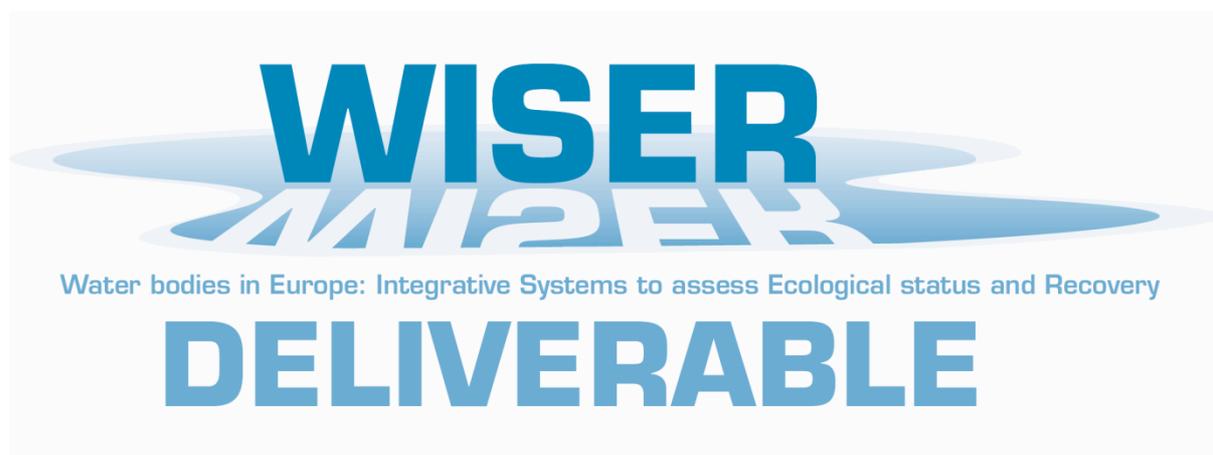


Collaborative Project (large-scale integrating project)
Grant Agreement 226273
Theme 6: Environment (including Climate Change)
Duration: March 1st, 2009 – February 29th, 2012



Deliverable 5.3-3: Comparison of mechanistic and statistical approaches for catchment management in coastal waters

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Due date of deliverable: **Month 32**

Actual submission date: **Month 33**

Project co-funded by the European Commission within the Seventh Framework Programme (2007-2013)

Dissemination Level

| | | |
|----|---|---|
| PU | Public | X |
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Summary

In this report, the aim was to compare mechanistic and statistical models including Bayesian techniques to identify linkage between pressure variables and the responses of biological quality elements in the non-tidal, oligohaline River Vantaa estuary in the Baltic Sea and in the meso-macro-tidal Nervión estuary in Basque coast.

In River Vantaa estuary, trophic conditions are elevated due to huge nutrient loading. Based on the model simulations, reducing TN concentrations in river water would decrease concentrations in the estuary, whereas reduction of phosphorus inputs from the river alone would not improve the trophic status, meaning that the reduction of the level of TP in the Gulf of Finland is required, as well.

However, exchange of water with the open sea did not alone explain elevated trophic level in the estuary. The huge loading in the 1970s and 1980s raised nutrient reserves in the bottom sediment and accelerated benthic release of nutrients, i.e. internal loading, proved by an atypical seasonal distribution of phosphorus and high amounts of chlorophyll *a* in summer.

Based on regression and LLR models, nitrogen was the best predictor for chlorophyll *a*. Logistic regression analyses showed that the probability for cyanobacterial blooms in the inner estuary increased along with the increase in TP and the decrease in TN, the results of which accorded with the model simulations on the transport of nutrients. The negative correlation of TN with phytoplankton was explained by light limitation caused by suspended solids.

Based on the LLR tool, the target load of TP with 50% confidence was estimated as $1.6 \text{ g m}^{-2} \text{ a}^{-1}$, corresponding to $12 \text{ } \mu\text{g L}^{-1}$ in summer conditions. The target load of TN was estimated as $33 \text{ g m}^{-2} \text{ a}^{-1}$, corresponding to $300 \text{ } \mu\text{g L}^{-1}$. A danger exists that the estuary will fail to reach good ecological status by 2015.

In Nervión estuary, ammonium concentrations have decreased and transparency increased after the start of sewage treatment in 1991. However, no statistical change in the chlorophyll *a* could be observed. The decreasing nutrients concentration should reduce the phytoplankton development, whereas an increasing deeper euphotic layer depth should increase it. The impact on the chlorophyll *a* biomass evolution was not visible with simple statistical relations.

Based on the preliminary results, model simulations will allow assessing the role of the different processes in the time evolution of the chlorophyll *a*. The model reproduces the Nervión plume and the extension of low salinity values through the estuary. The nitrate extension from the Nervión input appeared to be larger than the ammonium plume extension. Nitrate from Nervión River extends in the whole harbour until the open sea. The phytoplankton development could be simulated in the Nervión plume.

1. Introduction

Catchment management introduced by EU Water Framework Directive (Directive, 2000/60/EC, WFD) will have major impacts on the conservation and restoration of aquatic ecosystems throughout Europe. All European countries should implement management measures specified in River Basin Management Plans (RBMP) by 2012 to achieve good ecological status and prevent further deterioration of water bodies by 2015. The objective is challenging despite the directive allows the possibility for a delay until 2030.

Required pressure reductions assessed in RBMPs can, in principle, be defined from dose-response curves revealing causal relationships between pressures and impacts of the ecological quality elements of the WFD. However, non-linearity of many biological or geochemical processes may complicate nutrient pathways. For example, the response of nutrient concentrations in surface waters may be delayed after considerable reductions in agricultural nitrogen input as a consequence of the long residence time of groundwater reservoirs and from the large amount of organic nitrogen that has accumulated in soils during periods of higher application rates (Grimvall *et al.* 2000, Andersson and Arheimer 2003). Additionally, the relationships between pressures and impacts may show non-linearity. Removal of the stress may lead an ecosystem to recover but not necessarily along the same trajectory of decline, in which case the system does not return to its original level (Tett *et al.* 2007), the difference between the two trajectories being termed as hysteresis. Marine ecosystems regime shifts, i.e. shifting thresholds, have been usually explained by multiple stressors such as climate change, overexploitation of resources, nutrient enrichment or all of them in various combinations (e.g. Weijerman *et al.* 2005, Daskalov *et al.* 2007, Österblom *et al.* 2007, Möllmann *et al.* 2009). However, the recovery may also be unattainable.

Coastal areas and estuaries are considered as particularly vulnerable to the impacts of climate change because they are subject both to changes in marine and terrestrial environments, the fact which causes further uncertainties in assessing ecological status and the effects of catchment management measures. The impacts of global climate change on aquatic ecosystems include rise of water temperature, changes in rainfall patterns with increasing droughts and floods and sea level rise, which in combination of nutrient enrichment encompass changes like increase probabilities of harmful algal blooms, accelerate oxygen consumption in near bottom waters, cause changes in community structure of fauna et flora and distribution of individual species etc. Through shifting thresholds and changes in residence conditions, possible time-lags and hysteresis recovery processes will become complicated. Eventually, the initial goal of achieving good ecological status may be impossible to meet.

The dynamic of marine environment is by definition complicated, and the multiple stressors with various interactive impacts make catchment management even more challenging. Dynamic simulation modeling is useful in looking at responses of eutrophication to changes in loading

and natural forces in marine environment, whereas empirical approaches have less frequently been applied in coastal waters. Both approaches are applicable in the implementation of the WFD to get answers for questions like what is the magnitude of the required pressure reduction and efficiency of the present pressure reduction measures taking into account varying and changing environmental conditions, how great is the uncertainty of ecological effects to pressure reduction and how climate change together with other stressors will affect the ecological thresholds.

In this study, mechanistic and statistical models including Bayesian techniques will be applied to identify linkage between pressure variables and the responses of biological quality elements with a special attention to recovery processes. We will estimate required nutrient reductions to achieve good ecological status in meso-macro-tidal Nerviön estuary in Basque coast and in non-tidal Vantaa estuary in the Baltic Sea. The mechanistic and statistical models linking catchment management to ecological status will be compared to each other with particular emphasis on uncertainty of pressure reductions to derive guidelines for management of ecosystem restoration. Biological recovery processes in these estuaries will be analyzed under different climatic conditions.

2. Finnish coastal waters

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2.1 Study area

River Vantaa estuary is a non-tidal and semi-enclosed small estuary east of the City of Helsinki (Fig. 2.1). It receives fresh water from River Vantaa, the mean flow of which is $16.7 \text{ m}^3 \text{ s}^{-1}$ and watershed area $1\,686 \text{ km}^2$. It is connected to the open Gulf of Finland first by two narrow straits and then by three sounds, which restrict horizontal water exchange. Both of the inner and outer estuaries are shallow (mean depths 4 and 18.5 m, resp.) and small in size (5.5 and 15 km^2). The estuary is oligohaline, the average salinity in summer varying between 0 and 5 psu. The water is turbid with Secchi depth ranging from 0.1 to 1 m. There is no stratification in the inner estuary but the outer estuary is seasonally stratified by thermocline lying in the depth of 10 m in summer. Oxygen conditions in near-bottom waters are good. The seafloor consists mainly of soft sediments. According to the Finnish typology, the estuary belongs to the inner archipelago of the Finnish coastal waters (Kangas *et al.*, 2003, Vuori *et al.*, 2009).

Vantaa estuary was heavily polluted by municipal waste waters of the city Helsinki in the 1960s and early 1970s, when the inner and outer estuaries received 865 and 210 t of nitrogen and 177 and 68 t of phosphorus annually, in respectively. However, the load of total phosphorus (TP)

reduced remarkably in the late 1970s after the introduction of chemical removal of phosphorus. After the 1980s, TP load was insignificant in the outer estuary. The load of total nitrogen (TN) in the inner estuary increased during the operational time of the plants until 1986 whereas in the outer estuary TN loads started slightly to decrease already in the late 1970s. All of the local municipal treatment plants (LMTP) in the inner and outer Vantaa estuaries were closed by 1987. Nutrient fluxes through River Vantaa have varied annually during 40 years so that the minimum fluxes of TN and TP in 2003 were five to eight times smaller than the maximum ones of 1974. As a whole, the fluxes were greater during the 1970s and 1980s than during the 1990s and 2000s, which is partly as a consequence of hydrological variation and partly as a consequence of reduction of point source and diffuse loading in the catchment.

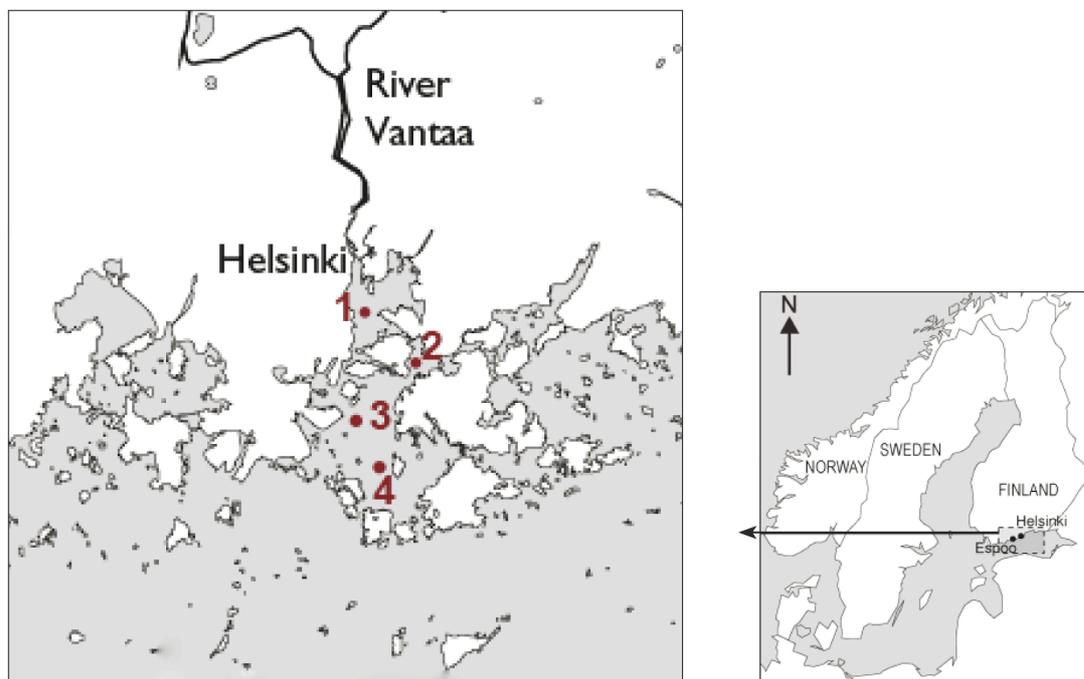


Fig. 2.1a River Vantaa estuary and the study sites off Helsinki in the Gulf of Finland, the Baltic Sea. The sites are (1) Vanhankaupunginlahti, (2) Tullisaarenselkä, (3) Katajanokka and (4) Vasikkasaari.

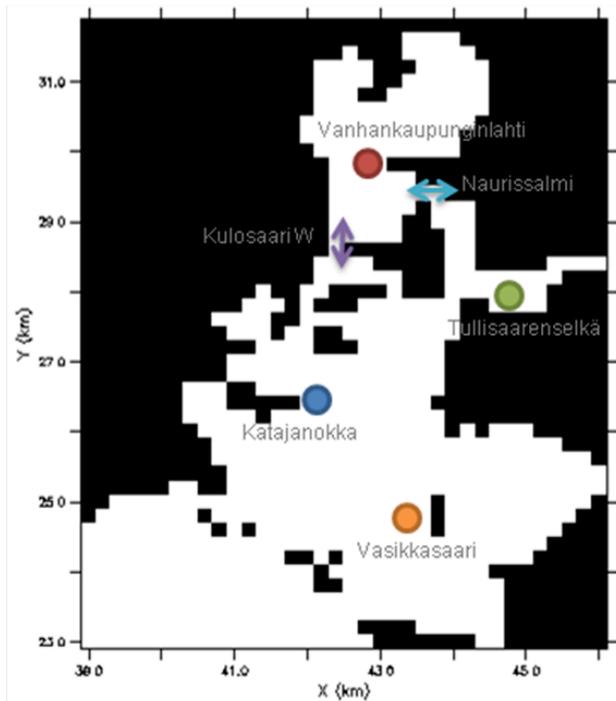


Fig.2.1b. Observation sites and two sounds between the inner and outer estuary of River Vantaa.

2.2 Material and methods

2.2.1 Dataset

The dataset was compiled for River Vantaa, River Vantaa estuary and its offshore waters in the eastern side of Helsinki City, the Baltic Sea (Fig 2.1). The data included the inputs of nitrogen and phosphorus from local municipal treatment plants, mean water flows (MQ), the fluxes of total phosphorus (TP) and total nitrogen (TN) of River Vantaa, and concentrations of TP and TN, salinity and temperature in surface and near-bottom water layers as well as concentrations of chlorophyll *a* and total phytoplankton biomass (as wet weight) in the productive surface water layer (0 to 4 m). Water quality samples were taken from four coastal monitoring stations of Helsinki City 3 to 14 times per mid-and late summer season (July to September) in 1970-2010. The data on coastal water quality, phytoplankton biomasses and nutrient inputs from point sources originated from the monitoring of Helsinki City, whereas the data on the water quality and water flows of River Vantaa originated from the national monitoring of the Finnish Environment Institute (SYKE). All the data excluding phytoplankton biomass are maintained in the register of SYKE.

Annual loads of TP and TN of River Vantaa were calculated by multiplying the mean monthly concentration by the mean monthly water flow and summing up the monthly loads (Kauppila and Koskiahho, 2003). Water samples were taken 10 to 12 times per year and water flow was measured daily. The flushing rate of River Vantaa as well as the water flows and nutrient fluxes in the surface and bottom water layers of the Vantaa estuary were calculated using Knudsen's

equation (e.g. Bowden, 1980). Total nutrient inputs into the inner estuary were estimated by summing up the point-source loads from the coast, nutrient fluxes of River Vantaa and near-bottom nutrient fluxes from the outer estuary.

2.2.2 Biological and chemical analyses

Chlorophyll *a* taken from the composite sample of phytoplankton (0-4 m) was analysed according to Lorenzen (1967). The chlorophyll samples were extracted with acetone from 1980 to 1994 and ethanol (Ethyl alcohol) thereafter. However, the results of the samples extracted with acetone have been corrected to correspond to those extracted with ethanol. The treatment of the phytoplankton samples of Helsinki City have varied in process of time. Here, composite samples of phytoplankton taken with a Ruttner sampler were preserved with Lugol's solution (Edler, 1979). Cells were counted with a Leitz Fluovert FS microscope employing Utermöhl's technique (1985). Cell numbers were converted to biomass (as carbon content) using the measurements of the Helsinki City (Huttunen and Kononen, unpubl. data) or the volumes reported by Edler *et al.* (1979) or Naulapää (1965). In the register of Helsinki City, carbon biomasses have also been converted to correspond to wet weight biomasses.

Phosphorus and nitrogen were determined from unfiltered samples spectrometrically and colorimetrically following the Finnish standard methods (Koroleff, 1983). Total nitrogen samples were first oxidized with potassium peroxodisulphate, then reduced to nitrate-nitrogen, using CD amalgam. Total phosphorus analyses were initiated by digestion with potassium peroxodisulphate, and determined by using the ammonium molybdate method (Murphy and Riley, 1962). Salinity has been measured electronically using a Salinometer or Conductivity meter.

2.2.3 Statistical models

2.2.3.1 Regression techniques

Stepwise regression analyses were used to explain the variation of chlorophyll *a* as a function of total nitrogen, total phosphorus, salinity and temperature in the inner and outer estuaries of River Vantaa. Additionally, single regression analyses were carried out to test further the effects of weather conditions on nutrient-chlorophyll relationships dividing the data to subsets which represented (i) wet and warm, (ii) wet and cold, (iii) dry and warm, and (iv) dry and cold summers. The summertime MQ of $10 \text{ m}^3 \text{ s}^{-1}$ or alternatively the surface salinity of 2.9 psu (median value) were set as boundaries between wet and dry summers. The surface water temperature of $18 \text{ }^\circ\text{C}$ was set as the boundary between warm and cold summers.

The relationship between nutrient concentrations and the probability of phytoplankton and cyanobacterial blooms were estimated using logistic regression, which is based on a generalized linear model for binomially distributed dependent variables that utilizes the logit link function (Venables and Ripley 1994). Observed cyanobacterial phytoplankton biomass was transformed into binomially distributed variables according to the threshold values of blooms. The boundary

of cyanobacterial blooms in the inner estuary was defined using a regression function between cyanobacterial biomass and chlorophyll *a* and the good (G)/moderate (M) boundary value of chlorophyll *a* ($4.7 \mu\text{g l}^{-1}$ according to Vuori *et al.*, 2009).

2.2.3.2 The LLR internet tool

LakeLoadResponse(LLR) internet tool, developed by Malve *et al.* (manuscript 2011), is described in WISER Deliverable D5.2-1: **Analysis of applied modeling approaches in the case studies**. Here the tool was applied to oligosaline Vantaa estuary to predict the responses of summertime (July to September) concentrations of TN, TP and chlorophyll *a* as well as phytoplankton total biomasses to nutrient inputs. The purpose was to estimate target nutrient loads (TNL) for the given quality standards of good ecological status in River Vantaa estuary, the ecological class boundaries in the inner archipelago of the Gulf of Finland obtained from the report by Vuori *et al.* (2009). Nutrient and salinity balances were calculated to be used in the end in the predictions of target nutrient loads of River Vantaa estuary. Here the local data were the only sources for calculations, but in future the models will be corrected with estuarial-specific data on water and nutrient balances.

The Lake Load Response (LLR) internet tool has been developed to predict the effects of nutrient loading into a lake. The internet tool is based on the LakeState (LS) model, that consists of three component models: Chapra's (1975) model for retention of total phosphorus and nitrogen, the hierarchical, linear regression model for chlorophyll *a* (Malve 2007) and the logistic regression model for phytoplankton biomass (Kauppila P., Lepistö L., Malve O. & Raateland A. 2011, Submitted to BER). In the LS model the mechanistic and statistic approach were combined using Bayesian inference with Markov Chain Monte Carlo (MCMC) simulation methods. This way, predictions about the water quality as well as about the model error can be made on a statistical basis, which gives more confidence into lake management planning.

Using Chapra's retention model (Equation 1) we estimated here the in- estuarial nutrient concentrations as a function of incoming loading and water outflow. The estimates were based on the assumption of equilibrium ($\frac{dC}{dt} = 0$), according to which the in-lake/estuarial nutrient concentration does not change during the retention period. Especially the internal loading of the water area can affect this equilibrium and weaken the reliability of the estimates.

$$V \frac{dC}{dt} = W - QC - v_s AC = 0 \quad (1)$$



V the volume (m^3) of an estuary

X change in nutrient concentration in time unit

- W external loading (mg d⁻¹)
 Q outflow (m³ d⁻¹)
 C in-estuarial nutrient concentration (µg l⁻¹, mg m⁻³)
 v_s Chapra's sedimentation rate (m d⁻¹)
 A estuarial surface water area (m²)

Because of the use of Bayesian inference, Chapra's retention model was actually used to calculate the expectation of the nutrient concentration. The estimated nutrient concentration (Equation 2) and Chapra's sedimentation rate (Equation 3) were presumed to be normally distributed (Equation 2). The sedimentation rate was first determined using observations of the other variables, after which the model was used to calculate concentration estimates with different loadings.

$$C \sim N(\mu, \tau^2) \quad (2)$$

- μ expected value for in-estuarial nutrient concentration (µg l⁻¹, mg m⁻³)
 τ² error variance for the model

$$v_s \sim N(\mu_s, \sigma_s^2) \quad (3)$$

- μ_s expected value for sedimentation rate (m d⁻¹)
 σ_s² variance for sedimentation rate

Chapra's sedimentation rate was calculated (Equations 8 and 9) using the trapping parameter and sedimentation rate values obtained from the models.

$$R_p = 0.60 - 0.0006W \quad (R^2 = 0.47) \quad (4)$$

$$R_N = 0.35 - 0.00016T_w \quad (R^2 = 0.48) \quad (5)$$

$$\sigma = 0.129(C_{in})^{0.549} T_w^{-0.549} \quad (6)$$

$$\sigma = \beta C_{in}^{\theta 1} T_W^{\theta 2} z^{\theta 3} \quad (7)$$



(8)



(9)

| | |
|------------|--|
| R_p | phosphorus retention coefficient |
| W | external loading (mg d^{-1}) |
| R_N | nitrogen retention coefficient |
| T_W | hydraulic detention time (a) |
| σ | sedimentation coefficient (a^{-1}) |
| C_{in} | nutrient concentration in inflowing water (mg m^{-3}) |
| β | for phosphorus = 3.0 (SD \pm 0.25), for nitrogen = 0.67 (SD \pm 0.10) |
| $\theta 1$ | for phosphorus = 0.53 (SD \pm 0.13), for nitrogen = 0 |
| $\theta 2$ | for phosphorus = -0.75 (SD \pm 0.06), for nitrogen = -0.75 (SD \pm 0.11) |
| $\theta 3$ | for phosphorus = 0.58 (SD \pm 0.19), for nitrogen = 0 |
| z | the mean depth (m) of the estuary |
| v_s | Chapra's sedimentation rate (m d^{-1}) |
| q_s | hydraulic surface loading ($\text{m}^3 \text{a}^{-1}$) |

The estimate of the estuarial specific model is based on observations from the River Vantaa estuary. The sedimentation rates were fitted to the data obtained from the data of River Vantaa estuary. The in-estuarial phosphorus and nitrogen concentrations were used to predict the in-estuarial chlorophyll *a* concentration with the hierarchical, linear regression model for chlorophyll *a* (Malve and Qian, 2006, Lamon *et al.*, 2008). The model may be summarized as follows (Malve 2007):

$$\log(y_{ijk}) \sim N(X\beta_{ij}, \tau^2) \quad (10)$$

$$X\beta_{ij} = \beta_{0,ij} + \beta_{1,ij} \log(\text{TP}_{ijk}) + \beta_{2,ij} \log(\text{TN}_{ijk}) + \beta_{3,ij} \log(\text{TP}_{ijk}) \log(\text{TN}_{ijk})$$

$$\beta_{ij} \sim N(\beta_i, \sigma^2_i)$$

$$\beta_i \sim N(\beta, \sigma^2)$$

| | |
|-----------------|---|
| $\log(y_{ijk})$ | k th observed $\log(\text{Chla})$ value from lake j of type i |
| X | matrix containing the observed total phosphorus (TP) and total nitrogen (TN) values from the estuary j of type i |
| β_{ij} | estuarial specific model parameter vector $[\beta_{0,ij}, \beta_{1,ij}, \beta_{2,ij}, \beta_{3,ij}]$ which consists of the intercept ($\beta_{0,ij}$) and slopes for $\log(\text{TP})$ ($\beta_{1,ij}$), $\log(\text{TN})$ ($\beta_{2,ij}$) and for the combined effect of $\log(\text{TP})$ and $\log(\text{TN})$ ($\beta_{3,ij}$) |
| τ^2 | model error variance |
| β_i | vector $[\beta_{0,i}, \beta_{1,i}, \beta_{2,i}, \beta_{3,i}]$ of the model parameter means for coastal water type i |
| σ_i^2 | vector $[\sigma_{0,i}^2, \sigma_{1,i}^2, \sigma_{2,i}^2, \sigma_{3,i}^2]$ of variances in model parameters between estuaries of type i |
| β | means for coastal water types $[\beta_0, \beta_1, \beta_2, \beta_3]$ |
| σ^2 | variance for coastal water types $[\sigma_0^2, \sigma_1^2, \sigma_2^2, \sigma_3^2]$ |

From the relation between nutrient and chlorophyll a concentration it was then possible to determine the relation between loading and chlorophyll a concentration. This resulted in predictions about the target load with which a good ecological status according to chlorophyll a concentration is achieved.

The logistic regression model for phytoplankton biomass (Equation 11) gives predictions about the probability of phytoplankton biomass to exceed the boundary of good water quality with different phosphorus and nitrogen loads.

$$\text{Logit}(\text{Pr}) = \beta_n + \beta_1 \log(C_P) + \beta_2 \log(C_N) \quad (11)$$

$$\beta_i \sim N(\mu_R, \sigma_R^2)$$

| | |
|--------------|---|
| Pr | probability of bloom |
| β_n | regression intercept |
| β_i | regression coefficient |
| C_i | observed total phosphorus/ nitrogen concentration |
| μ_R | expectation for regression coefficients |
| σ_R^2 | variance for regression coefficients |

2.2.4 Mechanistic models

The hydrodynamic model applied to River Vantaa estuary and its offshore waters is based on the Coherens model code (Luyten *et al.*, 1999) which was initially designed as a regional model for

the North Sea. The original model was designed to predict the effect of changing conditions on the biota and to simulate the input and dispersion of contaminants in coastal and shelf seas. The physical parameterizations of the Coherens model are similar to those used in the Princeton Ocean Model (Blumberg and Mellor, 1987). It uses the hydrostatic assumption and utilizes sigma-coordinates in the vertical direction, and mode splitting is used to resolve the barotropic and baroclinic modes with different time steps. The model code is highly modularized and configurable. Many options exist to change the numerical scheme, turbulence parameterization, surface and bottom flux parameterizations or the treatment of boundary conditions.

A high resolution 1-dimensional version of the model has been successfully used to study the ventilation of flows in the transition zone from the northern Kattegat to the Arkona Sea (Bendtsen *et al.*, 2006), giving estimates of vertical mixing. A 3-dimensional model version was applied by Bendtsen *et al.* (2008) to study the ventilation of bottom water between the North Sea and the Baltic Sea. It showed that the ventilation in the transition zone has a large impact on the oxygen conditions of the Western Baltic Sea. The Coherens model is also a part of the EMAPS model validation work for the Baltic Sea in which six different models were setup in the same way (Myrberg *et al.*, 2010). The results of that study showed that the Coherens model performs well but like the other models, it has problems in resolving the halocline in the Gulf of Finland. The Coherens model was applied by Podsechin *et al.* (2005) to Lake Längelmävesi-Roine in Finland to quantify the role of different factors affecting the timing and intensity of algal blooms. Important factors include among others re-suspension of bottom sediments caused by episodic wind events. Additionally, Malve *et al.* (2011) describe Lake Pyhäjärvi application in the context of WISER Deliverable D5.2-1: **Analysis of applied modeling approaches in the case studies.**

2.2.4.1 Water quality sub-model

The water-quality model is based on total nutrient values and three algal compartments comprising cyanobacteria, diatoms and other algae. The total-nutrient-type approach has previously been used by Malve *et al.* (2007) to predict algal biomasses in lakes. The amount of nutrients, bound in algal cells, is subtracted from the amount of total nutrients to produce a pool of nutrients which are available to algal compartment increase. The main state variables of this model are total nitrogen (c_n) and total phosphorous (c_p) in the water, and the biomass of cyanobacteria (c_c), diatoms (c_d) and other algae (c_a). The concentrations of nitrogen (c_{ndet}) and phosphorous (c_{pdet}) in detritus can also be calculated if required.

The different compartments of the biological model are advected both horizontally and vertically, and diffused by the physical model, and the nutrient loadings are introduced via boundary conditions so only the source term in the total time derivative needs to be described. The change in time of a variable (ϕ) is thus:

$$\frac{\partial \phi}{\partial t} = S \tag{1}$$

In the model application of this work $S=0$ and variable (ϕ) is not changing. See table 1 for a list of parameters used in the water quality sub-model.

Table 2.1. Parameters used in the water quality submodel of Coherens.

| Parameter | Description | Value | Unit |
|-----------|---|---------|-------------------|
| IRED | Radiation attenuation by ice | 0.5 | - |
| NSR | Nitrogen sinking rate | - 0.005 | m d ⁻¹ |
| PSR | Phosphorus sinking rate | -2.0 | m d ⁻¹ |
| G0 | Maximal detritus phosphorus rate | 0.043 | 1 d ⁻¹ |
| B0 | Maximal detritus nitrogen mineralisation rate | 0.018 | 1 d ⁻¹ |
| SNDET | Sedimentation rate of detritus nitrogen | 0.16 | m d ⁻¹ |
| VNDET | Settling rate of detritus nitrogen | 1.0 | m d ⁻¹ |
| SPDET | Sedimentation rate of detritus phosphorus | 0.16 | m d ⁻¹ |
| VPDET | Settling rate of detritus phosphorus | 1.0 | m d ⁻¹ |
| HMIX | Depth of mixing layer | 1.0000 | - |

2.2.4.2 Coherens model application for the Helsinki area

A 200 m x 200 m grid was created for the Helsinki area and the bathymetry of the grid (Fig. 2.2) shows that the maximum depth is 28 m. The minimum depth is set to 0.5 m.

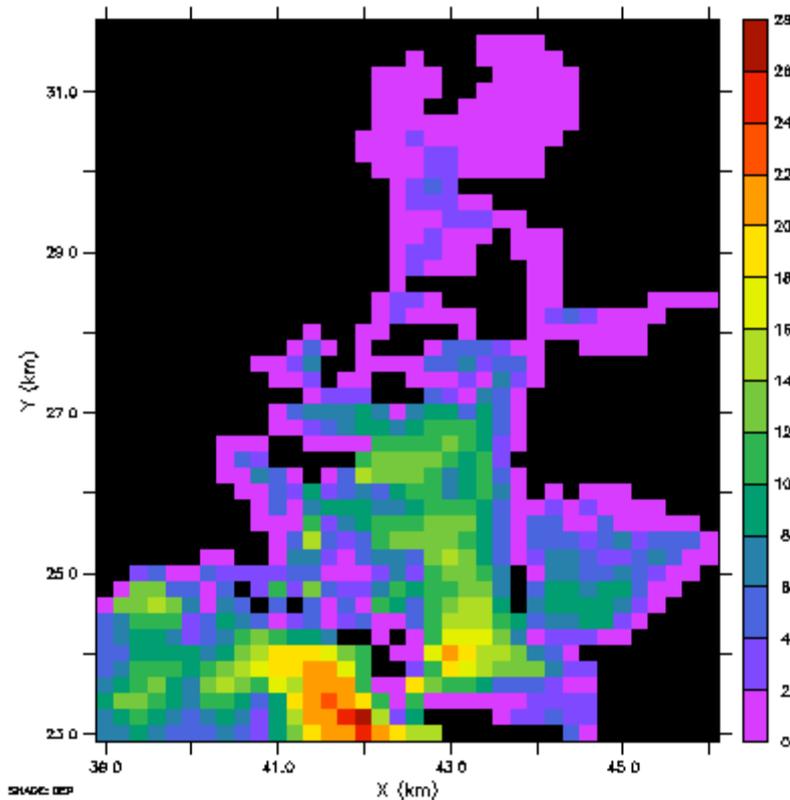


Fig. 2.2 The bathymetry of the 200m x200m resolution grid for the Helsinki region. River Vantaa comes in from the North.

Simulations with mechanistic Coherens model were run for two different summers, 1996 and 1999, of which the former represented a rather wet and the latter a very dry summer. Simulations were run from the 1st of April to the 1st of October in both years, and dynamic weather data were used in the calculations. Wind velocity, air temperature, relative humidity, cloudiness, air pressure and precipitation recorded in every three hours in Helsinki were obtained from a dataset of SMHI (Sveriges Meteorologiska och Hydrologiska Institutet), also utilized for the comparison study of the EMAPS model (Myrberg *et al.*, 2010). The wind speed in 1999 was limited to 11 m s⁻¹ at maximum because higher velocities caused instabilities in the open boundaries of Coherens model. The data of water flows and nutrient concentrations of River Vantaa were derived from the register of SYKE, as well as salinity and total nutrient concentrations of the sea water. The incoming Riemann characteristic (Luyten *et al.*, 1999) depends on zero inflow in the open boundaries. The initial values for temperature and total nutrients were obtained from the monitoring observations by interpolating to the first day of calculation periods. Different initial values were set for inner estuary of River Vantaa and the rest of the model area. Salinity was set to be initially zero in the whole model area.

Total phosphorus and total nitrogen models were calibrated by sinking rates using the observations of year 1996. Calibration resulted to sinking rate of -2.0 m d⁻¹ for total phosphorus and -0.005 m d⁻¹ for total nitrogen (see Table 2.1). Summer 1999 was a validation period.

2.3 Results

2.3.1 Changes in water quality in River Vantaa estuary since 1970s

As a result of the chemical removal of P in the late 1970s, summertime concentrations of TP in the inner Vantaa estuary dropped steeply from 700 to 200 µg l⁻¹. After the closing of the LMTP in 1992, TP has remained below 100 µg l⁻¹, varying depending on hydrological conditions. Exceptionally rainy summers such as in 2004 raised TP concentrations up to ca 300 µg l⁻¹. As a result of chemical removal of nitrogen in 1986, TN concentrations decreased first from 3500 µg l⁻¹ to the level below 2000 µg l⁻¹ but since 1992 they have varied from 500 to 2000 µg l⁻¹. The measures of water pollution control were reflected to amounts of inorganic nutrients, too. Thus, the proportion of inorganic N was highest (40% of TN in average) in the late 1980s, whereas in the late 1970s and 2000s the share was 20% on an average. The proportion of DIN varied between 5 to 90% of total nitrogen. Instead, the proportion of inorganic P was highest (60% on an average) in the 1970s. The proportion of DIP is also highly variable. Based on the DIN/TP ratio (see Ptacnik *et al.* 2010), the estuary has been mainly N limited since the late 1970s excluding the first half of the 1980s when the estuary was mainly P limited. The change back to N limitation in the mid' 1980s dated to the moment of starting removing N from municipal

waste waters. The change in nutrient limitation can also be seen in the trend of TN/TP ratio, which increased from the 1970s up in the turn of the 1990s, declining thereafter.

Chlorophyll *a* concentrations followed the trends of TN and TP: the highest concentrations (300 $\mu\text{g l}^{-1}$) were measured in the early 1980s after which concentrations have usually varied between 10 and 50 $\mu\text{g l}^{-1}$. Secchi depth has improved from its average of 0.3 m in the 1970s to 0.5 m. Based on chlorophyll *a* and Secchi depth, River Vantaa estuary is, however, in moderate status. Oxygen conditions in near-bottom waters have been good due to small residence time in the estuary as a consequence of discharge of River Vantaa.

During the 1970s and 1980s, chlorophyll *a* and phosphorus showed atypical seasonal distribution with summer maxima (Fig. x). After closing the LMTPs, the distribution moved more typical revealing spring maxima and summer minima (Fig. 2.3).

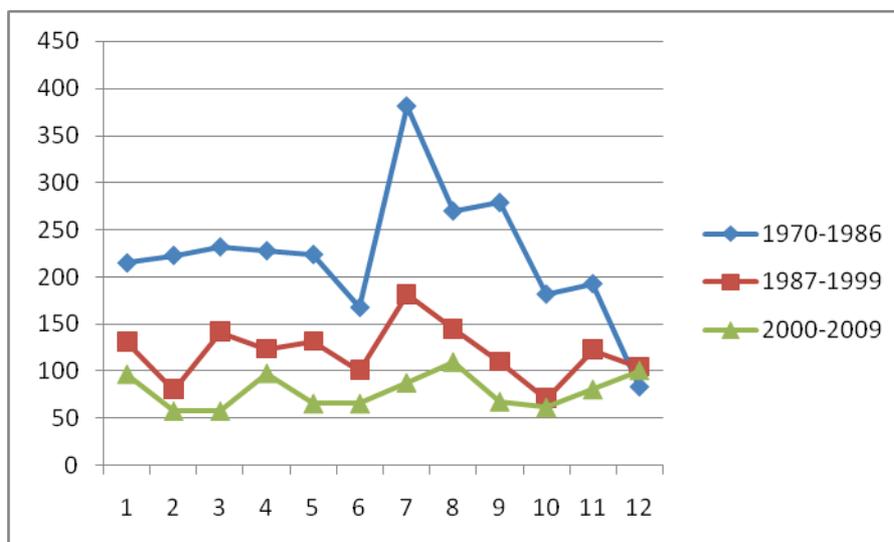
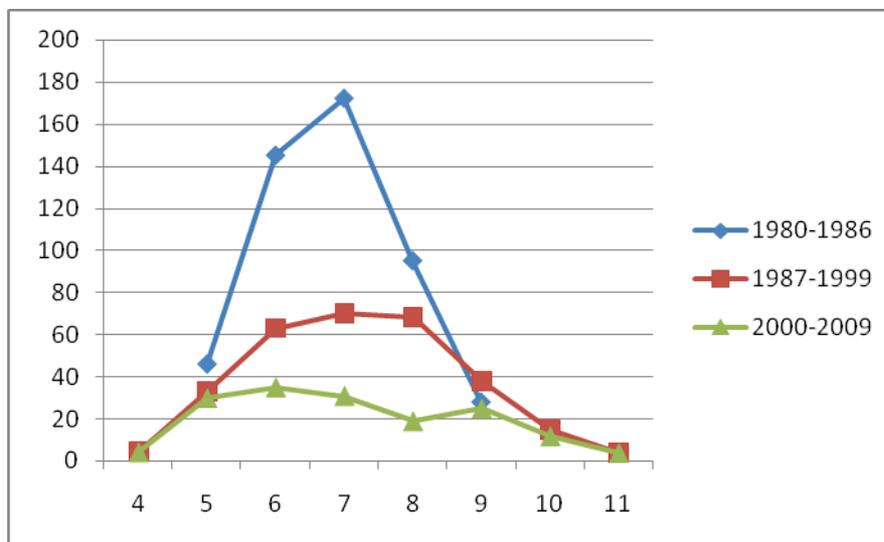


Fig. 2.3. Seasonal trends of average concentrations of (a) chlorophyll *a* ($\mu\text{g l}^{-1}$) and (b) total phosphorus ($\mu\text{g l}^{-1}$) in the inner estuary of River Vantaa during 1970 - 1986, 1987 - 1999 and 2000 - 2009.

2.3.2 Results on statistical approaches

2.3.2.1 Regression analyses

Based on stepwise regression analyses, concentrations of chlorophyll *a* in River Vantaa estuary could be predicted as a function of total nutrients, salinity and water temperature. In the inner estuary, total phosphorus or alternatively total nitrogen together with salinity and temperature explained 62 and 66% of the variation in chlorophyll *a*, in respectively (Table 2.2). In these statistically highly significant models, the effect of salinity was strongest (partial R-squares 0.28). However, the accuracy of the estimated averages varied, the root mean square error (RMSE) being 0.63 and 0.67, in respectively. The connection to nutrients was weaker (partial R-squares ca 0.20).

In the outer estuary, predictions were weaker, total nitrogen together with salinity and temperature explaining only 29% of the variation in chlorophyll *a*. However, accuracy in this highly significant model was better ($\text{rMSE} = 0.38$) than that in the inner estuary. Contrary to the model in the inner estuary, chlorophyll *a* linked stronger to nitrogen (partial R-square 0.20) than to salinity or temperature.

To further evaluate the effect of weather conditions, we divided the data to four subsets using salinity of 2.9 psu (median value) as the boundary between dry and wet summers, and water temperature of 18 °C as the boundaries between cold and warm summers. Predictions linking chlorophyll *a* to nutrients improved, but regression equations could only be produced for warm and dry, and cold and dry summers. In the best model, produced for warm and dry summers, total phosphorus and total nitrogen explained 85 and 92% of the variation in chlorophyll *a*, in respectively (Table 2.2). In the model built for cold and dry summers, the coefficients of determination were weaker, 0.63 and 0.53, in respectively. Furthermore, accuracy of the predictions improved in the case of warm and dry summer: RMSE was 0.30 when using total nitrogen as the predictor and RMSE was 0.40 when using total phosphorus as the predictor. In the case of cold and dry summers, accuracy did not improve notably compared to multiple regression models mentioned above.

Table 2.2. Models of phytoplankton chlorophyll *a* ($\mu\text{g l}^{-1}$) in the inner estuary of River Vantaa. Symbols: *n*, sample number; *rMSE*, root mean square error, R^2 , coefficient of determination; *p*, significance of the slope.

| | | n | rMSE | R^2 | <i>p</i> |
|----------|---|-----|-------|-------|----------|
| All data | $\text{LChl} = -11.5 + 1.1 \text{ LTN} + 0.7$ | 137 | 0.628 | 0.66 | < 0.0001 |

| | | | | | |
|--------------|--|-----|-------|------|----------|
| | LSal + 2.3 Ltemp | | | | |
| All data | LChl = - 6.93 + 1.15 LTP + 0.66 LSaI + 1.71 Ltemp | 137 | 0.666 | 0.62 | < 0.0001 |
| Warm and dry | LChl = -2.18 LTP | 26 | 0.400 | 0.85 | < 0.0001 |
| Warm and dry | LCh = -5.43 + 1.34 LTN | 26 | 0.295 | 0.92 | < 0.0001 |
| Cold and dry | LChl = -3.49 + 1.01 LTN | 38 | 0.667 | 0.53 | < 0.0001 |
| Cold and dry | LChl 0 -2.51 + 1.42 LTP | 38 | 0.588 | 0.63 | < 0.0001 |

2.3.2.2 The LLR model

The LLR tool was first applied using the whole data of the inner estuary of River Vantaa from the 1970s but the model overestimated the reductions of loading and concentrations of chlorophyll *a*. To get realistic predictions we used only the data gathered subsequent the closure of the LMTPs. Both the fit of the models and accuracy of the predictions were better for TN than TP (Fig. 2.4). The curve was fitted to estimate median concentrations of TP with 50% confidence. Good ecological status could then be achieved with the limit value of 12 $\mu\text{g l}^{-1}$ of TP and with the target P load of 1.6 $\text{g m}^2 \text{a}^{-1}$. Consequently, the average areal P load of 3.9 $\text{g m}^2 \text{a}^{-1}$ should be reduced by 60% to get good ecological status.

Regarding N, the target load was estimated as 33 $\text{g m}^2 \text{a}^{-1}$ with 50% confidence, corresponding to about half of the average areal load of N (72 $\text{g m}^2 \text{a}^{-1}$). However, if coastal managers would like to be more certain of the effects of the reduction, they could choose tighter confidence levels. For instance, using 80% confidence, the target N load should be set as 18 $\text{g m}^2 \text{a}^{-1}$. Considering the monitoring data, requirements for good ecological status have fulfilled so far only a couple of times when estimating chlorophyll *a* as a function of areal loads of both TN and TP with 50% confidence (Fig. 2.5).

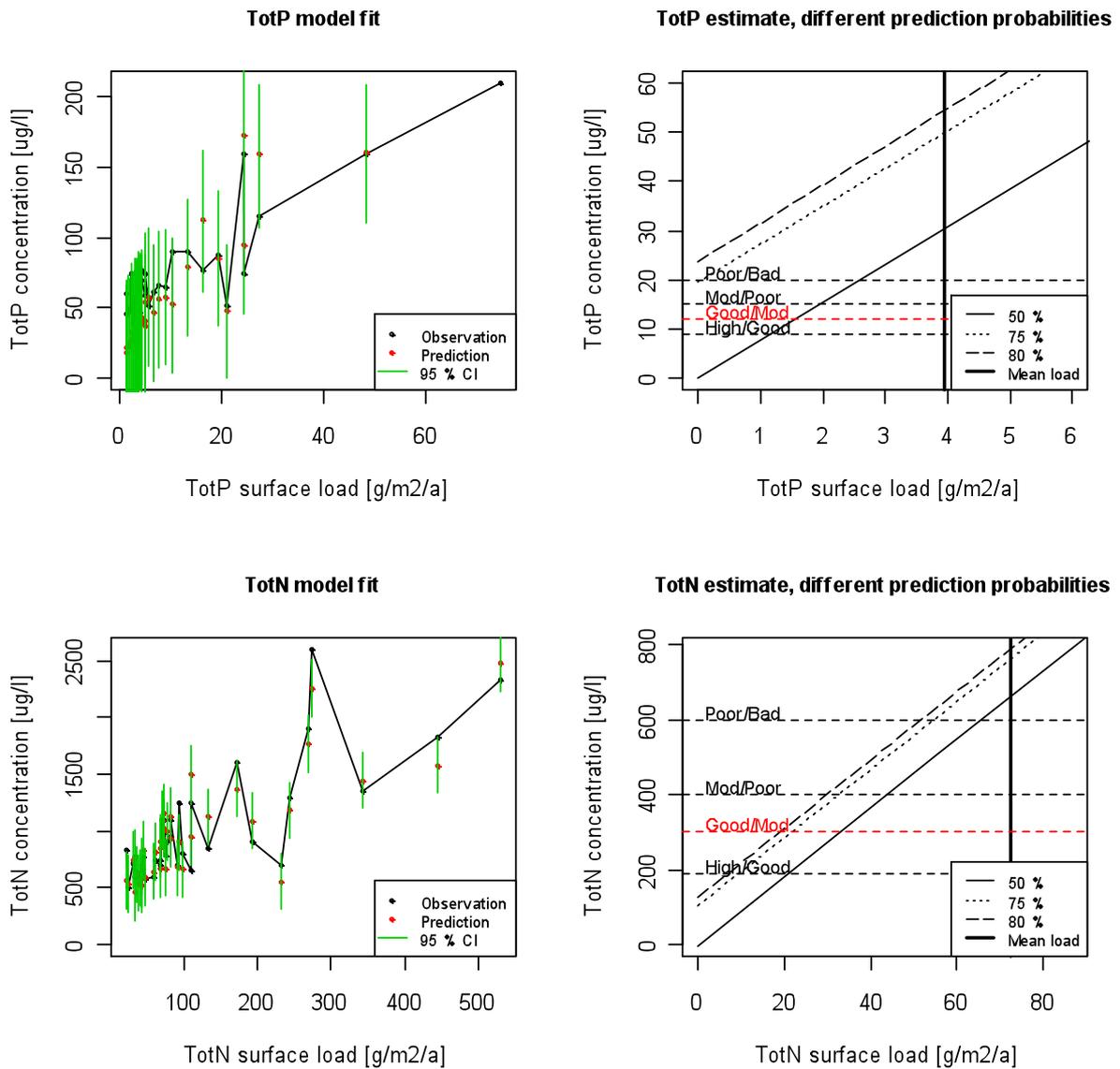
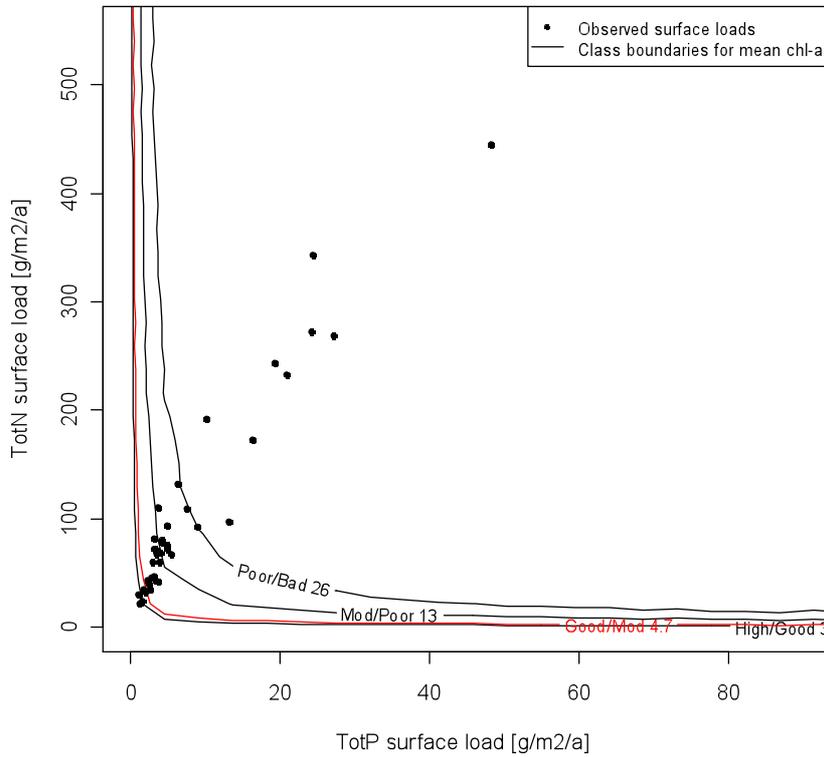


Fig 2.4. Models predicting concentrations of total phosphorus (TP) as a function of the areal TP load and concentrations of total nitrogen (TN) as a function of areal TN load. Figures in the right side show the confidences of 50, 75 and 80% to reach the status classes between high and good (H/G) status, good and moderate (G/M) status, moderate and poor (M/P) status and poor and bad (P/B) status. The upright lines show the average areal loads of nutrients in the period of 1988-2010.

Chl-a estimate as a function of incoming load



*Fig. 2.5 The LLR result on the concentrations of chlorophyll *a* estimated as a function of the areal loads of TP and TN ($\text{g m}^{-2} \text{a}^{-1}$) in River Vantaa estuary. The red curve shows the G/M boundary of chlorophyll *a* with 50% confidence in the combinations of TP and TN. The other boundaries of the ecological classification are also shown.*

Addition to phytoplankton biomass (here chlorophyll *a*), the WFD also requires other phytoplankton metrics in status assessments. In this study, cyanobacterial biomass was used to describe phytoplankton blooms, of which boundary was here defined based on chlorophyll *a*. In the inner River Vantaa estuary, chlorophyll *a* explained 63% of the variation in the biomass of cyanobacteria (Fig. 2.6). The boundary value of cyanobacteria was set as $185 \mu\text{g L}^{-1}$ using the G/M boundary of chlorophyll *a* ($4.7 \mu\text{g l}^{-1}$). Based on logistic regression, probabilities for cyanobacterial blooms increased along with the increase in phosphorus and the decrease in nitrogen concentrations (Fig. 2.7).

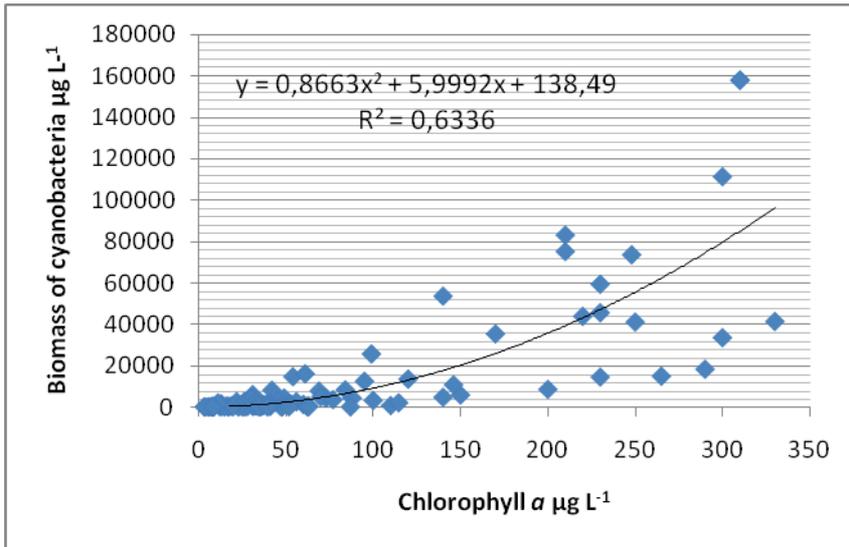


Fig. 2.6. The biomass of cyanobacteria estimated as a function of chlorophyll a in the inner estuary of River Vantaa.

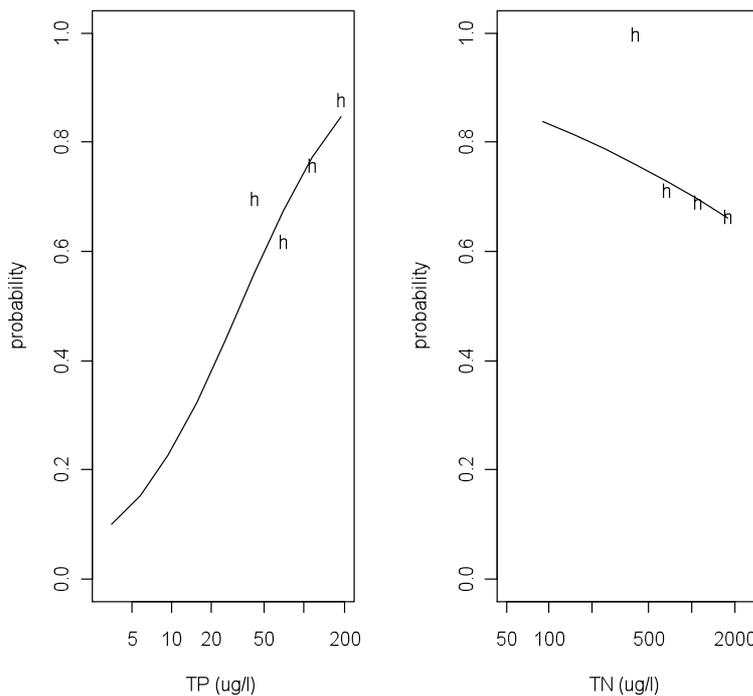


Fig. 2.7. Probabilities for cyanobacterial blooms estimated as a function of TP and TN in the inner estuary of River Vantaa.

2.3.4 Results on mechanistic approaches

Based on the results on current simulations, the direction of the flow varied between positive and negative values between the inner and outer estuary of river Vantaa (Figs 2.1b and 2.8). Consequently, the current direction changes frequently, and hence wind forcing strongly affects

the direction of currents in these sounds. In the main sound of the inner estuary of River Vantaa (Kulosaari W) a typical period of water flow to northwards is about 12-36 hours. In some cases, flow periods last even more than 150 hours.

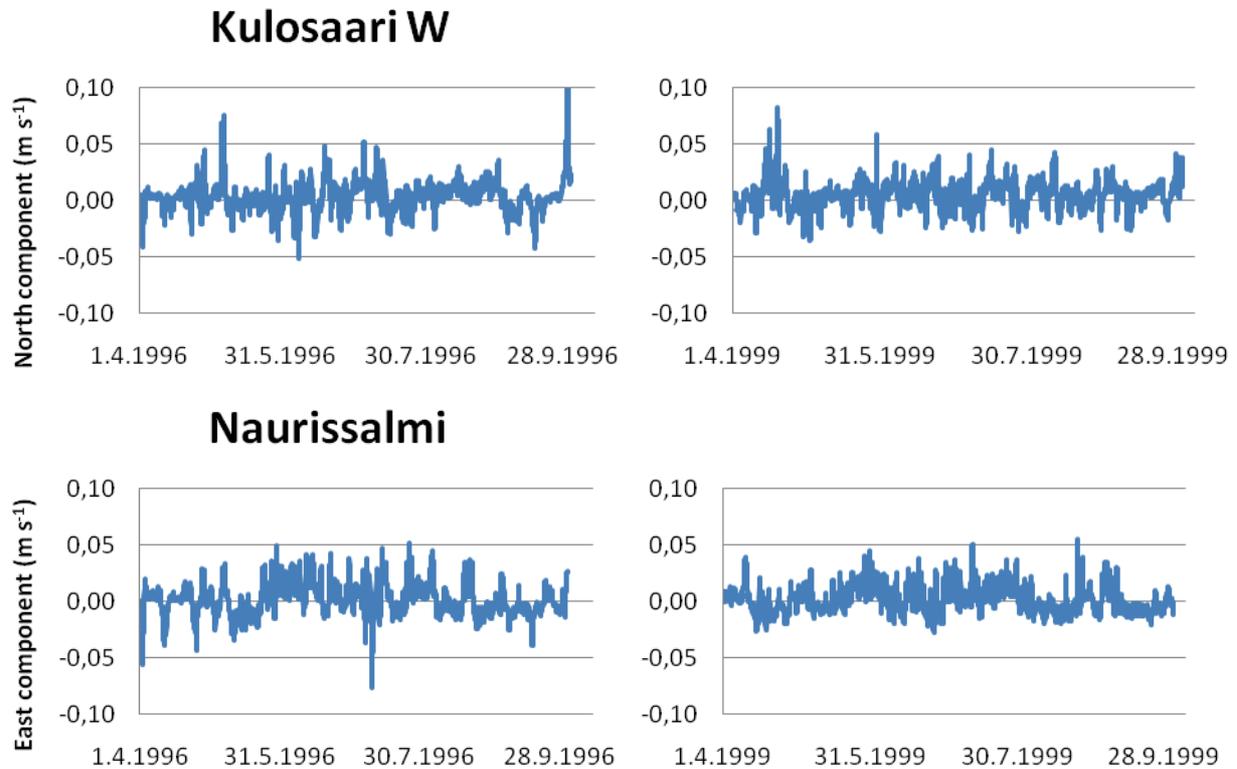


Fig. 2.8. Simulated currents in the inlets of the inner estuary of River Vantaa. The components of main direction of both inlets are shown.

Variation appeared when comparing the simulated and observed values of water temperature, salinity and nutrients at four stations of the estuary (Fig. 2.1b) in 1996 and in 1999 (Figs 2.9-2.12). The simulated temperature values followed quite well the observations at each station (Fig. 3) when considering both the calibration and validation periods. However, at the end of the summer season the simulated temperature values decreased too early in regard to the observations.

The simulated salinity values at Vanhankaupunginlahti seemed to be at too high level most of the time in regard to the monitoring data (Fig. 2.10), implying that too much water from the outer estuary comes to the inner estuary of River Vantaa in the model. Unfortunately, water current measures in these sounds were not available to model validation in this respect.

The simulated concentrations of total phosphorus corresponded well to the observations at all stations (Fig. 2.11). The simulations showed that the temporal variation is very intensive and the observations do not catch them at Katajanokka and Vasikkasaari stations. The simulated concentrations of total nitrogen matched very well to the observations at all stations (Fig. 2.12).

Additionally, simulated concentrations fitted to the observations both in the calibration period (1996) and in the validation period (1999), which makes the total nutrient model quite reliable.

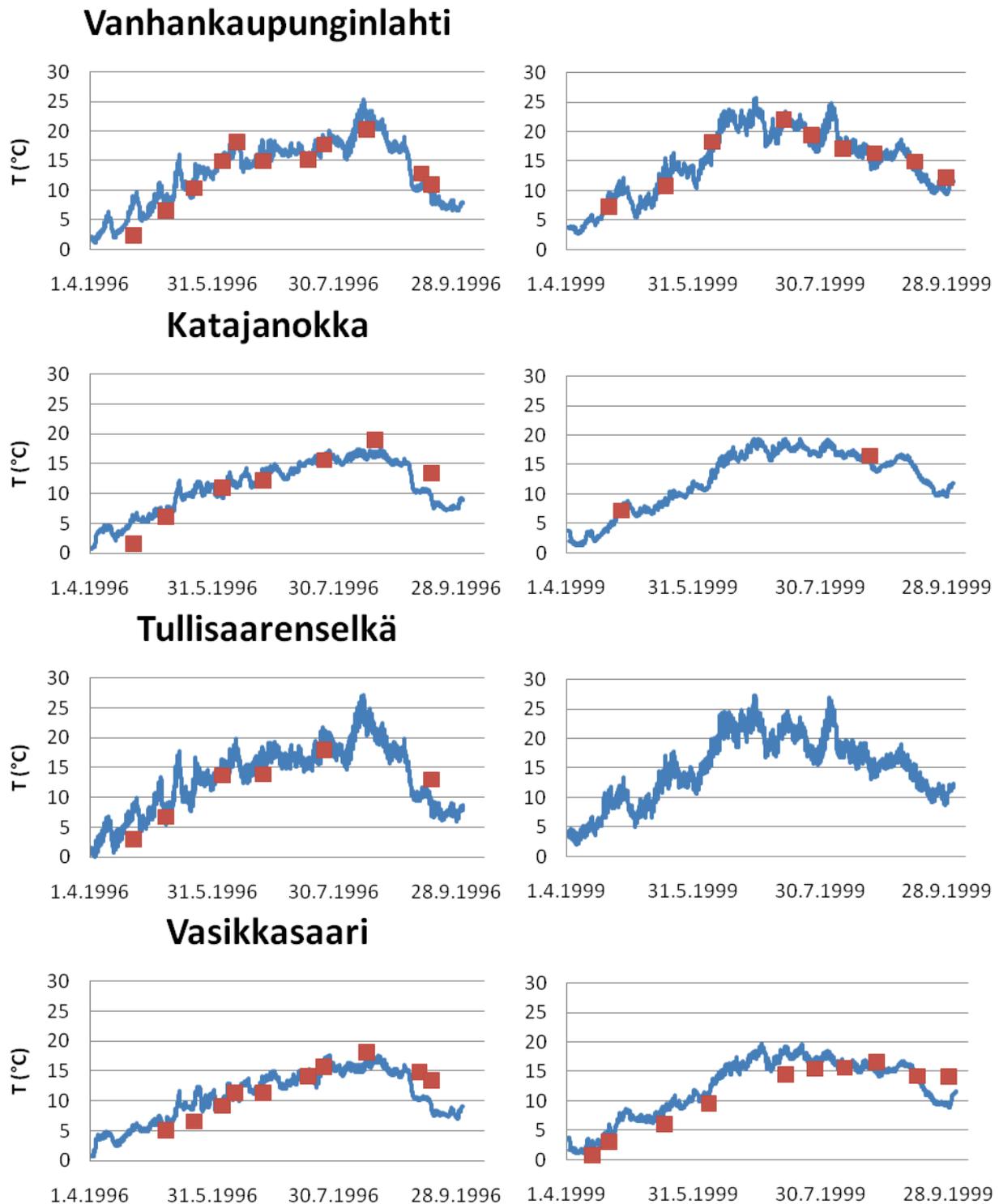


Fig.2.9. Simulated and observed temperature in surface water at Vanhankaupunginlahti, Katajanokka, Tullisaarenselkä and Vasikkasaari stations 1.4. - 1.10.1996 and 1.4. - 1.10.1999.

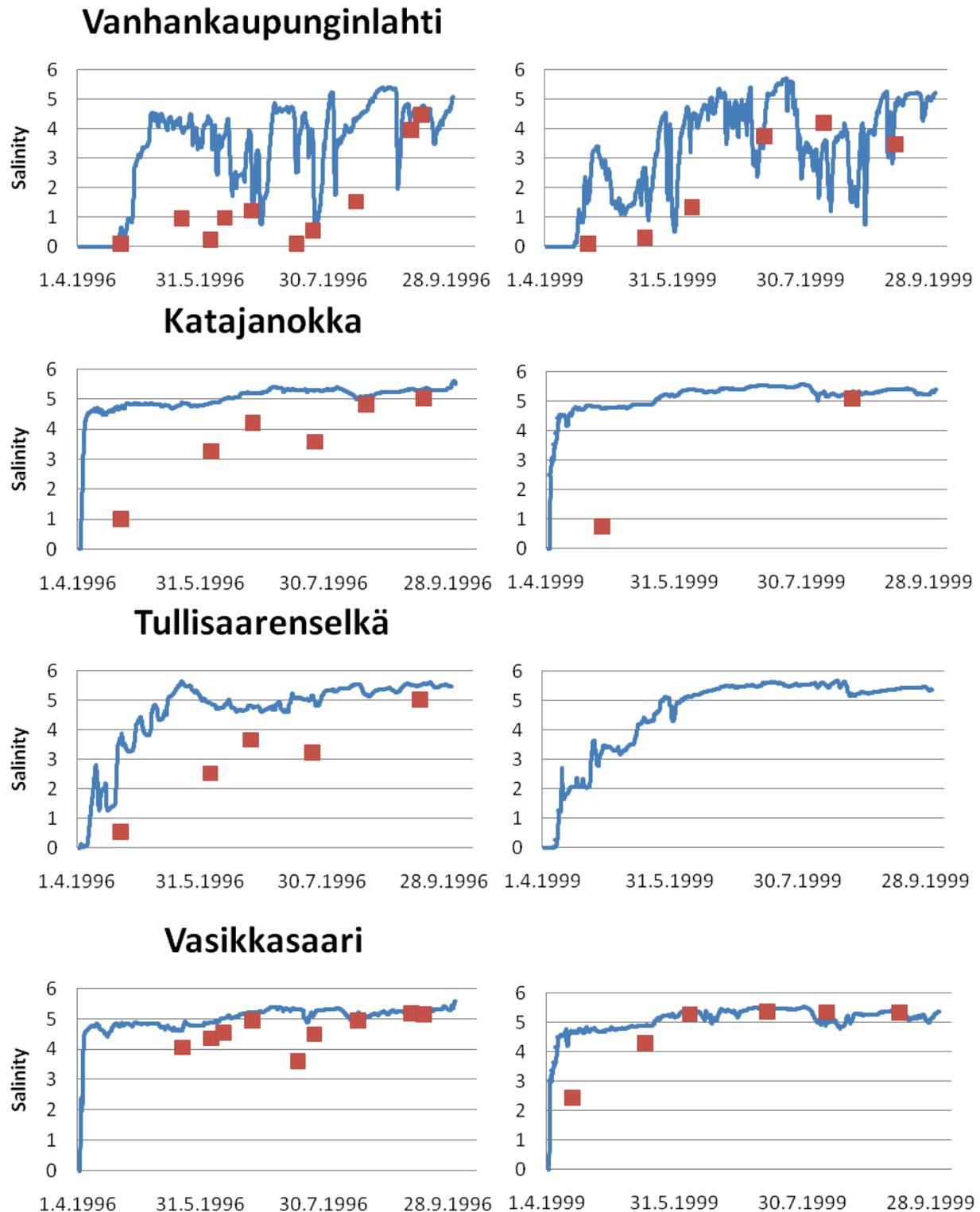
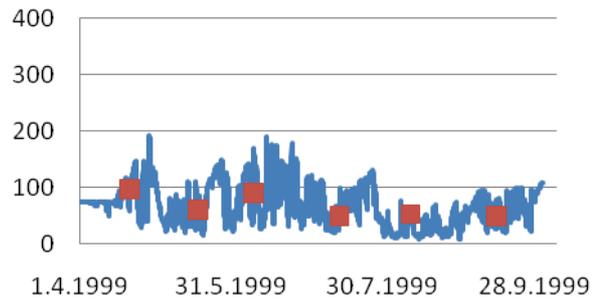
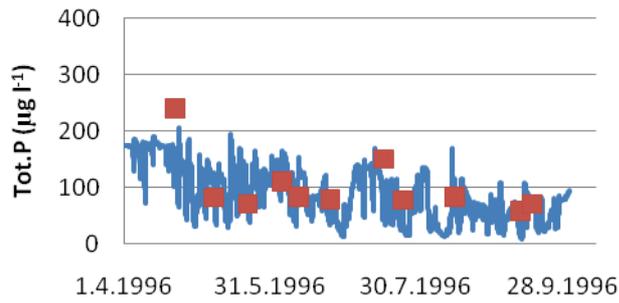
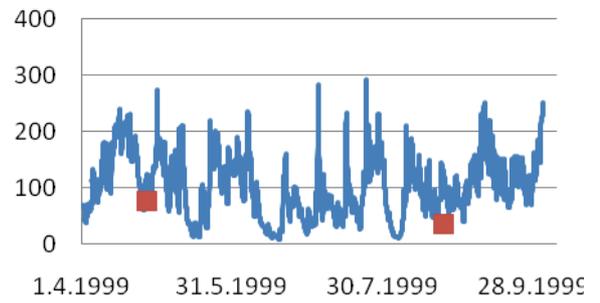
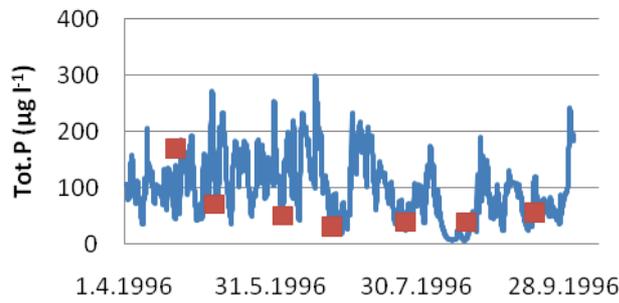


Fig. 2.10. Simulated and observed salinity (psu) in surface water at Vanhankaupunginlahti, Katajanokka, Tullisaarenselkä and Vasikkasaari stations 1.4.-1.10.1996 and 1.4.-1.10.1999. Salinity unit in simulations was PSU and in observations CN ‰.

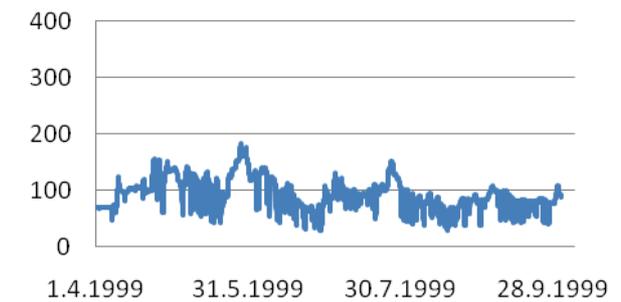
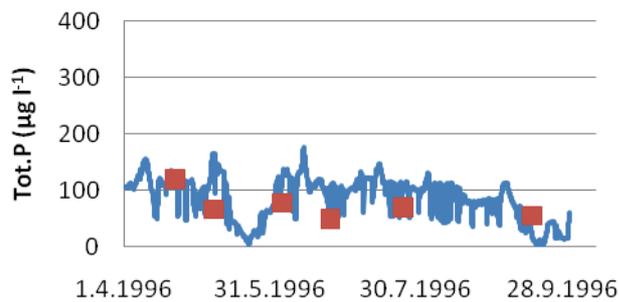
Vanhankaupunginlahti



Katajanokka



Tullisaarenselkä



Vasikkasaari

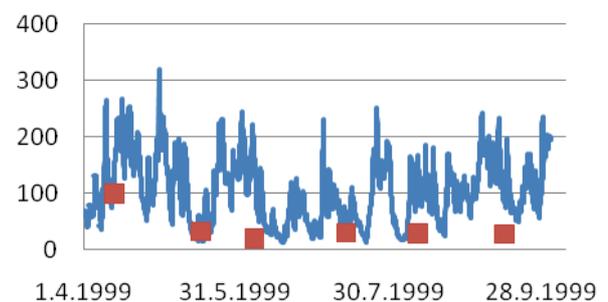
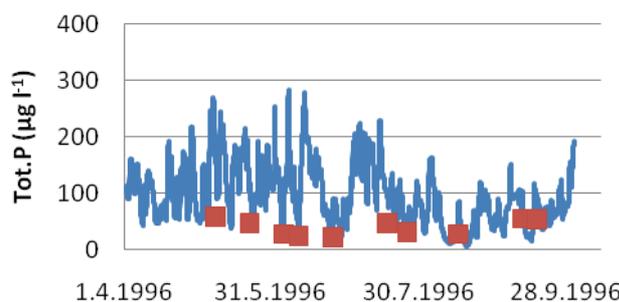
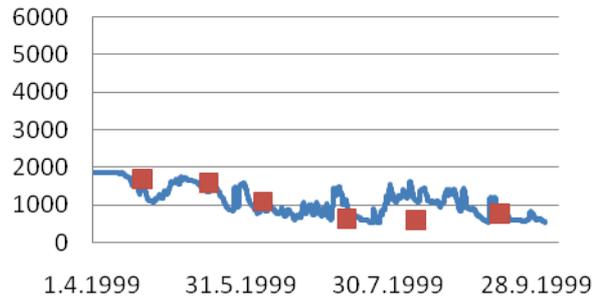
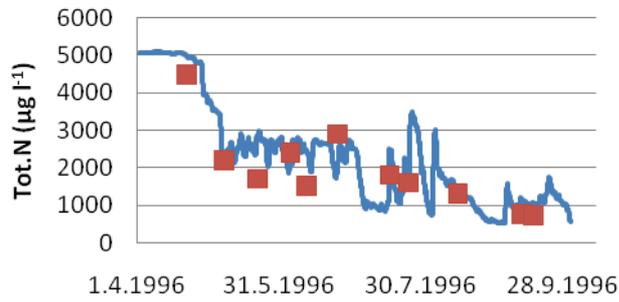
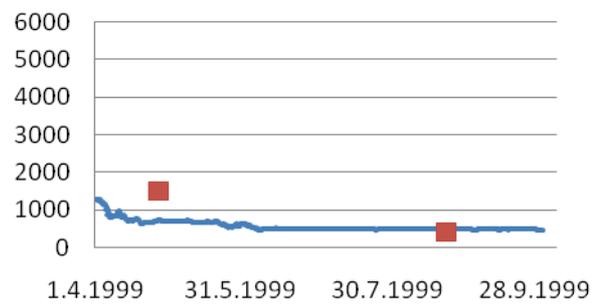
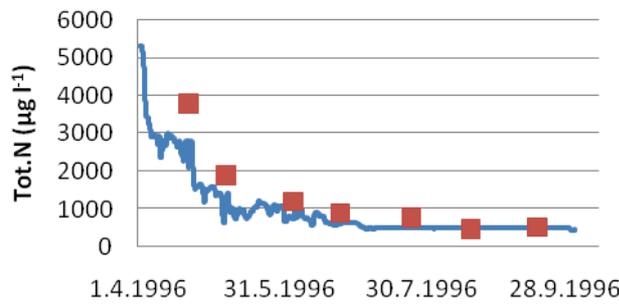


Fig. 2.11. Simulated and observed total phosphorus concentrations in surface water at Vanhankaupunginlahti, Katajanokka, Tullisaarenselkä and Vasikkasaari stations 1.4. - 1.10.1996 and 1.4.- 1.10.1999.

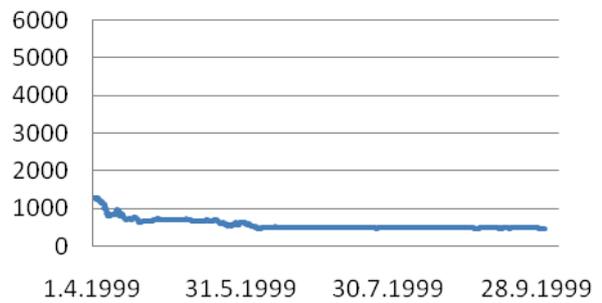
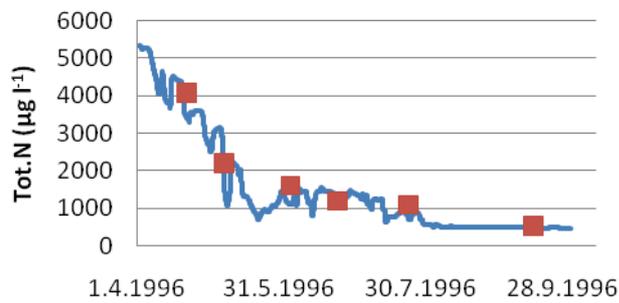
Vanhankaupunginlahti



Katajanokka



Tullisaarenselkä



Vasikkasaari

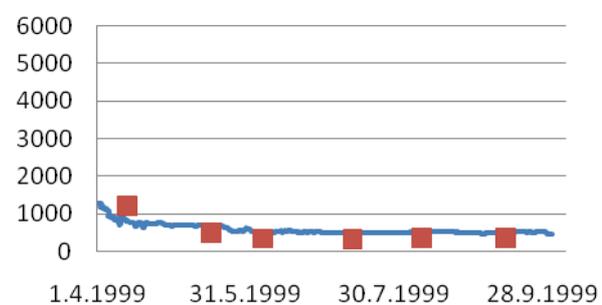
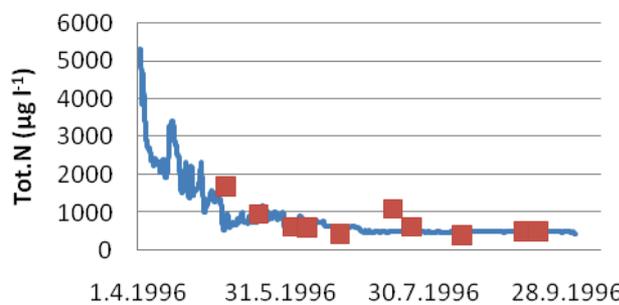


Fig. 2.12. Simulated and observed total nitrogen concentrations in surface water at Vanhankaupunginlahti, Katajanokka, Tullisaarenselkä and Vasikkasaari stations 1.4.-1.10.1996 and 1.4.-1.10.1999.

Exchange of water was demonstrated by simulations of the surface concentrations of total nutrients in certain moments of 1996. In each day (first days of each month of the period), total phosphorus concentrations near the river mouth were below $25 \mu\text{g l}^{-1}$, which indicated that minor amounts of total phosphorus entered the estuary from the river. By contrast, especially in the 1st June and the 1st October, a major input of phosphorus appeared to enter the estuary from the open sea. The model demonstrated that the advection from the sea is dominating over the advection from River Vantaa. Based on the time series, phosphorus concentrations in the outer estuary showed more variation than in the inner estuary (Fig. 2.11), which also supported our conclusion in that most of the in-flowing phosphorus into the estuary originated from the open sea.

The momentary concentrations of nitrogen showed that great amounts of total nitrogen flowed into the estuary from River Vantaa in first days of May, June and July. In each of the simulations, the concentrations were below $1000 \mu\text{g l}^{-1}$ near the open sea boundaries, revealing that major inputs of total nitrogen enter the estuary from River Vantaa. The time series of nitrogen in spring 1996 (Fig. 2.12) gave extra support to our hypothesis of the riverine origin of nitrogen. Otherwise, nitrogen levels stayed very stable outside the inner estuary, proving that no significant loads of nitrogen are coming from the open sea.

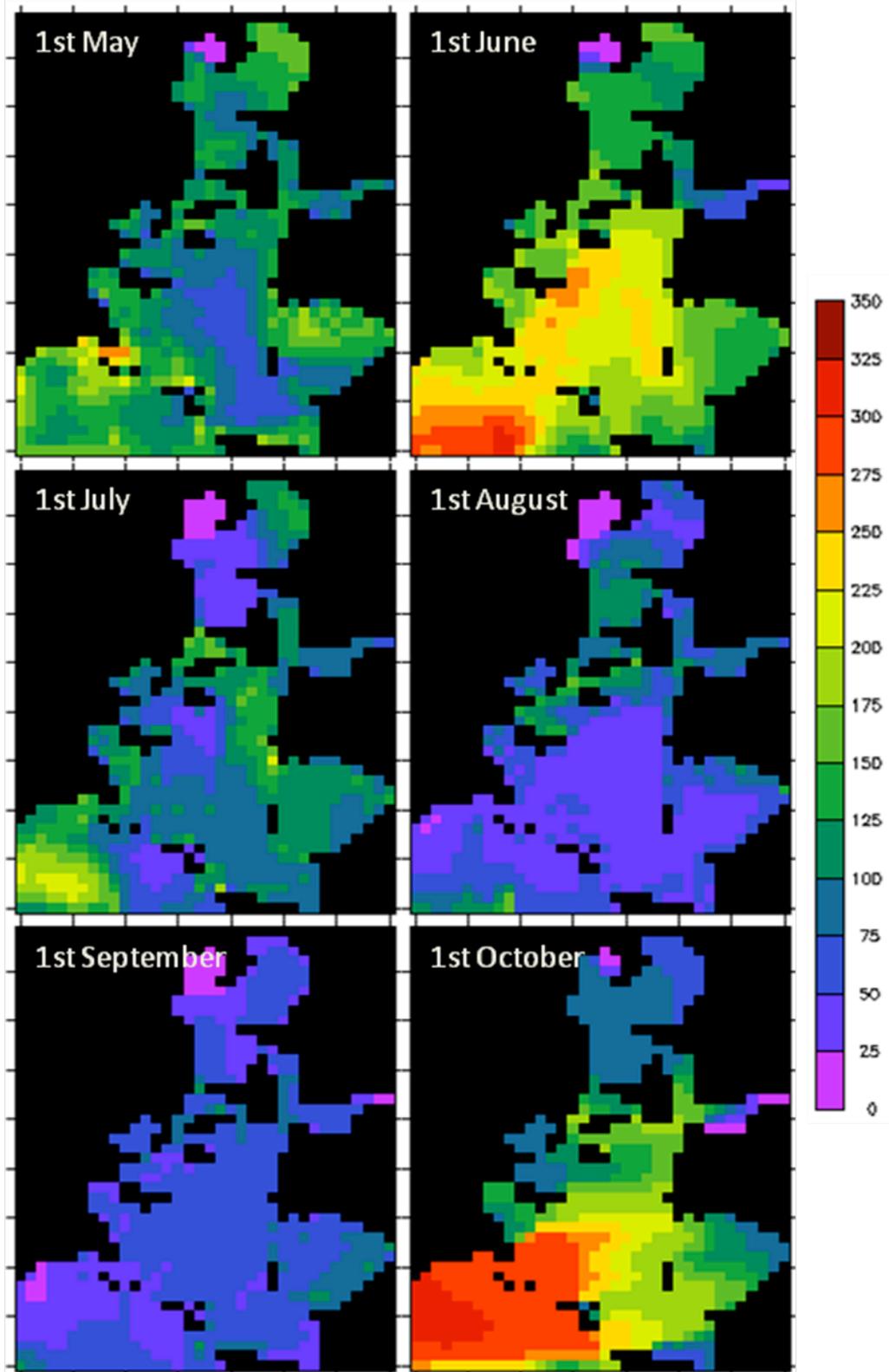


Fig. 2.13. Simulated total phosphorus concentrations ($\mu\text{g l}^{-1}$) in surface layer in 1996.

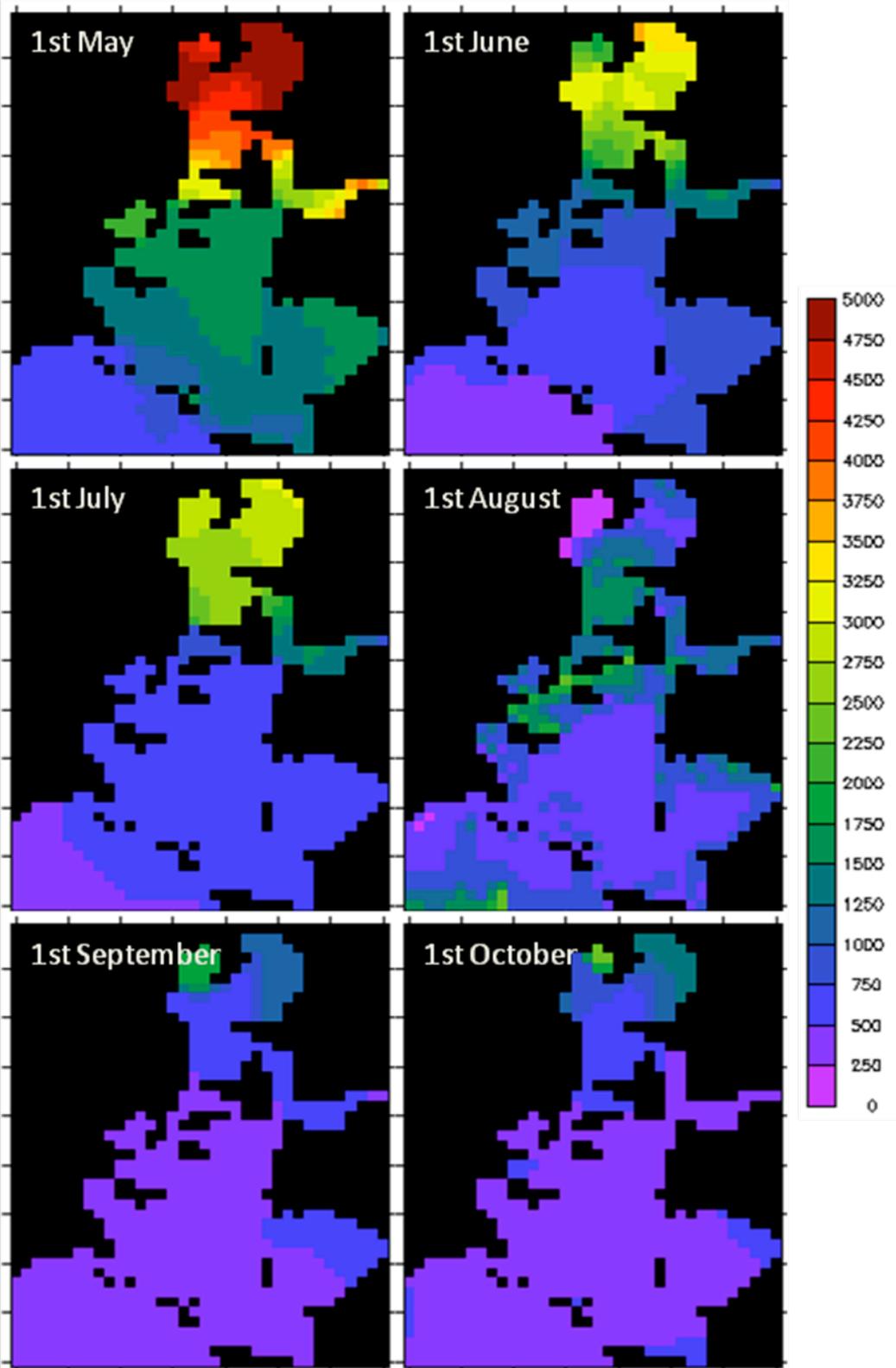


Fig. 2.14. Simulated total nitrogen concentrations ($\mu\text{g l}^{-1}$) in surface layer in 1996.

To show the influence of the varying weather conditions, we run two additional simulations for the year 1996. In the basic simulation of 1996 (Sim.1), a measured daily flow of River Vantaa was used as input data (Fig. 2.15), whereas in simulation 2, the water flow values of 1996 were multiplied by 0.5 to illustrate the affects of dry summers. The average water flow between the period of May and September was then estimated as $8.1 \text{ m}^3 \text{ s}^{-1}$ (Sim. 2). For simulation 3, the water flow values of 1996 were multiplied by 1.5 to illustrate the affects of wet summers in which case the average summertime water flow was estimated as $24.4 \text{ m}^3 \text{ s}^{-1}$. The concentrations of nutrients in the river water were same in each simulation.

Changing the water flow of River Vantaa caused only small differences to the simulation results of salinity and nutrient concentrations at Vanhankaupunginlahti station (Fig. 2.16). In some periods of time, salinity was a little bit higher in the simulation of dry summer than in the simulation of wet summer, since less river water entered to the estuary. However, in August salinity was higher during the wet summer, most probably due to an unusual intrusion of more saline seawater to the bay area.

Additionally, total phosphorus concentrations were at a higher level in the middle and at the end of the simulation 2 (dry summer) than in the simulation 3 (wet summer). A reduction in phosphorus loading from the river resulted in an increase in phosphorus concentration at the station, which supported the previous conclusion that a significant amounts of total phosphorus comes to the estuary from the open sea. Total nitrogen concentration could only minimally be affected by simulating the water flow of River Vantaa.

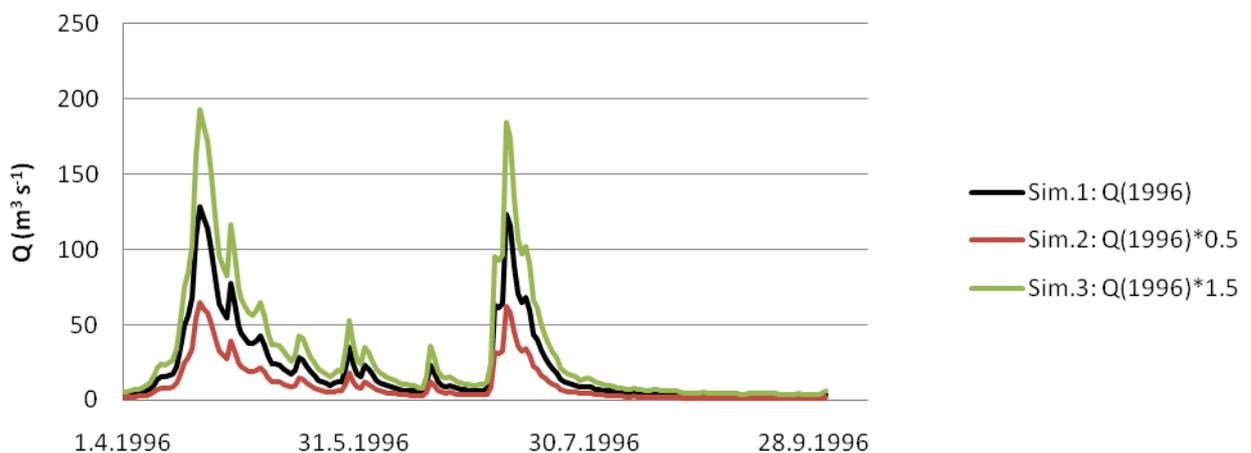
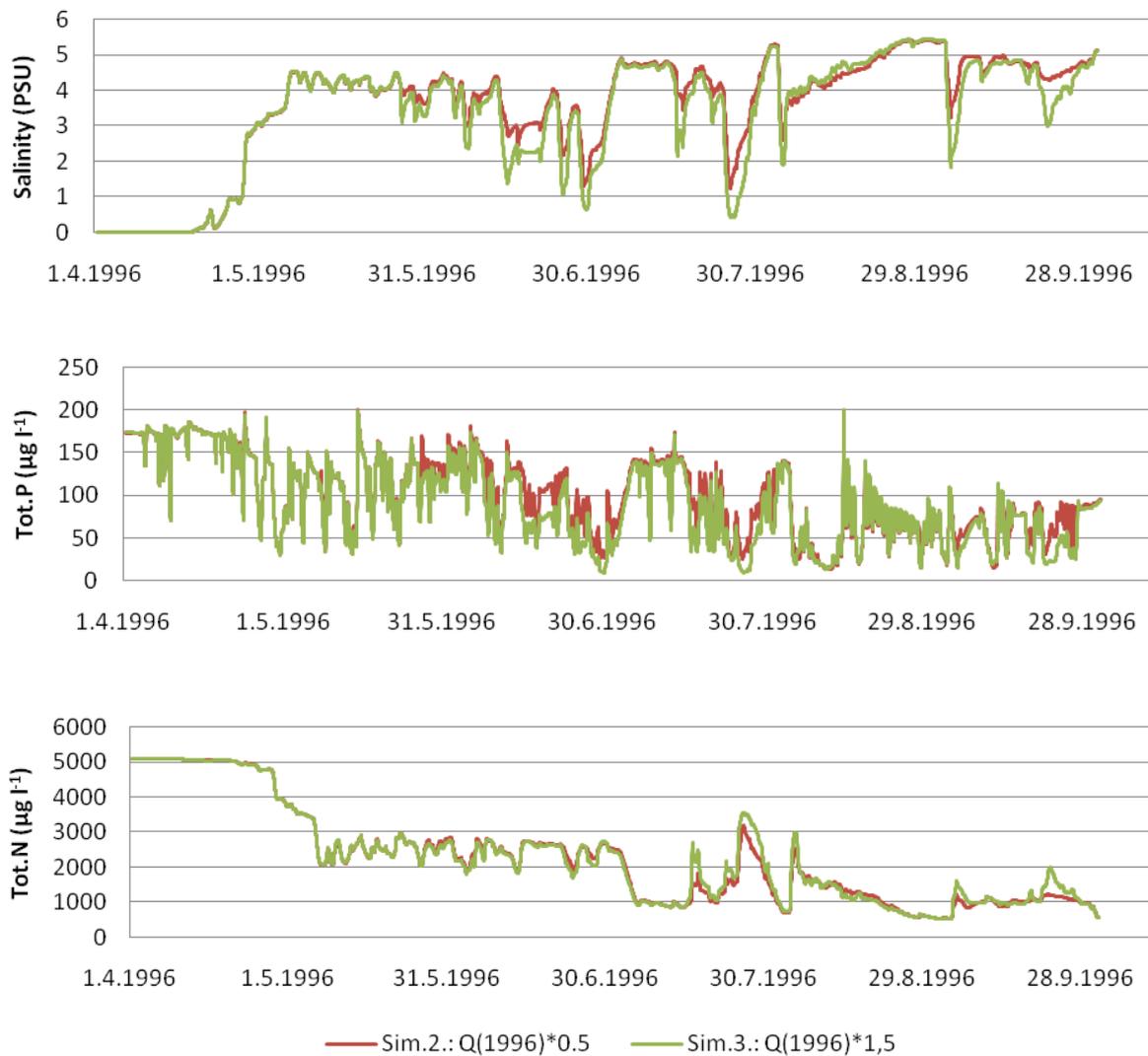


Fig. 2.15 Daily flows of River Vantaa in three simulations in 1996. Simulation 1 is the situation of 1996, where daily flow of River Vantaa is used. In simulation 2 the flow of 1996 was multiplied with 0.5 to illustrate a dry summer. In simulation 3 the flow of 1996 was multiplied with 1.5 to illustrate a wet summer.



Fig

Fig. 2.16. Salinity and total nutrient concentrations (in surface layer at Vanhankaupunginlahti station in 1996 in simulations with different flows in River Vantaa. In simulation 2 the daily flow of 1996 was multiplied with 0.5 to illustrate a dry summer. In simulation 3 the flow of 1996 was multiplied with 1.5 to illustrate a wet summer.

2.4 Discussion

Coastal management can benefit both of the results of statistical and mechanistic approaches, which provide different aspects in looking at responses of eutrophication to changing input and natural forces in coastal marine waters. In this study, the 3-D Coherens model represented a mechanistic approach, whereas traditional regression analyses together with the application of the LakeLoadResponse (LLR) tool were used as statistical approaches. In general, dynamic simulation models are complex and required a lot of computer capacity, whereas statistical models are simple and quick and they do not identify cause and effect relationships contrary to dynamic models. In the present case, calculating currents, salinity, temperature and nutrient

concentrations in each grid cell through time enabled the Coherens model to illustrate regional currents and transport of substances in River Vantaa estuary.

Additionally, it was possible by the LLR tool to estimate target nutrient loads to achieve good ecological status in the estuary. Together the preliminary results of these approaches gave information on the behaviour of the coastal system and the relationships of different environment factors in River Vantaa estuary. Due to large nutrient loading, elevated trophic conditions in River Vantaa estuary was evident based both on the simulation results of the Coherens model and the statistical results derived from regression and LLR analyses. The model simulations revealed that loading of total nitrogen in the estuary originated mainly from River Vantaa (Fig. 2.12). Hence it implied that decreasing nitrogen concentrations in the river would also decrease concentrations in the estuary. By contrast, the mechanistic simulations suggested that most of total phosphorus would come, not from the river, but from the outer coastal and open sea waters (Figs 2.11 and 2.16). Consequently, reduction in phosphorus inputs from the river alone would not be enough to improve the trophic status of the estuary. The exchange of the estuarial water with the open sea water actually means that it is impossible substantially to influence the phosphorus level of the estuary without decreasing phosphorus concentrations in the Gulf of Finland as well.

Exchange of water with the open sea does not, however, alone explain why phosphorus concentrations and general trophic level in the estuary are clearly elevated. The huge loading in the 1970s and 1980s (Pesonen 1988) raised nutrient reserves in the bottom sediment and accelerated benthic release of nutrients, i.e. internal loading, which could be proved by an atypical seasonal distribution of phosphorus and high amounts of chlorophyll *a* in summer (Fig. 2.3). In the 1970s and mid'1980s, the levels of total and inorganic phosphorus in the whole water column were on an average 30% and in some years up to ca 70% higher in summer than in spring. On the contrary, the contribution of resuspension was most probably small based on the small amounts of inorganic P and substantial amounts of organic phosphorus measured as chlorophyll *a* in summer. However, the high percentages of inorganic P of total phosphorus in the 1970s and inorganic N of total nitrogen in the late 1980s could be explained by the peaks of the above-mentioned nutrients in municipal waste waters.

After the closure of the LMTPs, the effect of internal loading decreased based on the seasonal distribution of phosphorus, which started to show spring maxima and summer minima. Typical seasonal distributions of phosphorus were also demonstrated by the simulations of the Coherens model (Figs 2.3 and 2.11). As a measure of organic production, the average concentrations of chlorophyll *a* in the summers of the 2000s decreased more than 70% from the level of the 1980s, supporting the reducing role of the benthic release of phosphorus. However, concentrations of chlorophyll *a* and phosphorus were still considerably high compared to the open sea (Räsänen *et al.* 2011), suggesting that the estuary was not entirely released from internal loading despite good oxygen conditions in near- bottom waters.

In statistical models, nitrogen was the best predictor for chlorophyll *a* (Table 2.2, Fig. 2.4). This was expected based on the ratio of DIN/TP: primary production has mainly been N limited since the 1970s. The only exception was the early 1980s, the short period when P-limitation prevailed

as a consequence of stating chemically to remove phosphorus from effluents. In fact, nitrogen limitation in Vantaa estuary is maintained by several processes. First, substantial amounts of inorganic P are released from the water-sediment interface also in good oxygen conditions (Rekolainen 1982). Additionally, the estuary exchanges water with the N-limited open coastal waters (cf. Kivi *et al.* 1993, Tamminen and Andersen 2007). The role of denitrification is probably also considerable in removing large amounts of N at the sediment – water interface on the basis of high organic production measured as chlorophyll *a*. Moreover, wintertime concentrations of nitrate-N (NO₃-N accounting 50-90% of TN) have also been constantly high in the water column since the mid' 1980s, and hence subject to denitrification.

Nutrient conditions strongly control phytoplankton amounts and community structure. According to Pellikka *et al.* (2007), the amounts of phytoplankton collapsed in River Vantaa estuary as a result of improving the purification efficiency and closing of local municipal treatment plants in the late 1980s. At the same time, phytoplankton composition changed from blue-green-algae-dominated to diatom-dominated, the main taxa being the non-N₂-fixing blue-green alga *Planktothrix agardhii* and the diatom genus *Skeletonema*. In the 1990s, the community was usually dominated by diatoms, whereas in the 2000s green algae displaced cyanobacteria. Our results showed that the probability for cyanobacterial blooms in the inner estuary increased along with the increase in phosphorus concentrations and the decrease in nitrogen concentrations (Fig. 2.7). The negative correlation with nitrogen could be explained by light limitation due to substantial amounts of river-driven suspended solids (cf. Korpinen *et al.* 2004). The results on bloom probabilities are in accordance with the simulation results in that nitrogen in the estuary is river-driven (Fig. 2.12).

Target nutrient loads in River Vantaa estuary could be estimated by the LLR tool, which was fitted to this individual coastal water area. With 50% confidence, the target load of phosphorus was estimated as 1.6 g m² a⁻¹, which corresponded to 12 µg L⁻¹ of nitrogen in summer conditions. In case of nitrogen, the target load was estimated as 33 g m² a⁻¹ and the corresponding nitrogen concentrations 300 µg L⁻¹, in respectively. Nutrient concentrations seemed realistic in regard that they are at the same level as the G/M boundaries of summertime TN and TP in the oligosaline waters (4-5 psu) of the outer Bothnian Sea and Askö archipelago (NFS 2008).

Based on the River Basin Management Plan (RBMP) of the Gulf of Finland, River Vantaa estuary will fail to achieve good ecological status by 2015, despite the closure of the local municipal treatment plants by 1987. Today, the estuary receives external nutrient loading only through River Vantaa: agriculture is responsible for 60% of the N load and scattered settlement for 60% of the P load (data not shown). Thus, the most important environmental targets listed in the present RBMP are measures that are related to the reduction of nutrients from field cultivation. Regarding settlement, building transfer sewers and reconstructing sewer network will be the most important actions.

The scenarios of VIHMA model present the effects of cultivation practises on nutrient inputs in Aurajoki river basin, the catchment resembling Vantaa River basin. Based on the scenarios, the potential reduction of erosion in Aurajoki river basin is 45 % at the maximum of the present

load taking into account the total effects of reduced tillage practices, buffer zones and constructive wetlands (Puustinen *et al.* 2010). In the case of TN and TP, the reductions are 39 and 15% at the maximum, in respectively. On the other hand, hydrological variation may be so great that major part of the reductions is counteracted by the annual variations of runoff and water flows (Puustinen *et al.* 2007, 2010).

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3. Spanish coastal waters: the Nervion estuary

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3.1 Study area and chlorophyll a evolution

River Nervión estuary is a mesotidal small estuary north of the city Bilbao, on the north Spanish coast. It receives fresh water from the Nervión River with a mean flow of $25\text{-}30\text{ m}^3\text{ s}^{-1}$. The Nervión is 22 km long with two main areas: an inner channel heavily stratified and less than 10 m deep, and an outer area slightly stratified with depth between 10 and 30 m. The estuary is connected to the open Bay of Biscay in the North Eastern Atlantic.

The natural features of the estuary have been drastically modified by urban, industrial and port settlements. Over the last 150 years the estuary received wastes from many sources (urban effluents, industrial wastes, mineral sluicing), and suffered many morphological pressures. The system was degraded: low oxygen, high levels of bacteria and contaminants, disappearance of fauna. It is classified as Heavily Modified Water Body. Then, a Sewerage Scheme was approved in 1979. The water treatment began in 1991 with physico-chemical treatment; the biological treatment came into operation in 2001. It serves a total population of ~ 1 million people. A monitoring program started in 1990 to assess the efficiency of the treatment.

Since this treatment in 1990, a decreasing temporal trend of ammonia in the estuary, and an increasing temporal trend of transparency are observed (Fig. 3.1). The first should reduce eutrophication risks and the second, changing the light conditions and thus implying a deeper photic layer, should increase the eutrophication risk. As we can observe on the figure 3.2, there is no statistically significant temporal trends in the annual mean chlorophyll biomass for the all stations (lower panel), but for some stations (Deusto, Axpe, Abra exterior – upper panel), the annual mean of chlorophyll *a* increases in the last years.

Thus, the objective of this study is to access which processes play roles on the time evolution of the chlorophyll *a*, taking in mind that the light “increasing” and the ammonium decreasing should have opposite effects.

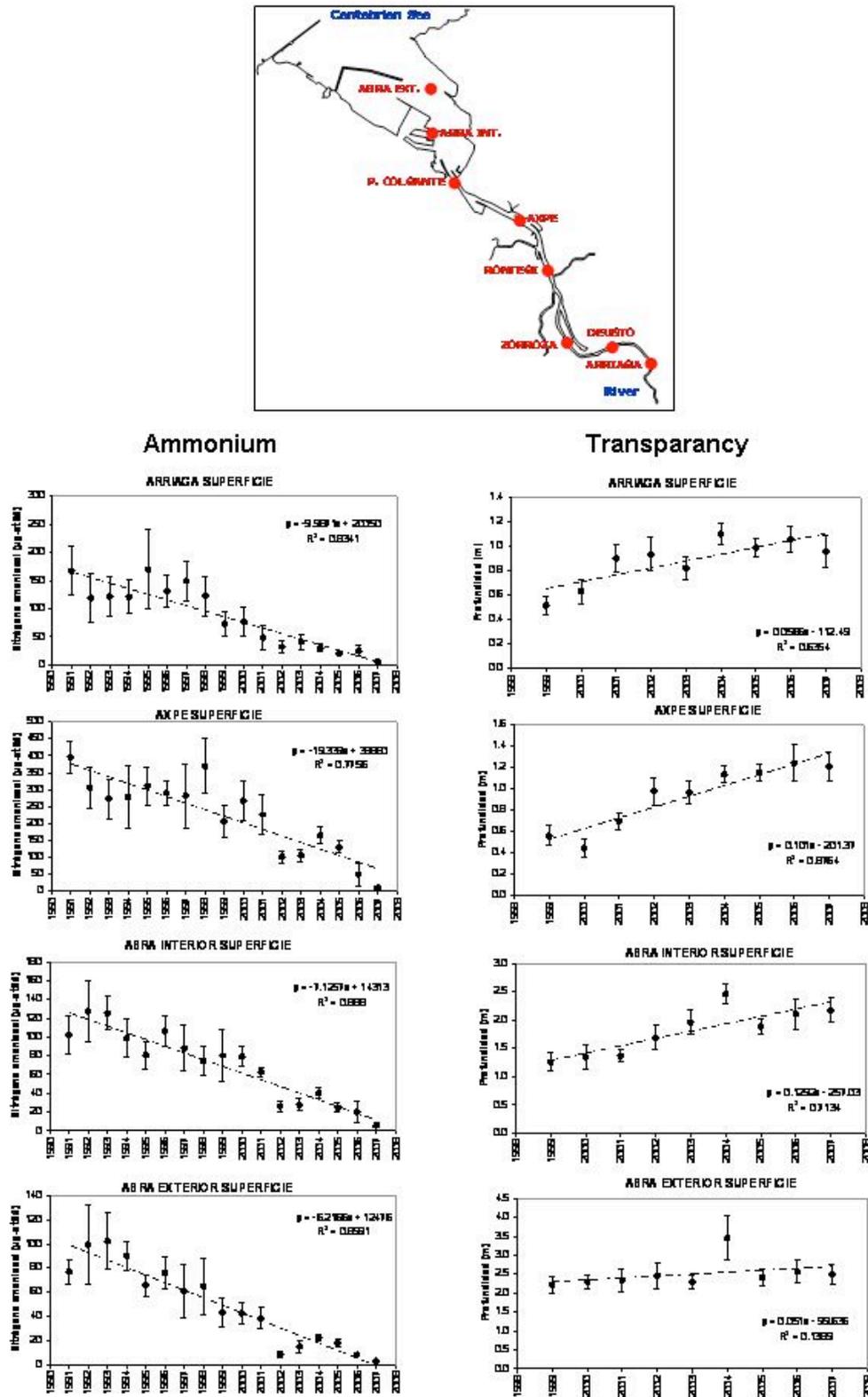


Figure 3.1: Upper panel: Maps of the Nervión estuary and situation of sampled stations. Lower panels: Ammonium concentration (left panels) and Sechi disc depth (right panels) at stations Arriaga, Axpe, Abra interior and Abra exterior.

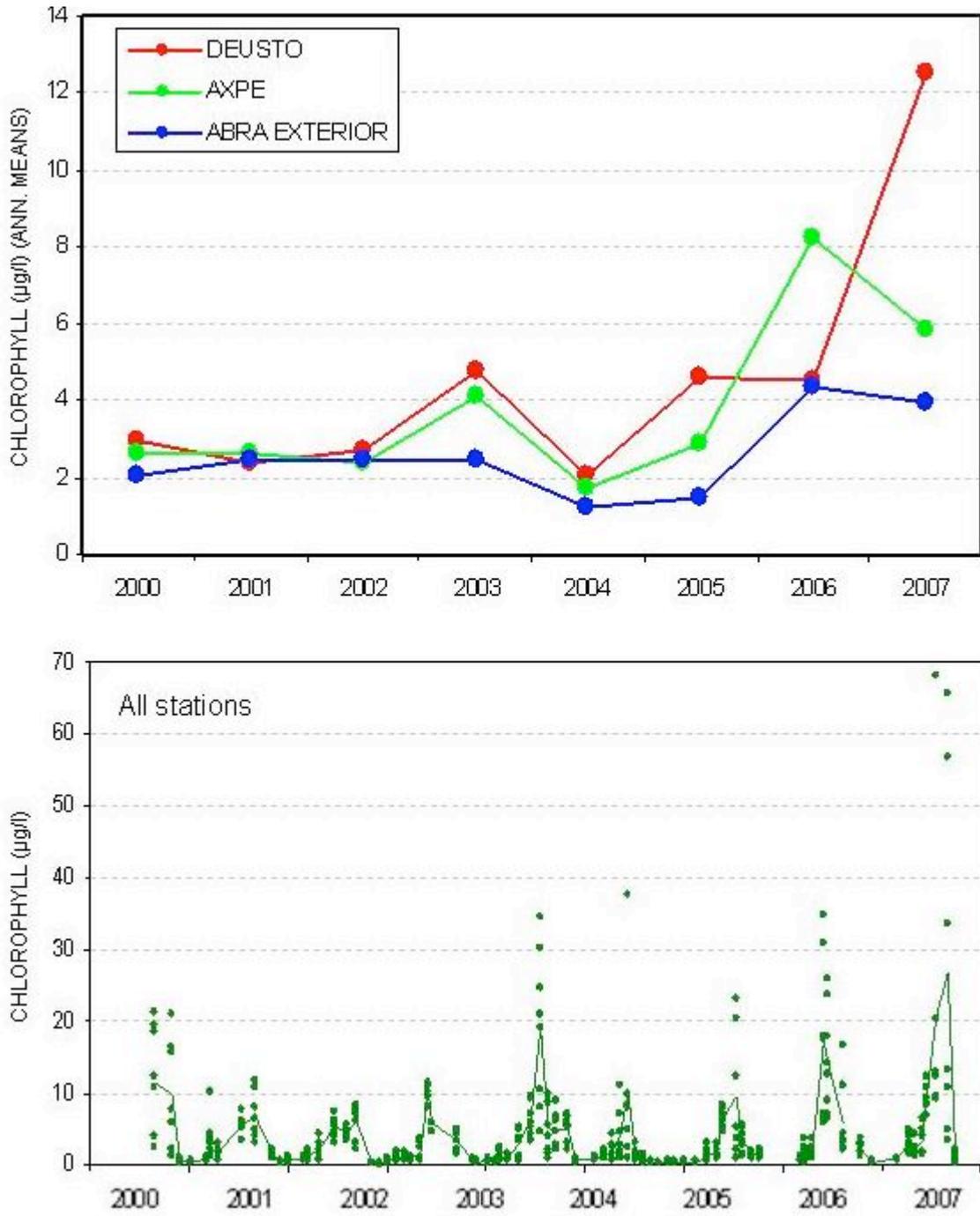


Figure 3.2: Chlorophyll a biomass evolution over the years 2000-2007: specific stations (upper panel) and all stations considered (lower panel).

3.2 Material and methods

3.2.1 Datasets

Daily observations of Nervion flow were used from the stations NB04 (Nerbioi at Zaratamo), IB03 (Ibaizabal at Amorebieta) and KD03 (Kadagua at Sodupe) of the “Diputación Foral de Viscaya”. The observations are available since September 1995.

Nervi3n River and Estuary monthly observations of temperature, salinity, ammonium, nitrate and chlorophyll *a* were imposed in the model as boundary conditions at the Nervi3n river points. Observations of temperature, salinity, ammonium and chlorophyll *a* were kindly provided by the “Consortio de Aguas Bilbao Bizkaia”, institution responsible of water supply and sewerage in the Bilbao area, which carries out water quality monitoring since 1989. (Franco *et al.*, 2011). There are 8 stations in this monitoring system (see map on Fig. 3.1), and observations at station 3 (P. Colgante, Fig. 3.1) situated at the transition between the river and the estuary, were used.

In order to complete the boundary conditions for nitrate, we have used observations from a station in the inner estuary from the “Red de calidad” (Borja *et al.*, 2011). As for the previous variable, the river boundary conditions for nitrate are monthly.

3.2.2 Statistical approach

Stepwise regression analyses were used to explain variation in chlorophyll *a* in a function of total nitrogen (or separated: ammonium or nitrate), total phosphorus, light, salinity and temperature in the inner estuary of the Nervion River. Additionally, these multiple regression analyses were also applied to spring and summer periods, when chlorophyll *a* usually develops.

3.2.3 Mechanistic model

3.2.2.1 Physical model

As the principal hydrodynamic forcing, only tidal currents within the estuary have been considered in this study. A depth integrated 2-D hydrodynamical model based on the Finite Element Method has been used and applied to the Nervi3n estuary (Gonz3lez *et al.*, 1995; John and Schmeier, 2008). A high-resolution finite element computational mesh of the estuary, with an average size of 85 m, a minimum of 3 m and a maximum of 250 m, has been constructed for the model implementation (Fig. 3.3). This grid covers the entire estuary from the river to the open sea. The model considers the main 7 tidal components to compute the currents. In the study area, the tidal current is dominated by M2 (principal lunar semidiurnal component) and S2 (principal solar semidiurnal component).

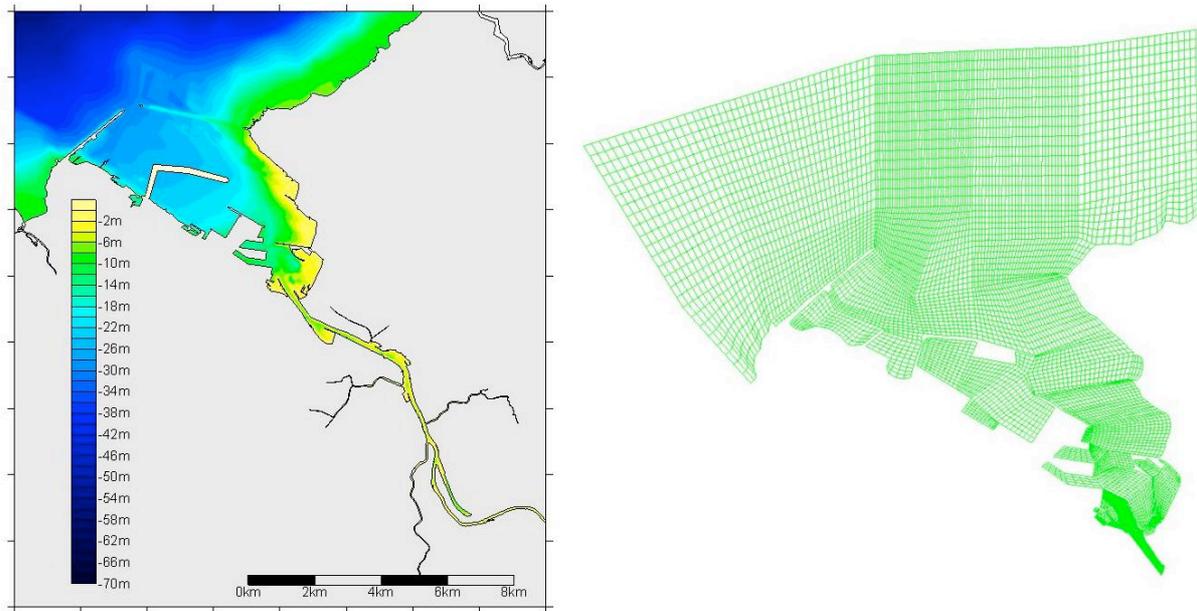


Fig. 3.3: Bathymetry of the suited area (left panel) and the Finite Element Method model grid (right panel) with an averaged size of 85 m.

3.2.2.2 Biogeochemical model

A biological NPZD-type model, coupled to this dispersion model, has been developed. The planktonic model takes into account 8 variables: nitrate, ammonium, small phytoplankton, large phytoplankton, small zooplankton, large zooplankton, end two classes of detritus. The model developed was from the NPZD-type model already developed in the ROMS model (Regional Ocean Model System, Shchepetkin and McWilliams, 2005; NPZD model from Fasham *et al.*, 1990).

3.2.2.3 Boundaries Conditions

The Nervión river runoff data was prescribed as boundary conditions on momentum. Daily flow data were used from from the stations NB04 (Nerbioi at Zaratamo), IB03 (Ibaizabal at Amorebieta) and KD03 (Kadagua at Sodupe) of the “Diputación Foral de Vizcaya”. When data were not available, before September 1995, we have computed daily climatology, and have performed linear interpolation for missing data during the available period. Finally a monthly climatology has been calculated to impose boundary condition in the model in term of flow at the Nervión river points.

Nervión River and Estuary monthly observations of temperature, salinity, ammonium, nitrate and chlorophyll a were imposed in the model as boundary conditions at the Nervión river points, using the observations provided by the “Consorcio de Aguas Bilbao Bizkaia” (see 3.2.1 Datasets)

The initial and open sea boundaries conditions for salinity and the 8 variables in the planktonic model, were interpolated on the Nervión model grid from a larger configuration of the ROMS model (Regional Ocean Model System, Shchepetkin and McWilliams, 2005) which covers the

entire Bay of Biscay. The initial and boundaries conditions of temperature were interpolated from observations.

3.2.2.4 Numerical experiments

To assess the effects of water features variations on the planktonic ecosystem and especially the phytoplankton responses, we plan to perform the following simulations.

1. A “realistic” reference simulation from 1991 to 2009 with daily runoff, and monthly mean boundaries conditions for the 10 variables (salinity, temperature and the 8 planktonic variables), with a 2-years spin-up (year 1991) to reach equilibrium.
2. The reference simulation with the 1991’s values of nutrients in the Nervión (first year reproduced from 1991 to 2009) to assess the effects of the nutrients decrease.
3. The reference simulation with a constant transparency to assess the treatment effects on the Nervión River and Estuary ecosystem.

3.3 Results

3.3.1 Statistical results

Based on regressions analyses, prediction of chlorophyll *a* (chl_a) in Nervion Estuary have been tested in function of the different nutrients, light, salinity and temperature. Values of R^2 show that, using data from “Consortio de Agua” or “Red de calidad” for all the periods, no equation explain more than 28.7% of the variation in the chl_a (Table 3.1).

The same multiple regressions analyses have been done only for the spring and summer periods (Table 3.1, lower part). Using the “Consortio de Agua” data during the spring, the regressions with temperature, salinity, light and/or ammonium explain between 37.1 and 41.2% of the chl_a evolution. Using the “Red de calidad” data during the summer, it appears that regressions using the light explain the best the chl_a evolution: between 51.8% and 55.3%. The regressions using nutrients do not explain a major part of the chl_a evolution during spring and summer, except regression taking into account temperature, salinity and ammonium which explains 30.8% of the chl_a evolution during the summer.

As we have seen in the paragraph “4.1 Study Area”, since the 90's, a decreasing temporal trend of ammonia, and an increasing temporal trend of transparency are observed (Fig. 3.1) in the Nervión River and Estuary. Despite these two processes, no statistical change in the chlorophyll *a* biomass has been observed. The decreasing nutrients concentration should reduce the phytoplankton development, but at the same time, an increasing deeper euphotic layer depth should increase it. The impact on the chlorophyll *a* biomass evolution is not visible with simple statistical relations. Indeed, the regressions equation do not show any and evident pattern on the role of light or ammonium on the chlorophyll ^a biomass evolution.

| Data | Equation | R ² | p |
|------------------------------------|--|----------------|--------|
| “Consortio” All data | $Chla = -3.57 + 0.45 * TEMP - 0.035 * salt - 0.003 * NH4$ | 27.4% | 0 |
| | $Chla = -3.19 + 0.39 * TEMP + 0.007 * salt - 0.13 * zeuph$ | 26.9% | 0 |
| | $Chla = -2.74 + 0.39 * TEMP + 0.000008 * salt - 0.004 * NH4 - 0.15 * zeuph$ | 27.2% | 0 |
| “Red de calidad” All data | $Chla = -5.39 + 0.23 * TEMP + 0.14 * salt - 0.024 * Light$ | 28.7% | 0 |
| | $Chla = -4.22 + 0.24 * TEMP + 0.055 * salt + 0.014 * NH4$ | 28.7% | 0 |
| | $Chla = -3.63 + 0.25 * TEMP + 0.04 * salt - 0.0009 * (NO3 + NO2)$ | 26.5% | 0 |
| | $Chla = -3.86 + 0.25 * TEMP + 0.04 * salt + 0.16 * (PO4)$ | 27.5% | 0 |
| “Consortio” Spring data | $Chla = -11.5 + 0.81 * TEMP + 0.068 * salt + 0.0017 * NH4$ | 37.1% | 0.002 |
| | $Chla = -11.36 + 0.81 * TEMP + 0.11 * salt - 0.19 * zeuph$ | 41.2% | 0.0001 |
| | $Chla = -11.07 + 0.81 * TEMP + 0.1 * salt - 0.0018 * NH4 - 0.2 * zeuph$ | 40.7% | 0.0004 |
| “Consortio” Summer data | $Chla = 18.95 + 0.04 * TEMP - 0.49 * salt - 0.017 * NH4$ | 19.7% | 0.0188 |
| | $Chla = 10.87 - 0.16 * TEMP - 0.055 * SALT - 0.4 * zeuph$ | 16% | 0.0556 |
| | $Chla = 14.65 - 0.17 * TEMP - 0.15 * salt - 0.0095 * NH4 - 0.41 * zeuph$ | 19% | 0.0592 |
| “Red de Calidad” Spring data | $Chla = -8.51 + 0.46 * TEMP + 0.13 * SALT - 0.013 * Light$ | 23% | 0.0580 |
| | $Chla = -9.47 + 0.69 * TEMP + 0.037 * SALT - 0.02 * NH4$ | 26.7% | 0.0132 |
| | $Chla = -9.34 + 0.52 * TEMP + 0.10 * salt - 0.004 * (NO3 + NO2)$ | 22.6% | 0.0632 |
| | $Chla = -8.93 + 0.68 * TEMP + 0.03 * salt - 0.52 * PO4$ | 28.5% | 0.0226 |
| | $Chla = -11.88 + 0.86 * TEMP + 0.01 * SALT + 0.01 * LIGHT - 0.014 * NH4 + 0.03 * (NO3 + NO2) - 0.72 * PO4$ | 32.5% | 0.1031 |
| | $Chla = 4.97429 - 0.04 * LIGHT - 0.005 * NH4 - 0.01 * (NO3 + NO2) + 0.02 * PO4$ | 7.02% | 0.7286 |
| “Red de calidad” Summer data | $Chla = -17.97 - 0.009 * TEMP + 0.84 * SALT - 0.12 * LIGHT$ | 52.9% | 0.0004 |
| | $Chla = 5 + 0.14 * TEMP - 0.16 * SALT + 0.07 * NH4$ | 30.8% | 0.0148 |
| | $Chla = -20.22 + 0.07 * TEMP + 0.61 * SALT + 0.09 * (NO3 + NO2)$ | 9.6% | 0.4112 |
| | $Chla = -4.7 + 0.11 * TEMP + 0.14 * SALT + 0.65 * PO4$ | 15.7% | 0.0182 |
| | $Chla = -14.48 + 0.11 * TEMP + 0.63 * SALT - 0.11 * LIGHT + 0.04 * (NO3 + NO2) + 0.03 * NH4 - 0.25 * PO4$ | 55.3% | 0.0054 |
| | $Chla = 8.74 + 0.005 * (NO3 + NO2) + 0.05 * NH4 - 0.3 * PO4 - 0.09 * LIGHT$ | 51.8% | 0.0016 |

Table 3.1: Multiple regressions equations predicting the chlorophyll a biomass (chla) in function of temperature (temp), salinity (salt), ammonium (NH4), nitrate and nitrite (NO3 and

NO₂), and light or euphotic layer depth (zeuph). It exists a statistically significant relationship between the variables at 99% confidence level if the p value is <0,05.

In this context, no efforts on more complex statistical approaches have been done. Indeed, the development of a mechanistic model has appeared more adequate to study the biogeochemical processes and fluxes in the Nervion estuary and then explain and predict the chlorophyll *a* biomass evolution.

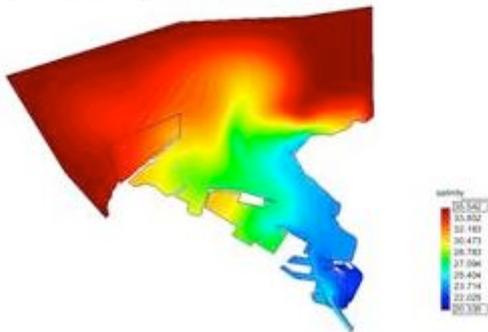
3.3.2 Mechanistic model preliminary results

The mechanistic model is running and first test simulation has been done for the year 1991, with a 2-years spin-up (thus, the simulation represents 3 times the year 1991).

The figure 4.3 shows the simulated salinity in spring and the simulated temperature at the end of the year. The model reproduces the Nervión plume and the extension of low salinity values through the estuary. The figure 4.4 shows the simulated ammonium, nitrate, and small and large phytoplankton, at days 212 (31th of July). The nitrate extension from the Nervión input is larger than the ammonium plume extension. Nitrate from Nervión River extend in the whole harbour until the open sea. The simulation shows also the phytoplankton development in the Nervión plume.

The results are preliminary and the work is in progress. However, the first preliminary results show that the model will allow us to assess the role of the different processes in the time evolution of the chlorophyll *a* biomass.

a) Salinity



b) Temperature

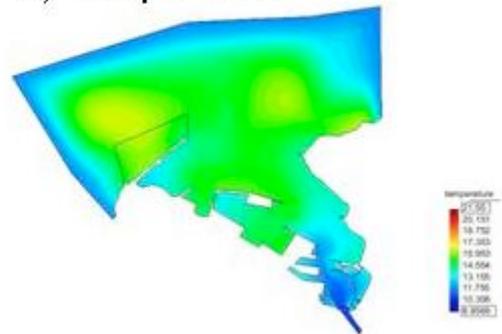
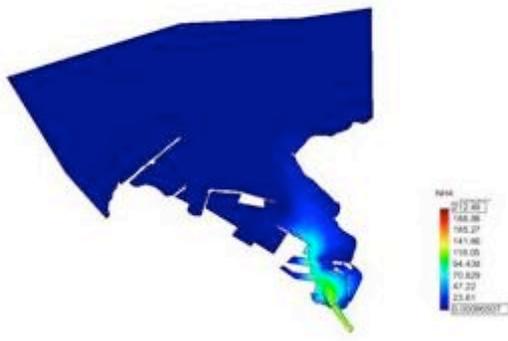
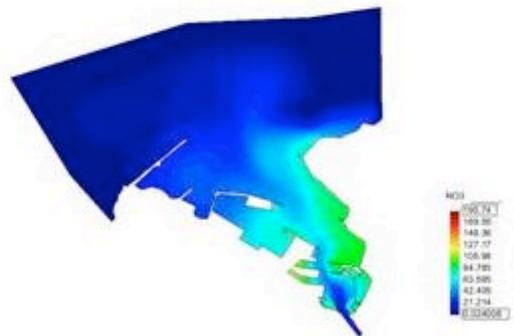


Figure 3.3: Simulated a) salinity at day 147(end of May) and b) temperature after 1 year of run.

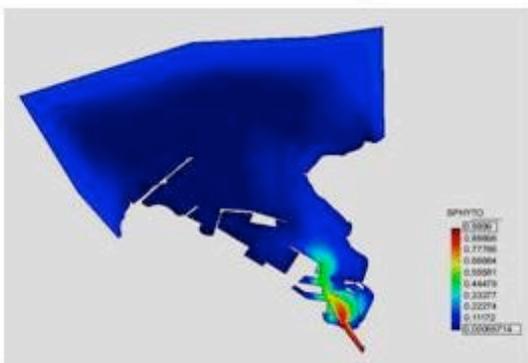
a) Ammonium



b) Nitrate



a) Small phytoplankton



b) Large phytoplankton

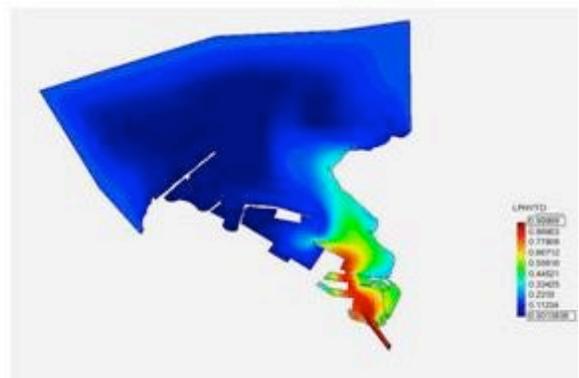


Figure 3.4: Simulated a) ammonium, b) nitrate, c) small phytoplankton and d) large phytoplankton, at day 212 (31th of July). All in $\mu\text{molN l}^{-1}$.

3.4 References

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