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SEVENTH FRAMEWORK PROGRAMME

WISER

Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D5.2-1: Analysis of applied modeling approaches in the case studies

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CO	Confidential, only for members of the consortium (including the Commission Services)	

Content

Non-technical summary

The usage of models in lake and river basin management is often cumbersome and confusing due to the insufficient monitoring data, the opacity of complex models and to the uncertainty of predictions. Thus, clear guidance and criteria for model usage in lake management is needed.

Deliverable 5.2-1:"Analysis of applied modeling approaches in the case studies" consists of results from the evaluation of case study models as a part of efforts of Task 4 and WP5.2 to provide river basin managers, stake holders and model end users with the guidance of model usage in the prediction of implications of pressure reduction and of management and in the optimal use of models in lake and catchment management. Interaction of monitoring, modeling and management will be particularly emphasized.

The usability of empirical and mechanistic models in the prediction of *impact of catchment management strategies and of climate change* on pressures *and ecological status of* Lake Veluwe in Netherlands and Lake Pyhäjärvi in Finland were analyzed. Communication between modelers and users throughout the modeling processes were reviewed. Supplementary experience from the modeling of Norwegian lakes was gained.

The analysis was based on the model benchmark criteria and the quality assurance (QA) guidance tools developed by BMW ("Benchmark models for the water framework directive"), the criteria of models in the implementation of TMDL analysis in US (Reckhow 2001) and HarmoniQuA (HArmonised Modelling Tools for Integrated Basin Management for implementing the Water Framework Directive") projects, respectively. The criteria and the guidelines were used to assess the choosing and using of models in the implementation of WFD.

In the end, the used models, the criteria and the guidelines for model usage in river basin management were evaluated and the lessons learnt were discussed and summarized:

1. A wide variety of models and modelling approaches were tested in the case study areas.
2. Climate and management scenarios differed among the case studies.
3. The evaluation was impacted by the different development stage of models.
4. Selected model selection criteria and modelling guidelines revealed different aspects in model selection and use: a. TMDL criteria (Reckhow 2001) - policy-relevant functioning of simple probabilistic lake models b. BMW criteria (Saloranta et al 2003) - technical standard of sophisticated hydrodynamic models c. HarmoniQuA guidelines (Refsgaard et al. 2005) - communication between modeller and user during the modelling process.
5. The applicability of models to policymaking depends on a. the user interface b. the model options c. the maturity and the extent of use of the model d. the input, output and complexity of the models e. the availability of manuals, both internal as external.

6. Uncertainty assessment and model validation were not included in some models and thus, the communication of them was lacking.
7. Simple probabilistic lake models proved to be specifically relevant in decision making due to the uncertainty estimates of predictions.
8. There is the shortage of technical and interactive guidelines of communication between modeller and user and of formalized review process. Obviously, modelling process should be developed in these respects.

Introduction

Background

The projected climate change can have a significant impact on the thermodynamic, chemical and biological properties of lakes. However, the response of a lake system to changing climate is individual and in many ways dependent on the lake type and characteristics. In addition to the direct climate impact on a lake, changes in external forcing factors, such as water discharge and nutrient loading from the river basin upstream of the lake, will determine the ecological status of a lake. Due to the complex interaction of aforementioned factors mathematical models are needed to estimate individual and combined effects of these factors. What is more, the usage of models in lake and river basin management is often cumbersome and confusing given the insufficient monitoring data, the opacity of complex models and the uncertainty of predictions. Thus, clear guidance and criteria for model usage in lake management is needed.

Deliverable 5.2-1:"Analysis of applied modeling approaches in the case studies" consists of results from the evaluation of case study models as a part of efforts of Task 4 and WP5.2 (Figure 1) to provide river basin managers, stake holders and model end users with the guidance of model usage in the prediction of implications of pressure reduction and of management and in the optimal use of models in lake and catchment management. Interaction of monitoring, modeling and management will be particularly emphasized.

We had two case studies i.e. Lake Veluwe, Netherlands and Lake Pyhäjärvi, Finland. Both lakes are shallow and eutrophied clear-water lakes with moderate to high alkalinity from Northern and Central Europe. Models were used to assess the impacts of climate change and catchment management measures on ecological status. Based on this experience and earlier experience, the guidelines on the use of different modeling approaches for the design of programme of measures will be developed in the Task 5 of WP5.2.

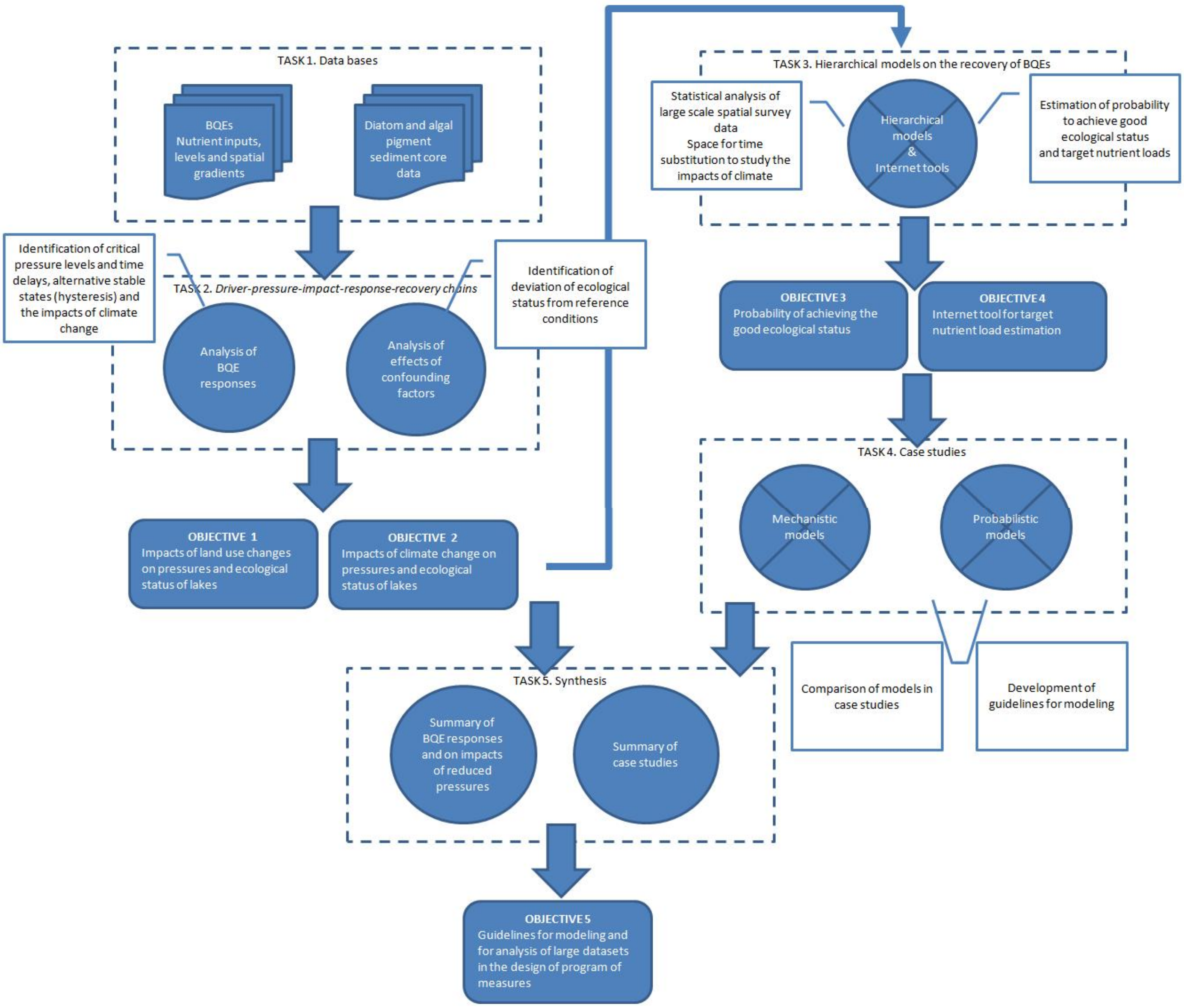


Figure 1: Workflow of WP5.2

Objectives

The goal of this deliverable was to provide WP5.2 with an analysis of usability of empirical and mechanistic models in the prediction of *impact of catchment management strategies and of climate change* on pressures *and ecological status of* Lake Veluwe in Netherlands and Lake Pyhäjärvi in Finland (Task 4 – Figure 2). Communication between modelers and users throughout the modeling processes were also analyzed. Supplementary experience from the modeling of Norwegian lakes was analyzed.

The analysis was based on the model benchmark criteria and the quality assurance (QA) guidance tools developed by BMW ("Benchmark models for the water framework directive") and HarmoniQuA (HArmonised Modelling Tools for Integrated Basin Management for implementing the Water Framework Directive") projects, respectively. The criteria and the guidelines were used to assess the choosing and using of models in the implementation of WFD.

In the end, lessons learnt were discussed and summarized and the used criteria and guidelines for model usage in river basin management were re-evaluated.

In model applications, we addressed:

1. *Land use*: Assess the impacts of catchment management strategies on BQEs and ecological status of lakes. The pressures addressed include eutrophication and hydro-morphological alterations (mainly lake level regulation).
2. *Climate change*: Assess the impacts of climate change on ecological status of lakes, focusing on impacts on the thresholds used to set the good/moderate class boundary for the various BQEs.
3. *Uncertainties*: Assess the uncertainty and risks of failing to achieve and maintain the good status objective under various climate change scenarios.
4. *Target nutrient loads*: Develop and apply models and tools that can be used for estimating the required pressure levels to achieve good ecological status.
5. *Guidelines*: Develop guidelines for using case-specific mechanistic models/empirical analysis of large datasets for designing the program of measures under climate change.

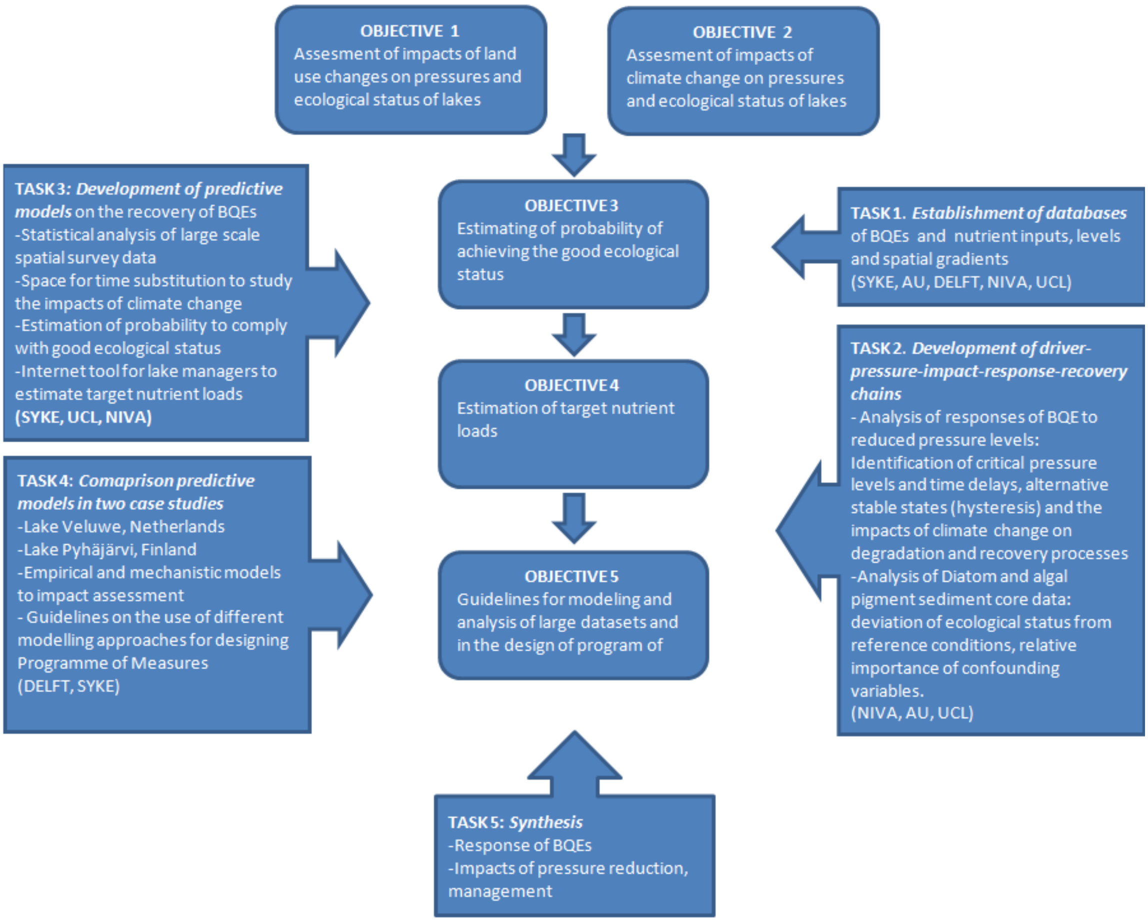


Figure 2: Objectives of WP 5.2.

MATERIALS – Description of case study areas

Lake Säkylän Pyhäjärvi

<u>River basin</u>

Pyhäjärvi is a large (155 km^2) but shallow mesotrophic lake in southwest Finland (Figure 3). Two major rivers, Yläneenjoki and Pyhäjoki, discharge into Lake Pyhäjärvi. Yläneenjoki river basin is considerably larger (233 km^2) than that of Pyhäjoki River (78 km^2). In addition, four main ditches (catchment areas between 6-20 km^2), located in the nearby catchment area, flow directly into the lake. The river mouths of both Yläneenjoki and Pyhäjoki have been regularly monitored and hence the loading estimates to the lake are easily available. On the contrary, the diffuse load from direct, nearby catchments is much more difficult to assess due to the contingency of water level, sediment and nutrient measurements.

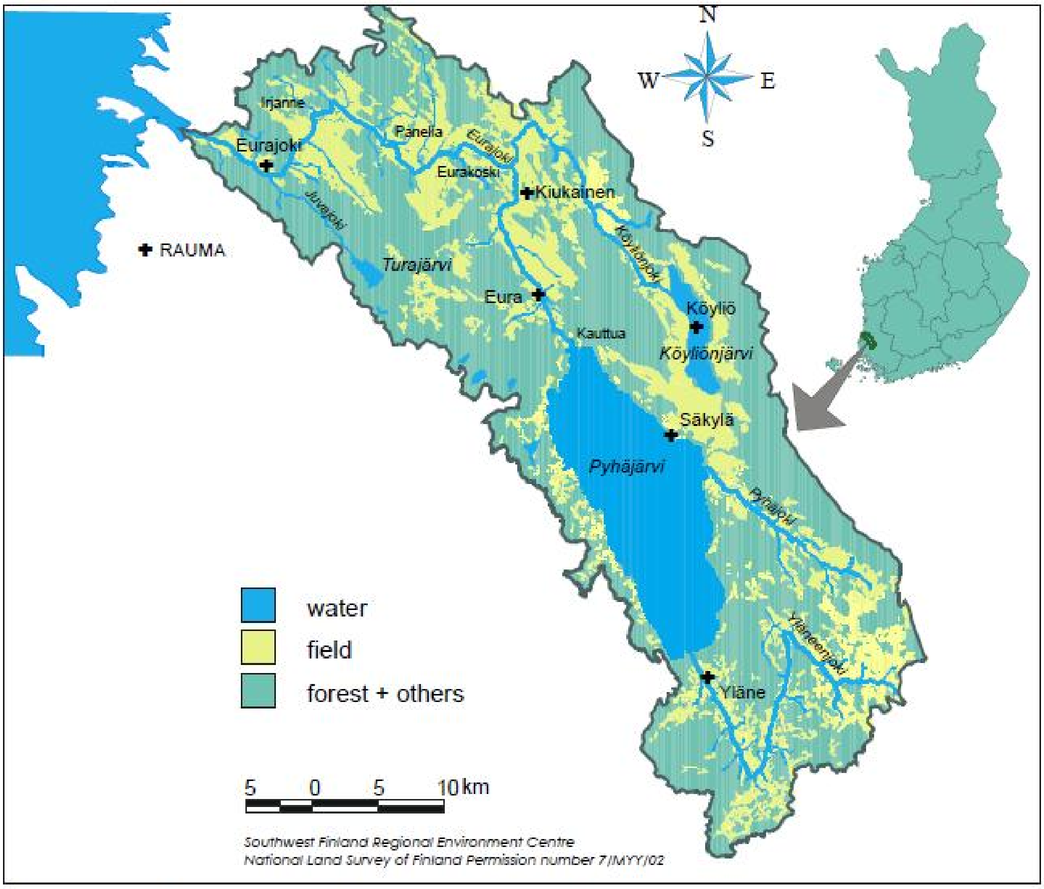


Fig I. Säkylän Pyhäjärvi (154 km^2), together with its 306-km^2 catchment, serves as the pilot area of the project. The whole Eurajoki catchment (1336 km^2) covers also areas between Lake Pyhäjärvi and the sea.

Figure 3: Lake Lake Säkylän Pyhäjärvi is situated on River Eurajoki catchment.

The soils in the Yläneenjoki river valley are mainly clay and silt, whereas tills and organic soils dominate elsewhere in the catchment. Forests and natural wetlands cover 65% of the catchment the rest being agricultural (34%) and urban (1%) areas.

Long-term (1965–1990) average annual precipitation is 628 mm yr^{-1} of which approximately 11% falls as snow. Average discharge (1991–2000) measured in the Yläaneenjoki main channel is 2.2 m^3s^{-1} and 0.7 m^3s^{-1} in Pyhäjoki, respectively (Hyvärinen, 2003). The highest discharges typically occur during the spring and late autumn months.. The portion of groundwater flow is not measured but according to typical annual water balances groundwater accounts for less than 20% of annual rainfall.

Agriculture in the catchment area consists mainly of cereal production and poultry husbandry (Figure 3), and is relatively intensive for Finland (Pyykkönen et al., 2004). According to surveys performed in 2000–2002, 75% of the agricultural area is planted for spring cereals and 5–10% for winter cereals (Pyykkönen et. al. 2004).

The Yläneenjoki catchment is responsible for 53–57% of the external phosphorus loading into Pyhäjärvi, while Pyhäjoki for 10–12%, respectively (Ekholm et al. 1997, Ventelä et al., 2007). Atmospheric deposition of P is 20 %.

Subsurface drainage is applied for major part of the agricultural soils in Yläneenjoki catchment. The recommended drain depth is circa 1.2 m and drain spacing around 20 m depending on hydraulic conductivity and drainage demands. All farmers in the Yläneenjoki area follow the recommended fertilizer levels which were derived from studies of the Finnish Agri-Environmental Programme (Palva at al. 2001). Both winter-time vegetation cover and reduced tillage have become more common during the recent years.

Erosion rates in Yläneenjoki area vary owing to local natural conditions and management practices. Agricultural field plots and catchments dominated by agriculture produce higher erosion rates than forested areas. The clayey soil of the south-western coastal plains is more susceptible to erosion than inland predominantly sandy and till soils (Tattari and Rekolainen, 2006).

The regular monitoring of water quality of the river Yläneenjoki has been started as early as 1970s. The nutrient load into the Lake Pyhäjärvi via the river has been calculated based on monthly averages from manual water sampling results (ca 12 samples per year) and daily water flow records obtained by site-specific head-discharge relation at one point, Vanhakartano (Kauppila and Koskiaho, 2003). Furthermore, water quality has been monitored on a monthly basis in three additional points in the main channel in the 1990s, and in 13 tributaries flowing into the river Yläneenjoki.

Lake Pyhäjärvi

Hydrological and hydrobiological research of the lake has been ongoing for decades in the environmental administration and in the University of Turku. Water chemistry and hydrology have been monitored from the 1960s to date. Intensified monitoring of nutrients and plankton community started in 1980, and the state of the lake since then has been summarised by Ventelä et al. (2007). Pyhäjärvi yields an unusually high fish catch (Sarvala et al., 1998).

Increased eutrophication became a major concern in the late 1980s: between 1970 and 1992 the nitrogen (N) concentration in the lake increased by 30 % (Ekholm et al., 1997) and the P concentration doubled, but both have decreased since then (Ventelä et al., 2007). In the Eurajoki catchment as a whole (Figure 3), total N load is estimated as 917 tonnes N a^{-1} of which, on average, 583 tonnes N a^{-1} reaches the sea; i.e. average N retention is 36% (Lepistö et al., 2006). Blue green algae blooms in the Lake Pyhäjärvi have been observed frequently in the 1990s. According to sediment studies, the lake productivity started increase in the 1950s in response to intensified cultivation and use of industrial fertilizers. Pyhäjärvi has an unusually high catch of fish, and the average annual catch has been estimated to be even three times higher than the average catch in Finnish lakes (Sarvala et al., 1998).

Figure 4 illustrates eutrophication of Lake Pyhäjärvi: changes in nutrient concentrations, Chl-a, and Secchi depth in Pyhäjärvi during the past 25 years (Ventelä et al., 2007). Total nitrogen (N) concentrations have varied from 364 to 534 µg l-1 during 1980–2005, increasing up to 1990, and slightly decreasing since then . Total P concentrations in the lake noticeably increased in the 1980s and 1990s. During the past few years total P has been decreasing, probably partly because of lower external loading due to a dry period and water protection actions in the catchment, and partly because of efficient fish removal. Also, Chl-a concentrations have been decreasing during the past few years due to lower phytoplankton biomasses (Fig. 4). Secchi depth has varied from 2.4 to 3.9 m during the open water season during 1980–2005 (Fig. 4). During the past 10 years Secchi depths have been lower than in the earlier years and seldom >3 m. (Ventelä et al., 2007).

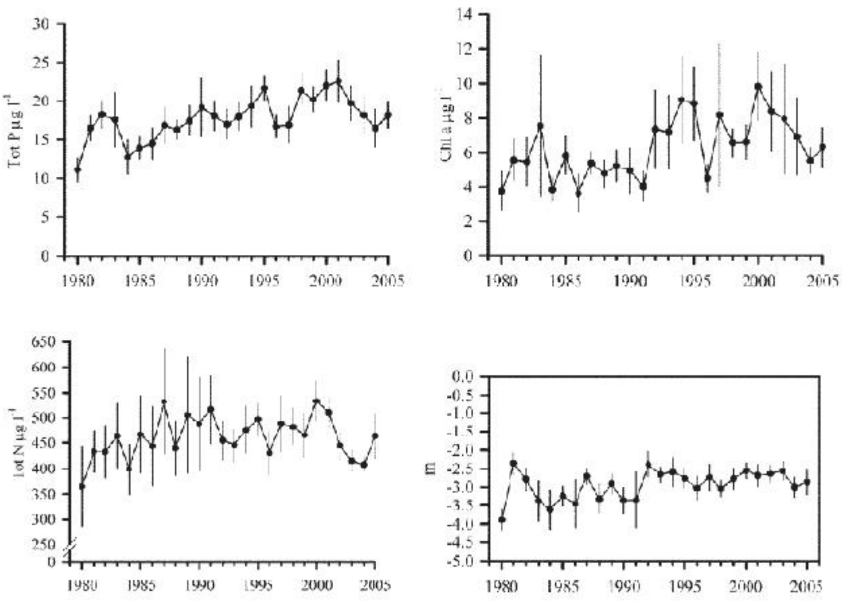


Fig. 2. Mean total P, total N and Chl-a concentrations and Secchi depth in Pyhäjärvi during the open water season (May-Oct) in 1980-2005. Vertical lines denote 95% confidence limits (Ventelä et al., 2007).

Figure 4: Mean total P, total N and Chla-a concentration as well as Sechhi depth in Pyhäjärvi during open water season (May-Oct) in 1980.2005. Vertical bars denote 95% condidence limits (Ventelä et al. 2007)

Internal processes contribute considerably to eutrophication of Pyhäjärvi (Ekholm et al., 1997). Total phosphorus concentrations in Pyhäjärvi rise in the late summer, although the external load is small at that time. It has been estimated that 2180 kg P a^{-1} (12, 5 % of the total phosphorus load) was released from the bottom sediments in 1997-1998. The annual nitrogen load from the sediment is estimated to be 106 000 kg N a^{-1}, which is 25 % of the total nitrogen load. Phosphorus release is due to both low oxygen concentration and to the resuspension of bottom sediments during high wind periods (Huttula 1994, Ekholm et al., 1997).

Sarvala et al.(1998) have shown that inter annual variations of the chlorophyll and P concentrations in Pyhäjärvi are associated with the changes in the total biomass of planktivorous fish. Strong stocks of planktivorous fish are accompanied by depressed zooplankton biomass, the practical disappearance of larger cladocerans, and high chlorophyll levels. One-third of the total variation in chlorophyll is attributed to changes in zooplankton biomass, and another third to changes in phosphorus concentrations. The commercial fishery keeps vendace stock, the dominant planktivore in Pyhäjärvi, small and water quality effects moderate. That is why Pyhäjärvi has been biomanipulated by commercial fisheries. After the reduction of vendace stock in the beginning of 1990s, however, other fish stocks increased deteriorating the water quality. During the 1990s the fishing of smelt, roach, ruffe and small perch has been subsidized and this fishing has successfully reduced smelt and roach (Sarvala et al., 2000).

Management actions

From the early 1990s on, Pyhäjärvi restoration project funded by local industry, municipalities, associations and by the state, have been testing and implementing several management actions in the lake and in its watershed. Southwest Finland Centre for Economic Development, Transport and the Environment (earlier Regional Environment Centre) has the main responsibility of the management planning and environmental monitoring of the lake. Finnish Environmental Institute (SYKE) has the responsibility in national level, to research and to monitor hydrology, chemistry and biology of surface waters, to develop and integrate modeling and management activities necessary for the improvement of usability and ecological status of surface waters. Pyhäjärvi Institute, located at lake shore in Eura since 1989, has been acting towards improving the water quality and ecological status of Lake Pyhäjärvi. Typically it works with local actors, maintaining ecological research and informing and educating partners. Departments of Biology and Geology have also had very active role in Lake Pyhäjärvi research.

The main management options applied in Pyhäjärvi area have been reduction of external nutrient load and management of fisheries (Ventelä et al., 2007). They were started by the Southwest Finland Regional Environment Centre (SFREC) since 1991. Farmers and fishermen have participated in these efforts in the water protection projects, Nutrient load abatement has a direct impact (often with a long delay) on algal blooming, but it seems that this option may become expensive. Fisheries management has proved to be a cost-efficient additional tool for the minimization of algal blooms. Intensified fishing in 2002–2004 reduced cyanobacterial blooms and restored the total phytoplankton biomass to the level of the early 1990s (Ventelä et al., 2007).

In recent years, several still various water protection measures have been taken in the catchment: fifty experimental sites have contributed to actions such as filtering ditches and sand filter fields, utilizing lime or other materials for binding of phosphorus, series of small dams, small chemical treatment units to treat waters from dairy and cattle farms, sewage water treatment of rural houses and village planning as a new method for promoting environmentally sound development of the rural areas.

Management measures studied in modelling were increased vegetation cover during winter, changes in and manure management, buffer zones, wetlands and bio manipulation (fisheries management, Figure 5).

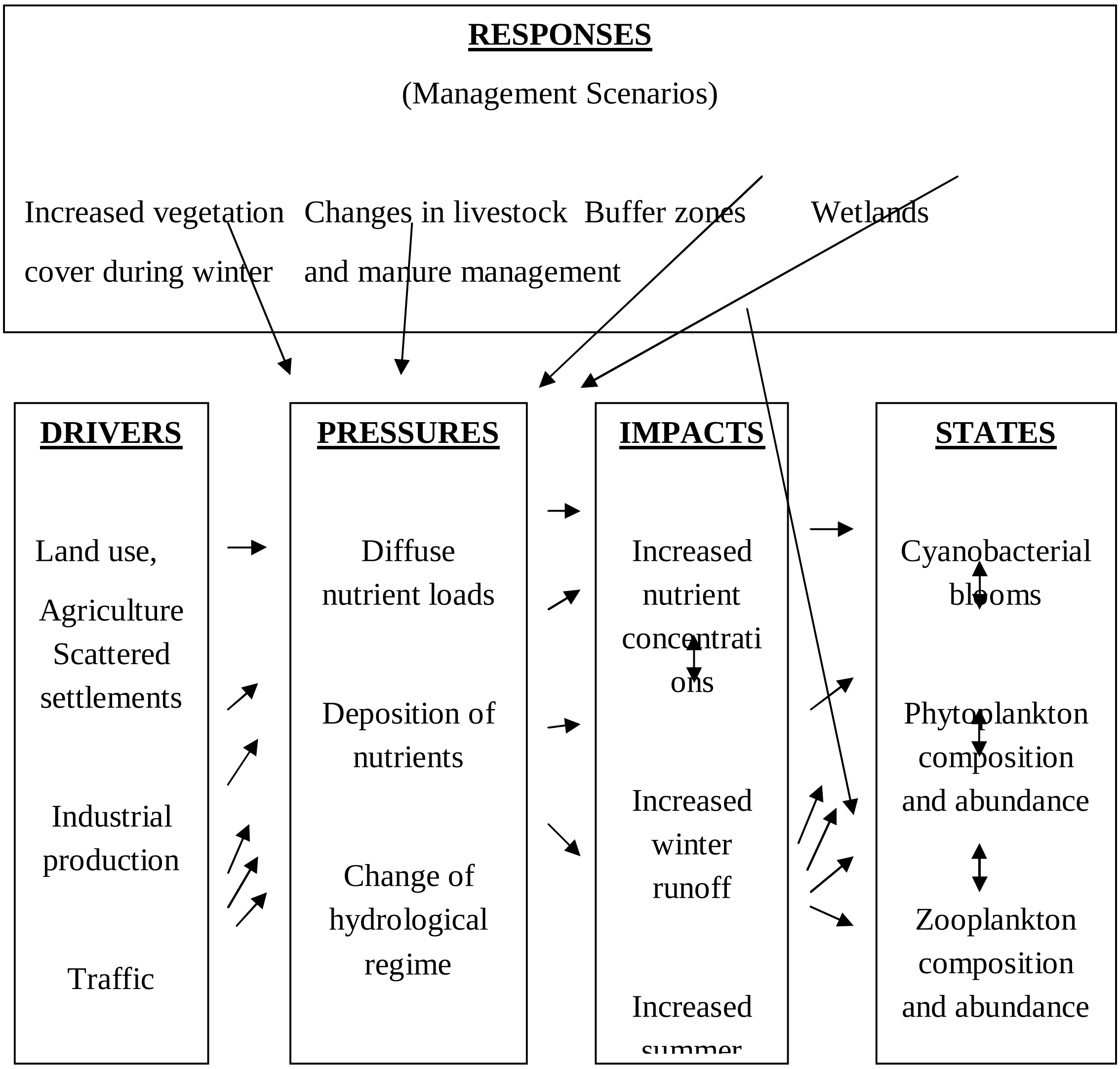


Figure 5: DPISR- chain of Lake Pyhäjärvi.

Lake Veluwe

Lake Veluwe (52°23’39 N, 5°41’41 E) is situated in the Netherlands and is part of the so-called “Veluwerandmeren”: a chain of shallow lakes that came into being due to land reclamation for the “Flevopolder” in 1957 (see figure 6). Lake Veluwe is about 3250 ha, and has two different profiles: a sandy, shallow part of about 1 meter depth bordering the old land and a clayey, deeper part of the lake, varying from 3 to 5 meters and borders the new land. Moreover, the two parts differ also in water flow: in the shallow part, seepage occurs from the old land while in the deeper part infiltration takes place. Additionally, the lake receives water from 17 streams (Van der Molen et al., 1998), a water purification plant and a pumping station. The streams are discharging from the old land where the land use is 54% recreational, 34% nature, 10% urban and 2% agriculture Lake Veluwe is connected to Lake Drontermeer and Lake Wolderwijd, of which the former is the most important discharge route (Smits, 2005). Lake Veluwemeer is of both national and international importance, as it contributes to the conservation of specific macrophytes, macrofauna and birds. To warrant proper protection of these species, it has been assigned as a bird and habitat directive area (Postema et al., 2005).

Figure 6: Map of the "Veluwerandmeren" and "Flevopolder" (Google Earth, 2010).

After the creation of Lake Veluwemeer, the lake was clear and the rich plant life was dominated by *Characeae* (Van der Molen, 1998). These macrophytes supported amongst other things, a large bird population (Leentvaar, 1961, 1966). However, in the late sixties a steady supply of nutrients from the old land and internal phosphate loading forced a switch from the clear, chara dominated state (Bak et al., 1998) to a turbid, algae and large bream (Abramis brama) dominated state (Noordhuis, 1997). The algae blooms were year round and dominated by Planktothrix (Berger, 1987). With the disappearance of the macrophytes also the bird population became severely reduced because of food shortages (Noordhuis et al., 1998).

To recover the clear state of the lake, the following measures were taken:

- Flushing the lake by pumping station Lovink with nutrient poor water. This flushing started in 1979 and was meant to flush only in winter period. However, after a few years the flushing got a more continuously character.
- In the winter of 1979, the dephosphatising efficiency of the sewage water treatment plant of Harderwijk (STP) was increased with circa 85 %.
- Between 1990 and 1995 biomanipulation, by means of catching breams, was conducted.
- In 2002 the obstruction between Lake Veluwemeer and Lake Wolderwijd was removed to increase the discharge of nutrients from Lake Veluwemeer towards Lake Wolderwijd
- In 2003 the third purification step for the STP Harderwijk was conducted as a mean to keep the nitrogen concentration not too high (Smits, 2005).

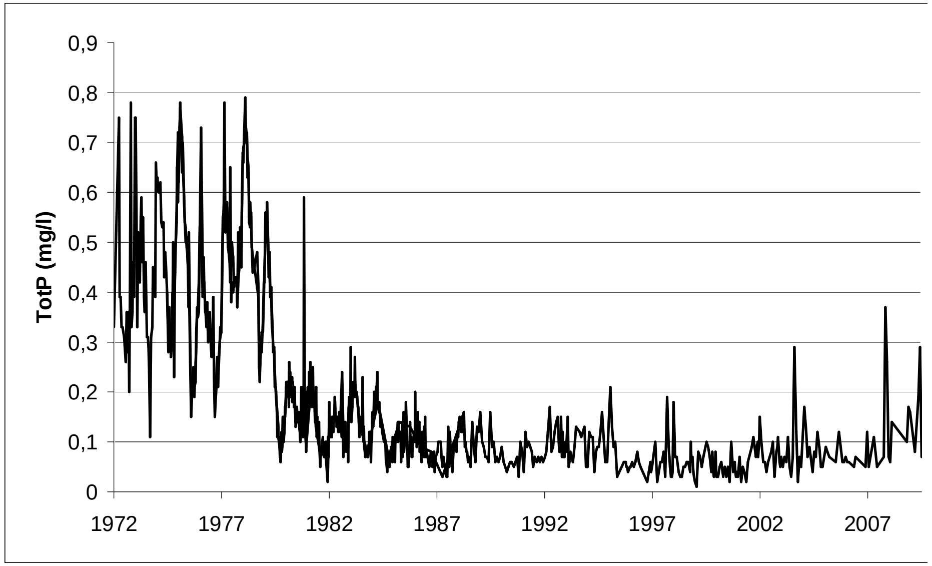
These measures should eventually lead to the following long term goals (Table 1).

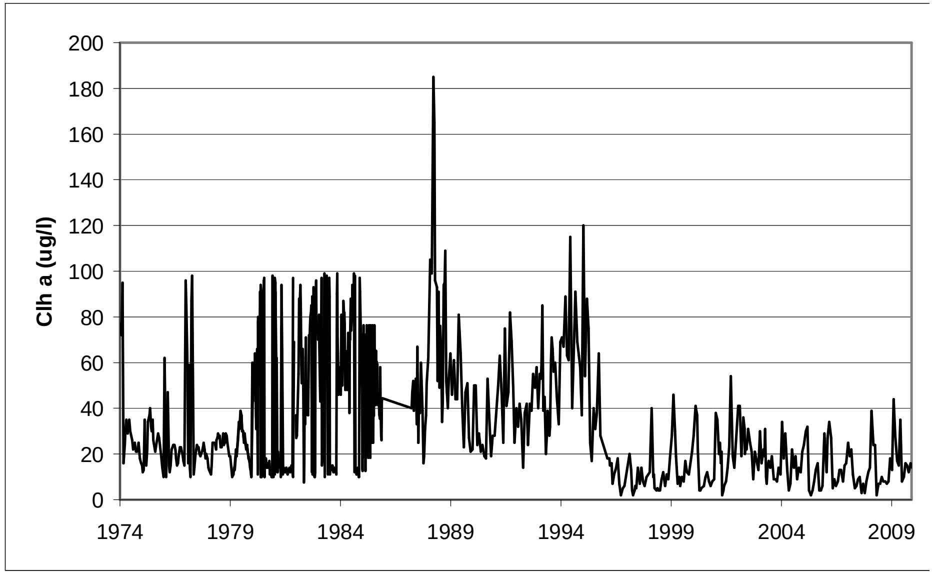
Table 1: Overview of long-term goals for lake Veluwemeer (Postema et al., 2005).

	Goal
Total P (mg/l)	0,06
Total N (mg/l)	1,00
Chlorofyll-a (µg/l)	10
Transparancy (m)	1,00

Until now, the long term goal of 0.06 mg/l for total P had been reached in 1996 until 2003. For total N, the long-term goal has never been reached, only the national minimum quality of 2,2 mg/l has been reached for the last couple of years. In addition, the chlorophyll-a concentrations are not reached, however, the water quality is well below the national water quality of 100 µg/l, but cyanobacteria were no longer problematic (Postema et al., 2005). After 1996, the overall lake became clear, sometimes reaching transparencies of 1 meter or more for longer periods (Ibelings et al., 2007). However, this is not the standard state.

From 1985, Chara has started to recolonize the lake, but it was a gradual process and a clear state in the shallow part coexisted with a turbid state in the deeper part of the lake. From 1994 onwards, Chara expanded rapidly together with a stepwise reduction of fish biomass and a reintroduction of the zebra mussel (Ibelings et al., 2007). During dredging activities in 2002, chara coverage of the lake decreased, with minimum values in 2003. However, when dredging activities were finished, Chara coverage increased (Ibelings et al., 2007) (figure 8).


TotP (mg/l)
0,9
0,8
0,7
0,6
0,5
0,4
0,3
0,2
0,1
0
1972
1977
1982
1987
1992
1997
2002
2007


Clh a (ug/l)
200
180
160
140
120
100
80
60
40
20
0
1974
1979
1984
1989
1994
1999
2004
2009

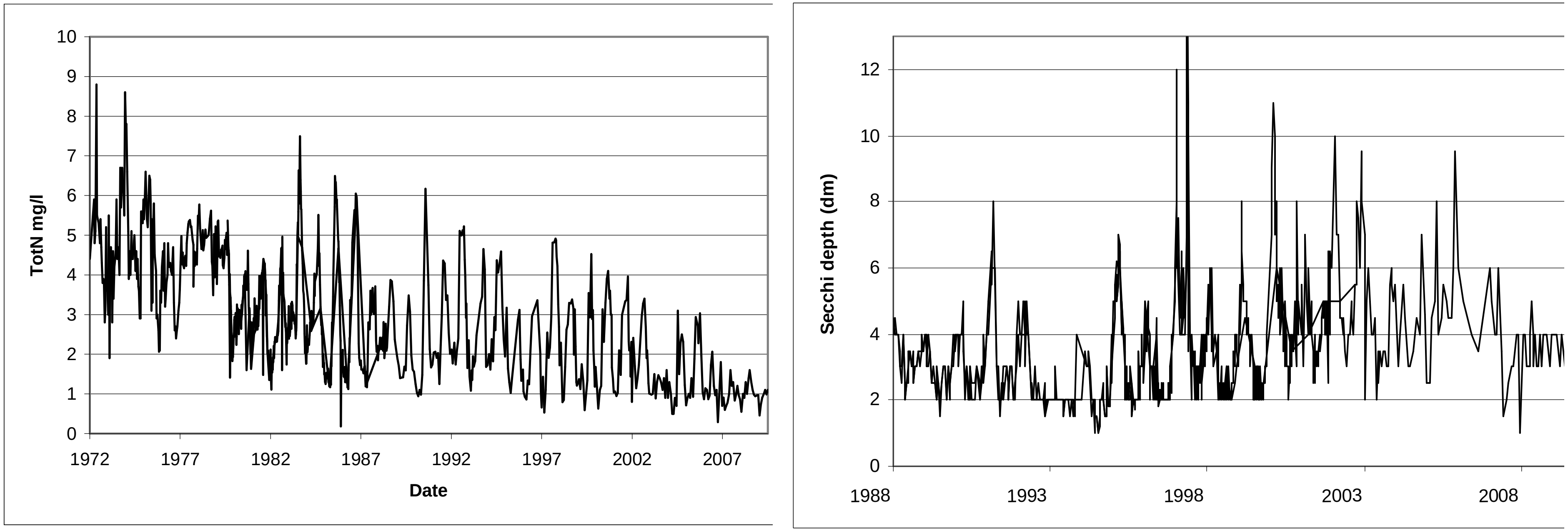


Figure 7: Measured, total P, total N, Chl-a concentrations and Secchi depth for Lake Veluwe for the period (of data is available) 1972-2009 (www.waterbase.nl).

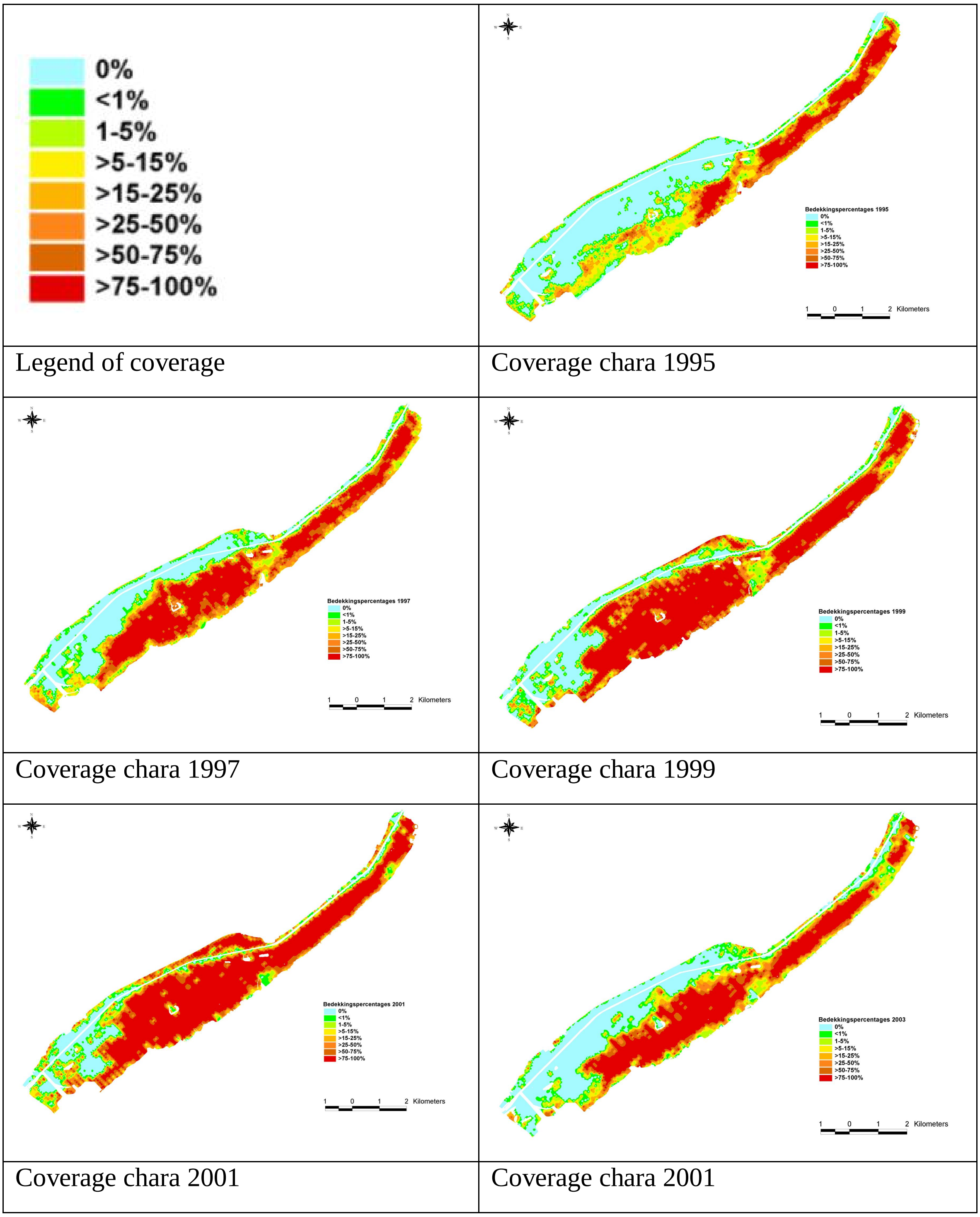


Figure 8: Overview of Chara coverage in Lake Veluwemeer for the period 1995-2001.

Nowadays, Chara is present in the system in reasonable large quantities, but is remains a fragile equilibrium. There are three threats that are seen as plausible for returning to a turbid state: 1) transparencies below 0.5 m: cyano-bacteria like Planktothrix can become dominant again, 2) phosphate concentrations above 0.01 mg/l: a gradual decline in chara species is expected and 3) a reduction in chara coverage by means of mowing due to recreational and/or shipping purposes, which is possible regarding the amount of functions the lake fulfils (Postema et al., 2005).

Norway

Previous studies have used both historical datasets (e.g. Blenckner et al., 2007) and batch model simulations (e.g. Fang and Stefan, 2009) to analyze patterns of overall lake response (rather than individual lake response) to climate change. In this study the latter batch model simulation method was merged into the MyLake model to simulate impacts of climate and river basin management changes on 181 Norwegian lakes. This set of lakes is based on the Euregi lake dataset, from which lake information for model setup was obtained (latitude, longitude, altitude, surface area, maximum depth, and water transparency (secchi depth)).

The Euregi lake dataset results from the regional eutrophication survey in 1988 (described in Lyche-Solheim et al. 2004). The dataset includes quantitative analyses of ca. 2550 samples from more than 400 lakes. The locations are selected in order to cover the broadest possible gradient of human influence. Parameters that typically represent eutrophication (chlorophyll a and total phosphorus) range over two orders of magnitude in this dataset. The dataset contains more oligotrophic than eutrophic lakes. Nevertheless, eutrophic lakes are probably overrepresented regarding the proportion of area covered by these lakes. This selection criterion implies that the selection is not completely representative for Norway geographically. All selected lakes are sampled at least four times, while a lower number are sampled more than 20 times. Almost 75% of the lakes are clear-water lakes, of which the majority is calcium-poor lakes. The remaining 25% are humic lakes; this group has equal proportions of calcium-poor and calcium-rich lakes.

The set of lakes considered for the simulations consisted of 345 lakes from the Euregi dataset with maximum depth larger than 4 m. For each lake the closest grid point in the CRU climatology was used as starting point, and the altitudes of that point and the neighboring eight grid points were compared to the actual altitude of the lake. The climatological time series from the grid point with the least altitude difference was selected for model forcing. If the altitude difference was larger than 50 m (or if no forcing data existed) the particular lake was not included in further simulations. After this selection process 181 (52 %) of the original 345 lakes were accepted for the final batch simulations.

These simulated 181 lakes ranged from 58 to 70 °N in latitude, from 0.1 to 200 km^2 in surface area, from 0.4 to 11 m in the secchi depth, from near sea-level up to 1100 m.a.s.l. in altitude, and from 4 to 300 m in maximum depth.

METHODS I - Used models

Assessment criteria of used models

The functioning and applicability of models was analyzed in terms of two complementary benchmark criteria of models

I. Benchmark criteria of models in the implementation of WFD in EU (BMW-project in Saloranta et al 2003) (Table 2).
II. Criteria of models in the implementation of TMDL analysis in US (Reckhow 2001) (Table 3).

The BMW criteria are comprehensive where as TMDL (Total Maximum Daily Load) criteria are compact but fast to apply.

The modeling processes were analyzed in terms of

I. HarmoniQuA guidelines in modeling process (Refsgaard et al. 2005).

HarmoniQuA guidelines (Table 4) were used to assess how well different aspects of modeling are communicated to and performed in accordance with managers and stakeholders.

Table 2: Benchmark criteria of models in the implementation of WFD in EU (BMW- project in Saloranta et al 2003). Look at Appendix A for more details.

1.1 Relevance 1.2 Scale and span 1.3 Tested 1.4 Complexity 1.5 Data requirements 1.6 Identifiability 1.7 Easy of understanding 1.8 Peer acceptance
2 Uncertainty and Sensitivity
3.1 Version control 3.2 User manual 3.3 Technical documentation 3.4 Suitability for end-user participation 3.5 Flexibility for adaptation

Table 3: Criteria of models in the implementation of TMDL analysis in US (Reckhow 2001).

the model focuses on the water quality standard
the model is consistent with scientific theory
model prediction uncertainty is reported
the model is appropriate to the complexity of the situation
the model is consistent with the amount of data available
the model results are credible to stakeholders
the cost for annual model support is an acceptable long-term expense
the model is flexible enough to allow updates and improvements.

Table 4: HarmoniQuA guidelines in modeling process (Refsgaard et al. 2005).

Availability of Internal technical guidelines Public technical guidelines Internal interactive guidelines
State of the art Maturity of discipline: availability of data and understanding of processes Maturity of markets: the number of and the competition between consultants
Motivation and the content and structure of communication Mutual understanding, dialogue and commitment of modelers, water manager and stakeholders Formalised review steps Uncertainty assessment Transparency and reproducibility Accuracy
Tasks to communicate with model users that is managers, stakeholders and public Definition of the purpose of the modeling study Collection and processing of data Establishment of a conceptual model Selection of code or alternatively programming and verification of code Model set-up Establishment of performance criteria Model calibration Model validation Uncertainty assessments Simulation with model application for a specific purpose Reporting

Lake Pyhäjärvi

Hydrodynamic models have been applied in Lake Pyhäjärvi already in 1970's by Virtanen (Virtanen, 1977). Later a horizontal 2D flow model together with a sediment transport model and a box model with water quality compartment were applied for the lake management planning project (Huttula 1994, Malve et al., 1994). Main purpose was to dimension external nutrient loading reduction to the attainment of lower and less frequent algal blooming.

Active work in applying mathematical aquatic models have happened during the last ten years in Finland and Lake Pyhäjärvi has served as a key case target in this respect. SYKE and also Institute of Pyhäjärvi have had active roles in this. Also the University of Turku, department of Biology has contributed to the model development significantly. During the EU-funded BMW-project (Benchmark Models for the Water framework directive) in 2002-2004 the lake and the excellent data sets were utilized in testing various models and their predictive capabilities. The models included in this comparison were MyLake, Delft 3D, LakeState (LS) –model in lake and The Soil and Water Assessment Tool (SWAT) on the catchment (Saloranta et al. 2004). Moreover, EU-project EUROHARP (2002–2006) utilized monitoring data from Eurajoki catchment when testing a set of nutrient leaching models, to provide end-users with quidance for choosing appropriate quantification tools (Kronvang et al. 2009). During the national TEKES-funded CatchLake project (2006-09) (Lepistö et al. 2008; Lepistö & Huttula, 2010), integrated catchment models SWAT and INCA-N (Granlund 2008) were applied to the Yläneenjoki catchment, in order to simulate the suspended solids, phosphorous and nitrogen loading of the lake. The 3D process-based lake model Coherens was also applied to the lake, utilizing input data from the SWAT and INCA models (Tattari et al, 2010). Cohrens model is under active development in SYKE and Lake Pyhärvi serves as excellent test case. In this project REFRESH, INCA model application was further developed, utilizing real time NO_3-N measurements in calibration (Etheridge et al., 2010)

During Comprehensive analysis of recovery of eutrophicated lakes (CARE) –project funded by the Finnish Academy in 2004-2006 changes in Lake Pyhäjärvi ecosystems during last decades and reasons for the change were studied and different models were used to predict lake's present state, recovery and future (Ventelä, 2007).

Today hydrologic rainfall-runoff model of Lake Pyhäjärvi catchment is a part of SYKE's nationwide flood and runoff forecasting system WSFS, that covers more than 80% of the state surface area.

Table 5 shows the models used in the management of Lake Pyhäjärvi.

Table 5: Models used in the management of Lake Pyhäjärvi.

Model	Assumptions	Domains	Key variables	Pressures	Management measures
WSFS VEMALA	Conceptual	River basin	TP, TN, runoff	Agriculture Climate Land use	Changes in agriculture and land use
INCA	Semi-distrib.	Catchment	TP, TN, runoff	Agriculture Climate Land use	Changes in agriculture and land use
SWAT	Semi-distrib.	River basin	TP, TN, runoff	Agriculture Climate Land use	Changes in agriculture and land use
LLR	CSTR, static, probabilistic	Lake	TP, TN, Chlor. a	Agriculture Climate Land use	Reduction of nutrient loads
LakeState	CSTR, dyn. , probabilistic	Lake	TP, TN, 4 algal groups	Agriculture Climate Land use	Reduction of nutrient loads, fisheries management
MyLake	1D, dyn.	Lake	DIP, DOP, DIN, PIP DOC, 2 algal groups	Agriculture Climate Land use	Reduction of nutrient load
Coherens	3D, dyn.	Lake	TP, TN, 4 algal groups	Agriculture Climate Land use	Reduction of nutrient loads
Influence diagram	Probabilistic, Integrated management model, meta	River basin, lake	Management measures, Cyanobacteria, zooplankton, planktivorous	Agriculture Climate Land use	Buffer strips, wet lands, forestation of fields, reduction of nutrient loads,

	model		fish, costs, benefits, decisions		fisheries management

WSFS and VEMALA-models

The WSFS-VEMALA model simulates hydrology and water quality for all river basins in Finland. The simulated water quality parameters are total phosphorus, total nitrogen and suspended solids. The model simulates on daily time step nutrient leaching from land areas, incoming load of each one hectare and larger lake, nutrient transport in rivers and finally loading into the sea. The model consists of hydrological and water quality parts. The hydrological part is based on HBV-model and it simulates on one day time step snow, soil moisture, ground water, runoff for each sub-basin and discharges and water levels in rivers and lakes. The model covers all Finland, about 400 000 km^2, which is divided into about 6000 sub-basins, which sizes are 50-100 km^2. Further model simulates hydrology for all 1 ha and larger lakes, which there are total of 59 072.

The time step in WSFS-VEMALA is one day and simulation unit in WSFS-VEMALA is the third level sub-catchment (50-100 km^2), but if there is a lake (bigger than 1 ha) in the sub-catchment, then the simulation unit is the lake's catchment. Each simulation unit is divided then into two land use classes – fields and other land areas (forests and peatlands) and these two land use classes are simulated separately. Nutrient diffuse loading from each land use class into the rivers and lakes is simulated. It is based on the daily relationship between nutrient concentration of runoff and the daily runoff, where the main dynamically changing variable is runoff and time independent parameters are calibrated for each sub-catchment, for each land use class and for each season of the year. Year is divided into four seasons: winter (snow cover and soil frost), spring (before thermic growth season), summer (growth season) and autumn (after growth season and before soil frost). Information about each fields' slope, crop and soil texture is processed and available in the model, but it's not directly used in the equations relating concentration and runoff. Each fields' slope, crop and soil texture information is used indirectly through the calibration procedure into the annual diffuse load calculations from fields.

Another tool VIHMA (Puustinen et al 2007) is used for simulation the nutrient load from fields. VIHMA estimates long term mean loading (kgXm^{2}y^{-1}) of each field plot based on plant, slope, soil type and agricultural practices applied on the field. The calculation is based on empirical specific load approach for different land use types. VIHMA is included in WSFS-VEMALA and the mean annual load for each field plot is calculated and then used in calibration to adjust the WSFS-VEMALA simulated mean annual load from subcatchment's filed area that it matches the VIHMA estimates. By using VIHMA the spatial distribution between subcatchments of simulated mean annual loadings has improved.

Also point load (point load form municipal waste water treatment plants, loading from scattered settlements and peat mining and fallout) is taken into account in WSFS-VEMALA loading simulations. Point load estimates and observations are taken into the model as input values.

Nutrient load transport, sedimentation, erosion, and denitrification in the river are simulated. Nutrient transport and retention through river is simulated in the similar way as the runoff routing through river. Rivers are divided into river stretches. Each river stretch is then simulated as a reservoir, respectively the inflowing load, mass of nutrients and outflowing load of the each river stretch is simulated. The sedimentation, erosion and nitrogen denitrifications are then simulated for each river reach.

Nutrient sedimentation, internal loading and denitrification in lakes are simulated. Nutrient balance for lakes is simulated according to the mass balance equation, where daily change of nutrient mass is equal difference between inflow loading and outflow loading, sedimentation, internal loading (in eutrophic lakes) and denitrification for nitrogen. Daily sedimentation coefficient constant in time is estimated by calibration for each lake separately.

Model parameters are estimated by automatic calibration, which minimizes the difference between observed and simulated concentrations in rivers and lakes and loads, VIHMA mean annual loads from fields. The number of calibrated parameters in the model is high and therefore implementation of the model requires adequate amount of water quality data. However the calibration of the model is not done subcatchment by subcatchment, but a total river catchment is calibrated simultaneously. In the case of low number of observations on one subcatchment, also the downstream observation points affect parameter values on that un-observed subcatchment. Using VIHMA mean annual load estimates improves the spatial distribution of parameter values particularly in cases of low observation subcatchments.

The WSFS-VEMALA model is used for real time simulation and forecasting of water quality. The daily updated forecasts are provided for the public by www pages. The

WSFS-VEMALA model and LakeState algae blooming model will be integrated. The resulting integrated model will be run daily for providing algae blooming forecasts for about 48 000 lakes. A set of important and well observed lakes will be selected among all lakes and forecast for these will be provided on a public web interface.

Mylake – 1D model

MyLake (Multi-year simulation model for Lake thermo- and phytoplankton dynamics) is a one-dimensional process-based model code for simulation of daily vertical distribution of lake water temperature (i.e. stratification), evolution of seasonal lake ice and snow cover, and phosphorus-phytoplankton dynamics and the code is described in Saloranta and Andersen (2007). The lake water simulation part of the model code is based on Ford and Stefan (1980), Riley and Stefan (1988), and Hondzo and Stefan (1993), while the ice simulation code is based on Leppäranta (1991) and Saloranta (2000). MyLake has been developed at the Norwegian Institute for Water Research (NIVA), and the model has been applied in the BMW, THERMOS and EUROLIMPACS projects (Saloranta et al. 2004, Lydersen et al. 2003). The strengths of the MyLake model code include:

- MyLake has a relatively simple and transparent model structure, it is easy to set up, and it is suitable both for making predictions and scenarios, and as an investigative tool.
- The short integration time allows comprehensive sensitivity and uncertainty analysis to be made as well as the simulation of a large number of lakes or simulations over long periods (decades).
- MyLake includes only the most significant physical, chemical and biological processes in a well-balanced and robust way.

The model building blocks are based on well-established science and have been used previously in numerous applications. The model building blocks are based on well-established science and have been used previously in numerous applications. Some users may also consider the model too simple for their purposes, as many processes, e.g. zooplankton grazing and fish population dynamics are left out of the current version (v.1.2) of the model code. Other limitations may be the model time step which is preset to 24 hours and cannot be changed, as well as the 1-dimensional vertical resolution approach, which may not be so well suited for some particular types of problems and lakes. Also the model code only runs in the Matlab environment but it is relatively easy to convert it into an independent program.

Generally the MyLake model code development aims to take into account the following five criteria adapted from Riley and Stefan (1988):

1. that the model code be general for use on different sites with a minimum of alterations,
2. that the model code be capable of simulating a wide range of treatment options,
3. that the model code incorporate the dominant physical, chemical and biological processes, especially processes directly affected by various treatment options,
4. that the physical, chemical and biological components be simulated with similar orders of detail to reduce the possibility of a weak link in the modelling process,
5. and that the model be economical enough to run to serve as management tool.

Required data for setting up the MyLake application for Pyhäjärvi were time series of meteorological variables, lake morphometry and initial profiles, and model parameter values.

The meteorological forcing data used in setting up a MyLake model application includes air temperature, relative humidity, air pressure, wind speed, and precipitation. Daily or sub-daily meteorological observations from Jokioinen station (Finnish Meteorological Institute, FMI) in 2001-2009 were used as model forcing. All the sub-daily time series were integrated to daily values. In addition, daily time series of global radiation from Jokioinen station are used until September 2008. After September 2008 the incoming solar radiation at the water or ice surface was calculated using astronomical equations and observed cloudiness from Pirkkala station (FMI).

The model calibration and evaluation against lake temperature observations was performed only by eye. The observations of temperature profiles were taken in the deepest part of Pyhäjärvi in 2001-2003 and 2005-2009. The calibration was done by using the parameter for the wind sheltering effect, i.e. effect of surrounding terrain on sheltering a water body from winds observed at a meteorological station that may be located at a distance from the water body. The wind sheltering affects the wind-induced vertical mixing of the lake in the open water period and thus the degree of temperature stratification. The calibration period was 2006-2007. The observations not used in calibration can be used as independent data for evaluation of model performance.

Most of the model parameters were the same as in Saloranta and Andersen (2007). The inflow volume scaling factor was set to 1 and the effect of the inflow temperature to the lake temperature was excluded.

In addition to the wind sheltering coefficient, the melting snow albedo was adjusted to agree better with lake observations. The melting snow albedo is constant in the MyLake model and it was set to 0.45 in order to the simulated ice-on and ice-off dates to agree with observations. This results in slightly thinner snow cover in wintertime, but it has scarcely any effect on the water temperature.

Results show that the model predicts lake water surface temperature very well (Figure 9). Only in early spring during the years 2002 and 2003 the model under estimated water temperature. In 2002 the heat content of the lake was later in the summer simulated correctly, where as in 2003 the heat content remained under estimated throughout the summer and also the stratification was not correctly simulated.

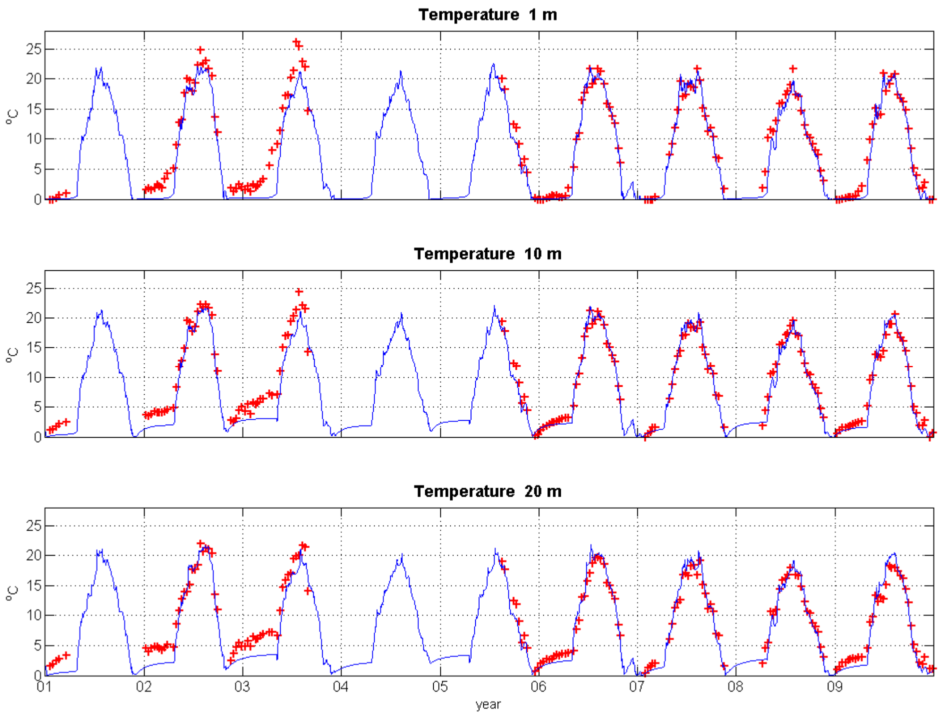


Figure 9: Simulated and observed water temperature in Lake Pyhäjärvi during the period of 1.1.2001-31.12.2009.

In this project MyLake-model was used for Lake Säkylän Pyhäjärvi only for demonstrative temperature calibration. More extensively it was used for 181 Norwegian lakes. This contribution is discussed later in this document. Anyhow model application to Lake Säkylän Pyhäjärvi proved very to be very promising and the management use potential of the model can be evaluated fairly straightforwardly.

Coherens – 3D lake model

The 3D lake model used in this study is based on the Coherens model code (Luyten et al. 1999) which was originally designed as a regional model for the North Sea. The aim of the work that produced the model was to predict the effect of changing conditions on the biota and to simulate the input and dispersion of contaminants in coastal and shelf seas. The physical parameterizations of the Coherens model are similar to those in the Princeton Ocean Model (Blumberg and Mellor, 1987). It applies the hydrostatic assumption, it utilizes sigma-coordinates in the vertical direction and mode splitting is used to resolve the barotropic and baroclinic modes with different time steps.

For this study the model was applied in cartesian coordinates with a grid interpolated from precise depth measurements provided by the Finnish Environment Institute (SYKE). The resolution of the mesh grid became 1000m x 1000m in the horizontal direction (Figure 10). It consisted of 24 cells in the y-direction, 14 cells in the x-direction and 8 sigma layers. This and the other basic features of the Coherens-model setup used in this study are outlined in Table 6.

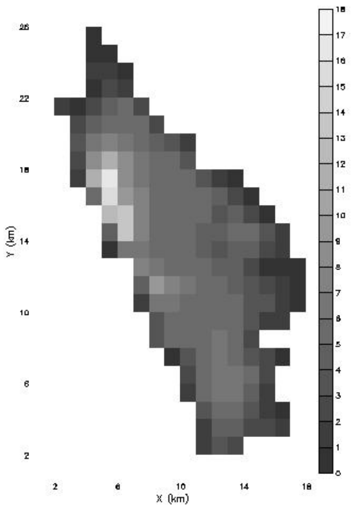


Figure 10: 1000m x 1000m grid bathymetry.

Table 6: The basic features of the COHERENS-model application.

Parameter	COHERENS
Horizontal grid and resolution	Cartesian, 1000m
Vertical grid and resolution	σ-coordinate 8 levels
Vertical turbulence scheme	k-ε model
Horizontal turbulence scheme for momentum	none
Horizontal turbulence scheme for T and S	none
Advection scheme for momentum	upwind scheme
Advection scheme for tracers (T, S and others)	TVD-superbee scheme
Convection	Hydrostatic model, convective adjustment
Equation of state	UNESCO (1981)
Surface heat fluxes:	
1) Short-wave radiation	Luyten et al. (1999)
2) Long-wave radiation	Luyten et al. (1999)
3) Sensible heat flux	Luyten et al. (1999)
4) Latent heat flux	Luyten et al. (1999)

The water-quality model used in this study is a total nutrient model where the growth of algae depends on the nutrient concentrations. The main variables are total nitrogen (c_n) and total phosphorous (c_p) in the water, biomass of cyanobacteria (c_c) and other algae (c_a) in the water and the concentrations of nitrogen (c_{ndet}) and phosphorous (c_{pdet}) in detritus. The last two compartments are inherently assumed to be included in the total nutrients.

The different compartments of the biological model are transported by diffusion and advection, both horizontally and vertically, by the physical model so only a source term in the time derivative needs to be described. The nutrient loadings are introduced via the boundary conditions. The change in time of a variable in the water column is thus:

$$\frac{\partial \phi}{\partial t} = S \qquad (1)$$

The source functions (S) for the different compartments are defined for cyanobacteria,

$$S_{cc} = (\mu_c - R_c)c_c \qquad (2)$$

for other algae,

$$S_{ca} = (\mu_a - R_a)c_a \qquad (3)$$

for nitrogen,

$$S_{cn} = \beta c_{N\det} - \frac{\mu_a N_{ina} c_a}{h_{mix}} - \frac{\mu_c N_{inc} c_c}{h_{mix}} \qquad (4)$$

for phosphorous,

$$S_{cp} = \gamma c_{P\det} - \frac{\mu_a P_{ina} c_a}{h_{mix}} - \frac{\mu_c P_{inc} c_c}{h_{mix}} \qquad (5)$$

for nitrogen in detritus,

$$S_{cn\det} = \frac{N_{ina} R_a c_a}{h_{mix}} + \frac{N_{inc} R_c c_c}{h_{mix}} - \beta c_{N\det} - \frac{S_{N\det}(z) c_{N\det}}{h_{mix}} \qquad (6)$$

for phosphorous in detritus,

$$S_{cp\det} = \frac{P_{ina} R_a c_a}{h_{mix}} + \frac{P_{inc} R_c c_c}{h_{mix}} - \gamma c_{P\det} - \frac{S_{P\det}(z) c_{P\det}}{h_{mix}} \qquad (7)$$

The growth and decay rates for the cyanobacterial algal compartment are

$$\mu_c = \mu_{c\max} f_{cn}(c_N, c_P) f_{cI}(I) f_c(T) f_{ac}(c_a, c_c) \qquad (8)$$

$$R_c = R_{c\max} f_l(T)\frac{c_c - C_{\min}}{c_c} \quad (9)$$

and for other algae

$$\mu_a = \mu_{a\max} f_{an}(c_N, c_P) f_{aI}(I) f_a(T) f_{ac}(c_a, c_c) \quad (10)$$

$$R_a = R_{a\max} f_l(T)\frac{c_a - A_{\min}}{c_a} \quad (11)$$

The rates for nitrogen and phosphorous mineralization are

$$\beta = \beta_0 f_n(T) \quad (12)$$

$$\gamma = \gamma_0 f_p(T) \quad (13)$$

The limiting factors in the equations are

$$f_{cn}(c_N, c_P) = \left(\frac{c_N}{c_N + K_{nc}}\right)\left(\frac{c_P}{c_P + K_{pc}}\right) \quad (14)$$

$$f_{ca}(c_N, c_P) = \left(\frac{c_N}{c_N + K_{na}}\right)\left(\frac{c_P}{c_P + K_{pa}}\right) \quad (15)$$

$$f_{cI}(I) = \frac{I}{I + K_{Ic}} \quad (16)$$

$$f_{aI}(I) = \frac{I}{I + K_{Ia}} \quad (17)$$

$$f_{ac}(c_a, c_c) = 1 - \frac{c_a + c_c}{A_{\max}} \quad (18)$$

The temperature dependence of the different processes is described by a function of the form

$$f_i(T) = e^{\int_{T_{opt,i}}^{T'} \ln\left(a_{T,i} + \frac{(1-a_{T,i})T'}{T_{opt,i}}\right) dT'} \qquad (19)$$

where the subscript i refers to the different equations (c, a, l, n and p). The integrand in Eq. (19) is calculated into a lookup table for each set of parameters and a range of temperature values at the beginning of each model run. See Table 7 for a list of the coefficients used in the equations.

Table 7: Water quality model coefficients

Coefficient	Definition	Value
N_{ina}	Nitrogen in other algae	0.027
N_{inc}	Nitrogen in cyanobacteria	0.027
h_{mix}	Mixing depth	7.0 m
P_{ina}	Phosphorous in other algae	0.004
P_{inc}	Phosphorous in cyanobacteria	0.004
S_{Ndet}	Sedimentation rate of detrital nitrogen, bottom layer only	0.16 m d^{-1}
S_{Pdet}	Sedimentation rate of detrital phosphorous, bottom layer only	0.16 md^{-1}
μ_{cmax}	Maximum growth rate of cyanobacteria	0.0465
R_{cmax}	Maximum decay rate of cyanobacteria	0.137
μ_{amax}	Maximum growth rate of other algae	0.0886
R_{amax}	Maximum decay rate of other algae	0.0845
A_{min}	Minimum biomass of other algae	0.01 g m^{-2}
A_{max}	Maximum biomass of other algae	300.0 g m^{-2}
C_{min}	Minimum biomass of cyanobacteria	0.01 g m^{-2}
β_0	Maximal detritus nitrogen mineralization rate	0.018 d^{-1}
γ_0	Maximal detritus phosphorous mineralization rate	0.043 d^{-1}
K_{Ic}	Half-saturation coefficient for radiation for cyanobacteria	6.7642
K_{Ia}	Half-saturation coefficient for radiation for other algae	1.9546
K_{nc}	Half-saturation coefficient for nitrogen for cyanobacteria	0.0000
K_{pc}	Half-saturation coefficient for phosphorous for cyanobacteria	0.9762
K_{pa}	Half-saturation coefficient for phosphorous for other algae	3.4524
K_{na}	Half-saturation coefficient for nitrogen for other algae	4.7515
$a_{T,cc}$	Coefficient for temperature limiting factor for cyanobacteria growth	1.0000
$a_{T,ca}$	Coefficient for temperature limiting factor for other algae growth	2.1082
$a_{T,l}$	Coefficient for temperature limiting factor for losses	1.0602
$a_{T,n}$	Coefficient for temperature limiting factor for detritus nitrogen mineralization	1.0179
$a_{T,p}$	Coefficient for temperature limiting factor for detritus phosphorous mineralisation	1.0575
$T_{opt,c}$	Optimal temperature for the growth of cyanobacteria	25°C
$T_{opt,a}$	Optimal temperature for the growth of other algae	15°C

$T_{opt,l}$	Optimal temperature for losses	25°C
$T_{opt,n}$	Optimal temperature for nitrogen mineralization	18°C
$T_{opt,p}$	Optimal temperature fot phosphorous mineralisation	18°C
c_{csr}	Sinking rate for cyanobacteria	1.0 m d^{-1}
c_{asr}	Sinking rate for other algae	-1.0 m d^{-1}
n_{sr}	Sinking rate for total nitrogen	0.0 m d^{-1}
p_{sr}	Sinking rate for total phosphorous	-0.00001 m d^{-1}

In this work, a special emphasis in the model development has been put on the erosion of particulate phosphorus. A sediment model needs to be coupled to the water column model because the deposition, burial of substances into the sediment and erosion of substances from the sediment are the processes behind the internal loading of the water column when oxygen is present. In anoxic conditions, the chemical reactions within the sediment need to be taken into account as well, and the amounts of other substances, such as iron, need to be considered.

The bottom sediment in the present model implementation has a two-layer structure. All substances are initially deposited into a so-called 'fluff'-layer. Erosion due to mechanical current interactions is possible only from this layer. For inorganic sediment and detrital matter, deposition and erosion are the only two processes acting on them in the fluff-layer. Nitrogen and phosphorous however can be drawn into the layer below the fluff-layer, in which they are buried and removed from the system. This process simulates the effect of benthic animals which actively bury nutrients.

The different biological compartments are not linked in this model because of the use of total nutrients. For example, the detrital nutrients are actually included in the total nutrients, but they are treated separately for completeness, and to retain the potential for converting the model into a dissolved nutrient model that can be used in marine applications. Also in reality there are processes acting in the sediment that convert the nutrients from one form into another, especially in the case of an anoxic bottom. These are not included in this version of the model code but the simulation of these requires separated compartments.

All biological compartments (algal groups, nutrients and detrital groups) and the inorganic sediment compartment have a sinking velocity, which together with the physical vertical velocity of the water determines their sinking rate. The sinking velocity can also be positive, as in the case of cyanobacteria, which means that the compartment is positively buoyant. This sinking rate also determines the deposition rate, except for the algal groups, which need to be turned into detritus before they can be deposited into the sediment. The flow of nutrients is shown in Figure 11.

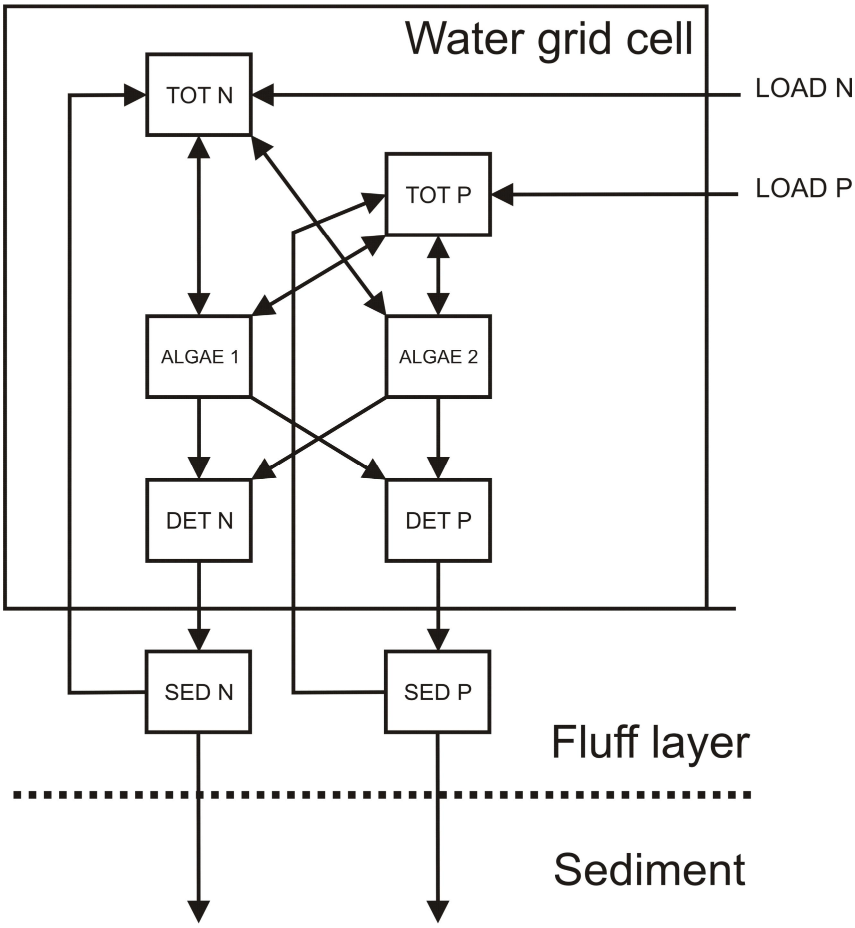


Figure 11: Flow chart showing the flow of nutrients through the biological and sediment models. Tot refers to total, det to detritus and sed to sediment.

The change in time of the concentration of compartment φ due to deposition and burial is

$$\frac{\partial c_{\varphi}}{\partial t} = ss \cdot c_{\varphi} - \tau_b \cdot c_{\varphi} \qquad (19)$$

where ss is the sinking rate and τ_b is the burial rate. Only the total nutrients are buried into the sediment and τ_b is 0 for the other compartments.

The erosion of a particle from the bottom is determined by the erosion rate from the fluff-layer into the water column. For a substance φ it is:

$$E(\varphi) = F(\varphi)\alpha_s \left| \frac{\tau_{100}}{\tau_{b,ref}} \right|^{n_s} \qquad (20)$$

Here τ_{100} is the bed shear stress at a reference height of 1m, α_s and n_s are adjustable coefficients. F(φ) is a function of the amount of any given fraction φ in the total eroded matter. It is set to 1 for all fractions. A nominal reference stress, $\tau_{b,ref}$, of 0.1Nm^{-2} is used to make the ratio dimensionless. The bed shear stress, τ_{100}, can be calculated either by using a quadratic bottom friction law:

$$\tau_{100} = \rho_0 C_{100}\left(u_{100}^2 + v_{100}^2\right) \qquad (21)$$

where the linearly interpolated horizontal velocity values at 1m are used (u_{100}, v_{100}) and ρ_0 is the reference density, or by using a linear law

$$\tau_{100} = \rho_0 k_{lin}\left(u_{100}^2 + v_{100}^2\right)^{\frac{1}{2}} \qquad (22)$$

The friction coefficient C_{100} depends on whether wave-current interactions are required or not. In the first case it is:

$$C_{100} = \left(\frac{\kappa}{\ln z_0}\right)^2 \qquad (23)$$

where κ is the Von Kàrmàn constant (0.4) and z_0 is the roughness length. In the second case it is:

$$C_{100} = \left(\frac{\kappa}{\ln\left(30/k_{bc}\right)}\right)^2 \qquad (24)$$

where k_{bc} is the apparent physical bottom roughness which is calculated from the wave height h_s and wave period T_w. In the present version of the model, these are not calculated. Because waves cause a lot of erosion in shallow areas, the simplicity of the wave-current interactions used in this model produce less erosion especially in shallow areas, than would be found in real conditions.

The only constraint set on the erosion E is that the erosion in a time step cannot exceed the amount of material in the fluff-layer.

It seems that the quadratic friction law produces the closest estimates when compared to observations Figure 12.

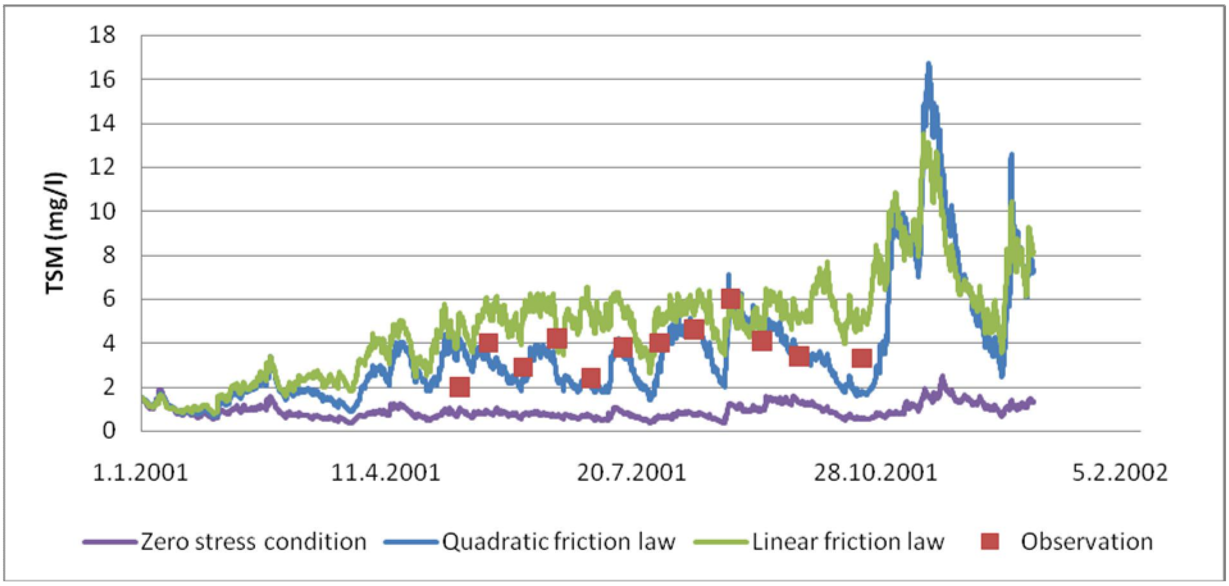


Figure 12: TSM estimates for different bed shear stress formulations compared to observations for the deepest part of lake Pyhäjärvi. The sinking velocity used was -0,0000031ms^{-1} and the value of αs was 0.006.

Figures 13 and 14show the water quality model calibration results in surface and in bottom water and also the simulated water quality in lake in the future.

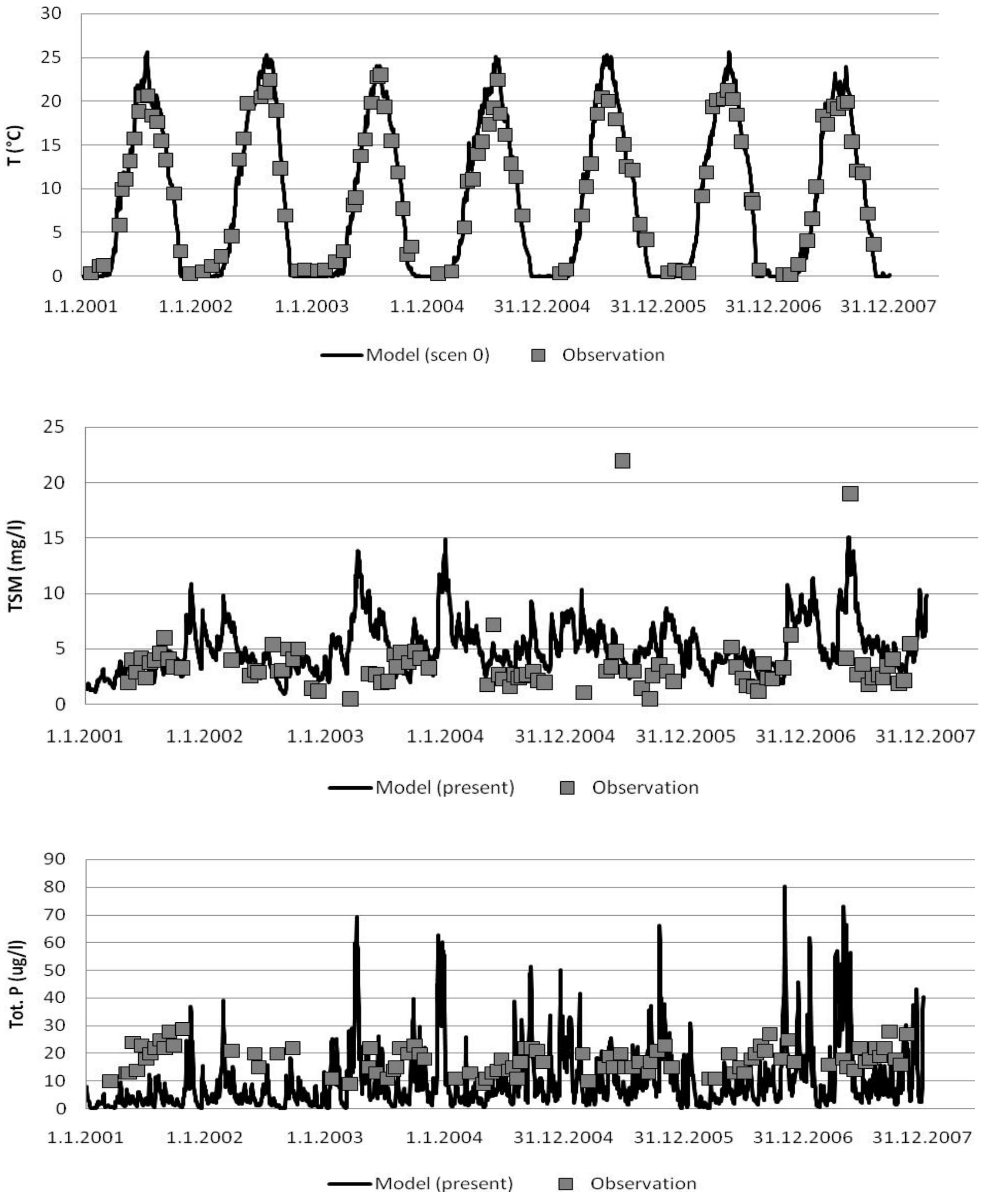


*Figure 13: Simulated temperature, suspended matter concentration and total phosphorus concentration in present case compared to observations **in surface water** at the deepest point of Lake Pyhäjärvi.*

In surface waters model predicts water temperature very well (Figure 13). Summer maximum temperatures are couple of degrees too high in the model, but the difference can be also due to the different water layer. In years 2001,2002 and 2003 at springtime water seems to stay too cool for too long in the model. Suspended solids concentration in the surface waters have low values in the observations, exept two obvious outlyers. The model seem to give too low values after 2002 and this may be due to the too high sinking velocity in model predictions. Variation of total phospohrus is higher in simulated values as in observations. Still the correspondce is quite good.

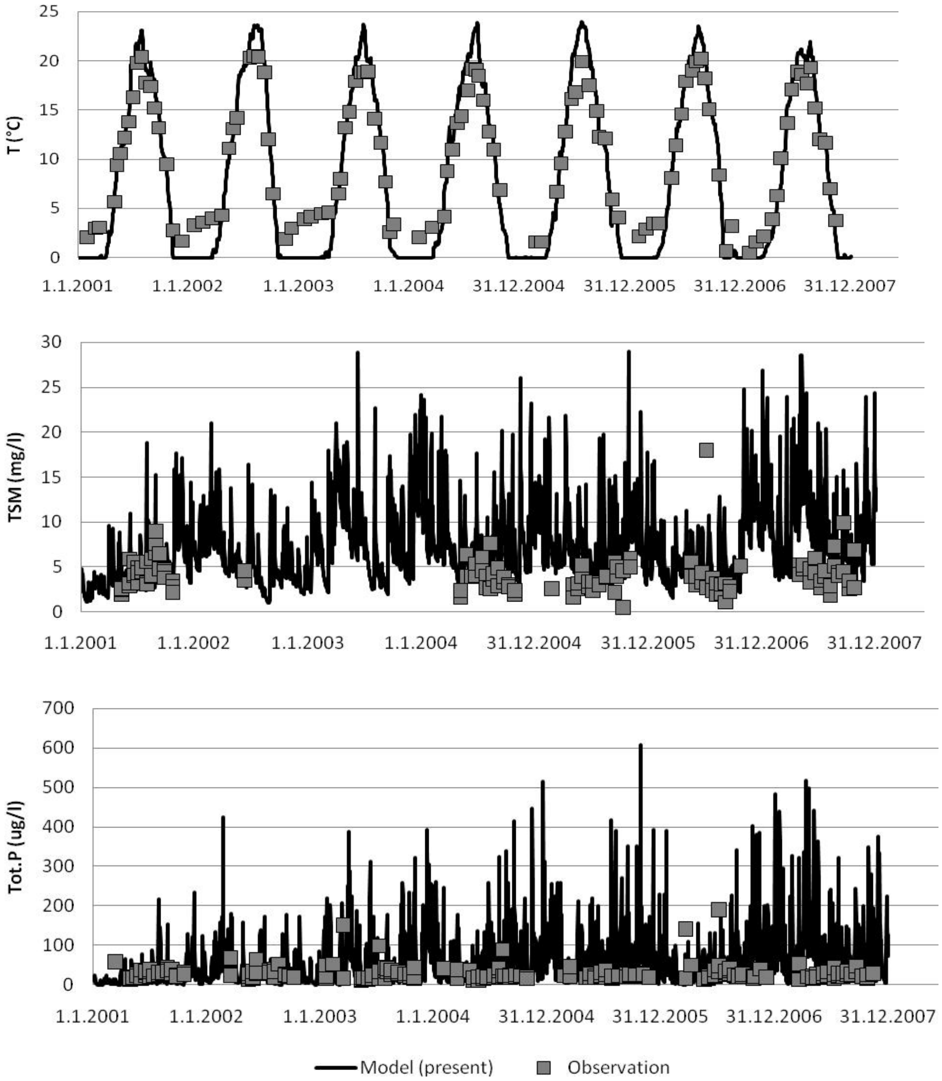


*Figure 14: Simulated temperature, suspended matter concentration and total phosphorus concentration in present case compared to observations **in bottom water** at the deepest point of Lake Pyhäjärvi. Simulated values are from the depth of 15m (bottom layer in the model). The real depth is 24m.*

Observations are from 15m (temperature) and from 15m and below that (TSM and P).

In deep waters model seems to give too high water temperatures in summer and too low values in winter time (Figure 14). The reason for this is in spatial resolution of the model and turbulence scheme, which lead to high mixing in summer. The winter time discrepance is due to the too high heat flux in the model as there is now ice model and part of the coming radiation is penetreting into the lake.

Both suspend solids concnetration and total phosphorus concentration in deep waters seem to be very high in the model results as compared to the observations.

Earlier Model applications

INCA-model

The Integrated Nutrients Model for Catchments – Nitrogen (INCA-N) (Whitehead et al. 1998, Wade et al. 2002) is a process-based and semi-distributed model that integrates hydrology, catchment and river N processes to simulate daily concentrations of NO_3-N and NH_4-N in the river system. The INCA model used in this application (Granlund, 2008) is version 1.11.1 . In the model, hydrologically effective rainfall (HER) is the input to the soil water storage driving water flow and N fluxes through the catchment system (Whitehead et al., 1998). Catchment hydrology is simulated with a simple three-box approach, including direct runoff and reservoirs of water in the reactive soil zone and deeper groundwater zone. Flows from the soil and groundwater zones are controlled by time constants, representing residence time in the reservoirs. The catchment is divided into sub-catchments. In each of them, INCA-N can simulate water flow and N processes in six land use classes. Base flow index (BFI) is used to calculate the proportion of water being transferred to the groundwater zone. Calculation of river flow is based on mass balance of flow and a multi-reach description of the river system. Flow variation within each reach is determined by a non-linear reservoir model (Whitehead et al., 1998). For the snow pack depth the calculation model utilizes a simple degree-day model. A process based function is used to calculate soil temperature from ambient air temperature (Rankinen et al., 2004a-c).

The INCA-N model was calibrated in Yläneenjoki catchment for period 1995-1999 and tested for period 2003-2007 (Granlund 2008). The daily hydrological input data (hydrologically effective rainfall, soil moisture deficit, air temperature and precipitation) was derived from the Watershed Simulation and Forecast System WSFS (Vehviläinen 1994, Vehviläinen & Huttunen 2002). In INCA-N-application the Yläneenjoki catchment was divided into five sub-catchments (Figure 15). The division was based on the 3rd level delineation included in the WSFS. The land cover and agricultural information for the different sub-catchments was based on the VEPS decision support system (area of forests and agricultural area) and the report of the MYTVAS-study (agricultural practices) (Palva et al. 2001). Six land cover classes were included in the simulations: one for forest and five for field crops (spring cereals, autumn cereals, grass, set-aside and others) (Figure 16).

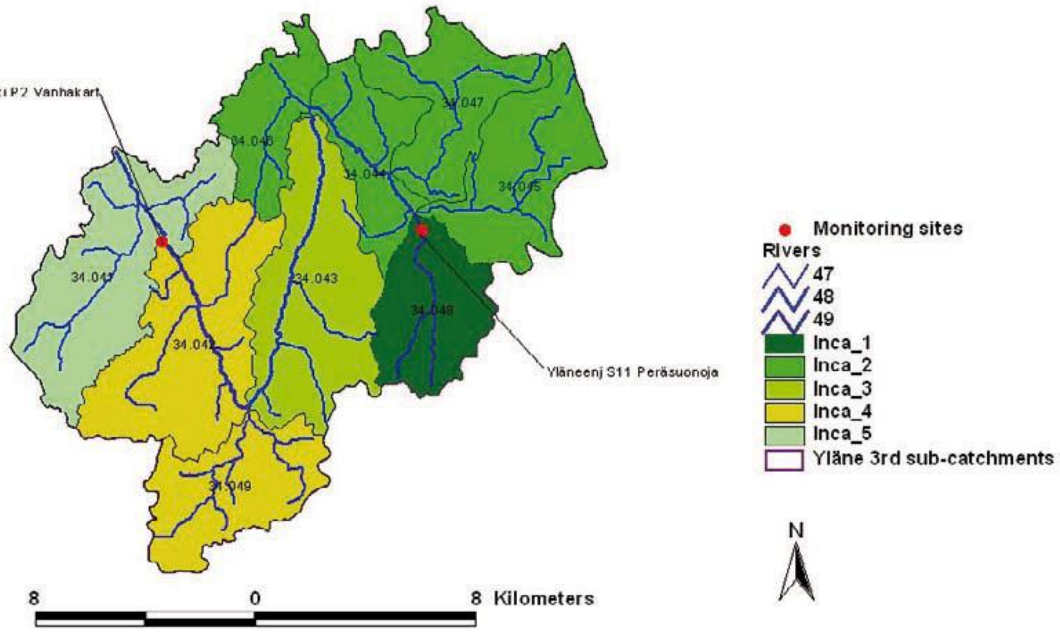


Figure 15: Yläneenjoki catchment delineation for INCA-N application.

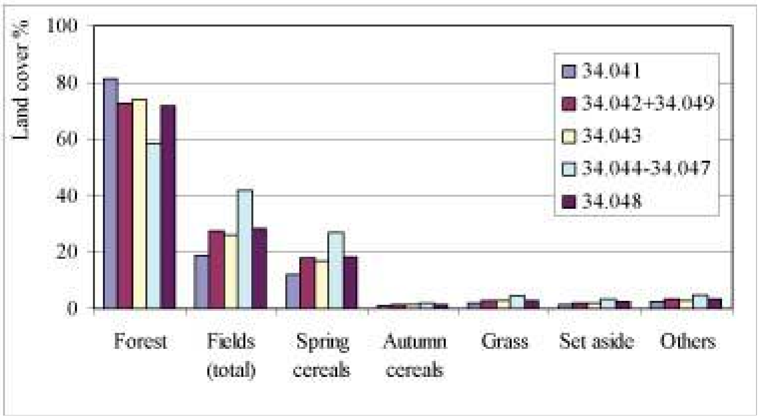


Figure 16: Land cover types in the Yläneenjoki catchment for INCA-N simulation. See Figure 15 for location of the sub-catchments 34.041–34.049.

The observed hydrological response of the Yläneenjoki catchment during 1995–1999 was rather fast. The correspondence between observed and simulated annual values was good at Vanhakartano; the Nash-Sutcliffe-coefficient had a value of 0.77 for simulated vs. observed daily discharge. The timing and magnitude of the flow peaks was simulated well for the whole calibration period. After hydrological calibration, representative literature information about typical N losses from different land use classes were used to further adjust the N process parameters. The calibrated model was able to simulate adequately the overall annual inorganic N dynamics in river water at Vanhakartano (Figure 17). Discrepancies in the simulated versus observed concentrations during single peaks are probably partly related to under- or overestimation of instantaneous discharge and simplifications made during the whole modeling process (Granlund 2008).

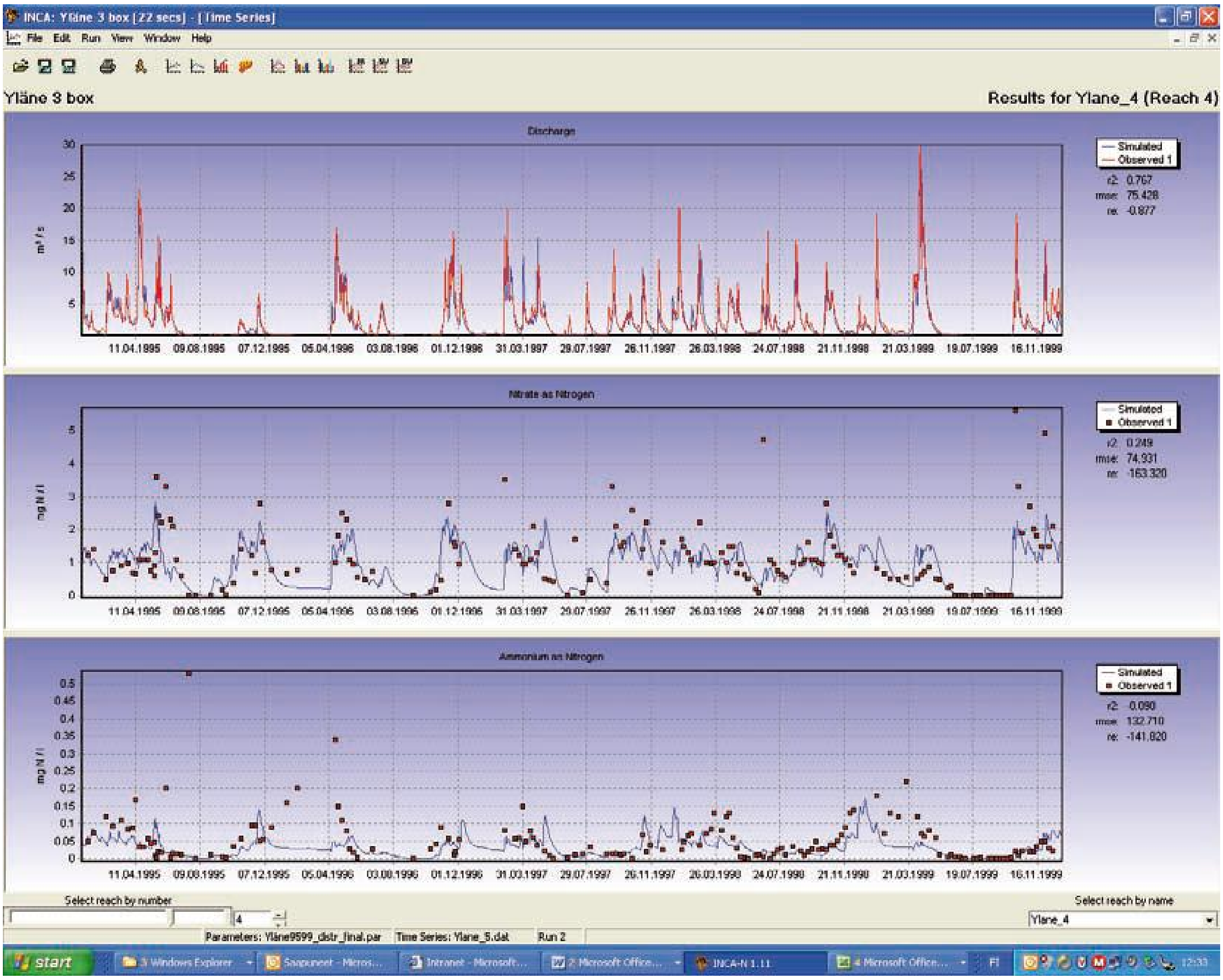


Figure 17: INCA-N calibration results for the period 1995–1999 in Vanhakartano: discharge (m3 s-1) (upper figure), NO3-N concentration (mg l-1) (in the middle) and NH4-N concentration (mg l-1) (lower figure).

The continuous NO_3-N sensor data proved to be extremely valuable for analyzing N concentration dynamics especially during autumn high flow. Interpretation of the data suggested several ideas for better calibration of N leaching in autumn. It is very important to be able to model autumn and winter losses correctly, because most of N is transported from catchments during the dormant season (Rankinen et al. 2004 a-c). Recently, INCA model application was further developed both in the Yläneenjoki and Pyhäjoki catchments, together with utilization of real-time NO_3-N measurements in calibration (Etheridge et al., 2010).

SWAT

The SWAT model (Soil and Water Assessment Tool) is a continuous time model that operates on a daily time step at catchment scale (Arnold et al. 1998, Neitsch et al. 2005). It can be used to simulate water and nutrient cycles in agriculturally dominated large catchments. The catchment is generally partitioned into a number of sub-basins. Division is based on the threshold area which defines the minimum drainage area required to form the origin of a stream. The smallest unit of discretization is a unique combination of soil and land use overlay referred to as a hydrologic response unit (HRU). SWAT is a partly process-based model and partly a distributed model as it includes many empirical relationships. The water quantity processes simulated by SWAT include precipitation, evapotranspiration, surface run-off and lateral subsurface flow, ground water flow and river flow. Water quality processes are calculated with various well-known equations. For example, erosion caused by rainfall and runoff is computed with the Modified Universal Soil Loss Equation (MUSLE) (Williams 1975). In terms of phosphorus (P), the primary mechanism of soluble fraction movement in the soil is by diffusion. Organic and mineral P attached to soil particles may be transported by surface runoff to the main channel (Neitsch et al. 2005). For channel flow simulation, SWAT uses Manning's equation coupled with variable storage or Muskingum routing method. Interactions and relationships of the river model QUAL2E (Brown and Barnwell 1987) are used as in-stream water quality processes in SWAT. The model has been world widely used and also further developed in Europe (e.g. Eckhardt et al. 2002, Krysanova et al. 1999, van Griensven and Meixner 2003).

SWAT was applied to Yläneenjoki catchment already earliear (Koskiaho et al. 2003 and Bärlund 2007). In this study new features like autocalibration and sensitivity analysis were tested For the SWAT simulations the available data on elevation, land use and soil types were aggregated. The resolution (5 m) of the digital elevation model (DEM) proved to be inaccurate for low-lying areas for successful set-up of the Yläneenjoki catchment. Hence, a modified DEM was used. There the main channels of the catchment were somewhat deepened to emphasize the actual routes of water. The agricultural crops and their location were available from TIKE (=Information Centre of the Ministry of Agriculture and Forestry in Finland). According to this database 100 different crop types are grown in the Yläneenjoki region. To build up a reasonable SWAT set-up this data was divided into 5 crop classes (autumn cereals, spring cereals, root crops, grasses and gardens) (Figure 18). All the rest land area was classified as forest. Soil type data was based on soil textural information of the Geological Survey of Finland in which soil type information covers southern Finland in a scale of 1:100 000 and it is available in 25 x 25 meter cells. Our SWAT project had 4 soil types namely clay, moraine, silt and peat .

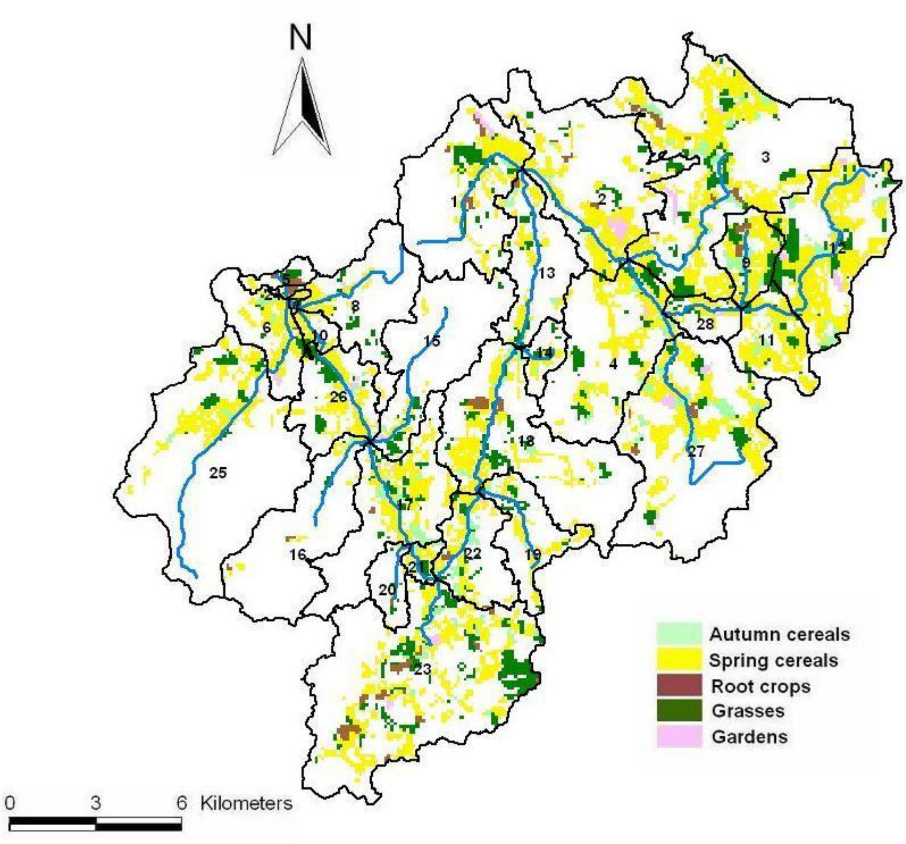


Figure 18: Distribution of agricultural land in the Yläneenjoki river basin.

The SWAT application was based on the earlier SWAT application. For example, the parameters of Bärlund's (2007) earlier application were set as initial values. In this work a more detailed GIS based land use map as earlier was used and in addition, the simulation years differed from the earlier set-up. The present SWAT application resulted in 29 sub-catchments. In this work threshold values were used to distinguish different land use and soil types within each sub-catchment. As an example if, more than 1% of a sub-catchment is under grass and these areas were divided on clay and silt soils both soils types representing more than 10% of the sub-catchment area, this would result in two HRUs (grass-clay and grass-silt) within this sub-catchment. In all, this approach resulted in 257 HRUs in the application.

First a sensitivity analysis was conducted. The build in sensitivity analysis in SWAT is Latin Hypercube One-factor-At-a-Time (LH-OAT) (Morris 1991). It was performed for a set of 32 parameters that have been found to be most influential for river discharge and sediment transport with many earlier modeling studies (Krysanova and Arnold 2008, Bärlund et al. 2007).

The sensitivity analysis revealed that in discharge simulation, none of the parameters got the ranking "very high sensitivity". The most influential parameters representing the sensitivity class "high sensitivity" were connected to equations describing processes such as ground water flow, evaporation and surface and subsurface runoff , snow melt, soil evaporation, soil water dynamic, surface and subsurface runoff . They all are highly relevant processes when simulating discharge in Finnish hydrological conditions.

In terms of sediment concentration representing general watershed properties and representing main channel characteristics, respectively, were ranked in the sensitivity class "very high sensitivity".

In the latest version of SWAT, autocalibration is implemented by using the Shuffled Complex Evolution (SCE-UA) algorithm that optimizes an objective function (SSQ= sum of the squares of the residuals) by systematically searching the entire parameter space (global optimization) (van Griensven and Meixner 2006)

In calibration simulated daily averages of discharge were compared against the observations at the Vanhakartano for the years 1995–1999. In terms of hydrology, the years differed quite a lot from each other.

Daily and monthly averages of flow were calibrated against the corresponding values determined from the observations made at Vanhakartano near Yläneenjoki river outlet. The calibration was done with the parameters which were ranked as most influential ones in the sensitivity analysis study. Autocalibration results of 6 simulation runs suggested the best parameter values including their good range in terms of discharge. For example, surface runoff parameter decreased from the initial value 4 to 0.4. This indicates that the surface runoff lag is very short in the Yläneenjoki area and a parameter regulating soil water holding capacity got a quite high value. Nash-Sutcliffe (NS) coefficients were calculated for the calibration results. NS coefficients were calculated for monthly values because for daily values it is very difficult to achieve a good fit even with intensive calibration efforts. The best coefficient value was 0.7. Figure 19 shows the clear improvement obtained by the calibration process. Particularly in spring and autumn the amendment was clear: not only were the simulated peaks were closer to the observed, but also the autumnal low flow period appeared much more realistic. In mid-winter and summer the results remained, even after calibration, relatively weak. There seems to be substantial snow melt in January and heavy summer rains, during July, that are missed by the model. As for annual average flow, the simulated values were generally lower than the measured.

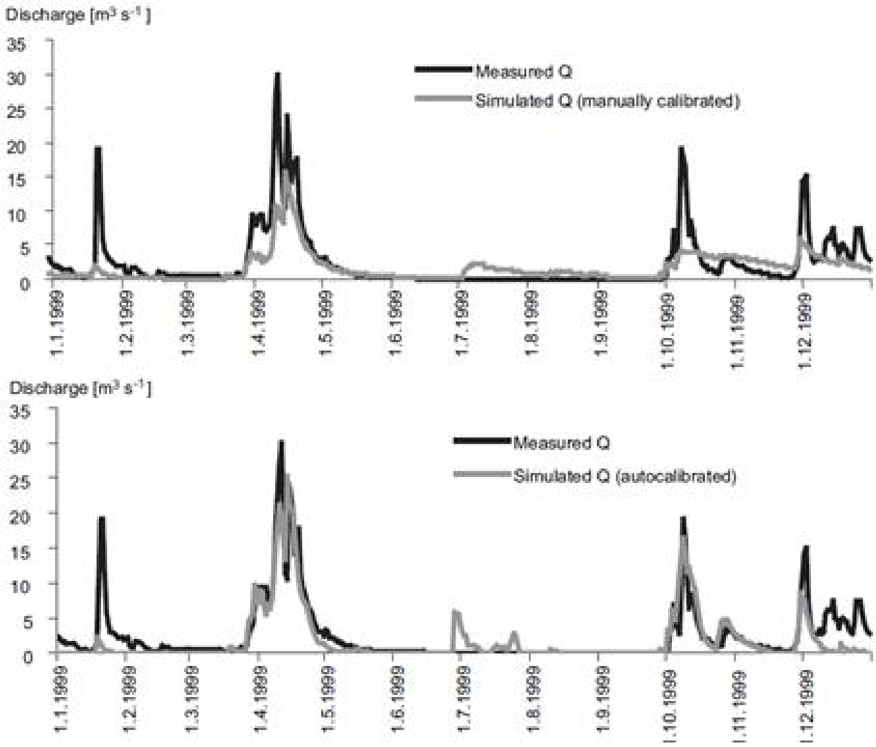


Figure 19: Measured and simulated discharge for Yläneenjoki outlet (Vanhakartano) with original manually calibrated parameter set (upper figure) and with the autocalibrated parameter set (lower figure).

In spite of the visual improvements in flow dynamics achieved by the autocalibration process, the fit between the simulated and observed daily flow remained poor when it was assessed by NS coefficients. For phosphorus and erosion, the compatibility between simulations and the calculated loads based on measurements were examined only with annual values (Figure 20). In the first year (1995) of calibration suspended sediment loading was highly overestimated and total phosphorus load, on the contrary, underestimated. One reason for this could be that the warm-up period of the model in this set-up was only one year. On the other hand, the load estimates based on manual grab sampling inherently lack reliability because long periods between the sampling occasions remain unknown. Our recent unpublished study shows that e.g. seasonal erosion can be considerably underestimated (19-72%) when based on manual sampling estimates. The years 1996-1999 show better fit exclusive of year 1999 for total P load.

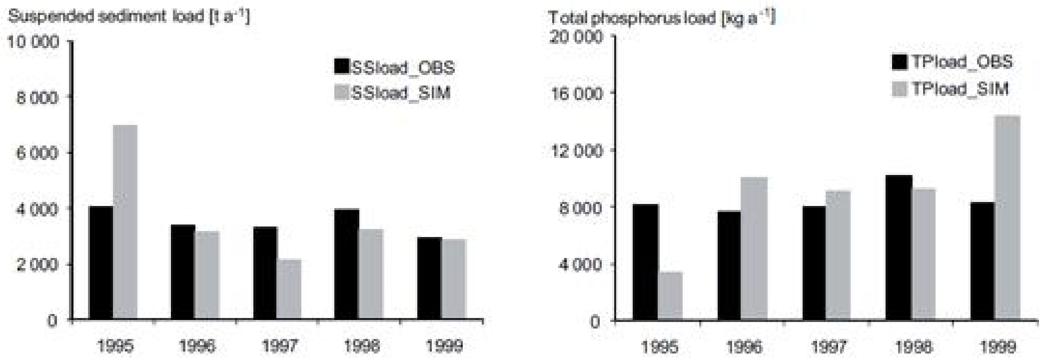


Figure 20: Measured (black bars, OBS) and SWAT-simulated (grey bars, SIM) sediment (SSload) and total phosphorus (TPload) loads during 1995–1999 for Yläneenjoki outlet.

LLR – Steady state probabilistic box model

The Lake Load Response (LLR) internet tool has been developed at SYKE to ease the use of models in making predictions about the effects of nutrient loading into a lake. The theoretical background of this tool is explained here in more detail than other models and tools since it will be used widely within WISER project within WP5.2..

LLR consists of three steady state models:

- Chapra's (1975) model for retention of total phosphorus and nitrogen including probabilistic estimates for model parameters and model predictions. It contains also different options for settling rate calculations,
- the hierarchical, linear regression model for chlorophyll-*a* (Malve 2007) and
- the logistic regression model for phytoplankton biomass (Kauppila P., Lepistö L., Malve O. & Raateland A. Unpublished).

With Chapra's retention model it is possible to estimate the in-lake nutrient concentrations as a function of incoming loading and water outflow (1).

$$V\frac{dC}{dt} = W - QC - v_s AC = 0 \qquad (25)$$

, where

V	lake volume (m^3)
$\frac{dC}{dt}$	change in nutrient concentration in time unit
W	external loading (mg d^{-1})
Q	outflow (m^3 d^{-1})
C	in-lake nutrient concentration (mg m^{-3})
v_s	Chapra's sedimentation rate(m d^{-1})
A	lake area (m^2)

In equilibrium state the in-lake nutrient concentration does not change during the retention period and ($\frac{dC}{dt} = 0$).

$$\Rightarrow C = \frac{W}{Q + v_s A} \qquad (26)$$

The model assumes complete mixing in the lake. In reality this is not the case in all lakes and also the variability of lake's internal loading affects the equilibrium and reduces the precision of estimates.

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

$$C \sim N(\mu, \tau^2) \qquad (27)$$

μ expectation for in-lake nutrient concentration (mg m^{-3})

τ^2 error variance for the model

$$v_s \sim N(\mu_s, \sigma_s^2) \quad (28)$$

μ_s expectation for sedimentation rate (m d^{-1})

σ_s^2 variance for sedimentation rate

The model parameter (expectation for sedimentation rate) estimation and predictions (expectation for in-lake nutrient concentration) are done according to the Bayesian paradigm using Markov chain Monte Carlo (MCMC) simulation methods.

If there are not enough observations from the study lake, the Chapra's sedimentation rate can be estimated within LLR by using regression models based on data from different kinds of lakes. The regression models of Dillon and Rigler's (1974) trapping parameter for phosphorus (4) and nitrogen (5) are fitted to total nutrient balance data from 12 Finnish lakes. Canfield and Bachmann's regression model for Vollenweider's sedimentation rate (6) (Reckhow 1988) is fitted to total phosphorus balance data from 723 North American and European lakes. Reckhow's regression model for Vollenweider's sedimentation rate (7) (Reckhow 1988) is fitted to total nutrient balance data from 70 lakes and reservoirs of the southeastern parts of North-America. Chapra's sedimentation rate can then be calculated (8 and 9) using the trapping parameter and sedimentation rate values obtained from the models.

$$R_P = 0.60 - 0.0006W \quad (R^2 = 0.47) \quad (29)$$

$$R_N = 0.35 - 0.00016T_W \quad (R^2 = 0.48) \quad (30)$$

$$\sigma = 0.129(C_{in})^{0.549} T_W^{-0.549} \quad (31)$$

$$\sigma = \beta C_{in}^{\theta 1} T_W^{\theta 2} z^{\theta 3} \quad (32)$$

$$v_s = \frac{R_{P/N}}{1 - R_{P/N}} q_s \qquad (33)$$

$$R_{P/N} = (1 + \frac{1}{T_W \sigma})^{-1} \qquad (34)$$

, where

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

User can predict LLR water quality estimates by using all the above mentioned alternative ways for calculating the sedimentation rate. The estimate of the lake specific model is based only on observations from the lake. The "Finnish lakes" -model calculates the estimates using the sedimentation rates obtained from the data of Finnish lakes. The "North American and European lakes" -model uses the sedimentation rates obtained from the data of North American and European lakes, as well as lakes and reservoirs of the southeastern parts of North-America. Predictions of the lake specific model are usually most reliable, because they are based on real observations from the study lake. However, if there are not enough observations from the lake, or if the observation values cover a very narrow range, the other two models can help to increase the reliability of the predictions. There may still be problems with how well regressions drawn from lakes from different areas or from a different kind of lake fit a certain lake.

The in-lake phosphorus and nitrogen concentrations can be used to predict the in-lake chlorophyll-a concentration with the hierarchical, linear regression model for chlorophyll-a (Malve & Qian 2006, Lamon et al 2008). The model may be summarized as follows (Malve 2007):

$$\log(y_{ijk}) \sim N(X\beta_{ij}, \tau^2) \quad (35)$$

$$X\beta_{ij} = \beta_{0,ij} + \beta_{1,ij}\log(TP_{ijk}) + \beta_{2,ij}\log(TN_{ijk}) + \beta_{3,ij}\log(TP_{ijk})\log(TN_{ijk})$$

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

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

,where

$\log(y_{ijk})$ *k*th observed log(Chla) value from lake *j* of type *i*

X matrix containing the observed total phosphorus (TP) and total nitrogen (TN) values from lake *j* of type *i*

β_{ij} lake specific model parameter vector $[\beta_{0,ij}, \beta_{1,ij}, \beta_{2,ij}, \beta_{3,ij}]$ which consists of the intercept ($\beta_{0,ij}$) and slopes for log(TP) ($\beta_{1,ij}$), log(TN) ($\beta_{2,ij}$) and for the combined effect of log(TP) and log(TN) ($\beta_{3,ij}$)

τ^2 model error variance

β_i vector $[\beta_{0,i}, \beta_{1,i}, \beta_{2,i}, \beta_{3,i}]$ of the model parameter means for lake type *i*

σ^2_i vector $[\sigma^2_{0,i}, \sigma^2_{1,i}, \sigma^2_{2,i}, \sigma^2_{3,i}]$ of variances in model parameters between lakes of type *i*

β means for lake types $[\beta_0, \beta_1, \beta_2, \beta_3]$

σ^2 variance for lake types $[\sigma^2_0, \sigma^2_1, \sigma^2_2, \sigma^2_3]$

From the relation between nutrient and chlorophyll-a concentration in the lake, it is possible to determine the relation between nutrient loading and chlorophyll-a concentration. This leads to predictions about the target load with which a good ecological status of the lake according to chlorophyll-a concentration can be achieved.

The hierarchy of the model means that it uses both the data from the study lake and from the lakes of same type to make the predictions. The lake type specific data, that includes observations from 2000 Finnish lakes, is already in the LLR database. The main basis for the usage of the hierarchical model is that lakes within the same type are assumed to have similar chlorophyll-a response to changing nutrient concentrations. It is also assumed that data from one lake type covers a wider range of observation values than that from a single lake.

In practice, chlorophyll-a predictions are based almost solely on the data from the study lake when there is plenty of it, or if there is a very large scatter, for any reason, in the chlorophyll-a response to nutrient concentrations within the lakes of same type. In the opposite situation the predictions are based on the lake type specific data, but usually the emphasis on different data sources is somewhere in between. The use of lake type specific data increases the reliability of the predictions, especially when the target loads are extrapolated outside the range of observational values from the study lake.

The logistic regression model for phytoplankton biomass (11) gives predictions about the probability of phytoplankton biomass to exceed the boundary of good water quality with different phosphorus and nitrogen loads. The model is fitted to two data sets that consist of observations from Finnish and Norwegian lakes. The first data set represents humic lakes (362 lakes, color > 40 mg Pt/l) and the second clear water lakes (852 lakes, color < 40 mg Pt/l). The division to two data sets is made because in clear water lakes the exceeding probability has shown to be most related to phosphorus concentration, but in humic lakes both to phosphorus and nitrogen concentration.

$$\mathrm{Logit}(Pr) = \beta_n + \beta_1 \log(C_P) + \beta_2 \log(C_N) \qquad (36)$$

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

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

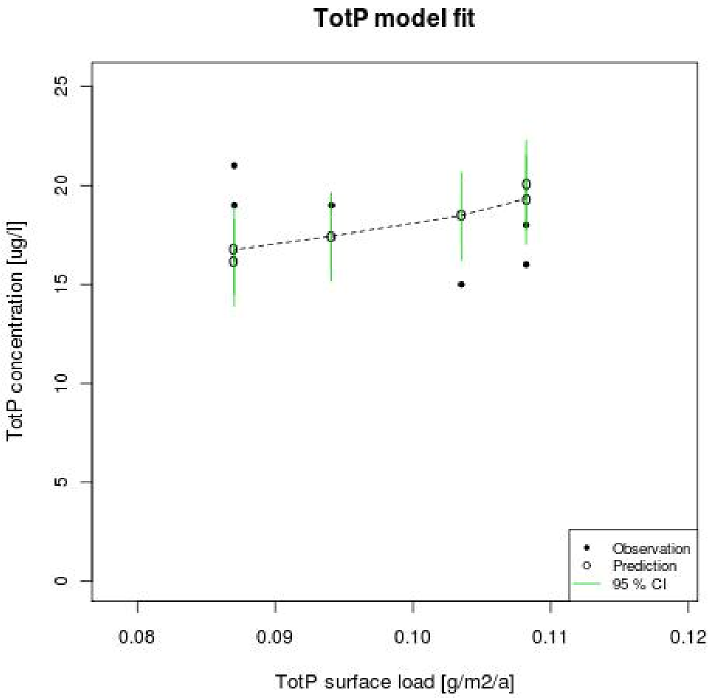


Figure 21: Fit of TP model / LLR.

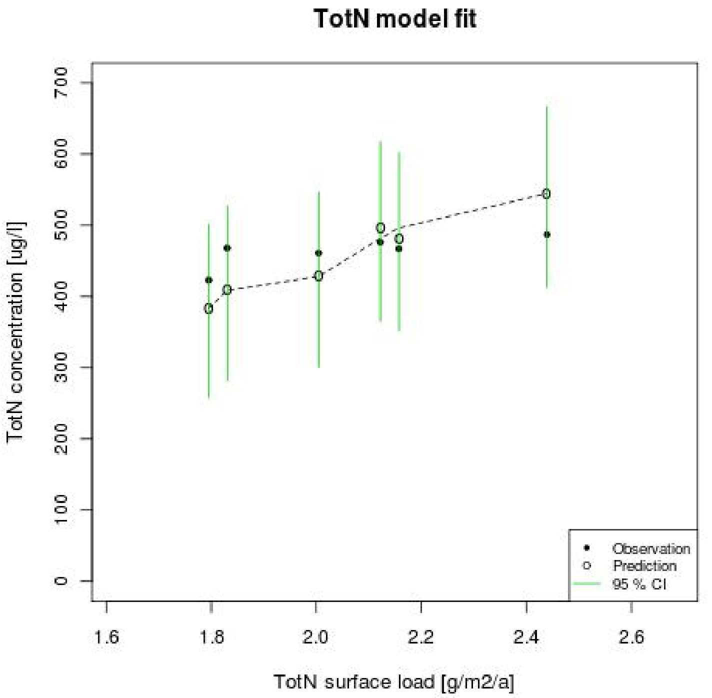


Figure 22: Fit of TN model / LLR.

LakeState – dynamic probabilistic box model

A non linear dynamic model called LakeState, based on total phosphorus and nitrogen mass balances and phytoplankton kinetics, was has been used for Lake Pyhäjärvi to simulate the main driving processes between nutrient loading, algae and zooplankton in a lake (Malve, 2007). This work is based on earlier work in EU BMW project. The computational period covered eight years of observations of the lake's water quality and hydrology. Fish is not included, but the impact of fisheries management can be simulated by manipulating zooplankton biomass concentration, which is an independent input variable in the model. Also in this model as in LLR the model parameter estimation and predictions were done according to the Bayesian paradigm using Markov chain Monte Carlo (MCMC) simulation methods. The predicted posterior distributions were constructed to reveal the consequences of the uncertainties in the predictions.

Predictive limits of the observations were quite high compared to the limits of the fitted model, indicating that model is capable to capture major dynamics of total P in 1990-2001, but not all of the variation of observed concentrations either deterministic or random. Phosphorus model (Figure 23) seems to fit to observations quite well and the fit is better than the nitrogen model. The predictive posterior distributions based on the MCMC and Monte Carlo (MC) simulation of LakeState model were analysed more closely to predict the consequences of loading reduction and fisheries management and to find optimal combination of these to actions with the given target summer maximum Cyanobacteria biomass. This means that the density estimate of mean TotP and summer maximum Cyanobacteria biomass, conditioned on a set of TotP- loading and zooplankton biomass (summer maximum) ranges can be calculated from the MC sample. Based on these calculations, external phosphorus loading level and zooplankton biomass that attain target summer maximum Cyanobacteria biomass with the given margin of safety (for example90 % percentile) can be estimated.

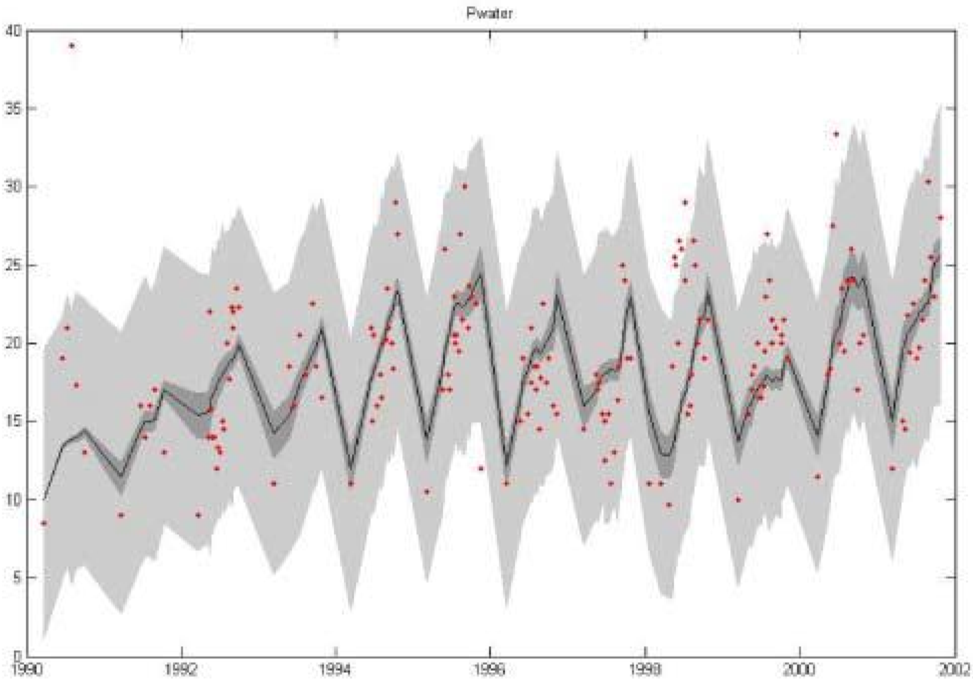


Figure 23: Observed and fitted total phosphorus concentrations (µg/l) 1990-2001. The darker gray area correspond to 95 % predictive limits of the fitted model. Solid line is the median algae concentration. Lighter gray gives 95 % prediction limits for the observations (Malve, 2007).

From the MC sample, all necessary percentiles of average TotP concentrations and summer maximum Cyanobacteria biomass were calculated as a function of MC sampled combinations of TotP load and grazing zooplankton biomass (summer maximum) (Figure 24). These results can be used to find the optimal combination of TotP load reduction and zooplankton biomass with the given range of certainty. Bayes network software "Hugin" (www.hugin.com) was used to learn causal relationships and conditional probability tables from the Monte Carlo simulations of LakeState model, to represent uncertainty in causal linkages between nutrient load and cyanobacteria summer maximum biomass, to link LakeState model with the catchment model using a simple probabilistic expression and to estimate expected attainability of the water quality criteria with given management options.

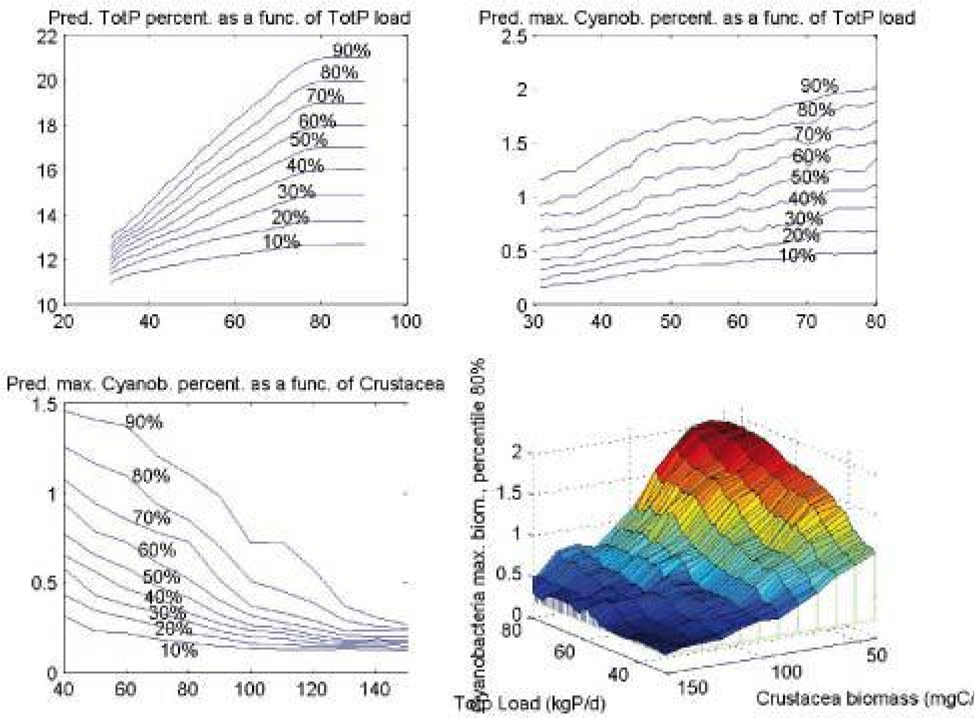


Figure 24: Estimated TotP and summer maximum Cyanobacteria biomass percentiles (10% - 90%) as a function of TotP load and summer maximum grazing zooplankton biomass. (a) mean TotP percentiles as a function of TotP load; (b) Max. summer Cyanobacteria biomass as a function of TotP load (zooplankton biomass summer maximum fixed to the level [30 50] mgC/l); (c) Max. summer Cyanobacteria biomass as a function of zooplankton biomass (TotP load fixed to the level [30 40] kgP); (d) Summer maximum Cyanobacteria biomass 80 % percentile as a function of TotP load and summer maximum grazing zooplankton biomass. This response surface can be used in the optimization of load reduction and fisheries management (Malve, 2007).

Influence diagrams

Representation of joint distribution of model variables and marginalizing conditional phytoplankton distributions with Bayes network method

Bayes network software “Hugin” (www.hugin.com) has been used to learn causal relationships and conditional probability tables from the Monte Carlo simulations of LakeState model, to represent uncertainty in causal linkages between nutrient load and cyanobacteria summer maximum biomass, to link LakeState model with the catchment model using a simple probabilistic expression and to estimate expected attainability of the water quality criteria with given management options.

Figure 25 shows a graphical representation of learned causal linkages between LakeState input and output variables. It was not a surprise that total nitrogen does not have a link to phytoplankton because Lake Pyhäjärvi is clearly phosphorus limited most of the time. These learned linkages reflect LakeState model equations, parameter distributions estimated from the lake data using MCMC method and external control variable distributions used in Monte Carlo analysis.

The constructed Bayes network can, for example, be used to calculate the conditional marginal distributions of N-fixing Cyanobacteria (CyanobMax) summer maximum biomass concentration (mg/l) and total P (TotP_C) average concentration (µg/l) with given Total Phosphorus loading (TotPLoad, 32.45 – 39.24 kg/d) and Crustacea (grazing zooplankton, ZooplMean) August-September average biomass concentration (54.94 – 74.79 mg C/l) (Figure 26).

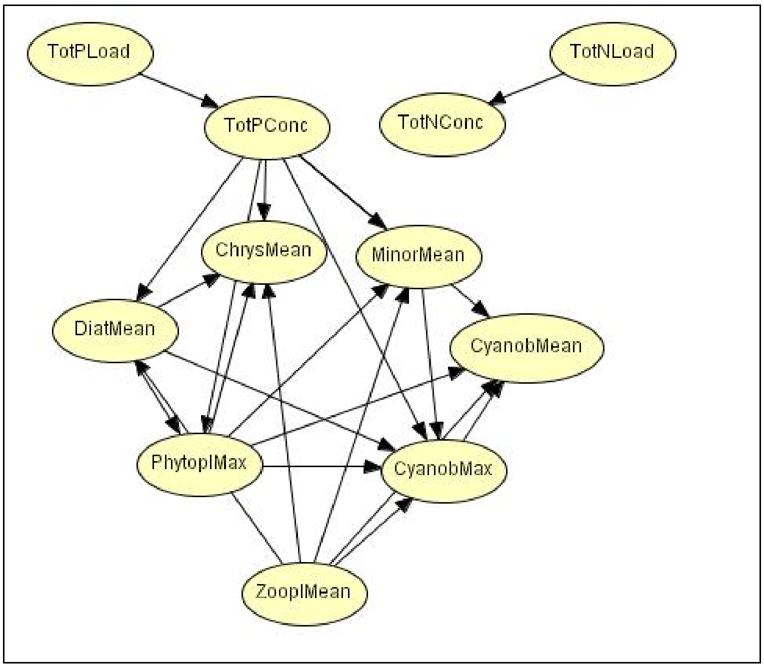


Figure 25: Bayes network representation of learned causal linkages between LakeState model input and output variables.

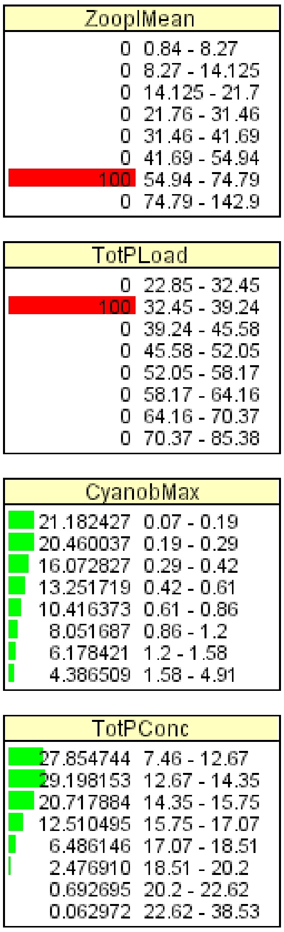


Figure 26: Conditional marginal distributions of Cyanobacteria (CyanobMax) summer maximum biomass concentration (mg/l) and Total Phosphorus (TotPConc) average concentration (µg/l) with given total P loading (TotPLoad, 32.45 – 39.24 kg/d) and Crustacea (grazing zooplankton, ZooplMean) August-September average biomass concentration (54.94 – 74.79 mg C/l).

Linking of lake and catchment models

As a part of an integrated river basin model, lake models can be linked to e.g. rainfall-runoff models, diffusion pollution models, river models and groundwater models. As many of the abatement measures are often done outside the lake domain, e.g. in the agricultural sector, there is a clear need to extend the modeling system “upstream” in order to be able to simulate the abatement measures more directly and realistically.

Linking of models can be implemented e.g. by using output series from one model as input data series to the next model, or by probabilistic linking, as shown in the following sections. Usually linking is done in form of one-way feeding of information from one model to the next, but sometimes models are even coupled so that the exchange of information between them is a two-way process, ongoing while the models are running.

In the Finnish case study, the LakeState model and a catchment model were successfully linked together with Bayes network learning method (Hugin software). This linked model system made a more direct simulation of the actual management (abatement) operations possible.

A catchment model that calculates phosphorus reduction percentage and the associated costs as a function of buffer strip width (B), forestation percentage (F) and wetland percentage (WL) was studied in Monte Carlo simulation to produce input for Hugin software Bayes network learning and estimation procedures (Figure 23). Simulations revealed conditional probabilities and correlations between variables.

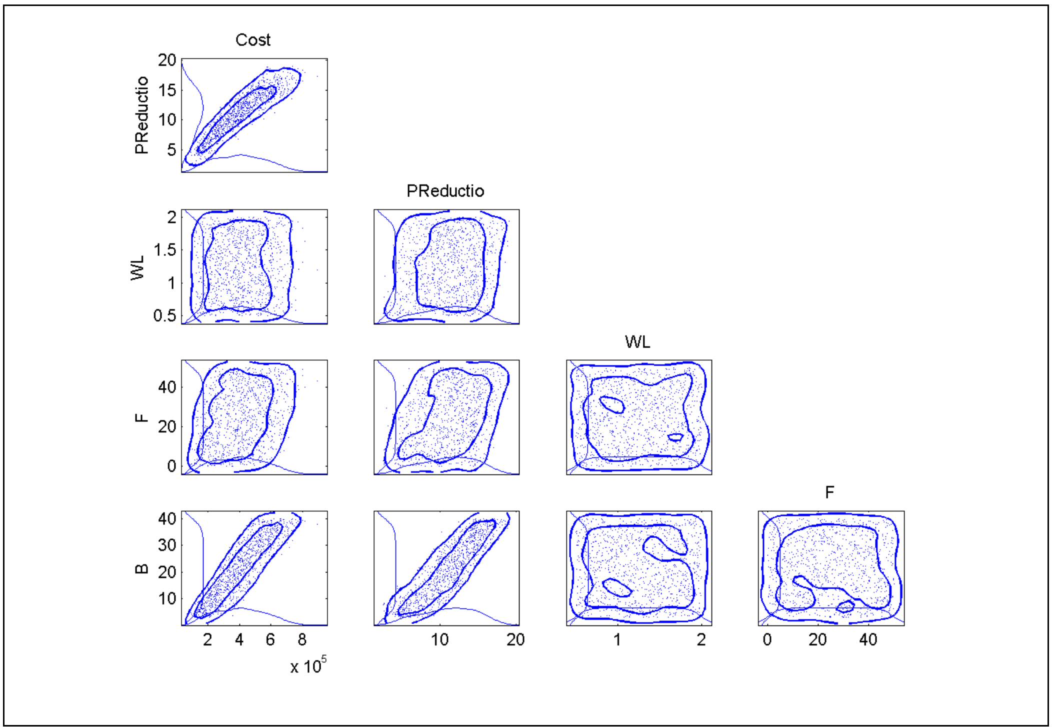


Figure 27: Bivariate plots of catchment model input and output variable MC-samples. Curves on the x- and y- axis are density functions of marginal distributions and closed lines are 50% and 95% confidence intervals of bivariate distributions.

Catchment Bayes network (Figure 28), which was learned from Monte Carlo simulation of the catchment model, was then used to calculate conditional marginal distributions of costs (459550 – 601175 €) and total phosphorus loading reduction percentage (14 – 15 %) (Figure 25) with given buffer strip width (31 – 35 m), wetland percentage (1.1 – 1.25 %) and forestation percentage (19 –25 %). Buffer strip width seems to be the most influential among the management actions (Figure 29).

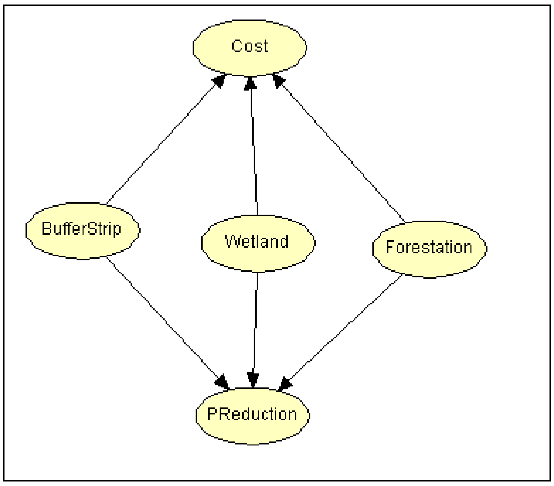


Figure 28: Graphical representation of the catchment Bayes network learned from Monte Carlo simulation of the catchment model. The network links costs, phosphorus loading reduction, buffer strip width, wetland percentage and forestation.

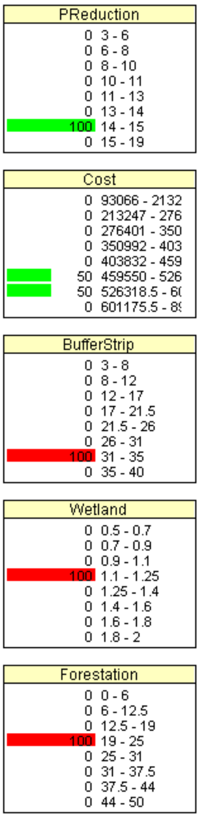


Figure 29: Conditional marginal distributions of costs and total phosphorus loading reduction percentage with given buffer strip width (31 – 35 m), wetland percentage (1.1 – 1.25 %) and forestation percentage (19 –25 %).

In the end, Bayes networks of LakeState and catchment models were linked (Figure 26) to estimate attainment of the designated water quality criteria or goal (Cyanobacteria summer maximum biomass) with the set of management options: buffer strip width, wetland percentage, forestation percentage and planktivorous fisheries management. Management options were implemented by decision nodes and the attainment of the water quality goal with a discrete change node and a utility node. A utility node relates a certain value to each of the states of the parent nodes, in this case 1 for the attainment and 0 for the non-attainment of the water quality goal.

Sarvala et al. (1998) and Helminen and Sarvala (1997) have published data that can be used to formulate conditional probability table for planktivorous fish biomass and grazing zooplankton (Crustacea) August-September average biomass. Linear regression was fitted to the data and zooplankton biomass was sampled using Monte

Carlo method and uniform distribution between observed ranges for planktivorous biomass. Conditional probability table (CPT) between planktivorous fish biomass and grazing zooplankton was calculated from the Monte Carlo sample and it was used to link these variables in the Bayes network (Figure 30).

We did not know exactly what kind of planktivorous fisheries management is needed to reduce planktivorous fish to a level that reduce Cyanobacteria biomass below water quality goal and what is the respective cost but the survey is going on to get these numbers. For demonstrative purposes we hypothesized fisheries management scenarios with related costs and impacts on planktivorous fish.

Bayes network, decision nodes and utility nodes together form an influence diagram that were used to study management decisions and their expected utilities in terms of Cyanobacteria summer maximum biomass and attainment or non-attainment of the water quality criteria (Cyanobacteria < 0.86 mg/l). As we can see, with the hypothetic fisheries management scenario 4 and with moderate catchment actions we obtain high probability (0.779) for the attainment of the water quality criteria (Figure 31).

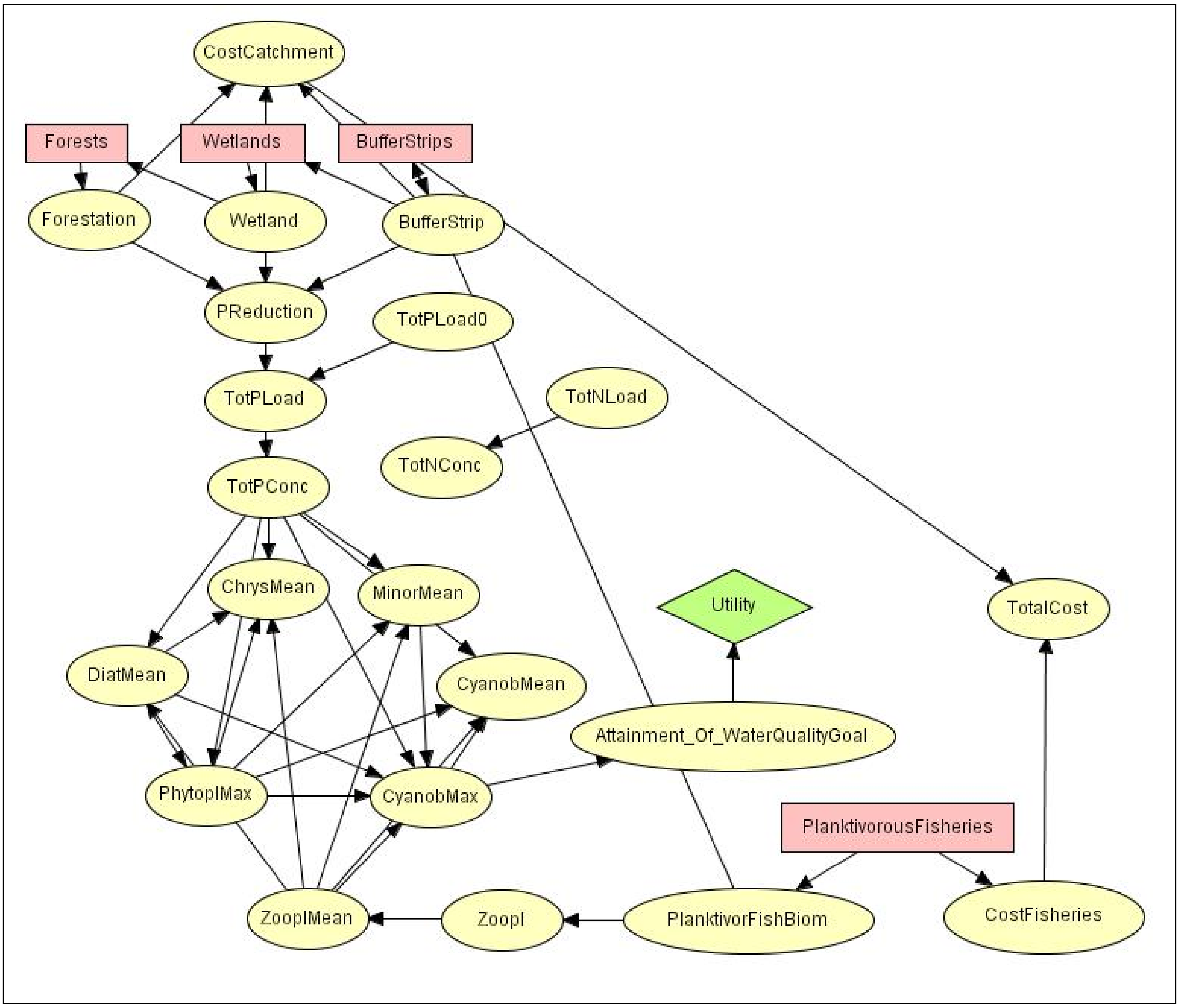


Figure 30: Bayes network structure that combines LakeState and watershed nutrient transport model and shows statistical relationships of the most important model inputs and outputs in the Lake Pyhäjärvi case.

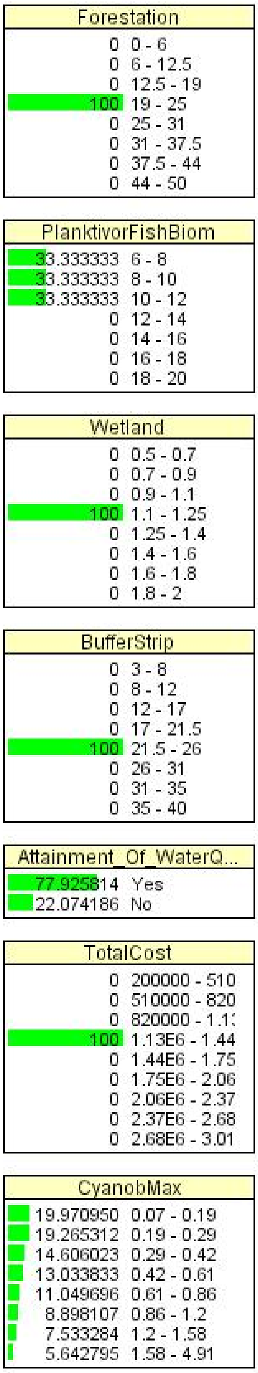


Figure 31: Conditional marginal distributions of attainment of water quality goal, costs, and Cyanobacteria (CyanobMax) summer maximum biomass with given buffer strip width (21 – 26 m), wetland percentage (1.1 – 1.25 %), forestation (19 –25 %) and planktivorous fisheries scenario 4 (biomass 6-12 kg ha^{-1}).

Benchmark evaluation of the integrated model

The ultimate focus of our Bayes network application to a multi-domain catchment-lake system was to make quantification of uncertainties and the transfer of uncertainties to risks which would be transparent and understandable to a wider audience, to policy makers and to stakeholders. That way we were able to serve rational decision making in water resources management and usage.

TheBayes network linking system was benchmarked with the additional benchmark questions for integrated models, developed by the integrated model work package of the BMW project.

No major shortcomings in the benchmarking questionnaire above indicated that Bayes networks are useful in the management decision making of multi-domain water resources.

Lake Veluwe

The water quality of the surface water is a function of in- and outgoing flows of water and in- and outgoing loads of substances and the processes working on those substances. For each of the models used to model Lake Veluwe, the WAQ module is used. The WAQ module that can be combined with Delft3D and SOBEK models but can also be used stand alone. The WAQ module makes it possible to calculate a wide range of water quality processes that incorporate the results of a FLOW module either from Delft3D or SOBEK or, in the stand alone version, predefined flow data. The WAQ module makes use of a so-called open processes library in which several processes are described, like biochemical reactions as the decay of BOD and nitrification, exchange of substances with the atmosphere, adsorption and desorption of contaminant substances and ortho-phosphate, resuspension of particles. Within these processes, the WAQ module supports over 140 substance, like 4 fractions of inorganic phosphorus, ammonium, nitrate and silica in the water column, heavy metals, temperature, oxygen. Furthermore, it also includes a sediment component in which differentiation of particulate inorganic and organic matter can be chosen and which interacts with the water column. Next to the WAQ module, all models presented below, expect the SOBEK2D model, incorporate BLOOM to be able to model algae growth and consequently algae blooms. BLOOM takes into account the following phytoplankton groups: diatoms, green algae and cyanobacteria and determines whether the species are phosphorus, nitrogen or energy limited. This is done be using a linear programming technique that maximize the total net phytoplankton production within each computational step. The availability of nutrients is calculated by WAQ and the energy is calculated based on light irradiance to the surface water. The specific growth rates of each phytoplankton type are given as fixed inputs to the model (www.delftsoftwarecentre.wldelft.nl).

Delft3D model

For lake Veluwe a modeling effort has been made with Delft3D. Delft 3D is a 2D/3D modeling system that can be used to investigate hydrodynamics, sediment transport and morphology and water quality for fluvial, estuarine and coastal environments. Within Delft3D, the FLOW module is the heart of model and can be used in a multi-dimensional way (2D or 3D). It can calculate non-steady flow and transport phenomena resulting from tidal and meteorological forcing on a curvilinear, boundary fitted grid. In 3D simulations the vertical grid is defined using the so-called sigma coordinate approach. This results in a high computing efficiency because of the constant number of vertical layers over the whole of the computational field. Within the FLOW module there are several standard features, like an advection-diffusion solver, a turbulence module, frictions laws (www.delftsoftwarecentre.wldelft.nl). For the lake Veluwe case, the FLOW module is combined with the WAQ module.

Input

As a start for the lake Veluwe case, a 3D grid was made and for the FLOW module a closed water balance for the period 1999-2000 was constructed, based on data of inflow from e.g. streams, precipitation, pumping station and outflow towards other lakes, infiltration and evaporation. For the WAQ module data on nutrients loads are necessary. However, this model ended up with a correct functioning FLOW module and only a first start with the WAQ and BLOOM module. Therefore, data on nutrients loads only existed out of made up and traceable quantities to check the technical function of the model. Nonetheless, this module can be extended to a complete FLOW-WAQ-BLOOM model.

SOBEK2DWAQ model

Lake Veluwe is used in this SOBEK2DWAQ model to validate a macrophyte module developed within WAQ. SOBEK2DWAQ simulations are based upon de Saint Venant equations describing 1D unsteady flow in channels and pipes, together with the complete shallow water equations for 2D flow. Robust implicit numerical techniques are applied efficiently to solve jointly the 1D and 2D equations and their links. The computed flow fields form the basis for the simulation the WAQ module (www.delftsoftwarecentre.wldelft.nl).

Input

For the model few checks on the hydrodynamics are made. One of these is the check on the water level. The water level shows to be kept on the desired level in both winter and summer of 1994, even though only a rough water balance is used (Meijers, 2005). The included water quality data concern two phosphor and nitrogen fraction, detritus from carbon, nitrogen and phosphor, chloride, oxygen and a fraction of inorganic matter. The inflow of these substances takes place at only two locations. This model can be used to give an idea of the functioning of the Lake Veluwe, but should be improved in the following ways:

1. the water balances of Lake Veluwe are needed to verify the model.
2. the schematization need to be broadened by including all streams and the concurrent WQ data.
3. If the model is to be used to model macrophytes properly, including their effects on water quality, extra bottom friction need to be introduced by the macrophytes in the lake.
4. Introduce a finer grid of 100*100 m or even 50*50 m (Meijers, 2005).

SOBEK1DWAQ model

This model is basically the same as the SOBEK2D model, but the flow is now calculated is 1D instead of 2D. This makes the model simpler, but maybe too simple.

Input

The model is run for the period 1999-2003 and water flows and water quality data for that period are incorporated. During validation of the model, it showed that the model is doing remarkably well. For the flow module, the water level is within range of the measured data and also the timing of the summer and winter water tables looks fine (Figure 32).

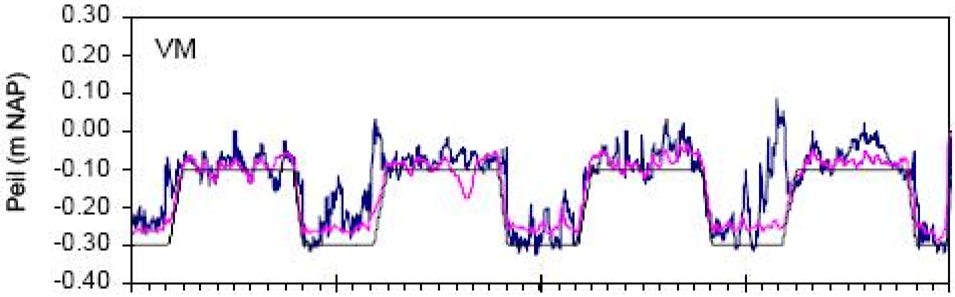


Figure 32: Water tables for lake Veluwe, the modeled flow data in pink, the measured data in blue and the desired water tables in black.

When looking at the water quality parameters, values seem to be in the right range and often follow the appropriate amplitude of oscillations. The only problematic parameter seems to be secchi depth (Figure 33). An explanation for this can be the restricted possibility to include fetch in the model.

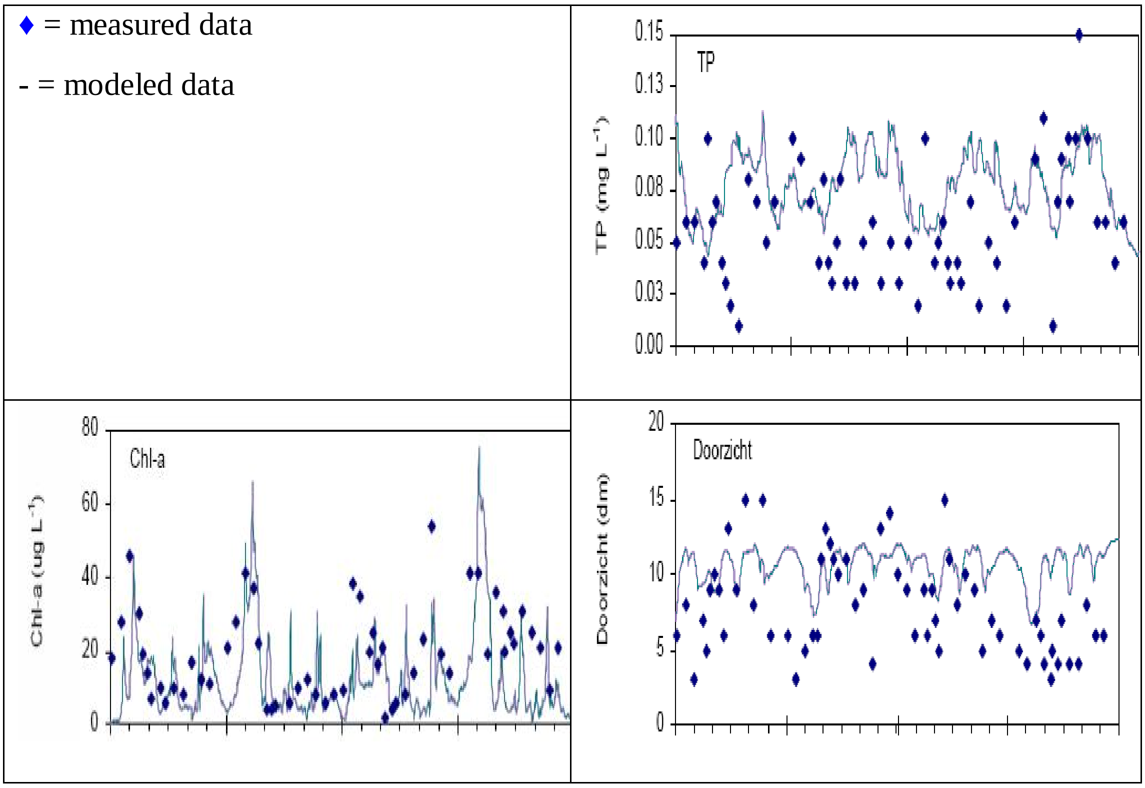


Figure 33: Calibration output for the SOBEKDWAQ model, at the upper right corner total P concentrations, lower left corner chlorophyll-a concentrations and lower right corner secchi depth, for the period 1999-2003 (Penning et al., 2007).

DBS and Delwaq-G

DBS and Delwaq-G are both stand alone versions of the earlier mentioned WAQ stand alone module. DBS and Delwaq-G differ only in two things: 1) DBS simulates the period 1976-1987 and Delwaq-G 1976-1992 and 2) DBS uses a four layer sediment module and Delwaq-G a ten layer sediment module. The extended sediment layer makes it possible to improve internal phosphate loading simulation. These water quality models use a coupling of three individual models: DELWAQ, BLOOM and a sediment layer model (for clarity reasons called SWITCH for both models). The coupling of these models makes it possible to model eutrophication problems in relation to water quality.

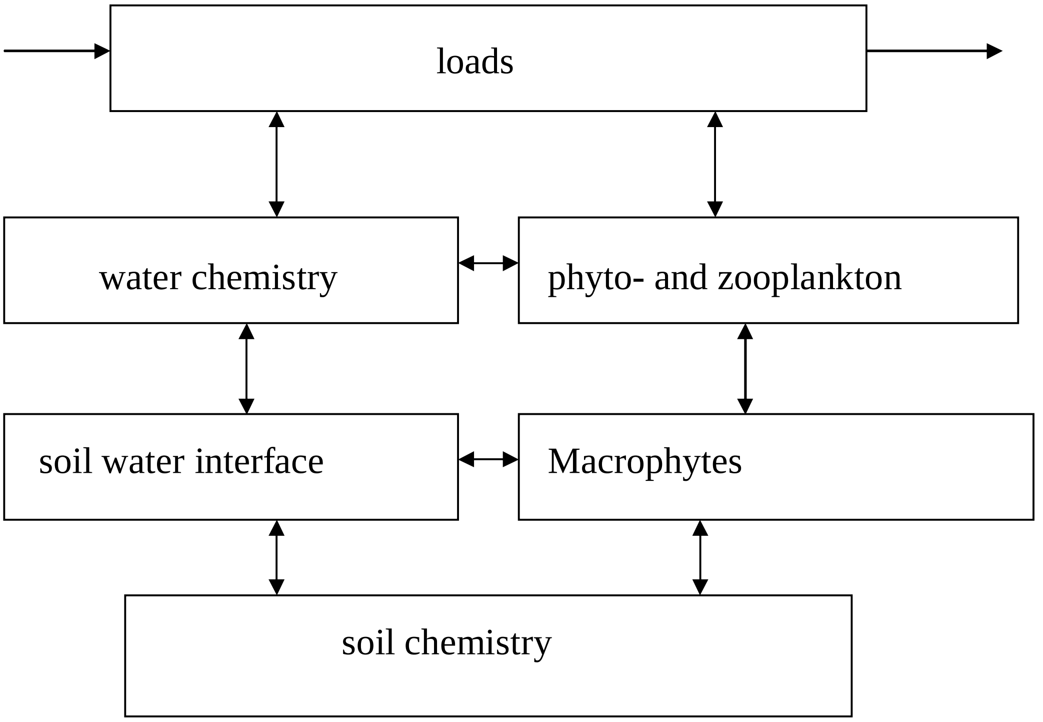
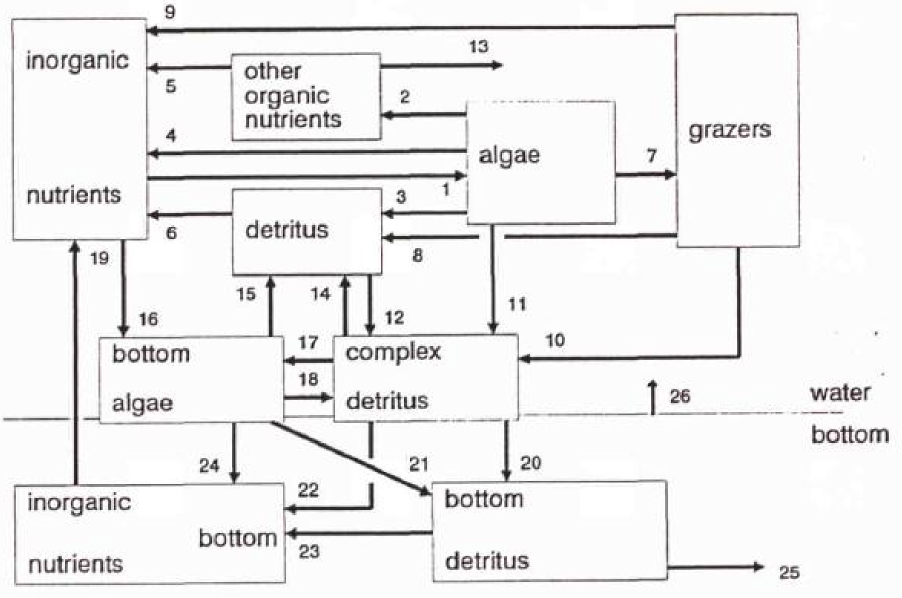


Figure 34: The interactions of the used model, however, the macrophyte module is still under development (after Los et al., 1994).

Input

The input needed for the model boils down to: 1) time dependent data regarding water flow, external loads and meteorology and 2) time independent information such as the chemical and biological system definition and the regional schematization. The time dependent processes are a combination of measured data, interpolated data and educated guesses. The processes involved in the model are depicted in Figure 35. However, this is a simplified figure, as only the routes between several compartments in the model are depicted and not the processes, like rearation, nitrification and the adsorption of phosphate. The regional schematization is simplified as two boxes: one for the shallow part of the lake and one for the deep part, with a diffusion coefficient between them and specified loads on each of the boxes.



1. Uptake of inorganic nutrients by phytoplankton. DBS assumes that ammonia is (completely) depleted first before the phytoplankton switch to nitrate.
2. Phytoplankton mortality flux to other organic substances.
3. Phytoplankton mortality flux to detritus.
4. Phytoplankton mortality flux to inorganic nutrients (autolysis). The sum of fluxes 2 + 3 + 4 equals the natural phytoplankton mortality.
5. Mineralization of other organic substances. It is assumed that this process is very slow and that it only takes place in the water column.
6. Mineralization of detritus. The actual rate depends not only on temperature, but also on the composition of detritus. It is maximal when there are sufficient nutrients relative to carbon.
7. Mortality of phytoplankton due to grazing.
8. Production of detritus due to zooplankton grazing (faeces).
9. Release of inorganic nutrients by grazers.
10. Production of complex detritus due to grazing by mussels (faeces).
11. Sedimentation of phytoplankton.
12. Sedimentation of detritus in the water phase.
13. Sedimentation of other organic nutrients, which become refractory.
14. Resuspension of complex detritus.
15. Resuspension of bottom phytoplankton.
16. Uptake of inorganic nutrients by bottom algae. This process is activated if process 17 cannot supply enough nutrients.
17. Uptake by bottom algae of dissolved nutrients released through mineralization of complex detritus in the water-bottom interphase.
18. Mortality flux of bottom algae to complex detritus.
19. Total flux of inorganic bottom nutrients to the water. This flux is influenced by all processes included within the bottom module of DBS.
20. Burial of complex detritus.
21. Burial of bottom algae.
22. Mineralization of complex detritus. For conceptual reasons this flux is handled by the bottom module; nutrients are not directly released into the surface water.
23. Mineralization of bottom detritus.
24. Part of bottom algae mortality flux to inorganic nutrients (autolysis).
25. Burial of bottom detritus. This process actually removes nutrients from the system.
26. Sediment growth. This results in dilution of the contents in sediment; usually it is set zero.

Figure 35: Processes involved in the used model (Los et al., 1994). Note: the term DBS is in this figure the same as Delwaq-G.

For the interaction between water and sediment, the following physical processes are included: sedimentation, net sedimentation, resuspension and burial. For the calculation of all processes occurring in the bottom, the sediment layer module is used. This module simulates the exchange of organic material, nutrients and dissolved oxygen between the sediment and the water columns, by using the following distinction of sediment layers: aerobic, denitrifying, upper reduced and lower reduced layer. It depends on the model how many layers of each type of layer is present.

Output

The output comprises concentrations of various water quality variables, such as secchi depth, oxygen, speciation of nitrogen and phosphorus, concentrations of 3 phytoplankton groups, the limiting factors for phytoplankton growth and so on in tables and graphs (Los et al., 1994).

DBS

The graphs presented below are from Van der Molen et al (1998). They show that total P concentrations are modelled reasonably well: trend and the amplitude of the oscillations are comparable, only at the end of the simulation the model and measured data diverge somewhat. The simulation of chlorophyll a give a comparable scene, however, the amplitude of the oscillations show more mismatch than is the case for total P concentrations. NO_3 concentrations show again quite good correlations between measured and modelled data.

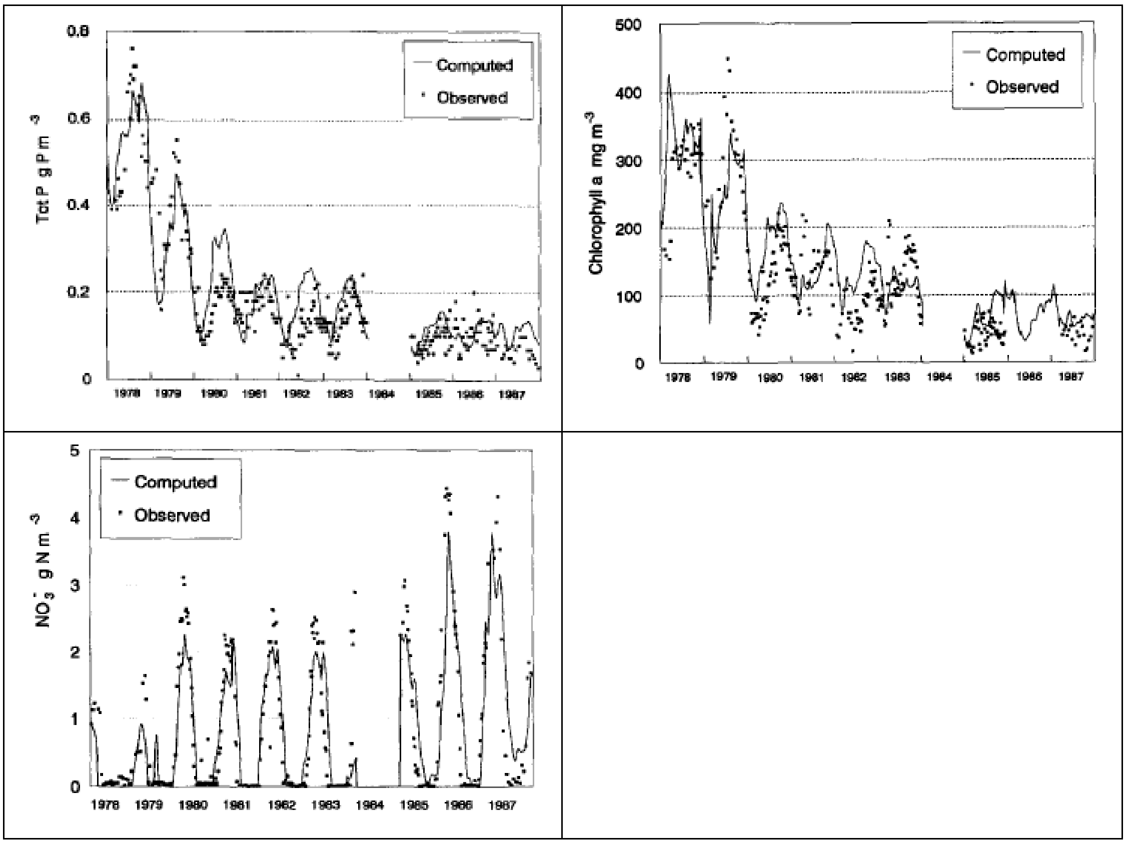


Figure 36: The calibration output of DBS given in Van der Molen et al (1998).

Delwaq-G

The calibration graphs are depicted in Figure 36. At the start of the simulation the trend for total P is visible in the model data, however, the peaks and dips are larger in the measured data. During the 1980s the model results correspond reasonably well with the model output but at the end of the simulation time total phosphate concentrations tend to be overestimated by the model. When looking at orthophosphate the trend is visible in the model data, however the yearly dynamics are not simulated but average values are in the data range. Chlorophyll a concentrations do show the trends in the simulated data, but the values are too high. The modelled secchi depth shows the same trends as the measured secchi depth, however, the measured data show higher values. The simulation of total N concentrations shows the same oscillations, but at the start of the simulations measured values are higher and towards the end they are lower. NO_3 concentrations seem to be modelled well.

For the purpose of simulation the effect of land and climate scenarios, past and future, only the DBS and Delwaq-G modules are more or less ready to be used as the period of 1976-1985 is important to hind cast regarding the measures taken to improve water quality. As Delwaq-G has the longer time series and proved to be better in internal phosphor loading (alas no dbs data available on this), Delwaq_G is chosen as model.

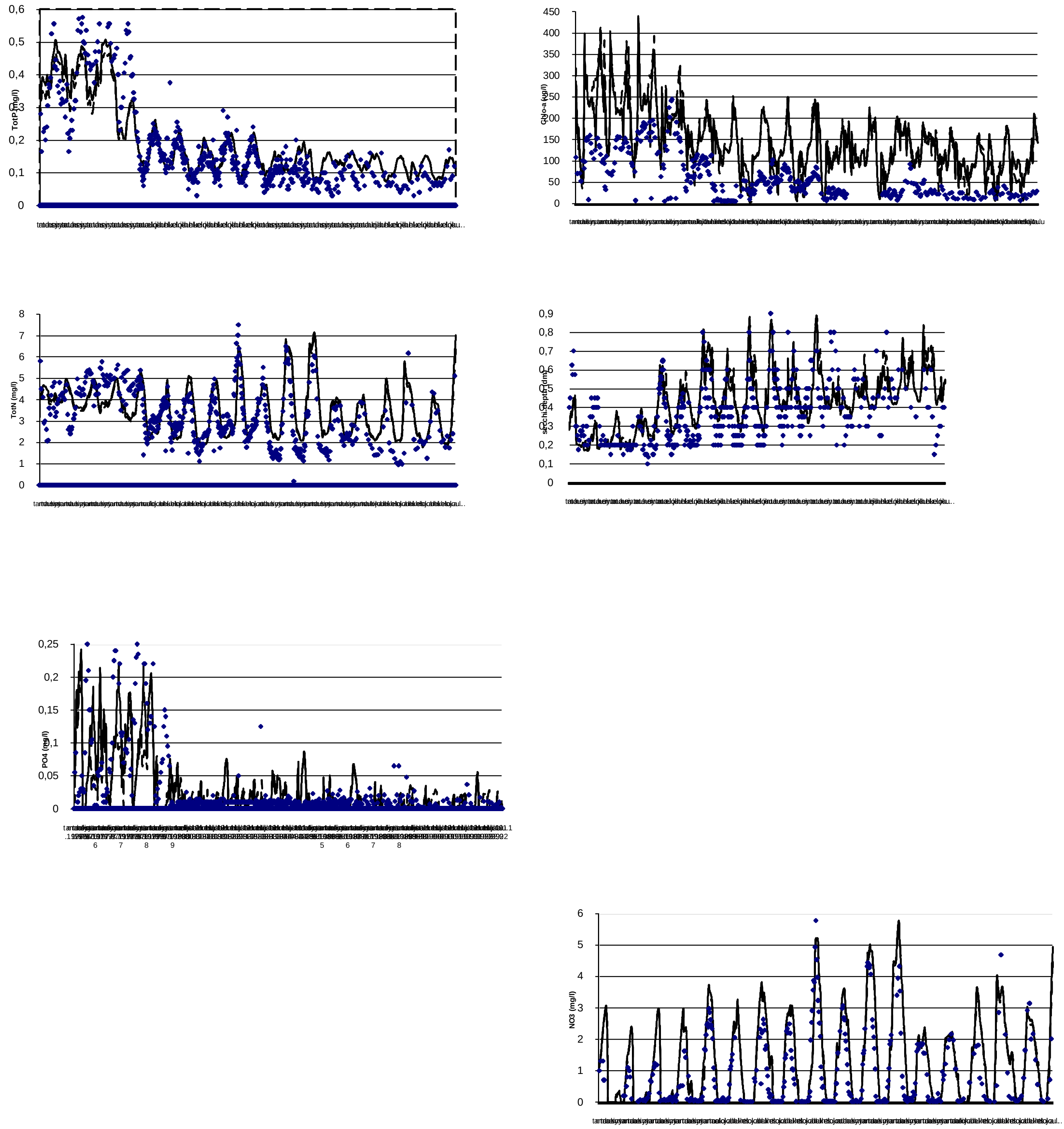


Figure 37: Depicts model output for Delwaq-G for the period 1976-1992. Substances showed are from top to bottom and from left to right concentrations in total P, chlorophyll a, total N, secchi depth, PO_4 and NO_3.

Norwegian lakes

MyLake application on set of lakes

Previous studies have used both historical datasets (e.g. Blenckner et al., 2007) and batch model simulations (e.g. Fang and Stefan, 2009) to analyze patterns of overall lake response (rather than individual lake response) to climate change. In this study we select the latter batch model simulation method and apply the MyLake model to simulate impacts of climate and river basin management changes on 181 Norwegian lakes. The general hypothesis is that temperature increase due to climate change may enhance algae growth in lakes and thus counteract management efforts of nutrient reduction to reach better water quality. We focus only on climate impacts directly on lakes and leave out any potential climate effects on river basin processes (landuse and farming practises, water discharge, and soil particle and nutrient transport).

The MyLake model v.1.2.1 (Saloranta and Andersen, 2007; Saloranta 2007) was used to simulate a larger set of Norwegian lakes in one “batch”. This set of lakes is based on the Euregi lake dataset, from which lake information for model setup was obtained (latitude, longitude, altitude, surface area, maximum depth, and water transparency (secchi depth)). Conical lake shape was assumed, and the depth where the secchi-disk disappears from view (secchi depth z_s) [m] was related to water light attenuation coefficient ε [m^{-1}] as in Hondzo and Stefan (1993):

i) $$\varepsilon = 1.84 / z_s \qquad (37)$$

For the meteorological forcing we used a high-resolution data set of surface climate (hereinafter referred as CRU-climatology; New et al., 2002) giving 1961-1990 monthly means of air temperature, precipitation, wind speed, relative humidity and sunshine duration in a grid with 10’ resolution in latitude and longitude.

For the hydrological forcing we used as a template the observed time series of water discharge, as well as suspended sediments and total phosphorus (P) concentrations from Vansjø-Hobøl river basin (station Kure) from year 2002. The discharge time series was scaled with the volume of each simulated lake so that the water residence time became equal to one year. Similarly, the total P time series were scaled for each simulated lake so that the discharge weighed mean concentration of total P in the inflow became 50 mgm^{-3}. The P scaling factor was also applied on time series of suspended solids and dissolved organic P. These one year long forcing time series were in the model simulations repeated for each simulation year, in order to keep the forcing similar on each simulation year. The discharge weighed mean concentration of total P in the unscaled inflow time series (Figure 38) was 48 mgm^{-3} and the yearly total P load 5.3 tons. Similarly the concentration of dissolved organic P in the unscaled inflow time series was kept constant at 7 mgm^{-3} throughout the year). 66 % of the total P load was in potentially bioavailable form (i.e. dissolved P and readily desorbable P in particles).

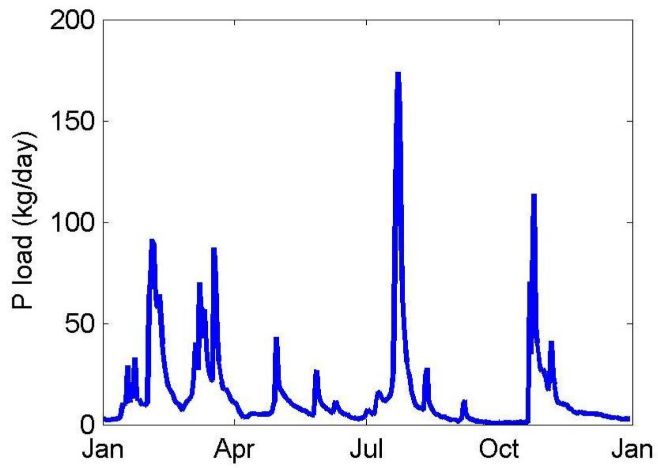


Figure 38: The (unscaled) time series of total P load in Vansjø-Hobøl river basin (year 2002, measured at station "Kure").

The set of lakes considered for the simulations consisted of 345 lakes in the Euregi data set with maximum depth larger than 4 m. For each lake the closest grid point in the CRU climatology was used as starting point, and the altitudes of that point and the neighbouring eight grid points were compared to the actual altitude of the lake. The climatological time series from the grid point with the least altitude difference was selected for model forcing. If the altitude difference was larger than 50 m (or if no forcing data existed) the particular lake was not included in further simulations. After this selection process 181 (52 %) of the original 345 lakes were accepted for the final batch simulations.

These simulated 181 lakes ranged from 58 to 70 °N in latitude, from 0.1 to 200 km^2 in surface area, from 0.4 to 11 m in the secchi depth, from near sea-level up to 1100 m.a.s.l. in altitude, and from 4 to 300 m in maximum depth.

The monthly climatological means and the approximately weekly observations of P concentration in the water discharge were linearly interpolated to form model forcing time series with daily resolution. The model was run for a 10.5 year period, but only results from the third whole year and last years were considered in further analysis. The first 2.5 years were regarded as model spin-up period, giving the model this period time to ”forget” the initial conditions. The 7 year time difference between the third and the last years was used to detect possible long-term developments in the lake properties (e.g. due to changes in internal P loading from sediments).

Nominal model parameter values were generally set as in Table 2 in Saloranta and Andersen (2007) (except for sinking velocities of chlorophyll-a and suspended solids which were both set to 0.2 $mday^{-1}$).

METHODS II - Climate and management scenarios

Global change scenarios

SRES (Special report on emission scenarios, IPCC 2000) greenhouse gas emission scenarios for Finnish conditions were developed in FINSKEN project (Carter et. al 2004). Global driving factors of environmental change for the scenarios were specified by IPCC (Intergovernmental Panel on Climate Change) in the Special Report on Emission Scenarios (2000). The SRES scenarios include different storylines or scenario families A1, A2, B1, B2 (Figure 39). The AIB-scenario, which is used in this project with WSFS-Vemala and Coherens lake model for Lake Pyhäjärvi, is a sub-scenario in family A1.

To produce the climate scenario the emission scenarios are given as input to climate models, which is then run globally or regionally. These computer models describe the climate systems based on physical equations. Besides the emission scenario used also the climate model used has a large influence on the projected changes in meteorological variables. In the near future, different emission scenarios do not yet differ markedly and thus in the largest differences are between different climate models. Because of limited computing power and data many phenomenon in the climate models have to be described in a simplified way. Since these problems are solved in different ways in different models, the model results differ from each other (Ruosteenoja 2007). The IPCC Fourth Assessment Report (IPCC 2007) includes climate scenarios from 23 global climate models using with a variety of emission scenarios. The range of global temperature change by the end of the century with different climate models and emission scenarios is shown in Figure 40 (IPCC, 2007).

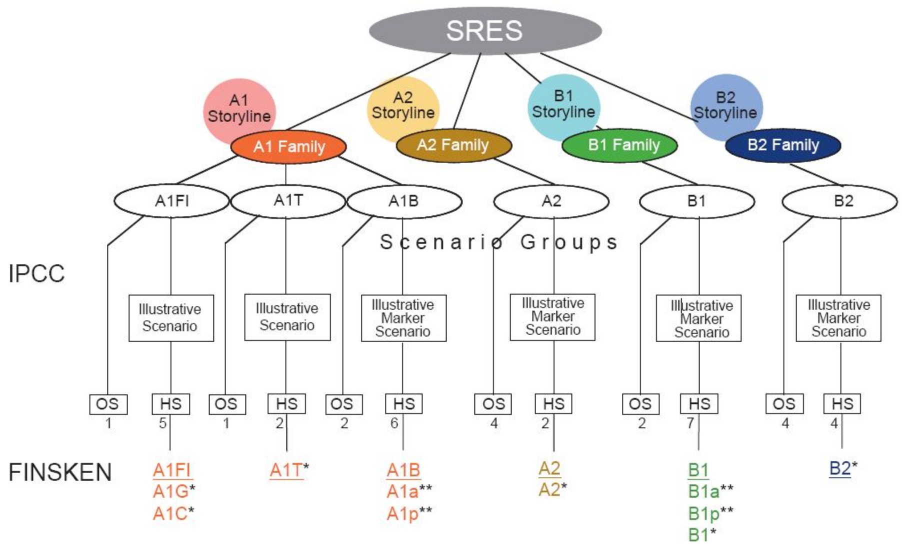


Figure 39: SRES scenarios

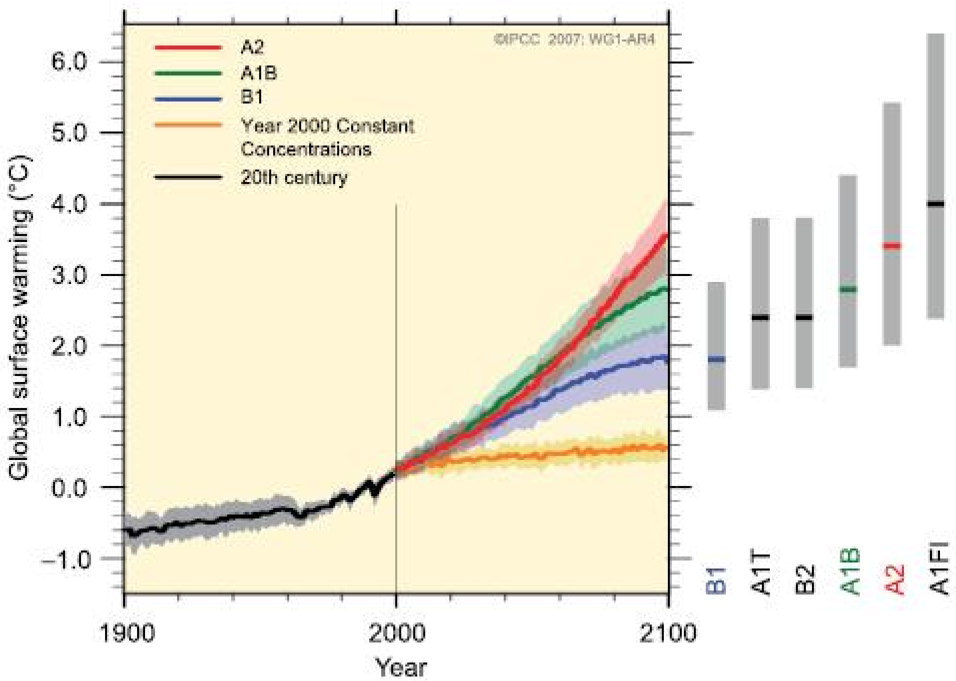


Figure 40: Solid lines are multi-model global averages of surface warming (relative to 1980–1999) for the scenarios A2, A1B and B1, shown as continuations of the 20th century simulations. Shading denotes the ±1 standard deviation range of individual model annual averages. The orange line is for the experiment where concentrations were held constant at year 2000 values. The grey bars at right indicate the best estimate (solid line within each bar) and the likely range assessed for the six SRES scenarios. (IPCC 2007)

Lake Pyhäjärvi

Climate scenarios

The climate scenarios for temperature and precipitation change used in the WSFS (Watershed Simulation and Forecasting System) and Coherens models in Finland were average scenarios calculated as the average of 19 global climate models with SRES A1B emission scenario (IPCC 2000) for four time periods. The time periods were 2010-29, 2020-49, 2040-69 and 2070-99 with the control period of 1971-2000. The 19 models were a subset of the 23 global climate models included in the IPCC Fourth Assessment Report (IPCC 2007). Four models from the IPCC Fourth Assessment Report were omitted because they either has scenarios for only one SRES emission scenario, they had an incorrect position for continent and oceans in Europe and North Atlantic or they could not reproduce the current climate in Finland satisfactorily (Ruosteenoja 2007). The scenarios were calculated by Finnish Meteorological Institute as part of ACCLIM-project (Ruosteenoja 2007) and were provided as monthly changes in temperature and precipitation in a grid over Finland. The use of a multi-model mean value for temperature and precipitation change provides more stable and arguably more reliable results than use of any single model.

The SRES A1B emission scenario includes assumptions of very rapid economic growth, global population that peaks in mid-century and declines thereafter, the rapid introduction of new and more efficient technologies and a technological change in the energy system balanced across all sources. In this scenario the greenhouse gas emissions grow until the middle of the 21st century and then begin to decrease. The emission and the greenhouse gas concentrations by the end of the century are mid-range in the A1B emission scenario compared with the other SRES scenarios. (IPCC, 2000) ´

Scenarios used in WSFS-VEMALA and Coherens model were:

- ▫Period 0: present situation
- ▫Period 1: A1B-scenario average scenario* of period 2010-2039
- ▫Period 2: A1B-scenario average scenario*2020-2049
- ▫Period 3: A1B-scenario average scenario* 2040-2069
- ▫Period 4: AIB-scenario average scenario* 2070-2099
- *average of 19 global climate models

Present situation includes data for period 1.1.2001-31.12.2007. In periods 1-4 the temperature and precipitation of period 2001-2007 is changed according to the average scenario for the period in question.

The method used to transfer the climate change from the global climate model to the hydrological model WSFS is the delta change approach (also called the perturbation or change factor approach). In the delta change approach the monthly changes of temperature (in ºC) projected by the climate scenarios are added to the observed temperature and monthly changes of precipitations (in %) are multiplied with the observed precipitation values of the reference period. Discharges and water levels are then simulated with the changed temperatures and precipitations as input and compared with the simulated values of the reference period.

For Coherens lake model river loading data (flow and concentrations of suspended matter and phosphorus) and meteorological data (air temperature and evaporation minus precipitation rate) for the periods 0 to 4 were got from WSFS calculations (Figure 41). The needed input data in Coherens model includes also data which was not offered by the WSFS. Wind, cloud coverage and air humidity data were from present situation in each scenario calculation and they were got from the Finnish Meteorological Institute. River water temperature data for the present situation was got from sampling database (HERTTA). This present situation data was also used for the climate change simulations.

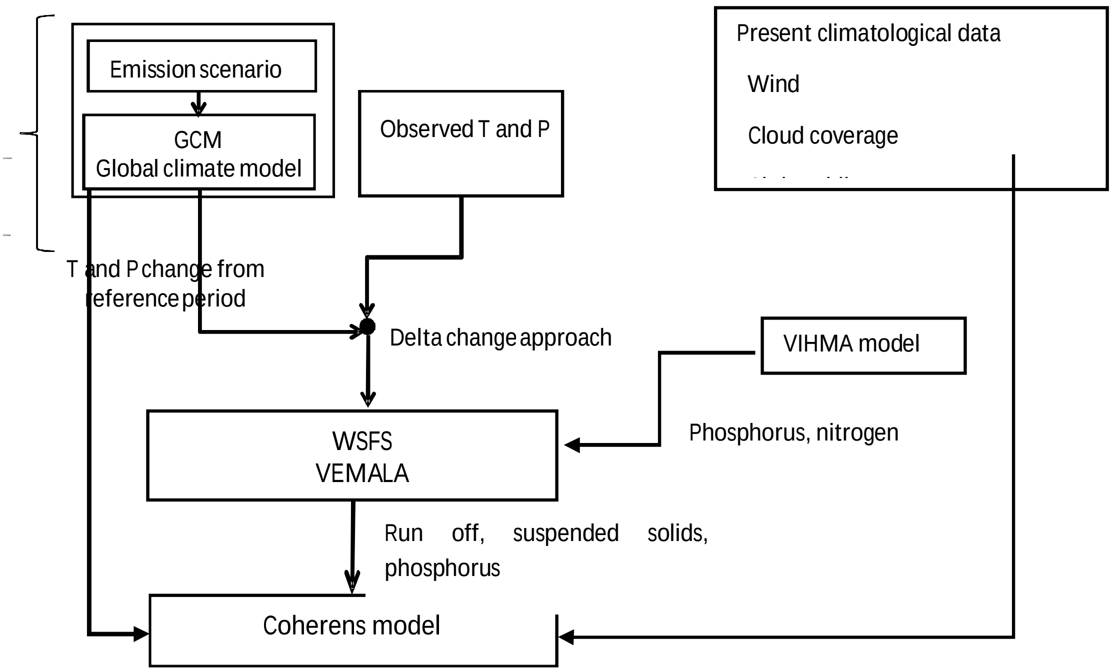


Figure 41: Data flow from Global Circulation Model to WSFS-VEMALA watershed model and Coherens Lake model in climate change simulations.

Input data for different time periods for Coherens model is shown in figures 42-45.

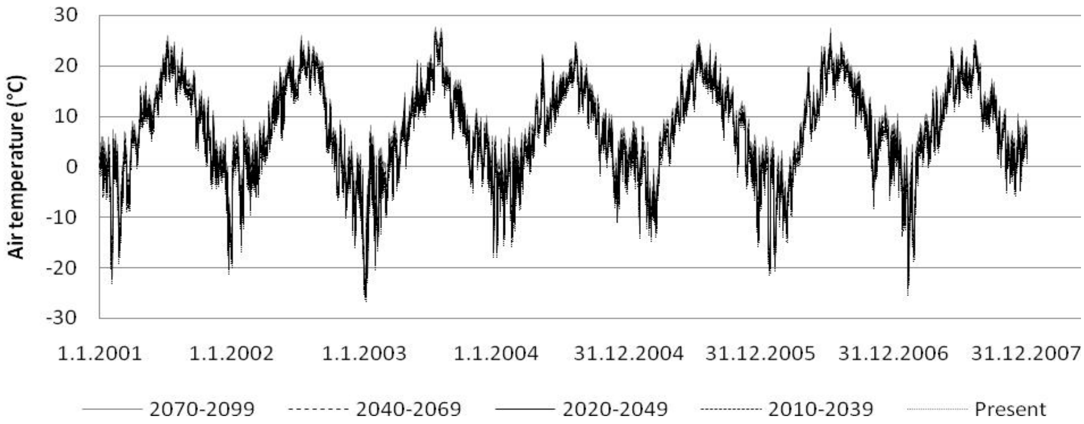


Figure 42: Air temperature data for Coherens-model.

From figure 42, it can bee seen that air temperature is highest in 2070-2099 and lowest in present situation. Difference is about 0-5 ^{0}C .

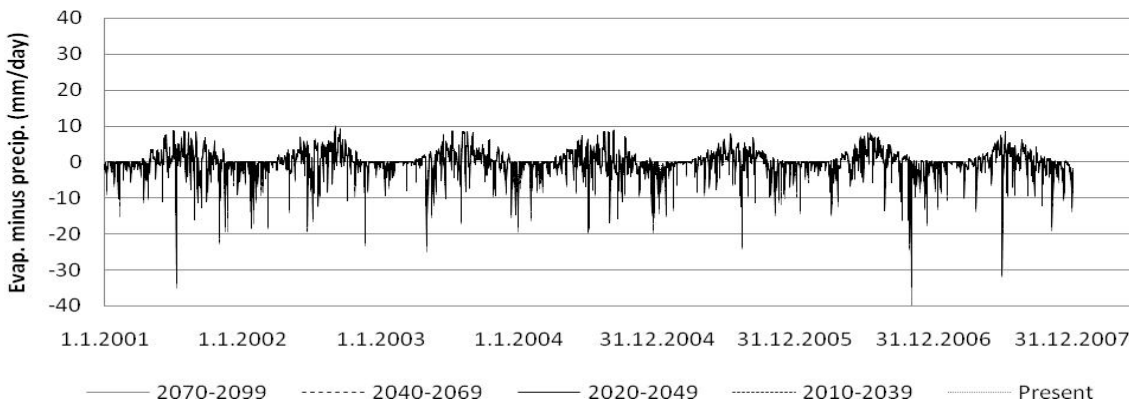


Figure 43: Evaporation minus precipitation data for Coherens model.

Evaporation minus precipitation values (Figure 43) are negative especially in winter, because neglible evaporation in winter. Peaks are greatest in 2070-2099 simulation as the winter precipitation increases. Increasing negative values indicate increasing evaporation and diminisheing rain during the summers towards the end of the century.

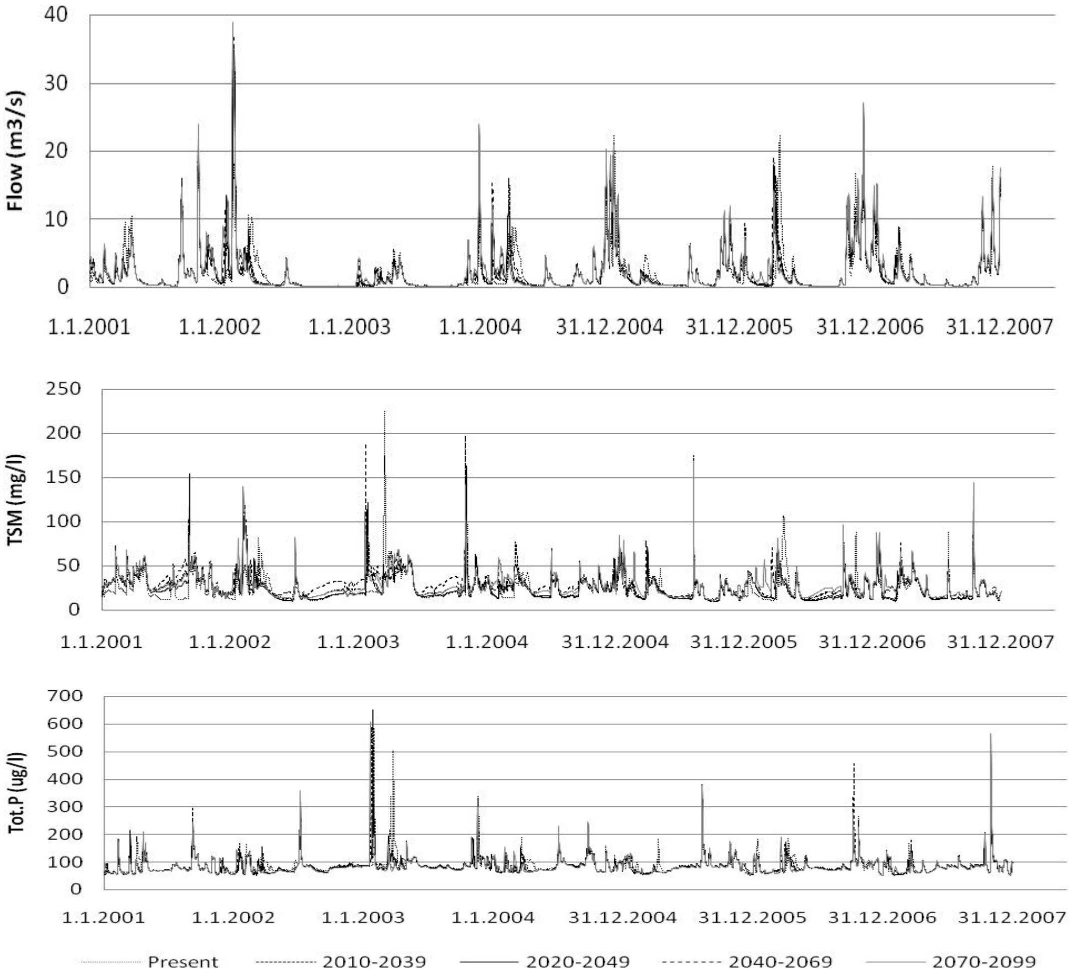


Figure 44: Flow, suspended matter and total phosphorus concentrations in River Yläneenjoki.

From figures 44 and 45 it can bee seen how Yläneenjoki brings more water the lake as Pyhäjoki now and also in future. Also concentration of suspended solids and total phosporus are higher and will higher there. The peak flows increase towars the end of the cencury and a clear shift towards winter floods is seen in several years. The change is more strong in Yläneenjoki as in Pyhäjoki. Similarly the time distrubtion of suspended solids and phospohorus load is markely shifted, especially in Yläneenjoki.

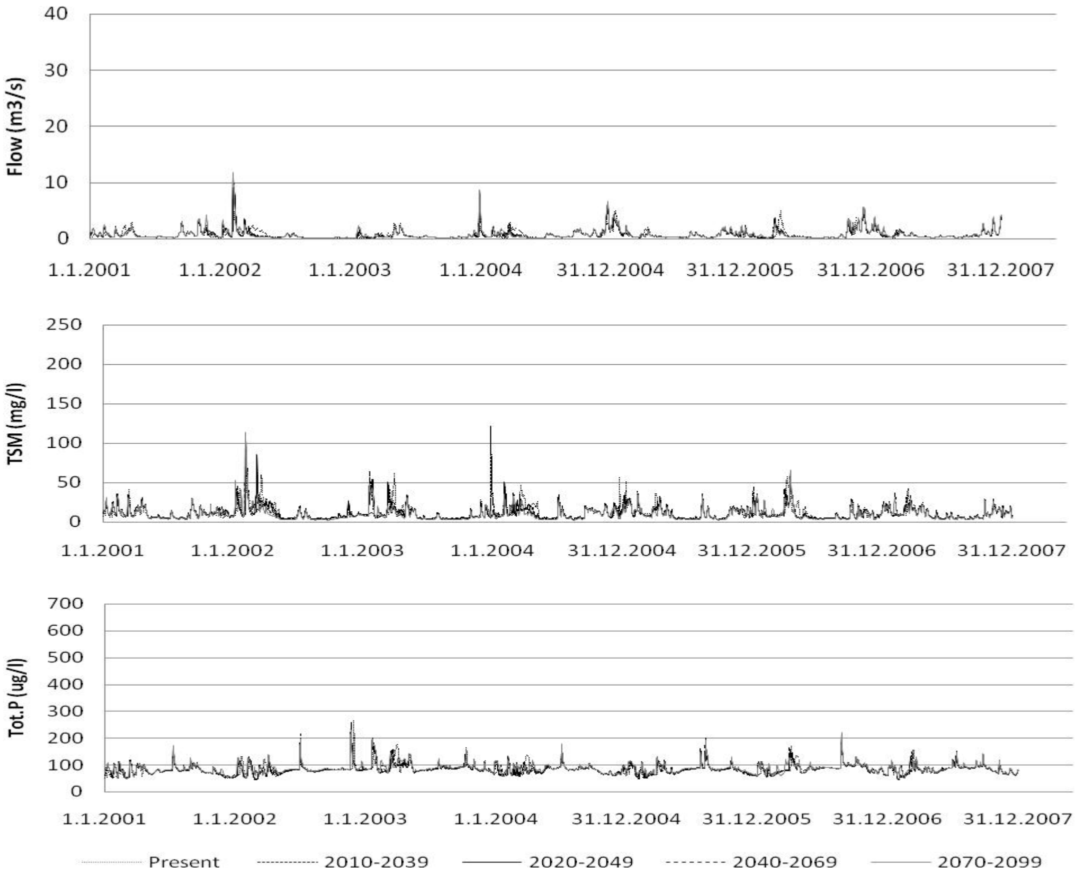


Figure 45: Flow, suspended matter and total phosphorus concentrations in River Pyhäjoki.

Management scenarios

Earlier SWAT model has been used for studying loading effect of constructed wetlands (Koskiaho et al. 2003). The effects of constructed wetlands were assessed by comparing the material fluxes produced by three different scenarios.

Also WSFS-VEMALA model was used to predict the effect of three types of management actions on Lake Pyhäjärvi watershed. The management actions have been chosen within TEHO-project lead by South-West Finland. All together 10 different scenarios have been simulated with WSFS-VEMALA-model. They were

- Buffer zones implemented as in master plan on 4152 fields with total surface area of 17500 ha (Scen 1)
- Increasing the winter time vegetation cover by 30, 50 and 70% from present. Two options were used: increment focused on all fields, increment focused on sloping fields. (Scenarios 2-7)
- Changes in animal husbandry and reduction of manure usage on fields. Loading reduced totally (Scen 8), loading doubled from present (Scen 9) and characteristic P load of manure fields reduced from class 14 to class 8-14 (Scen 10).

For lake model exercise the WSFS-VEMALA results from Scen 8 were used as input to 3D Coherens lake model.

Lake Veluwe and its river basin

Land use scenarios

The National Institute for Public Health and the Environment (RIVM), performed exploring studies to examine the effects of future developments in society on nature and landscape. This is done by the use of four scenarios, which describe two different contradictory trends: “individualization versus cooperation” and “globalization versus regionalization”. The four scenarios are named: Individualistic World (IW), Cooperating World (SW), Individualistic Region (IR) and Cooperating Region (SR). The land use changes are predicted for the year 2030. The trend for all scenarios is that the rural area decreases and urban and nature areas increase (Figure 46).

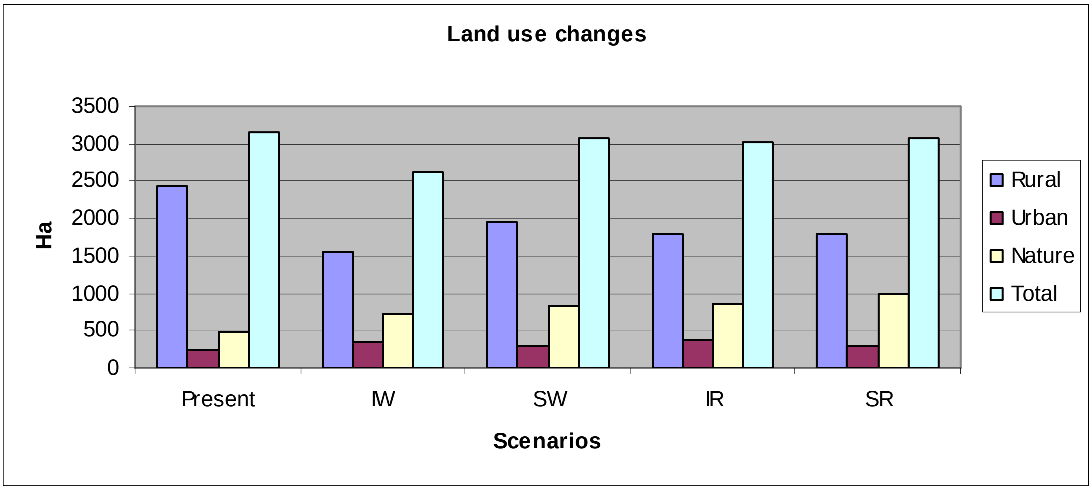


Figure 46: This figure shows the expected changes in land use: in all scenarios rural area decreases and urban and nature area increase. It depend on the scenario how the land use changes. Total means the total of the of the three land uses showed here, neglecting other types of lands uses that consist out of mixed forms.

Coupled to changes in land use are changes in emissions. If rural lands are altered in urban area, sewer water treatment plants (SWTPs) should clean more water and additionally, more overflow is likely. Moreover, as shown in Table 8, the change from rural to nature area implies less nutrients into the system and therefore in the discharging waters. Taking data from Meeusen et al. (2008), Statline (http://statline.cbs.nl/, the Dutch bureau on statistics) and Emmisie Registratie (www.emissieregistratie.nl, Dutch project on measuring nationwide emissions), the following relative changes in nitrogen and phosphorus are calculated and used to calculate altered loads upon lake Veluwe:

Table 8: Relative changes in N and P due to land use changes.

Relative change		Nature	Urban
IW	Nitrogen	-65	138
	Phosphorous	-41	585
SW	Nitrogen	-55	11
	Phosphorous	-25	351
IR	Nitrogen	-61	90
	Phosphorous	-34	514
SR	Nitrogen	-54	-10
	Phosphorous	-23	293

Climate scenarios

The climate scenarios put forward by KNMI (Dutch meteorological institute) are seen as guiding for the local climate of the Netherlands (Table 9). The climate scenarios exist out of four scenarios, which boil down to changes in air circulation patterns, temperature, precipitation and wind strengths. The scenarios are not based on the IPCC scenarios in which changes are based on technical and socio-economic developments. The KNMI scenarios are applicable to the four IPCC scenarios, as the scenarios are based on a broad spectrum of climate models that take increasing temperatures as its starting point. Scenarios consider the period onto 2050.

Table 9: Climate scenarios KNMI (Klein Tank en Lenderink, 2009)

Scenario	G	G+	W	W+
Wereldwijd				
Wind circulation patterns	Unchanged	Changed	Unchanged	Changed
Temperature (°C)	+1	+1	+2	+2
Summer				
Average temperature (°C)	+0,9	+1,4	+1,7	+2,8
Average precipitation (%)	+2,8	-9,5	+5,5	-19.0
Frequence of wet days (%)	-1,6	-9,6	-3,3	-19,3
Average precipitation on wet day (%)	+4,6	+0,1	+9,1	+0,3
Precipitation on 1% wettest day (%)	+12,4	+6,2	+24,8	+12,3
Potential evaporation (%)	+3,4	+7,6	+6,8	+15,2
Winter				
Average temperature (°C)	+0,9	+1,1	+1,8	+2,3
Average precipitation (%)	+3,6	+7.0	+7,3	+14,2
Frequence of wet days (%)	+0,1	+0,9	+0,2	+1,9
Average precipitation on wet day (%)	+3,6	+6.0	+7,1	+12,1
Percipitation on 1% wettest day (%)	+4,3	+5,6	+8,6	+11,2

Typical changes for the Netherlands

Temperature

Evidence show that temperatures of the Dutch regional climate have been risen twice as fast as global temperatures and this seems to be a trend rather than a natural oscillation (Figure 47). Increased temperatures in winter are caused by increased western, land inwards, winds and in summer increased incoming summer radiation due to less clouds and a cleaner atmosphere are the likely causes. To convert air temperature to water temperature, a conversion is used from EU project *SCENES* (Segrave et al., *in press*).

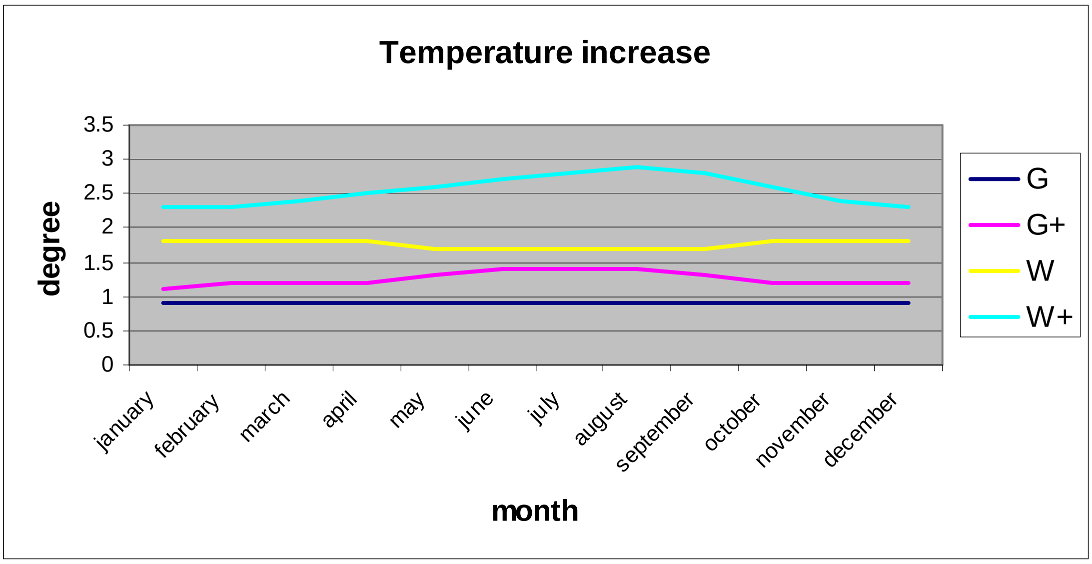


Figure 47: Changes in temperature for each month for 2050 by taking the period 1976-2005 as reference (KNMI, 2006)

Precipitation

Precipitation amounts in winter half year are likely to increase because a trend gearing towards this direction is already detected in the past couple of years (Figure 48). For the summer half year it is expected that the precipitation intensity will increase (according to some studies up to 14% per degree) as warmer air can contain more water than colder air. As the air is cooling, more precipitation will be formed. Also, increase in turbidity in a precipitation cloud will increase the amount of water that can be precipitated above a certain area. However, more intense precipitation is expected but the occurrence of precipitation will decrease because of changes in wind direction (coming from east (dry winds) instead of west (more humid air))(KNMI, 2006).

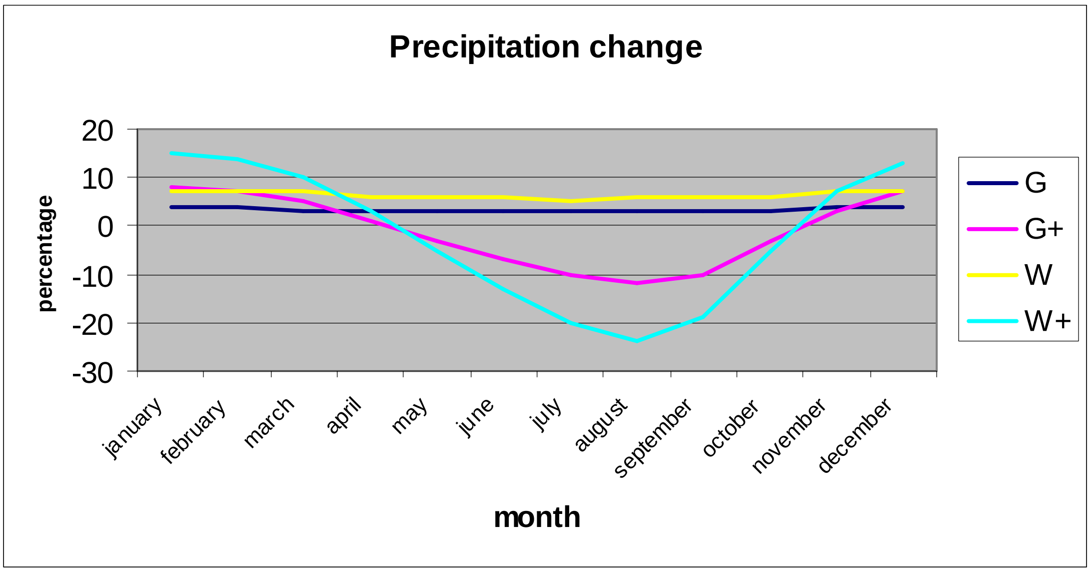


Figure 48: Changes in precipitation for each month for 2050 by taking the period 1976-2005 as reference (KNMI, 2006)

Norwegian lakes

To investigate the effects of increased air temperature and reduction in total P loading into the lakes for the different lake variables we run with MyLake nine different scenarios:

Scenario 1: control scenario (no changes);

Scenario 2: no change in air temperature, -20% decrease in P loading;

Scenario 3: no change in air temperature, -40% decrease in P loading;

Scenario 4: +2° C increase in air temperature, no change in P loading;

Scenario 5: +2° C increase in air temperature, -20% decrease in P loading;

Scenario 6: +2° C increase in air temperature, -40% decrease in P loading;

Scenario 7: +4° C increase in air temperature, no change in P loading;

Scenario 8: +4° C increase in air temperature, -20% decrease in P loading;

Scenario 9: +4° C increase in air temperature, -40% decrease in P loading;

The lake thermodynamic output variables recorded in the simulations included

- monthly mean water temperature [° C] in the 0-4 m layer

- maximum monthly ice thickness [m]
- number of days with ice cover per month
- number of lake ice freezing or melting episodes per month
- monthly mean depth of epilimnion (i.e. pycnocline) [m]

1 monthly mean effective light climate in the epilimnion [W m^{-2}]

The recorded lake water quality output variables included

- monthly mean as well as maximum total P concentration in the 0-4 m layer

1 monthly mean as well as maximum chlorophyll *a* (Chl) concentration in the 0-4 m layer

1 The monthly mean effective light climate in the epilimnion I_e was calculated as in Kalff (2002, eq. 38):

$$\text{i)} \quad I_e = \frac{I_0 - I_z}{\ln\left(I_0 / I_z\right)} \equiv \frac{I_0\left(1 - \exp(-\varepsilon z_{epi})\right)}{\varepsilon z_{epi}} \qquad (38)$$

1 where I_0 and I_z denote irradiance [W m^{-2}] at surface and bottom of the epilimnion, respectively, and z_{epi} the depth of the epilimnion (i.e. pycnocline).

Results

Scenario runs for Lake Pyhäjärvi

The results from earlier the SWAT model application suggest that even high increase of the number of constructed wetlands does not lead to substantial load reductions if their dimensioning is inadequate (Koskiaho et al. 2003). The most realistic and cost-effective approach is probably to try to concentrate the CWs in such parts of the catchment, where the above area is not very large and input concentrations are high (high field-%, steep slopes, high number of farms with animal husbandry).

In Figure x. we can see WSFS-VEMALA model results for Pyhäjärvi catchment for present situation and for A1B scenario for four time periods. The results show that in the climate change situation long term mean phosphorus loading into Pyhäjärvi will have following changes:

1. phosphorus load peak considerably decreases during the spring time, because snow melt flood peak decreased and shifted to winter months due to the rise of winter temperatures,
2. there is no any more clear phosphorus load peak, because there is no any more clear snowmelt peaks,

3. phosphorus loading is elevated during the all winter months and late autumn months (November, December) due to the both elevated runoff and higher concentrations, because soil is more often without the snow cover during the winter and erosion is more intensive,
4. phosphorus inflow loading during the summer stays about at the same level.

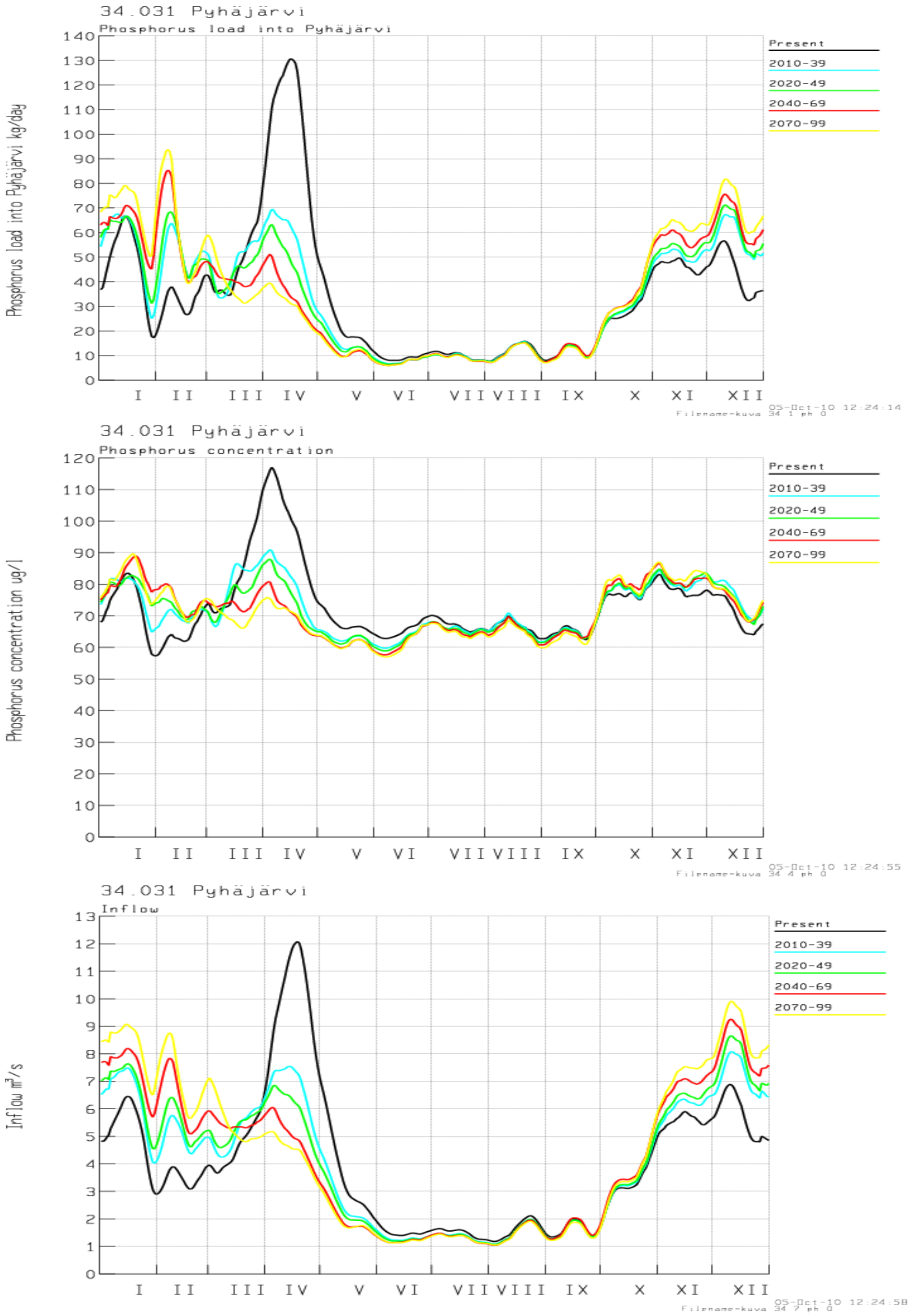


Figure 49: Phosphorus loading, phosphorus concentration and inflow to the Lake Pyhäjärvi simulated by WSFS-VEMALA for present situation and A1B climate change scenario.

The outputs from WSFS-VEMALA were used as inputs for 3D Coherens Lake model. The results are seen in figure 50. It shows the future surface water temperature in Lake Pyhäjärvi with concentration of suspended solids and total phosphorus towards the end of the century. It can be seen the maximum surface water temperatures will increase with few degrees and the cold water periods will be shorter that presently.

The concentrations of suspended solids and consequently the concentrations of total phosphorus will be markedly higher in winter time as in present climate.

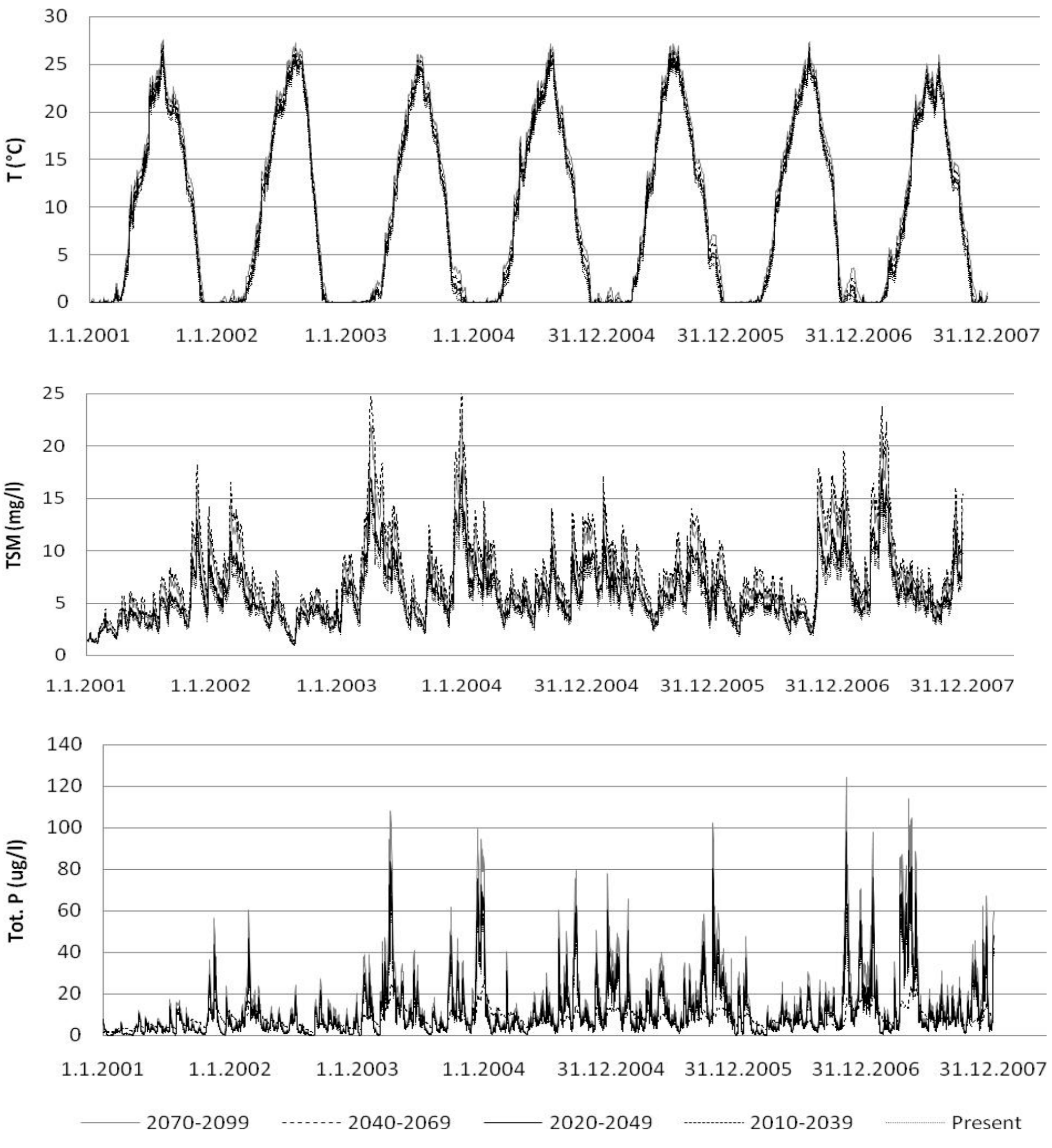


Figure 50: Simulated temperature, suspended matter concentration and total phosphorus concentration in surface water at the deepest point of Lake Pyhäjärvi.

WSFS-VEMALA model was run for Pyhäjärvi catchment also for a land-use scenario for period 2070-2099. In this scenario the loading from manure is remowed. Results of this scenario (scenario 8) were used as input data in Coherens model. Coherens model results are shown in Figure 51 for surface water total phosphorus concentration.

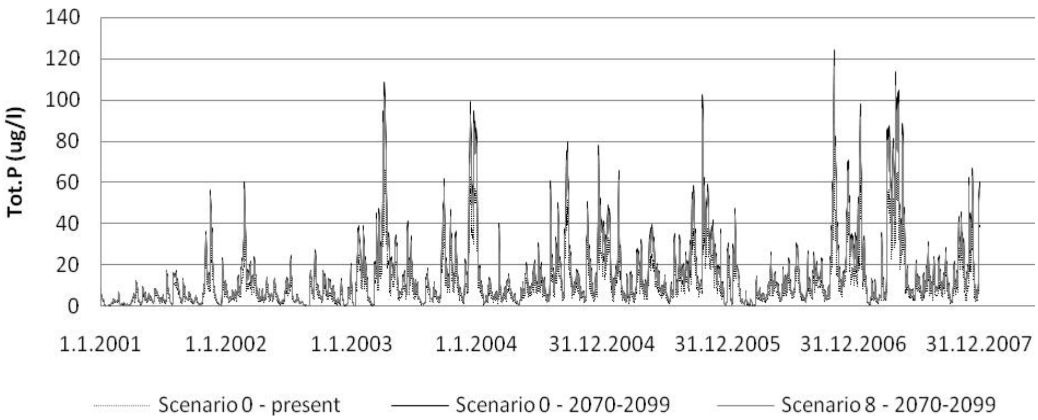


Figure 51: Total phosphorus concentrations in surface water at the deepest point of Lake Pyhäjärvi. Scenario 0 is climate change scenario AIB. Scenario 8 is a land-use scenario for removing the manure loading.

In this work LLR model was used to study the effect of nutrient load to chlorophyll a in Lake Pyhäjärvi and to estimate target nutrient loads given the lake type specific Good/Moderate boundary of TP, TN and chlorophyll a. Predictions of total phosphorus and nitrogen load are shown in Figures 52 and 53. According to lake specific model there is no need of reducing the loading, as the present load results to good water quality of the lake.

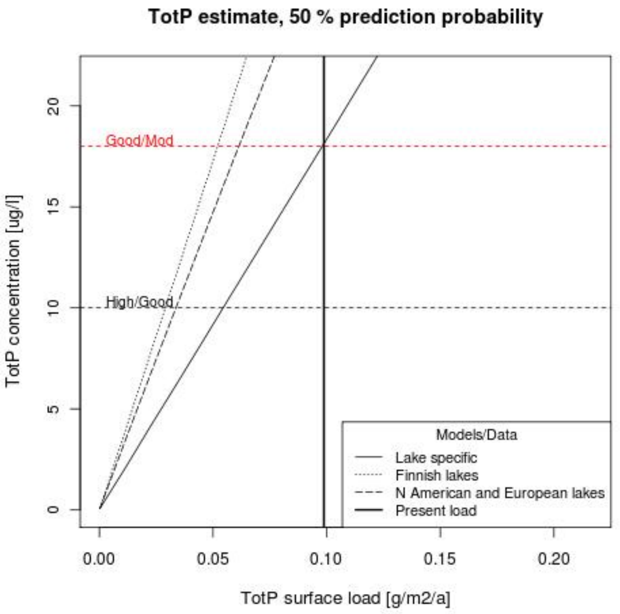


Figure 52: Estimate for phosphorus concentration (µg/l) in Lake Pyhäjärvi as a function of loading ($g/m^2/a$). Red horizontal dash line is the limit for good water quality according to WFD classification in Finland. The vertical indicates the present loading.

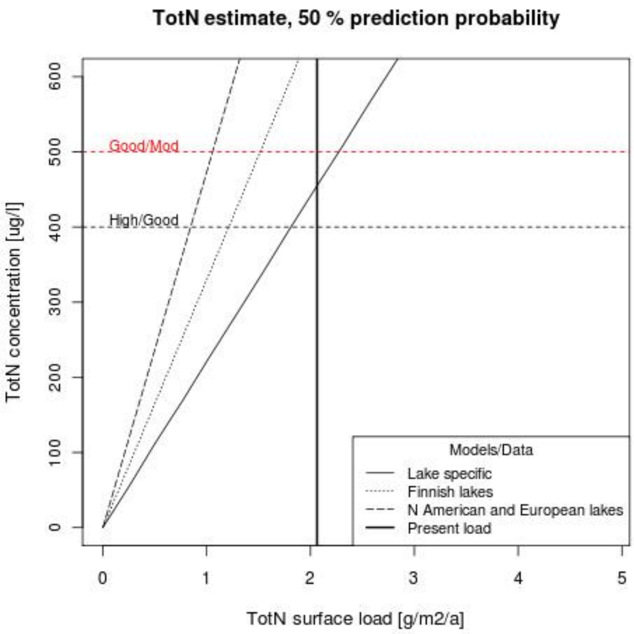


Figure 53: Estimate for nitrogen concentration (µg/l) in Lake Pyhäjärvi as a function of loading ($g/m^2/a$). Red horizontal dash line is the limit for good water quality according to WFD classification in Finland. The vertical indicates the present loading.

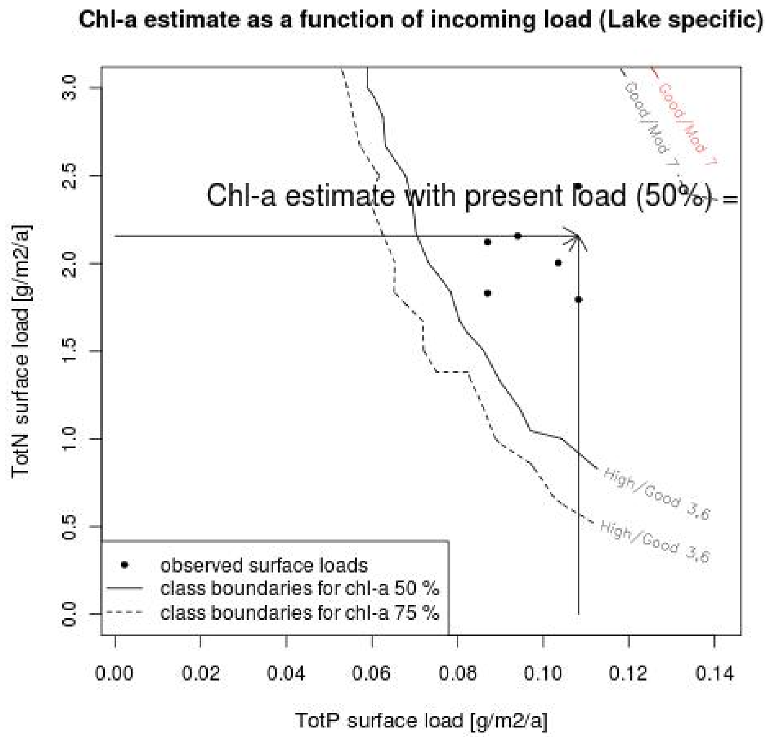


Figure 54: LLR-estimate for chlorophyll a concentration in Lake Pyhäjärvi as a function of phosphorus (X-axis) and nitrogen (Y-axis) loading (g/m2/a). The red curve shows the phosphorus - nitrogen loading combinations with which the chlorophyll a concentration will stay below good water quality limit with 50 % probability.

Table 10 shows that the reduction of phosphorus load is only marginal (1 %) according to lake specific model. Results for nitrogen load are shown in Table 11.

Table 10. TP target nutrient load calculated using LLR model.

	Load (g/m2/a)	Load (kg/d)	Sedimentation rate (m/d)	Load reduction (g/m2/a)	Load reduction (kg/d)	Load reduction (%)
Present load	0.10	42.00				
Target load: Lake specific	0.10	41.57	0.01	0.00	0.43	1.03
Target load: Finnish lakes	0.05	21.96	0.00	0.05	20.04	47.72
Target load: North American lakes	0.06	27.22	0.01	0.03	14.78	35.20

Table 11. TN target nutrient load calculated using LLR model.

	Load (g/m2/a)	Load (kg/d)	Sedimentation rate (m/d)	Load reduction (g/m2/a)	Load reduction (kg/d)	Load reduction (%)
Present load	2.06	877.00				
Target load: Lake specific	2.26	959.36	0.01	-0.19	-82.36	-9.39
Target load: Finnish lakes	1.52	644.21	0.01	0.55	232.79	26.54
Target load: North American lakes	1.06	449.12	0.00	1.01	427.88	48.79

Scenario runs for Lake Veluwe

For water quality management models can be used as a tool to determine what the best strategy is to improve water quality. However, it can also be interesting to know whether measures taken are indeed as useful as believed. In the first test to see if Delwaq-G is capable of giving insight in the effectiveness of measures, two taken measures are resimulated. As the Delwaq-G model encompasses the time period of 1976-1993, only the change in water quality of two measures and the combination of those two can be calculated. Measures taken in the time period are: 1) the increase in phosphate removal in the sewage water treatment plant (SWTP) of Harderwijk (also some nitrogen removal), which is discharging on the lake and 2) the start of flushing the lake by pumping station Lovink. As the model appears to have problems with accurate prediction of chlorofyl-a concentrations, these results will not be discussed in detail.

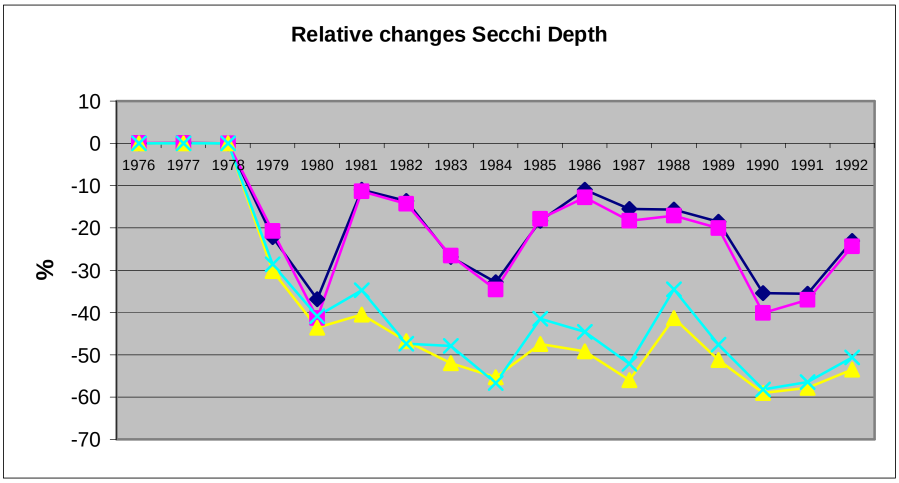
Scenario: no increase in efficiency of phosphate removal at SWTP Harderwijk

No increase in the efficiency of phosphate removal in the SWTP Harderwijk around 1978, secchi depth would have been far less than in the reference situation[1]. Figure 55 shows that at the midst of 1978, changes in secchi depth become evident. For the first two years secchi depth for all locations are the same as the reference situation. After 1978, winter and summer secchi depths are the same and up to 40% less than the reference situation. From 1981 onwards, the summer secchi depths are around 50% less and the winter secchi depth, with a rather irregular pattern, are around 25% less than the reference situation. The peak values of the reference situation of nearly 0.9 meter are never even approached.

[1] Note that this scenario shows the impact of only the flushing measure.

Legend

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



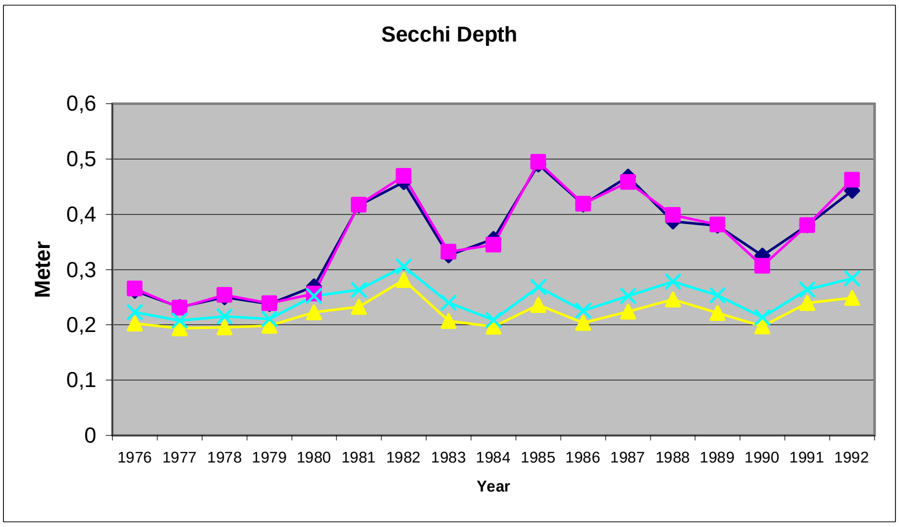


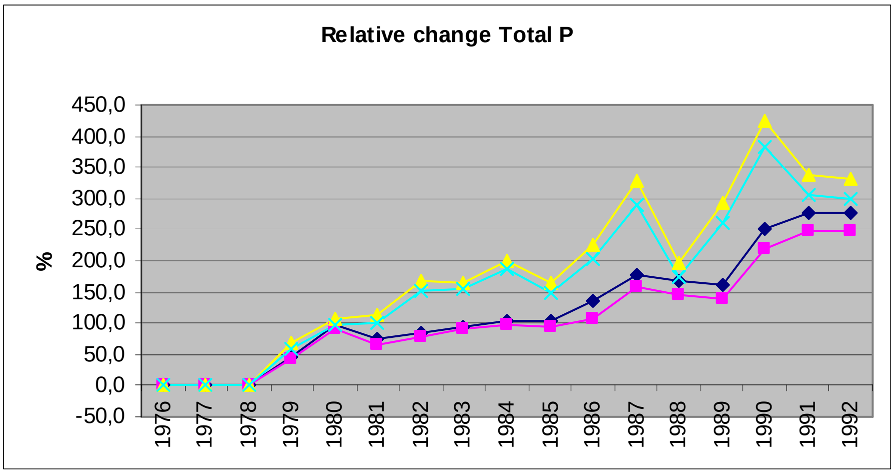
Figure 55: This figure shows the relative change in secchi depth for the scenario in which there was no increase in efficiency of phosphate removal at the SWTP of Harderwijk compared to the reference situation (above). The figure below show the absolute changes in secchi depth.

When regarding total P concentration, large differences can be seen in the yearly average values of the reference situation and this scenario. As there is no increase in phosphate removal in the largest discharging phosphate load, it is to be expected that the total P concentration oscillates around the starting value of 0.4 mg/l. However, after 1979 a small decrease in total P is visible, which increases again after the mid eighties to higher values for the summer period and somewhat lower values in the winter period towards the starting concentration of 0.4 mg/l. However, when looking to the relative changes in total P concentration, values are steadily increasing to 2.5 times the reference situation for winter period and about 3 times for summer period (Figure 56).

Legend

D_SecchiW
S_SecchiW
D_SecchiZ
S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



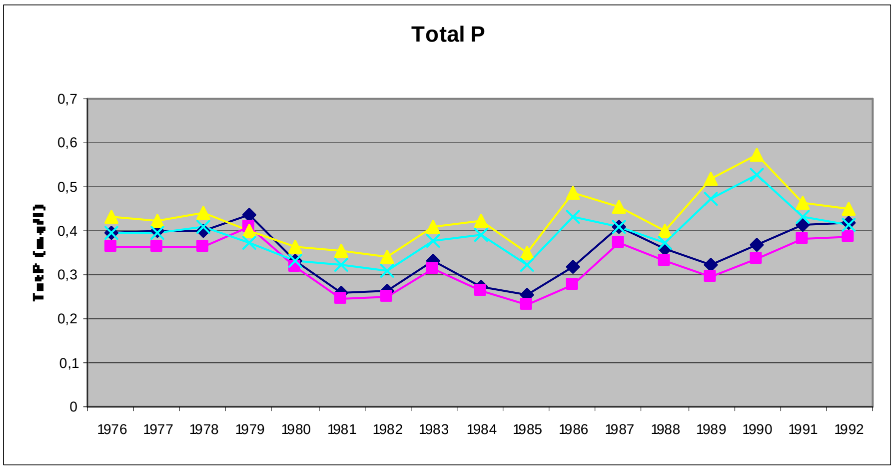


Figure 56: This figure shows the relative change in total P concentration for the scenario in which there was no increase in efficiency of phosphate removal at the SWTP Harderwijk compared to the reference situation (above). The figure below show the absolute changes in total P concentration.

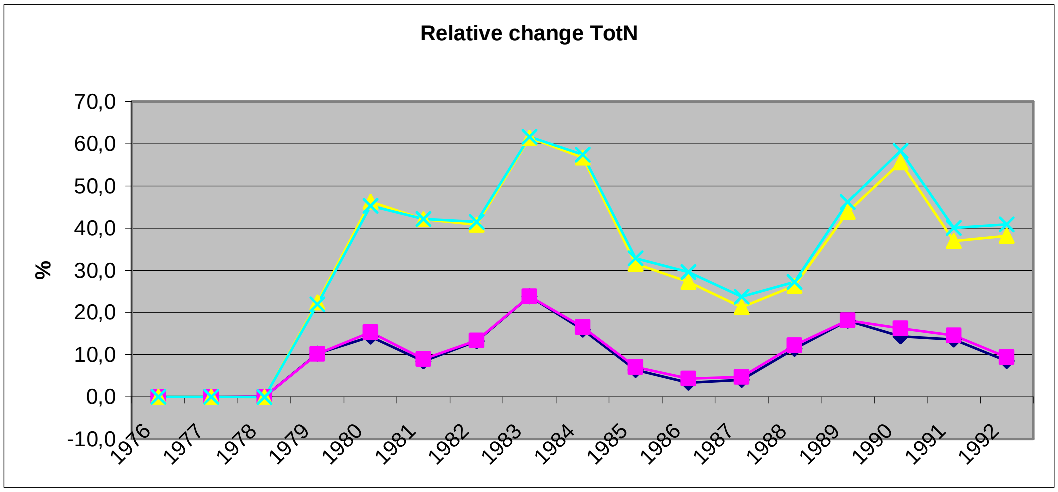
Next to total P concentration, changes in PO4 concentrations are also important for the ecological state of the system. PO_4 concentrations differ a lot between this scenario and the reference run. Especially the values in the deep part in summer increase tremendously. The other changes are also considerably, and starting to rise after 1984. The change in PO_4 concentration in the lake's sediment also increase, but values are highest in the deeper part, where infiltration takes place. Moreover, winter concentrations are higher than summer concentrations, which is similar to the PO_4 concentrations in the overlying water.

Total N concentrations in this scenario show an increasing trend in summer period and a gradually decreasing trend for winter period (Figure 57). For summer period values are around 10% of the reference situation and for winter period -15%. When looking at the absolute changes, summer and winter concentrations decrease to about 2.75 mg/ l and 3.5 mg/l, respectively.

Legend

D_SecchiW
S_SecchiW
D_SecchiZ
S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



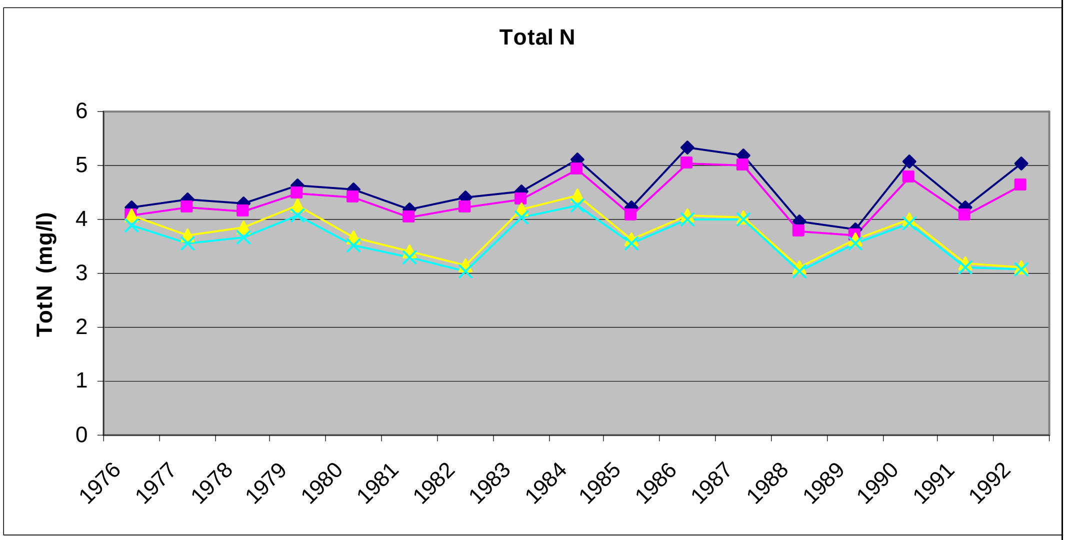


Figure 57: This figure shows the relative change in total N concentration for the scenario in which there was no increase in efficiency of phosphate removal at the SWTP of Harderwijk compared to the reference situation (above). The figure below show the absolute changes in total N concentration.

Also the fractions of nitrogen are important to consider as they support, amongst other things, algae growth. Considering nitrate and ammonium concentrations, the most remarkable difference is the almost disappearance of nitrate in the system during the summer period. As for the reference run, nitrate concentrations were alternating around 0.5 mg/l, but in this scenario, almost no nitrate is detected. Regarding the winter period, the values are somewhat lower than in the reference run, but the same trend can be seen. The same is true for ammonium concentrations in the winter, although values in this run are somewhat higher, but again, in the summer period ammonium almost disappears from the system. The disappearance of this substances can be explained with the doubled to tripled concentrations in chlorophyll-a.

Scenario: no flushing of the lake

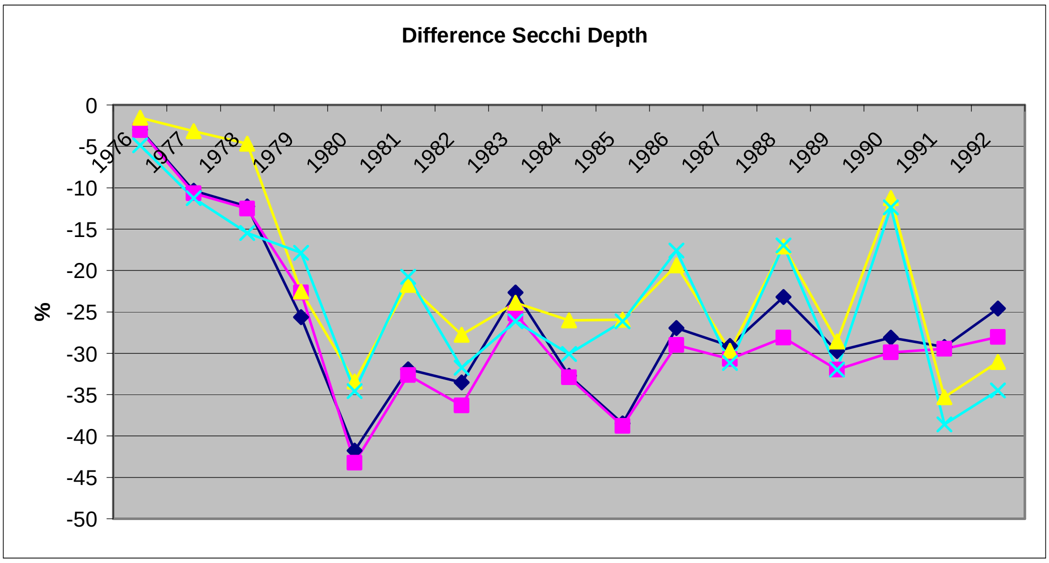
From the start of the simulation, the secchi depth for both locations decreases. It seems that the decrease in summer secchi depth is less than that of the winter secchi depth, around 75% and 70% of the secchi depth of the reference run. When looking at absolute changes, secchi depths for winter and summer period are almost the same, and stabilize around 0.35 meter and at the end of the simulation increase to 0.4 meter. Thus, the increase towards values of 0.9 meter does not show when the lake is not flushed and only increased phosphate removal of the SWTP is implemented. (Figure 58).

The relative change in total P concentration shows an increasing trend for the winter and summer period, with the former showing the largest increase (Figure 59). Especially at the end of the simulation, concentrations in the winter period seem to increase as the values in the summer period look more stable. However, when one looks at the absolute values, a decrease in total P concentration is visible: starting at values around 0.4 mg/l, the concentrations gradually drop to values between 0.15 and 0.2 mg/l. PO_4 concentrations of the water column are about 0.1 mg/l higher in winter compared to the reference run. For summer period, values are lower, except for 1982 and 1990. In the other years none or almost no PO_4 (0.005mg/l) is present in the system. When looking at the PO_4 concentration of the sediment in winter, values are decreasing from 1980 to 1990 from 0.07 mg/l to 0.015 mg/l, after which they increases towards values around 0.04 mg/l, which is not the case in the reference situation. In summer, a small decrease in values is detected and the values are comparable to the reference run.

Legend

D_SecchiW
S_SecchiW
D_SecchiZ
S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



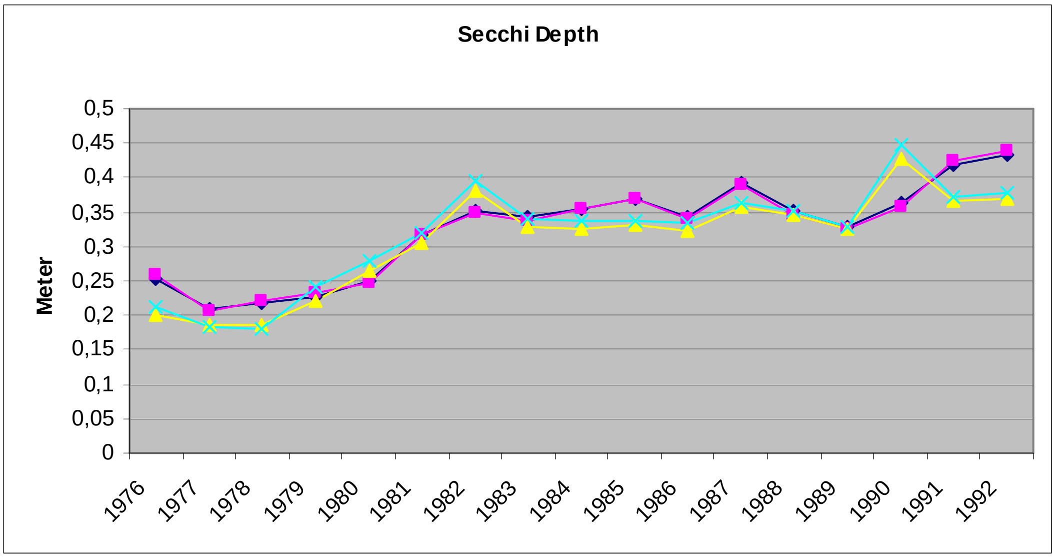
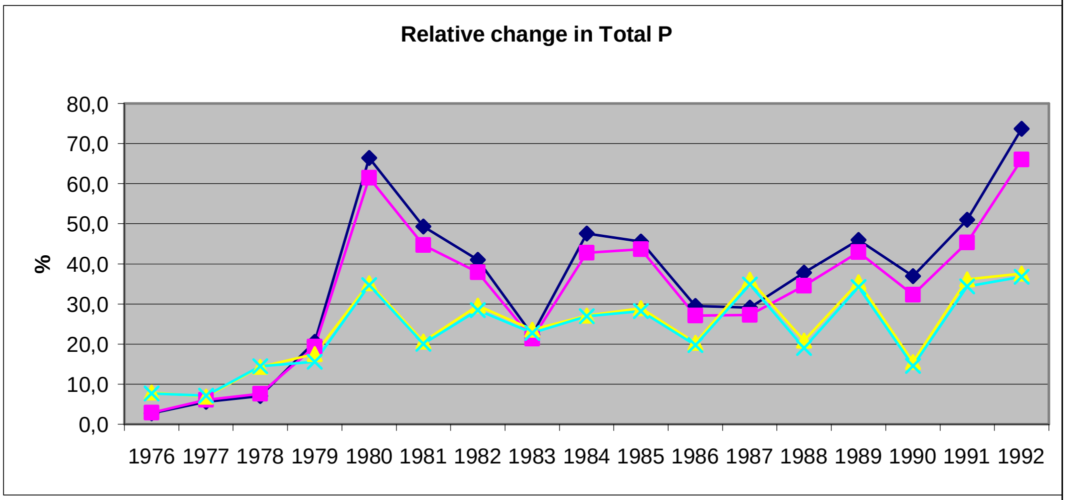


Figure 58: This figure shows the relative change of the secchi depth for the scenario in which there is no flushing of the lake compared to the reference situation (above). The figure below show the absolute changes in secchi depth.

Legend

- D_SecchiW
- S_SecchiW
- D_SecchiZ
- S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



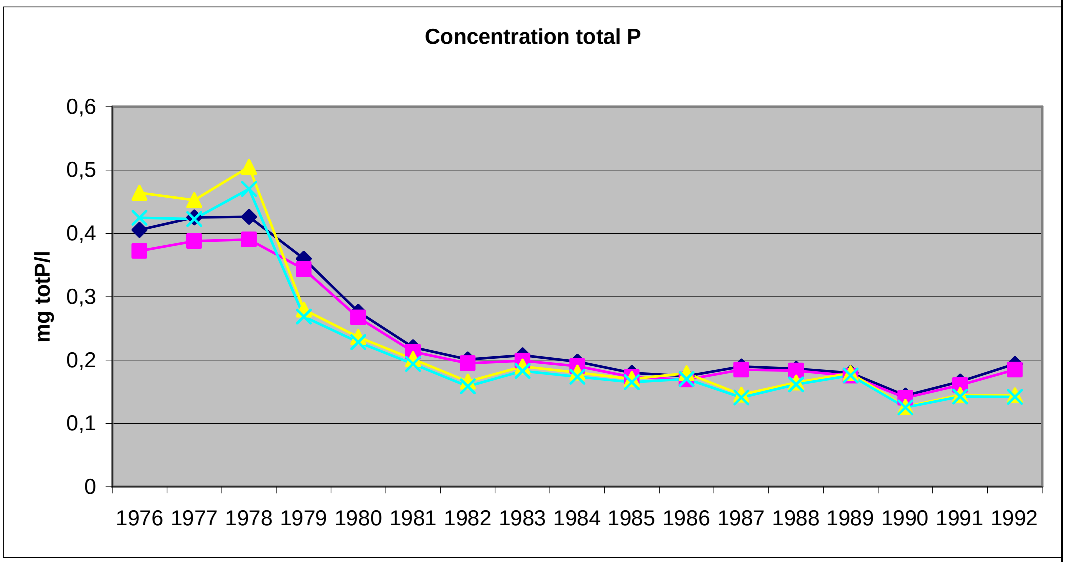


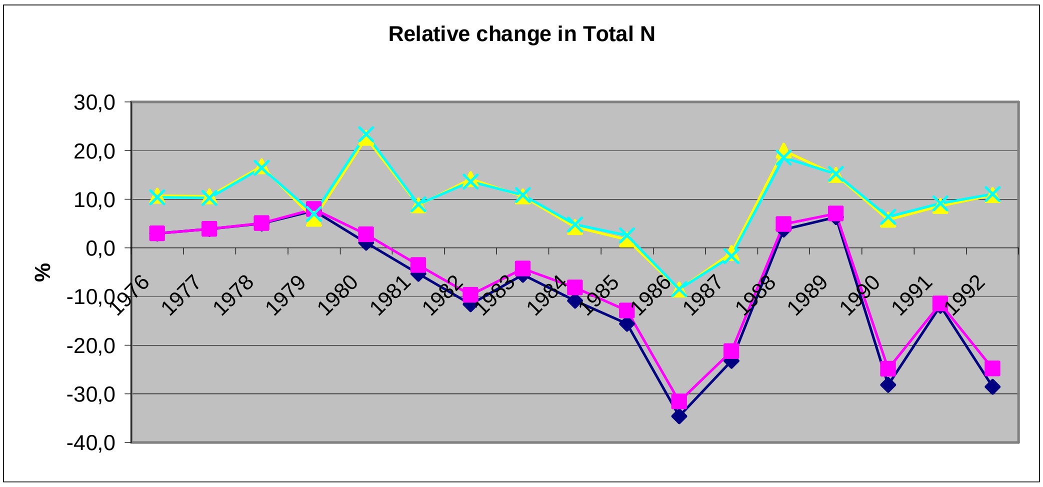
Figure 59: Figure XX: This figure shows the relative change of total P concentration for the scenario in which there is no flushing of the lake compared to the reference situation (above). The figure below show the absolute changes in total P concentration.

Total N concentrations show a decreasing trend for winter period and increasing trend for summer period compared to the reference situation (Figure 60). For winter period first a relative increase is visible, followed by a decrease that oscillates around -20% of the reference run. For the summer period, total N concentration show for the first couple of years a relative change around 10% higher than the reference situation. Later on, values show a more irregular pattern but tend to oscillate around 10% of the reference run. When looking at the absolute values, winter concentrations are after a first decrease, steady around 3.25 mg/l and in summer around 2.5 mg/l. However, as it seems that in this run the total N concentrations dropped for the winter period compared to the reference situation, this is only the case for average values. In the reference situation winter peaks are higher, reaching values of almost 5 mg/l, but the dips in summer are around 2 mg/l. This has consequences for the ecological status of the lake. For nitrate concentrations, for both time and location, concentrations are roughly halved and the original trend is conserved. After 1980, ammonium concentrations in the winter are higher in the deep segment than in the shallow segment. This is the other way around in the reference situation and moreover, the values are somewhat higher. In the summer period, ammonium concentrations are, as is also the case in the reference situation, depleted in the deep part. The difference between the runs can be found in the shallow part of the lake: NH_4 concentrations are depleted, except for 1982, 1990 and 1992. This is not the case in the reference situation: where values alternate between 0.05 mg/l and 0.1 mg/l, with the exception of 1986, where it became depleted.

Legend

D_SecchiW
S_SecchiW
D_SecchiZ
S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



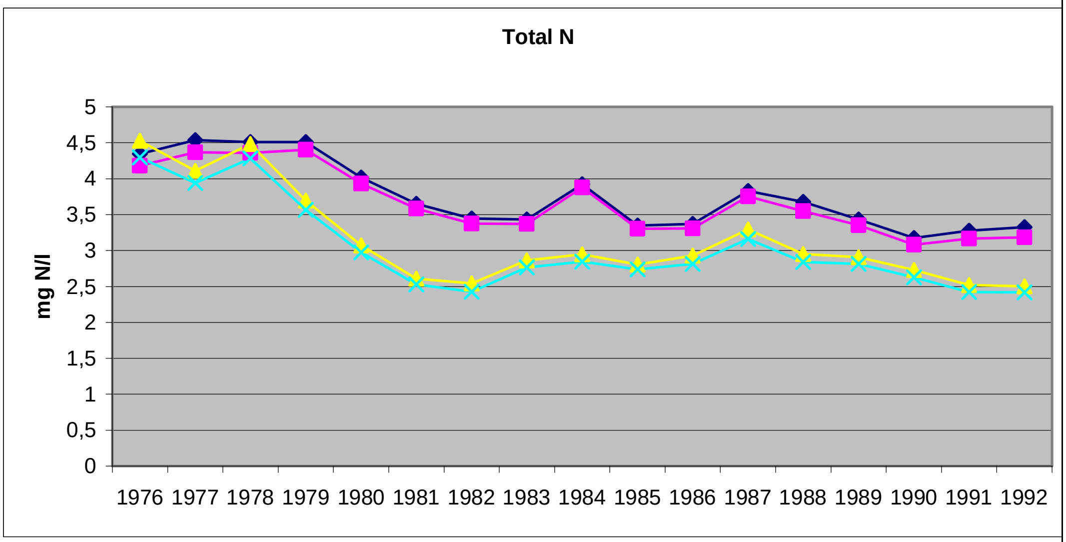


Figure 60: This figure shows the relative change of total N concentration for the scenario in which there is no flushing of the lake compared to the reference situation (above). The figure below show the absolute changes in total N concentration.

Remarkable is the depletion in PO_4 and NH_4 in alternating years, this suggests that the system is going from P limited towards N-limited state in this simulation. To get more insight in this situation, results of the limitation of algae growth can be helpful. It showed in the reference run, that algae growth was more limited by light at the last 5 years than is the case for this scenario, where both P and N start to become alternating the limiting factor.

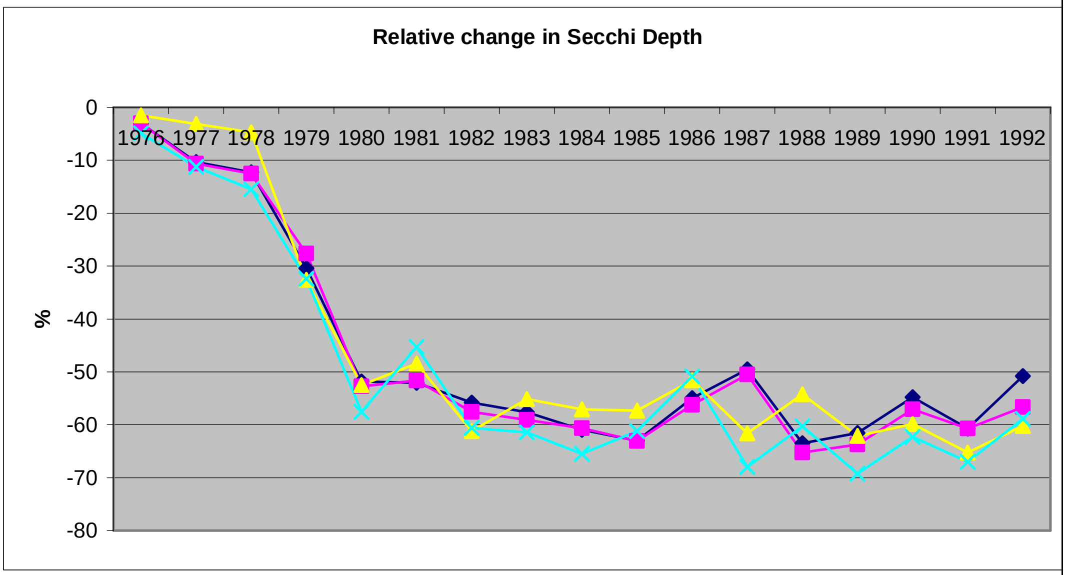
Scenario: no increase in efficiency of phosphate removal at SWTP Harderwijk and no flushing of the lake

Secchi depth for this scenario decreases for all locations and time periods to values around 30 to 40% of the reference run (Figure 61). The values of summer are in the same range as the scenario where there was no flushing of the lake. When looking at the absolute changes, secchi depth is until the mid-eighties stable around 0.2 meter, later in the simulation secchi depth tend to increase somewhat.

Legend

D_SecchiW
S_SecchiW
D_SecchiZ
S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



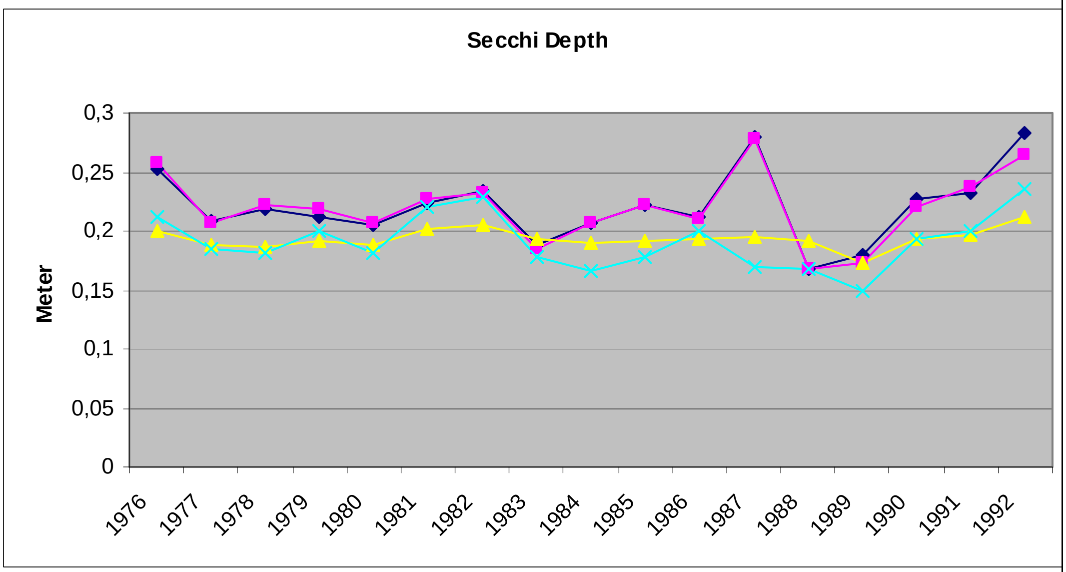


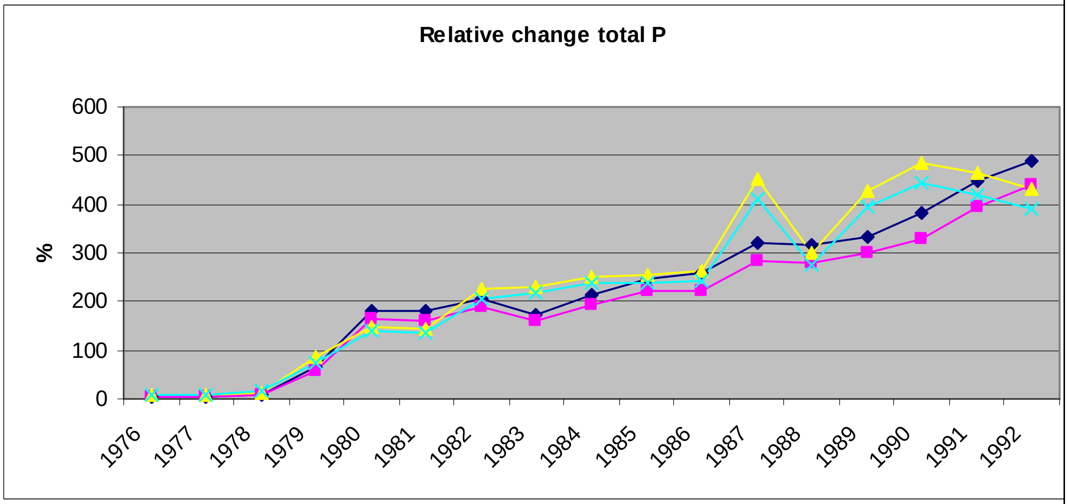
Figure 61: This figure shows the relative change in secchi depth for the scenario in which there was no increase in efficiency of phosphate removal at the SWTP of Harderwijk and no flushing of the lake compared to the reference situation (above). The figure below shows the absolute changes in secchi depth.

The total P concentrations gradually increase to values four to five times the reference situation, it seems like an increasing trend for winter period, but the summer period seems to flatten at the start of the nineties (Figure 62). The trend looks the same as for the scenario with no adaptations for the SWTP, but higher. The absolute values of total P concentration are stable around 0.45 mg/l until the mid-eighties for all locations and time periods. After that period, an increase to values of around 0.6 mg/l is visible. The trend of the not adapted SWTP scenario is present for PO_4 concentrations, but again, values are higher in this simulation. PO_4 concentrations in the sediment show an increasing trend, towards 0.2 mg/l in the deep part in the winter. For the shallow part, this increase is less, around 0.12mg/l. For the summer period, also an increasing trend can be seen, where the deep part has higher concentrations than the shallow part, 0.15 mg/l and 0.08 mg/l respectively. This could indicate internal phosphor loading.

For total N concentrations, the graph of the scenario with no flushing of the lake seems to be duplicated (Figure 63). Same trends and values are depicted. NO_3 concentrations are not present in the summer in the shallow part (except for 1976), but are present for the deep part for most of the years. At the start of the simulation, NO_3 concentrations are present, which is not the case for the reference run and the two other runs. In the winter of 1988, NO_3 becomes depleted. The trend in NO_3 concentrations in the winter are comparable with those of the scenario with no flushing of the lake, except that the values are almost halved. NH_4 concentrations are depleted for the whole simulation period during the summer in the shallow part of the lake, but for the deeper part, for a couple of years concentrations can be found around 0.5 mg/l. In the winter period, the same trend, but with higher concentrations, can be found compared to the reference run. However, in 1988 NH_4 is depleted, just for one year. It seems, that when no measures where taken, the system would have stayed phosphor limited, which is backed-up by the results of growth limitation of algae.

Legend

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



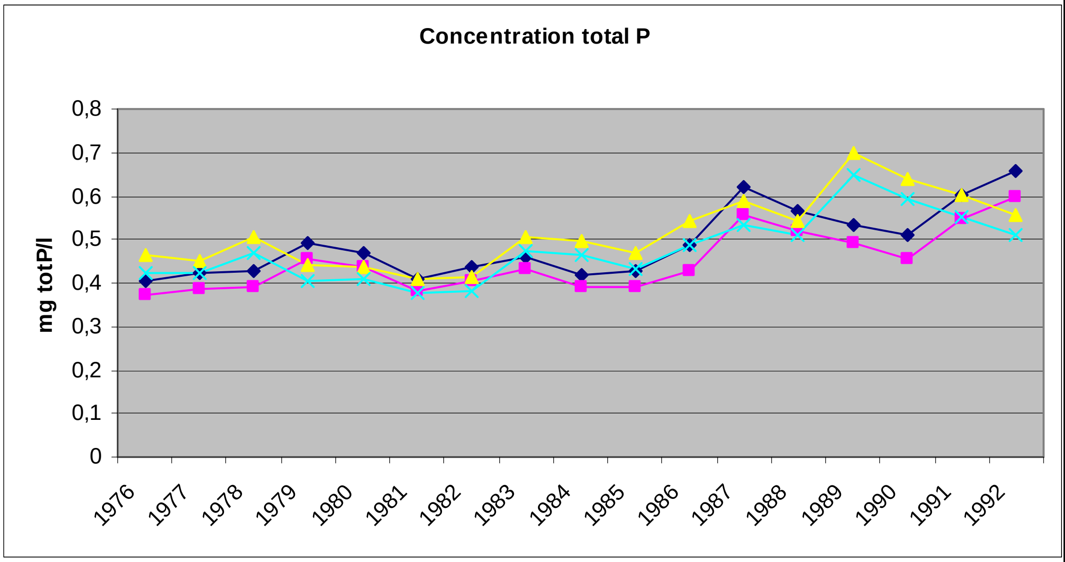
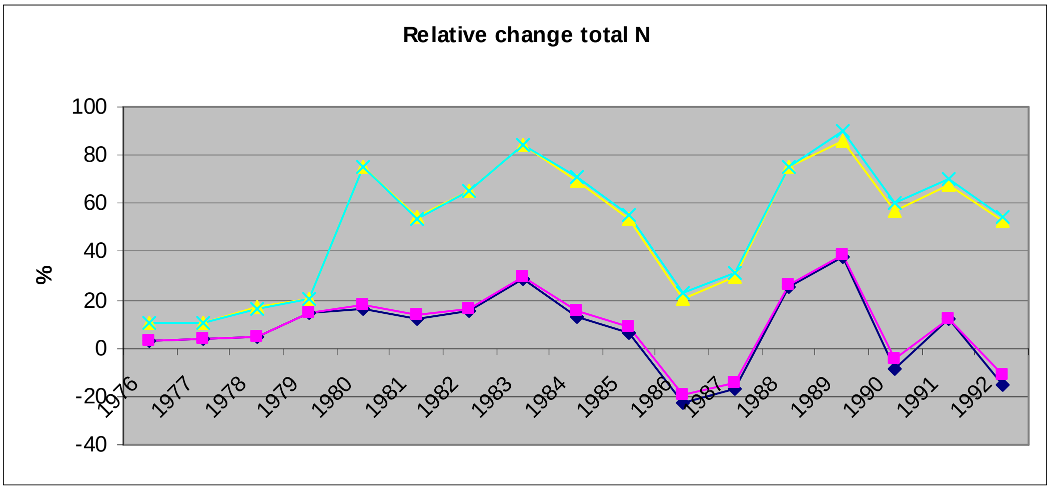


Figure 62: This figure shows the relative change in total P concentration for the scenario in which there was no increase in efficiency of phosphate removal at the SWTP of Harderwijk and no flushing of the lake compared to the reference situation (above). The figure below show the absolute changes in total P concentration.

Legend

D_SecchiW
S_SecchiW
D_SecchiZ
S_SecchiZ

The prefixes D_ and S_ mean deep and shallow part of the lake, respectively. The additional letters W and Z represent winter and summer period.



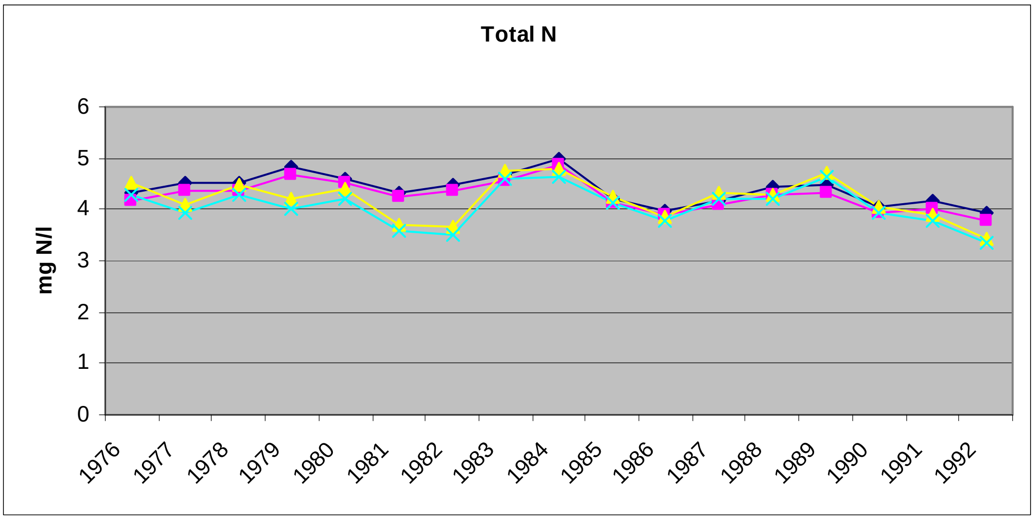


Figure 63: This figure shows the relative change in total N concentration for the scenario in which there was no increase in efficiency of phosphate removal at the SWTP of Harderwijk and no flushing of the lake compared to the reference situation (above). The figure below show the absolute changes in total N concentration.

Wrapping up results measures taken:

As shown in Figure 64 the flushing of the lake results in a lower secchi depth in summer than is the case when only phosphate removal at the SWTP takes place. It results in a difference of 0.1 meter. However, in winter period the secchi depth is higher when only flushing takes place, which is the case right after the start of the flushing regime. When none of the measures is executed secchi depth in summer remains around 0.2 meter and in winter around 0.25 meter. For recolonization of chara into the lake, this shows that it was indeed necessary to execute both measures: one of the measures would not have resulted in a secchi depth around 0.8 meter, required for Chara to be able to recolonize the lake. Total P concentrations where mostly lowered due to increased phosphate removal efficiency of the SWTP, as values dropped below 0.2 mg/l. Only flushing resulted in a more or less stable concentration: some reduction towards values around 0.25 mg/l in winter and 0.35 in summer, but at the end of the simulation values around 0.4 mg/l prevailed. Without any adaptations, Delwaq-G simulated that total P concentrations would have increased slowly to values around 0.6 mg/l at the end of the simulation. Total N concentrations remained quite stable under the flushing regime, with somewhat decreasing values in summer to concentrations of 3 mg/l and in winter an increase to values around 5 mg/l. In the improved efficiency case (also some extra nitrogen was removed), total N concentrations dropped in winter towards values of 3.25 mg/l and in summer 2.5 mg/l. No measures taken resulted in steady concentrations in winter and summer between 4 and 5 mg/l. These different changes in secchi depth, total P and total N and concurrent changes in the fractions of P and N, result in different limitations of algae growth and therefore different domination of algae type, N-, P-, light or other nutrient limited, would be the result (see figure X). This information is coming from the BLOOM module, which calculates which algae is dominating the ecosystem and by what the growth is hampered. When a system is phosphor limited, it means that the dominant algae cannot grow further because of shortage of phosphor. If in relation to phosphor more nitrogen becomes available, other algae start to grow and so on. Information presented below gives an overview of what type of limitation the lake encounters on a specific time for the lake on that moment, however, it does not show what type of algae is dominating the lake at that moment, as that goes too far this study. In the reference situation, the lake starts as being mostly nitrogen limited towards mostly phosphor limited. If only the flushing regime was introduced, the system would have stayed in a nitrogen-limited state. Improved efficiency of the SWTP would have resulted in a phosphor limited system, where the difference with the reference situation is the light limitation that does occur in the reference situation but not in the improved SWTP scenario. No measures at all results in a nitrogen limited system.

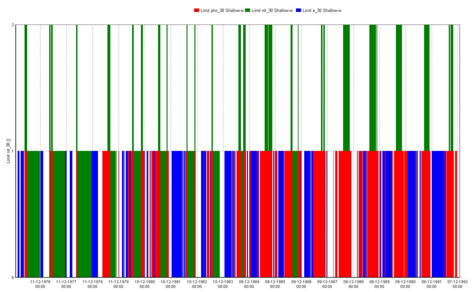
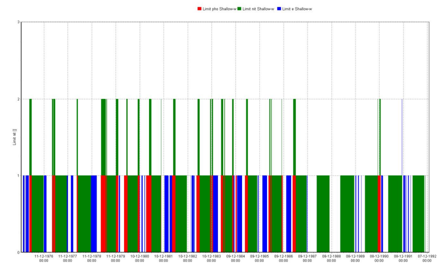
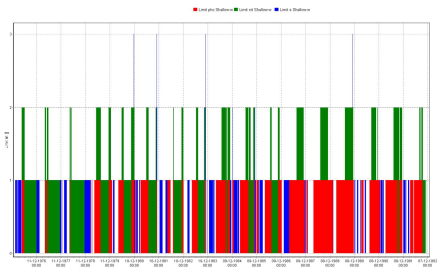
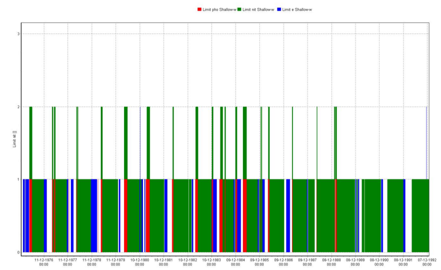
	
Original run	Only flushing
	
Only SWTP adapted	No measures executed

Figure 64: The pictures in this figure represent what the limited condition(s) is (are) for algae growth on a specific moment. On the y-axis there are2 to 3 units, meaning the total limiting factors (Energy, nitrogen and phosphor) and when there is a colored bar only from 0-1 this means that only one factor is limiting. If there is also a colored bar from to2 this means that there are two factors limiting algae growth and so on. = phosphor limited, = nitrogen limited and = energy limited.

Land use scenarios

As mentioned earlier, the model has only predefined inputs for the period 1976 to 1992; therefore future scenarios can not be run from the present state onwards. Just to get an idea of how land use changes in the river basin of Lake Veluwe affect the water quality of Lake Veluwe. Runs are made for the whole period, as initiation of the model is needed, but only results the period 1985-1992 are depicted. Note also that these runs are are independent from climate change.

When regarding secchi depth, the historic simulation renders the highest values. There is less difference between the four scenarios mutually than between the land use scenarios and the historic simulation. Within the land use scenarios a division can be seen between SR and SW scenarios and IR and IW scenarios. As in both S scenarios the least urban area is present, it seems that urban area can pose a serious pressure on water quality regarding secchi depth. Moreover, the two S scenarios differ in rural and nature area, where SW has nature area comparable to the IW scenario and SR has rural area comparable to IR. Total P summer concentrations are for al the years lowest in the reference situation, at the beginning two to three fold less than the land use scenarios. At the end of the simulation the difference has become less. All land use scenarios show the same trend, but the I scenarios have higher total P concentrations. This can be explained by the fact that the increase in urban area is for both I scenarios higher than the S scenarios. The increase in total P load from the SWTP and its overflow is likely to be the cause. The total N concentration for the reference situation and the S scenarios are about the same, but again, the I scenarios show highest concentrations. (see Figure 65).

More interestedly however, are the differences in PO_4, NO_3 and NH_4 concentrations, as these concentrations say something about the kind of limitation of algae growth, like light, P or N limited (Figure 66). This is useful to know as it offers insight in how to manage an ecosystem. The PO_4 for the historic simulation are almost zero for the deep part and low, but increasing for the shallow part. Moreover, it shows that for the land use scenarios the PO_4 concentrations are higher for at least the first 3 years of the shown results. After that, PO_4 concentrations remain lower for first the I scenarios and eventually also for the S scenarios. The opposite trend is visible when looking at the N fractions: NH_4 concentrations remain zero, at least for the summer period, for the four land use scenarios until 1988 and after that the summer concentrations remain well below the reference situation. Especially the IW scenario has low NH_4 concentrations throughout the whole simulation. NO_3.concentrations remains much higher in the reference situation than in all four land use scenarios, but never reach a depleted state. Looking at these results, it shows that the land use scenarios encounter N limitation in summer ealier than the reference situation. However, as NH_4 and NO_3 remain present in the nineties as well as now and then PO_4 concentrations, it seems that the system is going to another limited nutrient or light state. These results give therefore information which parameters should be considered when one wants to change or balance the lake's water quality.

Climate scenarios

Within the climate scenarios, only predicted change in temperature is incorporated, as it showed to be difficult to alter different water sources and sinks: an additional model is needed to calculate correct water balance. Thus, for simplicity reasons and showing the capability of Delwaq-G, changes in precipitation and evaporation are left out of the scenario. Moreover, only two climate scenarios will be considered: the least and most changing climate scenario, G and W+, respectively, as this will show the competence of the model to model the effect of temperature changes on water quality.

It appears that temperature changes exert not much effect on changes in the concentration of total P and N and secchi depth, as is shown in the graph below (Figure 67). However, changes in the P and N fractions are clearly visible. For PO_4 (Figure 68), it seems that higher temperatures result in eutrophic conditions (the period before 1980) in higher PO_4 concentrations. However, when the system becomes less eutrophic, there is not much difference between the reference situation and the scenarios with increased temperature. When looking at NO_3 concentrations only the W+ scenario show higher concentrations than the reference situation, but from 1980 onwards the NO_3 concentrations in the reference situation are higher than the climate scenarios. However, the difference become smaller towards the end of the simulation and it is shown that a change in temperature results in higher NO_3 concentrations for moderate temperature increase than for higher temperature increase. NH_4 shows the same trend as NO_3 with the exception that the differences are less pronounced and that NH_4 is depleted in the deep part, in contrast to NO_3. When looking at what limits algae growth in a specific state of the lake, one can see in Figure 69 that the reference situation remains longer N limited before it turns to P limited in comparison to moderate temperature increase (scenario G) and even longer in the case with high temperature increase (W+). The climate scenarios are at the start more nitrogen limited and are more energy limited at the end of the simulation. As such, one can see that increase in temperature interferes with the type of limitation for the type of algae in Lake Veluwe and hence, this can lead to altered management strategies, depending on the state of the lake.

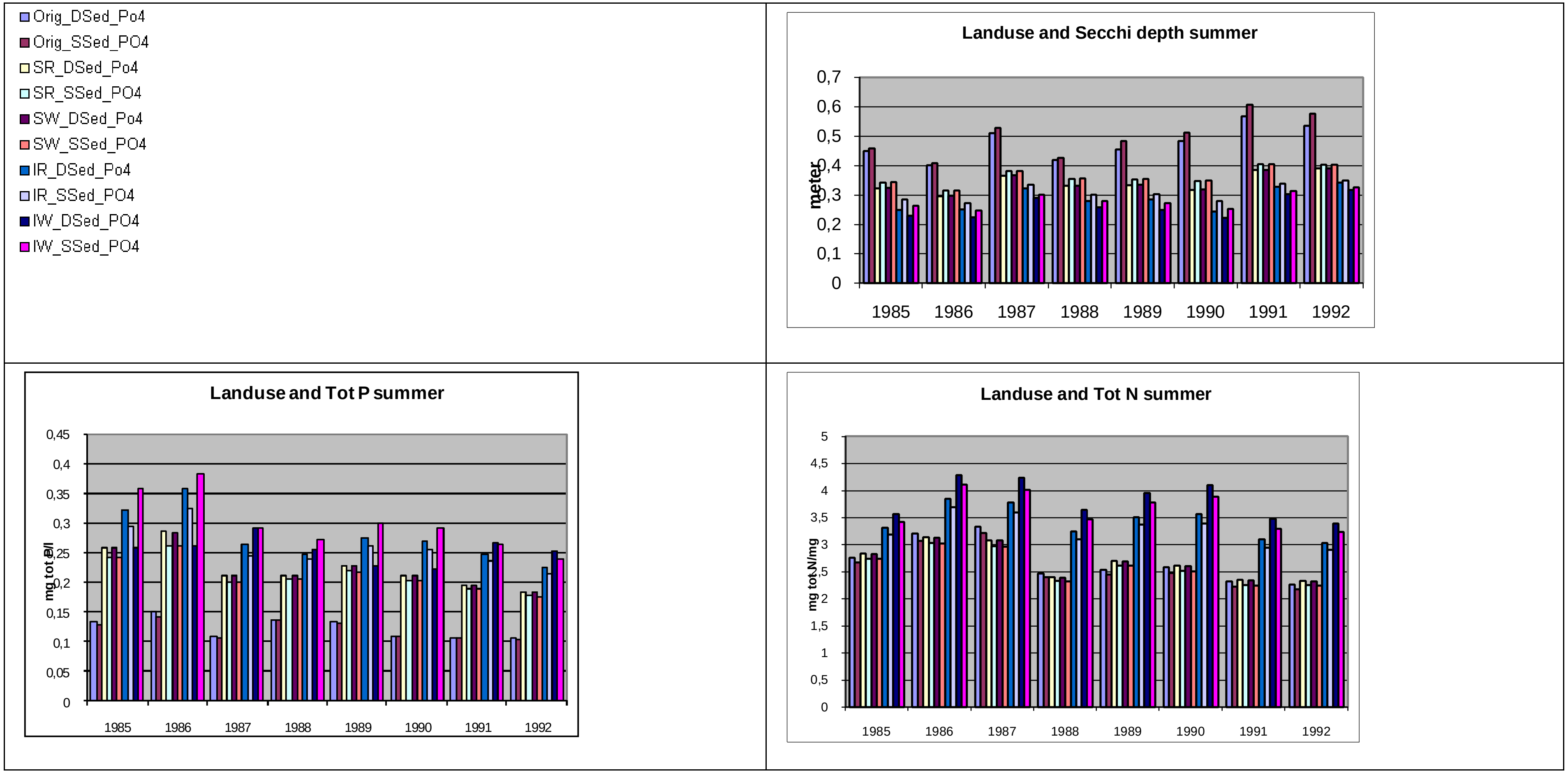


Figure 65: Changes in water quality due to land use changes for the reference situation and the 4 land use scenarios for the period 1985-1992.

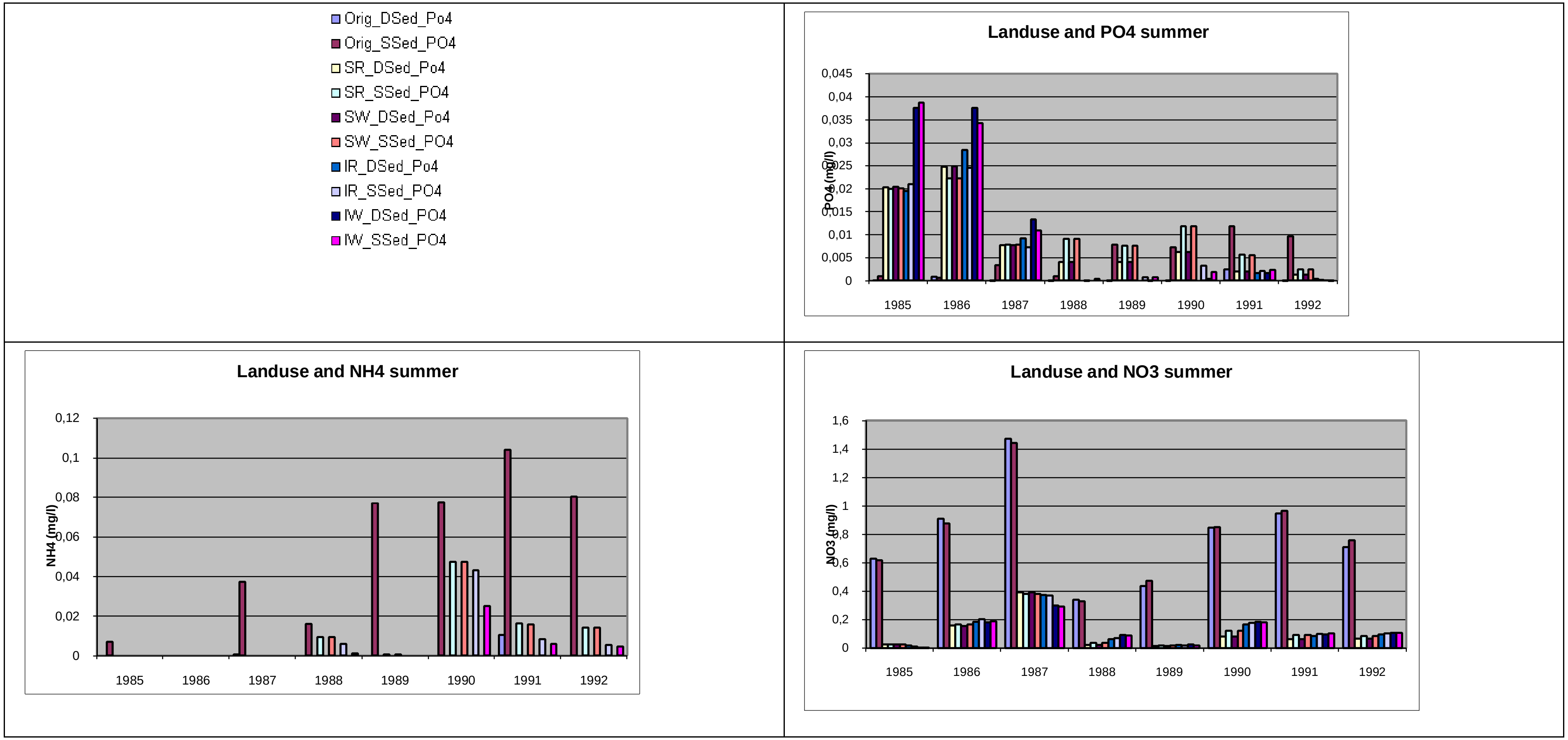


Figure 66: Changes in water quality due to land use changes for the reference situation and the four land use scenarios for the period 1985-1992.

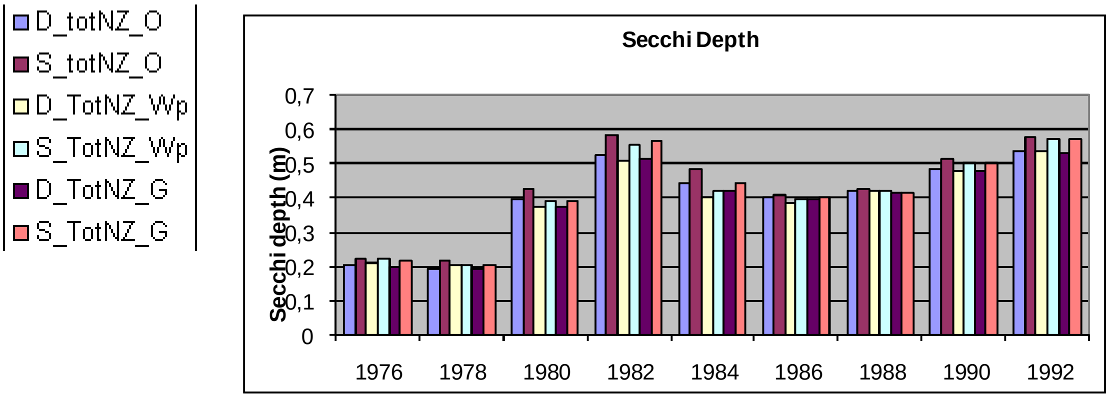


Figure 67: Changes in secchi depth for the reference situation and the G and W+scenarios.

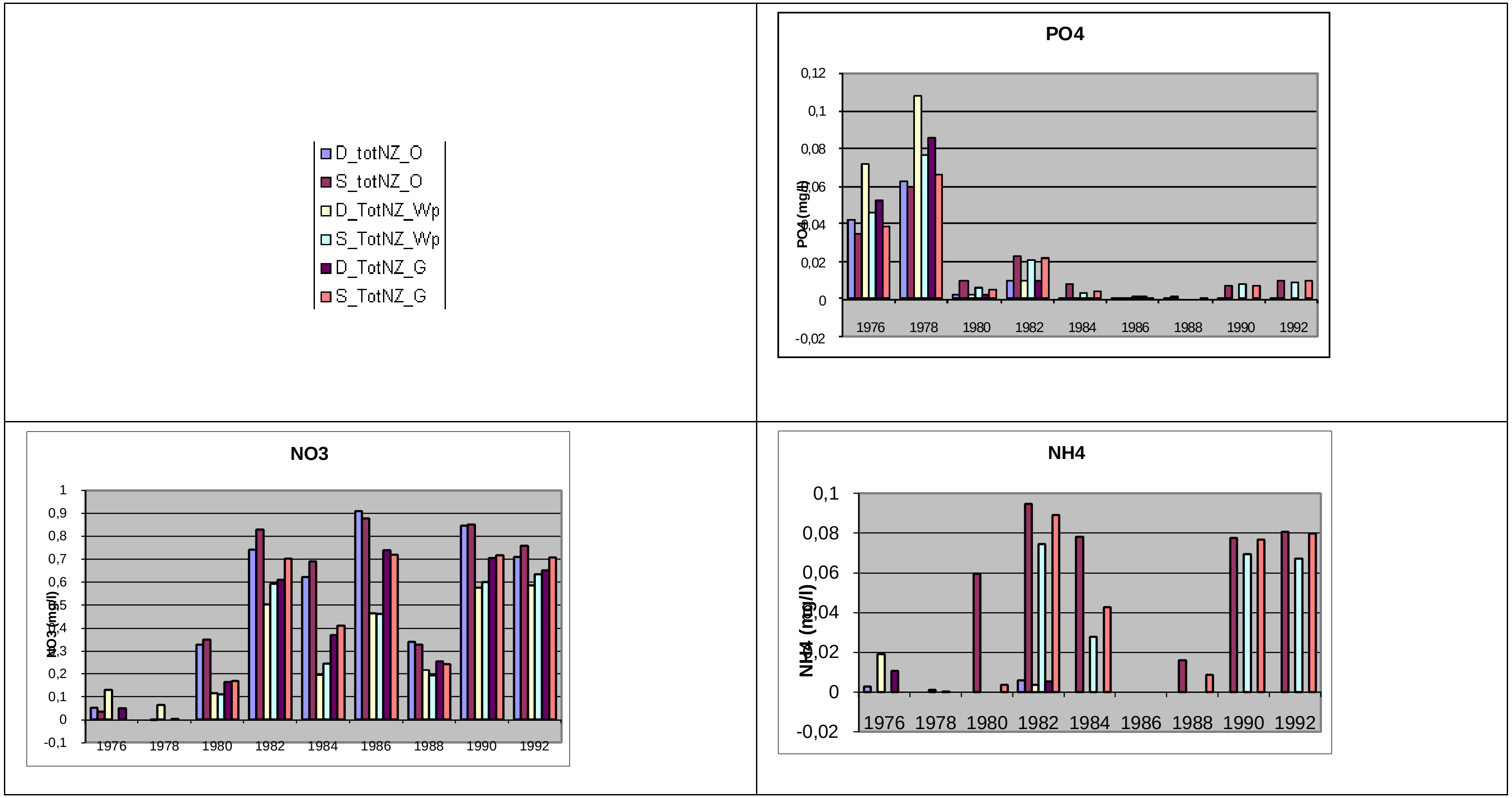


Figure 68: Overview of changes in PO_4, NO_3 and NH_4 concentration with moderate temperature (G) and high temperature (W+) increase compared to the reference situation.

= Energy limited

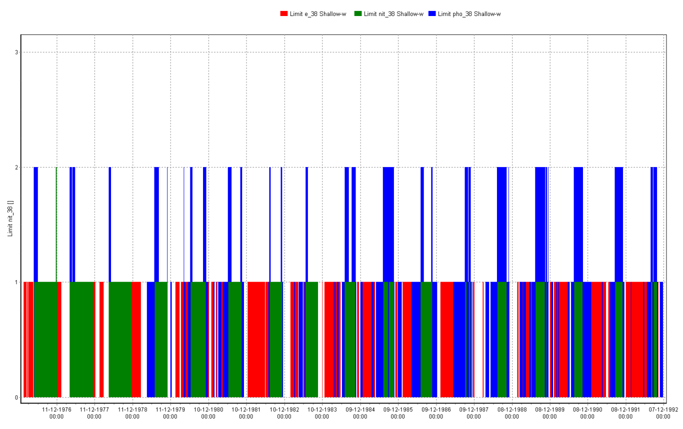
= Nitrogen limited

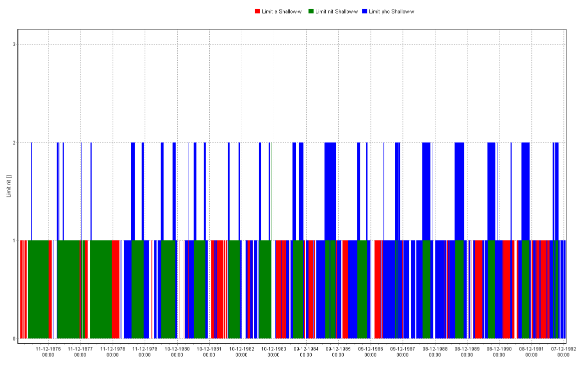
= Phosphor limited

Right upper corner the reference situation.

Left lower corner the G scenario

Right lower corner the W+ scenario





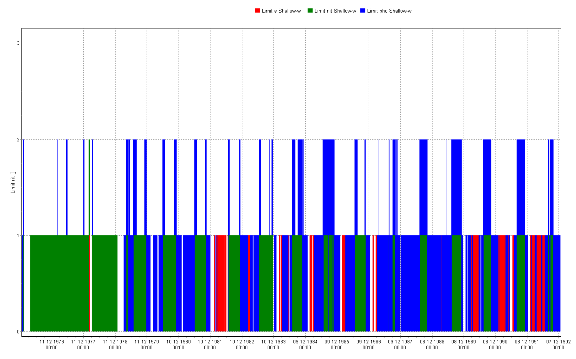


Figure 69: The pictures in this figure represent what the limited condition(s) is (are) for algae growth. On the y-axis there are 3 units, meaning the total limiting factors (Energy, nitrogen and phosphor) and when there is a colored bar only from 0-1 this means that only one factor is limited. If there is also a colored bar from 1-2 this means that there are two factors limiting algae growth and so on.

Macrophytes

It is only in the mid-nineties that macrophytes return to lake Veluwe. As is known, macrophytes, once established, exert feedback on water quality, like secchi depth and nutrient concentrations. However, once again the limiting time horizon of the Delwaq_G application and the difficulty of adapting and elaborating the water balance properly, makes it difficult to research the effect of macrophytes on the lake's water quality. Here, a first attempt is made with Delwaq-G to model the period while macrophytes where present in Lake Veluwe. For the water balance, the water balance of the years 1988 until 1992 is duplicated and attached as an extra time series to elongate the model run. Concentrations of the discharging inflows, however, are reconstructed from measured field data of the years 1999 until 2003. The goal of this modeling effort is to see whether the model is capable of modeling secchi depths of around 0.8 that are necessary to initiate macrophyte colonization of the lake. If not, the model needs to be calibrated, and if so, water plants can be build into the model as a second step. Thus, for now this working approach will do as it is merely an exploration of what the model is capable of, but in next modeling exercises the model needs to be updated with a proper water balance.

Results

The results show that even though the water balance is elongated by the last five years of the already existing water balance, the model's continuity is not consistent as can be seen in Figure 70. This inconsistency can be due to a mismatch of only one or two days when duplicating all the input required for the elongation of the run. As the files are large, this mistake could easily be made and is therefore a good case to show the models sensitivity to water balance adaptations. As the model after 1993 is alternating losing and gaining water and the concurrent loads, secchi depth results are different than expected, see fFgure 71. It is seen that the increasing trend is suddenly interrupted by lower values that, through time, start to increase again. However, due to the bad continuity results of the model, the results of secchi depth cannot be interpreted correctly. Recommendations for elongating the time series in the model are to fine tune the water balance, preferably with measured data. Moreover, the second step that is needed is to build in macrophytes into the model, with the proper feedbacks on the water quality. This makes is also possible to forecast the needed area colonized by macrophytes to sustain the lakes clear state and additionally, it can be used to hindcast trends observed in macrophytes coverage.

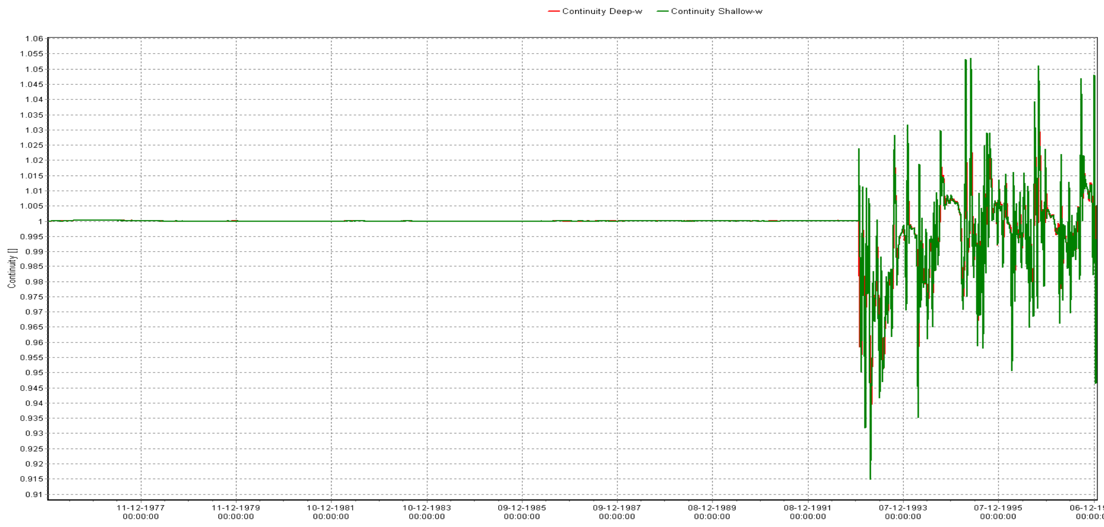


Figure 70: The continuity of the model for the period 1976 until 1997, in which the last five years are artificial constructed. It shows that more finetuning is needed, as continuity is fluctuating between -9 and +4%.

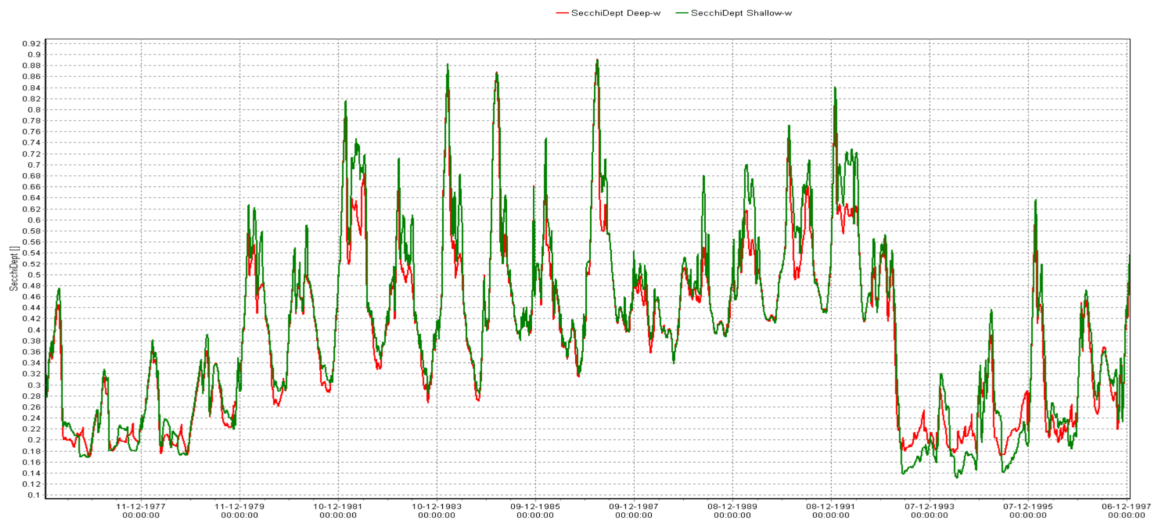


Figure 71: the secchi depth with the elongated run (1976 until 1997) shows a interruption of the trend of secchi depth for the last five years. However, problems with continuity prohibit proper interpretation of these results.

Scenario runs for Norwegian lakes

To detect whether the overall simulated lake response pattern to different factors seemed reasonable, we plotted several model output results from the control scenario (Figures 2 and 3).

Figure 72a shows daily simulation results from the control scenario for different P fractions during the last simulation year in an individual lake. This lake is situated at 59 °N at 27 m.a.s.l. and has 2 km^2 surface area, maximum depth of 15 m, and a secchi depth of 2 m. Figure 72b shows also the mean summer total P and chlorophyll-*a* concentration in the 0-4 m surface layer in all the 181 simulated lakes in the control scenario.

Figure 73 shows some results from the control simulations for different combinations of the lake characteristics and output variables. These results indicate a negative correlation between altitude, maximum depth and maximum monthly surface temperature. They also indicate a positive correlation between the summer epilimnion depth and lake surface area, as well as a negative correlation between the summer light climate in epilimnion and the PAR light attenuation coefficient. This type of response could be expected and thus the model outcome from the control simulations seem to be sensible.

We selected chlorophyll-*a* as our main metric of water quality, and Figure 74 shows how the mean and maximum yearly chlorophyll-*a* concentration in the 0-4 m surface layer changes in the eight different scenarios of temperature and/or P loading change, as compared to the control scenario.

The general result from the response of the 181 simulated lakes is that on average (for all the studied lakes) a +4° C increase in air temperature can outweigh the effect of -20% decrease in P loading on mean yearly chlorophyll-*a* concentration in the surface layer (Figure 4a, c). For the maximum yearly chlorophyll-*a* concentration the effect of reduction in P loading is somewhat stronger and the effect of temperature increase smaller than for the mean chlorophyll-*a* concentration (Figure 74 b, d).

The lakes responded generally quickly to the imposed changes in the scenarios, as no large differences were detected between the 3. and 10. years of simulations (Figure 74 a, b). Thus, long-term effects due to e.g. internal P loading from sediments were not included in the current model setup.

Finally, it is worth noting that the thermodynamic results shown in Figure 73 can be related to the 181 individual lakes, since all the strictly needed information for thermodynamic simulations were available from the Euregi lake data set (although not discharge, which might sometimes affect water temperature significantly). On the contrary, the water quality results shown in Figures 72 and 74 are hypothetic for each lake, as no site specific discharge or P loading time series are used in the batch simulations. As pointed out in the first section, the response of lakes to changes in external forcing (climate, discharge, nutrient loading) is individual and dependent on the lake and river basin characteristics. Therefore, it is worth bearing in mind that the results presented here apply for the given assumptions on water residence time, importance of internal loading from sediment, etc. for these lakes.

Future work on the batch simulations might continue e.g. along the following lines:

- more thorough statistical analysis of the simulated lake response data;
- new simulations with different discharge and P loading regimes (in terms of water residence time, timing and amount of P loading and the bioavailable P fraction);
- new simulations with new (forthcoming) version of MyLake including dissolved oxygen dynamics;
- new simulations with altered lake internal P loading and studies of lake response time to reduced P loading;
- new simulations with real loading time series (if available e.g. from simulation models);
- coupling of lake and river basin models, and new simulations on the combined climate change effects.

- extension of the simulated set of lakes to cover Scandinavia, Europe, etc.

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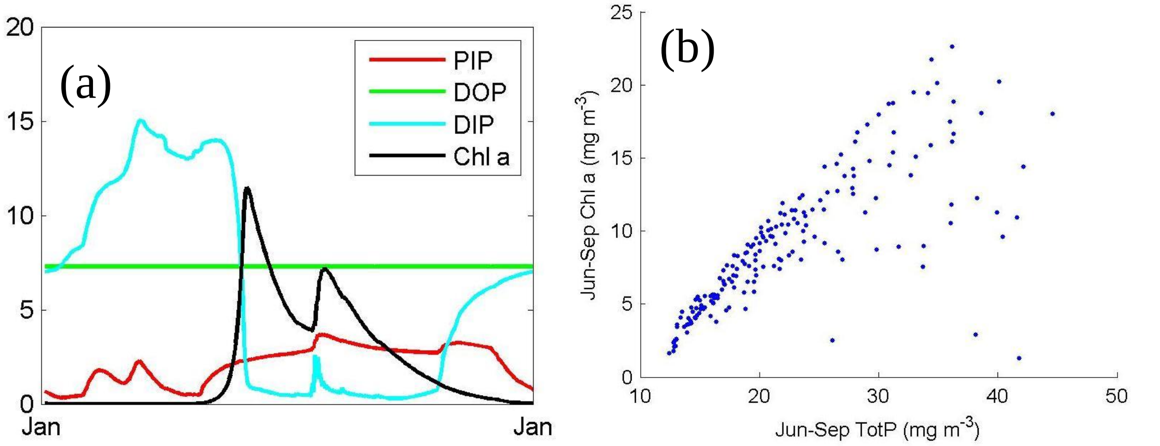

Figure 72: a) Daily simulation results for particulate inorganic P (PIP), dissolved organic P (DOP), dissolved inorganic P (DIP) and chlorophyll-a during the last simulation year in an individual lake in the control scenario (see text for lake properties). b) Simulated mean summer concentrations of total P vs. Chlorophyll-a (0-4 m layer) in the 181 Euregi lakes in the control scenario.

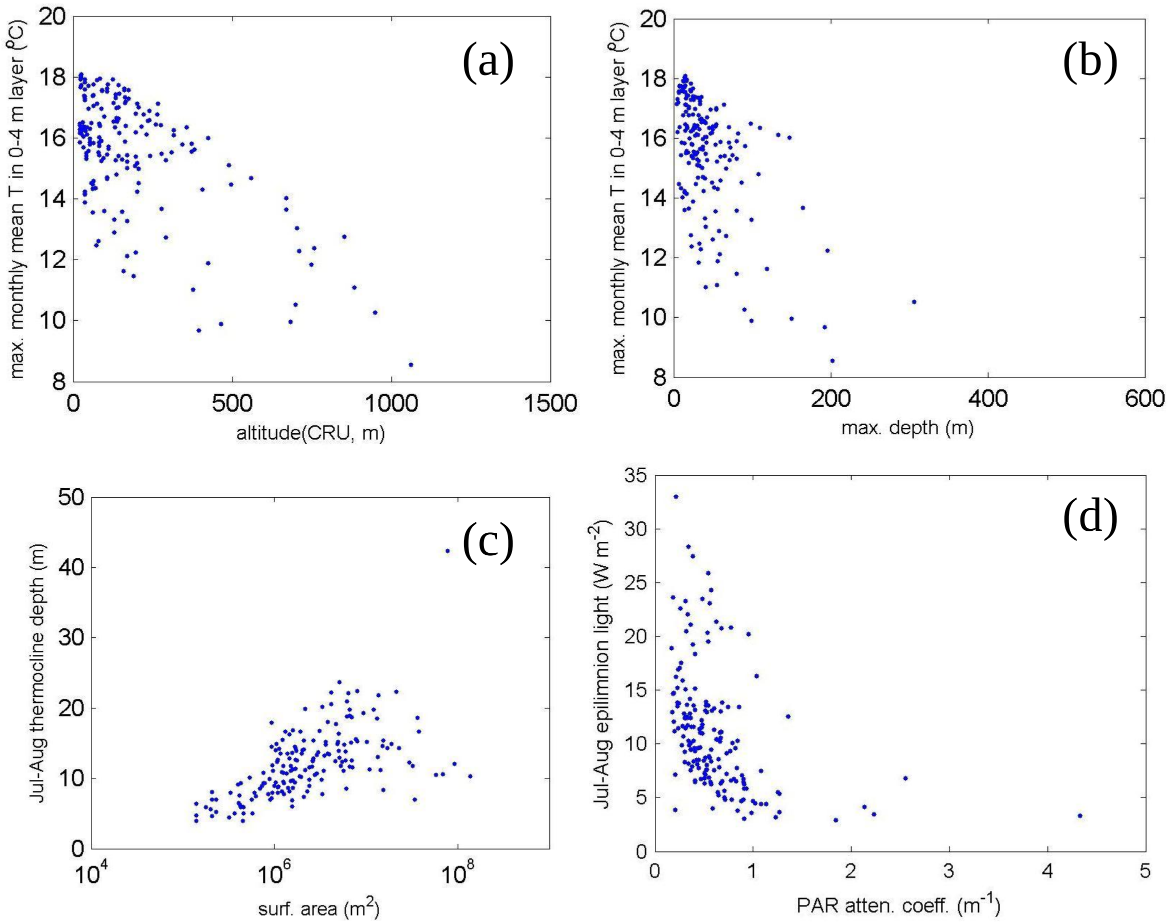

Figure 73: Simulated properties in the 181 Euregi lakes (means of the 0-4 m layer). Maximum monthly mean water temperature vs. a) altitude and b) maximum depth; c) mean July-August epilimnion (i.e. pycnocline) depth vs. lake surface area; d) mean July-August epilimnion light climate vs. PAR light attenuation coefficient.

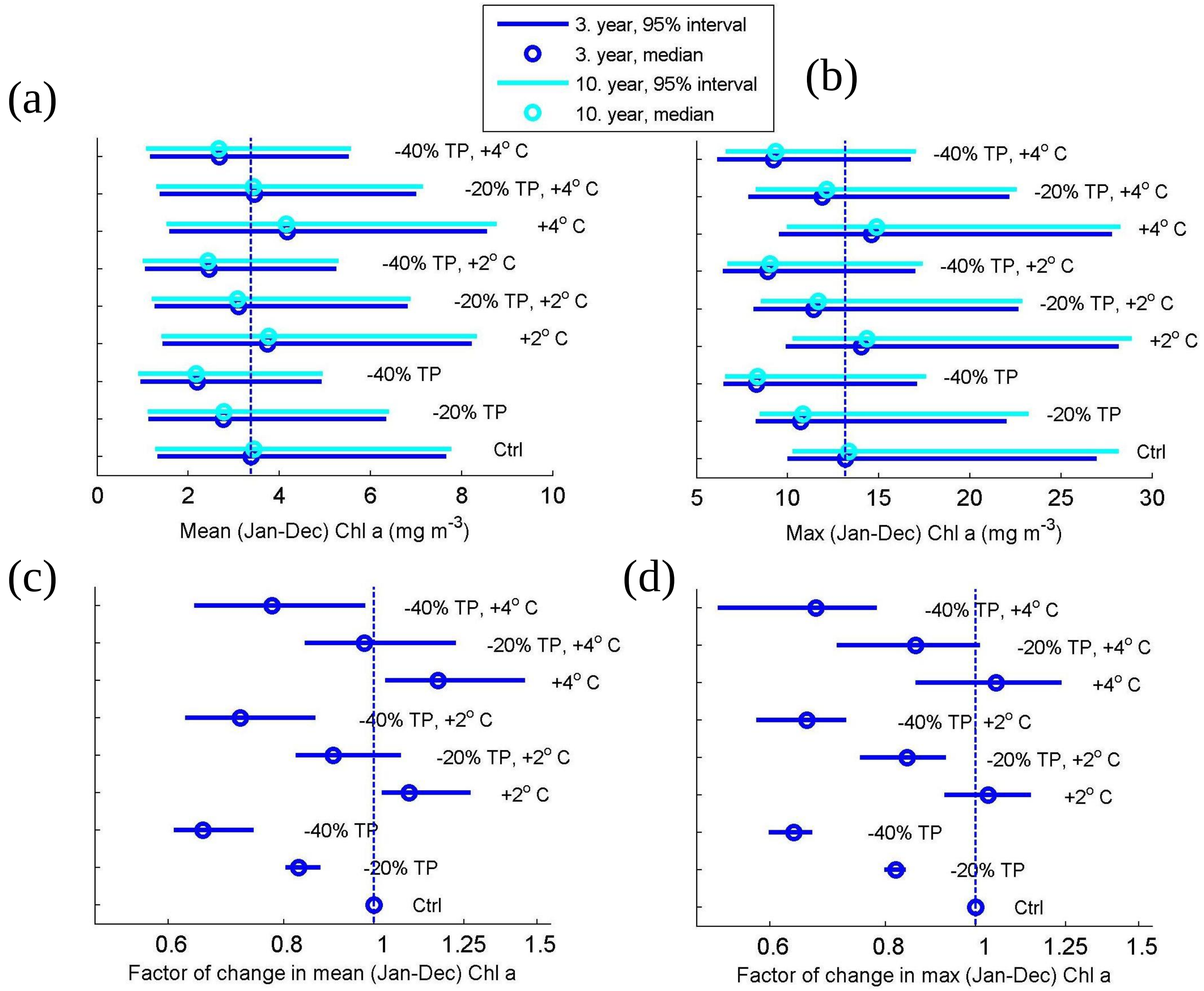

Figure 74: Change in the (a & c) mean and (b & d) maximum yearly chlorophyll a concentration (0-4 m layer) in the 181 Euregi lakes. The lines denote the control simulation and eight simulated scenarios of increased air temperature and reduction in P loading to the lake (see also figure legend). The upper panel shows absolute changes (in mg/m^3) in the distribution of the 181 lakes. The lower panel shows distribution of relative changes (factor of change) within each individual lake in each scenario compared to the control simulation. Note: logarithmic x-axis on (c & d).

Analysis and comparison of models and modelling processes

Lake Pyhäjärvi

Suitability of WSFS VEMALA, LLR, Coherens and MyLake models for river basin management were evaluated based on BMW criteria. The interaction with model users throughout the modeling process was assessed based on HarmoniQuA guidelines. Experience from earlier modeling works (INCA-N, SWAT, LakeStatea and Influence diagram) were used as a baseline for analysis.

No major technical differences other than the fractioning of nutrients were revealed in catchment models (Table 12). In contrast, applied lake models showed clearly wider spectrum of spatial and temporal scales that is from the lake average and steady state of LLR model to 1D&3D hydrodynamics of MyLake and Coherens models.

Table 12: Characteristics of models applied in Lake Pyhäjärvi: 1=WSFS VEMALA , 2= INCA-N, 3=SWAT, 4=LLR, 5=LakeState, 6=MyLake, 7= Coherens and 8=Influence diagram. Q=discharge, TP=total phosphorus, TN=total nitrogen, LU=Landuse, PL=point load, R=runoff, trans.=transport, ret=retention, Chla=chlorophyll a, Alg=algal biomass & composition, DIP=dissolved inorganic phosphorus, DOP=dissolved organic phosphorus, POP=particate organic phosphorus, SS=suspended solids, T=temperature, Load=nutrient load, Irrad=global irradiance, Wind= wind speed & direction, Cyanob=biomass of cyanobacteria, Fis man=fisheries management, Buf str=buffer stribs and Forest=forestation.

Model / **Crit/Char**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Key output variable	Q TP TN SS	Q NO_3 NH_4	Q DIP POP PIP	TP TN Chla	TP TN Alg	T DIP DOP POP Chla	T v TP TN SS Alg	Cyanob. Costs Benefits
Key input variables (pressures)	Rain, T, LU PL	Rain, T, LU PL	Rain, T, LU PL	TP_{Load} TN_{Load} Q_{in}	TP_{Load} TN_{Load} Q_{in} wind zoopl T	T Wind Irrad. Q Load	T Wind Irrad Q Load	Fish man. Wetlands Buf. Str. Forest.
Key processes	R trans. ret.	R trans. ret.	R trans. ret.	R trans. ret.	Water qual.	Therm & ice dyn. water qual.	Therm dyn. water qual.	-
Statistical infererence method.	no	no	no	Bayes MCMC	Bayes MCMC	no	no	Bayes
Num. of par.	>100	>50	>50	6	72	>25	>40	-
Num. of var.	>100	>20	>20	3	11	>15	>30	>15
Num. of inp.	>10	5-10	5-10	3	6	~10	~10	4

According to TMDL criteria (Reckhow, 2001), WSFS VEMALA was better suited for management compared to INCA and SWAT model in terms of prediction uncertainty and flexibility to updates and improvements (Table 13). Probabilistic lake models (LLR, LakeState and influence diagram) scored slightly better than complicated hydrodynamic models due to probabilistic error assessment and to lower long term costs.

Table 13. Attainment of TMDL criteria of models according to Reckhow, 2001. 1=WSFS VEMALA , 2= INCA, 3=SWAT, 4=LLR, 5=LakeState, 6=MyLake, 7= Coherens and 8=Influence diagram.

Model / Criteria	1	2	3	4	5	6	7	8
Consistency with science	1	1	1	1	1	1	1	1
Prediction uncertainty	1	0	0	1	1	0	0	1
Fits to the complexity of the situation	1	1	1	1	1	1	1	1
Consistency with data available	1	1	1	1	1	1	1	1
Crediblity to stakeholders	1	1	1	1	1	1	1	1
Acceptable long-term expense	0	0	0	1	1	1	0	1
Flexiblity for updates and improvements	1	0	0	1	1	1	1	1
Sum of scores	6	4	4	7	7	6	5	7

In contrast TMDL, SWAT and INCA models scored better according to BMW criteria due to the differences in data requirements, peer acceptance, version control, user manual, documentation. Respectively, MyLake and Coherens models scored better than others based on majority of criteria.

Interestingly, TMDL and BMW criteria showed contradictory performance of models due to the different focus and comprehension. For example, simple probabilistic lake models scored slightly better according to TDML criteria. This is because TMDL criteria emphasized more than BMW criteria practical policy relevant issues than poorly technical issues in management and decision making.

HarmoniQuA guidelines were used to assess how different aspects of modeling are communicated to and performed in accordance with managers and stakeholders (table 15). Comparison showed differences in communication in following respects: uncertainty assessment and model validation. Uncertainty assessment was not included in some models and thus, communication on it was lacking. Some models were not validated and thus, communication on validation was missing. Guidelines revealed the shortage of technical and interactive guidelines of communication between modeler and user and of formalized review process. Obviously, modeling processes should be developed in these respects.

Table 14. Attainment of benchmark criteria of models in the implementation of WFD (BMW). Scoring in Saloranta et al. 2003. 1=WSFS VEMALA , 2= INCA, 3=SWAT, 4=LLR, 5=LakeState, 6=MyLake, 7= Coherens and 8=Influence diagram.

Model / Criteria	1	2	3	4	5	6	7	8
1.1 Relevance	2	2	2	1	1	2	2	2
1.2 Scale and span	2	2	2	1	2	2	2	1
1.3 Tested	2	2	2	1	1	2	2	1
1.4 Complexity	2	2	2	1	2	2	2	1
1.5 Data requirements	1	2	2	2	2	2	2	2
1.6 Identifiability	1	1	1	2	1	1	1	2
1.7 Easy of understanding	2	2	2	2	1	2	1	2
1.8 Peer acceptance	1	2	2	0	1	2	2	0
2 Uncertainty and Sensitivity	1	0	1	2	2	0	0	2
3.1 Version control	0	1	1	1	1	1	1	1
3.2 User manual	1	2	2	1	0	2	2	0
3.3 Technical documentation	1	2	2	1	1	2	2	1
3.4 Suitability for end-user participation	1	1	1	2	1	2	1	1
3.5 Flexibility for adaptation	2	0	0	2	2	2	2	2
Sum of scores	19	21	22	19	18	24	22	18

Table 15. The attainment of HarmoniQuA guidelines in modeling process – how different aspects of modeling are communicated to and performed in accordance with managers and stakeholders. 1=WSFS VEMALA , 2= INCA, 3=SWAT, 4=LLR, 5=LakeState, 6=MyLake, 7= Coherens and 8=Influence diagram.

Model / Criteria	1	2	3	4	5	6	7	8
Internal technical guidelines	1	1	1	1	1	1	1	1
Public technical guidelines	0	0	0	0	0	0	0	0
Internal interactive guidelines	0	0	0	0	0	0	0	0
Maturity of discipline	1	1	1	1	1	1	1	1
Maturity of markets	1	1	1	1	1	1	1	1
Mutual understanding, dialogue and commitment of modelers, water manager and stakeholders	1	1	1	1	1	1	1	1
Formalised review steps	0	0	0	0	0	0	0	0
Uncertainty assessment	1	0	0	1	1	0	0	1
Transparency and reproducibility	1	1	1	1	1	1	1	1
Accuracy	1	1	1	1	1	1	1	1
Definition of the purpose of the modelling study	1	1	1	1	1	1	1	1
Collection and processing of data	1	1	1	1	1	1	1	1
Establishment of a conceptual model	0	0	0	0	0	0	0	1
Selection of code or alternatively programming and verification of code	1	1	1	1	1	1	1	1
Model set-up	1	1	1	1	1	1	1	1
Establishment of performance criteria	1	1	1	1	1	1	1	1
Model calibration	1	1	1	1	1	1	1	1
Model validation	1	1	1	1	1	1	0	0
Uncertainty assessments	1	0	0	1	1	0	0	1
Simulation with model application for a specific purpose	1	1	1	1	1	1	1	1
Reporting	1	1	1	1	1	1	1	1
Sum of scores	18	15	15	17	17	13	14	17

Lake Veluwe

When looking at different criteria sets available for modeling capability for the five models presented for Lake Veluwe, it shows that all three sets show more or less the same trend. This is a logical result because the models use more or less the same conceptual model. However, the aim for which they are made differ. This is one of the criteria which is not really highlighted in the guidelines used below (Tables 16-21), but can be essential to make a difference between models that are alike but are used with different intentions.

Table 16. Attainment of TMDL criteria of models according to Reckhow, 2001. 1= Delft3D, 2 = SOBEK2DWAQ, 3 = SOBEK1DWAQ, 4 = DBS, 5 = Delwaq-G

Model / **Criteria**	**1**	**2**	**3**	**4**	**5**
Consistency with science	1	1	1	1	1
Prediction uncertainty	0	0	0	0	0
Fits to the complexity of the situation	1	1	1	1	1
Consistency with data available	1	1	1	1	0
Crediblity to stakeholders	1	1	1	1	1
Acceptable long-term expense	1	1	1	1	0
Flexiblity for updates and improvements	1	1	1	0	0
Sum of scores	6	6	6	5	3

Table 17. Attainment of benchmark criteria of models in the implementation of WFD (BMW). Scoring in Saloranta et al. 2003. 1= Delft3D, 2 = SOBEK2DWAQ, 3 = SOBEK1DWAQ, 4 = DBS, 5 = Delwaq-G

Model / **Criteria**	**1**	**2**	**3**	**4**	**5**
1.1 Relevance	1	1	1	1	1
1.2 Scale and span	2	2	2	2	2
1.3 Tested	2	2	2	1	1
1.4 Complexity	1	1	1	1	1
1.5 Data requirements	2	2	2	1	1
1.6 Identifiably	2	2	2	2	2
1.7 Easy of understanding	1	1	1	1	1
1.8 Peer acceptance	2	2	2	1	1
2 Uncertainty and Sensitivity	1	1	1	1	1
3.1 Version control	2	2	2	1	1
3.2 User manual	2	2	2	0	0
3.3 Technical documentation	2	2	2	1	1
3.4 Suitability for end-user participation	1	1	1	1	1
3.5 Flexibility for adaptation	1	1	1	1	1
Sum of scores	22	22	22	15	15

Table 18. The attainment of HarmoniQuA guidelines in modeling process – how different aspects of modeling are communicated to and performed in accordance with managers and stakeholders. 1= Delft3D, 2 = SOBEK2DWAQ, 3 = SOBEK1DWAQ, 4 = DBS, 5 = Delwaq-G

Model / **Criteria**	**1**	**2**	**3**	**4**	**5?**
Internal technical guidelines	1	1	1	1	1
Public technical guidelines	0	0	0	0	0
Internal interactive guidelines	0	0	0	0	0
Maturity of discipline	1	1	1	1	1
Maturity of markets	1	1	1	1	1
Mutual understanding, dialogue and commitment of modelers, water manager and stakeholders	1	1	1	0	0
Formalised review steps	0	0	0	0	0
Uncertainty assessment	0	0	0	0	0
Transparency and reproducibility	1	1	1	1	1
Accuracy	1	1	1	1	1
Definition of the purpose of the modelling study	1	1	1	1	1
Collection and processing of data	1	1	1	1	1
Establishment of a conceptual model	1	1	1	1	1
Selection of code or alternatively programming and verification of code	1	1	1	1	1
Model set-up	1	1	1	1	0
Establishment of performance criteria	1	1	1	1	0
Model calibration	1	1	1	1	1
Model validation	1	1	1	1	0
Uncertainty assessments	1	1	1	0	0
Simulation with model application for a specific purpose	1	1	1	1	1
Reporting	1	1	1	1	0
Sum of scores	17	17	17	15	11

Norwegian lakes

MyLake scored well according TMDL and BMW criteria due to well documented uncertainty estimates (Tables 19 and 20).

Norwegian lakes scored well based on both criteria. Information on modeling process was not available (Table 21).

Table 19. Attainment of TMDL criteria of models according to Reckhow, 2001.

Model / Criteria	MyLake
Consistency with science	1
Prediction uncertainty	1
Fits to the complexity of the situation	1
Consistency with data available	1
Crediblity to stakeholders	1
Acceptable long-term expense	1
Flexiblity for updates and improvements	1
Sum of scores	7

Table 20. Attainment of benchmark criteria of models in the implementation of WFD (BMW). Scoring in Saloranta et al. 2003.

Model / Criteria	MyLake
1.1 Relevance	2
1.2 Scale and span	2
1.3 Tested	1
1.4 Complexity	2
1.5 Data requirements	1
1.6 Identifiability	1
1.7 Easy of understanding	1
1.8 Peer acceptance	1
2 Uncertainty and Sensitivity	2
3.1 Version control	2
3.2 User manual	2
3.3 Technical documentation	2
3.4 Suitability for end-user participation	1
3.5 Flexibility for adaptation	2
Sum of scores	23

Table 21. The attainment of HarmoniQuA guidelines in modeling process – how different aspects of modeling are communicated to and performed in accordance with managers and stakeholders.

Model / Criteria	?
Internal technical guidelines	1
Public technical guidelines	0
Internal interactive guidelines	0
Maturity of discipline	1
Maturity of markets	1
Mutual understanding, dialogue and commitment of modelers, water manager and stakeholders	?
Formalised review steps	?
Uncertainty assessment	?
Transparency and reproducibility	?
Accuracy	?
Definition of the purpose of the modeling study	?
Collection and processing of data	?
Establishment of a conceptual model	?
Selection of code or alternatively programming and verification of code	?
Model set-up	?
Establishment of performance criteria	?
Model calibration	?
Model validation	?
Uncertainty assessments	?
Simulation with model application for a specific purpose	?
Reporting	?
Sum of scores	

Discussion and summary

We had two case studies i.e. Lake Veluwe, Netherlands and Lake Pyhäjärvi, Finland. Eight mechanistic models (WSFS VEMALA, Coherens and MyLake in Lake Pyhäjärvi, Delwaq and BLOOM models - five models based on them - in Lake Veluwe and MyLake in Norwegian lake) were used to assess the impacts of climate change and catchment management measures on ecological status. The present and earlier models and modeling processes were evaluated based on BMW (Saloranta et al 2003) and TMDL (Reckhow 2001) criteria and HarmoniQuA guidelines (Refsgaard et al. 2005) to reveal their performance in river basin and lake management.

In Lake Pyhäjärvi case study, TMDL criteria highlighted policy-relevant functioning of simple probabilistic lake models (LLR, LakeState and Influence diagram) where as BMW criteria emphasized higher technical standard of sophisticated hydrodynamic models. However, it should be noted that the evaluation was impacted by the different development stage of models. HarmoniQuA guidelines for communication during the modelling process revealed the differences in communication in following respects: uncertainty assessment and model validation. Uncertainty assessment was not included in some models and thus, communication on it was lacking. Some models were not validated and thus, communication on validation was also missing. It revealed also the shortage of technical and interactive guidelines of communication between modeler and user and of formalized review process. Obviously, modeling processes should be developed in these respects.

When regarding the models that are available for Lake Veluwe, all are based on Delwaq for the water quality modeling and expect the 2D SOBEK model, make use of BLOOM. Moreover, all five models have more or less the same conceptual model at their base. The difference of the models presented boils therefore down on the user interface, the differences in model options, its maturity and the extent of use of the model. Regarding their applicability to policymakers, all can model the same scenarios, but the input, output and complexity of the models differ. As Delft3D and SOBEK 2D and 1D are internationally accepted and used often their ranking on the three criteria schemes are higher than the DBS and Delwaq-G models, which are only used for national purposes. Moreover, the DBS and Delwaq-G models are missing a user interface and are therefore not easily applicable for modeling by others than who are familiar with the model's structure and are therefore bad at the criteria of availability of available manuals, both intern as extern. Moreover, as the DBS and Delwaq-G are standalone versions of Delwaq, an additional flow model is needed to be able to construct a proper water balance when the model gets complicated. A simple water balance can be done manually. The Delwaq-G model scores especially bad regarding the HarmoniQua guidelines, as no report is made of the efforts done and so no information is present, outside the model itself, on the exact model set-up and the performance criteria.

Modelling of Norwegian lakes provided a good reference to policy oriented climate scenario analysis.

In the end, the used models, the criteria and the guidelines for model usage in river basin management were evaluated and the lessons learnt were summarized:

1. A wide variety of models and modelling approaches were tested in the case study areas.
2. Climate and management scenarios differed among the case studies.
3. The evaluation was impacted by the different development stage of models.
4. Selected model selection criteria and modelling guidelines revealed different aspects in model selection and use: a. TMDL criteria (Reckhow 2001) - policy-relevant functioning of simple probabilistic lake models b. BMW criteria (Saloranta et al 2003) - technical standard of sophisticated hydrodynamic models c. HarmoniQuA guidelines (Refsgaard et al. 2005) - communication between modeller and user during the modelling process.
5. The applicability of models to policymaking depends on a. the user interface b. the model options c. the maturity and the extent of use of the model d. the input, output and complexity of the models e. the availability of manuals, both internal as external.

6. Uncertainty assessment and model validation were not included in some models and thus, the communication of them was lacking.
7. Simple probabilistic lake models proved to be specifically relevant in decision making due to the uncertainty estimates of predictions.
8. There is the shortage of technical and interactive guidelines of communication between modeller and user and of formalized review process. Obviously, modelling process should be developed in these respects.

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