Simulation of the effect of riparian shading and nutrient reduction measures on phytoplankton in Middle Elbe basin (Germany)

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1. Introduction

The Middle Elbe basin is located in the North of Germany, and drain about 20% of the country. It ends with the dam Geesthacht about 50 km upstream of the city Hamburg and before any tidal influence. This middle part of the Elbe received about half of its water from the Labe basin in the Czech Republic and is further drained by the water-rich river Saale (30%) and three others (Schwarze Elster, Mulde, and Havel), each with a large basin mainly in the lowlands. The total Middle Elbe basin covers the area of 83,920 km^2, in which arable land use clearly dominates (55.5%), followed by 30% forests. In the sub-basins Spree (Havel) and Schwarze Elster large scale carbon and metal mining causes pollution and an imbalance of groundwater hydrology. Several urban areas are widely distributed in the Middle Elbe Basin sum up to total 21 million habitants. The most populated region is presented by the cities Berlin and Potsdam in the sub-basin Havel. Other centers are located in the sub-basin of river Saale (Halle and Leipzig) and along the Elbe river with the cities Dresden, Wittenberg and Magdeburg.

Under mean water level conditions the water has already an age of 63 hours (IKSE, 2005), when passing 367 km through the Czech Republic and arriving the main source of the German basin at Elbe station Schmilka. Water flow start in the region of the Giant´s Mountains and has crossed the Bohemian Cretaceous basin, confluence with river Vltava (Moldau) and cut partially volcanic bedrock. In the first German section (0-96 km) the river Elbe belongs to the Low Mountain region and cross the scenic river canyon in Elbsandsteingebirge with variegated sandstone and loess-covered lowlands. Near the city Meißen the river enters the North German lowlands (Figure 2). The river partially follows ancient glacial valleys formed during the Elster, Saale and Weichsel Glacial periods (Pusch et al. 2009). Active floodplains are to find subsequently in the plains of Middle Elbe (Gierk & de Roo 2008) which contribute to the nitrogen retention in the river system (Natho et al. 2012). Downstream of the city Magdeburg, the Elbe valley widens to about 20km and cross glacial deposits, and then enters a glacial valley partially used by the Havel tributary today (Pusch et al. 2009).

The river basin management plan (RBMP, FGGE 2009) has identified good ecological status only in 4.3% out of all the 2317 surface water bodies investigated. The water quality in the Middle Elbe basin is reduced by diffuse pollution (eutrophication and harmful chemical substances from former mining activities.) The good ecological status is additional disturbed by channelization and water regulation. 51% of total river length is classified as Heavily Modified Water Bodies (HMWB) or Artificial Water Bodies (AWB). Eutrophication is caused mainly via the pathway “diffuse pollution” from agriculture land. The current status of nutrient load is far from background conditions which have been reconstructed in former studies by the nutrient emission and transformation model MONERIS (Venohr et al. 2011; Wechsung et al. 2013; Becker & Venohr 2015). Scenarios were run to simulate the reduction effect of possible management measures listed in a catalog to reduce nutrient emission and to improve nutrient retention (retention ponds in drained agriculture) in the whole basin. According these future simulations the basin wide implementation of almost all measures will not be able to achieve sufficient nitrogen reduction to support high ecological status in coastal waters. Likewise, when applying a set of measures to reduce total phosphorus (TP) a model simulation of Quiel et al. (2011) revealed that also TP will remain on concentration level high enough to allow an enormous algal standing stock in Elbe river, on which grazers such as rotifers can establish a population size known for lakes in the spring bloom (Holst et al. 2002). Currently, high amounts of phytoplankton biomass are transported along the river and settle down in the tidal region, where they cause strong oxygen depletion as a secondary effect, which act as a barrier for diadrom-living fish species.

Therefore, the reduction of the eutrophication remains in the focus of management in the Middle Elbe basin. Supporting measures such as the increase in riparian vegetation is in debate not only to reduce nutrient input by erosion prevention and by improving denitrification. Furthermore, riparian vegetation can help to reduce effects of climate changes (heating and erosion), and reduce the incoming light for aquatic primary producers. As a support for this idea, a simulation carried out by a model (Hutchins et al. 2010) revealed that riparian tree shading has a high potential to reduce phytoplankton growth in small and mid-sized rivers. Regarding the river network of the Middle Elbe the share of rivers smaller than 10m is high (80%), but most of these naturally shaded rivers lost their riparian buffer zone when agriculture became intensified.

Therefore, we aim to establish a model chain between land use (including loss of riparian vegetation), nutrient emissions, water retention and nutrient transformation processes, to biological responses in a large freshwater river in Central Europe. The model is made to evaluate the ecological effects of multiple stressors acting in concert, e.g. nutrient loading, light increase and alterations in water flow duration. The chain of models is also used to run plausible future scenarios on climate, land use and management changes. As an ecosystem service the Middle Elbe basin provides already a strong nutrient retention in its tributaries which is modelled under improved riparian river conditions and future scenarios.

In this study, the undertaken modelling of Middle Elbe combine all stressors (climate, social, agro-technical measures) into three storylines for 2 future time frames.

2. Context for the modelling and storylines

The scenario concept is imbedded into the EU project MARS (Hering et al. 2015), in which three MARS –storylines are agreed to be implemented in the 16 European basins joining the project.

The projection to Middle Elbe result into the following scenarios:

Techno world (Storyline 1) has intermediate environmental stressor (overuse of resources), partly high technical or political measures are applied which is accompanied with relative high population increase (21.1% until 2050)

Consensus world (Storyline 2) put into practice improved sustainability (sustainable use of resources) and optimal technical or political measures accompanied with shrinking human population (-3.7% in 2050)

Fragmented world (Storyline 3) has the most strong environmental stressor (overuse of resources), and few or even reduced technical or political measures accompanied with strong shrinking human population (-15.9% in 2050)

Nutrient load and concentrations and phytoplankton status, biomass and composition are the modelled variables. The model output is used to evaluate ES and to provide advises for optimal measures under multi-stressor conditions.

a. Overall MARS model for the Basin

A statistic empirical model (EM Elbe) is derived from environmental and biological response data of monitoring station and combined with data for landscape morphology and landuse derived from common GIS maps and with WFD typology and status information of water bodies. EM Elbe is to identify the most important influencing factors and their interactions for biological response.

Process-based models are trained with current condition for simulating the MARS future scenarios which are driven by dynamic climate input data (RCP) and simulated discharge with the regional adopted model SWIM (Hattermann et al. 2015, Roers et al. 2016). The regional climate models RCMs RCP4.5 and RCP8.5 are applied in the MARS consortium driven by two global climate models, the ISI-MIP scenarios GFDL-ESM2M and IPSL-CM5A-LR and their results are feed into the hydrological model SWIM (Hattermann et al. 2015) in order to take into account the climate projection uncertainty (Roers et al. 2016).

Nutrient emission and transformation and the phytoplankton biomass of the large catchment Middle Elbe are simulated by MONERIS (Venohr et al. 2011) and the modul PhytoBasinRisk (Mischke et al., Annex 1), respectively driven by provided discharge simulated by the model SWIM (Roers et al. 2016). All three models are structured and computed for the regional scale into analytic units. By using the WFD river network, which designates water bodies, the model outputs can predict also ecological status for large water bodies and selected metrics.

Model applications focus on services for recreation and water purification (N and P-retention).

b. DPSIR model for the Middle Elbe Basin

Since the abiotic and biotic state of most water bodies are less than good because of high pressures, the river basin restrict response with management measures (FGGE 2009, 2015 a, b) which aim to improve also the Ecosystem Services (ES, Impact).

Using the Driving forces, Pressures, States, Impacts and Responses (DPSIR) framework (Figure 1), the modelling for Middle Elbe focus on the feedback of measures (Response) for reducing nutrient emission in urban waste water treatment plants (UWWTP) and in agriculture practice of the whole catchment with additional applications for the riparian buffer zone along the river net (develop riparian forest).

The observed and improved nutrient retentions in Elbe have a direct financial advantage for the society and can be directly translated as ES “Flow”. Any improvement by lowering “total algal biovolume and chlorophyll a” will improve the water quality and State, and help to maintain drinking water and raw water sources.

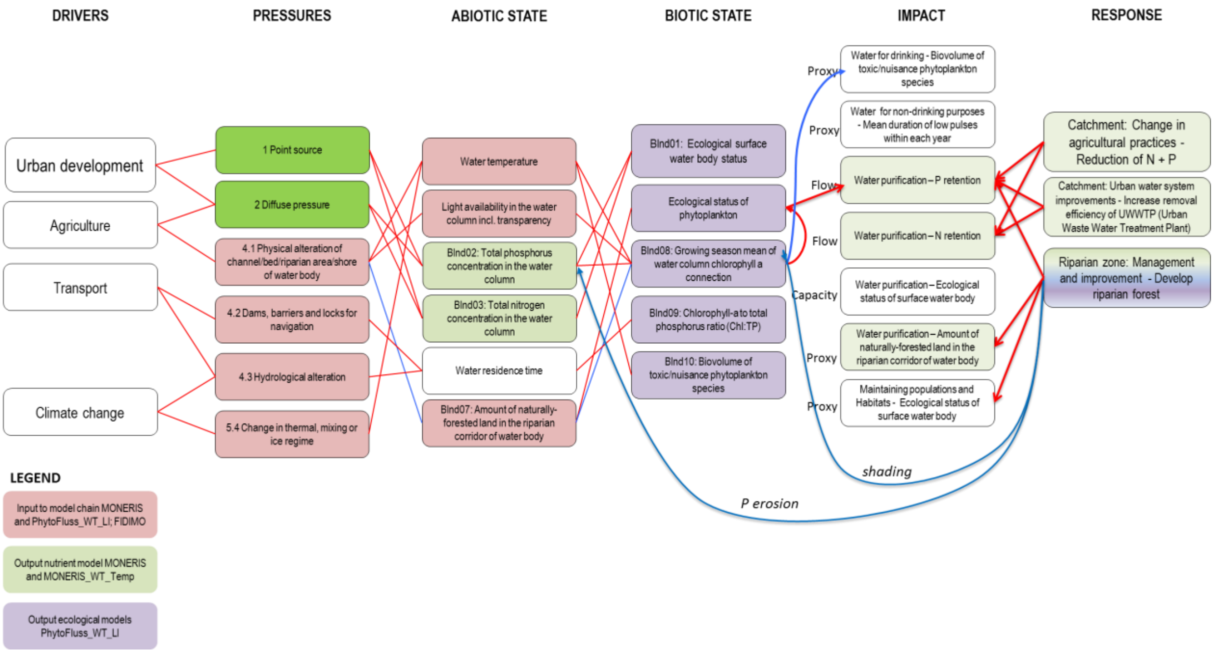


Figure 1: DPSIR model for the Middle Elbe basin

3. Data and methods

Training data

The Middle Elbe basin is an extensively monitored lowland basin (see table 1; FGGE, 2015b, 2nd RBMP). Here we refer only to a small part of all available data, which were selected for to have a nearby gauge station and for covering the period 2005-2010 almost continuously in monthly intervals at least for the vegetation period (April – October). Monitoring data were collected by the basin-sharing German Federal States under cooperation with the River Basin Community Elbe (FGGE). We linked the provided data with GIS information for population, municipal waste water treatment plants, landuse, slope (see Figure 2), soil type and river morphology and analytic units in a database.

Observation stations are evenly distributed (figure 2) within the four coordination regions (Table 1) and cover 73 different river water bodies.

Table 1: Number of rivers and lake water bodies (WB; 1st RBM-Plan (FGGE 2009)) and of observation stations for model fitting in the German coordination regions of Middle Elbe

		river WB	lake WB	stations	Analytic units
	Middle Elbe total	2318	344	104	722
River sections in regions	coordination regions				
Elbe - CZ/DE-border to Barby / Elde	MEL	409	69	28	146
Havel/Spree	HAV	982	213	21	210
Saale	SAL	354	35	18	181
Elbe from Barby to Geesthacht / Mulde and Schwarze Elster	MES	573	27	37	185

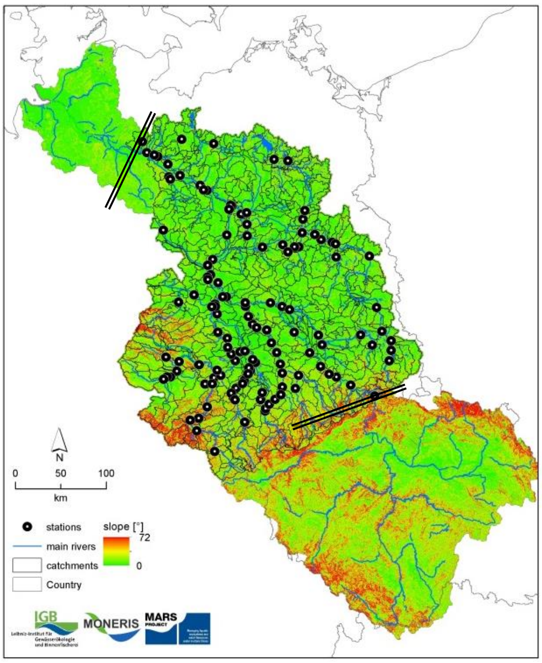


Figure 2: Landscape slope classes and observation stations with nearby gauges in the German Middle Elbe Basin with marker lines at main source station Schmilka at the Czech-German border and the basin outlet at dam Geesthacht.

Description of pre-processing of data

All spatial data for the study (information on land use and soils, and the digital elevation model) were projected to uniform GIS layers. The soil parameters are based on the German Soil Survey Map (BÜK 1000), and land-use data are based on the CORINE landcover classification published in the ATKIS data (© GeoBasis-DE / BKG (2013)) in 10x10m resolution. Watercourses are mapped as polylines, and as polygons when lakes or rivers are wider than 12m (for details see Annex I). The borders of analytic unit areas were taken from data of the Federal Environment Agency, which were used for nutrient emission modelling in former projects (Venohr et al 2015; Roers et al. 2016). The river network was joined to a map for water bodies provided by the river basin commission FGGE 2013 (DLM 5000 / BKG), which is used for WFD reporting (FGGE river network). Here, 3943 individual rivers segments belong to distinct river water bodies (EU_CD_RW) and were processed to extract following parameters for model application: a) borders of analytic unit, b) river category "main river (MR)" or "tributary (TRIB), c) river length, d) river width, e) land use in analytic unit, f) land use in the 10m-buffer at river shoreline and g) surface area of rivers in analytic unit. Also, 1247 lake polygons were analysized for: i) proportion of lake area tot total water body area in analytic unit, II) total lake lengths. The count of deep lakes was done by joining WDF reporting lakes (N = 283) to lake attribute list of DLM5000 /BKG. Other topographical parameters (slope in 100m, mean altitude), were derived from digital maps.

Climate data delivered by the external model SWIM were aggregated for each analytic unit for atmospheric conditions (short wave radiation at ground (named here as "global radiation"), air temperature, precipitation), and were additionally transformed to annual and summer means (see Table 7) and run-off were cumulated along the river network with the flow-net equation (Venohr et al.

2011). The calculation of incoming-light into the water body based on global radiation and water quality data and its reduction by riparian tree shading in described in Annex I.

Establishing an empirical model for Middle Elbe basin

An empirical model (EM) was established in order to identify interactions of stressors, which were included as abiotic variables into the model (Table 2).

A data set from monitoring and mapped data were compiled which include a list for environmental and abiotic variables. As a first step, a data screening was performed which follow the cookbook of Segurado et al. (2015a; Feld et al. 2016) and use a provided R script. Variables which show extreme positive skewness in the histogram were log-transformed (see “log” in legend in Table 2).

Table 2: Variables used to build up the empirical model for Middle Elbe and minimum and maximum of not transformed value in the vegetation period (Apr –Opct) in the years 2006-2010

type of variable	Legend	minimum	maximum
environmental			
site_PPTyp	river water body type		
ae	sub-basin size (km^2) log	186	125,413
slope	slope of analytic unit (m 0.1 km^{-1}) log	0.93	14.16
agri	% basin agriculture landuse	17.2	73.2
urb	% basin urban area	2.69	20.0
fors	% basin forest area	10.7	55.9
Q	Discharge (m^3 s^{-1})	0.01	1,126
prec	Precipitation at station (mm m^{-2} y^{-1})	0.56	146.5
abiotic			
TP	total phosphorus (mg/l)	0.04	0.70
PO4	dissolved reactive phosphorus (mg l^{-1}) log	0.005	0.43
NH4	ammonia (mg l^{-1}) log	0.005	0.77
abiNO3	nitrate (mg l^{-1}) log	0.005	8.9
O2	oxygen dissolved (mg l^{-1})	3.9	17.5
WTem	water temperature (°C)	6.0	27.3
response			
Chla	chlorophyll a (µg/L) log	1	375.9

For reducing the number of influencing variables they were checked for correlation to each other (Pearson's product-moment correlation), and dropped in case of correlation factor >0.6. For example, we kept in the two P variables (PO_4 and TP) although they correlate to each other with a coefficient of 0.56, but in the final model they have an opposite sign in the coefficient. When dropping one of both, the model has much higher residuals.

We first established a boosted regression tree (BRT) model and used the selected main explanatory variables to run GLM models (see Feld et al. 2016).

For analyzing the response of the benchmark indicator “chlorophyll a” to multiple stressors a GLM was derived according Segurado et al. (2015b). To reflect the spatial effect of the stations the catchment size (A_e) was included into model. A_e was strongly correlated to discharge at station (r^2=0.9), so we drop the latter variable. When including only the months within the vegetation period (April to October) the variable water temperature was less important for the model and was dropped by the GLM simplification step.

Process-orientated modelling with MONERIS and module PhytoBasinRisk

Nutrient emission, transformation and the phytoplankton biomass of the large catchment Middle Elbe are simulated with MONERIS (Venohr et al. 2009, 2011) in Version 3.0 (Venohr, unpubl.) and with the new module PhytoBasinRisk (Mischke et al. unpubl., Annex 1), respectively. Compared to other nutrient emission and water quality models the MONERIS model (MOdelling Nutrient Emissions in River Systems; BEHRENDT et al., 2000) and Modul PhytoBasinRisk work with a moderate demand of input data, requires only a short computing time and is applicable to large river basins.

In contrast to former model setups of Elbe basin with MONERIS (Behrendt et. al. 2000; Venohr et al. 2005; Becker & Venohr 2015, Wechsung et al. 2013) we have improved the setup with the following updated data: maps for landuse, N- and P-surplus, human population, connectivity to point sources and a new river net map, which is also used for EU-WFD reporting. By using the WFD river network, which designates water bodies, the model outputs can predict also ecological status for large water bodies and for selected metrics. The FGGE river network covers a total length of 33,000 km, to which main rivers contribute 18%. This detailed river network is still partly a simplification when comparing to rivers mapped in ATKIS data, where very small ditches are included, too.

The basin is divided into spatial analytic units and routed through the flow system to the catchment outlet at Elbe dam Geesthacht by a flow net equation (Venohr et al. 2011).

The process-based models are trained with current conditions (baseline runs) for simulating nutrients and phytoplankton biomass under conditions of the MARS future storylines driven by simulated climate variables and discharge conditions (see next chapter).

The MONERIS model simulate the substances total nitrogen (TN), TP and DIN on annual and monthly level as loads and concentrations (Venohr et al. 2011). The emissions are separated into pathways (see Table 3). The nutrient retention in the surface waters is calculated for each analytic unit and is calculated for the basin outlet as the cumulated basin retention by the help of the flow net equation for ecological service interpretations.

Table 3: Pathways separating the total emission in the model MONERIS

pathways emission	abbrivation for pathway
point sources	PS
urban systems without WWTP	US
ground water	GW
tile drainages	TD
Erosion	ER
surface run-off (dissolved fraction)	SR
atmoshaeric deposition on surface waters	AD

Climate and hydrological input data for scenario runs

Model MONERIS and module PhytoBasinRisk are driven by provided recently simulated discharge by the ecohydrological model SWIM (Soil and Water Integrated Model, Krysanova et al. 1998;

Hattermann et al. 2005), using the integrated module for computing the hydrology (Hattermann et al. 2015).

The baselines and scenarios are driven by external simulated monthly discharges for two different dynamic climate input data (RCP). In our study, we use long term mean (LT) for each month of the periods a) 1971-2001 b) 2020-2030 c) 2045-55 with climate variables precipitation (PP), global radiation (GR), air temperature (AT) and the modelled discharge in each modelled area (Q_AU).

The external simulated discharge data take into account the uncertainty of climate models (Hattermann et al. 2015, Roers et al. 2016) by applying two regional climate models (RCMs choice for MARS-project case studies is RCP4.5 and RCP8.5), driven by two global climate models four our setup (ISI-MIP scenarios GFDL-ESM2M and IPSL-CM5A-LR) and their results were feed into the hydrological model SWIM by several climate variables to calculate the discharge for all spatial analytic units for two training baselines (GFDL 1971-2001; IPSL 1971-2001) and for the future period 2010-2100. The hydrological simulation resulted in a high amount of daily data and further climate variables

In Table 7, we provide an overview of the discharge data and some climate variables finally used for our modelling and which were extracted for the different future periods.

According our model performance check, the monthly differences of climate variables and discharges are highly relevant for our model outputs. In case of PhytoBasinRisk, discharge fluctuations in the vegetation period can alter phytoplankton biomass strongly (see Annex I), while changes in the winter discharge are not directly relevant for phytoplankton development because of light limitation. As one example out of all 722 analytic units, climate driven differences in the cumulated mean discharges simulated by model SWIM for period April to October are listed at the basin outlet for baseline and for the future periods in Table 4.

The wettest long term mean is simulated for GFDL around the year 2050 with RCP8.5, and the driest projection is for IPSL around the year 2025 with RCP4.5. In contrast to former SWIM scenarios based on climate projections by STAR, which predicts dry summers for Elbe river in the future, the GCM-GFDL driven results revealed that summers will become near baseline (100 – 119% of baseline) or would be 86 – 91% of baseline in the future, when simulated discharges is driven with IPSL (Hattermann et al. 2015). Also, focusing on the vegetation mean the calculated water temperatures derived from simulated air temperatures are not very different from baseline simulations with a deviation in the range of 3 – 17% in future (see Table 4), while strongest climatic changes are simulated for winter months (not shown here).

Table 4: Simulated discharge ($m^3 s^{-1}$; cum Q) and water temperature (°C; WT) averaged for the vegetation period (Apr-Oct) at station Boitzenburg calculated as longterm means of each period, the baseline period and the two future periods and in two RCP-projections based on data provided by Potsdamer Climate Institute (Roers et al. 2016).

GCM-model ICI-MIP	Parameter as vegetation mean	*baseline 71-2001*	rcp4p5 2020-2030	rcp8p5 2020-2030	rcp4p5 20545-2055	rcp8p5 2045-2055
GFDL-ESM2M	cum Q	*627*	633	625	665	**745**
IPSL-CM5A-LR	cum Q	*656*	**565**	639	604	597
GFDL-ESM2M	WT	*17,4*	**17,3**	17,5	18,2	17,4
IPSL-CM5A-LR	WT	*18,1*	18,1	18,7	**19,7**	18,1

The full range of spatial (722 areas) and temporary differences (monthly for 11 years for 10 scenarios) in discharge and temperature are much stronger and additionally, high and low flow situations not occur synchronic in the whole basin, therefore simulations of specific sub-basins can differ from the overall mean at basin outlet.

Implementation of the scenarios in Basin Middle Elbe

The quantification of the scenario implementation is summarized in Table 6 and Table 7.

Land use: For the implementation of landuse into the MARS storylines 1-3 we followed global models summarized in Sanchez et al. (2015). For riparian buffer zones the current status of area without agriculture use was changed with minus 30% for scenario 1, plus 50% in scenario 2 and minus 80% for each individual analytic unit. The resulting change at all surface waters is provided in Table 6.

Human population and social effects: The German Federal Institute for Research on Building, Urban Affairs and Spatial Development (BBSR) analysed population demographic changes and trends in Germany and found a distinct decrease or rural population and increase of urban population (BBSR, 2016; “Rural exodus? Society on the move”). We take this trend into account by implementing it in different degree in the 3 MARS storylines (Figure 3; Table 6).

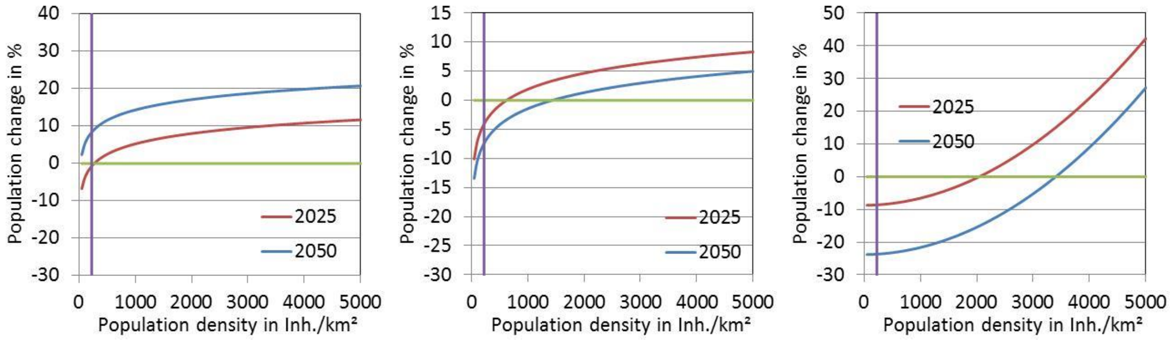


Figure 3: Assumed population changes in dependence of population density for Storyline 1 (left), 2 (middle) and 3 (right) for the years 2025 and 2050 compared to the zenus data 2010 for German municipalities (figure 11).

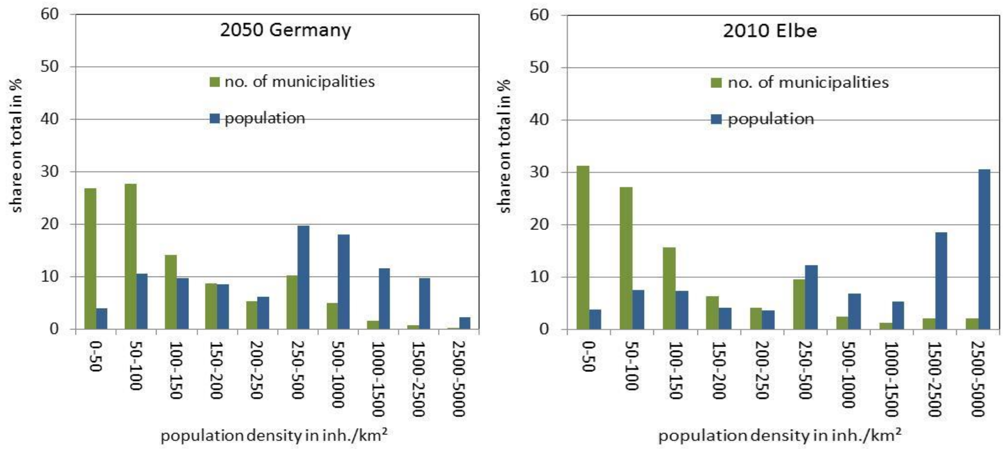


Figure 4: Share of municipalities and inhabitants in dependency of population density in Germany and the Elbe catchment according to zensus data 2010.

We assume that people in the cities will also change their nutrition, which will strongly effect the person specific P disposal (Table 6). In a more balanced diet with less meat, the P disposal will decrease strongest in storyline 1, less in storyline 2, but will be negatively change to more meat consumption in storyline 3.

Table 5: Final effluent of WWTP of different size classes for N and P assumed for the storylines (SL1 – SL3)

WWTP size	N			P		
class	<GK4	GK4	GK5*	<GK4	GK4	GK5*
SL1	20	18	10	1,5	1	0,5
SL2	10	8	5	1	0,5	0,3
SL3	30	25	20	3	2	1,5

*assumed only for the new connected habitants according population changes

Table 6: Change of emission input data to run MONERIS model for validation (2006-2010) and for the 3 MARS storylines in the future time periods 2025 and 2050.

	Story line				**1**	**1**	**2**	**2**	**3**	**3**
	period	**2006-2010**	**1971-2001**	**1971-2001**	**2025**	**2050**	**2025**	**2050**	**2025**	**2050**
arable land	%	41			23	22	23	22	23	22
Population	**Mio. Inhabitants**	**17.7**			**18.3**	**19.9**	**17.7**	**17.1**	**17.6**	**14.9**
connected to WWTP	% of population	90			90	90	90	93	91	91
person specific P disposal	g/inh./day	1.90			1.74		1.31		2.81	
N-surplus TS	kg/ha/yr	50			46		40		63	
P accumulation	kg/ha	730			621	619	555	548	831	824
DPS arable	%	77	77	77	67	67	59	59	80	80
buffer strips	% of surface waters	83			58		100		17	

Table 7: Climate input data summarized as longterm mean of the periods: validation period (2006-2010), climate baseline (1971-2001 for GFDL and IPSL) and future MARS storylines in 2020-2030 (2025) and 2045-55 (2050)

	period	**2006-2010**	**1971-2001**	**1971-2001**	**2025**	**2025**	**2025**	**2025**	**2050**	**2050**	**2050**	**2050**
	RCP				**4.5**	**4.5**	**8.5**	**8.5**	**4.5**	**4.5**	**8.5**	**8.5**
	Climate		**GFDL**	**IPSL**	**GFDL**	**IPSL**	**GFDL**	**IPSL**	**GFDL**	**IPSL**	**GFDL**	**IPSL**
Precipitation	mm	742	717	732	742	768	750	791	767	787	766	793
summer precipitation	mm	392	392	400	392	423	395	451	403	436	386	448
annual run-off	m³/s	769	843	862	867	840	874	862	925	823	971	839
summer run-off	mm	591	552	586	561	524	568	599	611	564	655	563
mean annual water temperature	°C	10.9	10.9	10.9	11.8	12.3	11.6	12.3	12.1	13.2	12.1	13.6
mean summer water temperature	°C	17.9	17.9	17.9	18.3	18.9	18.2	19.2	18.7	19.8	18.9	20.4
annual incoming short wave radiation at ground	w/m²	120	120	120	122	124	121	126	122	126	119	126
summer incoming short wave radiation at ground	w/m²	172	172	172	176	179	175	183	177	184	175	184

For emission pathways we keep the population connected to waste water treatment plants (WWTP) on the same level as baseline, but in storyline 2 the technical level is the highest, and in storyline 3 the lowest for WWTP. The final effluent from WWTP was assumed to change according Table 5.

The climate scenario data provided by Roers et al. (2016) includes the simulated annual run-off for each spatial analytic unit and were used as monthly longterm mean for each future period (annually summarized in Table 7).

The climate and abiotic data to run scenarios with module PhytoBasinRisk were taken from the climate model (SWIM) and from the nutrient model (MONERIS). In addition, current status of area with trees or bushes in the 10m buffer was changed with minus 30% for storyline 1, plus 50% in storyline 2 and minus 80% for storyline 3 for each individual analytic unit. In dependency to the mean river widths this results in less strong shading effect, since only a part of tributaries are smaller than 7m, for which optimal shading is assumed (see Annex 1). At tributaries the simulated mean tree shading factor is 0.83 in current status, 0.97 in storyline 1, strongest in storyline 2 with factor 0.53 (the measure response “develop riparian forest” is realized) and no tree shading in storyline 3 (shading factor 1).

Estimation of ecosystem services

The ecosystems services are calculated for nutrient retention and for improving ecological status.

a) Calculation of nutrient retention in tons per year by model MONERIS to costs, which would be necessary to reduce them in WWTP. The cost base is redrawn from a regional study for tributary Havel in Horbat et al. (2016).
b) Simulation of chlorophyll a concentration and their relation to ecological status (Mischke et al. 2011).

4. Results

Performance of nutrient emission modelling by MONERIS

Overall, the output of the MONERIS model was compared to other emission model in several studies and turned out to be in the same range of uncertainty than other (Kronvang et al. 2009; Malagó et al. 2015).

Modelled loads were transformed to nutrient concentration by cumulated discharge (Q). The simulated nutrient results were compared to monitoring data for main stations. The seasonal variation is sufficiently reflected by the model (Figure 5) and mainly within the ±30% confidence interval (Figure 6).

When comparing the simulated loads with the expected 1:1 line, there is no systematic under- or overestimation is simulated.

Additionally to the basin outlet the nutrient load for 12 further stations are compared. The correlation coefficient for TN loads for simulated to observed annual means range between 0.85 – 0.99, and for TP between 0.31 – 0.81, respectively. The lowest match is simulated for TP load for the outlet of river Havel, in which the water quantity management for operating a water bypass for a shipping course may have led to unusual discharge conditions in combination with retention in lake-river systems.

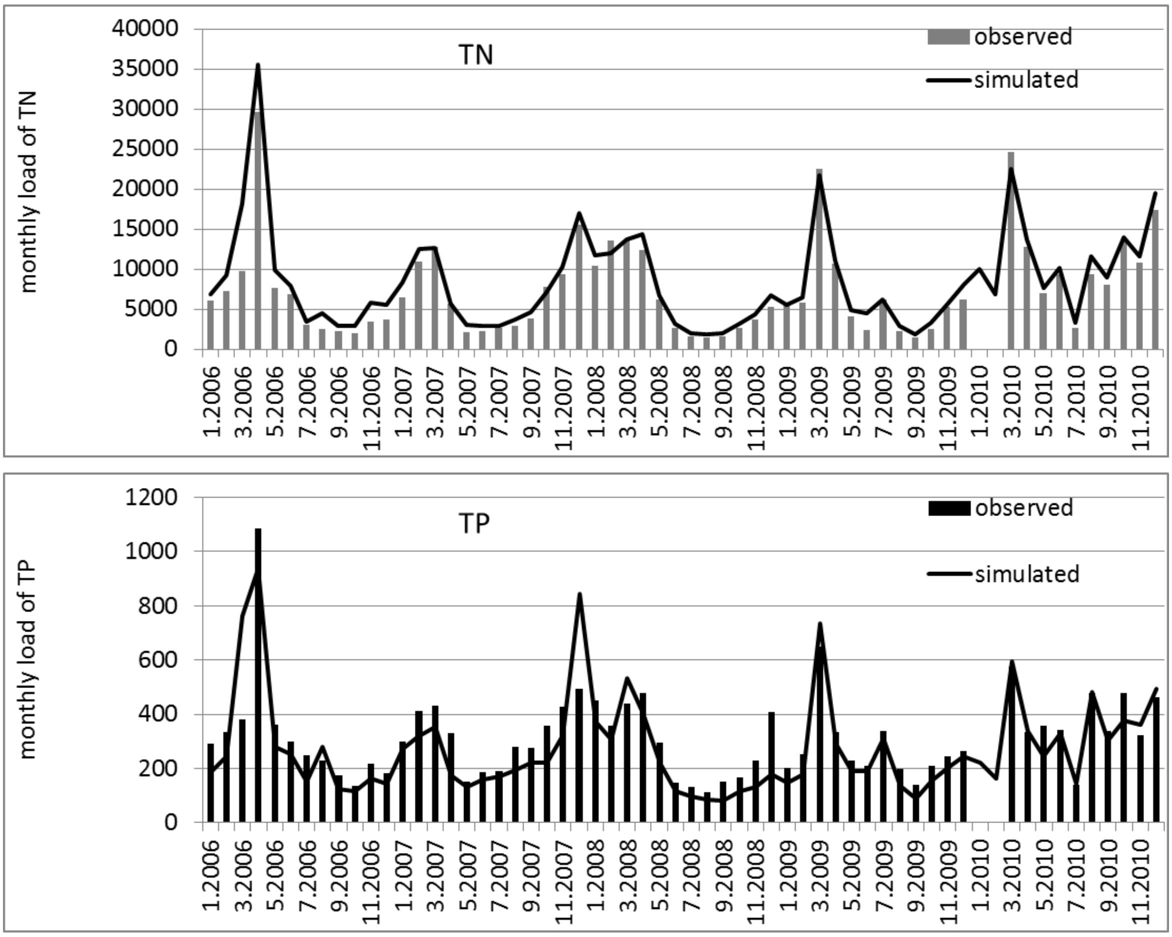


Figure 5: Observed and modelled monthly load of TP and TN at station near basin outlet (Neu Darchau)

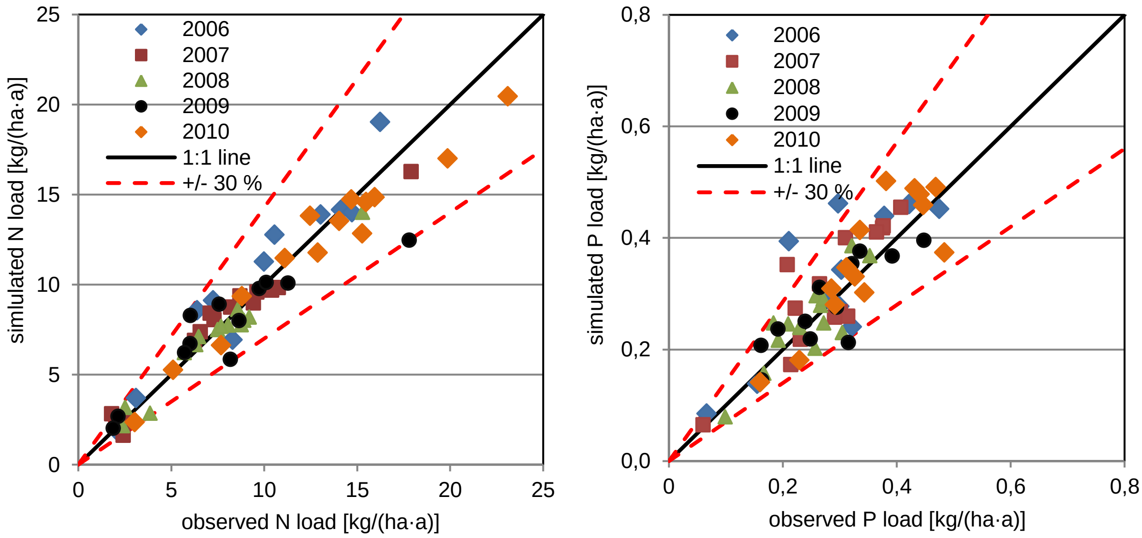


Figure 6: Modelled and observed monthly loads for TN (left) and TP (right figure) different colored for the years at basin outlets of Middle Elbe in years of validation period. Hatched lines indicate the ±30% confidence interval.

Performance analysis for module PhytoBasinRisk

Modelled chlorophyll a concentrations were compared to monitoring data for several main stations (see Figure 7). The seasonal variation is reflected by the model. For ecological status assessment the vegetation mean of the period April-October is used (Figure 8), so the low peak match is not in front of the simulation task.

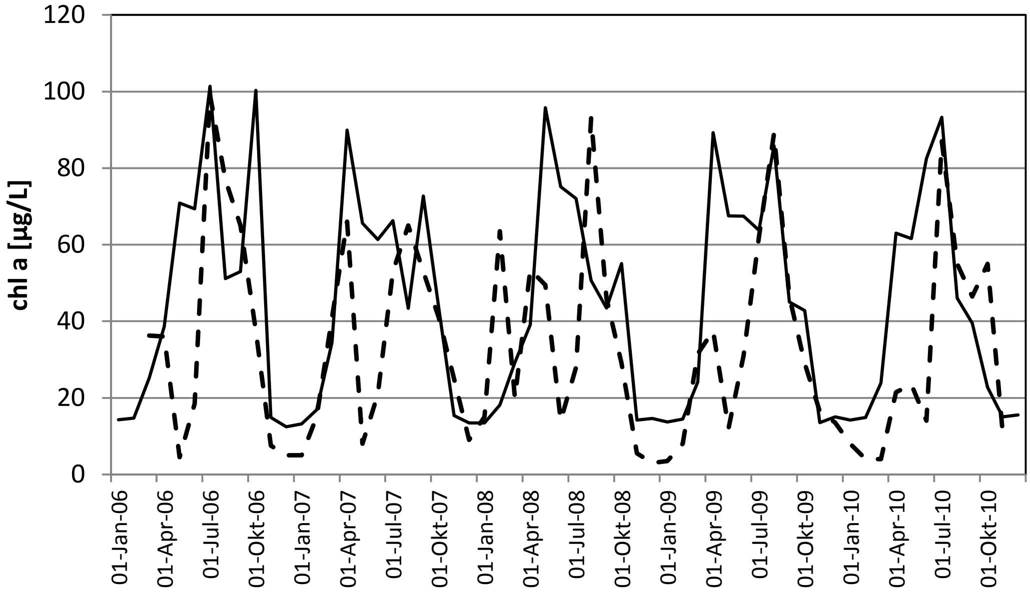
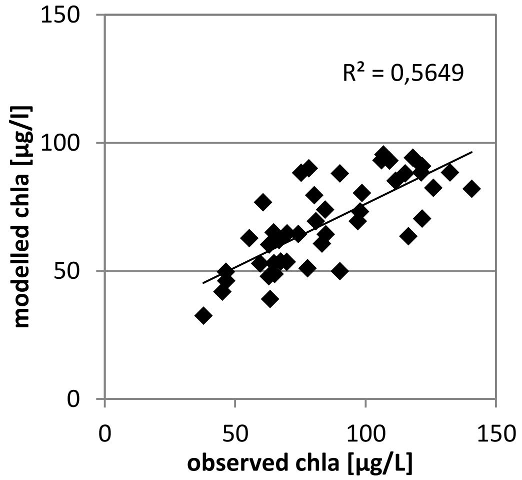


Figure 7: Observed (hatched black line) and simulated chlorophyll a concentration as monthly means modeled with PhytoBasinRisk in river Spree at station Jannowitzbrücke



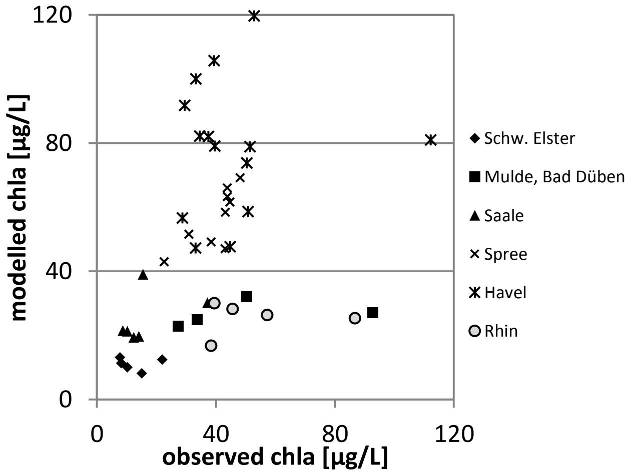


Figure 8: Observed and modelled chlorophyll a (chla) seasonal means in Elbe stations (left figure) and in tributaries (right figure)

A synthetic data set was produced to test sensitivity of the PhytoBasinRisk module for single variables. The input data of the longterm mean run for period 2005-2010 was altered with reducing P by 20% or 30%, increase water temperature with 2°C (pl2Temp) and run shading effect as assumed for storyline 2 (SL2_shade) for each of the spatial analytic units. The effect on simulated chlorophyll a is shown for a smaller catchment, in which shading effect can be expected because of smaller rivers (Figure 9).

The simulated summer concentrations of chlorophyll a do not response to P reduction because 30% reduced concentrations are still above the limitation thresholds. The chlorophyll a output clearly increase with increased temperature and are reduced by riparian shading when high share of trees in the 10m buffer were assumed as implemented for storyline 2 (upper Figure 9, SL2_shade).

These clear effects by temperature and tree shading vanish in larger catchments, because of biotic regulation: Higher grazing losses are simulated for rivers wider than 35m, when chlorophyll a concentrations > 50µg/L flow in from upstream sections, and temperatures are above 14°C (see Annex 1).

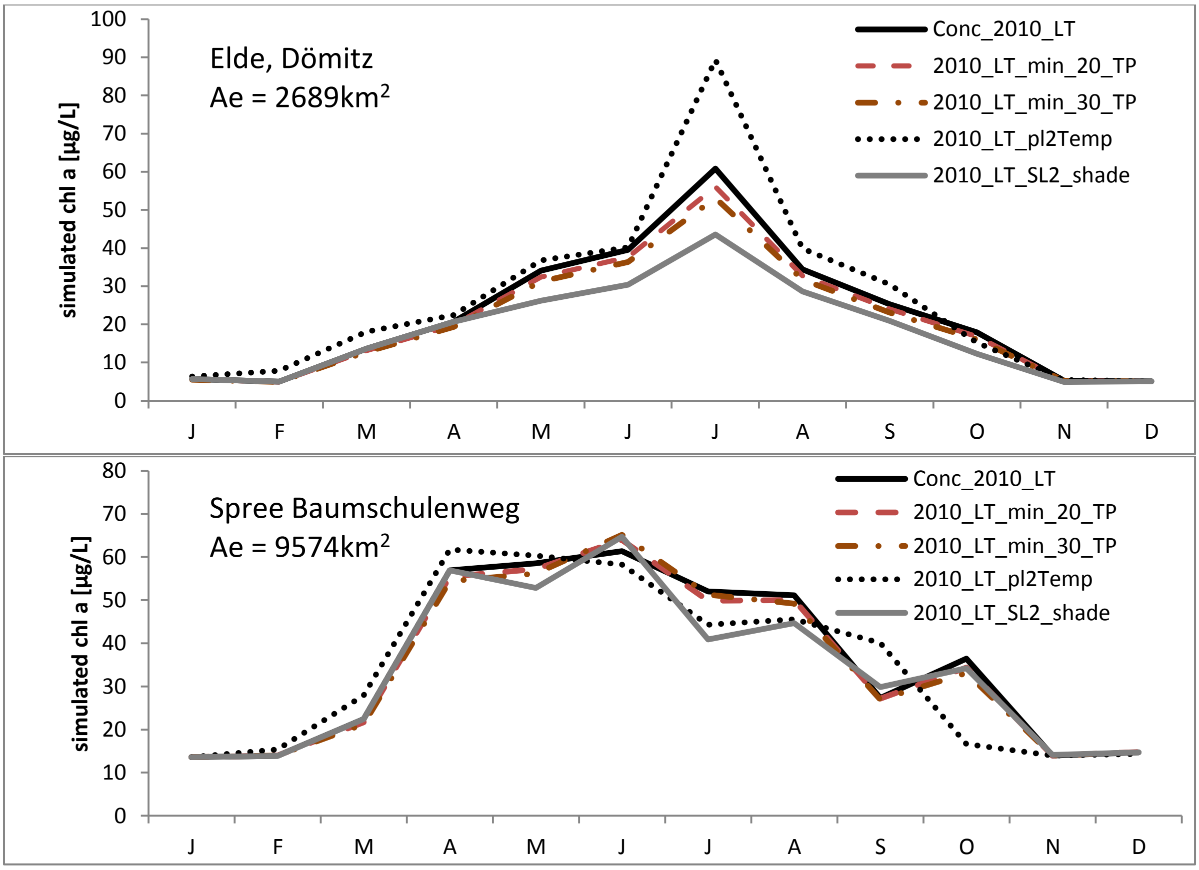


Figure 9: Monthly simulated chlorophyll a concentrations in sub-basins Elde (Dömitz) and Spree (Baumschulenweg) under synthetically altered input data (see text) in comparison to simulation for currents means of the period 2006-2010.

Uncertainty in the PM results

The Middle Elbe basin is large, so for several parameters the data are not available in high spatial resolution. The river net comprises high- and lowland regions to which more than 200 large lakes and reservoirs are connected. Therefore, uncertainties for nutrient emission and transformation result from several sources of the input data, the transformation in the lakes and in the estimation of the concentrations in the groundwater.

The PM model PhytoBasinRisk reflects the best the conditions in the main channel of the Elbe, while the different tributaries are mainly heavily modified by series of dams (Saale), water flow is managed by upstream reservoirs and dams (Spree), a chain of lake-river-systems are connected (Havel) or the river is dominated by macrophytes (Müggelspree). Estimations of water residence time (WRT) for all of these hydrological modified conditions are rough and causes the highest uncertainties in modelling chlorophyll a concentration. Flow times are available for only the main courses of the total river network (IKSE 2005). Uncertainty is also increased by lake outflows which are covered in the model by a common approach so far (sedimentation loss per lake-lengths; inoculum chlorophyll a in analytic units at the begin of the river net arms, WRT>30d for deep lakes, see Annex 1), but each lake has individual transformation processes (morphology, extent of macrophyte cover, fish-zooplankton interaction).

Empirical model results: Interaction of explanatory variables

The empirical model derived with a GLM model (GLM –model Middle Elbe, Gaussian family) with response variable "chlorophyll_a concentration" based on seasonal data for Middle Elbe was as followed (for abbreviations see Table 2):

log_Chla = -2.36253 + -10.99 * logPO4 + 19.43 * TP + 0.35 * NO3 + 0.65 * Ae + -0.065 * (Ae * NO3) + -1.52 (Ae * TP).

The two latter terms of the model are two synergistic interaction terms for catchment size to nutrient variables. No interaction was identified between two stressor variables.

Fitted directly to the training data set, the GLM model Middle Elbe is able to predict what is to expect when nutrient level is lowered by reducing management measures, as for storyline 2 in the Consensus World of the project MARS. For this purpose, the log-transformations (log10 +1) of variables required for statistical needs, must be converted back.

For example, starting with upstream station with 75,000km^2 catchment size and current nutrient concentrations (PO4 0.01; TP 0.12; NO3 1.4 mg/l) the model would predict 319µg/L chlorophyll a, which is occasionally realized in Elbe channel. I nasecond step, we can test the effect of a TP concentration of 0.085 mg/L in future simulation according nutrient modelling after setting a bundle of measures to reduce nutrients (see MONERIS in storyline 2): This reduction would restrict chlorophyll a to 53 µg/L, which is at the upper border for good ecological status according the German assessment method PhytoFluss (Mischke et al. 2011).

The model performance is shown in figures 8-9 and summarized in Table 8. All variables and both interaction terms are highly significant. Also the residual analysis for each of the variables revealed no trend (Figure 11). The simplified model revealed an cv correlation with 0.772. The other model performance indicators are se = 0.013; the mean total deviance = 1.228; mean residual deviance = 0.325; estimated cv deviance = 0.5; se = 0.027 and training data correlation = 0.859.

The Akaike Information Criteria (AIC) is lowest (AIC 2992.9) when both interaction terms are included in the final GLM model.

To take into account temporary random effect by analyzing several observation years, GLMs were built up for each of the 5 years. Each of the annual GLM revealed the same variables with slightly different influence proportion (for final model see Figure 12. Facing water residence times of only 8-10 days in main channel of Elbe in the German part and less than month in most of the tributary sections, we assume that the simulated value of the month before had not a strong influence on the next month. Instead, the exchange of the running water implies that for each month new nutrients emissions enter the surface waters and the recently produced algal biomass is washed out by longitudinal transport.

The indicated influence and sign of each of the variables in the GLM can be functionally explained by the following interpretations:

PO4

The negative sign in the coefficient of dissolved phosphorus (PO4) is functionally produced by algal uptake in our plankton-dominated river system. As in lake systems the PO4 is consumed up by phytoplankton or other plants, but extreme rapid P recycling from TP and sediment and rapid uptake causes the phenomenon that phytoplankton can still grow at very low concentrations of PO4, and growth is more depended on TP concentration. The integration of PO4 in the model operate as an indicator for the presence of algal biomass when low.

TP

Although P is in-cooperated into phytoplankton biomass, and phytoplankton is also a fraction of TP, the Pearson correlation between both variables is low (corr. = 0.1032). As expected from the Vollenweider model, TP is the most positive influencing variable for algal growth.

NO3

Nitrate (NO3) is available in the basin in surplus, except of river arms with a lake-river-system such as river Rhin or Müggelspree, in which summer denitrification can cause a depletion of NO3 and NH4 to critical limitation threshold below 0.15 mg/L. These limiting situation occur in the training data set in 3.7% of all cases.

Ae

The size of the catchment (Ae) at a sampling point trigger the water residence time for algal to grow, increase the risk to receiving larger emissions and is also strongly correlated to the discharge (corr. = 0.899).

Ae * NO3 and Ae * TP

Both interaction terms have a weak negative sign, which indicate that increasing site of the catchment decrease the effect of nutrients (NO3; TP), antagonistic effect (single variables are all positive). The integration of the interaction terms make the model much more realistic and prevent for overshooting for stations with large catchment size, and for overestimating in those with small catchment.

Table 8: Model summary for GLM for Middle Elbe with response variable chlorophyll a

Deviance Residuals:

Min	1Q	Median	3Q	Max
-3.4612	-0.6443	0.0111	0.6175	3.2129

Residual deviance: 839.65 on 1193 degrees of freedom

Null deviance: 1473.84 on 1199 degrees of freedom

Coefficients:

	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-2.36253	0.41241	-5.729	1.28e-08 ***
PO4	-10.98757	0.57190	-19.212	< 2e-16 ***
TP	19.42877	1.78226	10.901	< 2e-16 ***
NO3	0.35098	0.09160	3.832	0.000134 ***
Ae	0.64658	0.04765	13.571	< 2e-16 ***
NO3 : Ae	-0.06495	0.01044	-6.224	6.71e-10 ***
TP : Ae	-1.52323	0.19823	-7.684	3.20e-14 ***

Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

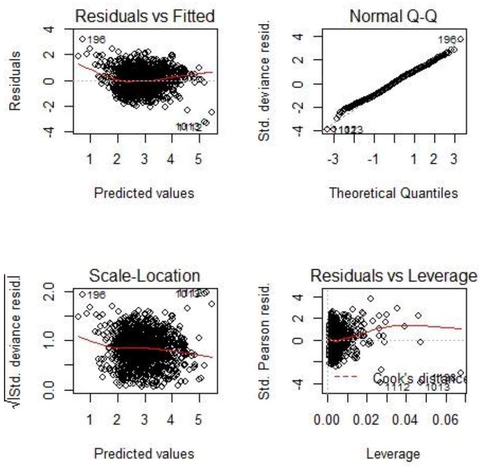


Figure 10: Model performance of the empirical model derived from GLM model.

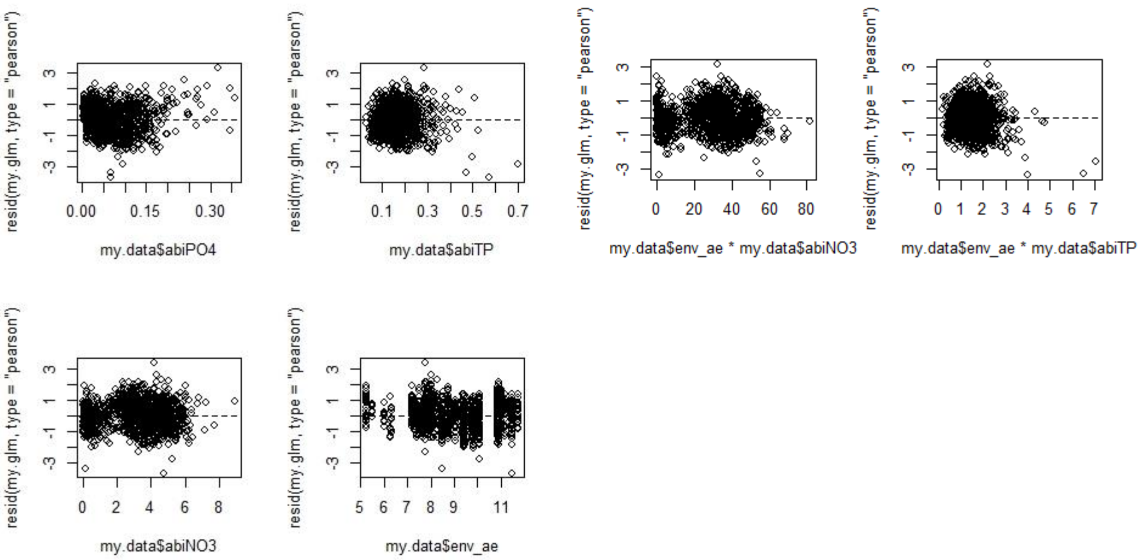


Figure 11: Residual for each of the terms used in the GLN model for Middle Elbe for response variable chlorophyll a-

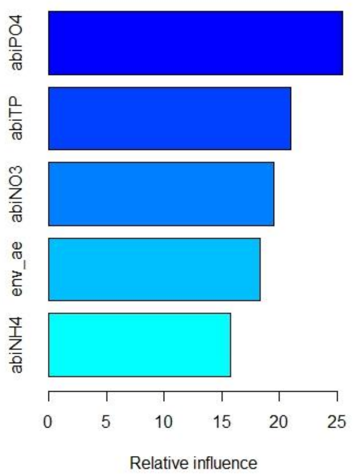


Figure 12: Relative influence of each of the variables to the GLM

Constrains to use the GLM model for predictions in Middle Elbe

The derived empirical model to predict the chlorophyll a concentration is simple in respect to the variables which are known to influence the phytoplankton growth (light exposure, growth time, silicon, nitrogen and phosphorus limitation) and losses (sedimentation, grazing by mussels and zooplankton, etc.).

Climate factors were dropped during the model simplification process such as precipitation and water temperature, but are known to be important. Furthermore, mean monthly conditions were analysed which do not reflect the short dividing rate of the phytoplankton, so uncertainties occur due to unknown interim disturbances and losses within a month.

The model setup suffer also from data limits: a) Especially upstream sites are less continuously monitored than those on the main channel (2005-2010) and b) data are not independent.

For developing a reliable model, the number of independent observations is crucial: Feld et al. (2016) recommend at least ≥150 independent observations and main stressors having ≥75% of the full gradient's lengths, to achieve low errors and a high goodness of- it (expressed as R^2) of the final averaged model. We do not have such a high number of independent observations at one site.

Furthermore, phytoplankton is transported along the river network, so upstream station will influence downstream phytoplankton biomass. Therefore, we can expect dependency of catchment size (Ae) to resulting response, but what we found was an low linear correlation (log: log r^2 = 0.104; raw data: r^2 = 0.21) using Ae to predict chlorophyll a for our 1,096 observations of 56 sites which partly are subsequently arranged along main channel Elbe or in its tributaries. This is in accordance with distribution of chlorophyll a concentration within the river net: Interim algal blooms in the tributaries Saale and Havel drop down before the confluence with the main river Elbe, and chlorophyll a concentrations not simply increase with catchment size at sub-sequence stations but interim drop downs can be observed (losses; see process- orientated module PhytoBasinRisk).

Results of modelling nutrient loads with process-orientated model MONERIS

At basin outlet, total nutrient emissions simulated for TP and TN for baseline and future scenarios show clearly the same trends when driven with ISI-MIP scenarios GFDL-ESM2M and IPSL-CM5A-LR (see Figure 13). In storyline 3 (Fragmented world) the emission is highest, and for storyline 2 (Consensus World) the lowest. Although highest human population is assumed for storyline 1, the implemented technical upgrade of WWTP, less N surplus and more balanced food will compensate human increase in the Techno World.

The GFDL socio-climate scenarios provide overall higher emissions for TN than IPSL. The TP difference is small. TN emission is dominated by the pathway groundwater, TP reach the surface waters from point sources (WWTP) and urban systems.

The nutrient emissions are transformed and retained within the river system (see next chapter) and are diluted by simulated discharge to final concentrations (Table 9) which effect the phytoplankton growth when limiting thresholds are surpassed.

While TN emission is reduced up to 33% of baseline, the resulting concentration reduction is less strong (Table 9).

The changes in monthly discharge frequently cause peak loads, while the resulting concentration remains low by dilution. This fact is demonstrated in Figure 14, for which the scenario with the highest TN emission on annual base was selected.

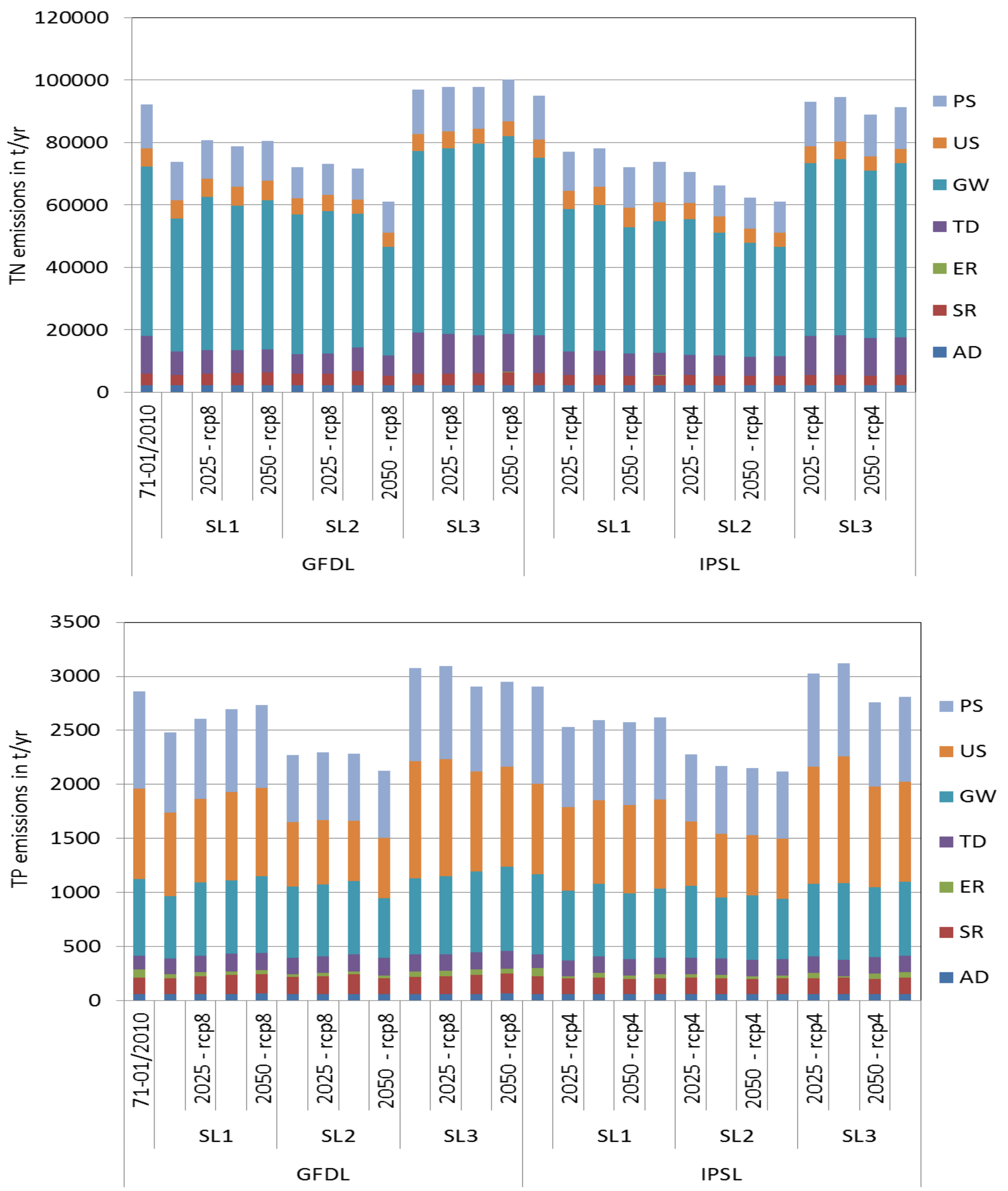


Figure 13: Simulated emission of TN (upper graph) and TP (lower graph) at outlet of Middle Elbe basin by model MONERIS (vs. 3.0) for longterm monthly baseline simulation (71-01/2010) and for all three storylines (SL1 – 3) driven by ISI-MAP climate scenarios GFDL or IPSL for RCP 8.5 or 4.5 and as a long term mean for the periods 2025 (2020-2030) and 2050 (2045-2055). The emissions are separated for the pathways listed in Table 3.

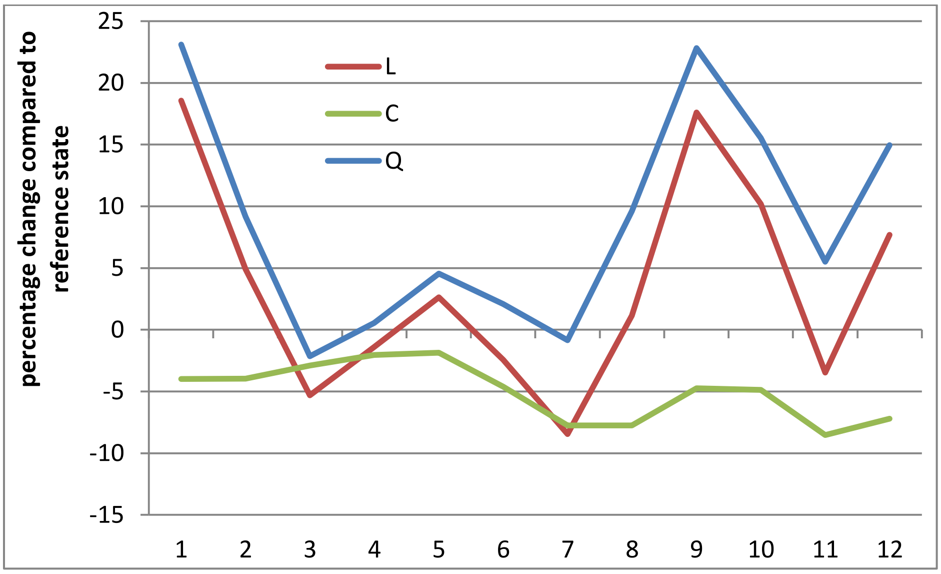


Figure 14: Comparison of simulated monthly values of cumulated discharge (Q), TN loads (L) and resulting TN concentration (C) for storyline 3 driven with climate GFDL and RCP 8.5 at basin outlet for the future period 2050.

Results of modelling phytoplankton biomass with PhytoBasinRisk

On monthly base for longterm means of each future periods, chlorophyll a concentration will decrease under GFDL climate simulation in all storylines and strongest in period 2050 with RCP 8.5 in comparison to GFDL baseline simulation for longterm period 1971-2001 (Figure 15). Spring bloom in April will be lowered by 40% reduction, while summer months differ from baseline with -10 – 15%. In contrast running the same landuse, population and management option changes for all storylines as in GFDL under IPSL climate simulations, at least for storyline 1 and 3 chlorophyll will increase up to 30-40% in some months (April, September) in comparison to baseline (IPSL 1991-2001).

On annual level, the differences between the two IPS-MIP climate scenarios are stronger than those between the different MARS storylines for station near basin outlet. As so, when a single month is simulated as extreme dry or wet for the longterm mean of a future period, this signal remains visible in all 3 storylines: For example wet month April in GFDL for 2050 in RCP 8 with 300m^{3} s^{-1} more than in its baseline run reduce simulated chlorophyll a by 40%.

The model was initially run with TP and TN simulated by model MONERIS, and a second run with using simulated dissolved inorganic nitrogen (DIN) instead of TN revealed very similar differences to the baseline, while the seasonal means of chlorophyll a- concentrations change with -5 -10%.

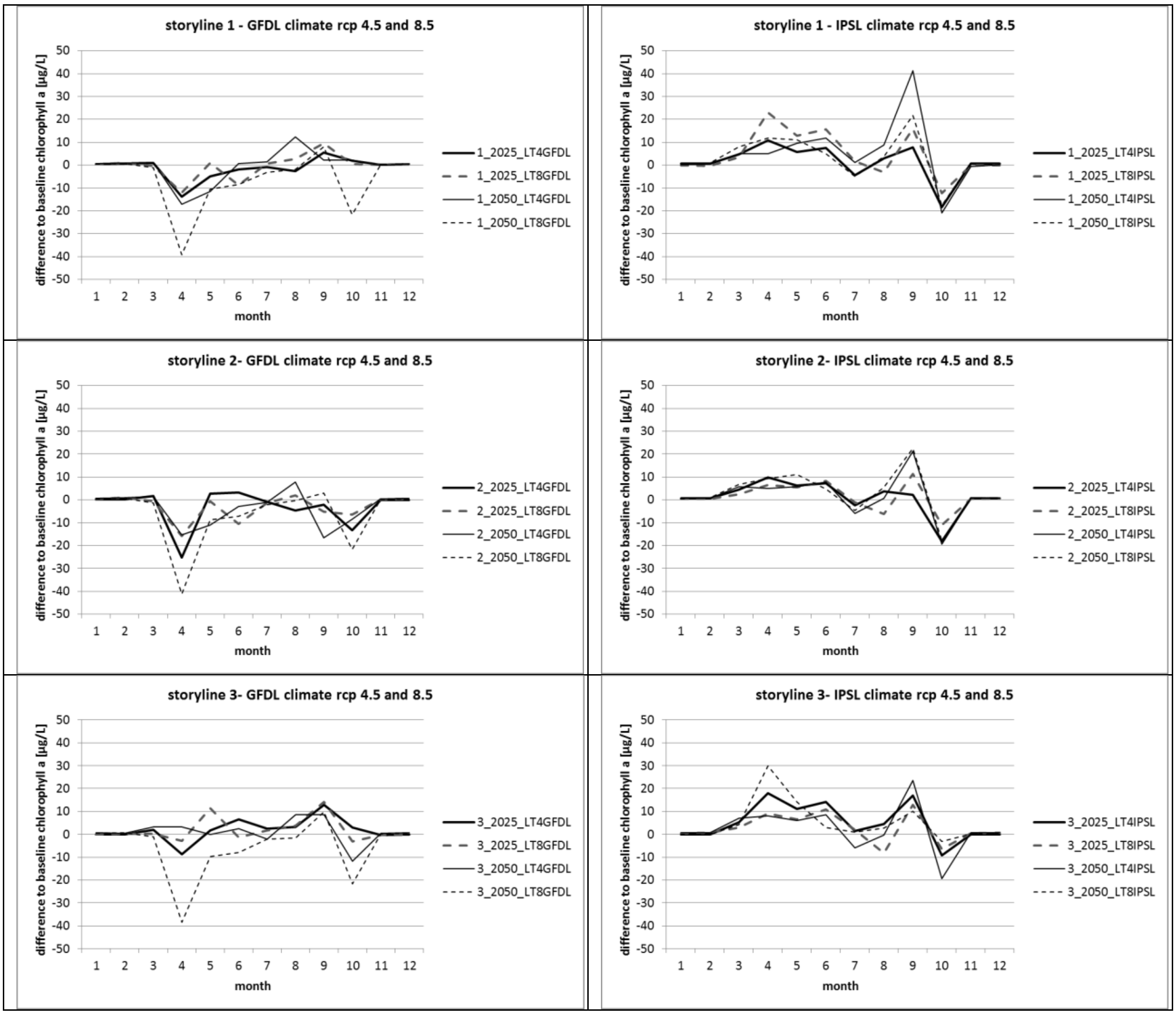


Figure 15: Change of chlorophyll a concentration to baseline (zero) for each month at basin outlet (station Boizenburg) modeled by PhytoBasin Risk in the three MARS storyline driven by simulated nutrient concentrations (MONERIS), and discharges (SWIM) for two climate scenarios and two future periods (2025; 2050)

c. Ecosystem service modelling results

Ecosystem service derived from nutrient retention and reduction of costs

In the German Middle Elbe basin occurs a strong nutrient retention in the river system. Comparing the emission with final loads, the retention contributes to reduce TP with a share of 39-41 % of the total emission at the basin outlet and TN with 25-26%. While TP retention is high in the tributaries only, the main channels contribute high denitrification rates in summer. Retentions simulated for all future scenarios are in the same range as the inter-annual variance observed for the period 2006-2010 (first 5 balks in Figure 16).

In absolute values 38,636 – 40,631 tons TN per year are retained or lost (denitrification) in the river system. To eliminate such high amount in WWTP it would cost the society 1932 – 2031 million Euro, when assuming cost of 50 € kg^{-1} N (see Horbat et al. 2016).

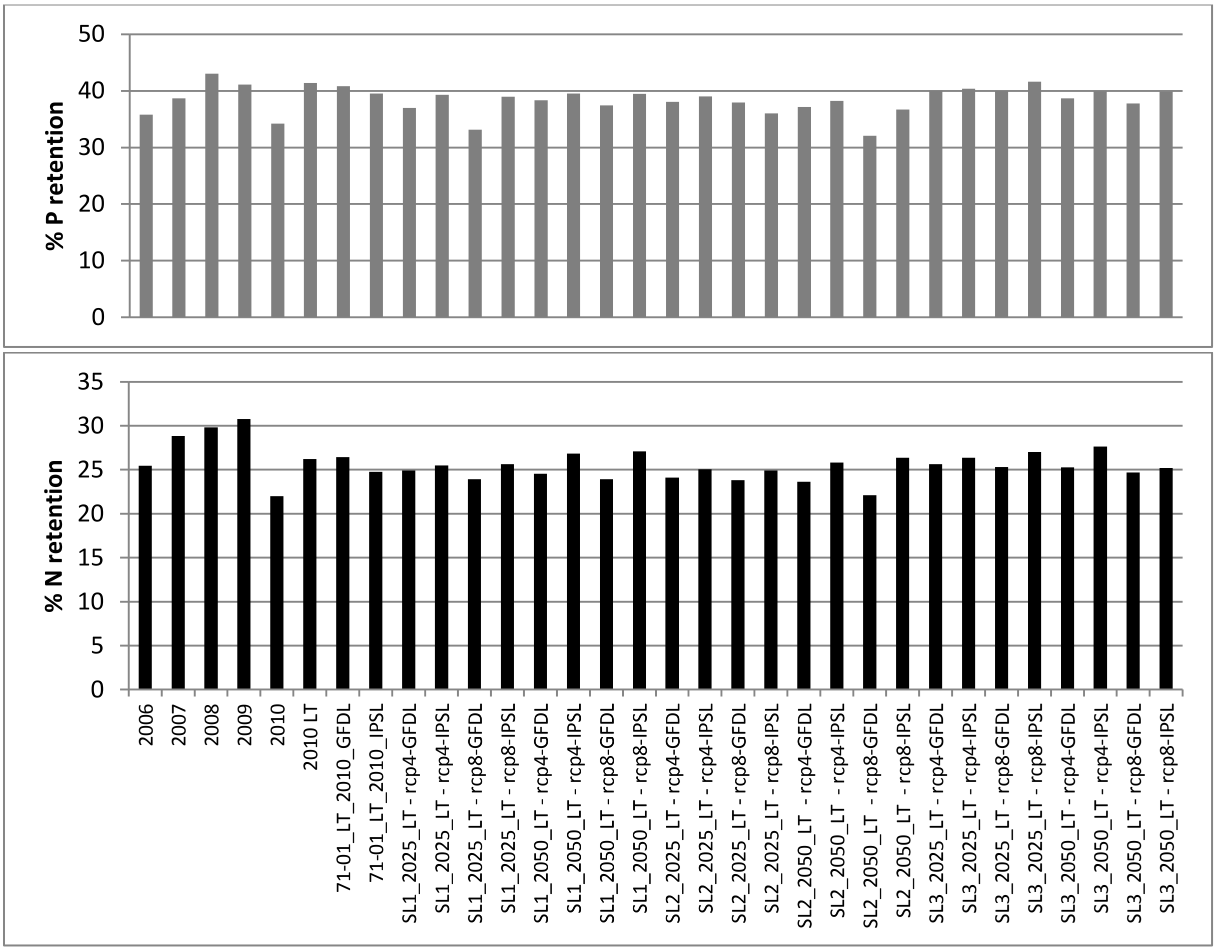


Figure 16: Proportion of calculated nutrient retentions (total P; upper graph; total N; lower graph) of total load in German area of Middle Elbe basin.

Ecosystem Services derived from status improvement (chlorophyll a)

For main river Elbe the chlorophyll a concentrations will change in a range of 58 - 81µg/L (near basin outlet; Table 9). This range corresponds to an ecological status class of “moderate” or “poor” according the current assessment classification (Mischke et al. 2011). Improvement from status class “moderate” to “good” (<26µg/L chla) is simulated for the sand – and clay dominated lowland rivers (type 15, 17), so at 3 stations of river Saale in the storyline 2.

TP concentrations higher than 0.075mg/L enable still high algal biomass with 58µg/L according our simulation with PhytoBaslin risk. In contrast, the empirical GML model predicts that resulting algal biomass would already decrease to a level of about 45µg/L chl a at this TP level, which would be in good status.

Key results of the scenario analysis

The achieved nutrient reduction in the Consensus World (storyline 2) with concentrations below 0.85µg l^{-1} TP is not long lasting for all months in the vegetation period and is not strong enough to limit vegetation mean of phytoplankton biomass, which is still above 65µg l^{-1} chlorophyll a in the seasonal mean (Table 9). Also, the assumed shading effect by riparian tree vegetation is not effecting the response output in the main channel. Tree shading in the tributaries reduce locally the algal growth (Figure 9), but as a consequence of optimized higher light availability (when more clear tributaries confluence) downstream, the optimized growth rates in all main channels compensate the regional light limiting effects.

Table 9: Key results of the scenario analysis as seasonal mean (Apr-Oct) for station Elbe, Boizenburg near basin outlet for concentrations of chlorophyll a, total phosphorus (TP, mg/l) and total nitrogen (TN). Minima are in shaded fields and maxima are in bolt values.

		Techno World				Consensus World				Fragmented World			
Var	year	RCP4_G	RCP4_I	RCP8_G	RCP8_I	RCP4_G	RCP4_I	RCP8_G	RCP8_I	RCP4_G	RCP4_I	RCP8_G	RCP8_I
chl_a	2025	72.6	74.8	73.5	81.2	69.2	74.3	68.1	75.2	77.9	81.2	76.2	75.9
chl_a	2050	74.0	80.7	64.0	77.4	69.5	75.0	63.8	77.4	78.3	76.8	64.7	81.3
TP	2025	0.088	0.084	0.112	0.086	0.077	0.079	0.077	0.084	0.093	0.096	0.093	**0.118**
TP	2050	0.083	0.089	0.081	0.089	0.075	0.080	0.079	0.084	0.087	0.093	0.085	**0.094**
TN	2025	2.88	2.98	3.08	2.96	2.86	2.84	2.86	2.72	3.27	3.25	3.29	**3.50**
TN	2050	2.85	2.86	2.78	2.77	2.75	2.67	2.45	2.50	3.15	3.14	3.07	**3.37**

5. Discussion

The GLM model derived from empirical nutrient and site data from Middle Elbe is able to predict the resulting chlorophyll a concentration with a high confidence within the vegetation period April to October, although it omit any climate dependent variables such as discharge, temperature or global radiation. The interactions of nutrient response with size of catchment improve the model strongly.

The used chain of models implicated a high uncertainty:

The strategy to combine climate and socio-economic future scenarios in the MARS storylines make it necessary to estimate the specific contribution of climate change and socio-economic changes in our simulated output for nutrients and phytoplankton biomass. Climate scenarios applied in our study will change the discharge not as strong as expected (Krysanova et al. 2008), when simulation results for IPSL and GFDL by model SWIM are used as longterm means of each future period, ranging between 565 – 745 $m^3 s^{-1}$ at basin outlet in the vegetation mean (Table 4). The relatively small range of climate changes has the lowest share on changes in nutrient concentration simulated by model MONERIS (Roers et al. 2016). We expect that simulation monthly changes for each single year will produce a much higher variability since extreme dry and extreme wet summers will be covered (Huang et al. 2013, 2015).

In contrast, another driver turns out be much more relevant in the Middle Elbe: a strong shrinking of human population especially in the rural areas is the officially expected change for the modelled future time frames. It has to be recognized that the change of this driver will almost neutralize the effect of overuse of resources, which is realized in our MARS storyline 3 (“Fragmented world”).

According to the results of our models the physical and nutrient conditions in Middle Elbe enable phytoplankton to build up high biomasses (>20- 150µg/L chla in seasonal mean) in a high share of the modelled spatial units within a wide range of discharge and nutrient conditions, which is in accordance to the observations at monitoring stations. Short time events as floods or nutrient depletion are rapidly compensated in the system by newly emitted inputs, by optimized growth rates when a turbid phase is flushed out of the system (within about 5-10 days) and by strong feedbacks within the food web (grazing). The remobilizing of nutrients from sediments is not fully reflected in the models, but is expected to be important especially in the lake-river chains. The mechanism of channel retentivity proposed by Reynolds and Descy (1996) influences the phytoplankton development in a strong manner when locations of almost no flow are frequent: Simulation is run here with a main velocity and assuming full mixed water bodies, we overlook the recruitment of phytoplankton from less turbulent areas.

Both models suffer from the fact that silica concentrations are not simulated for seasonal changes in future scenarios, but Si-depletion (<0.3 mg l^{-1}) is known to occur occasionally at Elbe stations and in the river Havel, when high biomasses of diatoms develop. In consequence we expect that some maxima values of simulated chlorophyll a concentrations are locally and occasionally overestimated. On the other hand, silicon (Si) is supplied by the regular geological wash-out in a amount to carry a high phytoplankton biomass without limitation, since the observed 3-5 mg/L Si enables to build up at least 300µg/L chla, when assuming Si content of 0.07 mg/mg in dry matter for planctonic diatoms (Quiel et al. 2011).

Implementing a bundle of measures in MARS scenario 2 (Consensus world) and running for future projections in 2025 and 2050 these measures will be able to compensate for climate change effects, but a level for strong phosphorus (P) nutrient limitation will not be achieved except of short summer periods, when simultaneously high phytoplankton biomasses occur. This is in accordance to future simulations by the process-based model QSIM applied to the main channel of Elbe river (Quiel et al. 2011), which predicts no or few P limitation. Nitrogen (N) is always available in surplus. The main pathway for P is point sources, while N is emitted mainly by diffuse pollution.

Therefore, a wider application of balanced fertilization and measures improving the nutrient retention directly near the agriculture fields such as installation of technically improved tidal drain ponds and much more areas with regular flooded wetlands is needed to achieve a reduction of the eutrophication risk in rivers and in the coastal zones.

The future development of the human population in the Middle Elbe region will lead to reduced emission since up to 19% less population are in the official actual prognoses. Only in combination with higher efforts to reduce nutrient emissions, the pressure relaxing by shrinking population will lead to good ecological status.

We identified antagonistic interactions between nutrients and catchment size when chlorophyll a is the response. Still, the full model predicts that smaller basins are less sensitive to nutrients and stronger eutrophication response is expectable in stations with a large catchment. This finding has the implication for water managers that a fail of ecological response observed in small sized rivers at high nutrient concentration is still a management problem, since eutrophication risk strongly increase in downstream water bodies or in the connected coastal zone. Much more effort has to be implemented in the Middle Elbe basin to further reduce the nutrient levels for improving ecological status.

6. Conclusion

The surface waters in the Middle Elbe basin are under high diffuse pollution and hydro-morphological pressure. Supported by specific environmental characters, such as extreme long residence times in reservoirs and river-lake-systems combined with intensive landuse in the riparian zones, the nutrient emissions are transformed effectively to phytoplankton biomass and lower the ecological status to the classes moderate or poor. Climate change will slightly intensify this transformation, if not suitable management options such as further reduction of nutrient surplus and increasing tree shading will be implemented in the whole basin including the smaller tributaries.

7. Acknowledgements

This work is part of the MARS project (Managing Aquatic ecosystems and water Resources under multiple Stress) funded under the 7th EU Framework Programme, Theme 6 (Environment including Climate Change), Contract No.: 603378 (http://www.mars-project.eu). We acknowledge Fred Hattermann and Michael Roers for delivering the modelled discharge data for the chosen socio-climate scenarios and the simulated input-data. Meteorological data were provided by the German Weather Service's Meteorological Observatory called Deutscher Wetterdienst (DWD). We kindly thank the Elbe basin river district (FGG Elbe) and all joining German Federal States for their cooperation and delivery of monitoring data and maps.

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9. Annex - Module description PhytoBasinRisk

Aim of the module

PhytoBasinRisk is a water quality module to simulate the risk of critical phytoplankton biomasses and composition for large river basins. The river phytoplankton is expected to be sensitive to following multiple pressures driven by human activity: a) changes in diffuse pressure (limiting or excess supply of nutrients), b) changes of land use in riparian area (functionally response to tree shading) and c) climate change effects on discharge, global radiation and temperature.

The module take over the localization of the analytic units and their flow net, and the modeled nutrient concentrations of the beforehand processed river basin by the nutrient emission and transformation model MONERIS (Venohr et al. 2011). Time series data for total runoff is essential to run MONERIS and PhytoBasainRisk, and these input data can be taken-over from regional hydrological models (for example see Hattermann et al. 2015).

Platform

The platform is Python 2.7 with packages pandas 0.17.1, numpy 1.10.4, networkx 1.11 and for database access use packages sqlalchemy 1.0.13, psycopg2 2.6.1, pypyodbc 1.3.3, pyodbc 3.0.10. The script loads all input data into a data frame redrawn from the Access data base (PBR_IN), in which the requested data are arranged in tables for a) characters of the analytic areas b) time series for climate and discharge data and c) time series for nutrient data. In PBR_IN also all constant values are defined, which are used for model equations in the Python script.

Input-Data processing for modul PhytoBasinRisk

As perquisites for module run, a) the river basin were spatially divided into analytic units with a flow net equation similar as for MONERIS (Venohr et al. 2011), and b) an up-to-date landuse map with a resolution of at least 10mx10m were analyzed for tree and bush vegetation in the 10m riparian buffer zone and for redrawing river bed characters. For each analytic unit atmospheric conditions (global radiation; temperature), stream bed characters (total length, mean depth under mean averaged flow conditions, river width (without lake sections), total length of connected lakes, number of deep lakes, percentage of connected lake areas to total water surface area), and topographical parameters (slope, altitude), were derived from digital maps and results for the input-table called AU_characters. The total runoff was provided by external model results (model SWIM, Roers et al. 2016).

The mean width was available from the river section polygons of the ATKIS GIS map for rivers wider than 12m (Figure 17). For smaller rivers the width is calculated based on catchment area, specific runoff, and mean slope of the catchment (Venohr et al. 2011). Mean widths of main rivers was 18.2m and 3.3m of tributaries in Middle Elbe basin.

For each analytic unit the dynamic of phytoplankton is calculated separately for two river categories: The first belonging to main rivers (MR; sections crossing the analytic unit) and the second to tributaries (TRIB; within the analytic unit), therefore all input values must be derived separately for these two river categories (Figure 17).

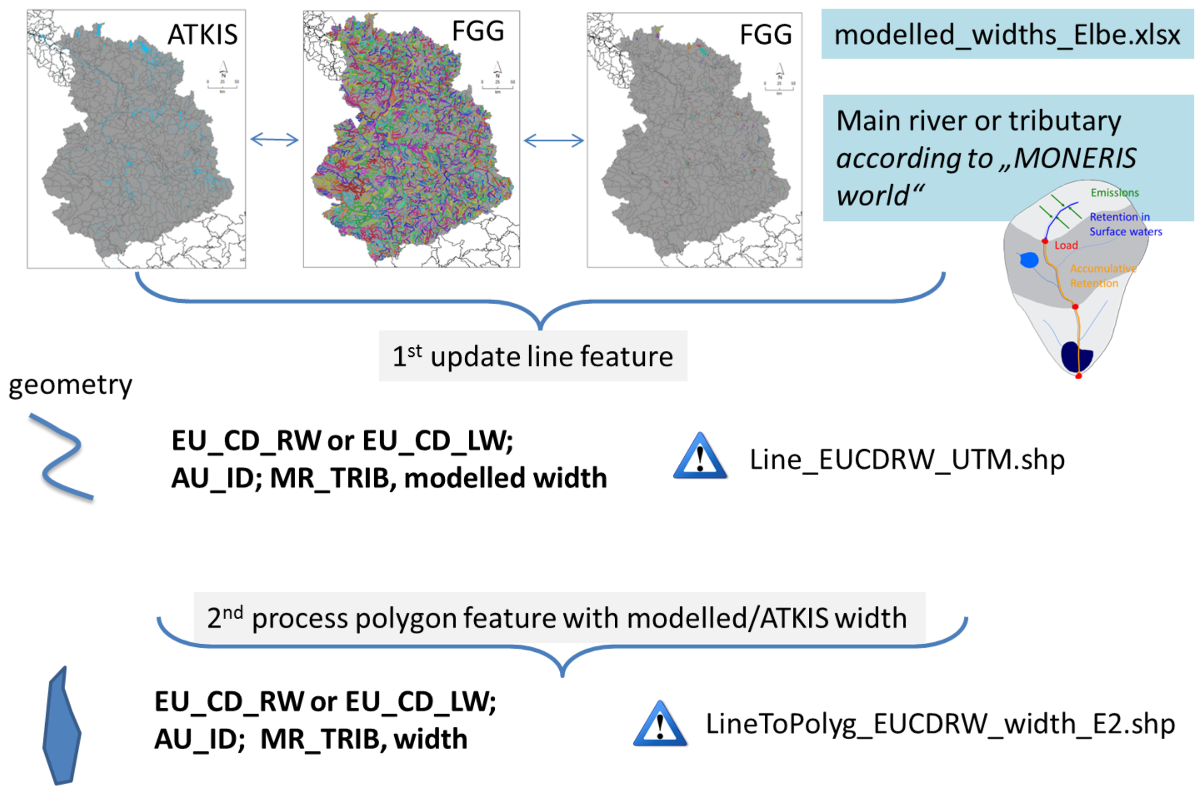

Figure 17: Joined GIS information of landuse in 10m buffer and surface water polygons with the river net linked to water body types (lake or river) used for WDF reporting by basin management (FGG Elbe) and tacked with label main river (MR) or tributary (TRIB) to derive mean river widths.

Data processing for Shading factor

For all rivers the shading factor (shad_fac) is derived based on the relative area covered by trees or bushes (% tree_bush_perc) in the 10-buffer zone, assuming maximum reduction of incoming light by trees with factor 0.8 and less shading by bushes (factor 0.6). The shade potential depends on the river width when full shade potential is expected up to widths of 5-6m (DeWalle, 2010), and no relevant shade effect when wider than 30m. In result, shade potential (p_sh) is calculated for each analytic unit by an exponential equation

Equation 1: p_sh = 1.5749 * e^-0.092 * width

Equation 2: shad_fac = tree_bush_perc * p_sh

Modul setup

The processes of algal growth and algal losses are calculated for each analytic unit and routed through the river net (Figure 18). When several analytic units gave their result into the next upstream unit, than the parameter values are weighted according their discharge contribution to total inflow. All processes can act as long as the calculated water residence time in the analytic unit, but not longer than 1 day, which is the temporal time step.

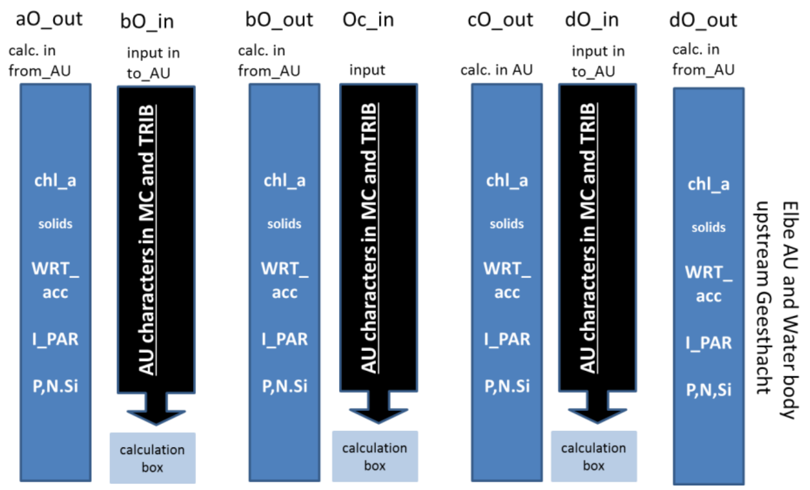

Figure 18: Data flow in module PhytoBasinRisk

Growth and limitation functions

Equation 3: Final biomass: growth + chla_in - losses

Maximum specific growth rates under ideal culture conditions are used following the strategy of Reynolds & Irish (1997) for model ProTech (Elliot et al. 2000, 2006). Maximum growth (chla_opt) might be limited by light, nutrients and residence time in a specific water body and be reduces by losses.

Analogous to the application of model Q-SIM (Quiel et al. 2011) any limitation is expressed as a factor (0-1) to maximal growth:

Equation 4: chla_netto = Chla_opt * fT * fN * fL

Light availability

Income global radiation is reduced by shading due to bank vegetation, reflection at water surface, extinct by various water components and lowered on average by the depth of the mixed water body, while rivers are assumed to be fully mixed (1D model). The calculation and constant values follows the Q-SIM approach of Hardenbicker et al. (2014), but river depth is dynamically modelled and riparian shading is added as an important factor for rivers smaller than 30m in width.

An estimation of water level according river depths for MR and TRIB was performed because this is a high-influence input-parameter not only to light but also for estimating growth (Köhler etal. 2002) and losses (e.g. mussels). Water levels at gauge stations were not available in sufficient spatial resolution and its extrapolation to up- and downstream stretches is limited because of artificial river bed cross-sections at gauges. The estimation of river depths in 722 analytic unitswas performed here in 3 steps: 1) defining a static depth (H_stat) under mean flow conditions by a depth-width- functions available for selected river stretches (IKSE, 2005) and applied on all analytic units, 2) manual corrections for registered river deepening and lateral banking (e.g. river Spree in the city Berlin) and 3) a dynamic fitting of water level depth based on the ratio of actual discharge to mean longterm discharge and established flow-depth curves documented for main river Elbe (IKSE 2005) and some of the tributaries.

Shading by bank vegetation was estimated by a transfer function between percentage of tree cover in the 10m buffer zone of the river sections and the river widths (see previous chapter). The shading factor range between 0.2 – 1, optimal tree shading or no shading respectively.

Limiting Nutrients

Nutrient limitation is defined by the most limiting nutrient. For each of the taxon which is modelled the half-saturation constants (k_P, k_N. k_Si) of each of the nutrients phosphorus, nitrogen and silica is needed (Quiel et al. 2011), which is compared to the incoming nutrients (conc_in) and transferred to a limiting factor operating between 0-1 by the equation (Quiel et al. 2011): P,N, Si_lim = conc_in / (k_ + conc_in).

Residence time

The residence time (WRT) within the analytic unit was estimated by their values for rivers (WRT_riv) and connected lakes (WRT_lak).

The modul River Basin Risk calculate algal growth for each day, so WRT longer than 1 day is set to 1, except of for identifying lake- dominated analytic unitsfor loss processes (sedimentation, zooplankton grazing).

WRT_riv was estimated based on total river lengths and flow velocities under mean flow conditions (velo_const) and resulting WRT modified according actual flow. Flow velocities observed for smaller rivers (BLfW 1995), for main river Elbe and some of the tributaries (IKSE 2005; Köhler et al 2002) were used to define velocity classes. These classes were linked to all analytic units by taking into account their landscape slope and river widths. Observed velocity-discharge relationships for mean, high and low flow conditions are different for MR and TRIB and were used to fit WRT_riv using with the ratio of actual discharge to mean longterm discharge (Q_act/Q_LT) as the variable:

Equation 5: MR_WRT_riv = riv_lenght * velo_const * e ^((Q_act/Q_LT) * -0,148)

Equation 6: TRIB_WRT_riv = riv_lenght * velo_const * e ^((Q_act/Q_LT) * -0,582)

The residence time of lakes connected to rivers (WRT_lak) was not available, but total length of lake water bodies and number of deep lakes was counted for each analytic unit. It was assumed that each km lake lengths increase WRT by factor 1.1 and by each count of deep lakes by 30 days.

Table 10: Classes of mean flow velocities under mean flow conditions

category	velo_const	unit	category	velo_const	unit
MR	0,2	m/s	TRIB	0,03	m/s
MR	0,5	m/s	TRIB	0,08	m/s
MR	0,75	m/s	TRIB	0,3	m/s
MR	1	m/s	TRIB	0,8	m/s
MR	1,2	m/s	TRIB	1,2	m/s
			TRIB	1,4	m/s
			TRIB	2,5	m/s

Losses

For each day the losses are calculated according following equation:

Equation 7: out_chla = netto_chla - chla_sed - chla_graz - chla_mort

The sedimentation and all other loss processes can act only as a share of daytime when water residence time is smaller than 1, but not longer than 1 day.

Following assumptions were made to derive the losses:

Sedimentation - when lake area more than half of all surface waters in analytic unit, than lake adopted phytoplankton (high buoyancy) have reduced sedimentation loss (max 20%), but when smaller portion of lake areas (<50%) high sedimentation loss with 10% per km of lake length.

The sedimentation of Bacillariophyceae (diatoms) were set as the strongest sedimentation loss in lakes (most heavy).

Grazing loss by zooplankton is set to act with 10% of produced chlorophyll if WRT >3 days (ciliats, rotifers) to simulate grazing in standing waters. Also in running waters rotifers can be important (Holst 2006). The potential abundance of rotifers is calculated in dependency of actual chlorophyll a and water temperature and for river sections wider than 35m. Grazing is calculated by individual ingesting rates 2,9µgC/d (Quiel et al. 2011) which corresponds to 0,0138µg/L chl a/Ind.

Grazing loss by mussels and macrozoobenthos are calculated in dependency to river depth, with maximal of 5% of produced chlorophyll a, when shallower than 1m for all river sections.

Mortality is low for algae and set as 0.1% (Quiel et al. 2011).

Performance discussion and sensitivity tests of module PhytoBasinRisk

The module training data set and the validation is described in the main report.

Riparian shading is one of the management options which should be reflected by the module PhytoBasinRisk, since this model simulated the phytoplankton development also in its small tributaries. The river network of Middle Elbe has a high potential for implementing riparian shading, because on average of total length, tributaries are small (3.3m) and even more than the half of main rivers would be influenced by tree shading (18.2m). Under conditions of maximal tree cover in the 10m riparian buffer in most tributaries incoming light is reduced to about 20% of total, and lowest limit for reduction is set for main river with 30m- width (mainly regulated shipping courses) with 8% reduction. This is much less light reduction as assumed for Ouse river with 39% (Hutchins et al. 2010), because we estimated less tree height (15m) and less canopy is allowed to hang over the water surface.

Satellite imagery (Google map) was used to compare with the ATKIS GIS map for land use data in 10m buffer. Summer snap shots of riparian areas covered with vegetation were of similar extents as those derived by GIS maps. Our estimate suggests that the combined areas covered by trees or by bushes currently contribute to approximately 25% of total capacity, which could be shaded along the total length of river sections with river width less than 30m.

A synthetic data set was produced to test sensitivity of the PhytoBasinRisk module for single variables. The input data of the longterm mean run for period 2005-2010 was altered with reducing P by 30%, increase water temperature with 2°C (pl2Temp) and run shading effect as assumed for storyline 2 (SL2_shade) for each of the spatial analytic units. The effect on simulated chlorophyll a is shown in the main report for a smaller catchment, in which shading effect can be expected because of smaller rivers (Figure 9).

For stations along main channel of river Elbe simulated chlorophyll a concentrations are continuously influenced by water temperature (Figure 19). In contrast, when doubling the discharge at all stations, an opposite effect is simulated for some of the subsequence stations: higher biomass in river sections of km 280 -416 (km counted with 0 at CZ/DE border), is simulated even when the shorter residence time is taken into account. The module simulate less self-shading and less grazing losses in the sections before so optimal growth rates can compensate for the wash out effect.

For phytoplankton in large rivers grazing losses by mussels can be very high (Schöl et al. 2002, Lindim 2015), but it is complicated to predict at which locations they occur and in which river depths their act the most. While Limdim (2015) simulated high grazing effect in tributary Havel, the sandy and moving river bed of Elbe river od allow mussel development only in few locations. Instead the rotifers are the most important zooplankton component and are widely distributed in all lowland sections of Elbe river system, when food carrying capacity is provided by high chlorophyll a concentrations.

Still, high discharge is an important negative influence factor for single month response, which can be demonstrated when comparing the simulated chlorophyll a for the most wet and most dry climate scenario (Figure 20; Table 4). In these scenarios the simulated discharge for the months April and May were most different (887 to 1372 m^3 s^{-1}; 652 to 957 m^3 s^{-1}) and so is the monthly response for chlorophyll a (for seasonal means see Table 9).

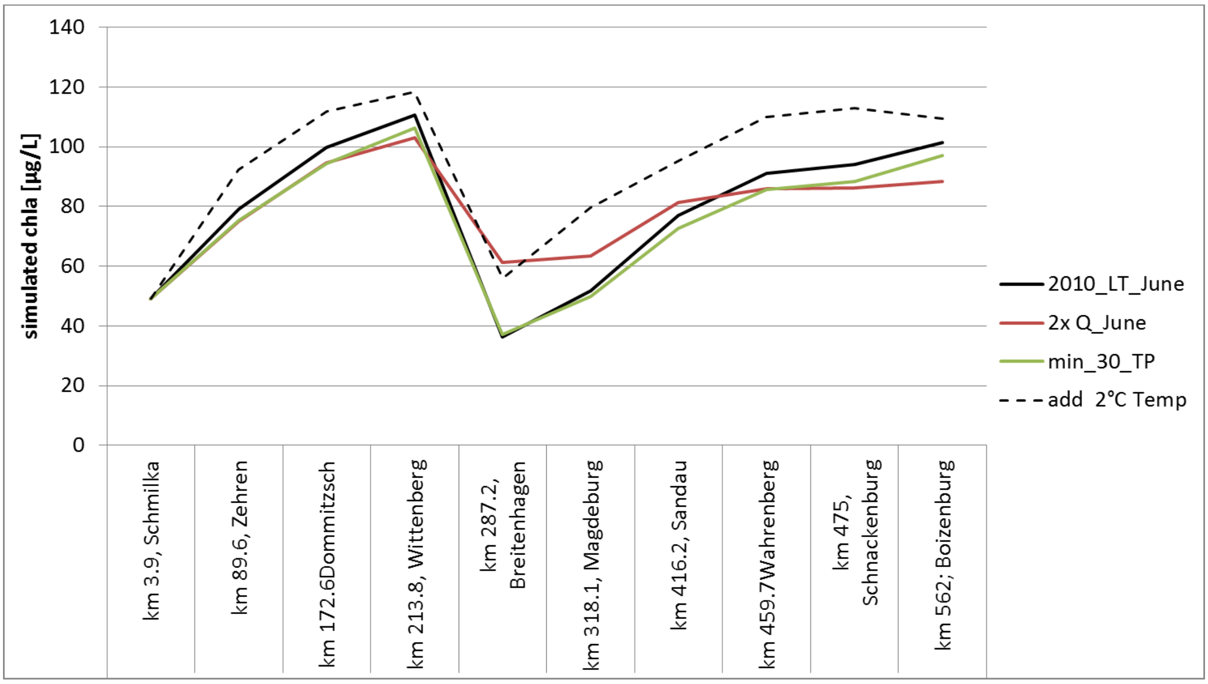

Figure 19: Effect of input variables synthetically changed to demonstrate single effect on simulated chlorophyll a at stations along the river Elbe for month June: longterm conditions in period 2006-2010 (2010 LT_June), discharged doubled for June (2x Q_June), -30% reduction of TP concentration (min_30_TP) and addition of 2°C to water temperature.

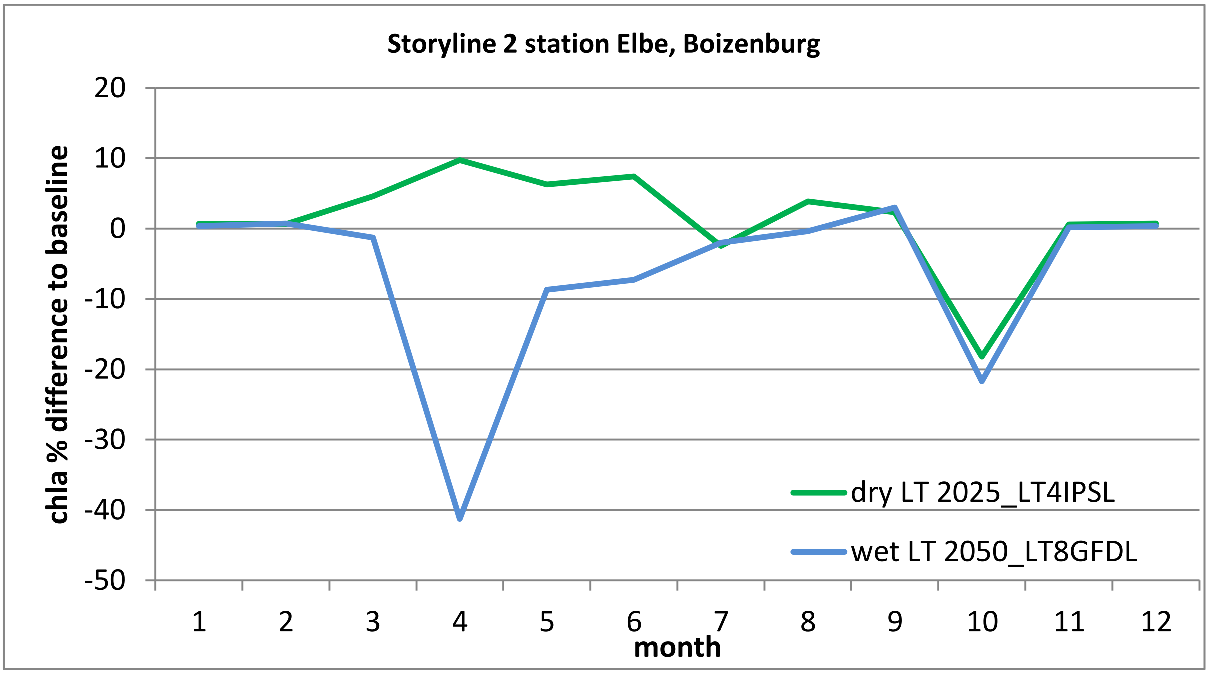

Figure 20: Difference of simulated chlorophyll a (storyline 2) to baseline simulation in the most dry and the wettest vegetation mean of discharge of all scenario runs (see Table 4).

Outlook

The module output is less sensitive to changes in discharge than expected, so the assumed compensatory processes (regulations by optimized growth and reduced losses) must be analyzed in more detail with simulated and empirical data.

Improvements of the PhytoBasinRisk model can be implemented in future updates, because the model set-up allows for following extensions:

- Refinement of time-step resolution: Modelling with daily input data
- Refinement of WRT with spatially individual discharge-flow time curves redrawn from gauge data analysis
- Refinement of specific taxa response: Include functional algal traits or species instead of the four algal classes
- Refinement of reflecting time-lags in response to climate variables: Direct the result of the spatial analytic unit to the downstream unit with a delay according the calculated water residence time, when time is more than one day.
- Improve the lake part of the model by using WFD monitoring data for 283 lakes with surface area larger than 0.5km^2 to train the specific lake models such as SALMO-1D (Simulation by an Analytical Lake Model, Benndorf und Recknagel 1982) or Protech (Elliot et al. 2000, 2006) for German WFD lake types.
- Loss factors can be refined by better field data for local depth variations (sedimentation) and mussel populations

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