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ACADEMIC DISSERTATION
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# Academic dissertation

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# Preface

This is a compilation and summary of results yielded in monitoring programmes and environmental studies carried out during more than 40 years in the sea areas surrounding the two Finnish nuclear power stations at Loviisa and Olkiluoto. The history of the studies covers the whole life span of nuclear power in Finland. Background studies were started at Loviisa in 1966, i.e., more than ten years before the first power plant unit came into operation, and five years before construction work was initiated on the island of Hästholmen. At Olkiluoto, the studies were started in 1972, before a single tree was cut down on the site. I personally took part in the first sampling expedition to the Hästholmen Island as a summer assistant and was responsible for setting in motion the field works at Olkiluoto.

During the long history of the studies, an enormous quantity of results has accrued. The long time-series of various hydrographical, biological and radioecological parameters collected in the two sea areas are unique and valuable not only concerning these specific areas, but also more widely, relating to the marine environment of the Finnish coast and even the whole northern Baltic Sea. The results of the monitoring programmes have been published either in Finnish or in English in the customary Annual Reports, but these results or those of the separate special studies have only sporadically been published in peer-reviewed journals. In the context of the latest international evaluation of the research activities of STUK (2005), the evaluation panel recommended that STUK should ensure that all data of general interest and relevance should be published by the laboratories themselves or in collaboration with other STUK units or universities.

A response to the recommendation was that it is better that the results are published by an expert or experts that have been closely involved in the work than to submit the material to external experts, e.g., in universities. At the worst, it could lead to the drawing of wrong conclusions. This gave me the impulse to write an extensive summary of the studies carried out at Loviisa and Olkiluoto during four decades. It would also be my legacy to posterity before retiring. The idea was supported by the facts that a decision was made at STUK to conclude the hydrographical and biological monitoring programme at Loviisa, and that my younger colleague Jukka Mattila decided to move onto new business. Earlier in my career I had not had time to write a doctoral thesis due to intensive field work over tens of years and later due to administrative tasks. Thus, why not do it now, when I had at last an opportunity to concentrate on it.

My strongest motive has been to get all the relevant results published and assessed. In fact, my feeling is that now is just the right time to summarize

the results, because some years ago many trends were still unclear. However, the processing of the huge material was a voluminous task. Although most of the results of the environmental radiation monitoring programmes and the hydrographical and biological monitoring programmes were saved in electronic form, much of the old data (esp. of separate special studies) were still only in paper form, and these had to be saved. The old databases had to be put in order, compiled and in some cases corrected. A great effort was made to dredge up all relevant data from desk drawers, and as a result most of them are now saved in the STUK databases. Most of the hydrographical monitoring results from Loviisa are also saved in the Water Quality Register of the Uusimaa Regional Environment Centre.

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**Keywords:** Nuclear power plants, thermal discharges, radioactive discharges, environmental effects, radioecological effects, aquatic environment, Baltic Sea, Gulf of Finland, Bothnian Sea

# Abstract

During recent decades, thermal and radioactive discharges from nuclear power plants into the aquatic environment have become the subject of lively debate as an ecological concern. Recently, an increasing demand for facts has appeared in context with the Environmental Impact Assessment procedures that are being in progress for planned new nuclear power units in Finland. The target of this thesis was to summarize the large quantity of results obtained in extensive monitoring programmes and studies carried out in recipient sea areas off the Finnish nuclear power plants at Loviisa and Olkiluoto during more than four decades. Especially in the conditions specific for the northern Baltic Sea, where biota is poor and adapted to relatively low temperatures and to seasonal variation with a cold ice winter and a temperate summer, an increase in temperature may cause increased environmental stress to the organisms. Furthermore, owing to the brackish-water character of the Baltic Sea, many organisms live there near the limit of their physiological tolerance. On the other hand, the low salinity increases the uptake of certain radionuclides by many organisms in comparison with oceanic conditions. The sea areas surrounding the Finnish nuclear power plants differ from each other in many respects (efficiency of water exchange, levels of nutrients and other water quality parameters, water salinity and consequent differences in species composition, abundance and vitality of biota). In addition, there are differences in the discharge quantities and discharge design of the power plants. In this thesis the environmental effects of the two power plants on the water recipients are compared and their relative significance is assessed.

There are four nuclear power plant units in Finland: two 488 $MW_e$ units at Loviisa, on the south coast, and two 840 $MW_e$ units at Olkiluoto, on the west coast. The units at Loviisa were commissioned in 1977 and 1980, and those at Olkiluoto in 1978 and 1980. Environmental studies were initiated at Loviisa about ten years, and at Olkiluoto six years before the start of operation of the power plants. Thus, 40-year-long time-series of results are available from the

hydrographical, biological and radioecological studies carried out for monitoring the environmental effects of the thermal and radioactive discharges from the power plants in the recipient sea areas.

Brackish water from the Baltic Sea is used for cooling in the Finnish nuclear power plants, and both the thermal and liquid radioactive discharges are let out into the sea. Each of the power plants use cooling water at a rate of about 40–60 $m^3 s^{-1}$, and the temperature rises in the condensers by about 10–13ºC. Loviisa NPP is located on the coast of the Gulf of Finland and Olkiluoto NPP on that of the Bothnian Sea. The state of the Gulf of Finland is clearly more eutrophic: the nutrient (total phosphorus and total nutrient) concentrations in the seawater are about 1½–2 times higher at Loviisa than at Olkiluoto, but the total phosphorus concentrations have still increased in both areas, even doubling at Loviisa between the early 1970s and 2000. The salinity is generally low in the brackish-water conditions of the northern Baltic Sea. However, the salinity of surface water is about 1‰ higher at Olkiluoto than at Loviisa (varying at the latter from near to 0‰ in early spring to 4–6‰ in late autumn). Thus, many marine and fresh-water organisms live in the Loviisa area close to their limit of existence, which makes the biota sensitive to any additional stress. The characteristics of the discharge areas of the two sites differ essentially from each other in many respects: the discharge area at Loviisa is a semi-enclosed bay in the inner archipelago, where the exchange of water is limited, whereas the discharge area at Olkiluoto is more open, and the exchange of water with the open Bothnian Sea is more effective.

The effects of the cooling water on the temperatures in the sea were most obvious in winter, when the conditions also most fundamentally differed from those of the natural state. Thermal discharges have significantly affected the ice conditions in the vicinity of the power plants. The formation of a permanent ice cover in the discharge areas has been delayed in early winter. On the other hand, the break-up of the ice occurs earlier in springs so that the growing season has been prolonged at both ends. From the biological point of view, the prolonging of the growing season and the disturbance of the overwintering time, in conditions where the biota has adjusted to a distinct rest period in winter, have been the most significant effects of the thermal pollution. Aquatic organisms in the northern Baltic Sea are acclimatized to a distinct annual winter period. The shortening of the ice winter or a total lack of ice cover has led to a blurring of the limits between the growing season and the winter season.

A temperature rise generally increases the metabolic activity and growth rate of aquatic organisms. This means an increased production of organic matter, and thus, thermal pollution promotes the eutrophication process in eutrophied environments. The raised temperature also increases the rate of decomposition

of organic matter in the receiving water bodies and leads to depletion of oxygen in deep water layers.

The hydrographical and biological results in the Loviisa area indicated a clearly higher level of eutrophy, which was based on the state of the whole Gulf of Finland. Thus, it was a challenge to distinguish the local effects of thermal discharges from the general eutrophication process of the Gulf of Finland. During the past 40 years the soft-bottom macrofauna has steeply deteriorated at many sampling stations, at some to the point of almost complete disappearance. A similar decline of the macrozoobenthos has been reported over the whole eastern Gulf of Finland. However, the local eutrophication process seems to have contributed to the decline of the bottom fauna in Hästholmsfjärden at Loviisa. Thermal discharges have increased the production of organic matter, which again has led to more organic bottom deposits unfavourable for benthic animals. Furthermore, these have increased the affinity of the isolated deeps for a depletion of oxygen, which has in turn caused a strong remobilization of phosphorus from the bottom sediments to the water phase. Phytoplankton primary production and primary production capacity doubled in the whole area between the late 1960s and the late 1990s, but started to decrease a little at the beginning of this century. The focus of the production shifted from spring to mid- and late summer. The general rise in the level of primary production was mainly due to the increase in nutrient concentrations over the whole Gulf of Finland, but the thermal discharge contributed to a stronger increase of production in the discharge area compared to that in the intake area of the cooling water. The eutrophication of littoral vegetation in the discharge area has been the most obvious, unambiguous and significant biological effect of the heated water. Spiked water milfoil *Myriophyllum spicatum,* the pondweeds *Potamogeton perfoliatus* and *Potamogeton pectinatus* and the vigorous growths of numerous filamentous algae as their epiphytes have strongly increased in the vicinity of the cooling water outlet, where they have formed dense populations in the littoral zone in late summer. However, the strongest increase of phytobenthos has extended only to a distance of about 1 km from the outlet, i.e., the changes in vegetation have been largest in those areas that remain ice-free in winter. A weaker eutrophication of littoral vegetation appeared in the whole area of Hästholmsfjärden Bay, but outside this area the phenomenon was slighter and observed only locally.

At Olkiluoto, the studies focusing on the effects of warm water discharge were more concise. Similar trends to those noticed in the Loviisa area regarding to increasing eutrophication were also discernible at Olkiluoto, but to a clearly smaller extent; this was due to the clearly weaker level of background eutrophy and nutrient concentrations in the Bothnian Sea, and to the local hydrographical

and biological factors prevailing in the Olkiluoto area. The level of primary production has also increased at Olkiluoto, but has remained at a clearly lower level than at Loviisa. In spite of the analogous changes observed in the macrozoobenthos, the benthic fauna has remained strong and diversified in the Olkiluoto area.

Radioactive discharges into the sea from the Finnish NPPs have been on average below 10% of the statutory limits. The discharged amounts of tritium were the most abundant, but those of other discharge nuclides were only a few percent of the limits, and still significantly decreased during recent years (during the last ten years to below 0.5%). Small amounts of local discharge nuclides were regularly detected in environmental samples taken from the discharge areas: tritium in seawater samples, and activation products, such as $^{60}$Co, $^{58}$Co, $^{54}$Mn, $^{110m}$Ag, $^{51}$Cr, among others, in suspended particulate matter, bottom sediments and in several indicator organisms (e.g., periphyton and the bladder-wrack *Fucus vesiculosus*) that effectively accumulate radioactive substances from the medium. The tritium discharges and the consequent detection frequency and concentrations of tritium in seawater were higher at Loviisa, but the concentrations of the activation products were higher at Olkiluoto, where traces of local discharge nuclides were also observed over a clearly wider area, due to the better exchange of water than at Loviisa, where local discharge nuclides were detected outside the Hästholmsfjärden Bay only quite rarely and in small amounts. At the farthest, an insignificant trace amount (0.2 Bq kg$^{-1}$ dw.) of $^{60}$Co originating from Olkiluoto was detected in *Fucus* at a distance of 137 km from the power plant. Discharge nuclides from the local nuclear power plants were almost exclusively detected at the lower trophic levels of the ecosystems. Traces of local discharge nuclides were very seldom detected in fish, and even then only in very low quantities, but not at all in birds nor in the inner organs and reproductive products of fish and birds.

The best indicators for $^{60}$Co were periphyton, the spiked water milfoil *Myriophyllum spicatum* and the bladder-wrack *Fucus vesiculosus*, whereas the intake of, e.g., Chernobyl-originated $^{137}$Cs was highest in predatory fish, perch and pike. As a consequence of the reduced discharges, the concentrations of local discharge nuclides in the environment have decreased noticeably in recent years at both Loviisa and Olkiluoto. Radioactive substances that originated from the Chernobyl accident and weapons-tests fallout (e.g. $^{137}$Cs, $^{90}$Sr, $^{239,240}$Pu) were still being detected in the environmental samples; the concentrations of $^{137}$Cs and natural radionuclides (e.g., $^{40}$K, $^{7}$Be) were in general higher than those of the local discharge nuclides.

The radiation doses to the public caused by discharges of radioactive substances from the Finnish nuclear power plants were small. The dose limit

set for members of the public from the normal operation of Finnish nuclear power plants is 0.1 mSv $a^{-1}$. This is approximately 1/40 of the average radiation dose received by Finns from different sources during one year. During the whole operational history of the power plants, the effective dose commitments of the critical groups have been at their highest less than 4%, and during more recent years clearly below 1% of the set limit. In general, the minor doses of local origin to the critical groups have been due to liquid discharges of $^{60}Co$ and people's shore occupancy. The environmental risk caused by the ionizing contaminants discharged from the Loviisa and Olkiluoto power plants was negligible: the doses to organisms were far below the conservative screening level of 10 µGy $h^{-1}$. Although the concentrations in environmental samples, and above all, the discharge data appear as seemingly large values, the radiation doses caused by them to the population and to biota are very low, practically insignificant. The effects of the thermal discharges have been more significant, at least to the wildlife in the discharge areas of the cooling water, although the area of impact has been relatively small. The results show that the nutrient level and the exchange of water in the discharge area of a nuclear power plant are of crucial importance.

*ILUS Erkki. Ydinvoimalaitosten radioaktiivisten aineiden ja lämminvesipäästöjen ympäristövaikutukset pohjoisen Itämeren murtovesiolosuhteissa. STUK-A238. Helsinki 2009, 372 s. + liitteet 8 s.*

**Avainsanat:** Ydinvoimalaitokset, lämminvesipäästöt, radioaktiivisten aineiden päästöt, ympäristövaikutukset, radioekologiset vaikutukset, meriympäristö, Itämeri, Suomenlahti, Selkämeri

# Tiivistelmä

Ydinvoimalaitosten radioaktiivisten aineiden ja lämminvesipäästöjen ympäristövaikutukset ovat viime vuosikymmeninä nousseet vilkkaan keskustelun kohteeksi ekologisena huolenaiheena. Suomeen suunniteltujen, uusien ydinvoimalaitosyksikköjen ympäristövaikutusten arvioinnit ovat aivan viime vuosina lisänneet niihin kohdistunutta tiedon tarvetta. Tämän tutkimuksen päämääränä oli laatia laaja yhteenveto Loviisan ja Olkiluodon voimalaitosten meriympäristöissä yli 40 vuoden aikana suoritettujen vesibiologisten ja radioekologisten tutkimusten, vesistötarkkailun ja säteilytarkkailun tuloksista. Lämpötilan nousu lisää ympäristöstä eliöille aiheutuvaa stressiä erityisesti pohjoiselle Itämerelle tyypillisissä olosuhteissa, missä eliöstö on niukkaa ja sopeutunutta suhteellisen alhaisiin lämpötiloihin: kylmään talveen ja lauhkeaan kesään. Lisäksi monet lajit elävät pohjoisen Itämeren vähäsuolaisissa olosuhteissa fysiologisen sietokykynsä rajoilla, mutta toisaalta alhainen suolapitoisuus lisää eräiden radionuklidien kertymistä eliöihin valtameriolosuhteisiin verrattuna. Suomen ydinvoimalaitoksia ympäröivät merialueet eroavat toisistaan monessa suhteessa: vedenvaihdon tehokkuuden, veden ravinnepitoisuuksien ja useiden muiden vedenlaatuparametrien sekä veden suolapitoisuuden ja siihen liittyen eliöstön lajikoostumuksen, runsauden ja elinvoimaisuuden puolesta. Lisäksi eroa on voimalaitosten päästömäärissä ja jäähdytysveden purkutavoissa. Tässä tutkimuksessa vertaillaan voimalaitosten ympäristövaikutuksia ja arvioidaan niiden merkitystä.

Suomessa on toiminnassa neljä ydinvoimalaitosyksikköä: kaksi 488 $MW_e$:n yksikköä Loviisassa ja kaksi 840 $MW_e$:n yksikköä Olkiluodossa. Loviisan yksiköt käynnistyivät 1977 ja 1980 ja Olkiluodon 1978 ja 1980. Ympäristötutkimukset aloitettiin Loviisassa noin kymmenen vuotta ja Olkiluodossa noin kuusi vuotta ennen ensimmäisen voimalaitosyksikön käynnistämistä, tavoitteena luoda pohja tulevien radioaktiivisten aineiden ja lämminvesipäästöjen ympäristövaikutusten seurannalle ja arvioinnille vastaan-

ottavissa vesistöissä. Näin ollen käytettävissä on 40 vuoden aikasarjat hydrografisten, biologisten ja radioekologisten tutkimusten tuloksista kyseisillä alueilla.

Suomen ydinvoimalaitoksissa käytetään Itämeren murtovettä jäähdytysvetenä, ja sekä radioaktiivisten aineiden että lämminvesipäästöt puretaan mereen. Voimalaitokset käyttävät jäähdytysvettä 40–60 $m^3 s^{-1}$, ja veden lämpötila nousee lauhduttimissa noin 10–13 ºC. Loviisan voimalaitos sijaitsee Suomenlahden ja Olkiluodon voimalaitos Selkämeren rannalla. Suomenlahden tila on selvästi rehevöityneempi; veden ravinnepitoisuudet (kokonaisfosfori ja kokonaistyppi) ovat Loviisassa noin 1½–2 kertaa suurempia kuin Olkiluodossa. Pintaveden fosforipitoisuudet kasvoivat edelleen molemmilla alueilla, ja Loviisassa ne jopa kaksinkertaistuivat 1970-luvun alun ja vuoden 2000 välisenä aikana. Sen lisäksi pintaveden suolapitoisuus on Olkiluodossa noin 1 ‰:a korkeampi kuin Loviisassa, jossa monet mereiset lajit elävät levinneisyysalueensa rajoilla, mistä johtuen eliöstö on herkkää kaikille muille ympäristömuutoksille. Loviisassa pintaveden suolapitoisuudet vaihtelevat lähes nollasta aikaisin keväällä, 4–6 ‰:een myöhään syksyllä. Loviisan ja Olkiluodon voimalaitosten jäähdytysveden purkualueet eroavat olennaisesti toisistaan. Loviisan purkualue, Hästholmsfjärden, on puolisuljettu lahti sisäsaaristossa, jossa veden vaihtoa avoimen Suomenlahden kanssa rajoittavat saaret, kapeat salmet ja vedenalaiset kynnykset. Sen sijaan Olkiluodon alue on avoimempi ja veden vaihto avoimen Selkämeren kanssa on tehokkaampaa.

Jäähdytysveden vaikutukset meriveden lämpötiloihin olivat näkyvimmät talvella, jolloin olosuhteet myös selkeimmin poikkesivat luonnontilaisista. Lämminvesipäästöt ovat vaikuttaneet merkittävästi jääolosuhteisiin voimalaitosten lähialueilla. Pysyvän jääpeitteen muodostuminen on viivästynyt alkutalvisin jäähdytysveden purkualueilla. Toisaalta jäiden lähtö on aikaistunut keväällä, jolloin kasvukausi on pidentynyt molemmista päistään. Biologisesta näkökulmasta katsoen kasvukauden piteneminen ja talvehtimisajan häiriintyminen olosuhteissa, missä eliöstö on tottunut selvään lepovaiheeseen talvella, ovat olleet merkittävimmät lämpökuormituksen ympäristövaikutukset. Pohjoisen Itämeren eliöstö on sopeutunut selvään jokavuotiseen talvikauteen. Jäätalven lyheneminen tai jääpeitteen täydellinen puuttuminen on johtanut kasvukauden ja lepokauden rajojen hämärtymiseen.

Lämpötilan nousu lisää yleisesti aineenvaihdunnan vilkkautta ja vesieliöiden kasvua. Se merkitsee lisääntynyttä orgaanisen aineksen tuotantoa, ja siten lämpökuormitus edistää rehevöitymiskehitystä rehevöityneessä ympäristössä. Lämpö nopeuttaa myös orgaanisen aineksen hajotustoimintaa vastaanottavassa vesistössä ja johtaa sen seurauksena hapen niukkuuteen pohjanläheisissä vesikerroksissa.

Vesistötarkkailun tulokset osoittivat rehevöitymisen olevan Loviisan alueella selvästi pidemmällä kuin Olkiluodossa, mikä johtuu koko Suomenlahden ylikuormitetusta tilasta. Tämä vaikeuttaa myös lämpimän veden paikallisten vaikutusten erottamista Suomenlahden yleisestä rehevöitymiskehityksestä. Pohjaeläimistö on taantunut voimakkaasti useimmissa Loviisan alueen havaintopaikoissa viimeisten 40 vuoden aikana. Seurauksena on paikoin ollut jopa pohjafaunan lähes täydellinen häviäminen. Vastaavasta pohjaeläinyhteisöjen heikkenemisestä on raportoitu koko itäisen Suomenlahden alueelta. Paikallinen rehevöitymiskehitys näyttää kuitenkin edistäneen Hästholmsfjärdenin pohjaeläimistön taantumista Loviisassa. Lämminvesipäästöt ovat lisänneet orgaanisen aineksen tuotantoa, mikä on edelleen johtanut orgaanisen aineksen lisääntymiseen pohjasedimenteissä ja niiden muuttumiseen pohjafaunalle epäsuotuisiksi. Hajotettavan orgaanisen aineksen lisäys on edelleen lisännyt syvänteiden pohjanläheisen veden altistumista happikadoille, mikä on aiheuttanut etenkin fosforin voimakasta remobilisointia pohjasedimenteistä alusveteen. Kasviplanktonin perustuotanto ja perustuotantokyky kaksinkertaistuivat koko alueella 1960-luvun lopun ja 1990-luvun lopun välisenä aikana, mutta kääntyivät jonkin verran laskuun tämän vuosikymmenen alussa. Samalla tuotannon painopiste siirtyi keväästä keski- ja loppukesään. Perustuotantotason yleinen kasvu johtui ensisijaisesti ravinnepitoisuuksien kasvusta koko Suomenlahdessa, mutta lämminvesipäästöt aikaansaivat sen, että tuotanto kasvoi jonkin verran voimakkaammin jäähdytysveden purkualueella kuin sen ottoalueella. Vesikasvillisuuden rehevöityminen jäähdytysveden purkualueella on ollut näkyvin, merkittävin ja kiistattomin lämpimän jäähdytysveden biologinen vaikutus. Tähkä-ärviä, ahvenenvita ja hapsivita sekä useiden levälajien muodostamat rihmaleväkasvustot niiden päällyskasveina ovat voimakkaasti runsastuneet jäähdytysveden purkupaikan läheisyydessä, missä ne muodostivat tiheitä yhdyskuntia rantavyöhykkeeseen loppukesäisin. Vesikasvillisuuden voimakkain runsastuminen on kuitenkin rajoittunut vain noin 1 km etäisyydelle purkupaikasta, ts. muutokset ovat olleet suurimpia niillä alueilla, jotka ovat jääneet talvella ilman jääpeitettä. Vähäisempää kasvillisuuden rehevöitymistä esiintyi koko Hästholmsfjärdenin alueella, mutta sen ulkopuolella sitä esiintyi heikompana vain paikoin.

Olkiluodossa tehtiin lämpimän veden vaikutuksia selvitteleviä tutkimuksia suppeammalla ohjelmalla kuin Loviisassa. Samanlaista, rehevöitymistä osoittavaa kehitystä kuin Loviisassa oli nähtävissä myös Olkiluodossa, mutta selvästi vähäisemmässä määrin, mikä johtui Selkämeren selvästi alhaisemmasta ravinnetasosta ja rehevöitymiskehityksestä sekä Olkiluodon edustan merialueelle tyypillisistä hydrografisista ja biologisista tekijöistä, kuten hyvästä vedenvaihdosta ja elinvoimaisesta eliöstöstä. Kasviplanktonin perustuotanto

on lisääntynyt myös Olkiluodossa, mutta se on jäänyt selvästi alhaisemmalle tasolle kuin Loviisassa. Huolimatta pohjaeläimistössä havaituista muutoksista, se on säilynyt elinvoimaisena ja monimuotoisena Olkiluodossa.

Suomen ydinvoimalaitosten radioaktiivisten aineiden päästöt mereen ovat olleet selvästi asetettujen päästörajojen alapuolella (keskimäärin alle 10 %). Tritiumin päästöt ovat olleet määrällisesti suurimpia, mutta muiden nuklidien päästöt ovat olleet vain muutamia prosentteja (viimeisten kymmenen vuoden aikana alle 0,5 %) päästörajoista, ja ne ovat viime vuosien aikana merkittävästi pienentyneet. Pieniä määriä paikallisia päästönuklideja havaittiin säännöllisesti jäähdytysveden purkualueilta otetuissa ympäristönäytteissä: tritiumia merivedessä ja aktivoitumistuotteita (kuten mm. koboltti-60:a, koboltti-58:a, mangaani-54:a, hopea-110$^{m}$:a ja kromi-51:a) sedimentoituvassa aineksessa, pohjasedimenteissä ja ns. indikaattoriorganismeissa (esim. perifyton ja rakkolevä), jotka keräävät tehokkaasti radioaktiivisia aineita ympäristöstä. Tritiumin päästöt, ja vastaavasti sen havaitseminen ja pitoisuudet merivesinäytteissä, olivat runsaampia Loviisassa, mutta aktivoitumistuotteiden pitoisuudet olivat suurempia Olkiluodossa, missä niitä havaittiin myös selvästi laajemmalla alueella. Purkualueen vedenvaihto ja radioaktiivisten päästöjen leviäminen on siellä tehokkaampaa kuin Loviisassa, missä paikallisia päästönuklideja havaittiin vain suhteellisen niukasti Hästholmsfjärdenin ulkopuolella. Merkityksettömän pieniä koboltti-60-pitoisuuksia (0,2 Bq kg$^{-1}$ kuivap.) havaittiin rakkolevässä etäisimmillään noin 137 kilometrin päässä Olkiluodon voimalaitoksesta. Paikallisia päästönuklideja havaittiin lähes yksinomaan ekosysteemin alimmilla trofiatasoilla. Kaloissa päästönuklideja tavattiin hyvin harvoin, ja silloinkin vain hyvin pieninä pitoisuuksina, mutta niitä ei tavattu lainkaan linnuissa eikä kalojen tai lintujen sisäelimissä tai lisääntymistuotteissa (mäti, maiti, munat, alkiot).

Parhaita koboltti-60:n indikaattoriorganismeja olivat perifyton, tähkäärviä ja rakkolevä, kun taas ahven ja hauki keräsivät tehokkaimmin Tshernobylin onnettomuudesta peräisin olevaa cesiumia. Voimalaitosten radioaktiivisten aineiden päästöjen vähentämisen ansiosta paikallisten päästönuklidien pitoisuudet pienenivät 1990- ja 2000-luvuilla merkittävästi ympäristönäytteissä sekä Loviisassa että Olkiluodossa. Tshernobylin onnettomuudesta ja 1950- ja 1960-lukujen ydinasekokeista peräisin olevia laskeumanuklideja (kuten cesium-137, strontium-90, plutonium-239,240) havaittiin edelleen ympäristönäytteissä, ja cesium-137:n sekä näytteissä havaittujen luonnon radionuklidien (kuten kalium-40 ja beryllium-7) pitoisuudet olivat yleensä suurempia kuin paikallisten päästönuklidien.

Suomen ydinvoimalaitosten radioaktiivisten aineiden päästöistä ympäristön asukkaille aiheutuvat säteilyannokset olivat erittäin pieniä. Voimalaitosten

normaalista käytöstä yksittäiselle ympäristön asukkaalle aiheutuvan säteilyannoksen raja-arvoksi on asetettu 0,1 mSv vuodessa. Tämä on noin neljäskymmenes osa keskimääräisestä säteilyannoksesta, jonka suomalaiset saavat eri lähteistä vuoden aikana. Voimalaitosten koko käyttöhistorian aikana altistuneimpaan väestöryhmään (ns. kriittinen ryhmä) kuuluvan henkilön säteilyannokset ovat olleet korkeimmillaan alle 4 %, ja viime vuosina selvästi alle 1 % asetetusta raja-arvosta. Kriittisellä ryhmällä tarkoitetaan tässä yhteydessä henkilöitä, jotka oleskelevat joko työssään tai vapaa-aikanaan paljon meren äärellä ja syövät runsaasti (päivittäin) paikallista kalaa. Samoin voimalaitosten radioaktiivisten aineiden päästöistä ympäristölle aiheutuva säteilyaltistus oli hyvin vähäistä: eliökunnan säteilyannokset olivat selvästi alle kansainvälisesti ympäristön säteilysuojelulle asetetun seulonta-arvon, 10 µGy tunnissa. Vaikka radioaktiivisten aineiden pitoisuudet ympäristönäytteissä, ja etenkin niiden päästömäärät tulevat esiin suurina numeroina, niiden väestölle ja luonnon eliöstölle aiheuttamat säteilyannokset ovat hyvin pieniä, käytännössä merkityksettömiä. Lämminvesipäästöjen vaikutukset ovat olleet merkittävämpiä erityisesti jäähdytysveden purkualueilla, vaikkakin vaikutusalue on ollut suhteellisen rajoittunut. Tämän tutkimuksen tulokset osoittavat, että jäähdytysveden purkualueen vedenvaihdolla ja ravinnepitoisuuksilla on ratkaiseva merkitys ympäristövaikutusten kannalta.

# Contents

## PART III
## SUMMARY, DISCUSSION AND CONCLUSIONS

# General introduction

During recent decades, the thermal and radioactive discharges from nuclear power plants into the aquatic environment have become the subject of lively debate as an ecological concern. Heat as a separate pollutant was first brought into the public eye in the UK a good 50 years ago; very few research programmes dealing specifically with the effects of thermal discharges were originated anywhere in the world before the early 1950s. By the mid-1960s there were many research projects concerned with thermal discharges in the UK, USA, USSR and Europe, and the term 'thermal pollution' was taken into general use. From 1960 to 1970 the literature concerned with pollution by heat grew to several hundred per year (Langford 1990). In 1974, the book 'Kylvatten – effekter på miljön' (Cooling water – effects on the environment) was published in Sweden (SNV 1974), and already in the 1960s discussion about the thermal effects of the planned nuclear power plants had started in Finland.

In 1974, the International Atomic Energy Agency (IAEA) organized a Symposium on 'Environmental Effects of Cooling Systems at Nuclear Power Plants' in Oslo (IAEA 1975a), and in 1975 a Symposium on 'Combined Effects of Radioactive, Chemical and Thermal Releases to the Environment' in Stockholm (IAEA 1975b). In 1980, the IAEA organized a Symposium on 'Impacts of Radioactive Releases into the Marine Environment' in Vienna (IAEA 1981), and since then the results of monitoring and studies carried out in the marine environments of nuclear power plants have been discussed in countless symposia and publications around the world.

Especially in the conditions specific to the northern Baltic Sea, where the biota is poor and adapted to relatively low temperatures and to seasonal variation with a cold ice-winter and a temperate summer, an increase of temperature may cause increased environmental stress to the organisms. Furthermore, owing to the brackish-water character of the Baltic Sea, many organisms exist near the limit of their physiological tolerance and have poor resistance to additional stresses (Dybern and Fonselius 1981). The effects of heated effluents are therefore of particular interest here. An increase of temperature stimulates the metabolism, increases the production of organic material and accelerates the growth of organisms; and consequently, it may lead to enhanced eutrophication (Autio et al. 1996). The distinction of the thermal effects from those caused by the increase of nutrients poses a challenge especially in the Gulf of Finland, where the levels of phosphorus and nitrogen have significantly increased during recent decades.

Linked with stimulated metabolism, a rise in temperature increases the uptake of several harmful substances (Grimås 1974) and most probably also

that of radionuclides in biota. The brackish-water character of the Baltic Sea, and especially the low salinity of water in the Finnish coastal areas, adds to the uptake of radionuclides, since the concentration factors of many radionuclides are much higher in low salinities than in real marine environments (*cf.* Hosseini et al. 2008). Even if the low salinity may make the life of many organisms of marine origin difficult, the warm water may on the other hand attract many immigrants to the discharge areas of cooling water. As another aspect, the effects of cooling water can be used to predict the possible biological changes in coastal waters caused, for example, by climatic change (Ilus and Keskitalo 2008).

There are four nuclear power plant (NPP) units in Finland: two pressurised water reactors at Loviisa (rated net electric power of each 488 MW), on the south coast, and two boiling water reactors at Olkiluoto (rated net electric power of each 840 MW) on the west coast of Finland (Fig. 1). The units at Loviisa were commissioned in 1977 and 1980, and those at Olkiluoto in 1978 and 1980. Brackish sea water is used for cooling in the Finnish nuclear power plants. When running at full capacity, the power stations discharge 50 – 60 $m^3 s^{-1}$ cooling water into the sea, the discharged water being 10 – 13ºC warmer than the intake water. Small planned and controlled discharges of radioactive substances (mainly of neutron activation products) are released into the recipient sea areas in the out-flowing cooling water. Extensive environmental monitoring and studies have been carried out in the sea areas surrounding the power stations since the Hästholmen and Olkiluoto islands were chosen as the sites for nuclear power plants in Finland. When the work was begun, the environmental effects of nuclear power were in general rather poorly known. In the course of more than 40 years, the extensive studies carried out have yielded a huge number of results to be utilized. During the last few years, the need for these results has increased with the adoption of the Environmental Impact Assessment procedures of the planned new nuclear power units in Finland.

The aim of this work was to compile the data and summarise the results in an extensive scientific publication, as a doctoral thesis and a legacy to posterity. It should be emphasised that the work is mainly based on monitoring results, and that these monitoring programmes were not possible to establish in the same way as a research plan in basic research. The radioecological special projects initiated followed this procedure better. All the results are not included in this publication. The phytoplankton results from Loviisa were earlier published elsewhere (Bagge and Niemi 1971, Ilus and Keskitalo 1987 and 2008) and were left out of this consideration. Fish and fishery research have never been included in the water quality and biological studies carried out by STUK, but have been conducted by consultants of the branch in question.

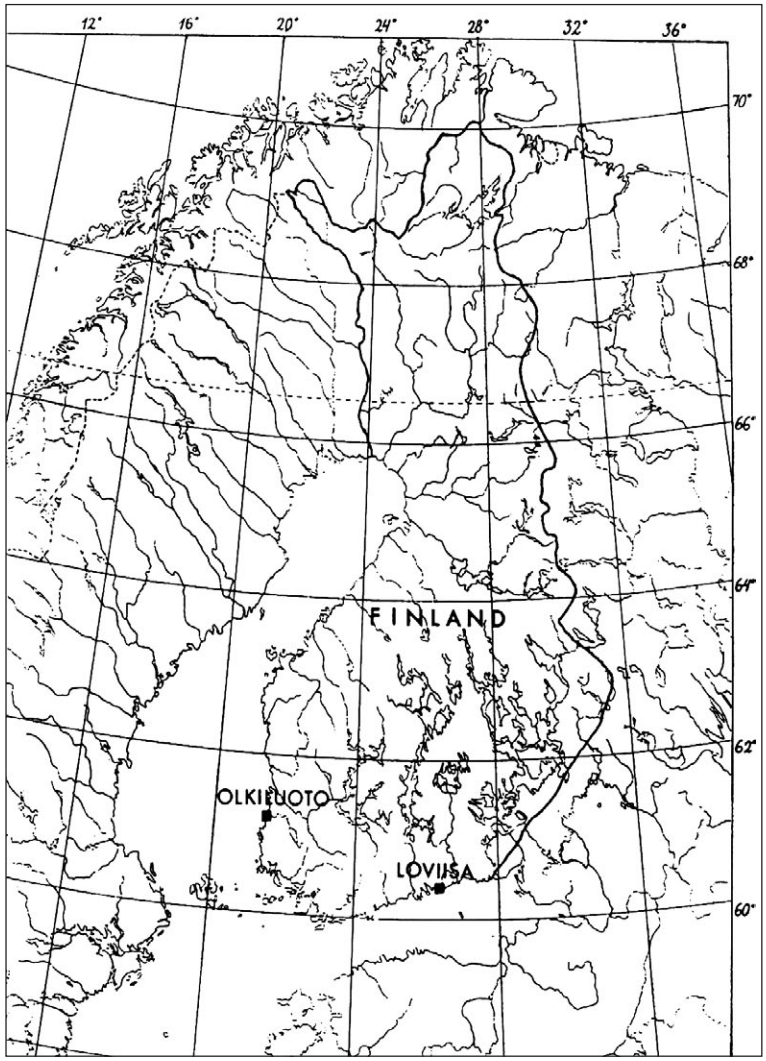


**Fig. 1.** Location of the Loviisa and Olkiluoto nuclear power stations.

# Part I

# Environmental effects of cooling water

# History of the hydrographical and ecological studies

Hydrographical and biological background studies were initiated at Loviisa and Olkiluoto in good time before the construction work on each of the power plants was started in order to establish a basis for monitoring potential environmental effects of future discharges of warm cooling water from the power plants into the sea in the northern temperate climate and the brackish-water conditions prevailing on the Finnish coast.

Hydrobiological baseline studies were started at Loviisa in 1966 by the Finnish Institute of Marine Research (FIMR) on the initiative and with the financing of the Ministry of Trade and Industry. In 1973, the implementation of the hydrobiological background studies was transferred from FIMR to the Institute of Radiation Physics, later the Radiation and Nuclear Safety Authority STUK. From 1976 onwards, the studies were carried out by STUK as a charged service to the Loviisa Nuclear Power Plant operated by Imatran Voima Oy, later Fortum Power and Heat Oy. At the same time, these studies began to be a part of the obligatory monitoring programme related to the water permit of the power plant. The construction work of the power plant was started in 1970 and the power plant units Loviisa 1 and Loviisa 2 were commissioned in February 1977 and November 1980. STUK was responsible for carrying out the obligatory monitoring programme of water quality at Loviisa continuously up to 2007. Consequently, the data set of a huge number of hydrographical and biological results constitutes an unbroken time-series more than 40 years long. The data consist of more than 130 000 observations from Loviisa and more than 30 000 from Olkiluoto.

The results have been reported annually to the power plant as bulky mimeographed Annual Reports in Finnish (Appendix 1), but only a few papers have been published in international scientific journals. The first scientific publication dealing explicitly with the results of the environmental studies carried out in the sea area surrounding the site of the Loviisa power plant was published by Bagge and Niemi (1971). In the 1970s, the Finnish Institute of Marine Research studied the water currents in the discharge area of the Loviisa power plant together with the temperature of the seawater and the energy exchange between the air and the sea surface by means of an oceanographic-meteorological mast constructed in the middle of Hästholmsfjärden Bay (Launiainen 1975, 1979, 1980). In addition, Imatran Voima Power Company and Fortum Power and Heat have commissioned several hydrographical and fishery surveys from other consultants.

At Olkiluoto, the hydrographical and biological baseline studies were initiated in 1972 with the aim of gaining information needed about the particular

ecological characteristics of the sea area. This was important because the water recipients at Loviisa and Olkiluoto differ essentially from each other in many hydrographical and biological features. The construction works at Olkiluoto were started in 1973 and the power plant units TVO I and TVO II were commissioned in September 1978 and February 1980. STUK was responsible for carrying out the obligatory monitoring programme related to the water permit of the Olkiluoto power plant up to 1983, but had then to relinquish the duty due to the lack of human resources. Since 1982, the obligatory monitoring has been executed by Water and Environment Research of Southwest Finland, and STUK has continued hydrobiological monitoring at Olkiluoto with its own concise programme up to 2007. As a part of the hydrobiological studies carried out by STUK at Olkiluoto, Dr. Jorma Keskitalo published his doctoral thesis on the "Effects of thermal discharges on the benthic vegetation and phytoplankton outside the Olkiluoto nuclear power station, west coast of Finland" in 1988 (Keskitalo, 1988).

# 1 Hydrographical and ecological studies at Loviisa

## 1.1 Study area

The Loviisa nuclear power plant is located on the island of Hästholmen on the north coast of the Gulf of Finland, about 12 km SSE of the town of Loviisa. Hästholmen is situated in the outer part of the inner archipelago; the open sea begins at Orrengrund Island, about 12 km south of Hästholmen. The sea area off Loviisa is characterized by successive water basins isolated from each other by shallow sounds and underwater sills. Therefore the water exchange of the basins with the open sea is limited. The exchange of water is especially bad in the deep areas of the basins below the sill depths (Bagge & Voipio 1967).

The power plant lies on the east coast of Hästholmen. The cooling water is taken from the west side of Hästholmen, Hudöfjärden, and discharged into Hästholmsfjärden situated east of Hästholmen (Fig. 2). The mouth of the intake channel is relatively deep: the upper edge is at a depth of 8.5 metres and the lower edge at 11.1 metres. Consequently, colder water is available for cooling and at the same time the drift of fish, macroalgae and ice into the intake channel is reduced. After passing through the condensers, the heated water is discharged into the sea over a curved 90-m wide embankment, which distributes the effluent over the surface of the receiving water body, thus promoting the heat transmission into the air. The temperature of the discharged water is about 8–12°C higher than that of the intake water. The flow of cooling water has been on average about 44 $m^3 s^{-1}$ (Fortum Power and Heat Oy, 1999 and 2008).

The discharge area of the heated cooling water, Hästholmsfjärden Bay, is a semi-enclosed basin between the mainland and the islands connected to the outer sea areas through narrow and shallow sounds to the south. The area of Hästholmsfjärden is about 9 $km^2$, the mean depth is 7.6 m and the water volume 68 500 000 $m^3$. The proportion of areas deeper than 10 m is about 28% and of those deeper than 15 m about 2% of the area. The deepest point of Hästholmsfjärden is about 18 m, in the southeast corner of the bay (Station 3, Fig. 3). Shallow underwater sills (sill depth about 8 m) isolate Hästholmsfjärden effectively from the currents and mixing processes of the surrounding sea area (Launiainen 1979). In the natural state, primarily only changes in seawater level caused currents and exchange of water in the sounds connecting with the open sea (Korhonen 1975, Launiainen 1975). Before the power plant came into operation, the natural theoretical retention time of the water in Hästholmsfjärden (incl. Klobbfjärden) was estimated as 50–60 days (Launiainen 1975).

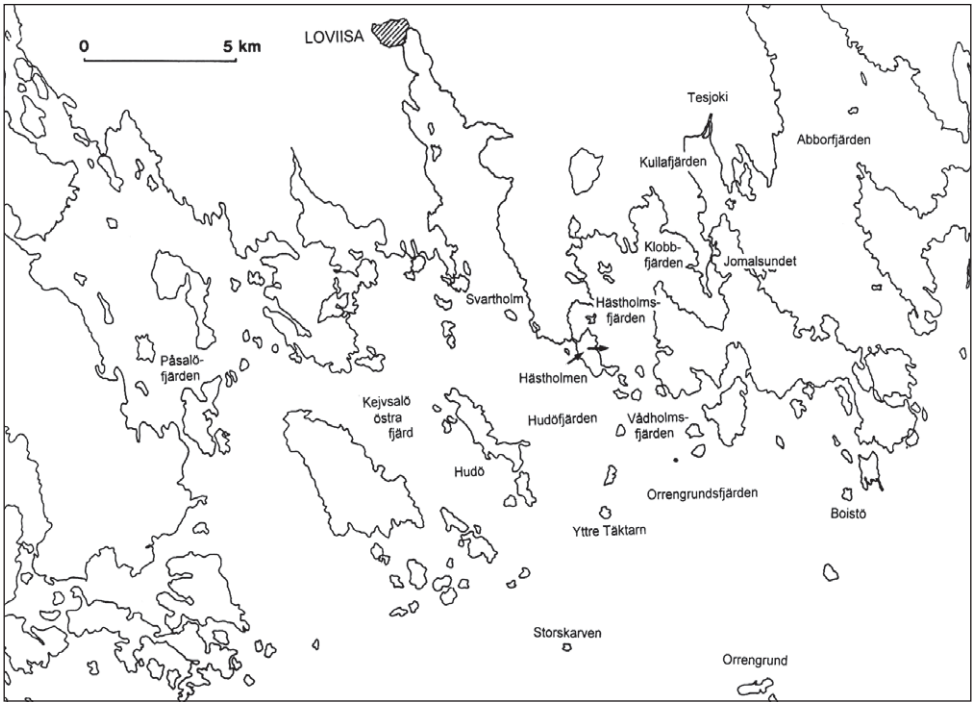


**Fig. 2.** The place-names in the sea areas surrounding the island of Hästholmen off Loviisa. The intake and outlet of the cooling water are indicated by arrows.

Klobbfjärden adjoins Hästholmsfjärden in the northeast. It is shallower than Hästholmsfjärden, the summed area of the two water bodies is 15.0 $km^2$, their mean depth 6.8 m and their summed volume 0.10 $km^3$ (Launiainen 1979). Water exchange between these two basins is limited by a shoal, which is broken only by a narrow 10 m deep channel. Klobbfjärden and Hästholmsfjärden are connected via Jomalsundet (in the northeast) to Kullafjärden and Abborfjärden, i.e., to the estuaries of the Tesjoki River and to that of the western fork of the Kymijoki River. Due to the river waters flowing through Jomalsundet into the area, Klobbfjärden and Hästholmsfjärden are naturally more limnic than the surrounding sea areas. The inflow of fresh water also strengthens the stratification and lengthens the stagnation periods of hypolimnion in Hästholmsfjärden. However, most importantly, the river waters are the most important factor affecting the load of nutrients and solid and oxygen-consuming matter into the whole sea area surrounding Hästholmen Island.

The intake area for the cooling water, Hudöfjärden, is clearly more marine than Hästholmsfjärden. Its water volume is larger and its connection with the open Gulf of Finland is more unrestrained. In spite of two threshold areas to

the south that limit the water exchange, the renewal of water is more rapid in Hudöfjärden than in Hästholmsfjärden (Bagge and Niemi 1971). However, below the sill depth the water masses are in a relatively resistant stagnation state, in the deep parts of the embayment (Bagge & Voipio 1967). The depth at the deepest point of Hudöfjärden is 24 m (Station 10, Fig. 3). The sill depth may be a little more than 10 m. Dredging of the fairway to Loviisa in the 1980s in the SW corner of Hudöfjärden may have slightly improved the exchangeability of water in the basin.

The near-bottom waters in Vådholmsfjärden and Orrengrundsfjärden, situated south of Hästholmsfjärden, seem to change more freely than those in the above basins, although they too are at least partly isolated by underwater sills and islands from the deeper parts of the Gulf of Finland. The depth at the deepest point of Vådholmsfjärden is 27 m, the sill depth being about 18 m. The southernmost permanent sampling station (Station 7, Fig. 3) is located in the northern part of Orrengrundsfjärden, where the depth is 33 m. Thus, the study area of the obligatory monitoring programme reaches to a distance of about 5 km from the power plant. In fact, however, the distance to the reference stations R1 and R2 in Påsalöfjärden and Kejvsalö östra fjärd (Figs. 3 and 63) is longer. At the deepest point of Orrengrundsfjärden, near Orrengrund Island, the depth is 66 m.

River waters also arrive in the sea area off Loviisa from the south via Orrengrundsfjärden and Vådholmsfjärden, as well as via Jomalsundet. Especially in early spring the salinity in the surface water is often lower at Stations 4 and 7 (Fig. 3), and the water is richer in nutrients than that in Hästholmsfjärden and Hudöfjärden. As the main flow direction is from east to west on the south coast of Finland, and the most important rivers causing nutrient load have their estuaries to the east of Loviisa, it is clear that nutrient-rich waters flow in from the east of Boistö, either from Abborfjärden or from a greater distance.

The salinity of the water is also low in other seasons in the sea area off Loviisa. The mean salinity in the surface water is 3.5–5‰ during the growing season, i.e., it is classified as alpha oligohaline. The biota consists of freshwater and brackish-water species. Owing to the low salinity, many marine species live there at their extreme limit of survival, e.g., the Common Mussel (*Mytilus edulis*), the Common Cockle (*Cerastoderma glaucum*) and some polychaetes. The eastern boundary line of their distribution range in the Gulf of Finland lies approximately to the seaward of Loviisa, and although they are still met with in the outer and western parts of the study area, they have been very seldom found in Hästholmsfjärden. The distribution boundary of Baltic Tellin (*Macoma balthica*) lies farther eastward in the outer archipelago, but it too has been met with relatively scarcely in Hästholmsfjärden, especially in recent years. It is clear that animal or plant species living in extreme salinity conditions are very

sensitive to all other factors causing stress in the environment, such as a decline in the water quality.

## 1.2 Thermal discharges

The cooling water is used for cooling the turbine condensers; both for after-cooling the primary and cooling the secondary seawater circuit. The cooling water is taken from the Hudöfjärden Bay located on the west side of Hästholmen Island. There are six holes (2.6 m × 5.15 m) in the intake construction near the bottom. The upper edge of the intake holes lies at a depth of 8.5 m and the lower edge at 11.1 m. The gap breadth in the intake grid is 85 mm.

From the condensers the warmed cooling water is led to the eastern shore of Hästholmen Island and discharged as a thermal load into Hästholmsfjärden Bay. The cooling water is discharged into the sea over a 90-m-broad embankment, which distributes the effluent over the surface of the receiving water body. The principle in this kind of surface layer discharge design is to disperse the effluent over a large surface area, and thus enhance the transfer of the heat into the atmosphere. The mixing of the effluent with the receiving water is generally poor, and the plume remains discrete for some distance from the outfall. Most of the heat loss in the surface plume is by evaporation (Langford 1990).

In the first operational years (1977–1980), the average flow rate of cooling water varied between 10 $m^3 s^{-1}$ (1980) and 22 $m^3 s^{-1}$ (1978 and 1979). Since 1981, the two units of the power plant have together used 37–46 $m^3 s^{-1}$ of cooling water given as annual average flow rates (Appendix 2). The amounts of heat discharged into the sea were $13–22.5 \cdot 10^3$ TJ $a^{-1}$ in 1977–1980 and $43.8–58.7 \cdot 10^3$ TJ $a^{-1}$ in 1981–2006. In 1981–1996, the average amount discharged into the sea was $50.1 \pm 2.6 \cdot 10^3$ TJ $a^{-1}$. In 1997, the net rated power of the two power plant units was raised from 445 MW to 488 MW, and consequently the average amount of heat discharged into the sea was $56.2 \pm 1.5 \cdot 10^3$ TJ $a^{-1}$ in 1997–2006. The highest temperature of the cooling water discharged into the sea has been 32.1°C, given as an hourly mean. The highest temperature of discharged cooling water permitted by the decision of the Water Court is 32.0°C. This limit was barely exceeded for a short time at the end of July in 1997 and in 2003 (Annual reports of Fortum Power and Heat Oy).

## 1.3 Nutrient load

When studying the thermal effects of the Loviisa power plant in the sea area, it is necessary to know the level of nutrients (phosphorus and nitrogen) in the water recipient. In this respect, it is important to sort out the load of nutrients

from different sources into the regional coastal waters: from the power plant, from other local sources and especially from the major sources, i.e., large rivers having their estuaries nearby.

The annual load of total phosphorus and total nitrogen from the Kymijoki and Tesjoki Rivers, from the sewage treatment plant of the town of Loviisa, from the local fish farms of Semilax and Loviisan Smoltti, and from the Loviisa power plant are given in Appendix 3. The Kymijoki River is the largest river flowing directly from Finnish territory into the Gulf of Finland (mean annual discharge 310 $m^3 s^{-1}$ in 1912–2004) and the data given in the Appendix apply to the annual load from the main (western) fork of the river mouth into Abborfjärden (Fig. 2). The Tesjoki River is a smaller river flowing into Kullafjärden, west of Abborfjärden (mean annual discharge 4.9 $m^3 s^{-1}$). Only a small share of the nutrient loads from Tesjoki and Kymijoki enters Klobbfjärden and Hästholmsfjärden via Jomalsundet. However, especially in spring, nutrients spread to the sea area of Hästholmen in river waters flowing from the east of Boistö. In addition, the average annual loads of total phosphorus and total nitrogen in 1996–2001 from the Loviisanjoki River (outlet at the town of Loviisa) were 5 100 kg and 74 900 kg, respectively (Finnish Environment Institute).

The outlet of the sewage treatment plant of the town of Loviisa is located at Östra Mörören nearby Svartholm Island in the north part of Hudöfjärden. The load from the town is mainly focused on Hudöfjärden, but taking into account the volume of cooling water flowing daily through the power plant from Hudöfjärden to Hästholmsfjärden, it is clear that the load of nutrients in Hudöfjärden may also affect the nutrient levels in Hästholmsfjärden.

Semilax operates two fish cultivation ponds in the sea area south of Hästholmen, in the near vicinity of the power plant. Loviisan Smoltti is a fish farm utilizing the warm water from the power plant in cultivating fingerlings. The operation was started in 1987. The farm is located on Hästholmen Island and the load is focused on Hästholmsfjärden. Sludge liquor from the fish farm has been treated in the sewage treatment plant of the power plant, but the data given in the Appendix indicate the direct load from the farm.

The load data of the power plant consist of summed values from all sources associated with the operation of the power plant on Hästholmen, including that of an accommodation area for temporary workers located on the mainland northeast of the power plant, but excluding that of the Loviisan Smoltti fish farm (see above). During the construction work of the power plant in the late 1970s, the load caused by communal waste water was high, because of the large number of employees working at the site. The latter waste water is discharged into Hudöfjärden, while the process waters are discharged with the cooling water into Hästholmsfjärden.

In addition to the above-mentioned sources, the internal nutrient load caused by remobilization of phosphorus and nitrogen from bottom sediments into the hypolimnion is an important factor affecting their concentrations in the water phase. Due to oxygen depletion in the near-bottom water in the deep basins during late summer and autumn, large quantities of nutrients in a form useable for algae are periodically remobilized into the near-bottom water. Lehtoranta and Mattila (2000) estimated that 675 kg of soluble reactive phosphorus and 1 860 kg of ammonium nitrogen were remobilized from the deep of Hästholmsfjärden in 1998. The phenomenon is discussed in detail in the chapters dealing with the oxygen and nutrient concentrations in water. The annual load of phosphorus from the approx. 400 summer cottages located within a 5 km radius of the power plant has been estimated as 20 kg and that of nitrogen as 50 kg (Mattila 2002).

The annual load of biological ($BOD_7$), chemical oxygen demand (COD) and solid matter caused by the communal waste water of the power plant are also given in Appendix 3. The load of all these emissions, but especially that of $BOD_7$, has considerably decreased during recent years in the communal waste waters of the power plant. In 1981–1992, the load of $BOD_7$ from the power plant was about 4% of that from the town of Loviisa and those of COD and solid matter were 0.03 and 0.02% of those of the Tesjoki River, respectively.

In 1995, the Gulf of Finland received in total 7 600 tonnes of phosphorus and 139 000 tonnes of nitrogen from overall sources (Pitkänen et al. 1997). Thus, the majority of the nutrient load comes from outside of the study area, and the total load affects the quality of water in the whole eastern Gulf of Finland.

## 1.4 Monitoring and research programmes

The aquatic environment of Hästholmen has been the object of intensive and versatile environmental studies for more than 40 years. When the nuclear power plant projects were started in Finland, very little knowledge was available regarding the effects of thermal discharges in our coastal waters. As a result of the intensive monitoring and research programmes, the sea area around Hästholmen has become one of the most studied coastal areas in Finland. Consequently, the long data sets obtained for various hydrographical and biological parameters provide a valuable contribution to the historical environmental data sets from the Gulf of Finland. For instance, the studies on phytoplankton primary production with the carbon-14 method were started in Loviisa as one of the first areas in Finland soon after a course held by Dr. E. Steeman-Nielsen at the Tvärminne Zoological Station (Bagge and Niemi 1971, Bagge & Lehmusluoto 1971). Fortunately, the measurements have in general

been made using the same methods, so that the results from the four decades are pretty well comparable.

The hydrobiological studies and the monitoring programmes of water quality have been focused on conventional hydrographical parameters (such as temperature, salinity, pH, oxygen concentration and its saturation state, total phosphorus, total nitrogen, transparency, etc.), phytoplankton and its primary production, benthic macrofauna and littoral vegetation. The monitoring programmes were reconsidered and amended, if needed, at intervals of about five years on the basis of the experience gained. As STUK took the responsibility for conducting the conventional ecological studies and the monitoring programmes, the motive was to get the necessary ecological background for the monitoring of environmental radiation and for the radioecological studies carried out in the area. At the same time, the institute has acquired a general view about all the impacts of the power plant on the aquatic environment, including those caused by the discharges of cooling water and other effluents.

Seawater samples were taken from various sampling stations 10–15 (–24) times per year in such a way that the majority of them were during the growing season (May–October). The number of sampling stations has slightly varied over the course of years, but at least there are long and quite unbroken data sets from eight sampling stations (1, 2, 3, 4, 5, 7, 8 and 10). These stations have also been continuously included in the obligatory monitoring programme (with Station 13 added in recent years). The location of the sampling stations is given in Fig. 3. The additional stations given in the figure have been included in the regular monitoring programme for a shorter time (Stations 9 and 11) or they have acted as supporting sampling stations for the regular monitoring programme (Stations 25, R1, R2, R3). The location of the sampling stations has been stable from year to year. The standard sampling times in the obligatory monitoring programme were the end of March, end of May, end of August and the beginning of November, but additional sampling was regularly carried out in connection with the measurements of primary production, and in autumn during the oxygen depletion periods at the deepest stations. The sampling in November was left out of the monitoring programme in 2000.

*In situ* primary production has been measured at the standard stations 2 and 8 since 1967 in parallel with the sampling of seawater. In 1973–1982, *in situ* primary production was also measured at Station 5 in Hästholmsfjärden and at Station 4 in Vådholmsfjärden, and in 1986–1987 and 1991 at the reference stations of Påsalöfjärden and Kejvsalö östra fjärd (Figs. 2 and 3). Since the late 1970s, measurements were made 10–12 times a year, but in the earlier years the frequency of the measurements was sometimes lower. The studies were

focused on the growing season (May–October) with special emphasis on the vernal maximum of phytoplankton in April–May.

The measurements of phytoplankton primary production capacity were started at Stations 2 and 8 in 1973. Until 1977, the primary production capacity was measured in samples taken from five separate depths. Regular surface water surveys were started in 1977 at Stations 1, 2, 3, 4, 5, 7 and 8 using mixed samples from 0–2 m. In some years, primary production capacity was measured at the reference stations R1, R2 and R3, too. The measurements were made in parallel with the *in situ* measurements.

The species composition and biomass of phytoplankton were studied annually at Stations 2 and 8 in 1967–1982, and after that every three years. The studies were carried out in parallel with the primary production measurements. The results have been published earlier by Bagge & Niemi (1971), Ilus & Keskitalo (1987) and Ilus & Keskitalo (2008).

Soft-bottom macrofauna (i.e., macrozoobenthos) has been studied in the sea area off Loviisa since 1966. At that time, samples of bottom fauna were taken for the first time from a transect between Orrengrund and Valko on board the research vessel Aranda (Bagge and Voipio 1967). At the same time, the first samples were taken with an auxiliary boat from Hästholmsfjärden, too. In 1967, a basic survey of benthic macrofauna was conducted in the littoral zone of Hästholmen Island. Occasional sampling at the deep soft bottom stations was also started in that year, but the regular monitoring of macrozoobenthos was started in 1973. The monitoring programme included ten standard stations: 1, 2, 3, 4, 5, 7, 8, 10, 51 and 52 (Fig. 3). Samples were taken twice a year, in May and August. In the first years, samples were even taken at some stations seven times a year; in these cases, mean values of the spring and late summer periods are considered here.

The first studies on littoral vegetation in Hästholmen were carried out in 1971. The study was implemented by snorkelling as a transect survey on nine 100-m long transects directed outwards from the shore line. In 1975–1982, aquatic macrophytes were studied annually by scuba diving and dredging along four permanent 100-m census transects around Hästholmen Island. (Ilus & Keskitalo 1986). Since that, the surveys have been repeated every three years along five transects: **a, b, c, d** and **e**. In 1999, the number of the census transects was increased to six by including transect **f** in the programme. The location of the transects is given in Fig. 4. The surveys were carried out in late August – early September, when the vegetation was best developed.

The results of the monitoring programme have been reported in Annual and Summary Reports in Finnish (see Appendix 1). Original data on the biological parameters (primary production, phytoplankton and benthos) are given in tables

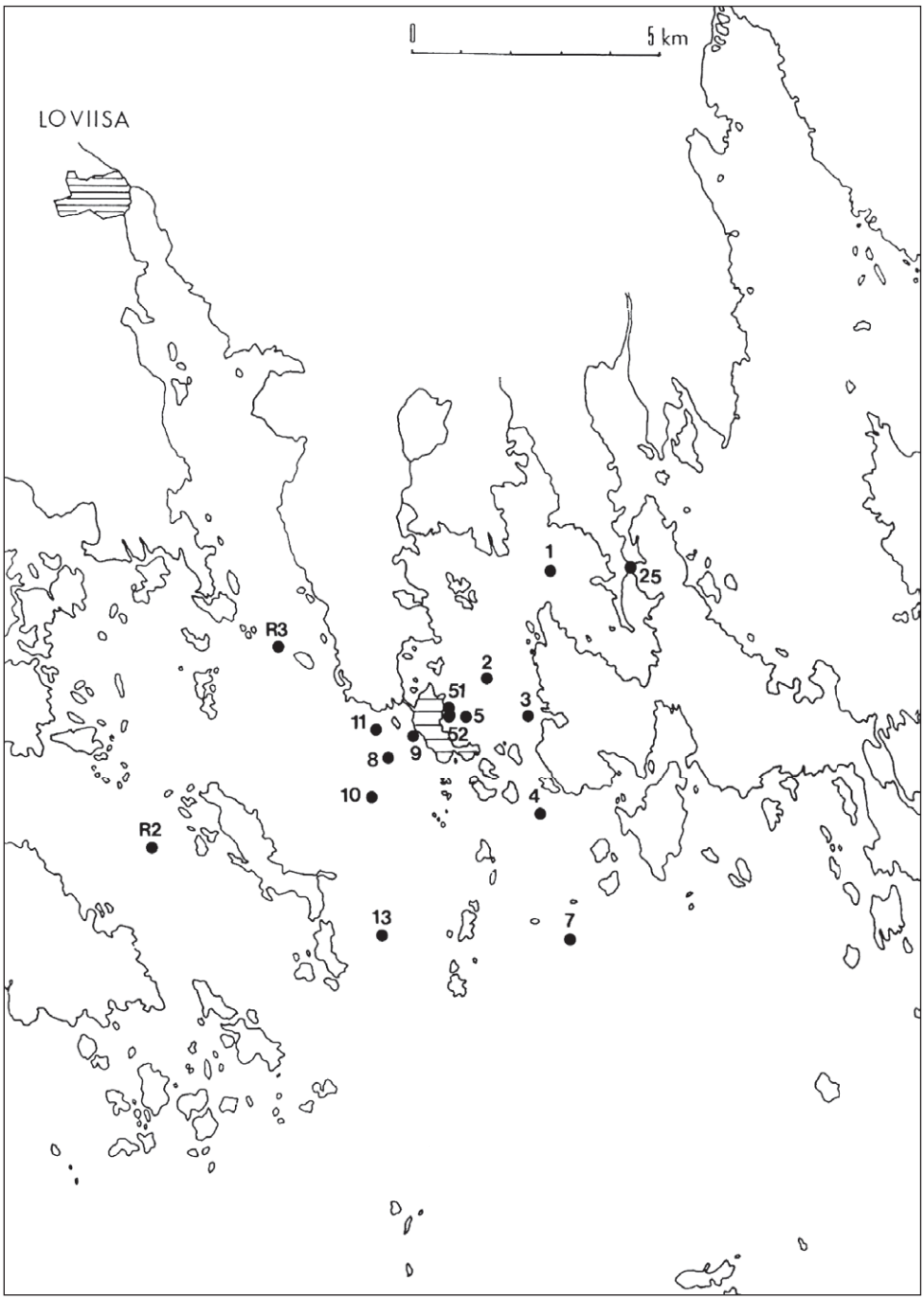


**Fig. 3.** Location of the sampling stations for hydrographical and hydrobiological studies at Loviisa. Reference station R1 is situated off the map, in Påsalöfjärden about 14 km west of the power plant (see Fig. 63).

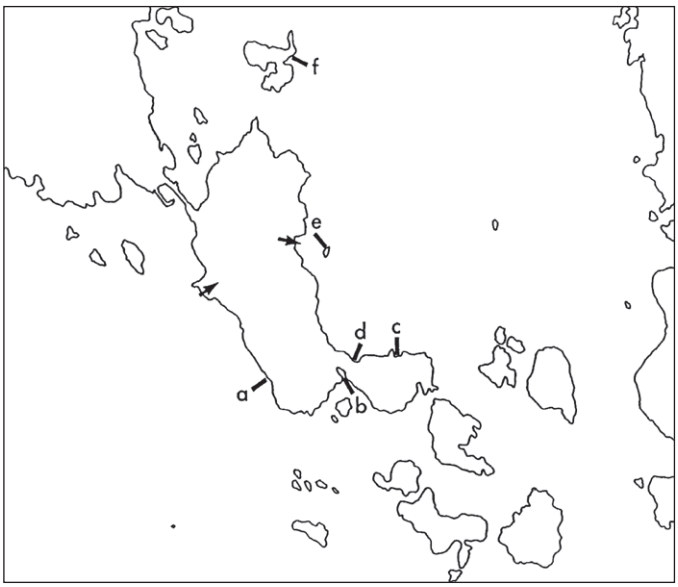


**Fig. 4.** Location of the transects for surveying littoral vegetation at Loviisa.

in these reports. Hydrographical data (as far as they belong to the obligatory programme) were given in tables in the reports since 1998; before that they were summarized in the text. Since 1983, the greater part of the hydrographical data has been entered into the national water quality register held by the Uusimaa Regional Environment Centre.

## 1.5 Material and methods

The material of this work consists of the results collected in the permanent monitoring programme and complementary hydrographical and hyrdobiological studies carried out in the marine environment of the Loviisa nuclear power plant since 1970. STUK has been responsible not only for implementing the monitoring programme, but also for designing and developing it, and for sampling and other field work connected with the monitoring.

The sampling and analysis methods used have been described in detail in the Annual Reports of the monitoring programme, which have been written in Finnish (Ilus 1980 … Mattila and Ilus 2007, Appendix 1), but some of them have also been described in English (eg. Ilus & Keskitalo 1986, 1987, 2008). The objective has been to follow as far as possible the same methods throughout the 40 years, so that the comparability of the results would endure in the long time-series. In such cases when the methods had to be modified or changed due to a

general development in sampling or analysis techniques, the comparability of the results was confirmed by intercomparison of the old and new methods.

**Position finding**

The permanent sampling stations were marked with surface buoys in spring. Benchmarks on the shore, an echo sounder and GPS were used when installing the buoys. The samples were regularly collected within a radius of 5–10 m of the buoy.

**Taking the water samples**

The water samples were taken with Ruttner or Limnos samplers. The height of the samplers varied between 32 and 62 cm and the volumes between 2 and 4.2 L. The marks on the hoisting wire were measured from the bottom plate of the sampler. In general, water samples were taken at depths of 0, 2 and 5 metres, and after that at intervals of five metres. The samples from the near-bottom water were taken so that the bottom of the sampler was 50 cm above the seabed.

The sub-samples were already separated in the boat as soon as possible into the sample bottles, with the oxygen, plankton and primary production samples being taken first. In particular, the samples for oxygen and primary production measurements were kept sheltered from sunlight in the boat. The sub-samples for pH measurements were separated later in the field laboratory from the salinity samples. In principle, the methods used followed the recommendations given by Mäkelä et al. (1992).

**Temperature**

The temperature of the water in the sampler was read with a mercury thermometer (accuracy 0.1°C) immediately after drawing up. The accuracy of the thermometer was checked at the beginning of the sampling period.

**Transparency**

During the open water period, the transparency of the water was measured from the shady side of the boat with a white, Ø 20-cm *Secchi disc*. In winter, transparency was measured from the cover plate of the water sampler through the hole in the ice.

**Salinity**

The samples were put into 250 ml glass or plastic bottles provided by the Finnish Institute of Marine Research (FIMR). The salinity was analysed at FIMR in accordance with its standard method using a salinometer. The method is

described in Grasshoff et al. (1999). The samples were kept cool before analysis. The results were given to two decimal places.

**pH**

The pH of the water samples was measured in the field laboratory immediately after the sampling trip. The measurements were made with portable Orion Research 401 or Orion Research 420A pH meters from temperature-adjusted samples according to the instruction manuals of the devices. Before the measurements the instrumentation was calibrated with buffer solutions of pH 7.00 and 10.00. The results were given to two decimal places.

**Oxygen**

The standard method of FIMR was used in the taking of oxygen samples and in carrying out their analysis (Koroleff 1979). The method was a modified Winkler method, in which the samples were put into 50 ml bottles (the volume of each bottle being individually determined), and the reagents (MnCl solution and alkalinic iodide solution) were added to the samples immediately after sampling before closing the ground-glass bung. The titration was done in the field laboratory as soon as possible after the sampling trip. The 0.015 N sodium thiosulphate solution used in the titration was changed twice during a sampling period. Solubility values from International Oceanographic Tables Vol. 2 were used in calculating the degree of saturation of oxygen.

**Total phosphorus and total nitrogen**

The samples were analysed at FIMR in accordance with its standard methods. The samples were kept cool before analysis. A manual method for total phosphorus and total nitrogen analyses was used until 1981 (Koroleff 1979), after which continuous-flow analysers and the methods described in Koroleff (1983) were used. The FIMR laboratory is the test facility T040 (EN ISO/IEC 17025) appraised by FINAS (Finnish Accreditation Service).

**Primary production**

Primary production was measured *in situ* with Steemann Nielsen's radiocarbon method (1952) in accordance with the Finnish SFS standard 3049 and the recommendations given by Lassig & Niemi (1972) and Gargas (1975).

The sampling was started in April–May and was continued until October–November at 1–3 week intervals in spring and 2–4 week intervals in summer and autumn. The samples were taken and the incubation was started in the morning, generally before 10 o'clock. The measurement depths were 0, 1, 2, 3, 5, 7.5 and 10 m (the lowest depth being 12 m in Hudöfjärden from 1967 to 1973).

The total depth is 11.5 m at permanent sampling station 2 in Hästholmsfjärden, and 17 m at sampling station 8 in Hudöfjärden. In the first few years, a measurement set consisted of one light and one darkened bottle at each depth. The employment of two parallel light samples was started in 1976, and in 1978–1981 parallel light samples were used regularly at each depth. From 1982 onwards, the measurement set regularly consisted of two light bottles at depths of 0, 1, 2 and 3 m; mean values of the parallel samples were used in reporting the results. Single light bottles were used at 5, 7.5 and 10 m. Dark fixation was measured at depths of 0, 2, 5, 7.5 and 10 m.

One ml of $NaH^{14}CO_3$ solution was added to 110 ml (volume of the bottles) of sample water, after which the sample bottles were put back as soon as possible into the sea and incubated for 24 h at their own depths in clear and darkened Jena glass bottles. Stout black plastic sheets were used in the light-shielding of the dark fixation bottles. From 1976 onwards, the $NaH^{14}CO_3$ solution was prepared and the activity was verified at STUK (Salonen 1979). Before that, the solution was prepared at Tracer Tekniikka Oy, Finland. The incubation was finished by adding 0.5 ml of conc. formalin (i.e., 0.2 ml of formaldehyde) to the sample. The samples were filtered through 0.45 µm cellulose-acetate filters in the field laboratory soon after drawing up the bottles from the water.

From 1970 to 1987 the activity concentration of $^{14}C$ in algae retained on the filters was determined with a Geiger-Müller counter according to Saxén and Lehmusluoto (1979); since 1987 a liquid scintillation counter (Wallac 1414 Guardian) has been used. Wallac OptiScint HiSafe scintillation solution was added to the bottles just before measurement. When the determinations were changed to liquid scintillation counting, the old and new methods were tested against each other with a large series of parallel samples in 1988. The results obtained with the new method at Loviisa were on an average 6% higher than those obtained with the old method. The results obtained with the Geiger-Müller method, and presented in this paper, have been corrected to be comparable with those obtained with liquid scintillation counting.

The concentration of dissolved inorganic carbon was calculated from the temperature, pH and salinity of the corresponding water samples according to Buch (1945). Dark fixation of carbon was subtracted from the light fixation to obtain the final primary production results.

**Primary production capacity**

An incubator of the Lehmusluoto type was used in incubation of the samples. It contains two narrow water vessels of plexiglass with two fluorescence tubes (AIRAM L18W-1XC DAYLIGHT 5000 DELUXE) between them. The luminous efficiency of the tubes at their centres was 8 800 lux, and the temperature of

the water bath, regulated by a circulation water thermostat, was 20 ± 1°C. The tubes were changed once a year before the beginning of the field season. The bottles were shaken thoroughly and transposed from the rims of the bowl to the centre and vice versa 5–6 times during the incubation to prevent the settling of plankton and to even out the light intensity. In the first few years, primary production capacity was measured at several depths at Stations 2 and 8, but from 1977 onwards the measurements were carried out using 0–2-m mixed samples. Otherwise, the methods were the same as for the *in situ* measurements. In an intercomparison exercise of primary production capacity measurements, arranged by the National Board of Waters in 1983 for 15 Finnish laboratories (Vuolas & Heinonen 1984) our results were consistent with the mean values.

**Benthic macrofauna**

The samples were taken with an Ekman-Birge sampler at the end of May and August. Over the course of time, three different Ekman-Birge samplers were used, the area of their orifices varying from 261 to 299 $cm^2$ and their weight from 4.4 to 5.0 kg. The varying area of the orifice was taken into account when calculating the results per $m^2$. In general, five parallel samples were taken at each station. Only at Stations 51 and 52, where the sediment contained a lot of sand and gravel, did one have to be satisfied with three parallel hauls. The parallel samples were collected into a 50 L plastic tub and mixed before sieving. However, from 2004 onwards the parallel samples were taken and handled separately. The sieving was done with a hand sieve (mesh size 0.6 mm) as soon as possible after the sampling. Before sieving, the bottom matter was thoroughly mixed with the water in the tub so that it easily passed through the sieve. The sludge produced was distributed with a plastic scoop in small batches onto the sieve so that the stay of the sludge on the sieve was as short as possible. The sieving residue was rinsed into a plastic box, and the animals were picked from the residue in the field laboratory as soon as possible after the sampling. Large-sized predators (*Saduria, Marenzelleria,* etc.) were already picked from the sample during the sieving. The sieving residue was gone through in small batches on a white plate using forceps and a light loupe.

The animals were preserved in 80% alcohol, which works better than formalin for samples that contain plenty of oligochaetes and chironomid larvae, because an exact determination of these groups generally requires the making of microscope preparates. The soft tissues of the animals become hard and less transparent in formalin. The species and biomass determinations were done from preserved samples within six months of sampling. The biomasses were weighed (accuracy 0.1 mg) species by species after the surface moisture had evaporated (less than 1 min on blotting paper). From 2004 onwards, the

parallel samples were determined separately, but the results are combined in this consideration.

**Littoral vegetation**

Littoral vegetation was studied on permanent 100-m-long census lines by scuba diving complemented by dredging with a Luther rake (Luther 1951). The method is described in Ilus (1980), Ilus & Keskitalo (1986) and Keskitalo & Ilus (1987); it is a modification of the line census method used earlier in fish studies at Loviisa (Bagge et al. 1975). A 100-m-long floating rope with numbered stryrofoam floats at 5-m-intervals was laid out between the shoreline and a buoy anchored 100 m from the shore. The diver slowly followed the transect near the bottom from the outer end to the shore. He/she was directed with a sounding lead from a rubber dinghy proceeding along the surface rope. The boat was usually staffed by two persons, one of whom rowed, while other communicated with the diver by underwater telephone (Fig. 5). Until 1994, a wired underwater telephone (Finn-Suit SP 1) connected to a tape-recorder was used to record the observations and description of the vegetation. From 1997 onwards, a wireless telephone Aquacom SSB-2001 was used. While proceeding along the transect, the diver described the vegetation and the character of the bottom on a 2-m-broad strip (1 m on each side of the diver), while the telephone-operator in the boat recorded distances from the shoreline on the tape according to the styrofoam floats along the surface rope. The diver paid special attention to dominant species and their coverages (estimated as percentages) and noted the limits of the different vegetation belts. The diver also used numbered plastic bags carried in a string bag to collect those samples of plants that were difficult to determine during the dive. In general, an auxiliary safety diver followed by the side of the main diver and assisted him/her in the sampling. The main part of the samples were

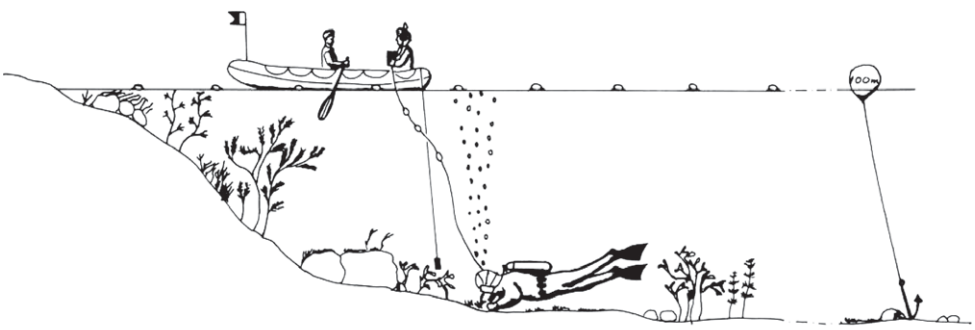


**Fig. 5.** The scuba diving method used in the studies of the littoral vegetation.

determined immediately after the dive, but some of them were conserved in 5% formalin or dried on waste paper for more exact examination.

To complete the information on the species composition, dredging was performed by the method of Luther (1951) on each transect after the dive. The tape recording made during the dive was transcribed into written form immediately afterwards.

## 1.6 Temperature of the seawater

The discharge of cooling water physically affects not only temperatures, but also currents in the receiving water body. At Loviisa, the thermal discharge is dispersed smoothly on the surface of the recipient. Thus, mixing of the heated water with the receiving water is generally poor, and the plume remains discrete for some distance from the outlet (*cf.* Langford 1990).

The temperature of the cooling water rises by about 8–12°C when passing through the condensers of the power plant, and consequently, the increase in seawater temperature in the discharge area is the most obvious environmental effect of the power plant (Ilus et al. 1997). This is especially true in winter, when the warm cooling water may affect ice and temperature conditions at distances of >10 km west of Hästholmsfjärden (Hari 1982).

Fortum has followed water temperatures in Hästholmsfjärden in winter by means of a temperature and ice monitoring programme, and in the past with three automatic measuring-buoys laid at a distance of 500 m from the cooling water outlet. In addition, FIMR followed water temperatures in the 1970s by means of a stationary oceanographic mast constructed in the middle of Hästholmsfjärden (Launiainen 1975, 1979, 1980). The results showed that, when the power plant was in operation, the changes of water temperature taking place in Hästholmsfjärden could also be very strong and rapid, especially under the ice in winter. In certain circumstances, the temperature could change > 4°C in half-an-hour or so. In light of this, our measurements carried out 10–15 (–24) times a year are not able to monitor the fast changes, but, on the other hand, they yield a valuable long time-series of measurements that have been made regularly at permanent sampling stations during 40 years.

**Winter**

The effects of the cooling water on the temperatures in the sea area around Hästholmen have been most obvious in winter: then the conditions have also most fundamentally differed from those of the natural state (Ilus & Keskitalo 1980). In consequence, the thermal discharges have significantly affected the ice conditions in the vicinity of the power plant. The formation of a permanent

ice cover in the discharge area has been delayed in early winter. In addition, the break-up of the ice has been advanced, so that the growing period has been prolonged at both ends. Areas close to the cooling water outlet are usually open throughout the winter, and the size of the open area may change day by day (ranging from several hectares to several square kilometres) according to the weather conditions. In general, the ice cover is thin in the sounds leading out from Hästholmsfjärden, and in late winter the ice melts quickly in the sounds when currents raise warm water into contact with the ice. In the northern part of Hästholmsfjärden and in Klobbfjärden the ice cover is often firm, even though the south part of Hästholmsfjärden and the sounds may be free of ice.

In fact, in winters milder than normal, the ice conditions were often already poor in the sea area before the existence of the power plant. According to the long-term statistics of FIMR, the number of real ice days at Svartholm (the northern part of Hudöfjärden) was on average 118 during the 30-year period 1961–1990. A permanent ice cover was formed on December 25 and ended on April 16. However, the ice winters 1960–61, 1974–75, 1988–89 and 1989–90 were mild in the whole Baltic Sea (Seinä and Peltola 1991), and more recently, the ice winters 1999–2000, 2000–2001 and 2001–2002 were short and mild, according to the Baltic Sea Portal (2008a). According to HELCOM (2007), there has been a general tendency toward milder sea-ice conditions during the past century in the Baltic Sea. The largest change has been in the length of the ice season, which has decreased by 14–44 days, mainly due to earlier ice break-up. During the past ten years, all ice winters have been average, mild or extremely mild.

During proper ice winters in the 1980s (such as 1979–80, 1984–85 and 1986–87), there was a quite permanent ice cover on the major part of Hästholmsfjärden (excluding small areas in the south-west) from late December to late April, and the duration of the ice winter in Hudöfjärden and Vådholmsfjärden was only a little longer. In the late 1980s and early 1990s, the duration of the ice winter was only a few weeks at the sampling stations of Hästholmsfjärden and 1–2 months in Hudöfjärden. In the 1990s and early 2000s, the winters were in general mild or very mild, and the duration and extension of the ice cover in the sea area around Hästholmen was very variable; i.e., the major part of Hästholmsfjärden was open throughout the winter. During the more proper winters of 1993–94 and 1995–96, there was a permanent ice cover in the area outside Hästholmsfjärden from December to the turn of April and May. The duration of the ice winter was then about 4 months at Stations 1, 4 and 8, while it was from a couple of weeks to a good month at Stations 2, 3 and 5.

In spite of the continuous thermal loading, the greater part of Hästholmsfjärden has been at least temporarily covered by ice in normal winters.

The formation of ice is encouraged by the flow of a fresh water layer originating from the runoff of the Tesjoki and Kymijoki Rivers onto the surface of the water body. Owing to its higher salinity, the heavier cooling water (taken from a depth of 8.5–11.1 m in Hudöfjärden) has 'dived' below the lighter freshwater layer and spread over wide areas as a thin sub-surface warm water layer. The cooling water has settled to a depth corresponding to its own density. After the formation of ice, the thermal stratification becomes stronger as the fresh water layer below the ice expands and the ice prevents the winds from mixing the water layers. At the same time, the transference of heat to the atmosphere is hindered, and the amount of warm water accumulated below the fresh water layer increases, whereupon it spreads over wide areas, often as a discrete layer (as 'warm water lenses'). Sinking cooling water plumes have also been reported from the sea areas off the Oskarshamn and Forsmark nuclear power plants on the east coast of Sweden (Ehlin 1974 and Sandström & Svensson 1990).

The highest temperatures measured under the ice in March–April have been 11.0–11.4°C at the stations in Hästholmsfjärden, 9.2°C in Klobbfjärden (Station 1), 7.4°C in Vådholmsfjärden (Station 4) and 9.9°C in Hudöfjärden (Station 8); most of them in 1987. The spreading of the heated water into Hudöfjärden means also that recirculation may occur under the ice. The cooling water usually settles down to a depth of 2–3 m (sometimes 4–5 m). Outside Hästholmsfjärden, the warm water plume turns west, following the main direction of the currents on the south coast of Finland. According to Hari (1982), it seems to be clear that during such winters when the Gulf of Finland has a firm ice cover, the heated water flows in a quite stable way under the ice along the south coast westward. Most of the water from the Kymijoki River spreads out in the outer archipelago to a distance of just a little over 10 km from the power plant. At the edge of the fresh water plume, the heated water comes in contact with the ice, and melts it from below. The ensuing difficult ice conditions can thus hinder winter fishing in quite extensive areas (Ilus et al. 1997).

In mild or very mild winters, the width of the ice-free area has often been 3–8 $km^2$. In ice-free conditions, the heat is freely transferred into the atmosphere, and the winds are effective in mixing the water masses. Thus, the highest temperatures occurring at the surface in open water are usually lower than those measured under the ice.

Aquatic organisms in the northern Baltic Sea are acclimatized to a clear winter period occurring every year. A shortening of the ice winter or a total lack of the ice cover leads to indistinct limits between the growing season and the wintering season. Most plants need a clear resting period, and the changed ice conditions disturb the natural annual rhythm of growth. For instance, it was already evident in the 1980s (Keskitalo & Heitto 1987) that, at Olkiluoto,

the increased illumination in the ice-free area and the direct influence of the temperature rise caused a significant advance in the start of the growing season of many species and to excessive occurrences of diatoms.

**Open water period**

Cooling water is discharged into Hästholmsfjärden over an curved 90-m wide embankment, which distributes the effluent over the surface of the receiving water body, where the warm water plume spreads out as a layer a few metres thick. Due to its lower density, the warm effluent does not easily mix with the water mass of the recipient, and this leads to a strengthening of the thermal stratification in the discharge area. Nevertheless, the mixing and the temperature decrease are most rapid in the near-zone, while the transfer of the heat into the atmosphere has a greater relative significance in the far-zone (Ehlin 1974). This is mainly due to the clearly lower mixing intensity when the velocity of the plume has lowered to that corresponding to the flow rate in the recipient, but also because the exposure surface of the plume for heat emission is larger in this zone. Most of the heat loss in a surface plume is due to evaporation (Langford 1990).

In open water circumstances, winds have an essential effect on the spreading out and the width of the distributive area of the warm water plume. The heated water flows with the wind and may become 'packed' into the northern or eastern part of the discharge area, or against the eastern bank of Hästholmen, or it may be forced to flow out from Hästholmsfjärden through the sounds in the south. Due to the effective transfer of the heat into the atmosphere and the more effective mixing of water masses in open water conditions, the thermal effects do not reach as far as in ice winters.

The seasonal course of the water temperatures from early spring to late autumn depends on the temperature of the intake water and the weather conditions in the course of the summer. Since the water is taken from a depth of 8.5–11.1 m, it is generally much cooler and its temperature range is narrower than that of the surface water. Already in the natural state, the warming in the spring was much quicker in Hästholmsfjärden than in Hudöfjärden and Vådholmsfjärden. Likewise, the water in Hästholmsfjärden cooled quicker in the autumn. This was due to the shallowness and the relatively small water volume of Hästholmsfjärden. In spring, the warming was also encouraged by the position of the bay, sheltered from strong winds, and by the dark colour of water caused by the river discharges. For the same reasons, the warming of the surface water in mid-summer was also more effective than in the outer sea areas, with, however, the warmed surface layer becoming thicker in the latter. In general, by late summer a relatively isothermal surface layer reached a depth

of 5 m in Hästholmsfjärden and 7.5–10 m in Hudöfjärden. These features give the background against which the thermal effects caused by the cooling water can be considered.

The exceptionally warm summer of 1972 gives us a good reference for the present day from a time prior to the cooling water discharge. The temperature of the surface water was then above 20°C down to a depth of 5 metres from late June to mid-August. On the 7th of July, the highest temperatures were 23.3°C in the surface water and 22.2°C at a depth of 5 metres at Station 2 in Hästholmsfjärden. At Station 8 in Hudöfjärden, the corresponding values were 22.8°C in the surface water layer and still as high as 21.0°C at a depth of 7.5 metres. According to Hansson and Omstedt (2008), the year 1975 was the warmest in the Baltic Sea in terms of water temperature for the past 500 years, but the temperatures in the Loviisa area did not reach similar values to those of 1972. The temperatures of the surface water were, at their highest, 21.2°C and 20.7°C at Stations 2 and 8, respectively, at the end of July.

Since the commissioning of the power plant, the highest temperatures in the surface layer have exceeded the values given above, but the warmest situations have in general been more momentary and the warming has not reached deeper layers to a similar extent. In years when the weather in May and early June has been warm and light winds have prevailed, the surface temperatures have already reached their maximum values in June, but in general the highest temperatures occur in July or August. The maximum values in the surface water were recorded at almost all sampling stations at the turn of July and August in 2003. Then the temperatures were:

| Station | 2 | 30.0°C | Station | 4 | 25.1°C | Station | R1 | 25.0°C |
|---|---|---|---|---|---|---|---|---|
| | 5 | 28.5 | | 8 | 24.1 | | R2 | 24.0 |
| | 3 | 27.7 | | 10 | 23.1 | | R3 | 24.1 |
| | 2 | 27.1 | | 7 | 23.2 | | | |
| | 1 | 26.4 | | 13 | 17.9 | | | |

However, due to the arrival of a low pressure area, the temperatures decreased a few days later, and by the 12th of August were below 20°C at all stations except 02 and 5. Only at Stations 8 and 7 were the records from the end of June 1988 slightly higher than those in 2003: 24.8 and 24.3°C, respectively. According to Hansson and Omstedt (2008), the highest annual air temperatures in the Baltic Sea region during the past 500 years occurred in 1989, but then the seawater temperatures hardly exceeded 20°C in the Loviisa area. Typical cool summers with low temperatures in the seawater were 1977 and 1987.

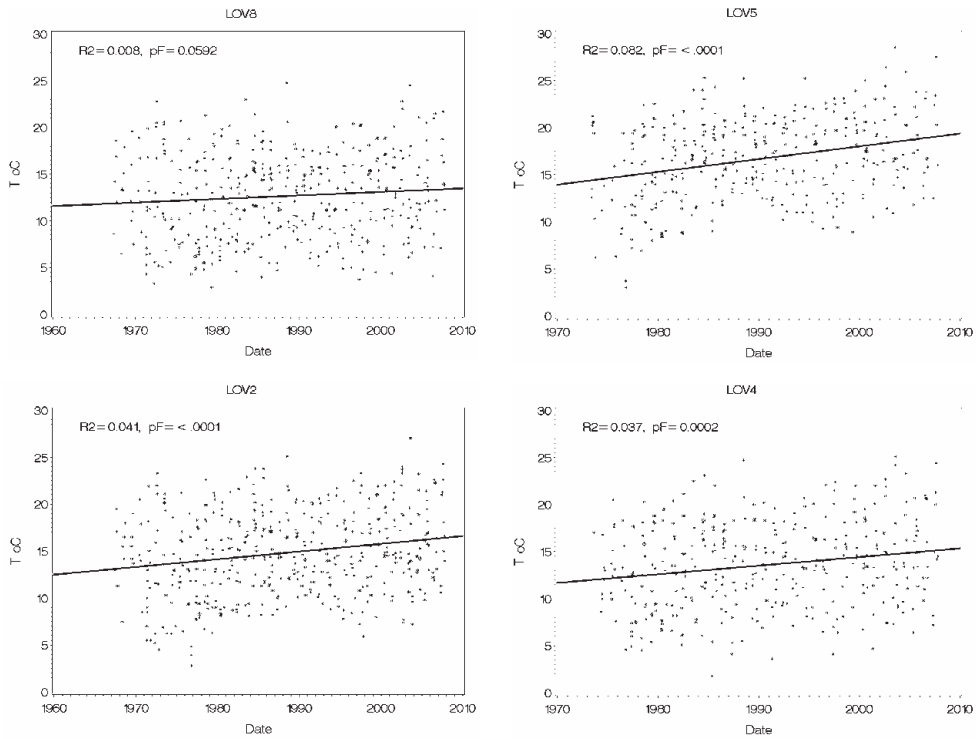


**Fig. 6.** Scatter plots of all temperature measurements of surface water carried out at Stations 8, 5, 2 and 4 during the whole study period at Loviisa. The regression lines show the trends in surface water temperature.

To assess the magnitude and extent of the thermal effects in the sea area, the temperatures measured at each station were compared with those measured at Station 8 situated in the intake area of the cooling water. This was justified by the monitoring history of Station 8, which is as long and frequent as that of Station 2, and longer than that of any other station. In addition, the location and environmental conditions of Station 8 are better comparable with those of the other stations, for instance, of Station 7, which is located in the outer archipelago. The regression lines in the scattering plots of all measurements of surface water carried out at four sampling stations during the whole study period (Fig. 6) show a significant rise in surface temperatures at the Stations 5, 2 and 4. The rise at Station 8 was not significant, but possibly due to the claimed general increase in surface temperatures of the Baltic Sea during recent decades (e.g. Döscher and Meier, 2003 and 2004). Siegel et al. (2006) stated that the yearly means show a slight positive trend with an increase of 0.8 K in 15 years (1990–2004). On the other hand, Hansson and Omstedt (2008) noted that from

1935 to the present no statistically-significant water temperature trend can be determined in the Baltic Sea. In a longer perspective, Makkonen et al. (1984) stated that the summer surface water temperatures in the early 1980s were 1ºC higher than during the second half of the 19th century and at the beginning of the 20th century.

The mean surface water temperatures of the growing season (May–October) show annual fluctuations caused by varying weather conditions in different years (Fig. 7). Before the commissioning of the power plant, the difference in the mean surface water temperature during the growing season between Stations 2 (Hästholmsfjärden) and 8 (Hudöfjärden) was 1°C, which was due to the shallowness and sheltered character of Hästholmsfjärden, as mentioned before. The temperatures decreased from the beginning of the study period (1967) to the commissioning year of the power plant (1977). Since then, temperatures have clearly increased at Stations 5 and 2 of Hästholmsfjärden, but a slight upward trend is also visible in the curves of Station 8 (Hudöfjärden) and in that of the outermost Station 7 (Orrengrundsfjärden).

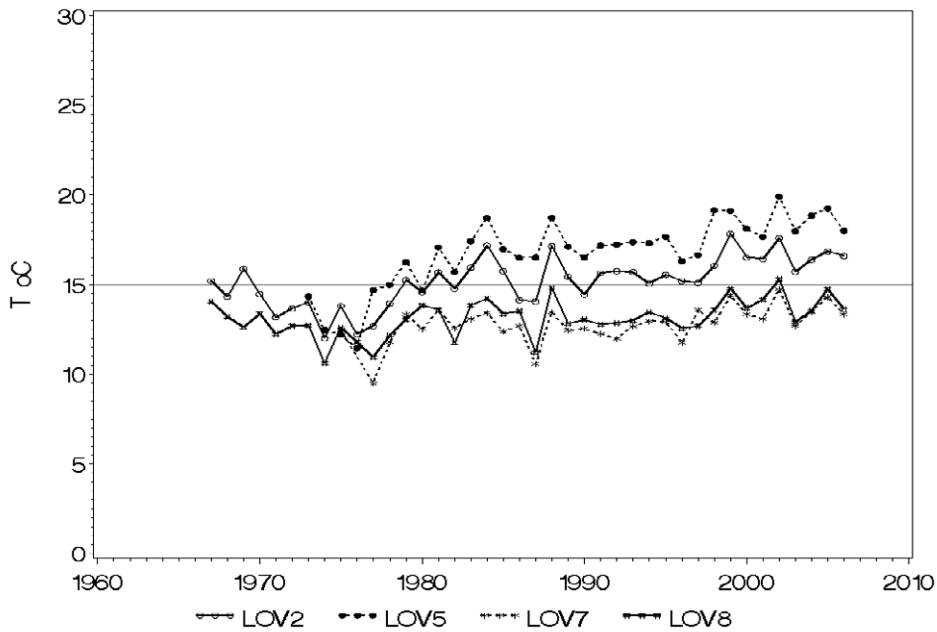


**Fig. 7.** Mean surface water temperatures (ºC) of the growing season (mean of monthly average values of six months, May–October, in each year) at Stations 2, 5, 7 and 8 at Loviisa in 1967–2006.

**Table 1.** Change in mean surface water temperature of the growing season at eight permanent sampling stations off Loviisa during 1976–2006.

| Area | Hästholmsfjärden | | | Vådholms-fjärden | Klobb-fjärden | Orren-grunds-fjärden | Hudöfjärden | |
|---|---|---|---|---|---|---|---|---|
| Station | 5 | 2 | 3 | 4 | 1 | 7 | 8 | 10 |
| Waterway distance from outlet [km] and direction | 0.4 E | 1.0 NE | 1.7 E | 2.8 SE | 3.4 NE | 5.4 SE | 3.0 SW | 3.4 SW |
| Change in mean surface water temperature of growing season (May–Oct.) in 1976–2006: | | | | | | | | |
| Probability $p_F$ | <0.0001 | 0.0002 | 0.0005 | 0.0017 | 0.0034 | 0.0527 | 0.0628 | 0.1343 |
| Average difference to Station 8 in 1997–2001 [°C] | 4.3 | 2.6 | 3.1 | 1.3 | 2.4 | -0.3 | 0 | -0.1 |
| Average difference to Station 8 in 2002–2006 [°C] | 4.9 | 2.6 | 3.3 | 1.2 | 2.4 | -0.3 | 0 | -0.3 |

The cooling water has raised the mean surface water temperatures during the growing season by 4–5°C at a distance of 0.4 km from the outlet and by 2.5–3°C at a distance of 1–2 km in Hästholmsfjärden (Table 1). The rise in temperature has also been statistically significant at Station 4 in Vådholmsfjärden (distance 2.8 km) and at Station 1 in Klobbfjärden (distance 3.4), but not at Station 7 in Orrengrundsfjärden. Compared to that at Station 8 in the intake area, the average increase has been 1.3°C at Station 4 and 2.4°C at Station 1. The F test of the change in the mean surface water temperature of the growing seasons during 1976–2006 shows that the change has been very significant at Stations 5, 2 and 3, and significant at Stations 4 and 1 (Table 1).

## 1.7 Water salinity

Water salinity is an important environmental factor in the brackish-water conditions of the coastal areas of the eastern Gulf of Finland. The survival of both marine and fresh water organisms is put to the test there due to the low salinity and, in particular, due to the large changes in salinity during the year. The species composition is poorer and more sensitive to other environmental changes than, for instance, in the western parts of the gulf, where the salinity of the surface water is 1–2‰ higher.

The salinity of the surface water varies in the sea area of Hästholmen from nearly 0‰ in early spring to 4–6‰ in late autumn. In general the salinities are at their lowest in March–April, and at their highest in October–November. As described above, a fresh water layer accumulates in early spring from the

runoff and river waters under the ice, where it forms a 1–2 m thick layer. In Jomalsundet the salinity is often 0‰ in April, and under the ice salinities close to zero have been measured at all sampling stations in the area. The minimum surface water salinities observed have been 0.13‰ in Hästholmsfjärden, 0.2‰ at Stations 4 and 7, and 0.26‰ at Station 8 (1980). After the melting of the ice, the salinities begin to rise, as winds can then mix the water masses. Sometimes, in the summer, the salinity of the surface water can rise strongly and rapidly due to up-welling of deep offshore waters into the archipelago (e.g., in 1996). The highest observed surface water salinities have been 6.17‰ in Hästholmsfjärden and 6.3‰ in Vådholmsfjärden and Hudöfjärden (November 1978).

In the near-bottom water, the variation in salinity is smaller, the values depending on the depth of the sampling station. The range of salinities in near-bottom water at the permanent benthos stations during the whole study period are given in Table 2. The highest salinity value measured in the area was 7.62‰ in near-bottom water at Station 7 in September 2006.

**Table 2.** Range of salinities (‰) in near-bottom water at the permanent benthos stations at Loviisa in 1967–2006.

| Station | Depth (m) | Minimum (Year) | Maximum (Year) |
|---|---|---|---|
| 1 | 8 | 2.69 (2006) | 5.65 (2006) |
| 2 | 12 | 3.51 (1995) | 6.23 (1977) |
| 3 | 18 | 3.69 (1989) | 6.31 (1978) |
| 4 | 23 | 3.81 (2001) | 6.94 (1985) |
| 5 | 11 | 3.54 (1989) | 6.25 (1977) |
| 7 | 33 | 3.59 (1992) | 7.62 (2006) |
| 8 | 17 | 3.74 (1989) | 6.41 (2006) |
| 10 | 24 | 4.11 (1992) | 6.57 (1978) |

Water salinities in the study area are regulated by the general trends in the salinity of the Gulf of Finland and the whole Baltic Sea, and by the variation in the quantities of runoff and river waters, which again depend on the amount of rainfall in the neighbouring area and in the drainage areas of the rivers discharging into the eastern Gulf of Finland. The local mean surface water salinities of the growing seasons (May–October) have considerably fluctuated during recent decades (Fig. 8). The fluctuation has been identical at all the stations monitored.

In spite of being at the far end of the Baltic Sea basin system, the Gulf of Finland is, however, hydrographically governed by the inflow of saline waters

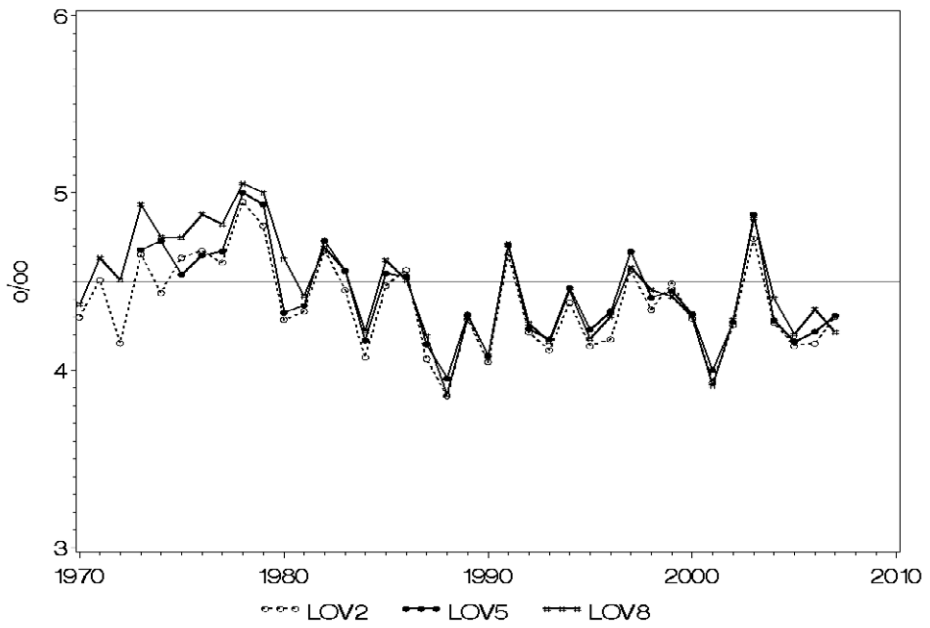


**Fig. 8.** Mean surface water salinity (‰) of the growing season (May–October) at Stations 2, 5 and 8 off Loviisa in 1970–2006.

through the Danish Sounds. In addition, approximately one-quarter of the total river water volume received by the Baltic Sea enters directly into the Gulf of Finland. The River Neva, with an annual inflow of some 77.7 km$^3$, is the largest river in the catchment area of the Baltic Sea. The large river runoff has a remarkable influence on the hydrography of the Gulf of Finland (Perttilä et al. 1995). According to Perttilä et al. (1980), the surface salinity in the Gulf of Finland increased by about 0.5‰ between 1962 and 1978. In the 1980s and 1990s, the salinities in the Baltic Sea and in the Gulf of Finland decreased, since none saline pulse from the North Sea reached the Baltic Sea, except in 1993 (Kauppila and Bäck 2001). In the 2000s, only one slighter pulse has occurred, in 2003 (Alenius and Lumiaro 2008).

In the sea area off Loviisa, there was a clear upward trend in the salinity of the surface water during the 1970s reaching maximum values in 1978 and 1979 (Fig. 8). This was in good agreement with the general trend in the Gulf of Finland (*cf.* Perttilä et al. 1980). The major inflow of saline water into the Baltic Sea in 1976 appeared in the surface water at Loviisa about two years later (*cf.* Alenius et al. 1998). During the 1980s and 1990s, the trend was slightly negative, retaining higher salinities, e.g., in the growing seasons of 1982, 1985, 1986, 1991 and 1997, and low salinities in, e.g., 1984, 1987, 1988 and 1990. In the 2000s,

the salinity of the surface water was exceptionally low in 2001 and exceptionally high in 2003. During the whole study period 1967–2006, the mean surface water salinities of the growing seasons varied between 3.85 and 4.95 ‰ at Station 2 in Hästholmsfjärden.

The cooling water flow from the power plant seems to have slightly affected the salinity in Hästholmsfjärden. Before the commissioning of the power plant, there was a difference of about 0.2‰ in the mean surface water salinities of the growing season between Stations 8 and 2. Since the power plant has been in operation, the difference has narrowed, so that in 1989, 1990, 2000 and 2002, for example, the mean salinity was about the same at Stations 8, 5 and 2. In some years (e.g., 1986, 1999 and 2006) the mean surface water salinity of the growing season has even been higher at Station 2 than at Station 8 (Fig. 8). The equalling-out of the salinity difference is explained by the facts that the cooling water is taken from Hudöfjärden from a depth of 8.5–11.1 m and that the average cooling water flow in 18 days equals the entire volume of Hästholmsfjärden. One would expect that the increased salinity would have a positive impact on the biota in Hästholmsfjärden, where many marine and brackish-water species live at their extreme limit of survival due to the low salinity.

## 1.8 pH of the water

pH is one of the conventional hydrographical parameters in the monitoring of water quality, and was routinely measured in almost all water samples, in the same way as temperature and salinity. The measurements were primarily taken on account of the phytoplankton primary production studies, because pH is used when calculating the amount of inorganic carbon in water, but the pH values can also be used in the monitoring of several processes in the sea area.

The freshwater layer accumulated under the ice in late winter and early spring consists of poor-quality river waters with low pH. Consequently, the lowest pH values in the surface water were usually measured in March–April. The minimum values recorded at that time were 5.9 in Hästholmsfjärden, 6.0–6.1 in Vådholmsfjärden and Hudöfjärden and 6.6 at Station 7 in Orrengrundsfjärden.

After the break-up of the ice, and in parallel with the vernal maximum of phytoplankton primary production, pH values increase strongly as carbon dioxide is fixed by photosynthesis. The pH has then often exceeded 9 in the surface water (maximum 9.3). After that, pH values have markedly decreased, being at their lowest during the growing season in late summer

or early autumn before the autumn turnover. The mean pH values of surface water during the growing seasons were 7.9–8.4 at Stations 2 and 8 in 1967–2006. In general, the differences between the sampling stations were small.

In the near-bottom water of the deep basins, the lowest pH values have occurred at the end of the summer stagnation period due to strong decomposition of organic matter in the sediments. In connection with oxygen depletion, the minimum pH value was 6.7 at Stations 3 and 10.

## 1.9 Water transparency

Water transparency, or *Secchi* depth, is a measure of the clarity of the water, and depends on the amount of particulate matter and dissolved substances in the water. In the open sea, the lowest values are recorded in July, the primary cause for the decreased transparency in summer time most probably being the increase in phytoplankton biomass (Fleming-Lehtinen et al. 2007). However, in coastal sea areas such as the Loviisa archipelago, lying close to river estuaries, the role of inorganic particulate matter is significant, especially in spring. This is due to the leaching of substances from the drainage areas, which results in an abundant diffusion of clayey river waters into the coastal zone. In any case, and especially in summer, the *Secchi* depth can be used as an indicator of eutrophication, because an increasing phytoplankton biomass probably has a high influence on water transparency in summer. An estimation of the share of phytoplankton in *Secchi* depth observations in the Gulf of Finland has shown that, during the summer, between 16 and 17% of the attenuation of the light is caused by phytoplankton (HELCOM 2009). Since the water transparency fundamentally affects the penetration of light and the thickness of the euphotic zone, it has a significant influence on the intensity of primary production and on the occurrence of aquatic vegetation in the various depth zones.

In the study area, the clay-turbidity caused by the river waters is strongest during the spring floods. At that time, the lowest transparency measured was only 10 cm in Jomalsundet (April 1999), but low values were also measured in areas outside the inner bays; 50–60 cm in Vådholmsfjärden and Orrengrundsfjärden, and 90 cm in Hudöfjärden (Table 3). The highest values recorded in the area during the whole study period were 650 cm at Station 13 in March 2000 and 600 cm at Station 7 in August 1975.

On average, water transparency increased when moving from the inner areas towards the open sea, so that the lowest mean transparency during the growing season was at Station 1 in Klobbfjärden (excl. Station 25 in Jomalsundet,

**Table 3.** The range of *Secchi* depths (cm) at different sampling stations of Loviisa during the growing seasons (May–October) in 1967–2006.

| Station | Minimum | | Maximum | |
|---|---|---|---|---|
| | cm | month(s), year(s) | cm | month(s), year(s) |
| 25 | 15 | V 94 | 90 | V 96 |
| 1 | 50 | V 85 | 400 | VIII 78, VI 91 |
| 2 | 100 | V 77 | 540 | VIII 75 |
| 5 | 120 | V 96 | 450 | VIII 76 |
| 3 | 110 | V 80,87, X 88 | 500 | VIII 77, VI 79 |
| 4 | 50 | V 85 | 560 | VIII 75 |
| 7 | 60 | V 85 | 600 | VIII 75 |
| R3 | 90 | V 96 | 420 | VII 03 |
| 8 | 90 | V 89 | 590 | VIII 75 |
| 10 | 90 | V 89 | 610 | VIII 75 |
| 13 | 250 | V 06 | 450 | VIII 99 |
| R1 | 40 | V 06 | 320 | VIII 94 |
| R2 | 110 | V 99 | 560 | VI 91 |

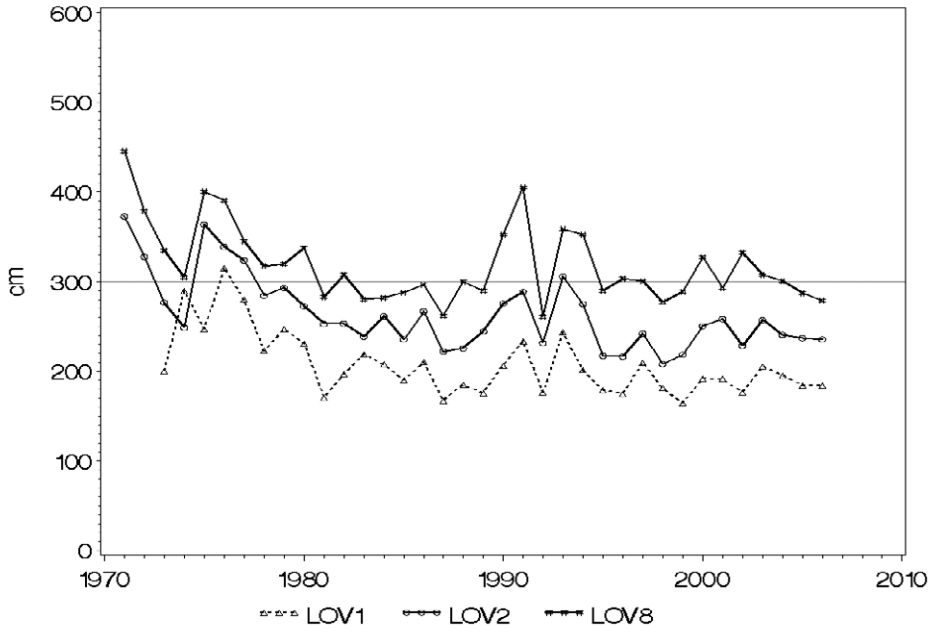


**Fig. 9.** Mean *Secchi* depths (cm) of the growing seasons at the Loviisa stations 1, 2 and 8 in 1971–2006.

which was usually monitored only in the spring) and the highest transparency was measured at Station 7 in Orrengrundsfjärden.

As a whole, the water transparency has clearly decreased in the study area during the 40-year period. What is worth noticing is that the decrease has been quite similar at all sampling stations (Fig. 9) except at the outermost Station 7, where the decrease has been a little smaller (Table 4).

During the growing seasons of 1971–1976, the mean of the average *Secchi* depths was 263 ± 50 cm at Station 1, 322 ± 49 cm at Station 2 and 376 ± 50 cm at Station 8. In 2001–2006, the corresponding values were 190 ± 11 cm, 243 ± 12 cm and 300 ± 19 cm, respectively. This means that the mean *Secchi* depth decreased by 73 cm at Station 1, by 79 cm at Station 2 and by 76 cm at Station 8 between these two 6-year periods. The Student's *t*-test was used to study whether the decrease in the *Secchi* depths was significant. At Station 3 the decrease was very significant and at Stations 5, 2, 1, 8 and 4 significant (Table 4). Irrespective of the fact that the decrease in the mean values was 69 cm at Station 10 and 33 cm at Station 7, the change was not significant, due to large dispersion of the results.

A decrease in summer-time water transparency has been observed in all open sea areas in all the Baltic Sea sub-regions over the last one hundred years (Fleming-Lehtinen et al. 2007). The decrease was most pronounced in the Northern Baltic Proper (from almost 9 m to 4 m) and in the Gulf of Finland (from 8 m to 4 m). In the Bothnian Bay and the Gulf of Finland the water was less transparent at the beginning of the 20$^{th}$ century, which was due to a higher natural turbidity and colouration caused by the leaching of substances from the drainage area. The water transparency status has significantly become lower in the Northern Baltic Proper, the Gulf of Finland and the Gulf of Bothnia (HELCOM 2009). Sandén and Håkansson (1996) statistically tested the trends in the *Secchi* depth values recorded at offshore stations in different parts of the Baltic Sea during two discrete time periods: 1919–1939 and 1969–1991. The tests showed that the *Secchi* depth decreased by ~0.05 m yr$^{-1}$ during both periods. They stated that the decrease could have been due to a rise in the concentration of humic substances, but was more likely to have been induced by an increase in the concentration of algae in the water. These conclusions indicate that primary production in the Baltic Sea has increased, both in the time perspective of the entire 20$^{th}$ century and during the period 1969–1991 (Sandén and Håkansson *op.cit*).

**Table 4.** Difference between the mean *Secchi* depths (cm) in the years until 1976 and in 2001–2006 at eight sampling stations off Loviisa. The Student's *t*-test was used to calculate whether the changes were significant.

| Station | –1976 | 2001–2006 | Difference | df | *t* | p |
|---|---|---|---|---|---|---|
| 3 | 320 | 257 | 63 | 8 | 6.03 | 0.000 |
| 5 | 322 | 263 | 59 | 8 | 4.62 | 0.002 |
| 2 | 322 | 243 | 79 | 10 | 3.80 | 0.004 |
| 1 | 263 | 190 | 73 | 8 | 3.56 | 0.007 |
| 8 | 376 | 300 | 76 | 6.06 | 3.58 | 0.012 |
| 4 | 371 | 310 | 61 | 3.51 | 4.57 | 0.014 |
| 10 | 379 | 310 | 69 | 2.35 | 1.81 | 0.193 |
| 7 | 384 | 351 | 33 | 9 | 1.33 | 0.215 |

In comparison with the above results, the decrease in water transparency has been clear but less pronounced in the inner sea area off Loviisa. Whilst the transparency has decreased 4 or 5 cm $yr^{-1}$ in the offshore areas, the decrease in Hästholmsfjärden and Hudöfjärden has been only 1–2 cm $yr^{-1}$ or 18–25% during the 40-year study period. On the other hand, the decrease in the Loviisa area has been more obvious in the inner areas than at Station 7, which best reflects the conditions in the open Gulf of Finland.

## 1.10 Oxygen

Depletion of oxygen is a common feature in the deep basins of inlets isolated from the open sea, in which the exchange of water is limited. The most critical dates for oxygen problems are the end phases of the stagnation periods, i.e., just before the break-up of the ice in spring and before the first autumn gales. If the hypolimnion is not renewed during the stagnation period as a result of storms, strong changes in sea level or up-welling, lively bacterial decomposition consumes the oxygen reserves to the end in the small-area deeps. In general, the situation improves only when the hypolimnion receives an oxygen supply either from the epilimnion in connection with the spring or autumn turnover, or when more saline oxygen-rich water flows from the open sea into the deeps.

In the sea area off Loviisa, the oxygen reserves of the hypolimnion are generally sufficient in winter. This is due to the small quantities of material to be decomposed in winter and the slowness of decomposition activity at low temperatures. Apart from the winter seasons, the oxygen conditions have been fairly good during most of the year in the whole study area. In general, the oxygen conditions have been good in the epilimnion through the year, but a deficiency or

even depletion of oxygen often appears in late summer and autumn in the near-bottom water. Since the 1970s, low oxygen concentrations have been repeated nearly every year in the deeps of Hästholmsfjärden (Station 3) and Hudöfjärden (Station 10), but especially in the 1990s and 2000s, when there was an almost regular depletion of oxygen at these stations (Fig. 10). In addition, the oxygen situation was at least occasionally problematic at other sampling stations too, for example at Stations 2, 5, 8, R3 and 7. In 1996, the oxygen conditions were very poor, and at the end of August the whole hypolimnion of Hästholmsfjärden seemed to be anoxic, with the water smelling strongly of hydrogen sulphide.

The deeps of Hästholmsfjärden and Hudöfjärden (Stations 3 and 10) are initially problematic with regard to the adequacy of oxygen reserves. Already in the natural state, a deficiency of oxygen was a characteristic of the deeps in late summer (Bagge & Voipio 1967). The character of the bottom in the deep of Hudöfjärden (black watery sludge smelling of hydrogen sulphide) indicated reducing conditions prevailing at the surface of the sediment in the 1960s, when the first samples were taken. Additionally, the degradation of sediments and zoobenthos in the deep of Hästholmsfjärden during the last 30 years indicates a weakening in oxygen conditions at that station. The oxygen problems in the deeps are due to the small volume of the hypolimnion and to the strong temperature and salinity stratification in the water column.

The primary reason for the poor oxygen conditions in the deeps of Hästholmsfjärden and Hudöfjärden is the problematic properties of the water exchange. It is clear that the eutrophication symptoms first become visible just in the oxygen conditions of the small-area deeps. Much smaller problems have been noticed in the oxygen conditions in the sea area off Olkiluoto, where the exchange of water is effective, although the thermal load is larger. The poor oxygen conditions in the Loviisa area are likely to have been caused by the higher quantity of nutrients in the water, characteristic for the Gulf of Finland, compared to those in the Bothnian Sea. The nutrients increase the biological production in the water phase, as well as the amount of organic matter settling onto the bottom to be decomposed there.

The exchange frequency of the near-bottom water in Hudöfjärden and Hästholmsfjärden is limited by the topographical factors typical for the Loviisa sea area, such as narrow and shallow sounds and underwater sills, more than one of these existing when coming from the open sea towards the inner archipelago. Besides the topographical factors, annual differences in weather conditions also have a significant influence on the exchange frequency of the near-bottom water. The restraints on water exchange and the deficiency of oxygen below the sill depth of 7–8 m in Hudöfjärden were already noticed in the first background studies carried out in the area in the 1960s (Bagge & Voipio 1967). Most probably,

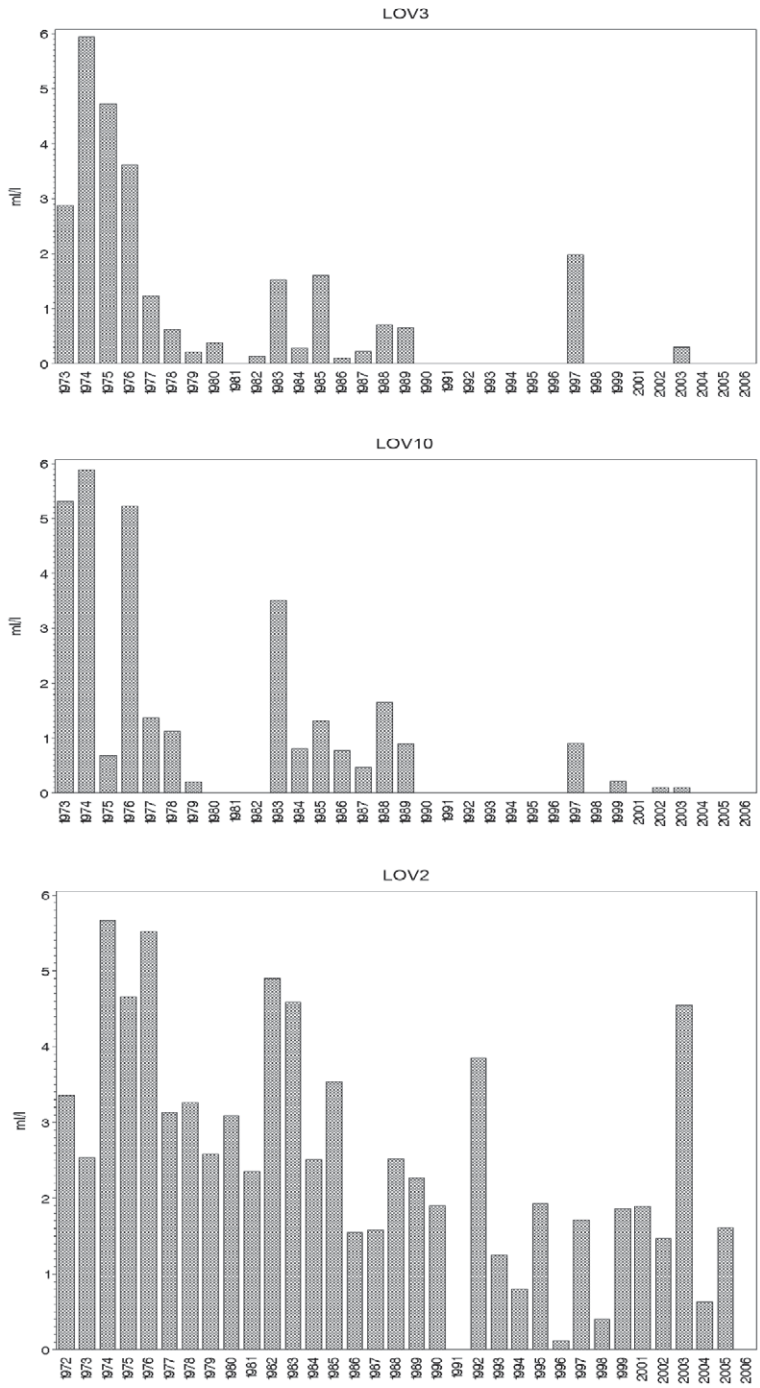


**Fig. 10.** Minimum oxygen concentrations (ml $l^{-1}$) in near- bottom water at Loviisa stations 3, 10 and 2 in 1972–2006.

the thermal discharges have played no part in the depletions of oxygen in the deep of Hudöfjärden, because the thermal effluents may only reach Hudöfjärden occasionally and then weakly in open water circumstances; also the seabed in the deep was already badly disturbed before the construction work of the power plant was started (Bagge & Voipio *op. cit.*).

In the case of the Hästholmsfjärden deep, the primary reasons for oxygen problems have also been the topographical factors described above. Nevertheless, it seems to be likely that the thermal discharges have to some extent increased the susceptibility of the deep area of Hästholmsfjärden to oxygen depletion. This has been affected by:

1. the increase of phytoplankton biomass caused by the warming in the surface water, which has led to revived decomposition in the hypolimnion when dead plankton sinks to the bottom,
2. the settling of the cooling water at the surface of the water recipient, which has led to strengthened thermal stratification in a bay sheltered from winds; this has further led to
3. decreased mixing between the water layers and to further isolation of the hypolimnion, and then
4. oxygen is consumed out of the near-bottom water, if no supplement arrives in connection with exchanges of water, and
5. the increase of temperature in near-bottom water during late summer, which has led to revived decomposition and an increase in oxygen consumption.

The increase in the mean temperature of near-bottom water in late summer in the deep of Hästholmsfjärden (Fig. 11) has been statistically significant ($p_F = 0.0015$).

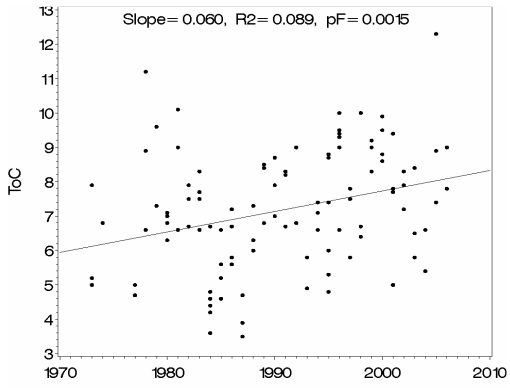


**Fig. 11.** The temperatures of near-bottom water in July at the Loviisa 3 station in 1972–2006.

Stations 3 and 10 have had a central position in the monitoring of the oxygen situation in the study area. However, short-term oxygen depletions appearing in these deeps are not very fateful considering the whole sea area, because fish, for instance, are able to keep away from the deeps when the oxygen situation has deteriorated. On the other hand, the situation is worse for benthic animals, which are exposed to a deficiency of oxygen repeatedly, and the character of the bottom sediment has gone through drastic changes as a result of oxygen depletions.

Extensive bottom areas of the Gulf of Finland have suffered from oxygen deficiency since the mid-1990s, exacerbating internal phosphorus loading and counteracting the decrease in external loading (HELCOM 2003). Low oxygen concentrations have also been observed in several coastal basins along the south coast of Finland. As recently as 2006, it was reported that oxygen concentrations in deep water were lower than ever before in the 2000s (HELCOM 2006), but since then the situation has slightly improved, at least in coastal areas (Baltic Sea Portal 2008b).

Eutrophication has increased primary production in marine ecosystems, and when large quantities of plankton die and sink to the sea floor, large amounts of oxygen are used up during their decomposition. In the Gulf of Finland, the near-bottom oxygen conditions are affected both by inflows of saline water from the Baltic Proper and by local conditions, especially in the heavily-loaded eastern Gulf and in the semi-enclosed basins of the northern archipelago (HELCOM 2003).

Oxygen concentrations in near-bottom waters are controlled by vertical mixing, water exchange and oxygen consumption by aquatic organisms. The oxygen consumption is in turn dependent on the amount of organic matter available for decomposition. Oxygen levels are good indicators of the indirect effect of eutrophication, especially in shallow waters, because they clearly reflect the amounts of organic matter being produced and decomposed. The lowest oxygen concentrations are typically measured at the end of the summer, when the decomposition of sinking organic material uses up oxygen reserves (HELCOM 2003). For fish in open water and animals living on the sea floor, oxygen deficiency causes stress at oxygen levels below 3 ml $l^{-1}$, and at levels below 2 mg $l^{-1}$ the situation becomes critical. To make matters worse, highly toxic hydrogen sulphide ($H_2S$) is commonly produced by chemical reactions in anaerobic conditions (HELCOM *op cit.*).

## 1.11 Nutrients

Nitrogen and phosphorus are the main nutrients at the bottom of all food chains. Primary producers fix these nutrients into their biomass through primary production. When excessive amounts of nutrients enter the sea, primary

production increases rapidly, and the natural ecological balance of the marine ecosystem is disturbed (HELCOM 2003). The nutrient concentrations of water in the study area are affected by the general nutrient levels in the Gulf of Finland, and more locally, by nutrient discharges from neighbouring rivers, local point sources, diffuse pollution and the internal nutrient load. Nutrient-rich waters from the neighbouring rivers are transported to the area especially in spring, while remobilization of nutrients from bottom sediments primarily occurs in the late summer and autumn. The annual load of total phosphorus and total nitrogen from the two major neighbouring rivers and four point sources are given in Appendix 3, and the degree of the internal load is outlined in Chapter 1.3.

The seasonal variation of nutrient concentrations in the study area is great. In the surface water, the concentrations are usually highest in spring, when a copious supply of nutrient-rich river water is flowing into the area, on the one hand through Jomalsundet, and on the other from the outside from east of Boistö. The effect of the river waters is most pronounced in such springs, when the area is covered by ice and the winds are not able to mix the nutrient-rich surface layer. The highest total phosphorus and total nitrogen values measured in Jomalsundet were 187 µgP $l^{-1}$ in April 1980 and 3 110 µgN $l^{-1}$ in April 2000, respectively. The maximum values observed under the ice at Station 2 in Hästholmsfjärden were 160 µgP $l^{-1}$ (in April 1978) and 2 560 µgN $l^{-1}$ (in April 1979). The maximum values observed at Station 4 in Vådholmsfjärden, 260 µgP $l^{-1}$ (in March 2005) and 1 900 µgN $l^{-1}$ (in May 1989), give evidence of the nutrient load coming into the area from the east of Boistö.

During the growing season (May–October), the nutrient concentrations in the surface water were clearly lower. In the 1970s, the average total phosphorus concentrations in the growing season varied between 11 and 30 µgP $l^{-1}$ in the surface water at Station 2, and between 11 and 31 µgP $l^{-1}$ at Station 8 (Fig. 12). In the 1990s, the corresponding means varied between 19 and 43 µgP $l^{-1}$ at Station 2, and between 21 and 46 µgP $l^{-1}$ at Station 8. In early summer, after the vernal maximum of phytoplankton, the concentrations usually decreased, but then increased again towards the autumn in parallel with the increasing impact of river waters as a consequence of autumn rains. Higher phosphorus concentrations have also sometimes appeared in summer, and in recent years these occasions have become more common.

Correspondingly, the average total nitrogen concentration in the growing season varied in the 1970s between 310 and 400 µgN $l^{-1}$ in the surface water at Station 2, and between 270 and 440 µgN $l^{-1}$ at Station 8 (Fig. 13). In the 1990s, the means varied between 440 and 590 µgP $l^{-1}$ at Station 2, and between 420 and 570 µgN $l^{-1}$ at Station 8. In general, the mean total phosphorus values were slightly higher in Hudöfjärden, while the total nitrogen values were higher in

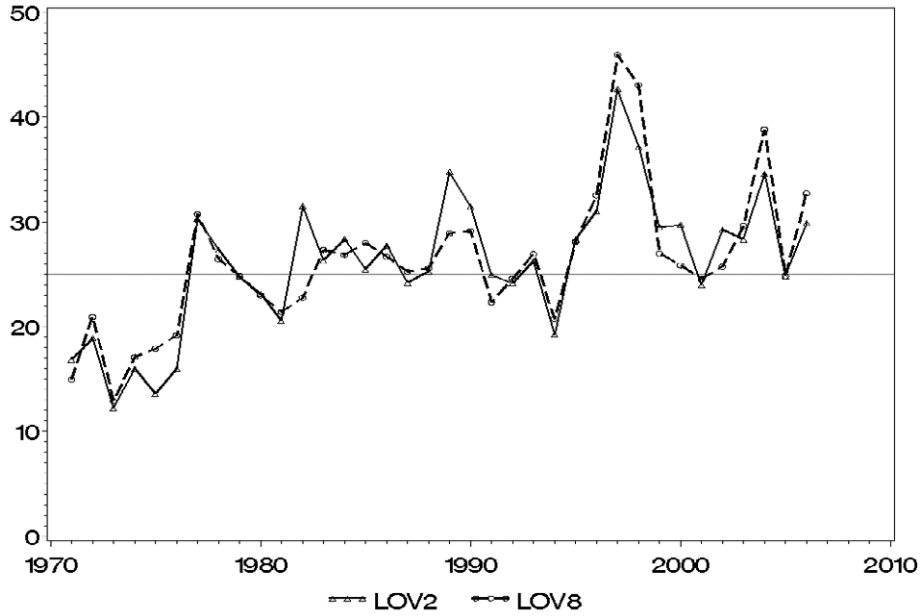


**Fig. 12.** Average total phosphorus concentrations (µgP $l^{-1}$) in the growing season (May–October) in surface water at the Loviisa stations 2 and 8 in 1971–2006.

Hästholmsfjärden, but the curves of the two stations were quite analogous. This difference may reflect the share of river waters in these two areas.

The nutrient concentrations in the near-bottom water were highest in the autumn. The total phosphorus values observed in connection with the oxygen depletions in the deeps of Hästholmsfjärden and Hudöfjärden were particularly high (Figs. 14 and 15). The highest values were recorded in both deeps in 1991. The phosphorus concentration was then 1 147 µgP $l^{-1}$ at Station 3 in September, and 965 µgP $l^{-1}$at Station 10 in October. Later in the 1990s and in the 2000s, the peaks were slightly lower, but elevated phosphorus values were regularly recorded in the end phase of the summer stagnation periods. At other sampling stations, the highest phosphorus values in the near-bottom water were 426 µgP $l^{-1}$ at Station 2 in September 1991, 295 µgP $l^{-1}$ at Station 8 in August 2004, 267 µgP $l^{-1}$ at Station 7 in October 2005, 260 µgP $l^{-1}$ at Station 5 in August 2000 and 113 µgP $l^{-1}$ at Station 4, repeatedly in recent years.

The remobilization of nitrogen from sediment into the near-bottom water has occurred analogously with that of phosphorus, but the intensity has been proportionately weaker. The peak values of total nitrogen in the near-bottom water were 2 665 µgN $l^{-1}$ at Station 10 (in October 1992) and 2 320 µgN $l^{-1}$ at Station 3 (in September 1996). Thus, the values were only about 4–5-fold

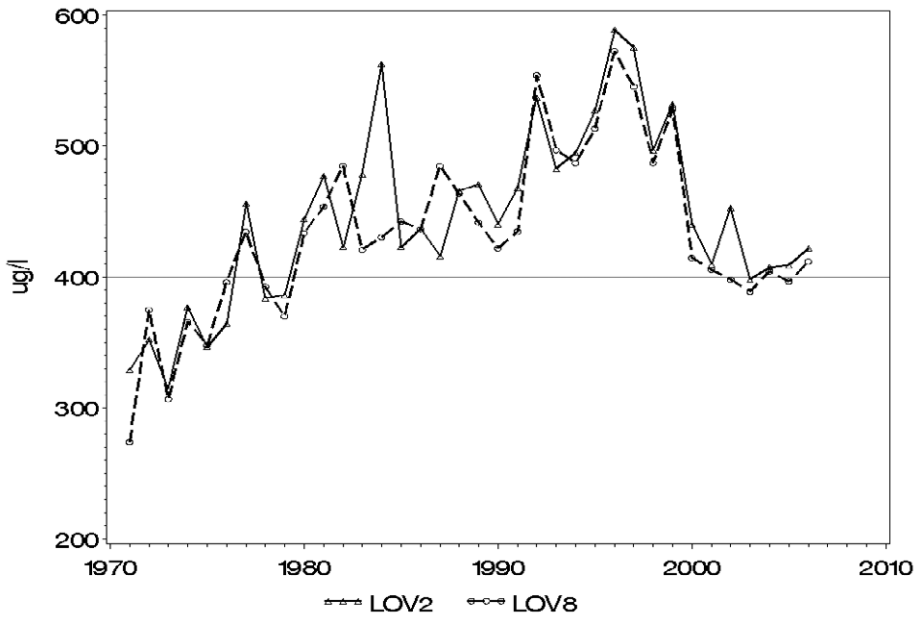


**Fig. 13.** Average total nitrogen concentrations (µgN $l^{-1}$) in the growing season (May–October) in surface water at the Loviisa stations 2 and 8 in 1971–2006.

compared to the average concentrations in surface water, whereas in the case of phosphorus the values were about 50-fold. It is worth underlining that exceptionally high phosphorus and nitrogen concentrations only appeared in the deeps in connection with the oxygen depletions; for most of the year the concentrations were closer to those in the surface water.

Lehtoranta and Mattila (2000) studied the remobilization of nutrients from the sediment in the deep of Hästholmsfjärden. They assessed that 675 kg of soluble reactive phosphorus and 1 860 kg of ammoniacal nitrogen were remobilized from the deep in 1998, and stated that the nutrients released to the near-bottom water were in a form available for algae. Respectively, Kauppila and Bäck (2001) stated that more inorganic phosphorus than inorganic nitrogen is remobilised from sediments, and that the internal load has increased the mass occurrences of nitrogen-binding blue-green algae and the total amount of phytoplankton in the eastern Gulf of Finland. The oxygen situation strongly affects the accumulation of phosphorus in the near-bottom water. In part the reason for this is that the phosphate bound in the sediment begins to dissolve again in the water, and in part that the sedimentation of the phosphates coming from the upper water layers with, e.g., dead organic matter is prevented (Perttilä and Voipio 1981).

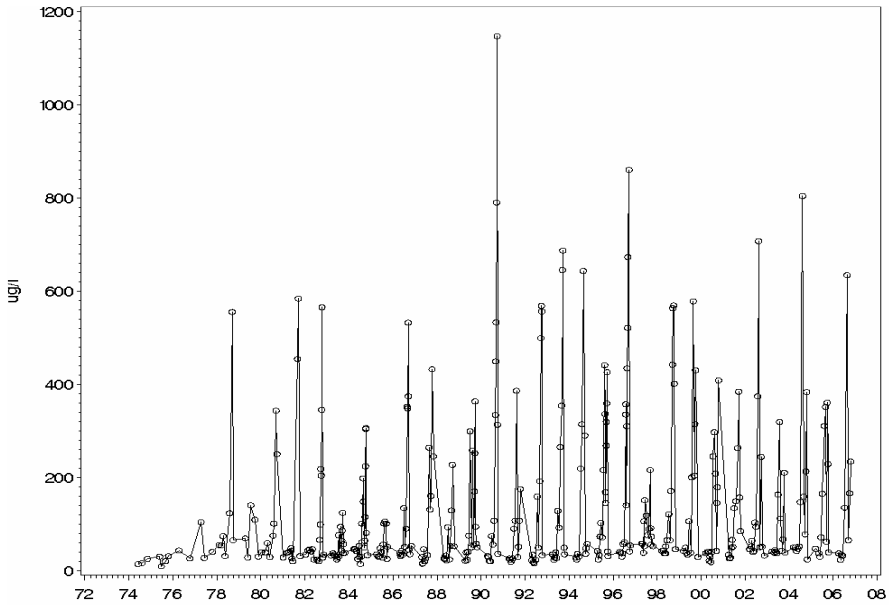


**Fig. 14.** Total phosphorus concentrations (µgP $l^{-1}$) in near-bottom water in Hästholmsfjärden's deep (Station 3) in 1974–2006.

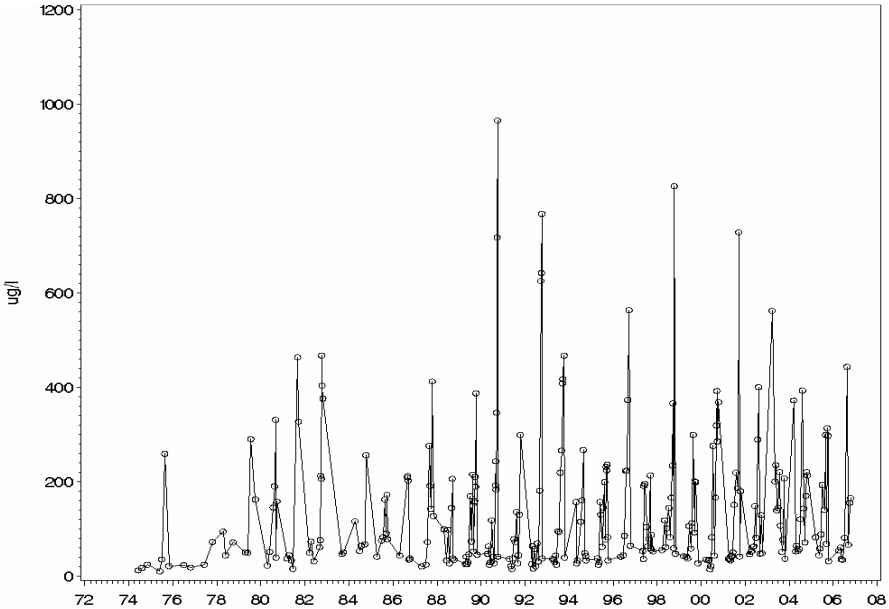


**Fig. 15.** Total phosphorus concentrations (µgP $l^{-1}$) in near-bottom water in Hudöfjärden's deep (Station 10) in 1974–2006.

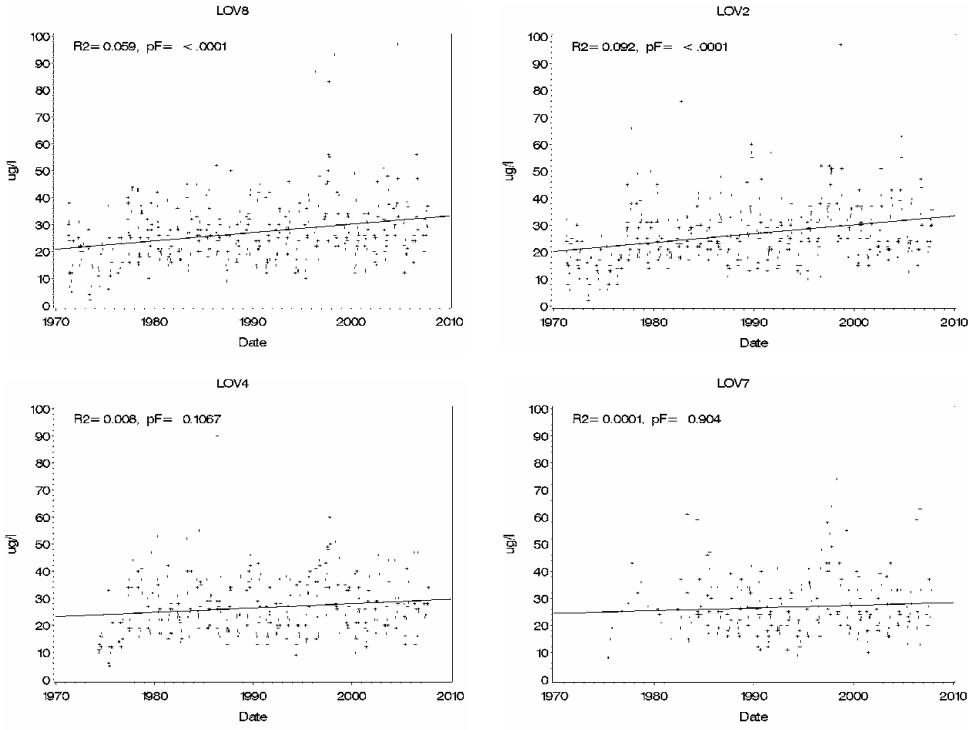


**Fig. 16.** Scatter plots of all total phosphorus measurements of surface water carried out at Stations 8, 2, 4 and 7 during the whole study period at Loviisa. The regression lines show the trends in total phosphorus concentrations.

In general, the nutrient concentrations have increased in the whole study area during the last 40 years. It is noteworthy that the increase has been fairly similar at all the sampling stations. The scatter plots of all total phosphorus measurements of the surface water carried out at Stations 8, 2, 4 and 7 during the whole study period show considerable divergence, but a clear increasing trend in the results (Fig. 16). However, the average concentrations in the growing seasons (Figs. 12 and 13) give a more illustrative picture of the trends. The phosphorus values rose strongly after the mid-1970s and, in spite of a large inter-annual variation, the rising trend continued until 1997, after which the concentrations have begun to decrease. The nitrogen concentrations have increased more steadily, so that the rising trend continued until 1996, after which the concentrations have taken a remarkable downward turn.

The mean concentrations of the 5-year periods demonstrate the parallel developments at five stations located in different parts of the study area (Figs. 17 and 18). On average, the total phosphorus concentrations in the surface water

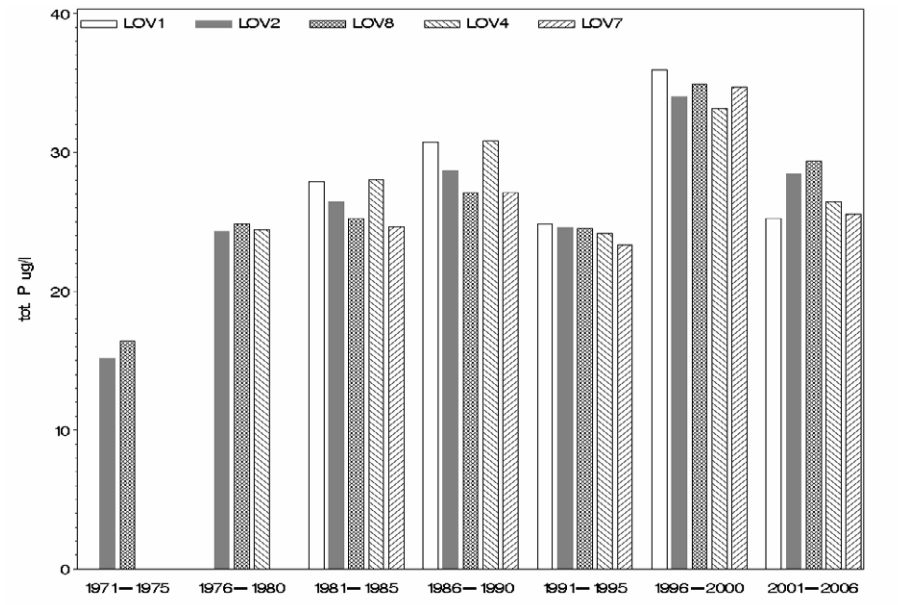


**Fig. 17.** Mean total phosphorus concentrations (µgP $l^{-1}$) in the growing season at Stations 1, 2, 8, 4 and 7 in 1971–2006.

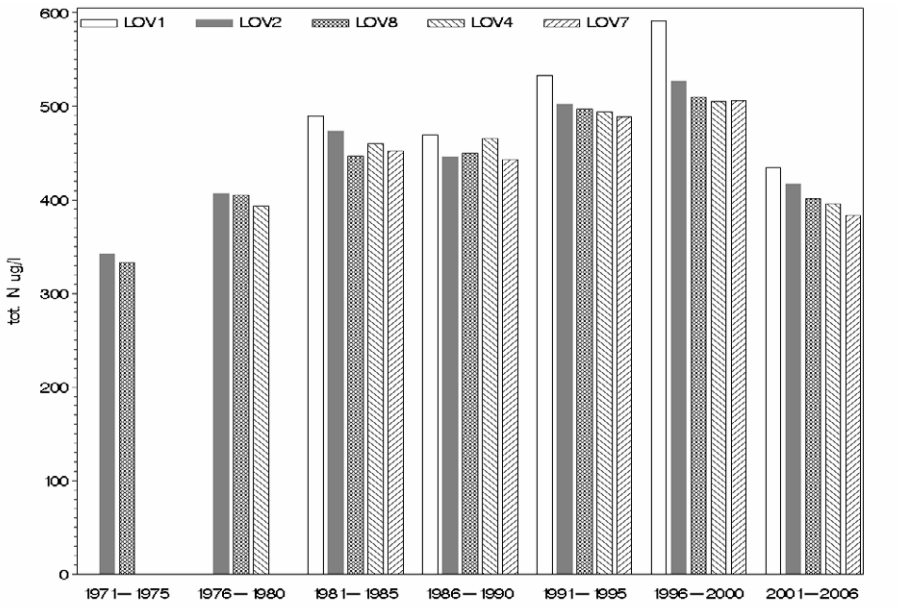


**Fig. 18.** Mean total nitrogen concentrations (µgN $l^{-1}$) in the growing season at Stations 1, 2, 8, 4 and 7 in 1971–2006.

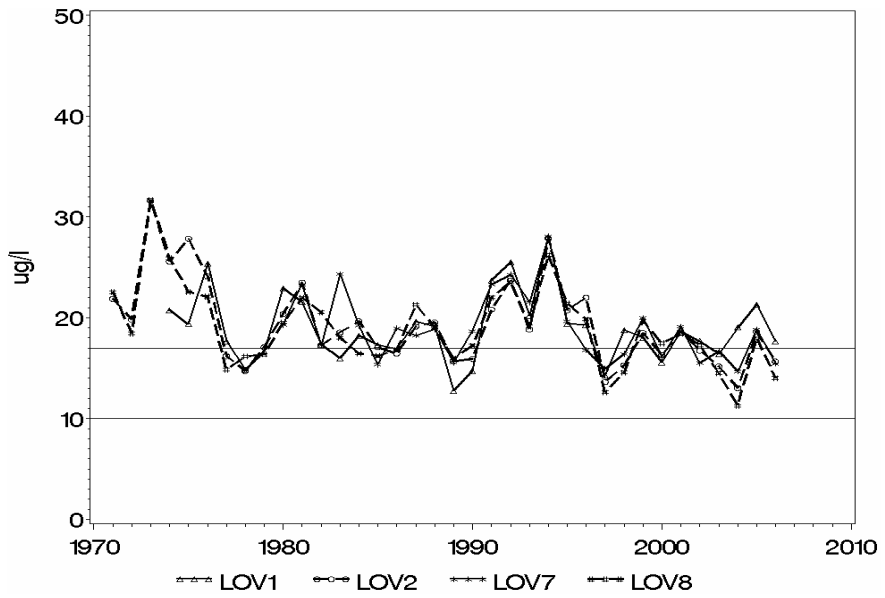


**Fig. 19.** Mean N/P-ratio in the growing season in surface water at Stations 1, 2, 7 and 8 in 1971–2006.

doubled and the total nitrogen concentrations increased 1½-fold over the whole area between the first half of the 1970s and 2000. During the growing season in 1995–2000, the mean phosphorus concentrations were 32–36 µgP $l^{-1}$ and those of nitrogen 510–530 µgN $l^{-1}$. Between the early 1970s and 2000, the change in the mean phosphorus concentration in the growing season was statistically very significant ($p_F < 0.0001$) at Stations 2 and 8, as was also the change in the nitrogen concentration at Stations 2, 4, 5, 8 and 10. The differences between the stations were statistically insignificant. In 2001–2006, the total phosphorus and total nitrogen concentrations were about 1/5 lower than those at the end of the 1990s.

The N/P-ratio in the study area has generally been above 17 during the growing seasons (Fig. 19), which indicates that phosphorus is clearly the limiting factor in the area (*cf.* Forsberg et al. 1978). The mean N/P-ratios were calculated as the means of separate ratios in separate water samples taken during the growing seasons. On average, the ratio has slightly decreased during the latest decade.

Phosphorus concentrations in the surface water have significantly increased in the whole Baltic Sea since the beginning of the 1960s (Perttilä et al. 1980, Perttilä & Voipio 1981). In the open Gulf of Finland, the phosphorus

concentration increased 40% between 1962 and 1978 (Perttilä et al. *op.cit.*). The nutrient load caused by the river waters increased in the 1970s and in the first years of the 1980s due to an average increase in river runoff and probably also to an increase in diffuse pollution. On the other hand, the phosphorus load caused by industry and communities decreased, especially in the 1970s, thanks to water-protective measures (Pitkänen et al. 1987). Diffuse pollution is estimated to cause the largest nitrogen load in the drainage area of the Baltic Sea, whereas communities and diffuse pollution are together the largest sources of phosphorus (Pitkänen 2008). About 25–35% of the total nitrogen load entering the whole Baltic Sea originates from airborne inputs (HELCOM 2003, Pitkänen 2008). The Gulf of Finland is clearly the most loaded sub-area of the Baltic Sea, despite the fact, that its nutrient load decreased by almost 40% between the end of the 1980s and the beginning of the 2000s (Pitkänen 2004). In 2000, Finland's share of the total load of nitrogen into the Baltic Sea was 12%, and its share of phosphorus 9% (HELCOM 2004).

A doubling of nutrient concentrations was noted in the open Gulf of Finland during the 1970s and 1980s. In the mid-1990s, the phosphorus concentrations began to increase again, due to a strengthening of the internal nutrient load (Kauppila and Bäck 2001). The phosphorus concentrations and budget in the Gulf were strongly affected by the enhanced benthic phosphorus input caused by reducing conditions in the sediment-water interface. Especially in the late summers and autumns of 1996 and 2001 the total areas of reduced bottom sediments were large enough to affect the phosphorus concentrations of the whole Gulf (Pitkänen et al. 2003). The nutrients accumulated in the near-bottom water end up in the trophogenic layer in connection with upwellings or strong mixing of water layers. In general, the increase in eutrophication in Finland's coastal waters continued during the 1990s and early 2000s, despite the simultaneous decrease in external nutrient load from direct and river point sources and via the atmosphere (nitrogen). Concentrations of nitrogen decreased slowly, but phosphorus concentrations, especially in the Gulf of Finland and the Archipelago Sea, increased during the 1990s, due to the release of phosphorus from sediments as a result of reducing conditions at the sediment-water interface (Kauppila et al. 2004). The recent decrease in the nutrient concentrations is obviously due to the decrease in the external nutrient load. For instance, the nitrogen load of the Gulf of Finland has decreased by 30% compared to that at the end of the 1980s (Kauppila and Bäck 2001).

Based on the load data given in Appendix 3, the nutrient load from the Loviisa power plant has constituted only a minor share of the total load of nutrients into the study area. In 1990–2006, the total loads of phosphorus and nitrogen from the Loviisa power plant were about 8% of the total point

source input (excl. the river discharges) into the sea area off Loviisa, but only 0.06% of the combined input of the Kymijoki and Tesjoki Rivers. In recent years, the nutrient load from the power plant has even notably decreased, while in particular the phosphorus loads from the fish farms of Semilax and Loviisan Smoltti have clearly exceeded those of the power plant (Appendix 3). In 2000–2006 the phosphorus load from Semilax was 13 times and that from Loviisan Smoltti 6 times higher than that of the power plant. Actually, the input of the internal load has been more significant as a local source than the input from the land-based sources (power plant + Loviisan Smoltti) in the total phosphorus load of Hästholmsfjärden (in 1998 three times higher, see Chapter 1.3). As a whole, however, the nutrient concentrations in the study area are unambiguously determined by the general level of nutrients in the eastern Gulf of Finland.

## 1.12 Phytoplankton primary production

### 1.12.1 *In situ* primary production

*In situ* primary production has been studied with the carbon-14 method at Stations 2 and 8 uninterruptedly for 40 years. In between, measurements were made at Stations 4, 5, R1 and R2 too, but the time-series of Stations 2 and 8 have been continuous. Since the late 1970s, the measurements were made 10–12 times a year; in earlier years the frequency was sometimes lower. The studies were focused on the growing season (May–October), but special attention was paid to the spring time, when the measurements were most frequent.

The studies were initiated in 1967 (Bagge and Niemi 1971). The measurement series is thus one of the longest continuous ones on the Finnish coast. In principle, the method has been kept the same, so that the results from the long study period stand comparison. In recent decades, measurements of chlorophyll *a* have often replaced primary production measurements in obligatory monitoring programmes as a cheaper and less time-consuming method, and the popularity of primary production measurements has declined. However, they still justify their use, because they measure the production itself in the water body. Niemi et al. (1985) stated that the amount of chlorophyll *a* gives some idea of the level of primary production, presupposing that the dynamics of the water body is known. Although the measurement of primary production provides information about the dynamics, it is perhaps not essential for assessing the degree of trophy. According to Leskinen et al. (1986), primary production measurements give information on the trophic status of the environment that cannot be obtained from nutrient and biomass estimates, especially when the nutrient concentration

and biomass are low, but light and temperature conditions are optimal for the growth of phytoplankton.

Primary production measurements were started at the two sampling stations (2 and 8) in April 1967, and have continued since then as an uninterrupted series of measurements. The present writer was engaged in the field work as a summer assistant at the beginning of June 1967, and was then responsible for the measurements from the early 1970s to the 2000s. This has guaranteed that the methods and the results have stayed comparable in the long measurement series. The results from the first 3-year period (1967–1969) were published by Bagge and Niemi (1971). The magnitude of production (*ca.* 30–40 g C (ass.) $m^{-2}$ $a^{-1}$) was found to be typical of the oligotrophic waters of the south coast of Finland. About 1/3 of this amount was produced during the vernal bloom of the phytoplankton. The magnitude of the vernal maximum was about 400–410 mg C (ass.) $m^{-2}$ $d^{-1}$ in 1968, and the daily values during the summer months varied between 69 and 335 mg C (ass.) $m^{-2}$ $d^{-1}$ in 1967–1969 (Bagge and Niemi, *op. cit.*).

The production values given here are net production values, i.e., the dark fixation values have been subtracted. In the natural state, the maximum of primary production occurred at the beginning of the open water period, about 1–2 weeks after the break-up of the ice. The rapid growth of production in spring was caused by the increase in nutrients brought into the trophogenic layer by the vernal turnover and by the changing of illumination conditions to optimal after the disappearance of the ice cover (*cf.* Ilus and Keskitalo 2008). Since the start-up of the power plant, the thermal discharges have changed the ice conditions and made them irregular. The date of the vernal maximum has advanced or become indefinite, depending on the changing ice conditions. On the other hand, the increasing intensity of solar radiation regulates the initiation of the vernal maximum, i.e., the illumination must be high enough for that. In general, the duration of the vernal maximum is very short (Niemi & Ray 1977), and the production decreases rapidly as the nutrient surplus is used up. In the natural state, a clear minimum was seen in June and a weak secondary maximum in August–September (Bagge and Niemi, 1971). The vernal maximum may be over in a few days, but since it may form as much as 1/3 of the annual production, the right timing of the measurements in spring is of crucial importance.

Already before the operation of the power plant was started, the catching of the peak vernal maximum was difficult in such years, when the ice winter was abnormal. After the start-up of the operation, the cooling water has resulted in an occurrence of the maximum at different times in different parts of the study area. Thus, the probability of failing to catch the top of the maximum has increased. Another problem in choosing the starting date for the primary

production measurements in spring is due to the fact that the production may already start quite intensively under the ice. For instance, on the 17th of April 1980, the *in situ* production was already 220 mg C $m^{-3}$ $d^{-1}$ in the surface water at Station 2, although the thickness of the ice was still 45 cm, and the ice was covered by frozen snow preventing penetration of the light into the water. The cooling water has also caused non-coincidence of the production peaks in the Olkiluoto area. The growing season may start 2–3 weeks earlier at stations close to the discharge point, making the estimation of the starting date difficult (Ilus 1983).

The most recent years since the late 1980s have been especially problematic regarding the starting date of the primary production measurements because the winters have been mild with no permanent ice cover at all, or the ice winter has been very short over the whole sea area. In these cases, the timing and significance of the vernal turnover has become indistinct, and the date of the spring maximum has been determined by the physiological readiness of the plankton community and by the moment when the light intensity has become optimal. Eloranta & Salminen (1984) studied primary production in a eutrophied inland cooling water pond, which was open almost the year around. They found that the main effect of the warm effluents on the yearly photosynthesis was the increase of production in the spring months due to the lack of ice cover. The spring maximum of the phytoplankton started there as early as February, and from midwinter till March the factor controlling the phytoplankton growth was light; in April it was nutrients (Eloranta 1980a and b).

In general, the vernal phytoplankton production reaches a peak in late April or early May, when great biological and chemical changes occur in the mixed layer after the break-up of the ice (Niemi 1975). Anyway, since the mid-1990s, the maximum values in Hästholmsfjärden have often been measured already in April. In 2003, the highest vernal production value (740 mg C $m^{-2}$ $d^{-1}$) in Hästholmsfjärden was already measured on the 26th of March, but in other years the illumination has appeared to be too low for notable production in March. In general, the vernal maximum was stronger in Hudöfjärden than in Hästholmsfjärden, which was obviously due to the higher turbidity of the water in Hästholmsfjärden as a result of the higher inflow of river waters into this area, which again limits the thickness of the illuminated layer (*cf.* Ilus and Keskitalo 2008). The highest vernal maximum values measured in the area were 2 170 mg C $m^{-2}$ $d^{-1}$ in Hudöfjärden (1983) and 1 830 mg C $m^{-2}$ $d^{-1}$ in Hästholmsfjärden (1984). The cooling water discharge seems not to have affected the magnitude of the vernal maximum in Hästholmsfjärden.

During the vernal maximum, the bulk of the production occurred in the uppermost 0–3-m layer and the peak value was usually measured just at the

surface (depth 0.5 m). The annual succession and vertical distribution of primary production at Stations 2 and 8 in 1972, 1983, 1997 and 2002 are shown in Figs. 20–23. After the spring phase, when the turbidity of water had decreased, the assimilation spread out into deeper layers, but even then almost all of the production took place in the uppermost 5 metres. However, the peak value was not always right at the very surface. This was probably due to the illumination and UV radiation being too strong at the surface, having an inhibitory effect on photosynthesis there (*cf.* Steemann Nielsen 1964, Ilmavirta & Hakala 1972, Niemi & Pesonen 1974a and b).

During the study period, the focus of daily primary production tended to move from the spring to the late summer (Figs. 20–23). While in the 1970s and 1980s the annual maximum values were almost regularly measured during the vernal bloom, in the 1990s they were often measured in June–July and in the 2000s in July–August. This was especially the case in Hästholmsfjärden (Fig. 24). On the other hand, the vernal maximum values remained relatively low in the 1990s and 2000s. The highest summer production values measured in the area were 1 510 mg C $m^{-2}$ $d^{-1}$ in Hästholmsfjärden and 1 280 mg C $m^{-2}$ $d^{-1}$ in Hudöfjärden on the 31$^{st}$ of July 1997. In the 1970s, the highest summer production values were 656 and 812 mg C $m^{-2}$ $d^{-1}$ in Hästholmsfjärden and Hudöfjärden, respectively.

The mean production in the summer months rose during the study period from 180–540 mg C $m^{-2}d^{-1}$ in 1970–1975 to 590–920 mg C $m^{-2}$ $d^{-1}$ in 1996–2000 at Station 2 in Hästholmsfjärden and from 280–460 to 540–680 mg C $m^{-2}$ $d^{-1}$ at Station 8 in Hudöfjärden (Table 5). In addition, high primary production values were measured in the late 1990s and the early 2000s even in autumn; 1 090 mg C $m^{-2}$ $d^{-1}$ at Station 8 on the 1$^{st}$ of October 1997 and 774 mg C $m^{-2}$ $d^{-1}$ at Station 2 on the 5$^{th}$ of September 2002. As a result of the increased production in summer and autumn, the significance of the vernal maximum has decreased. During the whole study period, the share of summer months represented about 37% of the total annual production in Hudöfjärden, but exceeded 50% in 1980, 1988, 1994 and 2003. In Hästholmsfjärden, the share of summer months was in general about 35%, but exceeded 47% in 1985, 1990 and 1997. The thermal discharges have affected the primary production values in Hästholmsfjärden most significantly in autumn, when the water temperature has already started to decrease elsewhere, but is still high in the discharge area (Ilus and Keskitalo 2008).

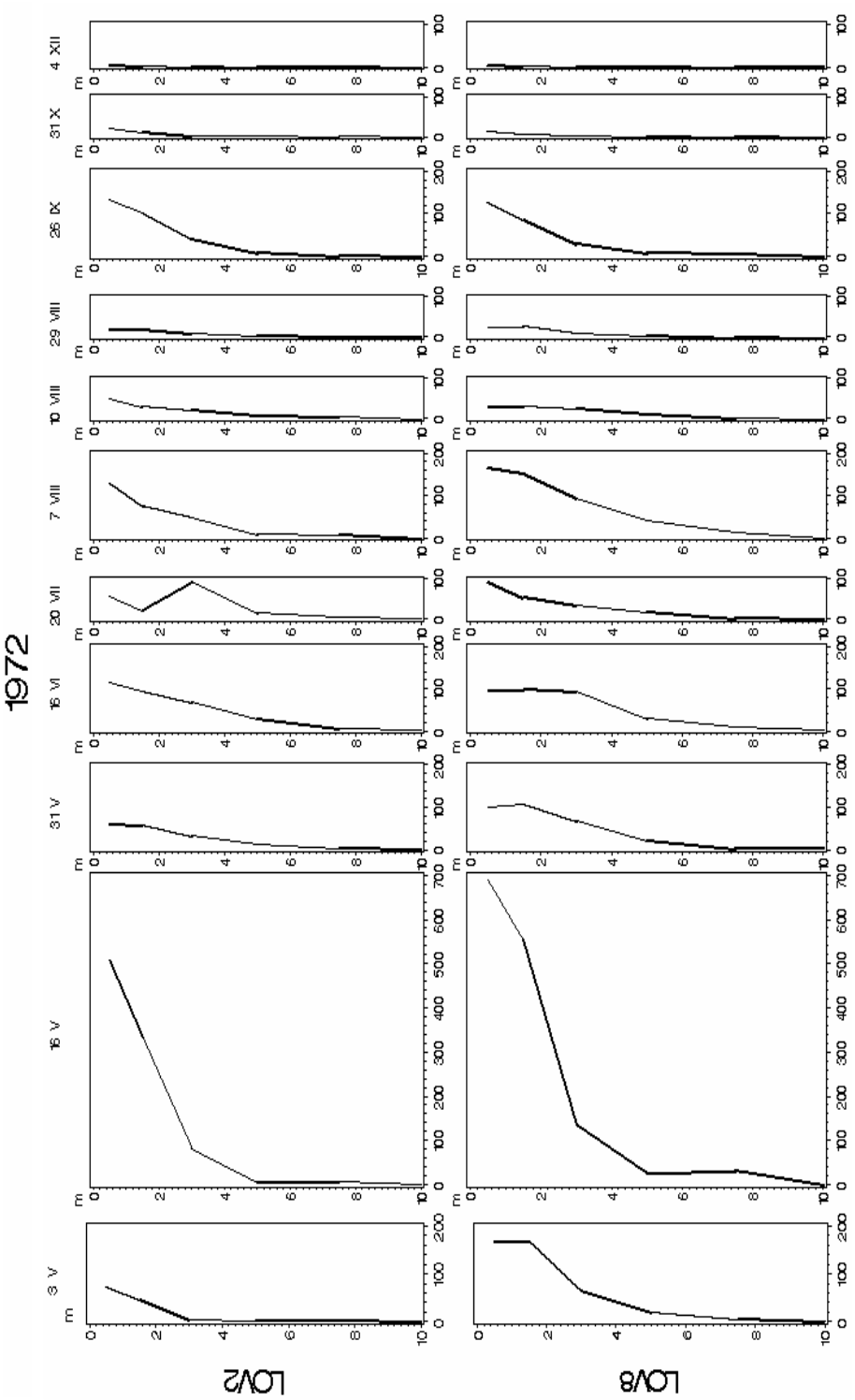


**Fig. 20.** Seasonal fluctuation and vertical distribution of phytoplankton primary production (mg C $m^{-3}$ $d^{-1}$) at the Loviisa stations 2 and 8 in 1972.

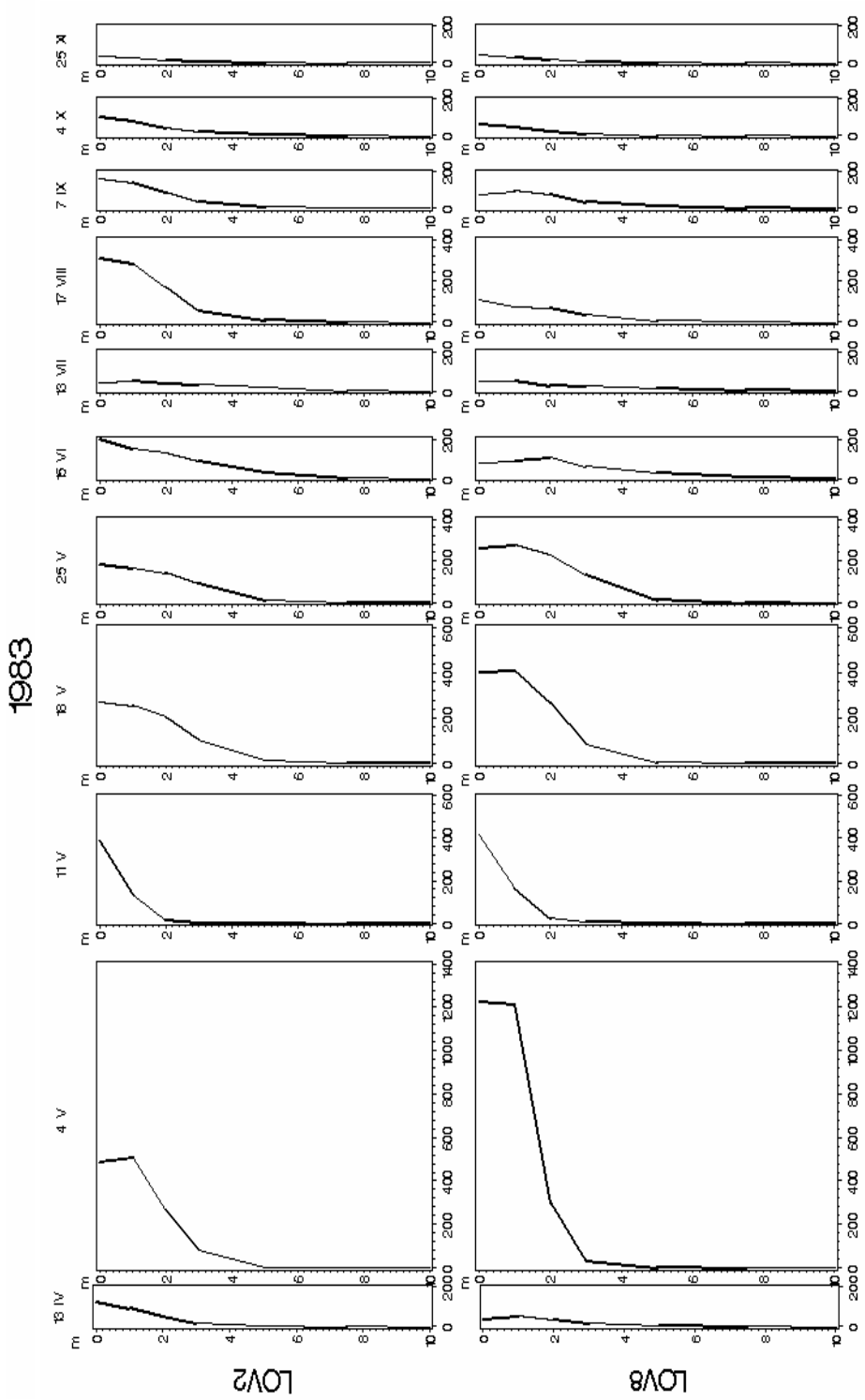


**Fig. 21.** Seasonal fluctuation and vertical distribution of phytoplankton primary production (mg C $m^{-3}$ $d^{-1}$) at the Loviisa stations 2 and 8 in 1983.

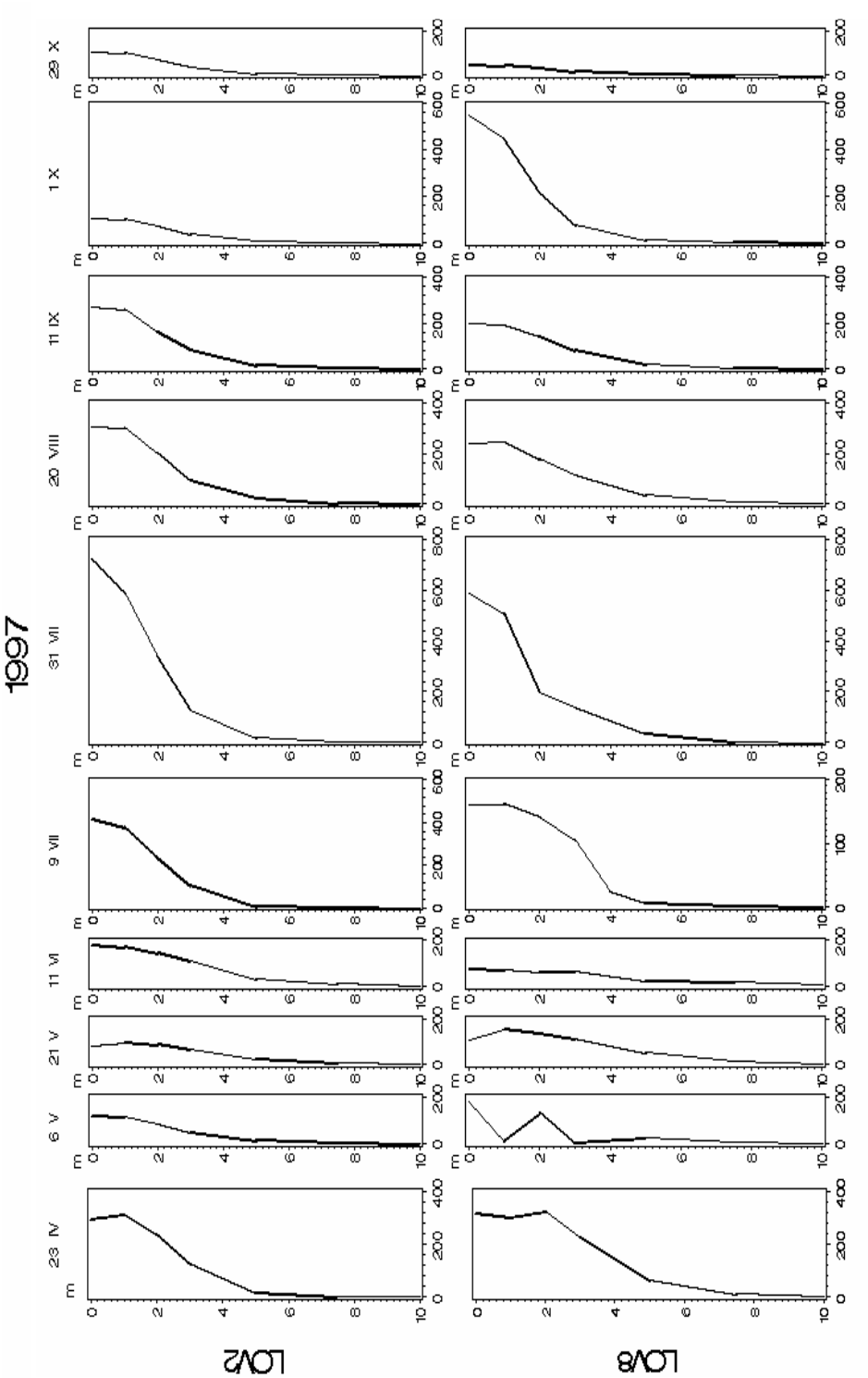


**Fig. 22.** Seasonal fluctuation and vertical distribution of phytoplankton primary production (mg C $m^{-3}$ $d^{-1}$) at the Loviisa stations 2 and 8 in 1997.

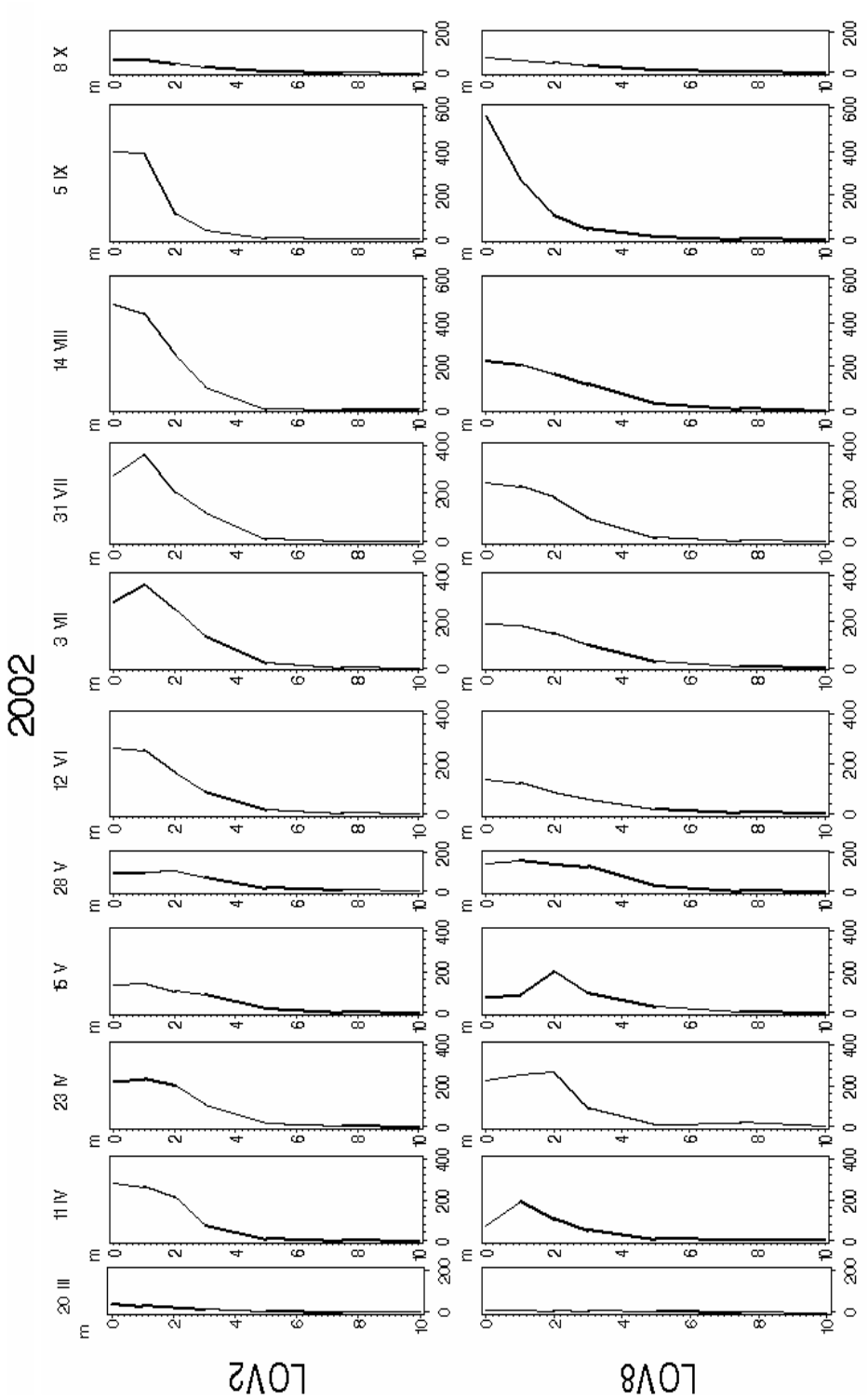


**Fig. 23.** Seasonal fluctuation and vertical distribution of phytoplankton primary production (mg C $m^{-3}$ $d^{-1}$) at the Loviisa stations 2 and 8 in 2002.

**Table 5.** Range of daily primary production in different periods and means of the summer months (June–Aug. $\overline{X}$) at the Loviisa stations 2, 8, 5, 4, R1 and R2 in 1970–2006 (mg C $m^{-2}$ $d^{-1}$).

| Period | Lov 2 | Lov 8 | Lov 5 | Lov 4 | Lov R1 | Lov R2 |
|---|---|---|---|---|---|---|
| **1970–1975** | | | | | | |
| March | – | – | – | – | – | – |
| April–May | 137–1 100 | 180–1 780 | 634–843[a] | 243–1 610[b] | – | – |
| June–Aug. | 66–566 | 82–812 | 411–784[a] | 203–419[b] | – | – |
| June–Aug. $\overline{X}$ | 180–544 | 283–458 | 563[a] | 292–310[b] | – | – |
| Sept.–Oct. | 6.0–455 | 5.9–411 | 191–403[a] | 144–351[b] | – | – |
| Nov.–Dec. | 9.4–66 | 3.7–11 | – | – | – | – |
| **1976–1980** | | | | | | |
| March | – | – | – | – | – | – |
| April–May | 7.9–1 500 | 98–1 170 | 116–1 390 | 93–1 450 | – | – |
| June–Aug. | 195–656 | 190–651 | 178–534 | 111–698 | – | – |
| June–Aug. $\overline{X}$ | 266–557 | 271–489 | 209–433 | 259–439 | – | – |
| Sept.– Oct. | 109–429 | 67–299 | 95–477 | 50–297 | – | – |
| Nov.–Dec. | 10 | – | – | – | – | – |
| **1981–1985** | | | | | | |
| March | 0.6–66 | 0.1 | – | – | – | – |
| April–May | 1.9–1 830 | 64–2 170 | 78–1 200[c] | 509–1 240[c] | – | – |
| June–Aug. | 189–1 070 | 131–1 160 | 249–422[c] | 156–844[c] | – | – |
| June–Aug. $\overline{X}$ | 254–873 | 294–756 | 278–362[c] | 302–413[c] | – | – |
| Sept.–Oct. | 26–485 | 13–387 | 25–488[c] | 15–246[c] | – | – |
| Nov.–Dec. | 2.7–28 | 3.2–17 | – | – | – | – |
| **1986–1990** | | | | | | |
| March | – | – | – | – | – | – |
| April–May | 287–1 580 | 118–1 520 | – | – | 190–720[d] | – |
| June–Aug. | 168–936 | 128–1 130 | – | – | 219–455[d] | – |
| June–Aug. $\overline{X}$ | 350–669 | 371–704 | – | – | 341–358[d] | – |
| Sept.–Oct. | 17–619 | 11–389 | – | – | 66–414[d] | – |
| Nov.–Dec. | 128 | 62 | – | – | – | – |
| **1991–1995** | | | | | | |
| March | – | –- | – | – | – | – |
| April–May | 253–1 370 | 246–1 600 | – | – | – | 433–762[e] |
| June–Aug. | 266–901 | 222–1 210 | – | – | – | 178–573[e] |
| June–Aug. $\overline{X}$ | 458–671 | 350–731 | – | – | – | 407[e] |
| Sept.–Oct. | 140–648 | 77–436 | – | – | – | 90–312[e] |
| Nov.–Dec. | 50 | 26 | – | – | – | – |

[a] = 1973, [b] = 1974–1975, [c] = 1981–1982, [d] = 1986–1987, [e] = 1991

**Table 5.** Continued.

| Period | Lov 2 | Lov 8 | Lov 5 | Lov 4 | Lov R1 | Lov R2 |
|---|---|---|---|---|---|---|
| **1996–2000** | | | | | | |
| March | 60–467 | 240 | – | – | – | – |
| April–May | 349–1 343 | 120–1 280 | – | – | – | – |
| June–Aug. | 285–1 510 | 321–1 280 | – | – | – | – |
| June–Aug. $\overline{x}$ | 588–923 | 538–680 | – | – | – | – |
| Sept.–Oct. | 200–715 | 150–1 090 | – | – | – | – |
| Nov.–Dec. | – | – | – | – | – | – |
| **2001–2006** | | | | | | |
| March | 106–737 | – | – | – | – | – |
| April–May | 106–980 | 182–1 080 | – | – | – | – |
| June–Aug. | 199–1 090 | 179–1 030 | – | – | – | – |
| June–Aug. $\overline{x}$ | 367–962 | 262–624 | – | – | – | – |
| Sept.–Oct. | 132–774 | 126–793 | – | – | – | – |
| Nov.–Dec. | – | – | – | – | – | – |

[a] = 1973, [b] = 1974–1975, [c] = 1981–1982, [d] = 1986–1987, [e] = 1991

Niemi (1975) stated that phytoplankton shows a characteristic seasonal succession, with the production having marked seasonal maxima and minima. He distinguished five different stages of phytoplankton production: a winter minimum, a vernal maximum, a summer minimum, a late summer maximum and a late autumn decline. Fig. 24 shows the seasonal succession of primary production at Stations 2 and 8 in eight years representing different stages of the study period. In 1972 and 1978, the succession followed the schema given by Niemi (*op. cit.*) quite well, having a clear vernal maximum, a summer minimum and a late summer maximum. However, the late summer maximum seemed not to occur until October. The curves for 1984, 1988 and 1995 show further the dominating but decreasing status of the vernal maximum, whereas the share and significance of the summer production is increasing. In the curves for 1998, 2001 and 2002, the significance of the summer and autumn production has further increased and that of the vernal maximum has decreased.

The production curves for 1972 and 1978 are typical for oligotrophic waters having a strong vernal maximum, followed by a minimum at the beginning of the summer and a weak secondary maximum in August–September (*cf.* Bagge and Niemi 1971). In contrast to that, the increase and high level of the summer and autumn values, which are visible in the curves since then, clearly indicate an on-going eutrophication process. A clear strengthening of the summer maximum has been considered to be characteristic of eutrophied waters (*cf.* e.g.Bagge &

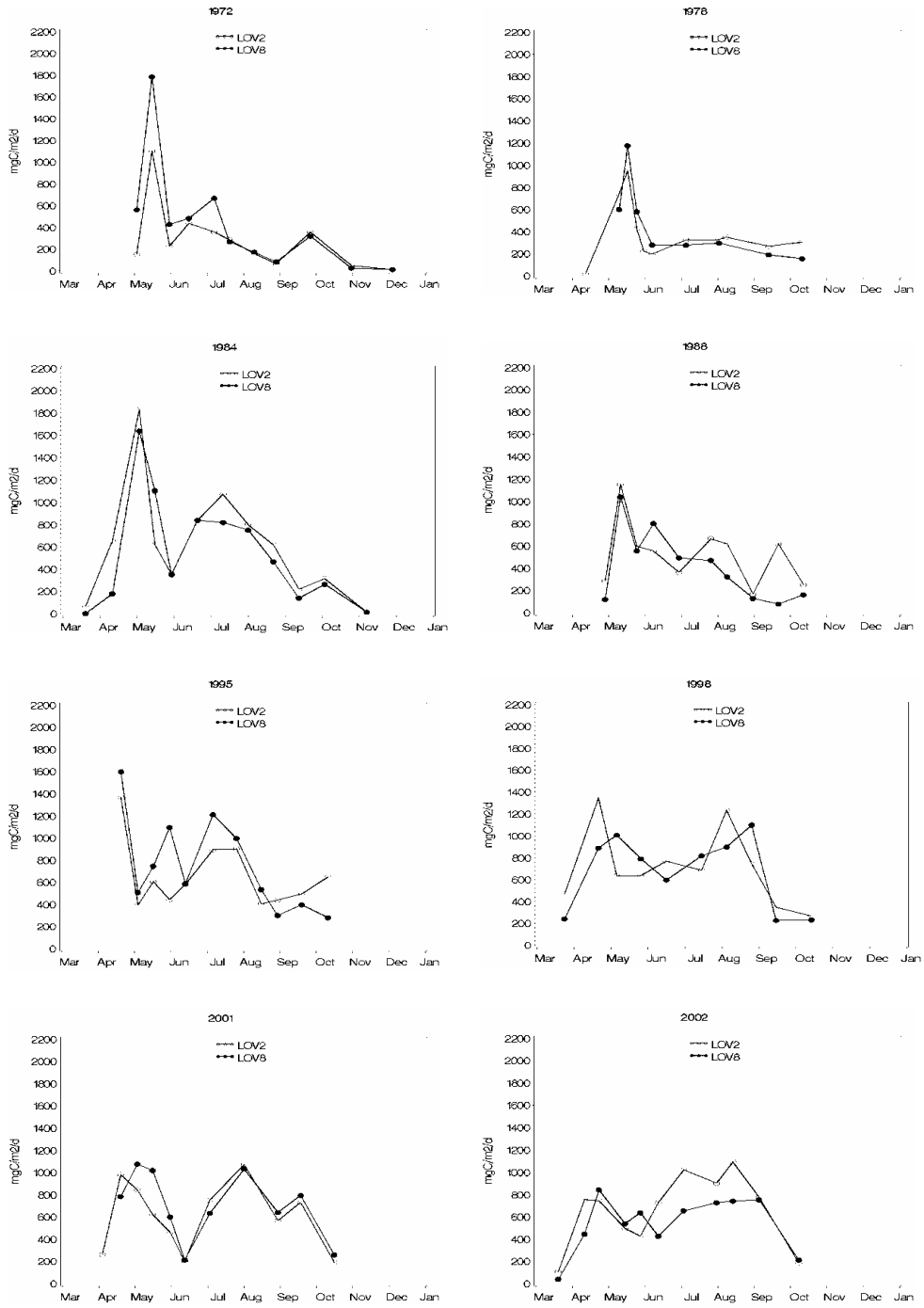


**Fig. 24.** Seasonal succession of phytoplankton primary production (mg C $m^{-2}$ $d^{-1}$) at the Loviisa 2 and 8 stations in 1972, 1978, 1984, 1988, 1995, 1998, 2001 and 2002.

Lehmusluoto 1971, Niemi & Pesonen 1974 a and b). Thus, the production curves give evidence for a progressive eutrophication process in the study area during the last 40 years. However, it is noteworthy that, apart from small differences between the curves, the eutrophication seems to have proceeded quite in parallel at these two stations.

The estimation of annual primary production has been considered to contain uncertainty elements, unless the depth profile is adequately represented and the sampling is frequent enough to cover the temporal variation (Niemi & Pesonen 1974a and b, Leskinen et al. 1986). In our case, the *in situ* measurements were generally made 10–12 times a year, and the vertical distribution can be regarded as adequate in the local conditions. Thus, in spite of the apparent annual variation, the long time-series of annual production results is an excellent interpreter of the development of the trophic level in the Loviisa sea area. The highest annual production values were 158.0 g C $m^{-2}$ $a^{-1}$ at Station 2 and 148.6 g C $m^{-2}$ $a^{-1}$ at Station 8 in 1998, and the second highest were 148.2 g C $m^{-2}$ $a^{-1}$ at Station 2 and 137.7 g C $m^{-2}$ $a^{-1}$ at Station 8 in 1997. While the annual production was on an average 62.1 ± 19.5 g C $m^{-2}$ $a^{-1}$ during the first half of the 1970s (1970–1975) at Station 2 in Hästholmsfjärden, it was 135.3 ± 21.0 g C $m^{-2}$ $a^{-1}$ there during the second half of the 1990s (1995–1999). Thus, the annual primary production more than doubled during this period (Fig. 25), although the values were clearly lower in the 3–4 last years in the early 2000s. It is noteworthy that the rise of annual production was fairly parallel at the two stations.

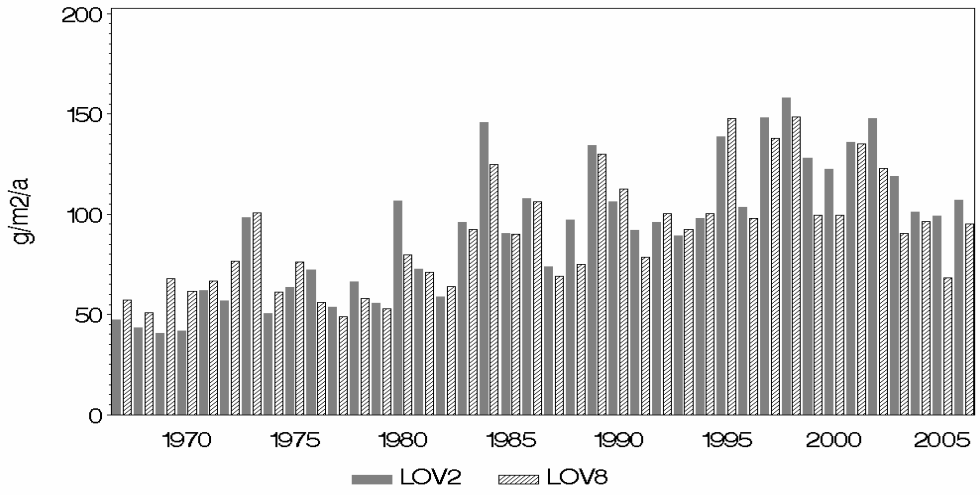


**Fig. 25.** Annual phytoplankton primary production (g C $m^{-2}$ $a^{-1}$) at the Loviisa 2 and 8 stations in 1967–2006.

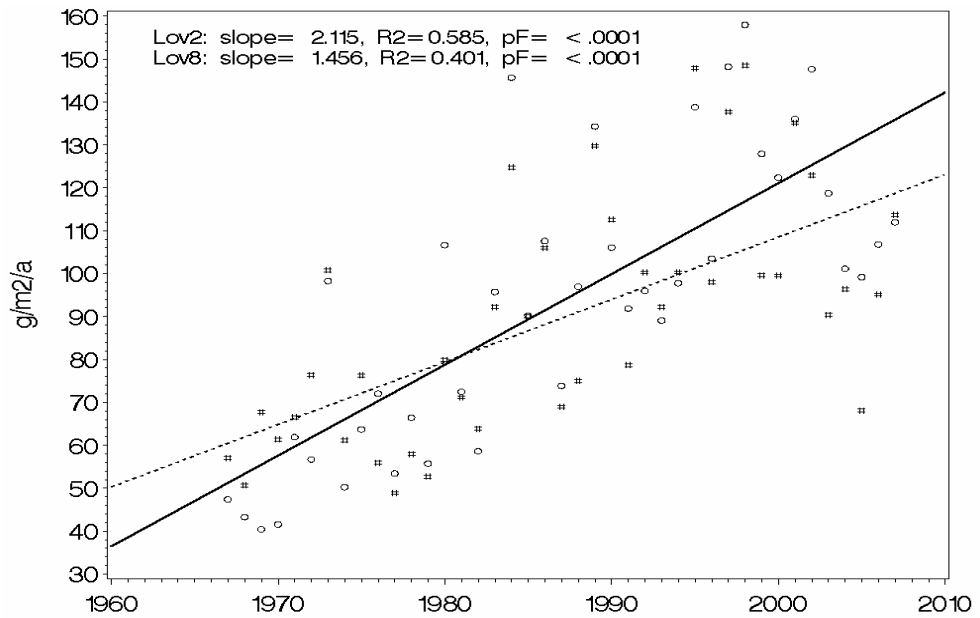


**Fig. 26.** The regression lines of annual primary production at the Loviisa 2 and 8 stations between 1967 and 2006.

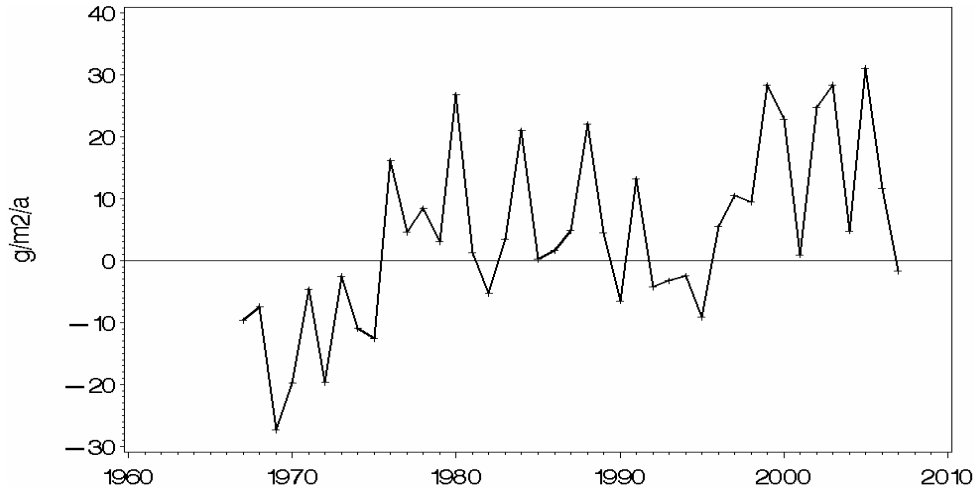


**Fig. 27.** The difference between the annual primary production values (Loviisa 2 – Loviisa 8) in 1967–2006.

The main reason behind the parallel increase of primary production in both the intake and discharge areas of the power plant has certainly been the general eutrophication in the whole area and in the whole Gulf of Finland. However, the thermal discharges have caused a change in the relationship between the annual production of Hästholmsfjärden and Hudöfjärden. In 1967–1975, the annual production was quite regularly higher in Hudöfjärden, but after that the relationship became reversed (Figs. 26 and 27). This change was statistically very significant ($p_F = 0.0001$).

Besides Stations 2 and 8, *in situ* primary production was also measured at Stations 5, 4, R1 and R2 in the 1970s, 1980s and in 1991 (Table 6). Accomplishment of an *in situ* measurement series with 7 sampling depths is laborious; it requires time and human resources and is expensive. Thus, the number of additional measurements had to be kept limited, because it was not wanted to break the continuous time-series at the Stations 2 and 8. The additional measurements were always made in parallel with those at Stations 2 and 8, and thus they give an idea of the simultaneous differences in the sea area. Stations 4, R1 and R2 can be considered as different kinds of reference stations to Station 8, which is situated in the intake area of the cooling water. In Hästholmsfjärden, Station 2 in the middle of the discharge area gives a point of comparison for Station 5 situated just in front of the cooling water outlet.

In general, the annual production was smaller at Reference Stations R1 and R2 than at Station 8, but in some years (1976 and 1982) it was slightly higher at Station 4 than at Station 8, even 25% higher in 1974 (Table 6). In general, however, the values at these two stations were close to each other. Larger differences were probably due to differences in the timing of the vernal or autumn maximum. It should be kept in mind that considerable changes in the primary production and composition of phytoplankton may take place in 2–3 days, irrespective of season (Tarkiainen et al. 1974). The clearly lower annual production at Station R1 in 1986 was probably due to the high turbidity of the water in Påsalöfjärden that weakened the formation of the vernal maximum.

In Hästholmsfjärden, the annual production was generally higher at Station 5 that at Station 2. The differences were probably due to the stronger thermal effect in the close vicinity of the cooling water outlet, or to the contradictory effects of the temperature rise shown in assimilation by phytoplankton passing through the cooling-water systems (*cf.* Langford 1990). Briand (1975) suggested that losses of cells during entrainment were a result of high temperatures, but it was clear from the data that there was no direct correlation, and that the maximum reductions occurred at temperatures in the middle and not the highest ranges. The algae cells driven with the cooling water through the power plant evidently remain in the discharge area for a period of from a few hours to

**Table 6.** Annual primary production at the Loviisa stations 4, R1, R2 and 5 in 1973–1991 (g C $m^{-2}$ $a^{-1}$). The annual production values at Loviisa 4, R1 and R2 are compared to those at Loviisa 8 in per cent (% </> Lov 8) and those at Loviisa 5 to those at Loviisa 2 (% </> Lov 2).

| Year | Lov 8 | Lov 4 | % </> Lov 8 | Lov R1 | % </> Lov 8 | Lov R2 | % </> Lov 8 | Lov 2 | Lov 5 | % </> Lov 2 |
|---|---|---|---|---|---|---|---|---|---|---|
| 1973 | 100.8 | – | – | – | – | – | – | 98.3 | 101.0 | >2.7 |
| 1974 | 61.2 | 76.8 | >25 | – | – | – | – | 50.3 | – | – |
| 1975 | 76.3 | 66.9 | <12 | – | – | – | – | 63.7 | – | – |
| 1976 | 56.0 | 58.6 | >4.6 | – | – | – | – | 72.1 | 73.3 | >1.7 |
| 1977 | 48.9 | 46.2 | <5.5 | – | – | – | – | 53.5 | 58.9 | >10 |
| 1978 | 58.0 | 56.0 | <3.4 | – | – | – | – | 66.5 | 68.8 | >3.5 |
| 1979 | 52.7 | 52.4 | <0.6 | – | – | – | – | 55.8 | 51.8 | <7.2 |
| 1980 | 79.9 | 78.2 | <2.1 | – | – | – | – | 106.7 | 110.3 | >3.4 |
| 1981 | 71.2 | 66.7 | <6.3 | – | – | – | – | 72.5 | 84.4 | >16 |
| 1982 | 63.9 | 65.2 | >2.0 | – | – | – | – | 58.6 | 63.5 | >8.4 |
| 1986 | 106.0 | – | – | 71.1 | <33 | – | – | 107.6 | – | – |
| 1987 | 69.0 | – | – | 64.3 | <6.8 | – | – | 73.9 | – | – |
| 1991 | 78.7 | – | – | – | – | 76.6 | <2.7 | 91.9 | – | – |

perhaps several days (Ilus and Keskitalo 2008), and the Station 5 lies directly in the outflowing cooling water stream.

The annual production values were naturally affected by the annual differences in weather conditions, but the significance of the water parameters was more pronounced. A stepwise regression analysis was used to discover which environmental factors best explain the changes in primary production at Stations 2 and 8. The dependent variable was annual primary production (g C $m^{-2}$ $a^{-1}$), and the independent variables were water temperature (°C), total phosphorus (µg $l^{-1}$), total nitrogen (µg $l^{-1}$), totN/totP ratio, water transparency (cm), air temperature (°C), global radiation (MJ $m^{-2}$) and the sum of sunshine hours. Time-weighted means for the surface layer during the growing seasons were used for the water variables and the means of the monthly means during the growing seasons for the meteorological variables. The meteorological data from the nearest observation station to Loviisa were taken from the Monthly Reports of the Finnish Meteorological Institute.

The regression analysis for the whole study period showed that on the first step the temperature of the water best explained the changes in primary production in Hästholmsfjärden, and thereafter the water transparency and total phosphorus, whereas total phosphorus was the best independent variable in Hudöfjärden and thereafter water temperature and total nitrogen (Table 7).

**Table 7.** Stepwise multiple regression analysis of the environmental factors explaining the annual primary production at the Loviisa stations 2 and 8 in 1967–2006. See the text for the variables. $R^2$ (%) = coefficient of determination. Pr>|t| = probability. Estimate = the slope of the regression line.

| Hästholmsfjärden (Station Loviisa 2) | | | | | Hudöfjärden (Station Loviisa 8) | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Parameter** | **Step** | **$R^2$ (%)** | **Pr>\|t\|** | **Estim.** | **Parameter** | **Step** | **$R^2$ (%)** | **Pr>\|t\|** | **Estim.** |
| Water temperature | 1 | 44 | <0.0001 | 15.33 | Total phosphorus | 1 | 22 | 0.0041 | 1.84 |
| Water transparency | 1 | 33 | 0.0003 | -0.44 | Water temperature | 1 | 18 | 0.0253 | 9.77 |
| Total phosphorus | 1 | 31 | 0.0005 | 2.60 | Air temperature | 1 | 11 | 0.0272 | 13.36 |
| Total nitrogen | 1 | 31 | 0.0005 | 0.25 | Total nitrogen | 1 | 13 | 0.0366 | 0.14 |
| Air temperature | 1 | 17 | 0.0090 | 19.14 | Water transparency | 1 | 10 | 0.0712 | -0.20 |
| tot N / tot P ratio | 1 | 4 | 0.2230 | -1.04 | Global radiation | 1 | 2 | 0.3237 | 0.03 |
| Global radiation | 1 | 1 | 0.5057 | 0.02 | tot N / tot P ratio | 1 | 1 | 0.5403 | -0.46 |
| Sunshine hours | 1 | 1 | 0.9837 | 0.01 | Sunshine hours | 1 | 1 | 0.7221 | 0.01 |
| Water temperature | 2 | 54 | 0.0004 | 11.64 | Total phosphorus | 2 | 35 | 0.0065 | 1.63 |
| Total phosphorus | | | 0.0130 | 1.61 | Water temperature | | | 0.0174 | 9.21 |
| Water temperature | 2 | 52 | 0.0008 | 11.52 | Total phosphorus | 2 | 33 | 0.0004 | 3.16 |
| Total nitrogen | | | 0.0309 | 0.14 | tot N / tot P ratio | | | 0.0281 | 2.06 |
| Water temperature | 2 | 49 | 0.0028 | 11.16 | Total phosphorus | 2 | 32 | 0.0041 | 1.76 |
| Water transparency | | | 0.0789 | -0.22 | Air temperature | | | 0.454 | 11.00 |
| Water temperature | 3 | 64 | 0.0001 | 11.40 | Total phosphorus | 3 | 49 | 0.0001 | 3.10 |
| Total phosphorus | | | 0.0003 | 3.36 | Water temperature | | | 0.0040 | 10.34 |
| tot N / tot P ratio | | | 0.0066 | 2.4 | tot N / tot P ratio | | | 0.0062 | 2.34 |

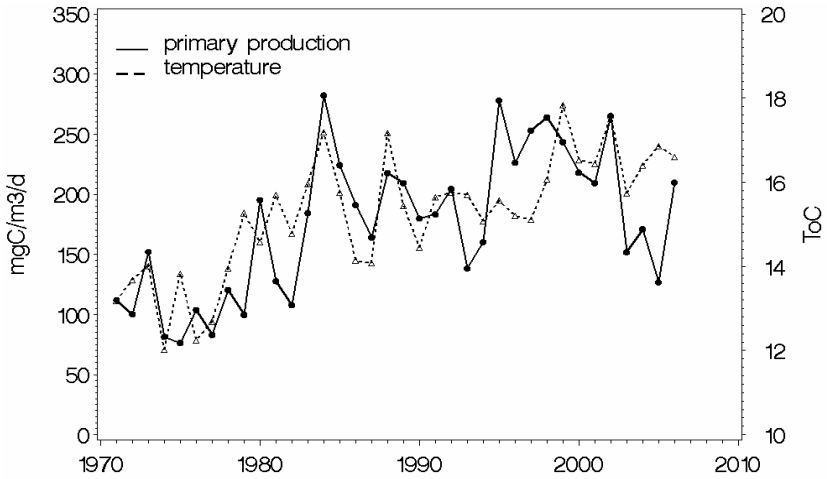


**Fig. 28.** Mean surface water primary production (mg C $m^{-3}$ $d^{-1}$) and the mean temperature of surface water (T °C) during the growing seasons 1971–2006 at Station Loviisa 2.

Water temperature and total phosphorus as a pair best explained the changes in primary production in Hästholmsfjärden and total phosphorus and water temperature those in Hudöfjärden. When three independent variables were used, water temperature + total phosphorus + N/P ratio best explained the changes in Hästholmsfjärden, and total phosphorus + water temperature + N/P ratio those in Hudöfjärden.

Fig. 28 illustrates the parallel succession of primary production and the temperature of the surface water at Station 2 based on the means of the growing seasons. Fig. 29, on the other hand, illustrates the partial dependence of the peak vernal maximum and the simultaneous concentration of total phosphorus in surface water at Station 8.

Lehmusluoto (1968) classified the level of eutrophication based on annual primary production in the different eutrophication zones of the sea area off Helsinki as follows:

| | | |
|---|---|---|
| I | very eutrophic | over 120 g C (ass.) $m^{-2}$ $a^{-1}$ |
| II | eutrophic | 80–120 |
| III | slightly eutrophied | 40–80 |
| IV | uneutrophied | below 40 |

Bagge & Lehmusluoto (1971) reported that, in unpolluted areas of the Gulf of Finland, the annual production values range from circa 15 to 60 g C $m^{-2}$. The classification of Lehmusluoto was, of course, aimed at local use (*cf.* Niemi and Pesonen 1974), but on this basis the annual production in Hästholmsfjärden and

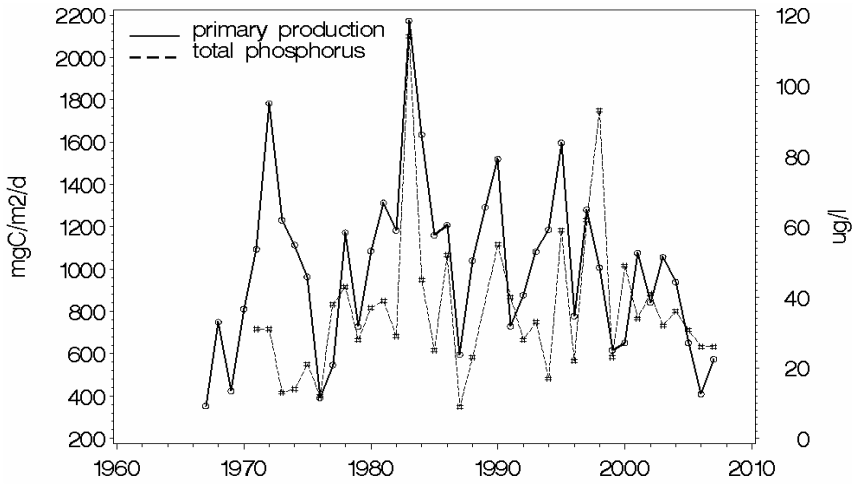


**Fig. 29.** The peak vernal primary production (mg C $m^{-2}$ $d^{-1}$) and the simultaneous total phosphorus concentration in surface water (µg $l^{-1}$) at Station Loviisa 8 in 1967–2006.

Hudöfjärden exceeded the limit of 'very eutrophic' for the first time in 1984, and then in 1989, seven times in the 1990s, and five times in the early 2000s.

Niemi and Pesonen (1974a and b) presented another classification based on daily production values and the ratios between the daily production at the depth of maximum production and that in the whole water column, and between the first-mentioned and the biomass of phytoplankton (all of these during the summer months, June–August). In this classification the limit for 'very eutrophic' was 1 000 mg C (ass.) $m^{-2}$ $d^{-1}$, with the limits for 'moderately eutrophic' being 600–1 000 mg C $m^{-2}$ $d^{-1}$, for 'slightly eutrophic' 300–600 mg C $m^{-2}$ $d^{-1}$ and for 'pure areas' 100–300 mg C $m^{-2}$ $d^{-1}$. In 1967–1971, in the undisturbed waters of the Tvärminne area (western Gulf of Finland), the vernal maximum values were 700–1 000 mg C $m^{-2}$ $d^{-1}$, the summer minimum values were 100–300 mg C $m^{-2}$ $d^{-1}$, the late summer values during the maximum of blue-green algae were 300–500 mg C $m^{-2}$ $d^{-1}$ and the values during the late autumn decline were below 100 mg C $m^{-2}$ $d^{-1}$ (Niemi 1975, 1976).

The limit of 1 000 mg C $m^{-2}$ $d^{-1}$ during the summer months was exceeded in Hästholmsfjärden once in the 1980s, twice in the late 1990s and three times in the early 2000s. In Hudöfjärden, the same limit was exceeded twice in the 1980s, five times in the 1990s and in 2001. Although these two classifications were not generally ratified, they do give a reference for the level of eutrophication in the Loviisa area.

Another area not far from Loviisa, where primary production measurements have been made since the late 1960s, is the sea area off Helsinki. During the 1960s and 1970s the annual production values in the Loviisa area evolved quite synchronously with those in the outer archipelago off Helsinki (Katajaluoto), which was classified as uneutrophied (*cf.* Pesonen 1981). The clear rise observed in the annual production at Katajaluoto during that period was considered to be associated with changes in weather conditions. During 1968–1979, the annual production varied at Katajaluoto between 24 and 99 g C $m^{-2}$ $a^{-1}$ (Pesonen 2003). In 1967–1971, the annual production at Tvärminne was about 50–65 g C $m^{-2}$ $a^{-1}$ (Niemi 1975), while it was 40–62 g C $m^{-2}$ $a^{-1}$ in the Loviisa area.

In the Gulf of Bothnia, the primary production is clearly lower than in the Gulf of Finland. According to our measurements, the annual production in 1972–1982 varied between 22 and 60 g C $m^{-2}$ $a^{-1}$ in the Olkiluoto area. According to Lassig et al. (1978), the mean annual primary production was 18 g C $m^{-2}$ $a^{-1}$ in the Bothnian Bay, 57 g C $m^{-2}$ $a^{-1}$ in the Bothnian Sea, and 101–103 g C $m^{-2}$ $a^{-1}$ in the Gulf of Finland in 1972–1975. Öström (1979) reported an annual production of 24 g C $m^{-2}$ $a^{-1}$ from the archipelago of Luleå (Bothnian Bay). According to Ackefors & Lindahl (1975) and Ackefors et al. (1975), the annual production was 94 g C $m^{-2}$ $a^{-1}$ in the southern Åland Sea in 1973, and about 70 g C $m^{-2}$ $a^{-1}$ in the

northern part of the Bothnian Sea in 1973–1974, but the slightly higher values were probably due to the short incubation time (4 h).

In the Baltic Proper and in the Southern Baltic Sea, the primary production is stronger than in the gulfs. In the northwestern Baltic Proper, the annual level of primary production was estimated at >100 g C $m^{-2}$ $a^{-1}$ in 1970–1971 (Hobro & Nyqvist 1972), and Larsson and Hagström (1982) reported a total primary production value of 160 g C $m^{-2}$ $a^{-1}$ from the outside of the southern archipelago of Stockholm in 1978. A fresh environmental status report of the Skagerrak, Kattegat and the North Sea region (Almroth et al. 2007) states that in the south-eastern parts of the North Sea the production exceeds 350–400 g C $m^{-2}$ $a^{-1}$, while the production in the Skagerrak exceeds 150 g C $m^{-2}$ $a^{-1}$.

### 1.12.2 Primary production capacity

Primary production capacity has been studied at Stations 2 and 8 since 1973, at Stations 1, 3, 4, 5 and 7 since 1977, and at Stations R2 and R3 since 1995. In 1973–1977, the incubation time was 4 hours, but since 1978 it has been 24 hours. Except for the change in the incubation time, the method has been the same throughout the whole study period. The measurements were made in parallel with the *in situ* measurements, so that in general the samples transferred to the incubation bottles for both measurements were taken from the same water sampler. The same difficulties concerning the timing of the *in situ* measurements in spring also applied to the primary production capacity measurements.

In general, the primary production capacity correlated closely with the *in situ* production. Consequently, the highest daily values of PP capacity usually occurred in spring during the vernal maximum of phytoplankton. The top values (1 080–1 230 mg C $m^{-3}$ $d^{-1}$) were measured at almost all the sampling stations during the vernal maxima in 1984 and 1985. Then the maximum value was measured at Station 3, but in general the highest daily PP capacity values were recorded at either Station 2 or 8. In the 1990s, the focus of the PP capacity started to move to the late summer and autumn in the same way as the *in situ* production. In 1992–1996, the highest values were measured at most stations in September or October, and the same trend was also continued in the 2000s.

The highest mean values of primary production capacity during the growing season (V–X) were about 520 mg C $m^{-3}$ $d^{-1}$ at Stations 1 and 2, 450–460 mg C $m^{-3}$ $d^{-1}$ at Stations 5 and 3, 420–430 mg C $m^{-3}$ $d^{-1}$ at Stations 4, 7, 8 and R3, and 320 mg C $m^{-3}$ $d^{-1}$ at Station R2 (Table 8). Although the highest daily PP capacity values were usually recorded at Stations 2 and 8, the mean values for the whole growing season generally decreased when moving from

**Table 8.** Mean primary production capacity in the growing seasons (May–October) 1973–2006 at the Sampling Stations Loviisa 2, 8, 1, 3, 4, 5, 7, R2 and R3 (mg C $m^{-3}$ $d^{-1}$).

| Year | LOV 2 | LOV 8 | LOV 1 | LOV 3 | LOV 4 | LOV 5 | LOV 7 | LOV R2 | LOV R3 |
|---|---|---|---|---|---|---|---|---|---|
| 1973 | 165 | 174 | – | – | – | – | – | – | – |
| 1974 | – | – | – | – | – | – | – | – | – |
| 1975 | 172 | 191 | – | – | – | – | – | – | – |
| 1976 | 256 | 233 | – | – | – | – | – | – | – |
| 1977 | 272 | 287 | 310 | 369 | 269 | 310 | 269 | – | – |
| 1978 | 197 | 125 | 212 | 201 | 123 | 179 | – | – | – |
| 1979 | 154 | 146 | 168 | 151 | 144 | 140 | – | – | – |
| 1980 | 258 | 219 | 302 | 291 | 233 | 260 | 250 | – | – |
| 1981 | 259 | 207 | 291 | 233 | 202 | 229 | – | – | – |
| 1982 | 250 | 244 | 251 | 249 | 234 | 218 | 257 | – | – |
| 1983 | 305 | 253 | 296 | 370 | 269 | 304 | 227 | – | – |
| 1984 | 362 | 369 | 382 | 396 | 319 | 388 | 270 | – | – |
| 1985 | 406 | 327 | 459 | 443 | 387 | 403 | 412 | – | – |
| 1986 | 314 | 260 | 381 | 319 | 251 | 317 | 260 | – | – |
| 1987 | 316 | 254 | 316 | 332 | 246 | 297 | 215 | – | – |
| 1988 | 439 | 297 | 424 | 422 | 382 | 410 | 308 | – | – |
| 1989 | 379 | 366 | 377 | 404 | 392 | 423 | 346 | – | – |
| 1990 | 275 | 220 | 262 | 270 | 217 | 280 | 204 | – | – |
| 1991 | 285 | 181 | 296 | 303 | 250 | 275 | 232 | – | – |
| 1992 | 362 | 336 | 384 | 365 | 323 | 363 | 312 | – | – |
| 1993 | 273 | 251 | 256 | 235 | 242 | 253 | 234 | – | – |
| 1994 | 325 | 305 | 259 | 281 | 252 | 263 | 224 | – | – |
| 1995 | 517 | 403 | 519 | 464 | 402 | 448 | 349 | 323 | 415 |
| 1996 | 400 | 318 | 338 | 371 | 328 | 317 | 218 | 222 | 326 |
| 1997 | 383 | 377 | 502 | 454 | 424 | 422 | 340 | 297 | 389 |
| 1998 | 456 | 385 | 432 | 415 | 396 | 389 | 360 | 289 | 353 |
| 1999 | 387 | 426 | 385 | 365 | 406 | 356 | 427 | 230 | 261 |
| 2000 | 330 | 283 | 332 | 285 | 256 | 304 | 209 | 184 | 244 |
| 2001 | 369 | 372 | 397 | 359 | 380 | 332 | 286 | 276 | 338 |
| 2002 | 379 | 288 | 332 | 352 | 246 | 309 | 204 | 189 | 265 |
| 2003 | 302 | 292 | 284 | 323 | 254 | 291 | 219 | 250 | 266 |
| 2004 | 231 | 248 | 240 | 232 | 217 | 215 | 165 | 184 | 274 |
| 2005 | 199 | 174 | 177 | 195 | 173 | 167 | 163 | 151 | 202 |
| 2006 | 160 | 223 | 149 | 173 | 162 | 147 | 224 | 184 | 189 |

the inner archipelago towards the outer areas, so that on average the highest values were found in Klobbfjärden and Hästholmsfjärden and the lowest in Orrengrundsfjärden and Kejvsalö östra fjärd. The highest average primary

production in Klobbfjärden was probably due to the fact that the nutrient load from the Tesjoki River is strongest in this area, but the high turbidity of the water in spring prevents the formation of a high daily PP capacity during the vernal maximum.

The inter-annual changes in primary production capacity occurred fairly well in parallel at all the sampling stations (Fig. 30) and the values showed a similar succession to those of the annual *in situ* production (Fig. 31). In 1996–2000, the average primary production capacity at Stations 2 and 8 was at least twice that in the early 1970s (Fig. 32), but dropped strongly in 2005–2006. However, a similar change to that demonstrated to have occurred in the relationship of annual primary production between Hästholmsfjärden and Hudöfjärden since the 1970s, was also valid for the mean primary production capacity values of Stations 2 and 8 (*cf.* p. 84). On the other hand, it is noteworthy that the thermal effect of the cooling water has not increased the primary production capacity at Station 5 as compared with Station 2 (Fig. 32).

It is obvious that the increase of primary production and primary production capacity in the study area has primarily been produced by the strong increase of nutrient concentrations in the whole Gulf of Finland. The decreasing trend of primary production capacity from the inner archipelago towards the outer areas reflects the gradients in hydrographical factors, such as the muddiness (transparency) of the water, within the study area. A similar gradient was also seen in the sea area off Helsinki. In the 1970s and early 1980s, the progression of the PP capacity at Sampling Stations 2 and 8 off Loviisa correlated nicely with those from Katajaluoto (a sampling station in the outer archipelago off Helsinki). In 1976–1980, the average PP capacity was 232 mg C $m^{-3}$ $d^{-1}$ at Katajaluoto, while the averages at Stations 2 and 8 off Loviisa were 227 and 202 mg C $m^{-3}$ $d^{-1}$. In the early 1990s, the values at Katajaluoto were much lower than at Loviisa, but again in 1996–2000, the average PP capacity was 368 mg C $m^{-3}$ $d^{-1}$ at Katajaluoto, while the averages at Stations 2 and 8 at Loviisa were 391 and 358 mg C $m^{-3}$ $d^{-1}$ (*cf.* Pesonen 2003).

Lehmusluoto (1969) submitted a classification for evaluation of the eutrophication level of waters based on the mean primary production capacity of the growing season as follows:

| | |
|---|---|
| oligotrophic | below 100 mg C $m^{-3}$ $d^{-1}$ |
| slightly eutrophic | 100–200 |
| eutrophic | 200–1 000 |
| very eutrophic | over 1 000 |

This classification has not been officially ratified, but based on it the values in the sea area off Loviisa generally exceeded the limit of eutrophic, except in the early 1970s and in the most recent years.

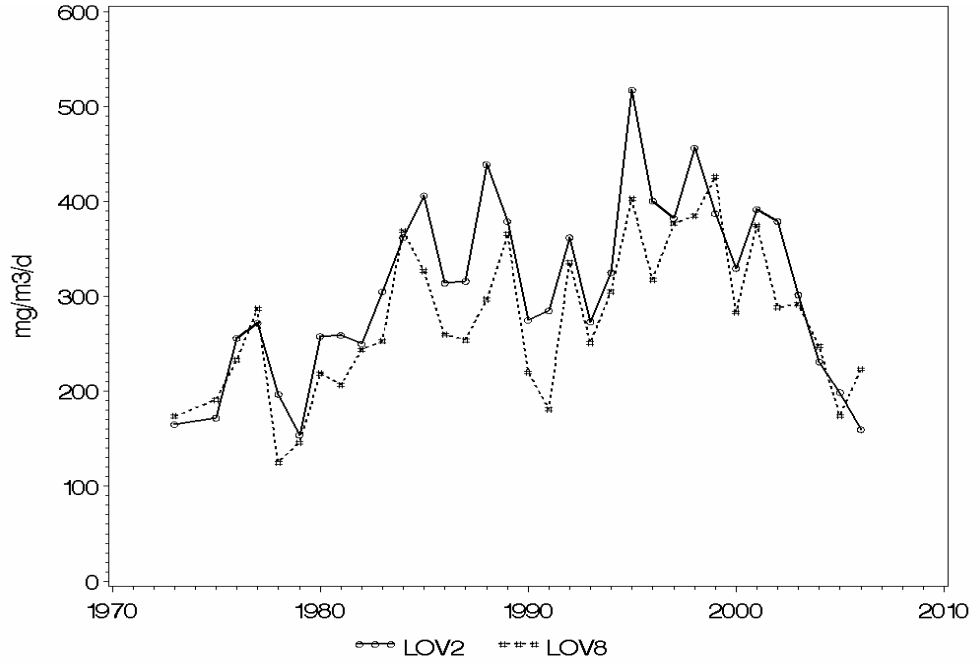


**Fig. 30.** Mean primary production capacity during the growing season (mg C $m^{-3}$ $d^{-1}$) at the Loviisa stations 2 and 8 in 1973–2006.

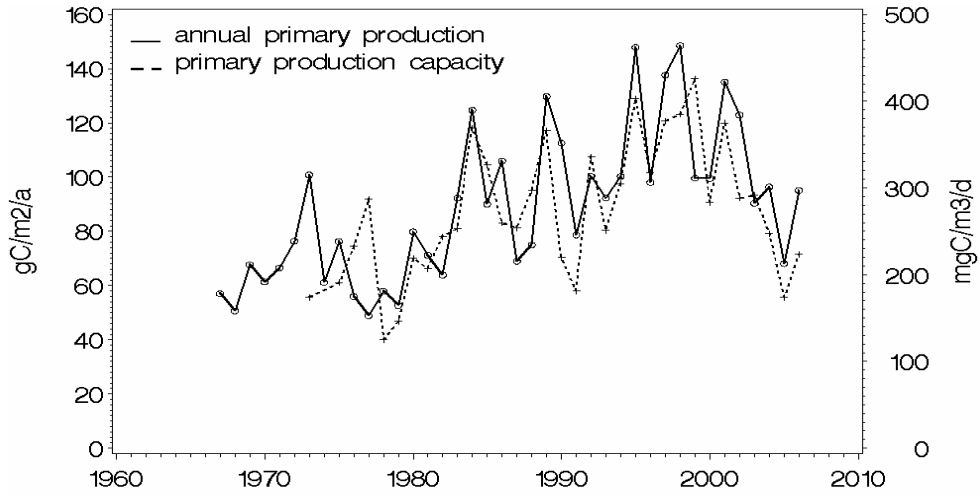


**Fig. 31.** Annual *in situ* primary production (g C $m^{-2}$ $a^{-1}$) and mean primary production capacity (mg C $m^{-3}$ $d^{-1}$) during the growing season at the station Loviisa 8 in 1967–2006.

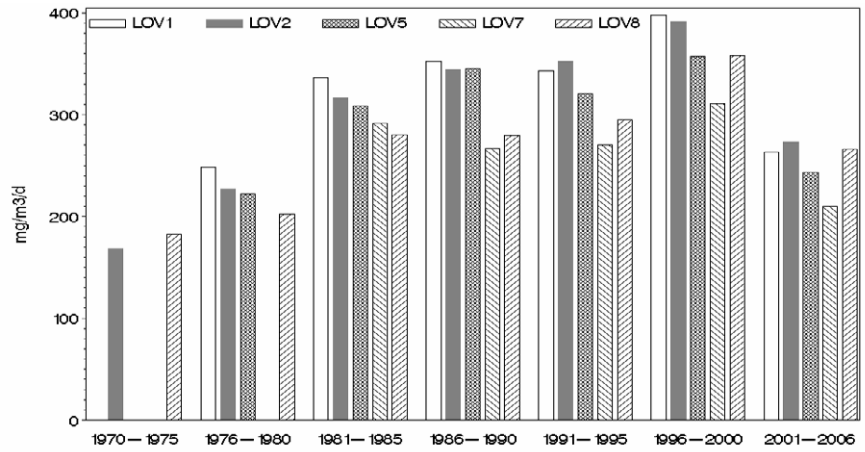


**Fig. 32.** Average primary production capacity (mg C $m^{-3}$ $d^{-1}$) of the growing season in five-year periods at the Loviisa stations 1, 2, 5, 7 and 8.

## 1.13 Littoral vegetation

Littoral vegetation has an important role in aquatic ecosystems. Apart from its significance in primary production, it gives shelter and nourishment for fish and other aquatic organisms. In general, the zonation of the benthic vegetation in the northern Baltic Proper consists of a belt of filamentous algae, a *Fucus* belt and a red algae belt. In the inner parts of the Gulf of Finland, the algal belts are poorer than in the Baltic Proper (*cf.* Hällfors and Niemi 1981), owing to the low salinity of water, but even there they form an important part of the ecosystem.

Aquatic macrophytes are widely used as bioindicators of various kinds of pollution and in monitoring the effects of industrial and sewage effluents. In the Baltic Sea, littoral vegetation has been studied in areas receiving cooling waters, e.g., outside the Swedish nuclear power stations (e.g., Andersson and Karås 1979, Nyquist 1979, 1983, Nyquist and von Braun 1980, Svensson and Wigren-Svensson 1982, 1983, Snoeijs 1985, 1986, 1988, 1990, Snoeijs and Mo 1987, Snoeijs & Prentice 1989, Sandström and Svensson 1990) and at Olkiluoto (Keskitalo and Ilus 1987). In the Gulf of Finland, aquatic macrophytes have also been used as bioindicators, e.g., in the sea area off Helsinki (Viitasalo 1990, Viitasalo et al. 2002, Autio et al. 2003, Ilmarinen and Oulasvirta 2008).

The first survey of aquatic macrophytes on the shores of Hästholmen was carried out in 1971. The study was realized as a transect census by snorkelling. The vegetation was described on nine 100-m transects directed outwards from the shore line. Three transects were situated on the west shore of Hästholmen, one on the south shore and five on the east shore.

In 1975–1982, aquatic macrophytes were studied regularly once a year on five transects, **a**–**e**, the locations of which are given in Fig. 4. During that period, scuba diving and an underwater telephone were brought into use in the studies, so that the diver gave a description of the vegetation by means of an underwater telephone, and this description was tape-recorded. In addition, the transects were studied by dredging using a Luther rake.

Since 1982, surveys of littoral vegetation on the same transects have been repeated every three years. In 1999, transect **f** was brought into the monitoring programme as a new study object. The surveys have been done regularly in late summer, when the vegetation is best developed. The study was implemented with a transect census method described earlier in Ilus (1980), Ilus and Keskitalo (1986) and Keskitalo and Ilus (1987).The results of the 1975–1985 surveys were published in Ilus and Keskitalo (1986). Detailed results from the surveys in 1988, 1991, 1994, 1997, 1999, 2002 and 2005 are given in the Annual Reports written in Finnish (see Appendix 1).

The archipelago area around Hästholmen is characterized by stony shores usually consisting of big boulders. Near the shore line, the bottoms are generally sand or gravel mixed with clay, containing plenty of large boulders. Rocky shores are relatively few, and these become stone fields just below the water level. Sandy and soft-bottom shores exist only here and there. Below water level the bottom generally slopes quite steeply, and therefore the littoral zone inhabited by aquatic plants is relatively narrow. At a depth of 7–8 m, the depth profile usually levels out and the bottom turns to soft clay or mud.

**Transect a**

Transect **a** is situated on the west side of Hästholmen (in the intake area of the cooling water), starting from the base of a craggy cliff. The transect is a steep underwater slope, with a profile sloping rapidly to a depth of 10 m. A uniform stony field extends to a depth of 2 m (to a distance of 10–15 m from the shore line). After the stony field the bottom slopes down, so that at a distance of 60 m from the shore line the depth is 10 m. On the slope, the bottom consists mainly of sand, gravel and stones. Mud starts to occur among the sand and gravel at a depth of 10 m. At the outer end of the transect (depth 13 m) the bottom is quite even, and consists of soft sulphidic gyttja clay.

Due to the slightly higher salinity and transparency of the water, and slightly more marine character of the biota, the number of species was greater (Table 9), the species composition was more diversified and the zonation of vegetation was more pronounced than on the other transects. The bulk of the vegetation occurred on the shallow stony field (depth 0–2 m), and beneath it on the upper part of the sandy slope (at a depth of 2–4 m). Owing to the wide

**Table 9.** List of taxa recorded on transects **a–f** at Loviisa.

| Species | a | b | c | d | e | f |
|---|---|---|---|---|---|---|
| **Cyanophyta (Cyanobacteria)** | | | | | | |
| *Tolypothrix* sp. | – | – | – | – | 1 | – |
| **Chlorophyta** | | | | | | |
| Ulvophyceae* | | | | | | |
| *Cladophora aegagrophila* | 1 | 1 | – | 2 | 12 | – |
| *Cladophora glomerata* | 12(+) | 12(+) | 2(+) | 12(+) | 12 | 2 |
| *Cladophora rupestris* | 12 | 1 | – | 1 | 1 | – |
| *Ulva compressa** (*Enteromorpha compressa*) | 1 | – | – | – | 1 | – |
| *Ulva flexuosa** (*Enteromorpha flexuosa*) | 1 | 1 | – | 12 | 1 | – |
| *Ulva intestinalis** (*E. intestinalis*) | 12 | – | 2 | 12 | 2 | 2 |
| *Ulva procera** (*Enteromorpha ahlneriana*) | 1 | – | 2 | – | 12 | – |
| *Ulva prolifera** (*Enteromorpha prolifera*) | – | – | – | – | 2 | – |
| *Ulva** sp. (*Enteromorpha* sp.) | 12(+) | 12 | 2 | 12 | 12 | 2 |
| Charophyceae | | | | | | |
| *Chara aspera* | 12 | – | – | 1 | – | – |
| *Tolypella nidifica* | 1 | – | – | 1 | – | – |
| **Chrysophyta** | | | | | | |
| *Vaucheria* sp. | 1 | 12 | 2 | 12 | 1 | – |
| **Phaeophyta** | | | | | | |
| *Chorda filum* | 1 | – | – | – | – | – |
| *Dictyosiphon chordarius* | 1 | – | – | – | – | – |
| *Dictyosiphon foeniculaceus* | 12(+) | 1 | – | – | – | – |
| *Ectocarpus siliculosus* | 12(+) | 12(+) | 2(+) | 12(+) | 12 | – |
| *Fucus vesiculosus* | 12 | 12 | 2 | 1 | 12(–) | 2 |
| *Pylaiella littoralis* | 12 | – | 2 | – | – | – |
| *Sphacelaria arctica* | 12 | 1 | – | – | – | – |
| *Stictyosiphon tortilis* | 1 | 1 | – | 1 | – | – |
| **Rhodophyta** | | | | | | |
| *Ceramium tenuicorne* | 12(–) | 1 | – | 1 | 1 | – |
| *Hildenbrandia* sp. | 12 | – | 2 | – | 2 | – |
| *Phyllophora* sp. | 1 | – | – | – | 1 | – |
| *Polysiphonia fucoides* | 12 | 12(–) | 2 | 1 | 1 | 2 |
| *(Polysiphonia nigrescens, P. violacea)* | | | | | | |
| **Bryophyta** | | | | | | |
| *Fontinalis* sp. | 1 | – | – | – | – | – |

1 = in 1975–1985, 2 = in 1994–2006, (–) = declined, (+) = significantly increased
* = nomenclatur according to Hayden et al. (2003).

**Table 9.** Continued.

| Species | a | b | c | d | e | f |
|---|---|---|---|---|---|---|
| **Magnoliophyta** | | | | | | |
| *Ceratophyllum demersum* | – | 12(+) | 2(+) | 2(+) | 2 | 2 |
| *Ranunculus peltatus* ssp. *baudotii* | 12 | 12 | 2 | 12 | 12 | 2 |
| *Myriophyllum spicatum* | 12(+) | 12(+) | 2 (+) | 12(+) | 12(+) | 2 |
| *Callitriche hermaphroditica* | – | – | – | 1 | 1 | – |
| *Potamogeton filiformis* | – | – | 2 | 2 | – | 2 |
| *Potamogeton pectinatus* | 12(–) | 12(+) | 2(+) | 12(+) | 1 | 2 |
| *Potamogeton perfoliatus* | 12(–) | 12 | 2(+) | 12(+) | 12 | 2 |
| *Ruppia maritima* | 1 | 2 | – | – | – | – |
| *Zannichellia palustris* var. *repens* | 12 | 12 | – | 12 | 1 | – |
| *Zannichellia palustris* var. *pedunculata* | 1 | 12 | – | – | – | – |
| *Najas marina* | – | 2 | – | 2 | – | 2 |
| *Phragmites australis* | – | 12(+) | – | 12 | – | 2 |
| *Lemna trisulca* | – | 2 | – | 1 | 1 | – |

1 = in 1975–1985, 2 = in 1994–2006, (–) = declined, (+) = significantly increased
* = nomenclatur according to Hayden et al. (2003).

depth range, the heterogenity of the vegetation was greater than on the other transects, and the species similarity between the years in 1975–1985 was high (Ilus and Keskitalo 1986). Since then, the species composition has changed to become somewhat poorer, and the abundance of the different species varied in the 1990s and 2000s to some extent. In general, the abundance of filamentous algae has increased and that of vascular plants decreased.

The zonation of benthic vegetation followed the pattern described by Hällfors and Niemi (1981), with a clear belt of filamentous algae on the stony field near to the shore, and a *Fucus* belt in the lower part of the stony field. A true belt of red algae was missing, but instead of that there was a clear belt of vascular plants (*Potamogeton perfoliatus, Potamogeton pectinatus* and later *Myriophyllum spicatum*) on the sandy slope below the *Fucus* belt. Red algae were relatively abundant on the transect in 1975–1985, but decreased significantly after that, even though *Polysiphonia fucoides* and *Ceramium tenuicorne* still belonged to moderately or scantily occurring species in the 2000s. In the 1990s and 2000s, the outer part of the transect was empty of haptophytic aquatic plants (from a distance of 35 m and a depth of 6 m downwards). In 1975–1985, the vegetation was sparse below a depth of 5 m, as well, but the lower limit of permanent vegetation was at a depth of 8–9 m (Ilus and Keskitalo 1986).

In 1975–1985, filamentous green, red and brown algae (*Cladophora glomerata, Ulva* spp.*, Ceramium tenuicorne, Ectocarpus siliculosus*) dominated near the shore line. The same species, excluding *C. tenuicorne,* were still the core species in the 1990s and 2000s, and in general the abundance of filamentous algae significantly increased in the whole sublittoral zone. Among others, the abundance of *Dictyosiphon foeniculaceus* increased and in 2002 it belonged to the dominant species. In addition, the abundance of *Ulva intestinalis* increased during the whole study period. Some individual vascular plants (*Zannichellia palustris, Ranunculus peltatus, Myriophyllum spicatum*, *Potamogeton perfoliatus, Potamogeton pectinatus*) were always found to grow between the stones.

A stand of bladder-wrack, *Fucus vesiculosus,* formed a belt at a depth of 1–2 m during the whole study period. The coverage of the stand was 100% until 1979, in 1981 and in 1985, but was only 60–70% in 1980 and 1982. As late as in 1988, the *Fucus* belt was totally covered by epiphytic filamentous algae and was in poorer shape than in 1985, but after that it started to recover. It was still abundantly covered by filamentous algae, but the *Fucus* stand showed distinct signs of strengthening. In 2001, the state of the *Fucus* belt had clearly improved, and the belt had become broader than in 1999. The most abundant epiphytes were *Ectocarpus siliculosus, Cladophora glomerata* and *Ceramium tenuicorne* in the early 1980s (Ilus and Keskitalo 1986), and *Dictyosiphon foeniculaceus, Ectocarpus siliculosus, Cladophora glomerata* and *Pylaiella littoralis* in the late 1990s and in 2000s.

The lower limit of the continuous zone of bladder-wrack has been considered as a classification criterion of water quality. In the Gulf of Finland, the most robust and continuous *F. vesiculosus* belt is observed on exposed shores, where the maximum growth depth is from 5 to 6 m, with the optimum at 2 to 3 m. The maximum growth depth varies geographically, with a decreasing trend towards the east, and correlates with the light intensity (Bäck & Ruuskanen 2000). In a water vegetation survey carried out in 2007 in the sea area off Helsinki and Espoo, the *Fucus* belt was recorded to reach to a depth of 4.9 m in the eastern outer archipelago of Helsinki (Ilmarinen and Oulasvirta 2008). On transect **a** in Loviisa, the lower limit of the continuous *Fucus* belt was about 2 m during the whole study period. However, single poorly growing, but haptophytic, *Fucus* specimens were found to a depth of 6–7 m in 1975–1985 (Ilus and Keskitalo 1986) and to a depth of 5.5 m in recent years.

The sandy slope below the stony field at a depth of 2–4 m was characterized in 1975–1985 by a growth of vascular plants and brown algae. During that period, the brown alga *Chorda filum* decreased, the vegetation was mosaic-like and the stands of single species were relatively thin. The slope was dominated by vascular plants: vigorous stands of *Potamogeton perfoliatus* and *P. pectinatus*

and a little fewer specimens of *Ranunculus peltatus, Myriophyllum spicatum* and *Zannichellia palustris* (Ilus and Keskitalo 1986). Since then, the abundance of vascular plants has declined, the species composition has become reduced and the abundance of filamentous algae has increased. In the late 1990s, the share of *Myriophyllum spicatum* increased, but *Chorda filum, Ruppia maritima, Stictyosiphon tortilis* and *Dictyosiphon chordarius* were not observed after the 1980s. *Chara aspera* was not observed in the 1990s, but was again seen in the 2000s.

Because transect **a** is situated in the intake area of the cooling water, and the spread of thermal discharges to the site is likely to be small, it can be considered as a reference site for the other transects when following general trends in the state of the littoral vegetation in the study area. In the 1980s, a strong increase was observed in the abundance of filamentous algae, and on the other hand, a decline in the *Fucus* belt and the growths of vascular plants. The abundance of red algae decreased significantly from that observed in the 1970s and early 1980s. In the 1990s, *Fucus* and the vascular plants seemed to recover, at least slightly, but the filamentous algae were left as their epiphytes and especially seemed to hinder the subsistence of the vascular plants.

**Transect b**

Transect **b** lies in a small bay on the south shore of Hästholmen. The distance by water from the cooling water outlet is ca. 1.8 km. Systematic monitoring of water temperature has not been done in the bay, but it is evident that warm water may travel around the SE point of Hästholmen to this sheltered and shallow bay, at least in winter and during calm weather in summer. The transect starts from two big boulders at a depth of 1 m and runs evenly and smoothly through shallow waters to a distance of 60 m from the boulders, after which the depth remains about the same, the maximum depth at the outer end being 3 m. The bottom consists of sand mixed with gravel, silt and stones with a layer of whirling organic matter on the surface, the amount of which has increased during the past few years.

In 1975–1982, a rare *Phragmites australis* stand grew at the bottom of the small bay and reached to the starting point of the transect. The transect was characterized by abundant occurrence of *Cladophora glomerata, Potamogeton perfoliatus, P. pectinatus* and *Fucus vesiculosus*. The vegetation was abundant throughout the transect, the heterogenity was low and the species similarity between the years was mostly high (Ilus and Keskitalo 1986). The *Phragmites* stand remained unchanged until 1982, but was wider and denser in 1985. *Cladophora glomerata* was found in all parts of the transect, but was more abundant near the shore. *Potamogeton perfoliatus* occurred extensively along the transect and showed only minor changes during the period 1975–1985.

*P. pectinatus* was abundant at a depth of 1.4–2.0 m, where its coverage, at its greatest, was 80%. *Myriophyllum spicatum* was not very abundant in 1975–1982, but increased in 1985. Until 1979, a dense stand of *Fucus vesiculosus* occurred in the stony area in the middle of the transect, but declined after that. Filamentous algae (*Cladophora glomerata, C. rupestris, Ulva flexuosa, Ectocarpus siliculosus* and *Vaucheria* sp.) covered the bottom almost totally, and in 1982–1985 tall plants were thickly covered by epiphytes (mainly *E. siliculosus*). *Ceratophyllum demersum*, which is known to favour eutrophied waters, was found for the first time on the transect in 1985.

Since the 1970s, several changes in the vegetation and bottom deposits have indicated increasing eutrophication on the transect:

1. The *Phragmites* stand at the bottom of the bay has strongly expanded and become taller and denser. In 1999, stems of *Phragmites* started to become abundant at a distance of 30 m from the starting point of the transect, and an unbroken dense reed stand reached to 25 m. *Potamogeton pectinatus* and *Myriophyllum spicatum* grew abundantly inside the reed stand and all the higher plants were covered by epiphytes (*Ectocarpus siliculosus* and *Cladophora glomerata*). In total, the growth was very dense.
2. The abundance of *Ceratophyllum demersum* has strongly increased. It was observed for the first time in 1985, and by 1999 its coverage was 20–30% at a distance of 35–40 m from the starting point. *Lemna trisulca* and *Najas marina* were observed on the transect for the first time in 1994 and 2002, respectively.
3. Red algae have disappeared; the last observation of *Ceramium tenuicorne* was in 1985.
4. The amount of soft whirling mud on the bottom has increased.
5. The abundance and the height of *Myriophyllum spicatum* and *Potamogeton pectinatus,* and the height of *Potamogeton perfoliatus* have increased in the middle and outer parts of the transect.
6. The abundance of filamentous algae has increased (esp. *Ectocarpus siliculosus and Cladophora glomerata*) on the transect.

The *Fucus vesiculosus* population on the stony field in the middle part of the transect weakened strongly in the early 1980s, but recovered in the 1990s and formed a clean and robust growth in 1999. The vegetation was slightly more one-sided and sparser in 2002 than in 1999, but returned in 2005.

The bay where the transect is situated is very shallow and sheltered. Warmed water occasionally drifts into the bay around the southeast tip of Hästholmen. Due to the sheltered nature of the bay, the warm water does not mix with the existing water mass, but in favourable weather conditions

stays there unmixed. Thus, the warmed water has probably contributed to the eutrophication process in the bay.

**Transect c**

Transect **c** is situated at the southwest edge of Hästholmsfjärden in a small cove on the north shore of Tallholmen (Fig. 4). Its distance from the cooling water outlet is 0.8 km. The north shore of Tallholmen is repeatedly exposed to the warm water plumes, and the growing season at the transect is significantly prolonged due to the absence of ice in early spring (*cf.* p. 45). The transect starts from the base of a steep cliff. After a narrow stony field at the shore line, the sand-silt bottom slopes down quite evenly within 70 metres to a depth of 10 m. The outer end of the transect extends to the soft-bottom area of Hästholmsfjärden.

The first surveys on the transect were done in the late 1970s. The stony field close to the shore line was then occupied by filamentous green algae (*Cladophora glomerata* and *Ulva* spp.), *Potamogeton* species (*P. perfoliatus* and *P. pectinatus*) grew sparsely on the sandy slope, and the lower limit of the haptophytic vegetation was at a depth of 7 m, 50 m from the shore line.

The vegetation started to increase strongly after the mid 1980s. In 1988 and 1991, tall vascular plants (*Potamogeton perfoliatus, Myriophyllum spicatum* and *P. pectinatus*) formed a dense stand at a depth of 2–3 m, reaching the surface of water, the plants being covered by a 'veil' of *Cladophora glomerata* (1988) and *Vaucheria* sp. (1991). In places the growth was so dense that it made the movement of the diver difficult. The stony field close to the shore line was covered by filamentous algae, *Ulva* spp., *Vaucheria* sp., *Ectocarpus siliculosus* and *Cladophora glomerata.*

The eutrophication process continued in the 1990s. *Ceratophyllum demersum* was observed moderately on the transect for the first time in 1994. Robust, dense stands of tall pondweeds (*Potamogeton* species) and spiked water milfoil (*Myriophyllum*) superdominated at a distance of 15–25 m from the shore line. A vital epiphyte growth consisted of *Pylaiella littoralis, Cladophora glomerata, Vaucheria* sp., *Ectocarpus siliculosus* and *Ulva* spp. In 1997, the general impression was of very luxuriant vegetation. A dense, almost impenetrable 'jungle' occurred at a distance of 15–28 m from the shore line. The dominant species was *Potamogeton perfoliatus*. *Myriophyllum spicatum* was also abundant, but *P. pectinatus* a little fewer in number. The stand was covered by a very dense, robust *Cladophora* growth. In addition, drifting *Cladophora glomerata* mixed with *Ulva intestinalis, Ulva procera* and *Vaucheria* sp. occurred on the bottom along the whole transect. At a distance of 10 m from the shore line, *Myriophyllum* became the dominant species and was again covered by the above-mentioned epiphytes. The onshore stony field was covered by abundant

*Cladophora glomerata* and *Ulva intestinalis*. A clear increase of *Ceratophyllum demersum* was recorded in 2005.

Since 1985, the vegetation on the transect has changed to become highly eutrophic. Besides the warming up of the water in summer, the eutrophication process has also been affected by the prolongation of the growing season; the illuminated period has lengthened due the absence of ice (see. p. 45). In recent years, the ice winter has generally been very short in the southern part of Hästholmen. In 1997, the eutrophication was also promoted by weather conditions favourable for biological production in summer and by the high concentrations of nutrients in the water. Besides the development of the underwater 'jungle' of the *Potamogeton* species and *Myriophyllum,* the abundance of filamentous algae as their epiphytes has strongly increased. On the other hand, the species composition has probably become poorer on the transect owing to the mass occurrence of the foregoing species. However, this conclusion may also arise from the fact, that the detection of smaller and more sparsely-occurring species was difficult for the diver in the tight growth, and especially in the rake samples, which were often terrible heaps of vegetation.

**Transect d**

Transect **d** is situated 200 m west of the transect **c** in the southwestern corner of Hästholmsfjärden, 0.7 km south of the cooling water outlet. The vegetation on the transect is exposed to the warm water plumes in nearly the same way as that on transect **c**. The depth profile slopes gently and evenly along the whole transect. After a zone of coarse gravel at the shore line, the bottom turns to sand, fine sand and silt. At the outer end of the transect (depth 6 m) the bottom is soft clay mixed with gravel.

In 1975–1985, the characteristic species in the vicinity of the shore line were *Cladophora glomerata* and *Ulva* spp., and in the deeper parts *Potamogeton perfoliatus, P. pectinatus, Myriophyllum spicatum, Fucus vesiculosus* and *Ectocarpus siliculosus* as their epiphyte. The species similarity between the study years was mostly high (Ilus and Keskitalo 1986). In late summer and autumn the amounts of *Cladophora glomerata* decreased slightly in favour of *Ulva* (mainly *U. flexuosa* and *U. intestinalis*). *Ranunculus peltatus* was recorded in 1981–1982.

After 1982, *Myriophyllum spicatum* increased significantly, by 1985 forming a dense, tall stand, which reached to the surface of water at a depth of 2–3 m. *Fucus vesiculosus* was common in 1975–1978, but declined in the following years. In later years (1981, 1982, 1985), tall vascular plants were covered by *Ectocarpus siliculosus* and other epiphytes (e.g. *Cladophora glomerata*), and the occurrence of drifting *E. siliculosus* increased on the bottom, as well.

The eutrophication process continued in the late 1980s and in the 1990s. The abundance of *Myriophyllum spicatum, Potamogeton perfoliatus* and *P. pectinatus* increased further, forming an impenetrable underwater 'jungle' similar to that on transect **c**, moving through which was difficult for the diver. In the late 1990s and early 2000s, the general impression was very luxuriant (Figs. 33 and 34). The transect was clearly the most eutrophied studied, the brush reaching to the surface of water being dense and wide-ranging, filling practically the whole bottom of the bay in late summer. In particular the proportion of *Myriophyllum* increased, as well as that of *Potamogeton pectinatus,* not forgetting *Potamogeton perfoliatus,* the abundance of which also remained high. *Ceratophyllum demersum* and *Potamogeton filiformis* occurred abundantly, and low *Ranunculus peltatus* in smaller quantities under the before-mentioned tall vascular plants. The abundance of *Ceratophyllum demersum* increased on the transect after its first observation in 1994.

Dense growths of epiphytic filamentous algae covered the tall vascular plants as an essential part of the 'jungle'. The dominant epiphytes were by turns *Cladophora glomerata, Ectocarpus siliculosus* and *Vaucheria* sp. In general the epiphytes formed an unbroken veil on the vegetation bed. Since 1994, the bottom in the shallow part of the transect was often covered by a thick mat of drifting filamentous algae; mainly *Cladophora glomerata* and *Ectocarpus siliculosus*, mixed with *Ulva intestinalis, U. flexuosa* and pieces of *Zannichellia palustris* and *Ranunculus peltatus*. A narrow zone of coarse gravel close to the shore line was covered by *Cladophora glomerata* and *Ulva intestinalis*. In the 1990s, *Phragmites australis* formed a scanty, narrow stand up to the shore line. In 2002 and 2005, the vegetation on the transect was still extremely luxuriant, but slightly less abundant than in 1999.

The increase in the littoral vegetation and its eutrophication trend (in particular in the south-western part of Hästholmsfjärden) has been the most visible biological effect of the power plant, which has obviously been the combined effect of an increase in the temperature and nutrient concentrations in the water. The species composition has probably become poorer on transect **d** as well, as a consequence of the eutrophication process. However, this conclusion may also arise from the fact that the detection of smaller and less commonly-occurring species was difficult among the dense vegetation, as mentioned before in connection with transect **c**. *Fucus vesiculosus, Ceramium tenuicorne, Polysiphonia fucoides, Chara aspera* and *Tolypella nidifica* were not detected on the transect after the mid-1980s.

At Forsmark, Snoeijs & Prentice (1989) noticed that the temperature increase favoured blue-green and green algae at the expense of red and brown algae. Red and brown algae and diatoms decreased in percentage cover-abundance

with increasing temperature. Blue-green and green algae, in contrast, tended to increase with temperature. *Ectocarpus siliculosus* was favoured and *Pilayella littoralis* was inhibited by artificial heating. *Ceramium tenuicorne* occurred in the heated water, but with lower abundances than at unheated sites; in summer it lost in competition with *Cladophora glomerata*.

**Fig. 33.** Dense vegetation on transect **d** (Photo by Anna Weckman).

**Fig. 34.** Dense vegetation on transect **d** (Photo by Anna Weckman).

**Transect e**

Transect **e** is located directly in front of the cooling water outlet (distance 150 m) at the north tip of the Halkokari islet. It starts from the base of the biggest boulder on the shore extending towards the nearest point of Hästholmen in the north. It is composed of the steep shore slope of the islet, which falls rapidly within 25 metres from the shore line to a depth of 6.5 m and then levels out to an even clay-mud bottom. At the outer end of the transect the depth is 7 m. The upper part of the transect is an evenly-sloping rocky bottom with stones and boulders to a distance of 15 m from the shore line. On the lower part of the slope, there are big boulders on gravel mixed with clay. Below the boulders, the bottom turns to clay mixed with gravel, and at the outer end of the transect the bottom is relatively soft clay. The warm cooling water flows out towards the transect, being restricted, however, to the uppermost 3–4 m layer, where it creates a relatively strong current (a rough estimate, 0.1 m $s^{-1}$).

The vegetation was sparser than on the other transects, and was mainly restricted to the rocky and stony shore slope. In 1975–1985, filamentous green algae (*Cladophora glomerata*, *Ulva* spp.) dominated on the upper part of the slope, and a stand of small *Fucus vesiculosus* occurred in the late 1970s in its deeper part, at depths of 2–5 m. Other species were recorded only occasionally (Ilus and Keskitalo 1986). *Cladophora glomerata* had a well-developed belt at depths of 0–2 m in 1975–1976, but covered the stones as a low mat, or formed only small tufts, in 1977–1985. *Ulva* spp. was recorded in 1976–1985, and especially in 1981 the *Ulva* belt was well developed.

The *Ulva* species observed on the transect in 1976–1985 were *U. procera, U. compressa, U. flexuosa* and *U. flexuosa* subsp. *paradoxa* (the last-mentioned abundant in 1982 and 1985). A rare stand of *Potamogeton perfoliatus* appeared on the transect in 1977–1978, but disappeared later. In 1976–1979 the biomass of *Fucus vesiculosus* was 130–150 g $m^{-2}$, but in later years it declined. After 1980 the thalli were dwarf-sized, brittle and covered with epiphytes, the most abundant of which was *Cladophora glomerata.* In 1985, *F. vesiculosus* was not observed, and the vegetation in general was very poor (Ilus and Keskitalo 1986).

As well as in 1985, the vegetation was also very poor in 1988 and 1991, although there was a vital *Cladophora glomerata* growth on the surface of a big boulder next to the transect. In 1988, all rocky surfaces were covered by withered *Cladophora* fluff, or more robust *Cladophora glomerata,* and *Ulva* spp. occurred only on the glaciated rock close to the shore line. In 1994, 1997 and 1999, the vegetation was clearly more vigorous than in the late 1980s and in 1991. *Myriophyllum spicatum* was the only vascular plant met with on the transect in 1991: a few small individuals were found on the sand bottom at a depth of 2–4 m. In 1994, *Myriophyllum* formed quite a healthy growth, and its

abundance continued to increase until 2005. In addition, *Potamogeton perfoliatus* and *Ranunculus peltatus* were occasionally observed on the slope. A last single tuft of *Fucus vesiculosus* was recorded in 1994. In 2005, the rocky surfaces were richly covered by a dreissenid bivalve *Mytilopsis leucophaeta* (see p. 113).

Filamentous algae occurred in general on the stones and the glaciated rock close to the shore line, but also as epiphytes on the few vascular plants. The dominant species were *Cladophora glomerata* and *Ulva* sp. In 1994, *Ulva intestinalis* formed a 0.5-m-broad belt at a distance of 3 m from the shore line and *Cladophora glomerata* a vigorous belt above it. *Ulva procera* and *Cladophora aegagrophila* occurred in the upper part of the transect in 1997. In 1999, 2002 and 2005, *Cladophora glomerata* was abundantly recorded on the glaciated rock close to the shore line.

The relatively strong current and the consequent erosion bottom in the upper part of the transect are probably the main reasons for the scantiness of the vegetation on the transect. Chlorination is not used for anti-fouling in the Loviisa power plant, but the process waters may contain other substances that may have inhibitory effects on littoral vegetation in the close vicinity of the cooling water outlet. The maximum temperatures are probably not the reason, although, e.g., *Fucus* has been noticed to be eliminated from a rocky shore where water temperatures reached 27–30ºC (Langford 1990). The highest temperature of the discharged water, given as an hourly mean, has been 32.1ºC (Appendix 2), but the cooling water has already mixed with larger water masses and has cooled down before reaching the site of the transect. Green algae again are reported to grow best at 25–35ºC (Snoeijs & Prentice 1989).

In general, the species composition has become poorer on the transect during the study period. Besides *Fucus vesiculosus,* e.g., the red algae *Ceramium tenuicorne, Phyllophora* sp. and *Polysiphonia fucoides* have disappeared. Snoeijs & Prentice (1989) noticed that thermal discharge lowered species diversity both in summer and winter at sites with flowing water, but not at sites with quiescent or stagnant water at Forsmark. Furthermore, Snoeijs (1988) stated that the main effect of heating was the persistence of some of the macroalgae (e.g. *Cladophora glomerata*, *Enteromorpha ahlneriana*) through winter, which allowed them to become more dominant earlier in the year than normal. A fast flow rate enhanced nutrient availability for algae, and absence of ice cover enhanced light availability and eliminated the mechanical abrading effect of ice.

**Transect f**

Transect **f** is located in Hästholmsfjärden about 0.9 km north of the cooling water outlet. It is directed southeastwards from a cove on the east shore of Åmusholmen Island. The profile slopes gently within 60 metres to a depth of

3 m, and after that slightly more rapidly, so that at a distance of 80 m the depth is 6 m and at the end of the transect 7 m. After a stony field close to the shore line, the bottom is sand and gravel in the first part of the transect, and a little more mud is found among the coarse material at the outer end. Along the whole transect there are boulders and stones on the bottom.

Transect **f** was added to the monitoring programme in 1999. Thus, no background data are available from the 1970s, and the results are based only on three surveys carried out in 1999, 2002 and 2005. In the outer part of the transect, the boulders were covered by a short *Cladophora* pile. The abundance of *Potamogeton perfoliatus* and *Myriophyllum spicatum* increased from a distance of 60 m towards the shore line, and the stems reached to the water surface. In addition, there were sparse patches of *Ranunculus peltatus* and *Ceratophyllum demersum* on the bottom. After the midpoint of the transect the coverage of *Potamogeton perfoliatus, Myriophyllum spicatum* and *P. pectinatus* varied between 20 and 30%, but very dense small-area stands also occurred in places. *Potamogeton filiformis* and *Najas marina* grew in smaller quantities under the taller plants. Some low *Fucus vesiculosus* tufts covered by *Cladophora glomerata* were noticed on the stones. A relatively sparse *Phragmites australis* stand reached to a distance of 12 m from the shore line. Close to the shore line the coverage of vegetation was nearly 100%: the stony field was covered by *Cladophora glomerata, Ulva intestinalis* and *Ulva* sp. Single *Ranunculus peltatus* specimens grew between the stones.

Although the cooling water flows mainly southwards from Hästholmsfjärden, the thermal effect is also noticeable in the northern part of the bay. Besides the raised temperature, the prolongation of the growing season is another factor influencing the biota, and the littoral vegetation on the transect clearly indicates eutrophy. This is shown in the dense growth of *Myriophyllum spicatum, Potamogeton perfoliatus* and *P. pectinatus* and the appearance of *Ceratophyllum demersum* and *Najas marina* in the species composition. However, the degree of eutrophication is clearly lower than on transects **c** and **d**.

**In conclusion**

The most obvious biological effect of the thermal discharge from the Loviisa power station has been the increase and eutrophication of the littoral vegetation in the southern and southwestern parts of Hästholmsfjärden Bay. Many perennial vascular plants, especially *Myriophyllum spicatum*, but also *Potamogeton perfoliatus, Potamogeton pectinatus* and *Phragmites australis,* and many filamentous algae have benefited from the warm water. In addition, the fast-growing filamentous algae have significantly benefited from the increase of nutrients in the water (*cf.* Mäkinen & Aulio 1986, Kauppila and Bäck 2001).

Thermal discharges have been noticed to strongly affect the overwintering of aquatic vegetation and the date of the start of the growing season (Keskitalo 1988). The increased illumination in the ice-free area in winter and the direct influence of the temperature rise cause a significant advance in the start of the growing season for many species (Keskitalo and Heitto 1987). Consequently, the changes in vegetation have been largest in areas that remain ice-free in winter.

The spring bloom dynamics of epilithic diatom communities were studied in the biotest basin of the Forsmark power plant, Sweden. The results showed that large blooms of fast-growing epilithic diatoms in large colonies can occupy the niche that emerges when the ice-free season is prolonged. Dramatic species shifts may only be expected if the winter ice cover is totally absent (Snoeijs 1990).

Although the warming of the water has significantly promoted the increase and eutrophication of aquatic vegetation in the vicinity of the cooling water outlet, where *Myriophyllum spicatum, Potamogeton perfoliatus, Potamogeton pectinatus* and filamentous algae as their epiphytes have formed dense growths in the littoral zone during late summer, the strongest eutrophication has extended only to a distance of about 1 km from the outlet. Slighter eutrophication was visible over the whole of Hästholmsfjärden, but outside it signs of eutrophication appeared in littoral vegetation only sporadically.

## 1.14 Benthic fauna

Studies of macrozoobenthos have established their position in most programmes monitoring the state of water bodies. Benthic animals are regarded as good indicators of water quality, due to their relatively stationary character, relatively long life span and ecological versatility. Different species respond in different ways to different kinds of environmental change. Certain species favour and benefit from eutrophication, others are sensitive to pollution and withdraw from it. Thus, the state of a water body and changes in it can be assessed on the basis of the species composition and abundance ratios within benthic animal communities. Benthic fauna is important in ecosystems, e.g., as food for fish.

The macrozoobenthos in the sea area around Hästholmen is very poor in character with respect both to the number of species and the abundance of specimens. This was already noticed in the first survey carried out in the area in 1966 (Bagge and Voipio 1967). The most significant factor affecting the scarcity of the benthic fauna is the low salinity of the water that makes the existence of many marine species difficult; but, on the other hand, the salinity is too high for the successful establishment of many freshwater species. The low salinity is reflected in the species composition as an abundant occurrence of oligochaetes

and chironomid larvae and as an almost total absence of many more marine species typical of Finnish coasts. The impact of the low salinity is clearly seen in the scarcity of Baltic Tellin *(Macoma balthica)*. Though this bivalve mollusc is generally an abundant and dominant type-species in the zoobenthos of our coastal waters, it is only sporadically met with at the deep soft-bottom stations in the Loviisa area. Because *Macoma* is relatively big, its poorness is a significant factor in the smallness of the benthic biomasses.

Bagge and Voipio (1967) stated, on the basis of their survey in 1966, that the sediments in the deep areas of Hudöfjärden and Hästholmsfjärden consist of black sulphidic clay (sapropel) with a very thin oxidized layer at the surface, which indicates that depletion of oxygen occasionally prevails at the mud-water interface. The soft-bottom areas in Loviisa Bay, Hudöfjärden and Hästholmsfjärden were characterised by low densities of bottom fauna. They concluded that the disturbed bottom and hydrographic conditions and the poor bottom fauna observed in the area are mainly due to the limited circulation of the water below the sill depth. They summarised that, owing to the occasional stagnation, these areas are sensitive even to small amounts of sewage and industrial wastes (Bagge and Voipio *op.cit.*).

As mentioned above, studies on macrozoobenthos have been carried out in the sea area off Loviisa since 1966. In that year, benthos samples were taken for the first time from the transect Valko–Orrengrund on a monitoring cruise of the r/v Aranda (Bagge and Voipio 1967), and the first samples from Hästhholmsfjärden were taken with an auxiliary boat on the same cruise. A basic survey of benthic fauna in the littoral areas of Hästholmen was arranged in 1967. The sampling of macrozoobenthos was started at many of the soft-bottom stations in the same year, but regular monitoring can be considered to have been initiated in 1973. The monitoring has been carried out at ten permanent sampling stations (1, 2, 3, 4, 5, 7, 8, 10, 51 and 52), shown in Fig. 3. Samples were taken twice a year, in May and August. In earlier years, samples were even taken at some stations seven times a year, but in these cases average values for the beginning and end phase of the growing season have been examined. The benthos studies on the littoral transects **a**–**e** (see p. 36) were continued in 1971 and 1974–1982; the original data are given in tables and commented upon in the Annual Reports written in Finnish (see Appendix 1). The methods used in the benthos studies are described on page 42. In addition, the zoobenthos in the Loviisa area has been studied, e.g., by Henriksson and Myllyvirta (1991) and Myllyvirta and Henriksson (2001).

The list of macrozoobenthic species determined in the samples from the 10 sampling stations is given in Table 10. This shows that the core species in the zoobenthic communities in the area have been the tubificid *Potamothrix*

*hammoniensis* and larvae of the *Chironomus plumosus* type; the polychaete *Marenzelleria* sp. in recent years, and the bivalve *Macoma balthica* and the amphipod *Monoporeia affinis* in the earlier years. A conspicuous feature in the list is the large number of chironomid larvae (22 taxa of a total number of 57). This group is an essential part in the local bottom fauna and consequently, the chironomid fauna and the emergence of aquatic insects, in general, were studied in the area by means of special funnel-net-traps in 1971. Altogether 76 species and 1 627 specimens of aquatic insects were sampled with the traps (Bagge et al. 1980), and 61 taxa of chironomids were identified by Lauri Paasivirta in the adults and pupae of the emerging insects (Paasivirta 2000), which predominantly belonged to the same genera as those presented in Table 10.

In addition to the taxa given in Table 10, a long list of other benthic animals was recorded in the area, though not in the regular samples of soft-bottom fauna. Altogether 132 species were identified in the zoobenthos studies carried out on the littoral lines, including, e.g., *Stylaria lacustris, Lumbriculus variegatus, Piscicola geometra*, *Hydracarina* sp., *Jaera praehirsuta, Asellus aquaticus, Gammarus oceanicus,* 4 *Valvata* species, *Hydrobia ventrosa, Hydrobia ulvae*, *Physa fontinalis,* 3 *Lymnaea* species, *Batyomphalus contortus* and *Cerastoderma glaucum*. Only the most abundantly observed species are mentioned here, excluding all supplementary insect larvae and turbellaria. The detailed species lists are available in the Annual Reports written in Finnish.

The vicinities of the cooling water outlets have been "first bridgeheads" for many nonindigenous species that have invaded Finnish coastal waters during recent decades. It seems to be obvious that the all-the-year-round elevated temperatures in the discharge areas of the cooling water promote the adaptation and spreading of the newcomers to a new habitat. The invasion of the spionid polychaete *Marenzelleria* sp., Conrad's false mussel *Mytilopsis leucophaeta* and the amphipod *Gammarus tigrinus* of the discharge area of the Loviisa NPP occurred in 1992–2005.

*Marenzelleria* sp. was observed for the first time in Finnish coastal waters at the entrance to the Gulf of Finland in 1990. Its introduction into the Baltic Sea probably occurred through ship transport (with the ballast water in ships) from estuaries on the north-eastern coast of North America. During 1990–1993, *Marenzelleria* expanded its distribution range, and the first observation in the eastern part of the Gulf of Finland was made in the spring of 1992 in the Loviisa archipelago near the cooling water outlet of the nuclear power plant (Stigzelius et al. 1997). At first, the newcomer was commonly determined to be *Marenzelleria viridis*, but subsequent and more exact studies showed that in fact probably three different species of the *Marenzelleria* genus had spread to the Finnish coast. *Marenzelleria* sp. is therefore used as the common name

**Table 10.** Macrozoobenthic species determined in samples taken from the Sampling Stations 1, 2, 5, 51, 52, 3, 4, 7, 8 and 10 in 1973–2006. For abbreviations see the footnote. The determination of chironomid larvae was carried out according to Bryce (1960) and the nomenclature is based on it.

| Species | 1 | 2 | 5 | 51 | 52 | 3 | 4 | 7 | 8 | 10 |
|---|---|---|---|---|---|---|---|---|---|---|
| Turbellaria | o | – | – | o | o | – | – | – | – | – |
| Nemertinea | | | | | | | | | | |
| *Prostoma obscurum* | o | o | o | b | b | – | o | – | o | – |
| Nematoda | o | – | o | – | – | – | – | – | – | – |
| Polychaeta | | | | | | | | | | |
| *Harmothoë sarsi* | – | – | – | – | – | – | o –79 | o –80 | – | – |
| *Hediste diversicolor* | – | – | – | o | o | – | – | – | – | – |
| *Polydora redeki* | – | – | – | o | o | – | – | – | – | – |
| *Marenzelleria* sp. | o | C 95– | C 96– | C 95– | C 95– | C 95– | C 00– | C 00– | C 99– | o 92– |
| Oligochaeta | | | | | | | | | | |
| Naididae | – | o | o | o | – | o | – | – | – | – |
| *Nais elinguis* | – | – | o | – | – | – | – | o | o | – |
| *Paranais litoralis* | – | – | – | o | – | – | o | o | o | – |
| Tubificidae | – | – | – | o | – | – | – | – | – | – |
| *Tubifex costatus* | – | – | o | – | – | – | b –82 | b –82 | – | – |
| *Psammoryctides barbatus* | – | – | – | o | – | – | o | – | – | – |
| *Limnodrilus* sp. | – | – | – | o | o | – | – | – | – | – |
| *Potamothrix hammoniensis* | C | C | C –94 | C | C | C –87 | C 85– | C 05– | C | – |
| *Clitellio arenarius* | – | – | – | – | – | – | – | o | – | – |
| Crustacea | | | | | | | | | | |
| *Mysis relicta* | – | – | – | – | – | o | o | o | b | o |
| *Mysis mixta* | – | – | – | – | – | – | – | o | o | o |
| *Neomysis integer* | o | o | o | o | o | – | o | – | o | – |
| *Saduria entomon* | – | o | o | o | o | o | b | C –88 | b | o |
| *Jaera albifrons* gr. | – | – | – | o | – | – | o | – | – | – |
| *Gammarus* sp. | o | – | o | o | o | o | o | o | o | – |
| *Gammarus salinus* | – | – | o | o | – | o | – | o | o | o |
| *Gammarus zaddachi* | – | – | – | – | – | – | – | o | o | – |
| *Gammarus tigrinus* | – | – | – | o 05 | – | – | – | – | – | – |
| *Monoporeia affinis* | o –74 | b –74 | b –74 | o | o | o | C –87 | C –85 | C –88 | o |
| *Leptocheirus pilosus* | – | – | – | – | o | – | – | – | – | – |
| *Corophium volutator* | – | – | o | o | o | – | – | – | – | – |

C = core species, b = common by-species, o = occasional species.
E.g. C 95– = core species since 1995, e.g. o –88 = occasional species up to 1988

**Table 10.** Continued.

| Species | 1 | 2 | 5 | 51 | 52 | 3 | 4 | 7 | 8 | 10 |
|---|---|---|---|---|---|---|---|---|---|---|
| Gastropoda | | | | | | | | | | |
| *Theodoxus fluviatilis* | – | – | – | – | – | – | – | – | o | – |
| *Bithynia tentaculata* | – | – | – | – | o | – | – | – | – | – |
| *Potamopyrgus antipodarum* | o | o | o | C 85– | C 85– | – | – | – | – | – |
| Bivalvia | | | | | | | | | | |
| *Macoma balthica* | b –83 | b –88 | b –87 | C –93 | C –93 | o | C –80 | b –86 | C –77 | o |
| Ephemeroptera | | | | | | | | | | |
| *Caenis horaria* | – | o | o | o | – | – | – | – | – | – |
| Ceratopogonidae | | | | | | | | | | |
| *Culicoides* sp. | o | – | o | o | – | o | – | – | o | – |
| *Palpomyia* sp. | o | – | – | o | o | – | – | o | o | – |
| Chironomidae | | | | | | | | | | |
| Tanypodinae | – | – | – | – | – | o | – | – | o | – |
| *Macropelopia* sp. | o | o | – | – | – | – | – | – | – | – |
| *Pentaneurini* sp. | – | – | – | – | – | – | o | o | – | o |
| *Procladius* sp. | b | b | b | b | b | b | b | o | b | o |
| *Tanypus* sp. | b | o | o | o | o | – | – | – | o | – |
| Orthocladiinae | – | – | o | – | – | o | – | o | o | o |
| *Corynoneura* sp. | – | o | – | o | o | – | o | – | – | – |
| *Chironomus* sp. | o | o | o | o | o | o | – | – | o | – |
| *Chironomus plumosus* gr. | C | b | b | C | C | b | C 02– | o | C | o |
| *Chironomus thummi* gr. | o | o | o | o | – | o | o | – | o | – |
| *Chironomus halophilus* gr. | o | o | o | o | o | o | b | o | o | o |
| *Chironomus salinarius* gr. | o | – | – | – | – | – | – | – | – | – |
| *Cladopelma* sp. | o | – | – | – | – | – | – | – | – | – |
| *Cryptochironomus* sp. | o | – | – | – | o | o | – | – | – | – |
| *Dicrotendipes* sp. | – | – | – | o | o | o | – | – | o | o |
| *Microchironomus tener* | o | – | – | – | – | – | – | – | – | – |
| *Microtendipes* sp. | – | – | – | – | – | o | – | – | – | – |
| *Parachironomus* sp. | o | – | – | – | – | o | – | – | – | – |
| *Paratendipes* sp. | o | – | o | o | o | – | – | – | – | – |
| *Phaenopsectra* sp. | o | o | – | – | – | – | – | – | – | – |
| *Tanytarsini* sp. | – | o | o | o | o | o | o | – | o | – |
| Coleoptera | o | – | – | – | – | – | – | – | – | – |

C = core species, b = common by-species, o = occasional species.
E.g. C 95– = core species since 1995, e.g. o –88 = occasional species up to 1988

in this publication. The spread of *Marenzelleria* in the Baltic Sea was very rapid and extensive, and its establishment was more successful either in more eutrophied areas or in more uniform biotopes (Kotta et al. 2003). In the Loviisa area, *Marenzelleria* was most abundant at Stations 51 and 52, i.e., in the close vicinity of the cooling water outlet, and in Hästholmsfjärden, although it later spread to all the sampling stations, but in lower densities. The macrozoobenthos communities in the northern Baltic Sea consist of few species, and thereby expansive immigrants, such as *Marenzelleria*, may have a strong influence on the established communities, and can be severe competitors to the dominant species on muddy and sandy bottoms (*cf*. Stigzelius et al. 1997).

In the autumn of 2003, a pronounced settlement of young dreissenid mussels was observed in the Loviisa archipelago close to the cooling water outlet of the power plant, which led thereafter to a dense adult population (Laine et al. 2006, Lehikoinen 2006). At first, the species was identified as the zebra mussel, *Dreissena polymorpha,* which had recently increased in numbers in the eastern Gulf of Finland. However, an examination of adult mussels collected in 2004 revealed the species to be Conrad's false mussel, *Mytilopsis leucophaeta.* This was the first record of the species in the northern Baltic Sea. In 2004, dense assemblages of adult *M. leucophaeta* were discovered in the area affected by cooling water from the nuclear power plant. The introduction of the species obviously took place via ballast water transport, resulting in a successful establishment in a favourable warm water environment. The mussel assemblages covered stones and boulders up to 100%, and were mainly found at a depth of 1–3 m. In these populations, densities and biomasses up to 28 000 ind. $m^{-2}$ and 9.8 kg $m^{-2}$ (wet weight) were observed , respectively (Laine et al. *op. cit.*). The most vigorous population has been concentrated in the vicinity of the cooling water outlet, but dense assemblages were found over the whole Hästholmsfjärden, and planktonic larvae stages were spread to a distance of at least 20 km from the power plant (Lehikoinen *op. cit.*, Mattila 2002). *Mytilopsis* was not detected in the ordinary bottom fauna samples.

The third recent observation of a new nonindigenous species in the vicinity of the cooling water outlet of the Loviisa power plant was that of the amphipod *Gammarus tigrinus*. The species was found for the first time in Finland in the vicinity of the port of Hamina (50 km east of Loviisa) in August 2003. The species is originally native to North America (as are also *Marenzelleria* and *Mytilopsis*); it was first discovered in Europe in 1931, in British and Irish waters (Baltic Sea Portal 2007). The first record in Loviisa waters was made in May 2005, when one specimen was detected in an ordinary benthos sample from Station 51 (Table 10). The species was also found abundantly in *Mytilopsis* samples taken from the vicinity of the cooling water outlet by scuba diving (K. Huusela, pers. comm).

In addition, the observations of *Hediste diversicolor, Pygospio elegans, Polydora redeki, Leptocheirus pilosus* and *Corophium volutator* in zoobenthos samples were also restricted to the vicinity of the cooling water outlet (Stations 5, 51 and 52), but this could as well be due to the more clayey bottom type at these stations. The only observation of the common mussel *Mytilus edulis* in the whole study area was likewise made in front of the cooling water outlet in Hästholmsfjärden. The salinity of the water in the Loviisa area is obviously too low for this cosmopolitan marine species, and its permanent distribution on the south coast of Finland is probably restricted to the Pellinki archipelago, 25 km west of Loviisa. In addition, the shrimps *Crangon crangon* and *Palaemon adspersus* were observed in Hästholmsfjärden in connection with the scuba diving studies. Other species, which are usually classified as benthic animals and were found abundantly in the study area, but were seldom found in benthos samples were *Cordylophora caspia, Laomedea loveni, Electra crustulenta* and *Balanus improvisus*. They were commonly found attached to all kinds of solid substrates (rock, stones, boulders, logs, ropes, etc.).

Changes in the abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Stations 1, 2, 5, 51, 52, 3, 4, 7, 8, and 10 during the study period are shown in Figs. 35–44.

**Station 1**

At Sampling Station 1 in Klobbfjärden, the depth is ca. 8 m and the character of the bottom is a little more solid sulphidic gyttja-clay than at the stations in the deep part of Hästholmsfjärden. The lowest oxygen concentration observed in the near-bottom water was 1.54 ml $l^{-1}$ in August 1996 and 2002 (23 and 25%, respectively). The core species in the benthic community were the tubificid *Potamothrix hammoniensis* (Oligochaeta) and the larvae of the *Chironomus plumosus* group (Chironomidae). The share of *C. plumosus* was much higher than at the stations in Hästholmsfjärden. The highest density of *C. plumosus* was 2 910 ind. $m^{-2}$ in September 2005 and that of *Potamothrix hammoniensis* 4 250 ind. $m^{-2}$ in May 2006. The highest total abundance and biomass at the station were 6 040 ind. $m^{-2}$ (September 2005) and 53.9 g $m^{-2}$ (May 2006). The change in the sampling method in 2004 perhaps affected the density values of oligochaetes in the results from September 2005 and May 2006 (Fig. 35), but probably not those of chironomid larvae. The most important by-species were the tanypodine larvae of *Procladius* sp. and *Tanypus* sp. (Chironomidae). *Macoma balthica* was a common by-species now and then in the early years of monitoring, and also some *Monoporeia affinis* (Crustacea) specimens were found at the station in 1973–1974. *Marenzelleria* sp. appeared in the species composition in 1999, but has so far occurred at low densities.

**Station 2**

The seabed in the central part of Hästholmsfjärden is characterised by a relatively even soft-bottom. The depth at Station 2, in the middle of Hästholmsfjärden, is a scant 12 m. The character of the bottom has changed during the 40 years from sulphidic gyttja-clay to more organic, watery deposits. The near-bottom water suffered from oxygen depletion in 1991 and 2006, and very low oxygen concentrations (0.12 and 0.40 ml $l^{-1}$, 2 and 5%) in 1996 and 1998 (Fig. 10). *Potamothrix hammoniensis* has always been a clear core species at this station, although *Marenzelleria* sp. was repeatedly more abundant in 1995–1999. *Marenzelleria* appeared at the station in spring 1995 and immediately achieved dominance, but its share then became smaller in the 2000s (Fig. 36). The most important by-species were the larvae of *Chironomus plumosus* gr. and *Procladius* sp., and in the past, the occasionally-occurring *Macoma balthica*, as well as *Monoporeia affinis*, which was even the dominant species in 1973. However, since the middle of the 1970s, only a few specimens of *Monoporeia* were occasionally recorded at the station. The abundance of macrozoobenthos increased from the late 1960s to the early 1980s. A clear decline took place between the mid-1990s and the early 2000s, but a strong increase of *Potamothrix* was again observed in 2002–2003 (Fig. 36). The highest total abundance was 1 930 ind. $m^{-2}$ in May 2003. The graph of biomasses has two peaks, with top values in 1980 and 1988, but after that the biomasses have decreased to a relatively low level. The highest total biomass was 32.4 g $m^{-2}$ in May 1980. In the 2000s, the total biomasses varied between 0.5 and 7.4 g $m^{-2}$.

**Station 5**

Station 5 is situated on the west side of Hästholmsfjärden at a distance of about 0.4 km from the cooling water outlet. The depth at the station is ca. 10.5 m, and the character of the bottom has changed during the 40 years from sulphidic gyttja-clay towards softer and more organic deposits. The near-bottom water suffered from oxygen depletion in the autumn of 1996, and from low oxygen concentration (0.70 ml $l^{-1}$, 10%) in August 2006. Traditionally the zoobenthos at Stations 2 and 5 was very similar: *Potamothrix hammoniensis* was the dominant species and *Chironomus plumosus* gr. and *Procladius* sp. were the most important by-species. *Monoporeia affinis* was relatively abundant at the station in 1973–1974, but since then it has been observed very seldom. *Macoma balthica* was a quite regularly observed by-species until 1987, but since then has also reverted. The abundance of *Potamothrix* reached its climax in the early 1980s, but then a fundamental change happened in the zoobenthos in the 1990s. *Potamothrix* disappeared almost totally from the species composition, and at the same time the total densities and biomasses of the macrozoobenthos

decreased radically (Fig. 37). *Marenzelleria* has occurred at the station since 1992, and in May 1996 it reached the dominant position for the first time. In May 2006, the maximum density of *Marenzelleria* was 307 ind. $m^{-2}$. Dead bottom (no macroscopic animals found) was observed at the station in August 2004 and in September 2006.

**Stations 51 and 52**

Stations 51 and 52 are situated near to the east shore of Hästholmen, directly in front of the cooling water outlet (distance about 100 m). The depth at the stations is ca. 7 m. The character of the bottom varies from muddy clay to clay mixed with gravel or sand. In consequence of the smaller depth and more solid and mixed bottom type, the benthic fauna has traditionally been more abundant and richer in species than that at the other stations studied. Altogether 38 taxa were identified at Stations 51 and 52 (Table 10). *Potamothrix hammoniensis* and the larvae of the *Chironomus plumosus* group were the core species (especially at Station 51) and *Procladius* sp. and the nemertinean *Prostoma obscurum* (others) typical by-species (Figs. 38 and 39). The abundance of *Chironomus plumosus* larvae lessened noticeably in the early 1980s, but in the 2000s they occurred in higher quantities again. *Macoma balthica* was one of the core species in the benthic community until 1993, but since then it has been clearly reduced in number. *Monoporeia affinis*, which is known as a species that prefers cold water and avoids pollution, was relatively abundant at the stations in the early 1970s, but since then has disappeared almost totally from the species composition. The small gastropod *Potamopyrgus antipodarum* has belonged to the core species since 1985 and has often occurred in noticeably high quantities. In 2005, the maximum densities of *Potamopyrgus* were 2 750 and 3 900 ind. $m^{-2}$ at Stations 51 and 52, respectively. *Marenzelleria* sp. appeared in the species composition in May 1992, and in 1995, it dominated at both stations for the first time. At its highest, the density of *Marenzelleria* was 860–870 ind. $m^{-2}$ at Station 52 in August 1999 and in May 2005. The highest total abundance of zoobenthos was recorded at both stations in September 2005 (4 280 ind. $m^{-2}$ at Station 51 and 4 320 ind. $m^{-2}$ at Station 52), at least partly due to a massive occurrence of *Potamopyrgus* (others in the Figures). The highest total biomasses were recorded in 1980 (58.4 g $m^{-2}$ at Station 51) and 1981 (36.7 g $m^{-2}$ at Station 52) due to the abundant occurrence of *Macoma* (Figs. 38 and 39).

At the same time as *Monoporeia* almost totally disappeared, *Macoma* noticeably decreased and *Chironomus plumosus* momentarily disappeared and then returned, some new and exceptional species appeared in the area in front of the cooling water outlet, as mentioned before. Many of them were of relatively marine origin, taking into account the low salinity prevailing in

Hästholmsfjärden (i.e., *Mytilus edulis, Hediste diversicolor, Pygospio elegans, Polydora redeki, Leptocheirus pilosus, Corophium volutator, Marenzelleria* sp., *Mytilopsis leucophaeta, Gammarus tigrinus*). The cooling water discharged from the power plant, apart from being warmer, is also more saline than the surface water in Hästholmsfjärden in general, because it is taken from a depth of 8.5–11.1 m in Hudöfjärden (p. 29). Therefore the higher salinity of the water, together with its elevated temperature, may attract marine species to the area. For instance, the salinity preference of the ragworm *Hediste diversicolor* is relatively high (18–40 PSU), but it can withstand a great variation in salinity. It was observed in front of the cooling water outlet for the first time in 1982, but its distribution never enlarged. Besides the higher salinity of the water, the elevated temperature might make its living in the area easier.

*Chironomus plumosus* is known as an indicator of eutrophied waters and for its tolerance to different kind of pollutants and anoxic conditions. Thus, the reason for its momentary disappearance was probably not the increase of temperature or eutrophy, but could be repeated failure in egg lying of the adult midges in an area subjected to a strong water flow. On the other hand, a similar momentary decrease was also found in the 1990s in the abundance of *Potamothrix hammoniensis*, which is also known as an indicator of increased eutrophy. *Potamopyrgus antipodarum* (*Paludestrina jenkinsi*) seems to thrive in the discharge areas of cooling water. Epilithic fauna was studied in the biotest basin at the Forsmark NPP in Sweden, and *Paludestrina jenkinsi* together with *Theodoxus fluviatilis* were noticed to be especially favoured by warm water. The ability of these snails to reproduce throughout the year may give them a competetive advantage at heated sites (Snoeijs 1989).

**Station 3**

Station 3 is situated in the southeast corner of Hästholmsfjärden at its deepest point. The depth is a scant 18 m. Anoxic conditions were recorded in the near-bottom water in 1981, and since 1990 every autumn except in 1997 and 2003 (Fig. 10). When the first samples were taken from the station in 1973–1974, the character of the bottom was described as black sulphidic gyttja with an oxic surface layer. During the 40 years, the seabed has become more organic, and in the 1990s it changed to very watery, black mud, indicating anoxia and the formation of hydrogen sulphide. The oxic surface layer has become thinner or disappeared, and a smell of hydrogen sulphide was commonly met with in connection with benthos sampling in late summer and in autumn. In the 1970s, and to a lesser degree in the 1980s, *Potamothrix hammoniensis* dominated, and the larvae of *Chironomus plumosus* gr. and *Procladius* sp. were the most important by-species in the macrozoobenthos. At its highest, the abundance of

*Potamothrix* was 1 230 ind. $m^{-2}$ and the total density of the bottom fauna was 1 470 ind. $m^{-2}$ in 1977 (Fig. 40). Since then, a decline in the zoobenthos continued until the mid-1990s. Dead bottom (no macroscopic animals found) was recorded in May 1991 and 1994, and in August 1996 and 2006. *Marenzelleria* sp. appeared at the station in May 1995. Since then, it has dominated in the zoobenthos, and in May 2000, the abundance of *Marenzelleria* reached a record high for the whole study area, 1 510 ind. $m^{-2}$ (Fig. 40). *Marenzelleria* seems to tolerate anoxia and the changed benthic conditions well, and effectively edges out the competitors. The total biomass at the station was at its highest 16.7 g $m^{-2}$ in May 1973. In the 1990s and 2000s, it has always been below 1.5 g $m^{-2}$ except in May 2000, when it was 3.2 g $m^{-2}$ due to the abundant occurrence of *Marenzelleria*.

**Station 4**

The depth at Station 4 in Vådholmsfjärden is 23 m. The character of the bottom has changed during the past 40 years from quite solid sulphidic gyttja-clay to a softer, more organic mud. The lowest oxygen concentration observed in the near-bottom water was 2.76 ml $l^{-1}$ (34%) in August 1993. In parallel with the deteriorating trend in the bottom deposits, the zoobenthos has gone through four different stages. Until 1980, a typical *Macoma balthica* community prevailed at the station, in which the tubificids *Potamothrix hammoniensis* and *Tubifex costatus* were the most important by-species (Fig. 41), and the polychaete *Harmothoë sarsi* and the relict crustacean *Saduria entomon* belonged quite regularly to the species composition. At that time the total biomass was at its highest 233 g $m^{-2}$ due to the abundance of *Macoma*. The *Macoma* community was followed by a *Monoporeia affinis* community (1980–1984), and the total density of bottom fauna peaked at 2 730 ind. $m^{-2}$ in 1983. The before-mentioned tubificids and *Macoma balthica* were the most important by-species of *Monoporeia.* This community was followed by a *Potamothrix hammoniensis – Chironomus plumosus* community (1985–1994), which was linked with a clear decrease in the total abundance of macrozoobenthos and a drastic decline in biomasses. Since 1995, the benthic fauna has been very scarce. It has been dominated in turns by chironomid larvae (mainly *Chironomus plumosus* gr. and *Procladius* sp.) and *Marenzelleria* sp., the total densities varying between 0 and 148 ind. $m^{-2}$ and the biomasses between 0 and 5.9 g $m^{-2}$. *Marenzelleria* appeared at the station in August 1995, but never did occur in noticeable quantities (max. 87 ind. $m^{-2}$). Dead bottom was noticed at the station three times: in May 1995, May 1996 and May 2006.

The reason for the deterioration of the macrozoobenthos at Station 4 was obviously linked with the change in the bottom deposits to those unfavourable for benthic animals. The abundance and biomass of the zoobenthos declined as the

amount of organic matter increased in the sediment. Although a real depletion of oxygen was never recorded in the near-bottom water at the station, anoxia and a consequent strengthening of the internal nutrient load have affected the bottom deposits widely in the eastern Gulf of Finland, and vast areas have begun to be inviable for benthic animals (*cf.* Myllyvirta and Henriksson 2001). It is clear that the oxygen samples taken from Station 4 at relatively long intervals from 0.5 m above the bottom do not give the real oxygen situation at the surface of the sediment.

The disappearance of *Macoma* in the 1980s was evidently due to the change of the bottom sediment to a more organic and watery composition, because the clam prefers mixed sand-mud bottoms, and the changing of the sediment to a more watery one may especially complicate the survival of the young mussels. The disappearance of *Monoporeia* later in the 1980s was clearly caused by the increased eutrophication and increase of organic matter in the sediment, since the eggs and embryos of the amphipod are very sensitive to a decrease in oxygen. However, the significant decrease of *Potamothrix hammoniensis* and even that of *Chironomus plumosus* was unexpected, since they are both considered to be indicators of polluted and eutrophied waters, and even tolerate poor oxygen conditions.

**Station 7**

Station 7 (depth 33 m) is situated in the northern part of Orrengrundsfjärden. The character of the bottom has changed during the 40 years from soft sulphidic gyttja-clay towards more organic and watery deposits. The lowest oxygen concentration recorded in the near-bottom water was 0.06 ml $l^{-1}$ (1%) in autumn 1996. In the late 1970s and early 1980s, the zoobenthos was characterised by large fluctuations, typical of the population dynamics of *Monoporeia affinis* (Segerstråle 1960). The maximum density of *Monoporeia* was 8 230 ind. $m^{-2}$ in1981, but since 1986 it disappeared almost totally from the fauna at the station. The most important by-species in the *Monoporeia* community were *Saduria entomon, Macoma balthica* and *Tubifex costatus.* The relict crustacean *Saduria entomon* was a core species later in the 1980s, but in lower densities. In the 1990s and early 2000s, macroscopic bottom animals were only occasionally caught at the station, most regularly separate specimens of *Saduria*. *Marenzelleria* has occurred in the benthos samples in low densities since 1996. In just recent years, oligochaetes (*Potamothrix hammoniensis* and *Nais elinguis)* have dominated in the samples. Dead bottom has been recorded at the station seven times since 1993. The occasional, eye-catching biomass peaks in the 1990s and 2000s were generally due to single large *Saduria entomon* specimens (Crustacea), while other species were missing (Fig. 42).

Since the effects of thermal discharges from the power plant have not been noticed to extend to Orrengrundsfjärden, it is clear that the destruction of the abundant *Monoporeia affinis* community in the early 1980s, which was followed by a general deterioration of the macrozoobenthos, had nothing to do with the environmental effects of the power plant, but was probably due to the general deterioration taken place in the benthic fauna of the Gulf of Finland.

**Station 8**

Station 8 is situated in the east part of Hudöfjärden, in front of the cooling water intake (distance 600 m). The depth is 17 m. During the past 40 years, the bottom has changed from black sulphidic mud to more organic and watery deposits. The lowest oxygen concentrations observed in the near-bottom water were 0.12 ml $l^{-1}$ (1%) and 0.38 ml $l^{-1}$ (5%) in 1994 and 1998, respectively. In the late 1960s and early 1970s, the station was inhabited by a typical *Macoma balthica – Potamothrix hammoniensis* community, and the total biomass was at its highest 68.4 g $m^{-2}$ in 1973 (Fig. 43). In the late 1970s, the fauna declined strongly, but recovered again in the early 1980s, when *Monoporeia affinis* was abundantly caught (at its highest 612 ind. $m^{-2}$) and the total density of the macrozoobenthos was 648 ind. $m^{-2}$. In the late 1980s and early 1990s, the species composition was most diversified (including *Potamothrix hammoniensis, Monoporeia affinis, Macoma balthica, Chironomus plumosus, Saduria entomon, Mysis relicta* and *Procladius* sp.) and the total biomass rose once again, but in the late 1990s the benthos was very sparse, consisting mainly of chironomid larvae. *Marenzelleria* sp. has been the dominant species since 1997, but in low densities, except in May 2000 (140 ind. $m^{-2}$). In 2001–2006, the total abundances and biomasses were 0–28 ind. $m^{-2}$ and 0–0.3 g $m^{-2}$, respectively. Dead bottom was recorded at the station six times since 1978.

**Station 10**

Station 10 is situated in the middle of Hudöfjärden at its deepest point. The depth in the small-area deep is 24 m. Bagge and Voipio (1967) reported that the sediments in Hudöfjärden consist of black sulphidic clay (partly sapropel) with a very thin oxidized surface layer. They also caught 45 ind. $m^{-2}$ of *Macoma balthica* from their Station 7 in 1966, but they did not hit upon the real deep (the depth of their Station 7 was 18 m). In our studies since 1973, the sediment in the deep (24 m) has been very watery, organic black sludge with a smell of hydrogen sulphide. Total depletion of oxygen was observed in the near-bottom water for the first time in 1980, and since 1990 every autumn except in 1997, 1999, 2002 and 2003 (Fig. 10). No permanent macrozoobenthos was found at the station in any phase of the study period. In general, only single chironomid

larvae or drifting crustaceans were caught. Even at its highest, the total density of bottom fauna was less than 100 ind. $m^{-2}$ (Fig. 44). Since 1992, *Marenzelleria* sp. has been occasionally met with, but in low densities (max. 40 ind. $m^{-2}$). The unique high peak in the biomass graph was caused by three large *Saduria entomon* individuals found at the station in May 1989 (Fig. 44).

It is clear that the cooling water of the power plant has not affected the deterioration of benthic conditions and zoobenthos in the deep of Hudöfjärden, because their state has been unchanged since the beginning of the 1970s, when the regular monitoring of zoobenthos was initiated at the station. It is more likely to be a question of a natural phenomenon due to a poor exchange of water, which is typical for this kind of deep basin inside a sheltered archipelago. No changes have been noticed in the quality of the sediments during the past 35 years; the bottom at the station was already totally disturbed as shown by the first sampling events in the 1970s.

**In conclusion**

The most noteworthy features in the trends of the macrozoobenthos in the study area during the 40-year study period have been:

1. the invasion of many nonindigenous species into the area. Most of them have started to spread from the immediate vicinity of the cooling water outlet, probably assisted by the raised temperature. Some of them are highly expansive (e.g., *Marenzelleria* sp., *Mytilopsis leucophaeta*), causing a threat and competition to the original species (*cf.* Paavola et al. 2008),
2. the almost total loss of *Monoporeia affinis* and *Macoma balthica* from the deep soft-bottom areas.
3. the strengthening of the status of *Potamothrix hammoniensis* and *Chironomus plumosus* gr. as core species; these were later threatened, and in places edged out, by *Marenzelleria*.
4. the general and, at some stations, drastic decrease of the zoobenthos, which was seen both in the abundances and biomasses at Stations 4, 5, 7 and 8, and in the biomasses at Stations 2 and 3.

Even though the first benthos studies carried out in the area already proved that the fauna is very poor in the sea area off Loviisa, the abundances and biomasses have further decreased at most of the sampling stations during the past 40 years. However, the deterioration of the zoobenthos has not primarily been caused by the effluents of the power plant, at least not at all of the sampling stations, but rather there has been a more wide-ranging trend in the background. Benthos studies carried out in the archipelago areas of the whole Itä-Uusimaa region and in the related outer areas of the Gulf of Finland have

proved that the benthic fauna has strongly declined or is disturbed over a great part of the Finnish south coast (Henriksson and Myllyvirta 1991, Myllyvirta and Henriksson 2001). It has been stated, that the repeated hydrogen sulphide periods in the deeps have reduced the zoobenthos, which even a few decades ago was much richer (Thorell 1989).

Macrozoobenthos communities have strongly fluctuated during the past 100 years, and generally deteriorated during the last 40 years in the open Baltic Sea, Archipelago Sea and Gulf of Finland (Andersin & Sandler 1991, Laine et al. 1997, 2003a, Kangas et al. 2001, Perus and Bonsdorff 2003). In the Baltic Proper, the prolonged stagnation period of deep water in 1977–1993 caused a deterioration of the macrozoobenthos as a consequence of anoxia and the formation of hydrogen sulphide in the sub-halocline areas (Laine et al. 1997). In the Åland archipelago, the abundance and biomass of the zoobenthos peaked in the late 1980s, after which a general decrease has taken place (Perus and Bonsdorff 2003). In the Gulf of Finland, even the deepest areas were inhabited at the beginning of the 20$^{th}$ century by macrofauna species typical of healthy bottoms. The first observations of an impoverishment came in the late 1940s. After a period of recolonization in the 1950s, a strong deterioration was observed, which lasted until the mid 1980s. During the later half of the 1980s, a remarkable recovery on deep bottoms east of Helsinki was noticed (Andersin & Sandler 1991), but in many coastal areas of the Gulf, a considerable decrease in the densities of macroinvertebrates has also been reported during the most recent decades (e.g. Laine et al. 2003a).

The combination of salinity, temperature, dissolved oxygen saturation and organic matter content of the sediment are stated to be the factors that best explain the species distribution and community structure of benthic fauna, and the changes in them (Bonsdorff et al. 2003, Laine 2003, O'Brien et al. 2003). A common phenomenon has been the decline or even total disappearance of *Monoporeia affinis*, its replacement with a community dominated by *Macoma balthica*, or still later, with an increasing abundance of oligochaetes and chironomid larvae (Kangas et al. 2001, Laine et al. 2003a). In the sea area off Helsinki, changes in the abundances of benthic animals connected with the improvement in the water quality during recent decades could be discerned in particular in the bay areas and in the inner archipelago, where the abundance of oligochaetes and chironomid larvae have increased (Laine et al. 2003b). In the outer archipelago, the changes were not clearly connected to the local loading, but rather a function of the general development of the whole sea area (Gulf of Finland), being quite similar to those observed in the Loviisa area.

Wiederholm (1971) noted larger biomasses of zoobenthos near to warm water discharges in Lake Mälaren, Sweden, principally because of the higher

abundance of oligochaetes. A rich supply of organic material to the sediments maintained a large biomass, and the fauna was composed of organisms with a high ability to withstand periods of oxygen deficit of the near-bottom water. The large number of oligochaetes was proposed to be a result of temperature stimulation of microbiological activities in the ooze, which favoured the sediment-feeding oligochaetes. In Loviisa, the original core species of the local benthic communities, *Potamothrix hammoniensis,* seems more to have suffered than benefited from the eutrophication process or the warm water discharges.

It is clear that the changes in the macrozoobenthos in the Loviisa area have been due to the general deterioration of the bottom conditions (black and watery sediments with a high content of organic matter and recurrent anoxic conditions) as a consequence of the eutrophication process. The augmented plankton production has increased the amount of sinking organic matter, which has to be decomposed at the bottom. The quality of the bottom has changed as the amount of organic matter has increased in the surface sediments, and the consumption of oxygen has increased, which has in turn led to a deficiency of oxygen and the formation of hydrogen sulphide into the near-bottom water, and to a deterioration in the benthic fauna. This process has partly been contributed to by the internal load of nutrients from the sediments back into the water phase. These processes connected to the general eutrophication of the eastern Gulf of Finland have probably been the reason for the changes in the zoobenthos in the outer archipelago of Loviisa (at least at Station 7), but have also been underlying factors in the changes occurred in Hästholmsfjärden. The effects of the cooling water have probably been seen in the changes that have taken place in Hästholmsfjärden, and possibly to a smaller degree at Station 4 in Vådholmsfjärden, at the farthest.

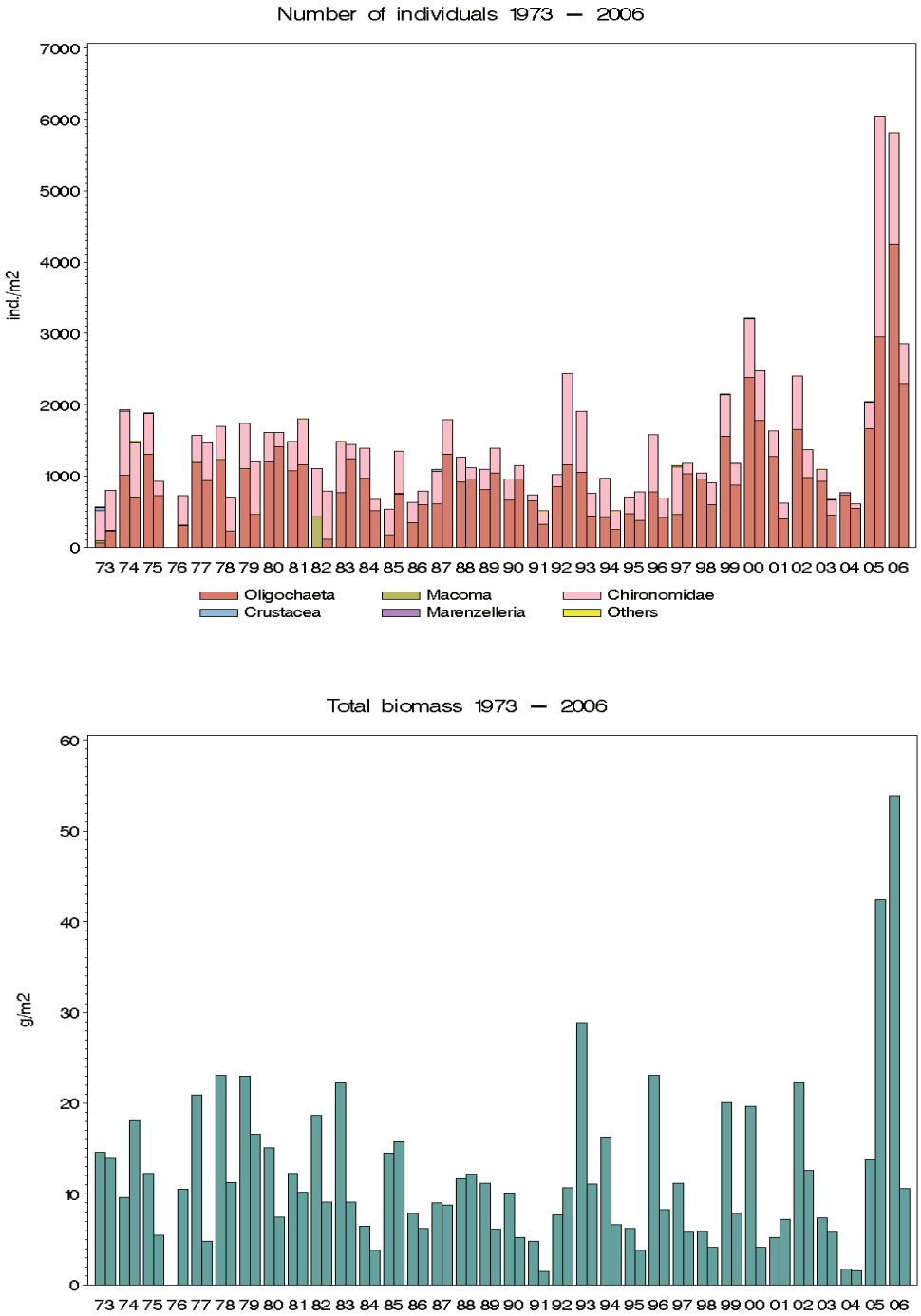


**Fig. 35.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 1 in 1973–2006.

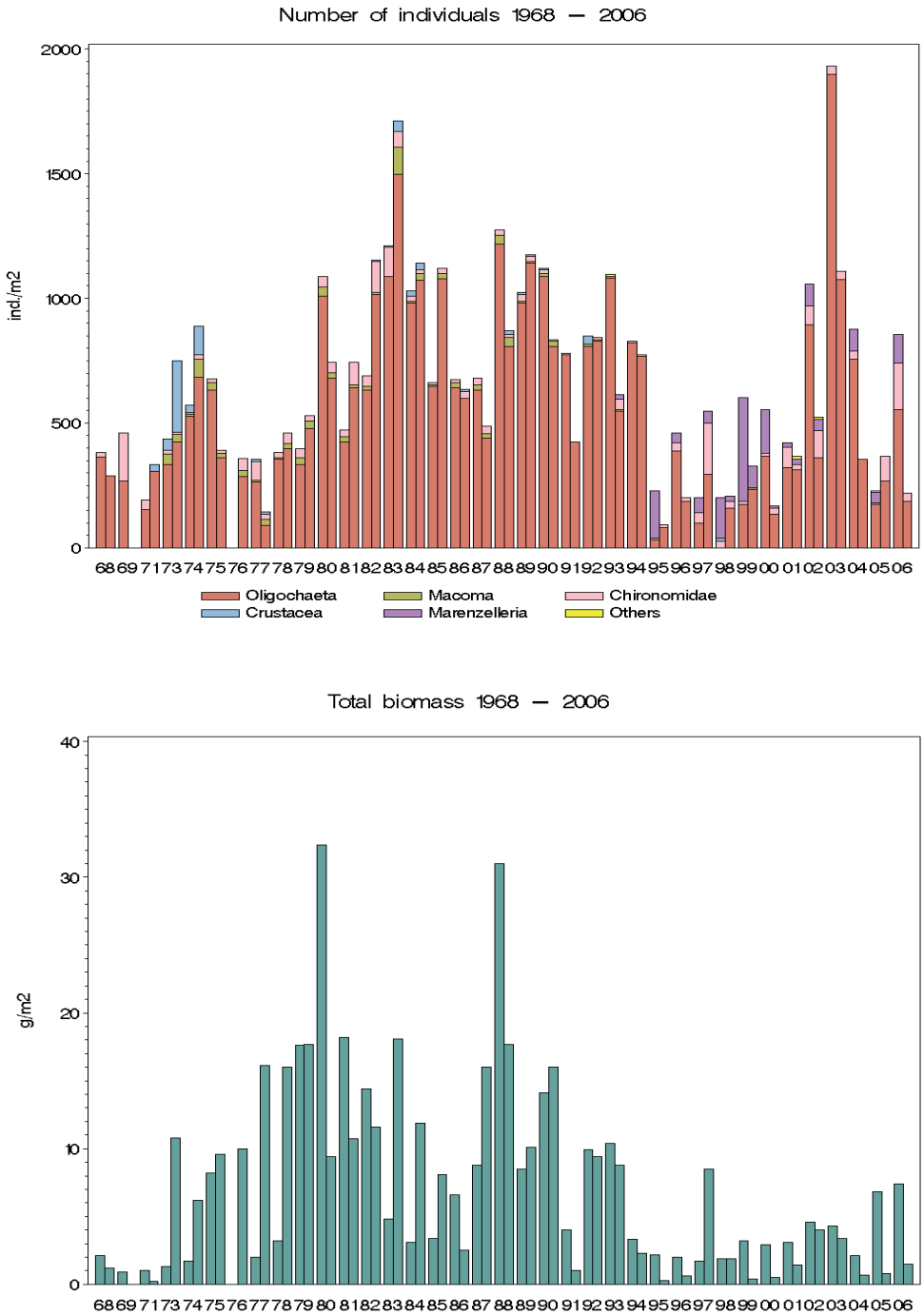


**Fig. 36.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 2 in 1968–2006.

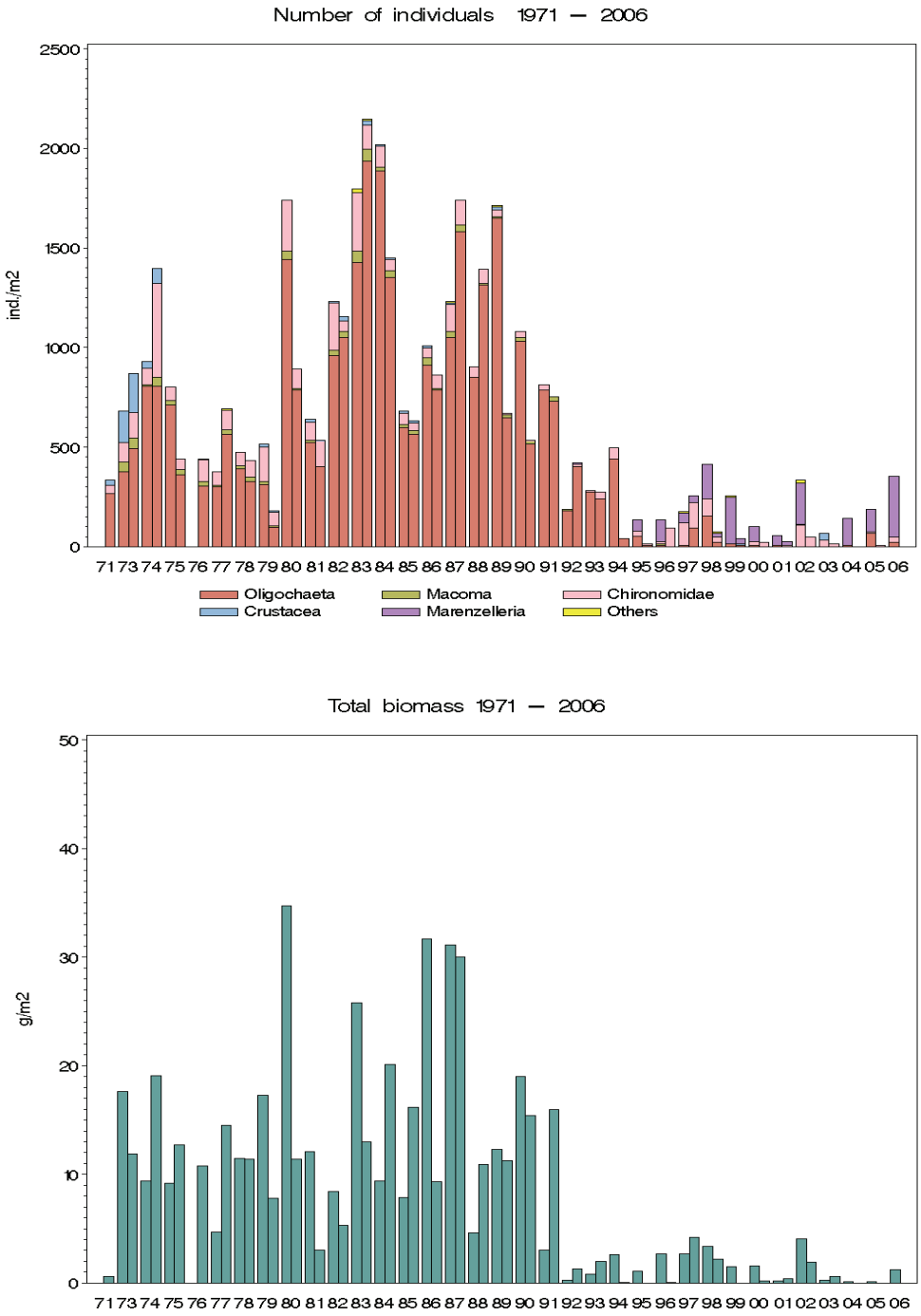


**Fig. 37.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 5 in 1971–2006.

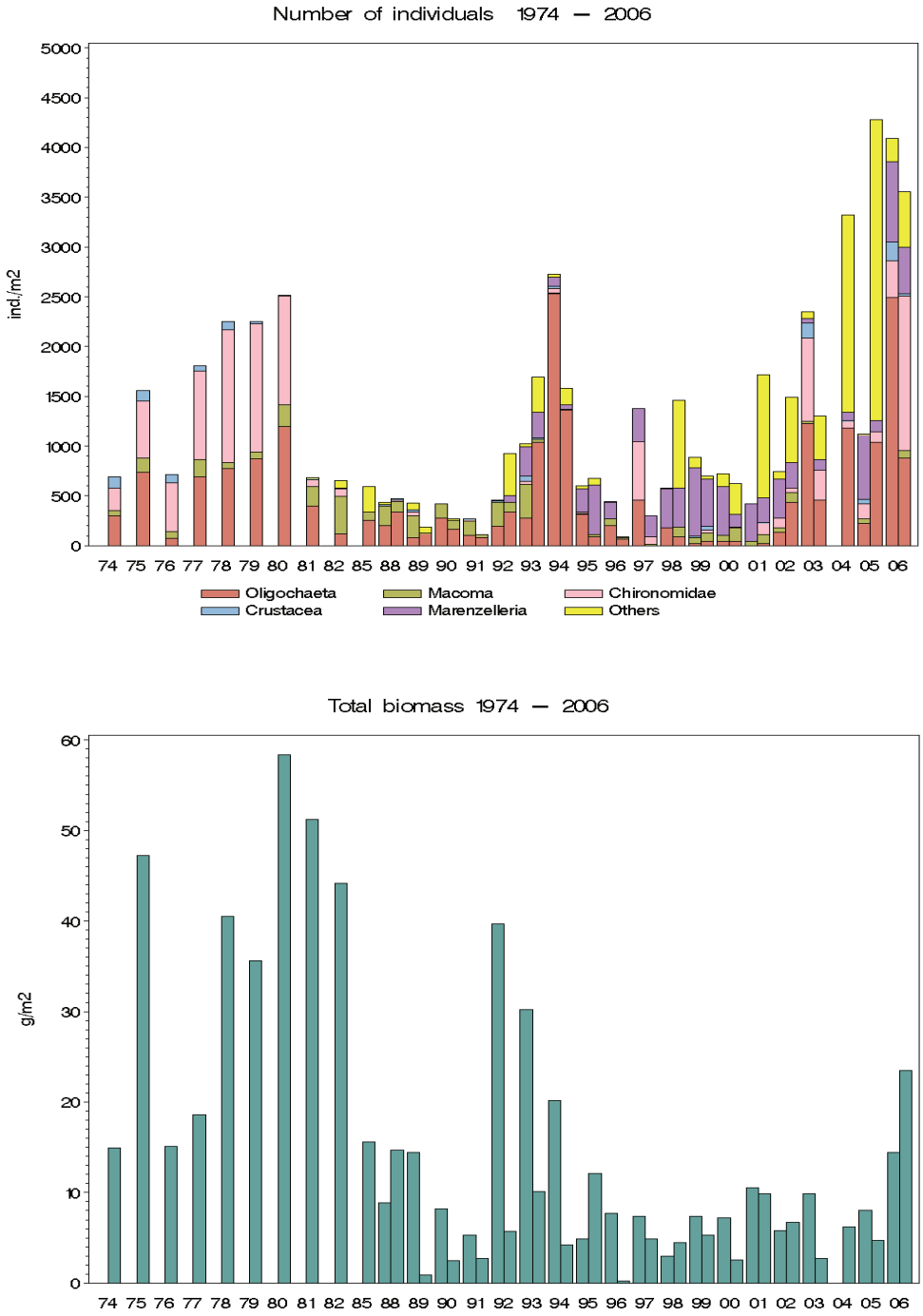


**Fig. 38.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 51 in 1974–2006.

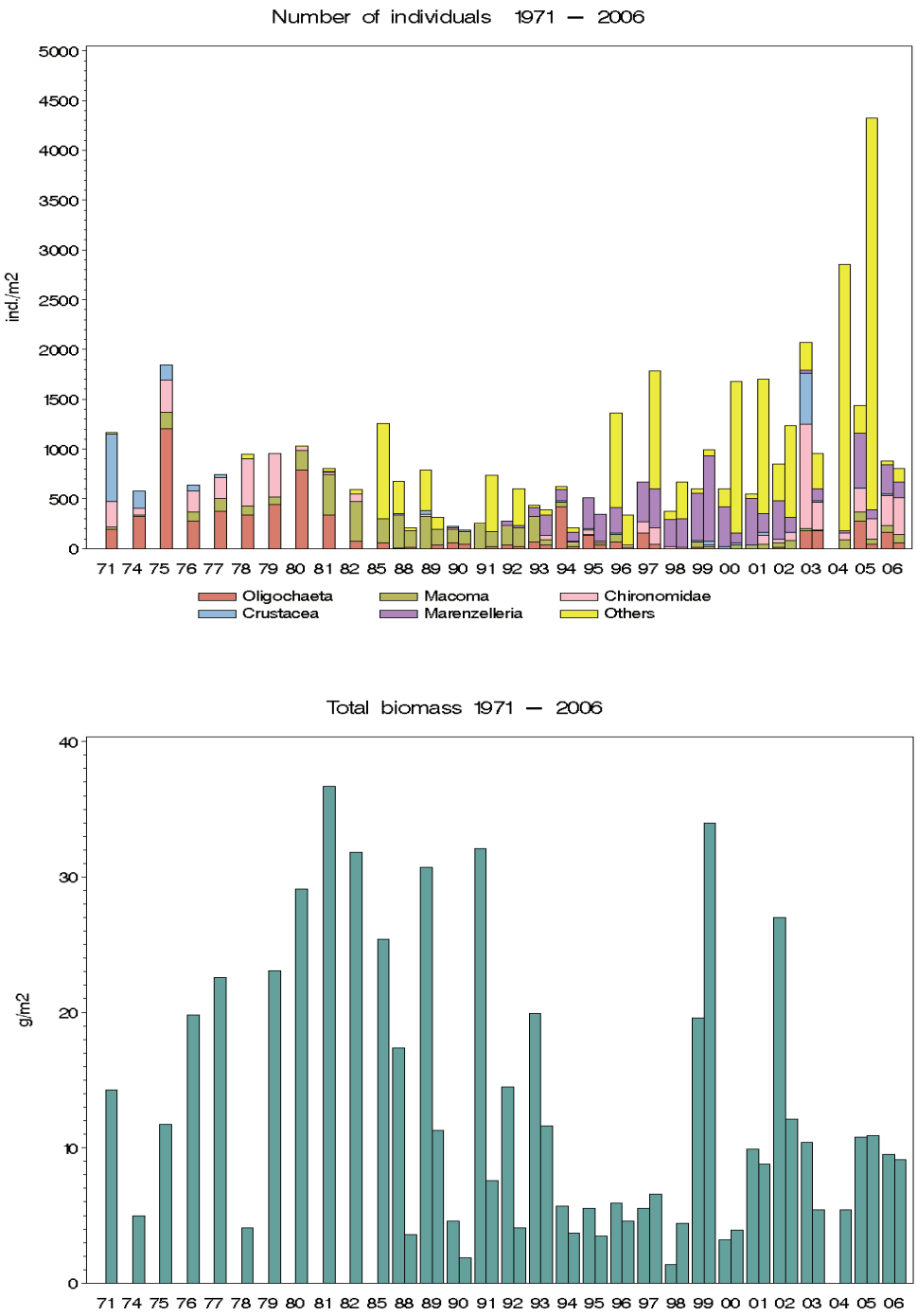


**Fig. 39.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 52 in 1971–2006.

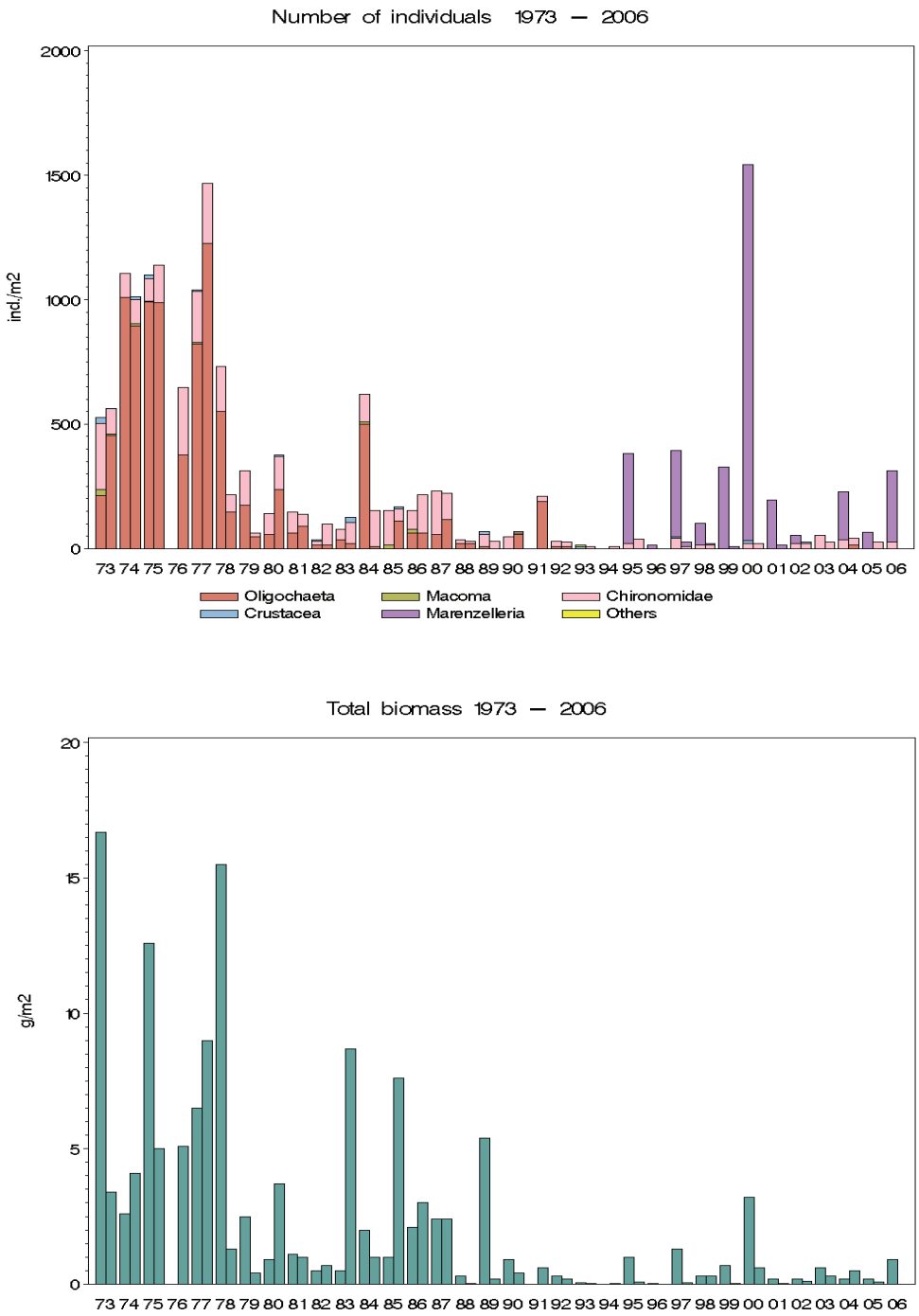


**Fig. 40.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 3 in 1973–2006.

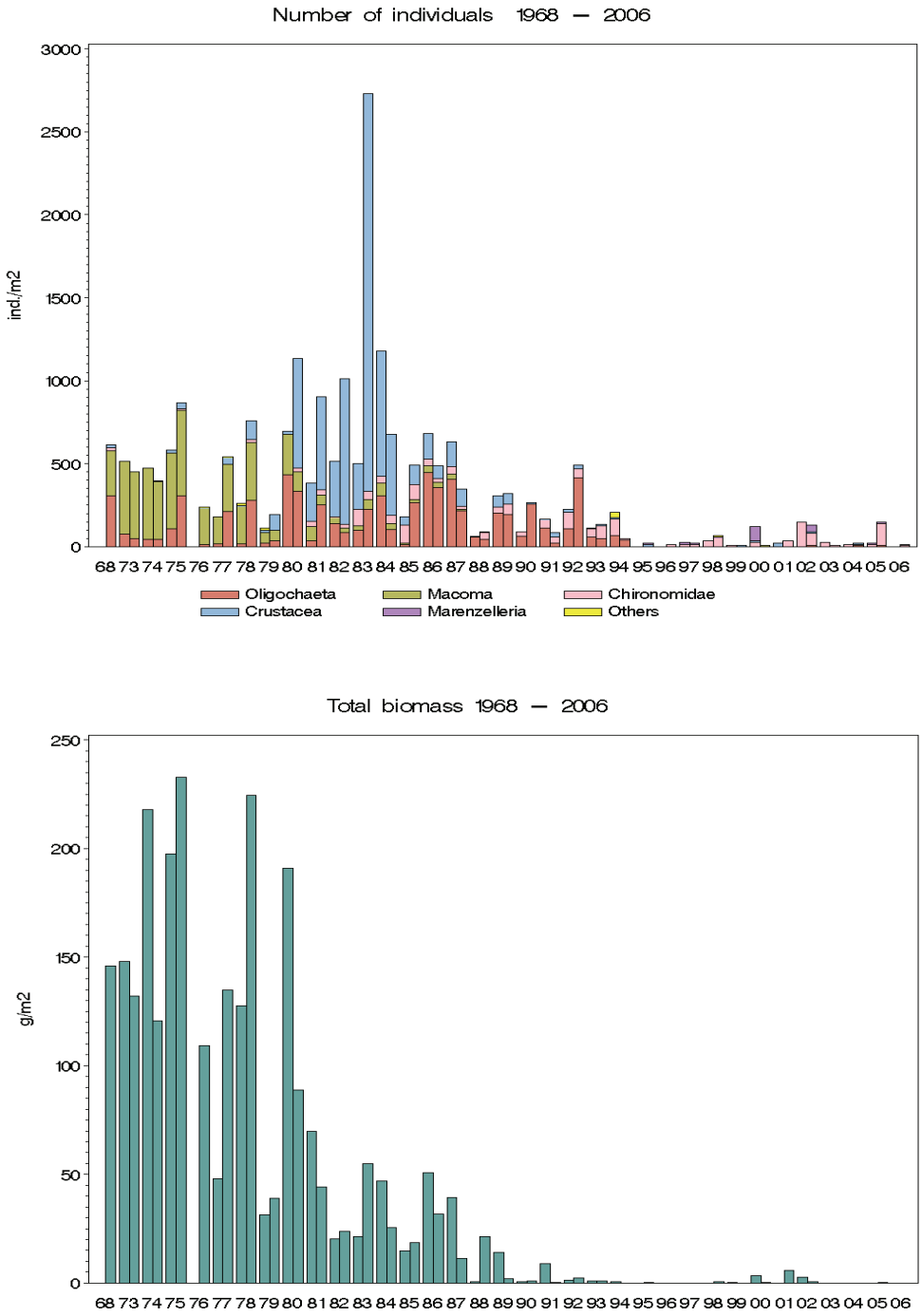


**Fig. 41.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 4 in 1968–2006.

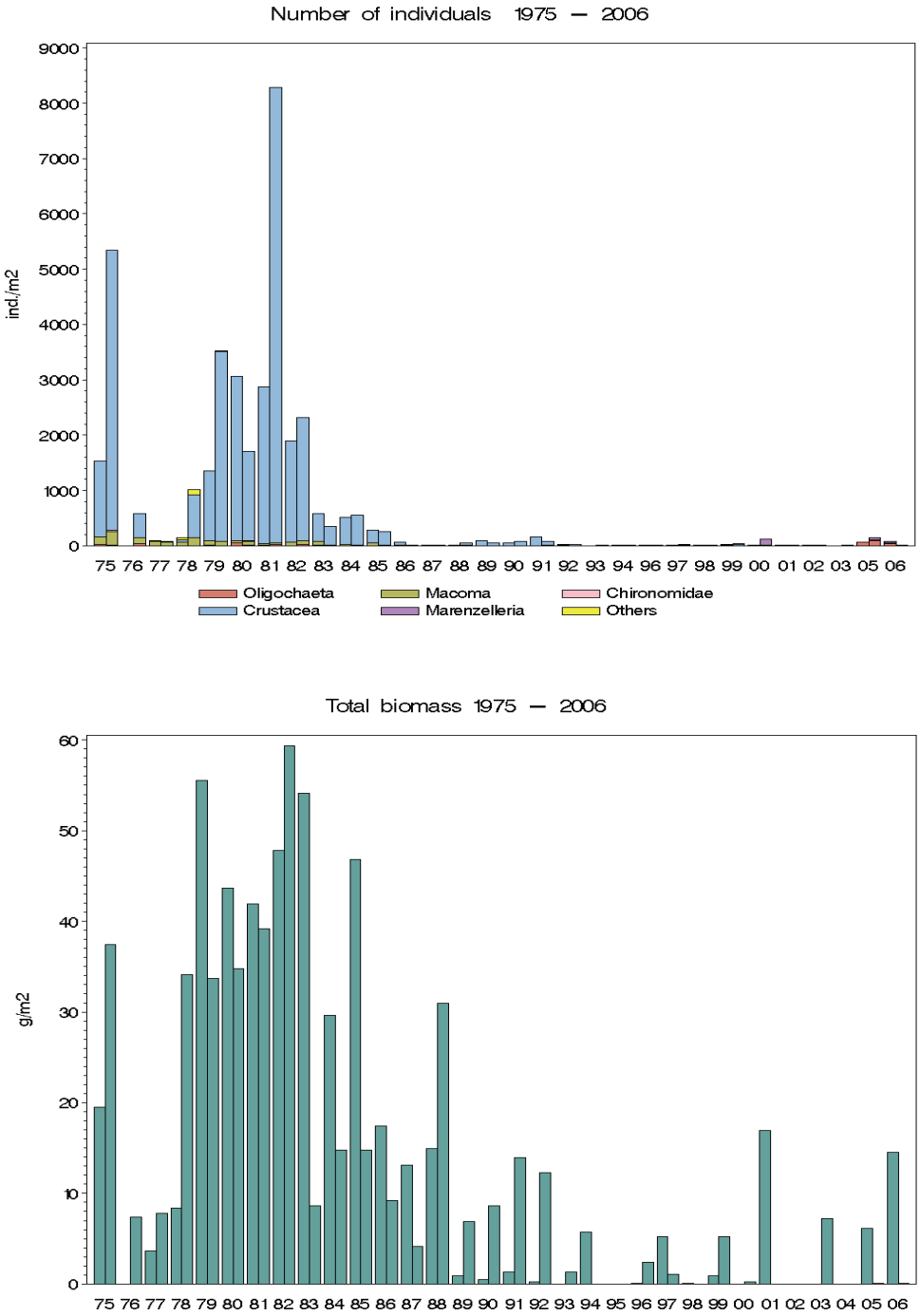


**Fig. 42.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 7 in 1975–2006.

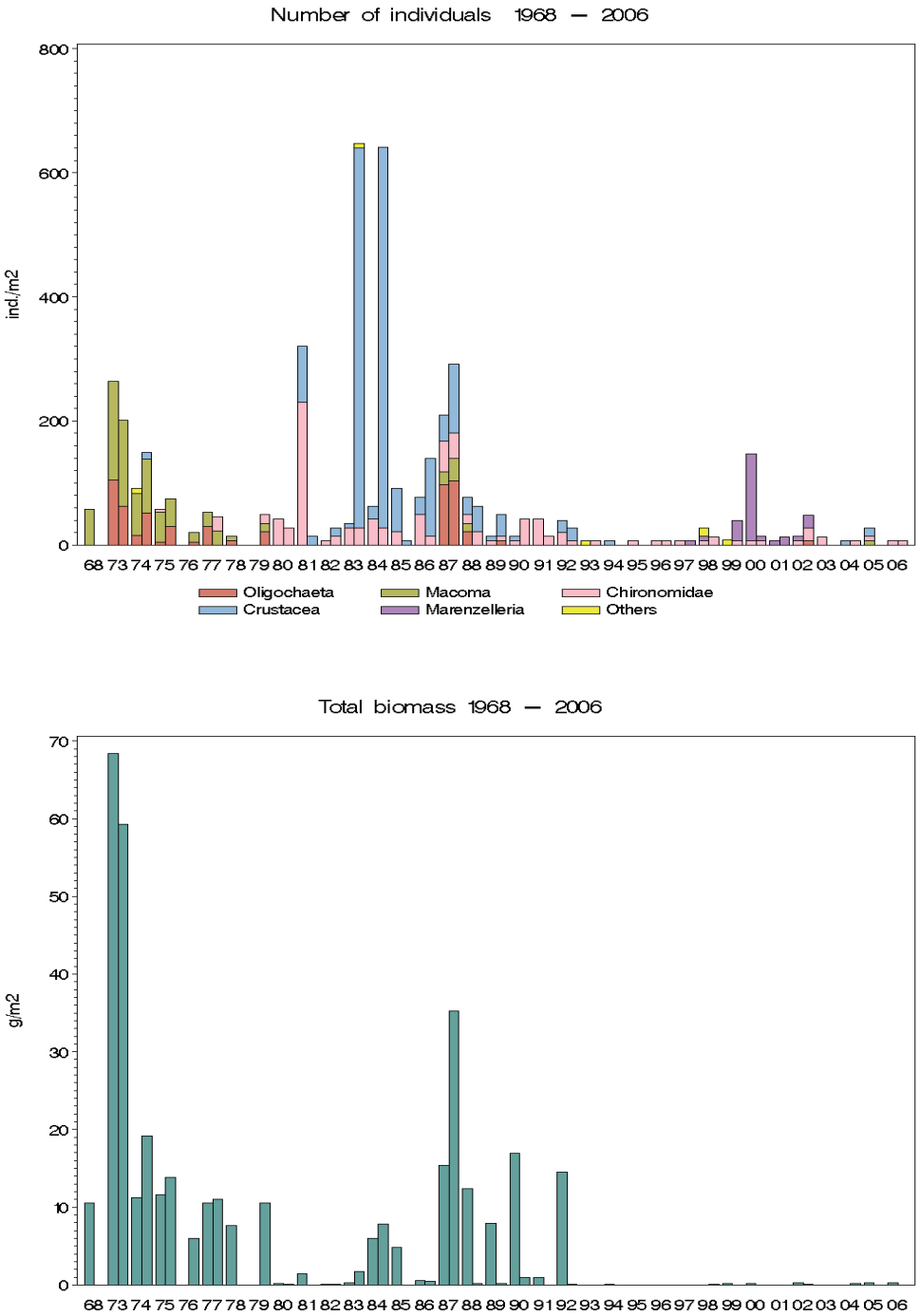


**Fig. 43.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 8 in 1968–2006.

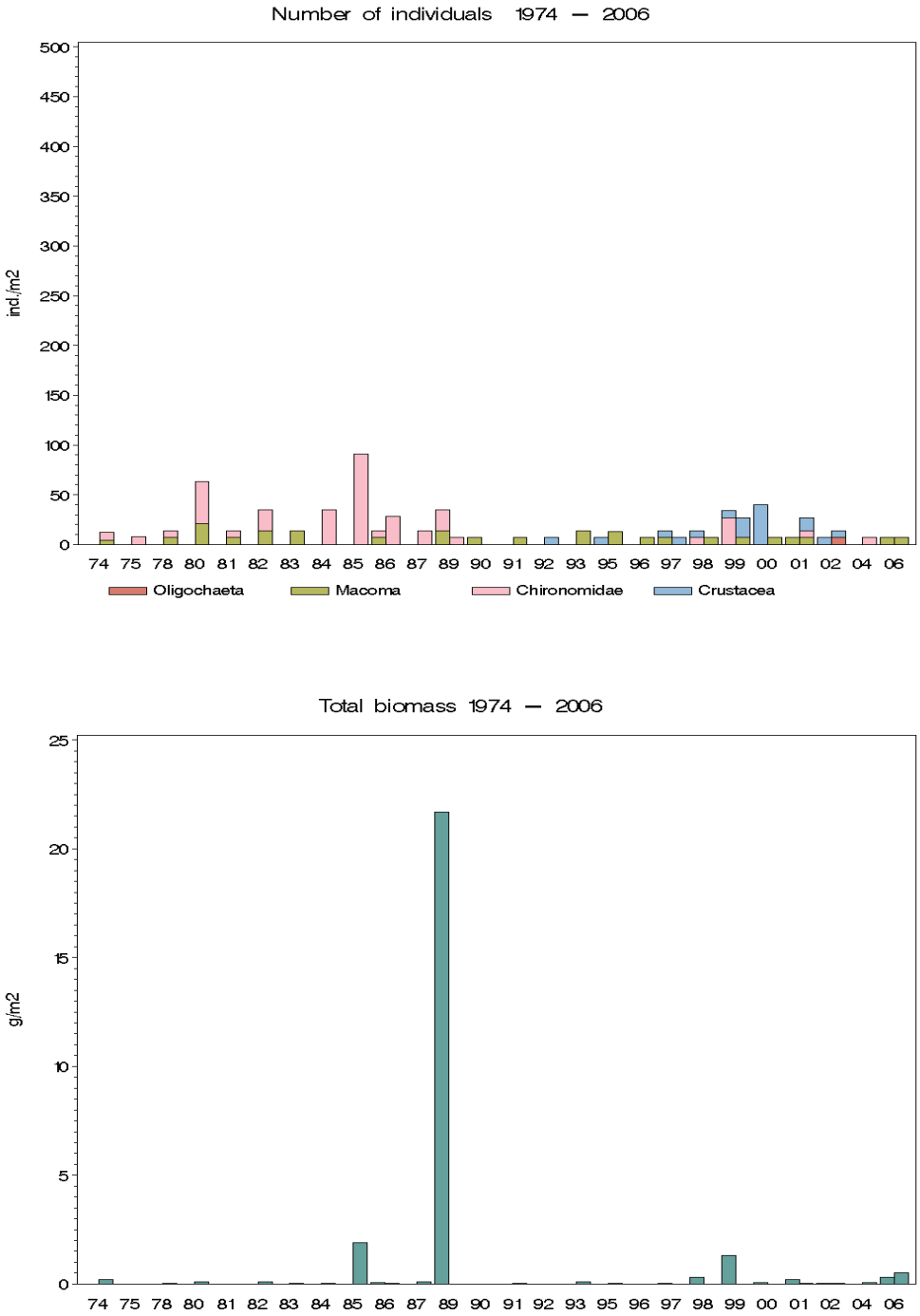


**Fig. 44.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Loviisa 10 in 1973–2006.

# 2 Hydrographical and ecological studies at Olkiluoto

At Olkiluoto, studies focusing on the effects of warm water discharges have been more limited than at Loviisa. Because of the narrower material, the results considered in this examination are mainly the average values of the summer months (June–August). The principal idea of this brief description is to give a basis for comparison with the Loviisa area considered before.

## 2.1 Study area

Olkiluoto is a 6-km-long and 2.5-km-broad island on the west coast of Finland about 12 km north of the town of Rauma. Capes directed towards the NW, shallow bays and small archipelagos between them are characteristic for this part of the west coast. The Bothnian Sea opens to the west of Olkiluoto. To the north, the island borders on the about 1.5-km-broad Eurajoensalmi, and to the south on the 1-km-broad Olkiluodonvesi, which again borders on the archipelago of Rauma (Fig. 45). Two small rivers empty into the sea to the east of Olkiluoto; the Eurajoki River into the eastern end of Eurajoensalmi, and the Lapinjoki River into a narrow sound between Olkiluoto and the mainland. The drainage area of the Eurajoki River is 1 336 $km^2$ and its mean discharge is 9.6 $m^3\ s^{-1}$ while those of the Lapinjoki River are 462 $km^2$ and 3.6 $m^3\ s^{-1}$ (TVO 1999, 2008). The power station is located at the seaward end of the island.

The cooling water for the power plant is surface water, which is taken from Olkiluodonvesi (from the south shore of Olkiluoto) and discharged as a jet stream through a 380-m-long channel into the shallow Iso Kaalonpuhti Bay on the west side of the island. After the discharge, the rapid jet mixes quickly with the receiving water, which means a faster drop in of the highest temperatures soon after the discharge, but a more wide-ranging spread of slightly elevated temperatures (Langford 1990). A rapid mixing of discharged cooling water, which brings about a rapid temperature decline, results in contrast in a slower dilution of the heat from the recipient (Ehlin 1974). On an average, the power station discharges 60 $m^3\ s^{-1}$ of cooling water into the sea, the discharged water being about 11–13ºC warmer than the intake water (TVO 2008). The flow of the cooling water from the intake to the outlet takes 4–5 minutes, and its velocity at the mouth of the discharge channel is about 2 m $s^{-1}$.

In contrast to the Loviisa area, the discharge area at Olkiluoto is quite exposed, and the water exchange with the open Bothnian Sea is more unimpeded. Off-shore of the Iso Kaalonpuhti Bay, the sea is relatively open except for

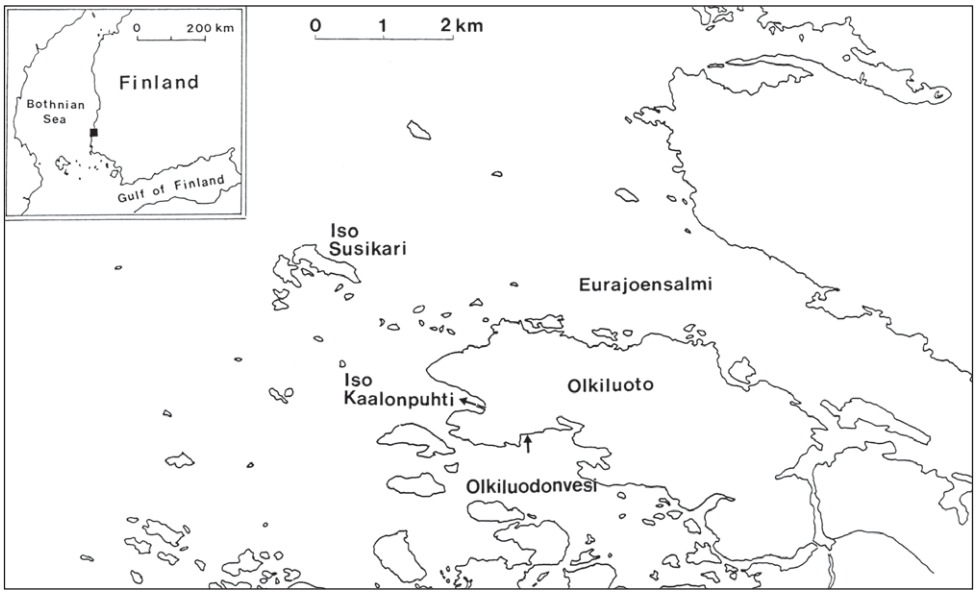


**Fig. 45.** The place-names of the sea areas surrounding the island of Olkiluoto in Eurajoki. The intake and outlet of the cooling water are indicated by arrows.

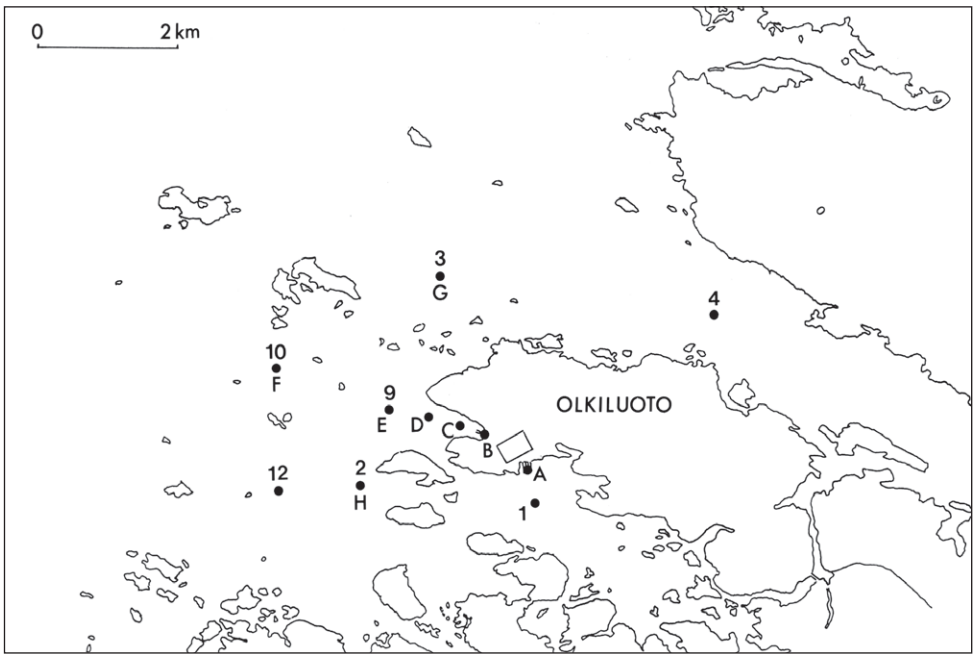


**Fig. 46.** Location of the sampling stations 1, 2, 3, 4, 9, 10, 12 and A–H used in the hydrographical and biological monitoring at Olkiluoto.

2–3 small islets and numerous skerries and rocks. However, the coastal area is quite shallow. The depths are generally less than 10 m, except for two small-area basins to the southwest and to the north of the Iso Kaalonpuhti Bay. The maximum depth in the first-mentioned basin is 15.5 m (Station 12) and in the last-mentioned basin 13 m (Station 3). In Eurajoensalmi and Olkiluodonvesi the depths are below 10 m. The Bothnian Sea coast becomes deeper steadily but very gradually. A depth of 10 m is usually reached outside the outermost islands, a depth of 20 m at a distance of 10–20 km from the mainland, while a depth of 50 m is not reached until about 30 km from the mainland.

The shores of the outer islets and skerries are mostly rocky or stony with a narrow stony field at the shore line. In the inner areas, the shores are characterized by stony fields. The seabed in the areas to the north and west of Olkiluoto is generally dominated by hard erosion bottoms (mainly sand, silt and gravel). A narrow area off-shore of Iso Kaalonpuhti Bay, the basin southwest of Olkiluoto, and the seafloor of Eurajoensalmi and Olkiluodonvesi are characterized by clayey sediments. In general the type of sediment is more solid than at Loviisa, but here too the sediments have changed towards a more organic gyttja-clay type at many stations during recent years.

During the summer months, the mean salinity of the surface water is about 5.5–6.0‰, but in winter the salinity may decrease under the ice to near to zero, especially in the inner bays. Nevertheless, the salinity is clearly higher than at Loviisa, and this difference is reflected in the species richness of the biota. The Common Mussel (*Mytilus edulis*), for example, is common but quite small-sized in the littoral zone. Furthermore, the Baltic Tellin (*Macoma balthica*) gets on much better than at Loviisa: it is a little larger in the Olkiluoto area and dominates there in the benthic fauna. In addition, many polychaetes and the Common Cockle (*Cerastoderma glaucum*) enrich the species composition of the zoobenthos.

## 2.2 Thermal discharges and nutrient load

The amount of heat discharged into the sea from the Olkiluoto power plant is close to double of that discharged from the Loviisa power plant. In recent years, the amount has increased by 13–15% compared to that in 1996, and in 2006 it was 98.8 PJ $a^{-1}$. About 60 $m^3$ $s^{-1}$ of cooling water is used; it is taken from the surface of the sea, and the temperature rises about 11–13ºC in the condensers.

River waters are the most significant source of nutrients in the area. In 2006, the annual load of total phosphorus from the Eurajoki River was 21 500 kg and that of total nitrogen 781 000 kg (Turkki 2007). The nutrient load from the

Lapinjoki River is generally 30–40% of that from Eurajoki. The phosphorus load of the Eurajoki River is comparable with that of Tesjoki at Loviisa, but the nitrogen load of the former is clearly higher. However, the total nutrient load from rivers and point sources other than the power station are clearly higher in the Loviisa area.

The annual load of biological oxygen demand ($BOD_7$), total phosphorus, total nitrogen, ammonium nitrogen and solid matter caused by the communal waste waters from the Olkiluoto power plant are given in Appendix 4. The nutrient load of the power plant has been somewhat higher at Loviisa, especially that of total phosphorus in the late 1970s and in the 1990s, but in recent years the difference has become smaller (*cf.* Appendix 3).

## 2.3 Material and methods

The hydrographical and biological monitoring programme carried out by STUK at Olkiluoto has always been more limited than that at Loviisa. At Olkiluoto, the background studies were initiates in 1972, six years before the construction work on the power station was started. In 1978–1982, STUK also took responsibility for carrying out the obligatory monitoring programme of water quality, but then had to give up this duty due to a lack of human resources. Since then, STUK has continued the hydrobiological monitoring at Olkiluoto up to 2007 with a concise programme for its own needs, i.e., to provide the necessary background information for the radiation monitoring and radioecological studies carried out in the area. In 1972–1982, seawater samples were taken at the permanent sampling stations 5–11 times a year; and primary production was measured 5–10 times a year, so that the majority were carried out during the growing season (May–October). The repertoire of hydrographical parameters has always been the same as that at Loviisa (temperature, salinity, pH, transparency, total phosphorus and total nitrogen). Oxygen measurements were, however, cut out in the 1980s, because the exchange of water seemed to be so effective in the area that no oxygen deficiency appeared.

Since 1982, the sampling frequency decreased, and in the 1990s and 2000s it was approximately 5 times during the growing season, mainly focusing on the summer months (June–August). The results discussed in this examination are thus in general the mean values for the summer months. The *in situ* primary production measurements were reduced as well, but as a counterbalance to this, primary production capacity measurements were continued at eight stations (A–H) at different distances from the cooling water outlet to monitor the distribution pattern of thermal effects in the area. The location of the sampling stations is given in Fig. 46. Soft bottom macrofauna was studied in the sea area

off Olkiluoto at Stations 1, 2, 3, 4, 9, 10 and 12 in 1973–2001. Samples were taken twice a year, in May and August. The results of the monitoring programme up to 1982 were reported in Annual Reports written in Finnish (see Appendix 1).

The results of the phytoplankton and phytobenthos studies carried out in the area in 1972–1982 were summarized in the doctoral thesis of Jorma Keskitalo (1988). In addition, results were published in Ilus (1983) and Ilus et al. (1986a) and in the original research publications of Keskitalo (1987a and b), Keskitalo and Heitto (1987) and Keskitalo and Ilus (1987).

The methods used in the studies at Olkiluoto were the same as those at Loviisa (see p. 38–44).

## 2.4 Temperature of sea water

Cooling water is discharged into Iso Kaalonpuhti Bay as a jet stream through a 380-m-long discharge channel. The stream is strong in Iso Kaalonpuhti Bay just outside the end of the channel, but soon starts to weaken, and is hardly detectable at Station 9, at a distance of 1 200 m from the discharge point. The mixing of the cooling water plume with the surrounding water masses is more efficient in the jet discharge than in the form used at Loviisa. Consequently, although at Olkiluoto the cooling water is taken from the surface, and the thermal load is larger, the average temperature of the surface water has generally been only some degrees higher at Station Olkiluoto 9 than in Hästholmsfjärden at Loviisa. In the 2000s, the maximum temperatures observed in our measurements during the summer months were 29.5ºC at Station 9, 24.0ºC at Station 3, 23.3ºC at Station 2 and 23.3ºC at Station 10. The values are only indicative, because the frequency of our measurements was only 1–2 times a month. The openness of the discharge area and the effective exchange of water play an important role in the lowering of the temperatures in the Olkiluoto area.

The mean surface water temperatures of the summer months (June–August) show a large annual fluctuation caused by the varying weather conditions in different years (Fig. 47). Before the commissioning of the power plant (in 1972–1977) the mean surface water temperatures at Stations 2, 3, 9 and 10 were fairly much the same. Since then, there has been an increasing trend in the values. Those at Stations 2, 3 and 10 have generally followed each other quite in parallel, whereas the increase of temperatures has been much stronger at Station 9 situated directly in front of the cooling water outlet. During the summer months of 1972–1977, the mean surface water temperatures were on an average 15.5 ± 1.4 and 15.3 ± 1.5ºC at Stations 2 and 3. In 2000–2006, the corresponding average values were 18.3 ± 1.6 and 18.3 ± 2.0ºC. This means that the surface water temperature of the summer months increased by 2.8–3.0ºC at

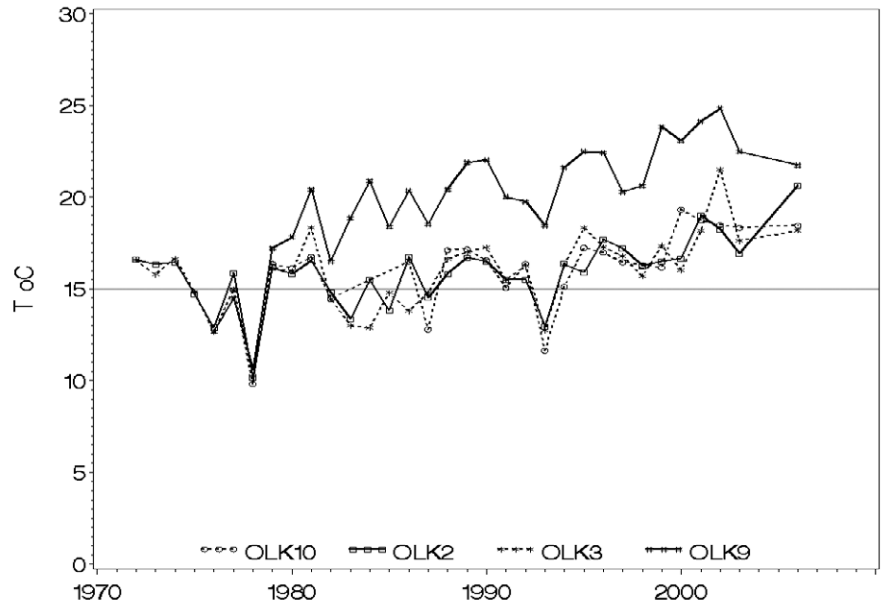


**Fig. 47.** Mean surface water temperatures of the summer months (June–August) at the Olkiluoto stations 2, 3, 9 and 10 in 1972–2006.

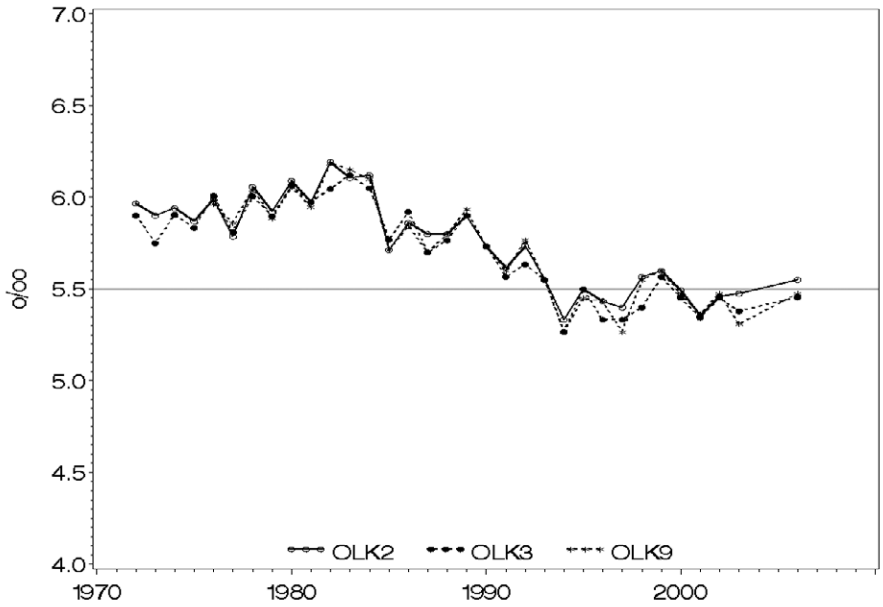


**Fig. 48.** Mean surface water salinity (‰) of the summer months (June–August) at the Olkiluoto stations 2, 3 and 9 in 1972–2006.

these stations during the 30 years. In 2000–2006, the corresponding value for Station 9 was 23.3 ± 1.2ºC. Since 1983, the average difference between Stations 2 and 9 has been 5.0 ± 1.3ºC (Fig. 47).

## 2.5 Water salinity

The salinity of the surface water is approximately 1‰ higher in the sea area off Olkiluoto than in Hästholmsfjärden at Loviisa. In Olkiluoto, too, the salinity varies significantly according to the season. In winter, river water from the Eurajoki and Lapinjoki Rivers may spread under the ice up to the study area, and may form a distinct freshwater layer on the surface of the recipient. Because fresh water also spreads to Olkiluodonvesi and the cooling water is taken from the surface layer there, the salinity of the discharged cooling water does not substantially deviate from that in Iso Kaalonpuhti Bay. Furthermore, because the warm water is efficiently mixed by the jet discharge in the wide ice free area in front of the outlet, similar "lenses" of warm water have not been noticed under the ice as at Loviisa. The lowest salinities observed in the study area in early spring in the 1970s were 0.6–0.7‰ in surface water at Stations 3 and 9.

Due to the shallowness of the area and the good exchange of water, the vertical salinity gradient in the water is quite insignificant; in general the salinity in the near-bottom water is only slightly higher than in the surface water. The highest salinity observed in the area was 6.74‰ at Station 2 in 1979. Until 1982, there was a slightly increasing trend in the salinity of the surface water, but after that the trend has been opposite (Fig. 48). In 1982, the mean salinity of the surface water during the summer months at Station 2 was 6.19‰, whereas it was 5.36‰ in 2001. Since then the salinity has slightly increased. The decrease of salinity has been a common trend in the whole Bothnian Sea (HELCOM 2002).

## 2.6 Water transparency

Basically, the transparency of the water is better in the Bothnian Sea than in the Gulf of Finland. In the background state, the transparency was more than 1 m higher in the discharge area of Olkiluoto than at Loviisa. In 1972, the mean *Secchi* disc value of the growing season was 446 ± 55 cm at Station 2 at Olkiluoto, while it was 330 ± 70 cm in Hästholmsfjärden off Loviisa. However, the transparency of the water also decreased in the Olkiluoto area during the 30 years (Fig. 49). The highest values recorded at the Olkiluoto stations 2, 3 and 9 were 620–720 cm and the lowest 50–110 cm. During the summer months of 1972–1977, the mean *Secchi* disc values at Stations 2 and 3 were on average

475 ± 73 cm and 483 ± 99 cm, respectively. In 2000–2003, the corresponding averages were 399 ± 56 cm and 396 ± 78. This means that the water transparency of the summer months decreased by 80–90 cm at these stations during the 30 years. The decreasing trend was common at all the stations; thus it was probably linked to the general eutrophication trend in the area, but might partly be due to the increased turbidity of the water caused by the cooling water current in the Iso Kaalonpuhti Bay. In 2000–2003, the average *Secchi* depth was 288 ± 56 cm at Station 9 situated at the mouth of the bay.

## 2.7 Total phosphorus and total nitrogen

In general, the levels of nutrients in the seawater are clearly lower in the Bothnian Sea than in the Gulf of Finland (HELCOM 2002, 2003); indeed the conditions in the whole Gulf of Bothnia differ considerably from those in the Gulf of Finland. The Gulf of Bothnia is sheltered from the main mass of the nutrient-rich deep waters of the Baltic Proper by a ridge, shaped by several underwater thresholds and the Archipelago Sea. Therefore, only small volumes of the eutrophied Baltic water below the halocline enter the Gulf of Bothnia. In addition, the external nutrient loading is smaller and areas receiving considerable internal loading are lacking in the Gulf of Bothnia (Kauppila and Bäck 2001, Pitkänen 2004).

During the summer months of 1972–1977, the mean total phosphorus values of the surface water were on average 10 ± 3 µgP $l^{-1}$ at Stations 2 and 3. In 2000–2003, the corresponding averages were 16 ± 2 µgP $l^{-1}$ at Station 2 and 17 ± 2 µgP $l^{-1}$ at Stations 3 and 9. This means that the concentrations had increased by 60–70% in the area, but that no significant difference had developed between Station 9 and Stations 2 and 3. In Hästholmsfjärden off Loviisa (Loviisa station 2), the average total phosphorus concentration during the summer months of 2000–2006 was 24 µgP $l^{-1}$. Thus, the concentrations were about 1.5 times higher at Loviisa. An extensive increase in the winter values of total phosphorus was also found in the open Bothnian Sea between 1979–1983 and 1991–1998 (Kauppila and Bäck 2002). In general, the coastal waters are more eutrophied compared with the offshore areas. This is due on the one hand to the nutrient load received by coastal waters and on the other hand to the intensified nutrient cycle between the bottom and the water in shallow coastal areas. Furthermore, the weakened exchange of water with the open sea due to islands and shoals intensifies the eutrophication sensitivity of coastal waters (Pitkänen 2004).

The maximum concentrations of total phosphorus observed in the surface water during the whole study period were 108 and 91 µgP $l^{-1}$ at Stations 3 and 9 in April 1987, and 68 and 69 µgP $l^{-1}$ at Stations 10 and 2 in May 1996. The peak

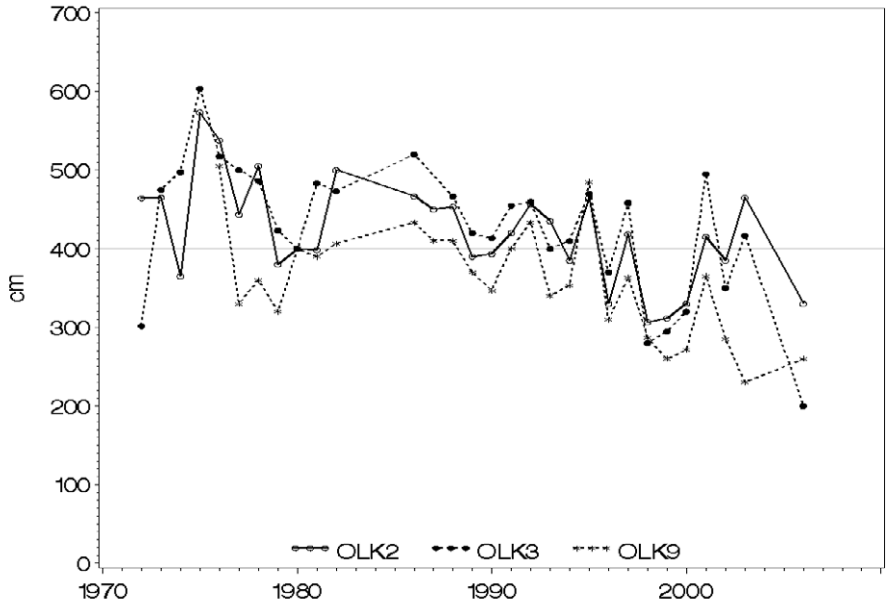


**Fig. 49.** Mean *Secchi* depths (cm) of the summer months (June–August) at the Olkiluoto stations 2, 3 and 9 in 1972–2006.

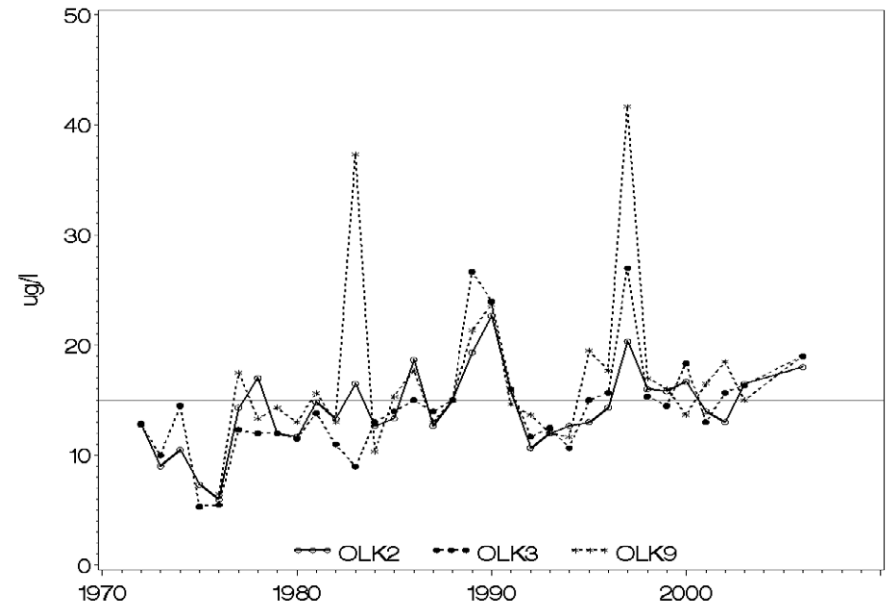


**Fig. 50.** Average total phosphorus concentrations (µgP $l^{-1}$) of the summer months (June–August) at the Olkiluoto stations 2, 3 and 9 in 1972–2006.

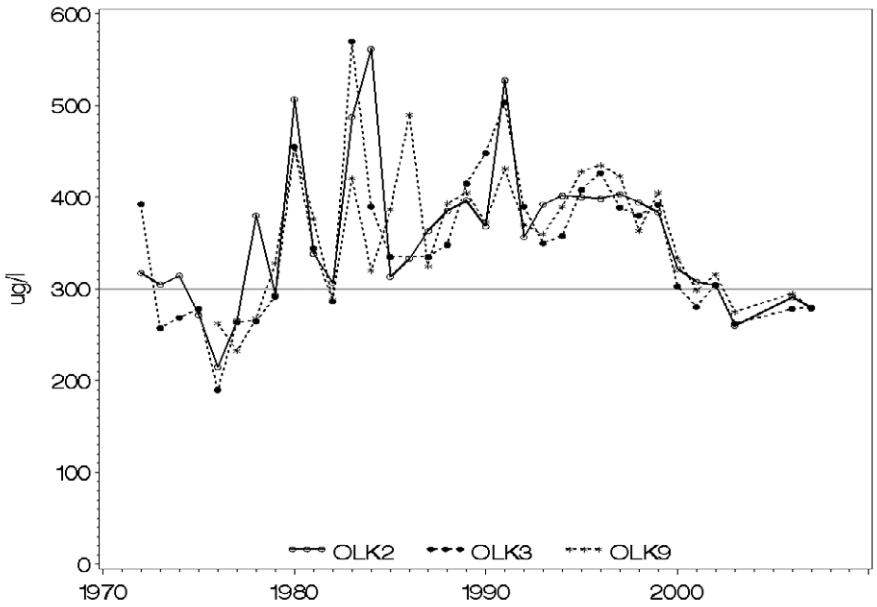


**Fig. 51.** Average total nitrogen concentrations (µgN $l^{-1}$) of the summer months (June–August) at the Olkiluoto stations 2, 3 and 9 in 1972–2006.

values in the average total phosphorus concentrations (Fig. 50) found at Station 9 in 1983 (37 µgP $l^{-1}$) and 1997 (42 µgP $l^{-1}$) were clearly due to local discharges.

The total nitrogen concentrations of surface water increased during the 1970s and 1980s until 1991, the mean values of the summer months being highest in the mid-1980s (Fig. 51). Since 1991, the values have decreased to the same level as in the early 1970s. During the summer months 1972–1977, the mean total nitrogen values of the surface water were on average 282 ± 39 µgN $l^{-1}$ and 275 ± 66 µgN $l^{-1}$ at Stations 2 and 3, respectively. In 2000–2003, the corresponding averages were 299 ± 27 µgN $l^{-1}$ at Station 2, 291 ± 19 µgN $l^{-1}$ at Station 3 and 306 ± 25 µgN $l^{-1}$ at Station 9. The mean of Station 9 was only slightly higher than those of Stations 2 and 3. In Hästholmsfjärden off Loviisa (Loviisa station 2), the average total nitrogen concentration during the summer months of 2000–2006 was 420 µgP $l^{-1}$. Thus, the concentrations were about 1.4 times higher at Loviisa.

As a conclusion, the total phosphorus concentrations of the surface water may still show a slowly increasing trend, but the total nitrogen concentrations have started to show a clear decrease. These trends are contradictory with the loading data of the power plant, which showed an increasing trend for nitrogen and a decreasing trend for the phosphorus load during recent years (Appendix 4).

This indicates that the concentrations of nutrients are mainly associated with the general trends along this part of the Bothnian Sea coast.

## 2.8 Primary production and primary production capacity

*In situ* primary production was measured at Olkiluoto at 2–4 stations in 1972–1982. The annual primary production values based on these measurements are given in Table 11.

**Table 11.** Annual phytoplankton primary production (g C $m^{-2}$ $a^{-1}$) at the Olkiluoto stations 2, 3, 9 and 10 in 1972–1982.

| Station | 2 | 3 | 9 | 10 |
|---|---|---|---|---|
| 1972 | 30 | 28 | – | – |
| 1973 | 60 | 60 | – | – |
| 1974 | 36 | 35 | – | – |
| 1975 | 35 | 33 | – | – |
| 1976 | 33 | 30 | 29 | – |
| 1977 | 26 | 27 | – | – |
| 1978 | 36 | 31 | – | 27 |
| 1979 | 28 | 22 | 28 | 24 |
| 1980 | 32 | 35 | 36 | 36 |
| 1981 | 38 | 35 | 46 | 33 |
| 1982 | 34 | 31 | 33 | 32 |

The level of primary production was low; the annual production values were generally about a half of those at Loviisa. Vernal maximum values were much lower, and the mean values of the summer months were about a half of those at Loviisa. The effect of warm water was visible only in 1981, when the annual production was about 30% higher at Station 9 than at the other stations. Since then, the level of primary production has increased in the area, but remained at a clearly lower level than at Loviisa. According to Turkki (2007), the annual primary production in 2006 was 69 g C $m^{-2}$ $a^{-1}$ at Station 9 and 56 g C $m^{-2}$ $a^{-1}$ at Station 2.

In 1983–1988, *in situ* measurements were still continued at 3–4 stations, but the frequency of the measurements was reduced, so that it was no longer relevant to assess the annual production. Since 1989, *in situ* measurements were continued only at Station 2, and these only 4–5 times a year, focusing on the summer months. The mean production of the summer months at Station 2 varied strongly, but tended to increase in the 1980s and 1990s (Fig. 52).

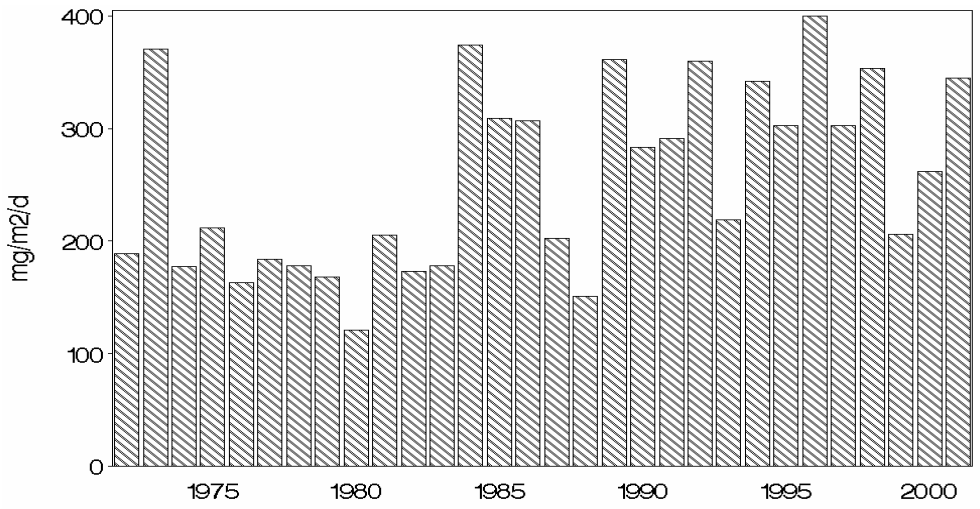


**Fig. 52.** Mean *in situ* primary production during the summer months (mgC $m^{-2}$ $d^{-1}$) at Olkiluoto 2 in 1972–2001.

While the average production of the summer months was 185 mg C $m^{-2}$ $d^{-1}$ in 1972–1982, it was 311 mg C $m^{-2}$ $d^{-1}$ in 1989–2001.

In the same way as the *in situ* values, the primary production capacity values also increased since the 1970s. The highest daily values of primary production capacity are given in Table 12.

**Table 12.** The highest primary production capacity values measured at Olkiluoto in 1978–2001.

| Station | Maximum PP capacity mg C $m^{-3}$ $d^{-1}$ | Date |
|---|---|---|
| 1 | 233 | 4 May 1994 |
| A | 242 | 4 May 1994 |
| B | 377 | 19 August 1992 |
| C | 430 | 19 August 1992 |
| D | 251 | 12 July 1989 |
| 9 | 355 | 23 April 1981 |
| 10 | 336 | 23 April 1981 |
| 2 | 273 | 11 August 1998 |
| 3 | 262 | 21 May 1980 |
| 4 | 384 | 21 May 1980 |

The highest value (430 mg C $m^{-3}$ $d^{-1}$) was measured at Station C, just outside the mouth of the cooling water channel, but the next highest (384 mg C $m^{-3}$ $d^{-1}$) was recorded at Station 4, situated in Eurajoensalmi north of Olkiluoto, where the nutrients from the Eurajoki and Lapinjoki Rivers probably raise the productivity. However, the mean primary production capacity values during the summer months only occasionally exceeded the limit of eutrophic (200 mg C $m^{-3}$ $d^{-1}$) at Station 9 in 1989 and 1998, but not once at Station 2 (Fig. 53). Thus, the values generally indicated only slight eutrophication.

In 1988–2001, primary production capacity was studied during the summer months at 8 stations. These were situated at the intake (Station A) and outlet (Station B) of the cooling water, and at different distances from the outlet (Stations C–H). The location of the stations is shown in Fig. 46. Station B was located just at the base of the cooling water channel, right where the heated water discharges from the tunnel into the open channel. The distribution pattern of primary production capacity in the samples taken at different distances from the outlet point in 1989, 1990, 1994, 1998 and 2001 is shown in Fig. 54. In some cases the PP capacity in the out coming water (Station B) was higher than that in the intake water, but in most cases the effect was the reverse. This means that in some cases the productivity of phytoplankton increased during entrainment through the cooling water systems, where it was exposed to a sudden temperature rise of 10–13ºC and to mechanical stress within the 4–5 minutes when passing from the intake to the outlet. However, the productivity seemed to increase more only after the exit from the cooling water channel. The productivity just seemed to reach its maximum at a distance of about 1 500 m from the outlet, where the temperature had already decreased and the plankton had had enough time to adapt to the prevailing conditions. It is quite another question, what share of the plankton at Station E had passed through the cooling system. In every case, the water at Station B was purely, that at Station C still rather purely, and that at Station D at least partly composed of the out-flowing cooling water. After Station E, the primary production capacity decreased with distance from the outlet (Fig. 54). Results from studies carried out at the Barsebäck NPP, Sweden, indicated that the passage through the cooling system hindered the assimilation of phytoplankton when the temperature in the intake area was lower than ca. 10ºC and stimulated the phytoplankton when the temperature was higher than ca. 10ºC (Edler et al. 1980).

Stuart & Stanford (1978) noted a significant (≈ 13 percent) stimulation of planktonic primary productivity due to power plant entrainment in a reservoir in south-western U.S.A.

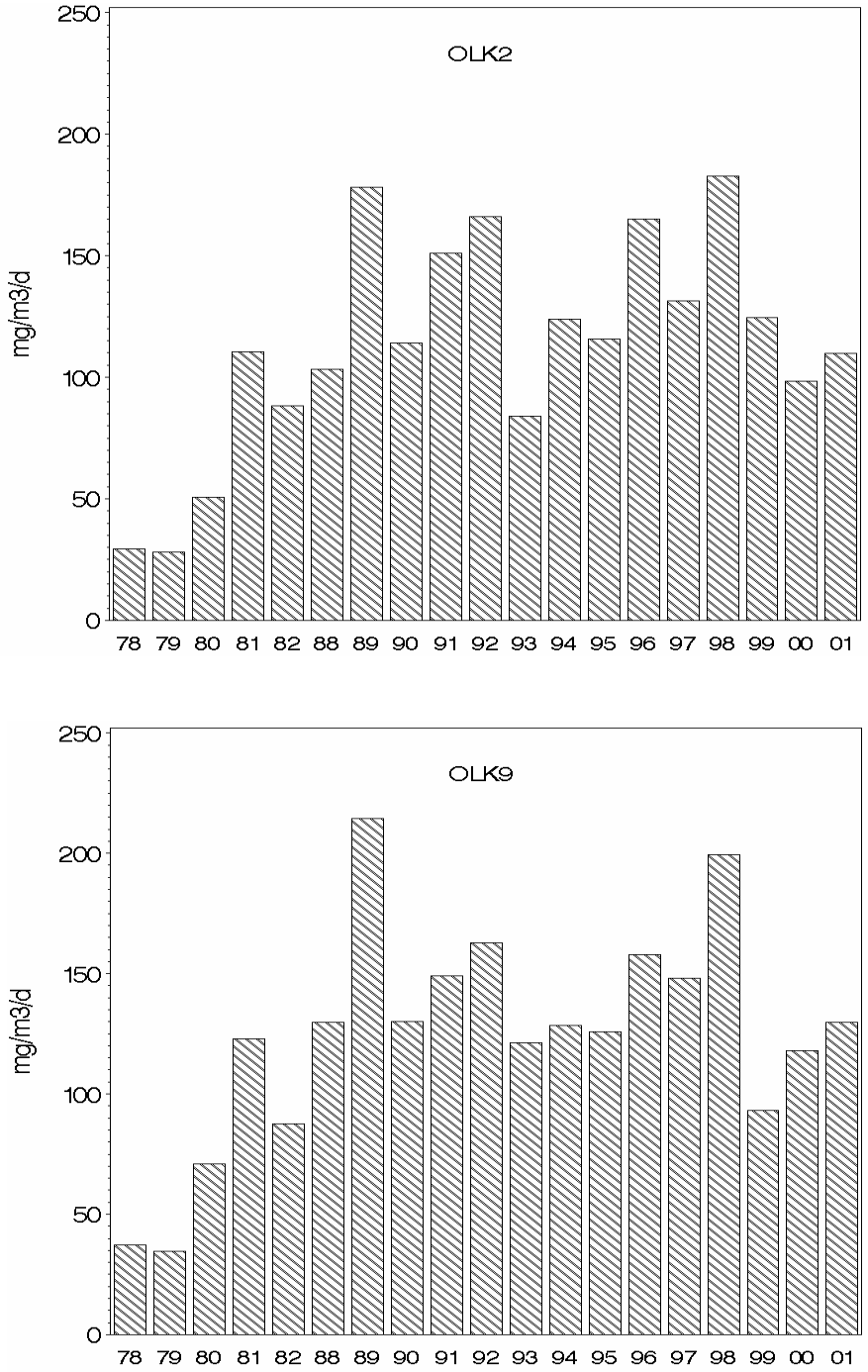


**Fig. 53.** Mean primary production capacity (mg C $m^{-3}$ $d^{-1}$) of the summer months (June–August) at the Olkiluoto stations 2 and 9 in 1978–2001.

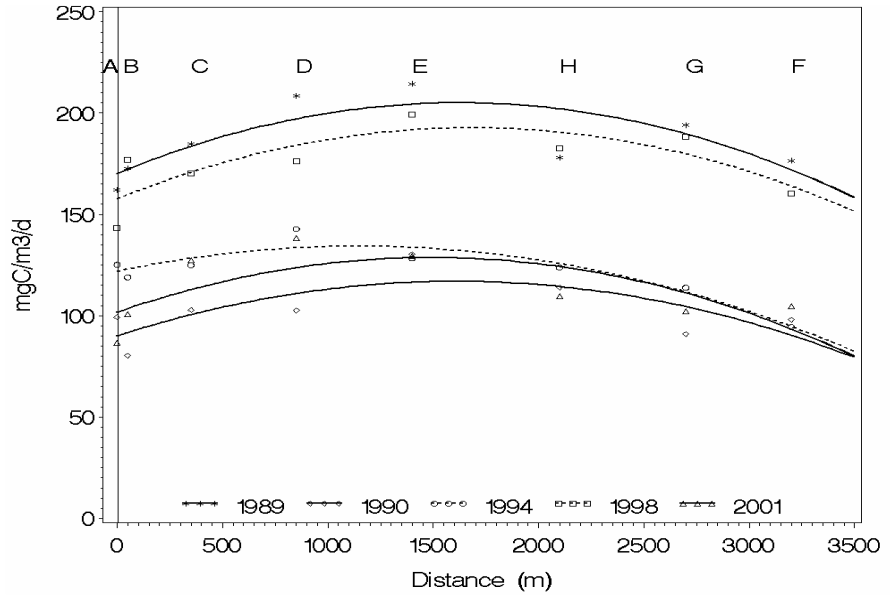


**Fig. 54.** Variation of mean primary production capacity (mgC $m^{-3}$ $d^{-1}$) of the summer months (June–August) at the Olkiluoto stations A–F as a function of distance from the cooling water outlet (m) in 1989, 1990, 1994, 1998 and 2001. Station A shows the reference level at the cooling water intake.

## 2.9 Benthic fauna

Owing to the higher salinity of the water, the macrozoobenthos in the Olkiluoto sea area is from the start richer in species, more diversiform and more abundant than at Loviisa. While the number of taxa detected at the permanent benthos stations was 57 at Loviisa, the corresponding number was 92 at Stations 1, 2, 3, 4, 9, 10 and 12 around Olkiluoto. While the tubificid *Potamothrix hammoniensis* and the larvae of the *Chironomus plumosus* group were the core species at Loviisa, and 22 of the 57 taxa were chironomid larvae, the benthic communities at Olkiluoto were dominated by *Macoma balthica* and polychaetes, and the species list included many more crustaceans (27 species) and molluscs (15 species). The immigrant polychaete, *Marenzelleria* sp., was observed for the first time at Olkiluoto in 1992, and since then it has also colonized the benthic communities there.

Since the seabed seaward of Olkiluoto is mainly characterized by erosion bottoms, two of the seven stations in our programme represent this bottom type: at Station 3 there is a relatively hard silt bottom and at Station 10 a sand and gravel bottom. At the soft bottom stations 1, 2, 4, 9 and 12, the sediment

in the early 1970s was sulphidic clay or relatively solid sulphidic gyttja-clay, but a clear change towards more organic and more watery deposits has also occurred at Olkiluoto, at least at Stations 12, 2 and 9. As was stated before, oxygen measurements were deleted from our programme in the 1970s, because, in the open and shallow sea area without any isolated deeps, the exchange of water seemed to be effective, and, a notable deficiency of oxygen did not appear in the near-bottom water at that time. However, the later monitoring studies carried out by the Water and Environment Research of Southwest Finland have shown that the state of oxygen has deteriorated in the area during recent years (Jumppanen 2002, Mattila 2004). Especially in front of Iso Kaalonpuhti Bay, the amount of algal debris has increased on the bottom, the decomposition of which has led to occasional depletion of oxygen in the near-bottom water.

Changes in the abundance of main species or taxonomic groups (ind. $m^{-2}$) and the total biomass of macrozoobenthos (g $m^{-2}$) at Stations 1, 2, 3, 4, 9, 10 and 12 during the study period are shown in Figs. 55–61.

**Station 1**

Station 1 is situated offshore of the cooling water intake, in the middle of Olkiluodonvesi. The depth at the station is 6.5 m; in 2003 the bottom was sulphidic clay with a 1–2-cm thick oxic layer on its surface. *Monoporeia affinis* (Crustacea) was the dominant species at the station in the mid-1970s and early 1980s. The highest density of *Monoporeia* was 5 790 ind. $m^{-2}$ in August 1983 (Fig. 55). Large fluctuations in abundance, typical for the normal population dynamics of *Monoporeia* (Segerstråle 1960), occurred in the 1970s, 1980s and 1990s, but the upturns in abundance were reduced in the 1990s, and since 1998 the species was no longer observed at the station. A permanent core species in the benthic community was *Macoma balthica*, the highest density of which was 2 170 ind. $m^{-2}$ in August 1977. *Marenzelleria* sp. appeared at the station in 1993, but has never reached dominance in the benthic community; the highest density was 680 ind. $m^{-2}$ in 1999. The amphipod *Corophium volutator* (Crustacea) occurred quite abundantly in 1973–1974, 1991 and in 1995–1996, when its maximum density was 480 ind. $m^{-2}$. Other by-species in the benthic community were *Hediste diversicolor, Saduria entomon, Potamopyrgus antipodarum, Procladius* sp. and *Potamothrix hammoniensis*. The total abundance of the macrozoobenthos was at its highest, 7 050 ind. $m^{-2}$, in August 1983 due to the peak occurrence of *Monoporeia.* The total biomass values fluctuated considerably, mainly according to the size-class-fluctuation of *Macoma balthica.* The maximum value of 283 g $m^{-2}$ was reached in May 1997, due to the abundance of large-sized individuals of *Macoma.* In 2001, the abundance of *Macoma* was quite average, but the individuals were small.

**Station 2**

Station 2 is situated to the south-west of Olkiluoto, to the west of the sound between the islands of Kuusinen and Lippo, in the easternmost part of the area of continuous accumulation bottom. The depth at the station is 14 m; in 2003 the bottom was sulphidic gyttja-clay with a 1–2-cm thick oxic layer on its surface. During recent years the character of the bottom has changed, becoming softer and more organic. In 2003, the gyttja-clay bottom was very soft just below the fluffy, oxic surface layer, but already turned to sulphidic clay at a depth of 10 cm. *Macoma balthica* has always been the unquestioned dominant species at the station, but its abundance has strongly fluctuated over the course of decades. Now and then, the mussel has almost totally disappeared (1985–1987), but then multiplied again (1989–1990). The fluctuation was probably associated with the population dynamics of the species, but the declined abundance in the late 1990s and early 2000s might also be associated with the changes in the quality of the bottom, with the progressive eutrophication process, or with competition/predation caused by *Marenzelleria* sp. This immigrant polychaete appeared at the station in 1992, and was the most abundant species in August–September 1996, 1997, 1999 and 2001 (Fig. 56). It may effectively prey on the juvenile stages of *Macoma.* The density of *Macoma* was, at its highest, 4 880 ind. $m^{-2}$ in August 1982, when the total abundance of the macrozoobenthos was 5 360 ind. $m^{-2}$. The highest density of *Marenzelleria* was 3 890 ind. $m^{-2}$ in September 1999, when the total abundance of macrofauna was 5 800 ind. $m^{-2}$. The biomasses followed closely the fluctuation of the large size-classes of *Macoma*. The most abundant by-species were *Potamothrix hammoniensis,* larvae of the *Chironomus halophilus* group, *Prostoma obscurum* and *Tubifex costatus.*

**Station 3**

Station 3 is located to the north-west of Olkiluoto and to the east of the island of Iso Susikari. The depth is 13 m and the seabed is fine sand – silt. In recent years, decaying algal residues have often been met with on the bottom. *Macoma balthica* has been the core species at the station, but as a whole the species composition has been more diversified than at the other stations. Altogether 16 crustaceans *(Praunus flexuosus, P. inermis, Neomysis integer, Gammarus oceanicus, G. salinus, G. zaddachi, Monoporeia affinis, Bathyporeia pilosa, Corophium volutator, Leptocheirus pilosus , Saduria entomon, Idotea baltica, I. chelipes, Jaera albifrons, J. praehirsuta, Asellus aquaticus)*, 6 polychaetes *(Harmothoë sarsi, Hediste diversicolor, Marenzelleria* sp.*, Pygospio elegans, Polydora redeki, Manayuncia aestuarina)*, the priapulid worm *Halicryptus spinulosus* and the soft-shell clam *Mya arenaria* were detected in the benthos samples. The *Gammarus* species were especially abundant in August 1983, and the larvae of

the *Chironomus halophilus* group in 1985 and in May 1986, when their density was 3 300 ind. $m^{-2}$ and the maximum total abundance of macrozoobenthos was 3 640 ind. $m^{-2}$ (Fig. 57). The density of *Macoma* was at its highest, 2 050 ind. $m^{-2}$, in 1993. *Marenzelleria* was observed at the station for the first time in 1993, and achieved dominance in the benthic community in August 1996, September 1998, September 1999 and September 2001. In September 1999 its density was 2 470 ind. $m^{-2}$. The small polychaete *Pygospio elegans* was abundant in 1992–1995, having a maximum density of 560 ind. $m^{-2}$ in 1994. The most common by-species in the benthic community were *Prostoma obscurum, Hediste diversicolor, Saduria entomon, Corophium volutator, Potamopyrgus antipodarum, Monoporeia affinis, Halicryptus spinulosus* and *Tubifex costatus.* Here too, the total biomass of the macrozoobenthos correlated closely with the amount of large-sized *Macoma* individuals in the samples; the maximum was 333 g $m^{-2}$ in 1981.

**Station 4**

Station 4 is situated in Eurajoensalmi north of Olkiluoto. The depth is 9 m, and the seabed is sulphidic clay with a thick oxic layer on its surface. *Monoporeia affinis* (Crustacea) dominated in the benthic fauna in 1977 and then in 1981–1985 (Fig. 58). In August 1984, its density was, at its highest, 6 930 ind. $m^{-2}$, and in August 1985 *Monoporeia* was still the main constituent in the benthic community, when the total abundance of the macrozoobenthos was 10 770 ind. $m^{-2}$, but after that it has completely disappeared from the species composition. *Monoporeia* has not been recorded at the station since May 1987. The tubificid *Potamothrix hammoniensis* (Oligochaeta) occurred abundantly at the station in 1982–1987, being most abundant in August 1985 (1 320 ind. $m^{-2}$). However, *Macoma balthica* has permanently been a core species in the benthic community. Its density varied during the whole study period between 360 and 3 140 ind $m^{-2}$ (average 916 ind. $m^{-2}$), and in 1990–2001 between 635 and 3 140 ind. $m^{-2}$ (average 1 276 ind. $m^{-2}$), the highest abundance being recorded in June 1993. *Marenzelleria* sp. appeared at the station in 1993 and was most abundant in August 1996 (1 190 ind. $m^{-2}$). The most abundant by-species were *Potamopyrgus antipodarum, Hediste diversicolor, Saduria entomon, Prostoma obscurum* and the chironomid larvae of *Procladius* sp. and the *Chironomus halophilus* and *Chironomus plumosus* groups. The total biomass of the macrozoobenthos correlated with the amount of large-sized *Macoma* individuals, and was at its highest 266 g $m^{-2}$ in May 1989.

**Station 9**

Station 9 is situated seaward of Iso Kaalonpuhti Bay at a distance of 1 200 m from the mouth of the cooling water channel. The depth at the station is 10 m;

in 2003 the bottom was sulphidic gyttja-clay with an oxic layer on its surface. During recent years, the character of the bottom has changed to become softer and more organic. In 2003, there was a 1–2-cm thick fluffy mud layer on its surface, then a gyttja-clay layer down to 10 cm, a black gyttja layer with gas bubbles at a depth of 10–14 cm, while sulphidic clay occurred from 25 cm downwards. *Macoma balthica* has been permanently the dominant species at the station, but its abundance has fluctuated strongly in the course of decades in the same way as at Station 2. Now and then, the mussel almost totally disappeared (1984, 1997–1998 and 2000), but then multiplied again (Fig. 59). The fluctuation was probably associated with the population dynamics of the species. The density of *Macoma* was highest in September 1993 (5 370 ind. $m^{-2}$) and June 2001 (4 930 ind. $m^{-2}$). In spite of its high abundance in 1999 and 2001, the biomasses were low due to the majority of small individuals in the population. The most common by-species were *Hediste diversicolor, Potamothrix hammoniensis, Potamopyrgus antipodarum,* and the larvae of the *Chironomus halophilus* group, *Prostoma obscurum* and *Procladius* sp. The density of *Chironomus halophilus* was at its highest 916 ind. $m^{-2}$ and that of *Potamothrix hammoniensis* 728 ind. $m^{-2}$. *Marenzelleria* sp. was detected at the station for the first time in 1994, and its density reached a maximum in 2001 (2 370 ind. $m^{-2}$). *Monoporeia* was found at the station only twice, in small quantities, and has not been observed at the station since May 1986. The total biomass of the macrozoobenthos correlated with the amount of large-sized *Macoma* individuals, and was at its highest, 335 g $m^{-2}$, in May 1989. The low biomasses in 1999 and 2001 arose from the low average weight of *Macoma* (Fig. 59).

**Station 10**

Station 10 is located seaward of Iso Kaalonpuhti Bay at a distance of 3 km from the cooling water channel. The depth at the station is 9 m and the seabed is sand and gravel. Since the power plant came into operation, drifting organic debris (mainly decaying plant residues) has been transported from Iso Kaalonpuhti Bay to the station by the cooling water flow, and has collected on the seabed (Ilus 1983). However, the debris does not remain there for long, but is swept away before any sediment has time to accumulate on the erosion bottom. Thus, the sieving residues of the benthos samples contained in general plenty of organic material and only a little sand and gravel. Consequently, chironomid larvae were the most abundant group found in the samples. In June 1994, small unidentified chironomid larvae (*Chironominae* sp.) yielded a record of 23 630 ind. $m^{-2}$, and the total abundance of the macrozoobenthos was 26 840 ind. $m^{-2}$ at the station (Fig. 60). Other taxa abundantly detected in the samples were *Chironomus halophilus* group, *Tanytarsini* sp., *Marenzelleria* sp. (since 1994), *Gammarus*

spp., *Asellus aquaticus*, *Macoma balthica, Nais elinguis, Tubifex costatus, Stylaria lacustris* and *Jaera albifrons*. In all, 54 taxa were recorded at the station. In spite of the high number of species and individuals, the total biomasses were low. In 1988 and 1994, the maximum values were 211 and 201 g $m^{-2}$, but in general the biomasses were below 50 g $m^{-2}$.

**Station 12**

Station 12 is situated south-west of Olkiluoto near to the Vähä Kivikkokari islet. It is the deepest point (15.5 m) in the soft bottom area south-west of Olkiluoto. The bottom sediments consist of sulphidic gyttja-clay with a 1–2-cm thick oxic layer on its surface, and sulphidic clay from 25 cm downwards. The character of the bottom has changed towards a softer and more organic type during recent years. *Macoma balthica* has remained the unquestioned dominant species at the station, although its abundance has fluctuated strongly in the same way as at Stations 2 and 9 (Fig. 61). Its density was at its highest, 5 710 ind. $m^{-2}$, in August 1982, and its lowest, 46 ind. $m^{-2}$, in August 1986, while the average density for the whole study period was 1 780 ind. $m^{-2}$. Thus, the abundance of *Macoma* remained at about the same level during the study period. *Marenzelleria* sp. appeared at the station in 1993, and was very abundant in August 1996 (5 880 ind. $m^{-2}$) and September 1999 (4 612 ind. $m^{-2}$), but otherwise its density was quite low. In August 1996, the total abundance of the macrozoobenthos was 9 140 ind. $m^{-2}$. The most abundant by-species were the tubificids *Tubifex costatus* and *Potamothrix hammoniensis*, the larvae of *Chironomus halophilus* gr. and *Prostoma obscurum*. The total biomass of the macrozoobenthos correlated with the abundance of large-sized *Macoma*. The highest biomass was 749 g $m^{-2}$ in June 1988, but tended to decrease in the late 1990s, although the number of *Macoma* did not show any decreasing trend, which proves the significance of the average biomass of *Macoma* in the biomass values (Fig. 62).

**In conclusion**

In spite of strong inter-annual fluctuation, the macrozoobenthos has remained quite robust in the sea area off Olkiluoto. The fluctuation was mainly due to the varying abundance of *Macoma balthica,* the occasional decline or size-class changes of which were especially eye-catching in the biomass values. Mattila (1993) reported on the deterioration of bottom fauna communities in the offshore areas along the Finnish coast of the southern Bothnian Sea during the 1980s. This general trend was stated as being associated with the overall eutrophication of the Baltic Sea. Some of the changes in the Olkiluoto area, such as the disapperance of *Monoporeia affinis,* were probably also due to the general progress of eutrophication in the Bothnian Sea.

Changes in the quality of the bottom were an obvious reason for the changes in the macrozoobenthos. In addition to the decaying algal residues found repeatedly on the seabed at Stations 3 and 10, a general change towards more organic, muddy deposits was also observed at most other stations (at least at Stations 2, 9 and 12), and this probably affected the abundance of, e.g., *Macoma balthica*. Snoeijs and Mo (1987) and Snoeijs (1988) recognized that, e.g., *Potamopyrgus antipodarum* (= *Paludestrina jenkinsi*) and oligochaetes were favoured by the cooling water discharge in the Forsmark area on the opposite side of the Bothnian Sea. Similarly, the abundance of *Potamopyrgus antipodarum,* the oligochaete *Potamothrix hammoniensis* and the chironomid larvae of the *Chironomus halophilus* gr. and *Procladius* sp. increased during the study period at Station 9 offshore of the cooling water outlet at Olkiluoto, which evidently interlocks with local effects of thermal discharges.

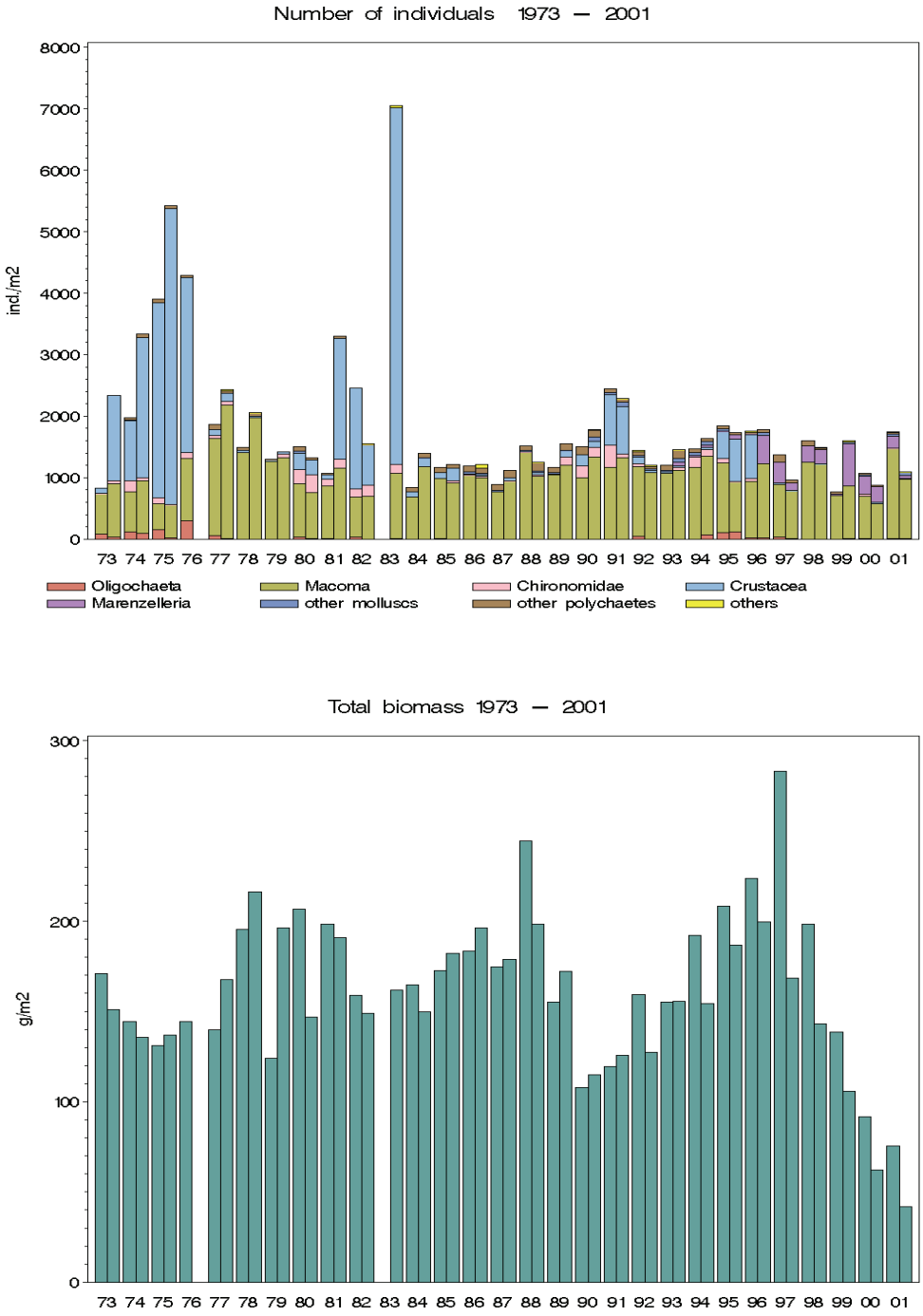


**Fig. 55.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 1 in 1973–2001.

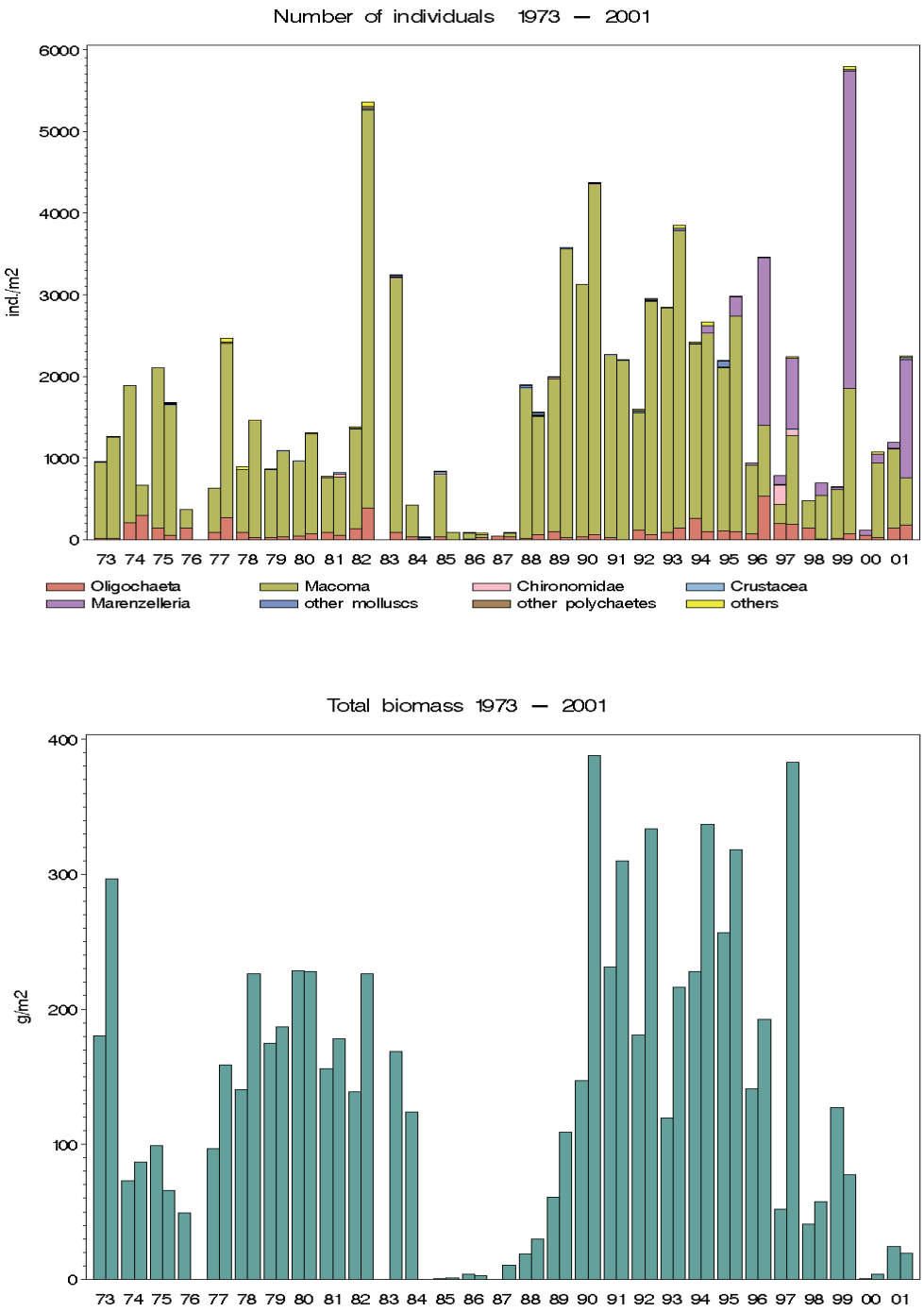


**Fig. 56.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 2 in 1973–2001.

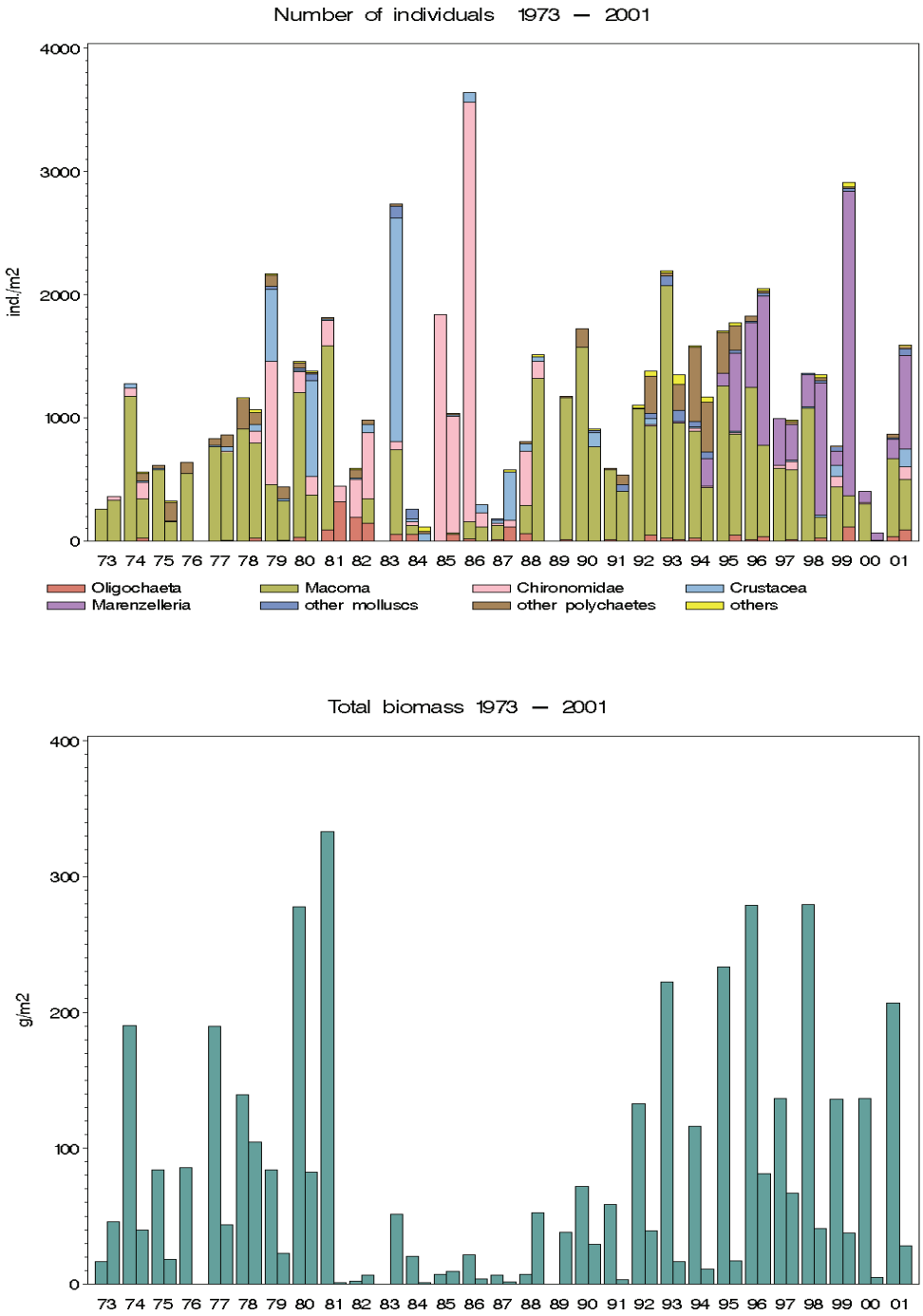


**Fig. 57.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 3 in 1973–2001.

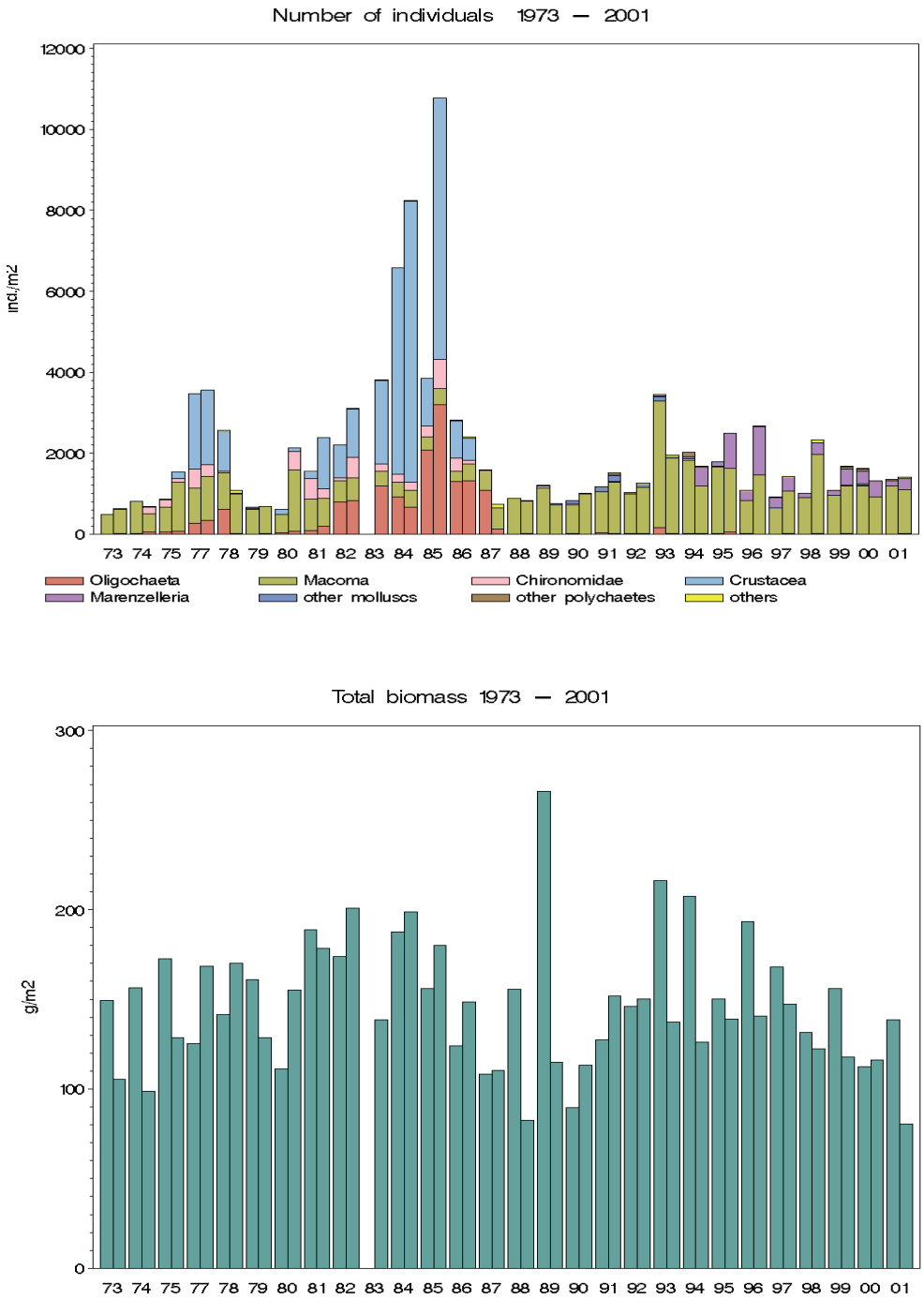


**Fig. 58.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 4 in 1973–2001.

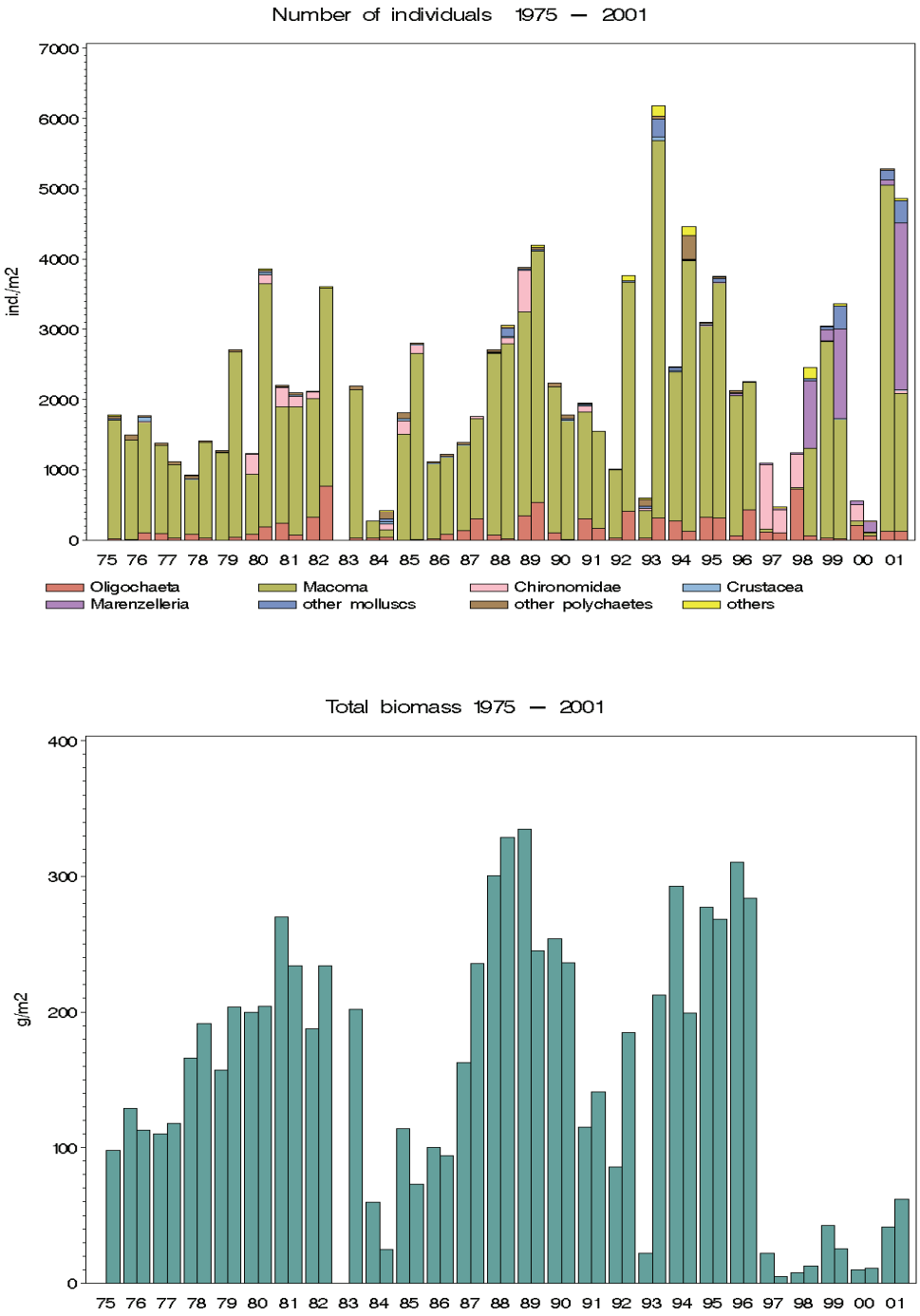


**Fig. 59.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 9 in 1975–2001.

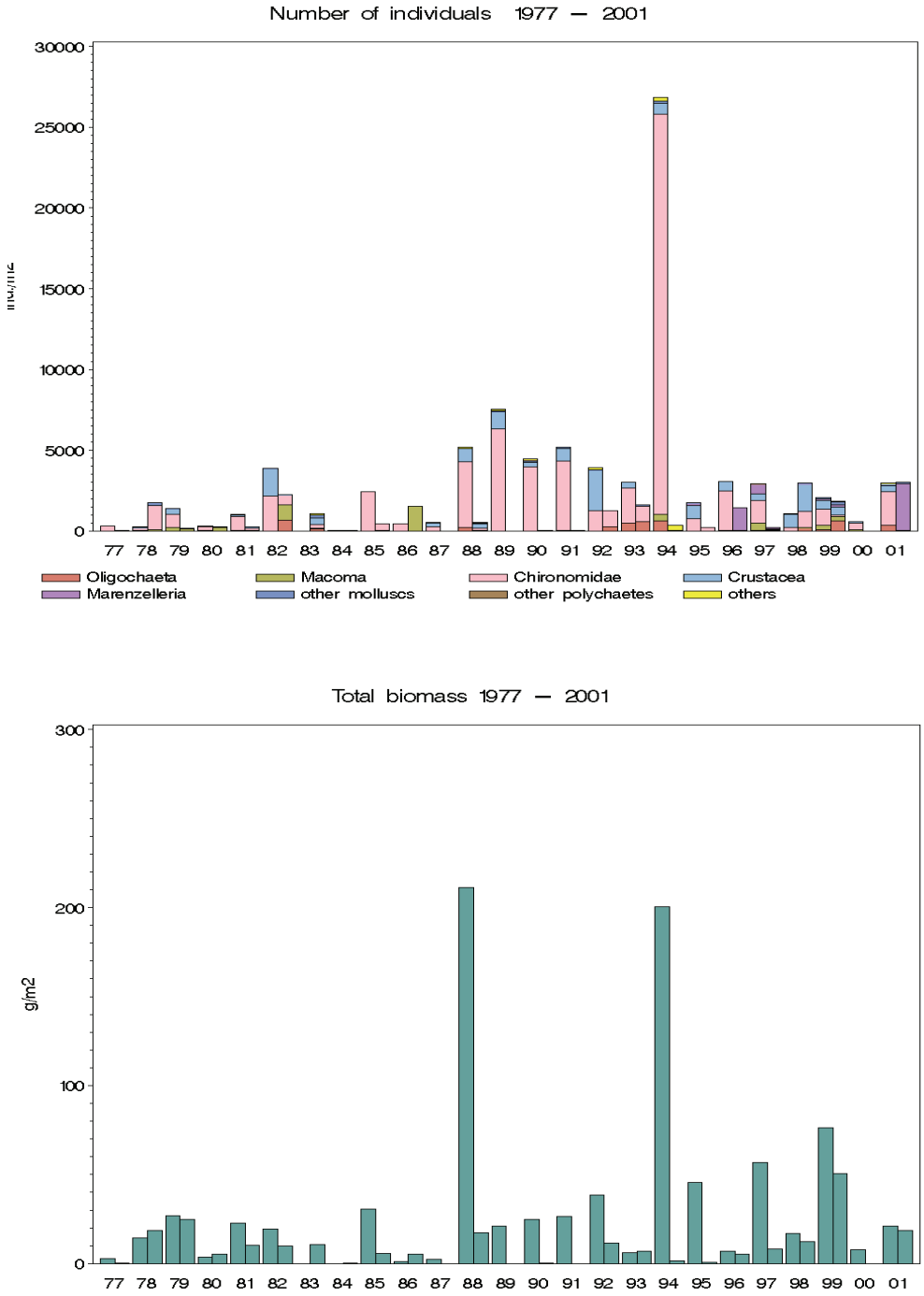


**Fig. 60.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 10 in 1977–2001.

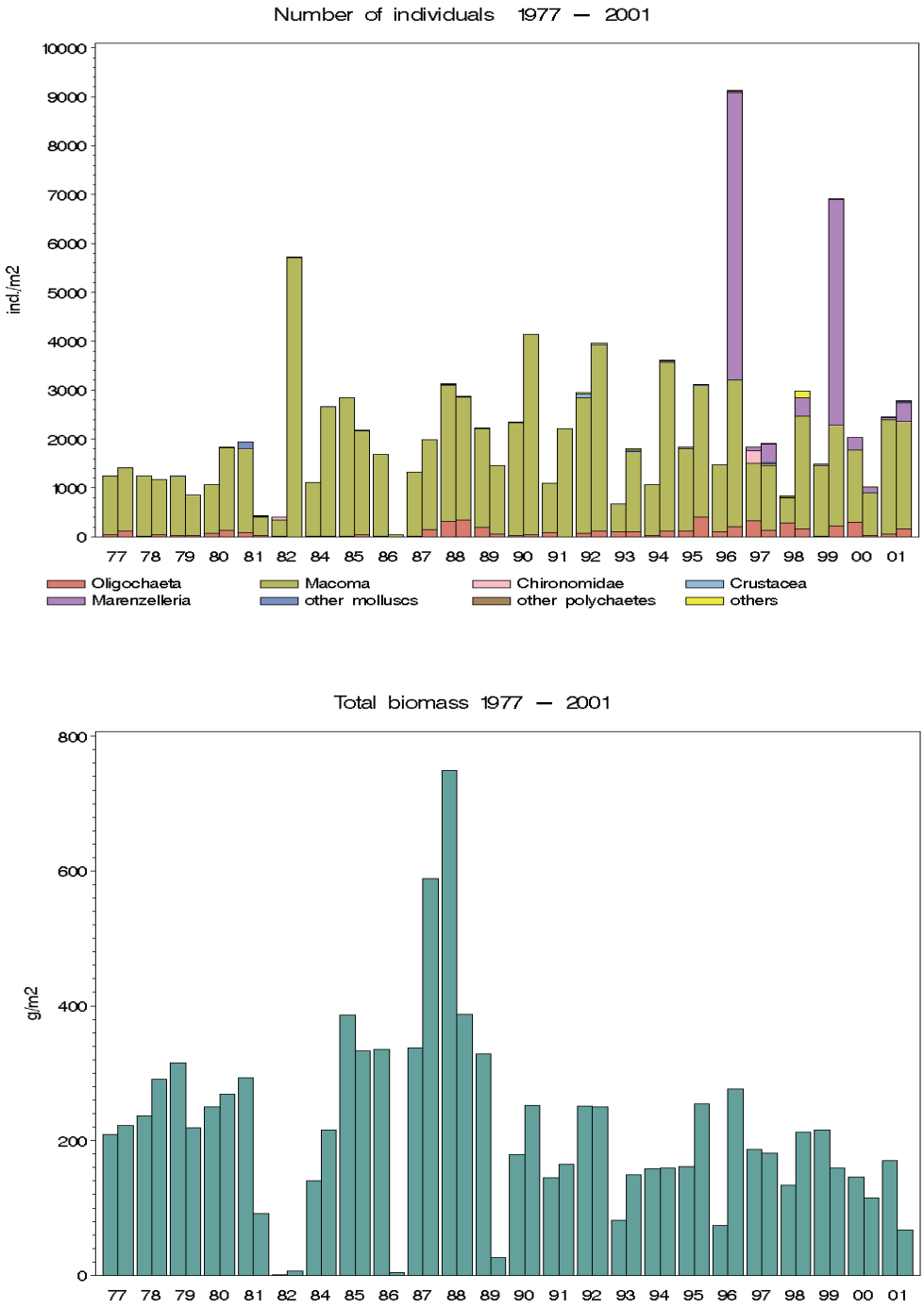


**Fig. 61.** Abundance of main species or taxonomic groups (ind. $m^{-2}$) and total biomass of macrozoobenthos (g $m^{-2}$) at Station Olkiluoto 12 in 1977–2001.

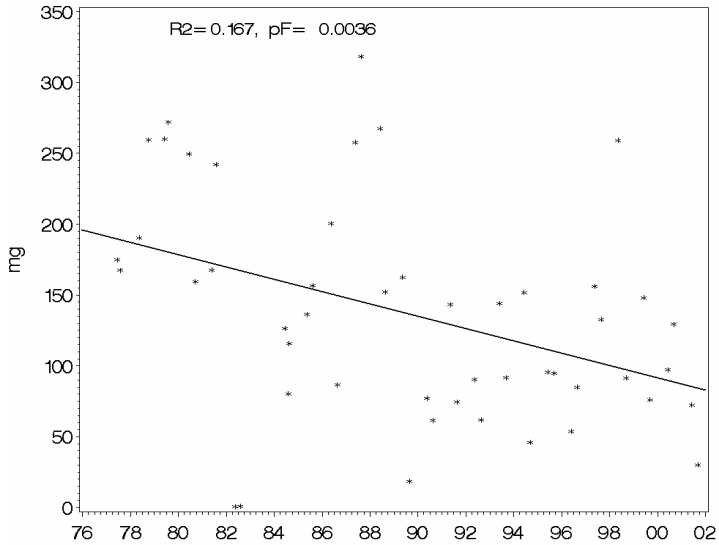


**Fig. 62.** The decrease in the average biomass of *Macoma balthica* at Station Olkiluoto 12 during 1977–2002.

# Part II

# Environmental radioactivity

# 3 Radioecological studies

## 3.1 Monitoring and research programmes

Background studies on radioactive substances in the aquatic environs of the Loviisa and Olkiluoto power plants were already initiated at both sites in good time before the construction works of the power plants were started. Baseline studies for monitoring the radioactivity in the environment and for radioecological studies were initiated at Loviisa in 1966 and at Olkiluoto in 1972. The construction works at Loviisa were started in 1970 and at Olkiluoto in 1973. The nuclear power plant units Loviisa 1 and 2 were commissioned in February 1977 and November 1980, and those at Olkiluoto (TVO I and TVO II) in September 1978 and February 1980.

Permanent programmes for monitoring radionuclides in the surroundings of the power plants were launched in both areas one year before the commissioning of the first power plant unit. From the very beginning, the Research and Environmental Surveillance at STUK and its organizational predecessors have been responsible for carrying out the permanent environmental monitoring programmes of radioactivity in the environs of the Finnish nuclear power plants. This task has been carried out as a charged service to the power plants. The volume of the monitoring programmes has remained at about the same level throughout the operational history of the power plants, though some details have been revised every five years.

Besides the permanent monitoring programmes, STUK has also run numerous radioecological studies in the sea areas off the power plants on its own account. Radiation monitoring in the environs of the power plants requires not only the implementation of permanent monitoring programmes, but also more profound research work focusing on smaller details in different sectors of the surveillance. The aims of the studies have been:

- to enlarge the scope and coverage of the monitoring programmes,
- to develop the monitoring programmes themselves
- to confirm that the conclusions based on the results of the permanent monitoring programmes are correct.

In addition, Research and Environmental Surveillance at STUK has been the Finnish partner in the work of HELCOM/MORS (Monitoring of Radioactive Substances in the Baltic Sea carried out under the auspices of the Marine Environment Protection Commission, HELCOM), which is run as an international co-operation project between all the Baltic Sea countries. The radioecological data colleted at STUK in the Baltic Sea studies and through this co-operation, provide a good reference for the radioecological monitoring and studies in the marine environment at Loviisa and Olkiluoto.

The environmental monitoring programmes of the Finnish nuclear power plants are relatively extensive, including altogether about 1000 samples and analyses per year. The aim is to confirm that the discharges from the power plants are within the permissible release limits, and to monitor the dispersion of these discharges in the environment. Local circumstances and different spreading directions on land and sea have been taken into account in planning the sampling network. The number of marine samples is about 100 per year. Samples are taken from seawater, aquatic indicator organisms (periphyton, seaweeds and benthic animals), fish, suspended particulate material and bottom sediments. The sampling programme is round-the-year, but it concentrates on late spring, summer and early autumn, when nature is free of ice and snow, and the organisms and natural processes are active.

The programmes for monitoring radionuclides in the environs of Finnish NPPs have been described in detail in the Annual Reports of STUK since 1986 and the results of the monitoring since 1971 (STL 1977, STL 1979, STL 1980a, STL 1980b, STL 1982a, STL 1982b, STL 1983, STL 1984, STUK 1987, Ilus et al. 1987b, Sjöblom et al. 1989, Klemola et al. 1991, Ilus et al. 1992b, Ikäheimonen et al. 1995, Klemola et al. 1998, Ilus et al. 2002a, Klemola et al. 2004, Ikäheimonen et al. 2006 and Ilus et al. 2008).

## 3.2 Material and methods

The material of this work consists of the results collected in the permanent monitoring programmes and special radioecological studies carried out since the 1970s in the marine environments off the Loviisa and Olkiluoto power plants. The place-names and locations of the sampling stations, sites and areas used in the permanent monitoring programmes in the sea areas surrounding the power plants are shown in Figs. 2, 45, 63 and 64. STUK has been responsible not only for the laboratory work, but also for designing of the monitoring programmes and for most of the sampling and other field work connected with the monitoring.

The sampling and analysis methods used in the environmental monitoring programmes at Loviisa and Olkiluoto are described in Ilus et al. (2008) and in the earlier Annual Reports of STUK. Detailed descriptions of the sampling, pre-treatment and analysis methods are also available in Finnish in the STUK/TKO Handbooks 3.1.2–3.1.6. In general, the samples taken for the monitoring programmes are analysed gammaspectrometrically. In addition, the seawater samples are analysed for tritium; a part of the seawater, indicator organism, fish and sediment samples are analysed for $^{89}$Sr and $^{90}$Sr, and a part of the indicator organism, sinking matter and sediment samples are analysed for $^{238}$Pu and $^{239,240}$Pu.

The development of a uniform and modern quality system for the whole STUK was started in 1997, and the regulation was completed in 1999. The quality system is based on Total Quality Management (TQM) according to ISO standard 9004 (ISO 2000). A major step in continuous improvement was reached in 1999 when accreditation was awarded by FINAS (the Finnish Accreditation Service). Accredited fields include tests of radiation safety (i.e., gammaspectrometric analyses, radiochemical analyses of tritium, radioactive strontium and transuranic elements in environmental and foodstuff samples) and related environmental sampling. The STUK laboratory is the test facility T167 (EN ISO/IEC 17025) appraised by FINAS.

In order to determine the quality of the sampling and analysis methods, our laboratory has participated in several intercomparison exercises in recent years conducted by various organizations, e.g., the International Atomic Energy Agency and its Marine Environment Laboratory, Nordic Nuclear Safety Research, Helsinki Commission, etc. Our results have always been consistent with the reference values.

The overall uncertainty of the analysis results includes statistical, calibration and analytical uncertainties expressed as relative error (%) at the 1σ confidence level, but not the uncertainty due to sampling.

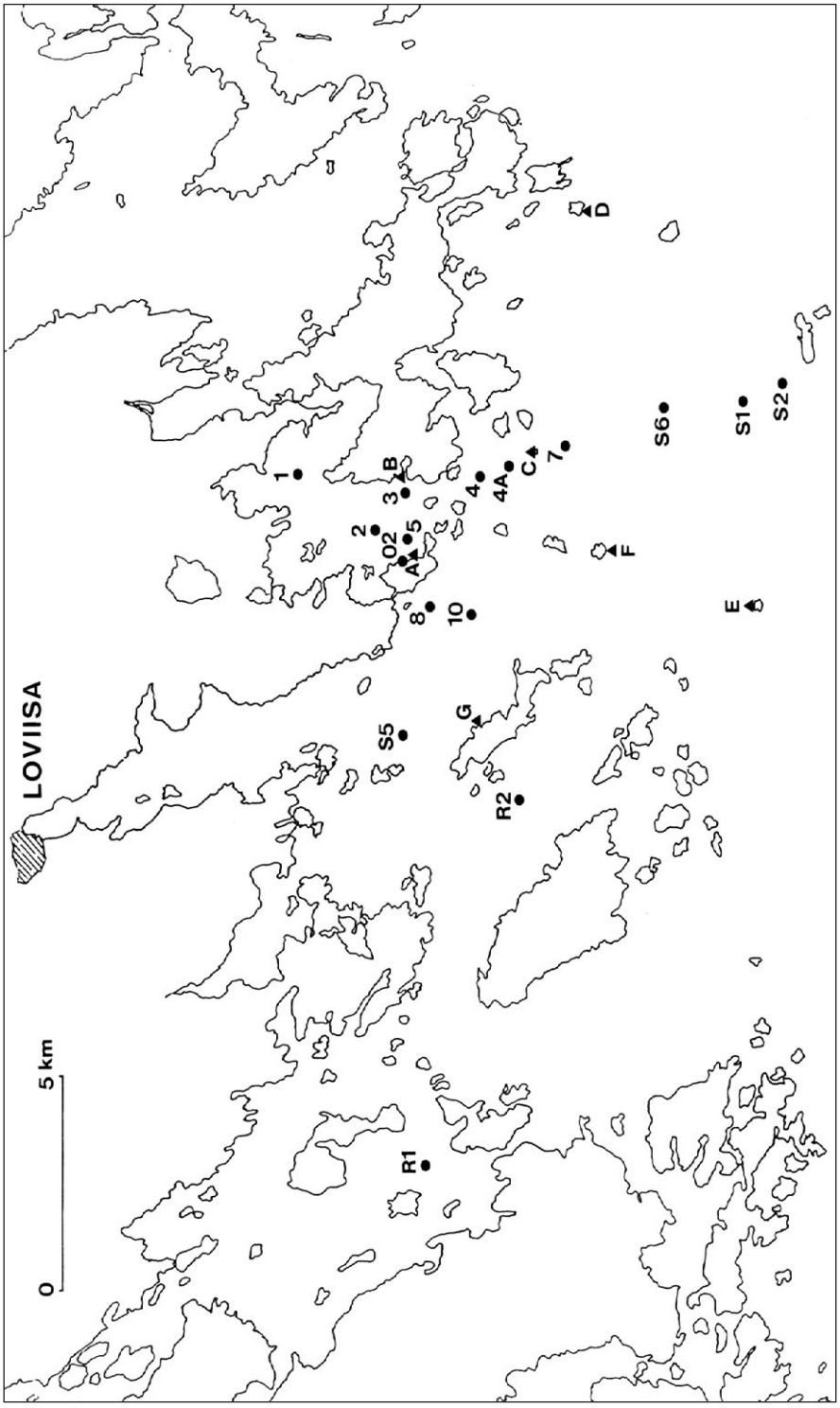


**Fig. 63.** Location of sampling stations and sites in the monitoring of environmental radioactivity at Loviisa (the sampling sites of biota are marked with triangles).

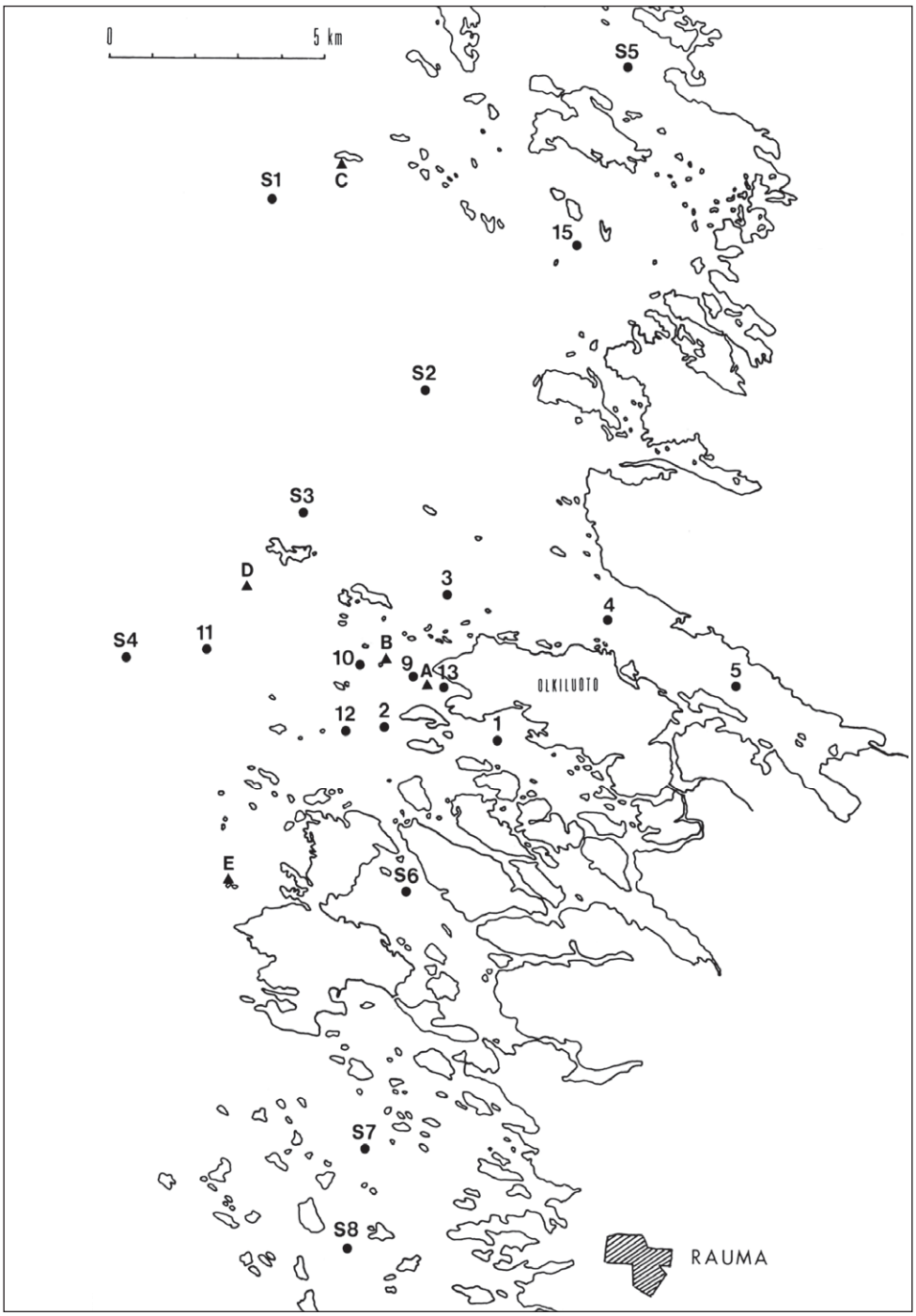


**Fig. 64.** Location of sampling stations and sites in the monitoring of environmental radioactivity at Olkiluoto (the sampling sites of biota are marked with triangles).

## 3.3 Radioactive discharges

During the normal operation of a nuclear power plant, small planned and controlled discharges, mainly of neutron activation products, are released into the recipient sea area together with the out-flowing cooling water. Today liquid and airborne discharges from the nuclear installations situated in the Baltic Sea area are reported annually to the Discharge Register of the Helsinki Commission (HELCOM), for instance. These data are published and considered, e.g., in the periodical assessment reports of HELCOM MORS-PRO and in the annual Indicator Fact Sheets of HELCOM. In international comparison, the discharges from the Finnish nuclear power plants have been small (cf., e.g., Ilus & Ilus 2000, Fig. 14). During the whole operational periods of the Loviisa and Olkiluoto power plants, the annual discharges of tritium have been, at their highest, 10–20% of the release limit, and the total activity of other nuclides released into the sea has been less than 4% (in the last ten years clearly less than 1%) of the set release limit.

The annual liquid discharges of $^{3}$H, $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{110m}$Ag, $^{124}$Sb, $^{134}$Cs and $^{137}$Cs from the Loviisa and Olkiluoto NPPs are shown in Figs. 65–72. A list of radionuclides that appeared regularly, frequently or occasionally in the annual liquid discharges reported from the Loviisa and Olkiluoto power plants in 1977–2006 is given below (in order of mass number):

Loviisa

- Regularly (every year):
  $^{3}$H, $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{110m}$Ag, $^{124}$Sb, $^{134}$Cs, $^{137}$Cs
- Quite frequently (10–14 times in 1977–2006):
  $^{51}$Cr, $^{59}$Fe, $^{90}$Sr, $^{95}$Zr, $^{123m}$Te
- Occasionally (1–8 times in 1977–2006):
  $^{7}$Be, $^{57}$Co, $^{65}$Zn, $^{75}$Se, $^{89}$Sr, $^{95}$Nb, $^{125}$Sb, $^{131}$I, $^{136}$Cs, $^{140}$Ba, $^{141}$Ce.

Olkiluoto

- Regularly (every year):
  $^{3}$H, $^{51}$Cr, $^{54}$Mn, $^{58}$Co, $^{60}$Co,
- Frequently (21–24 times in 1978–2006):
  $^{59}$Fe, $^{95}$Nb, $^{125}$Sb, $^{131}$I, $^{134}$Cs, $^{137}$Cs
- Quite frequently (11–17 times in 1978–2006):
  $^{89}$Sr, $^{95}$Zr, $^{110m}$Ag, $^{124}$Sb
- Occasionally (1–6 times in 1978–2006):
  $^{57}$Co, $^{65}$Zn, $^{85}$Sr, $^{90}$Sr, $^{103}$Ru, $^{113}$Sn, $^{125}$Sn, $^{136}$Cs, $^{140}$Ba, $^{141}$Ce, $^{144}$Ce, $^{181}$Hf, $^{238}$Pu, $^{239}$Pu, $^{241}$Am, $^{242}$Cm, $^{244}$Cm.

In general the discharges are very low. Many of the discharge nuclides are quite short-lived and are therefore not detected in environmental samples. Only those radionuclides with half-lives longer than one week are reported to the Discharge Register.

The Loviisa area received slightly more Chernobyl fallout in 1986 (on average about 23 kBq $m^{-2}$ of $^{137}Cs$) than the Olkiluoto area (on average about 20 kBq $m^{-2}$), both belonging, however, to the same deposition category 3 in the Finnish classification (Arvela et al. 1990). The emission plumes from Chernobyl on 28$^{th}$ and 29$^{th}$ of April 1986 spread over the whole of Finland from the southwest (STUK 1986).

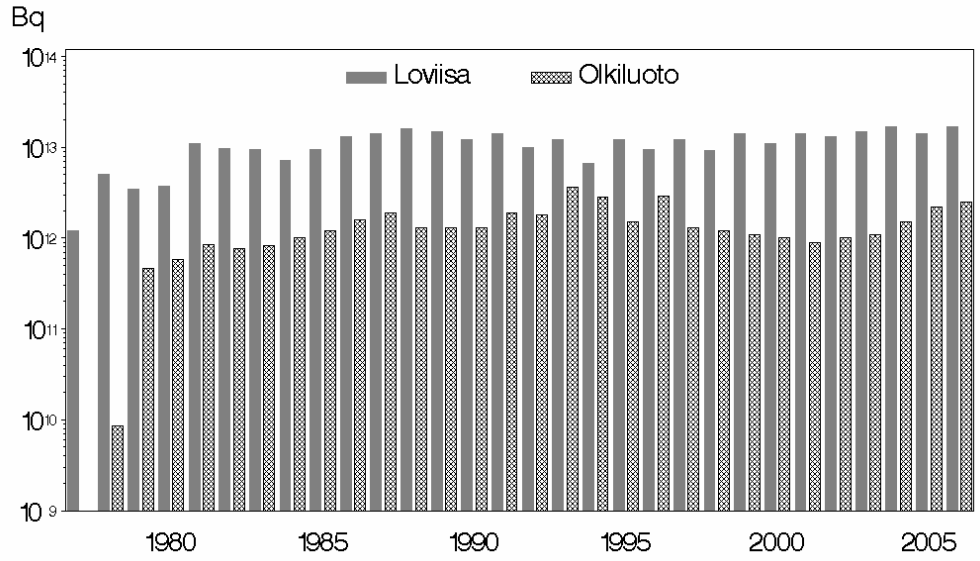


**Fig. 65.** Annual liquid discharges of $^{3}H$ from the Loviisa and Olkiluoto NPPs in 1977–2006.

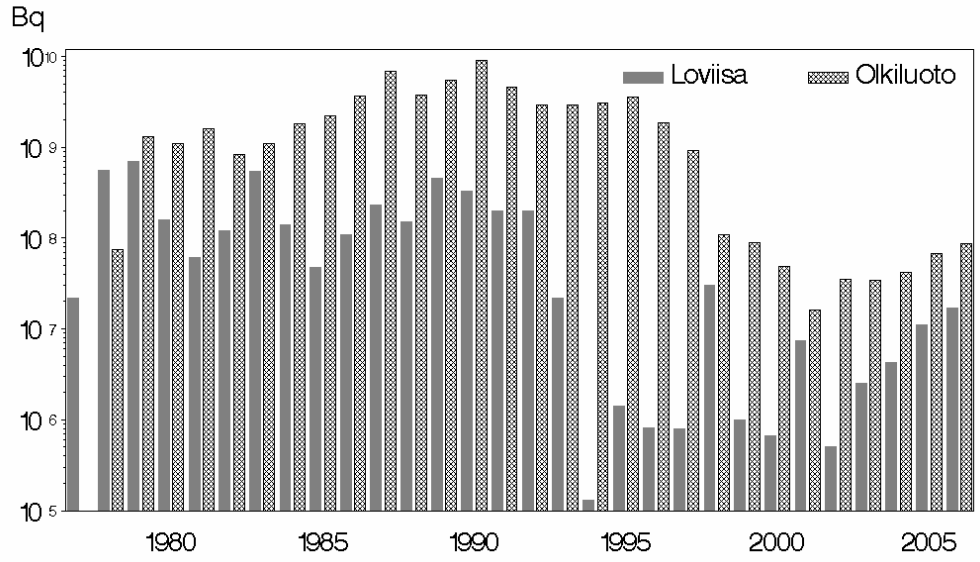


**Fig. 66.** Annual liquid discharges of $^{54}Mn$ from the Loviisa and Olkiluoto NPPs in 1977–2006.


**Fig. 67.** Annual liquid discharges of $^{58}$Co from the Loviisa and Olkiluoto NPPs in 1977–2006.

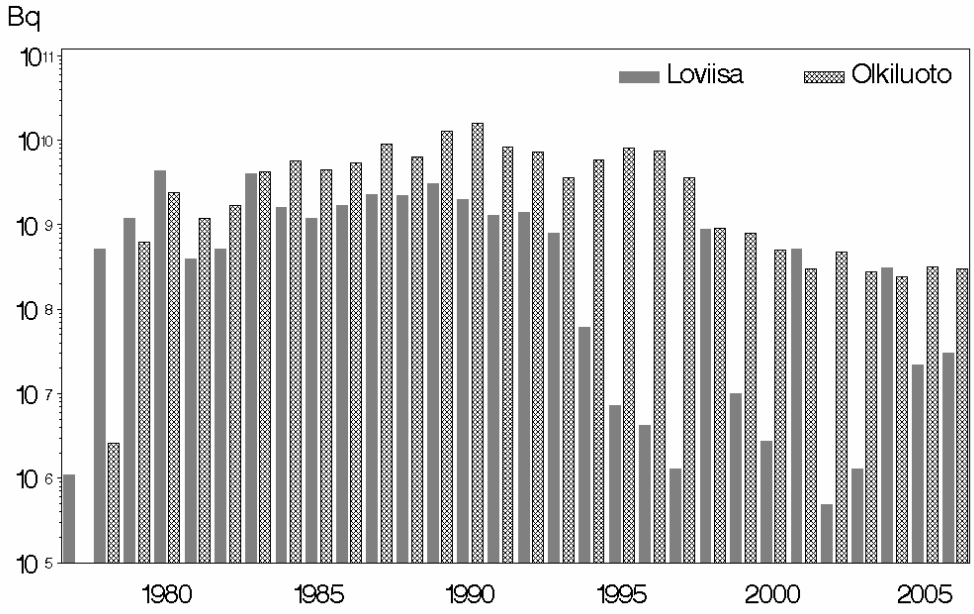


**Fig. 68.** Annual liquid discharges of $^{60}$Co from the Loviisa and Olkiluoto NPPs in 1977–2006.

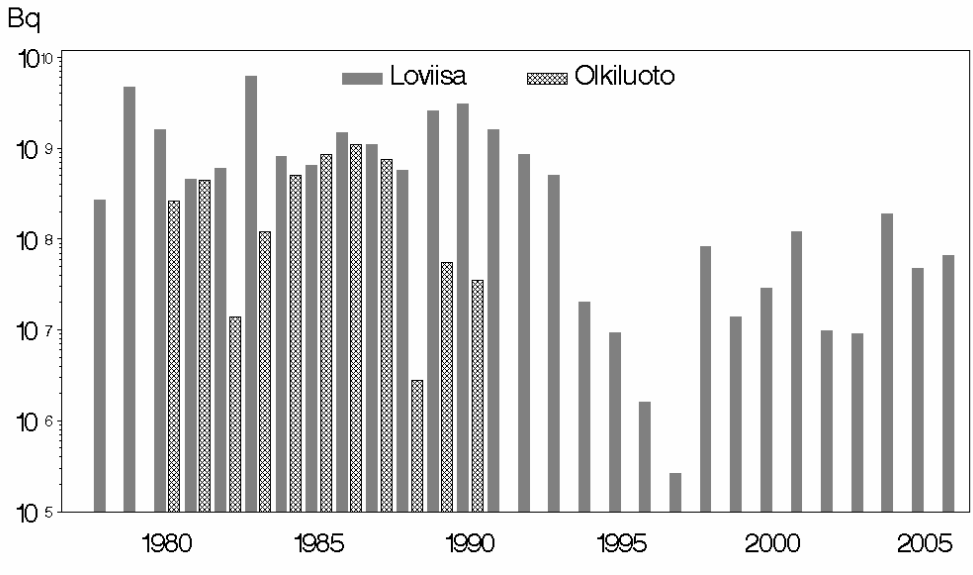


**Fig. 69.** Annual liquid discharges of $^{110m}$Ag from the Loviisa and Olkiluoto NPPs in 1977–2006.

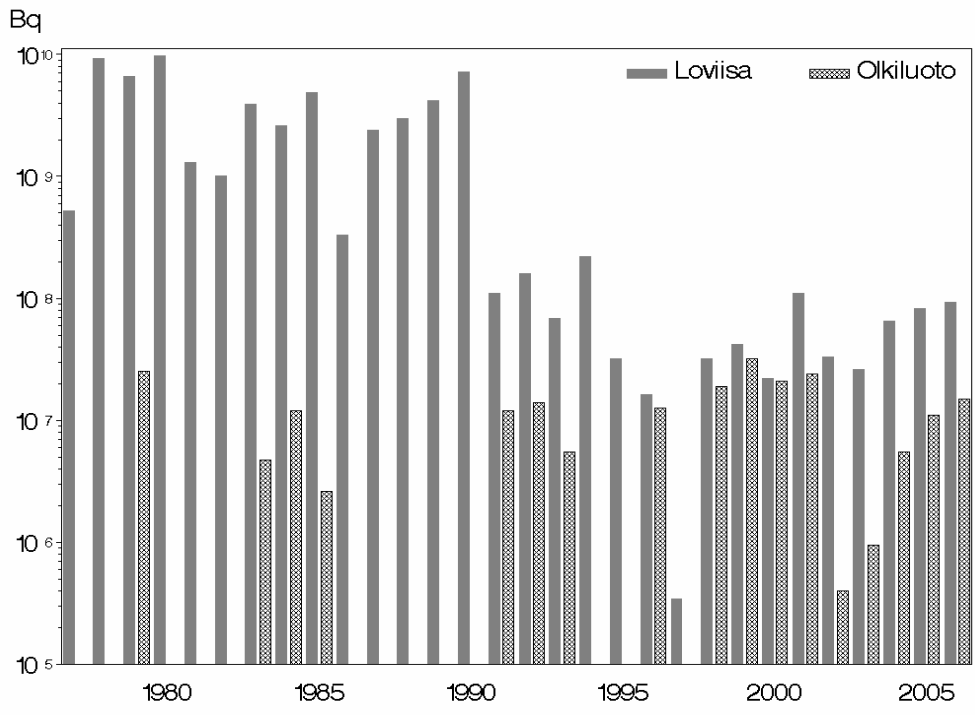


**Fig. 70.** Annual liquid discharges of $^{124}$Sb from the Loviisa and Olkiluoto NPPs in 1977–2006.

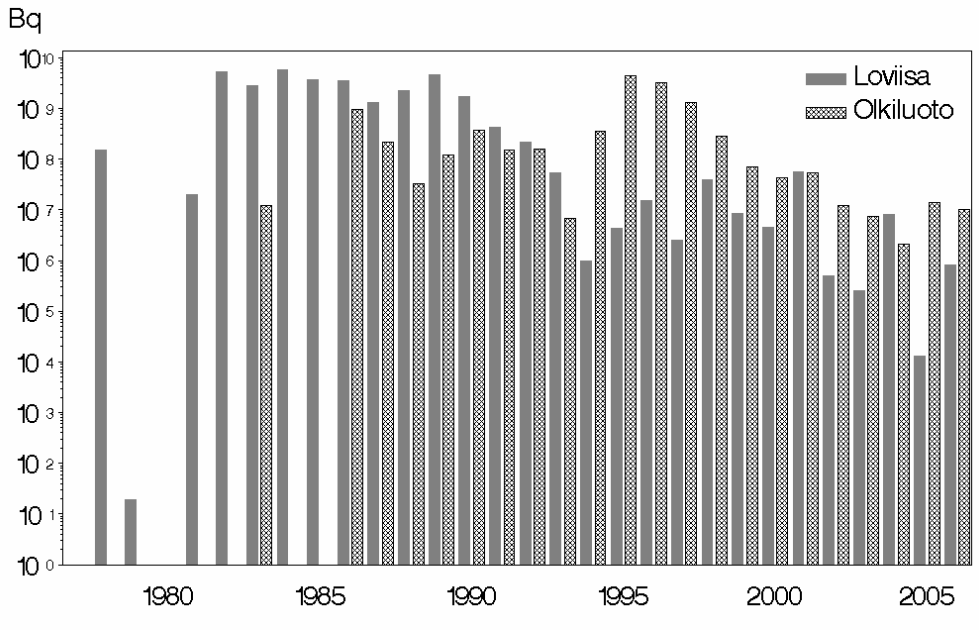


**Fig. 71.** Annual liquid discharges of $^{134}Cs$ from the Loviisa and Olkiluoto NPPs in 1977–2006

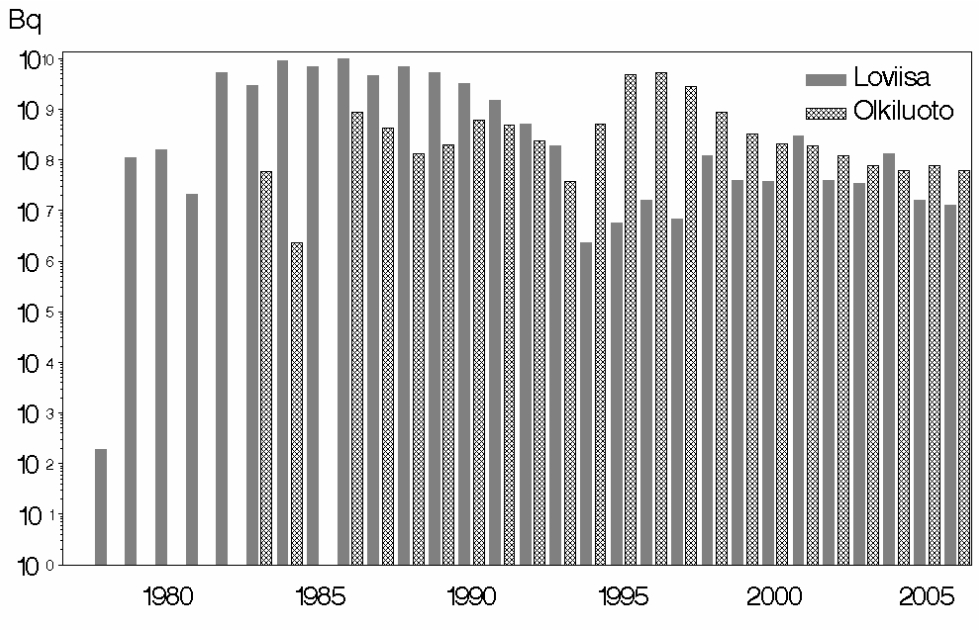


**Fig. 72.** Annual liquid discharges of $^{137}Cs$ from the Loviisa and Olkiluoto NPPs in 1977–2006.

## 3.4 Radioactive substances in seawater

### 3.4.1 Sampling network

In the regular environmental monitoring programme of the Loviisa NPP, activity concentrations of tritium and gamma-emitting radionuclides have been followed in surface seawater at 4–5 sampling stations 3–4 times a year since 1976. One of the sampling stations (Station Loviisa 02) was located right next to the cooling water outlet of the power plant, while Station R1 was a Reference Station located about 14 km west of the plant. The other sampling stations were Loviisa 1, 2 and 4 (Fig. 63). Activity concentrations of $^{90}$Sr were only analysed in the samples taken from the Loviisa 02 and R1 stations.

At Olkiluoto, activity concentrations of tritium and gamma-emitting radionuclides have been followed in surface seawater at 3–5 sampling stations 2–4 times a year since 1977. One of the sampling stations (Station Olkiluoto 13) was located directly in front of the cooling water channel of the power plant (at a distance of 600 m), while Station Olkiluoto 15 was located about 10 km north of the plant. The other sampling stations for seawater were Olkiluoto 2, 3, 10 and 11 (Fig. 64). $^{90}$Sr was analysed in the samples from the Stations Olkiluoto 13 and 15.

### 3.4.2 Tritium in seawater

**Loviisa**

Tritium concentrations in the surface seawater at the Loviisa 1, 2, 4 and R1 stations in 1976–2007 are shown in Figs. 73–76. The curves in the graphs present the radioactive decay of weapons-tests-originated tritium in Finnish coastal waters during the monitoring period, based on an annual average value ($14.7 \pm 1.2$ kBq m$^{-3}$) of $^{3}$H in 1976 and its physical half-life of 12.3 years. Thus, all the results that are clearly above these lines are assumed to be due to the local discharges.

The highest activity concentration of tritium detected in the seawater samples of the regular monitoring programme was 94 kBq m$^{-3}$ in a sample taken from the cooling water outlet (Station 02) in August 2002. At Station 4 in Vådholmsfjärden, the maximum concentration was 41 kBq m$^{-3}$ in May 1999, while at Stations 2 and 1 in Hästholmsfjärden and Klobbfjärden the maximum values were 33 kBq m$^{-3}$ and 24 kBq m$^{-3}$ in May 1981 and 1989, respectively. The temporal fluctuations of tritium concentrations in seawater were prompt and large in the vicinity of the cooling water outlet, depending on the movements of the discharge water plumes in the sea. At Reference Station R1, the concentrations

generally followed the decay rate of fallout tritium. From 1983 onwards, the concentrations started to drop below the detection limit, 7 kBq $m^{-3}$, and since 1998, all of them have been below the new detection limit, 4 kBq $m^{-3}$ (Fig. 76). However, the value from October 1991 (12 kBq $m^{-3}$) clearly exceeds the level of fallout tritium.

Besides the regular monitoring programme, the dispersion of liquid discharges from the Loviisa nuclear power plant was monitored in the recipient sea area in two special studies carried out in October 1978 and December 1979. Separate discharge events were followed by taking samples from different discharge batches (7–9) pumped from a neutralization tank into the out-flowing cooling water and from several sampling stations in the sea area within a tight timetable. Surface seawater samples were taken from 12–14 sampling stations 19 times during a discharge event, which lasted 7–9 days. The shortest sampling interval was 1 hour. The results of the first study (Project Severi) were published by Ilus et al. (1980). The highest tritium concentrations detected in seawater in these studies were 104–120 kBq $m^{-3}$ just in front of the cooling water outlet, but slightly elevated concentrations were momentarily measured at all the stations in the Hästholmsfjärden-Klobbfjärden area.

Fig. 77 illustrates the dispersion pattern of tritium in the surface water of Hästholmsfjärden Bay on the 27$^{th}$ of October 1978, after the release of three discharge batches. Clearly elevated tritium values were measured in the vicinity of the cooling water outlet, but the values rapidly decreased with increasing distance. The background level of fallout $^{3}H$ was estimated to be about 11–14 kBq $m^{-3}$ at that time. Fig. 78 illustrates the dispersion pattern of tritium on the 10$^{th}$ of November 1978, one week after the last discharge batch in the 1978 discharge event. Slightly elevated values were then dispersed over a larger area, and the values were lower close to the cooling water outlet, in a water mass coming from the cooling water intake situated at a depth of 8–11 m on the opposite side of Hästholmen Island.

**Olkiluoto**

Tritium concentrations in the surface seawater at the Olkiluoto 13 and 10 stations in 1977–2007 are shown in Figs. 79–80. The curves in the graphs present the radioactive decay of weapons-tests-originated tritium in Finnish coastal waters during the monitoring period, based on an annual average value ($14.7 \pm 1.2$ kBq $m^{-3}$) of $^{3}H$ in 1976 and its physical half-life of 12.3 years. Thus, all the results that are clearly above these lines are assumed to be due to the local discharges.

The maximum concentration of tritium detected in the seawater samples of the regular monitoring programme was 58 kBq $m^{-3}$ in a sample

taken from the Olkiluoto 13 station in March 2006 (Fig. 79). Station 13 is located in Iso Kaalonpuhti Bay, where the cooling water from the power plant is discharged through a long discharge channel, and consequently, elevated tritium concentrations are recorded there now and then. At the other sampling stations, elevated tritium concentrations were detected less frequently. Clearly elevated tritium concentrations were recorded at the Olkiluoto 3 station in 1983 (17 kBq $m^{-3}$) and at the Olkiluoto 10 station in 1991, 1992 and 2006 (17, 14 and 7 kBq $m^{-3}$), but all the other results were close to the equivalent level of the weapons-tests fallout, or below the detection limit. Station 3 lies about 2 km north and Station 10 about 3 km west of the cooling water outlet. Since 1996, the concentrations of tritium were below the detection limit (4 kBq $m^{-3}$) in all the seawater samples taken from all the other sampling stations except Stations 13 and 10. The temporal fluctuations of the tritium concentrations in seawater at Station 13 were prompt and large, depending on the movements of the discharge water plumes in front of the cooling water channel.

Tritium was not analysed in aquatic organisms or sediments at Loviisa and Olkiluoto.

### 3.4.3 Caesium-137 in seawater

**Loviisa**

Certain amounts of $^{137}Cs$ occurred in the mid-1970s in the seawater of the Gulf of Finland as a consequence of the nuclear weapons tests carried out in the 1950s and 1960s in the northern hemisphere. According to Salo et al. (1986), the estimated average concentration of $^{137}Cs$ in the Gulf of Finland was at its highest about 45 Bq $m^{-3}$ in 1966, decreasing to less than 20 Bq $m^{-3}$ by 1975. In the sea area surrounding the Loviisa NPP, the background level of $^{137}Cs$ in seawater was 12.4 ± 3.4 Bq $m^{-3}$ in 1975–1976 (Stations 1, 2, 4 and 10).

In 1982–1991, liquid discharges of $^{137}Cs$ from the Loviisa power plant varied between 1.5 and 10 GBq per year. In 1994–1997, they were significantly reduced to 2.3–16 MBq per year, and in the 2000s they varied between 13 and 300 MBq per year (Fig. 72).

Besides tritium, the activity concentrations of $^{137}Cs$ and other gamma-emitting nuclides were also monitored in the aquatic environment in the special studies carried out in 1978 and 1979 in connection with the discharge events mentioned above. Elevated $^{137}Cs$ concentrations were only detected in the close vicinity of the cooling water outlet, the concentrations then decreasing to the background level with increasing distance from the outlet (Fig. 81).

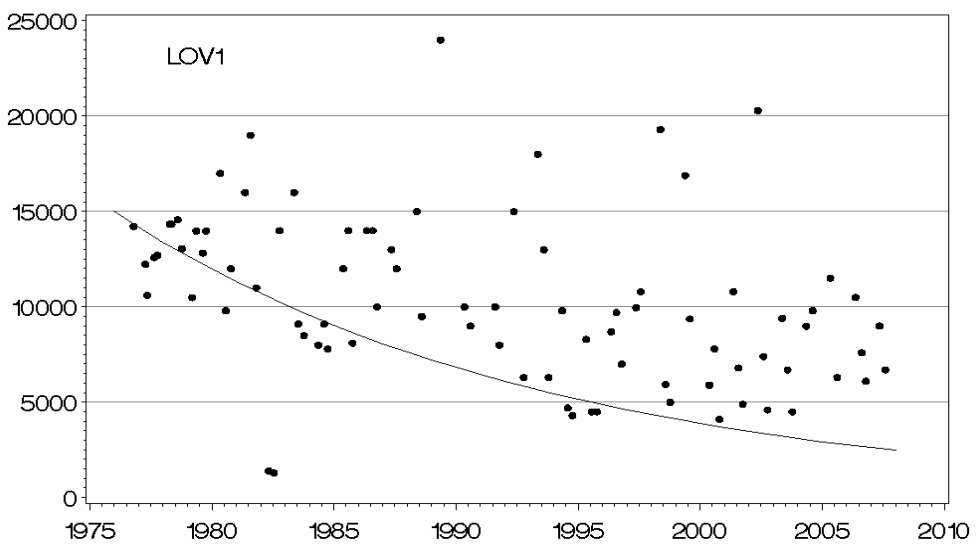


**Fig. 73.** Tritium concentrations in surface seawater (Bq $m^{-3}$) at Station Loviisa 1 in 1976–2007. The curve in the graph indicates the decay of weapons-tests tritium in Finnish coastal waters during the monitoring period.

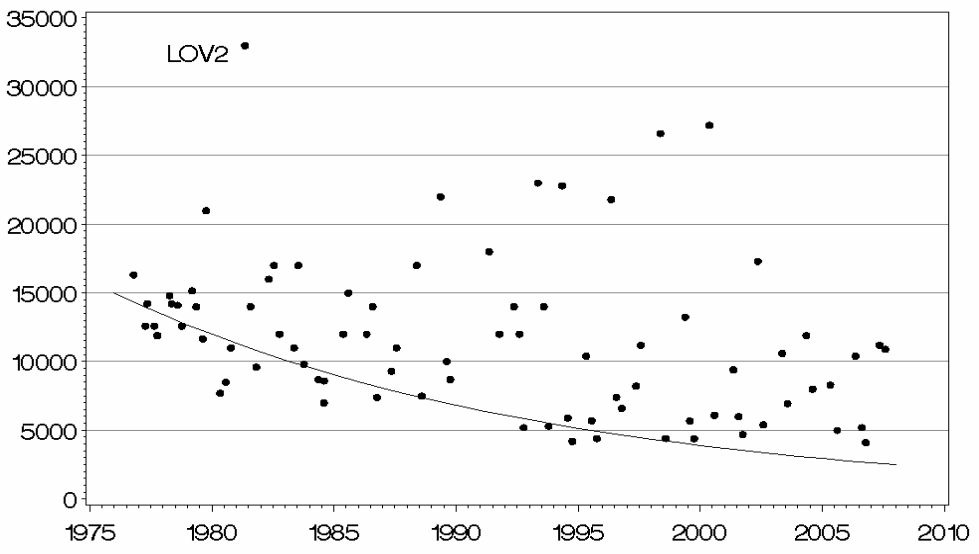


**Fig. 74.** Tritium concentrations in surface seawater (Bq $m^{-3}$) at Station Loviisa 2 in 1976–2007. The curve in the graph indicates the decay of weapons-tests tritium in Finnish coastal waters during the monitoring period.

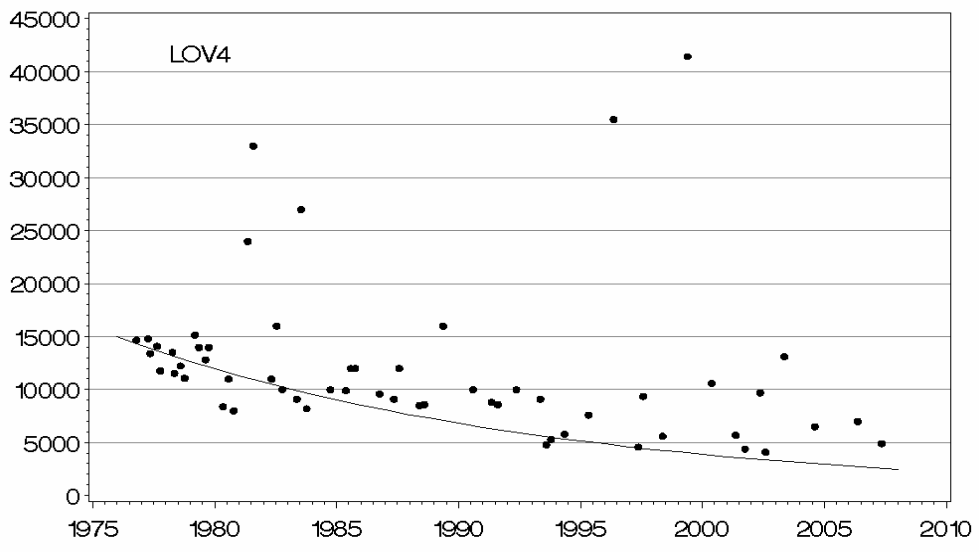


**Fig. 75.** Tritium concentrations in surface seawater (Bq $m^{-3}$) at Station Loviisa 4 in 1976–2007. The curve in the graph indicates the decay of weapons-tests tritium in Finnish coastal waters during the monitoring period.

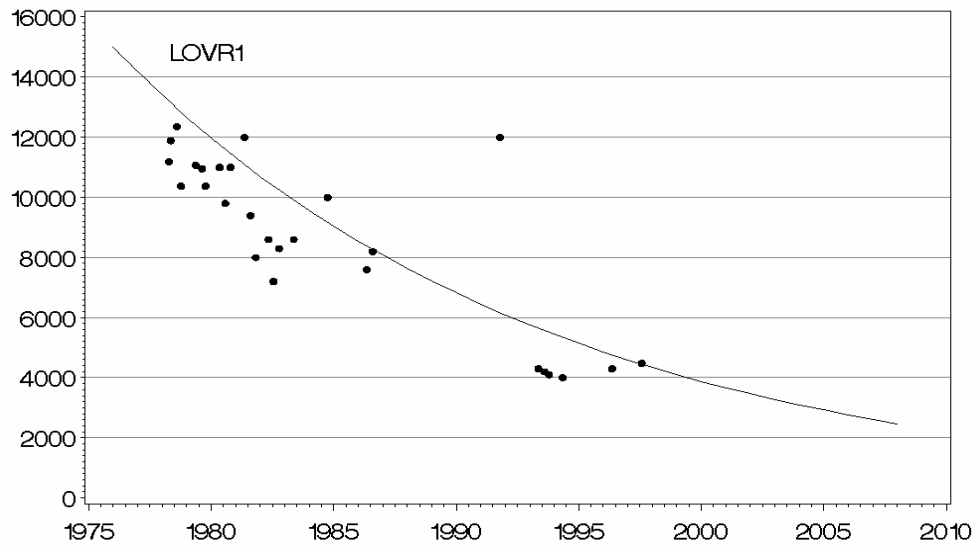


**Fig. 76.** Tritium concentrations in surface seawater (Bq $m^{-3}$) at Station Loviisa R1 in 1976–2007. The curve in the graph indicates the decay of weapons-tests tritium in Finnish coastal waters during the monitoring period.

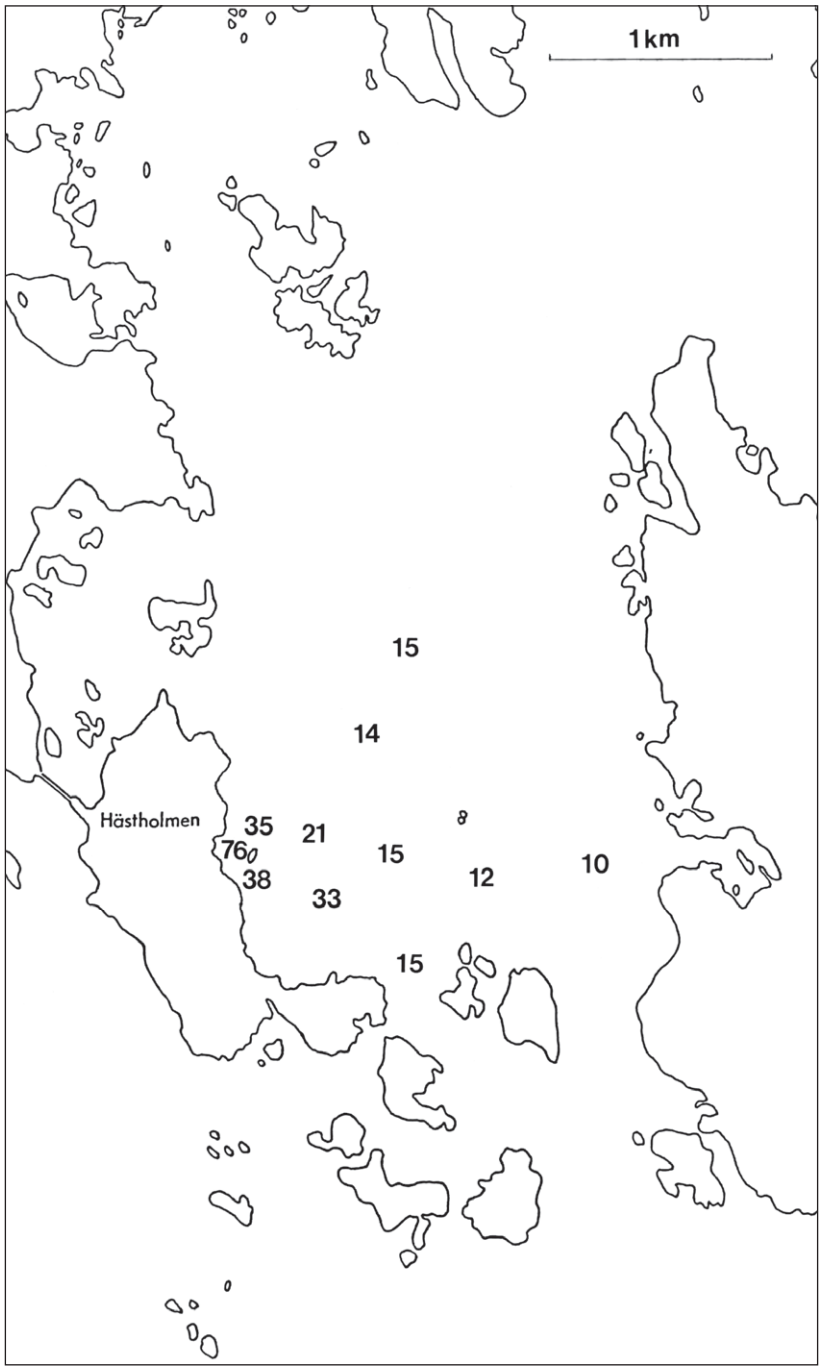


**Fig. 77.** Activity concentrations of tritium (kBq $m^{-3}$) in surface seawater at different distances from the cooling water outlet in Hästholmsfjärden Bay at Loviisa on 27 October 1978.

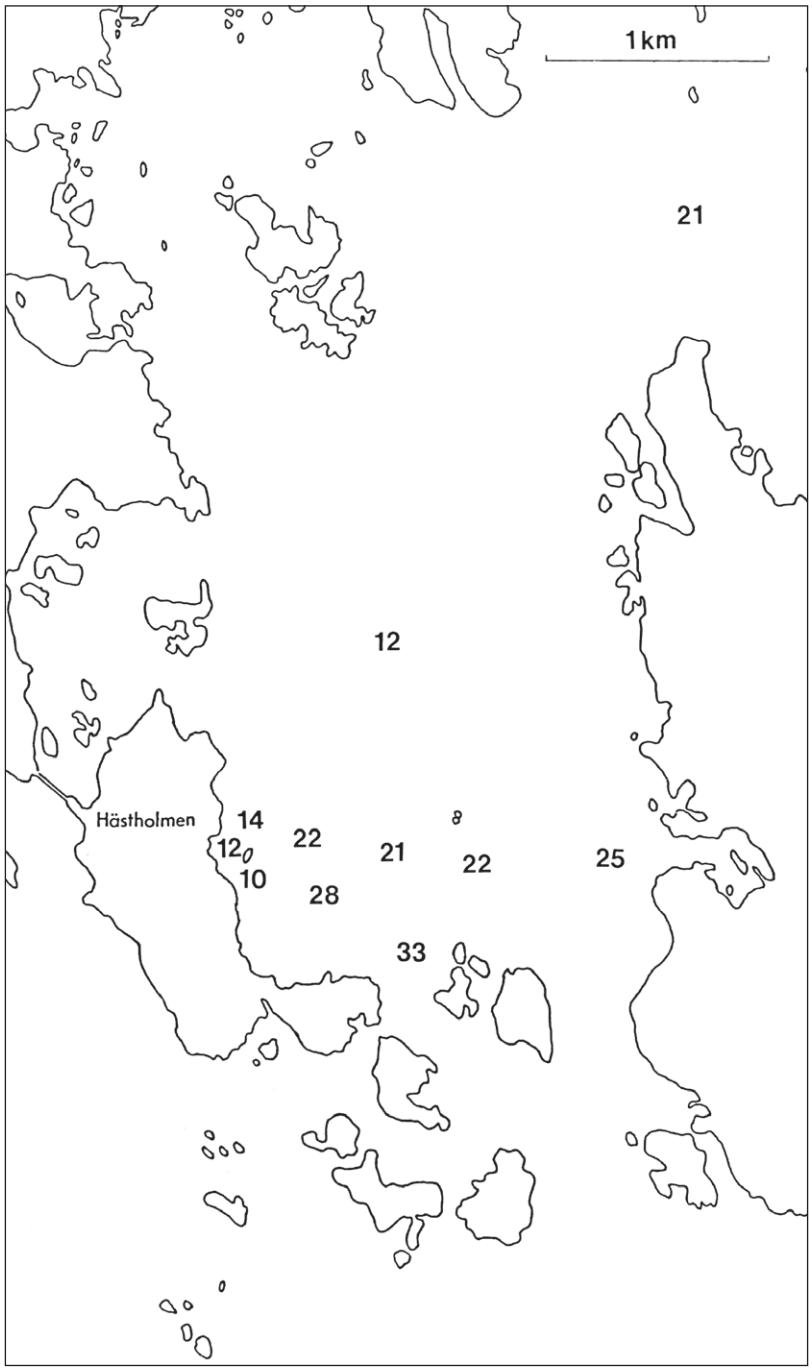


**Fig. 78.** Activity concentrations of tritium (kBq $m^{-3}$) in surface seawater at different distances from the cooling water outlet in Hästholmsfjärden Bay at Loviisa on 10 November 1978.

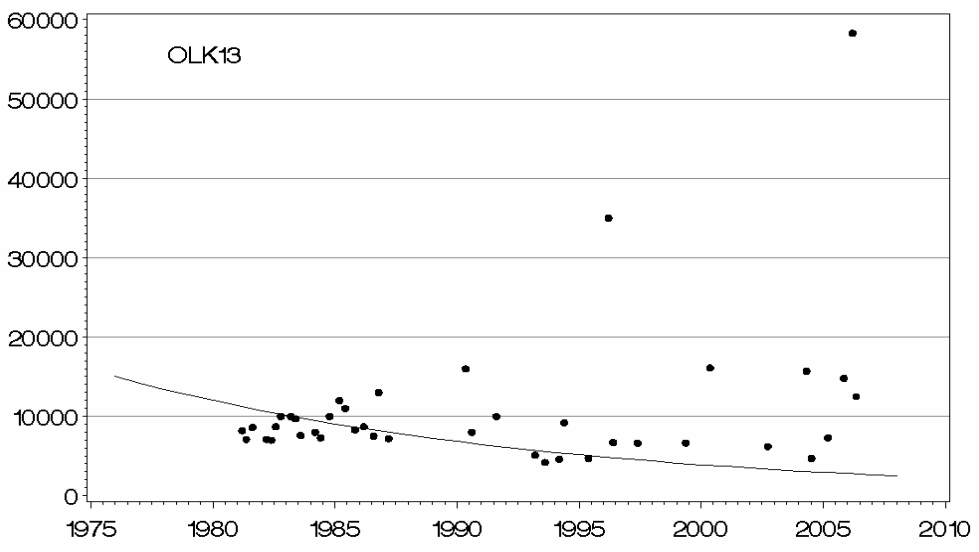


**Fig. 79.** Tritium concentrations in surface seawater (Bq $m^{-3}$) at Station Olkiluoto 13 in 1981–2007. The curve in the graph indicates the decay of weapons-tests tritium in Finnish coastal waters during the monitoring period.

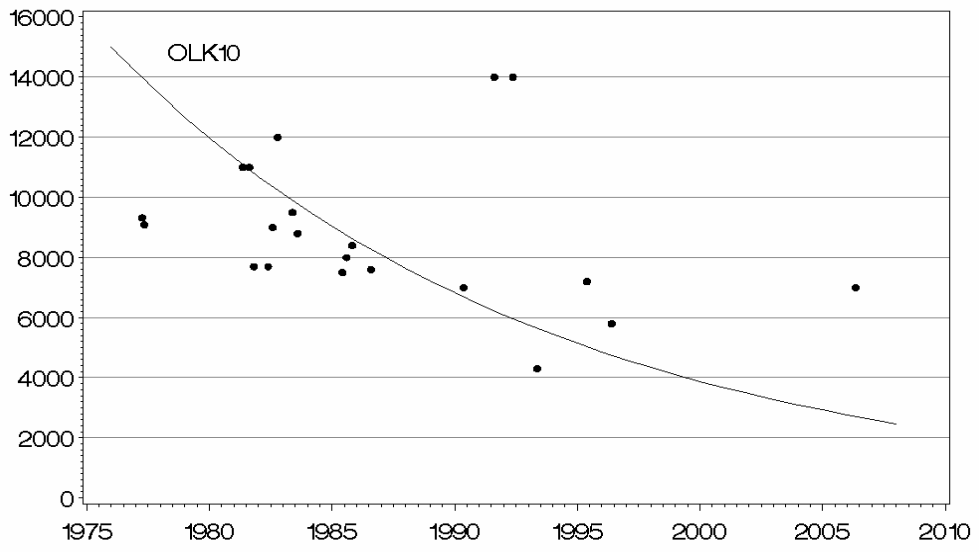


**Fig. 80.** Tritium concentrations in surface seawater (Bq $m^{-3}$) at Station Olkiluoto 10 in 1977–2007. The curve in the graph indicates the decay of weapons-tests tritium in Finnish coastal waters during the monitoring period.

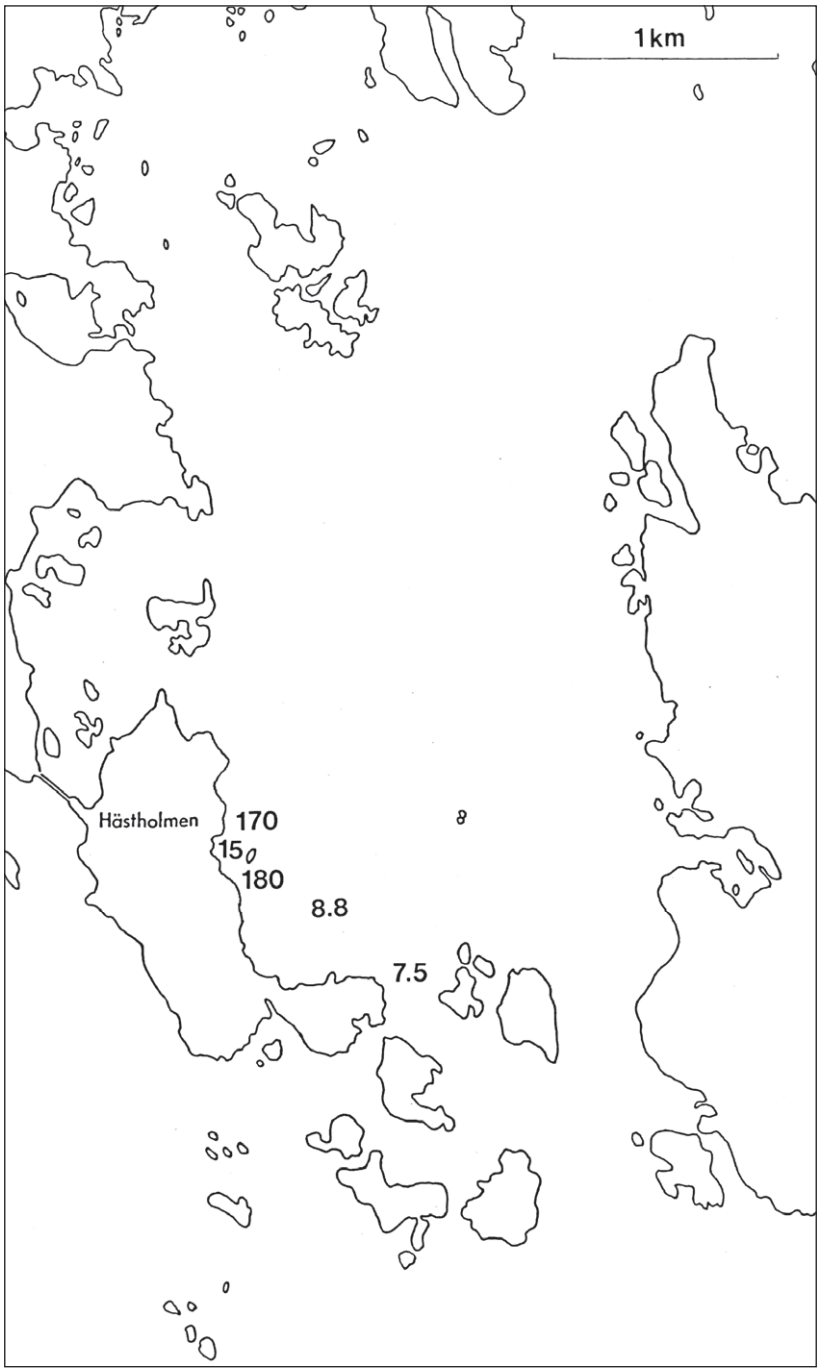


**Fig. 81.** Activity concentrations of $^{137}Cs$ (Bq $m^{-3}$) in surface seawater at different distances from the cooling water outlet in Hästholmsfjärden Bay at Loviisa on 17 December 1979.

Up to 1986, the levels of $^{137}$Cs had decreased to about 7–8 Bq m$^{-3}$ in the sea area surrounding the Loviisa power plant. In 1984–1985, the average concentrations in the surface water were 7.2–7.3 Bq m$^{-3}$ at Stations 02, 2 and 10 in Hästholmsfjärden and Hudöfjärden, whereas they were 8.3 ± 1.8 and 6.7 ± 1.7 Bq m$^{-3}$ at Stations 1 and 4 in Klobbfjärden and Vådholmsfjärden, respectively.

The fallout from the accident at the Chernobyl NPP on the 26$^{th}$ of April 1986 caused a sudden and drastic rise in the $^{137}$Cs concentrations in surface waters for a short while. The highest $^{137}$Cs concentration observed in seawater was 5 200 Bq m$^{-3}$ at Station 10 in Hudöfjärden on the 5$^{th}$ of May, and in general, the concentrations varied between 3 000 and 5 200 Bq m$^{-3}$ in the spring of 1986. The high caesium contents occurred simultaneously with low potassium contents: seawater was mixed with large quantities of rain and melt waters flowing into the area from the catchments (Ilus et al. 1987b). By August, when the deposited radionuclides were mixed with deeper water masses, the $^{137}$Cs concentrations had already fallen to 460–500 Bq m$^{-3}$, and in May 1988 (two years after the fallout) the concentrations were 190–220 Bq m$^{-3}$ at all the Loviisa sampling stations (Fig. 82).

After the Chernobyl fallout, the $^{137}$Cs concentrations in seawater have decreased steadily and relatively rapidly in the Loviisa archipelago. In 2000–2001, the average concentrations were 46–47 Bq m$^{-3}$ at Stations 1 and 2 in Klobbfjärden and Hästholmsfjärden, and 43 Bq m$^{-3}$ at Station 4 in Vådholmsfjärden and at Reference Station R1 situated 14 km west of the power plant. At Station Loviisa 02 located just in front of the cooling water outlet, the average concentration of $^{137}$Cs was 45 Bq m$^{-3}$. The effective ecological half-life of $^{137}$Cs in seawater at Loviisa was estimated at 8.6 ± 0.2 years between 1989 and 2007. During the first years after the Chernobyl accident the half-life was much shorter (Fig. 82). The half-life calculated for the period 1986–2007 was 5.7 ± 0.1 years. In the open Gulf of Finland, the effective half-life of $^{137}$Cs was estimated at 0.8 years in 1986–1988 and at 13 years in 1993–2006 (Herrmann et al. 2009).

**Olkiluoto**

The annual liquid discharges of $^{137}$Cs from the Olkiluoto power plant were at their highest 2.8–5.3 GBq in 1995–1997. Before that they varied considerably between 2.3 and 880 MBq per year, and in the 2000s they have been between 62 and 210 MBq per year (Fig. 72).

In 1977, the background level of $^{137}$Cs in the seawater in the sea area off Olkiluoto (Stations 2, 3 and 11) was 17.7 ± 3.0 Bq m$^{-3}$. In the period until 1986, the levels had decreased to about 9–11 Bq m$^{-3}$. In 1984–1985, the average

concentration in the surface water was 9.4 ± 1.6 Bq $m^{-3}$ at Station 13 in Iso Kaalonpuhti Bay and 10.9 ± 2.9 at Station 3, which is affected to some extent by river waters from the Eurajoki River.

The fallout from the Chernobyl accident suddenly and drastically raised the $^{137}$Cs concentrations in the surface water also in the Olkiluoto area. On the 2nd of June 1986, the $^{137}$Cs concentrations were 1 000 – 1 100 Bq $m^{-3}$ at all the sampling stations (Ilus et al. 1987b). On the opposite side of the Bothnian Sea, at Forsmark, the detected concentrations of $^{137}$Cs were 1 500 Bq $m^{-3}$ on the 3rd of June and 3 710 Bq $m^{-3}$ on the 15th of June (Grimås et al. 1986). By August 1986, the $^{137}$Cs concentrations had decreased to 280 – 370 Bq $m^{-3}$, and in May 1988 (two years after the fallout) the concentrations were 250 – 260 Bq $m^{-3}$ at all the sampling stations at Olkiluoto. After that, the $^{137}$Cs concentrations in seawater have decreased steadily (Fig. 83). In 2000 – 2001, the average concentrations were about 70 Bq $m^{-3}$ at the Olkiluoto 10 and 13 stations and about 71 Bq $m^{-3}$ at the Olkiluoto 2 and 15 stations.

The effective ecological half-life of $^{137}$Cs in seawater at Olkiluoto was estimated at 8.1 ± 0.4 years between 1989 and 2007. During the first years after the Chernobyl accident the decrease was faster but much slower than at Loviisa (*cf*. Figs. 82 and 83). The half-life calculated for the period 1986 – 2007 was 7.0 ± 0.2 years. In the open Bothnian Sea, the effective half-life of $^{137}$Cs was estimated at 2.5 years in 1986 – 1988 and at 9 years in 1993 – 2006 (Herrmann et al. 2009).

### 3.4.4 Other artificial radionuclides in seawater

**Loviisa**

In addition to $^{40}$K (naturally-occurring), $^{137}$Cs, $^{134}$Cs (from Chernobyl fallout), $^{90}$Sr (mainly from global fallout) and tritium, observations of other artificial radionuclides have been relatively few in the seawater samples of the regular environmental monitoring programme. The detected gamma-emitting radionuclides were mainly fission or activation products of local origin, such as $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{110m}$Ag, $^{124}$Sb and $^{125}$Sb, and were mainly detected in the seawater samples taken just in front the cooling water outlet.

Predominantly, local discharge nuclides appeared in seawater samples during the first operational years of the power plant. Small amounts (< 60 Bq $m^{-3}$) of e.g. $^{95}$Nb, $^{95}$Zr, $^{124}$Sb and $^{125}$Sb were then detected even more widely, i.e., at the sampling stations Loviisa 1, 2, 4 and 10 (Koivulehto et al. 1979, 1980). In the special discharge-monitoring-survey in December 1979 (see above), relatively high concentrations of $^{124}$Sb (maximum 5 500 Bq $m^{-3}$), $^{60}$Co (maximum 1 000 Bq $m^{-3}$)

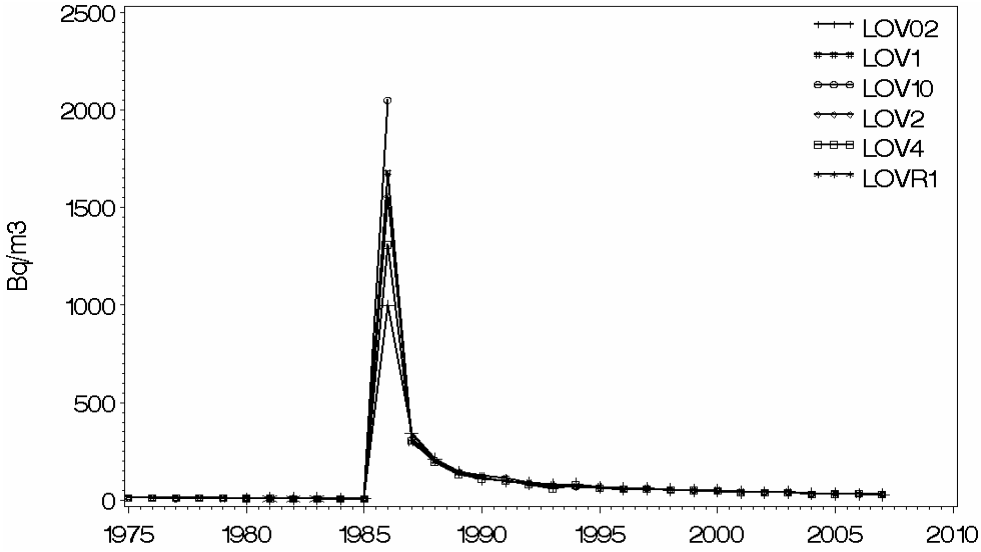


**Fig. 82.** Annual mean concentrations of $^{137}Cs$ in surface seawater (Bq $m^{-3}$) at the sampling stations at Loviisa in 1975–2007.

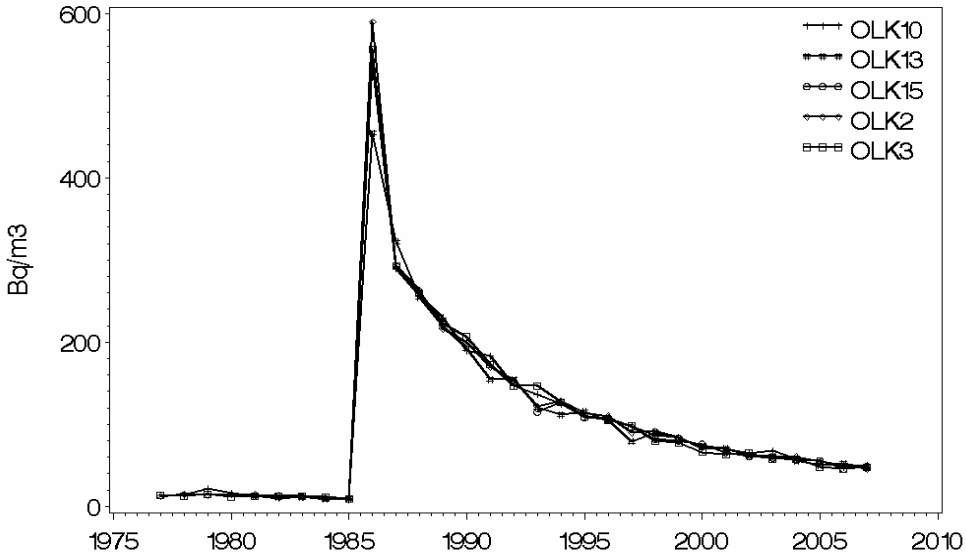


**Fig. 83.** Annual mean concentrations of $^{137}Cs$ in surface seawater (Bq $m^{-3}$) at the sampling stations at Olkiluoto in 1977–2007.

and $^{125}$Sb (maximum 670 Bq m$^{-3}$) were also detected in samples taken from the immediate vicinity (< 300 m) of the cooling water outlet, but almost without exception, not at the more distant stations. On the 28$^{th}$ of December 1979, four days after the last discharge batch, small amounts of $^{60}$Co (5.2 Bq m$^{-3}$) and $^{124}$Sb (35 Bq m$^{-3}$) were detected in a seawater sample taken from the strait leading to the south from Hästholmsfjärden Bay (distance 1 km from the outlet).

Since 1980, local discharge nuclides were detected only at the cooling water outlet (Station 02), except for three separate observations of $^{125}$Sb at Stations 1 and 4 in Klobbfjärden and Vådholmsfjärden in 1988 (Klemola et al. 1991). Small amounts of local discharge nuclides were detected for the last time in seawater samples in March 1993. The activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co and $^{110m}$Ag were then 19.5, 1.6, 5.6 and 3.9 Bq m$^{-3}$, respectively, at Station Loviisa 02 (Klemola et al. 1998). Since then, local discharge nuclides were not detected in the seawater samples of the regular monitoring programme.

The average activity concentrations of $^{90}$Sr in the surface water at the cooling water outlet (Station 02) and at Reference Station R1 during the operational period of the power plant are given in Table 13. The year 1986 was passed over in the average calculations, because of the momentarily higher values, which were certainly caused by the Chernobyl fallout. On the 9$^{th}$ of May 1986, the activity concentration of $^{90}$Sr was 91 Bq m$^{-2}$ at Reference Station R1 and 36 Bq m$^{-3}$ at Station 02 (Ilus et al. 1987). In August, the values had already decreased to 25 and 27 Bq m$^{-3}$, respectively.

The concentrations decreased in accordance with the half-life of $^{90}$Sr, and the results of the two stations did not significantly differ from each other. The slow decrease of strontium is explained by its slow transfer into sediments, while it remains in the water phase. The effective ecological half-life of $^{90}$Sr in seawater was 16.4 ± 0.1 years between 1989 and 2007, i.e., about half of the physical half-life. The difference is due to several ecological factors, such as sedimentation of strontium (slower than that of caesium), mixing of water masses, removal from coastal areas by sea currents, etc.

**Olkiluoto**

At Olkiluoto, local discharge nuclides were detected in regular seawater samples more often and more widely, but in smaller quantities than at Loviisa. All in all, $^{60}$Co was detected 35 times, $^{54}$Mn 15 times, $^{125}$Sb 9 times, $^{58}$Co 4 times and $^{110m}$Ag once in the water samples taken from the Olkiluoto 2, 3, 10, 13 and 15 stations in 1981–1998. The highest activity concentration was 32 Bq m$^{-3}$ of $^{60}$Co at the Station 13 in Iso Kaalonpuhti Bay in May 1991. Once (in August 1993), $^{60}$Co was detected in the seawater at Station 15,

**Table 13.** Average activity concentrations of $^{90}Sr$ (Bq $m^{-3}$) in the surface water of the Loviisa 02, 2 and R1 stations during 1973–2007.

| Period | Loviisa 02 | Loviisa 2 | Loviisa R1 |
|---|---|---|---|
| 1973–1976 | | 30.5 ± 2.4* | |
| 1980–1985 | 21.4 ± 3.2 | | 20.6 ± 3.4 |
| 1987–1990 | 20.3 ± 1.7 | | 20.4 ± 2.0 |
| 1991–1995 | 16.5 ± 1.6 | | 16.1 ± 2.3 |
| 1996–2000 | 13.2 ± 1.3 | | 13.1 ± 1.2 |
| 2001–2007 | 10.8 ± 1.8 | | 11.3 ± 2.5 |

* = Koivulehto et al., 1979

situated 10 km north of the power plant (5 Bq $m^{-3}$). Since 1998, local discharge nuclides were not detected in the seawater samples of the regular monitoring programme.

The activity concentrations of $^{90}Sr$ in the seawater were slightly lower than at Loviisa during the whole monitoring period. In 1977, the background level was 28 Bq $m^{-3}$ (Koivulehto et al. 1979). The Chernobyl fallout in 1986 raised the values only slightly (maximum 29 Bq $m^{-3}$), but on the other hand, the values returned to their former level only after five years. In 2001–2007, the average activity concentration of $^{90}Sr$ was 11.9 ± 2.5 Bq $m^{-3}$ at Station 13 in Iso Kaalonpuhti Bay and 11.4 ± 2.6 at Station 15 situated 10 km north of the power plant. At Olkiluoto, the effective ecological half- life of $^{90}Sr$ in seawater was about 20.6 years between 1989 and 2007.

**In conclusion**

Tritium ($^{3}H$) is ubiquitous in the aquatic environment and has a variety of sources (Jacobs 1968). It is a cosmogenic radionuclide with a half-life of 12.43 years, and is continuously produced in the upper atmosphere as a result of cosmic-ray-induced spallation and particle interaction with atmospheric nitrogen and oxygen (McCubbin et al. 2001). These processes produce most of the world's natural tritium (IAEA 2004a). The levels of tritium rose as a legacy of the atmospheric nuclear weapons tests between 1952 and 1962 that resulted in the injection of about 240 EBq into the earth's atmosphere (UNSCEAR 1993). During recent decades, the levels of fallout $^{3}H$ have continuously decreased, but plenty of tritium remains in the environment, mostly diluted in the oceans (IAEA 2004a).

Nevertheless, $^{3}H$ is also produced by a variety of processes in nuclear power plants, and consequently, it is the predominant radionuclide both in airborne and liquid discharges from the nuclear power plants, not only in Finland but also

elsewhere. Discharges from nuclear power plants into the aquatic environment result in locally enhanced water concentrations of tritium (McCubbin et al. 2001).

Tritium is produced in reactors by neutron activation of $^{2}H$, $^{3}He$, $^{6}Li$ and $^{10}B$ (IAEA 2004a). The formation of tritium by the activation reactions in pressurised water reactors (Loviisa) is considered to be mainly from the boron in the coolant water, which is used for reactivity control. In boiling water reactors (Olkiluoto) the contribution of tritium by activation reactions is mainly from the boron in the control rods. Tritium activities discharged from boiling water reactors into the environment are lower than those of pressurised water reactors, because less tritium is produced in or diffuses into the primary coolant (IAEA 2004a). Tritium is produced in light-water reactors in quantities that are relatively copious compared to other radionuclides. Fortunately, tritium, which emits beta particles of very low energy, ordinarily enters the environment in the form of water. It does not concentrate significantly in biological systems, and has a relatively rapid turnover rate (Eisenbud and Gesell 1997). Tritium emits low-energy β-particles and therefore does not pose an external radiological hazard (Phillips and Easterly 1981).

Annual discharges of tritium from the Loviisa NPP into the aquatic environment have varied between $1.2 \cdot 10^{12}$ Bq (1977) and $1.7 \cdot 10^{13}$ Bq (2004 and 2006) during the operational period of the power plant, composing 1.2–11.3% of the annual release limit ($1.5 \cdot 10^{14}$ Bq) set for the liquid effluents of tritium from this power plant. The total discharge of tritium from the Loviisa NPP into the aquatic environment since 1977 was $3.30 \cdot 10^{14}$ Bq, and the decay-corrected cumulative discharge until 2007 was $1.82 \cdot 10^{14}$ Bq.

Annual discharges of tritium from the Olkiluoto NPP into the aquatic environment have varied between $8.6 \cdot 10^{9}$ Bq (1978) and $3.6 \cdot 10^{12}$ Bq (1993) during the operational period of the power plant, composing 0.05–20% of the annual release limit ($1.8 \cdot 10^{13}$) set for the liquid effluents of tritium from this power plant. The total discharge of tritium from the Olkiluoto NPP into the aquatic environment since 1978 was $4.14 \cdot 10^{13}$ Bq, and the decay corrected cumulative discharge until 2007 was $2.28 \cdot 10^{13}$ Bq.

In Finnish coastal waters, the concentrations of $^{3}H$ from natural sources and nuclear weapons fallout decreased from about 10–15 kBq $m^{-3}$ to less than 4 kBq $m^{-3}$ between the late 1970s and the early 2000s (Ilus et al. 2008). The detection limit of our analyses of tritium in seawater was 7 kBq $m^{-3}$ until 1992, and 4 kBq $m^{-3}$ since 1993. Elevated $^{3}H$ concentrations were more frequent at Loviisa (Figs. 73–80), which was due to the larger discharges and to the slower exchange of water in the Loviisa discharge area.

In 2007, the mean activity concentration of $^{137}Cs$ in seawater was 28.3 ± 2.8 (range 23–32 Bq $m^{-3}$) at Loviisa and 43.7 ± 5.5 (33–50 Bq $m^{-3}$) at Olkiluoto. Since

the Chernobyl accident (1986), the caesium concentrations have decreased more rapidly at Loviisa, and in the whole Gulf of Finland, than at Olkiluoto and in the Bothnian Sea. In Figure 84, Station LL3a represents the open sea off Loviisa and Station EB1 that off Olkiluoto. Between 1986 and 2007, the decrease in $^{137}$Cs values was about 94% at Loviisa and 87% at Olkiluoto (including the physical decay of $^{137}$Cs). The main reason for the differently decreasing rates of caesium was the more effective water exchange between the Gulf of Finland and the Baltic Proper than that between the Bothnian Sea and the Baltic Proper. On the other hand, in the Loviisa area, the archipelago also retards the exchange of water more effectively than at Olkiluoto, and consequently, the exit rate of caesium from seawater in the discharge area. The activity concentrations of $^{137}$Cs in the vicinities of the Loviisa and Olkiluoto power plants did not differ from those in the open Gulf of Finland and Bothnian Sea (Fig. 84).

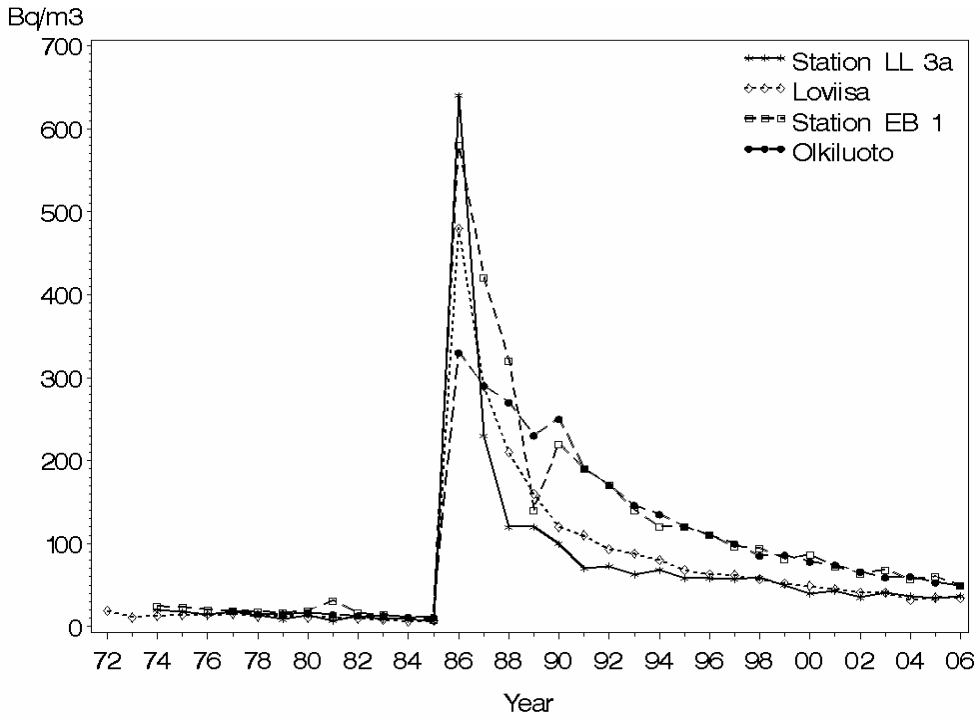


**Fig. 84.** Late-summer mean concentrations of $^{137}$Cs in surface seawater (Bq m$^{-3}$) at Loviisa and Olkiluoto, and the nearest offshore stations LL3a (Gulf of Finland) and EB 1 (Bothnian Sea) in 1972–2006.

## 3.5 Radioactive substances in aquatic indicator organisms

### 3.5.1 Sampling objects and network in the regular environmental monitoring programmes

From the very beginning, the occurrence of radioactive substances in the aquatic environment has been followed in the permanent monitoring programmes of the Loviisa and Olkiluoto NPPs by means of several indicator organisms that effectively accumulate radionuclides from water and sediments, thus facilitating the detection of small traces of radionuclides in the environment. The Bladderwrack, *Fucus vesiculosus,* is one of the most common indicator organisms used in radioecological studies almost worldwide, and consequently, it has also frequently been used in Finnish coastal waters (e.g. Ilus et al. 1988). *Fucus* is a large perennial brown alga, which forms dense growths in the littoral zone. In the Baltic Sea, the *Fucus* belts are often the dominant, and sometimes the only seaweeds in the littoral zone providing shelter and a source of nourishment for other organisms. In the oceans the thalli of *Fucus vesiculosus* can grow up to 100 cm or more, but in the areas off Loviisa and Olkiluoto they are not more than 30 or 40 cm in length, respectively, (Fig. 85, Ilus et al. 1981, 1983) due to the low salinity of the water.

The filamentous green algae *Cladophora glomerata* and *Ulva* spp. have also been used as indicators of radioactive substances in the cooling water recipients of the Loviisa and Olkiluoto NPPs right from the start of monitoring. They too are widespread algae, common in different kinds of water bodies, forming dense and uniform green growths near the shore line. A principal sampling object has been the Blanket weed, *Cladophora glomerata,* but it has sometimes been changed to Gut weed, *Ulva* spp., due to the alternation of these algae on the same growing base.

Benthic animals have been represented in the monitoring programmes by a Baltic glacial relict crustacean *Saduria entomon* at Loviisa, and by the bivalve mussels, the Baltic Tellin (*Macoma balthica*) and the Common mussel (*Mytilus edulis*) at Olkiluoto. *Saduria* is one of the largest crustaceans in the Baltic Sea (5–9 cm). It is a predator that feeds on other benthic animals; it is also a cannibal, as well as a carrion eater. *S. entomon* is an important prey for fish, at least for cod, eel, flounder and perch. *Macoma balthica* is the most common bivalve mussel on mud, clay and sandy mud bottoms all around the Baltic Sea. In Finnish coastal waters it grows to 1–2 cm long. It sucks a mass of bacteria and other micro-organisms from the surface of the sediment, and is an important prey for many benthic fish. *Mytilus edulis* is one of the key species of coastal ecosystems in the Baltic Sea. It is common in the sea area off Olkiluoto, but very scarce in the Loviisa area due to the low salinity of the water. The

common mussel is a cosmopolitan that often lives in big colonies attached to rocks and other hard substrates by strong byssal threads. In the Olkiluoto area, the Common mussel is generally less than 3 cm long. It filters nutritive particles from the water, and is an important prey for many coast-living animals (esp. eider ducks and other water fowls).

Since 1998, periphyton and the submerged seed plants, Spiked water milfoil *(Myriophyllum spicatum)* and Hair pondweed *(Potamogeton pectinatus)* have been used as new indicator organisms in the permanent monitoring programmes of both power plants. Periphyton is a complex mixture of algae, cyanobacteria, heterotrophic microbes and detritus that is attached to submerged surfaces in most aquatic ecosystems. It serves as an important food source for invertebrates and some fish, and is an important absorber of contaminants. Spiked water milfoil and Hair pondweed are tall submerged plants that thrive well in the warmed waters close to the outlets of cooling water from the power plants. In these habitats they can reach a length of several metres.

The samples of filamentous green algae were taken by snorkelling, while those of the bladder-wrack *(Fucus vesiculosus)*, submerged seed plants *(Myriophyllum spicatum* and *Potamogeton pectinatus)* and the common mussel *(Mytilus edulis)* were taken by means of scuba diving. The samples of the relict crustacean *Saduria entomon* were caught with bait nets, and the samples of the bivalve mussel *Macoma balthica* with an Ekman-Birge grab and a benthos sieve. The minimum sample size was 1.8 kg fresh weight for bladder-wrack and submerged seed plants, 0.5 kg for filamentous green algae and 200 g for *Mytilus, Saduria* and *Macoma*. Periphyton was collected on large collection plates (with a total collection area of 1 $m^2$). The large collection area is necessary to obtain a sufficient amount of periphyton for analysis. The plates were installed with buoys and anchors at a depth of 0.5 – 1 m close to the cooling water outlets of the power plants, and periphyton was scraped off the plate into a large collection vessel four times a growing season, and transported to the laboratory for pre-treatment and analysis. The methods are described in detail in Ilus et al. (2008).

In general, samples of the indicator organisms were taken once a year from one sampling site in the discharge area close to the cooling water outlet. The sampling sites are given in Figs. 63 and 64. Periphyton was collected continuously during the growing season (May-September) just in front of the cooling water outlet, and the accrued algal mass was recovered four times during the growing season. Until 1987, samples of *Fucus* were taken once or twice a year (May and August) from two sites (A and B) in both areas. In 1987 – 1992, *Fucus* was collected 3 times per year (May, end of June, August) from three sites (A, B and C) in both areas, and after that twice a year from five sites (A, B, C, D, E), except that at Loviisa the sites D and E were changed for sites F and G in 2003 (Fig. 63).

### 3.5.2 Caesium-137, strontium-90 and plutonium-239,240 in *Fucus*

**Loviisa**

In 1975–1976 (before the start-up of the power plant), the activity concentrations of $^{137}$Cs varied between 14 and 28 Bq kg$^{-1}$ d.w. in the *Fucus* samples collected from sampling site A, and between 16 and 23 Bq kg$^{-1}$ d.w. in those collected from sampling site B, both situated in Hästholmsfjärden Bay (Fig. 63). During the next ten years, the concentrations of the weapons-tests-fallout-originated $^{137}$Cs declined slowly, and in 1985, the concentrations were 12–13 Bq kg$^{-1}$ d.w. in the samples from site A and 8–11 Bq kg$^{-1}$ d.w. in those from site B. The Chernobyl fallout caused a sudden rise in the concentrations in 1986. At the end of May, the concentration of $^{137}$Cs was 4 900 Bq kg$^{-1}$ d.w. in *Fucus* from site A, and 2 700 Bq kg$^{-1}$ d.w. in that from site B. However, the concentrations soon started to decline at a good rate. Already by the end of August, the concentrations had decreased to a quarter of their top values: to 1 100 Bq kg$^{-1}$ d.w. at site A and to 700 at site B. Since then, the concentrations have steadily decreased. In 1990, the concentrations were 160–240 Bq kg$^{-1}$ d.w. at the site A and 150–190 Bq kg$^{-1}$ d.w. at the site B. In 2007, the corresponding values were 25–38 and 22–31 Bq kg$^{-1}$ d.w. Based on these results, the effective ecological half-life of $^{137}$Cs in *Fucus* was 4.5 ± 0.1 years between 1986 and 2007 at the Hästholmsfjärden sites.

At the sampling sites situated outside Hästholmsfjärden Bay, the activity concentrations of $^{137}$Cs were somewhat lower than at the sampling sites A and B situated closest to the power plant. In 1987, the concentrations were 240–570 Bq kg$^{-1}$ d.w. at sampling site C (Lilla Djupberget), when they were concurrently 400–770 Bq kg$^{-1}$ d.w. in Hästholmsfjärden Bay. In 1993, when sites D and E (Boistö and Storskarven) were brought into the programme, the concentrations were 67–88, 72–86 and 46–72 Bq kg$^{-1}$ d.w. at sites C, D and E, respectively. In 2003, sites D and E were changed for sites F and G (Yttre Täktarn and Hudö), and in 2007 the activity concentrations of $^{137}$Cs were 19–22, 21–25 and 23–26 Bq kg$^{-1}$ d.w. at sampling sites C, F and G, respectively. At site C, the calculated ecological half-life of $^{137}$Cs in *Fucus* was 5.5 years between 1986 and 2007, i.e. one year longer than in Hästholmsfjärden (see above).

In 1975–1976, the activity concentrations of $^{90}$Sr varied between 24 and 48 Bq kg$^{-1}$ d.w. in *Fucus* samples taken from the Hästholmsfjärden Bay. Just before the Chernobyl accident, the concentrations had declined to 14–18 Bq kg$^{-1}$ d.w. The fallout from the accident raised the values to 83 Bq kg$^{-1}$ d.w. at site A, and to 48 Bq kg$^{-1}$ d.w. at site B. In 1993, the $^{90}$Sr concentrations were 14 and 16 Bq kg$^{-1}$ d.w. at sites A and B, and in 2007, 11 and 9 Bq kg$^{-1}$ d.w. at sites A (Hästholmsfjärden) and G (Hudöfjärden), respectively. In 1976–1982, the activity concentrations

of $^{239,240}$Pu varied between 0.21 and 0.48 Bq kg$^{-1}$ d.w. in the samples taken from sampling sites A and B in Hästholmsfjärden Bay. In all samples taken from sites A, E and G in 1993–2007, the $^{239,240}$Pu concentrations were 0.027–0.18 Bq kg$^{-1}$ d.w. On the grounds of these results, the effective ecological half-lives of $^{60}$Sr and $^{239,240}$Pu in *Fucus* from site A were 8.4 and 6 years, respectively.

In 2008, the quality assurance of *Fucus* sampling was tested at sampling site C at Loviisa by taking five parallel samples from the littoral zone of the rocky islet of Lilla Djupberget. The samples were taken at the same time by two independent scuba divers from a depth of 1.7–2.3 m. The samples were pre-treated with the same standard method used in our laboratory, and analysed gammaspectrometrically. The activity concentrations of $^{7}$Be, $^{40}$K and $^{137}$Cs (Bq kg$^{-1}$ d.w.) and the relative uncertainties of the gamma analysis (1σ) were as follows:

| Sample | $^{7}$Be | $^{40}$K | $^{137}$Cs |
|---|---|---|---|
| 1 | 23.9 5 | 740 4 | 21.3 4 |
| 2 | 19.9 4 | 735 3 | 20.9 3 |
| 3 | 27.7 3 | 645 3 | 21.6 3 |
| 4 | 19.8 3 | 718 3 | 19.2 3 |
| 5 | 19.5 4 | 737 4 | 20.2 4 |

The results show that the uncertainty due to sampling was rather small. The ratio of the standard error to the mean value, expressed as a percentage, was 2% in the case of $^{137}$Cs and 2.5% in the case of $^{40}$K.

**Olkiluoto**

In 1974–1976 (before the start-up of the power plant), the activity concentrations of $^{137}$Cs were 8 Bq kg$^{-1}$ d.w. in the *Fucus* samples collected from sampling site A situated in Iso Kaalonpuhti Bay in front the cooling water outlet of the power plant (Fig. 64). During the next ten years, the concentrations of the weapons-tests-fallout-originated $^{137}$Cs stayed about at the same level, and in 1985, the concentrations were 7 Bq kg$^{-1}$ d.w. in the samples taken from sites A and B (Iso Kaalonpuhti and Kalliopöllä). In 1986, the Chernobyl fallout caused a sudden rise in the concentrations. On the 28$^{th}$ of April, the concentration of $^{137}$Cs was still 25 Bq kg$^{-1}$ d.w., but on the 21$^{st}$ of May it was 1 300 Bq kg$^{-1}$ d.w. in the *Fucus* sample collected from site A. However, in the same way as at Loviisa, the concentrations started to decline during the next months at a good rate. Already by the 14$^{th}$ of August, the concentrations had decreased to 280 and 300 Bq kg$^{-1}$ d.w. at sites A and B. After that, the concentrations have steadily decreased. In 1990, the concentrations were 110–140 Bq kg$^{-1}$ d.w. at sites A and B, and in

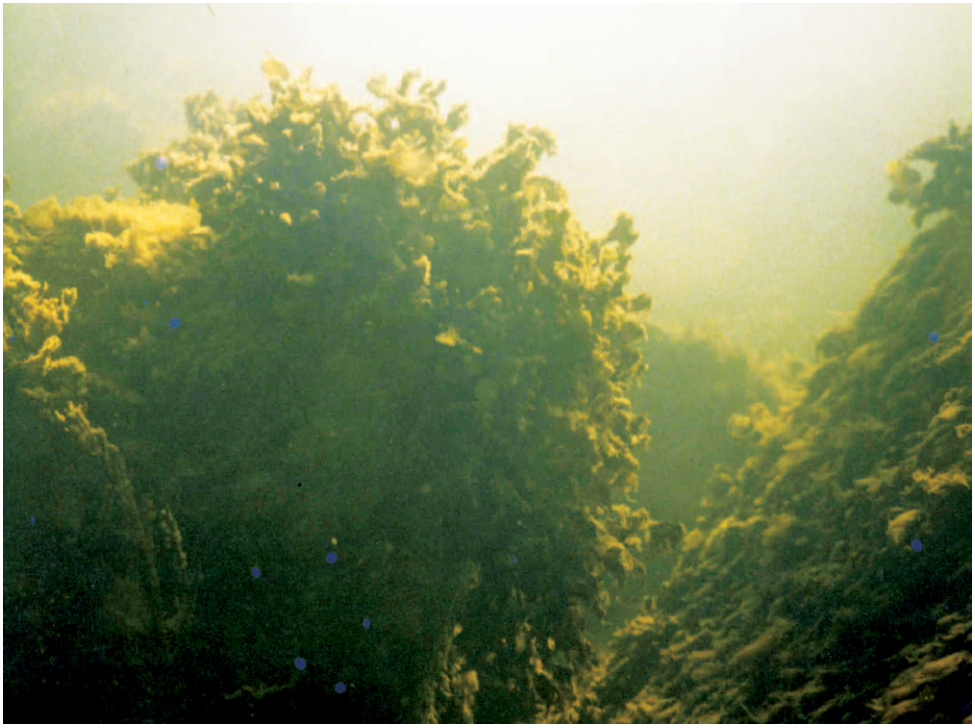

**Fig. 85.** Small-sized *Fucus vesiculosus* typical of the Loviisa area.

2006, the corresponding values were 31–39 and 24–37 Bq $kg^{-1}$ d.w., respectively. Based on these results, the effective ecological half-life of $^{137}Cs$ in *Fucus* was 6.8 ± 0.7 years between 1986 and 2007 at the sites A and B.

At sampling sites C, D and E (Iso Pietari, Iso Siiliö and Vähäkrunni), the activity concentrations of $^{137}Cs$ were somewhat smaller than at the sampling sites A and B situated closest to the power plant. In 1987, the concentrations were 150–220 Bq $kg^{-1}$ d.w. at sampling site C in Iso Pietari, at the same time as they were 220–370 and 230–410 at sites A and B in Iso Kaalonpuhti and Kalliopöllä. In 1993, when sites D and E (Iso Siiliö and Vähäkrunni) were brought into the programme, the concentrations there in *Fucus* were 69–81 and 54–78 Bq $kg^{-1}$ d.w., respectively, while they were concurrently 81–130, 70–120 and 77–86 Bq $kg^{-1}$ d.w. at sites A, B and C (Iso Kaalonpuhti, Kalliopöllä and Iso Pietari), respectively. In 2006, the concentrations were 31–39, 24–37, 26–28, 25–27 and 27–31 Bq $kg^{-1}$ d.w. the sites A, B, C, D and E, respectively. At site C, the ecological half-life of $^{137}Cs$ in *Fucus* was 6.7 years between 1986 and 2007.

In 1974–1976, the activity concentrations of $^{90}Sr$ were 15–17 Bq $kg^{-1}$ d.w. in *Fucus* samples taken from Iso Kaalonpuhti Bay. The Chernobyl fallout raised

the values only slightly, so that in 1987 the values were 25 and 24 Bq kg$^{-1}$ d.w. at sites A and B. In 1993, the $^{90}$Sr concentrations were 12–14 Bq kg$^{-1}$ d.w. at sites A, B and C (Iso Kaalonpuhti, Kalliopöllä and Iso Pietari), while in 2007, they were about 7 Bq kg$^{-1}$ d.w. at sites A and C. In 1976–1982, the activity concentrations of $^{239,240}$Pu varied between 0.05 and 0.70 Bq kg$^{-1}$ d.w. in the *Fucus* samples taken from sampling sites A and B, while in 1993–2007, the concentrations were 0.023–0.14 Bq kg$^{-1}$ d.w. in all the samples taken from sites A, B and C.

### 3.5.3 Other artificial radionuclides in *Fucus*

**Loviisa**

Between the start-up of the power plant (1977) and the Chernobyl accident (1986), 16 gamma-emitting radionuclides (excluding $^{137}$Cs) were detected in the *Fucus* samples taken from sites A and B situated in Hästholmsfjärden Bay. Some of these nuclides were detected only once or twice ($^{125}$Sb, $^{131}$I, $^{134}$Cs, $^{141}$Ce) during the period, while some of them ($^{54}$Mn, $^{60}$Co and $^{110m}$Ag) were detected almost regularly in the *Fucus* samples (Table 14).

$^{54}$Mn was detected in all samples taken in front of the cooling water outlet (site A) and also in nearly all samples taken from the opposite side of Hästholmsfjärden (site B). The highest concentration at site A was 120 Bq kg$^{-1}$ d.w. (1978) and 74 Bq kg$^{-1}$ d.w. at site B (1980). Other frequently-observed nuclides of local origin were $^{60}$Co and $^{110m}$Ag; their highest activity concentrations at site A were 89 and 140 Bq kg$^{-1}$ d.w., respectively. $^{131}$I was detected only once (August 1983) at both sites (Table 14).

The Chernobyl fallout contributed to a strong, sudden and transient appearance and rise of some 15 radionuclides in the *Fucus* samples (Table 15). Many of these nuclides occurred momentarily in relatively high concentrations, but because of their short half-lives (e.g. $^{140}$La, $^{131}$I, $^{140}$Ba, $^{141}$Ce, $^{129m}$Te, $^{95}$Nb $^{103}$Ru, $^{95}$Zr, $^{89}$Sr, $^{124}$Sb and $^{95}$Zr with half-lives from less than 2 to 65 days), the concentrations decreased and totally disappeared from the *Fucus* samples very rapidly. The activity concentrations of $^{106}$Ru, $^{110m}$Ag and $^{144}$Ce (with half-lives of 250–365 days) also rose significantly as a consequence of the Chernobyl fallout, but remained somewhat longer at elevated levels (Table 15).

$^{60}$Co, $^{110m}$Ag, $^{54}$Mn and $^{58}$Co are typical constituents in the liquid discharges of the local nuclear power plant (p. 170). However, at least the concentrations of $^{110m}$Ag increased markedly in connection with the Chernobyl fallout, and a slight increase could be seen in the concentrations of $^{54}$Mn, $^{60}$Co and $^{58}$Co, too (Table 15). After the Chernobyl accident, the concentrations of these nuclides have continuously decreased (Table 16), and especially during the last ten

**Table 14.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides, $^{90}$Sr and $^{239,240}$Pu in the *Fucus* samples taken from the sites Loviisa A and B in 1977–1985 (a = in all samples; * = not regularly analysed).

| | Loviisa A (Halkokari) Range [Bq kg$^{-1}$ d.w.] (f) | | Loviisa B (Bölsviken) Range [Bq kg$^{-1}$ d.w.] (f) | |
|---|---|---|---|---|
| **No. of samples** | **15** | | **15** | |
| $^{40}$K | 680–1 200 | (a) | 620–1 200 | (a) |
| $^{54}$Mn | 1.9–120 | (a) | 2.1–74 | (14) |
| $^{58}$Co | 1.0–48 | (11) | 3.7–13 | (7) |
| $^{60}$Co | 9.5–89 | (13) | 2.1–68 | (13) |
| $^{90}$Sr | 14–35 | (a) | 14–39 | (a) |
| $^{95}$Zr | 0.66–38 | (4) | 0.91–27 | (4) |
| $^{95}$Nb | 0.32–140 | (3) | 0.35–28 | (3) |
| $^{103}$Ru | 2.4–5.6 | (3) | 1.6–4.7 | (3) |
| $^{106}$Ru | 5.9–25 | (4) | 6.3–31 | (3) |
| $^{110m}$Ag | 3.8–140 | (13) | 2.1–50 | (12) |
| $^{124}$Sb | 2.7–12 | (4) | 1.0–14 | (3) |
| $^{125}$Sb | 4.5 | (1) | < | (0) |
| $^{131}$I | 10 | (1) | 3.0 | (1) |
| $^{134}$Cs | 1.3–1.6 | (2) | 1.0 | (2) |
| $^{137}$Cs | 10–19 | (a) | 7.8–23 | (a) |
| $^{141}$Ce | 2.6–6.4 | (2) | 1.7–6.7 | (2) |
| $^{144}$Ce | 12–66 | (6) | 12–61 | (6) |
| $^{155}$Eu | 4.4–24 | (2) | 1.3–33 | (4) |
| $^{239,240}$Pu* | 0.30–0.43 | (4) | 0.21– 0.41 | (5) |

< = below the detection limit

years their concentrations have sunk to very low values (Fig. 86), although small amounts of them were still detected quite frequently at the A and B sites in Hästholmsfjärden, and in ever-diminishing measure at the C site in Lilla Djupberget. The power station invested strongly in reducing the discharges, and this was visible in the environmental measurements as a whole in the 1990s and 2000s.

During the period 1993–2002, sites D and E (Boistö and Storskarven) were the most distant sampling sites for *Fucus* in the monitoring programme. Then $^{60}$Co was detected twice (May 1994 and August 1995) in low concentrations (0.26 and 0.27 Bq kg$^{-1}$ d.w.) at site D, but no other local discharge nuclides were detected. In 2003, sampling sites D and E were substituted by sites F and G (Yttre Täktarn and Hudö). Local discharge nuclides were not detected at the latter two sites during the period 2003–2007.

**Olkiluoto**

Between the start-up of the power plant (1978) and the Chernobyl accident (1986) nine gamma-emitting radionuclides (excluding $^{137}$Cs) were detected in the *Fucus* samples taken from sites A and B situated at distances of about 1 and 2.5 km from the cooling water outlet. $^{95}$Zr, $^{144}$Ce and $^{155}$Eu were detected only once or twice, while $^{54}$Mn, $^{60}$Co and $^{58}$Co were detected quite regularly in the *Fucus* samples (Table 17). The highest activity concentration of $^{60}$Co was 100 Bq kg$^{-1}$ d.w. at site A (Iso Kaalonpuhti) and 23 Bq kg$^{-1}$ d.w. at the site B (Kalliopöllä) in 1985. The highest activity concentrations of $^{54}$Mn and $^{58}$Co were 86 and 27 Bq kg$^{-1}$ d.w., respectively, at site A.

**Table 15.** Gamma-emitting radionuclides, $^{89}$Sr and $^{90}$Sr in *Fucus vesiculosus* (Bq kg$^{-1}$ dry weight) at sampling site Loviisa A in 1985–1987.

| Date | 9 Aug. 1985 | 29 May 1986 | 29 Aug. 1986 | 28 May 1987 | 25 Aug. 1987 |
|---|---|---|---|---|---|
| $^{40}$K | 1 100 | 1 100 | 1 100 | 1 200 | 1 000 |
| $^{54}$Mn | 6.7 | 31 | 19 | 7.7 | 5.7 |
| $^{58}$Co | 1.8 | < | 10 | < | 5.0 |
| $^{60}$Co | 36 | 44 | 35 | 58 | 24 |
| $^{65}$Zn | < | < | 7.1 | < | < |
| $^{89}$Sr | < | 710 | 53 | < | < |
| $^{90}$Sr | 15 | 83 | 28 | 21 | 24 |
| $^{95}$Zr | < | 210 | 17 | < | < |
| $^{95}$Nb | < | 400 | 36 | < | < |
| $^{103}$Ru | < | 5 900 | 260 | < | < |
| $^{106}$Ru | < | 2 100 | 410 | 110 | 42 |
| $^{110m}$Ag | 9.3 | 230 | 130 | 26 | 21 |
| $^{124}$Sb | 2.7 | < | 16 | < | < |
| $^{125}$Sb | < | 97 | 24 | 11 | 4.9 |
| $^{129m}$Te | < | 5 600 | 230 | < | < |
| $^{131}$I | < | 13 000 | < | < | < |
| $^{134}$Cs | < | 2 700 | 550 | 310 | 170 |
| $^{136}$Cs | < | 210 | < | < | < |
| $^{137}$Cs | 13 | 4 900 | 1 100 | 770 | 460 |
| $^{140}$Ba | < | 1 900 | < | < | < |
| $^{140}$La | < | 1 800 | 7.1 | < | < |
| $^{141}$Ce | < | 250 | 8.1 | < | < |
| $^{144}$Ce | < | 330 | 35 | 10 | < |

< = below the detection limit

**Table 16.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides, $^{90}$Sr and $^{239,240}$Pu in the *Fucus* samples taken from the Loviisa A, B and C sites in 1988–1997 and 1998–2007 (a = in all samples; * = not regularly analysed).

| | Loviisa A Range [Bq kg$^{-1}$ d.w.] ( f ) | | | | Loviisa B Range [Bq kg$^{-1}$ d.w.] ( f ) | | | | Loviisa C Range [Bq kg$^{-1}$ d.w.] ( f ) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1988–1997 | | 1998–2007 | | 1988–1997 | | 1998–2007 | | 1988–1997 | | 1998–2007 | |
| No. of samples | 25 | | 20 | | 25 | | 20 | | 25 | | 20 | |
| $^{40}$K | 690–1 200 | (a) | 310–960 | (a) | 720–1 000 | (a) | 330–950 | (a) | 720–920 | (a) | 490–900 | (a) |
| $^{54}$Mn | 0.81–4.4 | (24) | 0.21–2.0 | (9) | 0.44–2.8 | (24) | 0.14–1.1 | (8) | 0.36–0.50 | (4) | < | (0) |
| $^{58}$Co | 0.11–4.2 | (10) | 0.16–1.7 | (5) | 0.80–1.8 | (5) | 0.49–0.65 | (3) | – | (0) | < | (0) |
| $^{60}$Co | 2.0–41 | (a) | 0.17–6.9 | (19) | 1.3–31 | (a) | 0.27–5.3 | (16) | 0.27–2.1 | (24) | 0.16–0.22 | (3) |
| $^{90}$Sr* | 11–22 | (15) | 5.9–13 | (10) | 13–22) | (11) | < | (0) | < | (0) | < | (0) |
| $^{95}$Zr | 1.0 | (1) | < | (0) | 1.6 | (1) | < | (0) | < | (0) | < | (0) |
| $^{95}$Nb | 2.7 | (1) | < | (0) | < | (0) | < | (0) | < | (0) | < | (0) |
| $^{106}$Ru | 3.5–34 | (5) | < | (0) | 3.0–27 | (6) | < | (0) | 7.3–10 | (4) | < | (0) |
| $^{110m}$Ag | 0.67–19 | (a) | 0.30–4.6 | (14) | 0.54–15 | (24) | 0.37–2.4 | (11) | 0.38–2.1 | (9) | 0.13 | (1) |
| $^{124}$Sb | 1.2–15 | (2) | 0.19–0.91 | (3) | < | (0) | 0.41–0.51 | (2) | < | (0) | < | (0) |
| $^{125}$Sb | 2.3 | (1) | < | (0) | 1.8–3.0 | (5) | < | (0) | 1.0–2.0 | (2) | < | (0) |
| $^{134}$Cs | 1.4–130 | (a) | 0.21–1.6 | (8) | 1.1–100 | (a) | 0.23–1.0 | (8) | 1.0–100 | (a) | 0.18–0.69 | (7) |
| $^{137}$Cs | 59–450 | (a) | 13–87 | (a) | 56–350 | (a) | 13–72 | (a) | 57–340 | (a) | 18–49 | (a) |
| $^{239,240}$Pu* | 0.11–0.18 | (5) | 0.03–0.12 | (9) | 0.17 | (1) | < | (0) | < | (0) | < | (0) |

< = below the detection limit

**Table 17.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides (excl. $^{137}$Cs) in the *Fucus* samples taken from the Olkiluoto A and B sites in 1978–1985 (a = in all samples ).

| | Olkiluoto A Range [Bq kg$^{-1}$ d.w.] (f) | | Olkiluoto B Range [Bq kg$^{-1}$ d.w.] (f) | |
|---|---|---|---|---|
| No. of samples | 8 | | 8 | |
| $^{40}$K | 500–770 | (a) | 480–680 | (a) |
| $^{54}$Mn | 1.0–86 | (a) | 0.7–16 | (a) |
| $^{58}$Co | 2.2–27 | (7) | 1.8–6.0 | (4) |
| $^{60}$Co | 3.3 –100 | (7) | 4.1–23 | (6) |
| $^{65}$Zn | 5.7–21 | (4) | 2.7–3.6 | (2) |
| $^{95}$Zr | 18 | (1) | 14 | (1) |
| $^{110m}$Ag | 0.44–9.5 | (5) | 1.6–3.0 | (2) |
| $^{144}$Ce | 5.6 | (1) | 7.8 | (1) |
| $^{155}$Eu | < | (0) | 1.4 | (1) |

< = below the detection limit

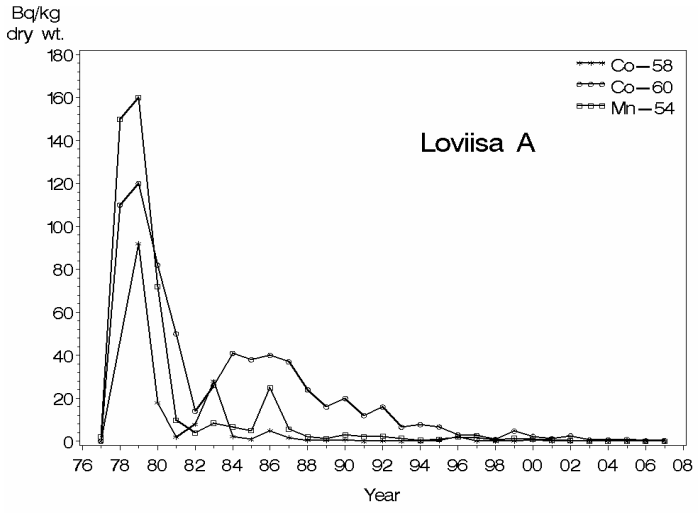


**Fig. 86.** Annual mean concentrations of $^{54}$Mn, $^{58}$Co and $^{60}$Co in *Fucus vesiculosus* (Bq kg$^{-1}$ d.w.) at sampling site Loviisa A in 1977–2007.

The Chernobyl fallout caused a strong, sudden and transient appearance and rise of some 20 radionuclides in the *Fucus* samples. The activity concentrations of gamma-emitting radionuclides and $^{90}$Sr in the *Fucus* samples taken from the nearest sampling site in front of the Olkiluoto NPP (Site A; depth 2 m) are given in Table 18. In August 1985, the sample still contained only small amounts of discharge nuclides from the local power plant ($^{60}$Co, $^{54}$Mn, $^{58}$Co, $^{65}$Zn and $^{110m}$Ag), 7 Bq kg$^{-1}$ of $^{137}$Cs from weapons-tests fallout and the usual amount of naturally-occurring $^{40}$K. The sample taken at 18:00 on the 28$^{th}$ of April 1986 (only some hours after the arrival of the fallout cloud) contained a long list of fresh fallout nuclides which were exotic to the area, and most of them short-lived. The activity concentration of $^{137}$Cs was still only 25 Bq kg$^{-1}$ and that of $^{131}$I was 410 Bq kg$^{-1}$. It is worth mentioning that this sample contained only dry deposition, because the first rain after the arrival of the fallout fell on the morning of April 29. On the 21$^{st}$ of May, the concentrations of $^{131}$I, and those of the caesium and ruthenium isotopes, as well as $^{140}$Ba and $^{128m}$Te, had reached their maximum values ($^{131}$I: 29 000 Bq kg$^{-1}$ and $^{137}$Cs: 1 300 Bq kg$^{-1}$), whereas many short-lived nuclides had already disappeared or started to diminish. In August 1986, the short-lived $^{131}$I (half-life of eight days) had almost completely disappeared and the $^{137}$Cs concentration was about a quarter of its highest value in May. In June 1987, the *Fucus* sample taken from the same sampling site contained only 220 Bq kg$^{-1}$ of $^{137}$Cs, 86 Bq kg$^{-1}$ of $^{134}$Cs and a slightly elevated amount of $^{110m}$Ag, in addition to the background levels of the local discharge nuclides (Ilus 2007).

**Table 18.** Gamma-emitting radionuclides and $^{90}Sr$ in *Fucus vesiculosus* (Bq $kg^{-1}$ dry weight) at sampling site Olkiluoto A in 1985–1987.

| Date | 15 Aug.1985 | 28 April 1986 (18:00) | 21 May 1986 | 14 Aug. 1986 | 9 July 1987 |
|---|---|---|---|---|---|
| $^{40}K$ | 650 | 590 | 690 | 690 | 750 |
| $^{51}Cr$ | < | < | < | < | 17 |
| $^{54}Mn$ | 86 | 18 | 37 | 57 | 140 |
| $^{58}Co$ | 25 | < | < | 14 | 7.8 |
| $^{60}Co$ | 100 | 70 | 110 | 94 | 84 |
| $^{65}Zn$ | 21 | 3.3 | 23 | 26 | 12 |
| $^{90}Sr$ | 15 | | | 14 | |
| $^{95}Zr$ | < | 690 | 110 | 4.2 | < |
| $^{95}Nb$ | < | 920 | 220 | 7.6 | < |
| $^{95m}Nb$ | < | 10 | < | < | < |
| $^{103}Ru$ | < | 570 | 1 900 | 52 | < |
| $^{106}Ru$ | < | 110 | 590 | 61 | < |
| $^{110m}Ag$ | 9.5 | 2.2 | 170 | 62 | 23 |
| $^{125}Sb$ | < | < | 24 | 4.0 | < |
| $^{129m}Te$ | < | 22 | 2 800 | 64 | < |
| $^{131}I$ | < | 410 | 29 000 | 6.8 | < |
| $^{132}Te$ | < | 500 | 300 | < | < |
| $^{134}Cs$ | < | 8.3 | 710 | 140 | 86 |
| $^{136}Cs$ | < | 3.4 | 150 | < | < |
| $^{137}Cs$ | 7.0 | 25 | 1 300 | 280 | 220 |
| $^{140}Ba$ | < | 1 000 | 4 700 | < | < |
| $^{140}La$ | < | 810 | 3 700 | 17 | < |
| $^{141}Ce$ | < | 690 | 94 | 1.7 | < |
| $^{144}Ce$ | < | 360 | 130 | 7.1 | < |
| $^{144}Pr$ | < | 320 | 65 | < | < |
| $^{147}Nd$ | < | 230 | < | < | < |
| $^{237}U$ | < | 38 | < | < | < |
| $^{239}Np$ | < | 3 500 | < | < | < |

< = below the detection limit

**Table 19.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides, $^{90}$Sr and $^{239,240}$Pu in the *Fucus* samples taken from the Olkiluoto A, B and C sites in 1988–1997 and 1998–2007 (a = in all samples).

| | Olkiluoto A (Iso Kaalonpuhti) Range [Bq kg$^{-1}$ d.w.] ( f ) | | | | Olkiluoto B (Kalliopöllä) Range [Bq kg$^{-1}$ d.w.] ( f ) | | | | Olkiluoto C (Iso Pietari) Range [Bq kg$^{-1}$ d.w.] ( f ) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1988–1997 | | 1998–2007 | | 1988–1997 | | 1998–2007 | | 1988–1997 | | 1998–2007 | |
| No. of samples | 25 | | 20 | | 25 | | 20 | | 25 | | 20 | |
| $^{40}$K | 620–860 | (a) | 550–870 | (a) | 490–800 | (a) | 340–770 | (a) | 490–980 | (a) | 480–1 100 | (a) |
| $^{51}$Cr | 4.0–16 | (5) | < | (0) | 5.4–12 | (3) | < | (0) | < | (0) | <) | (0) |
| $^{54}$Mn | 6.6–150 | (a) | 0.25–2.7 | (9) | 2.7–32 | (a) | 0.18–1.2 | (9) | 0.35–2.8 | (18) | < | (0) |
| $^{57}$Co | 0.14 | (1) | < | (0) | < | (0) | < | (0) | < | (0) | < | (0) |
| $^{58}$Co | 0.30–25 | (24) | 0.11–1.2 | (5) | 0.62–6.1 | (15) | 0.35 | (1) | 0.22–0.44 | (3) | < | (0) |
| $^{60}$Co | 1.5–170 | (a) | 0.52–12 | (19) | 7.1–42 | (a) | 0.35–7.1 | (a) | 1.2–5.3 | (a) | 0.12–1.0 | (14) |
| $^{65}$Zn | 0.70–9.3 | (13) | < | (0) | 1.3–2.9 | (6) | < | (0) | 0.52–1.3 | (3) | < | (0) |
| $^{90}$Sr* | 10–20 | (10) | 6.9–17 | (10) | < | (0) | < | (0) | 8.8–14 | (5) | 5.4–12 | (10) |
| $^{95}$Zr | 0.80 | (1) | < | (0) | < | (0) | < | (0) | < | (0) | < | (0) |
| $^{106}$Ru | 22 | (1) | < | (0) | 10–19 | (2) | < | (0) | 2.6–6.4 | (3) | < | (0) |
| $^{110m}$Ag | 1.1–6.9 | (5) | < | (0) | 0.90–6.0 | (5) | < | (0) | 0.33–8.6 | (9) | < | (0) |
| $^{124}$Sb | 0.34–0.76 | (2) | < | (0) | < | (0) | 0.23 | (1) | < | (0) | < | (0) |
| $^{125}$Sb | 0.80-4.8 | (6) | < | (0) | 0.74–3.6 | (7) | < | (0) | 0.90–2.0 | (2) | < | (0) |
| $^{131}$I | 8.0 | (1) | < | (0) | < | (0) | < | (0) | < | (0) | < | (0) |
| $^{134}$Cs | 1.4–86 | (a) | < | (0) | 0.75–79 | (a) | 0.16–1.0 | (7) | 0.50–50 | (a) | 0.08–0.62 | (10) |
| $^{137}$Cs | 65–280 | (a) | 29–73 | (a) | 40–270 | (a) | 18–64 | (a) | 38–180 | (a) | 22–49 | (a) |
| $^{144}$Ce | 5.6 | (1) | < | (0) | 8.2–9.8 | (2) | < | (0) | < | (0) | < | (0) |
| $^{239,240}$Pu* | 0.04–0.08 | (5) | 0.03–0.14 | (10) | 0.052 | (1) | < | (0) | 0.04–0.05 | (4) | 0.02–0.08 | (10) |

* = not analysed regularly, < = below the detection limit

**Table 20.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides in the *Fucus* samples taken from the Olkiluoto D and E sites in 1993–1997 and 1998–2007 (a = in all samples).

| | Olkiluoto D (Iso Siiliö) Range [Bq kg$^{-1}$ d.w.] ( f ) | | | | Olkiluoto E (Vähäkrunni) Range [Bq kg$^{-1}$ d.w.] ( f ) | | | |
|---|---|---|---|---|---|---|---|---|
| | 1993–1997 | | 1998–2007 | | 1993–1997 | | 1998–2007 | |
| No. of samples | 10 | | 20 | | 10 | | 20 | |
| $^{40}$K | 590–860 | (a) | 330–940 | (a) | 510–700 | (a) | 520–880 | (a) |
| $^{54}$Mn | 0.30–2.4 | (8) | 0.30 | (1) | 0.41–1.2 | (9) | < | (0) |
| $^{58}$Co | 0.36–0.38 | (2) | < | (0) | 0.24–1.3 | (3) | < | (0) |
| $^{60}$Co | 1.6–3.7 | (a) | 0.29–1.5 | (11) | 2.3–4.6 | (a) | 0.18–3.0 | (15) |
| $^{134}$Cs | 0.62–5.0 | (a) | 0.17–0.54 | (7) | 0.69–4.9 | (a) | 0.21–0.46 | (8) |
| $^{137}$Cs | 42–81 | (a) | 11–43 | (a) | 42–78 | (a) | 18–43 | (a) |

< = below the detection limit

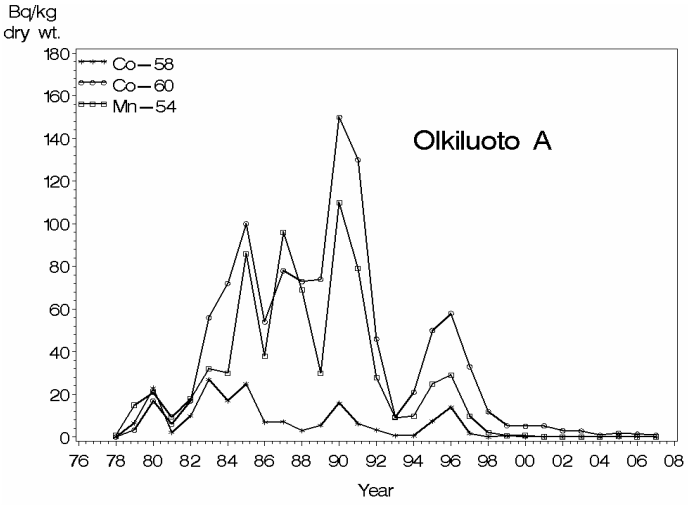


**Fig. 87.** Annual mean concentrations of $^{54}$Mn, $^{58}$Co and $^{60}$Co in *Fucus vesiculosus* (Bq kg$^{-1}$ d.w.) at sampling site Olkiluoto A in 1978–2007.

During the period 1988–1997, small amounts of $^{65}$Zn, $^{95}$Zr, $^{106}$Ru, $^{110m}$Ag, $^{124}$Sb, $^{125}$Sb and $^{144}$Ce were still measured in samples taken from sites A, B and C, but in general no longer in 1998–2007 (Table 19). For these nuclides, the physical half-life of $^{125}$Sb is 2.8 years, that of $^{65}$Zn, $^{110m}$Ag, $^{144}$Ce and $^{106}$Ru 244–368 days and that of $^{124}$Sb and $^{95}$Zr 60–64 days. Thus, these particular nuclides were most probably traces of the Chernobyl fallout. On the other hand, $^{125}$Sb, $^{124}$Sb, $^{110m}$Ag and $^{95}$Zr were frequently or quite frequently present in the liquid discharges of the power plant (p. 170).

$^{57}$Co and $^{131}$I were undoubtedly of local origin, and were detected only once during the period 1988–1997 (May 1996) at Olkiluoto A (Table 19). $^{51}$Cr was detected in *Fucus* collected at sites A and B now and then during the period 1988–1997, but no longer in 1998–2007. The concentrations of $^{60}$Co and $^{54}$Mn reached their maxima in 1990, and after a temporary decline in 1993 increased again in 1995 and 1996, but since then they have significantly decreased at all the sampling sites. Especially in the last nine years their concentrations have sunk to very low values (Fig. 87). In general, there was a clear difference in the concentrations of $^{60}$Co, $^{54}$Mn and $^{58}$Co between the sampling sites, in that the concentrations at the sites nearest to the cooling water outlet (A and B) were clearly higher than those at the outer sampling sites C, D and E (Table 20). Nevertheless, these nuclides were quite regularly detected at all the sampling sites. The power station invested strongly in reducing the discharges, and this was visible in the environmental measurements as a whole in the 2000s.

### 3.5.4 Radioactive substances in filamentous green algae

**Loviisa**

Before the Chernobyl accident (1975–1985), the activity concentrations of $^{137}$Cs in filamentous green algae at sampling site A in front of the cooling water outlet varied between 2.9 and 12 Bq kg$^{-1}$ d.w. $^{60}$Co, $^{58}$Co and $^{54}$Mn were detected only occasionally; their maximum values were 24, 9.0 and 28 Bq kg$^{-1}$ d.w, respectively. The highest values were those of $^{95}$Nb, $^{95}$Zr and $^{144}$Ce detected in 1977: 140, 110 and 82 kg$^{-1}$ d.w., respectively.

The general picture gained from the results of the green algae collected from Loviisa A after the Chernobyl accident was very similar to that of *Fucus vesiculosus* collected from the same site (*cf.* Tables 15 and 21). Green algae seemed to accumulate some radionuclides a little more effectively, but these were

**Table 21.** Gamma-emitting radionuclides and $^{90}$Sr in filamentous green algae (*Cladophora glomerata* / *Ulva* spp.) at sampling site Loviisa A in 1985–1988 (Bq kg$^{-1}$ d.w.)

| Date | 9 Aug. 1985 | 29 May 1986 | 29 Aug. 1986 | 25 June 1987 | 28 July 1988 |
|---|---|---|---|---|---|
| $^{40}$K | 850 | 1 100 | 970 | 2 100 | 2 200 |
| $^{54}$Mn | < | 19 | 4.6 | 1.2 | 3.0 |
| $^{58}$Co | 1.3 | < | 6.3 | < | < |
| $^{60}$Co | 2.4 | 12 | 6.5 | 3.5 | 7.6 |
| $^{90}$Sr | 1.9 | – | 2.1 | 5.0 | 3.3 |
| $^{95}$Zr | < | 1 000 | < | < | < |
| $^{95}$Nb | < | 1 500 | 4.5 | < | < |
| $^{103}$Ru | < | 7 100 | 71 | < | < |
| $^{106}$Ru | < | 2 600 | 110 | 41 | < |
| $^{110m}$Ag | 11 | 240 | 60 | 7.6 | < |
| $^{124}$Sb | 4.0 | < | 13 | < | < |
| $^{125}$Sb | < | 120 | < | < | < |
| $^{129m}$Te | < | 6 000 | 70 | < | < |
| $^{131}$I | < | 2 600 | < | < | < |
| $^{134}$Cs | < | 2 400 | 290 | 84 | 51 |
| $^{136}$Cs | < | 230 | < | < | < |
| $^{137}$Cs | 7.7 | 4 300 | 570 | 200 | 170 |
| $^{140}$Ba | < | 790 | < | < | < |
| $^{140}$La | < | 670 | < | < | < |
| $^{141}$Ce | < | 730 | < | < | < |
| $^{144}$Ce | < | 890 | < | < | < |

< = below the detection limit, – = not analysed

also more quickly removed. In late summer 1987, the activity concentration of $^{137}$Cs was 460 Bq kg$^{-1}$ d.w. in *Fucus*, while in green algae it was 200 Bq kg$^{-1}$ d.w.

Since 1988, local discharge nuclides ($^{60}$Co, $^{110m}$Ag and $^{54}$Mn) were detected in green algae only now and then. The maximum concentration of $^{60}$Co was 3.7 Bq kg$^{-1}$ d.w. in 1991. In 2007, the activity concentration of $^{137}$Cs in green algae from Loviisa A was 7.4 Bq kg$^{-1}$ d.w.

**Olkiluoto**

Before the Chernobyl accident (1974–1985), the activity concentrations of $^{137}$Cs in filamentous green algae at sampling site A in front of the cooling water outlet of the Olkiluoto power plant varied between 2.3 and 15 Bq kg$^{-1}$ d.w. After the start-up of the power plant (1978), local discharge nuclides $^{60}$Co, $^{58}$Co, $^{54}$Mn, $^{110m}$Ag and $^{51}$Cr were detected quite frequently in green algae. In 1982, the maximum concentrations of $^{54}$Mn, $^{51}$Cr, $^{60}$Co and $^{58}$Co were 190, 140, 83 and 74 Bq kg$^{-1}$ d.w., respectively.

In 1986–1988, the highest concentrations of $^{137}$Cs and $^{110m}$Ag were 120 and 100 Bq kg$^{-1}$ d.w. in green algae samples taken from Olkiluoto A. Since 1989, the concentrations of $^{137}$Cs and $^{134}$Cs have decreased according to their ecological half-lives. In 1990, the activity concentrations of $^{54}$Mn and $^{58}$Co were 400 and 17 Bq kg$^{-1}$ d.w., but after 1997 they were no longer detected. The maximum concentration of $^{60}$Co was 92 Bq kg$^{-1}$ d.w. in 1989, and it was regularly detected in the 1990s. In the 2000s, $^{60}$Co was the only gamma-emitting radionuclide besides $^{40}$K and $^{137}$Cs found in green algae. However, it was not detected in 2004–2006, and in 2007 its concentration was only 0.80 Bq kg$^{-1}$ d.w.

### 3.5.5 Radioactive substances in periphyton

Periphyton was brought into use as a new sampling object in the monitoring programmes of the power plants in 1998. Thus, the impact of the Chernobyl fallout on the activity concentrations of the fallout-originated nuclides, such as $^{137}$Cs and $^{134}$Cs, had largely faded out before that. Nevertheless, even in 1998–2007, $^{137}$Cs was still the dominating artificial gamma-nuclide in periphyton also (Table 22). The $^{7}$Be and $^{40}$K occurring abundantly in the samples are natural radionuclides. In 2007, the average concentration of $^{137}$Cs was 137 Bq kg$^{-1}$ d.w. in periphyton collected at the cooling water outlet at Loviisa and 113 Bq kg$^{-1}$ d.w. at Olkiluoto. In addition to the four above-mentioned radionuclides, only $^{60}$Co, $^{54}$Mn and $^{58}$Co were detected in periphyton samples from Olkiluoto. At Loviisa, $^{110m}$Ag and $^{124}$Sb were also frequently detected, with $^{51}$Cr, $^{59}$Fe, $^{95}$Zr, $^{95}$Nb and $^{123m}$Te being found more rarely in low concentrations. In 2007, the average content of $^{60}$Co was 4.6 Bq kg$^{-1}$ d.w. at Loviisa and 3.0 Bq kg$^{-1}$ d.w. at Olkiluoto. Since 2003, $^{54}$Mn was

not detected in periphyton collected from Olkiluoto A, except in one sample from the period May–June 2006. Periphyton seemed to accumulate caesium and the main local discharge nuclides more effectively than *Fucus,* taking into account, however, that periphyton was sampled right at the outflows of the cooling water, while *Fucus* was sampled a little further away.

**Table 22.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides in the periphyton samples taken from the Loviisa A and Olkiluoto A sites in 1998–2007 (a = in all samples).

| | Loviisa A Range [Bq kg$^{-1}$ d.w.] (f) | | Olkiluoto A Range [Bq kg$^{-1}$ d.w.] (f) | |
|---|---|---|---|---|
| **No. of samples** | **40** | | **40** | |
| $^{7}$Be | 60–2 100 | (39) | 66–1 400 | (a) |
| $^{40}$K | 400–1 300 | (a) | 200–960 | (a) |
| $^{51}$Cr | 19 | (1) | < | (0) |
| $^{54}$Mn | 0.55–40 | (23) | 0.58–6.1 | (13) |
| $^{58}$Co | 0.25–45 | (23) | 2.7 | (1) |
| $^{59}$Fe | 5.5–17 | (2) | < | (0) |
| $^{60}$Co | 0.61–83 | (34) | 1.0–54 | (38) |
| $^{95}$Zr | 1.6–3.4 | (2) | < | (0) |
| $^{95}$Nb | 5.8–7.3 | (2) | < | (0) |
| $^{110m}$Ag | 0.90–58 | (25) | < | (0) |
| $^{123m}$Te | 0.22–1.4 | (2) | < | (0) |
| $^{124}$Sb | 3.5–22 | (18) | < | (0) |
| $^{134}$Cs | 0.82–4.0 | (12) | 0.93–10 | (11) |
| $^{137}$Cs | 21–420 | (a) | 41–540 | (a) |

< = below the detection limit

### 3.5.6 Radioactive substances in *Myriophyllum spicatum* and *Potamogeton pectinatus*

Spiked water milfoil *(Myriophyllum spicatum)* and Hair pondweed *(Potamogeton pectinatus)* were also introduced into the monitoring programmes in 1998. $^{137}$Cs was also the dominating artificial radionuclide in these submerged seed plants (Table 23). As well as $^{7}$Be, $^{40}$K, $^{134}$Cs and $^{137}$Cs, ten other gamma-nuclides were detected in *Myriophyllum* and five in *Potamogeton* samples at Loviisa. At Olkiluoto, the corresponding numbers were six other gamma-nuclides in *Potamogeton* and three in *Myriophyllum*. In 2007, the activity concentrations of $^{137}$Cs were 3.7 and 4.1 Bq kg$^{-1}$ d.w. in *Myriophyllum* and *Potamogeton* at Loviisa, and 12 and 16 Bq kg$^{-1}$ d.w. at Olkiluoto. Thus, the concentrations were quite close to each other in the two plant species, but in both of them clearly higher at Olkiluoto than at Loviisa. At Olkiluoto, the $^{60}$Co concentration was 0.80 Bq kg$^{-1}$ d.w. in *Myriophyllum* and 1.2 Bq kg$^{-1}$ d.w. in *Potamogeton* (2007). In general, the $^{60}$Co concentrations tended to decrease in both species in both areas during 1998–2007, this tendency being more pronounced at Olkiluoto.

**Table 23.** Range of activity concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of gamma-emitting radionuclides in the samples of *Myriophyllum spicatum* and *Potamogeton pectinatus* taken from the Loviisa A and Olkiluoto A sites in 1998–2007 (a = in all samples).

| | ***Myriophyllum spicatum*** **Range [Bq kg$^{-1}$ d.w.] ( f )** | | | | ***Potamogeton pectinatus*** **Range [Bq kg$^{-1}$ d.w.] ( f )** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Loviisa A** | | **Olkiluoto A** | | **Loviisa A** | | **Olkiluoto A** | |
| **No. of samples** | **11** | | **10** | | **11** | | **11** | |
| $^{7}$Be | 62–290 | (a) | 21–69 | (9) | 14–100 | (a) | 11–85 | (a) |
| $^{40}$K | 440–700 | (a) | 290–700 | (a) | 470–820 | (a) | 370–630 | (a) |
| $^{51}$Cr | 7.9 | (1) | < | (0) | < | (0) | < | (0) |
| $^{54}$Mn | 0.66–4.0 | (9) | 0.46–3.8 | (7) | 0.28–3.7 | (8) | 0.38–2.3 | (4) |
| $^{58}$Co | 2.7–20 | (7) | 0.17–1.0 | (3) | 0.45–15 | (8) | 0.10–0.36 | (3) |
| $^{59}$Fe | 0.76 | (1) | < | (0) | < | (0) | 0.30 | (1) |
| $^{60}$Co | 0.74–19 | (10) | 0.80–18 | (a) | 0.50–3.6 | (9) | 0.75–9.8 | (a) |
| $^{95}$Zr | 0.42–1.7 | (4) | < | (0) | < | (0) | < | (0) |
| $^{95}$Nb | 0.61–5.1 | (3) | < | (0) | < | (0) | < | (0) |
| $^{110m}$Ag | 2.4–23 | (7) | < | (0) | 0.24–23 | (9) | 0.20 | (1) |
| $^{123m}$Te | 0.47–1.0 | (4) | < | (0) | < | (0) | < | (0) |
| $^{124}$Sb | 2.8–20 | (7) | < | (0) | 0.36–2.3 | (7) | 0.20 | (1) |
| $^{134}$Cs | 0.18–1.2 | (4) | 0.57–1.0 | (2) | 0.20–0.39 | (2) | 0.17–0.88 | (3) |
| $^{137}$Cs | 4.1–92 | (a) | 12–67 | (a) | 3.7–27 | (a) | 5.3–62 | (a) |

< = below the detection limit

**Table 24.** Activity concentrations or range of concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of $^{89}$Sr, $^{90}$Sr, $^{239,240}$Pu and gamma-emitting radionuclides in *Saduria entomon* caught from the deep area of Hästholmsfjärden (Loviisa 3) in 1976–2007.

| | ***Saduria entomon*** **[Bq kg d.w.] (f)** | | | | |
|---|---|---|---|---|---|
| **Date** | **1976–1985** | **15 May 1986** | **21 May 1987** | **1988-1997** | **1998–2007** |
| **No. of samples** | **14** | **1** | **1** | **10** | **10** |
| $^{40}$K | 150–350 | | 250 | 160–250 | 190–280 |
| $^{54}$Mn | 2.1–2.8 (2) | < | < | 11 (1) | < |
| $^{60}$Co | 1.1–19) (6) | 2.7 | 16 | 0.81–10 (10) | 0.70–1.4 (3) |
| $^{65}$Zn | 14 (1) | < | < | < | < |
| $^{89}$Sr | < | 34 | < | < | 3.0 (1) |
| $^{90}$Sr | 24–41 | 8.2 | 25 | 15–28 | 9.2–24 |
| $^{95}$Zr | < | 18 | < | < | < |
| $^{95}$Nb | < | 75 | < | < | < |
| $^{103}$Ru | < | 140 | < | < | < |
| $^{106}$Ru | < | 51 | 28 | < | < |
| $^{110m}$Ag | 1.2–44 (12) | 12 | 110 | 2.7–52 (10) | 0.46–3.8 (8) |
| $^{125}$Sb | < | 5.2 | < | < | < |
| $^{129m}$Te | < | 170 | < | < | < |
| $^{134}$Cs | < | 58 | 91 | 0.78–48 (7) | 0.50–0.92 (2) |
| $^{137}$Cs | 2.3–11 | 99 | 230 | 23–144 | 13–29 |
| $^{141}$Ce | < | 170 | < | < | < |
| $^{144}$Ce | < | 20 | < | < | < |
| $^{239-240}$Pu | 0.091–1.1 | | | | |

< = below the detection limit

*Myriophyllum* seemed to accumulate caesium and the main local discharge nuclides more effectively than *Fucus*. On the other hand, *Fucus* seemed to be a somewhat better indicator of caesium than *Potamogeton*, whereas the uptake of the main discharge nuclides by *Fucus* and *Potamogeton* was quite similar.

### 3.5.7 Radioactive substances in *Saduria entomon*

Samples of the relict crustacean *Saduria entomon* were caught once a year since 1976 from the deep area of Hästholmsfjärden Bay, south of the Sampling Station Loviisa 3. The distance from the cooling water outlet is ca. 1.8 km. Before the Chernobyl accident (1976–1985), the $^{137}$Cs concentrations in *Saduria* varied between 2.3 and 11 Bq kg$^{-1}$ d.w. and those of $^{90}$Sr between 24 and 41 Bq kg$^{-1}$ d.w. (Table 24). $^{110m}$Ag was regularly detected in *Saduria* since 1978, with the

concentrations varying between 1.2 and 44 Bq $kg^{-1}$ d.w. Other local discharge nuclides ($^{60}$Co, $^{54}$Mn and $^{65}$Zn) were also observed, but less frequently. The Chernobyl fallout was marked by the appearance of ten additional nuclides in the sample from May 1986. However, the concentrations of $^{137}$Cs, $^{134}$Cs and $^{110m}$Ag reached their maximum only in the samples from May 1987, by which time most of the other fallout-originated nuclides with shorter half-lives had already disappeared. Decreasing amounts of $^{110m}$Ag and $^{60}$Co were still detected in all samples from the period 1988–1997, but since 2001 the concentrations of $^{60}$Co were below the detection limit, and the concentrations of $^{110m}$Ag have also been lower than earlier in the 2000s. As a whole, *Saduria* has proved to be an excellent indicator of $^{110m}$Ag.

### 3.5.8 Radioactive substances in *Macoma balthica* and *Mytilus edulis*

Samples of the bivalve mussels Baltic Tellin *(Macoma balthica)* and Common mussel *(Mytilus edulis)* were collected once a year since 1977 in front of the cooling water outlet at Olkiluoto. The distance of the sampling sites from the outlet is ca. 1 km. *Mytilus* was collected from the littoral vegetation zone (depth 1–2.5 m), while *Macoma* was collected from the profundal zone (depth about 9 m). The mussels were analysed with their shells. Before the Chernobyl accident (1977–1985), the activity concentrations of $^{137}$Cs were < 5 Bq $kg^{-1}$ d.w. in both species, while those of $^{90}$Sr were 20–41 Bq $kg^{-1}$ d.w. in *Macoma* and 19–28 Bq $kg^{-1}$ d.w. in *Mytilus* (both analysed with shells). Additionally, only $^{60}$Co and $^{110m}$Ag were occasionally detected in low concentrations in *Macoma*, whereas in *Mytilus* $^{60}$Co, $^{54}$Mn and $^{58}$Co were detected more frequently and a little more abundantly (Table 25). As a consequence of the Chernobyl fallout, the $^{137}$Cs concentrations rose in 1986–1987 to 37–52 Bq $kg^{-1}$ d.w. in *Macoma* and to 14–28 Bq $kg^{-1}$ d.w. in *Mytilus*. $^{103}$Ru, $^{106}$Ru, $^{129m}$Te and $^{144}$Ce occurred in the mussels only in 1986–1987, and in addition, the Chernobyl fallout was reflected in variable concentrations of $^{89}$Sr, $^{110m}$Ag, $^{125}$Sb and $^{134}$Cs. On the other hand, the fallout did not affect the $^{90}$Sr concentrations. In general, the $^{90}$Sr, caesium and plutonium concentrations seemed to be slightly higher in *Macoma* than in *Mytilus*. In the post-Chernobyl period (1988–2007), the highest concentrations of $^{60}$Co and $^{54}$Mn found in *Mytilus* were 69 and 77 Bq $kg^{-1}$ d.w. (1990), whereas those in *Macoma* were 6.5 and 3.0 Bq $kg^{-1}$ d.w., respectively. During the last ten years, the concentrations of $^{60}$Co have decreased markedly in both species. Both mussels proved to be good indicators of strontium. *Mytilus* seemed to be a good indicator of cobalt and manganese, but weaker for caesium than *Fucus*. *Macoma* was a weaker indicator of cobalt and manganese than *Mytilus*, but better than *Mytilus* for caesium, and very good for silver.

**Table 25.** Activity concentration or range of concentrations (Bq kg$^{-1}$ d.w.) and detection frequencies (f) of $^{89}$Sr, $^{90}$Sr, $^{239,240}$Pu and gamma-emitting radionuclides in *Macoma balthica* and *Mytilus edulis* sampled from the area off the cooling water outlet at Olkiluoto (Olkiluoto A) in 1977–2007.

| | ***Macoma balthica*** **[Bq kg$^{-1}$ d.w.] (f)** | | | ***Mytilus edulis*** **[Bq kg$^{-1}$ d.w.] (f)** | | |
|---|---|---|---|---|---|---|
| **Date** | **1977–1985** | **1986–1987** | **1988–2007** | **1977–1985** | **1986–1987** | **1988–2007** |
| **No. of samples** | **9** | **2** | **21** | **9** | **2** | **21** |
| $^{40}$K | 36–120 | 60–100 | 25–95 | 49–120 | 45–65 | 32–85 |
| $^{54}$Mn | < | < | 0.36–3.0 (3) | 1.5–21 (6) | 33–62 (2) | 0.37–77 (14) |
| $^{58}$Co | < | < | 1.4–21 (2) | 2.4–3.6 (2) | 3.3 (1) | 0.50–3.9 (6) |
| $^{60}$Co | 3.9–4.1 (2) | 3.8–5.3 (2) | 0.46–6.5 (18) | 3.8–45 (5) | 37–50 (2) | 0.54–69 (20) |
| $^{89}$Sr | < | 30 (1) | 3.4 (1) | < | 24 (1) | < |
| $^{90}$Sr | 20–41 | 25–28 | 10–34 | 19–28 | 17–23 | 17–26 |
| $^{103}$Ru | < | 46 (1) | < | < | 57 (1) | < |
| $^{106}$Ru | < | 36 (2) | < | < | 70 (1) | < |
| $^{110m}$Ag | 0.77–6.6 (2) | 51–100 (2) | 4.0–18 (3) | < | 2.0–8.7 (2) | < |
| $^{125}$Sb | < | 4.3 (1) | < | < | 5.2 (1) | 2.0–2.1 (2) |
| $^{129m}$Te | < | 200 (1) | < | < | 150 (1) | < |
| $^{134}$Cs | < | 17–21 (2) | 0.54–7.5 (8) | < | 5.9–11 (2) | 1.0–3.1 (4) |
| $^{137}$Cs | 1.1–4.1 | 37–52 | 7.9–30 | 1.6–1.7 | 14–28 | 1.8–13 |
| $^{144}$Ce | < | 22 (1) | < | < | < | < |
| $^{239,240}$Pu | 0.030–0.074 | | | 0.017–0.041 | | |
| $^{241}$Am | | | | 0.019 | | |

< = below the detection limit

## 3.6 Radioecological special studies

### 3.6.1 *Fucus* surveys in the sea areas off Loviisa and Olkiluoto and along the Finnish coasts

The bladder-wrack, *Fucus vesiculosus,* has proved to be an excellent indicator of radioactive substances in the marine environment. It has been used successfully nearly worldwide for this purpose, and it is also the most common seaweed and most important constituent in the coastal ecosystems along almost the whole Finnish coast. Thus, a large number of special studies on radioactivity in *Fucus* have been carried out to map the appearance of fallout-nuclides in the coastal waters and the dispersion of discharge nuclides in the vicinities of the nuclear power plants. Many of these studies have been carried out as so-called "Pan Scandinavian *Fucus* Projects" sponsored by the NKS (Nordic Nuclear Safety

Research), i.e., similar surveys have been carried out in each Nordic country so that all the Nordic coasts have been covered.

In Finland, the first *Fucus* survey was made in 1980 along the Finnish coast of the Gulf of Finland and in the sea area surrounding the Loviisa NPP (Ilus et al. 1981). In 1981, a similar survey was made in the Archipelago Sea and Gulf of Bothnia including the sea area surrounding the Olkiluoto NPP (Ilus et al. 1983). In 1987, a *Fucus* survey was carried out along the whole Finnish coast and in the sea areas of Loviisa and Olkiluoto (Ilus et al. 1988). In 1991, an extensive survey was carried out in Finland as part of a joint Nordic *Fucus* project, in which all the Nordic coasts including Icelandic waters were monitored (Carlson et al. 1992). In 1995 and 1999, additional *Fucus* Projects were implemented at Loviisa and Olkiluoto and along the Finnish coast. The latter was part of an NKS Project entitled 'Radioactive tracers in Nordic Sea areas' and the aim was to determine whether Sellafield-originated $^{99}$Tc can be detected in Finnish coastal waters (Ilus et al. 2002b). The technetium data were further complemented with additional *Fucus* samplings in 2003 (Ilus et al. 2006b). All the *Fucus* samples were taken by scuba diving, the minimum sample size being 1.8 kg.

**Loviisa**

Activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{110m}$Ag and $^{137}$Cs in the *Fucus* samples collected from different sampling sites in the *Fucus* surveys at Loviisa are given in Table 26, and the location of the sampling sites in Fig. 88. In 1980, the concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co and $^{110m}$Ag were clearly higher in Hästholmsfjärden Bay than outside it, where the concentrations decreased rapidly in accordance with the increasing distance (Fig. 89). The maximum concentrations – 83, 47, 27 and 16 Bq kg$^{-1}$ d.w. of $^{110m}$Ag, $^{60}$Co, $^{54}$Mn and $^{58}$Co, respectively – were detected right in front of the cooling water outlet of the power plant (distance 200m). The farthest observation of $^{110m}$Ag was made at a distance of 4.4 km. These four nuclides were not detected in the samples collected elsewhere in the Gulf of Finland. On the other hand, $^{137}$Cs was detected in all samples. The concentrations of it were 6.7 – 15 Bq kg$^{-1}$ in the samples from Hästholmsfjärden, 6.9 – 11 Bq kg$^{-1}$ in the Loviisa area outside Hästholmsfjärden Bay and 7.4 – 11 Bq kg$^{-1}$ elsewhere in the Gulf of Finland (Ilus et al. 1981).

In 1987, the distribution pattern of $^{60}$Co was quite similar, except that the concentrations in Hästholmsfjärden were lower and observations were made over a little larger area (to a distance of 6.2 km). $^{60}$Co was not detected elsewhere in the Gulf of Finland. The concentrations of $^{110m}$Ag and $^{54}$Mn had decreased even more clearly, although the maximum values were still found at the sampling site nearest the cooling water outlet. Chernobyl-derived $^{54}$Mn was detected at

five sampling sites in the eastern Gulf of Finland and $^{110m}$Ag in all samples from the Gulf (maximum values in Haapasaari 1.4 and 31 Bq kg$^{-1}$, respectively). As a consequence of the Chernobyl accident, the activity concentrations of $^{137}$Cs were 570–670 Bq kg$^{-1}$ in Hästholmsfjärden, decreasing towards the open sea (Fig. 90). Outside Hästholmsfjärden, the concentrations varied between 370 and 590 Bq kg$^{-1}$. $^{137}$Cs was detected in all samples from the Gulf of Finland; the maximum in these more open-sea sites was 440 Bq kg$^{-1}$ in Ulkotammio (Ilus et al. 1988). In addition, the following nuclides were observed in the samples from the Gulf of Finland and Loviisa: $^{40}$K, $^{65}$Zn, $^{90}$Sr, $^{106}$Ru, $^{125}$Sb, $^{134}$Cs, $^{238}$Pu and $^{239,240}$Pu, and in four samples from Loviisa, also $^{144}$Ce.

In 1991, a strong decreasing trend became even more visible in the concentrations of the discharge nuclides considered before, and the concentrations of $^{137}$Cs had also started to decrease significantly. The $^{60}$Co concentrations were 6.6–6.9 Bq kg$^{-1}$ in Hästholmsfjärden, but small traces of it were detected up to 9.8 km from the power plant (Boistö). The distribution of $^{110m}$Ag and $^{54}$Mn had markedly diminished, the maximum values in front of the cooling water outlet being 3.3 and 1.2 Bq kg$^{-1}$, respectively. None of these nuclides were detected elsewhere in the Gulf of Finland. The highest $^{137}$Cs concentration was 124 Bq kg$^{-1}$, again at the nearest sampling site to the cooling water outlet; the concentrations varied between 103 and 124 Bq kg$^{-1}$ in Hästholmsfjärden and between 75 and 100 Bq kg$^{-1}$ outside it. $^{137}$Cs was detected in all *Fucus* samples taken along the coast of the Gulf of Finland, the highest concentration at the other sampling sites outside the Loviisa area being 87 Bq kg$^{-1}$.

In 1995, the distribution pattern and the concentrations of $^{60}$Co were quite similar to those in 1991.The concentrations in Hästholmsfjärden were 2.6–5.6 Bq kg$^{-1}$, and small traces of it were still detected as far away as Boistö (Site 19). $^{110m}$Ag was not detected at any sampling site, and small amounts of $^{54}$Mn and $^{58}$Co were observed only at the site nearest to the cooling water outlet. The concentrations of $^{137}$Cs varied between 56 and 85 Bq kg$^{-1}$ in Hästholmsfjärden (the highest value again in front of the outlet), and between 54 and 97 Bq kg$^{-1}$ outside it. Outside the Loviisa area, $^{137}$Cs was detected at all the sampling sites in the Gulf of Finland, the highest concentration being 64 Bq kg$^{-1}$ at Haapasaari.

In 1999, the $^{60}$Co concentrations in *Fucus* had further decreased; $^{60}$Co was detected at all the sampling sites in Hästholmsfjärden and at three sites outside it, the highest value being 2.6 Bq kg$^{-1}$ in front of the cooling water outlet (Fig. 91). Trace amounts of $^{54}$Mn were detected at four sampling sites in Hästholmsfjärden. $^{137}$Cs was detected at all the sampling sites off Loviisa and elsewhere in the Gulf of Finland, the maximum values being 52 Bq kg$^{-1}$ off Loviisa (Fig. 92) and 58 Bq kg$^{-1}$ at the other sites in the Gulf (Helsinki, Katajaluoto).

**Table 26.** Activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{110m}$Ag and $^{137}$Cs (Bq kg$^{-1}$ d.w.) in *Fucus vesiculosus* collected from different sampling sites in the special *Fucus* Projects carried out at Loviisa in 1980, 1987, 1991, 1995 and 1999 (– = not sampled, < = below the detection limit).

| Nuclide Year | Sampling Site, distance from the outlet (km) and direction | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1(A) | 2 | 3 | 4(B) | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14(C) | 15(F) | 16(G) | 17 | 18 | 19(D) | 20(E) | 21 |
| | 0.2 E | 1.0 S | 1.0 N | 2.0 E | 2.1 N | 2.2 NE | 2.8 NE | 2.0 S | 2.0 S | 3.2 W | 4.0 SW | 3.2 S | 3.9 S | 4.4 S | 6.2 S | 5.7 W | 5.1 NW | 8.1 W | 9.8 SE | 10.3 S | 11.9 S |
| $^{54}$Mn | | | | | | | | | | | | | | | | | | | | | |
| 1980 | 27 | 20 | 15 | 20 | 22 | 17 | – | 3.7 | 1.0 | 3.4 | < | 1.5 | < | 0.88 | < | – | – | – | < | < | – |
| 1987 | 6.0 | – | – | 3.3 | 5.2 | – | – | 3.2 | – | – | 1.2 | 1.7 | – | 1.1 | 0.8 | 1.6 | 2.4 | – | 1.0 | 1.2 | 1.1 |
| 1991 | 1.2 | – | – | 0.53 | < | – | – | < | – | – | < | 0.26 | – | < | 0.24 | < | < | – | < | < | < |
| 1995 | 0.98 | – | < | < | < | – | < | < | – | < | < | < | – | < | < | < | < | < | < | < | – |
| 1999 | 0.90 | – | < | 0.40 | 0.30 | – | 0.40 | < | – | < | < | < | – | < | < | < | < | < | < | < | < |
| $^{58}$Co | | | | | | | | | | | | | | | | | | | | | |
| 1980 | 16 | 16 | < | 3.7 | 3.8 | < | – | < | < | < | < | < | < | < | < | – | – | – | < | < | – |
| 1995 | 0.77 | – | < | < | < | – | < | < | – | < | < | < | – | < | < | < | < | < | < | < | – |
| $^{60}$Co | | | | | | | | | | | | | | | | | | | | | |
| 1980 | 47 | 28 | 32 | 23 | 39 | 30 | – | 6.5 | 1.8 | 4.2 | < | 1.8 | < | < | < | – | – | – | < | < | – |
| 1987 | 32 | – | – | 18 | 26 | – | – | 13 | – | – | 1.8 | 3.1 | – | 2.5 | 2.1 | 2.3 | 2.8 | – | < | < | < |
| 1991 | 6.9 | – | – | 5.1 | 6.6 | – | – | 3.3 | – | – | 0.6 | 1.3 | – | 1.3 | 0.6 | 0.9 | 1.2 | – | 0.4 | < | < |
| 1995 | 5.6 | – | 3.6 | 2.6 | 4.3 | – | 4.1 | 1.8 | – | 0.65 | < | 0.97 | – | 0.62 | < | 0.67 | 0.58 | < | 0.28 | < | – |
| 1999 | 2.6 | — | 1.7 | 1.7 | 1.4 | – | 1.6 | 0.6 | – | 0.30 | 0.10 | < | – | < | < | 0.30 | < | < | < | < | < |
| $^{110m}$Ag | | | | | | | | | | | | | | | | | | | | | |
| 1980 | 83 | 81 | 42 | 34 | 35 | 30 | – | 16 | 4.6 | 4.9 | 3.2 | 3.2 | 3.9 | 2.4 | < | – | – | – | < | < | – |
| 1987 | 22 | – | – | 16 | 15 | – | – | 15 | – | – | 14 | 12 | – | 18 | 19 | 11 | 16 | – | 29 | 19 | 22 |
| 1991 | 2.8 | – | – | 1.4 | < | – | – | 1.5 | – | – | < | 0.6 | – | < | < | < | < | – | < | < | < |
| 1999 | 0.30 | – | < | < | < | – | < | < | – | < | < | < | – | < | < | < | < | < | < | < | < |
| $^{137}$Cs | | | | | | | | | | | | | | | | | | | | | |
| 1980 | 13 | 6.7 | 13 | 12 | 12 | 15 | – | 11 | 8.8 | 8.8 | 8.7 | 9.1 | 8.9 | 7.8 | 9.4 | – | – | – | 7.8 | 6.9 | – |
| 1987 | 670 | – | – | 570 | 610 | – | – | 590 | – | – | 500 | 480 | – | 570 | 420 | 480 | 500 | – | 470 | 370 | 410 |
| 1991 | 120 | – | – | 100 | 110 | – | – | 93 | – | – | 86 | 87 | – | 84 | 92 | 75 | 89 | – | 100 | 79 | 81 |
| 1995 | 85 | – | 78 | 56 | 83 | – | 77 | 73 | – | 61 | 97 | 79 | – | 63 | 75 | 76 | 70 | 79 | 54 | 56 | – |
| 1999 | 52 | – | 45 | 43 | 44 | – | 39 | 41 | – | 41 | 41 | 43 | – | 40 | 40 | 44 | 49 | 45 | 46 | 34 | 37 |

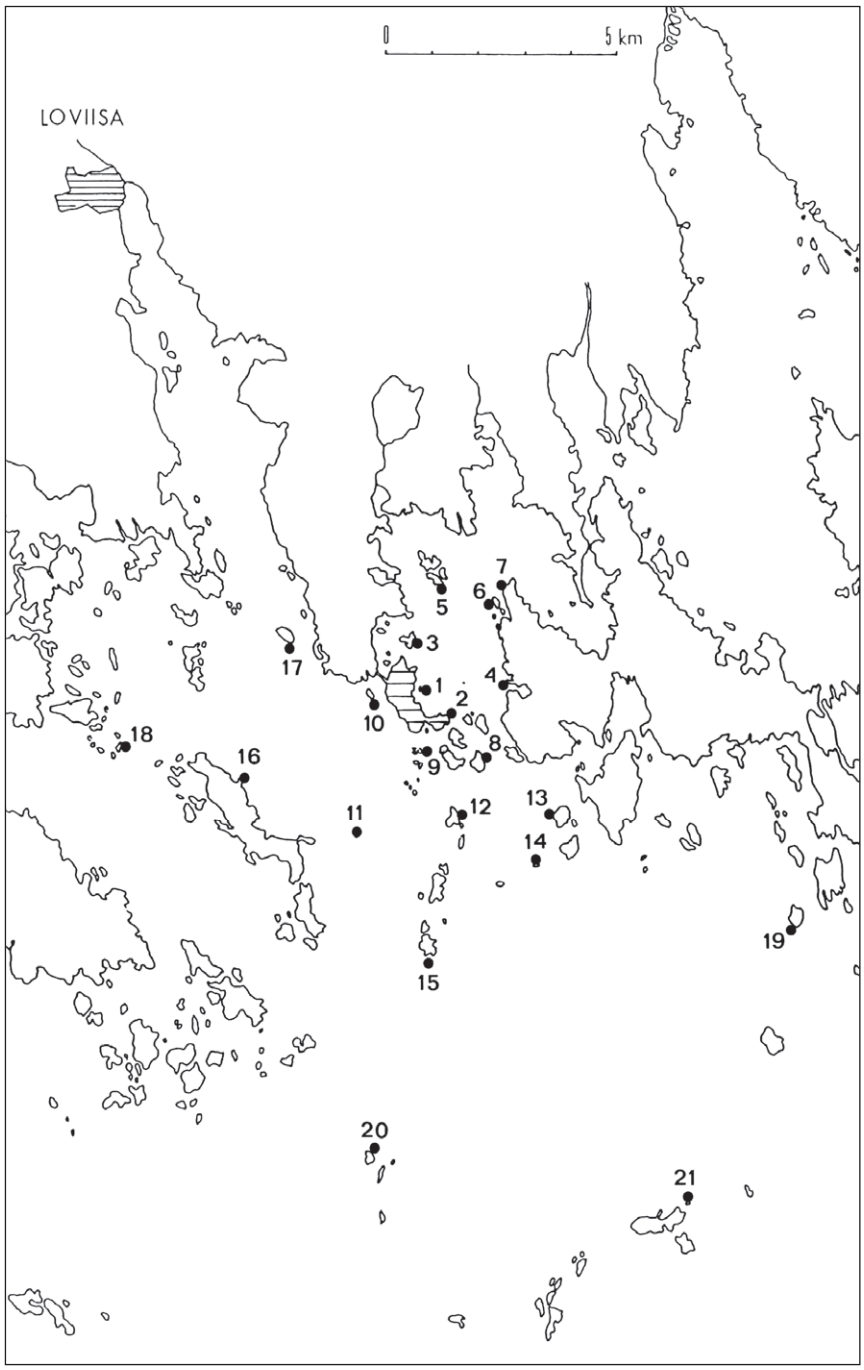


**Fig. 88.** The sampling sites in the *Fucus* surveys at Loviisa.

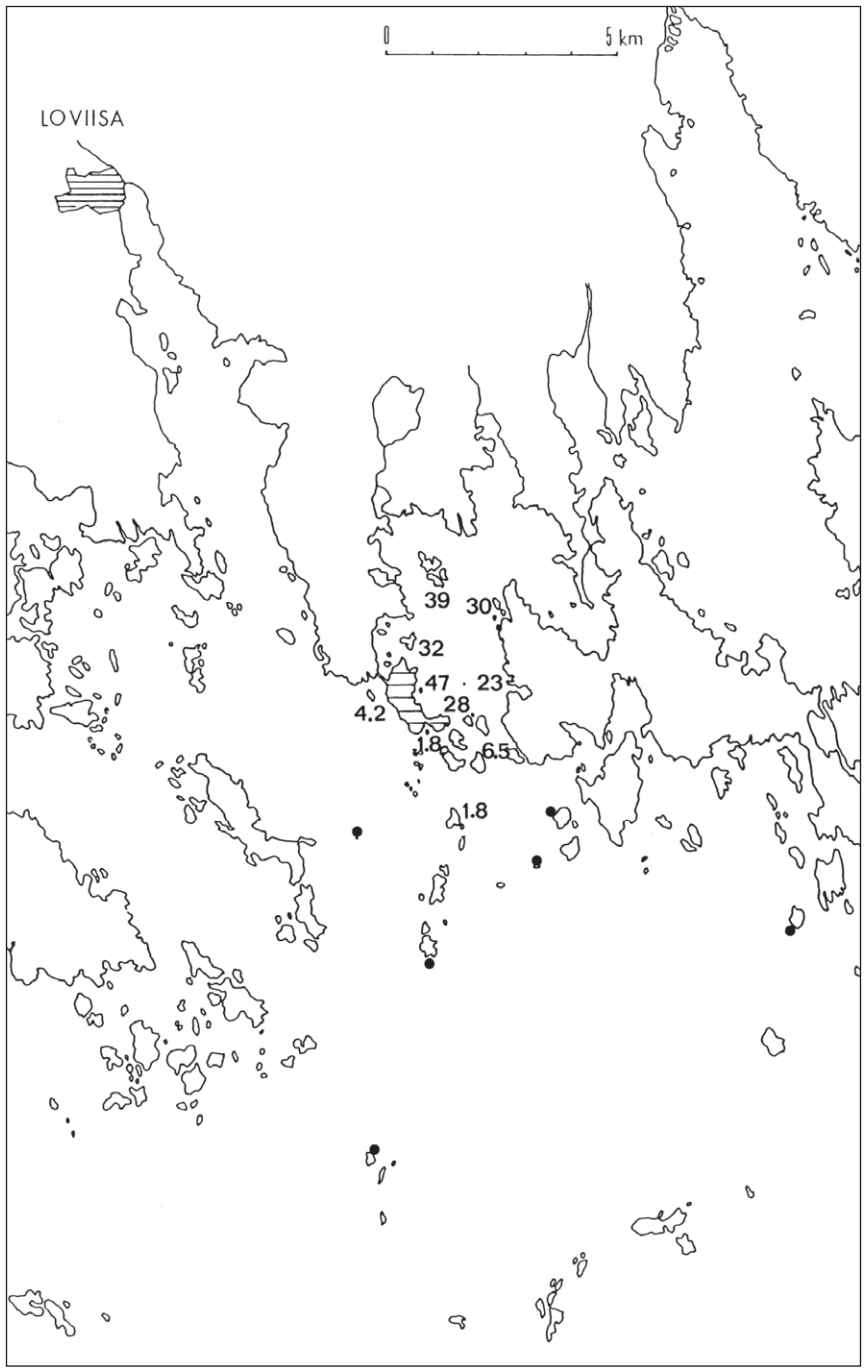


**Fig. 89.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites off Loviisa in the 1980 survey. A dot without a number means that the concentration was below the detection limit of 1 Bq $kg^{-1}$ d.w.

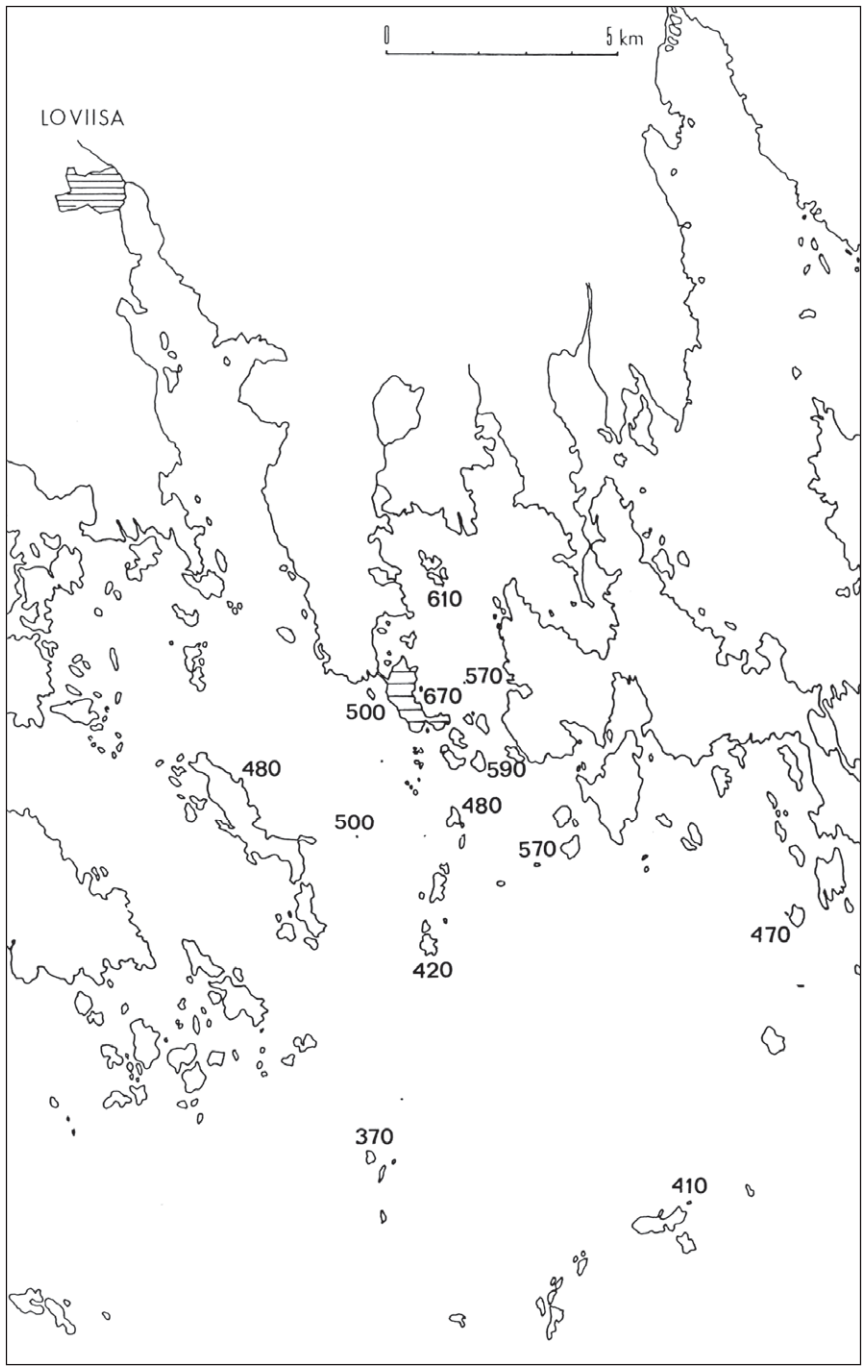


**Fig. 90.** Activity concentrations of $^{137}Cs$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites off Loviisa in the 1987 survey.

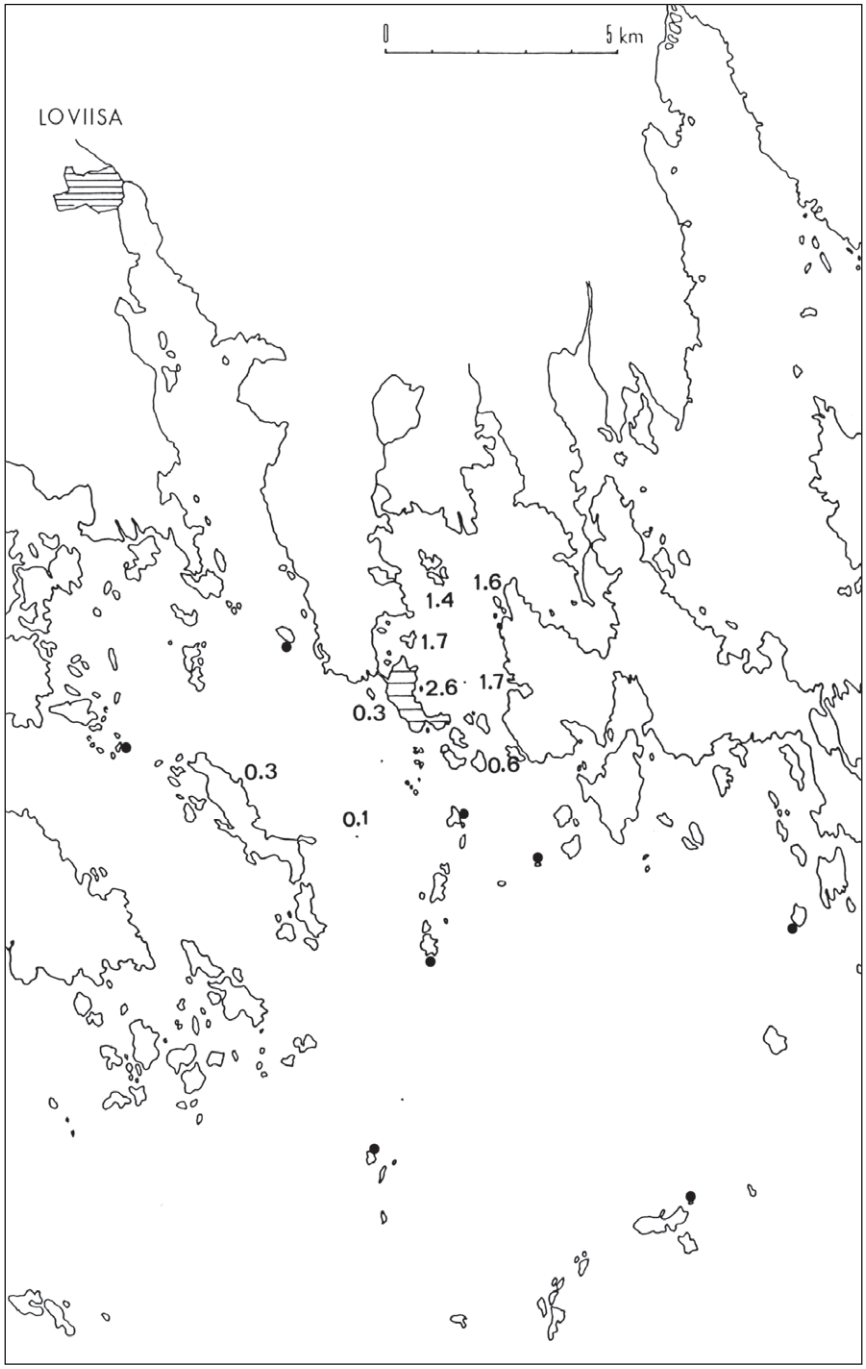


**Fig. 91.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites off Loviisa in the 1999 survey. A dot without a number means that the concentration was below the detection limit of 0.1 Bq $kg^{-1}$ d.w.

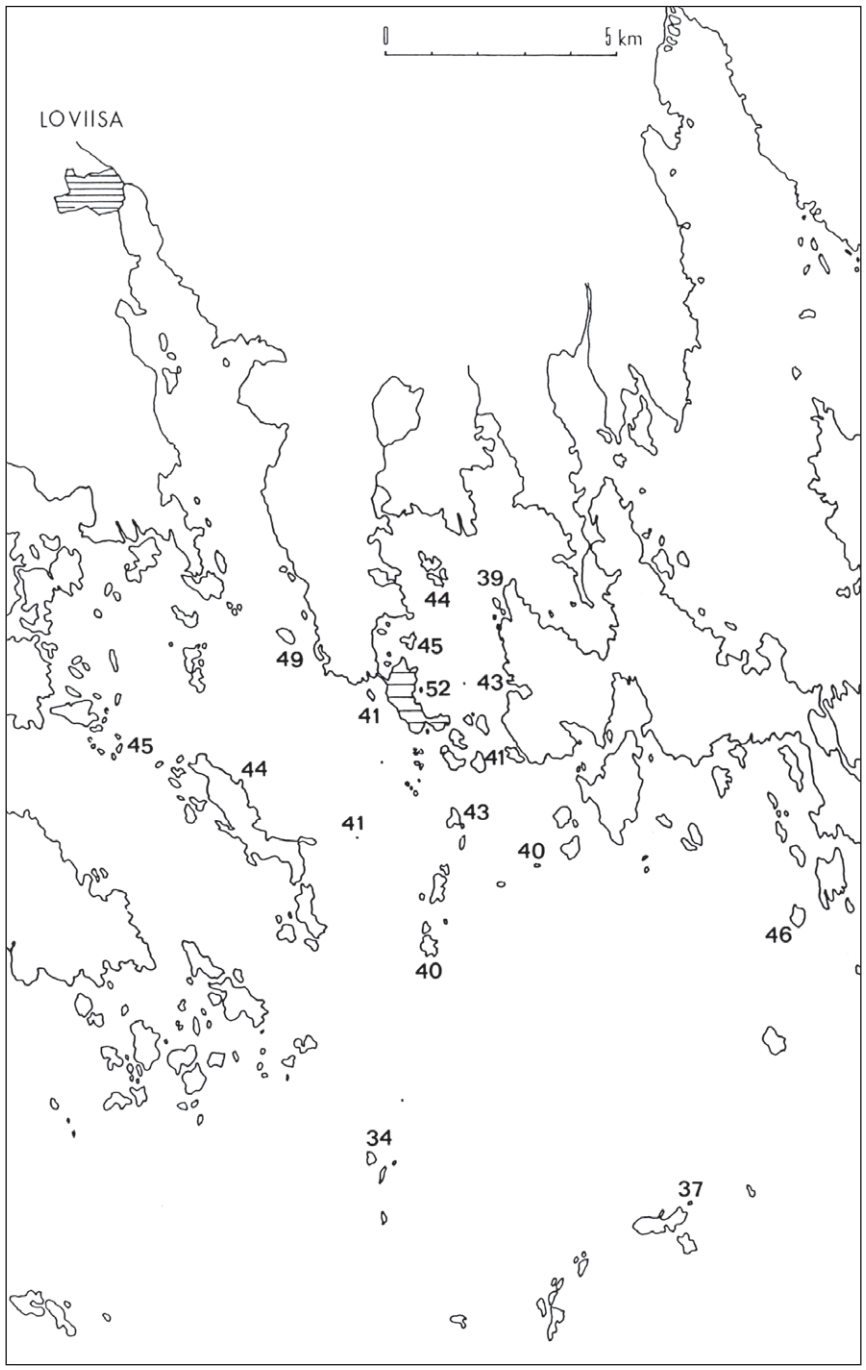


**Fig. 92.** Activity concentrations of $^{137}Cs$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites off Loviisa in the 1999 survey.

**Olkiluoto**

Activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{65}$Zn and $^{137}$Cs in the *Fucus* samples collected from different sampling sites in the *Fucus* surveys at Olkiluoto are given in Table 27, and the location of the sampling sites in Fig. 93. The distribution patterns of $^{60}$Co and $^{137}$Cs in *Fucus* in the Olkiluoto area and along the Finnish west coast are shown in Figs. 94–100. The results from the first survey in 1981 indicated a widespread dispersion of the local discharge nuclides $^{54}$Mn, $^{60}$Co and $^{58}$Co in *Fucus*, although the concentrations were lower than at Loviisa, even at the sampling sites located nearest to the cooling water outlet. The highest concentrations of $^{54}$Mn, $^{60}$Co and $^{58}$Co in Iso Kaalonpuhti Bay, at a distance of 400 m from the outlet, were 20, 9.6 and 6.7 Bq kg$^{-1}$, respectively. However, trace amounts of $^{54}$Mn and $^{60}$Co were detected not only at the northernmost Sampling Site 19 (Iso Pietari) in the Olkiluoto area, but also in the sample taken from Säppi situated 26 km north of Olkiluoto. The concentrations of $^{54}$Mn and $^{60}$Co at Säppi were 0.69 and 0.27 Bq kg$^{-1}$, respectively. $^{137}$Cs was detected in all the samples from the Olkiluoto area and the whole west coast; concentrations were between 8.0 and 10 Bq kg$^{-1}$ in the Olkiluoto area, and 5.4–10 Bq kg$^{-1}$ elsewhere in the Gulf of Bothnia (Ilus et al. 1983). In addition, small amounts of $^{65}$Zn were detected in three samples and $^{110m}$Ag in two samples at Olkiluoto. $^{239,240}$Pu from weapons-tests fallout was detected in all the samples, and $^{238}$Pu in five samples at Olkiluoto.

In 1987, $^{54}$Mn was detected not only in all samples taken from the Olkiluoto area, but also in all seven samples taken along the east coast of the Bothnian Sea and in four samples from the Åland Islands. The concentrations were 85 Bq kg$^{-1}$ at Sampling Site 2 in Iso Kaalonpuhti Bay (distance 700 m) and 1.4 and 1.7 Bq kg$^{-1}$, respectively, at the northernmost and southernmost sampling sites 19 and 22 (Iso Pietari and Kylmäpihlaja) of the Olkiluoto area (distance 12 and 13.2 km). The highest $^{54}$Mn concentrations in the sampling sites on the west coast of Finland and in Åland were 3.0 Bq kg$^{-1}$ (Ilus et al. 1988). As well as in all samples from the Olkiluoto area, $^{60}$Co was also detected in three samples from the west coast and two samples from Åland (Fig. 98). The highest concentration was 62 Bq kg$^{-1}$ in the sampling site of Iso Kaalonpuhti Bay, while at Sampling Sites 19 and 22 the concentrations were 1.7 and 3.0 Bq kg$^{-1}$, respectively. The highest $^{60}$Co concentration outside the Olkiluoto area was 3.8 Bq kg$^{-1}$ in Eckerö, Åland Islands. In the neighbouring area of Olkiluoto, the farthest observations were from Pujo, Pyhämaa in the south (1.4 Bq kg$^{-1}$, distance 29 km) and from Krookanlahti, Merikarvia in the north (1.0 Bq kg$^{-1}$, distance 68 km).

As a consequence of the Chernobyl accident, the activity concentrations of $^{137}$Cs were 220–230 Bq kg$^{-1}$ at Sampling Sites 2 and 6 off Olkiluoto in 1987, and decreased when going farther from the power plant (Fig. 94). $^{137}$Cs was

detected in all samples from the Bothnian Sea, Archipelago Sea and Åland Islands, the maximum being 340 Bq kg$^{-1}$ in the sample from Kristiinankaupunki (Ilus et al. 1988). Besides the elements considered before, the following nuclides were also observed in the samples from all the sea areas mentioned above and from Olkiluoto: $^{40}$K, $^{65}$Zn, $^{90}$Sr, $^{106}$Ru, $^{110m}$Ag, $^{125}$Sb, $^{134}$Cs, $^{238}$Pu and $^{239,240}$Pu, and in addition, $^{144}$Ce in four and $^{51}$Cr in one sample at Olkiluoto. The highest concentration of $^{65}$Zn at Olkiluoto was 8.4 Bq kg$^{-1}$ at the sampling site of Iso Kaalonpuhti Bay, but similar values of $^{65}$Zn also occurred in samples from Eckerö, Åland and Norrskär, the Quark (Ilus et al. *op. cit.*).

In 1991, the activity concentrations and distribution pattern of $^{54}$Mn and $^{58}$Co in *Fucus* were very similar to those of $^{54}$Mn in 1987 and $^{58}$Co in 1981. Trace amounts of $^{54}$Mn were also detected in three sampling sites in the Bothnian Sea. The $^{60}$Co concentrations in the Olkiluoto area were in general a little higher than in 1987. The maximum value in Iso Kaalonpuhti Bay was 138 Bq kg$^{-1}$ and the minimum 4.1 Bq kg$^{-1}$ at Iso Pietari (Fig. 95). Outside the Olkiluoto area, small amounts of $^{60}$Co were observed in five samples taken from the west coast and four samples taken from the Åland Islands (Fig. 99). In the samples from the west coast (excluding the Åland Islands) the farthest observations were from Isokari, Kustavi in the south (0.2 Bq kg$^{-1}$, distance 65 km) and from Sandskär, Kristiinankaupunki in the north (1.1 Bq kg$^{-1}$, distance 100 km). $^{137}$Cs was detected in all samples from the Bothnian Sea, the Archipelago Sea and the Åland Islands, the maximum being 170 Bq kg$^{-1}$ at the northernmost sampling site, Norrskär. In the Olkiluoto area, the highest concentrations of $^{137}$Cs were 130–140 Bq kg$^{-1}$, again at Sampling Sites 2 and 6, the values decreasing as the distance from the cooling water outlet increased. In addition, $^{65}$Zn was detected at three sites (maximum 1.8 Bq kg$^{-1}$) and $^{125}$Sb at four sites (maximum 1.5 Bq kg$^{-1}$) in the Olkiluoto area.

In 1995, trace amounts of $^{58}$Co were found more widely than earlier, as far north as Sampling Site 19 (Table 27). On the other hand, a clear decrease was visible in the concentrations of $^{54}$Mn and $^{60}$Co compared to the values from 1987 and 1991, their highest values being 22 and 40 Bq kg$^{-1}$, respectively, in Iso Kaalonpuhti Bay. $^{54}$Mn was not found outside the Olkiluoto area, but traces of $^{60}$Co were again observed in five samples taken from the west coast and two samples taken from Åland (Fig. 100). The farthest observations of $^{60}$Co of Olkiluoto origin (both 0.2 Bq kg$^{-1}$) were again from Isokari, Kustavi in the south, but from Nämpnäs, Närpiö in the north (distance 137 km). In the western Åland Islands, the detected activity concentrations of $^{60}$Co were 0.7 and 0.4 Bq kg$^{-1}$, but most probably these originated from the Forsmark power plant situated only about 75 km west of Åland (see below). $^{137}$Cs was detected in all samples from the Bothnian Sea, Archipelago Sea and Åland Islands, the maximum being

83 Bq kg$^{-1}$. In the Olkiluoto area, the highest concentration was 79 Bq kg$^{-1}$ at Sampling Site 14. In addition to the aforementioned nuclides, all the samples contained naturally-occurring $^{40}$K and Chernobyl-derived $^{134}$Cs.

In 1999, a strong decrease was obvious in the concentrations of the discharge nuclides $^{54}$Mn, $^{58}$Co and $^{60}$Co in *Fucus* from the Olkiluoto area. Only trace amounts of $^{54}$Mn and $^{58}$Co were detected in just five sampling sites nearest to the cooling water outlet, and these nuclides were not detected outside the Olkiluoto area. The highest concentrations at the sampling site in Iso Kaalonpuhti Bay were 0.8 and 1.2 Bq kg$^{-1}$ for $^{54}$Mn and $^{58}$Co, respectively. $^{60}$Co was still detected at all sampling sites in the Olkiluoto area (Fig. 96), but the concentrations had decreased to 5.6 Bq kg$^{-1}$ in Iso Kaalonpuhti Bay and to 0.7 and 0.6 Bq kg$^{-1}$ at Sampling Sites 19 and 22. Outside the Olkiluoto area, traces of $^{60}$Co were still detected in three samples from the west coast (Kristiinankaupunki 0.1 Bq kg$^{-1}$, Säppi 0.4 Bq kg$^{-1}$ and Pyhämaa 0.2 Bq kg$^{-1}$) and in one sample from the Åland Islands (Eckerö 0.6 Bq kg$^{-1}$). $^{137}$Cs was detected in all samples from the Bothnian Sea, Archipelago Sea and Åland Islands, the maximum being 53 Bq kg$^{-1}$. In the Olkiluoto area, the highest concentration was 62 Bq kg$^{-1}$ at Sampling Site 2 (Fig. 97). In addition to the aforementioned nuclides, all the samples contained only naturally-occurring $^{7}$Be and $^{40}$K and Chernobyl-derived $^{134}$Cs.

In the 2003 *Fucus* survey, $^{60}$Co was no longer detected outside the Olkiluoto area. The $^{99}$Tc activities in *Fucus* from the Olkiluoto area were 2.2–4.2 Bq kg$^{-1}$ in 1999 and 2.5 Bq kg$^{-1}$ in 2003, being typical values for Finnish coastal waters (Ilus et al. 2002b and 2006b).

In the 1980s, analogous results were published from the west coast of Sweden. Trace amounts of $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{65}$Zn and $^{110m}$Ag originating from the Barsebäck and Ringhals NPPs were detected in *Fucus* collected from the Sound, Kattegat and Skagerrak at distances of up to 150 km from the power plants (e.g. Mattsson et al. 1980a and b, Nilsson et al. 1980, Nilsson 1981). The activity concentrations in *Fucus* reflected very well the discharge rate of radionuclides from the power plants, and the distribution of activation products along the west coast of Sweden could be described with a power function. The activity concentration C (z) at a distance of z from the outlet of the power plant followed the model:

$$C(z) = \alpha \cdot z^{-\beta},$$

where z is the distance in kilometres and α is a constant that varies with the amount of activity that is discharged. The constant β reveals the mixing of the discharge into the water mass, and was found to be around 1.4 for all the nuclides mentioned above, except $^{110m}$Ag. Zakaria et al. (2008) repeated the study at Barsebäck in 2002–2003. Although the present-day concentrations were

considerably lower than those in the earlier studies, the decrease in activity concentration with distance from the power plant could be described by the same power function, with an exponent ranging from 1.4 to 2.4.

The power function was applied to the distribution pattern of $^{60}$Co on the Finnish west coast (Figs. 101–103). The exponent 1.4 also fitted the results from Olkiluoto very well, especially at shorter distances, and the decrease in activity concentrations was fairly well in accordance with the calculated model lines shown in the Figures. In a southward direction (Fig. 103), the sampling sites in Åland seemed to diverge from the model, which indicates that most probably the source of $^{60}$Co there was not Olkiluoto, but Forsmark, which is located on the Swedish east coast at a distance of 70 km, and is thus closer to Åland than Olkiluoto. This conclusion is supported by the fact that the currents on the east coast of Sweden are directed to the south, and that $^{60}$Co discharges from Forsmark have generally been larger than those from Olkiluoto. In the northward direction, the decrease in the concentrations followed the calculated model line quite nicely, except for the diverging result from Kristiinankaupunki (Skaftung in the Figure) in 1991 (Fig. 101).

**In conclusion**

Although the residues of discharge nuclides from the Olkiluoto NPP could be detected at relatively long distances from the power plant, their relevance regarding to environmental risk is insignificant. The minimal concentrations found in *Fucus* are much better regarded as interesting curiosities that highlight the accuracy of the analysis methods, the usability of *Fucus* as an indicator organism, and the potential of small amounts of radionuclides in tracer studies. The tiny amounts of the relatively short-lived radionuclides (for instance the physical half-life of $^{60}$Co is 5.3 years) are far below any risk level (see Chapter 7). The spreading of $^{60}$Co and some other radionuclides in detectable quantities merely to the Finnish coast of the Bothnian Sea and not to that of the Gulf of Finland, was partly due to the higher discharges of these nuclides from Olkiluoto than from Loviisa, but maybe more significantly due to the free exchange of water and better diffusion capability of discharges from Olkiluoto.

**Table 27.** Activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{65}$Zn and $^{137}$Cs (Bq kg$^{-1}$ d.w.) in *Fucus vesiculosus* collected from different sampling sites in the special *Fucus* Projects carried out at Olkiluoto in 1981, 1987, 1991, 1995 and 1999 (– = not sampled, < = below the detection limit).

| Nuclide Year | Sampling Site, distance from the outlet (km) and direction | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2(A) | 3 | 4 | 5 | 6(B) | 7 | 8 | 9 | 10 | 11(D) | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19(C) | 20(E) | 21 | 22 |
| | 0.4 SW | 0.7 W | 1.4 SW | 1.5 NW | 1.9 W | 2.3 W | 3.4 W | 3.5 SW | 4.1 W | 4.8 W | 5.3 W | 5.5 W | 2.5 NW | 2.6 N | 4.0 N | 4.6 N | 7.3 N | 10.5 N | 12.0 N | 6.8 SW | 9.3 SW | 13.2 SW |
| $^{54}$Mn | | | | | | | | | | | | | | | | | | | | | | |
| 1981 | 20 | 8.0 | 2.8 | 6.1 | 5.1 | 2.5 | 1.4 | 1.6 | 1.7 | 0.76 | – | 1.1 | 2.0 | – | 2.1 | – | – | – | 0.55 | 1.3 | – | < |
| 1987 | – | 85 | – | – | – | 17 | – | 16 | 9.8 | 4.3 | – | 3.3 | 7.7 | – | 5.6 | 4.5 | 2.4 | – | 1.4 | 6.0 | – | 1.7 |
| 1991 | – | 65 | – | – | – | 21 | – | 16 | 12 | 3.8 | – | 2.8 | – | 11 | 5.0 | 5.7 | 3.1 | – | 1.4 | 6.3 | – | 1.7 |
| 1995 | – | 22 | – | – | – | 10 | – | 6.0 | 4.0 | 2.6 | 2.4 | – | 13 | 19 | 9.4 | 4.8 | 1.8 | 1.7 | 1.1 | 0.64 | < | < |
| 1999 | – | 0.80 | – | – | – | 0.60 | – | 0.50 | < | < | < | < | < | 0.40 | < | 0.30 | < | < | < | < | < | < |
| $^{58}$Co | | | | | | | | | | | | | | | | | | | | | | |
| 1981 | 6.7 | 2.7 | 0.62 | 2.1 | 1.1 | < | 0.32 | < | < | < | – | < | 0.32 | – | < | – | – | – | < | 0.44 | – | < |
| 1987 | – | 5.8 | – | – | – | 1.2 | – | < | < | < | – | < | < | – | < | < | < | – | < | < | – | < |
| 1991 | – | 4.7 | – | – | – | 2.3 | – | 1.8 | 1.3 | < | – | < | – | 1.0 | < | < | < | – | < | 0.90 | – | < |
| 1995 | – | 4.2 | – | – | – | 1.5 | – | 1.4 | 1.2 | 0.83 | 0.38 | – | 3.5 | 5.0 | 3.2 | 2.5 | 2.4 | 1.7 | 0.42 | < | < | < |
| 1999 | – | 1.2 | – | – | – | 0.40 | – | 0.50 | 0.50 | < | < | < | < | 0.20 | < | < | < | < | < | < | < | < |
| $^{60}$Co | | | | | | | | | | | | | | | | | | | | | | |
| 1981 | 9.6 | 6.5 | 3.5 | 6.1 | 3.5 | 3.7 | 3.1 | 2.6 | 2.2 | 0.78 | – | 0.84 | 3.6 | – | 2.1 | – | – | – | 1.3 | 1.1 | – | < |
| 1987 | – | 62 | – | – | – | 22 | – | 24 | 15 | 5.9 | – | 4.5 | 14 | – | 6.5 | 9.7 | 5.9 | – | 1.7 | 8.3 | – | 3.0 |
| 1991 | – | 134 | – | – | – | 40 | – | 44 | 28 | 9.7 | – | 9.3 | – | 27 | 14 | 18 | 17 | – | 3.6 | 18 | – | 5.0 |
| 1995 | – | 40 | – | – | – | 12 | – | 9.4 | 5.8 | 3.8 | 3.6 | – | 12 | 15 | 8.1 | 6.7 | 6.8 | 4.9 | 5.6 | 2.3 | 1.5 | 1.3 |
| 1999 | – | 5.6 | – | – | – | 3.4 | – | 4.2 | 2.1 | 0.90 | 0.80 | 1.0 | 2.4 | 3.3 | 1.2 | 2.4 | 1.4 | 0.60 | 0.70 | 1.8 | 0.90 | 0.60 |
| $^{65}$Zn | | | | | | | | | | | | | | | | | | | | | | |
| 1981 | 1.6 | 0.45 | < | 2.3 | < | < | < | < | < | < | – | < | < | – | < | – | – | – | < | < | – | < |
| 1987 | – | 8.4 | – | – | – | 3.6 | – | 4.2 | 4.2 | 4.7 | – | 3.9 | 2.7 | – | 2.8 | 2.3 | 1.4 | – | 1.8 | 4.1 | – | 2.9 |
| 1991 | – | 3.8 | – | – | – | < | – | 1.8 | 1.0 | < | – | < | – | < | < | < | < | – | < | < | – | < |
| $^{137}$Cs | | | | | | | | | | | | | | | | | | | | | | |
| 1981 | 10 | 8.5 | 10 | 8.5 | 9.8 | 8.6 | 8.7 | 8.0 | 9.2 | 8.9 | – | 8.2 | 9.3 | – | 9.1 | – | – | – | 8.5 | 8.1 | – | 8.7 |
| 1987 | – | 220 | – | – | – | 230 | – | 170 | 190 | 190 | – | 160 | 170 | – | 170 | 190 | 160 | – | 150 | 150 | – | 150 |
| 1991 | – | 130 | – | – | – | 140 | – | 120 | 130 | 88 | – | 86 | – | 96 | 96 | 94 | 90 | – | 95 | 110 | – | 95 |
| 1995 | – | 77 | – | – | – | 68 | – | 78 | 70 | 59 | 60 | – | 78 | 79 | 57 | 51 | 55 | 61 | 58 | 52 | 49 | 56 |
| 1999 | – | 62 | – | – | – | 60 | – | 50 | 53 | 43 | 35 | 48 | 57 | 50 | 48 | 46 | 47 | 27 | 42 | 41 | 44 | 40 |

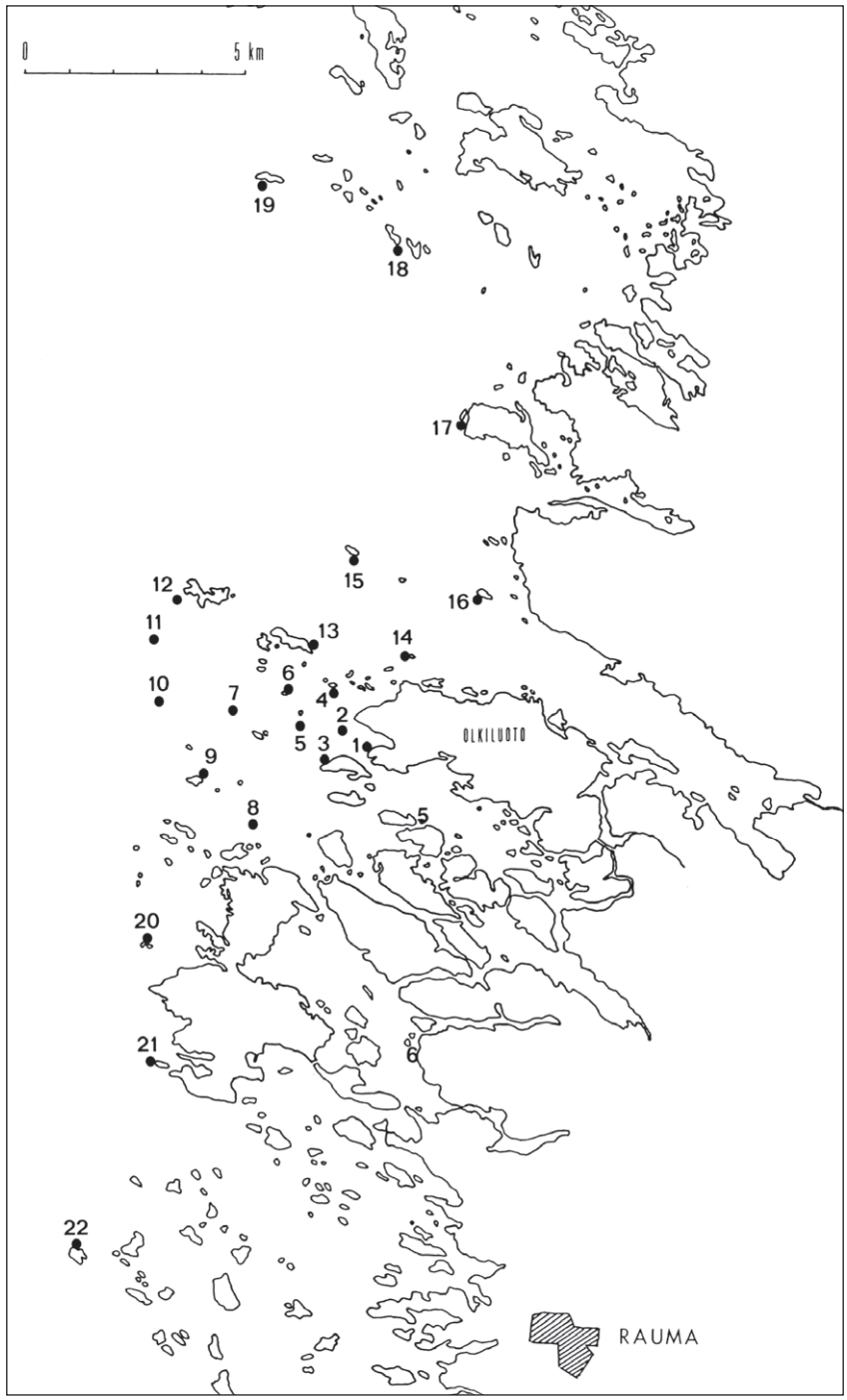


**Fig. 93.** The sampling sites in the *Fucus* surveys at Olkiluoto.

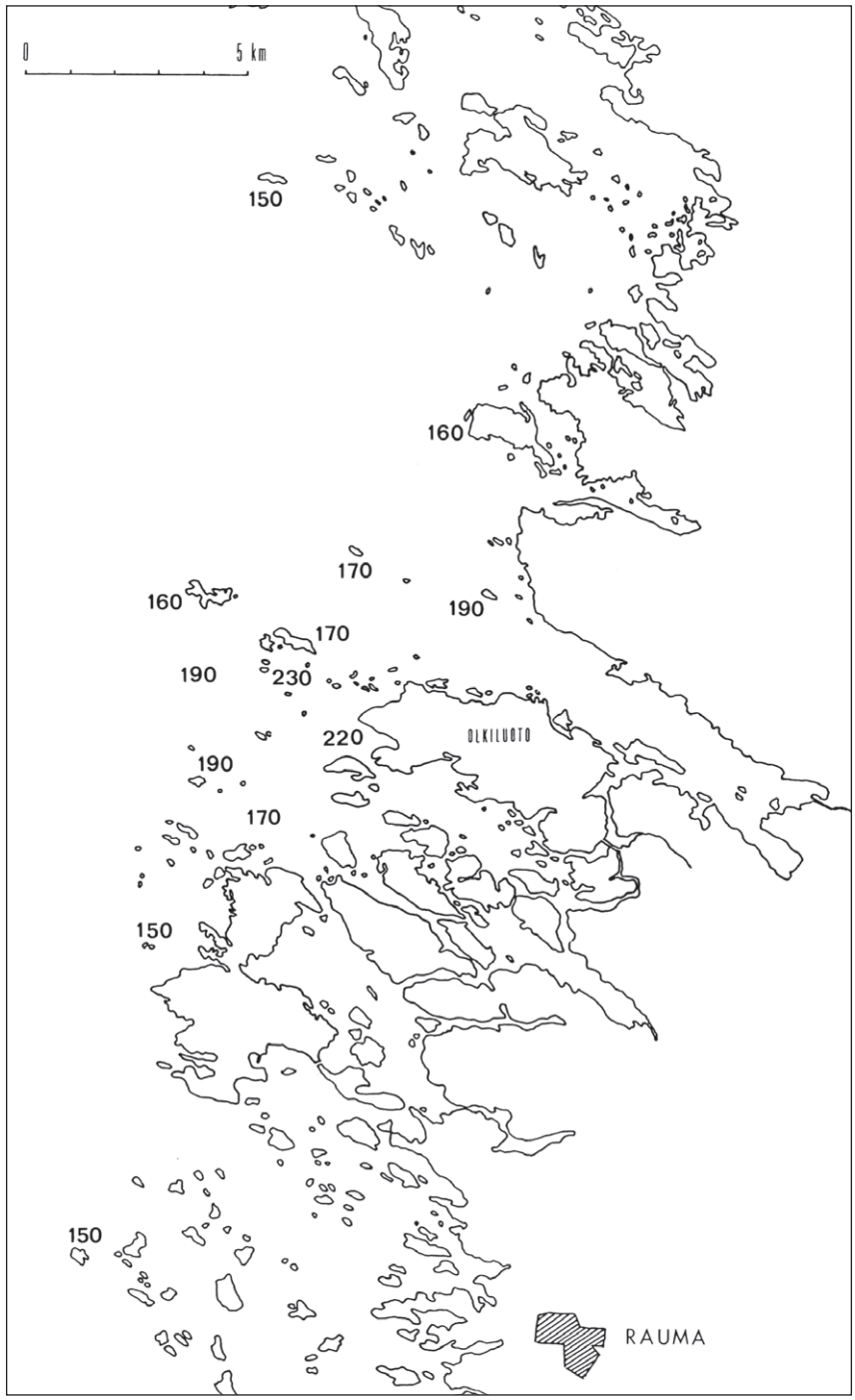


**Fig. 94.** Activity concentrations of $^{137}Cs$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites at Olkiluoto in the 1987 survey.

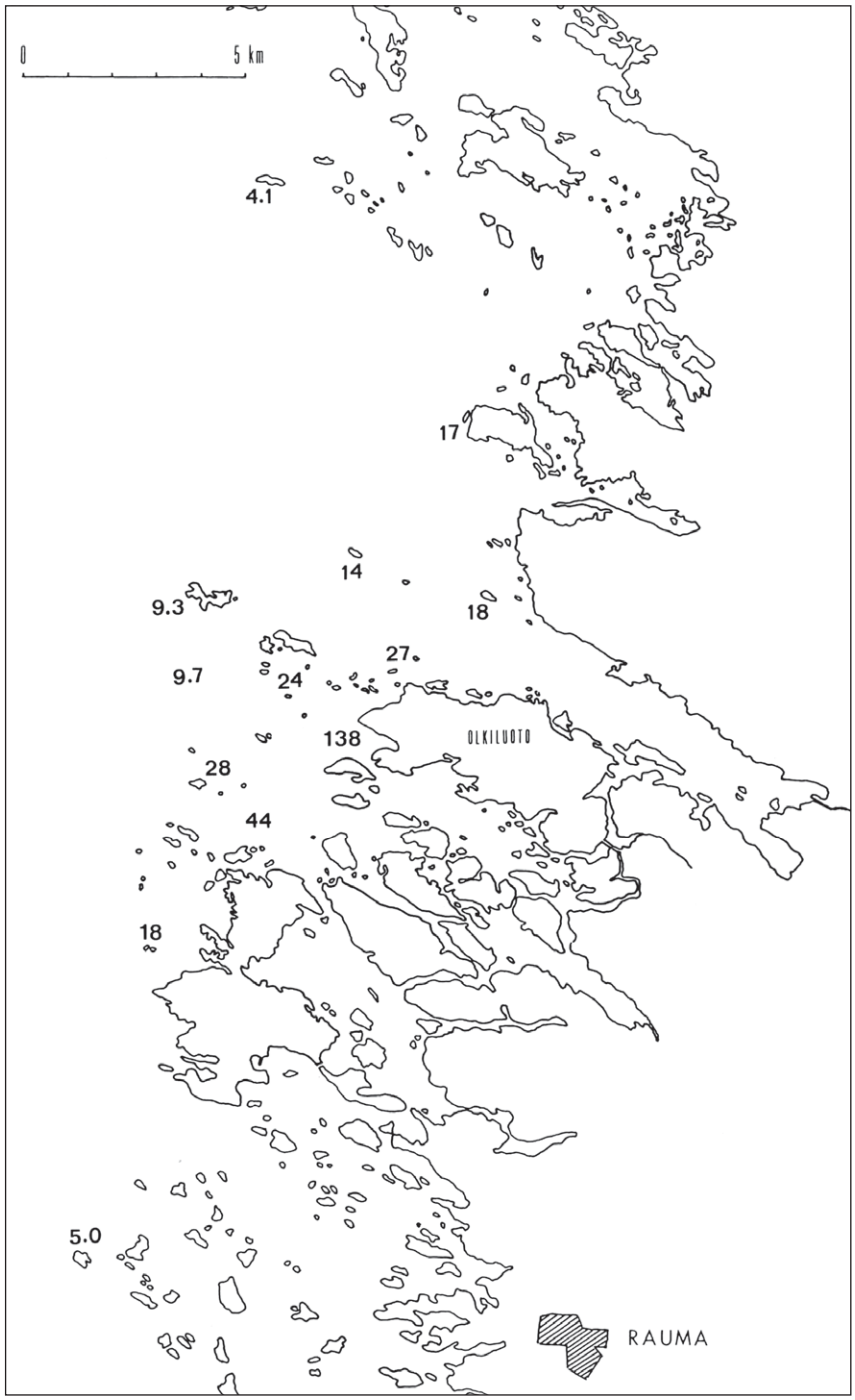


**Fig. 95.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites at Olkiluoto in the 1991 survey.

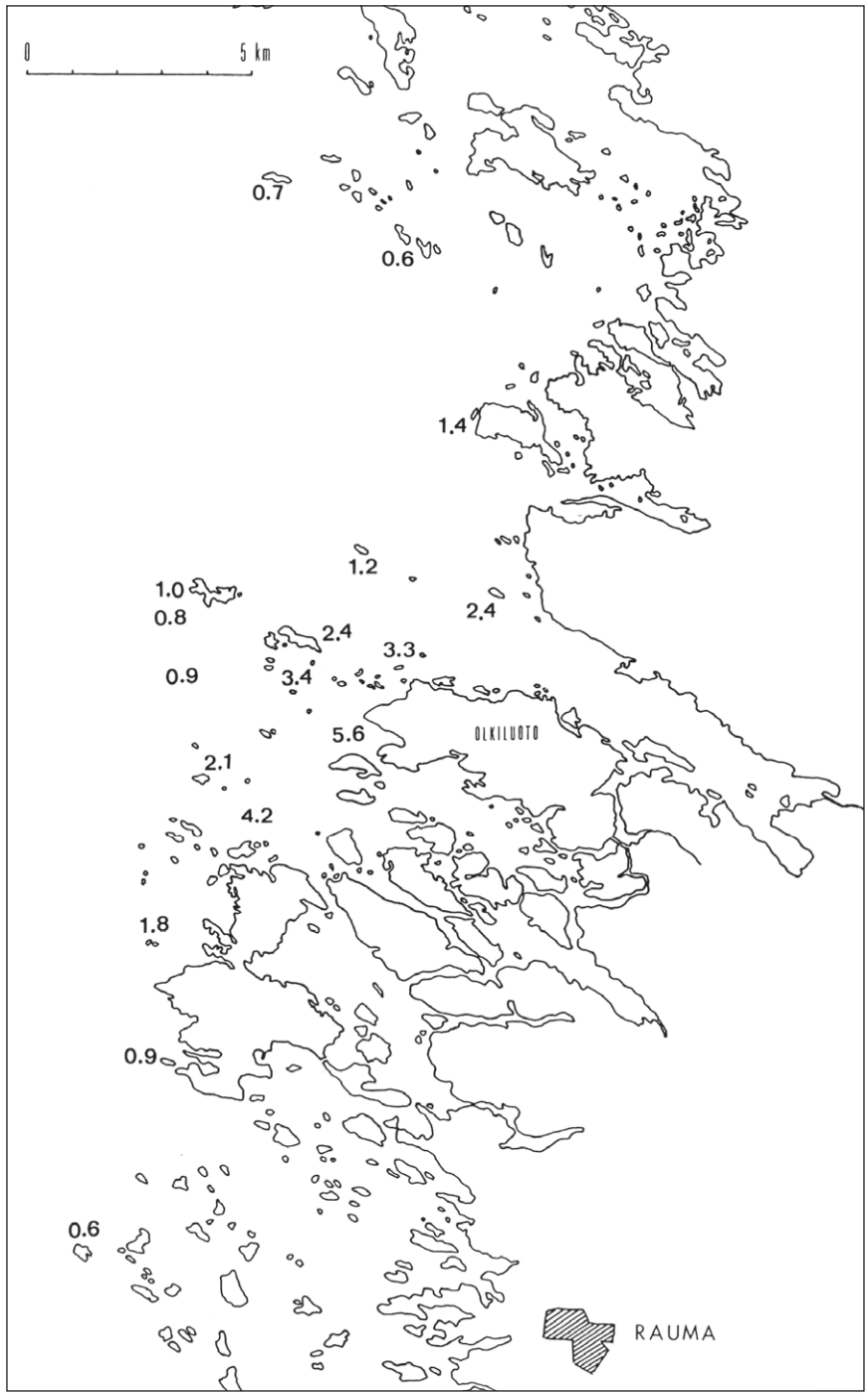


**Fig. 96.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites at Olkiluoto in the 1999 survey (detection limit 0.1 Bq $kg^{-1}$ d.w.).

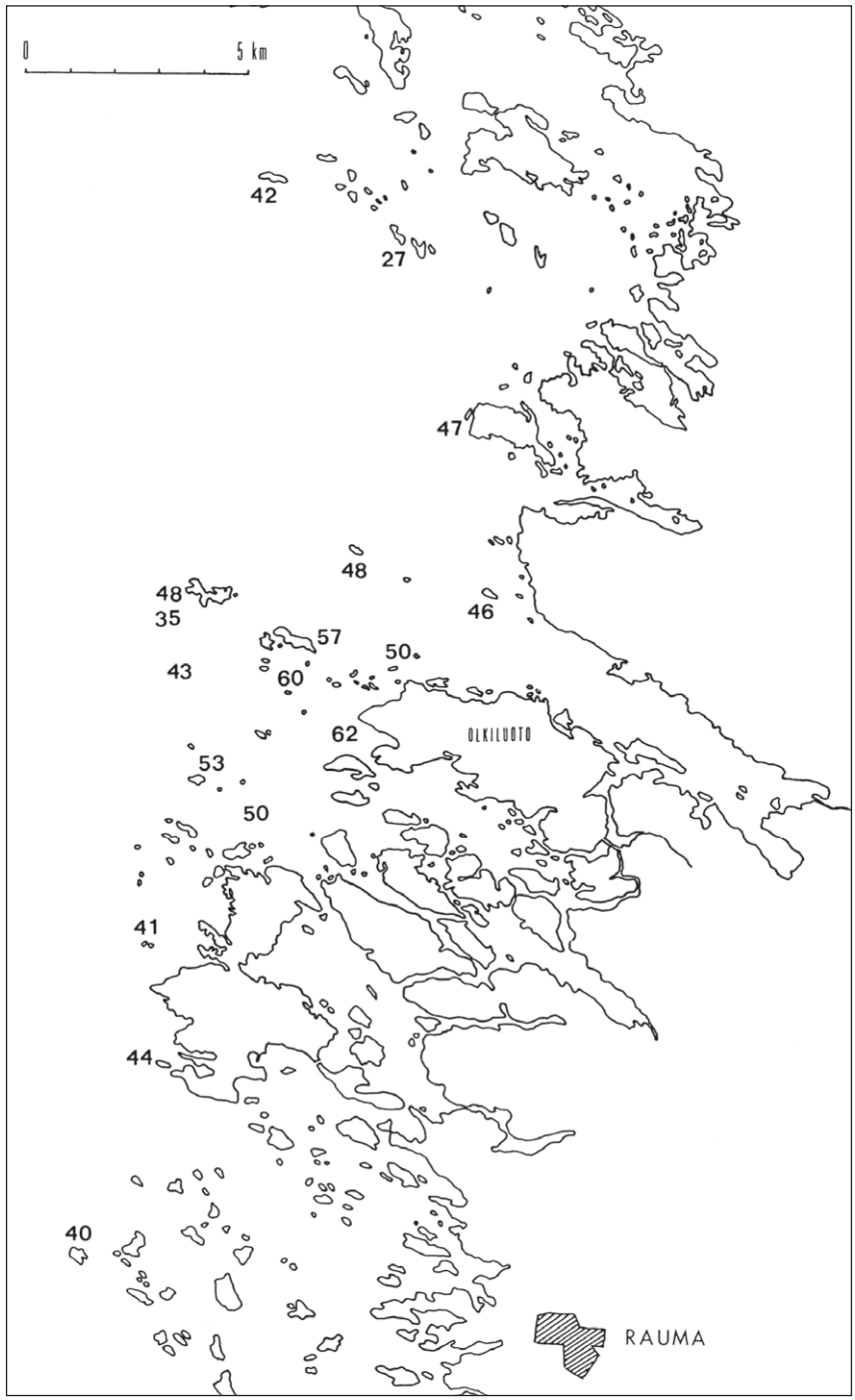


**Fig. 97.** Activity concentrations of $^{137}Cs$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* at the sampling sites at Olkiluoto in the 1999 survey.

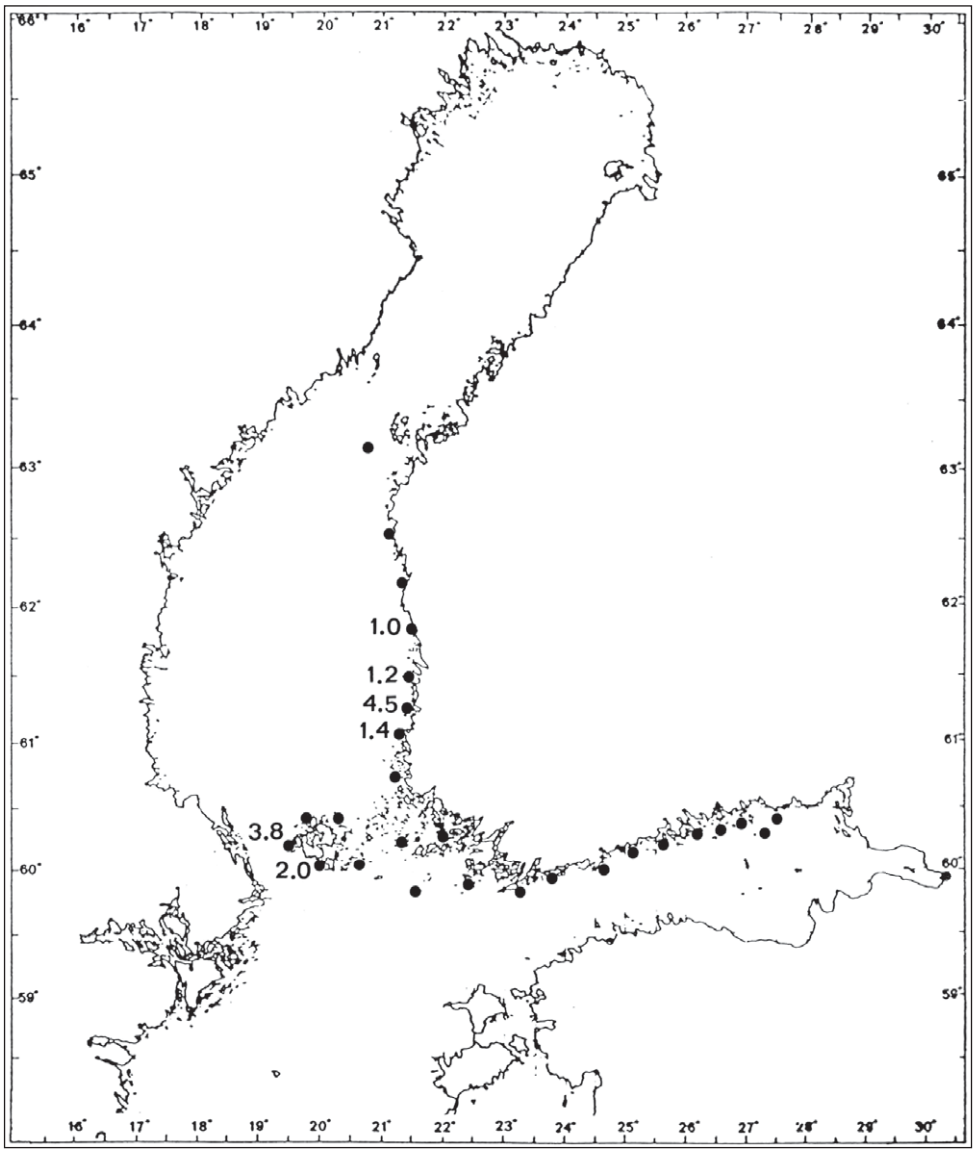


**Fig. 98.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* collected from 27 sampling sites along the Finnish coast in 1987. A dot without a number means that the concentration was below the detection limit of 1 Bq $kg^{-1}$ d.w.

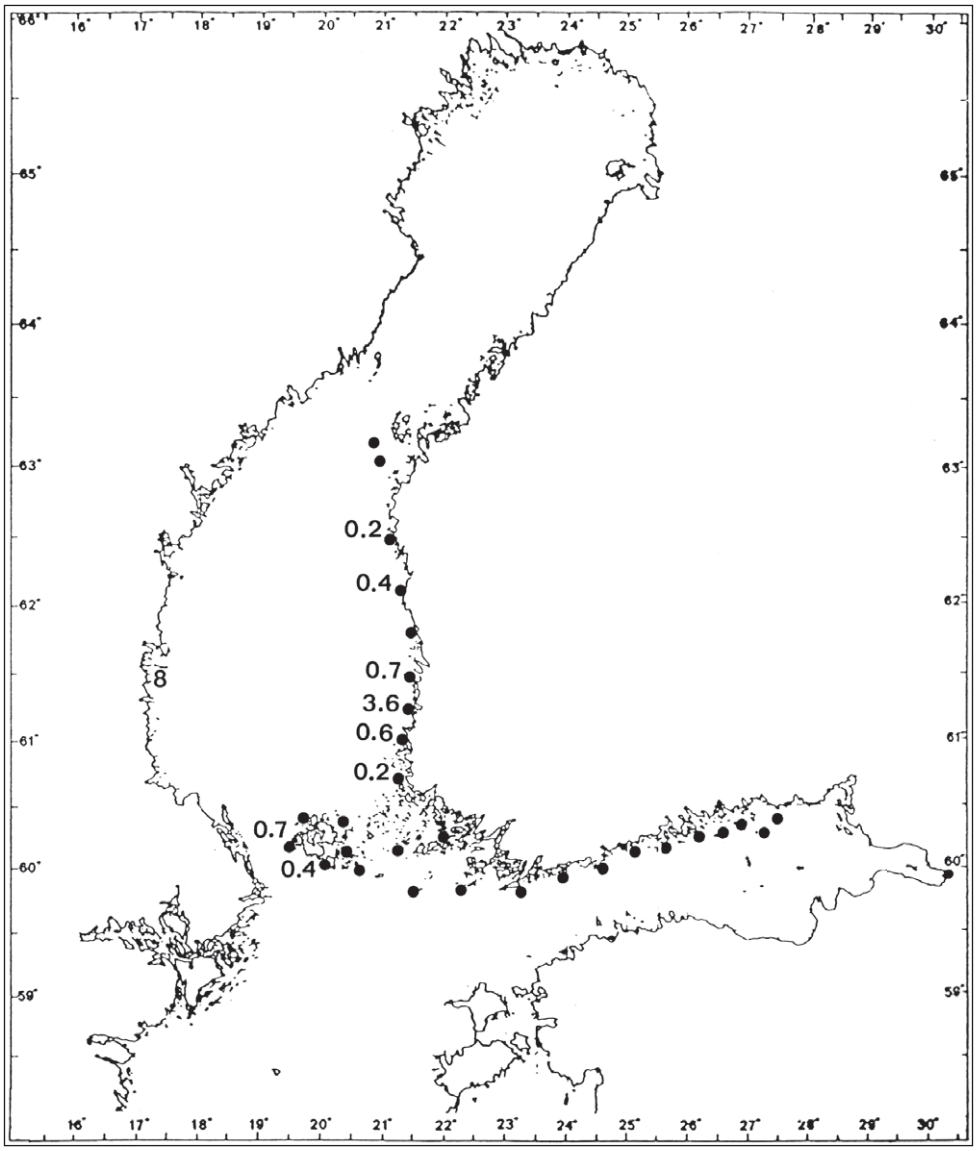


**Fig. 99.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* collected from 26 sampling sites along the Finnish coast in 1991. A dot without a number means that the concentration was below the detection limit of 0.1 Bq $kg^{-1}$ d.w.

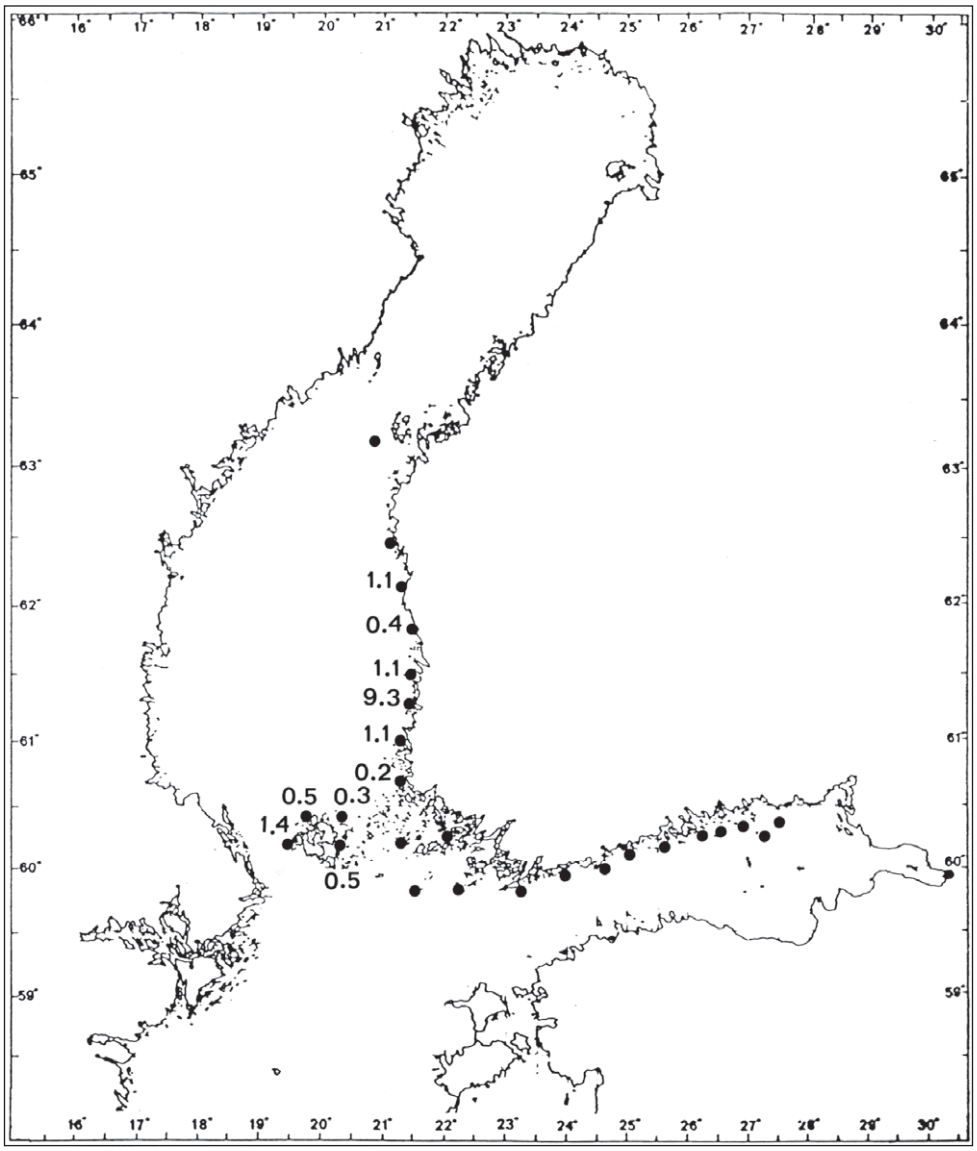


**Fig. 100.** Activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in *Fucus vesiculosus* collected from 29 sampling sites along the Finnish coast in 1995. A dot without a number means that the concentration was below the detection limit of 0.1 Bq $kg^{-1}$ d.w.

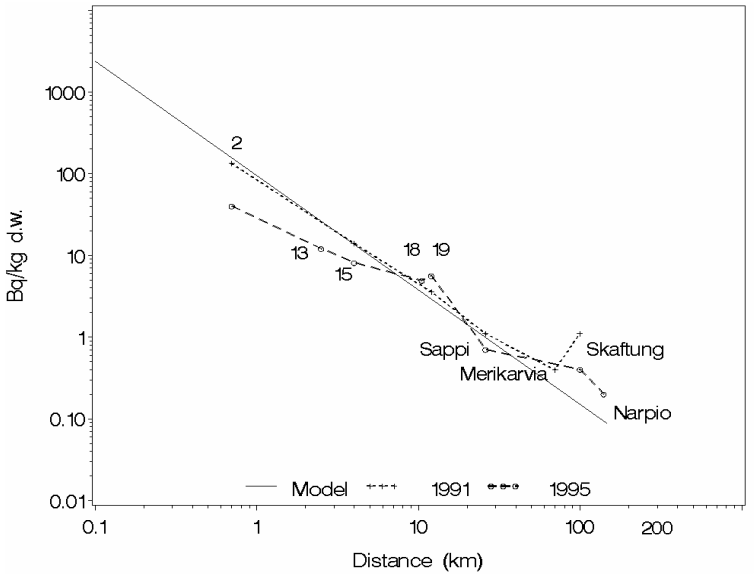


**Fig. 101.** Variation in activity concentration of $^{60}$Co in *Fucus* as a function of distance northwards from Olkiluoto in 1991 and 1995. For the numbers of the sampling sites, see Fig. 93. For the model line, see the text.

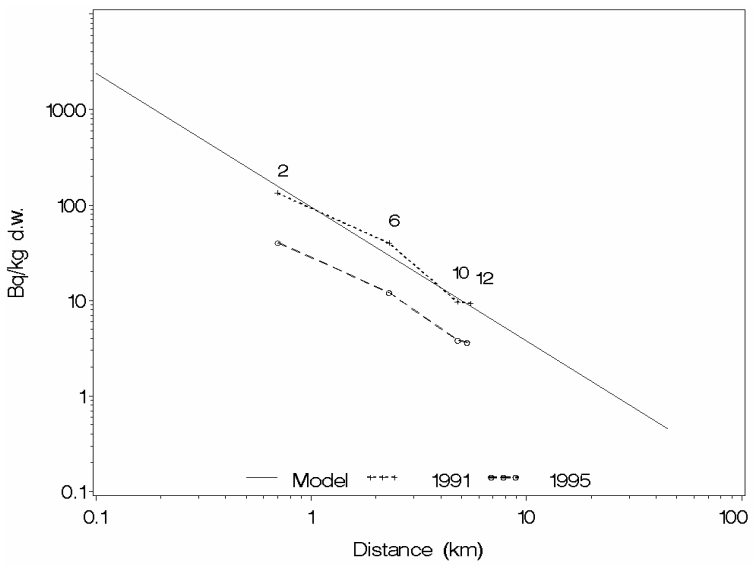


**Fig. 102.** Variation in activity concentration of $^{60}$Co in *Fucus* as a function of distance westwards from Olkiluoto in 1991 and 1995. For the numbers of the sampling sites, see Fig. 93. For the model line, see the text.

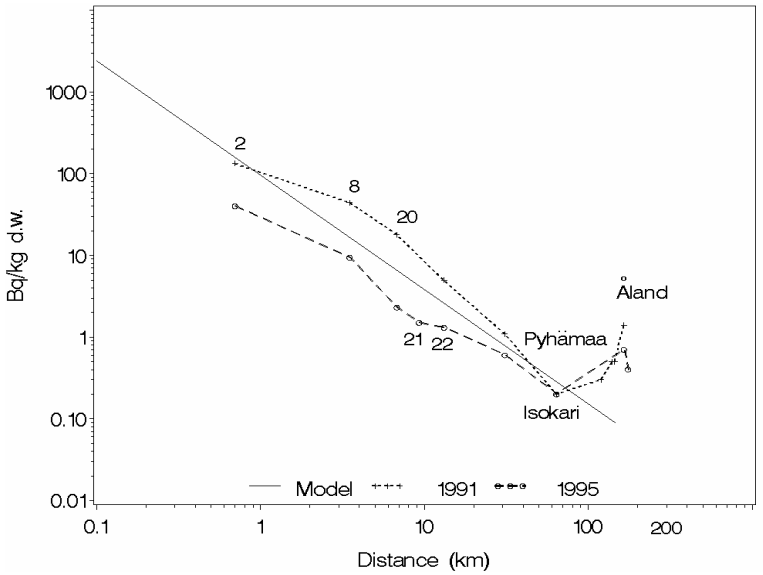


**Fig. 103.** Variation in activity concentration of $^{60}$Co in *Fucus* as a function of distance southwards from Olkiluoto in 1991 and 1995. For the numbers of the sampling sites, see Fig. 93. For the model line, see the text.

### 3.6.2 Seasonal fluctuations of radionuclide concentrations in *Fucus*

The seasonal fluctuations of gamma-emitting radionuclides in *Fucus vesiculosus* were studied in the discharge area of the Loviisa power plant in 1982 and 1983 and at Olkiluoto in 1985 and 1986. *Fucus* samples were taken once a month throughout the year from two sampling sites in Hästholmsfjärden (Sampling Sites 1 and 4 in Fig. 88) and from four sampling sites at Olkiluoto (Sampling Sites 2, 3, 6 and 14 in Fig. 93). The sampling sites were situated at distances of 0.2–2.2 km from the cooling water outlets. The samples were taken by scuba diving, the minimum sample size being 1.8 kg fresh wt. In winter (December–April) the sampling dives were done under the ice. Initially, the projects were planned to last two years in both areas, but in June 1986 the sampling was left off at Olkiluoto, due to the sample backlog in the laboratory caused by the Chernobyl accident. Because it was presumable that the radionuclide concentrations in *Fucus* depend not only on biological factors (such as seasonal changes in the vital functions and growth rate of the algae) but also on the discharges of radionuclides from the local power plants, weekly discharge data from the power plants were compiled for interpretation of the results.

**Loviisa**

In general, the activity concentrations of local discharge nuclides were higher at Sampling Site 1 situated near to the cooling water outlet than at Sampling Site 4 situated on the opposite side of Hästholmsfjärden Bay (Fig. 88). However, during the first half of 1982 the concentrations of $^{58}$Co and $^{60}$Co were repeatedly somewhat higher at Sampling Site 4. The highest activity concentrations of $^{60}$Co, $^{110m}$Ag and $^{54}$Mn in *Fucus* recorded from Sampling Site 1 were 170, 130 and 26 Bq kg$^{-1}$ d.w., respectively, in December 1983, whereas the highest concentration of $^{58}$Co (120 Bq kg$^{-1}$ d.w.) was recorded in January 1983 (Table 28). The $^{137}$Cs concentrations were also generally a little higher at Sampling Site 1.

In principle, liquid discharges from the Loviisa power plant are intermittent, but they are generally timed to occur during the maintenance and refuelling periods (generally in August–October) or at the end of the year. The activity concentrations of local discharge nuclides in *Fucus* seemed to be highly dependent on the discharge amounts (Ilus et al. 1986b) and they were in good agreement with the concentrations theoretically calculated from the discharge data (Ojala et al. 1986). During the first quarter of 1982, the weekly discharges of $^{58}$Co, $^{60}$Co and $^{110m}$Ag were less than $5 \cdot 10^6$ Bq and did not appear to raise the concentrations in *Fucus* (Figs. 104–106). On the other hand, the weekly discharges larger than $1 \cdot 10^8$ Bq in the last quarter and at the end of 1982, and after midsummer and at the end of 1983, raised the concentrations in *Fucus* substantially.

During the dischargeless periods in April–October 1982 (26 weeks) and January–June 1983 (23 weeks), the concentrations went down very rapidly. At Sampling Site 1, the loss of $^{58}$Co was 98 and 96% during these two periods. The corresponding percentage values were 79 and 84% for $^{110m}$Ag, 81 and 72% for $^{60}$Co and 67 and 73% for $^{54}$Mn (Figs. 104–106). This demonstrates that the 'memory' of *Fucus* is very short, the activity concentrations detected in the *Fucus* samples thus giving an indication of relatively recently-occurred discharges.

The theoretical ('calculated') values of $^{58}$Co, $^{60}$Co and $^{110m}$Ag in *Fucus* shown in Figs. 104–106 are based on the model used, e.g., by Mattsson et al. (1980a) and Nilsson et al. (1980) in their studies at Barsebäck. Accordingly, the time-dependent concentration C(t) in *Fucus* can be expressed as:

$$\frac{d\,C(t)}{dt} = a\;A(t) - (k + \lambda)\,C(t)$$

where a is the accumulation factor of a nuclide, describing the increase of activity concentration in *Fucus* compared with seawater, A(t) is the concentration of a nuclide in the seawater surrounding the *Fucus* plant, k is the elimination rate of activity of the nuclide, and $\lambda$ is the physical decay-rate constant of the nuclide.

The biological half-lives were calculated from the results from January–May 1983, when no new discharges occurred. The values were 75 days for $^{58}$Co and $^{60}$Co and 120 days for $^{110m}$Ag. The effective environmental half-lives (which also take into account the physical half-lives) were 36 days for $^{58}$Co, 72 days for $^{60}$Co and 81 days for $^{110m}$Ag (Ojala et al. 1986). At Barsebäck, the biological half-life of $^{58}$Co and $^{60}$Co in *F. vesiculosus* was found to be 60 ± 15 days, and approximately the same value was found for $^{54}$Mn and $^{65}$Zn (Mattsson 1980a). Dahlgaard and Boelskifte (1992) concluded, on the basis on their results obtained from the Ringhals area with the SENSI model, that the dilution of contaminant concentrations with new growth is the major cause of the decreasing concentrations of previously-accumulated zinc, cobalt and manganese in *Fucus*. The actual loss of caesium from *Fucus* was of the same order of magnitude as the growth dilution. On the Finnish coast the growth rate of *Fucus* is very slow. Thus the influence of new growth is certainly smaller than on the Swedish west coast.

Various factors have been observed to affect the uptake, accumulation and release of radionuclides in algae, e.g., season of the year, age of the tissue, salinity of the water, etc. (Carlson 1990). At Loviisa, the discontinual discharges and the seasonal variation in hydrological mixing conditions in Hästholmsfjärden Bay seemed to be the main factors influencing the uptake of radionuclides (Ojala et al. 1986). The impact of the season and the stage of the growing period seemed to be weaker than the impact of the discharged amounts, perhaps at least partly because the main part of the discharges took place at times other than the growing periods.

**Table 28.** Activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{110m}$Ag, $^{124}$Sb, $^{131}$I and $^{134}$Cs in *Fucus vesiculosus* (Bq kg$^{-1}$ d.w.) collected once a month from the Loviisa 2 and 4 sampling sites (see Fig. 88) during the Monthly *Fucus* Project in 1982 and 1983 (< = below the detection limit).

| Date | Loviisa 1 (A), Halkokari | | | | | | | Loviisa 4 (B), Bölsviken | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | $^{54}$Mn | $^{58}$Co | $^{60}$Co | $^{110m}$Ag | $^{124}$Sb | $^{131}$I | $^{137}$Cs | $^{54}$Mn | $^{58}$Co | $^{60}$Co | $^{110m}$Ag | $^{124}$Sb | $^{131}$I | $^{137}$Cs |
| 14.01.1982 | 14 | 55 | 26 | 25 | < | < | 11 | 11 | 68 | 34 | 26 | < | < | 11 |
| 25.02.1982 | 10 | 53 | 27 | 19 | < | < | 11 | 7.7 | 45 | 28 | 12 | < | < | 9.7 |
| 19.03.1982 | 9.7 | 42 | 33 | 18 | < | < | 11 | 5.2 | 34 | 20 | 9.6 | < | < | 13 |
| 15.04.1982 | 5.1 | 23 | 17 | 12 | < | < | 11 | 6.1 | 25 | 24 | 6.2 | < | < | 10 |
| 28.05.1982 | 4.7 | 15 | 19 | 11 | < | < | 14 | 4.3 | 10 | 14 | 4.4 | < | < | 15 |
| 24.06.1982 | 2.3 | 3.0 | 7.8 | 3.2 | < | < | 13 | < | 6.6 | 13 | 3.2 | < | < | 13 |
| 23.07.1982 | 2.7 | 3.5 | 9.3 | 2.7 | < | < | 11 | < | 2.1 | 7.0 | 1.0 | < | < | 12 |
| 25.08.1982 | 1.7 | 0.62 | 5.1 | 3.1 | < | < | 13 | < | < | 2.1 | < | < | < | 6.4 |
| 23.09.1982 | 1.7 | 0.56 | 3.2 | 2.5 | < | < | 14 | < | < | 2.6 | 0.74 | < | < | 10 |
| 15.10.1982 | 6.3 | 47 | 18 | 14 | 3.2 | < | 14 | 2.3 | 11 | 6.4 | 4.5 | < | < | 12 |
| 12.11.1982 | 7.5 | 45 | 20 | 8.5 | 1.8 | < | 12 | < | 8.8 | 4.5 | 2.8 | < | < | 8.8 |
| 16.12.1982 | 3.2 | 21 | 11 | 8.6 | 1.2 | < | 12 | < | 6.2 | 3.9 | 3.1 | < | < | 7.4 |
| 20.01.1983 | 21 | 120 | 46 | 30 | 1.9 | < | 16 | 4.6 | 19 | 10 | 7.3 | < | < | 10 |
| 18.02.1983 | 17 | 81 | 42 | 23 | < | < | 15 | 5.4 | 19 | 12 | 6.6 | < | < | 10 |
| 11.03.1983 | 13 | 50 | 31 | 18 | < | < | 15 | 3.5 | 11 | 8.3 | 3.8 | < | < | 10 |
| 15.04.1983 | 10 | 27 | 26 | 12 | 1.1 | < | 15 | 3.6 | 5.9 | 7.5 | 2.7 | < | < | 13 |
| 19.05.1983 | 11 | 15 | 20 | 6.9 | < | < | 16 | 2.2 | 3.7 | 7.0 | 2.8 | < | < | 15 |
| 22.06.1983 | 5.6 | 5.0 | 13 | 4.9 | < | < | 15 | 2.6 | 1.9 | 5.0 | 1.0 | < | < | 11 |
| 22.07.1983 | 5.0 | 4.6 | 12 | 18 | 1.4 | < | 12 | 1.8 | 1.5 | 5.4 | 8.9 | < | < | 8.8 |
| 19.08.1983 | 6.5 | 26 | 23 | 27 | < | 5.6 | 10 | 2.2 | 13 | 11 | 13 | < | 3.0 | 8.1 |
| 22.09.1983 | 6.4 | 17 | 21 | 48 | 5.4 | 7.2 | 12 | 4.5 | 13 | 16 | 49 | 6.4 | 4.3 | 9.3 |
| 14.10.1983 | 9.7 | 16 | 24 | 78 | 8.4 | < | 16 | 4.7 | 8.1 | 13 | 31 | 1.3 | < | 10 |
| 15.11.1983 | 7.3 | 17 | 21 | 60 | 4.2 | 5.0 | 16 | 5.5 | 14 | 15 | 45 | 2.2 | 8.4 | 12 |
| 20.12.1983 | 26 | 34 | 170 | 130 | 4.1 | < | 17 | 10 | 24 | 54 | 62 | 2.4 | 1.3 | 12 |

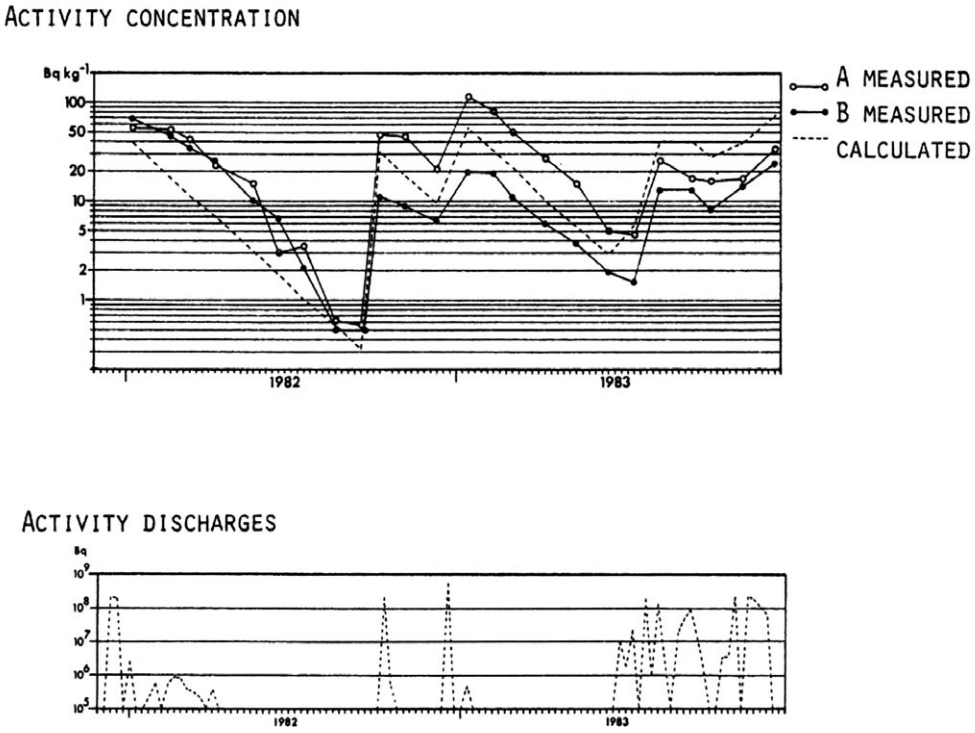


**Fig. 104.** Time variation of measured and calculated $^{58}$Co activity concentrations in *Fucus vesiculosus* at two locations (A = Site 1, B = Site 4) in the discharge area at Loviisa, and the weekly $^{58}$Co activity discharges from the Loviisa power plant in 1982–1983 (Ojala et al. 1986).

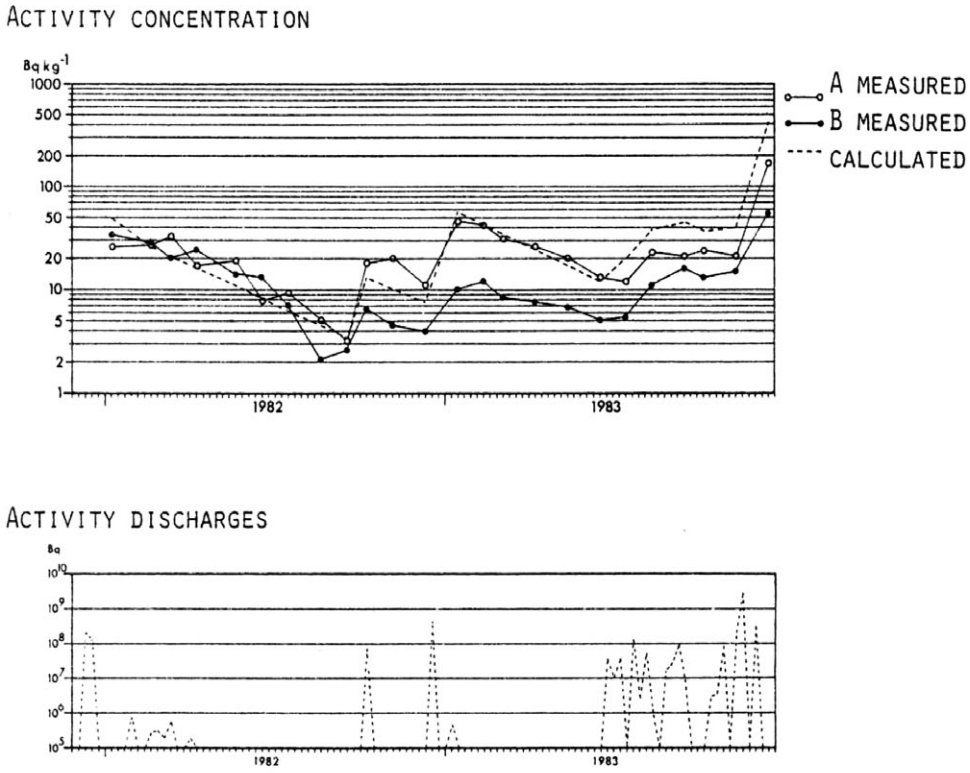


**Fig. 105.** Time variation of measured and calculated $^{60}$Co activity concentrations in *Fucus vesiculosus* at two locations (A = Site 1, B = Site 4) in the discharge area at Loviisa, and the weekly $^{60}$Co activity discharges from the Loviisa power plant in 1982–1983 (Ojala et al. 1986).

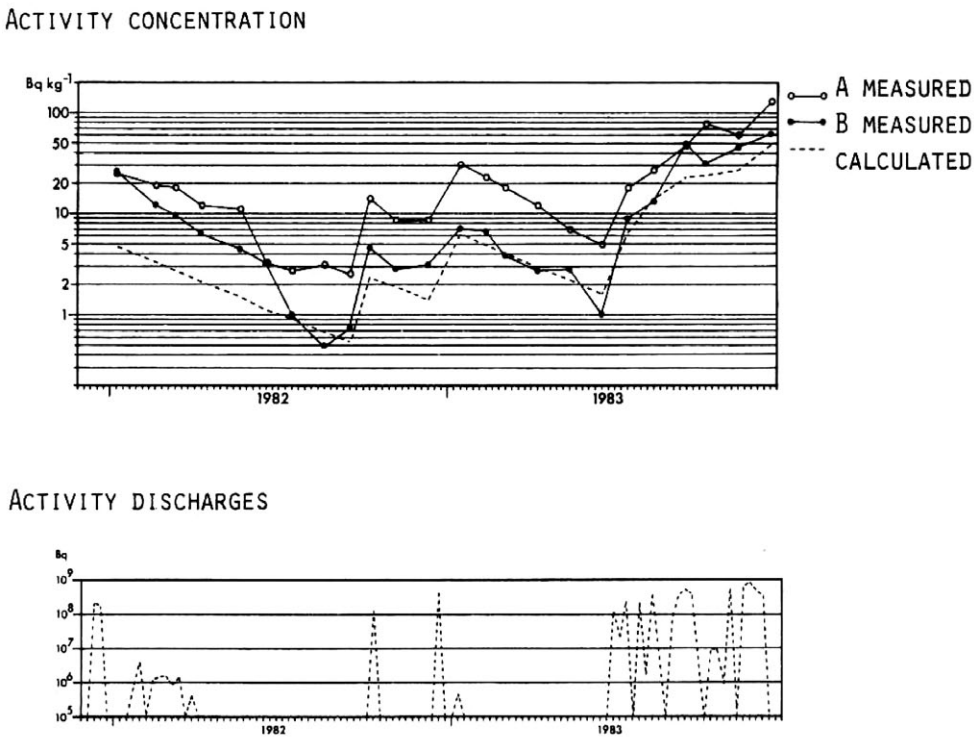


**Fig. 106.** Time variation of measured and calculated $^{110m}$Ag activity concentrations in *Fucus vesiculosus* at two locations (A = Site 1, B = Site 4) in the discharge area at Loviisa, and the weekly $^{110m}$Ag activity discharges from the Loviisa power plant in 1982–1983 (Ojala et al. 1986).

**Olkiluoto**

Contrary to the procedures at Loviisa, liquid discharges from the Olkiluoto power plant are more regular; they are not continuous but repetitious, at regular and shorter intervals. However, the largest discharges are usually timed to coincide with the maintenance and refuelling periods, i.e., to May–June.

Local discharges were most clearly visible in *Fucus* collected from Sampling Sites 2 and 3 located in Iso Kaalonpuhti Bay, but only to a much lesser degree at the farther Sampling Sites 6 and 14 located in Kalliopöllä and Valkiakari (Fig. 93). In general, the $^{137}$Cs concentrations too were a little higher in *Fucus*

from the sampling sites nearest to the cooling water outlet. The highest activity concentrations of $^{60}$Co and $^{58}$Co from Sampling Site 2 were 140 and 42 Bq kg$^{-1}$ d.w. in June 1985, while that of $^{54}$Mn was 100 Bq kg$^{-1}$ d.w. in July 1985. The impact of the Chernobyl accident was clearly visible in the results from the 21$^{st}$ of May 1986, especially in those of $^{137}$Cs (Table 29). At Sampling Sites 6 and 14, the synchronism between the activity concentrations of local discharge nuclides in *Fucus* and the discharges was not so clear (Table 30), but the concentrations were more even, reflecting a more even dispersion of discharge nuclides in the water at greater distances.

In Iso Kaalonpuhti Bay, the dependence of the activity concentrations of discharge nuclides in *Fucus* seemed to follow a quite similar connection with the magnitude of the discharges as in Hästholmsfjärden at Loviisa. Even the repetitious regular discharges of $^{54}$Mn and $^{58}$Co less than $2 \cdot 10^7$ Bq in July 1985–April 1986 seemed neither to raise the concentrations in *Fucus* nor keep them at the same level, but rather the concentrations seemed to decline (Figs. 107 and 108). Discharges of $^{60}$Co at a level of $2 \cdot 10^7 – 2 \cdot 10^8$ Bq seemed to keep the concentrations at about the same level or let them decrease slowly (Fig. 109). On the other hand, the peak value of $^{60}$Co discharges in the second half of June 1985 did not raise the concentration in *Fucus*, which was already at its highest in the first half of June. The discharges of $^{110m}$Ag were lower, and consequently, affected the concentrations in *Fucus* more slightly (Fig. 110).

In the same way as at Loviisa, the quantities of the discharges were more important in determining the concentrations in *Fucus* than the impact of the season or other biological factors connected to it.

In 1985, seawater samples were taken in parallel with the *Fucus* samples from Sampling Site 2 (Table 29). $^{60}$Co was detected in five samples (1.9–10 Bq m$^{-3}$) and $^{54}$Mn in one sample (4.7 Bq m$^{-3}$); in the other samples the concentrations were below the detection limit. There was a coincidence between the highest $^{54}$Mn concentration in *Fucus* and the detection of $^{54}$Mn in seawater, but there was no correlation between the $^{60}$Co concentrations in *Fucus* and in seawater. It is obvious that the radionuclide concentrations in seawater fluctuate very rapidly at this sampling site, because it is located just in front of the outlet channel and exposed to the current of the cooling water.

Concentration ratios CR (= concentration factors CF) calculated from these results for *Fucus* were 3 800 for $^{54}$Mn, 4 400 for $^{60}$Co and 140 for $^{137}$Cs on a fresh weight basis. The values for Mn and Cs were lower than those recommended by IAEA and those used in the EU ERICA Tool for marine macroalgae (*cf.* IAEA 2004b, Hosseini et al. 2008), whereas that for Co lay at about the centre of the range (6 000 and 2 100). At least the lower value for caesium at Olkiluoto is explained by the brackish-water conditions prevailing on the Finnish coast.

**Table 29.** Activity concentrations of $^{51}Cr$, $^{54}Mn$, $^{58}Co$, $^{60}Co$, $^{65}Zn$, $^{110m}Ag$, $^{137}Cs$ in *Fucus vesiculosus* (Bq $kg^{-1}$ d.w.) collected once a month from the Olkiluoto 2 and 3 sampling sites (see Fig. 93) during the Monthly *Fucus* Project in 1985 and 1986 and in seawater (Bq $m^{-3}$) from the Olkiluoto 2 sampling site in 1985 (< = below the detection limit, – = not sampled or not analysed).

| Date | Olkiluoto 2 (A), Kaalonpuhdin matalikko | | | | | | | Olkiluoto 3, Kuusinen W | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | $^{51}Cr$ | $^{54}Mn$ | $^{58}Co$ | $^{60}Co$ | $^{65}Zn$ | $^{110m}Ag$ | $^{137}Cs$ | $^{51}Cr$ | $^{54}Mn$ | $^{58}Co$ | $^{60}Co$ | $^{65}Zn$ | $^{110m}Ag$ | $^{137}Cs$ |
| 29.01.1985 | < | 29 | 7.2 | 120 | 12 | 5.5 | 7.0 | – | – | – | – | – | – | – |
| 06.03.1985 | 8.9 | 18 | 4.3 | 63 | 5.9 | 3.8 | 8.5 | 2.9 | 16 | 3.2 | 59 | 7.3 | 2.9 | 7.3 |
| 26.03.1985 | 12 | 13 | 4.0 | 48 | 5.5 | 2.9 | 8.8 | 2.7 | 14 | 2.6 | 53 | 4.5 | 1.7 | 8.1 |
| 16.04.1985 | 7.0 | 15 | 2.9 | 52 | 4.3 | 2.6 | 8.7 | 3.0 | 14 | 2.7 | 54 | 5.3 | 2.2 | 7.7 |
| 14.05.1985 | 17 | 22 | 6.6 | 100 | 6.6 | 2.7 | 11 | 7.8 | 11 | 2.3 | 42 | 3.6 | 1.6 | 11 |
| 12.06.1985 | 18 | 79 | 42 | 140 | 14 | 3.8 | 8.8 | 14 | 54 | 31 | 80 | 7.8 | 2.2 | 11 |
| 09.07.1985 | < | 100 | 26 | 99 | 14 | 8.7 | 7.4 | 11 | 53 | 17 | 64 | 9.7 | 4.5 | 7.8 |
| 14.08.1985 | < | 86 | 25 | 100 | 21 | 9.5 | 7.0 | < | 32 | 6.5 | 33 | 7.5 | 2.6 | 7.1 |
| 18.09.1985 | < | 43 | 11 | 70 | 9.5 | 3.1 | 6.5 | < | 26 | 3.7 | 36 | 5.6 | 2.2 | 6.9 |
| 15.10.1985 | < | 38 | 9.5 | 78 | 10 | 3.0 | 6.8 | < | 31 | 6.1 | 49 | 6.3 | 3.2 | 7.4 |
| 14.11.1985 | < | 32 | 5.8 | 64 | 9.7 | 3.1 | 7.0 | < | 34 | 6.3 | 50 | 8.1 | 2.6 | 7.0 |
| 12.12.1985 | < | 29 | 5.2 | 71 | 9.5 | 2.1 | 5.9 | < | 20 | 3.6 | 39 | 4.2 | 1.1 | 7.1 |
| 28.01.1986 | < | 24 | 3.7 | 69 | 6.8 | 2.4 | 6.2 | < | 26 | 2.7 | 52 | 6.4 | 2.1 | 6.5 |
| 20.02.1986 | < | 22 | 2.8 | 51 | 4.5 | 2.3 | 9.0 | < | 17 | 1.6 | 38 | 3.0 | 1.3 | 8.2 |
| 20.03.1986 | 5.8 | 18 | 1.5 | 48 | 3.9 | 1.8 | 8.9 | < | 17 | 1.2 | 44 | 4.5 | 1.5 | 8.1 |
| 28.04.1986 | < | 18 | < | 70 | 3.3 | 2.2 | 25 | – | – | – | 65 | – | – | – |
| 21.05.1986 | < | 37 | < | 110 | 23 | 170 | 1300 | – | – | – | 62 | – | – | – |
| **Seawater** | | | | | | | | | | | | | | |
| 29.01.1985 | < | < | < | 3.8 | < | < | 11 | – | – | – | – | – | – | – |
| 06.03.1985 | < | < | < | < | < | < | 10 | – | – | – | – | – | – | – |
| 26.03.1985 | < | < | < | 4.3 | < | < | 12 | – | – | – | – | – | – | – |
| 16.04.1985 | < | < | < | < | < | < | 9.6 | – | – | – | – | – | – | – |
| 14.05.1985 | < | < | < | 3.3 | < | < | 9.7 | – | – | – | – | – | – | – |

**Table 29.** Continued.

| Date | Olkiluoto 2 (A), Kaalonpuhdin matalikko | | | | | | | Olkiluoto 3, Kuusinen W | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | $^{51}$Cr | $^{54}$Mn | $^{58}$Co | $^{60}$Co | $^{65}$Zn | $^{110m}$Ag | $^{137}$Cs | $^{51}$Cr | $^{54}$Mn | $^{58}$Co | $^{60}$Co | $^{65}$Zn | $^{110m}$Ag | $^{137}$Cs |
| 11.06.1985 | < | < | < | < | < | < | 10 | – | – | – | – | – | – | – |
| 09.07.1985 | < | 4.7 | < | 10 | < | < | 11 | – | – | – | – | – | – | – |
| 15.08.1985 | < | < | < | < | < | < | 9.9 | – | – | – | – | – | – | – |
| 18.09.1985 | < | < | < | < | < | < | 11 | – | – | – | – | – | – | – |
| 15.10.1985 | < | < | < | 1.9 | < | < | 11 | – | – | – | – | – | – | – |
| 14.11.1985 | < | < | < | < | < | < | 10 | – | – | – | – | – | – | – |

**Table 30.** Activity concentrations of $^{51}Cr$, $^{54}Mn$, $^{58}Co$, $^{60}Co$, $^{65}Zn$, $^{110m}Ag$, $^{137}Cs$ in *Fucus vesiculosus* (Bq $kg^{-1}$ d.w.) collected once a month from the Olkiluoto 6 and 14 sampling sites (see Fig. 93) during the Monthly *Fucus* Project in 1985 and 1986 (< = below the detection limit, – = not sampled or analysed).

| Date | Olkiluoto 6 (B), Kalliopöllä NE | | | | | | | Olkiluoto 14, Valkiakari W | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | $^{51}Cr$ | $^{54}Mn$ | $^{58}Co$ | $^{60}Co$ | $^{65}Zn$ | $^{110m}Ag$ | $^{137}Cs$ | $^{51}Cr$ | $^{54}Mn$ | $^{58}Co$ | $^{60}Co$ | $^{65}Zn$ | $^{110m}Ag$ | $^{137}Cs$ |
| 29.01.1985 | < | 7.2 | 1.4 | 25 | 2.6 | 2.2 | 5.9 | < | 4.1 | 1.2 | 16 | 2.1 | 1.9 | 6.1 |
| 06.03.1985 | 3.0 | 7.8 | 1.8 | 31 | 3.2 | 2.2 | 6.2 | < | 4.4 | 0.7 | 19 | 2.8 | 1.9 | 5.8 |
| 26.03.1985 | 10 | 7.3 | 3.4 | 34 | 2.3 | 2.1 | 7.3 | < | 4.4 | 0.5 | 19 | 2.3 | 2.1 | 6.3 |
| 16.04.1985 | < | 5.3 | 1.1 | 21 | 2.7 | 1.5 | 7.0 | – | – | – | – | – | – | – |
| 14.05.1985 | 12 | 11 | 2.5 | 51 | 6.1 | 0.9 | 9.3 | < | 2.8 | < | 18 | 3.3 | 1.7 | 7.1 |
| 12.06.1985 | 12 | 14 | 7.4 | 39 | 3.0 | 2.2 | 11 | < | 4.8 | 5.0 | 18 | 1.3 | 1.1 | 7.1 |
| 10.07.1995 | < | 29 | 11 | 39 | 5.5 | 3.2 | 8.3 | < | 7.8 | 4.7 | 19 | 3.3 | 1.4 | 6.7 |
| 14.08.1985 | < | 16 | 4.7 | 23 | 2.7 | 1.7 | 6.6 | < | 5.1 | 2.7 | 16 | 2.4 | 1.0 | 7.1 |
| 18.09.1985 | < | 13 | 2.3 | 21 | 3.1 | 1.2 | 7.0 | < | 5.5 | 2.0 | 14 | 2.4 | 1.0 | 6.3 |
| 15.10.1985 | < | 6.5 | 1.6 | 17 | 2.7 | 0.8 | 6.5 | < | 5.5 | 1.3 | 16 | 2.2 | 0.7 | 5.6 |
| 14.11.1985 | < | 7.2 | 1.5 | 15 | 1.1 | 0.7 | 6.4 | < | 4.7 | 1.3 | 16 | 2.3 | 1.1 | 5.4 |
| 12.12.1985 | < | 7.4 | 1.0 | 16 | 1.2 | 0.9 | 7.1 | < | 4.6 | 0.9 | 14 | 1.0 | 0.7 | 5.6 |
| 28.01.1986 | < | 6.6 | 0.8 | 18 | 1.4 | 0.7 | 6.3 | < | 4.9 | 0.6 | 15 | 1.8 | 0.8 | 6.0 |
| 20.02.1986 | < | 7.0 | 1.4 | 25 | 2.9 | 1.0 | 8.5 | < | 4.3 | 0.5 | 14 | 1.5 | 0.6 | 6.2 |
| 20.03.1986 | < | 4.9 | 1.0 | 17 | 1.6 | 0.8 | 7.3 | < | 4.1 | 0.5 | 15 | 1.8 | 0.6 | 5.7 |
| 28.04.1986 | – | – | – | 18 | – | – | – | – | – | – | 18 | – | – | – |
| 21.05.1986 | – | – | – | 17 | – | – | – | – | – | – | 24 | – | – | – |

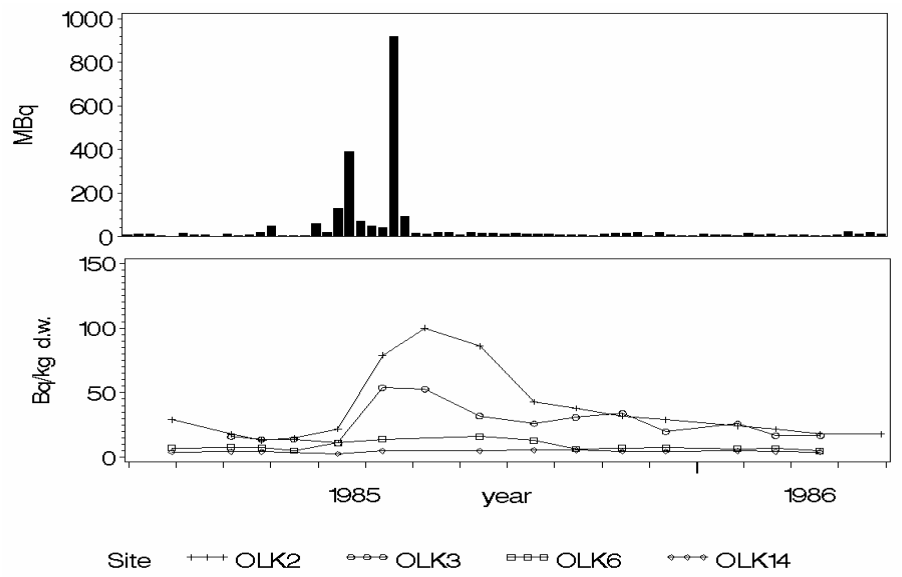


**Fig. 107.** Time variation of measured $^{54}$Mn activity concentrations in *Fucus vesiculosus* at four sampling sites in the discharge area of Olkiluoto (below) and the weekly $^{54}$Mn activity discharges from the Olkiluoto power plant in 1985–April 1986 (above).

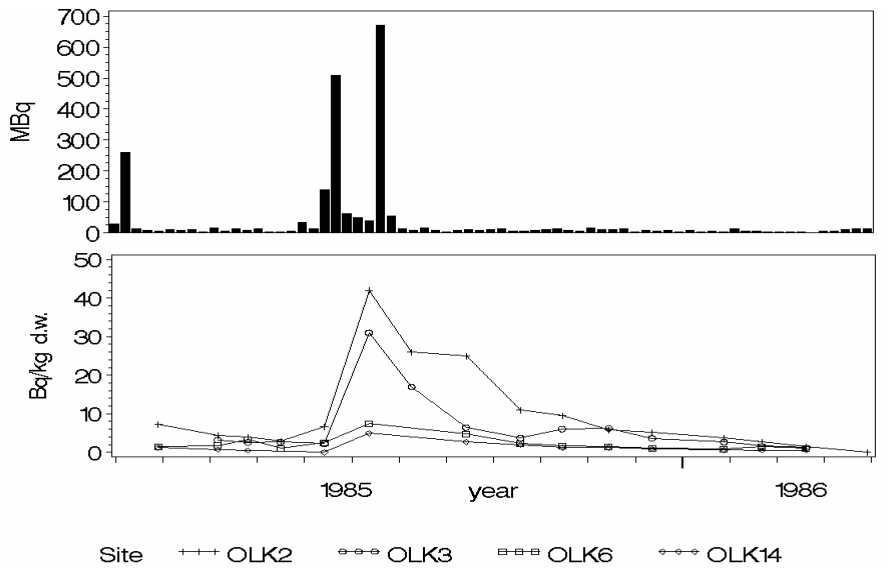


**Fig. 108.** Time variation of measured $^{58}$Co activity concentrations in *Fucus vesiculosus* at four sampling sites in the discharge area of Olkiluoto (below) and the weekly $^{58}$Co activity discharges from the Olkiluoto power plant in 1985–April 1986 (above).

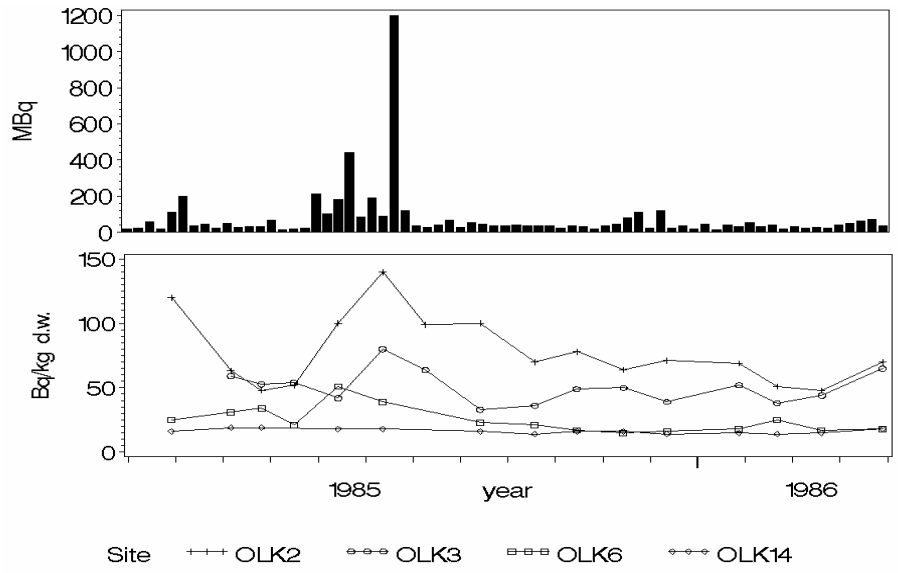


**Fig. 109.** Time variation of measured $^{60}$Co activity concentrations in *Fucus vesiculosus* at four sampling sites in the discharge area of Olkiluoto (below) and the weekly $^{60}$Co activity discharges from the Olkiluoto power plant in 1985–April 1986 (above).

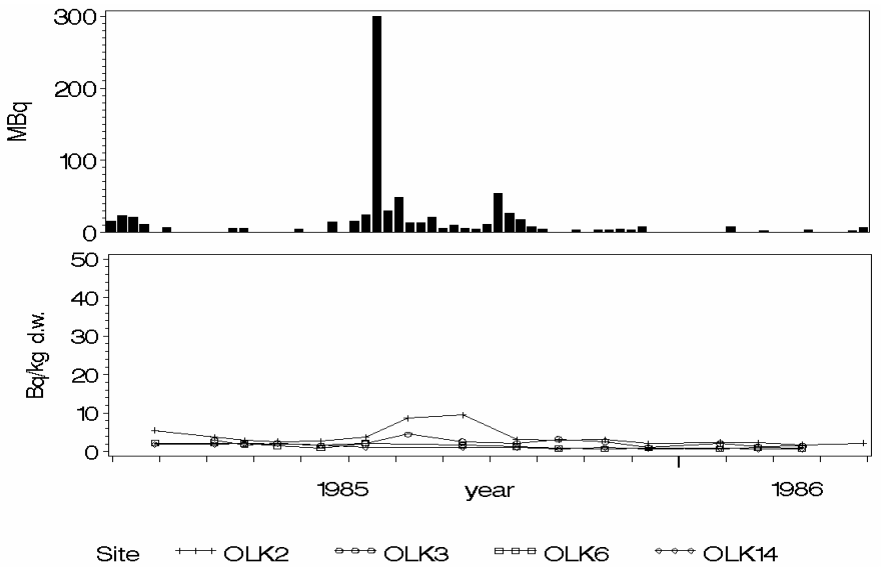


**Fig. 110.** Time variation of measured $^{110m}$Ag activity concentrations in *Fucus vesiculosus* at four sampling sites in the discharge area of Olkiluoto (below) and the weekly $^{110m}$Ag activity discharges from the Olkiluoto power plant in 1985–April 1986 (above).

### 3.6.3 Studies on indicator organisms

Indicator organisms and other indicator samples have been succesfully used in the monitoring of radioactive substances in the environment. The main advantage of indicator organisms is their ability to accumulate radionuclides effectively (and often very rapidly) from the surrounding medium, and consequently, small quantities of them in the environment can be easily detected in indicator organisms, even though their concentrations in the medium (e.g., in air, soil, water or sediments) are still below the detection limits. Some indicators are specific for certain radionuclides, which enhances their usefulness in monitoring programmes. Algae such as *Fucus vesiculosus* and related species have been widely used as indicator organisms (Dahlgaard 1994). On the other hand, the common awareness about the accumulation of many harmful substances in the top consumers in food chains (such as predatory fish and birds or seals) and to sensitive stages of reproduction, have focussed attention on these objects and the search for new indicator objects.

Indicator organisms have been broadly used in the monitoring of radioactivity in the environments of the Swedish nuclear power plants. Much weight has been given to *Fucus vesiculosus,* because it generally has high activity concentrations and discloses the presence of a large variety of radionuclides in comparison with other indicator organisms (Wallberg and Moberg 2002, Notter 1983, Neumann et al. 1991). In addition, benthic diatoms have been shown to accumulate radionuclides very efficiently, and have been abundantly used, especially in environmental monitoring at Forsmark (Notter and Snoeijs 1986, Sjoeijs & Notter 1993), but also at other Swedish NPP sites (Snoeijs & Simenstad 1995). Diatoms were not separately used in our studies, though diatoms were undoubtedly important constituents in the periphyton samples.

**Loviisa 1988–1989**

In a special study carried out in the discharge area of Loviisa in 1988–1989, the aim was to take samples from all important members of the local aquatic ecosystem and to survey the radionuclide concentrations at different trophic levels of the ecosystem. The study also aimed to search for potential 'top accumulators' at the highest level of the food web, and to find new sampling objects for the monitoring programmes (Ilus et al. 1992a, Ilus 1995). Furthermore, the objective was to make a total inventory of Chernobyl-derived caesium in Hästholmsfjärden Bay. The study was carried out about 2–3 years after the Chernobyl accident, when $^{137}$Cs had presumably reached equilibrium in the aquatic ecosystem, and most of the short-living nuclides had disappeared from nature. For instance, the activity concentrations of $^{137}$Cs in seawater had decreased to 1/20 of its values in May 1986.

Samples were taken from seawater, phytoplankton, zooplankton, aquatic plants, benthic animals, fish and waterfowl, and their inner organs and reproductive products (Table 31, Fig. 111). Seal samples were provided by the Game and Fisheries Research from the eastern Gulf of Finland outside the Loviisa area. $^{110m}$Ag, $^{60}$Co and $^{54}$Mn were almost exclusively detected only at the lower trophic levels of the ecosystem, i.e., in phytoplankton, zooplankton, macrophytes and benthic animals, but not in vertebrates, e.g., fish, waterfowl and seals, or in their inner organs. On the other hand, the Chernobyl-derived isotopes $^{137}$Cs and $^{134}$Cs were most abundant in these latter groups, especially in the muscle tissues of predatory fish and seals (Ilus 1995, Fig. 111 of this paper). In fish-eating waterfowl, the concentrations were a little higher than in roach and Baltic herring. The caesium concentrations were generally lower in the inner organs than in muscle tissues. *Fucus vesiculosus* seemed to be a weaker indicator for radiocaesium than predatory fish, but it was better than most other seaweeds and benthic animals. On the other hand, *Fucus* was proved to be a good indicator for $^{60}$Co and $^{54}$Mn, as was also the relict crustacean *Saduria entomon* for $^{110m}$Ag.

According to a rough estimate based on the results from 1988, about 95% of the total inventory of $^{137}$Cs in Hästholmsfjärden was bound in bottom sediments, about 4% was in seawater and less than 1% in fish and other biota (Ilus et al. 1992a).

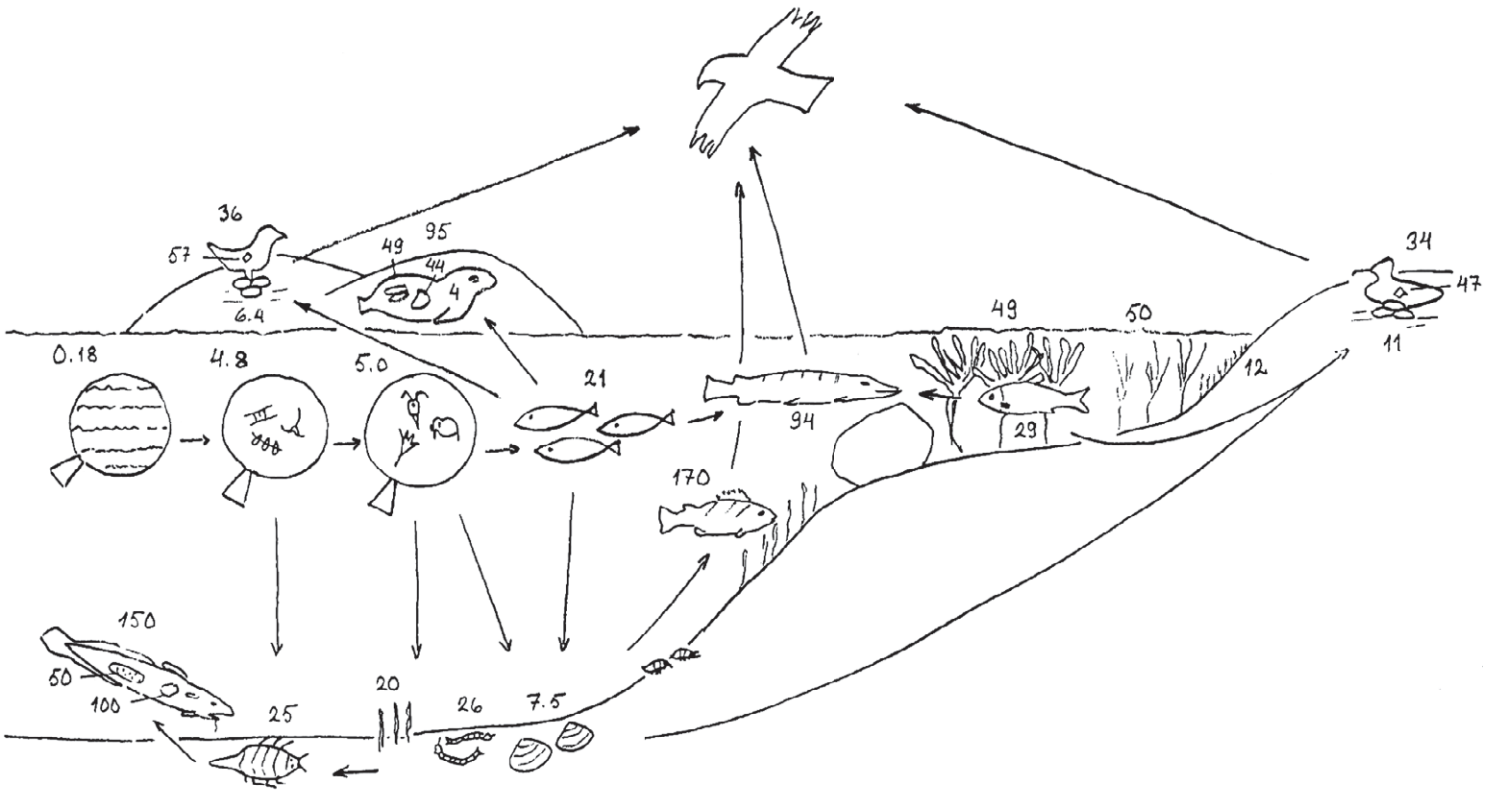


**Fig. 111.** $^{137}Cs$ in some members of the aquatic food web at Loviisa in 1988–1989 (Bq $kg^{-1}$ fresh weight). (Illus et al. 1992a). The osprey is not a member of the local fauna.

**Table 31.** Activity concentrations of gamma emitting radionuclides and $^{90}Sr$ in indicator samples taken from the sea area off the Loviisa Nuclear Power Plant in 1988–1989.

| Sample | | Bq kg$^{-1}$ dry wt. | | | | | | | | | | Bq kg$^{-1}$ f.w. | CR f.w. |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Organism** | **Tissue** | **$^{40}K$*** | **$^{54}Mn$** | **$^{58}Co$** | **$^{60}Co$** | **$^{90}Sr$*** | **$^{106}Ru$** | **$^{110m}Ag$** | **$^{125}Sb$** | **$^{134}Cs$** | **$^{137}Cs$*** | **$^{137}Cs$*** | **$^{137}Cs$*** |
| Phytoplankton | | 1 100 | 6.9* | < | 53* | – | < | 340* | < | 220* | 830 | 4.8 | 27 |
| Zooplankton | | 1 700 | 2.4* | < | 8.3* | – | < | 110* | < | 150* | 600 | 5.0 | 28 |
| **Macroalgae and other aquatic plants** | | | | | | | | | | | | | |
| *Cladophora glomerata* | | 1 900 | 0–3.0 | < | 1.0–7.6 | 2.8 | < | < | < | 31* | 110 | 12 | 67 |
| *Fucus vesiculosus* | | 930 | 0–2.6 | 0–1.2 | 0.8–41 | 20 | 0–34 | 1.7–7.4 | 0– 8.6 | 67* | 270 | 49 | 270 |
| *Myriophyllum spicatum* | | 1 100 | 2.0 | < | 16 | – | < | 9.4 | < | 180 | 630 | 50 | 280 |
| *Phragmites australis* | | 110 | < | < | 0.83 | – | < | 1.7 | < | 60 | 220 | 26 | 140 |
| **Benthic animals** | | | | | | | | | | | | | |
| *Macoma baltica* | | 52 | < | < | 4.3 | – | < | 9.8 | < | 5.2 | 22 | 7.5 | 40 |
| *Potamothrix hammoniensis* | | – | < | < | 7.2 | < | < | < | < | 28 | 120 | 20 | 110 |
| *Saduria entomon* | | 240 | < | < | 8.6 | – | < | 35 | < | 34 | 110 | 25 | 140 |
| *Chironimus plumosus* | | – | < | < | < | – | < | < | < | < | 150 | 26 | 140 |
| **Fish** | | | | | | | | | | | | | |
| *Clupea harengus memb.* | whole** | 470 | < | < | < | – | < | < | < | 21 | 89 | 21 | 120 |
| *Rutilus rutilus* | whole** | 420 | < | < | < | – | < | < | < | 31 | 120 | 29 | 160 |
| *Esox lucius* | fillets | 510 | < | < | < | – | < | < | < | 120 | 440 | 94 | 520 |
| *Lota lota* | flesh | 420 | < | < | < | – | < | < | < | 220 | 630 | 150 | 830 |
| | liver | 190 | < | < | < | – | < | < | < | 150 | 420 | 100 | 560 |
| | entrails | 270 | < | < | < | – | < | < | < | 120 | 370 | 90 | 500 |
| | bones+head+skin | 160 | 1.0 | < | 1.0 | – | < | 4.6 | < | 69 | 220 | 67 | 370 |
| | milt | 390 | < | < | < | – | < | < | < | 55 | 160 | 37 | 210 |
| | spawn | 340 | < | < | < | – | < | < | < | 61 | 190 | 50 | 280 |
| *Perca fluviatilis* | whole** | 460 | < | < | < | – | < | < | < | 170 | 650 | 170 | 940 |

* = mean value, – = not analysed, ** = flesh and bones analysed in one

**Table 31.** Continued.

| Sample | | Bq kg$^{-1}$ dry wt. | | | | | | | | | | Bq kg$^{-1}$ f.w. | CR f.w. |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Organism** | **Tissue** | **$^{40}$K*** | **$^{54}$Mn** | **$^{58}$Co** | **$^{60}$Co** | **$^{90}$Sr*** | **$^{106}$Ru** | **$^{110m}$Ag** | **$^{125}$Sb** | **$^{134}$Cs** | **$^{137}$Cs*** | **$^{137}$Cs*** | **$^{137}$Cs*** |
| **Birds** | | | | | | | | | | | | | |
| *Mergus merganser* | muscle | 330 | < | < | < | – | < | < | < | 37 | 120 | 34 | 190 |
| | heart | 210 | < | < | < | – | < | < | < | 43 | 130 | 35 | 190 |
| | liver | 310 | < | < | < | – | < | < | < | 54 | 160 | 47 | 260 |
| | stomach + intestine | 220 | < | < | < | – | < | < | < | 28 | 93 | 42 | 230 |
| | eggs | 200 | < | < | < | – | < | < | < | 14 | 47 | 11 | 60 |
| *Larus canus* | muscle | 200 | < | < | < | – | < | < | < | 39 | 110 | 36 | 200 |
| | heart | 480 | < | < | < | – | < | < | < | 51 | 78 | 26 | 140 |
| | liver | 170 | < | < | < | – | < | < | < | 56 | 180 | 57 | 320 |
| | stomach + intestine | 150 | < | < | < | – | < | < | < | 24 | 74 | 37 | 210 |
| | eggs | 150 | < | < | < | – | < | < | < | 7.0 | 26 | 6.4 | 36 |
| **Seals** | | | | | | | | | | | | | |
| *Phoca hispida* | muscle | 360 | < | < | < | – | < | < | < | 200 | 460 | 95 | 530 |
| | heart | 530 | < | < | < | – | < | < | < | 90 | 220 | 45 | 250 |
| | liver | 300 | < | < | < | – | < | < | < | 48 | 130 | 44 | 240 |
| | kidney | 230 | < | < | < | – | < | < | < | 69 | 150 | 49 | 270 |
| | train | < | < | < | < | – | < | < | < | 9.0 | 12 | 4.0 | 20 |

* = mean value, – = not analysed, ** = flesh and bones analysed in one

**The INDOFERN project**

In 2000–2001, the indicator values of some aquatic organisms for radioactive substances was studied in the sea areas off Loviisa and Olkiluoto as part of a Joint Nordic NKS Project, INDOFERN, coordinated by the author. The results of the whole INDOFERN project are summarised in Ilus (2006), and the studies carried out at Loviisa and Olkiluoto in Ilus et al. (2006a). The objective of the whole project was to identify new indicator organisms and biomarkers for the assessment of environmental radioactivity in normal and emergency situations. Indicator organisms are valuable monitoring tools in emergency situations, because they can give information about the nuclide composition and dispersion of radionuclides in the early stages of emergency situations. The aim of the study carried out at Loviisa and Olkiluoto was to compare the indicator values of different members of the local aquatic ecosystems with respect to environmental monitoring.

Samples were taken from 27 species, including phytoplankton (9 samples), zooplankton (9 samples), periphyton (12 samples), macroalgae and vascular plants (16 samples), benthic animals (8 samples), fish (20 samples) and birds (6). Special attention was paid to the different tissues and organs of fish and birds, such as flesh, liver, entrails, bones, milt, spawn, eggs, eggshells, etc. (in total 64 samples). The samples were taken from relatively small areas at both Loviisa and Olkiluoto; thus the results are well comparable inside each site.

In both areas, the activity concentrations of $^{40}$K were clearly highest in the filamentous green alga *Cladophora glomerata* and in other aquatic plants. The lowest $^{40}$K concentrations were in the eggshells of birds, in mussels and in the bones of birds and fish (Tables 32 and 33, Fig. 112).

The discharge nuclides from the local nuclear power plants ($^{51}$Cr, $^{54}$Mn, $^{58}$Co, $^{60}$Co, $^{95}$Zr, $^{95}$Nb, $^{110m}$Ag, $^{123m}$Te and $^{124}$Sb) were only detected at the lower trophic levels of the ecosystem. $^{60}$Co was detected in phytoplankton, zooplankton, periphyton (maximum), macroalgae and other submerged aquatic plants (especially Spiked water milfoil, *Myriophyllum spicatum*), and in two mussel species (Common mussel, *Mytilus edulis,* and Baltic Tellin, *Macoma balthica*) and in the relict crustacean *Saduria entomon* (Tables 32 and 33, Fig. 113). $^{54}$Mn, $^{58}$Co, $^{110m}$Ag and $^{123m}$Te were most abundantly detected in periphyton, in submerged aquatic plants (e.g., Hair pondweed, *Potamogeton pectinatus* and *Myriophyllum spicatum*), and in the Bladder-wrack (*Fucus vesiculosus*). $^{124}$Sb was most abundant in *Myriophyllum spicatum* and *Potamogeton pectinatus*. Local discharge nuclides were not detected in fish or birds, nor in their inner organs or reproductive products. This was in agreement with the results of the study carried out in the Loviisa area in 1988–1989 (Ilus et al. 1992a, Ilus 1995).

In both areas, the activity concentrations of $^{137}$Cs per dry weight were highest in periphyton (mean values 240 and 310 Bq kg$^{-1}$ dry weight). However, the $^{137}$Cs

contents were highest in fish flesh, if the results are given on a fresh weight basis (23–38 Bq $kg^{-1}$ fresh weight in perch, *Perca fluviatilis* and 18–26 Bq $kg^{-1}$ in pike, *Esox lucius*). In the muscle tissues of aquatic birds, the $^{137}$Cs concentrations were generally lower than in fish flesh. Likewise, in the inner organs, or in milt and spawn of fish, the $^{137}$Cs concentrations were lower than in fish flesh. In some fish-eating birds (Goosander, *Mergus merganser* and Great black-backed gull, *Larus marinus*), on the contrary, the highest concentrations were not in the muscles but in the liver and entrails. In bird's eggs the concentrations were generally very low, especially in eggshells. In benthic animals, the $^{137}$Cs concentrations were equal to or slightly lower than in aquatic plants (Figs. 114 and 115).

The concentration ratios of $^{137}$Cs (on a fresh weight basis) were calculated using the average $^{137}$Cs concentrations in seawater given before. For the sake of uniformity, the CR values were calculated in the same way for the aquatic birds, their organs and eggs, although their living habits differ markedly from the others (not living in water). The highest concentration factors were those for perch (470–580) and pike (370–380) and their inner organs. On the lower trophic level, the highest concentration ratios were those for *Marenzelleria* sp. (250), periphyton (180–210) and *Fucus vesiculosus* (130–210). In general, the concentration ratios were somewhat higher in the study carried out at Loviisa in 1988–1989, i.e., when the activity concentrations of Chernobyl-derived $^{137}$Cs were still at a higher level, than in those carried out at Loviisa and Olkiluoto in 2000 and 2001. The CR values for macroalgae, vascular plants, crustaceans and fish were somewhat higher in the samples from Loviisa and Olkiluoto, but those for phytoplankton, zooplankton and bivalve mussels were lower than the values given by IAEA and EU ERICA for analogous marine organisms (IAEA 2004b, Hosseini et al. 2008). The higher values at Loviisa and Olkiluoto were certainly due to the low salinity of water. However, all the CR values from Loviisa and Olkiluoto were also clearly lower than the values given in Hosseini et al. (2008) for fresh water organisms. The differences in the concentration ratios of plankton are probably due to the confusing mixed use of fresh and dry weight values in the literature. In 2000–2001, most of the Chernobyl-derived $^{134}$Cs had already disappeared from the environment; only traces were detected in some fish samples, in zooplankton, periphyton and the bladder-wrack.

The best indicators for the local discharge nuclides were organisms from the lower level of the ecosystem (e.g., periphyton, *Myriophyllum spicatum, Fucus vesiculosus,* zooplankton, *Potamogeton pectinatus, Cladophora glomerata, Saduria entomon* and phytoplankton for $^{60}$Co). The best indicators for $^{137}$Cs among the lower organisms were periphyton, zooplankton, *Fucus vesiculosus*, phytoplankton, *Cladophora glomerata,* and *Myriophyllym*, and in vertebrates the predatory fish: perch *(Perca fluviatilis)* and pike *(Esox lucius)*.

**Table 32.** Activity concentrations of gamma emitting radionuclides and $^{90}Sr$ in indicator samples taken from the sea area off the Loviisa Nuclear Power Plant in 2000.

| Sample | | Bq kg$^{-1}$ dry wt. | | | | | | | | | | | | | Bq kg$^{-1}$ f.w. | CR f.w. |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Organism** | **Tissue** | **$^{40}K$*** | **$^{51}Cr$** | **$^{54}Mn$** | **$^{58}Co$** | **$^{60}Co$** | **$^{90}Sr$** | **$^{95}Zr$** | **$^{95}Nb$** | **110mAg** | **$^{123m}Te$** | **$^{124}Sb$** | **$^{134}Cs$** | **$^{137}Cs$*** | **$^{137}Cs$*** | **$^{137}Cs$*** |
| Phytoplankton | | 330 | < | < | < | < | – | < | < | < | < | < | < | 34 | 0.19 | 4 |
| Zooplankton | | 420 | < | < | < | 0–0.49 | – | < | < | < | < | < | < | 49 | 0.37 | 8 |
| Periphyton | | 600 | 0–160 | 1.5–10 | 0–26 | 3.9–15 | – | 0–5.0 | 0–9.6 | 0–58 | 0–1.1 | 0–8.9 | 0–3.1 | 310 | 10 | 210 |
| **Macroalgae and other aquatic plants** | | | | | | | | | | | | | | | | |
| *Cladophora glomerata* | | 1 000 | < | < | < | 0.68 | – | < | < | < | < | < | < | 28 | 2.9 | 62 |
| *Fucus vesiculosus* | | 770 | < | 0–0.95 | 0–1.7 | 0.76–2.4 | 12 | < | < | 0–4.6 | < | < | 0.27–0.68 | 54 | 10 | 210 |
| *Myriophyllum spicatum* | | 490 | < | 3.0 | 6.5 | 6.1 | – | < | < | 16.8 | 0.31 | 2.4 | 0.35 | 24 | 1.9 | 41 |
| *Potamogeton pectinatus* | | 730 | < | 3.7 | 15 | 3.6 | – | < | < | 23 | 0.53 | 2.2 | < | 8.9 | 1.1 | 22 |
| **Benthic animals** | | | | | | | | | | | | | | | | |
| *Macoma balthica* | | 70 | < | < | < | < | – | < | < | < | < | < | < | 2.0 | 0.86 | 18 |
| *Marenzelleria* sp. | | 51 | < | < | < | < | – | < | < | < | < | < | < | 30 | 5.3 | 110 |
| *Saduria entomon* | | 230 | < | < | < | 0.70 | 15 | < | < | 0.46 | < | < | < | 25 | 5.5 | 120 |
| **Fish** | | | | | | | | | | | | | | | | |
| *Clupea harengus memb.* | whole** | 110 | < | < | < | < | 0.054 | < | < | < | < | < | 0.06 | 31 | 7.4 | 160 |
| *Rutilus rutilus* | whole** | 110 | < | < | < | < | – | < | < | < | < | < | 0.029 | 25 | 5.9 | 120 |
| *Esox lucius* | fillets | 110 | < | < | < | < | – | < | < | < | < | < | 0.14 | 84 | 18 | 370 |
| *Perca fluviatilis* | whole** | 100 | < | < | < | < | 0.15 | < | < | < | < | < | 0.15 | 88 | 23 | 470 |
| **Birds** | | | | | | | | | | | | | | | | |
| *Anas platyrhynchos* | egg white | 390 | < | < | < | < | – | < | < | < | < | < | < | 9.8 | 1.16 | 24 |
| | egg yolk | 70 | < | < | < | < | – | < | < | < | < | < | < | 1.6 | 0.73 | 15 |
| | egg shells | | < | < | < | < | – | < | < | < | < | < | < | 2.2 | 1.7 | 36 |

* = mean value, < = below the detection limit, ** = flesh and bones analysed in one

**Table 32.** Continued.

| Sample | | Bq kg$^{-1}$ dry wt. | | | | | | | | | | | | | Bq kg$^{-1}$ f.w. | CR f.w. |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Organism** | **Tissue** | **$^{40}$K*** | **$^{51}$Cr** | **$^{54}$Mn** | **$^{58}$Co** | **$^{60}$Co** | **$^{90}$Sr** | **$^{95}$Zr** | **$^{95}$Nb** | **110mAg** | **$^{123m}$Te** | **$^{124}$Sb** | **$^{134}$Cs** | **$^{137}$Cs*** | **$^{137}$Cs*** | **$^{137}$Cs*** |
| *Larus canus* | muscle | 400 | < | < | < | < | – | < | < | < | < | < | < | 43 | 12 | 260 |
| | liver | 280 | < | < | < | < | – | < | < | < | < | < | < | 29 | 9.2 | 190 |
| | heart+kidney+lung | 330 | < | < | < | < | – | < | < | < | < | < | < | 31 | 6.6 | 140 |
| | stomach+intestine | 430 | < | < | < | < | – | < | < | < | < | < | < | 28 | 7.2 | 150 |
| | egg entrails | 160 | < | < | < | < | – | < | < | < | < | < | < | 8.8 | 2.2 | 46 |
| | embryos | | < | < | < | < | – | < | < | < | < | < | < | 46 | 10 | 210 |
| | egg shells | 61 | < | < | < | < | – | < | < | < | < | < | < | 2.9 | 2.2 | 46 |
| *Sterna hirudo* | egg entrails | 200 | < | < | < | < | – | < | < | < | < | < | < | 4.5 | 1.1 | 24 |
| | embryos | 230 | < | < | < | < | – | < | < | < | < | < | < | 8.4 | 1.2 | 25 |
| | egg shells | | < | < | < | < | – | < | < | < | < | < | < | 3.0 | 2.8 | 58 |

* = mean value, < = below the detection limit, ** = flesh and bones analysed in one

**Table 33.** Activity concentrations of gamma emitting radionuclides in indicator samples taken from the sea area off the Olkiluoto Nuclear Power Plant in 2001.

| Sample | | Bq kg$^{-1}$ dry wt. | | | | | | Bq kg$^{-1}$ f.w. | CR f.w. |
|---|---|---|---|---|---|---|---|---|---|
| **Organism** | **Tissue** | **$^{7}$Be*** | **$^{40}$K*** | **$^{54}$Mn** | **$^{60}$Co** | **$^{134}$Cs** | **$^{137}$Cs*** | **$^{137}$Cs*** | **$^{137}$Cs*** |
| Phytoplankton | | 50 | 340 | < | 2.9–4.3 | < | 63 | 0.33 | 5 |
| Zooplankton | | 34 | 570 | 0–0.67 | 2.0–16 | 0–0.53 | 78 | 0.81 | 12 |
| Periphyton | | 790 | 620 | 0–0.58 | 0–19 | 0–1.2 | 270 | 12 | 180 |
| **Macroalgae** | | | | | | | | | |
| *Cladophora glomerata* | | – | 1 770 | < | 3.5 | < | 59 | 6.3 | 91 |
| *Fucus vesiculosus* | | – | 760 | 0–0.47 | 1.7–5.8 | 0–0.47 | 50 | 9.2 | 130 |
| *Ranunculus peltatus* | | 11 | 750 | 0.79 | 3.1 | < | 17 | 1.1 | 16 |
| *Myriophyllum spicatum* | | – | 600 | < | 8.5 | < | 24 | 2.0 | 29 |
| *Potamogeton perfoliatus* | | – | 350 | < | 1.7 | < | 7.7 | 0.7 | 10 |
| *Potamogeton pectinatus* | | – | 630 | < | 1.8 | < | 25 | 3.2 | 46 |
| **Benthic animals** | | | | | | | | | |
| *Mytilus edulis* | | – | 54 | < | 1.7 | < | 4.5 | 1.2 | 17 |
| *Cerastoderma glaucum* | | – | 50 | < | < | < | 2.5 | 0.91 | 13 |
| *Macoma balthica* | | – | 65 | < | 0.5 | < | 11 | 3.6 | 52 |
| *Marenzelleria* sp. | | – | | < | < | < | 140 | 17 | 250 |
| *Saduria entomon* | | – | | < | < | < | 24 | 8.6 | 130 |
| **Fish** | | | | | | | | | |
| *Clupea harengus memb.* | whole** | – | | < | < | < | | 11 | 160 |
| | milt | – | 350 | < | < | < | 28 | 5.8 | 84 |
| | spawn | – | 260 | < | < | < | 17 | 4.6 | 67 |
| *Abramis brama* | whole** | – | | < | < | 0–0.31 | | 5.7 | 83 |
| | spawn | – | 180 | < | < | < | 9.2 | 2.4 | 35 |
| *Rutilus rutilus* | whole** | – | | < | < | 0–0.06 | | 7.6 | 110 |
| *Esox lucius* | fillets | – | 570 | < | < | 0.54–1.2 | | 26 | 380 |
| | liver | – | 390 | < | < | < | 55 | 13 | 160 |
| | entrails | – | 270 | < | < | 0–0.91 | 100 | 24 | 340 |
| | bones | – | 72 | < | < | < | 9.1 | 3.9 | 57 |
| | spawn | – | 290 | < | < | < | 34 | 11 | 160 |
| *Perca fluviatilis* | whole** | – | | < | < | < | | 38 | 540 |
| | liver | – | 360 | < | < | < | 118 | 24 | 350 |
| | spawn | – | 200 | < | < | < | 78 | 16 | 230 |
| **Birds** | | | | | | | | | |
| *Somateria mollissima* | muscle | – | 330 | < | < | < | 13 | 2.7 | 39 |
| | heart | – | 400 | < | < | < | 12 | 2.1 | 30 |
| | liver | – | 330 | < | < | < | 8.5 | 2.0 | 29 |
| | entrails | – | 120 | < | < | < | 3.5 | 1.8 | 26 |
| | egg entralis | – | 110 | < | < | < | 2.0 | 0.86 | 12 |
| | egg shells | – | 27 | < | < | < | 0.63 | 0.43 | 6 |

* = mean value, – = not analysed, ** = flesh and bones analysed in one

**Table 33.** Continued.

| Sample | | Bq kg$^{-1}$ dry wt. | | | | | | Bq kg$^{-1}$ f.w. | CR f.w. |
|---|---|---|---|---|---|---|---|---|---|
| **Organism** | **Tissue** | **$^{7}$Be*** | **$^{40}$K*** | **$^{54}$Mn** | **$^{60}$Co** | **$^{134}$Cs** | **$^{137}$Cs*** | **$^{137}$Cs*** | **$^{137}$Cs*** |
| *Mergus merganser* | muscle | – | 380 | < | < | < | 23 | 6.7 | 96 |
| | heart | – | 300 | < | < | < | 19 | 5.3 | 77 |
| | liver | – | 310 | < | < | < | 27 | 7.9 | 110 |
| | entrails | – | 410 | < | < | < | 16 | 7.2 | 100 |
| | bones | – | 41 | < | < | < | 1.5 | 0.79 | 11 |
| | feathers | – | | < | < | < | < | 3.0 | 43 |
| *Cygnus olor* | egg white | – | 330 | < | < | < | 17 | 2.0 | 29 |
| | egg yolk | – | 66 | < | < | < | 0.80 | 0.42 | 6 |
| | egg shells | – | 10 | < | < | < | 0.50 | 0.41 | 6 |
| *Larus marinus* | muscle | – | 320 | < | < | < | 13 | 4.3 | 62 |
| | heart | – | 240 | < | < | < | 12 | 4.0 | 58 |
| | liver | – | 310 | < | < | < | 16 | 5.2 | 75 |
| | entrails | – | 160 | < | < | < | 10 | 4.9 | 71 |
| | fat | – | | < | < | < | 2.0 | 1.1 | 16 |
| | bones | – | 26 | < | < | < | 1.1 | 0.55 | 8 |
| | egg entrails | – | 160 | < | < | < | 5.1 | 1.2 | 17 |
| | egg shells | – | 15 | < | < | < | 0.71 | 0.47 | 7 |
| *Larus argentatus* | egg entrails | – | 160 | < | < | < | 2.0 | 0.49 | 7 |
| | egg shells | – | | < | < | < | 1.3 | 1.0 | 15 |

* = mean value, – = not analysed, ** = flesh and bones analysed in one

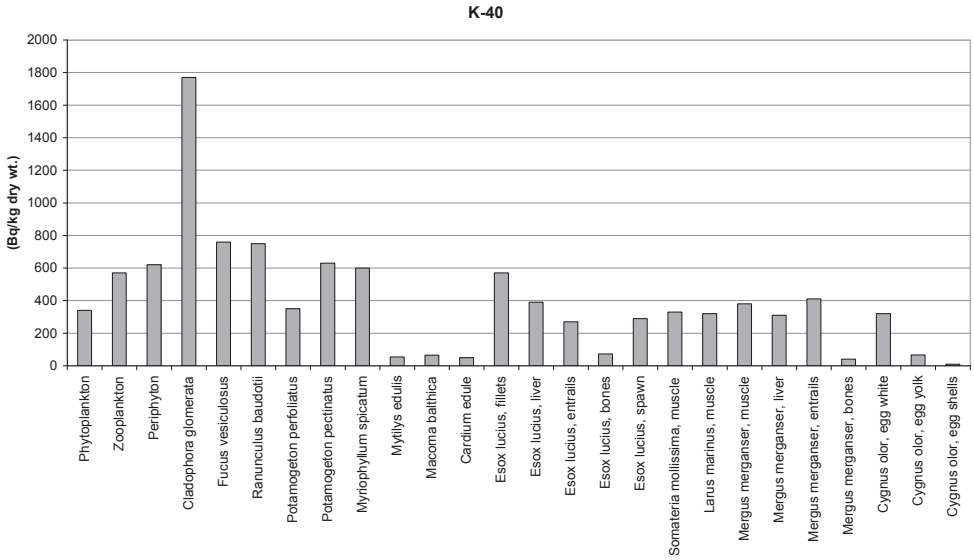


**Fig. 112.** $^{40}$K mean values (Bq kg$^{-1}$ dry weight) in some indicator samples in the sea area off Olkiluoto in 2001 (Ilus et al. 2006a).

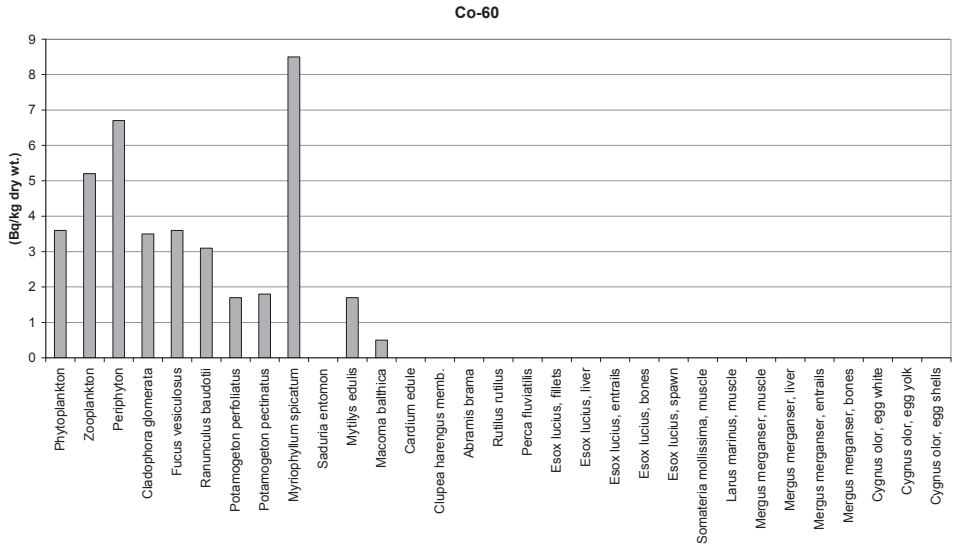


**Fig. 113.** $^{60}$Co mean values (Bq kg$^{-1}$ dry weight) in some indicator samples in the sea area off Olkiluoto in 2001 (Ilus et al. 2006a).

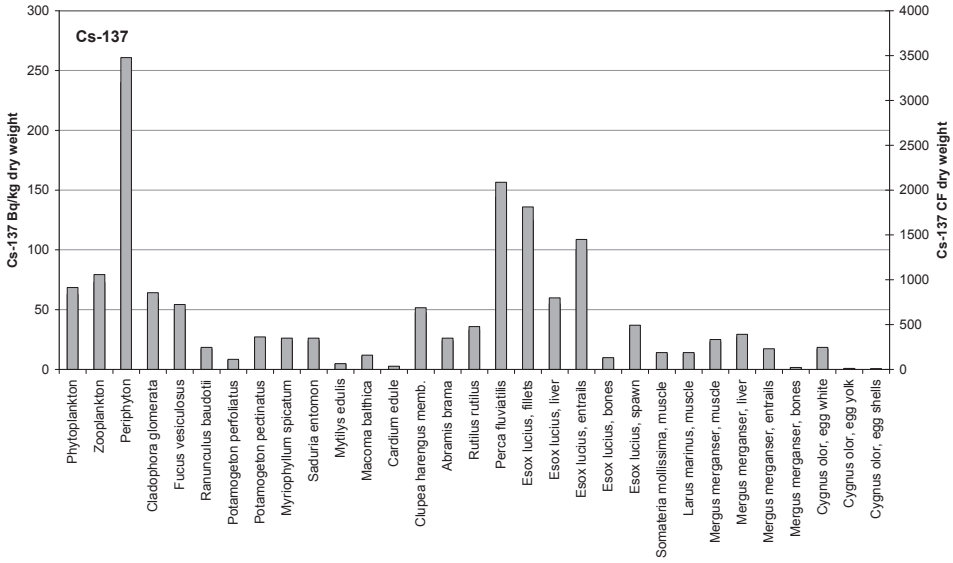


**Fig. 114.** $^{137}$Cs mean values (Bq kg$^{-1}$ dry weight) and mean concentration factors (on a dry weight basis) in some indicator samples in the sea area off Olkiluoto in 2001 (Ilus et al. 2006a).

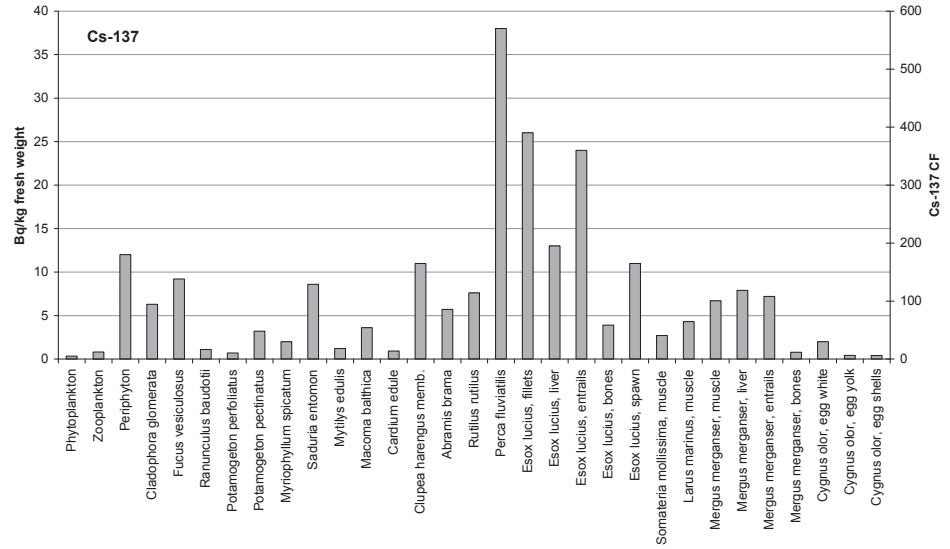


**Fig. 115.** $^{137}$Cs mean values (Bq kg$^{-1}$ fresh weight) and mean concentration factors (on a fresh weight basis) in some indicator samples in the sea area off Olkiluoto in 2001 (Ilus et al. 2006a).

### 3.6.4 Studies on shore soil

Exposure to radioactivity as a consequence of beach occupancy by people belonging to a 'critical group' is one of the most important pathways causing a radiation dose to the public from the aquatic environment in coastal areas (Nielsen 2000). This has also been proved in the vicinities of the Finnish nuclear power plants, being mainly due to the liquid discharges of $^{60}$Co (Klemola et al. 2004, Ikäheimonen et al. 2006 and Ilus et al. 2008).

In 1985–1986, a special survey of shore soil was carried out on the shores of Iso Kaalonpuhti Bay at Olkiluoto (Ilus et al. 1987a). Soil samples were taken with a soil drill at different distances from the shore line. The height of each sampling point above sea level was determined with a theodolite. It was proved that the $^{60}$Co content in the shore material depends on the height of the sampling site above the mean water level. It was likely that $^{60}$Co is accumulated in the shore material during high tide periods directly from the water; the $^{60}$Co content correlated with the time that the water stayed at different levels on the shore (Fig. 116). The highest $^{60}$Co content near the shore line was 660 Bq m$^{-2}$. It was estimated that if members of a theoretical critical group were to stay 700 hours per year on the shore, they would get an annual external radiation dose of $5 \cdot 10^{-7}$ Sv from $^{60}$Co (Ilus et al. *op. sit.*).

Surveys of shore soil were later carried out at Loviisa in 2004 and at Olkiluoto in 2005. In addition to soil, samples of vegetation were also taken from the shore profiles. At Loviisa, three survey transects were set up on the western shores of Hästholmsfjärden at distances of 0.6–1.0 km from the discharge point. The activity concentrations of $^{137}$Cs varied from 86 to 6 000 Bq kg$^{-1}$ d.w. in the soil samples, and from 0.64 to 140 Bq kg$^{-1}$ d.w. in the plant samples. The highest concentrations were detected on the transect located farthest (about 1 km) from the discharge point and at a relatively high altitude above sea level (0.5–1 m). Trace amounts of local discharge nuclides ($^{58}$Co, $^{60}$Co, $^{110m}$Ag and $^{124}$Sb) were detected in only two samples of aquatic green algae, but not in plants growing on land. The highest activity concentration of $^{60}$Co in the shore soil was 3.9 Bq kg$^{-1}$ d.w. (at an altitude of 0.8 m above the mean water level), while the maximum values of $^{60}$Co per square metre were 24 and 39 Bq m$^{-2}$ close to the shore line. Using the dose conversion coefficient of $^{60}$Co for the effective dose (Shleien et al., 1998) and the occupancy time of the critical group given above (700 hours/year), these quantities of $^{60}$Co would give an effective committed dose of 0.14–0.23 µSv per year.

At Olkiluoto, three survey transects were located on the shores of Iso Kaalonpuhti Bay at distances of 0.25–0.75 km from the mouth of the cooling water channel. The activity concentrations of $^{137}$Cs varied from 41 to 1 600 Bq kg$^{-1}$ d.w. in the soil samples, and from 0.98 to 130 Bq kg$^{-1}$ d.w. in the plant samples.

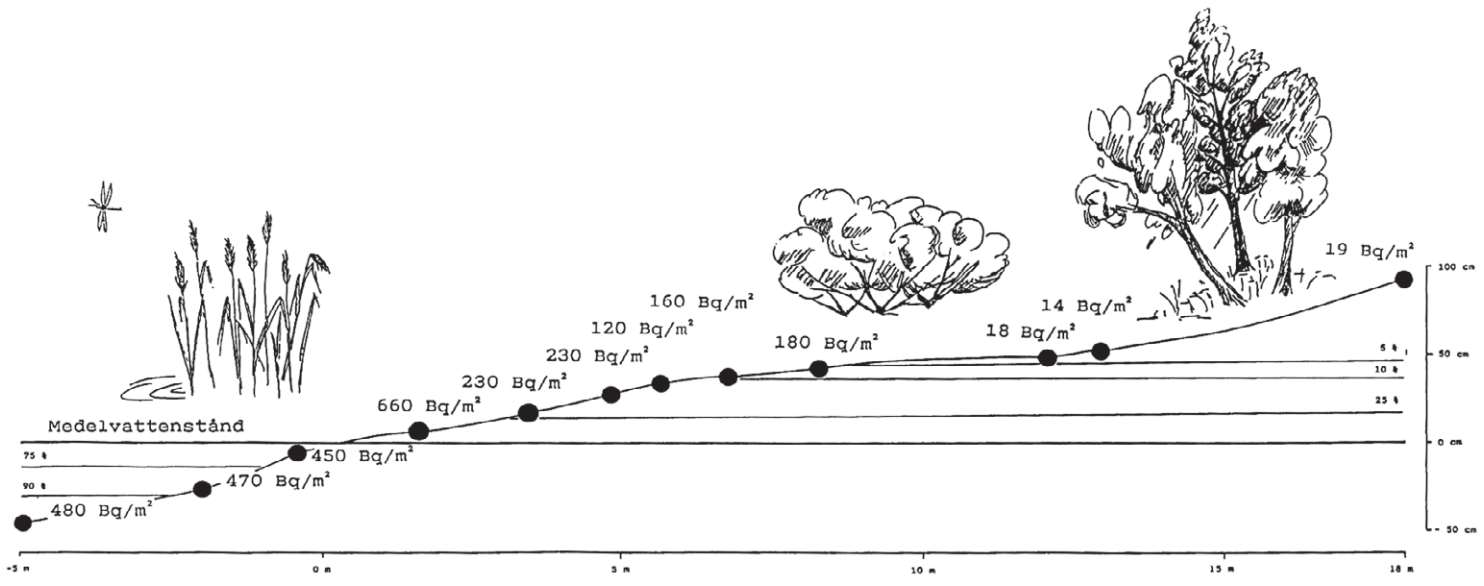


**Fig. 116.** $^{60}$Co content of shore soil in a shore profile of Iso Kaalonpuhti Bay at Olkiluoto in 1986. The percentages show the proportion of the year for which the water is above the marked level (Ilus et al. 1987a).

The $^{137}$Cs concentrations did not correlate with the altitude above sea level at Olkiluoto, either. $^{60}$Co was the only local discharge nuclide detected in the plant samples. The highest concentration was 4.5 Bq kg$^{-1}$ d.w. in dead and dry *Cladophora* that built up into heaps on the shore. The highest activity concentration of $^{60}$Co in shore soil was 14 Bq kg$^{-1}$ d.w. on the transect nearest to the discharge channel at an altitude of 26 cm above the mean water level. The maximum values of $^{60}$Co per square metre were 160 and 140 Bq m$^{-2}$ at altitudes of 21 and 27 cm above the shore line. Using the dose conversion coefficient of $^{60}$Co for effective dose (Shleien et al., *op. cit.*) and the occupancy time of the critical group given above (700 hours/year), these quantities of $^{60}$Co would give an effective committed dose of 0.83 – 0.95 µSv per year.

## 3.7 Radioactive substances in fish

### 3.7.1 Monitoring objects and sampling network

In the permanent environmental monitoring programmes of the Loviisa and Olkiluoto NPPs, radioactive substances are followed in four fish species

- Baltic herring *Clupea harengus membras*
- roach *Rutilus rutilus*
- pike *Esox lucius*
- perch *Perca fluviatilis.*

Baltic herring is the most important catch of commercial fishery in Finland. It is a subspecies of herring adapted to the brackish-water conditions of the Baltic Sea. It is smaller than herring and contains less fat. Baltic herring is a pelagic schooling fish that migrates widely in connection with spawning and feeding. Fry and young fish feed on zooplankton, and older specimens mainly on zooplankton, crustaceans and small fish.

Roach is one of the most common fish species in Finland. It is generally very stationary, occupying mainly benthic living habits. Its food consists of planktonic crustaceans, benthic animals (gastropods and mussels) and even plants.

Pike is the most common big predatory fish in Finnish coastal waters. It is one of the most important objects of free-time fishing in the country. It is very stationary, and favours vegetation-rich littoral areas. Big pikes feed on fish, frogs, young waterfowl, etc.

Perch, too, is one of the most common fish species in Finland, and likewise a very popular catch for free-time fishing. Similarly to pike, it favours the vegetation-rich littoral zone, but it also makes short-distance migrations between the feeding and reproduction areas. The nourishment of perch is wide-ranging: zooplankton, benthic animals and fish.

Samples of each fish species were taken twice a year (in May and in September) from two areas; from the discharge area (Area I) and from a reference area (Area II). At Loviisa, the reference area is located on the west side of Hästholmen Island, in Hudöfjärden, while at Olkiluoto it is on the north side of Olkiluoto Island (Figs. 117 and 118). The target sample size was 5 kg, consisting of different numbers of specimens, depending on the size of the fish. Pike were scaled and filleted for analysis, but only the heads and entrails were removed from perch and roach, which were also scaled. Baltic herrings were analysed without heads and entrails. All the samples were analysed for gamma-emitting nuclides; $^{89}$Sr and $^{90}$Sr were analysed only in the samples of Baltic herring and perch caught from one area (Area I).

Fish farms that utilize the warm cooling water from the power plants in the breeding of young salmon were operated in context with the Olkiluoto power plant between 1987 and 1997, and at Loviisa since 1987. Fingerlings of rainbow trout *(Oncorhynchus mykiss)*, salmon *(Salmo salar)*, brown trout *(Salmo trutta)* and whitefish *(Coregonus lavaretus)* were principally grown one year in the fish farms before continued their growing in coastal growing bags. 1 kg of fingerlings was taken as a sample once a month during the growing period, and was analysed gamma-spectrometrically as such.

### 3.7.2 Radioactive substances in wild fish

**Loviisa**

Activity concentrations of $^{137}$Cs, other artificial gamma-emitting radionuclides and $^{90}$Sr in the fish samples caught in the permanent monitoring programme at Loviisa are summarised in Table 34. During the 1970s and early 1980s, the concentrations of the weapons-tests $^{137}$Cs decreased steadily in all the fish species monitored. In 1985 (one year before the Chernobyl accident), the concentrations were on an average 4.7, 2.9, 1.5 and 1.1 Bq kg$^{-1}$ fresh wt. in perch, pike, Baltic herring and roach, respectively.

The Chernobyl fallout led to a rapid and noticeable increase in the $^{137}$Cs concentrations, but the above-mentioned order in the magnitude of the concentrations in the four species remained unchanged. The maximum concentrations recorded in the fish species after the Chernobyl fallout were 230 Bq kg$^{-1}$ in perch, 99 Bq kg$^{-1}$ in Baltic herring and 53 Bq kg$^{-1}$ in roach in September 1986. The maximum concentration in pike (140 Bq kg$^{-1}$) was reached later, in May 1988 (Fig. 119). The delay in the rise of caesium concentrations in predatory fish was also noticed in freshwater fish. Non-predatory fish already reached their maximum $^{137}$Cs contents in the same year as the deposition

occurred, while predatory fish like pike and large perch had their maxima a year or two later, due to longer food chains (more steps) (*cf.* Saxén 2007). By 1996 (ten years after the accident), the $^{137}$Cs concentrations had decreased on average 85%, and by 2006 (20 years after) 91% from the maximum values.

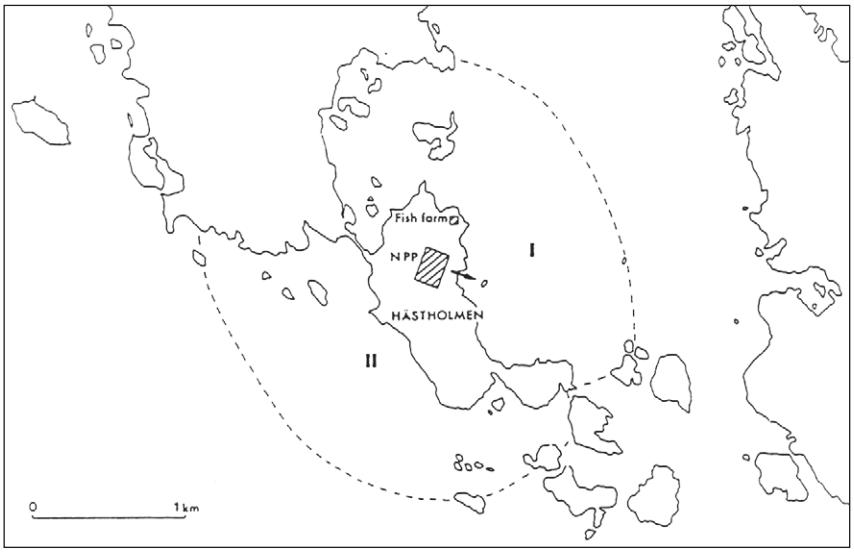


**Fig. 117.** Fishing areas at Loviisa.

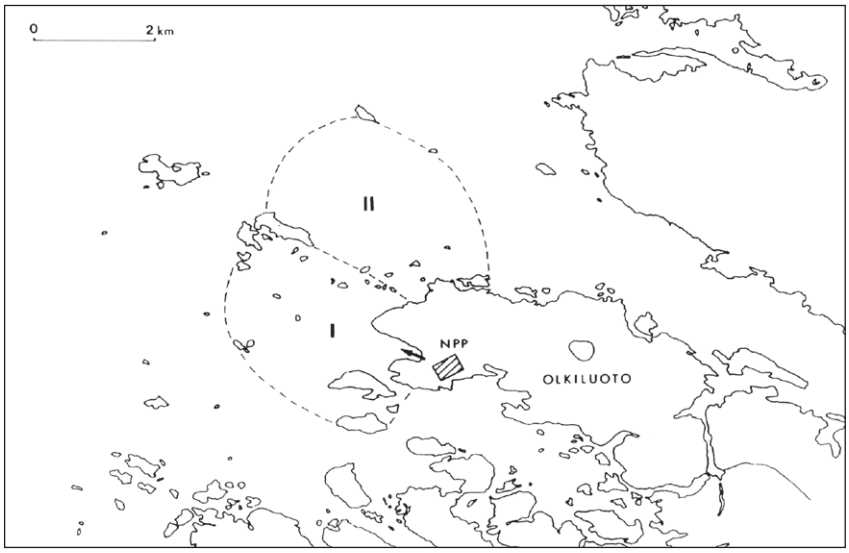


**Fig. 118.** Fishing areas at Olkiluoto.

In 1986–1990, the $^{137}$Cs concentrations in Baltic herring fluctuated regularly in both sampling areas at Loviisa, with the values decreasing in winter, but a new increase occurring in summer (Fig. 120). This was assumed to be associated with the seasonal migrations of Baltic herring. In general, the species migrates for the winter to deeper offshore waters, where the activity concentrations of $^{137}$Cs were lower than in the shallow coastal waters (Ilus et al. 1989, Sjöblom et al. 1989).

In 1986–1987, the fish samples also contained certain fallout nuclides that were only detected during this period, i.e., were not recorded before the accident nor in later years. In May 1986, small amounts of $^{131}$I (maximum 26 Bq kg$^{-1}$) were detected in almost all fish samples, but because of its short half-life (8 days), it was not detected later. In addition, several samples of each species contained small amounts of $^{140}$La, $^{140}$Ba, $^{103}$Ru, $^{65}$Zn and $^{110m}$Ag (maximum concentrations 10, 5.8, 4.5, 0.2 and 0.1 Bq kg$^{-1}$, respectively) during and immediately after the fallout period (1986–1987).

Since 1988, other nuclides than $^{40}$K, $^{90}$Sr, $^{134}$Cs and $^{137}$Cs have not been detected in the fish samples from Loviisa, excluding one detection of $^{110m}$Ag in a sample of Baltic herring from Hästholmsfjärden in 1989 (0.1 Bq kg$^{-1}$). The Chernobyl accident did not noticeably affect the $^{90}$Sr concentrations in fish. Strontium is mainly accumulated in the bones of fish, and the highest concentration of $^{90}$Sr in the fish samples from Loviisa was 1.6 Bq kg$^{-1}$ in a perch sample from 2005. Perch was analysed as scaled, without heads and entrails, but with the bones.

**Table 34.** Summary of data on $^{137}$Cs, $^{90}$Sr and other gammanuclides except $^{40}$K and $^{134}$Cs, in fish caught in the permanent monitoring programme at Loviisa.

| | Baltic herring | Roach | Pike | Perch |
|---|---|---|---|---|
| $^{137}$Cs: mean in 1985 (Bq kg$^{-1}$ f.w.) | 1.5 ± 0.1 | 1.1 ± 0.1 | 2.9 ± 0.5 | 4.7 ± 0.9 |
| $^{137}$Cs: max. after Chernobyl (Bq kg$^{-1}$ f.w.) | 99 | 53 | 140 | 230 |
| Date of max. | 5.9.1986 | 23.9.1986 | 8.5.1988 | 19.9.1986 |
| $^{137}$Cs: mean in 1996 (Bq kg$^{-1}$ f.w.) | 10.3 ± 0.3 | 7.5 ± 0.9 | 24.5 ± 6.4 | 35.2 ± 5.6 |
| $^{137}$Cs: mean in 2006 (Bq kg$^{-1}$ f.w.) | 5.4 ± 0.2 | 5.5 ± 1.9 | 16.6 ± 3.7 | 17.9 ± 2.7 |
| $^{90}$Sr: max in 1971–2007 (Bq kg$^{-1}$ f.w.) | 0.2 | 0.5 | 0.1 | 1.6 |
| Other nuclides except $^{40}$K, $^{90}$Sr, $^{134}$Cs, $^{137}$Cs | | | | |
| – before Chernobyl | – | – | – | – |
| – in 1986–1987: nuclide (frequency) | $^{131}I_{(2)}$, $^{140}Ba_{(1)}$, $^{103}Ru_{(2)}$, $^{65}Zn_{(1)}$ | $^{131}I_{(2)}$, $^{140}Ba_{(1)}$, $^{103}Ru_{(2)}$, $^{140}La_{(1)}$ | $^{131}I_{(2)}$, $^{140}La_{(2)}$, $^{103}Ru_{(1)}$, $^{65}Zn_{(1)}$, $^{110m}Ag_{(1)}$ | $^{131}I_{(1)}$, $^{140}La_{(1)}$, $^{103}Ru_{(2)}$, |
| – since 1988: nuclide (frequency) | $^{110m}Ag_{(1)}$ | | | |

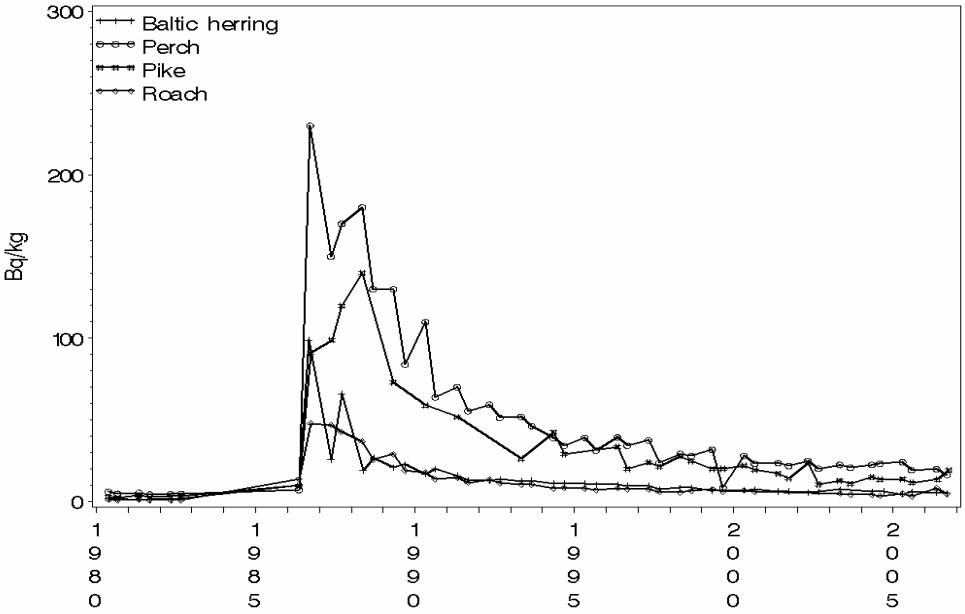


**Fig. 119.** Activity concentrations of $^{137}Cs$ (Bq $kg^{-1}$ f.w.) in four fish species at Loviisa II in 1980–2007.

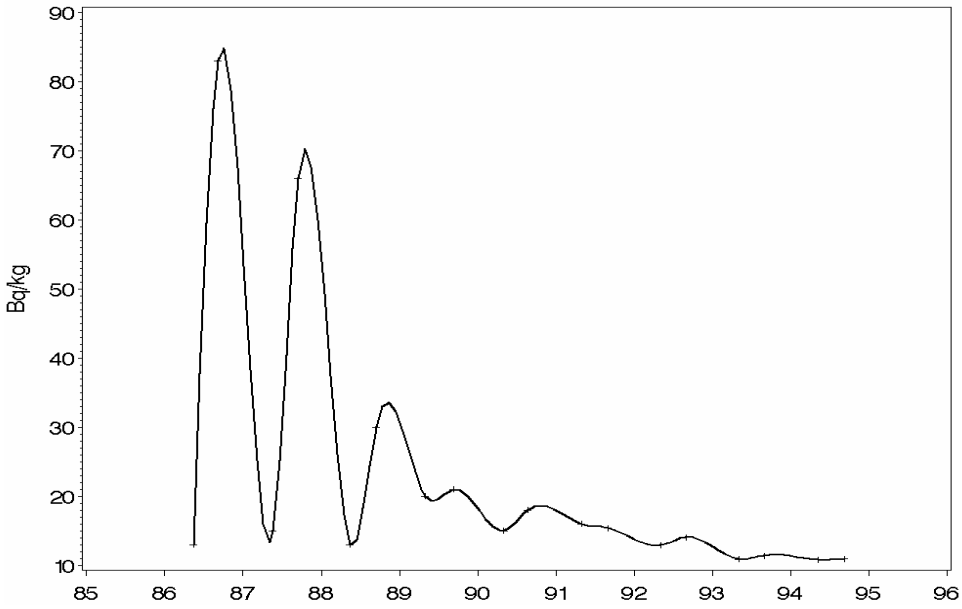


**Fig. 120.** Fluctuation of $^{137}Cs$ in Baltic herring at Loviisa in 1986–1994 after the Chernobyl accident.

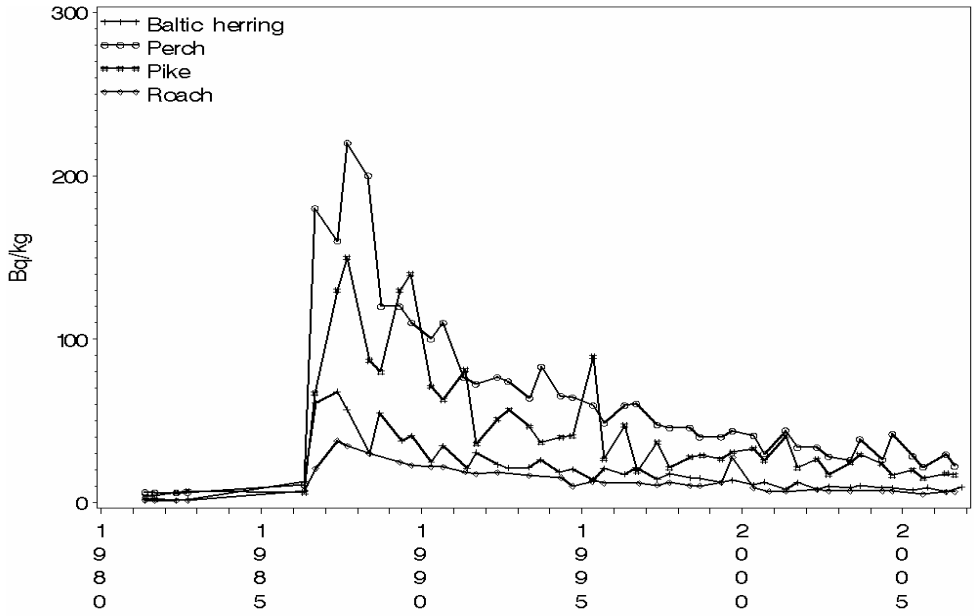


**Fig. 121.** Activity concentrations of $^{137}$Cs (Bq kg$^{-1}$ f.w.) in four fish species at Olkiluoto II in 1980–2007.

**Olkiluoto**

Before the Chernobyl accident, the temporal progress of the $^{137}$Cs concentrations in the four fish species at Olkiluoto was identical with that at Loviisa, and the average concentrations in 1985 were about the same in both areas (*cf.* Tables 34 and 35). In the same way, the maximum concentrations recorded after the Chernobyl fallout were also surprisingly similar at Olkiluoto: 220 Bq kg$^{-1}$ in perch, 150 Bq kg$^{-1}$ in pike, 77 Bq kg$^{-1}$ in Baltic herring and 38 Bq kg$^{-1}$ in roach; all of them in 1987 (Fig. 121). However, the $^{137}$Cs concentrations decreased slightly slower in the Olkiluoto area during the 10 and 20 years after the accident. At Olkiluoto, there were also signs of a fluctuation between the spring and autumn values of $^{137}$Cs in Baltic herring similar to that noticed at Loviisa in the first years after the fallout, but the fluctuation at Olkiluoto was not so regular.

Detection of nuclides other than $^{40}$K, $^{90}$Sr, $^{134}$Cs and $^{137}$Cs was much more frequent at Olkiluoto than at Loviisa. Before the Chernobyl accident, trace amounts (< 0.2 Bq kg$^{-1}$) of $^{58}$Co and $^{60}$Co were detected in one Baltic herring sample taken from Area II in 1984. In the same way as at Loviisa, small amounts of $^{131}$I (maximum 84 Bq kg$^{-1}$) were detected in almost all fish samples in May 1986, but because of its short half-life, it was not detected later. In addition, several samples contained small amounts of $^{129m}$Te, $^{132}$Te, $^{103}$Ru , $^{140}$La, $^{95}$Nb, $^{141}$Ce,

$^{110m}$Ag , $^{65}$Zn, $^{60}$Co and $^{54}$Mn (maximum concentrations 40, 29, 11, 9.0, 1.7, 0.97, 0.75, 0.38, 0.3 and 0.09 Bq kg$^{-1}$, respectively) during and immediately after the fallout period (1986–1987). Most of these observations were made in Baltic herring, and none in roach (Table 35).

Since 1988, small amounts of the local discharge nuclides $^{60}$Co, $^{65}$Zn, $^{54}$Mn and $^{110m}$Ag have every now and then been detected in fish samples taken from both sampling areas, I and II, at Olkiluoto. All the activity concentrations have been very low: from 0.07 to 0.44 Bq kg$^{-1}$ fresh weight. Nine of the 14 observations of local discharge nuclides were made in roach and four in perch, but local discharge nuclides were not found in pike. Ten of the 14 findings were of $^{60}$Co, the last ones being in 2003. Due to the very low concentrations, the findings were more a curiosity in the records than a matter of concern.

### 3.7.3 Radioactive substances in farmed fish

A clear majority of the growing batches in the fish farms of Loviisa and Olkiluoto were fingerlings of rainbow trout *(Oncorhynchus mykiss)*. During the whole operational history of the Loviisa Smoltti fish farm, the samples taken from the establishment contained only small amounts of $^{137}$Cs and $^{134}$Cs besides natural $^{40}$K. During ten years, the activity concentrations of $^{137}$Cs decreased from about 5 Bq kg$^{-1}$ to less than 1 Bq kg$^{-1}$ fresh wt. and the concentrations of $^{134}$Cs sank below the detection level.

In the Olkiluoto fish farm, the trend in the caesium values was almost the same as at Loviisa, but in addition, minor amounts of $^{54}$Mn and $^{60}$Co (0.05 and 0.13 Bq kg$^{-1}$ fresh wt., respectively) were detected once in a rainbow trout sample taken from the establishment in 1989. The low caesium concentrations in the farmed fish were proved to be due to the low content of caesium in the feed used in the farms. In a test analysis carried out in 1988, the $^{137}$Cs content of the feed used in the Olkiluoto farm was 3.7 Bq kg$^{-1}$ dry wt. (Klemola et al. 1991).

**In conclusion**

The activity concentrations of $^{137}$Cs detected after the Chernobyl accident in fish from the sea areas of Loviisa and Olkiluoto, and from the whole Baltic Sea, were notably lower than those reported, e.g., from Finnish lakes. The highest $^{137}$Cs concentration found in Baltic Sea fish was about 300 Bq kg$^{-1}$ fresh wt. in pike caught in 1990 from the sea area off Vaasa (Ilus 2007). The maximum concentrations in Loviisa and Olkiluoto were 230 and 220 Bq kg$^{-1}$ fresh wt. in perch, whereas the maximum found in freshwater fish in Finland was 33 000 Bq kg$^{-1}$ fresh wt. in a pike sample caught in 1987 from the area of highest deposition (Saxén 1990). It is well known that the uptake of caesium by fresh

**Table 35.** Summary of data on $^{137}Cs$, $^{90}Sr$ and other gammanuclides except $^{40}K$ and $^{134}Cs$ in fish caught in the permanent monitoring programme at Olkiluoto.

| | Baltic herring | Roach | Pike | Perch |
|---|---|---|---|---|
| $^{137}Cs$: mean in 1985 (Bq kg$^{-1}$ f.w.) | 1.8 ± 0.3 | 1.3 ± 0.2 | 3.4 ± 0.8 | 4.7 ± 0.2 |
| $^{137}Cs$: max. after Chernobyl (Bq kg$^{-1}$ f.w.) | 77 | 38 | 150 | 220 |
| Date of max. | 10.5.1987 | 8.5.1987 | 7.9.1987 | 9.9.1987 |
| $^{137}Cs$: mean in 1996 (Bq kg$^{-1}$ f.w.) | 19.2 ± 1.8 | 12.0 ± 0.9 | 29.7 ± 12.3 | 61.7 ± 3.1 |
| $^{137}Cs$: mean in 2006 (Bq kg$^{-1}$ f.w.) | 7.7 ± 1.7 | 7.2 ± 2.4 | 18.1 ± 0.7 | 23.1 ± 4.8 |
| $^{90}Sr$: max in 1971–2007 (Bq kg$^{-1}$ f.w.) | 0.1 | 0.2 | 0.1 | 0.8 |
| Other nuclides except $^{40}K$, $^{90}Sr$, $^{134}Cs$, $^{137}Cs$ | | | | |
| – before Chernobyl | $^{58}Co_{(1)}$, $^{60}Co_{(1)}$ | – | – | – |
| – in 1986–1987: nuclide (frequency) | $^{131}I_{(2)}$, $^{129m}Te_{(2)}$, $^{132}Te_{(1)}$, $^{103}Ru_{(2)}$, $^{140}La_{(2)}$, $^{95}Nb_{(1)}$, $^{110m}Ag_{(4)}$, $^{65}Zn_{(2)}$, $^{54}Mn_{(1)}$ | $^{131}I_{(1)}$ | – | $^{60}Co_{(4)}$ |
| – since 1988: nuclide (frequency) | $^{60}Co_{(1)}$ | $^{60}Co_{(5)}$, $^{65}Zn_{(2)}$, $^{54}Mn_{(1)}$, $^{110m}Ag_{(1)}$ | | |

**Table 36.** Concentration ratio CR (Bq kg$^{-1}$ f.w. in fish / Bq kg$^{-1}$ in water) of $^{137}Cs$ for perch, pike, Baltic herring and roach caught from Loviisa I and Olkiluoto I during the whole study period (mean and range).

| Species | Loviisa I | Olkiluoto I |
|---|---|---|
| Perch | 630 ± 170 (470–1 200) | 540 ± 96 (330–740) |
| Pike | 410 ± 81 (310–570) | 350 ± 85 (200–610) |
| Baltic herring | 170 ± 29 (100–240) | 170 ± 25 (130–250) |
| Roach | 110 ± 22 (78–180) | 130 ± 55 (95–390) |

water organisms is orders of magnitude higher than in marine or brackish-water organisms, due to the relatively high concentration of K (a chemical congener of Cs) in seawater (Eisenbud and Gesell 1997). In the same way, the CF for $^{90}Sr$ is greatly affected by the concentration of Ca in the water.

$^{137}Cs$ concentrations increased after the Chernobyl accident most rapidly in plankton-feeding fishes, because plankton was directly contaminated by the radionuclides deposited, and due to the coincidence of the vernal bloom of phytoplankton and the fallout. The delay was longer in predatory fishes, which get their nutriment through a longer food chain. In the lakes, the $^{137}Cs$ concentrations reached peak values in 1987, and in some cases in 1988 (Saxén et al. 1997, Saxén 2007). At Olkiluoto, the peak value in pike was reached in

September 1987 and at Loviisa in May 1988. Both in the lakes and in the coastal areas, the $^{137}$Cs concentrations were generally highest in perch (*cf.* Saxén and Rantavaara 1987).

Mean values of sample specific concentration ratios CR (=concentration factor CF) of $^{137}$Cs between fish and water in the catching areas Loviisa I and Olkiluoto I during the whole study period are given in Table 36. The values verify those given in Tables 31–33, and are somewhat higher than those given in the literature for marine fish (due to the low salinity of water), but clearly lower than those given for freshwater fish (*c.f.* e.g., Hosseini et al. 2008).

## 3.8 Radioactive substances in sinking matter (= suspended particulate matter)

### 3.8.1 Sampling network and methods

Suspended particulate matter can be considered as a non-living indicator of radionuclides in the aquatic environment, as many radionuclides tend to adsorb to particles. The affinity of caesium to clay particles is well-known, but many other nuclides seem to have a similar tendency as well. Since many problems are involved in the sampling of recently-settled particles from the surface of the sediment, proper sediment samples were taken in the monitoring programmes only once every 4 years, and the less frequent sampling was replaced by continuous round-the-year collection of sinking matter.

Sinking particulate matter was collected using buoy-borne sediment traps anchored 1 metre above the sediment surface. The collection was initiated at Loviisa in 1974 and at Olkiluoto in 1977. Over the course of time the sampling method underwent several changes. At the beginning, the collection was done with single funnel-shaped collection vessels supplemented with changeable sample tubes (volume 50 ml), and attached in pairs at different depths to the anchor rope. This procedure was followed at Loviisa in 1974–1979 and at Olkiluoto in 1977–1980. In the next phase, the sediment traps consisted of four parallel collection vessels of the same type as before fixed quadrilaterally to a plastic core, and anchored 1 metre above the sediment surface. These kinds of traps were used at Loviisa in 1980–1988 and at Olkiluoto in 1981–1992.

In 1989, a new type of sediment trap was brought into use at Loviisa. It consisted of four cylindrical collection vessels made of plexiglass, mounted quadrilaterally in a plastic frame. The inside diameter of the cylinders was 92 mm and the height was 300 mm. These traps were used at Loviisa in 1989–2000. The currently-used type of sediment traps consists of three cylinder-shaped collection vessels made of plexiglass, mounted in a triangle-shaped frame (Fig. 122).

**Fig. 122.** A triangle-shaped sediment trap on deck.

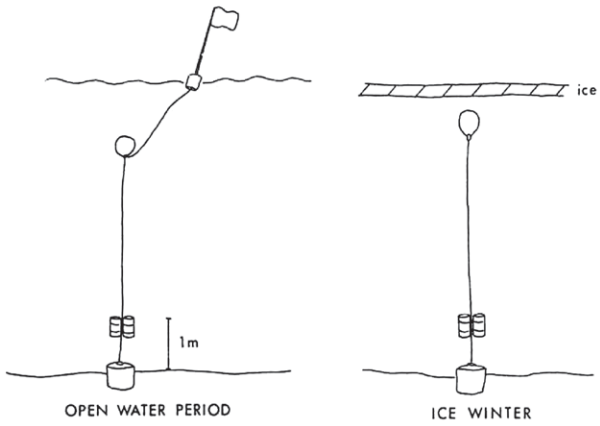


**Fig. 123.** Installation of the sediment trap during the open water and winter periods.

The inside diameter of the cylinders is 92 mm and the height 500 mm; thus the height/diameter (H/D) ratio is 5.43.

The collection vessels (= tubes) were changed four times a year: at the beginning of May, the end of July, the end of August and the beginning of November. The trap was raised to the surface, the tubes were detached, and clean tubes were inserted in their place. The trap was then carefully lowered down again into the water without disturbing the light surface layer of sediment. The boat was anchored during this work.

Sinking matter is collected all round the year; only in the early stages of monitoring was the collection suspended during the freezing and thawing seasons in autumn and spring. Fig. 123 illustrates the different ways the traps are installed during the open water and winter periods. In winter, a big signal buoy is anchored below the ice at a depth of 4–5 metres. As a result, the drifting ice does not remove the trap from its permanent position in the spring. During the open water period, the traps are marked with flag buoys. More detailed descriptions of the traps and the methods are given in Ilus et al. (2008).

At Loviisa, sinking mater was collected at Stations 1, 2, 3 and R1 until 1993. In 1993, the collection at Station 2 was moved to Station 4A situated in Vådholmsfjärden about 4 km south of Hästholmsfjärden (Fig. 63). Station 1 is located in Klobbfjärden and Station 3 at the deepest point of Hästholmsfjärden. Reference Station R1 is located about 14 km west of the power plant. At Olkiluoto, sinking matter was collected at Stations 2, 3 and 4 until 1993. In 1993, the collection at Station 2 was moved to Station 12, and Station 15 was added to the programme as a new sampling station for sinking matter (Fig. 64). Stations 2 and 12 are located southwest of Olkiluoto Island and Stations 3 and 4 to the north of it. Station 15 is located about 10 km north of the power plant. The samples were analysed for gamma-emitting radionuclides and annually-combined samples collected at two stations in both areas were also analysed for $^{238}$Pu and $^{239,240}$Pu.

### 3.8.2 Caesium-137 in sinking matter

**Loviisa**

During the open water periods of 1974–1976 (before the start-up of the power plant), the activity concentrations of $^{137}$Cs varied between 50 and 130 Bq kg$^{-1}$ d.w. in sinking matter collected at Stations 1, 2, 3 and R1. During the next ten years, the concentrations of the fallout-originated $^{137}$Cs declined slowly, and in 1985, the concentrations ranged from 60 to 100 Bq kg$^{-1}$. The Chernobyl fallout raised the caesium concentrations drastically in 1986. The concentrations were

already strongly elevated in the samples from the winter collection period, which terminated on the 8th of May 1986, but the highest concentrations occurred at each station in the samples for the period between the 8th of May and the 12th of June 1986. The maximum values were 48 000, 44 000, 32 000 and 24 000 Bq $kg^{-1}$ d.w. at Stations 3, 2, R1 and 1, respectively, but already during the next collection period between the 12th of June and the 9th of July, the concentrations had decreased to 3 600–10 000 Bq $kg^{-1}$ d.w. Since then, the $^{137}Cs$ concentrations have steadily decreased (Figs. 124–126). In 1990, the mean concentration was 1 700 ± 250 Bq $kg^{-1}$ d.w. at all the stations monitored, while in 2007 the means were 550 ± 29, 440 ± 40, 420 ± 120 and 400 ± 120 Bq $kg^{-1}$ at Stations 1, R1, 3 and 4A, respectively.

**Olkiluoto**

The activity concentrations of $^{137}Cs$ were 89 and 78 Bq $kg^{-1}$ d.w. in the sinking matter collected during the open water period of 1978 at Stations 2 and 4. After the Chernobyl accident, a significant rise occurred in the $^{137}Cs$ concentrations at Olkiluoto too, but somewhat smaller than at Loviisa (Fig. 127). In spring and early summer 1986, the maximum values were 13 000, 10 000 and 6 000 Bq $kg^{-1}$ d.w. at the Stations 3, 4 and 2, respectively. After that, a steady decline in the $^{137}Cs$ concentrations of sinking matter also occurred at Olkiluoto. In 1990, the mean concentration was 1 300 ± 200 Bq $kg^{-1}$ d.w. at all the stations monitored, while in 2007 the means were 340 ± 36, 330 ± 12, 300 ± 19 and 280 ± 17 Bq $kg^{-1}$ at Stations 12, 15, 4 and 3, respectively.

### 3.8.3 Other artificial radionuclides in sinking matter

**Loviisa**

The Chernobyl fallout caused a strong, sudden and transient appearance and rise of some ten radionuclides in sinking matter. Many of these nuclides occurred momentarily in relatively high concentrations in sinking matter samples collected at all the sampling stations (Table 37). However, because the nuclides were almost exclusively very short-lived (e.g. $^{140}La$, $^{131}I$, $^{140}Ba$, $^{141}Ce$, $^{129m}Te$, $^{95}Nb$ $^{103}Ru$ and $^{95}Zr$, with half-lives from less than 2 to 65 days), the concentrations decreased and very soon totally disappeared from the sinking matter samples. The activity concentrations of $^{106}Ru$, $^{110m}Ag$ and $^{144}Ce$ (with half-lives of 250–365 days) also rose significantly as a consequence of the Chernobyl fallout, but remained somewhat longer at elevated levels.

$^{60}Co$, $^{54}Mn$ and $^{58}Co$ are typical constituents of the liquid discharges of the local nuclear power plant. Nevertheless, their concentrations also seemed to

increase, at least slightly, in connection with the Chernobyl fallout. Moreover, in each case their concentrations decreased consistently during the next years after the fallout (Table 37), although relatively high concentrations of them already occurred in sinking matter before the Chernobyl accident (Table 38). In general, the activity concentrations of these three nuclides were clearly lower after the Chernobyl accident than before it, the concentrations decreasing significantly especially during the 1990s. In 2002 and 2006, small peaks in annual mean concentrations of these nuclides occurred in sinking matter collected from Loviisa 3 (Fig. 128). $^{110m}$Ag and $^{125}$Sb were detected in sinking matter both before (1980–1985) and after the Chernobyl fallout (1991–2007).

Local discharge nuclides were rather regularly detected in the sinking matter collected from Hästholmsfjärden and Klobbfjärden (Stations 1, 2 and 3). In addition, $^{60}$Co was abundantly detected in low concentrations at Station 4A, located in Vådholmsfjärden, in front of the straits leading from Hästholmsfjärden Bay to the outer archipelago (distance about 4 km from the outlet). Minor amounts of $^{60}$Co, $^{54}$Mn and $^{125}$Sb were also detected from time to time in the sinking matter collected from Reference Station R1 (Table 38) located in Påsalöfjärden (distance 14 km from the power plant).

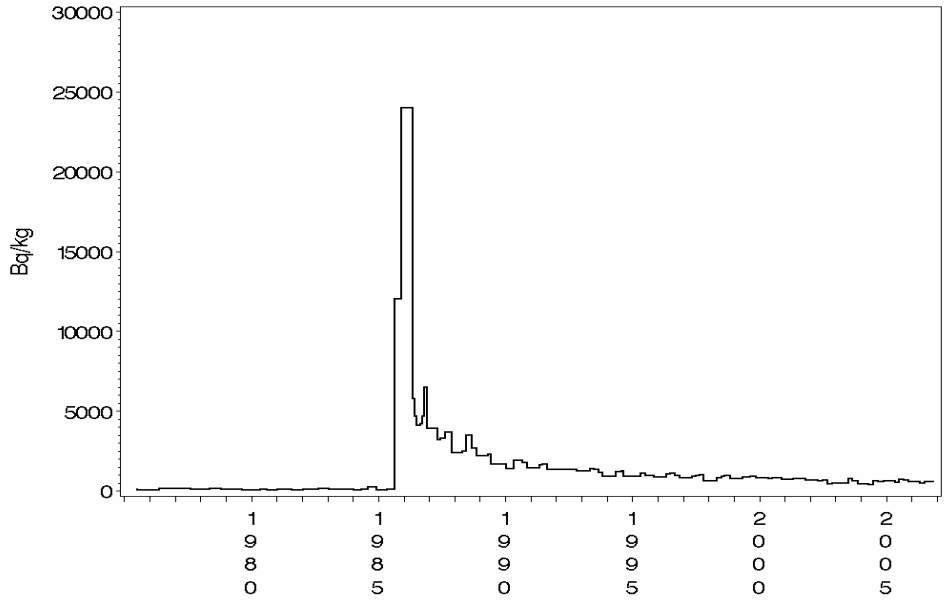


**Fig. 124.** Activity concentrations of $^{137}Cs$ in sinking matter (Bq $kg^{-1}$ d.w.) at Station Loviisa 1 in 1975–2007.

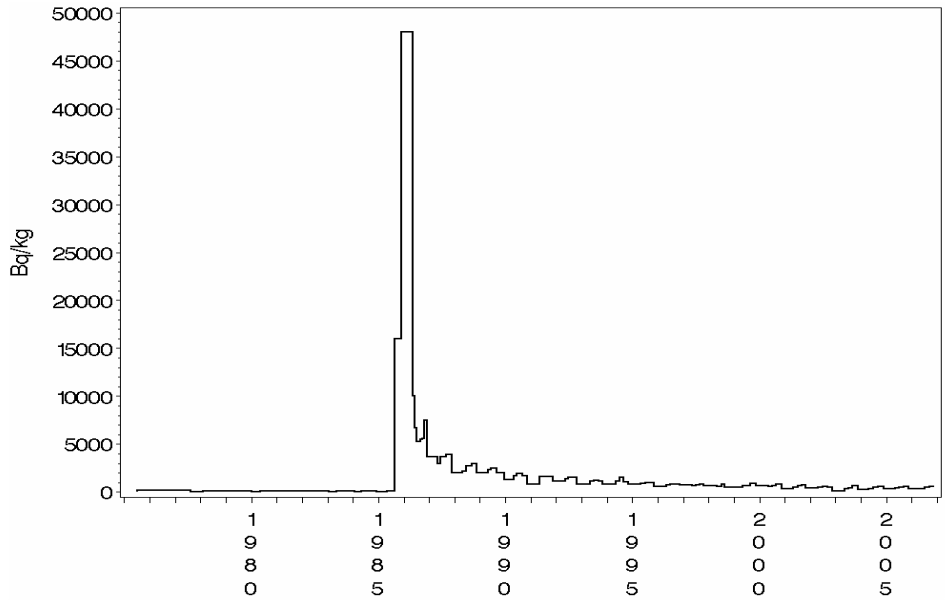


**Fig. 125.** Activity concentrations of $^{137}Cs$ in sinking matter (Bq $kg^{-1}$ d.w.) at Station Loviisa 3 in 1975–2007.

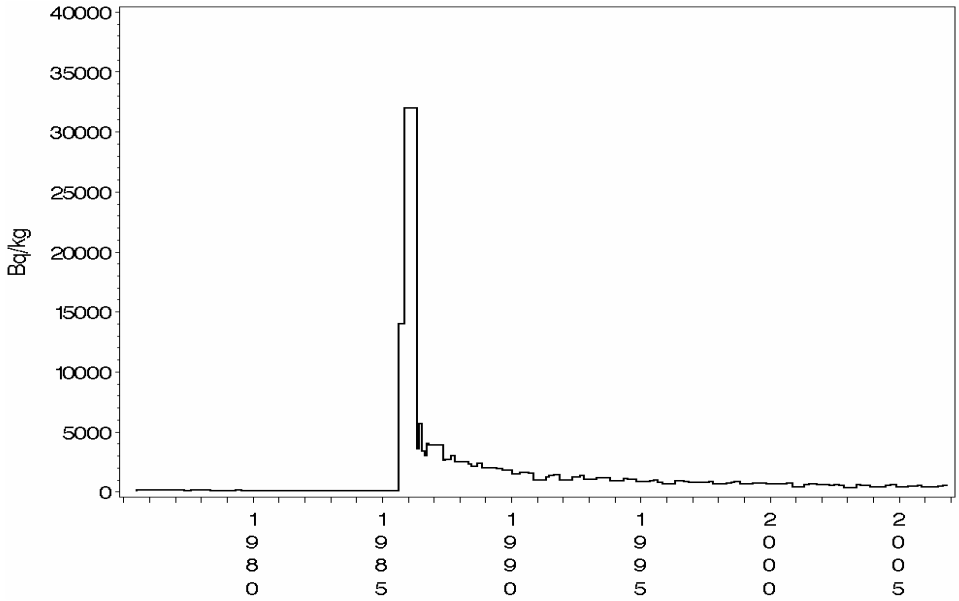


**Fig. 126.** Activity concentrations of $^{137}Cs$ in sinking matter (Bq $kg^{-1}$ d.w.) at Station Loviisa R1 in 1975–2007.

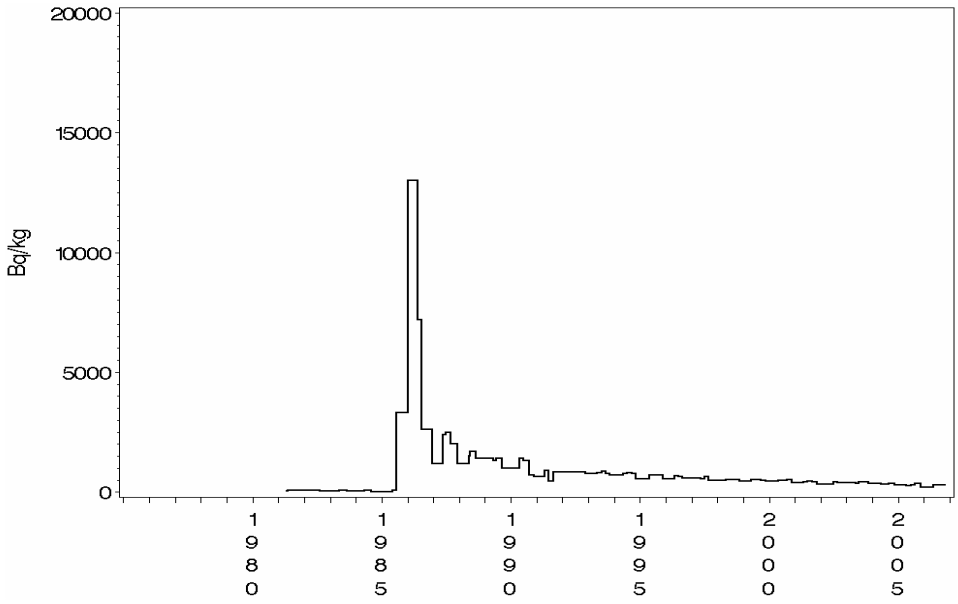


**Fig. 127.** Activity concentrations of $^{137}Cs$ in sinking matter (Bq $kg^{-1}$ d.w.) at Station Olkiluoto 3 in 1981–2007.

**Table 37.** Detection of other gamma-emitting radionuclides except $^{40}K$, $^{134}Cs$ and $^{137}Cs$ in sinking matter collected at the Loviisa 1, 2, 3 and R1 stations during and after the Chernobyl fallout (1986–1990). The observed maximum concentration (+ the year in parentheses), and the last observed concentration (+ the year and month in parentheses) are given. The arrow means that the observations continued after 1990. The concentrations are given in Bq $kg^{-1}$ dry wt.

| | Loviisa 1 | Loviisa 2 | Loviisa 3 | Loviisa R1 |
|---|---|---|---|---|
| Mn-54 | 56 (1986),<br>2.5 (1991 V) | 48 (1986),<br>11 (1991 V) | 110 (1986),<br>19 (1991 V) | 57 (1986),<br>2.6 (1987 XI) |
| Co-58 | | 41 (1987),<br>12 (1990 VIII) | | |
| Co-60 | 270 (1987),<br>18 (1990, →) | 430 (1987),<br>39 (1990, →) | 360 (1987),<br>33 (1990, →) | 4.0 (1988),<br>1.8 (1990 V) |
| Nb-95 | 8 200 (1986),<br>27 (1986 XII) | 7 700 (1986),<br>150 (1987 VIII) | 2 500 (1986),<br>21 (1986 XII) | 7 600 (1986),<br>170 (1986 X) |
| Zr-95 | 440 (1986),<br>57 (1986 IX) | 2 400 (1986),<br>29 (1987 VIII) | 1 100 (1986),<br>68 (1986 IX) | 3 000 (1986),<br>59 (1986 X) |
| Ru-103 | 25 000 (1986),<br>120 (1987 IV) | 42 000 (1986),<br>190 (1986 XII) | 42 000 (1986),<br>360 (1986 XII) | 24 000 (1986),<br>100 (1986 XII) |
| Ru-106 | 9 400 (1986),<br>17 (1990 V) | 14 000 (1986),<br>42 (1990 XI) | 14 000 (1986),<br>39 (1990 XI) | 8 500 (1986),<br>29 (1990 I) |
| Ag-110m | 630 (1986),<br>23 (1990, →) | 1 000 (1986),<br>36 (1990, →) | 1 000 (1986),<br>40 (1990, →) | 690 (1986),<br>2.0 (1990 I) |
| Sb-125 | 1 200 (1986),<br>18 (1990, →) | 1 900 (1986),<br>20 (1990 XI) | 2 300 (1986),<br>27 (1990, →) | 1 200 (1986),<br>12 (1990 XI) |
| Te-129m | 17 000 (1986),<br>770 (1986 VIII) | 27 000 (1986),<br>2 200 (1986 VII) | 25 000 (1986),<br>3 600 (1986 VII) | 16 000 (1986),<br>1 600 (1986 VIII) |
| I-131 | 6 800 (1986),<br>2 000 (1986 VI) | 13 000 (1986),<br>13 000 (1986 V) | 15 000 (1986),<br>4 600 (1986 VI) | 7 100 (1986),<br>1 900 (1986 VI) |
| Ba-140 | 9 700 (1986),<br>5 000 (1986 VI) | 17 000 (1986),<br>12 000 (1986 VI) | 15 000 (1986),<br>9 800 (1986 VI) | 11 000 (1986),<br>5 600 (1986 VI) |
| La-140 | 9 400 (1986),<br>4 500 (1986 VI) | 15 000 (1986),<br>9 600 (1986 VI) | 12 000 (1986),<br>9 700 (1986 VI) | 9 500 (1986),<br>5 700 (1986 VI) |
| Ce-141 | 1 500 (1986),<br>1 500 (1986 VI) | 3 200 (1986),<br>59 (1986 IX) | 1 700 (1986),<br>33 (1986 IX) | 3 300 (1986),<br>18 (1986 X) |
| Ce-144 | 1 700 (1986),<br>1 700 (1986 VI) | 4 600 (1986),<br>15 (1990 XI) | 1 400 (1986),<br>54 (1987 XI) | 2 700 (1986),<br>43 (1987 XI) |

**Table 38.** Detection of other gamma-emitting radionuclides except $^{40}K$, $^{134}Cs$ and $^{137}Cs$ in sinking matter collected at the Loviisa 1, 2, 3, 4A and R1 stations before the Chernobyl accident (1980–1985) and in 1991–2007. The range of observed concentrations (Bq kg$^{-1}$ d.w.) and the frequency of observations during the periods (in parentheses) are given. a = in all samples.

| Nuclide | Loviisa 1 | | Loviisa 2 | | Loviisa 3 | | Loviisa 4A | Loviisa R1 | |
|---|---|---|---|---|---|---|---|---|---|
| | 1980–1985 | 1991–2007 | 1980–1985 | 1991–2007 | 1980–1985 | 1991–2007 | 1993–2007 | 1980–1985 | 1991–2007 |
| Mn-54 | 4.0–58 (8) | 0.8–7.6 (10) | 4.4–75 (9) | 5.9–17 (4) | 4.4–68 (15) | 1.3–166 (22) | 1.2–12 (8) | 3.4–5.0 (2) | – |
| Co-58 | 16 (1) | 3.1 (1) | 30–83 (3) | 10 (1) | 2.9–58 (4) | 1.9–60 (7) | 3.0–6.7 (2) | – | – |
| Co-60 | 4.5–210 (15) | 1.2–41 (65) | 11–320 (15) | 13–140 (9) | 11–260 (a) | 1.3–120 (59) | 0.9–11 (23) | – | 1.0–3.0 (5) |
| Nb-95 | – | – | – | 2.1 (1) | – | – | – | – | – |
| Zr-95 | 7.0–260 (2) | – | 200 (1) | 9.0 (1) | 56–190 (2) | – | – | 59 (1) | – |
| Ru-103 | 220 (1) | – | – | – | – | – | – | – | – |
| Ru-106 | – | – | – | – | 63–69 (2) | – | – | – | – |
| Ag-110m | 2.2–140 (15) | 1.6–46 (21) | 6.8–330 (15) | 7.1–132 (9) | 7.4–240 (a) | 2.0–130 (35) | – | 6.6 (1) | – |
| Sb-124 | – | – | 77 (1) | – | 46 (1) | 2.1–15 (6) | – | – | – |
| Sb-125 | 17 (1) | 4.6–22 (4) | 22 (1) | 14 (1) | – | 7.0–15 (6) | – | – | 7.1–9.4 (2) |
| Ce-144 | 67–100 (2) | – | 61–290 (2) | – | 74–220 (2) | – | – | 52–75 (2) | – |

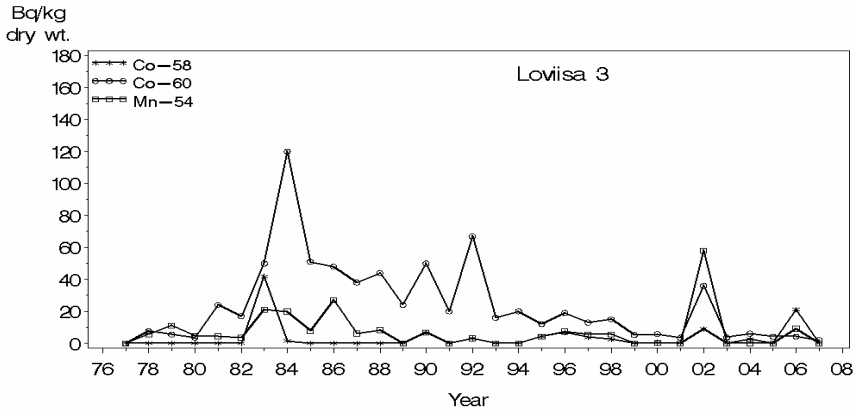


**Fig. 128.** Mean concentrations of $^{54}Mn$, $^{58}Co$ and $^{60}Co$ in sinking matter (Bq kg$^{-1}$ d.w.) at Station Loviisa 3 during the open-water periods in 1977–2006.

**Olkiluoto**

The Chernobyl fallout caused a similar sudden increase and a slow decrease in the concentrations of certain short-lived radionuclides in the sinking matter collected from the Olkiluoto area, as was presented above for the Loviisa area. However, the concentrations were clearly lower at Olkiluoto (Table 39).

In the same way as at Loviisa, the concentrations of $^{60}$Co, $^{54}$Mn and $^{58}$Co seemed to increase slightly in connection with the Chernobyl accident, and then decreased consistently during the years following the accident. Contrary to these three nuclides, $^{51}$Cr and $^{65}$Zn occurred in sinking matter only during the period 1–3 years after the accident. $^{125}$Sb occurred much more frequently at Olkiluoto than at Loviisa, while $^{110m}$Ag has been almost totally missing from sinking matter since 1989. Since the mid- 1990s, the concentrations of the local discharge nuclides decreased strongly, and remained at a very low level in the 2000s (Fig. 129). The power station invested strongly in reducing the discharges, and this was visible in the environmental measurements as a whole in the 1990s and 2000s.

**Table 39.** Detection of other gamma-emitting radionuclides except $^{40}$K, $^{134}$Cs and $^{137}$Cs in sinking matter collected at the Olkiluoto 2, 3 and 4 stations during and after the Chernobyl fallout (1986–1990). The observed maximum concentration (+ the year in parentheses), and the last observed concentration (+ the year and month in parentheses) are given. The arrow means that the observations continued after 1990. The concentrations are given in Bq kg$^{-1}$ dry wt.

| | Olkiluoto 2 | Olkiluoto 3 | Olkiluoto 4 |
|---|---|---|---|
| Cr-51 | 230 (1989), 160 (1989 XII) | 260 (1989), 70 (1989 XII) | 150 (1989), 150 (1989 VI) |
| Mn-54 | 61 (1987), 37 (1990, →) | 110 (1987), 49 (1990, →) | 34 (1986), 36 (1990, →) |
| Co-58 | 34 (1990), 3.5 (1991 VI) | 80 (1987), 13 (1989, →) | 9.0 (1989), 9.0 (1989 XII) |
| Co-60 | 330 (1988), 76 (1990, →) | 120 (1989), 73 (1990, →) | 70 (1990), 70 (1990, →) |
| Zn-65 | 5.5 (1986), 5.5 (1986 XII) | | 8.2 (1986), 8.2 (1986 XII) |
| Nb-95 | 4 600 (1986), 550 (1987 V) | 21 000 (1986), 9 400 (1986 XII) | 11 000 (1986), 220 (1987 V) |
| Zr-95 | 1 000 (1986), 100 (1987 V) | 4 700 (1986), 1 400 (1986 XII) | 5 500 (1986), 580 (1986 XII) |
| Ru-103 | 8400 (1986), 800 (1986 XII) | 14 000 (1986), 1 600 (1986 XII) | 23 000 (1986), 530 (1986 XII) |
| Ru-106 | 3400 (1986), 16 (1991 XII) | 6 600 (1986), 29 (1990 XI) | 5 700 (1986), 16 (1991 XI) |
| Ag-110m | 290 (1986), 30 (1990 V) | 640 (1986), 3.6 (1989 XII) | 420 (1986), 3.7 (1989 XII) |
| Sb-125 | 290 (1986), 13 (1991 XII) | 800 (1986), 4.5 (1991 XII) | 780 (1986), 17 (1991 XI) |
| Te-129m | 8 800 (1986), 8 800 (1986 VII) | 16 000 (1986), 16 000 (1986 VII) | 20 000 (1986), 4 000 (1986 VII) |
| I-131 | | 3 100 (1986), 3 100 (1986 V) | 7 400 (1986), 7 400 (1986 V) |
| Ba-140 | | 4 200 (1986), 4 200 (1986 V) | 12 000 (1986), 12 000 (1986 V) |
| La-140 | | 3 700 (1986), 3 700 (1986 V) | 9 900 (1986), 9 900 (1986 V) |
| Ce-141 | 770 (1986), 110 (1986 XII) | 3 300 (1986), 330 (1986 XII) | 5 200 (1986), 130 (1986 XII) |
| Ce-144 | 840 (1986), 36 (1989 XII) | 4 400 (1986), 9.3 (1989 XII) | 4 100 (1986), 27 (1990 V) |

**Table 40.** Detection of other gamma-emitting radionuclides except $^{40}K$, $^{134}Cs$ and $^{137}Cs$ in sinking matter collected at the Olkiluoto 2, 12, 3, 4 and 15 stations before the Chernobyl accident (1979–1985) and in 1991–2007. The range of observed concentrations (Bq kg$^{-1}$ d.w.) and the frequency of observations during the periods (in parentheses) are given. a = in all samples.

| Nuclide | Olkiluoto 2 | | Olkiluoto 12 | Olkiluoto 3 | | Olkiluoto 4 | | Olkiluoto 15 |
|---|---|---|---|---|---|---|---|---|
| | 1979–1986 | 1991–1992 | 1993–2007 | 1981–1985 | 1991–2007 | 1979–1985 | 1991–2007 | 1993–2007 |
| Mn-54 | 4.8–29 (12) | 7.6–26 (6) | 2.1–58 (15) | 3.8–20 (8) | 2.0–47 (26) | 4.2–28 (11) | 2.7–35 (25) | 1.9–6.5 (10) |
| Co-58 | 11–42 (7) | 3.5 (1) | 2.5–16 (3) | – | 2.7–48 (6) | 11–30 (3) | 2.0–10 (3) | – |
| Co-60 | 13–77 (14) | 23–60 (a) | 1.6–245 (54) | 14–54 (a) | 1.5–98 (60) | 7.4–73 (13) | 1.8–66 (61) | 0.8–26 (42) |
| Zr-95 | 65–120 (2) | – | – | 59–170 (2) | 15 (1) | 100 (1) | – | – |
| Ru-106 | – | 16–51 (3) | – | 49–64 (2) | 15 (1) | – | 16 (1) | – |
| Ag-110m | – | – | – | – | – | – | – | 2.8 (1) |
| Sb-124 | – | – | 8.8 (1) | – | – | – | – | – |
| Sb-125 | – | 9.8–20 (a) | 5.7–7.0 (3) | 15 (1) | 4.5–12 (3) | – | 4.0–17 (7) | 3.2–9.9 (4) |
| Ce-144 | 120–190 (4) | – | – | 130–230 (2) | – | 20–96 (4) | – | – |

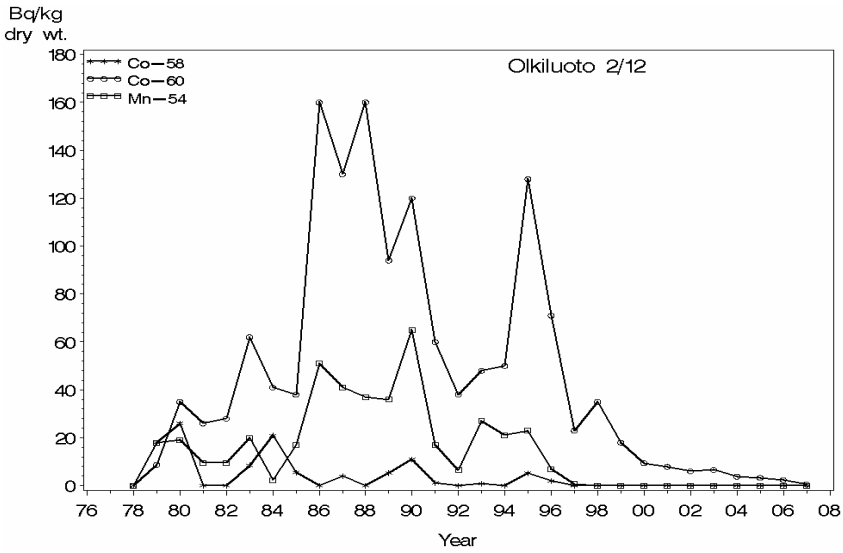


**Fig. 129.** Mean concentrations of $^{54}Mn$, $^{58}Co$ and $^{60}Co$ in sinking matter (Bq kg$^{-1}$ d.w.) at the Olkiluoto 2/12 stations during the open-water periods in 1978–2007.

Local discharge nuclides were mainly detected in sinking matter at Stations 2, 12, 3 and 4, i.e., in the close vicinity of the power plant, on the west and north side of Olkiluoto Island. However, $^{60}$Co was also detected quite regularly and $^{54}$Mn from time to time, in the sinking matter collected from Station 15 situated 10 km north of the power plant (Table 40).

**In conclusion**

The indicator value of sinking matter in the monitoring of radioactive discharges from a power plant is shown in Figs. 130 and 131. In general, there was a good agreement between the peaks in the discharge data and those in the observed concentrations in sinking matter.

The activity concentrations of $^{137}$Cs and $^{134}$Cs were at both Loviisa and Olkiluoto clearly higher in sinking matter than in the indicator organisms. This is due to the affinity of caesium to adsorb to clay particles.

The distribution factor $K_d$ (Bq kg$^{-1}$ d.w. in particulate matter or sediments / Bq kg$^{-1}$ in water) provides a convenient means of describing the relationship between radionuclide concentrations in suspended particulate matter or bottom sediments and in water. $K_d$ values for $^{137}$Cs based on annual mean values in sinking matter and seawater at Loviisa and Olkiluoto before (1978–1985) and after the Chernobyl accident (1987–2007) are given in Table 41.

The $K_d$ values at Loviisa were clearly higher than those at Olkiluoto, reflecting the strong tendency of caesium to associate with clay particles in the sea area off Loviisa. The $K_d$ value given by IAEA (2004b) for $^{137}$Cs in ocean margins is $4 \cdot 10^3$. Thus the values at Olkiluoto were only slightly higher, whereas the values at Loviisa were clearly higher than the values recommended by IAEA for model calculations, but still inside the range given in the publication. In both areas the values were about 1.3–1.5 times higher after the Chernobyl fallout than before it.

**Table 41.** Distribution factors ($K_d$) based on annual mean values of $^{137}$Cs in sinking matter and seawater at all sampling stations at Loviisa and Olkiluoto before and after the Chernobyl accident (mean and range).

| | Loviisa | Olkiluoto |
|---|---|---|
| 1978–1985 | $1.0 \cdot 10^4 \pm 2.4 \cdot 10^3$ ($4.8 \cdot 10^3 - 1.5 \cdot 10^4$) | $4.8 \cdot 10^3 \pm 6.3 \cdot 10^2$ ($3.7 \cdot 10^3 - 6.0 \cdot 10^3$) |
| 1987–2007 | $1.5 \cdot 10^4 \pm 2.0 \cdot 10^3$ ($9.8 \cdot 10^3 - 1.8 \cdot 10^4$) | $6.4 \cdot 10^3 \pm 9.6 \cdot 10^2$ ($3.8 \cdot 10^3 - 8.4 \cdot 10^3$) |

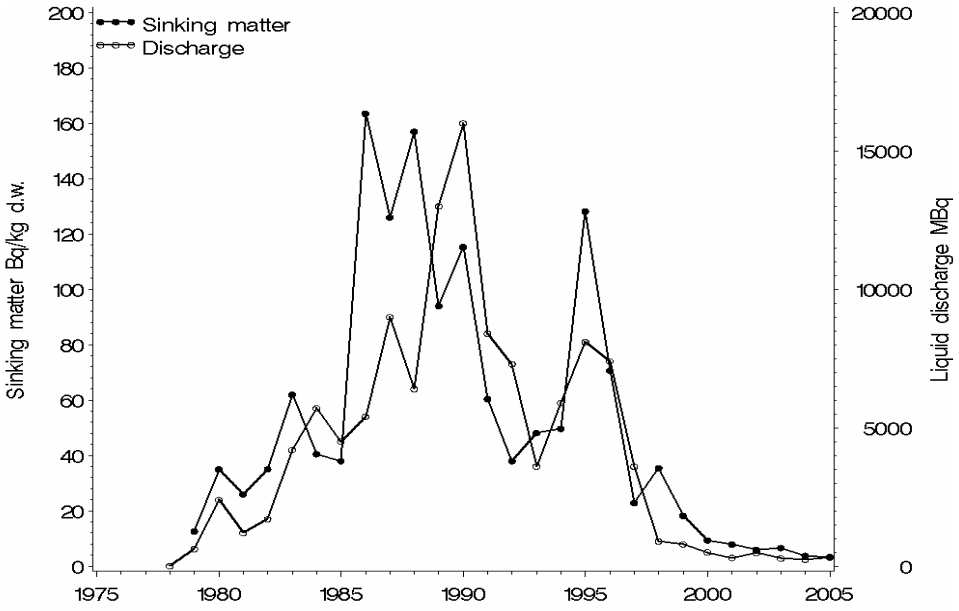


**Fig. 130.** Annual liquid discharges of $^{60}$Co from the Olkiluoto NPP and mean concentrations of $^{60}$Co in sinking matter during the open-water periods at Olkiluoto 2/12 station.

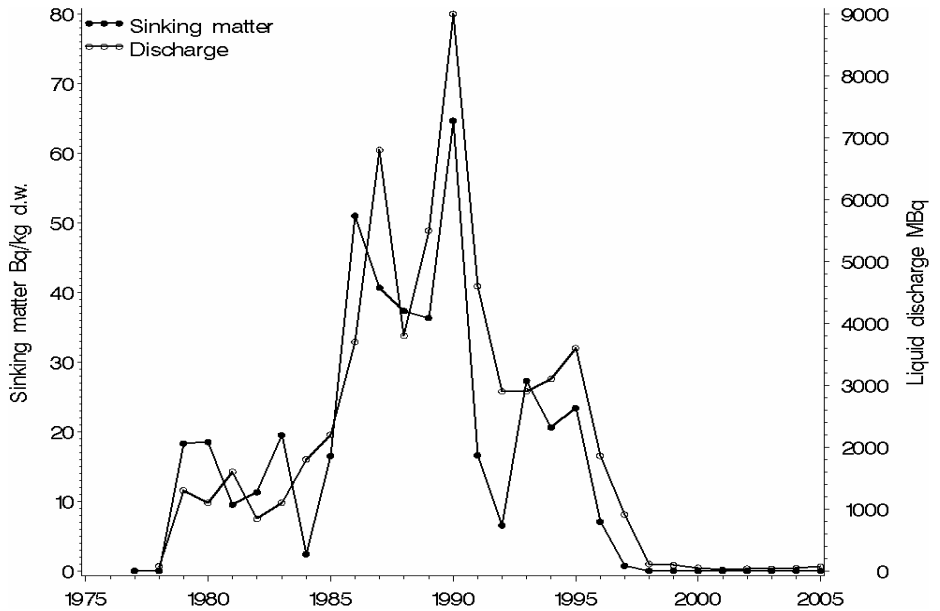


**Fig. 131.** Annual liquid discharges of $^{54}$Mn from the Olkiluoto NPP and mean concentrations of $^{54}$Mn in sinking matter during the open-water periods at Olkiluoto 2/12 station.

## 3.9 Bottom sediments

### 3.9.1 Sampling network and methods

Samples of bottom sediments were regularly collected from the vicinity of the Loviisa NPP beginning in 1975 and from the vicinity of the Olkiluoto NPP beginning in 1976, initially for determination of the background levels of $^{90}$Sr, $^{137}$Cs and $^{239,240}$Pu caused by atmospheric weapons tests, and to establish a basis for the regular monitoring of radionuclides following the start-up of the power plants. At Loviisa, proper sediment samples were taken annually in 1975–1980 at Stations 1, 2, 3 and R1. Since 1983 the samples have been taken every three or four years at 8–10 sampling stations. Besides the stations mentioned above, samples were also taken from Stations 4, 5, 7, 8 and 10 in 1983–1990, and from Stations R2, S1, S2, S5, S6 and 4a in 1994–2006. At Olkiluoto, sediment samples were taken annually in 1976–1980 at Stations 1, 2, 4, 9 and 12. Since 1983, the samples have been taken every four years at 5–11 sampling stations. Besides the stations mentioned above, samples were taken at stations 3, 5, 15 and S1–S8 in 1991–2003. The locations of the sampling stations used in the sediment surveys are shown in Figs. 63 and 64.

In 1975–1991, the sediment samples were primarily taken with a STUK corer. This consists of a stainless steel body containing an inner Plexiglass tube with an inner diameter of 64 mm. The corer is 50 cm long and weights 12.5 kg. The slicing equipment consists of an extruder piston and simple plastic slides for 2–5 cm thick subsamples. Since 1994, the principal equipment used in sediment sampling was a Gemini Twin Corer, which is a gravity corer consisting of two parallel coring tubes in a stainless steel body and two lateral rudder closers. The inner diameter of the Plexiglass coring tubes is 80 mm. The total length of the corer is 132 cm and that of the coring tubes 79 cm. The weight of the corer is 33 kg + 2 or 4 additional weights, 7.4 or 10.2 kg each. The sectioning apparatus of the Gemini Twin Corer consists of a screw-operated extruder piston and a wheel for adjusting the extrusion, a Plexiglass slicing base and a slicing ring with centimetre scaling. In addition to these two principal types of equipment, some other corers (Aquarius Box Corer, Niemistö Corer and Limnos sediment sampler) were used repeatedly for intercomparisons of the sampling techniques carried out at Loviisa (Klemola et al. 1991, Ilus et al. 2000), while an Ekman-Birge grab was employed in some samplings of sandy bottoms at Olkiluoto. All the devices are described in Ilus et al. (2000).

In general, six or five parallel cores were taken with the STUK corer, the cores then being sectioned at 5-cm intervals and the parallel slices combined for analysis. In 1994–1999, the parallel cores obtained by one haul with the Gemini Twin Corer were sectioned at either 1-cm or 5-cm intervals and

the parallel slices were combined for analysis, but since 2002 only one of the parallel Gemini cores was sectioned at 1-cm intervals; the 1-cm slices were then analysed either separately, or as bulk samples representing 5-cm subsamples. The 1-cm subsamples taken by the Niemistö Corer were combined from 10 (1986–1987) or 5 (1988→) parallel cores. In the sectioning process, the sediment slices were moved directly from the slicing base or sample slides into plastic boxes or Petri dishes, in which they were freeze-dried before homogenizing and analysing.

### 3.9.2 Caesium-137, strontium-90 and plutonium-239,240 in bottom sediments

**Loviisa**

The role of $^{137}$Cs in sedimentological studies is particularly important, because of the affinity of caesium to adsorb on clay particles. A certain amount of $^{137}$Cs already occurred in the sediments of Finnish coastal waters before the Chernobyl accident, as a result of the atmospheric nuclear weapons tests carried out in the northern hemisphere in the late 1950s and early 1960s. After the accident, the proportion of the 'old' caesium began to become insignificant, as a consequence of the clearly larger amounts of Chernobyl caesium and since the caesium peak of the old global fallout had already been buried into deeper sediment layers (Ilus et al. 2007).

Fig. 132 illustrates the vertical distribution of $^{137}$Cs in 1-cm-slices at the Loviisa 3 station in 1986–2002 (Ilus et al. 2008). The first profile shows the situation on the 10$^{th}$ of June in 1986, only six weeks after the arrival of the Chernobyl fallout. Fallout caesium (5 800 Bq kg$^{-1}$ d.w.) was then exclusively present in the uppermost centimetre of the sediment. The $^{137}$Cs in the lower slices originated from the weapons-tests fallout. Two years later, in 1988, the caesium peak caused by the Chernobyl accident was in the second centimetre, and in 1990 already in the fifth centimetre. In 2002, fresh sediment containing lower concentrations of caesium had settled on the bottom, and the Chernobyl peak was buried to a depth of 13–14 cm, which corresponds to a sedimentation rate of about 8 mm a$^{-1}$ (Ilus et al. *op cit.*). At the same time, the caesium peak at the Loviisa S5 station was at a depth of 16–17 cm, corresponding to a sedimentation rate of about 10 mm a$^{-1}$ (Table 42).

In the monitoring programmes, the less frequent sampling of sediments is replaced by a continuous collection of sinking matter (p. 270). If we compare the $^{137}$Cs concentrations in the uppermost 0–1-cm sediment layers with those in the recently-settled sinking matter, the concentrations in sinking matter seem to

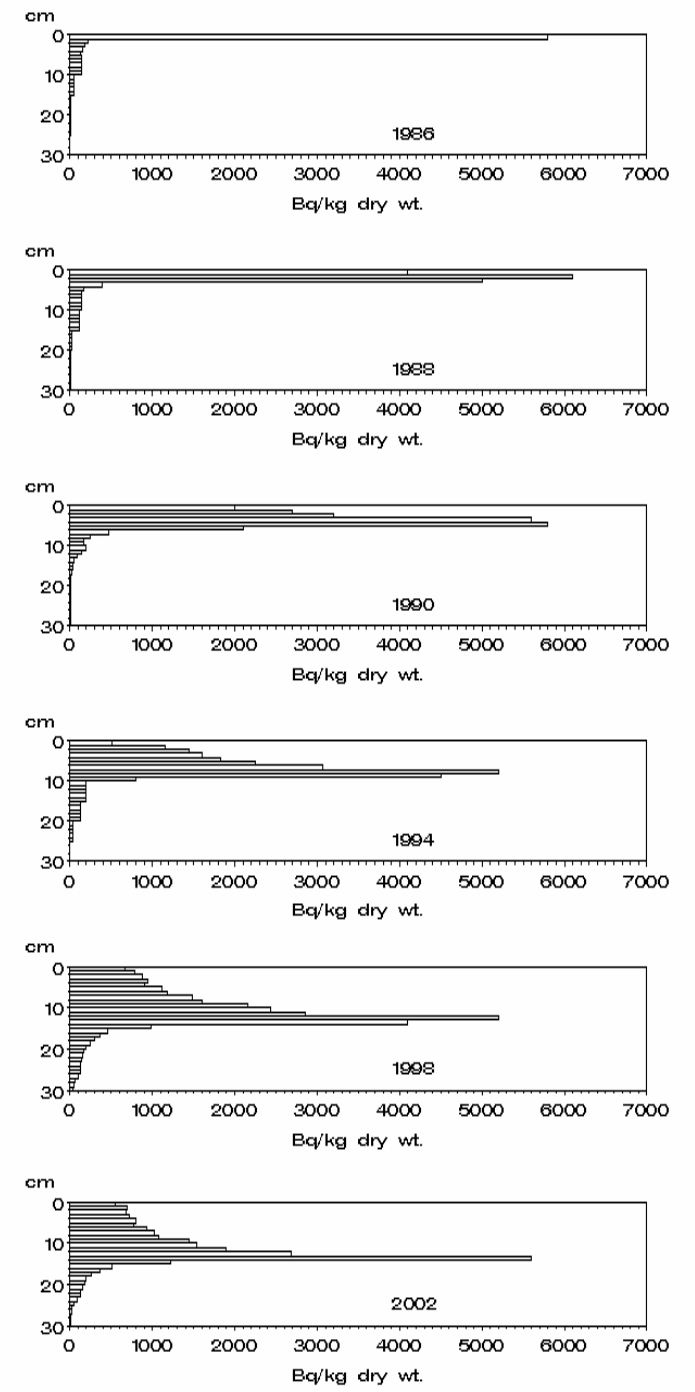


**Fig. 132.** Vertical distribution of $^{137}Cs$ concentrations at the Loviisa 3 station in 1986–2002 (Ilus et al. 2008).

**Table 42.** Activity concentrations of $^{137}$Cs in the surface (0–1cm) layer and at the 'peak depth' at some samplings stations at Loviisa in 2002.

| Station | $^{137}$Cs (Bq kg$^{-1}$ d.w.) in surface 0–1cm layer | $^{137}$Cs peak (Bq kg$^{-1}$ d.w.) | Depth (cm) of the $^{137}$Cs peak |
|---|---|---|---|
| Loviisa 1 | 650 | 1 210 | 10–11 |
| Loviisa 3 | 550 | 5 590 | 13–14 |
| Loviisa 4 | 640 | 3 590 | 15–16 |
| Loviisa 7 | 580 | 1 860 | 10–11 |
| Loviisa 10 | 450 | 4 700 | 14–15 |
| Loviisa R1 | 590 | 1 930 | 15–16 |
| Loviisa R2 | 700 | 1 370 | 6–7 |
| Loviisa S5 | 430 | 5 100 | 16–17 |
| Loviisa S6 | 570 | 1 970 | 9–10 |

**Table 43.** Average activity concentration of $^{137}$Cs in sinking matter collected between the 4$^{th}$ of July 2001 and the 2$^{nd}$ of July 2002 and in the surface sediment layer (0–1cm) of two parallel sediment cores taken in June 2002 at the Loviisa 1, 3 and R1 stations.

| Station | Average $^{137}$Cs conc. in sinking matter during 4 July 2001 – 2 July 2002 | Average $^{137}$Cs conc. in 0–1 cm sediment layer (2 parallel cores) in June 2002 |
|---|---|---|
| Loviisa 1 | 743 ± 35 | 690 ± 57 |
| Loviisa 3 | 585 ± 148 | 575 ± 35 |
| Loviisa R1 | 620 ± 34 | 590 ± 0 |

be a little higher than in the corresponding sediment samples. For example, the average $^{137}$Cs concentrations in sinking matter collected between the 4$^{th}$ of July 2001 and the 2$^{nd}$ of July 2002 at Stations 1, 3 and R1 were about 2–7% higher than those in the surface sediments at the same stations in June 2002 (Table 43). This was probably due to a loss of the flocculent particulate material from the surface layer in connection with the sediment sampling.

The total amounts per square metre at the various stations may vary considerably from year to year (Fig. 133). These differences may arise for several reasons, such as small differences in the exact position of the sampling site, faults versus success in sampling, etc. (Ilus et al. 2000). In general, however, the highest value obtained at a station should arguably be the most representative (Ilus 2001), since mistakes in sampling very seldom lead to increased values in the cumulative total amounts. On the contrary, lower values may well result

**Table 44.** The highest cumulative total amounts of $^{137}Cs$ at the sampling stations in the Loviisa sea area before and after the Chernobyl accident (Bq $m^{-2}$).

| Station | Max. total amount of $^{137}Cs$ before Chernobyl (year), Bq $m^{-2}$ | Max. total amount of $^{137}Cs$ after Chernobyl (year), Bq $m^{-2}$ |
|---|---|---|
| Loviisa 1 | 4 800 (1975) | 34 900 (1998) |
| Loviisa 2 | 2 300 (1978) | 42 400 (1998) |
| Loviisa 3 | 2 300 (1983) | 49 200 (1998) |
| Loviisa 4 | 4 900 (1983) | 61 100 (1990) |
| Loviisa 5 | 2 500 (1983) | 32 100 (1998) |
| Loviisa 7 | 8 500 (1983) | 23 300 (1998) |
| Loviisa 10 | 4 000 (1983) | 47 000 (1994) |
| Loviisa R1 | 8 000 (1978) | 81 300 (1990) |
| Loviisa R2 | | 19 100 (2002) |
| Loviisa S1 | | 10 000 (1994) |
| Loviisa S5 | | 49 600 (2002) |
| Loviisa S6 | | 26 000 (2002) |

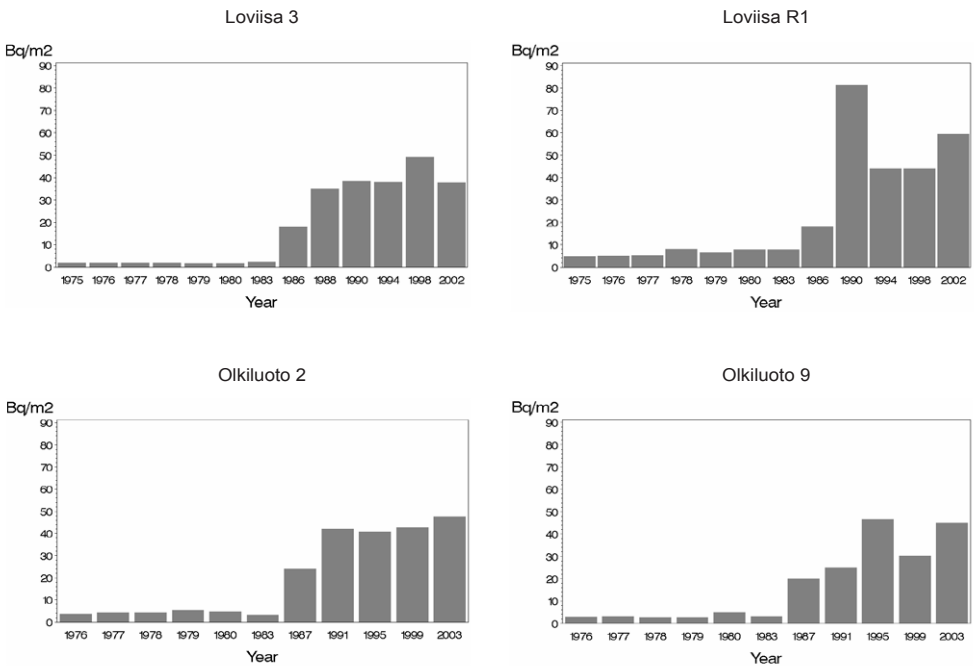


**Fig. 133.** Total amounts of $^{137}Cs$ in bottom sediments (Bq $m^{-2}$) at the Loviisa 3, Loviisa R1, Olkiluoto 2 and Olkiluoto 9 stations in 1975–2003.

from a failure in sampling. The highest cumulative total amounts of $^{137}$Cs at the sampling stations in the Loviisa sea area before and after the Chernobyl accident are given in Table 44.

Before the Chernobyl accident the maximum cumulative total amounts of $^{137}$Cs were 2 300–8 500 Bq m$^{-2}$ at the sampling stations at Loviisa. The highest values were at Stations 7 and R1, i.e., at the stations located farthest away from the power plant. After the accident, the maximum total amounts were recorded in 1990 at Stations R1 and 4: 81 300 and 61 100 Bq m$^{-2}$, respectively. At Stations 2, 3 and 5 in the discharge area of the power plant, the maximum values were 42 400, 49 200 and 32 100 Bq m$^{-2}$, respectively. The differences in the total amounts at different stations were due to differences in local deposition, sedimentation conditions and characteristics of the bottom sediments.

At the nearest open sea station LL3a, situated at the same longitude as Loviisa in the middle of the Gulf of Finland, the total amount of $^{137}$Cs in sediments was 27 000 Bq m$^{-2}$ in 1990, but the maximum, 56 900 Bq m$^{-2}$, was not reached until 1999. Thus, the settling of $^{137}$Cs was much faster in the shallow coastal waters, and the coastal effect (proximity of river mouths, higher amount of solid matter in water and higher level of eutrophication) was reflected in the higher amounts of caesium in the seabed, because caesium is transported to the bottom attached to particles.

The total amounts of $^{90}$Sr varied before the Chernobyl accident between 74 and 310 Bq m$^{-2}$ at the Loviisa 1, 2, 3, 7 and R1 stations, the highest values occurring at Stations 7 and R1. The Chernobyl fallout increased to some extent the total amounts of $^{90}$Sr in sediments. At the Loviisa 3 station, the values varied between 78 and 110 Bq m$^{-2}$ in 1975–1980, but were 210 and 390 Bq m$^{-2}$ in 1986 and 1990. Correspondingly, at Station R1, the values varied between 120 and 230 Bq m$^{-2}$ in 1975–1980, and were 210 and 320 Bq m$^{-2}$ in 1986 and 1990. Since 1986, strontium was generally analysed only from the uppermost 0–10 cm sediment layers. Thus, it was not possible to follow the trend in the total amounts per square metre.

In 1983, the activity concentrations of $^{90}$Sr varied in the surface sediments in the Loviisa area between 1.1 and 4.4 Bq kg$^{-1}$ dry wt. The Chernobyl fallout raised the values considerably, but differently to the caesium concentrations, the strontium concentrations had already returned to their former level by the beginning of the 2000s. The highest activity concentration of $^{90}$Sr was 31 Bq kg$^{-1}$ d.w. in the surface sediment at Station 3 in 1990, but as early as 1998 the concentration was 3.9 Bq kg$^{-1}$ d.w. In 2006, the $^{90}$Sr values were 1.4–3.6 Bq kg$^{-1}$ d.w. in all the 0–5 cm and 5–10 cm sediment samples taken and analysed from Stations 1, 2, 3, 4 and 5.

Contrary to these cases of caesium and strontium, the Chernobyl fallout did not affect the plutonium concentrations in the sediments in detectable amounts, but the quantities of the weapons-tests-originated $^{239,240}$Pu steadily decreased in the surface sediment layers during the whole period from 1983 to 2006. Whereas the activity concentrations had varied in the surface sediments (0–10 cm) between 1.9 and 6.7 Bq kg$^{-1}$ d.w. in 1983, they were 0.5–2.0 Bq kg$^{-1}$ d.w. in 2006. The cumulative total amounts of $^{239,240}$Pu varied between 70 and 190 Bq m$^{-2}$ and were clearly highest at Reference Station R1.

In 1986, the average $^{90}$Sr/$^{137}$Cs ratio was 0.87 ± 0.30% in the surface sediments at Loviisa, and decreased to 0.30 ± 0.06% until 2006. The average activity ratio of $^{90}$Sr/$^{137}$Cs in deposition was 3.5% in Finland after the Chernobyl accident (Saxén et al. 1987, Aaltonen et al. 1990). Thus, the lower ratio in the surface sediments indicates slower and weaker accumulation of strontium into bottom sediments. The analogous $^{239,240}$Pu/$^{137}$Cs ratio in surface sediments was on average 0.49 ± 0.25% in 1986 and 0.18 ± 0.05 in 2006.

**Olkiluoto**

Fig. 134 illustrates the vertical distribution of $^{137}$Cs in 1-cm slices at the Olkiluoto 2/12 stations in 1986–2003. It shows the corresponding time series of the depth profiles of caesium after the Chernobyl accident to that for Loviisa 3 in Fig. 132, except that the first profile is from 1987. By then the peak of Chernobyl caesium (4 000 Bq kg$^{-1}$ d.w.) was already in the second centimetre. In the four years intervals the peak was buried to ever deeper layers, as fresh sediment with lower concentrations of caesium settled on the bottom. In 2003, the highest activity concentration (2 000 Bq kg$^{-1}$ d.w.) was in the 18–19 cm sediment layer, which corresponds to a sedimentation rate of about 11 mm a$^{-1}$ (Ilus et al. 2008).

The highest cumulative total amounts of $^{137}$Cs at the sampling stations in the Olkiluoto sea area before and after the Chernobyl accident are given in Table 45. Before the accident, the maximum total amounts were 2 300–5 200 Bq m$^{-2}$. The highest values were at the Stations 2 and 9, i.e., at the stations just in front of the power plant. After the accident, the maximum total values were 47 600 Bq m$^{-2}$ at Station 2 in 2003, 46 600 Bq m$^{-2}$ at Station 9 in 1995 and 37 400 Bq m$^{-2}$ at Station 12 in 1999. The type of sediment at these three stations is softer than at any of the other stations at Olkiluoto, where the sediments in general are of more solid clay than those at Loviisa. At the Olkiluoto 3, S1, S2, S3, S4 and S7 stations with hard clay or sandy bottoms (where the samples were taken with the Ekman-Birge grab) the total amounts of $^{137}$Cs were 600–4 700 Bq m$^{-2}$. At the open sea station EB1 in the middle of the southern Bothnian Sea, the maximum total amount of $^{137}$Cs in sediments was already reached in 1992. The top value (46 000 Bq m$^{-2}$) was equal to the highest values found in the sea area off Olkiluoto.

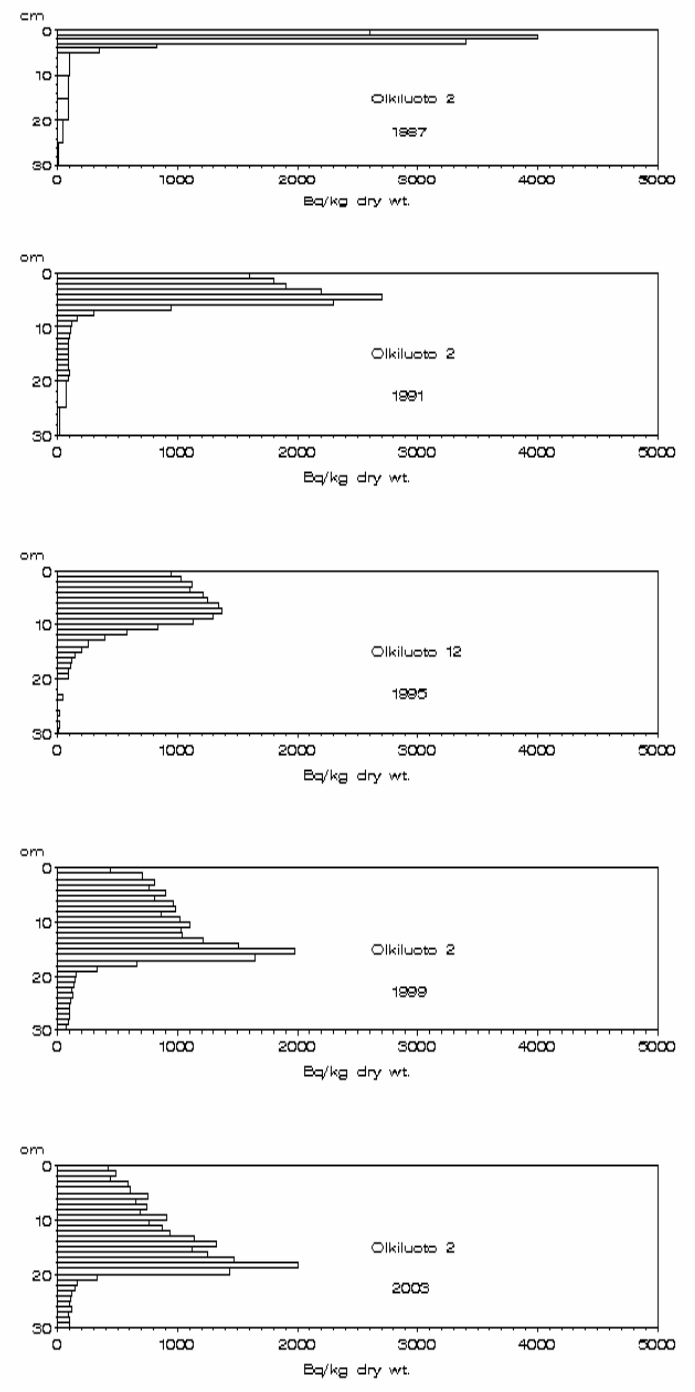


**Fig. 134.** Vertical distribution of $^{137}Cs$ concentrations at the Olkiluoto 2/12 stations in 1987–2003 (Ilus et al. 2008).

**Table 45.** The highest cumulative total amounts of $^{137}Cs$ at the sampling stations in the Olkiluoto sea area before and after the Chernobyl accident (Bq $m^{-2}$).

| Station | Max. total amount of $^{137}Cs$ before Chernobyl (year), Bq $m^{-2}$ | Max. total amount of $^{137}Cs$ after Chernobyl (year), Bq $m^{-2}$ |
|---|---|---|
| Olkiluoto 1 | 2 300 (1978) | 19 800 (1995) |
| Olkiluoto 2 | 5 200 (1979) | 47 600 (2003) |
| Olkiluoto 3 | | 3 500 (1991) |
| Olkiluoto 4 | 4 300 (1980) | 32 000 (2003) |
| Olkiluoto 5 | | 10 700 (2003) |
| Olkiluoto 9 | 4 800 (1980) | 46 600 (1995) |
| Olkiluoto 12 | 4 000 (1979) | 37 400 (1999) |
| Olkiluoto 15 | | 24 100 (1995) |
| Olkiluoto S1 | | 1 500 (1991) |
| Olkiluoto S2 | | 600 (1991) |
| Olkiluoto S3 | | 1 700 (1991) |
| Olkiluoto S4 | | 800 (1991) |
| Olkiluoto S5 | | 24 700 (1999) |
| Olkiluoto S6 | | 21 400 (2003) |
| Olkiluoto S7 | | 4 700 (1995) |
| Olkiluoto S8 | | 27 000 (2003) |

The total amounts of $^{90}Sr$ varied between 74 and 260 Bq $m^{-2}$ at the Olkiluoto 1, 2, 4, 9 and 12 stations before the Chernobyl accident. The highest values occurred at Stations 2 and 12; 260 and 240 Bq $m^{-2}$, respectively. After the accident (in 1987), the highest values, at Stations 4 and 2, were 520 and 490 Bq $m^{-2}$, respectively. Correspondingly, the activity concentrations of $^{90}Sr$ varied in the surface sediments in the Olkiluoto area between 1.8 and 2.5 Bq $kg^{-1}$ dry wt. in 1983. As a consequence of the Chernobyl fallout, the $^{90}Sr$ concentrations rose to 49 and 38 Bq $kg^{-1}$ d.w. in the surface sediments at Stations 2 and 4 in 1987, but in 2003 the values had already decreased to 1.8–5.6 Bq $kg^{-1}$ d.w. in all the 0–5 cm and 5–10 cm sediment samples taken and analysed from Stations 2, 9, S5, and S8. The average $^{90}Sr/^{137}Cs$ ratio was 1.6 ± 0.6% in the surface sediments at Olkiluoto in 1987, and decreased to 0.55 ± 0.14% until 2003.

The Chernobyl fallout did not affect the plutonium concentrations in the sediments in the Olkiluoto area, either: there too the quantities of the weapons-tests-originated $^{239,240}Pu$ decreased steadily in the surface sediment layers during 1983–2003. Whereas the activity concentrations varied in surface sediments (0–10 cm) between 2.2 and 5.7 Bq $kg^{-1}$ d.w. in 1983, they were 1.2–2.3 Bq $kg^{-1}$

d.w. in 2003. The cumulative total amounts of $^{239,240}$Pu varied between 77 and 340 Bq m$^{-2}$ and were generally highest at the Olkiluoto 2 station. At Olkiluoto, the $^{239,240}$Pu/$^{137}$Cs ratio in surface sediments was on average 0.41 ± 0.35% in 1987 and 0.29 ± 0.09% in 2003.

### 3.9.3 Other artificial radionuclides in bottom sediments

**Loviisa**

In addition to natural $^{40}$K, Chernobyl-derived $^{137}$Cs and $^{134}$Cs, and $^{90}$Sr, $^{238}$Pu and $^{239,240}$Pu that mainly originated from the weapons-tests-fallout, altogether 15 radionuclides were detected in ordinary sediment samples taken from Loviisa in 1975–2006. Many of them were very short-lived, and occurred principally only in the samples taken in 1986 soon after the Chernobyl accident (e.g. $^{140}$La, $^{140}$Ba, $^{141}$Ce, $^{129m}$Te, $^{95}$Nb, $^{103}$Ru, $^{89}$Sr and $^{95}$Zr).

The activity concentrations of the naturally-occurring $^{40}$K ranged from 590 to 1 100 Bq kg$^{-1}$ dry wt. Of the other nuclides, the areal distribution of $^{125}$Sb was widest, and its concentrations in sediments were highest in the next few years after the Chernobyl accident (1986–1990). Then $^{125}$Sb was also recorded at Reference Station R1, although the maximum value (160 Bq kg$^{-1}$ d.w.) was detected at Station 3 in 1988. In 1998, $^{125}$Sb was still analysed in low concentrations at a depth of 10–15 cm at Stations 2, 3 and 7. $^{144}$Ce was already widely found in sediments in 1983, but the highest concentrations were analysed in 1986, when the maximum value was 280 Bq kg$^{-1}$ d.w. at Station 5. The last observation of $^{144}$Ce (34 Bq kg$^{-1}$ d.w.) was recorded at Station 3 in 1988.

$^{60}$Co was detected in sediment samples at Stations 5 and 3 for the first time in 1983 (7.5 and 6.8 Bq kg$^{-1}$ d.w.). The highest values occurred in the uppermost sediment layers (0–5 cm) at these stations in 1986 and 1988 (69 and 56 Bq kg$^{-1}$ d.w., respectively). After that, the $^{60}$Co contents in the surface layer consistently decreased; since 1998 somewhat higher concentrations occurred in the deeper layers (Fig. 135). In 1998, $^{60}$Co was found down to a depth of 15–20 cm at Station 3. This was in good agreement with the operational history of the power plant and the sedimentation rate determined with the $^{137}$Cs method for this station (see p. 284). Supposing that the sedimentation rate is 8 mm a$^{-1}$, the sediment layer deposited in 1977 was at a depth of 17 cm in 1998. In all, $^{60}$Co originating from local discharges was observed in the sediment samples at all the other stations (Stations 1, 2, 3, 4, 5, 7, 8, and 10) except at Stations R1, R2, S1, S2, S5 and S6. Thus, the farthermost observation was at a distance of about 5 km from the power plant.

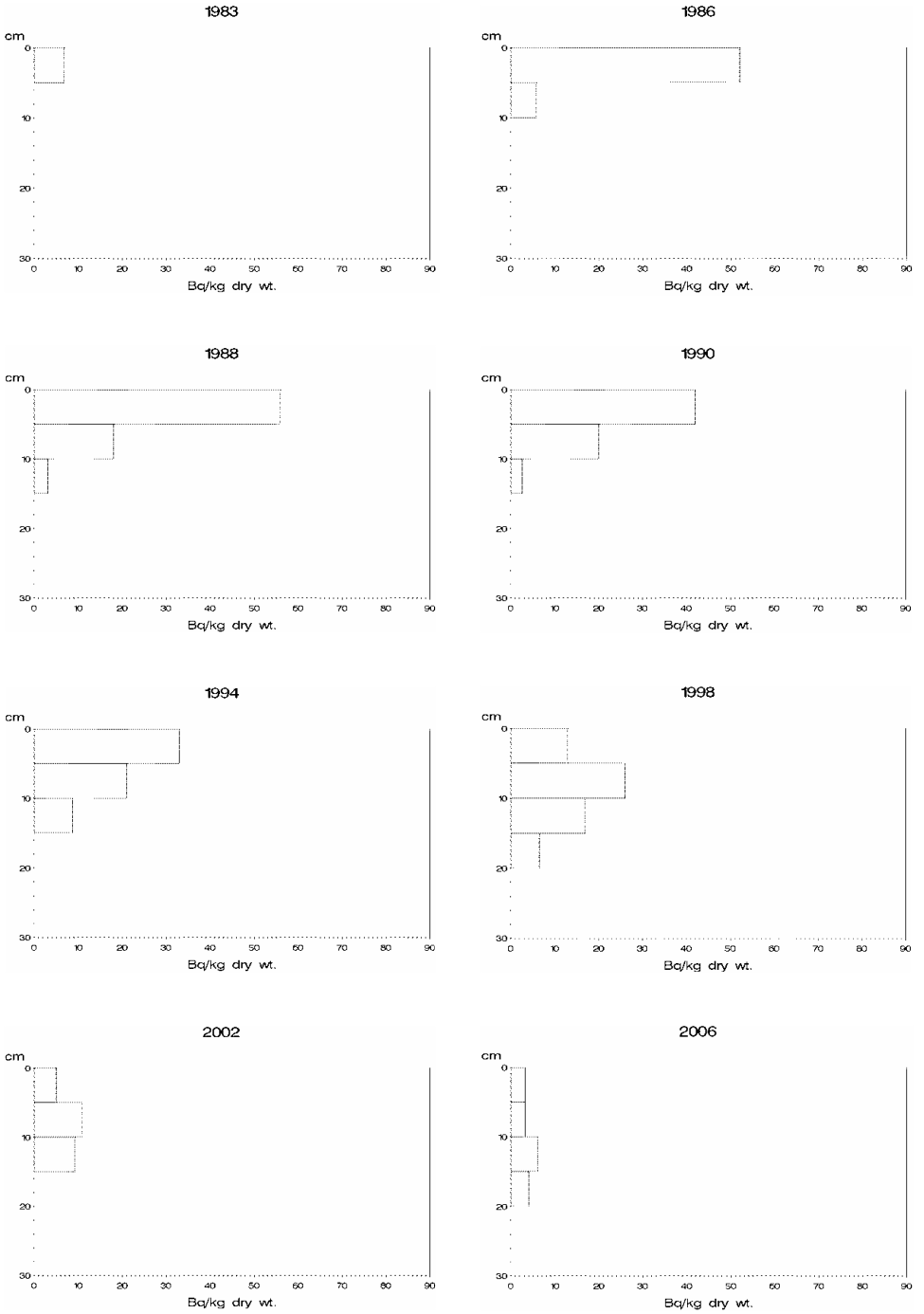


**Fig. 135.** Vertical distribution of $^{60}$Co concentrations at the Loviisa 3 station in 1983–2006.

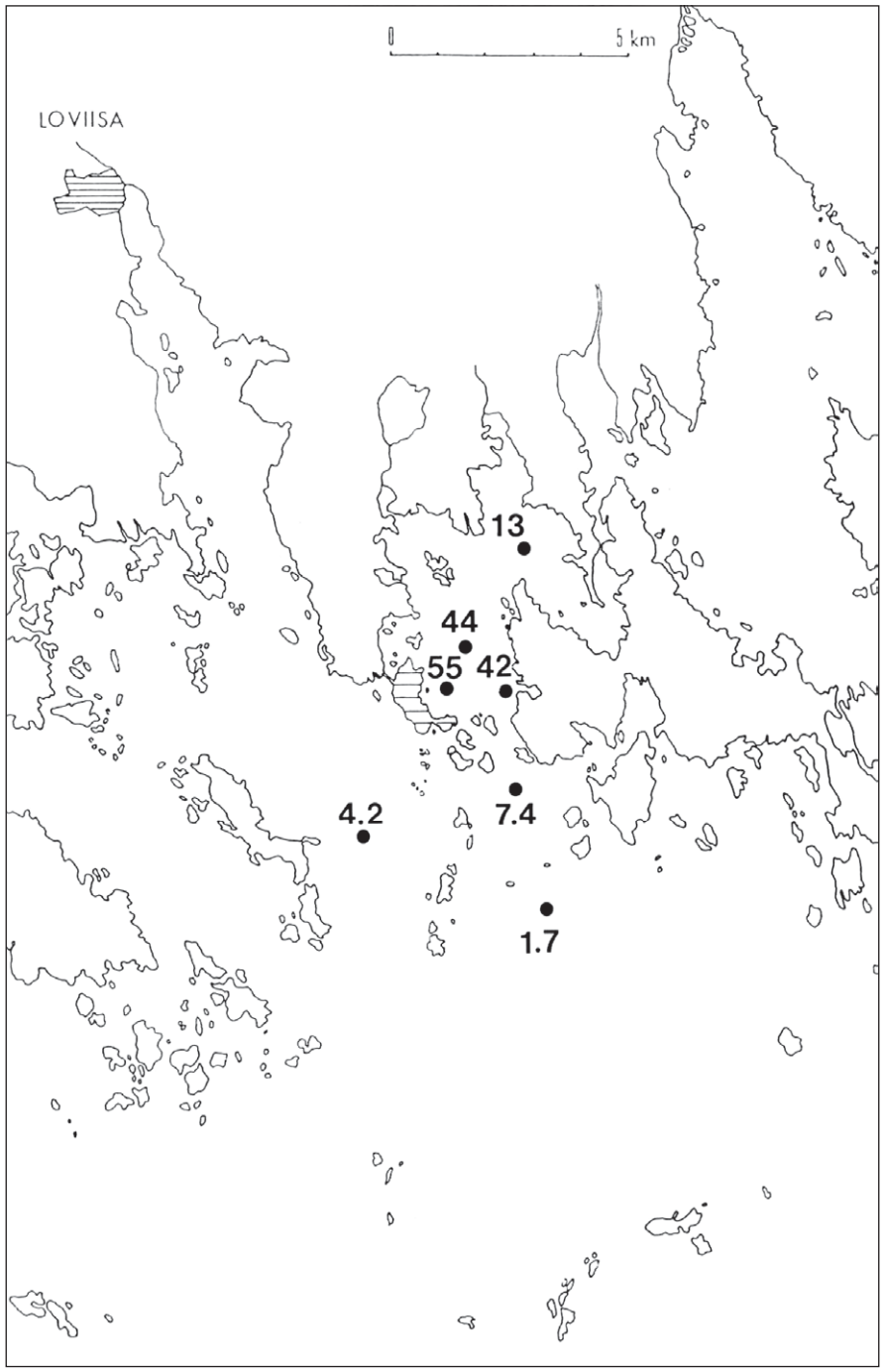


**Fig. 136.** Distribution of $^{60}$Co in the 1990 sediment survey at Loviisa. Maximum activity concentrations of $^{60}$Co (Bq kg$^{-1}$ d.w.) in the 5-cm-slices of the sediment profiles (in this case always in the surface sediment layer; see the text).

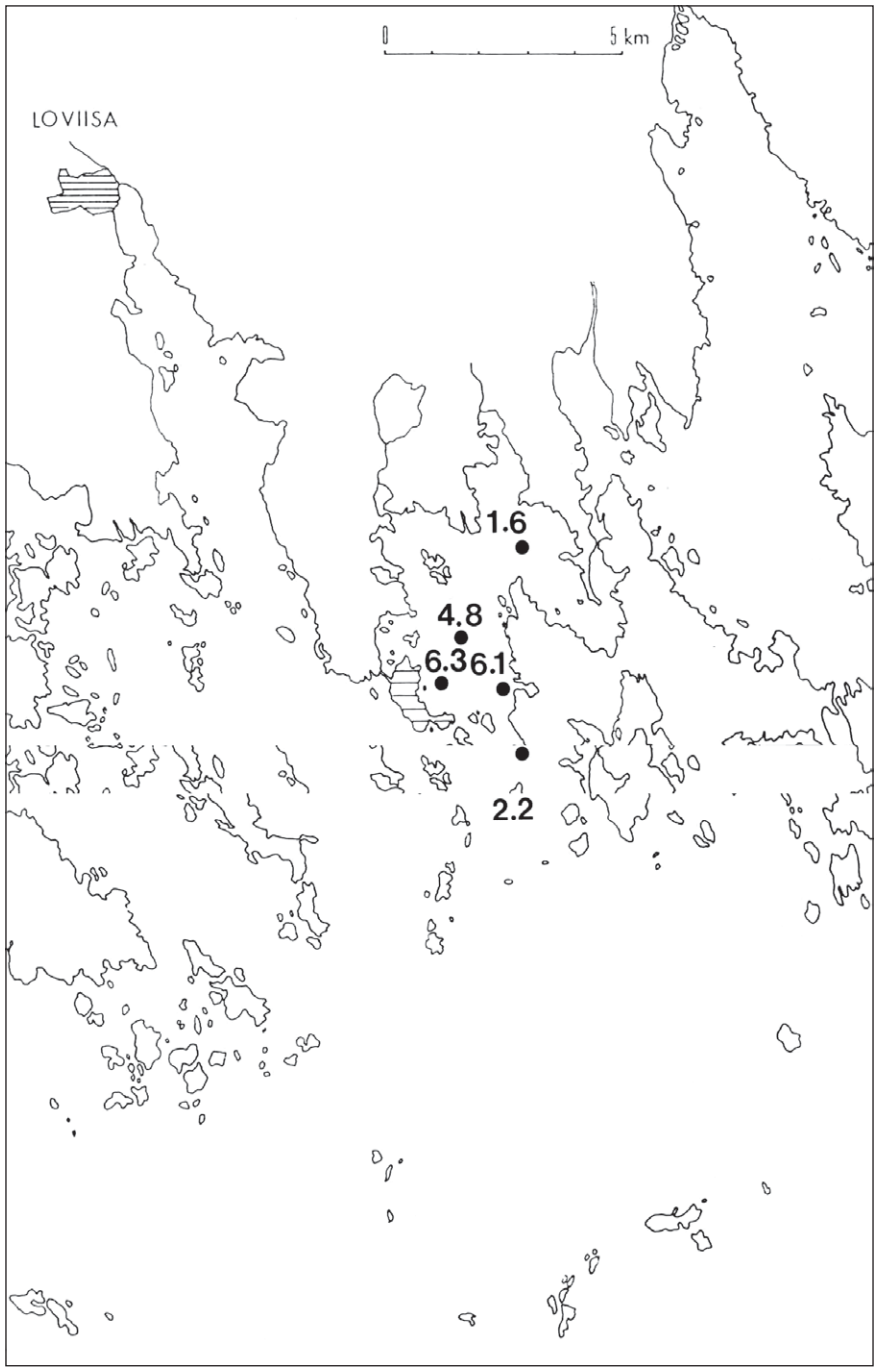


**Fig 137.** Distribution of $^{60}$Co in the 2006 sediment survey at Loviisa. Maximum activity concentrations of $^{60}$Co (Bq kg$^{-1}$ d.w.) in the 5-cm-slices of the sediment profiles (in this case at different depths according to the sedimentation rate; see the text).

Fig. 136 illustrates the maximum concentrations of $^{60}$Co at each sampling station in the 1990 sediment survey, excluding Reference Station R1, where cobalt was not found. The maximum concentrations occurred at each station in the uppermost 0–5-cm sediment layer. Fig. 137 illustrates the maximum concentrations in the 2006 sediment survey. Then the maximum at Station 5 (6.3 Bq kg$^{-1}$ d.w.) was in the slice of 15–20 cm. At Station 1 the maximum was in the surface layer, but at all the other stations in the 10–15-cm layer. Samples were also taken at Stations 4a, 7 and R1, but cobalt was not found in these samples.

The occurrence of $^{106}$Ru in sediments was clearly connected to the Chernobyl fallout. In 1986, the maximum concentrations (150–750 Bq kg$^{-1}$ d.w.) were at all the stations in the surface sediment layer. In 1990, the concentrations ranged from 27 to 91 Bq kg$^{-1}$ d.w., but after that $^{106}$Ru was not observed in sediment samples. Small amounts of $^{110m}$Ag and $^{54}$Mn were already found in the sediments in 1983, but the maximum values were analysed in 1986 and 1988 (50 Bq kg$^{-1}$ d.w. for $^{110m}$Ag at Loviisa 3 in 1986, and 9 Bq kg$^{-1}$ d.w. for $^{54}$Mn at Loviisa 3 in 1988). In 1986, $^{110m}$Ag was also detected at Reference Station R1 (11 Bq kg$^{-1}$ d.w. in the surface layer). $^{54}$Mn was not detected in sediment samples after 1988, and $^{110m}$Ag not after 1994.

**Olkiluoto**

In addition to $^{40}$K, $^{90}$Sr, $^{134}$Cs, $^{137}$Cs, $^{238}$Pu and $^{239,240}$Pu, nine other radionuclides were detected in ordinary sediment samples taken from Olkiluoto in 1976–2003. The activity concentrations of the naturally-occurring $^{40}$K ranged from 450 to 1 000 Bq kg$^{-1}$ dry wt. $^{95}$Zr, $^{95}$Nb, $^{106}$Ru, $^{110m}$Ag and $^{144}$Ce were almost exclusively found in 1987, soon after the Chernobyl accident, and, in smaller quantities, still in 1991. The highest activity concentrations of $^{106}$Ru, $^{110m}$Ag and $^{144}$Ce were 720, 51 and 270 Bq kg$^{-1}$ d.w., respectively, in the surface sediment layer at Station 2 in 1987. The concentrations of $^{125}$Sb were also highest in 1987 (maximum 100 Bq kg$^{-1}$ d.w. at Station 2), and small concentrations of it were still detected in 1995 at Stations 15 and S5.

$^{60}$Co originating from local liquid discharges was already detected in 1983 at all the stations monitored, i.e., at Stations 1, 2, 4, 9 and 12 (Figs. 138 and 139). At Station 9, in front of the cooling water outlet, the activity concentration of $^{60}$Co was 50 Bq kg$^{-1}$ d.w. in the uppermost 0–5-cm layer and 5.7 Bq kg$^{-1}$ d.w. in the 5–10-cm layer. At Stations 2, 12, 1 and 4, the concentrations in the surface layer were 23, 15, 11 and 7.7 Bq kg$^{-1}$ d.w., respectively. A clear rise occurred in the concentrations in connection with the Chernobyl accident, but the maximum values recorded in any of the sediment slices were detected in 1991–2003. The highest single activity concentration of $^{60}$Co (83 Bq kg$^{-1}$ d.w.) was recorded at a

depth of 7–8 cm in the 1-cm slices taken from Station 2 in 2003 (Ilus et al. 2008). At Station 9, the maximum value was 79 Bq $kg^{-1}$ d.w. in the 0–5-cm surface layer in 2003.

Figs. 140 and 141 illustrate the maximum concentrations of $^{60}Co$ at each sampling station in the 1995 and 2003 sediment surveys. In 1995, the maximum concentrations occurred at each station in the uppermost 0–5 cm sediment layer. In 2003, the maximum at Station 2 was at a depth of 7–8 cm. At Stations 4, S5, S6 and S8 the maximum was in the 5–10 cm sediment layer, and cobalt was no longer detected in the surface layer at Stations S5, S6 and S8. At the Stations 1, 5 and 9, the maximum was still in the surface layer. The farthest observations of local discharge nuclides in sediments, i.e., small amounts of $^{60}Co$, were recorded at Kuuskajaskari (2.6 Bq $kg^{-1}$ d.w.), 16 km to the south of Olkiluoto, and at Pirskerinfäärtti (4.9 Bq $kg^{-1}$ d.w.), 17 km to the north of Olkiluoto measured by sea (Figs. 140 and 141).

Another radionuclide in the sediment samples, which mainly originated from the liquid discharges of the local power plant, was $^{54}Mn$. It was detected in the sediment samples from Stations 1, 2, 3, 4, 9 and 12 in 1987, 1991 and 1995. The highest concentration was 16 Bq $kg^{-1}$ d.w. in the surface sediment layer at Station 9 in 1995. In 1995, a small amount of $^{54}Mn$ (1.5 Bq $kg^{-1}$ d.w.) was also found at Station S6.

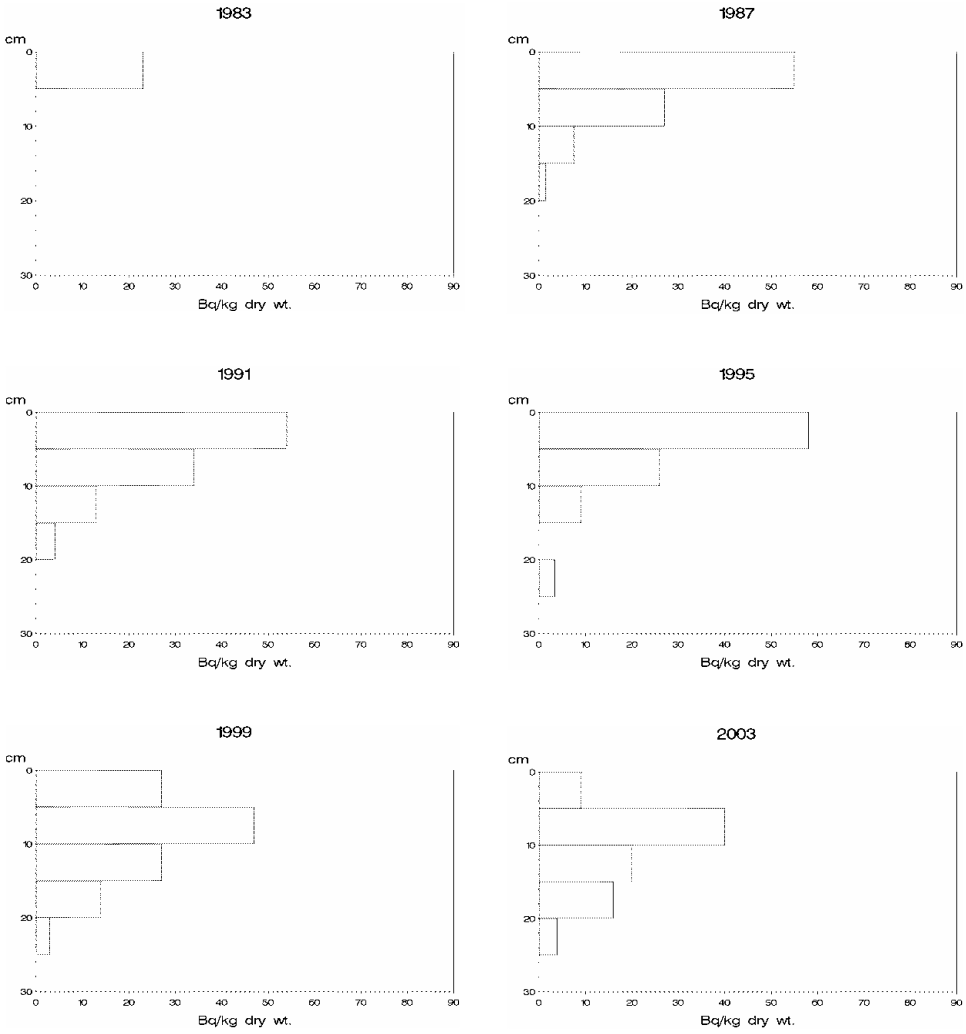


**Fig. 138.** Vertical distribution of $^{60}Co$ concentrations at the Olkiluoto 2 station in 1983–2003.

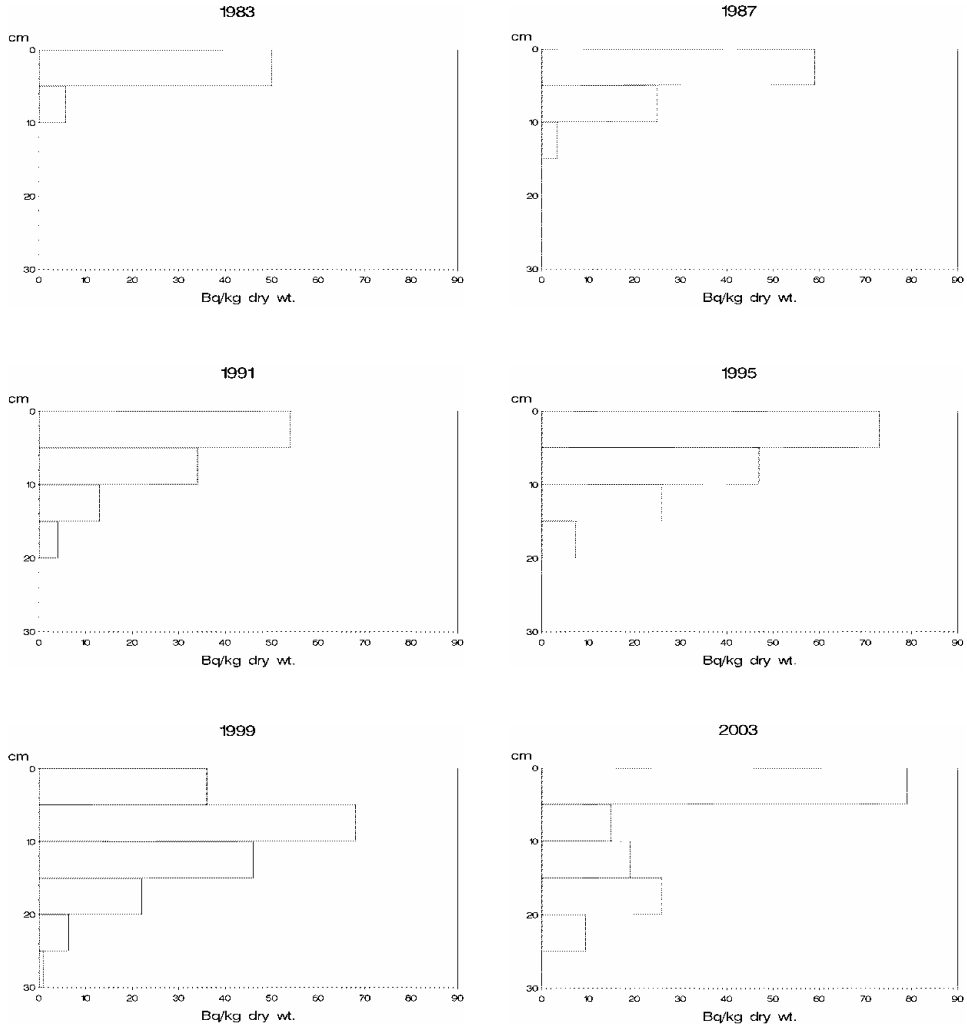


**Fig. 139.** Vertical distribution of $^{60}Co$ concentrations at the Olkiluoto 9 station in 1983–2003.

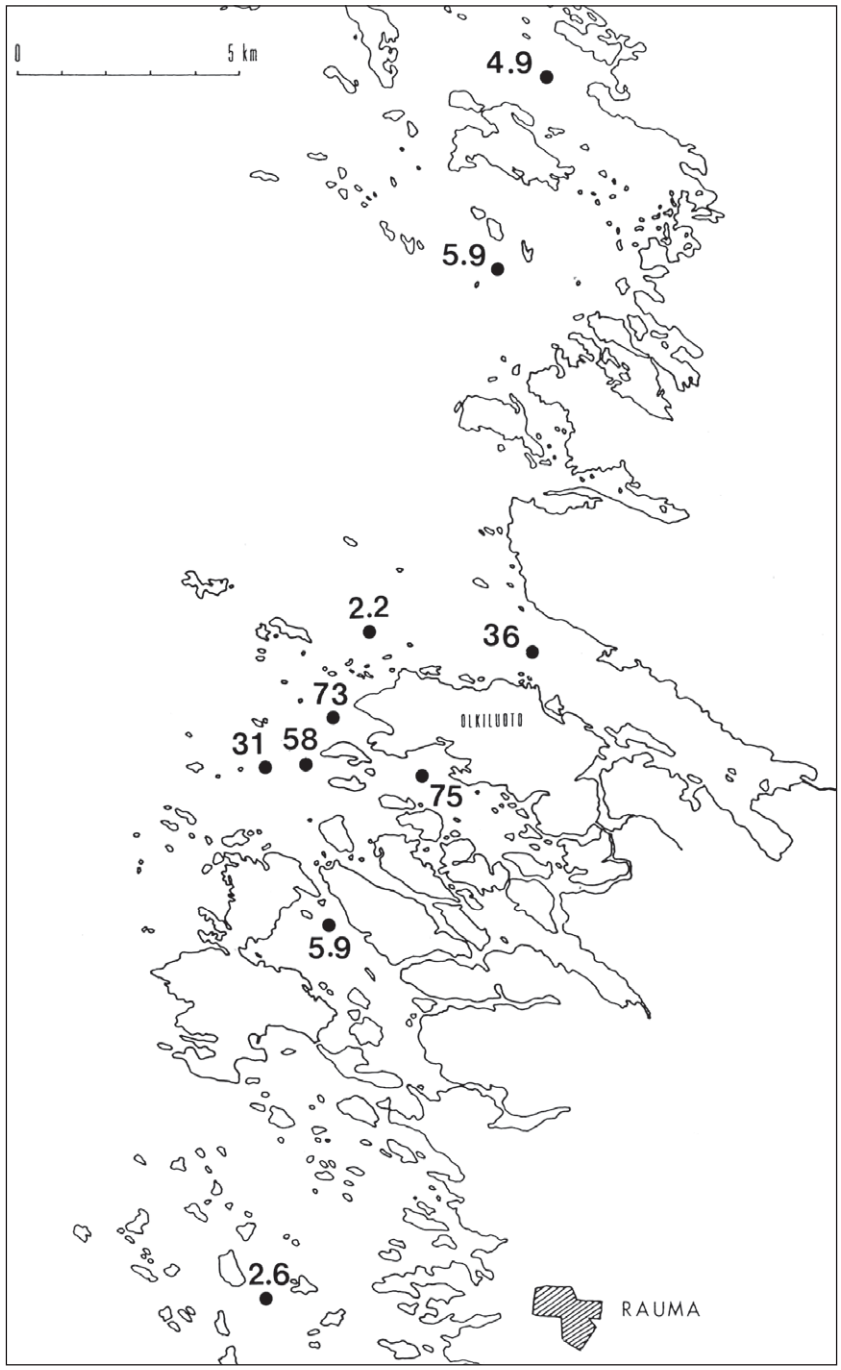


**Fig. 140.** Distribution of $^{60}Co$ in the 1995 sediment survey at Olkiluoto. Maximum activity concentrations of $^{60}Co$ (Bq $kg^{-1}$ d.w.) in the 5-cm slices of the sediment profiles (in this case always in the surface sediment layer; see the text).

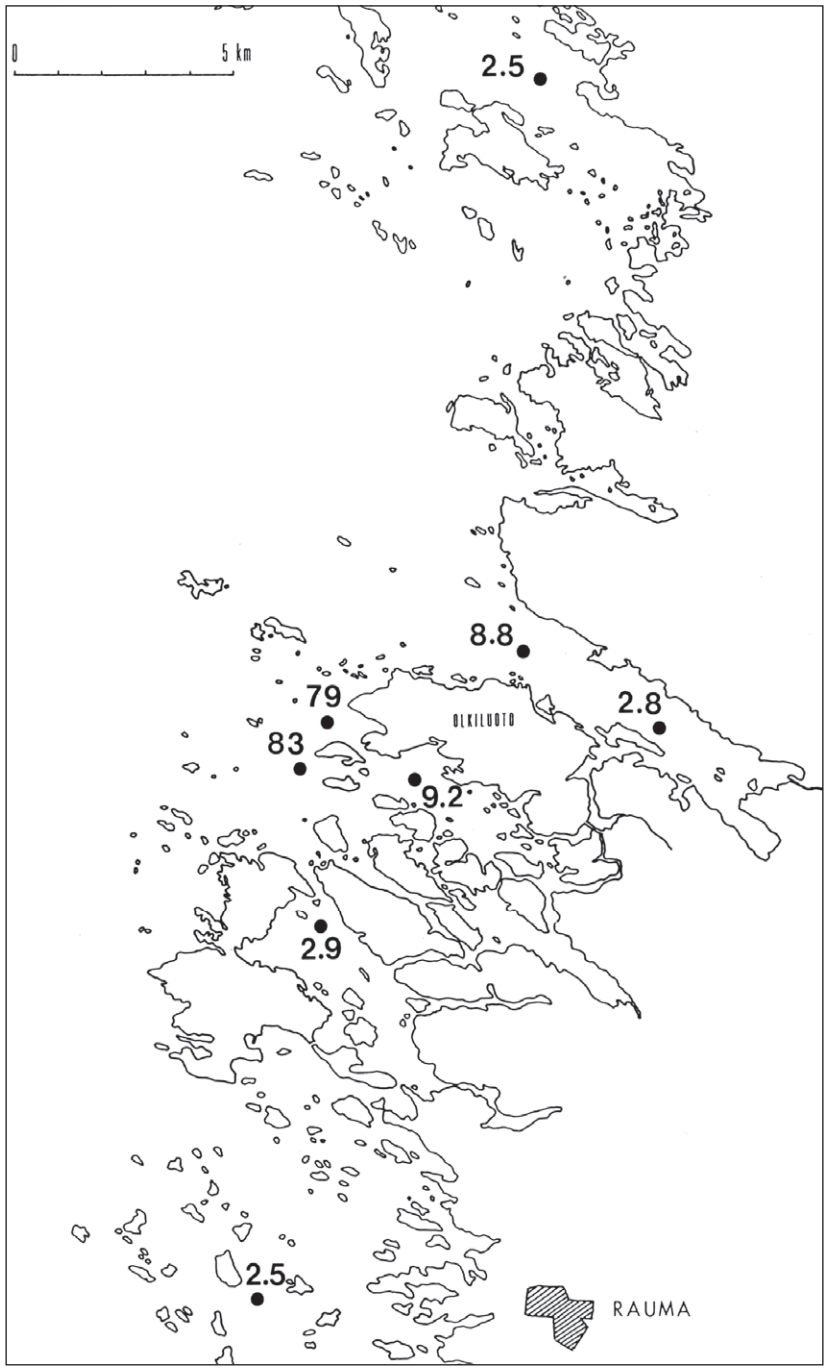


**Fig. 141.** Distribution of $^{60}$Co in the 2003 sediment survey at Olkiluoto. Maximum activity concentrations of $^{60}$Co (Bq kg$^{-1}$ d.w.) in the 5-cm slices of the sediment profiles (in this case at different depths according to the sedimentation rate; see the text).

# Part III

# Summary, discussion and conclusions

# 4 Thermal effects

## 4.1 Loviisa

The amount of heat discharged from the Loviisa NPP into the sea increased in the course of the study period from about 20 PJ $a^{-1}$ in the late 1970s to about 56 PJ $a^{-1}$ in 1997–2006. The temperature of the discharged water was about 8–12ºC higher than that of the intake water. In 1997–2006, the average flow rate of the cooling water was about 44 $m^3$ $s^{-1}$ and the annual maximum temperature of discharged water ranged from 28.4 to 32.1ºC.

The most obvious physical effect of the cooling water discharged from the power plant has been the increase in temperature of the seawater in the discharge area. The effects of the cooling water on the temperatures in the sea surrounding Hästholmen Island have been most eye-catching in winter, when the conditions have also most fundamentally differed from those in the natural state, as the thermal discharges have significantly affected the ice conditions in the vicinity of the power plant. The formation of a permanent ice cover in the discharge area has been delayed in early winter, while the break-up of the ice in spring has been advanced, so that the growing season has been prolonged at both ends. Areas close to the cooling water outlet are usually open throughout the winter; the size of the open area may change from day to day (ranging from several hectares to several square kilometres) depending on the weather conditions. In the 1990s and 2000s, the winters were in general mild and the extension of the ice cover in the sea area around Hästholmen was very variable, but overall, a major part of Hästholmsfjärden could be open throughout the season. In mild or very mild winters, the extent of the ice-free area has often been 3–8 $km^2$. On the other hand, if the sea is ice-covered, the warm cooling water may spread over wide areas under the ice as a discrete layer settled to a depth of 2–3 m. In such situations, the warm water has affected the ice and temperature conditions at distances of >10 km west of Hästholmsfjärden. When the heated water comes into contact with the ice, it may cause unexpected and treacherous conditions for people moving on the ice. From a human point of view, difficult ice conditions can thus hinder winter fishing over quite extensive areas.

From the biological point of view, the prolonging of the growing season and the disturbance of the overwintering time, in conditions in which the biota has adjusted to a distinct rest period in winter, are undoubtedly the most significant effects of the thermal pollution. Aquatic organisms in the northern Baltic Sea are acclimatized to a clearly recognisable annual winter season. A shortening of the ice winter or the total lack of an ice cover leads to indistinct limits between the growing period and winter season. Most plants need a resting period, and the changed ice conditions disturb the natural annual rhythm of growth.

At Olkiluoto, for instance, the increased illumination in the ice-free area and the direct influence of the temperature rise caused a significant advance in the start of the growing season of many species and an excessive occurrence of diatoms (Keskitalo & Heitto 1987). In Hästholmsfjärden at Loviisa, water temperatures exceeding 11ºC in March and the lack of an ice cover throughout the winter shortens the resting period, and leads to the start of the active period in spring as soon as the light intensity becomes adequate. Thus, the length of the growing season is primarily regulated by the limits of sufficient light intensity in late winter and late autumn.

Snoeijs (1988) stated that, in the discharge area of the Forsmark NPP in Sweden, the effects of thermal discharge were greatest in winter and early spring due to the extension of the active season for organisms. Some species persisted through winter in a 'pauperate' condition at sites affected by cooling water discharge, and their persistence favoured them in spring to start earlier and to become more abundant than under natural conditions. This behaviour was shown, e.g., by the filamentous green alga *Cladophora glomerata* and macrofauna (e.g., chironomid larvae) that seek food and shelter in the *Cladophora*-belt. Furthermore, Snoeijs (1990) stated that an increased water temperature resulted in a higher biomass, through a chain of ecological effects. The strong reduction in the ice cover was crucial in this chain, as it resulted in a reduced loss of diatoms by reduced abrading and higher primary production through the higher availability of light, nutrients and substrate.

In open water conditions, winds have an essential role in the spreading and maintenance of the distributive area of the warm water plume. The heated water flows with the wind and may be 'packed' into the northern or eastern part of discharge area, against the east shore of Hästholmen, or it may be forced to flow out from Hästholmsfjärden through the sounds in the south. Due to the effective transfer of the heat into the atmosphere and the more effective mixing of the water masses in open water conditions, the thermal effects are less visible and do not reach as far as in ice winters. The highest temperatures of surface water recorded in summer were 30ºC at a distance of 50 m from the outlet, 28.5ºC at a distance of 400 m, 27–28ºC at the other stations in Hästholmsfjärden, 26.5ºC in Klobbfjärden, about 25ºC at Stations 4 and 8 in Vådholmsfjärden and Hudöfjärden, and at the reference station in Påsalöfjärden, and about 23ºC at Station 7 in Orrengrundsfjärden.

The mean surface water temperatures of the growing season (May–October) show annual fluctuations caused by the varying weather conditions in different years. Before the commissioning of the power plant, the difference in mean surface water temperature between Station 2 of Hästholmsfjärden and Station 8 of Hudöfjärden during the growing season was 1ºC. The cooling water has

raised the mean surface water temperatures of the growing season by 4–5ºC at a distance of 0.4 km from the outlet and by 2.5–3ºC at a distance of 1–2 km in Hästholmsfjärden. The rise of temperature was also statistically significant at Station 4 in Vådholmsfjärden (at a distance of 2.8 km) and at Station 1 in Klobbfjärden (at a distance of 3.4 km), but not at Station 7 in Orrengrundsfjärden. The average increase was 1.3ºC at Station 4 and 2.4ºC at Station 1, compared to that at Station 8 situated in the intake area. This means that the thermal effect was clearly detectable during the growing seasons in Hästholmsfjärden and Klobbfjärden, but declined rapidly on leaving Hästholmsfjärden.

It is known that a temperature rise generally increases the metabolic rate, and the growth rate of aquatic organisms. This means an increased production of organic matter, and therefore, there are reasons to suspect that a temperature rise can have severe consequences, above all in nutrient-rich waters (Grimås 1974). Thus, thermal pollution probably promotes the eutrophication process in eutrophied environments. The heat also increases the rate of decomposition of organic matter in the receiving water bodies, and this can lead to depletion of oxygen in deeper water layers (Grimås 1975).

Nitrogen and phosphorus are the main nutrients at the bottom of all food chains. Primary producers fix the nutrients into their biomass through primary production. When excessive amounts of nutrients enter the sea, primary production increases rapidly, and the natural ecological balance of the marine ecosystem may be disturbed (HELCOM 2003). The nutrient concentrations of water in the study area are mainly affected by the general nutrient levels in the Gulf of Finland, and more locally, by nutrient discharges from neighbouring rivers, local point sources, diffuse pollution and the internal nutrient load. Nutrient-rich waters from neighbouring rivers are especially transported to the area in spring, while the remobilization of nutrients from bottom sediments is focused on the late summer and autumn.

During the last 40 years, the nutrient concentrations have strongly increased in the whole study area. On average, the total phosphorus concentrations of surface water doubled and the total nitrogen concentrations increased 1½-fold between the first half of the 1970s and 2000. During the growing seasons 1995–2000, mean phosphorus concentrations in surface water were 32–36 µgP $l^{-1}$ and those of nitrogen 500–590 µgN $l^{-1}$. Between the early 1970s and 2000s, the change in the mean phosphorus concentrations was statistically very significant at Stations 2 and 8, and the change in the nitrogen concentrations similarly at Stations 2, 4, 5, 8 and 10. However, the increase in nutrient concentrations occurred evenly over the whole area; the differences between the stations were statistically insignificant. In 2001–2006, the total phosphorus and total nitrogen concentrations actually decreased, being about

1/5 lower than those at the end of the 1990s; this was due to a decrease in the external nutrient load in the Gulf of Finland (Kauppila and Bäck 2001).

In a longer perspective, the increase in nutrient concentrations has been a general trend in the whole Baltic Sea, and especially in the Gulf of Finland. Phosphorus concentrations in surface water have significantly increased over the whole Baltic Sea since the beginning of the 1960s (Perttilä et al. 1980, Perttilä & Voipio 1981), and a doubling of nutrient concentrations was observed in the open Gulf of Finland during the 1970s and 1980s (Kauppila and Bäck 2001). In the mid-1990s, the phosphorus concentrations in the Gulf of Finland began to increase due to the strengthening of the internal nutrient load caused by remobilization of phosphorus from the sediments in the eastern Gulf of Finland (Pitkänen et al. 2003). In general, the progress of eutrophication in Finland's coastal waters continued during the 1990s and early 2000s, despite the simultaneous decrease in external nutrient load from direct and river point sources and via the atmosphere (Kauppila et al. 2004). The Loviisa power plant was apparently responsible for only a minor proportion of the total input of nutrients into the study area. Actually, regarding the total phosphorus load to Hästholmsfjärden, the input of the internal load has been more significant as a local source than the load from land-based sources (power plant + Loviisan Smoltti). However, the nutrient concentrations in the study area are unambiguously determined by the general level of nutrients in the eastern Gulf of Finland.

Nutrients have a focal role in the regulation of biological production. Primary producers fix the nutrients into their biomass through primary production. Other factors affecting the amount of annual phytoplankton primary production are the temperature of the water and the length of the growing season. The unstable ice conditions have advanced the start of the production period in spring and caused non-coincidence of the vernal maximum peaks at different sampling stations. In addition, the fact that the production may already start quite intensively under the ice, has made the attainment of the maximum difficult. The highest vernal values measured in the area were 2 170 mg C $m^{-2}$ $d^{-1}$ in Hudöfjärden (1983) and 1 830 mg C $m^{-2}$ $d^{-1}$ in Hästholmsfjärden (1984). The maximum was generally lower in Hästholmsfjärden due to the higher turbidity of the water, caused by the higher impact of river waters there in spring, and the cooling water discharge seemed not to affect the magnitude of the maximum there. On the other hand, the correlation between the magnitude of the vernal maximum and the concentration of phosphorus in the water, together with the decrease in phosphorus concentrations may explain the decrease in the vernal production values in recent years.

In fact, the emphasis of primary production tended to move from the spring to the late summer during the study period. While the maximum values in the

1970s and 1980s were rather regularly measured during the vernal bloom, in the 1990s they were often measured in June–July and in the 2000s in July–August. Such was especially the case in Hästholmsfjärden, where the thermal discharges affected the primary production most significantly in autumn, when the water temperature had already started to decrease elsewhere, but was still high in the discharge area. The highest summer production values measured at the end of July 1997 were 1 510 mg C $m^{-2}$ $d^{-1}$ in Hästholmsfjärden and 1 280 mg C $m^{-2}$ $d^{-1}$ in Hudöfjärden, being 2.3 and 1.6 times higher, respectively, than the highest values in the 1970s. The strengthening of summer production is characteristic of eutrophied waters (Bagge & Lehmusluoto 1971, Niemi & Pesonen 1974a and b). Thus, the increased summer values provide evidence of the progressive eutrophication that has taken place in the study area during the last 40 years. However, in spite of small differences, the eutrophication in the intake and discharge areas of the cooling water seems to have proceeded quite in parallel.

Annual primary production more than doubled in the area between the late 1960s and late 1990s, but turned downwards a little at the beginning of this century. In 1998, the highest annual production values were 158 g C $m^{-2}$ $a^{-1}$ in Hästholmsfjärden and 149 C $m^{-2}$ $a^{-1}$ in Hudöfjärden. The rise was fairly parallel in both areas. The main reason for the increased level of annual production was obviously the general eutrophication in the whole study area and in the whole Gulf of Finland. However, the thermal discharges have changed the relationship between the magnitude of annual production in Hästholmsfjärden and Hudöfjärden. In 1967–1976, the annual production was quite regularly higher in Hudöfjärden, but since then the relationship has been reversed. This change was statistically very significant.

In 1976–1982, *in situ* primary production was also measured at a station situated at a distance of 400 m off the cooling water outlet. The annual production there was on average 8% higher than at the permanent sampling station in mid-Hästholmsfjärden. The difference was probably due to a stronger effect of the temperature rise in the close vicinity of the discharge point.

The regression analysis for the whole study period showed that, in the first step, the temperature of the water best explained the changes in primary production in Hästholmsfjärden, and thereafter the transparency of the water and total phosphorus, whereas total phosphorus was the best independent variable in Hudöfjärden, and thereafter water temperature and total nitrogen. Water temperature and total phosphorus as a pair best explained the changes in primary production in Hästholmsfjärden, and correspondingly total phosphorus and water temperature in Hudöfjärden. When three independent variables were used, water temperature + total phosphorus + N/P ratio best explained the changes in Hästholmsfjärden, and total phosphorus + water temperature

+ N/P ratio in Hudöfjärden. The N/P ratio was generally above 17 both in Hästholmsfjärden and Hudöfjärden during the growing seasons, which indicates that phosphorus is the limiting factor in the area.

The primary production capacity correlated closely with the *in situ* production. Thus, the seasonal succession of the PP capacity followed that of the *in situ* values, as well as the moving of the highest productivity to the late summer and autumn in the 1990s. The inter-annual changes in PP capacity also occurred fairly much in parallel at all the sampling stations, and showed a similar progress to those of the *in situ* annual production. Consequently, a similar change as that demonstrated to have occurred in the relationship between the annual production of Hästholmsfjärden and Hudöfjärden since 1976, was also valid for the PP capacity values of these areas.

In 1996–2000, the average primary production capacity in Hästholmsfjärden and Hudöfjärden was at least two times higher than in the early 1970s, but dropped strongly in 2005–2006. The highest daily PP capacity values were recorded at the traditional sampling stations of Hästholmsfjärden and Hudöfjärden, but in general the mean values of the growing seasons decreased when moving from the inner areas towards the outer archipelago, so that on average the highest values occurred in Klobbfjärden and Hästholmsfjärden and the lowest in Orrengrundsfjärden and Kejvsalö östra fjärd. The occurrence of the highest average PP capacity in Klobbfjärden was probably due to the fact that the nutrient load from the Tesjoki River is strongest in this area, but the high turbidity of the water in spring arrests the formation of high daily PP capacity values during the vernal maximum. The highest mean values of primary production capacity during the growing season were about 520 mg C $m^{-3}$ $d^{-1}$ in Klobbfjärden and Hästholmsfjärden, and about 320 mg C $m^{-3}$ $d^{-1}$ in Keijvsalö östra fjärd.

Based on the classifications given by Lehmusluoto (1968 and 1969) and Niemi and Pesonen (1974a and b), the recorded levels of *in situ* primary production indicate that the waters were very eutrophic in the sea area around Hästholmen, at least in the late 1990s and at the beginning of this century. The primary production capacity exceeded the limit of eutrophic in the early 1970s and in very recent years.

The main factor affecting the transparency of the water in the Loviisa archipelago is inorganic particulate matter leaching from the drainage areas of the Kymijoki and Tesjoki Rivers, which results in an abundant diffusion of clayey river waters into the area, especially in spring. In addition, the transparency is affected by the amount of phytoplankton, which has increased due to eutrophication during the study period. Thus, the *Secchi* depth can also be used as an indicator of eutrophication. Since the transparency of the water

fundamentally affects the penetration of light and the thickness of the euphotic zone, it has a significant influence on the intensity of primary production and on the distribution of aquatic vegetation in different depth zones. In general, water transparency increases when moving from the inner areas towards the open sea, so that the lowest mean transparency was found in Klobbfjärden and the highest in Orrengrundsfjärden. As a whole, the water transparency has clearly decreased in the whole study area during the past 40 years. The decrease has been quite the same at all the sampling stations, except at the outermost station in Orrengrundsfjärden, where the decrease has been a little smaller. The mean *Secchi* depth values of the growing season decreased by 69 – 79 cm in Klobbfjärden, Hästholmsfjärden and Hudöfjärden, and by 33 cm in Orrengrundsfjärden between the beginning of the 1970s and the 2000s. Most of the decrease was evidently due to the general eutrophication process.

Eutrophication increases biological production in the water phase, and when the larger quantities of plankton die and sink to the sea floor, they increase the decomposition of organic matter at the bottom, and the oxygen reserves are easily used up. Depletion of oxygen is a common feature in deep areas of inlets isolated from the open sea, in which the exchange of water is limited. The deeps of Hudöfjärden and Hästholmsfjärden are, for natural reasons, problematic regarding the adequacy of oxygen reserves, due to their small volume and poor renewal of the near-bottom water below the sill depth. A deficiency of oxygen was already characteristic of the deeps in late summer in their natural state. The character of the bottom in the deep of Hudöfjärden (black, watery sludge that smelled of hydrogen sulphide) indicated reducing conditions prevailing at the surface of the sediment, and the seabed was already badly disturbed in the 1960s, when the first samples were taken (Bagge & Voipio 1967). However, in the deep of Hästholmsfjärden, the succession in the quality of the bottom and in the benthic fauna during the last 30 years have indicated yet further weakening of the oxygen conditions.

Since the 1970s, low oxygen concentrations have been repeated nearly every year in the deeps of Hudöfjärden and Hästholmsfjärden, but particularly in the 1990s and 2000s anoxic conditions have occurred rather regularly in late summer in their near-bottom water. Most probably the thermal discharges have played no part in the depletions of oxygen in the deep of Hudöfjärden, since the thermal effects may reach Hudöfjärden only occasionally in open water circumstances, and even then only slightly; and the seabed in the deep was already in bad condition before the construction work on the power plant was started. In the deep of Hästholmsfjärden, too, the primary reason for the oxygen problems have been local factors linked to the topographic characteristics and the limited exchange of water. Nevertheless, it seems likely that the thermal

discharges have increased the susceptibility of the deep area of Hästholmsfjärden to oxygen depletions, thus aggravating the development.

Strong remobilization of nutrients from the bottom sediments to the water phase has occurred in connection with the oxygen depletions in the deeps. In late summer, the total phosphorus concentrations have been, at their highest, about 50-fold, and the total nitrogen concentrations about 4–5-fold, in the near-bottom water, compared to their average concentrations in surface water. In 1998, about 675 kg of soluble reactive phosphorus and 1 860 kg of ammoniacal nitrogen were remobilized from the deep of Hästholmsfjärden (Lehtoranta and Mattila 2000).

The deeps of Hudöfjärden and Hästholmsfjärden have occupied a central position in the monitoring of the oxygen situation in the study area. However, short-term oxygen depletions appearing in these deeps are not very significant considering the whole sea area, because fish, for instance, are able to keep away from the deeps when the oxygen situation has deteriorated. On the other hand, the situation is worse for benthic animals, which repeatedly are exposed to a deficiency of oxygen, and because the bottom sediment has gone through drastic changes as a result of the oxygen depletions.

As well as in the deeps of Hudöfjärden and Hästholmsfjärden, the oxygen situation has been at least occasionally problematic at other sampling stations, too. In 1996, for instance, the oxygen situation was exceptionally poor, and at the end of August almost the whole hypolimnion of Hästholmsfjärden seemed to be anoxic, with the near-bottom water smelling strongly of hydrogen sulphide. At other times of the year, the oxygen situation has generally been reasonably good in the area.

The soft-bottom macrofauna has suffered a strong deterioration or even an almost total loss at many sampling stations in the study area during the past 40 years. A similar decline of the macrozoobenthos has been reported from the whole eastern Gulf of Finland (e.g. Andersin & Sandler 1991, Laine et al. 1997, Kangas et al. 2001). However, the local eutrophication process seems to have contributed to the decline of the bottom fauna in Hästholmsfjärden. Thermal discharges have increased the production of organic matter, which again has led to more organic bottom deposits. It is clear that the changes in the macrozoobenthos have been due to the general deterioration in the benthic conditions (black, watery sediments with a high content of organic matter and recurrent anoxic conditions) in the whole study area, as a consequence of the eutrophication process. The augmented plankton production has increased the amount of sinking organic matter, which has to be decomposed at the bottom. The quality of the bottom has changed as the amount of organic matter has increased in the surface sediments, and the consumption of oxygen has increased, which

in turn has led to a deficiency of oxygen and formation of hydrogen sulphide in the near-bottom water and to a deterioration of the benthic fauna.

Even though the first benthos studies carried out in the area in the 1960s already proved that the fauna is very poor in the sea area off Loviisa (Bagge and Voipio 1967), the abundance and biomass of the macrozoobenthos have further decreased, or the fauna has almost totally disappeared from many stations during the study period. The deterioration has been strongest at the outermost sampling stations in Orrengrundsfjärden and Vådholmsfjärden, but significant changes have also occurred in the deep of Hästholmsfjärden, at Station 5 situated nearest (400 m) to the cooling water outlet, and at Station 8 in Hudöfjärden (no permanent macrofauna was found in the deep of Hudöfjärden during the whole study period). Furthermore, the biomass of the zoobenthos has undergone a clear decrease since the late 1980s at Station 2 in Hästholmsfjärden, although the assemblage of the tubificid *Potamothrix hammoniensis* and the larvae of *Chironomus plumosus* gr. was still vigorous in very recent years. Actually, only the fauna at Station 1 in Klobbfjärden showed signs of strengthening in the 2000s through an increasing abundance of *Chironomus plumosus* gr. and *Potamothrix hammoniensis*.

The most noteworthy features in the trends of the macrozoobenthos during the past 40 years have been:

1. Invasion of many nonindigenous species into the area. Many of them have started to spread from the immediate vicinity of the cooling water outlet, probably assisted by the raised temperature. Some of them are highly expansive (e.g. *Marenzelleria* sp., *Mytilopsis leucophaeta*), causing a threat to and competition with the original species.
2. Almost total loss of *Monoporeia affinis* and *Macoma balthica* from the deep soft-bottom areas.
3. Strengthening of the status of *Potamothrix hammoniensis* and *Chironomus plumosus* gr. as core species; this was later threatened and in places displaced by *Marenzelleria* sp.
4. A general, and at some stations drastic, decrease in the zoobenthos, which is seen in both the abundance and biomass values at Stations 4, 5, 7 and 8, and in the biomasses at Stations 2 and 3.

The deterioration in the macrozoobenthos has not primarily been caused by the thermal effects of the power plant, but there has been a more wide-ranging trend in the background. The general eutrophication of the eastern Gulf of Finland has probably been the decisive reason for the changes in the zoobenthos in the outer archipelago of Loviisa (at least at Station 7), but it has also been an underlying factor in the changes recorded in Hästholmsfjärden. The

effects of the cooling water have probably been concerned in the changes that have occurred in Hästholmsfjärden, and possibly to a smaller degree at Station 4 in Vådholmsfjärden, at the farthest.

The most obvious, unambiguous and significant biological effect of the thermal discharge from the Loviisa power station has been the increase and eutrophication of the littoral vegetation in the discharge area. Spiked water milfoil *Myriophyllum spicatum* and the pondweeds *Potamogeton perfoliatus* and *Potamogeton pectinatus* together with the vigorous growths of several filamentous algae (e.g. *Cladophora glomerata, Ectocarpus siliculosus, Pilayella littoralis, Vaucheria* sp.) as their epiphytes have strongly increased in the littoral zone in the near vicinity of the cooling water outlet. The strongest increase in the phytobenthos has extended to a distance of about 1 km from the outlet. Weaker eutrophication of the littoral vegetation has appeared in the whole area of Hästholmsfjärden Bay, but outside it the phenomenon has been slighter and only observed in some places. The general increase in nutrient levels in the study area and in the whole Gulf of Finland has been the primary background reason for this succession, but in the end the eutrophication of the littoral vegetation has been a combined effect of the raised temperature and nutrient concentrations in the water.

In the studies of littoral vegetation, the eutrophication was most pronounced on the census transects situated on the south-western shore of Hästholmsfjärden, where the above-mentioned vascular plants and filamentous algae formed dense, luxuriant growths in late summer. The eutrophication began to strengthen in the 1980s, and progressed in the 1990s and 2000s. While the abundance of the dominant species increased, the overall species composition seemed to become poorer on the transects, although the detection of the smaller and scantier species was difficult among the very dense vegetation. Although the cooling water mainly flows southwards from Hästholmsfjärden, weaker eutrophication of the vegetation was also seen in the northern part of the bay. The phytobenthos was poorest on the transect situated just in front of the cooling water outlet (distance of 150 m). The relatively strong current and the consequent erosion of the bottom were probably the main reason for the scantiness of the vegetation on this transect.

Many perennial vascular plants, especially *Myriophyllum spicatum*, but also *Potamogeton perfoliatus, Potamogeton pectinatus* and *Phragmites australis* and many filamentous algae (*Cladophora glomerata, Ectocarpus siliculosus, Pilayella littoralis*, *Vaucheria* sp. and *Ulva* spp.) have benefited from the thermal effect and from the increase of nutrients in the water. Thermal discharges have a strong effect on over-wintering and the date of the start of the growing season. The increased illumination in the ice-free area in winter and the direct influence

of the temperature rise cause a significant advance in the start of the growing season for many species (*cf.* Snoeijs 1988). The changes in vegetation have been largest in areas that remain ice-free in winter. In addition to the above-mentioned species, the increased abundance of *Ceratophyllum demersum, Najas marina* and *Lemna trisulca*, as well as the disappearance of the bladder-wrack *Fucus vesiculosus,* the red algae *Ceramium tenuicorne* and *Polysiphonia violacea* and the stoneworts *Chara aspera* and *Tolypella nidifica* indicated increasing eutrophication on the transects in Hästholmsfjärden. On the other hand, *Fucus vesiculosus* has shown signs of recovery on the reference transect in Hudöfjärden after a decline in the early 1980s.

## 4.2 Olkiluoto

The amount of heat discharged into the sea from the Olkiluoto power plant is close to double that discharged from the Loviisa power plant. In the recent years, the amount has increased by 13–15% compared to that in 1996, and in 2006 it was 98.8 PJ $a^{-1}$. The flow rate of the cooling water is about 60 $m^3$ $s^{-1}$; it is taken from the surface of the sea, and the temperature rises about 11–13ºC in the condensers.

The hydrographical and biological studies carried out at Olkiluoto were more concise than those at Loviisa. The background studies were initiated with a relatively extensive programme in 1972, but since 1982 the monitoring has been carried out with a more restricted programme, primarily with the goal of providing necessary background data for the radiation monitoring and radioecological studies carried out in the area. Nevertheless, it was the ambition to maintain knowledge of the effects of the thermal discharges into the water recipient of the Olkiluoto power plant. As a consequence, the results discussed in this examination are mainly focused on the mean values of the summer months (June–August) only.

Ice conditions and water temperatures were not systematically monitored by STUK in winter, but in mild winters during the most recent decade, the ice-free area in front of Olkiluoto was often unbrokenly connected with the open Bothnian Sea almost throughout the winter season. During the open water periods in 1978–2006, there was an increasing trend in the mean surface water temperatures of the summer months at all sampling stations located off Olkiluoto. At Stations 2, 3 and 10, situated at distances of 2.1–3.2 km from the cooling water outlet, the mean surface water temperatures of the summer months increased by 2.8–3.0ºC during the 30 years. Since 1983, the average difference between the mean temperatures at Station 2 and that at Station 9 situated just seaward of Iso Kaalonpuhti Bay (at a distance of 1.2 km from

the mouth of the cooling water channel) was 5.0ºC. The highest surface water temperature observed was 29.5ºC at Station 9, while those at Stations 2, 3 and 10 were 23–24ºC. The openness of the sea area and the jet discharge mode certainly contribute to the lowering of the maximum temperatures at Olkiluoto.

The level of phosphorus in seawater is clearly lower in the Bothnian Sea than in the Gulf of Finland and in the Baltic Proper. Nevertheless, the mean total phosphorus concentrations in the surface water in the sea area off Olkiluoto increased by 60–70% between the early 1970s and the 2000s. In 2002–2003, the mean values were 16–17 µgP $l^{-1}$, with no significant difference occurring between the sampling stations. Most probably, the increase was associated with the general eutrophication trend on this part of the Bothnian Sea coast, corresponding to the trend observed in the open Bothnian Sea (HELCOM 2009). On average, coastal waters are more eutrophied compared with the adjacent open sea areas (Pitkänen 2004). The total nitrogen concentrations of the surface water increased during the 1970s and 1980s until 1991, after which the mean values of the summer months decreased to the same level as in the early 1970s. In 2000–2003, the mean values of the summer months in the sea area off Olkiluoto were 290–310 µgN $l^{-1}$.

Owing to the low nutrient concentrations, the level of phytoplankton primary production is lower in the Gulf of Bothnia than in the Gulf of Finland and the Baltic Proper. However, the annual production was also doubled in the Olkiluoto area between the early 1970s and the 2000s. Similarly, while the average production of the summer months in 1972–1982 was 185 mg C $m^{-2}$ $d^{-1}$, it was 311 mg C $m^{-2}$ $d^{-1}$ in 1989–2001 at the permanent monitoring station Olkiluoto 2.

The primary production capacity increased in parallel with the *in situ* primary production values. The highest single value (430 mg C $m^{-2}$ $d^{-1}$) during the summer months was measured just beyond the mouth of the cooling water channel in August 1992. However, the mean PP capacity values of the summer months only occasionally exceeded 200 mg C $m^{-2}$ $d^{-1}$ at Station 9 situated seaward of Iso Kaalonpuhti Bay, but never at the standard station Olkiluoto 2, thus indicating only slight eutrophication. The productivity of phytoplankton was often noticed to increase during entrainment through the cooling water systems, where plankton was exposed to mechanical stress and to a sudden temperature rise of 10–13ºC within 4–5 minutes when passing through the power plant, although in most cases the productivity decreased during the entrainment. The productivity seemed to increase to a greater degree only after the exit from the cooling water channel, and just reached a maximum at a distance of about 1.5 km from the outlet, where the temperature had already decreased and the plankton had had time enough to adapt to the prevailing conditions.

The salinity of water is approximately 1‰ higher at Olkiluoto than at Loviisa, which means that the biota is more vigorous and diversified. The abundance and biomass of the macrozoobenthos fluctuated strongly, however, at the Olkiluoto sampling stations, mainly due to the fluctuating occurrence of the core species *Macoma balthica*. Now and then, the mussel almost totally disappeared from the fauna at many stations, but then again multiplied at the same sites. The strong variability in the size-classes of *Macoma* was especially reflected in the total biomasses. The fluctuation was probably associated with the population dynamics of the species, but the lowered abundance in the late 1990s and early 2000s could also be associated with the changes in the quality of the bottom caused by progressive eutrophication. At some stations the decline of *Macoma* might also be due to competition/predation by *Marenzelleria* sp., which appeared in the area in 1992, and reached the status of dominant species repeatedly in the late 1990s and early 2000s, especially at stations having organic soft bottoms. *Marenzelleria* may especially cause predation pressure to the juvenile stages of *Macoma*.

A general change towards more organic and watery deposits was an obvious trend in the character of the seabed at the soft bottom stations in the sea area of Olkiluoto during the study period. As a whole, a general deterioration of bottom fauna communities was observed in the offshore areas along the Finnish coast of the southern Bothnian Sea during the 1980s (Mattila 1993). Some of the changes in the Olkiluoto area (such as the disappearance of *Monoporeia affinis*) might also be due to the general progress of the eutrophication in the Bothnian Sea, because the latter species also disappeared from Station 4 of Eurajoensalmi, which was not affected by the cooling water discharge. On the other hand, some species, e.g., *Potamopyrgus antipodarum,* the oligochaete *Potamothrix hammoniensis*, the chironomid larvae of the *Chironomus halophilus* group and *Procladius* sp. at Station 9 situated seaward of Iso Kaalonpuhti Bay seemed to be favoured by the cooling water discharge. In spite of the changes observed, the benthic fauna has remained strong and diversified in the Olkiluoto area.

# 5 Results of radioecological monitoring and studies

## 5.1 Loviisa

Tritium is clearly the most abundant radionuclide discharged from the Loviisa nuclear power plant into the sea. Annual discharges have varied between $1.2 \cdot 10^{12}$ and $1.7 \cdot 10^{13}$ Bq during the operational period of the power plant, representing 1–11‰ of the annual release limit ($1.5 \cdot 10^{14}$ Bq) set for liquid discharges of tritium from the Loviisa NPP. Between 1977 and 2007, the decay-corrected cumulative discharge of tritium was $1.82 \cdot 10^{14}$ Bq (decay-corrected to the end of 2006).

The concentrations of tritium originating from global fallout decreased in Finnish coastal waters from about 10–15 kBq $m^{-3}$ to less than 4 kBq $m^{-3}$ between the late 1970s and the early 2000s. Activity concentrations in excess of the decreasing level of fallout tritium were relatively frequently detected in Hästholmsfjärden, but more seldom in Klobbfjärden and Vådholmsfjärden. At the Reference Station R1 in Påsalöfjärden, concentrations that slightly exceeded the fallout level were detected 1–3 times. The highest concentration (120 kBq $m^{-3}$) was detected just off the cooling water outlet in the context of a special study, in which the spreading of liquid discharges in the recipient was monitored with a tight schedule. At the standard stations of Hästholmsfjärden and Klobbfjärden, the highest values were 33 kBq $m^{-3}$ and 24 kBq $m^{-3}$, and in Vådholmsfjärden 41 kBq $m^{-3}$. In general, the concentrations decreased below the fallout level or detection limit when passing out of Hästholmsfjärden.

During the first operational years of the power plant, small amounts (< 60 Bq $m^{-3}$) of local discharge nuclides other than tritium (e.g. $^{95}$Nb, $^{95}$Zr, $^{124}$Sb and $^{125}$Sb) were occasionally detected in seawater samples from Hästholmsfjärden, Klobbfjärden and Vådholmsfjärden, and once from Hudöfjärden. In a special discharge-monitoring survey carried out in December 1979, relatively high concentrations of $^{124}$Sb (maximum 5 500 Bq $m^{-3}$), $^{60}$Co (maximum 1 000 Bq $m^{-3}$) and $^{125}$Sb (maximum 670 Bq $m^{-3}$) were detected in samples taken from the immediate vicinity (< 300 m) of the cooling water outlet, but not at the more distant sampling stations. Four days after the last discharge batch, only small amounts of $^{60}$Co (5.2 Bq $m^{-3}$) and $^{124}$Sb (35 Bq $m^{-3}$) were detected in a seawater sample taken from the strait leading from Hästholmsfjärden to the south (at a distance of 1 km from the outlet). Since 1980, local discharge nuclides were detected only at the cooling water outlet, except for three separate observations in 1988 of $^{125}$Sb at the stations in Klobbfjärden and Vådholmsfjärden. During recent decades, discharges of radioactive substances from the Loviisa power

plant have significantly decreased, except those of tritium. Small amounts of local discharge nuclides were detected for the last time in seawater samples in 1993, when the activity concentrations of $^{54}$Mn, $^{58}$Co, $^{60}$Co and $^{110m}$Ag were 2–20 Bq m$^{-3}$, just off the cooling water outlet. Since then, local discharge nuclides were not detected in seawater samples.

The background level of $^{137}$Cs in seawater due to the nuclear weapons tests, carried out in the 1950s and 1960s in the atmosphere of the northern hemisphere, decreased until 1986 in the Loviisa area from about 45 Bq m$^{-3}$ to about 7–8 Bq m$^{-3}$. In 1979, elevated concentrations of $^{137}$Cs were detected in the context of the special discharge-monitoring survey mentioned above. 170–180 Bq m$^{-3}$ of $^{137}$Cs was then momentarily recorded in the close vicinity of the cooling water outlet, but beyond that point the concentrations decreased very soon to the background level with increasing distance from the outlet.

The Chernobyl fallout in 1986 caused a sudden and drastic rise in the $^{137}$Cs concentrations of surface water (3 000–5 200 Bq m$^{-3}$) for a short while. After the fallout, the concentrations decreased steadily and relatively rapidly in the Loviisa archipelago. In the whole Gulf of Finland, the decrease of $^{137}$Cs concentrations was clearly faster than that in the Bothnian Sea, due to the effective exchange of water with the Baltic Proper. After 1986, any traces of locally-discharged $^{137}$Cs could not be detected in seawater, owing to the dominating amount of Chernobyl-originated $^{137}$Cs in all the environmental samples. In general, the concentrations reflected the distance from the mainland, the values being a little higher at the innermost stations and decreasing towards the open sea.

Indicator organisms are widely used in the monitoring of radioactivity in the environment and in radioecological studies because of their ability to accumulate effectively radioactive substances from the surrounding medium; even small quantities are thus easily detected in indicator organisms, even though their concentrations in the medium (e.g., water) are below the detection limit. In a special study carried out at Loviisa in 1988–1989, local discharge nuclides, $^{110m}$Ag, $^{60}$Co and $^{54}$Mn, were almost exclusively detected only at the lower trophic levels of the ecosystem, i.e., in phytoplankton, zooplankton, macrophytes and benthic animals, but not in vertebrates, e.g., fish, waterfowl and seals, nor in their inner organs. On the other hand, the Chernobyl-originated isotopes $^{137}$Cs and $^{134}$Cs were most abundant in the last-mentioned groups, especially in the muscle tissues of predatory fish and seals. In fish-eating waterfowl, the concentrations were a little higher than in roach and Baltic herring. The caesium concentrations were generally lower in the inner organs than in the muscle tissues. *Fucus vesiculosus* seemed to be a less effective indicator for radiocaesium than fish flesh, but it was better than most other seaweeds and benthic animals.

On the other hand, *Fucus* was proved to be an excellent indicator for $^{60}$Co and $^{54}$Mn, as well as the relict crustacean *Saduria entomon* for $^{110m}$Ag.

From the very beginning, *Fucus vesiculosus* has been one of the key objects in the radioecological monitoring programmes and studies carried out in the vicinities of the Finnish nuclear power plants. The Chernobyl fallout caused a strong, sudden and transient appearance and rise of some 15–18 radionuclides in the *Fucus* samples. Many of these nuclides occurred momentarily in relatively high concentrations, but because of their short half-lives, the concentrations decreased and then totally disappeared very rapidly. The most important radionuclide in the fallout was $^{137}$Cs with a physical half-life of 30 years. At the end of May 1986, the highest concentrations of Chernobyl-originated $^{137}$Cs were 4 900 and 2 700 Bq kg$^{-1}$ d.w. at the Hästholmsfjärden sampling sites. By 2007, the concentrations had decreased to 22–38 Bq kg$^{-1}$ d.w. In general, the activity concentrations of $^{137}$Cs were somewhat higher in Hästholmsfjärden than outside it, which was probably due to the stimulating effect of the heated water on the accumulation rate of caesium, and to the effect of coastal proximity. Nevertheless, the $^{137}$Cs discharges from the power plant may also contribute to the concentrations in the vicinity of the cooling water outlet.

$^{60}$Co, $^{110m}$Ag, $^{54}$Mn and $^{58}$Co were the most frequently-detected local discharge nuclides in the *Fucus* samples. They were regularly detected since the start-up of the power plant, but their concentrations increased at least slightly or even markedly (in the case of $^{110m}$Ag) in connection with the Chernobyl fallout. Since then, the concentrations of these nuclides have continuously decreased; especially during the last ten years, their concentrations have sunk to very low values, thanks to the reduced discharges from the power plant. Despite this, small amounts of them have still been detected quite frequently at the A and B sites in Hästholmsfjärden, and in ever-decreasing amounts at the C site of Lilla Djupberget. In the samples of the regular monitoring programme, $^{60}$Co was, at its farthest, detected twice at Boistö (at a distance of 10 km) in the mid-1990s. In the special *Fucus* surveys carried out in the 1980s and 1990s along the Finnish coast, and separately at the NPP sites, the highest concentrations of $^{110m}$Ag, $^{60}$Co, $^{54}$Mn and $^{58}$Co were measured in 1980 in Hästholmsfjärden, while outside it the observations and concentrations declined rapidly. Soon after the Chernobyl accident (1987), small amounts of $^{110m}$Ag and $^{54}$Mn were recorded as far out as Storskarven and Orrengrund, but local discharge nuclides were never detected at the sampling sites of the open Gulf of Finland outside the Loviisa area. In 1999, only trace amounts of $^{60}$Co were found in the samples from Hästholmsfjärden and Hudöfjärden, and the detections of $^{54}$Mn, $^{110m}$Ag and $^{58}$Co were reduced to a very few, low values.

Studies on the seasonal fluctuation of radionuclide concentrations in *Fucus* showed that the activity concentrations in *Fucus* were highly dependent on the discharge amounts, and were in good agreement with the concentrations theoretically calculated from the discharge data. During the dischargeless periods, the loss of $^{58}$Co from *Fucus* in 23–26 weeks was 96–98%, that of $^{60}$Co 72–81% and that of $^{54}$Mn 67–73% in the same time. The effective half-lives for $^{58}$Co, $^{60}$Co and $^{110m}$Ag were estimated at 36, 72 and 81 days, respectively. In Barsebäck, Sweden, the biological half-lives of $^{58}$Co, $^{60}$Co, $^{54}$Mn and $^{65}$Zn in *Fucus* were found to be 60 ± 15 days (Mattsson 1980a). Dahlgaard and Boelskifte (1992) concluded that dilution through new growth is the main cause of the decreasing concentrations on the west coast of Sweden. On the Finnish coast the growth rate of *Fucus* is very slow, and the influence of new growth is certainly smaller. Various factors have been observed to affect the uptake, accumulation and release of radionuclides in algae, e.g., the season of the year, age of the tissue, salinity of the water, etc. (Carlson 1990). In Loviisa, the discontinuous discharges and the seasonal variation in hydrological mixing conditions in Hästholmsfjärden Bay seemed to be the main factors influencing the uptake of radionuclides. The impact of the season and the stage of the growing season seemed to be less than the impact of the discharged amounts, perhaps at least partly because the main part of the discharges took place at times other than the growing seasons.

In Finland, fish is the only seafood caught from the Baltic Sea and consumed by man. Radioactive substances have been monitored during the whole operational history of the Finnish NPPs in four fish species (Baltic herring, roach, pike and perch) twice a year in two catching areas at each NPP site. The Chernobyl fallout led to a rapid and noticeable increase of $^{137}$Cs in fish, though the concentrations remained relatively low compared to those in freshwater fish; this was due to the lower uptake of caesium in the marine environment owing to the relatively high potassium concentration in seawater. Accumulation of $^{137}$Cs was strongest in perch and pike, and lowest in Baltic herring and roach. The highest $^{137}$Cs concentration recorded in perch at Loviisa was 230 Bq kg$^{-1}$ f.w. Radionuclides that would unambiguously have originated from local discharges were not observed in fish samples caught at Loviisa. Only $^{40}$K, $^{90}$Sr, $^{134}$Cs and $^{137}$Cs were found in analyses of fish before the Chernobyl accident. In 1986–1987, the samples also contained small amounts of $^{110m}$Ag, which since then was again recorded in one sample of Baltic herring caught in Hästholmsfjärden in 1989 (0.1 Bq kg$^{-1}$ f.w.), but its origin might also have been the Chernobyl fallout. Traces of local discharge nuclides were not detected in the fingerling samples taken from the Loviisan Smoltti fish farm.

Due to the tendency of caesium to adsorb on minerals, soils and sediments (Cornell 1993), the caesium contents in sinking matter rose drastically soon

after the Chernobyl accident in 1986. The maximum concentration of $^{137}$Cs in sinking matter collected in May from the deep of Hästholmsfjärden was 48 000 Bq kg$^{-1}$ d.w. After that, the concentrations began to decrease rapidly as fresh particulate material settled on the bottom and the concentrations in the water phase became smaller. By September 1986, the concentration had already decreased to 5 500 Bq kg$^{-1}$ d.w., and the concentrations continued to diminish steadily, so that in 2007 the mean concentration of $^{137}$Cs was 420 Bq kg$^{-1}$ d.w. in sinking matter collected from the deep of Hästholmsfjärden. In addition, the Chernobyl fallout contributed to the strong and transient appearance of some ten other radionuclides in sinking matter. However, because the nuclides were almost exclusively very short-lived, their concentrations decreased and then very soon totally disappeared from the sinking matter samples.

Local discharge nuclides ($^{60}$Co, $^{110m}$Ag, $^{58}$Co, $^{54}$Mn and $^{125}$Sb) were rather regularly detected in sinking matter collected from Hästholmsfjärden and Klobbfjärden. In addition, $^{60}$Co was abundantly detected in low concentrations in the samples collected from Station 4a in Vådholmsfjärden lying seaward of the straits leading from Hästholmsfjärden Bay to the outer archipelago (at a distance of about 4 km from the outlet). Minor amounts of $^{60}$Co, $^{54}$Mn and $^{125}$Sb were also detected from time to time in sinking matter collected from Reference Station R1 located in Påsalöfjärden (14 km from the power plant). In general, the activity concentrations of the local discharge nuclides were clearly lower after the Chernobyl accident than before it, and the concentrations decreased markedly during the 1990s. In 2002 and 2006, there were small peaks in the annual mean concentrations of these three nuclides in sinking matter collected from the deep of Hästholmsfjärden.

The Chernobyl fallout significantly increased the total amounts of $^{137}$Cs and $^{90}$Sr in the seabed at Loviisa, but not those of $^{239,240}$Pu. Before 1986, the highest cumulative amounts of $^{137}$Cs in the study area were 2 300–8 500 Bq m$^{-2}$. After the accident, the highest total inventories of $^{137}$Cs were 61 100–81 300 Bq m$^{-2}$ at the stations located farthest from the power plant, and 32 100–49 200 Bq m$^{-2}$ at the stations in Hästholmsfjärden. At Station 3 in the deep of Hästholmsfjärden, the total amounts of $^{90}$Sr varied between 78 and 110 Bq m$^{-2}$ in 1975–1980, while they were 210 and 390 Bq m$^{-2}$ in 1986 and 1990. The sedimentation rate was estimated to be about 8 mm a$^{-1}$ at Station 3, based on the depth of the Chernobyl-originated $^{137}$Cs peak in 2002.

$^{60}$Co was detected in the sediment samples taken from Hästholmsfjärden for the first time in 1983. The highest concentrations (56–69 Bq kg$^{-1}$ d.w.) occurred in the uppermost sediment layers in 1986 and 1988. Since then, the $^{60}$Co contents in the surface sediment layer have continuously decreased, and since 1998 the top values have occurred in the deeper layers, 5–10 and 10–15 cm.

In 1998, the depth range of $^{60}$Co extended to 15–20 cm at Station 3, which was in good agreement with the operational history of the power plant and the estimated sedimentation rate at the station. At the same time as the cobalt peaks became buried deeper into the underlying sediment layers, the concentrations decreased. In 2006, the highest concentrations of $^{60}$Co were about 6 Bq kg$^{-1}$ d.w. in Hästholmsfjärden, and the maximum concentration occurred in the 15–20 cm slice at Station 5 situated 400 m off the cooling water outlet. The farthest observation of $^{60}$Co in sediments (1.7 Bq kg$^{-1}$ d.w.) was recorded in 1990 from Station 7 of Orrengrundsfjärden at a distance of 5 km from the power plant.

## 5.2 Olkiluoto

Tritium is clearly also the most abundant radionuclide discharged into the sea at Olkiluoto. Annual discharges of tritium have varied between $8.6 \cdot 10^9$ and $3.6 \cdot 10^{12}$ Bq during the operational period of the power plant, representing 0.05–20‰ of the annual release limit ($1.8 \cdot 10^{13}$ Bq) set for liquid discharges of tritium from the Olkiluoto NPP. The decay-corrected cumulative discharge up to 2007 was $2.28 \cdot 10^{13}$ Bq (decay-corrected to the end of 2006). Activity concentrations exceeding the decreasing level of fallout tritium were now and then detected at the sampling station situated in Iso Kaalonpuhti Bay in front of the cooling water outlet, but only very seldom at other stations. The highest activity concentration of tritium found in Iso Kaalonpuhti Bay was 58 kBq m$^{-3}$.

Small quantities of other local discharge nuclides (e.g., $^{60}$Co, $^{54}$Mn, $^{125}$Sb, $^{58}$Co) were detected in seawater more often and more widely than at Loviisa. The highest activity concentration detected in Iso Kaalonpuhti Bay was 32 Bq m$^{-3}$ of $^{60}$Co. A trace amount of $^{60}$Co (5 Bq m$^{-3}$) was once detected at Station 15, situated 10 km north of the power plant. During recent decades, discharges of radioactive substances from the Olkiluoto power plant have significantly decreased, except those of tritium. Consequently, local discharge nuclides were not detected in the seawater samples of the regular monitoring programme since 1998. The highest $^{137}$Cs concentrations detected in seawater at Olkiluoto after the Chernobyl fallout were 1 000–1 100 Bq m$^{-3}$. After that, the caesium concentrations decreased steadily and relatively rapidly. However, in the Bothnian Sea the decrease of $^{137}$Cs has been clearly slower than in the Gulf of Finland, due to the retarded exchange of water with the Baltic Proper. In 2000–2001, the average concentrations in surface water were 70–71 Bq m$^{-3}$, while they were 43–47 Bq m$^{-3}$ at Loviisa. Since 1986, traces of locally-discharged $^{137}$Cs were not able to be detected in seawater samples, owing to the dominating amount of Chernobyl-originated $^{137}$Cs in all the environmental samples.

Indicator organisms are an excellent tool in the monitoring of radioactive substances in the environment, especially if the concentrations in sea water fall below the detection limit. In a special study carried out in 2001 at Olkiluoto, the indicator value of different members of the local aquatic ecosystem was compared with regard to environmental monitoring purposes. As at Loviisa, discharge nuclides from the local nuclear power plant ($^{60}$Co and $^{54}$Mn) were detected only at the lower trophic levels of the ecosystem, but not in fish or birds, nor in their inner organs or reproductive products. The activity concentrations of the Chernobyl-originated $^{137}$Cs were highest in the flesh, liver and entrails of perch and pike and in periphyton. In the milt and spawn of fish and in the muscle tissues of aquatic birds, the concentrations were generally lower than in fish flesh. However, in some fish-eating birds, the highest $^{137}$Cs concentrations were not in the muscles but in the liver and entrails. In bird's eggs, the concentrations were generally very low. The best indicators for locally-discharged $^{60}$Co were periphyton, *Myriophyllum spicatum, Fucus vesiculosus* and zooplankton. Algae have been successfully used in monitoring discharges of radionuclides also in the environments of Swedish nuclear power plants. *Fucus* is also a commonly-used indicator in Sweden (Notter 1983, Neumann et al. 1991), but in addition diatoms (Snoeijs & Notter 1993) and algal growths on installed substrates (Snoeijs and Simenstad 1995) have proved to be excellent monitoring objects. Algal samples provide useful complements to water and sediment samples in the monitoring programmes, since radionuclide concentrations are much higher in algal samples and are proportional to the discharges (Snoeijs and Simenstad *op.cit.*).

In Finland, diatoms have not specifically been used, but our periphyton samples are in principle quite comparable with the algal growth samples monitored in Sweden, although the species composition has not been determined. The chief indicator organism, at Olkiluoto too, has in any case been *Fucus vesiculosus*. The Chernobyl fallout caused a strong, sudden and transient appearance and rise of some 20 radionuclides in the *Fucus* samples at the end of April 1986. A sample collected off Olkiluoto only some hours after the arrival of the fallout cloud (dry deposition) contained a long list of fresh fallout nuclides which were exotic to the area, and most of them short-lived. Many of these nuclides occurred momentarily in relatively high concentrations, but because of their short half-lives, the concentrations decreased and totally disappeared very rapidly. However, the most important radionuclide in the fallout was $^{137}$Cs, with a physical half-life of 30 years. The maximum concentration of Chernobyl-originated $^{137}$Cs, 1 300 Bq kg$^{-1}$ d.w., was not reached until the 21$^{st}$ of May, but by August the concentration had already decreased to 280 Bq kg$^{-1}$ d.w. In 2006, the concentrations were 24–37 Bq kg$^{-1}$ d.w. in the sea area off Olkiluoto. Thus, the concentrations in *Fucus* had decreased to the same level as at Loviisa,

though the caesium concentrations in seawater had lessened much slower in the Bothnian Sea. In general, the activity concentrations of $^{137}$Cs have been highest just in front of Iso Kaalonpuhti Bay, which is probably due to the stimulating effect of the heated water on the accumulation rate of caesium, and in part also, to the effect of coastal proximity. Nevertheless, the $^{137}$Cs discharges from the power plant may also have contributed to the concentrations in the vicinity of the cooling water outlet.

$^{60}$Co, $^{54}$Mn and $^{58}$Co were the most frequently-detected local discharge nuclides in the *Fucus* samples: they were regularly detected after the start-up of the power plant at the sampling sites of Iso Kaalonpuhti Bay and Kalliopöllä. In contrast to Loviisa, their concentrations did not increase in connection with the Chernobyl fallout, but somewhat later. The concentrations of $^{60}$Co and $^{54}$Mn reached their maxima (170 and 150 Bq kg$^{-1}$ d.w.) in 1990, and after an interim decline in 1993 they increased again in 1995 and 1996, but since then they have significantly decreased at all the sampling sites. In general, there has been a clear difference in the concentrations of $^{60}$Co, $^{54}$Mn and $^{58}$Co between the sampling sites, with concentrations at the sites nearest of the cooling water outlet (Iso Kaalonpuhti and Kalliopöllä) being clearly higher than those at the outer sampling sites (Iso Pietari, Iso Siiliö and Vähäkrunni). Nevertheless, these nuclides have been quite regularly detected at all the permanent sampling sites. The distance of Iso Pietari from the cooling water outlet is 12 km.

In the special *Fucus* surveys carried out in the 1980s and 1990s along the Finnish coast and separately at the NPP sites, $^{60}$Co was detected in practically all samples taken in the near vicinity (12–13 km) of the Olkiluoto power plant. The concentrations were highest (maximum 134 Bq kg$^{-1}$ d.w.) in Iso Kaalonpuhti Bay and decreased with increasing distance, being generally less than 5 Bq kg$^{-1}$ d.w. at the outermost sites. $^{54}$Mn was detected in lower concentrations at all the sites in the near vicinity of Olkiluoto in 1987 and 1991, and at almost all the sites in 1981 and 1995, but in 1999 the observations of $^{54}$Mn were restricted to only the five neighbouring sites. Outside the Olkiluoto area, trace amounts (generally 1 Bq kg$^{-1}$ d.w., or less) of $^{60}$Co were also detected in *Fucus* samples collected along the west coast of Finland. At the greatest distance, small quantities of $^{60}$Co (0.2 Bq kg$^{-1}$ d.w.) were detected in 1995 at Närpiö in the north (137 km) and at Isokari in the south (61 km).

A power function, established by Mattsson et al. (1980a) and Nilsson (1981) for the distribution of discharged activation products from the Barsebäck NPP on the west coast of Sweden, was applied to the distribution pattern of $^{60}$Co on the Finnish west coast. The exponent 1.4 also fitted the results from Olkiluoto very well, especially at shorter distances, and the decrease in activity concentrations was fairly well in accordance with the calculated values. In the

southward direction, the results from the sampling sites off Åland seemed to diverge from the model, which probably means that the source of $^{60}$Co there was not Olkiluoto, but Forsmark, which is located on the Swedish east coast closer to Åland (distance 70 km) than Olkiluoto is. This conclusion is strengthened by the fact that the currents on the east coast of Sweden are directed southwards, and that the $^{60}$Co discharges from Forsmark have generally been larger than those from Olkiluoto.

Although residues of discharge nuclides from the Olkiluoto NPP could be detected at relatively long distances from the power plant, their relevance regarding an environmental risk is insignificant. The minimal concentrations found in *Fucus* are much rather interesting curiosities that highlight the sensitivity of the analysis methods, the usability of *Fucus* as an indicator organism, and the potential of small amounts of radionuclides in tracer studies. The minor amounts of the relatively short-lived radionuclides are far below any risk level. The spreading of $^{60}$Co and some other radionuclides in detectable quantities only to the Finnish coast of the Bothnian Sea, but not to that of the Gulf of Finland, was probably due to the larger discharges of the nuclides from Olkiluoto than from Loviisa, but maybe more significantly due to the free exchange of water and better diffusion capability of discharges from Olkiluoto.

Studies on the seasonal fluctuation of radionuclide concentrations in *Fucus* demonstrated that the quantities of the discharges were more important in determining the concentrations in *Fucus* than the impact of the season or other related biological factors. In Iso Kaalonpuhti Bay, the dependence between the activity concentrations in *Fucus* and the magnitude of the discharges was clear, although the radionuclide concentrations in seawater fluctuate very rapidly at this site, because it is located just off the outlet channel and is directly exposed to the cooling water flow. At the more distant sampling sites of Kalliopöllä and Valkiakari, the synchronism between the activity concentrations of local discharge nuclides in *Fucus* and the discharges themselves was not so clear; the concentrations were more constant, reflecting a more even dispersion of the discharge nuclides in the water at greater distances.

Radioactive substances have been monitored in Baltic herring, roach, pike and perch at Olkiluoto during the whole operational history of the power plant twice a year in two catching areas. The Chernobyl fallout led to a rapid and noticeable increase of $^{137}$Cs in fish, though the concentrations remained relatively low (*cf.* Loviisa). The accumulation of $^{137}$Cs was strongest in perch and pike, and lowest in Baltic herring and roach. The highest $^{137}$Cs concentration recorded in perch at Olkiluoto was 220 Bq kg$^{-1}$ f.w. Discharge nuclides from the local power plant were detected in low concentrations more frequently than at Loviisa. Before the Chernobyl accident, trace amounts ($< 0.2$ Bq kg$^{-1}$) of $^{58}$Co and

$^{60}$Co were detected in one Baltic herring sample caught in 1984. Since 1988, small amounts (0.07 to 0.44 Bq kg$^{-1}$ fresh weight) of $^{60}$Co, $^{65}$Zn, $^{54}$Mn and $^{110m}$Ag were detected now and then in fish samples taken from Olkiluoto. Nine of the 14 observations were in roach and four in perch, but local discharge nuclides were not recorded in pike. Ten of the 14 findings were of $^{60}$Co, the last ones being from 2003. Due to the very low concentrations, the findings are more a curiosity in the records than a matter of concern. Minor amounts of $^{54}$Mn and $^{60}$Co (0.05 and 0.13 Bq kg$^{-1}$ fresh wt.) were detected once in a rainbow trout fingerling sample taken from the Olkiluoto fish farm in 1989. The low caesium concentrations in the farmed fish were traced to the presence of a low content of caesium in the feed used in the fish farms.

The Chernobyl accident also caused a significant rise in the $^{137}$Cs concentrations of sinking matter at Olkiluoto, although the increase was much smaller than at Loviisa. The maximum concentrations in the samples collected in the spring and early summer of 1986 were 6 000–13 000 Bq kg$^{-1}$ d.w. After that, the concentrations began to decrease rapidly as the concentrations in the water phase became smaller and fresh particulate material settled on the bottom. In 2007, the mean concentrations of $^{137}$Cs in sinking matter collected from Olkiluoto were 280–340 Bq kg$^{-1}$ d.w. Moreover, the Chernobyl fallout caused a sudden increase and a slow decrease in the concentrations of certain short-lived radionuclides in sinking matter samples at Olkiluoto similar to that presented for the Loviisa area, but the concentrations were clearly lower at Olkiluoto.

Local discharge nuclides (primarily $^{60}$Co, $^{54}$Mn, $^{58}$Co and $^{125}$Sb) were mainly detected in sinking matter collected from the close vicinity of the power plant, on the west and north side of Olkiluoto Island. In addition, $^{60}$Co was quite regularly, and $^{54}$Mn from time to time, detected in samples collected from Station 15 situated 10 km north of the power plant. Since the mid-1990s, the concentrations of the local discharge nuclides decreased strongly and also stayed at a very low level in the 2000s. The indicator value of sinking matter in the monitoring of radioactive discharges from the power plant was good. In general, there was a good agreement between the peaks in the discharge data and in those of the observed concentrations in sinking matter.

The total amounts of $^{137}$Cs and $^{90}$Sr in the seabed increased significantly as a consequence of the Chernobyl fallout, but not those of $^{239,240}$Pu. Before 1986, the highest cumulative amounts of $^{137}$Cs at the sampling stations in the Olkiluoto sea area were 2 300–5 200 Bq m$^{-2}$. After the accident, the highest total inventories of $^{137}$Cs were 46 600–47 600 Bq m$^{-2}$ at the soft bottom stations and 600–4 700 Bq m$^{-2}$ on the erosion bottoms. The total amounts of $^{90}$Sr varied between 74 and 260 Bq m$^{-2}$ before the Chernobyl accident, while they were 490–520 Bq m$^{-2}$ in 1987. The sedimentation rate was estimated to be about

11 mm $a^{-1}$ at Station 2, based on the depth of the Chernobyl-originated $^{137}Cs$ peak in the sediment in 2003.

After 1983, $^{60}Co$ was detected in bottom sediments at all the sampling stations monitored. At Station 9, in front of the cooling water outlet, the activity concentration of $^{60}Co$ was then 50 Bq $kg^{-1}$ d.w. in the uppermost 0–5 cm sediment layer, and 5.7 Bq $kg^{-1}$ d.w. in the 5–10 cm layer. A clear rise was noticed in the concentrations in connection with the Chernobyl accident, but the highest values recorded in any of the sediment samples were not detected until 1991–2003. The highest single activity concentration of $^{60}Co$ (83 Bq $kg^{-1}$ d.w.) was recorded at a depth of 7–8 cm in a 1-cm slice taken from Station 2 in 2003. At Station 9, the maximum value was then 79 Bq $kg^{-1}$ d.w. in the 0–5 cm surface layer. The depth profiles at Station 2 indicate decreasing discharges of $^{60}Co$. The most distant observations of $^{60}Co$ in sediments were recorded in 1995 at Kuuskajaskari (2.6 Bq $kg^{-1}$ d.w.), 16 km to the south of Olkiluoto, and at Pirskerinfäärtti (4.9 Bq $kg^{-1}$ d.w.), 17 km to the north of the cooling water outlet measured by sea. In 2003, cobalt was no longer found in the surface layer, and the concentrations in the deeper layers had slightly decreased at the more distant stations, but the maximum concentrations had slightly increased in the vicinity of the power plant.

# 6 Comparison of the results from the two areas

The characteristics of the sea areas surrounding the Finnish nuclear power plants differ from each other in many details. The Finnish south coast is characterized by a narrow archipelago zone between the mainland and the open sea. In the Bothnian Sea, the basically open coast is only in places sheltered by a narrow archipelago. The Loviisa NPP lies in the outer part of the inner archipelago and the recipient of the thermal and radioactive discharges is a semi-enclosed bay between the mainland and the islands, which is connected to the outer sea only through narrow and shallow sounds. Islands and a successive series of underwater sills isolate Hästholmsfjärden Bay from the currents and mixing processes of the surrounding sea areas, and this leads to a limited exchange of water in the recipient. The Olkiluoto site is clearly more exposed to the sea. The recipient sea area is quite open. It is bordered by smaller islets, skerries and rocks, but the water exchange with the open Bothnian Sea is more or less free. In addition, the sea area at Loviisa is characterized by isolated small-area deeps inside the archipelago, whereas the seafloor off Olkiluoto is shallower and more even without notable deeps.

The nutrient level of seawater in the archipelago off Loviisa, and in the whole Gulf of Finland, is much higher than in the Olkiluoto area and in the whole Gulf of Bothnia. In general, the total phosphorus and total nitrogen concentrations in the surface water were at least 1½ times higher at Loviisa. Based on this long-time difference in nutrient levels between the Gulf of Finland and the Gulf of Bothnia, the underlying eutrophication process has progressed much farther in the Loviisa area. The salinity of the water is approximately 1‰ higher in the sea area off Olkiluoto than in Hästholmsfjärden at Loviisa. This apparently small difference is, however, quite crucial, because the salinity in Loviisa is so low that many more marine organisms live there at the limit of their distribution range, which again makes the biota sensitive to any additional stress. Furthermore, as a result of the low salinity the biota in the Loviisa area is much poorer, whereas the biota in Olkiluoto is more diversified and robust.

The cooling water systems of the Loviisa and Olkiluoto power plants differ from each other on two counts. At Olkiluoto, the coolant is taken from surface water, whereas at Loviisa it is taken from a depth of 8.5 – 11.1 m. The latter mode provides colder intake-water and ensures lower temperatures of the discharged water in summer. At Loviisa, the heated water is discharged into the sea over a broad, curved embankment, which spreads the effluent out on the surface of the receiving water body. The principle in this kind of surface layer discharge design is to disperse the effluent over a large surface area, and thus enhance

the transfer of heat into the atmosphere. The mixing of the effluent with the receiving water is generally poor, and the plume remains discrete for some distance from the outfall.

At Olkiluoto, the heated water is discharged into the sea as a strong jet flow through a long convergent discharge channel. The rapid jet flow mixes quickly after discharge with the receiving water, which means a faster drop in the highest temperatures soon after the discharge, but a more wide-ranging spread of slightly-elevated temperatures.

The rated power of the Olkiluoto units (840 $MW_e$ each) is much higher than those of Loviisa (488 $MW_e$ each), and hence, the amount of heat discharged into the sea from the Olkiluoto power plant is close to double that discharged from the Loviisa power plant. Although the cooling water at Olkiluoto is taken from the surface, and the thermal load is larger, the average temperatures of surface water have generally only been some degrees higher in front of Iso Kaalonpuhti Bay at Olkiluoto than in Hästholmsfjärden at Loviisa. The jet discharge mode and the openness of the discharge area, and consequently, the effective exchange of water, play an important role in lowering the temperatures in the Olkiluoto area. At Loviisa, the cooling water has raised the mean surface water temperatures of the growing season by 4–5ºC at a distance of 0.4 km from the outlet, and by 2.5–3ºC at a distance of 1–2 km in Hästholmsfjärden. At its farthest, the temperature rise was statistically significant in Klobbfjärden (2.4ºC) at a distance of 3.4 km and in Vådholmsfjärden (1.3ºC) at a distance of 2.8 km from the outlet. At Olkiluoto, the mean surface water temperatures of the summer months rose by 2.8–3.0ºC at distances of 2.4 and 2.0 km from the outlet. The average difference between the station situated just in front of Iso Kaalonpuhti Bay and the before-mentioned stations was about 5ºC during the summer months. The discharge of cooling water is directly focused on this station.

Insufficient observational material was available about the ice conditions at Olkiluoto in winter for comparative consideration. However, in mild winters during the most recent decade, the ice-free area seaward of Olkiluoto has often been unbrokenly connected with the open Bothnian Sea almost throughout the cold season. On the other hand, it is likely that, at Olkiluoto, the heated effluent does not spread to wide areas as discrete warm water 'lenses' under the ice as it does at Loviisa, because the salinity of the discharged water does not substantially deviate from that of the recipient water and the effluent is effectively mixed by the jet discharge in the wide ice-free area in front of the outlet. This probably lessens the risk to fishermen moving on the ice. At Loviisa, the extent of the ice-free area has been 3–8 $km^2$ in the mild or very mild winters typical of recent years. Under the ice, the warm water has affected the ice and

temperature conditions in normal ice winters at distances of >10 km west of Hästholmsfjärden.

Nutrient concentrations increased strongly in the Loviisa area and in the whole Gulf of Finland during the 40-year study period. On average, the total phosphorus concentrations in surface water doubled and the total nitrogen concentrations increased 1½-fold between the first half of the 1970s and 2000. Consequently, the annual primary production of phytoplankton more than doubled in the Loviisa area between the late 1960s and the late 1990s. The rise of primary production was fairly parallel both in the intake and outlet areas of the cooling water, but the thermal discharges contributed to a slightly stronger increase in Hästholmsfjärden than in Hudöfjärden, so that after the start-up of the power plant the production was generally higher in Hästholmsfjärden, while the situation was the opposite before that. In the 1990s and 2000s, the annual primary production values demonstrated recurrently a significant eutrophy in both areas (Hästholmsfjärden and Hudöfjärden).

At Olkiluoto, the nutrient concentrations also increased, but to a lesser extent. Between the early 1970s and the 2000s, the total phosphorus concentrations increased by 60–70%, but in spite of that, the total phosphorus and total nitrogen concentrations of the surface water at Olkiluoto were only about half of those at Loviisa at the turn of the millennium. Consequently, although the phytoplankton primary production also doubled at Olkiluoto between the early 1970s and the 2000s, the annual values there were, even at their highest, still less than half of the highest values at Loviisa. The level of annual production indicated only slight eutrophication at Olkiluoto. The primary production capacity increased in parallel with the *in situ* production in both areas. As a consequence of the lower level of eutrophication in the Bothnian Sea, the transparency of the water was more than 1 m greater in the discharge area off Olkiluoto than at Loviisa.

Besides the increased nutrient levels and water temperatures, the prolonging of the growing season as a consequence of the shortened ice winter has been the main stimulating factor for increased primary production. At the same time as the temperature rise increases the metabolic rate of organisms and the production of organic matter, it also accelerates the decomposition of organic matter in the receiving water bodies. In the deeps of Hudöfjärden and Hästholmsfjärden, the increased decomposition has led to depletion of oxygen in near-bottom water in late summer. The anoxic conditions have become an annual event, and they have affected a strong remobilization, predominantly of phosphorus, but also of nitrogen, from the bottom sediments to the water phase. This internal nutrient load has become the most important local source of nutrients in the area. The depletion of oxygen in late summer has been

a natural feature of the deeps, but the thermal load seems to have increased the susceptibility of the deep area of Hästholmsfjärden to oxygen depletions. Although the eutrophication process has also occasionally led to slightly decreased oxygen situations in the near-bottom water at Olkiluoto, notable remobilization of nutrients from the sediments to the water phase has not been observed there.

Due to the low salinity, the macrozoobenthos at Loviisa is by nature very poor in species and few in number, in contrast to that at Olkiluoto, which is more vigorous and diversified. An essential difference is the scarcity of the Baltic Tellin, *Macoma balthica* in the soft-bottom fauna at Loviisa, whereas this typical species of Finnish coastal waters furthermore dominates in the benthic communities at Olkiluoto. Continuing from this scant starting-point, the soft bottom macrofauna has suffered a strong deterioration or even an almost total loss at a majority of the sampling stations at Loviisa during the past 40 years. Although a similar decline in the macrozoobenthos has been reported from the whole eastern Gulf of Finland, the local eutrophication process seems to have contributed to the decline of the bottom fauna in Hästholmsfjärden. Thermal discharges have increased the production of organic matter, which again has led to more organic bottom deposits unfavourable for the macrozoobenthos. The general eutrophication of the Gulf of Finland has probably been the decisive reason for the deterioration of the fauna at the outermost sampling stations, and it has also been an underlying factor in the changes recorded in Hästholmsfjärden, but the local impact of the cooling water has strengthened the decreasing trend there.

The organic content of the bottom sediments has also increased at Olkiluoto, and consequently slight indications of deteriorated zoobenthos assemblages have been registered. However, the benthic fauna has in general remained strong and diversified in the Olkiluoto area. As a common feature, the highly expansive immigrant polychaete, *Marenzelleria* sp., appeared in the discharge areas of the Loviisa and Olkiluoto power plants in 1992. It seems to favour and benefit from heated water, and colonized the benthic communities in both areas very quickly, causing a threat to and competition with the original species there.

The most obvious, unambiguous and significant biological effect of the thermal discharge from the Loviisa power station has been the increase and eutrophication of the littoral vegetation in the discharge area. Spiked water milfoil *Myriophyllum spicatum*, the pondweeds *Potamogeton perfoliatus* and *Potamogeton pectinatus* and vigorous growths of several filamentous algae (e.g. *Cladophora glomerata, Ectocarpus siliculosus, Pilayella littoralis, Vaucheria* sp.) as their epiphytes have strongly increased in the littoral zone in the vicinity of the cooling water outlet. The littoral vegetation at Olkiluoto has not

been investigated by us in recent years, but very similar trends were already observed there in our studies in 1975–1982 (Keskitalo and Ilus 1987). Cursory observations since then have shown that the eutrophication process of the littoral vegetation has proceeded in Iso Kaalonpuhti Bay and in the shallow area south of Susikari Island.

The detection of local discharge nuclides in environmental samples is principally determined by the discharged amounts. In addition, the mode of the cooling water discharge and the exchange rate of water in the discharge area affect the distribution and concentrations of the released radionuclides in the water recipient. The salinity of the water may slightly affect the uptake of certain radionuclides in biota (e.g., a higher salinity reduces the uptake of caesium and strontium), but on the other hand low salinity may retard the vital functions and the growth rate of organisms, for instance in the salinity condition typical of the Loviisa area, where many marine or brackish water organisms live at the limit of their distribution range. In principle, increased temperature stimulates the vital functions and the uptake of radionuclides and many harmful substances.

The discharges of tritium from the Loviisa power plant have been larger than those from Olkiluoto, and consequently, the detection of tritium in seawater samples was more frequent, and the concentrations higher and more widespread in the Loviisa area. In general, the concentrations decreased below the fallout level or the detection limit of tritium when passing out of Hästholmsfjärden. However, elevated concentrations of tritium were recurrently recorded in Vådholmsfjärden, and small quantities of it were rarely detected at the reference station in Påsalöfjärden. At Olkiluoto, locally discharged tritium was now and then detected in Iso Kaalonpuhti Bay, but very seldom at the other stations. The highest activity concentrations of tritium recorded in seawater were 120 kBq $m^{-3}$ at Loviisa and 58 kBq $m^{-3}$ at Olkiluoto.

The effect of the Chernobyl fallout was clearly stronger in the aquatic environment at Loviisa than at Olkiluoto. The highest concentrations of Chernobyl-originated $^{137}Cs$, observed soon after the fallout in seawater, sinking matter and *Fucus* were 5 200 Bq $m^{-3}$, 48 000 Bq $kg^{-1}$ d.w. and 4 900 Bq $kg^{-1}$ d.w. at Loviisa, and 1 100 Bq $m^{-3}$, 13 000 Bq $kg^{-1}$ d.w. and 1 300 Bq $kg^{-1}$ d.w. at Olkiluoto, respectively. The concentrations decreased in both areas very rapidly during the first days, weeks and months after the fallout, but, on a longer time-scale, the $^{137}Cs$ concentrations in seawater clearly decreased faster in the Loviisa area and the whole Gulf of Finland than in the Olkiluoto area and the Bothnian Sea. This was due to the more effective exchange of water between the Gulf of Finland and the Baltic Proper compared with that between the Bothnian Sea and the Baltic Proper. In 2000–2001, the average concentrations of $^{137}Cs$ in seawater were 70–71 Bq $m^{-3}$ at Olkiluoto, when they were 43–47 Bq $m^{-3}$ at Loviisa. In fish, the

caesium concentrations at Loviisa and Olkiluoto decreased in the same ratio as in seawater, but in *Fucus* the concentrations decreased much slower at Loviisa. Thus, in the 2000s the $^{137}$Cs concentrations in *Fucus* were slightly higher at Loviisa than at Olkiluoto, although the concentrations in seawater were clearly lower at Loviisa. In sinking matter, the $^{137}$Cs concentrations at Loviisa were clearly higher than at Olkiluoto, due to the higher content of solid matter (clay particles) in the water.

Excluding tritium, the most important local discharge nuclides detected in environmental samples were $^{60}$Co, $^{110m}$Ag, $^{54}$Mn $^{58}$Co and $^{124}$Sb. In earlier years, the discharges of $^{60}$Co, $^{54}$Mn and $^{58}$Co from Olkiluoto were clearly higher than those from Loviisa (Figs. 66–68), whereas those of $^{110m}$Ag and $^{124}$Sb were higher from Loviisa (Figs. 69–70). In the most recent decades, both nuclear power plants have invested and succeeded in reducing radioactive discharges, and this was visible in the environmental measurements as a whole in the 1990s and 2000s. Discharges of $^{60}$Co, $^{54}$Mn $^{58}$Co and $^{124}$Sb have significantly decreased, and $^{110m}$Ag was not registered in the discharges from Olkiluoto since 1990. The discharges of $^{137}$Cs and $^{134}$Cs have decreased in the same way (Figs. 71–72), but the differentiating locally-discharged caesium from the dominating bulk amounts of Chernobyl-originated caesium has in general been difficult right up to the present time. To be sure, $^{134}$Cs has practically disappeared from environmental samples at the same time as the Chernobyl-originated $^{134}$Cs has decayed according to its short half-life (2 years).

$^{60}$Co has become the main object in monitoring the appearance and distribution of radionuclides discharged from the local nuclear power plants in the environment, because it is effectively transferred not only to most of the indicator organisms but also to the suspended particulate matter and the bottom sediments. Furthermore, in recent years it has often remained as the only local discharge nuclide detected in various environmental samples when the concentrations of the others have fallen below the detection limit, although its discharges have decreased as well. Up to 2007, the total amount of $^{60}$Co discharged into the sea from the Olkiluoto NPP was $1.2 \cdot 10^{11}$ Bq, and the cumulative amount, which takes into account the decay of $^{60}$Co (half-life 5.3 a), was $1.8 \cdot 10^{10}$ Bq. The corresponding amounts discharged from Loviisa were $3.1 \cdot 10^{10}$ Bq and $3.1 \cdot 10^{9}$ Bq. The discharges from Olkiluoto have markedly decreased since 1997. The discharges from Loviisa varied considerably after 1993, but were in general significantly lower than those from Olkiluoto (Fig. 68).

The decrease in the $^{60}$Co, $^{54}$Mn and $^{58}$Co discharges was clearly seen in the concentrations of these nuclides in the *Fucus* and sinking matter samples taken up to 2007 from the sampling sites situated closest to the cooling water outlets in both areas (Figs. 86, 87, 128, and 129). In the *Fucus* samples taken

for the regulatory monitoring programme at Loviisa, the concentrations of these nuclides have decreased continuously, and especially during the last ten years their concentrations have sunk to very low levels, although small amounts of them were still detected quite frequently at the sampling sites of Hästholmsfjärden, and in ever-diminishing measure at the site of Lilla Djupberget, where $^{60}$Co was last detected in 1999. At the farthest, $^{60}$Co was found in the samples of the regulatory monitoring programme at Boistö (at a distance of 10 km) in the mid-1990s. In the *Fucus* samples taken from Olkiluoto for the regulatory monitoring programme, the concentrations of $^{60}$Co and $^{54}$Mn reached their maxima in 1990, and after an interim decline in 1993 they increased again in 1995 and 1996; since then they have decreased significantly at all the sampling sites. In general, the concentrations of $^{60}$Co, $^{54}$Mn and $^{58}$Co were clearly higher at the sampling sites nearest to the cooling water outlet, but they were quite regularly detected at all the permanent sampling sites, including Iso Pietari and Vähäkrunni at distances of 12 and 7 km from the outlet.

$^{60}$Co, $^{110m}$Ag, $^{58}$Co, $^{54}$Mn and $^{125}$Sb were rather regularly found in sinking matter samples collected at Loviisa from Hästholmsfjärden and Klobbfjärden. In addition, smaller quantities of $^{60}$Co were frequently detected in the samples collected from Station 4a of Vådholmsfjärden lying seaward of the straits leading to the outer archipelago from the Hästholmsfjärden Bay (at a distance of about 4 km from the cooling water outlet). Minor amounts of $^{60}$Co, $^{54}$Mn and $^{125}$Sb were also detected from time to time in sinking matter collected from Reference Station R1 located in Påsalöfjärden (at a distance of 14 km from the power plant). In general, the activity concentrations were clearly lower after the Chernobyl accident than before that, and the concentrations decreased, especially during the 1990s. At Olkiluoto, $^{60}$Co, $^{54}$Mn, $^{58}$Co and $^{125}$Sb were mainly detected in sinking matter collected from the close vicinity of the power plant, on the west and north side of Olkiluoto Island. In addition, $^{60}$Co was quite regularly, and $^{54}$Mn from time to time, detected in samples collected from Station 15 situated 10 km north of the power plant. Since the mid-1990s, the concentrations of the local discharge nuclides decreased strongly and remained at a very low level in the 2000s.

The major components in ordinary sediment samples were the naturally-occurring radionuclides and long-lived fallout nuclides, and the appearance of local discharge nuclides was much smaller compared to that in sinking matter. Of the last-mentioned, only $^{60}$Co was regularly detected in both areas; $^{54}$Mn occurred in sediment samples at Olkiluoto until 1995, and observations of $^{125}$Sb decreased in both areas since the 1990s. At Loviisa, the highest activity concentrations of $^{60}$Co occurred in the surface sediment layer of Hästholmsfjärden in 1986–1994. Since then, the concentrations have decreased, and the peak concentrations have

been buried into deeper layers as fresh sediment has settled on the surface. In parallel with the decreasing concentrations, the distribution of $^{60}$Co has shrunk from 13.5 km (Station 7) in 1990 and 1998 to 9.5 km (Station 4a) in 2006, as a consequence of the decreased discharges. Transfer of the peak concentrations to deeper sediment layers was also noticed in nearby areas at Olkiluoto. The highest $^{60}$Co concentrations found at Olkiluoto were only slightly higher than those at Loviisa, but the most notable difference was the clearly wider distribution of the $^{60}$Co records in sediments at Olkiluoto. Minor amounts of $^{60}$Co were found in sediments at distances of 16 km to the south and 17 km to the north of the Olkiluoto power plant.

Even more eye-catching was the difference in the distribution range of $^{60}$Co observations in *Fucus vesiculosus* on the west versus the south coast of Finland. At the greatest distance from Olkiluoto, minimal quantities of $^{60}$Co (0.2 Bq kg$^{-1}$ d.w.) were detected in 1995 at Närpiö in the north (distance 137 km) and at Isokari in the south (distance 61 km) on the west coast. On the south coast, $^{60}$Co was not detected in *Fucus* outside the Loviisa area at the coastal sampling sites of the Gulf of Finland. The spreading of $^{60}$Co in detectable quantities just on the west coast of Finland was at least partly due to the higher discharges of the nuclides from Olkiluoto than from Loviisa, but maybe more significantly due to the free exchange of water and better diffusion capability of discharges from Olkiluoto. The widespread observations correlate with the discharge trends of $^{60}$Co from Olkiluoto. Although the 2003 *Fucus* survey was a little briefer, $^{60}$Co was no longer detected outside the immediate Olkiluoto area in this survey. The minimal concentrations found in *Fucus* at relatively long distances from the power plant are mainly interesting curiosities that highlight the sensitivity of the analysis methods, the usability of *Fucus* as an indicator organism, and the potential of small amounts of radionuclides in tracer studies. Their relevance regarding environmental risk is insignificant.

As a conclusion, the environmental effects of the thermal discharges were clearly more pronounced in the sea area off Loviisa, whereas the effects of the radioactive discharges were more pronounced in the sea area off Olkiluoto. The reasons for the more notable biological effects of the warm water discharges in Loviisa were the much higher nutrient level in the Gulf of Finland and the topographic characteristics resulting in a limited exchange of water in the area. Although similar trends were also found in the biological elements at Olkiluoto (e.g., a rise in the level of primary production and changes in the zoobenthos), their significance remained smaller owing to the lower level of the eutrophication trend in the Bothnian Sea. The clearly more significant outcome of the effects of the radioactive discharges at Olkiluoto was obviously due to the higher discharges of certain radionuclides ($^{60}$Co, $^{54}$Mn, $^{58}$Co) from the site in the past. Other factors

increasing the environmental detectability of the radioactive discharges were probably due to the features associated with the diffusion capability of aquatic discharges from Olkiluoto, including the jet discharge mode, which increases the currents, and the free exchange of water, which contributes to the involvement of the discharges in the currents and mixing processes of the surrounding sea area.

# 7 The significance of environmental radiation and thermal pollution

When examining the effects of radioactive discharges from the nuclear power plants into the environment, the main attention should be paid to the radiation effects of the discharge nuclides, i.e., the radiation doses to people and organisms living in the vicinities of the power plants, not to the findings in the environment.

Radiation doses to the public caused by discharges from the Finnish nuclear power plants are estimated at STUK using the VALTO computer model (Blomqvist and Kara 1982). The calculations are based on the discharge and meteorological data reported to STUK by the power companies. Individual doses to the critical group since the beginning of the power production at Loviisa and Olkiluoto are presented in Fig. 142. The critical group is a hypothetical group of people living in the vicinity of the power plants and eating maximum amounts of local foodstuffs such as, e.g., fish and other marine products. The dose limit set by the Government for members of the public from the normal operation of Finnish nuclear power plants is 0.1 milliSv $a^{-1}$. This is approximately 1/40 of the average radiation dose received by Finns from different sources (half of this from indoor radon) during a year. During the whole operational history of the power plants, the effective dose commitments of the critical groups have been, at their highest, less than 4%, and during recent years clearly below 1% of the limit set by the Government. In general, a major part of these minor doses has been due to the liquid discharges of $^{60}Co$ and peoples' shore occupancy.

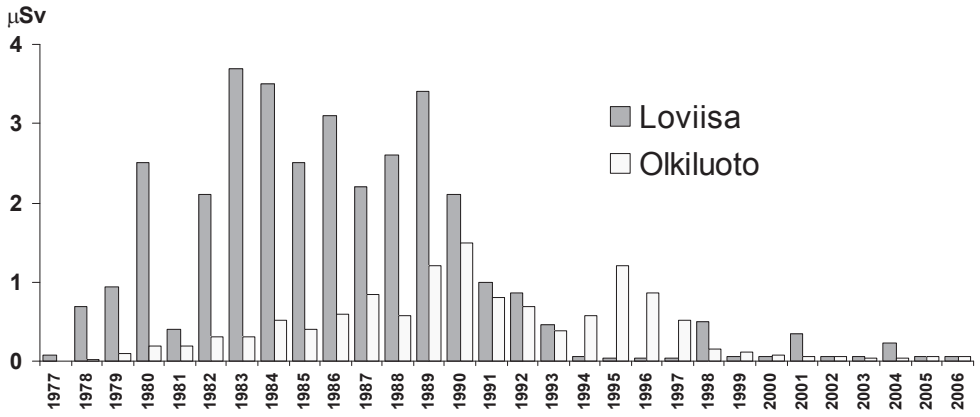


**Fig. 142.** Average doses in micro Sv $a^{-1}$ to the critical groups of people in the vicinities of Finnish NPPs since the beginning of power production.

In 2004, the dose estimate obtained with the VALTO model for aquatic discharges from the Loviisa power plant was 0.19 µSv, while the estimates obtained from the results of the shore-soil survey carried out in the same year were 0.14 – 0.23 µSv. In 2005, the dose estimate obtained with the VALTO model for aquatic discharges from Olkiluoto was 0.031 µSv, whereas the shore soil survey resulted in 0.83 – 0.95 µSv (see p. 262). The dose estimates from the soil surveys are based on the maximum values observed, and are therefore the most conservative ones. In any case, the doses are very low and practically insignificant. In contrast to these external doses, it has been difficult to estimate internal doses to the members of the critical group, because of the very rarely met with, and the very low quantities of local discharge nuclides in anything ingested by humans (almost exclusively fish). In the few cases in which local discharge nuclides have been detected in fish samples (maximum 0.44 Bq $kg^{-1}$ fresh wt.), the doses that would be caused by them have been insignificant.

The exposure of the population to radiation caused by ingesting Baltic Sea fish and other marine products, and from living by the sea after the Chernobyl accident, was evaluated in the EC-funded Marina Balt Project in 1996 – 1998 (Nielsen 2000). The dose for the critical groups living on the coasts of the Bothnian Sea and the Gulf of Finland peaked in 1986 at a value of 0.2 mSv. For the sake of comparison, on average a Finn receives an annual dose of 3.7 mSv from other sources, and it should be borne in mind that, after 1986, the effect of the Chernobyl fallout decreased significantly and very soon after the accident. At the same time, individuals in the critical groups received a yearly dose of 0.7 mSv from $^{210}$Po occurring naturally in the sea.

In the past decade, the radioecological research has been focused worldwide on topics relating to the protection of the environment (or biota) from the effects of ionizing radiation. Within the European Community, the issue of the exposure of wildlife to ionising radiation was first considered in the FASSET Project (Framework for Assessment of Environmental Impact) in 2000 – 2004, which developed an assessment framework incorporating the transfer of radionuclides to biota, the estimation of doses and the collation of data on effects (Larsson 2004). It was followed by the EC-funded ERICA Project (Environmental Risks from Ionising Contaminants: Assessment and Management) in 2004 – 2007, that concluded with the publication of two main outputs: 'the ERICA Integrated Approach to the assessment and management of environmental risks from ionising radiation', and 'the ERICA Tool', which is a software programme with its supporting databases for assessing the radiological risk to biota (Larsson 2008, Brown et al. 2008). The ERICA Project was further followed by the PROTECT co-ordinated action (Protection of the Environment from Ionising Radiation in

a Regulatory Context) in 2006–2008, the objective of which was to evaluate the practicability and relative merits of different approaches to protection of the environment from ionising radiation (Andersson et al. 2008).

The ERICA Tool was applied to the assessment of environmental risks from ionising radiation caused by aquatic discharges from the Loviisa and Olkiluoto power plants during normal operation. The examination of environmental risk from ionising contaminants was principally focused on the maximum concentrations of local discharge nuclides detected at Loviisa and Olkiluoto in the environmental samples.

Tier 1 is the most conservative stage of the ERICA Tool and is universal for all reference organisms. It requires minimal data input, and maximal measured media concentrations are suggested as such (Brown et al. 2008). It does not give dose rates to biota, but it can release one from further consideration (more exact assessment is not needed) if the contamination is proved to be so low that the result does not reach the generic screening value 10 µGy $h^{-1}$ agreed by the ERICA and PROTECT Projects (Larsson 2008, Andersson et al. 2008).

The maximum concentration of tritium detected in the environs of the Finnish nuclear power plants was 120 kBq $m^{-3}$ in seawater from Loviisa (p. 177). For this maximum concentration in seawater, the output of the estimation was that, for all reference organisms, the dose rates are below the 10 µGy $h^{-1}$ screening level. The limiting reference organism was phytoplankton (Tier 1). The total dose rate for phytoplankton was $4.12 \cdot 10^{-4}$ µGy $h^{-1}$ (Tier 2) assuming that the tritium concentration is the same in phytoplankton (120 Bq $kg^{-1}$ f.w.) as in seawater. For macroalgae (*Fucus vesiculosus*) the estimated total dose rate caused by this maximum concentration of tritium was $9.90 \cdot 10^{-4}$ µGy $h^{-1}$.

Tritium is clearly the most abundant radionuclide in both the liquid and airborne discharges of the power plants, and it is also that most abundantly found in the environmental (seawater) samples. The doses to organisms from tritium are almost entirely from internal sources, owing to the very short ranges of the beta particles (UNSCEAR 1996). Traditionally, the risks from tritium are considered to be small, because it is a low-energy beta emitter that behaves chemically like water and thus has a relatively short biological half-life (12 days); in general, no evidence has been found of the biomagnification of tritium. According to Weaver et al. (1969), tritium is one of the least hazardous radionuclides produced in nuclear reactors. On the other hand, it is clear from the wealth of tritium data now available that the relative biological effectiveness (RBE) values for tritium beta rays are higher than those used earlier in radiation protection (e.g. Straume and Carsten 1993).

In addition, the dose rate for the brown alga *Fucus vesiculosus* was assessed on the basis of a sample taken on 14$^{th}$ of August 1990 in front of the cooling water

outlet at Olkiluoto, representing the maximal case of local discharge nuclides detected in biota. The sample contained:

| Nuclide | Conc. in *Fucus* Bq kg$^{-1}$ d.w. | Conc. in *Fucus* Bq kg$^{-1}$ f.w. | Conc. in water Bq l$^{-1}$ |
|---|---|---|---|
| $^{54}$Mn | 150 | 26 | $<2 \cdot 10^{-3}$ |
| $^{58}$Co | 20 | 3.4 | $<3 \cdot 10^{-3}$ |
| $^{60}$Co | 170 | 29 | $1.7 \cdot 10^{-3}$ |
| $^{65}$Zn | 9.3 | 1.6 | $<3 \cdot 10^{-3}$ |
| $^{90}$Sr | 15 | 2.6 | $2.0 \cdot 10^{-2}$ |
| $^{134}$Cs | 92 | 3.1 | $4.3 \cdot 10^{-2}$ |
| $^{137}$Cs | 120 | 21 | $2.3 \cdot 10^{-1}$ |

The concentration ratios (CR) for marine macroalgae (Hosseini et al. 2008) were used in the assessment. The limiting reference organism was the polychaete worm (Tier 1), but again for all reference organisms the dose rates were below the 10 µGy h$^{-1}$ screening level. The total dose rate per organism obtained for *Fucus* was $6.74 \cdot 10^{-3}$ µGy h$^{-1}$ (Tier 2), i.e., less than 1/1 000 of the screening level, and the maximum proportion ($2.81 \cdot 10^{-3}$ µGy h$^{-1}$) was caused by the Chernobyl-originated $^{137}$Cs.

Furthermore, the dose rate for the relict crustacean *Saduria entomon* was assessed on the basis of a sample taken on 21$^{st}$ of May 1987 from Hästholmsfjärden in Loviisa, exemplifying the maximal case of local discharge nuclides in bottom-dwelling organisms. The sample contained:

| Nuclide | Conc. in *Saduria* Bq kg$^{-1}$ d.w. | Conc. in *Saduria* Bq kg$^{-1}$ f.w. | Conc. in water Bq l$^{-1}$ | Conc. in sedim. Bq kg$^{-1}$ d.w. |
|---|---|---|---|---|
| $^{60}$Co | 16 | 3.6 | – | 46 |
| $^{90}$Sr | 26 | 5.8 | – | 24 |
| $^{110m}$Ag | 110 | 24 | – | 37 |
| $^{134}$Cs | 92 | 20 | $1.6 \cdot 10^{-1}$ | 1 300 |
| $^{137}$Cs | 230 | 51 | $3.7 \cdot 10^{-1}$ | 4 100 |

The limiting reference organism was again the polychaete worm (Tier 1); for all reference organisms the dose rates were below the 10 µGy h$^{-1}$ screening level. The total dose rate per organism obtained for *Saduria* was 1.19 µGy h$^{-1}$ (Tier 2), and the maximum proportion of both the external and internal dose

rates was caused by the Chernobyl-originated $^{137}$Cs (external $5.95 \cdot 10^{-1}$, internal $9.18 \cdot 10^{-3}$ µGy h$^{-1}$). The proportion of $^{60}$Co in the external dose rate was $2.99 \cdot 10^{-2}$ and that of $^{110m}$Ag $2.59 \cdot 10^{-2}$ µGy h$^{-1}$.

The dose rate caused by the Chernobyl-originated $^{137}$Cs was also assessed for perch using a concentration of 230 Bq kg$^{-1}$ f.w. for perch (the maximum value at Loviisa in 1986) and the corresponding value of $5.0 \cdot 10^{-1}$ Bq l$^{-1}$ for seawater. Dividing the occupancy factor equally for water and the sediment surface, the total dose rate per organism for benthic fish was $1.89 \cdot 10^{-1}$ µGy h$^{-1}$, mainly caused by the external dose rate ($1.50 \cdot 10^{-1}$ µGy h$^{-1}$).

For the sake of comparison, the dose rates to submerged hydrophyte roots (e.g. *Myriophyllum, Ceratophyllum, Potamogeton* spp.) and above sediment parts due to the major radionuclides discharged into the cooling water pond of the Ignalina NPP (Lithuania) were $4.4 \cdot 10^{-2}$ and $4.0 \cdot 10^{-3}$ µGy h$^{-1}$, when the exposure of biota was assessed based on data from 1988–1999 (Nedveckaite et al. 2007). The internal exposure dose rate due to the natural background α-emitters $^{210}$Po, $^{238}$U and $^{226}$Ra was estimated to be 1.24 µGy h$^{-1}$.

In conclusion, the environmental risk caused by the ionizing contaminants discharged from the Loviisa and Olkiluoto power plants is far below the conservative screening level 10 µGy h$^{-1}$. Although the concentrations in the environmental samples, and above all, in the discharge data appear as seemingly large values, the radiation doses caused by them to humans and to biota are very low, practically insignificant.

The harmful effects of the thermal discharges, as such, would probably also be small without the underlying general eutrophication trend in the recipients, and the consequent combined effect of elevated nutrient concentrations in the water and the rise of temperature. From the biological point of view, the most important environmental effect of the cooling water is certainly the prolonging of the growing season as a result of reduced ice conditions in winter. This again, together with increased temperatures during the growing season, has led to accelerated and increased biological production and an increased amount of organic material to be decomposed, which in turn has led to changes in the bottom deposits and in the macrozoobenthos. On the species level, the increased temperatures have led to changes in biological communities, in which thermophilic species have benefited from the heat and expanded their vital space, while thermophobic or eutrophy-avoiding species have disappeared.

# 8 Finnish coastal waters as a recipient for NPP discharges

The boreal location is an undisputed advantage when considering the applicability of Finnish coastal waters as a source for and recipient of cooling water from nuclear power plants. The temperate climate, and consequent low seawater temperatures provide a good basis for using them as coolant. Nevertheless, the shallow coastal waters in Finland may also reach unwelcomely high temperatures in summer, and the temperature of the discharge water may rise close to the permitted maximum values, considering the tolerance of organisms in the immediate vicinity of the cooling water outlets. In the Finnish nuclear power plants, the problems caused by the maximum temperatures in summer are reduced by a deep-water intake or the rapid jet discharge of cooling water. However, the first-mentioned may hide a risk, if the deep water in the intake area is rich in nutrients. The combined effect of heat and nutrients can then lead to an undesirable eutrophication process in the discharge area. On the other hand, the formation of ice on the cooling water intake in winter may cause difficult problems for the nuclear power plants.

The Baltic Sea is neither an ocean nor a lake, but a large brackish-water basin (Voipio 1981). Brackish water is a mixture of seawater and fresh water, and river estuaries are world-wide typical brackish water habitats. The Baltic Sea, as a whole, is the world's largest pool of brackish water. The salinity decreases from around 20‰ in the surface water of the Kattegat to 1–2‰, or even less, in the bottoms of the Gulf of Bothnia and the Gulf of Finland, compared to 35‰ in the open oceans. On the Finnish coast the salinity of the surface water varies between 7‰ and almost fresh water. Consequently, only relatively few animal and plant species live in the brackish ecosystems of the Baltic Sea, compared to other aquatic ecosystems – although this limited biodiversity does include a unique mix of marine and freshwater species adapted to the brackish conditions, as well as a few true brackish water species (HELCOM 2008).

The low salinity of the water may incorporate a slight disadvantage regarding the Finnish coastal areas as a recipient for thermal discharges compared to sea areas with a higher salinity. In the extreme case, some species may live at the limit of their distribution range, and besides that, the low diversity of biota may sensitize the organisms to any additional environmental stress, such as thermal pollution. In addition, low salinity is known to promote the uptake of certain radionuclides by organisms. Relatively little is known about the impact of salinity on the uptake of radionuclides, but at least the uptake of caesium, strontium and iodine have been shown to increase by decreasing salinity

(e.g. Eisenbud and Gesell 1997, Carlson and Erlandsson 1991, Nakahara 1999). More results are available about the augmenting effect of increasing temperature on the uptake of radionuclides.

The effectiveness of the water exchange in the discharge area is one of the most important characteristics for all kinds of industrial discharges, including the thermal and radioactive discharges from nuclear power plants. Effective exchange of water guarantees a rapid mixing of the discharges into large water masses, and thus weakens the local point source effects in the nearness of the outlets, diluting the effluents and lowering their concentrations in the aquatic environment. On the other hand, a fast jet discharge and effective exchange of water are shown to bring about a more wide-ranging spread of slightly elevated temperatures and slower total removal of the heat from the recipient (Ehlin 1974, Langford 1990), as well as a more wide-spread distribution of low-level concentrations of discharged radionuclides in environmental samples (this study).

An ideal alternative for NPP discharges would be an open offshore site exposed to the currents and mixing processes of the surrounding open sea area, with an adequately deep and evenly seaward-sloping seabed without any sheltering island, skerries or rocks. As the thermal effects are found to be most pronounced in the littoral vegetation, the disadvantage can be lessened by a long-distance discharge, i.e., by an outfall tunnel leading the discharge into the sea well off the coast. This procedure is commonly used in many cities for sewage discharges, and analogous thinking has been applied to the cooling water discharge at the Forsmark NPP in Sweden. Long-distance discharge by an outfall tunnel may also enable the choice of a site lying at a greater distance from the shore line.

This study has shown that the nutrient level and the exchange of water in the discharge area are of crucial importance regarding the environmental effects of thermal discharges in the receiving sea area. Thus, if the goal is to minimize the environmental effects of the cooling water, archipelago areas with high contents of nutrients and a limited water exchange are not ideal recipients for heated discharges. The thermal effects as such may possibly be restricted, if the level of eutrophy is low, but the combined effect of elevated temperature and nutrients may significantly accelerate the progress of eutrophication, and may lead to consequences exceeding the tolerance of the ecosystem to retain its customary capacity for vital activity, and to maintain the healthy structure of the biological communities. On the other hand, when choosing sites for potential supplementary nuclear power plants, one standpoint would be to concentrate all the discharges to previously loaded areas and thus save virgin areas. A soft alternative for the marine environment would certainly be the utilization of the huge amount of waste heat on land, e.g., for district heating.

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# APPENDIX 1

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# APPENDIX 2

## Thermal discharges from Loviisa NPP

**Table 1.** Annual data on discharges of warm cooling water into the sea from the Loviisa nuclear power plant (Annual reports of Imatran Voima Oy / Fortum Power and Heat Oy).

| Year | Average flow rate of cooling water $m^3s^{-1}$ | Amount of heat discharged into sea $10^3$ TJ $a^{-1}$ | Max. temperature of discharged water (hour mean) °C |
|---|---|---|---|
| 1977 | 19.6 | 19.3 | 27.7 |
| 1978 | 22 | 22.5 | 25.6 |
| 1979 | 22 | 22.0 | 27.3 |
| 1980 | 10 | 13.0 | 16.5 |
| 1981 | 37.0 | 43.8 | 29.7 |
| 1982 | 39.0 | 47.0 | 29.0 |
| 1983 | 40.0 | 51.0 | 29.6 |
| 1984 | 38.5 | 51.2 | 30.7 |
| 1985 | 42.5 | 53.0 | 26.3 |
| 1986 | 40.2 | 50.0 | 27.9 |
| 1987 | 41.7 | 54.4 | 26.0 |
| 1988 | 40.0 | 51.6 | 31.9 |
| 1989 | 41.3 | 52.4 | 27.8 |
| 1990 | 41.2 | 48.7 | 27.3 |
| 1991 | 42.8 | 51.1 | 30.4 |
| 1992 | 39.2 | 49.4 | 27.0 |
| 1993 | 41.6 | 51.9 | 28.0 |
| 1994 | 38.1 | 49.2 | 29.7 |
| 1995 | 40.4 | 47.5 | 26.6 |
| 1996 | 39.1 | 50.0 | 28.6 |
| 1997 | 43.0 | 54.9 | 32.1 |
| 1998 | 44.9 | 55.4 | 28.4 |
| 1999 | 44.3 | 58.5 | 31.1 |
| 2000 | 43.4 | 55.4 | 29.0 |
| 2001 | 46.0 | 57.2 | 32.0 |
| 2002 | 42.5 | 54.1 | 30.5 |
| 2003 | 43.7 | 56.0 | 32.1 |
| 2004 | 44.2 | 55.8 | 29.8 |
| 2005 | 46.1 | 58.7 | 29.3 |
| 2006 | 43.3 | 55.6 | 29.7 |

# APPENDIX 3

## Nutrient load from main sources into the study area off Loviisa

**Table 1.** Annual load of total phosphorus (kg $a^{-1}$) from different sources into the sea area surrounding the Loviisa NPP in 1970–2006.

| Year | River Kymijoki[1)] | River Tesjoki[1)] | Town of Loviisa[1)] | Semilax[1)] | Loviisan Smoltti[1)] | Loviisa NPP[2)] |
|---|---|---|---|---|---|---|
| 1970 | – | 15 768 | – | – | – | – |
| 1971 | – | 12 921 | 4 745 | – | – | – |
| 1972 | – | 14 819 | 4 015 | – | – | – |
| 1973 | – | 10 074 | 6 205 | – | – | – |
| 1974 | – | 21 754 | 6 935 | – | – | – |
| 1975 | 137 000 | 10 403 | 803 | – | – | – |
| 1976 | 88 000 | 5 037 | 840 | – | – | 293 |
| 1977 | 187 000 | 17 338 | 1 862 | – | – | 208 |
| 1978 | 132 000 | 10 403 | 4 015 | – | – | 177 |
| 1979 | 172 000 | 14 819 | 2 300 | – | – | 88 |
| 1980 | 148 000 | 13 870 | 1 971 | – | – | 124 |
| 1981 | – | 24 273 | 1 643 | – | – | 55 |
| 1982 | 149 000 | 16 389 | 1 095 | – | – | 35 |
| 1983 | 155 000 | 13 870 | 1 679 | – | – | 22 |
| 1984 | 192 000 | 25 550 | 511 | – | – | 54 |
| 1985 | 185 000 | 16 389 | 548 | – | – | 50 |
| 1986 | 176 000 | 15 768 | 584 | – | – | 17 |
| 1987 | 186 000 | 17 338 | 730 | – | – | 13 |
| 1988 | 234 000 | 18 615 | 438 | – | 215 | 20 |
| 1989 | 161 000 | 15 768 | 803 | – | 274 | 23 |
| 1990 | 168 000 | 16 389 | 550 | 38 | 534 | 77 |
| 1991 | 139 000 | 16 300 | 511 | 40 | 384 | 87 |
| 1992 | 144 000 | 19 500 | 700 | 45 | 524 | 143 |
| 1993 | 114 000 | 12 420 | 273 | 64 | 492 | 120 |
| 1994 | 139 000 | 17 070 | 400 | 58 | 467 | 160 |
| 1995 | 137 000 | 18 060 | 443 | 235 | 216 | 211 |
| 1996 | 120 000 | 20 340 | 319 | 296 | 209 | 187 |
| 1997 | 97 000 | 13 730 | 354 | 367 | 136 | 131 |
| 1998 | 134 000 | 21 820 | 365 | 526 | 142 | 80 |

[1)] Provided by Helsinki Water and Environment District (1970–1990) and the Finnish Environment Institute (1991–2006).
[2)] Data collected from the Annual monitoring reports of cooling water and effluents of the power plant by summing the values of communal wastewaters, those of process waters since 1990, and those of wastewaters from the construction work of the depository for power plant waste in 1993–1996.

**Table 1.** Continued.

| Year | River Kymijoki[1)] | River Tesjoki[1)] | Town of Loviisa[1)] | Semilax[1)] | Loviisan Smoltti[1)] | Loviisa NPP[2)] |
|---|---|---|---|---|---|---|
| 1999 | 108 000 | 15 410 | 327 | 385 | 130 | 12 |
| 2000 | 117 000 | 24 440 | 197 | 347 | 128 | 19 |
| 2001 | 102 000 | 15 310 | 602 | 472 | 145 | 50 |
| 2002 | 77 000 | 12 130 | 200 | 342 | 91 | 16 |
| 2003 | 71 000 | 12 560 | 223 | 347 | 191 | 10 |
| 2004 | 138 000 | 28 530 | 183 | 305 | 205 | 57 |
| 2005 | 125 000 | 17 710 | 269 | 267 | 230 | 12 |
| 2006 | 91 000 | 21 690 | 309 | 403 | 166 | 25 |

[1)] Provided by Helsinki Water and Environment District (1970–1990) and the Finnish Environment Institute (1991–2006).
[2)] Data collected from the Annual monitoring reports of cooling water and effluents of the power plant by summing the values of communal wastewaters, those of process waters since 1990, and those of wastewaters from the construction work of the depository for power plant waste in 1993–1996.

**Table 2.** Annual load of total nitrogen (kg $a^{-1}$) from different sources into the sea area surrounding the Loviisa NPP in 1970–2006.

| Year | River Kymijoki[1)] | River Tesjoki[1)] | Town of Loviisa[1)] | Semilax[1)] | Loviisan Smoltti[1)] | Loviisa NPP[2)] |
|---|---|---|---|---|---|---|
| 1970 | – | 252 215 | – | – | – | – |
| 1971 | – | 206 955 | 19 710 | – | – | – |
| 1972 | – | 237 250 | 20 075 | – | – | – |
| 1973 | – | 161 330 | 25 550 | – | – | – |
| 1974 | – | 348 210 | 27 740 | – | – | – |
| 1975 | 3 222 000 | 166 440 | 22 995 | – | – | – |
| 1976 | 1 698 000 | 80 665 | 26 280 | – | – | 9 603 |
| 1977 | 3 380 000 | 277 400 | 33 215 | – | – | 7 910 |
| 1978 | 2 288 000 | 166 440 | 37 960 | – | – | 6 100 |
| 1979 | 2 565 000 | 237 250 | 58 400 | – | – | 4 200 |
| 1980 | 2 679 000 | 221 920 | 35 405 | – | – | 2 900 |
| 1981 | – | 388 360 | 34 310 | – | – | 1 070 |
| 1982 | 3 456 000 | 262 435 | 52 195 | – | – | 1 880 |
| 1983 | 3 166 000 | 221 920 | 43 800 | – | – | 980 |
| 1984 | 4 205 000 | 408 800 | 25 915 | – | – | 1 170 |
| 1985 | 3 422 000 | 262 435 | 31 025 | – | – | 980 |
| 1986 | 3 439 000 | 252 215 | 24 820 | – | – | 760 |

[1)] Provided by Helsinki Water and Environment District (1970–1990) and the Finnish Environment Institute (1991–2006).
[2)] Data collected from the Annual monitoring reports of cooling water and effluents of the power plant by summing the values of communal wastewaters, those of process waters since 1990, and those of wastewaters from the construction work of the depository for power plant waste in 1993–1996.

**Table 2.** Continued.

| Year | River Kymijoki[1)] | River Tesjoki[1)] | Town of Loviisa[1)] | Semilax[1)] | Loviisan Smoltti[1)] | Loviisa NPP[2)] |
|---|---|---|---|---|---|---|
| 1987 | 3 914 000 | 277 400 | 23 725 | – | – | 550 |
| 1988 | 4 809 000 | 297 840 | 24 820 | – | 1 205 | 1 000 |
| 1989 | 3 531 000 | 252 215 | 28 470 | – | 3 066 | 1 124 |
| 1990 | 3 956 000 | 262 435 | 23 694 | 238 | 3 406 | 2 252 |
| 1991 | 3 456 000 | 242 360 | 29 930 | 250 | 2 719 | 2 730 |
| 1992 | 4 011 000 | 300 000 | 28 784 | 295 | 3 689 | 5 021 |
| 1993 | 3 236 000 | 200 000 | 22 306 | 458 | 3 471 | 5 037 |
| 1994 | 3 159 000 | 320 000 | 26 447 | 483 | 2 921 | 1 719 |
| 1995 | 3 362 000 | 310 000 | 25 788 | 1 850 | 2 245 | 1 731 |
| 1996 | 2 846 000 | 300 000 | 22 139 | 2 576 | 2 156 | 1 690 |
| 1997 | 2 498 000 | 210 000 | 26 369 | 2 802 | 1 990 | 1 396 |
| 1998 | 3 277 000 | 350 000 | 23 747 | 3 098 | 1 423 | 3 742 |
| 1999 | 2 701 000 | 310 000 | 24 686 | 2 324 | 1 424 | 837 |
| 2000 | 3 470 000 | 320 000 | 17 313 | 2 342 | 910 | 916 |
| 2001 | 3 348 000 | 250 000 | 24 388 | 3 268 | 1 244 | 3 021 |
| 2002 | 2 716 000 | 200 000 | 20 019 | 2 299 | 1 073 | 1 108 |
| 2003 | 2 310 000 | 180 000 | 18 091 | 2 713 | 1 339 | 1 130 |
| 2004 | 4 269 000 | 460 000 | 19 561 | 2 125 | 1 767 | 3 399 |
| 2005 | 3 748 000 | 320 000 | 19 791 | 2 091 | 2 111 | 1 006 |
| 2006 | 3 141 000 | 340 000 | 26 421 | 2 982 | 1 865 | 1 510 |

[1)] Provided by Helsinki Water and Environment District (1970–1990) and the Finnish Environment Institute (1991–2006).
[2)] Data collected from the Annual monitoring reports of cooling water and effluents of the power plant by summing the values of communal wastewaters, those of process waters since 1990, and those of wastewaters from the construction work of the depository for power plant waste in 1993–1996.

**Table 3.** Annual mean load of biological ($BOD_7$) and chemical oxygen demand (COD) and solid matter caused by the communal waste waters of the Loviisa power plant during four periods in 1976–2006 (kg $a^{-1}$).

| Years | $BOD_7$ | COD | Solid matter |
|---|---|---|---|
| 1976–1980 | 6 956 ± 6 488 | 5 594 ± 3 430 | 4 649 ± 2 654 |
| 1981–1990 | 1 537 ± 518 | 805 ± 327 | 964 ± 262 |
| 1991–2000 | 673 ± 528 | 528 ± 187 | 768 ± 381 |
| 2001–2006 | 196 ± 103 | 458 ± 178 | 699 ± 388 |

# APPENDIX 4

## Waste water load from Olkiluoto NPP

**Table 1.** Annual mean load of biological oxygen demand ($BOD_7$), total phosphorus, total nitrogen, ammonium nitrogen and solid matter caused by the communal waste waters of the Olkiluoto power plant during four periods in 1985–2006 (kg $a^{-1}$). SD in parentheses. (Turkki 2007).

| Years | 1985–1989 | 1990–1994 | 1995–1999 | 2000–2006 |
|---|---|---|---|---|
| $BOD_7$ | 2 500 (980) | 2 620 (1 680) | 890 (890) | 350 (95) |
| Total P | 19 (0.99) | 30 (17) | 19 (21) | 13 (7) |
| Total N | 1 020 (320) | 1 300 (410) | 1 230 (400) | 1 760 (680) |
| $NH_4$-N | 1 020 | 1 030 (500) | 950 (300) | 1 430 (600) |
| Solid matter | 730 | | 770 (630) | 410 (140) |

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