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NUTRIENT REDUCTIONS IN THE GOTHENBURG WASTE WATER TREATMENT PLANT AND THEIR EFFECTS ON NUTRIENT CONCENTRATIONS AND CHLOROPHYLL IN THE ESTUARY OF RIVER GÖTA ÄLV

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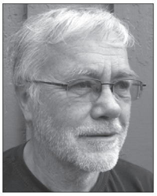

Abstract

Response to variations in nutrient inputs to the southern estuary of river Göta älv on the Swedish west-coast is investigated by means of estuarine nutrient and chlorophyll (*Chl-a*) data from 1990–05. The estuary receives nutrients from the river, and from the Gothenburg Waste Water Treatment Plant (WWTP). Since 1972, when the WWTP was built, the estuary has experienced more than 50 % decrease of phosphorus inputs, and about 20% decrease of nitrogen inputs. Estuarine concentrations of phosphate and total phosphorus (TP) and, more recently ammonium and total nitrogen (TN) have decreased substantially. Early reductions of phosphorus also gave a downward trend in *Chl-a*, while effects of recent reductions of nitrogen in the WWTP have been obscured by the larger and more variable water and nitrogen fluxes from the river. Today, annual inputs from the WWTP are 1200 tons TN and 50 tons TP, compared to river inputs of 5000 tons TN (now decreasing) and 100 tons TP. The estuarine phosphorus budget is dominated by the inflow of Kattegat waters (≈300 tons TP). Estuarine *Chl-a* decreased during the latest 4-y period (2002–05), compared to preceding periods, mainly because of a substantial decrease in river discharge and river nitrogen load. Lower discharge makes nutrient poor Kattegat waters (with lower *Chl-a*) more frequent in the estuary. However, lower nitrogen inputs, including those from the WWTP also contribute to this decrease. The importance of initial mixing of waste water is highlighted.

Key words – nutrient circulation, waste water, chlorophyll, nutrient limitation, Kattegat, Göta älv, Gothenburg, estuarine circulation, nitrogen, phosphorus

1. Introduction

This paper is a revised version of an earlier report in Swedish (Rydberg 2005), discussing the need for additional phosphorus removal in the city of Gothenburg Waste Water Treatment Plant (RYA WWTP), situated in the river Göta älv estuary (Fig. 1). The aim with this paper is to reach an international community, but also to add estuarine data from another couple of years, up to 2005, and to discuss those in the light of recent decreases in discharge and nutrient supply.

The earlier report (henceforth LR) contained a detailed study of nutrient concentrations and nutrient fluxes in the estuary. Conditions were compared for different periods, from 1970 and onwards, with the aim to investigate whether previous reductions of anthropogenic nitrogen and phosphorus have brought about any effects on nutrient concentrations and *Chl-a*. Similar studies were carried out by Anon. (2005a, b), all with the particular purpose to investigate effects of an additional 12 ton TP reduction in the WWTP.

Nutrient loads from anthropogenic sources are generally supposed to contribute to increased primary production (i.e. Bock et al. 1999), with lowered visibility and oxygen deficit in bottom waters as possible effects (Ross et al. 1993, 1994; Cloern 1996; Gowen et al. 2000).The primary production (PP), in turn is expected to increase with increasing *Chl-a*, as discussed by Söderström (1986) for the estuary of Göta älv, and in more general terms by i.e. Shiomoto (1998) and Marra et al.

Explanatory text, abbreviations

CMP = Coastal Monitoring Program
DIN = nitrate + nitrite + ammonium
NO_3-N = nitrate
NH_4-N = ammonium
NO_2-N = nitrite
PO_4-P = phosphate
PP = Primary Production
psu = practical salinity unit (≅*o/oo*)
TN = total nitrogen
TP = total phosphorus
WWTP = Waste Water Treatment Plant
1µM = 1µgat/l = 30.9µg/l P = 14µg/l N

(2003). Changes in nutrient supply can bring about both immediate and long-term consequences, the latter with links to sedimentation and burial. In semi-enclosed basins, such as the Baltic Sea, where deep-water exchange is limited, the long-term effects and the interchange with the seabed are of great importance (Carman and Rahm 1997). Processes such as denitrification and nitrogen fixation can radically change the nutrient conditions.

The city of Gothenburg built its WWTP in the early 1970s. Since then, the input of nitrogen (TN) and phosphorus (TP) to the estuary has been reduced from 3000 and 600 tons/y, respectively to about 1200 and 50 tons/y (Fig. 2). However, reports on improvements of the estuarine water quality have been rare (LR). This in turn, may be due to large river nutrient inputs. The southern branch, alone carries some 5000 and 100 tons/y respectively. Including the northern branch, which also, intermittently affects the estuary and the Gothenburg archipelago (see Fig. 1), the river nutrient loads are about 15000 and 300 tons/y, respectively.

This new article, as aforementioned, will employ another 3 years of observations, during which the WWTP fluxes and the river discharge have been lower. Focus will be on estuarine response (nutrient concentrations and *Chl-a*) to variations in river discharge and nutrient fluxes. The distribution of river and waste water beyond the estuary is also discussed, as in LR. Another two reports (Anon. 2005a; 2005b) as mentioned were simultaneously with LR evaluating the effects of reduced phosphorus input. Their results will be discussed as well.

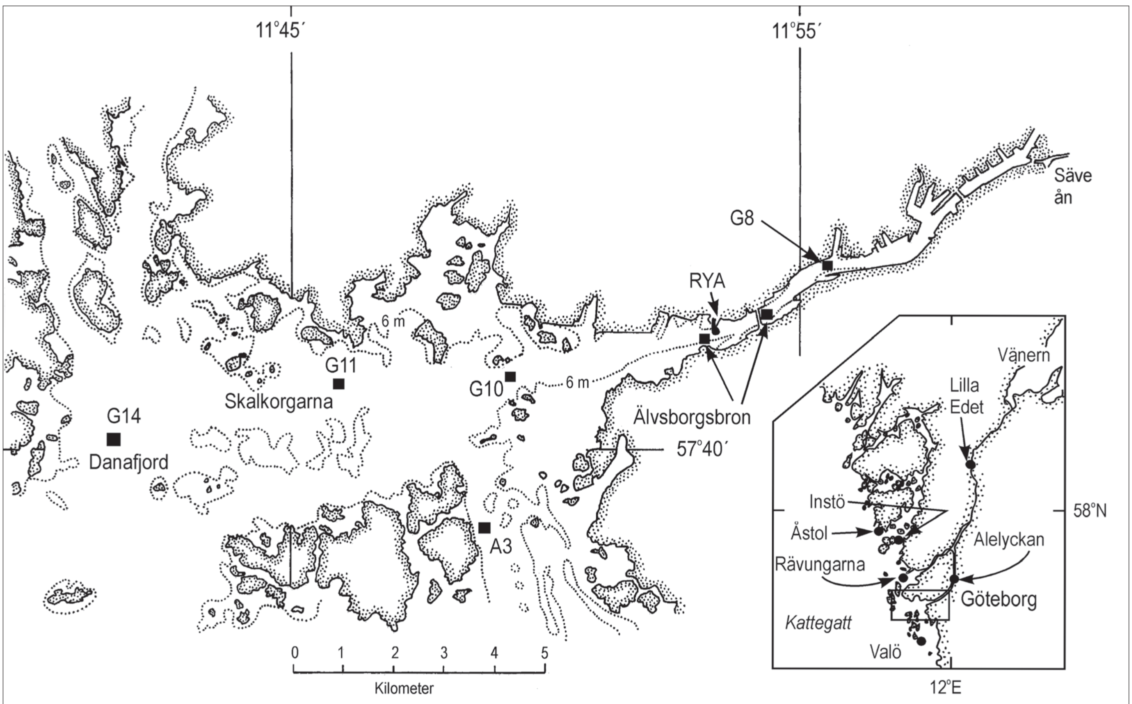

Figure 1. *The Göta Älv southern branch estuary, with surrounding archipelago. Marked stations are part of the station networks of the Coastal Monitoring Program and others. Stn Älvsborgsbron (G9) was moved to its upstream position in 1994.*

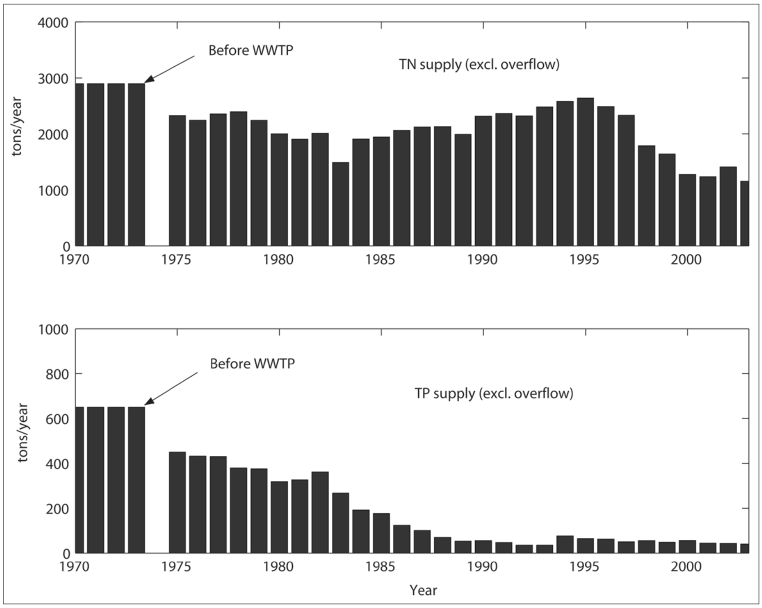

Figure 2. *Supply of TN and TP from Rya WWTP to the estuary. Occasional overflows due to storm water are not included. Nitrogen removal was introduced in the period 1994–97, phosphorus removal 1982–84. Preparation for nitrogen removal occasionally increased the load of phosphorus, during the 1990s. Increase of TN loads from 1980s is due to additional inputs (more customers).*

2. River discharge, estuarine hydrography and nutrient conditions

The Waste Water Treatment Plant (Rya WWTP) is located in the southern branch of the river Göta Älv, which runs through Gothenburg. Mean discharge in the river is $550 m^3 s^{-1}$ at the outlet from Lake Vänern (Fig. 1). About 30 % of the river discharge, controlled by a dam in the northern branch, is diverted into the southern branch and the Gothenburg estuary. The estuary is of salt wedge type, with a spring tidal range of 0.2 m (Selmer and Rydberg, 1993). Mixing is due to shear in its inner end, winds and coastal currents further out. Also the receiving Kattegat waters are strongly stratified. Low saline surface waters from the Baltic Sea, are brought northward with currents along the Swedish west coast and in Kattegat. A halocline, situated at about 15 m, separates the low saline surface waters (15–30 psu) from high saline ocean water (30–35 psu). Mixing takes place with both the surface and the deep waters (i.e. Selmer and Rydberg, 1993). The hydrography varies on a short term basis, dominated by winds. At times, ocean water occupies the whole water column.

River nitrogen and phosphorus fluxes have been observed monthly since about 1970 (www.slu.se). However, for nitrogen (TN), the technology was not consolidated until 1987. The fluxes vary on time scales of a few years, from 10 000–20 000 tons TN/y, and from 120–350 tons TP/year (Selmer and Rydberg 1993). There are also seasonal variations with lower discharge in summer.

The inner parts of the estuary, upstream of Stn G10 are strongly influenced by river water. Secchi depths vary from about 1 m in the river water to 5 m at Stn G14 (Anon., 1981). There is a restricted visibility, low salinity and a limited number of marine phytoplankton, thus also a low nutrient uptake and PP (Selmer and Rydberg 1992; 1993). Further out, the production is limited either by light (mainly in winter, from Nov–Feb) or by phosphorus (Söderström 1986; Rydberg and Selmer 1993). According to Anon. (2005a), Stn G9 is light limited throughout the year, G11 is light limited in winter and phosphorus limited in summer. Station G14 is occasionally limited by nitrogen in summer, otherwise like G11. Annual mean values of *Chl-a* are larger (at Stn G10–14 3–3.5 µg/l) than elsewhere on the Swedish west coast (2–2.5 µg/l; Anon. 2002a). Unfortunately, PP is not regularly observed in the estuary, but Söderström (1986) discussed the relation between *Chl-a* and PP, based upon a restricted number of observations in the estuary. He estimated an annual PP of 125 gC m^{-2}, at Stn G10. Open sea values on the west coast are 135–250 gC/m^{-2} (Lindahl et al. 1998; Rydberg et al. 2006). Thus, although the chlorophyll concentrations are higher in the estuary than in the open sea, the observed primary production seems lower.

Earlier investigations

Investigations in the estuary started in the early 1970s when the WWTP was taken into service. Hydrographic observations, comprising salinity and nutrient measurements, incl. chlorophyll, were undertaken from 1970–75 (Eriksson and Peippo 1975). Björn-Rasmussen (1976) focussed on the estuarine plankton community and the relationship between phytoplankton and *Chl-a*, but her work also includes raw data from the hydrographic program and comments on those. Referring to average estuarine conditions, the author found substantial decreases in TN and TP (from 61–49 µM TN and from 2.0–1.34 µM TP). The author also studied relationship between *Chl-a* data and phytoplankton and their seasonality, but did not explicitly explored changes in *Chl-a* over the study period. Anon. (1981), for example, summarizes hydrographic data from several stations from 1970–81, incl. current measurements, but there are no *Chl-a* data.

Chemical precipitation of phosphorus was introduced in the WWTP between 1982–85. This led to a successive decrease in TP input from 400 to 100 tons/y (Fig. 2). A new monitoring program, with measurements of nitrogen and phosphorus, Secchi depths, *Chl-a*, and PP was carried out during those years. Söderström (1986) calculated water exchange and fluxes of TN and TP at Stn G8 and G10. He compared the years 1982–85 and found that a reduction from 420 to 300 tons TP/y lowered mean TP concentrations from 1.55 to 1.26 µM. Söderström (1986) also pointed out that the inner estuary is light-limited, and that nitrogen removal would have marginal effects, as phosphorus limits the production further out in the estuary.

Selmer and Rydberg (1993) mapped nutrients and salinity distribution in the estuary more in detail, with measurements, in addition of planktonic nitrogen uptake, as a preparation for nitrogen removal in the WWTP. Uptake measurements indicated a low PP, while their nutrient data indicated rapidly decreasing nitrogen values towards the outer part of the estuary, thus suggesting an underestimated uptake. A box model by Selmer and Rydberg (1993) based on twelve expeditions indicated that 40 and 65 % respectively of the nitrate and ammonium supply to the estuary disappears from the surface waters inside Stn G14 (Fig. 1).

In 1990, a mutual Coastal Monitoring Program (CMP) was launched for the waters of the northern Swedish west-coast. Since then, the estuary of Göta älv including the archipelago is covered by 6–8 stations, from Valö in the south to Åstol in the north (see Fig. 1). An evaluation of the CMP for the period 1990–2003, (Anon. 2004) showed decreasing phosphate concentrations in the estuary. These were suggested to depend upon generally lower concentrations in the waters of Skagerrak and Kattegat and not upon local change. However, this report failed to see a significant decrease also of ammonium, as this took place parallel to an increase of nitrate concentrations and the inorganic nitrogen components were lumped together, shown as dissolved inorganic nitrogen (DIN).

Anonymous (2002a) deals with the conditions in Skagerrak and Kattegat. Salinity and chlorophyll data from 1970–02 are evaluated. In Kattegat, winter concentrations of nitrate and phosphate were found lower during the 1990s than during the 1980s, with a minimum during two dry years, 1996–97. The surface water (0–15m) *Chl-a* during the season of active PP, as afore mentioned is about 2 µg/l, but close to the coast 2.5–3 µg/l. Rydberg et al. (2006) on the other hand, point to the fact that there is also a substantial decrease in PP and according to Rasmussen et al. (2003) also in *Chl-a* and in phosphorus and nitrogen in the open Kattegat waters. Substantial reductions of river-borne nutrient fluxes and improved waste water treatment on the Danish side (Rydberg et al. 2006) are suggested to have affected the conditions.

Related bentic and sediment studies

There are monitoring programs also for benthic flora and fauna and for sediment. Some of these cover the whole west-coast, others focus on the estuary. HydroGIS, in a series of annual reports (e.g. Anon 2000; 2002b; 2003a) studies the influence of nitrogen removal on adherent algae and on hard and soft bottom fauna in northern Göteborg archipelago. The investigations cover the period 1996–2002. Occurrence and distribution of filamentous algae along the Bohus coast has been studied annually since 1994 (Moksnes and Pihl 1995; Pihl et al. 1999; 2001; Nilsson and Pihl 2002). Within the frame of these investigations, the Göta Älv estuary has a much lower coverage of algae than the northern areas of the coast. Sedimentation, burial and sediment nutrient content has been investigated by Cato (1992; 1997), also as a part of the CMP. The distribution of suspended matter from erosion and transport in the river has been discussed in a thesis by Brack (2000), and by Stevens and Åkermo (2003), particularly regarding ship traffic and dredging.

3. Data and methods

Several monitoring programs have been initiated to follow up changes in the estuary, in relation to lowered nutrient supply from the WWTP. However, the use of different stations and sampling depths including the lack of unified methods has limited precise evaluations of long term changes. Even the Coastal Monitoring Program (www.bvvf.se) which has been running continuously since 1990, is subject to changes (stations have been moved or withdrawn, sampling depths have been altered etc). However, data from the CMP formed the basis for the LR study. The data were used to compare estuarine water quality for the periods 1990–93, 1994–97 and 1998–02. Here, a new 4-y period, 2002–05 is added, whereby the preceding period is changed to 1998–2001.

The CMP data comprises monthly observations at several stations and depths. The following stations (see Fig. 1) were employed; Åstol, Instö Ränna, Rävungarna, Danafjord (Stn G14), Skalkorgarna (Stn G11), Älvsborgsbron (Stn G9) and Valö. Data on salinity, S, total phosphorus, phosphate, total nitrogen, ammonium, nitrate, nitrite, oxygen, O_2 and *Chl-a* were employed.

Discharge and nutrient concentrations in the river were obtained from the Swedish Electricity Board and from the Department of Environmental Analysis, Uppsala at www.slu.se. Discharges are delivered as daily data from Lilla Edet and Alelyckan (see Fig. 1). Nutrients (same as above) are measured once per month at Alelyckan. Nutrients and discharge from the river Säveån (Fig. 1) and other smaller tributaries were obtained from yearly reports, i.e. Anon (2003b).

Nitrogen and phosphorus loads from the WWTP, were obtained from the WWTP as monthly mean concentrations and fluxes of total phosphorus, phosphate, total nitrogen and ammonium, with added intermittent overflows. Such overflows occur mainly during heavy rains, because "storm water" from the streets is channelled through the waste water system, and there is an upper capacity level at WWTP to take care of the floods.

Raw data treatment

The CMP data were divided into four main periods, 1990–93, 1994–97, 1998–01 and 2002–05. Periodwise (summer, Mar–Oct and winter, Nov–Feb) salinity, *Chl-a* and nutrient concentrations, in surface and deep water, respectively, were calculated at Stn G9 (surface water 0–2 m, deep water 5–10 m), Stn G11 (surface water 0–5 m, deep water 10–14 m) and Stn G14 (surface water 0–10 m, deep water 15–30 m). Monthly nutrient fluxes from the river Göta älv were calculated by multiplying the monthly mean discharge, q_G by the concentrations, C_A at Alelyckan. The discharge from the river Säveån and other smaller tributaries, q_S was added to q_G assuming a seasonal variation with a maximum of 35 m^3s^{-1} in winter and a minimum of 17 m^3s^{-1} in summer (i.e. Anon. 2003b). Nutrient concentrations for the parameters that are regularly observed (TP, TN and nitrate) are similar to those in the Göta älv. Thus, for calculations of nutrient inputs to the estuary, q_N, season- and periodwise, the equation $q_N = (q_G + q_S)C_A$ was used.

Nutrient and volume fluxes in the estuary

Water exchange in the estuary is calculated season- and period-wise, basically in order to estimate the supply of nutrients from the sea, i.e. nutrients which enter the estuary with the undercurrent from the Kattegat (Fig. 3). As already indicated, this water may have quite variable nutrient concentrations. Calculations were made for Stn G8 (Älvsborgsbron) and Stn G14 (Danafjord) data, but the results are strictly valid for the input salinities; if the deep water inflow has a salinity, S_o and the surface water outflow a salinity S_1, and the river discharge is $q_f = q_G + q_S$, (see above) then, for steady state, salt- and volume conservation, as shown in Eq. 1–2 can be used for the calculation of the exchange flows, q_o, and q_1 (see Fig. 3).

$$q_f = q_1 - q_o \qquad \text{Eq (1)}$$

$$q_1 S_1 = q_o S_o \qquad \text{Eq (2)}$$

$$q_0 = \frac{q_f S_1}{\Delta S} \qquad \text{Eq (3)}$$

For fluxes of nutrients, $q_1 C_1$ and $q_o C_o$ the following equation is used;

$$q_1 C_1 = q_o C_o + R \qquad \text{Eq (4)}$$

where R is a sink (or source) term, e.g. the uptake of nutrients within the estuary, and C is the nutrient concentration.

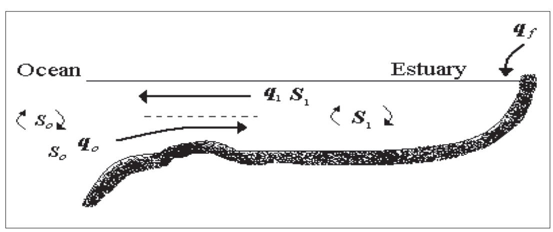

Figure 3. *A model estuary, indicating water exchange as in- and outflow (q_0, q_1) of water with salinities S_0 and S_1, respectively.*

4. Results

River nutrient fluxes and discharge

Figure 4 shows the long-term, monthly mean discharge in the river, based on 1986–05 data. The flow in the southern branch, passing Alelyckan is regulated by means of a mobile hatch in the northern branch. A minimum discharge of 120 m^3s^{-1} is needed to prevent the salt wedge from reaching the freshwater intake at Alelyckan. The annual mean discharge at Lilla Edet is 562 m^3s^{-1} and, at Alelyckan 170 m^3s^{-1}. The river Säveån and other smaller tributaries entering south of Alelyckan add some 26 m^3s^{-1}. Thus the mean discharge into the southern branch estuary therefore, averages 196 m^3s^{-1}. This additional input is accounted for in the model.

River discharge is regulated by means of a hydro-electric plant and a dam at the outlet of Lake Vänern (Fig. 1). The lake is large enough to buffer most seasonal and other intermittent variations in rainfall and inflow. Thus, the discharge is normally determined by the need for electric power, with seasonal variations as indicated in Fig. 4. However, if a long period of heavy rainfall takes place when the water level in Lake Vänern is near its maximum limit, then discharge must be kept high independent of season, although below a limit of about 1200 m^3s^{-1}, to avoid high erosion and landslides. Such conditions were at hand, particularly in the spring of 2001. For a long period, the discharge was twice normal values, with accordingly higher nutrient loads to the estuary (Karlsson and Andersson, 2003; see discussion).

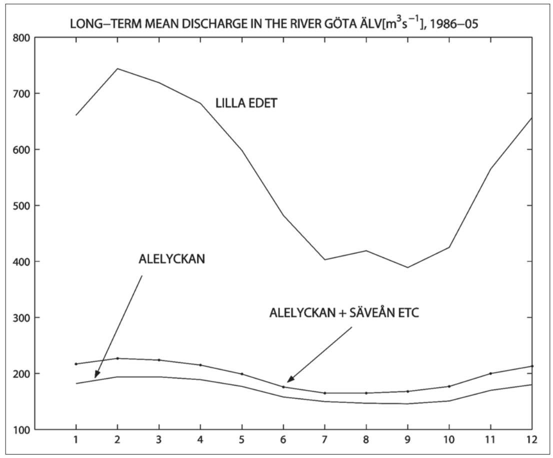

Figure 4. *Long term monthly mean discharge (1986–2005) in the river Göta Älv, at Lilla Edet and at Alelyckan (see Fig. 1). The total discharge, q_f in the southern branch is determined by adding seasonal mean discharges in the river Säveån and other smaller tributaries south of Alelyckan.*

Figure 5 shows monthly nutrient concentrations at Alelyckan. Lake Vänern has a dominating effect on the river nutrient concentrations. Nitrate and TN are always high, and their annual and seasonal variations small. Summer nitrate values are some 20 % lower than winter (Fig. 5). Concentrations of TP and phosphate are very low, 20 and 5 µg/l, respectively, thus lower than in the estuary. Lack of

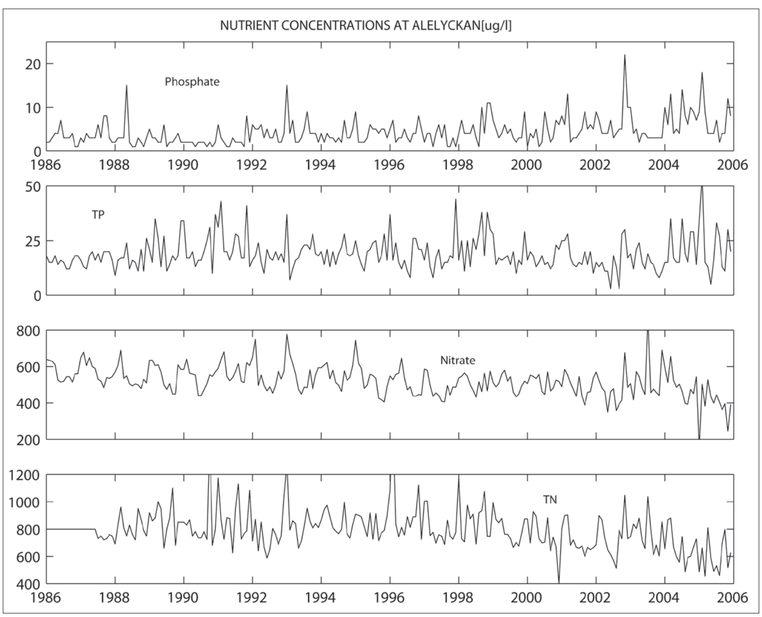

Figure 5. *Concentrations of phosphate, TP, nitrate and TN, measured once per month at Alelyckan, 1986–05.*

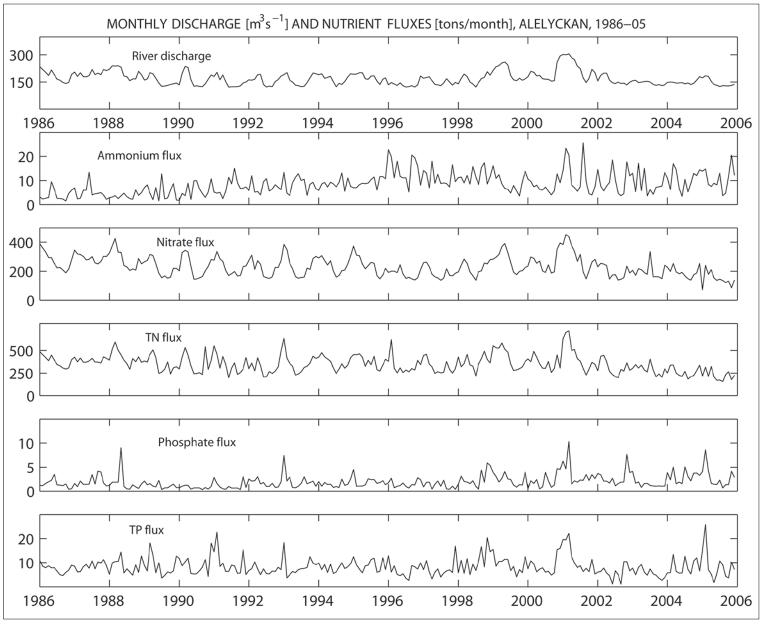

Figure 6. *Monthly river discharge and nutrient fluxes at Alelyckan, 1986–05.*

phosphorus in Lake Vänern results in a low primary production therein and high river nitrogen fluxes.

Since year 2000, there is a tendency for lower nitrogen concentrations, which can be seen also in Fig. 6, showing the corresponding nutrient fluxes at Alelyckan. Annual mean fluxes are shown in Table 1. The fluxes of TN and TP are 2–3 times larger than those from the WWTP (cf Fig. 2). In the southern branch, deviations from the mean flux are generally small. For the river as a whole, however, variations between seasons and years become larger. The period from 1998–01, for example deviates strongly from the mean, as indicated in Table 2, showing 4-y seasonal mean discharge and nutrient fluxes, data which are used to calculate nutrient uptake and circulation within the southern branch estuary. This figure clearly indicates that the long-term variations in river discharge and the consecutive nutrient inputs may easily obscure effects of reduced nitrogen and phosphorus inputs from the WWTP.

Between 1998–01 and 2002–05, the river TN and TP input decreased by 140 and 3 tons/month, respectively, which in itself is larger than any recent reduction undertaken in the WWTP. Thus, in order to make evaluations of estuarine response to the WWTP measures, in terms of nutrient concentrations and *Chl-a*, variations in the river fluxes has to be taken into account.

The nutrient fluxes are larger during winter than during summer (Table 2), with a tendency for decreasing

Table 1. *Discharge (m^3s^{-1}), concentrations and long-term monthly mean transport of nitrogen and phosphorus at Alelyckan (1986–05). For the river as a whole, the transports are approx. 3 times larger, with a larger standard deviation (Std).*

	Mean concentration (µg/l)	Mean transport (tons/month)	Std	Mean transport (tons/year)
Freshwater discharge (m^3s^{-1})	–	169.9	37.2	–
Ammonium	19.9	8.5	4.5	102
Nitrate (+Nitrite)	525	232	69	2784
Total Nitrogen, TN	794	349	99	4188
Phosphate	4.3	1.9	1.5	23
Total Phosphorus, TP	18.8	8.3	3.7	100

Table 2. *4-y seasonal mean discharge (m^3s^{-1}) at Lilla Edet and in the southern branch (incl. tributaries) and input of nitrogen and phosphorus to the southern branch (tons/month; 1986–05).*

	1986–89	1990–93	1994–97	1998–01	2002–05
Summer (Mar–Oct)					
Discharge (Lilla Edet)	639.4	380.5	440.3	703.2	410.2
Discharge (southern branch)	200.7	169.8	173.4	213.3	168.1
Phosphate Flux	2.1	1.3	1.5	2.5	2.3
TP Flux	9.1	8.5	7.9	9.7	7.4
Nitrate Flux	289.9	240.7	239.6	289.3	214.1
TN Flux	424.4	353.4	365.3	430.5	295.6
Winter (Nov–Feb)					
Discharge (Lilla Edet)	708.1	651.0	646.4	732.5	591.3
Discharge (southern branch)	223.6	207.4	200.4	246.2	185.5
Phosphate Flux	1.4	2.3	2.2	4.1	4.4
TP Flux	10.0	12.6	11.3	14.3	10.3
Nitrate Flux	348.3	341.4	300.1	345.0	245.9
TN Flux	462.8	477.5	479.9	491.4	348.6

nitrogen fluxes as well. Mean concentrations of nitrate and TN are also decreasing, as seen from Table 3. During the last 4-y period the nitrogen flux was 35 % lower than the mean of the period 1986–2001. The annual flux from 2002–05 was 3500 tons, only. The reasons behind ought to be investigated.

It is important to note that variations in river discharge also affect mixing and circulation in the estuary. Increasing discharge will force fronts and more intensive mixing further out into the estuary and also bring more turbid freshwater further out. Thus, independent of the nutrient loads, areas of high PP and Chl-a will be located further out in the estuary. These fundamental issues will be discussed further below.

Nutrient supply from Rya WWTP

Figure 7a, b shows monthly nutrient fluxes (in and out) from the Rya WWTP during 1986–05, including overflows. Mean waste water discharge since 1990 averages 4 m^3s^{-1}. The nutrient fluxes into the WWTP are subject to seasonal variations, mainly due to discharge variations. In addition, the wastewater treatment depends on the temperature, being more efficient during summer. This enhances seasonality in the outflow, particularly for nitrogen (Fig. 7a). While phosphorus removal reached its present levels in the mid 1980s (Fig. 2), investment for nitrogen reduction took place from 1994–97. Since then, the TN fluxes have decreased gradually, partly due to more efficient reduction, partly due to better capacity

Table 3. *4-y mean concentrations of TN and TP, phosphate and ammonium (µM), at Alelyckan.*

	1986–89	1990–93	1994–97	1998–01	2002–05
Summer (Mar–Oct)					
Phosphate	0.12	0.10	0.11	0.14	0.17
TP	0.56	0.63	0.57	0.57	0.55
Nitrate	38.7	37.4	35.9	35.7	33.4
TN	58.3	57.3	57.9	55.5	48.2
Winter (Nov–Feb)					
Phosphate	0.08	0.14	0.14	0.20	0.30
TP	0.57	0.76	0.72	0.72	0.68
Nitrate	42.3	44.2	38.9	37.1	34.5
TN	57.4	63.3	66.7	56.1	51.8

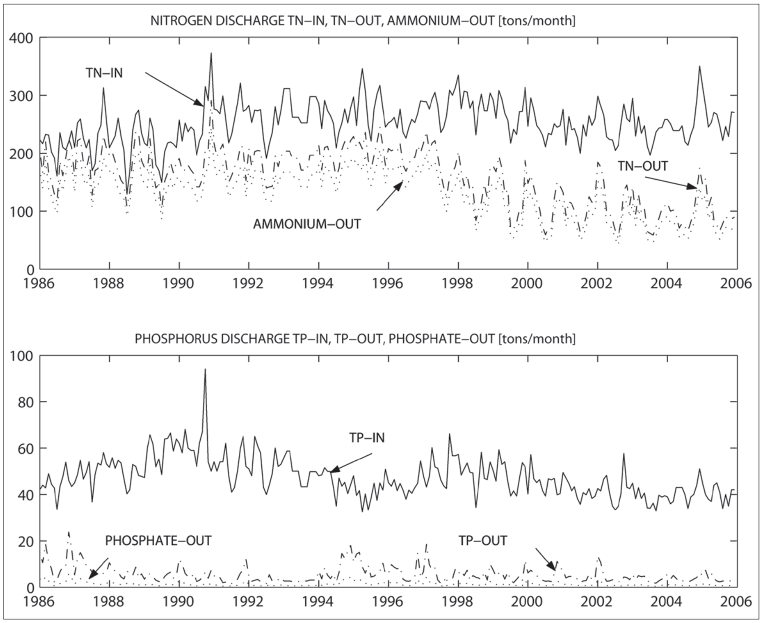

Figure 7 a–b. *Monthly in- and outflow of nitrogen and phosphorus in the Rya WWTP from 1986–05.*

for handling overflows during heavy rains. Today, the TN output averages100 tons/month (2003–06), including overflows. TP output averages 4 tons/month, corresponding to 60 and 90 % removal, respectively. As aforementioned, the river nutrient supply (Table 2), is for phosphorus 8–9 tons TP/month and for nitrogen 300–400 tons TN/month.

LR studied the ratio NH_4-N /TN, finding a mean of 0.83 in the outflow from the WWTP. During the period from 1998–2003, the mean was 0.809, with a slight decrease at hand, related to more efficient removal of nitrogen. From 2003–06, the ratio was 0.793. For phosphorus, the corresponding ratio PO_4/TP was 0.244, with small variations. Inorganic loads from the WWTP (Table 4) are calculated from TN and TP loads, using the ratios as above.

Table 4. *4-y seasonal mean river and waste water nutrient supply (tons/month), and river discharge (m^3s^{-1}). Small amounts of ammonium in the river and nitrate in the waste water are not included.*

	River 1990–93	WWTP 1990–93	River 1994–97	WWTP 1994–97	River 1998–01	WWTP 1998–01	River 2002–05	WWTP 2002–05
SUMMER								
Discharge	170	–	173	–	213	–	168	–
Phosphate	1.3	0.8	1.5	1.5	2.5	1	2.3	0.8
TP	8.5	3.3	7.9	6.3	9.7	4.1	7.4	3.2
Nitrate	241		240		289		214	
Ammonium		150		163		87		71
TN	353	178	365	194	431	107	296	88
WINTER								
Discharge	207	–	200	–	246	–	185	–
Phosphate	2.3	1.3	2.2	2.4	4.1	1.6	4.4	1.3
TP	12.6	5.5	11.3	9.8	14.3	6.8	10.3	5.4
Nitrate	341	–	300	–	345	–	246	–
Ammonium	–	168	–	172	–	121	–	105
TN	477	199	480	204	491	149	349	125

Table 5. *4-y seasonal mean values of salinity and nutrients, oxygen and Chl-a at Älvsborgsbron, Skalkorgarna and Danafjord. Stn Älvsborgsbron (G9) was moved upstream in June 1994 (see Fig 1) which resulted in a substantial decrease in the surface water salinities and increase in nitrate concentrations.*

SUMMER Surface Water	Salinity	PO_4-P	TP	NO_3-N	NH_4-N	TN	Chl-a
Älvsborgsbron (G9)							
1990–93	11.54	0.32	0.87	18.76	5.44	–	2.80
1994–97	7.40	0.22	0.96	27.88	4.02	48.76	2.36
1998–01	5.63	0.23	0.58	28.49	2.70	48.33	3.03
2002–05	5.33	0.17	0.64	26.45	3.23	49.34	2.57
Skalkorgarna (G11)							
1990–93	18.93	0.17	0.65	5.75	2.89	26.17	4.61
1994–97	18.85	0.15	0.60	7.03	2.72	29.85	4.92
1998–01	17.19	0.15	0.52	8.03	2.58	29.04	4.87
2002–05	17.78	0.12	0.53	6.06	1.76	26.10	4.12
Danafjord (G14)							
1990–93	21.29	0.15	0.57	2.74	1.20	21.07	3.43
1994–97	21.67	0.14	0.48	3.51	0.82	21.34	3.38
1998–01	20.00	0.12	0.48	3.97	0.94	22.42	4.23
2002–05	20.84	0.11	0.47	2.12	0.65	18.90	3.57

SUMMER Deep Water	Salinity	PO_4-P	TP	NO_3-N	NH_4-N	TN	Oxygen
Älvsborgsbron (G9)							
1990–93	21.85	0.40	0.92	3.77	3.43	–	6.30
1994–97	21.01	0.41	1.03	8.06	3.60	29.31	6.01
1998–01	19.20	0.49	1.01	8.13	3.67	29.35	6.39
2002–05	21.58	0.32	0.84	4.54	3.42	24.91	6.21
Danafjord (G14)							
1990–93	30.67	0.55	0.82	3.85	1.71	17.57	5.67
1994–97	30.86	0.48	0.65	4.72	1.86	18.38	5.67
1998–01	30.25	0.48	0.69	4.97	2.05	18.66	5.60
2002–05	30.25	0.45	0.69	4.30	1.73	17.38	5.65

WINTER Surface Water	Salinity	PO_4-P	TP	NO_3-N	NH_4-N	TN	Chl-a
Älvsborgsbron (G9)							
1990–93	8.98	0.51	1.03	27.52	5.00	–	1.58
1994–97	9.30	0.27	0.84	32.43	2.86	46.06	0.93
1998–01	6.81	0.37	0.71	31.48	2.40	49.86	1.87
2002–05	6.49	0.19	0.67	30.28	2.55	49.08	1.85
Skalkorgarna (G11)							
1990–93	20.59	0.50	0.85	11.19	4.38	31.61	1.79
1994–97	22.14	0.40	0.74	9.86	4.70	32.18	1.91
1998–01	20.87	0.46	0.76	12.23	3.43	31.01	1.53
2002–05	21.24	0.44	0.72	9.97	2.89	27.71	1.34
Danafjord (G14)							
1990–93	24.06	0.50	0.83	6.59	1.73	23.26	1.85
1994–97	24.83	0.38	0.70	5.82	1.76	22.65	2.45
1998–01	24.40	0.45	0.77	7.17	1.48	22.16	1.77
2002–05	23.34	0.42	0.68	6.00	1.25	21.16	1.63

WINTER DeepWater	Salinity	PO_4-P	TP	NO_3-N	NH_4-N	TN	Oxygen
Älvsborgsbron (G9)							
1990–93	23.32	0.69	1.15	9.21	2.48	–	7.06
1994–97	24.43	0.54	1.01	9.24	3.22	25.17	6.90
1998–01	24.46	0.65	1.12	9.48	2.69	28.30	6.87
2002–05	23.90	0.49	0.91	6.96	2.14	23.72	7.23
Danafjord (G14)							
1990–93	30.67	0.76	1.06	6.33	1.05	19.70	6.42
1994–97	29.75	0.56	0.79	5.88	1.05	19.74	6.39
1998–01	30.37	0.66	0.91	6.62	1.14	18.97	6.44
2002–05	29.12	0.59	0.79	5.53	1.12	17.55	6.40

As already indicated, WWTP inputs are lower than those from the river. A fundamental difference is also that the waste water nitrogen is totally dominated by ammonium, whereas the river input is totally dominated by nitrate. This difference is utilized in the forthcoming tracing of river and waste water.

Nutrient concentrations and nutrient fluxes in the estuary

Period-wise mean salinities, oxygen and nutrient concentrations incl. *Chl-a,* separated between summer (Mar–Oct) and winter (Nov–Feb), are shown in Table 5. The nutrient concentrations are expected to show variations in time, because of varying inputs and concentrations in Kattegat, but also because of variations in river discharge, which affect the estuarine dynamics. Larger river discharge will move the mixing areas (fronts) further out, which results in lower PP (and relatively higher nutrient concentrations) in the inner parts of the estuary. During 1998–01, the discharge was some 25 % higher than during the other periods (i.e. Table 3). Therefore, the salinities are lower than during the other periods, but also, nitrate and TN concentrations are higher, even though the river concentrations are low.

It is also obvious that the ammonium concentrations are much lower after the reduction of nitrogen was carried out, in 1997. The ammonium input averaged 2000 tons/y during 1990–97 and 1300 tons/y from 1998–05. In summer, surface water ammonium decreased from 2.8 to 2.1 µM at G11 and from 1.0 to 0.8 µM at G14 (Table 5). The decrease is even more obvious in winter, from 4.5 to 3.2 µM and from 1.7 to 1.3 µM, respectively. The waste water outlet is at 4 m depth right outside Rya WWTP, which means that ammonium is mixed and spread both upstream with the deep water and downstream with the surface water (Selmer and Rydberg, 1993). The decreased nitrogen load is readily seen in the deep-water at Stn G9 (Älvsborgsbron).

Phosphate and TP also show a decrease since the early 1990s (Table 5). This decrease is univocal in both deep and surface waters, although there is no change in river or waste water inputs (Table 3). However, also Kattegat data shows a general downward trend. Danish countermeasures from the late 1980s and further on have reduced phosphorus inputs to the Belt Sea and the Kattegat by some 4000 tons/year (Rydberg et al. 2006), much enough to affect phosphate and TP concentrations in the open sea.

Oxygen concentrations in the deep water are high with mean values between 6–8 ml/l (Table 5). Lowest oxygen concentrations, 4–5 ml/l usually occur in September (Karlson and Andersson 2003). There are no trends in deep-water oxygen.

The variations in *Chl-a* between different periods are relatively large (Table 5). In LR, the conclusion was that there was no trend in *Chl-a* from 1990–02. However, during 2002–05, *Chl-a* is lower at all stations compared to the earlier periods. It could indicate a downward trend, particularly as high values during the preceding period (1998–01) were caused by a much higher discharge (Karlsson and Andersson, 2003). Tables 2–3 indicate that the discharge is 20 % less and the nitrogen flux 30 % less than during the preceding period, when the *Chl-a* values are higher. Surplus of inorganic nitrogen (nitrate+ammonium) in the estuary, precludes the decrease in nitrogen load from affecting *Chl-a* at these stations. This is the essence of phosphorus limitation. However, we shall return in the discussion to this important change, to see if a higher resolution in data can improve our insight.

Figure 8 shows the development of *Chl-a* in the surface water of the estuary from 1990. There is a clear seasonal cycle at all station and a mutual connexion between G11 and G14, but not with G9. The long term *annual mean* is 3.60 µg/l at G11 and 3.04 µg/l at G14. These values are higher than elsewhere on the Swedish west coast, where the values are typically 2–2.5 µg/l. As elsewise, data from G14 are 0–10 m average, and G11 are 0–5 m average. Taking a 0–5 m average at G14 would increase the average to 3.15 µg/l, only. Thus Stn G11 features by far, the highest values. Below, estuarine chlorophyll will be compared with data from the northern branch estuary.

Early phosphorus reductions

LR studied data back to 1970, to see if the earlier TP reductions (Fig. 2) brought about any decrease in phosphorus concentrations and *Chl-a.* As mentioned, several earlier series exist, but compatibility with the CMP series were mostly poor. Table 6 shows data from1971 and

Table 6. *TP and Chl-a data from 1971 and 1974 (µM and µg/l respectively) compared to recent CMP data.The methods of compiling data are the same (basically data from June–Sep, 0–5 and 0–10 m, respectively).*

(June–Sep)	Skalkorgarna G11		Danafjord G14	
	TP	Chl-a	TP	Chl-a
1971	1.18	6.9	0.8	2.7
1974	0.77	5.3	0.63	2.3
1990–93	0.67	5.5	0.58	3.3
1994–97	0.55	5.3	0.48	3.3
1998–01	0.47	5.7	0.41	4.2
2002–05	0.50	4.2	0.41	2.8

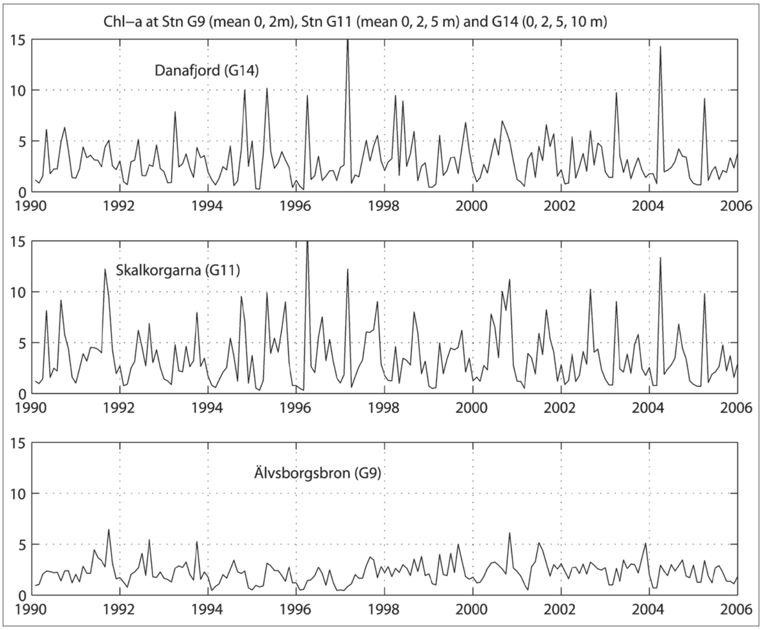

Figure 8. *Vertical mean monthly Chl-a concentrations (µg/l) at Danafjord (Stn G14), Skalkorgarna (Stn G11) and Älvsborgsbron (Stn G9) from 1990-05. Overall mean values for this period are 3.04 µg/l for Danafjord, 3.60 µg/l for Skalkorgarna and 2.25 µg/l for Älvsborgsbron.*

1974, compared to CMP data, using the same type of compilation. Between 1971 and 1974, when the WWTP was built, *Chl-a* at G11 decreased from 6.9 µg/l (June–Sep) to 5.3 µg/l, and TP concentrations from 1.18 to 0.77 µM. Lower values are seen also at G14. Phosphorus reduction in the 1980s (Fig. 2) has given still lower TP concentrations, as pointed out by Söderström (1986). However, *Chl-a* is unchanged or even higher at G14. Only during the most recent period there was a decrease in *Chl-a*. This is surprising, because the TP load decreased quite substantially (see further below).

Flux calculations from model

Fluxes of surface and deep water (q_1, q_0), calculated from Eqs 1–3 (Ch 3) were discussed in detail in LR. Calculations were carried out for each of the periods 1990–93,..,–02. The results indicated some 40 % higher inflows (q_0) during 1998–02, when the discharge was 25 % larger than during the other periods. Table 7 shows long-term mean volume fluxes. The division between summer and winter shows that the flows are 50 % larger in winter, partly due to a higher discharge and partly because of more efficient wind mixing, seen in Eq 3 as less vertical stratification (cf Table 4).

Mean nutrient fluxes, calculated according to Eq. 4, are shown in Table 8. The sink term, *R* inside of Stn G14, is remarkably large for nitrate and ammonium. In summer it's about 80 % for both nitrate and ammonium, but even in winter *R* indicates about 50 % loss of nitrate and 70 % loss of ammonium. As *R* should be basically influenced by planktonic uptake, related to primary production, the losses are too high. Several explanations are at hand; i) the nitrogen concentrations of inflowing Kattegat water are overestimated, ii) river and waste water takes a northward track and the simple model used here cannot correctly account for estuarine gradients in salinity and nutrients, iii) horisontal diffusion (mixing) is responsible for a large part of the exchange. At present, this is not a serious disadvantage, but the sink term for nitrogen components, should be regarded as too high, the inflows as too high and the outflows as too low. This problem is less important for phosphorus though, where gradients are small.

The average sink for TP is about 125 tons/y (summer). The sink corresponds to a loss of 2.5 gP $m^{-2}y^{-1}$, which in turn corresponds to a carbon sink of about

Table 7. *Seasonal mean discharge (m^3s^{-1}) in the southern branch of Göta älv, q_f and deep and surface water volume fluxes (q_0, q_1) at Älvsborgsbron (Stn G9, suffix a) and Danafjord (Stn G14, suffix d) calculated on the basis of Eqs. (1–3). Data from 1990–02.*

	River	Älvsborgsbron G9		Danafjord G14	
	q_f	q_{oa}	q_{1a}	q_{od}	q_{1d}
WINTER	229	124	353	993	1222
SUMMER	193	131	323	614	807

Table 8. *Seasonal mean nutrient fluxes (tons/month) in the southern branch of Göta älv, Q_f and deep and surface water nutrient fluxes (Q_0, Q_1) at Älvsborgsbron (Stn G9, suffix ä) and Danafjord (Stn G14, suffix d) calculated on the basis of Eqs. (1–4). Numbers in bold are (slightly) overestimated. Data from 1990–02.*

	River	Älvsborgsbron G9			Danafjord G14			Rya
	Q_f	$Q_{0ä}$	$Q_{1ä}$	$R_Ä$	Q_{0d}	Q_{1d}	R_D	
WINTER								
Phosphate	2.5	4.8	8.3	1.0	40.1	33.4	–10.9	1.75
TP	10.3	8.3	19.1	0.5	55.9	57.7	–15.8	7.31
Nitrate	334	41.3	390	25.2	**231**	289	**–278**	≤3
Ammonium	13.4	12.3	43.9	18.2	**38.8**	71.8	**–130**	149
TN	515	110	610	–15.3	**707**	1006	**–396**	180
SUMMER								
Phosphate	1.3	3.1	4.9	0.5	12.2	5.0	–9.4	1.08
TP	6.5	7.1	15.0	1.6	17.6	18.1	–10.5	4.51
Nitrate	232	24.7	262	5.3	**64.5**	69.2	**–231**	≤3
Ammonium	10.3	15.0	44.6	19.3	**26.1**	20.0	**–146**	129
TN	364	89.8	477	23.2	**258**	**444**	**–335**	155

125 gC $m^{-2}y^{-1}$, which fits in well with the primary production given by Söderström (1986). As the weight ratio N/P in phytoplankton is about 7, it means that if losses are due to planktonic uptake, the loss of nitrogen (nitrate + ammonium) should not exceed 900 tons/y, compared to the total loss according to Table 8 of more than 2500 tons/y.

For the inner estuary, inside Stn G9, the calculations are more reliable. Here, *R* is generally positive, meaning that the upper estuary is a production zone. The more detailed observations by Selmer and Rydberg (1993) showed clearly that both ammonium, nitrate and are added to this part of the estuary, presumably to mineralization of organic material from the river, which settles when the freshwater meets the saltwedge, upstream of Stn G9.

The residence time *T* for the water in the estuary can be roughly estimated from the formula $T=V/q_1$, where V denotes the volume of water inside which the flow q_1 is calculated. For the whole volume inside Danafjord the residence time in summer is about 6 days and in winter 3–4 days. For the volume inside Älvsborgsbron, the residence time is less than 24 hrs. The rapid water exchange explains why oxygen concentrations are always high in the deep water.

Further spreading of river and waste water

In LR, spreading of the river water beyond the southern branch estuary was studied by means of salinity, nitrate and TN. For waste water, ammonium was used as a tracer. Table 9 shows winter and summer mean values for all CMP stations between Valö and Åstol (for positions, see Fig. 1). As seen from Fig. 1, the northern branch of river Göta älv, with its more than twice as large discharge (Table 2) has its estuary some 20 km north of the southern branch. During northerly winds it will affect the conditions at G14. Normally however, the mixed river waters will move northwards due to the Coriolis force and the north-going coastal current.

Outside the archipelago, a natural south-north salinity gradient exists due to Baltic water outflow. From Valö to Åstol the surface water gradient is about 1 psu per 20 km; in summer from 22 to 25, in winter from 25 to 27 (Rodhe 1996). More river water reaches Åstol than Valö. The salinity difference between the stations is just about 1 psu, and Valö with salinities (Table 9) similar to the open sea, is virtually non-influenced by river water. If the salinity at Åstol is $S_å$, then the proportion of river water at Åstol may be estimated from the expression $(S_å-S_o)/S_o$ where S_o denotes the open sea salinity (in winter 27, in summer 25) outside Åstol. Thus, in summer about 10 % of the surface water at Åstol is water from the river Göta Älv (mainly from its northern branch), and about 7 % in winter. One might have expected the winter values to be higher, because of the larger discharge during winter. However, the salinity is increased by the stronger mixing in winter. In the southern branch estuary, similar rough calculations indicate some 10 % river water at Stn G14, some 25 % at G11 and 75 % at G9, on average. Likewise, the proportions of river water at Instö Ränna and at Rävungarna, are 20 and 30 %, respectively.

During winter, waste water ammonium gives a clear signal northwards to Instö Ränna (Table 9). In summer the signal is weak, presumably due to higher nutrient uptake. The high ammonium values at Rävungarna and Instö Ränna indicate that the waste water does not mix

Table 9. *Seasonal mean (1990–02) surface water salinity (psu), nutrient concentrations (µM) and Chl-a (µg/l) at all stations in the estuary of Göta älv, including the northern branch (see Fig. 1). Stn Rävungarna contains data for 3 years only. The depths included in the definition of surface water are indicated in the last column.*

	Salinity	Phosphate	TP	Ammonium	Nitrate	TN	Chl-a	Depths
SUMMER								
Valö	21.27	0.13	0.49	0.45	1.26	18.63	3.10	0–10
Älvsborgbron (G9)	8.14	0.26	0.81	4.06	24.96	48.65	2.70	0–2
Skalkorgarna (G11)	18.29	0.16	0.59	2.68	6.9	28.08	4.75	0–5
Danafjord (G14)	21.02	0.14	0.51	0.98	3.33	21.55	3.63	0–10
Rävungarna	15.73	0.16	0.58	0.87	12.33	28.84	3.91	0–5
Instö Ränna	20.04	0.15	0.56	0.66	4.29	22.34	3.73	0–10
Åstol	22.61	0.14	0.49	0.45	2.39	19.22	3.38	0–10
WINTER								
Valö	24.72	0.44	0.75	0.89	4.79	20.55	2.45	0–10
Älvsborgbron (G9)	8.36	0.37	0.86	3.41	30.52	48.03	1.48	0–2
Skalkorgarna (G11)	21.24	0.45	0.78	4.17	11.09	31.49	1.74	0–5
Danafjord (G14)	24.40	0.44	0.76	1.63	6.52	22.73	2.04	0–10
Rävungarna	18.4	0.36	0.71	2.12	15.21	30.81	2.60	0–5
Instö Ränna	22.31	0.40	0.71	1.77	10.41	26.20	2.01	0–10
Åstol	25.19	0.44	0.72	1.14	6.61	21.76	2.12	0–10

thoroughly with the river water in the southern branch. This is also seen in a detailed cross-section shown in Anon. (1981). In winter, the surface mean values at Instö Ränna are higher than at Stn G14, showing that waste water is routed northward along the coast. Simulations done by Anon. (2005a) also indicate this routing of the waste water, although dilution is overestimated in their figures, because of an assumed too high initial mixing. As discussed in evaluation of the box model fluxes, this northgoing flow means that the uptake (particularly of nitrogen) becomes overestimated.

Summer *Chl-a* (Table 9) data shows that enhanced production is at hand within the archipelago northwards to Åstol, and possibly somewhat further north. However, only about 3 % of the river water will enter the Orust-Tjörn fjords (see Björk et al. 2000). Karlson and Andersson (2003) studied the recipient during a period of extreme river discharge in 2001 (Table 2; Fig. 6). The discharge during the spring of 2001 was about 1200 m^3s^{-1}, 300 m^3s^{-1} of which in the southern branch. They showed increased nitrogen concentrations at Åstol, but also enhanced *Chl-a*. The conclusion is that a high river discharge is needed to affect the conditions at Åstol. In fact, Åstol is a station where nitrate from the river may *actively* affect the *Chl-a*, as at this station during 3–6 months per year, nitrate values are low and nitrate may limit production. Thus, as a coarse estimate, the area of influence of river water and its nitrogen is about 500 km^2, from Åstol to Valö and approximately to the outer rim of the archipelago.

5. Discussion and conclusions

The aim in LR was to investigate whether variations in the discharge of the river and/or earlier reductions of the nitrogen and phosphorus supply from the Rya WWTP has or has had any effect on nutrient concentrations, flows and *Chl-a* in the estuary. LR used data from the Coastal Monitoring Program divided into 3 periods. Here, another three years of observations have been added, such that there is a fourth period available for comparison with earlier data and investigations. A particular issue in LR was whether another12 tons/y reduced phosphorus would yield any effects. The conclusion was that the reduction would lead to a marginal outward displacement of zone affected by river water, a 2.5 % increase of the zone where phosphorus is the limiting nutrient (i.e. where nitrate+ammonium are available in excess). The change in area was estimated by comparing the total supply of TP, ≈4–500 tons/y (100 tons/y from the river, 300 tons from the Kattegat and 50 tons from the WWTP) with the phosphorus reduction. Similar conclusions, but differently formulated, were drawn by Anon (2005a), expressing the change in terms of *Chl-a*; a decrease of 1.7–3.4 % for 16 tons/year, and in concentrations of TP (0.2–0.8 %) and phosphate (0.6–0.9 %). Anon. (2005b) indicates 12 tons corresponding to 5 % less phosphorus input, but uses a smaller area in their calculation of phosphorus flux from the sea.

In this study, it is shown that TP and phosphate concentrations in the estuary have continued to decrease during recent years, and also that *Chl-a* was substantially lower during 2002–05 (Table 5). The decrease in *Chl-a* was 30 %, compared to the preceding period. Although the long term decrease in TP (Table 6) is obviously related to reduced phosphorus input from the WWTP (from 600 to 50 tons/y since 1971; Fig. 2), the decrease in TP and phosphate from 1990 (Table 5) is more likely just a part of a general phosphorus decrease, related to a lower input to the Kattegat and the Belt Sea (Rasmussen et al. 2003; Rydberg et al. 2006). Lower *Chl-a,* on the other hand, is most likely due to lower river discharge and nitrogen input. As may be evaluated from Table 4, the land-based inputs of nitrogen decreased from about 7500 tons/y (1990–01) to about 4500 tons/y (2002–05). Detailed studies of the hydrographic records from Skalkorgarna (G11) show that it is extremely rare that Kattegat waters, which are depleted in nitrate during summer and has lower *Chl-a*, is dominating at this station. At Stn Danafjord, however, pure Kattegat water is present typically during three months (three times) per year, but during the latest period, 2002–05, it happened more often. With few exceptions, these occasions were simultaneous with low *Chl-a.* Thus, while phosphorus limits the production at Skalkorgarna, nitrogen limitation (see also Anon. 2005a) was more frequent at Danafjord, because of a combination of low river discharge and nitrogen input (both are important).

Table 10 compares, with optimum depth resolution, the conditions at Skalkorgarna and Danafjord during the periods 1998–01 and 2002–05, respectively. The large difference in nitrate concentrations between the two periods is readily seen, and it is obvious that the river front is further out during 1998–01, when the discharge is higher. Comparing nitrate and *Chl-a* at Skalkorgarna shows that maximum *Chl-a* appears when the (mean) nitrate concentrations are 3–7 µM while for Danafjord maximum appears at lower mean concentrations, typically 2–4 µM, but also the maximum *Chl-a* concentrations are lower. When pure Kattegat water with very low nitrate concentrations appears at Danafjord, then *Chl-a* is also low. In summary, the variations in river discharge have a large impact on estuarine nutrient conditions and phytoplankton growth. It determines the area of influence of river nutrients, and similarily for waste water nutrients. The waste water nutrients, as pointed out, seem to take a more northerly track, than the river water as judged from ammonium concentrations. However, also the waste water nutrients will be contributing to *Chl-a* and primary production. The reduction of nitrogen in the WWTP (>1000 tons/y) resulted in a decrease of ammonium at Danafjord from about 1 to 0.65 µM (Table 5), and although not explicitly shown, this reduction presumably contributed to lowering *Chl-a* at Stn Danafjord.

Today, nutrient reductions in the WWTP amount to 60 % for nitrogen and 90 % for phosphorus, whereby inputs to the estuary are 1200 tons TN and 50 tons TP per year. Therefore, the estuarine nitrogen budget is dominated by the river input (5000 tons TN/y) and the phosphorus budget by inflow of Kattegat waters (3–400 tons/y), both of which are also decreasing at the moment. On the other hand, the waste water is also mixed with river water from the northern branch and waste water nutrients inputs, in fact should be compared to the total fluxes in river Göta älv, and to the total estuarine exchange with Kattegat waters. Surprisingly high ammonium concentrations from the WWTP appear at Rävungarna and Instön in the northern branch estuary (see Fig. 1), indicating a restricted initial mixing with the southern branch waters at the outlet near Älvsborgsbron.

Table 10. *Summer (Mar-Oct) mean salinity (psu), nitrate and phosphate concentrations (µM) and Chl-a (µg/l) at Skalkorgarna (Stn G11) and Danafjord (Stn G14) during the most recent 4-y periods, 1998–01 (98) and 2002–05 (02), respectively. The first period experienced 20 % higher discharge and 40 % higher nitrogen input from the river.*

	Depth	Sal 98	Sal 02	Nitrate98	Nitrate02	Phos98	Phos02	Chl98	Chl02
Danafjord G14									
	0	17.27	19.12	7.06	3.17	0.12	0.10	4.24	3.68
	2	18.79	19.65	4.65	2.33	0.11	0.09	4.95	3.77
	5	20.55	20.72	2.62	1.39	0.11	0.09	4.32	3.27
	10	23.35	23.24	2.14	1.60	0.14	0.14	3.01	3.53
Skalkorgarna G11									
	0	13.59	15.12	14.44	10.48	0.18	0.13	3.80	3.55
	2	17.53	17.68	7.54	5.60	0.15	0.12	5.97	4.42
	5	20.33	20.27	3.35	2.13	0.14	0.11	4.47	4.37
	10	22.59	23.48	2.52	2.04	0.16	0.17	3.25	3.52

If so, it is obvious that further reduction of nutrients in the WWTP will have a marginal impact on the southern branch estuary (at Danafjord), while further effects should be looked for in the northern branch. At least, an improved initial mixing might restrict export of WWTP towards north.

The river waters are turbid with a low visibility (1 m), as mentioned and production is therefore light limited out to where estuarine mixing becomes efficient (2–5 m in the area between G10–G14, i.e. Anon. 2005a). A lower discharge, as during the latest 4-y period will bring about improved visibility and therefore a more efficient production further in. This scenario was not investigated. Phosphorus in sediments from earlier periods when the load was larger may leak to the bottom waters of the estuary (Anon. 2005a). Investigations of the exchange between bottom waters and sediments will improve estuarine modelling.

Söderström (1986) carried out a limited number of primary production measurements in the estuary and correlated those with *Chl-a* data in an effort to do ecological modelling. Data on primary production were in the range 110–140 gC $m^{-2}y^{-1}$, evaluated about halfway between Stn G9 and G11. The production values are lower than those of the open Kattegatt (e.g. Rydberg et al. 2006), which may seem surprising, knowing that *Chl-a* is substantially higher than anywhere else in the area. Söderströms assessment yields a short and intense peak in production in the period June–August, while long term data shown here seem to have a more evenly distributed production. Rydberg and Selmer (1993) studied phytoplankton and nutrient uptake but, like Söderström with too few data. In summary, it is quite clear that the internal processes that govern the dynamics of nutrients in the estuary should have been studied much more in detail. Internationally, as indicated in the introduction, this is an important subject (i.e. Ross et al. 1994; Gowen et al. 2000; Seitzinger and Sanders 1997). In Sweden, the conditions in the Himmerfjärden have been extensively studied (Elmgren and Larsson 1997; Savage 2003). Few reports, if any, have tried to go deep into the nutrient dynamics of the river Göta älv estuary. It its obvious with the great amount of available data, that more can be done.

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References

Anon. 1981 Recipientundersökningar år 1980–1981. Göteborgsregionens Ryaverk AB, Vattenvårdsanläggningar. Rapport no 15.

Anon., 2000 Kvävereningens inverkan på bottnarna i Göteborgs norra skärgård. HydroGIS. Rapport 226.

Anon., 2002a OSPAR Assessment 2002 Swedish National Report on Eutrophication Status in the Kattegat and the Skagerrak. SMHI Rapport 79.

Anon., 2002b Uppföljning av kvävereningens inverkan på bottnarna i Göteborgs norra skärgård. HydroGIS Rapport 266.

Anon., 2003a Uppföljning av kvävereningens inverkan på bottnarna i Göteborgs norra skärgård. HydroGIS Rapport 311.

Anon., 2003b Rapport avseende 2002 års vattendragskontroll. Del B: Säveån. Göta älvs vattenvårdsförbund, april 2003.

Anon., 2004 Sammanställning och utvärdering av hydrografiska mätningar längs Bohuskusten. SMHI Rapport 57, 2004.

Anon., 2005a Utredning av effekterna av fosforutsläpp från Ryaverket. DHI. Slutrapport. Mars 2005. Bohuskustens vattenvårdsförbund, Uddevalla (available in English at www.bvvf.se).

Anon., 2005b Utredning kring effekterna av ett minskat utsläpp av fosfor från Ryaverket, NIVA. Bohuskustens Vattenvårdsförbund. LNR 4947-2005. Uddevalla (available in English at www.bvvf.se).

Bock, M.T., Miller, B.S. and Bowman, A.W., 1999 Assessment of Eutrophication in the Firth of Clyde: Analysis of Coastal Water data from 1982 to 1996. Marine Pollution Bulletin, Vol. 38, No 3, 222–231.

Björk, G., Liungman, O. and Rydberg, L., 2000 Net circulation and salinity variations in an open ended Swedish fjord system. Estuaries 23(3):367–80.

Björn-Rasmussen, S., 1976 Phytoplankton in an eutrophicated estuary. PhD Thesis. Department of Marine Botany. University of Gothenburg. Gothenburg. 157pp.

Carman, R. and Rahm, L. 1997 Early diagenesis and chemical characteristics of interstitial water and sediments in the deep deposition bottoms of the Baltic proper. Journ. Sea Res. 37 (1–2): 25–47.

Cato, I. 1992 Sedimentological investigations of the Bohus Coast 1990 – the coastal water monitoring program of the Göteborg and Bohus County. Sveriges Geol. Undersökning, Uppsala. 97 pp.

Cato, I. 1997 Sedimentological investigations of the Göta älv River 1995 – the water monitoring program of Göta älv and changes after 1990. 158-92 Sveriges geologiska undersökning, Uppsala.

Eriksson, E. and Peippo, J., 1975 En modell för vatten- och fosforomsättningen in Göteborgs skärgård. Vatten, 2:106-19.

Elmgren, R. and Larsson, U. 1997 Himmerfjärden, förändringar i ett näringsbelastat kustekosystem i Östersjön. Rapport 4565 Naturvårdsverket. Stockholm.

Gowen, R.J. and Bloomfield, S.P., 1996 Chlorophyll st anding crop and phytoplankton production in the western Irish Sea during 1992 and 1993. *Journal of Plankton Research*, 18 (9): 1735–1751.

Gowen R.J., Mills D.K., Trimmer M., and Nedwell, D.B., 2000 Production and its fate in two coastal regions of the Irish Sea: the influence of anthropogenic nutrients. *Marine Ecology-Progress Series* 208: 51–64.

Karlson, B. and Andersson, L. 2003 The *Chattonella* bloom in the year 2001 and effects of high freshwater input from river Göta älv to the Kattegat-Skagerrak area. SMHI Report No 32. 2003.

Lindahl, O., Belgrano, A., Davidsson, L., Hernroth, B., 1998. Primary production, climatic oscillations, and physico-chemical processes, the Gullmar Fjord time series data set (1985–96). ICES J. Mar Sci. 55, 723–729.

Marra, J., Ho, C. and Trees, C.C. 2003 An alternative algoritm for calculation of primary production from remote sensing data. Report Lamont-Doherty Earth Observatory of Columbia University, LDEO 2003-1. 27 pp.

Moksnes, P-O. and Pihl, L. 1995 Utbredning och produktion av fintrådiga alger i grunda mjukbottensområden in Göteborgs and Bohuslän. Länsstyrelsen i O-län. Miljöavdelningen 1995:10.

Nilsson, H. C. and Pihl, L. 2002 Förekomst, utbredning och biomassa av fintrådiga alger i grunda mjukbottenområden i Bohuslän under 2002. Marine Monitoring AB, Kristineberg, 2002.

Pihl, L., Svensson, A. and Nilsson, H. C. 2001 Förekomst, utbredning och biomassa av fintrådiga alger i grunda mjukbottenområden i Bohuslän under 2001. Marine Monitoring AB, Kristineberg, 2001.

Pihl, L., Svensson, A. and Nilsson, H. C. 1999 Förekomst, utbredning och biomassa av fintrådiga alger i grunda mjukbottenområden i Bohuslän under 1998–99 Marine Monitoring AB, Kristineberg, 1999.

Rasmussen, B., Gustafsson, B.G., Stockenberg, A., Ærtebjerg, G., 2003. Nutrient loads, advection and turnover at the entrance to the Baltic Sea. J. Mar. Syst. 39, 43–56.

Rodhe, J., 1996. On the dynamics of the large-scale circulation of the Skagerrak. *Journal of Sea Research* 35:9–21.

Rydberg, L. 2005 Fosforrening vid Ryaverket, GU. Reports 2005, www.bvvf.se

Rydberg, L., Aertebjerg, G. and Edler, L., 2006 Fifty years of primary production measurements in the Baltic entrance region, trends and variability in relation to land-based input of nutrients. *Journal of Sea Research* 56:1–16.

Ross, A.H., Gurney, W.S.C., Heath, M.R., Hay, S.J. and Henderson, E.W. 1993. A strategic simulation model of a fjord ecosystem. *Limnol. Oceanogr*. 38, 128-153.

Ross, A.H., Gurney, W.S.C. and Heath, M.R. 1994. A comparative study of the ecosystem dynamics of four fjordic sea-lochs. *Limnol.Oceanogr*. 39, 318-343.

Savage, C. 2003 Fate retention and effects of sewage nitrogen in a coastal ecosystem. PhD Thesis in Systems Ecology. Stockholm University, SE 10961 Stockholm Sweden.

Shiomoto, A. 1997 Size-fractionated Chlorophyll-a concentration and primary production in the Ochotsk Sea. Journal of Oceanography 53:601-10.

Seitzinger, S.P. and Sanders, R.W. 1997 Contribution of dissolved organic nitrogen from rivers. Mar. Ecol. Progr. Ser. 159:1–12.

Selmer, J. S. and Rydberg, L. 1993 Effects of nutrient discharge by river water and waste water on the nitrogen dynamics in the archipelago of Göteborg, Sweden Mar. Ecol. Progr. Ser. 92:119-33.

Selmer, J. S. and Rydberg, L. 1992 Hydrographical and hydrochemical measurements and phytoplankton counts in the estuary of River Göta älv, Sweden 1988-90. Department of Oceanography, University of Gothenburg. Red Ser. 16. 2pp.

Stevens, R. and Ekermo, S. 2003 Sedimentation and erosion in connection with ship traffic, Göteborg Harbour, Sweden. Environmental Geology 43(4):466-75.

Söderström, J., 1986 Kustvattnet I Göteborgsregionen 1982–84. En analys av tre års hydrografisk-kemiska och biologiska undersökningar. Rapport 1986:5, Naturvårdsenheten, Länsstyrelsen, Göteborgs och Bohuslän. Göteborg.