

Recent eutrophication of coastal waters in southern Finland
– A palaeolimnological assessment

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Academic dissertation

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Abstract

Eutrophication of estuaries and coastal waters has become a significant problem worldwide. In the Baltic Sea, where the nutrient load has strongly increased from its natural level, this has led to marked changes in the coastal ecosystems. The European Water Framework Directive (WFD) requires all surface waters in Europe to fulfill the criterion of “good ecological status”, where the biological and chemical status departs only slightly from undisturbed baseline conditions by 2027. The WFD requires the determination of these baseline conditions against which the extent of anthropogenic eutrophication and the present ecological status of a system can be assessed.

To provide a means for water quality assessments in coastal waters of the Baltic Sea area, a near-shore calibration data set of 49 embayments was collected. It consisted of measured environmental data and surface sediment diatom assemblages from the southern coast of Finland. The study sites were small, shallow and sheltered with a mean surface area of 211 ha and a mean depth of 4.3 m. Salinity showed a clear overall increasing trend from east to west ranging from 0.7 to 6.4‰. Higher than average nutrient concentrations occurred in estuaries with significant fresh water input, or close to municipalities, emphasising the significance of local sources to nutrient concentrations; no clear impacts of loading from the Neva River estuary were detected. Only ca. 5 % of all sites had good water quality, while most sites (65 %) fell into the class “moderate water quality” based on a national classification scheme by the Finnish Environment Institute.

The collected calibration data set was used to explore the relationship between surface sediment diatom assemblages and 15 environmental variables with special emphasis on nutrients. The main environmental gradients in this data set were represented by nutrients and depth. Total dissolved nitrogen (TDN), total phosphorus (TP), depth and salinity all accounted for significant and independent fractions of variation in the diatom data and together explained 34% of the total variation. There were clear changes in diatom assemblage structures along the nutrient gradients. Although these changes were gradual, it was possible to identify a number of taxa that were more abundant in a particular nutrient environment. Diatom assemblages that were least affected by nutrient enrichment included a variety of benthic species (e.g. *Rhoicosphenia curvata*, *Staurosirella pinnata*, *Opephora mutabilis* and *Bacillaria paxillifer*) and a relatively high species richness. Several small planktonic taxa such as *Cyclotella atomus*, *Cyclotella meneghiniana* and *Thalassiosira pseudonana* were good indicators of highly elevated nutrient concentrations (>600 $\mu\text{g l}^{-1}$ TDN and >60 $\mu\text{g l}^{-1}$ TP) together with low species richness. These taxa could be used as potential indicators of the quality of coastal waters in the Baltic Sea. Their first appearance in regular monitoring could be used as an early warning sign for deteriorating water quality.

In order to define baseline or reference conditions and reconstruct a detailed history of eutrophication in the study area, long sediment cores were collected from five of the study sites with varying anthropogenic influence. Two sites (Töölölahti and Laajalahti) were located in Helsinki, the capital of Finland, with predominantly urban catchment areas, while the remaining three sites

(Fasarbyviken, Pieni Pernajanlahti and Hellänlahti) were mostly impacted by agriculture. The cores were dated using ^{210}Pb , ^{137}Cs and, at two sites, spheroidal carbonaceous particles (SCPs). Diatom species richness, % abundance of planktonic taxa, and detrended correspondence analysis (DCA) primary axis scores, which provide a numerical means to summarise the compositional change of diatom assemblages over time, were used to study changes in the trophic status of these sites in addition to the ecological information contained in changes in the abundance of individual species.

A diatom-based weighted-averaging partial least squares (WA-PLS) transfer function model for TDN was developed with a prediction accuracy of $0.09 \mu\text{g L}^{-1}$ (\log_{10} units). Its performance was assessed using ca. 30 yr records of measured water quality data at four of the sites along with numerical measures such as the modern analogue technique (MAT) at all sites. Overall, the model tracked well the trends in the nutrient record, although it underestimated very high nutrient concentrations during time periods of heavy nutrient loading to the urban embayments. Encouraging, however, is the ability of the model to relatively accurately infer nutrient concentrations, which are in the range of sites included in the used calibration data set. The potential of the multi-proxy approach in a coastal environment was assessed at Laajalahti, where reference conditions and eutrophication history were defined using the stratigraphy of diatoms, sediment geochemistry, stable isotopes, sedimentary pigments, and long-term monitoring results of water quality and nutrient loading.

The urban sites showed marked increases in the abundance of planktonic diatoms and diatom-inferred total dissolved nitrogen (DI-TDN), and a decrease in species richness starting in the 19th – early 20th century with increased urbanisation. At both sites a clear recovery was observed after the cessation of waste water loading by the early 1980s. The presently planktonic diatom assemblages of these embayments, however, show no change back to the pre-disturbance diverse benthic communities. The changes observed in the rural sites were only moderate and occurred later starting in the 1940s. No distinct increase in DI-TDN was seen; however, all sites showed an increase in small planktonic taxa indicating increased nutrient enrichment and turbidity. These small floristic changes could be seen as an early warning signal despite little change in the inferred nutrient concentrations.

At four of the sites, significant changes in diatom assemblages and associated DI-TDN occurred after the 1920s, hence background or reference conditions could be set at early 1900s. The multi-proxy study from the urban site Laajalahti defined the mid to late 1800s as realistic reference conditions. In the case of Töölönlahti, a longer sediment core would have to be taken in order to establish pre-urbanisation conditions at the site. This would mean recreating conditions that existed in the embayment over 200 years ago. At that time, however, the embayment did not exist in its present form due to post glacial land uplift and urban development, but was a more open system affected by the open sea. On the basis of these results realistic reference conditions for management purposes could be set at the late 1800s. The reference conditions were defined by generally diverse benthic diatom assemblages (> 80 % benthic taxa) and DI-TDN concentrations

of ca. 400 $\mu\text{g l}^{-1}$. The general productivity as indicated by sedimentary organic phosphorus (OP), chlorophyll *a* + its degradation products (Chl*a*D) and diatoxanthin was low and bottom water oxygen conditions were good (Cu:Zn ratio).

The present study shows that in urban estuaries major changes in the nutrient status and species assemblages have taken place and that we are still far from a good ecological state of these systems. On the other hand, the results indicate that in rural embayments the effects of eutrophication on the biota have perhaps been less pronounced than expected. Although these sites have crossed an ecologically important threshold, the changes are still only small to moderate. At such sites management efforts may be well rewarded: internal loading from sediments would likely be a minor problem, as external loading has not been heavy (cf. the urban sites) and hence no large nutrient pool has accumulated in the sediments.

The results of this thesis have clear implications for the European Water Framework Directive. Diatoms could be applied to water quality classification and monitoring purposes in the coastal waters of the Baltic Sea area using techniques such as weighted-averaging regression and calibration. Analysing dated sediment stratigraphies using diatoms as key indicators or using several biological and geochemical proxies can help to assess ecological changes in coastal waters over longer time scales (ca. 100-200 yr) and provide information on background conditions. These techniques are also a valuable means for detecting early community changes, where water quality problems are not yet evident. Thus, they provide a management tool not only for ecosystem restoration but also for ecosystem protection.

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List of original publications

This summary aims to synthesise the key findings of the original publications, which are referred to by their Roman numerals in the text.

I Weckström, K., A. Korhola & P. Shemeikka, 2002. Physical and chemical characteristics of shallow embayments on the southern coast of Finland. *Hydrobiologia* 477: 115-127.

II Weckström, K. & S. Juggins. Coastal diatom-environment relationships from the Gulf of Finland, Baltic Sea. *Journal of Phycology*. Accepted.

III Weckström, K., S. Juggins & A. Korhola, 2004. Quantifying background nutrient concentrations in coastal waters: A case study from an urban embayment of the Baltic Sea. *Ambio* 33(6): 324-327.

IV Weckström, K. Assessing recent eutrophication in coastal waters of the Gulf of Finland (Baltic Sea) using subfossil diatoms. *Journal of Paleolimnology*. Accepted.

V Kauppila, P., K. Weckström, S. Vaalgamaa, H. Pitkänen, A. Korhola, N. Reuss & S. Drew, 2005. Tracing pollution and recovery using sediments in an urban estuary, the northern Baltic Sea: Are we far from ecological reference conditions? *Marine Ecology Progress Series* 290: 35-53.

Major contributions of all authors. KW = Kaarina Weckström, AK= Atte Korhola, PS = Petri Shemeikka, SJ = Steve Juggins, PK = Pirkko Kauppila, SV = Sanna Vaalgamaa, HP = Heikki Pitkänen, NR = Nina Reuss, SD = Simon Drew

Paper I

Idea + planning: KW and AK

Field work: KW

Laboratory analyses: KW, except for TN and TP, which were analysed at the University of Joensuu

Data analysis: KW: water chemistry data, PS: GIS

Figures: KW

Writing: KW: main part of the text, PS: GIS methods, AK: comments and contributions to text

Paper II

Idea + planning: KW and SJ

Field work: KW

Laboratory analyses: KW

Data analysis: KW and SJ

Figures: KW (1,2,3,4), SJ (4)

Writing: KW, SJ commented on the text

Paper III

Idea + planning: KW and SJ

Field work: KW

Laboratory analyses: KW

Data analysis: KW and SJ

Figures: KW

Writing: KW, SJ and AK commented on the text

Paper IV

All phases: KW

Paper V

Idea + planning: PK, KW, SV, HP, AK

Field work: KW and SV collected the sediment material

Laboratory analyses: KW: diatom analysis, SV: geochemistry, NR: pigments, SD: stable isotopes

Data analysis: PK: water chemistry data, KW: sediment data

Figures: PK (1,2,5), KW (6,7,8), SV (1,3,6,8)

Writing: PK: water chemistry analyses, results and discussion; introduction & general discussion, KW: diatom analysis, results and discussion; data analysis on sediment data, SV: sediment geochemistry analysis, results and discussion, HP: water chemistry analysis, AK: data analysis on sediment data, NR: pigment analysis, results, SD: stable isotope analysis, results. All authors actively commented on different draft versions

1. Introduction

Coastal seas are highly diverse and productive ecosystems, which makes them a very valuable resource from a human perspective. However, as more than three-quarters of the world's population live in coastal catchment areas, they are subject to growing pressures (Vitousek et al. 1997; Jickells 1998). The increase in the loading and transport rates of the limiting plant nutrients nitrogen (N) and phosphorus (P) during the last century (Conley 2000; de Jonge et al. 2002) has led to undesirable eutrophication effects in receptive areas. Consequently, marine eutrophication of coastal waters has become a significant and wide-spread problem (e.g. Turner & Rabelais 1994; Nixon 1995; Billen & Garnier 1997; Paerl et al. 2003). Eutrophication induces changes in the structure and function of coastal foodwebs (Bonsdorff et al. 1997; Jackson et al. 2001; Grall & Chauvaud 2002) and results in anoxia and hypoxia of the sea bottom (Jørgensen et al. 1990; Gray et al. 2002).

The trend of recent post-industrial nutrient enrichment can be observed in the Baltic Sea, a large semi-enclosed brackish water basin with an extensive, diverse and densely populated drainage area. The Baltic Sea is one of the main sea areas of the European Union and, hence, has strategic importance to the economic and social development of Europe (EurOCEAN 2004). It is, however, extremely vulnerable to anthropogenic disturbance due to its special hydrographical conditions, simple ecosystem structures and poor water exchange with the North Sea (Voipio 1981). The nutrient load of the Baltic Sea has strongly increased from its natural level (Cederwall & Elmgren 1990; HELCOM 1990; Wulff et al. 1994). Larsson et al. (1985), for example, estimated an eightfold increase in P and a fourfold increase in N loading during the last century. Since the bulk of nutrient loading is land-derived, shallow coastal areas with often restricted water exchange are the areas most impacted by increased nutrient levels. This is manifested as changes in phytoplankton species composition and increased plankton biomass (HELCOM 1991), which has led to decreased water transparency (Cederwall & Elmgren 1990; Bonsdorff et al. 1997). The amounts of filamentous algae have increased (Rosenberg et al. 1990; Mäkinen et al. 1994), and nuisance blue-green algal blooms have become a common phenomenon (Melvasalo 1978; Plinski 1992; Kauppila & Lepistö 2001). Resulting from increased turbidity and competition with fast-growing filamentous algae, the depth penetration of the bladder wrack (*Fucus vesiculosus*) belts has declined (Kautzky et al. 1986; Bäck et al. 2001). Increased primary production increases sedimentation and decomposition of organic matter, which has led to anoxia near the bottom in susceptible semi-enclosed coastal areas (Rosenberg et al. 1990; Pitkänen et al. 2001). Excessive nutrient enrichment of coastal waters can alter the biogeochemical cycle of silica causing dissolved silicate limitation (e.g. Conley et al. 1993). This may ultimately change the species composition of the spring bloom towards a flagellate-dominated community (Officer & Ryther 1980), which will also affect higher trophic levels. Benthic fauna has responded to increased organic sedimentation with decreased species diversity (Pearson & Rosenberg 1978; Bonsdorff et al. 1991; Bonsdorff et al. 1997) and increased biomass, except in anoxic areas, where numbers

have decreased drastically (see Rosenberg et al. 1990). Although natural phenomena such as fluctuations in weather conditions, the large freshwater supply via rivers and the geomorphology of the coast regulate the hydrodynamics of coastal waters in the Baltic Sea, it is evident that these recent changes have been human-induced.

The setting of functional management targets requires knowledge about the present structure and biogeochemical nutrient cycles of coastal ecosystems. A prerequisite for intelligent management, however, is to also have information about how community structures and nutrient concentrations have varied through time and what the background conditions have been. When attempting to assess the onset, trends and magnitude of recent eutrophication one is hampered by the lack of reliable long-term data as most contemporary monitoring programs only cover the last ca. 30 years. Although hydrographic features have been documented in the Baltic Sea since the late 19th century (HELCOM, 1990), nutrients and biota have only been monitored since the 1960s (Kohonen 1974). The recently implemented European Water Framework Directive (WFD) (Anon. 2000) requires the surface waters of the European Union to be of “good ecological status” by 2015-2027, where biological and chemical elements depart only slightly from those expected under undisturbed conditions. EU member states are expected to 1) develop classification schemes that describe the present ecological status of surface waters based on information on background conditions against which recent anthropogenic disturbance can be assessed, and 2) develop efficient and cost-effective monitoring strategies focusing on biological parameters.

Baseline conditions in marine environments can be estimated using methods such as numerical models, integrating data from multidisciplinary projects (e.g. Billen & Garnier 1997; Nielsen et al. 2003) and data mining from early studies and historical data. At present, however, most computer models are limited in their direct relationships to complex, spatially and temporally variable ecosystem processes. Data mining, on the other hand, can be compromised by the poor quality of archived data.

The constraints discussed above limit our ability to effectively manage and protect our aquatic resources. Without knowledge of past conditions it is difficult to monitor the effectiveness of nutrient reduction policies, or even know what the appropriate restoration targets for degraded ecosystems are. Information about the effect of long-term nutrient enrichment on past ecosystem structure and functioning can best be answered using palaeolimnological methods, which use the record of chemical and biological changes contained in sediment cores. These archives can be opened by analysing the sediment itself (its physical and chemical structure) and the remains of different organisms preserved in the sediment, e.g. diatoms, chrysophyte and dinoflagellate cysts, cladocerans, chironomids, pollen and plant macrofossils (e.g. Smol et al. 2001a, 2001b). By dating the sediment using, for example, radioactive isotopes (²¹⁰Pb, ¹³⁷Cs, ¹⁴C) the information contained in the sediment can be transformed into a historiography of a water body (e.g. Appleby 2001).

Diatoms (Bacillariophyceae), which are single-celled aquatic algae, are the most widely used indicator group in palaeolimnology, as they 1) are present in diverse, numerically abundant

assemblages, 2) play an important role in primary production and food web dynamics in aquatic ecosystems, 3) have a siliceous cell wall that generally preserves well in freshwater and marine sediments, 4) can be identified to species or sub-species level due to their taxonomically distinct frustules, and 5) are known to be sensitive to environmental change (e.g. Battarbee et al. 2001). Due to these characteristics, diatoms have also increasingly been used in contemporary monitoring programs for assessing the present water quality of rivers and lakes (Dixit & Smol 1994; Kelly & Whitton 1995; Kelly et al. 1998; Eloranta & Soininen 2002). To date, similar use of diatoms has not been made in coastal waters.

Many studies have used knowledge of the ecology and distribution of contemporary species to qualitatively interpret past aquatic conditions from subfossil assemblages (e.g. Brugam 1978; Engstrom et al. 1985; Stabell 1985; Fritz 1989; Douglas et al. 1994; Anderson et al. 1995; Korhola & Blom 1996; Andren et al. 2000; Cooper et al. 2004). The development of weighted-averaging (WA) regression and calibration statistical techniques (ter Braak & van Dam 1989; ter Braak & Juggins 1993) allow quantitative reconstruction of ecological change from diatom sedimentary records using species optima for the environmental variable of interest derived from modern calibration data sets. These data sets consist of the present limnology and surface sediment diatom assemblages of a sufficient number of sites (usually > 40) along an environmental gradient. The abundance-weighted average of all species' optima calculated for each analysed fossil sample gives a good estimate of the e.g. nutrient or pH status of the time represented by that sample, with statistically reliable errors of prediction (RMSEP) estimated by cross-validation. These approaches have been successfully applied to fresh water ecosystems using diatoms with regard to e.g. surface water acidification (e.g. Birks et al. 1990; Renberg et al. 1993; Battarbee 1994, Hall & Smol 1996; Dixit et al. 1999), eutrophication (e.g. Jones & Juggins 1995; Bennion et al. 1996, Hall et al. 1997; Bradshaw & Anderson 2001) and climate change (e.g. Fritz et al. 1991; Korhola et al. 2000; Bigler et al. 2003). This methodology is not yet common in coastal marine systems. Diatom-based transfer functions have been developed by Juggins (1992) for salinity in the tidal part of the River Thames, by Jiang et al. (2002) for sea-surface temperature around the shelf-seas of Iceland, and by Campeau et al. (1999) for water depth in coastal areas of the southeastern Beaufort Sea. The first diatom-based transfer functions for nutrients in coastal systems have been developed in the Baltic Sea area (Clarke et al. 2003; Paper III in this thesis).

By combining qualitative chemical and biological proxies representing a range of ecosystem components with quantitative reconstructions of individual water chemistry variables, the palaeolimnological approach becomes more integrated. Uncertainty in interpretations is reduced when using complementary information from changes in a number of different indicators. These multi-proxy studies, although laborious, provide a multifaceted insight into past environments and their structure (e.g. Lotter et al. 1998; Hall et al. 1999; Birks et al. 2000; Korhola et al. 2002; Cooper et al. 2004; paper V in this thesis).

The term palaeolimnology commonly refers to limnic or lake environments, where the majority of studies in this field have been conducted. Palaeolimnological data are much sparser

from estuarine and coastal waters due to the dynamic, high-energy nature of these environments. Resuspension and sediment mixing from tides, waves and currents can result in homogenized sediments that lack temporal resolution suitable for palaeolimnological studies. Another concern in applying palaeolimnological methods in coastal sites is bioturbation caused by large benthic macrofauna. However, the tidal activity is negligible in the Baltic Sea, and the coastline is generally broken with fjords and embayments providing environments sheltered enough (e.g. fladas) for the accumulation of fine grained sediments. Bioturbation, though, may still be of concern in these sheltered environments.

The overall aim of this work was to assess post-industrial eutrophication in coastal waters of the Gulf of Finland, which is one of the most affected parts of the Baltic Sea (Pitkänen et al. 1990; Kauppila and Bäck 2001). The specific objectives of this study were to:

- 1) describe the present state of 49 embayments along the southern coast of Finland in terms of their physical and chemical characteristics, and to examine the effects of different land-use practices on their water quality;
- 2) determine which environmental variables mostly contribute to the composition and distribution of coastal diatom assemblages in the Gulf of Finland, and to describe the present diatom assemblage structure along the defined environmental gradients;
- 3) generate diatom-based transfer functions for quantifying long-term nutrient changes in coastal waters and evaluate their performance;
- 4) define long-term trends in eutrophication and establish ecological background or reference conditions at five embayments affected by varying land-use practices;
- 5) demonstrate the potential of multi-proxy studies in coastal environments by constructing a detailed history of disturbance and subsequent recovery in an urban embayment using a range of biological and chemical parameters preserved in the sediment record; and
- 6) assess the use of diatoms for contemporary water quality classification and monitoring purposes, and demonstrate their applicability for defining background conditions according to requirements of the European Water Framework Directive.

2. Study area

The study area was located on the southern coast of Finland from 22°93'E to 27°77'E (Fig. 1), which belongs to the hemi- and southern boreal vegetation zones. The bedrock in the eastern part of the study area consists entirely of Rapakivi granite, whereas in the western part it mostly consists of granite and to a lesser extent of granodiorite and quartz-feldspar schist and gneiss. The main soil types are evenly distributed along the coast consisting of till, clay and silt (Suomen kartasto 1990). The coastline is characteristically indented and contains many small, sheltered, and relatively shallow embayments, which freeze over in the winter. This thesis focused on 49 such sites (Appendix 1).



Figure 1. The index map shows the Baltic Sea and its bordering countries. In the detailed map the calibration data set is outlined and the five long-core sites are indicated by stars.

The catchment areas of the study sites are small; 62% of the embayments have a catchment area less than 10 km². All main types of land use in southern Finland are represented in the catchments including agriculture, industrial activities, municipalities and fish farming. A few catchments are less developed with only some summer cottage dwellings. Many of the catchment areas are dominated by forests (mean ca. 73 %), while agricultural land comprises between 0 and ca. 34 % with a mean of ca. 13 % (Appendix 1). Only sites in Helsinki have a marked proportion of densely built-up urban area in their catchments, while the proportion of sparsely built-up urban area (mostly summer cottages) varies considerably.

Five of the study sites with differing land use practices in the catchment area were chosen for a detailed assessment of their eutrophication history (Table 1). The two urban sites are located in Helsinki, the capital of Finland. Töölönlahti lies in the centre of the city surrounded by a densely built residential area, a railway yard, and a park; Laajalahti is located in the western part of the

greater Helsinki area. Extensive urbanisation of the catchment area of Töölönlahti took place in the late 19th century, while urbanisation in the Laajalahti catchment area did not start until the 1920s. Both embayments have received waste water from sewage treatment plants during the 1900s (1910-1959 at Töölönlahti, and 1957-1986 at Laajalahti). These sites are still eutrophic with annual mean total phosphorus (TP) concentrations of ca. 70 $\mu\text{g l}^{-1}$ and 65 $\mu\text{g l}^{-1}$, and a total dissolved nitrogen (TDN) concentration of ca. 600 $\mu\text{g l}^{-1}$ and 400 $\mu\text{g l}^{-1}$, respectively.

Table 1. Description of the five long-core sites. Values for catchment and surface area are rounded up. Approximate mean depth was estimated from charts. To = Töölönlahti, La = Laajalahti, PP = Pieni Pernajanlahti, Fa = Fasarbyviken, Ha = Hellänlahti

	<u>Urban sites</u>			<u>Rural sites</u>	
	To	La	PP	Fa	Ha
Location	60.18°N, 24.93°E	60.18°N, 24.87°E	60.38°N, 25.90°E	60.37°N, 25.98°E	60.58°N, 27.77°E
Catchment area (km ²)	4.7 / 0.4	52	356	18	386
Surface area (km ²)	0.2	5.3	9.0	1.3	2.0
Appr. mean depth (m)	2.0	2.4	5.0	2.0	1.6
Transparency (m)	0.7	1.0	1.2	1.0	0.8
Salinity (‰)	4.8	5.4	4.4	4.8	3.7
TDN ($\mu\text{g l}^{-1}$)	600	393	398	365	385
TP ($\mu\text{g l}^{-1}$)	68	63	45	39	48
Chl a ($\mu\text{g l}^{-1}$)	46	14	17	13	17

Two of the rural sites, Pieni Pernajanlahti and Hellänlahti, are estuaries, where riverine loading is the main external nutrient source. Consequently, these sites have large catchment areas (356 km² and 386 km², respectively) of which agricultural land comprises ca. 26 % (Pieni Pernajanlahti) and 15 % (Hellänlahti), while the rest is almost entirely covered by forests. At Hellänlahti, three fish farms, which are located in the outer part of the estuarine complex Virolahti, which Hellänlahti is part of, may also affect the water quality of the site. Both sites are presently eutrophic with an annual mean TDN concentration of ca. 400 $\mu\text{g l}^{-1}$ and 390 $\mu\text{g l}^{-1}$, and a TP concentration of ca. 45 $\mu\text{g l}^{-1}$ and 50 $\mu\text{g l}^{-1}$, respectively. The third rural site, Fasarbyviken, is a small embayment. Its catchment area is dominated by forests (ca. 69 %), while agricultural land, which is mainly located in the immediate vicinity of the site, covers ca. 24 %. A relatively large pig farm (appr. 700 heads) located in the catchment may affect the water quality of the embayment, as the manure is spread on nearby fields. Fasarbyviken is moderately eutrophied with an annual mean TDN concentration of ca. 370 $\mu\text{g l}^{-1}$ and a TP concentration of ca. 40 $\mu\text{g l}^{-1}$.

3. Materials and Methods

3.1. Sample collection and laboratory analyses

The 49 sites chosen for the calibration data set of modern limnology and surface sediment diatom assemblages were all lake-like, i.e. relatively small and semi-enclosed to ensure undisturbed sedimentation and to ascertain that the fossil diatom assemblages of the sediments originate from the assemblages of the overlying waters. Little or no previous information on the water quality of these embayments was available. Forty-five of the study sites were visited six times between August 1996 and February 1998 to encompass the seasonal variability in water chemistry. Four additional sites were included in the data set and were sampled in February and August 1998 to increase the number of sites at the high end of the nutrient gradients (sites 46-49).

An estimate of maximum water depth of each embayment was undertaken in August 1996, except for the four sites sampled in 1998. Each site was traversed by boat along a few transects (number of transects depending on the size of the embayment) with a portable echo sounder and the maximum depth recorded in meters. In many cases the depth could be read off a chart. An approximate surface area was determined from 1:50 000 maps using a digitizer. Sampling procedures for physical and chemical analyses are given in detail in paper I (p. 117-118). In summary, water samples were collected close to the deepest point of each site at 1 m depth. One surface sample each visit was considered to be representative as the sites are shallow and generally not stratified during the summer months. For the determination of chlorophyll *a* (Chl-*a*), however, a composite of three samples was taken. Nutrient analyses and field measurements of physical and chemical variables were carried out four to six times during the sampling period, except for the four additional sites with only two measurements. Other variables, mainly ionic chemistry, were measured less frequently (two to four times) as their concentrations tend to fluctuate less.

Surface sediment samples were collected at the same location as the water samples with a small gravity corer (Glew 1989). The top 1 cm was extracted for analysis of modern diatom assemblages and stored in small plastic bags at 4 °C. The long-core sites were sampled with a Mini- Mackereth corer (Mackereth 1969) in September 1998, except for Töölönlahti, which was sampled in October 2003. The cores were taken from the deepest area at all sites, and sectioned into 1 cm intervals. The main characteristics of the obtained sediments were described and sub-samples were stored at 4 °C in small plastic bags.

All laboratory and statistical analyses carried out in this study are listed in Table 2. The papers in which a detailed description of the methods can be found are also indicated.

Table 2. Analyses used in papers I-V. Papers with a detailed description of individual methods are indicated in bold.

Parameter	Paper	Analytical method
Laboratory techniques		
<i>Water chemistry</i>		
pH	I	<i>in situ</i> measurement
Alkalinity	I	Potentiometric titration
Conductivity	I, II	<i>in situ</i> measurement
Salinity	I, II, III, IV	<i>in situ</i> measurement
Total dissolved nitrogen (TDN)	I, II, III, IV, V	Spectrometry. Oxidation with peroxodisulfate
Nitrate & nitrite (NO ₃ +NO ₂)	I, II	Phenoldisulphonic acid spectrometric method
Ammonium (NH ₄ ⁺)	I, II	Spectrometric determination with indophenol blue
Total phosphorus (TP)	I, II, III, IV	Ammonium molybdate spectrometric method
Total dissolved phosphorus (TDP)	I, II	Ammonium molybdate spectrometric method
Dissolved inorganic phosphorus (DIP)	I, II	Ammonium molybdate spectrometric method
Chlorophyll <i>a</i> (Chl <i>a</i>)	I, II, III, IV	Spectrometric method. Extraction with acetone
Dissolved silica (DSi)	I, II	Molybdenum blue spectrometric method
Sulphate (SO ₄)	I	Turbidimetric method. Spectrometric determination
Chloride (Cl)	I	Colorimetric method. Potentiometric titration
Calcium (Ca), Magnesium (Mg)	I	Flame atomic absorption spectrometry
Sodium (Na), potassium (K)	I	Flame emission photometric method
Iron (Fe), manganese (Mn)	I	Flameless atomic absorption spectrometry
<i>Sediment geochemistry</i>		
Organic content	V	Loss-on-ignition at 550°
Grain size	V	Measured on a Malvern Mastersizer
Total carbon (TC), total nitrogen (TN)	V	Leco-analyser
TP, inorganic P (IP); (organic P (OP) = TP-IP)	V	Ammonium molybdate spectrometric method

Biogenic silica (BSi)	V	DeMaster – method
Copper (Cu), Zinc (Zn)	V	Atomic absorption spectrophotometry
Stable isotope $\delta^{15}\text{N}$	V	Continuous flow isotope mass spectrometry
Sedimentary pigments	V	High performance liquid chromatography (HPLC)
Diatoms	II, III, IV, V	Oxidation using H_2O_2
Dating of long cores	III, IV, V	^{210}Pb , ^{137}Cs (CIC, CRS models), SCP (paper V)
Numerical techniques		
Catchment area outlining & land use %	I	GIS (ESRI's ArcView 3.2 & Spatial Analyst)
Correlations between limnological variables	I	Pearson correlation coefficients
Relationships between limnology & land-use	I	RDA
Significance of monotonic trends in monitoring data	V	Kendall Tau B
Magnitude of significant trends in monitoring data	V	Linear regression analysis
Relationships between Chl <i>a</i> and TN and TP in monitoring data	V	Linear regression analysis
Effective no. of occurr. of taxa in the calibration data set	II	Hill's N2 values
Species turnover (gradient length) of the calibration data set	II, III, V	DCA
Diatom-environment relationships	II, III, V	CCA, partial CCA
Response of individual taxa to environmental variables	II	Hierarchical set of taxon response models
Species richness	II, IV, V	Rarefaction analysis
Transfer function development	III, IV, V	WA-PLS
Diatom-inferred TDN reconstructions	III, IV, V	WA-PLS
Reliability of WA-PLS reconstructions	IV	MAT, goodness-of-fit using CCA
Statistical zonation of stratigraphical data	IV, V	Optimal sum of squares partitioning & broken-stick model
Overall changes in diatom community structure of core data	IV, V	CA, DCA
Trends in biological and chemical variables of core data	V	PCA

3.2. Sediment dating

The most widely used methods for dating recent (post-industrial) sediments covering the last ca. 150 years include radioisotopic techniques such as ^{210}Pb and ^{137}Cs . ^{210}Pb is a naturally occurring radioisotope in the uranium (^{238}U) decay series, which is constantly produced by natural processes and constantly decaying. It has a half life of 22.26 years suitable for constructing chronologies over ca. the last century. The ^{210}Pb activity of sediments has two components: supported and unsupported ^{210}Pb . Unsupported ^{210}Pb is deposited from the atmosphere at a constant rate, while supported ^{210}Pb is derived from the *in situ* decay of the parent radionuclide ^{226}Ra . Problems using this dating method may arise with e.g. unsupported ^{210}Pb transport from the catchment area via erosion, loss of ^{210}Pb inputs via outflow, and spatial redistribution of ^{210}Pb deposits with e.g. sediment focusing (e.g. Smol 2002). Despite these potential problems this method has given reliable results in the great many studies in which it has been employed. Two models are generally used for calculating the sediment dates: the constant rate of supply model (CRS) and the constant initial concentration model (CIC) (Appleby et al. 1978). The former is more frequently used, the latter, however is more suitable in cases with large changes in sediment accumulation rates (e.g. slump events, sediment focusing).

To pinpoint certain time periods in sediment stratigraphies, it is possible to use radioisotopes produced by the nuclear industry. The most common isotope used is ^{137}Cs . Stratospheric testing of atomic weapons has been a major source of ^{137}Cs starting in 1952 and peaking in 1962-1963. After the banning of stratospheric testing the concentrations dropped steadily. This radioisotope was also released in the Chernobyl (Russia) nuclear power plant accident in 1986, the fallout from the plume was, however, confined to areas in the former Soviet Union, Europe and Turkey, where this isotope can be detected as a clear peak (exceeding the 1963 marker) in recent sediments. The main problem associated with this dating method is post-depositional mobility of ^{137}Cs . This often results in a “tail” extending to depths preceding stratospheric nuclear weapons testing; the 1963 peak, however, generally remains fixed (Smol 2002). For a detailed description of ^{210}Pb and ^{137}Cs dating see Appleby (2001).

Spheroidal carbonaceous particles (SCPs), which are formed when fossil fuels are burned at high temperatures, can be used as additional chronological control in sediment profiles (Tolonen et al. 1992; Rose, 2001). There are easily discernible changes in SCP concentrations, which are notably consistent wherever analysed, as the concentrations follow major changes in global fossil fuel combustion. This method does not produce a detailed chronology, but provides a few chronological tie-points (e.g. the start of the record in ca. the mid 1800s and the concentration peak in ca. the 1970s, depending on the region).

The four sediment cores collected in 1998 were analysed for ^{210}Pb , ^{226}Ra and ^{137}Cs by direct gamma assay in the Liverpool University Environmental Radioactivity Laboratory, using Ortec HPGe GWL series well-type coaxial low background intrinsic germanium detectors. The

technical procedures are described in Appleby et al. (1986, 1992). Sediment chronologies were calculated using the constant rate of supply (CRS) and the constant initial concentration (CIC) ^{210}Pb dating models together with chrono-stratigraphic dates determined from the ^{137}Cs record. To obtain a chronology for the Töölönlahti core, it was correlated using % LOI with another core from a previous study, which was dated using ^{210}Pb , ^{137}Cs and spheroidal carbonaceous particles (SCP) (Tikkanen et al. 1997). This was possible, as there were distinctive changes in % LOI. Chronologies for the cores collected in 1998 are given in Paper IV, Table 3. Spheroidal carbonaceous particles were also analysed from Laajalahti to provide additional support for the dating.

4. Results and discussion

4.1. The present

4.1.1. *Limnological properties of the study sites*

The study sites had a mean surface area of ca. 2 km² and a mean depth of 4.1 m. Their surface water salinity ranged from 0.7 to 6.4‰, spanning the salinity gradient in the Gulf of Finland (Haapala & Alenius 1994). Salinity showed a clear overall increasing trend from east to west following the open sea salinity gradient caused by the large River Neva estuary at the eastern end of the Gulf of Finland. The lowest levels, however, were always associated with sites receiving river input of fresh water. Water transparency was lowest in the shallow and more eutrophied embayments, ranging between 0.3 m and 3.5 m. Generally, the sites were turbid having a mean transparency of only 1.5 m. Reductions in Secchi depth readings during the last decades are clearly visible in long-term data series from the open Baltic Sea (Launiainen et al. 1989) as well as from coastal areas (Bonsdorff et al. 1997).

The nutrient concentrations of the sites ranged between 248-2068 $\mu\text{g l}^{-1}$ total dissolved nitrogen (TDN) and 15-86 $\mu\text{g l}^{-1}$ total phosphorus (TP) with mean concentrations of 419 $\mu\text{g l}^{-1}$ and 37 $\mu\text{g l}^{-1}$, respectively. Chlorophyll-*a*, reflecting primary productivity varied between 3 and 152 $\mu\text{g l}^{-1}$ with a mean of 13 $\mu\text{g l}^{-1}$. Only ca. 5 % of all sites had good water quality according to the water quality classification used by the Finnish Environment Institute (Antikainen et al. 2000). Ca. 65% had moderate water quality with the rest of the sites mostly falling into the class of poor rather than bad water quality (Table 3). The sites of the present study were classified using mean TP and Chl-*a* concentrations. TDN concentrations could not be used, as this water quality classification scheme does presently not include nitrogen as a variable for coastal waters. The mean values for nutrients and Chl-*a* showed no correlation with longitude (Paper I: Fig. 2), and thus did not follow the trend of increasing concentrations from west to east observed at

more pelagic stations (Pitkänen et al. 1990; Perttilä et al. 1995). Higher nutrient values regularly occurred in the vicinity of large rivers or close to urbanised areas (Paper I: Fig. 2).

Concentrations of dissolved silica (DSi) ranged between 0.24 and 2.25 mg l⁻¹ with a mean of 0.56 mg l⁻¹ (Appendix 1). The concentrations decreased markedly during the spring bloom and were low throughout the summer, being, on average, 0.30 mg l⁻¹ or less. Concentrations of DSi have decreased over the last few decades in the Baltic Sea as a result of increased diatom production and the resultant increase in sedimentation of silica in the spring (Wulff & Rahm 1988; Sandén et al. 1991). Increased diatom spring blooms are a clear indication of increased nutrient availability, which has been reported in several studies both in freshwater and marine systems (Conley et al. 1993). A detailed description of the physical and chemical properties of the study sites can be found in paper I.

Table 3. An extract of the water quality classification used by the Finnish Environment Institute for coastal waters (Antikainen et al. 2000). Only total phosphorus and chlorophyll *a* are shown.

Variable	High	Good	Moderate	Poor	Bad
Total phosphorus (µg l ⁻¹)	< 12	12-20	20-40	40-80	> 80
Chlorophyll <i>a</i> (µg l ⁻¹)	< 2	2-4	4-12	12-30	> 30

4.1.2. Diatom assemblages

Due to the shallowness of the calibration data set sites (< 10 m, with one exception), the majority of diatom taxa found in this study were benthic, many of which occurred at low relative abundances. Only 14 taxa occurred at a maximum abundance over 10%, most of these were either planktonic or belonged to the genus *Fragilaria* (incl. revised genera), which composed the bulk of benthic assemblages (on average 35%). *Fragilaria* spp. were dominant in the shallow sites and had a strong, negative correlation to the percentage of planktonic taxa ($r = -0.87$, $p \leq 0.01$), which clearly increased with depth (Fig. 2). The correlation between the percentage abundance of planktonic taxa and depth ($r = 0.53$, $p \leq 0.01$), although being statistically significant, was not as strong as might have been expected due to the increased amount of small, weakly silicified planktonic taxa in shallow, eutrophic embayments. Species richness of the modern diatom assemblages (as estimated by rarefaction) varied considerably ranging from 39 at the deepest site of the data set (22 m, site 39), which was heavily dominated by *Cyclotella choctawhatcheana*, to 103 at a relatively large site (13) with low nutrient and chlorophyll-*a* concentrations. In general, species richness was lowest at both ends of the depth range. The average species richness of all sites was 75 taxa.

According to this study, both nitrogen and phosphorus, together with depth and salinity, account for significant and independent fractions of variation in the composition of coastal diatom

assemblages of the Gulf of Finland and, altogether, explained 34% of the total variation. Depth explained most variance (10.4%), followed by TDN (7%), TP (7%) and salinity (6.3%). The low conditional effects between the variables emphasise the largely independent influence of each variable on diatom distributions in this dataset (Paper II, Table 2).

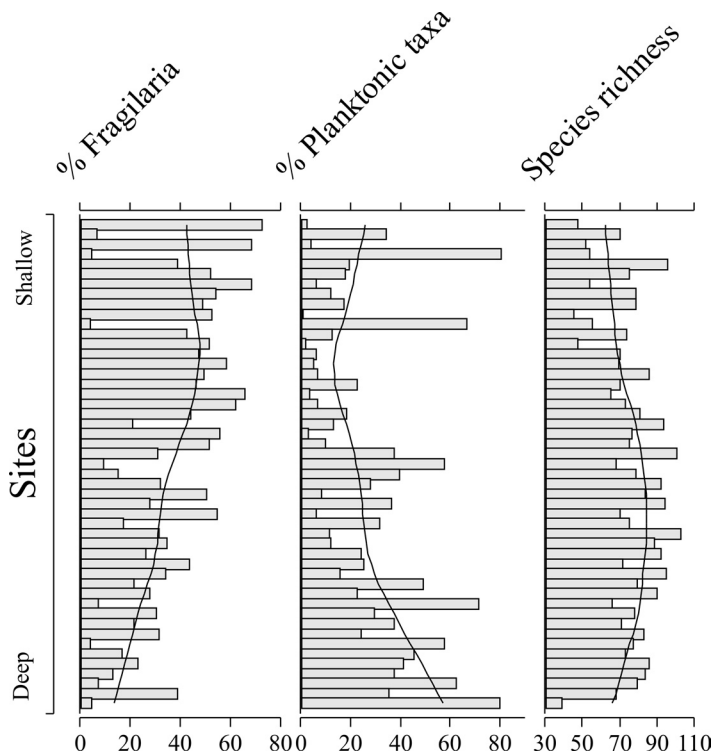


Figure 2. Selected characteristics of the surface sediment diatom assemblages: % abundance of *Fragilaria* spp. and planktonic taxa and species richness. The sites were sorted according to depth. A locally weighted scatterplot smooth (lowess, span 0.45) was fitted to all graphs to help detect major trends.

4.1.3. Linking limnology, land use and diatom community structure

The main environmental gradients in the modern calibration set of diatoms and associated environmental data were represented by nutrients and nutrient-related variables and depth (Fig. 3). The nutrient gradients were strongly related to urban development (municipalities, industry) rather than agriculture. Agriculture is an important factor influencing water quality (and consequently diatom assemblages); however, the proportion of agriculture in the catchment areas of this study was low to moderate (mean 13%, range 0-34%, Appendix 1). The relationships between catchment land use, nutrient gradients and diatom assemblages will most likely change with the inclusion

of sites with predominantly agricultural catchments, such as those being included as part of an ongoing national research programme BIREME funded by the Finnish Academy.

The deeper, slightly more saline sites in the data set (e.g. 39, 43, 44, 35) were least affected by nutrient enrichment, and consequently had high transparency and relatively low chlorophyll *a* concentrations. The catchments of most of these sites were dominated by forests (Figure 3). These embayments supported relatively diverse (species richness ca. 75 taxa) diatom communities including mostly benthic taxa, e.g. *Rhoicosphenia curvata*, *Staurosirella pinnata*, *Opephora mutabilis*, *Navicula phyllepta* and *Bacillaria paxillifer*, except for the deepest site (39), which was dominated by the planktonic *Cyclotella choctawhatcheeana* (abundance 72% of the whole assemblage).

The catchment areas of moderately eutrophied embayments (e.g. 3, 4, 7, 20, 27, 34, 36, 45, 49) were, in general, less forested (more disturbed) than the previous group. These sites showed higher nutrient concentrations and productivity, and lower water transparency compared to the least affected sites in this data set. Species richness was highest (on average > 80 taxa) in the deeper sites of this group, defined by planktonic taxa such as *Thalassiosira levanderi*, *Skeletonema costatum*, *Aulacoseira subarctica*, *Asterionella formosa* and *Stephanodiscus parvus*. The higher species richness could possibly be attributed to the moderate nutrient enrichment, as intermediate levels of productivity have been shown to increase the diversity of aquatic communities (e.g. Dodson et al. 2000; Irigoien et al. 2004). It may, alternatively, be explained by the freshwater input received by some of these sites, which is also indicated by the higher abundance of the latter three mostly freshwater taxa. The generally small and almost enclosed embayments in this group clearly exhibited lower species richness (mean ca. 65 taxa) due to the dominance of small *Fragilaria* taxa (incl. revised genera), such as *Fragilaria elliptica* agg., *Fragilaria amicornum*, *Staurosira* cf. *construens* var. *venter* and *Fragilaria exigua*. The predominance of *Fragilaria* spp. in shallow coastal embayments and in recently developed coastal lakes is a typical feature of the lake isolation succession in the Baltic Sea area caused by post glacial isostatic land uplift (Stabell 1985; Seppä & Tikkanen 1998). Mass occurrences of opportunistic and fast-reproducing *Fragilaria* spp. appear to be associated with environmental instability due to increased turbidity, changes in salinity or higher nutrient supply (e.g. Haworth 1976; Denys 1990).

The most eutrophied sites in the data set (e.g. 23, 24, 33, 46, 47) had a large proportion of urban area in their catchments (mean 41 %, range 9-81 %). These sites had high nutrient and chlorophyll *a* concentrations (mean TDN, TP and Chl-*a*: ca. 990, 70 and 55 µg l⁻¹, respectively); due to the high productivity they were also very turbid (mean water transparency ca. 0.7 m).

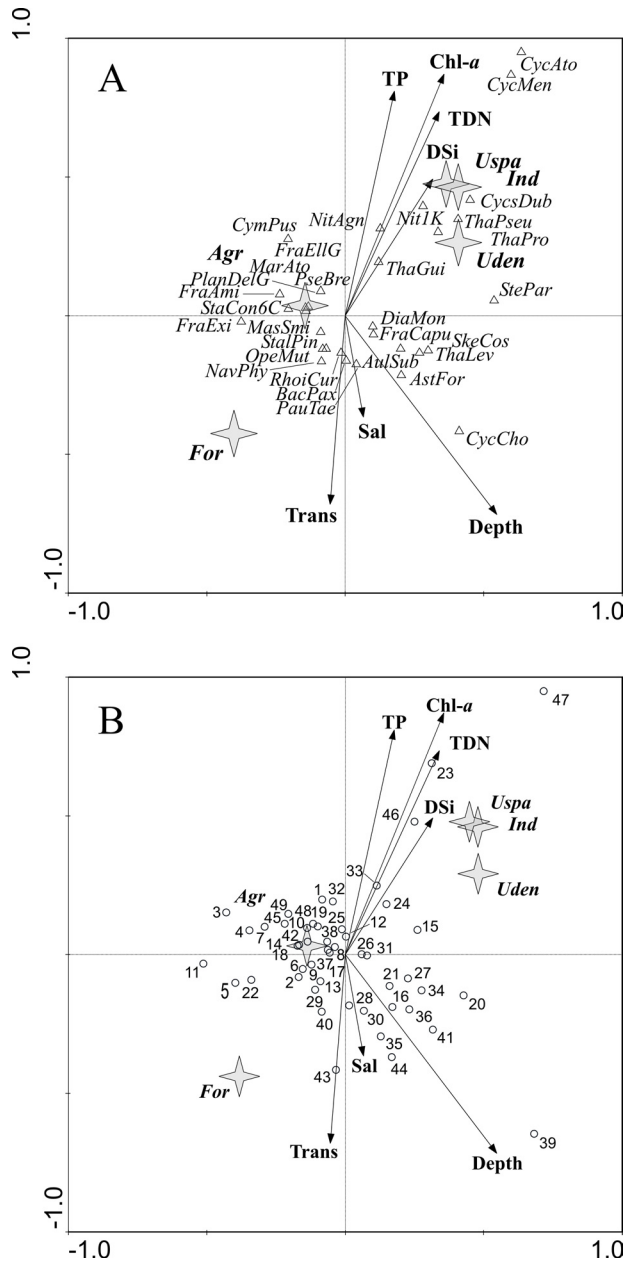


Figure 3. CCA ordination diagram showing the relationship between species A), sites B), and selected environmental variables. The first two ordination axes were statistically significant ($p \leq 0.001$) and together accounted for 24 % of the variance in the species data. Catchment land use classes were included in the CCA analysis as passive variables, which do not influence the definition of the ordination axes, but are added to an existing ordination by projection. These five classes are indicated by stars (Agr = agriculture, For = forests, Usps = sparsely built-up urban area, UDen = densely built-up urban area, Ind = industrial area)

These sites supported an average species richness of only ca. 64 taxa, which is concordant with the finding that anthropogenic eutrophication often decreases the species richness of algal communities (e.g. Wetzel 2001). Taxa typical of these sites included small planktonic species such as *Cyclotella atomus*, *C. meneghiniana* and *Thalassiosira pseudonana* (Fig. 3). The increased relative abundance of small planktonic taxa may be a result of several factors, including e.g. higher nutrient requirements of these species, high turbidity of the sites limiting benthic diatom growth, and dissolved silicate limitation, which is caused by prolonged nutrient enrichment of a water body (e.g. Schelske et al. 1983; Conley et al. 1993). Under such conditions taxa with low silica requirements, like the small planktonic taxa in this study, will be strong competitors (Tilman et al. 1982). There are, however, no clear trends in the dissolved silicate concentration of the studied sites (paper I).

4.2. The past

4.2.1. Constructing chronologies

The dating results are given in detail in Paper IV, hence only a shorter summary will be given here. All sites sampled in 1998 contain reasonably good ^{210}Pb records with gradually declining concentrations of unsupported ^{210}Pb (Fig. 4). At these four sites, the period represented by the total ^{210}Pb activity before reaching equilibrium with the supporting ^{226}Ra was about 100 years. At Laajalahti and Hellänlahti, this occurred at a depth of ca. 20–25 cm, whereas at Pieni Pernajanlahti and Fasarbyviken equilibrium was reached at a greater depth of 60 and 75 cm, respectively.

All four cores taken in 1998 had a well-resolved subsurface peak of ^{137}Cs activity, recording the fallout from the 1986 Chernobyl accident. At Laajalahti and Hellänlahti, it was not possible to distinguish a peak recording the 1963 nuclear weapons fallout maximum, whereas at Pieni Pernajanlahti and Fasarbyviken this incidence was observed as a small shoulder on the ^{137}Cs profile at depths of 31.5 and 48.5 cm, respectively.

Applying the 1986 ^{137}Cs date as a reference point, composite model ^{210}Pb chronologies (using both CIC and CRS) were constructed for Laajalahti and Hellänlahti. ^{210}Pb dates for Pieni Pernajanlahti and Fasarbyviken were calculated using the CRS model, together with the 1986 and 1963 depths of the ^{137}Cs stratigraphy, following methods outlined in Appleby (1998). At Laajalahti and Hellänlahti, the sediment accumulation rate appears to have been relatively uniform during the first half of the 20th century. During the early 1980s accumulation rates increased dramatically and contemporary values are calculated to be ca. 0.6 cm y⁻¹ and 0.83 cm y⁻¹, respectively. At Pieni Pernajanlahti, the sediment accumulation rates were relatively uniform until 1940.

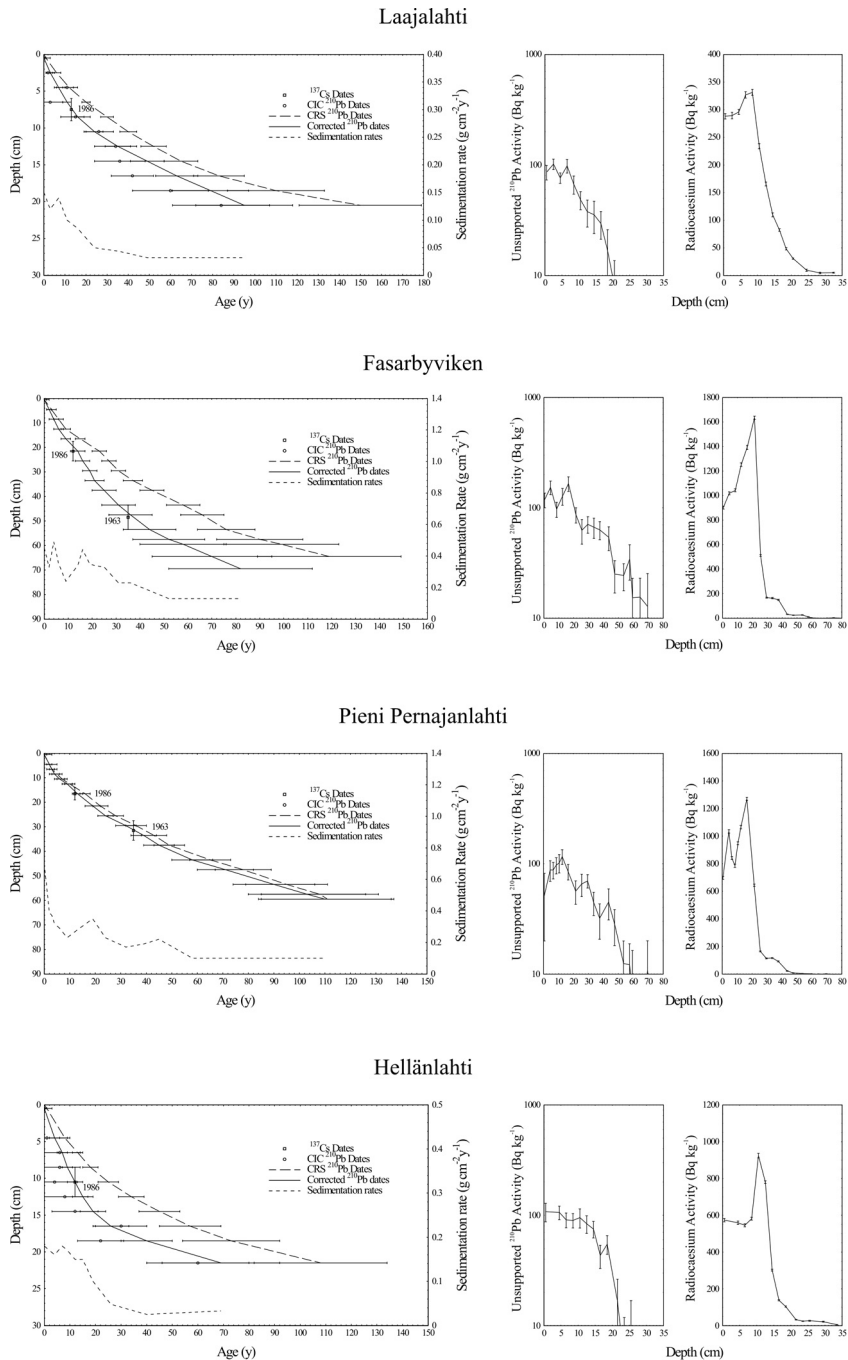


Figure 4. Radiometric chronology of Laajalahti, Fasarbyviken, Pieni Pernajanlahti and Hellänlahti showing CRS and CIC model ^{210}Pb dates together with the 1986 depth determined from the ^{137}Cs stratigraphy. Also shown are the corrected ^{210}Pb dates and sedimentation rates. Unsupported ^{210}Pb and ^{137}Cs are plotted on the right side.

Since then accumulation rates at Pieni Pernajanlahti have increased generally, with peaks in the early 1950s, ca. 1980, and during the past few years. The mean post-1963 sediment accumulation rate has been 0.96 cm y^{-1} . At Fasarbyviken, accumulation rates increased sharply between 1950-80 reaching peak values of more than $1.5\text{-}2 \text{ cm y}^{-1}$ in the 1980s and 1990s.

The core taken from Töölönlahti in 2003 was correlated with a previously dated core using %LOI, as easily discernible changes occurred in the % LOI profiles. The original core was dated using ^{210}Pb , ^{137}Cs and spheroidal carbonaceous particles (SCP). As the ^{210}Pb dating proved to be problematic, more weight was given to the ^{137}Cs and SCP dating, which were in good agreement, and were supported by the pollen record of park trees and known land use changes in the catchment area. The mean sedimentation rate of the core during the 20th century was ca. 0.6 cm y^{-1} . A more detailed discussion regarding the chronology of the site is given in Korhola & Blom (1996) and Tikkanen et al. (1997).

The Laajalahti core was additionally analysed for SCPs. The established ^{210}Pb chronology agreed well with the SCP record from this site, which showed a rapid increase in particle concentration starting in the late 1930s with a clear sub-surface peak at 1975, consistent with the known energy consumption data from the Helsinki area (P. Leeson, unpublished data). The SCP method has been shown to give fairly reliable age determinations for lake and coastal sediments in Finland (e.g. Tolonen et al. 1992), and can thus be used as an independent control for ^{210}Pb dating when determined from the same core.

In many cases it is difficult to find undisturbed high-resolution sediment records in marine environments due to sediment reworking caused by e.g. tides and currents. The sites chosen for this study are sheltered and are likely to have more in common with lake sediment records than many marine basins. This is indicated by the relatively undisturbed ^{210}Pb profiles shown in Figure 3, which clearly contrast to the near uniform ^{210}Pb profiles that occur in a number of marine coastal sediments.

4.2.2. Changes in diatom assemblage structure during the last century

The main changes in diatom assemblages of the five study sites are shown as trends in % planktonic diatoms, detrended correspondence analysis (DCA) axis 1 scores, and species richness in Figure 5. DCA axis 1 scores are a numerical means to summarise the compositional change of diatom assemblages over time. The DCA scores highly correlated with the abundance of planktonic diatoms (Töölönlahti: $r = 0.90$, Laajalahti: $r = -0.99$, Fasarbyviken: $r = -0.92$, Pieni Pernajanlahti: $r = -0.96$, Hellänlahti: $r = -0.82$; $p \leq 0.01$ in all cases) indicating that the largest change has occurred in diatom life forms, which can be attributed to changes in habitat availability. The urban and rural sites showed a very different development of diatom assemblages during the last

century (Fig. 5). The two urban sites displayed marked increases in the abundance of planktonic diatoms and a decrease in species richness starting in the 19th– early 20th century with increased urbanisation. At both sites a clear recovery (expressed as a decrease in the abundance of planktonic taxa and an increase in species richness) was observed after the cessation of waste water loading by the mid 1980s (Table 4). Diatom taxa defining the hypertrophic periods of these sites included *Cyclotella atomus*, *C. meneghiniana*, *Thalassiosira guillardii* and *Thalassiosira pseudonana*. At Töölönlahti, in particular the two *Thalassiosira* taxa were dominant when the ineffective waste water treatment plants were operational (and the nutrient loading presumably highest) between 1910 and 1959. All of these taxa have been reported thriving in eutrophic waters (e.g. EDDI 2001; Bradshaw et al. 2002; Gell et al. 2002; MOLTEN 2004; Cooper et al. 2004). In contrast, the changes observed in the rural sites were only moderate and occurred later starting in the 1940s with the most pronounced changes occurring after the 1960s (Table 4). The relatively late timing of the first changes in the diatom assemblages of the rural sites coincided with the intensification of agriculture in Finland after the Second World War. New fields were cleared and the use of artificial fertilisers increased dramatically (IFA 1996-2005). All rural sites showed an increase in small planktonic taxa, although this increase was not in the same order of magnitude compared to the heavily impacted urban sites and the taxa favoured by advancing eutrophication were partly different. These taxa included *Cyclotella choctawhatcheeana*, *Thalassiosira proschkinae* and *Thalassiosira pseudonana*, all reported to increase with anthropogenic disturbance (e.g. Cooper 1995; Andren et al. 1999; MOLTEN 2004). A small general decrease could be detected in species richness at Hellänlahti and Pieni Pernajanlahti over the last century, but this was, again, not as dramatic as the decrease observed at the urban sites during the period of waste water loading. It should be noted that Töölönlahti showed first signs of eutrophication as early as the beginning of the 19th century, as urban development around this site started in the early 1800s (Tikkanen et al. 1996). The assemblage changes prior to the 20th century at this site are discussed in Paper IV, but are not included in this summary.

The principal change in the diatom assemblages of all cores occurred in the proportions of diatom life forms. An increase in the abundance of planktonic taxa has frequently been reported as a response to eutrophication in fresh waters and marine environments (e.g. Cooper and Brush 1991; Andren et al. 1999; Sayer et al. 1999; Bennion et al. 2004; Cooper et al. 2004), as nutrient enrichment increases turbidity (via increased planktonic productivity), which favours planktonic taxa over benthic. The increased relative abundance of small planktonic taxa with eutrophication may also be attributed to other factors such as higher nutrient requirements of these species and silica limitation, as discussed earlier. As coastal planktonic diatom communities include far fewer taxa than benthic ones, a clear decrease in species richness can be observed when the abundance of planktonic taxa is high (Fig. 6).

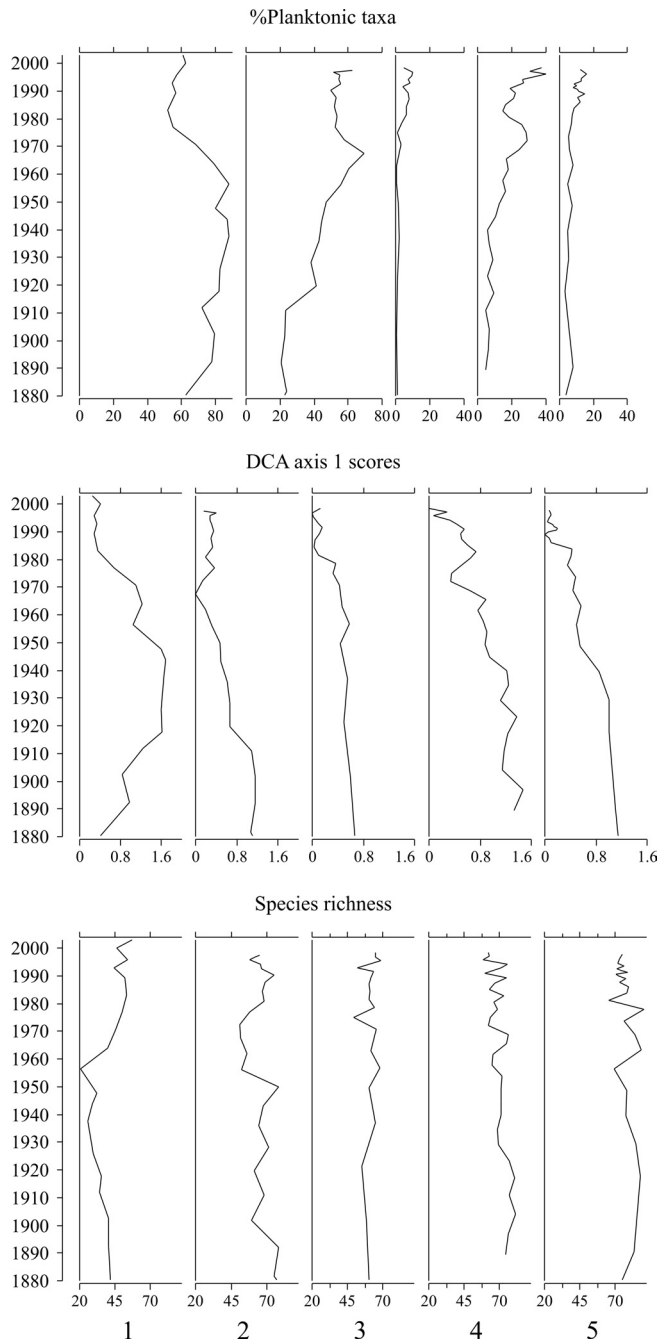


Figure 5. Summary diagram of the sites showing the main trends in % abundance of planktonic diatoms, diatom assemblage composition (DCA axis 1), and diatom species richness since ca. 1880. 1 = Töölönlahti, 2 = Laajalahti, 3 = Fasarbyviken, 4 = Pieni Pernajanlahti, 5 = Hellänlahti. Sites 1 and 2 are urban, and sites 3-5 rural.

Although a clear recovery at the urban sites was observed, the presently planktonic diatom assemblages of these two embayments show no change back to the pre-disturbance diverse benthic communities (Table 4) dominated by *Fragilaria* spp. but also characterised by several other taxa such as *Planothidium delicatulum* agg., *Navicula gregaria*, *Diatoma moniliformis* and *Rhoicosphenia curvata*. A similar situation, where diatom assemblages did not change back to the ones characteristic of pre-disturbance conditions after a reduction in nutrient loads, was observed by e.g. Anderson et al. (1990) and Lotter (2001). At present, the planktonic assemblages at both sites are maintained by internal loading from the sediments (Tikkanen et al. 1997; Paper V) as the embayments continue to be nutrient enriched despite decreased external loading, and by high turbidity, which has resulted in a loss of submerged macrophytes (Häyren 1921, 1937). The resultant decrease in sediment stability further enhances turbidity. This phytoplankton-dominated state may be preserved despite nutrient reductions due to the inability of aquatic plants to recolonise in highly turbid waters.

Table 4. Summarised trends in eutrophication, recovery, and estimated departure from chemical and biological reference conditions between urban and rural embayments of the Gulf of Finland. Biological reference conditions for these shallow embayments, which would naturally support mostly benthic diatom communities, were defined as the mean value of % planktonic taxa in 1850 (Laajalahti) / 1880 (rural sites) - 1900, and chemical reference conditions as the mean of diatom-inferred TDN during the same time period. Töölönlahti was not included when defining background conditions. The values in brackets indicate present % abundance of planktonic taxa.

Trend in eutrophication	Urban sites	Rural sites
Onset of eutrophication	early 1800s-1920	After 1940
Worst state	ca. 1915-1975	Since 1980s
Recovery	Partial since 1980s	No recovery
Present departure from chemical reference conditions	ca. 40 %	0-20%
Biological reference conditions	< 20% planktonic taxa (60)	< 10% planktonic taxa (8-36)

4.2.3. Diatom-nutrient models and their applicability to coastal environments

One of the basic requirements for the development of weighted-averaging (WA)-based transfer functions is that the reconstructed variable is ecologically important in the system of interest, or is linearly related to such a variable (Birks et al. 1990). In the present data set, variables accounting for significant and independent fractions of variation in the diatom data included depth, TDN, TP and salinity and the influence of each variable on diatom distributions in this dataset was largely independent as suggested by variance decomposition. The variance explained by these

four variables was relatively low (34 %), but it is typical of datasets containing many taxa and many zero values in the species matrix; more important are the associated permutation tests showing the statistical significance of the explanatory variables (Paper II, Table 2).

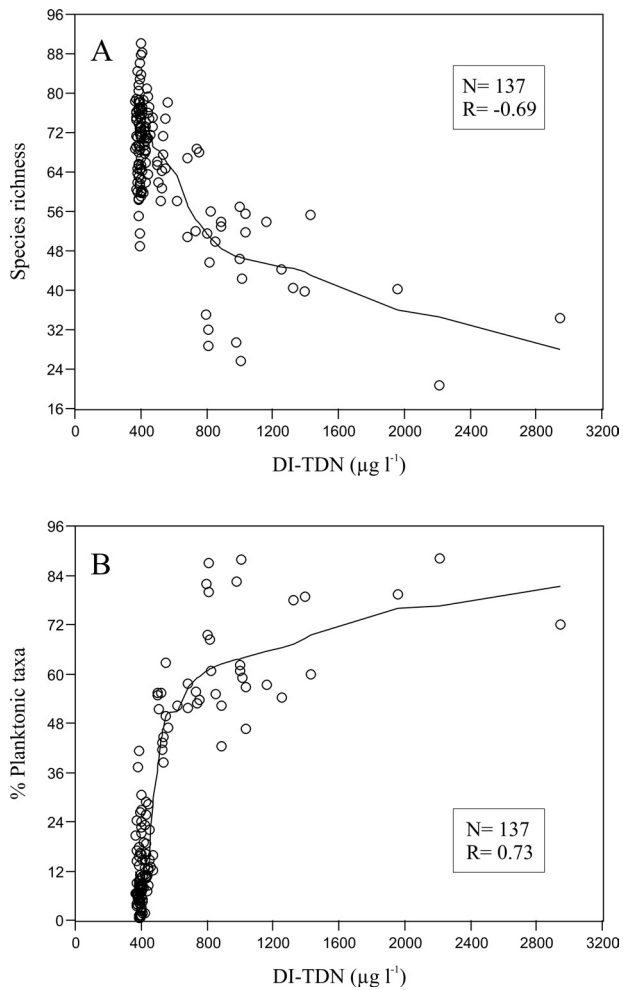


Figure 6. The relationship between diatom-inferred TDN and A) species richness and B) % abundance of planktonic taxa in the 137 fossil samples from all long-core sites.

WA-based transfer functions for freshwaters are generally robust, as the variable being reconstructed (e.g. pH, TP, salinity) tends to be the dominant variable controlling diatom communities in the collected calibration data sets. Problems do occur when secondary gradients are reconstructed (e.g. Anderson 2000). In coastal marine environments, the development of WA transfer functions for nutrients is less straightforward than in freshwaters because of the strong salinity gradients that can occur in estuaries and coastal waters (e.g. Juggins 1992; Conley et al. 2000b). Therefore, it is essential that the salinity signal in a coastal calibration set is not overriding

the nutrient signal. The salinity gradient along the southern coast of Finland is relatively short; hence both nutrients (TDN, TP) explain a higher proportion of the variance in the diatom data compared to salinity in the present data set.

Of several WA techniques assessed (see Birks 1995; Olander et al. 1999), weighted averaging partial least squares (WA-PLS) performed best and was used to develop a model for both nutrients. The model for TDN performed better than the TP-model in terms of the squared correlation between the observed and diatom-inferred values ($r^2 = 0.73$ and 0.57 , respectively), and the root mean square error of prediction ($RMSEP = 0.09_{\log}$ and 0.10_{\log} , respectively), as assessed by leave-one-out cross-validation. A likely explanation can be found in the fact that nitrogen is generally the limiting nutrient to phytoplankton growth in most parts of the Baltic Sea with the exception of the low-saline Bothnian Bay (e.g. Granéli et al. 1990; Pitkänen 1994). Consequently TDN was adopted for further analyses. The fact that TP also significantly explained variation in diatom assemblages could be attributed to seasonal limitation of TP observed in estuaries (Pitkänen & Tamminen 1994; Conley 2000; Meeuwig et al. 2000).

The two urban sites Töölönlahti and Laajalahti showed similar responses to urbanization and waste water loading in their diatom-inferred TDN (DI-TDN) (Figure 7). Concentrations of DI-TDN in both cores increased concurrent with increasing urbanization of the catchment areas (Laakkonen and Lehtonen 1999; Laurila & Laakkonen 2004). The inferred TDN concentration in the lower part of the core was substantially higher (Fig. 7) and the increase occurred somewhat earlier in Töölönlahti, most likely due to the more central location of the embayment in the gradually growing city in the late 1800s. Highest DI-TDN ($2948 \mu\text{g l}^{-1}$ and $826 \mu\text{g l}^{-1}$) were observed during the period of waste water loading to these embayments (1910-1959 and 1957-1986) followed by a clear decrease in concentrations after closure of the waste water treatment plants and the diversion of wastewaters elsewhere. There was, however, an unexpected decrease in DI-TDN concentrations at Töölönlahti between ca. 1920-1950, when diatom assemblage structure and known land use changes in the catchment area would have suggested the opposite. This will be discussed in more detail in the following paragraph.

At the three rural sites DI-TDN concentrations were considerably lower ($< 480 \mu\text{g l}^{-1}$) and displayed very different trends. Although diffuse nutrient loading from agriculture is the most important single anthropogenic source of nutrients to Finnish coastal waters (Pitkänen 1994), the effects are likely to be moderate in this study, as the catchment areas of the rural sites are mostly forested with agriculture comprising $< \text{ca. } 30 \%$ of the catchments. The reconstructed TDN for Fasarbyviken showed little variation throughout the core (Fig. 7), whereas DI-TDN fluctuated moderately in the Pieni Pernajanlahti core (Fig. 7) and appeared to be to some extent positively correlated with the mean flow of the River Iloanjoki ($r = 0.48$) (data, which are not shown here, was provided by the Uusimaa Regional Environment Centre from the 1970s onwards). As River Iloanjoki is the main source of external loading into Pernajanlahti, changes in the mean flow would affect the nutrient concentrations of the embayment. At Hellänlahti there was a distinct

increase in the mid 1980s after relatively stable concentrations throughout most of the 1900s. This could be attributed to increased riverine nutrient loading and the onset of fish farming at the site.

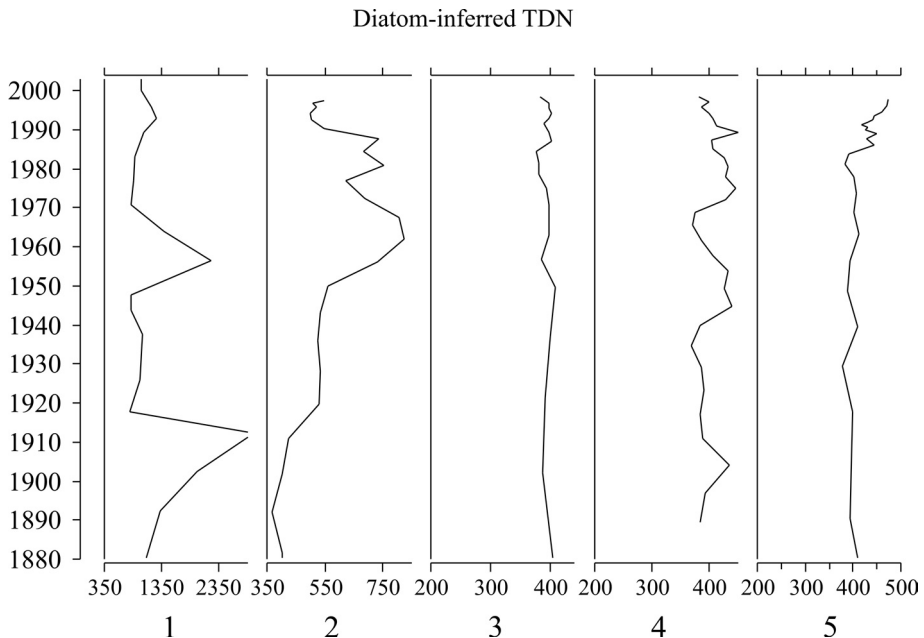


Figure 7. Summary diagram of the sites showing the main trends in the diatom-inferred TDN since ca. 1880. 1 = Töölönlahti, 2 = Laajalahti, 3 = Fasarbyviken, 4 = Pieni Pernajanlahti, 5 = Hellänlahti.

Although diatom-nutrient inference models are a sophisticated tool offering a quantitative means for reconstructing past nutrient concentrations directly, there also are potential problems and errors associated with these models, for example: 1) Inclusion of benthic diatom taxa in models, which are essentially designed to estimate open-water nutrient concentrations. The distribution of benthic diatoms is likely to be related to factors such as water depth, light availability and variations in substrate rather than to open water nutrients directly (e.g. Allen 1971; Bennion 1995; Michelutti et al. 2003). This problem applies to the rural cores of this study, which were dominated by benthic species. However, studies from the Baltic Sea show that nutrient enrichment of shallow coastal areas also affects benthic diatom assemblages by increasing production and changing the assemblage composition (e.g. Sundbäck and Snoeijs 1991). 2) Species shifts in the diatom stratigraphy could be caused by factors other than the variable being reconstructed. This was likely to be the case at Töölönlahti. The decrease in DI-TDN concentrations was mainly due to a marked decline in the relative abundance of *Cyclotella atomus*, which has the highest TDN optimum in our calibration data set and the dominance of *Thalassiosira guillardii*, which, although a eutrophic species, has a clearly lower TDN optimum (Appendix 2). The observed shift

between the two taxa may have been caused by silica limitation of the system with increasing nutrient concentrations, more intensive diatom blooms and resulting depletion of silica from the water column (e.g. Schelske et al. 1983; Conley et al. 1993). 3) Species optima may be poorly modeled if a) their response curves to the variable of interest are truncated at one end of the environmental gradient (Birks et al. 1990), and b) they have a broad range of tolerance, as the optima for such taxa tend to lie at the centre of the gradient covered (see Bennion et al. 2001). An example of each case in this data set is shown in Fig. 8. 4) High seasonal variability in nutrient concentrations requires a sufficient amount of measurements for defining a reliable yearly mean to be used in the WA-models. It is important that the measurements also include the periods relevant to diatom growth (see e.g. Bradshaw et al. 2002). High-frequency water sampling however, is both time-consuming and costly (in cases where no monitoring data is available). In this study, the yearly mean for TDN is based on four measurements, due to the large geographical coverage of the study sites and limited person-power. Monitoring data for most study sites were not available. The measurements, however, cover all seasons and should provide a good estimate of the general trophic status of the study sites. 5) Preservation problems of diatom frustules may occur for reasons related to e.g. water chemistry and physical abrasion. The composition of fossil assemblages can be distorted due to differential dissolution of taxa or diatoms could be completely missing due to poor preservation (Ryves et al. 2001). In this study diatom preservation was generally good; valves were neither severely dissolved nor broken beyond identification. 6) Poor analogues of fossil assemblages with the modern calibration data set (Birks 1998), which decreases the reliability of inferences made using present day species assemblages and their environment.

As there are a number of potential problems involved with diatom-nutrient model development, there is a need to carefully evaluate and validate the obtained reconstructions. This can be done by using e.g. long-term monitoring records of water chemistry and statistical measures such as the modern analogue technique (MAT) and the measure of goodness-of-fit. Four of the sites have a measured total nitrogen (TN) record of ca. 30 years (provided by the Finnish Environment Institute, data not shown), which enabled the validation of the reconstructions for the upper part of the cores. The DI-TDN concentrations were compared with the annual, seasonally weighted mean TN concentrations. When comparing the records, it should be noted that in the study area TDN composes only ca. 60-70% of TN (Paper III, pp. 326-327).

The main trends in the measured TN of the urban sites were recorded reasonably well by the diatom-inferred TDN. However, the actual TN concentrations were systematically underestimated by the transfer model during the most pronounced eutrophication period of Laajalahti (Paper III, Fig. 3); the inclusion of more sites at the high end of the TDN gradient could increase the accuracy of the model over this part of the gradient. There was a reasonably good agreement between the measured and diatom-inferred values since the 1990s at Laajalahti and since 1975 at Töölönlahti (Paper IV) giving some confidence to the DI-TDN reconstruction of these sites over nutrient

ranges covered by the calibration data set. The clearly underestimated TDN-reconstruction at Laajalahti could partly be explained by poorly modelled optima for the common eutrophic taxa (e.g. *Cyclotella atomus*, *C. meneghiniana*, *Thalassiosira pseudonana*), which showed truncated response curves to TDN (e.g. Fig. 8). This could strongly influence the inferred TDN concentrations. Although the model was capable of inferring TDN concentrations as high as 3000 $\mu\text{g l}^{-1}$ at Töölönlahti, concentrations during the time of waste water loading at this site were likely to be much higher, as is indicated by monitoring data from Laajalahti, where concentrations as high as ca. 5000 $\mu\text{g l}^{-1}$ were measured.

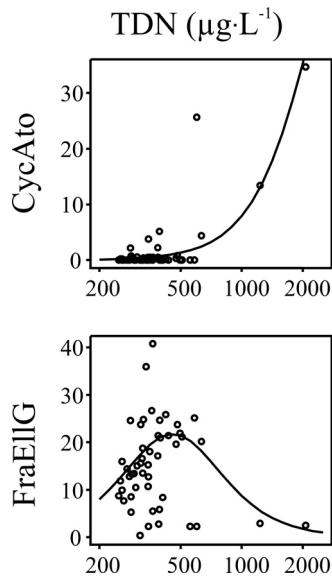


Figure 8. Plots of species % percentage abundance against annual mean TDN concentration for the calibration data set of 49 surface sediment samples. *Cyclotella atomus* shows a truncated response curve to TDN, while *Fragilaria elliptica* agg. exhibits a broad range of tolerance to TDN.

The TN values at Fasarbyviken were moderately underestimated by the model, whereas at Hellänlahti, the measured and diatom-inferred values compare quite well both before and after the increased concentrations in the mid 1980s (Paper IV). The monitoring data at both sites displayed peaks of high ($> 1000 \mu\text{g l}^{-1}$) TN concentrations, which were not recorded in the DI-TDN. These peaks, were, however only short-lived (1-3 years), and may have been smoothed out in the diatom-inferred TDN reconstruction. The explanation for the underestimated N values at Fasarbyviken may be the dominance of benthic *Fragilaria* taxa throughout this core. *Fragilaria* spp. have a wide range of ecological tolerance, which make them poor indicator taxa, and their optima for a given variable tend to lie close to the centre of the covered environmental gradient, influencing the performance of the transfer model (see Bennion et al. 2001). At Pieni Pernajanlahti, which has no continuous monitoring record, the DI-TDN value for the surface sample (387 $\mu\text{g l}^{-1}$) was in

good agreement with the measured annual mean TDN value ($398 \mu\text{g l}^{-1}$), which adds confidence in the reconstruction of the site.

Only Töölönlahti appeared to have many fossil samples with poor modern analogues in the training set (Paper IV, Fig. 3). This lack of modern analogues especially during the most pronounced eutrophication period results, in part, from the high abundance of *Thalassiosira guillardii*, a taxon that only occurred at low abundances in the training set, and from the rarity of extremely eutrophied sites in the training set (Paper III, pp. 326-327). Many of these samples also had a poor fit to TDN. This would indicate that the TDN reconstruction for Töölönlahti should be interpreted with some caution. Laajalahti showed good analogues with the modern training set throughout the core with only one sample having a poor fit to TDN (Paper IV, Figure 3). This is somewhat surprising, as the monitoring record from Laajalahti indicates concentrations as high as ca. $5000 \mu\text{g l}^{-1}$, whereas the training set only includes sites $< 2000 \mu\text{g l}^{-1}$. This would suggest that the diatom assemblages of Laajalahti have not responded to these anomalously high nutrient concentrations occurring in the embayment in the 1960s and 1970s, which is supported by the clearly underestimated TDN-reconstruction. The fossil samples of the rural sites generally had good modern analogues in the training set. Samples with poor analogues usually had a high abundance of taxa, which were rare in the training set, e.g. *Cocconeis placentula* at Hellänlahti and *Diatoma tenuis* at Pieni Pernajanlahti.

4.2.4. The multi-proxy approach in a coastal environment

Combining quantitative inferences of nutrient concentrations from diatom-based transfer functions with geochemical and biological records preserved in long sediment cores allows us to provide environmental reconstructions based on both empirical modelling and the ecological information contained in a range of geochemical and biological proxies. By observing changes in a number of different indicators, it is possible to obtain a more complete picture of ecosystem response to the changing environment and reduce uncertainty in interpretation.

The urban embayment Laajalahti was chosen for a detailed multi-proxy study. It was analysed for the stratigraphy of diatoms, sediment geochemistry, stable isotopes and sedimentary pigments (Paper V). The results of these analyses were compared with long-term monitoring results of water quality, nutrient loading and known land-use changes in the catchment area. Five phases were distinguished in the eutrophication history of Laajalahti: 1) the pre-industrial phase (before ca. 1815), 2) the phase of slight human disturbance (ca. 1815-1900), 3) the onset of anthropogenic impact (ca. 1900-1955), 4) the severe pollution phase (ca. 1955-1975), and 5) basin recovery and the phase of internal loading (from ca. 1975 onwards) (Paper V, Fig. 8). These phases were based on a principal components analysis (PCA) including diatom indices and the geochemical variables BSi, TN, TC, TP, OP and Cu:Zn ratio.

During phase 1, sedimentary nutrient levels in Laajalahti were quite stable, but showed a slow increase during phase 2, which was interpreted as an indication of slight nutrient enrichment. This could be attributed to post-glacial isostatic land uplift, as embayments become more enriched by nutrients during their isolation process, which was also supported by changes in the grain size of the sediment and in the diatom community structure, where small *Fragilaria* taxa dominated the assemblages. Annual average concentrations of DI-TDN were ca. 400 $\mu\text{g l}^{-1}$. The predominance of benthic over planktonic taxa and the overall diverse diatom assemblages suggest undisturbed conditions in the embayment during both periods, although the abundance of planktonic diatom taxa started to increase slowly after the 1820s, indicating an increase in anthropogenic disturbance.

Phase 3 was characterised by a slow rise in the population and the lack of treatment of wastewaters (Laakkonen & Lehtonen 1999). Wastewater loading spread to a wider area, as the network of wastewater pipelines extended to the western part of Helsinki. The nutrient loads into the embayment were estimated to be similar to the loads in 1986, which was the last year the municipal treatment plant was operational. The increase in nutrient enrichment, indicated by DI-TDN and sedimentary $\delta^{15}\text{N}$, and the subsequent increase in organic production (measured as organic phosphorus (OP)) resulted in deterioration of oxygen conditions near the bottom as indicated by the sedimentary copper: zinc (Cu:Zn) ratio. Nutrient enrichment also changed the biotic community structure; planktonic diatom taxa increased markedly while species richness decreased relative to its natural level. Increased chlorophyll *a* + its degradation products (Chl*a*D) concentrations indicated a general increase in algal biomass.

Phase 4 was characterised by heavy wastewater loading and a clear start of the recovery due to purification activities in the treatment plant at the end of this phase. The nutrient levels of the estuary during this phase were highly elevated despite the decline of nutrient concentrations at the start of the 1970s (Paper V, Fig. 2). Changes in the biotic community structure reflected changes in nutrient levels. The abundance of *Cyclotella atomus* and *C. meneghiniana*, both indicators of eutrophication, increased during this stage. The increased dominance of planktonic taxa suggests significant nutrient enrichment evidenced by simultaneous peaks of nutrients (sedimentary TN, OP and DI-TDN) and pigments (Chl*a*D and diatoxanthin). Additionally, a clear loss of species richness was observed. The peak of the Cu:Zn ratio most likely indicates anoxia, which could be verified by the monitoring data showing low oxygen concentrations in the near-bottom water layer in the early 1970s.

Phase 5, starting from ca. 1975, was characterised by substantial internal loading and the termination of external wastewater loading in 1986. The decline of DI-TDN and OP since the late 1980s reflected the closure of the wastewater treatment plant. However, nutrient levels in the water column have not decreased since the 1990s, which is explained by internal loading from the sediments. The clear increase in OP in the reactive sediment surface (top 3 cm) is mainly a consequence of remineralisation and decomposition of fresh organic material. Contrary to

expectations, the stable isotope $\delta^{15}\text{N}$ concentrations did not decrease after termination of external waste water loading. The further increasing concentrations after the closure of the waste water treatment plant could be explained by increased denitrification activity.

Based on present measured nutrient concentrations, productivity and the community structure of the biota, Laajalahti is still eutrophic. This is indicated by declining diatom species richness since the late 1980s, the continuing dominance of planktonic taxa in the diatom assemblages, frequent cyanobacterial blooms and high abundance of green algae (Pesonen et al. 1995).

There was generally a good agreement between the sedimentary record and the monitoring data. The steep decline in DI-TDN in the mid 1970s could be validated by the data on N loading and concentrations in the water (Paper V, Fig. 2, 6). Sedimentary OP as a rough measure of organic production showed a decreasing trend from the 1970s towards the top of the core. Similar decreases could also be seen in the monitoring data of P. The decline of DI-TDN and OP since the late 1980s reflected the closure of the wastewater treatment plant. Despite the lack of data before 1968, the wastewater loading most probably peaked at the end of the 1960s, which was estimated on the basis of the length of the wastewater pipeline and the population density of Helsinki (see Laakkonen & Lehtonen 1999).

Some sediment proxies did not follow trends in monitoring data or known changes in land use. This was especially the case with the sediment records of TC, TN and TP, as their concentrations increased towards the top of the sediment core. However other geochemical proxies reliably reflected changes in the monitoring data. A detailed discussion on problems and potential of the geochemical proxies will be presented in the doctoral thesis of Sanna Vaalgamaa, and hence no further discussion will follow here with the brief exception of biogenic silica. Concentrations of biogenic silica (BSi) in the sediment reflect diatom production of aquatic systems (Conley & Schelske 2001). Surprisingly no significant increase in BSi concentrations could be observed in Laajalahti during the peak nutrient loading at the end of the 1960s (Paper V, Fig. 6). This contrasts with data from the open sea areas of the Baltic, where large increases in down-core BSi have occurred during the last ca. 50 years attributed to eutrophication (A. Clarke, pers. comm.). The results from Laajalahti may be explained by silicate limitation of diatoms and competition with other phytoplankton groups such as cyanobacteria, green algae and flagellates, which might have responded more strongly to the highly elevated nutrient concentrations.

By using several proxies for tracing the eutrophication history of Laajalahti, it was possible to gain insight into longer-term changes in nutrient concentrations (DI-TDN, OP), general productivity of the system (OP, *Chl a*D), diatom productivity (diatoxanthin), diversity patterns (diatom species richness) and bottom water anoxia (Cu:Zn ratio). The diatom indices (DI-TDN, species richness, and % planktonic diatoms) employed here proved to be good indicators of changes in the trophic status when validated against the existing monitoring data, and hence could be applied alone in similar studies. By analysing other proxies in addition to diatoms, the story becomes more complete. According to this study, proxies best suited for defining the trophic

development of coastal sites of a similar setting included diatom indices, sedimentary pigments and organic phosphorus.

5. Use of diatom indices in coastal management of the Baltic Sea

Diatoms are increasingly being used in the management of freshwaters (e.g. Dixit et al. 1992; Battarbee 1994; Dixit & Smol 1994; Kelly & Whitton 1995; Kelly et al. 1998; Passy & Bode 2004). The situation is, however, very different in coastal monitoring and management. For example, diatoms are presently not part of the quality elements of the Water Framework Directive in coastal waters. Although they are indirectly included in the WFD as part of the phytoplankton, many diatom taxa can not be identified to species level in live plankton counts, which hampers the accuracy of ecological interpretations. This study suggests that surface-sediment diatom assemblages could provide an additional and useful means to describe and monitor the state of coastal waters. Although the changes in the diatom assemblages along the sampled nutrient gradients were gradual, there were a number of taxa that were more abundant in a particular nutrient environment. These taxa could be used as potential indicators of the quality of coastal waters in the Gulf of Finland. Taxa such as *Rhoicosphenia curvata*, *Staurosirella pinnata*, *Opephora mutabilis*, *Navicula phyllepta*, *Bacillaria paxillifer* and *Pauliella taeniata* preferred lower N concentrations ($< \text{ca. } 400 \mu\text{g l}^{-1}$), whereas taxa preferring low P environments were scarcer. Many *Fragilaria* spp. were most abundant in moderate nutrient concentrations. However, as other factors such as depth and morphology of coastal embayments also play an important role in the ecology of these taxa, their use as indicators of moderate nutrient concentrations may be problematic. High occurrences of *Fragilaria* spp. are frequently associated with environmental instability (e.g. Haworth 1976, Denys 1990). Thus, the dominance of these species could also be a first indication of increased anthropogenic disturbance. Small, centric, planktonic taxa such as *Cyclotella atomus*, *Cyclotella meneghiniana* and *Thalassiosira pseudonana* appear to be good indicators of highly elevated nutrient concentrations ($> \text{ca. } 600 \mu\text{g l}^{-1}$ TDN and $\text{ca. } 60 \mu\text{g l}^{-1}$ TP). Coastal systems, where these taxa are abundant, already have a poor water quality (Paper II). The first appearance of these taxa (and other small, centric species) indicates the start of biological change and could hence be used in regular monitoring as an early warning sign of deteriorating water quality (see Battarbee 1999). Compared to e.g. water chemistry monitoring, which requires numerous point measurements for assessing changes in nutrient concentrations over longer time scales, surface sediment diatom assemblages represent an integrated sample of all habitats over one to several years, which greatly economises routine monitoring. New techniques in ecology such as Bayesian Belief Networks (Trigg et al. 2000) could be applied to the classification of the ecological status of coastal waters *sensu* WFD using similar diatom data sets. Calibration data sets like these can also be used for biomonitoring schemes by calculating species nutrient optima using weighted-averaging based techniques (e.g. Schönfelder et al. 2002, DeNicola et al. 2004);

The abundance-weighted average of all species' optima gives a good estimate of the nutrient status of a site with statistically reliable errors of prediction.

The recent European Water Framework Directive (WFD) (Anon., 2000) requires all surface waters in Europe to fulfill the criterion of “good ecological status”, where the biological and chemical status departs only slightly from undisturbed baseline conditions, during the realisation period 2015-2027. These baseline conditions need to be determined in order to assess the extent of anthropogenic eutrophication and the present ecological status of a system (see Andersen et al. 2004). The long-term perspective gained from reliably dated sediment records can provide details of reference conditions, the timing and rate of ecosystem response to nutrient enrichment, and, in cases of sufficient temporal resolution, an indication of natural variability beyond that offered by most neo-ecological studies (e.g. Smol 1992; Anderson 1995; Hall & Smol 1999) or the most advanced dynamic model.

At four of our sites, significant changes in diatom assemblages and associated DI-TDN occurred after the 1920s, hence background or reference conditions could be set at early 1900s. The multi-proxy study from the urban site Laajalahti defined the mid to late 1800s as realistic reference conditions. At Töölönlahti, the length of the sediment core was not sufficient to establish pre-urbanisation conditions. Defining reference conditions at this site may be problematic as it requires recreating conditions that existed in the embayment centuries ago. Over this time scale other factors, such as post glacial land uplift, will also affect the physical and chemical environment and consequently the biota. Such a situation was observed at Laajalahti, where the diatom assemblage structure pre-1800 reflected a more open system compared to the present (Paper V). These examples emphasise the importance of taking natural variability of these systems into account in order to establish realistic reference conditions, which could be used as a basis for restoration schemes. The results suggest that reference conditions for management purposes could be set at the late 1800s in coastal waters of southern Finland.

The established reference conditions were defined by generally diverse benthic diatom assemblages that were characterised by small *Fragilaria* spp. Based on the multi-proxy study, the general productivity as indicated by sedimentary organic phosphorus (OP) and sedimentary pigments was low and bottom water oxygen conditions were good (Cu:Zn ratio). The diatom-inferred TDN, however, was surprisingly high (ca. 400 $\mu\text{g l}^{-1}$ TDN corresponding to ca. 600 $\mu\text{g l}^{-1}$ TN in the study area) (Table 4), which could be attributed to longer term (10^2 - 10^3 yr⁻¹) anthropogenic influence and higher natural leaching in catchment areas with clay-rich soils such as those in this study (Suomen kartasto 1990) (see paper IV for a more detailed discussion). Another reason for the higher than expected reference TDN concentrations could be the dominance of *Fragilaria* spp., as their optima for TDN lie close to the centre of the sampled TDN gradient of the calibration data set (mean ca. 420 $\mu\text{g l}^{-1}$ / median ca. 350 $\mu\text{g l}^{-1}$ in the present study). The established reference conditions seem to, however, represent relatively stable systems before recent, post-industrial anthropogenic eutrophication (Paper IV, Fig. 3, 5).

In addition to detailed studies of individual sites, diatom indices can be used for multi-

site regional assessment of coastal waters in the Baltic Sea area by applying them to the top 1-cm interval of sediment (representing present conditions) and to deeper sediment material (representing pre-disturbance conditions) to determine the present departure from the background nutrient status across a more extensive area (the “top-bottom” approach) (e.g. Cumming et al. 1992; Bennion et al. 2004). A prerequisite for this method is knowledge of the approximate sediment accumulation rates of the area to be sampled in order to estimate the required depth for the bottom sample. As sediment accumulation can vary considerably from site to site, and as it is time-consuming and expensive to date a large number of sites, this may be problematic. However, as studies on well-dated cores become more abundant, they may give a relatively reliable range of accumulation rates to be expected in these kinds of coastal embayments. As this kind of work is well in progress in the Baltic Sea area (undertaken in projects such as DETECT and DEFINE <http://www.helsinki.fi/bioscience/ecru/projects>), this sampling strategy could provide an effective management tool in the near future in coastal areas of the Baltic Sea with clear applications to the EU Water Framework Directive.

6. Conclusions

The 49 study sites scattered along the southern coastline of Finland exhibited a wide range of physical and chemical conditions reflecting patterns of different land-use types, river inputs and other local influences. All sampled embayments were generally small, sheltered, shallow, and characterised by relatively turbid water. There was a clear increasing trend in salinity from east to west due to the strong influence of the large River Neva estuary. Most studied sites fell into the category of moderate water quality according to the classification used by the Finnish Environment Institute (Antikainen et al. 2000). At present only ca. 5 % of all sites have good water quality. The mean values for nutrients and Chl-*a* did not follow the trend of increasing concentrations from west to east observed in the open sea (Pitkänen et al. 1987), but higher nutrient values were observed at sites with freshwater input from rivers or close to the urbanised areas emphasising the importance of local influence on the nutrient status of sheltered coastal embayments.

The main environmental gradients in the modern calibration set of diatoms and associated environmental data were represented by nutrients and nutrient-related variables (Chl-*a*, turbidity), depth and salinity. The nutrient gradients were strongly related to urban development (municipalities, industry) rather than agriculture. Although agriculture is an important factor influencing water quality, the proportion of agriculture in the catchment areas of this study was only low to moderate (mean 13 %). Of the key gradients, depth, TN and TP best explained the variation in the composition of diatom assemblages along the southern coast of Finland. Additionally salinity accounted for a significant and independent fraction of variation in the diatom data, but explained less of the total variation than the three other variables, which can be

explained by the relatively short salinity gradient in this study (< 6 ‰). Although the changes in diatom assemblage structures along the nutrient gradients were gradual, it was possible to identify clear response patterns. Taxa that were more abundant in a particular nutrient environment were also observed. Diatom assemblages that were least affected by nutrient enrichment included a variety of benthic species (e.g. *Rhoicosphenia curvata*, *Staurosirella pinnata*, *Opephora mutabilis* and *Bacillaria paxillifer*) and a relatively high species richness, whereas the most eutrophied sites were defined by a marked presence of small planktonic taxa such as *Cyclotella atomus*, *Cyclotella meneghiniana* and *Thalassiosira pseudonana* and a clearly reduced species richness. These taxa could be used as potential water quality indicators in coastal environments of the Baltic Sea together with indices such as species richness and the abundance of planktonic species.

The sediment record at five sites chosen for a detailed study of changes in trophic status indicated that the principal change in the diatom assemblages of all cores occurred in the proportions of diatom life forms. As nutrient enrichment increases turbidity, planktonic taxa are favoured over benthic forms. Since coastal planktonic diatom communities include far fewer taxa than benthic ones, a clear decrease in species richness can be observed at those sites showing a marked increase in planktonic taxa.

The quantitative transfer function approach showed both potential and problems when applied to these five sites in order to study post-industrial trends in nutrient concentrations. Overall, the diatom-TDN models performed reasonably well as shown by assessment using long-term monitoring data and statistical measures such as the modern analogue technique. The general trends in TDN concentrations were reconstructed reliably, but very high concentrations during heavy nutrient loading were systematically underestimated. Short-lived peaks of high TN concentrations in the monitoring data were also not recorded in the diatom-inferred TDN most likely due to the “smoothing” nature of sediment records. Important, however, is the ability of the model to relatively accurately infer nutrient concentrations, which are in the range of the sampled gradient of the used calibration data set.

The urban sites showed marked increases in the abundance of planktonic diatoms and diatom-inferred TDN, and a decrease in species richness starting in the 19th – early 20th century with increased urbanisation. At both sites a clear recovery was observed after the cessation of waste water loading by the mid 1980s. In contrast, the observed increase in the rural sites was only moderate and occurred later starting in the 1940s, which could be attributed to intensification of agriculture in Finland after the Second World War. No distinct increase in diatom-inferred TDN was seen at the rural sites, except at Hellänlahti. However, all sites showed an increase in small planktonic taxa. These floristic changes could be seen as an early warning signal despite the lack of change in the inferred nutrient concentrations.

By using the multi-proxy approach for tracing the eutrophication history of one urban site, Laajalahti, it was possible not only to assess longer-term changes in nutrient concentrations (DI-TDN) and diversity patterns (diatom species richness), but also trace changes in general

productivity (OP, Chl a D), diatom productivity (diatoxanthin), and bottom water anoxia (Cu: Zn ratio). Although diatom indices alone proved to be good indicators of trophic change at the studied site, the history of eutrophication became more comprehensive when additional proxies were analysed. Proxies that appear to be best suited for defining the trophic development of similar coastal sites in the Baltic Sea included diatom indices, sedimentary pigments and organic phosphorus.

Although a clear recovery at the urban sites was observed (expressed as a decrease in the abundance of planktonic taxa and DI-TDN, and as an increase in species richness), the presently plankton-dominated diatom assemblages of these two embayments show no change back to the pre-disturbance diverse benthic communities. Both urban embayments continue to be nutrient enriched due to internal loading (Tikkanen et al. 1997; Paper V), which supports high productivity and high turbidity, and favours planktonic diatom assemblages. This suggests that decreased external loading can be counteracted by internal loading from a nutrient pool accumulated in the sediments during the period of high external loading. Hence, after the initial improvement following reduced nutrient loads, no further improvement occurs. This phytoplankton-dominated state may be preserved despite nutrient reductions due to the inability of aquatic plants to recolonise in highly turbid waters.

The present study shows that in urban estuaries major changes in the nutrient status and species assemblages have taken place and that these systems are still far from good ecological status. On the other hand, the results indicate that in rural embayments the effects of eutrophication on the biota have perhaps been less pronounced than expected. Although these sites have crossed an ecologically important threshold, the changes are only small to moderate. At such sites management efforts may be well rewarded: internal loading from sediments would likely be a minor problem, as external loading has not been heavy (cf. the urban sites) and hence no large nutrient pool has accumulated in the sediments.

The results presented here show that palaeolimnological methods, which are widely used in fresh waters, can also be successfully applied to the more complex coastal environments in the Baltic Sea area for reconstructing long-term (> 100 yr) changes in coastal ecosystems affected by eutrophication. The diatom indices used in this study (DI-TDN, species richness, and % planktonic diatoms) proved to be good indicators of changes in the trophic status when validated against the existing monitoring data and known land use changes. Whether the quantitative transfer function approach is applicable to coastal areas outside the Baltic requires further study. These methods, however, will not be applicable everywhere, but in suitable environments of constant sediment accumulation. Palaeolimnological techniques show clear applicability within the EU Water Framework Directive for defining trophic reference conditions as well as for current water quality monitoring in coastal waters of the Baltic Sea area.

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Appendix 1. Code, location, environmental characteristics and catchment land use types of the 49 study sites. C-area = catchment area, Perim = perimeter, Trans = water transparency, Temp = temperature, Cond = conductivity, Sal = salinity, Alk = alkalinity, TDN = total dissolved nitrogen, TP = total phosphorus, TDP = total dissolved phosphorus, DIP = dissolved inorganic phosphorus, DSI = dissolved silica, Chl *a* = chlorophyll *a*, Uden = densely built-up urban area, Usps = sparsely built-up urban area, Ind = industrial area, Agr = agriculture, For = forest

Site number	Lon (°E)	Lat (°N)	C-Area (km ²)	Area (km ²)	Perim km	Depth (m)	Trans (m)	Temp (°C)	Cond (mS/cm)	Sal (per mil)	pH (units)
1	27.77	60.58	386	1.95	10.1	2.00	0.77	11.5	6.9	3.7	7.9
2	27.62	60.52	121	2.62	10.9	2.00	1.15	12.1	6.9	3.7	7.3
3	27.57	60.50	57	2.40	11.7	2.20	2.10	11.2	7.7	4.2	8.3
4	27.37	60.50	16	0.61	5.1	2.00	1.53	12.4	7.8	4.2	7.9
5	26.78	60.48	17	1.76	6.6	0.70	0.46	12.9	7.5	4.1	8.0
6	26.65	60.43	37061	7.41	17.1	3.60	1.50	12.4	4.5	2.3	7.9
7	26.52	60.37	0	0.40	3.3	3.00	1.98	13.4	3.2	1.5	7.9
8	26.42	60.38	1	1.57	8.8	3.20	1.25	13.9	7.9	4.3	8.4
9	26.43	60.37	2	0.87	5.0	5.20	2.35	11.8	7.4	4.0	8.4
10	26.18	60.40	49	4.46	15.1	2.55	1.12	12.3	8.9	4.8	8.0
11	26.20	60.38	3	0.59	6.2	2.10	2.03	13.6	8.7	4.8	8.3
12	25.98	60.37	18	1.29	7.5	2.80	1.00	13.9	8.7	4.8	8.2
13	26.02	60.30	9	3.88	14.2	4.15	1.56	11.7	9.0	4.9	8.3
14	26.00	60.28	1	0.48	4.8	4.30	2.54	11.1	8.9	4.9	8.3
15	25.90	60.38	356	9.06	29.3	4.60	1.18	13.0	8.1	4.4	8.1
16	25.78	60.30	36	5.59	19.2	4.45	1.41	13.1	8.8	4.8	7.9
17	25.78	60.30	5	0.95	6.1	3.60	1.50	11.8	9.7	5.3	8.4
18	25.52	60.28	4	0.39	3.1	2.40	1.02	11.8	9.8	5.4	8.3
19	25.48	60.25	1	0.21	3.5	2.80	1.39	12.9	9.5	5.3	8.5
20	25.33	60.27	224	0.62	6.1	6.10	0.91	12.2	9.4	5.2	8.3
21	25.25	60.25	29	1.02	8.1	4.90	1.08	12.0	9.8	5.5	8.4
22	25.15	60.20	2	0.90	3.6	2.90	1.73	10.1	9.9	5.4	8.4

23	25.02	60.20	1713	4.80	18.0	1.20	0.33	12.8	3.7	1.8	8.0
24	24.87	60.18	52	5.13	12.9	3.20	0.95	12.7	9.7	5.4	8.1
25	24.58	60.10	19	1.17	7.4	2.00	0.68	13.2	9.9	5.5	7.8
26	25.10	60.20	27	2.89	13.2	4.00	1.18	12.7	9.7	5.4	8.2
27	24.60	60.07	2	0.59	5.0	5.85	1.49	12.3	10.3	5.7	8.3
28	24.47	59.98	5	0.74	5.8	6.45	2.34	11.2	10.2	5.6	8.4
29	24.47	60.05	95	1.60	8.1	3.40	1.46	11.3	10.5	5.8	8.2
30	24.13	60.03	5	1.17	8.3	6.05	1.99	11.3	10.7	6.0	8.2
31	24.08	60.03	4	0.87	7.7	4.40	1.40	12.8	10.6	5.9	8.1
32	23.78	59.98	2	1.42	5.4	2.20	1.35	14.1	10.2	5.7	8.1
33	23.52	59.98	46	5.21	17.5	2.60	1.15	13.9	5.7	3.0	7.7
34	23.53	60.08	2216	3.59	50.6	8.50	2.44	13.2	1.8	0.7	7.6
35	23.20	60.03	33	5.97	24.6	5.40	2.18	13.5	11.0	6.2	8.1
36	23.70	59.95	1	1.27	5.8	7.50	2.11	13.2	10.7	6.0	8.3
37	23.65	59.92	1	0.35	4.0	3.45	1.53	13.5	10.9	6.1	8.2
38	23.53	59.92	1	1.03	9.8	3.05	1.38	13.6	10.6	5.9	8.2
39	23.12	60.02	4	1.24	6.3	22.00	3.51	13.0	9.2	5.1	8.1
40	22.93	60.08	7	0.65	6.1	3.20	1.39	13.5	9.2	5.1	8.2
41	23.05	60.12	7	0.88	5.8	5.30	1.16	13.0	9.5	5.3	8.0
42	23.23	59.85	1	0.38	7.5	2.35	1.05	13.0	10.3	5.8	8.2
43	23.20	59.92	2	0.57	4.2	7.30	3.20	13.0	11.1	6.2	8.1
44	22.98	59.92	3	2.63	11.2	9.20	3.42	13.7	11.3	6.4	8.2
45	22.97	59.93	10	1.63	6.4	1.60	1.22	14.3	11.4	6.4	8.3
46	60.18	24.93	5	0.21	2.1	2.15	0.70	9.2	8.8	4.8	7.9
47	60.33	25.60	1	0.76	4.1	1.70	0.45	8.4	8.0	4.3	8.0
48	60.53	27.02	76	3.66	14.6	2.80	1.80	9.0	7.1	3.8	7.9
49	60.53	27.14	7	0.81	7.8	2.00	1.40	9.2	7.8	4.2	8.2
Mean			872	2.05	9.9	4.09	1.53	12.4	8.8	4.8	8.1
Median			7	1.17	7.5	3.20	1.40	12.8	9.2	5.1	8.2
Min			0	0.21	2.1	0.70	0.33	8.4	1.8	0.7	7.3
Max			37061	9.06	50.6	22.00	3.51	14.3	11.4	6.4	8.5

Site number	Alk (mmol/l)	TDN (µg/l)	NH4 (µg/l)	NO3+NO2 (µg/l)	TP (µg/l)	TDP (µg/l)	DIP (µg/l)	DSi (mg/l)	Chl a (µg/l)	Na (g/l)	Mg (mg/l)
1	1.06	385	23	35	48	16	8	0.73	17	0.83	109
2	1.00	498	54	48	25	6	2	1.15	5	0.72	101
3	1.16	325	13	17	29	14	6	0.43	13	1.00	133
4	1.08	360	20	19	32	11	3	0.53	8	0.97	127
5	1.02	435	82	22	25	9	2	1.02	4	0.96	122
6	0.64	348	11	27	29	14	6	0.41	7	0.48	66
7	0.79	395	13	42	24	11	3	0.48	12	0.65	87
8	1.20	345	8	12	32	16	10	0.35	11	1.17	147
9	1.23	273	6	10	33	19	10	0.26	8	1.27	163
10	1.24	328	15	11	41	18	11	0.42	14	1.30	159
11	1.32	338	16	6	28	17	7	0.25	3	1.28	159
12	1.21	365	17	11	39	14	7	0.55	13	1.20	148
13	1.34	258	8	11	36	20	13	0.32	7	1.38	176
14	1.35	345	6	10	34	21	15	0.26	6	1.45	174
15	1.26	398	21	9	45	17	6	0.57	17	1.27	158
16	1.21	388	63	8	21	7	3	0.75	11	1.31	158
17	1.39	318	6	7	40	23	11	0.43	10	1.48	183
18	1.38	283	17	10	41	20	9	0.35	11	1.53	186
19	1.30	353	12	12	40	16	9	0.48	12	1.42	179
20	1.31	348	18	11	47	21	13	0.53	8	1.32	171
21	1.36	253	8	5	41	20	11	0.30	11	1.56	189
22	1.39	263	14	7	35	24	17	0.24	4	1.62	197
23	1.02	1230	59	202	86	16	5	2.25	42	0.38	66
24	1.49	393	30	15	63	22	11	0.66	14	1.51	181
25	1.45	423	74	11	46	16	6	0.53	10	1.57	190

26	1.45	285	102	10	49	19	10	0.40	10	1.60	195
27	1.39	300	9	6	39	17	12	0.30	10	1.69	204
28	1.41	248	8	7	33	19	11	0.27	6	1.69	202
29	1.38	288	12	8	32	14	7	0.42	6	1.67	199
30	1.43	280	37	6	36	27	18	0.28	4	1.81	209
31	1.53	295	21	6	48	31	18	0.35	9	1.80	210
32	1.35	585	22	14	35	15	3	0.37	11	1.41	170
33	1.01	633	157	34	38	12	4	1.25	20	0.68	93
34	0.72	558	100	90	25	10	4	0.74	5	0.28	33
35	1.41	408	13	12	20	12	3	0.64	4	1.76	203
36	1.44	285	5	5	31	14	8	0.32	8	1.77	206
37	1.45	305	9	5	41	16	6	0.36	9	1.82	211
38	1.49	325	11	11	39	17	9	0.41	11	1.74	205
39	1.21	315	118	19	15	8	2	0.55	4	1.36	170
40	1.32	365	9	11	22	10	4	0.49	5	1.50	184
41	1.22	390	5	28	25	13	4	0.64	7	1.41	157
42	1.49	323	19	5	39	17	9	0.34	10	1.64	195
43	1.35	258	8	8	18	13	6	0.43	3	1.59	190
44	1.38	318	7	5	20	9	8	0.42	3	1.80	209
45	1.43	508	46	11	36	12	4	0.35	6	1.79	206
46		600	49	31	68	23	14	1.02	46		
47		2068	1567	36	81	21	11	1.17	152		
48		474	13	22	31	16	11	0.87	9		
49		483	22	12	32	18	10	0.63	10		
Mean	1.27	419	61	20	37	16	8	0.56	13	1.34	164
Median	1.34	348	16	11	35	16	8	0.43	9	1.42	176
Min	0.64	248	5	5	15	6	2	0.24	3	0.28	33
Max	1.53	2068	1567	202	86	31	18	2.25	152	1.82	211

Site number	K (mg/l)	Ca (mg/l)	Cl (g/l)	SO4 (mg/l)	Fe (µg/l)	Mn (µg/l)	Uden	Uspa	Ind %	Agr	For
1	31	43	1.53	242	198	32	0.1	1.4	2.9	16.0	79.7
2	27	41	1.25	234	138	117	0.0	1.3	3.0	19.4	76.2
3	37	52	1.83	295	105	21	0.0	1.4	3.0	22.4	73.1
4	37	50	1.75	279	68	33	0.0	3.2	2.5	11.6	82.7
5	35	49	1.65	274	75	88	0.1	5.0	5.0	29.4	60.6
6	19	28	0.88	136	85	23	0.1	2.0	3.4	9.5	84.9
7	26	35	1.21	192	63	22	0.0	0.0	1.8	0.0	98.2
8	41	57	2.12	327	45	40	0.2	3.7	5.4	4.8	86.0
9	45	61	2.33	346	94	23	0.0	3.1	3.6	2.4	90.9
10	47	61	2.03	333	75	65	0.0	1.0	3.1	17.4	78.5
11	46	61	2.37	353	30	7	0.0	0.7	2.8	3.7	92.8
12	43	58	2.24	337	109	28	0.0	4.2	3.2	23.7	69.0
13	49	62	2.63	392	40	20	0.0	4.9	4.3	27.1	63.7
14	52	71	2.14	381	30	25	0.0	14.0	5.9	9.7	70.5
15	46	62	2.29	359	138	45	0.0	3.0	3.4	27.1	66.4
16	46	65	2.31	357	48	70	0.0	4.2	5.8	32.6	57.4
17	52	69	2.29	399	78	17	0.0	3.6	2.7	5.3	88.5
18	55	72	2.86	415	78	20	0.0	2.9	11.6	2.8	82.6
19	51	67	2.63	399	68	16	0.0	27.1	10.2	2.3	60.4
20	47	65	2.37	388	146	35	0.3	6.5	4.8	33.7	54.8
21	56	66	2.92	422	83	50	0.1	6.1	3.9	7.3	82.6
22	58	68	3.01	436	33	9	24.0	11.9	18.8	1.3	44.1
23	16	34	0.62	162	295	50	1.9	7.2	7.5	26.2	57.2
24	54	71	2.90	414	43	6	20.5	17.8	17.9	5.3	38.4
25	57	73	2.99	417	150	20	0.1	8.2	3.3	25.3	63.1

26	58	74	3.16	424	93	8	24.1	18.0	17.0	7.2	33.7
27	61	76	2.54	430	223	8	0.0	2.6	3.3	0.2	93.9
28	61	76	2.55	440	88	12	0.1	7.8	5.4	4.8	81.9
29	62	74	3.25	454	63	27	1.3	8.7	4.9	26.0	59.0
30	65	77	3.15	472	121	35	0.0	3.3	3.2	5.6	87.9
31	64	79	2.44	447	84	11	0.1	4.0	3.7	4.3	88.0
32	52	64	2.64	367	40	25	0.0	7.4	5.9	7.2	79.5
33	26	40	1.28	217	78	27	0.4	3.6	4.8	28.7	62.4
34	8	20	0.35	87	125	20	0.2	4.6	4.1	21.0	70.1
35	64	79	3.58	462	75	15	0.0	2.2	3.4	34.2	60.2
36	64	76	3.07	465	63	7	0.0	8.8	5.0	5.2	81.0
37	67	78	3.85	490	55	7	0.0	8.2	3.9	8.6	79.3
38	63	77	3.29	469	75	10	0.0	5.9	7.0	0.6	86.6
39	50	65	2.83	392	53	30	0.0	1.6	2.6	0.5	95.3
40	55	70	2.87	386	58	12	0.1	4.3	3.9	23.3	68.5
41	53	59	2.91	353	146	17	0.0	2.6	3.4	15.5	78.6
42	59	78	3.29	452	75	34	0.0	3.0	4.3	10.4	82.3
43	58	71	2.63	399	70	8	0.0	0.8	4.0	3.9	91.3
44	64	78	3.99	465	20	2	0.0	7.0	4.1	3.3	85.6
45	64	79	3.14	393	50	17	0.0	1.2	2.9	12.9	83.0
46							33.3	16.7	30.8	0.8	18.5
47							0.6	29.9	13.1	3.8	52.6
48							0.1	3.5	5.2	15.7	75.5
49							1.2	13.2	16.0	5.3	64.3
Mean	49	63	2.44	363	88	27	2.2	6.4	6.2	12.6	72.7
Median	52	66	2.55	392	75	21	0.0	4.2	4.1	8.6	78.5
Min	8	20	0.35	87	20	2	0.0	0.0	1.8	0.0	18.5
Max	67	79	3.99	490	295	117	33.3	29.9	30.8	34.2	98.2

Appendix 2. Taxon codes, names and authority, number of occurrence, Hill's N2 values and maximum abundance of all taxa encountered in this study (excluding spp.). The use of aggregate taxa follows the taxonomic conventions of the MOLTEN project (MOLTEN 2004).

Taxon code	Taxon name and authority	N. occur.	Hill's N2	Max. abund.
AchAcaC	<i>Achnanthes</i> cf. <i>acares</i> Hohn & Hellerman	15	12.6	0.5
AchAmo	<i>Achnanthes amoena</i> Hustedt 1952	1	1.0	0.2
AchBre	<i>Achnanthes brevipes</i> C.A. Agardh 1824	1	1.0	0.2
AchBre2	<i>Achnanthes brevipes</i> var. <i>intermedia</i> (Kützing) Cleve 1895	3	3.0	0.2
AchKri	<i>Achnanthes kriegei</i> Krasske 1943	4	2.7	7.8
AchLem	<i>Achnanthes lemmermannii</i> Hustedt 1933	36	20.9	2.1
AchMinuC	<i>Achnanthes</i> cf. <i>minuscula</i> Hustedt 1945	9	6.2	0.7
AchNit	<i>Achnanthes nitidiformis</i> Lange-Bertalot	1	1.0	0.3
AchObl	<i>Achnanthes oblongella</i> Østrup 1902	6	3.1	1.3
AchPar	<i>Achnanthes parvula</i> Kützing 1844	2	1.8	0.4
AchPer	<i>Achnanthes pericava</i> Carter 1966	6	6.0	0.2
AchSubm	<i>Achnanthes submarina</i> Hustedt 1956	4	2.1	4.2
AchSubs	<i>Achnanthes subsalsa</i> Petersen 1928	1	1.0	0.2
AchSuc	<i>Achnanthes suchlandtii</i> Hustedt 1933	1	1.0	0.2
AcnMinG	<i>Achnanthidium minutissimum</i> agg.	39	9.0	12.5
ActNor	<i>Actinocyclus normanii</i> (Gregory) Hustedt 1957	1	1.0	0.2
ActOct	<i>Actinocyclus octonarius</i> Ehrenberg 1837	1	1.0	0.2
ActOct2	<i>Actinocyclus octonarius</i> var. <i>crassus</i> (W. Smith) Hendey 1954	12	6.7	1.6
AmpAcu	<i>Amphora acutiuscula</i> Kützing 1844	13	6.3	1.4
AmpAeq	<i>Amphora aequalis</i> Krammer 1980	6	5.4	0.3
AmpCof	<i>Amphora coffeaeformis</i> (C.A. Agardh) Kützing 1844	12	10.0	0.5
AmpCom	<i>Amphora commutata</i> Grunow in Van Heurck 1880	1	1.0	0.2

AmpCop	<i>Amphora copulata</i> (Kützing) Schoeman & Archibald 1986	33	18.9	2.8
AmpExiC	<i>Amphora</i> cf. <i>exigua</i> Gregory 1857	4	3.5	0.4
AmpFog	<i>Amphora fagediana</i> Krammer 1995	1	1.0	0.2
AmpHol	<i>Amphora holsatica</i> Hustedt 1925	9	8.0	0.4
Amplna	<i>Amphora inariensis</i> Krammer 1980	1	1.0	0.2
AmplIn	<i>Amphora lineolata</i> Ehrenberg 1838	19	8.2	3.3
AmpOva	<i>Amphora ovalis</i> (Kützing) Kützing 1844	11	5.8	1.2
AmpPed	<i>Amphora pediculus</i> (Kützing) Grunow in A. Schmidt et al. 1875	46	24.2	6.6
AmpVen	<i>Amphora veneta</i> Kützing 1844	3	2.3	0.5
AneMin	<i>Aneumastus minor</i> (Hustedt) Lange-Bertalot 1993	4	4.0	0.2
AstFor	<i>Asterionella formosa</i> Hassall 1850	4	2.3	3.7
AulAmb	<i>Aulacoseira ambigua</i> (Grunow) Simonsen 1979	9	7.5	1.9
AulDis	<i>Aulacoseira distans</i> (Ehrenberg) Simonsen 1979	2	2.0	0.2
AulGra	<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen 1979	4	3.6	0.3
AullIsl	<i>Aulacoseira islandica</i> (O. Müller) Simonsen 1979	2	2.0	0.2
Aullac	<i>Aulacoseira lacustris</i> (Grunow) Krammer 1990	1	1.0	1.1
Aullir	<i>Aulacoseira lirata</i> (Ehrenberg) Ross 1986	1	1.0	0.2
AulSub	<i>Aulacoseira subarcfica</i> (O. Müller) Haworth 1988	12	2.6	5.9
BacPax	<i>Bacillaria paxillifer</i> (O.F. Müller) Hendey 1951	39	24.0	2.4
BerFen	<i>Berkeleya fennica</i> Juhlin-Dannfelt 1882	8	7.1	0.4
BerRut	<i>Berkeleya rutilans</i> (Trentepohli) Grunow 1880	12	10.7	0.4
BraBre	<i>Brachysira brebissonii</i> R. Ross in B. Hartley 1986	7	2.1	6.4
BraVit	<i>Brachysira vitrea</i> (Grunow) R. Ross in B. Hartley 1986	13	8.4	1.3
BreLan	<i>Brebissonia lanceolata</i> (C.A. Agardh) Mahoney & Reimer 1984	1	1.0	0.2
CalAmp	<i>Caloneis amphisbaena</i> (Bory) Cleve 1894	2	2.0	0.2
CalAmp2	<i>Caloneis amphisbaena</i> f. <i>subsalina</i> (Donkin) Van der Werff & Huls (1960)	3	2.7	0.3
CalBac	<i>Caloneis bacillum</i> (Grunow) Cleve 1894	9	7.7	0.5
CalPer	<i>Caloneis permagna</i> (Bailey) Cleve 1894	2	2.0	0.2
CalSil	<i>Caloneis silicula</i> (Ehrenberg) Cleve 1894	6	6.0	0.2
CalWes	<i>Caloneis westii</i> (W. Smith) Hendey 1964	6	5.6	0.4

CamBic	<i>Campylodiscus bicostatus</i> Smith in Roper 1854	1	1.0	0.2
CamCly	<i>Campylodiscus clypeus</i> Ehrenberg 1840	6	4.7	0.5
CamEch	<i>Campylodiscus echeneis</i> Ehrenberg 1840	7	5.6	0.5
CatAdh	<i>Catenula adhaerens</i> (Mereschkowsky) Mereschkowsky 1902-1903	12	7.4	1.2
CavCoc	<i>Cavinula cocconeiformis</i> (Gregory) D.G. Mann & A.J. Stickle in Round et al. 1990	5	4.6	0.3
CavPse	<i>Cavinula pseudoscutiformis</i> (Hustedt) D.G. Mann & A.J. Stickle in Round et al. 1990	2	1.6	0.5
ChaSpp	<i>Chaetoceros</i> spp. Ehrenberg 1844	26	14.7	3.7
CocDis	<i>Cocconeis disculus</i> (Schumann) Cleve in Cleve & Jentzsch 1882	10	6.9	0.7
CocHofC	<i>Cocconeis</i> cf. <i>hoffmannii</i> Simonsen	3	3.0	0.2
CocNeod	<i>Cocconeis neodiminuta</i> Krammer 1991	4	4.0	0.2
CocNeot	<i>Cocconeis neothumensis</i> Krammer 1991	46	20.3	6.3
CocPed	<i>Cocconeis pediculus</i> Ehrenberg 1838	22	15.4	1.0
CocPla	<i>Cocconeis placentula</i> Ehrenberg 1838	47	25.7	3.6
CocScu	<i>Cocconeis scutellum</i> Ehrenberg 1833	15	10.9	0.9
CosAst	<i>Coccinodiscus asteromphalus</i> Ehrenberg 1844	11	8.3	0.7
CtePul	<i>Ctenophora pulchella</i> (Ralfs ex Kützing) Williams & Round 1986	44	33.0	1.1
CycoDub	<i>Cyclostephanos dubius</i> (Fricke) Round 1982	5	2.9	2.6
CycoInv	<i>Cyclostephanos invisitatus</i> (Hohn & Hellerman) Theriot, Stoermer & Håkansson 1987	1	1.0	0.7
CycAto	<i>Cyclotella atomus</i> Hustedt 1937	24	4.5	34.7
CycCho	<i>Cyclotella choctawhatcheeana</i> Prasad 1990	41	9.1	71.8
CycDis	<i>Cyclotella distinguenda</i> Hustedt 1927	1	1.0	0.2
CycMen	<i>Cyclotella meneghiniana</i> Kützing 1844	23	5.2	9.7
CycPse	<i>Cyclotella pseudostelligera</i> Hustedt 1939	9	5.0	1.9
CycRad	<i>Cyclotella radiosa</i> (Grunow) Lemmermann 1900	5	3.5	0.7
CycRos	<i>Cyclotella rossii</i> Håkansson 1990	7	3.0	1.3
CycSte	<i>Cyclotella stelligera</i> Cleve & Grunow in Van Heurck 1882	7	5.1	0.9
CympSol	<i>Cymatopleura solea</i> (Brébisson) W. Smith 1851	1	1.0	0.2
CymCis	<i>Cymbella cistula</i> (Ehrenberg) Kirchner 1878	2	2.0	0.2
CymDes	<i>Cymbella descripta</i> (Hustedt 1943) Krammer & Lange-Bertalot 1985	3	2.6	0.3
CymPus	<i>Cymbella pusilla</i> in A. Schmidt et al. 1875	4	2.6	1.2

DenKue	<i>Denticula kuetzingii</i> Grunow 1862	1	1.0	0.3
DenTen	<i>Denticula tenuis</i> Kützing 1844	5	2.2	1.3
DiaMon	<i>Diatoma moniliformis</i> Kützing 1833	46	10.9	35.8
DiaTen	<i>Diatoma tenuis</i> C.A. Agardh 1812	35	22.2	1.3
DiaVul	<i>Diatoma vulgare</i> Bory 1824	4	3.6	0.5
DipAlpC	<i>Diploneis</i> cf. <i>alpina</i> Meister 1912	1	1.0	0.2
DipDec2	<i>Diploneis decipiens</i> var. <i>parallela</i> A. Cleve 1915	6	3.1	2.1
DipDid	<i>Diploneis didyma</i> (Ehrenberg) Ehrenberg 1854	1	1.0	0.2
DipModC	<i>Diploneis</i> cf. <i>modica</i> Hustedt 1945	1	1.0	0.4
DipOcu	<i>Diploneis oculata</i> (Brébisson) Cleve 1894	1	1.0	0.5
DipSmi	<i>Diploneis smithii</i> (Brébisson) Cleve 1894	28	19.0	1.6
EllAre	<i>Ellerbeckia arenaria</i> (Moore) Crawford 1988	1	1.0	0.3
EncCae	<i>Encyonema caespitosum</i> Kützing 1849	4	4.0	0.2
EncHeb	<i>Encyonema hebridicum</i> Grunow ex Cleve	1	1.0	0.3
EncMin	<i>Encyonema minutum</i> (Hilse in Rabenhorst) D.G. Mann	1	1.0	0.2
EncPro	<i>Encyonema prostratum</i> (Berkeley) Kützing 1844	1	1.0	0.2
EncSil	<i>Encyonema silesiacum</i> (Bleisch) D.G. Mann in Round et al. 1990	3	2.3	0.5
EncsMic	<i>Encyonopsis microcephala</i> (Grunow) Krammer 1997	1	1.0	1.7
EntPal	<i>Entomoneis paludosa</i> (W. Smith) Reimer in Patrick & Reimer 1975	21	13.1	1.1
EntPse	<i>Entomoneis pseudoduplex</i> Osada & Kobayasi 1990	14	10.0	0.9
EpiAdn	<i>Epithemia adnata</i> (Kützing) Brébisson 1838	18	11.6	1.2
EpiSor	<i>Epithemia sores</i> Kützing 1844	37	20.5	3.0
EpiTur	<i>Epithemia turgida</i> (Ehrenberg) Kützing 1844	19	13.3	0.9
FalCle	<i>Fallacia clepsidroides</i> Witkowski 1994	14	10.4	0.9
FalCry	<i>Fallacia cryptolyra</i> (Brockmann) A.J. Stickle & D.G. Mann in Round et al. 1990	3	3.0	0.2
FalFlo	<i>Fallacia florinae</i> (Møller) Witkowski 1993	9	7.2	0.5
FalPse	<i>Fallacia pseudony</i> (Hustedt) D.G. Mann in Round et al. 1990	1	1.0	0.2
FalPyg	<i>Fallacia pygmaea</i> (Kützing) A.J. Stickle & D.G. Mann in Round et al. 1990	11	8.1	0.7
FalTen	<i>Fallacia tenera</i> (Hustedt) A.J. Stickle & D.G. Mann in Round et al. 1990	22	17.2	0.7
FraAmi	<i>Fragilaria amicornum</i> Witkowski & Lange-Bertalot 1993	45	20.5	21.0

FraCapu	<i>Fragilaria capucina</i> Desmazzières 1825	35	16.3	3.3
FraEllG	<i>Fragilaria elliptica</i> agg.	49	36.5	40.7
FraExi	<i>Fragilaria exigua</i> Grunow in Cleve & Möller 1878	30	6.7	21.4
FraFam	<i>Fragilaria famelica</i> (Kützing) Lange-Bertalot 1980	5	3.7	0.6
FraGed	<i>Fragilaria gedanensis</i> Witkowski 1993	31	20.5	2.3
FraPar	<i>Fragilaria parasitica</i> (W. Smith) Grunow in Van Heurck 1881	5	3.2	0.9
FrapCyl	<i>Fragilariopis cylindrus</i> (Grunow) Krieger in Helmcke & Krieger 1954	25	19.5	0.9
FraSubC	<i>Fragilaria cf. subsalina</i> (Grunow) Lange-Bertalot	19	7.3	2.8
FraVau	<i>Fragilaria vaucheriae</i> (Kützing) J.B. Petersen 1938	10	9.3	0.4
FraFbic	<i>Fragilariforma bicapitata</i> (Mayer) Williams & Round	1	1.0	0.2
FraVvir	<i>Fragilariforma virescens</i> (Ralfs) Williams & Round	4	4.0	0.2
FruRho2	<i>Frustulia rhomboides</i> var. <i>saxonica</i> (Rabenhorst) De Toni 1891	4	1.5	2.2
GomGra	<i>Gomphonema gracile</i> Ehrenberg 1838	4	4.0	0.2
GomAngu	<i>Gomphonema angustum</i> Agardh 1831	1	1.0	0.4
GomOli	<i>Gomphonema olivaceum</i> (Hornemann) Brébisson 1838	16	13.4	0.5
GomPar	<i>Gomphonema parvulum</i> (Kützing) Kützing 1849	15	9.8	0.9
GomTru	<i>Gomphonema truncatum</i> Ehrenberg 1832	1	1.0	0.2
GomVen	<i>Gomphonema ventricosum</i> Gregory 1856	1	1.0	0.2
GompPse	<i>Gomphonemopsis pseudexigua</i> (Simonsen) Medlin 1986	14	10.9	0.7
GyrAcu	<i>Gyrosigma acuminatum</i> (Kützing) Rabenhorst 1853	6	5.4	0.3
GyrAtt	<i>Gyrosigma attenuatum</i> (Kützing) Rabenhorst 1853	7	7.0	0.2
GyrDis	<i>Gyrosigma distortum</i> (W. Smith) Cleve	8	6.5	0.5
GyrFas	<i>Gyrosigma fasciola</i> (Ehrenberg) Griffith & Henfrey 1856	11	6.9	0.9
GyrMac	<i>Gyrosigma macrum</i> (W. Smith) Griffith & Henfrey 1856	15	5.9	2.9
GyrObs	<i>Gyrosigma obscurum</i> (W. Smith) Griffith & Henfrey 1856	9	6.6	0.7
GyrPar	<i>Gyrosigma parkerii</i> (Harrison) Elmore 1921	14	11.6	0.5
GyrSca	<i>Gyrosigma scalproides</i> (Rabenhorst) Cleve 1894	1	1.0	0.2
GyrStr	<i>Gyrosigma strigilis</i> (W. Smith) Cleve 1894	16	5.3	3.4
KarCle	<i>Karayevia clevei</i> (Grunow in Cleve & Grunow) Round & Bukhtiyarova nov. comb.	5	2.8	1.0
KarLat	<i>Karayevia laterostrata</i> (Hustedt) Round & Bukhtiyarova nov. comb.	1	1.0	0.2

LutMut	<i>Luticola mutica</i> (Kützing) D.G. Mann in Round et al. 1990	1	1.0	0.4
MarAto	<i>Maryana atomus</i> (Hustedt) Snoeijs 1991	49	34.2	9.0
MarMar	<i>Maryana martyi</i> (Héribaud) Round in Round et al. 1990	3	2.3	0.5
MarSch	<i>Maryana schulzii</i> (Brockmann) Snoeijs 1991	19	10.7	1.4
MasBal	<i>Mastogloia baltica</i> Grunow in Van Heurck 1880	1	1.0	0.2
MasBra	<i>Mastogloia braunii</i> Grunow 1863	3	3.0	0.2
MasEil	<i>Mastogloia elliptica</i> (C.A. Agardh) Cleve 1893	1	1.0	0.2
MasExi	<i>Mastogloia exigua</i> Lewis 1861	1	1.0	0.3
MasPum	<i>Mastogloia pumila</i> (Cleve & Möller) Cleve 1895	16	13.0	0.5
MasSmi	<i>Mastogloia smithii</i> Thwaites 1856	28	16.0	1.7
MeiArc	<i>Melosira arctica</i> Dickie 1852	5	4.5	0.5
MeiLin	<i>Melosira lineata</i> (Dillwyn) C.A. Agardh 1824	17	13.2	0.7
MeiMon	<i>Melosira moniliformis</i> (O.F. Müller) C.A. Agardh 1824	3	2.7	0.3
MerCir	<i>Meridion circulare</i> (Greville) C.A. Agardh 1831	1	1.0	0.2
NavAboC	<i>Navicula</i> cf. <i>aboenensis</i> (Cleve) Hustedt 1952	1	1.0	0.2
NavAng	<i>Navicula angusta</i> Grunow 1860	2	2.0	0.2
NavAto	<i>Navicula atomus</i> Kützing) Grunow 1860	1	1.0	0.5
NavCap2	<i>Navicula capitata</i> var. <i>hungarica</i> (Grunow) R. Ross 1947	37	18.4	4.7
NavCap3	<i>Navicula capitata</i> var. <i>lueneburgensis</i> (Grunow) Patrick in Patrick & Reimer 1966	5	4.5	0.4
NavCar	<i>Navicula cari</i> Ehrenberg 1836	9	5.8	0.7
NavCin	<i>Navicula cincta</i> (Ehrenberg) Ralfs in Pritchard 1861	5	3.1	0.9
NavCosC	<i>Navicula</i> cf. <i>costulata</i> Grunow in Cleve & Grunow 1880	3	2.6	0.4
NavCruc	<i>Navicula crucicula</i> (W. Smith) Donkin 1872	28	21.1	0.7
NavCryc	<i>Navicula cryptocephala</i> Kützing 1844	12	4.4	2.6
NavCryfC	<i>Navicula</i> cf. <i>cryptotenella</i> Lange-Bertalot 1985	4	4.0	0.2
NavDifC	<i>Navicula</i> cf. <i>difficillima</i> Hustedt 1950	1	1.0	0.3
NavDig	<i>Navicula digitoradiata</i> (Gregory) Ralfs in Pritchard 1861	11	9.2	0.5
NavDue	<i>Navicula duerenbergiana</i> Hustedt in A. Schmidt et al. 1934	1	1.0	0.2
NavEid	<i>Navicula eidrigiana</i> J.R. Carter 1979	4	3.6	0.4
NavEri	<i>Navicula erifuga</i> Lange-Bertalot 1985	4	3.6	0.4

NavFla	<i>Navicula flantica</i> Grunow 1860	38	25.5	1.4
NavFon	<i>Navicula fonticola</i> Grunow in Van Heurck 1880	1	1.0	0.5
NavGer1	<i>Navicula germainii</i> Wallace 1960	1	1.0	0.7
NavGerM	<i>Navicula germanopolonica</i> Witkowski & Lange-Bertalot 1993	17	14.0	0.9
NavGre	<i>Navicula gregaria</i> Donkin 1861	49	39.0	3.6
NavGre2T	<i>Navicula gregaria</i> tp.2	8	6.2	0.7
NavGre3T	<i>Navicula gregaria</i> tp.3	14	10.0	0.8
NavHal	<i>Navicula halophila</i> (Grunow) Cleve 1894	1	1.0	0.4
NavIgnC	<i>Navicula</i> cf. <i>ignota</i> Krasske 1932 emend. Lund 1948	2	2.0	0.2
NavJae	<i>Navicula jaernefeltii</i> Hustedt 1942	4	3.5	0.5
NavKueC	<i>Navicula</i> cf. <i>kuelbsii</i> Lange-Bertalot 1985	4	3.0	0.7
NavLan	<i>Navicula lanceolata</i> (C.A. Agardh) Ehrenberg 1838	9	7.9	0.4
NavLenC	<i>Navicula</i> cf. <i>lenzii</i> Hustedt in A. Schmidt et al. 1936	4	2.2	0.9
NavMar	<i>Navicula margaritana</i> Witkowski 1994	25	21.2	0.7
NavMen1	<i>Navicula menisculus</i> Schumann 1867	6	4.8	0.5
NavMens	<i>Navicula meniscus</i> Schumann 1867	16	12.8	0.5
NavMicC	<i>Navicula</i> cf. <i>microdigitoradiata</i> Lange-Bertalot 1993	1	1.0	0.2
NavMin	<i>Navicula minuscula</i> var. <i>murialis</i> (Grunow) Lange-Bertalot 1981	1	1.0	0.2
NavMini	<i>Navicula minima</i> Grunow in Van Heurck 1880	7	7.0	0.2
NavObl	<i>Navicula oblonga</i> (Kützing) Kützing 1844	1	1.0	0.5
NavPauC	<i>Navicula</i> cf. <i>paul-schulzii</i> Witkowski et Lange-Bertalot sp. nov.	22	6.6	3.6
NavPere	<i>Navicula peregrina</i> (Ehrenberg) Kützing 1844	18	13.8	0.7
NavPerm	<i>Navicula perminuta</i> Grunow in Van Heurck 1880	46	26.5	5.7
NavPhy	<i>Navicula phyllepta</i> Kützing 1844	25	17.5	1.6
NavPhyT	<i>Navicula</i> tp. <i>phyllepta</i>	1	1.0	2.6
NavPor	<i>Navicula porifera</i> Hustedt 1944	1	1.0	0.2
NavPor2	<i>Navicula porifera</i> var. <i>opportuna</i> (Hustedt) Lange-Bertalot 1985	1	1.0	0.2
NavPra	<i>Navicula praeterita</i> Hustedt 1945	2	2.0	0.2
NavPro	<i>Navicula protracta</i> (Grunow) Cleve 1894	1	1.0	0.2
NavRad	<i>Navicula radiosa</i> Kützing 1844	1	1.0	0.2

NavRam	<i>Navicula ramosissima</i> (C.A. Agardh) Cleve 1895	1	1.0	0.2
NavRec	<i>Navicula recens</i> (Lange-Bertalot) Lange-Bertalot 1985	14	11.1	0.7
NavRhy	<i>Navicula rhynchocephala</i> Kützing 1844	48	35.3	3.3
NavRotC	<i>Navicula cf. rotunda</i> Hustedt 1945	1	1.0	0.2
NavSal	<i>Navicula salinarum</i> Grunow in Cleve & Grunow 1880	24	9.6	2.7
NavSch	<i>Navicula schoenfeldii</i> Hustedt 1930	1	1.0	0.2
NavSem	<i>Navicula seminulum</i> Grunow 1860	12	6.3	1.5
NavSle	<i>Navicula slesvicensis</i> Grunow in Van Heurck 1880	9	7.4	0.5
NavSoeC	<i>Navicula cf. soehrensii</i> Krasske 1923	3	2.6	0.3
NavStaC	<i>Navicula cf. starmachii</i> Witkowski & Lange-Bertalot sp. nov.	2	2.0	0.2
NavSubcC	<i>Navicula cf. subcostulata</i> Hustedt 1934	1	1.0	0.3
NavSubtC	<i>Navicula cf. subtilissima</i> Cleve 1991	1	1.0	0.3
NavSup	<i>Navicula supralittoralis</i> Aleem & Hustedt 1951	28	21.3	1.0
NavTrip	<i>Navicula tripunctata</i> (O.F. Müller) Bory 1822	1	1.0	0.2
NavTriv	<i>Navicula trivialis</i> Lange-Bertalot 1980	13	9.3	1.1
NavVan	<i>Navicula vanhoeffenii</i> Gran 1897	3	2.7	0.3
NavVen	<i>Navicula veneta</i> Kützing 1844	14	6.2	1.6
NavVir	<i>Navicula viridula</i> (Kützing) Ehrenberg 1838	5	5.0	0.2
NavVir2	<i>Navicula viridula</i> var. <i>rostellata</i> (Kützing) Cleve 1895	2	2.0	0.2
NavVit	<i>Navicula vitiosa</i> Schimanski 1978	2	2.0	0.2
NavWik	<i>Navicula wiktoriae</i> Witkowski & Lange-Bertalot sp. nov.	23	16.5	0.9
NeiAff	<i>Neidium affine</i> (Ehrenberg) Pfutzer 1871	3	2.7	0.3
Nit1K	<i>Nitzschia</i> sp.1K	26	7.3	7.0
NitAci	<i>Nitzschia acicularis</i> (Kützing) W. Smith 1853	5	4.4	0.4
NitAgn	<i>Nitzschia agnita</i> Hustedt 1957	16	10.3	1.3
NitAngaC	<i>Nitzschia cf. angustatula</i> Lange -Bertalot 1987	10	8.3	0.5
NitAngi	<i>Nitzschia angustiforaminata</i> Lange-Bertalot 1980	1	1.0	0.4
NitArcC	<i>Nitzschia cf. archibaldii</i> Lange-Bertalot 1980	1	1.0	1.0
NitBac	<i>Nitzschia bacillum</i> Hustedt in A. Schmidt et al. 1922	1	1.0	0.2
NitBreC	<i>Nitzschia cf. bremensis</i> Hustedt in A. Schmidt et al. 1921	1	1.0	0.2

NitCap	<i>Nitzschia capitellata</i> Hustedt in A. Schmidt et al. 1922	9	7.9	0.4
NitComp	<i>Nitzschia compressa</i> (Bailey) Boyer 1916	5	5.0	0.2
NitDis	<i>Nitzschia dissipata</i> (Kützing) Grunow 1862	26	20.4	0.9
NitDub	<i>Nitzschia dubia</i> W. Smith 1853	10	8.1	0.5
NitEle	<i>Nitzschia elegantula</i> Grunow in Van Heurck 1881	2	1.6	0.5
NitEpiC	<i>Nitzschia</i> cf. <i>epithemoides</i> Grunow in Cleve & Grunow 1880	1	1.0	0.2
NitFil	<i>Nitzschia filiformis</i> (W. Smith) Van Heurck 1896	6	4.7	0.6
NitFri	<i>Nitzschia frigida</i> Grunow in Cleve & Grunow 1880	4	3.6	0.3
NitFruG	<i>Nitzschia frustulum</i> agg.	43	26.7	4.2
NitGei	<i>Nitzschia geitleri</i> Hustedt 1959	17	9.6	1.1
NitGraf	<i>Nitzschia graciliformis</i> Lange-Bertalot & Simonsen 1978	1	1.0	0.2
NitGras	<i>Nitzschia gracilis</i> Hantzsch 1860	2	2.0	0.2
NitLor	<i>Nitzschia lorenziana</i> Grunow in Cleve & Grunow 1880	9	7.1	0.5
NitMic	<i>Nitzschia microcephala</i> Grunow in Cleve & Möller 1878	24	17.4	1.1
NitPal	<i>Nitzschia palea</i> (Kützing) W. Smith 1856	36	27.5	1.1
NitPalec	<i>Nitzschia paleacea</i> (Grunow) Grunow in Van Heurck 1881	17	12.1	0.7
NitPar	<i>Nitzschia parvula</i> W. Smith 1853 non Lewis 1862	2	2.0	0.2
NitPer	<i>Nitzschia perminuta</i> (Grunow) M. Peragallo 1903	6	5.4	0.4
NitPus	<i>Nitzschia pusilla</i> Grunow 1862 emend. Lange-Bertalot 1976	7	7.0	0.2
NitRec	<i>Nitzschia recta</i> Hantzsch in Rabenhorst 1861-1879	3	3.0	0.2
NitRev	<i>Nitzschia reversa</i> W. Smith 1853	13	10.0	0.6
NitSca	<i>Nitzschia scalaris</i> (Ehrenberg) W. Smith 1853	1	1.0	0.2
NitSig	<i>Nitzschia sigma</i> (Kützing) W. Smith 1853	25	17.8	0.9
NitSubl	<i>Nitzschia sublinearis</i> Hustedt 1930	1	1.0	0.2
NitSupC	<i>Nitzschia</i> cf. <i>supralitorea</i> Lange-Bertalot 1979	2	2.0	0.2
NitThe	<i>Nitzschia thermaloides</i> Hustedt 1955	8	7.3	0.4
NitTub	<i>Nitzschia tubicola</i> Grunow in Cleve & Grunow 1880	18	13.5	0.7
NitUmb	<i>Nitzschia umbonata</i> (Ehrenberg) Lange-Bertalot 1978	5	5.0	0.2
NitValc	<i>Nitzschia valdecostata</i> Lange-Bertalot & Simonsen 1978	5	4.5	0.3
NitVals	<i>Nitzschia valdestriata</i> Aleem & Hustedt 1951	23	18.8	0.5

OpeMut	<i>Opephora mutabilis</i> (Grunow) Sabbe & Vyverman	41	19.1	3.7
PauTae	<i>Pauliella taeniata</i> (Grunow) Round & Basson 1997	44	20.4	26.9
PetdGem	<i>Petrodictyon gemma</i> (Ehrenberg) D.G. Mann in Round et al. 1990	5	4.5	0.3
PetMar	<i>Petroneis marina</i> (Ralfs in Pritchard) D.G. Mann in Round et al. 1990	4	4.0	0.2
PinApp	<i>Pinnularia appendiculata</i> (C.A. Agardh) Cleve 1895	1	1.0	0.2
PinGib	<i>Pinnularia gibba</i> Ehrenberg 1841 (1843)	1	1.0	0.2
PinHemC	<i>Pinnularia cf. hemiptera</i> (Kützing) Rabenhorst 1853	1	1.0	0.2
PinKro	<i>Pinnularia krockii</i> (Grunow) Cleve 1891	8	7.3	0.4
PinLag	<i>Pinnularia lagerstedtii</i> (Cleve) Cleve-Euler 1934	1	1.0	0.2
PinLun	<i>Pinnularia lundii</i> Hustedt 1954	2	1.2	1.9
PinMic	<i>Pinnularia microstauron</i> (Ehrenberg) Cleve 1891	3	3.0	0.2
PinSub	<i>Pinnularia subcapitata</i> Gregory 1856	3	3.0	0.2
PlacCle	<i>Placoneis clementis</i> (Grunow) E.J. Cox 1987	17	15.2	0.4
PlanDelG	<i>Planothidium delicatulum</i> agg.	49	35.8	18.2
PlanLanG	<i>Planothidium lanceolatum</i> agg.	4	3.1	0.5
PlatLep	<i>Plagiotropis lepidoptera</i> (Gregory) Kuntze 1898	25	13.6	2.2
PleElo	<i>Pleurosigma elongatum</i> W. Smith 1852	1	1.0	0.4
PleSal	<i>Pleurosigma salinarum</i> Grunow 1880	8	7.3	0.4
PsaAlt	<i>Psammolithidium altaicum</i> Bukhtiyarova comb. nov., nom. nov.	1	1.0	0.4
PsaHel	<i>Psammolithidium helveticum</i> (Hustedt) Bukhtiyarova & Round comb. nov.	1	1.0	0.2
PsaKue	<i>Psammolithidium kuelbsii</i> (L.-B. in L.-B. & Kram.) Bukhtiyarova & Round comb. nov.	18	2.7	17.5
PsaMar	<i>Psammolithidium marginulatum</i> (Grunow) Bukhtiyarova & Round comb. nov.	1	1.0	0.2
PsaPun	<i>Psammolithidium punctulatum</i> (Simonsen) Bukhtiyarova & Round 1996	4	4.0	0.2
PsaRevC	<i>Psammolithidium reversum</i> (L.-B. & Kram.) Bukhtiyarova et Round comb. nov.	40	12.0	7.5
PsaRevT	<i>Psammolithidium tp. reversum</i>	1	1.0	0.5
PsaRos	<i>Psammolithidium rossii</i> (Hustedt) Bukhtiyarova & Round 1996	2	2.0	0.2
PsaSub	<i>Psammolithidium subatomoides</i> (Hustedt) Bukhtiyarova & Round comb. nov.	5	3.6	0.9
PsaVen	<i>Psammolithidium ventralis</i> (Kraske) Bukhtiyarova & Round comb. nov.	1	1.0	0.2
PseBre	<i>Pseudostaurisira brevistriata</i> (Grunow) Williams & Round 1987	49	33.8	20.5
RhoBre	<i>Rhopalodia brebissonii</i> Krammer in Lange-Bertalot & Krammer 1987	1	1.0	0.2

RhoGib	<i>Rhopalodia gibba</i> (Ehrenberg) O. Müller 1895	19	14.4	0.7
RhoiCur	<i>Rhicospheria curvata</i> (Kützing) Grunow	43	24.9	4.8
RosLin	<i>Rossithidium linearis</i> (W. Smith) Round & Bukhtiyarova nov. comb.	1	1.0	0.2
RosPet	<i>Rossithidium petersenii</i> (Hustedt) Round & Bukhtiyarova nov. comb.	1	1.0	0.3
RosPus	<i>Rossithidium pusillum</i> (Grunow) Round & Bukhtiyarova nov. comb.	9	6.2	0.7
SelIPup	<i>Sellaphora pupula</i> (Kützing) Mereschkowsky 1902	6	4.8	0.5
SkeCos	<i>Skeletonema costatum</i> (Greville) Cleve 1878	22	9.0	9.3
StaCon1	<i>Staurosira construens</i> var. <i>construens</i> Ehrenberg 1843	6	5.4	0.3
StaCon6C	<i>Staurosira</i> cf. <i>construens</i> var. <i>venter</i> (Ehrenberg) Hamilton 1992	37	12.4	7.4
StalLap	<i>Staurosirella lapponica</i> (Grunow in Van Heurck) Williams & Round 1987	36	12.7	13.8
StalPin	<i>Staurosirella pinnata</i> (Ehrenberg) Williams & Round 1987	35	14.1	9.2
StanPho	<i>Stauroneis phoenicenteron</i> (Nitzsch) Ehrenberg 1843	4	4.0	0.2
StanPro	<i>Stauroneis producta</i> Grunow 1880	1	1.0	0.3
StanSmi	<i>Stauroneis smithii</i> Grunow 1860	1	1.0	0.2
StanSpi	<i>Stauroneis spicula</i> Hickie 1874	22	16.7	0.7
StapSal	<i>Staurophora salina</i> (W. Smith) Mereschkowsky 1903	3	2.6	0.5
SteHan	<i>Stephanodiscus hantzschii</i> Grunow in Cleve & Grunow 1880	2	1.8	0.3
SteMed	<i>Stephanodiscus medius</i> Håkansson 1986	1	1.0	0.2
StePar	<i>Stephanodiscus parvus</i> Stoermer & Håkansson 1984	5	1.7	6.4
SurBre	<i>Surirella brebissonii</i> Krammer & Lange-Bertalot 1987	16	13.2	0.5
SurLin	<i>Surirella linearis</i> W. Smith 1853	1	1.0	0.2
SurOval	<i>Surirella ovalis</i> Brébisson 1838	1	1.0	0.4
SurStr	<i>Surirella striatula</i> Turpin 1828	17	13.6	0.5
SynAcu	<i>Synedra acus</i> Kützing 1844	2	1.4	0.9
SynUln	<i>Synedra ulna</i> (Nitzsch) Ehrenberg 1832	2	1.8	0.3
TabeFlo	<i>Tabellaria flocculosa</i> (Roth) Kützing 1844	11	7.3	0.9
TabeQua	<i>Tabellaria quadriseptata</i> Knudson 1952	2	2.0	0.2
TabeVen	<i>Tabellaria ventricosa</i> Kützing 1844	1	1.0	0.2
TabFasG	<i>Tabularia fasciculata</i> agg.	45	31.8	2.9
ThaBal	<i>Thalassiosira baltica</i> (Grunow) Ostenfeld 1901	31	17.2	2.4

ThaGui	<i>Thalassiosira guillardii</i> Hasle 1978	34	17.9	3.3
ThaHyp2	<i>Thalassiosira hyperborea</i> var. <i>lacunosa</i> (Berg) Hasle 1989	10	9.1	0.4
ThaHyp3	<i>Thalassiosira hyperborea</i> var. <i>pelagica</i> (Cleve-Euler) Hasle 1989	8	6.5	0.5
ThaLev	<i>Thalassiosira levanderi</i> Van Goor 1924	18	8.8	4.0
ThaPro	<i>Thalassiosira proschkiniae</i> Makarova 1979	30	10.0	17.5
ThaPseu	<i>Thalassiosira pseudonana</i> Hasle & Heimdal 1970	32	11.3	13.1
TroDan	<i>Tropidoneis dannfeltii</i> Cleve-Euler	2	1.8	0.4
TryApi	<i>Tryblionella apiculata</i> Gregory 1857	18	14.0	0.7
TryCir	<i>Tryblionella circumscuta</i> (Bailey) Ralfs in Pritchard 1861	6	5.5	0.4
TryGra	<i>Tryblionella gracilis</i> W. Smith 1853	11	7.8	1.3
TryHun	<i>Tryblionella hungarica</i> (Grunow) D.G. Mann in Round et al. 1990	25	17.5	0.9
TryLev	<i>Tryblionella levidensis</i> W. Smith 1856	18	15.9	0.4
TryLit	<i>Tryblionella littoralis</i> (Grunow) D.G. Mann in Round et al. 1990	3	2.7	0.3
TrySal	<i>Tryblionella salinarum</i> (Grunow) Pelletan 1889	33	20.1	1.5
