Collaborative Project (large-scale integrating project)

Grant Agreement 226273

Theme 6: Environment (including Climate Change)

Duration: March 1st, 2009 – February 29th, 2012

WISER

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

DELIVERABLE

Deliverable D5.2-3: Guidelines on the use of different modelling approaches for designing Program of Measures

Lead contractor: **Deltares**

Contributors: **Harm Duel, Olli Malve, Timo Huttula, Valesca Harezlak and Ellis Penning**

Due date of deliverable: **Month 35**

Actual submission date: **Month 36**

Project co-funded by the European Commission within the Seventh Framework Programme (2007-2013) Dissemination Level		
PU	Public	X
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	

Content

Executive summary

A large variety of lake ecosystem models is available, ranging from simple lineair regression models to complex dynamic deterministic models, from expert judgement based models to (large sets of) data driven models. Based on the lessons learned from the WISER case studies where different type of models were applied, and after reviewing existing guidelines, a new set of guidelines is developed to support the selection of models and to apply the models selected conform the protocol of good modelling practice.

For a successful application of lake ecosystem models in the context of supporting lake managers in designing restoration and mitigation measures to improve the ecological status of lakes, the following 10 steps are commended:

Step 1: Close cooperation with end-users in the formulation of management questions that need to be addressed in the model application.

Step 2: Designing conceptual models together with end-users to have a common understanding and to create a clear picture about the issues that should be addressed.

Step 3: Identify together with the end-users the model requirements within the constraints of time, money and data-availability.

Step 4: Select together with the end-users the models that are needed to analyse and quantify the effectiveness of restoration measures.

Step 5: Set up your lake ecosystem modelling framework by translating the conceptual model into a set specific model, including the processing of data in order to prepare input files necessary for executing the model

Step 6: Calibrate and validate the lake ecosystem model according to the procedures of good modelling practice.

Step 7: Conduct a sensitivity analysis to identify the appropriateness of the model structure and parameter estimation

Step 8: Quantify the (magnitude of) uncertainties in the model structure, model parameters and model predictions.

Step 9: Discuss the results of the validation and sensitivity and uncertainty analysis with the end-users.

Step 10: Conduct the simulation and evaluation activities, including non scientific reporting of the impacts of management measures to the end-users.

1. Introduction

1.1. The WISER project

The majority of European freshwater ecosystems are degraded due to water pollution and alternations in hydrology and morphology. Consequently, many aquatic species have disappeared from entire eco-regions. To provide knowledge and tools to design cost-effective restoration measures, the WISER project started within the 7th Framework Program of the European Commission. The main objective of WISER is to support the implementation of the European Water Framework Directive (WFD) by developing tools for the integrated assessment of the ecological status of rivers, lakes and coastal waters. Besides improving existing and developing new assessment methods for all WFD biological quality elements, analysis and assessment of biological recovery processes in selected cases studies, is one of specific objectives of the WISER project. Therefore, a variety of modelling techniques is applied to address pressure-response relationships and to evaluate the effectiveness of restoration measures and to analyse how changes in climate may affect the ecological status and recovery processes.

There is a long tradition in modelling lake ecosystems to analyse the biological responses to increased loading of nutrients. In recent years, different modelling techniques are applied to study the recovery processes and climate change impacts. Within the WISER project, the applicability of existing modelling approaches to analyse the impact of restoration measures and lake water management on lake ecosystem processes is reviewed. A limited set of existing modelling approaches were applied in two case studies: Lake Pyhäjärvi (Finland) and Lake Veluwe (the Netherlands): the results are reported in Deliverable 5.2-1. In this report, we review the applicability of different modelling approaches to analyse climate and land-use scenarios and restoration strategies based on literature and the experiences from case studies. In addition, this report provides guidance on lake ecosystem modelling in order to design and implement cost-effective restoration programmes and to develop climate robust lake management strategies.

1.2. European Water Framework Directive and the role of models

The general goals of the Water Framework Directive (WFD), introduced by the European Union on 22 December 2000, are to achieve a "good status" in all water bodies by and to protect the aquatic ecology, unique valuable habitats, drinking water resources and bathing water with reasonable costs. Planning and implementation of water management is organized at a river basin scale in order to ensure that local factors and the need for water protection measures are taken into account efficiently. The adaptation to changing environment and climate were underlined. The targeting of the pollutant load reduction and the designing and implementation of solutions to restore or rehabilitate the aquatic environment are the challenging key ingredients of the river basin planning, and all our existing science, technology, mathematics and practical experience in this field is needed to achieve compliance with the water quality standards with regard to chemical substances and ecological status. Hydrological and biogeochemical cycling, in particular, and the resource conditions for the assembly of the plankton and macrophyte communities must be considered comprehensively. Until recently, the theoretical foundation for ecology was empirical rather than theoretical, ranging from deterministic to stochastic approaches, and hence there is no equivalent comprehensive biological foundation analogous to Newtonian mechanics or hydrodynamics that can be employed for the control of eutrophication and pollution in lakes and rivers. In addition, the determination, calibration and validation of predictive models are hampered by the overwhelming number of factors affecting the aquatic ecosystem processes and species diversity and by the limited experimental and observational resources available. Hence, the translation of scientific theories, specific observations on river basin and mathematical approaches into forms which are useful for river basin planning is difficult. Before starting the planning of lake or a river basin management it is crucial to identify expected ecological and social impacts of intended measures. Here, data and models are coming in. However, without clear guidance on the use of scenarios from models it is hard for lake and river basin managers and stake holders to make realistic choices between management options.

1.3. The concept of models

This report uses the word model as a term for all kind of representations of essential aspects with respect to lake ecosystems and its catchments. This frequently refers to a computer program with corresponding input. However, the word 'model' may also refer to some notes on paper, a mathematical model, a diagram or a figure, representing the lake ecosystem.

Conceptual models are qualitative representations of relationships that are addressing the key aspects of – in this case – the lake ecosystem (Figure 1). A mathematical model is the mathematical translation of the conceptual model. Examples of mathematical models are: algebraic equations, differential equations, ordinary differential equations, partial differential equations, neural networks, statistical models, knowledge rules on dose-response relationships and combinations of these.

A model is dynamic if it describes changes over time; it is stationary or static if it does not (steady state models, examples are mass balance models). A mathematical model has one or more independent variables and one or more dependent variables. In a dynamic model, time is the minimum independent variable present. In a spatial model, at least one spatial dimension is another independent variable. A dynamic 3D model has four independent variables: time and three spatial dimensions.

Dynamic models on the basis of a non-stochastic representation are referred to as deterministic models: the knowledge of the modelled system is fully determined in the model and repetitive use of the model produces the same results. Just like the system of which is it a representation, a model has a model structure (state variables and relations which are defined by auxiliary variables) and a model behaviour (how does the model behave along the axis or axes of the independent variables: what are the changes in the result of a model over time and/or along the spatial axis (axes)).

Empirical models are based on large data sets and are representations of the relationships are based on statistics (e.g. linear regression models, decision trees, neural networks) – see Figure 2.

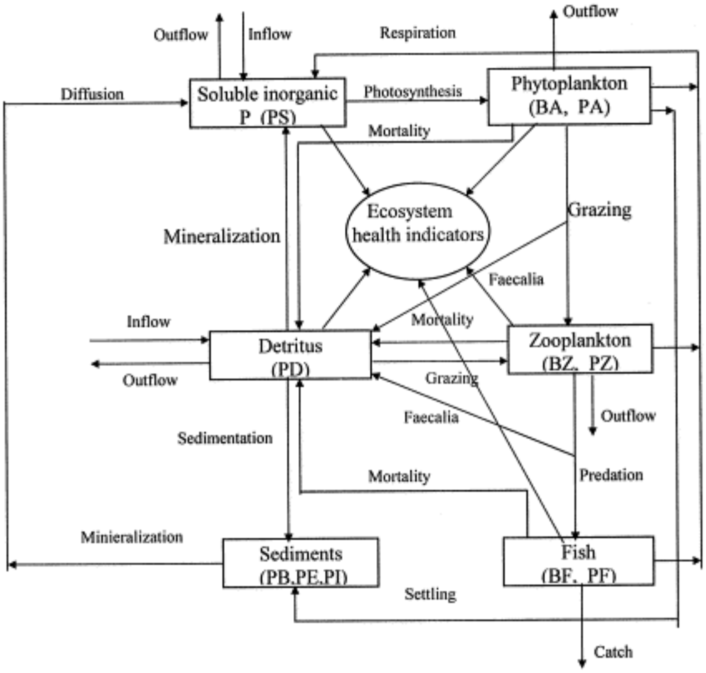

Figure 1: An example of a conceptual lake ecosystem model to identify ecosystem health indicators for Lake Chao (Xu et al., 2001).

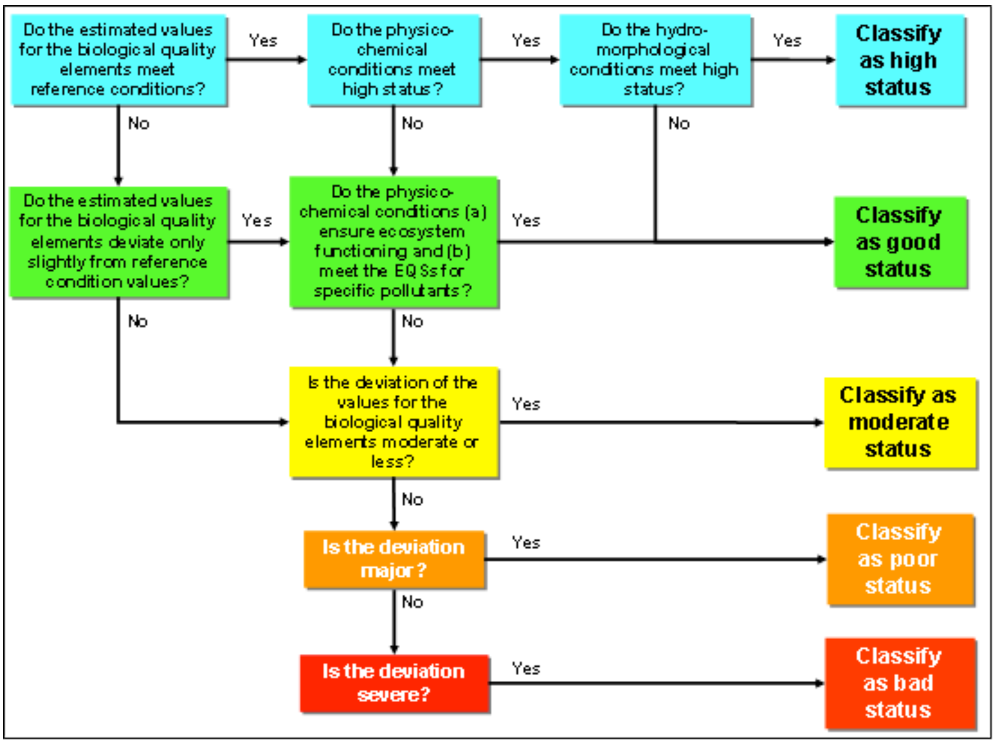

Figure 2: A Decision Tree for classifying the chemical and ecological status of water bodies according the Water Framework Directive.

2. Modelling lake ecosystems and their responses to changing climate and land use in lake catchments: a brief overview

2.1. Introduction

The ecological quality of lakes is threatened by a large number of anthropogenic stress factors, in particular eutrophication, pollution of various types, overexploitation and invasive species, changes in land use and hydrology in the catchment, and climate change (e.g. Gulati and Van Donk 2002; Mooij et al. 2005; Jeppesen et al. 2009; MacKay et al. 2009). Before starting a lake ecosystem modelling project, it is essential to be aware of existing models and concurrent approaches and to properly conceptualise the issues, the variables, the time and space scales and the desired outcomes for the model simulations (Van Waaveren et al., 1999). To improve our understanding of lake ecosystem processes and the ecosystem responses to changing conditions (multiple stressors or pressures) as a result of changing climate and land use in lake catchments, lake management and other human activities, modelling tools can be very useful to capture the essential ecological processes in shallow and deep lakes. Over the years, many different modelling systems have been developed and their applications have been reported in scientific journals. Recently, Reichert & Mieleitner (2008), Mooij et al (2010) and Jørgensen (2010) have made an overview of different lake ecosystem modelling approaches. In this chapter, we review different lake ecosystem modelling approaches in the context of implementation of WFD.

2.2. Lake ecosystems: degradation and recovery processes

Increased nutrient loads from anthropogenic sources have led to eutrophication of many lake ecosystems world wide and consequently resulted in dramatic changes in species composition, and foodweb structure and availability of suitable habitats for lake species (Wetzel, 1992; Jeppesen et al, 1997; Zhang et al., 2006; Burger et al., 2008). In eutrophic lakes in particular, nitrogen (N) and phosphorus (P) are the main factors that are driving lake ecosystem productivity and foodweb structure, including the resulting species composition of aquatic communities (Jeppesen et al., 2000, 2005). In shallow lakes, eutrophication has resulted in very turbid lakes due to recirculation of suspended sediments, in addition to the occurrence of algal blooms. As a consequence, macrophytes are disappeared and benthic vertebrates and fish communities are changed dramatically. Nowadays there is a strong focus on implementing measures to restore lake ecosystems. Attempts to restore lake ecosystems or mitigate ecosystem responses to eutrophication, such as algal blooms, generally focus on reductions of point sources from the lake catchment (Meijer et al, 1999; Søndergaard et al, 2003). There are good examples of lake ecosystem recovery recorded, however in many cases the dominance of cyanobacteria and white fish species (e.g. bream) pre-exist despite significant improvement of the nutrient levels in the lakes (Søndergaard et al., 2007; Gulati et al, 2008). Even with severe constraints on point sources loads to lakes, recovery from eutrophication has frequently delayed due to high rates of internal recycling of nutrients between sediments and overlying water column and wind driven recirculation of sediments (Jeppesen et al, 2003; Penning, 2012). Scheffer (1990), Scheffer et al (1993), Scheffer & Van Nes (2007) reported on multiplicity of stable states of lake ecosystems and different thresholds for turning into different stable states: thresholds for nutrient levels with respect to lake ecosystem recovery from eutrophic conditions into a clear lake with macrophytes are much lower then the thresholds related to the impacts of nutrient enrichment. For cold lakes, restoration methods encompass both bottom-up and town-down controls, whereas for warm water lakes bottom-up or nutrient control methods appear to be most significant for eutrophication control (Beklioglu et al., 2011). Warm lakes tend to be more productive than in cold lakes with similar nutrient concentrations. With a diverse and abundant omnivorous fish community, the predation pressure on zooplankton is strong, top-down control becomes less important and nutrient control emerges as a decisive factor for the water clarity. As for climate change, the structure and functioning of cold temperate European shallow lakes are expected to become more similar to those of the shallow lakes in the Mediterranean region, as the temperature increase will enhance the top-down controls of omnivorous and benthivorous fish as well as the nutrient cycling (Beklioglu et al., 2011).

2.3. Empirical models

Lake ecosystem responses to increased nutrient loadings have triggered a numerous series of model approaches. A classic modelling approach of ecosystem response to eutrophication is the mass balance equations approach, based on the Vollenweider concept. Models based on this concept are empirical steady state models. There is a large suit of empirical models available focusing on predicting chlorophyll-a concentration from nutrient loading to lakes. These types of models are widespread applied, especially to quantify the impact of eutrophication. Many applications are focusing on phosphorus (P), however, other nutrients, including nitrogen (N), silica and macronutrients can also limit phytoplankton growth as well as changes in light conditions (Elser et al., 1990; Burger et al., 2007; Kosten et al., 2009; Los, 2009). Despite its limitations, empirical regression models on nutrient loading and phytoplankton biomass are a useful tool to make a first assessment of the response of phytoplankton biomass to increases in nutrient loading (Figure 3), especially because of they are easy to use. Empirical regression models are based on large datasets. If such large datasets are not available expert knowledge models and fuzzy logic models could be developed (see chapter 2.7).

Many of the existing empirical lake ecosystem models were developed for phytoplankton-dominated lakes without presence of macrophytes. However, there are a few empirical models that are addressing the role of macrophytes to water transparency (Hamilton and Mitchell 1996). To analyse the role of submerged macrophytes in shallow lakes, more complex models were developed (Scheffer 1998; Jeppesen et al. 1998).

The empirical regression models have been extensively used by water managers world-wide for setting targets for acceptable nutrients concentrations and nutrient loadings in lakes, including defining ecological quality ratios for phytoplankton to support the implementation of the Water Framework Directive (Duel et al, 2007, Phillips et al., 2010). To cope with the high coefficient variation in the empirical relationships, different lake types have been distinguished.

Besides regression models on nutrient-phytoplankton biomass relationships, there are also regression models that are linking nutrient levels (with additional lake features such as water depth and lake size) with other biological elements such as bacterioplankton biomass, zooplankton biomass, zoobenthos biomass, macrophyte coverage, fish biomass and production, bird numbers and richness (see also Mooij et al, 2010).

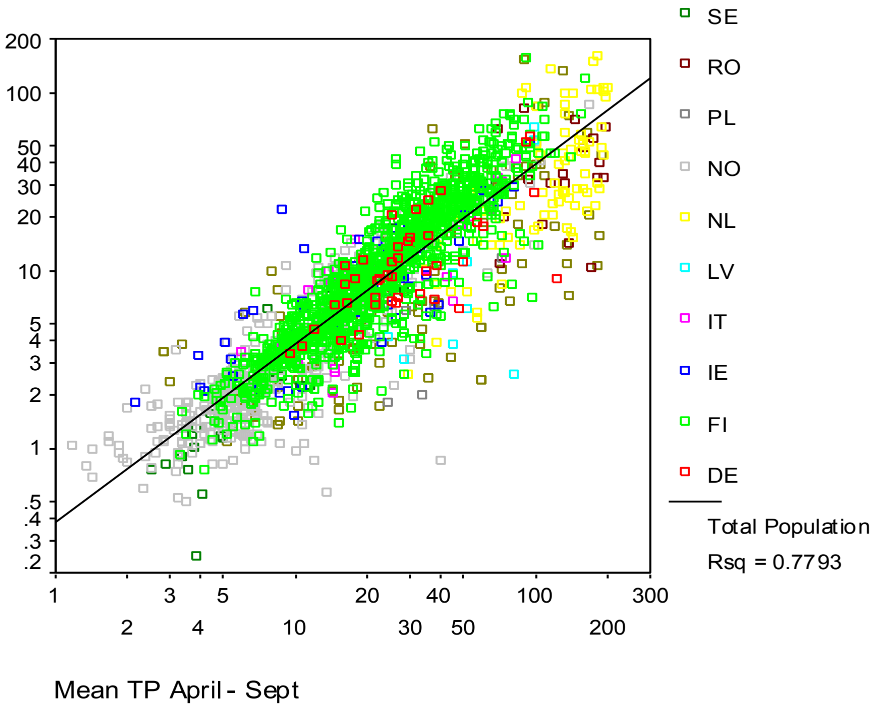

Figure 3: A linear regression of summer-average chlorophyll-a versus summer average Total-P.

2.4. Complex dynamic models

As (in many cases) empirical steady state models and expert models do not capture the complex ecological phenomena that occur in natural systems, such as response time to measures, foodweb relationships, temporal variations and feedback mechanisms. Complex dynamic models that are describing hydrodynamics – sediment dynamics, physio-chemical processes and biological processes are therefore often used to capture ecosystem processes and in relation to both hydrodynamics and bio-geochemical cycles. There is a large number of software packages available that are capturing hydrodynamic processes, biogeochemical process in water column and sediments, and biological processes. In general, the biological processes are focusing on plankton communities and have included simple foodweb structures (primary producers, grazers, filter feeders) and population dynamics (of plankton). Examples of such models are (in alphabetic order):

- CAEDYM (Computational Aquatic Ecosystem Dynamics Model), is a process-based library of water quality, biological and geochemical (sub)models that may run independently or coupled with hydrodynamic models 1Dv DYRESM (Dynamic Reservoir Simulation Model) or the 3D ELCOM (Estuary and Lake Computer Model) (ELCOM) to account for hydrodynamic processes such as transport and mixing. CAEDYM consists of a series of mathematical equations representing the major biogeochemical processes influencing water quality and its biological responses (Hipsey et al, 2007; Leon et al, 2011). Although CAEDYM can model multiple phytoplankton functional groups, zooplankton and fish, benthic biological communities (macro algae, macrophytes and benthic invertebrates) the model is mainly applied around the world for effects of increased nutrient loading on algal blooms and changes to phytoplankton succession.

- CE-QUAL-W2 is a 2D hydrodynamic and water quality model that simulates vertical stratification and longitudinal variability in key ecosystem components: chlorophyll-a as well as biotic groups, including periphyton, phytoplankton, zooplankton and macrophyte groups (Berger & Wells 2008). W2 models basic eutrophication processes as temperature-nutrient-algae-dissolved oxygen-organic matter-sediment relationships. W2 has also been used to drive models of food web dynamics (Saito et al. 2001) and to support studies of fish habitat (Sullivan et al. 2003).
- Coherens (Coupled hydrodynamical ecological model for regional shelf seas) is a 3D hydrodynamic model for coastal and shelf seas and it is coupled to biological, resuspension and contaminant models to simulate biological cycling processes in relation to currents, salinity, temperature and nutrients (Luyten et al, 1999). Recently, Coherens is also applied to Finnish lakes to simulate phytoplakton biomass (Liukko et al., 2009).
- Delft3D is a flexible integrated modelling suite, which simulates 2D (in either the horizontal or a vertical plane) and 3D hydrodynamics, sediment transport, sediment-water exchange, morphology, water quality and ecological processes for lake, fluvial, estuarine and coastal environments. Phytoplankton kinetics is simulated by the model BLOOM (Figure 4), which is based on a competition principle using the ratio between the actual growth rates and the resource requirements (Los 2009). The model maximises the net production of the phytoplankton community in a certain time period consistent with the environmental conditions and existing biomass levels. Algal diversity in freshwater applications is represented in three species groups: diatoms, flagellates and green algae and three genera of cyanobacteria: *Microcystis*, *Aphanizomenon* and *Planktothrix*. Delft3D can also be linked with HABITAT that is describing the habitat requirements of individual species or species groups. Delft3D has been applied to lakes all over the world, from small to large lakes, from shallow to deep lakes, from tropical to temperate lakes.
- EwE is an ecosystem modelling framework consisting of three main components: Ecopath, Ecosim and Ecospace. Ecopath is a static ecosystem model depicting mass-balances of different functional groups and their trophic interactions (Christensen & Walters, 2004). The model establishes mass-balances by solving sets of linear equations that describe the production and consumption of each group. Ecosim is dynamic model to describe temporal variations of the flows identified by Ecopath mass-balances. Ecospace is a spatial and temporal dynamic module primarily for exploring the impact and placement of protected areas. EwE is mostly used to study effects of fisheries' management policies in both marine and freshwater systems.

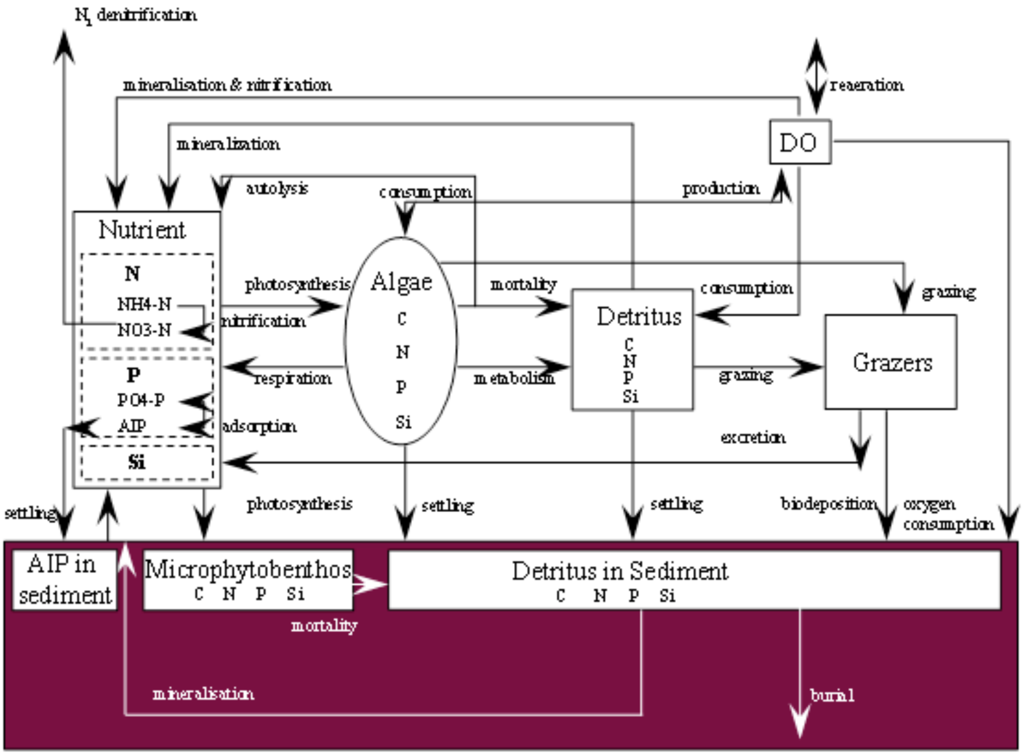

Figure 4: A schematic overview of the BLOOM model.

- LakeWeb is a dynamic model to predict phytoplankton biomass and production. LakeWeb also accounts for production and biomass of bacterioplankton, two types of zooplankton (herbivorous and predatory), two types of fish (prey and predatory) as well as zoobenthos, macrophytes and benthic algae (Håkanson & Boulion, 2002). Fundamental concepts include consumption rates, metabolic efficiency ratios, distribution coefficients, migration of fish and predation pressure. LakeWeb has been tested against empirical data sets from many lakes. LakeWeb can be applied to address the ecological responses of lake management issues, such as biomanipulation, changes in land-use (eutrophication and humification), acidification and global temperature changes. LakeWeb can simulate such measures and predict the positive and negative consequences of remedial measures (Håkanson & Boulion, 2002).
- MyLake (Multi-year Lake simulation model) is a 1D lake simple mechanistic model for predicting daily vertical distribution of lake water temperature and simulates also the evolution of seasonal lake ice and snow cover as well as nutrient cycles, sediment-water interactions and phytoplankton dynamics (Saloranta & Andersen 2007). MyLake has been applied to lakes in Norway and Finland (Kankaala et al. 2006; Saloranta & Andersen 2007).

- PCLake is an integrated ecological model of shallow non-stratifying lakes, describing phytoplankton, macrophytes and a simplified food web, within the framework of closed nutrient cycles (Janse et al., 2008, 2010). The model describes a completely mixed water body and comprises both the water column and the sediment top layer (10 cm), with the most important biotic and abiotic components (Figure 5). The upper sediment layer is included, to take into account sediment-water exchange and deposition history.

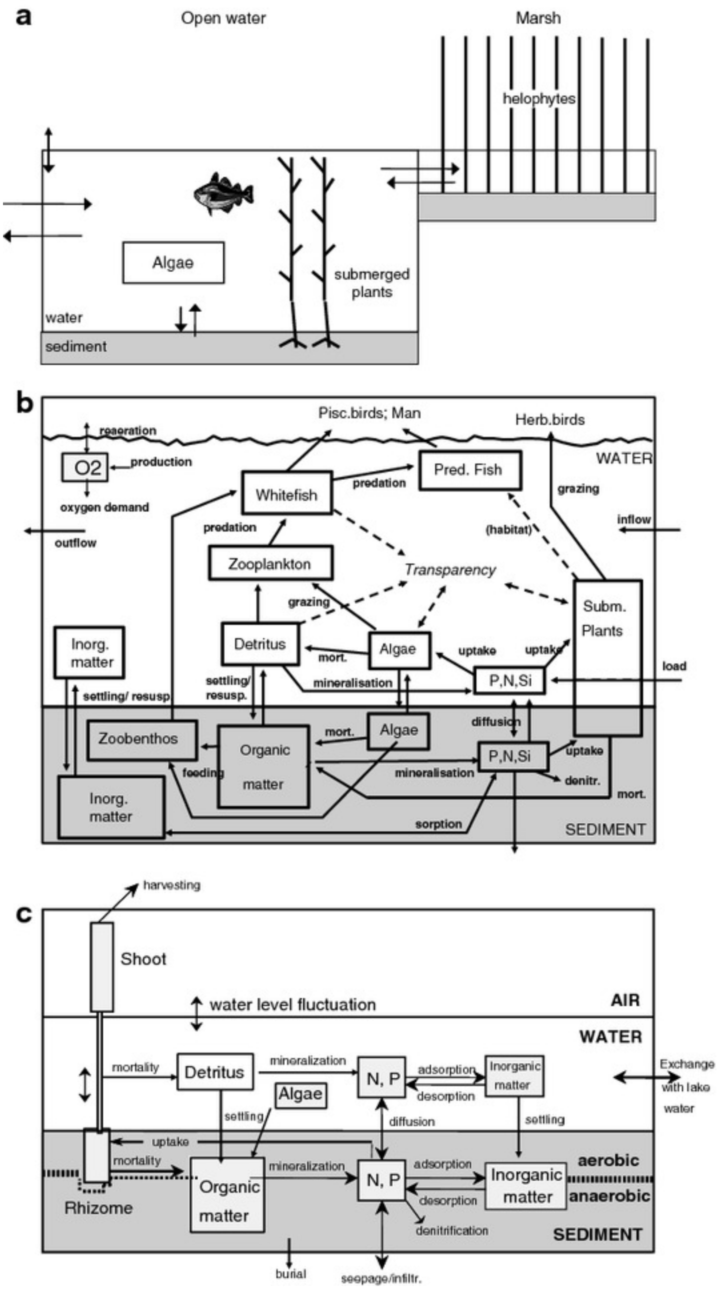

Figure 5: Schematic picture of the PCLake model. (*a*) general overview; (*b*) open water module; (*c*) marsh module (Janse ,2005).

Optionally, a wetland zone with helophytes can be added. No further horizontal (like depth variations) or vertical distinction within the lake is taken into account. All biota are modelled as functional groups. The main groups in the water phase are three groups of phytoplankton (diatoms, greens and cyanobacteria), zooplankton, planktivorous, benthivorous and piscivorous fish. Submerged macrophytes are included, consisting of a shoot and a root fraction. Further groups in the top layer of the sediment are the settled fractions of the three types of phytoplankton, as well as zoobenthos. PCLake has been used to estimate the critical nutrient loading levels for both forward and backward switches between the clear and the turbid state of shallow lakes, to identify the key processes determining the switch and the way critical loading levels depend on lake features and management factors (Janse et al. 2008).

- PROTECH (Phytoplankton responses to environmental change) is a rule based 1D model that simulates the dynamic responses of up to 10 species of phytoplankton (from a library of over 100) to environmental variability in lakes and reservoirs (Reynolds et al., 2001, Elliot and Thackeray, 2004). PROTECH calculates exponents describing growth and loss processes (mortality, sedimentation, consumption by grazing zooplankton), on the basis of the maximum growth rates of algal species in culture. These maximum growth rates are derived from relationships established between the algae's morphology and its growth rate subject to defined thresholds of light, temperature and nutrients. PROTECH has been extensively applied (Elliott et al., 2010).
- SALMO (Simulation of an Analytical Lake Model) is a 1D dynamic ecological model that simulates the most important pelagic food-web compartments of lakes and reservoir (Benndorf and Recknagel 1982). It simulates the seasonal development of temperature, stratification and turbulence as well as the concentrations of phosphorus, nitrogen, phytoplankton (three or more functional groups), zooplankton, oxygen, DOC (with a focus on humic substances) and suspended matter (four particle classes). With the exception of three lake specifc parameters, all other model parameters are considered general constants. SALMO-2 is a two box version, a spatially aggregated model with two horizontal layers (epilimnion and hypolimnion) and a variable mixing depth (Figure 6). SALMO is used for scenario analysis in decision making and as a research tool.
- SOBEK is a 1D/2D hydrodynamic model that simulates the complex flows and water related processes, including morphology, sediment-water interactions, water quality and biological processes in rivers and lakes. Similar to Delft3D, phytoplankton dynamics and biomass are simulated with BLOOM model (Los, 2009).

2.5. Ecotoxicological models

Risk assessments for aquatic ecosystems are mostly based on acute toxicity tests, bio-assays or analysis of accumulation of toxics in organisms (Hendriks, 1995). In recent decade, several modelling tools have been developed to predict the fate and effect of toxic substances in aquatic ecosystems (Koelmans et al., 2001). For risk assessments of the sediment quality in aquatic ecosystems, the TRIAD approach has been developed combining physico-chemical, ecotoxicological and biological information based on macroinvertebrates (Van de Guchte, 1992). However, there is not always a clear relationship between the chemical status and ecological status, even in water bodies with high toxic pressures. The impact of toxic pressures is due to interactions with both hydro-morphological conditions and nutrient levels highly complex and may influence all the levels of the aquatic ecosystems. For the prediction of the effects of toxic substances the bio-availability as well as the fate of the contaminants has to be considered.

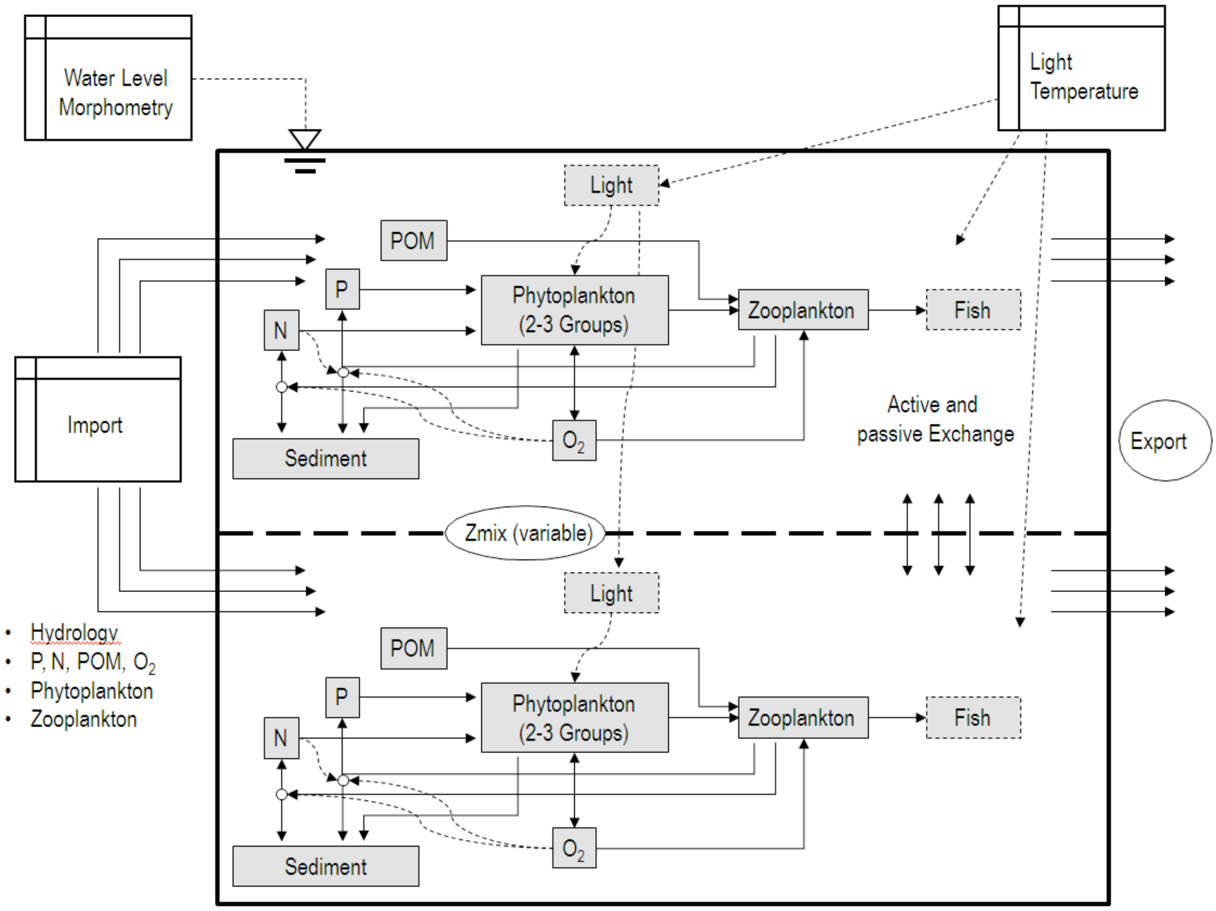

Figure 6: A schematic representation of the two layer version of SALMO-2 (Baumert et al., 2005).

OMEGA is a modelling tool for risk assessment of multiple toxic pressures based on species sensitivity distributions to toxic pressures (Posthuma et al, 2002). Species sensitivity distribution functions (SSD) are ecotoxicological extrapolation models that are representing the variation in sensitivity of species to a toxic substance by a statistical or empirical distribution function of responses for a sample of species (Figure 7). The basic assumption of the SSD concept is that the sensitivities of a set of species can be described by some distribution, usually a normal or logistic distribution. The available ecotoxicological data are seen as a sample from this distribution and are used to estimate the parameters of the SSD. The SSD is derived from sensitivity data obtained from acute or chronic toxicity tests, for example LC_{50}-values and no-observed-effect concentrations (NOECs) respectively. The number of data to construct SSDs varies widely. It is evident that the number of data is highly important for the derivation of the SSD and the conclusions based on them. The current OMEGA database contains validated toxicity datasets for a wide range of toxic substances. These datasets were used to derive environmental quality standards in the United States and Europe.

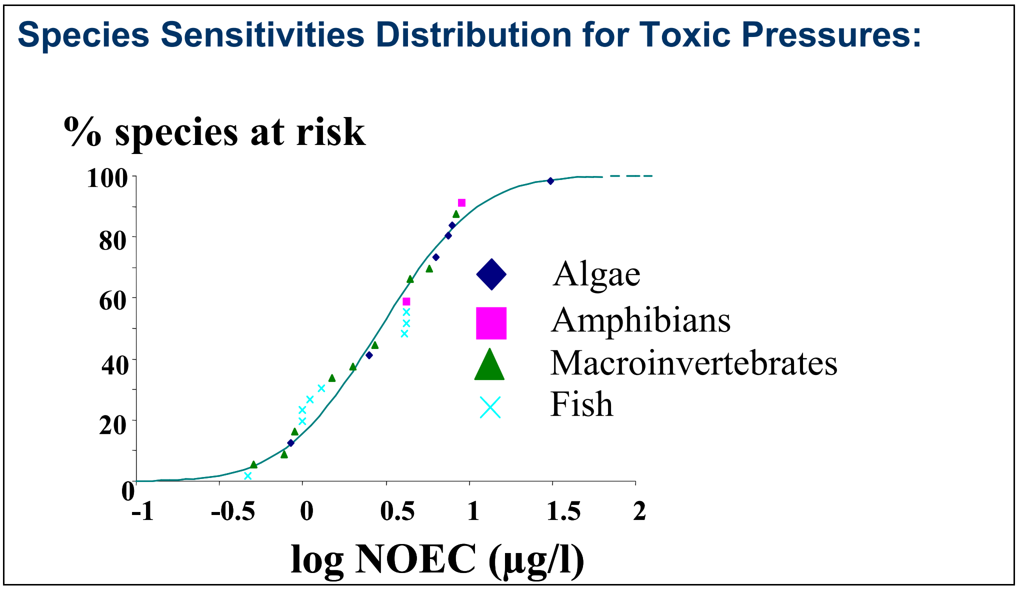

Figure 7: An example of combining individual NOEC levels of specific species to a combined assessment (Duel et al., 2007).

A basic assumption in ecotoxicological risk assessment using SSD is that laboratory-generated single-species toxicity data provide useful information about the communities to be protected. One of the issues of extrapolation of toxicity data from the single species (laboratory conditions) to the community level (field conditions) that should be addressed is the importance of ecosystem processes and functioning, such as the variation in biological availability of toxic substances, exposure routes and the occurrence of ecological interactions. It is evident that statistical extrapolation from a relatively small set of toxicity data to the real world by the use of SSD's and associated extrapolation techniques contains relatively large uncertainty.

The ecotoxicological risks of toxic substances on community level are expressed as the potential affected fraction (PAF). The use of the word potential indicates that PAF refers to a risk, the fraction of species estimated to be exposed beyond an effective concentration, and not at empirically observed fraction of species in a community that are affected. The risk assessment based on PAF is not taken into account temporal and spatial variability of the aquatic communities within water bodies: species abundance and diversity may differ in time and space within a water body.

To estimate ecotoxicological risks from multiple toxic substances different methods are used. When mode of action of the different toxic substances is known, risks will be estimated by addition of toxic units. In other cases, risks from multiple contaminants will be estimated by addition of PAFs (msPAF).

With respect to biological responses to metals in the aquatic environment, the biotic ligand modelling (BLM) approach is a valuable tool to evaluate quantitative the manner in which water chemistry affects the speciation and biological availability of metals in the aquatic systems (Figure 8). The BLM approach has gained widespread interest amongst regulatory governmental bodies because of its potential for use in developing water quality criteria and in performing aquatic risk assessments for metals. Specifically, the BLM does this in a way that considers the important influences of site specific water quality conditions (Paquin et al., 2002;

Verschoor et al., 2011). Based on the BLM concept, there is a set of different modelling tools available (Vink & Verschoor, 2010).

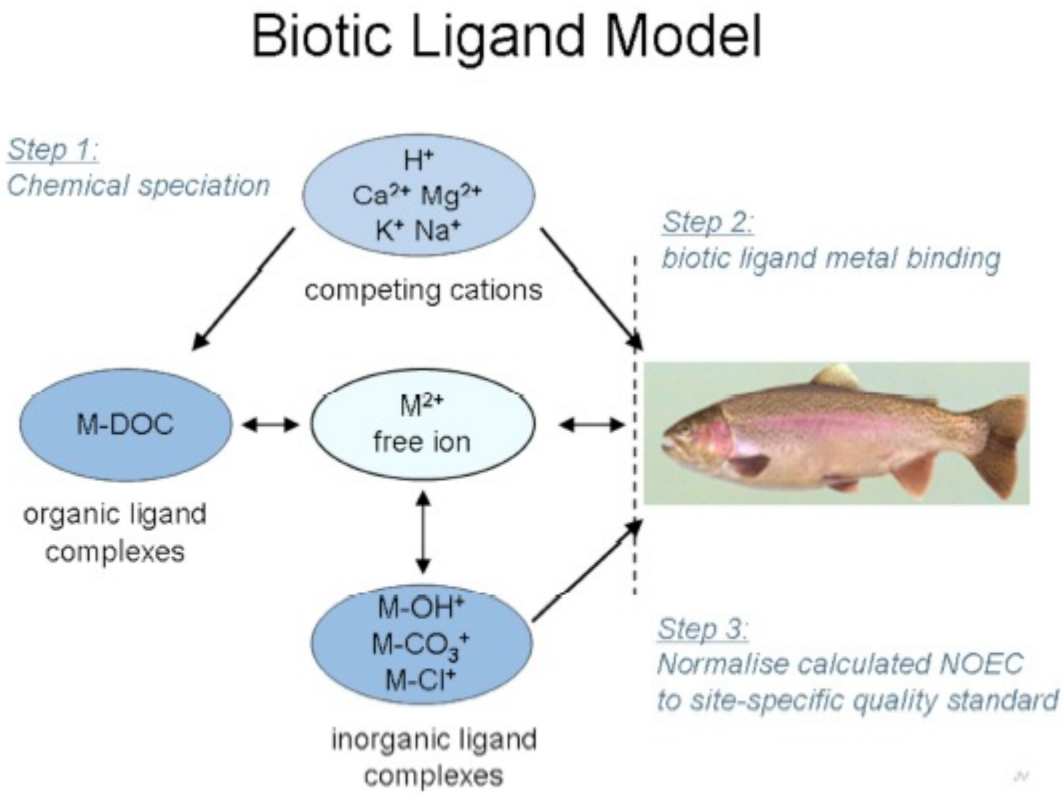

Figure 8: Concept of biotic ligand model, which includes calculation of chemical speciation, binding to biota and a normalisation procedure to calculate site-specific quality standards (Vink & Verschoor, 2010).

2.6. Data driven modelling approaches

The availability of large datasets and deterministic modelling techniques allows the development of ecosystem models with high reliability (see section 2.2.3). However, during recent years, new techniques such as artificial neural networks, fuzzy logic, classification trees and Bayesian belief networks proved to have a high potential in ecological modelling as well, as they combine reliable predictions with gaining insight in ecosystem interactions (Lek and Guégan, 1999; Recknagel, 2003; Goethals, 2005; Adriaenssens et al., 2006). Artficial neural networks and classification trees are data driven methods, whereas fuzzy logic and Bayesian network models are knowledge based methods that can be of considerable importance, in particular when enough data of good quality are missing to develop data driven models.

Artificial Neural Networks (ANNs)

Artificial Neural Networks are non-linear modelling approaches to classify current ecological state or predict future ecological state as a result of changes in environmental conditions. Various types of neural networks exist, but most popular ANNs are multilayer feed forward neural networks with back propagation algorithm (Hagan et al., 1996; Goethals, 2005). The back propagation network constructs a model based on existing data (e.g. environmental conditions) with known outputs (e.g. biological indicator values), for example the relationship between macro-invertebrate species index and environmental variables such as water depth, sediment types, dissolved oxygen, pH, nutrient concentrations etc. The architecture of the back propagation network is a layered feed-forward neural network in which the non-linear elements (neurons) are arranged in successive layers, and the information flows from input layer to output layer, through one or more hidden layers (Figure 9). There are no lateral connections within any layer, nor feedback connections are possible.

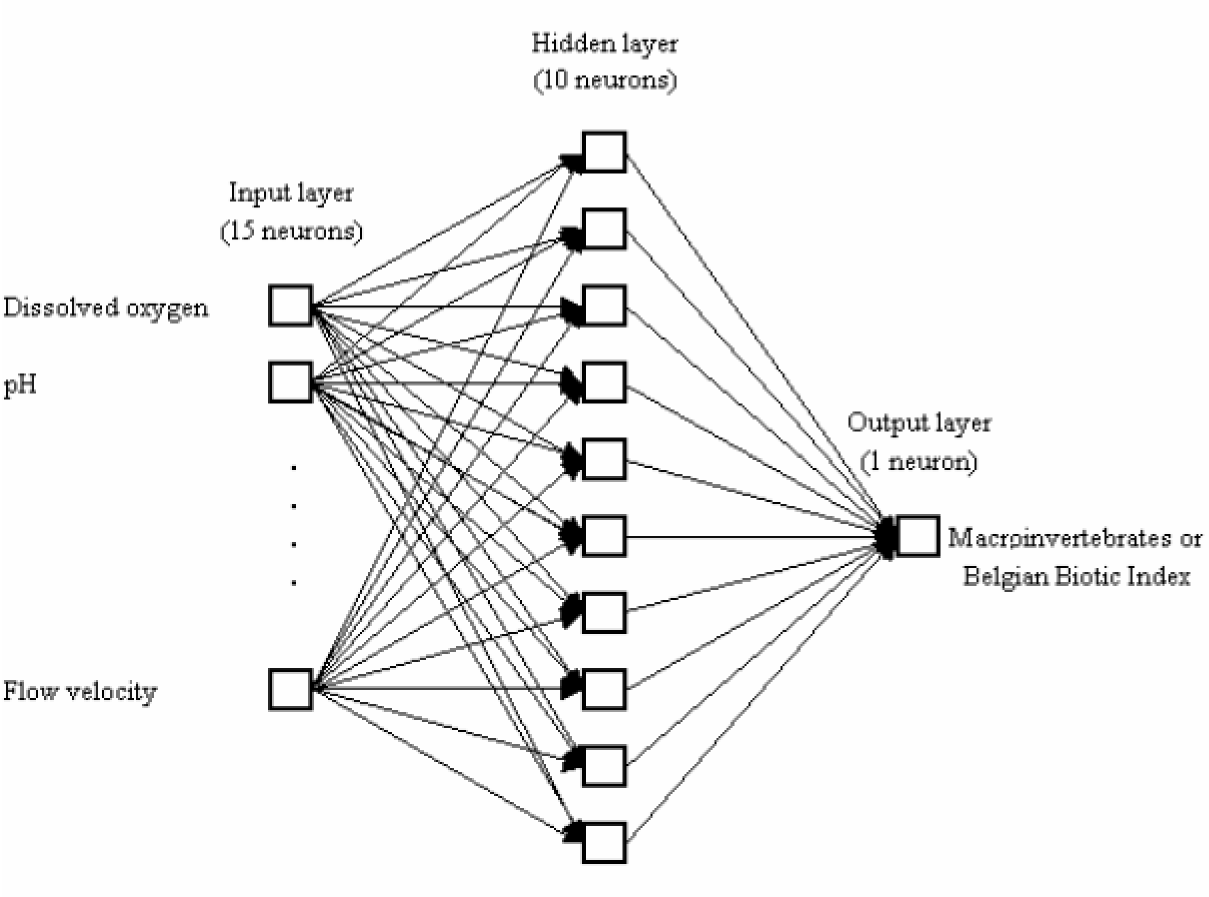

Figure 9: Illustration of a three-layered artificial neural network with input layer containing 15 environmental variables, one hidden layer with 10 neurons and one output layer (from Dedecker et al., 2002).

Classification trees

Classification and regression trees (often referred to as decision trees) is another technique to develop predictive models based on complex datasets (Figure 10). However, compared to ANN, applications of classification trees in modelling aquatic ecosystems are rather limited (Goethals, 2005). The common way to induce rules in the form of decision trees is the so-called 'Top-Down Induction of Decision Trees'. Tree construction proceeds recursively, starting with the entire data set. At each step, the most informative input variable is selected as the root of the sub-tree and the data set is split into subsets, according to the values of the selected input variable. In this manner, rules are generated that relate the values of input variables with the ecological parameters (e.g. the presence/absence of species, biological indices etc). For discrete input variables (classification trees), a branch of a tree is typically created for each possible value of that particular variable. For continuous input variables (regression trees), a threshold is selected and two branches are created based on that threshold. Tree construction stops when all examples in a node are of the same class. In order to reduce the noise in the data and to improve the predictive results with regard to complexity and accuracy of the predictions, several optimisation methods can be applied. Major examples are pruning, bagging and boosting.

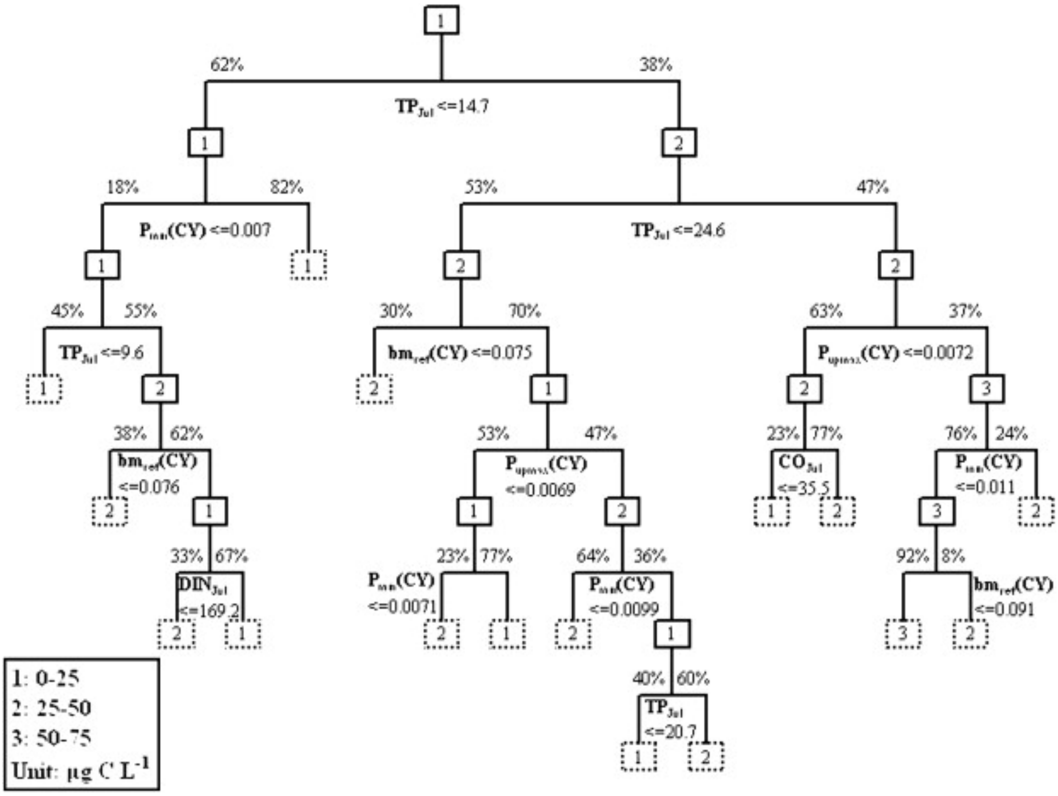

Figure 10:. Classification tree diagram of diatom biomass under nutrient enrichment conditions (Zhao et la., 2008).

2.7. Expert models

Expert models are generally applied in cases when there is no large dataset available to develop empirical regression models or when existing knowledge on specific ecosystem processes or habitat requirements of species is limited.

Fuzzy logic models

Fuzzy logic models consist of a number of conditional “if-then” rules to describe relationships or processes that cannot be easily described by deterministic equations. For example: when environmental variables are difficult to quantify or classify due to high level of uncertainty as a result of high level of temporal or spatial variability (e.g. substrate type, wind exposure, light conditions) and therefore often not appropriate to use as conditional values for prediction of biological processes or population densities, fuzzy logic modelling is a valuable technique for this purpose. The fuzzy logic approach enables to process imprecise qualitative information by means of membership functions into quantitave information (real values). The membership functions are explaining the degree of truth of any fuzzy statement by values between 0 and 1 (Figure 11). Fuzzy logic models are also applied to lake ecosystems, especially to predict the occurrence of cyanobacteria blooms in lakes (Chen & Mynett, 2003; Ibelings et al., 2003).

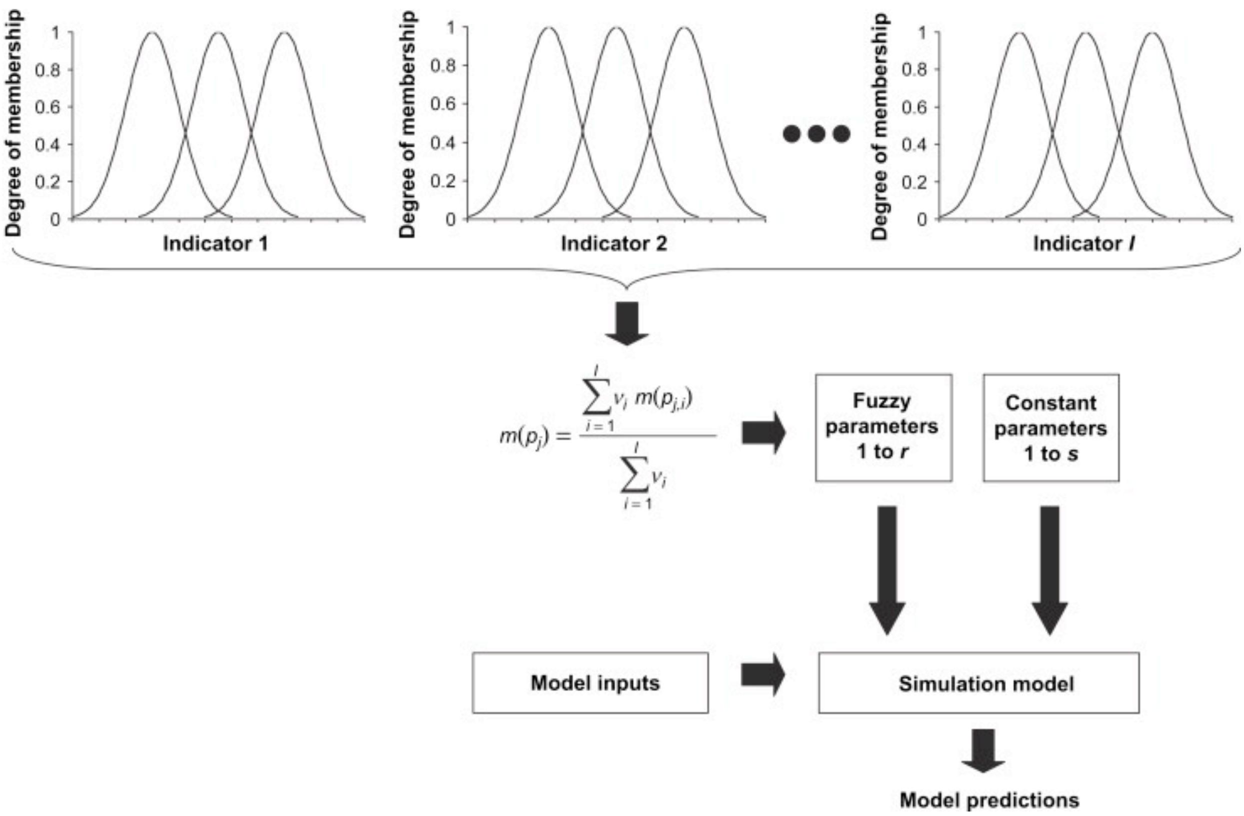

Figure 11: Schematic representation of a hybrid fuzzy-mechanistic simulation model (Lauzon & Lence, 2007).

Bayesian belief networks (BBNs)

BBNs are models with a network structure that focus on the explicit representation of 'cause-and-effect' relationships between variables, representing in this case ecosystem components. The network architecture is linked to probability distributions that allow it to deal with variability and uncertainty in the models. This is particularly useful for the description of ecological systems (Regan, 2002). Bayesian belief networks are probabilistic expert systems in which the knowledge base has two components: (1) a network of causal relationships between variables and (2) a set of conditional probability matrices that relate each variable to its causal variables (Figure 12). Bayesian models are based on the concept that defines the formalism of updating a belief about a hypothesis (or a priori probability) in the light of new evidence (e.g. new data).

The updated probability is called the posterior probability. A common application of Bayesian models is stock assessment especially in fish ecology and fish management (Baran & Jantunen, 2003; Baran et al, 2006). Bayesian models are considered to be a powerful tool to increase knowledge about aquatic ecosystems by the integrative analysis of probabilities of models and observation data (Lek et al., 2005).

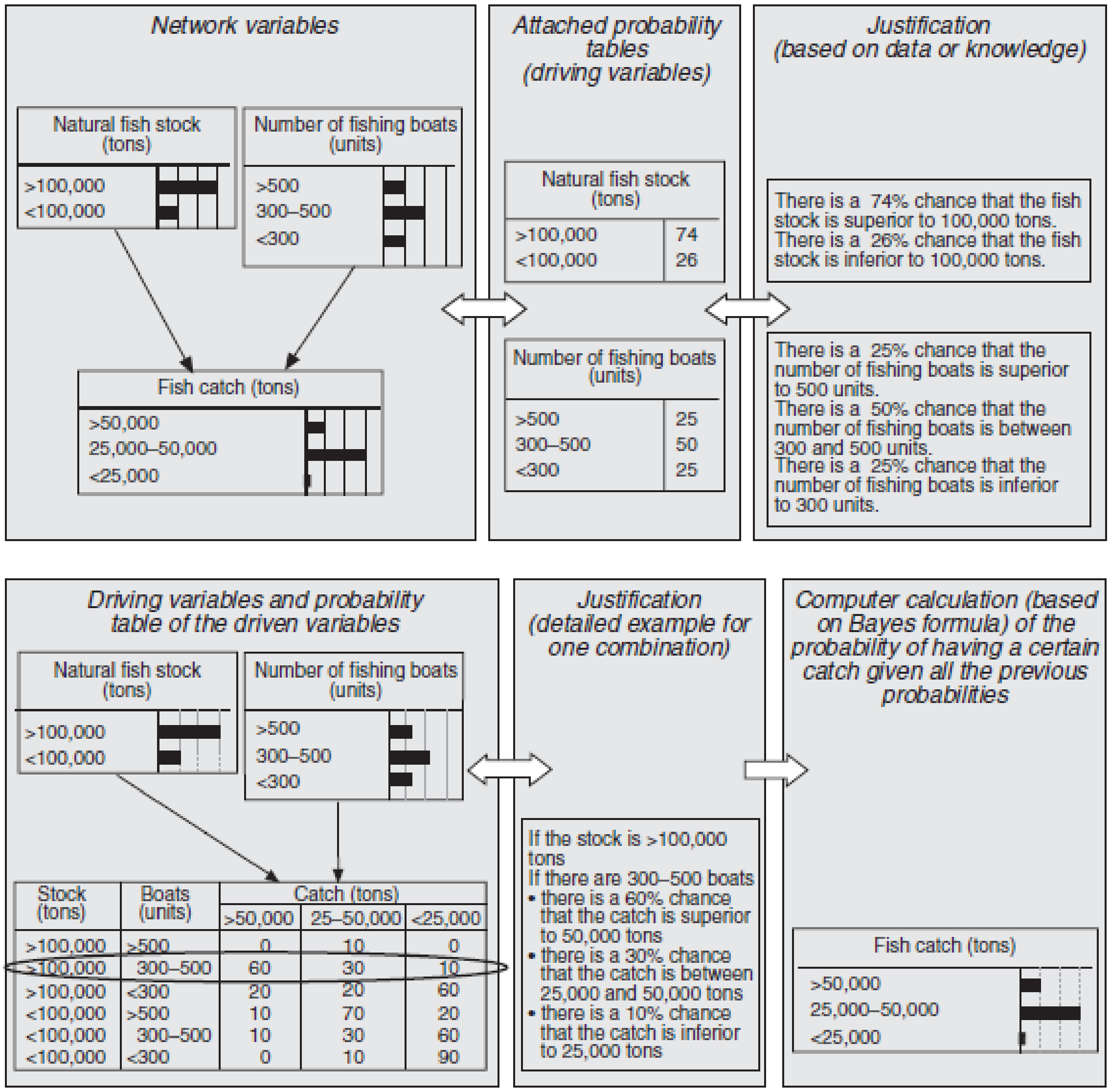

Figure 12: An example of a Bayesian model representing a hypothetical fishery (Baran et al, 2006).

2.8. Structurally dynamic models

In structurally dynamic models, the parameters are constantly varied to account for adaptation and shifts in the species composition (Zhang et al., 2010). These models are more popular in fundamental science than in applied science, as they aim at development of theory of complex lake ecosystems rather than making realistic predictions to support lake ecosystem management (Mooij et al., 2010). In general, structurally dynamic models can be valuable to answer questions related to specific biological processes, but should be used with great caution when predicting ecological effects of changing environmental conditions, e.g. due to climate change or measures to restore the lake ecosystem.

There are different types of structurally dynamic models. Minimal dynamic models are very simple structurally dynamic models and have been mostly applied to study predator – prey interactions (Scheffer et al., 1997). Individual based models usually focus on the population structure (e.g. size, weight, age) of a few ecological groups (especially macrophytes, fish). For modelling large populations in lakes, it is generally too computationally demanding to model all individuals seperately. Therefore super-individual models have been developed. In the super-individual approach each individual has an extra property, namely the number of individuals that it represents. This approach has been applied in macrophyte model Charisma (Van Nes et al, 2003), the fish model Piscator (Van Nes et al., 2002) and the zebra mussel model Driessena (Van Nes et al., 2008). Charisma describes the seasonal cycle of macrophytes in shallow lakes. It is especially detailed in the description of photosynthesis, and can model self shading and shading among different species. Piscator has eight interacting fish species, three types of fishery, piscivorous birds and a simple representation of the fish food. The model can include size differences among year classes by defining different super individuals with slightly different growth rates. Dreissena model describes the growths of the zebra mussels (Dreissena polymorpha). The model is spatially explicit and predicts length frequency distributions of zebra mussels.

Another type of structurally dynamic model is based on the Dynamic Energy Budget theory (Kooijman, 2012). Dynamic Energy Budgets (DEB) models are linking physiological processes of individual organisms such as ingestion, assimilation, respiration, growth and reproduction, in a framework of ecosystem modelling. The DEB theory assumes that every organism has a certain amount of energy as input (food) which is stored after assimilation (Figure 13). Since assimilation is never 100% efficient, a part of the energy ends up in faeces. The assimilated energy is stored and can be used for growth (structure), somatic maintenance (metabolic rate), maturity maintenance (metabolic processes) or reproduction. Recently, there is a great interest in developing DEB models to study the changes in environmental conditions on the population size of species, especially key species in the food chain (e.g. Wijsman et al, 2009; Troost et al, 2010). Janssen (2012) has applied DEB models to simulate the response of changes in water temperature and water quality on the carrying capacity of a lake ecosystem for mussels, as changes water temperature and nutrient concentrations have a large impact on food availability for mussels.

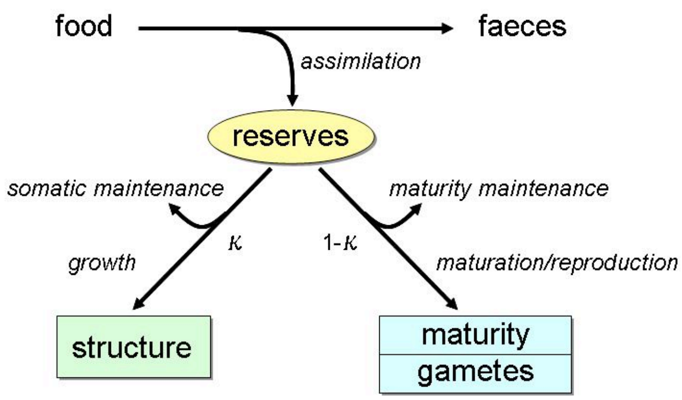

Figure 13: A schematic representation of a DEB model.

2.9. Habitat models

Habitat models are species specific models that are providing habitat information to evaluate the impact on availability and suitability of habitats for species resulting from changes in environmental conditions. Habitat models generally define habitat suitability based on the provision of certain habitat attributes required for living and/or reproduction. Empirical relationships, scientific literature and professional expertise are incorporated into the equations to describe the influence of environmental conditions on the suitability of habitats available (Figure 14). When monitoring data on both species abundance and environmental parameters are available, habitat preference curves can be developed. With respect to aquatic ecosystems, habitat models are commonly used to analyse habitat suitability for fish species in rivers. Nowadays, habitat models are widespread used for water management and as a result there is a large set of habitat models available for species that are characteristic for river and lake ecosystems (Duel et al., 2005; Goethals, 2005).

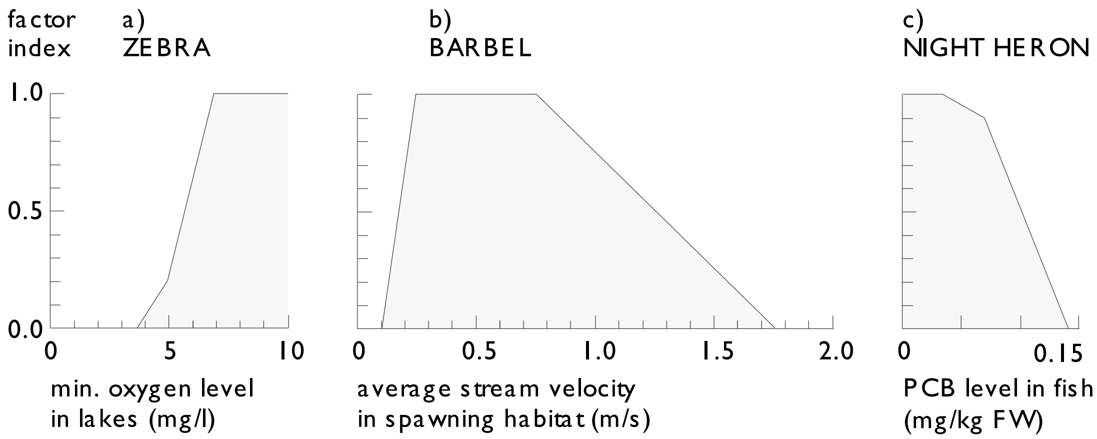

Figure 14: An example of habitat suitability curves (Duel et al., 2005).

3. Modelling approaches in WISER case studies

3.1. Introduction

In this project, different model approaches were tested with respect to their applicability to address the ecological responses to changes in hydrodynamics and water quality and to quantify ecological impacts of mitigation and restoration measures. A selection of existing lake ecosystem models were applied in two case studies: Lake Veluwe, a shallow lake in the Netherlands where ecosystem recovery took place after implementing restoration measures and Lake Pyhäjärvi in Finland, where pressures on lake ecosystem are increasing due to land use changes in lake catchment. In this chapter, the different modelling approaches are described briefly. For more details about the application of the selected models for the two case studies, we refer to WISER deliverable D5.2-1: Analysis of applied modelling approaches.

3.2. Case studies

3.2.1 Lake Pyhäjärvi

4.3.1. Study area

Pyhäjärvi is a large (155 km^2) but shallow mesotrophic lake in southwest Finland (Figure 15). Two major rivers, Yläneenjoki and Pyhäjoki, discharge into Lake Pyhäjärvi.

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 17), 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., 2008). 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. Lake 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).

Internal processes contribute considerably to eutrophication of Lake Pyhäjärvi (Ekholm et al., 1997). Sarvala et al. (1998) have shown that inter annual variations of the chlorophyll and P concentrations in Lake 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 Lake Pyhäjärvi, small and water quality effects moderate. That is why Lake 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).

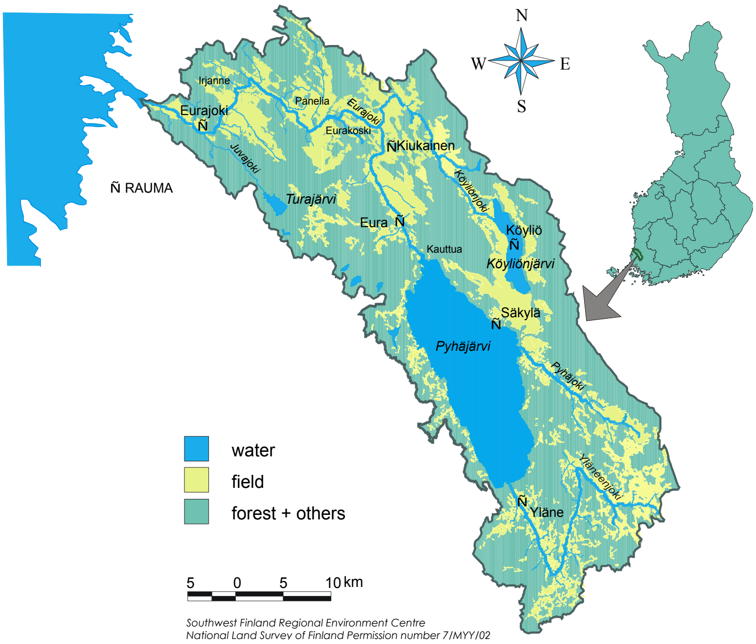

Figure 15: Case study Lake Pyhäjärvi

3.2.2 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 (Figure 16). Lake Veluwe is about 3250 ha, and has two main sections: a shallow section of about 1 meter depth bordering the old land (Pleistocene sand deposits) and a deeper section, varying from 3 to 5 meters that borders the new land (old marine clay deposits). In the shallow part, seepage occurs from the old land while in the deeper part infiltration takes place to the Flevopolder. Additionally, the lake receives water from 17 streams (Van der Molen et al., 1998), a waste water treatment plant and the pumping station 'Lovink' that is responsible for the largest quantity of incoming water to the lake. Lake Veluwemeer is of both national and international importance, as it contributes to the conservation of specific macrophytes, macroinvertebrates and birds. To warrant proper protection of these species, it has been assigned as a bird and habitat directive area

Figure 17: 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 et al., 1998). These macrophytes supported a large bird population. 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 to a turbid, algae and large bream (*Abramis brama*) dominated state (Scheffer et al., 1994). The algae blooms were year round and dominated by *Planktothrix*. Concurrent with the disappearance of the macrophytes, the bird population became severely reduced because of food shortages (Noordhuis et al., 2002).

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

- Increased flushing with nutrient poor water of the lake by pumping station Lovink. 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 waste water treatment plant of Harderwijk (WWTP) was increased with circa 85 %.
- Between 1990 and 1995 biomanipulation, by means of catching breams, was conducted (Lammens et al., 2004).
- In 2002 the obstruction between Lake Veluwe and Lake Wolderwijd was removed to increase the discharge of nutrients from Lake Veluwe towards Lake Wolderwijd
- In 2003 the third treatment step for the WWTP Harderwijk was implemented to further reduce nitrogen concentrations.

During recovering of the lake, several measuring and modeling studies were conducted (e.g. Van den Berg et al., 1998; Van der Molen et al., 1998; Scheffer et al., 2004). Measuring water quality and biological components lead to insight in trends of those parameters in Lake Veluwe. Modeling studies were mostly performed for hind casting purposes: how effective were the measures taken? What were the key measurements? However, modeling studies were also used to gather knowledge on how the system works during steady states and system shifts (e.g. van Nes et al., 2002; Ibelings et al., 2007). Nowadays, Lake Veluwe is in a clear water state with high abundance of macrophytes in the shallow part of the lake and chlorophyll-a concentrations around 15 µg/l, except during the early spring diatom bloom when concentrations are around 40 µg/l. Although the current ecological status of the lake appears to be good, climate change and upstream land use changes may alter this status.

3.3. Modelling framework for Lake Pyhäjärvi

Several catchment models and lake models have been applied in Lake Pyhäjrvi and its catchment. They are listed with their main characteristics in Table 1.

WSFS-VEMALA

The operational WSFS-VEMALA model simulates hydrology and water quality for all river basins in Finland. 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 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). 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. Year is divided into four seasons. Another tool in WSFS-VEMALA is VIHMA. It estimates long term mean loading (kg * m^{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. Nutrient load transport, sedimentation, erosion, and denitrification in the river are simulated. Nutrient sedimentation, internal loading and denitrification in lakes are simulated. Nutrient balance for lakes is simulated according to the mass balance equation. 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 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.

Table 1: 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 agri-culture and land use
INCA	Semi-distrib.	Catchment	TP, TN, runoff	Agriculture, climate, land use	Changes in agri-culture and land use
SWAT	Semi-distrib.	River basin	TP, TN, runoff	Agriculture, climate, land use	Changes in agri-culture 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 model	River basin, lake	Management measures, Cyanobacteria, zooplankton, planktivorous fish, costs, benefits, decisions.	Agriculture, climate, land use	Buffer strips, wetlands, forestation of fields, reduction of nutrient loads, fisheries management

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.

WSFS-VEMALA model results show that in the climate change situation long term mean phosphorus loading into Pyhäjärvi will have following changes:

- 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,
- there is no any more clear phosphorus load peak, because there is no any more clear snowmelt peaks,
- 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,
- phosphorus inflow loading during the summer stays about at the same level.

INCA-N

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. 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-storage approach, including direct runoff and reservoirs of water in the reactive soil zone and deeper groundwater zone. The catchment is divided into sub-catchments. INCA-N can simulate water flow and N processes in six land use classes. The INCA-N model was calibrated in Yläneenjoki catchment for period 1995-1999 and tested for period 2003-2007. 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.

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 timing and magnitude of the flow peaks was simulated well for the whole calibration period. The calibrated model was able to simulate adequately the overall annual inorganic N dynamics in river water. Discrepancies in the simulated versus observed concentrations during single peaks were probably partly related to under- or overestimation of instantaneous discharge and simplifications made during the whole modelling process (Granlund, 2008).

The SWAT model (Soil and Water Assessment Tool) can be used to simulate water and nutrient cycles in agriculturally dominated large catchments (Arnold et al., 1998; Neitsch et al., 2005). The model has been world widely used and also further developed (e.g. Eckhardt et al. 2002; Krysanova et al., 1999; Van Griensven and Meixner, 2003).In the model application 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.

SWAT was applied to Yläneenjoki catchment (Koskiaho et al., 2003; Bärlund, 2007). For the 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 the main channels were somewhat deepened to emphasize the actual routes of water. The agricultural crops data were divided into 5 crop classes (autumn cereals, spring cereals, root crops, grasses and gardens). All the rest land area was classified as forest. Soil type data was based on soil textural information of the Geological Survey of Finland with four soil types namely clay, moraine, silt and peat.

The present SWAT application resulted in 29 sub-catchments and 257 HRUs. In calibration simulated daily averages of discharge were compared against the observations at the Vanhakartano for the years 1995–1999. Daily and monthly averages of flow were calibrated against the corresponding values determined from the observations made at Vanhakartano.

The results from 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. 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).

LLR

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. It 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,
- a hierarchical, linear regression model for chlorophyll-*a* (Malve, 2007) and
- a logistic regression model for phytoplankton biomass (Kauppila P., Lepistö L., Malve O. & Raateland A. Unpublished).

The model assumes complete mixing throughout the lake. 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. Chapra's sedimentation rate is also expected to be normally distributed. The model parameter estimation and predictions 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 fitted to Finnish, European and North-American data sets published elsewhere. 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). 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 pools information 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 logistic regression model for phytoplankton biomass 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 (clear and humic lakes) that consist of observations from Finnish and Norwegian lakes.

LLR model was used to estimate 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. According to the model there is no need of reducing the loading, as the present load results to good water quality of the lake.

LakeState

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

The model was capable to capture major dynamics of total P in 1990-2001. Phosphorus model fitted to observations quite well and the fit was better than in the nitrogen model. 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). 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. The model was linked with catchment model and results are discussed below.

MyLake

MyLake 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 (Saloranta & Andersen, 2007).

The set up of MyLake model for Lake Pyhäjärvi was quite straitforward as required data were 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. Model predicted lake water surface temperature very well. Futher results will be published soon.

Coherens

The 3D Coherens lake model was originally designed as a regional model for the North Sea (Luyten et al., 1999), but can be applied for lakes as well. The water quality model used in this study is a total nutrient model where the growth of algae depends on the nutrient concentrations during . In this work, a special emphasis in the model development has been put on the erosion of particulate phosphorus. The model was used to simulate the water quality constituents for a period of seven years (2001-2007). The model was able to predict the epilimnetic water temperature very well. In suspended solids computation soem discrepansies between obeserved and simulated values were found due to the the too high sinking velocity in model predictions. Variation of total phospohrus was higher in simulated values as in observations. Still the correspondce is quite good. In deep waters model gave too high water temperatures in summer and too low values in winter time. 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 was 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 seemed to be very high in the model results as compared to the observations.

The outputs from WSFS-VEMALA were used as inputs for 3D Coherens Lake model. Results showed that the maximum surface water temperatures will increase with few degrees and the cold water periods will be shorter that presently. The maximum concentrations of suspended solids will increase up to 10 ug/l and consequently the concentrations of total phosphorus will be markedly higher in winter time and even in summer the peaks will increase up to 40 ug/l higher as in present climate. 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 removed. Results of this scenario (scenario 8) were used as input data in Coherens model.

Influence diagrams in water quality simulations of a lake and linking of lake and catchment models.

Linking of models can be implemented e.g. by using output series from one model as input data series to the next model as described above, or by probabilistic linking, as shown in the following. LakeState model and a catchment model were linked together with Bayes network learning method (Hugin software). This linked model system made a more direct simulation of the actual management (abatement) operations easier. A catchment model that calculates phosphorus reduction percentage and the associated costs as a function of buffer strip width, forestation percentage and wetland percentage was used in Monte Carlo simulation to produce input for Hugin software Bayes network learning and estimation procedures. Simulations revealed conditional probabilities and correlations between variables.

In the end, the two Bayes networks of LakeState and catchment models were linked to estimate attainment of the designated water quality criteria or goal (the level of Cyanobacteria summer maximum biomass). Management options were implemented by decision nodes and the attainment of the water quality goal with a discrete change node and a utility node. 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 cost, Cyanobacteria summer maximum biomass and of attainment or non-attainment of the water quality criteria (Cyanobacteria < 0.86 mg/l). The results show that with a certain fisheries management scenario and with moderate catchment actions we obtain high probability for the attainment of the water quality criteria and associated costs.

3.3. Modelling framework for Lake Veluwe

Lake Veluwe is located in the Lake IJsselmeer area, and this area consists of a number of large and small shallow lakes. Since the mid- 80s of past century, efforts have been made to improve the water quality and ecological status of the water bodies. The lake ecosystems are recovering slowly. Unfortunately, the possible impacts of climate change were ignored when the measures for improvement of the ecological quality were selected. Therefore, it is unclear what will be the future of the lake ecosystems in the Lake IJsselmeer area. To answer this question, the Ministry of Infrastructure and Environment and regional governmental bodies have initiated a series of research projects. A modelling framework is developed to study the ecological processes in the lakes and to identify what are the impacts of changing climate and adaptation strategies (Figure 18). This modelling framework was the starting point for this case study in WISER, however the 1D/2D hydrodynamic model SOBEK-flow was used to investigate the hydrodynamic processes in the lake. To address water quality and ecological responses to changes in water temperature and water quality, a deterministic water quality and primary productivity model DELWAQ-G – BLOOM is applied (Figure 19). DELWAQ-G - BLOOM is a part of the Sobek modelling suite (see chapter 2.2.4) and suitable for modeling water quality and primary production in both fresh water and marine ecosystems (Los et al., 1988; Rip et al., 2007; Blauw et al., 2009). The model code has been previously published under various names, depending on the study and is also known as GEM (marine ecosystems) and DELWAQ-G (freshwater ecosystems). In this model chemical water quality processes (combined in a subset of processes called DELWAQ-G) and primary productivity dynamics (combined in a subset of processes called BLOOM) is available.

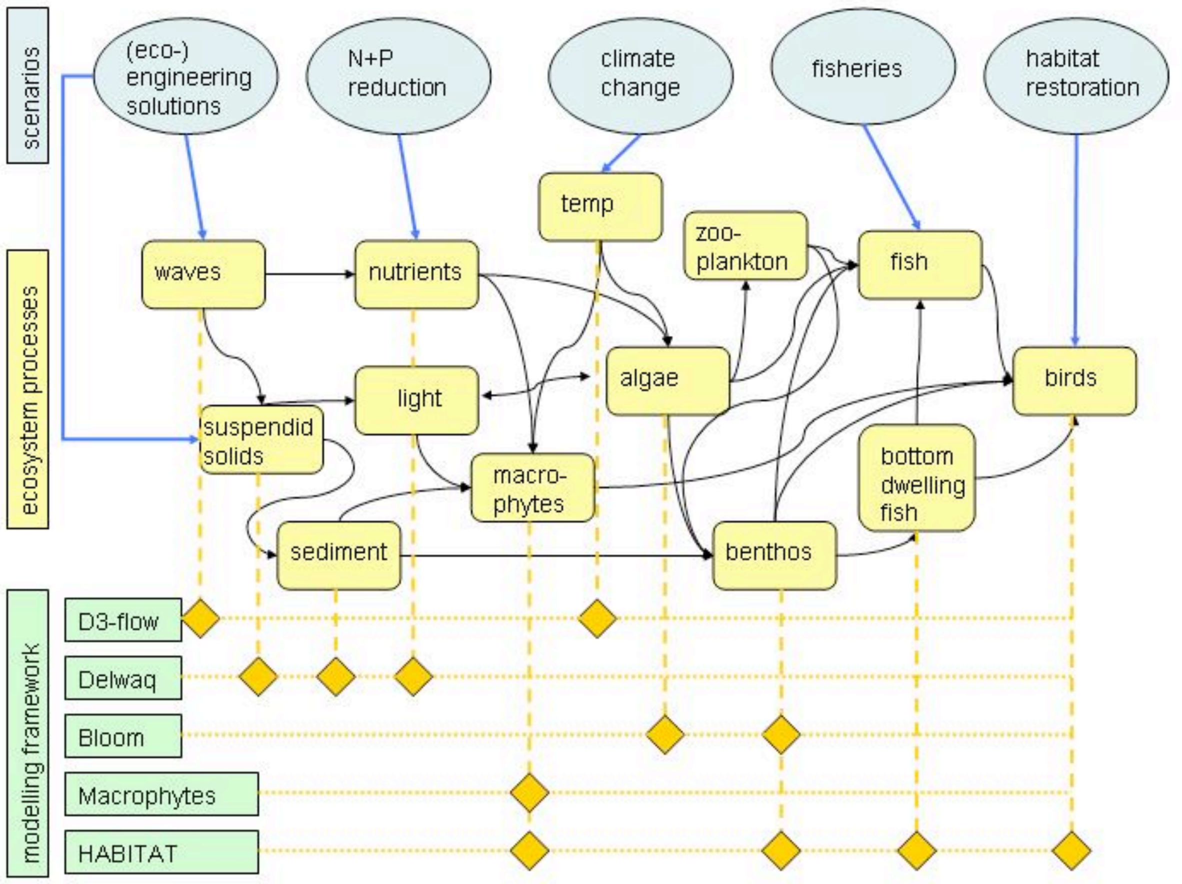

Figure 17: Modelling framework for lake ecosystems in Lake IJsselmeer area.

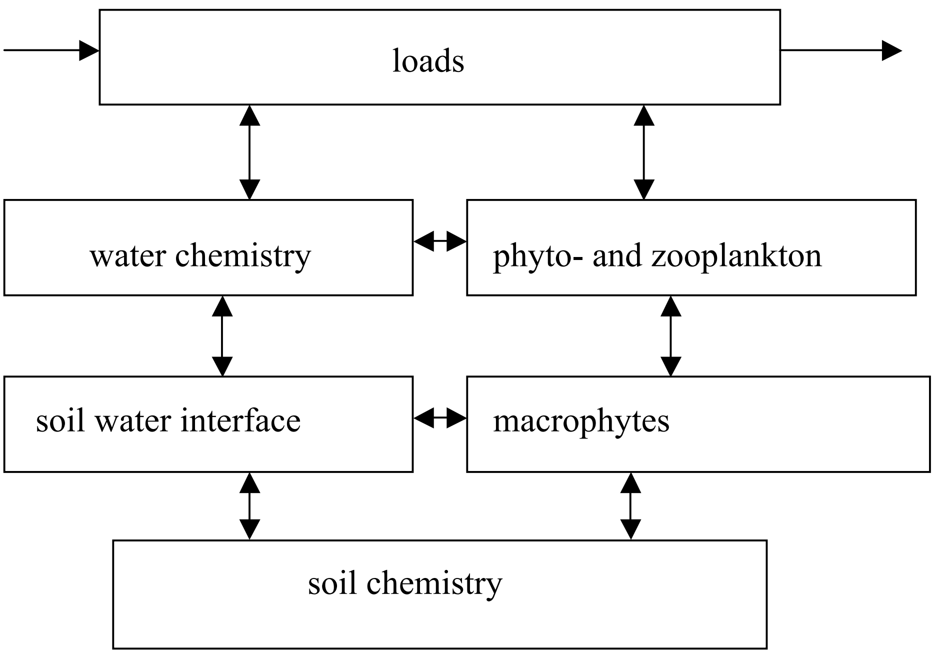

Figure 18*:* The different components of Delwaq-G.

DELWAQ-G simulates the chemical processes within in the water column, the sediment and the interaction of those processes between those compartments (see also Figure 4). The involved processes in the model are not only the routes between several compartments but also the processes such as re-aeration, nitrification and the adsorption of phosphate. For the interaction between water and sediment, the following physical processes are included: sedimentation, net sedimentation, resuspension and burial. For the simulation of all processes occurring in the sediment, the sediment layer module is used. This module simulates the exchange of organic material, nutrients and dissolved oxygen between the sediment and the water column. BLOOM uses linear programming as base for its calculation. Limiting factors being light, nutrients (N, P and Si), growth and dying maximum. BLOOM has been used in many applied studies to model algal biomass and composition. The model is extensively calibrated and validated and good results have been obtained for a large number of studies under both light limiting conditions and nutrient limiting conditions (Los and Brinkman, 1988; Los and Wijsman, 2006; Los et al., 2008; Blauw et al., 2009).

4. Lessons learnt from case studies

4.1. Introduction

Models are nowadays indispensable tools to gain a better understanding of the complexity of eutrophication processes and to guide water managers in making decisions. Specially, models may help to analyse and detect relationships between different (groups of) substances or species and environmental conditions, as well as to guide water managers to select the optimal strategy for managing a specific water body. In this chapter, we will review the model applications in the case studies according the criteria of "good modelling practice" (Van Waaveren et al., 1999), with a special attention to the interactions between modellers and end-users (in this case lake managers). Important here is to make a distinction between credibility and acceptability of model applications. Credibility is the technical appropriateness of the model whereas acceptability is the perception of the manager of the practical value of the modelling results. Analysis of the lake ecosystem and the ecological impact of proposed measures by using modelling tools is performed mostly by modellers, but model results are used by managers. Therefore, managers have to judge if model results and the uncertainties are acceptable to be used in decision making. Model credibility may contribute to the acceptability of a model or its results, but managers may decide that the model results are the best guess, irrespective of uncertainties or lack of significant discrimination between alternative courses of action (Van der Molen, 1999).

4.2. Good modelling practice

Recent guidelines on developing and applying models to support water managers in making adequate decisions to restore the chemical and ecological status of rivers and lakes and to promote sustainable use of the water resources, emphasise the importance of integrated approaches and collaboration between modellers and stakeholders (Van der Molen, 1999; Van Waaveren, 1999; Refsgaard et al, 2005, 2007). In general, the interactions between the modelling process and the water management process are clear in the beginning of the modelling process, where the modeller receives the specifications of objectives and requirements for the modelling study from the water management process, and towards to end of the modelling study, where the modelling results are provided as input to the water management process. According to Refsgaard et al. (2007), the modelling process consists of five major steps (Figure 19):

- STEP1 (model study plan). This step aims to agree on a Model Study Plan comprising answers to the questions: Why is modelling required for this particular model study? What is the overall modelling approach and which work should be carried out? Who will do the modelling work? Who should do the technical reviews? Which stakeholders/ public should be involved and to what degree? What are the resources available for the project? The water manager needs to describe the problem and its context as well as the available data. A very important (but often overlooked) task is then to analyse and determine what are the various requirements of the modelling study in terms of the expected accuracy of modelling results. The acceptable level of accuracy will vary from case to case and must be seen in a socio-economic context. It should, therefore, be defined through a dialogue between the modeller, water manager and stakeholders/public. In this respect an a priori analysis of the key sources of uncertainty is crucial in order to focus the study on the elements that produce most information of relevance to the problem at hand.

- STEP 2 (data and conceptualisation). In this step the modeller should gather all the relevant knowledge about the study basin and develop an overview of the processes and their interactions in order to conceptualise how the system should be modelled in sufficient detail to meet the requirements specified in the model study plan. Consideration must be given to the spatial and temporal detail required of a model, to the system dynamics, to the boundary conditions and to how the model parameters can be determined from available data. The need to model certain processes in alternative ways or to differing levels of detail in order to enable assessments of model structure uncertainty should be evaluated. The availability of existing computer codes that can address the model requirements should also be evaluated.
- STEP 3 (model set-up). Model set-up implies transforming the conceptual model into a site-specific model that can be run in the selected model code. A major task in model setup is the processing of data in order to prepare the input files necessary for executing the model. Usually, the model is run within a graphical user interface (GUI) where many tasks have been automated.
- STEP 4 (calibration and validation). This step is concerned with the process of analysing the model that was constructed during the previous step, first by calibrating the model, and then by validating its performance against independent field data. Finally, the reliability of model simulations for the intended type of application is assessed through uncertainty analyses. The results are described so that the scope of model use and its associated limitations are documented and made explicit.
- STEP 5 (simulation and evaluation). In this step, the modeller uses the calibrated and validated model to make simulations to meet the objectives and requirements of the model study. Depending on the objectives of the study, these simulations may result in specific results that can be used in subsequent decision making (e.g. for planning or design purposes) or to improve understanding (e.g. of the hydrological/ecological regime of the study area). It is important to carry out suitable uncertainty assessments of the model predictions in order to arrive at a robust decision. As with the other steps, the quality of the results needs to be assessed through internal and external reviews that also provide platforms for dialogues between water manager, modeller, reviewer and, often, stakeholders/public.

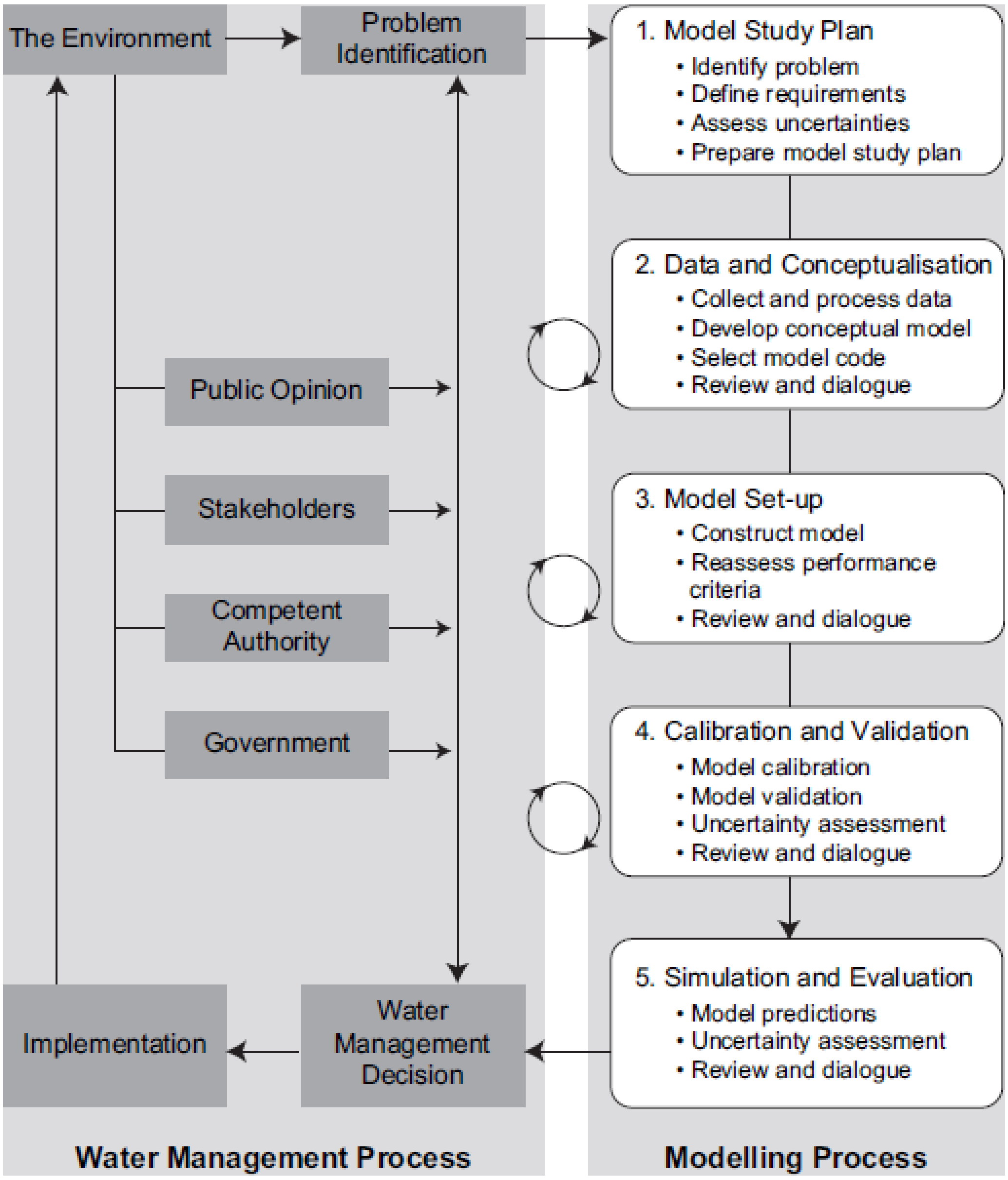

Figure 19: The interactions between the five steps of the modelling process and the water management process (Refsgaard et al., 2007).

4.3. Lake Pyhäjärvi

4.3.1 Feedback from stakeholders

The feedback from the stakeholders of Lake Pyhäjärvi about models and their usage were gathered in a workshop. It took place on 10'th of October 2011 at the Pyhäjärvi Institute, Eura Lake and catchment models applied to the region were presented to the participants and following questions were discussed:

1. Are you familiar with a model?
2. Have you used the results of model in the planning of program of measures?

   a. if yes – how
   b. if not – why
3. How to improve the usability of model results?

The full set of applied models was presented as in table x with remarks presented in Appendix 1. and Still the the discussion was focused on a set of models applied and developed from 2000 onwards. The were: WSFS-VEMALA SWAT, INCA, LakeLoadResponse (LLR), COHERENS, MyLake/PROTECH, and decision making models (influence diagrams).

LLR model discussed first was not recognized among the stakeholders. The model can be used to estimate target nutrient load reduction (TNLR) for the lake type specific good/moderate ecological class boundaries in terms of total phosphorus, total nitrogen and chlorophyll a. The model was not familiar and has not been approved by the experts of Pyhäjärvi Institute, Turku University and environment authorities due to the omission of food web impacts of fisheries. Also, decades long records of hydrology, water chemistry, food web and fisheries support dynamic approach and it is not used in LLR model. However, the approximation used in model was approved by the coordinator of agricultural water protection in a local environment center. Her opinion was based on the fact that in most lake cases the poor data situation prevails and this case study is a good test case with large amount of data. The dissemination of model results was required to make LLR model noted by its strengths, limitations and by the scope of management measures estimated using the model.

The catchment scale SWAT model which is designed to inform the selection of crop and site and of agricultural techniques and the timing of agricultural activities was neither known by stakeholders. The popularization of model based comparison of impacts of agricultural measures was required to gain approval from local peers and to promote the usage of SWAT model.

The INCA with similar scope but more coarse overall spatial resolution compared to SWAT model was recognized by the stakeholders. But here again model results were not used due to the lack of popular publication of model results.

COHERENS 3D hydrodynamic lake hydrodynamics and water quality model was well known by stakeholders. Latest climate and agricultural scenarios were not yet used. Internal loading and fishery's food web impacts were needed to put results fully into practice.

Fine-featured simulation results for the detailed food web model MyLake/PROTECH were recognized with satisfaction. A new version of the model is expected soon and the usage of the model has encouraging prospects.

The hydrological part of the Finnish Water Shed Forecasting System WSFS-VEMALA was best known of the models and climate and land use scenarios were widely used. The usage of local data and the fitting of model to it were required reduce uncertainties and achieve the full potential of the model. The fresh results of water quality part were used but more tests and experience is needed to approve the usage of results.

Decision making tools using Bayesnets and influence diagrams were not recognized earlier. However during the discussion the potential of these tools were realised and the testing in the selection and dimensioning of measures was put forward.

4.3.2. Usability of models in design and decision making

Usability of case study models in design and decision making is be evaluated according the principles of good modelling practice. The criteria (with cursive font) are listed below with the comments related to the case study.

The integration of models and simulations

The integration of models and simulations was practiced well within two models, namely WSFS-VEMALA-COHERENS and Bayesnet. The first one is comprehensive, gives lot of information but is a laborious approach and is also limited with spatial resolution. The results are produced by a specialist operated automatic system. Stakeholders are not able to direct them. Bayesnet is a very compact and easy to use also for model oriented stakeholders.

Communication of uncertainty

Communication of uncertainty is fully implemented to LakeState, LLR and Bayesnets. In WSFS-VEMALA it is included by cloud simulations.

Data demands

Data demands vary according to model complexity and required accuracy in planning and in decision making. In general, mechanistic models, especially those with water quality and hydrobiological compartments are much more data intensive than statistical models and bayesnets.

Public participation

Pyhäjärvi Institute has provided unique environment for public participation and regional environmental centre has supported the model development and data collection in many ways. However, due to some reason as indicated in stakeholder workshop model results above models were not familiar to them and they were not approved nor disseminated by them.

Communication with model users

All models have been developed and applied by SYKE or it's collaborators (for example Deltares). The main communication with local potential users has been on the data collection and selection of scenarios for simulations. The model results have been published mainly in scientific publications and the number of popular publications has been too low. In addition, the local usage of models has been small. Therefore, the communication with the river basin managers and stakeholders should be improved.

Delivery of knowledge rules

Model based knowledge rules can be useful for river basin management but they have not been systematically collected from model applications. In the future existing results should be carefully analyzed in this respect. In fact, there are numerous applications.

Usability in design and decision making

Most of the catchment management measures have been included in simulations. There as, fisheries management and internal loading need more attention. Uncertainties have been included as well but the communication to stakeholders should be improved.

Climate change scenarios

The main climate change scenario in Finland is A1B and it has been used also in this case study. WSFS-VEMALA has been used for simulating the effects of various global model simulation outputs to accuracy of hydrological predictions. The output was used in Coherens runs. WSFS-VEMALA output produced with different climate change change scenarios should be used systematically in the all models. In order to achieve sustainable mitigation under changing climate also the land use scenarios should be combined.

The climate scenarios for temperature and precipitation change used in the WSFS (Watershed Simulation and Forecasting System) and Coherens models (see below) 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 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).

Management and restoration scenarios

There has been fairly limited number of land use and social scenarios available for this region. The ongoing discussion is increasing the number and diversity of scenarios for wide and active model usage.

4.4. Lake Veluwe

The modelling framework for Lake Veluwe is a part of the national modelling framework for water management in the Netherlands. At this moment, this national modelling framework is updated to address the future challenges in water management in the Netherlands, including climate change and adaptation. Important step in this process are consultations with stakeholders and decision makers to identify the model requirements for a wide range of future applications. Within the context of the WISER project, there was no workshop held to discuss the model application for Lake Veluwe.

Integration of models and simulations

All the modelling tools that were applied to Lake Veluwe are part of the Sobek modelling suite and thus fully integrated.

Uncertainty analysis

Uncertainty analysis was not a part of this case study. The applied models have been used in many studies to predict algal biomass and composition. The modelling framework is extensively calibrated and validated and good results have been obtained for a large number of studies under both light limiting conditions and nutrient limiting conditions (Los, 2009). Calibration and validation were carried out for this case study as well.

Climate projections have a high level of uncertainty. In this case study, climate scenarios from the Royal Dutch Meteorological Institute (KNMI) were used. These scenarios are the starting point for developing a national strategy for climate adaptation and accepted by a wide range of stakeholders and decision makers in the Netherlands.

Data demands and availability

Large data sets on water quality and chlorophyll-a are available. As this model study is meant merely to explore the relative contribution of climate change and water management to water quality, some simplifications are made. Changes in solar radiation patterns, precipitation patterns, stream velocity (both climate and management induced), macrophyte growth and dynamic grazing were not taken into account. The effects of the measures carried out in Lake Veluwe by the National Water Board were assessed by altering the water and/or nutrient balance underlying the original long term calculation.

Public particaption

None, when referring to the WISER project. However, this case study is connected to a set of studies on the impact of climate change on lake ecosystems and possible lake management strategies to adapt lake ecosystems to climate change. Within this context, there are intensive interactions with stakeholders to discuss possible lake management strategies.

Delivery of knowledge rules

There was no systematic collection of model based knowledge rules from model applications.

Usability in design and decision making

Since the mid-80s, a set of measures have been implemented to restore the lake ecosystem. The model validation was taken into account this period and the model application was able to correctly represent the changes that occurred over time as a result of changes in management. Therefore we conclude that the modelling framework is suitable for analysis of restoration measures in shallow lakes.

Climate change scenarios and nutrient scenarios

Temperature runs were made with both the long-term and the short-term models. Temperature scenarios that were used were deduced from climate scenarios from the Royal Dutch Meteorological Institute (KNMI) and boiled down to four scenarios: 1. +0.9°C (G), 2. +1.3°C *(G+), 3. +1.8*°C *(W) and 4. +2.6*°C *(W+).*

The nutrient scenarios were only run with the short-term model. Nutrient scenarios were set for both increased and reduced concentrations of nitrogen and/or phosphorous relative to the reference run. Increased nutrient concentrations stands for increased eutrophication due to climate change and reduced nutrient concentrations for water management efforts. Increased nutrient concentrations were set on +10% nitrogen and/or +10% phosphorous and reduced nutrient concentrations on -10% nitrogen and/or -10% phosphorous

The short-term run was also used to explore to combined effect of temperature increase and change in nutrient concentrations. Six scenarios were simulated being: 1. +0.9°C and -10% nitrogen reduction, 2. +0.9°C and -10% phosphorus reduction, 3 +0.9°C and -10% nitrogen and -10% phosphorus reduction, 4. +2.6°C and -10% nitrogen reduction, 5. +2.6°C and -10% phosphorus reduction, 6 +2.6°C and -10% nitrogen and -10% phosphorus reduction,

We acknowledge that climate change has many more pathways through which it affects the aquatic ecosystem: changing hydrology, solar radiation, wind and precipitation patterns, more erratic weather and temperature fluctuations during the year and between years and the effect of all these on biogeochemical cycling are currently not included in the analysis. We merely demonstrated that deterministic models that are well calibrated and validated can help assessing the potential effects of such processes by taking only one example parameter representing climate change.

4.4. Key messages

Based on the feedback from stakeholders the following messages were identified:

- Close co-operation with end users in the formulation of management questions and the conceptual framework.
- Designing conceptual models together with stakeholders to have a common understanding and to create a clear picture about the issues that should be addressed
- Non scientific reporting of impacts of management measures directly to the end users
- More time and resources to collateral data collation and analysis together with end user
- Flexible selection of temporal and spatial scales
- Use of Bayesian networks throughout the modeling process in the analysis and dissemination of ecological and social impacts.
- The storing of input and output data into a generic database.

5. Guidance to apply models for analysing lake ecosystem responses to restoration and mitigation measures

Models that are describing the relationships between the biological quality elements and the physico-chemical conditions of surface waters are indispensable tools for water managers of rivers, lakes and coastal waters. Firstly, to gain a better understanding of the complexity of physico-chemical processes that have an impact ecological quality of surface waters. Secondly, to guide water managers in making decisions about the implementation of the Water Framework Directive and strategies to adapt to climate change. Lake ecosystem models vary from simple rules of thumb, to statistical relationships (regression models) and finally to more sophisticated descriptions of dynamic processes (deterministic models). As presented in this chapter 2 of this report, there is a wide range of modelling tools available to address the ecological processes in lakes.

Important criteria for a successful application of lake ecosystem models are related to credibility and acceptability (Van der Molen, 1999). The ideal situation is a credible model, that is accepted by decision makers based on its performance during the analysis of lake ecosystem processes, including the effects of changing climate, and assessment of effectiveness of restoration measures.

Important steps in assessing the credibility of model applications are (Van der Molen, 1999; Van Waaveren, 1999; Refsgaard et al., 2007):

- Objectives of the model applications are specified and the expected output of the models is in agreement with this objectives.
- Dimensions of the modelling system (components of the lake ecosystem: e.g. species groups, deep water, shallow water, littoral zones, riparian zones) and the temporal and spatial scales meet the objectives and the availability of data.
- Data availability is utilised sufficiently and the model application is not hampered by lack of input data and observations; uncertainties in data are addressed.
- Appropriateness of the model structure and parameter estimation (calibration) is examined through sensitivity analysis.
- Model validation is based on an independent set of observations / monitoring data and the results are quantified and related to the objectives.
- The uncertainties in the model structure, model parameters and model predictions are addressed and quantified.

Criteria for assessing the acceptability are (Van der Molen, 1999):

- The motivation for the initiation of a modelling project is known.
- Constraints in time and money for model development and application are specified.

- Arguments for approval of the model (results) are made explicit
- Consequences of the use of the model (results) are discussed.

From perspectives of scientists and modellers, there is a strong focus on credibility aspects of the model application, especially data availability, calibration, validation, sensitivity analysis and uncertainty analysis (Figure 20).

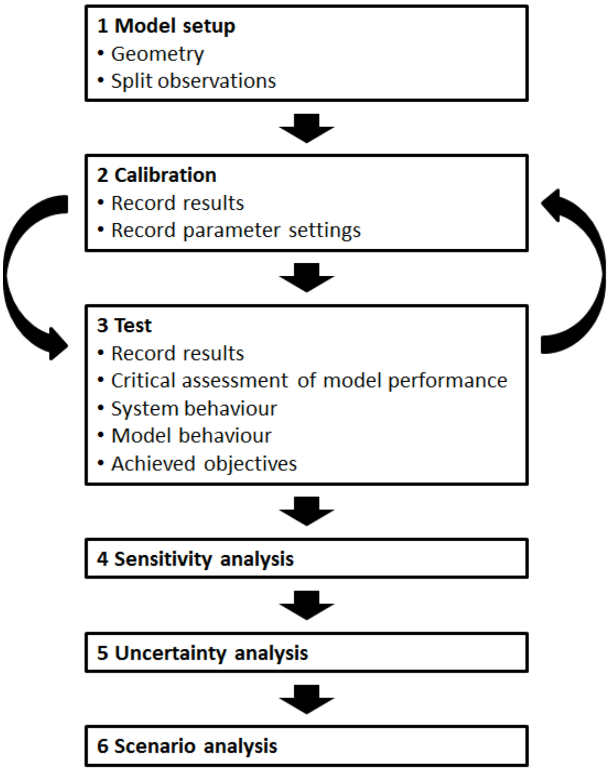

Figure 20: Steps for setting up a credible model application (Wade et al., 2008)

The interactions with stakeholders and decision makers are generally minimised to two moments in the modelling process, namely, in the beginning of the modelling process, where the modeller receives the specifications of objectives and requirements for the modelling study, and towards to end of the modelling study, where the modelling results are provided to stakeholders and decision makers. To support lake managers adequately, this process need to be improved and will not only improve the acceptability of the models applied, but also in a better set up of the models. Therefore, a close collaboration between the modellers and the stakeholders and decision makers right from the beginning of the modelling process is of great importance and starts with a detailed analysis of the lake management problems and the formulation of the objectives of the model application. Setting up a conceptual framework is a valuable approach to structure the discussion between modellers and stakeholder / decision makers, and will provide more clearly the information needed to achieve the objectives defined in the problem analysis. This is one of the lessons learned from the case studies. As a result of common conceptual model development, it will be apparent which ecological components and processes should be addressed, which model domains need to be taken into account, which variables should be included in the model, which processes and relationships between variables should be included, what spatial schematisation will represent the modelled systems properly, how the temporal scales need to be addressed, what kind of data are needed and what kind of interactions with the environment need to be considered (Figure 21).

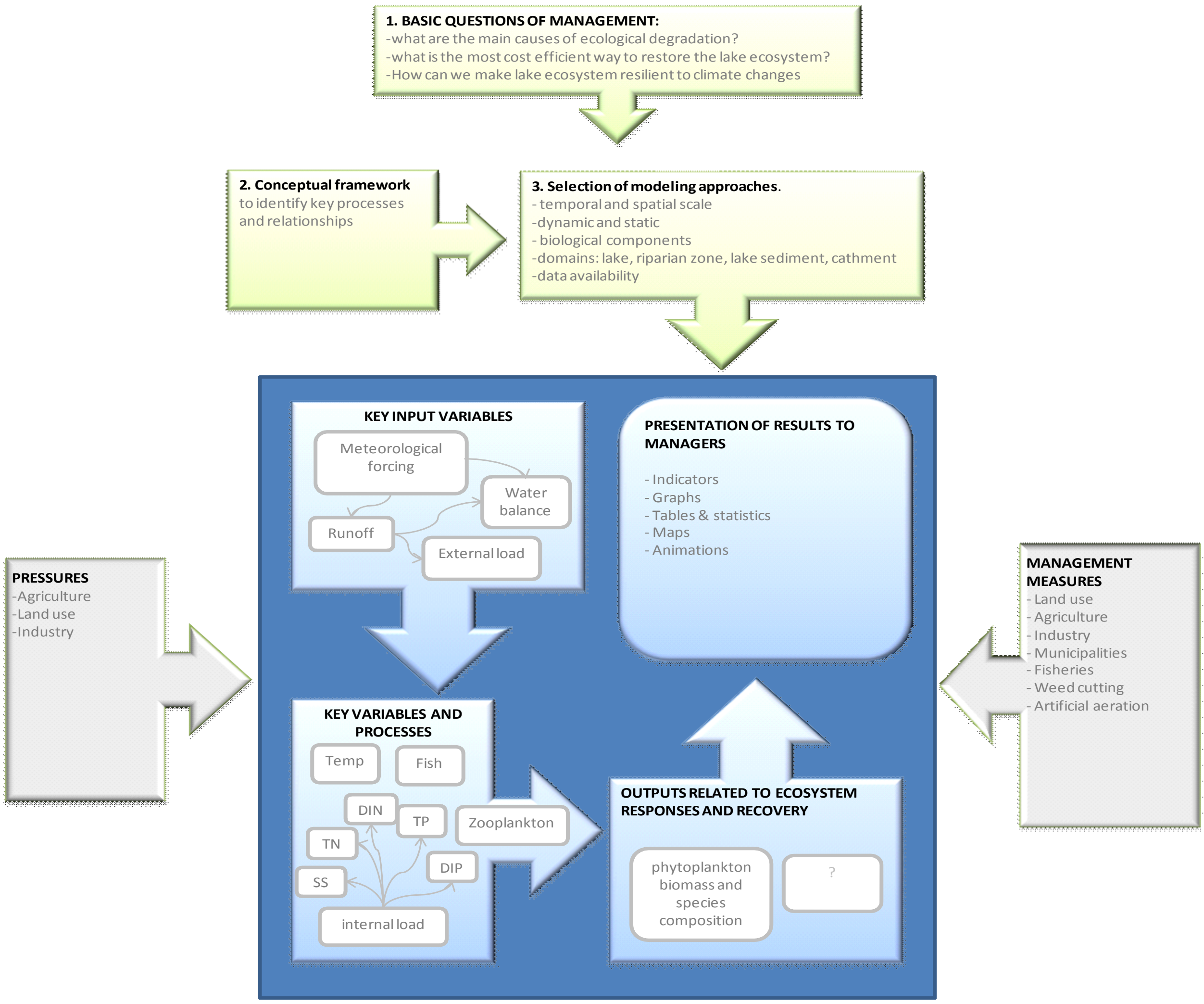

Figure 21: Framework for application of lake ecosystem models for designing program of restoration measures, taken into account the ecological impact of climate change and adaptation.

For a successful application of lake ecosystem models in the context of supporting lake managers in designing restoration and mitigation measures to improve the ecological status of lakes, the following 10 steps are commended:

Step 1: Close cooperation with end-users in the formulation of management questions that need to be addressed in the model application.

Step 2: Designing conceptual models together with end-users to have a common understanding and to create a clear picture about the issues that should be addressed.

Step 3: Identify together with the end-users the model requirements within the constraints of time, money and data-availability.

Step 4: Select together with the end-users the models that are needed to analyse and quantify the effectiveness of restoration measures.

Step 5: Set up your lake ecosystem modelling framework by translating the conceptual model into a set specific model, including the processing of data in order to prepare input files necessary for executing the model

Step 6: Calibrate and validate the lake ecosystem model according to the procedures of good modelling practice.

Step 7: Conduct a sensitivity analysis to identify the appropriateness of the model structure and parameter estimation

Step 8: Quantify the (magnitude of) uncertainties in the model structure, model parameters and model predictions.

Step 9: Discuss the results of the validation and sensitivity and uncertainty analysis with the end-users.

Step 10: Conduct the simulation and evaluation activities, including non scientific reporting of the impacts of management measures to the end-users.

References

Adriaenssens, V., Goethals, P.L.M. and De Pauw, N., 2006. Fuzzy knowledge-based models for prediction of macroinvertebrate taxa in watercourses in Flanders, Belgium. Ecological Modelling 195 (1/2): 3-10.

Arnold, J. G., R. Srinivasan, R. S. Muttiah, and J. R. Williams. 1998. Large□area hydrologic modeling and assessment: Part I Model development. J. American Water Resour. Assoc. 34(1):73-89.

Baran, E. & Jantunen, T.A., 2003. BayFish – Tonle Sap: a Bayesian model of the fish production in Tonle Sap Great Lake, Cambodja.

Baran, E., Jantunen, T.A. & Cheng, P., 2006. Developing a consultative Bayesian model for the integrated assessment of aquatic resources: an inland coastal zone case study. In: Hoanh, C.T., Gowing, T.P., Hardy, B. (eds.): Environment and livelihoods in tropical coastal zones. Managing fishery-agriculture conflicts. CAB International, Wallingford UK, p. 206-218.

Baran, E., Jantunen, T.A., Cheng, P. & Hakalin, M., 2007. BayFish – Tonle Sap: a model of the Tonle Sap fish resource. World Fish Centre, Pnom Penh, Cambodja.

Baumert, H.Z., Benndorf, J., Bigalke, K., Goldmann, D., Nöhren, I., Petzoldt,Y., Post, J. & Rolinski, S., 2005. Das hydrophysikalisch ökologische Talsperren- und Seenmodel SALMO-HR Modelldokumentation und Leitfaden für den Anwender. Dresden,

Benndorf, J. & Recknagel, F., 1982. Problems of application of the ecological model SALMO to lakes and reservoirs having various trophic states. Ecological Modelling 17:129-145

Beklioglu, M., Meerhoff, M., Søndergaard, M. & Jeppesen, E., 2011. Eutrophication and restoration of shallow lakes from a cold temperature to a warm Mediterranean and (sub)Tropical Climate. In: Eutrophication: causes, consequences and control. Springer. p. 91-108.

Berger, C. & Wells, S., 2008. Modeling the Effects 1460 of Macrophytes on Hydrodynamics. Journal of Environmental Engineering 134:778-788.

Bärlund, I., Kirkkala, T., Malve, O. and Kämäri, J. 2007. Assessing SWAT model performance in the evaluation of management actions for the implementation of the Water Framework Directive in a Finnish catchment Environmental Modelling & Software, 22 (5) (2007), pp. 719–724.

Blauw, A. N., Los, F.J., Bokhorst, M., & Erftemeijer, P. L. A. (2009). GEM: a generic ecological model for estuaries and coastal waters. *Hydrobiologia*. doi:DOI 10.1007/s10750-008-9575-x

Burger, D.F., Hamilton, D.P., Pilditch, C.A. & Gibs, M.M., 2007. Benthic nutrient fluxes in a eutrophic, polymictic lake. Hydrobiologia 584: 13-25.

Burger, D.F., Hamilton, D.P. & Pilditch, C.A., 2008. Modelling the relative importance of internal and external nutrient loads on water column nutrient concentrations and phytoplankton biomass in a shallow polymictic lake. Ecological modelling 211:411-423.

Chapra SC 1975. Comment on ‘An empirical method of estimating the retention of phosphorus in lakes’ by WB Kirchner and PJ Dillon. Water Resour Res 11:1033–1034.

Chen, Q. & Mynett, A.E., 2003. Integration of data mining techniques and heuristic knowledge in fuzzy logic modelling of eutrophication in Taihu Lake. Ecological Molelling 162 (1/2): 55-67.

Christensen, V. & Walters, C.J., 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling 172: 109-139.

Dedecker, A.P., Goethals, P.L.M. & De Pauw, N., 2002. Comparison of artificial neural network (ANN) model development methods for prediction of macroinvertebrate communities in the Zwalm river basin in Flanders, Belgium. Scientific World Journal 2: 96-104.

Duel, H., Van der Lee, G.E.M., Penning W.E. & Baptist, M.J., 2003. Habitat modelling of rivers and lakes in the Netherlands: an ecosystem approach. Canadian Water Research Journal 28 (2): 231-248.

Duel, H., Los, F.J., Rekolainen, S., Kaas, H., Heiskanen, A-S., Carstensen, J., Lyche-Solheim, A., Boorman, D., Dunbar, M.J., Friberg, N. & Krause-Jensen, D., 2005. Relationships between ecological status of surface waters and both chemical and hydromorphological pressures. In: Lawson (ed.): River basin management: progress towards implementation of the European Water Framework Directive. Taylor&Francis/Balkema, p. 269-280.

Duel, H., Groot, S. & Boorman, D., 2007. Relationships between biological qulaity elements and physico-chemical pressures: validation and practual use of relationships. D18 report. REBECCA project.

Eckhardt, K., S. Haverkamp, N. Fohrer, and H. G. Frede. 2002. SWAT□G, a version of SWAT99.2 modified for application to low mountain range catchments. Phys. Chem. Earth 27(9:10):641-644.

Ekholm, P., Malve, O. and Kirkkala, T, 1997. Internal and external loading as regulators of nutrient concentrations in the agriculturally loaded lake Pyhäjärvi (southwest Finland). Hydrobiology 1:3-14.

Elliott, J.A., Irish, A.E. & Reynolds, C.S., 2010. Modelling phytoplankton dynamics in fresh waters: affirmation of the PROTECH approach to simulation. Freshwater Reviews 3:75-96.

Elser, J. J., Marzolf, E.R. & Goldman, C.R., 1990. Phosphorus and Nitrogen Limitation of Phytoplankton Growth in the Freshwaters of North America: A Review and Critique of Experimental Enrichments. Canadian Journal of Fisheries and Aquatic Sciences 47: 1468-1477.

Goethals, P.L.M., 2005. Data driven development of predictive ecological models for benthis macroinvertebrates in rivers. Thesis. Ghent University.

Gulati, R.D. & Van Donk, E., 2002. Lakes in the Netherlands, their origin, eutrophication and restoration: state-of-the-art review. Hydrobiologia 478:73-106

Gulati, R.D., Dionisio Pires, L.M., & Van Donk, E., 2008. Lake restoration studies: failures, bottlenecks and prospects for new ecotechnological measures. Limnologica 38: 233-247.

Hagan, M.T., Demuth, H.B. & Baele, M., 1996. Neural networks design. PWS Publishing Company. Boston, USA.

Håkanson, L. & Boulion, V., 2002. The Lake Foodweb - modelling predation and abiotic/biotic interactions. Backhuys Publishers, Leiden. 344 pp.

Hendriks, A.J., 1995. Concentrations of microcontaminants and response of organisms in laboratory experiments and Rhine-Delta field surveys. Thesis. Utrecht: Utrecht State University.

Hipsey, M.R., Romero, J.R., Antenucci, J.P. & Imberger, J., 2007. A computational aquatic ecosystem dynamics model (CAEDYM): a versatile water quality model for coupling with hydrodynamic drivers. Proceedings of 7th Internationaal Conference on Hydroinformatics, Nice, France. Volume 1: 526-533.

Ibelings, B.W., Vonk, M., Los, F.J., Van der Molen, D.T. & Mooij, W.M., 2003. Fuzzy modelling of cyanobacterial surface water blooms: validation with NOAA-AVHRR Satellite images. Ecological Applications 13: 1456-1472.

Ibelings, B. W., Portielje, R., Lammens, E. H. R. R., Noordhuis, R., Berg, M. S., Joosse, W., & Meijer, M. L., 2007. Resilience of Alternative Stable States during the Recovery of Shallow Lakes from

Eutrophication: Lake Veluwe as a Case Study. Ecosystems 10(1): 4-16. doi:10.1007/s10021-006-9009-4

Janse, J.H., 2005. Model studies on the eutrophication of shallow lakes and ditches. Wageningen University, Wageningen

Janse, J.H., De Senerpont Domis, L.N., Scheffer, M., Lijklema, L., Van Liere, L., Klinge, M. & Mooij, W.M., 2008. Critical phosphorous loading of different types of shallow lakes and the consequences for management estimated with the ecosystem model PCLake. Limnologica 38:203-219.

Janse, J.H., Scheffer, M., Lijklema, L., Van Liere, L., Sloot, J.S. & Mooij, W.M., 2010. Estimating the critical phosphorus loading of shallow lakes with the ecosystem model PCLake: Sensitivity, calibration and uncertainty. Ecological Modelling 221:654-665.

Janssen, A.B.G., 2012. Internactions between Driessena polymorpha and food quality and quantity in Lake IJsselmeer. Report. Wageningen University & Deltares.

Jeppesen, E., Jensen, J.P., Søndergaard, M., Lauridsen, T.L., Pedersen, L.J. & Jensen, L., 1997. Top down control in freshwater lakes: the role of nutrient state, submerged macrophytes and water depth. Hydrobiologia 342/343:151-164

Jeppesen, E., Søndergaard, M. & Christoffersen, K., eds., 1998. The structuring role of submerged macrophytes in lakes. Springer Verlag

Jeppesen, E., Jensen, J.P., Sondergaard, M., Lauridsen, T. & Landkildehus, F., 2000. Trophic structure, species richness and biodiversity in Danish lakes: Changes along a phosphorus gradient. Freshwater Biology 45:201-218

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K., Anneville, O., Carvalho, L., Coveney, M.F., Deneke, R., Dokulil, M., Foy, B., Gerdeaux, D., Hampton, S.E., Kangur, K., Köhler, J., Körner, S., Lammens, E.H.R.R., Lauridsen, T.L., Manca, M., Miracle, R., Moss, B., Nõges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C.L., Straile, D., Tatrai, I., Willén, E. & Winder, M., 2005. Lake responses to reduced nutrient loading- an analysis of contemporary long-term data from 35 case studies. Freshwater Biology 50:1747-1771

Jeppesen, E., Meerhoff, M., Jacobsen, B. a., Hansen, R. S., Søndergaard, M., Jensen, J. P., Lauridsen, T. L., et al., 2007. Restoration of shallow lakes by nutrient control and biomanipulation—the successful strategy varies with lake size and climate. Hydrobiologia 581(1): 269-285. doi:10.1007/s10750-006-0507-3

Jeppesen, E., Kronvang, B., Meerhoff, M., Søndergaardm, M., Hansen, K.M., Andersen, H.E., Lauridsen, T.L., Beklioglu, M., Ozen, A. & Olesen, J.E., 2009. Climate change effects on runoff, catchment phosphorus loading and lake ecological state, and potential adaptations. Journal of Environmental Quality 38:1930-1941

Jorde, K., Schneider, M. & Zoellner, F., 2000. Analysis of instream habitat quality – preference functions and fuzzy models. In: Wang and Hu (eds.): *Stochastic Hydraulics*: 671-680. Rotterdam: Balkema.

Jørgensen, S.E., 2010. A review of recent developments in lake modelling. Ecological Modelling 221:689-692

Kankaala, P., Huotari, J., Peltomaa, E., Saloranta, T. & Ojala, A., 2006. Methanotrophic activity in relation to methane efflux and total heterotrophic bacterial production in a stratified, humic, boreal lake. Limnology and Oceanography 51:1195-1204

Koelmans, A.A., Van der Heijde, A., Knijff, L.M. & Aalderink, R.H., 2001. Integrated modelling of eutrophication and organic contaminant fate and effects in aquatic ecosystems. A review. *Water Research 15*: 3517 – 3536.

Kooijman, S.A.L.M., 2000. Dynamic energy and mass budgets in biological systems (2nd edition). Cambridge University Press.

Koskiaho, J. , Ekholm, P., Räty, M., Riihimäki, J. and Puustinen, M. 2003. Retaining agricultural nutrients in constructed wetlands—experiences under boreal conditions Ecological Engineering, 20 (1) (2003), pp. 89–103.

Kosten, S., Huszar, V.L.M., Mazzeo, N., Scheffer, M., Sterberg, L.S.L. & Jeppesen, E., 2009. Lake and watershed characteristics rather than climate influence nutrient limitation in shallow lakes. Ecological Applications 19: 1791-1804.

Krysanova, V., Wechsung, F., Becker, A., Poschenrieder, W. and Graefe, J. 1999. Mesoscale ecohydrological modelling to analyse regional effects of climate change. Environmental Modelling and Assessment, 4 (4) (1999), pp. 259–271.

Lammens, E. H. R. R., van Nes, E. H., Meijer, M.-L., & van den Berg, M. S. (2004). Effects of commercial fishery on the bream population and the expansion of Chara aspera in Lake Veluwe. Ecological Modelling177(3-4): 233-244. doi:10.1016/j.ecolmodel.2003.10.002

Lauzon, N. & Lence, B.J., 2007. Hybrid fuzzy-mechanistic models for addressing paramter variability. Environmental Modelling and Software 23 (5): 535-548.

Lek, S. & Guégan, J.F., 1999. Artificial neural networks as a tool in ecological modelling, an introduction. *Ecological Modelling*, 120, 65-73.

Leon, L.F., Smith, R.E.H., Hipsey, M.R., Bocaniov, S.A., Higgens, S.N., Hecky, R.E., Antenucci, J.P., Guilford, S.J. & Imberger, J., 2011. Application of 3D hydrodynamic – biological model for seasonal and spatial dynamics of water quality and phytoplankton in Lake Erie. Journal of Great lakes Research 37 (1): 41-53.

Lepistö, A., Huttula, T., Bärlund, I., Granlund, K., Härmä, P., Kallio, K. Y., Kiirikki, M., Kirkkala, T., Koponen, S., Koskiaho, J., Kotamäki, N., Lindfors, A., Malve, O., Pyhälahti, T., Tattari, S. and Törmä, M. 2008. New measurement technology, modelling, and remote sensing in the Säkylän Pyhäjärvi area - CatchLake. Helsinki: Finnish Environment Institute, 76 p.

Liukko, N., Rasmus, K., Huttula, T., Lepistö, A., Tattari, S., Koskiaho, J. & Granlund, K., 2009. The Coherens 3D water quality model: a possible tool for the planning of lake management actions. Cost 869 Workshop, Keszthely, Hungary.

Los, F.J., & Brinkman, J.J., 1988. Phytoplankton modelling by means of optimizing: A 10-year experience with BLOOM II. *Verh. Internat. Verein. Limnol.*, *23*, 790-795.

Los, F.J. & Wijsman, J.W.M., 2006. Application of a validated primary production model (BLOOM) as a screening tool for marine, coastal and transitional waters. *Journal of Marine Systems, 64, 201-215.*

Los, F.J., Villars, M.T & Van der Tol, M.W.M., 2008. A 3-dimensional primary production model (BLOOM/GEM) and its applications to the (southern) North Sea (coupled physical-chemical-ecological model). *Journal of Marine Systems,* 74, 259-294/doi:101016/j.jmarsys.2008.01.002.

Los, F.J. 2009. Eco-hydrodynamic modelling of primary production in coastal waters and lakes using BLOOM. Wageningen University, Wageningen.

Luyten, P. J., Jones, J. E., Proctor, R., Tabor, A., Tett, P. & Wild-Allen, K. 1999. COHERENS - A coupled hydrodynamical-ecological model for regional and shelf seas: user documentation. MUMM report. Management Unit of the Mathematical Models of the North Sea, 911 pp.

MacKay, M.D., Neale, P.J., Arp, C.D., De Senerpont Domis, L.N., Fang, X., Gal, G., Jöhnk, K., Kirillin, G., Lenters, J.D., Litchman, E., MacIntyre, S., Marsh, P., Melack, J., Mooij, W.M., Peeters, F., Quesada, A., Schladow, S.G., Schmid, M., Spence, C., Stefan, H.G. & Stokes, S.L., 2009. Modelling lakes and reservoirs in the climate system. Limnology and Oceanography 54:2315-2329

Malve, O. 2007. Water quality prediction for river basin management. Doctoral dissertation. Helsinki University of Technology.Espoo, Finland. TKK-DISS 2292. ISBN 978-951-22-87499. URL:http://lib.tkk.fi/Diss/2007/isbn9789512287505/

Meerhoff, M., Clemente, J.M., de Mello, F.T., Iglesias, C., Pedersen, A.R. and Jeppesen, E., 2007. Can warm climate-related structure of littoral predator assemblies weaken the clear water state in shallow lakes? Global Change Biology 13: 1888–1897.

Meijer, M.L., De Boois, I., Scheffer, M., Portielje, R. & Hosper, H., 1999. Biomanipulation in shallow lakes in the Netherlands: an evaluation of 18 case studies. Hydrobiologia 408: 13-30.

Mooij, W.M., Hülsmann, S., De Senerpont Domis, L.N., Nolet, B.A., Bodelier, P.L.E., Boers, P.C.M., Dionisio Pires, L.M., Gons, H.J., Ibelings, B.W. & Noordhuis, R., 2005. The impact of climate change on lakes in the Netherlands: a review. Aquatic Ecology 39:381-400

Mooij, W.M., Trolle, D., Jeppesen, E., Arhonditsis, G., Belolipetsky, P.V., Chitamwebwa, D.B.R., Degermendzhy, A.G., DeAngelis, D.L., De Senerpont Domis, L.N., Downing, A.S., Elliott, J.A., Fragoso, Jr. C.R., Gaedke, U., Genova, S.N., Gulati, R.D., Håkanson, L., Hamilton, D.P., Hipsey, M.R., 't Hoen, J., Hülsmann, S., Los, F.J., Makler-Pick, V., Petzoldt, T., Prokopkin, I.G., Rinke, K., Schep, S.A., Tominaga, K., Van Dam, A.A., Van Nes, E.H., Wells, S.A. & Janse, J.H., 2010. Challenges and opportunities for integrating lake ecosystem modelling approaches. Aquatic Ecology: DOI:10.1007/s10452-010-9339-3.

Neitsch, S. L., J. G. Arnold, J. R. Kiniry, and J. R. Williams. 2005. Soil and Water Assessment Tool (SWAT), Theoretical Documentation. Temple, Tex.: USDA□ARS Grassland Soil and Water Research Laboratory.

Noordhuis, R., Van der Molen, D.T. & M. van den Berg, 2002. Response of herbivorous water-birds to the return of Chara in Lake Veluwemeer, the Netherlands. Aquatic Botany 72 (3-4): 349-367.

Paquin, P.R., Gorsuch, J.W., Apte, S., Batley, G.E., Bowles, K.C., Campbell, P.G., Delos, C.G., Di Toro, D.M., Dwyer, R.L., Galvez, F., Gensemer, R.W., Goss, G.G., Hostrand, C., Janssen, C.R., McGeer, J.C., Naddy, R.B., Playle, R.C., Santoro, R.C., Schneider, U., Stubblefield, W.A., Wood, C.M. & Wu, K.B., 2002. The biotic ligand model: a historic overview. Comparative Biochemistry and Physiology Part C 133 (1/2): 3-35.

Penning, W.E., 2012. Ecohydraulics in large shallow lakes: implications for management. Thesis. Technical University Delft.

Phillips et al, 2010. Lake phytoplankton composition metrics. WISER D3.1-1.

Posthuma, L., Sutter, G.W. & Traas, T.P., 2002. *Species sensitivity distributions in ecotoxicology*. Lewis Publishers.

Recknagel, F., Cao, H., Kim, B., Takamura, N. & Welk, A.,, 2006. Unravelling and forecasting algal population dynamics in two lakes different in morphometry and eutrophication by neural and evolutionary computation. Ecological Informatics 1:133-151

Refsgaard, J.C., Henriksen, H.J., Harrar, W.G., Scholten, H. & Kassahun, A., 2005. Quality assurance in model based water management – Review of existing practice and outline of new approaches. Environmental Modelling & Software, 20, 1201-1215.

Refsgaard, J.C., Van der Sluijs, Højberg, A.L., Vanrolleghem, P.A. 2007. Uncertainty in the environmental modelling process – A framework and guidance. Environmental Modelling & Software, 22, 1543-1556.

Regan, H.M., 2002. A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecological Applications, 12, 618-628.

Reichert, P. & Mieleitner, J., 2008. Lake Models. In: Jørgensen, S.E. & Fath, B.D. (eds.), Ecological Models. Vol. 3 of Encyclopedia of Ecology, 2068-2080. Oxford: Elsevier

Reynolds, C.S., Irish, A.E. & Elliott, J.A., 2001. The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). Ecological Modelling 140:271-291

Rip, W. J., Ouboter, M., & Los, F. (2007). Impact of climatic fluctuations on Characeae biomass in a shallow, restored lake in The Netherlands. Hydrobiologia 584: 415-424.

Saito, L., Johnson, B.M., Bartholow, J. & Hanna, R.B., 2001. Assessing Ecosystem Effects of Reservoir Operations Using Food Web-Energy Transfer and Water Quality Models. Ecosystems 4:105-125

Saloranta, T.M. & Andersen, T., 2007. MyLake—A multi-year lake simulation model code suitable for uncertainty and sensitivity analysis simulations. Ecological Modelling 207:45-60

Sarvala, J., Helminen, H. and Auvinen, H. 1998. Portrait of a flourishing freshwater fishery: Pyhäjärvi, a lake in SW-Finland. Boreal Environment Research 3: 329–345.

Sarvala, J., Helminen, H. and Karjalainen, J. 2000. Restoration of Finnish lakes using fish removal: changes in the chlorophyll-phosphorus relationship indicate multiple controlling mechanisms. Verhandlungen Internationale Vereinigung für Limnologie, 27 (2000), pp. 1473–1479

Scheffer, M. ,1990. Multiplicity of stable states in fresh water systems. Hydrobiologia 200:475-486

Scheffer, M., Rinaldi, S., Kuznetsov, Y.A. & Van Nes, E.H., 1997. Seasonal dynamics of Daphnia and algae explained as a periodically forced predator-prey system. Oikos 80: 519-532.

Scheffer, M., 1998. Ecology of shallow lakes. Chapman & Hall, London. 357 pp.

Scheffer, M., Hosper, S.H., Meijer, M.L., Moss, B., Jeppesen, E., 1993. Alternative equilibria in shallow lakes. Trends in Ecology and Evolution 8:275-279

Scheffer, M., van den Berg, M. S., Breukelaar, A., Breukers, C., Coops, H., & Meijer, M. L., 1994. Vegetated areas with clear water in turbid shallow lakes. Aquatic Botany 49: 193-196.

Scheffer, M. & Van Nes, E.H., 2007. Shallow lakes theory revisited: various alternative regimes driven by climate, nutrients, depth and lake size. Hydrobiologia 584: 455-466

Søndergaard, M., Jensen, J.P. & Jeppesen, E., 2003. Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia 506-509: 135-145.

Søndergaard, M., Jeppesen, E., Laurdsen, T.L., Skov, C., Van Nes, E.H., Roijackers, R., Lammens, E. & Portielje, R., 2007. Lake restoration: successes, failures and long-term effects. Journal of Applied Ecology 44: 1095-1105.

Sullivan, B.A., Jager, H.I. & Myers, R., 2003. Modeling white sturgeon movement in a reservoir: the effect of water quality and sturgeon density. Ecological Modelling 167:97-114.

Troost, T., Wijsman, J., Saraiva, S. & Freitas, V., 2010. Modelling shellfish growth with dynamic energy budget models: an application for cockles and mussels in the Oosterschelde (Southwest Netherlands). Philosophical Transactions of the Royal Society: Biological Sciences 365: 3567-3577.

Van de Guchte, C., 1992. The sediment quality TRIAD: an integrated approach to assess contaminated sediments. In: River water quality. *Ecological assessment and control*: 417-431. Brussels: Commission of the European Communities.

Van den Berg, M. S., Coops, H., Meijer, M. L., Scheffer, M., & Simons, J., 1998. Clear water associated with a dense Chara vegetation in the shallow and turbid Lake Veluwe, The Netherlands. In E. Jeppesen, M. Sondergaard, & K. Christoffersen (Eds.), The structuring role of macrophytes. Springer-Verlag, New York, p. 339-352.

Van der Molen, D.T., Portielje, R., Boers, P. C. M., & Lijklema, L., 1998. Changes in sediment phosphorus as a result of eutrophication and oligotrophication in Lake Veluwe, the Netherlands. Water Research 32(11): 3281-3288.

Van der Molen, D.T., 1999. The role of eutrophication models in water management. Thesis. Wageningen University, the Netherlands.

vanGriensven, A. and Meixner, T., 2003. Sensitivity, optimisation and uncertainty analysis for the model parameters of SWAT. In: SWAT2003: 2nd International SWATConference, Bari, Italy, 1–4 July.

Van Nes, E.H., Lammens, E.H.R.R. & Scheffer, M., 2002. PISCATOR, an individual-based model to analyze the dynamics of lake fish communities. Ecological Modelling152: 261-278.

Van Nes, E.H., Scheffer, M., Van den Berg, M.S. & Coops, H., 2003. Charisma: a spatial explicit simulation model of submerged macrophytes. Ecological Modelling 159: 103-116.

Van Nes, E.H., Noordhuis, R., Lammens, E.H.H.R., Portielje, R., Reeze, B. & Peeters, E.T.M., 2008. Modelling the effects of diving ducks on zebra mussels Dreissena polymorpha in lakes. Ecological Modelling 211 (3-4): 481-490.

Van Waaveren, H. et al., 1999. Smooth modelling in water management. Good modelling practice handbook. Stowa report 99-05. Dutch Department of Public Works, Institute for Inland Water Management and Wastewater Treatment. Report 99.036. The Netherlands.

Vehviläinen, B. 1994. The watershed simulation and forecasting system in the National Board of Water and the Environment. Publications of Water and Environment Research Institute, 17 (1994), pp. 3–16.

Vehviläinen, B. and Huttunen, M. (2002) The Finnish watershed simulation and forecasting system (WSFS). XXI Conference of Danube Countries on the Hydrological Forecasting and Hydrological Bases of Water Management, Romania, September 2-6.

Ventelä, A.-M., Tarvainen, M., Helminen, H. and Sarvala, J. 2007. Long-term management of Pyhäjärvi (southwest Finland): eutrophication, restoration – recovery? Lake and Reservoir Management 23: 428–438.

Verschoor, A.J. Vink, J.P.M., De Snoo, G.R. & Vijver, M.G., 2011. Spatial and temporal variation of water type specific no-effect concentrations and risks of Cu, Ni and Zn. Environmental Science & Technology 45(14): 6049-6056.

Vink, J.P.M. & Verschoor, A.J., 2010. Biotic Ligand Models: availability, performance and applicability for water quality assessment. Deltares report.

Wade, A.J., Whitehead, P.G. and O'Shea L.C.M.2002.The prediction and management of aquatic nitrogen pollution across Europe: an introduction to the Integrated Nitrogen in European Catchments project (INCA). Hydrol Earth Syst Sci, 6 (2002), pp. 299–313.

Wade, A.J., Jackson, B.M. & Butterfield, D., 2008. Over-parameterised, uncertain 'mathematical marionettes' – how can we best use catchment water quality models? An example of an 80 year catchment-scale nutrient balance. Science of the Total Environment 400 (1-3): 52-74. doi: 10.1016/j.scitotenv.2008.04.030.

Wentzel, R.G., 1992. Clean water: a fading resource. Hydrobiologia 243/244: 273-281.

Wijsman, J.S.M., Brummelhuis, E.B.M. & Smaal, A.C., 2009. DEB model for cockles (Cerastoderma edule) in the Oosterschelde. IMARES Report. Yerseke.

Whitehead, P.G. , Wilson, E.J. and Butterfield, D. 1998. A semi-distributed Integrated Nitrogen Model for multiple source assessment in Catchments (INCA): Part I — model structure and process equations. Sci Total Environ, 210/211 (1998), pp. 547–558.

Xu, F-L., Tao, S., Dawson, R.W., Li. P. & Cao, J., 2001. Lake ecosystem health and assessment: indicators and methods. Water Research 35(13): 3157-3167.

Zhang, Y.L., Qin, B.Q., Zhu G.W., Gao, G., Lui L.C. & Chen, W.M., 2006. Effects of sediment resuspension on underwater light field in shallow lakes in the middle and lower reaches of the Yangtze River: a case study in Longgan Lake and Taihu Lake. Science in China; Series D Earth Sciences 49(1): 114-125.

Zhang, J., Gurkan, Z. & Jørgensen, S.E., 2010. Application of eco-exergy for assessment of ecosystem health and development of structurally dynamic models. Ecological Modelling 221: 693-702.

Zhao, J., Ramin, M., Cheng, V. & Arhonditsis, G.B., 2008. Competition patterns among phytoplankton functional groups: How useful are the complex mathematical models? Acta Oecologica 33(3): 324-344.