

Evaluating the effects of nutrient load reductions on the biomass of toxic nitrogen-fixing cyanobacteria in the Gulf of Finland, Baltic Sea

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The effects of nutrient load reductions on the biomass of N-fixing cyanobacteria were evaluated in the scale of the Gulf of Finland. The two analysed reduction scenarios were Finnish national agenda and the improvement of phosphorus removal in the present purification plants of St. Petersburg. The effects of load reduction scenarios were tested by using a 3D-ecosystem model with a horizontal resolution of 5 km. According to the results the Finnish national agenda cannot decrease the biomass of N-fixing cyanobacteria, but it seems to be able to reduce the total phytoplankton biomass in the coastal waters. The phosphorus reduction in St. Petersburg decreases the N-fixing cyanobacteria significantly in the central parts of the Gulf of Finland. Improvement of the phosphorus purification efficiency in the present sewage treatment plants offers us an opportunity to try to control the intensity of the toxic blooms of nitrogen-fixing cyanobacteria in the scale of the Gulf of Finland.

Introduction

During the first week of July 1997, the Gulf of Finland (GOF) became covered by floating ac-

cumulations of cyanobacteria. Two main genera responsible for the bloom were *Nodularia* and *Aphanizomenon*, which both are capable of fixing atmospheric nitrogen. *Nodularia*-species

are found to be commonly toxic in the GOF (Sivonen *et al.* 1989). The bloom lasted almost a month, over the most famous summer holiday season in July. The whole open sea became so densely covered by the potentially toxic accumulations that swimming as well as many other recreational activities were largely hampered. All this happened simultaneously with one of the warmest and sunniest summers of the century. The bloom resulted in a public debate about the ecological state of the GOF and possible actions to prevent similar blooms in the future.

The blooms of N-fixing cyanobacteria have probably been a natural part of the Baltic Sea ecosystem for thousands of years (Bianchi *et al.* 2000). One of the first bloom reports originates from the year 1884, when the prince of Monaco, Albert I, was exploring the Baltic Sea. He described the scenery in the following words: "Algal vegetation covered the whole Baltic Sea spanning from Gotland to Preuss and to the entrance of the Gulf of Finland colouring water olive green." (Pouchet and de Guerne 1885).

The growth of these blooms is controlled by the phosphorus availability (Kononen *et al.* 1994). The main natural phosphorus sources in the Baltic Sea are in the deep anoxic sediment areas (e.g. Fonselius 1969). The last big bloom in the GOF was preceded by an oxygen deficiency in the bottom water in late summer 1996 (Pitkänen and Välipakka 1997). Under anoxic conditions in the sediment–water interface phosphate is released from the sediments mostly due to reduction of oxidised iron compounds. This phenomenon is called internal loading and in 1996 it was estimated to correspond to one-year's anthropogenic load of phosphorus from the drainage area to the GOF.

The main reason for the sudden oxygen deficit was the lack of vertical mixing in winter 1995–1996 (Pitkänen and Välipakka 1997). In the following throughout mixing in winter 1996–1997, the released phosphorus load entered the surface layer increasing the average DIP concentrations by 10%–30% in the whole GOF. When the spring bloom 1997 was over, there was still plenty of excess DIP available in the surface water for the growth of N-fixing species.

Even though the hydrographical and meteorological factors largely control the outcome of

the blooms of cyanobacteria, anthropogenic eutrophication of the GOF is supposed to have a clear effect on the intensity of the blooms. Despite some decrease in the nutrient load during the 1990s, GOF is still receiving a 3 to 5 times higher nutrient load than the Baltic Sea on average (Pitkänen 1994) and there is no doubt that the origin of the phosphorus buried in the sediment is mainly anthropogenic. Furthermore, the nutrient load keeps the new biomass production and sedimentation high (Heiskanen and Kononen 1994). Decomposition of this material consumes oxygen both in the deeper water layers and on the sediment surface and accelerates the formation of anoxia during the periods of pronounced stratification.

In this paper we aim to evaluate the effects of nutrient load reduction scenarios on the biomass of the potentially toxic nitrogen-fixing cyanobacteria. The task requires development and testing of an ecosystem model. A realistic ecosystem model for the GOF includes at least descriptions of the dynamic water exchange with the Baltic Proper, oxygen concentration dependent sediment processes, a detailed pelagic food web as well as a high spatial resolution along the coastal zone and capability for long model simulations. Due to limiting computer capacity and comprehensive knowledge of the key processes all of these requirements are difficult to reach. Our choice has been to focus on one algal group, the nitrogen-fixing cyanobacteria with high spatial resolution and reasonably long model simulations. Our approach do not take into account the phenomena caused by water exchange with the Baltic Proper and changes in the oxygen conditions, but in this paper we prove that a realistic cyanobacterial model for the Gulf of Finland is possible to construct with a less complicated approach.

Material and methods

Load reduction scenarios

The only bloom promoting factor which can be controlled is the anthropogenic nutrient load. The extensive bloom of summer 1997 speeded up the preparation of a Finnish national agenda

“Water Protection Targets to 2005” (Anonymous 1998), which aims at cutting 50% of the Finnish anthropogenic load of both N and P before the year 2005. The reference years for the load reduction are set to the beginning of 1990s and therefore the targets have already partly been reached. However, the agenda is still estimated to cut the GOF entering bioavailable N-discharges by 2100 tonnes y^{-1} and P-discharges by 62 tonnes y^{-1} .

The main anthropogenic nutrient source for the GOF is the Russian city of St. Petersburg with its 4.5 million inhabitants. According to the information provided by the Russian authorities, one third of the sewage is lead directly to the GOF without any treatment. Two thirds are treated biologically. The reduction of biological oxygen demand is good in the sewage treatment process, but only ca 50% of total phosphorus is removed. According to studies on nutrient limitation in the GOF (Kivi *et al.* 1993, Pitkänen and Tamminen 1995) phosphorus seems to be the main limiting nutrient in the easternmost estuarine waters, while the role of nitrogen becomes more important towards west. However, the potentially toxic blooms of N-fixing cyanobacteria in the middle and western parts of GOF are also controlled by phosphorus (Kononen *et al.* 1994).

Treatment of the last third of the sewage requires building of a new large sewage treatment plant. The new plant would cut both the nitrogen and phosphorus load to the GOF significantly. Preliminary plans already exist but the financing is still open. Meanwhile, more phosphorus could be removed at the present sewage treatment plants by optimising the biological process (Rantanen 1994) or by using ferrosulphate precipitation. These should be relatively cheap procedures which need only minor technical changes to the present treatment plants. These processes could result in a reduction of 570 tonnes y^{-1} of bioavailable phosphorus. The Finnish national agenda can only reach roughly 10% of this.

Model development

The working principles of our ecosystem model follow the oceanic phytoplankton model de-

scribed by Tyrrell (1999). Phytoplankton is defined as two competing groups of species: nitrogen-fixing cyanobacteria and the other phytoplankton. The other phytoplankton grows faster and out-competes the N-fixers in all temperatures if both DIN and DIP are available. When DIN is consumed close to zero but DIP is still abundant, the N-fixers gain a competitive advantage. N-fixation is an energy consuming process, thus the growth of N-fixers is not effective in low temperature. Their optimum temperature for growth varies between 16–28 °C in the GOF (Kononen and Leppänen 1997). The availability of trace elements needed for the N-fixing enzyme complex is not limiting their growth in the brackish waters of GOF unlike in the open oceans (Falkowski 1997). Both algal groups use nutrients in the model according to the Redfield ratio (Redfield 1958).

All losses of biomass are described by one loss rate term which is temperature dependent. The maximum loss rate is lower for the nitrogen fixers because they are avoided by grazers due to their toxins (Sellner 1996). The nutrient and light limitation of growth rate is calculated according to the Michaelis–Menten kinetics and temperature limitations according to Frisk (1982). Light availability is limited by the presence of ice cover. The growth is further limited by shelf-shading when the actual biomass is reaching the maximum biomass describing the carrying capacity of the area. When the actual biomass reaches the minimum biomass, which mimics the over-wintering stages of the phytoplankton species, the loss rate approaches to zero and the algae cannot get extinct during the unfavourable season.

When the algal cells die, they are converted into detritus N and P, which starts to settle at a constant speed and release DIN and DIP back into the water column at temperature dependent rates. The rate of phosphorus regeneration is roughly twice as fast as nitrogen regeneration (Garber 1984). When detritus N and P reach the lowest grid cell of the 3D model, they start to be converted into sediment at constant rates, permanently out the nutrient cycle. Denitrification is not described as a separate process but included to the sedimentation term of detritus nitrogen. Ecosystem model equations, vari-

ables and parameters are presented in Tables 1–3. The previous version of the ecosystem model with only one algal group not capable for N-fixing has been presented for the GOF by Kuusisto *et al.* (1998).

The ecosystem model has been built on the top of a 3D-water quality model (Virtanen *et al.* 1986, Koponen *et al.* 1992). The model is applied to the GOF with a horizontal grid resolution of 5 km and thirteen vertical layers. The Baltic Proper facing border of the GOF is closed and no inflow is possible. There is only outflow from the GOF equal to the flow of the main rivers Neva, Narva and Kymi.

The model is run by using synoptic wind observation from the coastal weather station of the Finnish Meteorological Institute (FMI). Water temperature data for the ecosystem model are collected from the intensive monitoring sites of the Finnish Environment Institute (FEI) and light intensity as total radiation data in Helsinki weather station by FMI. Data on ice cover is provided by the Finnish Institute of Marine Research (FIMR). The transport of all variables is calculated 3-dimensionally, except for the biotic interactions between dissolved nutrients and algal biomass in the mixed surface layer. The algal biomass is presented as biomass per area [g (ww) m⁻²]. There is no algal growth and no interactions between the dissolved nutrients and algal biomass below the mixed layer. The

reason for this partly 2D-approach is the lack of information on the depth distribution of algae in the data used for the model calibration.

Calibration

The data for model calibration were collected at five intensive monitoring sites by FEI and FIMR (Fig. 1). Biomass [g (ww) m⁻³] was calculated from chlorophyll *a* measurements (mg m⁻³) with a statistical formula (Kuusisto *et al.* 1998) based on parallel measurements in the GOF. The original measurements were combined samples covering the whole euphotic zone defined as twice the Secchi depth. Thus the biomass concentrations (g m⁻³) were further converted to biomass per area (g m⁻²) by multiplying them by the depth of the euphotic zone (m).

The calibration was started with a 0-dimensional version of the growth model using the two algal groups. All data available from the five calibration sites (Fig. 1) for 1990s were used. Biomass data of N-fixing cyanobacteria was not available and therefore the model was calibrated by using total phytoplankton biomass. The model was developed with the AQUASIM programme version 2.0 (Reichert 1995, Reichert *et al.* 1995). AQUASIM is a computer programme for dynamic model development in aquatic systems. It can be used for estimation of missing or uncertain parameters in cases where sufficient material is available. The programme is able to utilise several independent data sets in the estimation process.

The estimated parameters were transferred to the 3D-model and the calibration process was continued by using total phytoplankton biomass, surface DIN and DIP data for three years: 1995, 1997 and 1998. The years were selected to represent completely different conditions for the growth of cyanobacteria. Year 1995 was the last year when practically all phosphorus was used by the spring bloom of phytoplankton and there was no marked excess DIP left for the N-fixing cyanobacteria. Year 1997 was the extensive bloom summer when the high DIP-concentrations coincided with sunny weather and exceptionally high surface water temperatures. Year 1998 started like the previous year with equally

Table 1. Model variables.

Symbol	Definition	Unit
C_C	Biomass of cyanobacteria (wet weight)	g m ⁻²
C_A	Biomass of the other algae (wet weight)	g m ⁻²
C_{DIN}	DIN concentration	mg m ⁻³
C_{DIP}	DIP concentration	mg m ⁻³
C_{Ndet}	Detritus nitrogen	mg m ⁻³
C_{Pdet}	Detritus phosphorus	mg m ⁻³
L_{DIN}	DIN load	mg m ⁻³ d ⁻¹
L_{DIP}	DIP load	mg m ⁻³ d ⁻¹
I	Total radiation	MJ m ⁻² d ⁻¹
T	Temperature	°C
Ice	Ice-cover (0.1)	–
t	Time	d
h	Depth of grid cell	m

high DIP concentrations. The summer weather was completely opposite with occasional heavy winds and rain combined with low surface temperature. Due to the weather conditions in 1998 the blooms remained moderate compared to the

previous summer. Calibration results for the westernmost (Längden) and easternmost (Huovari) sites are presented in Fig. 2. The calibrated parameters together with references are presented in Table 2.

Table 2. Model parameters

Symbol	Definition	Reference	Value	Unit
N_{inC}	Nitrogen in cyanobacteria	Redfield 1958	0.0193	–
P_{inC}	Phosphorus in cyanobacteria	Redfield 1958	0.00268	–
N_{inA}	Nitrogen in the other algae	Redfield 1958	0.0193	–
P_{inA}	Phosphorus in the other algae	Redfield 1958	0.00268	–
μ_{Cmax}	Maximal growth rate of cyanobacteria	calibration	0.5	d^{-1}
μ_{Amax}	Maximal growth rate of the other algae	Olli <i>et al.</i> 1996	0.7	d^{-1}
R_{Cmax}	Maximum loss rate of cyanobacteria	calibration	0.1	d^{-1}
R_{Amax}	Maximum loss rate of the other algae	calibration	0.15	d^{-1}
K_{NC}	Half-saturation coefficient of DIN for cyanobacteria	Tyrrell 1999	0	$mg\ m^{-3}$
K_{PC}	Half-saturation coefficient of DIP for cyanobacteria	Kononen & Leppänen 1997	2	$mg\ m^{-3}$
K_{NA}	Half-saturation coefficient of DIN for the other algae	calibration	7	$mg\ m^{-3}$
K_{PA}	Half-saturation coefficient of DIP for the other algae	calibration	1	$mg\ m^{-3}$
K_{IC}	Half saturation coefficient of radiation for cyanobacteria	calibration	20	$MJ\ m^{-2}\ d^{-1}$
K_{IA}	Half saturation coefficient of radiation for the other algae	calibration	15	$MJ\ m^{-2}\ d^{-1}$
C_{min}	Minimum biomass of cyanobacteria	calibration	0.5	$g\ m^{-2}$
A_{min}	Minimum biomass of the other algae	calibration	0.01	$g\ m^{-2}$
A_{max}	Maximum total biomass of algae	calibration	300	$g\ m^{-2}$
β_0	Maximal detritus nitrogen mineralisation rate	Garber 1984	0.018	d^{-1}
γ_0	Maximal detritus phosphorus mineralisation rate	Garber 1984	0.043	d^{-1}
V_{Ndet}	Settling rate of detritus nitrogen	Heiskanen & Tallberg 1999	1	$m\ d^{-1}$
V_{Pdet}	Settling rate of detritus phosphorus	Heiskanen & Tallberg 1999	1	$m\ d^{-1}$
S_{Ndet}	Sedimentation rate of detritus nitrogen	calibration	0.16	$m\ d^{-1}$
S_{Pdet}	Sedimentation rate of detritus phosphorus	Lehtoranta 1998	0.00	$m\ d^{-1}$
T_{opt}	Optimal temperature	Kononen & Leppänen 1997	25	$^{\circ}C$
	for the growth of cyanobacteria	calibration	15	$^{\circ}C$
	for the growth of the other algae	calibration	25	$^{\circ}C$
	for losses	calibration	25	$^{\circ}C$
	for detritus nitrogen mineralisation	Garber 1984	18	$^{\circ}C$
	for detritus phosphorus mineralisation	Garber 1984	18	$^{\circ}C$
a_T	Coefficient for temperature limiting factor			
	for the growth of cyanobacteria	calibration	1.14	–
	for the growth of the other algae	calibration	1.001	–
	for losses	calibration	1.05	–
	for detritus nitrogen mineralisation	Garber 1984	1.31	–
	for detritus phosphorus mineralisation	Garber 1984	1.60	–
I_{red}	Radiation attenuation by ice	calibration	0.5	–
h_{mix}	Depth of mixing layer	calibration	20	m

Table 3. Model equations, rates and limiting factors.

Equations

$$\frac{\partial c_C}{\partial t} = (\mu_C - R_C)c_C \quad (1)$$

$$\frac{\partial c_A}{\partial t} = (\mu_A - R_A)c_A \quad (2)$$

$$\frac{\partial c_{\text{DIN}}}{\partial t} = \beta c_{\text{Ndet}} - \mu_A N_{\text{inA}} c_A h_{\text{mix}}^{-1} - \mu_C N_{\text{inC}} c_C h_{\text{mix}}^{-1} + L_{\text{DIN}} \quad (3)$$

$$\frac{\partial c_{\text{DIP}}}{\partial t} = \gamma c_{\text{Pdet}} - \mu_A P_{\text{inA}} c_A h_{\text{mix}}^{-1} - \mu_C P_{\text{inC}} c_C h_{\text{mix}}^{-1} + L_{\text{DIP}} \quad (4)$$

$$\frac{\partial c_{\text{Ndet}}}{\partial t} = N_{\text{inA}} R_A c_A h_{\text{mix}}^{-1} + N_{\text{inC}} R_C c_C h_{\text{mix}}^{-1} - \beta c_{\text{Ndet}} - v_{\text{Ndet}} c_{\text{Ndet}} h^{-1} - S_{\text{Ndet}} c_{\text{Ndet}} h^{-1} \quad (5)$$

$$\frac{\partial c_{\text{Pdet}}}{\partial t} = P_{\text{inA}} R_A c_A h_{\text{mix}}^{-1} + P_{\text{inC}} R_C c_C h_{\text{mix}}^{-1} - \gamma c_{\text{Pdet}} - v_{\text{Pdet}} c_{\text{Pdet}} h^{-1} - S_{\text{Pdet}} c_{\text{Pdet}} h^{-1} \quad (6)$$

Rates

$$\mu_C = \mu_{\text{Cmax}} f_{\text{CN}}(c_{\text{DIN}}, c_{\text{DIP}}) f_{\text{CI}}(I) f(T) f_{\text{AC}}(c_A, c_C) \quad (7)$$

$$\mu_A = \mu_{\text{Amax}} f_{\text{AN}}(c_{\text{DIN}}, c_{\text{DIP}}) f_{\text{AI}}(I) f(T) f_{\text{AC}}(c_A, c_C) \quad (8)$$

$$R_C = R_{\text{Cmax}} f(T) (c_C - C_{\text{min}}) / c_C \quad (9)$$

$$R_A = R_{\text{Amax}} f(T) (c_A - A_{\text{min}}) / c_A \quad (10)$$

$$\beta = \beta_0 f(T) \quad (11)$$

$$\gamma = \gamma_0 f(T) \quad (12)$$

Limiting factors

$$f_{\text{CN}}(c_{\text{DIN}}, c_{\text{DIP}}) = \frac{c_{\text{DIN}}}{c_{\text{DIN}} + K_{\text{NC}}} \times \frac{c_{\text{DIP}}}{c_{\text{DIP}} + K_{\text{PC}}} \quad (13)$$

$$f_{\text{AN}}(c_{\text{DIN}}, c_{\text{DIP}}) = \frac{c_{\text{DIN}}}{c_{\text{DIN}} + K_{\text{NA}}} \times \frac{c_{\text{DIP}}}{c_{\text{DIP}} + K_{\text{PA}}} \quad (14)$$

$$f(T) = \exp \left[\int_{T_{\text{opt}}}^T \ln \theta dT \right], \text{ where } \theta = a_T + (1 - a_T) T / T_{\text{opt}} \quad (15)$$

$$f_{\text{CI}}(I) = \frac{I(1 - \text{Ice}I_{\text{red}})}{I(1 - \text{Ice}I_{\text{red}}) + K_{\text{IC}}} \quad (16)$$

$$f_{\text{AI}}(I) = \frac{I(1 - \text{Ice}I_{\text{red}})}{I(1 - \text{Ice}I_{\text{red}}) + K_{\text{IA}}} \quad (17)$$

$$f_{\text{AC}}(c_A, c_C) = 1 - \frac{c_A + c_C}{A_{\text{max}}} \quad (18)$$

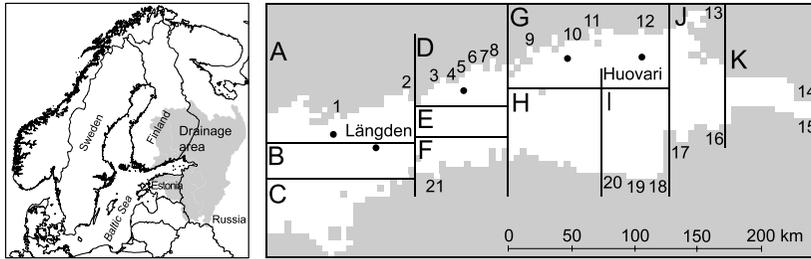


Fig. 1. The calculation area for the 3D model of the Gulf of Finland, the Baltic Sea. Grid size of the map is 5 km. The area is divided into 11 parts (A–K) for starting values of the nutrient concentrations, which are presented in Table 5. The most important point sources are presented as numbers (1–21) and their nutrient load is given in Table 4. The five sampling stations used for model calibration are presented as black circles. Calibration results of the westernmost (Längden) and the easternmost station (Huovari) are given in Fig. 2.

Validation

The N-fixing cyanobacteria are the only species forming floating accumulations in the open GOF. Most of these accumulations are formed by *Nodularia*. *Aphanizomenon* is more common close to the coastline, but it often grows in the deeper water layers contributing only a minor part to the floating accumulations (Kononen *et al.* 1998). The *Nodularia* accumulations can be detected in satellite images during cloudless conditions (Kahru *et al.* 1994). The model was validated by using NOAA satellite image interpretations from the on-line data bank of the Stockholm University (Naturgeografiska institutionen) (<http://www.marin.natgeo.su.se/~ab/>). Representative sets of images were available for two sunny summers 1997 and 1999. The number of images with detected accumulations in the GOF were 16 and 13, respectively. The combination of these image interpretations was compared with the modelled mean biomass of N-fixing cyanobacteria for the corresponding periods.

Simulating the effects of nutrient load reductions

The calibration runs lasted only one year and each of them was started with a separate set of starting values. There is no reason to run cali-

bration of several years in a sequence, because the model at its present state cannot simulate the oxygen dependent sediment processes like the massive phosphorus release in 1996–1997. For simulating the effects of nutrient load reductions, one year is not a long enough period to demonstrate the effects in the scale of GOF, where the theoretical residence time of water is close to 3 years. (e.g. Alenius *et al.* 1998). Thus, we have to extend the simulations to last clearly longer than the calibration periods — even though this increases the uncertainty of the results considerably. The uninterrupted six-year simulations were started with initial nutrient concentrations measured in January–March 1999. The simulations ended to the year 2005, which is the target year of the Finnish national agenda. The simulations were run by using real weather data from 1993–1998.

The water quality model uses precalculated flow fields and wind data for the computation of hydrodynamics. The precalculated fields are flow fields for certain steady wind conditions. They are used in combination with real wind data to calculate an approximation of the real flow field. In addition to the wind induced flows, also the riverine fresh water imposes a flow component to the GOF. Only the largest rivers are included and their river flow dependent flow fields are added to the calculation. The procedure is explained in detail by Virtanen and Koponen (1985).

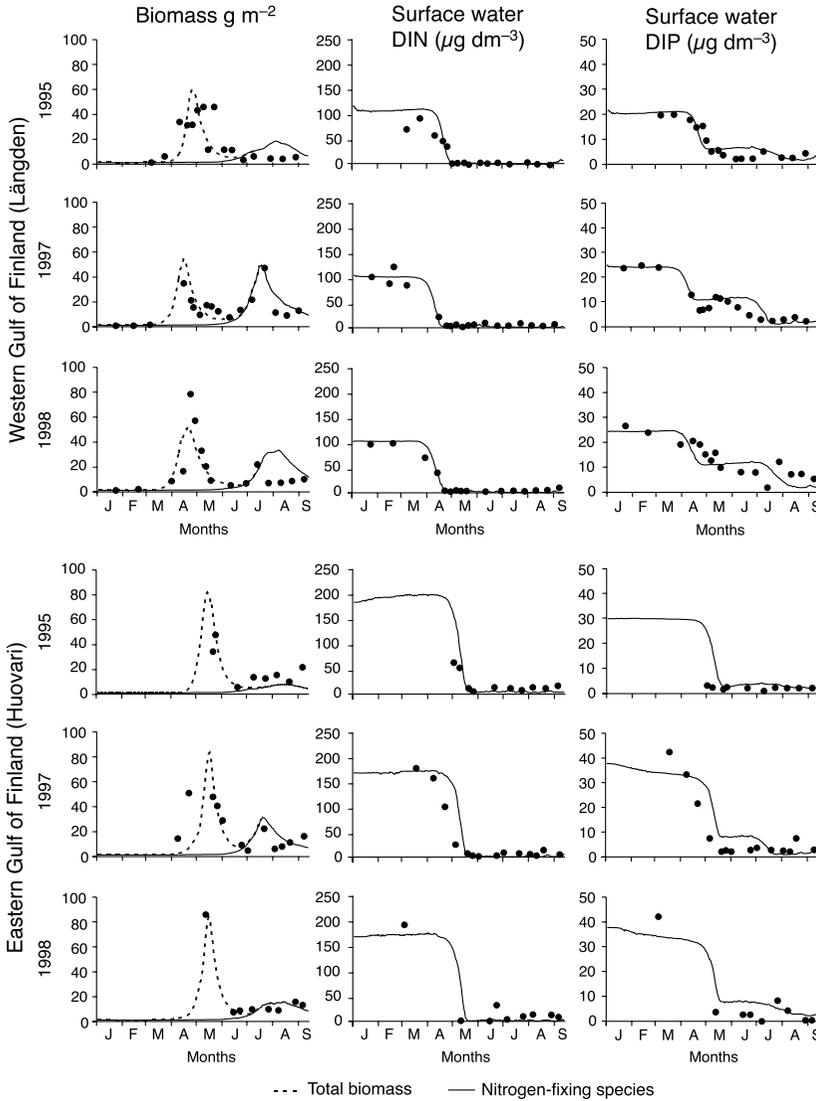


Fig. 2. Example of the ecosystem model calibration at two intensive monitoring sites on the western and the eastern Gulf of Finland (see Fig. 1). The calibrated parameters were total phytoplankton biomass and surface water DIN and DIP. Observations are described as black circles and model results as lines.

This approximation requires much less computing time compared to online flow calculation making long simulation periods possible.

Both the wind induced flow fields and the river flow fields are calculated separately for thermally stratified and non-stratified conditions representing summer and winter periods. In summer (March–September) the thermocline is set permanently to the depth of 12 meters. The end concentration fields for algae, soluble nutrients and detritus nutrients were saved when the set of flow fields was changed and these concentration fields were used for the starting values for calculating the following season.

To be able to run 6-year simulations we had to make a presumption that with the present nutrient load and without any significant changes in the internal loading, GOF is in a steady state, in which the nutrient input and output are equal. When the load is cut, output increases input and the nutrient concentrations start to decrease. In the model, the steady state was set up by calibrating the sedimentation rate of detritus nitrogen so that the nitrogen levels remained stable throughout the simulation period with the present nutrient load. The steady state assumption is close to the real situation in the 1990s when the rise in the DIN concentrations have

been stabilising or even a slight decrease have been observed (Perttilä *et al.* 1995, Anonymous 1999, Kauppila and Bäck 2000). The main reason for this change is supposed to be a decreased N-load due to the collapse of the Soviet Union. The net sedimentation rate of detritus phosphorus was assumed to be close to zero (Lehtoranta 1998) due to the poor oxygen conditions, so that the only output of phosphorus was the outflow from the GOF. This assumption made it possible to keep the phosphorus concentrations in the model simulation at the relatively high level of the late 1990s.

The simulation was first run with the present load and then with reduced loads according to the chosen scenarios. The average biomasses of both algal groups were recorded for the last growing season (March–September) of the 6th calculation year. The biomass of each scenario was then compared with the biomass calculated by using the present load. The results are presented as relative change in the average biomass of the whole growing season. Due to the simplicity and possible semiquantitative nature of the model, we decided to present the results, instead of actual percentages, to slight (< 15%) or significant (> 15%) changes. Changes lower than 2% were neglected.

Nutrient load and starting values

Information on the loads of total N and P from point sources and via rivers from Finland to the GOF was obtained from the databases maintained by FEI. The total nutrients were converted into bioavailable nutrients by using results of algal tests (Ekholm 1994, Ekholm and Krogerus 1998). The main nutrient sources to the GOF are the from the city of St. Petersburg and the river Neva discharges to the eastern part of the GOF. Information on their bioavailable nutrient discharges are based on the river flow measurements, nutrient analysis and availability tests carried out in co-operation between Russian research institutes, environmental authorities and FEI. Other nutrient load information from Russian and Estonian sources are obtained directly from the local environmental authorities under the Estonian–Finnish–Russian co-operation

(Pitkänen *et al.* 1997). Atmospheric supply of nitrogen is based on model calculations (Bartnicki *et al.* 1998). The annual nutrient discharges for 1998 are presented in Table 4 together with the nutrient load reduction scenarios. For model input the discharges have been converted to monthly values which take into account seasonal variation in river flow.

The starting values of DIN and DIP concentrations were given for 11 sub-areas of GOF (Fig 1). Different sets of starting values were used for years 1995, 1997/1998 and 1999 (Table 5) because of the benthic DIP release in late summer 1996. Starting values are based on winter measurements by FEI and FIMR during January–March when the water column has been vertically well mixed down to the permanent halocline. Sampling stations do not cover the whole GOF evenly. Best data sets were available from the Finnish coastal waters and from the middle and western GOF. The available information from the Estonian and Russian coastal waters was scarce.

Results

Validation

The nutrient starting values used for validation as well as calibration were most comprehensive along the Finnish coast. In 1997 some measurements were available from the Estonian and Russian territorial waters, but we lacked all near-coast measurements. In 1999 the lack of current information was even worse and we had to use the 1997/1998 starting values for the southern and eastern sides of the GOF. These values were most likely too high for DIP, because in other parts of the GOF a clear decreasing trend was observed. Therefore the quantitative validation was carried out only for the northern side of GOF where the best sets of starting values were available.

The satellite detected frequency of the N-fixing cyanobacteria accumulations (mainly *Nodularia*) was generally twice as high in 1997 as in 1999 (Fig. 3). The clearest change in the distribution of the accumulations between the validation years can be seen in the coastal areas of the

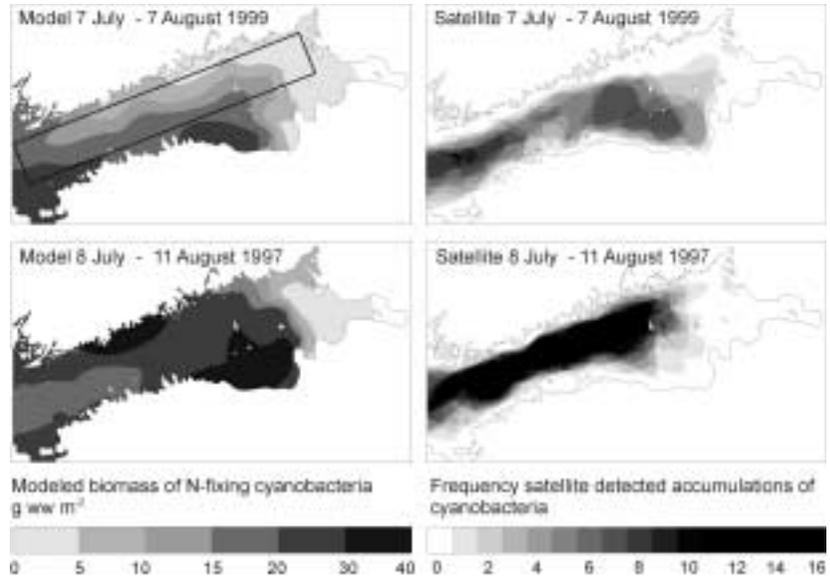
Table 4. Bioavailable nutrient load to the Gulf of Finland (GOF). The present load is based on data and estimates for the year 1998. The load reductions “Finnish national agenda” and “P-removal in St. Petersburg” are calculated on the base of this data. The load reduction data is presented only for those sources where reductions take place.

Number in Fig. 1	Source	Type of discharge	Present (1998)		Finnish national agenda		P-removal in St. Petersburg	
			DIN t y ⁻¹	DIP t y ⁻¹	DIN t y ⁻¹	DIP t y ⁻¹	DIN t y ⁻¹	DIP t y ⁻¹
1	Karjaanjoki	river	422	7	304	5	–	–
2	Kirkkonummi	municipal	73	0	41	0	–	–
3	Espoo	municipal	447	5	443	5	–	–
4	Helsinki	municipal	1 311	13	1 197	7	–	–
5	Vantaanjoki	river	934	24	708	17	–	–
6	Porvoo	municipal	92	0	18	0	–	–
7	Mustijoki	river	256	7	186	6	–	–
8	Porvoonjoki	river	900	16	578	11	–	–
9	Koskenylänjoki	river	146	4	116	3	–	–
10	Kymijoki	river	3 130	64	2 555	44	–	–
11	Kotka	municipal/industrial	158	4	27	4	–	–
12	Virojoki	river	102	3	84	2	–	–
13	Viipuri	municipal/river	380	40	–	–	–	–
14	Neva	river	22 000	500	–	–	–	–
15	St. Petersburg	municipal	16 500	1 980	–	–	16 500	1 410
16	Koporskaja	river	370	55	–	–	–	–
17	Luga	river	1 580	45	–	–	–	–
18	Narva	municipal/industrial	520	45	–	–	–	–
19	Narvajoki	river	2 780	390	–	–	–	–
20	Kohtla-Järve	municipal/industrial	2 090	30	–	–	–	–
21	Tallin	municipal	1 100	35	–	–	–	–
	Finland non-point		2 103	83	1 712	64	–	–
	Estonia non-point		5 300	245	–	–	–	–
	Atmospheric		17 250	0	–	–	–	–
	Total discharge to GOF		79 944	3 595	77 839	3 533	79 944	3 025
	Change				-2 105	-62	0	-570

Table 5. Starting values for the bioavailable nutrients in the Gulf of Finland. Values are based on measurement carried out between 1 January and 31 March and represent well mixed winter conditions. Missing values replaced with earlier measurements are written in italics.

Area code in Fig 1.	No. of sampling stations	1995		1997 and 1998		1999	
		DIN (mg m ⁻³)	DIP (mg m ⁻³)	DIN (mg m ⁻³)	DIP (mg m ⁻³)	DIN (mg m ⁻³)	DIP (mg m ⁻³)
A	4	120	22	105	25	148	31
B	4	89	18	97	23	108	30
C	2	100	19	87	21	81	20
D	40	194	30	188	34	167	37
E	3	118	26	118	32	104	30
F	1	141	25	109	26	<i>109</i>	<i>26</i>
G	8	181	30	168	38	202	35
H	3	142	29	156	41	160	33
I	1	150	28	122	29	<i>122</i>	<i>29</i>
J	3	220	31	200	33	<i>200</i>	<i>33</i>
K	0	<i>250</i>	<i>30</i>	<i>250</i>	<i>30</i>	<i>250</i>	<i>30</i>

Fig. 3. Validation of N-fixing cyanobacteria during the bloom periods of summers 1997 and 1999. Modelled biomass and frequency of satellite detected accumulation in the Gulf of Finland. The satellite images are processed and interpreted by the Naturgeografiska institutionen of the Stockholm University. The area used for quantitative validation (Fig. 4) is indicated by a rectangle.



eastern GOF. In 1997, a high frequency of accumulations followed the coastline to the Finnish–Russian border. In 1999, this area remained practically free of cyanobacteria accumulations. The model was generally able to reproduce the observed spatial distribution and annual variation in the accumulations. In the open GOF, the model was able to explain 50% of the observed pattern (Fig. 4). Along the coast the predicted biomass was generally too high.

3.2. Scenarios

According to the model results, the Finnish national agenda doesn't seem to be an effective way to fight against accumulations of cyanobacteria (Fig. 5). The results indicate that there may be a risk for slight biomass increase in the N-fixing cyanobacteria along the densely populated part of the Finnish coastline. However, the national agenda will decrease the total phytoplankton biomass in the vicinity of the Finnish coastline, but its effect to the whole GOF is practically negligible. The coastal biomass decrease takes place both during the spring and the summer period (Fig. 6A)

The model results indicate that phosphorus removal in St. Petersburg would affect the biomass of N-fixing cyanobacteria practically in the

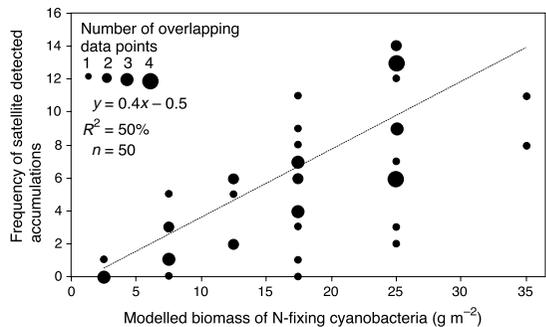


Fig. 4. Quantitative validation of N-fixing cyanobacteria during the bloom periods of summers 1997 and 1999. Modelled biomass versus frequency of satellite detected accumulation in the Gulf of Finland. The area used for the quantitative validation is indicated by a rectangle in Fig. 3.

whole GOF (Fig 5). Their biomass will decrease mainly in the central parts of GOF but the positive effects might be detectable even at the western border of the calculation area. The effect will be negligible outside of St. Petersburg, because this area is mainly phosphorus limited and thus outside the distribution of N-fixers. The total phytoplankton biomass will decrease in the whole open GOF except for a narrow zone of slight increase across the eastern GOF. The spring bloom biomass is decreasing slightly even in this zone. The main increase

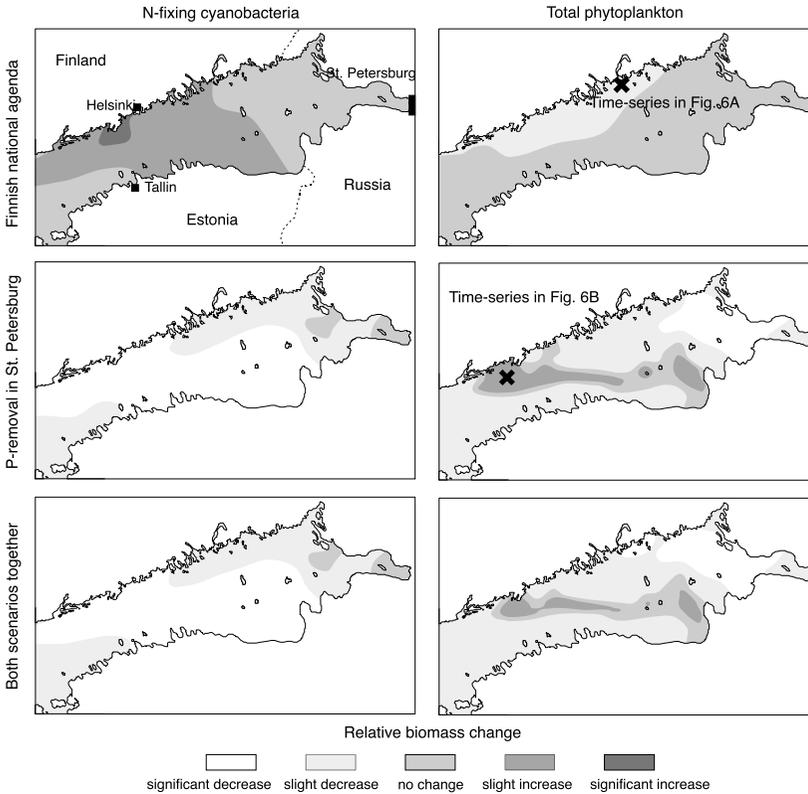


Fig. 5. The relative effect of the nutrient load reductions on the biomass of N-fixing cyanobacteria and the total phytoplankton biomass (N-fixing cyanobacteria + the other phytoplankton) in the Gulf of Finland. Finnish national agenda aims at 50% reduction of both N and P discharges in Finland compared to the level of early 1990s. P-removal in St. Petersburg reduces only the P-load but it does not have any effect on the N-load.

takes place in the summer (Fig. 6B).

If both nutrient reduction scenarios are combined, their adverse effects seem to diminish. The possible increase in the biomass of N-fixing cyanobacteria due to the Finnish national agenda can no longer be detected in the model simulation if the scenarios are combined. Furthermore, the increase in the total phytoplankton biomass in the zone across the GOF will decrease.

Discussion

According to the validation made with satellite images, the outcome of our model appeared to be reasonably good in simulating the biomass of the toxic genus *Nodularia*. Finding good validation data proved to be difficult due to the extremely patchy character of the blooms. Thus routine water sampling rarely is suitable for estimating the magnitude of cyanobacterial blooms. The semiquantitative validation data

gathered by using satellite images proved to be useful until other means of automated data collection are available.

The validation runs lasted only one growing season and they were started with separate year-specific sets of nutrient starting values. The good results make us believe that the outcome of the blooms is highly dependent on the starting values, especially on the amount of excess DIP left over when the spring bloom has consumed nutrients according to the Redfield ratio. However, if temperature is not high enough for the growth of cyanobacteria, all excess DIP cannot be utilised and the bloom do not reach its maximum potential biomass. The starting value dependence offers us a possibility to use the model for short-term forecast purposes. The winter nutrient concentrations can be used to forecast the magnitude of the blooms for the coming summer.

The most serious problem in the bloom modelling was obvious along the coastline. In our validation data the blooms seemed to be

absent along the coast where, however, high biomasses were shown by the model. One explanation for the mismatch might be difficulties to detect the biomass close to the coastline and in the archipelago from the satellite images. The main genus close to the coastline is commonly *Aphanizomenon*, which do not form floating accumulations like *Nodularia* (Kononen *et al.* 1998). If *Aphanizomenon* is not visible in the satellite images, it is possible that the model results are not as far from the reality as appears in our validation. An other possible explanation is the low sedimentation rate of detritus phosphorus used in the model. This may cause too high recycling and availability of phosphorus in the shallow areas, where detritus cannot settle below the productive zone, promoting the growth of N-fixing cyanobacteria.

The model results indicate that N-dominated nutrient load reductions, like the Finnish national agenda, cannot decrease the biomass of N-fixing toxic cyanobacteria, but can even have a slight increasing effect. Some experiments support this results (Elmgren and Larsson 1997). According to the model, the possible increase will occur mainly in the coastal area where the biomass of N-fixing cyanobacteria is generally lower than on the open sea. The reason for the lower biomass can be too high N:P ratio or also the influence of the bacterial production. As bacteria are superior competitors with regard to DIP uptake and they are often P-limited in the coastal waters, they may alter the P dynamics considerably (Heinänen and Kuparinen 1992). Thus the interactions of bacteria and DIP are difficult to predict in a simple model simulation. Near the coastline the combination of additional DIP and a high availability of dissolved organic substances may at least partly compensate the simulated increase by the increased phosphorus uptake and production of bacteria. An other process able to compensate the increase in the N-fixing cyanobacteria is the influence of the decreasing total phytoplankton biomass to the oxygen consumption and P-release from the deep bottom areas. The present model do not describe this connection.

In winter, when the algal growth is close to zero, the nutrient retention capacity of the Neva estuary is negligible in the model simulations.

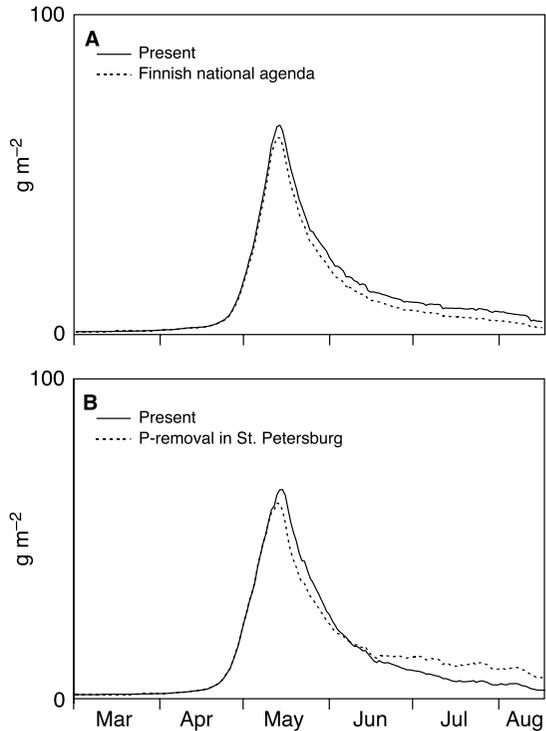


Fig. 6. Time-series of the influence of the nutrient load reductions to the seasonality of the other phytoplankton (total phytoplankton without N-fixing cyanobacteria). Both time series represent the last 6th calculation year. — **A:** the area where the influence of the Finnish national agenda is clearly detectable. — **B:** the area with the highest increase in the total phytoplankton biomass caused by P-load reduction in St. Petersburg. The exact locations of the time-series points are presented in Fig 5.

The same phenomenon has already been demonstrated by Savchuk (2000). Especially during the ice covered period, when the wind induced vertical mixing is excluded, the fresh water flow from the river Neva can often be detected along the Finnish coastline (e.g. Alenius *et al.* 1998). In spring when the ice-cover breaks-up and the phytoplankton production starts, the nutrients carried by this flow are rapidly consumed by the spring bloom. The P-load reduction seems to alter the N:P ratio of this westward flowing water mass so that less DIP is left for the cyanobacteria after the spring bloom of phytoplankton. We believe that this is the main process explaining the model indicated significant decrease in the cyanobacteria biomass in the

central parts of GOF.

According to the model results, both scenarios seem to decrease the spring bloom of phytoplankton in their impact area. The biomass produced during the spring bloom is not grazed effectively and it is responsible for the main oxygen consuming load to the sediment bottoms (Heiskanen and Kononen 1994). Thus the both scenarios should have a positive effect also on deep sediment bottoms. However, the spring bloom dynamics is difficult to mimic in a simple phytoplankton model. In the Baltic Sea several of the most important cold-water diatoms and dinoflagellates produce cysts, which settle rapidly at the end of the bloom (Heiskanen and Kononen 1994). These cysts contain a large quantity of high-energy compounds for the resting period. They are a mode of long-term burial of both organic carbon and nutrients, which complicate the modelling of nutrient circulation. It is very difficult to estimate what happens to the cyst production when there are less nutrients available. It is even possible that the cyst production is stimulated leading to a higher loss of nutrients and carbon out of the system (Kuosa *et al.* 1997).

The model results are sensitive to the N:P ratio of the algae. In our model this ratio is fixed to the widely accepted and used Redfield ratio, 7.2 by weight. If the algal community is able to accommodate its N:P ratio or to perform luxury uptake of the less limiting nutrient, the above described negative effects of the nutrient load reductions may not be any more valid. This kind of alternations from the Redfield ratio have been reported in diatoms, which are an important part of spring bloom in the Baltic (Istvanovics *et al.* 1994, Glibert *et al.* 1995). Whether these processes are important in the case of the GOF is not known. Therefore the model results should be considered rather as caricatures of the changes caused by the load reductions. At least the negative effects are more likely to be over than underestimates.

Phenomena occurring in the spatial scale close to the 5 km horizontal model resolution cannot be reliably simulated due to so called numerical diffusion. This means that running the model in a coarse resolution induces excess diffusion to naturally sharp boundaries found

e.g. in the coastal zone and archipelago. In the case of the Finnish national agenda, the model probably spreads its effects too far from the coast and therefore underestimates the biomass changes along the coastal zone. The model indicated slight reduction in the total phytoplankton biomass is likely to be a clear underestimate for areas close to the most important Finnish point sources and river mouths.

In the model results, reduction in the phosphorus load from St. Petersburg decreases nitrogen retention in the phosphorus limited Neva estuary during the growing season and thus promote export of nitrogen westward in summer. This process have been documented in a smaller scale in the Stockholm archipelago (Brattberg 1986). We believe that the zone of increased total phytoplankton biomass shown in Fig. 5 is formed when this nitrogen load from the Neva estuary meets the nitrogen limited parts of the GOF. The location of this zone is not necessarily fixed to a certain part of the GOF. In our test runs with altered weather data or starting values, it moves between the eastern and the central parts of the GOF.

In recent model simulations dealing with the effects of the nutrient load reductions in the GOF (Kuusisto *et al.* 1998, Inkala and Pitkänen 1999, Savchuk and Wulff 1999, Savchuk 2000), the authors have concluded that the most effective way to reduce eutrophication in the Gulf of Finland is to cut nitrogen load. Only in the Neva estuary the cutting of phosphorus load is found to have positive effects. Our results are not in conflict with these previous results. In the mainly nitrogen limited GOF, nitrogen removal is without doubt the most effective means to cut the primary production and to slow down the progress of eutrophication. However, the potentially toxic blooms of cyanobacteria in the open GOF are mainly phosphorus limited and their occurrence is dependent on the summertime DIP availability in the surface water. Even though our model approach do not take into account the water exchange with the Baltic Proper, it seems to be clear that phosphorus removal will affect the N:P ratio to a less favourable direction for the nitrogen fixing cyanobacteria.

Management guidance

The effective nutrient load reduction in Finland has started by the chemical phosphorus precipitation. By now, phosphorus load from most of the point sources is minimal and only diffuse load mainly from agriculture can be further diminished. The logical next step is reduction of the nitrogen load. Nitrogen dominated load reductions in Finland do not necessarily affect the biomass of N-fixing toxic cyanobacteria, but they decrease the total phytoplankton biomass close to the coastline. The phosphorus load can still be cut significantly in the city of St. Petersburg. According to our results, this cutting might be a realistic option to decrease the biomass cyanobacteria in the GOF. Combination of improved nitrogen removal in Finland and phosphorus removal in St. Petersburg would decrease both the toxic blooms of cyanobacteria and the total phytoplankton biomass along the Finnish coastline.

We have to remember that blooms of cyanobacteria are a natural part of the Baltic Sea ecosystem. We may be able to manipulate their intensity but we cannot stop them. Their occurrence is not only dependent on the absolute nutrient load or sea water nutrient concentrations but also the N:P ratio. Therefore the decrease in the total nutrient load do not necessarily guarantee a decrease in their outcome.

References

- Alenius P., Myrberg K. & Nekrasov A. 1998. The physical oceanography of the Gulf of Finland: a review. *Boreal Env. Res.* 3: 97–125.
- Anonymous 1998. Water Protection Targets to 2005. *Suomen ympäristö* 226: 1–82. [In Finnish with English summary].
- Anonymous 1999. Nutrients in European ecosystems. *Environmental assessment report* 4: 1–155. European Environment Agency.
- Bartnicki J., Barrett K., Tryso S., Erdman L., Gusev A., Dutchak S., Pekar M., Lükewille A. & Krognos T. 1998. Atmospheric supply of nitrogen, lead, cadmium, mercury and lindane to the Baltic Sea. *EMEP/ MSC-W* Note 3/98.
- Bianchi T.S., Westman P. & Rolff C. 2000. Cyanobacterial blooms in the Baltic Sea: Natural or human induced? *Limnology and Oceanography* 45: 716–726.
- Brattberg G. 1986. Decreased phosphorus loading changes phytoplankton composition and biomass in the Stockholm archipelago. *Vatten* 42: 141–153.
- Eckholm P. 1994. Bioavailability of phosphorus in agriculturally loaded rivers in southern Finland. *Hydrobiologia* 287: 179–194.
- Eckholm P. & Krogerus K. 1998. Bioavailability of phosphorus in purified municipal wastewaters. *Water Research* 32: 343–351.
- Elmgren R. & Larsson U. 1997. Himmerfjärden. Changes in a nutrient enriched coastal ecosystem. *Naturvårdsverket Rapport* 4565: 1–197. [In Swedish with English summary].
- Falkowski P.G. 1997. Evolution of the nitrogen cycle and its influence on the biological sequestration of CO₂ in the ocean. *Nature* 387: 272–275.
- Fonselius S.T. 1969. Hydrography of the Baltic deep basins III. *Fishery Board of Sweden, Series Hydrography*, Report 23.
- Frisk T. 1982. An oxygen model for Lake Haukivesi. *Hydrobiologia* 86: 133–139.
- Garber J.H. 1984. Laboratory study of nitrogen and phosphorus remineralisation during the decomposition of coastal plankton and seston. *Estuarine, Coastal and Shelf Science* 18: 685–702.
- Glibert P.M., Conley D.J., Fischer T.R., Harding L.W.Jr. & Malone T.C. 1995. Dynamics of the 1990 winter/spring bloom in Chesapeake Bay. *Marine Ecology Progress Series* 122: 27–43.
- Heinänen A. & Kuparinen J. 1992. Response of bacterial thymidine and leucine incorporation to nutrient (NH₄, PO₄) and carbon (sucrose) enrichment. *Archiv für Hydrobiologie* 37: 241–251.
- Heiskanen A.-S. & Kononen K. 1994. Sedimentation of vernal and late summer phytoplankton communities in the coastal Baltic Sea. *Archiv für Hydrobiologie* 131: 175–198.
- Heiskanen A.-S. & Tallberg P. 1999. Sedimentation and particulate nutrient dynamics along a coastal gradient from a fjord-like bay to the open sea. *Hydrobiologia* 393: 127–140.
- Inkala A. & Pitkänen H. 1999. The effect of load reductions on algal biomass in the eastern Gulf of Finland estimated by the FINNALGA model. *Boreal Environment Research* 4: 357–366.
- Istvanovics V., Padisak J., Petterson K. & Pierson D.C. 1994. Growth and phosphorus uptake of summer phytoplankton in Lake Erken (Sweden). *Journal of Plankton Research* 16: 1167–1196.
- Kahru M., Horstmann U. & Rud O. 1994. Satellite detection of increased cyanobacteria blooms in the Baltic Sea: Natural fluctuation or ecosystem change? *Ambio* 23: 469–472.
- Kauppila P. & Bäck S. 2001. State of the Finnish coastal

- waters in the 1990's. *Finnish Environment* 427, 134 pp.
- Kivi K., Kaitala S., Kuosa H., Kuparinen J., Leskinen E., Lignell R., Marcussen B. & Tamminen, T. 1993. Nutrient limitation and grazing control of the Baltic plankton community during annual succession. *Limnology and Oceanography* 38: 893–905.
- Kononen K., Lahdes E.O. & Grönlund L. 1994. Physiological and community responses of summer plankton to nutrient manipulation in the Gulf of Finland (Baltic Sea) with special reference to phosphorus. *Sarsia* 78: 243–253.
- Kononen K. & Leppänen J.-M. 1997. Patchiness, scales and controlling mechanisms of cyanobacterial blooms in the Baltic Sea: Application of a multiscale research strategy. In: Kahru M. & Brown C.W. (eds.), *Monitoring algal blooms: New techniques for detecting large-scale environmental change*. Landes Bioscience, Austin, pp. 63–84.
- Kononen K., Hällfors S., Kokkonen M., Kuosa H., Laanemets J., Pavelson J. & Autio R. 1998. Development of a subsurface chlorophyll maximum at the entrance to the Gulf of Finland, Baltic Sea. *Limnology and Oceanography* 43: 1089–1106.
- Koponen J., Alasaarela E., Lehtinen K., Sarkkula J., Simbirowicz P., Vepsä H. & Virtanen M. 1992. Modelling dynamics of large sea area. *Publications of the Water and Environment Research Institute* 7: 1–91.
- Kuosa H., Autio R., Kuuppo P., Setälä O. & Tanskanen S. 1997. Nitrogen, silicon and zooplankton controlling the Baltic spring bloom: an experimental study. *Estuarine, Coastal and Shelf Science* 45: 813–821.
- Kuusisto M., Koponen J. & Sarkkula J. 1998. Modelled phytoplankton dynamics in the Gulf of Finland. *Environmental Modelling & Software* 13: 461–470.
- Lehtoranta J. 1998. Net sedimentation and sediment-water nutrient fluxes in the eastern Gulf of Finland (Baltic Sea). *Vie Milieu* 48: 341–352.
- Olli K., Heiskanen A.-S. & Seppälä J. 1996. Development and fate of *Eutreptiella gymnastica* bloom in nutrient enriched enclosures in the coastal Baltic Sea. *Journal of Plankton Research* 18: 1587–1604.
- Perttilä M., Niemistö L. & Mäkelä K. 1995. Distribution, development and total amounts of nutrients in the Gulf of Finland. *Estuarine, Coastal and Shelf Science* 41: 345–360.
- Pitkänen H. 1994. Eutrophication of the Finnish coastal waters: Origin, fate and effects of riverine nutrient fluxes. *Publications of the Water and Environment Research Institute* 18: 1–45.
- Pitkänen H., Kondratyev S., Lääne A., Gran V., Kauppila P., Loigu E., Marcovets I., Pachel K. & Rummyantsev V. 1997. Pollution load on the Gulf of Finland from Estonia, Finland and Russia in 1985–1995. *Suomen ympäristökeskuksen monistesarja* 105: 9–18. Finnish Environment Institute, Finland.
- Pitkänen H. & Tamminen T. 1995. Nitrogen and phosphorus as production limiting factors in the estuarine waters of the eastern Gulf of Finland. *Marine Ecology Progress Series* 129: 283–294.
- Pitkänen H. & Välipakka P. 1997. Extensive deep water oxygen deficit and benthic phosphorus release in the eastern Gulf of Finland in late summer 1996. *Suomen ympäristökeskuksen monistesarja* 105: 51–59. Finnish Environment Institute, Finland.
- Pouchet G. & de Guerne J. 1885. Sur le faune pelagique de la mer Baltique et du Golfe de Finlande. *Comptes Rendus des Seances de l'Academie des Sciences* 100: 919–921.
- Rantanen P. 1994. Biological phosphorus removal study at the Suomenoja research station. *Vatten* 50: 321–328.
- Redfield A.C. 1958. The biological control of chemical factors in the environment. *American Scientist* 46: 205–221.
- Reichert P. 1995. Design techniques of a computer program for the identification of processes and the simulation of water quality in aquatic systems. *Environmental Software* 10: 199–210.
- Reichert P., von Schulthess R. & Wild D. 1995. The use of AQUASIM for estimating parameters of activated sludge models. *Water Science and Technology* 31: 135–147.
- Savchuk O. 2000. Studies of the assimilation capacity and effects of nutrient load reductions in the eastern Gulf of Finland with a biogeochemical model. *Boreal Environment Research* 5: 147–163.
- Savchuk O. & Wulff F. 1999. Modelling regional and large-scale response of Baltic Sea ecosystem to nutrient load reductions. *Hydrobiologia* 393: 35–43.
- Sellner K.G., Olson M.M. & Olli K. 1996. Copepod interactions with toxic and non-toxic cyanobacteria from the Gulf of Finland. *Phycologia* 35 (suppl.): 177–182.
- Sivonen K., Kononen K., Esala A.-L. & Niemelä S.I. 1989. Toxicity and isolation of the cyanobacterium *Nodularia spumigena* from the southern Baltic Sea. *Hydrobiologia* 185: 3–8.
- Tyrrell T. 1999. The relative influence of nitrogen and phosphorus on the oceanic primary production. *Nature* 400: 525–531.
- Virtanen M. & Koponen J. 1985. Simulations of transport under irregular flow conditions. *Aqua Fennica* 15: 65–75.
- Virtanen M., Koponen J., Dahlbro K. & Sarkkula J. 1986. Three-dimensional water-quality-transport model compared with field observations. *Ecological Modelling* 31: 185–199.