GOCE 036822

SCENES

Water Scenarios for Europe and for Neighbouring States

Instrument: Integrated Project

Thematic priority: Global change and ecosystems

D4.6
Socio-economic and ecological impacts of future changes in Europe's freshwater resources

Volume C: Water for Nature

Due date of deliverable: 31.03.2011
Actual submission date: 29.04.2011

Start date of project: 1.11.2006 Duration: 54 months

Lead contractor: Deltares Revision version 1

Project co-funded by the European Commission within the Sixth Framework Programme (2002-2006)		
Dissemination Level		
PU	Public	X
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	

This report is a result of contributions by the following persons (in alphabetical order by last name):

Name	**Institute**
Mike Acreman	Centre for Ecology and Hydrology, UK:
Martina Flörke	Centre for Environmental Systems Research, Germany
Helen Houghton-Carr	Centre for Ecology and Hydrology, UK
Cedric Laizé	Centre for Ecology and Hydrology, UK
Sibren Loos	Deltares, The Netherlands
Tomasz Okruszko	Warsaw University of Life Sciences, Poland
Christof Schneider	Centre for Environmental Systems Research, Germany

Houghton-Carr, H. (ed.), 2011. Socio-economic and ecological impacts of future changes in Europe's freshwater resources. SCENES project. D4.6: Volume C: Water for Nature. Centre for Ecology and Hydrology, Wallingford, United Kingdom.

Preface

SCENES is a four year European research project developing scenarios for the changes in the quantity and quality of fresh water resources in pan-Europe due to climate change, land use change and socio-economic development. The water scenarios are developed based on the SAS-approach that combines storylines with simulations. The storylines are developed by a Pan-European Panel (PEP). This report describes impacts of future changes in Europe's freshwater resources in terms of indicators for 'Water for Nature'.

This report is deliverable D4.6 of the FP6 Project SCENES (EU contract GOCE 036822).

Contents

1 Introduction

SCENES impact indicators

This report is an appendix to deliverable D4.6 of the SCENES Project. Deliverable D4.6 is reporting the results of an analysis of the socio-economic and ecological impacts of future changes in Europe's freshwater resources. In the SCENES project water scenarios have been developed describing possible future climate and socio-economic developments and the impacts of these scenarios. The impacts are expressed through a set of indicators covering a wide range of topics.

Within SCENES, we distinguish two types of impact indicators:

- Generic hydrological impact indicators: indicators that are addressing the hydrological changes in freshwater availability and quality in terms of too much (flood events) or too little (drought events, water stress).
- Impact indicators for water system services: indicators that are addressing the environmental, ecological and socio-economical consequences of changes in the state of fresh water resources on water system services: Water for Food, Water for Nature, Water for People and Water for Industry and Energy.

The total set of impact indicators is listed in Table 1.1. The indicator ID's refer to water system services. The generic hydrological indicators have "Water" as ID.

Table 1.1 Overview of SCENES impact indicators

ID	Name
Water 1	Water Consumption Index
Water 2	Water Stress Index
Water 3	Water Scarcity Index
Water 4	Change in frequency of flood events
Water 5	Change in flood hazards
Water 6	Change in frequency of river low flow
Water 7	Change in magnitude of river low flow
Water 8	Change in mean annual river flow
Food 1	Agricultural crop production
Food 2	Irrigation water withdrawals
Food 3	Water stress in irrigation
Nature 1	Environmental flows
Nature 2	Floodplain wetlands
Nature 3	Ecosystem services of wetlands
Nature 4	Change in water supply to wetlands
Nature 5	Aquatic macrophyte diversity in lakes
Nature 6	Habitat suitability for river water temperature for fish
People 1	Domestic water stress
People 2	Flood risk
People 3	Risk for harmful algal blooms in shallow lakes and reservoirs
People 4	Domestic water availability
Industry 1	Extra demand for cooling water
Industry 2	Navigability of large rivers
Industry 3	Cooling water stress

SCENES scenarios and indicator quantification

For quantification of future scenarios, four socio-economic scenarios are combined with two climate change scenarios. The socio-economic scenarios are based on UNEP's GEO4 scenarios and adjusted in a participatory exercise with key European scientists. Four scenarios resulted which are called: Economy First (EcF), Fortress Europe (FoE), Policy Rules (PoR), and Sustainability Eventually (SuE). Two climate scenarios are used which were generated by two different global circulation models (GCM's): MIMR and IPCM4, following the SRES A2 emission pathway. The reference period (2000s) is represented by the climate normal period (1961-1990) for river discharges and considers the water uses of the year 2005 (except for irrigation for which demand is influenced by the variation in evaporation and precipitation).

These eight scenarios have been used as input for the global water model WaterGAP (Water – Global Assessment and Prognosis; Alcamo et al. 2003; Döll et al. 2003). The resulting output for a baseline (2000s) and eight future (2050s) situations has formed the basis for the quantification of the indicators.

This report

The indicators are discussed in detail in five Appendices:

- Volume A: Generic indicators
- Volume B: Water for Food
- Volume C: Water for Nature (this volume)
- Volume D: Water for People
- Volume E: Water for Industry & Energy

This report, Volume C, discusses the Water for Nature indicators. Each indicator chapter starts with an introduction to the indicator, followed by the method that was used to calculate the indicator. Next, the results are described. Each chapter ends with a synthesis and the most important key messages that could be derived from the analysis.

Chapter 8 of this volume discusses the key findings that can be drawn from the analysis of the nature indicators.

The method applied to analyse the regional variations in impacts as well as to assess whether climate change or socio-economic development is the more dominant driving force for changes in the indicator, used in chapter 8, is discussed in chapter 2 of Volume A.

Chapter 3 of Volume A provides an overview of the results for main input data used for the computation of the indicators, consisting of either input for or output from WaterGAP.

References

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T. & Siebert, S., 2003. Development and Testing of the WaterGAP 2 Global Model of Water Use and Availability, *Hydrological Sciences Journal,* 48 (3): 317–337.

Döll, P., Kaspar, F. & Lehner, B., 2003. "A Global Hydrological Model for Deriving Water Availability Indicators: Model Tuning and Validation", *J. Hydrol.*, 270, pp. 105-134.

2 Water for Nature 1 – Environmental flows

2.1 Introduction

Various factors determine the health of a river ecosystem (Moss, 2010; Norris and Thoms, 1999), including light, temperature, nutrient levels, water discharge, channel structure, physical barriers to connectivity, species interactions and the level of management, such as macrophyte cutting and dredging, fishing and stocking. Many of these factors are not independent; for example, discharge, channel structure and macrophyte growth interact to determine water depth and velocity, which in turn influence food delivery, light penetration and oxygen levels. Discharge (flow, measured in units of volume ÷ time) is a key variable, which changes naturally through time. Various authors have suggested that all elements of the flow regime influence freshwater ecosystems, including floods, average and low flows (Junk *et al.,* 1989; Richter *et al.,* 1996; Poff *et al*., 1997; Biggs *et al*., 2005; Arthington *et al.*, 2006; Kennen *et al*., 2007). In many rivers, discharge is heavily influenced by anthropogenic activities, such as water abstraction, storage in reservoirs and effluent returns associated with public supply, agriculture and industry. The Millennium Ecosystem Assessment (2005) showed that many ecosystems were being degraded or lost, with aquatic systems suffering particularly from the withdrawal of water for direct human needs, many impacts directly resulting from fragmentation by dams (Nilsson *et al*., 2005). Thus, there is a pressing need to assess the degree of alteration of discharge to determine likely impacts on river ecosystems. The development of environmental flow regimes for rivers and associated systems is receiving increasing attention (Poff *et al*. 2010, Dyson *et al*. 2003). One approach to defining an environmental flow regime is to base it on an acceptable departure of the flow regime from a baseline. Normally the baseline is the natural flow regime and any departure signifies a degradation of the river ecosystem. One key area of current research is to envisage future impacts of climate change, rising populations, varying global markets and government policies on river ecosystems through alterations to the hydrological regime. This paper reports the results of research undertaken to assess hydro-ecological response(s) under future scenarios for Europe.

2.2 Method

Calculation approach

It is based conceptually on the Range of Variability Approach (RVA) using Indicators of Hydrological Alteration (IHA), a desk-top technique for defining environmental flow requirements introduced by Richter *et al*. (1996, 1997). The IHA/RVA recognises that all characteristics of the flow regime (e.g. low and high flows and flood events) and their magnitude, duration, timing, frequency and rate of change are all ecologically important. First, the hydrological regime prior to an impact, whether due to, for example, the building of a structure, an abstraction point or climate change, is described by the IHA and constitutes the baseline against which post-impact conditions are assessed. The underlying assumption is that, if an ecosystem exists under the baseline conditions, then any departure from the baseline beyond some admissible thresholds will affect the ecosystem significantly.

Details of the SCENES methodology development can be found in deliverables D4.3 and D4.4, and in Laize et al (2010). Flow regimes are characterised by nine monthly time-step parameters (second column in Table 2.1). From the parameters (one value per year of record per site), indicators (one value per period of record per site) were derived in order to capture parameter magnitude and variability.

Percentiles (i.e. 50th percentile to describe magnitude, and span between 25th and 75th percentiles to describe variability) were chosen because: (i) percentiles are less sensitive to outliers than mean and standard deviation; (ii) parameters are not necessarily normally-distributed, hence, percentiles would better describe skewed distributions. An exception was made for flood and minimum flow timing parameters. Indeed, these parameters are the months (i.e. integers ranging from 1 to 12) when flood and low flow events happen and are best summarised over the period of record by their mode. Consequently, there are 16 indicators (third column in Table 2.1). All 16 indicators are computed for the baseline data and for all considered scenarios. Departure from the baseline can be due to any combination of change in magnitude (shift in 50th percentile) and/or variability (shorter or longer 25th-75th percentile span).

Table 2.1 Environmental flow indicators

Regime characteristic	*Parameter monthly (one value per year)*	*Indicator (one value per record)*
Flood Magnitude & Frequency	Number of times that monthly flow exceeds threshold (all-data naturalised Q5 from 1961-1990)	50th Percentile (magnitude) Span 25th-75th Percentiles (variability)
Flood Timing	Month (as number Jan=1, Dec=12) of maximum flow	Mode of month
Seasonal Flow	January flow (mm runoff)	50th Percentile (magnitude) Span 25th-75th Percentiles (variability)
	April flow (mm runoff)	Idem
	July flow (mm runoff)	Idem
	October flow (mm runoff)	Idem
Low Flow Magnitude & Frequency	Number of months that flow is less than threshold (thresholds = all-data naturalised Q95 from 1961-1990)	Idem
Minimum Flow Timing	Month (as number Jan=1, Dec=12) of minimum flow	Mode of month
Low Flow Duration	Number of times that two consecutive months are less than threshold (all-data naturalised Q95 from 1961-1990)	50th Percentile (magnitude) Span 25th-75th Percentiles (variability)

Input data

Modelled monthly river flows ($m^3 s^{-1}$) were generated for more than 35,000 cells (0.5^o latitude x 0.5^o longitude) from the WaterGAP model. They cover major rivers and their tributaries (very small catchments were excluded). Series were generated for nine different model runs corresponding to different climate models and socio-economic scenarios. The first run is of naturalised flows for the standard period 1961-1990 based on climate data from the Climate Research Unit (University of East Anglia, UK); this is used as the baseline describing the current situation excluding impacts of dam management and consumptive water use. The eight other runs are for the 2040-2069 time period ('2050s') under combinations of two climate models and four socio-economic scenarios. The climate models are: IPCM4 (GCM IPSL-CM4, Institut Pierre Simon Laplace, France) and MIMR (GCM MICRO3.2, Center for Climate System Research, University of Tokyo, Japan) both using the SRES A2 emission scenario.

The four socio-economic scenarios are Economy First (EcF), Policy Rules (PoR), Fortress Europe (FoE), and Sustainability Eventually (SuE). To allow comparison between catchments of different sizes, flow data were converted to runoff (mm).

Thresholds and critical values

Based on common expert knowledge (e.g. WFD flow thresholds; Acreman *et al*., 2008), for a given parameter, scenarios are therefore considered not significantly different from the baseline if the total indicator difference is within 30% with the exception of the mode indicators (flood timing, minimum flow timing) for which a threshold of 1 month was retained. For practicality and ease of display and interpretation, differences are aggregated via a colour coding scheme: a site is assigned blue, green, amber, or red when its number of parameters different from baseline are 0 (i.e. no impact), 1-5 (low impact), 6-10 (medium impact), and 11-16 (high impact), respectively.

Validation

We make direct use of WaterGAP output, which has already been validated.

Uncertainty and sensitivity

The overall efficiency of the underlying WaterGAP model was assessd using a subset of cells (Laize, et al, 2010). Given the pan-European scale of the model, the overall efficiency was considered acceptable, especially as this study focuses on the relative changes in flows rather their absolute magnitudes.

2.3 Results

2.3.1 Baseline scenario

Not applicable given the methodology, which considers the differences between the baseline and the various scenarios.

2.3.2 Future scenarios

See Figures 2.1-2.8.

General pattern

Most rivers are impacted. Table 2.2 summarises how many percents of the cells used (ie >35,000) fall in which impact level category; regardless of scenario, more than half of the cells are medium impact, and roughly 15-20% high impact meaning about two thirds of the cells have at least medium impacts. The picture is very consistent within as well as between climate models.

Table 2.2 *Distribution of impact levels per runs (% of cells)*

		None	Low	Medium	High
IPCM4	*Natural*	5	28	51	15
	EcF	5	21	53	21
	FoE	5	21	54	20
	PoR	5	22	54	19
	SuE	5	23	54	19
MIMR	*Natural*	5	29	53	13
	EcF	5	27	53	16
	FoE	5	27	53	15
	PoR	5	28	53	14
	SuE	5	29	53	14

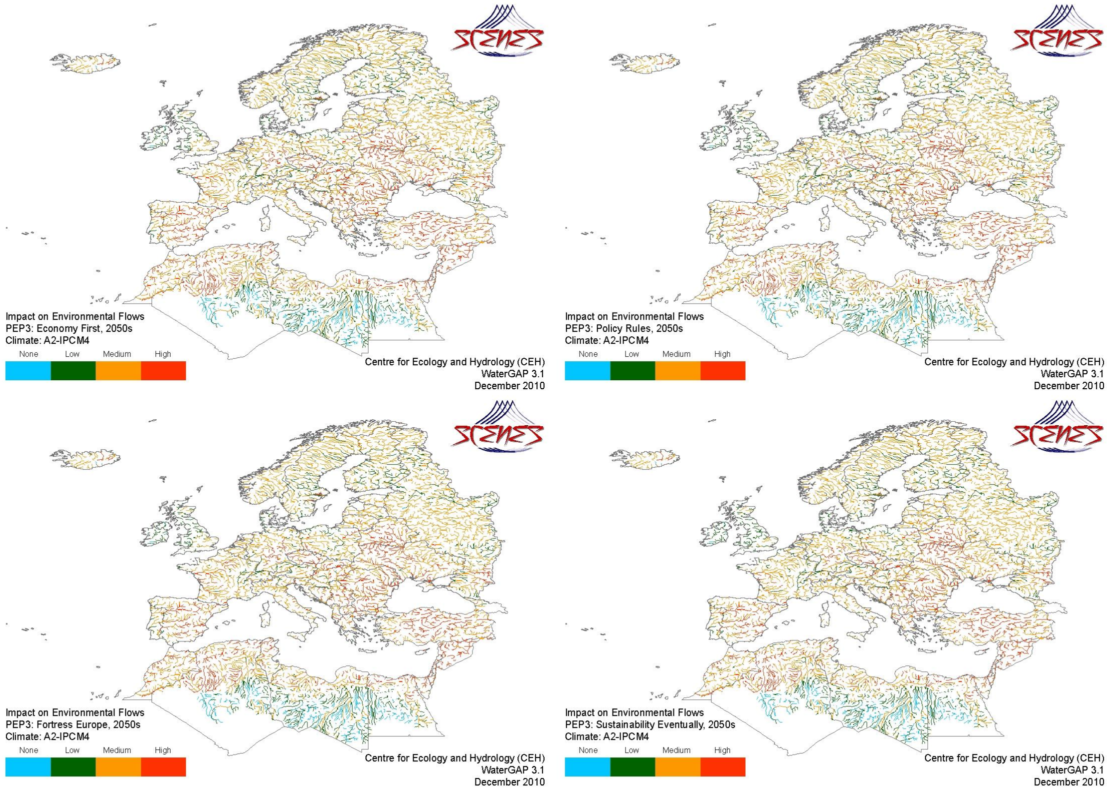

Figure 2.1 until 2.4 (left to right). Impact on environmental flows under the IPCM scenario. Economy First: Figure 2.1. Policy Rules: Figure 2.2. Fortress Europe: Figure 2.3. Sustainability eventually: Figure 2.4.

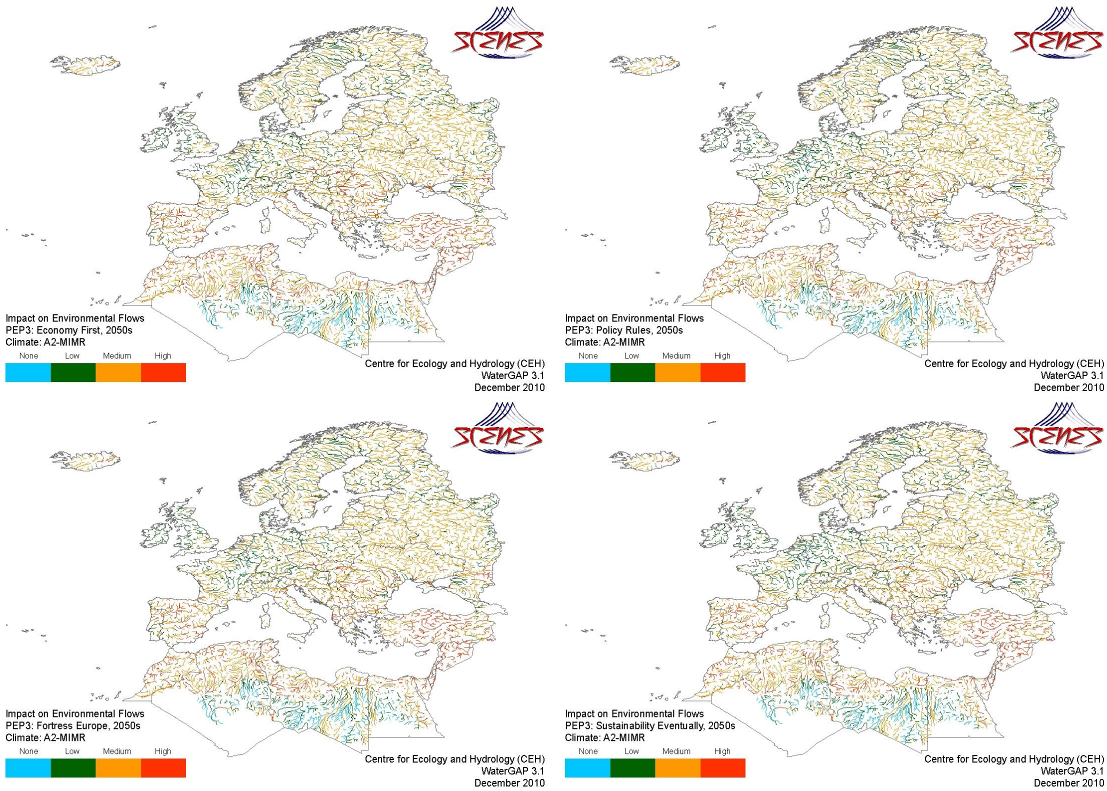

Figure 2.5 until 2.8 (left to right). Impact on environmental flows under the MIMR scenario. Economy First: Figure 2.5. Policy Rules: Figure 2.6. Fortress Europe: Figure 2.7. Sustainability eventually: Figure 2.8.

Socio economic and climate scenarios

From Table 2.2, the main difference between climate models is that IPCM4 runs have slightly more high impact cells while MIMR ones have slightly more low impact cells. Although the total numbers of cells in each impact level category is very similar, the scenarios introduce some significant variation, at the local scale, in terms of where the impacts occur.

Table 2.3 summarises how many percents of the cells get different impact levels when comparing runs against each others. First, climate is the primary driver; socio-economic scenarios only cause differences of around 20% under IPCM4 and up to 10% under MIMR. MIMIR runs are generally about third different from IPCM4 ones. Looking the the maps, this translates in particular with some parts of the UK, France, Germany, etc. being less impacted under MIMR runs than IPCM4 ones. Second, the socio-economic scenarios are rather similar; differences between them range from 4 to 9% under both climate scenarios. Yet, given the large scale of the study and the underlying WaterGAP model, even a 1% variation represents several hundred km of river.

Table 2.3 Summary of differences in impact levels between all runs (% of different cells)

		IPCM4					MIMR			
		Natural	*EcF*	*FoE*	*PoR*	*SuE*	*Natural*	*EcF*	*FoE*	*PoR*
IPCM4	*EcF*	21								
	FoE	20	5							
	PoR	18	7	5						
	SuE	17	9	6	4					
MIMR	*Natural*	35	37	36	35	34				
	EcF	37	34	33	33	32	10			
	FoE	36	35	34	33	33	8	5		
	PoR	36	37	36	34	34	5	8	5	
	SuE	36	37	36	35	34	3	9	7	4

2.4 Synthesis

Climate is the primary driver, setting the broad patterns at the pan-European scale. The socio-economic scenarios are secondary drivers that can introduce some variation at the more local scale. Under all projections, most rivers have medium to high impacts. For a summary of observed changes in all regions see Table 2.4.

Table 2.4 Regional observations on changes with respect to the baseline scenario

		Northern Africa	Western Europe	Northern Europe	Southern Europe	Central Eastern Europe	Eastern Europe	Western Asia
IPCM	EcF	-	-	-	--	-	-	--
	FoE	-	-	-	--	-	-	--
	PoR	-	-	-	--	-	-	--
	SuE	-	-	-	--	-	-	--
MIMR	EcF	-	-	-	--	0	-	--
	FoE	-	-	-	--	0	-	--
	PoR	-	0	-	--	0	-	--
	SuE	-	0	-	--	0	-	--

2.5 References

Acreman, M.C., Dunbar, M.J., Hannaford, J., Wood, P.J., Holmes, N.J., Cowx, I., Noble, R., Mountford, J.O., King, J., Black, A., Extence, C., Crookall, D. and Aldrick, J. 2008. Developing environmental standards for abstractions from UK rivers to implement the Water Framework Directive. *Hydrol. Sci. J.-J. Sci. Hydrol*., **53** (6), 1105-20.

Arthington A.H., Bunn S.E., Poff N.L. & Naiman R.J. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. *Ecol. Appl.,* **16**, 1311-1318.

Biggs, B.J.F., Nikora, V., Snelder, T. 2005. Linking scales of flow variability to lotic ecosystem structure and function. *River Res. Appl.,* **21**, 283-298.

Dyson, M., Bergkamp, G. and Scanlon, J. (eds.), 2003. *Flow: essentials of environmental flows*. IUCN, Gland, Switzerland and Cambridge, UK.

Junk, W.J., Bayley, P.B. and Sparks R.E. 1989. The flood pulse concept in river-floodplain systems. *Can. J. Fisheries Aquat. Sci.,* **106**, 110-127.

Kennen J.G., Henriksen J. A. and Nieswand S. 2007. *Development of the hydroecological integrity assessment process for determining environmental flows for New Jersey streams. US Geological Survey Scientific Investigations Report 2007-5206*. http://pubs.er.usgs.gov/usgspubs/sir/sir20075206.

Laize, Cedric L.R.; Acreman, M.; Dunbar, M.; Houghton-Carr, H.; Flörke, M.; Schneider, C. 2010 Monthly hydrological indicators to assess impact of change on river ecosystems at the pan-European scale: preliminary results. In: *British Hydrological Society Third International Symposium Role of Hydrology in Managing Consequences of a Changing Global Environment, Newcastle upon Tyne, United Kingdom, 19-23 July 2010*.

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington, DC.

Moss, B. 2010. *Ecology of freshwaters - a view for the twenty-first century*. Wiley-Blackwell, Chichester

Nilsson C., Reidy C., Dynesius & M., Revenga C. 2005. Fragmentation and flow regulation of the world's large river systems. *Science*, **308**, 405-408.

Norris, R.H. and Thoms, M.C. 1999. What is river health? *Freshwater Biol.,* **41**, 197-209.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. and Stromberg, J. C. 1997. The natural flow regime. *Bioscience* **47**, 769-784.

Poff N. L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M.C., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keeffe, J.H., Olden, J.D., Rogers, K., Tharme, R.E. and Warner, A. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards, *Freshwater Biol.* **55**: 147-170. *doi:10.1111/j.1365-2427.2009.02204x*

Richter, B.D., Baumgartner, J.V., Powell, J. and Braun, D.P. 1996. A Method for Assessing Hydrologic Alteration within Ecosystems. *Conserv. Biol*., **10** (4), 1163-1174.

Richter, B.D., Baumgartner, J.V., Wigington, R. and Braun, D.P. 1997. How much water does a river need? *Freshwater Biol.*, **37** (1), 231-249.

3 Water for Nature 2 – Floodplain wetlands

The future of European floodplain wetlands under a changing climate

Christof Schneider[a,*], Martina Flörke[a], Gertjan Geerling[b], Harm Duel[b], Mateusz Grygoruk[c], Tomasz Okruszko[c]
[a]Center for Environmental Systems Research, University of Kassel, Kurt-Wolters-Straße 3, D-34109 Kassel, Germany
[b]Deltares, P.O. Box 85467, 3508 AL Utrecht, The Netherlands
[c]Department of Hydraulic Engineering and Environmental Restoration, Warsaw University of Life Sciences, ul. Nowoursynowska 159, 02-776 Warsaw, Poland
[*]Corresponding author. Tel.: +49 561 804 3913. Email address: schneider@usf.uni-kassel.de

3.1 Abstract
In the future, climate change may severely alter flood patterns over large regional scales. Consequently, besides other anthropogenic factors, climate change depicts a potential threat to river ecosystems. The aim of this study is to evaluate the impact of climate change on floodplain inundation for important floodplain wetlands in Europe and to place these results in an ecological context. This work is performed within the SCENES project considering three different climate change projections for the 2050s. The global scale hydrological model WaterGAP is applied to simulate current and future river discharges which are then used to i) estimate bankfull flow conditions, ii) determine three different inundation parameters, and iii) evaluate the hydrological consequences and their relation to ecology. Results of this study indicate that in snow affected catchments (e.g. in Central and Eastern Europe) inundation may appear earlier in the year. Duration and volume of inundation are expected to decrease. This will lead to a reduction in habitat for fish, vertebrates, water birds and floodplain specific vegetation causing a loss in biodiversity, floodplain productivity and fish production. Contradictory results occur in Spain, France, Southern England and the Benelux countries. This reflects the uncertainties of current climate modelling for specific seasons.

Keywords: climate change, ecological impacts, floodplain wetlands, floods, partial duration series, WaterGAP

3.2 Introduction
Floodplain wetlands are defined by their recurring inundation caused by flooding of adjacent rivers and often, the health of riverine ecosystems is dependent on the natural pattern of these inundation events (Junk et al. 1989). Hence, changes in flood flows have severe consequences on ecological and biological processes in river ecosystems. They may drain wetlands close to the river, reduce the productivity of river banks, lower the dynamics of delta regions, and eradicate communities of organisms in the water (Nilsson et al. 2005). Today, river systems are already regarded as the most threatened ecosystems on the planet (Malmqvist & Rundle 2002) and the loss of biodiversity in riverine ecosystems has proceeded faster over the past 30 years than in terrestrial or marine ecosystems (Jenkins 2003). Besides other anthropogenic factors such as river regulation, channelization, wetland drainage and water abstractions, climate change may severely alter the natural pattern of inundation over large regional scales in the future. Due to increasing temperatures, evapotranspiration will be raised nearly everywhere causing a reduction in runoff (Frederick & Mayer 1997).

Precipitation patterns are also altering under climate change with regionally and seasonally different developments (IPCC 2007) leading to higher or lower runoff values in the future (Alcamo et al. 2007). In addition, in snow affected catchments, runoff is influenced by a decreased volume and duration of snow cover in the winter time (Verzano & Menzel 2009). All these effects interact differently at different locations leading to unfavourable changes in the river flow regimes with large geographical differences in directions and causes.

The hydrological impacts, in turn, can have significant consequences on the quality and functioning of floodplain wetland ecosystems which have evolved under, and become dependant on, regular inundation. Inundation (flooding) steers floodplain aquatic connectivity and the transport of matter and organisms through the system (Tockner et al. 2000; Junk et al. 1989). The floodplain landscape, and hence its connectivity gradient, is formed by interaction between the hydro-morphological regime and changing biota. Along these gradients, called ecotones, environmental factors vary and provide specific habitats for flora and fauna species. The formation of the floodplain landscape as well as the abiotic conditions in floodplain ecotones are highly influenced by the magnitude, timing and duration of inundation (Petts & Amoros 1996; Hughes 1997; Tockner et al. 2000). Here, the inundation magnitude causes the disturbance generated by a flood by i) determining the surface area inundated, ii) determining the amount of matter and organisms transported, and iii) influencing the floodplain shape. But reshaping the floodplain by erosion and sedimentation processes is essential for creation of pioneer habitat (Hughes 1997; Marston et al. 2001; Petts 2000; Geerling et al. 2006). Sediment and associated nutrients, transported in the main channel, are deposited in floodplains during floods. The distribution of these sediments and nutrients affect ecological succession and production of various floodplain elements, such as floodplain forests and floodplain lakes (Amoros & Wade 1996; van Geest et al. 2005).
The timing of inundation has an influence on the ecological functioning of floodplains, like affecting life cycles of biota on various levels of scale. Some examples are given below: spawning habitat availability for fish species varies greatly at different flood levels, and spawning is mostly confined to a certain time in the year (Van de Wolfshaar et al. 2009). Also settlement of seeds, like poplar or willow, depends on the flood level during seed dispersal. The timing determines if suitable habitat can be colonised and additionally, a subsequent summer floodplain inundation can remove young seedlings and prevent settlement entirely (Merritt et al. 2009). In plant community research, summer floodplain inundations are regarded as more influential than winter inundations, because they determine the zonation of grassland communities, while winter inundations seem to sustain the zonation (van Eck et al. 2004). Also tolerance to inundation of floodplain forest species is lower in the growing season than in winter (Glenz et al. 2006). In the decomposition phase of floodplain life cycles, inundation of floodplains in winter is related to higher decomposition rates of floodplain litter than summer inundations (Langhans & Tockner 2006).

As the last parameter, flood duration accounts for the abiotic soil conditions, the amount of settlement of fine sediments and amount of groundwater contact. The adaptation to flood duration is a selective pressure to floodplain species. For example, Casanova & Brock (2000) state that duration determined the zonation of plant communities. Floodplain forest species survival under flood duration stress varies greatly and determines short-term (extreme) and long-term (chronic influence) community composition (Glenz et al. 2006).

While flooding can cause damages with enormous costs, it is beneficial at natural locations and stimulates important ecosystem services of floodplain wetlands such as detoxification and nutrient removal, biomass and fish production, carbon storing, as well as biodiversity maintenance. In addition, healthy floodplains contain recreation and aesthetic values. The aim of this study is an ecological based assessment on flooding by quantifying the changes in magnitude, timing and duration of overbank flows for major European rivers affected by climate change impacts.

For our analysis we used the global scale hydrological model WaterGAP for simulating floods on European scale. In addition, three climate change projections for the 2050s (2040-2069) were selected, calculated by three different GCMs (General Circulation Models): two representing the IPCC SRES A2 and one the SRES B1 scenario. As a flood indicator we used the "bankfull flow approach" for deriving the inundation parameters. Finally, the expected effects on selected ecological components are qualitatively demonstrated.

This paper is organized as follows. In 'methods' we describe the selection process of floodplain wetlands for this analysis, the estimation of bankfull flow as an indicator for inundation and the analysis of all overbank flows by different hydrological parameters. Furthermore, the modelling approach of current and future river discharge data by WaterGAP and the selected climate change projections are described in this section. In 'results and discussions', at first, the impacts of climate change on volume, duration and timing of inundation are depicted. Subsequently, the hydrological changes are discussed and their impact on ecology is qualitatively evaluated.

3.3 Methods

Selection of rivers

The analysis focused on major European rivers which have a high biological potential due to the flooding of adjacent floodplain wetlands. As there is no commonly accepted database of valuable European floodplains, following procedure has been implemented in order to select the rivers of interest. In a first step, a database of vast (i.e. area bigger then 5000 ha) European wetlands has been created and the riparian (river fed) wetlands were taken out for further analysis. In a second step, a spatial analysis of the Corine Land Cover (CLC) map (EEA 2004) has been performed in order to find the river valleys with specific classes of vegetation assuming that this indicates the potential or the need of flood pulsing.

The database of European wetlands was established in the context of the SCENES[1] project and consists of 102 objects including spatial data and attributes. Thereby, the following data-filtering and preparation procedure was applied:

1. Data about protected wetlands were collected from available European sources (NATURA 2000, Ramsar Convention) and national wetland surveys supported by expert knowledge of wetland specialists in Europe. This database contains more than 4000 objects.
2. The database was filtered by means of area (i.e. larger than 5000 ha) which resulted in more than 400 objects.
3. The database was divided into countrywide sets, which were sent to national experts for verification. The main question of the survey was about the "real" extend of the wetland area. Within the database, protected area is a mix of different habitats and the wetland of interest can cover an insignificant part of the protected territory. Additionally, national experts were asked for further characteristics of the selected objects such as water feeding and wetland type. This part of the procedure resulted in 102 objects.
4. In parallel, a database containing the spatial boundaries of the selected wetlands was created and related to the WGS84 (World Geodetic System 1984) coordinate system. The reviewed database was unified into one comprehensive dataset consisting of tables and ArcGIS shapefiles (*.shp) associated in topologically correct layers.

---

[1] *SCENES (Water Scenarios for Europe and for Neighbouring States), contract nr. GOCE 036822, integrated project in the 6th framework programme.*

A first subset of rivers which fed wetlands during high flows was identified by means of spatial analysis based on the created wetland datasets. Thereby the focus was on hydrological dependent wetlands (so called riparian wetlands or wetlands of fluviogenic type of hydrological feeding). As the spatial extent of each of the wetland objects was known and precisely mapped in the WGS-84 coordinate system, topological correctness allowed to relate the wetland database to the drainage direction map (DDM5; Lehner et al. 2008) as used in WaterGAP. Subsequently, 44 rivers were selected for the further hydrological analysis.

A second subset of rivers was identified by the analysis of land cover in the river valleys. As a most coherent and prospective data source for the presented study, the Corine Land Cover (CLC) dataset was used. During the analysis following land use categories were taken into consideration: wetlands, inland marshes, peat bogs, natural grasslands, pastures scrub and herbaceous vegetation associations. If the combined size of such labelled area was bigger than 5000 ha and crossed by a major river of the DDM5 map, then the river has been chosen for the further analysis. This phase of the selection resulted in the choice of 30 rivers.

Finally, 74 rivers were taken into consideration; all representing major European rivers responsible for the hydrological feeding of important European wetlands or associated with floodplains still covered with the vegetation resulting from high moisture conditions.

Bankfull discharge approach and application

The floodplain analysis performed in this study is based on the concept that in a riverine ecosystem different flows have different functions. Floodplains are hydrology-dependent ecosystems and in order to maintain their multitude of crucial ecological as well as socio-economic functions, they depend on high flows which lead to inundation. In the analysis of flood dynamics and their ecological impacts, the scientific community has largely adopted magnitude and frequency of bankfull discharge as one of the important concepts (Navratil et al. 2006). Bankfull discharge is the flow at which the channel is full of its capacity (to the top of the banks), whereas the flow just begins to enter the active floodplain (Leopold 1964). Above this discharge, all in-channel secondary channels and in-channel wetlands are generally hydraulically connected and so it provides important information for the ecological functioning of the river (Navratil et al. 2006).

The determination of bankfull discharge, however, is a complex analysis, and a choice has to be made between different existing methods. In order to estimate bankfull discharge for large scale modelling purposes, an approach needed to be found which does not require in-situ river characteristics or hydraulic data (such as cross-sectional area) that are not available on a continental grid. Using flood frequency analysis in order to estimate bankfull flow is based on the assumption that on the long-term average, bankfull discharge occurs at a certain time interval (this does not imply regularity of occurrence). This assumption is not true for all types of rivers (e.g. bankfull events occur more frequently within the Coastal Plain as shown by Sweet & Geratz 2003), but still good estimates of bankfull flow can be gained. Leopold et al. (1964) stated that there is a remarkable similarity in the frequency of bankfull stage on a variety of rivers in diverse physiographic settings and sizes. Although some localities may diverge greatly from a specific frequency (Williams 1978; Mosley 1981), a number of studies worldwide have proven a correlation between certain flood return periods and bankfull discharge (Woodyer 1968; Harman 1999; Castro & Jackson 2001). Therefore, in this study a statistical approach on flood frequency analysis has been chosen which was applied on daily discharge data of a 42-year time series (1961-2002) simulated by WaterGAP.

Two methods in flood frequency analysis do exist which can be used to estimate bankfull discharge, (i) the Annual Maximum Flood (AMF) approach and (ii) the Partial Duration Series (PDS) approach.

Due to its simpler structure, the AMF approach has been often used and best approximation is obtained by considering a recurrence period of 1.5 years (Dury 1977; Dunne & Leopold 1978). However, a direct comparison has shown that the PDS approach should generally be preferred (Madsen et al. 1997a). The PDS approach takes into account all flood peaks above a certain threshold and thus, has several important advantages in contrast to the AMF approach (Begueria 2005). The most important advantage for our analysis is the enhanced resolution for high frequency events (Sweet & Geratz 2003) such as the inundation of floodplains. The return period of bankfull discharge is very close to the smallest value which can be obtained by using annual series (i.e. one year). The PDS approach, however, is able to provide sub-annual recurrence intervals and respects that in some years, rivers can have more than one bankfull flow event. In addition, the PDS approach adapts better to heavy-tailed distributions which are common in hydrological applications (Madsen et al. 1997b) and makes more sufficient use of simulated hydrographs as it includes more flood peaks (Kite 1977). Thus to be more precisely in estimating bankfull discharge, we made use of the PDS approach whereas a return period of 0.92 years was applied as suggested by Dunne & Leopold (1978). In Europe, the PDS approach was applied by Petit & Paucet (1996) to determine the return period of bankfull discharge of 33 gravel-bed rivers in Belgium. For these rivers, an average return period of 1.2 years was found.

PDS approach

The PDS approach is based on the selection of flood peaks above a fixed threshold and comprises the assumptions that these are mutually independent, exponentially distributed, and their number per time period follows a Poisson distribution (Langbein 1949; Shane & Lynn 1964; Todorovic & Zelenhasic 1970; Davison & Smith 1990). Hence, the choice of an appropriate threshold is the crucial part of the PDS approach which, however, represents one of the most difficult issues in its appliance (Nguyen 2002). In general, a threshold that is too low makes the threshold exceedances too close in time and thus, introduces serial dependence. On the other hand, a threshold that is too high, leads to an important loss in information of the hydrograph.

In the scientific literature, different systematic methods for the choice of threshold have been proposed and applied whereas the determination of the optimal threshold selection still requires more research (Adamowski et al. 1998; Deidda & Puliga 2009). Many researchers suggested methodologies that are based on the mean number of threshold exceedances per year. Most frequently a value between one and five has been cited (Choulakian et al. 1990; Begueria 2005). However, Lang et al. (1999) stated that no unique specific value exists for precise modelling and hence, an increasing threshold censoring procedure is recommended which is based on mathematical tests. One of these tests is the dispersion index (DI) which was proposed by Cunnane (1979). The DI is used to verify the adequacy of the Poisson assumption and is defined as the ratio of the average number of threshold exceedances per year to its variance. If the threshold exceedances follow a Poisson process, then the DI should be close to one. Consequently, the DI was used in our study to find the most suitable threshold within a range of one to five[2] threshold exceedances per year. Ashkar & Rousselle (1983) showed that if a specific threshold has been found which follows a Poisson process, then any higher threshold produces independent flood peaks. Thus, we started with the lowest threshold of our range (i.e. five) and raised it step-by-step until the DI was close to one (i.e. the DI must be within the limits of a confidence interval around one which was calculated by testing against a chi-square distribution with a significance level of 0.05). In addition, the assumption of independence was relaxed by applying a declustering scheme. As hydrological events occur grouped in clusters (i.e. multiple peaks correspond to the same flood event), only the single highest flood peak within a cluster was included in the PDS.

[2] *Applied values for threshold setting: 1.0 / 1.2 / 1.6 / 2 / 2.5 / 3 / 3.5 / 4.5 / 5*

Thereby, one flood event is characterised by an up-crossing of the threshold level and the subsequent down-crossing.

The gained flood peaks were then arranged in order of magnitude and fitted to a Generalized Pareto distribution (GPD), which is a special case of both exponential and Wakeby distribution. The GPD was introduced by Pickands (1975). Since then, it was often used in hydrology and especially for the distribution of independent exceedances over a certain threshold (Hosking & Wallis 1987; Davison & Smith 1990; Wang 1991; Rosbjerg et al. 1992) because of its inherent properties. Finally, estimating the inherent parameters of the GPD enabled the calculation of bankfull discharge by applying a recurrence period of 0.92 years.

The procedure to determine bankfull discharge includes the modelling of a 42-years time series on a daily resolution, threshold setting, declustering and distribution fitting. All was done individually for each relevant WaterGAP grid cell (5 x 5 arc minutes) in Europe. However, our continental scale approach is connected with a number of uncertainties such as the setting of an appropriate threshold, the ambiguity of bankfull stage and the assumption of a specific return period that characterizes bankfull discharge for all types of rivers.

Parameter of interest

In general, all components of a natural flow regime have a certain ecological significance. Low, medium and high flows create and maintain different habitat features and aquatic species have evolved life history strategies primarily in direct response to them (Bunn & Arthington 2002). As it is important for this study to know about the flow events associated with overtopping of the banks and inundation of the floodplain, any flow greater than bankfull flow is considered a critical flow to investigate. But besides the magnitude of these overbank flows, it is also important to analyse how long they last and at which time of the year they occur. Flood magnitude and volume account for the extent of inundation whereas the duration of inundation determines whether a particular life-cycle phase of aquatic species can be completed. The timing of inundation assesses if life-cycle requirements are met, because key life-cycle phases are linked to the timing of annual extremes.

In this study, these parameters were regarded as crucial for describing hydrological alterations and are defined as follows (see also Figure 3.1):

- Flood volume for inundation (i.e. the cumulative amount of water above bankfull flow)
- Duration of overbank flows (i.e. the number of days flow exceeds bankfull flow)
- Timing of inundation (i.e. the month of the year with the highest flood volume)

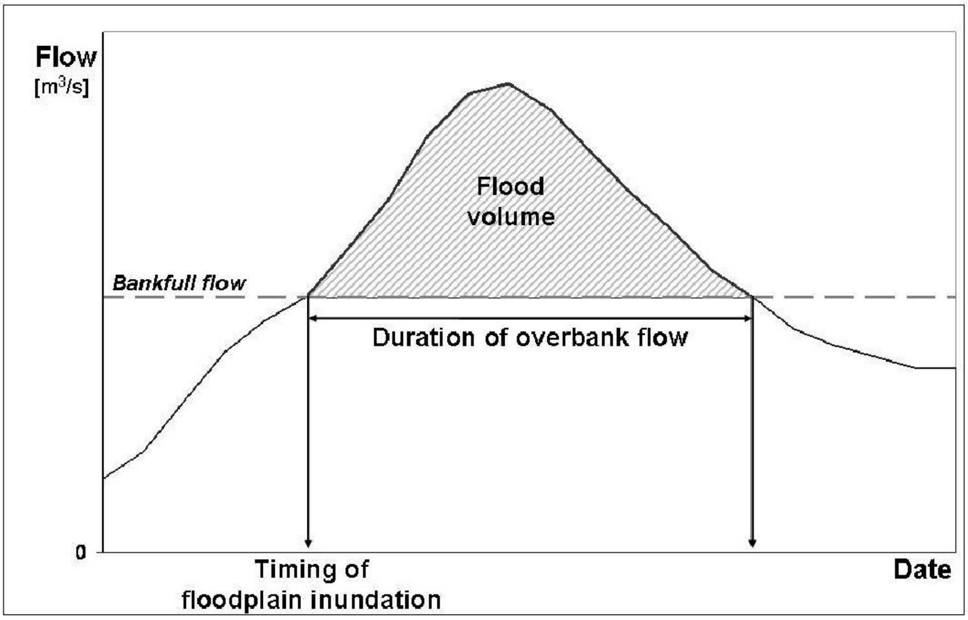

Figure 3.1 Parameters applied in this study to analyse floodplain inundation.

Modelling of current and future daily time series

To compute the impact of climate change and other important driving forces on future water resources, the WaterGAP model (Water – Global Assessment and Prognosis) (Alcamo et al. 2003, Döll et al. 2003) was used. The model version applied in this study, WaterGAP3, herein referred as WaterGAP, computes both water availability and water uses on a 5 by 5 arc minutes grid (longitude and latitude), covering the whole of Europe. WaterGAP consists of two main components: a Global Hydrology Model to simulate the terrestrial water cycle and a Global Water Use Model (Flörke & Alcamo 2005) to estimate water withdrawals and water consumption of five different water use sectors. Since this study focuses on the impact of climate change on hydrological alterations of river discharges, the influence of water uses has not been taken into account. Thus, only the hydrological model of WaterGAP is described in more detail.

The aim of the Global Hydrology Model is to simulate the characteristic macro-scale behaviour of the terrestrial water cycle in order to estimate water availability. Herein, water availability is defined as the total river discharge, which is the sum of surface runoff and groundwater recharge. The upstream/downstream relationship among the grid cells is defined by a global drainage direction map (DDM5) which indicates the drainage direction of surface water (Döll & Lehner 2002). Additionally, the flow length per grid cell is enhanced by applying an individual meandering factor for each grid cell derived from a high-resolution DDM (Lehner et al. 2008). In a standard model run, river discharges are simulated in 19254 river basins in Europe. The effect of changing climate on runoff is taken into account via the impacts of temperature and precipitation on the vertical water balance.

Next to others, the main improvements of the model were done with special focus on the models ability to simulate floods. First, the storage of precipitation as snow is a crucial process within the hydrological cycle, since snow melt in spring induces increased river discharges and even floods in snow affected watersheds. Second, with the calculation of dynamical flow velocities the differentiation between mountainous rivers with steep river bed slopes and rivers in lower regions is possible. Therefore the inclusion of snow related processes, with snow melt calculated by a simple degree-day algorithm on sub-grid scale (Verzano & Menzel 2009) and the consideration of dynamical flow velocity (Verzano et al. 2005) were implemented in the model code.

The parameters of interest indicating hydrological alterations in major European floodplain wetlands have been derived from the 30-year time series of gridded daily river discharge results calculated by WaterGAP for the reference period (1961-1990) and for three GCM-scenario combinations representing the 2050s (2040-2069). Thus time series were modelled for selected European floodplain wetlands taking into account a daily resolution.

Climate change scenarios

The baseline climate input including monthly information on precipitation, temperature and others covers the timeframe 1961 – 1990. For the model simulations we used a combination of the datasets CRU TS 2.1 (Mitchell & Jones 2005) and CRU TS 1.2 (Mitchell et al. 2004). Although the CRU TS 1.2 dataset has a higher spatial resolution (10 arc minutes) it covers only the predominant part of Europe. In order to get information for grid cells that were not covered, the CRU TS 2.1 dataset with a spatial resolution of 30 arc minutes was applied. Then both datasets were simply downscaled to a 5 arc minutes grid. Both CRU datasets, TS 2.1 and TS 1.2, provide monthly values for precipitation, temperature, cloud cover and the number of wet days per month. However, the WaterGAP model simulates river discharges on a daily time step. Therefore, the monthly climate input had to be downscaled from monthly to daily values. In this context, temperature and cloudiness were downscaled with a cubic-spline-function between the monthly averages, which were assigned to the middle of each month. Precipitation was first distributed equally over the number of wet days per month and then distributed between the wet days within a month. The latter calculation was mathematically realized by using a two-state, first-order Markov Chain, for which the parameters were chosen according to Geng et al. (1986). Both, downscaling of temporal and spatial data are associated with uncertainties as local sub-grid features and dynamics are neglected. However, Prudhomme & Davis (2009) showed that these uncertainties are generally lower than uncertainties caused by using different GCMs. In the scope of this paper, only one downscaling method was used.

The impact of climate change on water resources is expected to be stronger in 2050 compared to 2025. For this reason, we have drawn our attention only to the 2050s time period. To take into account the uncertainty of climate modelling, two SRES emission scenarios from three different GCMs were analyzed. Within the SCENES project, the following model and scenario combinations were selected: (1) The IPSL-CM4 model from the Institute Pierre Simon Laplace, France representing an A2 scenario (IPCM4-A2). This GCM-scenario combination indicates high temperature increase and low precipitation increase or decrease in Europe (warm & dry); (2) The MICRO3.2 model from the Center for Climate System Research, University of Tokyo, Japan representing an A2 scenario (MIMR-A2). In accordance with the IPCM4 model, the MIMR model projects a high temperature increase over Europe, but in combination with a high precipitation increase or low decrease (warm & wet); (3) The ECHAM5/MPI-OM model from the Max-Planck Institute for Meteorology, Germany representing a B1 scenario (MPEH5-B1). In contrast to the A2 scenario, the B1 scenario predicts a small temperature increase and an average precipitation change (moderate). These models were chosen to compute climate projections under changed levels of greenhouse gas emissions as specified for the SRES A2 and B1 scenarios for the 2050s represented by a time series covering the years 2040 to 2069. The original GCM outputs have a spatial resolution of 1.875° x 1.875° (T63, longitudinal and latitudinal) and have been downscaled to the 5 arc minutes grid cells by applying a simple bilinear interpolation approach. Here, monthly temperature (T) and precipitation (P) results were used from the selected GCMs described above. The number of rain days per month and the cloudiness were taken from the reference period (1961-1990), and then the climate values were downscaled to daily climate as described in the section above. Hence, a possible increase of climate variability at the daily scale was not taken into account.

This simple approximation of pseudo-daily future climate input was initially implemented in WaterGAP for studies of climate change impacts on long-term average discharge and may affect the simulated magnitude of high flows. The future climate input was scaled in consideration of the difference between observed and simulated climate of the reference period (Henrichs & Kaspar 2001, Lehner et al. 2006). For temperature, the observed CRU time series were scaled by adding the respective difference between the future and present-day temperature values from the GCM. For precipitation, observed precipitation time series were scaled by multiplication with the respective ratio between future and present-day precipitation. An exception to this rule occurs when present-day precipitation is close to zero (< 1mm); in this case the respective value was added. Following this method, monthly values for 30-year climate time series were constructed for the 2050s. This scaling approach is frequently applied to force global scale hydrological models for climate change studies.

3.4 Results and discussion

Although high flows leading to floodplain inundation can occur the whole year, usually, they accumulate within a specific season or month of the year. Therefore, our results strongly depend on seasonal climatic conditions. Within this study we assumed that the more uniform the results for a river and the larger a regional pattern, the higher the significance of changes in future floodplain inundation. In the following, the results indicating future changes in (i) flood volume, (ii) duration, and (iii) timing of overbank flows are presented.

Change in flood volume

Flood volume determines the extent of inundation. The three different climate change realisations generally imply a change in flood volume for almost all regions of Europe (Figure 3.2). In Central and Eastern Europe, all three climate projections show an agreement in flood volume causing floodplain inundation which is likely to be reduced in the 2050s. Thereby, the climate impacts are stronger under the IPCM4-A2 and MIMR-A2 projections compared to the MPEH5-B1 scenario realisation. Likewise, there is agreement in Ireland, Scotland and Western England. In this area more water will be available for floodplain inundation in the future.

An area of high uncertainty indicated by contradictory results in our analysis occurs in France, Spain and the Benelux countries as well as on the Thames and Derwent River. Depending on the climate change data used, flood volume is reduced (IPCM4-A2) or increased (MIMR-A2) in the future. However, in these regions, agreement can be found in mountainous areas where rivers originate at high altitude. A reduced flood volume is predicted under all three climate change projections for river reaches at the Sisterna Iberico (Turia), the Pyrenees (Cinca and Garonne), Massif Central (Dordogne, Loire and Lot), the Alps (Rhone, Rhine, Enns, Mura, Drava, Isar, Inn, and Po), and likewise at the Dinaric Alps (Neretva) and Rila mountains (Maritza). The reduced flood peak is then carried forward along the rivers, at least for some distance.

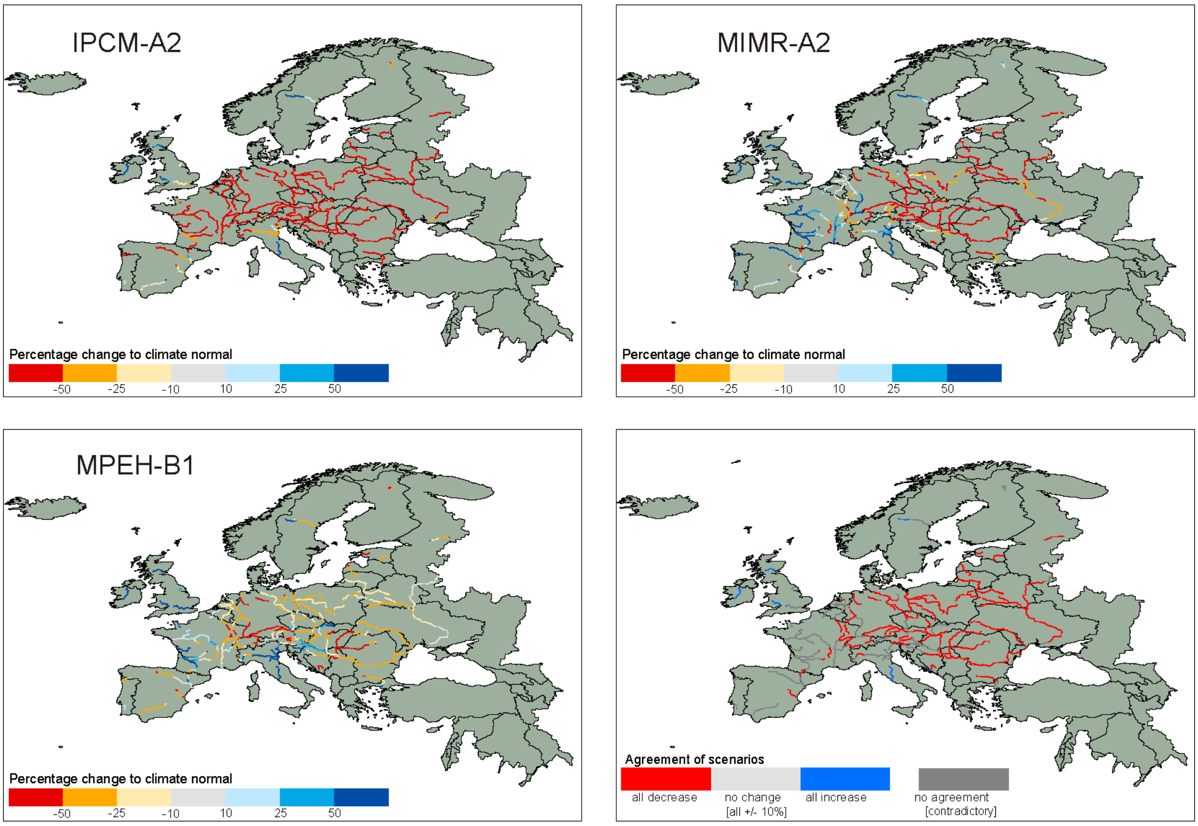

Figure 3.2 Change in flood volume in the 2050s compared to the baseline period (1961-90) under different future GCM-scenario combinations (IPCM4-A2, MIMR-A2 and MPEH5-B1). Agreement between these scenarios is shown in the map on the bottom right. Here, grid cells are labelled as 'no change', when the change in flood volume is within the range of +/- 10% under all three applied scenarios. If the change in flood volume increases under all three scenarios and the increase is higher than 10% in at least one scenario, then the grid cell is marked as 'all increase'. The same applies for 'all decrease' in case of a reduction.

Change in duration of overbank flows

The sensitivity of floodplains is often based on the duration of floodplain inundation. For the parameter duration a quite similar picture is drawn as for the flood volume (Figure 3.3). In Central and Eastern Europe as well as in mountainous areas the duration of overbank flows is reduced, while in Ireland, Scotland and Western England duration of overbank flows is increased. The results based on MPEH5-B1 climate predict for most rivers only minor changes. Again, in France, Northern Spain and the Benelux countries as well as on the Thames River, no agreement can be found under the three GCM-scenario realisations. In contrast to the flood volume, there is also no agreement for the rivers Elbe, Havel, Warta and Narew, as well as for parts of the Oder and Dnieper.

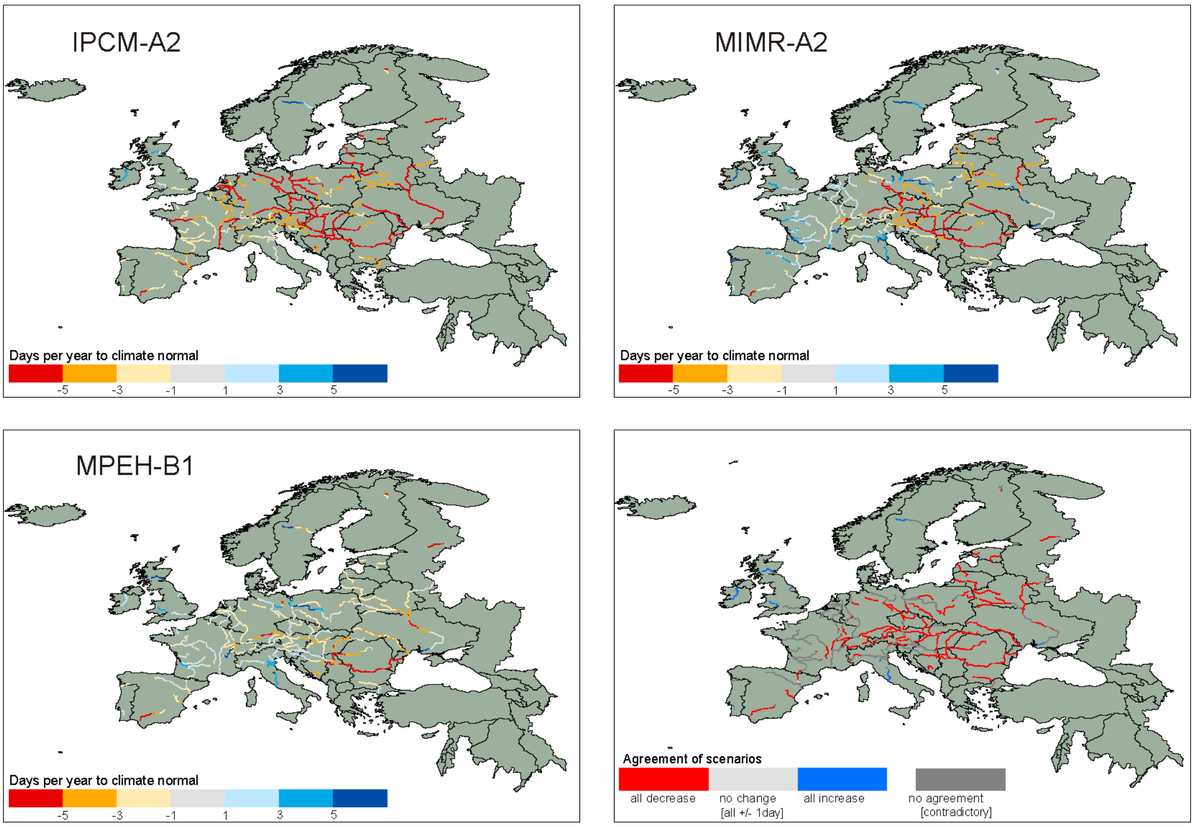

Figure 3.3 *Change in duration of overbank flows in the 2050s compared to the baseline period (1961-90) under different future GCM-scenario combinations (IPCM4-A2, MIMR-A2 and MPEH5-B1). Agreement between these scenarios is shown in the map on the bottom right.*

Change in timing of floodplain inundation

The timing of floodplain inundation is important as access to and availability of floodplain habitats must coincide with life-cycle requirements of flood dependent local flora and fauna. Figure 3.4 depicts the month of the year with the highest flood volume in the baseline period, i.e. the time of the year where usually floodplain inundation occurs. According to this, in Western and Southern Europe, floodplain inundation accumulates in the winter time (blue coloured grid cells) while in Central, Eastern and Northern Europe inundation usually occurs in spring (green coloured grid cells). The North of Fennoscandia and mountainous areas stand out with the highest flood volume occurring mostly in June.

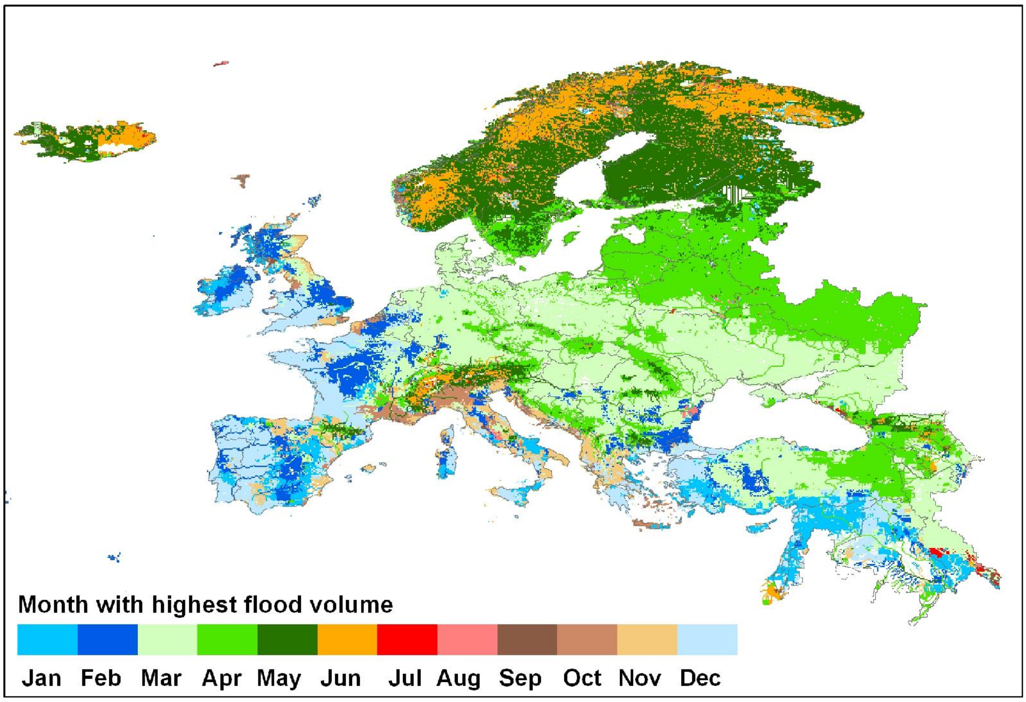

Figure 3.4 Month of the year with the highest flood volume within the baseline period (1961-90) as simulated by WaterGAP for each grid cell in Europe.

In the 2050s, there will be a shift in timing of floodplain inundation on many rivers in Europe, especially under the IPCM4-A2 and MIMR-A2 climate change projections (Figure 3.5). In Eastern and Northern Europe, floodplain inundation is expected to occur earlier than under baseline conditions (i.e. at least one month). In Southern and Central Europe, timing of floodplain inundation is likely to be earlier for many rivers, but there are also some rivers, especially in northern Italy, where timing can also be later within the year.

Assessment of the hydrological changes

The impact of climate change on floodplain inundation is induced on the one hand by increasing temperatures and on the other hand by spatial and temporal changes in precipitation patterns. Northern and Eastern Europe are characterised by cold or continental climate with strong winters often permanently below 0°C. Floodplain inundation in this area often occurs as a consequence of snow melt in spring falling together with strong precipitation events during this time (see also Figure 3.4). Due to increasing temperatures under climate change, extent and duration of snow cover are significantly reduced in this area in the 2050s. In the Northern Hemisphere, a reduction of approximately 10% in snow cover has been observed since 1966 (IPCC 2001) and Arnell (1999) showed that in the 2050s, snow cover will be considerably decreased over large parts in Central and Eastern Europe by the end of the winter. In addition, precipitation falls more often as rain instead of snow, leading directly to runoff in the winter time. Hence, in Central and Eastern Europe, discharges are likely to be increased in the winter, but the resulting snow melt induced flood peak in spring is expected to be decreased in the 2050s as less water is stored in the snow pack. This development in snow affected river basins was demonstrated on an example in Belarus by Arnell (1999) and is exemplified here for the Narew River in Figure 3.6.

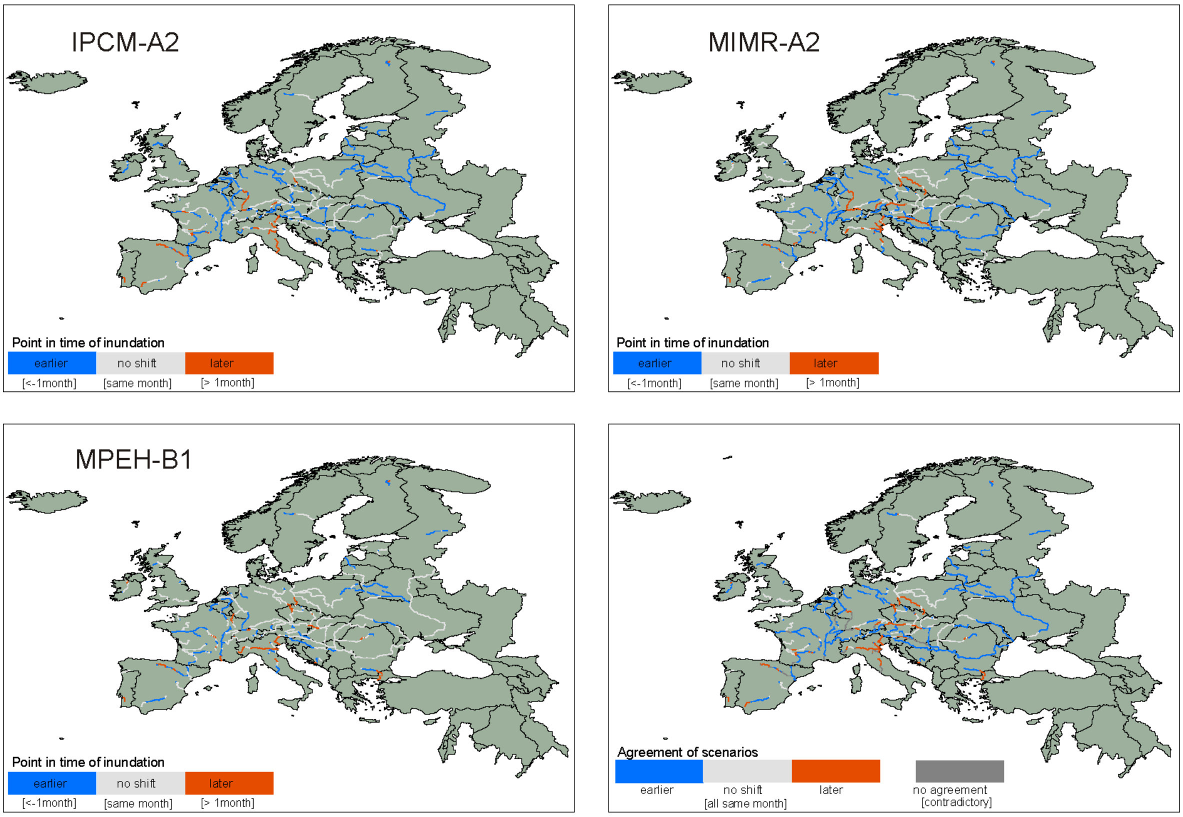

Figure 3.5 *Change in timing of floodplain inundation in the 2050s compared to the baseline (1961-90) under different future GCM-scenario combinations (IPCM4-A2, MIMR-A2 and MPEH5-B1). Agreement between these scenarios is shown in the map on the bottom right. Here, grid cells are labelled as 'no shift', when the highest flood volume occurs under all three scenarios in the same month as in the baseline. Grid cells are labelled as 'earlier' or 'later', when the shift has the same direction under all three scenarios and is at least one month in one scenario.*

The reduced flood volume for inundation and the reduced duration of overbank flows as identified in our study for rivers in Central and Eastern Europe can be explained by the major role of snow melt in this region. Here, in the snow affected river basins, the three scenarios show high agreement. An analysis of flood hazards was also conducted by Dankers & Feyen (2008). They also found a considerably decrease in flood hazards in the northeast of Europe. While in their study, this applies for the Baltic States, Finland and Northern Russia, in our study reduced flood volumes were already found in Poland and Eastern Germany. These differences could be explained by a much higher return period (i.e. 100 year) and the choice of a different GCM input. Dankers & Feyen (2008) expected in their analysis an increase in flood hazards in France and Northern Italy. This development corresponds to our analysis at least in two scenarios (MIMR-A2 and MPEH5-B1).

The duration of overbank flows shows a similar development as the flood volume. But considering MIMR-A2 climate, the duration of overbank flows shows an increase for a few rivers in Central and Eastern Europe (i.e. for Elbe, Havel, Warta and Narew, as well as parts of Oder and Dnieper). However, this can be explained by the increased runoff in the winter which causes some minor discharge peaks above bankfull flow for this climate projection, but does not lead to widespread inundation of the associated floodplains. As the 0°C level is crossed earlier in the year, snowmelt is induced earlier in the year, too. Therefore, floodplain inundation is likely to occur earlier within the year in the 2050s. The same effect on flood

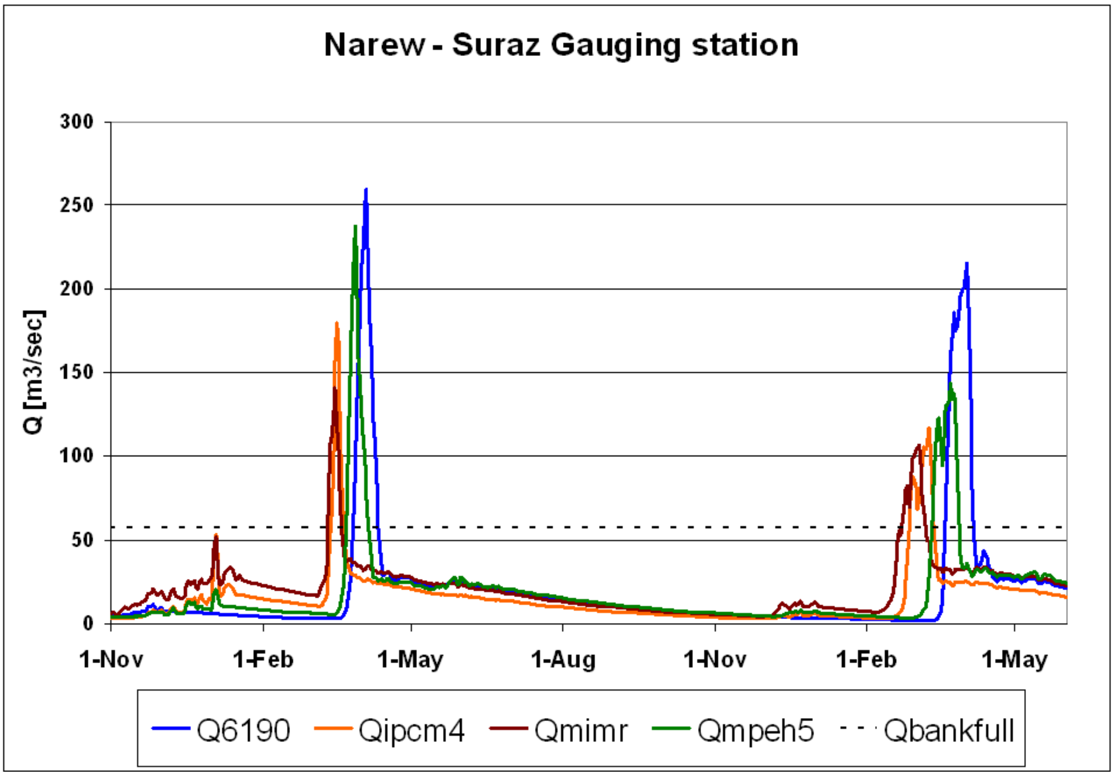

Figure 3.6 Two year example of the Narew River in Poland simulated by WaterGAP showing changing discharges in the 2050s under the three applied scenarios compared to the baseline period (blue hydrograph).

patterns applies to rivers originating in mountainous areas (e.g. Alps, Massif Central, and Pyrenees) where snow melt influences the different indicators for some river distance. The shift in flood patterns to earlier seasons in the year in European mountain regions could also be shown by Dankers & Feyen (2008).

Western and Southern Europe are characterised by maritime climate with milder winters where snow melt does not play a crucial role in the formation of high flows. Here, floodplain inundation is often caused by winter rains at a time of the year where evapotranspiration is low. Hence, in this area, future predictions of floodplain inundation strongly depend on the GCMs' precipitation patterns rather than temperature. However for France, Spain, South England and the Benelux countries, the three different climate projections applied in our study predict contradictory results for precipitation in winter (Figure 3.7), the time where usually overbank flows occur in these regions.

Under the IPCM4-A2 climate projection, less winter precipitation in France, Spain, South England and the Benelux countries is leading to a reduction in floodplain inundation, while under the MIMR-A2 climate higher winter precipitation is causing an increase in floodplain inundation. The MPEH5-B1 climate shows moderate changes in winter precipitation with higher winter precipitation in parts of France, Benelux and South England, but less winter precipitation in Spain, parts of Turkey and Middle East. Consequently, for the selected rivers in France, Spain, South England and the Benelux countries, our analysis provides contradictory results which reflect the uncertainties of current climate model calculations.

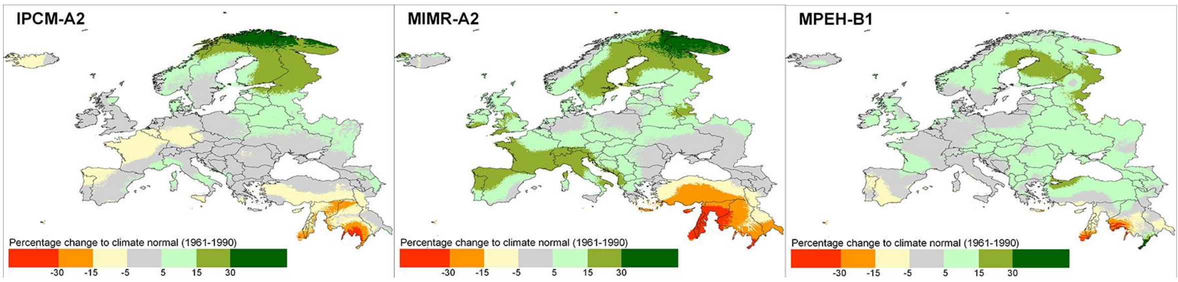

Figure 3.7 Absolute change in average precipitation in winter between the three different scenario realisations (IPCM4-A2, MIMR-A2, MPEH-B1) for the 2050s and the baseline conditions.

In this study, the scaling approach was chosen to calculate future projections of temperature and precipitation to force WaterGAP (see Methods). This approach was applied on CRU climate data which provide a significantly higher spatial resolution (0.167°) in contrast to time series calculated by GCMs (1.875°). Therefore, snow induced flood peaks were represented significantly better, especially in the comparable small mountains of Europe. On the other side, the appliance of time series calculated by GCMs would directly respect the enhanced future climate variability which is again supposed to increase flooding in the future in many areas (IPCC 2007; Kundzewicz et al. 2007). For future analysis, uncertainties due to spatial downscaling could be reduced by applying Regional Climate Model (RCM) output to force WaterGAP. State-of-the-art RCMs possess a spatial extent between 12 km and 50km over Europe (Christensen and Christensen, 2007). Uncertainties due to temporal downscaling could be minimized by using lately established climate data based on daily time steps (Weedon et al. 2010).

Ecological impacts

Here, the results regarding changes in volume, duration and timing of inundations are placed in an ecological context, focusing on the effects on two major floodplain vegetation types and fish. To assess the effect of climate change on floodplain vegetation, a literature survey was carried out. The challenge is to fit the knowledge available to the applied model scale and outputs.

Floodplain vegetation

As shown above climate change has an impact on the hydro-morphological regime of rivers and wetlands, and as such can influence the spatial arrangement of floodplain vegetation. In particular the vegetation communities that reach their climax stage after long development cycles and depend on specific hydro-morphological regimes are sensitive to climate change. Those include the hardwood floodplain forests and the dry river grasslands that also inhabit the so-called river corridor species or "Stromtalpflanzen" (Burkhart 2001; van Looy & Meire 2009). Drier river grasslands and riparian mixed forest habitat is already marginalised through direct anthropogenic impacts such as deforestation and the cut-off of the lower dynamic floodplain area from the river by embankments. Expert judgement based on the literature reviewed gives the following climate related factors and ranges, which are needed to sustain hardwood forests and dry floodplain grasslands (see Table 3.1).

Table 3.1 Climatic factors and their responses for floodplain forests and dry river meadows. The vegetation types focused on are in the Natura 2000 habitat list (CEC 1992): (91E0) Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae); (91F0) Riparian mixed forests of Quercus robur, Ulmus laevis and Ulmus minor, Fraxinus excelsior or Fraxinus angustifolia, along the great rivers (Ulmenion minoris); (6120) Dry river grasslands of calcareous soils. In the NL these are of the Medicagini-avenetum pubescens type (van Looy and Meire 2009) also including the Stromtalpflanzen (Burkhart 2001); (6440) Alluvial meadows of river valleys of the Cnidion dubii (Germany) also including the Stromtalpflanzen (Burkhart 2001). Growing season is defined as the months May to September.

Climatic factor	**Habitat type**	
	Alluvial hardwood forest	Alluvial 'dry river meadows'
Magnitude		
Extreme flood	At least every 10-20 years to block succession to pure terrestrial forest.	Winter floods maintain habitat gradients shaped by summer floods.
Sedimentation	Fundamental for succession towards less frequent inundated area.	Increased input from sediment rich in nutrients will deteriorate habitat conditions.
Bankfull	125% bankfull can be important to create new pioneer sites along meandering rivers to colonize by succession precursors for hardwood forest development (Richter & Richter 2000).	125% bankfull can be important to create new pioneer sites along meandering rivers to colonize (Richter & Richter 2000). New pioneer sites are important for dispersal and recruitment for river corridor species (van Looy and Meire 2009).
Duration		
Inundation (days/year)	< 40 days/y	2 – 20 (max) days/y
Duration of flood event	Direct vital range: not longer than 60% of growing season and not two seasons in a row (no recovery; Glenz et al. 2006) Recruitment: no recruitment when flooded more than 30-40% of growing season (Glenz et al. 2006)	Less than 1 week in growing season
Timing		
Summer flooding	Chronic increase in inundating in summer can influence the species composition of existing sites.	Very sensitive to inundations in summer. Floodplain inundations in summer influence habitat zonation, (van Eck 2004). An increase of inundations in summer will lead to decrease of habitat suitability.
Summer drought	Can tolerate summer drought (Glenz et al. 2006)	River corridor species (Stromtalpflanzen) seem to have the advantage on being able to withstand flooding, while also being able to cope with dry circumstances due to high drainage capacity of the elevated floodplain parts they occupy. Changes in either of these parameters will influence distribution negatively. (Burkhart 2001)

Exact dry grassland and hardwood forest community composition varies across Europe depending on eco-region and adaptation to existing flood regimes and local management. Management such as grazing, mowing or cutting, and flood regime both influence vegetation composition (Gerard et al. 2008). Therefore, to assess effects of climate change on the European scale, the relative shift of parameters is the most important factor. Important to distinguish are the acute sensitivity of individuals and the sensitivity of populations or communities to chronic hydrologic alteration (Merritt et al. 2009). The latter is important to sustain the dry grasslands and floodplain forests on the long term (Geerling et al. 2006).

Changes in flood magnitude as found under most climate projections seem to point toward a reduction of flood volumes (Figure 3.2). Consequently, the floodplain area can decrease and formerly inundated floodplain forests and dry grasslands will be colonized by more terrestrial species or invaders (Predick & Turner 2008). This will directly lead to a decline of habitat area for these vegetation types.

Secondly, a decrease of flood volume can lead to morphologically less active systems, especially in the upland areas. However, this can lead to an initial increase in softwood forest establishment as deduced from Marston et al. (2001). The loss of floodplain dynamics will eventually lead to a loss of floodplain diversity, also for dry grasslands.

An increase in flood volume will increase floodplain area. A gradual change will affect current locations of dry grassland and hardwood stands as they are inundated more frequently or with longer durations. However, available habitat may shift towards formerly dry areas. Secondly, the affected rivers can become more morphologically active and can rejuvenate existing habitats.

Duration of overbank flows is decreasing for most rivers with agreement of all three climate projections (Figure 3.3). Change is up to 5 days per year or more. Overall consequences are similar as to reduction in flood magnitude, a decrease of habitat availability. However, as other more frequently inundated areas become suitable, i.e. less inundated, habitat for dry grassland and floodplain forest may shift towards these.

The results show that the timing of flood peaks may shift, in most agreement the floods appear earlier in the season. Most influential is timing for reproduction, or seed dispersal for genera like Willow (Salix spp.) or Poplar (Populus spp.). These genera time their seed release in such a way that conflicts between different species of Salix spp. and Populus spp. are minimised. Floods affect the habitat availability, and change in flood timing can decrease the available habitat for some of these species. Additionally, vegetation reacts also to spring temperature changes and timing of these may also change due to climate change. This can either mitigate or enhance the consequences of changes in flood timing. Recent studies support the notion that floods during the growing season may be particularly important from the ecological point of view by affecting plant distribution and survival. In contrast, the effect of winter flood timing on vegetation is not easily determined and regarded less influential (van Eck et al. 2004). A differentiation between summer and winter floods was not carried out in this study, but could lead to improvements in future ecological impact assessments.

Fauna (Fish)
A reduction in flood magnitude influences the connectivity of the landscape, notably for aquatic habitats (Petts & Amoros 1996). In combination with the expected reduction of flood duration, habitat availability will be less for aquatic species. Fish production will probably decrease when flood volume decreases as can be deduced from Lindholm et al. (2007) for tropical systems. Additionally the expected change of flood timing towards earlier spring can make habitats unreachable. A decrease in morpho-dynamics for upper reaches can lead to a less diverse riverine landscape with lower availability of aquatic habitat.

3.5 Conclusion

River ecosystems including floodplain wetlands belong to the most threatened ecosystems on the planet with a proceeding loss in biodiversity. Regarding the health of these ecosystems, flows above bankfull flow play a crucial role as ecological and biological processes change when the river is linked to the associated floodplain. However, floodplain inundation is often disturbed by anthropogenic factors such as river regulation, channelization, wetland drainage, and water abstractions. Climate change is altering volume, duration and timing of future floodplain inundation events, and therefore constitutes an additional threat to river ecosystems.

Results of this study indicate that climate change impacts floodplain inundations over large regional scales in Europe in the 2050s. In snow affected catchments (i.e. in Central, Eastern and Northern Europe as well as in mountainous areas) duration and volume of inundation are expected to decrease and inundation may appear earlier in the year.
Here, inundation usually occurs in spring when snow melt falls together with strong precipitation. Due to an increased temperature, the proportion of precipitation falling as snow is reduced as well as extent and duration of snow cover. This leads to earlier snow melt within the year and considerably reduced snow melt induced flood peaks. According to this, on the selected floodplains in Central, Eastern and Northern Europe, the extent of floodplain habitat is reduced in the 2050s compared to current conditions. Consequently, floodplain forests and dry grasslands are expected to be colonized by more terrestrial species or invaders. As habitats for spawning, nursery, foraging and escaping from predation are narrowed, fish can be negatively affected. All in all, important ecosystem services such as biodiversity maintenance, nutrient removal, detoxification, carbon storing, floodplain productivity and fish production are likely to decrease on the selected floodplain wetlands in Eastern and Northern Europe. Hence, to avoid economic losses and to assure a natural pattern of floodplain inundation, it is important to consider the impact of climate change in the planning of future adaptation measures.

In warmer regions, inundation strongly depends on the simulated precipitation patterns. Here, the choice of the climate projection has a bigger influence on the hydrological results compared to areas where snow melt induced flood peaks occur. In our analysis, precipitation patterns modelled by three different GCMs representing two different emission scenarios lead to contradictory results for future changes in volume, duration and timing of floodplain inundation. This finding reflects the uncertainties of current climate modelling for specific seasons and therefore, no consistent conclusions could be drawn for rivers in Spain, France, Southern England and the Benelux countries.

The simulation of flood scenarios and hence bankfull flow events could be improved by the usage of daily climate to force WaterGAP. Instead of applying the 'scaling approach' to get a higher spatial resolution climate input according to measured data, time series as calculated by RCMs (Regional Circulation Model), but bias-corrected, could be used to consider changes in future climate variability. One aim in riparian ecology is to build a general framework for predictions of e.g. vegetation response to altering inundation conditions. In this study, the ecological impact analysis has been performed in an indicative and qualitative manner. For our future work, we will improve ecological consequences for fish and fauna by distinguishing upland and lowland rivers as well as incorporating a more systematic approach by considering functional classifications of species that respond in similar ways to components of hydrological regimes.

Acknowledgements
The financial support from the EU's Research Framework Programme (FP6) under contract no. 036822 (SCENES) is gratefully acknowledged. The authors express their appreciation to Prof. Mike Acreman from the Centre for Ecology and Hydrology in Wallingford for fruitful discussions.

References

Adamowski, K., Liang, G.C. & Patry, G.G. 1998 Annual maxima and partial duration flood series analysis by parametric and non-parametric methods. Hydrol. Processes 12, 1685-1699.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T. & Siebert, S. 2003 Development and testing of the WaterGAP 2 global model of water use and availability. Hydrological Sciences 48(3), 317-337.

Alcamo, J., Flörke, M. & Märker, M. 2007 Future long-term changes in global water resources driven by socio-economic and climate changes. Hydrol. Sci. J., 52(2) April 2007.
Amoros, C. & Wade, P.M. 1996 Ecological successions. Chap. 10 in G. Petts & C. Amoros (eds). Fluvial Hydrosystems, Chapman & Hall, London, 211-241.

Arnell, N.W. 1999 Climate change and global water resources. Global Environ. Change, 9, 31-49.

Ashkar, F. & Rousselle, J. 1983 Some remarks on the truncation used in partial flood series models. Water Resour. Res. 19(2), 477–480.

Begueria, S. 2005 Uncertainties in partial duration series modelling of extremes related to the choice of the threshold value. J. Hydrol., 303, 215-230.

Bunn, S.E. & Arthington, A.H. 2002 Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. Springer-Verlag New York Inc. 30(4), 492-507.

Burkhart, M. 2001 River corridor plants (Stromtalpflanzen) in Central European lowland: a review of a poorly understood plant distribution pattern. Global Ecol. Biogeogr., 10(5), 449-468.

Casanova, M.T. & Brock, M.A. 2000 How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? Plant Ecology 147 (2), 237-250.

Castro, J.M. & Jackson, P.L. 2001 Bankfull Discharge Recurrence Intervals And Regional Hydraulic Geometry Relationships: Patterns In The Pacific Northwest, USA. J. Am. Water Resour. Assoc. 37(5), 1249-1262.

Commission of the European Community (CEC) 1992 Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Official journal of the European communities No L 206/7.

Choulakian, V., El-Jabi, N. & Moussi, J. 1990 On the distribution of flood volume in partial duration series analysis of flood phenomena. Stochastic Hydrology and Hydraulics 4, 217-226.

Christensen, J.H. & Christensen, O.B. 2007 A summary of the PRUDENCE model projections of changes in European climate by the end of this century, Clim. Change 81, 7-30.

Cunnane, C. 1979 A note on the Poisson assumption in partial duration series models. Water Resour. Res. 15(2), 489-494.

Dankers, R. & Feyen, L. 2008 Climate change impact on flood hazard in Europe: An assessment based on high-resolution climate simulations. J. Geophys. Res., 113, 1-17.

Davison, A.C. & Smith, R.L. 1990 Models for exceedances over high thresholds. J. R. Stat. Soc. B 52(3), 393–442.

Deidda, R. & Puliga, M. 2009 Performances of some parameter estimator of the generalized Pareto distribution over rounded-off samples. Phys. Chem. Earth., 34, 626-634.

Döll, P., Kaspar, F. & Lehner, B. 2003 A global hydrological model for deriving water availability indicators: model tuning and validation. J. Hydrol., 270, 105-134.

Döll, P. & Lehner, B. 2002 Validation of a new global 30-min drainage direction map. J. Hydrol., 258, 214-231.

Dunne, T. & Leopold, L.B. 1978 Water in Environmental Planning, W.H. Freeman and Company, New York, 33-490.

Dury, G.H. 1977 Underfit streams: retrospect, perspect and prospect. In Gregory, K.J. (Ed.), River Channel Changes, 281-293.

EEA 2004 Corine Land Cover 2000 - Mapping a decade of change Document Actions, Tech. Rep. Brochure No 4/2004, EEA (European Environment Agency).

Flörke, M. & Alcamo, J. 2005 European Outlook on Water Use, Technical Report prepared for the European Environment Agency. Kongens Nytorv. 6. DK-1050. Copenhagen, DK URL: // http: scenarios.ewindows.eu.org/reports/fol949029, 2005.

Frederick, K.D. & Major, D.C. 1997 Climate change and water resources. Clim. Change **37**, 7-23.

Geerling, G.W., Ragas, A.M.J., Leuven, R.S.E.W., van den Berg, J.H., Breedveld, M., Liefhebber, D. & Smits, A.J.M. 2006 Succession and rejuvenation in floodplains along the river Allier (France). Hydrobiologica 565, 71-86

Geng, S., Penning, F.W.T. & Supit, I. 1986 A simple method for generating daily rainfall data', Agric. For. Meteor. 36, 363-376.

Gerard, M., El Kahloun, M., Rymen, J., Beauchard, O. & Meire, P. 2008 Importance of mowing and flood frequency in promoting species richness in restored floodplains. J. Appl. Ecol. 45(6), 1780-1789.

Glenz, C., Schlaepfer, R., Iorgulescu, I. & Kienast, F. 2006 Flooding tolerance of Central European tree and shrub species. Forest Ecology and Management 235(1-3), 1-13

Harman, W.A., Jennings, G.D., Patterson, J.M., Clinton, D.R., Slate, L.O., Jessup, A.G., Everhart, J.R. & Smith, R.E. 1999 Bankfull Hydraulic Geometry Relationships for North Carolina Streams. In Proceedings of the Wildland Hydrology Symposium, ed. Olsen D.S., Potyondy J.P.. Bozeman, MT: American Water Resources Association.

Henrichs, T. & Kaspar, F. 2001 Baseline-A: A reference scenario of global change. In: Lehner, B., T. Henrichs, P. Döll, J. Alcamo (Eds.): EuroWasser: Model-based assessment of European water resources and hydrology in the face of global change. Kassel World Water Series – Report No. 5, 4.1-4.8, Center for Environmental Systems Research, University of Kassel.

Hosking, J.R.M. & Wallis, J.R. 1987 Parameter and Quantile estimation for the Generalized Pareto Distribution. Technometrics 29(3), 339-349.

Hughes, F.M.R. 1997 Floodplain biogeomorphology. Progress in Physical Geography 21(4), 501-529

IPCC 2001 Intergovernmental Panel on Climate Change: Climate Change 2001 - The Scientific Basis, J. T. Houghton et al., Eds., Cambridge University Press, Cambridge, UK, New York, NY, USA.

IPCC 2007 Intergovernmental Panel on Climate Change: Climate Change 2007—The Physical Science Basis, S. Solomon et al., Eds., Cambridge University Press, Cambridge, UK, New York, NY, USA.

Jenkins, M. 2003 Prospects for biodiversity. Science 302, 1175–1177.

Junk, W.J., Bayley, P.B. & Sparks, R.E. 1989 The flood pulse concept in river-floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences, 110–127.

Kite, G.W. 1977 Frequency and Risk Analysis in Hydrology. Water Resources Publications, Fort Collins, CO.

Kundzewicz, Z., Mata, L., Arnell, N., Döll, P., Kabat, P., Jimenez, B., Miller, K., Oki, T., Sen, Z., and Shiklomanov, I.: Freshwater resources and their management. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK, 2007.

Lang, M., Ouarda, T.B.M.J. & Bobee, B. 1999 Review - Towards operational guidelines for over-threshold modelling. J. Hydrol. 225, 103-117.

Langbein, W.B. 1949 Annual floods and the partial duration series. EOS Trans., Am. Geophys. Union 30, 879-881.

Langhans, S.D. & Tockner, K. 2006 The role of timing, duration, and frequency of inundation in controlling leaf litter decomposition in a river-floodplain ecosystem (Tagliamento, northeastern Italy). Oecologia 147 (3), 501-509.

Lehner, B., Döll, P., Alcamo, J., Henrichs, T. & Kaspar, F. 2006 Estimating the impact of global change on flood and drought risks in Europe: A continental, integrated analysis. Clim. Change 75, 273-299.

Lehner, B., Verdin, K. & Jarvis, A. 2008 New global hydrography derived from spaceborne elevation data, Eos, Transactions, AGU, 89(10),93-94, http://hydrosheds.cr.usgs.gov (21-Oct-2008).

Leopold, L.B., Wolman, M.G. & Miller, J.P. 1964 Fluvial Processes in Geomorphology, Dover Publications (June 28, 1995).

Lindholm, M., Hessen, D.O., Mosepele, K. & Wolski, P. 2007 Food webs and energy fluxes on a seasonal floodplain: The influence of flood size. Wetlands 27(4):775-784.

Madsen, H., Rasmussen, P.F. & Rosbjerg, D. 1997 Comparison of annual maximum series and partial duration series methods for modeling extreme hydrologic events 1. At-site modeling. Water Resour. Res. 33(4), 747-757.

Madsen, H. & Rosbjerg, D. 1997 The partial duration series method in regional index-flood modeling. Water Resour. Res. 33(4), 737-746.

Malmqvist, B. & Rundle, S. 2002 Threats to the running water ecosystem of the world. Environ. Conserv. 29(2), 134-153.

Marston, R.A., Girel, J., Pautou, G., Piegay, H., Bravard, J.-P., Arneson, C. 2001 Channel metamorphosis, floodplain disturbance, and vegetation development: Ain river, France. Geomorphology 13, 121-131.

Merritt, D.M., Scott, M.L., Poff, N.L., Auble, G.T. & Lytle, D.A. 2009 Theory, Methods and tools for determining environmental flows for riparian vegetation: riparian vegetation-flow response guilds. Freshwater biology (in press).

Mitchell, T. D., Carter T.R., Jones, P.D., Hulme, M. & New, M. 2004 A Comprehensive Set of High-resolution Grids of Monthly Climate for Europe and the Globe: the Observed Record (1901-2000) and 16 Scenarios (2001-2100), Tyndall Centre for Climate Change Research, Working Paper 55.

Mitchell, T. D. & Jones, P. D. 2005 An Improved Method of Constructing a Database of Monthly Climate Observations and Associated High-resolution Grids, Int. J. Climatol. 25(6), 693-712.

Mosley, M.P. 1981 Semi-determinate hydraulic geometry of river channels, South Island, New Zealand. Earth Surface Processes and Landforms 6, 127-137.

Navratil, O., Albert, M.B., Herouin, E. & Gresillon, J.M. 2006 Determination of bankfull discharge magnitude and frequency: comparison of methods on 16 gravel-bed river reaches. Earth Surface Processes and Landforms 31, 1345-1363.

Nguyen, V.T.V. 2002 On Modelling of Extreme Hydrologic Processes, International Workshop on Non-Structural Measures for Water Management Problems, London, Ontario, October 18-20, UNESCO IHP-V Technical Documents in Hydrology No. 56, 167-180.

Nilsson, C., Reidy, C.A., Dynesius, M. & Revenga, C. 2005 Fragmentation and Flow Regulation of the World's Large River Systems. Science 308, 405-408.

Petit, F. & Pauquet, A. 1997 Bankfull Discharge Recurrence Interval in Gravel-bed Rivers. Earth Surf. Processes Landforms 22, 685-693.

Petts, G.E. & Amoros, C. 1996 Fluvial Hydrosystems. Chapman and Hall, London, Weinheim, New York, Tokyo, Melbourne, Madras, 322 pp.

Petts, G.E. 2000 A perspective on the abiotic processes sustaining the ecological integrity of running waters. Hydrobiologia 422-423(0), 15-27.

Pickands, J. 1975 Statistical inference using extreme order statistics. Ann. Stat. 3(1), 119-131.

Predick, K.I. & Turner, M.G. 2008 Landscape configuration and flood frequency influence invasive shrubs in floodplain forests of the Wisconsin River (USA). Journal of Ecology 96(1), 91-102.

Prudhomme, C. & Davies, H. 2009 Assessing uncertainties in climate change impact analyses on the river flow regimes in the uk. part 1: baseline climate. Clim. Change 93, 177–195.

Richter, B.D. & Richter, H.E. 2000 Prescribing Flood Regimes to Sustain Riparian Ecosystems along Meandering Rivers. Conserv. Biol. 14(5), 1467-1478.

Rosbjerg, D., Madsen, H. & Rasmussen, P.F. 1992 Prediction in Partial Duration Series With Generalized Pareto-Distributed Exceedances. Water Resources Research 28(11), 3001-3010.

Shane, R. M. & Lynn, W. R. 1964 Mathematical model for flood risk evaluation. J. Hydraul. Div. Am. Soc. Civ. Eng., 90(HY6), 1-20.

Sweet, W.V. & Geratz, J.W. 2003 Bankfull Hydraulic Geometry Relationships and Recurrence Intervals for North Carolina's Coastal Plain. J. Am. Water Resour. Assoc. 39(4), 861-871.

Tockner, K., Malard, F. & Ward, J.V. 2000 An extention of the flood pulse concept. Hydrol. Processes 14, 2861-2883

Todorovic, P. & Zelenhasic, E. 1970 A stochastic model for flood analysis. Water Resour.Res. 6(6), 1641-1648.

Van de Wolfshaar, K.E., Ruizeveld De Winter, A.C., Straatsma, M.W., Van den Brink, N.G.M. & de Leeuw, J.J. 2009. Estimating spawning habitat availability in flooded areas of the river Waal, the Netherlands. River Res. Appl. (not issued yet). DOI: 10.1002/rra.1306

Van Eck, W.H.J.M., Van der Steeg, H.M., Blom, C.W.P.M. & De Kroon, H. 2004 Is tolerance to summer flooding correlated with distribution patterns in river floodplains? A comparative study of 20 terrestrial grassland species. Oikos 107(2), 393

Van Geest, G.J., Wolters, H., Roozen, F.C.J.M., Coops, H., Roijackers, R.M.M., Buijse, A.D. & Scheffer, M. 2005 Water-level fluctuations affect macrophyte richness in floodplain lakes. Hydrobiologica 539(1), 239-248.

Van Looy, K. & Meire, P. 2009. A conservation paradox for riparian habitats and river corridor species. J Nat Conserv 17(1), 33-46.

Verzano, K., Hunger, M. & Döll, P. 2005 Simulating river flow velocity on global scale, Adv. Geosci. 5, 133-136.

Verzano, K. & Menzel, L. 2009 Hydrology in Mountain Regions: Observations, Processes and Dynamics, IAHS-Publication 326, chap Snow conditions in mountains and climate change - a global view, pp 147–154.

Wang, Q.J. 1991 The POT model described by the generalized Pareto distribution with Poisson arrival rate. J. Hydrol. 129, 263-280.

Weedon, G.P., Gomes, S., Viterbo, P., Österle, H., Adam, J.C., Bellouin, N., Boucher, O. & Best, M. 2010 The WATCH Forcing Data 1958-2001: A meteorological forcing dataset for land surface- and hydrological-models. WATCH Technical Report 22. http://www.eu-watch.org.

Williams, G.P. 1978 Bank-Full Discharge of Rivers. Water Resour. Res. 14(6), 1141-1153.

Woodyer, K.D. 1968 Bankfull frequency in Rivers. J. Hydrol. 6, 114-142

4 Water for Nature 3 – Ecosystem services of wetlands

4.1 Introduction

An appropriate hydrological regime within a wetland is essential to maintain goods and services. This regime is related to the source of water which is different for particular kinds of wetlands. This indicator provides an overview of ecosystem services of European wetlands based on representative sample of 103 protected wetlands larger than 5000 ha. Six major ecosystem services of wetlands were classified namely: biodiversity in terms of plants and birds, biomass production, nutrient removal, carbon storage and fish spawning. Each of the six services was treated equally in the evaluation approach. For each of the analyzed ecosystem services, hydrological drivers were defined that become responsible for the proper function of each service. Hydrological processes responsible for the proper wetland function were defined according to the wetland type and modelled in different climate change and socio-economic scenarios for the year of 2050 with the WaterGAP model. Information on hydrological drivers in particular scenarios were obtained and referred to the ecosystem services demanding. Modelled alteration in hydrology of wetlands was responsible for either the proper function of particular services in future or the loss of analyzed services.

4.2 Method

Calculation approach

The analyzed sample contains the set of protected wetlands of Europe whose size is bigger than 5000 ha. The set of wetlands was prepared according to the procedure presented on Figure 4.1.

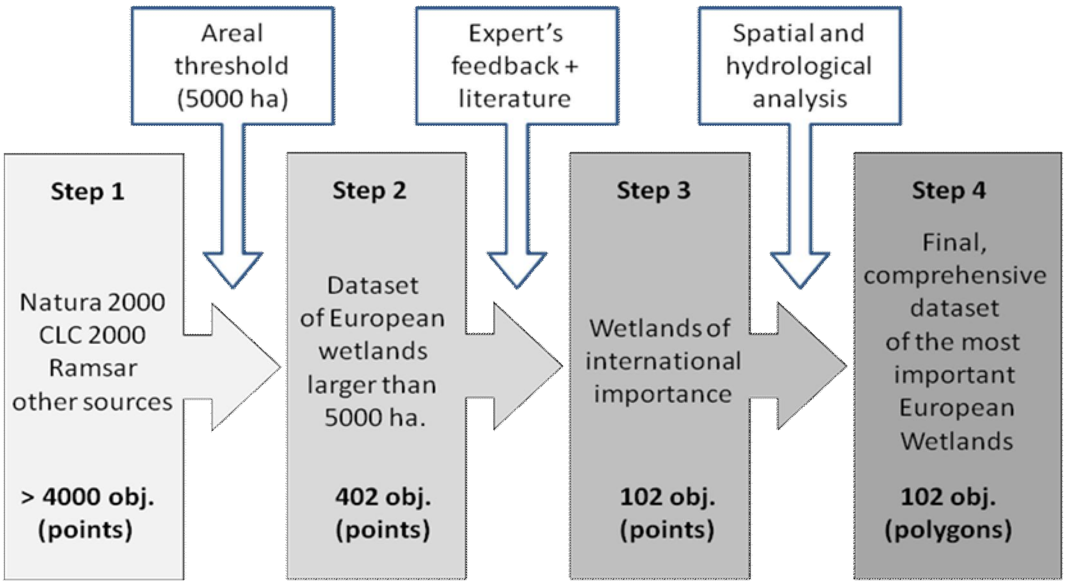

Figure 4.1 Wetland selection procedure

In the first step of the approach, data on wetlands were collected from readily and practically available sources. For example, Natura 2000 and Corine Land Cover 2000 were used as a spatial reference data. In some cases remote sensing open-source data were used to precisely locate wetlands. Such an approach brought together more than 4000 wetlands within Europe. These are necessarily large wetlands as analysis of this European scale did not permit small wetlands to be incorporated. In the second step, results of the analysis were selected with the areal threshold, which for the purpose of this study was set at 5000 ha.

As a result of this filtering, 402 wetlands were captured as an input to our database for the subsequent steps of the analysis. In the third step, the database of 402 wetlands was divided into regional, country-wide sets. Datasets were sent to national wetland experts for their feedback. Additionally, experts were asked for the details regarding selected wetlands, mainly on water supply mechanisms and a hydrological description of particular wetland. Literature studies were also undertaken to permit access to scientific publications on wetlands of interest. The output of this exercise was a wetland dataset with 102 entries. In the fourth step, a dataset of verified wetlands spatial extent was created. For the obtained data, the geodatabase was set up with GIS software. The final reviewed database was unified into one comprehensive dataset that consists of tables and ESRI shapefiles (*.shp) associated in topologic layers (Figure 4.2). All the data provided in feature sets are referred to WGS84 coordinate system.

Ecosystem services of wetlands were described as the individual wetland's impact on particular components of the environment. Information on the status of wetlands gathered into the comprehensive dataset was taken from scientific publications and official websites that described particular areas of interest. Some of the data were supplied by national wetland experts and authorities. In the presented approach, six ecosystem services were described. Three services out of six were focused on biotic components of the environment and the remaining three services on the physical and economical dimension of wetlands.

Service 1 - Habitats of rare bird species, that have strict requirements for constant water availability. Evaluation of this service was achieved through the species of bird, which need seasonally inundated wetlands or wetlands with a shallow groundwater level. For each wetland, bird species were selected by their protection status and a level of their population. Although most of the bird species present in wetland areas are associated with certain elements of the hydrological regime, only birds directly dependent on open-water and groundwater were valued with 1. Those birds were: Aquatic warbler (*Acrocephalus paludicola*), Black-throated Diver (*Gavia arctica*), Red-throated diver (*Gavia stellata*), Common snipe (*Gallinago gallinago*), Geese (*Anser anser, Anser erythropus*), Crane (*Grus grus*), species of waders (*Charadriiformes*), Herons (*Ardea cinerea, Ardea alba, Ardea purpurea*), Night heron (*Nycticorax nycticorax*), Great-crested Grebe (*Podiceps cristatus*), Pelican (*Pelecanus*), Yellow wagtail (*Motacilla flava*), Great cormorant (*Phalacrocorax carbo*), certain species of ducks (*Anas clypeata, Aytha nyroca*), Corncrake (*Crex crex*), Eurasian curlew (*Numenius arquata*), Common shelduck (*Tadorna tadorna*), Osprey (*Pandion haliaetus*), Smew (*Mergellus albellus*), Little tern (*Sternulla albifrons*), Eurasian golden plover (*Pluvialis apricaria*), Ringed plover (*Charadrius hiaticula*), Pink flamingo (*Phoenicopterus roseus*), Glossy ibis (*Plegadis falcinellus*), Dunlin - (*Calidris alpina*), Wood sandpiper (*Trinia glareola*), Ruff (*Philomachus pugnax*), Meadow pipit (*Anthus pratensis*), Little ringed plover (*Charadrius dubius*), Little grebe (*Podiceps ruficollis*) and the Pied avocet (*Recurvirostra avosetta*). Other birds were valued with 0. There is a number of wetlands with birds specified above, but if there was any written remark, that the population of the certain bird is weak and endangered, the value 0 of the wetland bird service was given.

Service 2 - Fish spawning. In cases of wetlands seasonally and constantly inundated, flooded areas play an important role in fish spawning and population development. As the database of wetlands consists only of a general hydrological description, in the method presented, it was assumed that every wetland ecosystem defined as marsh and swamp can play an important role in fish spawning processes. Floodplains and lowlands inundated in spring provide relatively warmer water, than in the river channel. Such areas are widely used by the early foraging fry (Górski et al, 2010). In cases of some freshwater fish species, a correlation was observed between the flood frequency and the efficiency of natural spawning. Within Europe, the main fish that is adapted to natural spring flooding in the valleys is the Northern pike (*Esox lucius*).

Therefore, all the ecosystems that have been developed in seasonally flooded conditions with shorter (marshes) and longer inundation periods (swamps) and that were defined within the natural spatial extent of the Northern pike population (Backiel, 1965) were valued in the fish spawning service. Other types of inland wetlands – fens and bogs were valued with 0. Estuaries were valued with 1, as the brackish water conditions become specific spawning conditions of a number of species of freshwater saltwater fish.

Service 3 - Vegetation service. Hydrological conditions of wetlands, such as shallow groundwater occurrence and seasonal or permanent flooding, lead to the development of rare plant communities. Thus, the natural function of water circulation systems within a wetland permits valuable plant communities to develop. In terms of feedback, the presence of particular plant species is often used as an indicator of water supply mechanisms and the hydrological status of the wetland. The main criterion of this study with regard to the wetland vegetation service was the presence of certain plant communities within the range of the wetland. Only the wetland plant communities that are strongly dependent on wet conditions were taken into account. Criteria established by Natura 2000 were used as an indicator (Table 4.1). Value 1 was assigned for the wetland vegetation service, if at least one of the valuable plant communities is present within the range of particular wetland.

Table 4.1 Wetland plant communities that were taken into account in the analysis of wetland vegetation ecosystem service

Natura 2000 code	Plant community
3150	Ox-bow lakes and natural eutrophic waters with *Nympheion* and *Potamion*
3270	Flooded muddy river banks
6120	*Koelerion glaucae*
6410	*Moninion*
6430	*Adenostylion alliarie* and *Convulvuletalia sepium*
6510	*Arrhenatherion elatrioris*
7140	Transitional bogs, mostly with *Scheuchzerio-Caricetea*
7230	Mountain and lowland alcalic mires
9170	*Galio-Carpinetum, Tilio-Carpinetum*
91 D0	*Vaccinio uliginosi-Betuletum pubescens, Vaccinio uliginosi-Pinetum, Pino-mugu Sphagnetum, Sphagno girgensohnii Piceetum;*
91 E0	*Salicetum albo-fragilis, Populetum albae, Anenion-glutinoso-incanae, Alnetum;*
91 F0	*Ficario-Ulmeteum*
91 I0	*Quercetelia pubescenti - petraeae*

Service 4 - Carbon storage. Peatlands ecosystems function as a sinks for carbon, when the peat forming processes are the main force driving to the accumulation of organic matter. Thus, all the mires which hydrological conditions have not been significantly degraded were valued with 1.

Service 5 - Nutrient removal, which is a crucial process in the case of wetlands situated in the river valleys, where over-bank inundation occurs. In this approach, marshes were classified as providing the nutrient removal service. Also certain swamps located in floodplains provide the service of nutrient removal.

Service 6 - Production of wetlands' specific goods, that is, an effect of wetland function was classified as an economic service of the ecosystem. Within Europe, the main goods that come from wetlands for human economic activity are reed for roofing and willow harvest as well as extensive meadows harvest (those one which are economically supported by agro-environmental schemes).

For the purpose of this study, an arbitrarily chosen threshold was used to identify the hydrological threats to wetland ecosystem services. The ability of wetlands to provide the services depends on number of factors, particularly the hydrological regime. Due to the general character of this study we have concentrated on the concept of a “necessary” condition rather than on “fair” condition for wetlands functioning. Thresholds have been defined in such a manner that meeting them ensures functioning of the wetland. The other factors which can also impact the particular service (e.g. water quality, small modification of hydrological regime, land use forms on floodplain and in the surrounding areas, management options for open vegetation including grazing and mowing, etc) were not included in this exercise. It means that the results are biased to the situations where our current knowledge says with high level of certainty that particular ecosystem service will be lost:

Habitats of rare bird species – the service is lost if there is a reduction of flood volume in riparian wetlands (swamps or marshes) or there is a change in timing of flooding by more than one month. In the case of mires, change of surplus of water to water deficit indicates a decrease of groundwater level which impacts negatively the wading birds.

Fish spawning – the service is lost if there is no flooding in riparian wetlands (swamps or marshes) or there is a change in the timing of flooding by more than one month. In the case of estuaries, the loss of at least 50% of freshwater inflow to an estuary results in a salinity increase.

Habitat for wetland vegetation – is lost when there is a lack of flooding in riparian wetlands or the peat forming process reverses (moorshing) in the case of bogs and fens. The lack of flooding is the most often referred reason for losing riparian forests and sedge vegetation. The mire habitat becomes a peatland habitat with an abundance of nutrients compared to nutrient limitation in case of growing peat. This second process in mires is indicated by changing from surplus water to a water deficit in multi-year water balances.

Carbon storage – changes to carbon emission in mires (bogs and fens) when, instead of a peat growing process, the decay of peat occurs due to a moorshing process. Again it is indicated by the change in water balance.

Nutrient removal – halting this function of floodplain (riparian wetlands) is first of all caused by stopping the flooding process.

Production of goods - is lost when there is an absence of floods onto riparian wetlands or groundwater-fed wetlands (fens and bogs) become too wet for any agricultural practices due to swampy conditions. The lack of inundations has fatal implications for the reed beds (source of so called “roofing reed”) and the lack of flooding impacts the willow communities. We assumed that the mire habitat becomes too wet for any agricultural purposes when positive water balance has been doubled.

In the first step, the number of ecosystem services provided by wetlands in the reference year was estimated (Figure 4.2). In the second step, the impact of future climate change and water management scenarios was introduced. Using results from the WaterGAP model, the major changes in hydrological regime (i.e. precipitation, groundwater recharge and river flow) were indentified. If the changes in the components were greater than arbitrary chosen threshold the particular service was flagged as endangered.

Input data
- Daily discharges (WaterGAP)
- Monthly precipitation and PET data form MIMR/ A2 and IPCM4/A2 climate models
- Yearly average groundwater recharge (WaterGAP)
- National wetlands inventories

Spatial and temporal scales
Annual averages basing on daily or monthly data were applied in analysis. For estuaries and riparian wetlands calculations were performed for the main grid where wetland is located. For bogs and fens it was an average of all grids which are touching the wetland area. The services were identified for the area of wetland only.

Thresholds and critical values
The thresholds used to define the level of water stress are:

0 - 1 service lost	= minor change
2 – 3 services lost	= change
> 4 services lost	= significant change

Validation
We make direct use of WaterGAP output, which has already been validated. There was no additional validation method for the Nature 3 indicator aspect, as we have no empirical proof for the system dysfunction due to water stress. The threshold values for the loss of particular ecosystem services were based on literature.

Uncertainty and sensitivity
- WaterGAP and Climate models (Modelling rainfall-runoff and water use at the large scale to cover entire Europe will have uncertainties as a result of scale itself and gaps in data. Projecting water use and availability for future scenarios is uncertain by its very nature. Alcamo *et al.* (2000) provide more information on the uncertainties involved and their order of magnitude).
- Survey and classification methods of wetlands
- Expert based services given in the wetlands inventories
- Thresholds choice for values where ecosystem service loss occurs

4.3 Results

4.3.1 Baseline scenario
According to the given criterions, the number of ecosystem services was defined for each of wetlands taken into account for the reference year of 2000 (Figure 4.2 and Table 4.2).

4.3.2 Future scenarios
The extent of changes in ecosystem services of wetlands fulfilled in applied scenarios are presented in Figures 4.3-4.6 (IPCM4A2 model) and Figures 4.7-4.10 (MIMRA2 model). A summary of ecosystem services alteration in particular scenarios is given in Tables 4.2, 4.3, 4.4 and 4.5.

General pattern
Within 2050 time horizon the losses of ecosystem services predicted using the IPCM4A2 climate data are approximately 50% higher than modelled with the MIMRA2 climate. Moreover, under the IPCM4A2 climate realisation, the significant difference between the two socio-economic scenarios analysed was that the modelled lost of services in the EcF scenario is approximately 20% higher than in the SuE scenario.

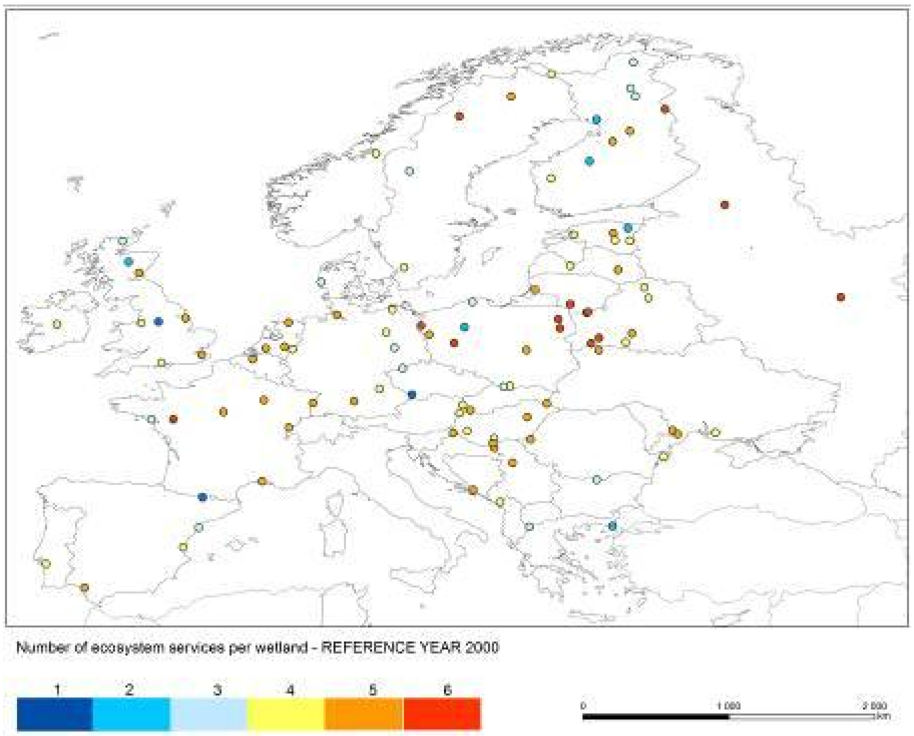

Figure 4.2 Number of ecosystem services in reference year 2000

Table 4.2 Statistics of ecosystem services of analyzed wetlands in reference year of 2000 and in particular scenarios in year of 2050

Year	**2000**	
Total number of services of analyzed wetlands	441	
Year	**2050**	
Number of services in analyzed scenarios	**IPCM4A2**	**MIMRA2**
Economy First	234	324
Fortress Europe	282	342
Policy Rules	277	342
Sustainability Eventually	272	319

Table 4.3 Percentage service lost within particular types of wetlands

Dominant type of wetland	**MIMRA2**					**IPCM4A2**				
	Economy First	Fortress Europe	Policy Rules	Sustainability Eventually	**Average lost of services**	Economy First	Fortress Europe	Policy Rules	Sustainability Eventually	**Average lost of services**
	%					**%**				
Bog	6	7	7	6	**6,5**	17	18	18	17	**17,5**
Fen	30	23	23	31	**26,8**	50	40	42	35	**41,8**
Marsh	31	24	24	34	**28,3**	55	40	40	47	**45,5**
Swamp	36	30	31	36	**33,3**	56	40	44	43	**45,8**
Estuary	24	30	30	24	**27,0**	51	43	43	43	**45,0**

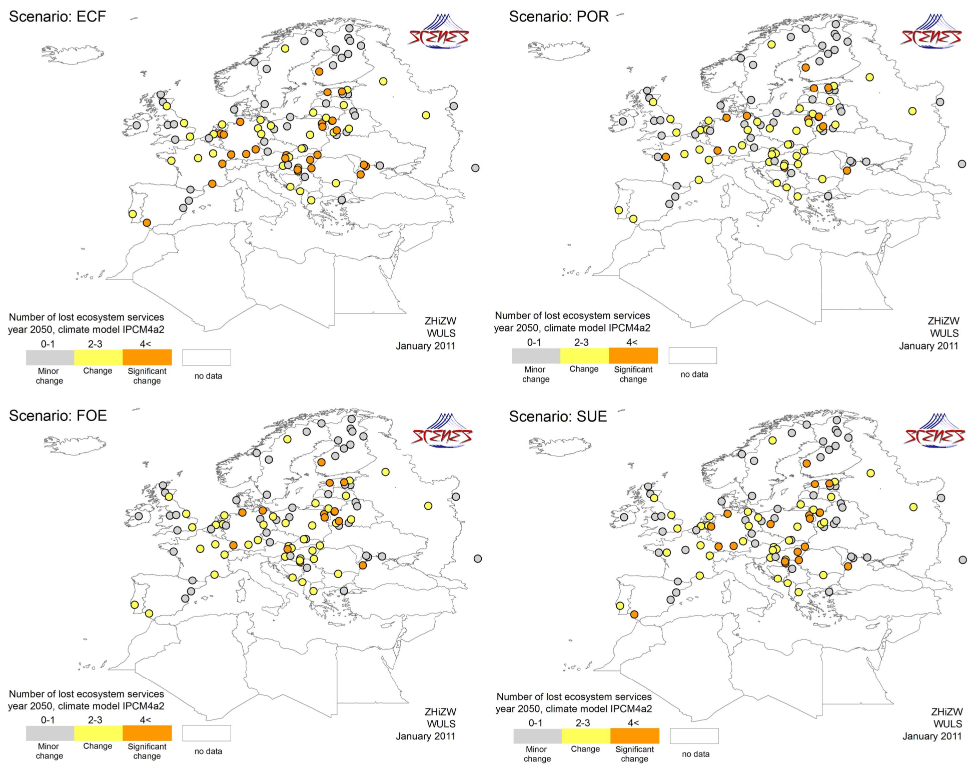

Figure 4.3 until 4.6 (left to right). Change in number of ecosystem services under the IPCM scenario. Economy First: Figure 4.3. Policy Rules: Figure 4.4. Fortress Europe: Figure 4.5. Sustainability eventually: Figure 4.6.

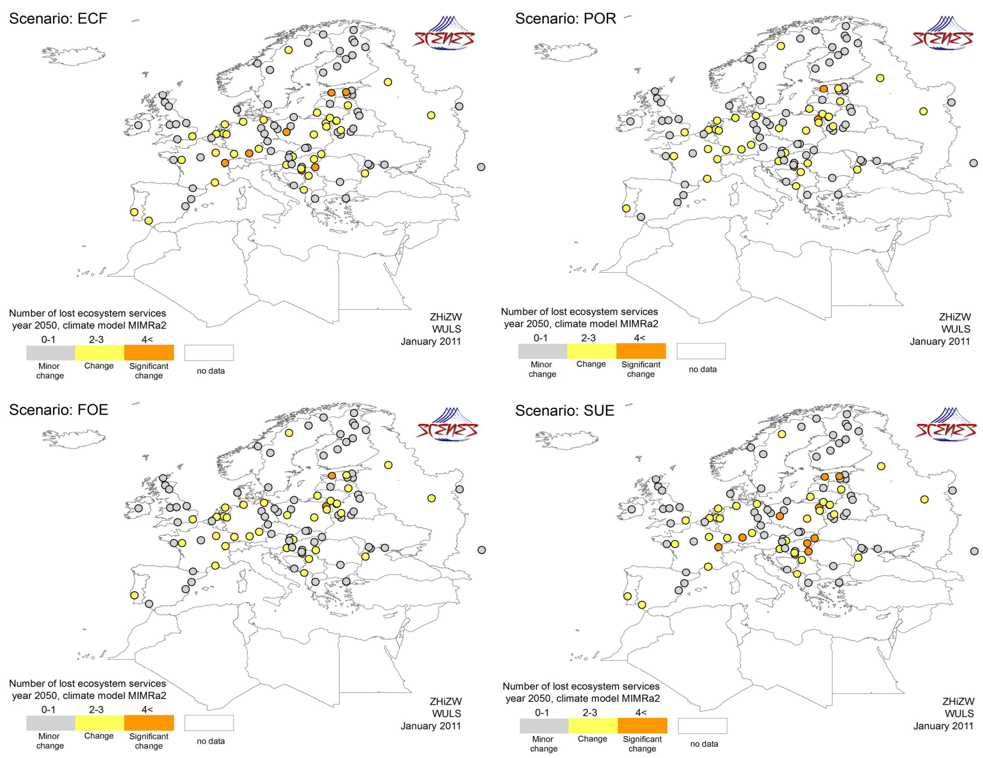

Figure 4.7 until 4.10 (left to right). Change in number of ecosystem services under the MIMR scenario. Economy First: Figure 4.7. Policy Rules: Figure 4.8. Fortress Europe: Figure 4.9. Sustainability eventually: Figure 4.10.

Table 4.4 Number of wetlands that provide particular ecosystem services in analyzed scenarios

Service	Number of objects in reference year	MIMRA2				IPCM4A2			
		Economy First	Fortress Europe	Policy rules	Sustainability Eventually	Economy First	Fortress Europe	Policy rules	Sustainability Eventually
Wetland vegetation	77	67	75	75	67	47	53	51	55
Wetland birds	75	32	35	35	30	21	35	35	28
Fish spawning	75	31	31	31	28	21	37	37	30
Carbon Storage	52	46	49	49	46	36	39	39	40
Nutrient Removal	83	75	80	80	75	54	60	58	60
Production	79	73	72	72	73	55	58	57	59
TOTAL	**441**	**324**	**342**	**342**	**319**	**234**	**282**	**277**	**272**

Table 4.5 Percentage reduction of number of wetland objects that provide particular ecosystem services in analyzed scenarios

Service	MIMRA2				IPCM4A2			
	Economy First	Fortress Europe	Policy rules	Sustainability Eventually	Economy First	Fortress Europe	Policy rules	Sustainability Eventually
	%				%			
Wetland vegetation	13	3	3	13	39	34	31	29
Wetland birds	57	53	53	60	72	53	53	63
Fish spawning	59	59	59	63	72	51	51	60
Carbon Storage	11	6	6	11	31	25	25	23
Nutrient Removal	10	4	4	10	35	30	28	28
Production	8	9	9	8	30	29	27	25

that within the applied assumptions in ecosystem services criterions, as well as in the parameter application used in WaterGAP to model particular elements of water balance, the hydrological and climatic stress for wetlands appear as most negatively impacted in the long-time horizon. Under the MIMRA2 climate realisation, a significant difference between socio-economic scenarios was not found. However, this can be explained by the land use changes simulated by LandSHIFT especially for cells in Belarus, Hungary, Ukraine and Lithuania. Here, water consumption is also higher under the EcF scenario and has a stronger impact on the hydrograph. However, this effect is outweighed by land use changes in the 2050s modelled for the two scenarios and the underlying climate conditions.

Socio economic and climate scenarios

The most significant loss of ecosystem services modelled for the 2050s can be observed in central Europe (Hungary, Germany, France, Belarus, Poland) under both climate realisations and socio-economic scenarios. Similarly, as within the short-time horizon, the greatest loss was observed in services of wetland birds and fish spawning. In the EcF scenario simulated with the IPCM4A2 climate projection, the most significant loss of all the services was noticed: the Wetland Bird service was reduced by 78%, Carbon Storage service was reduced by 60%, Fish Spawning service was reduced by 52%, Wetland Vegetation service was reduced by 39%, Nutrient Removal service was reduced by 35% and the Production service was reduced by 30%. In general, in both analysed time horizons, wetlands of Scandinavia and the British Isles have not lost as many services as wetlands in the European Lowlands.

It is likely that the water balance, assumed in this study to be the main indicator of changes of fens, bogs and the volume of the overbank flow (indicator for riparian wetlands) is more stable in regions of a mild, oceanic climate and the low variation and relatively high amount of precipitation. Services which seem to be the most sensitive to the analysed hydrological and climatic stress are associated with the avifauna and ichtyofauna.

The production of goods on wetlands seem to be the most resistant to hydrological and climatic pressures applied within analysed scenarios and hence the most constant among the analysed ecosystem services. It is important to stress that under the worst case scenario, Europe can anticipate losing almost half (207 of 441) of the services provided currently by wetlands.

4.4 Synthesis

Results from the analysis indicate that, within the whole pan-Europe area, the most extreme reduction of ecosystem services appeared in the Economy First scenario. Changes in water regime due to the socio-economical developments are clearly visible. In general, however. climate change dominates over socio-economic change.

Some changes can be observed in the Netherlands, Hungary and Baltic States. The negative impact on wetlands in the PoR scenario was observed mainly within wetlands of Poland and Belarus. Significant changes in a number of ecosystem services can be observed in SuE scenarios for the most of Europe. The most threatened are wetlands of Belarus, Croatia, France and Hungary. In general no major changes in ecosystem services were observed within the British Isles and Scandinavia in any of the scenarios. For a summary of observed changes in all regions, see Table 4.6.

Table 4.6 Regional observations on changes with respect to the baseline scenario

		Northern Africa	Western Europe	Northern Europe	Southern Europe	Central Eastern Europe	Eastern Europe	Western Asia
IPCM	EcF	*no data*	--	-	--	--	-	*no data*
	FoE	*no data*	-	-	-	--	-	*no data*
	PoR	*no data*	-	-	-	--	-	*no data*
	SuE	*no data*	-	-	-	--	-	*no data*
MIMR	EcF	*no data*	0	-	-	-	-	*no data*
	FoE	*no data*	0	-	-	-	0	*no data*
	PoR	*no data*	0	-	-	-	0	*no data*
	SuE	*no data*	0	-	-	-	0	*no data*

According to obtained results the biggest impact on wetland function seems to be the one of climate change. Social and economical aspects did not play significant role in analysed scenarios. In result, presented shifts in ecosystem services become mostly the consequence of climate forecasts applied in particular scenarios.

4.5 References

Acreman, M.C. & Miller, F. 2007 Practical approaches to hydrological assessment of wetlands lessons from the UK. In: Okruszko, T., Maltby, E., Szatyłowicz, J., Świątek, D., Kotowski, W. (eds) Wetlands; monitoring, modelling and management: Taylor & Francis, London.

Acreman, M.C. & Mountford, J.O. 2009 Wetlands. In: Ferrier, R., Jenkins, A. (eds) *Handbook of catchment management*. Blackwell, Oxford

Acreman, M.C., Blake, J.R., Booker, D.J., Harding, R.J., Reynard, N., Mountford, J.O. & Stratford, C.J. 2009 A simple framework for evaluating regional wetland ecohydrological response to climate change with case studies from Great Britain. *Ecohydrology* 2, 1-17.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T. & Siebert, S. 2003, Development and testing of the WaterGAP 2 global model of water use and availability. Hydrological Sciences 48(3), 317-337.

Aus der Beek, T., Flörke, M., Lapola, D.M., Schaldach, R., Voß, F., and Teichert, E. 2010, Modelling historical and current irrigation water demand on the continental scale: Europe. Adv. Geosci. (in press).

Backiel T. 1965, Freshwater fisheries in Poland, XVI Limnologorum Conventus in Polonia MCMLXV, Cracow, Polish Academy of Sciences, Hydrobiological Committee, Cracow, PWN.

Barbier, E.B. , Wetland as Natural Assets. Hydrological Sciences Journal this volume

Biggs, B.J.F., Ibbitt, R.P., Jowett, I.G. 2008, Determination of flow regimes for protection of in-river values in New Zealand: an overview Ecohydrology and Hydrobiology, 8, 1, 17-29

Bullock, A., & Acreman, M.C. 2003, The role of wetlands in the hydrological cycle. Hydrology and Earth System Sciences. 7,3, 75-86.

Byrne, K.A., Chojnicki, B., Christensen, T.R., Drösler, M., Freibauer, A., Friborg, T., Frolking, S., Lindroth, A., Mailhammer, J., Malmer, N., Selin, P., Turunen, J., Valentini, R., & Zetterberg, L. 2004, EU Peatlands: Current Carbon Stocks and Trace Gas Fluxes. Proceedings of the workshop of the Concerted Action CarboEurope-GHG, Lund, Sweden.

Döll, P., Kaspar, F. & Lehner, B. 2003, A global hydrological model for deriving water availability indicators: model tuning and validation. J. Hydrol., 270, 105-134.

Dugan, P. J., 1990. *Wetland conservation - a review of current issues and required action*. IUCN - The World Conservation Union, Gland, Switzerland, 96p Dunne, T. & Leopold, L.B. 1978, Water in Environmental Planning, W.H. Freeman and Company, New York.

Finlayson, M & Moser, M. 1991, Wetlands. Facts on File, Oxford.

Fischer, B., Turner, R.K. & Morling, P. 2009, Defining and classifying ecosystem services for decision making. Ecological Economics 68, 643-653 doi:10.1061/j.ecolecon.2008.09.014

Flörke, M. & Alcamo, J. 2004, European Outlook on Water Use, Center for Environmental Systems Research, University of Kassel, Final Report, EEA/RNC/03/007, 83 pp. Available online
http://scenarios.ew.eea.europa.eu/reports/fol949029/fol040583/Water_stress_final_report.pdf.

Górski K., Winter H. V., De Leeuw J. J., Minn A. E. & Naglekerke L. A. J., 2010, Fish spawning in a large temperate floodplain: the role of flooding and temperature, Freshwater Biology 55 issue 7, 1509-1519.

Gore A.J.P., 1983, Ecosystems of the Word 4A. Mires: swamp, bog, fen and moor. General Studies. Amsterdam, Oxford, New York 1983.

GRDC (2004), Long Term Mean Monthly Discharges and Annual Characteristics of Selected GRDC Stations. The Global Runoff Data Centre: Koblenz, Germany.

Junk, W.J., Bayley, P.B., Sparks R.E., 1989, The flood pulse concept in river-floodplain systems. [w:] Proceedings of the International Large River Symposium. D.P. Dodge (ed.). J. Can. Fish. Aquatic Sci. Special Issue 11: 106-127.

Keddy, P. 2010, Wetland Ecology: Principles and Conservation. Cambridge University Press, Cambridge.

Maltby, E. 1986, Waterlogged wealth. Earthscan, London. 200 pp.

Millennium Ecosystem Assessment 2005 Ecosystems and human well-being. Island Press, Washington DC, USA.

Mitsch, W.J., Gosselink, J.G., Anderson C.J., & Zhang L. 2009. Wetland Ecosystems, John Wiley & Sons, Inc., New York, 295 pp.

Okruszko T., Kijańska M., 2003: Viewing Wetlands as Water Users in Integrated River Basin Management Plans. International Journal of Ecology and Environmental Sciences, Special Issue: Wetlands and Agriculture, Vol. 29, No 1.

Okruszko T., Kiczko A., 2008: Assessment of water requirements of swamp communities: the river Narew case study. Publications of the Institute of Geophysics of the Polish Academy of Sciences 2008, Vol. E-9 (405), s. 27-39, Warszawa

Petr T., 2005, Fisheries and Aquaculture topics - Floodplains. Topics Fact Sheets. FAO. Rome.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. & Stromberg, J. C., 1997, The natural flow regime. Bioscience 47, 769-784

Richter, B. D., Baumgartner, J. V., Powell, J. & Braun D. P., 1996, A Method for Assessing Hydrological Alteration within Ecosystems. Conserv. Biol., 10, 1163-1174

Schneider C., Flörke M., Geerling G., Duel H., Grygoruk M. & Okruszko T. 2010, The future of European floodplain wetlands under a changing climate, Journal of Water and Climate, IWA Publ, in press.

Verhoeven J. T. A., Arheimer B., Yin C. Q., & Hefting M. M. 2006, Regional and global concerns over wetlands and water quality. Trends in Ecology & Evolution, 21:96-103.

Verzano, K., Hunger, M. & Döll, P. 2005, Simulating river flow velocity on global scale, Adv. Geosci. 5, 133-136.

Welcomme, R. 1979 Fisheries ecology of floodplain rivers. Longman, London

Wheeler, B.D., Gowing, D.J.G., Shaw, S.C., Mountford, J.O. & Money, R.P. 2004. Eco-hydrological Guidelines for Lowland Wetland Plant Communities. Environment Agency, Bristol, UK. 85 pp

5 Water for Nature 4 – Change in water supply to wetlands

5.1 Introduction

Wetlands belong to the most vulnerable ecosystems for future climate change as well as for local changes in water balance due to the human use. Analysis of wetland response strongly depends on their main source of water: rain, groundwater or river water. The representative sample of 103 wetlands of different types has been chosen, namely: bogs, fens, riparian wetlands and estuaries in order to check how much they can be affected by changes in water regime. Relative change in comparison to the baseline (reference year 2000) was an indicator for the possible changes.

5.2 Method

The analysed sample comprised the set of protected wetlands of Europe whose size is bigger than 5000 ha. The selection procedure was presented in chapter 4 of this volume. Depending on the type of wetland (i.e. the main source of water), different indicators were calculated.

Calculation approach

Bogs - Change in precipitation/PET
Fens - Change in (groundwater recharge + precipitation)/PET
Riparian wetlands - Change in flood volume
Estuaries - Change in volume of freshwater inflow to the estuary

Changes were calculated comparing the indicated values for the precipitation, PET and discharges for data of Climate 2050 (for the particular PEP3 scenario) to climate normal (1961-90).

Input data

- Daily discharges (WaterGAP)
- Monthly precipitation and PET data from MIMR/A2 and IPCM4/A2 climate models
- Yearly average groundwater recharge (WaterGAP)

Spatial and temporal scales

Annual averages were based on daily or monthly data. For estuaries and riparian wetlands, calculations were performed for the main grid square where the wetland was located. For bogs and fens, an average of all grids which are touching the wetland area was used.

Thresholds and critical values

The thresholds used to define the level of changes are:

> 60 %	significant change
20 – 60 %	change
-20 / + 20%	minor change
-20 / - 60 %	change
< - 60%	significant change

Validation

We make direct use of WaterGAP output, which has already been validated.

Uncertainty and sensitivity

- WaterGAP and climate models (Modelling rainfall-runoff and water use at the large scale to cover the entire pan-Europe area will have uncertainties as a result of scale itself and gaps in data. Projecting water use and availability for future scenarios is uncertain by its nature. Alcamo *et al.* (2000) provide more information on the uncertainties involved and their order of magnitude).
- Survey and classification methods of wetlands
- The use of PET instead of actual ET of wetland vegetation
- Recharge areas for groundwater supply
- Thresholds choice

5.3 Results

5.3.1 Baseline scenario

In the baseline scenario we use the WaterGAP modelled flows which are supplying the wetlands for the reference years.

5.3.2 Future scenarios

In Figures 5.1-5.8, results are shown for the four scenarios simulated with both IPCM4a2 and MIMRa2 climate models, showing the relative change in the indicator.

General pattern

In general, the climate change factor dominates. Changes in water regime due to the socio-economic developments are not clearly visible. The biggest reduction of indicators can be observed within all the Europe, except the British Isles and northern Scandinavia. Also wetlands located within the Iberian Peninsula seem not to be impacted in terms of changes in the hydrological indicators analyzed for the purpose of this study. Among the wetlands taken into account in the analysis, the negative changes in indicators occurred in most of the examples. The most severe changes can be observed in scenarios that use the results of IPCM4a2 climate model as an input.

Socio economic and climate scenarios

In the Economy First scenario, the most negative changes in indicators appeared, of which the region with the biggest reduction is Central Eastern Europe. The most significant positive changes in indicators were noticed within the Balkans. In the Fortress Europe and Policy First scenarios, a significant reduction of wetland water supply was seen for the whole pan-Europe area. Within Finland and the Benelux countries, some changes were observed due to the differences in models used to model the climate change impact scenarios independent from the scenarios themselves – IPCM4a2 model brings more of the significant negative changes than the MIMRa2 model.

5.4 Synthesis

For a summary of observed changes in all regions, see Table 5.1. The climate change factor dominates. There is no big difference between the socio-economic scenarios. The wetlands located in the northern part of Europe and those which are supplied by groundwater seem to be most secure in their future water supply. Central Europe, riparian wetlands under the “dry and hot” scenario are the most endangered in their water supply.

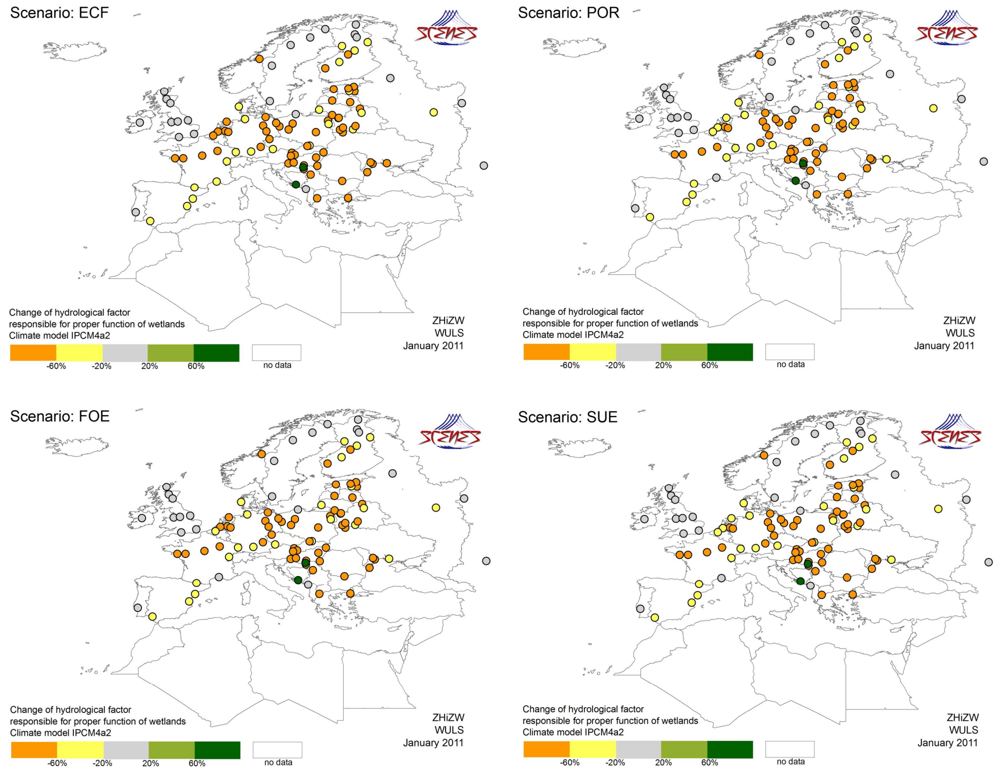

Figure 5.1 until 5.4 (left to right). Change in water supply to wetlands under the IPCM scenario. Economy First: Figure 5.1. Policy Rules: Figure 5.2. Fortress Europe: Figure 5.3. Sustainability eventually: Figure 5.4.

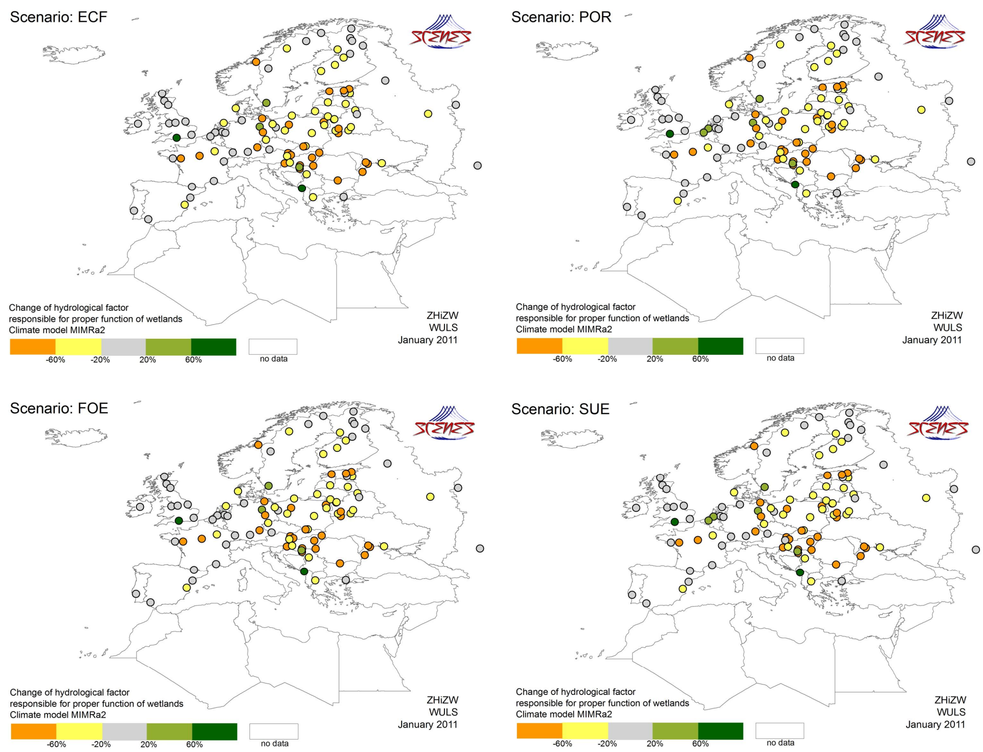

Figure 5.5 until 5.8 (left to right). Change in water supply to wetlands under the MIMR scenario. Economy First: Figure 5.5. Policy Rules: Figure 5.6. Fortress Europe: Figure 5.7. Sustainability eventually: Figure 5.8.

Table 5.1 Regional observations on changes with respect to the baseline scenario

		Northern Africa	Western Europe	Northern Europe	Southern Europe	Central Eastern Europe	Eastern Europe	Western Asia
IPCM	EcF	*no data*	--	0	--	--	-	*no data*
	FoE	*no data*	--	0	--	--	-	*no data*
	PoR	*no data*	--	0	--	--	-	*no data*
	SuE	*no data*	-	0	--	--	-	*no data*
MIMR	EcF	*no data*	0	0	-	-	0	*no data*
	FoE	*no data*	0	0	-	-	0	*no data*
	PoR	*no data*	+	0	0	-	0	*no data*
	SuE	*no data*	+	0	0	-	0	*no data*

5.5 References

Acreman, M.C. & Miller, F. 2007 Practical approaches to hydrological assessment of wetlands lessons from the UK. In: Okruszko, T., Maltby, E., Szatyłowicz, J., Świątek, D., Kotowski, W. (eds) Wetlands; monitoring, modelling and management: Taylor & Francis, London.

Acreman, M.C. & Mountford, J.O. 2009 Wetlands. In: Ferrier, R., Jenkins, A. (eds) *Handbook of catchment management*. Blackwell, Oxford

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T. & Siebert, S. 2003, Development and testing of the WaterGAP 2 global model of water use and availability. Hydrological Sciences 48(3), 317-337.

Aus der Beek, T., Flörke, M., Lapola, D.M., Schaldach, R., Voß, F., and Teichert, E. 2010, Modelling historical and current irrigation water demand on the continental scale: Europe. Adv. Geosci. (in press).

Barendregt A., NieuwenhuiS J.W., 1993: ICHORS, hydrological relations by multidimentional modeling of observations. [in:] The use of hydro-ecological models in The Netherlands. Hooghart J.C, Posthumus C.W.S (Eds.) TNO Committee on Hydrological Research, Proceedings and Information No 47, Delft: 11-30.

Batelaan O., De Smedt F., Triest L., 2002: A Methodology for Mapping Regional Groundwater Discharge Dependent Ecosystems. Third International Conference on Water Resources and Environment Research, 22nd–25th July 2002, Dresden University of Technology.

Biggs, B.J.F., Ibbitt, R.P., Jowett, I.G. 2008, Determination of flow regimes for protection of in-river values in New Zealand: an overview Ecohydrology and Hydrobiology, 8, 1, 17-29
Bullock, A., & Acreman, M.C. 2003, The role of wetlands in the hydrological cycle. Hydrology and Earth System Sciences. 7,3, 75-86.

Byrne, K.A., Chojnicki, B., Christensen, T.R., Drösler, M., Freibauer, A., Friborg, T., Frolking, S., Lindroth, A., Mailhammer, J., Malmer, N., Selin, P., Turunen, J., Valentini, R., & Zetterberg, L. 2004, EU Peatlands: Current Carbon Stocks and Trace Gas Fluxes. Proceedings of the workshop of the Concerted Action CarboEurope-GHG, Lund, Sweden.

Denny, P., 1993: Water management strategies for the conservation of wetlands. IWEM Annual Symposium on The Management of Scarce Water Resources, 13-14 Oct. 1992, Brighton, J. IWEM 7.

Döll, P., Kaspar, F. & Lehner, B. 2003, A global hydrological model for deriving water availability indicators: model tuning and validation. J. Hydrol., 270, 105-134.

Dugan, P. J., 1990. *Wetland conservation - a review of current issues and required action*. IUCN - The World Conservation Union, Gland, Switzerland, 96p Dunne, T. & Leopold, L.B. 1978, Water in Environmental Planning, W.H. Freeman and Company, New York.

Finlayson, M & Moser, M. 1991, Wetlands. Facts on File, Oxford.

Flörke, M. & Alcamo, J. 2004, European Outlook on Water Use, Center for Environmental Systems Research, University of Kassel, Final Report, EEA/RNC/03/007, 83 pp. Available online

http://scenarios.ew.eea.europa.eu/reports/fol949029/fol040583/Water_stress_final_report.pdf.

Gore A.J.P., 1983, Ecosystems of the Word 4A. Mires: swamp, bog, fen and moor. General Studies. Amsterdam, Oxford, New York 1983.

GRDC (2004), Long Term Mean Monthly Discharges and Annual Characteristics of Selected GRDC Stations. The Global Runoff Data Centre: Koblenz, Germany.

Junk, W.J., Bayley, P.B., Sparks R.E., 1989, The flood pulse concept in river-floodplain systems. [w:] Proceedings of the International Large River Symposium. D.P. Dodge (ed.). J. Can. Fish. Aquatic Sci. Special Issue 11: 106-127.

Keddy, P. 2010, Wetland Ecology: Principles and Conservation. Cambridge University Press, Cambridge.

Larson J.S., Mazzarese D.B., 1994: Rapid assessment of wetlands: history and application to management. [w:] Global wetlands. W.J. Mitach (ed.). Elsevier Science Publishers, Amsterdam: 625-636.

Linacre E.T.,1976 Swamps w: J.L. Monteith (Edytor). Vegetation and Athmosphere, vol 2., Case Studies, Academic Press, London, pp. 329 – 347.

Lonard R.J., Clairain E.J., 1995: Identification of methodologies and the assesment of wetland functions and values. [w:] Proceedings of the National Wetland Assessment Symposium. Kusler J.A., Riexinger P. (eds.). Portland, ME: 66-71.

Maltby, E. 1986, Waterlogged wealth. Earthscan, London. 200 pp.

Mitsch, W.J., Gosselink, J.G., Anderson C.J., & Zhang L. 2009. Wetland Ecosystems, John Wiley & Sons, Inc., New York, 295 pp.

Okruszko T., Kijańska M., 2003: Viewing Wetlands as Water Users in Integrated River Basin Management Plans. International Journal of Ecology and Environmental Sciences, Special Issue: Wetlands and Agriculture, Vol. 29, No 1.

Okruszko T., Kiczko A., 2008: Assessment of water requirements of swamp communities: the river Narew case study. Publications of the Institute of Geophysics of the Polish Academy of Sciences 2008, Vol. E-9 (405), s. 27-39, Warszawa

Okruszko T., Ignar S., Recognition of wetlands hydraulics as a tool for their protection and restoration. [w:] Environmental hydraulics and Sustainable Water Management. Proceedings of the „4th International Symposium on Environmental Hydarulics and the 14th Congress of IAHR-APD", Hong-Kong, 15-18.12.2004

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. & Stromberg, J. C., 1997, The natural flow regime. Bioscience 47, 769-784

Richter, B. D., Baumgartner, J. V., Powell, J. & Braun D. P., 1996, A Method for Assessing Hydrological Alteration within Ecosystems. Conserv. Biol., 10, 1163-1174

Schneider C., Flörke M., Geerling G., Duel H., Grygoruk M. & Okruszko T. 2010, The future of European floodplain wetlands under a changing climate, Journal of Water and Climate, IWA Publ, in press.

Verhoeven J. T. A., Arheimer B., Yin C. Q., & Hefting M. M. 2006, Regional and global concerns over wetlands and water quality. Trends in Ecology & Evolution, 21:96-103.

Verzano, K., Hunger, M. & Döll, P. 2005, Simulating river flow velocity on global scale, Adv. Geosci. 5, 133-136.

Wheeler, B.D., Gowing, D.J.G., Shaw, S.C., Mountford, J.O. & Money, R.P. 2004. Eco-hydrological Guidelines for Lowland Wetland Plant Communities. Environment Agency, Bristol, UK. 85 pp

6 Water for Nature 5 – Aquatic macrophyte diversity in lakes

6.1 Introduction

Eutrophication is defined as an excess of chemical nutrient emissions to such an extent, that it increases the natural primary production of the ecosystem. Phosphorus as well as nitrogen are often identified as the main culprits. Eutrophication is a threat to a wide range of ecosystems, endangering water quality and biodiversity through changes in plant communities (EEA, 2007b).

Aquatic macrophyte diversity is a good indicator for eutrophication. This indicator relates the total nitrogen (TN) concentrations to the aquatic macrophyte diversity in lakes to give an indication of the ecological status of lakes across Europe. Thresholds for TN concentration have been identified and applied to the modelled TN concentrations for the baseline and all SCENES scenarios.

Various diffuse and point sources contribute to the total N loading to surface waters and as a result, several Directives have been put in place to reduce the nitrogen loading. The Nitrates Directive (91/676 EEC) aims to reduce nitrogen pollution from diffuse agricultural sources by defining various abatement measures. Other measures concerning load reduction relate to sewage control, and the operation and extent of WWTP's are described in the Urban Waste Water Treatment Directive (91/271/EEC) and the IPPC Directive (Integrated Pollution Prevention and Control (2008/1/EG)).

6.2 Method

Calculation approach

TN is calculated was calculated with the HABITAT nitrogen model (Malotaux, 2010). This model estimates nitrogen concentrations within large river basin across Europe on a grid basis. The model estimates TN concentrations from calculated TN loads in surface waters and the annual discharge (natural availability) as calculated by the WaterGAP3.1 model (CESR). The TN loads were calculated by the HABITAT nitrogen model from nitrogen emissions (both point and diffuse sources) and by applying retention factors to the different compartments within a river basin (retention in the soil- and groundwater, in agricultural fields, and in the river network) and applying a correction factor to translate river TN concentrations to TN concentrations in lakes. The correction factor was estimated by comparing two water quality dataset for lakes and rivers (EEA Waterbase, 2010). The TN concentrations for both lakes and rivers were grouped by country and plotted to derive a factor of 0.42 (Figure 6.1).

The nitrogen retention in the river network, f_{river}, was calculated following Behrendt and Opitz (2000) based on the hydraulic load of the upstream river basin and expressed as a the fraction of the nitrogen load supplied from upstream:

$$f_{river} = 1.9 \cdot HL^{-0.49} \qquad \text{and} \qquad HL = \frac{q}{W}$$

where q specific runoff ($l \cdot km^{-2} \cdot s^{-1}$) ; W = upstream water area coverage (%). The coefficients used for the calculation of f_{river} are derived from the study on the Danube by Schreiber et al. (2003).

The sources that have been used to derive the N loading are natural background N, atmospheric deposition, fertilizer- and manure application on agricultural fields, industries, households, and urban litter. Part of the emission from point sources is lost by removal in waste water treatment plants (WWTPs) based on data for sewage connectivity and treatment levels. The individual sources are discussed in the *Input data* section of this chapter. A more detailed description of the HABITAT nitrogen model can be found in deliverable 4.3 – Annex E (Malotaux, 2009). For each SCENES scenario, relative macrophyte diversity in lakes was derived for the A2 scenario run by two climate models IPCM4-A2 and MIMR-A2.

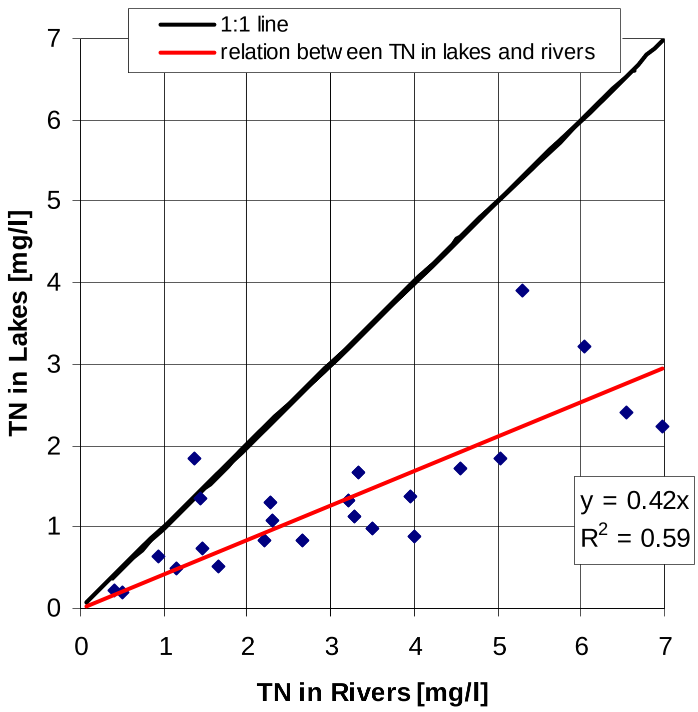

Figure 6.1 Comparison of TN concentration in lakes and rivers.

Input data
Various sources contribute to the total N loading to surface waters.

The Natural background N emission was based on European background concentrations of NO_3^--N of 0.2 – 0.3 mg l^{-1} (Meybeck, 1982) observed in natural rivers. Converting this estimate from concentration to a specific rate per unit area results in an estimation that lies close to the background Nitrogen emission estimated by the EEA of 1 – 2 kg ha yr^{-1} (EEA 2005). The background N emissions rate was set at 1.5 kg ha^{-1} yr^{-1}. All areas and scenarios are assumed to release the same amount of background N emissions.

N loading from atmospheric N deposition has been obtained from the website of the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP; Jonson et al., 1998) which contained 50 x 50 km gridded N deposition data for the year 2000.

Manure application was estimated from data on livestock units (CESR, 2007). Livestock Units represents a value for which all livestock types are normalized to non-dairy cattle based on their relative excretion rates. An excretion rate of 65 kg N per head was used to represent non-dairy cattle (Schreiber et al., 2003).

Fertilizer application rates were calculated from FAO country totals (metric tones per year; FAOSTAT, 2009) and agricultural area (from reclassification of Corine land class 2000 data; EEA, 2007a). High application rates in for example Finland result from a small agricultural area cover. In general the application rates correspond well to the rates as presented in Miterra (Velthof et al., 2009).

From the SCENES storylines a qualitative interpretation of fertilizer use was drawn for each scenario (Table 6.1). To translate this to a quantitative estimate the long-term projections of fertilizer consumption by region (Tenkorang and Lowenberg-DeBoer, 2008; FAO, 2003), were used to estimate the general trend in nitrogen fertilizer use. The study by Tenkorang and Lowenberg-DeBoer (2008) estimate a 50% increase for Europe by 2050, in FAO (2003) for industrial countries the change in fertiliser consumption from 1997/1999 to 2030 is estimated to be around 25%. Based on the qualitative analysis (FoE: Maximizing the agricultural production in order to secure the food production; EcF: higher production in areas where economic yield is highest; PoR: realization of the policy target has priority; SuE: realization of optimal ecological status) the scenario specific values were estimated in range with the general projection, with Fortress Europe showing the largest increase for water rich countries and Sustainability Eventually showing an overall decrease in fertilizer use.

Table 6.1 Qualitative analysis of future fertilizer use translated into quantitative fertilization rates.

Fertilizer use	**SuE**	**PoR**	**EcF**	**FoE**	**Remarks**
Water Poor	- -20%	- -20%	- -20%	- -20%	Southern & parts of central Europe
Water Rich	- -10%	+ +10%	++ +30%	+++ +50%	Western, eastern & northern Europe
Maximum use (kg N ha^{-1})	100	170	250	250	

The Nitrates Directive obliges Member States to limit the use of animal manure to a maximum of 170 kg of nitrogen per hectare in order to realize a good ecological status (Fraters et al., 2007). The European Commission granted the Netherlands the right to derogate from the obligation, implying that farmers could use up to 250 kg of nitrogen per hectare on grasslands. For an optimal ecological quality the fertilization rate is lower, up to 100 kg N per hectare. Based on this numbers a limit to the nitrogen fertilization rate was used for the different scenarios. The change in fertilizer use for the different scenarios is summarized in Table 6.1.

N loading from industries was quantified with a dataset from the website of the European Pollutant Emission Register (EPER, 2001) which contains data on N emissions of industrial facilities that emit more than 50,000 kg N per year. This dataset covers the 15 EU member states, Hungary and Norway and represents emission data of the year 2001. Inherently, emission data is missing in the EPER due to the threshold of 50,000 kg N that needs to be exceeded before N emissions from a facility are registered. Comparing data from the Dutch Emission Register (DER) and EPER data on Dutch industrial N emissions, the DER data gave N emission (95 Gg N) that clearly exceeds the amount as registered in the EPER database (6.6 Gg N) for the Netherlands. A correction factor of 7 was applied to the whole of pan-Europe after a calibration procedure using a detailed dataset for the Rhine river basin (De Wit, 1999). It is recognized that this data has a high uncertainty in regard to the 'real' industrial emissions.

N loading from households was estimated from population data, connectivity and treatment levels, and an estimated value of N emission per capita. Data on connectivity and treatment levels as well as the calculation method (as described in Malotaux, 2010) were derived from the Centre for Ecology and Hydrology (CEH). Population data was available on a 5 x 5 min scale covering the pan-Europe area from the Centre for Environmental Systems Research (CESR), at the University of Kassel, Germany (CESR, 2007). For the N emissions per capita a value of 11 g N cap^{-1} day^{-1} was used which is in range with values derived from literature, 9 - 18 g N cap^{-1} day^{-1} (De Wit, 1999; Van Drecht et al., 2003), and is close to the guideline value of 12 g N cap^{-1} day^{-1} as reported by OSPAR. It was assumed that per capita emissions were equal in all regions.

Data on the connectivity of household to sewage systems, the fraction of the waste water that is treated in WWTPs, percentage of waste water treated per treatment type was available per country from the SCENES database.

The maps showing the contribution of the total N emission to the surface water (N loading) can be found in Figure 6.2 (left) for the baseline and in Figures 6.3-6.6 for the scenarios (change in total N loading). The source apportionment of the different emission sources is shown in Figure 6.2 (right).

Thresholds and critical values
Thresholds for TN have been identified by deriving a relationship between TN and the amount of aquatic macrophyte species. To this end, data from various datasets (Pot, 2001; Wiser, 2009) have been analyzed. Pot (2010) contains water quality and macrophyte data for Dutch lakes. Using Van der Molen and Pot (2007), non-aquatic species were filtered out. Annual average TN values were calculated and coupled with samples of aquatic species taken in the same year. From WISER (2009), aquatic macrophyte species and TN values ranging from 1 up to 30 measurements per year were present; these were also converted into annual average TN values. This yielded a total of 1056 sites: 569 from WISER (2009) and 587 from Pot (2010). 75-percentile values for the number of species were calculated and TN thresholds were set at concentrations coinciding with a steady decrease in the number of species, expressed as relative species richness (see Table 6.2). Box plots from the sample distribution are shown in Figure 6.7.

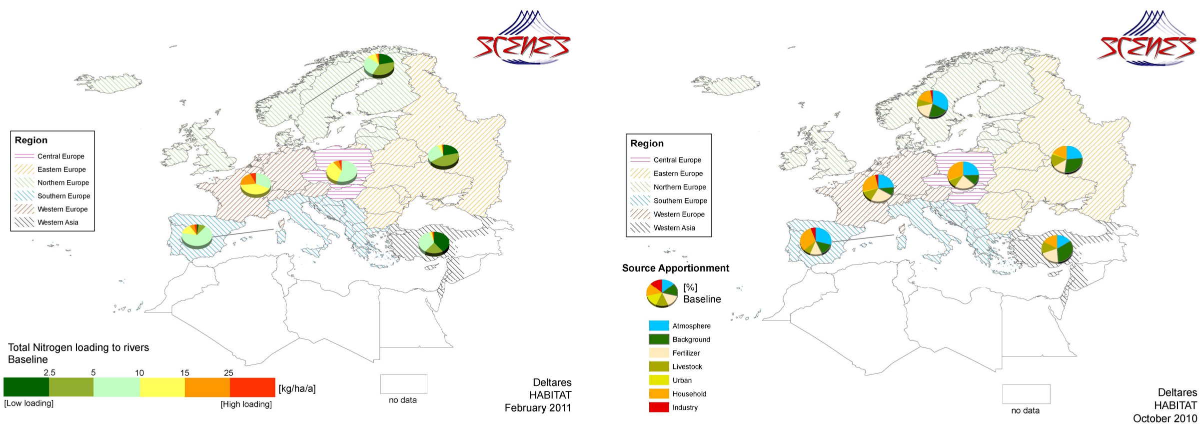

Figure 6.2 Total N loading results (left) and source apportionment (right) for the baseline

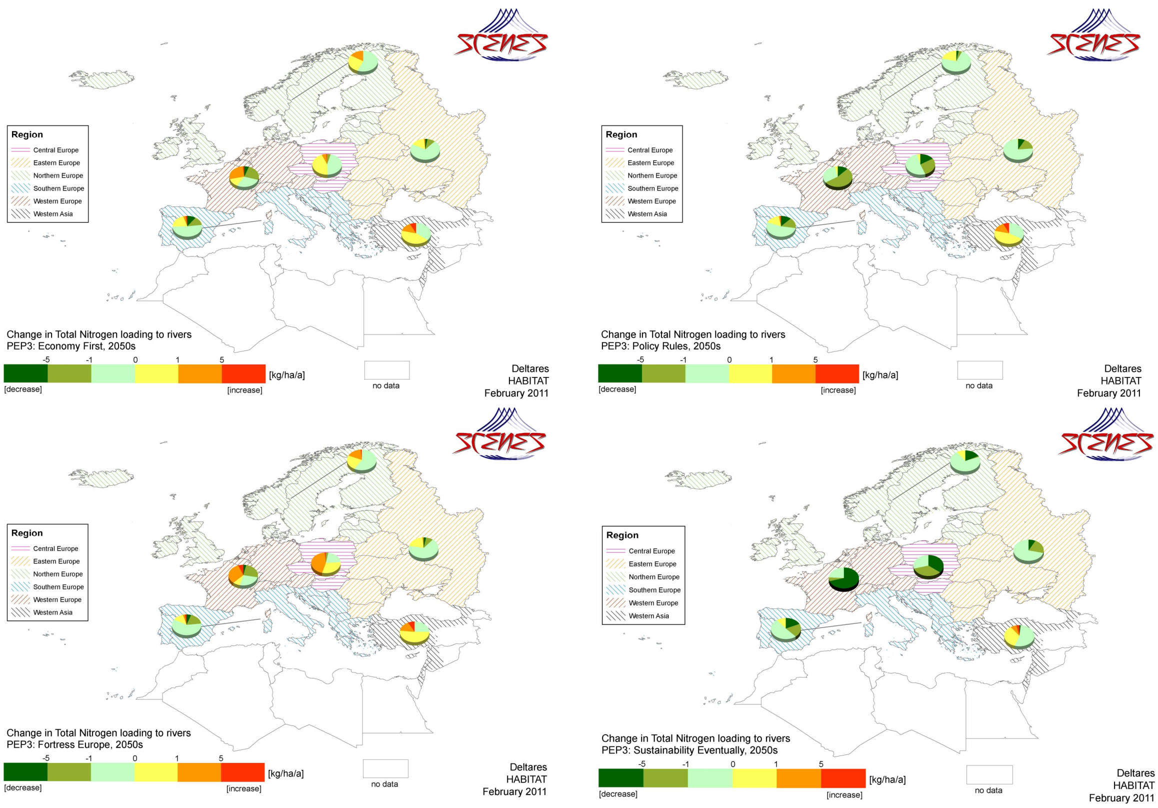

Figure 6.3 until 6.8 (left to right). Change in N-loading to surface waters by 2050. Economy First: Figure 6.3. Policy Rules: Figure 6.4. Fortress Europe: Figure 6.5. Sustainability eventually: Figure 6.6.

Table 6.2 Status of the aquatic macrophyte diversity according to TN-thresholds for freshwater lakes derived from the relative species richness.

Aquatic macrophyte diversity	TN-threshold in mg Nl^{-1}	75-percentile	Relative species richness	# of sites
Very Low	> 2.2	5	18%	291
Low	1.6 – 2.2	10	36%	117
Intermediate	1.1 – 1.6	14	50%	98
High	0.75 – 1.1	24	87%	81
Very High	≤ 0.75	28	100%	473

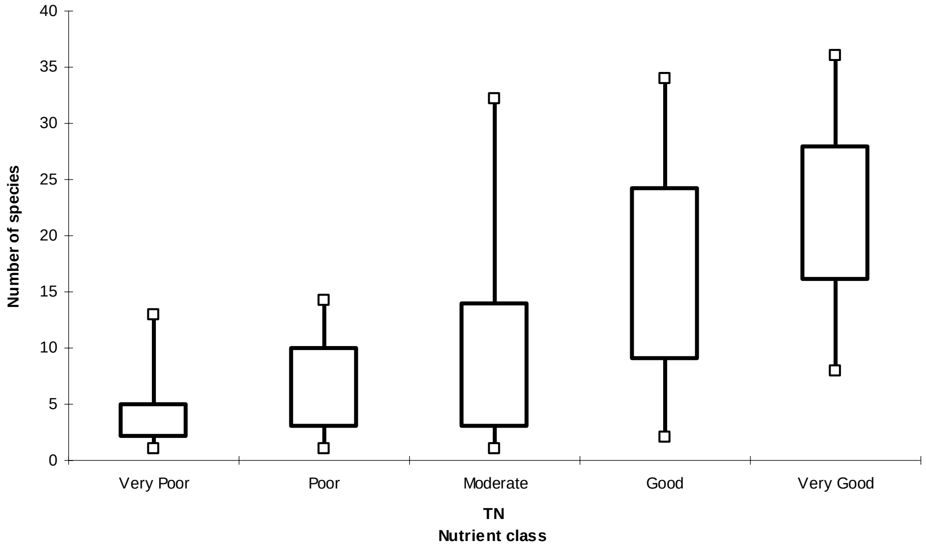

Figure 6.4 *Box plot for the nutrient classes based on TN thresholds and the number of species from Table 6.2.*

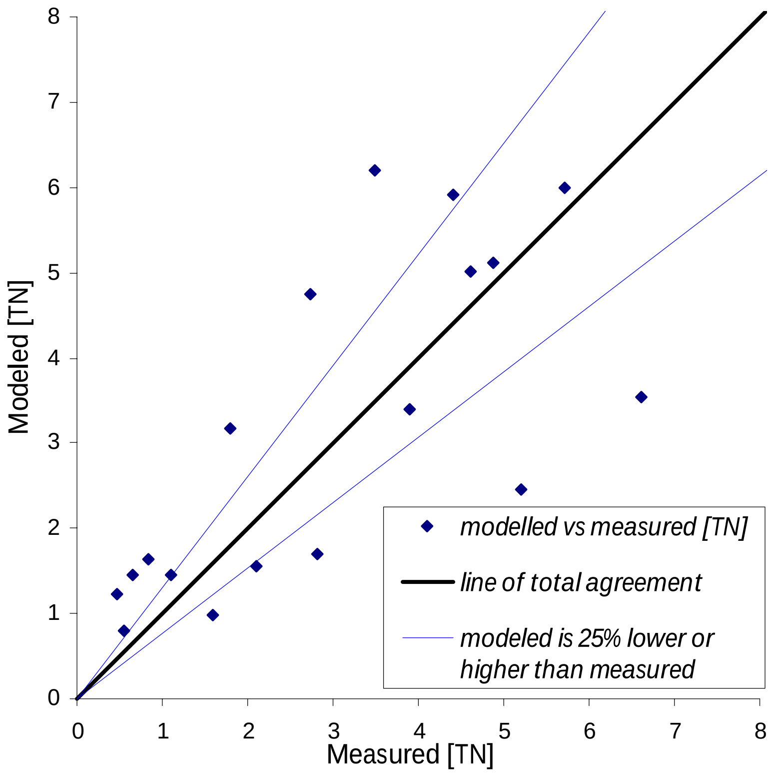

Figure 6.5 *Modelled versus measured TN concentration for a set of river basin outlets for which measured data was available.*

Validation
The modelled TN concentrations were compared to measured N concentrations at river outlets (EEA, 2005; De Klijne et al, 2007; Leeks et al., 1997; Salvetti et al., 2006). Most of these values fall within a 25% confidence band (Figure 6.8).

Cross-scale analysis (See also Deliverable DIA2.5) revealed that the estimated nutrient load for Estonia is in range with the estimate from the pilot area (3-4 kg N ha^{-1}). However, for the Russian part the estimate seems to be underestimated, which is probably due to the definition of the background load that is just too low for the Russian part (and probably also for Belarus) of the Lake Peipsi catchment. These levels should be more or less the same as in Estonia (for the northern part of Russia) or Lithuania (for central part of Russia and Belarus). Air load seems to be underestimated for these areas as well that results very low loading levels for Russia and in some parts of Belarus.

A comparison with baseline estimates from the pilot area Narew River basin in Poland (about 3 kg N ha^{-1} yr^{-1}), showed that the N loading is overestimated, which can most likely be attributed to too high estimation of atmospheric deposition on forested areas and also too high households emission due to development of local infrastructure (i.e. municipal waste water treatment plants). The storylines made by stakeholders during Pilot Area workshops assume fertilizer use reduction in agriculture (in Sustainability Eventually). This is line with the pan-European projection.

Uncertainty and sensitivity
For the estimation of emissions at the European scale it is inevitable to make a number of assumptions that involve a level of uncertainty. The threshold of 50,000 kg N a^{-1} that needs to be exceeded for registration of industrial emission implies that small scale activity might not have been taken into account. However, to also account for the emission from smaller industries a correction factor was applied using a detailed dataset for the Rhine river basin (De Wit, 1999).

Since fertilizer forms a major source of nitrogen to surface waters, the results for the scenarios show a strong relation to the change in fertilizer use estimates. These estimates are an interpretation of the storylines that indicate agricultural intensification for water rich countries and a reduction for water poor countries. The quantitative estimates are associated with a high degree of uncertainty; consequently it is hard to provide a clear judgement of the future change in TN loading.

Uncertainties also apply to the estimated excretion rates from livestock, the wastewater treatment level in the future and N retention factors between point of emission in both agricultural and non-agricultural fields and the loading to surface waters. The N retention factors were assumed to be equal in all river basins.

Although hydraulic load shows a good fit with nitrogen retention in the river network (Behrendt and Opitz, 2000), the application of the relationship gives overestimates of retention in the northern part of Europe where lake area is relatively high.

Uncertainties in N emissions for the various sources as well as the impact on model results have been summarized in Table 6.3.

Table 6.3 Uncertainties associated with nitrogen emission data and consequences for model outcome.

Pollution source	Magnitude of uncertainty	Impact on model outcome
atmosphere	high	medium
industry	high	medium
urban waste water	low	low
rural domestic waste water	medium	low
agriculture	high	high
natural background	low	low

Besides N loading the model results are sensitive to the discharge from WaterGAP3.1. An underestimation of discharge will directly lead to overestimation of TN concentrations.

Concerning the threshold validation using the various data sources, sampling strategies may have differed between databases and even within databases, as Pot (2010) contains data from various sources. However, this is not a critical uncertainty since species richness is not as dependent on sampling strategy and effort as abundance. Secondly, the large amount of samples will most likely compensate for this.

6.3 Results

The TN threshold has been applied to the results derived from the HABITAT TN model and were aggregated for each SCENES region for both climate models. Maps have been created depicting the share of grid cells falling in a quality class indicating the relative aquatic macrophyte diversity in lakes (Figures 6.9-6.17).

6.3.1 Baseline scenario

Areas with a high diversity of aquatic macrophytes in lakes are mainly found in Northern Europe and parts of Eastern and Western Asia. Also areas with a high diversity are found in Southern Europe. Areas with low diversity are found in Western and Central Europe. For Western Europe about equal shares in areas with low and high diversities in lakes can be observed. For Central Europe over 75% falls in the lower diversity range.

6.3.2 Future scenarios

General pattern

For all climate and socio-economic scenarios, Northern Europe contains the highest share of cells (over 75%) in the very high diversity class (≤ 0.75 mg l^{-1}). For Eastern Europe, about 50% is in the very high diversity class, followed by Southern Europe (> 25%), Western Asia and Western Europe (around 25%). The Central European region only has a small share of cells falling into the very high diversity class. About 50% of Central Europe falls within the very low diversity class.

High N loading occurs in areas where high population densities exist and areas with intensive agriculture. Urban and agricultural areas can easily be depicted. Western Europe shows the highest total N loading, whereas Northern Europe and Western Asia show the lowest rates.

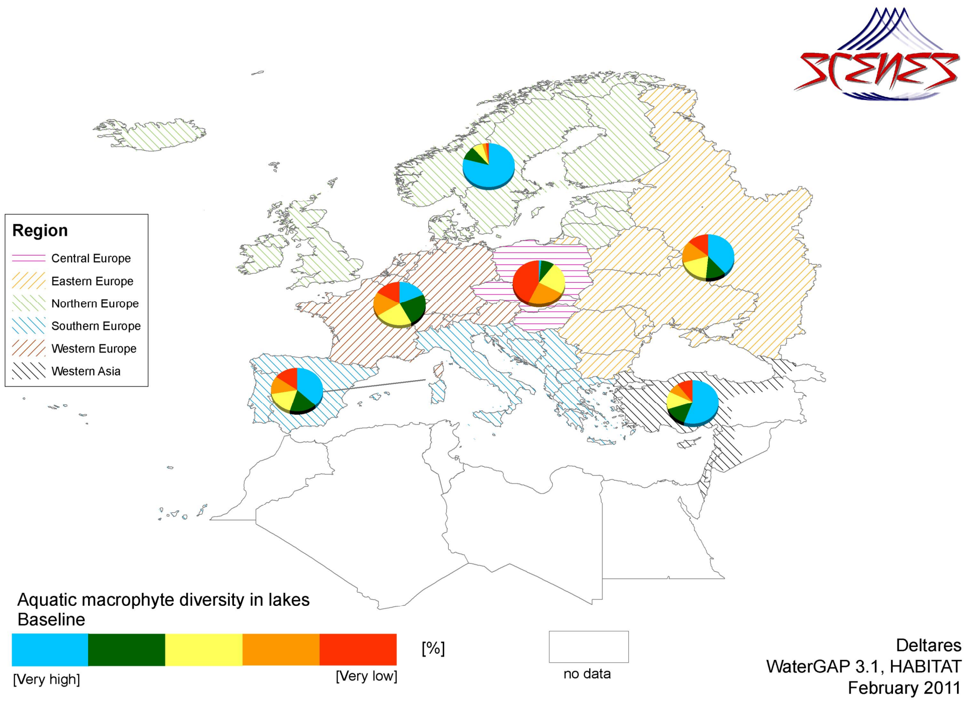

Figure 6.6 Aquatic macrophyte diversity in lakes for the baseline situation

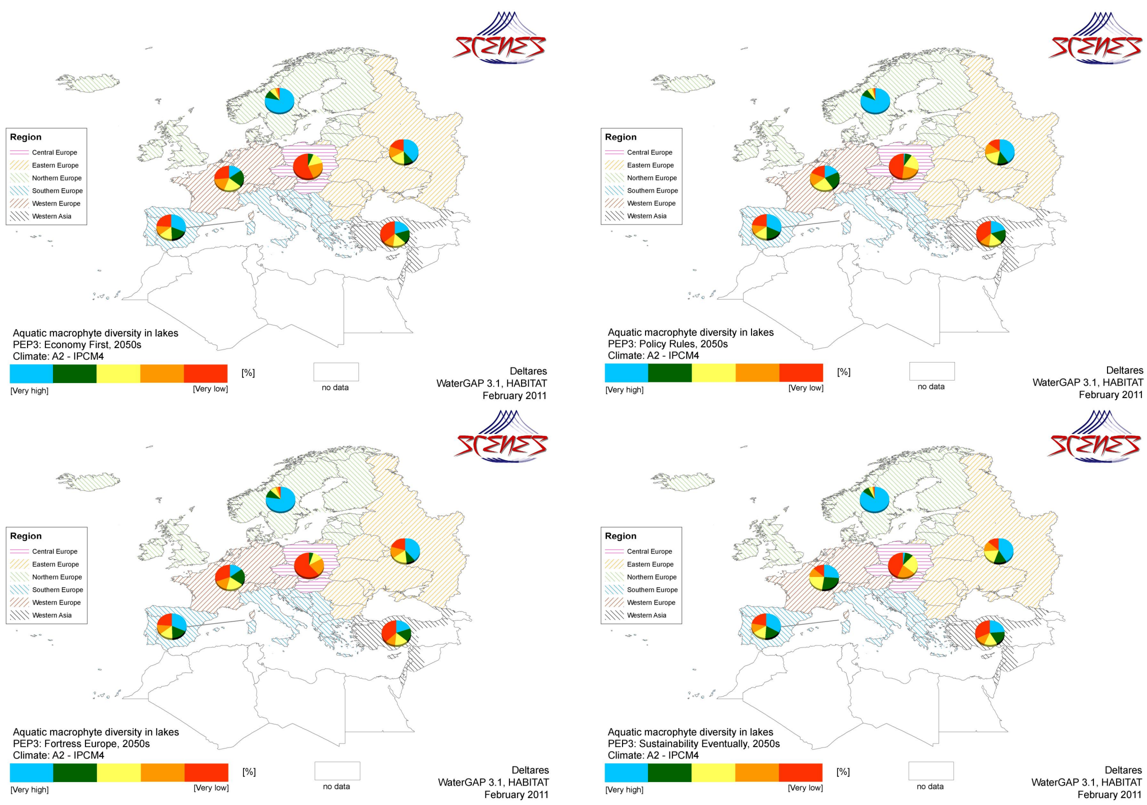

Figure 6.7 until 6.10 (left to right). Aquatic macrophyte diversity in lakes by 2050 under the IPCM scenario. Economy First: Figure 6.10. Policy Rules: Figure 6.11. Fortress Europe: Figure 6.12. Sustainability eventually: Figure 6.13.

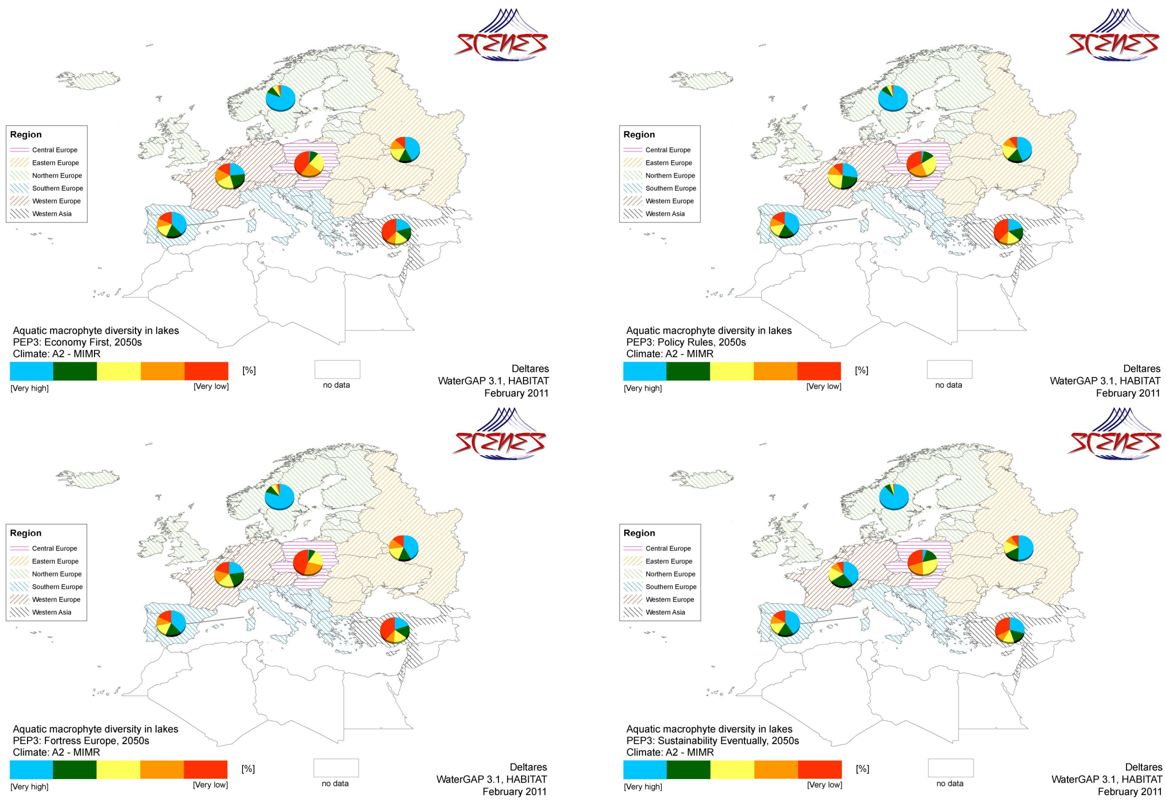

Figure 6.8 until 6.11 (left to right). Aquatic macrophyte diversity in lakes by 2050 under the MIMR scenario. Economy First: Figure 6.8. Policy Rules: Figure 6.9. Fortress Europe: Figure 6.10. Sustainability eventually: Figure 6.11.

N loading to surface waters

Nitrogen loading from agricultural and urban areas is highest in Western Europe, followed by Central Europe, where atmospheric deposition, households and fertilizer are the largest emission sources. In some areas in Western Europe with intensive agriculture or high urbanization N loading rates exceed 25 kg ha^{-1} a^{-1} that is more than 15 times the natural background N loading. The lowest nitrogen loading can be found in Northern and Eastern Europe and Western Asia, where N loading from background emission followed by atmospheric deposition are the dominant sources.

The highest increase in N loading for 2050 is under the Fortress Europe scenario, where Western Europe shows the highest increase. For Sustainability Eventually as well as for Policy Rules a decrease in N loading can be observed all over Europe except for a few areas, mainly in Western Asia. In the other two scenarios most parts in Western Asia show an increase, whereas the other regions show more areas with a decreased N loading than with an increased N loading, except for Central Europe in the Economy First scenario.

Socio economic and climate scenarios

Sustainability Eventually features the least deterioration for all regions and climate models. For the combined Sustainability Eventually and IPCM4-A2 scenario almost no change with the baseline is observed, except for Western Europe. The only improvement towards a higher diversity is observed for the combination of the Sustainability Eventually and MIMR-A2 scenario, as well as Policy Rules/MIMR-A2 and Sustainablity/IPCM4-A2 for Western Europe.

This improvement for Western Europe is that the current fertilizer use is higher than the policy target of 170 kg N ha^{-1} $year^{-1}$ in several countries. Western Asia is an exception to this, as this region shows a decrease in diversity for all scenarios. The results for MIMR-A2 show higher diversity than results for IPCM4-A2 for all socio economic scenarios. Differences in climate model are most pronounced for Central Europe. Fortress Europe shows the poorest results: for IPCM4-A2 only around 5% of the cells reaches high relative diversity and most of the cells fall in the low/very low class for Central Europe. For MIMR-A2 about 10% meets a high relative diversity.

Applying the MIMR-A2 model, results in a higher relative diversity, due to higher precipitation and runoff, this allows for the dilution of nutrients.

6.4 Synthesis

For a summary of the observed changes in all regions, see Table 6.4. The Nordic region shows the highest relative diversity and the Central European the lowest, followed by Western Europe and Eastern Asia. This is in line with the N loading to surface waters (Figure 6.2), except that Western Europe has a higher N loading rate than Central Europe. Because Central Europe has a drier climate and less runoff than Western Europe the relative diversity of macrophytes in lakes is lower in the Central European region as compared to Western Europe. Sustainability Eventually has the best results. Differences between the socio-economic scenarios are most pronounced for Central and Western Europe.

Table 6.4 Regional observations on changes with respect to the baseline scenario

		Northern Africa	Western Europe	Northern Europe	Southern Europe	Central Eastern Europe	Eastern Europe	Western Asia
IPCM	EcF	*no data*	-	0	-	-	0	--
	FoE	*no data*	-	0	-	--	-	--
	PoR	*no data*	0	0	-	-	0	--
	SuE	*no data*	0	+	-	0	0	--
MIMR	EcF	*no data*	0	0	0	0	0	--
	FoE	*no data*	0	0	0	-	0	--
	PoR	*no data*	0	+	0	0	0	--
	SuE	*no data*	+	+	0	+	+	--

This picture is supported by EEA (2007b) although here it is reported that Western Continental Europe is more affected by eutrophication (70%) than Central Europe. Southeastern Europe, covering part of the SCENES region Central Europe, is mentioned to have reports of eutrophication as well. In other regions the problem is less prominent but nonetheless often reported. The damage level is expected to decline only slightly by 2020 (EEA, 2005).

Future scenarios will not show a significant improvement of the nutrient levels in rivers and lakes in comparison to the current situation, consequently many rivers and lakes (~50% in populated areas) will still support a good ecological status according the WFD requirements. Western Asia shows a decreasing trend for all scenarios.

6.5 References

Behrendt, H. & Opitz, D., 2000. *Retention of nutrients in river systems: Dependence on specific runoff and hydraulic load*. Hydrobiologica 410: 111-122.

CESR, Center for Environmental Systems Research. University of Kassel, Germany. *www.cesr.de*.

CESR, 2007. Centre of envrionmental Systems Research SCENES. *Gridded data: population and livestock numbers.*

De Wit, M., 1999*. Nutrient fluxes in the Rhine and Elbe basins*. Utrecht Koninklijk Nederlands Aardrijkskundig Genootschap/Faculteit Ruimtelijke Wetenschappen Universiteit Utrecht.

EEA, 2005. *Source apportionment of nitrogen and phosphorus inputs into the aquatic environment*. EEA Report. European Environmental Agency. 46. Copenhagen, Denmark. Retrieved Access Data 2005.

EEA, 2007a. *Corine land cover map European Envrionmental Agency*. Retrieves Acces Data Acces 2007 from http://www.eea.europa.eu/themes/landuse/clc-download

EEA, 2007b. *Europe's environment – The fourth assessment*. European Environment Agency, Copenhagen, Denmark, 453 pp.

EEA Waterbase, 2010, The European Topic Centre on Water. Version 10, 08 April 2010. http://www.eea.europa.eu/.

EPER, 2001. The *European Pollutant Emission Register*. European Commission. Retrieves Acces Data Acces 2001 from http://eper.ec.europa.eu/eper/

Tenkorang, F. & Lowenberg-DeBoer, J., 2008. *Forecasting Long-term Global Fertilizer Demand.* Food and Agriculture organization of the United Nations (FAO), Rome, 44 pp.
FAOSTAT, 2009. data obtained from *http://faostat.fao.org/*

Jonson, J. E., Bartnicki, J., Olendrzynski, K., Jakobsen, H. A. & Berge, E., 1998. EMEP Eulerian model for atmospheric transport and deposition of nitrogen species over Europe. *Environmental Pollution* 102(1S1): 289-298.

De Klijne, A., Hooijboer, A.E.J., Bakker, D.W., Schouwmans, O.F. & Van den Ham, A., 2007. *Milieukwaliteit en nutriëntenbelasting. Achtergrondrapport milieukwaliteit van de Evaluatie Meststoffenwet 2007*. RIVM rapport 680130001, p. 60.

Leeks, G.J.L., Neal, C., Jarvie, H.P., Casey, H. & Leach, D.V, 1997. *The LOIS river monitoring network: strategy and implementation*. Science of the Total Environment 194/195: 101-109.

Malotaux, J.M., 2009. Total Nitrogen Concentration model for European Surface Waters (in Methodology of indicator development and initial validation of core set of indicators - Deliverable 4.3 - Annex E), Deltares.

Malotaux, J.M., 2010. *Total nitrogen concentration modeling for European river basins.* Master thesis sustainable development, Department of Innovation and Environmental Sciences, Utrecht University.

Meybeck, M., 1982. *Carbon, nitrogen, and phosphorus transport by world rivers.* American Journal of Science Vol. 282(40): 401-450 .

Pot, R. 2010. *Toestand en trends in de waterkwaliteit van Nederlandse meren en plassen*. Onderzoeksrapport voor Rijkswaterstaat Waterdienst; Roelf Pot, Oosterhesselen.

Salvetti, R., Azzellino, A. & Vismara, R., 2006. *Diffuse source apportionment of the Po river eutrophying load to the Adriatic sea: Assessment of Lombardy contribution to Po river nutrient load apportionment by means of an integrated modelling approach.* Chemosphere 65(11): 2168-2177

Schreiber, H., Constantinescu, L. T., Cvitanic, I., Drumea, D., Jabucar, D.,Juran, S., Pataki, B., Snishko, S., Zessner, M. & Behrendt, H. 2003. *Harmonised Inventory Of Point And Diffuse Emissions Of Nitrogen And Phosphorus For A Transboundary River Basin, Research Report*. Institute Of Freshwater Ecology And Inland Fisheries, 159 P., Berlin.

Van der Molen, D. & Pot, R., 2007. *Referenties en maatlatten voor natuurlijke watertypen voor de Kaderrichtlijn Water*. STOWA, Utrecht, 362 pp.

Van Drecht, G., Bouwman, A. F., Knoop, J. M., Beusen, A. H. W. and Meinardi, C. R., 2003. *Global modeling of the fate of nitrogen from point and nonpoint sources in soils, groundwater, and surface water*. Global Biogeochemical Cycles 17(4): 1115.

Velthof, G. L., Oudendag, D., Witzke, H. P., Asman, W. A. H., Klimont, Z. & Oenema, O., 2008. *Integrated assessment of nitrogen emissions from agriculture in EU-27 using MITERRA EUROPE*. J. Environ. Qual. 38: 402–417, doi:10.2134/jeq2008.0108, 2009

Wiser, 2009. WISER database, received from Bernard Dudley, last update: 06-03-2009

7 Water for Nature 6 – Habitat suitability for river water temperature for fish

7.1 Introduction

Water temperature has been identified as one of the main pressures on habitat suitability for fish resulting from climate change (Verdonschot et al., 2007). Not only directly, but also indirectly temperature can play an important role due to an influence on phosphorus mobilization (Jeppesen et al., 2009). Both will in turn lead to changes in the composition of the fish community due to increased primary production.

Increased water temperature also touches on other subjects, such as the cooling water capacity of rivers, as there are set maximum temperatures for river water (e.g. Van der Grinten et al., 2007) resulting from the requirement of the WFD (Water Framework Directive; European Commission, 2000) to define hard temperature boundaries. An increasing trend in electricity generation across Europe requires more cooling water and consequently can lead to higher water temperatures.

As both fish community composition and structure and physico-chemical conditions (including temperature) within the WFD are identified as Quality Elements, river water quality critical values have been identified for selected fish species communities in the Nordic, Atlantic, Central, Southern, and Eastern European region.

7.2 Method

Calculation approach

Water temperature was calculated with the HABITAT water temperature model from both natural background water temperature, and the temperature surplus related to the discharge of cooling water from industrial activity. The temperature surplus was routed over a river network map, thereby taking into account the cooling effect of the surplus temperature in downstream direction using a cooling factor and added to the natural background water temperature. This background temperature was derived from an air-water temperature region specific relationship. For an elaborate description of the HABITAT water temperature model, see the indicator "Industry 1 - Extra demand for cooling water" chapter. For each socio-economic scenario, the habitat suitability of river water temperature for fish was derived for the climate models IPCM4-A2 and MIMR-A2.

Input data

Air temperature for the different scenarios was derived from the CRU and climate scenarios used within SCENES. July was chosen to be representative for the warmest month. Water availability during low flow conditions (Q_{90}) was used to calculate discharge, since low flow conditions occur in the summer months and can have a severe impact on elevated water temperature. Q_{90} means that 90% of the monthly values during the total 30 year period are higher than the provided discharge.

National data on Thermal Energy Production for the calculation of discharge of cooling water was obtained from the Centre for Ecology and Hydrology (CEH). A likelihood map for the location of thermal energy plants was used to translate the country totals to a gridded map (see Industry 1 for a detailed method description).

Temperature thresholds and critical values

A literature research was conducted for key fish species. Firstly, eco-regions (Figure 7.1) were clustered by SCENES regions (Table 7.1). After the selection of key species, critical temperatures for fish species were collated from literature (Table 7.2) and thresholds were set based on the amount of key species for which it is identified as being critical. Thresholds are listed in Table 7.3.

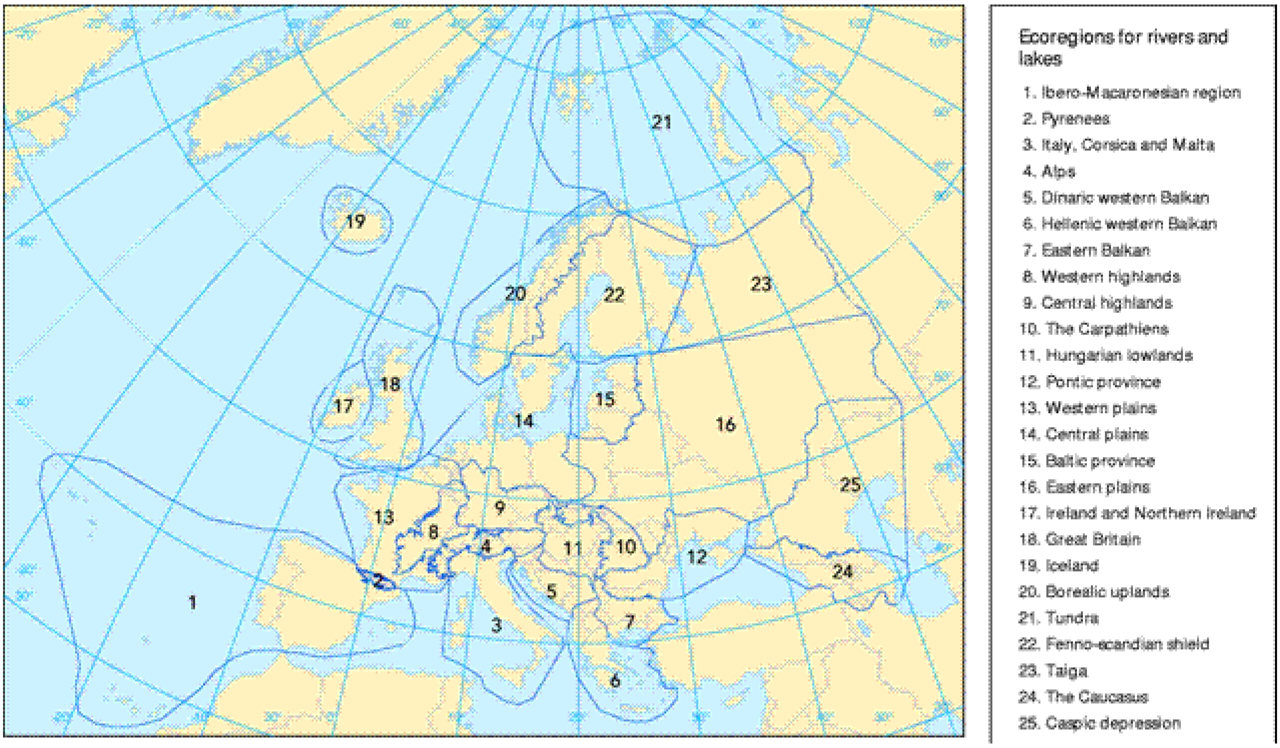

Figure 7.1 Ecoregions for rivers and lakes according to Illies (1968).

Table 7.1 Clustered ecoregions in SCENES from Illies (1968).

SCENES region	Ecoregion
Nordic & Alpine	2, 4, 15, 19, 20, 21, 22, 23 +18 (Scotland)
Atlantic	8, 13, 14, 17, 18 (UK)
Central & Eastern	9, 10, 11, 12, 16, 24, 25
Southern	1, 3, 5, 6, 7 & Turkey

Validation

The air-water relationship has been validated for various rivers across Europe (Segrave, 2009). The HABITAT temperature model has been validated for only the Rhine River. The temperature surplus in the Rhine River closely meets the computed value (~4°C versus ~3°C for the baseline situation).

Thresholds have been validated with values from the literature (Table 7.2) and by the European Directive on water quality for fish (European Commission, 2006). The Directive sets the threshold for Salmoniformes (including salmon and trout) at 21.5°C, which approximately coincides with the lower range in the Nordic & Alpine region. For Cypriniformes (including carps) it is set at 28°C, coinciding with the lower range of the Southern region.

Table 7.2 Selected fish species, scientific names, their occurrence in the SCENES regions and critical summer temperatures inferred from literature.

Species	Scientific name	Nordic & Alpine	Atlantic	Central & Eastern	Southern	Critical summer temp. (ºC)
Beluga / European sturgeon	*Huso huso*			X		30[1]
European river lamprey	*Lampetra fluviatilis*	X	X	X	X	30[2,3]
Atlantic Salmon	*Salmo salar*	X	X			28[3]
Sea trout	*Salmo trutta trutta*	X	X	X	X	26[3]
Common whitefish	*Coregonus lavaretus*	X				26[4]
Houting	*Coregonus oxyrinchus*	X				23[5]
Grayling	*Thymallus thymallus*	X	X	X		26[6,7]
Common dace	*Leuciscus leuciscus*	X	X	X		28[8]
European chub	*Squalius cephalus*		X	X	X	24 - 30[8]
Ide	*Leuciscus idus*	X		X		36[9,10]
Common nase	*Chondrostoma nasus*			X		26 – 29[8,11]
Gudgeon	*Gobio gobio*		X	X	X	30[8,12]
Common barbell	*Barbus barbus*		X	X		2713,14
European eel	*Anguilla anguilla*	X	X	X	X	33 – 39[15,16]
Burbot	*Lota lota*	X	X	X		23[17,18]
European bullhead	*Cottus perifretum/gobio)*		X	X		28[18,19]

[1]Reinartz (2002); [2]Maitland et al. (2002); [3]Wolters et al. (2003); [4]Flüchter et al. (1980); [5]Rosenthal and Munro (1985); [6]Kraiem and Patee (1980); [7]Elliot (1981); [8]Alabaster and Lloyd (1982); [9]Cazemier and Wiegerinck (1993); [10]Ruremonde (1988); [11]Herzig and Winkler (1985); [12]Gaumert (1986); [13]Philipart (1982); [14]Banarescu and Bogutskaya (2003); [15]Sadler (1979); [16]Boëtius and Boëtius (1967); [17]Hochleitner (2002); [18]Elliot and Elliot (1995); [19]Kainz and Gollman (1989).

Table 7.3 Temperature thresholds in °C for SCENES ecoregions

Fish temperature class	Nordic & Alpine	Atlantic	Central & Eastern	Southern
Good[1]	< 22	< 24	< 26	< 28
Critical[2]	22-24	24-26	26-28	28-30
Poor[3]	> 24	> 26	> 28	> 30

[1]water temperature is reaching critical value for a maximum of 1 – 2 key species;
[2]water temperature is exceeding critical value for one or two key species;
[3]water temperature is exceeding critical values to high extend.

Uncertainty and sensitivity

Uncertainties associated with the HABITAT temperature model are described in the indicator "Industry 1 - Extra demand for cooling water" chapter.

The eco-regions (Figure 7.1) were clustered and assigned to one of the four SCENES regions. Uncertainties can arise from countries that overlap with more than one eco-region (e.g. France was assigned to the Atlantic region).

Information concerning critical fish temperatures has been extracted from various sources and critical temperatures may have been defined in a different fashion. For example, field experiments may yield different values than laboratory experiments due to the presence of different pressures.

In some cases individuals have been allowed to adapt to higher temperatures, which can give higher critical values. As such, the definition of critical temperature may vary.

Fish also have preferences for certain altitudes and temperatures: more upstream parts have a different composition with more stenothermic species. As we have not considered this, upstream stretches may come out positively biased as the level of detail did not allow us to correct this.

7.3 Results

The temperature thresholds have been applied to the socio-economic scenarios in combination with the IPCM4-A2 and MIMR-A2 models. Maps have been created depicting the grid cells of large river and its tributaries falling in classes ranging from good to poor habitat suitability (Figures 7.2-7.10).

7.3.1 Baseline scenario

Most rivers in Europe have a good habitat suitability for fish when looking at the water temperature. Exceptions are all located in the western and southern parts of Europe in downstream reaches of the large rivers.

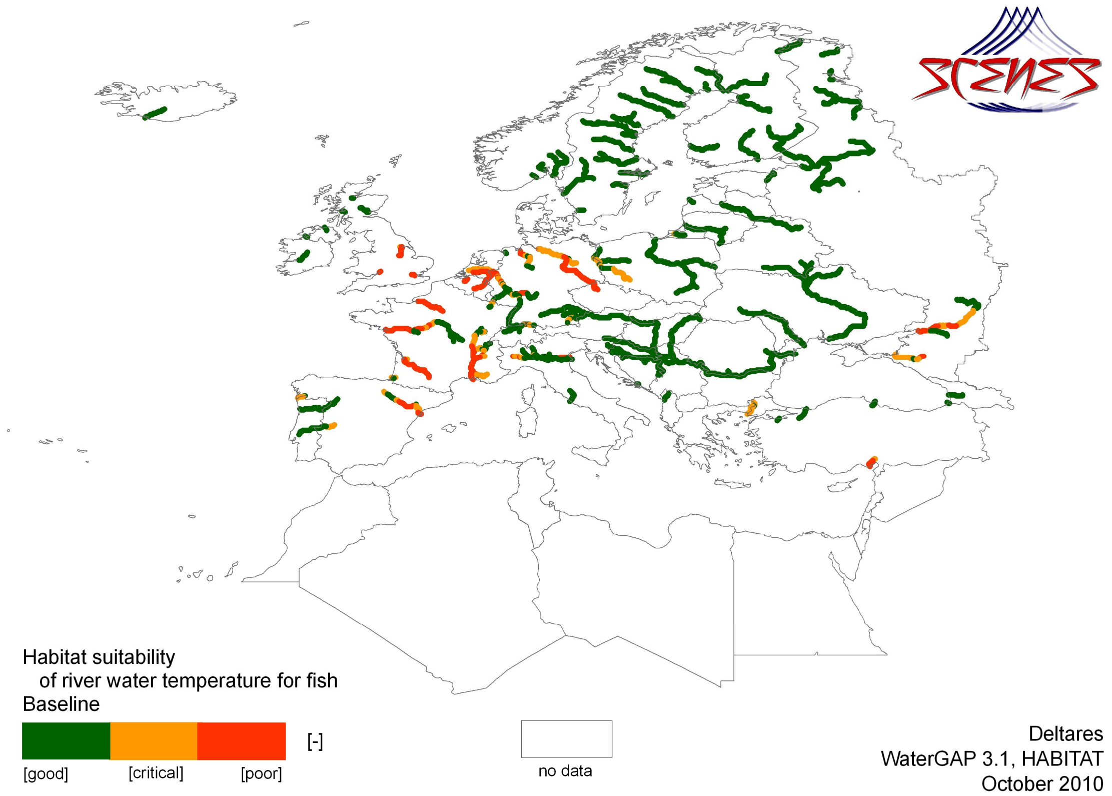

Figure 7.2 Habitat suitability of river water temperature for fish for the baseline situation

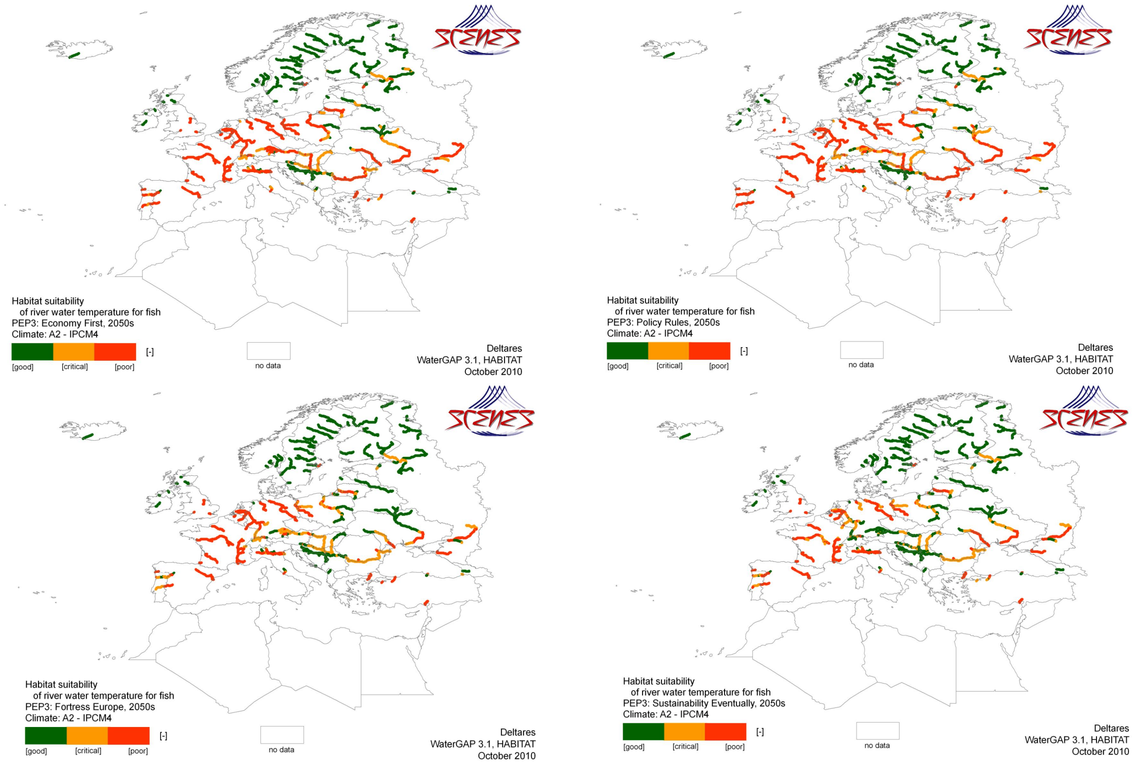

Figure 7.3 until 7.6 (left to right). Habitat suitability of river water temperature for fish by 2050 – IPCM climate scenario. Economy First: Figure 7.3. Policy Rules: Figure 7.4. Fortress Europe: Figure 7.5. Sustainability eventually: Figure 7.6.

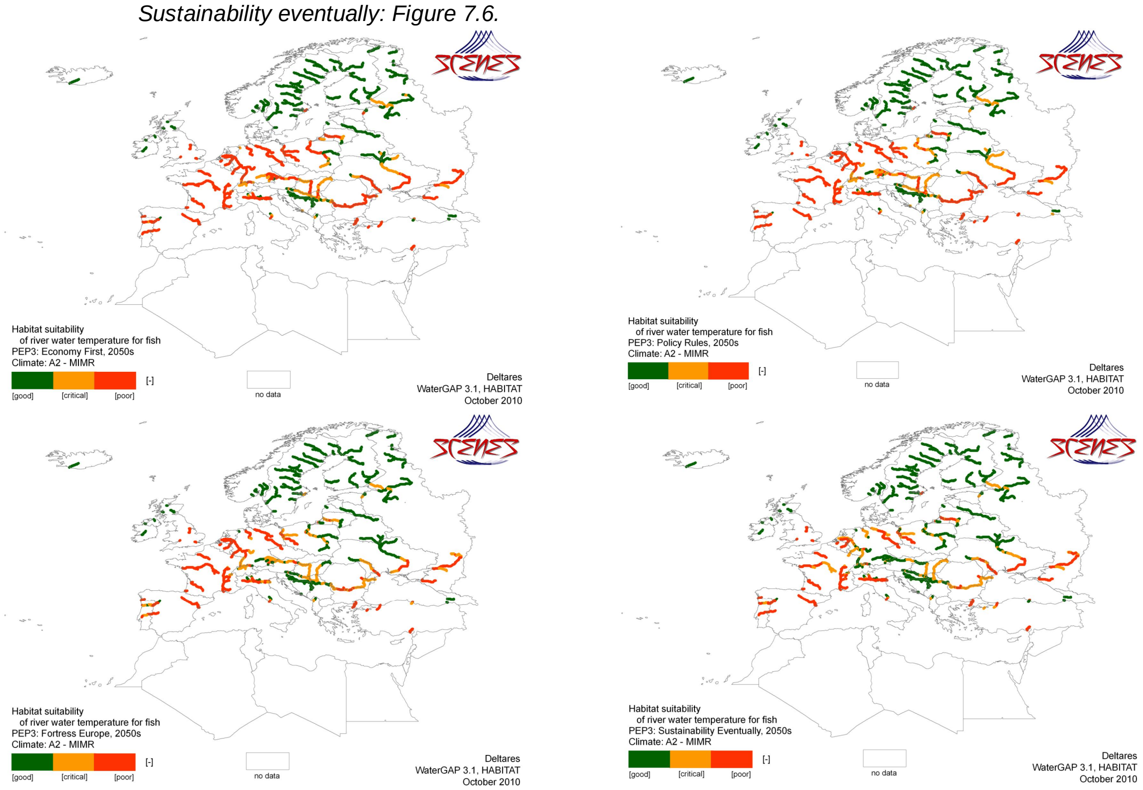

Figure 7.7 until 7.10 (left to right). Habitat suitability of river water temperature for fish by 2050 – MIMR climate scenario. Economy First: Figure 7.7. Policy Rules: Figure 7.8. Fortress Europe: Figure 7.9. Sustainability eventually: Figure 7.10.

7.3.2 Future scenarios

General pattern
Results for all four socio economic scenarios show good habitat suitability for most rivers in the Nordic and Alpine region for both IPCM4-A2 and MIMR-A2 climate scenarios. However, when a comparison with the baseline situation is made, large parts of Central, Eastern and Southern Europe show critical to poor habitat suitability for fish. For Western Europe many upstream parts of the rivers change to poor habitat suitability. In general, habitat suitability of river water temperature for fish is identified as being good in the upstream sections and is deteriorating when moving downstream.

Natural and excess water temperatures
The water temperature changes in future as a result of changing natural temperature (climate) and a temperature surplus (excess temperature from cooling water discharge). Both climate scenarios (IPCM4-A2 and MIMR-A2) lead to an increase in natural water temperature for all rivers, ranging between 2 - 4 °C, and slightly higher for Northern Europe. The excess temperature may both decrease and increase (see Volume E, section 2.3). When looking at the combination of the change in natural and excess temperature, the estimated decrease in excess temperature for the scenario with the least impact (SuE) is compensated by the natural temperature increase. Therefore, except for very few sections in Western Europe, all scenarios show an increase in water temperature when looking at the combined effect of climate and socio-economic impact.

Socio economic and climate scenarios
The Atlantic region is mainly classified as being in critical to poor condition for Sustainability Eventually, except for Ireland (all in good condition). In other scenarios river water temperature in the Atlantic region is mainly classified as being in poorly suited for fish. Policy Rules and Econonmy First show the worst conditions for fish in all regions. Applying the IPCM4-A2 scenario results in river water temperature being even less suitable for fish than for the MIMR-A2 scenario.

7.4 Synthesis
It is clear that there is a significant risk that future habitat suitability of river water temperature for fish will be reduced. Where currently many populated and industrialized areas in the Atlantic region and Southern Europe already show poor habitat suitability, for the scenarios large parts of Europe show potential problems for region specific fish species, except for the Nordic region. For this indicator the focus has been on direct effects of temperature change; indirect effects of temperature on habitat suitability were not taken into account. For a summary of the observed changes in all regions, see Table 7.4.

7.5 References
Alabaster, J.S. & Lloyd, R., 1982. *Water Quality Criteria for Freshwater Fish - second edition*. FAO/Butterworth Scientific, London, 361 pp. In: Van Emmerik, W.A. & H.W. de Nie, 2006. De zoetwatervissen van Nederland - ecologisch bekeken. Sportvisserij Nederland, Bilthoven. 267 pp.

Banarescu, P.M & Bogutskaya, N.G., 2003. *The Freshwater Fishes of Europe; Cyprinidae 2, part II: Barbus.* AULA-Verlag GmbH Wiebelsheim, 454 p. In: Wijmans, P.A.D.M., 2007. Kennisdocument barbeel, Barbus barbus (Linnaeus, 1758). Kennisdocument 14. Sportvisserij Nederland, Bilthoven, 76 pp.

Table 7.4 Regional observations on changes with respect to the baseline scenario

		Northern Africa	Western Europe	Northern Europe	Southern Europe	Central Eastern Europe	Eastern Europe	Western Asia
IPCM	EcF	*no data*	--	0	--/0	--	--	-/0
	FoE	*no data*	--	0	-/0	-	-/0	-
	PoR	*no data*	--	0	--/0	--	--	--
	SuE	*no data*	-	0	--/0	-	-/0	-/0
MIMR	EcF	*no data*	--	0	--/0	--	--	-/0
	FoE	*no data*	--	0	-/0	-	-/0	--/0
	PoR	*no data*	--	0	--/0	--	--/0	--
	SuE	*no data*	-	0	--/0	-	-/0	-/0

Boëtius I. & Boëtius J, 1989. *Ascending elvers, Anguilla anguilla, from five European localities. Analysis of pigmentation stages, condition, chemical composition and energy reserves*. Dana 7: 1-12. In: Klein Breteler, J.G.P., 2005. Kennisdocument Europese aal of paling, Anguilla anguilla (Linnaeus, 1758). Kennisdocument 11. Sportvisserij Nederland, Bilthoven, 78 pp.

Cazemier, W.G. & Wiegerinck, J.A.M., 1993. *Oecologische randvoorwaarden voor Nederlandse zoetwatervissoorten*. RIVO-DLO rapport C 005/93. In:: Van Emmerik, W.A. & H.W. de Nie, 2006. De zoetwatervissen van Nederland - ecologisch bekeken. Sportvisserij Nederland, Bilthoven. 267 pp.

Elliot, J. M., 1981. *Some aspects of thermal stress on freshwater teleosts*. Stress and Fish (A. D. Pickering, ed), pp. 209-245. London: Academic Press. In: Küttel, S., A. Peter & A. Wüest, 2002. Rhône Revitalisierung - Temperaturpräferenzen und -limiten von Fischarten Schweizerischer Fliessgewässer.

Elliot, J.M. & Elliot, J.A., 1995. *The critical thermal Limits for the Bullhead, Cottus gobio, from three populations in North-West England*. Freshwater Biol. 33: 411-418. In: Küttel, S., A. Peter & A. Wüest, 2002. *Rhône Revitalisierung - Temperaturpräferenzen und -limiten von Fischarten Schweizerischer Fliessgewässer*.

European Commission, 2000. *Directive 2000/60/EC of the European parliament and of the council – of 23 October 2000 – establishing a framework for Community action in the field of water policy.* Office for official Publications of the European Communities, Luxembourg.

European Commission, 2006. *Directive 2006/44/EC of the European Parliament and of the Council of 6 September 2006 on the quality of fresh waters needing protection or improvement in order to support fish life*. Office for official Publications of the European Communities, Luxembourg.

Flüchter, J., 1980. *Review of the present knowledge of rearing whitefish (Coregonidae) larvae*. Aquaculture 19: 191-208. In: Otto, S.A. & S. Zahn, 2008. Temperatur- und Sauerstoff-Toleranz ausgewählter Wanderfischarten der Elbe. Institut für Binnenfischerei e.V., Potsdam-Sacrow, 43 pp.

Gaumert, D., 1986. *Kleinfische in Niedersachsen. Hinweise zum Artenschutz.* Mitteilungen aus dem Niedersächsisches Landesamt für Wasserwirtschaft. Heft 4. Hildesheim, 71 pp. In: Van Emmerik, W.A. & H.W. de Nie, 2006. De zoetwatervissen van Nederland - ecologisch bekeken. Sportvisserij Nederland, Bilthoven. 267 pp.

Herzig, A. & Winkler, H., 1985. *Der Einfluss der Temperatur auf die embryonale Entwicklung der Cypriniden*. Osterreichs Fischerei 38: 182-196. In: Küttel, S., A. Peter & A. Wüest, 2002. Rhône Revitalisierung - Temperaturpräferenzen und -limiten von Fischarten Schweizerischer Fliessgewässer.

Hochleitner, M., 2002. *Die Quappe (Lota lota L.) - Biologie und Aquakultur*. In: Die Quappe (Lota lota)-Fisch des Jahres 2002. Hrsg. Verband Deutscher Sportfischer e.V., Offenbach am Main, S.23-37. In: Otto, S.A. & S. Zahn, 2008. Temperatur- und Sauerstoff-Toleranz ausgewählter Wanderfischarten der Elbe. Institut für Binnenfischerei e.V., Potsdam-Sacrow, 43 pp.

Illies, J. (Ed.), 1967. *Limnofauna Europaea*. G. Fischer, Stuttgart.

Jeppesen, E., Kronvang, B., Meerhoff, M., Søndergaard, M., Hansen, K.M., Andersen, H.E., Lauridsen, T.L., Liboriussen, L., Beklioglu, M.B., Özen, A. & Olesen, J.E., 2009. *Climate Change Effects on Runoff, Catchment Phosphorus Loading and Ecological State, and Potential Adaptations*. J. Environ. Qual. 38: 1930-1941.

Kainz, E. & Gollmann, H. P., 1989. *Beiträge zur Verbreitung einiger Kleinfischarten in österreichischen Gewässern - Teil 1: Koppe, Mühlkoppe oder Groppe (Cottus gobio L.). Österreichs Fischerei* 42: 204-207. In: Küttel, S., A. Peter & A. Wüest, 2002. Rhône Revitalisierung - Temperaturpräferenzen und -limiten von Fischarten Schweizerischer Fliessgewässer.

Kraiem, M. & Pattee, E., 1980. La *tolérance à la température et au déficit en oxygène chez le Barbeau (Barbus barbus L.) et d'autres espèces provenant des zones voisines.* Archiv für Hydrobiologie 88: 250-261. In: Küttel, S., A. Peter & A. Wüest, 2002. Rhône Revitalisierung - Temperaturpräferenzen und -limiten von Fischarten Schweizerischer Fliessgewässer.
Maitland P.S., 2003. *Ecology of the River, Brook and Sea Lamprey*. Conserving Natura 2000 Rivers Ecology Series No. 5. English Nature, Peterborough.

Philippart, J.C., 1982. *Mise au point de l'alevinage controlé du barbeau Barbus barbus (L.) en Belgique*. Caiers d'Ethologie appliquée 2(2):173-202. In: Wijmans, P.A.D.M., 2007. Kennisdocument barbeel, Barbus barbus (Linnaeus, 1758). Kennisdocument 14. Sportvisserij Nederland, Bilthoven, 76 pp.

Reinartz, R., 2002. *Sturgeons in the Danube River - Biology, Status, Conservation.* International Association for Danube Research (IAD), Bezirk Oberpfalz, Landesfischereiverband Bayern e.V., 154 pp.

Rosenthal, H. & Munro, A.L.S., 1985. *Der aquatische Lebensraum, Umweltbedingungen in natürlichen Gewässern und Aquakulturen.* In: Grundlagen der Fischpathologie. R.J. ROBERTS & H.J. SCHLOTFELDT, Berlin und Hamburg: 1 - 22. In: Otto, S.A. & S. Zahn, 2008. Temperatur- und Sauerstoff-Toleranz ausgewählter Wanderfischarten der Elbe. Institut für Binnenfischerei e.V., Potsdam-Sacrow, 43 pp.

Ruremonde, R. van, 1988. *Veranderingen van de visfauna in het Nederlandse rivierengebied: een historisch overzicht.* Doctoraalscriptie, Katolieke Universiteit Nijmegen, 65 pp. In Van Emmerik, W.A. & H.W. de Nie, 2006. De zoetwatervissen van Nederland - ecologisch bekeken. Sportvisserij Nederland, Bilthoven. 267 pp.

Sadler, K., 1979. *Effects of temperature on the growth and survival of the European eel, Anguilla anguilla L.* Journal of Fish Biology 15: 499-507.

Segrave, A.J., 2009, *River water temperature for industrial cooling (in Methodology of indicator development and initial validation of core set of indicators* - Deliverable 4.3 - Annex H), KWR Watercycle Research Institute.

Van der Grinten, E., van Herpen, F.C.J., van Wijnen, H.J., Evers, C.H.M., Wuijts, S. & Verweij, W., 2007. *Afleiding maximumtemperatuurnorm voor goede ecologische toestand (GET) voor Nederlandse grote rivieren*. In opdracht van het Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer. RIVM Rapport 607800003, 86 pp.

Verdonschot, R.C.M., de Lange, H.J., Verdonschot, P.F.M. & Besse, A., 2007. *Klimaatverandering en biodiversiteit. I. Literatuurstudie naar temperatuur.* Alterra-rapport 1451. Alterra, Wageningen, 128 pp.

Wolter, C., Arlinghus, R., Grosch, U.A. & Vilcinskas, A., 2003. Zeitschrift für Fischkunde, Suppl.Bd. 2, 164pp. In: Otto, S.A. & S. Zahn, 2008. *Temperatur- und Sauerstoff-Toleranz ausgewählter Wanderfischarten der Elbe*. Institut für Binnenfischerei e.V., Potsdam-Sacrow, 43 pp.

8 Key messages

Based on the findings for the generic indicators, this Chapter provides an answer to four general questions:

- What are the key messages?
- What is the overall image per region?
- Are there big differences between regions?
- Can socio-economic changes (SE) or climate changes (CC) be identified as dominant driving forces of these changes?

To answer these questions the analysis for all scenarios is aggregated into an indication per indicator and per region of where the focus lies (positive, negative, no change, or a combination) and what the uncertainty is with respect to future changes (do the different scenarios point in the same direction or not) as presented in Table 8.1.

In Table 8.1, the indicators are grouped slightly differently:

- Indicators based on climate change:
    - Floodplain wetlands: flood volume
    - Floodplain wetlands: flood duration
    - Floodplain wetlands: flood timing
- Indicators in which climate change and socio-economic change have been combined:
    - Environmental flows
    - Ecosystem services of wetlands
    - Change in water supply to wetlands
    - Aquatic macrophyte diversity in lakes
    - Habitat suitability for river water temperature for fish

The main climate-related input data include climate (which generates the natural flow) and, for Nature 6, natural river water temperature. The main socio-economic input data include sectoral water demands and, for Nature 5, Nitrogen loading to surface waters, and for Nature 6, excess river water temperature.

What are the key messages?

- Under the two 2050 climate scenarios, the vast majority of freshwater ecosystems in Europe experience significant ecological change (with respect to significant alterations in flow regime).
- On the basis of the PEP3 results, climate change is a more important driver than socio-economic change for the water quantity indicators (involving magnitude and timing of extreme events). Impacts are more severe under IPCM4 scenarios than under MIMR scenarios.
- Both river ecosystems and wetland ecosystems are vulnerable to climate change.
- Future scenarios will not show a significant improvement of the nutrient levels in rivers and lakes in comparison to the current situation, consequently many rivers and lakes will not support a good ecological status according the WFD requirements.

- In the current situation, water temperature is a limiting factor for fish in rivers in highly industrialized and urbanized catchments due to cooling water discharges, especially in Western Europe. For future scenarios temperature rise is mainly caused by climate change and will affect fish communities in rivers in many catchments in Europe. Only in Northern Europe, fish populations are not affected significantly by river water temperature.

What is the overall image per region?

Northern Africa
Overall result: water for the environment will be negatively impacted in terms of quantity; for water quality there is no information. Results are only available for one indicator: environmental flows, which shows low impacts inland to high impacts in the Morocco-Algeria-Tunisia coastal zone. This is a composite index based on many aspects of the flow regime, so may be representative of other quantity-based nature indicators.

Western Europe
Overall result: the future is highly uncertain due to a high level of inconsistency and uncertainty across the region. That the flow regime will change is clear, but the direction of that change is not, varying significantly across the region and with both climate and socio-economic scenario.

Table 8.1 Aggregation of Nature indicator results

Region	**Impacts - Climate**						**Impacts – Climate and socio-economic**									
	Floodplain wetlands						Environmental flows		Ecosystem services		Water supply to wetlands		Macrophyte diversity in lakes		Habitat suitability (T) for fish	
	Volume		Duration		Timing											
	Focus	Uncertainty	Focus	Uncertainty	Focus	Uncertainty	Focus	Uncertainty	Focus	Uncertainty	Focus	Uncertainty	Focus	Uncertainty	Focus	Uncertainty
North Africa							-	L								
Western Europe	++/--	H	+/-	H	-	M	-/0	M	--/0	H	+/--	H	+/-	H	--/-	M
Northern Europe	+	M	+	M	-/0	M	-	L	-	L	0	L	+/0	M	0	L
Med/S Europe	++/--	H	+/-	H	+/-	H	--	L	--/-	L	--/0	H	-/0	M	--/0	H
Cen/E Europe	-	L	-	M	0	M	-/0	M	--/-	M	--/-	M	+/--	H	--/-	M
E/E Europe	-	L	-	M	-	L	-	L	-/0	M	-/0	M	+/-	H	--/-	H
Western Asia							--	L					--	L	--/0	H

Northern Europe
Overall result: the water for nature indicators show relatively little impact across the region with many remaining unchanged or showing a small increase or decrease from the baseline. Southern UK shows more similarity to the neighbouring Western Europe region than to the rest of the Northern Europe region.

Mediterranean Southern Europe
Overall result: the future is highly uncertain due to a high level of inconsistency and uncertainty across the region. That the flow regime will change is clear, but the direction of that change is not, varying significantly across the region and with both climate and socio-economic scenario.

Central Eastern Europe
Overall result: the water for nature indicators show consistently negative impacts across the region for all scenarios except SuE. The flow regime will change and flood volumes and durations will decrease and may occur slightly earlier due to changes in snow/glacial melt patterns. Other quantity and quality indicators show losses in ecosystem services, and decreases in wetland water supply, aquatic macrophyte diversity and fish habitat suitability, with a medium to high level of uncertainty.

Eastern Eastern Europe
Overall result: the water for nature indicators show consistently negative impacts across the region, particularly in the southern part bordering Western Asia. The flow regime will change and flood volumes and durations will decrease and may occur slightly earlier due to changes in snow/glacial melt patterns. Other quantity and quality indicators show losses in ecosystem services, and decreases in wetland water supply, aquatic macrophyte diversity and fish habitat suitability, with a medium to high level of uncertainty.

Western Asia
Overall result: water for the environment will be negatively impacted in terms of both quantity and quality. Results are only available for one water quantity indicator: environmental flows, which shows moderate to high impacts across the region. The two water quality indicators show that a decrease in aquatic macrophyte diversity and a decrease in fish habitat suitability are to be expected, though there is a high level of uncertainty.

Are there big differences between regions?
Table 8.1 shows that whilst the water for nature indicators show impacts across pan-Europe, the severity and direction of that impact is greater in some regions than in others. The least change will be felt in Northern Europe, which has relatively high water availability and low demand so can absorb any decrease in the former and decrease in the latter to some extent. This is followed by Eastern Eastern Europe and Central Eastern Europe which are likely to experience more severe impacts in terms of water quality than water quantity. The situation is more serious for Western Europe and Mediterranean Europe where the direction and magnitude of impacts is highly variable and uncertain. Water for nature quantity indicators are assessed after the other sectors are satisfied, so in these two regions the results are indicative of uncertainty across the whole water use sector. In Western Asia and North Africa less information is available, but it is likely that the areas already experiencing problems will see these worsen and new areas may start to see negative impacts.

Can socio-economic changes or climate changes be identified as dominant driving force of these changes?

Table 8.2 summarises whether climate change (CC) or socio-economic change (SE) seems dominant.

Table 8.2 Dominant driving force per indicator

Indicator/driver	**CC or SE?**
Environmental flows	CC
Floodplain wetlands	*no data*
Ecosystem services of wetlands	CC
Change in water supply to wetlands	CC
Aquatic macrophyte diversity in lakes	CC/SE
Habitat suitability for river water temperature for fish	CC