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PIRKKO KAUPPILA

Phytoplankton quantity as an indicator of eutrophication
in Finnish coastal waters

Applications within the Water Framework Directive

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Pirkko Kauppila

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List of original publications and author's contribution

(I) Kauppila, P., Hällfors, G., Kangas, P., Kokkonen, P. & Basova, S. 1995. Late summer phytoplankton species composition and biomasses in the eastern Gulf of Finland. *Ophelia* 42: 179-191.

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List of abbreviations

AS	Archipelago Sea
BS	Bothnian Sea
BB	Bothnian Bay
ANOVA	analysis of variance
Chl	chlorophyll <i>a</i>
ChlaD	Chlorophyll <i>a</i> and its degradation products
CIS	Common Implementation Strategy
Cu	copper
DIN	inorganic nitrogen
DI-TDN	diatom-interred total dissolved nitrogen
DIP	inorganic phosphorus
EC	European Commission
EEA	European Environment Agency
EU	European Union
FIMS	Finnish Institute of Marine Research
EQR	Ecological Quality Ratio
FEA	Finnish Environment Administration
GF	Gulf of Finland
HELCOM	Helsinki Commission
ICES	International Council for the Exploration of the Sea
Ls	Southwestern inner archipelago
Lu	Southwestern outer archipelago
Lv	Southwestern middle archipelago
Ms	Quark inner archipelago
Mu	Quark outer archipelago
MWWTP	municipal wastewater treatment plant
N ₂	nitrogen gas
NO ₂ -N	nitrite-nitrogen
NO ₃ -N	nitrate-nitrogen
OECD	Organization of Economic Co-operation and Development
OP	organic phosphorus
OSPAR	commission for the Protection of the Marine Environment of the North-East Atlantic
Ps	Bothnian Bay inner coastal waters
Pu	Bothnian Bay outer coastal waters
r ²	coefficient of determination
Ses	Bothnian Sea inner coastal waters
Seu	Bothnian Sea outer coastal waters
Ss	Gulf of Finland inner archipelago
Su	Gulf of Finland outer archipelago
TN	total nitrogen
TP	total phosphorus
TRIX	trophic index
UNEP	United Nations Environment Programme
WFD	Water Framework Directive
ww	wet weight
Zn	Zink

Phytoplankton as an indicator of eutrophication in coastal marine waters. Applications under the Water Framework Directive

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The tackling of coastal eutrophication requires water protection measures based on status assessments of water quality. The main purpose of this thesis was to evaluate whether it is possible both scientifically and within the terms of the European Union Water Framework Directive (WFD) to assess the status of coastal marine waters reliably by using phytoplankton biomass (ww) and chlorophyll *a* (Chl) as indicators of eutrophication in Finnish coastal waters. Empirical approaches were used to study whether the criteria, established for determining an indicator, are fulfilled.

The first criterion (i) was that an indicator should respond to anthropogenic stresses in a predictable manner and has low variability in its response. Summertime Chl could be predicted accurately by nutrient concentrations, but not from the external annual loads alone, because of the rapid affect of primary production and sedimentation close to the loading sources in summer. The most accurate predictions were achieved in the Archipelago Sea, where total phosphorus (TP) and total nitrogen (TN) alone accounted for 87% and 78% of the variation in Chl, respectively. In river estuaries, the TP mass-balance regression model predicted Chl most accurate when nutrients originated from point-sources, whereas land-use regression models were most accurately in cases when nutrients originated mainly from diffuse sources. The inclusion of morphometry (e.g. mean depth) into nutrient models improved accuracy of the predictions.

The second criterion (ii) was associated with the WFD. It requires that an indicator should have type-specific reference conditions, which are defined as “conditions where the values of the biological quality elements are at high ecological status”. In establishing reference conditions, the empirical approach could only be used in the outer coastal waters types, where historical observations of Secchi depth of the early 1900s are available. Most accurate prediction was achieved in the Quark. However, the average reference values in the outer coastal types are underestimated in sites near the zone of the inner coastal waters. In the inner coastal water types, reference Chl, estimated from present monitoring data, are imprecise - not only because of the less accurate estimation method – but also because the intrinsic characteristics, described for instance by morphometry, vary considerably inside these extensive inner coastal types. As for phytoplankton biomass, the reference values were less accurate than in the case of Chl, because it was possible to estimate reference conditions for biomass only by using the reconstructed Chl values, not the historical Secchi observations. A paleoecological approach was also applied to estimate reference conditions for Chl. In Laajalahti, an urban embayment off Helsinki, strongly loaded by municipal waste waters until 1986, reference conditions prevailed in the mid- and late 1800s. The recovery of the bay from pollution has delayed as a consequence of benthic release of nutrients. Laajalahti will probably not achieve the good quality objectives of the WFD on time.

The third criterion (iii) was associated with coastal management including the resources it has available. Analyses of Chl are cheap and fast to carry out compared to the analyses of phytoplankton biomass and species composition; the fact which has an effect on number of samples to be taken and thereby on the reliability of assessments. However, analyses on phytoplankton biomass and species composition provide more metrics for ecological classification, the metrics which reveal various aspects of eutrophication contrary to what Chl alone does.

Keywords: phytoplankton biomass, chlorophyll *a*, eutrophication, indicators, pollution history, empirical modeling, reference conditions, Water Framework Directive, coastal waters, Baltic Sea

1 Introduction

1.1 Eutrophication and research supporting coastal management

Coastal eutrophication is a major environmental threat worldwide (Vollenwieder 1975), and the Baltic Sea is particularly at risk from this process (Rosenberg *et al.* 1990, Wulff *et al.* 1990). Eutrophication is most frequently described as enrichment of mineral nutrients (primarily nitrogen and phosphorus) to surface waters (Richardson and Jørgensen 1996), and as an increase in the rate of supply of organic carbon to an ecosystem (Nixon 1995). Nixon proposed a classification schemes which describes oligotrophic, mesotrophic, eutrophic and hypertrophic status in marine coastal waters based his classification on phytoplankton primary production. Within the EU, a common legislative approach defines eutrophication as the enrichment of water by nutrients especially compounds of nitrogen and phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms and the quality of the water concerned (Urban wastewater Treatment Directive, C.E.C. 1991).

Eutrophication manifests itself in a number of ways, for example: as an increase in the biomass of phytoplankton (Harding and Perry 1997) and macroalgae (Valiela *et al.* 1997), increased incidence of phytoplankton blooms (Kahru *et al.* 1994, Richardson 1997), anoxia and hypoxia (Matthäus 1990, Rosenberg *et al.* 1990, Kiirikki *et al.* 2006), and fish and benthos kills (Baden *et al.* 1990, Hansson and Rudstam 1990, Norkko and Bonsdorff 1996). In quantifying eutrophication, phytoplankton biomass, measured as chlorophyll *a*, is most often used to measure the trophic status of a body of water. The factors controlling phytoplankton biomass include nutrients, mainly nitrogen (Hecky and Kilham 1988, Kivi *et al.* 1993), phosphorus (Krom *et al.* 1991, Andersson *et al.* 1996) and silica (Turner and Rabalais 1994, Zimba 1998). However, phytoplankton biomass is influenced not only by nutrient concentrations but also by the ratios of nutrients (Prairie and Kalff 1989, Molot and Dillon 1990, Tamminen and Andersen 2007, Andersen *et al.* 2007), the rate of nutrient

turnover (Smith 1984, Levine *et al.* 1997) and other environmental factors such as turbidity (Fisher *et al.* 1988, Irigoien and Castel 1997), hydrography (Cloern 1987), herbivory (Meeuwig *et al.* 1998, Kotta and Møhlenberg 2002) and grazing (e.g. Kuosa and Kivi, 1989; Kuosa, 1991, Uitto 1997, Setälä 2004). Furthermore, phytoplankton biomass may also be associated with hypoxia and anoxia, because following a period of oxygen deficit the release of phosphorus from sediment may raise the phytoplankton biomass in the productive water layer (Richardson and Jørgensen 1996).

Eutrophication of the Baltic Sea is a consequence, on one hand, of real external loading and the intrinsic properties of this brackish seawater basin, and on another hand, due to huge resources of organic material, which, for several decades, have been stored into the bottom sediments of the seabed (Conley *et al.* 2002, Vahtera *et al.* 2007). The properties that makes the Baltic Sea extremely sensitive to eutrophication include shallowness, small water volume, low salinity, restriction of vertical mixing due to semi-permanent stratification, and slow water exchange through the Danish Straits (Table 1, Voipio 1981, HELCOM 1996, 1998). The Baltic Sea receives nutrients fivefold the amount of nutrients from the catchment vis-a-vis its water surface area (HELCOM 1996, 1998). Saltwater inflows from the North Sea renew irregularly oxygen resources in the bottom waters of the naturally hypoxic deep basins of the Baltic Proper, and push old hypoxic and nutrient-rich water towards the Gulf of Finland (Pertilä *et al.* 1996). The hydrodynamics of the Baltic Proper are reflected at least as far as the eastern and middle Gulf of Finland, where it affects on stratification, the levels of nutrients and near-bottom oxygen conditions (Kahru *et al.* 2000).

Today, eutrophication in the Baltic Sea is in a self-sustaining “vicious circle”, due to accelerated benthic release of nutrients, i.e. internal loading, associated with anoxic bottom sediments and huge amounts of organic material (Lehtoranta 2003, Conley *et al.* 1997, 2007, Vahtera *et al.* 2007), which appears to counteract decreases in the external loads of phosphorus at least in the Gulf of Finland (Pitkänen *et al.* 2001). In the Baltic Sea, the pool of inorganic phosphate dissolved in the water correlates positively to the area of bottom covered by hypoxic water, but not to changes in total external

Table 1. Main characteristics of the Baltic Sea, the Gulf of Finland, the Bothnian Sea and the Bothnian Bay. The Archipelago Sea is included into the Bothnian Sea. Sources: HELCOM 1996, 1998.

Characteristics	Baltic Sea	Gulf of Finland	Bothnian Sea	Bothnian Bay
Drainage basin, km ²	1 641 650	421 000	228 000	277 000
– In Finland	300 000	107 000	48 000	146 000
Water area, km ²	422 000	29 600	79 256	36 260
Volume, km ³	21 000	1 100	4 889	1 500
Mean depth, m	55	38	68	43
Maximum depth, m	450	123	230	147
Fresh water flow, km ³ a ⁻¹	540	100-125	88	105
Residence time, years	22	8-10	3	5

phosphorus load (Conley *et al.* 2002). In fact, oxygen conditions in open deep waters of the Baltic Sea appear indirectly to control nutrient dynamics through benthic release of inorganic phosphorus and through denitrification – the loss process of nitrogen converting nitrate-N into nitrogen gas (Raateoja *et al.* 2005, Vahtera *et al.* 2007).

Effective control of coastal marine eutrophication requires variety of monitoring, experimental and modelling research. Mesocosm studies offer an example of experimental approach to examine cause and effect relationships in marine systems. These small-scale studies are used to examine, for instance, nutrient limitation (Seppälä *et al.* 1999, Tamminen and Andersen 2007, Andersen *et al.* 2007) and community responses to nutrient enrichment (Lignell *et al.* 2003, Olsen *et al.* 2006). Extrapolation of mesocosm studies into natural systems is not straightforward as the results of experiments are often affected by artificial boundaries and a lack or limited contact with sediment (Richardson 1996). Information on functioning of marine systems may be used for modelling purposes, in which case it provides an appropriate tool for managers in controlling eutrophication.

Dynamic simulation modelling opens up a potential approach to look at responses of eutrophication to changing input and natural forces in coastal marine waters. The models describe the behaviour of the system taking into account the presumed interrelationships of sub-processes. The predictive capability of these models are based on natural processes such as primary and secondary production, sedimentation, denitrification and nitrogen fixing of algae. Examples on simulation studies in the Baltic Sea are presented by Virtanen *et al.* (1986) and Kiirikki *et al.* (1998, 2001, 2006) to

estimate phytoplankton biomass and concentrations under varying loads of nutrients, the model designed by Savchuk and Wulff (1999, 2007) to simulate regional and large-scale ecosystem responses to nutrient reductions, and the model by Janssen *et al.* (2004) to investigate inter-annual variability of cyanobacterial blooms controlled by wintertime hydrographical conditions. Similar studies in the Atlantic Ocean include the model created by Soetaert *et al.* (1994) to estimate net phytoplankton growth, and the model of Madden's and Kemp's model (1996) for investigating growth responses of submerged vascular plants to eutrophication. Dynamic simulation models are complex and require often a lot of computer capacity. However, despite the argument put forward by Visser and Kamp-Nielsen (1996), computers are cheap once the required programs have been developed. Moreover, the use of numerical models may provide a deeper understanding of marine ecosystems, and also offer improved opportunities for predicting future trends, especially when the responses between different factors are non-linear (Dippner 2006).

Empirical approaches provide an alternative to dynamic simulation models. They are simple, cheap and quick, and require less data. However, in contrast to dynamic simulation models they are not site-specific, and they do not identify cause and effect relationships. That said, they have been successfully used to predict eutrophication in lakes (Vollenweider 1975, OECD 1982), but have less frequently been applied in coastal waters. This is mainly because lakes are clearly defined, with measurable in- and outflows, which contrasts with marine systems (Visser and Kamp-Nielsen 1996) - with the exception of semi-enclosed estuaries. Examples of comparative models in

marine environments are the model of Boynton *et al.* (1982) to predict phytoplankton chlorophyll *a* as a function of nitrogen and phosphorus loads and the model of Monbet (1992) used to predict chlorophyll *a* as a function of inorganic nitrogen in micro- and macro-tidal estuaries. The mass-balance approach of Vollenweider has been applied by, for instance, Jordan *et al.* (1991) and Nixon *et al.* (1995). Moreover, Meeuwig and Peters (1996) demonstrated that regression models based on land-use also accurately predict chlorophyll *a* and are an alternative to the phosphorus-based mass-balance approach applied to North Atlantic estuaries.

Empirical approaches are also useful tools in trend analyses and for building a picture of historical patterns. In marine systems, for instance, it should be remembered that the detection of a change in a trend requires decades of monitoring (Visser and Kamp-Nielsen 1996). Empirical approaches also offer a potential method to hind-cast historical nutrient and trophic status. For example, Smith *et al.* (2003) predicted natural background concentrations of nutrients in streams and rivers, and Dodds *et al.* (2006) determined ecoregional reference conditions for nutrients, Secchi depth and chlorophyll *a* in lakes and reservoirs, and Greve and Krause-Jensen (2005) predicted the depth limits of eelgrass, *Zostera marina*, in pristine conditions in coastal waters.

The last approach, namely paleoecological techniques, provide another tool to trace historical pollution and pristine conditions of waters. In essence, these techniques are built on empirical relationships by way of transfer functions. Nutrient concentrations can be inferred quantitatively from the remains of organisms preserved in the sediment. Thus, diatoms and algal pigments are known to be especially sensitive indicators of trophic conditions (Battarbee 1991, Korhola and Blom 1996, Andren *et al.* 1999, Leavitt and Hodgson 2001). Sediment archives have been successfully used in assessing past anthropogenic impacts and cultural eutrophication in freshwater ecosystems (Bennion *et al.* 1996, Rippey and Andersson 1996), but their application to coastal systems has been limited.

In order to manage coastal eutrophication, it is important to study and conceptualize the cause and effect relationships in coastal marine waters (Cloern 2001). In the final analyses, the two questions that concern coastal managers are these: by how much

must nutrients be reduced in order to restore a water body at least to good ecological status, as targeted by the Water Framework Directive (2000/60/EC), and how much time is needed in order to return that water body to its “original regime” (Sheffer *et al.* 2001) before periodic hypoxia was common (Conley *et al.* 2007). The foregoing examples of monitoring, experimental and modelling researches may offer tools to resolve these kinds of questions.

1.2 Coastal eutrophication and water pollution control policy

Under Finnish water protection policy, coastal eutrophication is considered a priority issue. Present water protection actions concerning the reduction of nutrients coming from point and diffuse sources include Water Protection Targets for 2015 (Nyroos 2006) and Finland’s Programme for the Protection of the Baltic Sea, ratified in 2005. Finland’s current water legislation is mainly based on the revised Environmental Protection Act (86/2000) and Environmental Protection Decree (169/2000). Additionally, protection of national waters is directed by many political actions and programmes, such as Ministerial Programme for Sustainable Development (1998), and the Environmental Programme of Agriculture (2000-2006).

Finland’s national legislation is influenced by acts and directives coming from European Union. Eutrophication, for instance, is addressed by several directives, among them the Water Framework Directive (2000/60/EC), the Urban Wastewater Treatment Directive (91/271/EEC), Nitrate Directive (91/676/EEC) and the EU Marine Strategy directive (2005/0211(COD)). The Water Framework Directive (WFD) of 1995, which established the basic principles of sustainable water policy in the European Union, aims to maintain surface waters at least at the status of good, or to restore them where necessary to that level, by 2015. The Urban Wastewater Treatment Directive deals mainly with waste water discharges from municipal sources, whereas the Nitrate Directive addresses the diffuse nitrogen loading arising from agricultural activity. By taking an ecosystem-based approach, EU Marine Strategy directive integrates all pressures and impacts, with the purpose of achieving good environmental status by 2021.

Finland is also a party to many international conventions and proceedings concerning the protection of marine waters from pollution. Through the Helsinki Convention in 1974, seven coastal states around the Baltic Sea established a commission, namely the Baltic Marine Environment Protection Commission, also known as the Helsinki Commission (HELCOM). The convention came into force in 1980, after ratification by the seven states. The revised Convention, signed by all nine Baltic coastal states and the European Community in 1992, entered into force in 2000. The broad aim of HELCOM is to “protect the marine environment of the Baltic Sea from all sources of pollution, and to restore and safeguard its ecological balance”. The HELCOM Recommendations to the governments of the Contracting Parties are based on unanimous decisions. This Commission works in close cooperation both with other intergovernmental organizations (e.g. the International Council for the Exploration of the Sea, ICES, and the United Nations Environment Programme, UNEP) and with other non-governmental international organizations (e.g. World Wildlife Fund, WWF).

Sustainable Development, defined by the Rio Declaration at the United Nations conference in 1992 and reiterated at the 2002 World Summit, is one of the essential principles in national and international water protection policies. Status assessments, and thematic and indicator reports published, for instance, by HELCOM and the European Environment Agency (EEA), together support sustainable development by producing relevant information for use by decision makers and the public.

Essentials of the Water Framework Directive

The intent of the European Union Directive 2000/60/EC is to control water pollution. The overall aim of the Water Framework Directive (WFD) is to maintain

and improve the ecological quality of surface waters and, ultimately to achieve good environmental quality by controlling the pollution sources that impact them. The assessment of surface water status according to the WFD requires an ecological classification, which is based on four biological quality elements: phytoplankton, zoobenthos, macrophytes and fishes (Table 2). Fishes, however, is excluded from the biological quality elements of coastal waters. Nutrients, near-bottom oxygen conditions and Secchi depth are among the elements supporting the classification. Member states were ordered to incorporate the directive in their national legislations in 2003. By 2015 all surface waters need to achieve Good Ecological Status.

In assessing ecological status, surface waters in each water category (i.e. rivers, lakes, transitional waters, coastal waters and heavily modified water bodies) should be differentiated into various types, which operate as the classification and management units of the Directive. The types need to be characterized by obligatory and optional factors. In the Baltic Sea, the obligatory factors are latitude, longitude, tidal range and salinity, whereas the optional factors are descriptors such as wave exposure, water residence time, mixing conditions and the range of average temperature. The requirement is that the types are ecologically relevant to ensure the reliable establishment of type-specific reference conditions (Anonymous 2003).

Reference conditions are defined as a description of a biological quality element at high status. In other words, a surface water body which exhibit either no or only very minor anthropogenic disturbances resulting from human activities, and which possess the values of the biological quality elements along with the physico-chemical and hydromorphological quality elements that reflect undisturbed conditions (EU directive 2000/60/EC). Moreover, reference conditions should reflect natural variability, both

Table 2. Phytoplankton variables and the hydromorphological and physico-chemical variables given by the Water Framework Directive (WFD).

Phytoplankton variable	Hydromorphological variable	Physico-chemical variable
Phytoplankton composition, abundance and biomass	Depth variation Structure and substrate of coastal bed Structure of intertidal zone Direction of dominant currents Wave exposure	Transparency Thermal conditions Oxygen conditions Salinity Nutrient conditions

spatial and temporal, and attempts should be made to minimize it within a type. Among inland waters lakes in pristine conditions still exist (e.g. Lepistö *et al.* 2006a), but in coastal environments, such kind of waters are rare (HELCOM 2006). The directive gives tools for establishing reference conditions, including the use of historical data (e.g. Krause-Jensen *et al.* 2004), palaeoreconstructions (e.g. Clarke *et al.* 2003), mathematical models (e.g. Schernewski and Neumann, 2004) and empirical models (e.g. Sahlsten and Hansson 2004).

Classification is based on Ecological Quality Ratio (EQR), which is the relationship between the measured value and reference value, the numerical value lying between 0 and 1. Ecological status is divided into five classes (excellent, good, moderate, poor and bad). Member states are allowed to set their own class boundaries, but the values of the EQR set for each status class is supposed to meet the normative definition for that status class given by the Directive (Directive 2000/60/EC Annex V). Harmonization between EU member states of the boundaries of the two upper classes must be carried out by way of intercalibration.

1.3 Assessing trophic status using phytoplankton metrics

The balance of water ecosystems is disturbed by eutrophication, which, in turn, leads to increases in phytoplankton quantity and primary production, changes in phytoplankton community structure, decrease in diversity, and increase in intensity and frequency of harmful algal blooms. Metrics based on phytoplankton quantity and productivity are widely used indicators of eutrophication in the status assessments of surface waters (e.g. HELCOM 2002, EEA 2007, Nixon *et al.* 2003, OSPAR 2003). These metrics include phytoplankton abundance, biomass measured by wet weight or by assimilated quantity of carbon, concentrations of chlorophyll *a*, primary production and productivity. Several of these metrics are used to classify surface waters. For instance, Rodhe (1969) and Nixon (1995) each used organic carbon supply to measure primary production in their respective classification schemes for lakes and marine coastal waters. The trophic classification schemes developed by the Organization of Economic Co-operation and Development (OECD)

in lakes are based on chlorophyll *a*, total P and Secchi depth (Vollenweider and Keres 1982), and these classification schemes appear to be applicable in the Baltic coastal environment, too (Kauppila, unpublished data).

Aggregated indices, built on mathematical equations, are another type of sum parameters. Vollenweider *et al.* (1998) developed a trophic index, TRIX, for the Adriatic Sea, applying Carlsson's (1977) example of aggregating variables in inland waters. The TRIX index indicates both direct productivity such as chlorophyll *a* and oxygen percentage saturation, and nutritional factors such as total N, total P, inorganic N and phosphate-P (Vollenweider *et al.* 1998). However, despite good experiences in coastal water management in the Mediterranean Sea (Giovanardi and Volleinweider 2004), the index cannot be directly applied to other marine coastal waters without prior validation. Hence, for example, according to Vascetta (unpubl. data), this index requires further testing at least in the northern Baltic Sea, because the water is ice-covered in winter, and, additionally, concentrations of nutrients, oxygen and chlorophyll *a* show strong seasonality. In the Adriatic Sea, where the open water period lasts the whole year round, the TRIX index represents the annual averages of the variables.

However, using only sum variables, such as chlorophyll *a* or the TRIX index, to assess trophic status may be misleading, because they give virtually no information on species composition. For example, low concentrations of chlorophyll *a* cannot justifiably be used to describe water quality as good if toxic species are present. Lepistö *et al.* (2005) showed that even low density cyanobacterial blooms containing *Anabaena lemmermannii* P. Richer may be highly toxic. Similarly, Smayda (1997) classifies many toxic dinoflagellates and diatoms as being harmful even at low levels of abundances and biomasses. A shift in species composition may also be an early warning signal of eutrophication, a signal which is not revealed in the measurements of chlorophyll *a*. Hence, the OSPAR commission for the Protection of the Marine Environment of the North-East Atlantic (OSPAR), for instance, aware of the importance of species composition in status assessments, includes not only sum variables, such as maximum and mean concentrations of chlorophyll *a*, but also region or area specific phytoplankton

indicator species, that are categorized as either nuisance species or toxin producing species in its classification schemes (OSPAR 2003). Similarly, in its ecological classification, the WFD requires that member states include not only phytoplankton abundance and biomass but also species composition and blooms (Table 1).

In marine coastal waters, checklists exist on phytoplankton species that also indicate eutrophication (e.g. Hällfors 2004). According to Brettum and Andersen (2005), a species is a good indicator of water quality when that species is found frequently and in great numbers of individuals, and when the highest fraction of the total biovolume lies only within narrow intervals along the scale of trophic level. Carstensen and Heiskanen (2007) found the cyanobacterial *Planktothrix agardhii* (Gomont) Anagnostidis & Komárek to be a potential indicator species in the Baltic Sea based on probability of presence related to increased nutrient levels. We found that a single species in one area of the Baltic Sea may indicate oligotrophy in that area, whereas that same species may indicate meso-/eutrophy in another area, the phenomenon which may reflect different kinds of adaptations and life strategies (Kauppila, unpublished data).

Different taxonomic groups of phytoplankton (i.e. phyla) are known to be sensitive to eutrophication (e.g. Reynolds 1980, 1984, Reynolds *et al.* 2002). Phytoplankton community structure may be described by functional groups, species dominance relationships, size groups, diversity indices, and phytoplankton photosynthetic pigments. An example of a promising functional group is cyanobacteria; for instance, abundance of *Microcystis aeruginosa* Kützinger, *Nodularia spumigena* Mertens and *Planktothrix agardhii* are typically associated with eutrophication (e.g. Niemi 1988, Kahru *et al.* 1994, Johansson and Wallström 2001). Regarding size groups, small phytoplankton cells have been found to dominate under oligotrophic conditions, whereas the abundance of larger phytoplankton cells increases under eutrophic conditions (Kuosa 1990, Irwin *et al.* 2006). Diversity indices in coastal marine waters have also been investigated (e.g. Karydis and Tsirtsis 1996, Danilov and Ekelund 2001, Arhondsitsis *et al.* 2003), but exclusion of rare species from analyses, for instance, due to insufficient taxonomical expertise limits their

wider application for describing phytoplankton community structure. Finally, phytoplankton photosynthetic pigments, such as chlorophyll *a* and β -carotene, provide a chemical approach to analyzing phytoplankton at taxonomic (i.e. phylum) group level (Schlüter *et al.* 2000, Pearl *et al.* 2003). They would be easy to incorporate in water-quality monitoring programmes for assessing the effect of environmental controls on ecosystem structure and function over varying spatial and temporal scales (Pearl *et al.* 2003). The major drawback, however, is that toxic taxa cannot be identified by pigment analyses.

Phytoplankton blooms (algal mass occurrences), besides being a regular phenomenon in many coastal water areas, appear to have increased in frequency, intensity and extent during recent decades (Hallegaeff 1993, Kahru *et al.* 1994, Anderson *et al.* 2002). Phytoplankton blooms may exhibit features, such as exceptionality, toxicity (Smayda 1997), and patchiness (Kononen and Leppänen 1997, Reynolds 2006). Mass occurrences of phytoplankton may occur as either surface accumulations or mixed in the water column. Additionally, harmful algal blooms may occur, not only among cyanobacteria but also in other algal groups, such as dinoflagellates and diatoms (e.g. ICES 2006). Efforts to define phytoplankton blooms include the study by Tett (1987), who set a 100 mg of chlorophyll m^{-3} limit for a bloom event, and Flemming and Kaitala (2006), who presented phytoplankton spring bloom intensity index based on automatically sampled fluorescence and chlorophyll *a* measurements carried out using equipment set up on cargo ships. Carstensen *et al.* (2004) used long-term monitoring data obtained from shallow Baltic Sea estuaries, and based their definition of bloom on the Gaussian distribution of the observations of chlorophyll *a* exhibiting a significant increase in the concentrations of chlorophyll *a*.

In conclusion, several metrics are used to describe phytoplankton quantity or production, but only few of them fulfill the requirements of being a good indicator of eutrophication. According to Dale and Beyeler (2001), an ecological indicator should be straightforward and inexpensive to measure, be sensitive to stresses in the system and respond to those stresses in a predictable manner, be anticipatory before substantial changes in ecosystem integrity

occur, and be integrative. Finally, an ecological indicator should exhibit low variability in its response. These requirements are demanding when considering, for instance, the complex interactions of phytoplankton with other organisms in the water (e.g. Kuosa *et al.* 1997, Kuuppo *et al.* 1998), not to mention the complexity of whole ecosystem with its multiple stressors and sensitivity factors (Cloern 2001, Kononen *et al.* 1999, Kononen 2001). Moreover, phytoplankton species composition is usually highly variable inside large regions, such as the Baltic Sea (e.g. Kononen *et al.* 1999, Kauppila and Lepistö 2001, HELCOM 2002, Gasiūnaitė *et al.* 2005). The fact is that, to date, at best only few phytoplankton indicators have been developed to describe phytoplankton community structure that would be applicable for the WFD purposes. As a result, it is very likely that many member states of the European Union will start assessing the ecological status of their coastal waters using chlorophyll *a* as a proxy variable for phytoplankton biomass in their national classification schemes. One reason for this is that the chemical analyses of chlorophyll *a* are cheaper and faster to carry out than analyses on phytoplankton biomass (ww) and species composition. The second reason is that the prediction of phytoplankton chlorophyll *a* as a function of nutrients has proved successful in the context of lake management (e.g. Dillon and Rigler 1974, Vollenweider 1975, Canfield and Bachmann 1981). Thirdly, along with the implementation of the WFD, quantification of relationships between chlorophyll *a* and nutrients and overall evaluation of the applicability of phytoplankton indicators for management purposes various types of coastal marine waters have recently received greater attention, too.

1.4 Objectives and structure this study

The main purpose of this thesis is to evaluate the applicability of phytoplankton quantity, as measured by phytoplankton biomass (wet weight) and chlorophyll *a*, as an indicator of eutrophication in Finnish coastal waters, mainly from the standpoint of the Water Framework Directive (WFD). I first discuss factors controlling phytoplankton quantity and species composition, and secondly, evaluate

how applicable indicator phytoplankton quantity is to assess trophic status in Finnish coastal waters. The factors discussed in this thesis comprise nutrient loads and concentrations, morphometry and catchment properties, hydrographical and meteorological factors, light conditions and near-bottom oxygen conditions (Fig 1). Loss processes, such as grazing and herbivory, are not embraced by this thesis. Nor are the internal biogeochemical processes of coastal ecosystem (e.g. sedimentation, denitrification, nitrogen fixation).

A two pronged approach was used to establish the applicability of phytoplankton quantity as an indicator of eutrophication in Finnish coastal waters: (a) the applying of scientific criteria, and (b) the applying of the criteria laid out in the WFD. When applying scientific criteria, an indicator should response to disturbances and anthropogenic stresses (e.g. nutrient loading) in a predictable manner, and this response should have low variability. Additionally, the WFD requires establishment of type-specific and well-defined reference conditions.

(a) The fulfillment of the scientific criteria were studied using empirical approaches on the data on water quality in Finnish coastal waters (Table 3). First, the objective was to evaluate reliability of the data regarding annual loads of total N and total P discharging into Finnish coastal waters (paper II). Reliability of the estimates of nutrient loads is important, because the estimates affect predictions of the amounts of phytoplankton biomass. Secondly, the aim is to study the reliability of empirical models predicting chlorophyll *a* as a function of (i) nutrient concentrations, (ii) nutrient loads, specified using land-use regression models and mass-balance equations, and finally (iii) nutrient loads combined with morphometry (e.g. mean depth, water volume, residence time), hydrography (e.g. salinity) and meteorological factors (e.g. wind conditions) (papers III, IV and this summary paper).

Three hypotheses were formulated concerning relationships between descriptors of coastal eutrophication and external controlling factors. The formulating of the first hypothesis was based on the knowledge that agricultural diffuse loading is the main source of nutrients into Finnish coastal waters (e.g. Pitkänen 1994, Vuorenmaa *et al.* 2002) and that land-use integrates a number of anthropogenic factors affecting phytoplankton biomass (Meeuwig

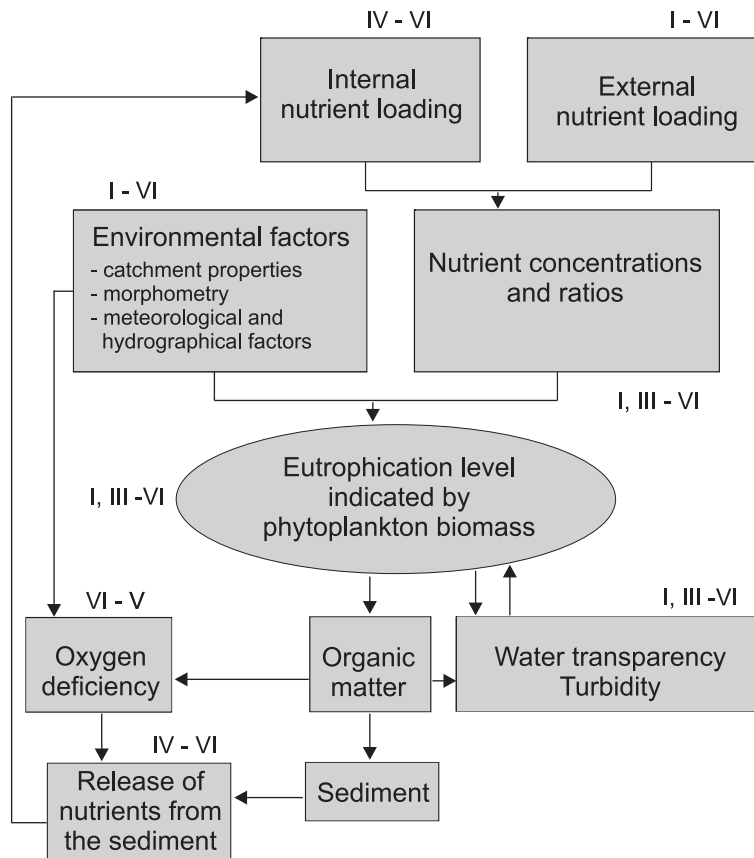


Fig. 1. Simplified picture of factors controlling phytoplankton biomass. Roman numerals (I-VI) refer to the individual papers discussing these factors. The symbol (Su) refer the this summary paper.

Table 3. Descriptions of the data sets used to achieve the objectives of the papers I-VI and this summary paper (Su). Symbols: Chl, chlorophyll *a*; TN, total nitrogen; TP, total phosphorus; rMSE, root mean square error.

Objectives	Location of the study area	Years of sampling	Number of sites	Papers
Evaluation of the importance of nutrients, and hydrographical and meteorological factors on phytoplankton biomass and species composition.	Eastern Gulf of Finland	1990-1992	35	I
Evaluation of reliability (rMSE) in the annual loads of TN and TP.	Finnish rivers	1986-1995	24	II
Evaluation of the TN and TP prediction of Chl alone, and combined with morphometry, hydrography and near-bottom oxygen conditions.	Finnish estuaries Finnish coastal waters	1989-1993 1985-2006	64 (19 estuaries) 763	III, IV Su
Use of empirical approach to estimate reference conditions for Chl and biomass	Finnish coastal waters	1960-2006	763	Su
Use of paleoecological techniques to trace pollution history and reference conditions for TN and Chl.	Laajalahti	Water samples in 1977-2003	1	V
Assessing ecological status of its Laajalahti and Bay recovery from pollution.	Laajalahti	1970-2003	1	VI

1999). Thus, the hypothesis being tested was that a land-use model predict chlorophyll *a* more accurately in Finnish estuaries than do concentrations of total P or a phosphorus-based mass-balance model (paper III). Secondly, it is well-known that phosphorus may be released from the sediment into the water column during conditions of an oxygen deficit at the interface of the sediment and water, which process in turn may generate an increase in phytoplankton biomass in productive surface water layer. In other words, the hypothesis to test was that near-bottom oxygen conditions are linked to chlorophyll *a* in Finnish estuaries (paper IV). The third hypothesis being tested was based on the studies by Wallin and Håkanson (1991) and Meeuwing and Peters (1996), who previously showed that the model combining morphometric (e.g. mean depth) predicts coastal eutrophication better than a model that uses nutrients alone. The validation of these three hypotheses was tested in Finnish coastal waters using, chlorophyll *a*, near-bottom oxygen concentrations and oxygen percentage saturation as predictors (papers III, IV and this summary paper).

(b) From the standpoint of water protection policy, the objective was to study whether phytoplankton biomass and chlorophyll *a* are useful indicators in assessing the ecological status of Finnish coastal waters according to the WFD (Table 3). The WFD directs member states to establish type-specific reference conditions for biological quality variables in order to have a baseline against which the changes can be measured. In this thesis, reference conditions are established only for chlorophyll *a* and phytoplankton biomass (wet weight). Although chlorophyll *a* is basically a chemical variable, it has generally been accepted as a means to describe phytoplankton biomass in the ecological classifications of the WFD (Anonymous 2006a). Set against this backdrop, the aim of this thesis is to establish reference conditions for phytoplankton biomass (ww) and chlorophyll *a* in Finnish coastal waters by employing (i) an empirical approach using Secchi depth (this summary paper) and (ii) a N-based diatom-transfer function using paleoecological techniques (paper V).

A multi-proxy approach was applied in order to trace the history of the pollution of the Laajalahti Bay in order to determine reference conditions, and in turn the recovery of the bay from a polluted

state (Table 3). This was achieved by connecting long-term monitoring results of water quality and loading with sediment data (Weckström *et al.* 2004, paper V), which consisted stratigraphy of diatoms (Weckström *et al.* 2004), sediment geochemistry (Vaalgamaa 2004), stable isotopes (Weckström *et al.* 2004) and sedimentary pigments (Reuss *et al.* 2005). Different classification scenarios were evaluated for the Laajalahti Bay based on concentrations of total nitrogen and chlorophyll *a* (paper VI). The class boundaries were determined using particular percentage deviations from reference values, as suggested previously by Andersen *et al.* (2004) and Sahlsten and Hansson (2004).

2 Study areas

2.1 River catchments

The study area in paper II consisted of 24 river basins, which cover 87% of the Finnish catchment (Fig. 2). The basins vary greatly in their morphometric and land-use characteristics (Ekholm 1993, Pitkänen 1994). The surface areas range between 566 and 51 127 km², and the mean water flow between 6 and 397 m³ s⁻¹. Six of the basins are large, more than 14 000 km², whereas the surface area of the smaller basins is below 5 000 km². The proportion of agricultural land varies between 0.5 to 44%. The basins were defined as agricultural in cases where the percentage of land given over to fields exceeds 10% of the total land area. All other basins are mainly forested. The density of lakes ranges between 0.2 and 17%. In most of the agricultural river basins, the percentage of lake is low, below 5%. In view of this variability, the basins were divided into five classes according to the main characteristics in order to examine the influences of different kinds of the rivers on load estimates: (i) large rivers with low lake percentage, (ii) large rivers with high lake percentage, (iii) small agricultural rivers with low lake percentage, (iv) small agricultural rivers with high lake percentage, and (v) small forested rivers with low lake percentage (paper II).

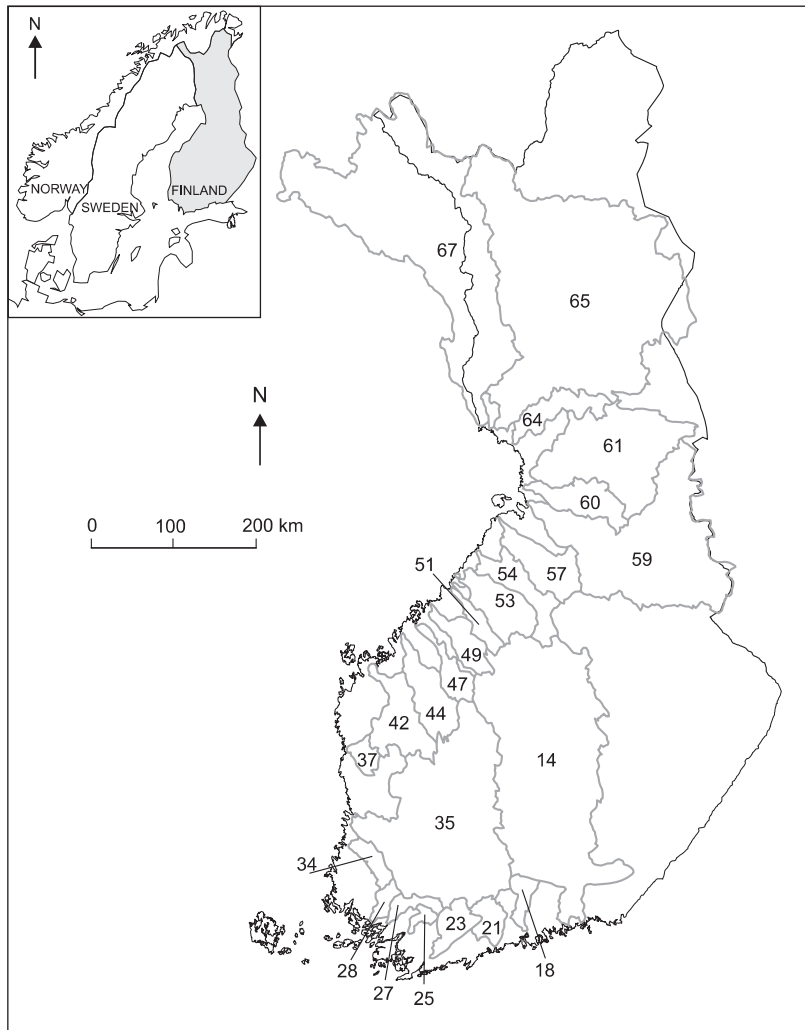


Fig. 2. Rivers discharging from Finland into the Baltic Sea (from paper II).

2.2 Estuarial waters

Estuarial waters in the Baltic are atypical in the sense that they are non-tidal. However, a broad definition of estuaries allows for a continuum of different types of systems (Day *et al.* 1989). In the broad scale, the Gulf of Finland and the Gulf of Bothnia may even be considered as large estuaries because there are strongly influenced by river waters. The Neva estuary in the eastern Gulf of Finland is an open estuary where water is mixed by saline and freshwater. In paper I, the eastern Gulf of Finland was divided into five sub-areas on the basis of

geo-morphological and hydrographical features according to Pitkänen *et al.* (1993) and presented in Fig. 3. Water in the shallow Bay of Neva (sub-area I, depth below 6 m) lying inside the flood-protection barrier is mainly of freshwater origin. Covering the water area of 3 200 km², the river Neva estuary (sub-areas II and III) extends from the Bay as far as the island of Seskar (Pitkänen *et al.* 1993, Table 4). In addition to sharp depth changes (from about 8 to 40 m) the bay is also characterized by strong vertical mixing, indicated by steep vertical and horizontal salinity gradients, and absence of clear halocline. Surface salinity in the estuary ranges typically

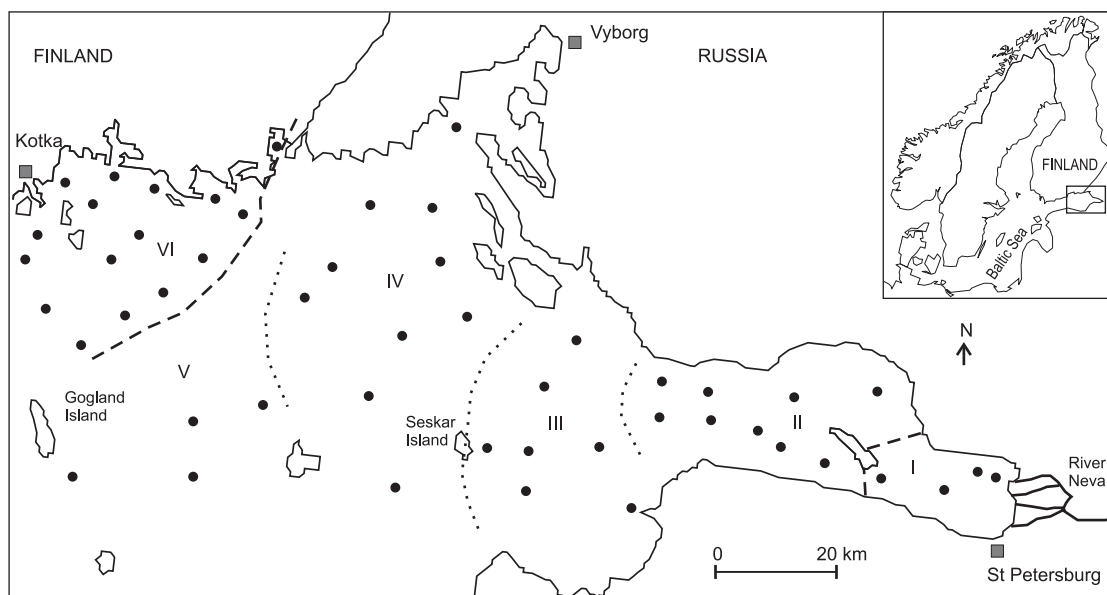


Fig. 3. Study area and the sampling sites studied in the eastern Gulf of Finland in August 1990-1992.

Table 4. Main characteristics in 19 Finnish estuaries, the Laajalahti Bay and the Neva estuary outside the flood-protection barrier. Data given from papers I, III-IV. For the Neva estuary, water area was given by Pitkänen (1991), mean depth, maximum depth and volume calculated in this study and the remainder of the variables given by HELCOM (1998).

Morphometry and Catchment properties	Finnish estuaries		Laaja-lahti Bay	Neva estuary
	Median	Range		Median
Water area, km ²	34	2.0 - 145	5,3	3 200
Mean depth, m	6.4	3.1 - 18	2.4	21
Max. depth, m	20	7.0 - 49	3	36
Volume, 10 ⁶ m ³	219	7.5 - 1 452	12.7	67 200
Mean water flow, m ³ s ⁻¹	10	0.7 - 256	-	2 488 ⁽¹⁾
Residence time, years	0.7	0.01 - 8.18	0.11	0.5
Urban population, %	1.3	0.3 - 6.7	54	2
Agriculture, %	25	9.5 - 43	12	12
Forestry, %	69	54 - 87	34	55 ⁽¹⁾
Watershed, km ²	992	70 - 27 046	25	215 600
TP-load, t a ⁻¹	28	1.8 - 467	0.9	3 526 ⁽²⁾
TN-load, t a ⁻¹	537	49 - 10 770	12	5 8105 ⁽²⁾

⁽¹⁾ River Neva in 1859-1988

⁽²⁾ including 21 rivers in 1995

between 1 and 3 psu, the residence time being about six months. In contrast to the deeper open Gulf of Finland (sub-area IV, general depth between 20 and 60 m) with a salinity ranging between 4 and 5 psu, the Finnish archipelago (sub-area V, general depth between 20 and 40 m) forms its own sub-area, which comprises several semi-enclosed basins and somewhat lower salinities (below 4.5 psu). Trophic status, as measured by concentrations of nutrients

and chlorophyll *a*, is elevated in the easternmost Gulf of Finland compared with the open parts of the Gulf (Pitkänen *et al.* 1993, Pitkänen and Tamminen 1995, Table 5).

Along with extensive estuaries, the broken shoreline of the Baltic Sea characterised by numerous small estuaries and embayments. Very often, the estuaries of Finland are relatively enclosed systems, or winding and fjord-like systems, or island-rich

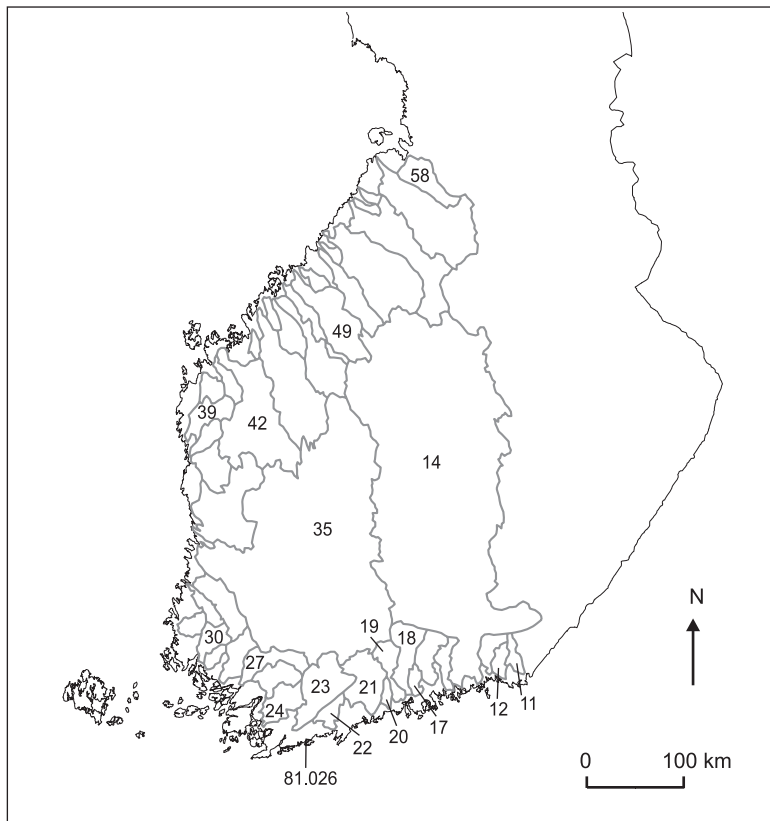


Fig. 4. The location of catchments of Finnish estuaries of this study: 11, Virojoki; 12, Vehkajoki; 14, Kymijoki; 17, Iloanjoki; 18, Porvoonjoki; 19, Mustijoki; 20, Sipoonjoki; 21, Vantaanjoki; 22, Siuntionjoki; 23, Karjaanjoki; 24, Kiskonjoki; 27, Paimionjoki; 30, Laajoki; 35, Kokemäenjoki; 39, Närpiönjoki; 42, Kyrönjoki; 49, Perhonjoki; 58, Temmesjoki; and 81.026, Fagerviken (from paper III and IV).

Table 5. Water quality of 19 Finnish estuaries in June/July to August 1989-1993, the Laajalahti Bay in July to August 1987-2002 and the inner and outer Neva estuary outside the barrier in August 1990-1992. Data given in papers I, III-IV.

Physico-chemical Variables	Finnish estuaries		Laaja-lahti Bay	Neva Estuary
	Median	Range	Mean	Median
Secchi depth (m)	1.6	0.6 - 2.8	0.8	2.3
Salinity (psu)	4.3	1.9 - 6.2	4.9	2.2
Chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$)	7.7	2.6 - 31	24	30
Total P ($\mu\text{g l}^{-1}$)	33	47 - 91	67	30
Total N ($\mu\text{g l}^{-1}$)	466	269 - 1404	672	450
Bottom oxygen (mg l^{-1})	7.8	5.9 - 10	8.5	4.1
Bottom oxygen (%)	80	52 - 97	95.7	-

systems or relatively simple pocket estuaries, (papers III and IV, Fig. 4). Morphometry varies between well-mixed and stratified estuaries. Additionally, the estuaries are generally relatively small and shallow, the water area ranging from 2 to 145 km² and the mean depth from 3 to 18 m (Table 4). They have low salinities (below 6 psu), short residence times and are loaded with nutrients originating both from point

and diffuse sources. Agriculture is the main source of nutrients in those estuaries lying along the southwestern coast of Finland (Vuorenmaa *et al.* 2002). Most of the estuaries are eutrophied compared with their neighbouring coastal water areas; summertime concentrations of chlorophyll *a* are range from 2.9 to 31 $\mu\text{g l}^{-1}$ (Table 5)

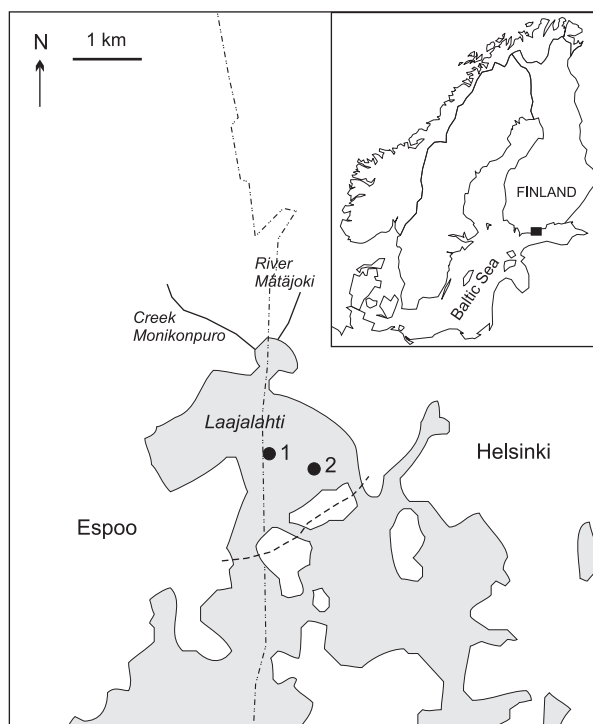


Fig. 5. Location of study area of the Laajalahti Bay and sampling sites for water quality (1) and coring (2). Redrawn from paper V.

The Laajalahti Bay, west of Helsinki City is an urban estuary having a long eutrophication history (Fig. 5, papers V and VI). It is semi-enclosed and small: its surface area is 5.3 km² and its mean depth of 2.4 m (Table 4, Lappalainen and Pesonen 2000). The bay receives fresh water from two brooks, and is connected to the Gulf of Finland first by two narrow straits and subsequently by two sounds, both of which restrict horizontal water exchange. The theoretical residence time is 0.105 years and the average salinity (4.6 psu) of the bay is close to that of the open archipelago. The water is turbid with Secchi depth varying between 0.5 and 1 m. There is no clear density stratification in the bay, therefore oxygen conditions near the bottom are usually good both during summer and the period of ice cover between December and April. In the 1960s, the Laajalahti Bay was one of the most eutrophied areas off Helsinki (Varmo *et al.* 1988, Lappalainen and Pesonen 2000). However, the bay has recovered from severe pollution after the closing of the municipal treatment plant in 1987. The present-day water quality of the bay is presented in Table 5.

2.3 Finnish coastal waters

Coastal waters around Finland can roughly be divided into the Gulf of Finland, the Archipelago Sea and the Gulf of Bothnia (Fig. 2 insert). The last of these three consists of two distinct basins, the Bothnian Sea and the Bothnian Bay, which are separated from each other by a sill lying 20 m deep and the shallow archipelago of the Quark. Southwards, the Archipelago Sea and the Åland Sea partly isolate it from the northern Gulf. Thus, the deep water of the Baltic Proper is connected to the Bothnian Sea only by a narrow channel between Sweden and Åland (Fonselius 1996).

The combined drainage area of the Gulf of Finland, the Archipelago Sea and the Gulf of Bothnia is 897 000 km², of which the Finnish catchment (301 300 km²) accounts for 34% (Table 1.1, HELCOM 1996). The average depths varies from 23 m in the Archipelago Sea to 68 m in the Bothnian Sea. The morphometry of the Finnish coast is characterised by a broken shoreline and a multitude of islands. The rectilinear length of the

Finnish mainland coast is 1 100 km, but if all the islands (ca. 73 000 islands) are included, the actual length rises to 39 000 km (Granö and Roto 1986). A mosaic of islands and skerries unique in the whole world dominates the northern coasts of the Baltic Sea.

Meteorological factors determine many physical and hydrographical conditions of the northern Baltic Sea. The seasonal variation of water temperature is considerable (0 to 20 °C), and the water is ice-covered more than 90 days during the winter (Leppäranta *et al.* 1988). The tide in the northern Baltic Sea is insignificant, only 1-2 cm. The irregular fluctuations of the water level are responses to changes in barometric conditions and in the direction and force of the wind. During periods of westerly winds and low pressure conditions, the inflows from the Danish Sounds may bring more saline and nutrient-rich water into the Baltic Sea, which pushes the “old” deep waters towards the Gulf of Finland (e.g. Matthäus 1982, 1990). Locally, westerly and northerly winds may cause an up-welling of cooler water below the thermocline. This, in turn, raises the salinity and nutrient concentrations in the surface water.

The hydrographical features of the Gulf of Finland and the Gulf of Bothnia differ in many respects from each other. The former is a direct extension of the Baltic Proper; thus there is no threshold hampering the flow of deep, saline water into the Gulf of Finland. This leads to a strong salinity stratification at the depth of 60-70 m in the western part, which is the area where the gulf reaches its maximum depth of 120 m (Perttilä *et al.* 1996). The halocline becomes weaker towards the east due to a slow vertical mixing, but strengthens again in the Neva estuary and rises to a depth of 10-30 m as a result of the large fresh water input from the River Neva (Pitkänen and Tamminen 1995). Additionally, vertical mixing is prevented by the thermocline that occurs in the summer at a depth of 10 to 30 m, which further promotes conditions for oxygen deficiency in near-bottom waters.

The Gulf of Bothnia is sheltered from the bulk of the deep waters by a ridge formed by several underwater thresholds, and also by the shallowness of the Archipelago Sea. Only small volumes of water from the saline deep water below the halocline enter the Gulf of Bothnia. This in the combination with the

effects of large discharges from the northern rivers sustain the low salinities of the Gulf: the salinity ranges from 1 to 4 psu in the Bothnian Bay and from 4 to 6 psu in the Bothnian Sea (Kullenberg 1981). In the spring and autumn, the weak stratification and the seasonal turnover of the whole water body extends down to the bottom promoting high oxygen concentrations in the near-bottom water layers. In contrast to the Gulf of Finland, anoxic conditions have never been observed in the open Gulf of Bothnia (Wikner 1996).

The hydrographical features also lead to differences in the sensitivity of the two northern Gulfs of the Baltic Sea to eutrophication. The Gulf of Finland is affected by considerable nutrient load from land areas, mainly originating from the River Neva and St. Petersburg region (Pitkänen *et al.* 1993). In the Gulf of Finland the nutrient load per unit water area is 2-3 times to the average of the whole Baltic Sea (Pitkänen *et al.* 2001a). In the open Gulf of Finland nutrient supplies become available for phytoplankton through upwelling and strong mixing events in late autumn and winter. In general, water mixing does not regularly and completely reach the sea bottom, except in the eastern Gulf of Finland, where, due to the lack of the permanent halocline, nutrient reserves on the sea bed relatively easily reach the productive surface layer (Pitkänen and Tamminen 1995). In the eastern Gulf of Finland, benthic release of nutrients was accelerated in the mid 1990s by the strengthened stratification and incomplete wintertime mixing (Pitkänen *et al.* 2001b). In the Gulf of Bothnia, the amount of phytoplankton is noticeable smaller than in the Gulf of Finland due to the smaller external nutrient loading and the lack of areas receiving substantial internal loading. In the Archipelago Sea, the general trophic status in summer lies between that in the Gulf of Finland and the Gulf of Bothnia.

2.4 Finnish coastal typology

The national typology of Finnish coastal waters (Fig. 6) is based on the proposal by Kangas *et al.* (2003), the ecological relevance of which was tested by using zoobenthos assemblages (Perus *et al.* 2004). In characterization of coastal waters, the main five types were first differentiated from each other by location (longitude and latitude), salinity

and the duration of ice cover. The last-mentioned factor, derived from the descriptor of the range of average water temperature (Anonymous 2003), is ecologically significant in the northern Baltic Sea. The duration of ice cover underlines the unique nature of the Bothnian Bay, where ice cover lasts more than 150 days, in contrast, to the easternmost Gulf of Finland, which has low salinity (below 3 psu) but where the duration of ice cover is shorter, i.e. between 90 and 150 days (Kullenberg 1981, Leppäranta *et al.* 1988). These main coastal types differ from the division used by HELCOM Commission, especially in the southern coastal water areas, where the longitudinal boundary in the Gulf of Finland was set off Porkkala instead of off Hanko peninsula (Perus *et al.* 2004).

The further division into inner and outer coastal waters was mainly based on water residence time and wave exposure. Consequently, the inner coastal waters differed from the outer types in that they have longer residence time - weeks or months - and the fact that they are more sheltered against the wind than the outer coastal waters, where renewal of waters lasts only days. Moreover, mixing conditions also supported the division into the inner and outer coastal types, although the complex bottom topography, especially in the Gulf of Finland and Archipelago Sea, made it in many cases difficult to draw a clear the line between the well-mixed waters near the coast and the seasonally stratified offshore waters. More detailed descriptions of the physical characteristics of Finnish coastal types are found in Kangas *et al.* (2003) and Perus *et al.* (2004). A type-

Table 6. Median, minimum and maximum values of physico-chemical variables in the coastal water types of Finland in July to August 1990-2006. The chemical variables were sampled in surface waters and oxygen %-saturation near the bottom. Symbols: GF, Gulf of Finland; AS, Archipelago Sea; BS, Bothnian Sea; BB, Bothnian Bay. Locations and abbreviations of the coastal water types given in Fig. 6.

Marine area / Coastal water type		Depth of the sites (m)	Salinity (psu)	Secchi depth (m)	Chlorophyll ($\mu\text{g l}^{-1}$)	TN ($\mu\text{g l}^{-1}$)	TP ($\mu\text{g l}^{-1}$)	Near-bottom oxygen-% saturation
Median; range								
GF	Ss	16; 3-40	4.3; 0.8-5.5	2.2; 0.9-4.8	7.4; 1.6-48	420; 210-820	30; 14-67	62; 5-102
	Su	42; 26-70	4.2; 2.8-5.7	3.4; 1.3-5.6	5.0; 1.5-16	370; 220-540	22; 12-75	57; 0-86
AS	Ls	32; 9-119	5.7; 1.5-6.1	2.3; 0.8-5.1	5.6; 2.1-32	380; 200-600	26; 12-63	53; 0-99
	Lv	40; 28-93	5.9; 5.2-6.3	3.0; 1.4-6.5	3.9; 1.4-16	342; 220-610	19; 9-31	66; 25-89
	Lu	38; 15-84	5.9; 4.9-6.4	3.8; 1.6-7.6	3.9; 1.1-19	330; 180-520	22; 9-55	66; 0-90
BS	Ses	12; 4-35	5.4; 4.7-6.1	2.5; 0.8-4.9	2.7; 1.0-19	308; 220-685	20; 3-36	86; 51-102
	Seu	17; 7-42	5.4; 4.8-6.0	3.7; 1.3-7.5	2.3; 0.2-9.2	260; 175-382	13; 7-28	81; 55-108
Q	Ms	4; 0.6-13	4.3; 2-5.5	1.8; 0.5-4.7	5.3; 0.1-42	403; 220-3100	16; 7-96	97; 45-138
	Mu	20; 10-65	3.6; 3.1-5.7	4.0; 2.1-9.0	2.0; 0.1-3.7	368; 200-411	9; 5-18	84; 21-101
BB	Ps	7; 1.3-16	3.1; 0.3-3.6	2.1; 0.4-6.0	4.9; 0.1-21	335; 235-2132	14; 2-76	93; 51-125
	Pu	16; 0.7-42	2.9; 0.1-4.0	3.2; 0.6-8.0	2.7; 0.5-11	286; 81-690	10; 2-92	91; 42-110

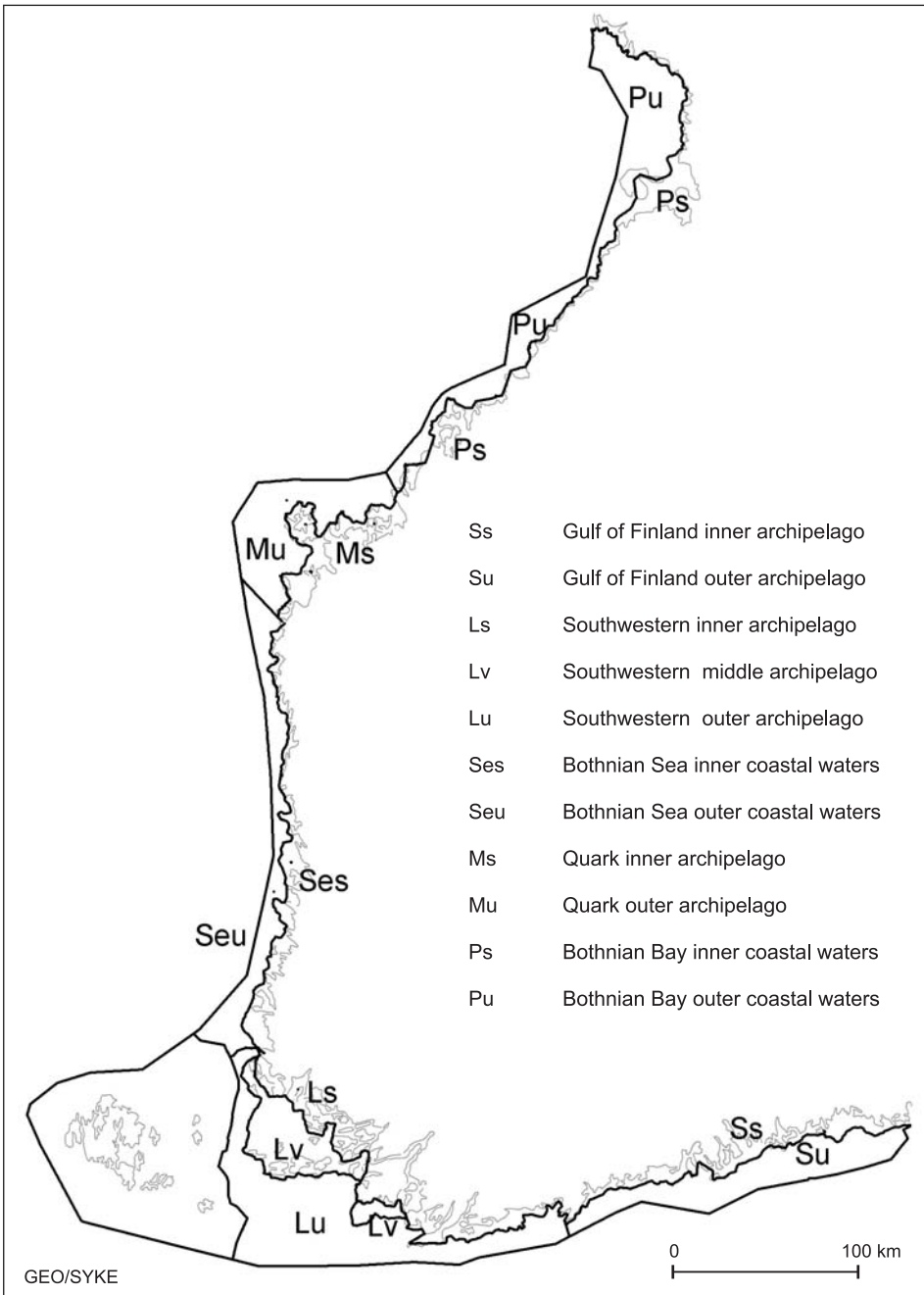


Fig. 6. National coastal water types of Finland, defined according to the Water Framework Directive.

specific description of water quality is presented in Table 6.

3 Materials and methods

3.1 Data sets of the coastal water monitoring

Six separate data sets were compiled for this thesis, with data originating from five surface water areas: (i) 24 Finnish rivers (paper II), (ii) Eastern Gulf of Finland (paper I), (iii) 19 Finnish estuaries (papers III, IV), (iv) the Laajalahti Bay (papers V, VI),

and Finnish coastal waters (this summary paper). Altogether 35 sampling sites were visited in the easternmost Gulf of Finland and the Neva estuary in August 1990-1992 (Fig 3). The data derived from 19 estuaries (Fig. 4) consisted of total 72 sampling stations visited between June and August in 1989-1993, the number of stations in each estuary varying from 1 to 17. City of Helsinki Environment Centre has one water quality monitoring station in the Laajalahti Bay; there samples were taken in 1966-2001 (Fig. 5). In Finnish coastal waters, altogether 763 sampling stations comprised the coastal monitoring network, which were sampled in July to August 1985-2007 (Fig. 7). The monitoring data sets

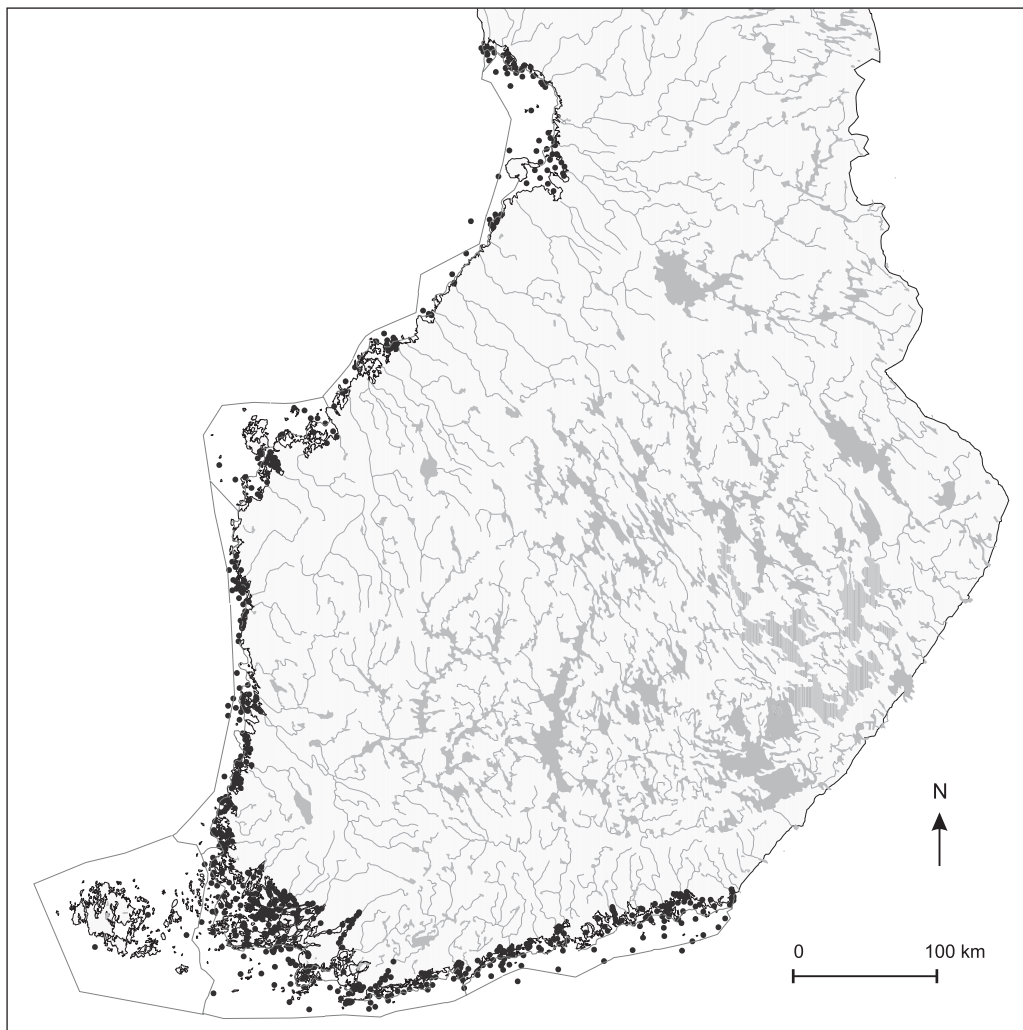


Fig. 7. Sampling stations of Finnish coastal waters.

Table 7. Variables and analyses used in water and sediment. Sediment geochemistry has mostly been described in more detailed in separate publications of the other authors of Paper V (See Reuss 2005, Weckström 2005, Vaalgamaa 2007). Symbols: Su, summary paper.

Materials	Variable	Methods	Used in papers
Water	Phytoplankton biomass and species composition	Utermöhl 1958, Edler 1979	I
Water	Chl	Spectrometrically and colorimetrically according to Lorenzen (1967)	I, III-VI, Su
Water	Secchi depth		I, III-V, Su
Water	TN, NH ₄ -N, NO ₃ -N, TP, PO ₄ -P	Spectrometrically and colorimetrically according to Koroleff (1983) and Murphy and Riley (1962)	I-VI, Su
Water	Salinity	Conductimetrically	I, III-VI, Su
Water	Oxygen	Winkler method, titrimetrically according to Grasshoff <i>et al.</i> (1983)	IV-V, Su
Sediment	TN, TC	Analysed by Leco-analyser	V
Sediment	Bsi	Modification of the method of DeMaster (1981)	V
Sediment	TP	Ammonium molybdate method with ascorbic acid reduction (SFS-EN 1997, 1189)	V
Sediment	IP	Digesting in HCl (Aspila <i>et al.</i> 1976), analyzing (SFS 1997, 1189)	V
Sediment	Cu, Zn	Digesting by autoclaving (SFS 1980, 3044), spectrophotometer	V
Sediment	¹⁵ N	Continuous-flow isotope mass spectrometry (CF-IRMS)	V

compiled for this thesis comprise information on phytoplankton biomass (measured as wet weight and concentrations of chlorophyll *a*), water chemistry, coastal morphometry, land-use, total nutrient loads, and mean water flows of rivers (Table 7). All data, originating both from Finnish national monitoring programmes and local water quality monitoring surveys, are stored in the database of Finnish Environment Administration (FEA). Summarized descriptions of the coastal monitoring programmes are presented in the study by Kauppila and Bäck (2001).

In estuaries, average values were estimated by calculating monthly, seasonal and annual means for each station and then averaging the station-specific annual means into estuarial-specific mean values for the period 1989 to 1993 (equation in paper III). In the Laajalahti Bay, the annual average values were calculated for winter (January to March) and for summer (July to August). In Finnish coastal waters, average values were estimated for the summer period 1985-2006.

Variables characterising coastal morphometry consist of catchment size, surface water area, mean depth, volume, residence time, openness and fetch. The land-use data comprise the percentage values of the watershed that is urban, agricultural or forested, respectively. Catchment size and land-used data from the early 1990s were derived from the databases

of Finnish Environment Administration. Estuarial surface area and mean depth were calculated from 1 : 50 000 bathymetric charts (Finnish Institute of navigation 1996-1998). Mean depth was estimated using a grid technique whereby the depth under each square of the grid covering the estuary was recorded and the average of all of these depth values was calculated. The theoretical residence time (years) was calculated using Bowden's (1980) saltwater fraction method (Table 8). Fetch is the measure of the longest diameter of the water area in the direction of the prevailing wind. Prevailing wind directions, calculated by the Finnish Meteorological Institute (1990-1995), were based on measurements at 11 meteorological stations close to the estuaries.

3.2 Load calculations

Annual loads of nutrients for 24 rivers were calculated using data on concentration of total N, total P and suspended solids, each of which were sampled 10 to 12 times per year, together with the daily measured water flow data (paper II, Fig. 2). The annual loads of each substances were calculated by using six methods: averaging, linear interpolation, periodic, correlation, partially flow-stratified and flow-stratified methods, all of which are described in detail in paper II (Table 8). The averaging method is generally used to estimate Finland's national figures

Table 8. Numerical and statistical techniques used in papers I-VI and this summary (Su). Statistical techniques for paleoecological data are described, in addition to paper V, in more detailed in separate articles of the other authors of paper V (see Reuss 2005, Weckstöm 2005, Vaalgamaa 2007).

	Numerical and statistical methods	Used in papers
<i>(1) Water quality data</i>		
Morphometry (e.g. estuarial water area, mean depth) and land-use	1:50 000 bathymetric charts GIS (ESRI-s ArchView 3.2 and spatial analyses)	I, II, III, IV, V, VI
Residence time	Knudsen's equation (Bowden 1980)	III, IV
Fetch, the longest diameter of the water area in the direction of the prevailing wind	1:50 000 bathymetric charts, prevailing wind direction from the Finnish Meteorological Institute	IV
Load estimates for TP and TN	Averaging method Linear interpolation method Periodic method Correlation method Partially flow-stratified method Flow-stratified method	II, III, IV II II II II II
Significance of monotonic trends	Kendall Tau B	V
Common variation	Variance analyses, ANOVA	Su
Magnitude of significant trends	Simple linear regression analysis	IV
Relationships between different water quality variables	Spearman's correlation Simple linear regression analyses Multiple regression analyses Two-step mass-balance regression analysis	I, II II, IV, V, Su III, IV, Su III
Accuracy and precision, and general reliability	Monte Carlo method MSR, mean square residual VSR, variance of the squared residual rMSE, root mean squared error	II III III II, III, Su
Model validation	Adjusted R-square PRESS statistics (Stevens 1996)	IV IV
<i>(2) Paleoecological data</i>		
Species turnover (gradient length) of the calibration data set	DCA, detrended correspondence analysis	V
Diatom-environment relationships	CCA, partial CCA	V
Species richness	Rarefaction analyses	V
Diatom-inferred TDN reconstructions	WA-PLS, weighted-averaging partial least squares regression and calibration	V
Statistical zonation of stratigraphical data	Optimal sum of squares partitioning and broken-stick model	V
Overall changes in diatom community structure of core data	CA, DCA, detrended correspondence analysis	V
Trends in biological and chemical variables of core data	PCA, principal component analysis	V

for the annual riverine loads of nutrients (see e.g. HELCOM 2004).

Accuracy and precision of the load estimates derived from the five calculation methods were evaluated by applying the Monte Carlo procedure, previously used for example by Richards and Hollaway (1987) and Rekolainen *et al.* (1991). The daily data sets were constructed by linear interpolation of the weekly monitored water quality data collected from the River Paimionjoki in 1985 and during the period 1988 to 1991 in order to incorporate the variation of daily water flow into the

calculation. By using the flow-dependent strategy of 12 annual samples (Paper II), one hundred replicate data sets were randomly sampled from the daily data set. The replicate data sets for each method were used to calculate the accuracy and precision of the load estimates (Table 8). The root mean squared error (rMSE, Bickel and Doksum 1977) was calculated to describe the general reliability of the methods under study. Details of the Monte Carlo procedure and associated equations are given in Paper II.

3.3 Biological and chemical analyses

Composite samples of phytoplankton (surface to twice the Secchi depth) were taken with a Ruttner-sampler and preserved with acid Lugol's solution (Willén 1962). Cells were counted with a Zeiss IM35 microscopy employing Utermöhl's technique (1958). Cell numbers were converted to biomass (ww) using the volumes of the phytoplankton database of the Finnish Environment Administration (FEA), most of which have been calculated according to Edler (1979). Chlorophyll *a* was analysed from the composite sample of phytoplankton was analysed according to Lorenzen (1967). The chlorophyll samples were extracted with acetone from 1980 to 94, and ethanol (Ethyl alcohol) thereafter.

Both nitrogen and phosphorus amounts were determined from unfiltered samples following the Finnish standard methods (Table 7). Total nitrogen samples were first oxidized with potassium peroxodisulphate, then reduced to NO_3 using CD amalgam and, finally, determined spectrometrically and colorimetrically. The analyses of nitrate-N and nitrite-N followed the same procedure, and ammonium-N samples were determined by the indophenol blue method. The concentrations of nitrate-N and nitrite-N were reported largely as a sum ($\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$), which usually deviates only slightly from nitrate-N. Total phosphorus and phosphate-P were determined spectrophotometrically and colorimetrically by using the ammonium molybdate method, described by Murphy and Riley (1962). The analyses of total P samples were initiated by digestion with potassium peroxodisulphate. Suspended solids were filtered through a glass fibre filter ($< 70 \text{ g m}^{-2}$, GF/C) and analysed gravimetrically. Oxygen concentration and oxygen %-saturation were determined by the Winkler method (Grasshoff *et al.* 1983).

3.4 Statistical analyses

Regression analyses:

Multiple regression techniques were used to explain variation in three descriptors of eutrophication by using variation in ambient water chemistry, the descriptors being (i) phytoplankton quantity (chlorophyll *a*, and biomass measured as wet

weight), and (ii) near-bottom oxygen conditions (oxygen concentration and %-saturation), and (iii) Secchi depth. In addition, land-use and morphometry were used as explanatory variables in all three cases as well (Tables 2 and 8). To avoid multicollinearity, correlated explanatory variables were not included in final regression models. All the variables were \log_{10} -transformed to stabilize variance (Zar 1999). Land-use variables that were presented as percentages of the catchment area were transformed into $\log_{10}(x + 1)$ in order to avoid problems arising from the presence of zero values in some of the land-use categories. The set of preliminary models for regression was identified using the R-square option in the SAS procedure AProc Reg@. The Abest@ model was chosen on the basis of the highest coefficient of determination (r^2) and the smallest residual mean square error (rMSE). The mass-balance approach to predicting chlorophyll *a* involves two steps. First, total P is estimated as a function of the load of total P, sedimentation and flussing (Vollenwider 1975) using Canfield and Bachmann's (1981) mass-balance equation (paper III). The estimated concentrations of total P are then applied to predict chlorophyll *a* using an empirical regression analysis.

Establishment of reference conditions:

Reference conditions were established for Secchi depth, chlorophyll *a* and biomass by applying three different approaches: (i) the use of historical data, (ii) empirical modelling and (iii) the use of frequency distribution data collected since the 1960s (Table 9). The historical observations of (i) Secchi depth (altogether 19 sites and 80 samples) originated from the summer research cruises of the Finnish Institute of Marine Research (FIMR) in the northern Baltic Sea between 1925 and 1934. The observation sites were mainly located in the open sea and in the outer coastal waters of the northern Baltic Sea. Comparability between the past and present Secchi values were ensured using a correction factor given by Launiainen *et al.* (1989).

The number of the past Secchi observations varied a lot in different coastal water areas around Finland: least observations were found in the outer Quark (5) and most in the Gulf of Finland (25). Consequently, the accuracy of the estimated averages varies, larger sample sizes resulting in better accuracy. This accuracy, i.e. the variation

of the estimated mean across samples represented the standard error of the mean. It is calculated by dividing the sample variance by the square root of the number of observations. The problem arising from the small number of observations in certain coastal water areas was solved by assuming that the amount of variation within the outer coastal water areas is similar. This common variation can be estimated as the error variation in Analysis of variance (ANOVA, Table 8). Confidence limits of the mean Secchi depth were calculated using this common estimate. The same approaches were applied to chlorophyll *a* and phytoplankton biomass.

The empirical approach (ii) was applied in the calculation of reference conditions for (a) phytoplankton chlorophyll *a* and (b) biomass (ww) in the outer coastal waters (Table 9). Reference values for (a) mid-summer phytoplankton chlorophyll *a* were calculated by applying the historical observations of Secchi depth from the early 20th century. The past values of Secchi depth were used as an input to the linear regression equations, whose parameters had been estimated using the current data of chlorophyll *a* and Secchi depth. However, when phytoplankton biomass (ww) was regressed on Secchi depth, the coefficients of determination (r^2) remained very low. Hence, (b) the average reference values for phytoplankton biomass in the outer coastal waters had to be reconstructed in another way, viz the reference values estimated for chlorophyll *a* and the linear regression equations predicting phytoplankton biomass as a function of chlorophyll *a*. The data of the model came from 16

sites (altogether 80 samples) monitored in Finnish coastal waters in July to August in 1995-2005. The similar procedure as in the case of Secchi depth was used when estimating the upper confidence limits of the means for chlorophyll *a* and phytoplankton biomass.

In the inner coastal water types (see Fig. 6), reference conditions for phytoplankton could not be determined empirically, because historical Secchi observations were not available. However, the fact that the smallest values of the present Secchi data in Finnish coastal waters are covered by the scatter of the historical observations in the Baltic Sea (Table 6, Launiainen *et al.* 1989) may justify a comparison between the historical observations of the early 1900s and the percentiles of the current frequency distribution of data gathered since the 1960s. As a result of the above, (iii) the mean Secchi values of the past data were compared against different percentiles derived from the present frequency distribution data (Table 9). The purpose was to find a percentile in the present data that was equivalent to the type-specific mean values of the past observations. Consequently, the average reference values for Secchi depth in the inner coastal waters were estimated using the ratio between the mean Secchi values of the past data and the given percentile value of the present data calculated for the inner coastal waters. Reference conditions for (iii) phytoplankton chlorophyll *a* and biomass in the inner coastal waters were estimated in a similar way to the reference Secchi values of the inner coastal waters.

Table 9. Methods used to establish reference conditions for Secchi depth, chlorophyll *a* (Chl) and phytoplankton biomass (ww) in the inner and outer coastal waters of Finland.

	Establishment of reference conditions		
	Method		
	Use of historical data	Empirical relationship	Frequency distribution
Secchi depth	Outer coastal waters in the period 1914-1934		Upper 95% percentile in inner coastal waters
Chl		between Chl and Secchi depth in outer coastal waters	Lower 5% percentile in inner coastal waters
Biomass		between biomass and Chl in outer coastal waters	Lower 5% percentile in inner coastal waters

3.5 Paleolimnological datasets and techniques

Reconstruction of reference conditions

Paleolimnological techniques were used to reconstruct reference conditions for total N and chlorophyll *a* and to trace different pollution phases in the Laajalahti Bay (paper V). In this procedure, the results of a transfer function were applied to the results of the sediment core of Laajalahti to estimate long-term trends in total N concentrations. The transfer-function has been developed from independent training set composed of diatom relative abundance data and associated variables in water-chemistry (Weckström *et al.* 2004). The training set, in turn, is based on 49 sites sampled between 1996 and 1998 from small, shallow and sheltered embayments in the Gulf of Finland (Weckström *et al.* 2004). The original 49-site calibration data set was used to infer past annual concentrations of chlorophyll *a* (Weckström *et al.* 2004, paper V). A detailed description of the modern calibration data set in terms of water sampling, analysing techniques and water quality parameters is given in Weckström *et al.* (2002). Methods used to analyse geochemistry are given in Table 7. Statistical and numerical methods are presented in Table 8.

4 Control of phytoplankton biomass

4.1 Physical factors

4.1.1 Morphometry

Coastal morphometry, i.e. such characteristics as mean depth, water surface area, volume, residence time and fetch, affects the flow of energy and nutrients through the coastal water ecosystem and thereby determines the sensitivity of an coastal water area to eutrophication (Wallin and Håkanson 1991). Among Finnish estuaries, models combining mean depth and the percentage of watershed under forestry improved the predictions of the amounts of chlorophyll *a* (paper III), which supports the hypothesis of "the loading and sensitivity effects" by Wallin and Håkanson (1991). In contrast to their hypothesis, another descriptor of eutrophication,

namely near-bottom oxygen conditions, was predicted either as a function of mean depth or land-use, but no model combining these two parameters could be formulated (paper IV). This suggests that in Finnish small estuaries oxygen conditions near the bottom respond, at least partly, differently to external disturbance than phytoplankton biomass does.

Water residence time is another descriptor of coastal morphometry. Empirical testing of its effect on phytoplankton in the Baltic Sea has usually been unsuccessful (e.g. Wallin and Håkanson, 1991, paper III) despite the fact that the models by Wallin and Håkanson (1991), which combine residence time and nutrients, are able to predict two other indicators of eutrophication, i.e. Secchi depth and the amount of sedimentation. In the Baltic coastal waters, the theoretical residence time is typically below 0.08 years, if we exclude estuaries and coastal lagoons (Schernewski and Wielgat 2004). If we look at Finnish estuaries, the range was estimated to be greater than this, i.e. from 0.01 to 5.62 years, being greatest in the semi-enclosed inlets such as the bay of Paimionlahti and least in estuaries of the larger river basins such as Kymijoki and Kokemäenjoki (paper III). However, the ratio of chlorophyll *a* to total P or total N in the above-mentioned estuaries did not diverge from the average ratio (paper III). This suggests that the range of water residence time around the coast of Finland is probably too small to explain the variation in the amounts of chlorophyll *a* in general. In the Bay of Neva, the situation seems to be different: the short residence time of a few days, inside the flood-protection barrier, is probably the main reason for the low concentrations of chlorophyll *a* (8-10 µg l⁻¹) relative to total P (Pitkänen *et al.* 1993, paper I).

In addition to the effect of chlorophyll – nutrient yields, water residence time affects the structure of phytoplankton assemblages, too, by determining the bioavailability and utilization of nutrients. Thus, for example in the Neva Bay, phytoplankton assemblages characterized by the dominance of cryptomonadales (paper I) are consistent with the discovery that fast-growing phytoplankton, for instance flagellates, tolerate a short residence time (Ollrik 1994). Additionally, cryptomonads and green-algal species of the Ulotrichales order have an optimal growth under reduced salinity conditions (Pinckney *et al.* 1999), which also explains their

occurrences in the salt-free Neva Bay. In general, species benefiting from their competitive ability to grow fast in unstable environmental conditions include small sized green algae and flagellated taxa (Lepistö 1995). Furthermore, some diatoms benefit from rapid intrinsic rates of growth ('R-selected species'), the majority of which are also tolerant of high-frequency hydraulic disturbance ('R-strategists') according to a classification by Reynolds (1987a,b).

The effect of wind and wave expose on biota (Kahma 1986, Cattaneo 1990, Ekeboom *et al.* 2003, paper IV) can be measured by fetch, which in this study was defined as the longest diameter of the water area in the direction of the prevailing wind. In Finnish estuaries, the model combining the percentage of watershed under urban population and fetch accounted for 44% of the variation in chlorophyll *a* (paper IV). The coefficient of determination (r^2) was not very high, but the slope was highly significant. The model suggests that the stronger the mixing by wind of estuarial waters that receive nutrients from anthropogenic sources, the greater the yield of chlorophyll *a*. This accords with the study in the Gulf of Riga, which showed that both low and high wind speeds may benefit phytoplankton growth (Anonymous 2006a). Calm conditions are known to promote the growth of cyanobacteria (Kanoshina *et al.* 2003, Wasmund 1997), whereas strong winds break down the thermocline and transport nutrients up to the productive water layer, triggering there the growth of non-nitrogen fixing algae. Diatoms, because of their fast rate of growth, are another group having an ability to cope with strong water mixing conditions (Walsby and Reynolds 1980).

4.1.2 Light conditions

Light conditions control phytoplankton growth, which in turn - along with losses through grazing, settling and flushing - determine the amount of phytoplankton in the productive water layer. Humus content especially near marshy catchments (Heikkinen 1994, Pettersson *et al.* 1997), contents of resuspended material in shallow coastal areas (Mallin and Paerl 1992, Chen *et al.* 2005) and phytoplankton blooms may reduce light penetration considerably (Pustel'nikov and Shmatko 1971, May

2005). Reduced transparency, in turn, inhibits the penetration of light into the water and decreases the depth of the euphotic zone, which affects phytoplankton amounts and species composition in the water column; various phytoplankton species have different life strategies to adapt to light limitation (e.g. Talling 1971, Reynolds 1988). Water turbidity restricts phytoplankton growth especially in shallow tidal areas (Chen *et al.* 2005), where nutrient levels are typically elevated. Similarly, light limitation occurring primarily in the water column of permanent turbidity maxima enables the ecosystem to contain considerable amounts of nutrients without increasing phytoplankton biomass (Irigoiien and Castel, 1997).

The simplest measure of water transparency is Secchi depth, which shows a negative relationship with phytoplankton biomass (e.g. Carlson 1977, Nielsen *et al.* 2002, Larsson *et al.* 2006). In this thesis, the mid-summer Secchi depth in Finnish coastal waters accounted for 16 to 89% of the variation in chlorophyll *a* (Table 10). The weak correlation in the outer coastal waters of the Gulf of Finland was affected by the small range of chlorophyll *a* in the coastal type. Root mean square error (rMSE) in the model, describing the average concentrations of chlorophyll *a* was small, at its worst 9% of the average values of chlorophyll *a* of the coastal type (Table 10). Good correlations have been found in the southern Baltic Sea (Sandén and Håkansson 1996) and in the Stockholm archipelago (Larssen *et al.* 2007). Factors affecting the relationships between chlorophyll *a* and Secchi depth include phytoplankton amounts and species composition, age structure of phytoplankton assemblages, together with humic colored substances, suspended solids and clayey waters (Ollrik 1994, Tolstoy 1979) brought down by rivers to the coast.

4.1.3 Meteorological and hydrographical factors

Meteorological and hydrological conditions influence the trophic status of water and algal blooms, among others, by affecting the development of stratification, mixing of water masses and flow conditions, which in turn control many internal processes such as sedimentation. At the Baltic wide scale, specific meteorological conditions, characterised by strong westerly winds and frequent

Table 10. Models predicting summertime chlorophyll *a* as a function of Secchi depth in the outer coastal types of Finland. All the variables are log-transformed. Symbols: *n*, sample number; *r*MSE, root mean square error; *r*², coefficient of determination; *p*, significance of the slope. Locations and abbreviations of the outer coastal water types in Fig. 6.

Sea area	Outer coastal water type	Equation	<i>n</i>	<i>r</i> MSE	<i>r</i> ²	<i>p</i>
Gulf of Finland	Su	LgChl = 1.11 – 0.71 LgSec	32	0.088	0.16	< 0.0001
Archipelago Sea	Lu	LgChl = 1.23 – 1.20 LgSec	70	0.146	0.28	< 0.0001
Bothnian Sea	Seu	LgChl = 0.67 – 0.54 LgSec	43	0.071	0.36	< 0.0001
Quark	Mu	LgChl = 1.02 – 1.14 LgSec	13	0.031	0.89	< 0.0001
Bothnian Bay	Pu	LgChl = 0.88 – 0.80 LgSec	60	0.079	0.66	< 0.0001

passage of low pressure systems, cause the intrusion of saline water through Danish straits into the Baltic Sea (Weidemann 1950), which in turn may influence trophic conditions in various Baltic basins. For instance, the massive blooms of N₂-fixing *Nodularia spumigena* Mertens that were recorded in the central and eastern Gulf of Finland in 1995 were explained by the decline of the N:P ratio (Kahru *et al.* 2000). The reduced nutrient ratio was a consequence of the release of phosphate from sediment resulting from the arrival of saline and oxygen-depleted water from the Baltic Proper (Kahru *et al.* 2000). Experimental studies have shown that the growth of *N. spumigena* is stimulated either by the addition of phosphorus or a combination of phosphorus and nitrogen (Graneli 1981, Wallström 1991).

At the local scale, blooms of other cyanobacterial species may also be associated to specific meteorological and hydrographical conditions. For instance, *Microcystis aeruginosa* Kützing formed massive blooms in the south-eastern Finnish archipelago in October 1987, when conditions for transportation of the low-saline and nutrient-rich water from the River Neva were favourable (Niemi 1988, Pitkänen *et al.* 1993). The pattern was similar to what was observed in the easternmost Gulf of Finland in August 1992, when the extensive bloom of *Planktothrix agardhii* (Gomont) Anagnostidis & Komárek was associated with south-easterly winds and warm weather at a time when the mixed surface layer was thin (paper I, Pitkänen and Tamminen 1995). These exceptional weather and hydrographical conditions caused nutrient-rich waters from the River Neva and the St. Petersburg region to extend as far west as the easternmost Finnish archipelago, where salinity (2-3 psu) in the surface water layer at that time was clearly lower than normally. This contrast with typical hydrographical conditions in the eastern Gulf of Finland, in which N₂-fixing

cyanobacteria *Aphanizomenon* sp. usually dominate (paper I, Basova and Lange 1998, Rantajärvi *et al.* 1998, Kauppila and Lepistö 2001). The nearly optimal inorganic N/P ratio for phytoplankton growth appears to favour more *Aphanizomenon* than the non-N₂-fixing *P. agardhii* (Lepistö *et al.* 2005, paper I).

Vertical mixing, another important hydrographical factor, usually favours the dominance of diatoms, among others, because diatoms typically grow more rapidly than other algal groups (Walsby and Reynolds 1980). Many diatoms tolerate sinking because they benefit of the ability to take up nitrate and ammonium nitrogen in the dark (MacIsaac and Dugdale 1972). These characteristics explain the dominance of *Skeletonema subsalsum* (A. Cleve) Bethge in the Neva estuary during the mid-summer season of 1992, when the mixing depth of ca. 20 m significantly exceeded the euphotic depth of ca. 5 m (paper I). A third point is that phytoplankton species that are capable of regulating their position in the water column have a competitive advantage over immobile species in conditions where the euphotic depth is greater than the mixing depth (Reynolds 1984). This phenomenon probably explains why the abundances of cryptomonadales in the Neva estuary and the eastern Gulf of Finland were higher in 1992 than during the two previous summers (paper I).

Besides controlling phytoplankton dominance relationships directly, mixing conditions and stratification also influence indirectly phytoplankton growth by regulating many internal processes in water and sediment. In Finnish coastal waters, for instance, Niemi (1982) and Heiskanen and Tallberg (1999) have investigated resuspension and sedimentation, and Lehtoranta (2003) sediment release of phosphorus. In this thesis, sediment release of phosphorus will be discussed in the context of near-bottom oxygen conditions in Chapter 4.3.

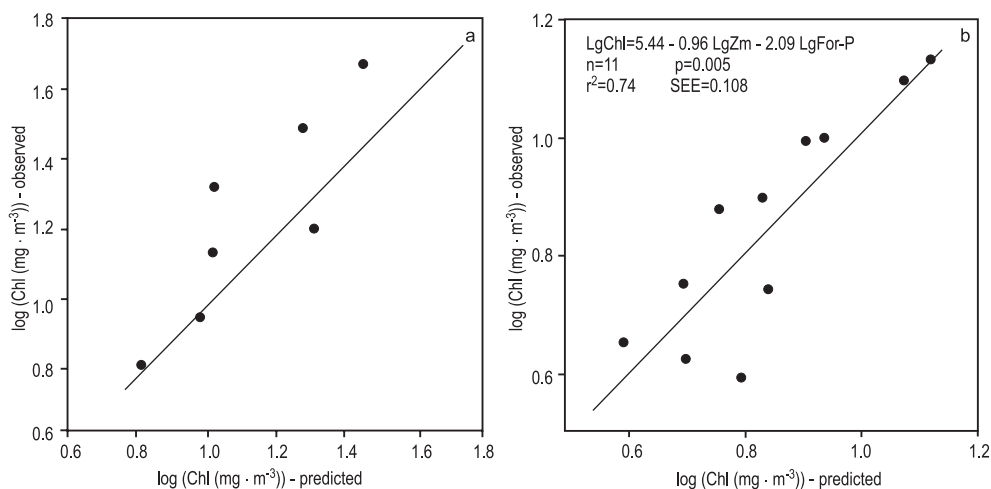


Fig. 8. Observed versus predicted values for chlorophyll *a* (Chl) estimated by the mass-balance model for the Finnish estuaries dominated by point-source loading, and (b) observed versus predicted values for log mean Chl estimated as a function of log mean depth (LgZm) and log percentage forest (LgFor-P) for the Finnish estuaries dominated by diffuse loading, where r^2 is the coefficient of determination and SEE is the model standard error of estimate (from paper III).

4.2 Chemical factors

4.2.1 Nutrient loading

Phytoplankton biomass is elevated in many coastal water areas around Finland, mainly due to considerable nutrient inputs via rivers and from local sources of nutrients (Pitkänen *et al.* 1994, papers II- IV, Tables 5 and 6). However, the trends of phytoplankton biomass do not necessarily follow the trends of nutrient inputs. This is the case especially in the Gulf of Finland, where eutrophication has continued despite reduction of external nutrient loading during the 1990s, which is, in turn, a consequence of an extensive release of phosphorus from the sediment (Pitkänen *et al.* 2001b, Lehtoranta 2003). In general, the closer the source of nutrients is to the recipient water body, the clearer is the response of phytoplankton to changes in the loading of nutrients, and vice versa (e.g. Niemi 1973, Pitkänen 1994, Lappalainen and Pesonen 2000, Elmgren and Larsson 2001, Kauppila and Bäck 2001, HELCOM 2004). In marine coastal environments, cause and effect relationships have been demonstrated by dynamic simulation models (Kiirikki *et al.* 2001, 2006).

Additionally, responses of phytoplankton to changes in nutrient loads have been proved by empirical mass-balance models, which, along with incorporating nutrient loads, employ morphometric and process parameters, such as mean depth, residence time and sedimentation coefficients. Initially, mass-balance models were developed for lake management (e.g. Vollenweider 1975, Kirchner and Dillon 1975, Meeuwig and Peters 1996), but recently they have also been applied in coastal environments (paper III). In Finnish estuaries, a two-step mass-balance model, developed based on Canfield and Bachmann's (1981) mass-balance equation, accurately estimated the variation in chlorophyll *a* (paper III). The procedure was first to predict TP concentrations as a function of both TP load, sedimentation and flushing, and, secondly, to predict chlorophyll *a* as a function of the estimated TP values. A similar approach in the Baltic Sea was applied, by for instance, Nordvarg and Håkanson (2002) and by Gyllenhammar and Håkanson (2005) to estimate the impact of the specific TP loads from fish-farms to eutrophication. These models were good, because they accounted for 72 to 84% of the variation in chlorophyll *a* and 57% of the variation in Secchi depth.

Table 11. Models predicting phytoplankton chlorophyll *a* as a function of total nitrogen (TN), total phosphorus (TP), land-use and morphometry in the Gulf of Finland (Ss, Su), the Archipelago Sea (Ls, Lv, Lu), the Bothnian Sea, (Ses, Seu), the Quark (Ms, Mu), the Gulf of Bothnia (Ps, Pu) and Finnish estuaries. Symbols: Sal, salinity; Depth, depth of sites; Zm, mean depth; For, percentage of watershed under forestry; Urb, percentage of watershed under urban population; Fetch, the measure of the longest diameter of water surface area in the direction of the prevailing wind. See the symbols of statistic as in Table 10.

Coastal water area	Equation	n	rMSE	r ²	p
Nitrogen models					
Gulf of Finland	LgChl = -4.38 + 2.01LgTN	135	0.121	0.71	< 0.0001
Archipelago Sea	LgChl = -4.24 + 1.92LgTN	274	0.123	0.78	< 0.0001
Bothnian Sea	LgChl = -3.62 + 1.64LgTN	128	0.140	0.58	< 0.0001
Quark	LgChl = -3.80 + 1.74LgTN	49	0.170	0.70	< 0.0001
Quark	LgChl = -5.77 + 0.217LgTN	33	0.089	0.93	< 0.0001
Bothnian Bay	LgChl = -1.79 + 0.95LgTN	124	0.177	0.25	< 0.0001
Bothnian Bay	LgChl = -4.29 + 1.95LgTN - 0.10LgSal	48	0.128	0.54	< 0.0001
Estuaries	LgChl = -2.60 + 1.32LgTN	19	0.184	0.75	< 0.0001
Phosphorus models ⁽¹⁾					
Gulf of Finland	LgChl = -1.02 + 1.31LgTP	135	0.117	0.73	< 0.0001
Gulf of Finland	LgChl = -0.83 + 1.26LgTP - 0.21LgSal	109	0.103	0.80	< 0.0001
Archipelago Sea	LgChl = -1.20 + 1.35LgTP	297	0.094	0.87	< 0.0001
Bothnian Sea	LgChl = -0.91 + 1.10LgTP	142	0.125	0.64	< 0.0001
Quark	LgChl = -0.90 + 1.32LgTP	49	0.170	0.70	< 0.0001
Quark	LgChl = -1.25 + 1.81LgTP - 0.22LgSal	33	0.119	0.87	< 0.0001
Bothnian Bay	LgChl = -0.44 + 0.93LgTP	124	0.117	0.67	< 0.0001
Bothnian Bay	LgChl = -0.53 + 1.06LgTP + 0.30LgSal	48	0.085	0.79	< 0.0001
Estuaries ⁽¹⁾	LgChl = -0.82 + 1.16LgTP	19	0.175	0.74	< 0.0001
Lang-use models					
Estuaries ⁽²⁾	LgChl = 5.44 - 0.96 LgZm - 2.09 LgFor	11	0.110	0.74	< 0.005
Estuaries ⁽¹⁾	LgChl = 0.95 + 0.74 LgUrb - 0.39 LgFetch	17	0.213	0.44	< 0.0001

⁽¹⁾ Equations from paper IV

⁽²⁾ Equation from paper III

Beyond the above, incorporating land-use patterns into empirical equations enables the estimation of the effect of nutrients from diffuse sources on eutrophication (Meeuwig 1999, papers III and IV, Valiela 2001, Bradshaw *et al.* 2005, Hong and Huang 2006). In Finnish estuaries, a land-use regression model using the percentage of catchment that is forested together with estuarial mean depth best predicted chlorophyll *a* in the non-point dominated estuaries ($r^2 = 0.74$), whereas the mass-balance approach was the most accurate model for the point-source loaded estuaries (paper III, Table 11, Fig. 8). In Finland, agriculture is the main source of nutrients (Pitkänen, 1994; Rekolainen *et al.* 1995; Vuorenmaa *et al.* 2002, Pitkänen and Räike 2004). The equation of the land-use model is logical, because the greater

the percentage of forested catchment, the smaller is the yield of chlorophyll *a*. An other reason why the percentage of the catchment area under agriculture did not explain the variation in chlorophyll *a* is because the chlorophyll values represented summer season, when the effect of agriculture is relatively small. Additionally, coastal rivers carry the main portion of nutrients from agricultural sources into coastal waters in spring and late autumn (Rankinen *et al.* 2006).

That said, however, when assessing dose-response relationships, uncertainty occurring in the estimation of nutrient fluxes, should be taken into account. This is especially important in the case of diffuse loading, because such loading depends substantially on hydrological conditions. Reliable

estimations of nutrient fluxes require an appropriate calculation method along with a good sampling programme. Underestimation and lack of precision are known to cause serious problems (Dolan *et al.* 1981, Richards and Holloway 1987, Preston *et al.* 1989). For example, the averaging method – which is generally used to calculate national nutrient fluxes for pollution load compilations of the Baltic Sea (e.g. HELCOM 2004) – was found to underestimate the loads of phosphorus and suspended solids by 10 to 21%, respectively, in agricultural low-lake rivers flowing into the southern and south-western coastal water areas of Finland, respectively (paper II). By contrast, the flow-stratified method overestimated the loads of phosphorus and suspended solids by about 20% in large low-lake rivers of the Bothnian Bay (paper II). Factors affecting great variation in the load estimates include differences in the methods employed to take into account, not only correlation between observed concentrations and water flow, but also sensitivity of the method to the characteristics of river basins (Lathrop 1986, Preston *et al.* 1989). In Finnish rivers, the general reliability (rMSE) for the estimates of phosphorus and suspended solids were best when using the correlation method, and for the estimate of nitrogen when using the periodic method based on Monte Carlo simulations (paper II). This is consistent with the results of a study by Ekholm *et al.* (1995). Accuracy in the loads of particulate-associated substances is reduced, for instance, by a lack of coincidence of the peaks of concentration and water flow in the beginning of snow-melt period (Walling and Webb 1988, Ulén 1995, Posch and Rekolainen 1993).

4.2.2 Nutrient concentrations

In summer, the bulk of nutrients are bound to algae, which gives rise to inter-correlation between phytoplankton biomass and total nutrients. In Finnish estuaries, the model predicting phytoplankton biomass as a function of both total N and total P accounted for ca. 75% of the variation in chlorophyll *a* (paper IV, Table 11). In Finnish coastal waters, the best predictions were achieved in the Archipelago Sea, where total P accounted for 85% and total N for 78% of the variations in chlorophyll *a* (Table 11). By contrast, in the Bothnian Bay total N predicted only 25% of the variation in chlorophyll *a*. Wallin

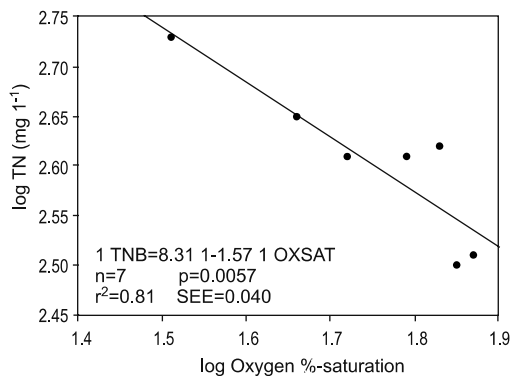


Fig. 9. Regression equation for bottom total N (TNB) as a function of bottom oxygen percentage saturation (OXSET) for seven 'pits', where r^2 is the coefficient of determination and SEE is the model standard error of the estimate. Redrawn from paper IV.

and Håkanson (1991) and Kauppila *et al.* (paper V) obtained even better results ($r^2 = \text{ca. } 0.90$) in selected coastal water areas of the Baltic Sea. The predictions made for Finnish coastal waters accord with the models developed for Swedish coastal waters, where, for example, total N alone explained 36 to 77% of the variation in chlorophyll *a* (Larsson *et al.* 2007). Additionally, nutrients in combination with morphometric or hydrographical factors improved to some extent predictions of chlorophyll *a*; salinity improved the coefficient of determination and accuracy especially in the Gulf of Bothnian (Table 11). This accords with the hypothesis of the dose - sensitivity factor presented by Wallin and Håkanson (1991). Here, dose means nutrients and sensitivity factors include morphometry (e.g. mean depth).

The results regarding the relationships between chlorophyll *a* and nutrients support the fact that phosphorus is the main limiting nutrient in the Bothnia Bay in summer, whereas N and P together limits phytoplankton primary production in the Gulf of Finland, the Archipelago Sea and the Bothnian Sea (Kivi *et al.* 1993, Tamminen and Andersen 2007). A strong nitrogen limitation seemed to improve predictions. This is the case in the Laajalahti Bay, where primary production of phytoplankton is clearly N-limited in mid-summer conditions (paper V). Similarly, the strengthening of N-limitation in the Archipelago Sea in the 1990s (Kirkkala *et al.* 1998) most probably explains the relatively high coefficient of determination ($r^2 =$

0.67) of the predictions there (Table 11). Correlation between phytoplankton and nutrients is weakened by humic substances; this fact is explained by phosphorus which is unavailable to phytoplankton when bound to stable humic matter (Salonen *et al.* 1984, Jones 1992). Thus, high humic contents (Nehring 1981, Heikkinen 1994) partly explain the weaker correlations found between phytoplankton and phosphorus in the Gulf of Bothnia compared to correlations in the southern coastal areas.

4.3 Connection to near-bottom oxygen conditions

During the recent years, many coastal water areas of the Baltic Sea have suffered from extensive oxygen deficiency at the interface of water and sediment (Pitkänen *et al.* 2001b, Lehtoranta 2003, Conley *et al.* 2006). Oxygen deficiency is partly explained by increased sedimentation of organic matter caused by considerable external nutrient loading and partly by natural factors such as coastal topography and physical stratification. In multiple regression analyses, organic matter and Fe concentrations account for 90% of the variation in sediment total phosphorus broadly across the Gulf of Finland (Lehtoranta and Pitkänen, 2003). However, despite the high nutrient loading, bottom oxygen conditions are usually better in the Neva estuary than elsewhere in the Gulf of Finland, due to more effective vertical mixing of estuarial water (Lehtoranta and Pitkänen 2003). Additionally, increases in phytoplankton biomass in the Archipelago Sea have been associated with weakened oxygen conditions occurring since the late 1990s either locally or as a result of increased nutrient transport from the Gulf of Finland and the Baltic Proper (Helminen *et al.* 1998, Kirrkala *et al.* 1998, Hänninen *et al.* 2000, Suomela 2001, Karlson *et al.* 2002, Virtasalo *et al.* 2005). Moreover, external nutrient enrichment is not the only important factor in stimulating the growth of different phytoplankton groups (Lagus *et al.* 2007) and sustaining the eutrophic state of the shallow waters of the Archipelago Sea, the presence of sediment also plays a role (Suomela *et al.* 2005).

In this thesis, relationships between summertime near-bottom oxygen conditions and chlorophyll *a*, and total nutrients were tested by regression analyses in Finnish estuaries. Neither chlorophyll

a in productive surface layer nor near-bottom concentrations of total phosphorus or total nitrogen explained variation in near-bottom oxygen concentrations or oxygen percentage saturation (paper IV). This contradicts the general understanding that oxygen usually decrease in nutrient enriched systems (OECD 1982, Conley *et al.* 2007) and that near-bottom oxygen conditions are typically associated with elevated phytoplankton production and biomass (e.g. Pitkänen *et al.* 2001b). The fact that no link between near-bottom oxygen conditions and chlorophyll *a* or nutrients was found in small Finnish estuaries arises from estuarine nutrient dynamics which are controlled not only by river and marine water inflows, but also by annual sedimentation and material fluxes between sediment and water. Much of the phosphorus and nitrogen sink out of the mixed surface layer during spring (Heiskanen and Tallberg 1999). By contrast, only a very small proportion of annual external nutrients reaches the estuaries in summer (Pitkänen 1994), when most of the nutrients in coastal waters come from the sea. Additionally, the summer concentration of phosphorus in the near-bottom water of the estuaries is a function of numerous other processes (Nixon and Pilson 1984) that have enough time to change the actual concentrations after the sedimentation peaks. Furthermore, the small range of oxygen concentrations probably weakened the results of the regression analyses (paper IV). Thus, generally speaking oxygen deficiency is not a problem in small Finnish estuaries.

In contrast to small Finnish estuaries in general, the oxygen conditions in seven 'pits', i.e. the sub-thermocline waters of deep basins, could be associated with near-bottom total nitrogen, which explained 76 and 81% of the variation of oxygen concentrations and oxygen percentage saturation, respectively (paper IV, Fig. 9). The coefficients of determination were high probably due to weakened denitrification at the sediment surface and decreased oxygen concentration in sub-thermocline waters. The effect of mixing conditions on oxygen concentrations near the bottom has been quantified in, among others, Danish estuaries and coastal areas where wind speed in late-summer coupled with nitrogen inputs from the land explained 52% of the inter-annual variation in near-bottom oxygen (Conley *et al.* 2006).

5 Developing reference conditions for phytoplankton

European waters are generally considered to have been pristine before industrialization in the mid-1800s and early 1900s (e.g. Bennion *et al.* 2004, paper V). However, little historical information describing phytoplankton biomass exists regarding Finnish coastal waters (e.g. Levander 1901, 1915, Leegaard 1920, Holmberg 1935), nor there are any data on nutrient concentrations from that time (Buch 1932). Furthermore, information on phytoplankton in those days cannot be compared with present analyses because of the qualitative nature of the historical data (Finni *et al.* 2001a, Heiskanen *et al.* 2005). However, variations in phytoplankton biomass can be estimated from other variables, which describe conditions before the period when eutrophication actually started. In defining reference conditions, Secchi depth has been found useful, because this method is easy to carry out and past observations are comparable to current monitoring data. Additionally, good correlations, found for example between chlorophyll *a* and Secchi depth in lakes and coastal marine waters (Carlson 1977, Nielsen *et al.* 2002), confirm the usefulness of the method in estimating reference conditions.

Recently, reference conditions have been defined based on historical data analyses (e.g. Krause-Jensen *et al.* 2004), empirical modeling (e.g. Larsson *et al.* 2007, Lepistö *et al.* 2006b), numerical simulations (e.g. Nielsen *et al.* 2003, Schernewski and Neumann 2005) and paleolimnological approaches (e.g. Clarke *et al.* 2003, Weckström *et al.* 2004, paper V). The study by Nielsen *et al.* (2003) provides an example of the combined approach in which simulation modeling was first used to estimate reference nutrient levels, and which then were used in empirical equations to reconstruct reference chlorophyll *a*.

5.1 Comparative approach

5.1.1 Historical observations of Secchi depth

Historical observations of Secchi depth demonstrate that oligotrophic conditions prevailed in the northern Baltic Sea in the early part of the 20th century. At that time, Secchi depth (an average 9.5 m, SD 2.5 m) was 2.5 to 3 m more than it is today (Launiainen *et al.* 1989, Flemming-Lehtinen *et al.* 2006). Observations of Secchi depth from the period 1925 to 1934 partly encompassed Finnish coastal water area. In the outer archipelago, the Secchi values of that time period varied on an average from 6.7

Table 12. Observations of Secchi depth (m) in the outer coastal water types of Finland in July 1925-1934. Reference values of Secchi depth in the inner coastal water types were calculated by proportioning the mean reference values of the outer types with the 99% deviation values of the present-day frequency distribution data in 1960-2005. The lower 99% confidence limits were calculated for the standard error of the means. Symbols: SD, standard deviation.

	Gulf of Finland		Western coastal waters			Bothnian Sea		Quark		Bothnian Bay	
	Inner (Ss)	Outer (Su)	Inner (Ls)	Middle (Lv)	Outer (Lu)	Inner (Ses)	Outer (Seu)	Inner (Ms)	Outer (Mu)	Inner (Ps)	Outer (Pu)
Secchi depth in 1925-1934											
Number of sites		7			4		3		1		4
Sample number		25			20		10		5		20
Mean	4.9 [*]	5.9	5.7 [*]	7.1 [*]	8.9	7.1 [*]	9.2	4.2 [*]	7.8	5.1 [*]	6.9
Min		2.8			5.4		4.5		5.0		2.3
Max		10.8			14.4		16.1		8.8		11.5
SD		2.03			2.03		2.93		1.73		2.42
Lower 95% confidence limit	4.8 [*]	5.8	5.6 [*]	6.9 [*]	8.7	6.7 [*]	8.7	3.6 [*]	6.7	4.9 [*]	6.7
Data on Secchi depth in 1960-2005											
99% upper percentile	4.8	5.8	5.0	6.2	7.8	6.3	8.2	4.1	7.7	4.8	6.5
95% upper percentile	4.0	5.0	4.3	5.5	7.0	5.0	6.6	3.4	5.5	3.8	5.7

^{*} Calculated by proportioning the mean reference values of the outer coastal water type with 99% percentile.

m in the Gulf of Finland to 8.9 m in the Bothnian Sea (Table 12). In the inner coastal waters, where the historical observations are absent, the average Secchi values were estimated to vary from 4.5 m in the Bothnian Bay to 7.0 m in the Bothnian Sea. These calculations were based on an assumption that the relationships between 99% percentiles analyzed from modern frequency distribution data and the average values of the historical Secchi depth are comparable in the inner and outer coastal waters (Table 12).

The present observations of Secchi depth are on average ca. 3 to 6 m less than the type-specific average reference values; the greatest deviations from the reference values being found in the outer coastal waters of the Archipelago Sea and the Bothnian Sea (Tables 6 and 12). On the other hand, the maximum values of the current Secchi depth data, especially in the outer northern Gulf of Bothnian, at the same level than their reference values, suggesting that single water bodies might in specific summer conditions still reflect even a status of excellent. Despite the above, caution is required when interpreting the figures. For example, the average Secchi depth of 5.0 m, recorded in the eastern archipelago of the Gulf of Finland in the summer of 1990 (paper I), would suggest a status of excellent (compare Table 12). However, this is inconsistent with the present knowledge on the trophic status

of the eastern Gulf of Finland (e.g. Pitkänen *et al.* 1993, paper I, Rantajärvi *et al.* 1998). Moreover, if we take into consideration the eutrophying effects of untreated sewage outfall from the St. Petersburg area that ended up in the sea already in the early 1900s (Leegaard 1920), the timing of reference conditions seem questionable at least in the case of the easternmost Gulf of Finland.

5.1.2 Reconstructing phytoplankton chlorophyll *a*

Reference concentrations of chlorophyll *a* were calculated by substituting the historical values of Secchi depth into the type-specific equations on the relationships between chlorophyll *a* and Secchi depth (Tables 10 and 12). The reconstructed chlorophyll *a* values are estimated to vary on an average between 1.6 and 3.0 $\mu\text{g l}^{-1}$ in the Gulf of Finland and Archipelago Sea, and between 1.4 and 2.8 $\mu\text{g l}^{-1}$ in the Gulf of Bothnia (Table 13). The smallest values, 1.4 $\mu\text{g l}^{-1}$, were achieved in the outer coastal waters of the Bothnian Sea and the greatest, 3.0 $\mu\text{g l}^{-1}$, in the inner archipelago of the Gulf of Finland. The higher concentrations of chlorophyll *a* in the Bothnian Bay than in the Bothnian Sea were caused by differences in seasonality, because in the Bothnian Bay, phytoplankton vernal bloom is still going on in June (Alasaarela 1980, Kauppila and Lepistö 2001).

Table 13. Reference values of chlorophyll *a* ($\mu\text{g l}^{-1}$) in Finnish coastal water types in July to August, estimated empirically based on Secchi depth. Reference values for the inner coastal types were calculated by proportioned the mean reference values of the outer types with the 5% lower percentiles of the present-day frequency distribution data in 1977-2005. Explanations in more detailed as in Fig. 12.

	Gulf of Finland		Western coastal types			Bothnian Sea		Quark		Bothnian Bay	
	Inner (Ss)	Outer (Su)	Inner (Ls)	Middle (Lv)	Outer (Lu)	Inner (Ses)	Outer (Seu)	Inner (Ms)	Outer (Mu)	Inner (Ps)	Outer (Pu)
Reconstructed chlorophyll <i>a</i>											
Number of sites		7			4		3		1		4
Total number of samples		25			20		10		5		20
Mean	3.0*	2.5	2.7*	1.8*	1.6	1.6*	1.4	2.1*	1.2	2.8*	1.6
Min		1.6			1.0		0.9		1.0		0.9
Max		5.2			2.6		2.0		1.6		3.4
SD		0.8			0.5		0.3		0.3		0.6
Upper 95% confidence limit	3.0	2.5	2.7	1.9*	1.7	1.8*	1.5	2.5*	1.4	2.8*	1.6
Data on chlorophyll <i>a</i> in 1977-2005											
5% lower percentile	2.6	2.2	2.1	1.6	1.2	1.2	0.8	1.8	1.0	1.7	1.0
1% lower percentile	1.7	1.5	1.3	1.2	0.7	0.6	0.4	1.1	0.4	1.1	0.7

* Calculated by proportioning the mean reference values of the outer coastal water type with 95% percentile.

The reconstructed concentrations of chlorophyll *a* in the Gulf of Bothnia accord with the corresponding reference values reported by Larsen *et al.* (2007) in the Swedish side of the Gulf of Bothnia. Additionally, the preliminary results of the reference chlorophyll *a* in the coastal and open waters around Finland appear to be consistent with each other (HELCOM 2006, Vuori *et al.* 2006, Laamanen *et al.* 2007a,b). In contrast, Finland and Sweden reported clearly different reference values for EU intercalibration in the middle Archipelago Sea and the comparable coastal water types in the Stockholm archipelago (Anonymous 2006a), which probably results from a failure to characterize this common intercalibration type. Typological characteristics (e.g. salinity and duration of ice cover) inside this common type are similar (Anonymous 2006a), but the basic hydrographical features of the Archipelago Sea and the Stockholm archipelago are different. The Archipelago Sea receives a part of its waters from the Gulf of Finland and a part from the Baltic proper (Helminen *et al.* 1998, Hänninen *et al.* 2000). The open sea area outside the Stockholm archipelago receives a part of its waters from the Bothnian Sea, where nutrient concentrations are among lowest in the scale of the Baltic Sea, and a part from northern Baltic proper, where, due to the stable stratification, nutrient content of the water layer above the halocline is lower than in the Gulf of Finland (Fonselius 1978, Håkansson *et al.* 1996, Wulff *et al.* 1996).

Besides errors in typology, differences in the estimated reference values may arise from methodology. In a comparative study by Nielsen *et al.* (2003), a 2-layer numerical simulation model and an empirical approach produced similar estimates in the outer zone of a Danish estuary, but the differences increased towards the inner estuarial waters. This was probably due to differences in the sensitivity of these models to response to the high variability of characteristics that contribute phytoplankton growth

and biomass in the estuary. However, comparison of methods used in various studies is often complicated by differences in the averages are calculated. Paleolimnological methods usually employ annual means, in contrast to empirical methods, which present seasonal and summertime means.

According to HELCOM (2006), today hardly any coastal water area around Finland nor in the Baltic Sea reflect pristine conditions. In Finnish coastal waters, the present chlorophyll *a* values are two to five times higher than the estimated reference values (Tables 6 and 13, papers I, III, IV, Pitkänen 1994, Kauppila and Lepistö 2001, Kauppila *et al.* 2004). However, based on the joint Finnish-Swedish intercalibration report some water bodies may still reflect a status of excellent in the Bothnian Sea and the Quark (Anonymous 2006b).

5.1.3 Reconstructing phytoplankton total biomass

Secchi depth proved to be a weak predictor of midsummer (July to August) phytoplankton biomass (ww) in Finnish coastal waters, because Secchi depth and biomass did not correlate with each other. Therefore, phytoplankton biomass was predicted using chlorophyll *a* as an independent variable. Chlorophyll *a* accounted for 24 to 50% of the variation in phytoplankton biomass (Table 14). Best predictions were achieved in the Archipelago Sea. The relationship between biomass and chlorophyll *a* was also good ($r^2 = 0.53$) in the eastern Finnish archipelago and the Neva Estuary (paper I), whereas Niemi (1971) found even better predictions (r^2 ca. 0.9) in the Kimito Estuary in southwestern Finland. In those estuarial waters, phytoplankton amounts decrease markedly from the eutrophied inner parts out towards to the more oligotrophic open sea.

Biomass - chlorophyll relationships vary greatly in the Baltic Sea (Larsson *et al.* 2006), which is affected among others by taxonomical differences of phytoplankton (Ollrik 1994), enumeration

Table 14. Models of phytoplankton biomass as a function of chlorophyll *a*. Symbols: GB, Gulf of Bothnia; AS, Archipelago Sea; and GF, Gulf of Finland. See the symbols of statistic as in Table 10. Locations of the coastal water types in Fig. 6.

Sea area	Outer coastal types	Equation	n	rMSE	r^2	p
GB	Seu, Mu, Pu	$LgBiom = 2.20 + 0.84 LgChl$	55	0.361	0.30	< 0.0001
AS	Lv, Lu	$LgBiom = 2.26 + 1.22 LgChl$	86	0.343	0.34	< 0.0001
AS	Lu	$LgBiom = 2.23 + 1.22 LgChl$	32	0.322	0.50	< 0.0001
GF and AS	Su, Lv, Lu	$LgBiom = 2.36 + 1.02 LgChl$	200	0.350	0.24	< 0.0001

Table 15. Reference values of phytoplankton biomass (mg l^{-1}) in the outer coastal types of Finland, estimated based on the reference concentrations of chlorophyll *a* ($\mu\text{g l}^{-1}$) and the type-specific relationships between phytoplankton biomass and chlorophyll *a* (see Tables 13 and 14).

	GF	Western coastal types GF and AS	BS	Q	BB
	Outer (Su)	Outer (Lu)	Outer (Seu)	Outer (Mu)	Outer (Pu)
n (sites)	7	4	3	1	4
n (samples)	25	20	10	5	20
Mean	0.67	0.34	0.21	0.16	0.21
Min	0.41	0.19	0.15	0.11	0.12
Max	1.28	0.55	0.29	0.22	0.51
SD	0.22	0.10	0.03	0.04	0.09

of large algal cells (Lepistö *et al.* 2006c), light conditions (White *et al.* 1988) and nutritional status of phytoplankton relative to amounts of cellular chlorophyll *a* (Rhee 1978, Jensen and Sakshaug 1973). Variability in the relationships are also caused by the fact that picoplankton is typically not included in the counted total biomass, although it is contained in the water samples where chlorophyll *a* is measured. Additionally, phytoplankton biomass and chlorophyll *a* have been found to respond differently to nutrient enrichment (Lagus *et al.* 2004). According to Olrik (1994), the relationships between biomass and chlorophyll *a* are highest during green algal dominance ($r^2 = 0.78$), medium during diatom dominance ($r^2 = 0.76$) and lowest during bluegreen algal dominance ($r^2 = 0.55$). The amounts of chlorophyll *a* are also relatively low among dinoflagellates (Reynolds 1984). Thus, the taxonomical differences in the amounts of chlorophyll *a* partly explain the low relationships between biomass and chlorophyll *a* achieved in the Gulf of Finland (Table 14), where cyanobacteria often predominate summertime phytoplankton assemblages (Kononen and Niemi 1984, Leppänen *et al.* 1988, Kahru *et al.* 2000, Kauppila and Lepistö 2001). In diatom dominated phytoplankton assemblages, typical for the Bothnian Bay (Kauppila and Lepistö 2001), the weak correlation may at least partly be explained by species which have relatively small amounts of cellular chlorophyll *a* in relation to their biovolumes (Reynolds 1984). Seasonal light conditions most probably contributed to the overall low relationships between biomasses and chlorophyll *a* in Finnish coastal waters (Table 14), because the relationship between light intensity and chlorophyll *a* has shown to be highest during

spring and winter and lowest in summer (White *et al.* 1988).

The midsummer reference biomasses, estimated empirically by the relationships between phytoplankton biomass and chlorophyll *a* (Table 14), varied a lot in the outer coastal waters around Finland. They ranged on an average from 0.16 to 0.21 mg l^{-1} in the Gulf of Bothnia, they were ca 0.34 mg l^{-1} in the Archipelago Sea and ca. 0.67 mg l^{-1} in the Gulf of Finland (Table 15). Larsson *et al.* (2007) estimated similar reference biomass (0.15 to 0.21 $\text{mm}^3 \text{l}^{-1}$) on the Swedish side of the coastal Gulf of Bothnia. In the Gulf of Finland especially, the reconstructed biomasses appear to be high compared to total biomasses reported for the late 1960s and early 1970s – in the period when phytoplankton biomasses and species composition in many offshore water areas indicated undisturbed conditions (e.g. Niemi *et al.* 1970, Bage and Niemi 1971, Niemi 1973, Niemi and Ray 1975, 1977). However, inter-annual variation in phytoplankton biomasses should also be taken into account. For example, according to Kononen and Niemi (1984), many dominant phytoplankton species and groups at the entrance to the Gulf of Finland during the 1970s revealed great year-to-year fluctuations, a phenomenon which the authors were unable to link to the hydrographical and chemical changes in the Baltic Proper at that time. However, total biomasses in the Gulf started to increase after the mid-1970s, which was associated with increases in surface salinity and the levels of nutrients (Kononen and Niemi 1984), as a result of saline water intrusions from the Baltic Proper (Matthäus 1982).

5.2 Multi-proxy approach

5.2.1 Tracing history of pollution from sediment and water

Five historical phases of pollution including reference conditions were identified in bay of Laajalahti (paper V). There were based on a multidimensional index of water quality, generated by principal component analysis on the sediment data (paper V). These sets of data consisted of stratigraphy of diatoms (Weckström *et al.* 2004), sediment geochemistry (Vaalgamaa 2004), stable isotopes (Vaalgamaa 2004), sedimentary pigments (Reuss 2005), longterm monitoring results of water quality and nutrient loading (paper V). The identified phases were

- (i) a pre-industrial phase (before ca. 1815),
- (ii) a phase of slight human disturbance (ca. 1815 to 1900),
- (iii) an onset of anthropogenic impact (ca. 1900 to 1955),
- (iv) a severe pollution phase (ca. 1955 to 1975)
- (v) basin recovery coupled with a phase of internal loading (from ca. 1975 onwards).

(i) In **prehistorical phase** before ca. 1815, human disturbance in the Laajalahti Bay area was minor and sedimentary nutrient levels were stable.

(ii) In the **phase of slight human disturbance** after 1815, nutrient levels started to increase slightly, most probably as a consequence of postglacial isostatic land-uplift. During this process, water exchange in the bay weakened, because the bay became partly isolated from the open sea. At that time, there was no general sewage network in Laajalahti. The predominance of benthic over planktonic taxa and the overall diverse diatom assemblages indicated by species richness suggest that the bay experienced little if any disturbance, despite the fact that abundance of planktonic diatom taxa started to increase slowly after the 1820s.

(iii) **The onset of anthropogenic impact** spanned the years from ca. 1900 to 1955. This period was characterised by a slow rise in the population along with the lack of treatment of wastewaters (Laakkonen and Peltonen 1999). The nutrient loads into the bay were estimated to be 9.9 t of total P a⁻¹

and 110 t of total N a⁻¹. The onset of eutrophication is indicated by clear changes in the sediment chemistry and biotic community structure. The rise in nutrient enrichment (expressed as diatom-inferred total dissolved nitrogen, DI-TDN and $\delta^{15}\text{N}$) and subsequent increase in organic production (measured as organic phosphorus, OP) led to oxygen deficiency (expressed as the Cu:Zn ratio) in the sediment. Algal biomass increased, as indicated by chlorophyll *a* plus its degradation products (Chl*a*D), and the abundance of benthic diatoms began to decrease simultaneously with a distinct increase in planktonic taxa, although benthic communities still dominated during this phase. According to Finni *et al.* (2001b), cyanobacterial blooms were common near Helsinki at those days.

(iv) **The severe pollution phase** covered the time period from ca. 1955 to 1975. This period was characterised by heavy wastewater loading (on an average 40 t total P a⁻¹ and 180 t total N a⁻¹) and the start of the recovery due to the introduction of the water-treatment plant built in 1957. The sedimentary record generally confirms the main trends of nutrients in the water column. The peak of the Cu:Zn ratio in the middle of the period suggest an anoxic period, which is verified by water monitoring data, which show that oxygen concentrations were low in the nearbottom water layer. Sedimentary OP showed a declining trend from the 1970s towards the top of the sediment cores. Similarly, total P in the water showed a steep decline. The increased dominance of planktonic taxa over benthic ones suggests significant nutrient enrichment. Maximum eutrophication in the middle of the period was reflected as simultaneous peaks of sedimentary nutrients (total N and DI-TDN) and sedimentary pigments (Chl*a*D and diatoxanthin), which could be evidenced by water chemistry data. Additionally, a clear loss of species richness in the bay was concomitant with the period of greatest eutrophication.

(v) **Basin recovery and the phase of internal loading**, spanning the period from ca. 1975 to the present day, represents conditions of substantial internal loading and the termination of external wastewater loading in 1986, which may be indicated by the decline of DI-TDN and organic phosphorus since the late 1980s. The Laajalahti Bay is still clearly N-limited in the summer, indicated by an

strong interdependence between chlorophyll *a* and total N ($r^2 = 92\%$). The role of denitrification is probably substantial, based on high OP and chlorophyll *a*, increase in the sediment $\delta^{15}\text{N}$ profile towards sediment surface, and constantly high wintertime nitrate nitrogen in the water. Despite a clear recovery, high nutrient concentrations and productivity mean that the Laajalahti Bay remains eutrophied. Additionally the community structure is characterised by the decline in species richness since the late 1980s and the continuing dominance of planktonic taxa over benthic taxa in the diatom assemblages. Phytoplankton is dominated by cyanobacteria and the abundance of green algae indicates significant eutrophication during the 1990s (Pesonen *et al.* 1995, Autio *et al.* 2003).

5.2.2 Establishing reference conditions for phytoplankton

Palaeolimnological analyses on sediment geochemistry and diatom community structure suggested that natural conditions in Laajalahti prevailed in the mid- to late 1800s (paper V). The time before the early 1800s does not represent realistic reference conditions, because the postglacial land-uplift increased nutrient enrichment in the estuary, which in turn was reflected in the community structure of biota. However, the kinds of changes, driven by for instance climate and hydrology, are taken into account in the Directive, which advises that 'in controlling anthropogenic pressures it is unrealistic to base reference conditions upon historic landscapes that no longer exist in modern Europe'. On the other hand, due to rise, albeit show, in the population and a lack of wastewater treatment (Laakkonen and Peltonen 1999) the status of Laajalahti in the early 1900s did not reflect undisturbed conditions either.

Based on sets of the calibration data (Weckström *et al.* 2004), past concentrations of total N were inferred to be ca. $600 \mu\text{g l}^{-1} \text{a}^{-1}$ and chlorophyll *a* to be ca. $10 \mu\text{g l}^{-1} \text{a}^{-1}$ (paper V). However, methodological uncertainties remain in the analyses. For instance, the diatom inferred function was worse for chlorophyll *a* than for total N (Weckström *et al.* 2004). That said, the reference values appear high compared to presentday figures for the annual average concentrations of total N (430 to $517 \mu\text{g l}^{-1} \text{a}^{-1}$) and of chlorophyll *a* (8.2 to $11.6 \mu\text{g l}^{-1} \text{a}^{-1}$)

in some small embayments of the Gulf of Finland affected by external nutrient loading (paper VI). This may suggest that some inner coastal areas might still today be possess good ecological status in the northern Baltic Sea, especially in those cases where human pressure to these bays is slight. In view of the proximity of the coast, and in conjunction with restricted water exchange and resuspension of nutrients from the bottom, concentrations of chlorophyll *a* and nutrients in shallow semi-enclosed bays are typically elevated compared with the offshore waters (Pitkänen *et al.* 2001a), which must also have been the case also in pristine conditions.

The data obtained from sediments provided information on the states of plankton communities at time when the Laajalahti Bay was pristine. Benthic diatoms predominated over planktonic taxa and species diversity of the assemblages was high based on the species richness index (Weckström *et al.* 2004, paper V). However, the slow increase in planktonic diatoms at the expense of benthic diatoms indicates the onset of human impact in the early 1900s. In contrast, the attempt to trace extra information on species composition by studying data on sedimentary pigment proved unsuccessful. According to Reuss *et al.* (2005), the 100 year set of records of sediment pigment in the Laajalahti Bay is too limited for this purpose. However, looking at historical records, instead, the abundances of phytoplankton appear to be small and species few in number in the early years of the 1900s, which implies undisturbed conditions. The assemblages consist mainly of the cyanobacteria *Aphnizomenon* sp. and the diatom *Chaetoceros minimus* (Levander) Marino, Giuffrè, Montresor&Zingone (Välikangas 1926). Even then, Finni *et al.* (2001b) classified the bay as moderately eutrophied based on the zooplankton species, *Filinia longiseta*, which dominated the assemblages in Laajalahti in 1919.

5.2.3 Recovery of Laajalahti from pollution based on water monitoring data

The recovery of Laajalahti from pollution can be divided into two periods: (a) the period during the municipal wastewater treatment plant (MWWTP) was operating in 1957-1986 and (b) the internal loading phase after the MWWTP was closed in 1986. The first period was characterized by rapid

declines of the concentrations of nutrients and chlorophyll *a*, and the accompanying improvements in transparency and wintertime oxygen conditions, resulting from a reduction in the loads of nutrients and organic matter (Fig. 10). The reduction was greatest for the wintertime concentrations of total P and inorganic nitrogen (DIN): total P declined by 244 mg total P m⁻³ a⁻¹ and DIN by 412 mg DIN m⁻³ a⁻¹. In contrast, summertime total P dropped on average by 24 mg total P m⁻³ a⁻¹ and 155 mg total N m⁻³ a⁻¹, respectively. The summertime DIN did not show any trend, because practically all nitrogen was bound in phytoplankton. Chlorophyll *a* decreased from 75 to 45 mg m⁻³.

As a consequence of the closure of the municipal wastewater treatment plant in 1986, the amounts of nutrients and chlorophyll *a* first dropped, but the status of the bay did not continue to improve

thereafter. The recovery was delayed by the considerable release of phosphorus from the sediment. The daily release of phosphorus from the sediment is estimated to be 4 mg m⁻² d⁻¹ (Munne and Autio 2005), which is less than the average value (13 mg m⁻² d⁻¹) found in the reduced surface sediments of the coastal Gulf of Finland (Pitkänen *et al.* 2001b), but much more than was found in oxidized bottoms in Tvärminne off Pojo Bay in the western Gulf of Finland (< 2 mg m⁻² d⁻¹, Lehtoranta and Heiskanen 2003). In the Laajalahti Bay, which is shallow, high water temperature in bottom water layer (in summer ca. 18 °C) probably strongly accelerates internal loading, because the release of phosphorus from the sediment is a microbial process, which is dependent on temperature (Holdren and Armstrong 1980, Kelderman and Van der Repe 1982). According to Munne and Autio

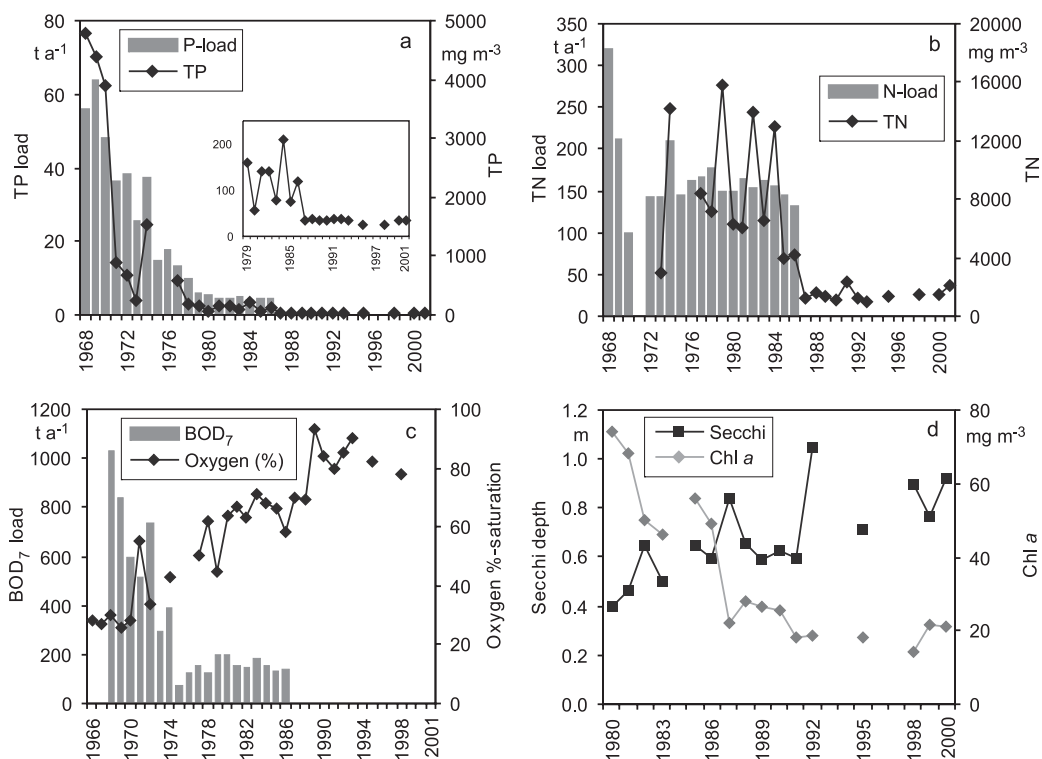


Fig. 10. Trends in water-quality and annual municipal loading of nutrients and organic matter into Laajalahti Bay during 1966-2001. (a) Surface wintertime concentrations and municipal load of total P; (b) surface wintertime concentrations and municipal load of total N; (c) near-bottom summertime oxygen percentage saturation and municipal load of organic matter (as BOD₇); (d) summertime concentrations of phytoplankton chlorophyll *a* and Secchi depth. Loading data during 1968 to 1971 based on calculations by Rekolainen (1982). The figure redrawn from paper V.

(2005), the internal loading of phosphorus (1.7 t a^{-1}) equals to external loading of phosphorus (1.1 to 2.2 t a^{-1}), whereas external loading of nitrogen (30 to 46 t a^{-1}) is ten times greater than the internal loading of phosphorus (2.9 t a^{-1}).

Internal loading may be associated with other processes in the Laajalahti Bay, too. For instance, it maintains N-limitation by releasing substantial amounts of phosphate-P from the sediment-water interface, even under good oxygen conditions in near-bottom waters (Rekolainen 1982, Munne and Autio 2005, paper V). Another process that enhances N-limitation in the bay is denitrification, which, based on high organic production (indicated by sedimentary organic P and chlorophyll *a*) and constantly high wintertime concentrations of nitrate-N in the water (paper V), probably removes large amounts of N at the sediment-water interface. Furthermore, the increase in the $\delta^{15}\text{N}$ profile in the sediment core site indicated that increasing amounts of N have been subjected to denitrification (paper V). The estuary also receives water from N-limited open coastal waters, although water exchange with the open Gulf is restricted by sounds. Additionally, the increase in the proportion of the heterocystic cyanobacteria species *Aphanizomenon flos-aquae* and *Anabaena* sp. in the summers of the late 1990's and the early 2000's suggest that N_2 -fixing has a role (Autio *et al.* 2003). The earlier dominating nonheterocystic species *Planktothrix agardhii* has not occurred in the bay in the 1990s.

6 Overall evaluation of the applicability of phytoplankton quantity as an indicator of eutrophication

The applicability of phytoplankton quantity - i.e. biomass and chlorophyll *a* - as an indicator of eutrophication can be evaluated from at least three viewpoints. First, by examining to what extent phytoplankton biomass and chlorophyll *a* fulfill the main scientific criteria used for determining a good status indicator. In short, this requirement means mainly that it is possible to predict accurately the amounts of phytoplankton biomass and chlorophyll *a* from amounts of nutrient. Secondly, by examining to what extent the indicators studied fulfil the specific

requirements set by the EU Water Framework Directive (WFD). The Directive requires among other things that biological quality elements should have type-specific and well-defined reference conditions. Finally, by examining the prerequisites that are required by coastal monitoring in order to produce reliable information on phytoplankton biomass and chlorophyll *a* for status assessments – the prerequisites naturally depend on available resources. Generally speaking, this third requirements can, in the end, be tied in with coastal management. Altogether, these three viewpoints form a three-tiered structure in which scientific criteria forms the base element establishing the reliability of indicators; water protection policies such as the WFD with their special requirements lie in the middle; and finally, coastal management including the resources it has available, forms the top of layer the structure.

The main scientific criterion established for determining a good indicator is that it responses to disturbances and anthropogenic stresses in a predictable manner and has low variability in its response (Dale and Beyeler 2001). Based on the statistical models, developed separately for estuaries and the other Finnish coastal water areas, this criterion was fulfilled: the summertime concentrations of chlorophyll *a* were predicted quite accurately from nutrient concentrations (papers III and IV, Table 11); root mean square error (rMSE) was less than 9% of the average chlorophyll concentrations in the coastal types (Tables 6 and 11). However, external annual nutrient loads alone did not explain variation in chlorophyll *a* (paper III). This is understandable, because nutrient loads entering the sea are rapidly affected by primary production and sedimentation in the close vicinity of the loading source in summer. However, the two step mass-balance equation using reconstructed concentrations of total P, predicted chlorophyll *a* most accurately in estuaries receiving nutrients from point-sources, whereas the land-use model, which, in fact, incorporate diffuse loading and turbidity, gave most accurate predictions in estuaries receiving nutrients from diffuse loading (Fig 8, paper III).

Additionally, the hypothesis was tested to establish whether nutrients in conjunction with morphometry (e.g. mean depth, fetch) or hydrographical factors (e.g. salinity) would improve predictions of coastal

Table 16. Hypotheses and conclusions of the statistical analyses in Finnish coastal waters and estuaries.

Hypotheses (H)	Location of	Main conclusions
Land-use predict better Chl than TP or TP based mass-balance model alone	Estuaries	H. supported
A model combining nutrients and morphometry more accurately estimate Chl than nutrients alone	Estuaries Finnish coastal waters	H. supported
A model combining nutrients and morphometry more accurately estimate near-bottom DO and oxygen-% saturation than nutrients alone	Estuaries	H. supported
Near-bottom DO and oxygen-% saturation are linked to Chl	Estuaries	H. not supported

eutrophication. Generally speaking, the hypothesis was supported in the set of data studied (Table 16). In estuaries, chlorophyll *a* could be predicted as a function of mean depth and land-use (paper III). Similarly, in different coastal water areas around Finland the models that combined nutrients and morphometry and hydrographical factors in most cases improved accuracy (rMSE) and the coefficient of determination (r^2 , Table 11). By contrast, the hypothesis was not supported by models predicting other descriptors of eutrophication, such as near-bottom oxygen conditions, because no multiple regression could be developed (paper IV). Land-use, described as percentage of watershed under agriculture, was the best predictor of near-bottom oxygen conditions, which emphasized the importance of diffuse loading for eutrophication. In contrast, unsuccessful attempts to describe relationships between near-bottom oxygen conditions and other descriptors of eutrophication, viz. chlorophyll *a* and near-bottom concentrations of nutrients, were explained by the fact that biogeochemical product processes have enough time to change the actual concentrations after the sedimentation peaks.

Besides pure scientific requirements, the applicability of eutrophication indicators should be evaluated in the light of requirements set by water protection policy, specified in thesis as a policy dictated by EU's Water Framework Directive. The WFD requires that biological quality elements have well-defined reference conditions in order to enable ecological classification. According to the Guidance Document on the Common Implementation Strategy (CIS) of the WFD given for coastal and transitional waters, (1) reference conditions should be type-specific; that is to say, the typology must lead to the reliable derivation of biological reference conditions (Anonymous 2003); additionally, (2) reference conditions should incorporate natural variability – both spatial and temporal; and finally, (3) variability

in natural conditions should be minimized within a type to enable valid comparisons between biological communities.

In this thesis, the indicator value of phytoplankton chlorophyll *a* in Finnish coastal waters was evaluated against the three above-mentioned criteria (1-3) concerning the establishment of reference conditions. According to the first criterion (1), the type-specific reference conditions should be reliable (Anonymous 2003), meaning that (a) the empirical models built are accurate, and (b) the reconstructed average concentrations of chlorophyll *a*, derived using the past Secchi values, are accurate. In this thesis, (a) the chlorophyll models developed for various Finnish coastal areas were relatively accurate: the most accurate models were constructed for the Quark, whereas the model developed for the Archipelago Sea was less accurate (Table 10). As for phytoplankton biomass (ww), predictive power of the model was weakened by the fact that the reference values of phytoplankton biomasses were based on the empirically reconstructed concentrations of chlorophyll *a*, and not on measured values.

Regarding accuracy, i.e. the variation of the reconstructed chlorophyll *a* across samples (b), problems occurred in a couple of the coastal water types where the number of samples was low, such as in the Quark (see chapter 3.3). Accuracy was improved by estimating confidence limits for the average mean values by using the common variation among the outer coastal types. However, it is also important to be aware that reliability in the inner coastal waters is worse than in the outer coastal types, where historical observations are available for predictions. Moreover, the intrinsic characteristics, described for instance by morphometry, vary more within the inner coastal types than within the outer types, which naturally complicates the establishment of the type-specific reference conditions for the inner coastal types. Krause-Jensen *et al.* (2005) has

demonstrated using eelgrass that the use of type-specific classification carries a risk of misinterpreting ecological conditions, especially in cases where reference conditions are less well defined.

The second criterion (2) of reference conditions is that they should incorporate natural variability – both spatial and temporal (Anonymous 2003). Based on the historical Secchi depth of the 19 observation sites, the spatial variability of the reconstructed chlorophyll values was low, but temporal large, especially when considering the broader picture of Finnish coastal water areas, for which annual observations of Secchi depth were available between the mid-summers 1925 and 1934. However, temporal variability differed depending on the coastal water area: it was best in the Gulf of Finland, and worst in the middle Archipelago Sea and the Quark. In fact, the criterion of natural variability may be associated with accuracy of the average reference conditions already described above (see the first criterion 1b). Regarding paleoecological approaches in general, it is possible to describe spatial variability within a type provided that sites within a water body are numerous. However, paleoecological techniques provide no means whatsoever of tracing seasonal variability of natural conditions. Having evaluated the site-specific reference values in the Laajalahti Bay, the conclusion is that methodological uncertainties surely played a role (papers V and VI).

The third criterion (3) set for reference conditions is that natural variation should be minimized within a coastal water type in order to enable valid comparisons between biological communities (Anonymous 2003). This criterion aims at homogeneity of a type, which was carried out in the characterization, i.e. typology, of coastal waters by hydromorphological variables, which, in turn, according to the Directive are meant to support the use of biological quality elements in the ecological classification. However, when defining reference conditions empirically, the minimized variability, i.e. homogeneity of a type, may be contradictory, because empirical modeling requires that variation must be measured. Cases showing small range of dependent variables may justify the use of more general models instead the use of type-specific models.

Finally, the applicability of phytoplankton chlorophyll *a* and biomass as indicators of

eutrophication can be assessed from the standpoint of coastal management and the resources available for coastal water monitoring. At least two important criteria for a good indicator can be found among the criteria suggested by Dole and Beyeler (2001), the criteria which can be associated with coastal management: (i) Indicators should be inexpensive to measure, (ii) and they should be anticipatory, meaning that a change in the indicator should be measurable before a substantial change in ecosystem occurs. The chemical analyses of chlorophyll *a* are, of course, cheap compared to the analyses of phytoplankton biomass and species composition, in spite of the fact that the prices of quantitative phytoplankton analyses vary to some extent depending on an analyser, the number of samples to be analysed, and the enumeration method used. However, the price should not be the only criterion. The quantitative analyses of phytoplankton biomass and species composition provide more metrics for ecological classification, the metrics which reveal various aspects of eutrophication contrary to what chlorophyll *a* alone does. Toxicity and a shifts of species composition, for instance, may be anticipatory signals of eutrophication.

The purpose of coastal water monitoring is to describe reliably enough both spatial and temporal variability in water quality. Reliable assessments of coastal water status, in turn, can be carried out by an extensive monitoring network and an sufficiently frequent sampling, which takes seasonal variation into account. For example, to assess growing season averages or to catch a vernal phytoplankton peak more frequent sampling is required than when estimating only summertime mean values. In this thesis, the statistical analyses were based on the summer period between July and August, when the number of samples can be kept to a minimum because of the stable water column and relatively small variations of nutrients and chlorophyll *a* at that time. Carstensten *et al.* (2005) constructed an approach to reduce sampling frequencies in annual and seasonal data analyses without any loss of reliability. This method produced accurate results in Danish coastal waters (Carstensten *et al.* 2005) in contrast to in the Laajalahti Bay, where it failed probably mainly due to great seasonal variation in nutrients and chlorophyll *a* (Anonymous 2006b).

7 Summary

This thesis investigated the adequacy of phytoplankton biomass (ww) and chlorophyll *a* as indicators of eutrophication in Finnish coastal waters. The process of evaluation comprised the following: carrying out statistical tests (a) on the predictability of the responses of summertime phytoplankton to nutrient concentrations and loading and (b) on the accuracy of reference conditions established separately for summertime phytoplankton biomass and chlorophyll *a*.

Concentrations of total N and total P accounted for 60 to 90% of the variation in phytoplankton chlorophyll *a* in Finnish coastal waters in summer, when most of phytoplankton biomass is bound to algae. Nutrient loads did not predict directly the variation in chlorophyll *a*, because nutrient loads entering the sea are rapidly affected by primary production and sedimentation close to the loading source in summer. Incorporating coastal morphometry (e.g. mean depth), or hydrography (e.g. salinity) into nutrient models improved accuracy (rMSE) of the predictions. A land-use regression model incorporating morphometry was most accurate in estuaries receiving nutrients from diffuse sources, whereas a phosphorus-based mass-balance equation was most accurate in estuaries dominated by point-source loading.

These regression models can be applied to two purposes: (i) predicting the average level of chlorophyll *a* in small water bodies where the concentration data are missing, and (ii) assessing the responses of chlorophyll *a* to changes in nutrient concentrations when considering restoration activities for river basin management plans. In estuaries, near-bottom oxygen conditions were associated with land-use and fetch, i.e. the diameter of the area in the direction of the prevailing wind. On the other hand, oxygen conditions near the bottom cannot directly be used to assess the level of chlorophyll *a*. In general, the high level of accuracy demonstrates the relevance of limnological approaches to predict eutrophication in Finnish estuaries, characterized by low salinity and the lack of tide.

Control of eutrophication is difficult in complex coastal marine systems like the Baltic Sea, especially when benthic release of phosphorus delays the recovery of water bodies from pollution, as in the case

of the Laajalahti Bay. Thus, numerical simulation models, incorporating biogeochemical processes in water and sediment, are probably more useful – although much more laborious and expensive – for estimating the effects of nutrient reductions on phytoplankton in complex and extensive water bodies, where the quality objectives set by the Water Framework Directive (WFD) will probably not be achieved. Whichever approach is chosen, efforts should also be made to minimize the uncertainty inherent in the estimates of riverine nutrient fluxes when deciding on how to allocate resources for the control of water pollution. Based on the root mean square error (rMSE) describing general reliability, correlation method is recommended to estimate riverine phosphorus loads and periodic method to estimate nitrogen loads. The traditional method averaging riverine nutrient fluxes underestimates phosphorus loads especially in the Archipelago Sea, where seasonal variations of water flow and water quality are usually great.

Assessing ecological status according to the WFD requires the establishment of type-specific and well-defined reference conditions. It was possible to apply empirical modeling to define reference conditions for chlorophyll *a* in outer coastal waters, where the historical observations of Secchi depth from the early 1900s partly encompassed Finnish coastal water. The coefficient of determination (r^2) varied considerably in different sea areas, which could be associated with the differences in the homogeneity of the areas relative to concentrations of chlorophyll *a* and Secchi depth. Root mean square error (rMSE), describing accuracy of the models, varied from 3 to 16% of the average concentrations of chlorophyll *a* in the coastal water types. As for phytoplankton biomass, rMSE was ca. 12% of the average biomass of the coastal water types. The predictive power of the biomass model was weakened by the fact that the reference values of phytoplankton biomasses were based on the empirically reconstructed concentrations of chlorophyll *a*, and not on measured values. Both the average reference values of phytoplankton biomass and chlorophyll *a* represent an extensive and heterogenic coastal water type. Consequently, the average reference values underestimate the "real" reference conditions in sites near the zone of the inner coastal types, because the historical

Secchi observation originated from sites near the open sea.

In the inner coastal types, reference conditions were estimated using the present frequency distribution data, which give no information on the spatial nor temporal variation of phytoplankton quantity in its pristine conditions. The inner coastal types are more heterogenic in respect to coastal morphometry and hydrographical factors. The same type-specific reference conditions represent a shallow coastal bay, a semi-enclosed deep estuary and an inner archipelago inside the coastal type, increasing uncertainty of the ecological classification.

Today, chlorophyll *a* is the only phytoplankton metric providing extensive information for the ecological classification of Finnish coastal waters. The WFD requires the inclusion of other phytoplankton indicators, too. The quantitative analyses of phytoplankton biomass and species composition are more expensive but they provide more metrics for ecological classification, the metrics which reveal various aspects of eutrophication contrary to what chlorophyll *a* alone does.

Yhteenveto

Työn tarkoituksena oli arvioida kasviplanktonin biomassaa (ww) ja *a*-klorofylliä rehevyyden indikaattorina Suomen rannikkovesialueilla, tutkimalla voidaanko kasviplanktonin määrää arvioida luotettavasti ravinteiden avulla ja voidaanko kasviplanktonille luoda referenssiolot vesipuitedirektiivin vaatimusten edellyttämällä tavalla.

Suomen rannikkovesissä kesäaikainen kokonaisuus ja -fosfori selittivät suurimman osan, noin 60-90 % kasviplanktonin *a*-klorofyllin vaihtelusta kesällä, jolloin suurin osa ravinteista on sitoutuneena kasviplanktonin biomassaan. Keskimääräistä *a*-klorofyllipitoisuutta kuvaavan regressiomallin jäännöshajonta jäi alle 9 % keskimääräisestä klorofyllipitoisuudesta rannikkovesityypeillä. Ulkoiset ravinnekuormat eivät suoraan selittäneet *a*-klorofyllin vaihtelua, koska kesäaikainen kasviplanktonin perustuotanto ja sedimentaatio vaikuttavat nopeasti merialueille päätyviin ravinnekuormiin lähellä rannikon ravinnelehteitä. Empiiristen mallien jäännöshajontaa voitiin pienentää sisällyttämällä ennusteesiin rannikon muotoa (esim. vesialueen keskisy-

vyys) tai hydrografiaa kuvaavia muuttujia (esim. suolaisuus). Jäännöshajontaa voitiin pienentää myös jakamalla jokiestuaarit pääasiallisen kuormituslähteen mukaan kahteen luokkaan. Maankäyttöön perustuva regressiomalli oli parempi jokiestuaareissa, joissa ravinteet päätyivät merialueille pääasiassa hajakuormituslähteistä, kun taas fosforipohjainen aineiden tasapainoon perustuva regressiomalli oli parempi estuaareissa, joissa pistekuormitus on valitseva kuormituslähde.

Yllä kuvattuja regressiomalleja voidaan käyttää hyväksi, kun arvioidaan vesipuitedirektiivin mukaista ekologista tilaa Suomen rannikkovesialueilla: niitä voidaan käyttää (i) arvioimaan pienten vesimuodostumien keskimääräistä *a*-klorofyllitasoa silloin, kun pitoisuustiedot puuttuvat ja (ii) arvioimaan *a*-klorofyllin vastetta ravinne-*a*erien muutoksiin esimerkiksi silloin, kun harkitaan vesialueiden tilan ylläpitoa tai palauttamista vähintään hyvään tilaan. Jokien estuaareissa pohjan läheisen hapen pitoisuus ja kyllästysaste voitiin yhdistää maankäyttömuotoihin ja rannikon morfometriaan kuten vesialueen halkaisijaan eli ”fetchiin” vallitsevan tuulen suuntaisesti määritettynä. Sen sijaan pohjanläheisten happiolojen perusteella ei voida suoraan arvioida keskimääräisen *a*-klorofyllin määrää estuaarissa. Regressiomallien jäännöshajonnan pienuus osoittaa, että nämä alunperin limnologiset lähestymistavat ovat käyttökelpoisia Suomen rannikkovesialueilla, joita luonnehtii alhainen suolaisuus ja vuorovesivaihtelujen puuttuminen.

Rehevyyden hallinta Itämeren mutkikkaissa ekosysteemeissä on vaikeaa, etenkin kun pohjasedimentistä vapautuva fosfori viivyyttää rehevöityneen vesialueen palautumista hyvään tilaan, mistä Laajalahti on hyvänä esimerkkinä. Biogeokemiallisia prosesseja kuvaavat numeeriset simulointimallit ovat todennäköisesti yksinkertaisia regressiomalleja käyttökelpoisempia silloin, kun arvioidaan miten kasviplankton vastaa ravinnekuormien vähentämiseen monimuotoisilla rannikkovesialueilla - simulointimallit ovat tosin paljon työlämpiä ja kalliimpia kuin empiiriset mallit. Kumpi tahansa lähestymistapa valitaankin, jokien ravinnevirtojen estimaattien epävarmuuteen tulisi myös kiinnittää huomiota varsinkin silloin, kun päätetään kuinka resursseja jaetaan vesiensuojelutyössä. Laskenta-kaavojen yleistä luotettavuutta kuvaavan jäännöshajonnan (rMSE) antamien tulosten perusteella

korrelaatiomenetelmää suositellaan käytettäväksi silloin, kun arvioidaan jokien kuljettamia fosforikuormia, kun taas periodimenetelmää suositellaan käytettäväksi jokien typpikuormien arviointiin. Ravinnevirtojen keskiarvoihin perustuva perinteinen menetelmä aliarvioi jokien fosforivirtoja erityisesti Saaristomerellä, jossa jokivirtaamien ja vedenlaadun kausivaihtelut ovat suuria.

Vesiputedirektiivin mukainen ekologisen tilan arviointi edellyttää, että jokaiselle rannikkovesityypille luodaan vesityypikohtaiset referenssiolot. Empiiristä mallinnusta oli mahdollista soveltaa *a*-klorofyllin referenssiolojen määrittämiseen Suomen ulommilla rannikkovesityypeillä, joilta vanhoja, 1900-luvun alun näköhavaintotietoja oli saatavilla. Regressiomallin selitysaste (r^2) vaihteli suuresti eri merialueilla: se oli pienin ($r^2 = 0.16$) Suomenlahdella ja suurin ($r^2 = 0.89$) Merenkurkussa. Tulos kuvastaa hyvin sitä, että Suomenlahti on ylipäättään klorofyllipitoisuudeltaan ja näkösyvyydeltään homogeenisempi alue kuin Merenkurkku. Keskimääräistä *a*-klorofyllipitoisuutta kuvaavan mallin jäännöshajonta vaihteli 3:sta 16 prosenttiin alueen keskimääräisestä klorofyllipitoisuudesta. Keskimääräistä kasviplanktonin biomassaa kuvaavan mallin jäännösvarianssi oli noin 12 % alueen keskimääräisestä biomassasta. Kyseistä mallia heikensi kuitenkin se että biomassaa ei voitu laskea suoraan näkösyvyyden vaan estimoitujen klorofyllipitoisuuksien funktiona. Kummassakin tapauksessa keskimääräiset referenssiarvot edustavat laajaa, heterogeenistä rannikkovesityyppiä. Keskimääräiset kasviplanktonin biomassan ja *a*-klorofyllin referenssiarvot aliarvioivat todellisia referenssiarvoja lähellä sisemmän rannikkotyypin rajaa, koska näkösyvyyden vanhat havaintopaikat sijaitsivat usein lähellä avomerta.

Sisemmillä rannikkoalueilla ei voitu käyttää empiiristä mallinnusta, vaan referenssiarvot laskettiin nykyisen seurantadatan frekvenssijakaumia hyväksi käyttäen. Kasviplanktonin *a*-klorofyllin alueellisesta tai ajallisesta luonnonvaihtelusta ei näin ollen ole mitään tietoa. Sisemmät rannikkovesityypit ovat syvyyssuhteiltaan ja avoimuudeltaan ulkoisia rannikkotyyppejä heterogeenisempiä. Yksi tyypikohdainen referenssiarvo edustaa sekä matalan lahden suualueen, puoliavoimen syvän jokiestuaarin kuin myös saaristaisen rannikkoalueen keskimääräistä *a*-klorofyllipitoisuutta. Tämä lisää ekologisen luokituksen epävarmuutta.

Tällä hetkellä *a*-klorofylli, analyysinsä halpuuden ja helppouden ansiosta, on ainoa kasviplanktonia kuvaava muuttuja, josta on Suomen rannikkovesistä pitkältä ajanjaksolta alueellisesti kattavaa tietoa. Vesiputedirektiivi edellyttää muidenkin kasviplanktonmittareiden luomista ekologista luokitusta varten. Kasviplanktonin biomassaa ja lajistokoostumus vaativat enemmän resursseja, mutta ne mahdollistavat monipuolisempien mittareiden kehittämisen. On selvää ettei rehevyyden vaihtelun kuvaaminen onnistu yksin *a*-klorofyllin avulla.

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