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IMPACT OF EFFLUENTS FROM LYBY WASTEWATER TREATMENT PLANT ON THE NITROGEN CONTENT OF LAKE RINGSJÖN AND RÖNNE Å RIVER, SWEDEN

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Abstract

Lake Ringsjön is situated in the south of Sweden, in the county of Skåne. It consists of three shallow lakes in series: Sätoftasjön, Östra Ringsjön and Västra Ringsjön, with the outlet in the river Rönne å. The lake has been eutrophic since the 1960s. One large point-source to the lake is the municipal wastewater treatment plant (WWTP) Lyby where most of the wastewater from Hörby is treated. In this paper, the impact of this point-source on the water quality of the lake and the outlet, Rönne å, has been modeled with regard to the nitrogen content. A model of the three basins and six of the biggest effluents was built to try to see how the system would reply to changes in the inlets and if a decrease in the nitrogen concentration at Lyby WWTP would affect the concentration in Rönne å. The model indicated that the retention time was the most important factor for the outlet concentration of nitrogen in the lake, i.e. the closer to Rönne å a point source is, the more impact it has on the outlet. The results from the model were used when Hörby municipality got permit from the environmental court to continue to discharge wastewater to the lake without nitrogen treatment.

Key words – Ringsjön, Nitrogen model, retention time, municipal wastewater treatment

Introduction: The lake Ringsjön

Lake Ringsjön has 14 tributaries, 5 out of which have Sätofta Basin as a recipient, 7 discharge into Östra Ringsjön and 2 into Västra Ringsjön. The total lake surface area is about 40 km2, out of which Sätofta Basin represents 11 %, Östra Ringsjön 52 % and Västra Ringsjön 37 %. There are two wastewater treatment plants (WWTP): one (Lyby WWTP) discharges into Hörbyån and the other (Ormanäs WWTP) directly into Västra Ringsjön. The capacity of the first is 18 000 pe with 3000 m^3/d treated on average, while the latter treats 10 500 pe and 2 500 m^3/d (see Figure 1). Extensive research on the eutrophication process of Ringsjön has been going on since 1970, resulting in a number of suggestions on how to reduce the eutrophic condition. During 1988–1992, fish, mainly roach and bream, was caught from the lake to improve the water quality. In total, 720 ton of fish were removed from the three water basins.

The main characteristics of the largest inflows and outflows of the area can be simplified as shown in Figure 2.

Process of eutrophication

It is well known that nitrogen and phosphorus are the main plant nutrients causing eutrophication and related impacts on aquatic life and water quality. Excess concentrations of such nutrients in water bodies lead to a

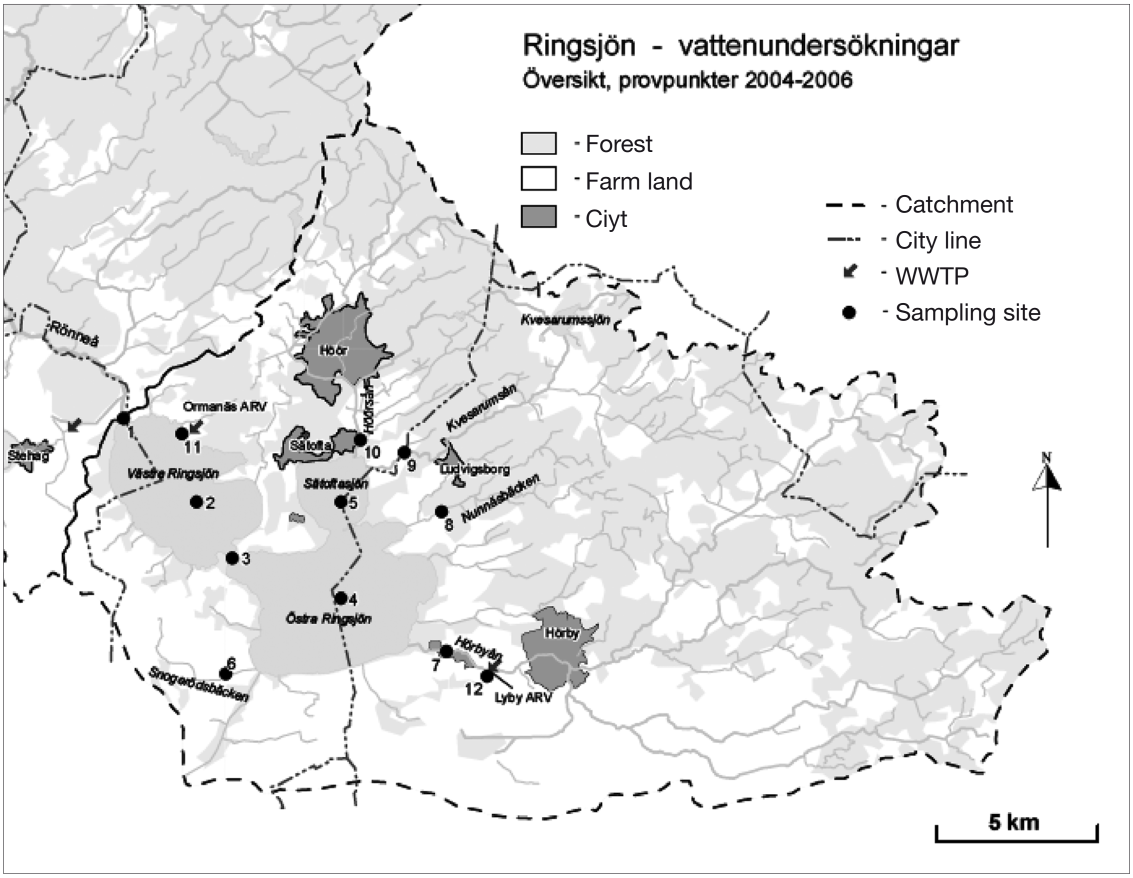

Figure 1. *Map of Lake Ringsjön in southern Sweden: the three basins with the largest streams and rivers connected to the lake.*

number of negative effects on the water bodies, such as over-nourishment, proliferation of aquatic plants (blue-green algae), reduced light penetration, depletion of dissolved oxygen in surface water, disappearance of benthic invertebrates, production of toxins which are potentially poisonous to fish, cattle and humans, and changes in biological structure. Nitrogen and phosphorus stem from various point and non-point sources such as do-

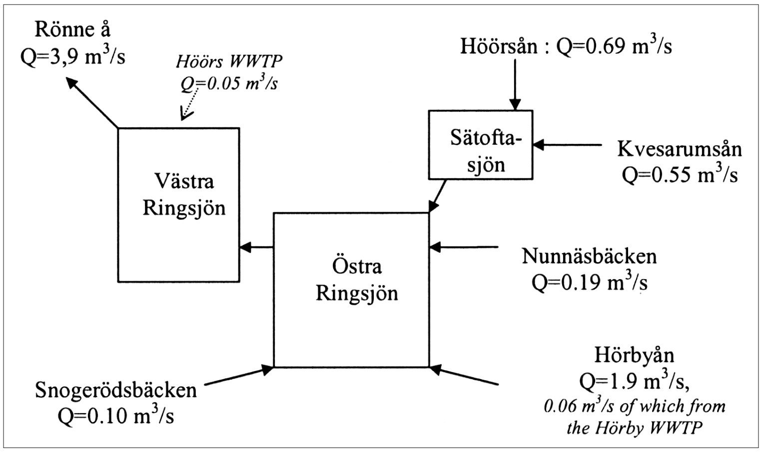

Figure 2. *Scheme of Lake Ringsjön (average flow data for 2006 [1]).*

mestic and industrial wastewater discharges, runoff from forestry and agriculture, and atmospheric deposits. However, much of the excess phosphorus load in inland waters originates from point sources, especially the two WWTPs, although inputs from agricultural land can also be significant The northern part of the catchment area is dominated by forests and wetlands, whereas there are mainly agricultural lands in the southern part. Nitrogen loading is primarily due to agricultural activity, and especially to the use of nitrogen fertilizers and manure. Nitrogen emissions from the domestic and industrial sectors are much lower than agricultural emissions. Atmospheric deposition of N that originates from fossil fuel combustion and from agricultural sources makes further addition to the nutrient loading. Much of the nitrate and phosphorous remain in the lake, settling to the bottom with the decaying mass of algae.[2]

The generally accepted upper concentration limits for lakes to be free of algal nuisance are at the time of spring overturn: 0.3 mg/l of ammonia plus organic nitrogen, and 0.02 mg/l of orthophosphate phosphorus. Lakes with annual mean total N and P concentration greater than 0.8 mg/l and 0.1 mg/l, respectively, exhibit algal blooms and nuisance weed growths during most of the growing season. The lake Ringsjön is wholly eutrophic with a mean of 1.04 mg/l of Tot-N and 0.096 mg/l of Tot-P in 2006 for instance, see Figure 3. Enell (1983) assumed the pool of N and P to be stored in the top 10 cm of the sediment layer. The storage represent many years of external loading. Ringsjön is very shallow and has a mean depth of less than 2 meters. In small lakes that are less than 10 m deep, the windy and rainy weather period in summer again frees nutrients from the hypolimnion into the surface water where the nutrients become available to plankton. The lowering of the water level of 1.5 m in the 19th century corresponded to a water volume of about 25%. The basins are not normally stratified and the areas around are flat and windexposed: that is why the wind redistribution of sediment material is effective [4]. The nutrient cycle becomes intensive.

The lake Ringsjön became eutrophicated during the 1960s and 1970s. There are four main reasons that may explain this process:

- The increase of agricultural production that required more fertilizers.
- The doubling of the population during the 1950s which had led to the increase of the nutrient content of discharges from insufficient treated water from municipal sewage treatment plants [3].
- The water level that lowered because of the water extraction for drinking water and release of downstream of Lake Ringsjön to guarantee a minimum water flow in the river Rönne å [3].
- The sewage water that came from households in rural areas as well as summer houses [3].

Leaching of the nutrients in the sediment can take place under specifically physical and biological condition. In the profound sediments, bulk P content is not exceptionally high, while the pore water phosphate concentrations, especially in Sätofta Basin, were very high, indicating large potential for phosphorus release to the water. This is also indicated by the large proportion of Fe- and Al bound P in the sediments of Sätofta Basin [4]. Although there are no direct quantifications of phosphorus release from the sediments in Lake Ringsjön, measurements of phosphorus concentrations in the water mass as well as budget calculations for the three basins clearly show a high capacity for internal loading. Nutrients buffer the ecosystem and cause a more or less pronounced inertia with respect to effects from changes in the external nutrient loading [3, 4]. The rate of phosphorus in the lake water increases during summer, contrary to what could be expected at least for dissolved inorganic phosphate. The increase of the amount of

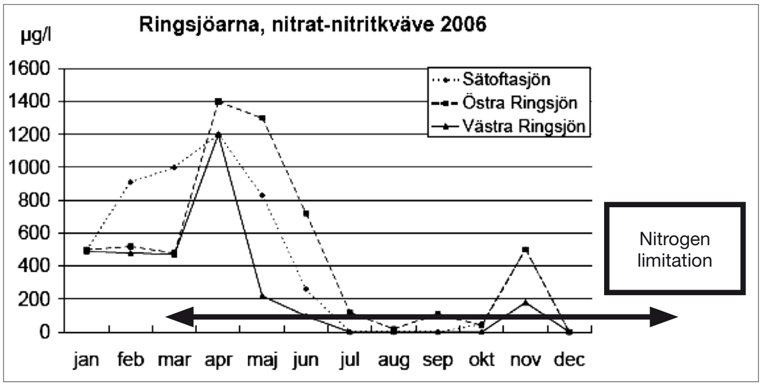

Figure 3. *Nitrate concentration in the three lakes in 2006[1].*

phosphorus in summer coincides with low external loading. This increase also coincides with an increase in pH often reaching 9.5–10 in late summer, but in the 1990s, the summer increase in P had been less dramatic, indicating reduced internal loading [3]. Even if seasonal internal loading is large, losses through outflow may still be small since internal phosphorus loading is at its maximum in summer, when the inflow and outflow of water are minimal. In the autumn, before the lake is flushed due to high late autumn and winter run-off, phosphorus concentration are markedly lowered.

Different theories on cyanobacteria blooming

Thomas listed the first explanations of why harmful algae are controlled by limiting inflow of phosphates [9]. P is present only in traces in oligotrophic lakes. An addition of phosphate alone to lake water is sufficient to increase the growth of bacteria and blue-green algae. Out of putrefied parts of organisms and sludge, nitrogenous compounds return in larger quantities than phosphates compounds in the biochemical cycle. In eutrophic lakes, nitrates are eliminated from time to time by the process of denitrification. The growth of cyanobacteria is limited by phosphorus loading. They are not nitrogen limited, because they use nitrogen fixation when phosphate is in excess. Nevertheless they grow using dissolved inorganic nitrogen (DIN) if DIN is available. Their colony form limits their access to nutrient (that is why they have a lower maximum growth rate than phytoplankton) and makes them less edible to zooplankton. Moreover, the presence of ammonium nitrogen fixation is reported to inhibit nitrogen fixation. [10–12]

Nutrient limitation is modeled with a Michaelis-Menten expression with only one limiting nutrient at the time (phosphate or dissolved inorganic nitrogen). There are seven differences that could explain the success of cyanobacteria on eukaryotic phytoplankton[6]:

1. A low TN/TP benefits to cyanobacteria, [13]
2. Low light intensity gives advantage to them,
3. They can regulate their buoyancy and thus their position in water,
4. They can resist against zooplankton grazing,
5. They require more trace elements to eukaryotic microalgae [14].
6. As they migrate from the bottom to the top, they have stored internal phosphorus from the sediment, they are thus more competitive[15].
7. Whereas nitrogen-fixing species are favored by nitrogen scarcity, non-nitrogen-fixing cyanobacteria are fostered by ammonium-nitrogen and eukaryotic microalgae by nitrate-nitrogen.

What is the real impact of Lyby discharge on the lakes?

There is no general regulation on inland WWTP discharges in Sweden, but each WWTP is considered case by case. It is always advised to decrease as much as possible the nitrogen load into a lake. Nevertheless because of the negative feed-back of nitrogen concentration on blue-green algae blooms and as the phosphorus content is the limiting factor on cyanobacteria growth, a systematic analysis would rather focus on improving phosphorus treatment. In addition of that reason, it is all the easier to extend a WWTP when there is no standard limit on the discharge.

A model was built for studying the impact on Lyby discharge on the lake outlet, the river Rönne å. This was accomplished in two steps. First the impact of the WWTP discharge on the river Horbyån was characterized. Then, a model was developed based on a total mass balance of P and N, describing to what extent the Lyby WWTP contributed to the outlet nitrogen content in Rönne å. Rönne å is highly polluted by human activity: agriculture, sewage treatment and atmosphere deposit. The model is supposed to give the outlet concentration (in Rönne å) based on concentration and flow from Lyby WWTP and the analyzed degradation rate.

The evolution of nitrogen and phosphorus concentrations along the river Hörbyån was analyzed in order to study the impact of Lyby discharge. It was compared with the change in chloride concentration in Hörbyån, since chloride is a conservative ion that cannot bind to the mostly negatively charged soil particles in the river sediments.

The mass balance is given by equation 1. For notation see Figure 4.

$$Cd = \frac{Qu\ Cu + Qo\ Co}{Qu + Qo} = \frac{Qu\ Cu + a\ Qu\ Co}{(1 + a)\ Qu} \qquad (1)$$

where $a = \dfrac{Cd - Cu}{Co - Cd}$ = dilution factor

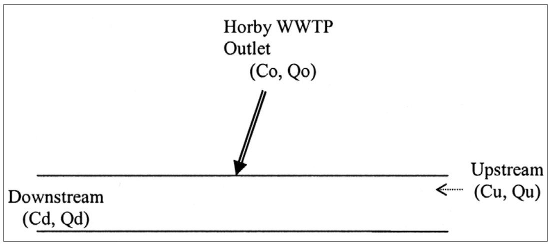

Figure 4. *Flow and concentration at the outlet from Lyby WWTP.*

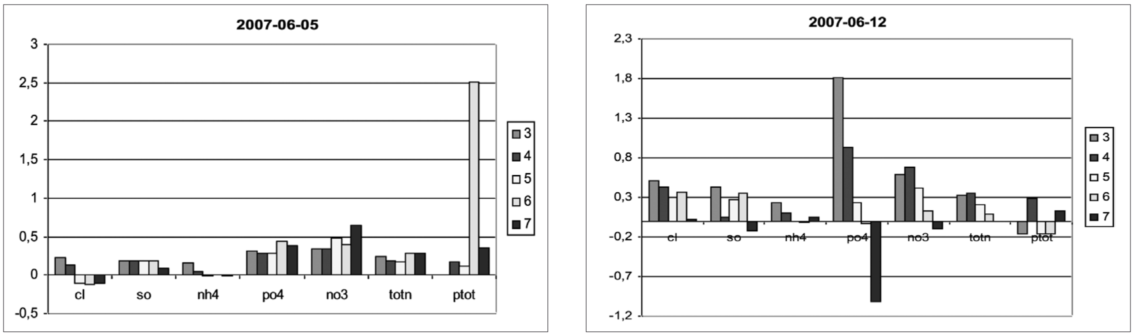

Figure 5. *Dilution factor of 2006-06-05 and 2006-06-12 for sampling points 3–7.*

Eight samples were collected twice in June in the river water of Hörbyån and analyzed on accredited laboratory for concentrations of total nitrogen, ammonia, nitrate, nitrite, orto-phosphate, total phosphate and chloride. The samples were taken at eight different spots specified on the maps, two upstream the WWTP location, one in the WWTP outlet and five downstream. On the first sampling occasion, a mill dam in the river had just been emptied the day before. It was then refilled before the next occasion of sampling. The results show an unexpected increase of nitrate, total nitrogen, orthophosphate and total phosphorus concentration just downstream the pond for the first samples (see Figures 5–9). The sampling point number 6 is situated just after the pond location and sampling point number 7 is 1 620m downstream. The emptied dam released phosphorus (see the peak of 190 µg/l for Tot-P, at no. 6) at high concentration levels, the Tot-P content then decreased until the lake whereas the phosphate concentration is quite steady after the emptied dam. Furthermore, when the pond was emptied, the nitrification after the WWTP still seems to occur contrary to the samples of the 12th June when some nitrogen reduction is observed between the dam and the lake. The owner took away the sediment layer and then refilled it with water. The dam is thus a good buffer zone between the WWTP outlet and the lake; it helps to decrease the phosphorus, phosphate, nitrate and nitrogen content that will be loaded into the lake.

However, as expected, the 12th June-samples show a decrease in concentration after the peak due to the Lyby discharge. The nitrate and nitrogen concentration almost reach their upstream value before the lake.

Model design

The water flow in the three basins of Ringsjön was modeled according to the following assumptions:

The mean depths of the Lake Ringsjön are similar to or less than the depth of the epilimnion during summer

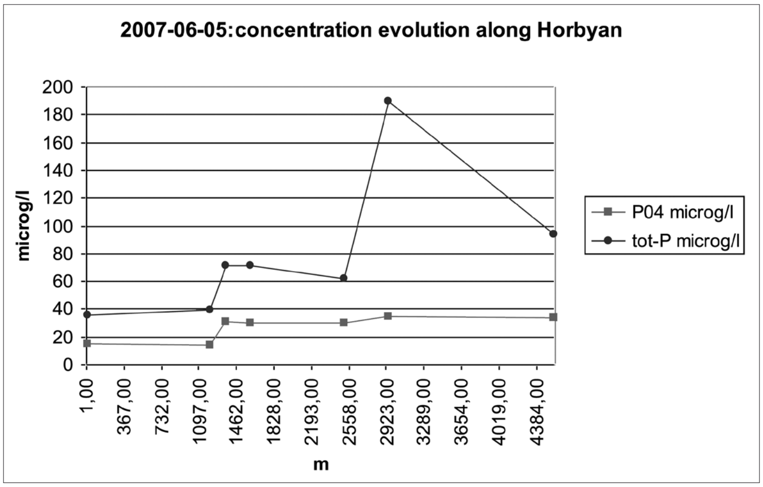

Figure 6. *Concentration evolution along Horbyan the 5th of June 2007: PO4 and Tot-P. The WWTP is located at X = 1100 m while the empty pond is located near X = 2500 m.*

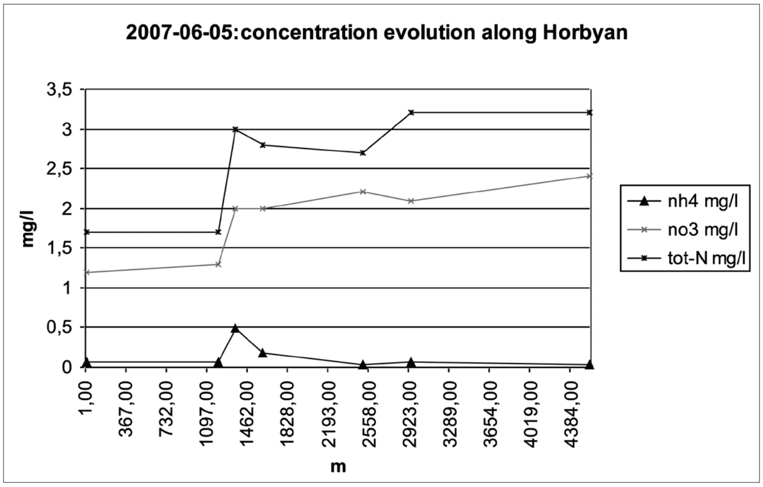

Figure 7. *Concentration evolution along Horbyan the 5th of June 2007: ammonium, nitrate and Tot-N.*

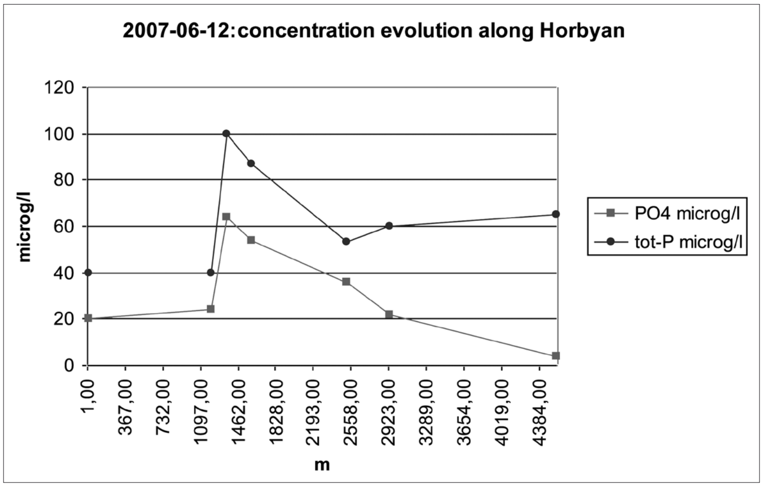

Figure 8. *Concentration evolution along Horbyan the 5th of June 2007: PO4 and Tot-P.*

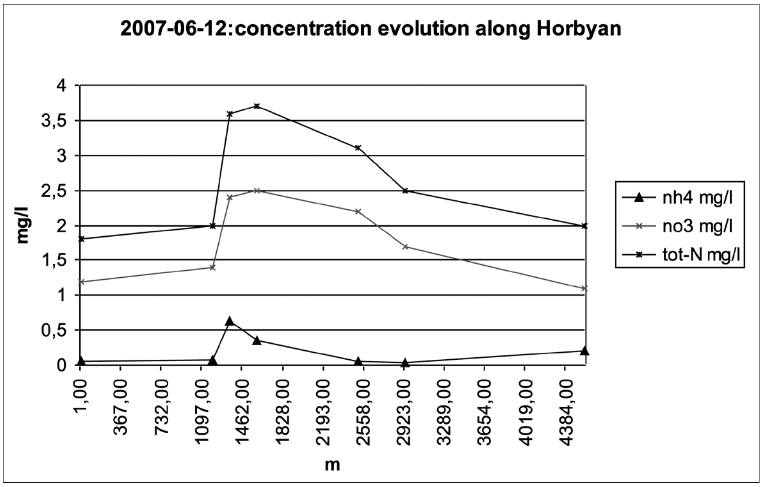

Figure 9. *Concentration evolution along Horbyan the 5th of June 2007: ammonium, nitrate and Tot-N.*

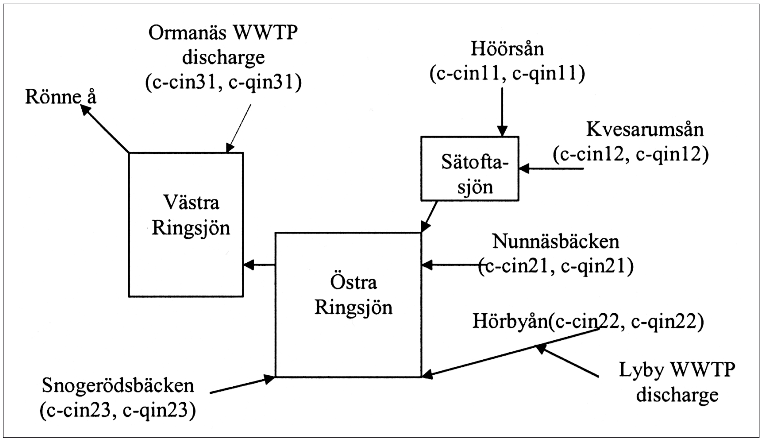

Figure 10. *The Ringsjön lake system as three CSTRs lakes-in-series.*

and the changes in nutrients (N and P) are similar for surface, intermediate and deep water, illustrating that Lake Ringsjön is a shallow lake and well mixed during most of the year.

The three water basins are regarded as continuous stirred tank reactors (CSTR), i.e. the following assumptions are made[17]:

1. There is no spatial variation of the concentration in each lake,
2. The region of varying concentration at the inlet of a lake is so small that it is negligible when compared to the entire lake,
3. The time required to achieve complete mixing in a lake is very small when compared to the residence time,
4. The lakes are all completely mixed and the outlet concentration is equal to the lake concentration.

The water basins can then be modeled as three tanks in series with inlets and outlets shown in Figure 10. Since long-term data for the small tributaries not was available for more than some few years, only the six largest were taken into account. For these, water flow and water quality data for 30 years are available.

Each lake is modeled according to equation 2, which is based on mass conservation.

$$dC / dt = ((\text{mass flux}_{in} - \text{mass flux}_{out})/\text{Volume}_{lake}) - kC \qquad (2)$$

where k = degradation rate (day^{-1}), (see Equation 3)

The water quality was modeled with 24h time-steps from 1994 to 2006, i.e. 12 years, mainly because data of the WWTPs were only available from 1994.

The nutrient retention was taken into account with an empirically derived formula for the degradation rate, k, see equation 3. The processes accounted for are sedimentation or settling of particle-bound nutrients, uptake and storage of nutrients by plankton, macrophytes and fish, and the biochemical cycling between different forms that finally may lead to disappearance from the river system via denitrification.

$$k = \frac{t}{t-h} b + d \cos[(r_t + t / 365.25)\, 2\pi] \qquad (3)$$

where k = degradation rate (day^{-1}), t = time (days) and h, b, d, r_t = empirical parameters.

The values of the empirical parameters giving the best fit for Rönne å are presented in Table 1. In Figure 11, the model calculation of total nitrogen content compared to the measured in Rönne å is presented.

The correlation was $R^2 = 0.66$ which of course is not perfect but the best that could be accomplished for the present model circumstances. The model has difficulties to follow the rapid changes in concentrations. The model can be improved if necessary but it will also make it more complex. The three water bodies of Ringsjön could be divided into more boxes, assuming that each water body not is one single well-stirred tank reactor. Using different degradation rates, *k*, for each of the water bodies, assuming that the biochemical reactions and nutrient retention vary in the different water bodies might give a more correct picture of the waters. Taking into

Table 1. *Values of empirical parameters of eq. 3 giving the best fit for Rönne å.*

Parameter	Value
h	0
b	0,0035
d	0,0002
r_t	0,76

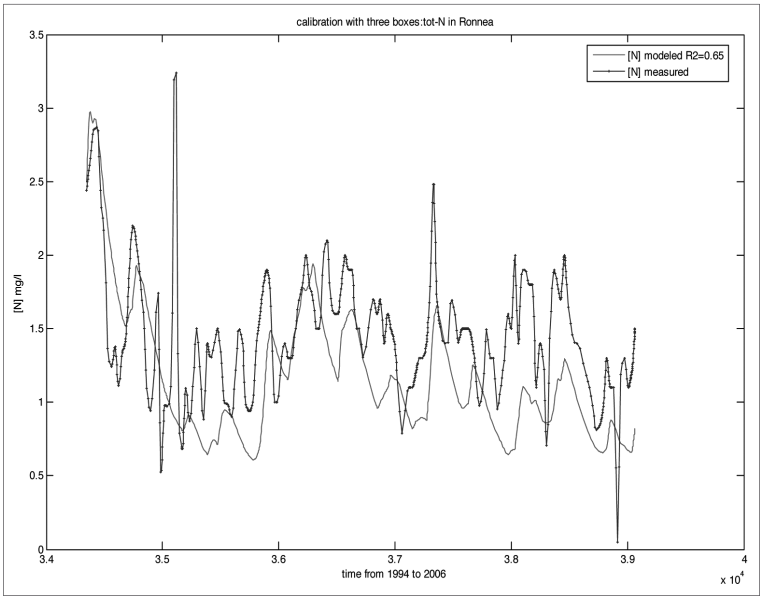

Figure 11. *Model calibration of the Tot-N in Rönne å (point 1 in figure 1) with three boxes.*

account the change of the water level with season and year. By dividing the water body Östra Ringsjön into two boxes due to topography (the volume of the first is $9.9 \cdot 10^6$ m^3 and of the second is $114.9 \cdot 10^6$ m^3), the model was recalibrated and R^2 improved from 0.66 to 0.68, see Figure 12.

The model was verified by setting up parameters and systematically examining the predictions of the model to changes in input conditions. This involves checking how the model answers to concentration changes in every effluent. The model can now be considered as calibrated; the next step is to see the impact of any concentration change in Lyby outlet on Rönne å. First, the model was used for calculating changes in the system due to changes in the effluent nitrogen concentrations from Lyby WWTP and Ormanäs WWTP.

New concentrations downstream in Hörbyån were calculated and directly fed into the model assuming different nitrogen reduction rates at Lyby WWTP, ranging from 10 to 100 %. The model could be used for calcu-

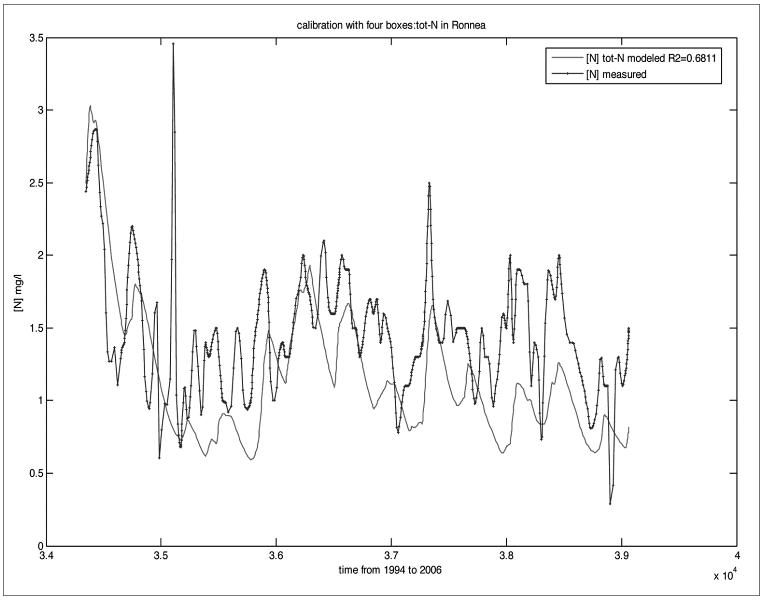

Figure 12. *Model calibration of the Tot-N in Rönne å with four boxes.*

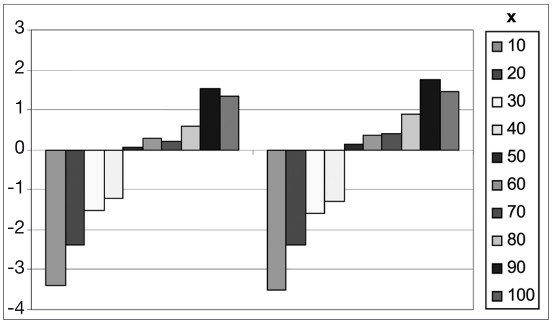

Figure 13. *Changes (%) in Tot-N concentration in the two lakes – Västra Ringsjön on the left and Östra Ringsjön on the right – resulting from different reductions (x %) of effluent concentration from Lyby WWTP.*

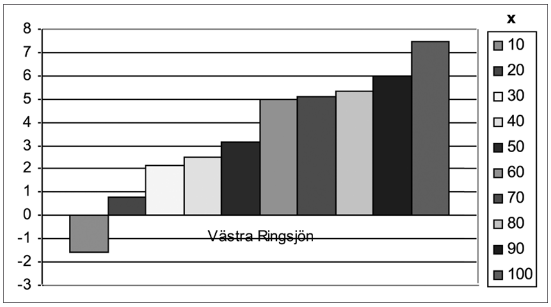

Figure 14. *Impact (%) on the lake Tot-N concentration with x % reduction of the TotN concentration in Ormanäs outlet.*

lating to what extent changes in WWTP outlet nitrogen concentration affected the concentrations of nitrogen in Rönne å. Figure 13 and 14 show the results on Rönne å nitrogen content if Ormanäs or Lyby WWTP have nitrogen treatment.

The bigger the reduction is, the bigger the changes in Rönne å and in Västra Ringsjön are. Reducing the tot-N in Lyby outlet has a smaller effect on Rönne å than reducing nitrogen in Ormanäs WWTP. The outlet of Ormanäs is close to the Rönne å river but the mass flow is little compared to Lyby WWTP, see Figures 7 and 8. The turning point when there is a decrease in the total nitrogen content concentration in the lakes comes earlier for the case of Ormanäs effluent (around a reduction of 10 %) than for Lyby (around a reduction of 40–50 %). The negative correlations can be explained by the term kC in equation 2, the implication being that when the concentration increases, so does the reduction rate.

Furthermore, the contribution of Lyby WWTP is 4 % while Ormanäs WWTP only contributes with 2 % of the total mass flow, see Figures 15 and 16.

So if the goal is to reduce the nitrogen content in Rönne å, it is more effective to start with the nitrogen treatment in Ormanäs WWTP than in Lyby. Some nitrogen will return to the atmosphere in the lake and some will settle to the bottom and be released next year, when it either will be used by the aquatic fauna and flora or be found again in the Rönne å. With only those two curves it is quite obvious that the nitrogen content in Rönne å decreases more when nitrogen reduction in Ormanäs WWTP occurs than in Lyby WWTP.

Tot-N reduction in another effluent: Snogerödsbäcken

The impact of Snogerödsbäcken on Rönne å outlet was also modeled, see Figure 17. Its catchment area is mainly agricultural, compared to the other effluents.

Treating the Tot-N in this river leads to an improvement in Rönne å and in Västra Ringsjön that is almost seven times better then when we treat in Lyby WWTP outlet, mainly for two reasons:

It contributes to 4 % of the total mass flow inlet and

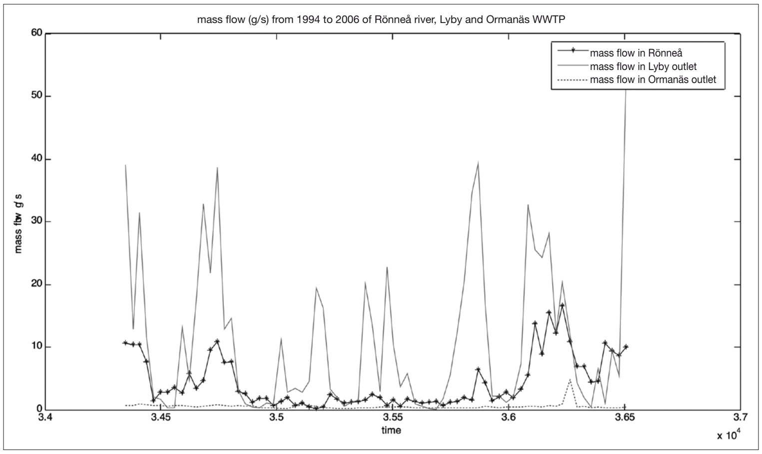

Figure 15. *Mass flow in g/s in Rönne å, Lyby and Ormanäs outlets.*

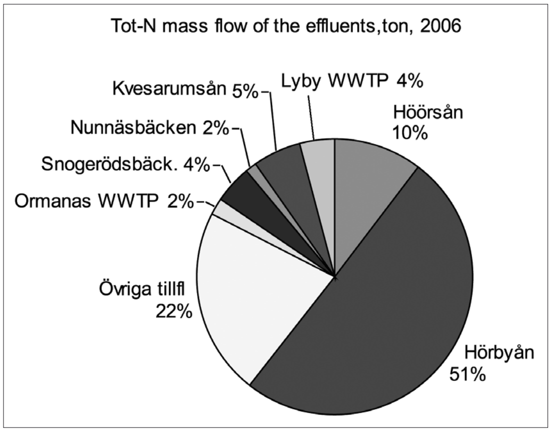

Figure 16. *Tot-N mass flow of the effluents in 2006.*

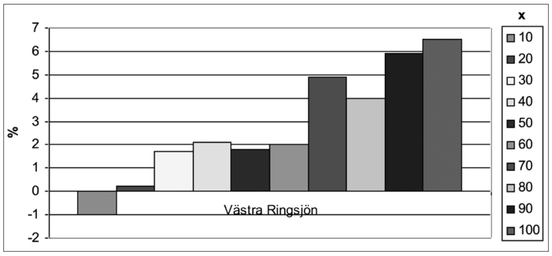

Figure 17. *Impact (%) on the lake Tot-N concentration with x % reduction of the Tot-N concentration in Snogerödsbäcken.*

above all it carries most of the Tot-N leaching from the agricultural areas, see Figure 18, and the discharge is closer to the Västra Ringsjön Lake than the discharge of Horbyan. There is then less time for denitrification to occur.

Finally, we plotted the Tot-N in Rönne å when we treated some of the nitrogen in Snogerödsbäcken, in Lyby or Ormanäs outlet and again the best value obtained is when we treated the outlet of Ormanäs, see Figure 19.

An interesting and important issue is whether the impact on Rönne å and Västra Ringsjön is more correlated with the mass flow or the retention time. If the two most upstream effluents we can notice a greater correlation with the mass flow that enters the first lake (Höörsan (10 % of the mass flow) and Kvesarumsån (5 % of the mass flow)). However, Nunnäsbäcken has a great impact with only 2 % of the total contribution whereas the Lyby WWTP is contributing to the mass inflow at the same level with a smaller impact on the lakes outlet. However, this difference can be explained by the retention time. The map of the lakes indeed shows that Nunnäsbäcken discharges in the corner of lake Östra Ringsjön close to the following corner-entrance of the third lake (see scheme of Lake Ringsjön), it has thus a smaller retention time (not the total volume of the second lake is affected by this effluent, that is also why we have divided the Östra Basin into two boxes) whereas the Horbyån effluent (in which the WWTP discharges) enters the lake Östra Ringsjön in the opposite corner of the entrance to Västra Ringsjön letting more time do denitrification to occur. Finally, for the most two downstream effluents (Snogerödsbäcken and Ormanäs), it is the retention that seems to play a major role, as Snogerödsbäcken contributes to 4 % of the total mass inflow compared to the 2 % of Ormanäs's one.

It could be unexpected that the Tot-N reduction in Höörsan had such a significant impact on Rönne å whereas it is the most upstream river; but it can yet be partly explained by its 10 % contribution to the total mass flow inlet.

Still, it is when the Tot-N of Ormanäs WWTP is treated that the bigger improvement in nitrogen content in Rönne å is observed.

The regional environmental authority in Scania de-

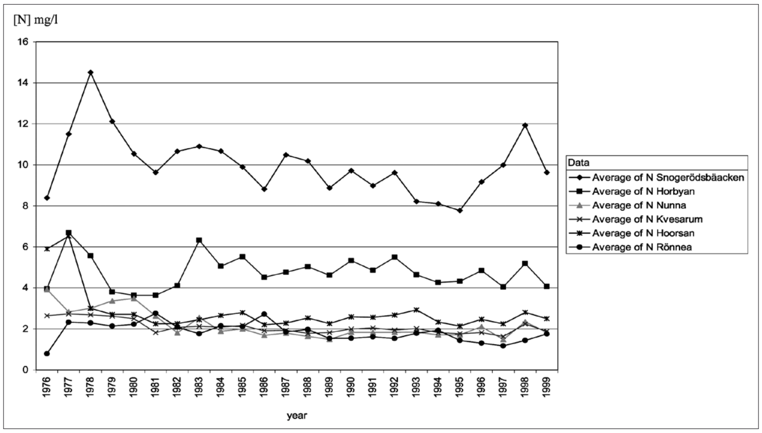

Figure 18. *Annual mean of Tot-N load in every effluent.*

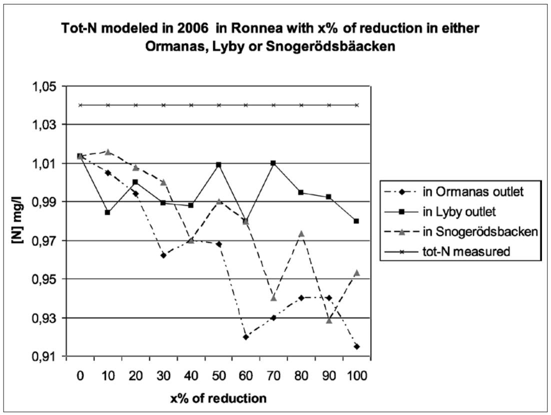

Figure 19. *Tot-N modeled in Rönne å in 2006 with x % of reduction in Ormanäs, Lyby or Snogerödsbäcken.*

manded nitrogen reduction at Lyby wastewater treatment plant because of the public notice Swedish EPA NFS 1994:7. Both the regional environmental authority in Scania and the Swedish environmental protection agency argued for the use of another nitrogen retention model (the TRK model) for estimating the nitrogen retention. Even the developers of the TRK model say that the model is not accurate for high nitrogen retention and long retention time in lakes. It is only valid for wintertime condition with low nitrogen retention, i.e. not useful for water catchments where nitrogen retention is high. The present work was used as argument when first the environmental court and later the environmental Supreme Court ruled that the TRK model was not applicable for deciding if NFS 1994:7 applies to a wastewater treatment plant. The legal view, that the TRK model is not valid for deciding whether a sewage treatment plant according to NFS 1994:7 needs nitrogen treatment, may change the requirements for some of the Swedish wastewater treatment plants. This is especially valid if they are located inland. The relatively simple model used here gave a much more correct estimation of nitrogen retention then the more complex TRK model.

Conclusion

Most of the time, the excess of nutrient coming from point sources in spring is because the WWTPs are surcharged and the network is flooded, as a consequence some of the water cannot be treated and goes directly to the natural environment through by-passes. The retention time is a very important factor in this lake system. Discharges close to the Rönne å will affect the outlet concentrations more than discharges far away. While reductions in point sources of nitrogen would contribute to achieving eutrophication reduction goals, these goals cannot be approached without significant reductions in diffuse sources such as the use of fertilizers and atmospheric deposition. Among the more interesting observation from the used model and sampling is the anomalistic nitrogen concentration (probably due to fixation) in the presence of cyan-bacteria and the extremely high nitrification rate in Hörbyån in spite of the relatively short retention time in the river.

Literature

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