–5.5 ‰. The sediments consist of moraine with a low organic content, about 1 % (loss on ignition). Erosion and abrasion are heavy on the bottoms and the northern parts especially are strongly affected by scraping ice.

### 2. METHODS

In early August 1981, the benthic macrofauna was sampled with an  $0.10\text{-m}^2$  van Veen grab at 30 stations (1 sample per station). The depth range sampled was 7–43 m. The treatment of samples followed the BMB recommendations (Dybern et al. 1976).

### 3. RESULTS AND DISCUSSION

The low salinity of the area sets the northern distribution limit for most of the marine elements

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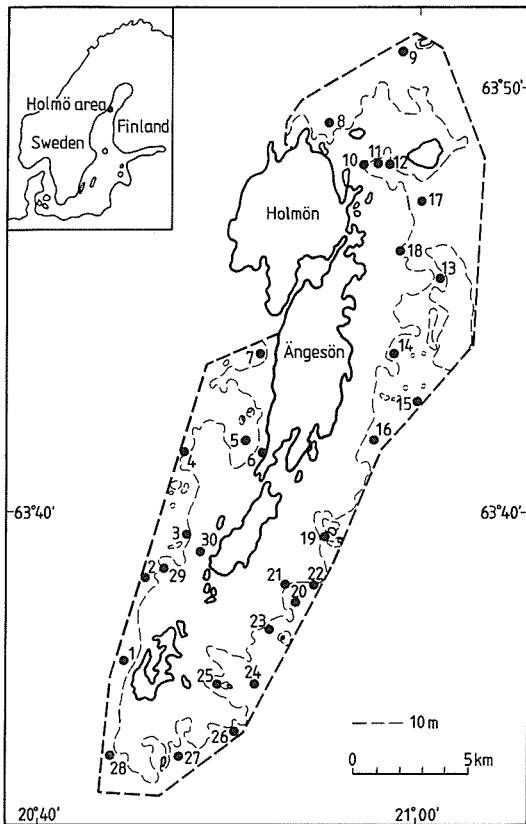


Fig. 1. Map of the Holmö area and the sampling localities.

of the Baltic macrofauna. This was reflected in the low number of species present, on average 3.5 taxa  $0.1 \text{ m}^{-2}$  (Table 1). Of the 10 taxa obtained, 4 were of marine origin. Nearly 100 % of the biomass (wet weight including shells) was contributed by *Mesidothea entomon* (3 %), *Pontoporeia affinis* (23 %) and *Macoma balthica* (74 %). *P. affinis* was the dominant of the total abundance, with a percentage of 83 %, and a maximum density as high as 25 000 ind  $\text{m}^{-2}$  was observed at one site (40 m depth).

The total mean abundance and biomass amounted to  $3\,310 \pm 1\,020 \text{ ind m}^{-2}$  and  $43.9 \pm 9.9 \text{ g m}^{-2}$  (Table 1), or to  $2\,500 \text{ ind m}^{-2}$  and  $40 \text{ g m}^{-2}$ , respectively, when weighted according to the bottom depth distribution. Comparisons with other Baltic areas show that the macrofauna diversity of the study area is slightly higher and the biomass considerably higher than what is normally found in adjacent areas, except for the coastal area at Norrby, situated 50 km south-west of the study area (Fig. 2). In the Norrby area, Cederwall and Leonardsson (1984) found a mean biomass in the upper 43 m of  $100 \text{ g m}^{-2}$ , more than twice that of the study area. The biomass in the Askö area (northern Baltic Proper, Ankar and Elmgren 1976) is three times as high as in the Holmö region. Generally, the macrofaunal biomass shows the highest values on shallow bottoms and the values decrease with increasing depth, but in the Holmö area the depth distribution of the biomass showed a minimum ( $33 \text{ g m}^{-2}$ ) in the upper depth interval, 7–20 m, and increased to a maximum ( $127 \text{ g m}^{-2}$ ) at 31–40 m depth, decreasing deeper down. This may be due to the heavy erosion and abrasion on the shallow bottoms.

The macrofaunal production, estimated by means of P/B ratios, amounted to  $90 \text{ kJ m}^{-2} \text{ a}^{-1}$ .

Table 1. The benthic macrofauna obtained from the Holmö area in August 1981 (van Veen grab samples,  $n = 30$ , depth interval 7–43 m).

| Taxon                                | Mean abundance                      |     | Mean biomass                      |     |
|--------------------------------------|-------------------------------------|-----|-----------------------------------|-----|
|                                      | ind $\text{m}^{-2} \pm \text{S.E.}$ | %   | $\text{g m}^{-2} \pm \text{S.E.}$ | %   |
| <i>Mesidothea entomon</i> (L.)       | $21 \pm 6$                          | 1   | $1.1 \pm 0.3$                     | 3   |
| <i>Pontoporeia affinis</i> Lindström | $2\,730 \pm 900$                    | 83  | $10.0 \pm 3.8$                    | 23  |
| <i>Macoma balthica</i> (L.)          | $480 \pm 150$                       | 14  | $32.5 \pm 8.4$                    | 74  |
| <i>Prostoma obscurum</i> Schultze    | 82                                  | 2   | 0.3                               | < 1 |
| <i>Oligochaeta</i>                   |                                     |     |                                   |     |
| <i>Gammarus salinus</i> Spooner      |                                     |     |                                   |     |
| <i>Corophium volutator</i> (Pallas)  |                                     |     |                                   |     |
| Chironomidae                         |                                     |     |                                   |     |
| <i>Theodoxus fluviatilis</i> (L.)    |                                     |     |                                   |     |
| <i>Lymnaea peregra</i> (Müller)      |                                     |     |                                   |     |
| Total                                | $3\,310 \pm 1\,020$                 | 100 | $43.9 \pm 9.9$                    | 100 |

Mean number of taxa  $3.5 \text{ } 0.1 \text{ m}^{-2}$

That is, 1/5 of the production calculated for the southern Baltic Proper, half that of the Askö area, but three times that of the Luleå area (Bothnian Bay, Table 2).

Seventy-eight percent of the production was attributed to deposit feeders, 15 % to suspension feeders and 7 % to carnivores. The corresponding values in the Askö area have been estimated at 58, 33 and 9 %, respectively (Fig. 3). Due to the low salinity around Holmöarna, all filter-feeding marine bivalve species except *M. balthica* are absent, so that the deposit feeder production is more important than is the case within the same depth interval in the Baltic Proper. The deposit- and suspension-feeding *M. balthica* is here assumed to be mainly a filter feeder. Thus in the Holmö area, a much higher percentage of the pelagic production enters the benthic food webs directly after its sedimentation, whereas in the Baltic Proper a high portion of the pelagic production is first channelled via filter feeders before it reaches the bottom.

In the Holmö area, *P. affinis* was responsible for 75 % of the macrofaunal production, compared with 46 % in the Askö area (Fig. 3). It showed a two-year life cycle (Fig. 4), which was shorter than expected, since a 3–4 year life span has been reported for neighbouring regions (Samberg 1977). In the southern Baltic Proper, the species has a life cycle of 1 year and in the northern Baltic Proper the life span is 1 or 2 years (Ankar 1977). From Fig. 4 it is clear that the growth in body length of the population in the Holmö area can be at least as high as for populations in the northern Baltic Proper. However, the mean individual weight, calculated from biomass determinations and the body length distributions, showed that individuals of a given length were heavier in the northern Baltic Proper than in the Holmö population. Thus

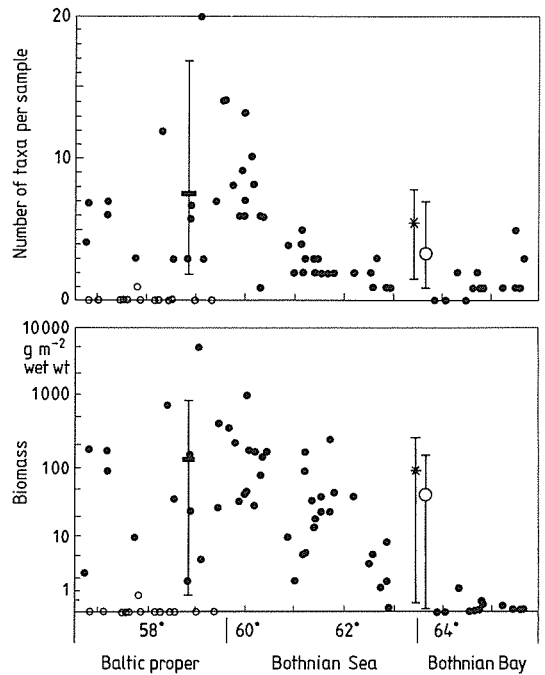


Fig. 2. Geographical distribution of number of macrofaunal taxa (upper) and biomass (lower figure) ( $\text{g m}^{-2}$  wet weight including shells) in the Baltic Sea (filled circles). — the Askö area (9–50 m,  $n = 38$ ),  $\circ$  the Holmö area (7–43 m,  $n = 30$ , Ankar & Leonardsson 1981), \* the Norrby area (7–43 m,  $n = 15$ , number of taxa  $0.3 \text{ m}^{-2}$ , Cederwall & Leonardsson 1984). Vertical lines indicate range of maximum to minimum values and unfilled circles values from oxygen depletion areas. Logarithmic biomass scale =  $\log_{10} [1 + (\text{biomass in g})]$ . (Figures redrawn from Elmgren 1978.)

Table 2. Annual pelagic primary production, sedimentation and benthic macrofaunal production ( $\text{kJ m}^{-2} \text{ a}^{-1}$ ) in different areas of the Baltic Sea.

|                           | Kiel Bay <sup>1</sup><br>southern<br>Baltic<br>Proper<br>15 m | Askö area <sup>1</sup><br>northern<br>Baltic<br>Proper<br>9–50 m | Askö area <sup>2</sup><br>northern<br>Baltic<br>Proper<br>45 m | Holmö area <sup>3</sup><br>northern<br>Bothnian<br>Sea<br>7–43 m | Luleå <sup>4</sup><br>Bothnian<br>Bay<br>0–7 m |
|---------------------------|---------------------------------------------------------------|------------------------------------------------------------------|----------------------------------------------------------------|------------------------------------------------------------------|------------------------------------------------|
| Primary<br>production     | 9 200                                                         | 6 000                                                            | 6 000                                                          | 2 500 <sup>5</sup>                                               | 500                                            |
| Sedimentation             | 4 900                                                         | 2 400                                                            | 2 400                                                          | ..                                                               | ..                                             |
| Macrofaunal<br>production | 440                                                           | 220                                                              | 105                                                            | 90                                                               | 30                                             |

1. Compiled by Ankar 1977, 2. After Cederwall 1979, 3. This study, 4. After Kautsky et al. 1984, 5. After Hagström 1984.

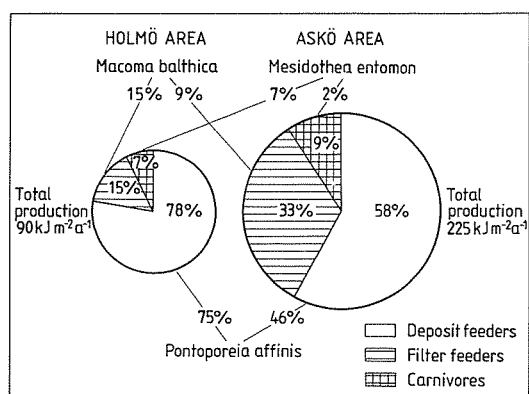


Fig. 3. Composition of benthic macrofaunal production in the Holmö and Askö areas. Percentage share of trophic groups and dominant species common to the two areas.

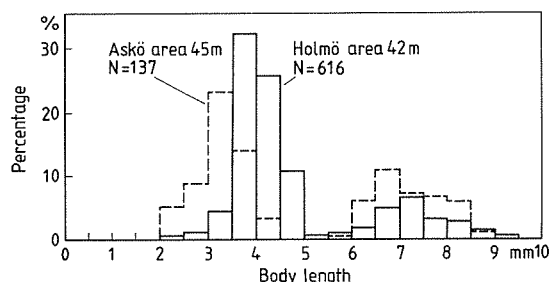


Fig. 4. Percentage size distribution of *Pontoporeia affinis* in the — Holmö area (42 m, n = 616) and — Askö area (45 m, n = 137). The sampling dates were 9 August 1981 and 2 August 1971, respectively.

the somatic growth rate seems to be slower in the latter area.

High metal concentrations (As, Cd, Cr, Cu, Pb and Zn) have been measured in the organic fraction of the sediment in the area (Länsstyrelsen i Västerbottens Län 1975), but the high abundance and biomass of the macrofauna do not reveal any negative effect on the macrofaunal community. Most probably, the metal concentrations are influenced by a nearby metal work, Rönnskärsverket, but leakage from underwater ore-bearing ground can not be excluded.

#### 4. CONCLUSIONS

The Holmö area is poor in macrofaunal species, but the species composition is typical of the northern Bothnian Sea.

Nearly 100 % of the macrofaunal abundance, biomass and production can be attributed to *Mesidothea entomon*, *P. affinis* and *Macoma balthica*. The annual macrofaunal production, estimated at 90 kJ m<sup>-2</sup>, is half or less than half that calculated for areas in the Baltic Proper, but three times that of the Luleå archipelago (Bothnian Bay).

The 2-year life cycle of *P. affinis* found in the study area is shorter than that normally found in adjacent waters. The growth in length seems to be similar to that of populations of the northern Baltic proper, but the somatic growth is most probably slower in the study area.

*P. affinis* contributes as much as 75 % of the macrofaunal production and must be a very important food item for the demersal fish fauna in the area.

Despite the high metal concentrations of the sediments around Holmöarna, the macrofauna showed a high density and biomass.

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## A SHALLOW-WATER FISH COMMUNITY STRUCTURED BY LAND UPLIFT AND DREDGING

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BLOMQVIST, E.M. 1986. A shallow-water fish community structured by land uplift and dredging. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Shallow vegetated bays serve as important spawning and feeding grounds for many fish species in the Baltic. Due to land uplift, areas of this type gradually become isolated from the sea. In some cases dredging has been used to prevent the formation of coastal lakes.

Studies on the fish community structure and fish migration activity were conducted in a semi-isolated bay on Åland before and after dredging.

The fish community size and species composition remained relatively unchanged until 1984, but dredging resulted in increased overall migration activity and distinct changes in species dominance and population densities. The changed conditions after dredging (increased feeding and spawning area) altered the interspecific interactions of the species; some species benefited by the habitat changes, other species decreased and some were unaffected.

Index words: Fish ecology, community structure, dynamics, land uplift, dredging

### 1. INTRODUCTION

Along the Finnish coast, certain shallow bays rich in vegetation are slowly isolated from the sea as a consequence of post-glacial land uplift. Such areas, in both brackish and marine environments, are ecologically important as spawning and feeding grounds for fish (Allen 1982, Blomqvist 1984).

Fish community studies have been conducted in inner archipelagoes of the Baltic by e.g. Anttila (1972), Hansson (1979, 1980), Neuman (1977, 1982), Andreasson and Petersson (1982), Blomqvist (1982, 1984) and Thorman (1983), who described different aspects of the biology and ecology of shallow-water fish assemblages. In the present study, the fish community in one such area, the bay Gloet on NW Åland (Fig. 1), was assessed during an isolation phase (1975, 1976), when the bay was connected with surrounding waters only

by a narrow, shallow (0.3 m) inlet, and a later phase (1978 onwards), when dredging had reconnected Gloet with the adjacent archipelago.

The aims of this paper are to characterize the fish community structure, and to examine the ecological changes induced by the dredging.

### 2. MATERIAL AND METHODS

The fish community was monitored by regular gill net sampling during 1975, 1976 (isolation phase) and 1978—1984 (post-isolation phase). The migration activity of the fish was assessed during 1975, 1976 and 1978, 1979 by closing the inlet with fish

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traps and catching the fish leaving and entering the bay (Fig. 2). The diet of the most abundant species of fish was determined by taking stomach samples from gill net, trap and seine catches. A more detailed description of the study area and methods is given in Blomqvist (1984). The basic hydrographical features are shown in Fig. 3.

### 3. RESULTS

The most abundant species in the fish community were: roach *Rutilus rutilus* (L.), perch *Perca fluviatilis* (L.), white bream *Blicca bjoerkna* (L.) and rudd *Scardinius erythrophthalmus* (L.). Other species caught were bleak *Alburnus alburnus* (L.), pike

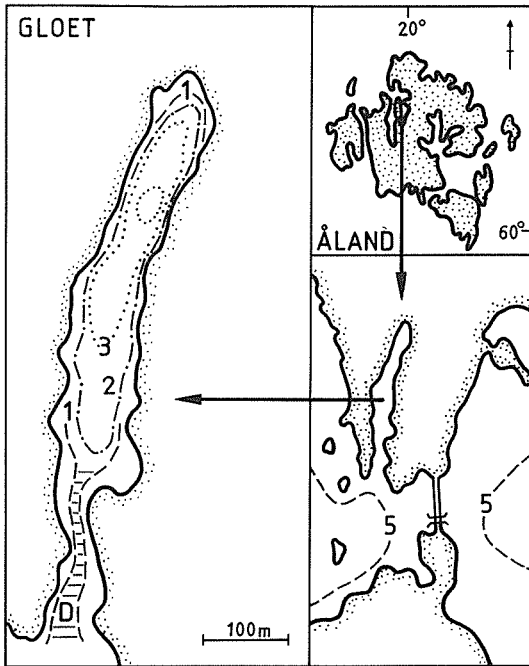


Fig. 1. The study area, Gloet bay, in the NW Åland archipelago. D = dredged area. Depth curves (m) indicated.

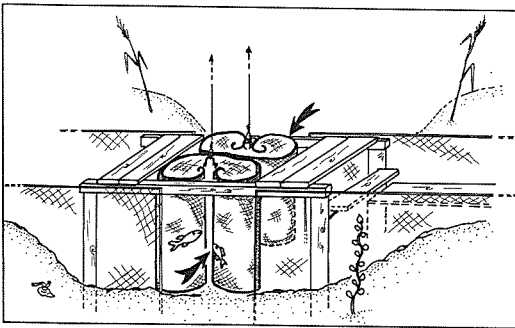


Fig. 2. The traps used to assess the fish migration in and out of Gloet bay.

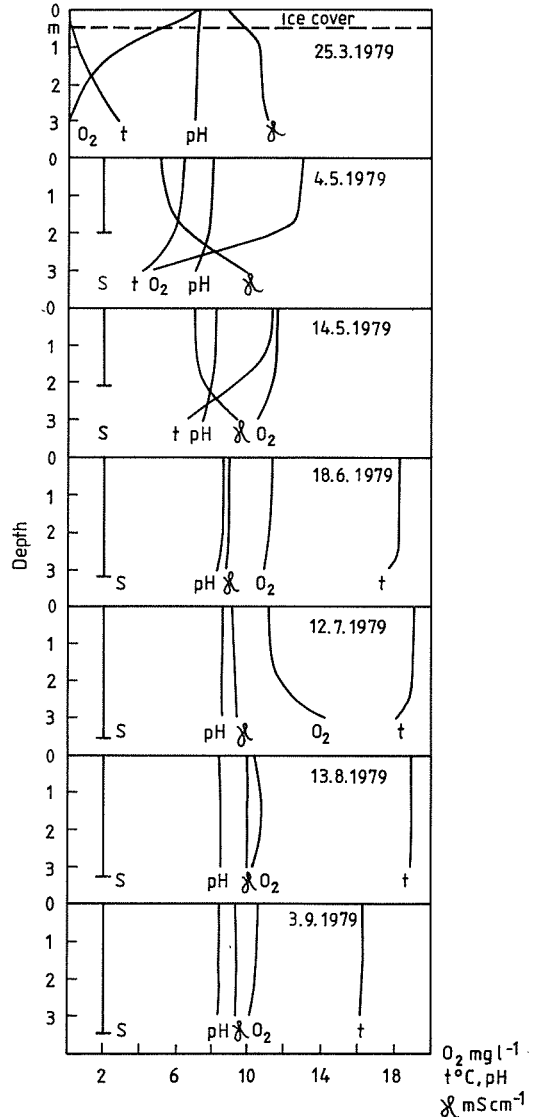


Fig. 3. Hydrographical conditions in Gloet bay in 1979: Secchi depth (S), oxygen ( $O_2$   $mg\ l^{-1}$ ), temperature ( $^{\circ}C$ ), pH and conductivity ( $ms\ cm^{-1}$ ).

*Esox lucius* (L.) and bream *Abramis brama* (L.). Species present in the gill net catches only after dredging were ruffe *Acerina cernua* (L.), smelt *Osmerus eperlanus* (L.), ide *Leuciscus idus* (L.) and crucian carp *Carassius carassius* (L.). The catch data (number of species, total abundance and total biomass) are given in Fig. 4. The dominance pattern of the constituent species (perch, roach, rudd and white bream) is presented in Fig. 5.

The migration study (Fig. 6) revealed a more than tenfold increase in the numbers of fish frequenting Gloet after dredging.

The food choice of the most important species, briefly presented in Table 1, shows a distinct overlap in diet between e.g. perch, roach and ruffe. These species also showed a considerably wide food choice spectrum.

#### 4. DISCUSSION AND CONCLUSIONS

The catch data clearly demonstrate the importance of shallow archipelagoes as spawning and feeding grounds for most fish populations. In both the gill net (Fig. 4) and trap catches (Fig. 6) peak values

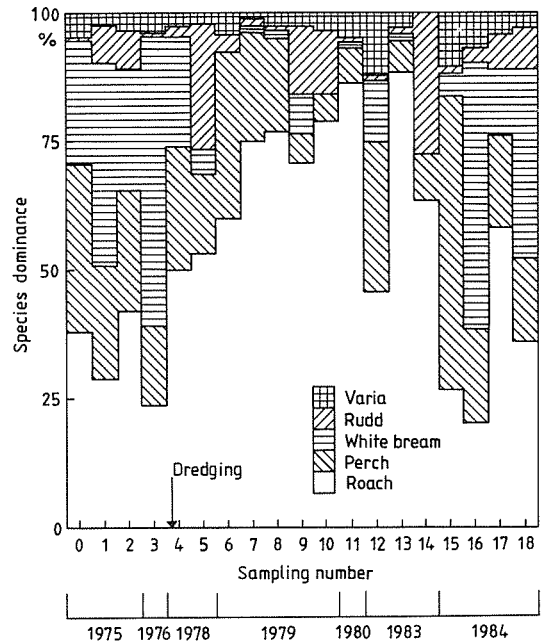


Fig. 5. Species dominance (% of total abundance) in the fish community in 1975–1984 according to gill net samplings. (Sampling no:s 0–18; see Fig. 4).

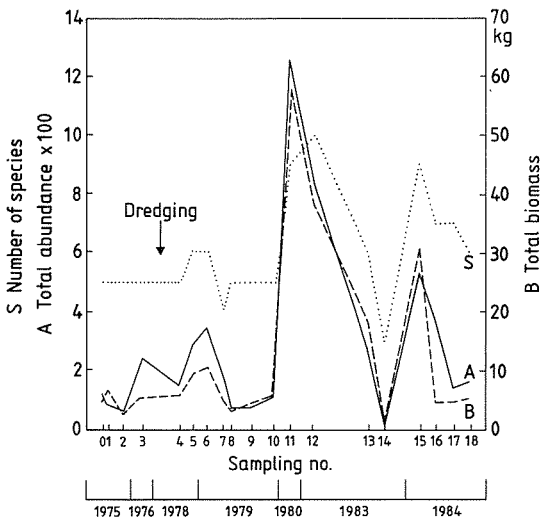


Fig. 4. Total catch data (S = number of species, A = total abundance, B = total biomass) from the gill net samplings before (1975, 1976) and after (1978–1984) dredging. Sampling dates: 1975: no 0 = 9.7, no 1 = 15.7, no 2 = 16.8, 1976: no 3 = 6.7, 1978: no 4 = 29.6, no 5 = 24.8, 1979: no 6 = 5.6, no 7 = 4.7, no 8 = 25.7, no 9 = 4.9, no 10 = 23.10, 1980: no 11 = 13.5, 1983: no 12 = 17.5, no 13 = 10.9, no 14 = 12.11, 1984: no 15 = 16.5, no 16 = 18.6, no 17 = 24.7, no 18 = 15.8.

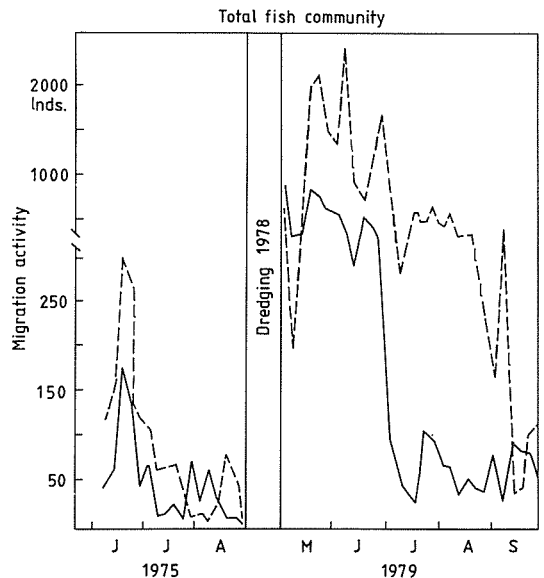


Fig. 6. The migration of the fish community before (1975) and after (1979) dredging (total catches in the traps for 5-day periods). Solid line = inward migration, broken line = outward migration.

Table 1. Food choice of the most important fish species of the bay Gloet.

| Food                  |             |             |              |          |        |               |          |             |           |       |        |         |         |           |          |           |              |          |             |         |         |        |            |         |       |        |              |                |             |       |             |          |   |
|-----------------------|-------------|-------------|--------------|----------|--------|---------------|----------|-------------|-----------|-------|--------|---------|---------|-----------|----------|-----------|--------------|----------|-------------|---------|---------|--------|------------|---------|-------|--------|--------------|----------------|-------------|-------|-------------|----------|---|
| Species               | Insecta spp | Trichoptera | Chironomidae | Sialidae | Nereis | Crustacea spp | Gammarus | Pontoporeia | Corophium | Iaera | Idotea | Asellus | Mysidae | Ostracoda | Copepoda | Cladocera | Mollusca spp | Bithynia | Hydrobiidae | Lymnaea | Cardium | Macoma | Pisces spp | Rutilus | Perca | Clupea | Gasterosteus | Pomatoschistus | Plantae spp | Chara | Potamogeton | Detritus |   |
| Perca fluviatilis     | +           | +           | +            | +        | +      | +             | +        | +           | +         | +     | +      | +       | +       |           | +        | +         | +            | +        | +           | +       |         | +      | +          | +       | +     |        | +            | +              | +           | +     | +           |          |   |
| Rutilus rutilus       | +           | +           | +            | +        |        | +             | +        | +           |           |       |        | +       |         | +         | +        | +         | +            | +        |             | +       | +       | +      | +          | +       | +     |        |              |                | +           | +     | +           | +        |   |
| Blicca bjoerkna       | +           | +           | +            |          |        | +             |          |             |           |       |        |         |         | +         | +        | +         | +            |          |             | +       |         | +      | +          |         |       |        |              |                | +           | +     | +           | +        |   |
| Scardinius erythropt. | +           |             | +            |          |        | +             |          |             |           |       |        |         |         |           |          |           | +            |          |             | +       |         |        |            |         |       |        |              |                | +           | +     |             |          | + |
| Esox lucius           |             | +           | +            |          | +      | +             | +        | +           |           |       |        | +       |         |           |          |           | +            |          |             |         |         |        |            | +       | +     | +      | +            |                |             |       |             |          |   |
| Acerina cernua        | +           | +           | +            | +        | +      | +             | +        | +           | +         |       | +      | +       |         |           | +        | +         | +            |          |             |         | +       |        | +          | +       | +     | +      |              |                |             | +     | +           | +        |   |
| Abramis brama         | +           |             | +            |          |        |               |          |             |           |       |        | +       |         |           | +        |           | +            |          | +           |         |         |        |            |         |       |        |              |                | +           |       |             |          | + |

were always obtained during spring and early summer, when adults frequented Gloet for spawning and juveniles utilized it as a nursery.

The community size and biomass values varied greatly within and between years, so that no significant increase could be demonstrated during 1975–1980 (see also Blomqvist 1984). The catches in 1984, however, indicate a slight increase in abundance and biomass values (Fig. 4, and Blomqvist and Mattila, unpublished data).

The alteration of the habitat by dredging (increased area for dispersal) resulted in a distinct change in the species dominance pattern. During the isolation, 1975, 1976, roach and white bream were dominant in the fish assemblage, along with perch. Due to changed interspecific interactions (competition for food and space) after dredging, roach and rudd increased in numbers and importance. This was also shown for ruffe and pike (Blomqvist 1984). The population of white bream decreased immediately after dredging (Fig. 5), while the perch population remained relatively stable. In 1984, however, drastic changes in the fish community structure occurred. White bream, and to some extent perch, regained their relative numerical importance at the expense of roach (Fig. 5, and Blomqvist and Mattila, unpublished data). This may indicate that a stabilization phase is being reached in the fish community after dredging, as the standing stocks in the area now seem higher (Fig. 4) with on average more species present than in pre-dredging years.

Thus the reconnection of the bay with the sea resulted in an overall increase in the diversity and production of the fish community. The initial changes in the dominance pattern after dredging may have been caused partly by altered niche-dimensions within the bay due to the increased

amount of fish utilizing the food resources (cf. Fig. 6; increased overall fish migration). This seems likely, as the dominant species, roach, perch and white bream, feed partly on the same food items (Table 1). This hypothesis is further supported by data indicating changes in the diets of several species after dredging (Blomqvist and Mattila, unpublished data).

In summary, this study shows the importance and potentials of shallow inner archipelago areas and sheltered bays for fish production, when the effects of land uplift are counteracted.

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## PATTERNS IN POST-IMPACT SUCCESSIONS OF ZOOBENTHOS FOLLOWING PHYSICAL AND CHEMICAL DISTURBANCE IN THE NORTHERN BALTIC SEA

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& Carl-Sture Österman<sup>1)</sup>

BONSDORFF, E., LEPPÄKOSKI, E. & ÖSTERMAN, C.-S. 1986. Patterns in post-impact successions of zoobenthos following physical and chemical disturbance in the northern Baltic Sea. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Several reports exist on man's negative impact on benthic assemblages in the northern Baltic Sea, but few studies describe the recovery after instantaneous physical destruction (dredging, sand suction) or diffuse chemical disturbance (municipal waste water, industrial pollution). The recovery potential of Finnish coastal waters is discussed on the basis of 7 long-term case studies. The response of the zoobenthic community is presented in species-abundance-biomass graphs, and compared with experimental colonization data. Generally the magnitude, chemical duration and secondary effects of the impact influence the pattern of recovery. Three categories of community recovery are distinguished: 1. Permanent, 2. Intermittent, and 3. Delayed. Field experiments are suggested as an alternative means of monitoring and measuring the recovery potential of coastal waters.

Index words: Zoobenthos, pollution abatement, recovery potential, succession, dynamics, defaunation experiments

### 1. INTRODUCTION

From the northern Baltic Sea and southern parts of the Bothnian Sea several reports exist on man's impact on the benthic ecosystem. There is evidence not only of chemical pollution (including eutrophication), but also of several other forms of disturbance, affecting the biotic and abiotic properties of the coastal and archipelago waters. Thus the wash effects of ship's traffic, the effects of dredging and exploration of subaquatic mineral resources, and different aspects of engineering works in coastal waters have been studied (for references, see Bonsdorff 1983, Bonsdorff et al. 1984, Leppäkoski and Bonsdorff 1984). Informa-

tion exists on the chemical fate and short-term biological effects of chemical pollution (e.g. Linden et al. 1979), and the immediate negative effects of engineering works (Karlsson 1983) but, there are few reports on the recovery of physically or chemically damaged or destructed Baltic ecosystems. Pearson and Rosenberg (1978), Boesh and Rosenberg (1981) and Pearson (1981) have tried to review the responses (including restoration) of marine benthic ecosystems to different kinds of mainly organic pollution, but no such analysis exists for Baltic conditions, although the general ecology of the Baltic Sea is well documented (Elmgren 1984). The aim of this paper is to give a brief survey of some recovery processes along the

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Finnish SW coast mainly using the macrozoobenthic communities as an example. In this context critical questions are: What happens if the sources of pollution or physical disturbance are eliminated or significantly reduced? How long a time is needed for recovery from seriously damaged conditions to a natural or semi-natural state? Can the recovery process be predicted, and can some general classification be made on the improvement at community level? And is there any realistic alternative to costly long-term field studies or toxicity tests in aquaria? A brief introduction to the problem was given by Leppäkoski and Bonsdorff (1984).

## 2. CASES STUDIED

The cases presented in this paper are shown in Fig. 1, and all represent several years of post-impact development. Cases nos. 1, 2 and 3 (the recipient of a former sulphite pulp mill, the area off a former fish industry plant and the recipient of an oil refinery) represent former point-sources of chemical pollution, whereas case no. 5 (a municipal waste water recipient) has been affected by a diffuse eutrophication load. Examples 4, 6 and 7 represent recovery from mechanical destruction of the benthic communities (dredging and sand suction). Locality no. 7 has also been used for colonization experiments since 1982 in order to develop a sensitive tool for "recovery monitoring".

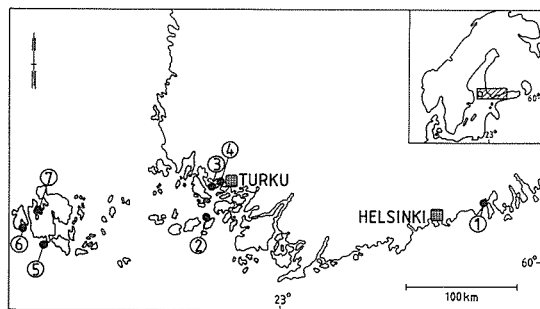


Fig. 1. Locations of cases studied. 1 = former sulphite pulp mill (Tolkis), 2 = former fish industry plant (Nauvo), 3 = oil refinery (Naantali), 4 = large scale dredging (Raisio Bay), 5 = municipal sewage (Mariehamn), 6 = subaquatic exploitation of sand (Degersand Bay), 7 = small scale dredging, and site of field experiments (Gloet Bay).

## 3. MATERIAL AND METHODS

In all cases, 5–10 replicate grab samples of the zoobenthos were taken on each sampling date (at least once a year). The sampling design varied to some extent (Bonsdorff 1983, Leppäkoski and Bonsdorff 1984), but the recommendations given for the Baltic Sea were generally followed (Dybern et al. 1976). For the colonization experiment boxes ( $30 \times 40 \times 11$  cm) filled with defaunated (deep frozen, 48 h) sediment were used. The method is described in more detail by Bonsdorff (1983) and Bonsdorff and Österman (1985). The results are mainly presented as primary data in graphical form (species-abundance-biomass curves, as used by Pearson and Rosenberg 1978).

## 4. RESULTS

The zoobenthic recovery following the abatement of the effluents from a sulphite paper mill has been slow, and displays a pattern governed by depth and distance from the former point of pollution (Fig. 2). In an area within 1.0 km of the mill benthic community establishment has been poor, but the biomass values 1.5 km from the mill (12 m depth) are comparable to those in other archipelago areas. During the first three years of recovery, benthic immigration into the formerly more or less lifeless bottoms was slow, and in this case even 10 years seems to be insufficient for full recovery of the benthos.

The long-term community responses to the wastes of a fish industry plant are shown in Fig. 3. The plant was in operation 1971–75, causing total elimination of the fauna below 12–15 m depth in the vicinity of the plant. Leppäkoski et al. (1983) described the initial phase of recovery, which was rapid, being slowest at the deepest locality (Fig. 3 c). In this case zoobenthic recovery can be coupled also with changes in the sediment, as the loss on ignition (a rough estimate of organic content) has declined from 8.5 to 3.0 %.

The benthic community showed an immediate positive response to a drastic reduction in the effluents from an oil refinery in 1973 (Fig. 4). The bivalve *Macoma balthica* (L.) was especially favoured by the improved quality of the water and sediment (Fig. 4 b). The detoxification of the sediment and the initial recovery were described by Leppäkoski and Lindström (1978), and Leppäkoski

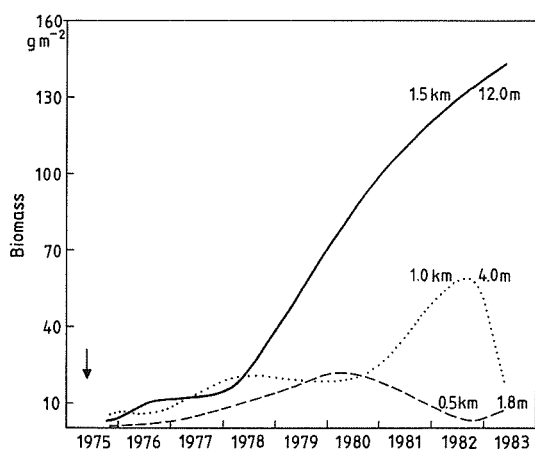


Fig. 2. The course of the macrozoobenthic biomass ( $\text{g ww m}^{-2}$ ) after 75 years of effluent discharge from a sulphite pulp mill (arrow indicates termination of pollution) (depth: m, distance: km).

and Bonsdorff (1984) recorded stabilization of community recovery at nearshore (3–11 m depth) localities. In deeper parts of the recipient (Fig. 4; 24 m) the shortterm response was dramatic, with an increasing number of species, a peak in community abundance, but a time lag in biomass development (Fig. 4 a). Between 1976 and 1982 a stabilization phase occurred in the community, and in 1982–83 the community seemed to have reached equilibrium with the nearby recruitment areas. The time lag in development of the biomass is due to the slow growth and long life span of *Macoma balthica* (Fig. 4 b). Thus it took 8–10 years for the community to recover from severe chronic oil pollution.

The recovery process following dredging and sand suction (localities 4, 6 and 7 in Fig. 1) has been thoroughly described by Bonsdorff (1980, 1983), Bonsdorff et al. (1984) and Leppäkoski and Bonsdorff (1984). These studies indicate that the recovery potential of shallow brackish water ecosystems is high following mechanical destruction, although it may take about 5 years before dynamic equilibrium is reached.

The response of the zoobenthic community to an improvement in the treatment of municipal

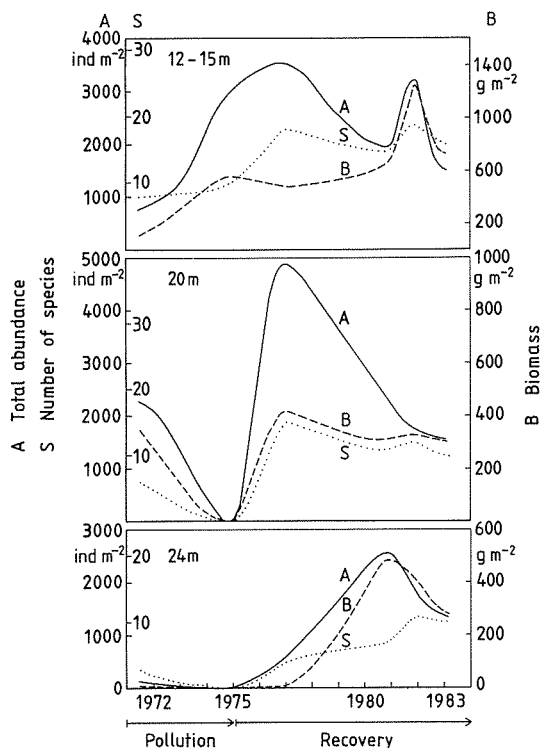


Fig. 3. Dynamics of the zoobenthic community in different depths before (1971–75) and after (1975–83) the abatement of pollution by a fish industry plant. S = number of species, A = total abundance ( $\text{ind m}^{-2}$ ), B = biomass ( $\text{g ww m}^{-2}$ ).

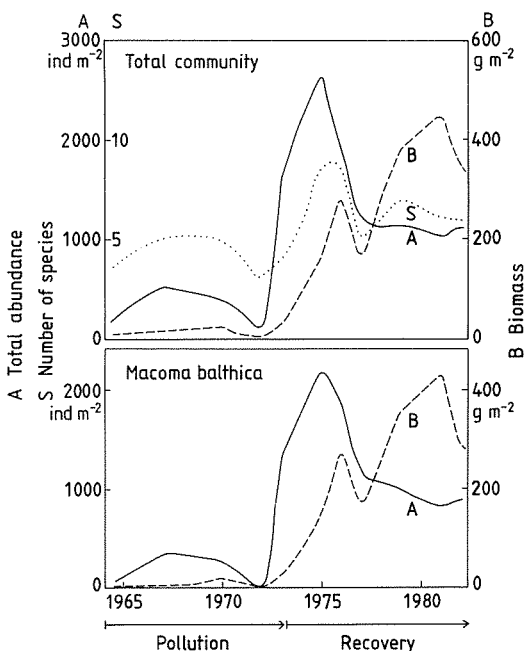


Fig. 4. Dynamics of the zoobenthic community (upper part) and *Macoma balthica* population (lower part) before (1965–71) and after (1971–82) a decrease in the discharge of oil refinery effluents. For explanations of symbols, see Fig. 3.

waste water in 1979 (locality 5, Fig. 5) was remarkable at a shallow (2 m) station; the abundance values peaked within two years, diversity (species number) and biomass peaked about one year later and in 1983–84 the community parameters seemed to have stabilized at an intermediate level. Thus a real improvement has taken place, especially compared with the situation in the years immediately before the treatment change. This improvement is also occurring on deeper (12–15 m) bottoms, although the process is somewhat slower (Leppäkoski and Bonsdorff 1984).

In the experimental study, the colonization by the benthos was rapid, basically due to the short immigration distance, but largely followed the same pattern shown in Figs. 3, 4 and 5 (Fig. 6). This field experiment was conducted in a previously dredged area (Bonsdorff 1983), and thus a direct comparison can be made between the natural and experimental patterns of recovery (Fig. 6 a, b). The community development during the 6 months of the experiment and the 6 years following dredging is surprisingly similar. The biomass does not reach stability within 6 months, but reaches the level of the surrounding area. The abundance values even exceed those in the recruitment community, whereas the number of species stabilizes at the same level. Thus a small-scale, short-term experiment contained as much information about the recovery potential of the community, as 6 years of monitoring studies.

## 5. DISCUSSION AND CONCLUSIONS

These case studies suggest that the zoobenthic recovery patterns are fairly uniform. This is in accordance with data from other areas (see Bonsdorff 1983 and Bonsdorff et al. 1984 for references). Gray (1982) distinguished between mechanical interference, and chemical stress, listing adaptations in zoobenthic communities to both types of disturbance. In cases of dredging and submarine exploitation of sand, the initial factor affecting the benthos was the instantaneous, physical removal of the habitat. In these cases initial recovery was rapid, and dominated by r-strategists. In cases of high chemical stress the productivity of the individual is reduced (or the fauna is totally eliminated) over time. Reduction of stress will give gradual recovery, with *Macoma*

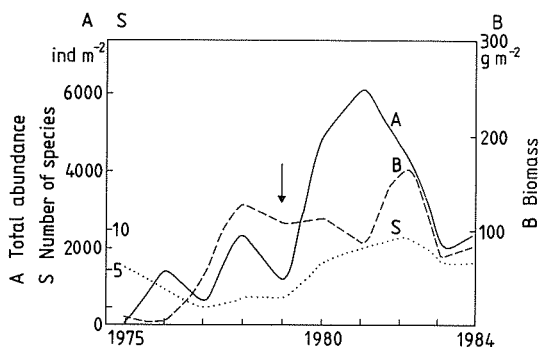


Fig. 5. Dynamics of the zoobenthic community in relation to improved treatment of municipal sewage water (arrow indicates the onset of improved treatment). For explanations of symbols, see Fig. 3.

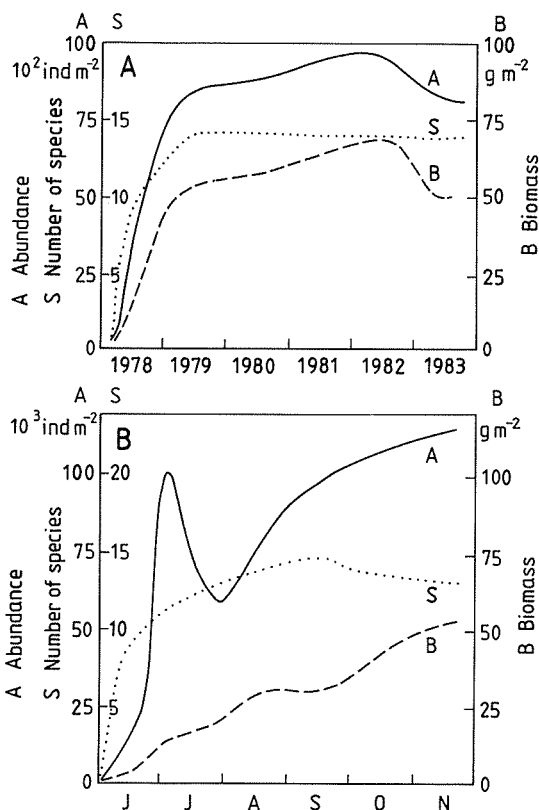


Fig. 6. Dynamics of the zoobenthic community establishment and succession after small scale dredging (A), and during a field colonization experiment (B). For explanations of symbols, see Fig. 3.

*balthica*, for example playing an important role from the onset of recolonization. Details of the succession following physical and chemical disturbance are given by Bonsdorff (1983) and Leppäkoski and Bonsdorff (1984).

This information allows classification of the types of recovery, but it must be borne in mind that the pattern is influenced by the magnitude of the impact, duration of the chemical pollution and secondary effects. The secondary effects may be even more serious than those of the original source of stress, as they may persist in the benthic assemblages for an indefinite period. In case 1 (Fig. 2), for example the reason for the slow recovery close to the former point of pollution is the character of the sediment (a thick layer of cellulose fibres covers the bottom), and in case 6 subaquatic sand exploitation altered the bottom topography sufficiently to cause an annual oxygen deficiency due to stagnation in the water column. This has led to a cyclic recovery pattern, where equilibrium or stabilization is never reached (Bonsdorff 1983).

Three categories of recovery can be distinguished for the benthos of Finnish coastal waters (Leppäkoski and Bonsdorff 1984):

1. **Permanent recovery;** a state of dynamic equilibrium is reached within about 5 years (e.g. case 4 in Fig. 1). The colonization pattern is fairly predictable.
2. **Intermittent recovery;** the final stabilization is prevented by secondary effects of the initial impact (case 6). Short-term predictions can be made, when the secondary effects are known.
3. **Delayed recovery;** the final stabilization is delayed by slow recovery of the sediment (case 1), and the recovery process is more or less unpredictable.

As recovery processes always seem to demand several years, monitoring the succession is tedious and expensive. Due to the relative similarity of recovery patterns between the cases described, colonization experiments (Fig. 6 b) can be a useful tool in monitoring any particular case (Bonsdorff and Österman 1985). Thus we suggest that this method should be considered for the purpose of offensive and preventive marine environmental protection.

## ACKNOWLEDGEMENTS

We thank the Maj and Tor Nessling Foundation for financial support in 1982–84.

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## STATE OF THE SWEDISH COASTAL ZONE OF THE GULF OF BOTHNIA

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CEDERWALL, H. 1986. State of the Swedish coastal zone of the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Descriptions are given of areas selected in relatively unaffected regions and in regions directly influenced by human activities. In the northern Bothnian Sea the mean annual phytoplankton primary production was 64 g m<sup>-2</sup> C and in the southern part the values were 20 % higher. An increase of the benthic biomass has been found in areas not directly affected by pollution, which may indicate slight eutrophication. Conditions have improved in most of the areas along the coast receiving pollutant loads; the concentration of dissolved oxygen has increased and the benthic fauna has recovered. Marked improvements have been noted in only a few cases, however.

Index words: Baltic Sea, Gulf of Bothnia, coastal zone, water quality, nutrients, primary production, sedimentation, soft bottom macrofauna, pollution load

### 1. INTRODUCTION

This assessment of the condition of the Swedish coastal zone of the Gulf of Bothnia was made on the initiative of the executive group of the Committee for the Gulf of Bothnia. For practical reasons the paper is divided into two parts; the first deals with areas without direct anthropogenic influence (according to results from research projects and the National Swedish Monitoring Programme) and the second with areas directly influenced by industries and municipalities (results from local programmes for monitoring waste outfall areas).

### 2. MATERIAL AND METHODS

The materials and methods differ between the many projects cited in this paper. Since there is not

room for a description of all the methods here, the reader is referred to the references given in the text.

### 3. RESULTS

#### 3.1 Areas not directly exposed to pollution

##### 3.1.1 The pelagic zone

Since 1979, intensive studies have been carried out in the Norrby archipelago, south of Umeå (Fig. 1), 15—18 samplings being performed per year.

The nutrient levels measured here do not deviate from those recorded in the open northern Bothnian Sea (Erland Bergstrand, pers. comm.).

Primary production has been measured with the unfiltered water technique (Larsson and Hagström

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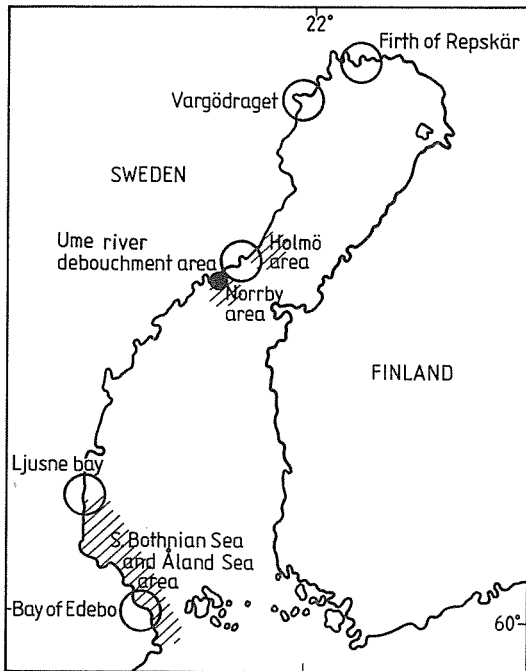


Fig. 1. The site of intensive pelagic monitoring (small filled circle), the areas of extensive macrobenthos studies (shaded) and the waste discharge areas (large circles) dealt with in this paper.

1982). During the period 1979–1983 the mean annual production was  $64 \text{ g m}^{-2} \text{ C}$  (Fig. 2). The production seems to have decreased from 1980 to 1981. At present there is no explanation for this, but the decrease lies within the range of inter-annual variation measured in the Askö area (Larsson 1983). Measurements carried out in the Öregrund area, southern Bothnian Sea (Torbjörn Willén, pers. comm.), suggest that a decrease may have occurred there as well.

The bacteria account for an unusually large part of the primary production in the Norrby area (Hagström et al. 1983, Hagström 1984) (Fig. 2).

The sedimentation in the Norrby archipelago (expressed as  $\text{g m}^{-2} \text{ year}^{-1}$  dry weight) is about 3/4 of that in the Askö area (Hagström unpubl., Larsson unpubl.), but the primary production is only half as great. This fact, together with the high proportion of bacterial production, is very likely due to the high supply of organic matter from rivers debouching near the archipelago.

An increase of the primary production towards the south in the Baltic (Table 1 and Fig. 3) has also been found earlier (Elmgren 1978). The measurements carried out in the Öregrund area in 1977–1978 (Lindahl and Wallström 1983) gave values

about 20 % higher than the mean value for the Norrby area. If only the Norrby values from 1979 and 1980 are considered, however, the production seems to be about the same.

### 3.1.2 The benthos

Extensive studies of the macrozoobenthos have been carried out in recent years (Fig. 1). In 1980, an inventory of the benthos in the Holmö area was made in preparation for the establishment of a marine reserve (Ankar and Leonardsson 1986). Within the National Swedish Environmental Monitoring Programme, monitoring of the soft-bottom macrofauna was started in the Gulf of Bothnia in 1983. Sampling was carried out in the Norrby archipelago and in the southernmost region, the Åland Sea and the southern Bothnian Sea (Fig. 1).

Looking at the biomass values obtained in these and some other studies (Fig. 3), we find that they increase towards the south. The difference between the Bothnian Bay and the Bothnian Sea is especially obvious. This phenomenon is not restricted to the coastal areas and archipelagos but is also observed in the open sea (Elmgren 1978).

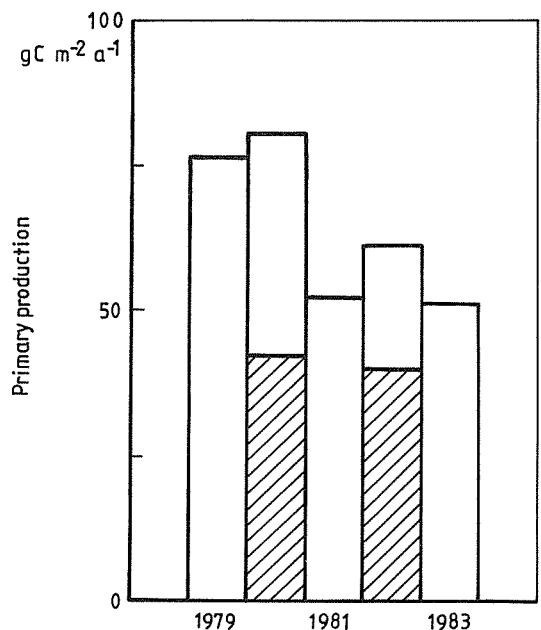


Fig. 2. Annual primary production in 1979–1983 in the Norrby area, northern Bothnian Sea. The shaded parts of the bars show the proportion contributed by the bacteria. (Compiled from Hagström et al. 1983, Hagström 1984 and unpublished).

Table 1. Pelagic primary production and benthic secondary production ( $\text{kJ m}^{-2} \text{a}^{-1}$ ) in different parts of the Baltic Sea. Data for the Norrby area and the Southern Bothnian Sea from 1) Hagström et al. 1983, Hagström 1984 and unpubl., 2) Lindahl and Wallström 1983 and 3) Cederwall and Leonardsson unpubl. Other data from Ankar and Leonardsson 1986.

|                                | Kiel Bay | Askö area | Southern Bothnian Sea | Norrby area        | Holmö Islands | Luleå area |
|--------------------------------|----------|-----------|-----------------------|--------------------|---------------|------------|
| Benthos depth range (m)        | > 15     | 9–50      | 9–199                 | 8–126              | 7–43          | 0–7        |
| Primary production             | 9 200    | 6 000     | 3 000 <sup>2</sup>    | 2 500 <sup>1</sup> | —             | 500        |
| Macrofauna production (summed) | 440      | 220       | 160 <sup>3</sup>      | 160 <sup>3</sup>   | 90            | 30         |

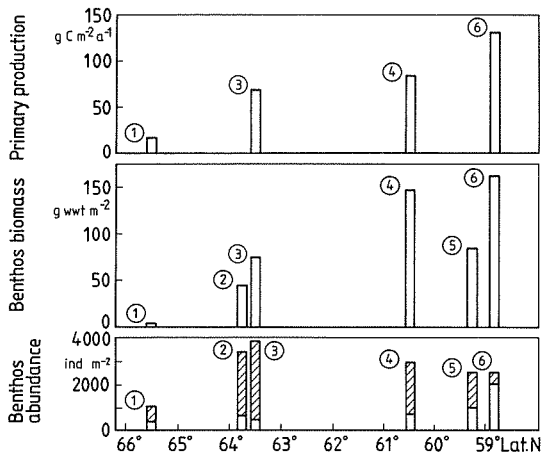


Fig. 3. Annual primary production, benthos biomass and benthos abundance in some coastal areas in the northern Baltic Sea. The shaded parts of the bars in the lowest diagram show the proportion of *Pontoporeia affinis*. 1 = the archipelago of Luleå, 2 = the Holmö area, 3 = the Norrby area, 4 = southern Bothnian Sea and Åland Sea, 5 = the archipelago of Stockholm, 6 = the Askö-Landsort area. (Compiled from Wulff et al. 1977, Hagström et al. 1983, Hagström 1984 and unpubl., Lindahl and Wallström 1983, Ankar and Leonardsson 1986, Cederwall and Leonardsson unpubl. and Ankar unpubl.).

The secondary production differs greatly between the Bothnian Bay and the Bothnian Sea (Table 1), but seems to be about the same in the northern and southern parts of the Bothnian Sea. This can possibly be explained by a greater input of organic matter by rivers in the northern part. On the other hand, the difference in the primary production between the northern and southern Bothnian Sea may not be very significant (see above). The fact that the biomass is higher in the southern part of the Bothnian Sea, in spite of the secondary production being the same, is due to differences in the species dominance. In the

southern part *Macoma balthica* is more common, and this species has a lower P/B ratio than *Pontoporeia affinis*, which is more dominant in the northern part of the Bothnian Sea.

The benthos stations visited in 1983 in the Åland Sea and the southern part of the Bothnian Sea had been sampled in the 1920s (Hessle 1924) or 1950s (Sellerberg 1961). The biomass values found in 1983 were significantly higher than those recorded earlier (Cederwall et al., unpubl.). Preliminary results indicate that they may have been 4-fold those for the 1920s and 2.5-fold those for the 1950s.

### 3.2 Areas receiving pollutant wastes

Of the many areas receiving waste loads, five have been selected for this paper (Fig. 1). The choice was mainly based on the scope of the monitoring programmes and the amount of data available.

#### 3.2.1 Firth of Repskär

The load of organic matter (measured as BOD<sub>7</sub>) from the pulp and paper mill Karlsborgsverken was reduced by 40 % in the middle of the 1970s. From then on there were no changes until 1982, when the load was decreased by a further 30 % (Fig. 4).

The oxygen supply seems to have increased in the area close to the outlet from the industry, while no change has been recorded further out in the firth (Fig. 4) (Råghall 1982).

Decreases in the abundance of the macrozoobenthos were recorded during the 1970s at the station adjacent to the outlet and the nearest 10 stations (Fig. 5) (Landner et al. 1977, Lindeström and Landner 1976, Lindeström 1980). In 1982, however, both the abundance and the species

number seem to have increased (Lindeström 1982), though unfortunately the station closest to the outlet had to be moved 100 m in 1982. Similar changes were not found in the outer area of the firth, though the 1982 abundance value was the highest measured (Fig. 5).

In 1976, 1979 and 1982, 20 samples were collected each year from an intensive sampling area. The changes recorded (Lindeström 1982) disagree with the results given above, and indicate a deterioration of the situation. Taxa sensitive to pollution have decreased, and tolerant taxa have increased (Fig. 6).

In 1974 low concentrations of chlorophyll *a* coupled with high concentrations of phaeophytin were found at several stations (Länsstyrelsen i Norrbottens län 1978). This was regarded as an effect of the pollution load. In 1979 a similar situation was noted at only one station (Larsson 1980).

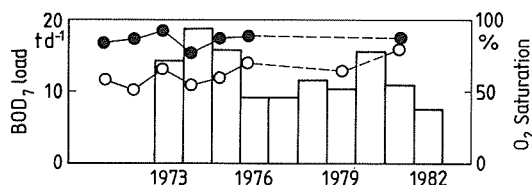


Fig. 4. Annual load of BOD<sub>7</sub> imposed by Karlsborgsverken on the Firth of Repskär (columns; data from Karlsborgsverken) and annual mean values of oxygen saturation measured in the inner (open circles) and outer part (filled circles) of the firth (After Råghall 1982).

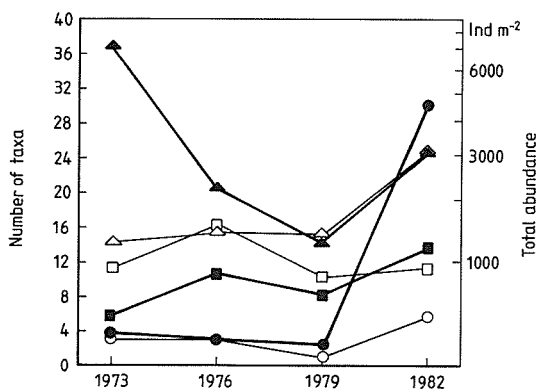


Fig. 5. Variations in the number of taxa (open symbols, thin lines) and abundance of the macrozoobenthos (filled symbols thick lines) close to the outlet (circles), and in the inner (triangles), and outer part (rectangles) of the Firth of Repskär. Compiled from Ladner et al. 1977, Lindeström and Ladner 1976, Lindeström 1980 and Lindeström 1982.

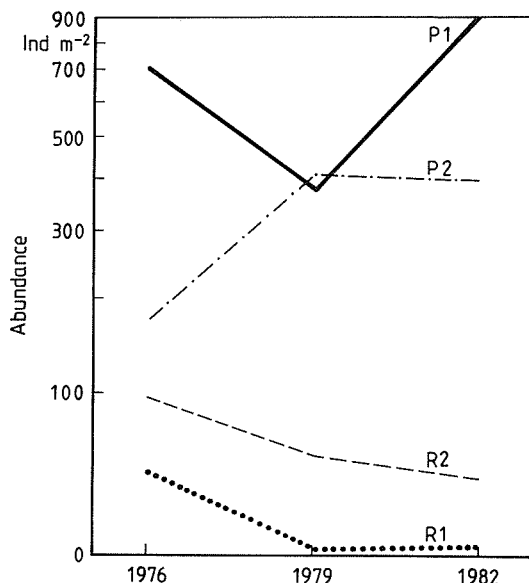


Fig. 6. Variations in the abundance of the four indicator species groups (Acc. to Leppäkoski 1975) in a sampling area in the inner Firth of Repskär. P1 = progressive species of the first order, P2 = progressive species of the second order, R2 = regressive species of the second order, and R1 = regressive species of the first order. (Compiled from Lindeström & Landner 1976, Lindeström 1980 and Lindeström 1982).

### 3.2.2 Vargödraget

The pollution load increased considerably in the early 1970s, when a part of the waste water outfall to the inner Pit firth was moved to Vargödraget (Fig. 7). The load decreased after a few years and has subsequently been about 12–13 tonnes BOD<sub>7</sub> per year.

The oxygen saturation values increased significantly during the period 1973–1978 (Fig. 8) (Råghall 1980). They decreased during 1979 and 1980, only to increase again during 1981–1982 (Lövgren 1981, 1982, 1983).

The benthos was studied in 1974 and 1979 (Landner et al. 1977, Råghall 1980). No significant changes were found. The benthos of the area is very poor, but seems to have been so already in 1923 (Hessle 1924).

### 3.2.3 Ume River debouchment area

The phosphorus concentrations in the Umeälven debouchment area decreased greatly during the late 1960s and have since been about the same as in the open Bothnian Sea (Bergstrand 1983a). The

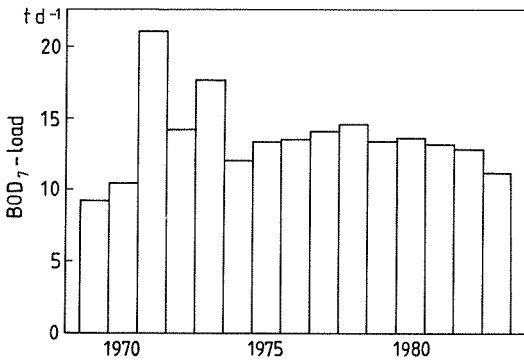


Fig. 7. Annual load of BOD<sub>7</sub> imposed on Vargödragnet in 1969–1983 (data from Lövholmens bruk).

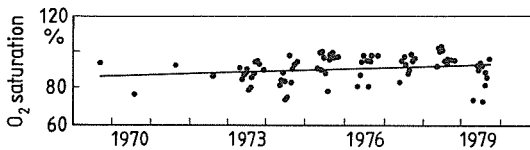


Fig. 8. Variations in oxygen saturation at two intensively studied stations in Vargödragnet in 1969–1979. The increasing trend of the regression line is statistically significant. (From Råghall 1980).

nitrogen concentrations, however, seem to be higher in the debouchment area than in the open sea (Bergstrand 1983a).

The benthos investigation carried out in 1977 showed that between Obbola and Holmsund the fauna was severely impoverished on bottoms deeper than 10 m (Persson and Cederwall 1983). High numbers of tubificids were found along the outer transects, indicating an impact here as well. Another benthos study was carried out in 1983 (Umeå kommuns laboratorium 1984). Since only the shallow stations along the inner transects were sampled then, no conclusions can be made regarding possible changes.

### 3.2.4 Ljusne Bay

Since the early 1970s the BOD<sub>7</sub> load has been reduced by 90 %. The reduction of the total phosphorus load has been 80 %. The change from the sulphite to the sulphate method at Vallvik alone, resulted in a reduction of over 50 % from 1974 to 1975 (Cederwall and Persson 1986).

Since the measurements began in 1975 the

phosphorus concentrations have not been significantly higher than in the open Bothnian Sea but the nitrogen concentrations are definitely higher in the bay (Bergstrand 1983b).

On some occasions, reduced phytoplankton growth has been recorded close to Vallvik (Willén 1983).

Concurrently with the reduction of the pollutant load from Vallvik, the abundance, and most probably also the biomass, increased in the bay (Cederwall and Persson 1986). One earlier (1959) and one later (1980) study indicate that the values were low even before 1975, and that they have remained higher also after 1977.

### 3.2.5 Bay of Edebo

The pollution load decreased in the early 1970s (Fig. 9), but has subsequently been about 10 tonnes BOD<sub>7</sub> per year. In 1983 the manufacture of sulphite pulp was discontinued, and the load should now have decreased considerably.

The oxygen situation in the inner part of the bay improved in the mid-1970s (Fig. 9). No changes have been detected since then. In the outer bay no trend can be seen.

The area where the macrofauna is missing (the primary minimum, Leppäkoski 1975) seems to have been reduced (Fig. 10) (Hydroconsult 1969, Vattentest 1972, 1974, Nyman 1976, 1979, Odelström 1984). The station net was altered in 1982, however, and nothing certain can be said about changes during the 1980s. No significant changes have been recorded in the areas of the primary maximum and secondary minimum.

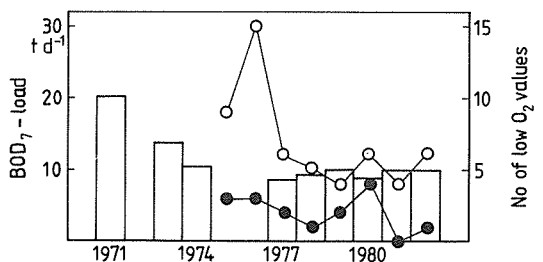


Fig. 9. Annual load of BOD<sub>7</sub> discharged from Hallstavik to the Bay of Edebo in 1971–1982 (columns) and the annual number of low oxygen values (5 mg per litre, 12–15 measurements per year) in the inner (open circles) and outer (filled circles) part of the bay (based on data from Hallstaviks bruk).

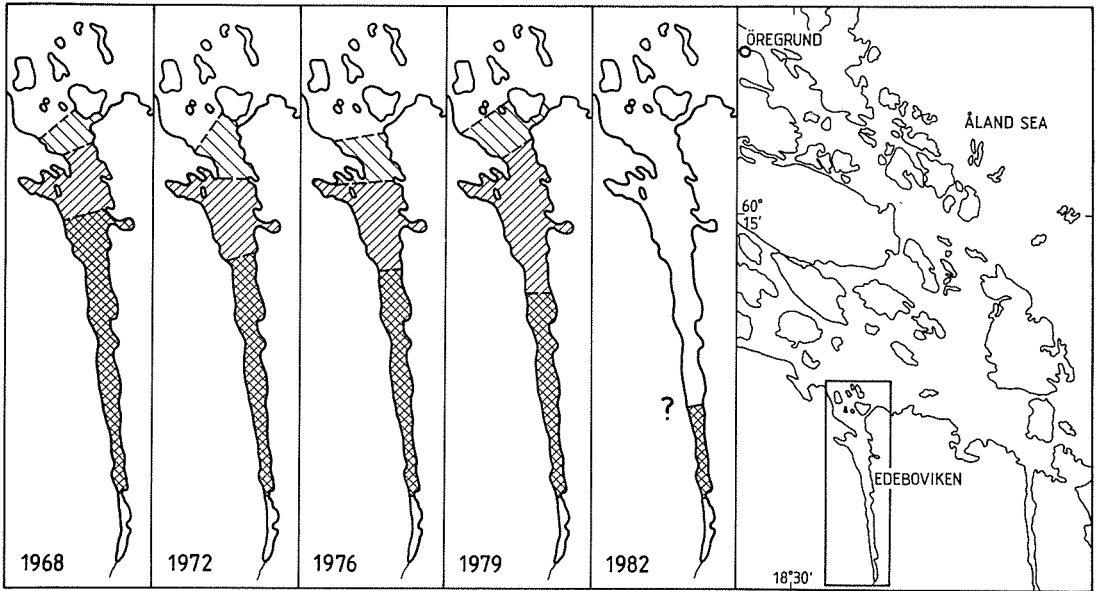


Fig. 10. Zoobenthic zonation (Acc. to Leppäkoski 1975) of the Bay of Edebo 1968–1982. Cross-hatching = area of primary minimum (practically "dead"), heavy hatching = area of primary maximum (high numbers of nonsensitive species) and light hatching = area of secondary minimum (low numbers of both nonsensitive and sensitive species). Note that the station net was changed in 1982, so the results are not fully comparable. (Compiled from Hydroconsult 1969, Vattentest 1972, 1974, Nyman 1976, 1979 and Odelström 1984).

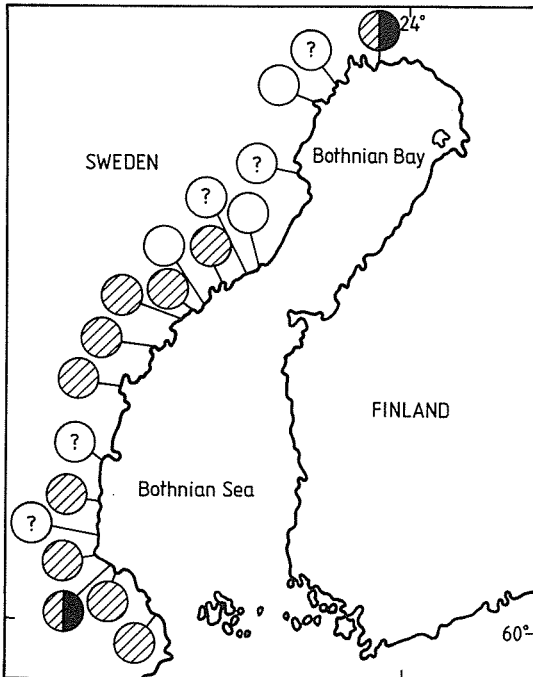


Fig. 11. Changes in the state of waste-discharge areas along the Swedish coast of the Gulf of Bothnia 1970–1982. White symbols, no signs of change; shaded, signs of improvement; black, signs of deterioration.

#### 4. CONCLUSIONS

Very few signs of deterioration have been reported for the period 1970–1982 (Fig. 11). On the contrary, signs of improvement have been noted in a great number of the areas receiving pollutant wastes. The improvements have usually been small, however; marked improvements been shown in only a few cases, in which processes have been changed or operation closed down.

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## RESULTS OF MACROBENTHOS STUDIES IN LJUSNE BAY EXAMINED IN RELATION TO THE POLLUTION LOAD

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CEDERWALL, H. & PERSSON, B.-G. 1986. Results of macrobenthos studies in Ljusne bay examined in relation to the pollution load. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The results indicate that the benthos has increased during the 1970s. There was a significant increase in its abundance from 1975 to 1976, which was probably caused by a decrease in the load of organic matter. The abundance remained at the higher level in 1977 and studies carried out in 1959 and 1980 suggest that the abundance and biomass were low before 1975 and remained high after 1977 as well.

Index words: Baltic Sea, Gulf of Bothnia, pollution load, soft bottom macrofauna

### 1. INTRODUCTION

During 1983 the National Swedish Environment Protection Board carried out an assessment of the trophic status of some selected Swedish water systems, including the adjoining coastal areas. Among these coastal areas was Ljusne Bay, in the southern part of the Bothnian Sea (Fig. 1).

One of the aims of this study was to find out what information on changes in the environment was provided by the various benthos studies carried out in the area.

Ljusne Bay is influenced by effluents from sewage plants and the wood fibre industry, discharged to the bay or the lower reaches of the River Ljusnan, which empties into the bay.

During the 1970s there was a considerable decrease in the pollution load received by the bay (Ånell 1983). The discharges of organic substances made directly to the bay decreased by 90 % and the total phosphorus load was reduced by 80 % (Fig. 2). At the same time the amount of organic substances transported by the river Ljusnan decreased by 50 % (Ahl 1983).

### 2. MATERIAL AND METHODS

During the period 1959 to 1980, five investigations of the macrozoobenthos were carried out in Ljusne Bay (Table 1). The first of these studies was basic research, those of 1975–77 were pollution effects studies and the last a special study made on account of the proposed location of a power plant by the bay.

The sampling was done in early summer, except in 1980, when the samples were taken in September. In 1959 and 1980 a 0.1 m<sup>2</sup> van Veen grab was used, but the rest of the sampling was done with a standard Ekman grab.

In sieving the samples a mesh size of 0.6 mm was generally used. In 1959 the mesh size was 1.0 mm and in 1980 0.5 mm. The samples were preserved in 4 % formaldehyde solution. The biomass values were determined as formalin wet weight. No biomass determinations were carried out in 1980.

The station net was not the same, except in the period 1975–1977. In order to evaluate the data, mean values for the area had to be calculated and compared. Post-sampling stratification was carried

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Table 1. Studies of the benthic soft bottom macrofauna in Ljusne Bay.

| Year | Grab type | Mesh size mm | No. of stations | No. of samples/stations | Investigator               |
|------|-----------|--------------|-----------------|-------------------------|----------------------------|
| 1959 | van Veen  | 1.0          | 4               | 2                       | Sellerberg (1961)          |
| 1975 | Ekman     | 0.6          | 12              | 2—3                     | Ånell (1975)               |
| 1976 | Ekman     | 0.6          | 13              | 2                       | Ånell (1976)               |
| 1977 | Ekman     | 0.6          | 14              | 2                       | Ånell (1978)               |
| 1980 | van Veen  | 0.5          | 5               | 5                       | Blomqvist and Smith (1980) |

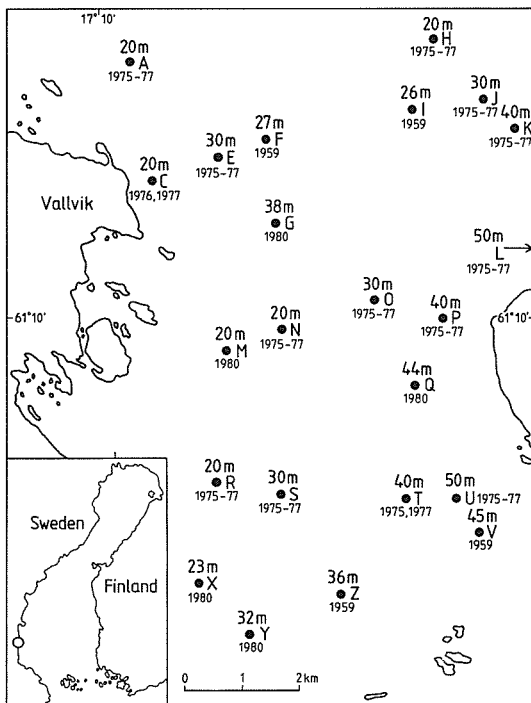
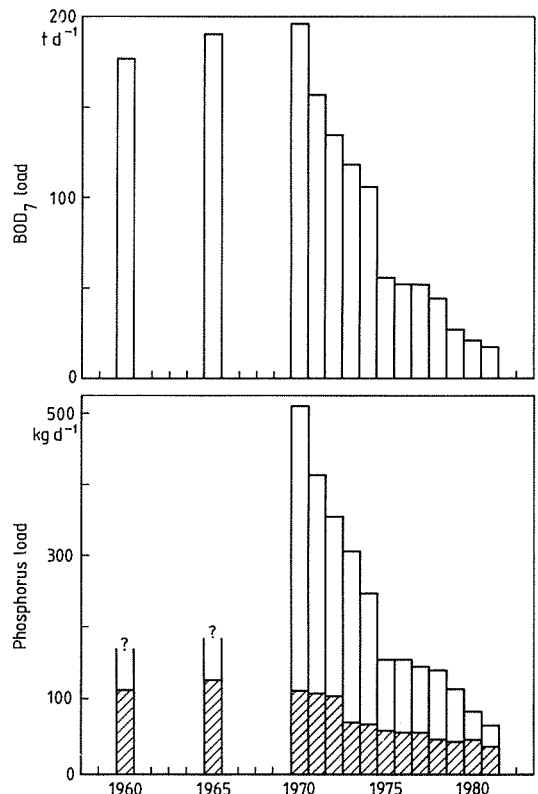


Fig. 1. Sampling stations in Ljusne Bay. Capital letters indicate station codes, figures above letters the depth and figures below the letters sampling years.

Fig. 2. Variations in annual load of organic matter (measured as BOD<sub>7</sub>) and phosphorus in Ljusne Bay 1960–82. The unshaded parts of the bars in the lower diagram show the proportion with industrial origin, the shaded parts the municipal. (Based on data from Ljusnan-Voxnans vattenvårdsförbund).

out and only data from the depth interval 20–50 m were used in the calculations, since seasonal and interannual variations are greater on shallower bottoms.

Earlier studies in which both 1-mm and 0.5-mm sieves were used (Cederwall 1977 and unpublished, Ankar and Leonardsson 1981) allowed conversion of the figures for 1959 (1-mm sieve) to "0.5-mm values". No correction was made for differences between 0.5- and 0.6-mm mesh sizes.

Areal mean values, variances, standard deviations and standard errors were calculated for the abundance of *Pontoporeia affinis* and *Macoma balthica*, and for total abundance and total biomass. Since the station net was the same during the years 1975–1977, sign test (Hodges and Lehman 1970) was used to determine statistically significant differences between those years. The Mann-Whitney U-test (Elliott 1971) was used to determine significant differences between all the sampling years. Correlation coefficients were calculated for the correlations between the pollution load (given as BOD<sub>7</sub>) and total abundance and biomass.

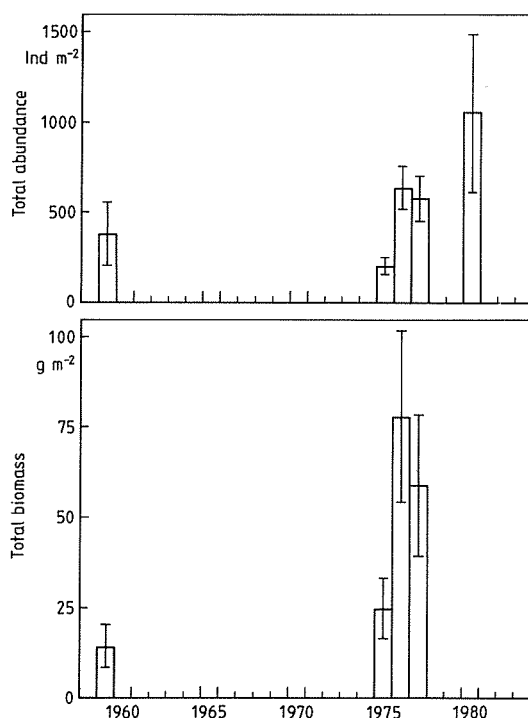


Fig. 3. Total abundance and biomass of soft bottom macrofauna (mean and S.E.) in Ljusne Bay 1959–80.

### 3. RESULTS

The areal mean values for total abundance and biomass are shown in Fig. 3. The increase in abundance between 1975 and 1976 is statistically significant (sign test (= s. t.):  $p = 0.012$ , Mann-Whitney U-test (= M.-W.):  $p < 0.05$ ). The abundances found in 1977 and 1980 were also significantly higher than in 1975 (1977, s. t.:  $p = 0.003$ ; 1980, M.-W.:  $p < 0.025$ ). No significant increases in biomass were recorded, though  $p$  values less than 0.10 were obtained for the differences between 1975–1976, 1975–1977 and 1959–1977.

Compared with the value for 1975, the abundance of the dominant species, *Macoma balthica*, was significantly higher in 1976 (s. t.:  $p = 0.022$ , M.-W.:  $p < 0.01$ ), 1977 (s. t.:  $p = 0.012$ , M.-W.:  $p < 0.005$ ) and 1980 (M.-W.:  $p < 0.05$ ).

For *Pontoporeia affinis* the only significant change in abundance was a decrease between 1959 and 1975 (M.-W.:  $p < 0.05$ ).

The coefficient calculated for the correlation between total abundance and the annual load of organic matter (measured as BOD<sub>7</sub>) during the five study years was  $-0.50$ . For biomass and the organic load, the coefficient was  $-0.68$ . If instead

of the BOD-values for the study years, the values for the immediately preceding year were used, correlation coefficients of  $-0.72$  and  $-0.89$  were obtained for total abundance and biomass, respectively. In all calculations the BOD-value for 1960 was paired with the macrofaunal figures for 1959, since no BOD-values were available for the 1950s. We have, however, no reason to believe that the values for 1959 or 1958 differed significantly from those in 1960.

### 4. DISCUSSION

There are several possible sources of error when the results of different benthos investigations are used. The changes in the station net raise the question of representativeness, as to which nothing certain can be said. A somewhat higher proportion of sandy bottoms in 1980 and a somewhat higher proportion of clayey bottoms in 1959 could have led to over- or underestimation (cf. Ankar and

Elmgren 1976).

The depth range 20–24 m was not sampled in 1959, but that should not affect the abundance or biomass figures significantly (cf. Ankar and Elmgren 1976, Cederwall and Leonardsson 1984).

Except in the period 1975–1977, the number of stations differed greatly between the sampling years. If the station nets were all reasonably representative of the distribution of the benthos in the bay, this would mainly affect the possibility of establishing statistically significant differences between years.

The later sampling time in 1980 should result in higher abundance values, than sampling in June–July in the same year. Whether the annual recruits of *Macoma balthica* are caught quantitatively on the 0.5-mm sieve in September depends on their growth rate. The growth rates given in the literature (Segerstråle 1960, Bergh 1974, Ankar 1980) suggest that only the recruits from the two shallowest stations were caught on the 0.5-mm sieve in September 1980. This means that the abundance figures for *Macoma balthica* may be 50 % higher than in June–July. For *Pontoporeia affinis* the difference may be +30 % and for *Harmothoe sarsi* –75 % (Cederwall 1977). The seasonal variations in the other species can be disregarded.

If the total abundance is corrected according to the figures estimated above, the value for June–July becomes 640 ind m<sup>-2</sup>. It seems likely that the abundance in June–July 1980 was about the same as in 1976 and 1977.

Since the van Veen grab is a better gear for sampling sandy bottoms, the figures for 1975–1977 could be somewhat underestimated. As mentioned earlier, the figures for 1959, when only a 1-mm sieve was used, have been converted. The conversions can be expected to give an over- rather than an underestimate of the figures for 1959.

The macrozoobenthos has comparatively long generation times, and will therefore react rather slowly to a decrease in the pollution load. Also, the macrobenthos cannot be sampled quantitatively until autumn, which delays detection of possible changes. The effects of a change in the annual pollution load should not become evident until the year after the change. Thus it is more correct to correlate the figures for the benthos with the figures for the pollution load of the preceding year. When this was done the correlation coefficients were higher.

During 1975–1977 the station net and the methods were the same. The abundance and biomass recorded in 1976 were more than 200 % higher than in 1975. For comparison, it may be

mentioned that the greatest difference found between two consecutive years in the Askö-Landsort area (19–20 stations) was 20 % (Ankar 1985). The abundance increase from 1975 to 1976 was statistically significant.

At the same time (1975) the load of organic matter was cut by half, mainly because the process was changed at the Vallvik pulp mill. This decrease in the pollution load was most probably responsible for the increase in the benthos. The 1977 and 1980 figures were also significantly higher than the 1975 values, which indicates that the change was not temporary.

The data from 1975–1977 contain much information about the shallower areas (< 20 m), which has not been used in the calculations presented here (Ånell 1975, 1976, 1978). Those data show that there has been a recovery in the fauna of the areas close to the river mouth and the outlet of the Vallvik pulp mill. At some stations devoid of macrofauna in 1975, recolonization had taken place in 1976–1977. At nearby stations the abundance of taxa favoured by pollution, such as oligochaetes and chironomids, had decreased and taxa intolerant of pollution had increased.

Similar changes in the benthos, in response to a reduced load of organic matter have been reported from several areas affected by wood fibre industries along the coast of the Gulf of Bothnia (Cederwall 1986, Bonsdorff et al. 1986). Those results support the conclusion that the changes found in the benthos of Ljusne Bay have been caused by the considerable decrease in the pollution load.

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# THE EFFECTS OF ANTHROPOGENIC ACIDIFICATION ON THE STOCKS AND FISHERIES OF BREAM AND BURBOT IN THE SEA AREA INFLUENCED BY THE RIVER KYRÖNJOKI IN THE GULF OF BOTHNIA

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HUDD, R., HILDÉN, M. & URHO, L. 1986. The effects of anthropogenic acidification on the stocks and fisheries of bream and burbot in the sea area influenced by the River Kyrönjoki in the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The bream, *Abramis brama* (L.), and burbot, *Lota lota* (L.), stocks were studied in the archipelago area off the estuary of the Kyrönjoki, Gulf of Bothnia, in order to analyse the effects of anthropogenic acidification. The most serious changes in the stocks and fisheries occur as a result of reproductive failures due to acidification of spawning and nursery areas. These results emphasize the importance of estuaries and shallow waters for the reproduction of the fish stocks in the Gulf of Bothnia.

Index words: Acidification, fish stocks, fisheries, environmental changes, estuaries, *Abramis brama* (L.), *Lota lota* (L.)

## 1. INTRODUCTION

Acidification is common in the coastal areas of the Gulf of Bothnia, where it was observed to cause mass mortality of fish as far back as last century (Anon. 1896, Högbom 1921). Although the problem has long been recognized (Högbom 1921), few studies have been devoted to the effects of acidification on fish stocks and fisheries. In this paper we present the first results of a study aimed at analysing the effects of anthropogenic acidification on the stocks and fisheries of bream, *Abramis brama* (L.), and burbot, *Lota lota* (L.). Besides providing an example of the effects of a rapid environmental change on fish stocks and fisheries, this and similar studies in already acidified waters can also serve to broaden our understanding of acidification.

## 2. THE STUDY AREA, MATERIAL AND METHODS

The river Kyrönjoki is one of the watercourses seriously affected by acidification due to the oxidation of sulphide clays found in areas of former sea beds (Purokoski 1959, Eriksson 1982). The drainage area of the Kyrönjoki (around 5 000 km<sup>2</sup>) contains an area of 350 to 400 km<sup>2</sup> in which sulphide Littorina clays predominate (Erviö 1975). This area has largely been drained, dredged or altered by water construction works, such as embankments, dams and reservoirs, all which activities may cause acidification. Many of the changes in the drainage area of the Kyrönjoki have occurred since the late 1960's (Hudd et al. 1984).

The bream and burbot stocks which are the objects of this study are fished in the Northern

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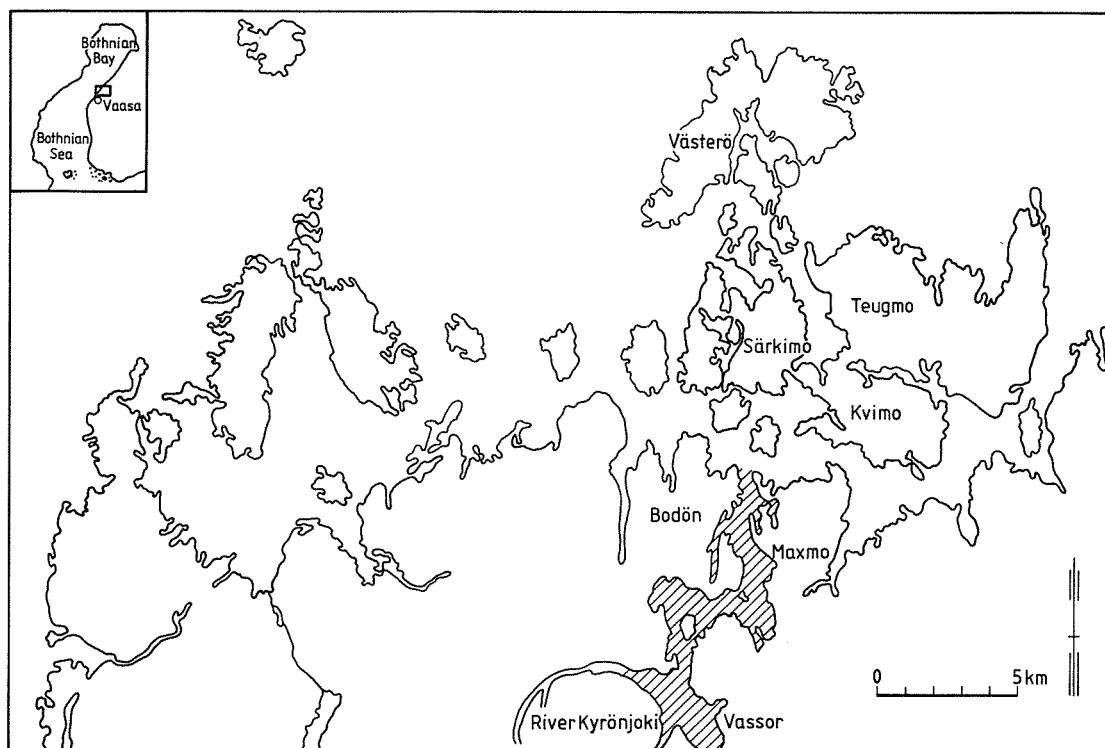


Fig. 1. The study area. The estuary of the Kyrönjoki is hatched.

Quark over an area of c. 1 000 km<sup>2</sup> (Fig. 1).

The study is based on determination of the home ranges by tagging spawning fish on the spawning grounds. Fishery statistics for commercial catches have been obtained from the receipts of wholesale fish firms operating in the area covered by the home ranges of the fish stocks. The age structure of the stocks has been determined from yearly samples of the commercial catch.

### 3. RESULTS

#### 3.1 Changes in fishing of bream and burbot

During the 1970's the burbot fishing grounds in the inner parts of the estuary were totally lost and nowadays no burbot are observed there in winter or spawning time. In the bream fishing areas the most notable change has been the loss of the places

in the estuary where wintering and spawning bream were fished.

Both the bream and the burbot fisheries have been disturbed by abrupt changes in the migration pattern and by excessive mortality of fish in the gear or in storing cages. These changes are due to rapid and sudden acidification of the river water and the effects have been observable over a large archipelago area (for details see Hudd et al. 1984).

#### 3.2 Catches

During the late 1960's the commercial catches of burbot were apparently fairly stable (Fig. 2). The statistics do not give complete coverage of the catches, because the receipts of some of the wholesale fish firms have been lost. Back-calculated from the period when nearly complete coverage of the commercial catches was accomplished, the level of the commercial burbot catches in the late 1960's was 20–25 tonnes each fishing season (late autumn—early spring). The total catch was higher,

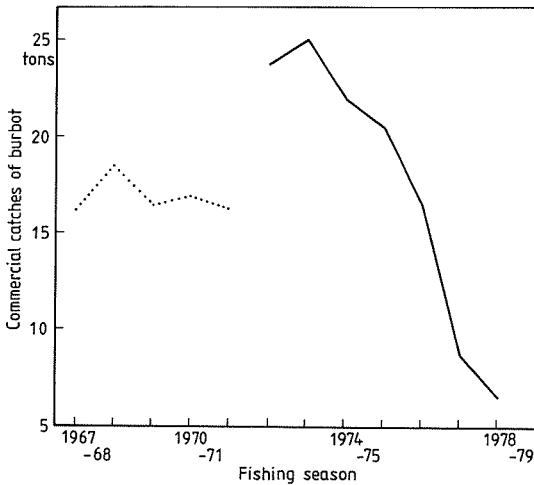


Fig. 2. The commercial catches of burbot. The dotted line indicates incomplete statistics (see text).

but no data are available on the non-commercial catches. However, the trends of the total catches are fairly well reflected in the commercial catches, this conclusion being supported by interviews. From the fishing season 1973–74 the catches declined continuously until 1980–81, since when an increase has been observed.

Two periods of change can be noted in the commercial bream catches. At the end of the 1960's the catches were at a level of 40–60 tonnes. In 1971 and 1972 the catches were low, but they increased again in 1974–76. From 1976 the catches declined continuously to 1979 (Fig. 3) and have remained at a low level from 1980 onwards, though according to interviews, the spring catches in 1984 have been fairly good.

### 3.3 The stocks and their age distribution

Taggings of burbot and bream spawners in the estuary and its vicinity have shown that the same stocks are fished over the entire study area. At present there is no important immigration of either bream or burbot (Hudd et al. 1984). Observations of larvae and juveniles have shown that the spawning and nursery areas of bream and burbot are located in the estuary and its vicinity. The spawning areas of the bream lie chiefly in the inner parts of the estuary, whereas at present those of the burbot are situated in the archipelago in the vicinity of the river mouth (Hudd et al. 1983, 1984).

The age distribution of the spawning stock of burbot in the years 1979–82 shows that the year classes from the early 1970's are weak, as are also the year classes 1976 and 1977 (Fig. 4). The year class 1977 appeared in significant numbers during one single year. The year class 1975 dominated the spawning stock in 1979–82. The same year class has also predominated in the catches in 1981, 1982 and probably also in 1983 and is apparently responsible for the increase of the catches in the beginning of the 1980's. A conservative estimate of the loss of catches due to reproductive failure from 1973 to 1982 is about 50 tonnes.

The age composition of the bream stock shows similarities to the age composition of the burbot stock, with very weak year classes at the beginning of the 1970's. The changes in the bream stock have been simulated with the aid of a population analysis model (e.g. Gulland 1983). The simulation shows a drastic decline of the exploited stock in the latter part of the 1970's (Fig. 3). Since the exploitation has remained at a fairly low level (Hudd et al. 1984) it is apparent that the decline has not been caused by the fishery but by the weakness of several successive year classes in the early 1970's. The model runs also suggest that 20–30 tonnes of bream were lost due to the direct

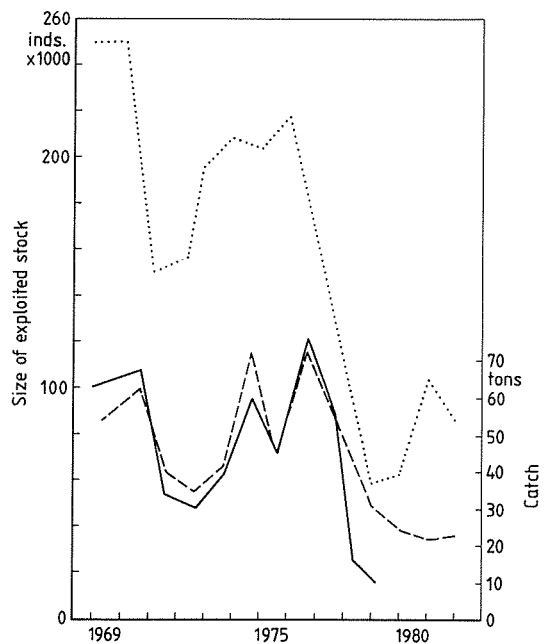


Fig. 3. The commercial catches of bream (—), the estimated catches of bream according to the population model (---) and the size of the exploited stock (.....).

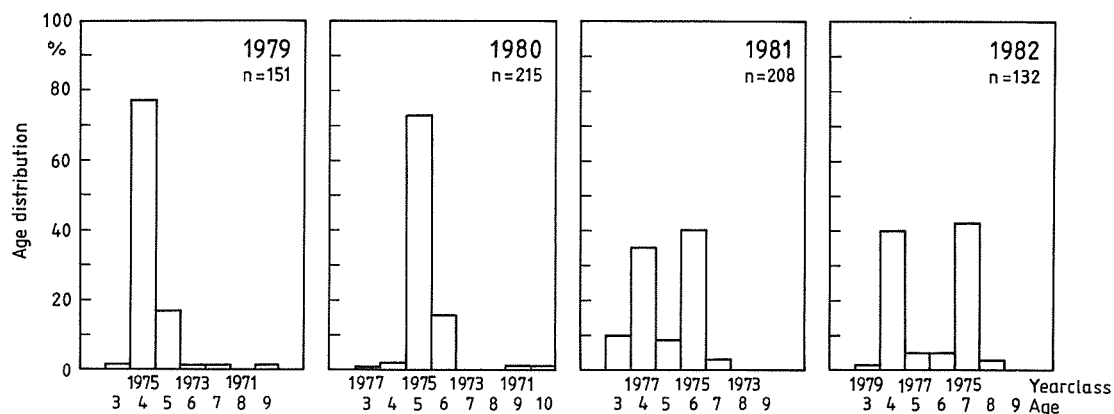


Fig. 4. The age structure of the spawning population of burbot from 1979 to 1982.

effects of the mass mortalities in 1970–73. A conservative estimate of the loss of catches due to reproductive failure is about 70 tonnes. Preliminary observations indicate that the year classes of the middle 1970's are relatively strong. Their recruitment to the fishery apparently explains the catch increase observed in 1984.

#### 4. DISCUSSION

The results above indicate that although the catches of bream and burbot have developed differently, the changes in the stocks have had the same cause. The age structures of the catches reveal that successes and failures of reproduction have occurred in the same years in the two species. The differences in the trends of the catches have thus been caused by the differences in the ages at which the species recruit to the fisheries. The burbot recruits to the fishery when 4–6 years old; the bream at an age of 10–12. This shows that the most important effect on the fish stocks and fisheries has been the loss of reproduction. There is a lag between the actual environmental change (the acidification of the estuary) and the time when its effects become generally observable (decline of the catches). From the recruitment ages it is apparent that the lag is longer for bream than for burbot. Consequently the burbot fishery responds more rapidly to changes in year class strength than the bream fishery.

As the most important effects of the acidifica-

tion occur during the early developmental stages of the fish, it is possible to forecast the trends of the catches from observations of larval and juvenile densities. Thus, although the catches have increased recently, the increase cannot be expected to be permanent, because larval observations suggest that the reproductive success has varied in recent years due to fluctuations in acidity.

It should be noted that the mass mortalities of adult fish caused fairly small declines of the catches; a decrease could be observed in the bream fishery (Fig. 3), but the burbot fishery was almost unaffected. Thus important parts of the stocks survived. Among bream, the main survivors were apparently immature fish, which did not ascend to the spawning areas, where the fish kills mostly occurred. The burbot survived because when the fish kills took place (early summer, autumn) the adult fish had left the shallow parts of the archipelago. On the other hand, the burbot in early developmental stages were hit hard by the acidification, because their distribution coincided with the spread of the acid river water (which probably contained high concentrations of toxic metals as well).

This detailed study of the situation in the estuary of the Kyrönjoki and its archipelago confirms the suggestion of Hildén et al. (1982) that the main reason for the decline of fish stocks and fisheries in the Northern Quark is the destruction of spawning and nursery areas. This is due to the fact that many fish species are dependent on estuaries and shallow vegetated inlets for reproduction. For instance it is interesting to note that although the ecology of adult bream and adult burbot is very different, their larval ecology has

some common features. These features of their larval ecology partly explain why the environmental changes have had similar effects on the fish stocks. The dependence on estuaries increases as one proceeds northward in the Gulf of Bothnia (Eriksson and Müller 1982), clearly because the conditions in the sea becomes harsher. The estuaries offer areas rich in nutrients with warm water when the larvae hatch. This makes them important for many of the fish species in the Gulf of Bothnia and absolutely essential for some. The bream and the burbot in the area off the Kyrönjoki are examples of the latter group.

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**PHYTOPLANKTON OF THE OPEN SEA AREAS  
OF THE GULF OF BOTHNIA —  
OBSERVATIONS MADE IN THE FIRST STAGE  
OF THE BALTIC MONITORING PROGRAMME  
IN 1979—1983**

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HUTTUNEN, M., KONONEN, K., LEPPÄNEN, J.-M. & WILLÉN, T.  
1986. Phytoplankton of the open sea areas of the Gulf of Bothnia —  
Observations made in the first stage of the Baltic monitoring programme in  
1979—1983. Publications of the Water Research Institute, National Board of  
Waters, Finland, No. 68.

Phytoplankton has been studied in the open sea areas of the Gulf of Bothnia  
according to the Baltic Monitoring Programme since 1979. This study  
summarizes the data obtained at Finnish and Swedish laboratories on the  
phytoplankton species composition, biomass and chlorophyll *a* at the main  
monitoring stations. No changes were observed during the years 1979—1983  
and the chlorophyll *a* values did not differ significantly from those recorded in  
the early 1970s. Recommendations are made concerning the sampling strategy  
and microscopical studies.

Index words: Phytoplankton, chlorophyll *a*, Baltic Monitoring Programme,  
Gulf of Bothnia

## 1. INTRODUCTION

The Baltic Monitoring Programme was intended to provide information on the state, fluctuations and long-term trends in the Baltic Sea. In the first stage of the programme four sampling stations in the Gulf of Bothnia were visited by the Finnish and Swedish research vessels.

This study summarizes the data on the phytoplankton species composition, biomass and chlorophyll *a* reported by the Finnish and Swedish laboratories. In addition to evaluating the results with respect to the state of the Bothnian Sea, attention is paid to some points that became evident when the values from different laboratories were combined.

## 2. MATERIAL AND METHODS

The samples were collected at four stations in the Gulf of Bothnia: F2, BO3, US5b and SR5, four times a year, mostly in March, May—June, July—August and November (Fig. 1).

The analyses were performed according to the Guidelines of the Interim Baltic Marine Environment Protection Commission (1979). The phytoplankton biomass values were calculated using the mean volumes of the species adopted at the Finnish Institute of Marine Research. Three laboratories are responsible for the analyses, as follows:

Chlorophyll *a*: The Finnish Institute of Marine Research and the National Board of Fisheries, Sweden.

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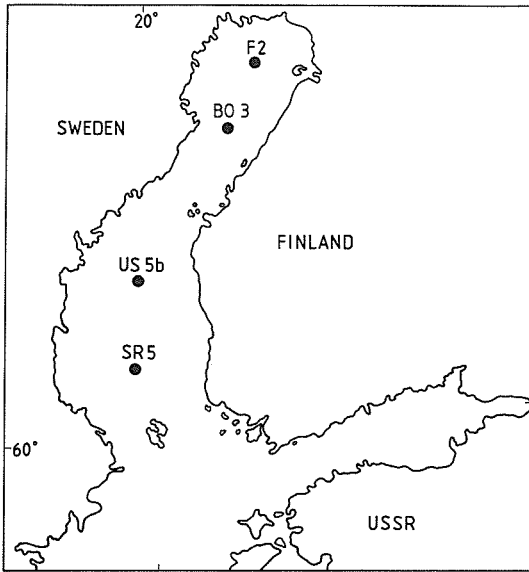


Fig. 1. The sampling stations.

Phytoplankton: The Finnish Institute of Marine Research and the National Swedish Environmental Protection Board.

### 3. RESULTS

The phytoplankton species composition revealed a clear difference between the Bothnian Bay and the Bothnian Sea (Fig. 2–6), reflecting the different nutrient and salinity conditions (e.g. Lassig et al.

Table 1. The dominant species (the nomenclature follows Edler et al. 1984) at the different sampling stations in 1979–1983. 1 = March, 2 = May–June, 3 = July–August, 4 = November.

| Species name             | F2   | BO3  | US5b | SR5  |
|--------------------------|------|------|------|------|
| CYANOPHYCEAE             |      |      |      |      |
| Gomphosphaeria spp.      |      |      | 34   | 4    |
| Microcystis reinboldii   | 1    | 4    | 4    | 34   |
| Aphanizomenon flos-aquae |      |      | 34   | 3    |
| CRYPTOPHYCEAE            |      |      |      |      |
| Cryptomonas spp.         | 1234 | 1 3  | 1 4  | 1 4  |
| DINOPHYCEAE              |      |      |      |      |
| Gymnodinium spp.         | 12   | 12   |      | 2 4  |
| Katodinium rotundatum    |      |      |      | 4    |
| Glenodinium spp.         | 1    | 1    | 1 3  | 1    |
| Gonyaulax catenata       | 2    | 12   | 12   | 12   |
| Peridinium spp.          |      | 2    |      |      |
| Protoperidinium bipes    |      |      | 1    |      |
| CHRYSTOPHYCEAE           |      |      |      |      |
| Dinobryon petiolatum     |      | 2    |      |      |
| Calycomonas wulffii      |      |      |      | 3    |
| DIATOMOPHYCEAE           |      |      |      |      |
| Actinocyclus octonarius  |      |      | 4    | 4    |
| Chaetoceros ceratosporus | 2    |      |      |      |
| Chaetoceros wighamii     | 2 4  |      | 3    | 3    |
| Chaetoceros spp.         |      | 2    | 2    | 2    |
| Thalassiosira baltica    | 12   | 1 4  | 2 4  | 2 4  |
| Thalassiosira spp.       | 4    | 234  | 3    |      |
| Achnanthes taeniata      | 12   | 2    | 2    | 2    |
| Diatoma elongatum        | 2    | 3    |      | 3    |
| Navicula vanhoffeni      |      |      |      | 1    |
| Nitzschia longissima     |      |      |      | 2    |
| PRASINOPHYCEAE           |      |      |      |      |
| Pyramimonas spp.         | 3    | 3    | 2    | 2    |
| CHLOROPHYCEAE            |      |      |      |      |
| Monoraphidium contortum  | 12 4 | 12   | 2 4  |      |
| Planktonema lauterbornii |      |      | 4    | 4    |
| OTHER GROUPS             |      |      |      |      |
| Flagellata               | 123  | 1234 | 123  | 1234 |

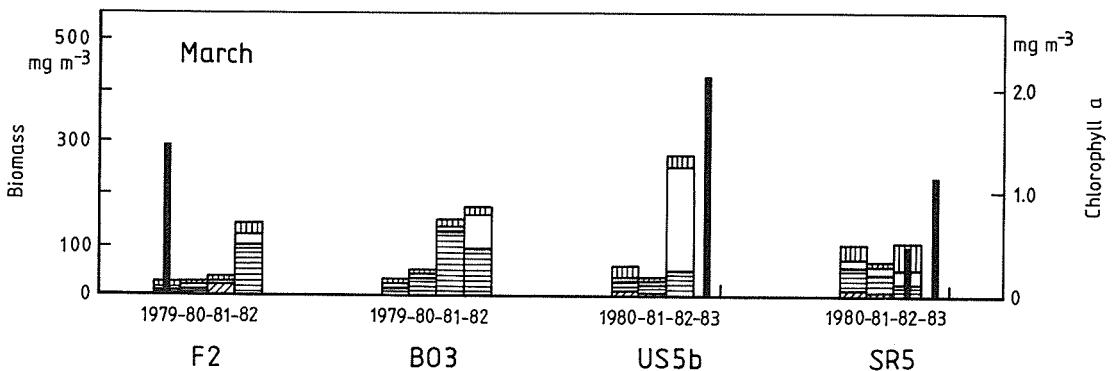


Fig. 2. Phytoplankton biomass (broad columns), species composition (▨ = *Cyanophyceae*, ▤ = *Dinophyceae*, □ = *Bacillariophyceae*, ▧ = other groups) and chlorophyll *a* (narrow columns) at four monitoring stations in the Gulf of Bothnia in March.

1978). The most noticeable feature was the sparse occurrence of nitrogen-fixing blue-green algae in the Bothnian Bay. The dominant phytoplankton species are presented in Table 1. At station BO3 the amount of blue-green algae, mostly *Microcystis reinboldii*, increased in July–August 1981–1982. The group Dinophyceae was poorly represented in the northernmost areas; *Dinophysis acuminata*, *D. norvegica* and *Prorocentrum bipes*, which are typical species in the Baltic Proper and the Gulf of Finland, were not encountered in the Bothnian Bay at all. On the other hand, the group Chloro-

phyceae, including many freshwater species (e.g. *Dictyosphaerium pulchellum*, *Elakatothrix gelatinosa*, *Scenedesmus* spp. and *Sphaerocystis Schroeteri*), occurred abundantly in the area.

The values for phytoplankton biomass and chlorophyll in general varied greatly and no temporal trends could be observed during the monitoring years. Comparison of the July–August chlorophyll values from 1979–1983 with the values from the years 1969–1975 (Lassig and Niemi 1973, 1975, 1976) failed to reveal any statistically significant differences. The chlorophyll

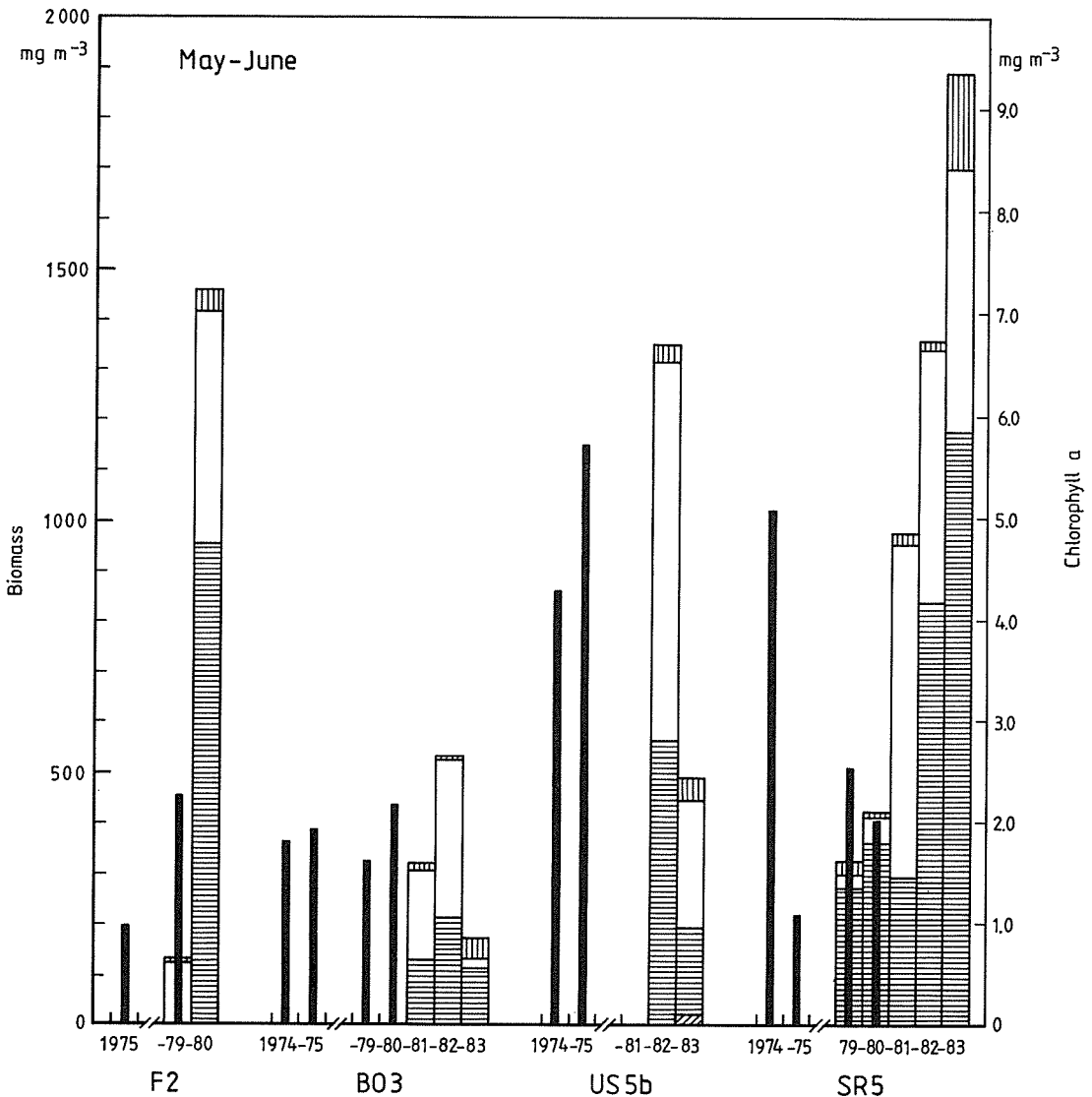


Fig. 3. Phytoplankton biomass, species composition and chlorophyll *a* at monitoring stations in the Gulf of Bothnia in May–June. Units and symbols as in Fig. 2.

and biomass values ranged around the same level as observed by Niemi and Ray (1975, 1977), Lassig et al. (1978), Alasaarela (1979) and Forsskåhl (1980).

#### 4. CONCLUSIONS

The phytoplankton and chlorophyll *a* data collected in the first stage of the Baltic Monitoring Programme did not reveal any significant changes during the years 1979–1983 or significant differences from the chlorophyll data from the

years 1969–1975. The results can therefore be considered a baseline for future monitoring.

During the present combined study, certain observations were made concerning sampling strategy and microscopic analyses of phytoplankton that deserve to be considered in future programmes.

Four samplings a year have been recommended as a compromise between the need for continuous monitoring and the resources available. This limited number makes evaluation of the data difficult, since the annual fluctuation of phytoplankton is high. The situation could be improved, however, by paying more attention to the sampling time. Though winter samplings (March–Novem-

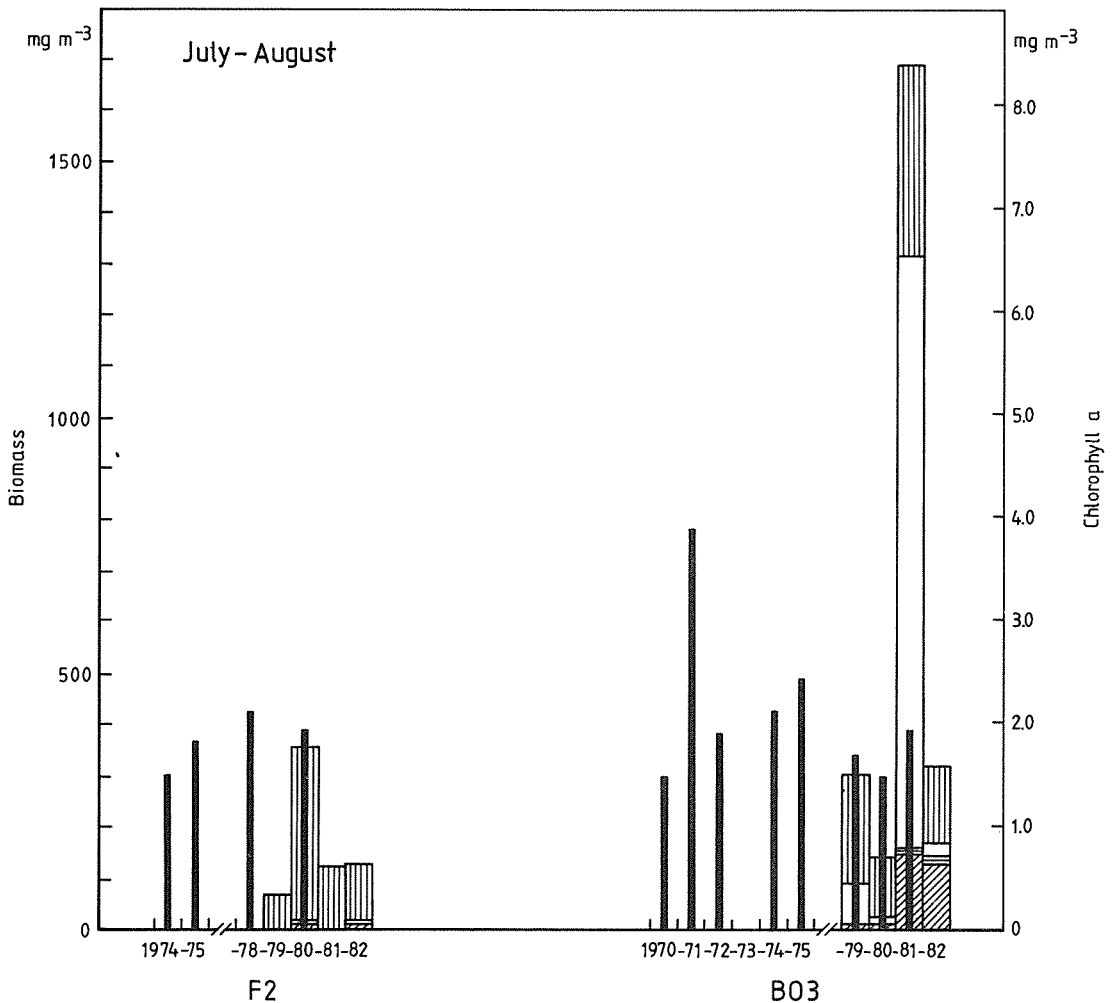


Fig. 4. Phytoplankton biomass, species composition and chlorophyll *a* at stations F2 and BO3 in the Gulf of Bothnia in July–August. Units and symbols as in Fig. 2.

ber) are required for nutrients, they are not suitable for phytoplankton; the sampling should be concentrated on the growing season. One sampling in May—June does not give sufficient information on either the duration or the magnitude of the phytoplankton bloom in spring—early summer, since these depend closely on the increase in the nutrient level. The four monitoring stations in the Gulf of Bothnia are situated far apart and a confusing picture results if they are visited on successive days. In the south the vernal maximum is reached in May but in the north in June. Therefore 2 single May—June sampling does not represent the bloom at all the stations. We recommend several successive cruises during the vernal bloom period.

We found that taxonomic revision is needed, especially for the Cryptophyceae, small flagellates, *Gymnodinium* spp. and *Thalassiosira* spp.

The different microscopic magnifications used when counting the samples seriously hampered comparison of the results from different years and stations. We recommend that the microscopic magnification should be agreed exactly for each phytoplankton species.

Counts of the 6—9 dominant species give 90 % of the total biomass with sufficient accuracy (Willén 1976) and this method has been recommended to facilitate the laborious counting procedure. Environmental changes may, however, first be reflected as changes in the occurrence of the less important species. In our material the sea

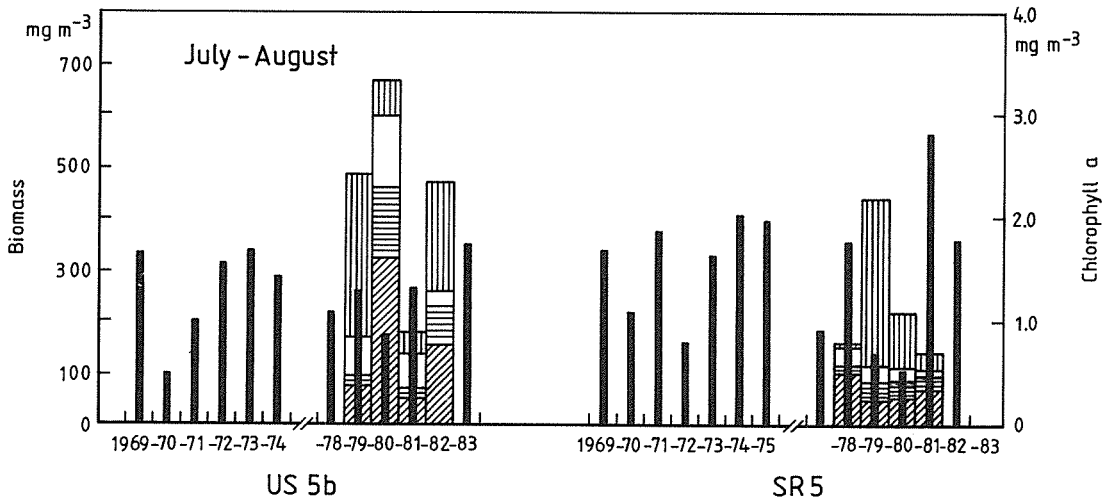


Fig. 5. Phytoplankton biomass, species composition and chlorophyll *a* at stations US5b and SR5 in the Gulf of Bothnia in July—August. Units and symbols as in Fig. 2.

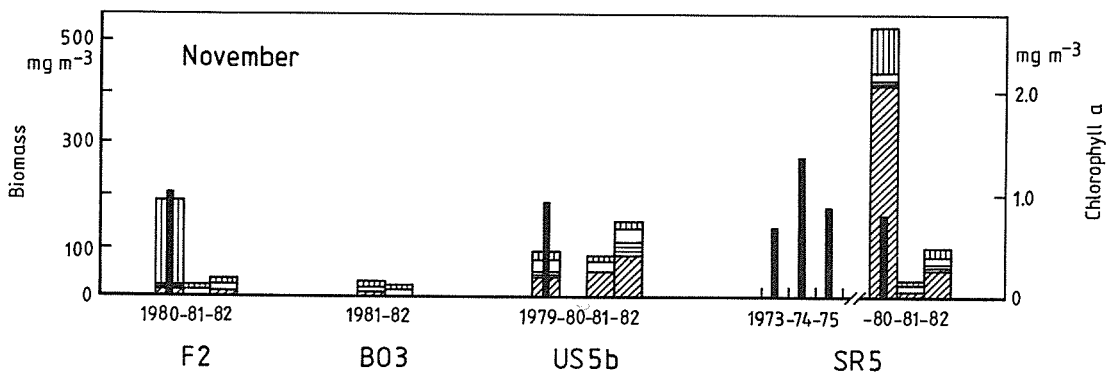


Fig. 6. Phytoplankton biomass, species composition and chlorophyll *a* at the monitoring stations in the Gulf of Bothnia in November. Units and symbols as in Fig. 2.

areas differed most clearly in the less abundant species and we recommend that every species should be recorded, even if they are not all included in the calculation of the total biomass.

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## EFFECT OF DISCHARGES FROM THE OLKILUOTO NUCLEAR POWER STATION ON THE RECEIVING WATERS

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ILUS, E., OJALA, J. & SJÖBLOM, K.-L. 1986. Effect of discharges from the Olkiluoto nuclear power station on the receiving waters. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

A brief review is given of the environmental impact of cooling water from the Olkiluoto power station on the discharge area in 1978–83. The most noticeable effect was the change in the temperature and ice conditions in winter. During summer the hydrographical changes were relatively small. The biological effects were most distinct in the littoral vegetation and bottom fauna near the discharge canal.

The radioactive discharges from the power station were most easily detected in the algae, sedimenting material and bottom sediment of the neighbouring sea area. The main radionuclides detected were <sup>54</sup>Mn, <sup>58</sup>Co and <sup>60</sup>Co. The concentrations were low and insignificant as regards the radiation burden on man.

Index words: Ecology, radioecology, nuclear power station, cooling water, radioactive releases

### 1. INTRODUCTION

The Olkiluoto nuclear power station is situated on the west coast of Finland, about 30 km south of Pori. The first 660 MW unit went into operation in 1978 and the second in 1980.

Since 1977, an extensive radiological surveillance programme has been carried out by the Finnish Centre for Radiation and Nuclear Safety in both the terrestrial and aquatic environments of the power station. The results of the latter part of this programme are discussed in this paper.

To clarify the thermal effects of cooling water on the receiving waters, hydrographical and biological studies have also been made in the area.

These studies are designed to give an ecological background for the radiological surveillance programmes.

### 2. STUDY AREA

The study area and the sampling stations are presented in Fig. 1. The cooling water for the power station is taken from the south side of Olkiluoto and discharged towards the west, as indicated by the arrow in the map. When running at

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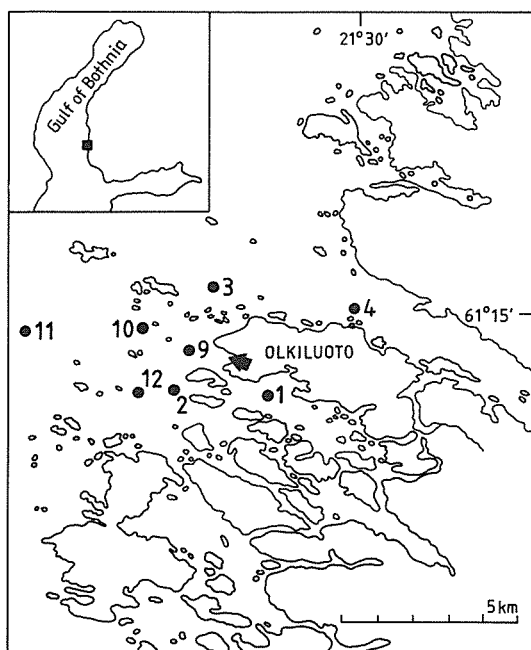


Fig. 1. The study area and the sampling stations.

full capacity the power station uses about  $60 \text{ m}^3 \text{ s}^{-1}$  of cooling water. At the outlet, the temperature of the water is about  $10^\circ\text{C}$  higher than that of the intake water, and the velocity of flow is about  $2 \text{ m s}^{-1}$ .

### 3. RESULTS AND DISCUSSION

The most noticeable of all the environmental effects of the power station has been the impact of the cooling water on the temperature and ice conditions in the discharge area in winter. A rather large area outside the outlet has been ice-free throughout the year. Thus, the growing period of the phytoplankton and littoral vegetation has been markedly prolonged in the vicinity of the outlet.

Owing to the openness and the effective water exchange in the area, the hydrographical changes in summer were relatively small, and so were also the effects of the cooling water on the phytoplankton production (Ilus 1983). During the summer months (June–August), the cooling water caused an average increase of  $2\text{--}4^\circ\text{C}$  in the surface water temperature at a distance of 1 km from the outlet, compared with more distant areas (Fig. 2).

The clearest biological effects have been noted in the littoral vegetation and bottom fauna near the discharge canal. *Myriophyllum spicatum* and *Cladophora glomerata* have increased significantly in the shallow bay where the cooling water is led out (Keskitalo and Ilus 1986). At the nearest soft-bottom station (Station 9) the abundance and biomass of the bivalve *Macoma baltica* increased markedly in 1978–83, but decreased in 1984 (Fig. 3) due to decomposing algal material deposited on the bottom. This material obviously originates from the area near the outlet and is transported by strong currents into small basins outside the bay.

The radiological surveillance programme of the Olkiluoto nuclear power station includes regular sampling of sea water, fish, algae, bottom animals, sedimenting material and bottom sediment in the neighbouring sea area.

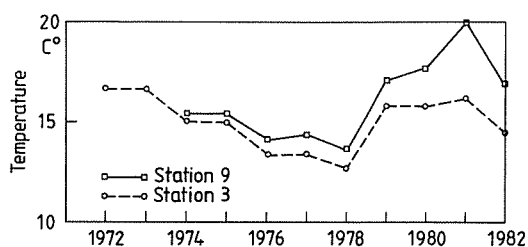


Fig. 2. The mean temperature of the surface water at Stations 9 and 3 during the summer months (VI–VIII) of 1972–1982.

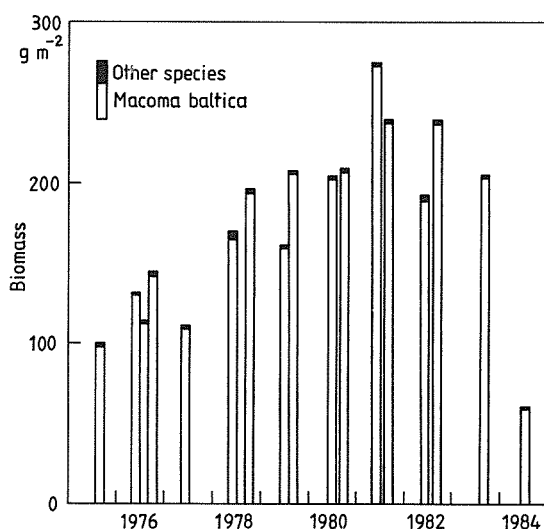


Fig. 3. The biomass of *Macoma baltica* and other bottom animals at Station 9 in 1975–1984.

Table 1.  $^{54}\text{Mn}$ ,  $^{58}\text{Co}$ ,  $^{60}\text{Co}$  and  $^{137}\text{Cs}$  in sedimenting material and bottom sediment samples in 1978 and 1983 ( $\text{Bq kg}^{-1}$  dry wt).

| Station | Nuclide           | Sedimenting material |      | Bottom sediment 0–5 cm |      |
|---------|-------------------|----------------------|------|------------------------|------|
|         |                   | 1978                 | 1983 | 1978                   | 1983 |
| 2       | $^{54}\text{Mn}$  | 0                    | 12   | 0                      | 0    |
|         | $^{58}\text{Co}$  | 0                    | 3.8  | 0                      | 0    |
|         | $^{60}\text{Co}$  | 0                    | 51   | 0                      | 23   |
|         | $^{137}\text{Cs}$ | 88                   | 76   | 99                     | 80   |
| 3       | $^{54}\text{Mn}$  | ..                   | 3.5  | ..                     | 0    |
|         | $^{60}\text{Co}$  | ..                   | 49   | ..                     | 0    |
|         | $^{137}\text{Cs}$ | ..                   | 59   | ..                     | 7.7  |
| 4       | $^{60}\text{Co}$  | 0                    | 16   | 0                      | 7.7  |
|         | $^{137}\text{Cs}$ | 78                   | 55   | 83                     | 58   |
| 9       | $^{54}\text{Mn}$  | ..                   | ..   | 0                      | 5.8  |
|         | $^{60}\text{Co}$  | ..                   | ..   | 0                      | 50   |
|         | $^{137}\text{Cs}$ | ..                   | ..   | 96                     | 79   |
| 12      | $^{60}\text{Co}$  | ..                   | ..   | 0                      | 15   |
|         | $^{137}\text{Cs}$ | ..                   | ..   | 110                    | 71   |

0 below detection limit

.. not analysed

Table 1 shows that the reactor-induced nuclides  $^{54}\text{Mn}$ ,  $^{58}\text{Co}$  and  $^{60}\text{Co}$  have appeared in sedimenting material and sediment samples since the power station started to operate in 1978. At the same time the content of  $^{137}\text{Cs}$ , originating from nuclear weapon tests, has consistently decreased.

Discharge nuclides have not been found in fish flesh samples and only occasionally in sea water.

The bladder-wrack, *Fucus vesiculosus*, has proved to be an excellent bioindicator of radioactivity in the aquatic environment. Table 2 presents concentrations of reactor-induced  $^{54}\text{Mn}$ ,  $^{58}\text{Co}$ ,  $^{60}\text{Co}$ ,  $^{65}\text{Zn}$  and  $^{110m}\text{Ag}$  in *Fucus* near the cooling water outlet (distance 800 m) in 1977–83. The concentrations are in good correlation with the discharges, as can be seen in Fig. 4 with reference to  $^{60}\text{Co}$ . The amounts of  $^{137}\text{Cs}$  originating from fallout have slightly decreased. A special *Fucus* survey in 1981 showed that the main directions in which the radionuclides spread in the Olkiluoto

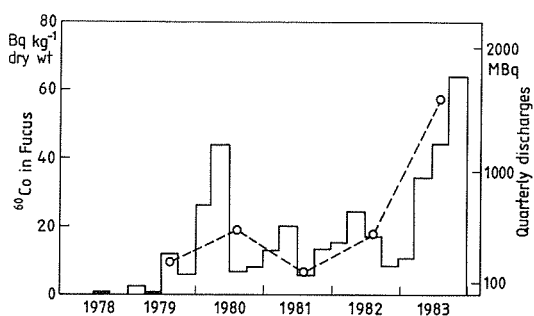


Fig. 4.  $^{60}\text{Co}$  in *Fucus vesiculosus* collected near the cooling water outlet in 1978–1983 (dotted line) and the quarterly discharges of  $^{60}\text{Co}$  (solid line).

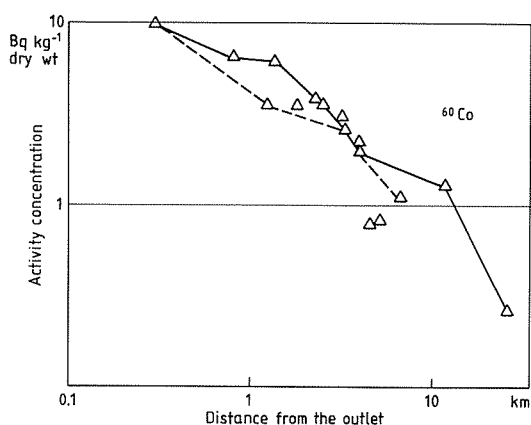


Fig. 5.  $^{60}\text{Co}$  in *Fucus vesiculosus* in relation to the distance from the power station in 1981.

area are northwest and southwest (Ilus et al. 1983). The farthest point at which  $^{60}\text{Co}$  was detected was 27 km north of Olkiluoto (Fig. 5).

The amounts of radionuclides in all the environmental samples are very low. The radiation burden on man, estimated from the discharge measurements of the Olkiluoto power station, is about  $2 \cdot 10^{-7} \text{ Sv a}^{-1}$  (Blomqvist 1984), which is 1/10000 of that of the natural radiation background in the area.

Table 2.  $^{54}\text{Mn}$ ,  $^{58}\text{Co}$ ,  $^{60}\text{Co}$ ,  $^{65}\text{Zn}$ ,  $^{110m}\text{Ag}$  and  $^{137}\text{Cs}$  in *Fucus vesiculosus* collected near the cooling water outlet in 1977–1983 ( $\text{Bq kg}^{-1}$  dry wt).

| Nuclide            | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 |
|--------------------|------|------|------|------|------|------|------|
| $^{54}\text{Mn}$   | 0    | 0.37 | 54   | 27   | 9.4  | 18   | 32   |
| $^{58}\text{Co}$   | 0    | 0    | 26   | 20   | 2.2  | 10   | 27   |
| $^{60}\text{Co}$   | 0    | 0    | 8.0  | 19   | 5.9  | 17   | 56   |
| $^{65}\text{Zn}$   | 0    | 0    | 5.0  | 5.9  | 0    | 5.9  | 5.7  |
| $^{110m}\text{Ag}$ | 0    | 0    | 0    | 1.6  | 0    | 0.44 | 3.4  |
| $^{137}\text{Cs}$  | 13   | 8.5  | 13   | 7.5  | 9.8  | 8.0  | 7.3  |

0 below detection limit

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## PHYTOPLANKTON COMPOSITION AND BIOMASS IN THE ARCHIPELAGO SEA, NORTHERN BALTIC, IN 1966—1970 AND 1979—1982

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KIPPO-EDLUND, P. & NIEMI, Å. 1986. Phytoplankton composition and biomass in the Archipelago Sea, Northern Baltic, in 1966—1970 and 1979—1982. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The phytoplankton biomass and composition in the Archipelago Sea are compared between the periods 1966—1970 and 1979—1982. The changes in the hydrography and nutrient level of the Baltic Proper in the 1970s are reflected in increased levels of salinity and nutrients in the Archipelago Sea. In consequence, the phytoplankton biomass has increased and the species composition become more marine. Nitrogen-fixing blue-green algae, favoured by the surplus of phosphorus, were more abundant in the late 1970s. In the inner archipelago the phytoplankton biomass had increased but no marked change was found in the species composition.

Index words: Phytoplankton biomass, composition, Archipelago Sea, Baltic Sea

### 1. INTRODUCTION

The year 1969 saw the end of a long period of stagnation in the Central Basin of the Baltic Proper (Fonselius 1976, Nehring and Francke 1981, Matthäus 1982). Mixing of the basin water with deep and surface water led to a rise in the surface salinity in the seventies, which was also noted at the entrance to the Gulf of Finland and in the SE Bothnian Sea (Niemi 1975, Pietikäinen et al. 1978, Lassig et al. 1980, Koljonen et al. 1981, Hällfors et al. 1982, Launiainen and Koljonen 1982).

At the same time, the winter levels of  $\text{PO}_4\text{-P}$  and  $\text{NO}_3\text{-N}$  increased in the mixed surface layer in the Central Basin (Nehring 1981), as also at the entrance to the Gulf of Finland (Niemi 1975, Lassig et al. 1980, Hällfors et al. 1982). This

change in the hydrographic conditions of the Baltic Proper may have been responsible for the increase in production and the change in the algal and faunal species composition (Kangas et al. 1982, Hällfors et al. 1984, Kononen and Niemi 1984). On the other hand, the cause may have been changes in the nutrient load from the land (Melvasalo et al. 1981), since there have been many reports of deterioration of the coastal waters (e.g. Melvasalo 1971, Niemi 1972, Melin and Lindahl 1973, Melvasalo and Viljamaa 1975, Alasaarela 1979a, b).

The aim of the present study is to assess the influence of the hydrographic-chemical changes in the Baltic Sea on the phytoplankton in the Archipelago Sea by comparing phytoplankton material from 1966—1970, i.e. the period before the changes, with material from 1979—1982.

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## 2. STUDY AREA, MATERIAL AND METHODS

The Archipelago Sea contains several basins of different size and form, and has a complicated pattern of currents (Koljonen 1979). The sill formed by this sea prevents saline Baltic deep water rich in phosphorus from flowing into the Bothnian Sea; only Baltic surface water flows through the area (Pietikäinen et al. 1978). The sampling stations used in this study are located in the eastern part of the Archipelago Sea (Fig. 1).

Phytoplankton was sampled (by the Turku Water District of the Board of Waters) in late July or early August. The samples from 1966–1970 were preserved with formalin, those from 1979–1982 with Keefe's (1926) solution. This difference decreases the comparability of the results (Hällfors et al. 1979). The counting (Utermöhl technique) was made by the methods of the National Board of Waters (Kippo-Edlund 1984), in the Water Research Institute. The material from 1966–1970 was determined and counted by Ainikki Naulapää

and that from 1979–1982 by Kippo-Edlund (1984). Unless otherwise mentioned, the nomenclature follows Hällfors (1979). The hydrographic and chemical data originate from the water quality register of the National Board of Waters, Finland.

## 3. HYDROGRAPHIC-CHEMICAL DETERMINANTS

During the sampling period 1979–1982 the surface temperature was in general higher than in 1966–1970 (Fig. 2). The salinity had clearly increased in the outer archipelago and Paimionlahti, but not in the Airisto area (Fig. 2). The water exchange between Paimionlahti and the open Baltic seems to be good.

The total phosphorus level had increased in the outer archipelago and the Airisto area, but not in

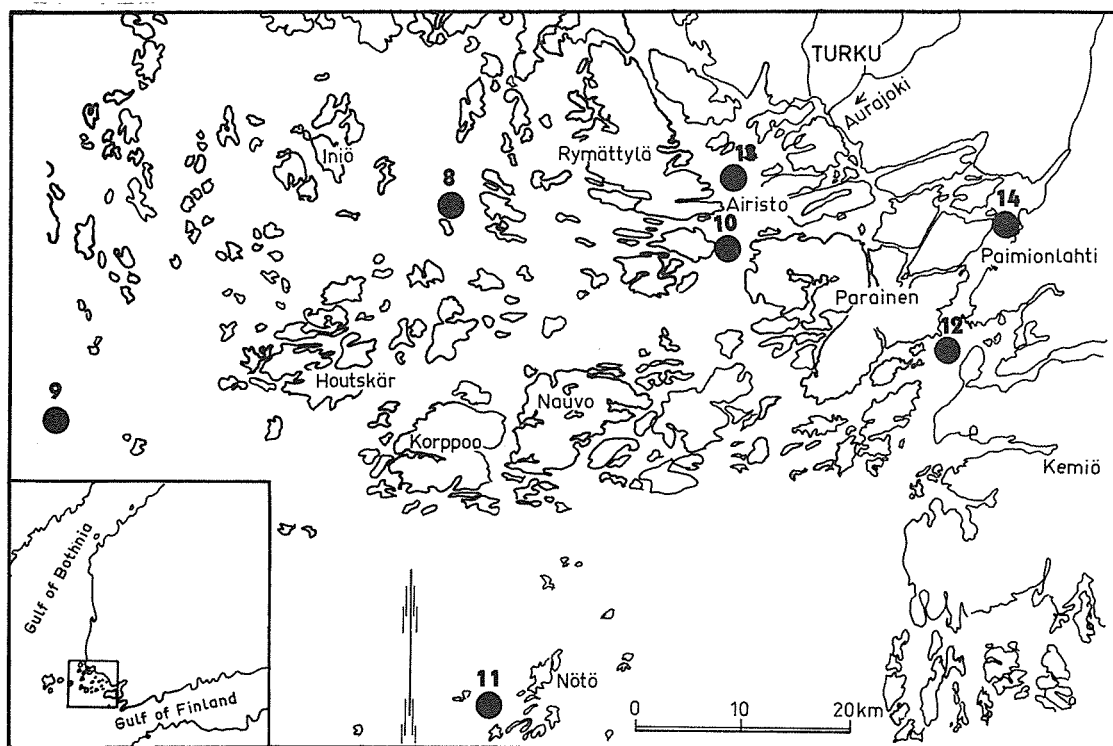


Fig. 1. Study area and sampling stations. Outer archipelago: st. 8 Pakinainen (depth 35 m), 9 Kihti (80 m), 11 Nötö (95 m); inner archipelago: 10 Airisto, Airismaa (75 m), 12 Paimionlahti (25 m), 13 Airisto, Rajakari (52 m), 14 Paimionlahti, Tryholm (47 m) Archipelago zones according to Jaatinen (1960).

Paimionlahti (Fig. 2). There was also an apparent rise in total nitrogen, but owing to a change in the method of analysis in the yearly 1970s (Erkoma et al. 1977), it is not certain whether this was real.

#### 4. CHANGES IN PHYTOPLANKTON BIOMASS AND COMPOSITION

Throughout in the study area, both the total phytoplankton biomass and the number of individuals/counting units were apparently higher in 1979–1982 than in 1966–1970 (Table 1).

In the outer archipelago and Paimionlahti the proportion of blue-green algae in the total biomass was smaller in 1979–1982, but at Airisto no difference was observed between the periods (Table 2). Species fixing molecular nitrogen (*Nodularia spumigena*, *Aphanizomenon flos-aquae*, *Anabaena lemmermannii*) were more abundant in 1979–1982 than in the earlier period. Changes are evident in the species composition of the blue-green algae (Table 3), but these may be partly due to different interpretations of the species in the determinations.

In 1979–1982 more Cryptomonads were counted, but the Euglenomonads showed no clear change (Table 2). Prymnesiophycean species (*Chrysoschromulina* spp. recorded in 1979–1982) were not observed in the early 1970s. In the outer archipelago and Paimionlahti, the proportion of diatoms in the total biomass was smaller in 1979–1982, but at Airisto no changes were observed (Table 2).

*Coscinodiscus granii* was common in 1979–1982. Apparent changes were observed in the species composition of the diatoms from 1979–1982 and also in the composition of the green algae (Table 3). In 1979–1982 the number of green algal species (freshwater element) in the outer archipelago was smaller than in 1966–1970. In 1979–1982 the proportion of green algae in the total biomass had decreased in the inner but not in the outer archipelago (Table 2).

#### 5. CONCLUSIONS

The increase in salinity probably affects the species composition, favouring marine species and diminishing the freshwater element (Kononen & Niemi 1984). The nutrient level increased as well. The higher phytoplankton biomass and the alteration in species composition observed in the later period seem to be the results of the hydrographic-chemical changes in the Baltic Sea. The increase in the abundance of species fixing molecular nitrogen is probably connected with the

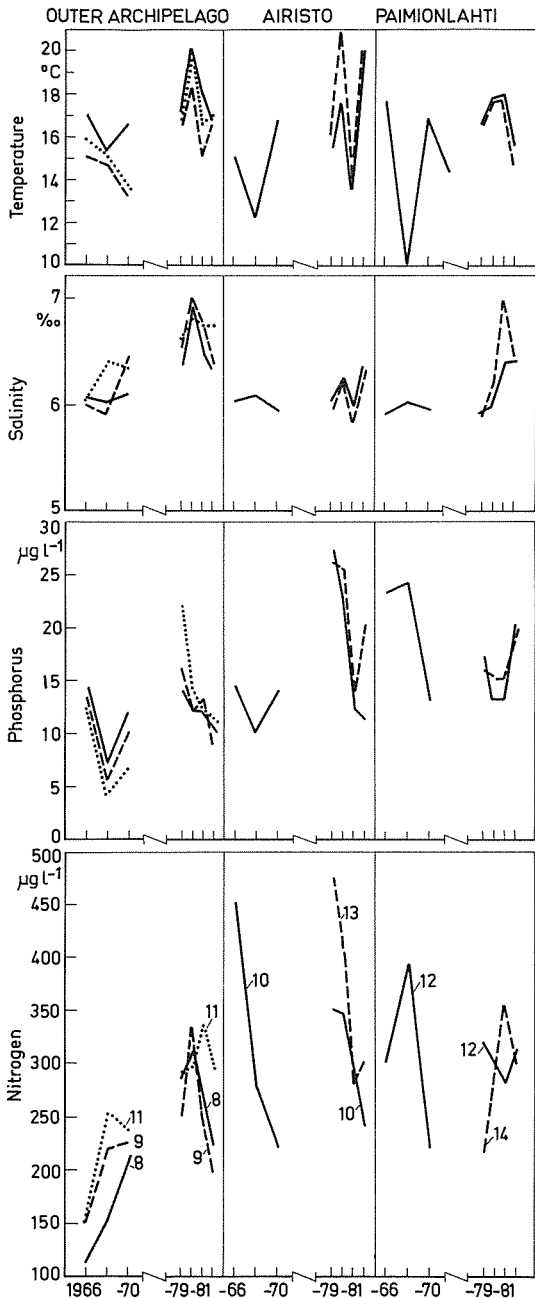


Fig. 2. Temperature, salinity, total phosphorus and total nitrogen in the surface layer (0–10 m) in late summer of 1966–1970 and 1979–1982. The sampling stations are indicated in the lowest part of each column.

Table 1. Biomass and total number of phytoplankton individuals/counting units in the surface layer ( $\bar{x}$  = 0–10 m) in late summer of 1966–1970 and 1979–1982. (For the sampling stations, see Fig. 1).

| Area              | Biomass mg l <sup>-1</sup> |           | Number of individuals/<br>counting units per litre |           |
|-------------------|----------------------------|-----------|----------------------------------------------------|-----------|
|                   | 1966–1970                  | 1979–1982 | 1966–1970                                          | 1979–1982 |
| Outer archipelago | 0.187                      | 0.444     | 61 800                                             | 149 400   |
| Airisto           | 0.155                      | 0.840     | 24 300                                             | 106 800   |
| Paimionlahti      | 0.184                      | 0.633     | 55 100                                             | 128 200   |

Table 2. The proportions of the primary phytoplankton groups (%) in the total phytoplankton biomass of the surface layer ( $\bar{x}$  = 0–10 m) in late summer of 1966–1970 and 1979–1982. (For the sampling stations, see Fig. 1., the Prasionophyceae are included in the Chlorophyceae.)

| Phytoplankton<br>group | Outer<br>archipelago |               | Airisto       |               | Paimionlahti  |               |
|------------------------|----------------------|---------------|---------------|---------------|---------------|---------------|
|                        | 1966—<br>1970        | 1979—<br>1982 | 1966—<br>1970 | 1979—<br>1982 | 1966—<br>1970 | 1979—<br>1982 |
| Cyanophyceae           | 17                   | 11            | 1             | 0             | 24            | 2             |
| Cryptophyceae          | 4                    | 38            | 27            | 33            | 5             | 37            |
| Bacillariophyceae      | 13                   | 2             | 34            | 35            | 31            | 15            |
| Euglenophyceae         | 3                    | 1             | 1             | 2             | 8             | 8             |
| Chlorophyceae          | 27                   | 24            | 8             | 20            | 6             | 13            |

surplus of phosphorus as suggested by e.g. Niemi 1979, Rinne et al. 1981.

In the inner archipelago areas the algal production and species composition are affected by the influence of water flowing from the land. Although the biomass had increased, no marked change could be found in the species composition.

Besides being reflected in the phytoplankton, major changes in the hydrography and nutrient balance of the Baltic Proper are also mirrored in the macroalgal communities of the outer archipelagoes, in waters under the direct influence of the open sea, as has been demonstrated recently on the southern coast of Finland (Kangas et al. 1982, Hållfors et al. 1982, 1984).

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- Alasaarela, E. 1979b. Phytoplankton and environmental conditions in central and coastal areas of the Bothnian Bay. *Annales Botanici Fennici* 16: 241–274.
- Table 3. Blue-green algae, diatoms and green algae observed in the surface layer of the Archipelago Sea ( $\bar{x}$  = 0–10 m) in 1966–1970 and 1979–1982. fq = frequent in every sample, p = common, usually present, r = rare, occurring seldom. (The present authors have not been able to check the determinations of the material from 1966–1970.)
- | Species                                 | 1966–1970 | 1979–1982 |
|-----------------------------------------|-----------|-----------|
| <i>Chroococcus minutus</i>              | —         | r         |
| <i>Gomphosphaeria aponina</i>           | p         | —         |
| <i>G. lacustris</i>                     | r         | p         |
| <i>Merismopedia warmingiana</i>         | r         | r         |
| <i>Microcystis reinboldii</i>           | p         | p         |
| <i>Microcystis</i> sp.                  | —         | r         |
| <i>Anabaena augstumalis</i> Schmidle    | r         | —         |
| <i>A. baltica</i>                       | r         | —         |
| <i>A. circinalis</i>                    | r         | —         |
| <i>A. constricta</i> (Szafer) Geitl.    | r         | —         |
| <i>A. flos-aquae</i> (Lyngb.) Bréb.     | r         | —         |
| <i>A. lemmermannii</i>                  | —         | fq        |
| <i>A. subcylindrica</i>                 | r         | —         |
| <i>Anabaena</i> sp.                     | —         | r         |
| <i>Anabaenopsis elenkinii</i>           | —         | r         |
| <i>Aphanizomenon flos-aquae</i>         | fq        | fq        |
| <i>A. gracile</i>                       | p         | —         |
| <i>Nodularia spumigena</i>              | p         | fq        |
| <i>Oscillatoria limnetica</i>           | fq        | —         |
| <i>Spirulina jenneri</i> (Stiz.) Geitl. | r         | —         |
| <i>S. subsalsa</i>                      | r         | —         |
| <i>Actinocyclus octonarius</i>          | r         | —         |
| <i>Chaetoceros danicus</i>              | r         | —         |
| <i>C. holsaticus</i>                    | p         | —         |
| <i>C. subtilis</i>                      | r         | —         |
| <i>C. wighamii</i>                      | p         | p         |

Table 3. Continued.

| Species                                | 1966–1970 | 1979–1982 |
|----------------------------------------|-----------|-----------|
| <i>Chaetoceros</i> sp.                 | r         | p         |
| <i>Coscinodiscus granii</i>            | r         | fq        |
| <i>Cyclotella ocellata</i> Pantocsek   | r         | —         |
| <i>C. praetermissa</i>                 | r         | —         |
| <i>Melosira binderana</i> Kütz.        | r         | —         |
| <i>Melosira</i> sp.                    | —         | r         |
| <i>Skeletonema costatum</i>            | r         | r         |
| <i>Stephanodiscus hantzschii</i>       | fq        | —         |
| <i>Thalassiosira baltica</i>           | fq        | —         |
| <i>T. levanderi</i>                    | r         | —         |
| <i>T. fluviatilis</i>                  | r         | —         |
| <i>Biddulphiales</i> $\phi$ 10 $\mu$ m | —         | fq        |
| <i>Achnanthes taeniata</i>             | r         | r         |
| <i>Diatoma elongatum</i>               | r         | —         |
| <i>Gomphonema</i> sp.                  | r         | —         |
| <i>Synedra acus</i>                    | r         | —         |
| <i>S. ulna</i>                         | r         | —         |
| <i>Synedra</i> sp.                     | r         | r         |
| <i>Tabellaria fenestrata</i>           | r         | —         |
| <i>Bacillariales</i>                   | p         | r         |
| <i>Pyramimonas</i> sp.                 | p         | fq        |
| <i>Carteria</i> sp.                    | fq        | —         |
| <i>Chlamydomonas</i> sp.               | r         | r         |
| <i>Dictyosphaerium pulchellum</i>      | —         | r         |
| <i>Crucigenia quadrata</i>             | r         | —         |
| <i>Kirchneriella contorta</i>          | r         | r         |
| <i>Oocystis borgei</i>                 | r         | p         |
| <i>O. parva</i>                        | p         | —         |
| <i>O. submarina</i>                    | p         | —         |
| <i>Scenedesmus bicellularis</i>        | p         | —         |
| <i>Monoraphidium contortum</i>         | p         | p         |
| <i>M. mirabile</i>                     | r         | —         |
| <i>M. setiforme</i>                    | r         | —         |
| <i>Planctonema lauterbornii</i>        | —         | p         |
| <i>Ulotrichales</i>                    | —         | r         |
| <i>Cladophora</i> sp.                  | r         | —         |
| <i>Mougeotia</i> sp.                   | r         | —         |

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## THE RESPONSE OF PERIPHYTIC ORGANISMS TO THE LOAD OF ORGANIC AND INORGANIC NUTRIENTS FROM A FISH FARM

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LESKINEN, E., KOLEHMAINEN, O. & ISOTALO, I. 1986. The response of periphytic organisms to the load of organic and inorganic nutrients from a fish farm. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

During summer 1983 the growth of periphyton was studied in the surroundings of a fish farm in Houtskari, an oligotrophic area in the Archipelago Sea. The annual loads of phosphorus, nitrogen and organic matter from the fish farm were estimated at 1 300 kg, 10 000 kg and 20 000 kg respectively, the maximum monthly load occurring in September–October (45 % of the total). Panels of glass were used as artificial substrata to collect the periphytic material during exposure times of two weeks. Chlorophyll *a* and organic carbon were measured on the periphyton. The values obtained were compared with the results of water analyses. In some cases the total phosphorus values were higher at the sampling stations nearer the fish farm. Otherwise, the water analyses showed no differences between the sampling stations. The periphyton was sampled at three different times. The results revealed a gradient starting from the fish farm. The gradient was especially marked in the last sampling period in August–September, when the load of wastes reached the maximum for the research period.

Index words: Periphyton, artificial substratum, fish farm, water quality

### 1. INTRODUCTION

During the last decade production of fish in net cages in open water has become a popular occupation in the Archipelago Sea. The resultant load of organic matter and nutrients is known to have harmful effects in the surroundings, such as formation of mucus on fishing gear in the vicinity of the fish farms. In the freely flowing water systems of the archipelago, however, it can be difficult to determine the dispersal of the load by standard physical and chemical water analysis. In

many instances periphyton has been shown to be a good indicator of water quality (e.g. Collins and Weber 1978, Cairns 1978). In Finland it has mostly been used to detect the effects of fish farms in limnic surroundings (Pitkänen and Poikolainen 1983).

The aim of this study was to test how well the growth of periphyton indicated the extent of the effects of the load in the brackish water environment of the Archipelago Sea. The results were compared with the chlorophyll *a* and nutrient content of the surrounding water.

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## 2. MATERIAL AND METHODS

The research area was the waters around a fish farm, near Houtskari in the Archipelago Sea, southwest of Finland. The fish farm is located in a strait with a width of about 300 m and length of 2 000 m, between the islands Mussalo and Björkö (Fig. 1). The time during which the fish are fed is from May to November. The monthly load of organic matter and nutrients was calculated from the fish farmer's figure for the amount of fodder given to the fish (Fig. 2). In 1983 the total loads of organic matter, phosphorus and nitrogen were estimated at 20 000 kg, 1 300 kg and 10 000 kg, respectively, the maximum monthly load occurring in September–October.

The periphyton was collected at 16 stations (Fig. 1) from panels of glass hanging vertically at a depth of 0.3 m for a period of two weeks. The periphyton

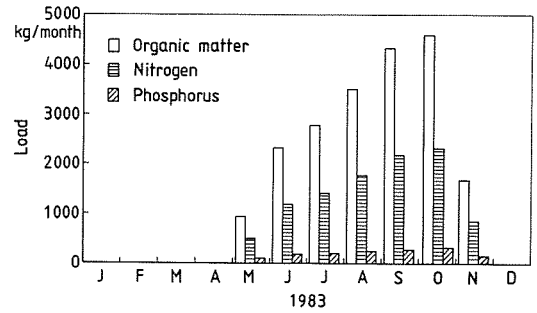


Fig. 2. The estimated monthly loads of organic matter, nitrogen and phosphorus coming from the fishfarm in 1983. The figures were calculated from the amount of fodder reported by the fish farmer.

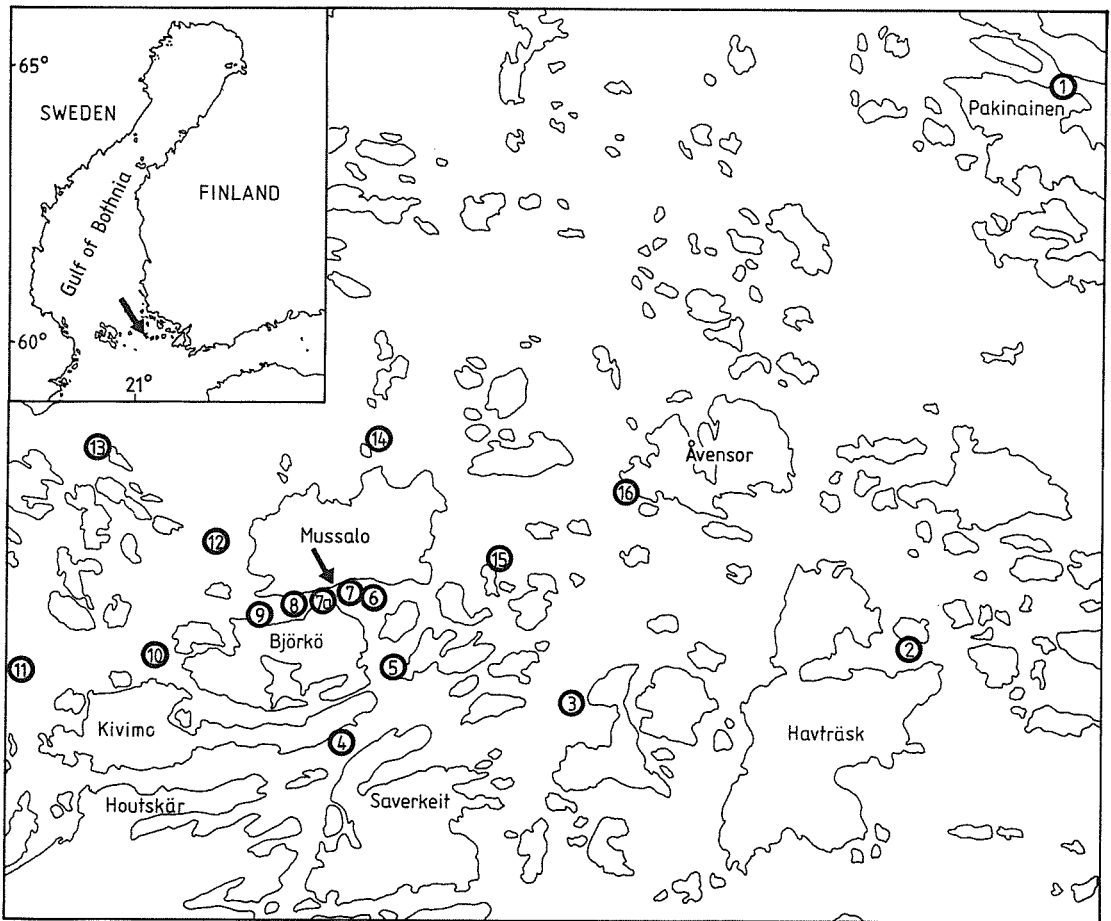


Fig. 1. Study area in the Archipelago Sea. The sampling sites are indicated with numbered circles. The location of the fish farm is marked with an arrow.

was sampled three times during summer 1983 (22.6.—6.7., 20.7.—3.8. and 22.8.—5.9.1983). Water samples were taken four times during the sampling period at the same stations (22.6., 6.7., 3.8. and 5.9.1983), and the water was analysed for chlorophyll *a*, total nitrogen, total phosphorus and phosphate phosphorus, by the methods of the National Board of Waters, Finland (1981). The periphytic variables measured were chlorophyll *a* and organic carbon. For the measurement of chlorophyll *a* five replicate glass panels with the attached periphytic material were submerged separately in the extraction solution of 94 % ethanol for about 24 h. The measurements of chlorophyll *a* were made with a spectrophotometer at wavelengths of 665 and 750 nm or with a fluorometer. Correction for phaeopigments was made according to Marker et al. (1980), with the final HCl concentration of 0.01 M. The results were calculated after Lorenzen (1967) using 83.4 as extinction coefficient (Wintermans and DeMots 1965) and 1.7 as maximum acid ratio (Marker et al. 1980). The organic carbon content of the periphyton was measured from five replicates by wiping the attached material from the glass surface on to an ignited glassfibre filter. The samples were stored in ignited foil cups until the measurement with an infrared gas analyzer according to Salonen (1979).

### 3. RESULTS

At every sampling (Fig. 3), the values for the periphytic variables were higher near the fish farm. During the first sampling period, in 22.6.—6.7.1983, the growth of periphyton was most rapid at sampling sites to the west of the fish farm. The panels positioned nearest the net cages had unfortunately disappeared before the sampling, and thus the nearest stations from which periphyton was collected were about 500 m from the fish farm. Even so, the values of periphytic chlorophyll *a* and organic carbon in the most strongly affected area were 10-fold those in the control area (sites 1—2 and 11—14, Fig. 3). During the second sampling period, in 20.7.—3.8.1983 the east wind had apparently resulted in the accumulation of periphyton on substrata west of the fish farm. At the last sampling time, in 22.8.—5.9.1983, there was a gradient both east and west of the farm (Fig. 3). The ratio of chlorophyll *a* to organic carbon varied between the sampling stations and the sampling

Table 1. The ratio of periphytic chlorophyll *a* to organic carbon at the three sampling times.

| Sampling station | Sampling time |          |          |
|------------------|---------------|----------|----------|
|                  | 6.7.1983      | 3.8.1983 | 6.9.1983 |
| 1                | 0.025         | 0.021    | 0.038    |
| 2                | 0.081         | 0.060    | 0.076    |
| 3                | 0.049         | —        | 0.130    |
| 4                | 0.045         | 0.094    | 0.046    |
| 5                | 0.066         | 0.060    | 0.150    |
| 6                | 0.212         | —        | 0.084    |
| 7                | —             | 0.092    | 0.101    |
| 8                | —             | 0.032    | 0.107    |
| 9                | 0.114         | 0.037    | 0.020    |
| 10               | 0.147         | 0.106    | 0.017    |
| 11               | 0.188         | 0.058    | —        |
| 12               | 0.117         | 0.064    | —        |
| 13               | 0.139         | —        | —        |
| 14               | 0.074         | —        | —        |
| 15               | —             | —        | 0.008    |

periods (Table 1). The trend at all samplings, however, was an overall increase in the ratio towards the fish farm, except for the nearest stations to the west of the farm, where the ratio was usually lower.

The water analyses showed the low level of nutrients typical of the research area. A decrease in the values of chlorophyll *a* and nutrients could be observed throughout the sampling period until the last sampling time at the beginning of September, when chlorophyll *a* and total phosphorus had increased. The variables measured on the water did not seem to be affected by the load coming from the fish farm. Only at the second sampling time, on 6.7.1983, did the values for total phosphorus show a slight rise by the fish farm.

### 4. DISCUSSION

The effects of the fish farm were reflected by an increase in periphytic growth. Similar results were obtained in experiments made in limnic surroundings (Heinonen 1981, Marja-aho 1982, Törmälä and Markkanen 1983). The water analysis gave no reliable evidence of differences between the sampling stations (Fig. 3), as could be expected in an archipelago area where the hydrological conditions allow free exchange of water. The periphytic sampling revealed that at all times the direct effects of the load were limited to the relatively narrow and shallow strait where the fish farm is

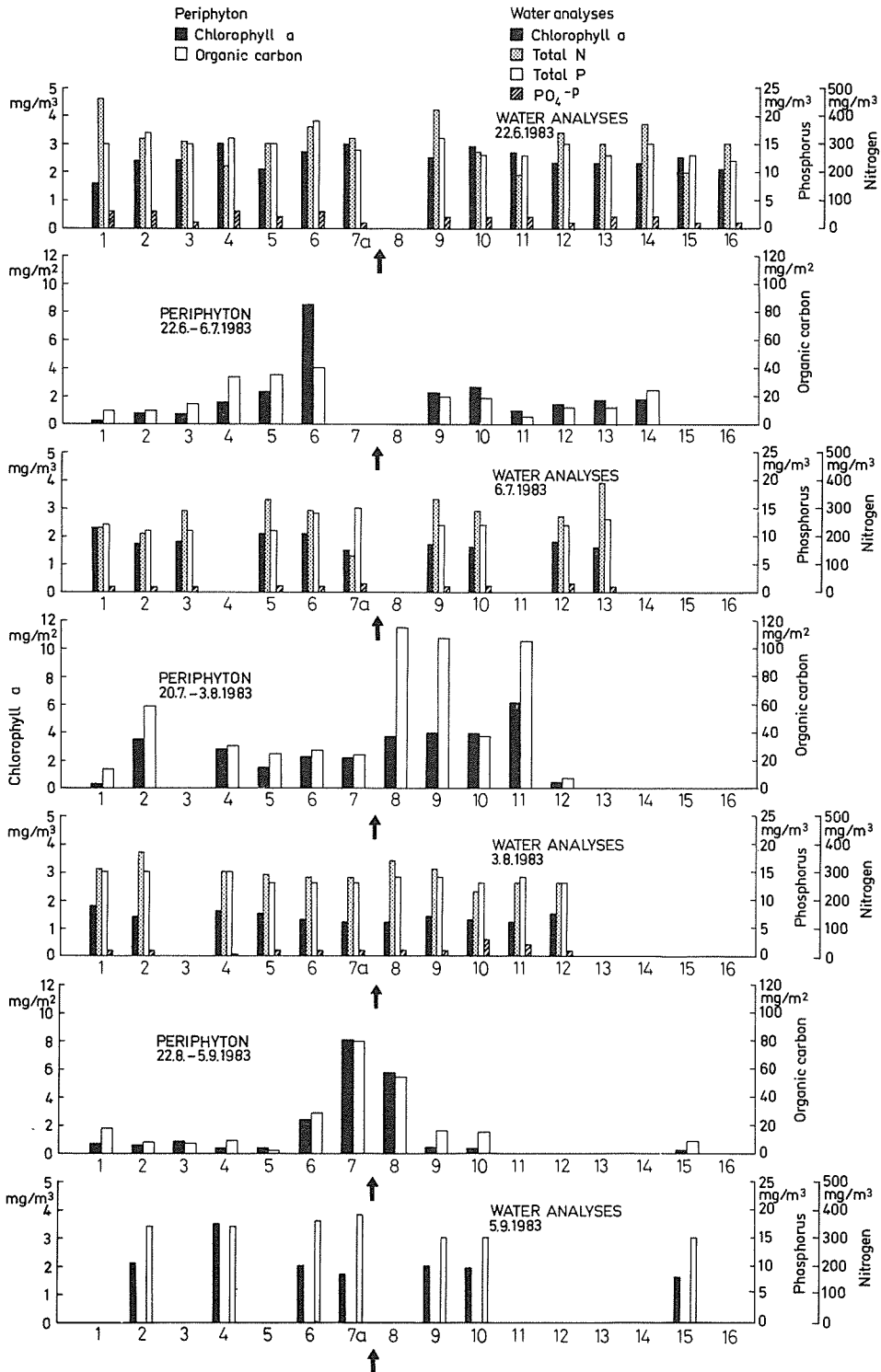


Fig. 3. The results of periphytic sampling and water analyses. The sampling sites are indicated with numbers and the location of the fish farm with an arrow.

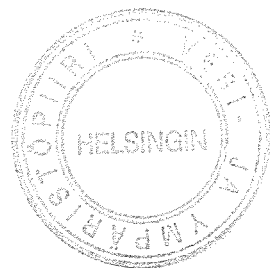
situated. The dispersal area did, however, vary according to the direction of the currents.

The growth of periphyton varied between sampling times (Fig. 3). The periphytic biomass was best expressed by organic carbon, which was highest in July–August, when the conditions were most favourable for the growth of periphyton. The relative amounts of chlorophyll *a* also differed at the three sampling times. The level of chlorophyll *a* varies according to the species composition and the level of autotrophy on the substrata and according to the available light. The species composition and the level of autotrophy change with the water quality and time of year. The available light also depends on the weather conditions. The spatial differences due to differences in water quality are, however, most prominent and thus the periphytic chlorophyll *a* seems to be a suitable variable for monitoring purposes, as suggested by Heinonen (1984).

The differences between sampling periods raise the question of the most appropriate sampling time and the number of samplings needed to give reliable results. Both will depend on the aims of the monitoring programmes and the resources available. The sampling should take place when the load is at its maximum, but should also take account of the annual succession of periphytic organisms.

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## CLADOPHORA GLOMERATA (CHLOROPHYTA) AS AN INDICATOR OF COASTAL EUTROPHICATION

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MÄKINEN, A. & AULIO, K. 1986. *Cladophora glomerata* (Chlorophyta) as an indicator of coastal eutrophication. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The biomasses and contents of nitrogen and phosphorus were measured on samples of the filamentous green alga *Cladophora glomerata* from the Archipelago Sea, SW Finland. The biomasses were highest in areas of intensive fish farming, and the values recorded near the major population centre (Turku) also clearly exceeded those obtained on the outer archipelago. The tissue levels of the major nutrients were high in the vicinity of Turku and in the fish farming areas. The algal tissues showed excess accumulation of nitrogen, but phosphorus appeared to be a limiting factor for the growth of *Cladophora*. Samples taken throughout the growing season revealed that the maximum biomasses are reached earlier than is generally assumed. Evident interactions were found between the trophic status of the sea and the biomass of *Cladophora*.

Index words: Algae, biological monitoring, *Cladophora*, eutrophication, fish farming, nutrient ecology, primary production

### 1. INTRODUCTION

The process of eutrophication has been well studied with regard to the nutrient enrichment of seawater and the reactions of phytoplankton and invertebrates. Much less is known about the effects of nutrient loadings on littoral belts of macroalgae.

The waters receiving anthropogenic nutrient loads (from urban centres, agriculture or fish farming) can support massive stands of filamentous algae, since these organisms are very tolerant of the unfavourable conditions often associated with eutrophication (e.g. occasional anaerobic conditions, toxic substances). On the southwestern

coast of Finland, the nutrient load imposed on the sea by the expanding fish farming industry already exceeds the effects of urban waste discharges (Eklund 1984).

The species composition of macroalgal communities has been used as an index of changes in the Archipelago Sea (Peussa and Ravanko 1974, Mäkinen et al. 1984). In the present study, the biomass of the dominant littoral species, *Cladophora glomerata* (L.) Kuntz., was examined in relation to the trophic status of the sea in six sampling areas representing varying degrees of eutrophication. Both spatial and temporal variation was studied.

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## 2. MATERIAL AND METHODS

Algal samples were collected from six sampling locations in the Archipelago Sea, SW Finland (Fig. 1). The sampling was designated to include habitats with varying trophic status. The site in Hirvensalo is situated near the major population centre in the area (Turku; 160 000 inhabitants). The sites in Kustavi and Houtskär represent areas of intensive fish farming, where the annual production of rainbow trout is 435 and 150 tons, respectively. The sites in Pakinainen and Stenskär were control areas, where the influence of anthropogenic eutrophication was negligible. The water at Hirvensalo and Kustavi is markedly eutrophicated, and at Houtskär and Seili the nutrient concentrations are slightly elevated, but the water of the control areas represents the natural trophic status (Jumppanen 1981).

The algal samples were taken from similar habitats with regard to quality of bottom, degree of exposure and point of compass. This ensured that environmental factors other than trophic status had as little effect as possible (cf. Jansson 1974). At Seili, five independent subsamples of 125 cm<sup>2</sup> were taken at intervals of two weeks during the growing season in 1982 and 1983. At the other sites three to five replicate samples were taken at the end of the growing season in 1983. The method of sample collection is described in detail by Jansson (1974). Biomass values per unit area were calculated after oven-drying at 60 °C for 24 h (Dybern et al. 1976).

The nitrogen contents of the algal tissues were determined by the micro-Kjeldahl method after hydrogen peroxide-sulphuric acid digestion (Allen 1974). For the phosphorus analyses, colorimetric measurements were made by the molybdenum blue procedure after mixed acid digestion were used for the phosphorus analyses (Allen 1974). The concentrations of total nitrogen and total phosphorus in the water samples were determined by the local water laboratory of the National Board of Waters, Finland, using official standard methods.

## 3. RESULTS AND DISCUSSION

### 3.1 Variation between the sites

Table 1 presents the biomasses, tissue nutrient contents and accumulation per unit area of nitrogen and phosphorus in the filamentous green alga *Cladophora glomerata*.

The algal biomasses revealed a clear spatial pattern. The highest values were measured near the fish farming centre in Kustavi (max. 461 g m<sup>-2</sup>). These exceed the levels obtained at the control site at Stenskär by a factor of three to four (Table 1). In the neighbourhood of Turku the biomass values also exceeded those recorded at the control sites, the values being lower, however, than in the fish

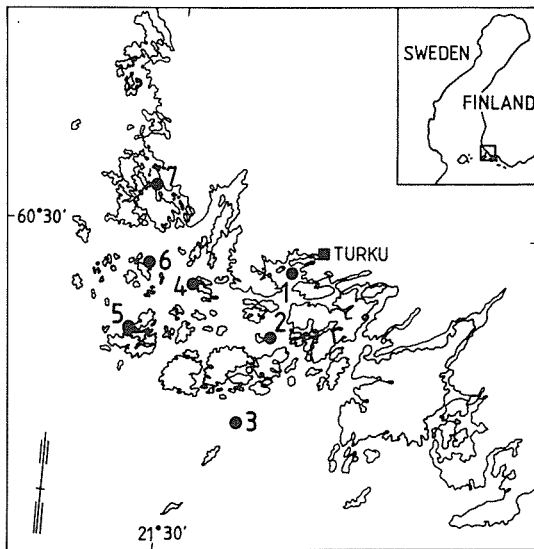


Fig. 1. The study area. 1. Hirvensalo, 2. Seili, 3. Stenskär, 4. Pakinainen, 5. Houtskär, 6. Iniö, 7. Kustavi.

Table 1. Biomass values, tissue nutrient contents and accumulation per unit area in stands of *Cladophora glomerata*. The samples were taken within one week (27.07.83—03.08.83). All the results are expressed on a dry weight basis.

| Site       | Biomass           | Nitrogen           |                   | Phosphorus         |                   |
|------------|-------------------|--------------------|-------------------|--------------------|-------------------|
|            | g m <sup>-2</sup> | mg g <sup>-1</sup> | g m <sup>-2</sup> | mg g <sup>-1</sup> | g m <sup>-2</sup> |
| Hirvensalo | 239.9             | 26.4               | 6.3               | 1.8                | 0.44              |
| Houtskär 1 | 222.1             | 37.1               | 8.2               | 1.6                | 0.34              |
| ” 2        | 168.9             | 29.3               | 5.0               | 1.3                | 0.34              |
| ” 3        | 271.4             | 30.7               | 8.3               | 1.4                | 0.38              |
| ” 4        | 70.1              | 22.9               | 1.7               | 0.6                | 0.05              |
| ” 5        | 106.1             | 30.7               | 3.3               | 0.7                | 0.08              |
| ” 6        | 183.9             | 40.0               | 7.4               | 1.4                | 0.26              |
| Kustavi 1  | 460.6             | 16.4               | 7.6               | 0.5                | 0.25              |
| ” 2        | 216.8             | 18.6               | 4.0               | 1.1                | 0.25              |
| ” 3        | 433.1             | 28.6               | 12.4              | 1.2                | 0.52              |
| ” 4        | 248.6             | 22.9               | 5.7               | 0.8                | 0.19              |
| ” 5        | 340.3             | 22.9               | 7.8               | 0.8                | 0.26              |
| ” 6        | 183.9             | 22.1               | 4.1               | 0.7                | 0.13              |
| Pakinainen | 229.9             | 23.6               | 5.4               | 0.6                | 0.36              |
| Stenskär   | 131.8             | 18.6               | 2.5               | 0.9                | 0.13              |

farming areas.

The biomass values of the *Cladophora* belts were high throughout the study area. The present figures for algal biomass at both the control and eutrophicated sites markedly exceed the values reported earlier from the Archipelago Sea (Rönnerberg 1981) and the Tvärminne area (Hällfors et al. 1975). Wide variation in biomass values is caused by yearly fluctuation in the growth dynamics, and by differences between sampling dates and microhabitats, which hampers direct comparisons of reported values (cf. Jansson 1974).

The nutrient contents of the algal samples were highest in Hirvensalo and in the fish farming centre near Houtskär. The maximum level of phosphorus ( $1.84 \text{ mg g}^{-1}$ ) in Hirvensalo indicates nutrient discharges from Turku. Nearly as high values were recorded in Houtskär (max.  $1.55 \text{ mg g}^{-1}$ ). The contents of nitrogen showed a similar spatial pattern, the maximum levels being recorded in Houtskär (max.  $40 \text{ mg g}^{-1}$ ). The values of Hirvensalo slightly exceeded the levels obtained from the other fish farming area at Kustavi. The elevated contents of nitrogen in the material from Houtskär indicate loadings of N from fish farming (cf. Eklund 1984). In Kustavi the algal nutrient levels were lower than in Houtskär, although fish farming is much more intense in the former area. This paradoxical situation can be explained by the stronger water circulation in the fish farming centre at Kustavi.

In most of the algal samples the nitrogen contents exceeded the critical concentration of  $21 \text{ mg g}^{-1}$  (this is the nutrient level of the thalli, below which optimal growth would be limited; Birch et al. 1981). In contrast, all values for phosphorus were below the critical concentration of  $3.3 \text{ mg g}^{-1}$  (Birch et al. 1981). According to the tissue nutrient contents, the availability of phosphorus is the chief factor regulating the production of *Cladophora* in the study area.

The ratio of nitrogen to phosphorus in the algal tissues ranged from 14 (in Hirvensalo) to 38 (in Stenskär). In most samples it varied between 20 and 30. The N/P ratios indicate accumulation of nitrogen in the marine nutrient reserves. In other studies on littoral filamentous algae the N/P ratio has generally varied between 10 and 20 (cf. Kornfeldt 1982). The variation between the most eutrophicated and the control areas directly reflects the nutrient levels recorded in the water analyses.

The accumulation of nutrients by the algal biomass was greatest in the fish farming area at Kustavi. The maximum amounts bound by the *Cladophora* belt were  $12 \text{ g m}^{-2}$  for nitrogen and

$0.5 \text{ g m}^{-2}$  for phosphorus. The variation in the biomass values and nutrient contents was high even within the sampling sites. The eutrophicated sites in Hirvensalo and Kustavi, however, showed clearly higher values than the control areas (Table 1). The capacity to retain large amounts of nutrients gives macroscopic algae an important role in the nutrient cycling processes in coastal waters. By filtering nutrients from the water, the filamentous algae with long life cycles can limit the growth of phytoplankton, thus affecting the rate of primary production in the littoral zone.

### 3.2 Temporal variations

The annual growth cycle of *Cladophora* was studied on Seili island (Fig. 1) in 1982 and in 1983. The study site was a sheltered rocky shore with no significant anthropogenic influence. The waters in the Seili area are slightly eutrophicated. The variation in the algal biomass values and nutrient contents of the thalli is presented in Fig. 2.

Very rapid growth of *Cladophora* thalli began in early May, and the biomasses reached the peak values (max.  $515 \text{ g m}^{-2}$ ) as soon as late June. During six weeks (19 May to 29 June) a 40-fold increase in biomass values occurred. After the highest biomass was reached, the algal belt declined sharply (Fig. 2). In 1983 the trend was essentially the same, but the peak biomass values were not reached until late July. The results of the two successive growing seasons revealed that much of the earlier research on the production of *Cladophora* has been conducted too late in the growing season, mostly in August when the detachment of algal biomass has already begun.

During the most active period at the beginning of the growing season, the nutrient contents and the biomass values of *Cladophora* showed opposite trends. High nutrient contents in spring can be regarded as a competitive advantage, because the nutrient resources in the ambient water tend to decrease simultaneously with the increase of macroalgal biomass. Excess consumption (accumulation of nutrients above the level required for optimal growth) ensures the adequacy of nutrient resources (nitrogen in the present study) during the growth period.

The contents of nitrogen and phosphorus in the algal tissues reflect the variation in the nutrient concentrations of sea water as well as the changes in the biomass values (Fig. 3). The marked increase in nutrient concentrations in sea water coincides

with the reduction of the biomass of the algal belt (cf. Figs. 2 and 3) and could thus be associated with the breakdown of algal biomass, with a short time lag.

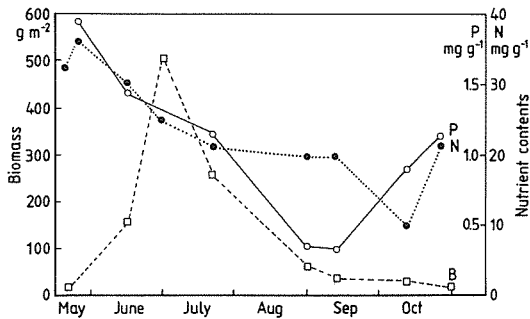


Fig. 2. Seasonal variation in the biomasses and in the nutrient contents (P = phosphorus, N = nitrogen) of *Cladophora glomerata* at Seili in 1982. All the values expressed on a dry weight basis.

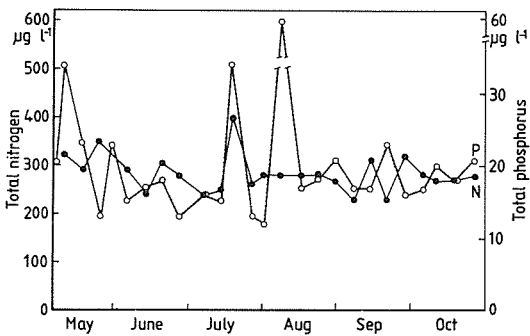


Fig. 3. Seasonal variation in the nutrient concentrations (P = phosphorus, N = nitrogen) of sea water in the *Cladophora* belt at Seili in 1982.

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## PULP MILL EFFLUENTS AND FISH — A CASE STUDY

Olof Sandström<sup>1)</sup>

SANDSTRÖM, O. 1986. Pulp mill effluents and fish — a case study. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

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The effects of bleached kraft mill effluent on fish were studied in the area receiving waste water from the Norrsundet pulp industry, SW Bothnian Sea. The catches were always small in the subarea closest to the pulp mill, but outside these waters was a zone with high densities of fish, especially roach and ruff. Growth studies and checks for external injuries were made on perch, a species showing stationary behaviour in this area. The fish in subareas near the effluent outfall had grown significantly better than their controls. Epithelial damage was often noted and the caudal fins of perch from the vicinity of the outfall were significantly shorter than the fins of control fish.

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Index words: Fish, pulp mill effluents, Baltic Sea

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### 1. INTRODUCTION

Pulp mill effluents can cause harm to fish in a variety of ways, either directly by modifying behaviour or physiological functions, or by their impact on fish habitats (Kelso 1977, Oikari and Soivio 1977, Leach and Thakore 1977, Lehtinen and Oikari 1980). Large areas of the sea may be affected by the discharge of such effluents, as in Sweden, where pulp production is mainly found along the coasts of the Gulf of Bothnia. The effects on the fish stocks, however, are still incompletely known.

This paper presents information from an ongoing research project, started in 1983 by the Swedish national environment protection board, on the effects of bleached kraft mill effluent (BKME) on fish (Neuman et al. 1984). The presentation considers three common demersal and stationary species in Baltic coastal areas, i.e., perch (*Perca fluviatilis*), roach (*Rutilus rutilus*), and ruff

(*Gymnocephalus cernua*), and concentrates on the distribution, growth and some external defects in adult fish.

The research project covers both pelagic and demersal species. The ultimate aim is to reach a general understanding of the effects of BKME on the habitats, reproduction, growth and health of the fishes and to discover whether these cause changes in fish production and distribution.

### 2. MATERIAL AND METHODS

The studies were made in the coastal area receiving BKME from the Norrsundet pulp industry, in the SW part of the Bothnian Sea (Fig. 1). This system was chosen mainly as it lacks influence from rivers. The effects on distribution and reproduction are

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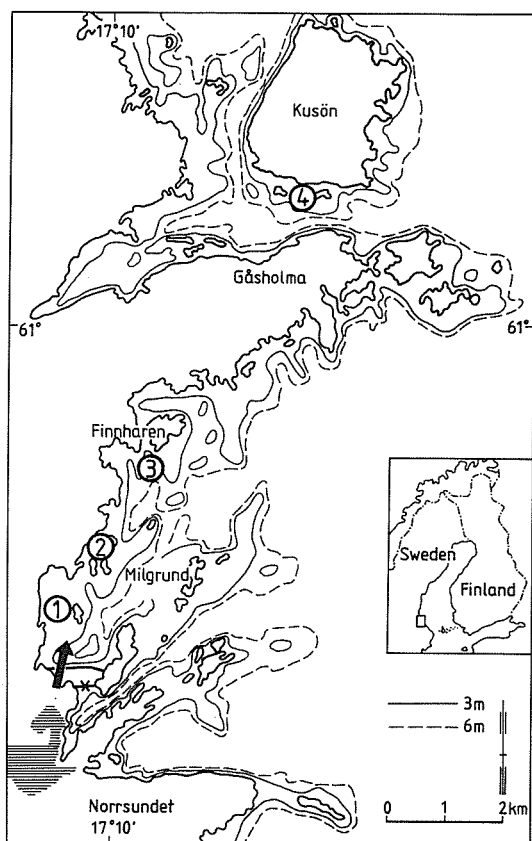


Fig. 1. The study area and its division into subareas. The waste water discharge is indicated by an arrow.

more easily distinguished when there is no migration to freshwater areas.

Fishing was done at eight stations located in four subareas; two stations per area. Bottom set nets, 3 m deep and 35 m long, consisting of five parts with different mesh sizes (17–50 mm), were used. Two nets were placed at each station. As the water depths at all stations were close to 3 m, the main part of the water column was covered by the nets. The nets were set in the afternoon (15.00–16.00) and lifted the following morning (07.00–09.00).

The captured fish were weighed (nearest 0.1 g). Their total lengths, and in perch caudal fin lengths, were measured to the nearest mm. Scales or opercula (in perch) were preserved for growth analyses. All fish were examined for external defects. The material used in this report was collected in late June and July 1983 (6 fishings) and in May and early June 1984 (12 fishings). Control material concerning individual growth rates was collected in Finbo, NW Åland.

### 3. RESULTS

The abundances and distributions of the species followed a definite pattern (Fig. 2). Results from samplings during the spawning period in spring did not differ markedly from the summer catches. Closest to the discharge, practically no fish were caught. In subareas 2 and 3, however, the densities of roach and ruff were considerable and higher than in the control subarea, 4. The catches of perch were low at all stations except in the control subarea.

The external examination revealed that most perch from subareas 1, 2 and 3 suffered from fin injuries, evident as losses of the soft, outer parts of the fins. Regression lines were constructed for caudal fin lengths against total body lengths (Fig. 3). As can be seen in the figure, the fish from waters near the outfall had shorter fins, regardless

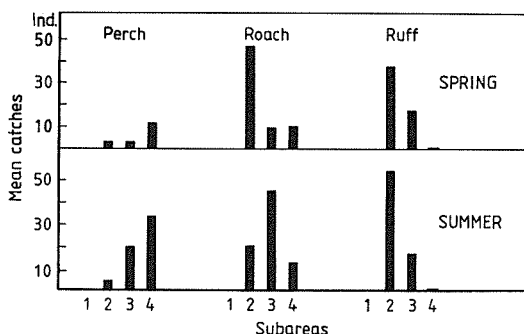


Fig. 2. The distribution of perch, roach and ruff illustrated as the mean catches per station in different subareas during the spring and summer seasons.

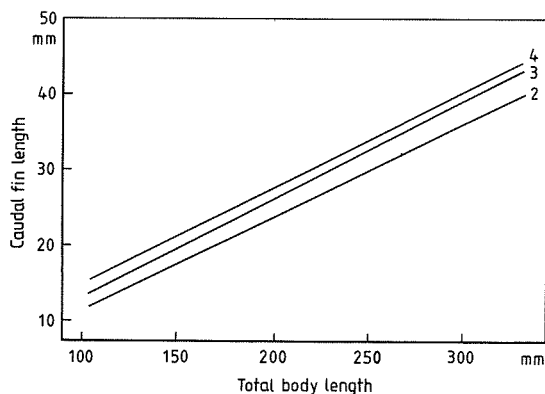


Fig. 3. Regression lines constructed for caudal fin lengths against total body lengths. The lines are drawn for subareas 2, 3 and 4.

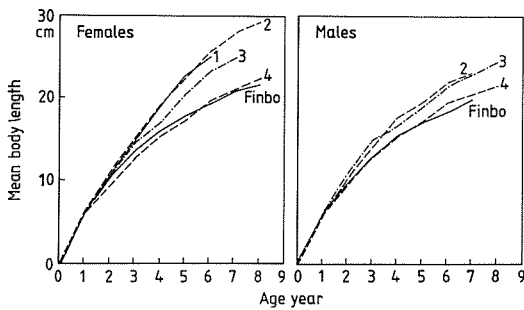


Fig. 4. Growth curves for male and female perch caught in different subareas. The mean length at a certain age was established by back-calculations on gill covers. Data from a natural population at Finbo, Åland, are added to the figure.

of the size of the fish. The difference between subareas 2 and 4 was statistically significant (*t*-test).

Analyses of growth expressed as mean yearly length increments, are presented for 391 perch, 248 females and 143 males (Fig. 4). Control data from Finbo, NW Åland, are added to the figure. As the age distributions differ between samples, strict comparisons cannot be made. The observations indicate, however, that fish captured in the subareas near the effluent outfall had grown faster than those from the control waters.

#### 4. DISCUSSION

The distribution pattern demonstrated here was stable during the two study years, although the total catch varied with the seasons. Roach and ruff were very abundant in the moderately polluted subareas, perhaps an effect of organic enrichment. Perch seem to be more sensitive to the deleterious impact of the effluent and were never common near the effluent outfall. Earlier studies (Neuman and Sandström 1982, Hanson 1982) have yielded similar results, which points to the conclusion that the observed distribution pattern may be of general character.

The low abundances of perch are evidently at least partly due to toxic effects, as most of the fish caught in subareas 1–3 had fin injuries. As it is hard to find comparable information on injured

fish in the literature, further studies are needed to test the generality of these observations.

Although most growth studies do not reveal any effect of exposure to the concentrations expected in the waters near outfalls, BKME has been shown to stimulate growth in at least one experiment (McLeay and Brown 1974). Effects on internal regulatory systems were suggested as an explanation. Regardless of whether the faster growth reflects an action on the physiology or is the effect of, for example, low competition pressure, it is astonishing to find enhanced growth in fish living in habitats with properties causing malfunctions in such vital parts as the epithelial system.

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## REVIEW ON HARMFUL SUBSTANCES IN THE GULF OF BOTHNIA

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& Mats Olsson<sup>4)</sup>

KRISTOFFERSSON, R., SANDLER, H., MIETTINEN, V. & OLSSON, M. 1986. Review on harmful substances in the Gulf of Bothnia. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The latest information published on the occurrence and effects of harmful substances in the Gulf of Bothnia is reviewed. The concentrations of some harmful substances, e.g. DDT and PCB, seem to be decreasing in the environment. However, new harmful substances, for example chlorinated terpenes, chlordanes, polychlorinated dibenzodioxins and dibenzofurans and compounds in discharges from the pulp and paper industry, are being discovered.

The effects of pollution are found as physiological and morphological changes in fish and mussels. Enforcement of the research essential to an evaluation of the degree of pollution by harmful substances is stressed.

Index words: Heavy metals, chlorinated hydrocarbons, effects of pollution, Gulf of Bothnia

### 1. INTRODUCTION

The first survey of compounds toxic to the environment in the Gulf of Bothnia was presented in 1981 (Edgren et al. 1983). The conclusion was that some heavy metals and organic compounds, especially DDT and PCB, had accumulated in fish, sea birds and seals to levels believed to cause ecological damage. The most alarming effects of the accumulated substances were observed in seals and white-tailed eagles, whose existence was threatened because of reproductive disturbances. As a result of the survey, the Committee for the Gulf of Bothnia agreed at its annual meeting in 1981 that further measures were needed to protect the seal populations in the Gulf of Bothnia.

The research on toxic substances has continued in monitoring programmes, in regional and special investigations and in local recipient control programmes. Some of the major research programmes commenced in recent years, but not reported at the seminar, are introduced below.

Monitoring studies are conducted on a national basis in coastal waters and in the open sea areas. Some of them are performed within the frame of the Baltic Monitoring Programme (BMP) of the Helsinki Commission, which includes analysis for the heavy metals Zn, Cu, Pb, Cd and Hg, and for DDT and PCB compounds in fish and benthic invertebrates in the open sea. Annual sampling was started in 1979. In the Bothnian Sea the fish species studied are Baltic herring and cod, in the Bothnian

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Bay only Baltic herring. The only benthic animal studied is the crustacean *Mesidotea entomon*.

For some years, chlordanes have also been determined on the monitoring material, including water and sediment samples.

In the Finnish coastal monitoring programme, *Mesidotea entomon* and *Macoma baltica* samples are collected annually at 10 stations and analysed for heavy metals and PCB and DDT compounds, and similar analyses are performed on fish samples collected every fifth year in 12 areas.

Within the Swedish environment monitoring programme, samples are collected and analysed annually in one archipelago area and two open sea areas believed not to be locally polluted.

The most ambitious regional investigations concerning heavy metals have been conducted in the waters receiving the effluents of the Rönnskär Works, a complex sulphide ore smelter in Skellefteå, and the Vuorikemia Works, a titanium dioxide plant in Pori.

The possible reasons for the decline of trawl catches of autumn spawning herring are being studied in the locally polluted waters outside Pori. The major pollutants of the area are iron, heavy metals and sulphuric acid.

The project "Fish and metals" in Sweden and the projects dealing with the effects of pulp and paper mill effluents in Finland and Sweden are concerned with the problem of complex pollution by entire industrial branches. Part of this research has been carried out in the Gulf of Bothnia.

A joint Scandinavian project, "Ecotoxicological methods for the aquatic environment", coordinated by Nordforsk, was completed in 1982 (Nordforsk 1982). This work formed the basis for the development of methods of studying the effects of toxic and harmful compounds. The toxicity tests and physiological methods have been adapted to the conditions typical of our two countries.

A great deal of research is in progress in Sweden and Finland, and not all the results from the Gulf of Bothnia are presented at the triennial seminars organized by the Committee or in the reference list compiled each year for the annual reports. The group hopes that the list can be improved, possibly by establishing a computer register of the literature.

New results obtained from studies on toxic compounds in the Gulf of Bothnia were presented in 11 poster presentations and can be grouped as follows:

- Toxic compounds in water and sediments
- Bioaccumulation
- Biological effects of toxic compounds.

## 2. TOXIC COMPOUNDS IN WATER AND SEDIMENTS

The distribution of some chlorinated phenolic compounds discharged from a sulphate pulp mill was studied in the receiving waters by Fogelqvist et al. (1986). The compounds were determined in both the water and sediments to elucidate their pathways and accumulation in the sea. Chloroform was used in parallel as a tracer to measure the plume and the dilution of the effluent. The results showed that the transport of chlorophenolics was dominated by dilution and adsorption processes, and their behaviour in the receiving area was correlated to the apparent lipophilicity of the chlorophenolics. The most lipophilic chlorophenolics settle in the inner part of the discharge area while the more hydrophilic components are transported farther seaward.

Perttilä and Haahti (1986) presented concentrations of chlorinated hydrocarbons recorded in the water and sediments in the Gulf of Bothnia. The situation has improved as regards DDT and PCB compounds. Due to the bans and limitations on their use, the concentrations are lower in the upper layers of the sediments. It is interesting to note that chlordanes can be found in the sediments deposited as early as the mid 1960s.

## 3. BIOACCUMULATION

Aulio (1986) has studied aquatic macrophytes as indicators of heavy metal pollution. The plants were sampled in the estuary of the River Kokemäenjoki. The submerged taxa accumulated the highest concentrations of metals (Cu, Zn, Mn, Fe), while less efficient bioaccumulation capacity was shown by the floating-leaved and emergent plant species, especially the latter. Some species showed selective accumulation of some metals.

Autio and Kangas (1986) have determined heavy metal concentrations of the algae *Fucus vesiculosus*, *Pilayella littoralis*, *Cladophora glomerata* and *Ceramium tenuicorne*. Only *Fucus* seems to be of value as a bioindicator of heavy metal pollution.

Rappe et al. (1986) have identified highly toxic polychlorinated dioxins and dibenzofurans in fish, seal and bird samples from the Baltic Sea. The levels are very low, i.e. in the ppt range. The study strongly indicates general contamination of our environment by these compounds. Incineration of

municipal and hazardous waste cannot be excluded as a possible source.

The study of Sandler (1986) presents heavy metal concentrations recorded for *Mesidotea entomon*, *Pontoporeia affinis* and mysids. The iron and copper concentrations were high on a global scale, while the lead, cadmium and mercury concentrations were lower than reported elsewhere. *M. entomon* seemed to be suitable as a bioindicator only of lead, cadmium and mercury pollution.

#### 4. BIOLOGICAL EFFECTS OF TOXIC COMPOUNDS

The frequency of spinal deformities in the fourhorn sculpin was studied in relation to the concentrations of heavy metals and chlorophenols in the fish by Bengtsson and Miettinen (1986). The frequencies of defects in seven areas off the Finnish coast in the Bothnian Bay varied between 8 % and 25 % of the populations. The highest frequencies were recorded in the areas subjected to elevated loads of heavy metals. The connection between the frequency of defects and heavy metal concentrations in the fish cannot yet be definitely established, since further studies are necessary on factors contributing to the defects.

Larsson et al. (1986) have studied the physiology of fish in waters with lead, cadmium and complex metal pollution. The physiological functions most strongly affected were the immune defence, erythrocyte ALA-D activity and ion balance. Good agreement was found between the toxic effects detected in nature and those observed in laboratory tests.

Lindström (1986) sampled *Mytilus edulis* L. in the discharge area of an iron and steel works. He observed shell deformities, whose frequency decreased gradually from c. 45 % of the population near the works to c. 3 % at a distance of c. 4 km. Lindström concluded that the method appears to provide a promising tool for environmental monitoring in the vicinity of pollution sources.

Sunila and Lindström (1986) presented the long-term effects of short-term exposure (24 h) of *Mytilus edulis* L. to copper and cadmium. The shell morphology, length growth and survival of the mussel were observed one year after exposure. The growth of the test mussels was retarded and about half of them had shell deformities; only about one quarter of the control mussels had similar

deformities and only 3 % of the natural population.

Södergren (1984) gave a review of a project entitled "Biological effects of effluents from pulp industries". The 16 subprojects cover both chemical and biological aspects. The main attention is given to bleaching plant effluents. As part of this project, Sandström (1986) studied the effects of bleached kraft mill effluent on fish in coastal waters. Close to the mill was a zone with high densities of roach and ruffe. In the inner zone the perch had grown significantly better than perch from the control area, even though they showed epithelial damage and their caudal fins were shortened.

In order to obtain an overview of the role of harmful substances in the Gulf of Bothnia, the material presented at the seminar by the other three working groups should also be taken into account.

#### 5. CONCLUSIONS

It can be noted that the accumulation of information from the current research only exceptionally results in a reduction of the problems to be solved. An advantage of the work is that it provides continuous information on the impact of human activities. If we look at the situation pessimistically, however, we are forced to admit that the personnel and financial resources are probably too small to ensure the identification of new serious environmental threats. The development of new analytical methods, chemical, physiological and ecological, is so time-consuming and the funds required so vast that the resources available can hardly be said to correspond to the needs of research.

Certainly, it is encouraging to observe that the concentrations of some harmful substances, e.g. DDT and PCB, seem to be decreasing in our environment, as a result of extensive research and the wise decisions of the authorities. It is, however, important to emphasize that at the same time new harmful compounds are being discovered in the environment, for example chlorinated terpenes, chlordanes, polychlorinated dibenzodioxins and dibenzofurans and some other compounds in discharges from the pulp and paper industry. Our knowledge of these compounds is quite insufficient. Firstly, the chemical analysis is both

expensive and difficult so that the concentrations in the environment are seldom determined and the pollutant loadings are not known. Secondly, knowledge of the physiological and ecological effects is scarce and the cost of obtaining such information would be considerable. Yet this information, together with the loading assessment, is essential to an evaluation of the degree of pollution.

It seems that many years of research are needed to collect sufficient results and scientific information for an understanding of the ecological impact of various current discharges. A complication is that the pollutant load changes with time. For instance, it took 10–15 years for DDT and PCB to reach a critical pollution level in the Baltic Sea. Although the use of these and many other compounds has been banned for several years in Sweden and Finland, problems still remain, since both countries receive important quantities of airborne pollutants. The only way to fight this is to work through international organizations for cooperation between the countries concerned.

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## AQUATIC MACROPHYTES AS INDICATORS OF HEAVY METAL POLLUTION

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AULIO, K. 1986. Aquatic macrophytes as indicators of heavy metal pollution. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Samples of 15 plant species representing all the major life forms of aquatic vascular plants were collected from the Kokemäenjoki River estuary, western Finland, and their photosynthetic tissues were analysed for heavy metals. In general, the submerged taxa had the highest metal concentrations (Cu, Zn, Mn, Fe), while the floating-leaved and especially the emergent plants showed weaker metal accumulation. Of the individual plant species, the pondweeds *Potamogeton perfoliatus* and *P. obtusifolius* proved to be the most promising candidates for biological monitoring of heavy metal pollution in aquatic environments. The patterns of metal accumulation were further studied by calculating the discrimination factors for the dominant helophytes. These revealed specific iron enrichment in the horsetail, *Equisetum fluviatile*, and selective accumulation of Zn in relation to Fe by the cattail *Typha latifolia*.

Index words: Aquatic plants, heavy metals, pollution monitoring

### 1. INTRODUCTION

The number of studies describing the ability of aquatic plants to accumulate heavy metals from the aquatic environment has increased greatly in recent years. The reasons for the choice of biological indicators instead of conventional physico-chemical approaches can be summarized as follows (cf. Phillips 1977):

- (1) biological accumulators give an idea of the status of a particular water body over a long period of time,
- (2) several indicator organisms have selective accumulation of certain metals,
- (3) the strong accumulation of metals by macrophytic plants makes it possible to analyse even very

dilute solutions without fractionation and/or preconcentration procedures,

- (4) the analytical routines are easier to perform with biological materials than with abiotic environmental samples.

The present paper summarizes the patterns of metal accumulation in 15 species of littoral and aquatic plants collected from the Kokemäenjoki River estuary in western Finland. The accumulation capacity of each species was evaluated in material obtained from one particular sampling area to ensure that the availability of metals was as uniform as possible. Thus the differences in the metal concentrations could be attributed to species-specific characteristics rather than to variation in environmental conditions.

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## 2. MATERIAL AND METHODS

The plant samples were collected from the Kokemäenjoki River estuary in western Finland (61°34'N, 21°40'E) in August 1982. The material comprised 15 species of littoral and aquatic plants representing all the major life form categories, viz. tall emergent species (= helophytes), floating-leaved species and constantly submerged taxa. For helophytes and floating-leaved species, only leaf blades were used for the analyses, while shoot tips of about 5 cm long were used for submerged plants. The heavy metals (Cu, Zn, Mn, Fe) were determined with conventional methods of atomic absorption spectrophotometry after dry ashing of 0.5 g samples of dried tissues (Allen 1974). All the results are expressed as  $\mu\text{g g}^{-1}$  of dry weight.

With the sediment samples, the exchangeable fractions of metals were extracted with 0.5 N hydrochloric acid. The pH values of the sediment samples were measured with a glass electrode, and the soil organic matter was determined by dry ashing (at 475 °C for 4 h) (Allen 1974).

## 3. RESULTS AND DISCUSSION

The heavy metal concentrations in the 15 species of vascular aquatic plants are presented in Table 1. Comparison of the life forms reveals clear

differences in metal accumulation capacity between the three categories. The submerged species appeared to be the most efficient metal accumulators, while the floating-leaved and emergent taxa showed a weaker capacity for heavy metal enrichment (Table 1). The differences can be associated with different mechanisms of element uptake. The submerged taxa, which are able to accumulate metals both from the water via the shoots and from the sediments through root uptake (Denny 1980), were rich in all the metals studied (Table 1). The emergent taxa, depending solely on uptake from the sediments via the roots, were poor in metals as compared to the true aquatic plants.

The ranking of the elements by abundance was very regular in the 15 species studied: in the order of decreasing concentration, the sequence of the four heavy metals was  $\text{Mn} > \text{Fe} > \text{Zn} > \text{Cu}$ . The only exception to the general pattern was that the iron concentration exceeded the level of manganese in *Butomus umbellatus* (Table 1), the difference being, however, statistically nonsignificant.

All the metal concentrations in the plant tissues were high as compared to the data presented in the literature. The average levels of copper ( $14.1 \mu\text{g g}^{-1}$ ), zinc ( $120 \mu\text{g g}^{-1}$ ), iron ( $1421.7 \mu\text{g g}^{-1}$ ) and manganese ( $2794.4 \mu\text{g g}^{-1}$ ) clearly exceed the average concentrations recorded in aquatic plants (cf. the reviews by Hutchinson 1975 and by Dykijova 1979). These elevated levels indicate enhanced availability of metals in the study area. The load imposed on the Kokemäenjoki River

Table 1. The concentrations of copper, zinc, iron and manganese ( $\mu\text{g g}^{-1}$  dry wt) in 15 species of aquatic vascular plants collected from the Kokemäenjoki River estuary, western Finland.

| Species                    | Concentrations ( $\mu\text{g g}^{-1}$ ) |       |       |       |
|----------------------------|-----------------------------------------|-------|-------|-------|
|                            | Cu                                      | Zn    | Fe    | Mn    |
| Emergent species:          |                                         |       |       |       |
| Phragmites australis       | 5.8                                     | 30.5  | 235   | 573   |
| Schoenoplectus lacustris   | 6.1                                     | 27.2  | 163   | 1 355 |
| Typha latifolia            | 6.8                                     | 49.5  | 117   | 951   |
| Equisetum fluviatile       | 7.3                                     | 60.7  | 806   | 1 961 |
| Acorus calamus             | 4.7                                     | 41.0  | 169   | 1 168 |
| Butomus umbellatus         | 11.7                                    | 98.3  | 2 900 | 2 617 |
| Floating-leaved species:   |                                         |       |       |       |
| Nuphar lutea               | 16.7                                    | 195.2 | 759   | 1 917 |
| Nymphaea candida           | 10.6                                    | 117.4 | 540   | 1 223 |
| Polygonum amphibium        | 16.0                                    | 71.5  | 1 384 | 2 187 |
| Potamogeton natans         | 20.8                                    | 141.4 | 1 118 | 2 315 |
| Sparganium emersum         | 13.5                                    | 151.0 | 2 690 | 3 456 |
| Submerged species:         |                                         |       |       |       |
| Potamogeton perfoliatus    | 27.3                                    | 460.0 | 2 377 | 7 916 |
| Potamogeton obtusifolius   | 39.7                                    | 162.0 | 2 078 | 4 825 |
| Potamogeton gramineus      | 14.5                                    | 124.0 | 3 012 | 4 575 |
| Myriophyllum verticillatum | 10.2                                    | 83.2  | 3 600 | 4 875 |

Table 2. Correlation coefficients between the metal concentrations of the six emergent plants and the corresponding metal levels, pH and soil organic matter in the plant rooting medium (N = 30, significance levels: \*\*\* =  $P < 0.001$ , \*\* =  $P < 0.01$ , \* =  $P < 0.05$ ).

| Tissue element | Edaphic characteristics |         |        |        |         |       |
|----------------|-------------------------|---------|--------|--------|---------|-------|
|                | Fe                      | Mn      | Zn     | Cu     | org. %  | pH    |
| Fe             | .141                    | .979*** | .019   | .094   | -.067   | -.084 |
| Mn             | -.470                   | .818*** | -.670* | -.653* | -.720** | .541  |
| Zn             | .234                    | .937*** | -.204  | -.071  | -.314   | -.052 |
| Cu             | -.163                   | .630*   | -.365  | -.349  | -.620*  | -.117 |

Table 3. Concentrations of manganese, iron and zinc in plant tissues and in exchangeable fractions of the plant rooting medium. Utilization of the heavy metals is characterized by the discrimination factor values.

|                                 | Concentrations, mg g <sup>-1</sup> |      |       |      |       |      | Utilization of metals |       |        |
|---------------------------------|------------------------------------|------|-------|------|-------|------|-----------------------|-------|--------|
|                                 | Mn                                 |      | Fe    |      | Zn    |      | Fe/Zn                 | Mn/Zn | Mn/Fe  |
|                                 | plant                              | soil | plant | soil | plant | soil |                       |       |        |
| <i>Phragmites australis</i>     | 0.57                               | 0.24 | 0.24  | 19.6 | 0.03  | 0.51 | 0.21                  | 40.37 | 193.96 |
| <i>Schoenoplectus lacustris</i> | 1.38                               | 0.28 | 0.16  | 11.3 | 0.03  | 0.22 | 0.10                  | 36.14 | 348.08 |
| <i>Typha latifolia</i>          | 1.00                               | 0.23 | 0.12  | 17.3 | 0.05  | 0.20 | 0.03                  | 17.39 | 626.81 |
| <i>Equisetum fluviatile</i>     | 1.96                               | 0.45 | 0.71  | 16.9 | 0.07  | 0.29 | 0.17                  | 18.04 | 103.67 |

estuary does, indeed, include several metals, the levels of which markedly exceed the normal background values of Finnish waters (Isotalo 1979).

The processes of mineral nutrition are complex and several interrelated factors affect the enrichment of each individual element (Epstein 1972). Thus it is not surprising that the metal levels of plant tissues are weakly correlated with the corresponding metal concentrations of the plant substrata. In the present study the relationships between the heavy metal concentrations in the tissues of the six emergent species and the metal levels of the sediment layer were analysed together with the data on the acidity and organic matter fraction of the substrata.

Manganese was the only element showing a significant positive correlation between the plant tissues and sediments; the other three showed nonsignificant (Fe) or even negative (Zn, Cu) correlations (Table 2). The negative correlations of the metal levels with the pH values and soil organic matter were expected, because the mobility of metals is usually enhanced in acidic conditions (Lucas and Davis 1966).

The absence of positive correlations between the metal levels in plants and sediments can be associated with species-specific utilization and/or competitive interactions in the mineral nutrition processes. These specific features could be evaluated by calculating the discrimination factor (Df) values, i.e. the comparative element ratios for the

plant samples and for their substrata (cf. Aulio and Ojala 1983). For example, the Df values were calculated for the four dominant helophytes (Table 3). In spite of the fairly uniform metal concentrations in the plant tissues (Table 3), the accumulation capacity was shown to vary significantly in the taxa studied. Specific iron enrichment was apparent in *Equisetum fluviatile*, whereas *Typha latifolia* showed a tendency to accumulate zinc in preference to iron (Table 3).

The present material strongly suggests that the pondweeds *Potamogeton perfoliatus* and *P. obtusifolius* are suitable indicator organisms for heavy metal pollution monitoring. *P. perfoliatus* can be regarded as a selective accumulator of manganese, while *P. obtusifolius* shows a marked capacity for accumulating copper (Aulio and Salin 1982; cf. the review by Dykyjova 1979). Of the emergent species, *Butomus umbellatus* was the most promising candidate as a selective accumulator of iron (Table 1). Examination of the ability to utilize the metals available to the plants further demonstrated the selective accumulation of iron by *Equisetum fluviatile* and of zinc by *Typha latifolia* (Table 3). The floating-leaved taxa showed an 'intermediate' accumulation capacity (Table 1), thus appearing to be of limited value as monitoring organisms. Being very tolerant of water currents as well as metal pollution, yellow water lilies (*Nuphar lutea*) have proved to be especially suitable indicator organisms in lotic environments (Aulio 1980).

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# SPINAL DEFORMITIES AND CONCENTRATIONS OF HEAVY METALS AND CHLOROPHENOLS IN THE FOURHORN SCULPIN (*MYOXOCEPHALUS* *QUADRICORNIS* L.) OFF THE FINNISH COAST OF THE BOTHNIAN BAY

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BENGTTSSON, Å. & MIETTINEN, V. 1986. Spinal deformities and concentrations of heavy metals and chlorophenols in the fourhorn sculpin (*Myoxocephalus quadricornis* L.) off the Finnish coast of the Bothnian Bay. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

More than 2 000 fourhorn sculpins caught in seven areas off the Finnish coast in the Bothnian Bay were examined by X-ray for the occurrence of vertebral defects. The samples showed defect frequencies ranging from 8.1 % to 24.6 %. The two highest frequencies originated from the areas exposed to an increased load of heavy metals (Kokkola and Vaasa). The background level seemed to lie around 10 %.

The muscle and liver of fish with normal spines and with spinal defects were analysed for mercury, cadmium, lead, zinc, copper and chlorophenol. Mercury was the only metal differing significantly between the areas ( $p < 0.001$  for both muscle and liver). A correlation seems to exist between the mercury content in muscle and the occurrence of spinal defects ( $p = 0.06$ ). No muscle or liver sample contained chlorophenols over the detection limit of  $0.5 \mu\text{g g}^{-1}$ .

A connection between the defect frequencies and heavy metal concentrations in the fourhorn sculpin cannot be definitely established, as the sex ratio may contribute to the variation in the defect frequencies. To demonstrate the impact of pollution and identify the source, further studies are necessary including investigations of the mechanical and biochemical properties of the skeleton and the physiological condition of the fish.

Index words: Fourhorn sculpin, spinal deformities, heavy metals, chlorophenols, the Bothnian Bay

## 1. INTRODUCTION

The potential value of spinal defects in describing or measuring the impact of pollution on fish has been pointed out by both international conservation committees, e.g. ICES (McIntyre and Pearce 1980) and individual scientists (Bengtsson 1979). Observations of defects have actually been used by some authors in field studies (e.g. Wunder 1975, Sloof 1982).

Studies on the frequency of spinal deformities in fourhorn sculpins (*Myoxocephalus quadricornis* L.) have been conducted for some years in Sweden. The results have indicated correlations between elevated defect frequencies and discharges from the metal industry and pulp mills (Bengtsson and Bengtsson 1983a). During the studies it appeared that the "background" levels of defect frequencies were somewhat lower off the Finnish coast. To verify this and to obtain information on the

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occurrence of pollution sources causing raised levels of vertebral damage, this study was initiated by the National Board of Waters. The impact of pollution was studied by concurrent measurements of the concentrations of heavy metals and chlorophenols in fish.

## 2. MATERIAL AND METHODS

Fourhorn sculpins (*Myoxocephalus quadricornis* L.) were caught in seven areas along the Finnish coast of the Bothnian Bay during autumns 1981 and 1982 (Fig. 1). The aim was to obtain around 300 individuals per area. The sculpins were taken in gill nets, individually marked and packed before deep-freezing. Afterwards their sex, length and weight were determined and each fish was examined for vertebral defects by X-ray technique (Bengtsson and Bengtsson 1983b). Fish were also sampled according to the area and vertebral defects for chemical analysis of liver and muscle. Each analysed sample consisted of subsamples from three individuals. Heavy metals were determined by the methods of the National Board of Waters (1981) and chlorophenols by the method presented by Oikari et al. (1984).

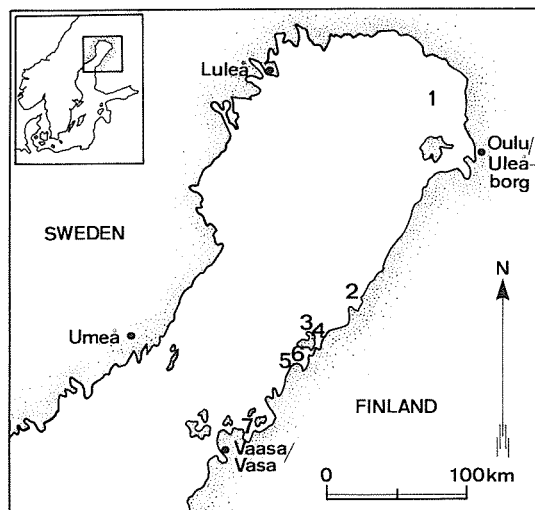


Fig. 1. Sampling areas for fourhorn sculpins.

## 3. RESULTS AND DISCUSSION

The proportion of fish showing severe spinal defects averaged 13.8 %. This is fairly low compared with data from the Swedish part of the Bay.

The frequency of defects was lowest in 1981 at Pietarsaari, Storgrundet, an area receiving effluents from a pulp mill, among other sources (Table 1). One year later, the frequency was somewhat higher, 14.7 %, in the nearby area Mässkär. The variation in the results does not seem to differ much from the 95 % confidence limits based on the binomial distribution. These results are interesting, as extremely high defect frequencies (up to 69 %) have been observed outside pulp mills in Sweden, although this phenomenon is not general.

Another area studied in both years, Kokkola, receives metals: amounts in tons in 1982 iron 771, zinc 48, cobalt 28, nickel 24 and copper 3.5 (National Board of Waters 1983), discharged from zinc, cobalt and artificial fertilizer plants. In 1981 the sample was collected at Tankar, situated about 15 km from the factories. It did not show any increase in the occurrence of fish with spinal defects (11.9 %). In the following year the sample was taken at Ykspihlaja, near the industrial area, and an elevated defect frequency of 24.6 % was found. This was the highest frequency observed in this study. The second highest frequency (16.8 %) was recorded at the mouth of the River Kyrönjoki. The river runs through Litorina alum soils and is rich in metals. In 1982 the mean concentrations were iron 2.2 mg l<sup>-1</sup>, aluminium 1.6 mg l<sup>-1</sup>, zinc 27 µg l<sup>-1</sup>, nickel 14 µg l<sup>-1</sup>, cobalt 6 µg l<sup>-1</sup> and copper 4 µg l<sup>-1</sup> (Matti Verta unpublished data). It is doubtful whether this is a real increase in the defects or simply background variation. The "background level" seems to lie around 10 % along this part of the Finnish coast, which is somewhat lower than the 16.8 % observed at the mouth of the River Kyrönjoki. The "background level" obtained along the Swedish coast is, however, around the Kyrönjoki value.

The concentrations of metals in fish liver and muscle are presented in Table 2. The analysis of variance and covariance (Classical ANOVA, Kim and Kohout in Nie et al. 1975) was used to test the influence of the locality and spinal condition on the metal content in the muscle and liver.

Mercury was the only metal showing a significant difference between the areas ( $p < 0.001$  for both liver and muscle). A correlation seems to exist between the mercury content in muscle and the occurrence of spinal defects ( $p = 0.06$ ). The copper concentration in

Table 1. Frequencies of vertebral damage (% fish with defects), sex ratios (no. males/no. females) and sizes of fourhorn sculpins.

| Code | Area                    | Year | Week  | n   | % with defects | 95 % C.I. | Sex ratio m/f | Mean length cm |
|------|-------------------------|------|-------|-----|----------------|-----------|---------------|----------------|
| A    | Ulkokrunni              | 1981 | 44    | 321 | 11.5           | 8—16      | 0.32          | 19.0           |
| B    | Kalajoki, Hevoskari     | 1981 | 39—44 | 341 | 12.0           | 8—16      | 0.41          | 19.0           |
| C    | Kokkola, Tankar         | 1981 | 42    | 344 | 11.9           | 8—16      | 0.41          | 19.5           |
| D    | Kokkola, Ykspihlaja     | 1982 | 45    | 284 | 24.6           | 19—30     | 0.72          | 18.9           |
| E    | Pietarsaari, Storgundet | 1981 | 41—42 | 370 | 8.1            | 5—12      | 0.52          | 18.6           |
| F    | Pietarsaari, Mäskär     | 1982 | 44    | 102 | 14.7           | 8—23      | 0.48          | 19.6           |
| G    | Vaasa, Kyrö River mouth | 1981 | 45—48 | 291 | 16.8           | 12—22     | 0.60          | 19.9           |

Table 2. Concentrations of mercury, cadmium, lead, zinc and copper (mg kg<sup>-1</sup> fresh weight) in muscle and liver of fourhorn sculpins without (ND) and with (D) vertebral damage.

| Code | Locality                | Hg   |      | Cd    |       | Pb   |      | Zn   |      | Cu   |      | Length cm |
|------|-------------------------|------|------|-------|-------|------|------|------|------|------|------|-----------|
|      |                         | ND   | D    | ND    | D     | ND   | D    | ND   | D    | ND   | D    | ND/D      |
| A    | Ulkokrunni              |      |      |       |       |      |      |      |      |      |      |           |
|      | muscle                  | 0.51 | 0.74 | 0.005 | 0.005 | 0.05 | 0.05 | 4.64 | 8.58 | 0.46 | 0.35 | 21.7/21.5 |
| B    | Kalajoki, Hevoskari     | 0.50 | 0.58 | 0.19  | 0.24  | 0.10 | 0.10 | 56.3 | 61.3 | 5.49 | 10.4 |           |
|      | muscle                  | 0.11 | 0.18 | 0.018 | 0.006 | 0.05 | 0.05 | 5.71 | 5.86 | 0.46 | 0.39 | 18.4/20.2 |
| C    | Kokkola, Tankar         | 0.19 | 0.10 | 0.11  | 0.11  | 0.10 | 0.10 | 64.0 | 74.6 | 5.54 | 4.97 |           |
|      | muscle                  | 0.13 | 0.24 | 0.014 | 0.005 | 0.05 | 0.05 | 5.82 | 7.20 | 0.29 | 0.46 | 20.0/22.2 |
| D    | Kokkola, Ykspihlaja     | 0.11 | 0.15 | 0.27  | —     | 0.10 | 0.10 | 65.3 | 73.0 | 6.14 | 8.43 |           |
|      | muscle                  | 0.17 | 0.17 | 0.037 | 0.010 | 0.01 | 0.01 | 9.50 | 10.6 | 0.58 | 0.60 | 20.7/21.2 |
| E    | Pietarsaari, Storgundet | 0.14 | 0.15 | 0.25  | 0.16  | 0.05 | 0.02 | 40.9 | 25.8 | 1.54 | 2.38 |           |
|      | muscle                  | 0.14 | 0.18 | 0.005 | 0.005 | 0.05 | 0.05 | 5.17 | 6.33 | 0.48 | 0.46 | 19.7/21.7 |
| F    | Pietarsaari, Mäskär     | 0.10 | 0.14 | 0.21  | 0.20  | 0.10 | 0.10 | 77.4 | 90.0 | 6.07 | 4.50 |           |
|      | muscle                  | 0.08 | 0.09 | 0.006 | 0.015 | 0.01 | 0.01 | 8.90 | 9.90 | 0.47 | 0.90 | 20.5/20.6 |
| G    | Vaasa, Kyrö River mouth | 0.05 | 0.07 | 0.12  | 0.21  | 0.03 | 0.04 | 24.3 | 43.2 | 2.14 | 2.52 |           |
|      | muscle                  | 0.26 | 0.29 | 0.005 | 0.008 | 0.05 | 0.05 | 9.42 | 5.08 | 0.61 | 0.34 | 22.2/19.7 |
|      | liver                   | 0.19 | 0.17 | 0.30  | 0.16  | 0.10 | 0.10 | 95.4 | 43.0 | 10.7 | 6.88 |           |

liver showed a weak correlation with the locality ( $p = 0.07$ ), but this did not exceed the limit of significance ( $p = 0.05$ ). The differences in the mercury content seem mainly to be caused by the high levels at Ulkokrunni. This area was considered to be less polluted than the others but the mercury concentrations in fish muscle reveal the effect of industrial activities, probably those in Oulu. The levels of mercury are comparable to those obtained by Nuorteva and Häsänen (1975) in sculpins caught in the polluted water area off Pori.

The content of zinc in sculpins caught off the River Kyrönjoki seemed to differ when spinal defects existed, but due to the great variation in the data on zinc the difference is doubtful. It is somewhat surprising that the fish in that area do not show higher levels of zinc than in the other areas, since the

levels of that metal in the river water are known to be elevated.

The levels of chlorophenols in the fish liver were in all cases below the detection limit of  $0.5 \mu\text{g g}^{-1}$  fresh weight.

The lack of significant differences in metal content between fish showing spinal defects and those without defects is not surprising, although metals may be the factor causing the defect. The spinal defect may be the cost the fish have to pay for detoxification of the metals or lowering of the metal content in the tissues. One major problem with studies of the frequency of vertebral damage is that the history of the defect is not known. The defect may have been caused by pollutants long ago and the fish may have migrated since then, which will cause errors in the evaluation of the results.

Other studies made along the Swedish coast suggest that sex and size may influence the defect frequency obtained. Multiple linear regression analysis (Ryan et al. 1976) was used to test the influence of these factors. The sex ratio (no. of males/no. of females) proved to be correlated ( $p < 0.05$ ) with the defect frequencies. This may be a factor causing, or contributing to the variation in defects. But it does not explain why defects exist or what factors cause the defects.

Although major doubts exist, the results presented here give an indication of the pollution sources that raise the frequency of vertebral damage in fish, especially fourhorn sculpins. They cannot be used to prove any impact or to identify the source that causes the impact. To solve this problem more intensified studies should be made that take account of the frequencies of vertebral damage, the mechanical and biochemical properties of the skeleton and physiological condition of the fish.

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## THE USE OF LIPOPHILICITY IN STUDIES OF THE DISTRIBUTION PATHWAYS OF CHLORINATED ORGANICS FROM A PULP MILL

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FOGELQVIST, E., ABRAHAMSSON, K., JOSEFSSON, B. & XIE, T.M. 1986. The use of lipophilicity in studies of the distribution pathways of chlorinated organics from a pulp mill. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The distributions of some chlorinated phenolic compounds discharged from a sulphate pulp mill were studied. The compounds were measured both in the water column and in the bulk sediment. In water, the concentrations were compared with the concentration of chloroform, which was used as a tracer to delineate and monitor the dilution of the effluent plume. This comparison, together with a correlation with the lipophilicity of the chlorophenolics, supports some conclusions regarding the pathways and the accumulation of these compounds in the sea. The lipophilicity was measured by the partition coefficients in the octanol-water system and corrected for pH. The results show that the transportation of chlorophenolics was dominated by dilution and adsorption processes. In organic-rich sediments the fate of chloroguaiacols and chlorocatechols was observed to be strongly influenced by bioactivity.

Index words: Chlorinated phenolic compounds, chloroform, pulp bleachery effluent, water analysis, marine environment, sediment, octanol-water partition coefficient, capillary gas chromatography

### 1. INTRODUCTION

The fate of organic pollutants in aquatic environments can depend on: dilution, adsorption on to particles and sedimentation, chemical and photochemical decomposition, volatilization, biodegradation and bioaccumulation. The adsorption of organic pollutants on to particles is strongly related to their lipophilicity, described by their log P values — the logarithm of the partition coefficient in a octanol-water system (Hansch

1979). This parameter has been used in environmental studies in a river (Lopez-Avila and Hites 1980). The lipophilicity parameters of chlorophenolics (Xie et al. 1984) offer a useful means of studying the marine biogeochemistry of these types of compounds, as do also the simple and sensitive GC methods presented recently for determining trace amounts of chlorophenolics in water and sediment, (ng l<sup>-1</sup> and ng g<sup>-1</sup>, respectively) (Abrahamsson and Xie 1983).

An extended version of this article, including all data, will be published elsewhere (Xie et al., 1986).

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## 2. MATERIAL AND METHODS

The sampling was performed on five occasions during 1982 and 1983. Details of the sampling locations and dates are given in Fig. 1. Chlorinated phenolics and chloroform in the water samples were determined according to the procedures described by Abrahamsson and Xie (1983) and Fogelqvist et al. (1982), respectively. The three-phase extraction acetylation method for the determination of chlorinated phenolics in sediment samples was employed according to Xie (1983). The water content in the sediment was determined by heating at 110 °C to constant weight. The organic content was estimated by the loss of weight on ignition at 475 °C for 5 hours. Before analysis the sediment samples were stored at a temperature of -20 °C.

## 3. RESULTS AND DISCUSSION

The plume of the bleaching effluent from the pulp mill was investigated using chloroform as a tracer. On four occasions, between July 1982 and November 1983, the distribution patterns were determined, one of them being presented in Fig. 1. The isopleth map is based on the sea water samples collected in November 1983 at 30 different stations. Fig. 1 shows that the north-east effluent flow gyrates southward, due to the main southward current.

### 3.1 Chlorophenolics in water

The dilution factors of the chlorophenolics differ from each other, indicating that other factors also play an important role. The high lipophilicities of chlorophenolics imply that they can be adsorbed on to organic particles suspended in the water. Since the phenolics are weak acids, their adsorption potentials are not only dependent on their lipophilicities but are also strongly influenced by the pH of the ambient water. Table 1 lists the half-distances of some chlorophenolics together with their acidities ( $pK_a$ ),  $\log P$  and  $\log P_a$ .  $P_a$  is the apparent partition coefficient, which takes account of the phenolate ions at pH 7. The half-distance is the distance from the first sampling station near the outlet to the point where the concentration of an individual chlorophenolic is reduced by half.

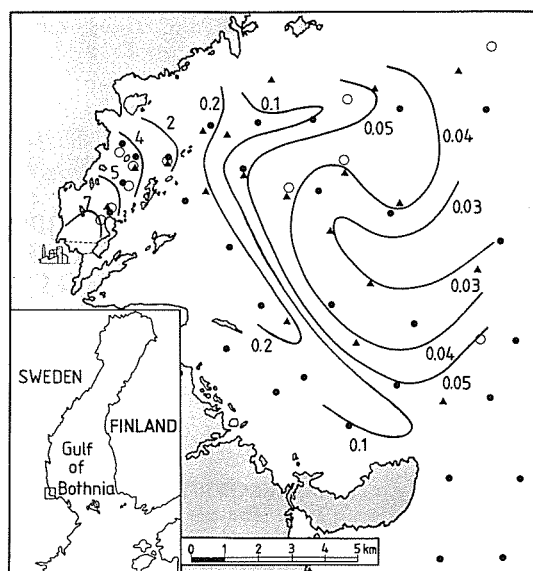


Fig. 1. Sampling area and sampling stations.

▲ water samples from September 1982.

● water samples from November 1983.

○ sediment samples.

The isopleths describe the concentration of chloroform (in  $\mu\text{g l}^{-1}$ ) in the 1983 investigation.

Table 1. The  $pK_a$ ,  $\log P$ ,  $\log P_a$  and half-distance values of some chlorophenolics.

| Compound                              | $pK_a^a$ | $\log P^b$ | $\log P_a^c$      | half-dist.<br>(km) |
|---------------------------------------|----------|------------|-------------------|--------------------|
| Tetrachloro-<br>catechol (TeCC)       |          | 4.31       | 2.70 <sup>c</sup> | 1.1                |
| 2,4,6-trichloro-<br>phenol (TrCP)     | 6.21     | 3.75       | 2.89              | 1.0                |
| Pentachloro-<br>phenol (PCP)          | 4.90     | 5.04       | 2.94              | 0.9                |
| 2,3,4,6-tetrachloro-<br>phenol (TeCP) | 5.62     | 4.42       | 3.02              | 0.8                |
| Tetrachloro-<br>guaiacol (TeCG)       | 6.26     | 4.45       | 3.64              | 0.7                |
| 3,4,5-trichloro-<br>guaiacol (TrCG)   | 7.56     | 4.11       | 4.00              | 0.5                |
| Chloroform, $\text{CHCl}_3$           |          | 1.97       |                   | 1.9                |

<sup>a</sup>) data from ref. 3.

<sup>b</sup>) data from ref. 1.

<sup>c</sup>) calculated from eq.  $\log P_a = \log P - \log (1 + K_a \cdot 10^{\text{pH}})$  using the previous data, considering  $\text{pH} = 7$ .

<sup>d</sup>) measured from the concentration-distance curves of chlorophenols using the concentration at the station nearest to the outlet as the initial concentration.

<sup>e</sup>) directly determined by shake-flask GC method at  $\text{pH} = 7$  (Xie et al. 1984).

These data show that the half-distances of the chlorophenolics are not related to their log  $P$  values, but to their log  $P_a$  values. The half-distances decrease as the log  $P_a$  values increase, because the effluent is rich in large organic particles, mainly fibres, which adsorb chlorophenolics and sink rapidly to the bottom. This interpretation is supported by the high concentrations of chlorophenolics in the sediments near the source.

### 3.2 Chlorophenolics in the sediment

The concentrations of the chlorophenolics in the sediments are dependent on the abundance and nature of the organic matter. In the sediments from transportation bottoms, the organic content and composition vary with time. The accumulation bottoms are more stable and have a more constant organic content.

The ratio between the log  $P_a$  values of the chlorophenolics and their half-distances in the water phase did not show a significant correlation with the sediment data. It could be expected that the most lipophilic chlorophenolics with the shortest half-distances would be accumulated to a greater extent in the sediments. The concentration ratios in the sediment and water were compared. These ratios were then normalized against one of the most conservative chlorophenolics due to the lack of chloroform data for the sediments. Only occasionally, in sediments from transportation bottoms, could any trend be noted.

In transportation bottoms the concentrations of TeCG and 3,4,5-TrCG in the top layer are relatively high, which indicates that the sediments were deposited recently. This is confirmed by the fact that the organic matter mainly consists of fibres. At accumulation bottoms the ratios between the sediment and water concentrations showed no correlation with the log  $P_a$  values of the different chlorophenolics. These sediments contain about 40 % organic matter, but no fibres were observed.

The conclusion is that other factors have an impact on the chlorophenolic concentrations in sediment, such as chemical or biological processes. Due to accumulation rates on the order of centimetres/month, the sediment samples must be regarded as being integrated over a long time period, while the water samples reflect a more instantaneous situation.

The bioactivity in accumulation bottoms results

in a rapid decline of the 2,4,6-TrCP, TeCG and 3,4,5-TrCG contents. The chlorophenolic concentration ratios between sediment and water are only about 200 for these compounds. In contrast, for TeCC this ratio is 10 000, which is unexpectedly high. Possible transformation of tetrachloroguaiacol to tetrachlorocatechol in fish was observed by Landner et al. (1977). Some phenomena can also be explained by bacterial impact, as was recently demonstrated by Neilson et al. (1983), who showed that the chloroguaiacols were transformed to chloroveratrols by bacterial methylation. The high concentration of the chlorocatechols may also be the result of bacterial degradation of high-molecular-weight chlorinated lignin. The bacterial impact is further emphasized by the rapid decline of the chlorophenolic concentrations deeper in the sediments.

### 4. CONCLUSIONS

Chloroform was used as a measure of the dilution of the effluent water from a bleachery. The deviations of chlorinated phenolics from the observed dilution pattern indicate the importance of factors having an impact on the biogeochemistry of the chlorophenolics in the aquatic environment.

The lipophilicities of the chlorophenolics, described by their partition coefficients in the octanol-water system, are very helpful in studies of their fates in the aquatic environment. However, the influence of the pH of the ambient water must be taken into account, because of the acid-base character of these compounds. Biotransformation and biodegradation of chlorophenolics in bioactive sediment play an important role and need more investigation.

### ACKNOWLEDGEMENTS

We want to thank Mr U. Karlström for great help in these investigations. Chlorinated guaiacols and catechols were kindly supplied by Mr J.S. Knuuti-nen. This work was financed by the National Swedish Environment Protection Board, the Chinese Education Ministry and the Ministry of Light Industry of China.

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## MACROALGAE AS INDICATORS OF HEAVY METAL POLLUTION

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AUTIO, H. & KANGAS, P. 1986. Macroalgae as indicators of heavy metal pollution. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The concentration of copper, zinc, iron, lead and cadmium were measured on the brown alga *Fucus vesiculosus* and three annual filamentous algae in the Tvärminne area, northern Baltic Sea. Accumulation of heavy metals by the algae, and the seasonal and spatial variation in the metal levels of algae are discussed, and recommendations are given regarding the use of algae in monitoring heavy metal pollution.

Index words: Zinc, copper, iron, lead, cadmium, *Fucus vesiculosus*, filamentous algae, northern Baltic Sea

### 1. INTRODUCTION

The concentrations of heavy metals in sea water are rather low and irregular water movements make it difficult to interpret the results. In algae the concentrations are much higher than in water and may indicate the conditions prevailing for some length of time. According to several authors, macroalgae can give an indication of the heavy metals present in a sea area (e.g. Preston et al. 1972, Bryan and Hummerstone 1973, Melhuus et al. 1978).

In Öresund, southwestern Baltic Sea, heavy metals have been determined on a number of algal species (Hägerhäll 1973), including *Fucus vesiculosus* (Phillips 1977). Off the Finnish coast some common filamentous algae and *Fucus*, which are frequent and abundant in southern and southwestern archipelagoes, may be suitable as bioindicators of heavy metal pollution.

The choice of algae as indicators of heavy metals is complicated by lack of information on the

accumulation of metals by these organisms. Data are needed, for example, on the extent to which accumulation varies between species, seasons, different parts of the algae or algae of different ages. The uptake mechanisms may also be affected by the environmental conditions and competition between different metals.

The aim of this study was to investigate the usefulness of macroalgae as indicators of heavy metal pollution and to develop and test a suitable method of determining heavy metal concentrations in algae as well as to map the metal concentrations in algal tissues in the Tvärminne area.

### 2. MATERIAL AND METHODS

Four algal species were chosen: the perennial *Fucus vesiculosus* (L.), and three annual filamentous algae *Cladophora glomerata* (L.) Kütz., *Pilayella littoralis*

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(L.) and *Ceramium tenuicorne* (Kütz), all of which were abundant in the study area. The algal samples were collected from the Tvärminne area in 1979–1981 and water samples were taken in 1981 (Fig. 2, p. 186). *Fucus*, *Ceramium* and *Pilayella* were sampled by diving, *Cladophora* was taken from rocks just below the water surface. The samples were collected in a plastic bag. In the laboratory the epiphytic plants and animals were removed from the samples, being washed off the filamentous algae with sea water. Pieces were cut from the *Fucus* fronds and stipes for analysis. All the algal samples were dried at +90 °C for 24 hours and then ground to fine powder with an agathe mill.

Problems have been encountered in determining metals on algae by atomic absorption spectrophotometry (AAS). Algal tissue, especially *Fucus*, is not easily disintegrated and other ions contained by the algae disturb the analyses.

Dry ashing (+500 °C), wet digestion with  $\text{HNO}_3/\text{H}_2\text{SO}_4$  and wet digestion with  $\text{HNO}_3/\text{HCl}$  were applied to *Fucus*. The following procedure proved most suitable for this alga: 0.5 grams of dried algal powder was autoclaved in an ordinary pressure cooker with 10 ml  $\text{HNO}_3$  (30 min. at 2 atm., approx. 120 °C). After cooling to room temperature, 2.5 ml  $\text{HCl}$  was added, heated slowly and evaporated. The digesta were diluted to 25 ml. Iron and zinc were determined by flame (air-acetylene) AAS (Perkin-Elmer 305), and copper with a tube atomizer (HGA 74).

The determinations of lead and cadmium proved to be unreliable when only wet digestion was used. The metals must be separated from the disturbing matrix. Lead and cadmium were determined at the Technical Research Center of Finland with flame AAS after wet digestion and chelation with ammonium pyrrolidine dithiocarbonate (APDC).

The results of intercalibration carried out by the Institute of Botany of Aarhus University indicate that heavy metals, with the exception of small lead concentrations, can be determined reliably by the method used in this study (see Brix et al. 1983, laboratory no. 11). Algal tissue, however, is more difficult to analyse than the vascular plant *Zostera marina* used in the intercalibration. The lead and cadmium analyses, in particular, proved to be unreliable unless the metals were separated from the disturbing matrix. Therefore in this study lead and cadmium were determined after wet digestion and chelation with APDC.

The water samples were analysed for heavy metals according to Koroleff (1980) in the Research Laboratory of the National Board of Waters.

### 3. RESULTS AND DISCUSSION

The metal levels were determined in the three species of filamentous algae and *Fucus vesiculosus*. In the *Fucus* analyses attention was also paid to spatial and seasonal variation and to the accumulation on metals in different parts of the thallus. The results are presented as  $\mu\text{g g}^{-1}$  algal dry weight.

#### 3.1 Heavy metals in filamentous algae

The zinc concentration was lower in *Cladophora* than in *Ceramium* and *Pilayella* (Table 1), and lower in the annual filamentous algae than in *Fucus*. This may be due to their different chemical composition. Copper was at about the same level in all the annual species but was lower than in *Fucus*.

Iron varied most widely between the species studied and also seasonally. In *Cladophora* and *Pilayella* the level even exceeded  $5\,000\ \mu\text{g g}^{-1}$ , while in *Ceramium* the highest value was  $1\,500\ \mu\text{g g}^{-1}$ . The highest iron levels were measured at station 3, where the algae were covered by a dense growth of epiphytes.

The lead concentrations of *Cladophora* and *Ceramium* were higher at inner than at outer stations (Table 2). All the cadmium concentrations were lower than  $2\ \mu\text{g g}^{-1}$ .

The rate of growth affects the chemical composition of annual algae and thus also the recorded metal uptake. When the growth is vigorous, the concentrations do not increase rapidly (Rice & Lapointe 1981). The filamentous algae thus cannot be regarded as good heavy metal indicators, because it is difficult to distinguish differences caused by different exposure times (age of alga) from differences due to real variation in the metal concentrations of sea water.

#### 3.2 Heavy metals in *Fucus*

##### 3.2.1 Variations in thallus

The concentration of zinc consistently increased from the tops to the stipes of *Fucus* ( $167\ \mu\text{g g}^{-1}$  to  $343\ \mu\text{g g}^{-1}$ ) (Fig. 1). Copper behaved in the same manner (increase from  $6.8$  to  $9.8\ \mu\text{g g}^{-1}$ ), but the concentration of iron varied irregularly.

The high metal concentration in *Fucus* stipes is due to slow and irreversible accumulation (Bryan

Table 1. Concentrations of zinc, copper and iron in filamentous algae ( $\mu\text{g g}^{-1}$  dry wt) in Tvärminne in 1980. N is 3 if not given in parentheses.

| Station no.      | Date    | Cladophora glomerata |         |            |
|------------------|---------|----------------------|---------|------------|
|                  |         | Zn                   | Cu      | Fe         |
| 6R <sup>1)</sup> | 11 June | 58±0.4               | 18±1    | 313± 6     |
|                  | 18 July | 80±2                 | 15±0.2  | 2175± 34   |
|                  | 9 Sept  | 75±3                 | 12±0.1  | 611± 7     |
| 3                | 21 July | 102±2                | 25±3    | 5541±338   |
| 8                | 6 Sept  | 44±1                 | 15±0.3  | 200± 4     |
|                  |         | Ceramium tenuicorne  |         |            |
|                  |         | Zn                   | Cu      | Fe         |
| 6R               | 11 June | 152±1(4)             | 24±1(4) | 1068± 8(4) |
|                  | 18 July | 179±2                | 22±0(2) | 528±36(2)  |
|                  | 9 Sept  | 97±2                 | 20±1    | 1535±15    |
| 8                | 6 Sept  | 115±3                | 17±1(2) | 215± 8     |
|                  |         | Pilayella littoralis |         |            |
|                  |         | Zn                   | Cu      | Fe         |
| 6R               | 11 June | 153± 3               | 19±1    | 790± 34    |
|                  | 18 July | 189±15               | 24±1    | 5055±193   |
|                  | 21 July | 146± 1               | 20(2)   | 5367±443   |

<sup>1)</sup> Station 6R is situated at Kvarnskärsgrundet 500 m W of St. 6.

Table 2. Concentrations of lead and cadmium in filamentous algae ( $\mu\text{g g}^{-1}$  dry wt) in Tvärminne in 1980.

| Station | Date    | Pb         |          |           |
|---------|---------|------------|----------|-----------|
|         |         | Cladophora | Ceramium | Pilayella |
| 3       | 21 July | 9.9        | ..       | 3.7       |
| 6R      | 9 Sept  | 6.9        | 9.9      | ..        |
| 8       | 6 Sept  | 1.5        | 2.2      | ..        |
|         |         | Cd         |          |           |
|         |         | Cladophora | Ceramium | Pilayella |
| 3       | 21 July | 0.2        | ..       | 1.6       |
| 6R      | 9 Sept  | 0.7        | 0.2      | ..        |
| 8       | 6 Sept  | 1.4        | 0.6      | ..        |

<sup>1)</sup> see Table 1.

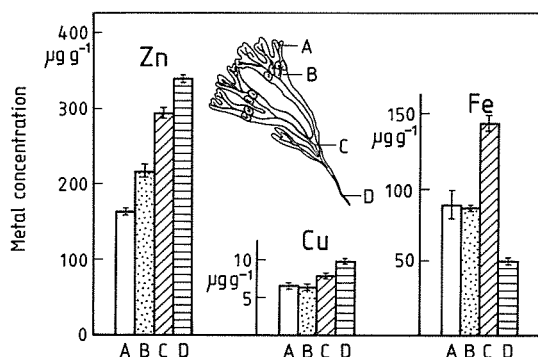


Fig. 1. Concentrations of zinc, copper and iron in different parts of the *Fucus* thallus. Individuals sampled at Långskär (station 7) in July 1979.

and Hummerstone 1973, Bohn 1979, Tomlinson et al. 1980). Metals are not transferred along the thallus from older parts to younger (Strömberg 1979).

The irregular distribution of iron is difficult to explain. The thallus surface may be contaminated by unknown substances (Bryan and Hummerstone 1973). The mucus covering the thallus may also contain iron, which means that not all the iron measured was bound to the algal tissues (Romeril 1977).

If *Fucus* is used as an indicator of the metal levels in water, the parts of the thallus sampled must always be of the same age. Samples from the tops will give the accumulation in the last season and samples from the stipes the accumulation of several years. If whole individuals are analysed, the sampling must be done at a fixed time, so that the ratio of young to old tissues is the same every year.

### 3.22 Spatial variation

At the innermost stations (1—4) the concentrations are higher than at the outer ones (6—8). Discharge of substances from the land and inland waters, both minerogenic and anthropogenic, is reflected as elevated concentrations of metals in *Fucus*.

In 1979—81 the copper concentration of *Fucus* varied from 6 to 16  $\mu\text{g g}^{-1}$  in the fronds and from 7 to 33  $\mu\text{g g}^{-1}$  in the stipes (Fig. 2, Table 3). The concentrations are of the same order of magnitude as those measured in Öresund (Hägerhäll 1973). In Cornwall, *Fucus* sp. in waters heavily loaded by municipal sewage was observed to have copper concentrations as high as 2750  $\mu\text{g g}^{-1}$  (Romeril 1977).

The load of zinc discharged from the iron and steel plant at Koverhar is 13.2 tons a year and is the greatest of the loads of heavy metals imposed on the area (Luotamo and Luotamo 1977). The zinc levels in *Fucus* were highest at station 5 near the outfall: algal fronds 1 133—1 372  $\mu\text{g g}^{-1}$  and stipes 1 190—2 672  $\mu\text{g g}^{-1}$  (Fig. 3, Table 3). At the other stations the levels were notably lower. This agrees with earlier reports that algae actively accumulate zinc and thus indicate the concentrations (Bryan and Hummerstone 1973, Romeril 1977, Bohn 1979).

The iron concentration in the *Fucus* thallus varied greatly, 25—1 082  $\mu\text{g g}^{-1}$  (Fig. 4, Table 3). The spatial distribution of iron resembled that of zinc, but the concentrations were also high east of Koverhar (stations 9—11). Natural waters may contain large amounts of iron and the rate of

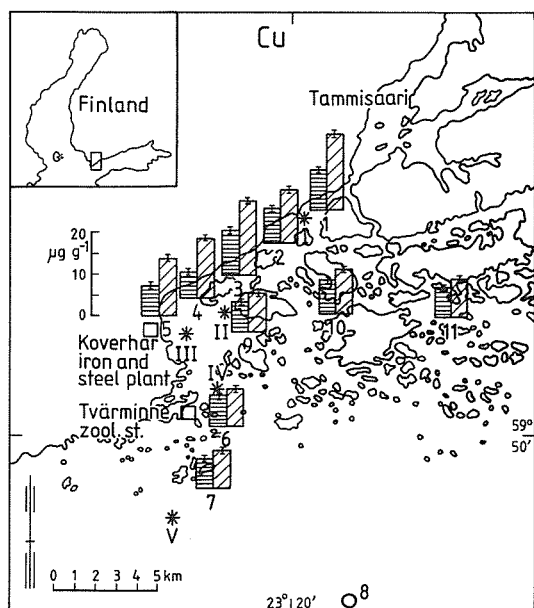


Fig. 2. Concentrations of copper in *Fucus* ( $\mu\text{g g}^{-1}$  dry wt) in the Tvärminne area in September 1979. Sampling stations marked with arabic numerals. Stars with roman numerals indicate water sampling stations.

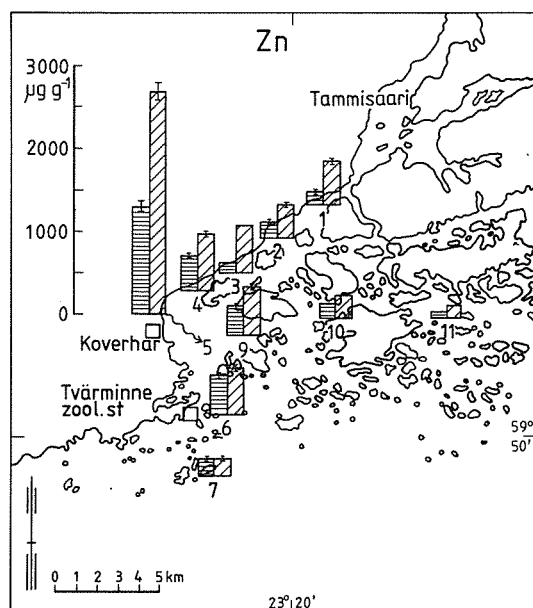


Fig. 3. Concentrations of zinc in *Fucus* ( $\mu\text{g g}^{-1}$  dry wt) in the Tvärminne area in September 1979.

accumulation in algae is higher in areas where the contents of other metals are low. The metal ions compete for binding sites, which are limited in algae. According to the literature, the iron in algae always varies greatly and irregularly, so that *Fucus* is useless as an indicator of the iron content of water (see also subsection 3.21).

The lead concentration in *Fucus* near the discharge point was 2.8 and 19  $\mu\text{g g}^{-1}$  (Table 4). At

other stations it was below the detection limit. Accumulation of cadmium by *Fucus* was slight (Table 4). In waters with high zinc concentrations, the zinc ions may compete with cadmium and thus inhibit cadmium adsorption (Bryan and Gibbs 1983). In a Norwegian fjord, however, in waters polluted by zinc, *Fucus vesiculosus* contained large amounts of both zinc and cadmium (Melhuus et al. 1978).

Table 3. Concentrations of zinc, copper and iron in *Fucus vesiculosus* ( $\mu\text{g g}^{-1}$  dry wt) in the Tvärminne area in September 1980–81. N is 3 if not given in parentheses.

| Station | Zn             |                | Cu           |              | Fe             |                 |
|---------|----------------|----------------|--------------|--------------|----------------|-----------------|
|         | Frond          | Stipes         | Frond        | Stipes       | Frond          | Stipes          |
| 1980    |                |                |              |              |                |                 |
| 3       | 160 $\pm$ 2    | 477 $\pm$ 3    | 8 $\pm$ 0.2  | 33 $\pm$ 1.2 | 105 $\pm$ 10   | 94 $\pm$ 2      |
| 7       | 160 $\pm$ 2    | 238 $\pm$ 3    | 6 $\pm$ 0.1  | 10 (1)       | 87 $\pm$ 4     | 37 $\pm$ 2      |
| 8       | 107 $\pm$ 4    | 180 $\pm$ 6    | 7 $\pm$ 0.5  | 9 $\pm$ 1.1  | 115 $\pm$ 9    | 25 $\pm$ 3      |
| 11      | 57 $\pm$ 0.2   | 148 $\pm$ 1    | 8 $\pm$ 0.1  | 11 $\pm$ 0.1 | 206 $\pm$ 5    | 87 $\pm$ 3      |
| 1981    |                |                |              |              |                |                 |
| 1       | 375 $\pm$ 30   | 457 $\pm$ 13   | 16 $\pm$ 0.5 | 16 $\pm$ 0.7 | 1 082 $\pm$ 53 | 308 $\pm$ 52    |
| 3       | 304 $\pm$ 2    | 432 $\pm$ 6    | 13 $\pm$ 0.5 | 15 $\pm$ 0.3 | 262 $\pm$ 4    | 379 $\pm$ 94    |
| 5       | 1 133 $\pm$ 38 | 1 190 $\pm$ 40 | 7 $\pm$ 0.8  | 19 $\pm$ 0.5 | 255 $\pm$ 35   | 1 401 $\pm$ 153 |
| 6       | 496 $\pm$ 6    | 321 $\pm$ 7    | 6 $\pm$ 0.3  | 9 $\pm$ 0.1  | 79 $\pm$ 2     | 56 $\pm$ 3      |
| 7       | 337 $\pm$ 4    | 248 $\pm$ 12   | 8 $\pm$ 0.3  | 12 $\pm$ 0.2 | 116 $\pm$ 1    | 39 $\pm$ 6      |

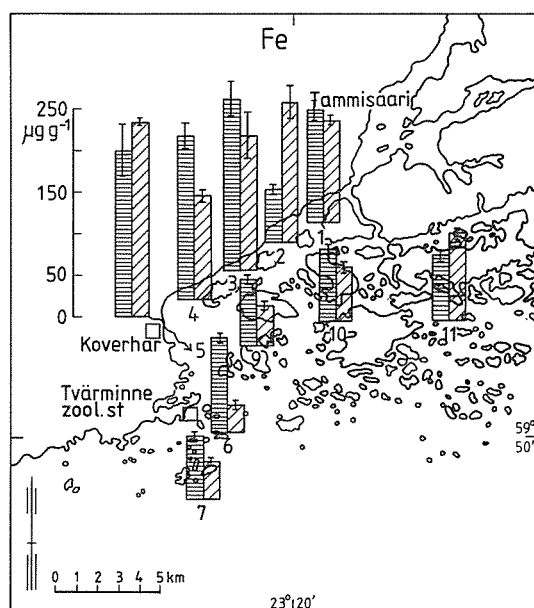


Fig. 4. Concentrations of iron in *Fucus* ( $\mu\text{g g}^{-1}$  dry wt) in the Tvärminne area in September 1979.

Table 4. Concentrations of lead and cadmium in *Fucus vesiculosus* ( $\mu\text{g g}^{-1}$  dry wt) in the Tvärminne area in 1980.

| Station | Pb     |        | Cd     |        |
|---------|--------|--------|--------|--------|
|         | Fronds | Stipes | Fronds | Stipes |
| 3       | <0.05  | <0.05  | 1.3    | 3.8    |
| 5       | 2.8    | 19.0   | 3.0    | 2.7    |
| 7       | <0.05  | <0.05  | 3.6    | 5.6    |

Table 5. Seasonal variation of zinc, copper and iron in *Fucus vesiculosus* ( $\mu\text{g g}^{-1}$  dry wt) in Tvärminne in 1980. N is 3 if not given in parentheses.

| Station | Date    | Zn    |        | Cu       |            | Fe    |        |
|---------|---------|-------|--------|----------|------------|-------|--------|
|         |         | Frond | Stipes | Frond    | Stipes     | Frond | Stipes |
| 7       | 5 June  | 278±7 | 291±11 | 7±0.2    | 10±0.3     | 125±7 | 75±5   |
|         | 15 July | 305±2 | 297±3  | 8±0.1    | 11±0.01(2) | 90±6  | 138±6  |
|         | 7 Sept  | 160±2 | 238±3  | 6±0.1(2) | 10 (1)     | 87±4  | 37±2   |
| 8       | 20 July | 161±2 | 166±8  | 8±0.2    | 9±0.1      | 42±1  | 43±1   |
|         | 6 Sept  | 107±4 | 180±6  | 7±0.5    | 9±1.1      | 115±8 | 25±3   |

Table 6. Concentrations of zinc, copper, iron, lead and cadmium in surface water ( $\mu\text{g l}^{-1}$ ) in the Tvärminne area in 1981. Stations marked in Fig. 2.

| Station | Zn        |          | Cu        |         | Fe        |       | Pb        |         | Cd         |
|---------|-----------|----------|-----------|---------|-----------|-------|-----------|---------|------------|
|         | $\bar{x}$ | range    | $\bar{x}$ | range   | $\bar{x}$ | range | $\bar{x}$ | range   |            |
| I       | 2.6       | 2.0—3.2  | 1.2       | 0.9—1.8 | 40        | 33—48 | 1.3       | 1.2—1.3 | <0.05      |
| II      | 2.3       | 1.6—3.2  | 1.0       | 0.9—1.1 | 33        | 29—39 | 1.3       | 1.2—1.4 | <0.05      |
| III     | 4.0       | 1.5—9.0  | 0.9       | 0.9—1.2 | 28        | 23—42 | 2.1       | 1.6—3.2 | <0.05—0.10 |
| IV      | 4.1       | 1.1—21.0 | 0.9       | 0.8—1.1 | 25        | 19—37 | 2.3       | 1.6—4.2 | <0.05—0.13 |
| V       | 0.8       | 0.6—1.0  | 0.8       | 0.8—0.8 | 7         | 6—9   | 1.1       | 1.0—1.3 | <0.05      |

### 3.23 Seasonal variation

The zinc concentration of *Fucus* was highest in mid summer and lowest in autumn, in both the tops and stipes (Table 5). This agrees with the observations of Fuge and James (1974) in the Bristol Channel. The copper concentration of *Fucus* did not vary remarkably during the growing season. The variation of iron was greater than that of zinc, but it was irregular. Bryan and Hummerstone (1973) did not find any annual variation for these three metals in *Fucus vesiculosus* or *F. serratus*.

The proportion of different algal parts varies as the alga grows. Although no clear seasonal variation was found in this study, for monitoring purposes it is safest to take samples at a fixed time of the year.

### 3.24 Metal accumulation

The metal concentrations in water sampled in July—September 1981 (Fig. 2) are presented in Table 6, the zinc concentration was highest (ca.  $4 \mu\text{g l}^{-1}$ ) at stations III and IV near Koverhar. The copper concentration was always about  $1 \mu\text{g l}^{-1}$ . The iron concentration was highest ( $40 \mu\text{g l}^{-1}$ ) at the innermost station and lowest ( $7 \mu\text{g l}^{-1}$ ) in the open sea. The lead and cadmium concentrations were highest at Storfjärden (IV).

Much clearer spatial variation was evident in the heavy metal concentrations determined on *Fucus* than in the concentrations in water. Accumulation of zinc by the alga is clearest near Koverhar. Where

the concentration in water is only four to five times as high as in "clean" areas. The level of copper also varies more in *Fucus* than in water, but the variation of iron is about the same in the alga and the water.

The cell walls of most seaweeds contain negatively charged polysaccharides (algin, fucoidin, etc.), to which metals attach. The nature of the accumulated ions is determined by the content in sea water and by the ion selectivity of the polysaccharides (Haug 1972). Brown algae contain ion-binding substances with a higher affinity for zinc than for other cations in sea water (Skipnes et al. 1975). According to Bryan (1969) e.g. the  $Zn^{2+}$  content of a *Fucus* plant is the result of irreversible zinc accumulation rather than an equilibrium established between the plant and surrounding water.

Free metal ions are most readily used by unicellular algae (Sunda and Guillard 1976) and probably also by *Fucus*. Ionic metal, however, is only one part of the total metal content in water. Thus the concentration of a metal in an alga does not necessarily indicate the total amount of this metal in the water, but only the soluble fraction.

#### 4. CONCLUSIONS

Filamentous algae cannot be recommended as indicators of metals in water.

In analyses of *Fucus vesiculosus* for zinc, copper and iron, atomic absorption spectrophotometry after wet digestion with  $HNO_3/HCl$  is suitable. Determination of lead and cadmium also requires prior chelation with APDC.

The material for analyses should be sampled at a fixed time of the year.

Samples for analyses should be cut from a fixed part of the *Fucus* thallus.

The metal level in algae does not necessarily indicate the total amount of metal in the water.

#### ACKNOWLEDGEMENTS

We are deeply grateful to the Walter and Andrée de Nottbeck Foundation for financing this study, to the staff of Tvärminne Zoological station for

good working facilities and to Dr. Paula Kankaala for critical reading of the manuscript.

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## FIELD APPLICATION OF PHYSIOLOGICAL METHODS ON FISH EXPOSED TO METAL POLLUTION

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LARSSON, Å., HAUX, C. & SJÖBECK, M.-L. 1986. Field application of physiological methods on fish exposed to metal pollution. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Physiological methods previously used to study the health of fish exposed to environmental pollutants in the laboratory have been tested on fish living in metal-polluted coastal waters and lakes. The most important results and observations are briefly reported. Aberrant values noted for several physiological parameters in fish from contaminated waters indicated that metal pollution affects physiological functions, such as the immune defence, the erythrocyte ALA-D activity and the ion balance. Good agreement was found between the toxic effects detected in fish exposed to metal pollution in nature and the effects observed in laboratory tests.

Index words: Field studies, brackish water, freshwater, teleost, fish physiology, sublethal effects, metal pollution, cadmium, lead

### 1. INTRODUCTION

Laboratory studies have shown that many physiological methods can be used to reveal adverse toxic effects in fish exposed to low levels of pollutants (Larsson et al. 1985). In the present study such diagnostic methods were tested on feral fish in metal-polluted waters during 1979–1983, as part of the Swedish Fish and Metal Pollution Project. The work comprised both research on methods and studies on fish living in lakes polluted by single metals (e.g. cadmium or lead) and in lakes and coastal waters polluted by emissions of several metals from a sulphide ore smelter. A comprehensive final report from the project is presented by Larsson et al. (1985).

### 2. MATERIAL AND METHODS

The experimental procedure had been thoroughly tested and standardized (Haux et al. 1985a). The fish were caught with gill nets, handled carefully and allowed to recover for 2–4 days in wooden fish chests before blood and other tissue samples were taken (Fig. 1). The clinical parameters mainly related to three aspects of physiology:

Hematology (the cellular and biochemical blood cell picture). Hematocrit, hemoglobin, red blood cell (RBC) count, mean cellular hemoglobin concentration (MCHC), mean cellular hemoglobin content (MCH), mean cellular volume (MCV), erythrocyte ALA-D activity, lymphocyte count, neutrophilic granulocyte count, thrombocyte

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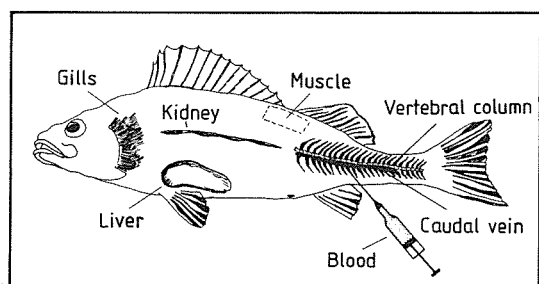


Fig. 1. Sampling of blood and other tissues in fish.

count.

**Metabolism.** Liver-somatic index (LSI), liver glycogen, muscle glycogen, blood glucose, blood lactate, total plasma protein.

**Osmotic and ionic balance.** Muscle water content, plasma sodium, plasma chloride, plasma potassium, plasma calcium, plasma magnesium, plasma inorganic phosphate.

The analytical procedures for determination of the physiological parameters have been described in previous reports (Dave et al. 1975; Johansson-Sjöbeck and Larsson 1978; Haux and Larsson 1979).

### 3. RESULTS AND DISCUSSION

#### 3.1. Methodological observations

The experimental procedure was found to function well and to give reliable baseline values for the physiological parameters of fish from uncontaminated waters (Haux et al. 1985a). However, the great number of investigations, carried out in different field conditions, revealed possible pitfalls and problems of crucial importance for the test results. Thus, in physiological field studies it is essential to take account of sex- and body size-dependent variation, selection of sampling stations, and not least seasonal variation (Larsson et al. 1985).

#### 3.2 Complex metal pollution

Perch (*Perca fluviatilis*) captured in August in lakes and in a part of the Gulf of Bothnia contaminated by a sulphide ore smelter, Rönnskärverken,

showed a strongly inhibited activity of the enzyme delta-aminolevulinic acid dehydratase (ALA-D) in the erythrocytes (Fig. 2), a reduced number of lymphocytes (Fig. 3), and slightly altered ion composition in the blood plasma (Larsson et al. 1985). The marked ALA-D inhibition suggests that the fish in these waters are affected by lead pollution. The low lymphocyte count for this time of the year, together with the low number of neutrophilic granulocytes and abnormal seasonal

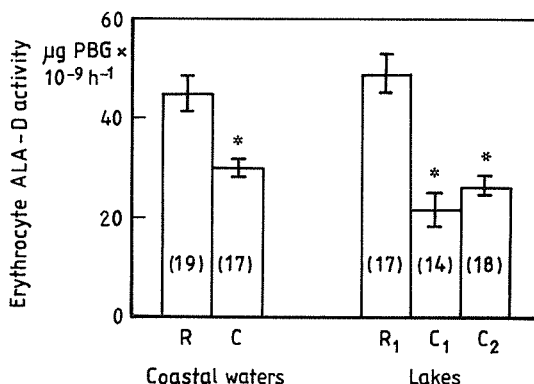


Fig. 2. Erythrocyte ALA-D activity in perch living at reference (R, R<sub>1</sub>) and metal polluted (C, C<sub>1</sub>, C<sub>2</sub>) stations in coastal waters and lakes. The results are expressed as mean values S.E. Number of fish within parenthesis. \* indicates a statistically significant difference between metal-loaded and reference fish.

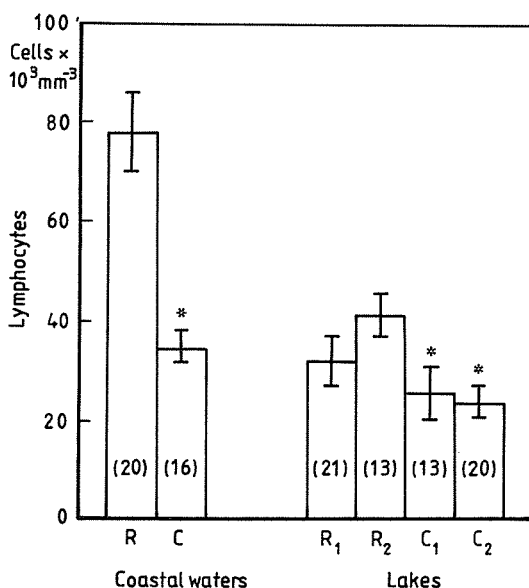


Fig. 3. Lymphocyte count in perch living at reference (R, R<sub>1</sub>, R<sub>2</sub>) and polluted (C, C<sub>1</sub>, C<sub>2</sub>) stations in coastal waters and lakes, respectively. Symbols as in Fig. 2.

variation in the white blood cell picture, indicates that the immune defence is affected by the complex metal pollution. Depression of the immunological defence system increases susceptibility to infectious diseases (Wedemeyer and McLeay 1981).

### 3.3 Cadmium pollution

Perch (*P. fluviatilis*) living in a cadmium polluted river (Emån) revealed typical symptoms of cadmium poisoning (Sjöbeck et al. 1984). The markedly enhanced lymphocyte count (Fig. 4), slight anemia and, in some fish, slight alterations of plasma potassium and magnesium are in good agreement with findings from laboratory experiments with fish exposed to cadmium (Larsson et al. 1985).

### 3.4 Lead pollution

Whitefish (*Coregonus* sp.) from two lakes receiving waste water from a lead-mine showed marked inhibition (by 87 %) of the erythrocyte ALA-D activity (Fig. 5; Haux et al. 1985b). This result confirms previous laboratory findings (Johansson-Sjöbeck and Larsson 1979) with fish exposed to lead and shows that the inhibited ALA-D activity in fish erythrocytes is a very sensitive and probably the earliest measurable toxic response in fish living in lead polluted waters. Besides the ALA-D response, carbohydrate metabolism and ion balance were slightly affected.

## 4. CONCLUSIONS

The experience from this field application of physiological techniques shows that certain clinical methods might be useful in detecting and diagnosing sublethal physiological effects in fish living in metal-polluted waters. The diagnostic value of many parameters is increased by the good agreement between the toxic effects detected in fish exposed to metals under natural field conditions and those in corresponding laboratory tests. Furthermore, some of the effects noted in the field and laboratory tests with cadmium and lead, resemble the clinical symptoms found in cadmium- and lead-poisoned mammals.

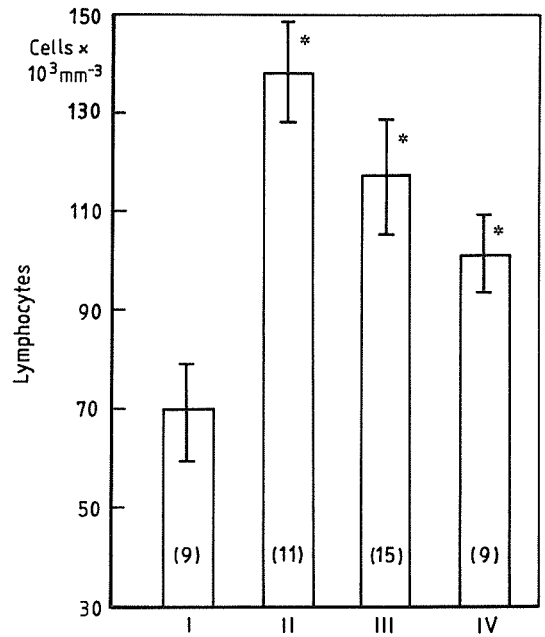


Fig. 4. Lymphocyte count in perch from a cadmium polluted river system. I = reference station; II, III, IV = cadmium-polluted stations. Symbols as in Fig. 2.

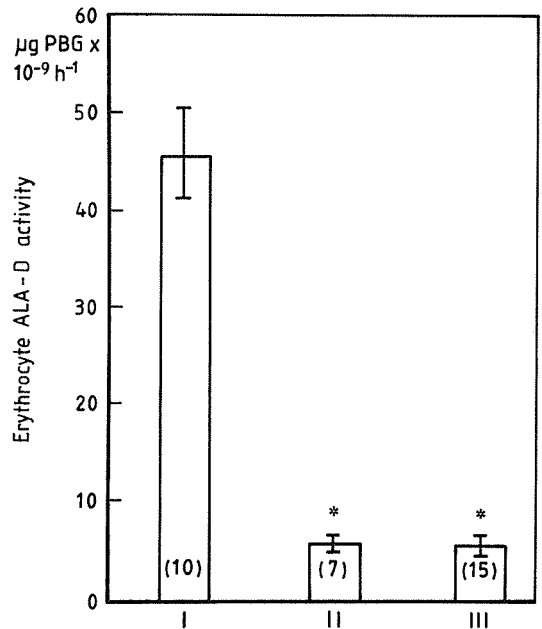


Fig. 5. Erythrocyte ALA-D activity in whitefish from a reference lake (I) and from lead-polluted lakes (II, III). Symbols as in Fig. 2.

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## DEFORMATION OF *MYTILUS EDULIS* L. SHELLS AS INDEX OF WATER QUALITY

Reijo Lindström<sup>1)</sup>

LINDSTRÖM, R. 1986. Deformation of *Mytilus edulis* L. shells as index of water quality. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

*Mytilus edulis* samples were collected from natural mussel populations in the Tvärminne area of SW Finland in summer 1983. The sampling area extended from Koverhar Iron and Steel Works to the open sea. The percentages of deformed shells and deformity types and the degrees of severity were determined at different sampling stations. The proportion of deformities decreased gradually from about 45 % at the dock of the Koverhar plant to about 3 % at a point 4 km southeast of Koverhar by water. At the same time, the severity of the deformities decreased, but the types of deformity remained unchanged.

Index words: Heavy metal pollution, monitoring of water quality, deformities of mussel shells, *Mytilus edulis* L.

### 1. INTRODUCTION

The typical pollutants discharged by steel and iron works are arsenic, ammonium, cyanide, phenols, oils and heavy metals, of which the most important are cadmium, chromium, lead, zinc and iron (Luotamo and Luotamo 1976). The levels of iron, zinc and lead in the tissues of *Mytilus edulis* and *Macoma baltica* were significantly higher in the vicinity of Koverhar than at the reference points in clean waters (Luotamo and Luotamo, 1976, 1979).

*M. edulis* has proved useful in the marine "Mussel Watch" monitoring programs (Anon. 1980, Davies and Pirie 1980, Phelps et al. 1981, Goldberg et al. 1983). In this study shell deformities in natural populations of *M. edulis* were examined in relation to their distance from the Koverhar Iron and Steel Works.

### 2. MATERIAL AND METHODS

Samples of *Mytilus* were collected from mussel fields at certain distances from the Koverhar plant in summer 1983 (Fig. 1). The depth was 4.5–6 m and the salinity 7 ‰. The sample sizes varied from 589 to 1000 individuals.

While the mussels were still alive, they were counted, their shell deformities were detected, counted and typed, and the severity of the deformity was determined. The types of deformities were divided into two main groups: continuous and discontinuous. The continuous deformities were classified as follows: 1. posterior score, 2. ventral score, 3. dorsal score, 4. flattening of posterior parts and 5. acutely pointed posterior parts. The discontinuous deformities were 6. humpy shells, in which normal growth re-emerged

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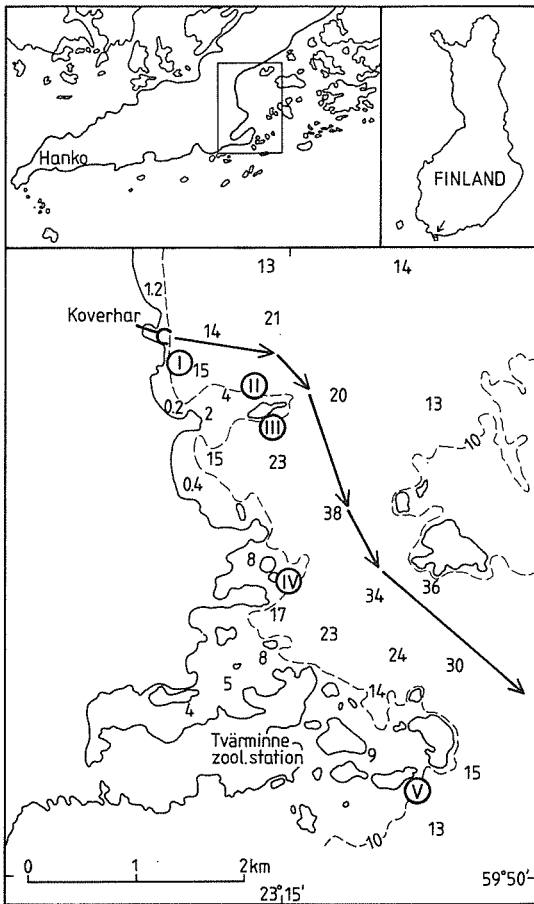


Fig. 1. Monitoring area. Circled numbers (from one to five) indicate sample points, other numbers the depths of water. Arrowed line shows the main direction of water flow at bottom.

following a temporary humpy growth and 7. streaks, i.e. brownish streaks or collars representing growth during exposure to pollutants. The severity of damage was classified as severe, moderate and slight. Severe damage is expected to make it difficult for the mussels to survive, while moderate damage is not a serious threat to survival and slight deformities merely affect the appearance.

### 3. RESULTS

Close to Koverhar (sampling station I in Fig. 1) the percentage of deformities was 44.5 ( $n = 843$ ), and at station II, about 1 000 m from the Koverhar it was 31.6 ( $n = 683$ ). At station III, located on the other side of an islet from station II, about 1 500 m from Koverhar by water, the percentage was 20.9 ( $n = 589$ ). At station IV (3 000 m) it was 6.6 and at station V (5 600 m) 2.8 ( $n = 1\,000$ ).

Thus, the percentages of deformed shells decreased with increasing distance from Koverhar (Fig. 2 and Table 1). At the same time the severity of deformity decreased, but the types of deformity remained unchanged (Table 1).

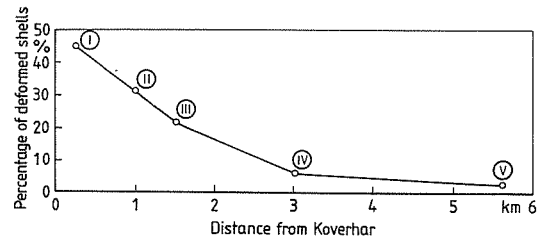


Fig. 2. Percentage of deformed shells at different sample points. Circled numbers indicate sample points, other numbers distances from Koverhar by way of water.

Table 1. Deformities at different sampling stations. Deformity types: 1. = posterior score, 2. = ventral score, 3. = dorsal score, 4. = posterior flattening, 5. = acute posterior pointing, 6. = humpy shell, 7. = streak.

| Station | Samples |          |      | Deformity    |          |        |           |      |     |      |     |      |      |
|---------|---------|----------|------|--------------|----------|--------|-----------|------|-----|------|-----|------|------|
|         | Total   | Deformed |      | Severity (%) |          |        | Types (%) |      |     |      |     |      |      |
|         | n       | n        | %    | severe       | moderate | slight | 1.        | 2.   | 3.  | 4.   | 5.  | 6.   | 7.   |
| I       | 843     | 375      | 44.5 | 21.7         | 32.4     | 45.9   | 2.7       | 51.6 | 2.3 | 2.0  | 0.0 | 8.5  | 32.7 |
| II      | 683     | 216      | 31.6 | 17.0         | 34.3     | 43.7   | 3.5       | 24.8 | 7.8 | 6.5  | 0.0 | 8.3  | 49.1 |
| III     | 589     | 123      | 20.9 | 15.1         | 29.5     | 55.4   | 2.9       | 14.4 | 6.5 | 2.2  | 0.7 | 14.4 | 59.0 |
| IV      | 882     | 58       | 6.6  | 15.7         | 20.0     | 64.3   | 1.4       | 30.0 | 1.4 | 8.6  | 0.0 | 15.7 | 42.9 |
| V       | 1 000   | 28       | 2.8  | 3.1          | 15.7     | 81.3   | 9.4       | 46.9 | 6.3 | 12.5 | 3.1 | 0.0  | 21.9 |

## 4. DISCUSSION

The shell deformities in natural populations of *Mytilus* attracted my attention years ago while I was studying the shells of molluscs in order to examine their life cycles and determine their age by scanning electron microscopy. It was easy to see that some shells had developed in a very regular manner and others had not. Also, shells collected in Finland in the 19th century were generally much more symmetrical and better formed than shells of the present. The reason for this is that the world is now more disordered than in older times, even from the point of view of the mussels. So the irregularities in the shells of *Mollusca* are correlated with disturbance of their environment.

Shell deformities have been reported to occur in mussels exposed to crude oil (Le Pennec and Le Roux 1979) and antimony (Walz 1979). Exposure to copper and cadmium for 24 hours caused certain types of deformities in shells of *Mytilus edulis* L. during the following year (Sunila and Lindström 1985). The distributions by deformity type after exposure to different metals, control manipulation or even within a natural population were not very different. Accordingly, environmental disturbance in general seems to give rise to about the same kind of shell deformities.

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Anonymous (National Academy of Sciences and National Research Council Environmental Studies Board

## CHLORINATED HYDROCARBONS IN THE WATER AND SEDIMENTS OF THE SEA AREAS AROUND FINLAND

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PERTTILÄ, M. & HAAHTI, H. 1986. Chlorinated hydrocarbons in the water and sediments of the sea areas around Finland. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

The concentrations of DDT and PCBs were determined in Baltic marine sediments and sea water. Large areal variation was detected; the surface sediments of the Gulf of Bothnia contained 2–5 ng g<sup>-1</sup> dry weight of DDT and 5–10 ng g<sup>-1</sup> of PCBs and the surface sediments at two sampling stations in the Gulf of Finland contained 10–20 ng g<sup>-1</sup> of DDT and 20–60 ng g<sup>-1</sup> of PCBs. In areas where the rate of sedimentation is high, the maximum organochlorine concentrations were found in sediment layers corresponding to the years 1970–1975.

Index words: Chlorinated hydrocarbons, sediments, water, Baltic Sea

### 1. INTRODUCTION

In the Baltic Sea countries the use of the pesticide DDT, was abandoned and restrictions were laid on the use and discharge of PCBs in the late 1960s and early 1970s. Since then, considerable attention has been devoted to possible trends in the concentrations of these substances in the marine environment. Methodological advances have been rapid since the 1970s, however, and the new methods have not always been thoroughly compared with the old ones before they came into use. This makes the results difficult to evaluate, especially in the case of PCBs. Also, the type and age of the samples (especially biological samples) often vary widely, and this affects the comparability of the results. There are several reports that the concentrations

of chlorinated hydrocarbons in biological material depend strongly on the age of the organisms (see, e.g. Phillips 1980: 237–260, Pyysalo et al. 1983, Perttilä et al. 1982), and thus emphasis should be laid upon the homogeneity of the samples. The monitoring of the Baltic marine environment will probably yield adequate data for trend analyses in a few years. So far the data are too scarce for statistical analyses, but studies conducted in several laboratories indicate that the concentrations of DDT and possibly also PCB are decreasing in the biota of the Baltic Sea (Paasivirta and Linko 1980, Moilanen et al. 1982, Olsson 1978). The sediments can be regarded as a kind of specimen bank. When the rate of sedimentation is stable, and the effects of certain disturbances (currents, bioturbation) are small, the sediment layers reflect the situation in the water mass at the time of deposition.

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Fig. 1. Sampling stations.

## 2. MATERIAL AND METHODS

Sediment samples (Fig. 1.) were taken on board the research vessel *Aranda* in 1981–1983 with a gravity corer. Immediately after sampling, the sediment core was cut into 1-cm-thick slices, which were stored in a deep-freeze. In the laboratory, the samples were freeze-dried and homogenized. 2,2', 3, 4, 5', 6, 6'-heptachlorobiphenyl was used as an internal standard. The samples were extracted with 50/50 acetone-hexane mixture for one hour in an ultrasonic bath. The extract was treated with concentrated sulphuric acid, and elemental sulphur was removed by reducing it to sulphide by means of elemental copper. Recovery for PCBs was 60–70 %, and for DDT 70–75 %. Water samples (20–40 l) were first filtered through a GF/C glass fibre filter and then run through an XAD-2 column approximately 2 bedvolumes/min. The column was later extracted with acetone for 4 hours. Extracted organics were then back-extracted to hexane, concentrated and cleaned with sulphuric acid and a silica/alumina column. Recovery for organochlorines was 70–80 %.

Determinations were carried out using an HP5730 GC, with splitless injection and temperature programming, 160–240°C in 16 minutes. PCBs were calculated using Clophen A50 as a standard.

## 3. RESULTS AND DISCUSSION

The DDT and PCB concentrations in the sediment cores are given in Figs. 2–5. The concentrations of DDT metabolites at three stations are given in Figs. 6–8. It should be noted that the results for different stations are not directly comparable with each other, because the sedimentation rate varies greatly. Sedimentation is most rapid in the eastern Gulf of Finland; at XV-1 the sedimentation rate averages 7.4 mm a<sup>-1</sup>. At Teili 1 the rate is 3.4 mm a<sup>-1</sup>, at EB1, US5b and F2a the rates are 2.4, 2.0 and 1.2 mm a<sup>-1</sup>, respectively. These sedimentation rates have been measured as the average of the 10 uppermost centimetres (Niemistö and Voipio 1981). Thus the concentrations found in a 1-cm-thick sample represent the floating average of a period, the length of which is different at different stations. At XV-1 the period is 1.3 years; at F2a it is 8.3 years.

At all sampling sites, the organochlorine concentrations start from very low values and rise with decreasing depth in the sediment. In regions where the rate of sedimentation is relatively low (Bothnian Sea and the Bothnian Bay), both the DDT and PCB concentrations increase almost evenly towards the sediment surface. This is to be expected, because the PCB and DDT discharges have been diminishing — or have ended — during the last 10–15 years, which is too short a period for changes to be seen in the sediments of this area, bearing in mind the very low level of concentrations, and the slow rate of sedimentation.

The concentrations are higher in the Gulf of Finland. At XV-1, which has an exceptionally high

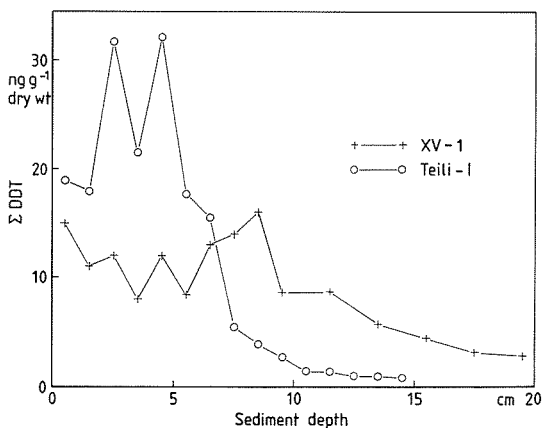


Fig. 2. Sum-DDT as a function of sediment depth at the stations XV-1 (eastern Gulf of Finland) and Teili-1 (Northern Baltic Proper).

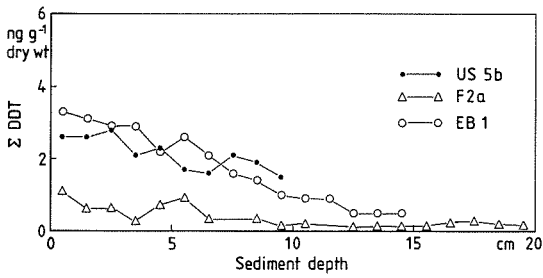


Fig. 3. Sum-DDT as a function of depth in the sediment cores in the Gulf of Bothnia.

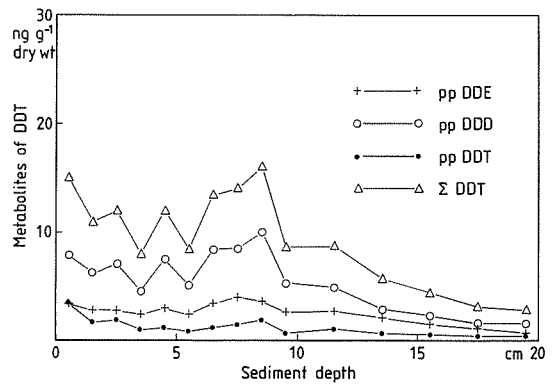


Fig. 6. DDT with its metabolites as functions of sediment depth at the station XV-1 (eastern Gulf of Finland).

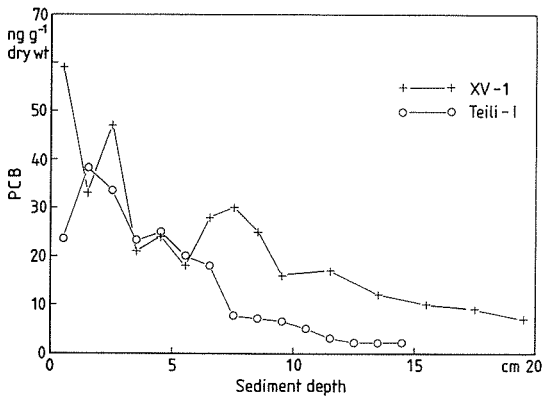


Fig. 4. PCB as a function of sediment depth at the stations XV-1 (eastern Gulf of Finland) and Teili-1 (northern Baltic Proper).

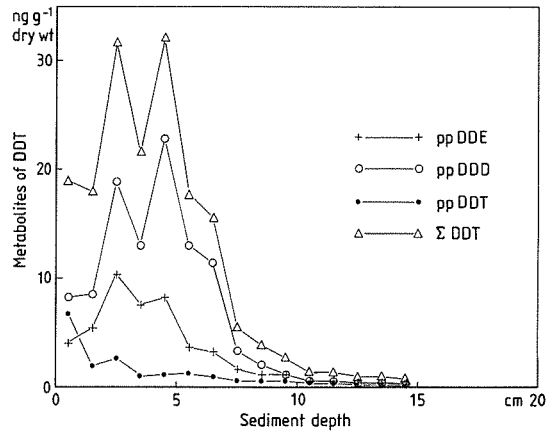


Fig. 7. DDT with its metabolites as functions of sediment depth at the station Teili-1 (northern Baltic Proper).

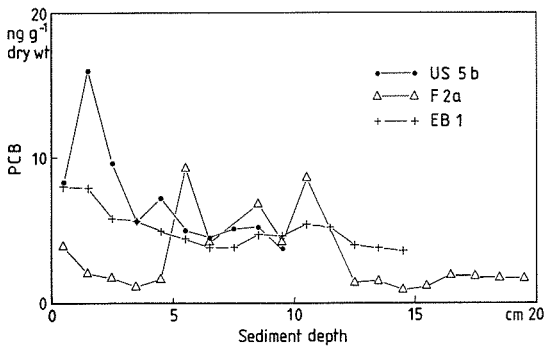


Fig. 5. PCB as a function of sediment depth in the sediment cores in the Gulf of Bothnia.

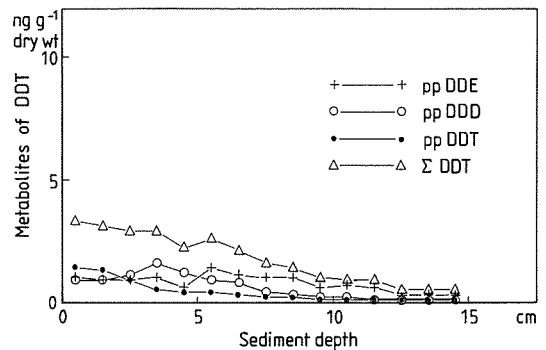


Fig. 8. DDT with its metabolites as functions of sediment depth at the station EB1 in the Bothnian Sea.

rate of sedimentation, the maximum DDT values are found 8–10 cm from the sediment surface, in a layer corresponding to the years 1969–1973. After this period, the values vary somewhat, giving a slope not differing significantly from zero. At Teili 1, where the rate of sedimentation is also fairly high, the maximum DDT values are found at a depth of 3–5 cm, which corresponds to the years 1968–1974. In the case of the PCBs, no peak is evident; the concentrations increase steadily towards the sediment surface. Figs. 6–8, giving the concentrations of the DDT metabolites, show that, with the exception of the uppermost layers, the most abundant DDT component is DDD. In biogenic material the most abundant component is generally DDE. The DDT values in the sediments of the Gulf of Finland and the northern Baltic Proper accord with the reduction of DDT discharges to the Baltic Sea, that has taken place since restrictions were laid upon the use of this insecticide. Considerably higher organochlorine concentrations have been reported for the sediments of the BOSEX area in the southern Baltic Proper (Niemistö and Tervo 1978). Concentrations up to 100 ng g<sup>-1</sup> dry weight of PCBs and 30 ng g<sup>-1</sup> dry weight of DDT were detected in the sediment surface layer.

In the water of the Baltic Sea, the organochlorine concentrations vary from approximately 0.5 to 2 ng l<sup>-1</sup>. The lowest values have been detected in the Bothnian Bay. Otherwise, there do not seem to be significant variations in the PCB and DDT concentrations. These results are somewhat lower than those reported earlier for the Baltic Proper (Osterroth 1977).

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## ANALYSIS OF AQUATIC BIOTA FOR POLYCHLORINATED DIOXINS AND DIBENZOFURANS

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RAPPE, C., BERGQVIST, P.-A. & MARKLUND, S. 1986. Analysis of aquatic biota for polychlorinated dioxins and dibenzofurans. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

A series of highly toxic polychlorinated dioxins and dibenzofurans has been identified in various aquatic samples from the Gulf of Bothnia and the Baltic Sea. The levels are in the ppt range. The same congeners have also been identified in samples of human adipose tissue and mother's milk from Northern Sweden and Germany. The ultimate origin of these hazardous contaminants is still unclear, but a correlation with incineration of chloro-organic compounds cannot be excluded.

Index words: Polychlorinated dioxins, polychlorinated dibenzofurans, aquatic biota

### 1. INTRODUCTION

Polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are two series of tricyclic almost planar aromatic compounds which exhibit similar physical, chemical and biological properties and have been the subject of much concern in recent years. They have been involved in accidents like that in Seveso, Italy, in 1976, the fire in the Binghamton State Office Building in 1981, the Love Canal incident in Niagara Falls in 1979, the intoxications at horse arenas in Missouri in 1971 and at Times Beach, Missouri 1982—83, the Yusho accidents in Japan in 1968 and in Taiwan in 1979, and also in the herbicide spraying program in Vietnam in the late 1960s. The chemical structures and numbering of these hazardous compounds are given in Fig. 1. The number of chlorine atoms in the compounds can vary between one and eight, producing up to 75 PCDD and 135 PCDF positional isomers. Animal studies and *in vitro* experiments have indicated

that there are pronounced differences in toxic and biological effects among the PCDD and PCDF isomers. A 1 000—10 000 -fold difference can be found between such closely related isomers as 2,3,7,8- and 1,2,3,8-tetra-CDD, and 1,2,3,7,8- and 1,2,4,7,8-penta-CDD. The isomers with the highest activity and the highest acute toxicity are those having 4—6 chlorine atoms and all lateral (2,3,7 and 8) positions substituted for chlorine. The

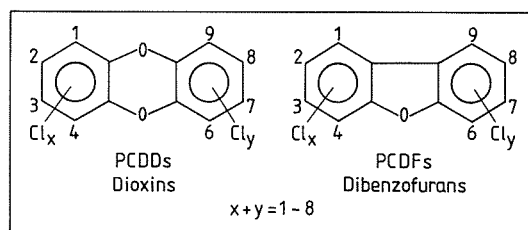


Fig. 1. Formulas for PCDDs and PCDFs.

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LD<sub>50</sub> values for guinea pigs for 2,3,7,8-tetra-CDD, 1,2,3,7,8-penta-CDD, 2,3,7,8-tetra-CDF and 2,3,4,7,8-penta-CDF are all below 10  $\mu\text{g kg}^{-1}$ . Because of the extreme toxicity of some of the isomers of PCDDs and PCDFs, highly sensitive and specific analytical techniques are required for the measurements. Detection levels for human and ecological samples should be orders of magnitude below the usual detection levels obtained in pesticide analysis. Another complication is the large number of isomers and the large variation in toxicity and biological potency between closely related isomers. A risk evaluation should be based on the levels of the highly toxic isomers found in isomer-specific analyses. In recent years a great number of analytical methods have been developed for the determination of trace amounts of PCDDs and PCDFs and especially the most toxic 2,3,7,8-tetra-CDD in technical and environmental samples. For a recent review on PCDDs and PCDFs, see Rappe (1984).

PCDDs and PCDFs are known contaminants in a series of industrial chemicals such as chlorophenols, chlorophenoxy acetic acid derivatives and polychlorinated biphenyls (PCBs). Together with other polyhalogenated aromatics, they have also been detected in emissions from both municipal and industrial incinerators (Rappe 1984).

Rappe et al. (1981) identified a series of tetra to octa-CDFs in fat samples from a gray seal in the Gulf of Bothnia and in a snapping turtle from the Hudson River. The total levels of PCDFs in these samples were 40  $\text{pg g}^{-1}$  and 3  $\text{ng g}^{-1}$ , respectively. In both samples the major PCDFs consisted of the most toxic isomers (2,3,7,8-tetra-, 2,3,4,7,8-penta- and 1,2,3,4,7,8- and 1,2,3,6,7,8-hexa-CDF).

Norstrom et al. (1982) have analyzed pooled samples of herring gull eggs collected in 1982 from various parts of the Great Lakes. In all samples 2,3,7,8-tetra-CDD was found at levels ranging from 9 to 90  $\text{pg g}^{-1}$ . In another study Stalling et al. (1983) were not able to identify measurable levels of tetra-CDDs and other PCDDs in fish samples from Lake Superior (the detection level was 2–5 ppt). The difference could be explained by the migration of the herring gulls during the winter. On the other hand a series of PCDFs could be identified in the Lake Superior fish samples, indicating more widespread background levels for the PCDFs than for the PCDDs. Stalling et al. (1983) also reported the results of analysis of fish samples from Lakes Michigan, Huron and Ontario. The toxic 2,3,7,8-substituted PCDDs and PCDFs were present in all samples, the highest levels being found in samples from Lake Huron, Lake Ontario and the Tittabawasee River. The residue pattern

found in the fish and locally high levels suggest contamination from point sources.

## 2. MATERIAL AND METHODS

The experimental conditions for the clean-up, the capillary gas chromatography and the mass spectrometry are given by Rappe et al. (1981), Stalling et al. (1983) and Rappe et al. (1983).

## 3. RESULTS

We have analyzed a series of environmental samples from the Gulf of Bothnia and the Baltic Sea. In the present study we are reporting on the levels of PCDDs and PCDFs found in six representative samples:

1. Osprey, egg from Holmöarna, Sweden, collected in 1981.
2. Ringed seal, fat, from Simo, Finland, collected in 1978.
3. Salmon, muscle, from Umeälven, Sweden, collected in 1980.
4. Guillemot, muscle, from Stora Karlsö, Sweden, collected in 1971.
5. Herring, muscle, from Utlängan, Sweden, collected in 1981.
6. Cod, liver, from Travemünde, Germany, collected in 1984.

A series of polychlorinated dioxins and dibenzofurans has been identified in all these samples and the results are presented in Table 1. The toxic 2,3,7,8-substituted isomers are dominant within each group of congeners. Due to technical problems, the analyses for 2,3,7,8-tetra-CDD has not been completed.

## 4. CONCLUSIONS

This study strongly indicates that we have a general background level of the toxic 2,3,7,8-substituted PCDDs and PCDFs in the Gulf of Bothnia and the Baltic Sea. We also have a strong indication that there is a background level of the

Table 1. PCDD and PCDF levels in environmental samples.

|                                 | Sample number         |                                 |                                   |                                      |                                    |                                |
|---------------------------------|-----------------------|---------------------------------|-----------------------------------|--------------------------------------|------------------------------------|--------------------------------|
|                                 | 1<br>Osprey<br>pg/egg | 2<br>Seal<br>pg g <sup>-1</sup> | 3<br>Salmon<br>pg g <sup>-1</sup> | 4<br>Guillemot<br>pg g <sup>-1</sup> | 5<br>Herring<br>pg g <sup>-1</sup> | 6<br>Cod<br>pg g <sup>-1</sup> |
| 2378-TCDF                       | 63                    | 12                              | 62                                | 0.22                                 | 3.9                                | 5.2                            |
| 12378-PeCDF                     | 27                    | 3.7                             | 24                                | 1.8                                  | 1.2                                | 3.3                            |
| 23478- —"—                      | 767                   | 125                             | 82                                | 97                                   | 5.6                                | 9.0                            |
| 123478/123479-HxCDF             | 17                    | 1.1                             | 9.1                               | 4.7                                  | 0.6                                | 0.36                           |
| 123678- —"—                     | 19                    | 2.4                             | 6.3                               | 10                                   | 0.4                                | 1.0                            |
| 234678- —"—                     | 16                    | 1.5                             | 5.1                               | 4.5                                  | 0.7                                | 1.9                            |
| 1234678-HpCDF                   | 7.5                   | 0.20                            | 41                                | 0.50                                 | 0.5                                | 0.95                           |
| 1234679- —"—                    | ND                    | ND                              | 0.56                              | ND                                   | ND                                 | ND                             |
| 1234689- —"—                    | ND                    | ND                              | 2.2                               | ND                                   | 0.2                                | ND                             |
| 1234789- —"—                    | ND                    | ND                              | 0.86                              | T                                    | ND                                 | ND                             |
| OCDF                            | 5.9                   | 1.8                             | 61                                | 0.04                                 | ND                                 | 0.17                           |
| 2378-TCDD                       | NA                    | NA                              | NA                                | NA                                   | ND                                 | NA                             |
| 12378-PnCDD                     | 258                   | 60                              | 22                                | 26                                   | ND                                 | 2.1                            |
| 123478-HxCDD                    | 33                    | 2.9                             | 0.78                              | 0.90                                 | ND                                 | 0.82                           |
| 123678- —"—                     | 141                   | 69                              | 6.6                               | 18                                   | ND                                 | 2.8                            |
| 123789- —"—                     | 87                    | 3.6                             | 0.24                              | 1.6                                  | ND                                 | 1.4                            |
| 1234679-HpCDD                   | 11                    | 0.40                            | 0.53                              | ND                                   | ND                                 | T                              |
| 1234678- —"—                    | 136                   | 1.2                             | 0.90                              | 0.62                                 | ND                                 | 3.1                            |
| OCDD                            | 275                   | 2.5                             | 4.3                               | 0.24                                 | ND                                 | 1.0                            |
| Recovery <sup>13</sup> C-TCDF % | 134                   | 75                              | 125                               | 83                                   | 88                                 | 70                             |
| Recovery <sup>13</sup> C-OCDD % | 109                   | 85                              | 87                                | 91                                   | 44                                 | 76                             |

ND = Not detected

T = Trace quantities, not quantified

NA = Not analyzed

same isomers in human adipose tissue and mother's milk in Northern Sweden and in Germany (Rappe et al. 1984 a).

Of special interest is the observation of 1,2,3,7,8-penta-CDD, normally at levels of about 10 ppt or higher. In the human samples, this isomer is found at higher levels than 2,3,7,8-tetra-CDD. This particular isomer has never been reported as a contaminant in any commercial product, but it is always present in samples from municipal and hazardous waste incinerators (Rappe et al. 1984 b). At the present time the possibility cannot be excluded that incineration of municipal and hazardous waste contributes to the background levels of PCDDs and PCDFs found in the environment and in the general population.

It is of fundamental importance to complete the studies of the background levels of the highly toxic PCDDs and PCDFs in the environment and in the general population. Special attention must be given to the possible correlation between these background levels and emissions from various incinerators.

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## HEAVY METALS IN BENTHIC CRUSTACEANS AND MYSIDS IN THE BOTHNIAN SEA

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SANDLER, H. 1986. Heavy metals in benthic crustaceans and mysids in the Bothnian Sea (northern Baltic Sea). Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Fe, Zn, Cu, Pb, Cd and Hg concentrations were determined on field collections of the isopod *Mesidotea entomon*, the amphipod *Pontoporeia affinis* and the mysids *Mysis mixta* and *M. relicta* from the Bothnian Sea.

The iron concentrations varied widely in all species and were high on a global scale. The zinc values were very even, indicating physiological regulation. The copper concentrations in *M. entomon* and *P. affinis* were about four times those in the mysids and were high on a global scale. The lead, cadmium and mercury concentrations were very low. The lead concentrations of *P. affinis* were three times those of *M. entomon* and the mysids. No areal trends were observed in the Bothnian Sea. *M. entomon* seems to be suitable as a bioindicator only for lead, cadmium and mercury pollution.

Index words: Heavy metals, field study, *Mesidotea entomon*, *Pontoporeia affinis*, Mysidaceae

### 1. INTRODUCTION

Good knowledge of concentrations of heavy metals in biota is important as it gives information about the load on the sea. The heavy metal concentrations of sediments are generally three or four orders of magnitude higher than the concentrations in the water. Thus benthic animals are exposed to high concentrations and the heavy metals may be accumulated along the food web, e.g. in fish that prey on benthic organisms. The present study examines the heavy metal concentrations of benthic crustaceans and mysids in the Bothnian Sea and tries to establish whether any areal or interspecific differences exist. The metals studied are: Fe, Zn, Cu, Pb, Cd and Hg. The species studied are the isopod *Mesidotea entomon*, the amphipod *Pontoporeia affinis* and the mysids *Mysis*

*mixta* and *M. relicta*. All these species are important food items of several commercially used fish species in the northern Baltic. *M. entomon* is used as an indicator of harmful substances, e.g. in the Baltic Monitoring Programme (BMP) and in the coastal monitoring programme of the National Board of Waters, Finland. Properties which make this species a suitable monitoring organism are its benthic life, large size and long life span. Its value as a bioindicator of heavy metal pollution will be further elucidated in the present study. *P. affinis* is numerically overwhelmingly predominant in the macrobenthic communities in the Gulf of Bothnia. In spite of its great importance, data on the concentrations of heavy metals in the species have seldom been published. The mysids have also been very poorly studied as regards heavy metals. They differ from *M. entomon* and *P. affinis* in not being

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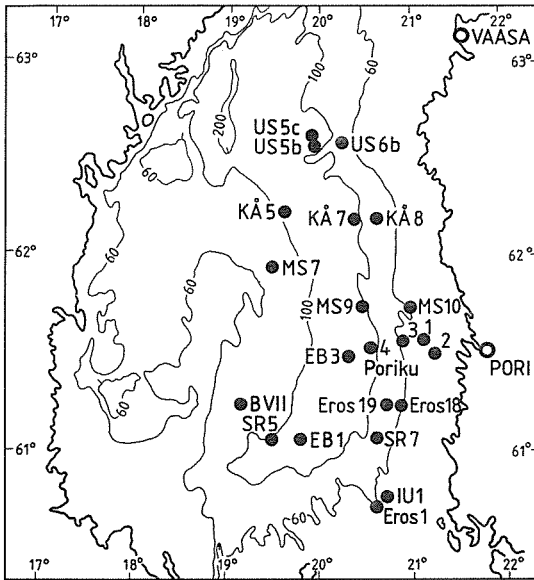


Fig. 1. The study area with depth curves and the sampling localities.

true benthic animals, as they undertake nocturnal vertical migrations to feed in the pelagial (Beeton and Bowers 1982) and live an epibenthic life during the daylight hours. This habitat difference may be mirrored in their heavy metal concentrations.

## 2. MATERIAL AND METHODS

The sampling was carried out on board the r/v Aranda between July 1981 and September 1982 and samples were taken at 22 localities (Fig. 1). *P. affinis* and *M. entomon* were sampled with a bottom dredge and the mysids with an epibenthic sledge.

In order to remove sediment and faeces, the specimens were kept alive in containers with surface water for 12 hours, after which the samples were kept deep-frozen until further treatment.

Prior to analysis the samples were freeze-dried, homogenized and extracted with  $H_2O_2$  and  $HNO_3$ . The analyses were carried out by flameless AAS (Perkin Elmer 4000 equipped with a HGA 400 furnace). The concentrations are expressed as  $\mu g$  metal  $g^{-1}$  dry tissue.

In the case of *M. entomon* and *P. affinis*, the heavy metals were analysed for different size-classes and, whenever possible, five parallel samples

were collected and analysed for each size-class and locality. In the present paper, however, only the population means are given; another paper (Sandler in prep.), based on the same material, deals with accumulation of the heavy metals and with both seasonal and age-dependent variation in their concentrations.

## 3. RESULTS AND DISCUSSION

### 3.1. Iron

The mean iron concentration in *M. entomon* is  $351 \mu g g^{-1}$  (Fig. 2). For *P. affinis* the mean is  $412 \mu g g^{-1}$  and for the mysids  $510 \mu g g^{-1}$ . The highest iron concentrations in *M. entomon* were found at station KÅ 5 (Table 1, Fig. 1), where high concentrations of iron were also observed in *P. affinis*, together with high values for zinc and copper (Table 2). The highest iron values in the mysids (Table 3) were observed at station MS 7, which is situated near KÅ 5. At both stations, especially KÅ 5, the surface of the sediments was observed to be formed of a hard layer of concretions. The average concentration of iron in Baltic Sea concretions is c. 30 % (weight percentage) (Winterhalter 1966) and it was not possible in this study to determine whether the high iron concentrations in the animals from the area of stations KÅ 5 and MS 7 reflect contamination by concretion particles or whether iron is really accumulating in their tissues in these environments with exceptionally high iron concentrations.

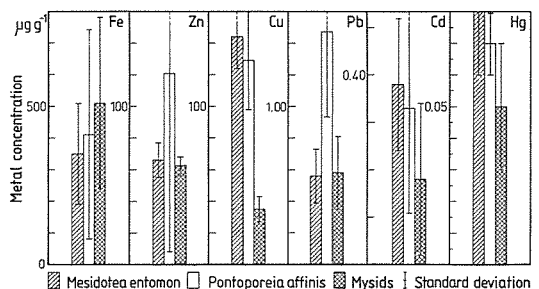


Fig. 2. The heavy metal concentrations ( $\mu g$  metal  $g^{-1}$  dry tissue) of *Mesidotea entomon*, *Pontoporeia affinis* and mysids in the Bothnian Sea. The figure presents the mean concentrations of all localities. For stations see Fig. 1.

Table 1. Concentrations of heavy metals in *Mesidotea entomon* ( $\mu\text{g metal g}^{-1}$  dry tissue) in the Bothnian Sea. For stations see Fig. 1.

| Station  | Date    | Depth m | Fe  | Zn | Cu  | Pb   | Cd   | Hg   |
|----------|---------|---------|-----|----|-----|------|------|------|
| Poriku 1 | July 81 | 37      | 406 | 75 | 157 | 0.60 | 0.41 |      |
| —”—      | Jan. 82 | ”       | 324 | 67 | 139 | 0.42 | 0.36 |      |
| —”—      | May 82  | ”       | 313 | 58 | 140 | 0.36 | 0.48 | 0.03 |
| Poriku 2 | July 81 | 26      | 538 | 93 | 145 | 0.52 | 0.49 |      |
| —”—      | Jan. 82 | ”       | 286 | 70 | 129 | 0.31 | 0.36 |      |
| —”—      | May 82  | ”       | 498 | 58 | 137 | 0.34 | 0.40 | 0.07 |
| Poriku 3 | July 81 | 56      | 482 | 85 | 178 | 0.60 | 0.53 |      |
| —”—      | Jan. 82 | ”       | 526 | 65 | 127 | 0.90 | 0.53 |      |
| —”—      | May 82  | ”       | 261 | 60 | 151 | 0.41 | 0.20 | 0.04 |
| Poriku 4 | May 82  | 99      | 213 | 78 | 208 | 0.36 | 0.61 | 0.12 |
| EB 1     | July 81 | 125     | 330 | 61 | 86  | 1.44 | 0.51 |      |
| —”—      | Jan. 82 | ”       | 348 | 53 | 133 | 0.25 | 0.30 | 0.11 |
| IU 1     | July 81 | 37      | 320 | 88 | 133 | 0.68 | 0.55 |      |
| Eros 1   | July 81 | 75      | 256 | 78 | 123 | 0.71 | 0.36 |      |
| —”—      | Jan. 82 | ”       | 267 | 67 | 127 | 0.68 | 0.34 |      |
| —”—      | May 82  | ”       | 276 | 52 | 159 | 0.42 | 0.16 | 0.05 |
| Eros 19  | July 81 | 70      | 480 | 69 | 113 | 0.87 | 0.69 |      |
| BVII     | July 81 | 49      | 406 | 75 | 112 | 1.66 | 0.34 |      |
| SR 7     | May 82  | 77      | 178 | 60 | 155 | 0.35 | 0.30 | 0.09 |
| MS9      | May 82  | 103     | 576 | 52 | 140 | 0.49 | 0.25 | 0.04 |
| KÅ5      | May 82  | 101     | 819 | 66 | 134 | 0.64 | 0.45 | 0.04 |
| KÅ8      | May 82  | 70      | 529 | 57 | 189 | 0.40 | 0.32 | 0.07 |
| US5b     | May 82  | 200     | 243 | 57 | 152 | 0.25 | 0.27 | 0.11 |
| US5c     | May 82  | 120     | 376 | 58 | 140 | 0.31 | 0.18 | 0.10 |
| US6b     | May 82  | 82      | 181 | 59 | 195 | 0.32 | 0.18 | 0.09 |
| SR5      | May 82  | 125     | 176 | 61 | 150 | 0.35 | 0.30 | 0.10 |

Table 2. Concentrations of heavy metals in *Pontoporeia affinis* ( $\mu\text{g metal g}^{-1}$  dry tissue) in the Bothnian Sea. For stations see Fig. 1.

| Station  | Date    | Depth m | Fe    | Zn  | Cu  | Pb   | Cd   | Hg   |
|----------|---------|---------|-------|-----|-----|------|------|------|
| PORIKU 1 | July 81 | 37      | 429   | 68  | 92  | 1.87 | 0.30 |      |
| PORIKU 3 | July 81 | 56      | 384   | 76  | 125 | 1.30 | 0.34 |      |
| PORIKU 3 | Jan. 82 | 56      | 125   | 138 | 146 | 2.00 | 0.57 |      |
| Eros 1   | July 81 | 75      | 298   | 70  | 94  | 1.27 | 0.21 |      |
| Eros 18  | July 81 | 70      | 219   | 242 | 107 | 0.90 | 0.70 |      |
| Eros 19  | July 81 | 70      | 658   | 96  | 128 | 2.30 | 0.71 |      |
| EB 1     | July 81 | 125     | 171   | 58  | 105 | 0.90 | 0.36 |      |
| MS 9     | May 82  | 103     | 256   | 56  | 125 | 1.76 | 0.09 | 0.08 |
| KÅ 5     | May 82  | 101     | 1 355 | 434 | 208 | 0.51 | 0.13 | 0.05 |
| US 5b    | May 82  | 200     | 200   | 39  | 135 | 2.00 | 0.13 | 0.09 |
| US 5c    | May 82  | 120     | 435   | 71  | 160 | 1.36 | 0.12 | 0.08 |

The ranges of the concentrations reported for *M. entomon* in the northern Baltic are shown in Table 4. Only two studies have dealt with iron concentrations, and both have been conducted in the Bothnian Sea. The concentrations are high compared with the values for related crustaceans from other regions reported in the review by Eisler (1981). The values of Häkkinä (1980) (Table 4) are especially high, but his study was carried out near

the shore in the surroundings of a titanium dioxide plant. A probable reason for the high iron concentrations in the animals in general is the fact that high iron concentrations have been observed in the water (and sediments) of the Bothnian Sea (Tulkki et al. 1983) and that the natural background level of iron in the sediments is high in the Baltic Sea.

Table 3. Concentrations of heavy metals ( $\mu\text{g metal g}^{-1}$  dry tissue) in mysids (*Mysis mixta* and *Mysis relicta*) in the Bothnian Sea. For stations see Fig. 1.

| Station | Date    | Depth<br>m | Fe    | Zn | Cu | Pb   | Cd   | Hg   |
|---------|---------|------------|-------|----|----|------|------|------|
| SR5     | Jun. 82 | 125        | 434   | 68 | 47 | 1.02 | 0.16 | 0.08 |
| SR5     | July 82 | 125        | 415   | 53 | 25 | 0.40 | 0.17 | 0.06 |
| EB 3    | July 82 | 130        | 334   | 56 | 28 | 0.63 | 0.11 | 0.03 |
| EB 3    | Sep. 82 | 130        | 475   | 66 | 46 | 0.59 | 0.25 |      |
| US6b    | May 82  | 80         | 276   | 52 | 38 | 0.37 | 0.03 | 0.05 |
| US6b    | July 82 | 80         | 465   | 62 | 27 | 0.79 | 0.12 | 0.05 |
| KÅ7     | July 82 | 111        | 595   | 91 | 46 | 4.56 | 0.64 | 0.04 |
| MS10    | July 82 | 51         | 310   | 55 | 28 | 0.74 | 0.11 | 0.05 |
| MS 7    | July 82 | 73         | 1 269 | 54 | 31 | 0.47 | 0.12 | 0.05 |
| Eros 1  | July 82 | 75         | 529   | 58 | 34 | 0.62 | 0.09 | 0.02 |

Table 4. Ranges of published heavy metal concentrations of *Mesidotea entomon* ( $\mu\text{g metal g}^{-1}$  dry tissue) in the northern Baltic Sea.

| Author                                  | Area                                      | Fe        | Zn      | Cu      | Pb         | Cd        | Hg        |
|-----------------------------------------|-------------------------------------------|-----------|---------|---------|------------|-----------|-----------|
| Nuorteva & Häsänen (1975) <sup>x)</sup> | Gulf of Finland<br>Bromarv area           |           |         |         |            |           | 0.06      |
| Niemi (1977)                            | Bothnian Bay<br>off Kokkola               |           | 104—125 | 194—268 |            |           |           |
| Voipio et al. (1977) <sup>x)</sup>      | Bothnian Bay off<br>Oulu and Kokkola      |           | 68—120  | 79—261  | 0.68—7.89  | 1.42—4.54 | 0.31—0.75 |
|                                         | Bothnian Sea                              |           | 69—114  | 122—240 | 0.90—2.96  | 0.91—3.04 | 0.02—0.13 |
|                                         | Gulf of Finland                           |           | 73—131  | 121—126 | 1.05—1.86  | 0.88—0.91 | 0.03      |
| Häkkinä (1980)                          | Bothnian Sea<br>off Pori                  | 495—3 240 | 108—154 | 65—206  | 0.70—2.50  | 1.30—1.80 | 0.16—0.66 |
| Tervo et al. (1980)                     | Bothnian Sea                              |           | 70—93   | 164—174 | 0.74—0.82  | 0.66—0.81 | 0.12      |
|                                         | Bothnian Bay <sup>xx)</sup>               |           | 62      | 258     | 1.36       | 1.15      | 0.30      |
| Kauppinen (1980)                        | The Quark<br>off Vaasa                    |           | 87—97   | 159—170 |            |           |           |
| Seisuma et al. (1983) <sup>xxx)</sup>   | Gulf of Riga                              |           | 35      |         | 5.00—10.00 | 0.50—0.70 | 0.01—0.12 |
| Sandler (1984)                          | Bothnian Sea<br>open areas                |           | 67—90   | 105—242 |            |           |           |
| The present study                       | Bothnian Sea<br>open and coastal<br>areas | 256—819   | 61—93   | 86—178  | 0.31—1.66  | 0.16—0.69 | 0.03—0.12 |

<sup>x)</sup> the concentrations are reported per wet weight unit; calculated to correspond to dry weights

<sup>xx)</sup> only one, very big/old specimens (length 92 mm)

<sup>xxx)</sup> the values are calculated from diagrams

### 3.2 Zinc

With the exception of a few values for *P. affinis* (Table 2), the zinc concentrations of all the species studied vary within a narrow range (Tables 1, 2 and 3). This indicates physiological regulation which makes the uptake independent of the ambient concentrations. Regulation of zinc in crustaceans is today a well-established fact; for example, it has been observed by Bryan (1967) in some species of decapod crustaceans and regulation has been also suggested for the isopod *M. entomon* (Sandler 1983, 1984).

The zinc values published for *M. entomon* in the northern Baltic (Table 4) are of the same order in all investigations, regardless of the fact that some of the studies were carried out in polluted sea areas. The zinc concentrations reported for mysids by Seisuma et al. (1983), however, are only c. 15  $\mu\text{g g}^{-1}$ , and thus only one quarter of the values of the present study (Table 3). This arouses suspicions of methodological differences.

Due to the physiological regulation of zinc, the concentrations of this biologically essential metal in aquatic crustaceans seem to be maintained at the same level in all environments.

### 3.3 Copper

The copper values vary only slightly and the levels in *M. entomon* and *P. affinis* are remarkably similar (Fig. 2), while the copper level in the mysids (Table 3) is considerably lower. Like the levels of zinc, the concentrations of copper reported for *M. entomon* in the northern Baltic (Table 4) are of the same order in all investigations, although some of these were conducted in areas polluted by heavy metals. On the other hand, the copper concentrations of

northern Baltic crustaceans seem to be remarkably high when compared with the data in the review by Eisler (1981) on crustaceans from other marine, coastal and estuarine areas.

### 3.4 Lead, cadmium and mercury

The lead concentrations vary considerably, within each species, without any areal trends. A possible explanation of the variations is the moults typical of crustaceans, as most of the lead seems to be located in the exoskeleton, with low concentrations in other tissues (Table 76 in Eisler 1981). This does not, however, explain the interspecific differences. The lead concentrations in *P. affinis* are on average more than twice as high as in *M. entomon* and the mysids.

In contrast to the lead levels, the cadmium and mercury concentrations of all the species studied vary only slightly and behave very similarly (Tables 1, 2 and 3) (Fig. 2).

The lead, cadmium and mercury concentrations of *M. entomon* vary widely in different studies in the northern Baltic (Table 4). Compared with the values of the present study, the lead concentrations of *M. entomon* from the Gulf of Riga (Seisuma et al. 1983) are, on average, one order of magnitude higher, indicating high levels of lead in the environment. Table 4 shows that the concentrations of cadmium and mercury are also higher in studies conducted in the surroundings of industry centres, as can be seen in the studies of Voipio et al. (1977) in the Bothnian Bay and of Häkkinä (1980). On the other hand, the Gulf of Riga shows very low values for the metals cadmium and mercury, of the same order as in the present study.

Table 5 presents the few data published on heavy metals in *P. affinis*. Both the lead and

Table 5. Ranges of published heavy metal concentrations of *Pontoporeia affinis* ( $\mu\text{g metal g}^{-1}$  dry tissue) in the northern Baltic Sea.

| Author                                    | Area                                      | Fe        | Zn     | Cu      | Pb        | Cd        | Hg        |
|-------------------------------------------|-------------------------------------------|-----------|--------|---------|-----------|-----------|-----------|
| Lithner (1974)                            | Bothnian Bay<br>Bay of Skellefteå         |           | 50—100 | 120—195 | 19—28     | 0.50—2.30 | 0.43—1.00 |
| Nuorteva & Häsänen<br>(1975) <sup>x</sup> | Gulf of Finland<br>Bromarv area           |           |        |         |           |           | 0.05      |
| Sandler (1984)                            | Bothnian Sea<br>open areas                |           | 56—137 | 90—130  |           |           |           |
| Present study                             | Bothnian Sea<br>open and coastal<br>areas | 125—1 355 | 39—434 | 92—208  | 0.51—2.30 | 0.09—0.71 | 0.05—0.09 |

<sup>x</sup>) the concentrations are reported per wet weight unit; calculated to correspond to dry weights

mercury values recorded outside the Rönnskärs-verken (metal industry) (Lithner 1974) are among the highest observed in the biota of the Baltic Sea and are located at the upper end of the global range. The results of the present study showed, however, that the concentrations of lead in *P. affinis* are considerably higher than in *M. entomon* and the mysids in unpolluted open sea areas as well.

The lead concentrations of mysids in the Gulf of Riga are  $1\text{--}2\ \mu\text{g g}^{-1}$  (Seisuma et al. 1983) and thus a little higher than in the present study (Table 3). Nuorteva and Häsänen (1975) reported comparatively high concentrations of mercury ( $0.27\text{--}0.41\ \mu\text{g g}^{-1}$  dry weight) in mysids from a coastal area at the entrance to the Gulf of Finland and considerably lower values in both *M. entomon* and *P. affinis* (Tables 4 and 5). The last observation is not supported by the results of the present study.

The lead, cadmium and mercury concentrations of northern Baltic crustaceans (Table 4) can generally be considered lower than reported in the review of Eisler (1981) for related crustaceans elsewhere.

#### 4. CONCLUSIONS

The concentrations recorded for the four species in the present study reveal no clear areal trends in the Bothnian Sea, except in the case of iron. The scarcity of iron data and the high concentrations observed make further investigations desirable.

- On a global scale, it seems that the concentrations of iron and copper are high in *M. entomon*, *P. affinis* and the two mysids in the Bothnian Sea, while the concentrations of lead, cadmium and mercury are low. The zinc concentrations in the species studied are very even in the Bothnian Sea and on the same level as in related aquatic crustaceans in other marine, coastal and estuarine areas.
- *M. entomon* can be recommended as a bio-indicator of lead, cadmium and mercury pollution, but not of zinc or copper.

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## EFFECTS OF COPPER AND CADMIUM ON THE SHELL MORPHOLOGY, LENGTH GROWTH AND SURVIVAL OF THE MUSSEL *MYTILUS EDULIS* L.

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SUNILA, I. & LINDSTRÖM, R. 1986. Effects of copper and cadmium on the shell morphology, length growth and survival of the mussel *Mytilus edulis* L. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

Mussels (*Mytilus edulis* L.) were exposed to elevated concentrations of copper or cadmium in the laboratory and then placed in cages in August 1979, in the sea (s 7 ‰). One year later the maximum lengths of the mussels were measured and the shells screened for deformities. Growth was 0.6 cm a<sup>-1</sup> in the control cages and was retarded in the cages of exposed mussels. Shell deformities were found in 63 % of the mussels exposed to cadmium and 46 % of the mussels exposed to copper. In the control cages 26 % were deformed, but in a natural population only 3 %. The proportion of deformities was inversely related to growth. Low concentrations of the metals proved to be injurious in this long-term test, probably due to a closing reaction.

Index words: Heavy metals, deformity, survival, growth, *Mytilus*

### 1. INTRODUCTION

The common mussel, *Mytilus edulis* L., has proved to be a useful bioindicator in marine monitoring programmes (Goldberg 1980). Besides the accumulation of pollutant substances by mussels, their physiological responses should be followed to establish water quality criteria.

As the growth rate is a measure of the general physical condition, we placed mussels exposed to copper and cadmium in cages in the sea and followed their growth for a year. The mussel shells were screened at the end of the year, to detect possible deformities.

The copper and cadmium concentrations of the water of the Baltic Sea are high in comparison with the levels in marine conditions: the copper concentrations vary between 5 and 15 nmol and the cadmium between 0.3 and 0.6 nmol dm<sup>-3</sup> (Brüg-

mann 1981). Accumulation of heavy metals by *Mytilus edulis* is greater in areas of low salinity than in areas of high salinity (Phillips 1977). In the Tvärminne area, on the southern coast of Finland, the copper concentrations of *Mytilus edulis* vary from 1.5 to 2.5 µg g<sup>-1</sup> and the cadmium content from 0.3 to 2.9 µg g<sup>-1</sup> (fresh weight) (Luotamo and Luotamo 1977). This poster paper is an abridged version of the article by Sunila and Lindström (1985).

### 2. MATERIAL AND METHODS

A sample of *Mytilus* was collected from a mussel field (s 7 ‰, depth 6 m) near the Tvärminne

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Zoological Station, on the southern coast of Finland. The maximum lengths of the mussels were measured and they were placed in cages, 110 specimens of each of two size classes, 2.0–3.0 cm and 0.5–1.5 cm, in each cage. The cages were constructed of nets, and contained a board on to which the mussels could fasten. For 24 hours, the mussels were exposed in their cages to 0.1, 0.2 and 0.4 mg l<sup>-1</sup> of copper ( $\text{CuSO}_4 \cdot 5 \text{H}_2\text{O}$  analytical grade) and 1.25, 2.5 and 5.0 mg l<sup>-1</sup> of cadmium ( $3 \text{CdSO}_4 \cdot 8 \text{H}_2\text{O}$  analytical grade, 7‰, 16 °C). The cages were then placed in the sea and anchored to the seabed, 0.7 m above the bottom at a depth of 3.5 m at the Tvärminne Zoological Station.

After three weeks, dead mussels were removed and one year later, in August, the cages were removed. The mussels that were still alive were measured and their shells screened for deformities. A new sample was collected later from the sampling station, to search for the observed shell deformities in a natural population.

### 3. RESULTS

#### 3.1 Survival rates

Three weeks after exposure, the LC 50 values were 0.32 mg l<sup>-1</sup> for copper and 2.5 mg l<sup>-1</sup> for cadmium. In the control cages the mortality during the year was 28 %. The proportion of dead mussels in the different concentrations is illustrated in Fig. 1.

#### 3.2 Growth

The average growth in the control cages was 0.6 cm a<sup>-1</sup>. Since the small specimens grew faster than the large ones, the two size classes overlapped during the year, which resulted in a bimodal size distribution. According to Fig. 2, copper and cadmium retarded growth.

#### 3.3 Shell deformities

Morphological changes were observed in the valves of the mussels grown in the cages. The deformities, always occurring in both valves, were located in the last growth zone and must thus have

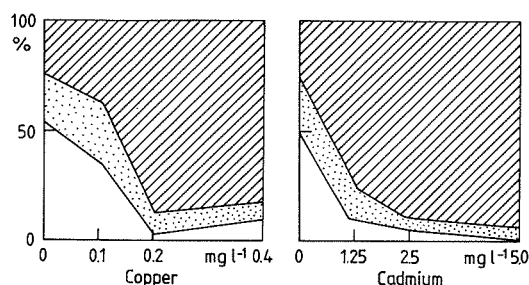


Fig. 1. The percentages of dead (diagonal), deformed (hatched) and undamaged (white) mussels one year after exposure to different copper and cadmium concentrations.

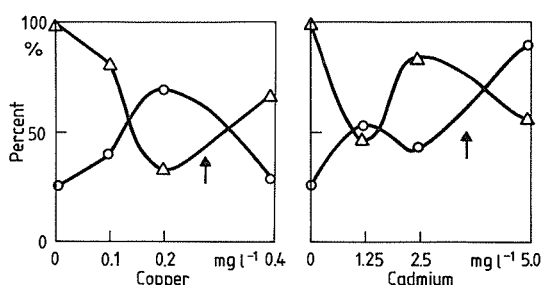


Fig. 2. Growth of mussels (Δ—Δ) and proportion of deformities (O—O) one year after exposure to copper and cadmium. Average growth rate in percent of control and proportions of deformed mussels in percent of survivors plotted against concentrations. Arrows indicate concentrations causing permanent closing due to the exposure.

been induced by the treatment.

The following types of deformity were observed (Fig. 3): I A posterior score, appearing in the slightest cases in the form of a stroke and in the most severe cases as a deep hollow. II A ventral score situated near the byssus cleft. III A dorsal score on the ligament side. IV A change in the direction of growth, resulting in the valves growing straight toward each other. The new growth zone was often soft, due to incomplete deposition of the mineral contents of the shell. V A change in the direction of growth from inward toward the other valve to parallel with the other valve. VI Humpy growth in the last growth zone, subsequently returning to normal. VII A yellow streak following the pattern of a growth ring.

While all these types of deformity were also observed in the control cages, their proportion was higher in the exposed specimens (Fig. 4). Growth and the proportions of the deformities were inversely related as functions of the concentrations (Fig. 2).

#### 4. DISCUSSION

The period of acute mortality (LC 50 values) seems to indicate the level of toxicity accurately enough for monitoring purposes. The LC 50 values reported by different authors vary between 0.2 and 0.3 mg l<sup>-1</sup> for copper (e.g. Martin et al. 1975; Abel 1976), and 1.5 and 2.5 mg l<sup>-1</sup> for cadmium (Martin et al. 1975).

The growth of *Mytilus* in the brackish water of the Baltic Sea has been described in detail by Kautsky (1982). Although the growth rate was reduced in every concentration tested in this study, no correlation between the concentration and the growth rate could be found. The concentrations in which the growth rate decreased most were those in which the highest percentage of deformities developed (Fig. 2). Thus, injury can be considered to retard growth.

The injuries at low concentrations revealed in these long-term tests seem to be due to a closing reaction. The concentrations causing permanent closing during the exposure period are between 0.2

and 0.4 mg l<sup>-1</sup> for copper and between 1.25 and 2.5 mg l<sup>-1</sup> for cadmium (arrows in Fig. 2). Such a closing reaction has been described, for example, by Davenport and Manley (1978). Strömberg (1982), who used laser diffraction to study the short-term effects of continuously applied heavy metals on the length growth of *Mytilus*, reported

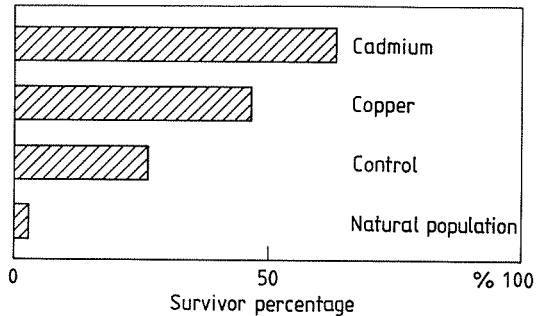


Fig. 4. The proportions of deformed mussels in a natural population and in test groups (control, copper and cadmium) in percent of survivors one year after exposure.

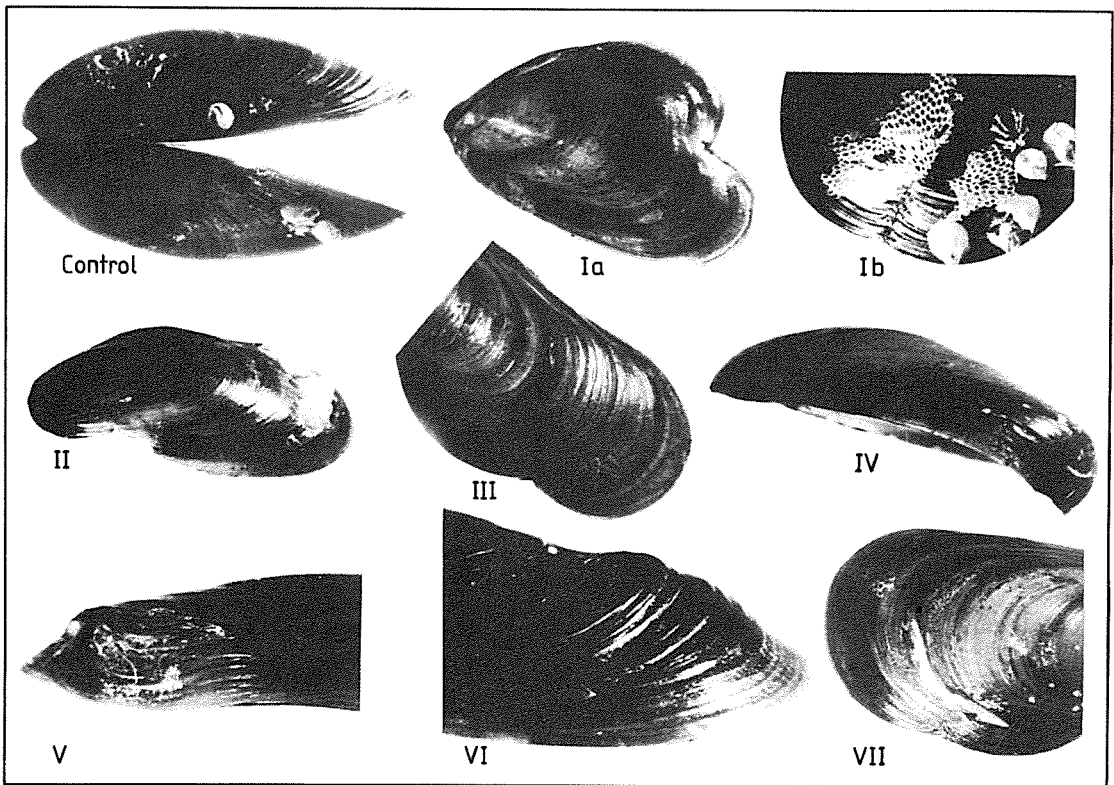


Fig. 3. Classification of deformity types. Control (6 ×). I. Posterior score. a. severe (5 ×) b. slight (7 ×). II. Ventral score (5 ×). III. Dorsal score (6 ×). IV. Flattening of posterior parts (6 ×). V. Thin-pointed posterior parts (7 ×). VI. Humpy shell (12 ×). VII. Streak (6 ×).

that copper and cadmium retarded growth.

None of the types of deformity observed in this study was clearly characteristic of either of the heavy metals. The increase of the proportion of deformities after exposure seems to be due to a general disturbance of shell formation rather than to any particular effect.

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## SUMMARY OF THE GULF OF BOTHNIA SEMINAR IN PORI 1984

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FONSELIUS, S.H. 1986. Summary of the Gulf of Bothnia Seminar in Pori 1984. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68.

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A summary is given of the results presented in papers on the hydrography, metals, organic toxic substances, material balances and effects of pollution. Some conclusions are drawn from these results and suggestions for future work are given.

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Index words: Hydrography, metals, organic substances, toxic matter, eutrophication, material balance, water balance, pollution effects, future work

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### 1. INTRODUCTION

The papers presented at the seminar dealt with many different aspects of the state of the Gulf of Bothnia and the pollution problems in those waters. They show that extensive work has been carried out in most fields. The following text will summarize the most important results.

### 2. HYDROGRAPHY

The search for long-term trends in the hydrographic data is continuing and interesting results have been

obtained. It has been found that the salinity of the Gulf of Bothnia has undoubtedly increased during the time measurements have been carried out, that is since the end of the 1800s (Fonselius 1986). The most interesting observation is that there are signs of a decreasing trend in the deep water during recent years. We do not yet know whether this trend will soon change direction again or become permanent.

The nutrients have also increased and it is tempting to try to find a relation between the increase in salinity and in nutrient concentrations, but this is difficult to confirm statistically. Even if we obtain a good correlation between these factors, the relation remains doubtful, due to the great supply of nutrients from land and from the atmosphere. The relation may be a coincidence. To obtain a real understanding of these problems, we

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must know more about the water exchange between the Baltic Proper and the Gulf of Bothnia and between the different basins in the Gulf of Bothnia. Promising results have been presented during the seminar, but many problems remain and more results are needed for better water and material balance calculations. Ulf Ehlin presented examples of models for the Gulf of Bothnia, with which horizontal and vertical circulation in the surface water can be computed. The models can also be used for computing the spreading of pollutants. Satellite pictures can be used as background material.

### 3. METALS

Monitoring of metals is in progress on both the Finnish and the Swedish sides of the Gulf and we have been given an account of the results. They show that the iron concentration has decreased slightly since 1973, but this trend is difficult to explain.

For other metals no trends have been established. Among the reasons for this are the uncertainty of earlier analytical techniques, the relatively short time series and the scarcity of data. Close to industrial discharge points higher concentrations have been found, but it is difficult to show that these discharges really have influenced the whole water mass.

We do not know enough regarding accumulation in organisms, flocculation, different chemical forms, exchange with the sediments, etc. Metal analyses are difficult to perform, and the risk of contamination during sampling and analysis is great. Very few researchers work with metal analysis in the open sea area in the Gulf of Bothnia. More scientists have undertaken investigations of metal discharges from industries and mining work, and the spreading of these effluents.

Unfortunately, not all major discharges are investigated and coordination of sampling, analytical techniques and choice of parameters is lacking. Intercalibration and standardization of working techniques are needed. All major discharges should be investigated in a uniform way, in order to give us a composite picture of the amounts of metal compounds reaching the open sea and the amounts sedimented in the neighbourhood of the discharge area. Unfortunately this kind of research is expensive and it is difficult to find sufficient funds for studies. In addition it has to be

remembered that metal compounds enter the Gulf of Bothnia with the water which flows in from the Baltic Proper and with atmospheric fallout.

Radioactive metal compounds also enter the Gulf of Bothnia, partly from the atmosphere and partly from leakages at nuclear power plants. Studies of such compounds have been presented during the seminar. It is encouraging to see that our power plants do not appear to have increased the load of radioactive strontium, caesium or plutonium.

### 4. ORGANIC TOXIC SUBSTANCES

It is even more difficult to obtain a picture of the discharge of organic compounds. Countless different compounds exist and many of them are still unknown. Even if the concentrations in certain cases may be high, e.g. for humus and lignin compounds, the analytical techniques in these cases are uncertain. In many other cases trouble in analyses is caused by the low concentrations. Rappe (1986) presented new facts about chlorinated dioxines and furanes, explaining that hundreds of such compounds exist and that some of them are extremely poisonous. Very little is known about this completely new field. The concentrations in the Gulf of Bothnia are extremely low, in some cases the total amount of the substance is of the order of magnitude of few lumps of sugar dissolved in the whole water mass of the Gulf of Bothnia. Such low concentrations can be detected analytically and although they are not really alarming, we must watch out for problems. The greatest difficulty at present is the high cost of instruments and analysis. According to Perttilä and Haahti (1986), DDT and PCB compounds are decreasing in the sediments, but a new threat is posed by the increase of chlordanes and toxaphenes.

Decomposition and oxidation consume oxygen and therefore the changes in oxygen saturation in the deep water and the BOD values are of great interest. Both urban and industrial oxygen-consuming discharges must be taken into account. Of the industrial discharges, the most important are the effluents from the forest industry. Wood-processing plants have existed since the middle of the 1800's and the discharges have been large. We have been able to change the trend, however, and the discharges are decreasing. In the summary from the Pollution Load Group (Enckell-Sarkola 1986) it was said that the maximum of industrial

discharges was reached in the 1960's. Since then the discharges have decreased greatly, due to closing of old plants, introduction of recirculation of wastes and modernization of techniques, and also in many cases owing to introduction of biological treatment of the waste water. This is not yet enough, but we are on the right road.

## 5. MATERIAL BALANCES

For all questions, computations of material balances are of great importance. Material balance models give us an idea of where to look for the problems. Water balance models are the foundation for material balance models. Without these transports of e.g. harmful and toxic substances cannot be calculated. During earlier seminars attempts at such modelling have been described. During this seminar no large-scale water or material balance models have been presented and this is scarcely surprising. With the existing data, further progress is hardly possible; we have inter alia to wait for more accurate results from the work on the water exchange through the Åland Sea and especially through the Archipelago Sea. Without considerable expenditure, it is unlikely that we will be able greatly to improve the water and salinity balance models in the near future. Moreover, all material balance models contain so many uncertain factors, that it will not make much difference if we increase the accuracy of the water balance models by some percentage units. It is important to continue material balance studies, however, and to try to obtain more exact values, especially for the sediments, which are normally the weakest link in the model. Smaller models for estuaries, etc., are of course important pieces in the larger jig-saw puzzle. The exchange between coastal waters and the open sea is also of a very great interest and we have been given an account of such work. Of still greater interest is sedimentation. As I mentioned above, sedimentation is often shown as a remainder, calculated as the difference between material added and removed. During the Second Seminar on the Gulf of Bothnia (Luleå, Sweden, June 16—17, 1981) more reliable figures, obtained from real measurements, were presented. These agreed quite well with the values computed earlier, but too much uncertainty still exists and more work is needed. What is chiefly required is computations of total amounts and measurements at transport and deposit bottoms, for both total

sediments and different sediment parameters, e.g. nutrients, trace metals and organic compounds.

Two important factors may influence the material balance, the acidification of the precipitation, which may affect the transport of material from land, and the regulation of river discharge, which may influence the spring flood. Large water reservoirs retain a great part of suspended matter, which settles in the reservoir, and since the spring flood has been reduced, material will not be transported as far out to sea as before.

## 6. POLLUTION EFFECT STUDIES

The most important problem in the Gulf of Bothnia is the effects of the pollution load. All the other studies provide a foundation for studies of these effects. Although the water is not used for drinking, fish living in it are eaten by man. The fish are a part of the food chain and accumulate toxic matter in their organs or are injured in different ways by poisons. Therefore the monitoring of biological parameters is of the utmost importance. But biological analyses are even more difficult to carry out than chemical analyses of the water. The methods are still less accurate and the distribution of organisms is patchy. Thorsten Ahl pointed out the difficulty in biological investigations of obtaining normal values. Both chemists and physical oceanographers can base their work on long series of observations and compute the deviation from the mean values for such series. A lively discussion of these problems arose among the biologists.

A series of studies on biological effects was presented by Andersin et al. (1986). The composition of phytoplankton in the open sea has been investigated since 1969, partly in collaboration between Finnish and Swedish scientists. Changes in the phytoplankton composition and biomass have been shown in the Archipelago Sea. Algae have been used as indicators of heavy metal pollution. The effects of industrial discharges have been studied outside industry plants and river mouths. The effects of increasing nutrient concentrations on e.g. *Cladophora glomerata* have been studied, and the changes in its nutrient composition have been analysed. Several studies have been presented on the effects of discharges from the forest industry on the marine fauna. Discharges of metals and organic compounds have been found to cause spinal defects in e.g. the fourhorn sculpin.

Physiological changes in the blood composition of perch have also been reported. Other established effects are changes in the shells of *Mytilus* and changes in the composition of the zoobenthos. It is very encouraging that a good correlation has been found between the decreasing industrial discharges and the increasing macrobenthos in the Ljusne area.

## 7. CONCLUSION

All results point to the conclusion that the large discharges have not yet caused any serious damage to the Gulf of Bothnia. Local deleterious effects have been found, but out in the open sea the condition of the Gulf of Bothnia seems to be satisfactory except for the situation of the seals. We have noted a certain "oceanization" of the conditions, due to the fact that the salinity has slightly increased. A certain eutrophication of the water is also taking place as the salinity increases. A major reason for this seems to be that the nutrient concentrations in the Baltic Proper are increasing and that due to the good water exchange the nutrient concentrations in the Gulf of Bothnia are also rising. But we must not forget that the supplies of nutrients from the land and the atmosphere are of the same order of magnitude as the addition from the Baltic Proper. Eutrophication should cause an increase in the plankton production and therefore also in the fish population. This should be good for the fisheries, but here the question arises. Which is more important, a bigger fish yield or clearer water for recreation? Input from land causes heavier eutrophication in coastal areas than in the open sea. A certain eutrophication may have to be accepted, but the limit of hypertrophication in coastal areas must not be passed. This would damage the ecological conditions.

## 8. SUGGESTIONS

It was suggested by Ulf Ehlin that the Committee for the Gulf of Bothnia should plan a Gulf of Bothnia Year and try to coordinate all investigations. The suggestion awoke a lively discussion and received warm support. It was recommended that

the executive board should begin planning such a year, to take place around 1987.

All the investigations mentioned have contributed to elucidate the problem of pollution in the Gulf of Bothnia. Professor Gunnar Kullenberg (1983) summarized the results from the Luleå Seminar and made suggestions regarding continuing investigations and the problems to be given priority. All that he said is still valid and I see no reason to repeat it. What I would really like to see, is an interdisciplinary investigation covering the whole problem complex, or at least the most important parts of it. This should be done by one or two persons as full-time work e.g. as a Ph D thesis. We need scientists to devote all their time to the Gulf of Bothnia and to penetrate all the aspects of the problems, that is, to put the jig-saw puzzle together. Why not a Finnish-Swedish doctor's thesis, written jointly by a biologist and a physical oceanographer or chemist?

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