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IMPACT OF CLIMATE CHANGE ON RIVER HYDROLOGY AND ECOLOGY: A CASE STUDY FOR INTERDISCIPLINARY POLICY ORIENTED RESEARCH

«SUDEM-CLI»

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FINAL REPORT

IMPACT OF CLIMATE CHANGE ON RIVER HYDROLOGY AND ECOLOGY: A CASE STUDY FOR INTERDISCIPLINARY POLICY ORIENTED RESEARCH "SUDEM-CLI"

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1. SUMMARY

A. Context

The impact of climate change on river hydrology and ecology is a subject that receives increasing attention and has strong implications for hydrological, ecological, economic and social policy. Because climate change affects such wide variety of disciplines, pursuing research in this field requires an interdisciplinary approach. This need to simultaneously understand and project the climate change, and to project and effectively deal with its impacts on the present and future aquatic ecosystem, presents a great challenge to the global research community. While it is important to understand sources and magnitudes of climate change uncertainty, there is also need to understand how and in what form policy makers can deal with uncertainties. The question arising here is how to address in both communication and decision making the uncertainties associated with regional climate change projections. The steps adopted by the policy makers are of two types: the setting up of mitigation measures through a reduction in GHG emissions and the setting up of adaptation measures aimed at decreasing the impacts and protecting both population and ecosystems faced with the climate risks of these coming years. This research aims to link nature management and development with water management as they both face adaptation challenges for climate change and because it is expected that both adaptation needs can be organized in a far more efficient way if the interrelationships between both are taken into account.

B. Objectives

The objective of the project was bringing together key experts from the climatological, hydrological and ecological research communities, as well as water managers and policy makers, in order to improve the decision making regarding the impact of climate change on aquatic and floodplain ecosystems. In the framework of the ADAPT project and in synergy with CCI-HYDR project, the impact of flood scenarios (frequency, duration, water height and season) on floodplain vegetation communities (habitats) and aquatic ecosystem was already under investigation with application to the ecological impacts.

Therefore a series of workshops have been organized, bringing together all sectors (climatologists, hydrologists / water engineers, biologists / ecologists and policy makers). Also hydro-meteorologists, sociologists and economists collaborating in the ongoing ADAPT and CCI-HYDR projects were invited to take part of these workshops and put their expertise in the general discussion around climate change and environmental friendly adaptation measures. The research focused on the case study of “Grote Nete & Grote Laak”. This allowed us to cover relevant issues regarding the environmental impact of Climate change induced changes in river hydrology. We specifically looked into the impact of changes in flooding regimes. This constraint on the research scope was due to several reasons, among which the specific focus of the CCI-HYDR and ADAPT projects on the impact of climate change on flooding regimes.

The research and workshops focused on the case study of “Grote Nete & Grote Laak”. This allowed covering relevant issues regarding the environmental impact of Climate change induced changes in river hydrology, both in terms of management options as for elements at stake (ecosystem types - vegetation types). The Grote Nete case raises awareness on several cross-policy challenges for water managers, nature development/conservation organisations, waste water treatment agencies which need strong interdisciplinary cooperation among hydrologists, ecologists and climatologists. More specifically, we adressed two important impact mechanisms which pose huge challenges for interdisciplinary research. The first topic is the effects of CC on urban run-off and associated inter-basin water transfers and how this can pose problems for hydrological modelling.

The second topic covers the positive and negative effects of macrophyte growth in relation to CC and how to potentially address variable flow resistance in hydrological modeling approaches.

In this project, we tested for the Grote Nete case the option to calibrate a simplified conceptual model to the full hydrodynamic model and run the long-term simulation in the simplified model. This test focused on the Zammelsbroek floodplain area. For this floodplain, we succeeded to obtain a conceptual (reservoir-type) model for which the results are close to the full hydrodynamic MIKE11 model. Hourly time series available for the period 1986-2005 were simulated in that conceptual model in order to obtain long-term information on water levels and inundation depths in the floodplain, the spatial extent and temporal variations in the inundated area, and the duration of the inundations. Given that the Zammelsbroek area has frequent inundations, 26 inundations (including very small ones) were simulated in the period 1986-2005 and the simulation results for these events were statistically summarized (table with inundation depths, duration, and extent). In this way, information could be obtained on flood events with high frequency, which is of high importance for the ecological impact investigation.

C. Conclusions

The impact of climate change on ecology through changes in flooding regimes is only one element of the many impact mechanisms that affect ecological values. Whether the ecological impacts through changes in flooding regimes will be important in comparison to other (climate related) stressors (drought, invasive species, ecological mismatching) or traditional stressors (eutrophication, acidification, fragmentation, pollution) remains an open question. Biodiversity values have been declining for decades and it seems that this trend is not changing, even without climate change. In relation to changes in flood regimes, one also has to acknowledge that the largest changes in flood regimes have been induced in the past through normalization of streams, increased run-off and a reduction in floodplain acreage.

Nevertheless we can learn important lessons from this study. If we put this in perspective and make the linkages to other disciplines, processes and mechanisms, we can see the other parts of the puzzle and identify the interdisciplinary challenges we need to tackle.

The first chapter is an introduction to key issues regarding global climate change uncertainty, with a focus on greenhouse gas emission scenarios. **Global climate change uncertainty comes from two main sources: emission scenarios and the limitations of our ability to model climate.** Emission scenarios must reflect the range of potential socio-economic futures. Most 3D climate model simulations to date have been based on the SRES scenarios presented by the IPCC in 2000. However these scenarios do not consider any explicit climate policy, and thus do not include very low emission cases. **Selecting a level of global warming that would be regarded as "acceptable" requires value judgments about the level of risk that is deemed acceptable; it is a political decision, although science can provide relevant information.** In 2009, the Parties to the UN Convention on Climate Change "took note" of the Copenhagen Agreement that lead in 2010 to the decision that the long-term objective would be the limitation of global warming to 2°C above pre-industrial average temperature. However, current emission reduction "pledges" by individual countries for 2020 do not represent enough efforts to make this objective likely (> 66% chances) to be satisfied. From a scientific viewpoint, a wide range of future emissions remains plausible. This wider range of possible scenarios is taken into account in a new process for developing and using scenarios for the next assessment report of the IPCC (AR5), which also aims at integrating researches on mitigation and adaptation in time for AR5.

The second chapter discusses issues related to climate scenarios used for hydrological studies, including natural climate variability and the comparison of model results and observations. Climate change scenarios for the 21st century specifically adapted to hydrological studies were developed in the CCI-HYDR and following projects. The underlying methodology deals with uncertainty associated to both emission scenarios and climate modelling by combining the results from an ensemble of runs with diverse emission scenarios in a set of 3 "climate change scenarios" representing comparatively wet, medium, or dry alternatives, with a definition that is tailored to each application. We consider the possibility and need to distinguish strong mitigation cases from other scenarios and conclude that while this was not a priority when the range of emission scenarios taken into account in climate models was limited, the appearance of strong mitigation cases in 3D climate simulations may require the separation of these cases from non-intervention scenarios, especially in the context of studies that look at the benefits from mitigation. RCM simulations match the statistics of observed precipitation extremes relatively well, depending on models. **The statistics from the CCLM model (used at UCL in the framework of ABC-impacts), are very close to observed values for version 3 of the model, but results from the new version 4 are deviating from observations in summer, requiring further investigations.** The statistical analysis done within CCI-HYDR shows that **precipitation extremes in Europe involve substantial natural variability (multidecadal oscillations) as well as a trend to more precipitation in winter.** This must be taken into account both for model validation and for impact analysis.

The third chapter elucidates the concept of ecosystem based adaptation through non-technical adaptation measures to Climate Change Impacts. **Ecosystem based adaptation to Climate Change is a concept where natural regulating processes are protected and/or restored and this provides opportunities to both society and biodiversity.** The physical characteristics of a catchment play a crucial role in the hydrological dynamics of its rivers. However, society's desire to control rivers and exploit floodplains for agricultural and industrial development has had enormous impact on riverine systems throughout Europe. **It is recognised increasingly that it is often more cost-effective to maintain, or even restore or create, water-related ecosystems than to try to provide the same services through expensive engineering structures, such as dams, embankments or water-treatment facilities**. River normalisation and increased soil sealing has turned flood regimes in a flood type that is atypical compared to any natural flood regime. Either regular floods are avoided through embankments and/or the extreme floods lack the flood-pulse properties to rejuvenate floodplain vegetation communities. Natural floodplains and riparian zones are dynamic environments and usually harbour a high biodiversity. The differentiation of the landscape by naturally functioning river systems enhances biodiversity on both the landscape and the local scale.

The fourth chapter gives an overview of recent progress and insights in the assessment of ecological impacts through changes in flooding regimes (UA). The impact mechanisms of flood events on ecosystems are described. Flood impact in lowland rivers occur mainly through drowning of vegetation (oxygen depletion in the root zone), external eutrophication and internal eutrophication. But, **regular flooding allows the fysiological adaptation of vegetation to flood events.** From the literature we deduct that especially flood timing, duration and regularity are crucial parameters. Based on the Biological Valuation Map we derive flood vulnerability map. **Practical challenges exist since most flood predicition models only provide data on the return period and maximal flood extent for extreme flood events.** In addition, there is often no detailed vegetation mapping available. A detailed study on flood timing, duration, depth and regularity is undertaken for the Zammelsbroek floodplain. **With respect to strategies to protect and restore floodplain biodiversity it is important to have regular, but less extreme flood events and the presence of topographical gradients within the floodplain in order to maintain species diversity.**

This brings us to the fifth chapter where the "state of art" and challenges regarding floodplain and river modeling are described. In this chapter, the challenges regarding floodplain and river modeling are described and illustrated based on the Grote Nete case application. **Most hydrological models are orientated towards flood prediction application.** They make use of techniques (synthetic rainfall events, composite storms, conceptual models) to allow fast calculation of many scenarios. **This is at the expense of the capability to evaluate ecological impacts and/or the evaluation of soft measures such as infiltration restoration, distributed (upstream) water retention, land-use change…** Furthermore the models are evaluated on their capability to accurately predict extreme events, while their performance on regular flows might be much lower. Long term simulations on the original models are seldom used, which makes it difficult to establish a reference condition to which changes can be compared.

The problems and challenges related to the hydrological modeling also relate to the increasing complexity of the hydrological system. *In chapter 6, we explore 2 important mechanisms that significantly increase the complexity of catchment functioning and require interdisciplinary research.* **We have identified and documented 2 mechanisms that significantly affect catchment hydrology.** The role of these mechanisms will become increasingly important, given the future climate projections. **Important sewage water transfers occur between and within catchment boundaries and these transfers have serious consequences for modeling and the water balance of the catchment.** The sewage infrastructure can be seen as a separate hydrological system that interacts with the river system. Not only is there a displacement of water across hydrological boundaries. Also water is transferred between compartiments: a) parasitic drainage (groundwater to sewage) b) run-off (rainfall to sewage) c) overflows (sewage to surface water) d) discharge at treatment plants (sewage to surface water). **Macrophytes can have a profound effect on the catchment hydrology under climate change scenarios.** Depending on climate conditions, species composition, morphology and nutrient availability they will alter flow resistance and hydraulic head through many non-linear mechanisms. **Prolonged periods of low flow, more sunlight, higher temperatures and higher nutrient availability (less dilution) will increase macrophyte growth and decrease the drainage capacity of streams. This is desirable for water conservation, but may pose local problems of summer flooding.** Further research on these mechanisms is needed to progress on the modeling of water quantity and quality. **Incorporation of sewage transfers and macrophyte growth into modeling approaches requires substantial effort, but is urgently needed as these mechanisms are very climate sensitive.**

D. Contribution to scientific support of sustainable development policy

A final chapter on policy support concludes the most relevant findings of the interdisciplinary research and formulates recommendations for nature and water management. Most important conclusions are summarized below.

Due to substantial uncertainty on the magnitude of regional / local climate change, policy measures need to take a range of possible futures into account, in particular by increasing resilience and looking for "no-regret" options. Actions and measures that are planned and executed today in the Grote Nete catchment are oriented towards the restoration of natural processes and may counterbalance the « additional » impact of climate change up to a certain level. The largest changes in flood regimes have been induced in the past through normalization of streams, increased run-off and a reduction in floodplain acreage.

Floodplain ecosystems may be relatively resilient against gradual changes in the magnitude of the flood regimes (depth-duration) if the floodplain exhibits a wide range of topographical gradients and is subjected to regular flooding. A more « ecological design » of controlled floodplains, can be a strategy to increase biodiversity.

Several hydrological-ecological interfacing problems were identified, which need further focus and research. It is interesting to notice that the current hydraulic model limitations are a natural result of improvements in flood risk modeling. Especially for ecological impact assessments and evaluation of soft measures (land-use change, infiltration restoration, upstream water retention), the recent progress in hydrological modelling has reduced the applicability, rather than improving the usefulness for these applications.

For the determination of ecological impacts of flooding also other variables such as the flood duration, the temporal evolution of the floodplain filling, the flood season, etc., are required. These outputs are by default not provided, neither validated. Also extraction of information on the flood season requires additional post-processing and validation. Even if the model architecture would allow the integration of these variables, calibration data on the duration of historical floods is most often not available. Consequently, the emptying of the flood plains cannot be modeled truthfully, which is a prerequisite. More attention thus should be given to the modeling, calibration and validation of flood duration and the underlying processes that affect flood duration.

It became clear that the water quality model is based on a huge number of assumptions, which are all due to lack of sufficient details (temporal frequency, locations) in the available pollution data. Averaged estimated loads and monthly measurements are not sufficient to allow accurate simulation of the daily or hourly concentration variations. It is this daily or hourly timescale that is of importance for an ecological impact analysis. In addition, the Grote Nete is however also largely influenced by short-duration pollution impacts from Combined Sewer Overflows (CSOs). The CCI-HYDR project has shown that due to climate change in the case study, CSO frequencies tend to increase, as well as the CSO pollutant concentrations due to prolonged dry weather periods during which sediments accumulate (higher storm flush for same CSO discharge; Willems et al., 2010). Also in the river, the same CSO discharge can lead to a higher impact, due to prolonged low flow periods in summer and increased eutrophication.

The present excessive macrophyte growth during low flows increases the hydraulic head, decreases valley drainage and results in more stable and higher groundwater levels. The water retention has significant positive effects on water quality and base flow if droughts persist, but may cause problems for harvesting crops in the valleys and may cause floods during summer storms. These interaction mechanisms need further study. The incorporation of a variable flow resistance (and macrophyte growth models) in hydraulic models is a huge challenge (research recommendation).

Climate change should be put in perspective and be linked to the « traditional » environmental stressors (eutrophication, dessication, acidification, soil sealing) which have not been tackled up to now and still cause further changes in the hydrological and ecological status of rivers and floodplains. Determining the impact of climate change on already heavily impacted ecosystems is rather ambivalent and in that case natural reference situations could be of use.

There should be continued efforts to monitor precipitation and flood changes, analyze their statistical properties, and compare observations to the models which are also used for future projections. Research projects that would allow integrated monitoring and modelling of several floodplain sites would contribute to a better understanding of the biogeochemical processes, which would allow to derive better evaluation criteria.

E. Keywords

Climate change, Climate adaptation, Climate modeling, Hydrological modeling, Floodplain ecology, Ecosystem based Adaptation, Macrophytes, Sewage transfers

2. METHODOLOGY AND RESULTS

CHAPTER 1: Emissions scenarios, international negotiations, and their consequences for adaptation and mitigation

This chapter is an introduction to issues regarding future projections of greenhouse gas emissions. It summarizes key global impacts on climate and how international negotiations may help reducing climate change and its effects. The end of the chapter presents the current evolution of IPCC-related work on scenarios and the integration of research on climate, impacts and adaptation.

1.1 Introduction: greenhouse gas emissions, global impacts and uncertainty

In this cluster project, the focus is on climate change impacts at the level of river subbasins and sewage systems. The focus on local/regional scale is a usual feature of this type of studies, because impacts are generally local in nature, as humans and natural species suffer from changes in a given place, which are often different from what they are in other locations because a number of geographically variable climatic and non-climatic parameters will influence these impacts. However climate is global in nature because changes in one region interacts with others, and change in long-lived greenhouse gases (GHGs) concentrations occur everywhere as they stay in the atmosphere for much longer than it is needed to transport them over the planet. Impacts also have a global dimension when it comes to issues such as food security, because some of the world agricultural production is traded internationally. Moreover, there is an ethical dimension in looking at impacts that are not only here and in the near future, but also in other regions and later. Therefore, while we are looking at local impacts, it is essential to keep in mind that climate change is a global problem.

Uncertainty is also a key aspect of future climate change projections; it comes from two main sources: emission scenarios (unknows regarding future socio-economic choices and changes) and climate models (unknowns about the climate system and its representation in models). Uncertainty in emission scenarios comes from unknowns regarding the evolution of human societies, such as population growth, economical changes, technological changes, policy and individual choices. The climate projections available to date, and used in this report, are all based on a set of scenarios presented in the Special Report on Emission Scenarios (SRES) by the IPCC (IPCC 2000). All these scenarios are “baseline” scenarios in the sense that they do not include any explicit climate policy (mitigation), although emission reduction may result from other environmental concerns that are taken into account in some scenarios. The CO_2 emissions from the most frequently used SRES scenarios are shown on Figure 1 (coloured lines).

Uncertainty in climate models has many sources. First, the conversion from emission to atmospheric concentrations introduces uncertainty, as the level of accumulation of greenhouse gases in the atmosphere results from complex biological, chemical or physical processes, in particular within the carbon cycle. Then uncertainties regarding the climate system may have consequences for global average change estimates, in particular the range of climate sensitivity (how much warming correspond to a doubling of CO_2 concentrations, associated to uncertainties regarding for example feedbacks from changes in water vapour concentrations) is estimated in IPCC AR4 to be in the range 2°C to 4.5°C with 66% chances, and the best guess estimate is 3°C. Other uncertainty sources are more regional, averaging out at the global scale but complicating local analyses (for example regarding precipitation changes in parts of Africa).

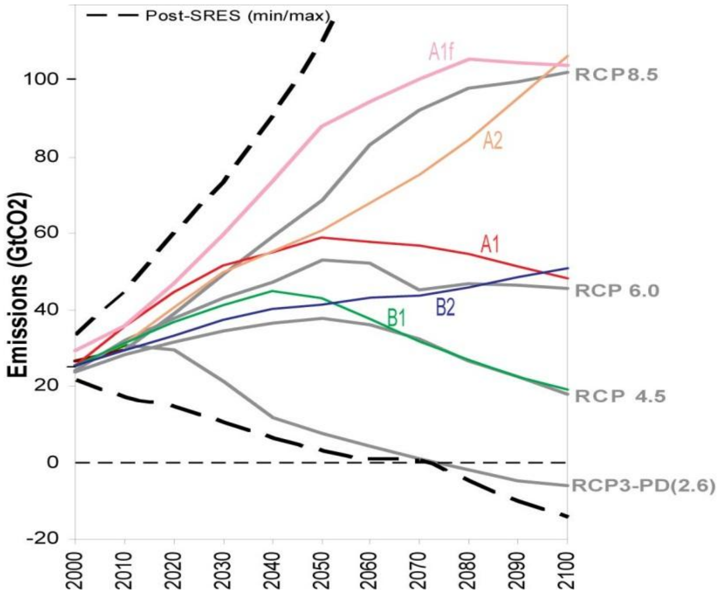


Figure 1: Coloured lines: Total carbon dioxide emissions for the SRES scenarios (4 "marker" scenarios and A1 Fossil Intensive scenario (IPCC 2000; IPCC 2007). Grey lines: illustrative carbon dioxide emissions for each of the representative concentration pathways (Moss and et al. 2008), see Section 1.3.

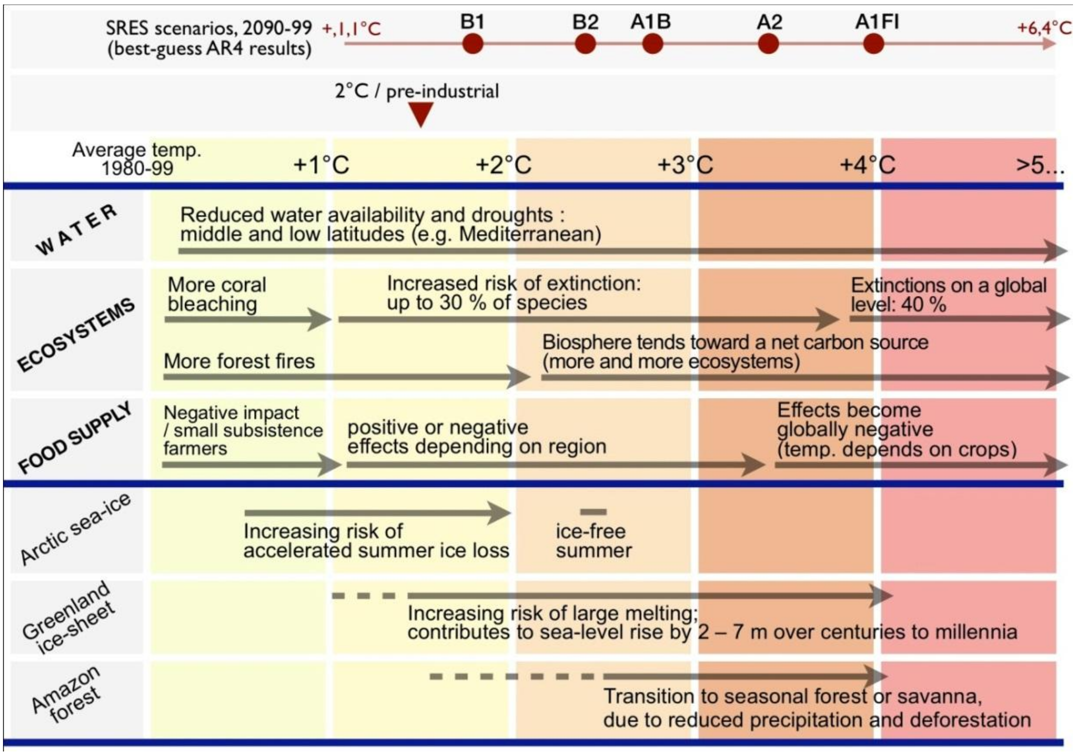


Figure 2: Top rows: global average temperature increase from 1980-99 until 2090-99 for key SRES scenarios, best guess results from IPCC, 2007 (the arrow indicates the full range, taking uncertainties into account). Middle rows: examples of important impacts (impacts starts approximately at the beginning of each arrow and increase with temperature; based on IPCC, 2007). Bottom rows: examples of important "tipping points" were a change in the climate system could accelerate substantially, become large and/or irreversible (the arrows indicates ranges from publications, and the dashes indicates that the change may start at lower temperatures according to some studies; based on (IPCC 2000; IPCC 2007; Salazar and et al. 2007; Lenton, Held et al. 2008; Boé, Hall et al. 2009; Malhi, Aragão et al. 2009; Nobre and Borma 2009; Fee, Johansson et al. 2010). The range given for tipping points reflects recent literature; note that uncertainties are large, and may not be entirely reflected in the ranges given due to incomplete knowledge of some of the processes.

Providing a global overview of impacts in diverse sectors is an extremely difficult task that can only be approached by large summaries such as IPCC assessment reports, therefore we restrict to a selection of important examples, shown in Figure 2 (for more information, see references in the caption). Some impacts, such as changes in the range of vegetal and animal species, can already be confidently linked to climate change, others are emerging or expected to start with even limited additional warming, such as more frequent coral bleaching events or adverse impacts on specific cultures. Larger levels of global warming would bring more severe impacts on ecosystems and would negatively affect the living conditions of an increasing share of people. In addition to impacts that are often increasing progressively with temperature, researches have looked at the possibility of thresholds levels in the climate system, called “tipping points”. These include phenomenon that could accelerate beyond a certain level of warming, such as suggested by several models for the decrease of sea-ice cover in the Arctic. They may also involve irreversibility, such as following the onset of large melting of continental ice in Greenland or West Antarctica, and subsequent long-term sea-level rise. Best-guess global warming levels reached by the SRES emission scenarios in 2100 are also shown in the figure (upper panel), suggesting that all these non-mitigation scenarios may result in significant impacts. However this does not define a level of “acceptable” warming, which would involve value judgements and political decisions.

1.2 International negotiations as an input for climate scenarios

Concerns regarding climate change started several decades ago, leading in 1988 to the creation of the Intergovernmental Panel on Climate Change (IPCC) by the World Meteorological Organization and the United Nations Environment Program. The mandate of the IPCC is to inform the policy making process by providing an objective assessment of the scientific findings on the risk of human induced climate change, its possible impacts, and options for adaptation and mitigation (IPCC, 2011). The first report of the IPCC was completed in 1990, and played a decisive role in the creation of the United Nations Framework Convention on Climate Change (UNFCCC). This convention was adopted in 1992 at the UN Conference on Environment and Development in Rio de Janeiro (UNFCCC 1992).

The objective of the UNFCCC convention is the “stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system” (UNFCCC 1992). The convention also states that the change should remain slow enough to enable adaptation of the ecosystems and food production as well has economic development. However, it does not provide any quantified objectives regarding the rate and level of mitigation efforts. Deciding about what is a “dangerous interference” is a political issue, as it is not possible to avoid all impacts (some are already happening), and also because there is significant uncertainty regarding how large the future impacts of current emissions could be, requiring decisions about the level of risk that is deemed acceptable. In 1995, the council of the European Union decided that its long-term objective would be to limit the increase in global average temperature to 2°C above pre-industrial levels (EU 1995). Although the UNFCCC convention introduced emissions reporting and the aim to return to 1990 emissions by year 2000 for the industrialized countries (those listed in its Annex I), binding commitments only appeared with the Kyoto Protocol (KP), adopted in 1997. The protocol establishes a “first commitment period” with emission reduction targets for a total of -5% of 1990 emissions in Annex I countries, to be met on average over the period 2008-2012. The Kyoto Protocol entered into force in 2005, but was never ratified by the United States. While all other large industrialized countries ratified the protocol, the fact that the U.S. remained outside the group reduces the demand for emission allowances in the framework of emission trading, thus potentially lowering the cost of these allowances. However, current emissions of some of the participating countries are still significantly above the amount that corresponds to their commitment, in particular for Canada (UNFCCC 2010) and possibly Japan (although GHG emissions in Japan decreased a lot in 2008 and 2009, resulting in a level close to the Kyoto target, e.g . (Nies 2010)).

In principle, these countries should buy the missing emission allowances, either from the unused amounts of countries such as Russia, and/or from projects that reduces emissions in developing countries or economies in transition. If, by contrast, countries do not comply with their engagements, this would hinder their participation into a subsequent commitment period – beyond 2012.

The first commitment period of the Kyoto Protocol is thus limited in time and in scale of action. As we will see in more detail in the next Section, curbing and ultimately stopping global warming requires much larger emission reductions, especially for the biggest emitters. While most "rich" countries, in particular the U.S., still have higher emissions/capita compared to developing countries (excluding land-use change, which complicates the figures), some emerging nations now emit a significant share of the world greenhouse gases – in particular, the total amount of emissions from China is now larger than from the U.S. Therefore, curbing world emissions does require both a continuation of efforts from countries that took part in the KP first commitment period, and a mechanism to promote mitigation efforts in the other countries.

In parallel of discussions on mitigation, adaptation has become an increasingly important part of the negotiations. An Adaptation Fund was created in the framework of the KP, based on a 2% share of purchases of emission reduction certificates from mitigation activities in developing countries (clean development mechanism projects, a part of emission trading activities in the KP) and other funding sources (UNFCCC 2002), see also (UNFCCC 2011c). Further development includes the so-called "Nairobi Work Programme on Impacts, Vulnerability and Adaptation" (2006) and subsequent decisions on helping adaptation, especially in developing countries.

In order to move beyond the first commitment period of the KP, the "Bali road map" was setup in 2007 at the 13th Conference of the Parties to the UNFCCC (COP 13). The intention was to achieve agreement by the Copenhagen conference (COP15) in 2009, following an agenda of negotiation issues known as the Bali Action Plan, and regular meetings of two "Ad-hoc Working Groups" (AWGs). The first of these groups existed already since 2005, and deals with further commitments for Annex I (i.e. industrialized) countries under the Kyoto Protocol (AWG-KP). The other group deals with Long-term Cooperative Action (AWG-LCA) in the framework of the convention – therefore only this second group involves the US.

While the Copenhagen conference resulted in refined texts under both AWG negotiation tracks, it did not finalize decisions. The work period of the AWGs was extended twice for one year, in preparation for the next COP (now scheduled for December 2011 in Durban). The "Copenhagen Accord" (CA) was both a significant development on several aspects such as the first inclusion of a limit to global warming (2°C) in an UNFCCC document, but it was generally regarded as deceptive because it did not contain binding commitments, the plenary did not adopt the CA but only "took note" of it, and very little was said on pathways needed to stay below the 2°C limit. Most, but not all, UNFCCC participating states associated themselves with the CA, by letter to the Secretariat of the Convention in early 2010. Most emitters, including developing countries, also provided "pledges" for emission reductions in 2020; however, these pledges are generally regarded as insufficient to put the World on a pathway that would not result in more than a 2°C warming from pre-industrial (see next Section). Industrialized countries also pledged to provide a "fast track" financing to increase near term (2010-2012) support to both adaptation and mitigation in developing countries.

Further progresses were made in Cancun (COP16) at the end of 2010, resulting in the Cancun Agreements and suggesting that the negotiation may eventually come to a successful conclusion, but there is still a long way to go. A substantial difficulty is that the United States are unlikely to be in a position to sign a binding commitment, as shown by the current activity of the Congress against climate change regulation (PEW Centre, 2011, and U.S. Congress, 2011). Progresses were done regarding support to developing countries for adaptation and mitigation, through financing, technology transfer and capacity building (UNFCCC 2011a).

As during each of the recent years, the next conference of the Parties (COP17, to take place in Durban), will be prepared during several meetings in 2011. The first of these meetings involving the AWGs was in Bangkok in April, the next will be in Bonn in June. An ongoing development is the setting up of the "Green Climate Fund" decided in Cancun to support a range of activities regarding mitigation and adaptation in developing countries. This fund will manage part of the financing that developing countries committed to provide in Copenhagen and Cancun, with a total amount increasing to 100 billion US$ per year in 2010, including diverse sources and financing modes (van Kerkhoff, Ahmad et al. 2011; UNFCCC 2011b). In spite of progresses, the negotiations are still facing substantial difficulties. In particular, it may become difficult to have a second commitment period within the KP fully ready by the end of 2012, so that an interim "fix" might be needed (ENB 2011). In addition, commitments are likely to remain on a "bottom-up" basis as the Copenhagen/Cancun emission reduction "pledges" suggests, at least for some time. In this context, it appear unlikely that commitments for 2020 will be strengthened in the coming years, and thus, as explained in the next Section, the probability that it will be possible to limit the global warming to less than 2°C - as the Cancun Agreement calls for - may decrease substantially.

1.3 How does the 2° C temperature limitation objective relates to climate science?

Two recent reports summarize the literature on the emissions scenarios that may be compatible with a limitation of global warming to less than 2°C (Den Elzen et al. 2010; Fee, Johansson et al. 2010). These two studies provide similar results, which is quite logical since the two reports are essentially based on the same literature. According to Fee, Johansson et al. 2010, to ensure a likely (> 66%) chance of limiting global warming to 2°C about pre-industrial temperatures requires:

- A peak in emissions by approximately 2015. The later the peak occurs, the steeper the decline in emissions would need to be in the subsequent decades. Delaying the emissions peak past this window will result in annual reduction rates that potentially exceed feasibility while substantially raising the costs of mitigation.

- A decrease in emissions of 50-70% in comparison to 1990 by 2050. This assumes further emission reductions after 2050.

- Reductions of long-lived greenhouse gases, such as carbon dioxide, which are essential, as well as reduction of short-lived forcing agents. In addition reductions of the short-lived greenhouse gases, black carbon aerosols, tropospheric ozone, and aviation-induced cloudiness, could also make an important contribution by lowering the rates of temperature increase in the near term. It would also counteract the warming resulting from reductions in sulphate aerosol concentrations due to reduced fossil-fuel use and air quality policies. Thus, efforts regarding all constituents contributing to global warming may be necessary, although the magnitude of abatement may be different for each gas. Technologies that achieve negative CO_2 emissions may be necessary in the long term (post 2030), and many studies suggest biomass energy with carbon capture and storage may be crucial for maintaining a 2°C limit.

The emissions reduction pledges associated with the Copenhagen Accord and Cancun Agreements fall short of a 2020 milestone that maintains a likely chance of achieving a 2°C limit without requiring potentially infeasible post-2020 reduction rates. Even the most optimistic interpretation of the current pledges suggests that to have a likely (66%) chance of limiting the warming to less then 2°C, an additional mitigation effort of 2 to 6 Gigatonnes of CO_2 equivalents would be required. Excluding the conditional pledges and other optimistic hypotheses, this gap is approximately 10 Gigatonnes of CO_2 equivalents. If a probability of staying below 2°C larger than 66% is required then greater emission reductions would be needed. By contrast, if a lower probability is considered acceptable (for instance 50%) then emission reductions could be somewhat lower.

While tightening the pledges would rather seems possible, given in particular the technical potentials, it is not clear that this will happen in the current negotiation context, as explained in the previous Section. In conclusion, a wide range of possible emission futures remains plausible from a scientific viewpoint.

1.4 Towards IPCC AR5: "new scenarios" process and integration of research on climate, impacts and adaptation

In this section, we summarize the ongoing and planned changes regarding scenarios in preparation for the 5th Assessment Report (AR5) of the IPCC. The new process will be an important change from previous assessments, and it is useful for those dealing with impacts and adaptation to be aware of that. The move originates from a need to replace the set of scenarios used so far in climate models – the SRES (Section 1.1), and to cover the whole range of published scenarios, including strong mitigation cases. The role of the IPCC in this process is to "catalyze" the preparation of new scenarios, in particular through organising expert meetings. The first of these meetings worked out the foundations for the new methodology (Moss and et al. 2008).

The central concept of this new framework is a set of 4 benchmark scenarios now referred to as "Representative Concentration Pathways - RCPs". By contrast to the previously used SRES emission scenarios, the RCPs are not based on storylines defining the drivers behind the emissions. Rather, the RCPs were defined by selecting concentrations pathways and the associated radiative forcing[1] in 2100 so as to cover the full range of scenarios available in the scientific literature. The RCPs are referenced by the radiative forcing reached in 2100, namely RCP8.5 (8.5 W/m^2, largest emissions), RCP6, RCP4.5, and RCP3-PD. In the name of the "RCP3-PD" scenario, PD stands for Peak-and-Decline: rather than increasing than stabilizing to a certain value, the radiative forcing is passing through a peak (at 3 W/m^2), than declining and eventually stabilising (the radiative forcing in 2100 was set to 2.6 W/m^2 following an evaluation of the plausibility of such low scenarios). The two lower scenarios are in the range of concentrations typical for mitigation scenarios, and the lowest one is representative of emissions that would follow from substantial mitigation efforts compatible with a limitation of global warming around 2°C, so that the coverage of possible future is much more comprehensive than with the previous non-mitigation SRES scenarios (Figure 1).

A key idea is that this set of pathways can be used to run climate models while new socio-economic scenarios are simultaneously developed. This parallel process is illustrated in Figure 3. When new socio-economic and emission scenarios will be ready, it is expected that it will be possible to link these to the RCPs so as to obtain climate change information from the climate runs based on the RCPs, thus avoiding a need for new climate simulations. A practical consequence for some of the impact and adaptation studies is that they do not only need to wait for the climate simulation results, but they may also need to wait for the availability of consistent socio-economic information from fully defined new scenarios with associated storylines. The RCP process helped to start this process more quickly than would the previously used "linear" approach (Figure 3) but it should be clear that the RCPs themselves do not provide complete socio-economic information so that further development is still needed in this area.

The process was designed to allow for an early start of the climate model simulations, but the selection of the lowest scenario was only confirmed in April 2009, and the data made available later. The delay in the selection of the RCP and preparation of emission/concentration now results in a start of model simulations later than expected. While climate modellers may still be in time for the AR5 (to be finalised in 2013/2014), the RCM simulations based on the RCP were not publicly available by the end of this project. The schedule will also be tight for impacts modellers wanting to take the

[1] *Radiative forcing* is a measure of the imbalance of incoming and outgoing energy in the Earth-atmosphere system, due to climate altering factors

climate model results into account in their own studies that could enter the AR5 writing process. However, progresses have been achieved in the validation of the recently developed 4th version of the CLM model.

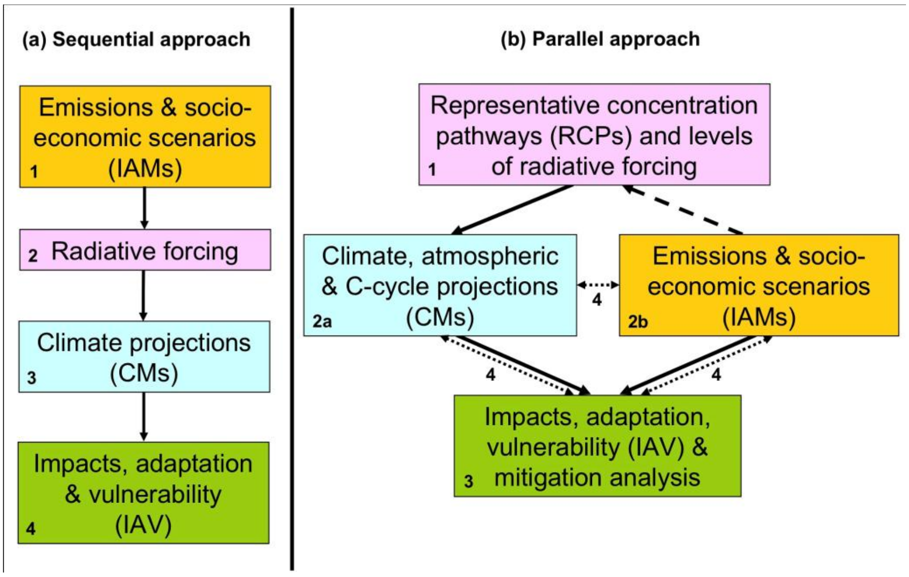


Figure 3: Approaches to the development of global scenarios: (a) previous *sequential* approach; (b) proposed *parallel* approach. Numbers indicate analytical steps (2a and 2b proceed concurrently). Arrows indicate transfers of information (solid), selection of RCPs (dashed), and integration of information and feedbacks (dotted) (from Moss et al., 2008).

As soon as sufficient climate model runs based on the RCP become available for Belgium, it is clear that additional research would be needed to study the effect of these changes in the scenarios (e.g. the effect of mitigation). Questions that may need to be answered are (among others): Is the range of scenarios used within CCI-HYDR (Belspo project SD/CP/03) sufficiently complete, and are climate simulations available to widen the range if needed? Could impact studies based on the CCI-HYDR scenarios be somehow "connected" to the new RCP process, and if not, what would be necessary to allow this? It will take time before a full evaluation becomes possible, including linking of the new scenarios selected for the AR5 to climate simulations, detailed assessment of extremes for an ensemble of models, and a range of impact studies. The treatment of uncertainty from scenarios in CCI-HYDR and the possible inclusion of lower emission scenarios in future work are further discussed in the next Chapter, Section 2.2.

CHAPTER 2: Precipitation extremes: variability and change in observations and models

In this chapter, we summarize the methodology developed for local climate change projections in earlier projects, and then discuss important issues related to these climate scenarios: combination of uncertainty from models and emission scenarios, differences between model and observations, and combination of natural variability and climate change.

2.1 Climate change scenarios adapted to hydrological impact studies

Climate change scenarios for the 21st century specifically adapted to hydrological and hydraulic river and sewage systems in Belgium were developed in the CCI-HYDR research project and extended in a project for the Flemish Institute for Nature and Forest Research (INBO). The methodology will only be briefly summarized here, as we will focus on issues discussed within this project (for more information, see (Ntegeka, Willems et al. 2008a; Ntegeka and Willems 2008c; Baguis, Roulin et al. 2010; Willems, Baguis et al. 2010).

The CCI-HYDR climate scenarios have been developed for specific study areas in Belgium, after statistically analyzing about 30 simulations with 11 different regional climate models (RCMs) and more than 20 simulations with global climate models (GCMs). Simulation results have been processed for the variables rainfall, temperature and potential evapotranspiration (ETo) till 2100. The climate model simulations assess future climate trends based on the projections of future greenhouse gas (GHG) from the SRES IPCC report (IPCC 2000). The regional climate model simulations with the SRES A2 and B2 regional scenarios were obtained from the European PRUDENCE project, where these RCMs were nested in a rather limited number of GCMs. To cover a wider range of GCMs and emission scenarios, additional GCM runs (A1B and B1 scenarios) were extracted from the IPCC AR4 database.

A specific algorithm was developed to obtain local climate change scenarios that can be applied to impact studies, on the basis of past observations and model simulations for the past and future. The algorithm imparts a perturbation based on the model results to the observed series to generate time series for the future. It intrinsically involves statistical downscaling (from daily to hourly time scale, and from grid scale to point scale) and bias correction (removal of the systematic deviation between the climate model results and the observations). It is applied to rainfall and potential evapotranspiration. For rainfall, the calculations involve two steps: the first step takes into account the changes in the number of wet days, and the second step takes into account the intensity of rain. The changes are quantile based to account for the fact that the changes might depend on the magnitude or return period of the event (Figure 4). Changes in the number of wet days are being made using a stochastic procedure.

The algorithm uses time series at hourly and daily time steps. These are time scales relevant for river subbasins. The scenarios were developed mainly for catchments up to 1000 km2. The time series perturbation procedure was developed from the PRUDENCE regional climate models which mainly dealt with a 30-year control period of 1961-1990 and a 30 year scenario period of 2071-2100. Interpolation is made for other periods to account for potential differences between the period covered by the input series and the standard 1961-1990 control period. In addition, a 30-year period roughly corresponds to an average climate “oscillation'' cycle (see following Sections and Ntegeka and Willems, 2008). More details about the perturbation procedure can be found in Chapter five of the CCI-HYDR Phase 1 Technical Report II “Study of climate change scenarios” (Ntegeka, Baguis et al. 2008b).

2.2 Dealing with uncertainty from climate models and emission scenarios

As summarized in Section 1.1 uncertainty in climate change scenarios comes from two main sources: the emission scenarios and the climate models. Taking these large uncertainties into account is a key issue for impact modelling, as the relevant climate information needs to be taken into account while avoiding unnecessary complexity. In the framework of CCI-HYDR, it was decided to simplify the climate scenarios by constructing sets of 3 scenarios to represent larger ensembles of model results: "high/wet", "mean/mild" and "low/dry" (Figure 4). The high scenario may be referred to as wet, and is thus adapted to studies of the risk of flooding, while the low scenario may be referred to as dry, and is thus critical for low flows. It is notable that the mean scenario represents mean conditions and is not the best future guess. The definition of high/mid/low is not unique, it is "tailored" to the application: it depends on time scale, return period and season / month, and is based on the expected hydrological impacts.

While providing a simplified view on the range of model results with a set of 3 climate scenarios is very useful for impact studies, it is also relatively difficult due to the need for adapting the selection of scenarios to the variables of interest and their application. For example, as correlations between the changes in precipitation and in potential evapotranspiration were found, the definition of high/low scenarios is based on the combined effect of rainfall and ETo. In other words, the variables are combined to generate an impact, which can then be classified as high, mean and low. Application of this methodology to other regions would require the same care in designing the scenarios.

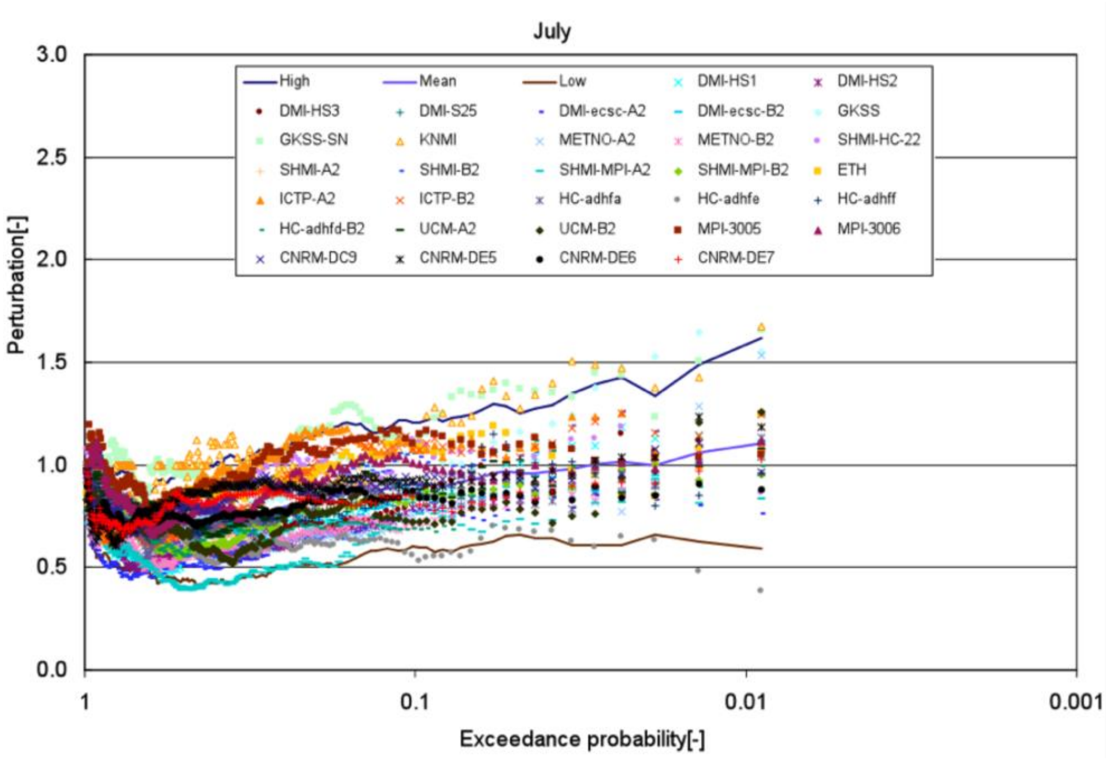


Figure 4: Example of perturbations in wet day rainfall intensities as function of the return period or exceedance probability based on the results of the PRUDENCE RCM runs for the month of July. The constructed low, mean and high scenarios are also shown.

Simulation of the three scenarios in the hydrological impact models allows assessing the range of uncertainty that is revealed by differences between the more than 50 climate model simulation considered and due to differences between the IPCC SRES GHG emission scenarios. This is an advantage of the methodology but also a potential difficulty, as there is only one uncertainty range for all the emission scenarios. This raises the question of whether we need to discriminate between emission levels when looking at impacts studies or not. The answer to this question likely depends on the application of the impact study as well as on the emission scenarios considered. Studies on adaptation would need to consider the whole range of possible futures; a possible way to deal with the considerable uncertainty that is obtained at the regional and local scales is then to start from the knowledge of current vulnerabilities and take action in a way that would reduce risks for a wide range of possible future evolutions, to the extent possible. By contrast, studying the benefits from mitigation requires discrimination among emissions levels in the climate scenarios.

When considering all impacts globally and in many sectors, there can be no doubt that lowering emissions reduce the risks (Chapter 1). But our ability to study this difference between emission levels in specific sectors at local scale, given the uncertainties, is a more difficult issue. Until very recently, the regional climate simulations were always performed for the SRES scenarios, which are non-mitigation scenarios, and most often ignoring the lower of these (the B1 family, although it was taken into account here on the basis of existing GCM runs). The first simulations with a low scenario assuming mitigation towards a 2°C global warming limitation (pre-industrial) were performed during the ENSEMBLES project.

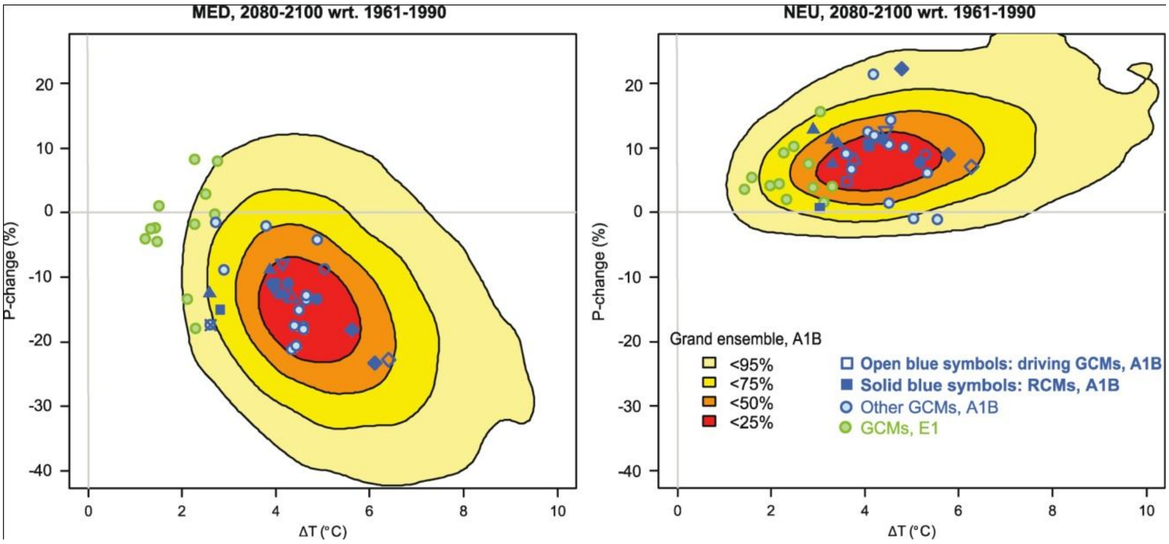


Figure 5: Annual changes in temperature (T) and precipitation (P) in the Mediterranean Basin (left) and northern Europe (right) from 1961-1990 to 2080-2100. Coloured areas depict probabilistic projection percentiles based on a statistical emulation of various sources of uncertainty, for the A1B scenario. The symbols show RCM and GCM simulations for the A1B scenario (blue) and for the E1 scenario (green, GCM only). Source: EU ENSEMBLES project (van der Linden and Mitchell 2009).

The ability to differentiate results from a though mitigation scenario (E1) from those of a "medium" non-mitigation scenario (A1B) is illustrated on Figure 5. For the Mediterranean region, the simulations based on E1 clearly result in reduced climate change compared to A1B (results from E1 would have very little chances to occur under A1B). For northern Europe, the difference is more modest, although E1 results tend to form a cluster outside the range of most A1B results. On smaller regions, we would expect a reduced ability to distinguish between scenarios, in particular over Belgium because it is located between the northern regions with increased precipitation and the southern ones with reduced precipitation. However, the data shown in this figure are annual averages; as seasonal changes are generally larger, there might be a possibility to distinguish very low emission scenarios from non-mitigation cases, and the situation may also be different for extremes. This would require more investigations that were not possible during this project in part because the discussions about the lowest emission case within the IPCC RCPs took more time than expected. A specific issue that might be important when analysing simulations with low emissions is that sulphate aerosols concentrations have links with fossil fuel emissions, so that the cooling effect from aerosols is expected to decline faster in though mitigation cases. This may complicate the picture of differences between mitigation and non-mitigation cases (John 2011).

2.3 Evolution of regional climate models and validation for use in hydrological studies

K.U.Leuven analyzed the climate model simulation results available from the more recent EU project ENSEMBLES (John 2011). Within the ENSEMBLES project, new indicators for evaluating the performance of RCMs were evaluated. Most indicators were based on regional temperature and precipitation statistics, but one of the indicators considered large-scale circulation and weather regimes. The objective was to identify models that perform well for all these "metrics", therefore combinations of metrics were considered to provide an aggregated score. While this exercise was regarded as exploratory, the results suggest that at least one of the models (ICTP–RegCM) for which precipitation statistics did not well match observations in Uccle within CCI-HYDR is performing quite well for other criteria. Model simulations can be erroneous, therefore eliminating simulations on the basis of performance metrics may be necessary to avoid taking inaccurate results into account, but designing such metrics and deciding about excluding some simulations remains a difficult task, which needs to take many aspects into account.

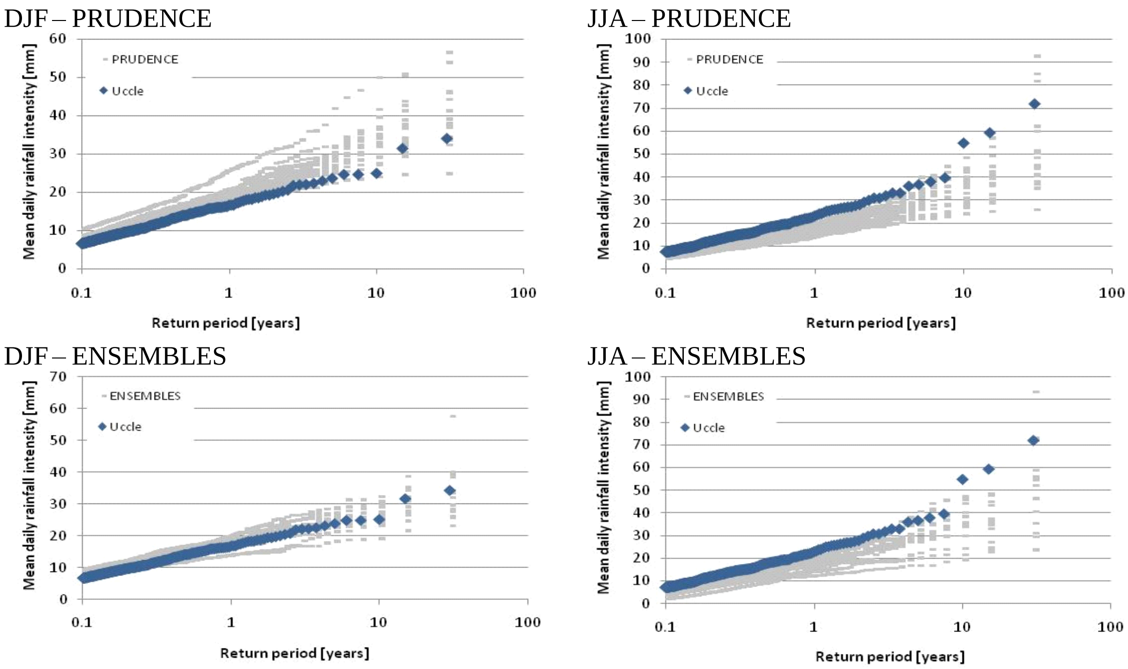


Figure 6: Validation of daily precipitation extremes. Top row: PRUDENCE RCM runs for the grid cell covering the Uccle meteo-station. Bottom row: ENSEMBLES RCM runs

Figure 6 shows the results of a comparison between the PRUDENCE and ENSEMBLES results for all RCM runs with available results for the main Belgian meteorological station at Uccle. It is clear from this figure that, while the PRUDENCE runs show systematic overstimation for the winter (DJF) season, these strongly reduced in the more recent ENSEMBLES runs. For the summer season, the systematic underestimation remain, which may be due to the fact that current RCMs are still too coarse to resolve convective precipitation, so that they must use approximate parametrisations (short time-slice convection-resolving experiments are however possible, see e.g. (Knote, Heinemann et al. 2010). For the winter season, lower rainfall intensities are expected for the RCM results in comparison with the observed point intensities, because of the spatial scale difference (grid averaged precipitation versus point precipitation), so that the results suggest that precipitation is better represented in the last simulations performed.

Through research cooperation between UCL and K.U.Leuven, the model-observation deviations were further investigated in this project based on simulations with the CLM regional climate. This was done in the same way as was done in the CCI-HYDR project for GCM and RCM results. Results from two different versions of the CLM model were considered: CLM3.0 and CLM4.0 model (CLM is also referred to as CCLM or COSMO-CLM, see ABC-Impacts (A.B.C.-Impacts 2011) and http://www.clm-community.eu)

In order to eliminate the influence of the GCM in which the CLM is nested (influence of the boundary conditions), CLM results driven by lateral boundary conditions from re-analysis data (ECMWF ERA-40 and NCEP/NCAR) were considered. This re-analysis data represents "real-time" meteorology based on the assimilation of observations rather than "unforced" general circulation models. In this way we focus on the ability of the regional climate model to produce extreme precipitations with statistical properties that match observations (the state of the atmosphere at large scales is similar in models and observations, therefore the model-observations differences resulting from natural variability are strongly reduced).

Figure 7 shows that CLM3.0 ERA40 results match well the Uccle historical daily rainfall extremes, also in the summer season. This shows that this RCM is able to reproduce rainfall extremes well when the model is forced by historical large-scale atmospheric circulation information at its boundaries (however, there may still be some overestimation in the model because grid-average extremes would be expected to show lower amounts than station data). Rainfall biases in the RCM results (PRUDENCE, ENSEMBLES) thus appear to mainly result from biases in the GCM forcings. This is consistent with other studies that found that the GCM explains a large fraction of the biases, although the regional model may have a larger role in summer due to local effects such as convection e.g. (Rummukainen 2010).

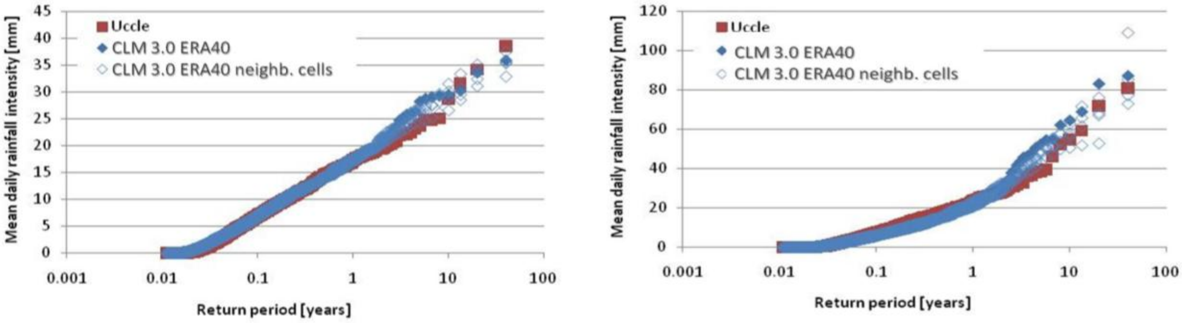


Figure 7. Validation of daily precipitation extremes, based on the CLM3.0 ERA40 results for the grid cell covering the Uccle meteo-station. Left : DJF (winter); right : JJA (summer)

Precipitation totals aggregated over time scales longer than a day are shown by IDF curves on Figure 8. As for the 1-day time scale, the ERA40 forced CLM3.0 results tend to show smaller biases than the mean value from ENSEMBLES simulations, although some overestimation of the strongest events (10 years return period) is again noticeable, especially for the shorter time scales shown (1 day).Finally, the analysis was extended with a new run done with the last version of CLM (4.0, which was finalised in 2010). For this simulation, we have an archive of hourly precipitation over more than 60 years, starting in 1948, and based on lateral boundary forcing from NCEP/NCAR reanalysis2. Figure 9 shows a preliminary comparison between the CLM3 and CLM4 runs. CLM4 results constantly show less precipitation than CLM3 ones, which were close to observations but slightly too large. For the winter season, the difference is small and the results from CLM4 may be as close to the actual climate as CLM3 ones. However, for summer, CLM4 results show a systematic underestimation in the rainfall extremes (compare with Figure 7).

[2] We thank Dr. Beate Geyer, from Helmholtz-Zentrum Geesthacht, Germany, and Dr. Daniel Luethi, Institute for Atmospheric and Climate Science, ETH, for giving us access to the long CLM runs.

The IDF curves (not shown) suggest that the results strongly depends on the aggregation time, ranging from very close to observations at a time scale of a few hours to substantial underestimation of the hourly values. As we have shown that the bias essentially occurs in summer (and for relatively short events), the change between model versions might be related to the representation of convection. However, various changes have been made between model versions, including changes in the representation of clouds, and due to the large CPU time required by such simulations, we could not yet explore these differences, including the ones that may come from the differences in lateral boundary forcing (here one run uses reanalysis from ECMWF while the other uses NCEP). While changes in a model that introduces more detailed and comprehensive representation of climate processes may in principle degrade specific results, further investigations are required to better understand this CLM4 simulation. The internal variability of the model may also explain part of the differences between two simulations.

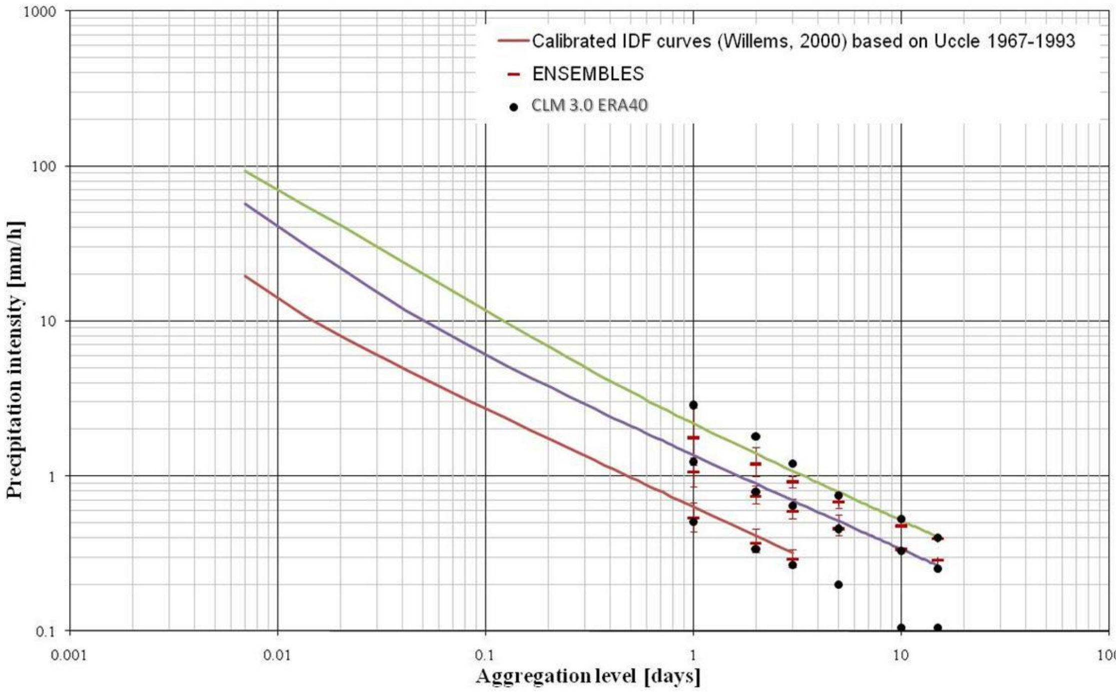


Figure 8. Comparison of rainfall IDF relationships for Uccle with ENSEMBLES and CLM3.0 ERA40 results. Three return periods are shown: 0.1 year (red line), 1 year (blue line) and 10 years (green line).

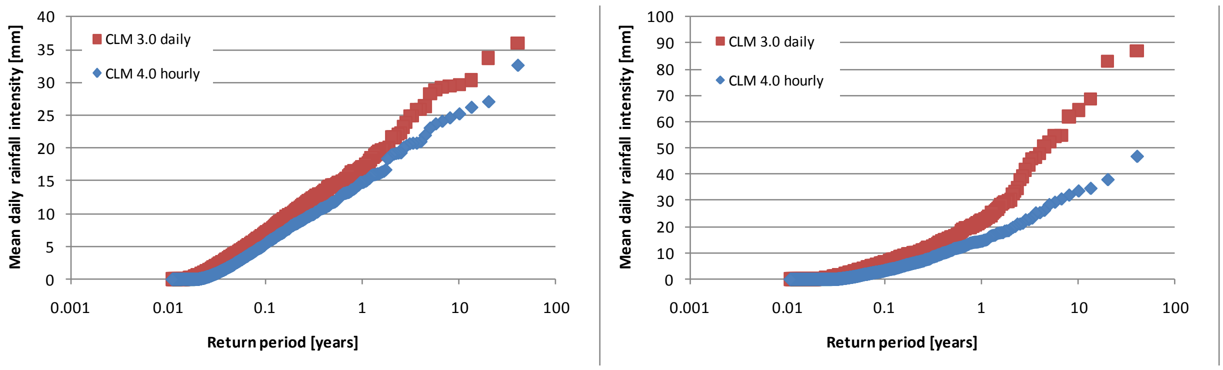


Figure 9. Preliminary comparison of daily precipitation extremes, based on daily CLM3.0 and CLM4.0 results for the grid cell covering the Uccle meteo-station. Left: winter (DJF), right: summer (JJA).

Several methods exist to perform the statistical downscaling of the climate model results. These methods transfer the climate changes at RCM/GCM scales, generally larger or equal to 20 km, to changes at hourly and point scales (hydrological impact scales). Each of these methods is based on a number of underlying assumptions, which introduce uncertainties in the statistical downscaling.

These uncertainties have not been analysed in this project, but have been analysed in large scale research projects such as ENSEMBLES and previous projects (e.g. EU STARDEX). Some statistical downscaling methods, e.g. based on weather typing, have the advantage that no direct use is made of the precipitation results of the climate models. Instead, changes in precipitation are assessed from changes in large-scale atmospheric circulation, temperature and other climatic variables for which the RCM/GCM results have a higher accuracy than the precipitation accuracy. However, a key issue with these methods is the hypothesis of stationarity, that is, to what extent statistical relations derived from present climate are still valid in future climate. By contrast, dynamical downscaling (i.e. regional climate models) involves physically-based relations which contribute to provide confidence in model results for perturbed climate. It is generally considered that statistical and dynamical downscaling methods both have merits and involve uncertainties.

2.4 Natural variability and its consequences for climate projections and risk assessment

In the CCI-HYDR project, statistical trend analysis on the precipitation series of the Royal Meteorological Institute of Belgium at Uccle has shown that series of precipitation extremes are influenced by multidecadal climate oscillations, as shown in Figure 10 (Ntegeka and Willems, 2008; Willems and Yiou, 2010). They were found significant at the 5% significance level. The climate oscillations have been observed for the whole European region, with anti-correlations between northern and southern Europe (Willems and Yiou 2010), and are mostly attributable to climate variability.

In this project, we analysed how climate models are representing these climate oscillations in the frequency of extreme events. As the explanation of these oscillations in extreme precipitation over Europe probably involve large scale climate dynamics, related to mechanisms that are not yet entirely identified but may involve complex interactions between climate system components (e.g. involving the oceans), these investigations may be very useful to better understand and improve the climate models.

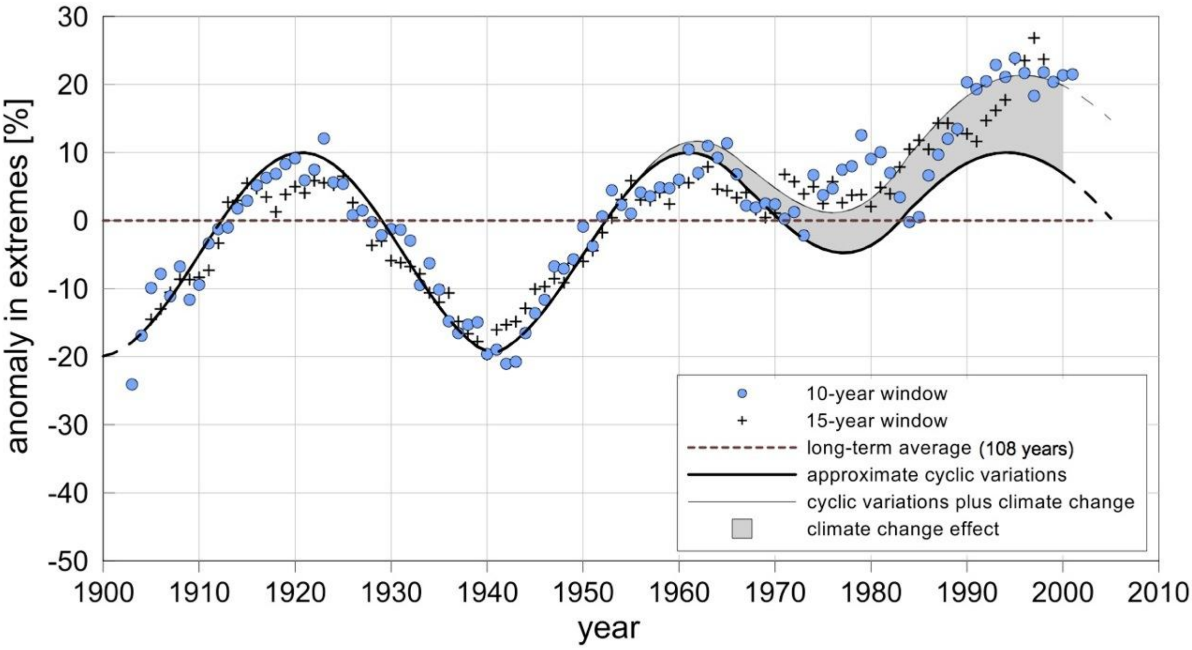


Figure 10: Anomalies and multidecadal oscillations in rainfall extremes based on the historical Uccle series, for the winter season (DJF) (10-minutes, 1898-2005) adapted from (Ntegeka and Willems 2008c).

In order to investigate this issue with more chances of success, we first analysed regional climate simulations driven by ECMWF ERA-40 re-analysis data. In this way we focus on the ability of the regional climate model to produce extreme precipitations with statistical properties that match observations. This work may also contribute to the understanding of the origin of the observed extremes. Figure 11 shows the comparison between our extreme precipitation index for the Uccle historical data and for CLM3, with lateral boundary forcing from ERA40 for the period 1961-2000 (red line). Despite the limited time period considered, it appears clearly that the CLM results follow the historical observations quite closely, showing part of an oscillation. This confirms that given the appropriate large-scale conditions, regional models can provide the correct precipitation statistics.

We then turn to a general circulation model (ECHAM5) to analyse its ability to provide multi-decadal oscillations. An ensemble of 17 simulations driven by SRES A1B emissions, each with slightly different initial conditions, is shown in Figure 11 (grey lines). The results also show an oscillatory behaviour in which the period and amplitude characteristics are similar to past observations for at least some of the ensemble members, although the amplitude of the oscillations provided by the model may be smaller, and oscillations may sometimes be faster. This preliminary analysis thus suggests that general circulation models may represent the physical processes that are responsible for the oscillations, and could therefore be a useful tool to investigate their origin. This is encouraging but further investigations are necessary to confirm that the oscillations found in the model results are due to the same mechanisms as the one operating in the real climate system, and understand these. The oscillations shown by the model are not in phase with the observed ones, but this was expected since the oscillations are attributed to (unforced) natural variability (the initialisation of the model does not relate to a particular date in the past, and differs between the ensemble members). The fact that the RCM simulations forced by re-analysis were in phase with the observations confirms that the oscillation signal is found in the large-scale atmospheric circulation (although its origin is likely to involve other climate system components).

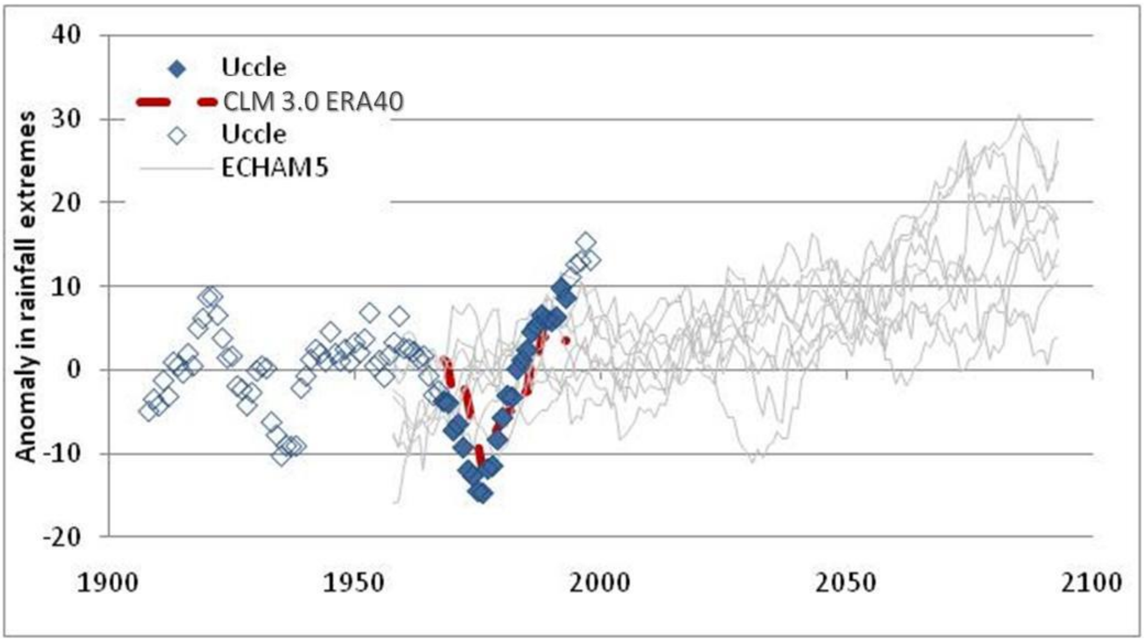


Figure 11: Anomalies in rainfall extremes based on the CLM3.0 results at Uccle, driven by ERA40 data, and comparison with 17 ECHAM5/MPI-OM A1B runs (ESSENCE project, (Sterl, Severijns et al. 2008).

These multidecadal oscillations must be taken into account when building climate change projections and conducting impact analyses. Indeed, if the study period is shorter than an oscillation, the rainfall-runoff modeller has to keep in mind that the results may be biased from the long-term averaged climate. Due to the oscillations, an input series period covering an oscillation peak may provide overestimated risk figures (but the maximum will be addressed), while using an input series that covers an oscillation low would result in underestimation of the risk. This is also important for model validation when it is not performed on the basis of re-analysis forcing (e.g. in the case of GCMs), because if the validation period does not cover an integer number of oscillations, then phase differences between the model and the observations may result in model-observations differences, even in the computed average climate statistics, while there is no actual model deficiency (bias) behind these differences. Further insight into these issues may possibly come from ongoing research efforts regarding predictability of decadal scale climate change (which will be addressed in the 5th assessment report of the IPCC).

The presence of climate oscillations makes it difficult to detect climate change trends in historical series. It also makes it difficult to create awareness on the potential impacts of climate changes. Long periods of relatively lower probabilities for extreme precipitation events and related floods also lower the public and political awareness. It is moreover difficult to explain to the public the difference between natural variability and patterns, and the superimposed climate change. Due to this complexity, a risk exists that awareness on flood vulnerability only appear in irrational waves of short-term actions without long-term perspectives. Decision makers need to be aware that following decades with relatively less extreme rainfall in their region, the risks associated with extreme rainfall are expected to peak again, and that at least in the winter season, the anthropogenic climate change is expected to increase the magnitude of the following peak.

CHAPTER 3: Non-technical adaptation measures to Climate Change Impacts: The ecosystem services approach

In this chapter we investigate hydrological processes and how manipulation of the hydrology can affect the delivery of ecosystem services. It is illustrated how ecosystem management can be used to reinforce certain ecosystem services as part of a climate adaptation strategy. In most of the Flemish catchments the hydrological system is strongly manipulated and does not little resemble its potential natural state. Different types of changes have been made: draining of marshlands for the purpose of agriculture, straightening and canalization of streams for shipping traffic and agriculture, loss of infiltration due construction of paved surface and buildings, diking of flood zones, etc… Besides economical benefits these changes had an impact on many of the hidden ecosystem services. Only gradually it becomes clear what we pay for it. Floods, water shortages, erosion, loss of biodiversity and eutrophication are signs that the regulation of hydrological extremes is seriously disturbed. In many cases we have become dependent upon expensive technical measures to replace those regulating functions.

3.1 Introduction

The water system provides direct or indirectly numerous goods and services (Figure 12). Since long time these goods and services are used to support our society in various ways. We can distinguish visible and fast renewable resources as fish, crops, timber and drinking water that distinctively can be linked to the water system. A combination of growing needs and a technological ability has resulted in an increased control and manipulation of the water system. These developments have finally led to a serious degeneration of the system's carrying capacity. The past and present large scale exploitation of marketable ES (Infrastructure, Agriculture, Forestry) has led to such a degenerated environmental quality that there is a severe impact on society (flooding, water shortage, desiccation, pollution, land-erosion, pests…and biodiversity losses). These problems are usually solved by technical solutions such as water retention basins, sewage infrastructure, treatment plants, canalization and normalization, dredging, dams, pumping, drainage, irrigation wells, embankments, etc….The implementation often brings about secondary effects (further disturbance of hydrological cycle and nutrient cycles leads to further loss of ecosystem functioning). This pathology of command and control is still relevant today and there is a profound pressure on policy makers to solve environmental problems by quick and visible solutions.

Ecosystem based land-use planning is considered as an important measure to increase resilience against flooding, droughts and associated water quality problems. Several concepts have been established in respect to land-use patterns and functionality such as Land Quality (Bouma 2002), Leakiness Indices (Doran and Zeiss 2000; Doran 2002). Shared by these approaches is that fluxes of water and substances determine the sustainability at certain locations given a certain land-use. Land-use patterns that reckon with the physical properties of soil and hydrology cause less interaction with the water system whilst a high discrepancy between actual land-use and physical suitability urges a more intense adaptation of the system and thus to a higher impact of land-use on the water system. Adaptation measures for CC can result in increase or decrease in the level of manipulation and control. Functional analysis of ecosystems and management measures for the deliverance of ecosystem services receives growing attention (Kremen 2005; Tscharntke, Klein et al. 2005). This is especially valid for natural and semi-natural structures such as wetlands, floodplains and forests. Especially these land-use types deliver important ecosystem services to society and resilience against flooding, droughts and associated water quality problems. At the different catchment scales, attention must be paid to attain a balanced land-use in which nature should be allowed to function. Mitsh and Gosselink (Mitsch and Gosselink 2000; Mitsch, Lefeuvre et al. 2002) state that a sustainable wetland acreage should range between 3 and 7 % of the catchment in order to regulate hydrology and water quality. Often small valley bottom wetlands and riparian strips in the headwater basins are neglected in wetland-policy (Merot, Hubert-Moy et al. 2006). Especially these wetlands provide a high functionality in regulating hydrology, water quality, and biodiversity over the whole catchment area (Merot, Hubert-Moy et al. 2006). Wetlands substrate biogeochemical hot spots and hot moments within the landscape, where hydrological flow paths converge with other flow paths or substrates containing complementary reactants (McClain, Boyer et al. 2003). Hot spots occur where hydrological flow paths converge with other flow paths or substrates containing complementary reactants. These biochemical interactions are often enhanced at terrestrial-aquatic interfaces (McClain et al, 2003) where system processes are more intense and provide an important functionality to local and downstream environment in regulating water and nutrient fluxes (Newson 2010).

Some of these hotspots can be pinpointed rather easily as they depend largely on abiotic conditions (e.g. landscape depressions, seepage areas, moorlands, frequently inundated areas). Protecting these areas is not sufficient as their ecosystem function (ability to generate a range of goods and services) is also dependent on larger scale processes and interactions that need to be preserved. The importance of ecosystems in reducing the occurrence of undesirable environmental phenomena (flooding, anoxia, eutrophication, erosion etc…) depends on the characteristics of a landscape and the existence of efficient structures that are capable to assimilate these perturbations. These phenomena are to some extent mitigated in magnitude, frequency, recovery time and scale by ecosystem functioning.

The occurrence and extent of these hotspots is primarily dependent on morphology and soil characteristics but also require functional ecosystems (self-organizing & adaptive) to provide the services in regulating water and nutrient cycles. Restoring these elements within landscapes would provide an increased resilience to system disturbances. In this approach, small scale wetlands within urban and agricultural landscapes are equally important in providing important. These small-scale potential wetlands can be detected by analysis of morphological properties such as wetness indices (Mérot, 2006). Although wetlands provide diverse valued services to humans, the incentives that private property owners have to protect wetlands may nevertheless remain low. Wetlands owners can neither easily capture the social benefits that accrue when wetlands are not protected nor produce those benefits independent of the cooperation of many others in pursuit of the same goals.

The key challenge of modern policy making in the domain of ecosystem management is to prevent or reduce the degradation of ecosystems and their services while meeting increasing demands. The potential of landscape management and planning for ecosystem service generation needs to be examined as a true option to reach both environmental and societal policy objectives at different scales. Many ecosystems in Western Europe have been degraded, fragmented and have been contained in their succession stage through active management. Climate change can be a catalyst to restore the coherence between human activities and natural processes. The concept of ecosystem services clearly demonstrates that it is more efficient and cost-effective on the long-term to make use of ecosystem services than to destroy them and replace these services through technical engineering. The challenge for management and planning is to restore a natural and characteristic diversity of ecosystems and create semi-artificial opportunities for ecosystem development that can compensate for climate changes and anthropogenic impact. The scheme in Figure 12 illustrates the linkages between climate change, land-use, environmental degradation and the opportunity to anticipate and plan for ecosystem service generation.

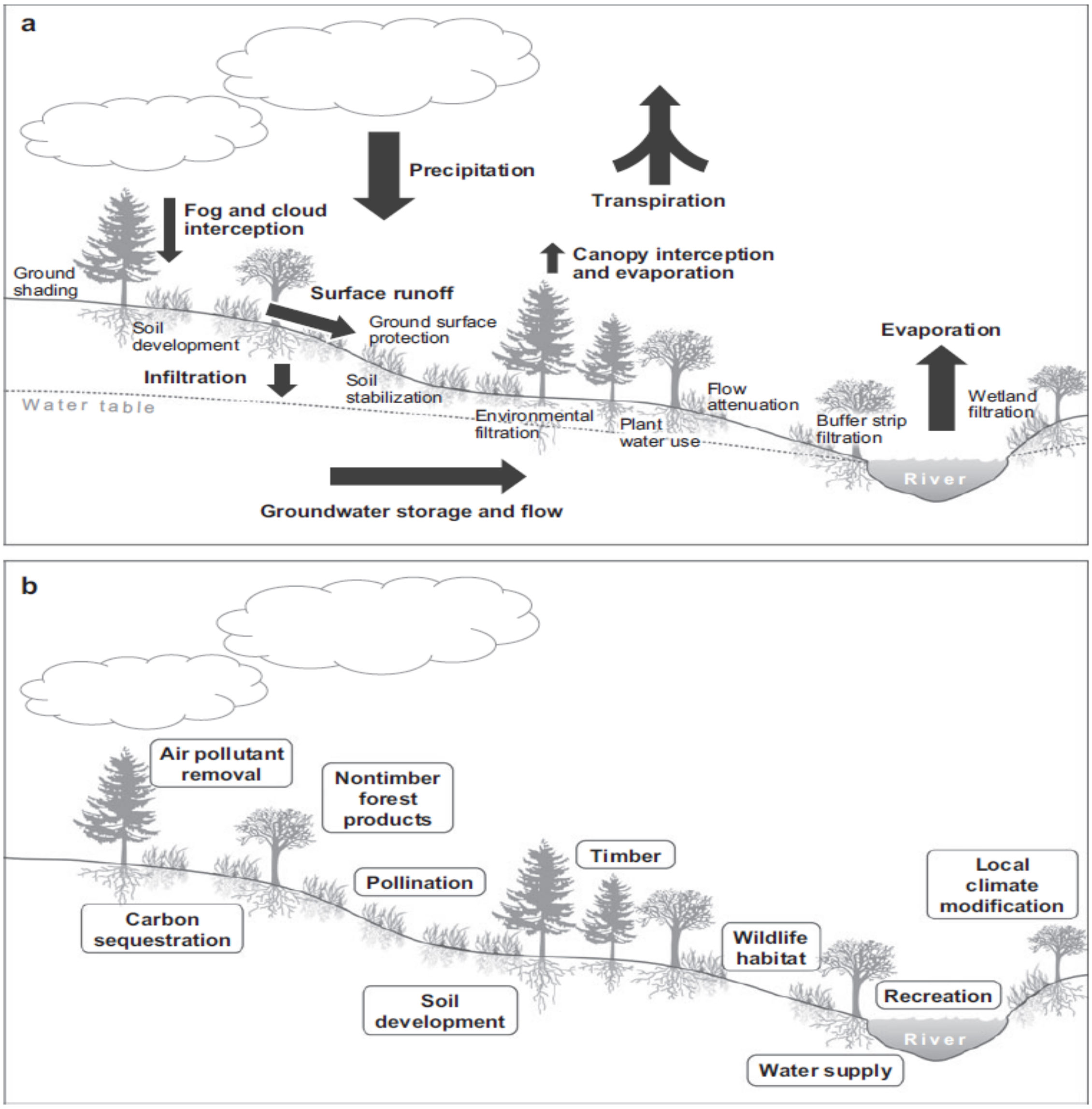


Figure 12: Ecosystem services provided by hydrological processes (Brauman, Daily et al. 2007).

Integrated catchment management is a crucial aspect because many ES are strongly related to water dynamics. It is necessary to evaluate human activity, land use and ES generation from a catchment perspective. In order to define a sustainable balance of ecosystem service generation and ecosystem service exploitation, there is a need to find a balance between the upstream supply of goods and services and the downstream use of goods and services. River mediated ES that are generated upstream will have a higher potential to generate benefits, since the downstream trajectory is longer. While upstream human activities have a (negative) impact on the downstream river system. Within our research group we try to develop methodologies that will allow us to research these considerations and implement them into policy tools.

3.2 Climate adaptation strategies in Flanders

As we will experience serious climate change effects during the next decades in Flanders, regardless the level of our worldwide mitigation policy, the need for an adequate adaptation strategy is obvious. With the Flemish context and its key challenges regarding water management, spatial planning and nature policy in mind, we can conclude that the current scientific and policy attention regarding climate change effects is lacking a strategy that links the different challenges that exist across the different policy domains. This is perhaps also because the current scientific and policy attention regarding climate change effects on the water system are somewhat biased towards flooding. Recent flooding problems in Flanders will only extend that bias. There is a need to recognize other impact mechanisms and put more effort towards solving drought related issues, the urban heat island effect as well as implementing measures that tackle drought, flooding and heat issues at the same time. To a certain extent, policy makers are hiding behind the uncertainties regarding climate change (emission reduction, model uncertainty, feedback effects, etc…) and its impact mechanisms. The largest uncertainty however is generated by the uncertainty in emission reduction scenarios, being at the beginning of the entire uncertainty chain. The decidedness, efficiency and effectiveness of the measures for emission reduction and/or adaptation is a crucial uncertainty. The fact that policy requires a higher accuracy and reliability is rather a psychological problem than a scientific problem. Policy makers are used to make decisions that are based on highly uncertain facts, trends, and predictions. Decisions with a positive message do not need to be well-substantiated, even if there is a potentially high societal and economic impact. Decisions on strong climate mitigation and adaptation strategies are of a fundamentally similar nature. But there is no strong political profiling on the topic because the risks are not perceived. Currently adaptation measures for water management in Flanders generally come down to a limitation of the likelihood of flooding by means of structural interventions. In addition the Flemish government is also developing flood predictors to allow the timely anticipation to potential floods (Brouwers et al 2009). There is currently no bearing capacity at political level to fully introduce land use related measures in flood risk management or adaptation policy in the large sense.

At last, a primary overarching difficulty is the modeler's ability to integrate social and biophysical processes from multiple disciplines, processes with inherently different spatial, temporal, and thematic foci, into one comprehensive model describing process and causality with respect to land-use change. So a final key challenge will be the integration of all modeling efforts in Flanders by spatial planners, hydrologist, ecologists …in order to provide sustainable land use solutions to the policy audience as the most important but often neglected part of our Flemish adaptation policy.

More integrative and conceptual assessment of mitigation and adaptation measures against climate change impacts are needed. Climate change can be a catalyst to restore the coherence between human activities and natural processes. The challenge for management and planning is to restore a natural and characteristic diversity of ecosystems and create semi-artificial opportunities for ecosystem development that can compensate for climate changes and anthropogenic impact. To a large extend rather expensive technical end-of pipe solutions have been applied to solve environmental problems (flood control areas, purification plants, groundwater extraction, embankments, drainage). Some of them are cost-effective in the current setting, but it is likely that climate change will push the limits of technical solutions. In the meantime both macro-scale socio economic developments and environmental degradation might push certain land-use activities to unfavorable cost-effectiveness situations.

A conclusion might be that we should recognize that the complex, adaptive nature of ecosystems and the biosphere will never be apprehended completely (Levin 1998). It is simply not possible to model and quantify all aspects of climate change induced impacts and risks. To build in environmental security, we need to preserve a level of ecosystem functioning and learn from their behaviour and the services they provide. A thorough analysis of the conceptual functioning of the catchment can bring insight into the water system and perhaps provide solutions that are more climate proof or cost-efficient than technical measures. Climate change will exacerbate the need for ecosystem services restoration. There are a number of non-technical adaptation and mitigation measures that can provide multiple benefits with respect to climate change impact mitigation. In general, measures should reduce evapotranspiration losses, increase infiltration and groundwater recharge, retain water in the headwater wetlands and increase flood storage downstream. The urban run-off is a major challenge as the frequency and intensity of extreme events will increase.

Local buffering and the promotion of infiltration infrastructure may contribute to both a decrease of the flood risk and a reduction of the low flow episodes. The hydro morphological integrity of rivers is crucial for biodiversity and many ecosystem services that can potentially be delivered (Darby 2010; Elosegi, Diez et al. 2010).

3.3 Setting of a reference situation

Drainage of wetlands for agricultural use has also existed for hundreds of years and has progressed along with increasing technological ability. Headwater wetlands are located in upstream colluviums and are mainly rain fed. They easily accumulate organic material under acid, anoxic conditions. Over time, the vast peat layer acts as a sponge in retaining water, which promotes aquifer recharge and provides base flow to headwater streams (Holden and Burt 2003; Holden, Chapman et al. 2004; Haigh 2006; Křeček 2006). These headwater wetlands provided a unique hydrological retention function. Most of the headwater wetlands were drained centuries ago by deepening and straightening the headwater streams and by building drainage channel networks. Because of their topographical placement, gravitational drainage by hand-dug drainage channels was feasible. The second type of wetland is the upstream valley bottom wetland. These narrow riparian zones act as buffers and delay the contribution of upstream catchments to the peak flow. Riparian wetlands contribute to base flow stability by retaining interflow, and increasing groundwater levels and deeper groundwater recharge (Bradley 2002; Merot, Hubert-Moy et al. 2006). The water table in these wetlands has seasonal fluctuations, allowing mineralisation and occurrence of high productive ecosystems. These wetlands have also disappeared to a large extent since the 19th century when machinery became available for canalisation of large streams. In contrast, downstream valley bottom wetlands have been preserved relatively well. Drainage of these wetlands used to be very difficult as there is little gravitational drainage potential. Embankments and active drainage by pumping only became economical in the post war period when self-reliant agricultural production became a European policy priority and received great financial support for large-scale land conversion. These wetlands can act as run-off generating areas by their shallow water table, but can equally dissipate peak flows from upstream by flooding. The floodwaters are temporarily stored, and as the discharge velocity drops, sediments and nutrients deposit. However, with embankments, this mass and energy is pushed further downstream and aggravates flood events. The remaining wetlands suffer from more frequent and more intensive flooding. This increase of extremities is caused by a reduction of floodplain acreage, increased urban run-off and the disappearance of upstream valley bottom and headwater wetlands. Downstream valley bottom wetlands are also water consumers as they receive a constant flux of groundwater - surface water (Bullock and Acreman 2003). They possibly reduce the base flow during periods of drought.

This gradual loss of wetlands indicates that in planning and designing for rewetting, it is not sufficient to restore valley bottom wetlands alone. Restoration of infiltration and re-creation of headwater wetlands are as important to restore hydrological functioning. Generation of wetland restoration scenarios can be built upon many methodologies found in literature (Kremen 2005; Tscharntke, Klein et al. 2005). Functional analysis of (potential) wetland landforms may reveal opportunities for water management and climate adaptation measures (Mitsch and Gosselink 2000; Mitsch, Lefeuvre et al. 2002). Many ecosystems in Western Europe have been degraded, fragmented and have been contained in their succession stage through active management. Many ecosystems have been isolated from landscape gradients and are managed as islands detached from their landscape functionality.

3.4 Challenge of incorporating long-term visions on land-use planning within hydrological and ecological models

For these reasons it is necessary to balance the costs and benefits of different long-term visions. For that it is also crucial to consider and include the potential impact of several climate change scenarios.

This includes elements related to extremities in water shortages, water quality problems, irrigation-agriculture, transitions to drought resistant crops, etc…but also increased risk for extreme storms and associated problems with urban hydrology, sewage overflows, land-erosion, mudflows, hailstorm damage,… The adaptation response should be a risk based response where there is a focus to reduce the local vulnerability to both global and local climate change.

It is clear that different long-term visions will have a different vulnerability to these different phenomena. But if general principles of water system functioning are respected, the vulnerability can be decreased. These general principles aim to restore natural functioning of the water system. Natural processes tend to maximize energy dissipation by which extreme conditions are less severe. The functionality of ecosystems is aimed on the process-driven interactions that take place. The functionality of ecosystems is on the level of energy and nutrient fluxes, by transfer of energy and material between structures (biological, chemical and physical). The development of a credible long term vision for nature development and water management is desirable for the evaluation of climate scenarios. There are many ongoing and planned initiatives that will influence the water system. There is an added value to compare the climate change response of the current situation to the long term vision. In that way, the long term vision can be evaluated. The growing awareness of the value of natural ecosystems has resulted in various Ecosystem-based River Basin Management intervention efforts being initialized to reverse past anthropogenic changes. One such measure being considered in Flanders is floodplain restoration and river valley rewetting. Different wetland types might require specific eco-hydrology in order to preserve them or allow them to provide certain desirable ecosystem functions. The response of ecosystems to restoration measures is most unpredictable and may result in transitory unstable ecosystems that may persist for decades. Successful water management requires knowledge on which wetlands perform different and/or specific hydrological functions. In depth study on the influence of wetlands on hydrology is necessary as there is little information available on the biological behaviour of different wetland types of interest in hydrological modeling. Successful water management requires knowledge on which wetlands perform different and/or specific hydrological functions. This information would improve the assessment of wetland restoration and rewetting. With this prospect of large scale nature development and floodplain restoration, it is necessary to investigate the current state of the catchment and the restoration potential. In this aspect it is also crucial to manage drainage, infiltration and evapotranspiration to restore the wetlands.

3.5 Case study Grote Nete

The Grote Nete catchment has a total surface of 385 km², has an average population density of 350 inhabitants/km² and approximately 486 km of categorised streams. The main part of the basin is situated in the province of Antwerp, close to the Dutch border. The region has predominant sandy soil coverage with the presence of parabolic land dunes in the upstream parts. These dunes were shaped by intense polar winds at the end of the Würm ice age. They created a distinctive gradient in the soil hydrology with areas of intense infiltration and seepage. Wide alluvial plains with shallow groundwater and distributed seepage can be found in the downstream parts of the study area. Like many European catchments, the Grote Nete Catchment has been experiencing an increase in extreme hydrological events (Estrela, Menéndez et al. 2001). In addition, extreme rainfall events by both duration and intensity have been observed more frequently during the last decade. The boundaries of the recent floods (09-1998, 12-1999, 02-2002 and 01-2003) extended to a large part beyond the natural alluvium of the river (Aerts, Van Orshoven et al. 2000; Van Damme, Defloor et al. 2005). This increase is observable both in intensity and increased frequency of flooding occurrence (De Rycke, Devos et al. 2004). Catchment hydrology of the Grote Nete is significantly impacted by urbanisation (25 %) and agriculture (17 % cropland and 16 % grassland). The ratio of impervious area strongly influences run-off and baseflow, increasing vulnerability to flooding and low-flows. A general approximation here is that the local recharge losses are approximately equal to the percentage of urbanised area. For the Grote Nete catchment, urbanisation originated on the ridges of dry, sandy land dunes. They were of no use for agriculture and were safe against floods. But these land dunes also have high infiltration capacity and determine the existence of intense seepage areas. Some scientists believe that the gradual increase in impervious area and implementation of sewage storm water drainage systems have reduced the role of these infiltration areas in the hydrological cycle and decreased both phreatic water tables and deep groundwater recharge. One way to restore wetlands is thus by restoring infiltration areas.

3.5.1 Derivation of scenarios for wetland restoration and infiltration restoration

The Grote Nete catchment has significant area classified as habitat directive area and bird directive area (Figure 13). Nowadays, these areas are still fragmented by Agricultural use and weekend cottages. The ongoing EU-LIFE-project "between Dune and Nete" (Province of Antwerp and Natuurpunt) gives strong impulse to nature restoration. There is a high synergy between nature development and water management, which will lead to a more natural water system and more robust nature development with ecosystem functions such as water conservation, groundwater replenishment, flood storage and nutrient retention.

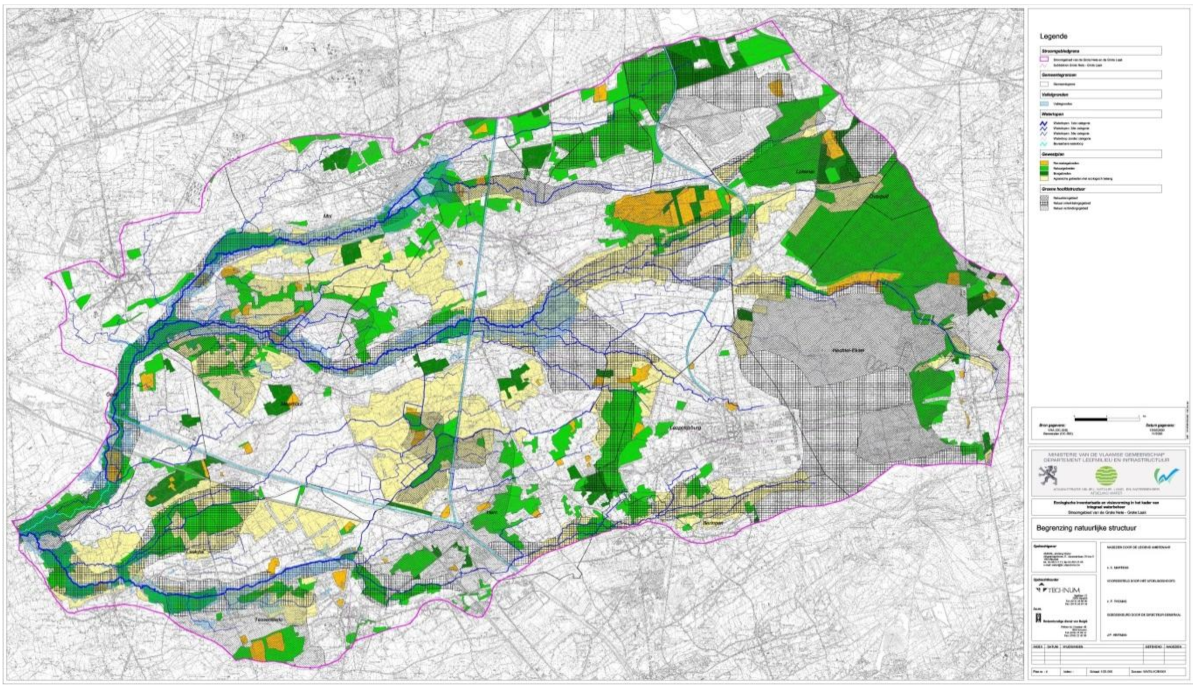


Figure 13: Special protection zones and Nature Structures in the Grote Nete catchment

Changes in land-use will affect catchment hydrology. The development of a credible long term vision for nature development and water management is desirable for the evaluation of climate scenarios. There are many ongoing and planned initiatives that will influence the water system. There is an added value to compare the climate change response of the current situation to the long term vision. In that way, the long term vision can be evaluated. Reconnection of formerly embanked areas (as one of possible adaptation measures) will provide opportunities for the restoration of valuable floodplain ecosystems. Detailed information about future changes in water quality and flood characteristics is crucial for the implementation of adaptation measures and their ecological assessment. The research focused on the position of the wetlands in the catchment and their typology. For each hypothesis, a restoration scenario was created for which the catchment response was obtained through distributed hydrological modeling.

In contrast to many classifications that exist, wetlands are no true landcover type and are often mapped as grassland, forest and other land-use types (e.g. Sedge communities and Alder forests). The land-use map of the basin management plan for the Nete Basin considers only 0.11 % as wetland. For other land-use maps such as Landsat TM and Corine land cover, 2.94% and 0.57% respectively of the basin acreage is mapped as wetland. It was assumed that former wetlands, although now drained, might still have relics of their original vegetation. Vegetation relics can be a valid method to determine degraded wetlands (Gisèle Weyembergh 2004). It is not uncommon that natural vegetations are forced back to small relics due to habitat fragmentation and habitat loss. Indicators of wetland vegetation were derived from the Biological Valuation Map (BVM), which is a standardized survey and evaluation of the biotic environment of Flanders and the Brussels Capital Region. The BVM is a ecotope classification map which is also used frequently as a land use map (Schneiders and Verheyen 1998; Van Eetvelde and Antrop 2005). Based on the first unit of the BVM (dominant vegetation community), about 6.5% of the catchment (Figure 14) has distinct wetland characteristics. The second and third unit of the BVM were screened for wetland and open water indicators (Figure 15). The percentage of wetland vegetation relics was used to assess the restoration potential and distance to target for wetland restoration. An additional 11 % of the catchment was found to have elements that indicate former (degraded) wetlands or complexes of ponds and pools.

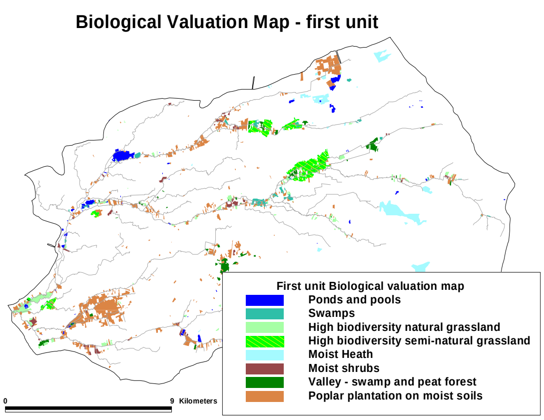


Figure 14: Water dependent terrestrial and aquatic ecosystems (first unit of BVM)

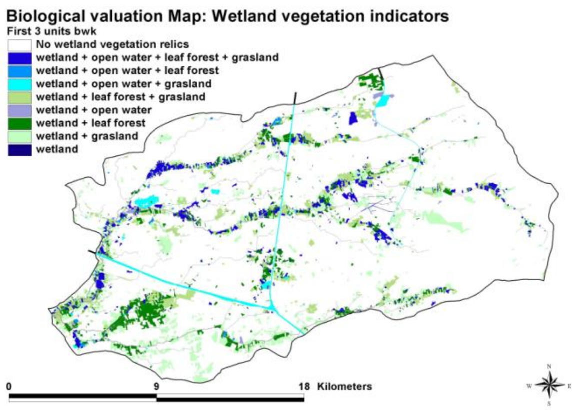


Figure 15: Wetland and open water indicators, based on first three units of BVM

3.5.2 Physical Indicators for Wetland Restoration

A Compound Topographic Index (CTI) was used to indicate phreatic groundwater conditions(Chaplot, Walter et al. 2000). This was based on the assumption that a CTI, which is based on surface runoff, can be used to predict soil hydromorphy especially in sandy homogeneous soils. Thus this method is primarily suitable to indicate interflow- groundwater flow dependent wetlands. This can be assumed valid for the sandy soils of the Grote Nete catchment, where accumulated water is easily infiltrated. A final combined CTI map was produced by a weighted average in which the weight-factor was proportional to the radius of the applied filters (Figure 16). The non-linear CTI classification was converted to a 50-class equal area classification.

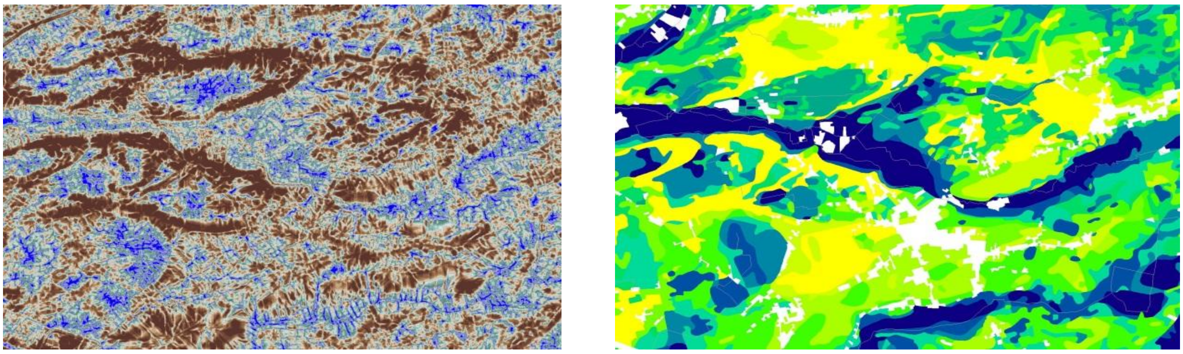


Figure 16: DEM wetness index (left) and soil drainage class (right)

The soil map of Flanders was used to verify the CTI-index. The soil map of Flanders is an extensive and reliable dataset and has been used for various applications (Dudal et al. 2005). From combinations of texture and drainage class, an indicative water table depth variation was derived (Figure 16 right). In valleys where the phreatic water level is high, groundwater gley soils can indicate water table depth in more detail. The essential common feature of poorly draining gley soils is that, under periodic or permanent water logging, the subsoil experiences a lack of oxygen within the pore space. Consequently under anaerobic conditions the insoluble iron oxides (which cause the characteristic yellow, brown or reddish-brown colour to soils with adequate aeration) are reduced chemically and the ferric iron changed to ferrous iron prior to translocation from the soil profile. Reference may be made to the table indicating depth at which reduction front (gley) occurs in sandy soils (Van Ranst and Sys 2000; Dudal, Deckers et al. 2005). Only 2% of the catchment has surfacing peat-soils, which can be found along the wettest parts in the valley system with permanent seepage of groundwater. Surface flow wetlands (SFW), or wetlands of which the groundwater table is strongly influenced by the river water height, can be regarded as water consumers from the river system. This is opposite to strictly groundwater dependent wetlands, which retain water that would otherwise drain. The relative height difference between the river itself and the valley is an indicator for surface water influenced wetlands. The following heights were calculated: -10 cm, -5 cm, 1 cm, 5cm, 10 cm, 20 cm, 50 cm and 100 cm (Figure 18). For this purpose we used the buffer by elevation change tool from the ESRI support center (Holzer 2005). In addition these areas must fall inside the alluvium of the rivers or have been flooded recently. This alluvium is determined by interpretation of the soil map and covers 16 % of the Grote Nete catchment.

3.5.3 Assessment of restoration measures: Wetland location

Among the aims of this study was to investigate whether the catchment was sensitive to the geographical distribution of restored wetlands. The upstream-downstream buffers were based on a topological analysis of the stream theme for a 250 meter interval along the streams. This involved the calculation of many possible indicators including Strahler, Shreve, upstream drainage area, and downstream length. The best performing indicator was the length downstream divided by the maximal downstream length (0-100%). The relative downstream length was squared in order to increase the differentiation between upstream and downstream. For the upstream buffer width, the resulting range (0-10000) has been multiplied with factors of 1/20, 1/10 and 1/5 (Figure 17). For downstream buffer, the relative length downstream has been first inverted, raised to its square and rescaled to respective maximal buffer widths of 0-250 m, 0-500 m and 0-1000 m (Figure 17).

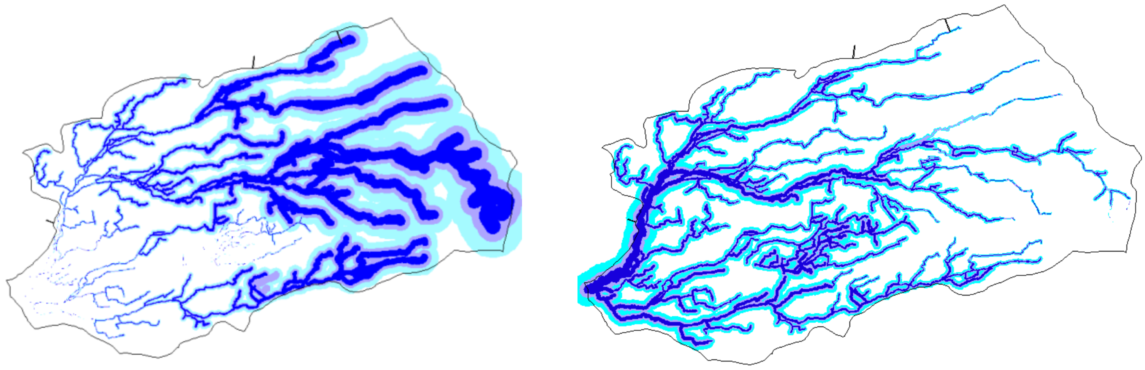


Figure 17: Upstream buffers (left) & Downstream buffers (right)

Minimal and maximal restoration scenarios were developed for respectively groundwater dependent wetlands (GFW) and river stage influenced wetlands (SFW). The initial GFW-min. and GFW-max. Scenarios were determined by selecting 10% and 20% respectively of the wettest parts of the catchment. Both are reduced by the requirement that land-use should be reversible without excessive costs (exclusion of built-up areas). The SFW-scenarios are restricted to the river alluvium. The height difference to the nearest stream is combined with a distance to stream (Table 1). For the practical surface flow wetland-scenario, Surface flow wetland achievability scoring should be above 8 (Table 1). Finally we obtain a gradual suitability map for restoration of river stage influenced wetlands Figure 18.

Table 1: Surface flow wetland achievability scoring

		Relative height difference to the nearest stream (cm)								
		-10	-5	1	5	10	20	50	100	> 100
Distance to nearest stream	< 25	20	19	18	17	16	15	13	11	5
	25-50	19	18	17	16	15	14	12	10	4
	50-75	18	17	16	15	14	13	11	9	3
	75-100	17	16	15	14	13	12	10	8	2
	100-150	16	15	14	13	12	11	9	7	1
	150-250	15	14	13	12	11	10	8	6	0
	>250	14	13	12	11	10	9	7	5	0

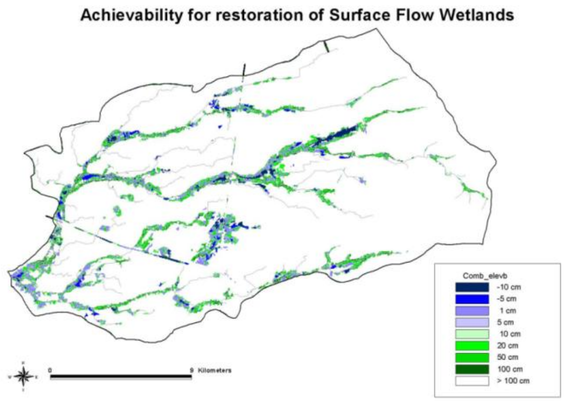


Figure 18: Height difference to nearest stream, within river alluvium.

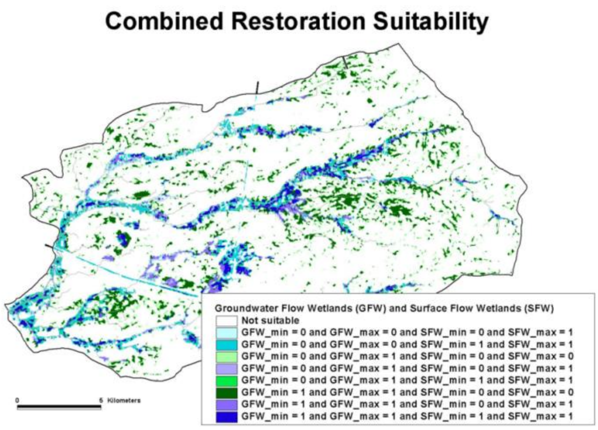


Figure 19: Combined restoration suitability for groundwater dependent wetlands (GFW) and river stage influenced wetlands (SFW).

Finally the combined restoration suitability for both surface water influenced and groundwater influenced wetlands was combined into Figure 19.

3.5.4 Results and discussion

This combined suitability map of Figure 19 became the basis to derive land-use change scenarios. The restoration scenarios are limited by the requirement that irreversible land-uses are excluded. For SFW-min, an additional condition excludes agricultural and recreational land-use if these conform to the spatial destination plans. The final scenarios for SFW-min and SFW-max are summarized in Table 2. The study generated the river valley rewetting scenarios for assessment as part of the restoration measures being considered in the Grote Nete catchment.

Table 2. Summary of scenarios (relative change to current land-use distribution)

Actual %		GFW min	GFW max	SFW min	SFW max	Inf. Rest.	D-S valley	U-S valley	U-S Headw.
Bare soil	3%	-0.1%	-0.4%	-0.1%	-0.2%	16.4%	0.0%	0.0%	-0.2%
Grassland	16%	-0.1%	-0.4%	0.0%	0.0%	-0.3%	-2.4%	-2.3%	-1.7%
Leaf forest	17%	-2.5%	-4.8%	-0.5%	-1.5%	-0.3%	-2.8%	-2.3%	-1.6%
Maize	17%	-2.7%	-4.9%	-1.3%	-2.4%	-0.3%	-1.5%	-1.7%	-2.0%
Open water	1%	-0.2%	-0.7%	-0.1%	-0.1%	0.0%	-0.6%	-0.3%	-0.1%
Paved	27%	-1.8%	-3.6%	-2.0%	-2.8%	-15.3%	-0.1%	-0.1%	-0.2%
Pine forest	11%	-0.1%	-0.2%	0.0%	0.0%	-0.2%	-0.1%	-0.2%	-0.5%
Wetland	8%	7.8%	15.5%	4.0%	7.2%	-0.1%	7.5%	6.9%	6.2%
Wetland cove	8%	15.8%	23.5%	12.0%	15.2%	7.9%	15.5%		14.9%

<u>EV analysis</u>: The results of the EV analysis revealed an exponential distribution for the river discharge, with a corresponding GPD as given in eqn (b), and γ =0 (Willems 2004). The parameters for eq. (b) and eq. (c) for the different scenarios were determined and are included in Table 3-2, where U-S refers to upstream, and D-S refers to downstream. The different scenarios did not result in a significantly different EV for stream flow. The peat layers in the upstream headwater wetlands were expected to act as water retainers. However, the EV analysis revealed that the presence of restored upstream headwater wetlands did not significantly reduce flow peaks through a water absorption action. A similar observation was made for the downstream valley bottom wetlands, which tend to dissipate peak flows by flooding; and the upstream valley bottom wetlands, which tend to retain water during intensive precipitation. EV theory has been applied before to a variety of problems in hydrology (Naveau et al. 2005). Naveau, et. Al . (2005) caution that the modeled distributions are by nature, associated with large margins of error. Owing to the rare occurrence of extreme values, estimates of parameters must be made from a limited sample of points, making possible the deduction of misleading conclusions. This is particularly so when dealing with complex non-linear and non-stationary processes (Naveau et al. 2005).

Table 3: Parameters for exponential EV distribution

	Original model	D-S valley	U-S valley	U-S headwater	Infiltration restoration
q_t, m^3/s	8.42	8.47	8.45	8.46	8.31
β	2.499	2.489	2.508	2.526	2.569
n, years	10	10	10	10	10
x	187	187	186	186	194

Actual evapotranspiration (AET) and Root Zone (RZ) water: Hydrographs for AET and RZ water were produced at 23 points, and were used to compare the cumulative values obtained from the calibrated catchment model against corresponding values obtained from the respective simulations. Across the three scenarios, there was an overall slight drop in the root zone water, with a corresponding decrease in the simulated AET (Figure 20).

The variability in RZ water can be seen in Figure 20 & Table 4 where points 1 and 2 did not have as much water in the root zone as point 3. Of note too is the fact that the average water in the root zone dropped at point 2, and increased at point 3. Thus, while the RZ water increased in some locations, it decreased in others. Points were selected with the objective of determining the distributed response of the catchment to a given scenario. For instance, three points were chosen when assessing the infiltration restoration scenario. The details of these points are contained in

Table Table 4. The difference between points 1, 2, and 3 arises from the location of the monitoring points. Points 2 and 3 are located in the valley of a relative large stream (high wetness), whilst point 1 is located on a headwater stream, surrounded by paved areas. Therefore Points 2 and 3 are expected to receive a steady lateral flow of groundwater from a large infiltration zone.

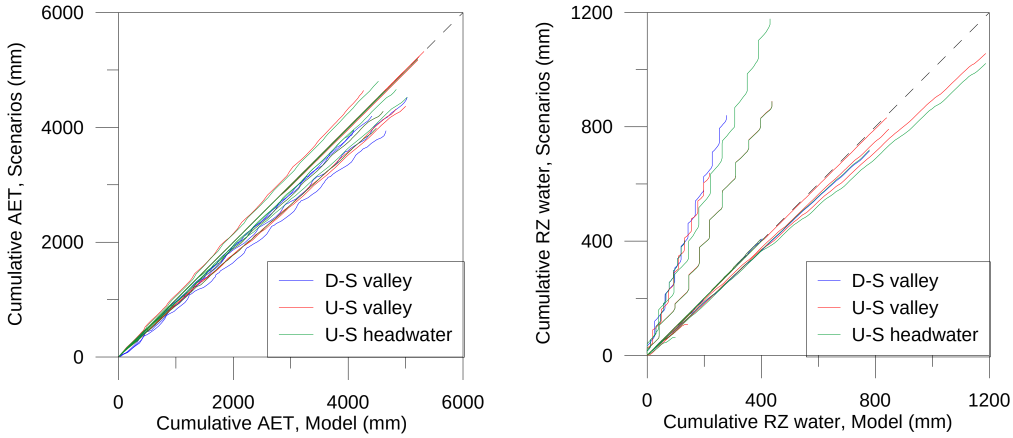


Figure 20: Cumulative AET (left) and cumulative root zone water (right)

Table 4: Properties and location of analysis points

Point	Landsat TM	Corine LC	BMP	Model	Distance to stream	U-S headwater	U-S valley	D-S valley
1	Paved	Paved	Maize	Maize	18m	Wetlands	Maize	Maize
2	Maize	Paved	Maize	Maize	183m	Maize	Maize	Wetlands
3	Maize	Grassland	Grassland	Wetlands	15m	Wetlands	Wetlands	Wetlands

In the case of infiltration restoration, analysis of the discharge hydrograph could not prove the hypothesis that a decrease in extreme hydrological events would occur. One possible explanation for this is that the increased infiltration into the sandy soils of the catchment leads to a quick rise in the phreatic levels, which in turn generate groundwater flow into the rivers, especially in the saturated soils of the valleys. It can be observed in Figure 21 that while the contribution to river discharge from the paved overland flow is decreasing, the contribution of saturated zone flow is increasing with the net effect that the total stream flow remains relatively unaffected.

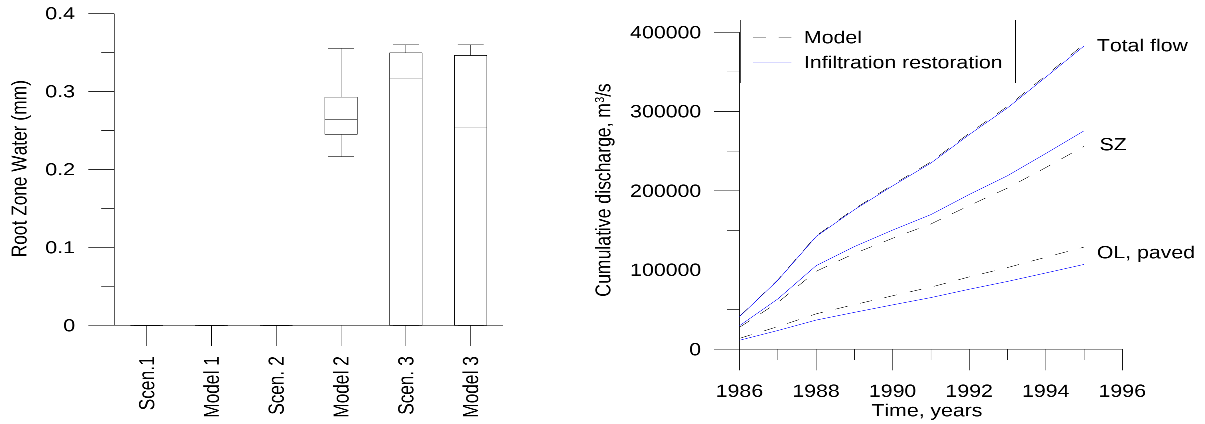


Figure 21: Root zone water (left) and discharge components for infiltration restoration scenario

A second explanation is that while urban areas near the streams may be most responsible for generating peak flows, they are unlikely to be restored in the scenario due to their high wetness index. In this case, the urban areas upstream that do get restored did not contribute as much to peak flows because of their remoteness from the stream. Also, the restoration of valley bottom wetlands (both upstream as downstream) will create conditions for increased manning coefficients. Vegetation development, and shallow and widened river beds, affect surrounding groundwater tables and decrease hydraulic head for the saturated zone flow component. This factor was not incorporated in the present model owing to the static nature of some model processes (Refsgaard and Storm 1995).

Finally, it was observed that there were a number of analysis points at which no significant response was obtained, particularly for the AET. This was not so in the case of the infiltration restoration (Figure 21) and downstream valley scenarios; but was true in 33% and 40% of the selected analysis points for upstream headwater and upstream valley, respectively. A recorded response will depend on a number of factors including, for instance, the position and relative movement of the phreatic table. In all cases the average phreatic level was observed to remain constant over time (Figure 22). The relationship between potential evapotranspiration (PET) and AET is another. If we are already operating close to the PET, as we would expect of points in the valleys, additional water is not likely to increase AET.

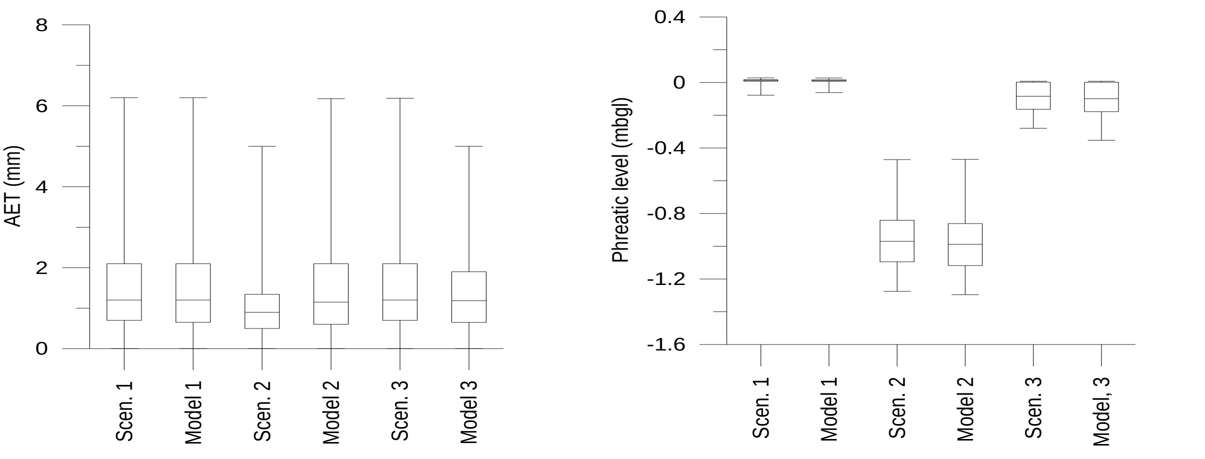


Figure 22: Cumulative AET (left) and phreatic level (right) for infiltration restoration scenario

3.6 Discussion

Wetlands are complex systems with intensive transformation of energy and matter of which we do not understand all relevant processes. Uncertainty remains since it is not possible to include certain elements of wetland behavior and development in models. Such aspects include:

1) The role of wetlands in energy dissipation, evapotranspiration and condensation of wetlands (Ripl 1995). Evapotranspiration is the most prominent of all hydrological process, yet there is little known about actual evapotranspiration for different hydrological conditions. Estimation of wetland ET remains insufficiently characterized due to the complexity of surface characteristics and the diversity of wetland types (Drexler, Snyder et al. 2004). Due to the variability and complexity of wetlands, there is no single approach that is the best for estimating wetland ET. Furthermore, there is no single foolproof method to obtain an accurate, independent measure of wetland ET (Drexler, Snyder et al. 2004). Apparently, factors other than water availability, such as atmospheric and stomata resistances to vapor transport, also can limit the PET rate of a wetland site (Shoemaker and Sumner 2006). Shimoyana (2004) reports that roughness change created by vegetation growth and surface wetness limit evapotranspiration to less than the potential evapotranspiration (Shimoyama, Hiyama et al. 2004). A horizon of moist-saturated air is formed, by which the atmospheric demand is reduced and transpiration rates are lower than expected. Consequently, the surrounding of a wetland and the size of the wetland becomes important and can also influence the AET. In the study of Andersen (2005), it is hypothesized that their results are conditioned by the proximity of the wetland to drier upland areas (Andersen, Hansen et al. 2005). Even more complex is regulation of plant transpiration through stomata as a response to climatical conditions or other stress-factors (Kutsch, Herbst et al. 2001).

2) Restoration of surface flow wetlands by re-meandering and morphological adaptation of streams could allow reestablishment of typical lowland floodplain wetlands. In-stream macrophyte development has significant effects on flow resistance and increases the hydraulic head and associated water retention (Vereecken, Baetens et al. 2006). The dynamic process of sediment trapping by extensive in-stream macrophyte growth during low flow, promotes wider, shallow and even braided river channels.

3) Incorporation of different wetland vegetation and their parameterization: To adequately study wetland hydrology, it is necessary to take species specific differences into account (Busch, 2000). There is a wide range of vegetation communities that is associated with wetlands, but little is known about their specific behavoir. The most important differences between these species are the different responses to a changing vapor pressure deficit (stomatal humidity). Within the study area there is occurrence of alder forests, poplar, open water, shrubs, sedges and moist grasslands. All of these types are influenced by periods of waterlogged conditions. For eutrophic, sedge dominated wetlands canopy transpiration can vary up to 25% higher or lower for different sedge species (Busch, 2000). Vegetation itself can alter hydraulic conductivity of soils by root type (Halabuk 2006). Alder forests under wet conditions have relative low LAI (< 5) (Herbst, Eschenbach et al. 1999; Johansson 1999) and evapotranspiration (0.35 – 3.87 mm/day) (Eschenbach and Ludger 1999). This means they can have a positive effect on wetland water budget (Dannowski and Ottfried 2006). This is in contrast to Populus Euramericana, which has high water consumption (”sap flow”) due to high metabolic activity of the poplar (Elias 2001). Poplar forests have been planted deliberately throughout the basin (2.8 - 3.2 % of the catchment area) to lower groundwater table at water logged sites. Cutting poplars stands would increase water budget. Analysis of the wetland restoration suitability reveals that poplar plantations comprise 10 % of all eligible sites, with increasing occurrence up to 25 % within the “GFW min and SFW min” - scenario. 4) Soil development in response to changing hydrology is also a factor of uncertainty (Holden and Burt 2003). Changing hydrological conditions can have large effects on soil biochemistry and influence vegetation development (Bio, De Becker et al. 2002). One such influence might be a change from a nutrient limited ecosystem to a high productive system. In general wetland soils pose a problem for modelers by their large lateral and vertical differences in hydraulic conductivity (Zeeb and Hemond 1998; Holden and Burt 2003). However it may help account for the high frequency of preferential flow pathways within what is otherwise a low matrix hydraulic conductivity peat

(Holden and Burt 2003). Ideally the model should correct soil properties for wetland restoration scenarios as saturated conditions generally decrease soil permeability and increase organic content.

CHAPTER 4: Recent progress and insights in ecological impact of changes in flooding regimes

4.1 Introduction

Notwithstanding the recognition of social and economic impacts of flood events, we also need to recognize that flood events are an essential part of the water system and that they are necessary for healthy water systems. The increase of land value has led to maximalisation of flood storage per square meter whilst disregarding the values of "natural flooding". One way to increase safety is by restoring water storage in floodplains. Natural(ised) floodplains have topographic heterogeneity and variability that result in gradients of flood return period and flood dept within the floodplain. Depending on the present vegetation types and the changes in flood parameters, there can be positive and negative influences of changes in flood depth, season, duration and frequency. First, a distinction needs to be made between peak flow storage and emergency water storage. In the first, it is the intention to store water frequently and to decrease the peak flow downstream. This means that the storage occurs also at regular precipitation events and with a minimal return period of once in every 25 years (Van Bommel and Hoekstra J.R. 2002). In the second case the floodplain is designed to act as an emergency flood zone, protecting urban settlements nearby, by storing water at critical and extreme water levels. In the case of peak flow flood storage, adapted vegetation can establish and wetland like ecosystems could develop. But preferably, the flooding frequency is as high as possible in order to maximise ecological and ecosystem service values. In the case of emergency storage, it is not likely that adapted vegetation will establish and it is possible that complete vegetation communities are wiped out during a long-lasting (> 14 days) flood event. In most cases, emergency floodplains are designed for quick release of floodwater, but it remains a highly dynamic environment with highly irregular behavior in which high quality ecological values are difficult to establish.

A flood event is a sequence of water levels and has to be translated to spatial patterns of flood depth and flood duration. In floodplains with topographic variation, each single location will have different exposure and affect vegetation differently. The depth-duration matrices of both regular and extreme flood events are crucial information for ecological impact assessment. The second step is to analyze whether this depth-duration matrix is changing with climate change scenarios. This in contrast to the data needed for flood risk assessment for properties, where the damage can be assessed by means of maximal water levels.

The uncertainties of ecological impact assessment are reflected by the wide variation of riparian vegetation communities and the wide variation in their flood resilience. In addition, there are many flood-parameters that influence the vegetation. It is the cumulative impact of individual flood events with different parameters that will determine the collapse of a vegetation type. It is utmost ambiguous that many high biodiversity floodplains are seldom flooded. Straightening of river systems has led to a decrease of flood frequency and intensity. This has also led to the situation that many ecosystems have become primarily groundwater dependent. To restore flooding on these sites, could mean a setback in ecological value, but there are many and complex processes involved. For ecological risk assessment, the prediction of structural changes in regular flows and floods is of primary importance. Especially the prediction of climate change impacts on flood duration and timing (season) is of great importance for ecological impact assessments. The most direct impact of inundation on vegetation is drowning of vegetation through oxygen depletion in the root system. This is determined by the combination of flood duration and timing. In this context the "timing" of the flooding relates to the general conditions of vegetation metabolism, microbial activity (temperature dependent) and water oxygen content (temperature dependent). Flooding during winter (colder) periods has lower ecological impacts compared to summer flooding because of climatic conditions:

a) Lower temperatures enable a higher dissolved oxygen content in the water

b) low temperatures reduce microbial activity and thus lower microbial oxygen consumption

c) Inundations during winter have less effect on the survival of plant species since the vegetation is not in active growth stage, which means that damage of eventual oxygen depletion is much lower. During active growth, the vegetation requires a much higher root system respiration.

Vegetation types may be adapted to certain flooding regimes if these occur on a regular basis (at least annually). Alternatively, also species may occur that recover quickly from flooding (grasses – shrubs). Bottomline is that species can only occur if their recovery time is less than the flood return period. Structural changes in timing and duration of flood events can thus have a large ecological impact. In addition, the impact of flooding on ecology is determined by the totality of all flood events. Changes in flood frequency (E.g. Bi-annual to Annual) of certain flood-events will also have an impact, even if the duration or timing remains unaffected. The elements at risk are thus exposed to a cumulative impact that is spatially differentiated. Concerning flood duration, frequency and timing, there is a wide range of classifications used in literature on floodplain ecology. In addition, they use rather qualitative descriptions of impacts on vegetation. It is evident that nature does not follow any arbitrary classification and associated assumptions and that uncertainty is an inherent part of ecology. In this aspect, there is also an indirect relation between climate change and phenomenology of nature. Rising temperatures do not only affect hydrology, but also the timing of the active growing season, soil microbiological activity and microbial species composition. Also the water quality of the flood water influences the vegetation communities. Essential parameters here are: initial oxygen levels, sediment load – dissolved nutrients – alkalinity – sulphate levels and soil conditions). Quantitative relationships between these parameters and vegetation types are difficult to find in literature because there is always interference by multiple factors. Furthermore, it is known that the hydrological effects of flood events do not only determine the drowning of vegetation by oxygen depletion, but that the hydrological parameters also determine or influence secondary effects and processes like (internal) eutrophication, alkalinisation and toxification. This means it is difficult to assess the impact of the hydrological aspects of changes in flooding parameters, solely based on changes in hydrological terms, without considering water quality data or soil characteristics (buffering capacity).

4.2 Concepts on catchment and floodplain functioning

Riverine ecosystems are not that well understood in term of their behavior. Much of the rivers in the North American and European temperate zones that have been studied by the early stream ecologists have been significantly altered by human activities and engineering before any research took place. The study of these modified ecosystems led to overestimation of their deterministic, static, homogenous and scale-independent behavior. A resilient ecosystem is one that has the self-regulating capability to deal with disturbances (both natural and manmade). Ecosystems are shaped by powerful adaptive and self-organizing forces. Groundwater and surface water dynamics are interconnected as rising water levels not only increase the wetted surface of the channel and eventually of the floodplain, but at the same time influence the exchange between groundwater and surface water either by allowing an upwelling of groundwater or by forcing a down-welling of the surface water into the aquifer vertically and laterally. Stability and resilience on a basin scale are a resultant of numerous interactions of ecosystems processes on different scales. Costanza and many others have put forward many thoughts and concepts of “ecosystem health”, “sustainability” and “ecological integrity”. The definition of Costanza is that a healthy ecosystem should be able to maintain structure (organization) and function (vigor) over time in the face of stress (resilience) (Costanza and Mageau 1999).

Flooding is a crucial and fundamental aspect of water system functioning (Gerritsen, M. Haasnoot et al. 2005). It is a regulating service, affecting the timing and quality of water flows. Natural floodplains and associated ecosystems develop themselves in such a way they retain water after flooding, allowing functions such as denitrification, carbon sequestration, water retention, nursery functions etc… For a long time, flood-associated sediment deposition was the natural fertiliser of valley agronomical (eco) systems. Within the current context, sediment retention, nutrient retention and denitrification in natural floodplains are beneficial to the in-stream water quality. When considering additional benefits of naturalised floodplains in stead of emergency flood storage designs, the cost-benefit ratio might favour the naturalised floodplains, despite their lower storage capacity per square meter. Naturalisation of rivers and adjacent floodplains is crucial to reaching the good ecological status of water bodies. Beside the biodiversity and landscape values, there are other important biochemical services. Denitrification is an important process that does not only occur during floods, but is a constant process that can take place within floodplain ecosystems on the condition of predominantly saturated soil conditions. The restoration of floodplains by topsoil removal and creation of gradients can strongly increase the denitrification potential. These conditions also favour the development of highly organic hydric soils, which can contribute to the worldwide carbon sequestration efforts. Especially the creation of new wetlands, can contribute to carbon sequestration.

Flooding characteristics are variables that affect the development of ecological communities. For certain there are many other important mechanisms and processes that affect species composition in floodplains. Both natural (e g. succession, sedimentation) as anthrophogenic stressors (dessication, acidification, invasive species). Hence, it is difficult to relate climate impacts on riparian vegetation through the specific mechanism of changing flood regimes. Resilience against climate change will not be determined by climate change induced changes in hydrological regimes. The major impact on ecological values has occurred in the past and more extreme floods may provide opportunities for ecological values. Not because extreme flooding is beneficial in se, but rather because it does not allow other land-uses and re-instates ecological rejuvenation (Figure 23). In addition, this is probably beneficial to the structural abiotic diversity of the floodplain (erosion, sedimentation, denitrification, carbon pools) and hence increases biodiversity.

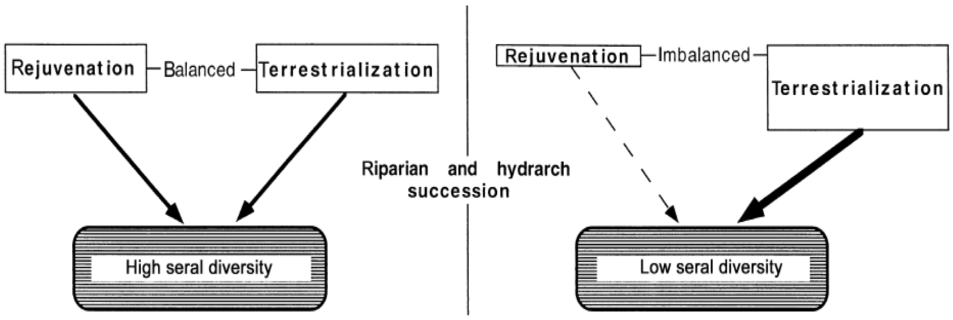


Figure 23: The diversity of successional stages in natural and regulated riverine landscapes as a function of the balance between rejuvenation and terrestrialization processes (Ward, Tockner et al. 2001)

How do we translate these conclusions into practice for ecosystem management? Within a changing environment, we do not seek methods to freeze ecosystems within their succession stage, but rather strategies to maintain structural biodiversity and enforce species capabilities to migrate and adapt. In ecosystem management for conservation there has been a strong focus on maintaining a certain succession stage. Nowadays nature conservation also focuses on the protection of the underlying processes and the generation of ecosystem services. Resilience and stability can be achieved by allowing certain natural processes to occur, as change is a natural resultant of their dynamic behaviour and thus a part of the resilience. Freezing ecosystems in a certain stage leads to failure (Pavlikakis and Tsihrintzis 2000).

In the long run there are difficulties to maintain these island ecosystems within a changing environment. To build in environmental security, we need to preserve a level of ecosystem functioning and learn from their behaviour and the services they provide. In the Flemish Region there is a positive correlation between recently flooded areas and the ecological value; One can argue whether this is because of the flooding processes or just because other land-uses are abandonned. But in either case, it provides opportunities for nature values. We need to distinguish ecological vulnerability from ecological risk. If the floodplain vegetation types are vulnerable to flooding, but display a low ecological value, the ecological risk is also low. Typical floodplain vegetation communities for frequently flooded ecosystems are considered to have a high ecological value.

4.3 Physical and chemical processes that determine the flood resistance of vegetation

Inundation is a driving force for many processes that can alter the abiotic conditions of an ecosystem. Whether these changes are permanently or temporal is dependent on the characteristics of both the inundation and the inundated site. Also the ecosystem's biotic component is an important factor (e.g. succession stage). The processes shown in Figure 24 will be described in the following paragraphs.

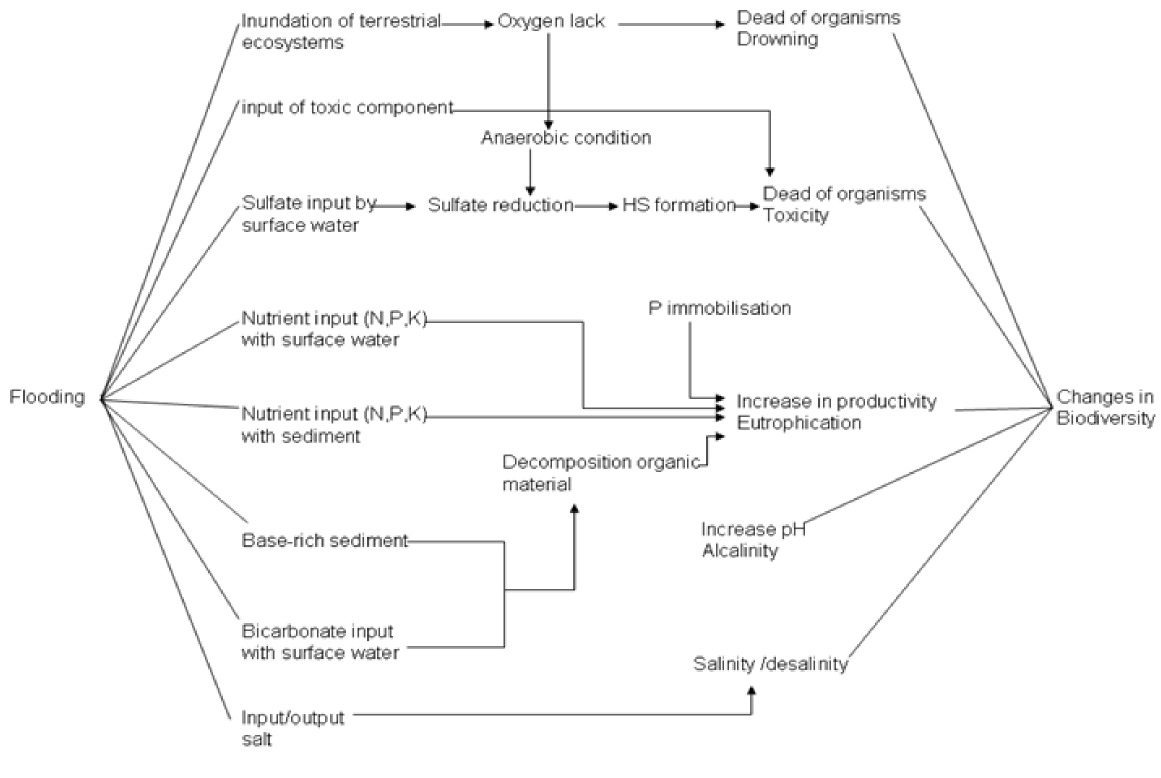


Figure 24: Determining processes for the effects on inundation on the the species composition of ecosystems (Runhaar, Talsma et al. 2004).

4.3.1 Direct effects: Oxygen depletion – drowning of vegetation

From a physiological point of view, water is not injurious to plants, but waterlogging is often accompanied by oxygen deficiency which affects root respiration. Because the oxygen content and the diffusion rate for oxygen is much lower in water then in air, oxygen depletion is a major risk for flooded terrestrial vegetation. To what extent the oxygen depletion occurs, depends on initial oxygen content, the turbulence of the flood water, the duration, microbial oxygen demand and the flood depth.

The rise rime of an inundation and the presence of high turbulence can be important for the oxygen content of the flood water. In case of very sudden inundation, oxygen can be enclosed when the water infiltrates the soil and can there be available to roots and bacteria. How fast the oxygen is depleted during the flood-event, depends mainly on temperature, the quantity of easily biodegradable organic material and the initial biological activity. The temperature is determinant for the maximal concentration of dissolved oxygen in the water. The colder the water, the higher the dissolved oxygen levels. With temperatures above 5° C, the bacterial decomposition and plant root activity rises and therefore also the associated oxygen demand rises. The presence of easily degradable organic material can promote bacterial decomposition and thus speed up the oxygen depletion. The time (season) of the flood event is thus determinant in many ways for the occurrence of oxygen depletion. Furthermore, in winter, the vegetation is inactive (not growth stage) and the effects of oxygen depletion are less severe. It is expected that, in comparison with a winter flood, the effects of oxygen depletion during the spring-summer season is much higher because of the active root growth (Knaapen 1990).

It is expected that oxygen depletion is primarily a critical factor for the flooding of vegetation on dry and moist soils (aerated soils). Vegetation on wet soils (periodically to permanently water saturated) is most likely adapted to anaerobic root zone conditions. One of such adaptations can be the presence of aerenchym – these structures allow cellular transport of air to the root system (Tiner 1996). Whether plants can use this strategy depends on the flood depth. Once the leaves are submersed the strategy fails evidently. A number of adapted species can stretch their leaves and branches to keep them above water. This mechanism fails during sudden and deep inundations since the stretching requires some time.

4.3.1.1 Possible effects of water storage to forests

Flooding vegetation communities impacts abiotic conditions and soil characteristics. Altered conditions ultimately impact the root system and eventually the whole tree. The magnitude of the impact depends on the duration and the timing of the flood (growth season). Flood events with a duration of several weeks (> 14 days), during the growing season will result in reduced growth for the entire growing season. The reduced growth is only visible several months after the flood event. Growth reduction or zero growht can lead to tree mortality. Some tree species have a higher tolerance against flooding. The effect of a flood event is determined by the effect of the flood even ton the soil structure. The soil structure is altered because the soil aggregates are disturbed and soil sealing occurs. The oxygen diffusion decreases CO_2 accumulates. A whole range of toxic components are produced by the anaërobic mineralisation of organic material in the soil. The toxic effects of reduced chemical compounds is usually limited to the duration of the flood. The effect is higher, when floods occur on soils that are normally moist to dry. Sites with a high soil moisture usually have adapted species that can deal with low soil oxygen diffusion and reduced, potentially toxic compounds. The first reaction to a flood event, is the reduction of stomata activity. This can be a reaction to the reduced hydraulic conductivity of the root system. Other studies claim that this is a reaction to a hormonal signal from the root system. Independent of the mechanism, stomata activity can be severely reduced within hours after the flood water has saturated the soil. A direct consequence is cessation of photosynthesis and transpiration. The production and transport of sugars therefore also stops. Flooding reduces the permeability of the roots for water transport, by the lowered oxygen levels. The root growth is strongly decreased. Eventually, roots and mycorrhizae die and decompose. Root mortality is enhanced through infection with funghi (e.g. *Phytophtora spp)*.. This will alter the root shoot balance, which often results in a higher drought intolerance after the flooding. Frequent floodings will also lead to shallow rooting trees, with a higher wind vulnerability. The impact of flooding on the growht and mortality of unadapted tree species is severe and occurs through different interacting processes of plant physiology.

4.3.1.2 Adaptation strategies of vegetation types to flooding

The ability of plants to resist waterlogged conditions depends on their ability to mobilise oxygen. This ability depends on plant species and variety, on the type of root system, on the stage of development as well as on external factors such as soil and air temperature and availability of oxygen in the root zone. Plants are, however, able to adapt themselves to waterlogged conditions. In the unsaturated part of the soil, plants may develop superficial roots to mobilise enough oxygen for respiration. Plants resistant to flooding conditions may develop physiological adaptations. These changes include the emergence of adventitious roots, the formation of hypertrophied lenticels at the stem base, the development of aerenchyma tissue and a gas transport system based upon the physico-chemical effect of thermo- osmosis (Eschenbach and Ludger 1999). These aerenchyma create a pathway of low resistance for the diffusion of oxygen from the air. Finally, oxygen may be translocated from leaves to the roots through the air cavities in the plant. Flood tolerance thus depends strongly on the species, the genotype, age and the timing, duration and water quality of the flooding. Evidently, floodings during the growth season are more harmfull. Firstly, because the vegetation needs more oxygen. Secondly, because water temperature is higher and contains less dissolved oxygen. Thirdly because soil microbial processes ar more active and consume more oxygen.

In general Angiosperms can withstand flooding better than Gymnosperms. Vital adult trees, can better withstand floods, than seedlings and overaged trees. Stagnating flood water is more harmfull that flowing flood water (flood zone is part of River bed) because the water is better mixed and has higher oxygen levels. During floods, the cambial tissue is also affected by the reduced oxygen transport (transpiration dependent). Certain flood tolerant tree species appear to have cambial cells that have a higher permeability for air. Metabolical adaptations to withstand decreased oxygen availability go along with the capacity to control anaerobic respiration. Flood tolerant trees could limit ethanol production by transforming them to non-toxic organic acids.

4.3.2 Indirect effects

4.3.2.1 External eutrophication

During flood events, nutrients can be brought into the floodplain ecosystem. This can lead to increased plant productivity and possibly to low biodiversity vegetation types, dominated by shrubs (Knaapen and Rademakers 1990; Higler, Beije et al. 1995). Together with the sediment that is brought in during the flood, nutrients can be adsorbed to the sediment particles. The amount of nutrients that is deposited per square meter is dependent on the sediment quantity, but also dependent on the grain size of the sediments. Finer particles like clay and silt stay longer in suspension and have relatively more capacity to adsorb nutrients. Furthermore because of their small grain size clay deposits are more compact and less permeable so that nutrients are less easily flushed away by water. The sand particles are usually deposited on the river banks, while the finer sediments accumulate in the stagnant floodplain water. Although in quantity the sand deposition is much higher per m^2, there is less nutrient content in the sand deposition. For lowland river systems, the knowledge basis is relatively poor. There is ongoing research on how sediments are transported into the floodplain and how much is deposited during a flood event. But this is a very complex topic, since the nutrient and sediment load and characteristics largely depend on the characteristics of the entire catchment. Not every peak flow will have the same load. A peak flow can be gradual from longlasting precipitation (interflow) or intense as a consequence of a storm. Also the season will play a role in the erodability of the land.

Nutrients can also be transported with the flood water in a dissolved form like nitrate, ammonium or phosphate. The quantities that are brought into the floodplain ecosystem are dependent on the stream typology and the concentration levels during peakflows. Ammonium can have a very quick increase of concentration levels during the quick rise of the peak flow. Once the flows do not increase significantly, the ammonium concentration drops.

Depending on the hydrograph of the flood event and the floodplain system (critical water level for the inlet of the floodplain system), there can be nutrients transported to the floodplain. The general consensus is that the input of dissolved nutrients is limited in comparison to the sediment adsorbed nutrients. In addition, much of the dissolved nutrients are again exported from the floodplain when water levels drop. The effective input is thus probably even lower; Furthermore the nitrate can easily be processed through denitrification (Venterink, Vermaat et al. 2005). It is not clear to which extent the surface water quality affects the floodplain ecosytems and it is even more uncertain if the current monitoring programs can capture the effective nutrient and sediment concentrations during a peak flow event. The concentrations and sediment, content can quickly change during these dynamic hydrological conditions. According to Runhaar and Jansen (2004), sedimentation – and the presence of seepage – would be a more determining factor for productivity.

4.3.2.2 Internal eutrophication

Indirectly, flooding can also lead to an increased nutrient availability by biochemical transformation from pools of biochemical unavailable nutrient form Figure 25. When oxygen is depleted from the soils, other chemical substances are used for oxidation of organic material (Mitch and Gosselink 1993). In a first phase, after depletion of oxygen, nitrogen is used as oxidator. Denitrifying bacteria transform nitrate to Nitrogen-gas. When all nitrogen is used, sulphates will be used as oxidator. Reduction of sulphate results in the formation of sulfides that can bind with $Fe2+$ to ironsulfide. When nitrogen removal from the ecosystem is high, inundation usually leads to an increased mobilisation and bioavailability of Phosphorous (Runhaar et al., 2004). The release of P is influenced by a number of processes that are site specific and cannot be generalized in thumb rules. Because Phosphates are commonly adsorbed to Ironhydroxide, processes that influence Iron also influence the availability of phosphate. One of the most important processes for P-mobilisation is the reduction of $Fe3+$ to the more mobile $Fe2+$. This takes place as a consequence of the low redox-conditions that result from anaerobic decomposition of organic material. Phosphate binds to iron oxide hydroxide through a direct ionic interaction between one or two negatively charged oxygen ions on the phosphate with the ferric ions ($Fe+++$) in the solid phase. When the reduction of $Fe3+$ to the more mobile $Fe2+$, takes place, the phosphate becomes available (Mitch and Gosselink 1993; RODEN and WETZEL 2002). Also the reduction of sulphate to sulfides could contribute to phosphate mobilisation. Sulfides have a higher capacity to bind Fe(II) hydroxide and could transform Fe(III) hydroxide-PO4 complexes (Fe3(PO4)2) to Fe(II)S and PO4. Phosphates can also become available due to mineralisation of organic material. This mineralisation is enhanced by a more neutral pH. The addition of the buffering Ca(CO3)2 due to flooding can thus enhance mineralisation. This effect can also play for addition of nitrates and sulphates, as they act as oxidators (by lack of oxygen). Because both the decomposition of organic material as well as the reduction of Fe(III), are temperature dependent processes, it is expected that the mobilisation of Phosphate is limited during winter flooding (Loeb and Lamers 2003; Loeb, Van Daalen et al. 2005). Finally, surplus phosphate can be released due to reduction of sulphates that are brought in with the flood water.

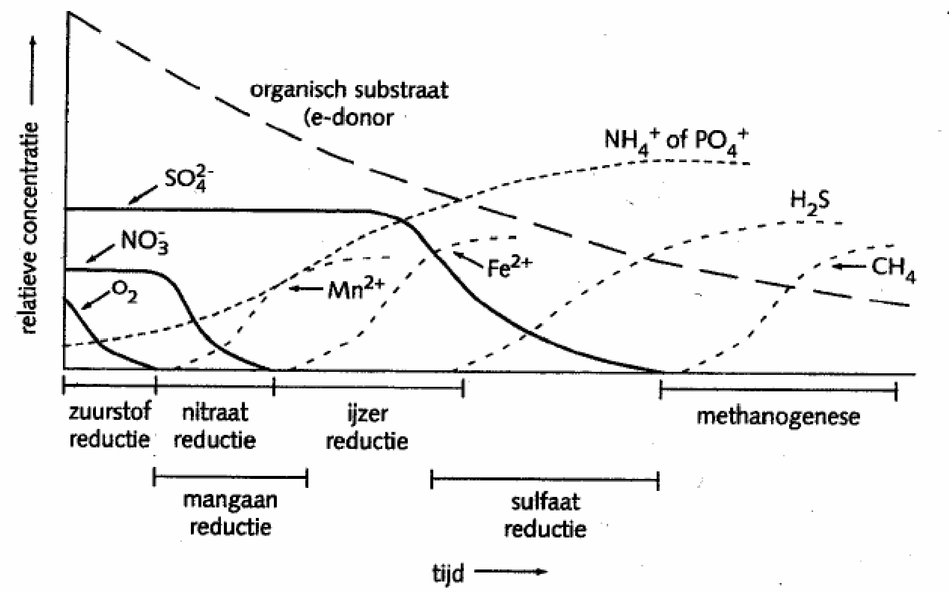


Figure 25: Reduction processes as a function of inundation duration and relative concentrations of the different substances that can be used for the oxidation of organic material (Mitch and Gosselink 1993).

The addition of sulphates in aquatic environment has a very high probability to mobilise phosphate. For terrestrial ecosystems, the addition of sulphates does not always lead to phosphate mobilisation. Lamers (1999) found that inundation with sulphate -rich water (200-400 mg/l) led to P-mobilisation (Lamers, Smolders et al. 1999). According to Loeb et al (2005), this surplus release that can be accredited to the sulphate addition, is neglectible, compared to the P-mobilisation from the iron reduction. The release of phosphate is dependent on the type of soil that is inundated. The inundation of iron rich soils with sulphate rich water leads to additional phosphate release, whilst this is not –or les- the case for soils with low iron content. Also in soils with low organic content, there is low phosphate mobilization because electron-acceptors lack to reduce the Fe (III). In contrast with this hypothesis, Kemmers et al (2003) found a much higher phosphate mobilization in soils with low iron and sulphate content (Kemmers, Delft et al. 2003). In the other sites, there was no difference found between inundation with sulphate rich and sulphate poor water.Whether phoshate mobilisation is a real threat for water storage in terrestrial ecosystems remains unclear. The processes that are described occur frequent in areas where the groundwater level is managed actively to sustain maximal groundwater levels until the summer period (drought mitigation). During water storage, phosphate mobilisation can only occur during a short period and it is expected that it is either washed away or re-fixated when the soils return from their waterlogged state (Smolders, Lucassen et al. 2003; Runhaar, Talsma et al. 2004). It is also unclear how much nutrients potentially can be released from a soil during a temporal flood event. After all, the local hydrology and groundwater regime will be determining for the level of internal eutrophication from inundation. The presence of seepage may prevent the floodwater intrusion into the soils and minimize the impact of the flood event. Also the seepage water usually contains enough iron to fixate both phosphates and sulphates.

4.3.2.3 Alkalinisation - acidification

Because the surface water is often alkaline through dissolved bicarbonate and sediment fixated calcium, the water storage has a pH-buffering function. Temporal flooding can contribute to the buffering capacity of the ecosystem and prevent acidification. According to Runhaar (2004) there is enough bicarbonate in the flood water to compensate the acidification-process. It is unclear if the bicarbonate can diffuse fast enough to the soils and whether the CO_2 gas that is produced from the buffering, can evacuate the soils quick enough to actively contribute to the buffering capacity.

In contrast, an inundation of natural weak-buffered acidic ecosystems can lead to a rise in pH and a loss of those species that are adapted to more acidious conditions. In addition, a rise of the pH in these acidious ecosystems can lead to enhanced mineralisation and the mobilisation of nutrients.

4.3.2.4 Toxification

As mentioned earlier, an oxygen deficit occurs in the soils during longlasting inundations and anaerobic decomposition processes will take place. As a consequence, there a number of toxic chemical substances are formed (H_2S, Fe^{2+}, NH_4, Mn^{2+}). Plant-species that occur at permanently waterlogged soils have adapted mechanisms to deal with these toxic compounds. Some species have aerated tissue to transport oxygen to the root system and to oxidate the toxic substances (which are all in reduced state). For species of more dry soil conditions, the oxygen deficit seems to be the most critical factor. In most cases the root system is heavily degraded before the toxic effects can take place.

4.3.3 Flood return period and vegetation: recovery, rejuvenation or adaptation

The capability of the ecosystem to adapt to inundation is dependent on the frequency and regularity of the inundation (Runhaar et al, 2004). As illustrated in Figure 26, there is a high occurrence of adapted species that are subjected to inundation on a regular basis (daily – yearly). In areas that are flooded less frequently (less than once each year), there will be no selection of inundation tolerant species. For return periods of more than one year, inundation sensitive species have time to recover from the flooding, but there will be no massive colonisation of inundation tolerant species. The time needed to recover depends on the vegetation type and thus the return period determines which vegetation types can occur given the return period. Vegetation types of dynamic systems can recover rather quickly (a few years), whilst vegetation types of low-dynamic environmental conditions (some forest types) can need recover times of decades to centuries (Runhaar et al., 2004). Bottomline is that vegetation types can occur if their recovery time is less than the return period. Finally, the potential to recover and reoccupate is also dependent on spatial aspects such as fragmentation-level, size of the area. The micro topography and topographical gradients of an area can be determining to the survival of relic populations (patches), which can reoccupate the area afterwards. In this way flooding can increase the biodiversity by emposing gradients of reoccupation and recovery.

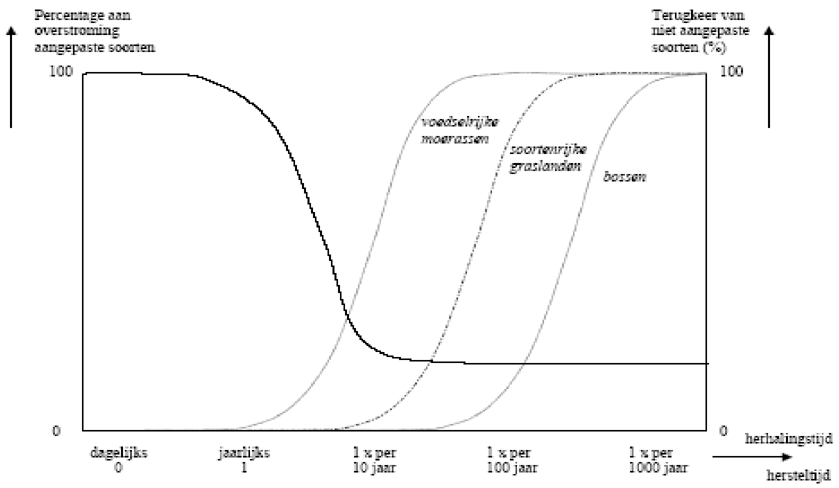


Figure 26: Completeness of the ecosystem in function of flood return frequency.

4.4 Conclusions regarding the effects of inundation on species composition.

Regarding the effects of flood storage on species composition, we can pose the following statements that summarise the relations that are displayed in Figure 27:

- The severity of oxygen depletion during floods is dependent on the turbulence of the flood water, the timing (season, temperature), the duration and the depth of the flood.
- The turbulence of the initial flood water is determining for the initial oxygen content of the water. The higher the initial oxygen levels, the longer it takes before it is depleted.
- The timing of the flood is probably one of the most determining factors. Flooding during winter periods has lower impact because of lower water temperatures, which enables a higher dissolved oxygen content and a lower microbial oxygen demands (lower microbial activity) both decreasing the probability of oxygen depletion during floods.
- Flood depth is determining for species that are capable to transport oxygen to their root system (like sedges and pitrus). If water levels exceed the vegetation height, this system of oxygen transport is disabled.
- Inundations during winter have less effect on the survival of plant species the vegetation is not in active growth stage, which means that damage of eventual oxygen depletion is much lower. During active growth flooding can have negative effects on plant survival. Especially species with a low regeneration capacity (which are sensitive to oxygen depletion), will experience negative effects. These species mostly occur on moderately moist to dry soils (Runhaar et al., 2004).
- Longlasting and deep inundations during the growth season will eradicate almost all species. After water retraction, species with a high regeneration capacity and species that depend on water transport of seeds will benefit from the flooding (Tandzaad, Dotterbloem, etc…).
- Frequent flooding will promote the settlement of species that are adapted to soil oxygen depletion.
- Dynamic flooding from riverbank overtopping can promote erosion and sedimentation patterns. Open eroded patches create dynamic conditions, where pioneer-vegetation can settle.
- Erosion and sedimentation pattern in the floodplain creates a wider range of abiotic conditions and promotes recolonisation from survival species on elevated patches.
- Floods can bring in sediments and substantial amounts of adsorbed nutrients.
- Dissolved nutrients can be brought in to the floodplain ecosystem, but the quantity is probably less important, compared to adsorbed sediment nutrients. Initial loading is less high and a large portion of dissolved nutrient retracts together with the floodwater. In addition, anaerobic conditions can remove nitrogen through denitrification.
- Longlasting inundation can lead to phosphate mobilisation and to eutrophication of the ecosystem. Reduction of Iron (III) releases phosphate from chemical complex with iron hydroxides.
- The presence of sulphates in the flood water can promote extra release of phosphate.
- The addition of sulphate rich water to soils with low iron content can promote the release of the toxic hydrogensulfide (H_2S).
- Short duration flood events cause less eutrophication because the stage of iron reduction is usually not reached.
- High groundwater levels and active seepage reduces the risk of internal eutrophication and H_2S-emmisssions.
- Nutrient deposition or release in lowproductive high biodiversity ecosystems, often leads to dominance of general species. Especially phosphate release will have sugnificant effects in phosphate limited high biodiversity ecosystems.
- In contrast, some vegetation types and species can benefit from the effects of water storage on abiotic conditions, specifically combinations of acidity and nutrient availability. For example “Dottergraslanden en Grote Zeggenvegetaties” largely depend on nutrient input from flooding.

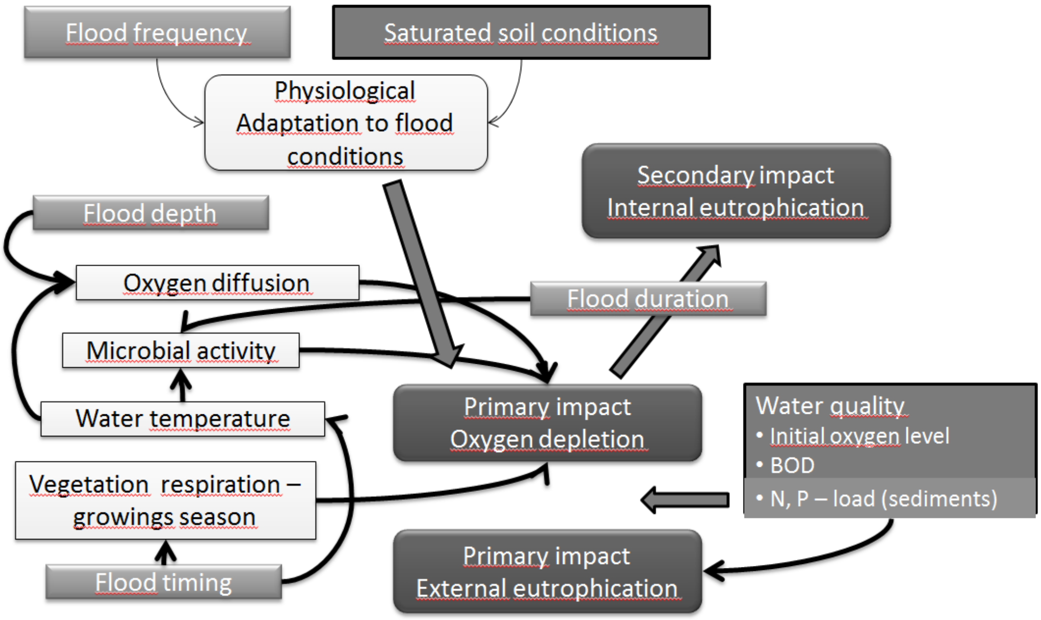


Figure 27: Complex interrelations between flood timing, duration, depth and the different impact mechanisms

The effect of water storage on vegetation thus largely depends on the characteristics of the vegetation itself and the frequency of flooding. Vegetation types that are less resistant to the effects of flooding will suffer negative effects. Depending on the season, duration and depth of the inundation, sensitive vegetation types will disappear, suffer temporal effects (recovery) or loss of the most sensitive species from the vegetation community. Frequent flooding will benefit inundation tolerant species. If current vegetation types are subjected to a frequent flooding regime, there will be a gradual shift to flood tolerant vegetation types. But also flood sensitive vegetation types can occur if their recovery time is less than the flood return period. Inundations thus can offer favorable conditions for the development of valuable flood resistant vegetaion types. Usually these are vegetation types that are adapted to wet soil conditions and thus belong in floodplains and valley. Flooding can create a certain degree of perturbation and variation (micro topography, soil sealing, soil texture, nutrient availability, succession stage renewal) within the floodplain and thus generate a wide range of abiotic conditions and associated biodiversity. The restoration of floodplain water storage capacity can provide opportunities to counter desiccation and acidification and restore valuable wet ecosystems.

One the one hand, opportunities for rewetting through flood storage are dependent on the hydrological characteristics of the flood events. It is determining that the flood water remains within the floodplain and contributes to groundwater level of the ecosystem. The morphology of the floodplain and the presence of drainage systems will determine whether there are opportunities for rewetting through water storage. In general it is assumed that temporal flood storage in a strongly drained area, cannot contribute to rewetting objectives. On the other hand, water storage with surface water can lead to external eutrophication and/or internal eutrophication wich will lead to low biodiversity vegetation types, dominated by highly productive schrubs.

4.5 A methodology to estimate flood vulnerability, based on the Biological Valuation Map

The **theoretical** assessment of ecological vulnerability to flood characteristics seems straightforward when the different mechanisms and variables are considered individually. However when assessing this vulnerability in climate context, we are confronted to **practical challenges** of data availability. The assessment methodology is not flawless and will exclude most of the parameters.

Firstly, because the parameters cannot be provided by models. Secondly because we do not have the spatial information on the floodplain characteristics. And thirdly because cannot determine a clear cause-effect relationship between flood chanracteristics and vegetation development (e.g. because of physiological adaptations).

Firstly, most hydrologic models only provide output on maximal depth and flood extent within a certain time window (5, 10, 25 … years). This is insufficient, since the vulnerability of ecosystems to flood is mainly depending on the season, duration and water quality. Also the hydrological models are focussing only on the main river and the urban areas at risk (the river tributaries are often not included in the models). **Secondly**, we often face the unavailability of detailed vegetation maps. The Biological Valuation Map is not a detailed vegetation map, but represents parcel level assessments. The mapping units are not comparable with international vegetation mapping units and needs to be translated in order to apply scientific insights from literature. In order to make a correct assessment, the spatial distribution of vegetation species should be known at a relatively high resolution. This would require a vegetation classification obtained from high resolution remote sensing data. **Thirdly**, floodplain biodiversity certainly not only determined by flooding characteristics only, but also by other pressures including land cover and pollution. In many cases, biodiversity has been impacted by fragmentation, eutrophication, drainage,… The effect of changes in flood characteristics induced by climate change will be dependent on how these changes relate to other pressures on biodiversity, both current and future. For certain flooding characteristics have changed in the past, as consequence of embankments, river normalisation, increased paved area run-off, storm drainage infrastructure…making flood event more extreme and irregular. In addition, floodplain biodiversity assessments should be analyzed from a landscape context. Floodplains with a structural diversity in topography and gradients will be more resilient against changes in hydrological regimes, as the abiotic conditions merely will shift from one location to another.

We present a methodology to evaluate the constraints of flooding characteristics on floodplain vegetation. Dependent on the flooding characteristics, vegetation will be adapted to flooding or will be recovering from flooding. Adapted vegetation is by definition not impacted by flooding, but supported by flooding. Not adapted vegetation is recovering from flooding and will be determined by the severity of the flood impact and the recovery window between impact events (time period between impacts). If this window is short, only grassland and shrubs are maintained. The presented methodology allows calculating an index score for a series of flood events within a certain time frame. This then will be confronted with the actual vegetation vulnerability scores. The Biological Valuation Map (BVM) is a standardized survey and evaluation of the biotic environment of Flanders and the Brussels Capital Region. The BVM is a ecotope classification map which is also used frequently as a land use map (Schneiders and Verheyen 1998; Van Eetvelde and Antrop 2005). The multiple layer ecotope-classification units were largely defined on the basis of vegetation, land use and small landscape elements and has a parcel level resolution. The Biological Valuation Map classification units are not recognized or not even comparable to standardized international vegetation classifications. Therefore a translation needs to be done to international recognized vegetation types. The 970 mapping units of the Biological Valuation Map were attributed one or more of the 82 different vegetation types that we derived from literature (see Annex A). The 82 vegetation types that have been selected are derived from the "Systematiek van natuurtypen voor Vlaanderen". Some vegetation types are to heterogeneous for the assessment of their flood tolerance. Where possible, these vegetation types are subdivided in subtypes. Many vegetation types occur in potential and/or effectively flooded areas. But their occurrence is not necessarily related to flooding. In many cases, flooding characteristics are of minor importance, compared to e.g. groundwater seepage. A problem for the determination of flood tolerance is that many flood parameters are relevant to potential effects on vegetation (return, period, season, duration, depth, flow velocity, sediment load, dissolved nutrients, alkalinity, sulphate concentration, soil texture and soil moisture). For many of these parameters, there is little quantitative knowledge on the cause-effect relations of changes in these parameters.

For each vegetation type information was gathered in order to complete the flood vulnerability assessment and information was sought that relates to inundation tolerance, appearance in relation to flood timing, frequency and duration of inundation, origin of flood water, productivity and acidity. The information was then compiled in one flood vulnerability assessment table containing all the characteristics of the 82 vegetation types relevant for flood impact through oxygen depletion. This translation exercise is ambiguous and many BVM-classifications can be attributed to multiple vegetation types. It is also possible that one of the layers has a mapping category that could not be attributed to a vegetation type. So we can conclude that there is no one to one relationship between the vegetation types and the BVM. The ecological vulnerability analysis considers the first three layers of the BVM (Figure 28). The first layer maps the dominant ecotope for the parcel, the second layer the co-dominant, the third layer indicates patches of secondary ecotopes, Each parcel can thus have a combination of layers and each of these layers indicates one of the 970 different mapping categories. For Flanders up to 4397 unique combinations were derived upon the first three layers of the BVM. It is the case that vegetation types, derived from the second and third layer of the BVM, are often confirming the vegetation types that were derived from the first layer. For each of the 82 vegetation types, the ecological vulnerability to the different flood types has been determined (Annex B). There is a wide range of flood type classifications used on literature. In addition, they use rather qualitative descriptions of impacts on vegetation. Furthermore the classifications used are also qualitative such as “periodical, incidental, long-lasting, short duration”. It is thus difficult to observe clearly demarcated classifications of frequency/regularity, duration and timing. The flood types that are considered were based upon literature study and essentially include the different aspects of flood events, timing, regularity, duration and depth. A flood event is no static process. Water levels rise and decrease during a flood event, making flood duration/depth spatially heterogeneous. If flood plains have high topographic variation, this creates gradients in flood conditions and associated ecological values. They will probably exhibit a higher biodiversity and ecological resilience to changes in flooding regimes. Ideally, a return period for each flood type (season, depth and duration), is needed for ecological vulnerability assessment. For each pixel we need the return period for each combination of duration, season and depth. Each flood type occurrence contributes to the development or collapse of a vegetation type. To make abstraction of a flood event, we classified 12 arbitrary conditions. The variables are timing (growing season, non growing season), duration (- 14 th day, +14th day) and depth (0-20 cm, 20-50 cm, + 50 cm). These flood types are expanded by a regularity classification. Each of these basic flood types can occur frequent (annual), regular (bi-annual), irregular (every 2-5 years) or seldom (more than 5 years between occurrence). If a certain flood type occurs regular, vegetation is likely to adapt to these conditions. As there are multiple vegetation types possible on a BVM-parcel, we need to aggregate ecological vulnerability. Because the actual vegetation type in the field is uncertain, we need to incorporate this uncertainty by designing weighting schemes. Each BVM unit can represent up to 5 vegetation types and there are three layers considered for ecological vulnerability assessment. The weighting options reflect different levels of precaution.

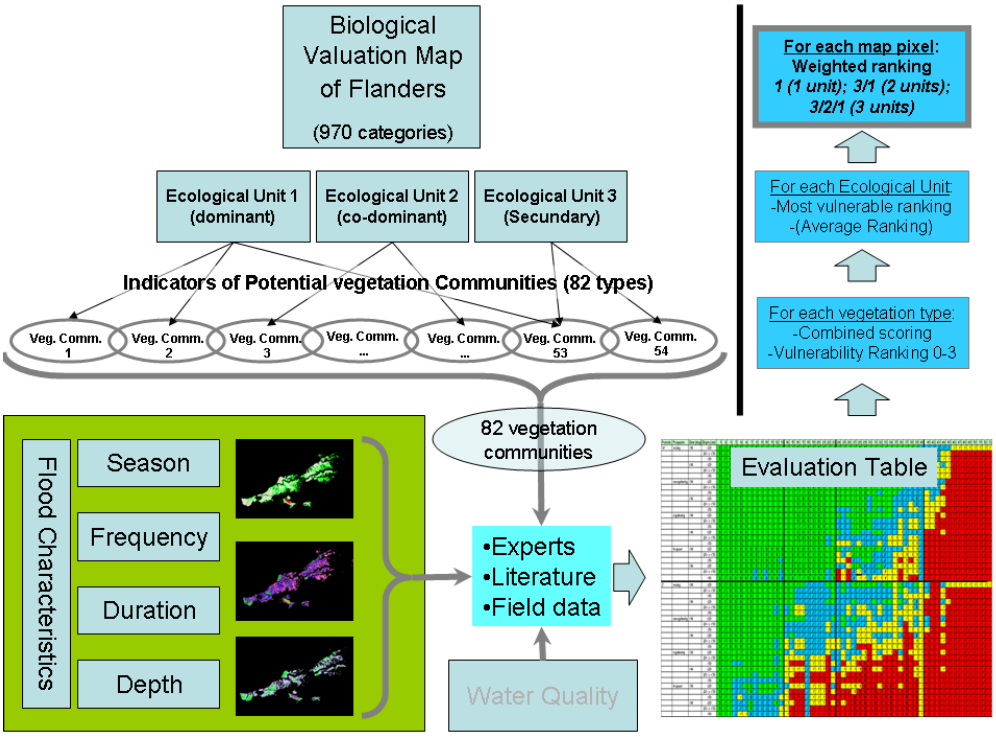


Figure 28: Basic methodology for the determination of flood type impacts on vegetation maps , derived from the Biological Valuation Map.

The total Ecological vulnerability was determined by 5 different methods and for 48 different floodtypes.

- Method 1: Most sensitive vegetation type for each layer, followed by a weighted average of the different layers (depending on the presence of BVM categories in the second and third layer).
- Method 2: Most sensitive vegetation type for each layer, followed by an average of the different layers
- Method 3: Most sensitive vegetation type for each layer, followed by selection of the most sensitive from the different layers
- Method 4: Averaged sensitivity for each layer, followed by a weighted average of the different layers (depending on the presence of BVM categories in the second and third layer).
- Method 5: Averaged sensitivity for each layer, followed by an average of the different layers

Ecological risk is not equal to the ecological vulnerability. One can have a very vulnerable vegetation types with a low ecological value (e.g. pine forest). On the other hand, there can be very valuable vegetation types (e.g. alder forest) that have a very low vulnerability when flooded. Therefore the total ecological vulnerability maps, needs to be combined again with the 7 class ecological valuation layer of the BVM. Also here, the sensitivity of the scoring needs to be investigated. It can be that the presence of very valuable elements as relics can be as important as the dominant vegetation type.

4.6 Results: Application of flood type classification on the Zammelsbroek Floodplains

The Zammelsbroek floodplains are located in the most downstream part of the Grote Nete Model at the confluence of the Grote Nete and the Grote Laak Figure 29. The area is also part of the SIGMA II project (http://www.sigmaplan.be/index.php?page=vallei-van-de-grote-nete).

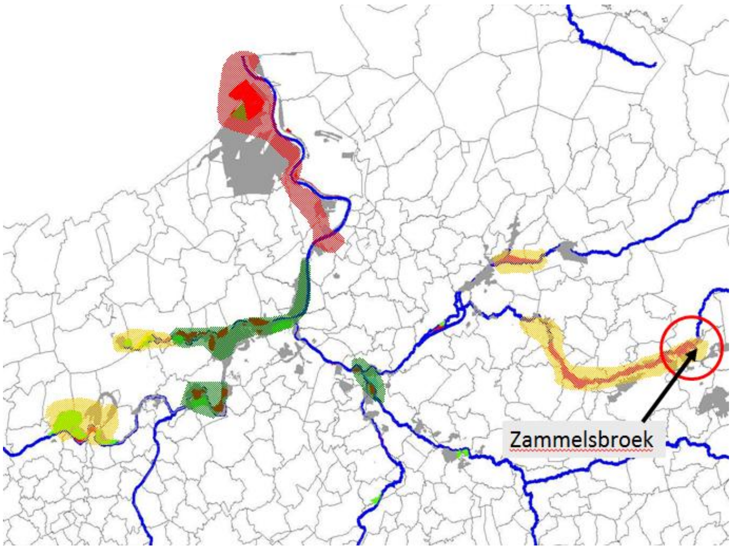
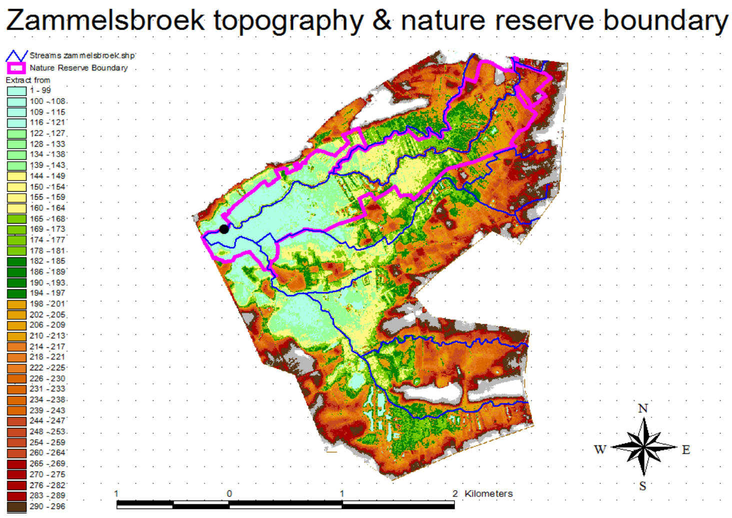


Figure 29: location of the zammelsbroek and the SIGMA-project zone

Because the current flood modeling approaches were unsuitable to provide the needed output for ecological impact assessment for the entire catchment, the hydraulics Lab of KUL tested for the Grote Nete case the option to calibrate a simplified conceptual model to the full hydrodynamic model and run the long-term simulation in the simplified model. This test focused on the Zammelsbroek floodplain area.

For this floodplain, we succeeded to obtain a conceptual (reservoir-type) model for which the results are close to the full hydrodynamic MIKE11 model. Hourly time series available for the period 1986-2005 were simulated in that conceptual model in order to obtain long-term information on water levels and inundation depths in the floodplain, the spatial extent and temporal variations in the inundated area, and the duration of the inundations. Given that the Zammelsbroek area has frequent inundations, 26 inundations (including very small ones) were simulated in the period 1986-2005 and the simulation results for these events were statistically summarized for the lowest point of the floodplain (Figure 31; Table 6). In this way, information could be obtained on flood events with high frequency, which is of high importance for the ecological impact investigation. Evidently, the statistics for the lowest point of the floodplain are not valid in the higher parts of the floodplain.



The highest ecological values are situated within the perimeter for the nature reserve (Figure 30). This perimeter is the desginated zone for subsidized land acquisition for the nature reserve. The floodable area extents beyond the reserve perimeter and flood events can originate also from the Grote Laak, which joins the Grote Nete at the reserve boundary.

Figure 30: the topography of the Zammelsbroek nature reserve (maximal extent) and the wider environment

Table 6: summary of the 26 flood events for the reference time series at the lowest point in the floodplain

Start of event	number of events	average duration	stdev duration	mean average depth	Stdev average depth	mean max depth	Stdev max depth
jan	3	12,3	5,86	0,34	0,20	0,76	0,43
feb	6	18,3	7,06	0,52	0,24	1,16	0,55
mar	4	12	4,55	0,39	0,24	0,94	0,55
apr	1	18	NA	0,52	NA	1,34	NA
may	0						
jun	1	20	NA	0,44	NA	0,89	NA
jul	1	18	NA	0,48	NA	1,00	NA
aug	0						
sep	2	16,5	0,71	0,49	0,13	1,11	0,27
okt	1	14	NA	0,34	NA	0,69	NA
Nov	2	14	1,41	0,36	0,05	0,75	0,08
Dec	5	18	8,89	0,50	0,26	1,07	0,54
Total	26						

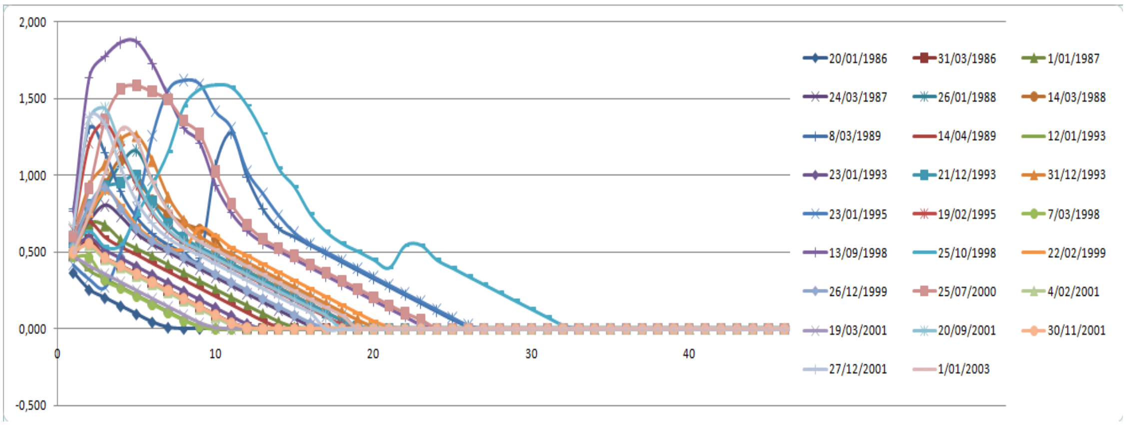


Figure 31: Depth-duration plots of the 26 flood events between 1987 and 2005 for the reference location.

The high climate change scenario was modeled for an identical period, but the precipitation time series of 1987-2005 was perturbated with the climate change effects. The results for the high scenario demonstrated that there was a significant increase of the flood frequency, depth and duration (Figure 32; Table 7).

Table 7: summary of the 36 flood events for the high climate change (wet scenario) time series at the lowest point in the floodplain

Start of event	number of events	average duration	stdev duration	mean average depth	Stdev average depth	mean max depth	Stdev max depth
jan	5	30,8	15,97	0,71	0,26	1,62	0,54
feb	6	30,5	17,82	0,70	0,37	1,37	0,68
mar	4	31	9,87	0,73	0,09	1,72	0,21
apr	1	27	NA	0,64	NA	1,84	NA
may	0						
jun	1	17	NA	0,51	NA	1,52	NA
jul	2	24	1,41	0,77	0,15	1,60	0,41
aug	0						
sep	3	20	2,65	0,76	0,27	1,87	0,58
okt	2	90	103,24	0,82	0,40	1,94	0,45
nov	4	19	1,83	0,64	0,14	1,47	0,33
dec	8	37,25	21,45	0,90	0,24	1,97	0,45
Total	36						

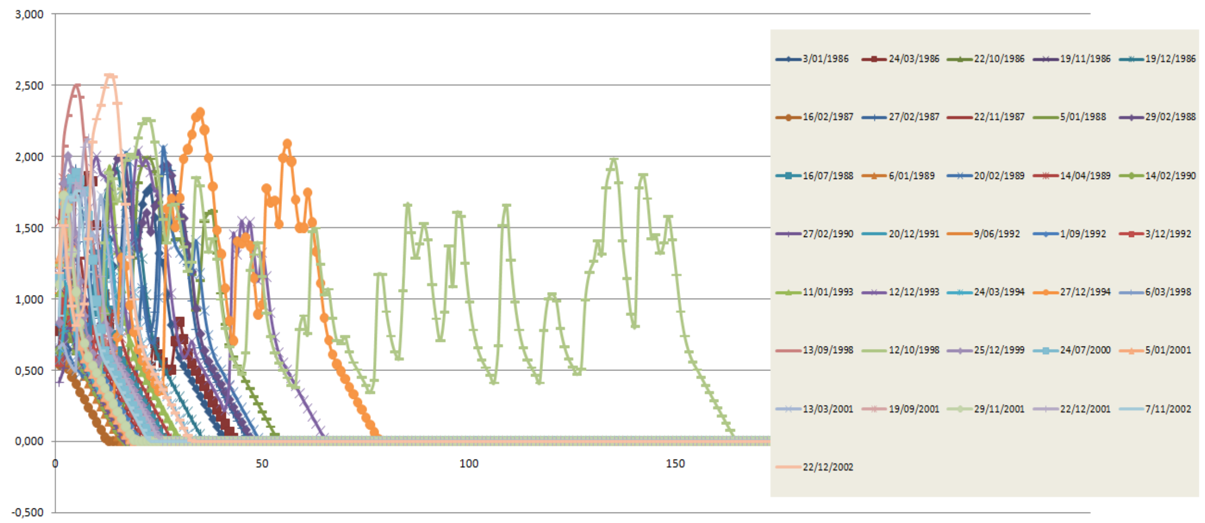


Figure 32: Depth-duration plots of the 36 flood events for the high climate change scernario and for the reference location (lowest point of the floodplain).

Table 8: Difference in number of flood events, duration and depth between the reference scenario and the high climate change scenario for the reference location

Start of event	number of events	average duration	stdev duration	mean average depth	Stdev average depth	mean max depth	Stdev max depth
jan	2	18,5	10,1	0,4	0,1	0,9	0,1
feb	0	12,2	10,8	0,2	0,1	0,2	0,1
mrt	0	19,0	5,3	0,3	-0,2	0,8	-0,3
apr	0	9,0	NA	0,1	NA	0,5	NA
mei	0	0,0	0,0	0,0	0,0	0,0	0,0
jun	0	-3,0	NA	0,1	NA	0,6	NA
jul	1	6,0	NA	0,3	NA	0,6	NA
aug	0	0,0	0,0	0,0	0,0	0,0	0,0
sep	1	3,5	1,9	0,3	0,1	0,8	0,3
okt	1	76,0	NA	0,5	NA	1,3	NA
nov	2	5,0	0,4	0,3	0,1	0,7	0,3
dec	3	19,3	12,6	0,4	0,0	0,9	-0,1
Grand Total	10	0,0	0,0	0,0	0,0	0,0	0,0

From the comparison of the reference scenario with the high climate change scenario for the lowest location of the Zammelsbroek floodplain (Table 8), we can deduct that the number of events increases from 26 to 36, but the increase is mostly in winter. The duration of the flood events increases significantly (almost doubles). For the lowest location of the floodplain we can observe "multiple peak flood events". This indicates that the actual number of events may be much higher, but is masked because the floodplain is not emptied. The mean depth increases with 50 % in average, compared to the reference scenario. The statistics for the lowest part of the floodplain give a comprehensive insight in the magnitude of the changes but are not valid for the entire floodplain. The Zammelsbroek floodplain has a distinct gradient in flood depth. Combined with the complex multiple peak flood events, there will be many more short flood events. Specific flood conditions in the lower parts of the floodplain for the reference scenario may occur in the higher parts for the high climate change scenario. If this hypothesis is valid, there may be opportunities for a gradual shift of vegetation communities along the gradient in flood characteristics.

Table 9: Flood type occurence (number of days) for each elevation interval within the floodplain for the reference scenario

floodtype	reference			0 cm	10 cm	20 cm	30 cm	40 cm	50 cm	60 cm	70 cm	80 cm	90 cm	100 cm	110 cm	120 cm	130 cm	140 cm	150 cm	160 cm	170 cm	180 cm	190 cm	200 cm	210 cm	220 cm	230 cm	240 cm	250 cm	260 cm	270 cm	280 cm	290 cm	300 cm
0	winter	< 14 d	< 20	18	14	18	30	30	30	21	20	12	14	7	5	6	9	1	2	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1	winter	< 14 d	20-40	23	30	34	54	47	33	30	26	21	12	11	15	10	3	7	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	winter	< 14 d	> 40	28	28	28	64	53	37	40	35	28	23	17	8	7	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	winter	> 14 d	< 20	27	28	28	7	10	15	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
4	winter	> 14 d	20-40	65	53	43	31	22	11	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
5	winter	> 14 d	> 40	199	162	122	41	28	24	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
6	summer	< 14 d	< 20	0	2	1	4	2	7	7	4	2	2	0	0	2	2	4	2	3	2	1	2	0	0	0	0	0	0	0	0	0	0	0
7	summer	< 14 d	20-40	0	3	3	6	7	11	6	4	2	0	2	4	6	6	5	5	3	3	2	0	0	0	0	0	0	0	0	0	0	0	0
8	summer	< 14 d	> 40	0	6	5	10	7	22	20	18	18	18	16	14	10	8	5	3	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	summer	> 14 d	< 20	6	6	5	5	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	summer	> 14 d	20-40	55	12	13	8	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	summer	> 14 d	> 40	56	41	35	23	19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 1: Flood type occurence (number of days) for each elevation interval within the floodplain for the high climate change scenario

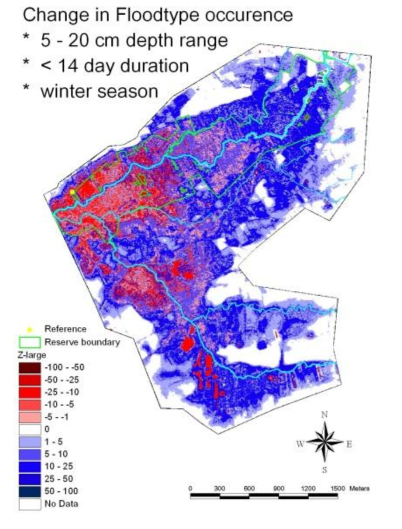
floodtype	high scenario			0 cm	10 cm	20 cm	30 cm	40 cm	50 cm	60 cm	70 cm	80 cm	90 cm	100 cm	110 cm	120 cm	130 cm	140 cm	150 cm	160 cm	170 cm	180 cm	190 cm	200 cm	210 cm	220 cm	230 cm	240 cm	250 cm	260 cm	270 cm	280 cm	290 cm	300 cm
0	winter	< 14 d	< 20	0	4	8	15	23	31	20	17	22	6	20	7	9	23	19	12	20	29	26	24	8	5	5	3	1	2	0	0	0	0	0
1	winter	< 14 d	20-40	0	6	18	40	39	28	26	25	24	27	13	21	23	25	30	36	39	44	32	13	10	8	4	3	2	0	0	0	0	0	0
2	winter	< 14 d	> 40	0	13	30	43	46	52	46	69	60	63	57	54	56	69	58	40	30	18	16	11	6	3	2	0	0	0	0	0	0	0	0
3	winter	> 14 d	< 20	651	45	48	49	49	57	41	23	22	18	22	13	16	23	10	13	9	2	0	0	0	0	0	0	0	0	0	0	0	0	0
4	winter	> 14 d	20-40	105	114	118	120	110	73	58	43	42	35	32	50	52	29	24	24	18	6	0	0	0	0	0	0	0	0	0	0	0	0	0
5	winter	> 14 d	> 40	708	634	546	447	383	337	299	252	219	196	177	134	103	65	47	34	18	6	0	0	0	0	0	0	0	0	0	0	0	0	0
6	summer	< 14 d	< 20	0	0	2	3	6	5	8	8	7	9	5	4	5	2	3	3	2	4	4	0	1	1	1	0	2	1	0	0	0	0	0
7	summer	< 14 d	20-40	0	0	3	7	8	5	11	16	14	9	9	7	5	6	5	6	8	4	1	2	2	1	2	3	1	0	0	0	0	0	0
8	summer	< 14 d	> 40	0	0	8	14	19	17	40	38	33	29	24	22	19	16	14	10	6	6	5	4	3	3	1	0	0	0	0	0	0	0	0
9	summer	> 14 d	< 20	200	14	11	10	9	10	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	summer	> 14 d	20-40	68	26	25	23	18	14	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	summer	> 14 d	> 40	113	100	79	59	43	37	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 2: Change in flood type occurence (number of days) for each elevation interval within the floodplain between the reference scenario and the high climate change scenario

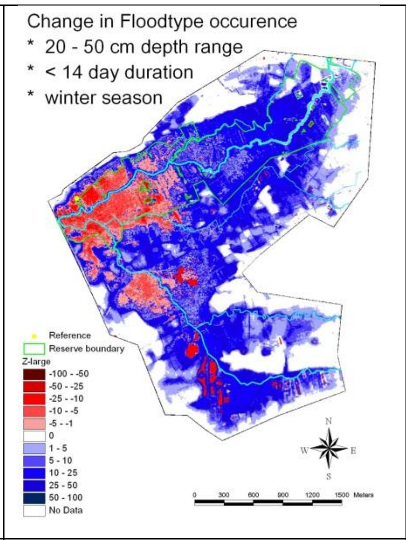
floodtype	difference			0 cm	10 cm	20 cm	30 cm	40 cm	50 cm	60 cm	70 cm	80 cm	90 cm	100 cm	110 cm	120 cm	130 cm	140 cm	150 cm	160 cm	170 cm	180 cm	190 cm	200 cm	210 cm	220 cm	230 cm	240 cm	250 cm	260 cm	270 cm	280 cm	290 cm	300 cm
0	winter	< 14 d	< 20	-18	-10	-10	-15	-7	1	-1	-3	10	-8	13	2	3	14	18	10	15	29	26	24	8	5	5	3	1	2	0	0	0	0	0
1	winter	< 14 d	20-40	-23	-24	-16	-14	-8	-5	-4	-1	3	15	2	6	13	22	23	31	39	44	32	13	10	8	4	3	2	0	0	0	0	0	0
2	winter	< 14 d	> 40	-28	-15	2	-21	-7	15	6	34	32	40	40	46	49	64	58	40	30	18	16	11	6	3	2	0	0	0	0	0	0	0	0
3	winter	> 14 d	< 20	624	17	20	42	39	42	38	23	22	18	22	13	16	23	10	13	9	2	0	0	0	0	0	0	0	0	0	0	0	0	0
4	winter	> 14 d	20-40	40	61	75	89	88	62	56	43	42	35	32	50	52	29	24	24	18	6	0	0	0	0	0	0	0	0	0	0	0	0	0
5	winter	> 14 d	> 40	509	472	424	406	355	313	290	252	219	196	177	134	103	65	47	34	18	6	0	0	0	0	0	0	0	0	0	0	0	0	0
6	summer	< 14 d	< 20	0	-2	1	-1	4	-2	1	4	5	7	5	4	3	0	-1	1	-1	2	3	-2	1	1	1	0	2	1	0	0	0	0	0
7	summer	< 14 d	20-40	0	-3	0	1	1	-6	5	12	12	9	7	3	-1	0	0	1	5	1	-1	2	2	1	2	3	1	0	0	0	0	0	0
8	summer	< 14 d	> 40	0	-6	3	4	12	-5	20	20	15	11	8	8	9	8	9	7	4	6	5	4	3	3	1	0	0	0	0	0	0	0	0
9	summer	> 14 d	< 20	194	8	6	5	4	10	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
10	summer	> 14 d	20-40	13	14	12	15	11	14	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	summer	> 14 d	> 40	57	59	44	36	24	37	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

A further in depth analysis can reveal the exact shift in flood type occurence. The different images of Figure 33 is a spatial visualisation of the changes in flood type occurence (Table 11).

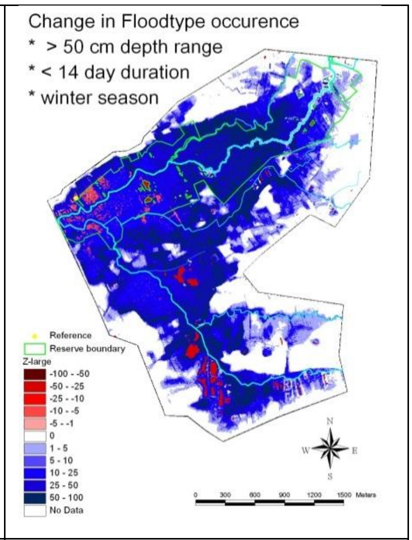
The figure is also available and printable at
http://www.belspo.be/belspo/ssd/science/pr_cluster_fr.stm



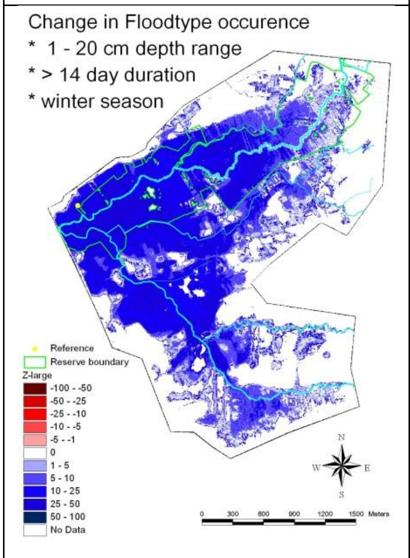
Floodtype 0



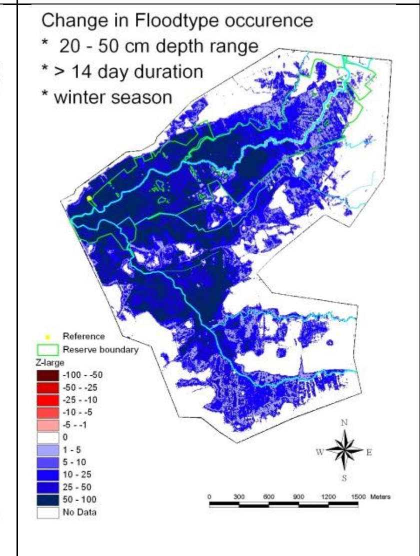
Floodtype 1



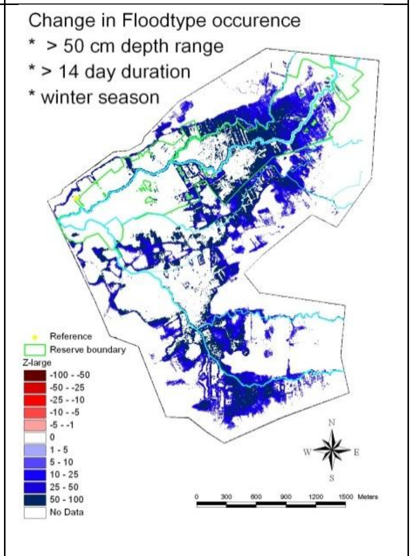
Floodtype 2



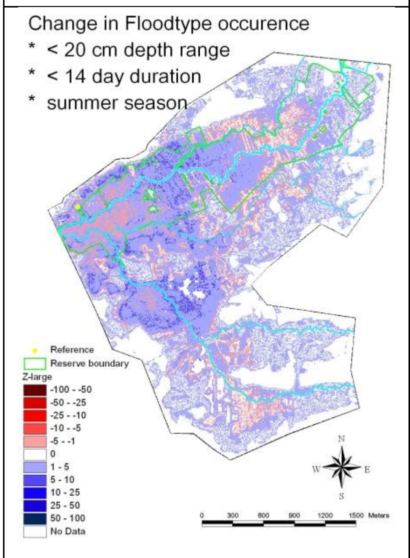
Floodtype 3



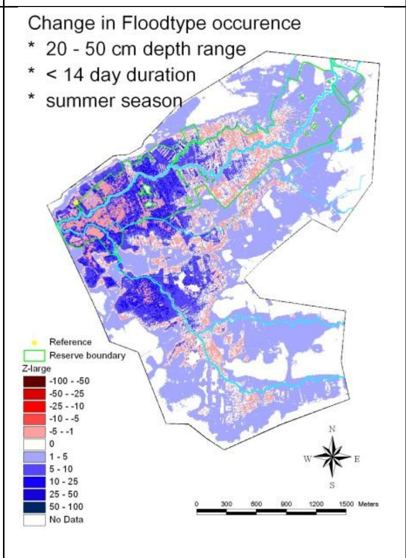
Floodtype 4



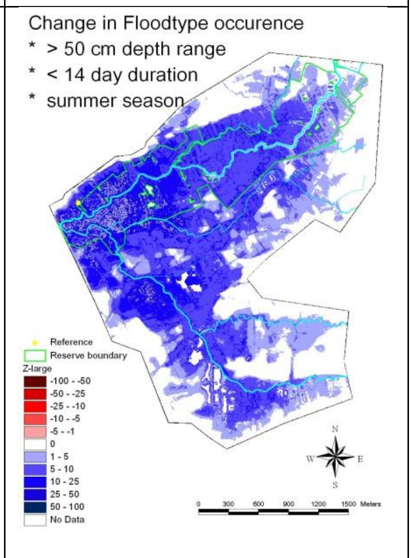
Floodtype 5



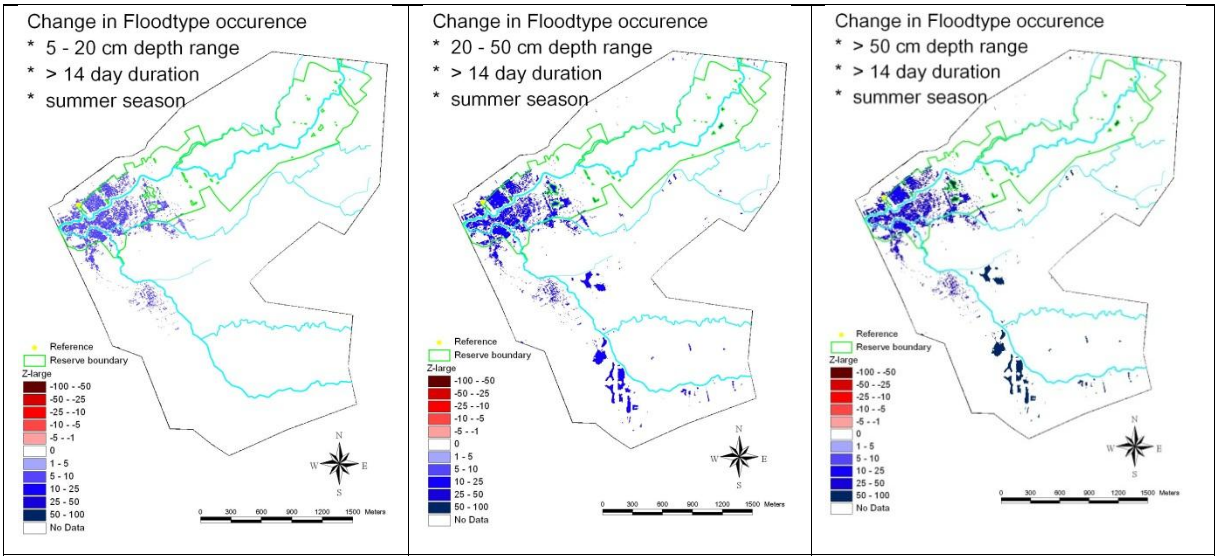
Floodtype 6



Floodtype 7



Floodtype 8



Floodtype 9	Floodtype 10	Floodtype 11

Figure 33: Change in flood type occurence (number of days) for each elevation interval within the floodplain between the reference scenario and the high climate change scenario

4.7 Discussion: considerations on the relevance of flood-impact studies on ecology

Climate change may drastically affect flooding regimes. It is also clear from the case of the Zammelsbroek that a relatively mild change in the river flow regime may drastically affect the flooding regime. This is valid for all floodplains that have some type of embankment between the river and the floodplain. The overtopping frequency and/or overtopping duration will then be the crucial factor. For floodplains without such embankments there will be a much more direct coupling between changes in the river flow and the flood regime. It can be expected, that the higher the embankment, the more drastic the impact will be if overtopping frequency and/or duration increases. Even if it is possible to quantify the ecological impact of changes in the flooding regime for a specific scenario and a specific floodplain with actual vegetation, it is of limited use. Beside natural succession, there are several other environmental stressors that impact floodplain ecological communities (dessication, eutrophication, acidification and invasive species). Many of these stressors can be indirectly related to climate change, but these stressors have already caused drastic changes to biodiversity in the past (without climate change). The dominant mechanism for changes in vegetation may be not flood related. It is to some extent possible to give each species a ranking on their intrinsic vulnerability for various stressors, based on their physiology, specialist/generalist aspects, area needs for viable populations, survival mechanisms, mobility of the species… A second aspect is that defining the impact of changes in flood regimes on current vegetation within 50 years is only illustrative. We can assume that climate change will occur rather gradual and that the changes in flooding regimes may also manifest gradually.

Nevertheless, there are ways to determine the intrinsic suitability/vulnerability of floodplains and riparian zones with respect to ecological values. Environmental heterogeneity indicators are repeatedly put forward as fundamental criteria for biodiversity conservation planning in the light of climate change. These areas have high potential for biodiversity, because there are many different abiotic and hydrological conditions present at small scales. In addition, the presence of many gradients on relatively small areas allows species to adapt, migrate and colonize within the boundaries of these areas along with the shifts in abiotic conditions (abiotic conditions can change/shift spatially along the gradient. Sites with more uniform conditions have higher risks for biodiversity losses, since many species have low mobility. Hence, heterogeneity on different scales is a proxy for biodiversity on different scales (species, vegetation types, habitats, ecosystems). The ecological vulnerability of the floodplain will be determined by the size of the flooplain and the presence of topographical gradients (Figure 34). A universal way to test floodplain/riparian zones on their resilience against changes in flood regimes may be the use of hypsometric analysis. This method indicates which water levels are critical because of high gradients. A linear hypsometric curve indicates that regardless of flood depth, equal areas of flood depth zonation remain present. If topographical gradients are present in a sufficient wide range, it is likely that the different flood types will also occur in any hypothetical future scenario. In the case of constructed controlled floodplains this will be seldom the case due to a limited topographical gradient (flat, with steep edges).

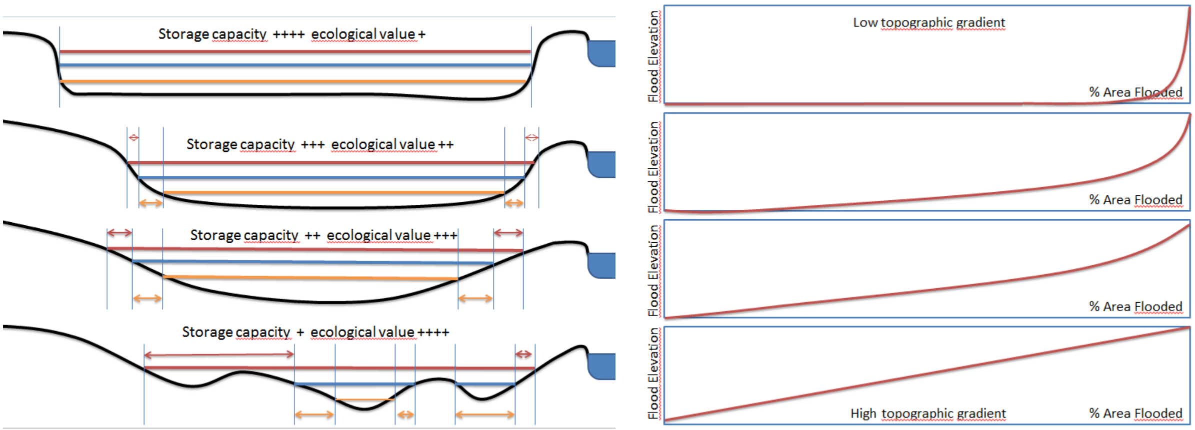


Figure 34: importance of topographical gradients for ecological resilience against CC

A hypsometric analysis of the Zammelsbroek floodplain shows that there is a very gradual increase of the flooded area for increasing water levels (Figure 35 – top). It is almost linear up to flood levels of 2 meter (for the lowest site of the entire floodplain). A hypsometric analysis of the nature reserve shows that the reserve selection is representative for the lager floodplain and that the same linear increase in flooded area can be observed (Figure 35 – bottom).

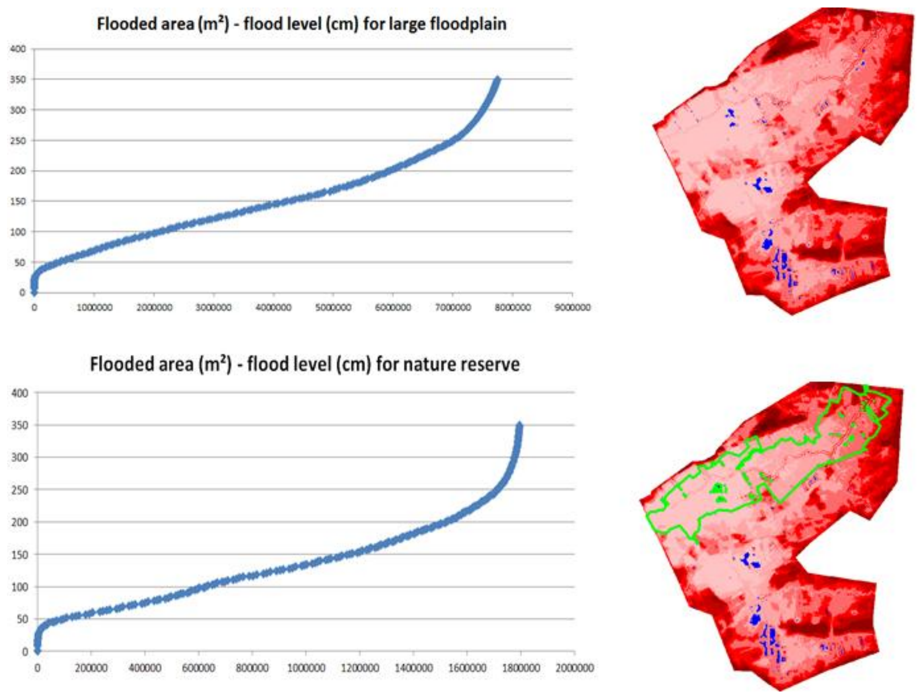


Figure 35: Hypsometric analysis of the Zammelsbroek (top) floodplain and the nature reserve area (bottom)

Chapter 5: Recent progress and insights in hydrological, river hydraulic and water quality modeling

In this chapter, the challenges regarding floodplain and river modeling are described and illustrated based on the Grote Nete case application. Most hydrological models are orientated towards flood prediction application. They make use of techniques (synthetic rainfall events, composite storms, conceptual models) to allow fast calculation of many scenarios. This is at the expense of the capability to include soft measures such as infiltration restoration, distributed water retention, land-use change… Furthermore the models are evaluated on their capability to accurately predict extreme events, while their performance on regular flows might be much lower. Long term simulations on the original models are seldom used, which makes it difficult to establish a reference condition to which changes can be compared.

Most state-of-the-art hydrological and river hydraulic models are oriented towards flood prediction. Typical applications are the assessment of flood risk along floodplains, flood damage evaluation for project planning, design of hydraulic structures for flood control, and flood-runoff forecasting in support of real-time operations. The flood modeling involves different steps, as summarized in Figure 36. Hydrologic analysis is performed first in order to get catchment runoff discharges, which are used as input to the hydraulic model for the river and surrounding floodplains. The water levels resulting from the hydraulic model are used to obtain flood maps. For this last step, a Geographic Information System (GIS) interface is employed.

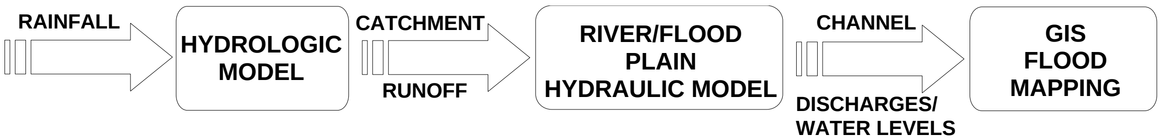


Figure 36. Schematic representation of river flood modeling.

5.1 Hydrological modeling

For the hydrological modeling, different modeling approaches exist. They vary from lumped to fully distributed models, and from deterministic to stochastic models. They describe the most dominant catchment rainfall-runoff components, including direct and indirect runoff, and the most dominant physical processes such as soil infiltration, groundwater recharge, soil moisture storage, surface and subsurface flow, evapotranspiration, etc. Usually they can simulate at various times steps (hourly, daily, monthly or seasonal).

In lumped hydrologic models, no spatial variability in the catchment runoff and processes is considered. Therefore, the response is evaluated only at the basin outlet or the location of the flow gauging stations. The response can be separated into slow flow (base flow; groundwater recharge) and quick flow (surface and subsurface) components. Typically, the lumped rainfall-runoff models lump, in a broad sense, the highly complex soil processes and properties into a few processes and parameter values. The model parameters do represent the physical features of hydrologic processes, but in a spatially aggregated (lumped) way and involve some degree of empiricism. Typically the only data input in these models is precipitation, evapotranspiration and the basin area. River flow data near the outlet of the catchment is also needed for calibration. Additionally air temperature and radiation is required if snowmelt is modeled. Geographical data such as topography, vegetation and soil type are not considered. The main advantages of lumped models are their simple structure, the minimum requirements of data, the fast setup and calibration of model parameters. They have a good performance compared with more complex models if the main interest is the calculation of river discharge.

Contrary to the lumped rainfall-runoff models, the spatially distributed hydrological models consider that the parameters fully vary in space at a resolution usually selected by the users. Compared with lumped models, distributed codes require a large amount of data for the parameterization in each grid cell increasing computational requirements. Because the physical processes are modeled in detail, they can provide the highest degree of accuracy if properly applied. In practice, however, data availability is limited, such that it might be difficult to identify all parameter values from the data. The number of parameters then might be too large for an accurate calibration on the basis of the limited amount of data. No unique set of 'optimal' parameter values exists in those cases; different sets of model parameters with equally good fit to the observed model output exist. The model is then called "overparameterized".

In the Grote Nete case, both lumped conceptual rainfall-runoff models (NAM, VHM) and a spatially distributed hydrological model (MIKE-SHE) have been implemented in the CCI-HYDR project. The NAM model has the general model-structure shown in Figure 37.

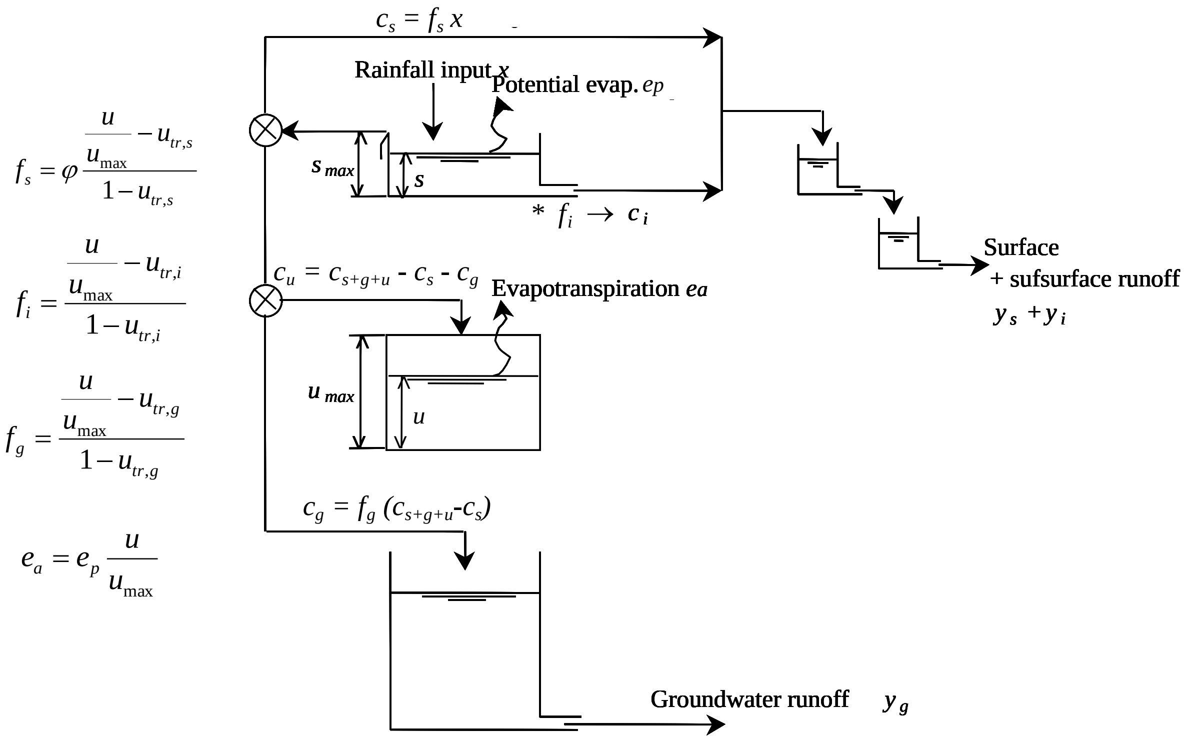


Figure 37. Model-structure of the NAM lumped conceptual rainfall-runoff model.

Rainfall input first fills up a surface storage reservoir (s). When this reservoir becomes full (s= s_{max}), overflow occurs. The surface storage reservoir is emptied by evapotranspiration (e_a), which depends on the potential evapotranspiration (e_p). When the soil is saturated ($u=u_{max}$), the actual evapotranspiration equals the potential evapotranspiration ($e_a=e_p$). When the soil is not fully saturated ($u<u_{max}$), the actual evapotranspiration is a fraction of the potential evapotranspiration (linearly depending on the relative soil water content: $e_a=e_p*u/u_{max}$). Part of the overflow volume contributes to quick or surface runoff. The rainfall fraction of this part is given by f_s, which linearly depends on the relative soil water content u/u_{max}, taking into account that no surface runoff will occur when the relative soil water content has a value lower than a threshold value $u_{tr,s}$. The remaining rainfall fraction ($1-f_s$) leads to soil infiltration, of which part (fraction f_g) percolates into the groundwater. The fraction f_g linearly depends on the relative soil water content.

When the relative soil water content has a value lower than a threshold value $u_{tr,g}$, no percolation occurs. Based on these equations, the total rainfall amount (x) per time step is split into rainfall contributions to surface runoff (c_s) and infiltration (c_{s+g+u}). Part of the infiltration water contributes to the groundwater (c_g). The rest fraction contributes to the soil water storage ($c_u = c_{s+g+u} - c_s - c_g$). Interflow (or subsurface runoff, or drain flow) is simulated as the outflow of the surface runoff reservoir. This outflow is modeled using the linear reservoir equation. The rainfall contributions to surface runoff and interflow is routed by means of two linear reservoirs in series. The recession constants of these reservoirs affects the response time of surface runoff to rainfall. These recession constants combined with the recession constant of the surface runoff reservoir determine the response time of interflow to rainfall. Also the outflow of the groundwater reservoir is modeled by means of a linear reservoir, where the groundwater reservoir recession constant determines the response time of groundwater to rainfall.

The use of NAM has the disadvantage that the model structure is fixed and cannot be modified by the user. Given that there are many assumptions underlying the NAM model structure, model performance may be insufficient in case these assumptions would not be fully valid. For that reason, the Hydraulics Division of K.U.Leuven developed a more generalized lumped conceptual modeling procedure, called VHM ("Veralgemeend conceptueel Hydrologisch Model"). The procedure starts from a generalized model structure framework (Figure 38) that is adjusted in a case-specific parsimonious way. The procedure makes sure that the derived model structure is 'identifiable' from the data, and avoids overparameterization.

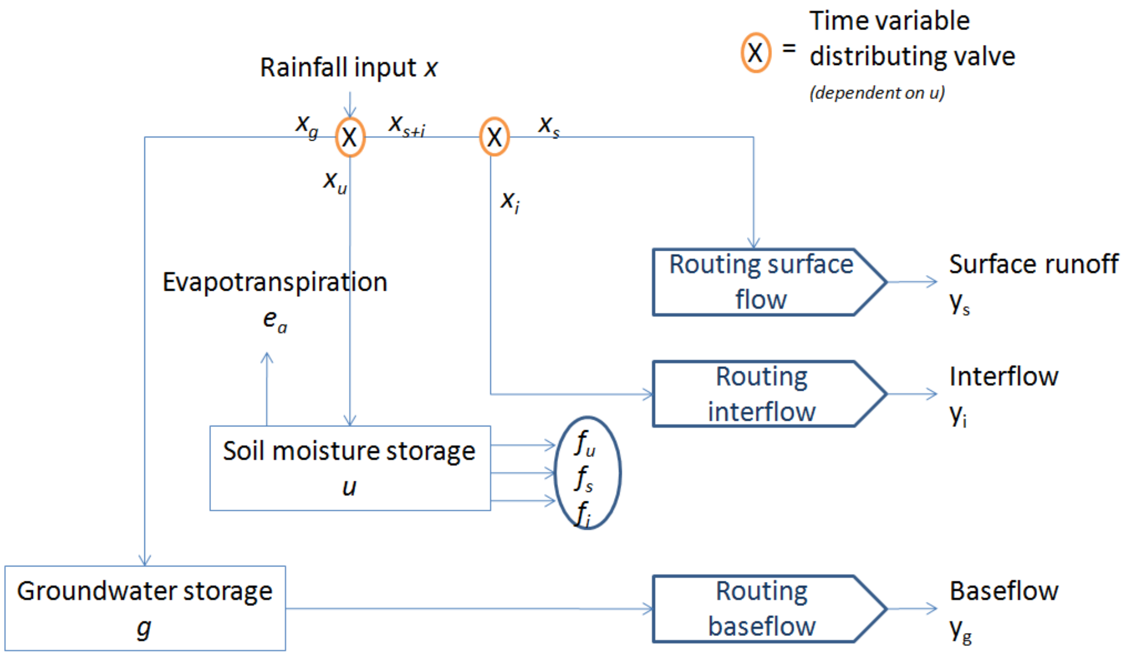


Figure 38. Generalized lumped conceptual rainfall-runoff model structure, considered by the VHM approach.

The VHM model-structure building is done in a transparent, step-wise way, where separate parts of the model structure are identified and calibrated based on multiple and non-commensurable information derived from river flow series by means of a number of sequential time series processing tasks. These include separation of the high frequency (e.g., hourly, daily) river flow series in subflows, split of the series in nearly independent quick and slow flow hydrograph periods, and the extraction of nearly independent peak and low flows. The subflow filtering is based on a digital numerical filter methodology (Willems 2009). This filter is used to split the total river flow series in its runoff components. More specifically, first a split is conducted between the quick flow and slow flow components based on a visual determination of the recession constant on a logarithmic plot (Figure 39).

In a next step, the quick flow component is further separated into the overland flow and interflow components by using this same method but then on the filtered time series. The result of these two steps combined is shown in Figure 40. The interflow component might physically represent the subsurface runoff in the catchment (if any); the slow flow component the baseflow or groundwater recharge. The subflow separation results enable the structure of the rainfall-runoff model components in VHM to be identified separately. They also enable the information base for calibration to be enlarged.

Next to the subflow filtering, the full time series is split in nearly independent quick and slow flow hydrograph periods, and nearly independent peak and low flows are extracted as the maxima during the quick flow periods and the minima during the slow flow periods. The model building and calibration accounts for the statistical assumptions and requirements on independency and homoscedasticity of the model residuals. Next to the separate identification of the subflow recession constants and related routing submodels, equations describing quick and slow runoff sub-responses and soil water storage are derived from the time series data. Model performance evaluation is based on peak and slow flow volumes as well as extreme high and low flow statistics.

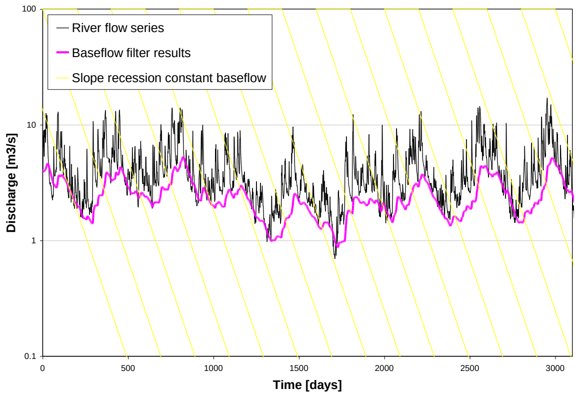


Figure 39. Baseflow filter results for the observed series at Varendonk station.

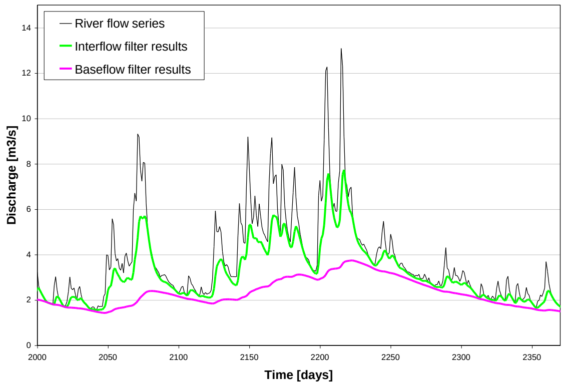


Figure 40. Baseflow and interflow filter results for the observed series at Varendonk station.

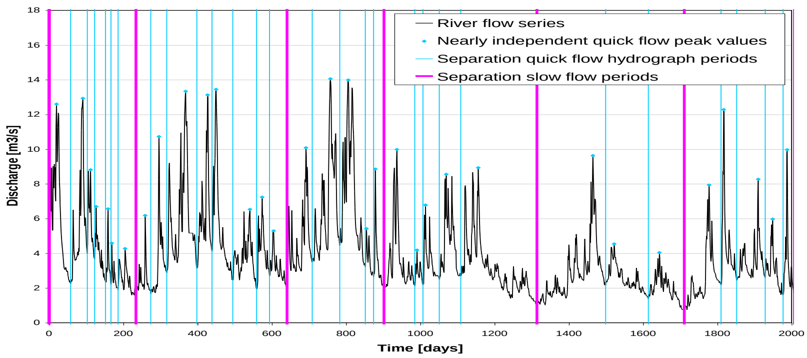


Figure 41 Split of the time series in nearly independent quick and slow flow periods based on the observed series at Varendonk station.

Along similar lines as for the calibration of NAM, a step-wise calibration approach is followed by the VHM model, making use of the results of the subflow filtering and the statistical time series analysis of the river flow series. The calibration steps are as follows:

- the recession constants of the quick flow (or surface runoff), interflow and groundwater are derived from the flow series and directly used as model parameters (of the linear reservoir models);
- the soil water storage volumes per slow flow event are optimized after comparison with the soil water storage volumes estimated from the time series data (by closing the water balance). The soil water storage volumes are estimated as rest fractions after subtracting the

total runoff observations from the rainfall input surface capacity and after assuming a model for computing the actual evapotranspiration from the potential evapotranspiration;

- the surface runoff, interflow and groundwater volumes per event (quick flow events for the surface runoff and interflow volumes, slow flow events for groundwater volumes) are optimised based on the respective subflow filter results.

The NAM and VHM lumped conceptual rainfall-runoff models have been constructed/simulated at the hourly time scale, but the model validation has been carried out over the full range of relevant time scales (the concentration times along the river network). The model validation was done both in real time based on continuous flow series and by statistically testing the agreement with the observed series. The statistical testing was done for the peak flows (flow maxima during the quick flow events), the low flows (flow minima during the slow flow events), the cumulative volumes per event for total flow as well as for the subflows, and in terms of frequency distributions and discharge/duration/frequency (QDF) relationships, following the method of Willems (2009). The maximum and minimum flow values were extracted from the simulated flow series during the quick and slow flow periods, and compared with the corresponding empirical values derived from the statistical time series analysis. Next to the lumped conceptual models, a detailed physically-based and fully distributed hydrological model has been implemented. The MIKE-SHE model considers the hydrological processes at a grid size that can be selected by the user and integrates the entire land phase of the hydrological cycle. It consists of separate modules for snow smelt, evaporation, overland flow, the unsaturated zone, the saturated zone and channel flow. In terms of process equations, the model is based on the kinematic wave equation for overland flow, the Richard’s equation for the unsaturated zone and applies 3 dimensional finite differences method to Darcy’s law. Model verification is done along similar lines as the lumped conceptual models, but additional verification data on groundwater levels (at groundwater wells spread over the basin) can be considered. The hydrological models were calibrated based on river flow time series available at the four river flow gauging stations (Varendonk/Geel-Zammel, Vorst, Meerhout and Tessenderlo), and for the MIKE-SHE model also based on groundwater well data at several pumping wells distributed over the basin (Figure 43).

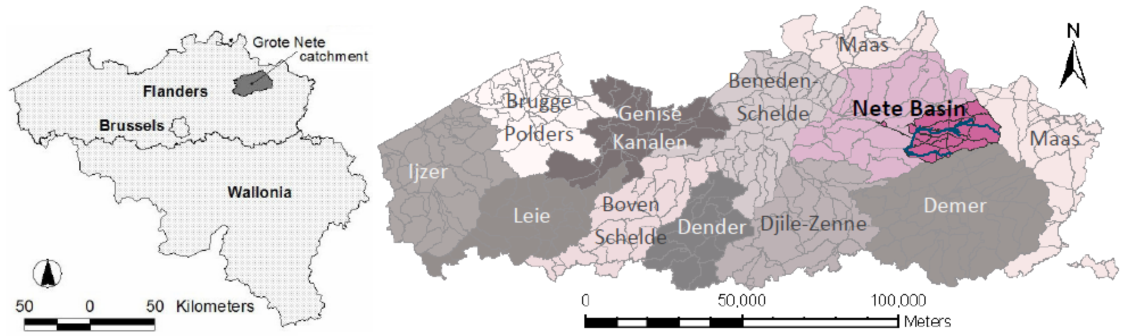


Figure 1. Location of the basin of the Grote Nete and Grote Laak.

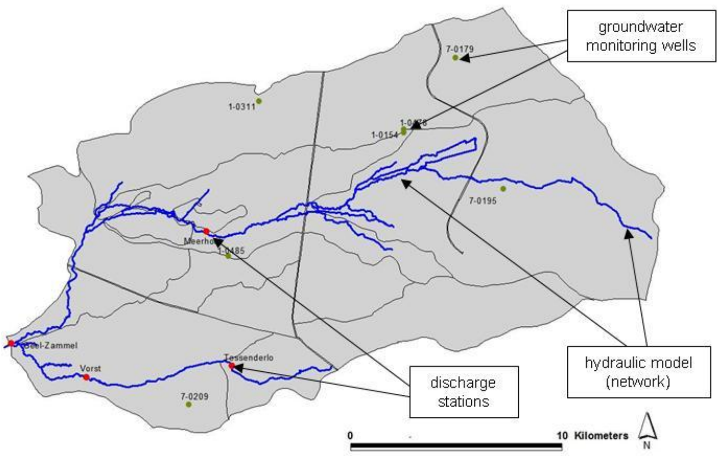


Figure 2. River flow gauging stations location and partial drainage areas for Molse, Grote Nete and Grote Laak sub-catchments.

Model parameters were derived in NAM and VHM in a lumped (spatially averaged) way upstream of the most upstream flow gauging stations (Meerhout and Tessenderlo), or in between two stations for the more downstream stations (Varendonk and Vorst). The calibration thus was done from up- to downstream. The number of calibrated lumped conceptual models consequently is equal to the number of flow gauging stations in the basin. For the MIKE-SHE model, a grid size of 500 m was considered. Rainfall input is based on the available hourly rainfall series at several locations in and outside the basin. They were provided by the Hydrological Information Service (HIC) and the Flemish Environment Agency (VMM). The point rainfall data are converted to areal rainfall estimates per subbasin based on the Thiessen polygons method. Figure 44 shows the VHM model results of total rainfall-runoff discharges at Varendonk station. Similar accuracy was obtained for the total rainfall-runoff discharges produced by the MIKE-SHE model. The additional spatially distributed results on groundwater levels also had good performance, as is shown in the example of Figure 45. For all groundwater wells, despite some systematic over- or underestimations for some periods, groundwater level fluctuations show that the dynamic processes are represented well by the model. Separate investigation of the accuracy of the peak and low flows is shown in Figure 46. These results show that based on the VHM model results, accurate extrapolations can be made to rainfall-runoff conditions for high return periods, which is important for the flood hazard calculations in this project.

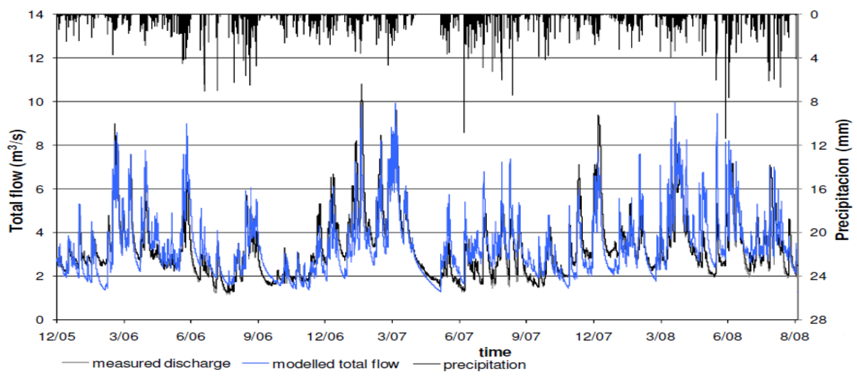


Figure 44. Comparison of hourly observed river flow series at Varendonk station with the VHM model results.

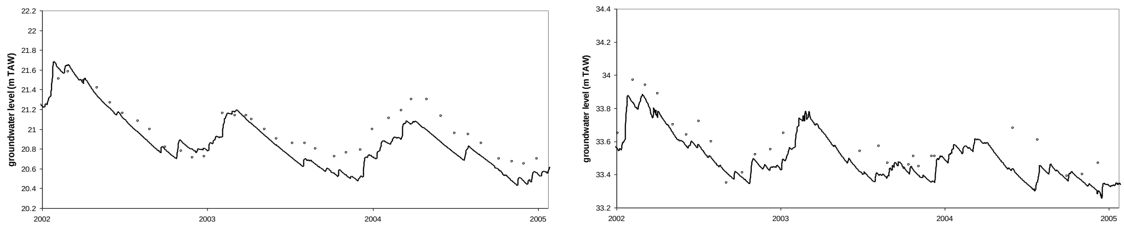


Figure 45. Comparison of observed groundwater levels at wells 209 (left) and 154 (right) with the MIKE-SHE model results.

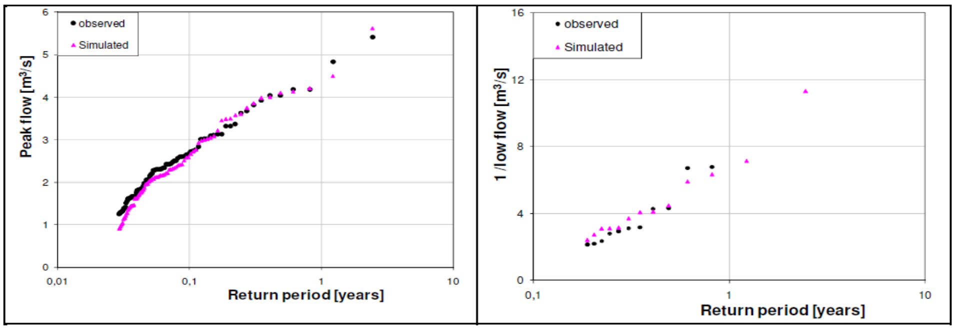


Figure 46. Comparison of observed and simulated (VHM model) peak and low flows vs. return period.

5.2 River hydraulic modeling

For the Grote Nete case, a full hydrodynamic model has been developed in the CCI-HYDR project for the Grote Nete, the Grote Laak and the main tributaries. This was done on the basis of the river modeling package MIKE11 (DHI Water & Environment), in combination with ArcGIS 9. MIKE11 is a modeling system for one-dimensional (1D) full hydrodynamic river modeling. Applying the rainfall-runoff model outputs as input to the river model, estimation can be made of river flow discharges, velocities and water levels at any point along the river network and at a continuous time scale. Available cross-section data were implemented to define the geometry of the river branches. River bed roughness coefficients were estimated and implemented along each section. They were validated based on simultaneous measurements of water levels and discharges at the flow gauging stations. Most hydraulic structures, such as bridges, weirs, culverts and hydraulic regulation structures were also implemented (Figure 47).

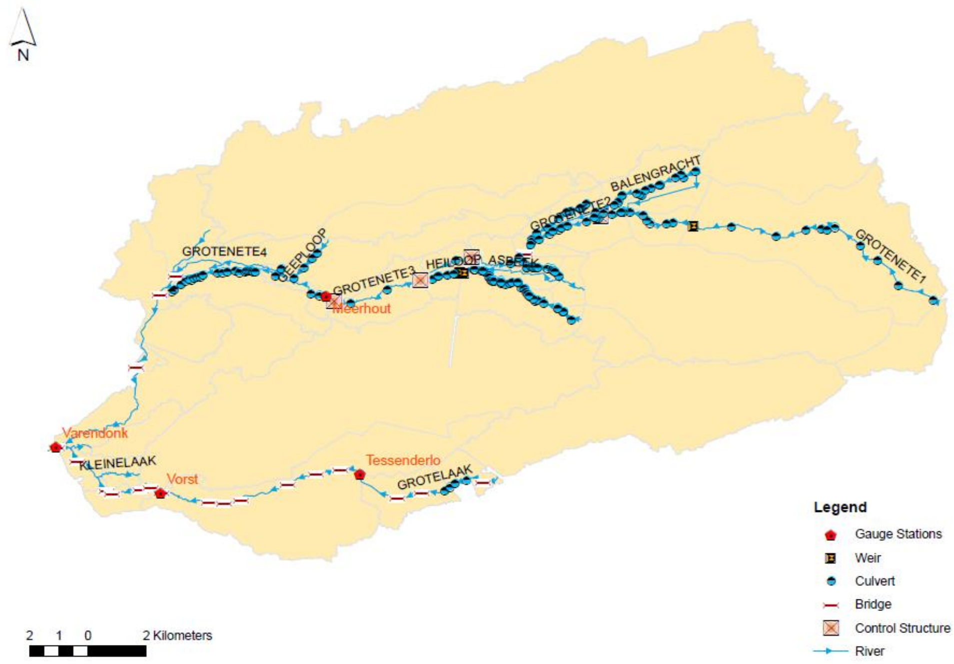


Figure 47. Overlay of the river hydraulic model on the Grote Nete river network.

The model has been upgraded in this SUDEM-CLI cluster project with the models of the upper reaches in the catchment. This increased model complexity, but also improved the model accuracy. This should be interpreted not only in terms of better predicting floods, but also in better representation of the physical processes. The model upgrade was done based on the field data obtained from the Province of Antwerp (Dienst Waterbeleid). The following rivers were additionally included in the model: Molse Nete, Kleine Laak, Hoofdgracht, Heiloop and Asdonkbeek. Data for the non-navigable rivers of 1st category were obtained from the Flemish Environment Agency (VMM), while for the 2nd category non-navigable rivers data were obtained from the Province of Antwerp (Dienst Waterbeleid) (Figure 48).

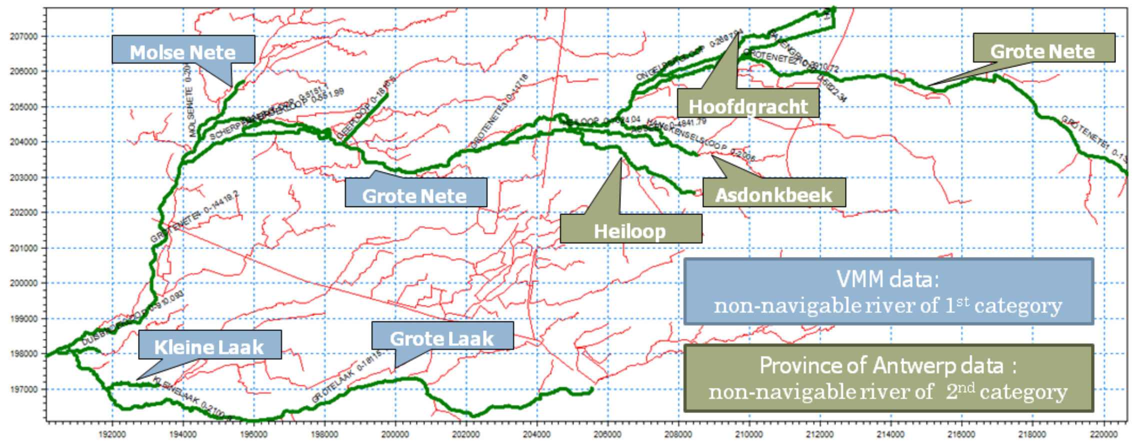


Figure 48 Modeled non-navigable rivers of 1st and 2nd category.

The VHM lumped conceptual rainfall-runoff results were used as input in the full hydrodynamic model. Two types of hydrodynamic simulations are conducted: simulation of historical flood events (all events in a continuous long-term simulation for the full period with available input data since 1986, with hourly time step), and simulation of synthetic events for given return periods (composite hydrograph method; return periods between 1 and 100 years).

The hydraulic model results were validated based on water level data for the historical flood events at the four flow gauging stations. Figure 49 shows example results of this validation for the internal Tessenderlo station.

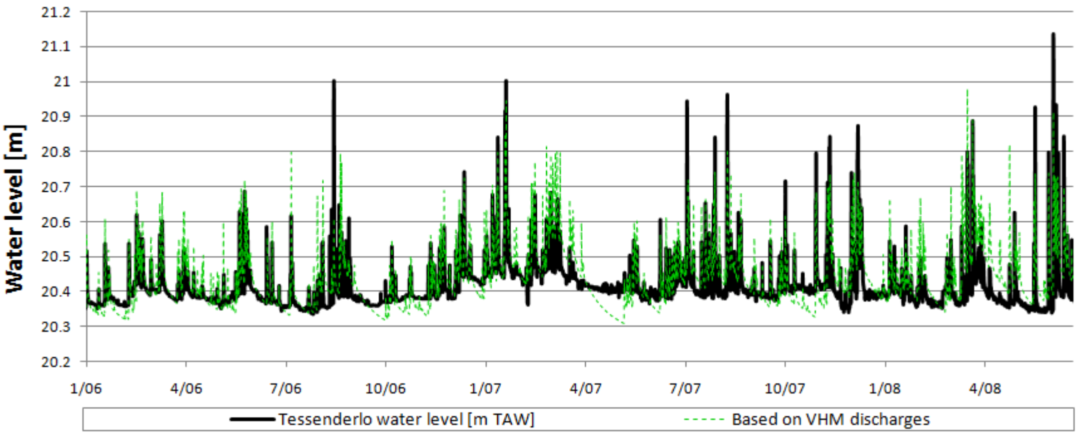


Figure 49. Comparison of hourly observed water level series at Tessenderlo station with the MIKE11 model results after simulation of VHM rainfall-runoff input.

5.3 River flood modeling

The floodplains in the basin were schematized by a 1D river hydraulic model based on a quasi-2D approach. In this approach the floodplains are modeled as flood cells or as a network of fictitious river branches and spills with the rivers (see Figure 50). The river branches represent the topographical depressions (the floodplains, most often with drainage canals at the bed), while the spills represent topographical elevations (e.g. roads, railways) in between these depressions. Also the dikes or the river embankments in between the river and the floodplains are represented by spills. By taking the cross-sections of the fictitious river-branches equal to those of the floodplains (cross-sections perpendicular to the axis of the river; see Figure 50), the storage of water in the floodplains (water levels, storage volumes and spatial extent) can be described in an accurate way. Also the discharges and the water velocities can be modeled for the floodplains, after appropriate values are chosen for the friction coefficients in the fictitious river-branches.

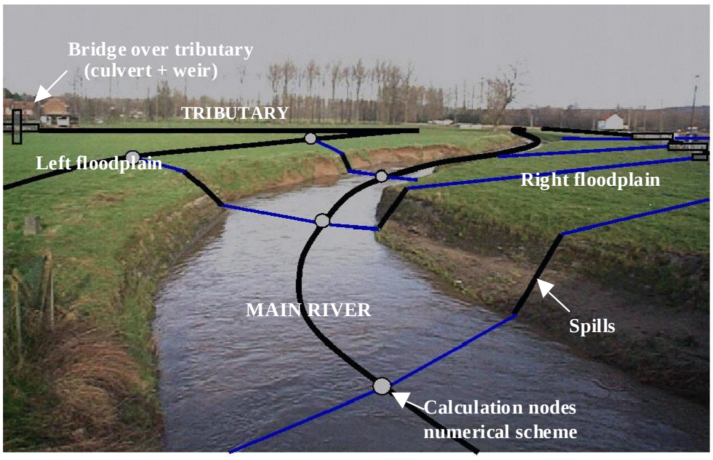


Figure 50. Schematic representation of the quasi-2D approach for river flood modeling (Willems, Vaes et al. 2002).

Based on the high resolution digital elevation model (DEM), MIKE-GIS was used to draw geometrical data for the river branches and spills/overflows, and to determine the preferential flow direction along the floodplain.

For the spills between the main river and the floodplains, the spill levels were derived from the DEM, but also more accurately from the cross-section survey. It is clear that the accuracy of the dike crest levels or embankment levels is very important for the flood model. It determines the return period of flooding and other flood related results.

The GIS system was also used to demarcate the potential flooded areas (areas to be modeled) and to visualize the spatial extent of the modeled floods. Flood maps were created which provide results on the depth, spatial extent and duration of the simulated inundations. The GIS system thus was applied both as a pre-processing tool and a post-processing tool. Figure 51 gives an overview of the 57 different floodplains demarcated and implemented.

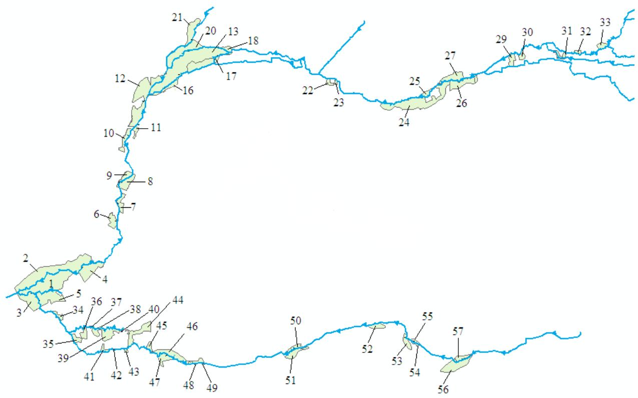


Figure 51. Overview of the floodplains modeled.

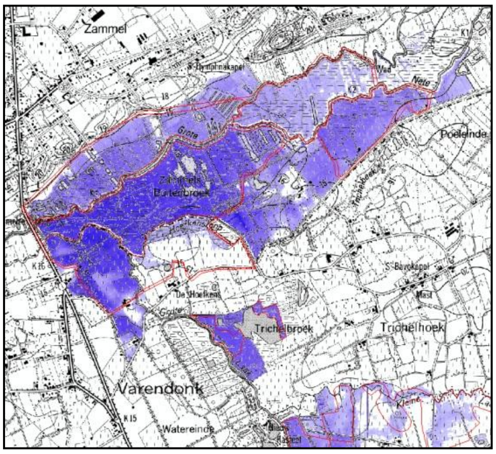


Figure 52. Overlay of model based flood maps for the historical flood of September 1998 (blue areas), with the ROG historical flood map (red polylines) for the Zammelsbroek area (floodplains no. 1-5)

Based on the quasi-2D approach, models were developed for all major floodplains along the hydrodynamically modeled rivers. The floodplain models were validated based on the historical flood map (ROG "Recent Overstroomd Gebied"; representing the maximum spatial extent of the inundated areas) of the largest recent historical flood of 1998. Figure 51 shows the model results for this flood of 1998 and the comparison with the ROG map for selected floodplains along the Grote Nete basin. The relatively good model performance showed that the quasi-2D approach is applicable to simulate the complex overland flow processes along the shallow floodplains.

5.4 River flood hazard maps

Based on the long-term time series of rainfall-runoff discharges (simulated by the hydrological model or based on the river flow data), rainfall-runoff discharge statistics, QDF-relationships and composite hydrographs have been derived. For each flow gauging station, statistical extreme value distributions were computed based on the (nearly independent) high flow extremes for each station (e.g. for Varendonk station based on the results shown in Figure 46). The distributions are hereafter called flood frequency distributions. They were calibrated to the empirical frequency distributions based on the extreme value theory (Willems 1998; Willems, Guillou et al. 2007) and describe the relationship between the magnitude of river peak flows and the recurrence interval or return period. They are typically used for the design of water engineering or hydraulic structures and bridges and for flood risk estimation. Apart from these water management applications, the flood frequency distributions can also be used to evaluate the hydrodynamic model performance against simulation of extreme flow conditions, and as basis for the construction of synthetic hydrographs (for given return periods).

The extreme value analysis was repeated for different aggregation levels (time scales over which the flow series were averaged).

The range of aggregation-levels is based on the concentration times along the river network considered from up- to downstream in the basin. Results of the extreme value analysis were summarized in the form of 'discharge/duration/frequency (QDF) relationships' and, more advanced, in the form of 'composite hydrographs'. These composite hydrographs are synthetic hydrographs constructed such that the average discharge equals a specific return period for all durations that are considered centrally in the hydrograph. Figure 53 shows an example (for Varendonk station) of composite hydrographs for return periods of 1, 5, 10, 50 and 100 years.

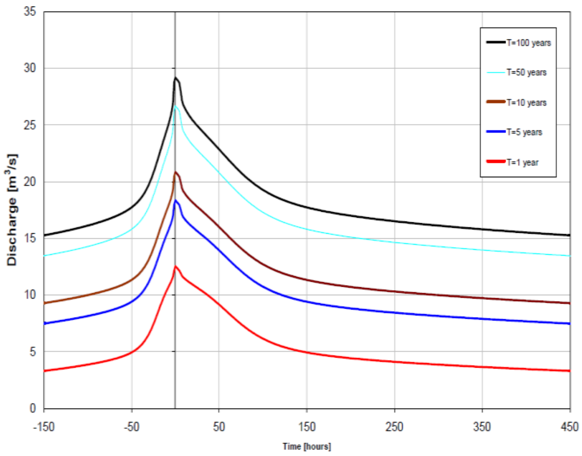


Figure 53. Composite hydrographs for 1, 5, 10, 50 and 100 years for the Grote Nete River at Varendonk station.

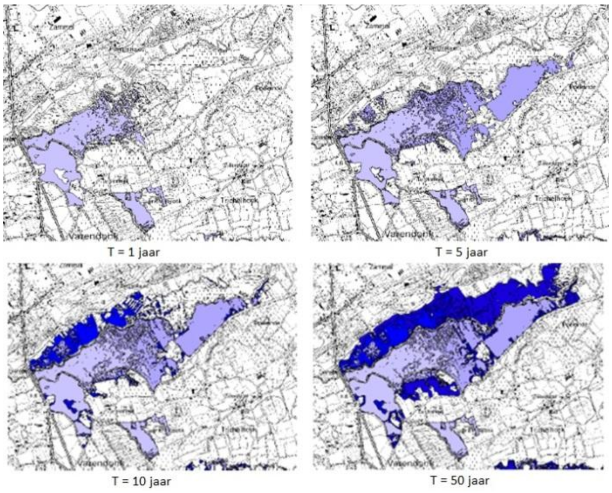
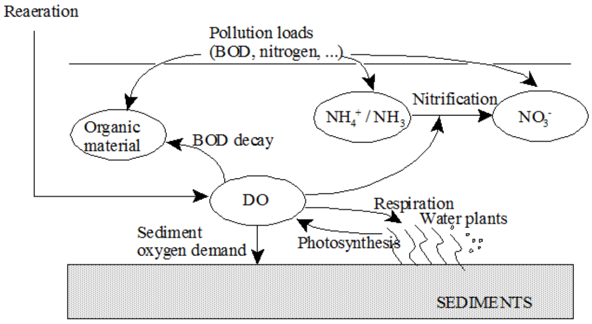


Figure 54. Flood hazard maps for Zammelsbroek area for return periods of 1, 5, 10 and 50 years.

The composite hydrographs were derived for each subbasin based on the observed river flow series at the flow gauging stations. They also could have been based on the NAM or VHM model results, but results should be close. The composite hydrographs have the important feature that river states with the same safety level at all locations along the river can be derived by one single short-term simulation. This is for instance shown by (Vaes, Willems et al. 2000). The composite hydrographs have been simulated in the river and floodplain hydraulic models to derive flood hazard maps (for the various return periods: 1, 5, 10, 50 and 100 years) along the modeled floodplains (Figure 54).

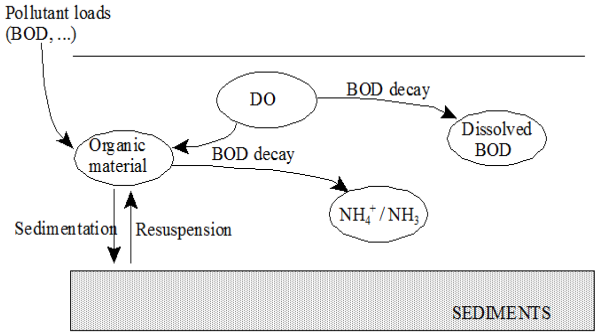
5.5 River (physico-chemical) water quality modeling

After implementation of the hydrological and hydraulic models and the flood hazard model, the MIKE11 hydraulic model of the main rivers and floodplains was in this SUDEM-CLI cluster project extended with a physico-chemical water quality model after implementation of domestic, industrial and agricultural pollution sources. The pollution sources were transferred to river water quality concentrations after consideration of advection, dispersion and the most important transformation processes for water quality variables describing components of the oxygen and organic pollution cycle (DO, BOD, temperature T) and the nitrogen cycle (total nitrogen, N_t, ammonia, NH_4^+-N, and Nitrates, assumed to be the sum of NO_2^--N and NO_3^--N). The input of nutrients on the floodplains during floods is crucial in determining the vulnerability/compatibility of habitats to floods. Figures 55, 56 and 57 give an overview of the different physico-chemical water quality processes considered in the model, together with the transformation process equations. These processes include: resuspension and sedimentation in the calculation of the BOD balance; the ammonia / nitrate balances, plus the oxygen consumption from the sediment oxygen demand and the nitrification process; and denitrification.



$$\frac{dDO}{dt} = + K_{reareation} * (C_S - DO) \qquad \text{(re-aeration)}$$
$$- K_{d3} * BOD_d * \theta_{d3}^{(T-20)} \qquad \text{(dissolved BOD)}$$
$$- K_{S3} * BOS_S * \theta_{S3}^{(T-20)} \qquad \text{(suspended BOD)}$$
$$- K_{b3} * BOD_b * \theta_{b3}^{(T-20)} \qquad \text{(deposited BOD)}$$
$$- Y_{nitr} * K_{nitr} * NH_4 - N * \theta_{nitr}^{(T-20)} \qquad \text{(nitrification)}$$
$$- R_s * \theta_{resp}^{(T-20)} \qquad \text{(respiration)}$$
$$+ P_h \qquad \text{(photosynthesis)}$$
$$- S_d \qquad \text{(sediment oxygen demand)}$$

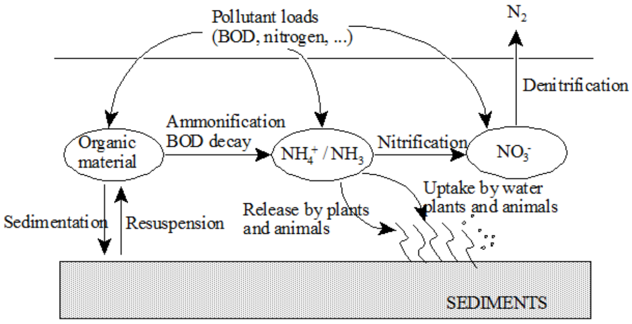
Figure 55. Overview of physico-chemical water quality processes considered in the model for the oxygen cycle.



$$\frac{dBOD}{dt} = - K_{d3} * BOD_d * \theta_{d3}^{(T-20)} \frac{DO}{M + DO}$$
$$- K_{S3} * BOS_S * \theta_{S3}^{(T-20)} \frac{DO}{M + DO}$$
$$- K_{b3} * BOD_b * \theta_{b3}^{(T-20)} \frac{DO}{M + DO}$$

Resp. Dissolved, suspended and deposited BOD

Figure 56. Overview of physico-chemical water quality processes considered in the model for the organic pollution cycle.



$$\frac{dNH_4 - N}{dt} = + Y_b * K_{b3} * BOD_b * \theta_{b3}^{(T-20)}$$
$$+ Y_d * K_{d3} * BOD_d * \theta_{d3}^{(T-20)}$$
$$+ Y_s * K_{s3} * BOD_s * \theta_{s3}^{(T-20)}$$
$$- K_{nitr} * NH_4 - N * \theta_{nitr}^{(T-20)}$$
$$- y_{resp} * (P_h - R_s)$$

Resp. BOD decay (line 1-3), nitrification (line 4), respirtation and photosysnthesis (line 5)

Figure 57. Overview of physico-chemical water quality processes considered in the model for the nitrogen cycle

The water quality model was implemented and calibrated based on water quality input data received from VMM. Figure 59 gives a summary of the pollutant sources introduced in the water quality model of the Grote Nete basin. Some pollution sources are punctual/point, while others are diffuse/distributed. Figure 58 shows the locations of the punctual sources. As can be seen in this figure, several punctiual pollution sources enter river branches that are outside the network of modeled branches. These were implemented at the most upstream location of the modeled network, but reduced in concentration to account for the attenuation due to the travel time of the pollutants from its source until the closer simulated river branch. The reduction factor was assumed proportional to the estimated distance of travel. In the case of the pollution sources in the Molse Nete subcatchment, some sources are so far from the start of the simulated river (max. 23km) that they were not taken into account.

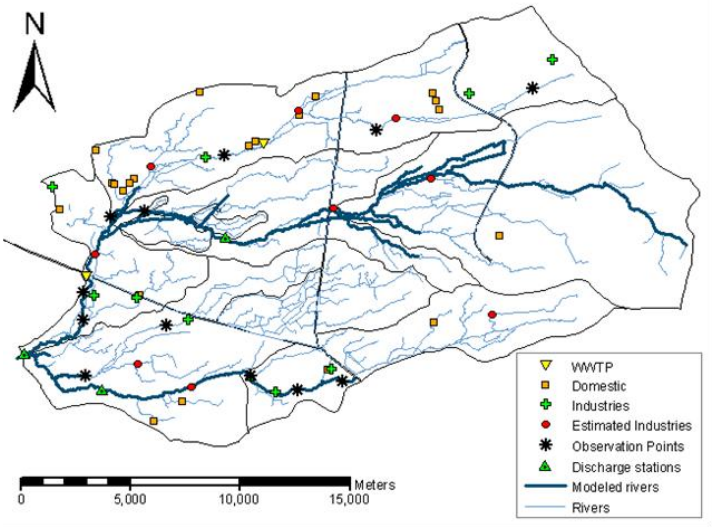


Figure 58. Locations of the punctual pollutant sources introduced in the water quality model of the Grote Nete basin.

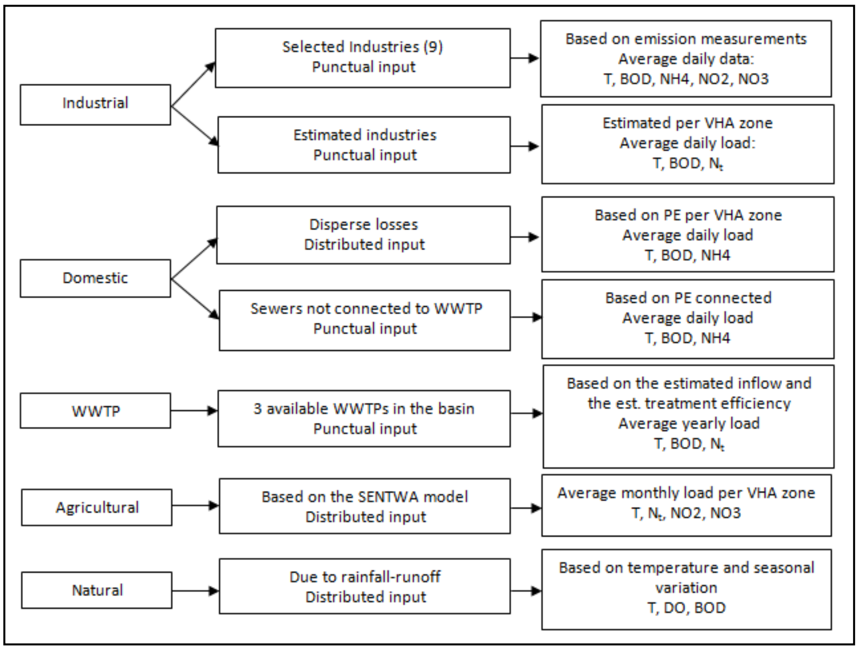


Figure 59. Summary of the pollutant sources introduced in the water quality model of the Grote Nete basin.

Industrial pollution sources

Two types of industrial pollution sources were identified. The first type comprises 9 industries with direct emission measurements available. The average daily load provided was based on approx. one emission measurement per year. For these industries, emission data were provided for T, BOD, NH4-N and Nitrates (NO2-N, NO3-N). The second type of remaining industrial pollution sources were estimated based on tax and municipality files. Daily average pollutant loads were estimated based on these files for T, BOD and total nitrogen.

Domestic pollution sources

Domestic pollution was calculated based on the estimated number of population equivalents (PE) and the domestic loads estimated per PE for T, BOD, NH4-N. They were subdivided in punctual and disperse losses. The punctual losses are due to sewer losses not connected to WWTPs. Their location and description is presented in Figure 58. The disperse losses were calculated based on the number of PE per hydrographic subcatchment (VHA zone), and spatially distributed in a uniform way based on the catchment areas.

Waste Water Treatment Plant pollution sources

There are 7 WWTPs treating water from the Grote Nete basin, but only 2 of them discharge their effluents inside the basin: Mol and Geel. The Mol WWTP discharges to the upper Molse Nete. Given that this branch was not modeled, its load was applied to the start of the modeled Molse Nete branch after applying a reduction factor due to attenuation of the pollutants. The Geel WWTP is very close to the lower Grote Nete. In addition, the Leopoldsburg-Legerkamp reed bed was considered, which is located upstream of the modeled part of the Grote Laak. Its pollution loads were also estimated from its served PE. The WWTP loads were considered for T, BOD and Nt based on yearly average estimated loads that take into account the estimated inflow and the estimated treatment efficiency of the WWTP. The pollution originating from urban stormwater runoff (e.g. combined sewer overflows) have not been taken into account. It is clear that this might cause significant underestimations in the domestic pollution loads during high stormwater periods. In cooperation with Aquafin, we are investigating how this can be improved.

Agricultural pollution sources

Agricultural pollution loads have been estimated based on the SENTWA model (System for the Evaluation of the Nutrient Transport to surface WAter). This model is being used by the VMM (Van Hoof, 2003) and enables calculation of the nutrient emissions from agriculture (manuring) to the surface waters. It is a semi-empirical model that quantifies orders of magnitudes of the nutrient emissions. It quantifies the load of total nitrogen (Nt) and total phosphorous on an annual and also monthly basis per river catchment (VHA zone). Seasonal variation due to rainfall is also taken into account. The model consists of seven routes of emission: atmospheric losses, direct losses, drainage losses, ground water losses, excess losses, erosion losses and runoff losses. The model demands input of agricultural land use, number of different kinds of animals (cattle), data on excretion coefficients for the different kinds of animals, use of fertilizers, data on transport of manure, precipitation quantities, and yields of different crops.

The Nt load, calculated by the SENTWA model in kg/month, was distributed proportional to the different types of discharge. This was done based on the subflow filter results (see hydrological model). The direct losses were assumed proportional to the total discharge, the drainage losses to the filtered interflow, the groundwater losses to the filtered baseflow and finally the erosion and runoff losses to the filtered overland flow.

Once the Nt concentration was determined per subbasin, it was split between ammonia and nitrates. This was done based on monthly factors obtained in previous studies in Belgium (Willems, Timmerman et al. 2005). The agricultural pollution was assigned in a distributed way over the basin.

Natural pollution sources

The rainfall-runoff is mixed with different pollutants in its path way to reach the rivers. The runoff water was assumed to have saturated DO concentration since it is mainly influenced by the natural reaeration process (Willems 2000). The DO concentration could be calculated with the following experimental equation (APHA 1985):

$$Cs=14.652+T[-0.41022+T(0.007991-0.000077774T)]$$

The organic pollution introduced by the rainfall runoff in the rivers was expressed by means of the BOD concentration. The organic matter might be attached to the transported sediment or due to falling leaves that follow a seasonal cycle. At the beginning of winter, BOD increases and at the start of summer it reduces. This cycle was described in a simplified way with a sinusoidal wave equation.

The seasonal variation of the BOD was empirically calibrated from BOD5 immision data for the Dender basin (Radwan, Willems et al. 2003). The same equation was used for the Grote Nete basin:

for years when February has 28 days: $BOD_{agr} = 4 + 3\cos\left(2\pi \frac{hour}{8760}\right)$

for years when February has 29 days: $BOD_{agr} = 4 + 3\cos\left(2\pi \frac{hour}{8784}\right)$

*where hour is the hour in any given year starting from 1st January.

Given that most of the physico-chemical processes in the environment are temperature dependent, temperature therefore is a key input to the water quality model. Daily temperature series for the studied period were selected from Herentals station (T measured at 5cm below the surface), where the missing summer period of 2006 was filled with data from Zonhoven (T measured 5cm above the surface) (Hydronet 2010). These temperature series show very similar behaviour to the measured water temperatures available for the observation points. The input temperature was assumed constant all over the basin. Due to the very detailed hydraulic model, the time step of the water quality model had to be taken very small (originally 10s), such that the simulation of 1 year took approximately 1 week. In an attempt to further increase the time step, the hydraulic model was modified by deleting or moving some conflictive cross sections. In this way the accuracy of the model reduced, but no important changes were observed in the results of the model. Selected water quality model results obtained are shown in Figure 60, Figure 61 and Figure 62. These results were obtained without calibration of the water quality process parameters; default values were used for these parameters. Model results match the observed concentrations and temperature values in order of magnitude. More detailed verification was, however, not feasible due to the limited temporal frequency of the available water quality measurements.

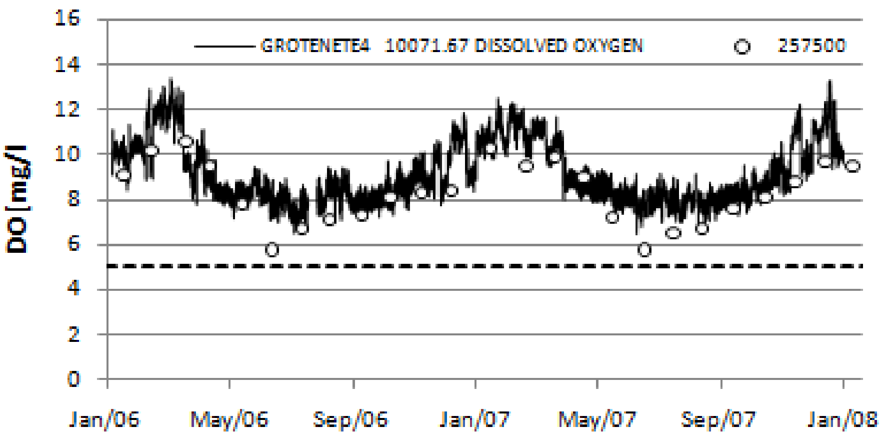


Figure 60. Comparison of observed and modeled DO concentrations at Varendonk station.

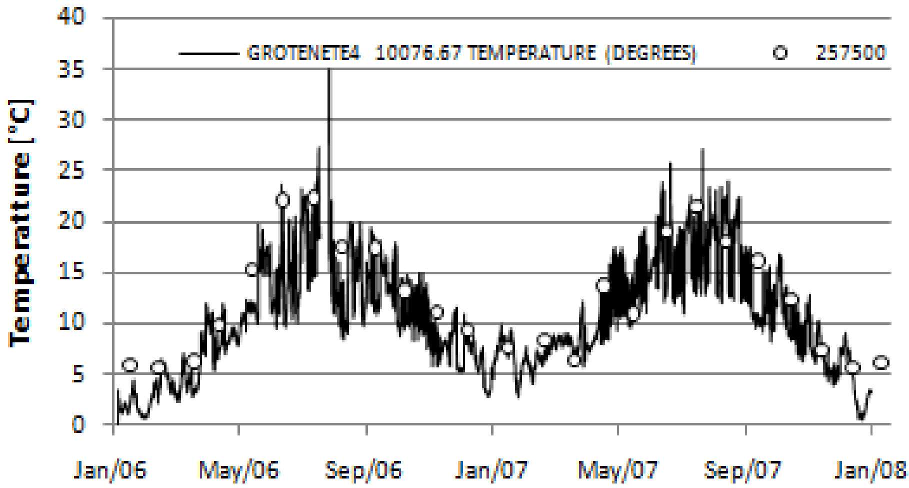


Figure 61. Comparison of observed and modeled water temperature values at Varendonk station.

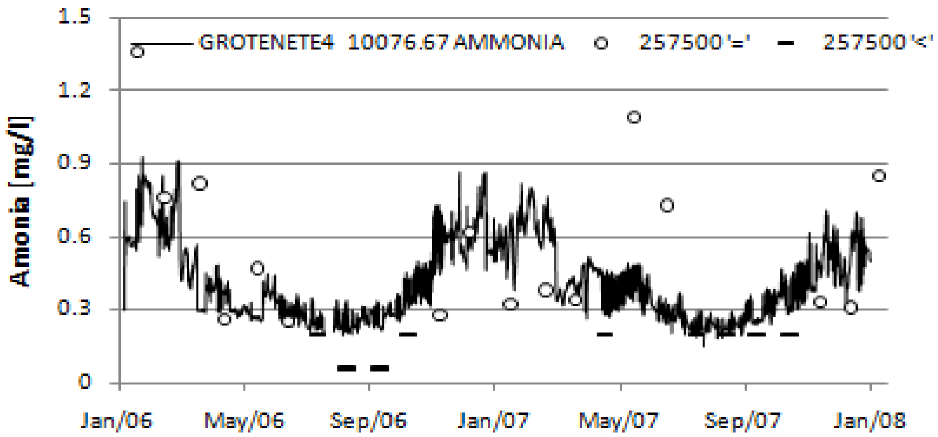


Figure 62. Comparison of observed and modeled NH4-N concentrations at Varendonk station.

5.6 Challenges for ecological impact modeling

The next step, the ecological impact analysis was found to be rather difficult. The hydrological and hydraulic models and impact results do not fully meet the needs of the ecological impact research. In the state-of-the-art hydrological and hydraulic river flood modeling (as described above), main focus goes to the accurate modeling of river inundations and their frequencies. This involves simulation of single flood events or synthetic events (for given return periods of flooding) derived from flow time series by statistical extreme value analysis in quasi-2D hydraulic models for the river and the floodplains, and mapping of the spatial extent of inundations using a GIS and topographical data. Ecological impact investigations, however, also require statistical information not only on extreme events, but also on less extreme events. The full time series of rainfall-runoff discharges thus needs to be simulated in the river hydraulic model, which is impossible because of the very long computational times of the hydraulic model. One solution would be to simplify the hydraulic model, for instance by calibrating a conceptual model to simulation results with the more detailed hydraulic model based on a number of (small, medium and high) events. After simplification, the full rainfall-runoff time series can be simulated and the results statistically post-processed also for the lower events.

Given that the statistical analysis on higher extremes is based on an extreme value theory (given the limited number of extreme events available in the time series) and for extrapolation purposes, the analysis for lower extremes requires a different approach (extreme value theory is not applicable below a threshold; but statistical analysis can be done empirically / non-parametrically given the larger number of less extreme events in the series).

Another, more important problem is that the floodplain modeling methodology, as outlined above, is developed for the quantification and mapping of the maximum spatial extent of specific flood events (historical or synthetic, and independent on the flood season). For the ecological impact study also other variables such as the flood duration, the temporal evolution of the floodplain filling, the flood season ("timing" of the flood events), etc., are required. These outputs are by default not provided. The flood duration can be modeled with the quasi-2D approach but largely depends on parameters describing the drainage or soil infiltration capacity. Calibration and validation is often done on the basis of the flood extent of large historical floods. For flood risk assessments, only the extent and depth of the flood is of importance. The emptying of the flood plains is therefore not modeled truthfully. The infiltration rate is often arbitrary and information on flood duration is not credible. Therefore, more attention should be given to the modeling, calibration and validation of flood duration and the underlying processes that affect flood duration. Also extraction of information on the flood "timing" requires additional post-processing and validation.

As a final challenge it can be noticed that the uncertainties on the water quality modeling results are an order of magnitude larger in comparison with the hydrological and hydraulic model results. This is mainly due to lack of detailed pollution input data and the limited temporal frequency of the water quality measurements. The water quality concentration results were found OK in order of magnitude for monthly values, but the model so far is not able to simulate daily or hourly concentrations in a reasonable way. It is this daily or hourly timescale that is of importance for an ecological impact analysis. The use of the water quality results in ecological impact investigations also requires further research.

CHAPTER 6: Interdisciplinary challenges for hydrological modeling

In this chapter we wish to identify and document the interrelations between the disciplines and point out the potential policy and management consequences. We also want to exemplify which variables are needed for interdisciplinary research and to which extent the disciplinary research is oriented towards adopting these interrelations. A special attention is given to the potential impact of CC on the ecological mechanisms.

We have identified and documented 2 mechanisms that significantly affect catchment hydrology.

Sewage system water transfers and the macrophytes as ecological engineers. The sewage water transfers occur between and within catchment boundaries and these transfers have serious consequences for modeling and the water balance of the catchment. The sewage infrastructure can be seen as a separate hydrological system that interacts with the river system. Not only is there a displacement of water across hydrological boudnaries. Also water is transferred between compartiments: a) parasitic drainage (groundwater to sewage) b) run-off (rainfall to sewage) c) overflows (sewage to surface water) d) discharge at treatment plants (sewage to surface water).

Climate change may result in an increase of:

a) excess rainfall during winter, which will affect parasitic water infiltration and sewage overflows
b) summer drought frequency and intensity, which will promote the accumulation of detritus and sediments within the sewage infrastructure.
c) intense convective summer storms, which will pose a threat to water quality because of increased risk of "first flush" sewage overflows after periods of drought.

Macrophytes can have a profound effect on hydrology. Depending on climate conditions, species composition, morphology and nutrient availability they will alter flow resistance and hydraulic head through many non-linear mechanisms. Prolonged periods of low flow, more sunlight, higher temperatures and higher nutrient availability (less dilution) will increase macrophyte growth and decrease the drainage capacity of streams. This is desirable for water conservation, but may pose local problems of summer flooding. Further research on these mechanisms is needed to progress on the modeling of water quantity and quality. Incorporation of these mechanisms into modeling approaches are needed as these mechanisms are very climate sensitive.

6.1 The macrophyte growth feedback mechanism

6.1.1 Introduction: Hydraulics and macrophytes

Macrophytes have re-appeared in many of our lowland rivers. For many decades, bad water quality and morphology disabled their occurrence. Increased waste water treatment efficiency for suspended solids has enabled the growth of macrophytes in many rivers. A crucial condition for macrophyte growth is light availability. The first generation of sewage treatment plants had a huge effect on the suspended solids, which improved light penetration and allowed re-establishment of macrophytes. Due to eutrophication, these macrophytes display excessive growth. The present excessive macrophyte growth during low flows increases the hydraulic head, decreases valley drainage and results in more stable and higher groundwater levels (Figure 63). The reduced drainage also increases denitrification because both the area of water saturated soils increases as the residence time of both surface and groundwater increases. The water retention has positive effects on water quality if droughts persist, but may cause problems for harvesting crops in the valleys and may cause floods during summer storms.

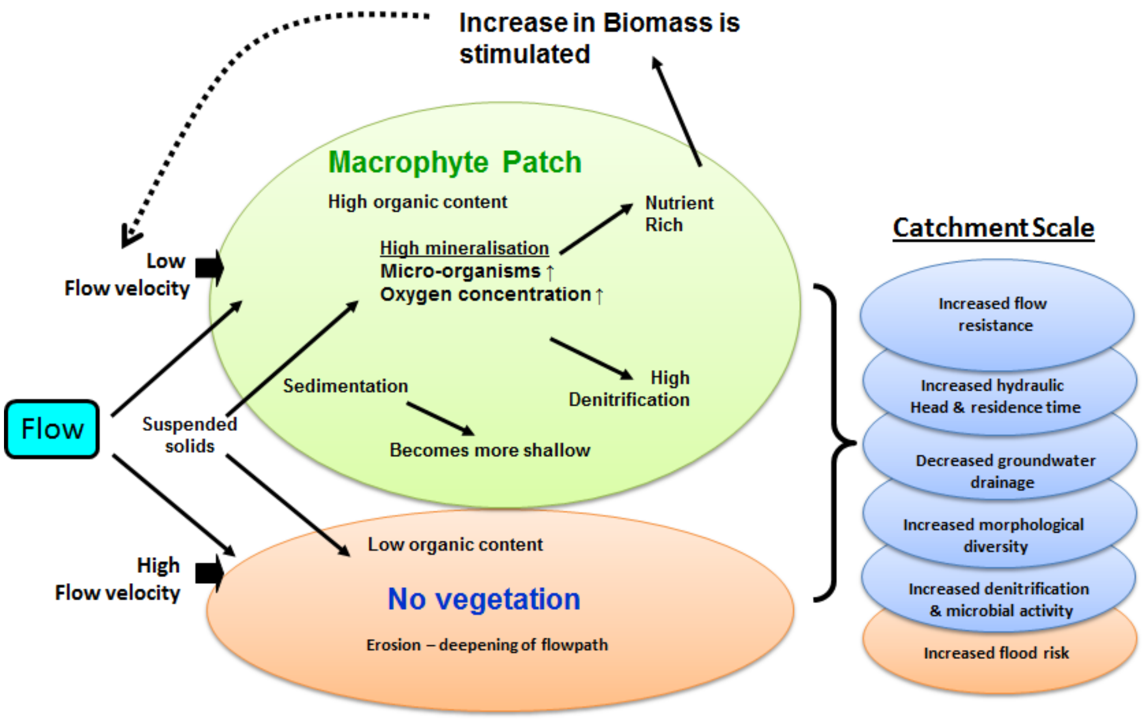


Figure 63: Feedback mechanisms between hydraulics, flow velocity, vegetation composition, nutrient availability, physical stress, and more….

Natural feedback mechanisms like this have long been underestimated and to a certain extent these mechanisms also have played a lesser role in the past. Many aspects of hydrology has been controlled through water management actions (drainage, straightening, dredging,…). Lowland rivers in northwest Europe have been subjected to an intensive management since long times (Moss 1998; Rackham 2000). The scope of this historical management was to drain land, prevent flooding and to create navigation routes for transport reasons (Iversen, Kronvang et al. 1993). In this way steep banks and uniform channels are created with a reduced connectivity between wetlands and rivers (Scholz and Trepel 2004). This generally creates a system with deteriorating water quality and high maintenance cost. This deterioration in river health has become unacceptable and many rehabilitation projects are nowadays carried out. A recent attention for river rehabilitation, nature development, fish migration passages increases the importance of understanding these mechanisms. The implications for management cannot be determined without interdisciplinary research. It is crucial to understand these mechanisms in order to produce truthful projections of management measures and climate change impacts.

6.1.2 A natural feedback regulation of river discharge

Aquatic vegetation plays a key role in the biogeochemistry and hydraulic functioning of rivers and provides a large series of ecosystem services. These systems are under increasing anthropogenic pressure and will be particularly vulnerable to climate change (Palmer, Liermann et al. 2008). When compared to terrestrial systems, the impact of these changes is scarcely investigated (Kosten, Lacerot et al. 2009). The main reason is that the biogeochemical and hydraulic functioning of aquatic vegetation is not straightforward: there potentially exist intricate complex interactions between hydraulics, biogeochemistry and the vegetation.

Where the different climate models are not always in agreement on the effect of climate change on river flows during winter, they do generally agree on the effects of climate change on the summer river flows. They point out that we can expect: an increase of persistent drought periods and an increase in heavy local thunderstorms.

Macrophytes and macrophyte patch growth shows self-organizing properties (Figure 63). Low flow conditions enhance macrophyte growth, which in their turn increases hydraulic friction. This friction increases the hydraulic head for the same flow. They thus provide a natural feedback mechanism to maintain a high water table. When macrophyte patches grow, they increase flow velocity at the edge of the patch to a point where the flow conditions do not allow the establishment of additional plants. The longitudinal length of the patch is generally limited by lower nutrient availability. The patch formation leads to the formation of preferential flowpaths, with deeper water table and high flow velocity. The patch itself traps sediments and becomes shallower. If water flows are very low and morphological quality is low (uniform depth and flow), there is no clear patch formation visible. This is understandable, because there are no differentiating factors for vegetation establishment. The down side of this is the complete filling of the stream with macrophytes. The way the river reacts to discharge changes depends on macrophyte densities, species composition and river morphology (Bal, Struyf et al. 2011). Different species have different reactions to changing flow velocities (e.g. some species are pressed downwards during high flow velocities). A good morphological quality favours a diverse macrophyte diversity and the creation of preferential flow paths. These preferential flowpaths can expand during peak flows. A low morphological diversity increases the risk of channel blocking by extreme densities.

Depending on the species, macrophytes exhibit different characteristics and effects on hydrology.

- Standing-Floating–submerged: Standing macrophytes have stiff stems and cause more friction then floating or submerged species.
- Water depth limitation: Certain species require shallow depths and can therefore not grow in a deeper preferential flow path.
- Rooting patterns and anchorage: Species with an extensive, robust root stem are able to store nutrients and are able to grow very fast after removal of the weed.
- Patch formation: Certain macrophyte species tend to grow in patches and capture sediments and nutrients by reducing local flow velocity within the patch. They therefore create favorable growth conditions. The patch size is limited by the nutrient availability at the end of the patch.
- Nutrient limitation: Macrophyte growth is limited by phosphate availability. In rural areas, phosphate is abundant in surface water and is no constraint.
- Light limitation: In relation to water depth and suspended matter,
- Flow velocity limitation: certain species may be limited by water flow velocities.

Water discharge and ecological values often have conflicting interests. To prevent flooding in areas with human occupation or agricultural benefits aquatic vegetation is removed. The regular removal of macrophytes favours species that have a quick recovery. These species tend to grow faster and denser and seem to aggravate the discharge problem. Mowing in patterns is a part of the solution but is not always applied. In order to better combine discharge capacity with ecological values, there is a need to improve further on the morphological quality. A good morphological quality allows the establishment of patches and preferential flow paths which can allow a better through flow when needed.

Climate change is likely to increase the occurence, severity and duration of droughts. Increased nutrient concentrations, because of less dilution due to lower summer rainfall (Baguis, Roulin et al. 2010) and increasing water abstractions, stimulates macrophyte growth (Carr and Chambers 1998). With increased nutrient availability leaf area increases (Andersen, Pedersen et al. 2005) and nutrient demand is satisfied by foliar uptake: aquatic plants tend to invest less in root development under high nutrient availability (Mantai and Newton 1982), causing a reduced anchorage strength (Schutten, Dainty et al. 2005). This reduction in root biomass will even be re-enforced due to: the reduced nitrification activity (De Boer 2001) with increased soil acidity, an indirect effect of increased carbon dioxide, and the lower nitrate concentrations in the interstitial pore water (Andersen, Pedersen et al. 2005). A strong pot(Bal, Brion et al. 2009)ential trade-off between nutrient availability and mechanical stress therefore exists with increased nutrients leading to higher biomass production but causing increased mechanical stress by the flow that can eventually be damaging aboveground biomass or dislodging the plant.

During period of stable low flows with high nutrient availability, macrophytes can fill up the entire channel, increase flow resistance and rise water tables. This causes natural feedback retention of both groundwater (decreased drainage) and surface water. By lowering drainage they mitigate desiccation of nature and agricultural land. Climate change is also likely to increase the occurrence of more extreme precipitation, which can lead to CSO-events. The sequence of drought, followed by summer storms is detriment to the water quality because of first-flush CSO-events. If this polluted water is brought into floodplain ecosystems, there can be severe ecological impact. The occurrence of summer flooding, due to extreme precipitation is not unlikely if river beds are completely overgrown by macrophytes and if the hydraulic head (water table) during low flows already reaches close to bankful situations.

Nevertheless, the macrophytes also have positive aspects like drought vulnerability mitigation and water quality improvement. Macrophyte growth increases flow resistance and hence increase hydraulic head and water residence time. The increased hydraulic head, decreases valley drainage and results in more stable and higher groundwater levels. Their impact on flow resistance can be desirable during droughts and reduce the impact of severe droughts on both nature values as agricultural yield.

Besides the hydraulic consequences also the nutrient cycling of rivers is influenced by macrophytes. Macrophytes take up nutrients during growth stage (ammonia is preferred) and release those during autumn and winter (mineralisation). Removal of macrophytes can remove nutrients from the river system, although this amount is insignificant compared to the total dissolved load in eutrophic streams. Macrophytes take up nutrients and store these in biomass. When the growing season ends, some macrophytes may store nutrients in their root system. During decomposition the majority of the accumulated nutrients are released again into the water. To investigate the seasonality of this release plant material was incubated in a lowland river. From these experiments clear seasonal differences were seen in decay. Highest break down was seen in spring because a lot of macro-invertebrate shredders were present. Between the tested species differences in decay were seen. *S. pectinata* decayed significantly faster than *P. natans*. As a consequence of this fast decay nutrients are released into the water column. When macrophytes are removed from the river, to reduce the friction for example, nutrient release is prevented with a decreased input towards downstream regions.

Phosporous should be the limiting nutrient in freshwater ecosystems, but there is no limitation of phosphorous and is the main cause for excessive growth of macrophytes. Macrophyte growth is limited by phosphorous availability in most cases. The uptake is primarily through the root system. Macrophyte patches are able to capture adsorbed nutrients through sedimentation within the patch. Usually nitrate is not a limiting nutrient within the Flemish context.

Beside the uptake of nutrients, also denitrification is promoted. The increase of the hydraulic head increases denitrification both by area (increased area of water logged soils) as by residence time. Opitz en Behrend (1999) developed a model that calculates the additional denitrification on basin scale through water retention. The effect of a higher hydraulic head (a decreased drainage) promotes denitrification in different ways. Residence time is an important variable in denitrification models (Seitzinger, Harrison et al. 2006). But an increase in hydraulic head does not only affect in stream denitrification (sediment-surface water), but the reduced drainage also increases denitrification in riparian zones because both the area of water saturated soils increases as the retention time of groundwater increases.

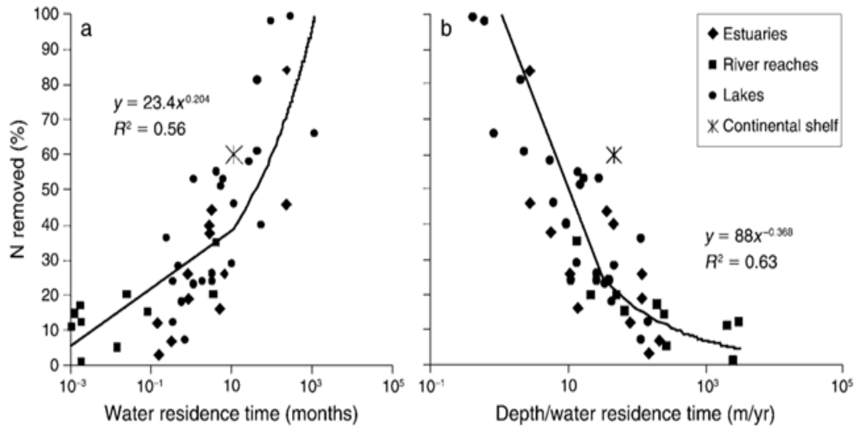


Figure 64: Effect of water residence time on relative N-removal

When we look at the parameters determining macrophyte growth the following results are available (Bal, Brion et al. 2009):

- The uptake of inorganic N is independent from the available light.
- No significant temperature effect could be detected on the NO3- uptake of *P natans* and *C platycarpa*. Except for their dependence on temperature, both macrophyte species displayed very similar DIN and DIC uptake behavior in spite of their very different morphology.
- Ammonium is the preferential source of nitrogen on the short run but can become toxic on the long run.

Macrophytes produce oxygen when photosynthesis is active. If vegetation is abundant, this may even lead to oversaturated oxygen levels. This saturation point (maximum dissolved oxygen concentration) is lower when the water temperature is higher and a general temperature increase (climate change) may thus increase the frequency of hyper saturated episodes. High oxygen availability may boost microbial activity if water quality is poor (high BOD). During daytime this will have a positive effect on water quality, but may aggravate oxygen depletion overnight as microbial activity continuous and photosynthesis does not. It is unclear how high oxygen levels and oxygen fluctuations affect other water quality regulating processes such as nitrification and denitrification. In the undesirable situation of very extreme fluctuations of oxygen levels, we can expect that denitrification in the water column is promoted. Very high oxygen levels allow a deeper penetration of oxygen into the sediments and create a wider gradient zone for river bed denitrification (nitrate rich groundwater that seeps into the stream through the sediment layer). Furthermore it has been stated that higher oxygen levels also allow more benthic diversity and activity. These benthic organisms create channels and bio-turbation, which allows a better oxygen penetration and thus contact area for denitrification (Henriksen 1983). Removal of nitrogen through denitrification in sediments as a function of eutrophication (increases towards the right end). Organic matter is deposited from the water column and consumes oxygen through degradation. As oxygen consumption increases with eutrophication, the sediment becomes completely oxygen free and the ecosystem switches into another state in which nitrogen is recirculated instead of transformed into nitrogen gas (Norberg 1999).

6.1.3 Simple macrophyte growth model as a solution?

A strongly simplified macrophyte growth model could be developed for inclusion in hydrological models. But it is not evident to incorporate this within the hydrological models. These models are often unable to deal with variable manning coefficients. A fundamental question here is whether hydraulic models predict flows or predict water levels. It is seen that flows remain relatively stable for increasing hydraulic head. It can very well be that the models may predict flows rather well, but fail to relate that to the correct hydraulic head. This again raises issues on the usefulness of hydrological models for ecological studies. The hydraulic head strongly determines the riparian groundwater levels and drainage potential of the streams. The impact of low flow episodes on riparian vegetation can thus not be predicted very well, even with advanced coupled groundwater – surface water models. Also for predicting summer flooding, this poses problems if the flow resistance and hydraulic head are underestimated.

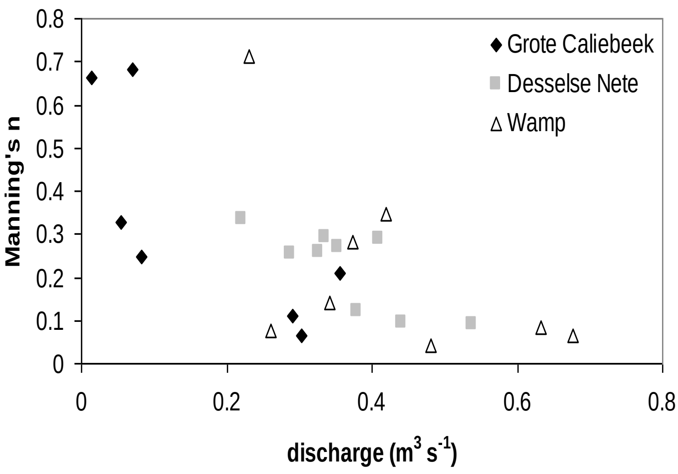
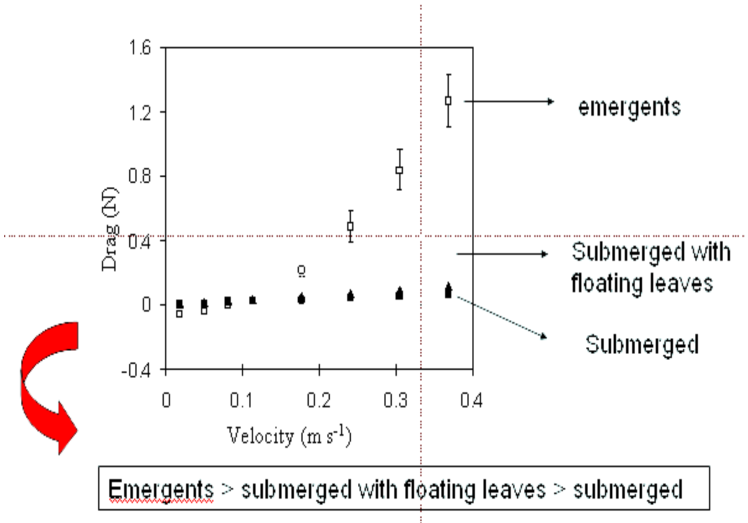


Figure 65 demonstrates that the relations between the flow resistance (manning) and discharge are not straightforward and depend on species composition and local morphology. This makes it very difficult for hydraulic studies as this would require a highly variable manning. Indeed, manning coefficients could only be derived from iterative calculations and would require the inclusion of macrophyte growth models (Manning ↑ => Flow velocity ↓ => Hydraulic head ↑ => Manning ↓).

Figure 65: Flow resistance – discharge plots for different river in the Nete catchment (Bal, Brion et al. 2009)

Species composition

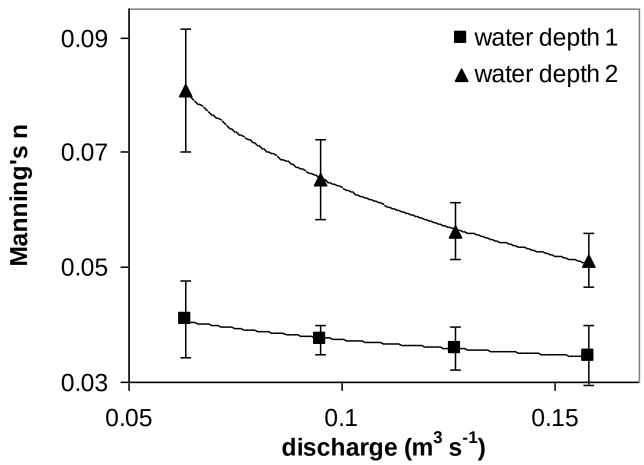


Flow resistance depends to a certain degree on the species composition (Figure 66).

Emergent species display the highest flow resistance. They also display a strong non-linear effect of flow resistance with respect to flow velocity. Submerged species do not display this non-linearity so strong. This is because the stems are flexible and are pushed down or simply break under higher currents.

Figure 66: Effects of macrophyte morphology on flow resistance (drag)

Water depth



The flow resistance also depends on the water depth. For an identical biomass, the friction increases for a lower water depth (Figure 67). This is evident as the biomass is compacted to a smaller volume. The flow resistance decreases with higher flow, but this is more outspoken for higher water levels. In reality, there is a complex relation between flow resistance, hydraulic head (water) level, flow velocity and flow volume. But these relations are dependent on species composition.

Figure 67: Effects of water depth on flow resistance for single species

Flow velocitiy

Different species have different reactions to changing flow velocities. Some species are pressed downwards during high flow velocities (Figure 68). But other effects may also occur (e.g. dislodgment). A good morphological quality favours macrophyte diversity and the creation of preferential flow paths. These preferential flow paths can expand during peak flows. A low morphological diversity increases the risk of complete channel blocking and extreme densities.

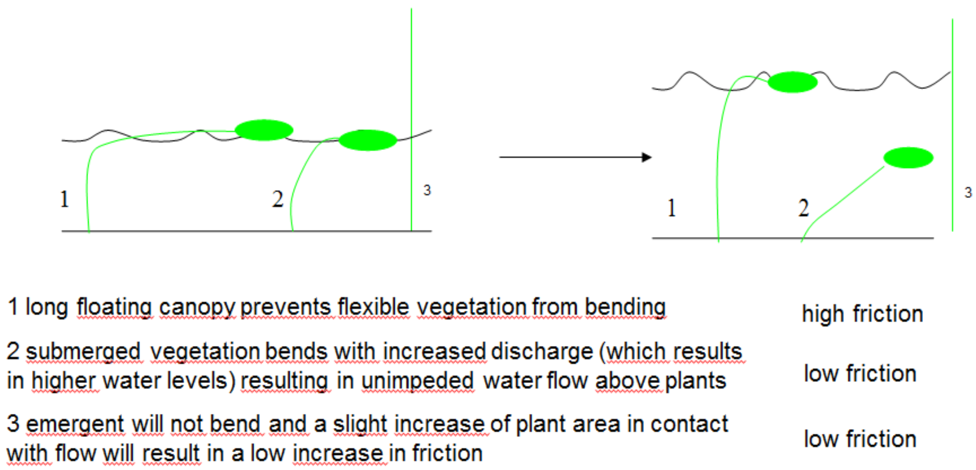


Figure 68: Effects of different floating vegetation length conditions (Bal, Brion et al. 2009)

Macrophyte response to removal

Macrophyte removal may appear a solution to the increased flood risk. To prevent flooding in areas with human occupation or agricultural benefits aquatic vegetation is removed. This favors species with extensive root anchorage since these species can recover better then those species that are removed along with the root system. Also these species stock nutrients inside the root system which allows them to grow very fast. Because re-growth can be very fast (less than 6 weeks) a second and even third vegetation removal becomes necessary. This re-growth capacity is strongly determined by the time when vegetation cutting occurred. For *Stuckenia pectinata* (former *Potamogeton pectinatus*) this re-growth was small when vegetation was removed later in the vegetation season (after July). Unfortunately this re-growth is also influenced by parameters like weather, light and temperature which should be incorporated into management of lowland rivers. In the end vegetation removal seems only to increase the problem since robust, fast growing species out-compete other species.

In pristine conditions various species with different resistance against disturbance (e.g. vegetation cutting) occur besides each other. Frequent vegetation cutting can thus lead towards a vegetation community with mainly fast growing species. Therefore vegetation removal can influence the species composition of a river with different consequences for the hydraulics. Experiments were therefore carried out which looked at the growth of a plant community before and after mechanical vegetation cutting. From the results it was seen that, for *P. natans* dominated communities, no difference in biomass production was observed before or after the vegetation cutting. In unmanaged river stretches a faster decline of the vegetation growth in time was observed when compared with frequently managed rivers.

When macrophytes are completely removed, the hydraulic resistance decreased, however the ecological damage is high. To reduce this impact one can remove parts of the aquatic vegetation. Hydraulic resistances of these partial vegetation cuttings increased maximal 23 % under different hydraulic regimes. Different species and water level also had an effect on the vegetation resistance. Three distinct plant groups with different hydraulic resistances were detected. Seasonally these hydraulic resistances changed with a factor 8 depending on vegetation development. When vegetation was removed friction decreased immediately, but increased again with a second peak later in the vegetation season.

Macrophyte response to elevated CO2 levels

How macrophytes will respond to elevated CO_2 levels is another pathway of CC-impact. There are strong indications that higher CO_2 levels positively affect macrophyte growth. Both leaf length as biomass increases with higher CO_2 availability (Anderson 2005). The effect is proportional to the growth and not absolute. This means that at sites with favourable light and nutrient conditions the additional growth in absolute biomass will be much higher than for less suitable locations. Not all macrophytes will benefit from this increased availability of carbon dioxide. Higher CO_2 concentrations will potentially favour C3 plants over the more carbon dioxide efficient C4 plants. The last have a lower phenotypic plasticity towards changing environmental conditions (Sage and McKown 2006) resulting in a competitive disadvantage at increased carbon dioxide concentrations. If climate change increases average and extreme temperature, this will also effectuate in higher water temperatures. Increased water temperatures will promote C4 carbon uptaking plants (tropical species) above the C3 pathway plants (= dependent of dissolved CO2). These exotic species are able to become dominant in these warmed up regions. Examples of this are Water Hyacinth (*Hydrocotyle ranunculoides*) and Kransvederkruid and (*Myriophyllum aquaticum*), which are already been reported problematic in parts of Northern Europe.

6.1.4 Conclusions

The natural feedback mechanisms that exist in natural(ised) rivers pose a huge challenge to water managers and hydrologists. The current hydrological models do not allow incorporating such mechanisms because these are primarily developed for flood prediction and situations with normalized rivers without vegetation (winter). The projected climate scenarios indicate that the drought problems in summer may become a larger problem then the risks of summer flooding. Yet, river naturalization may provide a solution to some of the climate adaptation challenges. The lowland rivers of the Grote Nete cachtment may serve as the water reservoir for a large part of Flanders. Infiltration of excess rainfall during winter can be a strategy to reduce the risk of winter floods and increase the base flow during summer. The sandy soil substrate in the catchment allows storing high amounts of water (20 - 30 % of volume) in the soils. This may even be higher (40 - 60 % vol.) for peaty soils. If droughts become more frequent and persistent, the urge to infiltrate and retain excess rainfall during winter becomes high. River restoration and extensive management can reduce groundwater drainage throughout the entire catchment and improve water quality through sedimentation, uptake and increased residence time (denitrification).

6.2 Sewage water transfers: implications for management and research

6.2.1 Sewer systems in Flanders

Urban hydrological regimes work at different time scales then the natural hydrological regime and they are much more diverse. The natural system generally reacts relatively slow to rain events because of the many buffering systems (infiltration, wetlands, etc.) that make part of the catchment. This results in a time lack between a rain event and the increase in flow at the river mouth. The anthropogenic hydrological system can shortcut several of these buffer systems and thus increase peak flow. Reaction time between rain events and flow changes are in principle much shorter in urban catchments. Urban areas are seldom fully paved and a mosaic of infiltration patches is present within a paved environment. The materials used for street paving are diverse and can vary in permeability. The effect of urbanization on the hydrology of a basin is however tremendous. This complexity is increased by the presence and operation of sewage systems. The awareness of water quality problems and the consequential sewage infrastructure has grown over the last decades and up till now, still new sewage piping is being installed. This results in a diversity of old and new infrastructure of which the exact functioning and state is not that well-known. Spatially distributed urbanization can be seen along the road-infrastructure. This increases investment and maintenance costs for connection to sewage treatment systems. The numerous kilometers of sewage infrastructure increase the risk for inflow of parasitic water. Beside the increased run-off water from roads and housetops, there is inflow from ditches and drainage-systems. This dilution decreases treatment efficiency and causes hydraulic instability during intense rainfall events and congestions during dry periods. Specifically, rainfall events that follow dry periods result in a significant load towards receiving surface waters by sewage system overflow events. The construction dates of the treatment plants and affiliated sewage piping ranges from 1964 to 1999. The current capacity (equivalent number of inhabitants) of the WWTP's has increased by technological updates and expansion. A 40-year time span between the first infrastructure and today illustrates the sheer diversity of technology and materials. The older 'sewage' infrastructure comprised local streams and man-made ditches that first served as open sewers and then were piped and covered. These old systems combine discharge of groundwater drainage, run-off drainage and actual sewage water. Actions were taken on different policy levels to start waste water treatment and new infrastructure was built, though in many cases the old systems were connected to the new ones.

6.2.2 What world do we model?

As anthropogenic influences increases, water regimes in a catchment are further diverted from their natural regime, thus increasing complexity. While these changes increase complexity they also have a huge impact on the uncertainties that we have to cope with. Complex catchments and its hydrological regime are a result of centuries of changing views of men on water and land use management. While many management principles are no longer valid and management no longer applied, the results of these practices are still present within the catchment. The outcome is a hydrological system that incorporates many different aspects, both natural and artificial. The development of reliable hydrological models involves both calibration and validation of the model. This requires input data (rain, temperature, evapotranspiration, gauging data etc.). But as the hydrological system is constantly subject to changes, the calibration and validation period applies to slightly different systems (e.g. large sewage infrastructure works in the late 90's). Models are often based on the available data. Most often there is a discrepancy between the time period used for the model design and the time period used to calibrate and/or validate the model. When we want to use the model for future predictions another discrepancy arises. Simulating future predictions inherently incorporates uncertainties concerning future catchment development (e.g. land-use change).

In the following section we briefly discuss and illustrate the different effects of the human environment on both spatial and temporal complexity of catchments and their effects on the different modeling uncertainties.

6.2.2.1 Spatial complexity

Spatial hydrological relations play a vital role within modern water management. In recent years focus has shifted from a local to a catchment view in which upstream-downstream dynamics take a central place. Upstream actions have downstream effects and management should be performed on a catchment scale. Recent model developments have tried to capture this dynamics by a stronger focus on physically based, catchment scale models. These are run-off models that incorporate physical attributes like land use, soil charateristics and slope. But the basic idea of these models is developed around natural, little disturbed, catchments. But we can wonder to what extend today's catchments, with an ever-increasing human impact, still fit within this catchment-scale concept and to what extend these models can incorporate these changes. Relevant questions that we can ask are: "Is rain the only driving force of river flow?" "Are catchment borders still the same as a few decades ago?" "Does water always run downstream?" "Can we actually use run-off based models to simulate hydrological regimens in heavily altered hydrological systems?"

Urbanization drastically changes the hydrology of a drainage basin. Roads and artificial surfaces cut down infiltration and storage while storm sewers speed up the flow of water into rivers. The most significant impact of urban development on water resources is an increase in overall surface runoff and the flashiness of the storm hydrograph (Praskievicz and Chang 2009). But besides this well-known effect there is a whole list of other elements that are relevant to urban hydrology like base flow, inter-basin water dynamics, flood protection, etc. These different elements lead to a highly complex hydrological system. The incorporation of these elements in the hydrological models, if they are known, is a difficult and time expensive undertaking. Physical based models have difficulties to incorporate the sewer infrastructure and are therefore generally neglected or compensated with statistical factors. In recent years methodologies have been developed that allow the incorporation of the storm water drainage systems in digital terrain models (Gironas, Niemann et al. 2010). But problems arise when sewer pipes cross stream reaches. These crossings cannot be incorporated in the digital terrain model. Combined physically based models are therefore hard to develop.

The conceptualization of paved run-off within distributed models is an Achilles tendon. Overland flow towards the streams is affected by: 1) the distance to the stream 2) the imperviousness of the land-use and 3) the slope. Paved run-off generated by remote urban areas has long travel distance towards the stream and partial or even full losses take place. These losses are usually compensated by a general overestimation of the actual impervious area. Large quantities of paved run-off need to be generated in order to contribute to stream peak flow. In this perspective, paved area configuration might play an equal important role as effective paved area. In an urbanized environment with sewage infrastructure, this might not only affect the performance of peak flow simulation, but also has indirect effects on parameterization of other land-uses and the over-underestimation of certain flow components. Paved run-off from remote urban areas is fully infiltrated along its way, by which groundwater recharge and phreatic level could be overestimated.

The non-linear behavior of storm water drainage makes generalization of this issue problematic. For mild precipitation, paved run-off is infiltrated through cracks and holes in the paved surface. The permeability of impervious materials is relative and difficult to quantify. Temperature and sunlight intensity then may play a significant role to the evapotranspiration – infiltration balance. Combinations of excessive rainfall in both intensity and/or duration might trigger sewage overflows.

From the ones that are monitored, we learn that overflows occur about 3 to 6 times a year. The principal dimensioning used is that a sewer pipe has a maximal hydraulic load of 6 times the dry weather flow (DWF14). Consequently overflow frequency and overflow volume are inversely correlated as they are triggered by extreme climatic events (Vaes, 1999). In addition it is observed that the occurrence of extreme precipitation and thus overflow events increase (Vaes, 1999).

The distortion of paved run-off stream flow contribution probably affects calibration and is one cause why peak-flows are underestimated in model simulations. Most of the time, run-off water is delayed through sewage infrastructure or even exported to outer basin. Peak precipitation has instant stream flow contributions through sewage overflow systems that bypass the sewage infrastructure. In addition this run-off contribution is not exported to outer basin. This threshold is likely to be very different for each overflow system and depends on local precipitation intensity and duration. However, these extreme conditions do not often occur and the weight in the calibration procedure is under normal conditions. Therefore the model cannot be representative for extreme hydrological conditions as the system fundamentally changes (flips).

6.2.2.2 Impervious area and households: peak flow vs. base flow

A very strong example of the influence of anthropogenic activities on the hydrological regime is the recent increase in impervious area and the development of sewer infrastructure. In many catchment systems the urban land use and the sewer infrastructure have become a determining factor for the water flow regime. Commonly urban land use encompasses only a low percentage of the catchment area. But it has a disproportionately large influence on both hydrology and biogeochemistry of receiving streams (Paul and Meyer 2001; Feminella and Walsh 2005; Cunningham, O'Reilly et al. 2009). In landscape indicator studies urban land use is generally aggregated to one class “impervious area” which has become an important indicator for anthropogenic impact on both landscape and aquatic ecosystems (Arnold and Gibbons 1996). Impervious area generally consists of roads, houses (or roofs), gardens etc. Considering the water flow regime the impervious areas can have two important , completely different impacts. A generally well known effect is the impermeability of the impervious area. Impermeability reduces the infiltration capacity of the soil and increases the runoff thus generating lower ground tables, increased peak flows and reduced base flow. Effects of increased imperviousness and runoff are a well-studied phenomenon because of the negative effect it imposes on flood safety. Not only real impervious areas have impact an impact on infiltration capacity. But almost every land use management decision that we take has a positive or negative impact on the infiltration capacity of the soil (Berglund, Ahyoud et al. 1981). Because of the reduced infiltration impervious areas have generally a negative effect on the ground water table and the river base flow. But households, homes and industries, which are a part of the impervious area, today also generate a part of the base flow. The constant water use by households during their day by day activities and the use of water for industrial processes generates an almost constant flow to the river system. Often the water used is captured outside the catchment and transported within the catchment. This eventually results in an increase in the amount of available water within the catchments river system. This water use, while relatively small on a household scale, can reach significant proportions when all households are taken into account. When we for example consider the Nete catchment we can easily demonstrate its potential. The Nete river discharges yearly on average 389 million m³ water in its receiving river the Rupel. Of this, 61 million m³ (mean 1999-2008) or 15% of the total discharge is water that comes from the different WWTP’s in the catchment. Part of this water comes from rainfall and seepage into the sewer system, but a major part is household generated. At the same time only 75% of the households is already connected to the sewer. Part of this water is transported from outside the catchment are pumped-up from the ground water but only at a limited number of places in the catchment. Thus locally affecting ground water regimes. The water discharged by the not connected households is not incorporated in this calculation. Especially in the summer when ground water levels are low and rainfall limited, the impact of households on the river flow might be relatively high.

If we want better understanding of the natural system within complex catchments, we may need to separate the anthropogenic base flow from the natural base flow. These sources of water need to be incorporated within hydrological models. Even when this might not be a relevant factor at the moment, it can become one in the future (e.g. reduced natural flow in summer).

6.2.3 Sewer infrastructure and water displacement

Water quantity models are a central factor in the progress of a good understanding of the water regime within peri-urban areas. They allow us to analyse both current and (hypothetical) future conditions. Water quantity models are largely dependent on information about the upstream areas that reflect the natural run-off conditions. The modeling of natural catchments tends to be relatively straightforward, water flows downstream and the direction is topography dependent. In contrary, sewer infrastructure completely disregards topography. Pipes and pumps allow the water to be transported upstream, downstream or between catchments and sub catchments. The objective of this exploratory research was to better understand the relative importance of the sewer infrastructure in terms of impervious surface fractions, and to quantify the impacts of these on the downstream river flows. As a result upstream areas can change drastically. Areas can be removed from the upstream area are added. Areas that used to be downstream might even become upstream situated areas because of pumping. Sewer infrastructure thus functions on a different spatial and temporal scale than the natural system. But both systems do interact and influence eachother (rainfall collection, groundwater drainage, pipe crack exfiltration, sewage overflows and WWTP discharge). The coupling of both systems is a huge challenge. Anthropogenic land use changes significantly impact flow regimes and water quality of river systems. Besides being a source of diffuse pollution, urban areas affect different hydrological components such as infiltration, surface run-off and evapotranspiration. Next to the direct influence through changes in the land surface, the growth of urban areas is often combined with an expansion of the sewer and wastewater treatment infrastructure. The latter might have a strong influence on the river high and low flows, as is shown for the Belgian case-study of the Grote Nete catchment. This catchment (380 km²) is a mosaic of semi-natural, agricultural and urbanized areas. Besides wastewater, the combined sewer systems collect run-off water from impervious areas and parasitic water from drainage systems and through seepage. This causes huge water transfers between neighboring catchments and can strongly change the hydraulic residence time of the system. The effects of urbanization have been extensively studied by researchers, but the effects of inter-basin water transfers by sewer infrastructure are seldom incorporated in impact studies. Their effects might however, be very large, as is shown for the Grote Nete case.

We studied the influence of the sewer and wastewater treatment plant (WWTP) infrastructure on the downstream river flows in the catchment of the Grote Nete, combining geographical analysis with hydrological modeling. Based on a high-resolution digital elevation model, the Grote Nete catchment has been split in fine scale sub-catchments (Figure 69). For 131 points within the catchment upstream land use fractions were derived from a land use map. A spatially explicit quantification was made of the impervious areas (rooftops and paved areas) and linked to the spatially variable inflow areas of sewer and WWTP infrastructure. Based on that analysis, the impervious areas were also classified as connected and not-connected to the different WWTPs. Three WWTPs are located inside the catchment (they receive surface water from outside the catchment); while five others are located outside the catchment (surface water is transported to neighboring catchments). Upstream areas were recalculated for the 131 selected points, with the upstream areas adjusted for WWTPs connected areas.

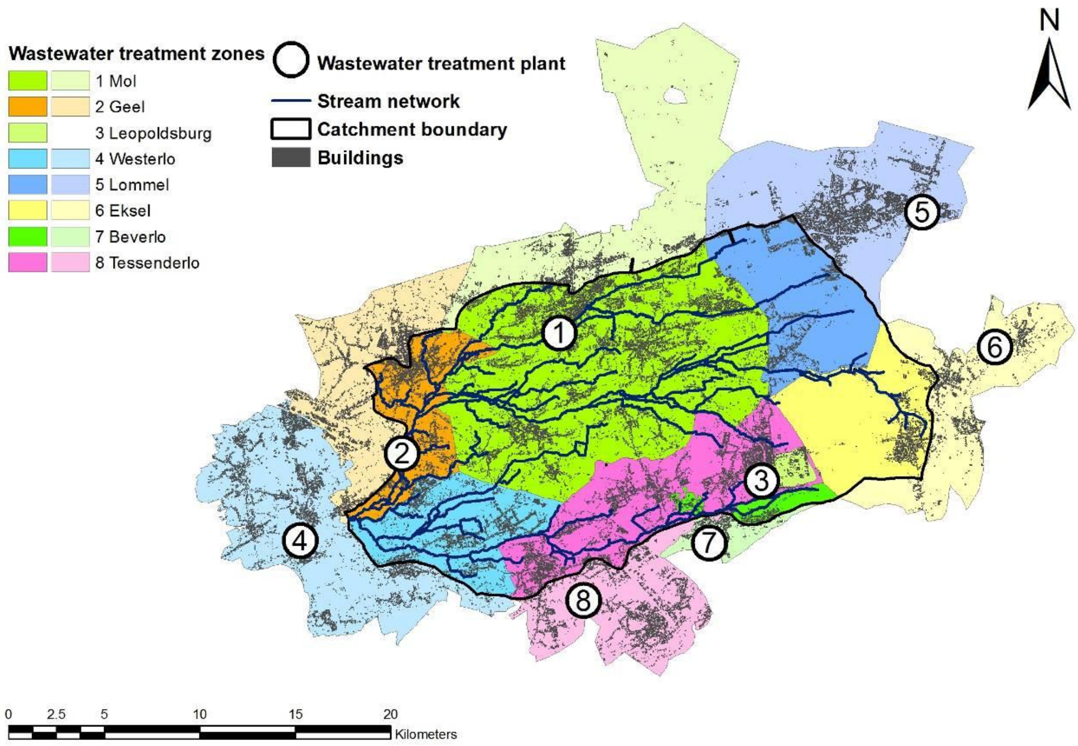


Figure 69: Different wastewater treatment zones that are, at least partially, situated within the Grote Nete catchment

From the analysis, we deducted that there are very significant water transfers between the Grote Nete catchment and adjacent catchments. In total, 2836 ha of impervious surfaces are present in the catchment. Of those, 1661 ha are connected to the WWTPs, 1175 ha are not. In total 761 ha of paved run-off is transported outside the catchment by the sewer system. At the same time waste- and rainwater from 461 ha is transported from outside to inside the catchment. Considering the 131 points, changes in upstream area across the Grote Nete were considerable. Upstream impervious areas could decrease by 99% or increase by 64%. Total upstream areas changed between -16% and 3% (Figure 70 a and b).

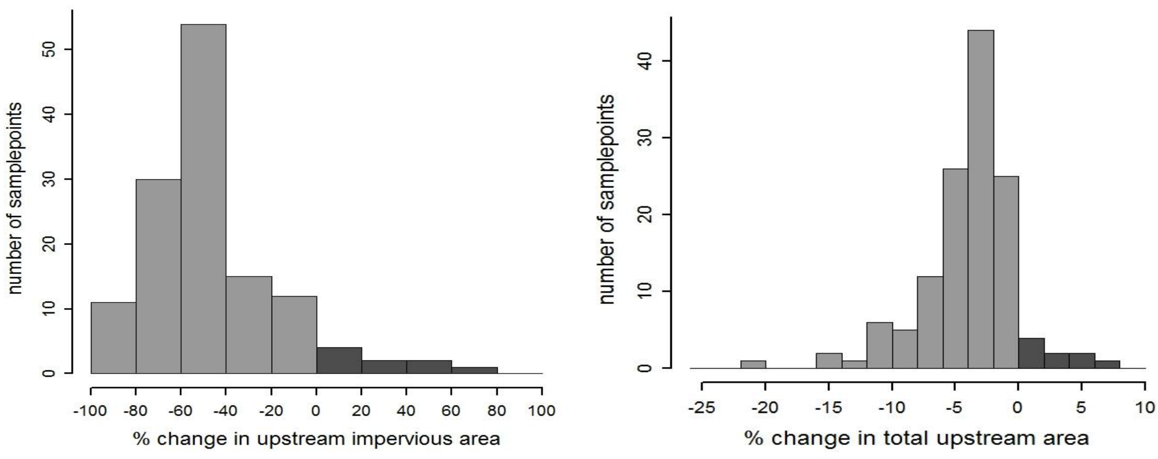


Figure 70 a and b: For the different points (n=131) the difference in upstream impervious areas was calculated between natural upstream areas and upstream areas with the WWTPs connected areas included. Upstream impervious area could decrease by 99% or increase by 64%. Total upstream areas changed between -16% and 3%.

The corresponding impacts on the downstream Grote Nete flows were found to be very significant, in terms of cumulative surface runoff volume, but also in terms of the contribution of the WWTP effluent discharges in the river baseflows (summer low flows). The WWTPs were found to contribute between 6% and 10% of the monthly average river flow at the Grote Nete catchment outlet (Table 12). The effect of sewer overflows (not considered in this study yet), will further increase the river peak flows and consequently also the river flood frequencies.

Table 12: Monthly relative contribution of the WWTPs to the Grote Nete river flow at the catchment outlet

Month	Avg. monthly flow at Varendonk	Geel WWTP (m^3/s)	Mol WWTP (m^3/s)	Geel WWTP (rel. %)	Mol WWTP (rel. %)	Sum (rel. %)
jan	7.03	0.20	0.22	2.8%	3.1%	6.0%
feb	5.86	0.22	0.26	3.8%	4.4%	8.2%
mrt	6.48	0.20	0.24	3.0%	3.8%	6.8%
apr	4.94	0.16	0.21	3.2%	4.2%	7.5%
mei	3.61	0.14	0.20	3.9%	5.4%	9.3%
jun	3.61	0.13	0.17	3.6%	4.7%	8.3%
jul	3.36	0.14	0.19	4.3%	5.6%	9.8%
aug	2.98	0.14	0.17	4.7%	5.8%	10.4%
sep	3.39	0.14	0.17	4.2%	5.0%	9.3%
okt	3.57	0.15	0.18	4.2%	5.0%	9.2%
nov	4.57	0.18	0.20	3.9%	4.5%	8.3%
dec	5.67	0.19	0.20	3.4%	3.6%	7.0%
Correlation to Avg. monthly flow		0.90	0.84	3.7%	4.6%	8.3%

The changes in the total upstream area and upstream impervious surfaces that contribute to natural catchment runoff in the Grote Nete were found to be large and will further increase in the future. These results and the relative contribution of the WWTPs to the river flow show the potential importance that sewer infrastructure has on the hydrological subflows in a river catchment, and on the water-transfers between neighbouring (sub-) catchments. This clearly demonstrates hydrological modelling in strongly urbanized catchment can encounter serious difficulties. To name a few of these difficulties:

- The catchment area and its land-use that is typically used for calibration of rainfall-runoff (e.g. for the areas upstream of flow gauging stations) might differ strongly from the real land-use. This might result in inaccurate calibrations and impact assessments. Overestimation of the impervious surfaces would bias the hydrological model parameters during the calibration (e.g. underestimate the surface runoff coefficient) When the model would be used for impact analysis of urbanization and climate change scenarios, the impacts on peak flows and flood frequencies would be underestimated.
- The presence of sewer infrastructure does not only increase the surface runoff volumes; they also increase the runoff velocity (sewer systems have a shorter concentration time). This increases the peak flows. This indicates that (natural) hydrological catchment models and sewer system models have to be considered and calibrated in an integrated/combined way.

In the future we need to find technical solutions to the difficulties faced in hydrological modelling of largely urbanized areas. In the discussion section of the paper, we will discuss a few possible solutions.

6.2.4 Conclusions

The main message is that sewer infrastructures in general and inter-basin water transfers specifically, are important factors to be taken into account when studying/modelling in peri-urban catchments. Not recognising these aspects can lead to strong biases in their impact quantifications. In 2008 zoning maps were approved in Flanders that will determine the further development of the sewer system: which houses will be connected to a waste water treatment plant, which won't.

Based on these upstream areas could be recalculated for the Nete catchment and estimates could be made of the further change in upstream areas. When the zoning maps are fully executed upstream areas will further be diverted from their natural state. Future upstream areas can experience a further decrease by 3.65% or an increase by 4.39%. The upstream households will be reduced in most reaches, but can also increase significantly. This might have a important impact on the base flow in the rivers. These changes will have impacts on water quantity and water quality. The socio-economic system is an ever-adapting and changing system. This implies constant changes in policy, society, economy. Changes in policy are reflected in land use and water management and eventually after implementation in the landscape/catchment and its hydrological dynamics. As previously mentioned spatial complexity is a result of socio-economic dynamics and the presence or absence of management actions. But it also has an important temporal factor. Human intervention in water management is very prominent and often policy induced. Often there are unforeseen externalities from technical interventions in water managmenent. This is especially true when we try to manage complex systems and we cannot predict all the (side)effects of the management. The water management that was considered to be good practice in previous decades has sometimes led to unacceptable risks because they were applied at to large scales. For example serious flood risks emerged downstream after impervious areas increased upstream and buffer functions of ditches were removed by replacing them with pipes.

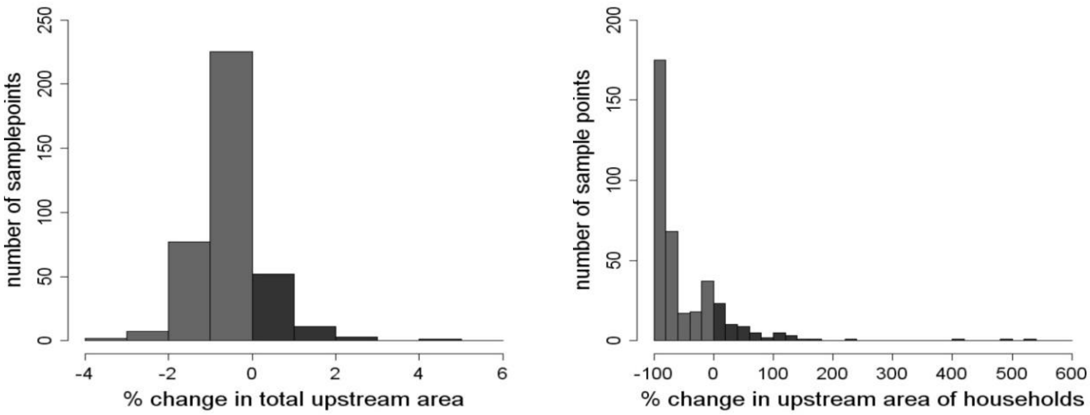


Figure 3: change in total upstream area (left) and connected rooftops (right)

In combination with the straightening of the rivers, common practice for a long time, has led to serious flooding in many downstream areas. As a result policy and management had to adapt to the new situation. Management implementation is often expensive and time consuming. As a result new policy and management practices cannot be implemented everywhere at the same time. One of the more distinct examples to prevent downstream flood risk and improve waste water treatment is the separation of rain water from waste water in urbanized areas and the development of buffer basins. But to implement this rather recent policy the entire sewer system has to be adapted. At the moment only small parts of the sewer system have been adapted. This patchiness of different pipelines, overflows and waste water treatment plant types within a catchment is difficult to incorporate in a catchment scale model.

3. POLICY SUPPORT

Policy support was achieved through contacts with relevant administrations and research institutes and through interdisciplinary workshops during which policy questions were highlighted and debated. The following paragraphs provides conclusions from these meetings and discussions, and formulate recommendations for nature and water management.

Due to substantial uncertainty on the magnitude of regional / local climate change, policy measures need to take a range of possible futures into account, in particular by increasing resilience and looking for "no-regret" options. While the range of possible greenhouse gas emission futures remains large, it remains challenging, but potentially more feasible now, to assess the consequences on ecosystems and infrastructures of excluding low, or high, emission cases. This could at least help in clarifying the potential benefits of mitigation in spite of uncertainties. **There should be continued efforts to monitor precipitation and flood changes, analyze their statistical properties, and compare observations to the models which are also used for future projections to further decrease the uncertainties.**

Actions and measures that are planned and executed today are oriented towards the restoration of natural processes and may counterbalance the « additional » impact of climate change up to a certain level. The largest changes in flood regimes have been induced in the past through normalization of streams, increased run-off and a reduction in floodplain acreage. The Determination of climate impacts through changes in flood regimes on the current vegetation within 50 years is only illustrative as biodiversity has been declining rapidly even without CC. While human activities impact the global climate change in a rather slow and progressive manner (though thresholds may exist), climate change may appear faster at specific times in a given region, due to its combination with natural variability (chapter 2). In addition, especially with embanked floodplains, a small increase in river water levels may result in significant changes in the floodplain water levels due to a sensitive overtopping threshold. The current nature values will also remain subjected to natural evolutions (e.g. succession) and anthropogenic stressors such as eutrophication, dessication and fragmentation. In addition, planned nature development goals and water management actions and measures should be included within such long term assessments.

Floodplain ecosystems may be relatively resilient against gradual changes in the magnitude of the flood regimes (depth-duration) if the floodplain exhibits a wide range of topographical gradients and is subjected to regular flooding. A more « ecological design » of controlled floodplains, can be a strategy to increase biodiversity. This is only valid if we assume that there will be no significant shifts in the temporal distribution of the flood events (winter > summer). Current climate projections still involve substantial uncertainty on the probability and magnitude of changes in the seasonal cycle (more precipitation in winter and less in summer) in mid-latitudes. The occurrence of extreme floods may pose a threat to existing nature values, but also opportunities for rejuvenation of the floodplain succession stages (grassland > reed & shrubs > willow > alder).

Internal and external eutrophication of floodplain ecosystems may be a more determining impact mechanism than drowning (oxygen depletion in the root zone), but depends on site characteristics (e.g. presence of seepage) and water quality. **Floodplain rewetting (reducing drainage) and further water quality improvement can decrease the risk of eutrophication.**

Several hydrological-ecological interfacing problems were identified, which need further focus and research. It is interesting to notice that the **current hydraulic model limitations are a natural result of improvements in flood risk modeling.** Most advances in the field of hydraulic modeling have been achieved in the domain of flood prediction methods for extreme events. Such methods include the use of design hydrographs or composite hydrographs and extreme value analysis (see chapter 5).

These developments have enabled to reduce model processing time (smart algorithms) in order to assess and compare the impact of as many possible scenarios for both climate change scenarios as for flood protection and mitigation strategies (dikes, retention basins). **Especially for ecological impact assessments and evaluation of soft measures (land-use change, infiltration restoration, upstream water retention), the recent advantages have been reducing the applicability, rather than improving the usefulness for these applications.**

Flood hazard mapping based on synthetic events, as traditionally done in flood risk management studies, is only feasible for large return periods (above the flow threshold considered in the statistical extreme value analysis). Contradictory to flood risk studies, estimation of flow regimes for small return periods (frequent events) is – for ecological impact studies – as important (or more important) as for high return periods (see also Chapter 4). **Estimation of changes in the occurence of frequent events requires continuous long-term simulations to be carried out and statistically post-processed also for the less extreme events. Such long-term simulations are feasible for the lumped conceptual rainfall-runoff models but unfeasible for the full hydrodynamic river and floodplain models because of the long computational times of the latter models.**

Given that the statistical analysis on higher extremes is based on extreme value theory (given the limited number of extreme events available in the time series) and is geared towards extrapolation , the analysis for lower extremes requires a different approach (extreme value theory is not applicable below a threshold; but statistical analysis can be done empirically / non-parametrically given the larger number of less extreme events in the series).

Another, more important problem is that the floodplain modeling methodology, as outlined in Chapter 3, is developed for the quantification and mapping of the maximum spatial extent of specific flood events (historical or synthetic, and independent on the flood season). **For the determination of ecological impacts also other variables such as the flood duration, the temporal evolution of the floodplain filling, the flood season, etc., are required. These outputs are by default not provided, neither validated.** Also extraction of information on the flood season requires additional post-processing and validation.

As explained above, one of the hydraulic model outputs which is not provided by default but is nevertheless very important for the ecological impact investigation, is the flood duration. This duration can be modeled with the (quasi-2D) hydraulic flood modeling approach but largely depends on parameters describing the drainage or soil infiltration capacity. These parameters need calibration, **while calibration data on the duration of historical floods is most often not available.** Arbitrary values are therefore often selected. **Consequently, the emptying of the flood plains cannot be modeled truthfully.** More attention thus should be given to the modeling, calibration and validation of flood duration and the underlying processes that affect flood duration. When no direct data on the flood duration are available, indirect validation data could be collected (e.g. based on interviews with local water managers or people living in the neighbourhood of the floodplains).

From the description of the physico-chemical water quality model in Chapter 3, **it became clear that the water quality model is based on a huge number of assumptions, which are all due to lack of sufficient details (temporal frequency, locations) in the available pollution data.** The same comment is valid for the verification data. The available water quality data allows to simulate general trends in the water quality state and related ecological behavior of the system. **Averaged estimated loads and monthly measurements are not sufficient to allow accurate simulation of the daily or hourly concentration variations. It is this daily or hourly timescale that is of importance for an ecological impact analysis.**

In the Grote Nete case study longer term (accumulated) domestic, industrial and agricultural pollution impacts were considered up to now. **The Grote Nete is however also largely influenced by short-duration pollution impacts from Combined Sewer Overflows (CSOs).** The catchment of the Grote Nete is intersected by 7 wastewater treatment zones, of which 5 drain to wastewater treatment zones (WWTPs) outside the catchment's boundaries. These water movements largely affect the flow regimes, mainly during the (extreme) high and low flow periods. Moreover, climate change will have different impacts on river catchment rainfall-runoff and sewer outflows, because of the different response times of river and sewer systems and the different flood seasons (mainly summer for sewer systems; most severe river floods in winter). Reliable assessment of the risk thus requires accurate climate simulation for all seasons and for a range of variables, including for heavy precipitation events associated with summer storms and convection, which is still difficult to simulate due to its small scale. **The CCI-HYDR project has shown that due to climate change in the case study, CSO frequencies tend to increase, as well as the CSO pollutant concentrations due to prolonged dry weather periods during which sediments accumulate (higher storm flush for same CSO discharge; Willems et al., 2010). Also in the river, the same CSO discharge can lead to a higher impact, due to prolonged low flow periods in summer and increased eutrophication.**

The effect of macrophyte growth in the rivers on the river hydrodynamics should be taken into account, especially during low flow periods. During past decades, the growth of macrophytes largely increased along the lowland rivers due to the increased waste water treatment efficiency for suspended solids (improved light penetration allowed re-establishment of macrophytes). **The present excessive macrophyte growth during low flows increases the hydraulic head, decreases valley drainage and results in more stable and higher groundwater levels.** The reduced drainage also increases denitrification because both the area of water saturated soils increases and the residence time of both surface and groundwater increases (Behrendt and Opitz 1999; Burt, Matchett et al. 1999; Seitzinger, Harrison et al. 2006). **The water retention has significant positive effects on water quality and base flow if droughts persist, but may cause problems for harvesting crops in the valleys and may cause floods during summer storms.** These interaction mechanisms need further study. **The incorporation of a variable flow resistance (and macrophyte growth models) in hydraulic models is a huge challenge (research recommendation).**

Climate change should be put in perspective and be linked to the « traditional » environmental stressors (eutrophication, dessication, acidification, soil sealing) which have not been tackled up to now and still cause further changes in the hydrological and ecological status of rivers and floodplains. Many other non-climatic factors determine the hydrological state (increase in paved areas, embankment of rivers, …) and ecological state (pollution, floodplain drainage, invasive species…) of the rivers and floodplains. **Determining the impact of climate change on already heavily impacted ecosystems is rather ambivalent and in that case natural reference situations could be of use.**

Knowledge concerning the relative importance of the processes affecting vegetation during floods (ecohydrological functioning of the floodplains) is rather limited and needs to be extended. **Research projects that would allow integrated monitoring and modelling of several floodplain sites would contribute to a better understanding of the biogeochemical processes, which would allow to derive better evaluation criteria.**

4. PUBLICATIONS

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ANNEX A: Biological Valuation Map to Vegetation Types

	Natuurtypen	Eenheden Biologische Waarderingskaart (potentieel)
1	Gemeenschap van smalle voedselrijke w aterlopen en poelen met Watertorkruid en Zw anebloem	k(mr+) / kt(mr+) / mr+ / mrb+ / / / / / / / / / / / /
2	Vlottende bies, pilvaren, ionenrijk w atertype	ao / ao- / ao+ / aom / aom- / aom+ / k(ao) / k(ao-) / k(ao+) / / / / / / /
3	Eutrofe plas met slibrijke bodem	ae / ae- / ae+ / aer / aer- / aer+ / aev / aev- / aev+ / / / / / / /
4	Grote Zeggegemeenschap met Scherpe Zegge en Oeverzegge	k(mc) / k(mc-) / k(mc+) / kt(mc) / kt(mc-) / kt(mc+) / mc / mc- / mc+ / / / / / / /
5	Waterlelie-gele plomp ionenrijk w atertype	ap / ap- / ap+ / apo / apo- / apo+ / app / app- / app+ / / / / / / /
6	Hoornblad-Watergentiaan ionenrijk w atertype	ap / ap- / ap+ / apo / apo- / apo+ / app / app- / app+ / / / / / / /
7	Rietmoerassen	k(mr) / k(mr-) / kt(mr) / kt(mr-) / mr / mr- / mrb / mrb- / / / / / / / /
8	Rietvegetatie met Haagw inde als constante soort	k(mr+) / kt(mr+) / mr+ / mrb+ / k(mru) / k(mru-) / k(mru+) / mru / mru- / mru+ / / / / / /
9	Struw elen met smalbladige w ilgen langs snelstromende rivieren	sf / sf- / sf+ / / / / / / / / / / / / /
10	Natte ruigten van het Moerasspirea-verbond (Filipendulion)	hf / hf- / hf+ / hfb / hfb- / hfb+ / hfc / hfc- / hfc+ / hft / hft- / hft+ / k(hf) / k(hf-) / k(hf+) / k(hfc)
11	Natte ruigten van het verbond van Harig w ilgeroosje (Epilobion hirsuti)	hf / hf- / hf+ / hfb / hfb- / hfb+ / k(hf) / k(hf-) / k(hf+) / kt(hf) / kt(hf-) / kt(hf+) / hr / hr- / hr+ / hrb
12	Zilverschoonverbond	hpr+ / / / / / / / / / / / / / / /
13	Wilgenstruw eel met breedbladige w ilgen in laagdynamisch milieu/geoorde w ilg	sf / sf- / sf+ / so / so- / so+ / sal / / / / / / / / /
14	Voedselarme vengemeenschappen met draadzegge	k(ms+) / kt(ms+) / ms+ / k(ms) / k(ms-) / kt(ms) / kt(ms-) / ms / ms- / / / / / / /
15	Verbond van grote vossenstaart	hf / hf- / hf+ / hfb / hfb- / hfb+ / k(hf) / k(hf-) / k(hf+) / kt(hf) / kt(hf-) / kt(hf+) / hp+ / hpr / hpr- / hu+
16	*associatie van boterbloemen en waterkruiskruid*	/ / / / / / / / / / / / / / /
17	*associatie van gewone engelwortel en moeraszegge*	/ / / / / / / / / / / / / / /
18	Pitrus-w olfspoot ionenarm type = *Rompgemeenschap*	hj / hj- / hj+ / hjb / hjb- / hjb+ / k(hj) / k(hj-) / k(hj+) / kt(hj) / kt(hj-) / kt(hj+) / aoo / aoo- / aoo+ / ao
19	Veenmos-snavelzegge ionenarm w atertype	aoo / aoo- / aoo+ / ao / ao- / ao+ / k(ao) / k(ao-) / k(ao+) / / / / / / /
20	Knolrus-veenmos ionenarm type = *Rompgemeenschap*	aoo / aoo- / aoo+ / ao / ao- / ao+ / k(ao) / k(ao-) / k(ao+) / / / / / / /
21	Drijvende w aterbeegbree oeverkruid ionen arm type	ao / ao- / ao+ / k(ao) / k(ao-) / k(ao+) / aom / aom- / aom+ / / / / / / /
22	Hoogveenslenken - verarmde gemeenschap Molinia	t / t+ / tm / tm- / tm+ / cm / cm- / cm+ / cmb / cmb- / cmb+ / cp / cp- / cp+ / cpb / cpb-
23	Basenrijke laagvenen en duinvalleinen met Parnassia, dw ergzegge en tw eehuizige zegge	mk / mk- / mk+ / mp / mp- / mp+ / / / / / / / / / /
24	Droog Heischraal grasland	hn / hn- / hn+ / hnb / hnb- / hnb+ / k(hn) / k(hn-) / k(hn+) / kt(hn) / kt(hn-) / kt(hn+) / / / /
25	Witbolgraslanden	hp / / / / / / / / / / / / / / /
26	Kamgrasland	hp+ / k(hp+) / kt(hp+) / hpr+ / / / / / / / / / / / /
27	Gagelstruw eel	sm / sm- / sm+ / / / / / / / / / / / / /
28	Overgang blauw grasl en dotterverbond	hme / hme- / hme+ / / / / / / / / / / / / /
29	*veldrus-associatie*	/ / / / / / / / / / / / / / /
30	*Verbond der droge stroomdalgraslanden*	/ / / / / / / / / / / / / / /
31	Natte heide met gew one dophei	ce / ce- / ce+ / ceb / ceb- / ceb+ / k(ce) / k(ce-) / k(ce+) / kt(ce) / kt(ce-) / kt(ce+) / / / /
32	Vochtige venige graslanden met biezenknoppen en pijpenstrootje	hm / hm- / hm+ / hmm / hmm- / hmm+ / hmo / hmo- / hmo+ / k(hm) / k(hm-) / k(hm+) / kt(hm) / kt(hm-) / kt(hm+) /
33	Natte heide met hoogveen elementen - Hoogveen	ces / ces- / ces+ / / / / / / / / / / / / /
34	Hoogveenslenken - met Witte snavelbies en Slank veenmos	ce / ce- / ce+ / ceb / ceb- / ceb+ / k(ce) / k(ce-) / k(ce+) / kt(ce) / kt(ce-) / kt(ce+) / / / /
35	Het verbond van Look-zonder-look (Galio-Alliarion)	hr / hr- / k(hr) / k(hr-) / kt(hr) / kt(hr-) / / / / / / / / / /
36	Doornstruw elen met eenstijlige meidoorn en sleedoorn	kh(sp) / kh(sp-) / kh(sp+) / kt(sp) / kt(sp-) / kt(sp+) / sp / sp- / sp+ / / / / / / /
37	*associatie van echte koekoeksbloem en gevleugeld hertshooi*	/ / / / / / / / / / / / / / /
38	Droge Heide met Pijpenstrootje	cm / cm- / cm+ / cmb / cmb- / cmb+ / cp / cp- / cp+ / cpb / cpb- / cpb+ / k(cm) / k(cm-) / k(cm+) / kt(cm)
39	Glanshaververbond	hu / hu- / hub / hub- / k(hu) / k(hu-) / kt(hu) / kt(hu-) / / / / / / / /
40	Drijftillen, sloten en oevers met Hoge Cyperzegge en w aterscheerling	k(mc) / k(mc-) / k(mc+) / kt(mc) / kt(mc-) / kt(mc+) / mc / mc- / mc+ / k(ms) / k(ms-) / kt(ms) / kt(ms-) / ms / ms- / m
41	Zuur laagveen met w ateraardbei en zw arte zegge	k(ms) / k(ms-) / kt(ms) / kt(ms-) / ms / ms- / / / / / / / / / /
42	*associatie van harlekijn en ratelaar*	/ / / / / / / / / / / / / / /
43	Verlandingsgemeenschap met Pluimzegge	k(mc) / k(mc-) / k(mc+) / kt(mc) / kt(mc-) / kt(mc+) / mc / mc- / mc+ / mm / mm- / mm+ / / / /
44	Braamstruw eel	kh(sp) / kh(sp-) / kh(sp+) / kt(sp) / kt(sp-) / kt(sp+) / sp / sp- / sp+ / / / / / / /
45	Het Marjolein-verbond (Trifolium medii)	hu+ / hub+ / k(hu+) / kt(hu+) / hu / hu- / hub / hub- / k(hu) / k(hu-) / kt(hu) / kt(hu-) / / / /
46	Droge heide met Bochtige smele	cd / cd- / cd+ / cdb / cdb- / cdb+ / k(cd) / k(cd-) / k(cd+) / kt(cd) / kt(cd-) / kt(cd+) / / / /
47	Kalkgrasland (Xerobromion, Mesobromion)	hk / hk- / hk+ / hkb / hkb- / hkb+ / k(hk) / k(hk-) / k(hk+) / kt(hk) / kt(hk-) / kt(hk+) / / / /
48	Buntgrasverbond	ha / ha- / ha+ / hab / hab- / hab+ / k(ha) / k(ha-) / k(ha+) / kt(ha) / kt(ha-) / kt(ha+) / / / /
49	Droge Heide met Struikhei (Calluna vulgaris)	cg / cg- / cg+ / cgb / cgb- / cgb+ / cv / cv- / cv+ / k(cg) / k(cg-) / k(cg+) / kt(cg) / kt(cg-) / kt(cg+) /
50	Dw erghaververbond	ha / ha- / ha+ / hab / hab- / hab+ / k(ha) / k(ha-) / k(ha+) / kt(ha) / kt(ha-) / kt(ha+) / / / /
51	Het verbond van gladde w itbol en havikskruiden (Melampyrion pratensis)	ha / ha- / ha+ / hab / hab- / hab+ / k(ha) / k(ha-) / k(ha+) / kt(ha) / kt(ha-) / kt(ha+) / / / /
52	Verbond van gew oon struisgras	ha / ha- / ha+ / hab / hab- / hab+ / k(ha) / k(ha-) / k(ha+) / kt(ha) / kt(ha-) / kt(ha+) / / / /
53	Bremstruw eel	kh(sg) / kh(sg-) / kh(sg+) / kt(sg) / kt(sg-) / kt(sg+) / kt(sgb) / kt(sgb-) / kt(sgb+) / sg / sg- / sg+ / sgb / sgb- / sgb+ / sgu
54	dotterbloemhooiland	hc / hc- / hc+ / k(hc) / k(hc-) / k(hc+) / kt(hc) / kt(hc-) / kt(hc+) / / / / / / /
55	schorren	da / da- / da+ / / / / / / / / / / / / /
56	duinen	dd / dd- / dd+ / dm / dm- / dm+ / / / / / / / / / /
57	slik	ds / ds- / ds+ / / / / / / / / / / / / /
58	oligotrofe sloten	k(ae) / k(ae-) / k(ae+) / / / / / / / / / / / / /
59	Voedselarme droge bostypes (Quercion-bossen)	kh(qb) / kh(qb-) / kh(qb+) / kt(qb) / kt(qb-) / kt(qb+) / qb / qb- / qb+ / fs / fs- / fs+ / kh(fs) / kh(fs-) / kh(fs+) / kt(fs)
60	Eutrofe en basicliene broekbossen en ruigten	k(mru) / k(mru-) / k(mru+) / mru / mru- / mru+ / kh(vn) / kh(vn-) / kh(vn+) / kt(vn) / kt(vn-) / kt(vn+) / vn / vn- / vn+ / ru
61	eiken of beukenbossen met w itte veldbies	kh(fl) / kh(fl-) / kh(fl+) / kt(fl) / kt(fl-) / kt(fl+) / kt(ql) / kt(ql-) / kt(ql+) / ql / ql- / ql+ / fl / fl- / fl+ /
62	Subatlantisch eikenmengbos, zure, arme variant :	kh(qa) / kh(qa-) / kh(qa+) / kt(qa) / kt(qa-) / kt(qa+) / qa / qa- / qa+ / fa / fa- / fa+ / kh(fa) / kh(fa-) / kh(fa+) / kt(fa)
63	Parelgras-beukenbos Melico-Fagetum	kjh(fm) / kjh(fm-) / kjh(fm+) / kt(fm) / kt(fm-) / kt(fm+) / fm / fm- / fm+ / / / / / / /
64		
65	Gierstgras-Beukenbos Milio-Fagetum	fe / fe- / fe+ / kh(fe) / kh(fe-) / kh(fe+) / kt(fe) / kt(fe-) / kt(fe+) / / / / / / /
66	Mesotrofe en basicliene droge bostypes (Fagion- en Carpinion-types)	fe / fe- / fe+ / qe / qe- / qe+ / kh(fe) / kh(fe-) / kh(fe+) / kt(fe) / kt(fe-) / kt(fe+) / kh(qe) / kh(qe-) / kh(qe+) / kt(qe)
67	Het voedselrijk Subatlantische eikenmengbos : Primulo-Carpinetum	qe / qe- / qe+ / kh(qe) / kh(qe-) / kh(qe+) / kt(qe) / kt(qe-) / kt(qe+) / / / / / / /
68	populier op natte grond	l / lh / lhb / lhb- / lhb+ / lhi / lhi- / lhi+ / / / / / / / /
69	populier op droge grond	ls / lsb / lsb- / lsb+ / lsh / lsh- / lsh+ / lsi / lsi- / lsi+ / n / n- / n+ / pop / potr /
70	basisch eiken of beukenbos	qk / qk- / qk+ / / / / / / / / / / / / /
71	Mesotroof elzenbroek Carici elongatae-Alnetum	kh(vm) / kh(vm-) / kh(vm+) / kt(vm) / kt(vm-) / kt(vm+) / vm / vm- / vm+ / / / / / / /
72	nitrofiel Essen - Elzenbos	kh(vn) / kh(vn-) / kh(vn+) / kt(vn) / kt(vn-) / kt(vn+) / vn / vn- / vn+ / / / / / / /
73	gemengd loofbos	gml / / / / / / / / / / / / / / /
74	Essenbronbos Carici remotae-Fraxinetum	kh(vc) / kh(vc-) / kh(vc+) / vc / vc- / vc+ / / / / / / / / / /
75	Alluviaal bos van de grote rivieren (Essen-Olmenbos Ulmo-Fraxinetum en Abelen-Iepenbos Violo odoratae-Ulmetum)	kh(vn) / kh(vn-) / kh(vn+) / kt(vn) / kt(vn-) / kt(vn+) / vn / vn- / vn+ / ru / ru- / ru+ / v / / /
76	Elzen-Essenbos met Slanke Sleutelbloem Primulo-Fraxinetum excelsioris	kh(va) / kh(va-) / kh(va+) / kt(va) / kt(va-) / kt(va+) / va / va- / va+ / / / / / / /
77	vochtig elzen-eikenbos	kh(vf) / kh(vf-) / kh(vf+) / vf / vf- / vf+ / / / / / / / / / /
78	Permanent zachthout-ooibos Salicetum triandro-viminalis	sf / sf- / sf+ / / / / / / / / / / / / /
79	naaldbos	p / pa / pa- / pa+ / pi / pi- / pi+ / pica / pics / pinn / pins / pint / pmb / pmb- / pmb+ / pmh
80	Oligotroof elzenbroek Carici laevigatae-Alnetum	v / vo / vo- / vo+ / vt / vt- / vt+ / / / / / / / / /
81	Esdoorn-Abelenbos met Veldhondstong en Witte w interpostelein	rud / qd / / / / / / / / / / / / / /
82	Permanent zachthout-ooibos Salicetum triandro-viminalis	sf / sf- / sf+ / / / / / / / / / / / / /

ANNEX B: Flood vulnerability matrix for Vegetation Types