EcoSys Bd. 8, 2000, 93-137

Models for wetland planning, design and management

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1 Introduction

The societal respect for wetlands has recently changed from former view of vast and wet lands, which have to be drained to fulfil economic purposes, to the now holistic view which regards wetlands as important multifunctional ecotones between terrestrial and aquatic ecosystems providing globally significant environmental, economical and social benefits. Hydrologically undisturbed wetlands are linked with their surrounding terrestrial areas via several hydrological pathways including groundwater inflow, surface runoff or river water inflow. These various of hydrological and hydrochemical conditions created by the mixing of different water sources promote a rich and abundant wildlife adapted to wet and often nutrient poor conditions. Important wetland functions include storage of water, carbon and nutrients, groundwater recharge, storm protection, flood mitigation, shoreline stabilisation, erosion control, and retention of carbon, nutrients, sediments and pollutants (DUGAN 1990). Economical benefits come from the direct use of wetlands for fishery, forestry, peat cutting, as well as the indirect ecosystems services as nutrient and pollutant abatement for water quality improvement, or global climate regulation (COSTANZA et al. 1997; DE GROOT 1992). Some wetlands are also valued highly for recreational activities with tourism as an important socio-economic factor. During the last decades, drainage and intensification of agricultural and forestry land use resulted in wetland degradation and loss (EEA 1999).

In many countries, wetland policy aims: (I) to conserve the remaining wetlands against present and future threats and (II) to implement landscape planning for the restoration and (re)construction of degraded wetland sites. The Ramsar Convention on Wetlands is a widely and internationally accepted political basis for world-wide wetland planning and conservation focusing originally on nature conservation and biodiversity. Water authorities have expressed the importance of wetlands as sinks and transformators in the water and nutrient cycle at the landscape level in many official documents e.g. European Nitrate Directive, the proposed European Water Framework Directive, the United States Clean Water Act or the Agenda 21.

In the future, wetlands will be important multifunctional landscape elements in a sustainable land use planning. In this process, models are valuable tools to improve wetland planning and management activity at different spatiotemporal scales. The aim of these guidelines is to promote the use of models for effective wetland conservation and management. Therefore, these guidelines present a scale based concept for wetland planning, design and management. In this concept, different models are described with their specific objectives, limitations, input and output data, and parameters and are illustrated with application examples. The model selection in this paper is based on the experiences of the young researchers, which were employed during the WET project, and is therefore limited.

2 Wetland definition

The term ‘wetland’ covers a wide spectrum of habitats under different hydrological and hydrochemical conditions including e.g. mires, fens, bogs, marshes, swamps, coastal areas, river valleys, ponds and small lakes. Important variables used in wetland classification are water level, nutrient status, water sources, vegetation composition and structure, and water flow patterns. The high variability of wetlands is expressed in official definitions like Article 1 of the RAMSAR Convention on Wetlands:

‚For the purpose of this Convention wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six metres.‘ http://www.ramsar.org/key_conv_e.htm

The European Environmental Agency has adopted this definition in their working programme (EEA 1999). The US-Environmental Protection Agency defines wetland in the Federal Manual for Identifying and Delineating Jurisdictional Wetlands (1989) in the following way:

‚Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.' US-EPA, 40 CFR 230.3 and CE, 33 CFR 328.3.

Such wide definitions are useful to include a broad spectrum of areas for administration purposes. But, they cause problems for wetland management which always has to take the specific hydrological and hydrochemical conditions into account. These guidelines are restricted to inland wetland ecosystems only and distinguish between surface flow (freshwater) and subsurface flow (groundwater) wetlands.

In surface flow wetlands the inflowing water is flowing over the soil surface at shallow depths. The main water source in the water budget is river inflow. The sediments in surface flow wetlands have a low hydrological conductivity. Typical examples for freshwater, surface flow wetlands are ponds, shallow lakes, terrestrialization fens, and frequently inundated areas along rivers and lakes.

In subsurface flow wetlands, the inflowing water is mainly flowing under the soil surface through a permeable soil layer e.g. weakly decomposed peat, sand or gravel. The main water source in natural subsurface flow wetlands is groundwater inflow. Typical examples are spring mires, percolation mires, or buffer zones along rivers.

In many natural wetlands, both freshwater, surface and groundwater, subsurface flow pathways occur. These hydrological conditions create a vegetation pattern depending on the quantity of different inflowing water sources.

The variety of wetland functions such as maintaining biodiversity, water quality improvement, harvesting or recreation leads to a diversification of goals in wetland planning and is visible in a branch of terms e.g. natural, seminatural, degenerated, constructed, reconstructed, restored, etc. wetlands. Constructed wetlands are principally designed to serve as water quality improvement or storm water control. However, in many cases constructed wetlands are designed for multifunctional purposes including recreation, fishing, hunting, etc. (PERSSON 1999; BENDORICCHIO et al. 2000).

3 Models as tools in environmental planning

Models are a valuable and widely used tool in environmental planning. Effective environmental planning often demands qualitative and quantitative predictions of the effect of future management activities as arguments for policy makers and administration. Models are applied to solve a wide range of wetland related problems under very different situations including:

- An environmental agency wants to quantify the effect of restored surface flow wetlands on water quality improvement at a catchment scale for the development of a wetland restoration programme.
- A municipality plans to restore (reconstruct) a degenerated wetland and wants to know the effect of different wetland water levels on the water levels in the surrounding terrestrial area to avoid conflicts with neighbouring landowners.
- A wetland manager has to maintain the efficiency of nutrient removal processes in a wetland and wants to know how different management strategies e.g. moving or sediment removal effect quantitatively the efficiency of removal processes to ensure a given water quality standard at the outlet.

Each question can only be answered through the application of models; but each problem requires a different type and complexity of modelling approach. In this context, a model is simply defined as a tool to solve problems.

The aim of this paper is to present different modelling strategies applicable in wetland planning, design or management. The following part gives only a short introduction into model theory, development and application, for a more comprehensive study the reader is referred to “Freshwater ecosystems: modelling and simulation” by STRASKRABA and GNAUCK (1985), the “Fundamentals of Ecological Modelling” by JØRGENSEN (1996), or CHAPRA (1997) “Surface Water-Quality Modeling”. SCHEFFER & BEETS (1994) have written a critique of dynamic ecological modelling concepts.

Tab. 1: Selection of different modelling possibilities for flow, transport and processes on different spatiotemporal scales (source: SCHENK & KAUPE 1998; changed)

SCALE		FLOW	TRANSPORT	PROCESS
SPATIAL	TEMPORAL			
- catchment	- decade	- steady-state / dynamic	- advection	- chemical
- subcatchment	- year	- saturated / unsaturated	- dispersion	- physical
- wetland	- week	- lateral / vertical	- diffusion	- biological
- plot	- day	- 1-; 2-, or 3-dimensional	- one phase	
- soil layer	- hour	- onelayer / multilayer	- multi phase	
combinations of:				
spatiotemporal scale		Flow		
		Flow +	Transport	
		Flow +	Transport	+ Process

The environmental model tool box consists of many different kind of models, some of them are easy to use while others require professional experience. Models are classified by several categories e.g. purpose, scale, complexity and mathematics. Table 1 gives an overview of the broad range of different modelling possibilities for flow, transport and processes on different spatiotemporal scales. The combination of flow, transport and processes in models is often only possible when simple solutions for each problem are applied to keep the mathematical complexity of the model low (SCHENK & KAUPE 1998).

Working with models

Models are composed of several elements: FORCING FUNCTIONS or external variables are functions or variables of an external nature that influence the state of a system. A forcing function can be (I) controllable, e.g. anthropogenic influence on drainage levels or land use or (II) uncontrollable, e.g. stochastic climate variables influencing biotic and abiotic components and process rates. STATE VARIABLES describe, as indicators, the state of the system in question. When modelling e.g. the water balance in a soil profile the moisture content in a given soil layer would be the state variable. MATHEMATICAL EQUATIONS are used to represent the biological, chemical and physical processes in a system. They describe the relationship between the forcing functions and state variables. PARAMETERS are coefficients in the mathematical representation of processes. A basic problem in ecological modelling is the parameter acquisition. Physical based models for water flow and transport use mainly universal constants such as gravity or atomic weights. Parameters describing ecological processes are varying in range and have no constant value. Presently, the majority of ecological models use single values to describe parameters in the simulation.

A Geographical Information System (GIS) is designed to manage and process spatial data. GIS's are also frequently linked with other models and used as a visualisation tool.

Working with models is always a step-wise, iterative processes where the research team follows three steps (e.g. JØRGENSEN 1996; BOUMA et al. 1998):

Problem definition

The first task in environmental planning and research should always be the identification of the problem in strong collaboration with the stakeholders. From the beginning, it is necessary to address the required accuracy of the final answers: Are they needed in a quantitative or qualitative way or somewhere in between. In what time is an answer expected? The answers will help to choose the right spatiotemporal working scale for the further research. Often a problem can be solved using different techniques, it is useful to describe and discuss the advantages and disadvantages of the different procedures with the stakeholders.

Model selection

Once the problem and the working scale has been identified, a model has to be selected. The model selection should be based on the data availability and the funds for a measuring programme. If it comes out that the available data are too limited for the selected model application at a given working scale, then the problem definition has to be reidentified and the working scale in collaboration with the stakeholder updated. This can be done by measuring the needed data or by reducing the model complexity to use the available data together with a discussion of the uncertainties.

Model application and evaluation

The model application itself is an iterative working process of stepwise calibration. First, the model is set up with the available data and knowledge. The simulation results are then compared with measured data e.g. water levels, nutrient concentrations or vegetation pattern. Based on these comparisons, parameters in the model input are calibrated to get a better fit. For details in the definition of calibration, evaluation, verification, validation etc. refer to the definitions on the Camase homepage (see Chapter 7).

3.1 A scale based concept for wetland planning, design and management

Wetland planning, design and management operates at a wide range of spatiotemporal scales ranging from meters-seconds to catchments-years. Being aware that wetlands are interacting at numerous interfaces with their surrounding environment it makes it a rather complex task to model wetlands. Therefore, we propose here a concept where models are used primarily as tools to solve specific problems during different stages of environmental wetland planning. In this scale based concept environmental activities related to wetlands are divided into three stages: planning, design and management. The major stages leading to effective wetland management and protection of a catchment and the interrelations between these stages are shown in figure 1. The data demands for model application will increase with narrowing the spatiotemporal scale. The required data during all stages are obtained in a database. Each stage will get data from the data base and the results are provided as new data to the basic data sets. The stages planning, design and management are closely interrelated and partly overlapping. However, following these stages will be an evolutionary, iterative and continuos process, with later stages of development providing new information that can be used to modify knowledge at earlier stages.

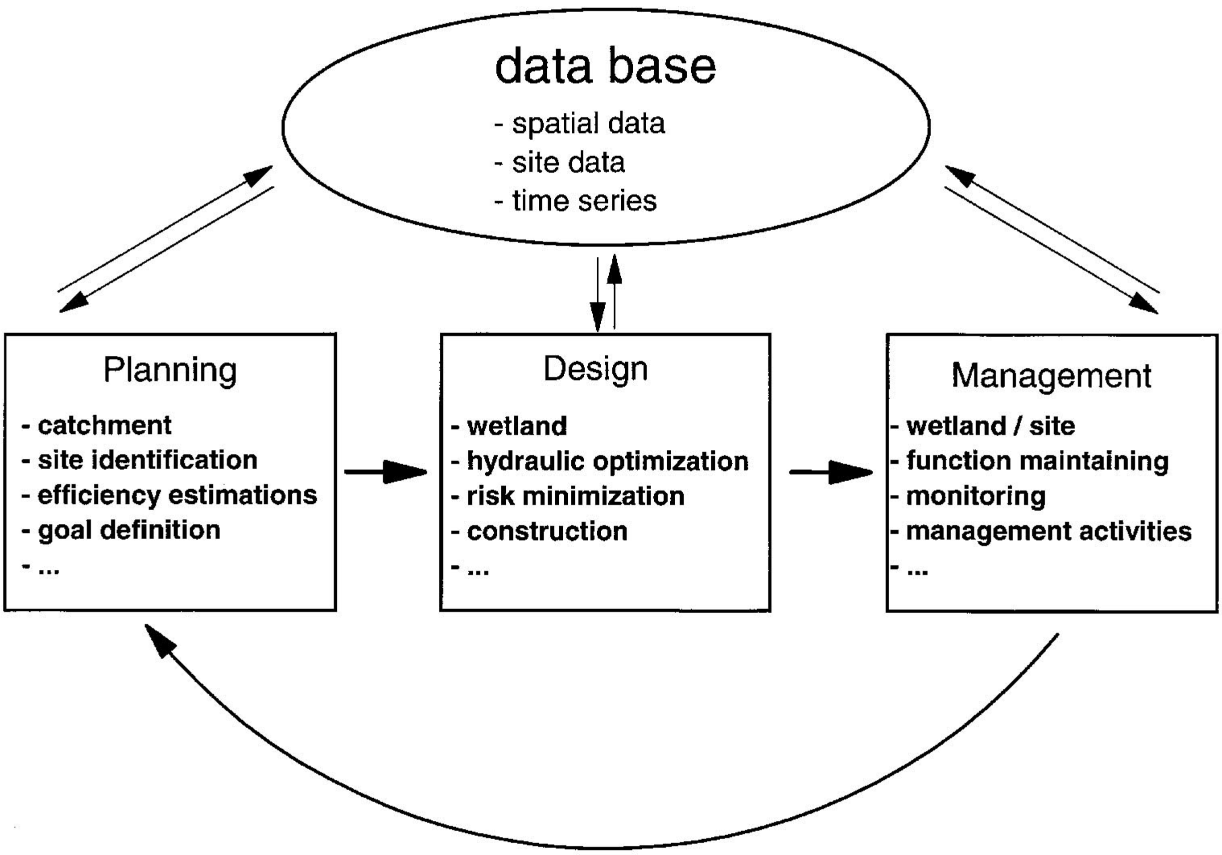

Fig. 1: A scale based concept for wetland planning, design and management.

Wetland Planning

The first stage “Wetland Planning” operates at the catchment scale. The basic goals during the planning stage are: (I) to define aims for wetland policy like maintaining biodiversity, conservation of natural dynamic processes, water quality improvement, and storm water retention and (II) to identify the most suitable wetlands in a catchment to achieve these goals. Wetland planning requires an indepth knowledge of water flows and nutrient loads entering the wetlands. Several models can be applied to obtain this essential information using best available spatial data and a Geographical Information System.

Wetland Design

The next stage is wetland design. Based on wetland policy aims and the site selection carried out in the planning stage, this stage operates at the wetland scale including the near wetland surrounding. The aim during this stage is to sharpen the planning of the wetland management plan, therefore additional hydrological and other data (vegetation, elevation, etc) are collected and evaluated. During this stage models can be applied for example to optimise hydrological flow patterns with the aim to increase retention time or to calculate the width of hydraulic buffer zones to ensure selfregulation processes. The result of the wetland design stage is a wetland management plan including construction works in the wetland and in the wetland surrounding.

Wetland Management

The third stage is wetland management where a wetland has been implemented in a catchment. The aim during wetland management is to maintain and ensure the pre-defined goals for the given wetland. To achieve these aims, it is necessary to install a monitoring programme adopted to the wetland specific goals identified in stage one (DAVIDSSON et al. 2000) During this stage, very specific models can be applied to analyse the development in the wetland and to assist in a cost efficient wetland management. The data and results obtained during the management phase will flow via feedback processes back in the data base and can be used to modify the previous stages.

4 Planning at the catchment scale

Environmental planning for wetland management and conservation at the catchment scale has first to define the environmental management goals and secondly to identify the most sensitive wetlands for achieving these management goals. International laws and convention form the boundary conditions for the regional formulation of wetland policy. According to the Dobris Assessment (EEA 1995) most European countries are committed to extending the protection of wetlands, while still only a small fraction of the continent’s wetland sites are directly protected (EEA 1999). Natural and seminatural wetlands with an undisturbed hydrology are most threatened by water management activities and nutrient input. Management goals for these wetlands are the protection against further nutrient input and drainage to maintain their biodiversity.

Degraded wetlands have lost their biodiversity value and their regulation value due to intensive land use for forestry and agriculture. Management goals for the restoration of degraded wetlands can focus on single functions e.g. restoration for biodiversity or water quality improvement or they can aim, according to the wise wetland use concept (RAMSAR 1987), to restore wetlands as multifunctional landscape elements (e.g. MALTBY et al. 1994).

While environmental goals are defined at a international or national level, the achievement of these goals always operates at the catchment scale. A catchment analysis aims to quantify water and nutrient fluxes by using models coupled to Geographical Information Systems.

4.1 Catchment analysis

The construction of wetlands or the conservation of existing wetlands aimed at the reduction of nutrient concentration in river water is most needed at locations where nutrient delivery from the upstream catchment is largest. The necessary size of the wetland further depends on the (variability) of the water discharge at the site of the wetland. Therefore discharge dynamics must also be quantified. At the scale of a large river catchment (e.g. Po, Rhine, Elbe, etc) only mean values of average discharge by channel flow, overland flow and baseflow (groundwater) are needed. In a second step in site selection, i.e. on regional to local scales (e.g. river Dommel, river Po-tributaries), the dynamics of these discharge flows also need to be quantified.

There are numerous existing models that describe the major processes involved in the transport of water and nutrients. Traditionally these models are based on physical, chemical, and biological descriptions (e.g. water fluxes: SHE (ABBOT et al. 1986), MODFLOW (MCDONALD & HARBAUGH 1984); nutrient fluxes: ANIMO (RIJTEMA et al. 1990); DAISY (HANSEN et al. 1991)). However, these models have been designed for relatively small scales, and require data that are not always available at the catchment scale For less time-consuming assessments one might consider the use of conceptual models (e.g. water fluxes: TOPMODEL, BEVEN 1986; LASCAM, SIVAPALAN et al. 1996; nutrient fluxes: AGNPS, YOUNG et al. 1989; CREAMS, KNISEL 1980). The choice of the most appropriate model to describe water and/or nutrient fluxes at a catchment scale depends on the spatial and temporal extent of the wanted analysis and the availability of data. Therefore, in most catchment studies, specially designed models are used (e.g. JOHNES 1996). Recently, the advances in dynamic modelling within Geographical Information Systems (e.g. PCraster, ArcInfo; GRASS), allow for the development of new concepts to model water and nutrient fluxes at the catchment scale leading to a fast growing number of new models. Here, we present examples of such new, conceptual GIS-based catchment models, which were recently applied in large European catchments (e.g. Elbe, Rhine, Po, and Dommel).

Water fluxes

The first requirement for a wetland is water. For a proper proceeding of ‘siting and sizing’ of a wetland, within the catchment in question, the distributed flow regime of the river and streams must be quantified first.

A conceptual modelling approach was used which treats hydrological response of a catchment as a spatially variable system and generates stream response for every location within the catchment. The model describes the water cycle in a catchment using simplified conceptual water storage elements, each element representing a part of the hydrological cycle. The PCraster model is GIS-based and describes dynamic soil moisture storage, overland flow, interflow, and groundwater recharge in interaction with vegetation, atmosphere and the drainage network. Interaction with deep groundwater is implemented with a coupled MODFLOW model. A detailed description is given in PIETERSE et al. (in prep.) and PIETERSE et al. (1998). Stream response is simulated on a 10-day basis, and is calibrated on two parameters for the outlet of the entire catchment. The calibrated parameters concern (*DIV)* the separation of infiltration to groundwater and direct runoff, and (*R)* for the drainage resistance. In contrast to statistical approaches this PCraster model can be validated with measured discharge within the catchment. Validation results are illustrated in Figure 2.

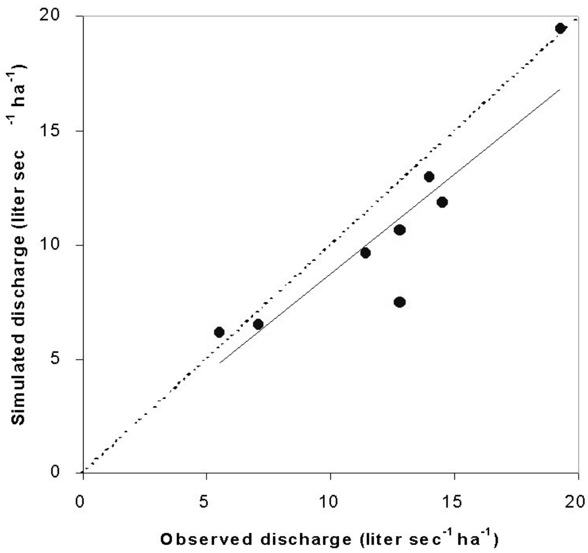

Fig. 2: Simulated vs. observed specific discharge. Modelled with a distributed conceptual water balance model $R^2 = 0.82$. (PIETERSE et al. in prep.).

Nutrient fluxes

The nitrogen load in a stream depends on the sources of nitrogen in the upstream basin and on the conditions that determine the transfer of nitrogen through the soil, groundwater, and river network:

$$L_{XY} = a \cdot (DE_{XY} + (b \cdot SSS_{XY}))$$

where L_{XY} is the simulated river nutrient load on location x,y (kg yr^{-1}), DE_{XY} is the direct (point) emissions to the river network upstream of location x,y (kg yr^{-1}), SSS_{XY} is the surplus (nitrogen input minus crop yield) at the soil surface, upstream from x,y (kg yr^{-1}), *a* is the fraction transferred through the river network (-), and *b* is the fraction transferred through the soil/groundwater system (-).

Nitrogen emissions (DE and SSS) can be computed and mapped using export coefficients applied to geo-referenced data on population numbers, wastewater treatment, industry, livestock numbers, agricultural land use, and atmospheric deposition. This came out for the rivers Rhine and Elbe (DE WIT 1999), and the PO (DE WIT & BENDORICCHIO in prep.). For a more detailed approach (second selection step in the 'siting and sizing' procedure at catchment scale) the export coefficients must be related to local hydrology, soil and land use characteristics. A method for the development of such locally validated export coefficient was performed for the Dommel (PIETERSE et al. in prep.). The different conceptual models were validated with nitrogen and phosphorus loads measured all over the Rhine, Elbe (see figure 3), Po and Dommel river networks.

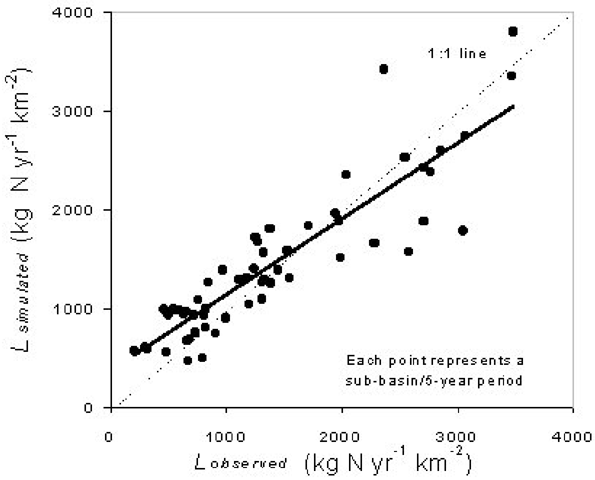

Fig. 3: Simulated vs. observed river N load in the Elbe basin. Modelled with PolFlow $R^2 = 0.80$. (DE WIT 1999).

Examples of application

Both the approaches of PIETERSE et al. (in prep.) and DE WIT & BENDORICCHIO (in prep.) use a dynamic modelling system within a GIS environment (PCRaster). Local drain direction maps (LDD) have been used to route overland flow and interflow water, added with drained groundwater, from upstream to downstream cells. The nitrogen fluxes of each of these water flows were modelled likewise. This implies that the models predict water and nitrogen fluxes at any given point in a river network, either averaged or by 10-days time step variation. Figures 4 and 5 show the kind of information that can be obtained from these models. This information can help planners to find the most appropriate locations for the (re)construction of wetlands.

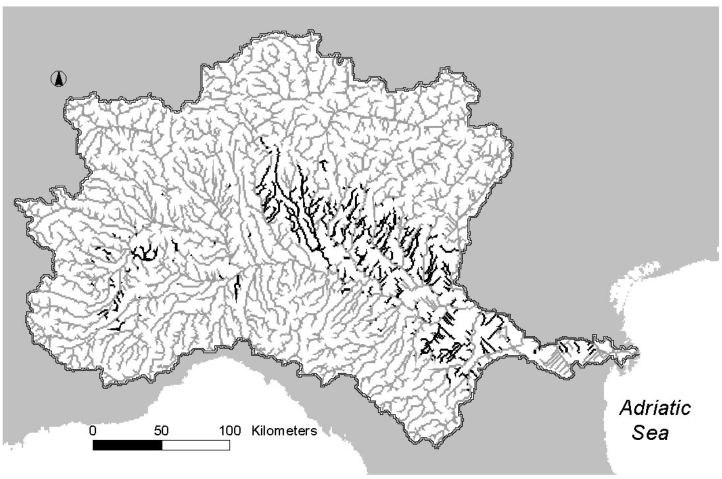

Fig. 4: Branches in the river Po network (Northern Italy) with an average N concentration of more than 10 mg l^{-1} (dark branches), (derived from DE WIT & BENDORICCHIO in prep.).

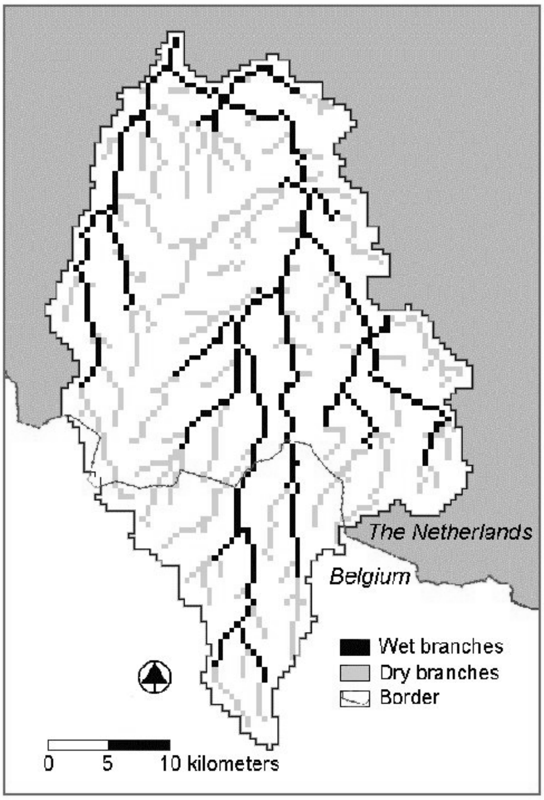

Fig. 5: Branches in the river Dommel basin with an average discharge of more than 100 l s^{-1} (derived from PIETERSE et al., in prep.).

4.2 Geographical Information Systems as a tool for spatial data management

A Geographical Information System (GIS) is needed to store, geo-reference and manipulate large amounts of spatial information. There are several GIS applications available for all platforms e.g. ARC/INFO, PCraster, Idrisi or GRASS. The strength of these tools lies in data analyses along with their extendibility and modularity. Other more non-specialised-user oriented software like MapInfo or ARC/VIEW are available for geographical data visualisation and manipulation. For its flexibility, the possibility of using floating point raster maps, the scripting facilities and finally because it is distributed as free software under the GNU General Public License (GPL) and runs over a wide variety of UNIX flavors, the Geographical Resource Analysis and Support System (GRASS 5.0) can be considered a good choice. This software, available through the internet, has been initially developed by USA CERL and is now maintained by Baylor University at `[http://www.baylor.edu/~grass/index2.html]`.

4.2.1 Selection of GIS based models

GIS software provides specific tools that are indispensable during the catchment analysis. For example, a first step in the determination of the required size for a wetland may be an estimation of the discharge Q that is flowing throughout the purposed surface flow wetland site. In order to achieve this a watershed model is needed. This model relies mainly on a Digital Elevation Model (DEM). Every professional GIS system provides software tools which, from this information, can produce the water flow pathways, the delineation and segmenting of channel networks and boundaries of sub-basins boundaries. This information is helpful in identifying the prominent watershed characteristics. The accuracy of the watershed model is highly dependent on the resolution of the DEM.

In GRASS the software that makes up the watershed model is a module called `r.watershed`. The outputs of this model are: a local drainage direction map containing in each cell a value indicating the direction of outflow of the water flowing in that cell, an accumulation map that, for each cell, gives the number of upstream cells that drain into it, a sub-basins map that groups all the cells that drain into the same stream into the same category and finally a flow streams map indicating the main surface water pathways in the watershed.

Flow streams and sub-basins maps generated by the watershed model can be used to verify the accuracy of the model by comparing them with the river network and basin maps obtained by aerial photographs and larger scale analysis.

In the following paragraphs some additional GIS based tools useful for catchment analysis are presented.

- *ANSWERS:*

Watershed simulation program integrated with GRASS. ANSWERS (Areal Nonpoint Source Watershed Environmental Response Simulation) is an event oriented, distributed parameter model that was developed to simulate the behaviour of watersheds having agriculture as their primary land use. Its primary applications are watershed planning for erosion and sediment control on complex watersheds, and water quality analysis associated with sediment associated chemicals. Because ANSWERS is a distributed parameter model that divides the watershed area into a series of grid elements, one of the primary inputs to the model are spatial data as each element requires input describing land use, soil type, slope and flow direction. Other spatial input data used by the model include the area of the watershed, the location of its outlet, elements containing channels, best management practices (BMPs), and subsurface drainage. The ANSWERS - GRASS integration uses GRASS map layers for these spatial ANSWERS inputs. Other data required by ANSWERS include physically descriptive parameters for each soil and land use type, the size (in meters) to be used for dividing the watershed into grid elements, and data describing the storm event(s) to be modelled.

- *AGNPS:* (`http://www.sedlab.olemiss.edu/agnps98.html`)

AGricultural Non-Point Source Pollution Model 98 (AGNPS 98) is a joint USDA Natural Resources Conservation Service and Agricultural Research Service system of computer models developed to predict non point source pollutant loading within agricultural watersheds. It contains a continuous simulation, surface runoff model designed for risk and cost/benefit analyses.

- *SWAT - GRASS - Interface (Soil and water assessment tool)* (http://www.brc.tamus.edu/swat/)

SWAT is a river basin scale model developed to quantify the impact of land management practices in large, complex watersheds. The model objective is to predict the effect of management decisions on water, sediment, nutrient and pesticide yields with reasonable accuracy on large, ungaged river basins. The SWAT model components are: weather, surface runoff, return flow, percolation, evapotranspiration, transmission losses, pond and reservoir storage, crop growth and irrigation, groundwater flow, reach routing, nutrient and pesticide loading, and water transfer.

- *r.water.fea - Finite element simulation for surface runoff*
  (http://www.baylor.edu/~grass/userman/html/r.water.fea.html)

`r.water.fea` is an interactive program that allows the user to simulate storm water runoff analysis using the finite element numerical technique. `r.water.fea` computes and draws hydrographs for every basin and stream junctions in an analysis area. It also draws animation maps at the basin level. The maps required by `r.water.fea` are: basins, streams, drainage direction, accumulation and slope.

The other data requirements of r.water.fea are the parameters needed to calculate infiltration and the channel roughness parameter. Model parameters may be provided either in the form of maps or as values: Manning roughness coefficient map or basin value, saturated hydraulic conductivity map or basin value, suction head at wetting front map or basin value, effective porosity map or basin value and degree of saturation or basin value.

4.3 Siting and sizing of surface flow wetlands with a GIS based Score system

This section presents an approach to identify and assess the suitability of possible wetland sites in a catchment with a GIS based score system (PALMERI & TREPEL in prep.). The methodology of the score system was developed to identify the site and size of surface flow wetlands for water quality improvement. The methodology can easily be adopted to other management goals by defining a score system and spatially distributed data layer for the goal in question.

Limitations

The model effectivety can be limited through the availability and quality of geographical data (i.e. spatial resolution etc.). The score system assumes mean annual conditions, therefore important temporal variation in discharge or nutrient concentrations are not included. However at the landscape level and in a pre-design phase this appraisal should be considered a first endeavour, sufficient at least for refining the siting LSS thus permitting the selection of those sites which, from an hydrological point of view, present an adequate amount of land.

Model description

The model couples raster based GIS analysis with the required area estimation of the wetland. The former is needed for addressing temporal and spatial variability of climatic, hydrological, geological, environmental and socio-economic data. The latter is basically a function of system parameters such as the hydraulic retention time and of the characteristics of the surrounding environment, for example discharge. The Land Score System (LSS) is implemented on the basis of an environmental policy objective (e.g. nitrogen or more in general nutrients abatement) by elaborating and combining the geographical data layers.

input data

The Land Score System is built by a set of spatially distributed data layers which reflect the hydrological, ecological and socio-economic conditions in the catchment. Three data layers (Slope, Depressions and Highlands /Lowlands) are constructed from a Digital Elevation Model. The other data layers (Soil Type, Land Use, Population density, historical distribution of wetlands, and river proximity) are generated from available data sources. The values in each data layer e.g. peat, sand or clay in the soil type layer are assessed according to their suitability for wetland restoration for water quality improvement.

The wetland size estimation is based on the calculation of hydraulic retention time and requires an estimation of discharge Q which is obtained from simple climatic water balance calculations.

output data

The final output of the methodology consists of two data layers: a land score layer (S) and a layer containing in each watershed cell the ratio of available land for wetland versus the area required for wetland based water treatment (H).

Technical details

The general idea for building a Siting LSS is to calculate for each of the information layers a spatially distributed land attribute, hence generating a set of layers with values in the range *[0,1]*. A land score layer (S) is obtained by clustering the land attribute layers by means of a weighted average. Finally the values obtained are grouped into six categories so that each cell in this last LSS layer will contain integer values in the range 0-5; 0 meaning that the cell is not suitable for wetlands and 5 that it is the most appropriate location.

As a first step the attribute layers with cell values in the interval *[0,1]* are calculated. For each layer L_i, for each cell *k* the data value is checked and *if it is favourable (positive)* in the scope of wetland (re)construction, a value of 1 is given in the resulting attribute layer, that cell in the attribute layer will otherwise be given a value of 0. Intermediate cases are given values in the interval *] 0,1 [*.

Once all the *nl* attribute layers have been produced the Land Score Layer (*S*) with cell values in the range *[0,1]* is obtained by calculating a simple weighted average on all the *nl* layers L_i

$$S = \frac{\sum_{i=1}^{nl} w_i L_i}{\sum_{i=1}^{nl} w_i}$$

Having selected the most suitable locations and given the local environmental and boundary conditions are met, how big does a wetland need to be in order to achieve the required efficiency in pollution abatement ?

A starting point in addressing this question is to obtain an initial rough size estimation, based on the quantity of water that discharges into the cell under question. The estimation relies on some general assumptions, notably on hydraulic retention times (*T*) required to achieve a significant abatement of a target substance and on typical wetland depths (*d*). The wetland's required surface area A_k is derived from the definition of the quantity of water that discharges in the raster cell *k*, Q_k, by the relation

$$A_k = \frac{Q_k \cdot T}{d} \quad [m^2]$$

Application example:

The **Adige-Bacchiglione** watershed case study (referred to hereafter as ADIBAC) is considered here. The scale of reference for the geographical data layers used for the Siting procedure is 2500 m^2 (0.25 ha) in space and 1 year in time.

The drainage basin of the ADIBAC, situated in the north-eastern part of Italy extends over a surface of ~50.000 *ha* distributed among the provinces of Padua and Venice and encompasses 31 municipalities. 47.130 *ha* drain into the Lagoon of Venice through one single outlet "La Botte delle Trezze", situated in the most eastern part of the basin. Up to 46.052 ha (the 94% of the total extension) are mechanically drained through a network of 650 *km* of ditches and small channels. This area has been exploited for agricultural purposes over many years. Extensive drainage activity has been carried out since the 16th century. The quantity of water per annum mechanically lifted is on average 150 million cubic meters.

Fig. 6: Score values assessing the suitability of land for the construction of surface flow wetlands for water quality improvement in the Adige-Bacchiglione catchment (Italy) (PALMERI & TREPEL in prep.)

The data for this analysis concern the altimetry (1.0), slope (1.0), depressions (1.0), distance to river network (1.0), soil type (2.0), land use (3.5), population density (3.0) and presence of historical wetlands (1.0). These eight geographical data layers are the minimal set of land attributes that are required in order to implement the LSS. The attribute layers can now be combined by means of equation in order to produce the Land Score Layer *S*. The distribution of weights (values in brackets) among the land attributes depends on the aspects on which the surveyors focus.

The map in Figure 6 contains continuous values that are grouped into 6 categories: < 0.5, 0.5-0.6, 0.6-0.7, 0.7-0.8, 0.8-0.9 and 0.9-1.0 indicating the suitability of areas for the restoration of wetlands. Most of the cells score in the first 3 lower categories. Some areas score in the two highest categories. It is then possible to conclude that, from an economical and geological point of view, the last areas are most suitable in terms of wetland (re)construction.

The next step is to calculate an estimation of the size required by a wetland placed in these areas and then select only those sites that present an adequate amount of land for the implementation of an efficient wetland system. In order to achieve this from equation (2) values for the daily discharge Q_k for each cell *k* in the watershed, the wetland depth *d* and the HRT *T* are needed. Once calculated a discharge map (D) by means of a watershed model, in order to apply equation (2) it is necessary to use some kind of typical or average values for the depth *d* and for the HRT *T*. Based on the literature, KADLEC and KNIGHT (1996) suggest an HRT of 3 to 6 days to be in general sufficient for nitrogen abatement. Since this is one of the most important factors for the water quality and by observing that this time interval is commonly adequate to remove the most part of suspended phosphorus and sediment, a value in the middle of this interval, e.g. *T*=5 days, is assumed as a customary estimation for HRT. Concerning the depth, it can be said that, by definition, a wetland system exhibits a water depth between 0 to 1 m. It is here assumed *d*=0.5 *m* as a typical value for the depth of a wetland system. Using these assumptions the ratio *T/d* is given a value of 10 and thus by multiplying the discharge map *D* produced with the watershed model by this value and dividing by 10000, a map (*A*) is produced with values expressing the number of hectares of wetland required in order to fully treat the water flowing in each cell.

$$A = \frac{D \cdot T}{d} \qquad [m^2] \qquad = \qquad \frac{D}{1000} \qquad [ha]$$

The map *A* obtained by means of this procedure for ADIBAC contains values in the range 0-210 *ha*

This calculation gives a primary rough estimation for the required wetland area. The size has to be defined precisely case by case in the following wetland design phase. However from these calculations some more criteria that will help in restricting further the available sites for wetland implementation are gained. Assuming that a box of 500 * 500 *m* (25 *ha*) is large enough in order to build a wetland, from the Land Score Layer *S* is calculated how much of the box's surface is available for wetlands. By scanning the *S* layer on the whole watershed area with a window of 500 *m* side it is possible to count how many hectares there are available for wetland, i.e. have a score in the last three classes. From this calculation the layer *W* is obtained, depicting the amount of land (in hectares) available for wetlands. The layer *W* has values in the range 0-25 *ha*.

In the meantime, with the same window, another layer A_{max} is constructed by assigning the maximum value presented by layer *A* in that window to the 500 *m* boxes. Finally, a last layer *H* is produced by calculating the ratio W/A_{max} giving a percentage value indicating how much of the water entering that window may be treated by wetland systems (figure 7).

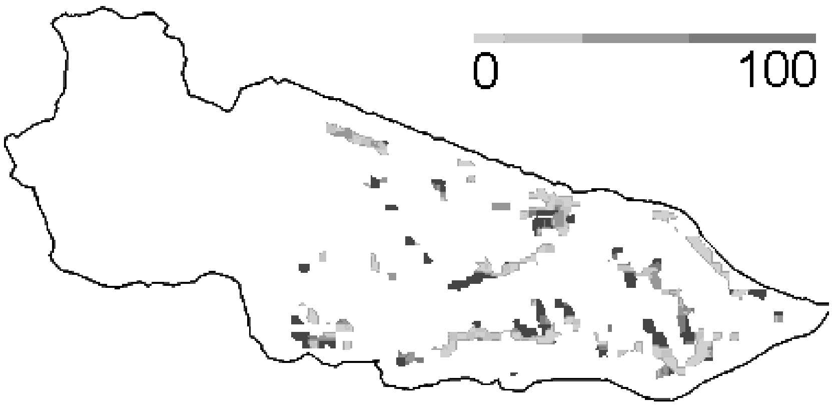

Fig. 7: Possible water treatment in surface flow wetlands (Layer H %) in the Adige-Bacchiglione catchment on the basis of the siting and sizing procedure.

Layer *H* presents large amounts of land that have scored a percentage higher than 25 %. Many of these sites can thus be considered favourable from an economical, geological and finally hydrological point of view in the scope of wetland implementation.

The methodology introduced here has the result that many of the areas of ADIBAC already selected by decision makers at the regional-level for wetland implementation are in fact suitable for this objective. This tool should then be considered a first unbiased address, useful to foster sound justifications of these choices to the attention of the policy makers and of the population involved.

4.4 Nutrient retention

The retention of nutrients and pollutants is an important landscape ecological function of wetlands. The specific hydrogeological and hydrochemical conditions in wetlands foster concentration reducing, physical, biochemical and biological processes including sedimentation, denitrification, absorption or plant uptake resulting in an overall water quality improvement. Water authorities require quantitative estimations of the retention efficiency as arguments for the implementation of wetland restoration programmes at a catchment scale.

At this stage in the planning process, models can calculate nutrient retention capacity of wetlands with available data sets. In southern Sweden, the complex GIS based HBV-N model has been used in an area of 145000 km² in the national decision-making process for best management practise including calculations of nitrogen leaching, nitrogen retention in groundwater, wetlands and rivers, and net transport to the sea on a daily basis (ARHEIMER & BRANDT 1998). The high data demand restricts the powerful HBV-N approach only to well equipped and experienced research institutions. On the other hand, very simple approaches for the calculation of nutrient retention, e.g. the linear relationship between wetland retention and wetland area used by for example JANSSON et al. (1998), are very rough and neglect the hydrogeomorphologic variability of wetlands.

Nutrient retention in surface flow wetlands

A useful tool for assessing the retention efficiency of freshwater, surface flow wetlands quantitatively requires minimum information about wetland morphology, hydrology and hydrochemistry, such a tool is PREWET.

Objective

PREWET was developed by DORTCH & GERALD (1995) for the U.S. Army Corps of Engineers at the Waterways Experiment Station as a predictive model that can be rapidly applied with minimal data input for estimating the amount of pollutant removal provided by surface flow wetlands (e.g. DORTCH 1996)

Limitations

The PREWET approach aims to minimise time and effort for model implementation to gain maximum information about removal rates. Simplifications are achieved by making assumptions that reduce complexity of the predictive mathematical formulations and input data requirements. The model calculates removal rates in surface flow wetlands under steady state conditions. Seasonal variation of discharge and pollutant concentration are not included.

Model description

PREWET assumes steady-state conditions and either fully mixed or one-dimensional, longitudinally varying concentrations. These simplifying assumptions allow rapid model implementation with minimal input data requirements. Given basic wetland characteristics and the pollutants of concern, PREWET estimates the amount of pollutant treatment (i.e. removal) provided by a wetland. PREWET addresses biochemical oxygen demand, suspended solids, coliform bacteria, total nitrogen, total phosphorus, and contaminants (organic chemicals and trace metals).

input data

The model requires information on wetland morphology e.g. wetland length and width, mean water table depth and the amount of inflowing water Q to calculate the hydraulic detention time. Selected pollutant mean inflowing concentrations are needed to run the model. For the specific removal processes, different decay or settling rates can be chosen.

output data

The model calculates the amount of pollution removed from the receiving water as removal efficiency in percent for the following water quality indicators: total suspended solid (TSS), or total inorganic suspended solids (ISS), total coliform bacteria (TCB), biochemical oxygen demand (BOD), total nitrogen (TN) and total phosphorus (TP).

Technical details

The model PREWET and documentation are available free of charge through the Internet. The detailed documentation includes all equations, thus enabling the user to calculate removal efficiency even without the software. The model runs under DOS.

Application example

The equations of the model PREWET were applied for three potential wetland areas in the Neuwührener Au catchment (40 km²) in northern Germany in order to test the effectivity of these potential wetlands for nitrogen abatement (TREPEL & PALMERI in prep.).

Tab. 2: Comparison of nitrogen removal efficiency of three wetlands in the Neuwührener Au catchment calculated with Prewet; The model calculations assume plug flow conditions, a mean nitrate concentration of 3.6 mg/l, a mean total nitrogen concentration of 5.0 mg/l and a denitrification rate of 0.12 mg N d^{-1}.

Wetland		Moorsee	Wellsee	Pohnsdorf
Upstream area	[ha]	1083.5	2068.3	3858.7
Wetland area	[ha]	66.3	56.5	62.8
Downstream area	[ha]	2803.0	1828.0	31.3
Average wetland length	[m]	1000.0	700.0	700.0
Average wetland width	[m]	663.0	807.1	897.1
Mean water depth	[m]	0.5	0.5	0.5
Hydraulic residence time	[d]	16.8	5.1	2.8
Removal efficiency for Total Nitrogen	[%]	76.6	35.6	21.5
Wetland load	[t N yr^{-1}]	17.0	32.5	60.6
Wetland retention	[t N yr^{-1}]	13.0	11.6	13.0
Catchment load	[t N yr^{-1}]	48.0	49.6	48.1
Catchment load reduction	[%]	22.7	20.1	22.6

The removal efficiency of these wetlands decreases with increasing catchment area from 75% in the most upstream wetland to 21% in the most downstream wetland, depending on the wetland size and length/wide ratio. The total amount of removed nitrogen is predicted in all three wetlands to be around 13 t N yr^{-1}. According to these calculations, restoration of any of these three wetlands with a free flowing water surface would reduce nitrogen load from the Neuwührener Au catchment for about 20%. These quantitative prediction about wetland efficiency can be used by water authorities together with other informations to decide spatially explicit where to start a wetland planning with a higher accuracy.

Nutrient retention in groundwater fed, subsurface flow wetlands

In groundwater fed wetlands, nutrient retention occurs along lateral flow pathways. Nutrient removal rates can only be quantified on the basis of a validated water budget (DAVIDSSON et al. 2000). The data needed to asses nutrient retention include information on groundwater levels, and geology; pollutant concentrations in different water layers and pollutant specific decay rates are presently not always available. A simple model approach for the quantification of nutrient removal rates in groundwater fed wetlands was not found. However, geohydrological models allow the quantification of nutrient and pollutant transformation under steady state conditions. The required decay rates are generally based on empirical relationships and do not include spatiotemporal variations or distinguish between different biochemical processes. Geohydrological models are discussed in chapter 6.2 in more detail.

5 Wetland design

Once environmental planners have identified an area as suitable to fulfil one or more functions of a wetland, they will ask for a detailed design phase to optimise this function in order to achieve the predefined goal. These construction plans are developed mostly by two different groups: wetlands with nature conservation as their aim are planned by biologists, while hydraulic engineers design wetlands for e.g. water quality improvement or flood protection. In an effective wetland restoration programme both groups are working together. In Sweden and Australia constructed wetlands as ponds etc. are, as a rule, designed by landscape architects (PERSSON 1999).

In the design phase, models are applied to analyse the water flow pattern under present conditions and to develop an optimised flow pattern in order to achieve the environmental goal. For example, the conservation of endangered plant communities e.g. mesotraphent small sedge reeds can only be successful if the hydrologic conditions in the surrounding are recognised. In this case, models are a useful tool to calculate the width of buffer zones around the wetland to ensure a predefined water level (VAN WALSUM & JOOSTEN 1994).

5.1 Designing surface flow wetlands

When designing water quality ponds or wetlands which aim to reduce nutrients, some basic aspects must be considered. The first and most obvious - if the wetland is to improve the water quality - it must be designed so that sedimentation and denitrification processes are as optimal as possible (BENDORICCHIO et al. 2000). The wetland can then have several other functions as a place for recreation, reducing high flow events, improving the natural environment and the landscape. All these functions place demands on aesthetics and on the flora and fauna habitat.

The first aspect, to create a good design for sedimentation and denitrification processes is achieved by a) having a low amount of mixing in the water body, and b) increasing the effective water volume. From an engineering and hydraulic standpoint, a rectangular water body with an inlet and outlet placed along the short sides would in principal be the best solution to cope with this.

A wise landowner would, however, not accept this in all situations, especially if aesthetic or recreational values are appreciated. This is also the case in technical, economical and ecological reasoning, as it is expensive to excavate large amount of soil and for the quality of the natural environment. This is then the task for the engineer to solve. How to create a good design for sedimentation- and denitrification processes (i.e. to optimise the hydraulic efficiency) and at the same time create good aesthetic and natural environments?

The answer to the last question is to use models and the most suitable models are the Computational Fluid Dynamic (CFD) models in 2- and 3-D.

Models for surface flow analysis and optimisation

A general 3-D flow analysis is, however, very complex and has become possible only with the rapid increase of computer capacity in the last few years. The basis is the solution of Navier-Stokes equations using different numerical methods. Restricted to two-dimensional (2-D) or two-dimensional depth integrated flow (2.5-D) the analysis is significantly faster.

There have been some studies on plain 2-D models, i.e. that the depth of the wetland is not considered. These simulations have however had several limitations to predict the reality. If one wants to use a 2-D model, it is better to use a depth-integrated version that takes depth into consideration.

Several commercial codes can be found on the market. The most common depth-integrated 2-D models are WASP/DYNHYDR5 (U.S. Environmental Protection Agency), SMS/RMA2 (U.S.

Army Corps of Engineers), and Mike21 (Danish Hydraulic Institute). Same examples of 3-D commercial codes are FLUENT, CFX, Mike3, FIDAP, and PHOENICS.

The basic principle in 3- and 2.5-D models is that a 3-D bathymetry is built up by many small cells, which together create any wanted volume. Then the water volume is defined within the bathymetry, as well as the physical characteristics of the water and the surfaces. In these kinds of models different factors can be added as evaporation, source (i.e. a pump or rain), barometric pressure and wind. The model is then basically driven by gravity, even though forces like barometric pressure and wind can be added.

The model can then simulate several different physical, chemical and biological processes, as water dynamics, tracer experiment, sedimentation, eutrophication or coliform bacteria.

Input and output data: an example of a 2.5-D model

To give an example of input data to simulate a tracer experiment in a 2-D depth-integrated model the following are used: bathymetry, simulation period, timestep, inflow, bed resistance, eddy viscosity (turbulent dynamic viscosity), component description of tracer, inlet boundary concentration and dispersion coefficients. Performing a simulation with this input data one can receive all flow velocities, water levels and concentrations in both time and space within the bathymetry.

Limitations

A two-dimensional, vertically integrated model is something between 2-D and 3-D, since it considers varying topography. Such a model can therefore be seen as a 3-D model, but with homogeneous vertical flow. It can not, however, consider 3-D effects in the flow, as in vertically stratified flow or in basins with steep slopes. But otherwise it can be assumed to represent the hydraulic conditions well. The advantage of a 2.5-D model compared to a 3-D is that it is less time-consuming, in regard to both grid-generating and simulation time, especially when simulating unsteady flow.

Examples

Flow patterns in ponds have been studied in 3-D by several researchers (SHAW et al. 1997, PETTERSSON 1999). Also tracer studies in basins and ponds have been simulated in 3-D and compared to field measurements (ADAMSSON et al. 1999; MATKO et al. 1996; MATTHEWS et al. 1997). TA and BRIGNAL (1998) have carried out a study on the effects of modifications of inlet and outlet in a storage reservoir.

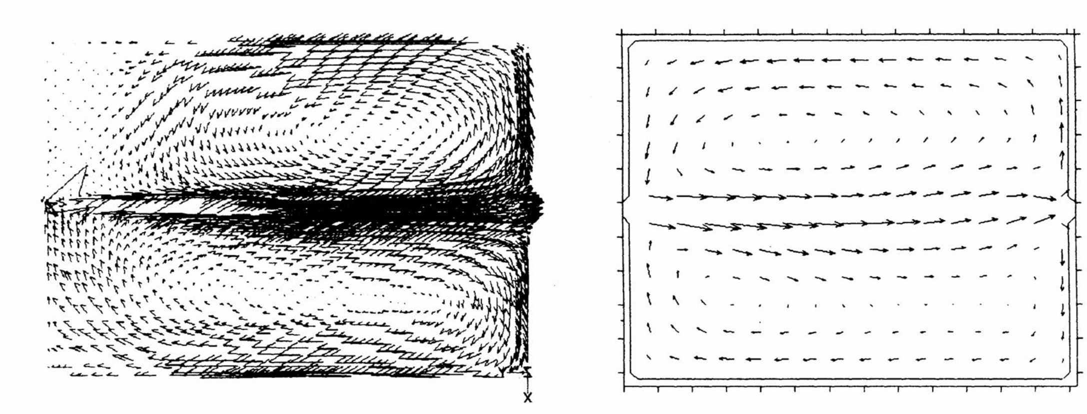

Fig. 8: Calculated flow pattern in 3-D (15 cm beneath the surface) (left) and 2.5-D (right) for 20 l/s (from: ADAMSSON et al. 1999)

In a study by GERMAN and KANT (1998) flow patterns in a pond were simulated with both a 3-D and a 2-D model. The 2-D model performed well when compared to the field measurement in some parts of the pond, but not in the entire pond. The 2-D model did not work in those parts where the topography was more complicated.

Some studies have been completed which compare 2.5-D numerical models to measured flow pattern or tracer response of ponds. BENELMOUFFOK and YU (1989) developed a 2.5-D code and compared the simulated flow pattern with field data from a small shallow pond. ADAMSSON et al. (1999) compared how well Mike 21 could simulate flow and tracer studies made in a test basin (Figure 8). These two studies were made in smaller water bodies that were similar to basins, contained no vegetation and had either flat or mildly sloping bottoms. PERSSON (1999), SOMES 1997 and BARRETT (1996) have all used a 2.5-D model on vegetated ponds/surface flow wetlands.

5.2 Designing groundwater fed wetlands

The flow patterns in many groundwater fed wetlands are effected by water management activities in the catchment like large scale drainage activities or pumping for drinking water. These activities change both the water level in the wetland and the water quality due to a different degree of mixing of different inflowing water sources (e.g. GROOTJANS et al. 1996). Hydrochemical changes of water quality composition influence the vegetation composition. In particular, plant species which are adapted to mesotrophic, base-rich conditions may decrease due to acidification processes in the plant available water layer. Restoration and management of groundwater fed wetlands require a hydrological analysis both on the site and the regional scale in order to compare the actual flow pattern with the natural situation and to develop a flow pattern which is less influenced by anthropogenic activities and nearer to the natural situation. The groundwater flow pattern of such sites has been analysed with 2-D or 3-D groundwater flow models (e.g. KLUGE et al. 1994; GROOTJANS et al. 1996; REEVE et al. 2000; SCHOT & MOLENAAR 1992). These models are presented in detail in chapter 6.2.

6 Management optimisation

Once a wetland has been identified to fulfil one or more functions in a catchment, the wetland manager has to maintain the functioning. The assessment of the functioning can be done only on the basis of a monitoring strategy, where the measured parameters give valuable information about the state of the wetland. Therefore, it is necessary to develop for each wetland an independent monitoring programme aiming to asses the specific environmental goals based on the specific hydrogeological conditions. DAVIDSSON et al. (2000) give a detailed overview about different monitoring techniques for wetland management.

Models can be applied during wetland management to analyse the present situation with a high spatiotemporal resolution. Based on these results, suggestions for effective management activities are made. Applying models during the stage of wetland management will always increase the understanding of the ongoing processes in the wetland. The selection of a model should be based on the most important question to be solved during wetland management because the data demands for modelling will be high. Very detailed and different models are applied presently for wetland management, therefore this section presents only three examples: The application of a detailed water quality model, the analysis of groundwater flow pattern and quantification of the water budget in groundwater fed wetlands and finally a selection of models for the analysis of biotic interactions in wetlands.

6.1 Water quality in freshwater, surface flow wetland

The efficiency of a wetland system in pollutant abatement is considered as the difference of water quality (pollutant concentrations) between inflow and outflow water where the hydraulic detention time has been taken into account.

Water quality is measured by means of well established quality indicators like dissolved oxygen (DO) or biological oxygen demand (BOD_5) (CHAPRA 1997), nutrient concentrations and other indicators depending on the quality goal in view. The dynamic of these indicators is strongly related to the HRT, to the pollutant loads, to the discharge Q as well as to design parameters. A water quality model is one that describes the fate and transport of these indicators.

The literature presents several complex models for the nutrient dynamics in wetlands (e.g. MARTIN & REEDY 1997; SPIELES & MITSCH 2000). DALL'O' (in prep) give a critical review of the current state and problems in wetland water quality modelling. BENDORICCHIO et al. (2000) present some simple water quality models. The variety of wetlands and investigation design led to a diverse evolution of models for nutrient dynamic in surface water wetlands. These models are applied at one site and mainly for scientific purposes. The WASP model system is a freely, distributed dynamic modelling system.

WASP5: Water Quality Analysis Simulation Program

The WASP modelling system is a generalised modelling framework for contaminant fate and transport in surface waters. Based on flexible compartment modelling, WASP can be applied in one, two or three dimensions. WASP is designed to permit easy substitution of user written routines into the program structure. This model is freely distributed and maintained by the Environmental Protection Agency of the United States (US-EPA) (AMBROSE et al. 1993).

Model objective

WASP is a compartmental dynamic model that can be used to analyse many different problems related to the water quality of shallow water bodies, rivers, estuaries and lakes. Problems that have been studied using WASP include biochemical oxygen demand, dissolved oxygen dynamics, nutrients/eutrophication, bacterial contamination, and toxic chemical movement.

Limitations

The great quantity of input data and the consequent difficulties in calibration are limitations. Due to its complexity under particular circumstances the model may exhibit numerical instability.

Model description

The distribution includes three different executables programs: *Dynhyd* to simulate the water movement, *Eutro* and *Toxi* to simulate transformations of substances present in the water and that may be used to evaluate the surface water quality. For a detailed description of Dynhyd refer to the software and model manual (AMBROSE et al. 1993).

While Eutro is intended to simulate the eutrophication phenomenon, Toxi permits to trace the fate of toxic micro-pollutants.

The term "WASP model system" is used to refer to the TOXI model, the EUTRO model, the DYNHYD program, and all associated support files and programs as a single unit. The term "WASP model" is synonymous with "WASP model system".

The Hydrodynamic Program (DYNHYD) is a simple link-node hydrodynamic program capable of simulating variable tidal cycles, wind and unsteady flows. It produces an output file that supplies flows, volumes, velocities and depths (time averaged) for the WASP modelling system.

The Toxic Chemical Model (TOXI) combines a kinetic structure adapted from the Exposure Analysis Modelling System (EXAMS) with the WASP transport structure and simple sediment balance algorithms. TOXI predicts dissolved and sorbed chemical concentrations in the bed and overlying waters.

The Eutrophication Model (EUTRO) combines a kinetic structure adapted from the Potomac Eutrophication Model with the WASP transport structure. This model predicts dissolved oxygen, carbonaceous biochemical oxygen demand, phytoplankton, carbon, chlorophyll-a, ammonia, nitrate, organic nitrogen, and orthophosphate in beds and overlying waters. WASP5 permits the modeller to structure one, two and three dimensional models; allows the specification of time variable exchange coefficients, advective flows, waste loads and water quality boundary conditions; and permits tailored structuring of the kinetic processes, all within the larger modelling framework without having to write or rewrite large sections of computer code. The two operational WASP5 models, TOXI5 and EUTRO5, are reasonably general. In addition, users may develop new kinetic or reactive structures. This, however requires an additional measure of judgement, insight, and programming experience on the part of the modeller. The kinetic subroutine in WASP (denoted "WASPB"), is kept as a separate section of code, with its own subroutines if desired.

input data

WASP traces each water quality constituent from the point of spatial and temporal input to its final point of export, conserving mass in space and time. To perform these mass balance computations, the user must supply WASP with input data defining seven important characteristics:

1. simulation and output control
2. model segmentation
3. advective and dispersive transport
4. boundary concentrations
5. point and diffuse source waste loads
6. kinetic parameters, constants, and time functions
7. initial concentrations

output data

DynHyd: water heads, volumes and velocities at the segments junctions.

TOXI: conservative tracer concentrations in each channel for the different advection /dispersion fields.

EUTRO: concentrations of many water quality relevant parameters such as the various forms of nutrients, dissolved oxygen, chemical oxygen demand and chlorophyll-a.

Technical details

The WASP5 program operates under MS-DOS. The software and documentation is available through the Internet at: http://www.cee.odu.edu/cee/model/wsp_desc.html

Application example

The WASP model (in particular TOXI) has been applied in conjunction with MIKE11 in the wetland of Castel Nuovo Bariano (Italy). This study aimed to obtain information on the prevailing factors that determine the hydraulic retention time. The hydrodynamic output of MIKE 11 has been used as an input for a single punctual tracer release event simulation with TOXI. The outputs of the model have been calibrated with experimental tracer data (Li), and different flow scenarios have been evaluated.

6.2 Water flow and water budget in groundwater fed wetlands

Effective wetland planning and management requires (at least) a basic understanding of wetland hydrology. Knowledge about quantity and quality of different water sources entering the wetland is especially needed. Most wetlands receive groundwater inflow from their surrounding catchment. The quantity and quality of lateral inflowing water controls vegetation composition and structure. Nutrient retention along lateral pathways is quantitatively of great importance in the landscape nutrient balance, because in these wetland-ecotones, nutrients from adjacent terrestrial ecosystems enter for the first time semiterrestrial systems. Using these ecotones for nutrient transformation and retention can decrease nutrient input into aquatic ecosystems significantly. However, quantifying nutrient retention along buffer zones is a complex task in both measuring programmes or models (e.g. BLICHER MATTHIESEN & HOFFMANN 1999; HAYCOCK et al. 1993).

This section presents two models for (I) analysing groundwater flow pattern and (II) calculations of wetland water exchange and balance. For the analysis of the regional flow pattern are commercial numerical groundwater flow models most suitable. However, the interface between saturated and unsaturated conditions or overflooding might be considered as boundary conditions in a limited temporal discretisation. With FEUWAnet which is specially designed for the simulation of short term groundwater dynamics and water balance of heterogeneous riparian wetlands these limitation can be solved.

6.2.1 Groundwater flow and transport models

Wetlands are hydrologically connected with their surrounding groundwater basin. The regional geohydrological conditions control directly or indirectly wetland features like vegetation composition or nutrient status. Geohydrological models are applied in wetland planning and management on different scales: first to analyse the water flow pattern from the surrounding catchment to the wetland (e.g. GROOTJANS et al. 1996) or secondly to study lateral water flow inside the wetland (e.g. KLUGE et al. 1994; REEVE et al. 2000). Geohydrological groundwater flow and transport models are a wide spread tool among geohydrologists. They use several 2 or 3dimensional codes e.g. MODFLOW, FLONET/TRANS, MIKE-SHE, etc.. Such models are applied frequently in order to understand the flow pattern, the seasonal variation and to quantify different inflowing water sources.

Limitations

Geohydrological models require, as input data, spatially explicit information about the distribution of different geological layers and their physical properties. This kind of information is gained by the evaluation of geological drillings and maps. However, most landscapes which were formed by glaciation processes show a high spatial variability in their distribution of geological strata at a local scale. The problems with the regionalisation of input data can be partly solved by the application of fuzzy-krigging technique (PIOTROWSKI et al. 1996).

Mathematical and conceptual problems occur during the application of geohydrological models, when groundwater enters the surface layer. At present, geohydrological models are restricted to saturated conditions. In many wetlands, saturated and unsaturated conditions change frequently in the upper soil layer. These water level fluctuations effect more importantly many processes in the nutrient cycle e.g. mineralisation of nitrogen.

Model objective

Choosing a geohydrological model depends on the purpose of your study and the available data. For simple analyses of regional or local groundwater flow conditions, 2-dimensional models like

FLONET/TRANS maybe already a successful tool. The code calculates flow pattern and water fluxes under steady-state conditions, but adapting input data (hydraulic heads, specific flux) to representative hydrological periods allows analysis of seasonal variation under quasi-steady state conditions. More complex model systems like MODFLOW or MIKE-SHE can be applied both in 2 or 3 dimensions under steady-state or dynamic conditions. They are mainly restricted by the availability of spatial distributed input data and hydrological boundary conditions.

The model complexity may increase with available data. In a good wetland hydrology concept, one would first develop a rough geohydrological model based on the available geological and hydrological informations. The simulations are then used to decide where more drillings are needed and where to place piezometers to monitor hydraulic heads. This additional information is needed to build a more precise geohydrological model.

input data

Groundwater flow models require data on the spatial distribution of geological substrate, their physical properties e.g. hydraulic conductivity, storage coefficient, data on hydraulic heads or specific flux, information about drainage and pumping activities and climate data (precipitation and evapotranspiration).

output data

Geohydrological model calculates e.g. equipotentials, streamlines, boundary fluxes and flow velocity vectors.

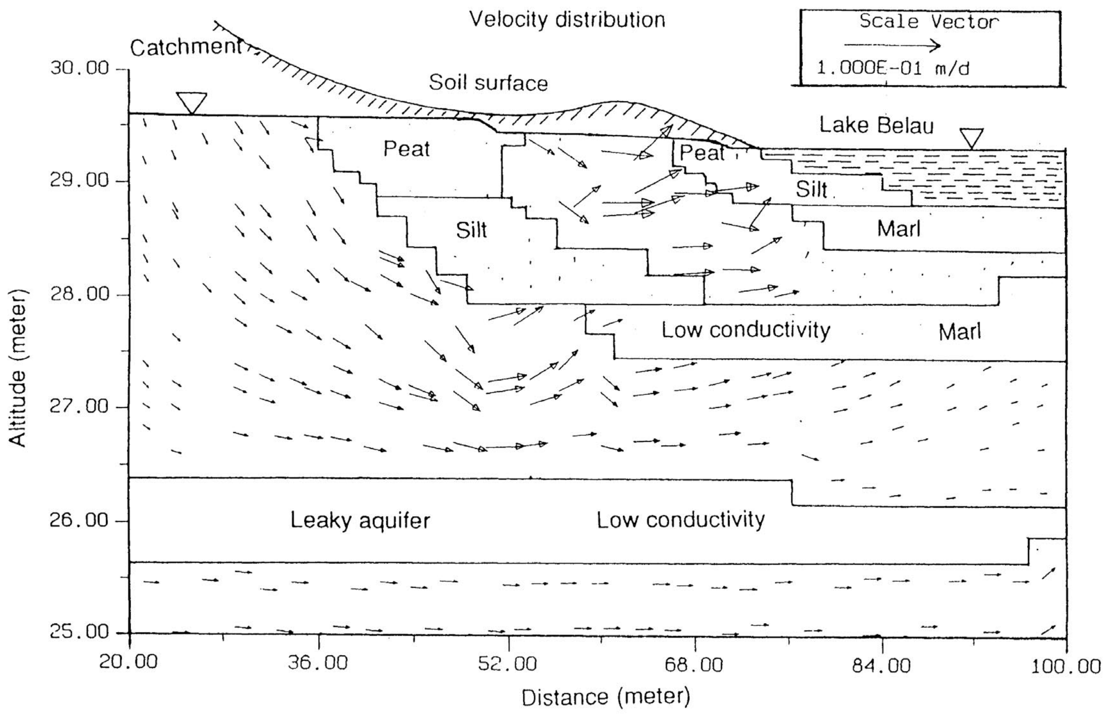

Fig. 9: Vertical cross-section of ground water velocity distribution in an alder peatland simulated with FLOTRANS (KLUGE et al. 1994).

Application example

Geohydrological models have been applied e.g. in a riparian wetland in order to study interactions between terrestrial, semiterrestrial and aquatic ecosystems (Figure 9)(KLUGE et al. 1994).

SCHOT and MOLENAAR (1992) have studied the influence of anthropogenic water management in a Dutch polder landscape during the last 400 years on groundwater flow pattern and groundwater quality with this modelling approach.

6.2.2 Water budget calculation with FEUWAnet

Model objective

Riparian wetlands are the environments placed between uplands and open water recipients (lakes, rivers,...). They are therefore typical interface systems: as such, they are characterised by the presence of processes characteristic of both systems. This fact confers to riparian wetlands the typical feature of high spatial and temporal heterogeneity (VANEK 1996; GOLD & KELLOG 1996) with the presence of different time scales which span over a wide range (FRÄNZLE & KLUGE 1996). A detailed model review has shown how present hydrologic models are not able to represent the complete array of the hydrologic processes in riparian areas (see DALL'O' et al. in prep.). Hence, to cover this gap, a new model, FEUWAnet, has been devised over the last decade and definitively developed in C++ language with Windows (95/98/NT) interface in the frame of the WET project (DALL'O' et al. in prep.). FEUWAnet allows the calculation of water balances and lateral water exchange in a transect parallel to the line of flow between the catchment area and the receiving water body.

Model description

The spatial boundaries of the model are represented in Fig. 9. The temporal range spans from days to months. To cope with the spatial complexity of riparian systems, structures are aggregated by means of a particular mathematical conceptualisation: the box concept. Hence the modelled transect is discretised in a number of boxes dependent on its geohydraulic structure. The boxes are then linked by means of a network of hydraulic resistances which reflects the path concept at a lower scale. The topology of the network consists of an upper and a lower layer of hydraulic resistances: this organisation is due to the need to represent both shallow and the immediately underlying groundwater lateral flow. The hydrologic processes taken into account in FEUWAnet are the (water) exchanges between the riparian wetland and: a) the catchment area; b) the receiving water body; c) the riparian wetland and the atmosphere; moreover, water exchanges between boxes and between the two vertical layers are taken into account. The relationships between the riparian wetland and the atmosphere are represented in each box by three state variables which model water content in the interception, snow and litter layer. The aim of the processes depicted in these layers is to yield the net recharge which enters the soil. In the soil box the unsaturated as well as the saturated zone are taken into account: the water volume is linked by a quasi-stationary relationship with the drainable water content. Finally, in the soil all the fluxes (shallow and deep lateral flows, net recharge, vertical flow, runoff, overflooding) are summed or subtracted to give the water balance, the resulting state variable being the piezometric level. FEUWAnet comprises also a particular calibration procedure: in presence of additional information, i.e. the observed piezometric level for each box of the modelled transect, it is possible to calibrate automatically the unknown hydraulic resistances of the network by using the Simplex Algorithm. Before calibration, however, two procedures specially developed for FEUWAnet, give reasonable (i.e. with physical meaning) initial values for the involved resistances.

Input data

Boundaries conditions are given by the groundwater level in the catchment area and by water level in the receiving open water system, by the potential evapotranspiration and by precipitation. Forcing functions are given by the LAI (leaf area index) of vegetation present along the modelled transect, by air temperature and by sunshine duration if evapotranspiration is calculated internally. Additional information for the optimisation procedure is represented by the time series of piezometric levels along the modelled transect; piezometers coincide approximately with the centre of each box.

Output data

All the state variables in each box, water levels in the interception, litter, snow layer and piezometric level, and the fluxes involved. The user friendly interface of FEUWAnet, inspired by the outlook of common spreadsheet, gives the possibility of easy pre and post-processing operations.

Limitations

The following assumptions build the structure of FEUWAnet:

- the set of boxes which discretise the transect is positioned upon a line connecting the catchment area to the receiving water body (transect concept).
- the prismatic box representing the whole riparian system is inferiorly closed by an aquiclude, therefore no exchanges take place with deep groundwater.
- the capillary water transport in the unsaturated/saturated soil zone is described in a peculiar way by a physical based quasi-stationary function and not by the Richard's equations. Therefore hysteresis effects are not modelled.
- evapotranspiration processes are modelled by means of empirical formulations, in opposition to energy driven models which would require intensive data (on hourly basis). The determination of reliable evaluations of evapotranspiration is an open issue in modelling wetlands.

Moreover, the set of differential non-linear equations which represents the mathematical structure of the model are solved numerically with the Euler method. FEUWAnet quantifies the hydrologic relationships along a riparian area; to know also the budget of pollutants, e.g. phosphorus or nitrogen, further models must be developed and linked to FEUWAnet.

Application example

FEUWAnet has been successfully applied to two riparian wetlands: an alder and a pasture wetland respectively (DALL'O' & KLUGE in prep.), both situated along the shore of Lake Belau, northern Germany. The input database has been originated form the Bornhöved Lake Project (Ecology Centre Kiel, Germany) and consists of time series which spans over ten years. The extreme variability of meteorological conditions in this period has demonstrated the robustness of FEUWAnet. Result of application show how FEUWAnet is able to replicate both the spatial and temporal heterogeneity present in the transects. This is realised by: a) the good accordance of predicted to observed values of piezometric levels (see Fig. 10); the good accordance of calculated water balance to previously estimated values of this quantity; c) the realistic values of unknown hydraulic resistances. This allows FEUWAnet to be considered a serious tool for different management options (scheduling and optimisation of experiments, determination of hydrologic relationships, ...).

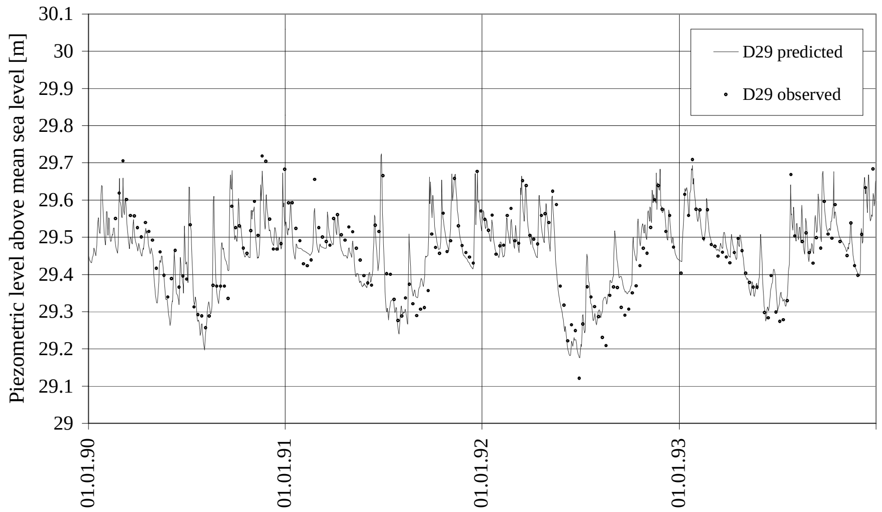

Fig. 10: Observed and predicted values in the box D_29 of the alder wetland transect (discretised with four boxes) in the years 1990-1993 (DALL'O' & KLUGE in prep.).

6.3 Water and nitrogen process based, dynamic model systems

The water and nitrogen dynamic of wetland ecosystems is characterised by complex spatio-temporal interactions between soil physical and chemical properties, microbial activities, species abundance, composition of vegetation types, hydrological and climatic conditions. Anthropogenic disturbance e.g. drainage, fertilisation, mowing or grazing influence the structure and functioning of ecosystems and the interactions between them up to the landscape scale are also characterised.

These complex interactions can either be quantified by establishing a long-term measuring programme for all important transformation processes and pathways, or by applying a validated, dynamic model system for the water and nitrogen transformation. The model application is, compared with long-term measurements, less expensive, and allows the identification of regulating factors in the water and nitrogen dynamics and the prediction of the effect of possible land use changes on the nitrogen balance (WALI et al. 1999). Activities to couple process based model for water and nutrient fluxes have developed powerful tools for the catchment analysis with a high spatio-temporal resolution (REICHE 1994; ARHEIMER & BRANDT 1998). However, these very complex, dynamic spatially distributed models are very data intensive and their present stage of development restricts their use mainly to research applications (JOLANKAI et al. 1999).

Model Objective

The model system described here briefly named ‚DILAMO' (Digital Landscape Analysis and Modelling') combines several tools for digital landscape analysis with a process based dynamic water and nitrogen model. The model simulates all relevant processes in the nitrogen cycle and is applied e.g. to analyse the effect of land use changes like reduction of fertiliser application, rewetting or changes of land use type on the water and nitrogen fluxes (REICHE et al. 1999).

Limitations

The parameter acquisition for GIS coupled, process based models can be solved only with simplification by using available digital maps. These data represent classified knowledge in form of soil or land use types, they do not reflect the specific, spatial variability of the input data. The complexity of dynamic model systems coupled to a GIS makes it also difficult to validate the simulation results (TREPEL et al. 2000). The integrated water and nitrogen model was developed for terrestrial ecosystems and has been successfully validated in semiaquatic ecosystems (TREPEL 2000), but still important processes in wetlands (e.g. lateral water flow, flooding) are described in a simplified way.

Model Description

The water and nitrogen model WASMOD is an integrated part of DILAMO and calculates water fluxes as well as carbon and nitrogen transformations of ecosystems dynamically with a high spatiotemporal resolution. The model is described in detail by REICHE (1994). The primary spatial units are single plots (ecosystems) characterised by the same vegetation, land use, soil properties and hydrological conditions. The soil profile is vertically segmented into 15 soil layers representing the unsaturated and saturated soil zone. The model can be applied for single plots or connected to a Geographical Information System for entire (sub)catchment areas. WASMOD consists of several submodels for transformation and transport processes involving water, heat, carbon, and nitrogen. For some of the processes, theories and mechanisms are well understood while for other processes existing knowledge is still limited. Processes in the water regime include daily measured precipitation as input, interception by plant canopy, infiltration into the upper soil layer, surface runoff, if infiltration capacity is exceeded, evapotranspiration and seepage. The vertical movement of water in the soil profile is solved by means of a numerical solution for the Richards equation. The soil mineral nitrogen processes are coupled with the carbon translocation and turn over processes according to HANSEN et al. (1990). Organic matter turnover is modelled by dividing the organic matter conceptually into three main pools: added organic matter of plant residues and manure

(AOM), soil organic matter (SOM) and soil microbial biomass (SMB). Each pool of organic matter is divided into two subpools characterised by particular C/N ratio and turnover times resulting in high and low mineralisation rates. The soil mineral processes include mineralisation of organic nitrogen, nitrification, immobilisation, denitrification, plant uptake by roots and vertical movement of nitrate and ammonia in the soil. Denitrification is simulated by defining a potential denitrification rate assumed to be related to the carbon dioxide evolution rate in the soil and the soil temperature. The vegetation is conceptually treated as vegetation type with characteristic annual biomass growth and decay rates, and specific carbon and nitrogen concentrations in the above and belowground biomass. The agricultural system management allows various management options, for example for fertilisation or harvest

Input data

The model input data are: soil physical and chemical properties, daily climate data, hydrological and topological site conditions, land use information, nutrient uptake capacity of different vegetation types. The DILAMO model system evaluates available spatial data (e.g. digital elevation model, digital land use map, soil profile description) and creates the input file for the Water and Nitrogen Model automatically.

Output data

The model output at the plot scale are time series with a user dependent variable time step from day to week for transformation and transport processes in the water and nitrogen dynamics as well as annual balances for all simulated processes in the water, nitrogen and carbon budget. At the mesoscale, only these last annual process rates are given as output for each spatial unit.

Examples

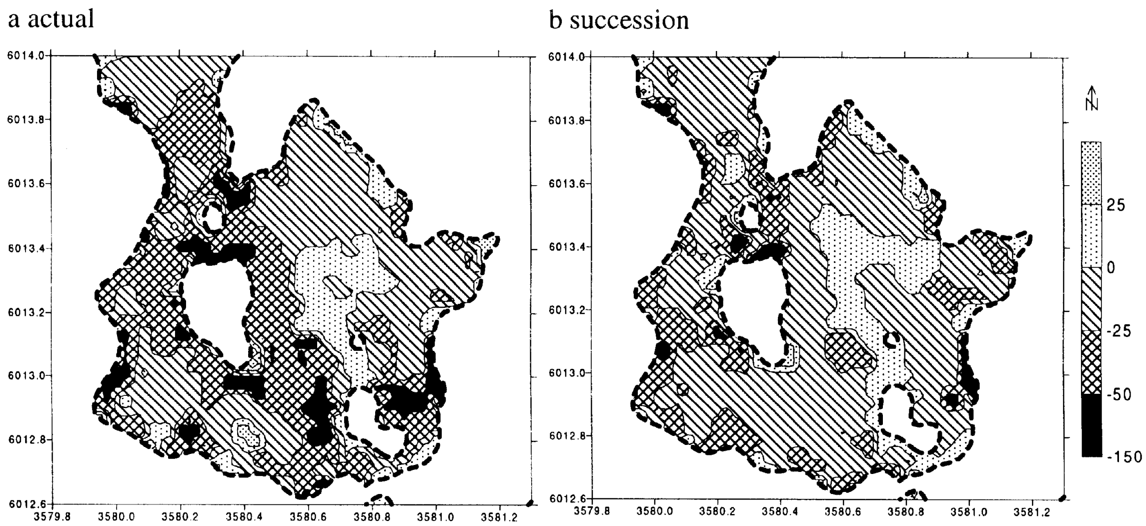

Fig. 11: Simulated nitrogen budget for minerotrophic peatsoils of the Pohnsdorfer Stauung; a = actual hydrological and land use conditions; b = succession and natural (rewetted) hydrological conditions. Mean annual nitrogen accumulation or deficit in [kg N ha^{-1} a^{-1}] for the simulation period: 10.'88 to 9.'97. (Source: TREPEL et al. 2000)

The model system has been successfully applied in a wide range of catchment analysis (e.g. KETELSEN et al. 1999, SCHIMMING et al. 1995; SCHIMMING et al. 2000; REICHE 1994) The effect of different land use types and land use changes on the nitrogen dynamics of minerotrophic peatlands were analysed by TREPEL (1999) and TREPEL et al. (2000) (figure 11).

6.4 Biological interactions in wetlands

6.4.1 Vegetation succession

Vegetation succession in wetlands is generally managed (1) to optimise plant productivity or retention and accumulation of ecosystems, (2) to prevent or control vegetation changes after changes in site conditions, (3) to restore former vegetation after wetland degradation or (4) to establish and stabilise vegetation in artificial (re)created wetlands. The aspects of vegetation succession under consideration may differ substantially with the spatiotemporal scale. The role of vegetation structure, plant functional types and plant productivity is often studied on long-term and global to regional scales. Management on short-term and local scales deals with the succession of plant communities, and plant populations with respect to properties like species growth, competition and dispersal, relation of plant species and communities with site conditions or the role in nutrient, water and carbon balance (PENNING DE FRIES 1983).

Succession modelling may be a useful tool at all steps of wetland management: (1) for a status quo-analysis or a functional analysis of the ecosystem, (2) during the process of finding management objectives, (3) to optimise concepts of vegetation monitoring and (4) for planning and evaluation of management activities. To analyse structures and simulate the development of vegetation in space and time, the partial or complete integration of succession models in geographical information systems (GIS) is opening up a lot of opportunities (RICHTER et al. 1997; DUTTMANN 1999). GIS-modelling is therefore a fast growing branch in vegetation science with a special emphasis on the prediction of vegetation changes. This chapter will give some brief examples on spatiotemporal model approaches with respect to the development of species composition and plant community change on local to regional scales.

Approaches of succession modelling

Models can be classified in many ways. The common separation between dynamic and steady-state (static) models may be of minor value in succession modelling as succession models are in the broader sense always “dynamic” since at least two different temporal stages of an ecosystem are related. Nevertheless real dynamic models which run a lot of time steps and contain at least one driving variable (which is a function of time) give a more realistic picture of ecosystem processes (PENNING DE VRIES 1983; JOHNSTON 1998).

Another model classification distinguishes between mechanistic models and empirical-statistical models (OLDE-VENTERING & WASSEN 1997; JOHNSTON 1998). Mechanistic models (also called deterministic, cause-effect- or functional models) describe ecosystem processes in mathematical terms. In general mechanistic models require detailed field measurements for each type of ecosystem under investigation. Empirical-statistical succession models (also called stochastic or correlative models) are based on correlations between abiotic variables and vegetation and contain one or more random variables. They are based on expert-knowledge or "rules" e.g. the indication values after ELLENBERG et al. (1992) or may be regression models based on field measurements and statistics. Many succession models however combine elements of both types (OLDE VENTERING & WASSEN 1997; BELDE & RICHTER 1997).

Empirical-statistical models

Empirical-statistical (response) succession models predict the probability of occurrence of plants or plant communities in a final succession stage (new equilibrium) as a response to habitat change. Linking only two time steps these models are less dynamic then mechanistic models. OLDE VENTERING & WASSEN (1997) give an overview of vegetation response models. The authors compared six models developed in the Netherlands which predict vegetation response to hydroecological changes. Expert-knowledge-based models like DEMNAT-2 (WITTE & al. 1992) use classified variables like species cover classes, pH classes, moisture classes etc. They are therefore less sensible than logistic regression models like ICHORS or ITORS which run with continuous scaled variables. Because most response models are based on indicative properties of plant species their use on greater spatial scales is limited (OLDE VENTERING & WASSEN 1997).

TONER & KEDDY (1997) developed a spatial-explicit logistic regression model which describes the probability of tree occurrence along riparian wetlands as a function of duration and time of flooding using tree species presence and hydrological data. As output for conservation purposes the model identified key predictors for wood occurrence and underlined the importance of certain hydrological parameters. Further examples of logistic regression succession models for wetland vegetation give VAN DE RIJT et al. (1996) and DE SWART et al. (1994). A possibility to run expert-knowledge models with incomplete informations about ecosystem processes is offered by fuzzy-logic approaches (RICHTER et al. 1997).

Predictions of final succession stages from empirical-statistical models will often be far from realistic and not easy to interpret (ERTSEN 1998). This may be due to mathematical errors (ERTSEN 1998) or because many regression models lack the initial vegetation as explanatory variable. In addition some environmental factors like life strategy characteristics of key species and their interactions during the process of succession are not well understood or not considered (DE SWART et al. 1994; RICHTER et al. 1997; TIMMERMANN 1999a). Furthermore the progress of succession in some wetlands never reaches an estimated equilibrium and can remain for decades in an intermediate state (TIMMERMANN 1999b).

Cellular automata models

Cellular automata (CA) represent the mechanistic model approach. A cellular automaton model can simulate spatial relations and developments with respect to neighbourhood relations. It consists of quadratic shaped cells within a regular grid, which states depend on the state of neighbour cells at the previous time step t-1. Changes take place within discrete time steps and simultaneously using the same set of transition rules (rate variables) for each cell. The theory of CA is well suited for succession modelling with a raster-based GIS because the cells can be used as CA units (BELDE & RICHTER 1997; WEBER et al. 1999).

The modelling procedure is divided in three steps: (1) definition of the initial state of each cell with respect to all variables, (2) time step t+1 with change of driving variables, (3) calculation of new states of each variable using a set of transition rules and related to both the cell itself and its neighbour cells. The transition rules have the form

$$a_{t+1}^{s} = f(a_t^{s-r},..., a_t^{s},... a_{t+1}^{s+r})$$

where a_t is state of cell s at time t; r is range of the neighbourhood of cell s; and f is local transition function, representing the transition rules.

Cellular automata are useful for succession modelling at all spatiotemporal scales (JOHNSTON 1998). Nevertheless they are especially suited to simulate small scale population dynamics combining intrinsic factors (life history traits of key species) and extrinsic (environmental) factors. BELDE and RICHTER (1997) give an example of CA adaptation to predict the succession in a wet grassland community after rewetting under different management scenarios. The model works with expert-knowledge from a set of 13 dominant plant species in a fen area of 140x98 m² (840 grid cells of 16 m² each). For different combinations of species with certain cover values several succession stages were calculated over a time span of 30 years. For each cell the following field data input was required: species cover, mowing regime and flood duration. BALTZER et al. (1998) give an example for small scale succession modelling of tree lawn plant species by combining a discrete-time Markov chain with a CA.

Concluding remarks

Succession models are still far from giving a realistic picture of the world. They have to face some problems and limitations which seem to be inherent and will therefore never solved completely: (1) the adoption of succession models on different ecosystems and plant-geographical regions is difficult due to changing ecological and physiological optima of plants, (2) single stochastic events like catastrophes (floods, droughts) and chance dispersal are not easy to model but may have essential effects for succession (VAN DE RIJT et al. 1996) and (3) also hysteresis effects occur which can not easily be reflected by stable transition rules of the model (DE SWART et al. 1994; BALTZER et al. 1998).

Predictive succession models should try to integrate most factors which determine vegetation development and emphasise the floristic composition at the initial succession stage. For future model development a combination of mechanistic elements, expert-knowledge and empirical-statistical relations seems to be most promising to come to more realistic predictions of vegetation succession (OLFF et al. 1995; OLDE-VENTERING & WASSEN 1997).

6.4.2 Trophic networks

Food chains do not occur as isolated units in an ecosystem but are connected to each other. These interconnected patterns are generally named ‘food web’ or more recently ‘trophic network’. In complex communities all organisms obtaining their food from the same element of the food chain belong to the same trophic level. This is a functional and not a taxonomic classification. According to the origin of assimilated energy a population can belong to several trophic levels.

Model Objectives

The model software described here briefly named ‘ECOPATH’ is designed to help the user to construct a (simple or complex) model of the trophic flows in an ecosystem (CHRISTENSEN & PAULY 1992a,b). The approach was initially applied to marine and freshwater ecosystems, but it has also been successfully applied to semiaquatic and terrestrial systems (e.g. DALSGAARD et al. 1995).

Limitations

- In ECOPATH the time scale is included only indirectly, therefore it is a static and not a dynamic model.
- It assumes the system to be in a steady state although it can accept accumulation and depletion of biomasses.
- Organic components are included into the model only as living or dead (detritus).
- Abiotic effects such as nutrient uptake by primary producers are not considered.
- The software can deal with a maximum of 50 compartments.

Model Description

ECOPATH (for Windows and predecessors) is both an approach for constructing static ecosystem models and a Public domain PC software. It includes routines for balancing of flows in an ecosystem and for estimating indices for ecosystem characterisation. These indices are primarily estimated from network analysis.

(1) Production + import = predation mortality + fishing mortality + other mortality + migration + biomass accumulation

forms the basic equation of the modelling approach. It assumes mass-balance, i.e. it balances the flow to and from each ecological group or compartment in the model. The predation mortality term is used to link predator and prey species, whereas

(2) Consumption = production + non-assimilated food + respiration.

A detritus compartment D receives flows originating from "other mortality M" and "non-assimilated food NA", so that D = M + NA.

The model can accept accumulation and depletion of biomasses during the time period modelled despite of the steady state assumption,. Thus, biomass accumulation or depletion rates can be quantified.

The trophic network is constructed based on the concept of trophic levels. Two different types of trophic levels are applied: 1) A fractional trophic level is assigned to each functional group to determine its position within the food web as suggested by ODUM and HEALD (1975) based on the quantitative composition of its diet: a trophic level of 1 is assigned to primary producers and detritus and a trophic level of 1 + the weighted average of the preys' trophic levels to consumers. 2) Prior to calculating quantity and efficiency of transfer of matter between trophic levels these fractional trophic levels are reversed by an approach suggested by ULANOWICZ (1995) into discrete trophic levels sensu LINDEMAN (1942).

Consumption is calculated by starting from top predators and subsequently proceeding down the trophic chain. The intake of each predator, coupled with a diet composition estimate, determines the grazing quote on the lower level.

The model software can be obtained from ICLARM (International Center of Living Aquatic Resources Management), Manila, Philippines or can be downloaded at `http://www.ecopath.org`

Input data

- A broad range of currencies can be applied, e.g. wet weight, dry weight, carbon, nitrogen, phosphorus, energy.
- Time scale is chosen freely by the user.
- For each living group, the following parameters are needed as inputs: biomass (B), production/biomass ratio (P/B), consumption/biomass ratio (Q/B). Gross efficiency rates (GE = production/consumption) are needed in cases where no estimate is available for either P/B or Q/B. Additionally, a diet composition estimate (DC, in percentages of volume or weight of food items), an estimate of the percentage of food that is not assimilated (NA), and the amount exported from the system by migration (E), are required as inputs for each ecological group. An additional parameter, usually ecotrophic efficiency (EE = predation mortality expressed as percentage of production), is then calculated using a set of linear equations. If known for a compartment, EE can also be entered and another unknown parameter (e.g. B) can be estimated.
- Primary producers are not classified as consumers. Therefore, these groups have no consumption term and do not appear as consumers in the diet matrix.

Outputs

- Based on the assumption of mass-balance, the model calculates in absolute numbers the following parameters for each compartment: biomass, accumulated/depleted biomass (BA), unassimilated food, flow to detritus, predation mortality (P*EE), respiration (R), assimilated food (A), food intake.
- It gives furthermore for each compartment the relationship R/A, P/R, R/B, the fractional trophic level, an omnivory index, a niche overlap index, a selection index, mortality coefficients.
- For the entire system the following summary statistics and indices are calculated: total throughput (total E + R + flow to detritus), net P, primary P/B, R/B, B/catches, efficiency of the fishery, connectance index, omnivory index, ascendency/capacity/overheads, cycling index.
- Mixed trophic impacts (assessment of the direct and indirect effects that changes of biomass of a group will have on the biomass of the other groups in a system).
- Primary P required to sustain harvest from the system.
- Ecological footprint.

Examples

To date a series of application examples have been published. The monograph 'Trophic models of aquatic ecosystems' (CHRISTENSEN & PAULY 1993) contains a worldwide collection of application examples for – amongst others - culture systems, lakes, rivers, and coastal areas including lagoons.

Besides a long list of published food webs using the ECOPATH software, two recent applications successfully applied the model to partial (or sub-) systems of a larger ecosystem complex. The first one treated the trophic web of a *Phragmites* covered littoral zone of a northern German lake (POEPPERL & OPITZ in prep.). The second application treated a shallow water area in the northern part of the lagoon of Venice based on data of a single summer season (Fig. 11) (CARRER & OPITZ 1999).

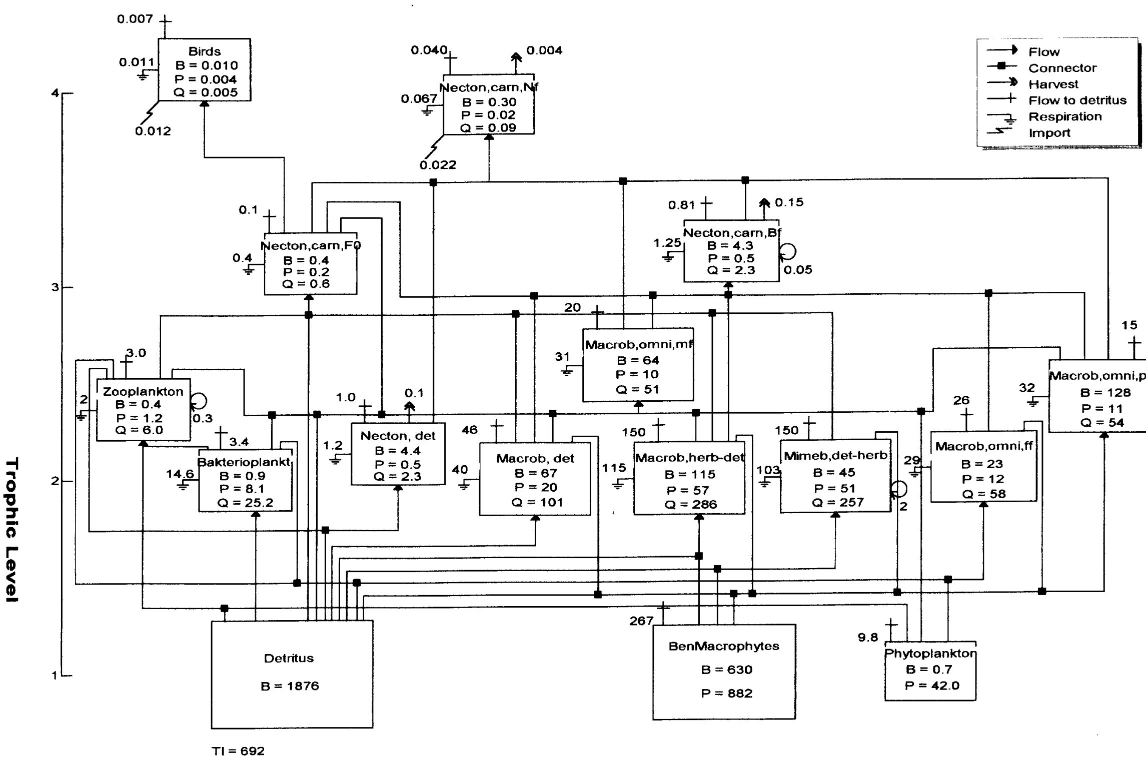

Fig. 12: Quantitative representation of trophic interactions within the food web of Palude della Rosa, Lagoon of Venice, during summer 1994. The area of each box is proportional to the logarithm of the biomass (B, in kcal m-2) of each group. Flows are in kcal m^{-2} $month^{-1}$. Q is total flow entering a compartment and P is the production of a compartment. (carn, carnivorous; det, detrivorous; herb, herbivorous; omni, omnivorous; prd, predators; Bf, benthic feeders; ff, filter feeders; Nf, nekton feeders; mf, mixed feeders). (from: CARRER & OPITZ 1999)

7 Model sources on the internet

The Internet is an excellent source for further information about models and their application to wetlands. The following annotated link list is a start for further searches.

WWW-Server for Ecological Modelling: This WWW-server provides easy access to available information about ecological modelling: A register and documentation system of ecological models: REM and ECOBAS, simulation-software and links, data sources and other information about modelling. It is also designed for modellers who want to make their models easily available. In addition, this WWW-Server integrates an interface to ECOBAS (Documentation of mathematical formulation of ecological processes). [http://dino.wiz.uni-kassel.de/ecobas.html]

GEOECODATA: This is a collection of Internet Data Resources for Geoecology and Ecological Modelling including Soil Science & Soil Physics Data Sources, General Earth Science Data Sources, Global Climate Data, GIS Data, Agriculture, Land Use and Water Specific Data.
[http://dino.wiz.uni-kassel.de/geoecodata/geoecodata.html]

CAMASE - Guidelines for modelling: The need for guidelines for modelling has been expressed several times, particularly by those out of the main stream of developments. This www-site present a first draft of guidelines for 'validation', 'sensitivity and uncertainty analysis' and 'calibration'. These topics are preceded by some relevant definitions and illustrated with examples and a reference list.
[http://www.bib.wau.nl/camase/modguide.html]

Environmental Model Library: Here is a list of Environmental/Hydrologic/Hydraulic/Water Quality/Water Resources computer models compiled and archived in the Civil & Environmental Engineering Department, Old Dominon University. All models are public domain, and have been compiled from various sources such as USEPA, USACE WES, USDA-ARS, etc.
[http://www.cee.odu.edu/cee/model/]

US EPA: Models and Model links: The US Environmental Protection Agency has provided an intensive link list about different environmental models.
[http://www.epa.gov/epahome/models.htm]

WES Environmental Models: The US Army Corps of Engineers have developed several environmental models for hydrology and hydrochemistry. Most of them are public domain software and have a user friendly documentation.
[http://www.wes.army.mil/el/elmodels/index.html]

RAMSAR: The RAMSAR server is an excellent entry for world wide wetland conservation and policy and offers many key documents and tools for wetland planning and management.
[http://www.ramsar.org/]

8 Discussion

The previous sections of this paper present several models and model approaches for wetland planning, design and management. The model descriptions are written in a heterogeneous way, reflecting different complexity, spatiotemporal working scale of the models and also different scientific interests of the model users. Table 3 summarises the application range and the availability of the presented tools for end users in environmental authorities.

The results in Table 3 indicate that not all models are readily applicable for end users in environmental authorities due to several reasons. Some good models, for example PolFlow or the wetland score system (DE WIT 1999; PALMERI & TREPEL in prep.) were recently successfully applied for scientific purposes. However, in these cases are no software tools available. A future user has to create the model – based on the scientific documentation - for him/herself again. The only available software for wetland planning described in this report is the nutrient retention model PREWET.

In hydrology, several commercial models are available for analysis of water flow pattern in surface flow wetlands (e.g. 2-D / 3-D Computational Fluid Dynamics) or groundwater flow wetlands (e.g. FLOTRANS; MODFLOW, etc.). The application of these tools requires a good training in hydrology and the application of models.

The application of GIS based models for wetland planning e.g. DomFlow (PIETERSE et al. in prep), PolFlow (DE WIT 1999) or the wetland score system (PALMERI & TREPEL in prep., TREPEL & PALMERI in prep.) are restricted through the availability of spatial data. The quality of the simulation results is related with the spatial resolution of the input data and the degree of how detailed informations (logical resolution) are classified (TREPEL et al. 2000).

Tab. 3: Assessment of the presented models according their availability for end users in environmental planning and their possible application range. Docu = documentation a = available, i.p. = in prep; Prog = Program available; Comp = complexity; Avail = availability; Application range: P = planning; D = design; M = management; capitals letter indicate main application range; small letters indicate minor application range.

Model	Docu	Prog	Comp	Avail	Application Range		
DomFlow	a	yes	medium	scientific	P	d	
PolFlow	a	yes	medium	scientific	P	d	
Site & Size	a	yes	simple	scientific	P	d	
Prewet	a	yes	simple	free	P	d	
CFD in 2-D / 3-D	a	yes	difficult	comm		D	M
WASP	a	yes	difficult	free		d	M
MODFLOW/MT3D	a	yes	medium	comm	P	D	
FLONET/TRANS	a	yes	simple	comm	P	D	m
FEUWAnet	a	yes	medium	scientific		d	M
succession		no	difficult	scientific			M
ECOPATH	a	yes	medium	free			M
DILAMO	a	yes	difficult	scientific	P	d	M

The majority of models suggested for application during the management stage are mainly scientific tools. Even if the software for WASP or DILAMO is available, the user will have to spend much time to learn how to handle the model. The ECOPATH software is a good example, how biological knowledge can be integrated in a software and how model developers discuss limitations of the model to avoid misuse and disappointments of future users (compare e.g.: Can Ecopath mass balance assessments provide information directly usable for policy analysis? in: CHRISTENSEN & WALTERS 2000). The section on vegetation succession models introduces a variety of simulation approaches. All tools however seem to be developed for purely scientific purposes.

The discrepancy between scientific models and practical models for environmental planning has been recognised earlier (e.g. RECKHOW 1994; KAMP-NIELSEN 1997; WALTERS & KORMAN 1999; ZÖLITZ-MÖLLER 1999) and is caused by the fundamental differences between these two user groups. In science, many modellers spend much time and effort in the development of very specialised, and complex models, which are then applied in a few case studies only. Even, when a model has been applied in several case studies for example the water and nitrogen simulation model WASMOD in the DILAMO modelling system, the model software is still not very user-friendly. The data handling restricts the application of such complex tools to well educated computer specialists. In science, there is due to financial constraints no time reserved for the development of a user friendly interface for the software. The environmental authorities at least in Europe have not spent much effort in the development of user friendly software tools compared to the US Environmental Protection Agency which offers a wide range of different models for download.

The scientific application of both simple, steady-state models and complex, dynamic models has been a fruitful way of identifying the relative importance of ecologically important mechanisms and enables us to learn more about the complex interactions in wetlands. The transfer of some of these valuable scientific models into user-friendly environmental software tools will help to design and manage wetlands in a more efficient way.

9 Acknowledgements

These guidelines were written during the last semester of the European funded WET project (TMR Research Network WET; Contract-Number: ERBFMRX-CT960051). The idea of transdisciplinary networking allowed us to see wetlands from different scientific viewpoints. We experienced that communication between hydrologists, ecologists, engineers and geographers is an effective way to increase our knowledge about the complex interactions in wetlands.

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