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Modelling the nitrate retention capacity of a restored riparian wetland

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Master's thesis for M.Sc. Water Science and Management, track Earth Sciences

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Preface

This thesis has been submitted in partial fulfillment of requirements for the degree of Master of Science at the Department of Earth Sciences, Faculty of Geosciences, Utrecht University, the Netherlands.

The work was carried out at the Water Resources Department of DHI from January to August 2015 under the academic supervision of Professor Marc Bierkens. Dr Mike Butts, Head of Innovation at the Water Resources Department of DHI, acted as external supervisor.

This thesis would not have been possible without the support of numerous people.

I would like to express my deep gratitude to my research supervisors Mike Butts and Marc Bierkens. Mike for always taking his time and his support on academic, practical, and personal level, his patient guidance and enthusiastic encouragement and useful critiques of my work. Marc for his valuable support along the way, his useful and constructive suggestions for improving my work, his help with planning, and for making this all possible by introducing me to DHI.

I would like to express my very great appreciation to Birgitte von Christierson, for offering her valuable input at all times, her approachability for any and all problems I encountered, and her thorough previous research which provided a very solid basis for my research project. Her willingness to give her time so generously and her support in this thesis process have been very much appreciated.

I would like to thank Prof. B. Kronvang and J. Poulsen, Department of Bioscience, Aarhus University and Prof. Peter K. Engesgaard, Department of Geosciences and Natural Resource Management, University of Copenhagen for providing data support. Data from Jensen et al. (2015) was acquired during the EU FP7 AQUAREHAB project as collaboration mainly between University of Copenhagen and Geological Survey of Denmark and Greenland.

My grateful thanks are also extended to Flemming Thorbjørn Hansen for his greatly valuable work and advice on the ECO Lab templates, and to Jannick Kolbjørn Jensen for the extensive and high quality fieldwork data. I wish to thank all of the colleagues of the DHI Water Resources Department for welcoming me as an intern. It has been an absolute pleasure to be part of such a friendly and diverse 'water geek' environment where everyone is passionate about modelling.

A special thanks to my partner Robin for always supporting me and his patience and encouragement along the way. Finally, I wish to thank my parents for their support and encouragement throughout my studies.

Laura Nieuwenhoven, Lund, August 2015

Summary

Riparian wetlands provide beneficial functions to the wider environment. They intercept surface as well as sub-surface flows and through water storage and retention serve as nutrient buffer zones for the river system. Eutrophication of surface waters is recognised as one of the most important challenges to achieving ‘good ecological status’ in Denmark. Implementing the Water Framework Directive (WFD) entails a significant reduction of nitrate loading in order to preserve and enhance the ecological status of water bodies. The restoration of streams and floodplains forms part of important strategy to reduce nutrient loadings. This study focuses on developing a model capable of predicting the nitrate reduction capacity of reconstructed wetlands using a dynamic ecological modelling tool ECO Lab, coupled with a detailed dynamic physically based numerical flow and transport model MIKE SHE. The restored riparian wetland Brynemade located next to Odense River on Funen Island in Denmark was modelled over seven years and the model outcomes were compared with field measurements. To quantify the impact of the restoration on nitrate reduction, a new model of the wetland under pre-restoration conditions was developed based on the calibrated model. By quantifying the nitrogen load reduction, the effectiveness of restoring riparian wetlands to achieve water quality objectives is assessed.

The results show that in the current wetland, denitrification patterns in the overland flow compartment follow river flood patterns, which indicate that most nitrate input originates from the river. The pre-restoration wetland receives fewer floods and has groundwater drainage, which results in a larger unsaturated zone and thus more denitrification compared to the restored situation, as well as less denitrification in the overland flow compartment. Denitrification in the saturated zone occurs mainly at the eastern boundary of the wetland, where the agricultural nitrate inflow occurs, as well as along the river channel, where nitrate seeps through the river bed to the saturated zone. The peat layer is the major contributor to total denitrification in the saturated zone. Most of the nitrate inflow from the adjacent agricultural fields is removed before it wells up to the surface.

Restoring a wetland to its natural state by restoring a river flood regime and removing drainage increases the nitrate retention capacity. Most of the extra denitrification capacity comes from the increase in floods and the subsequent denitrification in the overland flow compartment. In the pre-restoration wetland 391 kg N/year is removed, compared to 1287 kg N/year after restoration, which is close to a 230% increase. In addition, the nitrogen removal in the river doubled, from 0.23% to 0.47%. The post-restoration removal rates of 129 kg N/ha/year are within the typical range for a wetland of this type. However, since wetland restoration is such a space consuming measure, to reach WFD goals a mix of nitrate load reduction measures has to be implemented.

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1. Introduction

Riparian wetlands provide beneficial functions to the wider environment through their hydraulic connection to rivers and streams. They intercept surface as well as sub-surface flows and through water storage and retention serve as nutrient buffer zones for the river system. Riparian zones form a transition between terrestrial and aquatic ecosystems and often have a high level of biodiversity. They play a major role in water quality in watersheds, especially in low-order streams were the cumulative effects could have a large impact on downstream water quality (Rassam, Pagendam, & Hunter, 2008).

These characteristics mean that riparian wetlands have increasingly been included as an important element in river basin management strategies, for example within the River Danube and Tisza catchments (Fisher & Acreman, 2004). In Denmark, the restoration of streams and floodplains forms part of important strategy to reduce nutrient loadings. The four Danish Action Plans from 1985 to 1998 (Kronvang et al., 2005) aimed to transform 16,000 ha of farmland into wetland area and reduce nitrate loadings from agriculture by cutting N fertilizer quota by 10% in order to improve water quality and biodiversity of Danish water bodies. These plans have resulted in a decline of 50% of total land-based N-load to Danish coastal waters since 1990 (Windolf et al., 2012).

One example of a major restoration project is Odense River Basin, undertaken from 2002 to 2010 (Environment Centre Odense, 2007). The project was set up as one of the pilot projects for implementation of the European Water Framework Directive (WFD), which requires all surface and groundwater bodies to achieve “Good status” by 2015. It was funded by EU-LIFE Environment and the Danish Environmental Protection Agency and carried out by Funen County (later the Environment Centre Odense). Much of the Odense River has been assigned a more stringent objective than Good status, because of the occurrence of a number of species listed in Annex II of the Habitats Directive. The maximum nitrogen load to the Odense Fjord inlet while still permitting achievement of good ecological status in all parts of the fjord is estimated at 900 tonnes N. Taking the current nitrate load into account, this means one goal of the restoration project is to reduce the nitrate loadings to Odense Fjord by approximately 1200 tons per year (Environment Centre Odense, 2007).

To reduce nitrogen leaching in the Odense catchment, a number of measures were identified. These include higher utilization of the N content of manure, reduced N fertilization norms, increased area of catch crops, afforestation, and buffer zones around watercourses, as well as re-meandering of watercourses and re-establishment of wetlands (Kronvang et al., 2005).

This study focuses on modelling a restored riparian wetland along the Odense River, as well as modelling the wetland in its unrestored state. By quantifying the nitrogen load reduction, the effectiveness of restoring riparian wetlands to achieve water quality objectives is assessed. This chapter sketches the context in which this study was performed; why this study was done, its relevance, the main research question, and the body of research that predates this study. Chapter 2 gives the theoretical concepts behind the study and a description of the study area. Chapter 3 describes the approach taken to answer the research question. The model on which this study was based is described, as well as the range of adjustments and elaborations performed to make the model more suitable for representing the real-world situation. To quantify the impact of the restoration on nitrate reduction, a new model of the wetland under pre-restoration conditions was developed based on the calibrated model. The main results of the study are presented in chapter 4. In the discussion, the results and the model are analysed and recommendations for further research are given. The main conclusions are summarised in chapter 6.

1.1 Problem formulation

Societal challenge

Nutrient overload in soils and water bodies can lead to loss of biodiversity when faster growing species like grasses, reeds, and brambles take advantage of the higher nitrogen levels and overgrow the natural slower growing species. In aquatic environments, eutrophication can lead to toxic algal blooms. Implementing the Water Framework Directive entails a significant reduction of nitrate loading in order to preserve and enhance the ecological status of water bodies and terrestrial ecosystems. With its extensive coastline and intensive agricultural sector, eutrophication of surface waters due to diffuse pollution from agriculture is recognised as one of the most important challenges to achieving ‘good ecological status’ in Denmark. This study helps to identify how and to what degree restored wetlands can help reduce nitrate loading and thus reach the goals put forth in the WFD. In addition, wetland restoration is identified as a measure which has a substantial potential for cost-effective integration of climate change mitigation into nitrogen load reduction programmes (Kaspersen et al., 2015).

Scientific problem

It is important to include knowledge of the flow and transport processes in river and wetland restoration projects, both in order to predict the spatial and temporal flow and sediment deposition patterns and to assess the potential for nitrate removal on the riparian wetland and in the underlying aquifer. This knowledge helps to determine the potential of restored wetlands as measures to attain nitrogen load reduction goals. At this moment, few models include a linked physical representation of surface water flooding, unsaturated zone flow, and subsurface flow combined with nutrient processes. This study proposes a practical and innovative in its completeness model-based method to assess the potential of restored wetlands in reducing nutrient loadings. The project examines the nitrate retention capacity of the wetland Brynemade located next to Odense River on Funen Island in Denmark. The wetland is located in an environmental monitoring pilot basin used in a larger project on nitrate loadings called NOVANA which, if successful, can be upscaled to country level (Svendsen & Norup, 2005).

Stakes for DHI

DHI has developed an integrated surface water-groundwater hydrological model of a restored riparian wetland (Von Christierson et al., unpublished). By using the flow outputs from this model to run an ecological water quality model, the nitrate retention capacity of the wetland can be investigated. The outcome of this study will be valuable in understanding the flow and nutrient processes in riparian wetlands reconstruction in Denmark and abroad.

1.2 Literature review

The role of wetlands in nutrient loading reduction and water quality improvement has been studied extensively (Fisher & Acreman, 2004; Rassam et al., 2008). However, the effect of extensive alterations of the hydraulic interaction between streams and their floodplains and its effect on nutrient processes have received limited attention (Von Christierson et al., unpublished). For example, limited work has been undertaken to fully understand the implications of wetland reconstruction on water quality as most modelling studies have been focused mainly on the in-stream river hydraulics and the effect on habitats.

To obtain a full picture of the effects of wetland reconstruction, nutrient behaviour has to be taken into account.

Detailed modelling studies of the role of wetlands in nitrate retention have been conducted but few include an integrated model with surface water flooding, unsaturated zone flow, subsurface flow, as well as nutrient processes. Restrepo et al. (1998) describe the development of a numerical dynamic wetland flow module linked with the groundwater model MODFLOW but do not provide any means of calculating nitrate reductions, whereas Langergraber and Šimůnek (2005) have developed a multicomponent reactive transport model for constructed wetlands which only includes saturated flow. Rassam et al. (2008) have developed conceptual models for describing nitrate removal in riparian buffer zones using analytical mathematical functions, where nitrate attenuation capacity was restricted to potential denitrification only. Detailed models have previously been developed in which some processes are included, usually with a number of simplifications such as steady-state flow and simplified bank overtopping (Chavan & Dennett, 2008).

This study focuses on developing a model capable of predicting the nitrate reduction capacity of reconstructed wetlands using a dynamic ecological modelling tool ECO Lab (DHI, 2009), coupled with a detailed dynamic physically based numerical flow and transport model MIKE SHE (DHI, 2012; Graham & Butts, 2006). Coupled MIKE SHE-ECO Lab models have been used previously in studies to quantify the impacts of land management and water use on stream flow, temperatures and fish growth (Loinaz et al., 2013; Loinaz et al., 2014). The modelling approach developed here includes all flow processes in the saturated and unsaturated zones and overland flow linked with nitrate processes from the full terrestrial nitrogen cycle, in an integrated 3D dynamic environment. This comprehensive approach provides added descriptive value of the model over more simplified models.

1.3 Research objectives

The overall objective of the study is to understand and quantify the flow and nitrate retention processes in a restored riparian wetland, leading to an answer to the research question. This objective is achieved by:

- developing of an ecological water quality model linked to a 3D-integrated surface water-groundwater hydrological model,

- validating this model against observations of river flows and levels, groundwater levels and nitrate concentrations,

- estimating the removal of nitrate on the surface and in the peat and sand layers beneath the wetland during both wet and dry periods,

- estimating the impact of the restoration on nitrate retention by simulating the original situation with the unrestored wetland for comparison,

- understanding how the dynamic interaction of surface and subsurface flow affect flow, transport and nitrate retention in the restored wetland.

1.4 Research question

What are the important processes and characteristics that affect the nitrate retention capacity of a restored riparian wetland, how well are we able to represent the actual behaviour of such a wetland using numerical models and what is the impact of the wetland restoration on the nitrate retention capacity?

2. Theory and site description

2.1 Theoretic background

Nitrogen mainly occurs as an atmospheric gas. In the soil, in addition to the gas in occurring soil pore spaces, nitrogen occurs in both organic and inorganic forms. Organic forms of nitrogen make up a very high percentage of the total nitrogen found in the soil. However, plants are able to use only two inorganic forms of nitrogen: ammonium (NH_4^+) and nitrate (NO_3^-). Of these, nitrate is very mobile and can leach to groundwater and surface waters. In agriculture, when nitrogen is removed from soils by plant growth it needs to be replaced. For this reason as well as to increase crop yields, mineral fertilizers are applied. Nitrogen from mineral fertilizers is now the major source of nitrogen input to soils in EU countries (Pau Vall & Vidal, 1999). Nitrogen in commercial fertilizer is particularly soluble to facilitate uptake by crops. Excessive nitrogen application can pose a threat to the environment.

Nitrification and denitrification are major processes of the terrestrial part of the nitrate cycle. The conversion of ammonium and ammonia (NH_3) is primarily carried out by aerobic soil bacteria, which convert it to nitrite (NO_2^-) before oxidising it to nitrate. Denitrification is the process of nitrate reduction, carried out by soil bacteria. Denitrification generally proceeds through some combination of the following intermediate forms: $NO_3^- \rightarrow NO_2^- \rightarrow NO + N_2O \rightarrow N_2$ (g).

Wetlands are often considered as being effective at reducing nitrate loadings. This process is caused by encouraging sedimentation, adsorbing nutrients to sediments, plant uptake, and enhancing denitrification (Fisher & Acreman, 2004; Ballantine et al., 2014). Specifically, in natural wetlands the hyporheic zone (the surface water – groundwater interface adjacent to the stream channel) plays a major role in the nitrogen removal, via denitrification, anaerobic ammonia oxidation and plant uptake (Zhou et al., 2014).

The main nitrate removal processes in riparian wetlands are said to be denitrification (Chavan & Dennett, 2008; Hefting et al., 2013), plant uptake (Chavan & Dennett, 2008; Ballantine et al., 2014), and immobilization (Ballantine et al., 2014). The main ammonium removal processes identified by Chavan & Dennett (2008) are volatilization, nitrification, and plant uptake. They also identified mineralization (decomposition of organic N to ammonium) as another main process that takes place in the N cycle of riparian wetlands.

Of these processes, denitrification is the most important in wetlands, since the water-logged soils of wetlands provide anaerobic conditions optimal for denitrification (Hefting et al., 2013). Denitrification is primarily carried out by anaerobic bacteria, most of which are heterotrophic and use carbon as a source of energy. As such, the main limiting factors for denitrification in wetland soils include nitrate and carbon availability (organic carbon content), as well as low soil temperatures, pH and oxygen concentration (or degree of water-logging) (Fisher & Acreman, 2004; Rassam et al. 2008; Hefting et al. 2013; Ballantine et al., 2014; Song et al., 2014; Zhou et al., 2014). Other factors that were found to be of influence were retention time or flow rate of floods (Fisher & Acreman, 2004; Rassam et al., 2008), and vegetation, which provides organic carbon and microbial habitat and influences oxygen concentration (Fisher & Acreman, 2004; Song et al., 2014).

Normally, denitrification levels follow seasonal patterns, based on the fluctuating temperature and seasonal nitrate concentration differences from fertilizer usage and precipitation patterns (Fisher & Acreman, 2004; Song et al., 2014). Temperature is the main influence factor for nitrification and mineralisation (Chavan & Dennett, 2008).

2.2 Site description and data availability

The study site is a restored riparian floodplain of the Odense River on Funen Island in Denmark, at 55°13'12"N 10°17'33"E. The study site was chosen because of data availability, based on previous extensive research. The Brynemade wetland was restored in 2003 by re-meandering and reducing the flow capacity of the formerly straightened channel of the adjacent Tørringe Brook. The site had a fairly natural look at the project start. The channel bed level was on average raised by 1 m and the cross-sectional area was reduced by approx. 50%. The re-meandered channel has a length of approximately 6 km with 16 meanders (figure 1). The river valley is wide at this place and the bottom is relatively flat. The remeandered river encompasses a total of 125 ha riparian wetland area. The Brynemade wetland is 180 m in length, 200 m in width, and slopes 0.8 m from the margin of the riparian zone to the river. The upstream catchment area is 254 km^2 (Poulsen et al., 2014). The floodplain has been in use as permanent grazing meadow since the restoration. Along the river valley is a small pine tree nursery and behind that conventionally driven agriculture (Johnsen et al., 2011). The site is characterised by frequent flooding post restoration, both during the winter and during summer storms. The water level in the Odense River varies over 1 meter with the seasons. This leads to floods in Brynemade during winter (Johnsen et al., 2011).

The study area has a relatively simple geology, which consists of an upper layer of 1 m thick organic-rich peat and a lower layer of a sandy aquifer of a vertical thickness of 11-12 m (figure 2). Studies in the area found that the peat effectively separates surface water from groundwater. There are a limited number of spots in Brynemade where the groundwater seeps up through the grass. At very high water levels, the river water can possibly 'run behind' the peat and infiltrate adjacent to the fields. For the area as a whole these effects are however not very significant (Johnsen et al., 2011).

In the study area, a number of transects are present from previous studies. Jensen et al. (2014b) installed transects to record groundwater levels and nutrient concentrations, in order to examine the ability of the wetland to retain nitrate. Poulsen et al. (2014) conducted hydraulic measurements along a second set of transects including water depths and velocities. These measurements were used to study the deposition of sediment, organic matter and phosphorous in the wetland with a 2D dynamic river and floodplain model. The model predicts spatially and temporally changing zones of confluence on the floodplain due to variations in inundation depth.

Hydraulic conductivity (K) values are estimated to be about 10 m/d based on slug tests, which characterises the soil about 1 m under soil surface (Engesgaard & Nilsson, 2011). Jensen (2014)[1] developed a two-layer conceptual model of the area, with a contrast in vertical hydraulic conductivity of 80 between the overlying peat and the sand aquifer. The model included steady inflow of groundwater from the agricultural fields. He was able to model both observed flow patterns within the riparian zone and observed nitrate attenuation, with a calibrated first-order decay rate for nitrate reduction within the lower 10 m of the sand aquifer. The study showed that during flooding the groundwater-surface water

[1] Jensen (2015) is a revision of the papers included in the PhD thesis and is currently under revision.

interaction changes from that characterizing a groundwater-river system (no flooding) to that of a groundwater-lake system.

Based on measurements, Engesgaard and Nilsson (2011) estimated a residence time of 2500 days (6.8 years) for diffuse inflow, and a nitrate removal effectivity of about 40%. This can be understood as follows: if the breadth of the wetland is L and nitrate is measured until halfway (L/2), then effectivity is 50%. If nitrate is measured all the way at the river, then effectivity is 0%. So the 40% removal effectivity in Brynemade translate to a nitrate limit about 80 m away from the wetland edge, or 120 m away from the river. No traces of pesticides were found in the water samples (Engesgaard & Nilsson, 2011).

Available water quality measurements at the study area consist of sample measurements of concentrations of nitrate and other compounds (Cl, SO_4, NO_2, NH_4, O_2, Fe, Mn). These were taken at the piezometer locations along transects A-C (figure 3) at different depths (approximately 1.8 m, 3 m, 5 m and 7 m below ground) (Jensen, 2014). The piezometers are clustered in numbered groups, with a number after the decimal point indicating de depth of the piezometers. The higher the number after the point, the greater the depth, e.g. 11.2 is the second shallowest piezometer of group 11. On the map, only the shallowest x.1 piezometers are indicated. The measurements were carried out at three different times during 2010-2011: 13-15 April 2010, 4-6 October 2010 and 4-5 May 2011. In April and October samples were taken at all piezometers, and in May 2011 in only five piezometers. The NO_3^- concentration measurements are presented in figure 4.

A relationship between nitrate reduction rate and depth below ground surface was developed from a soil core take in the study area (Hoang, 2013). However, this relationship was not found to be valid for the upper 10 cm of the top soil, where nitrate reduction rates are much higher. This might be explained by the shallow root zone of grass and the facilitating effect of vegetation on denitrification (see 2.1 Theoretic background).

Estimates of the initial nitrate decay rates were determined from soil cores. Using these as a starting point the decay rates were then calibrated in the model to simulate the observed nitrate concentration. This resulted in a denitrification rate equal to a half-life of 200 days for the sand aquifer below 2 meter (Jensen, 2014). Another study used a 2D transport and nitrate reduction model and came to a nitrate reduction with a halftime of 80 days (Engesgaard & Nilsson, 2011).

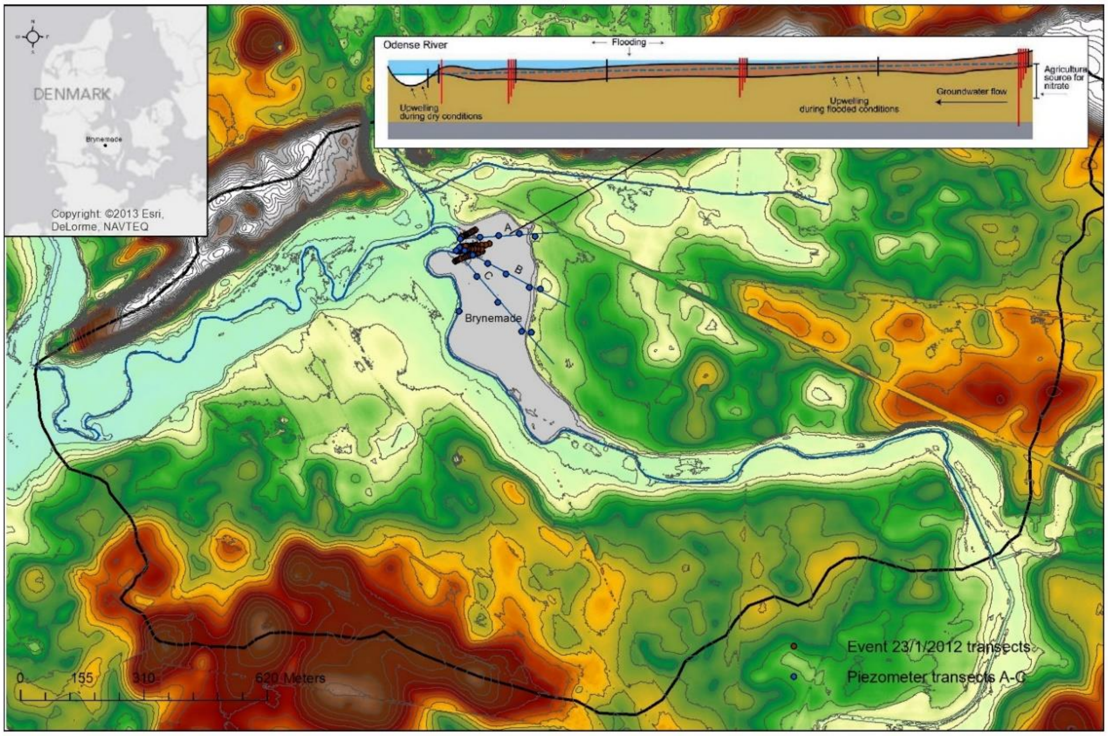

Figure 1 Overview of the study area (in grey) and surroundings (Von Christierson et al., unpublished)

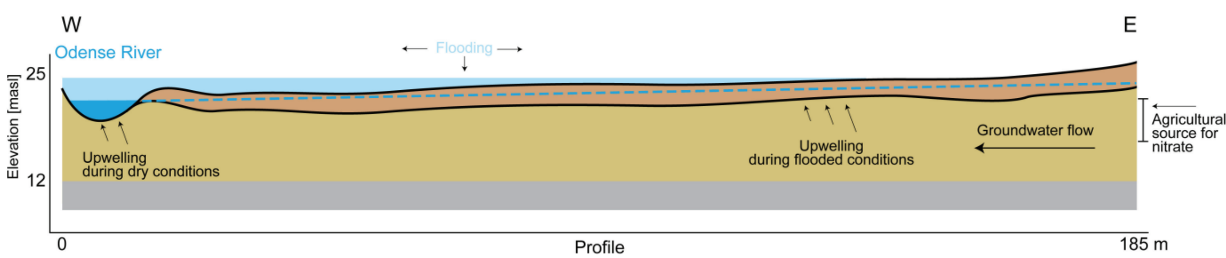

Figure 2 Conceptual model of the Brynemade riparian zone (Jensen et al., 2014a)

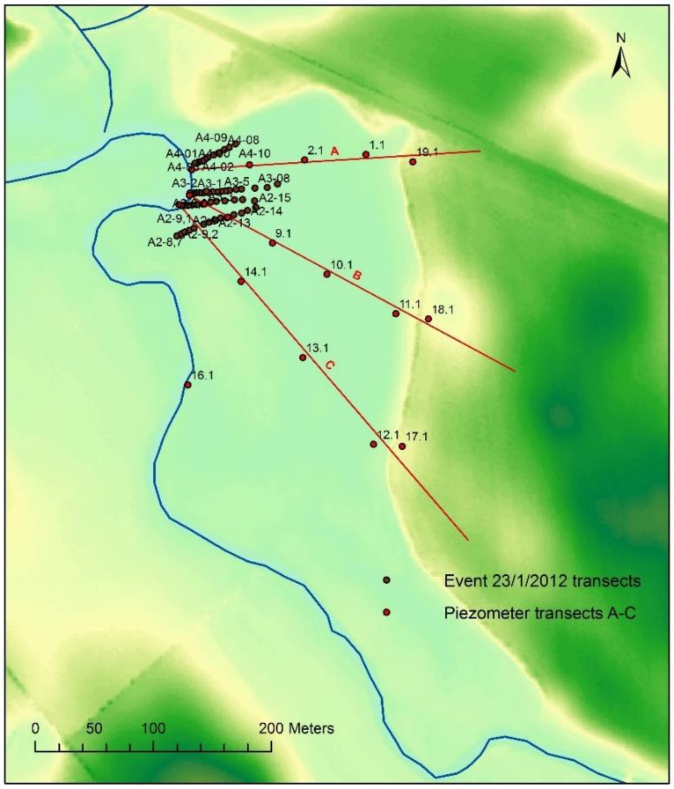

Figure 3 Location of piezometers (Von Christierson, Hansen, & Butts, unpublished)

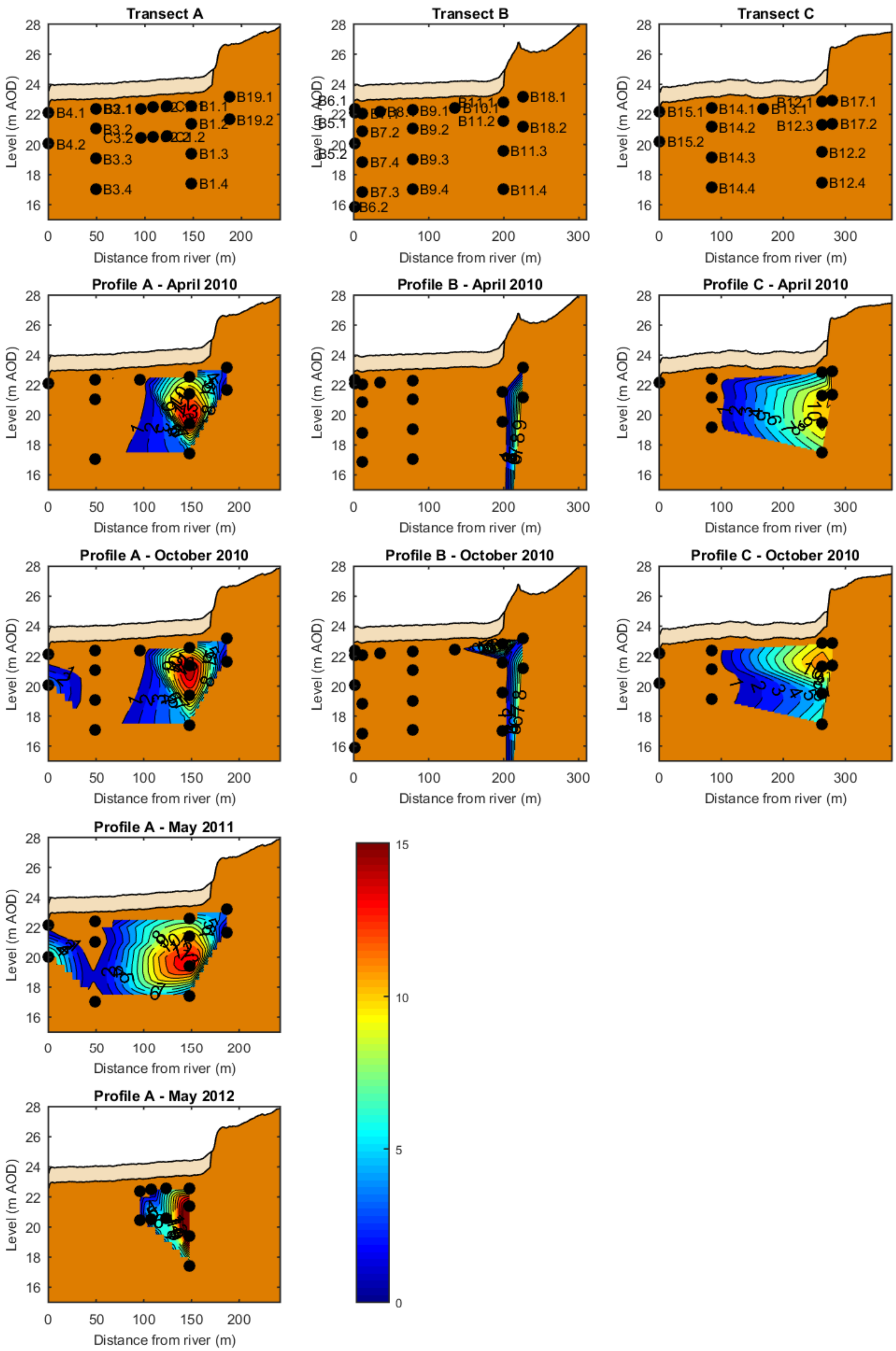

Figure 4 NO_3^- observations in the wetland (mg/L). Values lower than 1 mg/L are cut off (based on Jensen, 2014)

3. Method

In this chapter, first a general overview of the type of model used in this study is given. Then a detailed description follows of the flow model on which this study is based, with the adjustments and elaborations performed described and illustrated separately. The same is done for the base water quality model and the alterations which were needed to make the model outcome a suitable representation of the nitrate processes in the wetland. Subsequently, a description follows of the design of the pre-restoration model, i.e. a model of the situation before the channel was re-meandered.

3.1 Model of current situation

A dynamic 3D hydrological model (MIKE SHE) was set up in order to describe the hydrology of the site in a previous study (Von Christierson et al., unpublished); this model was used as a basis for this study. The flow model was extended and improved and the water quality module of MIKE SHE (MIKE SHE-ECO Lab) was used for investigating the nitrate retention capacity of the wetland.

MIKE SHE is a grid-based integrated surface water-groundwater model (DHI, 2012) which can represent flow processes within the river network, groundwater, the unsaturated zone and surface flows (Butts et al., 2012; Graham & Butts, 2006). It uses a finite difference formulation of a 2D model for overland flow combined with a 1D river model (MIKE 11), a 1D model of unsaturated flow and a 3D description of saturated flow. The model also has an evaporation and vegetation module for calculating percolation, as well as advection-dispersion (conservative) transport processes.

ECO Lab, an ecological process model developed by DHI handles the water quality simulations of the sinks and sources in the advection-dispersion equations. ECO Lab is incorporated in the user interface of MIKE SHE and is essentially a process equation solver that calculates the rate of change of any type of state variable given any number of related variables, processes and forcings (DHI, 2009). The biological and chemical transformation processes affecting the state variables of an ecosystem are represented as a set of coupled ordinary equations. The process functions can consist of mathematical functions, built-in functions, numbers, forcings, constants, and state variables. ECO Lab has a number of templates that describe well-known processes such as nutrient cycling, phytoplankton and zooplankton growth, growth and distribution of rooted vegetation and macro-algae in addition to simulating nitrogen and oxygen conditions. Additionally, customised process descriptions can be formulated.

ECO Lab relies on the MIKE SHE model to calculate flow and transport processes and acts as a post-processor at each time step to calculate the process dynamics. In this study, a coupled model was developed where the MIKE SHE modelling framework is used to model the flow and transport and then ECO Lab is used to model water quality and ecological processes within the ground water, surface water, channel system and soil water.

3.1.1 Base MIKE SHE models

The model developed by Von Christierson et al. (unpublished) consists of a larger catchment model of the area around the wetland (including the wetland) and a smaller wetland-only model of 10 ha (figure 5). The catchment model is used to simulate water flow, and the wetland model for both water flow and water quality. The larger catchment model results serve as input (hydrological boundary conditions) for the smaller model. The models run for seven years: from 01/01/2005 to 01/01/2012.

Both models are built up of geological layers, water quality layers, and computational layers. Of these, geological layers are the only ones that reflect reality while the other two represent computational layers. The model has two geological layers: an upper 1 m thick peat layer, and a lower 12 to 42 m thick sand layer. The peat layer has a horizontal conductivity of $1.16*10^{-6}$ m/s and a vertical conductivity of $1.16*10^{-7}$ m/s, whereas the sand layer has a horizontal conductivity of $9.26*10^{-5}$ m/s and a vertical conductivity of $9.26*10^{-6}$ m/s. These layers overlie an impermeable clay base layer, with a horizontal and vertical conductivity of $1*10^{-10}$ m/s and $1*10^{-11}$ m/s respectively. MIKE SHE assumes that the horizontal conductivity is isotropic in the x and y directions (DHI, 2012).

The water quality computational layers consist of peat, sand <0.5 m below the peat, sand <1 m below the peat, and sand >1 m below the peat. Seen from the ground surface, the first layer covers the unsaturated zone and is set to 1 m thickness in the wetland. Outside the wetland the thickness is variable and chosen experimentally so that even with the lowest modelled groundwater level, this layer doesn't dry out completely. This is necessary because of the way the model represents the exchange between the saturated and unsaturated zone and errors in the concentration calculations can occur (see Appendix A). These water quality layers are used to differentiate between water quality parameters in different depths. For example, the denitrification rate decreases significantly with depth.

The computational layers are the layers, which are actually used for the numerical finite difference computation. These do not have to coincide with either the geological or water quality layers, and MIKE SHE will take into account any differences. Seven layers were used: unsaturated zone, and sand <0.5 m, <1 m, <2 m, <3 m, <6 m, and <12 m below the unsaturated zone. Here too, the first layer covers the unsaturated zone and is set to 1 m in the wetland while outside the wetland the thickness is variable to ensure it doesn't dry out. The other eight layers cover the saturated zone, with increasing thickness with depth. This is because the nitrate plume is mostly moving in the shallow subsoil, so a higher precision is desired close to the ground surface. The three types of layers and their respective depths in a situation inside the wetland, where the unsaturated zone is 1 m thick, are illustrated in figure 6.

Drainage in both models is calculated with an empirical formula. Each cell requires a drain level and a time constant (leakage factor) to calculate the amount of drainage. In these models, the drain level is at a depth of -0.5 m and the time constant (and thus drainage) is only given in certain areas in the northeast and south of the catchment model area (see figure B-1 in Appendix B). The drainage is routed based on spatially varying grid codes. All drainage generated within one zone is routed to recipient nodes with the same drain code value. Per cell, the pre-processer routes the drainage in the following preference order: first, the nearest cell with a river link with the same grid code value, second, the nearest outer boundary cell with the same grid code, and third, the cell within the same grid code with the lowest drain level (DHI, 2012). In this way, grid codes can be thought of as drainage catchments.

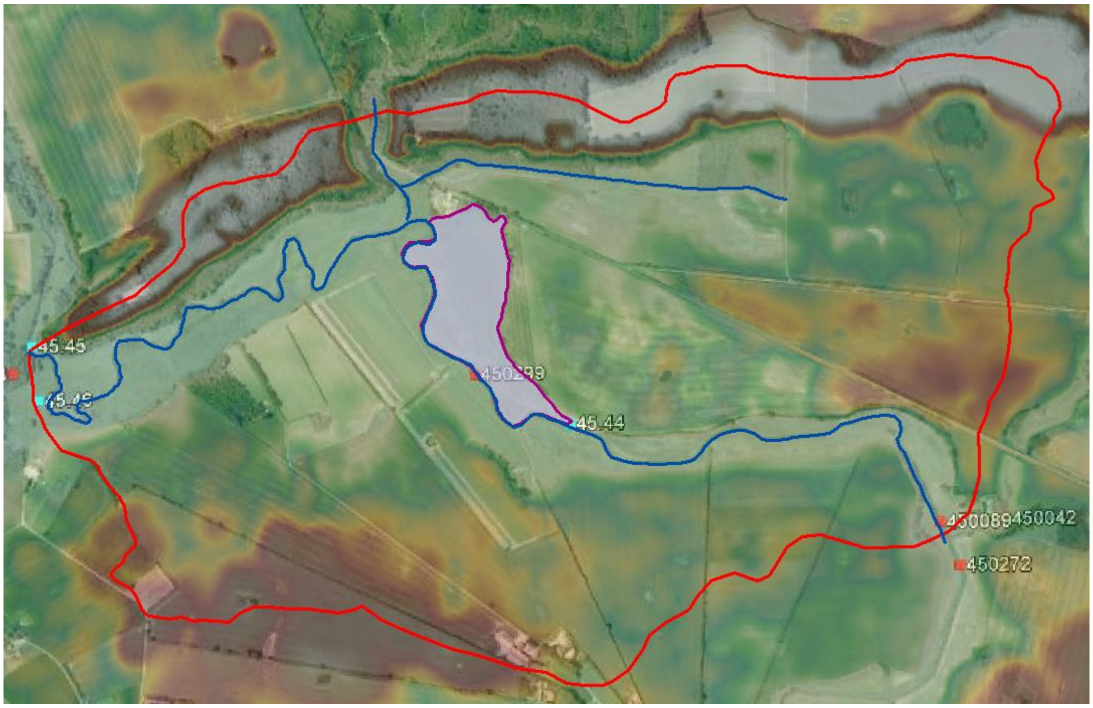

Figure 5 Extent of the catchment model (red line) and wetland model (highlighted area within purple line), projected on a composition of an aerial photo and elevation map. The Odense River is represented by a blue line.

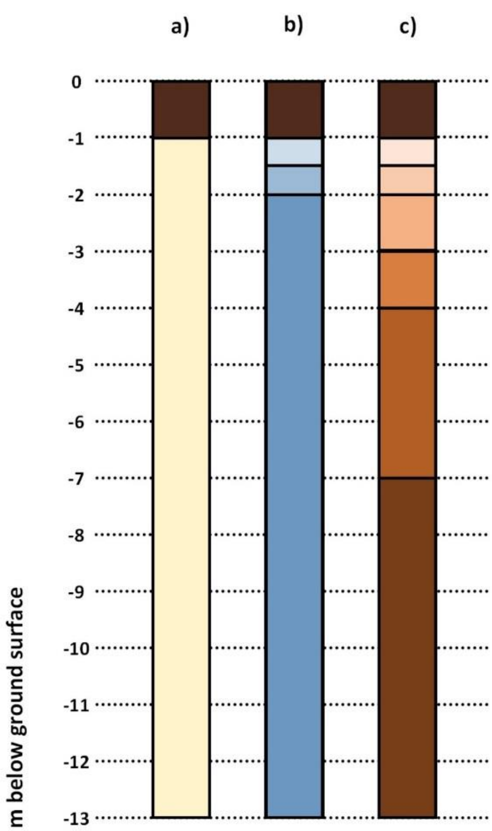

Figure 6 The three types of layers: a) geological layers, b) water quality layers, and c) computational layers

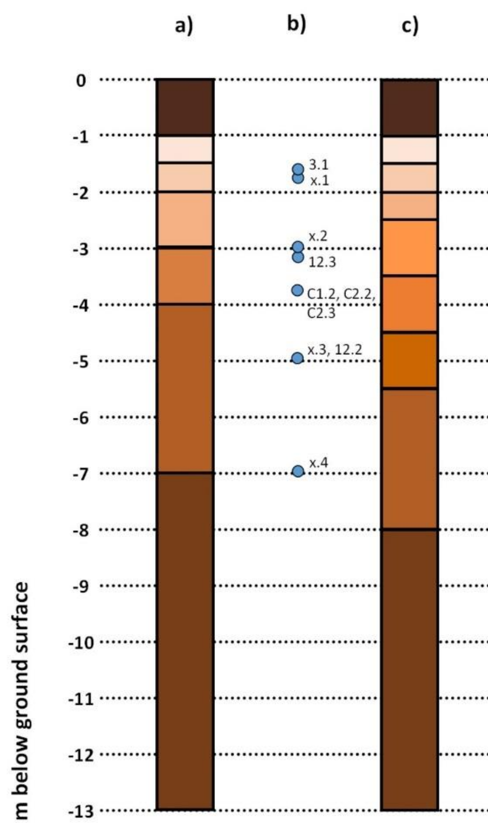

Figure 7 Original (a) and revised (c) computational layers, with the piezometer screen depths (b)

3.1.2 Extended MIKE SHE models

The alterations to the catchment and wetland flow models were kept limited, since it was already mostly calibrated. However, it was found that some adjustments had to be made to the flow model to increase the accuracy of the water quality model.

The main adjustment was performed on the computational layers. The original models consist of 7 computational layers. For presenting the results, the model looks at the value in the middle (depth-wise) of each layer. If the layer extends from -4 to -6 m, then the model result gives the value at -5 m. At any given location (x,y dimensions), one can choose to view the result of any of the 7 layers (z dimension). These can then be compared to field measurements in the same location. These measurements are made at particular depths, namely the depths of the filter screens of the piezometers.

However, the middle points of the 7 layers did not coincide with the filter screen depths. Some layer extents were such that a filter screen was exactly on the border between two layers. This made the model less suitable for comparing water quality simulations with the observations. Ideally the model result depths should coincide with the measurement depths, for realistic comparison. Both the larger and the smaller models were thus expanded to a 9 layer model. The computational layers were chosen to coincide with sample depths, while keeping a higher precision in the shallow soil to follow the nitrate plume (see figure 7). These layers boundaries were set at: variable unsaturated zone, and sand <0.5 m, <1 m, <1.5 m, <2.5 m, <3.5 m, <4.5 m, <7 m, and <12 m under the unsaturated zone. The effect of this adjustment is illustrated in Appendix B.

3.1.3 Base advection-dispersion and ECO Lab models

MIKE SHE can simulate conservative transport (advective-dispersive processes) and in addition has a small and fixed set of water quality processes. The hydrological model includes overland flow, river flow, unsaturated flow with gravity flow, evapotranspiration with plant uptake, and saturated flow. For the case of the wetland model, the water quality model is run subsequently.

At the first time step in the wetland model, the NO_3^- and NH_4^+ concentrations are assumed to be zero in the saturated, unsaturated and overland zones. This is assumed because the unsaturated zone is very small to non-existent (during floods) in this wetland, and because we assume a no-flood (no overland water) state at the beginning of the model. In the saturated zone, the model reaches equilibrium after about 1-2 years.

Nitrate input sources to the wetland are from the river during floods and the eastern boundary of the wetland (see figure B-2 in Appendix B). The nitrate load of the river is represented by a time series based on measurements at the station in Nørre Broby (downstream, east of the area shown in figure 5). The nitrate inflow boundary to the east is set to a constant value based on piezometer measurements and might stem from agricultural activity to the east of the wetland (Jensen et al., 2014b). The inflow is applied from 0 to 6 m below ground surface, and has a concentration of 13 mg/L NO_3^--N and 0.02 mg/L NH_4^+-N.

To perform more general water quality modelling, the ECO Lab model is used. During water quality simulation, the advective-dispersive N transport is calculated for each time step based on the hydrodynamics from MIKE SHE. The updated concentrations and forcing functions are then used in ECO Lab which evaluates all the expressions, integrates one time step, and returns the updated concentration values to the flow model, which then advances one time step. Simplified, the MIKE SHE process looks at N transport in and out of a cell, while the ECO Lab process looks at N processes within a cell.

ECO Lab templates can consist of:

- State variables – species concentration
- Constants – values always constant in time
- Forcings – variables of external nature which can vary in time
- Auxiliary variables – optionally
- Processes – describe the rate at which something changes
- Derived outputs – optionally

The unit for rate of change of state variables are g/m^2/day, mg/L/day (these are equal value), and undefined (dimensionless) (DHI, 2009). Time units are generally specified as day^{-1}.

Both MIKE SHE and ECO Lab process nitrogen only, not the individual species (such as NO_3^-) themselves. Instead, the different state variables represent the pools of nitrogen as they are distributed over the species. For example, the NO_3^- state variable output is actually only the N which is in NO_3^- form. To be able to compare the model output with field measurements, the output is converted to plain NO_3^- concentration by means of a simple molar conversion factor. For example, a model output of 4 mg/L NO_3^- -N translates to 4*4.426803 = 12 mg/L NO_3^-.

Forcings (such as temperature) can be specified in different ways. They can be user specified (e.g. as constants or timeseries), or built-in forcings, which can be selected from a list in the dialog. These are estimated and updated during simulation in MIKE SHE and then used as arguments in ECO Lab.

When an ECO Lab template is loaded to MIKE SHE, the default values of constants and forcings which are given in the template are not loaded. These have to be put in to MIKE SHE manually.

In the large catchment model no water quality modelling takes place, because of limited data availability. In the river, no water quality processes other than advection-dispersion are modelled because the flow rate is too high for the processes to have any effect on the concentrations before the water has flown out of the model area.

In the wetland model, the water quality processes are run in three different compartments: unsaturated zone, saturated zone and overland flow, each with different processes included (see figure 8 for a schematic representation).

In the unsaturated zone the processes are as follows:

$$\frac{d[NO_3^- - N]}{dt} = \left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right) - \left(K_t^{(t-20)} * K_d \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}\right)$$

$$\frac{d[NH_4^+ - N]}{dt} = -\left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right)$$

Where the first term between brackets is the nitrification and the second term the denitrification. These processes only take place when the nitrate or ammonium concentrations are above a certain threshold of 0.001 mg/L.

K_t = temperature constant with t = current temperature
K_n = nitrification rate at 20°C
K_{mn} = half-saturation concentration for nitrification in mg/L
K_d = denitrification rate at 20°C
K_{md} = half-saturation concentration for denitrification in mg/L

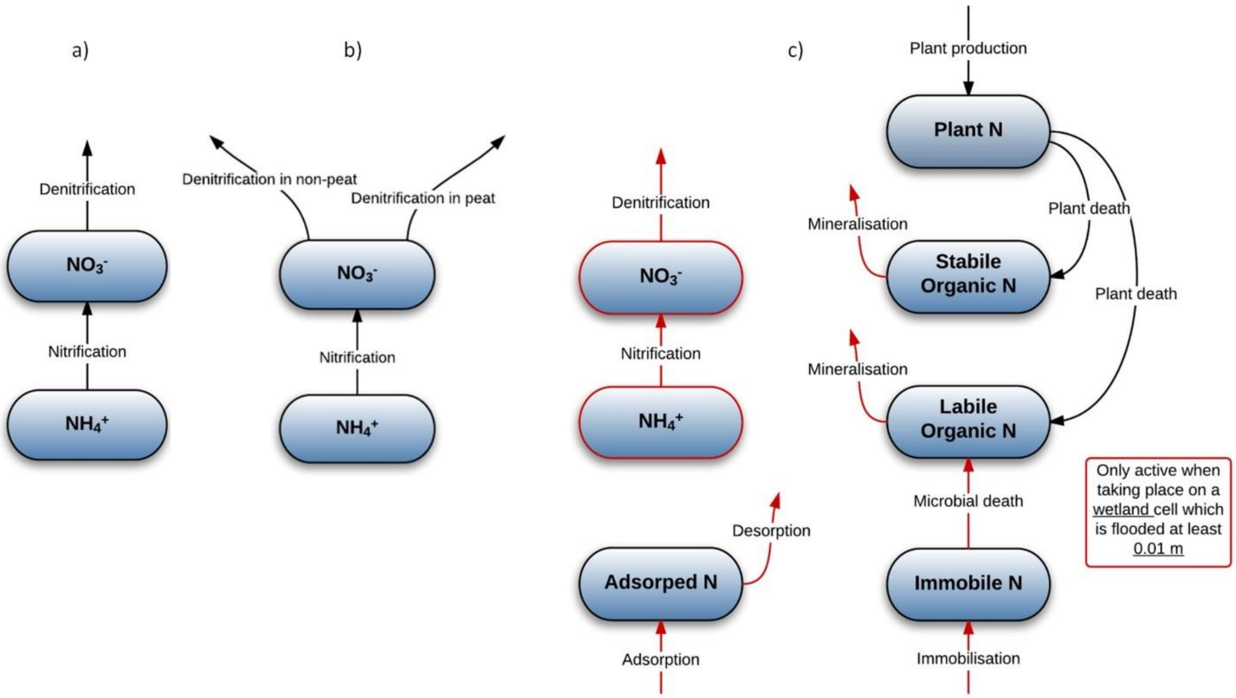

Figure 8 Flow charts of the a) Unsaturated Zone, b) Saturated Zone, and c) Overland Flow and Ground Surface ECO Lab models

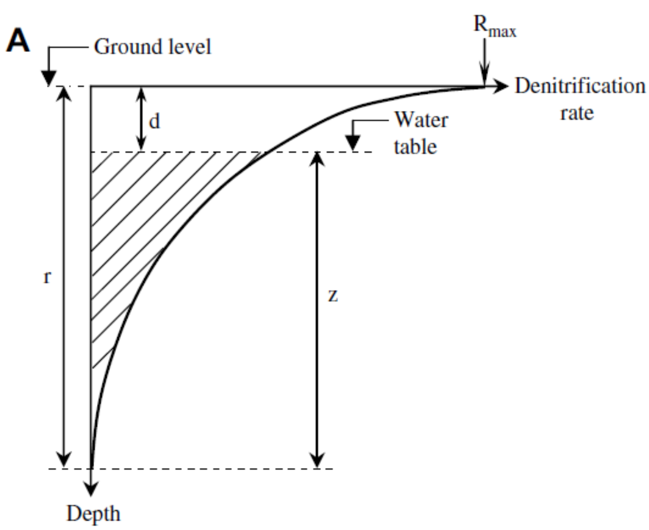

Figure 9 Distribution of denitrification rate through the riparian buffer (Rassam et al., 2008).

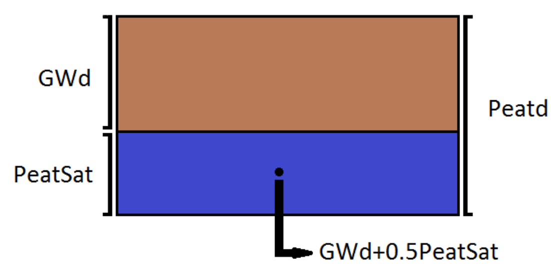

Figure 10 Derivation of GWd+0.5PeatSat from the ground surface level.

In the saturated zone, the total denitrification is composed of the denitrification occurring in peat soil, and in non-peat soil. As in the unsaturated zone, these processes only take place when the nitrate or ammonium concentrations are above 0.001 mg/L. If not named otherwise the terms in the equations are identical to the ones used in the unsaturated zone.

$$\frac{d[NO_3^- - N]}{dt} = \left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right) - \left(K_t^{(t-20)} * K_d \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}\right) - \left(K_t^{(t-20)} * K_{Peat} \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}\right)$$

The constant K_{Peat} is the denitrification rate in saturated peat, an expression which is based on Rassam et al. (2008). They model the distribution of denitrification potential with depth using an exponential decay function:

$$R_d = R_{max} \frac{e^{-kd} - e^{-kr}}{1 - e^{-kr}}$$

Where d = vertical depth below ground surface, R_d = nitrate decay rate at any depth *d*, R_{max} = maximum nitrate decay rate at soil surface, r = depth of root zone, and k = parameter describing the rate at which nitrate decay declines with depth (see figure 9).

In the model this equation is transcribed as follows:

$$K_{Peat} = (K_{max} - K_{min}) * \frac{e^{-k*(GW_d + 0.5*PeatSat)} - e^{-k*Peatd}}{(1 - e^{-k*Peatd}) + K_{min}}$$

Where

K_{max} = maximum denitrification rate in saturated peat
K_{min} = minimum denitrification rate in saturated peat
k = rate at which denitrification declines with depth in peat or rootzone
GW_d = groundwater depth relative to surface in m
PeatSat = saturated peat layer in m (= Peatd – GW_d)
Peatd = peat layer thickness in m (see figure 10).

In the overland flow compartment which describes the surface flows in the wetland, the same formulae as in the saturated and unsaturated zones are used. However, some extra conditions have been built in. The nitrification and denitrification processes only take place when four conditions are fulfilled: 1) the nitrate or ammonium concentrations are above 0.001 mg/L, 2) there is water flooding of at least 1 cm, 3) it is in the wetland area, and 4) input is available on the total annual solar radiation and total annual plant death, either by previous measurements or by running the model for a simulated year before the actual desired time period. The processes are in g/m^2/day so they have to be divided by the water flooding depth to get g/m^3/day. The temperature is set at 10°C. If not named otherwise the terms in the equations are identical to the ones used in the saturated and unsaturated zones.

$$\frac{d[NO_3^- - N]}{dt} = \left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right) - \left(K_t^{(t-20)} * K_d \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}\right)$$

$$\frac{d[NH_4^+ - N]}{dt} = -\left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right)$$

In addition, several other N processes are included in the overland flow compartment. These too only take place if certain conditions are fulfilled. In general, conditions 2, 3, and 4 have to be met, while certain processes have an additional concentration threshold condition. The equations describe the labile organic N, stabile organic N, plant N, immobilised N, and adsorbed N. As shown in figure 8, they do not act as either inputs or outputs for the state variables NH_4^+-N or NO_3^--N. Immobilised N and adsorbed N use the current NH_4^+-N concentration as input, but without affecting it. In short, these state variables are independent of the state variables NH_4^+-N or NO_3^--N.

$$\frac{dLON}{dt} = \frac{R_N * (W_d * P)}{S} + (K_{MM} * ImmN) - (K_t^{(t-20)} * K_{ML} * LON)$$

LON = labile organic N
R_N = ratio of dead Plant N (governing equation follows) decomposed into Labile N
W_d = wetland daily death of Plant N divided by accumulated death (dependent on light processes – see Appendix C)
P = annual plant production in g/m^2/year
S = wetland sediment thickness in m
(K_{MM} * ImmN) = a representation of wetland microbial mortality induced release of NH_4^+-N, which only takes place when the concentration of immobilised N is over 0.01 g/m^3; with K_{MM} = first order microbial mortality rate in day^{-1}, and ImmN is immobilised N (governing equation follows). As shown in figure 8, the release of NH_4^+-N only acts as a sink for LON and is not connected to the NH_4^+-N state variable.
($K_t^{(t-20)}$*K_{ML}*LON) = wetland mineralization of labile organic N, which only takes place when LON is over 0.001 g/m^3; with K_{ML} = first order mineralisation rate of labile N in day^{-1}.

$$\frac{dSON}{dt} = \frac{(1 - R_N) * (W_d * P)}{S} - (K_t^{(t-20)} * \frac{K_{MS}}{365} * SON)$$

SON = stabile organic N
($K_t^{(t-20)}$*K_{MS}/365*SON) = wetland mineralization of stabile organic N, which only takes place when SON is over 0.001 g/m^3; with K_{MS} = first order mineralisation rate of stabile N in $year^{-1}$.

$$\frac{dPlantN}{dt} = \frac{L_r}{L_{acc}} * \frac{P}{365} - W_d * \frac{P}{365}$$

PlantN = N stored in plant compartment
L_r = radiation at surface (see light processes in Appendix C)
L_{acc} = accumulated light (see light processes in Appendix C)
The first term represents the wetland plant production of nitrogen and the second term the plant death (release of N).

$$\frac{dImmN}{dt} = \left(K_t^{(t-20)} * K_{immo} \frac{[NH_4^+ - N]}{K_{mimmo} + [NH_4^+ - N]} \right) - (K_{MM} * ImmN)$$

ImmN = immobilised N

The first term between brackets represents the wetland immobilisation of NH_4^+-N, which only takes place when the concentration is over 0.001 mg/L; with K_{immo} = immobilisation rate of NH_4^+-N in g/m^3/day, and K_{mimmo} = half-saturation concentration in mg/L

(K_{MM} * ImmN) = wetland microbial mortality induced release of NH_4^+-N (as in LON)

Wetland adsorption of NH_4^+-N is divided into two scenarios.

If the NH_4^+-N concentration is over 0.001 mg/L, both adsorption (first term) and desorption (second term) take place:

$$\frac{dAdsN}{dt} = K_a \left(R_{ac} * \frac{[NH_4^+ - N]}{K_{ma} + [NH_4^+ - N]} - AdsN \right)$$

If the NH_4^+-N concentration is below 0.001 mg/L only desorption is in effect:

$$\frac{dAdsN}{dt} = K_a * -AdsN$$

AdsN = adsorbed N

K_a = first order adsorption rate of NH_4^+-N in day^{-1}

R_{ac} = adsorption capacity of NH_4^+-N per m^3 peat in g/m^3

K_{ma} = half-saturation concentration for adsorption of NH_4^+-N in mg/L.

3.1.4 Extended advection-dispersion and ECO Lab model

The changes described here apply only to the wetland model, since water quality modelling does not take place in the catchment model.

The nitrate input source on the eastern boundary of the wetland was changed to fit field measurements more precisely. It was differentiated by depth and the strength was adjusted (see table 1).

Table 1 Nitrate inflow, depths under surface level

	-3.5 to -4.5m	-4.5 to -5.5m	-5.5 to -6.5 m
NO_3^--N	8.62 mg/L	12.88 mg/L	8.62 mg/L
NH_4^+-N	0.013 mg/L	0.02 mg/L	0.013 mg/L

An isotropic diffusion term was added to the geological layer definition. Both the peat layer and the sand layer were set at a longitudinal dispersivity of 1 m, and transversal dispersivity of 0.01 m, based on a sensitivity analysis performed by Jensen et al. (2014b).

The ECO Lab processes in the overland flow template were extended. The changes are summarised in figure 11. In effect, the processes of the different state variables have been connected and a plant uptake term has been added. These changes are all incorporated in the expressions of NO_3^--N and NH_4^+-N. In short, the state variables labile organic N, stabile organic N, plant N, immobilised N, and adsorbed N are now affecting the state variables NH_4^+-N or NO_3^--N. The other expressions, as well as the templates for the saturated and unsaturated zones, remain unchanged.

For nitrate, two expressions are given. At each time step, both are calculated and the one with the highest outcome value is used as actual change. In this way, the more elaborate formula is used whenever the conditions are right, and otherwise only simple first order decay takes place. In the model code, these two equations are combined with a MAX() operator.

The first expression includes nitrification and denitrification, as well as plant uptake. This plant uptake is a sink, and does not actually connect to the PlantN state variable. This last term only takes place if either or both NH_4^+-N and NO_3^--N are above 0.005 mg/L. The second expression is a simple first-order decay rate.

$$\frac{d[NO_3^- - N]}{dt} = K_t^{(t-20)}\left(K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]} - K_d \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}\right) - \left(\frac{L_r}{L_{acc}} * \frac{P}{365} * \frac{[NH_4^+ - N]}{[NH_4^+ - N] + [NO_3^- - N]}\right)$$

$$\frac{d[NO_3^- - N]}{dt} = -0.1 * [NO_3^- - N]$$

K_t = temperature constant with t = current temperature
K_n = nitrification rate at 20°C
K_{mn} = half-saturation concentration for nitrification in mg/L
K_d = denitrification rate at 20°C
K_{md} = half-saturation concentration for denitrification in mg/L
L_r = radiation at surface (see light processes in Appendix C)
L_{acc} = accumulated light (see light processes in Appendix C)
L_{acc} = accumulated light (see light processes in Appendix C)
P = annual plant production in g/m^2/year

The expression for ammonium is expanded with input from the labile and stabile organic N state variables, as well as outputs to immobilised N and plant uptake. Again the plant uptake term is only taken into account at a sufficient high concentration of either or both species.

$$\frac{d[NH_4^+ - N]}{dt} = K_t^{(t-20)}(K_{ML} * LON * S + K_{MS} * SON * S) - K_t^{(t-20)}\left(K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]} + K_{immo} \frac{[NH_4^+ - N]}{K_{mimmo} + [NH_4^+ - N]} * S\right) - \left(\frac{L_r}{L_{acc}} * \frac{P}{365} * \frac{[NH_4^+ - N]}{[NH_4^+ - N] + [NO_3^- - N]}\right)$$

K_{ML} = first order mineralisation rate of labile N in day^{-1}.
S = wetland sediment thickness in m
K_{MS} = first order mineralisation rate of stabile N in year^{-1}
K_{immo} = immobilisation rate of NH_4^+-N in g/m^3/day
K_{mimmo} = half-saturation concentration in mg/L

A detailed description of the ECO Lab templates used in this study can be found in Appendix C.

The denitrification rates are based on laboratory measurements on soil cores from the wetland and used in a first-order model of the riparian zone (Jensen, Flow and Transport in Riparian Zones, 2014). These were then manually calibrated to mimic observed nitrate concentrations. Because preference was given to locally measured values over removal rates found in literature, these first-order rates were then approximated in Michaelis-Menten rates used in this study. The Michaelis-Menten rates are calibrated to represent the same absolute removal rates up to a concentration of around 10 mg/L, which is above the concentrations normally present in the model. Figure 12 shows decay rates based on a first-order rate of 0.24 day^{-1} (the rate in the peat layer in Jensen, 2014). The Michaelis-Menten rate was fitted and has a value of $K_{d,max}$ 10 g/m^3/day and K_m of 35 g/m^3. The same procedure was carried out for the other rates in different soil depths.

Similarly, the nitrate inflow boundary to the east was changed, in accordance to measurements done locally by Jensen et al. (2014b), which were transcribed to input values for the model. The inflow depth was changed from 0 to -6 m to -1.7 to -4.7 m, and the inflow concentration from 13 mg/L NO_3^- to 10.6 mg/L and 0.02 mg/L NH_4^+ to 0.016 mg/L. The effects of these adjustments can be found in Appendix B.

3.2 Pre-restoration model

To examine the impact the river restoration had on the nitrate retention capacity of the wetland, it is necessary to look at the situation in the wetland before the restoration. For this, a model was developed of the wetland in its pre-restoration state. The main changes were to the river channel's shape and course, to the wetland shape, and to the drainage in the wetland (see figure 13). The ECO Lab templates were identical.

The drainage extent was based on a land survey map from 1978 (Det Danske Hedeselskab, 1978), from when the wetland was in use as agricultural area. This map was digitised in ArcGIS and then added to the model by changing the time constants and making a new grid code area. In accordance with the drain map, care was taken that the drainage cannot take the shortest route to the river but rather gets routed to a certain point where according to the map it is pumped in the river (see figure B-3 in Appendix B). The topography was slightly adjusted as well, since the meandered river channel was visible in the 5 m resolution topography used. It was visually removed where possible.

The straightened river cross-sections and location were put in to MIKE 11. The wetland shape was changed to account for the new drainage point. Cells were added to extend the wetland on the north-west side, and the same amount of cells were removed along the western border, so that mass balance calculations could still be compared to the restored wetland situation. The western border was chosen for cell removal because there the wetland shape follows the meandered river, whereas the pre-restoration river lies far within the wetland boundaries (see figure 14).

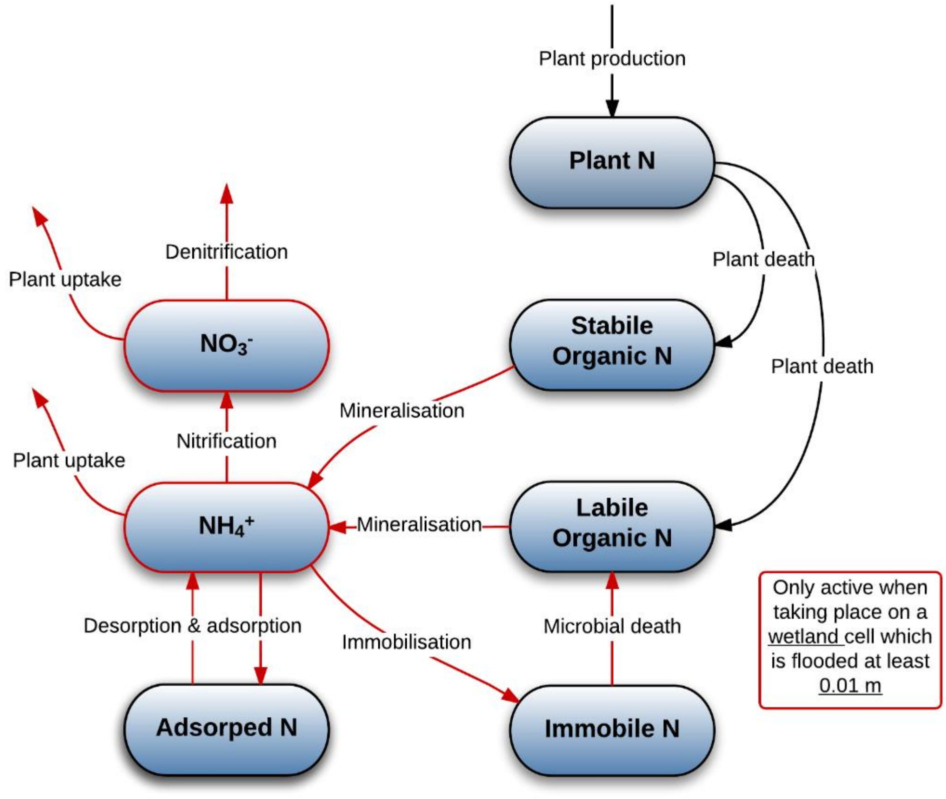

Figure 11 Elaborated overland flow template

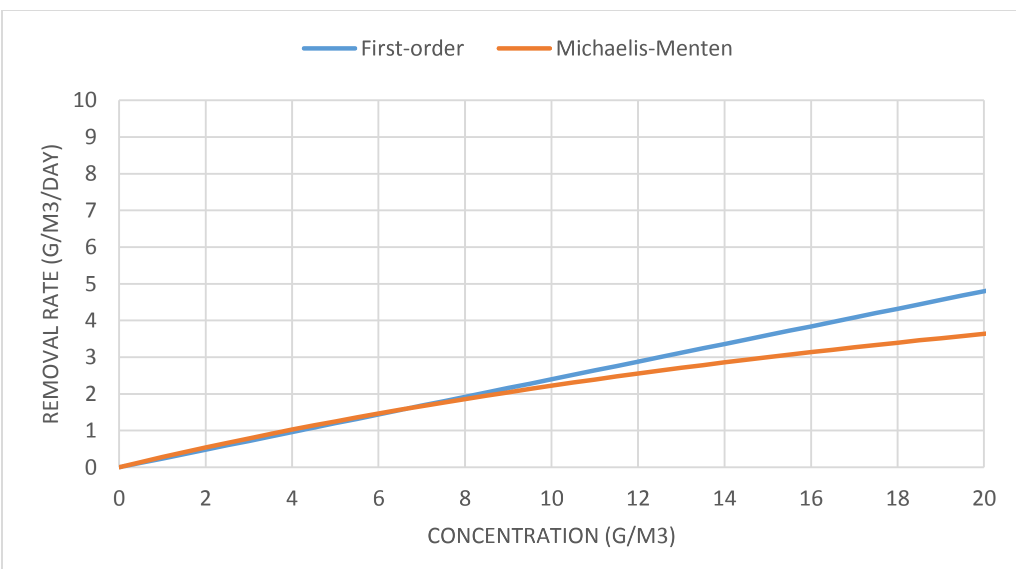

Figure 12 Removal rate vs concentration for first-order rate of 0.24 day^{-1} and calibrated Michaelis-Menten rate

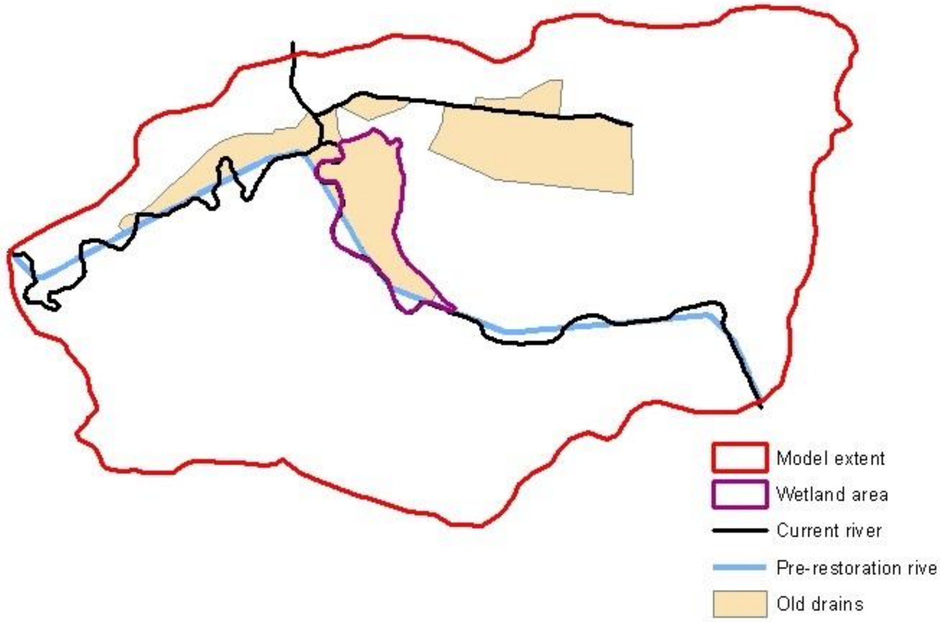

Figure 13 The pre-restoration and meandered current river channel, and the old agricultural drains. This image shows the restored wetland shape.

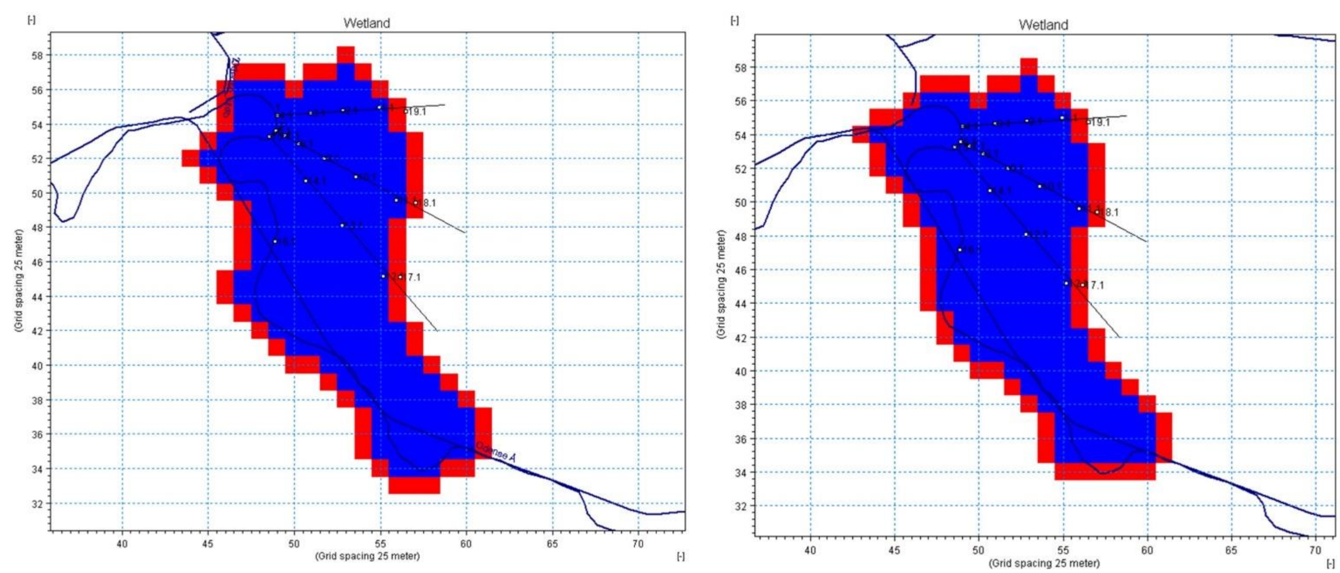

Figure 14 Restored (left) and pre-restoration (right) wetland model domain. Red cells indicate boundary. Shown are both the meandered river and the pre-restoration straight channel.

4. Results

The wetland model outcomes were analysed in different ways. To assess how well the model represents reality, the modelled NO_3^--N concentrations were compared with the measured concentrations in the field in time series and concentration profiles. In the profiles in figure 15 and 16, the right edge of the image coincides with the rightmost piezometers in each profile in figure 4.

Comparing figures 4, 15 and 16, it can be seen that the model reproduces the nitrate plume at a greater depth fairly well in both transects A and C. In the model, both transects show nitrate inflow from the surface. There are no piezometers at this shallow depth however so it is unknown if this plume is also present in the wetland. The two rightmost piezometers in transect A (19.1 and 19.2) show a very low concentration. These piezometers are relatively shallow considering their location on the hill ridge (their filter elevation corresponds to about -1.5 and -3m elsewhere in the wetland, respectively). The nitrate plume showing up in the measurements might thus originate from a deeper layer, or flow in from the top, where the hill ridge ends. In both cases piezometers 19.1 and 19.2 could be bypassed.

From an analysis of the mass balance follows the annual nitrate removal (denitrification and plant uptake), shown in table 2 and 3 for the restored and pre-restoration wetland, respectively. The removal rates per year are calculated averaged over the 7 years the model runs. The low percentage removal in the overland flow component compared to the large amount of kilograms removed, illustrates the large load of NO_3^- coming in to the wetland. In total, the wetland removes 5694 kg NO_3^- per year since the restoration, which was 1724 kg NO_3^- per year before the restoration. The nitrate load reduction due to the wetland restoration is thus 3970 kg NO_3^- per year, or 896 kg N from NO_3^- per year, which is a 229% increase in nitrogen removal. As discussed in 2.1 Theoretic background, nitrate is much more mobile than ammonium. Therefore, the nitrogen reduction from only the NO_3^--N is a good representation of the total N load reduction.

Table 2 Nitrate removal in the restored wetland

	kg NO_3^--N/year removed	kg NO_3^-/year removed	% of input/year removed
Overland flow	1171	5183	4.6%
Unsaturated zone	4	17	11.3%
Saturated zone	112	494	56.3%
Total/average	1287	5694	24.1%

Table 3 Nitrate removal in the pre-restoration wetland

	kg NO_3^--N/year removed	kg NO_3^-/year removed	% of input/year removed
Overland flow	198	876	5.9%
Unsaturated zone	13	53	14.3%
Saturated zone	180	795	44.3%
Total/average	391	1724	21.5%

Since the general groundwater flow direction is towards the river, the NO_3^- which does not get removed in any of the zones in principle flows into the river. By analysing the total N (all species) concentration time series in the river upstream and downstream of the wetland, it is seen that in the restored wetland 1171.3

mg N per litre per year is removed (averaged over 7 years), and in the pre-restoration state 584.6 mg N/L/year is removed. Thus the N load to the river is on average reduced by 586.7 mg N/L/year. In the pre-restoration wetland 0.23% of the river input is removed, and in the restored wetland 0.47%. Thus the wetland restoration leads to a change in reduction of N-load of 0.24%. This does not contradict the wetland N reduction of nearly 230%. Floods result in lower nitrate concentrations in the river, since the overland flow compartment carries out denitrification on flood water which subsequently flows back into the river. However, total denitrification within the wetland is also dependent on the N inflow from the eastern boundary. In addition, a transient event such as a flood where flood water mixes with groundwater with a high nitrate concentration, and then flows back before any significant denitrification can take place in the overland flow compartment, can result in less N reduction in the river compared to the wetland.

In table 2 and 3, a large reduction in nitrate input from the river to the overland flow component is observed, owing to a decrease in floods. This much lower input does however not result in a significant increase in the percentage removed. This may indicate that the transient nature of flood events does not provide enough time for a significant amount of denitrification to take place.

The flow model from Von Christierson et al. (unpublished) shows that the general groundwater direction in the wetland is from the eastern edge to the river. The groundwater wells up as it enters the wetland, which is visible from the nitrate plume direction figures 15 and 16. The upwelling is stronger in the pre-restoration situation, which may be caused by the drainage at 0.5 m depth.

The large load to the overland flow compartment can come from both the river floods, as well as upwelling of groundwater. The model is currently not able to discriminate between different sources of NO_3^-. A map view of the total accumulated denitrification per model cell, can show in what areas the denitrification takes place and thus give an indication of where the nitrate enters the wetland. Figures 17 through 19 show the accumulative denitrification at the end of the 7 year model run in the overland flow, unsaturated zone, and saturated zone, respectively. Figure 20 shows a typical flood pattern, in this case during a winter flood in 2007.

The resemblance of the flood patterns in both the restored and pre-restoration wetland in figure 20 to the denitrification pattern in the overland flow compartment (figure 17) indicates that the denitrification in that zone mainly takes place around the river area and in areas that are prone to flooding. This implies that the majority of the nitrate load entering the wetland originate from the river, not the groundwater.

The unsaturated zone in the restored wetland does not contribute a significant amount to the total denitrification (figure 18). This can be explained by the fact that the unsaturated zone is often not present or only at a very limited depth, since the groundwater levels are high and there are regular floods. There is a significantly larger accumulative denitrification in the pre-restoration wetland. This can be explained by the lower groundwater levels due to drainage and fewer floods in the wetland, which lead to a larger unsaturated zone than in the restored wetland.

The denitrification pattern in the peat layer of the saturated zone (figure 19) of the restored wetland indicates that the nitrate load in the lower soil layers mainly originates from the eastern boundary of the wetland, where the agricultural nitrate inflow occurs. There is also denitrification taking place at the river bed, the hyporheic zone. This pattern is even more visible in the pre-restoration wetland; here, the denitrification takes place along the straightened river channel and the eastern boundary. Some of the sand layers of the saturated zone (layers 2 and 7: at 0.5-1 m and 4.5-7m under the unsaturated zone) also see denitrification but only to a minor degree, in both the restored and pre-restoration wetland.

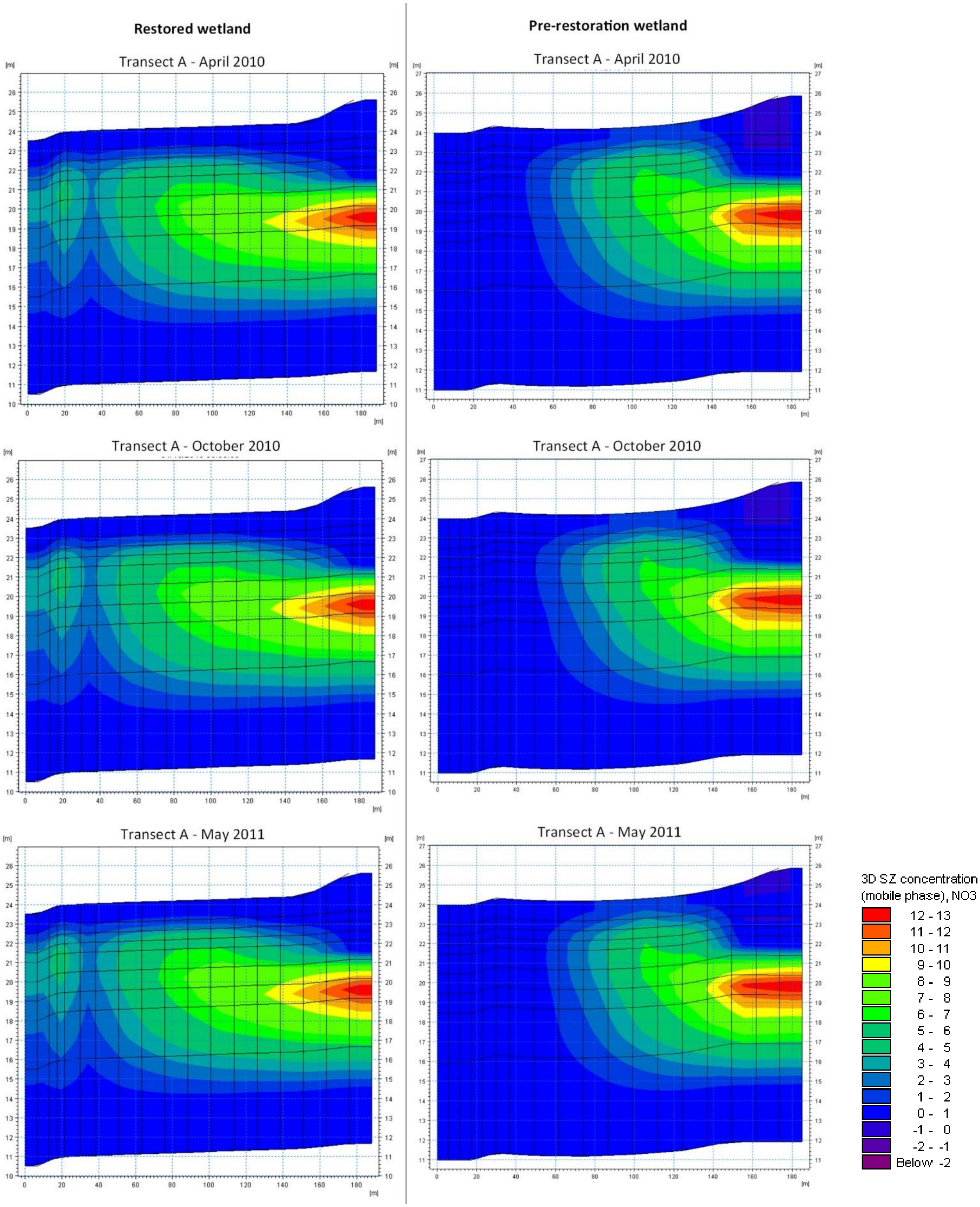

Figure 15 Modelled NO_3^--N profiles in transect A

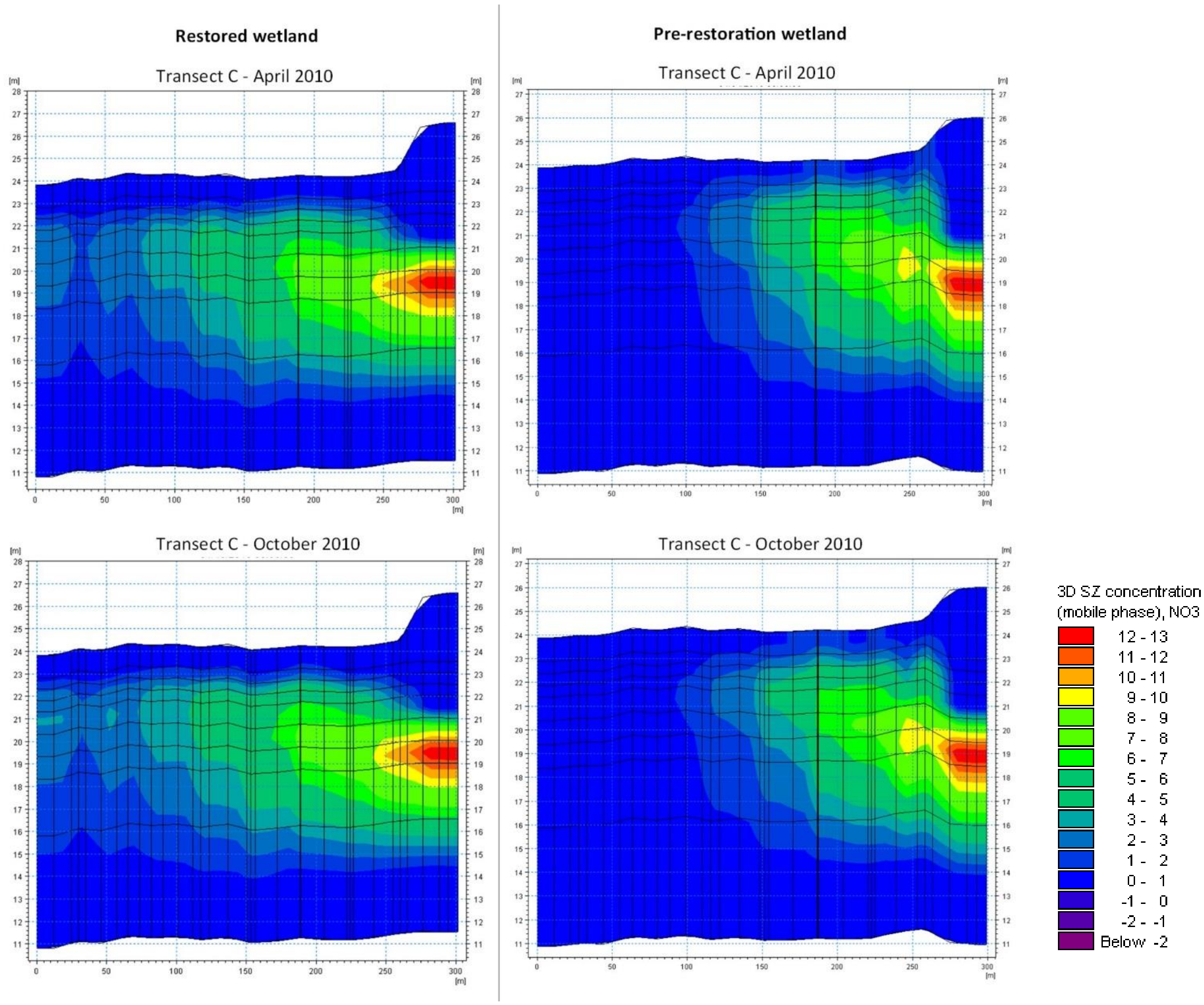

Figure 16 Modelled NO_3^--N profiles in transect C

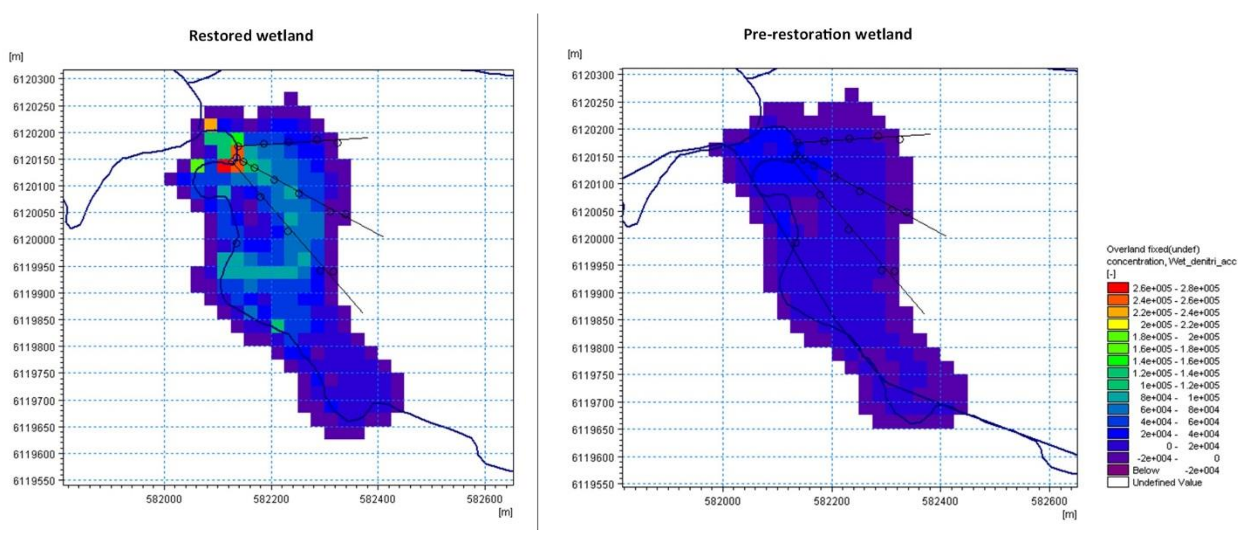

Figure 17 Accumulative denitrification in mg NO_3^--N/L in the overland flow compartment

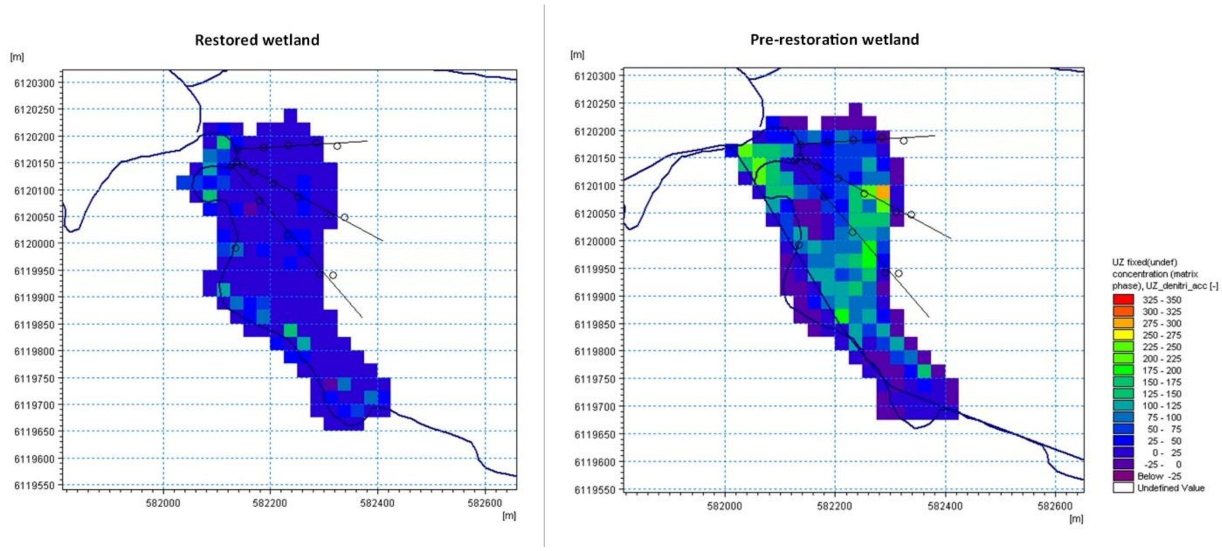

Figure 18 Accumulative denitrification in mg NO_3^--N/L in the unsaturated zone

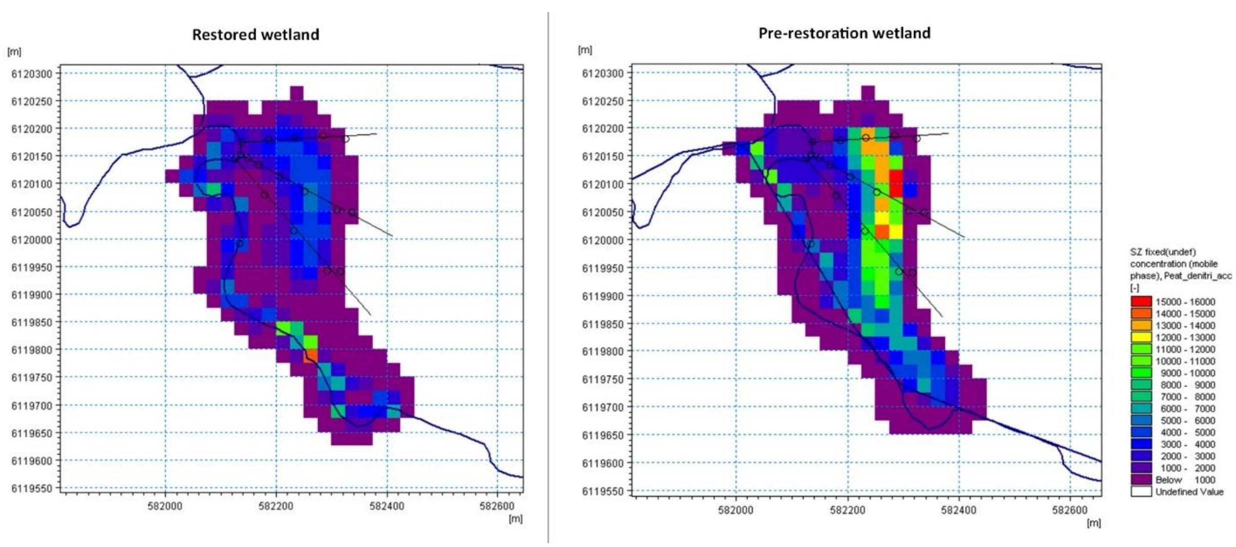

Figure 19 Accumulative denitrification in mg NO_3^--N/L in the saturated zone

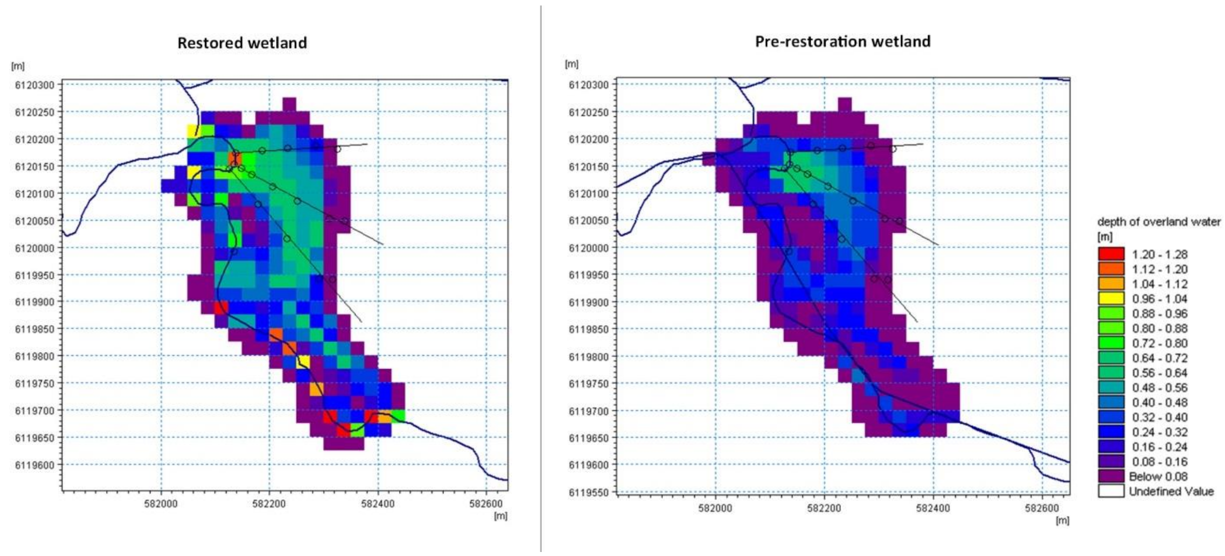

Figure 20 Modelled depth of overland water on 21st January 2007

5. Discussion

Modelling a riparian wetland can be challenging, even a well-documented wetland like Brynemade. Despite the large amount of data available, there are still some limitations in the data. A hydrological model can never provide a perfect representation of reality, but the question is whether the representation reaches the level of accuracy needed. In this case the model results show a large degree of conformity with the field measurements that the model can be assumed to be accurate enough to answer the research questions.

Within the wetland, pumping tests and soil cores show that the aquifer is very heterogeneous (Jensen, 2014). For example, from the nitrate measurements follows that in the region of transect B there is no nitrate inflow from the eastern boundary. No extra measurements have been made to map these heterogeneities and the path of the nitrate plume. Within the model, the heterogeneities found in the wetland are not accounted for. It is mainly at the eastern boundary around transect B that this affects the accuracy of the model somewhat.

In addition, the hyporheic zone is not accounted for separately. The model already shows a significant amount of denitrification taking place in the saturated zone at the river, but Zhou et al. (2014) state that the hyporheic zone plays a major role in the nitrogen removal not just via denitrification, but also via anaerobic ammonia oxidation and plant uptake. It would benefit the model's accuracy if a hyporheic zone could be included. Within the coupling of MIKE SHE and MIKE 11 this would provide a challenge and might demand a more conceptual approach.

Outside of the wetland, the water level and N measurements in the river are derived from a station downstream from the wetland, at Nørre Broby. The water levels can be easily adjusted to represent the wetland, by scaling them by the ratio of the size of the catchment at Brynemade and the size of the catchment at Nørre Broby. The N-species measurements are scaled in this manner as well. However, these downstream measurements already passed the wetland and thus have a lower N concentration than before passing/flooding the wetland. The nitrate load to the wetland might thus be underestimated. However, no upstream N measurements exist, and seeing the large N load in the river and the relatively small effect the wetland has upon it (see for example table 1), using downstream measurements instead should be a reasonable assumption.

In the model, oxygen is not included and as such no oxygen zonation effects are taken into account. Jensen (2014) performed oxygen measurements in the wetland and found mostly very low concentrations (below the 1 mg/L detection limit), with the highest being 2.9 mg/L near the wetland margin. He assumes anoxic conditions in his model of the area since the oxygen zone is very small, and the same approach has been used in this study. This provides a reasonable approximation, but in reality the denitrification could be slightly lower than found in this study, since oxygen counteracts the anaerobic process of denitrification.

In the literature, plant uptake is described as one of the most important nitrate removal mechanisms. In the model it is only implemented in a limited way, as a sink. No link exists between the NO_3^--N and NH_4^+-N removal process and the Plant-N production process. This is a limitation in the model, since plant root N uptake would occur in the saturated and unsaturated zones, and the Plant-N state variable exists in the overland flow template, and no links are possible between ECO Lab templates.

The Brynemade wetland model has an area of 10 ha, and the wetland restoration removes an extra 896 kg N (from nitrate) per year on average. This translates to a reduction of 89.6 kg/ha/year caused by restoration. As stated in the introduction, the nitrogen reduction goal for Odense Fjord is 1200 tonnes per year. If this were to be reached with wetland restorations only, the total area needed of wetlands with similar results as Brynemade would amount to 13,393 ha. For comparison, the island of Funen, on which Brynemade is located, is 309,970 ha.

Environment Centre Odense (2007) has identified a range of costs of nitrogen load reduction measures. They determine the cost of setting aside permanent low-lying grassland at 3226 DKK/ha, and the cost of setting aside low-lying arable land for wetlands at 4240 DKK/ha. Going by these figures, the one-off wetland restoration costs would range anywhere between 43,200,000 to 56,800,000 DKK. For comparison, the cost of implementing the Water Framework Directive in the Basin by a mix of measures has been estimated by the Environment Centre at 94,000,000 DKK, and the present total costs of water use in the Odense River Basin amount to approx. 612,000,000 DKK per year. The usability of setting aside arable or grassland for wetlands to reach nitrogen load reduction goals is limited mostly by the available extent of such lands.

The post-restoration Brynemade wetland has a nitrogen removal rate of 1287 kg/year, or 128.7 kg/ha/year. This is consistent with typical removal rates of 100-200 kg N/ha/year for a wetland of this type (Hoffmann et al., 2014). It has to be noted that Brynemade was not restored purely for optimizing nitrogen reduction. Tørringe Brook was remeandered to reduce physical pressures and restore natural hydromorphological conditions (Environment Centre Odense, 2007). Optimizing wetlands for nitrogen reduction would in essence mean mimicing a constructed wetland as close as possible, in which case the local ecological function usually is not prioritized.

6. Conclusions

An integrated surface water-groundwater model linked with an ecological water quality model has been used in this study to model flow and transport in a restored riparian wetland. This model has been validated against observations of river flows and levels, groundwater levels and nitrate concentrations. From the model results can be concluded that the behaviour of a restored riparian wetland can be represented with a sufficient degree of accuracy.

Denitrification patterns in the surface water component follow river flood patterns, which indicate that most nitrate input originates from the river, with upwelling from groundwater only playing a minor role. Since the pre-restoration wetland received fewer floods, there is much less denitrification taking place in the overland flow compartment.

In the restored wetland, the unsaturated zone plays a minor role in the total denitrification, since the high groundwater levels and the frequent floods cause only a very small unsaturated zone to be present. In the pre-restoration wetland, the groundwater drainage and fewer floods mean that there is a near permanent unsaturated zone, and total denitrification is much higher.

Denitrification in the saturated zone occurs mainly at the eastern boundary of the wetland, where the agricultural nitrate inflow occurs, as well as along the river channel, where nitrate seeps through the river bed to the saturated zone. The peat layer is the major contributor to total denitrification in the saturated zone. Most of the nitrate inflow from the adjacent agricultural fields is removed before it wells up to the surface.

Restoring a wetland to its natural state by restoring a river flood regime and removing drainage increases the nitrate retention capacity. Most of the extra denitrification capacity comes from the increase in floods and the subsequent denitrification in the overland flow compartment. There is close to a 230% increase in nitrogen removal in the wetland, and the nitrogen removal in the river doubled, from 0.23% to 0.47%. The post-restoration removal rates of 129 kg N/ha/year are within the typical range for a wetland of this type. A further 13,393 ha of similar wetland restoration on Funen Island would be needed to reach the Odense Fjord nitrogen reduction goal. On a total island area of 309,970 ha the feasibility of setting aside arable or grassland for wetlands to reach reduction goals is limited mostly by the available extent of such lands. It is desirable to implement a mix of nitrate load reduction measures in order to reach the nitrogen reduction goal.

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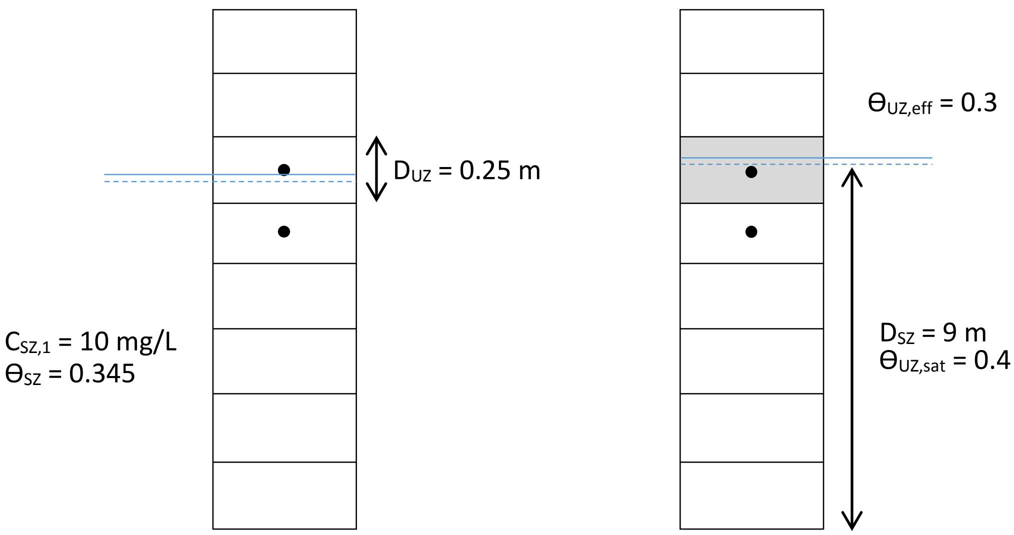
List of figures

Appendix A Unsaturated Zone Cell Size

Courtesy B. von Christierson

The UZ cell thickness must be small compared to the thickness of the saturated part of layer 1 in the saturated zone. Otherwise an error in the calculation of the concentration will occur as the water table drops or rises.

The example below shows how the new SZ concentration is calculated after the water table moves from just below the middle of an UZ cell to just above it. It is assumed that a cell belongs to the UZ until the water table reaches the calculation point in the middle of the cell:



$$C_{SZ,2} = \frac{\left(10\,\frac{mg}{L} * 9\,m * 0.345\right) + \left(10\,\frac{mg}{L} * 0.25\,m * 0.3\right)}{9\,m * 0.345} = 10.24\,\frac{mg}{L}$$

$$Mass_{before} = \left(10\,\frac{mg}{L} * 9\,m * 0.345\right) + \left(10\,\frac{mg}{L} * 0.25\,m * 0.3\right) = 31.8\,mg$$

$$Mass_{after} = 10.24\,\frac{mg}{L} * 9\,m * 0.345 = 31.8\,mg$$

The example below illustrates how the new concentrations in SZ and UZ are calculated when the water table is dropping and an UZ cell is activated (going from the figure on the right to the left):

$$C_{SZ,2} = \frac{10\,\frac{mg}{L} * 9\,m * 0.345}{9\,m * 0.345 + 0.25\,m * 0.4} = 9.69\,mg/L$$

$$C_{UZ,2} = \frac{9.69 \frac{mg}{L} * 0.4}{0.3} = 12.92\, mg/L$$

$$Mass_{before} = 10 \frac{mg}{L} * 9\, m * 0.345 = 31.05\, mg$$

$$Mass_{after} = \left(9.69 \frac{mg}{L} * 9\, m * 0.345\right) + \left(12.92 \frac{mg}{L} * 0.25\, m * 0.3\right) = 31.05\, mg$$

The graph below shows the saturated zone concentration using different UZ grid sizes. The actual concentration is constant at 10 mg/L. At a cell size of 0.2 m, there are still small fluctuations in the concentrations, but the maximum error is only approximately 1.5%.

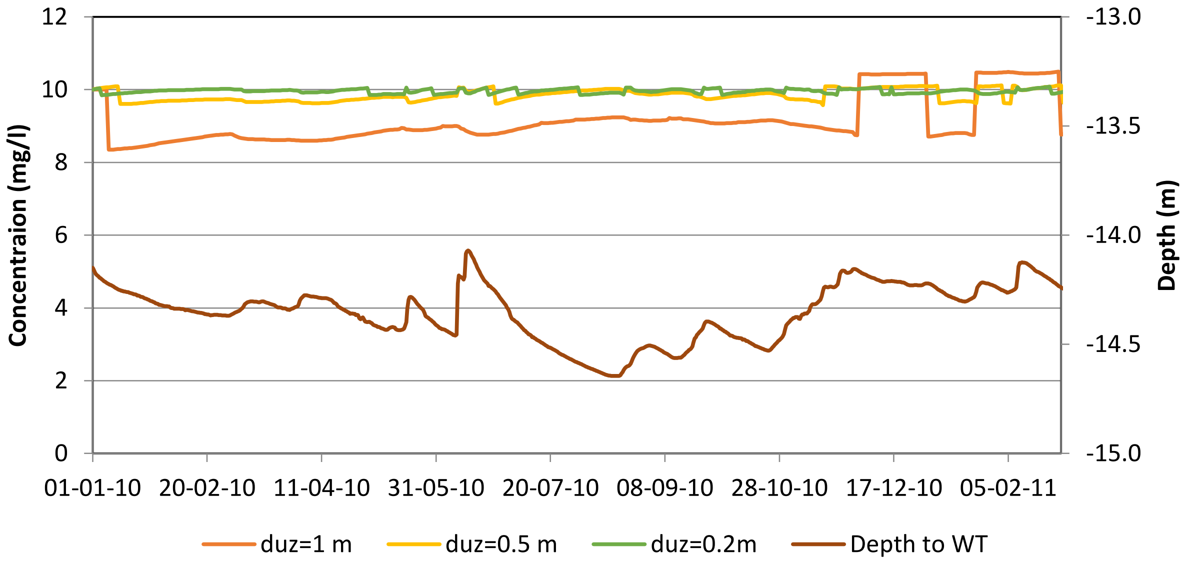


Figure A-1 SZ concentration with different UZ cell sizes

Appendix B Model inputs and calibration

Model inputs

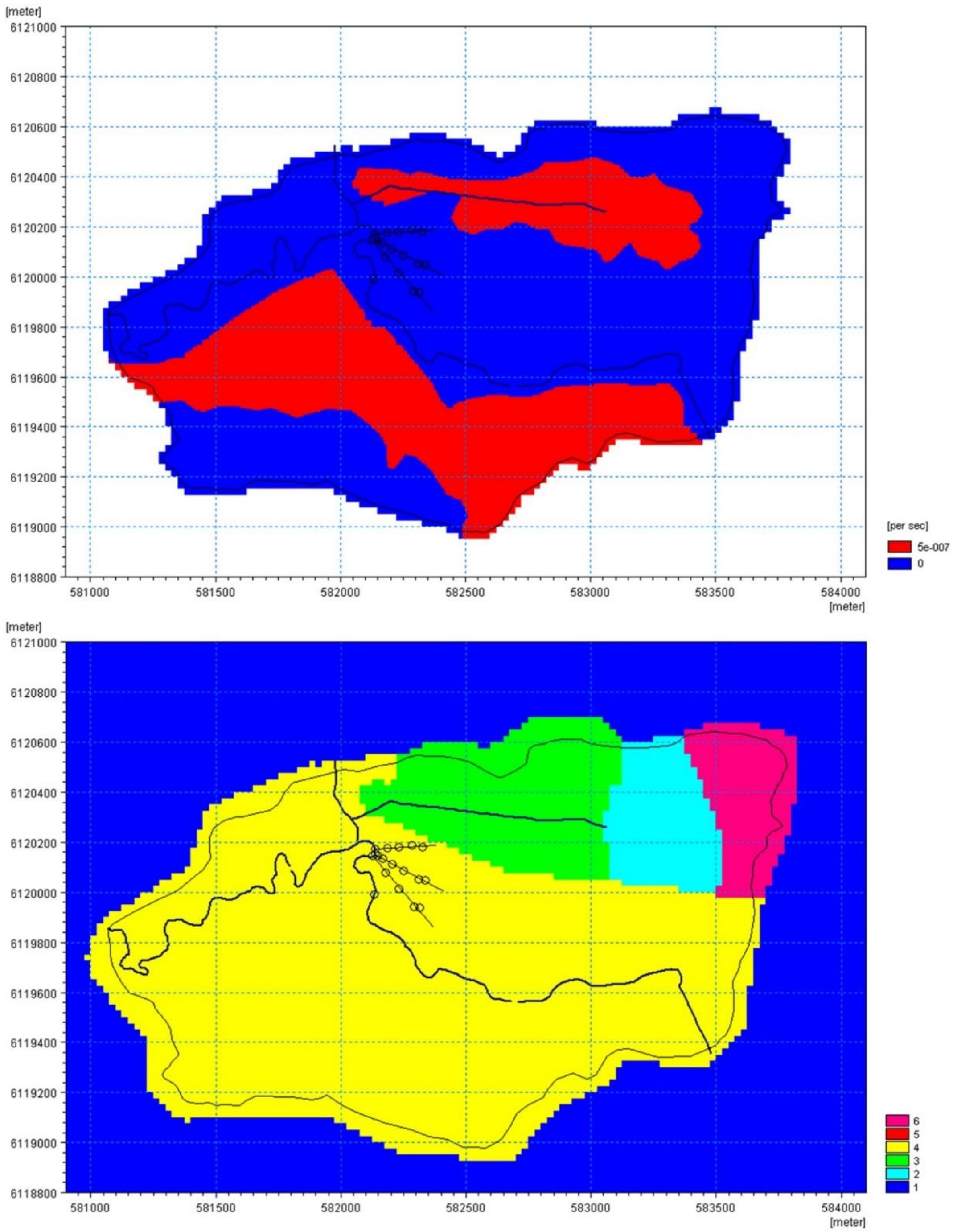


Figure B-1 Time constants (top) and grid code areas (bottom) of the base model

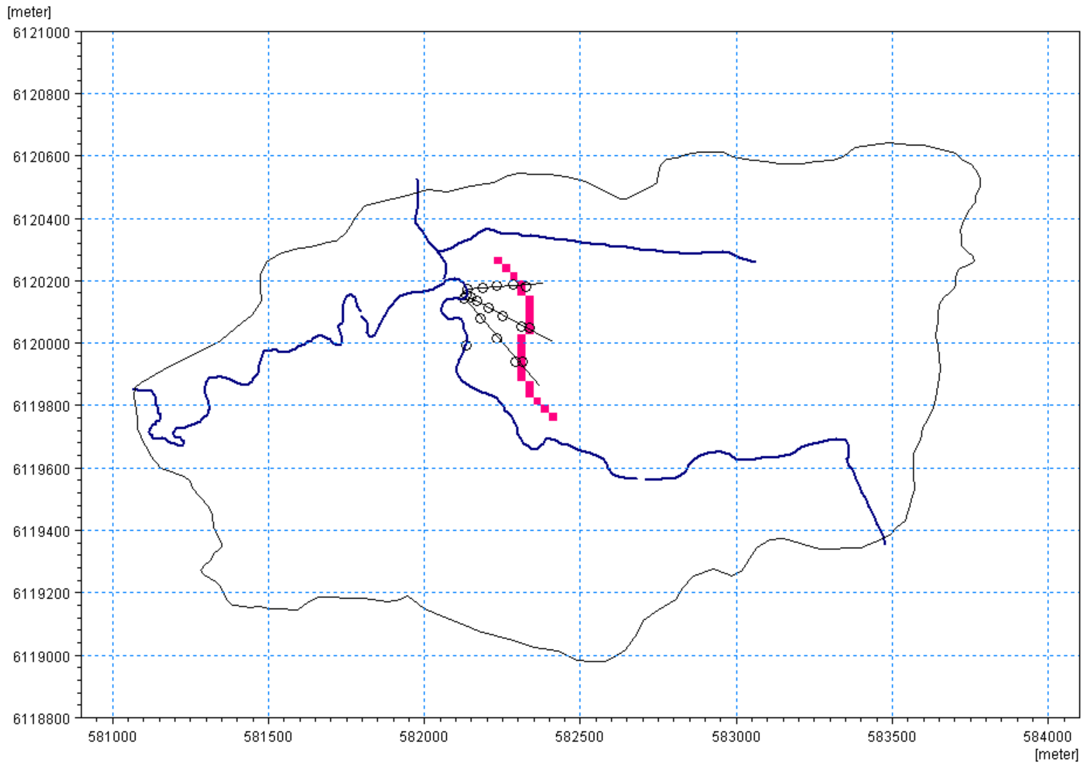


Figure B-2 Extent of the nitrate inflow boundary (pink line)

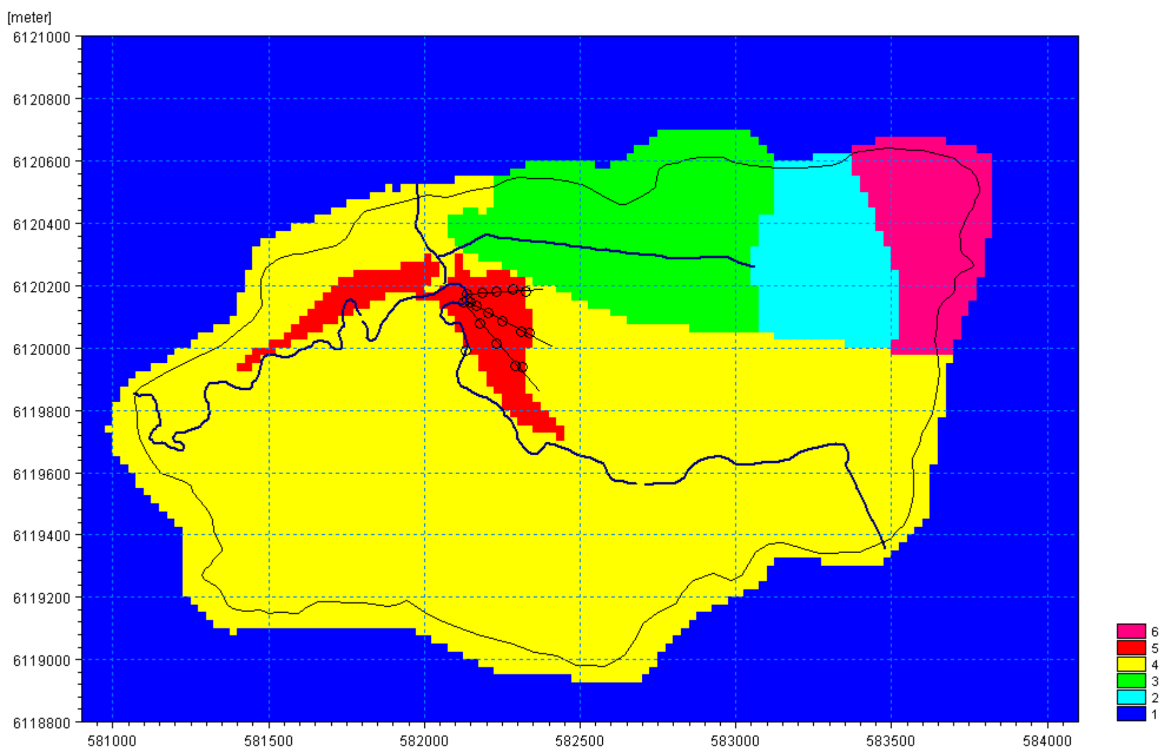
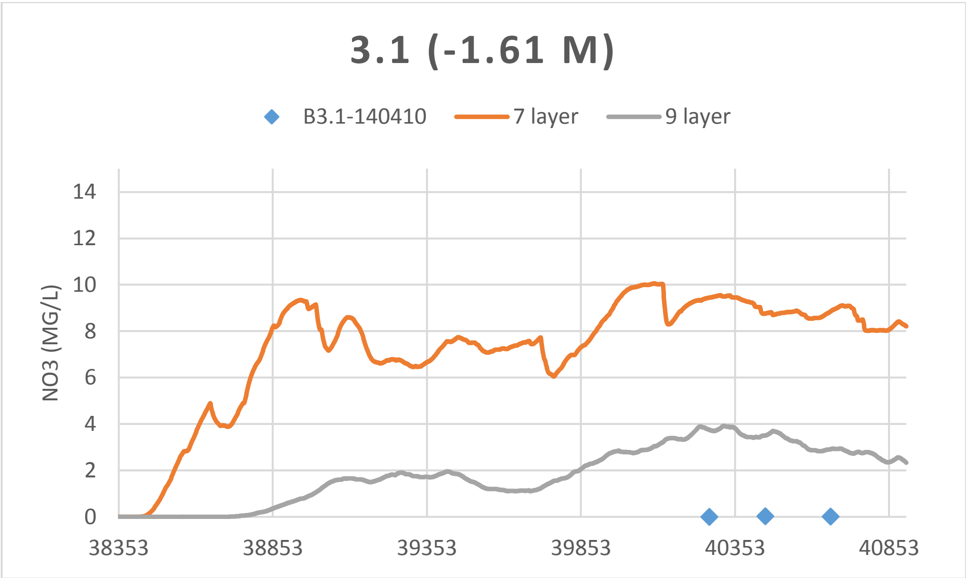
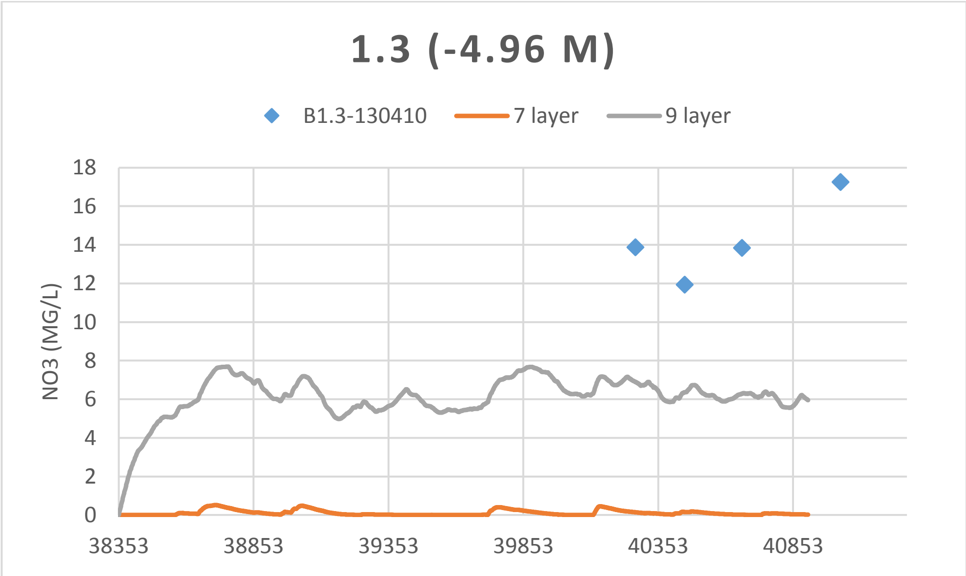


Figure B-3 Grid code areas of the pre-restoration model

Effect of adjustment from 7 to 9 layer

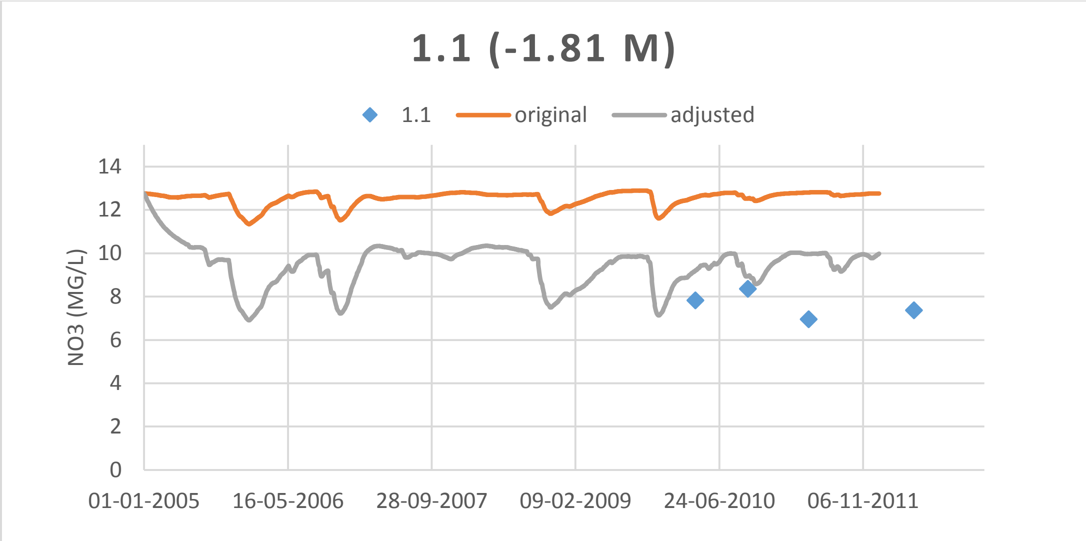
Especially closer to the river (piezometer 3.1) and at greater depths (piezometer 1.3), the 9 layer model gives a more accurate representation of the nitrate reduction than the 7 layer model. The following time series illustrate that. The piezometers are located in transect A, see figure 3 for precise locations.

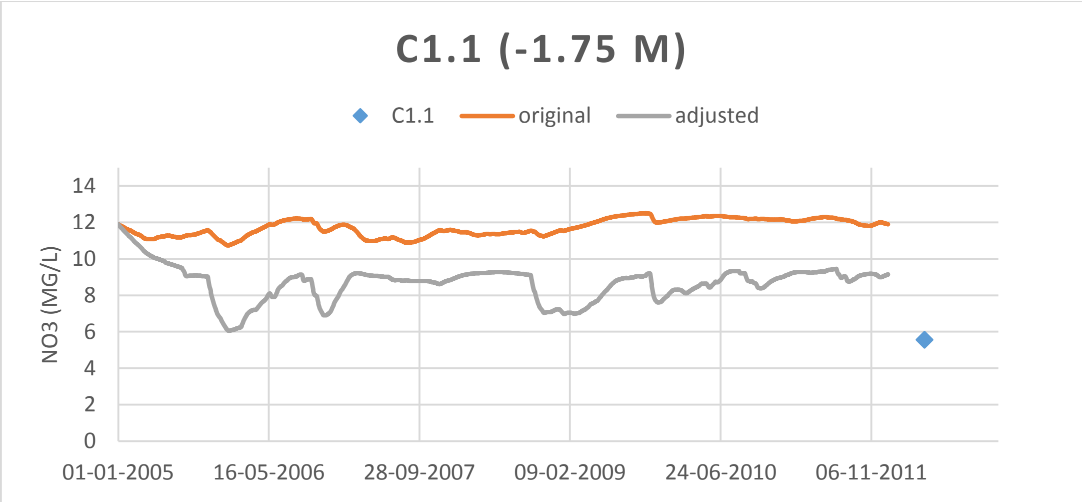


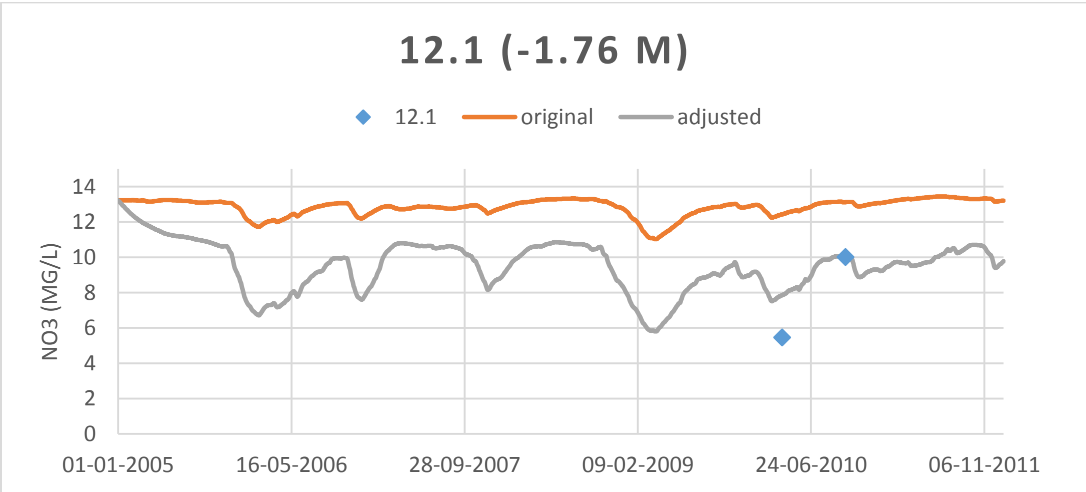


Effect of adjustment of nitrate inflow boundary and MM nitrate rates

The adjustment of the nitrate inflow boundary had the largest effect on the accuracy of the modelled nitrate concentration at the surface of the wetland.







Appendix C ECO Lab Templates

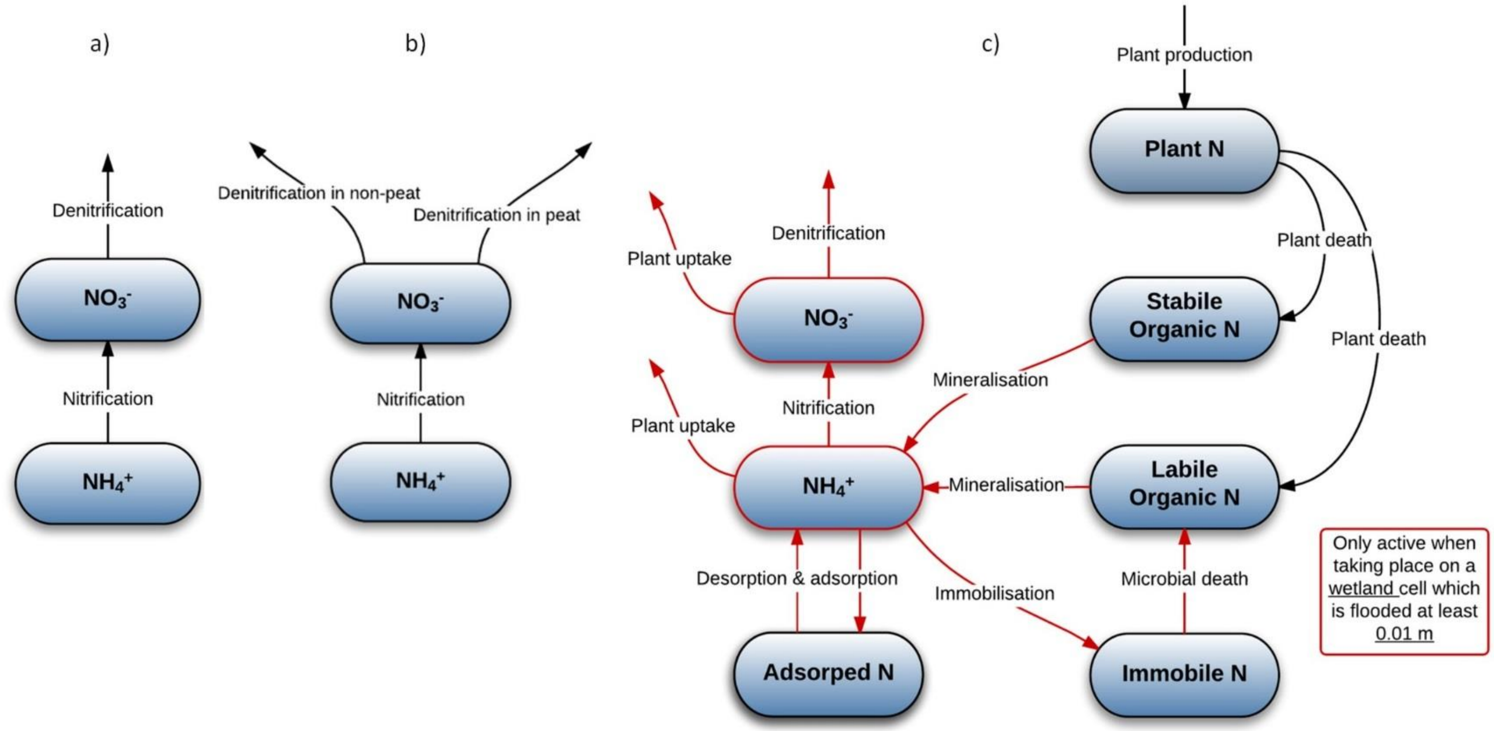


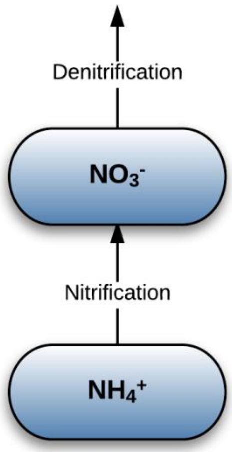
Figure C-1 Flow charts of the a) Unsaturated Zone, b) Saturated Zone, and c) Overland Flow and Ground Surface ECO Lab models

This appendix gives an overview of the equations used in the ECO Lab templates. In the following compartment descriptions, first the state variables for each compartment are given. The processes are in a table below the state variables and the constants and forcings are right below the process expressions. The values of the parameters can be found in Appendix D.

<u>Concentration dependent processes</u>

In UZ, SZ, OL, denitrification and nitrification only take place when NO_3^-/NH_4^+ concentration is over 0.001 mg/L.
In OL, Mineralisation of LON and SON only takes place when their concentration is over 0.001 g/m^3.
In OL, Immobilisation of NH_4^+ only takes place when the concentration is over 0.001 mg/L.
In OL, release of NH_4^+ through microbial death only takes place when the concentration of immobilised N is over 0.01 g/m^3.

UNSATURATED ZONE TEMPLATE



The processes in the unsaturated zone only take place when the nitrate or ammonium concentrations are above a certain threshold, 0.001 mg/L. In the model, this is written as an if-then-else statement, in both the nitrification and denitrification processes. For example, denitrification is represented with:

$$if(NH_4 > 0.001, ktempnitri * knitri * \frac{NH_4}{HalfNitri + NH_4}, 0)$$

With ktempnitri = $temp.coefficient^{(t-20)}$ with t = temperature; knitri = nitrification, first order rate at 20°C (unitless); and HalfNitri = half-saturation concentration (1/d).

The processes are temperature dependent. For the unsaturated zone, the (long-term average) temperature is set at 8°C.

The nitrification and denitrification equations are based on (Dørge, 1991). The ECO Lab equations are given here in differential equation form for easy comparison.

State variables	Expression (processes)	Initial value
$\boldsymbol{NO_3}$	$\frac{d[NO_3^- - N]}{dt} = \left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right) - \left(K_t^{(t-20)} * K_d \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}\right)$	0 (mg/L)
$\boldsymbol{NH_4}$	$\frac{d[NH_4^+ - N]}{dt} = -\left(K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}\right)$	0 (mg/L)

Processes	Dørge (1991)	Model
Nitrification	$K_t^{(t-20)} * K_{max} \frac{[NH_4^+]}{K_m + [NH_4^+]}$	$K_t^{(t-20)} * K_n \frac{[NH_4^+ - N]}{K_{mn} + [NH_4^+ - N]}$
	K_t = temperature constant with t = current temperature K_{max} = max. conversion rate per time unit K_m = half-saturation constant	K_t = temperature constant with t = current temperature K_n = nitrification rate at 20°C K_{mn} = half-saturation concentration for nitrification
Denitrification	$K_t^{(t-20)} * K_{max} \frac{[NO_3^-]}{K_m + [NO_3^-]}$	$K_t^{(t-20)} * K_d \frac{[NO_3^- - N]}{K_{md} + [NO_3^- - N]}$
	Same as Nitrification.	K_d = denitrification rate at 20°C K_{md} = half-saturation concentration for denitrification

The template also includes a control state variable, which can be used to identify total denitrification. It looks at the accumulative denitrification taking place in the unsaturated zone (in grams). The process registers the denitrification taking place in the water content of one cell ($length^3$), see figure C-2.

State variable	Expression (process)	Initial value
$\boldsymbol{UZ\_denitri\_acc}$	$UZda\_update$	0 (g)

Process	
$\boldsymbol{UZda\_update}$	$Denitri * (depth\_bot - depth\_top) * cell * cell * water\_content$

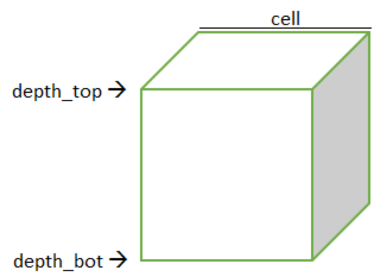
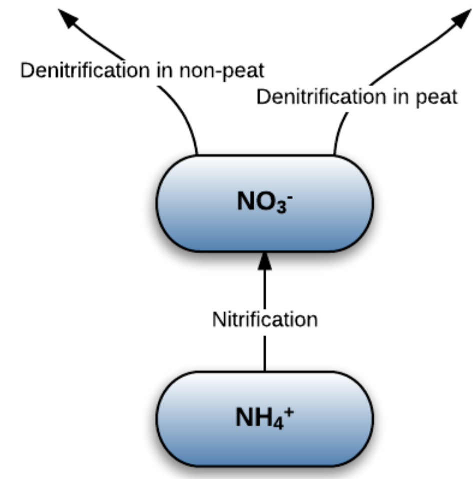


Figure C-2 Constants used in calculating accumulative denitrification, with depth_top being depth from ground surface to top of cell

Saturated Zone Template

State variables	Expression (processes)	Initial value
$\boldsymbol{NO_3}$	$Nitri - Denitri$	0 (mg/L)
$\boldsymbol{NH_4}$	$-Nitri$	0 (mg/L)
$\boldsymbol{Peat\_denitri\_acc}$	Pda_update	0 (g)
$\boldsymbol{SZ\_denitri\_acc}$	$SZda\_update$	0 (g)
$\boldsymbol{SZ\_nitri\_acc}$	$SZNda\_update$	0 (g)



Like the unsaturated zone, the processes in the saturated zone are based on Dørge (1991). For the saturated zone, the (long-term average) temperature is set at 8°C.

The total denitrification is composed of the denitrification happening in peat soil, and in non-peat soil.

Processes	
$\boldsymbol{Nitri}$	$if(NH_4 > 0.001, ktnitri * knitri * \frac{NH_4}{HalfNitri + NH_4}, 0)$
Constants and forcings used	Same as unsaturated zone, with ktnitri = ktempnitri. HalfNitri (mg/L), ten times lower than in the unsaturated zone
$\boldsymbol{Denitri}$	$Denit + DenitPeat$
$\boldsymbol{Denit}$	$if(NO_3 > 0.001, (1 - Peat) * ktdenitri * kd * \frac{NO_3}{HalfDenitri + NO_3}, 0)$
Constants and forcings used	Similar to unsaturated zone, with kd being the denitrification rate above the redox zone (1/d) Peat = 0 or 1 (indicate whether cell is peat or not)
$\boldsymbol{DenitPeat}$	$if(NO_3 > 0.001, Peat * ktdenitri * kdPeat * \frac{NO_3}{HalfDenitri + NO_3}, 0)$

Constants and forcings used	Same as Denit, with kdPeat = denitrification rate in saturated peat, based on Rassam et al. (2008). (see below)
$\boldsymbol{Pda\_update}$	$DenitPeat * SatThick * cell * cell * porosity$
$\boldsymbol{SZda\_update}$	$Denit * (depth\_bot - depth\_top) * cell * cell * porosity$
$\boldsymbol{SZNda\_update}$	$Nitri * SatThick * cell * cell * porosity$
Constants and forcings used	Similar to unsaturated zone, with SatThick = depth to phreatic surface (m) (MIKE SHE supplied forcing).

In the saturated zone there are three control variables, which monitor the accumulated denitrification in the peat layer and the non-peat layer, and accumulated nitrification (in both layers).

Saturated Peat Denitrification

The constant kdPeat is the denitrification rate in saturated peat, an expression which is based on Rassam et al. (2008). He models the distribution of denitrification potential with depth using an exponential decay function:

$$R_d = R_{max} \frac{e^{-kd} - e^{-kr}}{1 - e^{-kr}}$$

d = vertical depth below ground surface (L)
R_d = nitrate decay rate at any depth *d* (1/T)
R_{max} = maximum nitrate decay rate at soil surface (1/T)
r = depth of root zone (L)
k = parameter describing the rate at which nitrate decay declines with depth (1/L)

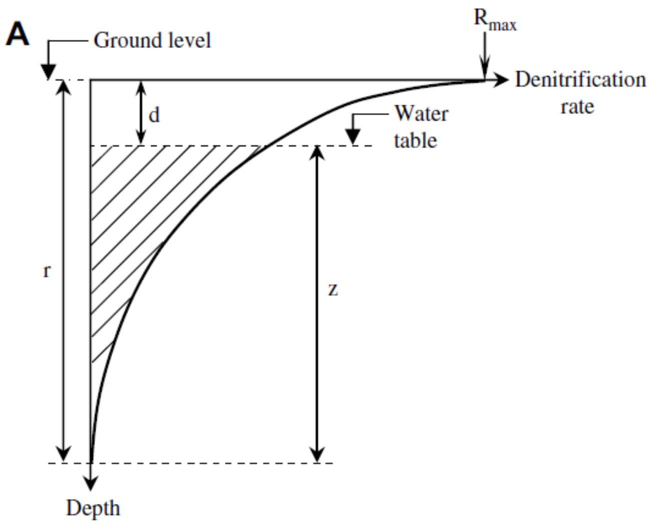


Figure C-3 Distribution of denitrification rate through the riparian buffer (Rassam et al., 2008)

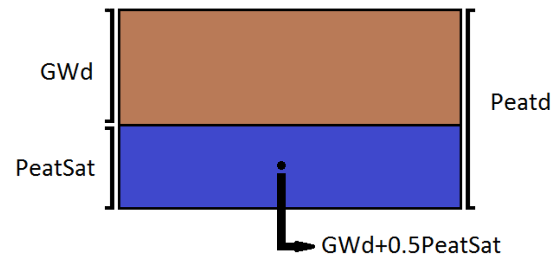


Figure C-4 Derivation of GWd+0.5PeatSat from the ground surface level

In the model this equation is transcribed as follows:

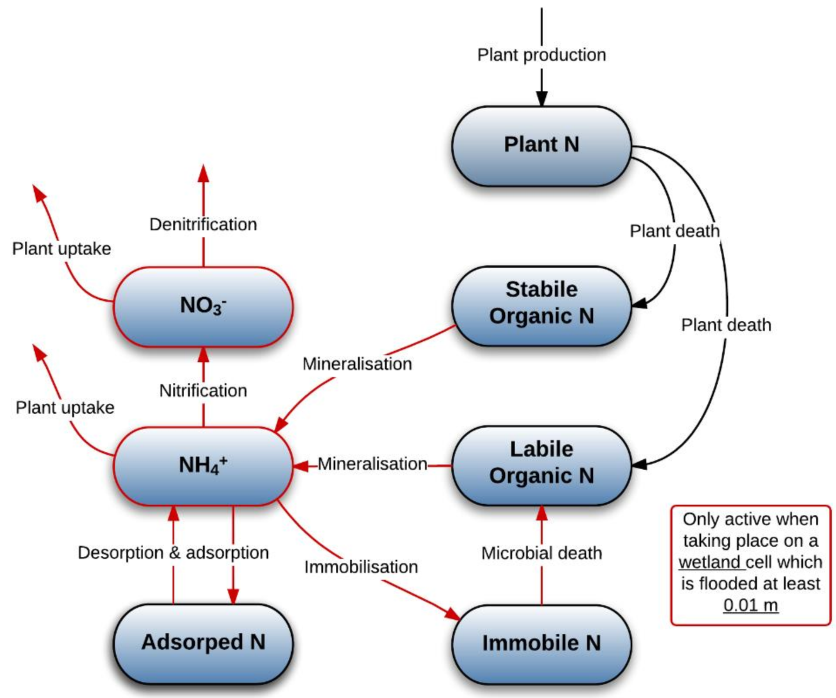
$$kdPeat = if(k == 0, 0, (kdpmax - kdpmin)\frac{e^{-k*\mathrm{MAX}(0,\ GWd+0.5*PeatSat)} - e^{-k*Peatd}}{(1 - e^{-k*Peatd}) + kdpmin}$$

k = rate at which denitrification declines with depth in peat or rootzone (-)
kdpmax = maximum denitrification rate in saturated peat (1/d)
kdpmin = minimum denitrification rate in saturated peat (1/d)
GWd = groundwater depth relative to surface = $\mathrm{MAX}(0, depth\_bot - depth\_top - sat\_thick)$ (-)
PeatSat = saturated peat layer = $\mathrm{MAX}(0, Peatd - GWd)$ (-)
Peatd = peat layer thickness (m)

OVERLAND FLOW TEMPLATE

This template is divided up in the (de)nitrification processes and control variables, with the other processes listed afterwards, for added clarity. The light processes are added at the end.

The nitrification and denitrification processes only take place when there is water flooding of at least 1 cm, in wetland areas and if model has run for more than a year (or initial total annual solar and total annual death are known on beforehand). The processes are in g/m^2/day so they have to be divided by the water depth to get g/m^3/day. The temperature is set at 10°C.



State variables	Expression9	Initial value
$\boldsymbol{NO_3}$	$if\left(depthc > 0, MAX\left(\frac{WetNitri - WetDeni - WetUptake\_N3}{depthc}, -NO3balance\right), 0\right) * (1 - ndc)$	1 (mg/L)
$\boldsymbol{NH_4}$	$if\left(depthc > 0, \frac{WetMineral\_LN + WetMineral\_SN - WetNitri - WetUptake\_N4 - WetAbsorbN - WetImmoN}{depthc}, 0\right) * (1 - ndc)$	0.5 (mg/L)
$\boldsymbol{Wet\_denitri\_acc}$	$Wetda\_update$	0 (g)
$\boldsymbol{Wet\_nitri\_acc}$	$WetdNa\_update$	0 (g)

Processes	Description	Expression
depthc	water depth, a check for division by 0	WetCover*depth
Constants and forcings used		WetCover = on-off switch for wet processes = $if\,(wetland == 1, if\,(depth > 0.01, 1, 0), 0)$ wetland = 0 or 1 (indicates whether cell is wetland or not) depth = depth of overland flow (MIKE SHE supplied)
WetNitri	Wetland nitrification	$if\,(NH_4 > 0.001, MAX\left(WetCover * ktemp * Nitri * \frac{NH_4}{HalfNitri + NH_4}, 0\right), 0)$
Constants and forcings used		ktemp = ktnitri and ktempnitri Nitri = nitrification, zero order rate constant NH_4 to NH_3. Similar to knitri (which is first order) (g/m^2/day) HalfNitri = half-saturation concentration, which is a different scale than unsaturated zone, but both rates lie at 20% of the maximum.
WetDeni	Wetland denitrification	$if\,(NO_3 > 0.001, MAX\left(WetCover * ktemp * Denitri * \frac{NO_3}{HalfDenitri + NO_3}, 0\right), 0)$
Constants and forcings used		Denitri = denitrification, zero order rate constant NO_3. Similar to kdenitri (which is first order) (g/m^2/day) HalfDenitri = half-saturation concentration (mg/L)
ndc	Day number (light process)	$if\,(day\_acc > 365{,}0{,}1)$
Constants and forcings used		day_acc = accumulated days (initial value = 367, if total annual solar and total annual death are known. If not, run model for one year before initiation of simulation)
Wetda_update		$if\left(depthc > 0, \frac{WetDeni}{depthc}, 0\right) * (1 - ndc) * depthc * cell * cell$
WetdNa_update		$if\left(depthc > 0, \frac{WetNitri}{depthc}, 0\right) * (1 - ndc) * depthc * cell * cell$
Constants and forcings used		Similar to unsaturated zone, with depthc acting as cell height (depth_bot – depth_top).

State variables	Expression (processes)	Initial value
$\boldsymbol{LON}$ **(labile organic nitrogen)**	$+\frac{(RPN * WetPDeadN + WetImmo\_Death - WetMineral\_LN)}{sedic} * (1 - ndc)$	10 (g/m^3)
$\boldsymbol{SON}$ **(stabile organic nitrogen)**	$+\frac{\left((1 - RPN) * WetPDeadN - WetMineral\_SN\right)}{sedic} * (1 - ndc)$	1000 (g/m^3)
$\boldsymbol{PlantN}$	$+(WetPProdN - WetPDeadN) * (1 - ndc)$	10 (g/m^2)
$\boldsymbol{ImmN}$	$+\frac{WetImmoN - WetImmo\_Death}{sedic} * (1 - ndc)$	1 (g/m^3)
$\boldsymbol{AdsN}$	$+\frac{WetAbsorN}{sedic} * (1 - ndc)$	1 (g/m^3)

Processes	Description	Expression
$\boldsymbol{RPN}$	RatioPlantNI as proces	RatioPlantNI
Constants and forcings used		RatioPlantNI = ratio of dead Plant N as Labile N [0-1] [0.2] (fraction)
$\boldsymbol{WetPDeadN}$	Wetland plant death nitrogen	$if(d\_acc == 0,0,\frac{DayPDead}{d\_acc} * PlantProNi)$
Constants and forcings used		d_acc = accumulated death (initial value 114.20) = $DayPDead * ndc$ DayPDead = wetland daily death of Plant N = $(d1 + d2 + d3 + d4 + d5 + d6)$ [see light processes] PlantProNi = annual plant production N (g/m^2/year)
$\boldsymbol{WetImmo\_Death}$	wetland microbial mortality induced release of NH_4	$if(ImmN > 0.01, MicroMor * ImmN * sedithickness * WetCover, 0)$
Constants and forcings used		MicroMor = first order microbial mortality rate (1/day) sedithickness = thickness of active upper layer (peat) (m)
$\boldsymbol{WetMineral\_LN}$	wetland mineralization of labile organic N	$if(LON > 0.001, (WetCover * \ ktemp * MineLabiN * LON * sedithickness, 0)$

Constants and forcings used		MineLabiN = first order mineralisation rate of labile N (1/day)
sedic	wetland sediment thickness as process	$if(sedithickness == 0{,}1, sedithickness)$
WetMineral_SN	wetland mineralization of stabile organic N	$if(SON > 0.001, (WetCover * \ ktemp * MineStabiN/365 * SON * sedithickness, 0)$
Constants and forcings used		MineStabiN = first order mineralisation rate of stabile N (1/year)
WetPProdN	wetland plant production nitrogen	$if(ios\_acc == 0{,}0, \frac{ios\_wet}{ios\_acc} * PlantProNi)$
Constants and forcings used		ios_acc = accumulated light (initial value 10803 for latitude = 55 and c = 0) = $ios\_wet * ndc$ ios_wet = radiation at surface (light process) = $ip * (1 - c) * (i1 + i2) * R$ [see light processes]
WetImmoN	wetland immobilisation of NH_4	$if(NH_4 > 0.001, \left(WetCover * \ ktemp * Immo * sedithickness * \frac{NH_4}{HalfImmo + NH_4}, 0\right)$
Constants and forcings used		Immo = Immobilisation of N: zero order immobilisation rate of NH_4 (g/m^3/day) HalfImmo = half-saturation concentration (mg/L)
WetAbsorN	wetland absorbtion of NH_4	$if(NH4 > 0.001, AbsorRateNi * \left(AbsorCapa * sedithickness * \frac{NH4}{HalfAbsorNi+NH4} - AdsN * sedithickness\right) * WetCover, AbsorRateNi * (-AdsN * sedithickness) * WetCover)$
Constants and forcings used		AbsorRateNi = first order adsorption rate of NH_4 (1/day) AbsorCapa = adsorption capacity of NH_4 per m^3 peat (g/m^3) HalfAbsorNi = half-saturation concentration for adsorption of NH_4 (mg/L)

LIGHT PROCESSES

The process DayPDead (wetland daily death of Plant N) has several plant death help variables (auxiliaries), where only one is active, depending on the current day number:

Auxiliary	Expression
$\boldsymbol{d1}$	$if(nd \geq 1, if(nd \leq maxiosday, 0.1{,}0), 0)$
	nd = day number = $daynumber(year, month, day)$ maxiosday = day number of maximum solar radiation (-)
$\boldsymbol{d2}$	$if(nd \geq maxiosday + 1, if\left(nd \leq maxiosday + 32, \left(\frac{0.9}{32}\right) * (nd - maxiosday) + 0.1{,}0\right), 0)$
$\boldsymbol{d3}$	$if(nd \geq maxiosday + 33, if(nd \leq maxiosday + 48{,}1{,}0), 0)$
$\boldsymbol{d4}$	$if\left(nd \geq maxiosday + 49, if\left(nd \leq maxiosday + 60, \left(-\frac{0.5}{12}\right) * \left(nd - (maxiosday + 48)\right) + 1{,}0\right), 0\right)$
$\boldsymbol{d5}$	$if(nd \geq maxiosday + 61, if(nd \leq maxiosday + 140{,}0.5{,}0), 0)$
$\boldsymbol{d6}$	$if(nd \geq maxiosday + 141, if\left(nd \leq 366, \left(-\frac{0.4}{366-(maxiosday+140)}\right) * \left(nd - (maxiosday + 140)\right) + 0.5{,}0\right), 0)$

The process ios_wet (radiation at surface) consists of three constants and two processes:

Term	Meaning	Value/expression
$\boldsymbol{ip}$	light fraction available for photosynthesis (-)	[0-1] [0.375]
$\boldsymbol{c}$	cloudiness factor (0=0%, 1=100%) (-)	[0-1] [0]
$\boldsymbol{R}$	solar radiation constant (E/m^2/day)	[0-1e^7] [517.039]
$\boldsymbol{i1}$	help function solar radiation (process)	$\sin(dcln) * \sin\left(lat * \frac{pi}{180}\right) * \frac{rdl}{2}$
$\boldsymbol{i2}$	help function solar radiation (process)	$\cos(dcln) * \cos\left(lat * \frac{pi}{180}\right) * \sin(\frac{rdl}{2})$
		dcln = sun's declination angle = $-0.406 * \cos(2 * pi * \frac{nd+12}{366})$ lat = latitude (degrees) rdl = relative daylength = $relative\_daylength(year, month, day, lat)$

RIVER TEMPLATE

MIKE 11 RIVER SIMULATION

This is a hydrodynamic, unsteady model. Simulation is from 01-01-2010 to 02-28-2011. The time step is 4.5 sec.

Of the input files needed for the simulation, the cross-section data and HD parameters are identical for the large catchment model and the wetland model for the wetland. The network file and boundary data are different.

MIKE 11 RIVER SIMULATION FOR WATER QUALITY

A model with advection-dispersion, simulation mode unsteady. Simulation is from 01-01-2010 to 01-10-2010. Time step is 4.5 sec.

Both the large catchment model and the wetland model use the same cross-sections, HD parameters, AD parameters, and ECO Lab parameters. The network file and boundary data differ for the two models. The HD results are stored with a frequency of 1 hour and the AD results are stored with a frequency of 100 time steps, in both models.

Appendix D Model parameters

Water flow parameters (catchment and wetland models) (Von Christierson et al., unpublished)

Model component		Parameters	Values
Overland flow		Manning's M	5 $m^{1/3}/s$
		Detention storage	0 mm
		Initial water depth	0 mm
		Leakage	0.0001 sec^{-1}
River flow (MIKE 11)		Manning's M	9 (summer) – 22 $m^{1/3}/s$
		River leakage coefficient	5 x 10^{-7} – 5 x 10^{-6} sec^{-1}
Land use		Leaf Area Index (LAI)	0.5 – 6
		Root depth	500 – 2100 mm
		Crop coefficient (K_c)	0.6 – 1.2
Unsaturated zone	Cells (0-0.4 m depth)	Saturated conductivity (K_s)	8.71 x 10^{-6} m/s
		Saturated water content (Θ_s)	0.34 – 0.424
		Residual water content (Θ_r)	0.01
		Water content at wilting (Θ_{wp})	0.05
		Water content at field cap. (Θ_{fc})	0.27
	Cells (0.4-0.8 m depth)	Saturated conductivity (K_s)	3.02 x 10^{-5} m/s
		Saturated water content (Θ_s)	0.34 – 0.424
		Residual water content (Θ_r)	0.01
		Water content at wilting (Θ_{wp})	0.05
		Water content at field cap. (Θ_{fc})	0.19
	Cells (0.8-31.5 m)	Saturated conductivity (K_s)	5.89 x 10^{-6} m/s
		Saturated water content (Θ_s)	0.34 – 0.424
		Residual water content (Θ_r)	0.01
		Water content at wilting (Θ_{wp})	0.07
		Water content at field cap. (Θ_{fc})	0.23
Saturated zone	Peat layer (0-1 m depth)	Saturated conductivity ($K_{s,horizontal}$)	1.16 x 10^{-6} m/s
		Saturated conductivity ($K_{s,vertical}$)	1.16 x 10^{-7} m/s
		Specific yield	0.15
		Specific storage	0.0001
	Sand layer (1-13 m depth)	Saturated conductivity ($K_{s,horizontal}$)	9.26 x 10^{-5} m/s
		Saturated conductivity ($K_{s,vertical}$)	9.26 x 10^{-6} m/s
		Specific yield	0.15
		Specific storage	0.0001

Water quality parameters (restored wetland model)

Model component		Parameters	Values
Overland flow		Temperature (T)	10 °C
		Denitrification rate at 20°C (K_d)	2 $g/m^2/day$
		Half saturation concentration for NO_3^--N (K_{mn})	4 mg/l
		Temperature coefficient (K_t)	1.06
		Nitrification rate at 20°C (K_n)	1 $g/m^2/day$
		Half saturation concentration for NH_4^+-N (K_{md})	4 mg/l
		Temperature coefficient (K_t)	1.06
		Annual plant production (P)	32 $g/m^2/year$
		Minimum concentration for NO_3^--N for plant uptake	0.005 mg/l
		Minimum concentration for NH_4^+-N for plant uptake	0.005 mg/l
		Fraction composed into LON (R_N)	0.02
		Mineralisation rate of SON (K_{MS})	1 $year^{-1}$
		Mineralisation rate of LON (K_{ML})	0.5 day^{-1}
		Immobilisation rate of NH_4^+-N (K_{immo})	5 $g/m^2/day$
		Half saturation concentration for NH_4^+-N (K_{mimmo})	25 mg/l
		Microbial mortality rate (K_{MM})	0.5 day^{-1}
		Adsorption rate of NH_4^+-N (K_a)	0.25 day^{-1}
		Half-saturation concentration of adsorbed nitrogen (K_{ma})	35 mg/l
		Adsorption capacity of NH_4^+-N per m^3 peat (R_{ac})	200 g/m^3
Unsaturated zone		Temperature (T)	10 °C
		Denitrification rate at 20°C (K_d)	1 $g/m^3/day$
		Half saturation concentration for NO_3^--N (K_{mn})	35 mg/l
		Temperature coefficient (K_t)	1.06
		Nitrification rate at 20°C (K_d)	0.5 $g/m^3/day$
		Half saturation concentration for NH_4^+-N (K_{md})	2 mg/l
		Temperature coefficient (K_t)	1.06
Saturated zone	Peat layer	Temperature (T)	10 °C
	(0-1 m depth)	Denitrification maximum rate at 20°C (K_{max})	10 $g/m^3/day$
		Denitrification minimum rate at 20°C (K_{min})	2 $g/m^3/day$
		Denitrification decline coefficient (k)	1.16 m^{-1}
		Half saturation concentration for NO_3^--N (K_{mn})	35 mg/l
		Temperature coefficient (K_t)	1.06
		Nitrification rate at 20°C (K_d)	0.5 $g/m^3/day$
		Half saturation concentration for NH_4^+-N (K_{md})	2 mg/l
		Temperature coefficient (K_t)	1.06
		Longitudinal dispersivity	1 m

		Transversal dispersivity	0.01 m
	Sand layer (1-1.5 m depth)	Temperature (T)	8 °C
		Denitrification rate at 20°C (K_d)	1 g/m^3/day
		Half saturation concentration for NO_3^--N (K_{mn})	35 mg/l
		Temperature coefficient (K_t)	1.06
		Nitrification rate at 20°C (K_d)	0.05 g/m^3/day
		Half saturation concentration for NH_4^+-N (K_{md})	2 mg/l
		Temperature coefficient (K_t)	1.06
		Longitudinal dispersivity	1 m
		Transversal dispersivity	0.01 m
	Sand layer (1.5-2 m depth)	Temperature (T)	8 °C
		Denitrification rate at 20°C (K_d)	0.1 g/m^3/day
		Half saturation concentration for NO_3^--N (K_{mn})	35 mg/l
		Temperature coefficient (K_t)	1.06
		Nitrification rate at 20°C (K_d)	0.05 g/m^3/day
		Half saturation concentration for NH_4^+-N (K_{md})	2 mg/l
		Temperature coefficient (K_t)	1.06
		Longitudinal dispersivity	1 m
		Transversal dispersivity	0.01 m
	Sand layer (2-13 m depth)	Temperature (T)	8 °C
		Denitrification rate at 20°C (K_d)	0.01 g/m^3/day
		Half saturation concentration for NO_3^--N (K_{mn})	35 mg/l
		Temperature coefficient (K_t)	1.06
		Nitrification rate at 20°C (K_d)	0.05 g/m^3/day
		Half saturation concentration for NH_4^+-N (K_{md})	2 mg/l
		Temperature coefficient (K_t)	1.06
		Longitudinal dispersivity	1 m
		Transversal dispersivity	0.01 m