

## Effects of land use, urbanization, and climate variability on coastal eutrophication in the Baltic Sea

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### Abstract

Sedimentary records of organic matter inputs (stable nitrogen isotopes [ $\delta^{15}\text{N}$ ], nitrogen [N], and carbon [C] content), phytoplankton abundance (pigments, stable carbon isotopes [ $\delta^{13}\text{C}$ ]), and community composition (pigments) were used to reconstruct the history and pathway to water-quality degradation in a Swedish Baltic coastal bay. Changes in nutrient sources and cycling began in the 1800s, but eutrophication intensified only after the 1950s, coincident with intensified agricultural practices. Specifically, sedimentary N and C content doubled,  $\delta^{13}\text{C}$  increased, and concentrations of pigments indicative of total algal biomass ( $\beta$ -carotene, Chlorophyll *a*), diatoms (fucoxanthin, diatoxanthin), chlorophytes (lutein-zeaxanthin, Chlorophyll *b*), and cyanobacteria (canthaxanthin) increased significantly after 1950. Enhanced algal abundance was most strongly related to total N influx and secondly to total P influx. In particular, historical change in algae was related to agricultural N influx and only weakly to agricultural P. In recent decades, wastewater N influx has further stimulated algal biomass, particularly diatoms. In contrast, colonial cyanobacteria were more correlated to total P than total N inputs. Variance-partitioning analysis explained 60% of the changes in algal abundance and community composition since 1880, with diffuse and point nutrients jointly explaining 36% of the long-term change in algal biomass. Climate variability has become more important as a factor influencing coastal eutrophication in recent decades, explaining 14% of the variance in the algal data since 1975. Both urban and agricultural sources of nutrients have degraded water quality, illustrating the need for cooperation between stakeholders at regional levels to achieve “good ecological status” in the Baltic coastal environment.

Anthropogenic activities have greatly increased the flux of nutrients to aquatic ecosystems worldwide, with an accelerating rate of increase in recent decades. This increased delivery of nutrients to estuaries and coastal waters has stimulated primary production and has accelerated the eutrophication process (Cloern 2001), particularly of coastal ecosystems limited by nitrogen (N; Nixon 1995). Eutrophication is recognized as one of the foremost threats to coasts and estuaries worldwide (Nixon 1995), causing degradation of water quality, ecological shifts, loss of biodiversity, and disruption of ecosystem function (Rabalais 2002). While the effects of coastal eutrophication are clear, there is still much debate about the relative importance of N vs. phosphorus (P) as controls of primary production in aquatic ecosystems (Lewis and Wurtsbaugh 2008; Schindler et al. 2008; Conley et al. 2009). Moreover, little is known of the unique effects of point and diffuse sources of N and P in degrading water quality and ecosystem function, nor of how these factors interact with climate variability to regulate coastal productivity. This lack of knowledge on the interactive effects of multiple stressors hampers development of predictive models and strategies to manage and restore sensitive nearshore environments (Cloern 2001).

The Baltic Sea provides an excellent model system to evaluate the relative effects of point and diffuse sources of N and P as drivers of coastal eutrophication. Basin-wide analyses estimate that total N (TN) influx to the Baltic Sea

has increased about fourfold during the last century, from  $2.3 \times 10^8 \text{ kg yr}^{-1}$  to  $\sim 1.1 \times 10^9 \text{ kg yr}^{-1}$  (Larsson et al. 1985), while water-column concentrations of TN have increased up to 85% in some coastal zones (Roskilde Fjord; Clarke et al. 2006). Simultaneously, P inputs to the Baltic have increased 3–4 times from  $9.6 \times 10^6 \text{ kg yr}^{-1}$  (Larsson et al. 1985) to  $\sim 3.3 \times 10^7 \text{ kg yr}^{-1}$  (HELCOM 2004). Concomitant with this increased nutrient availability, water transparency has declined (Bonsdorff et al. 1997), filamentous algae have partly replaced perennial brown macroalgae (Berger et al. 2003), and benthic fauna have been altered or eliminated due to changes in vegetation (Rönnerberg and Bonsdorff 2004), increased deposition of organic matter, and development of deep-water hypoxia (Cederwall and Elmgren 1990). Both point sources of nutrients (e.g., wastewater and riverine inputs; Voss et al. 2000; Savage and Elmgren 2004) and diffuse inputs (e.g., fertilizer and agricultural runoff; Clarke et al. 2006) have been implicated as causal factors of local eutrophication problems in the Baltic, while the ecological effects of climatic variability have also been recognized (Dippner et al. 2000).

The objectives of this study were the following: (1) to use annually-resolved sedimentary records to quantify baseline phytoplankton abundance in a Swedish Baltic coastal area at the beginning of the twentieth century, (2) to establish the pathway to modern water-quality degradation, (3) to assess the relative importance of N vs. P as determinants of water-quality degradation, and (4) to evaluate the unique and interactive effects of land use, urbanization, and climate variability on historical eutrophication.

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## Methods

**Site description**—Himmerfjärden (59°00'N, 17°45'E) is a mesohaline 30-km-long bay of the Baltic Sea proper located ~ 60 km south of Stockholm, Sweden (Fig. 1). The bay has a water surface area of 174 km<sup>2</sup>, a mean depth of 17 m, and a maximum depth of 52 m. The inner basins including Kaggfjärden experience long water residence, typically > 60 d in surface waters and > 140 d in bottom waters (Engqvist 1996). The bay is thermally stratified in summer, and bottom-water hypoxia results commonly in reduced sediments in deeper areas (Elmgren and Larsson 1997). The water is brackish with a mean salinity of 6, but unlike most estuaries, salinity fluctuations are small ( $\pm 2$ ), and there is no tidal forcing. The inner basins normally have an ice cover during winter that limits wind-induced mixing. Consequently, there are several sediment-accumulation areas suitable for high-resolution freeze-crust coring from the ice during winter.

**Field and laboratory methods**—Four sediment cores were taken from Himmerfjärden and surrounding bays in 2000 and 2001 (Fig. 1). All four cores showed comparable changes in pigment stratigraphies. Therefore, for brevity, we present only data from the 76-cm-long freeze-crust core (K1) collected from Kaggfjärden basin in March 2001. This core had the longest sequence of annually laminated sediments while displaying the same biogeochemical trends as our other cores and earlier studies (Bianchi et al. 2002; Savage et al. 2004). The presence of the sulfur-reducing bacterium *Beggiatoa* that lives at the sediment–water interface at the top of the core attested to the collection of the sediment surface. Chronology for the K1-core was determined by radiometric dating of radioactive isotopes <sup>210</sup>Pb and <sup>137</sup>Cs at the Liverpool University Environmental Radioactivity Laboratory and by varve counts.

The core was sectioned by hand in a –20°C freezer following the methods of Renberg and Hansson (1993). The outermost layers of the frozen core were removed using a wood plane to reveal laminations that, when present, were counted, measured, and sectioned according to cycles of annual deposition. Annual laminations (varves) included a dark-colored, organic-rich layer overlying a light-colored spring layer with abundant spring diatoms. The uppermost 33 cm of the core was clearly laminated and was sectioned by hand into 45 annual increments. The mean thickness of laminae was 1 cm in the top 20 laminae and 0.5 cm in laminae 21–45. Laminations were also present between 34 cm and 44 cm but were disturbed by bioturbation, and below 45 cm sediments changed from a dark, organic-rich matrix to more light-colored deposits. Consequently, the lowest region of the core (34–76 cm) was sectioned at 1-cm intervals.

Bulk sediments were freeze-dried, and water content was determined for each sample. Homogenized aliquots (~ 5 mg) were analyzed for stable isotope and elemental content using a Thermoquest (Finnigan-MAT) Delta Plus<sup>XL</sup> mass spectrometer interfaced with a Carlo Erba NC2500 elemental analyzer following Savage et al. (2004). Preliminary analyses indicated that stable carbon isotope

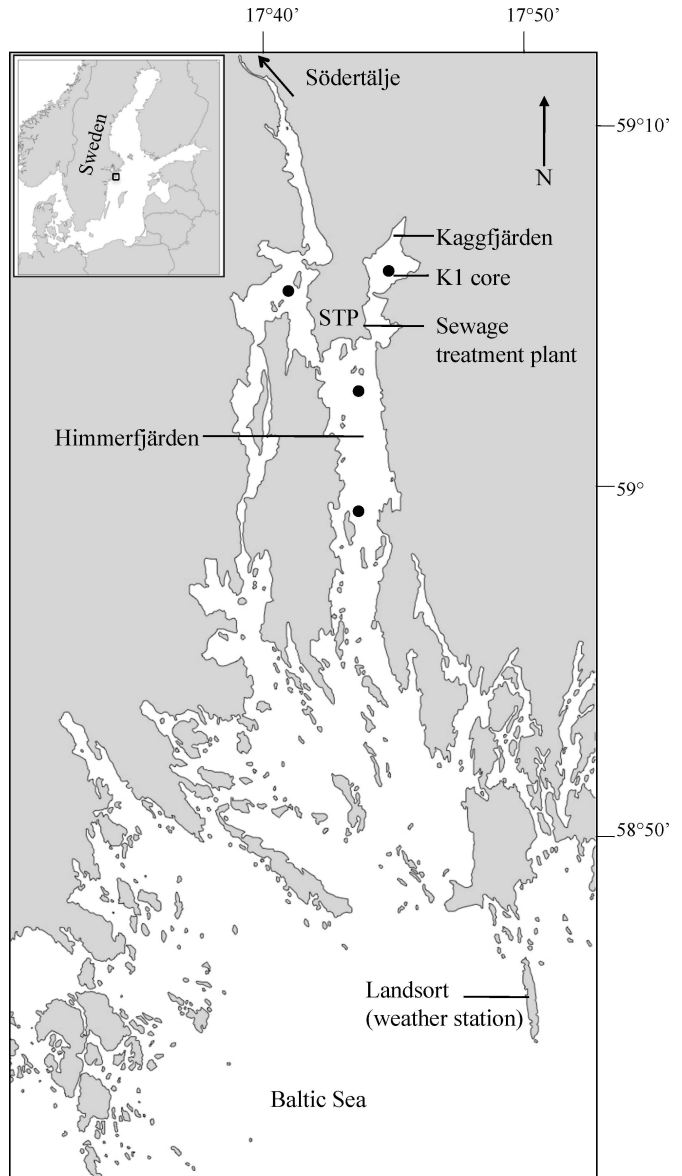


Fig. 1. Map of Himmerfjärden and Kaggfjärden embayment on the Swedish Baltic coast. The sampling location of the K1 frozen core and the other three cores are indicated as filled circles. The sewage treatment plant (STP) and Landsort weather monitoring station are also indicated in the figure.

( $\delta^{13}\text{C}$ ) determinations were not affected by acidification to remove carbonates; therefore, stable isotope analyses were conducted on whole sediments. Isotopes were measured relative to a working standard and were calibrated against the international standards, atmospheric  $\text{N}_2$  and Vienna Pee Dee Belemnite for stable nitrogen isotopes ( $\delta^{15}\text{N}$ ) and  $\delta^{13}\text{C}$ , respectively. Nitrogen and carbon stable-isotope ratios are reported as

$$\delta(\text{‰}) = \left[ \frac{(R_{\text{sample}} - R_{\text{standard}})}{R_{\text{standard}}} \right] \times 1000 \quad (1)$$

where  $R$  is the ratio  $^{15}\text{N}:^{14}\text{N}$  and  $^{13}\text{C}:^{12}\text{C}$ . Sedimentary  $\delta^{13}\text{C}$  values were corrected for the Suess effect, the time-dependent depletion in atmospheric carbon dioxide

( $\delta^{13}\text{C}_2$ ) since 1840 due to fossil-fuel burning (Schelske and Hodell 1995), following the corrected equation in Verburg (2007).

Forty-five intervals from the K1 frozen core were extracted for analysis of fossil-pigment concentrations (Leavitt and Hodgson 2001). Briefly, pigments were extracted into a standard mixture of acetone:methanol:water (80:15:5 by volume). Extracts were filtered, dried, and stored in oxygen-free, dark, and cold conditions. Carotenoid, chlorophyll and derivative concentrations were determined using a Hewlett Packard 1050 high-performance liquid chromatography (HPLC) system. Pigments identified included carotenoids characteristic of total algae [ $\beta$ -carotene, chlorophyll *a* (Chl *a*)], siliceous algae, diatoms and some dinoflagellates (fucoxanthin), diatoms, prymnesiophytes, some chrysophytes and dinoflagellates (diatoxanthin), cryptophytes (alloxanthin), cyanobacteria (canthaxanthin), and a mixture of chlorophytes and cyanobacteria (lutein-zeaxanthin), as well as *a*, *b*, and *c* phorbins, including undegraded chlorophyll and pheophytin derivatives. Pigment concentrations are expressed relative to organic matter (OM) concentrations (nmol pigment  $\text{g}^{-1}$  OM), determined as mass loss-on-ignition at 500°C for 1 h. Variations in the degree of pigment preservation were estimated using the ratio of Chl *a* to pheophytin *a*, since pheophytin *a* is a degradation product of Chl *a* (Leavitt and Hodgson 2001). Historical changes in algal exposure to ultraviolet radiation (UVR) were measured as changes in ratios of UVR-absorbing pigments:carotenoids (e.g., alloxanthin, diatoxanthin, lutein-zeaxanthin), an index that was linearly related to the depth of UVR penetration in whole-lake experiments (Leavitt et al. 1997).

*Numerical analyses*—Annually resolved sedimentary records of stable isotope ratios ( $\delta^{15}\text{N}$ ,  $\delta^{13}\text{C}$ ), elemental composition (molar C:N, %N, %C), and log-transformed pigment concentrations (nmol  $\text{g}^{-1}$  OM) were related to potential individual drivers of eutrophication (point sources and diffuse sources of N and P) using stepwise linear regressions in SYSTAT v. 12.0 (SYSTAT Software). Individual time series were tested for effects of lag responses using cross-correlation analyses following methods outlined in Bunting et al. (2007). In addition, variance partitioning analysis (Borcard et al. 1992) was used to assess the relative importance of urban point sources of nutrients, diffuse nutrient inputs from the catchment, and climatic variables as correlates of change in water quality since 1880.

*Historical data*—Continuous annual data of 16 environmental variables, including point and nonpoint nutrient sources and climate were collated for the variance partitioning analysis (VPA). Since historical data were missing or unreliable prior to 1880, the VPA was restricted to the period 1880–2000. Any missing data during this period were interpolated using a seven-point running average.

Urban point sources of nutrients: The head of the bay receives tertiary-treated wastewater that is discharged

primarily as dissolved inorganic nutrients at 25 m depth. The sewage treatment plant (STP) currently treats sewage from 250,000 people. The STP started operation in 1974 and initially discharged  $\sim 3.3 \times 10^5$  kg TN  $\text{yr}^{-1}$  and  $\sim 2 \times 10^4$  kg total phosphorus (TP)  $\text{yr}^{-1}$  (Fig. 2a,b). Sewage nitrogen inputs increased to peak loads of  $9 \times 10^5$  kg TN  $\text{yr}^{-1}$  by the mid-1980s when municipal wastewater was the predominant source of N in inner Himmerfjärden (Elmgren and Larsson 1997; Elmgren and Larsson 2001). During the 1990s, enhanced denitrification procedures were implemented that became fully effective in 1997, after which sewage N loads decreased to  $< 2 \times 10^5$  kg TN  $\text{yr}^{-1}$  (Savage and Elmgren 2004; Fig. 2a). Sewage phosphorus was generally  $\sim 1.3 \times 10^4$  TP  $\text{yr}^{-1}$  except in 1984 when  $\sim 3 \times 10^4$  TP was deliberately released from the STP to assess the effects of increased P availability on algal biomass in the bay. Monitoring data have shown that Chl *a* concentrations are correlated with annual mean TN concentrations ( $r^2 = 0.64$ ), and to a less extent TP ( $r^2 = 0.51$ ), suggesting that algal biomass is predominantly N-limited in the bay, except during spring and early summer in the years of peak N load (Elmgren and Larsson 2001).

Nutrient concentrations in the discharged wastewater are monitored frequently, providing accurate numbers for the inputs of tertiary-treated wastewater N and P into the bay (Fig. 2a,b). To assess the historical inputs of primary-treated sewage to the bay, we used human population census data for the drainage basin of Kaggfjärden and estimated person equivalents (PE) loading using 11.2 g N per person  $\text{d}^{-1}$  and 1.58 g P per person  $\text{d}^{-1}$  (Brandt et al. 2008). Long-term changes in sewage-derived nutrient loading to the bay thus reflect human population growth in the area, which increases steadily between 1880–2000 (Fig. 2a,b), and from 1974 also includes wastewater from distant areas transferred into the bay. Nitrogen and phosphorus loads from primary sewage and tertiary-treated wastewater were used in the data analysis as urban point sources of nutrients.

Diffuse nutrient inputs: Land use in the Kaggfjärden drainage basin (527  $\text{km}^2$ ) was quantified in terms of the areal extent of agriculture (arable land and pasture), forestry, open land, and built-up areas and the nutrient fluxes associated with each category. Historical records of N and P applications in commercial fertilizers and manure in Sweden (Statistiska centralbyrån, 1995; Hoffmann et al. 2000) were used to approximate long-term trends in fertilizer loading rates ( $\text{kg N km}^{-2}$ ,  $\text{kg P km}^{-2}$ ). Gross loads of fertilizer N and P applied to the catchment were calculated as national average fertilizer loading rates multiplied by 112  $\text{km}^2$ , the area under agriculture in the drainage basin. To calculate nutrient flux from agriculture to the receiving bay, we assumed 30% and 97% denitrification and retention in the soil for N and P respectively (Hoffmann and Johnsson 1999; Brandt et al. 2008) and multiplied the net nutrient load by a runoff factor ( $\text{L s}^{-1}$ ). The runoff factor was based on annual precipitation records (mm) so that agricultural leaching rates were linked to rainfall patterns. The resultant agricultural N and P loading rates in recent years were in agreement with

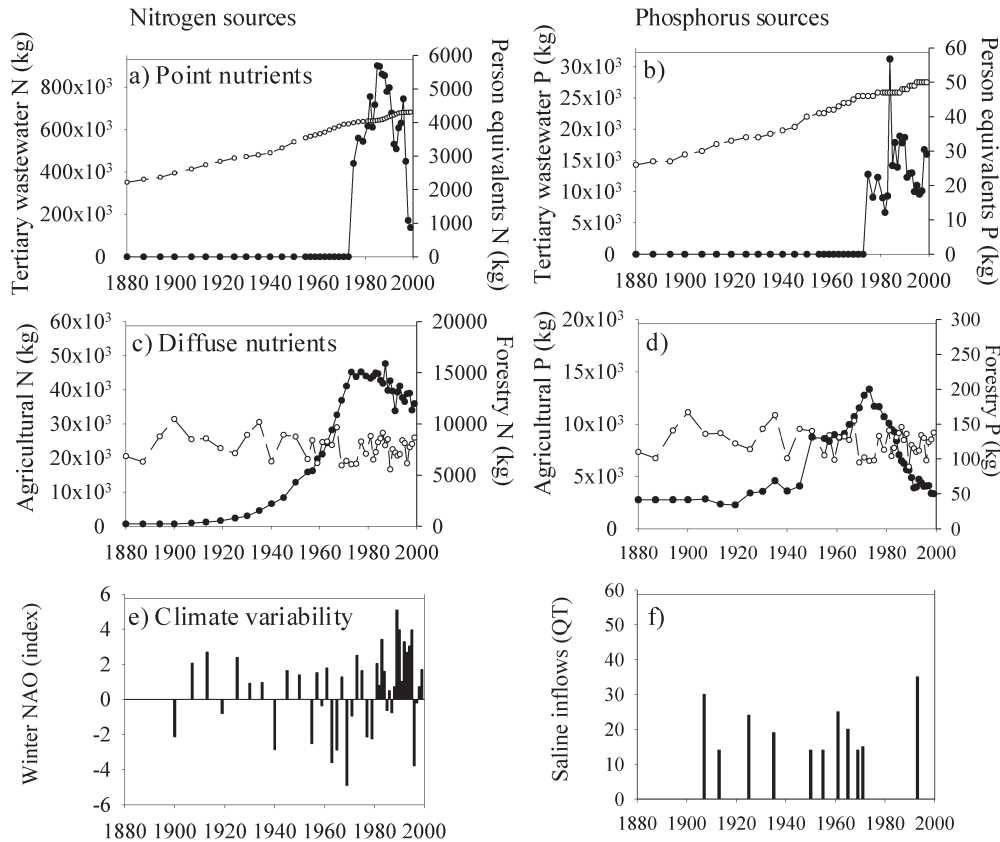


Fig. 2. Long-term data (1880–2000) for the Kaggfjärden drainage basin showing estimated changes in nitrogen and phosphorus sources and climate. (a, b) Tertiary-treated wastewater N and P inputs (filled circles) from Himmerfjärden sewage treatment plant and person equivalents nutrient loading in Kaggfjärden catchment (open circles). (c, d) Agricultural sources of nutrients (filled circles) and N and P leached from forestry land (empty circles). Climatic variables including (e) the winter NAO index, and (f) major Baltic inflows expressed as an intensity index (Q<sub>T</sub>). Further details given in Methods and Table 1.

modeled estimates for the Baltic region (Hoffmann and Johnsson 2000; Hoffmann et al. 2000; Brandt et al. 2008). Historical changes in agricultural N and P loads to the estuary show a rapid increase after 1950 coincident with increased fertilizer use, and they show a decrease after 1970 (Fig. 2c,d).

Forestry accounts for 248 km<sup>2</sup> (~ 50%) of the land area in the drainage basin, yet forests typically contribute only small amounts of nutrients to watercourses (Akselsson and Westling 2005). We used nutrient leaching rates from forest soils (Akselsson and Westling 2005) that incorporate a runoff factor based on rainfall to estimate N and P export from forested regions. Nutrient flux from forest soils thus exhibits large interannual variability that is driven by annual variation in rainfall (Fig. 2c,d).

Nutrient influx from built-up areas (29 km<sup>2</sup>) and untilled land (79 km<sup>2</sup>) was determined from estimates of hydrologic runoff using a leaching coefficient specific to the Baltic region (Hoffmann et al. 2000). However, these diffuse sources of nutrients were removed from the analysis because they were consistently neither significant nor unique (variance inflation factor, VIF > 20) as determi-

nants of algal abundance in Monte Carlo permutation tests.

**Climate:** Climatic data incorporated in the VPA included the winter North Atlantic Oscillation (NAO) index (Fig. 2e), indices of saline water inflow to the Baltic Sea from the North Sea (intensity index, Q<sub>T</sub>; Fig. 2f), annual total rainfall (mm), and annual average temperature (°C). The NAO is the dominant mode of low-frequency atmospheric variability over the North Atlantic and significantly affects Scandinavian climate variability (Hurrell 1995) and Baltic Sea level (Andersson 2002). Water conditions in the Baltic are also affected by the inflow of highly saline and oxygenated water from the North Sea, a climate–ocean phenomenon described using an intensity index (Q<sub>T</sub>), which can cause increased vertical mixing and higher nutrient concentrations in surface waters (Schinke and Matthaus 1998). Finally, annual total precipitation and annual average temperatures recorded at the Landsort coastal monitoring station were obtained from the Swedish Meteorological and Hydrological Institute.

The 16 environmental variables were tested for covariance using VIFs and for significance using Monte Carlo

Table 1. Explanatory variables used in the variance partitioning analyses and grouped into the three categories: (A) point nutrient inputs, (B) diffuse nutrient inputs, and (C) climate.

Variable name (units)	Start and end of record		Data source
<b>A) Point nutrient inputs</b>			
Tertiary wastewater nitrogen (kg yr <sup>-1</sup> )*	1974	2000	SYVAB (Himmerfjärden sewage treatment plant)
Tertiary wastewater phosphorus (kg yr <sup>-1</sup> )†	1974	2000	as above
Person equivalents nitrogen (kg yr <sup>-1</sup> )*	1820	2000	Human population in drainage basin (Swedish Statistical Bureau, SCB) multiplied by person equivalents from Brandt et al. (2008)
Person equivalents phosphorus (kg yr <sup>-1</sup> )†	1820	2000	as above
<b>B) Diffuse nutrient inputs</b>			
Agricultural nitrogen runoff (kg yr <sup>-1</sup> )*	1901	2000	Nutrients in commercial fertilizer and manure from Hoffmann et al. (2000); SCB (1995). Nutrient retention in soil from Hoffmann and Johnsson (1999); Brandt et al. (2008)
Agricultural phosphorus runoff (kg yr <sup>-1</sup> )†	1901	2000	as above
Nitrogen leaching from forest land (kg yr <sup>-1</sup> )*	1925	2000	Forestry in drainage basin from SCB. Nutrient leaching rates (Akselsson and Westling 2005). Rainfall (SMHI)
Phosphorus leaching from forest land (kg yr <sup>-1</sup> )†	1925	2000	as above
<b>C) Climate</b>			
Winter NAO, index	1900	2000	Hurrell (1995); <a href="http://jisao.washington.edu/data_sets/nao/">http://jisao.washington.edu/data_sets/nao/</a>
Major Baltic inflows, intensity index (Q <sub>T</sub> )	1900	2000	Schinke and Matthäus (1998)
Annual total precipitation (mm)	1880	2000	Swedish Meteorological and Hydrological Institute (SMHI)
Annual average temperature (°C)	1880	2000	as above

The \*nitrogen and †phosphorus sources were also combined into separate categories and their unique and interactive effects tested using VPA.

permutation tests. The data were assigned to three categories for each analysis, and the data set was refined to 12 variables with an equal number of factors in each category. In the first iteration, we assessed the importance of N vs. P by first classifying the data as nitrogen sources, phosphorus sources, or climate factors. Then, to evaluate the influence of point vs. nonpoint nutrient sources and climate-driven phenomena, the variables were assigned to categories connected with urban development, land use, or climate variability (Table 1). VPA quantifies how well these categories and their interaction terms explain the variance in the past algal abundance and community composition using a series of constrained and partial redundancy analyses (RDAs) in CANOCO version 4.02 (Microcomputer Power). RDAs were used since preliminary detrended correspondence analyses showed that the isotope and pigment data varied in a linear manner along environmental gradients. Analyses were performed for the periods 1880–2000, 1950–2000, and 1975–2000 to determine whether the potential drivers have evolved during the 1900s, in particular following increased fertilizer application (post-1950) and discharge of treated wastewater (post-1975).

## Results

**Core chronologies**—Analyses of radioisotope profiles demonstrated that total <sup>210</sup>Pb activity of the K1 core reached equilibrium with the supporting <sup>226</sup>Ra at a depth of ~ 32 cm (Fig. 3a). Unsupported <sup>210</sup>Pb activities were very low and declined irregularly with depth (Fig. 3b). In contrast, <sup>137</sup>Cs activity vs. depth (Fig. 3c) exhibited a

well-resolved peak at 7.5 cm that had a <sup>137</sup>Cs inventory (10,620 Bq m<sup>-2</sup>) characteristic of the 1986 Chernobyl reactor accident. Radiometric dating using the constant rate of supply (CRS) model gave a mean accumulation rate during the past 70 yr of 0.078 g cm<sup>-2</sup> yr<sup>-1</sup>. Although there are episodes of more rapid accumulation (~ 1950, 1986, 1995–2000), there was no systematic increase in deposition through time.

Uncertainty in the <sup>210</sup>Pb dates was greater in the lower sections of the core due to the low <sup>210</sup>Pb activity (Fig. 3d). Consequently, the chronology for the core was also validated using varve counts within the clearly laminated intervals (0–33 cm depth; years 1956–2000) and by extrapolation of the observed age–depth relationship for deeper sections. Specifically, age (determined by varves) was plotted against burial depth (cm) and fitted with a polynomial equation: age = 0.0259 depth<sup>2</sup> + 0.4171 depth + 1.8162 ( $r^2 = 0.99$ ,  $p < 0.0001$ ). This method assumes that mass accumulation rates did not change significantly with the onset of laminations, which is consistent with age estimates from <sup>210</sup>Pb and <sup>137</sup>Cs activities. Overall chronologies based on <sup>210</sup>Pb and varve counts were highly correlated ( $r^2 = 0.99$ ,  $p < 0.0001$ ).

**Sediment geochemistry**—The oldest sediment intervals in the coastal embayment had  $\delta^{15}\text{N}$  values within the range of values reported for modern sediments in the open Baltic Sea (Voss et al. 2000). Sedimentary  $\delta^{15}\text{N}$  increased from ~ 3.5‰ in the early 1800s to 4.5‰ at the end of the 1800s (Fig. 4a), then increased to a stable plateau of 5.5‰ between 1920 and 1970. There was no obvious change in  $\delta^{15}\text{N}$  with the onset of laminations in ~ 1950 (A in

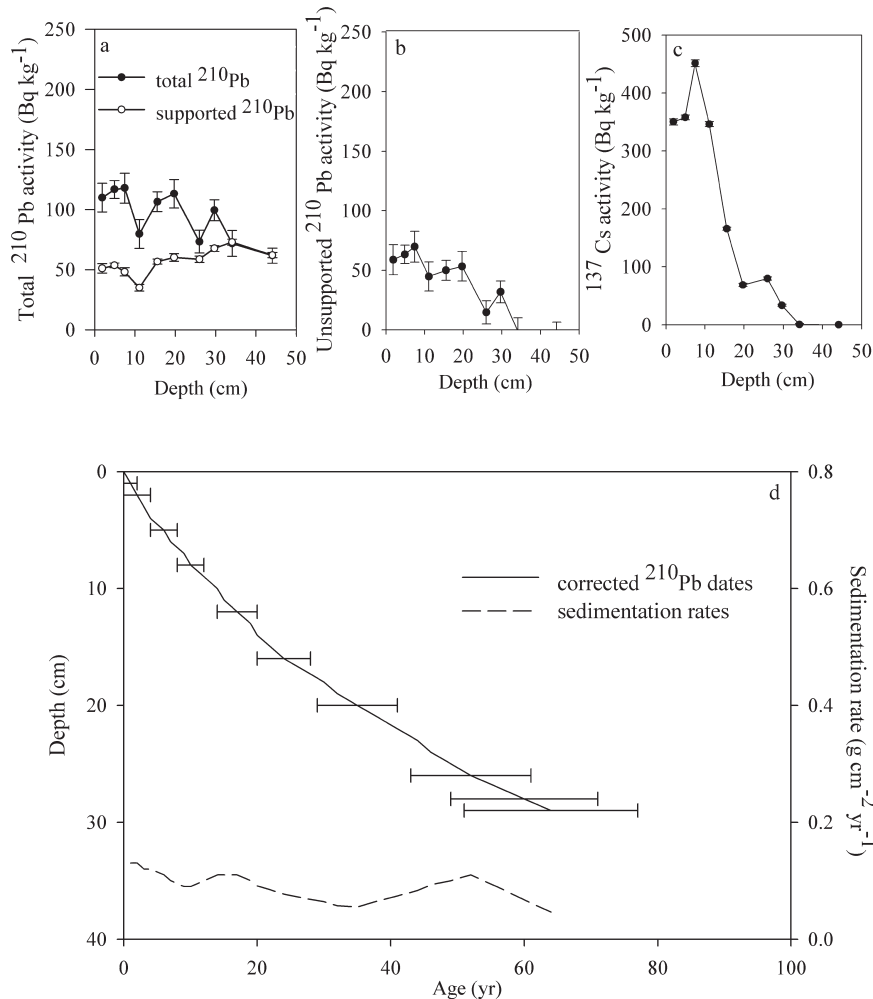


Fig. 3. Fallout radionuclides in the K1 core showing (a) total and supported  $^{210}\text{Pb}$ , (b) unsupported  $^{210}\text{Pb}$ , and (c)  $^{137}\text{Cs}$  concentrations vs. depth. (d) Radiometric chronology of the K1 core showing the CRS model  $^{210}\text{Pb}$  dates and the 1986 and possible 1963 depths determined from the  $^{137}\text{Cs}$  stratigraphy. Corrected  $^{210}\text{Pb}$  dates and calculated sedimentation rates are also shown.

Fig. 4a). The most distinctive increase in  $\delta^{15}\text{N}$  occurred following the initiation of treated wastewater discharge into the bay (B in Fig. 4a), when isotopic values increased from 5.5‰ in 1974 to a peak of 8.3‰ during years of maximum sewage loads (1985–1990). After 1997, sedimentary  $\delta^{15}\text{N}$  decreased to  $\sim 7.5$ ‰, following final commissioning of STP upgrades to remove waste N through advanced denitrification procedures. Long-term trends in sedimentary  $\delta^{13}\text{C}$  values (corrected for the Suess effect) ranged between  $-30$ ‰ and  $-33$ ‰, with high interannual variability after 1950, but a general enrichment occurred in the late 1970s–1980s and a pronounced decrease in  $\delta^{13}\text{C}$  in near-surface sediments (Fig. 4b). Sediment total N and C content (% dry weight) was relatively stable from  $\sim 1820$  to 1950, but doubled in sediments deposited during the last 50 yr (Fig. 4c). In contrast to  $\delta^{15}\text{N}$  profiles, there were few changes in N or C content prior to 1950, after which the content of both elements increased linearly to the surface. Similarly, C:N was very stable at  $\sim 10$  until  $\sim 1955$ , after which it progressively decreased to  $\sim 9$  in near-surface

sediments consistent with increasing N inputs. Overall, molar C:N ranged from 8.7 to 10.2, indicative of autochthonous phytoplankton inputs (Fig. 4d).

Changes in sedimentary isotope ratios ( $\delta^{15}\text{N}$ ) were related to variation in N inputs ( $R^2 = 0.57$ ,  $p < 0.0001$ ) and secondly to changes in P loads ( $R^2 = 0.49$ ,  $p < 0.0001$ ). In particular,  $\delta^{15}\text{N}$  signatures were correlated to inputs of urban N (sum of tertiary and primary sewage N inputs) ( $R^2 = 0.56$ ,  $p < 0.0001$ ) and, to a lesser extent, influx of diffuse N (sum of all catchment N sources) ( $R^2 = 0.39$ ,  $p < 0.0001$ ). Sediment  $\delta^{15}\text{N}$  was positively related to urban P inputs ( $R^2 = 0.39$ ,  $p < 0.0001$ ) but not to influx of diffuse P ( $R^2 = 0.04$ ,  $p = 0.14$ ).

*Sedimentary pigments*—Concentrations of most sedimentary pigments increased from  $\sim 1920$  to a peak in the 1980s, then declined in sediments deposited during the last 10–15 yr (Fig. 5; see Web Appendix: [http://www.aslo.org/lo/toc/vol\\_55/issue\\_3/1033a.pdf](http://www.aslo.org/lo/toc/vol_55/issue_3/1033a.pdf)). For example, profiles of pigments indicative of total algal abundance (Chl *a*,  $\beta$ -

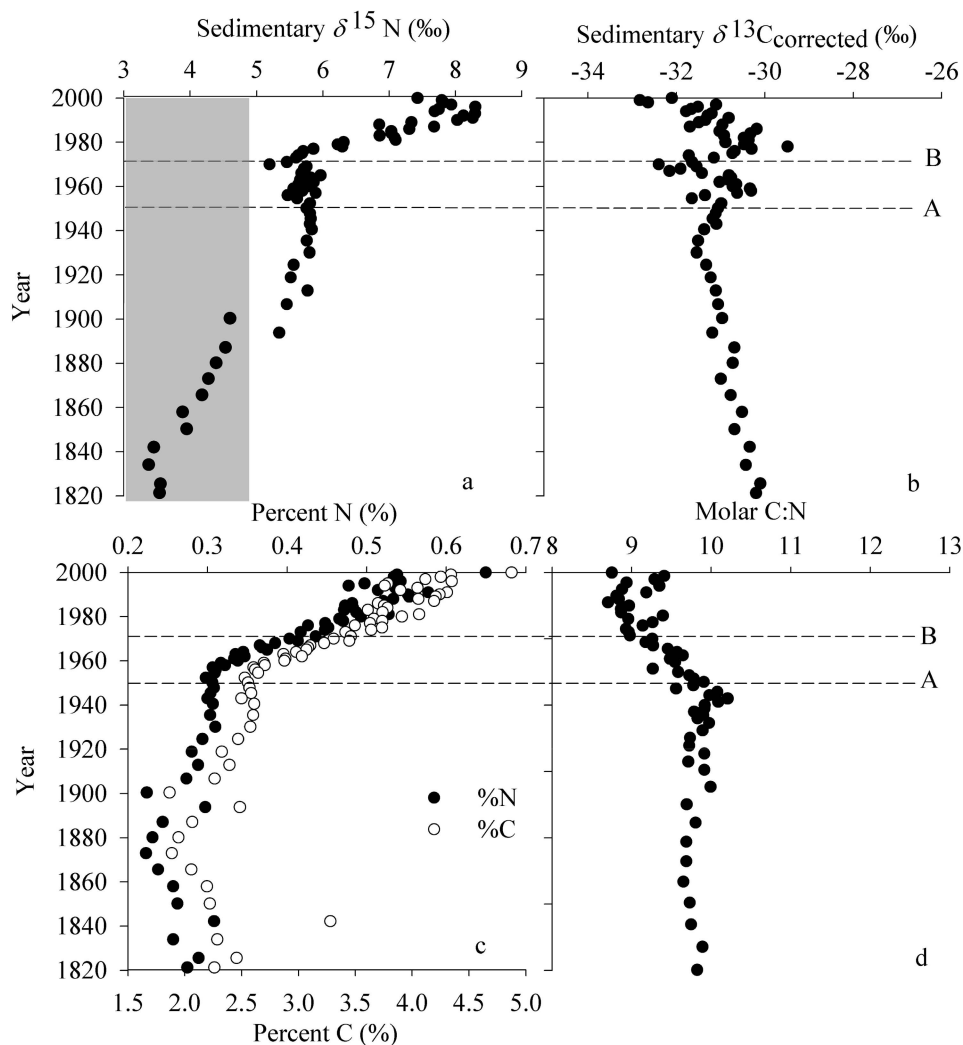


Fig. 4. (a) Sedimentary  $\delta^{15}\text{N}$  values of K1 core showing the timing of (A) the onset of clear laminations and (B) initiation of treated sewage discharge from the sewage treatment plant (STP). Reported sedimentary  $\delta^{15}\text{N}$  values for the open Baltic Sea (Voss et al. 2000) are illustrated as the shaded area. (b) Sedimentary  $\delta^{13}\text{C}$  values corrected for the Suess effect, (c) percent N (filled circles) and percent C (open circles), and (d) molar C:N of the K1 core.

carotene) started increasing  $\sim 1920$ , were unaffected by the onset of laminations (A in Fig. 5), and exhibited further increases post-1950 with maxima (e.g.,  $1353 \text{ nmol } \beta\text{-carotene g}^{-1} \text{ OM}$  in 1986) recorded during the years of peak wastewater N loads (1985–1990). Concentrations of  $\beta\text{-carotene}$  were still elevated during the 1990s ( $> 600 \text{ nmol g}^{-1} \text{ OM}$ ), but tended to decline in near-surface sediments ( $\sim 400 \text{ nmol g}^{-1} \text{ OM}$  after 1998). Similarly, concentrations of pigments derived predominantly from diatoms and other siliceous algae (fucoxanthin, diatoxanthin), green algae (Chl *b*), green algae and cyanobacteria (lutein-zeaxanthin), and cyanobacteria alone (canthaxanthin) all increased after 1950 and remained high or increased further (fucoxanthin, diatoxanthin, alloxanthin) following the onset of wastewater inputs. Overall, concentrations of fucoxanthin and diatoxanthin were tenfold greater than those of other biomarker pigments (up to  $2000 \text{ nmol g}^{-1} \text{ OM}$ ). In general, ratios of labile : stable pigments, such as Chl *a* : pheophytin

*a*, varied little with depth, suggesting that there were few changes in preservation during the core record. In contrast, analysis of the UVR index suggested that water clarity (as UVR exposure) was greatest prior to 1940 but has since decreased thereafter.

Sedimentary concentrations of  $\beta\text{-carotene}$  (total algal biomass) were most strongly related to changes in total N inputs ( $R^2 = 0.69$ ,  $p = 0.0001$ ; Fig. 6a). Cross-correlation analyses showed that algal abundance was strongly correlated to TN influx ( $r^2 = 0.84$ ,  $p < 0.001$ ), even when algal biomass was lagged 1–2 yr relative to N inputs ( $r^2 = 0.83$ ,  $p < 0.001$ ). In particular, sedimentary  $\beta\text{-carotene}$  concentrations were positively related to agricultural N runoff ( $R^2 = 0.69$ ,  $p < 0.0001$ ) and urban N inputs ( $R^2 = 0.64$ ,  $p < 0.0001$ ). Indicator pigments for diatom abundance (fucoxanthin+diatoxanthin) ( $R^2 = 0.54$ ,  $p < 0.0001$ ), cryptophytes (alloxanthin) ( $R^2 = 0.56$ ,  $p < 0.0001$ ), green algae and some cyanobacteria (pheophytin *b*; Chl *b*; lutein-

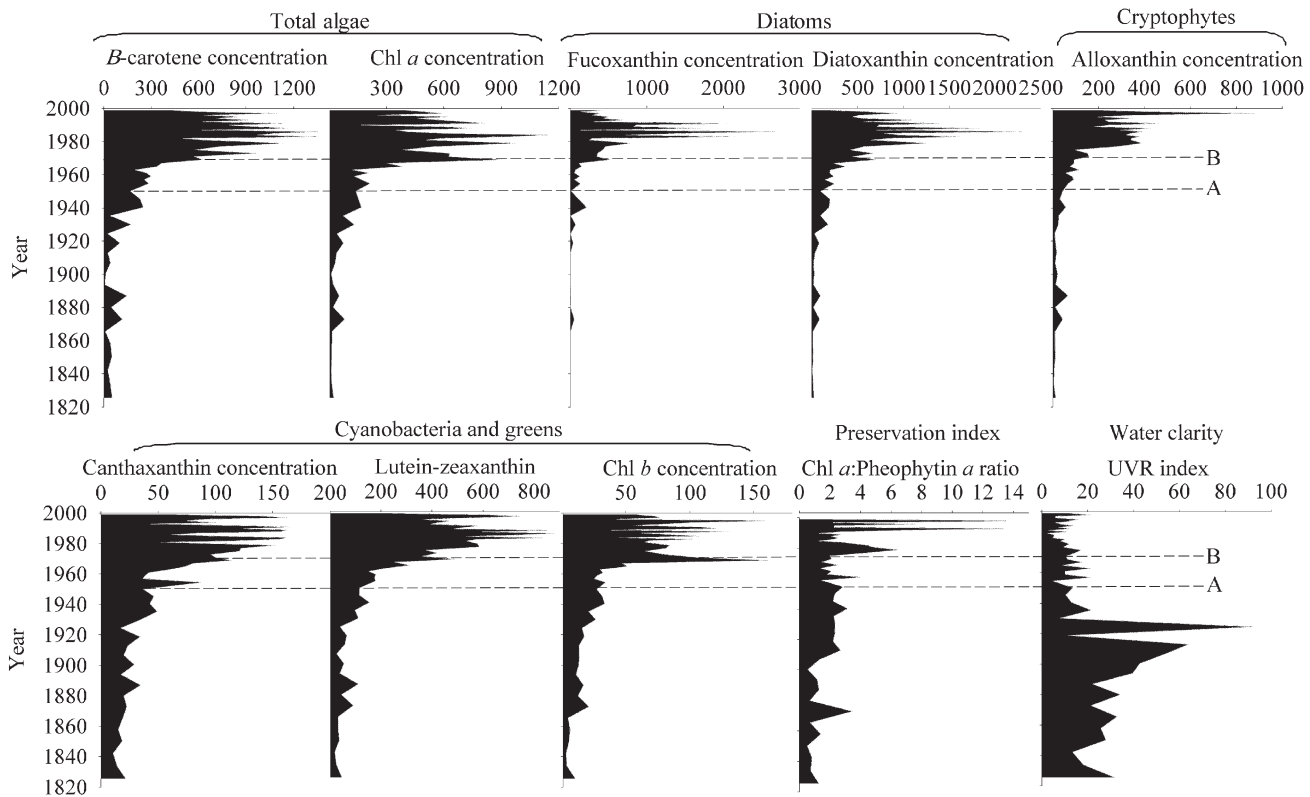


Fig. 5. Temporal change in sedimentary pigment concentrations, expressed as nmol pigment per gram of organic matter (OM). The timing of the two transitions: (A) onset of clear laminations, and (B) start-up of the STP is indicated on the figure.

zeaxanthin) were most strongly related to total N loads ( $R^2 = 0.52$ ,  $p < 0.0001$ ;  $R^2 = 0.40$ ,  $p < 0.0001$ ;  $R^2 = 0.76$ ,  $p < 0.0001$ , respectively; Fig. 6c). In particular, diatom abundance was positively related to wastewater N influx ( $R^2 = 0.73$ ,  $p < 0.0001$ ).

The relationship between  $\beta$ -carotene concentration and total P inputs was also significant ( $R^2 = 0.59$ ,  $p = 0.005$ ; Fig. 6b) but had a lower predictive power than for N influx. Cross-correlation analyses showed that algal abundance was positively related to P inputs ( $r^2 = 0.70$ ,  $p < 0.001$ ), and the correlation remained high when algal biomass was lagged 1–2 yr relative to P inputs ( $r^2 = 0.69$ ,  $p < 0.001$ ). However, with longer time lags, the relationship between  $\beta$ -carotene and P influx decreased (e.g.,  $r^2 = 0.40$ ,  $p < 0.001$  with a 10-yr lag). This proxy of overall algal biomass was positively related to P in urban wastewater ( $R^2 = 0.56$ ,  $p < 0.0001$ ) and was weakly related to agricultural P inputs ( $R^2 = 0.32$ ,  $p < 0.0001$ ). In contrast, regression analysis showed that canthaxanthin, a biomarker for colonial N-fixing cyanobacteria, was slightly better correlated with total P ( $R^2 = 0.51$ ,  $p < 0.0001$ ; Fig. 6f) than N influx ( $R^2 = 0.47$ ,  $p < 0.0001$ ; Fig. 6e).

VPA supported the linear regression results and demonstrated that N was a better unique predictor than P of variance in algal biomass and composition (Fig. 7a). Nitrogen influx alone explained 8%, 10%, and 24% variance in algal communities since 1880, 1950, and 1975, respectively. Thus, N became a better predictor of algal

biomass and composition in more recent history, explaining almost one-quarter of the variance since initiation of tertiary wastewater discharge in 1975. In contrast, P sources explained only 1%, 2%, and 13% variance for the same time intervals. Interestingly, the VPA showed that the total flux of nutrients (N and P combined) accounted for 38% of the explained variance in algal communities since 1880, as well as 25% since 1950, and 20% after 1975. Thus, N and P inputs individually and in concert explained about half (47%) of the variance in the long-term change (1880–2000) in algae in Kaggfjärden. Overall, climate had little unique explanatory power early in the time series (2% since 1880), but has been more important in recent decades, explaining 14% of the variance in the algal data since 1975.

Variance partitioning analysis also suggested that the combined effects of diffuse and point sources of pollutants (Fig. 7b) explained 36% of the change in algal abundance and community composition during the past century. Moreover, the interaction between point and diffuse nutrients and climate explained an additional 9% variance. Urban nutrient influx was the most important single predictor of algal change, in particular since 1975 when it explained 35% of the change in algal biomass. Similarly, the unique effects of diffuse nutrient inputs and climate have become more evident since 1975, accounting for 22% and 8%, respectively, of the historical change in algal communities. Overall, the VPA was able to explain 60% of the historical changes in algae since 1880. Thus, the three categories included in the model accounted for almost two-

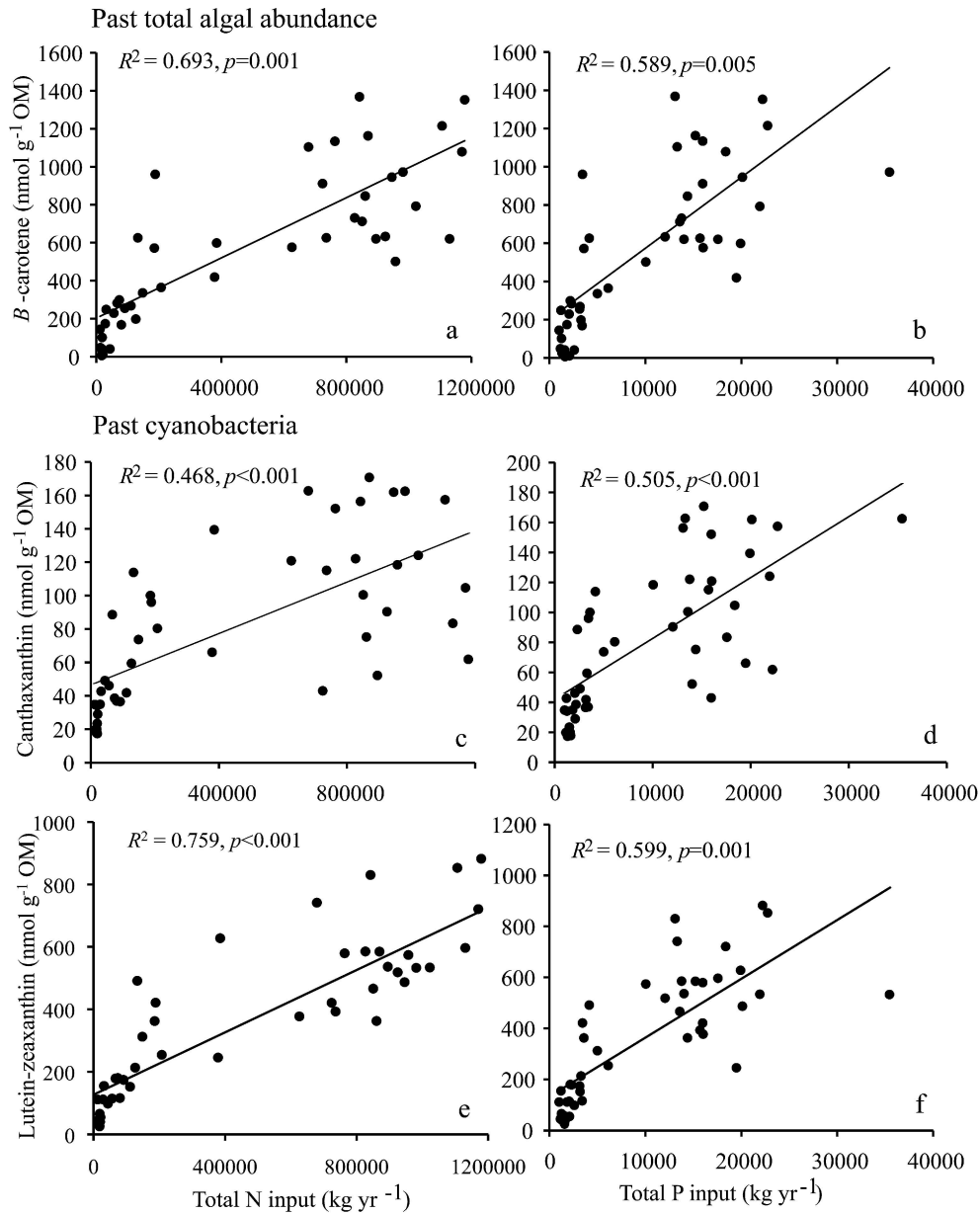


Fig. 6. Regressions between total algal abundance (as  $\beta$ -carotene) and (a) TN inputs and (b) TP inputs. Relationship between biomass of past cyanobacteria and green algae (lutein-zeaxanthin) and (c) TN inputs and (d) total P. Regressions between colonial cyanobacteria (canthaxanthin) and (e) TN and (f) TP influx.

thirds of the long-term change in algal communities and biomass.

## Discussion

Annually resolved sedimentary records of geochemistry and algal abundance and community composition revealed that nutrient sources and cycling have changed gradually in the studied Swedish coastal area since the mid-1800s, but that eutrophication effects (enhanced algal abundance) became pronounced only after 1950 concomitant with intensified farming and urbanization. Regression analyses demonstrated that N influx was a better predictor of long-

term development of coastal eutrophication than were P inputs, although variance partitioning analysis (VPA) suggested that the two nutrients often acted in concert to degrade water quality. In particular, nutrients from agricultural runoff and more recently wastewater discharge have increased algal abundance in Kaggfjärden several fold since 1950. Similarly, VPA was able to explain 60% of the total variance in algal communities since 1880, with point and diffuse nutrient sources jointly explaining over one-third (36%) of the long-term change in algal abundance and community composition. Climate variability has become more important in recent decades as a factor influencing local coastal eutrophication.

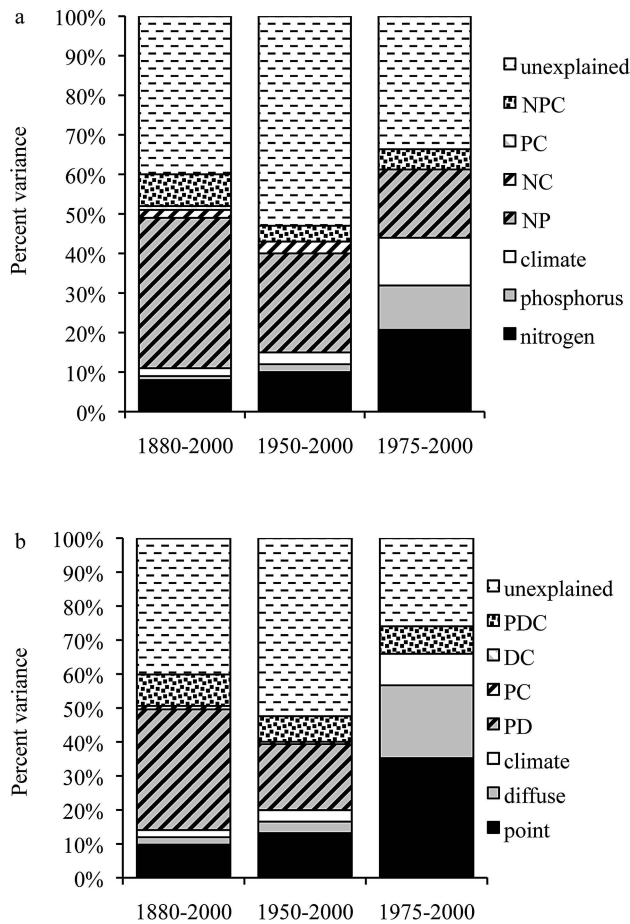


Fig. 7. Output of variance partitioning analyses to assess the relative importance of (a) nitrogen (N), phosphorus (P), and climate (C) as predictors of change in algal biomass and composition (as pigment data). (b) As above using point (P), diffuse (D), and climate (C) variables (see Table 1). The analyses were done separately for the time periods: 1880–2000, 1950–2000, and 1975–2000. The amount of explained variance and the relative contribution of the unique and interactive effects of the different factors in the model are presented as percentages.

*Inferred changes in algal abundance*—Analyses of sedimentary pigment biomarkers, C and N content, and C:N ratios ( $\sim 10$ ) suggested that OM input to sediments was predominantly of autochthonous origin and that algal abundance has increased since 1900 and substantially since 1950 (Figs. 4, 5). This enhanced algal biomass agrees with prior paleoecological studies in the Baltic that documented degraded water quality in German and Finnish coastal areas since  $\sim 1900$ , with particularly elevated nutrient loading since the 1950s (Voss et al. 2000; Clarke et al. 2006). Regression and multivariate analyses suggested that total algal abundance (as  $\beta$ -carotene) in Himmerfjärden was most strongly correlated with TN inputs ( $R^2 = 0.69$ ; Fig. 6a), including both agricultural ( $R^2 = 0.69$ ) and urban N sources ( $R^2 = 0.64$ ), while the combined effect of point and diffuse nutrients accounted for 36% of the long-term change in algal abundance and community composition (Fig. 7b). Consistent with this finding, we note that agriculture was most extensive in Sweden between 1880

and 1930; however, fertilizer use by farms was still very low (Hoffmann et al. 2000). Concomitant drainage of wetlands substantially reduced N retention by land, while after 1950 livestock density decreased and nitrogenous fertilizer use increased dramatically (Hoffmann et al. 2000). As a result, mean leaching rates and gross export of agricultural N to watercourses increased by 60% and 30%, respectively, after 1930, with both variables reaching a peak (Hoffmann et al. 2000) and a subsequent plateau (Stålnacke et al. 1999) by the mid-1970s (Fig. 2c). This increased loss of N from arable land due to increasing fertilization rates after WWII mirrors trends in Europe (Hoffmann and Johnsson 2000). Fertilizer P application rates also increased greatly from  $\sim 1950$  to  $\sim 1975$  and then decreased (Fig. 2d) because of P accumulation in agricultural soils (Andersson et al. 2000). Since agricultural N inputs ( $R^2 = 0.69$ ) were consistently better correlated to historical change in total algal biomass ( $\beta$ -carotene) than agricultural P inputs ( $R^2 = 0.32$ ), regression analyses suggest that mainly increased N availability from intensified post-WWII agricultural practices stimulated algal abundance in this coastal Baltic region (Fig. 5).

Downcore profiles of ubiquitous pigments (Chl *a*,  $\beta$ -carotene) indicated that total algal abundance peaked around the mid-1980s (Fig. 5), a pattern that is in agreement with other cores (Bianchi et al. 2002) and long-term phytoplankton monitoring data in the bay (Elmgren and Larsson 2001). Consistent with pigment profiles, sedimentary  $\delta^{13}\text{C}$  values were relatively enriched during this period (Fig. 4b), as expected when high productivity draws down the concentration of dissolved inorganic carbon (DIC) and increases algal reliance on atmospheric  $\text{CO}_2$  ( $-10\%$ ; Schelske and Hodell 1995). Since C:N ratios were stable and indicative of phytoplankton inputs ( $\sim 10$ ), we infer that there have been limited inputs of terrestrial OM (C:N  $> 20:1$ ) or ecological shifts in the dominant OM source (Long et al. 2008). Instead we conclude that the changes in biomarkers were mainly due to changes in enhanced primary production in the bay.

Pigment assemblages suggest that algal communities during the maximum of the mid-1980s were composed largely of diatoms, whose pigments (fucoxanthin, diatoxanthin) were tenfold more abundant than those of other algal groups. Elevated diatom biomass following wastewater inputs was also recorded in pigment analyses of modern sediments (Bianchi et al. 2002; Savage et al. 2004) and is consistent with monitoring records that show intense blooms of siliceous algae in Himmerfjärden, particularly in spring (Elmgren and Larsson 1997; Bianchi et al. 2002). Strong correlations between past diatom abundance (fucoxanthin + diatoxanthin) and wastewater N ( $R^2 = 0.73$ ,  $p < 0.0001$ ) and, to a lesser extent, wastewater P ( $R^2 = 0.64$ ,  $p < 0.0001$ ) support the hypothesis that urban wastewater promotes preferential development of diatom populations in coastal waters if silica is not limiting (Conley 1993). Consistent with the importance of urban point sources of nutrients as regulators of water quality, water-column monitoring data (Elmgren and Larsson 2001) and pigment stratigraphies (Fig. 5) record a decrease in algal biomass following enhanced wastewater treatment

(Fig. 2a). Similarly, improved P removal in an STP reduced Chl *a* concentrations and cyanobacterial abundance in Stockholm's inner archipelago, which receives large fresh-water inputs from Lake Mälaren (Brattberg et al. 1991). In the current study, the marked decrease in pigment concentrations after the mid-1990s (Fig. 5) following advanced wastewater N removal confirms that it is nonetheless important to decrease the supply of plant-available N to improve water quality in this coastal ecosystem.

*Effect of Baltic Sea chemical changes*—The regression and variance partitioning analyses considered only nitrogen and phosphorus inputs from urban point sources and diffuse nutrients associated with changing land-use practices in the catchment, rather than changes in the chemistry of the open Baltic Sea (Fonselius and Valderrama 2003). While inner basins of Himmerfjärden including Kaggfjärden undoubtedly receive some nutrients from the open sea due to estuarine circulation (Engqvist 1996), we did not include Baltic-derived nutrients in our analyses for four reasons. First, reliable nutrient data from the open Baltic Sea are available only since the late 1960s (Fonselius and Valderrama 2003). Second, prior studies of nutrient uptake along the main axis of Himmerfjärden demonstrate that algae assimilate, incorporate, and sediment substantial quantities of nutrients to the bottom deposits (Savage et al. 2004), while benthic denitrification removes substantial quantities of N as it is advected up the bay (Elmgren and Larsson 1997), making offshore nutrient concentrations unreliable as estimates of inputs to Kaggfjärden. Third, during the period for which comparative measurements are available (since 1977), concentrations of inorganic nitrogen (the main limiting nutrient) have been much higher inside Himmerfjärden than in the open Baltic, testifying to the paramount importance of local nitrogen sources (Elmgren and Larsson 1997). Finally, long-term water-exchange modeling shows that the average net flow from Kaggfjärden was outward into the main axis of Himmerfjärden (Engqvist and Stenström 2009), suggesting a considerable influence of nutrients from the local catchment on Kaggfjärden productivity.

*Cultural eutrophication*—Analysis of  $\delta^{15}\text{N}$  deposited in sediments during the past 200 yr revealed that the most dramatic shifts in N loading and inferred sources occurred coincident with the discharge of tertiary-treated sewage (Fig. 4a). Wastewater N loads accounted for over half ( $R^2 = 0.56$ ) of the explained variance in sedimentary  $\delta^{15}\text{N}$  values, which increased from 5.5‰ in 1974 when effluent discharge was initiated to 8.3‰ in 1990 when wastewater N loads peaked ( $8 \times 10^5 \text{ kg yr}^{-1}$ ; Fig. 2a) and were the predominant source of N in inner Himmerfjärden (Elmgren and Larsson 2001). We attribute this 3‰ enrichment in sedimentary  $\delta^{15}\text{N}$  during 15 yr to assimilation by phytoplankton of  $^{15}\text{N}$ -enriched sewage N ( $\delta^{15}\text{N}\text{-NO}_3$  [nitrates] = 26‰;  $\delta^{15}\text{N}\text{-NH}_4$  [ammonia] = 29‰ in 1988; Savage and Elmgren 2004) and its subsequent burial in sediments. Consistent with this interpretation, we have noted elsewhere that  $^{15}\text{N}$  content in surface sediments in Himmerfjärden declines linearly with distance from the wastewater outfall and hypothesized that spring diatoms are the main

vector of delivery to these sediments (Savage et al. 2004). In general,  $\delta^{15}\text{N}$  values in dissolved inorganic N (DIN) are higher for urban sources ( $9.9 \pm 2\%$ ) than agricultural inputs (3–7‰) due to intense denitrification processes in modern wastewater treatment facilities (Leavitt et al. 2006; Bunting et al. 2007). Further, the increased  $\delta^{15}\text{N}$  signature in Kaggfjärden sediments is similar to those recorded at southern Baltic coastal sites ( $\sim 13\%$ ; Struck et al. 2000; Voss et al. 2000) where there are substantial riverine inputs of urban N (Voss and Struck 1997). Together, these studies demonstrate that the past 50 yr represent the most dramatic period of alteration of N inputs to Baltic coastal areas.

Timing of changes in sedimentary  $\delta^{15}\text{N}$  at the beginning of the 1900s and the onset of clear laminations  $\sim 1950$  mirror patterns recorded at other coastal Baltic locations (Voss et al. 2000) and suggest that simultaneous changes in land use and urbanization within subbasins of the Baltic Sea catchment have initiated ecosystem change. For example, sedimentary  $\delta^{15}\text{N}$  in Kaggfjärden increased from 4.5‰ to 5.5‰ around 1900, which agrees with the timing of sedimentary  $\delta^{15}\text{N}$  changes (2.8–5.8‰) in the southern Baltic Sea (Voss et al. 2000). Similarly, sediments from the southern Baltic coast show further enrichment in  $\delta^{15}\text{N}$  values (4.3–6.5‰) after 1950 (Voss et al. 2000) that coincided with intensified industrial and agricultural activities in Germany and Poland (Larsson et al. 1985), elevated flux of dissolved nutrients in rivers, and documented increases in biological productivity (Struck et al. 2000). Interestingly, while the sediment record did not show a clear change in  $\delta^{15}\text{N}$  around 1950, this period marks the onset of permanent laminations in Kaggfjärden and other coastal areas of Sweden (Persson and Jonsson 2000). The permanent laminations were probably due to increased algal productivity and enhanced biological oxygen demand in bottom waters that favored development of deep-water anoxia and consequent limitations on bioturbation.

The sedimentary  $\delta^{15}\text{N}$  profile did not show any evidence of the gradual shift from the use of manure to synthetic fertilizers on farmland during the mid-1900s despite the  $\delta^{15}\text{N}$  value of manure often exceeding that of synthetic fertilizers immediately following application to fields (Heaton 1986). However, because microbial transformation and ammonia volatilization in catchment soils increase the  $\delta^{15}\text{N}$  of both manure and fertilizer, the degree of nitrate  $\delta^{15}\text{N}$  enrichment in drainage waters is often independent of the precise form of soil fertilization (Anderson and Cabana 2005; Voss et al. 2006). Thus while trends in the Kaggfjärden geochemical record cannot be used to distinguish fertilizer sources, the  $\delta^{15}\text{N}$  record remains essential to demonstrating increased influx of N from both agriculture and urban sources following increases in human population, urbanization, and resource use.

Sediments at the base of the core (Fig. 4a) exhibited similar isotopic signatures to those recorded in modern sediments in the central Baltic Sea (Voss et al. 2000). At this latter location, blooms of  $\text{N}_2$ -fixing cyanobacteria are thought to contribute isotopically light N ( $\sim 0\%$ ) OM to the offshore sediments (Struck et al. 2000). Not surprisingly then, we recovered detectable levels of carotenoids from cyanobacteria (e.g., canthaxanthin) from sediments

throughout the past 200 yr (Fig. 5), consistent with the hypothesis that cyanobacteria have been a common component of phytoplankton assemblages since at least the 1800s, and probably even for thousands of years (Bianchi et al. 2000a).

*Interpretation of fossil records*—Pigment profiles do not appear to have been biased substantially by changes in the preservation environment of Kaggfjärden. Although the ratio of native to degraded chlorophyll molecules increased briefly after the start-up of wastewater discharge (Fig. 5), these changes were not prolonged or directional. Similarly, there were few changes in the ratio of labile Chl *a* to stable pheophytin *a* at the onset of laminations, suggesting that there were negligible changes in pigment preservation despite development of deepwater hypoxia that restricted macrobenthic colonization and sediment disturbance. As noted by Leavitt and Carpenter (1989), bioturbation reduces the temporal resolution of time series but does not eliminate directional trends in algal abundance. Similarly, although previous analysis of pigments noted that resuspension and focusing could bias sedimentary records in Himmerfjärden (Reuss et al. 2005), that study used a core from a fully oxygenated site that lacks laminations and which experiences resuspension of lateral, shallow sediments (Elmgren and Larsson 1997). In contrast, the location of the freeze-crust core used in this study exhibits seasonal anoxia and is thus more suited to paleoecological analyses. This difference among studies confirms the need to select suitable core sites, preferably those with persistent anoxia (Bianchi et al. 2000b; Leavitt and Hodgson 2001), limited bioturbation and resuspension (Reuss et al. 2005), and reasonable proximity to sources of ecosystem disturbance. Moreover, the presence of laminations enables high-resolution sectioning that overcomes averaging of long-term trends in phytoplankton biomass (Bianchi et al. 2002).

*Management implications*—Continuing eutrophication is considered the most serious and challenging environmental problem facing the Baltic Sea (Elmgren and Larsson 2001). The keystone of eutrophication research is the concept of nutrient limitation, and the relative importance of N vs. P as drivers of eutrophication are still hotly debated (Lewis and Wurtsbaugh 2008; Schindler et al. 2008; Conley et al. 2009), particularly in Swedish Seas (Boesch et al. 2006). Our findings provide important new insights into this debate by demonstrating that the unique effects of N on total algal abundance were greater than those of P (Howarth and Marino 2006), while also suggesting that the combined influx of N and P had a powerful synergistic effect on water quality degradation (Elser et al. 2007; Conley et al. 2009). Moreover, our analysis suggests that P may have a dominant effect on the abundance of cyanobacteria (Fig. 6; Elmgren and Larsson 2001), consistent with findings from smaller boreal lake systems (Schindler et al. 2008). Taken together, these findings support management strategies that remove both N and P to achieve good ecological status in the Baltic coastal environment (Elmgren and Larsson 2001). This dual

nutrient removal may be particularly prudent in dynamic systems like the Baltic and other estuaries, where the nutrient limiting primary production may vary at different spatial and temporal scales (Granéli et al. 1990).

Using paleoecological proxies, we have shown that the combined effects of point and diffuse nutrient sources are the greatest drivers of long-term change in water quality in this study area. At centennial scales, diffuse nutrients from agriculture and urbanization have acted in concert to degrade water quality. However, in recent decades, urban wastewater has become the predominant control of environmental change in Kaggfjärden, while climatic effects have been discernable since 1975. Increased direct and interactive effects of climate may be anticipated in forthcoming decades, as elevated hydrologic variability is expected to increase the magnitude of extreme hydrologic events and their effect on terrestrial nutrient export. Because climate, land-use, and urbanization will interact most substantially at the landscape scale, we believe that it is now more imperative than ever for major regional and national stakeholders to cooperate to reduce both point and diffuse sources of nutrients to the Baltic Sea.

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