Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

MARS PROJECT

Managing Aquatic ecosystems and water Resources under multiple Stress

Deliverable 5.1: Five Reports on stressor classification and effects at the European scale

Lead beneficiary: National Technical University of Athens (NTUA)
Contributors: Multiple partners, see inside for details

Due date of deliverable: **Month 36**
Actual submission date: **Month 36**

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)	

Content

Overview

The Deliverable 5.1 is composed of 5 reports, the first report D5.1-1 contains 3 parts.

Summary D5.1-1 Part 1: Multi-stressors on surface water and effects on ecological status

Humans have increased the discharge of pollution, altered water flow regime and modified the morphology of rivers. All these actions have resulted in multiple pressures on freshwater ecosystems, undermining their biodiversity and ecological functioning. The European Union has adopted an ambitious water policy to reduce pressures and achieve a good ecological status for all water bodies. However, assessing multiple pressures on aquatic ecosystems and understanding their combined impact on the ecological status is challenging, especially at the large scale, though crucial to the planning of effective policies. Here, for the first time, we quantify multiple human pressures and their relationship with the ecological status for all European rivers. We considered ecological data collected across Europe and pressures assessed by pan-European models, including pollution, hydrological and hydromorphological alterations. We estimated that in one third of EU's territory rivers are in good ecological status. We found that better ecological status is associated with the presence of natural areas in floodplains, while urbanisation and nutrient pollution are important predictors of ecological degradation. We explored scenarios of improvement of rivers ecological status for Europe. Our results strengthen the need to halt urban land take, curb nitrogen pollution and maintain and restore nature along rivers.

Summary D5.1-1 Part 2: Analysis of pressure - response relations: classification of multiple pressures on broad river types

For this deliverable a unique and comprehensive collation of input data were derived. Information from different data sources, in varying formats, spatial resolution, comprising information on hydrology, physico-chemical water quality, geo-morphological characteristics, ecological status and other, were harmonized and merged to an extended database. The data were derived for about 100,000 sub-catchments (FECs) covering Europe, EFTA states and further, hydrologically connected areas to the east.

From this database pressure indicators were deduced and statistically compared to the ecological status reported by the EU-countries. An important and novel indicator is the impact of hydrological alteration on major flow characteristic like base flow, floods or duration of low flows. These were derived by comparing modelled flows for current conditions and for semi-natural conditions. The goal was identifying the most explanatory pressure indicators impeding a good ecological status.

First, the general statistics on the distribution of all pressure indicators were conducted, secondly, the pressure indicators were compared to the ecological status as assessed in 1st River Basin Management Plan (RBMP).

Importance of pressures for supporting good ecological state varies a lot among river types and regions in Europe. On large rivers, chemical stressors, percentage of broad leaved forest and share of agricultural land in floodplain are three most important pressures. On lowland, medium to large rivers, high flow hydrological characteristics become very important also. Share of coniferous forest in floodplain is important pressure in mid altitude rivers, whereas base flow and oxygen demanding substances are important for highland rivers.

Our results also suggest, that diffuse pollution of nutrients and decrease of riparian vegetation at present do not support good ecological status mainly in the Mediterranean and Atlantic regions. In the Central and Baltic region, the most important cause for a deterioration of ecological status is the combination of diffuse pollution of nutrients and hydrological alterations. In the Eastern Continental region all three types of pressures, namely, hydrological, morphological and chemical are equally important.

Classification of multiple pressures on European broad rive types presented here is closely related to JRC work (Grizzetti et al., 2017) and NTUA work (MARS, 2017) in the same work package of the MARS project. JRC has unveiled patterns between human pressures and ecological status of European rivers in non - stratified manner. NTUA has analysed relation of low flows and ecological flows (E-flows) to ecological status and contributed data for hydrological pressures. This contribution is very important, since in our study we indeed show that hydrological pressures are very important, and in some regions and river types even prevail over morphological pressures.

Our results serve as an input to scenario analysis tool at the European scale (namely work package 7.4) and will be expanded with additional data and expert knowledge.

Summary D5.1-1 Part 3: Multiple stressors and groundwater status analysis and statistical modelling at the European scale

The aim of the work is to analyse groundwater status and stressors (pressures) relevant for groundwater using available data at European scale reported by European countries (WISE-WFD and WISE-SoE datasets managed by the EEA). In particular, a definition of spatial extent of ground waters in poor status, acting single stressors (pollution, abstraction, saltwater intrusion) and stressor combinations including an identification of prevailing pollutants causing failure of good groundwater status. The aim of the statistical analysis is to use simple statistical models to investigate the large-scale pressures on the chemical status and quantitative status of groundwater reported by European Union Member states. In particular, to see if it is possible to use these models to investigate and understand any interactions between different pressures on groundwater status.

The analysis of stressors and status shows that prevailing stressor causing failure of good groundwater status is pollution, followed by groundwater abstraction. Pollution in combination with groundwater abstraction appears to be most common stressor combination in Europe. Salt water intrusion is almost always associated with groundwater abstraction or/and pollution, but it does not take place in all coastal areas. The most common type of groundwater pollutants are agrochemicals (nutrients and pesticides) affecting whole Europe and especially agricultural areas. When assessing pesticide pollution at European scale, one must take into account a bias induced by various monitoring strategies used by countries, there is lack of comparable data on pesticide metabolites that may occur more frequently and in higher concentrations than parent pesticides. EU WFD common implementation strategy does not assure sufficient harmonization of monitoring strategies among EU member states preventing comparable pan-European assessments.

The study demonstrated how 'data-led' methods, such as stepwise regression, can be used to suggest and estimate models of groundwater status. However, we note that they should be used with caution as such approaches can include spurious relationships which result from not accounting for multiple hypothesis tests. Only limited interactions have been investigated to date, however, there is some evidence for a synergistic interaction between arable farming and winter precipitation (when the regression does not include country as a random effect) on the chemical status of groundwater. There is, however, less confidence in the results of models of groundwater quantitative status which appears, as may be expected, to be largely driven by weather variables.

Summary D5.1-2: Relation of low flows, E-flows, and Ecological Status

The present report 'Relation of low flows, E-flows, and Ecological Status' presents a European scale analysis of hydrologic data at the resolution of the Functional Elementary Catchment (FEC). Simulated daily time-series of river flows from the PCR-GLOBWB global model were used based on a hypothetic near-natural scenario where water abstractions from water bodies do not exist and an anthropogenic scenario with water abstractions occurring.The latter practically represents the reality. Many hydrologic indicators expressing the characteristics of the rivers' hydrologic regime were calculated for all FECs with the Indicators of Hydrologic Alteration (IHA) methodology and software package and the deviations of the indicators' values between the two scenarios were used as proxy metrics of hydrologic alteration or hydrologic stress of rivers. Regressions between indicators with the rather limited dataset of EQR values of two BQEs (macroinvertebrates and phytobenthos) showed insignificant or very weak relationships when processed with the entire dataset for Europe or separately for each of the 20 Broad River

Types (BRTs). However, by conducting two examples at smaller scales (catchment or region) with better ecological response datasets clearer relationships were found.

Hydrologic alteration metrics were averaged per BRT without reference to any ecological response not showing remarkable hydrologic stress in certain BRTs or considerable differences in the degree of alteration among the various BRTs. Clearer results could be indicated by mapping the hydrologic alteration on Europe's geographical background. The mapped indicators, especially some of those connected with low flow conditions were the most informative showing that Southern Europe is more hydrologically stressed due to groundwater abstractions for irrigation. In the rest of Europe hydrologic conditions change less frequently within a single year and a multi-year period.

The determination of a minimum ecological flow connected with good ecological status needs further research with updated datasets, but the water community can already take advantage of the results produced herein to obtain a view of hydrologic stress in Europe, identify significant hydrologic stress on a local basis and try to interpret the impacts of this stress on river's ecology with the use of appropriate response data.

Summary D5.1-3: Impact of multi-stressors on ecosystem services and their monetary value

Which are the ecosystem services (i.e. the contribution of nature to human well-being) provided by European rivers, lakes, and coastal waters? Can we map and quantify them? Do enhanced ecosystem conditions and biodiversity support higher benefits for people? These are the questions addressed in this research.

We quantify the main ecosystem services provided by aquatic ecosystems at the European scale, including fish provisioning, water provisioning, water purification, erosion prevention, flood protection, coastal protection, and recreation. These services are provided by aquatic ecosystems, such as lakes, rivers, groundwater and coastal waters. We show European maps of ecosystem service capacity, flow (actual use), sustainability or efficiency and, when possible, benefit.

Our results indicate that the ecosystem services are mostly positively correlated with the ecological status of European water bodies (that is a measure of the ecosystem integrity and biodiversity), except for water provisioning, which strongly depends on the climatic and hydrographic characteristics of river basins. We also highlight how provisioning services can act as pressures on the aquatic ecosystems. Based on the relationship between ecosystem status and delivery of services, we explore qualitatively the expected changes of ecosystem services under scenarios of increase in different pressures.

Finally, we perform an economic valuation of the ecosystem services provided by European lakes, considering the current conditions and scenarios of improvement of the ecological status. Using a benefit transfer approach, we estimate that the average economic value of ecosystem

services delivered by a European lake is 2.92 million EUR per year. We also demonstrated that the ecological status of lake has an impact on valuation. The expected benefit from restoring all European lakes into at least a moderate ecological status is estimated to be 5.9 billion EUR per year, which corresponds to 11.7 EUR per person and per year.

Quantifying and valuing ecosystem services helps to recognise all the benefits that humans receive from nature, offering stronger arguments to protect and restore ecosystems and thus fostering the implementation of the European water policy. This study offers scientific evidence to this aim.

Summary D5.1-4: Effects of multiple stressors on ecosystem structure and services of phytoplankton and macrophytes in European lakes

The aim of this deliverable was to assess the impacts of multiple stressors on lake ecosystems at the European scale. We have examined ecological responses of two main biological groups (quality elements), namely algae (phytoplankton) and other aquatic plants (macrophytes), to a range of stressor combinations in large populations of lakes. Moreover, the impacts of future multiple stressor scenarios - future climate and nutrient concentrations - have been assessed for a phytoplankton community index.

While nutrients are a key stressor in all regions of Europe, MARS also focuses on the following key environmental changes for specific regions: water scarcity and flow alterations (Southern Europe); changes in hydrology and morphology (Central Europe); and changes in hydrology and temperature (Northern Europe). More specifically, the stressors that have been investigated in this report are related to increased air temperature and precipitation, hydropower and water abstraction for irrigation and public water supply, hydrological changes (flushing or water level changes), salinisation, or increase in humic substances (“brownification”). We have analysed effects on ecological status (ecological quality ratio values), and in addition a set of indicators of environmental stressors for both biological quality elements. For phytoplankton, the main indicators analysed were chl-a, abundance of cyanobacteria (a group of potentially harmful algae) and PTI (phytoplankton trophic index). For macrophytes, the main indicators were the water-drawdown index (WIc) for regulated lakes, a proportion of macrophyte coverage (%PVI), and other indices based on specific species or species groups. Interactions within the lake community, including also zooplankton (small crustaceans), were addressed by analysis of data from mesocosms across Europe. Potential effects of multiple stressors on ecosystem services (e.g. nutrient retention, nutritional value of fish, and cultural services to lake visitors) have also been investigated by case studies and national datasets. The main large-scale data sources used in our studies include the European Environment Agency's WISE-SoE datasets (Waterbase), data compiled during previous EU projects (WISER), and national monitoring data. Moreover, information on lake and catchment characteristics (such as land use) was obtained from the MARS geodatabase. The natural characteristics of lakes (such altitude, surface area, mean depth, alkalinity and humic level) were explicitly considered in most of the studies, either as co-variables or as determinands of lake types.

The analysis of EEA's water quality data in combination with land use data showed that, not surprisingly, total phosphorus (P) concentration in lakes clearly increased and Secchi depth (transparency) generally decreased with increasing proportion of arable and pasture lands in lake catchments across Europe. Total P was the stressor that correlated best with ecological status of phytoplankton, while Secchi depth better explained the ecological status of macrophytes. Climatic variables such as air temperature and precipitation, in contrast, had apparently no effect on the ecological status. This result does not contradict that climate change may cause additional stress for lake ecosystem. Instead, the space-for-time approach (using geographic variation in climate as a substitute for temporal variation) in these analyses may not be the most appropriate for detecting real effects of climate change. For the individual phytoplankton indicators (cyanobacteria and PTI), interactions between effects of nutrients and climatic stressors (temperature and/or precipitations) were found for some of the lakes or lake types. For example, the analysis of time series indicated that cyanobacteria are most favoured by nutrient stress in lakes of low nutrient status and sensitive to summer rainfall in short residence time lakes. However, the studies also revealed large variation in the combined stressor effects among the different lakes types. It was therefore difficult to generalise such results across lake types. For the PTI index (Northern Europe), the strongest interaction between nutrients and temperature stress was found for lowland siliceous lakes. We used this empirical relationship to predict the future PTI scores for this lake type under the MARS future climate scenarios. According to our model, increased temperature and precipitation will result in higher PTI scores, indicating impaired ecological status. In the short term (2030), however, climate-induced changes in PTI will probably not be sufficient to change the ecological status class of lakes (e.g., from Good to Moderate).

The analysis of Mediterranean (Turkish) lakes suggest that warming together with expected changes in land use in this regions may result in higher salinisation and eutrophication with more frequent cyanobacteria blooms and loss of biodiversity. Consequently, under such conditions, the ecosystem services potential (e.g. drinking and irrigation water, biodiversity etc.) are likely to be deteriorated if not lost completely. To counteract, stricter control of nutrients emissions and human use of water is urgently needed.

The interactions between nutrients and climatic stressors could most clearly be interpreted from the experimental data based on former mesocosm experiments. For example, these experimental results indicate that global climate warming might favour growth of macrophytes at moderate water level decrease southern regions, even under relatively eutrophic conditions. However, if the water level decrease becomes so extreme that macrophytes are directly negatively affected, and longer and intense drought periods become more common, the combined effects of eutrophication and extreme water level reductions may adversely affect the development of macrophytes. In contrast, warmer temperatures in northern regions may hamper macrophyte growth due to increased precipitation and, thus increased water levels and nutrient loading.

The MARS project have resulted in much new information on the combined effect of eutrophication and climate change and their interactions on trophic structure and dynamics - showing that combined effects through a series of cascading events can lead to deterioration in water quality and ecological status - there are still some knowledge gaps to be filled. Knowledge on differences in interactions along altitude, latitude and other biogeographical gradients are needed before firm and safe conclusions relevant for managers and WFD can be drawn. We also need more knowledge on the resilience of lake community structure and dynamics to extreme climatic events such as heat waves, drought, and heavy rainfall, since we can expect an increase of such events.

Summary D5.1-5: New functional diversity indices allowing assessing vulnerability in abiotic multi-stressor context

A community hosted by an ecosystem composed of species sharing the same characteristics i.e. species showing the same response to the environment and/or species with the same impact on their environment, can be defined as a community with high functional redundancy. Such community is supposed to be less vulnerable to species loss and the ecosystem functioning is also supposed to be less impacted than when communities are composed of species with different functional characteristics.

In this work, we first described the fish communities of lakes, rivers and estuaries of France, Spain and Portugal using species richness and functional diversity. Functional diversity was a measure of the extent of complementary among species considering five characteristics previously define by different sources (literature, available database): fish size, vertical position in the water body, spawning habitat, trophic group, and swimming mode. For the three aquatic systems, the number of species and functional diversity was generally higher in northern and western France than in the Mediterranean areas; this geographical pattern was explained by historical events (recolonization after the last glacial period). Higher functional diversity found in estuaries compared to lakes and rivers was explained by the importance of the connectivity between adjacent environments.

Analysing correlations between functional redundancy and species richness, results suggest that higher taxonomic richness in freshwater ecosystems is likely to increase the stability and resilience of fish assemblages after environmental disturbance because of higher species redundancy whereas it is not the case in estuaries.

Studying the impact of species loss following different scenarios, we also demonstrated that, in rivers and estuaries, rare species support singular ecological functions not shared by dominant species. Our results suggest also that functional diversity of fish assemblages in rivers can be more affected by environmental disturbances than in lakes and estuaries.

Finally, using functional redundancy and taxonomic vulnerability, we proposed a composite index of functional vulnerability, minimised for highly redundant assemblages composed of species with low extinction risk. Fish communities of estuarine ecosystems appear less vulnerable to species loss in comparison with assemblages of lakes and rivers. Although these latter systems obtained comparable scores, the functional vulnerability was not influenced by the same component. Fish assemblages in lakes are often redundant but composed of a large part of vulnerable species, whereas river assemblages are in general poorly redundant but composed of species with low intrinsic vulnerability. This new score is proposed to be used in conservation perspective to define management priorities.

Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

Deliverable 5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis.

D5.1-1 Part 1: Multi-stressors on surface water and effects on ecological status[1]

Lead beneficiary: **Joint Research Centre (JRC)**
Contributors: **Bruna Grizzetti, Alberto Pistocchi, Camino Liquete, Angel Udias, Faycal Bouraoui, Wouter van de Bund**

Due date of deliverable: **Month 36**
Actual submission date: **Month 36**

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)	

[1] This work is submitted to a peer review journal for publication: Grizzetti, B., Pistocchi, A., Liquete, C., Udias, A., Bouraoui, F., van de Bund, W., Human pressures and ecological status of European rivers. Scientific Reports. Submitted.

Content

Non-technical summary

Humans have increased the discharge of pollution, altered water flow regime and modified the morphology of rivers. All these actions have resulted in multiple pressures on freshwater ecosystems, undermining their biodiversity and ecological functioning. The European Union has adopted an ambitious water policy to reduce pressures and achieve a good ecological status for all water bodies. However, assessing multiple pressures on aquatic ecosystems and understanding their combined impact on the ecological status is challenging, especially at the large scale, though crucial to the planning of effective policies. Here, for the first time, we quantify multiple human pressures and their relationship with the ecological status for all European rivers. We considered ecological data collected across Europe and pressures assessed by pan-European models, including pollution, hydrological and hydromorphological alterations. We estimated that in one third of EU's territory rivers are in good ecological status. We found that better ecological status is associated with the presence of natural areas in floodplains, while urbanisation and nutrient pollution are important predictors of ecological degradation. We explored scenarios of improvement of rivers ecological status for Europe. Our results strengthen the need to halt urban land take, curb nitrogen pollution and maintain and restore nature along rivers.

Introduction

In the second half of the 20th century economic activities flourished in Europe while the status of rivers, lakes and coastal waters chronically deteriorated (Meybeck et al. 2003). Human activities have produced multiple pressures on waters, including nutrient pollution (Sutton et al. 2011; Fowler et al. 2013), modifications of river morphology (Belletti et al. 2015; Sweeney et al. 2004), alterations of water flow regime (Acreman and Dunbar 2004; Poff and Zimmerman 2010) and the introduction of alien species (Strayer 2010). Multiple pressures from land-based activities pose threats to human water security and freshwater biodiversity (Vörösmarty et al. 2010), and have produced cumulative effects in oceans and coastal waters (Halpern et al. 2015).

Natural spatio-temporal heterogeneity in rivers and floodplains is essential to support ecosystem biodiversity (Ward et al. 1999). However river regulation, such as flow alterations, channelization, dredging and river bank stabilization, have reduced the connectivity in the riverine landscape and altered the fluvial dynamics that support habitat heterogeneity (Ward et al. 1999). Similarly, the widespread construction of dams has diminished the natural disturbance patterns in rivers, homogenizing flow regional differences and creating cumulative transboundary effects (Poff et al. 2007; Ziv et al. 2012). Freshwater biodiversity is further threatened by water pollution related to human activities in the catchment, fish overexploitation and the increase in the number of alien species (Butchart et al. 2013). All these actions have resulted in multiple pressures on freshwater ecosystems that undermine their biodiversity and ecological functioning.

Disentangling and quantifying the cause and effect relationship between multiple pressures and ecological functioning is challenging, especially when addressing large geographical areas like Europe. Firstly, the quantification of pressures on water systems is hampered by limited and spatially heterogeneous data. Secondly, multiple pressures are acting concurrently on water bodies and their combined effect is poorly understood (Nõges et al. 2016). Thirdly, ecological conditions are the result of impacts building up over time, local natural conditions and climatic variability (Nõges et al. 2007; Brucet et al. 2013). Finally, ecological systems could change following non-linear patterns and regime shifts, and restoration measures do not necessarily return the ecological systems to their original state (Scheffer et al. 2001). All these aspects contribute to a great complexity in the link between multiple pressures and ecological status in water bodies. Yet understanding this relationship is necessary to plan effective policies (Hering et al. 2015; Navarro-Ortega et al. 2015) and restoration measures (Teichert et al. 2016), as long-term availability of water resources and many benefits for people depend on healthy aquatic ecosystems (MEA 2005; Guerry et al. 2015).

To protect and enhance water resources and aquatic ecosystems, since 2000 the European Union has adopted an ambitious water policy, the Water Framework Directive (WFD, European Parliament and Council 2000), with the objective of reducing pressures and achieving good ecological status for all European water bodies. With this aim, EU Member States had to assess the ecological status of rivers, lakes and coastal waters in their territory, and established programmes of measures to reduce significant anthropogenic pressures affecting the status.

Here, for the first time, we have characterised the main pressures acting on European rivers and explored their relationship with the ecological status reported by EU Member States. Our analysis addressed three main questions: 1. How do multiple pressures affect the ecological status of European rivers? 2. To what extent has the EU water policy target of good ecological status been achieved? and 3. How and where would measures to improve the ecological status of rivers be effective?

Results

How do multiple pressures affect the ecological status of European rivers?

To address this first question we quantified multiple pressures on European rivers and examined their relationship with reported data on the ecological status.

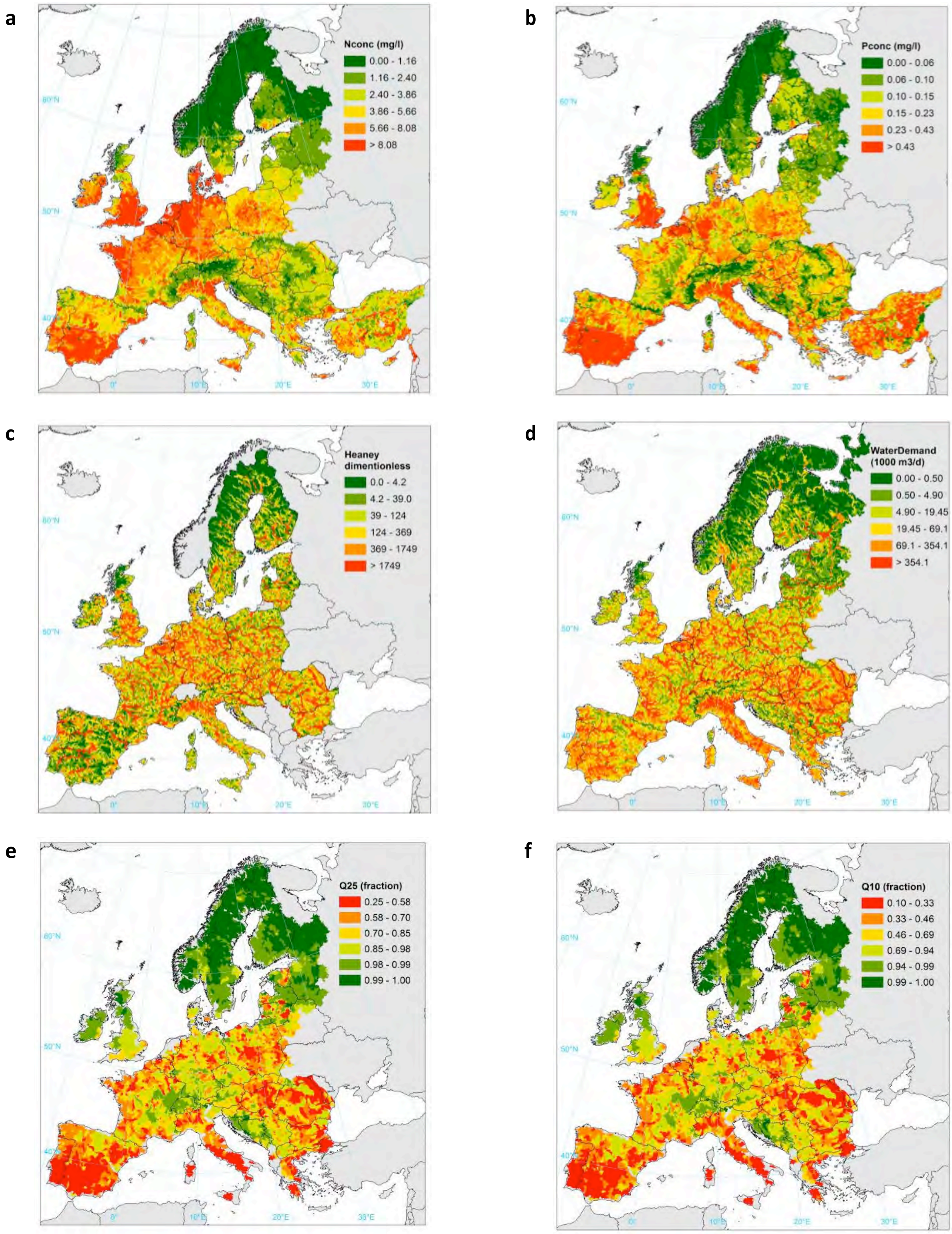
According to a recent European Commission report (European Commission, 2015a), the major pressures acting on European rivers are related to pollution, hydrological changes and hydromorphological alterations. We considered 12 indicators that could inform on these pressures (Table 1.1): nitrogen and phosphorus concentration; pollution from urban runoff; water demand; alteration of natural low flow regimes (at 10th and 25th percentiles); density of infrastructure in floodplains; natural areas in floodplains; artificial and agricultural land cover in floodplains; and artificial and agricultural land cover in the drained area. We quantified these indicators at the spatial resolution of catchments (180 km^2 on average), using pan-European models and data sets (we used best available data for the period 2004-2009, see ‘Methods’). The maps of pressures on European inland waters are shown in Figure 1.1.

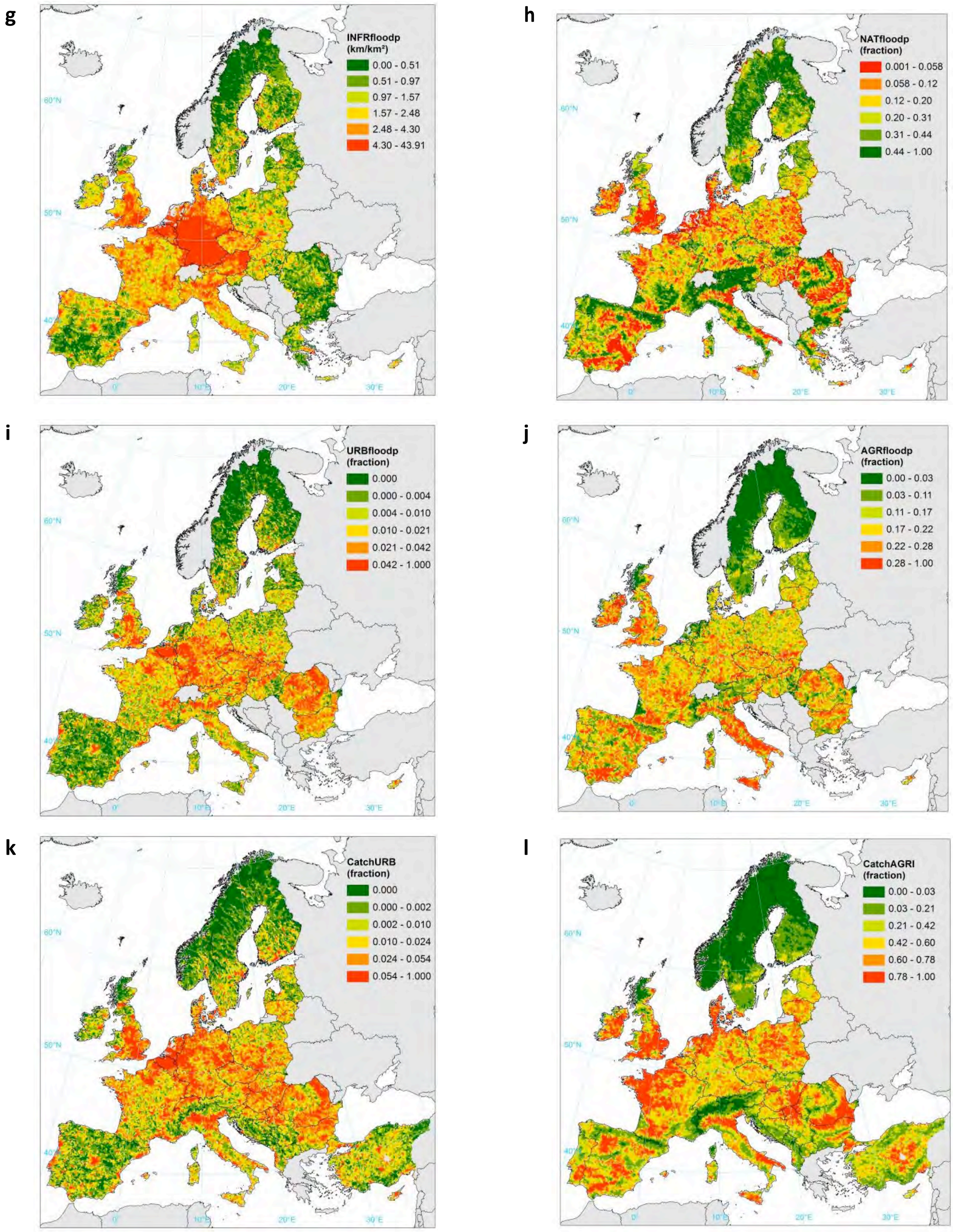
Table 1.1. Pressures considered in the study and the respective indicators.

Pressure	**Indicator** (*acronym*)	**How the indicator is estimated** (reference year and available spatial coverage*)
Pollution	Nitrogen concentrations in rivers (*Nconc*)	Estimated nitrogen concentration in rivers (mgN/l), based on the model GREEN (Grizzetti et al. 2012). (2005; EU-28+)
	Phosphorus concentrations in rivers (*Pconc*)	Estimated phosphorus concentration in rivers (mgP/l), based on the model GREEN (Grizzetti et al. 2012). (2005; EU-28+)
	Diffuse pollution from urban runoff (*Heaney*)	Relative intensity of the potential pollution load from urban runoff (dimensionless), estimated by the Heaney model (Pistocchi et al. 2015; Heaney et al. 1976). The indicator is designed to reproduce potential pollution and not specific contaminants, based on urban land cover (CLC 2006), annual precipitation and population. (2006; EU-28, without GR and CY)
Hydrological alterations	Total water demand (*WatDemand*)	Total water demand in the catchment upstream (mm/day) (Pistocchi et al. 2015 based on De Roo et al. 2012). (2006; EU-28, without CY)
	Low flow alteration at 25%-ile (*Q25*)	Ratio between the number of days the water flow is below the 25%-ile with and without water abstractions (fraction) (Pistocchi et al. 2015). The flow duration curve without abstractions is used to define the flow threshold of Q25%-ile. The indicator is computed using the estimations of the hydrological model LISFLOOD (De Roo et al. 2012), considering baseline conditions including water abstractions and an ideal undisturbed case with no abstractions. (2006; EU-28, without CY)
	Low flow alteration at 10%-ile (*Q10*)	Ratio between the number of days the water flow is below the 10%-ile with and without water abstractions (fraction) (Pistocchi et al. 2015). The flow duration curve without abstractions is used to define the flow threshold of Q10%-ile. The indicator is computed using the estimations of the hydrological model LISFLOOD (De Roo et al. 2012), considering baseline conditions including water abstractions and an ideal undisturbed case with no abstractions. (2006; EU-28, without CY)
Hydro-morphological alterations	Density of infrastructures in floodplains (*INFRfloodp*)	Density of infrastructure (roads and railways) in the floodplains (km/km^2) (Pistocchi et al. 2015; OpenStreetMap 2014). (dates not available, data extracted in 2014; EU-28, without HR)
	Natural areas in floodplains (*NATfloodp*)	Fraction of the floodplain occupied by natural elements (Pistocchi et al. 2015; Clerici et al. 2013). (2000; EU-28, without HR)
	Artificial land cover in floodplains (*URBfloodp*)	Fraction of urban land use (CLC 2006 class: artificial areas) in the floodplains (Pistocchi et al. 2015). (2006; EU-28, without GR and HR)
	Agricultural land cover in floodplains (*AGRfloodp*)	Fraction of agricultural land use (CLC 2006 class: arable land and permanent crops) in the floodplains (Pistocchi et al. 2015). (2006; EU-28, without GR and HR)
Integrated	Artificial land cover in catchment area (*catchURB*)	Fraction of catchment area which is urban (CLC 2006 class: artificial areas) (Pistocchi et al. 2015). (2006; EU-28, without GR and HR)
	Agricultural land cover in catchment area (*catchAGRI*)	Fraction of catchment area which is agricultural (CLC 2006 class: arable land and permanent crops) (Pistocchi et al. 2015). (2006; EU-28, without GR and HR)

(*) As at January 2017 the European Union (EU) is composed of 28 Member States (MS): Belgium (BE), Bulgaria (BG), Czech Republic (CZ), Denmark (DK), Germany (DE), Estonia (EE), Ireland (IE), Greece (GR), Spain (ES), France (FR), Croatia (HR), Italy (IT), Cyprus (CY), Latvia (LV), Lithuania (LT), Luxembourg (LU), Hungary (HU), Malta (MT), Netherlands (NL), Austria (AU), Poland (PO), Portugal (PT), Romania (RO), Slovenia (SI), Slovakia (SK), Finland (FI), Sweden (SE) and United Kingdom (GB).

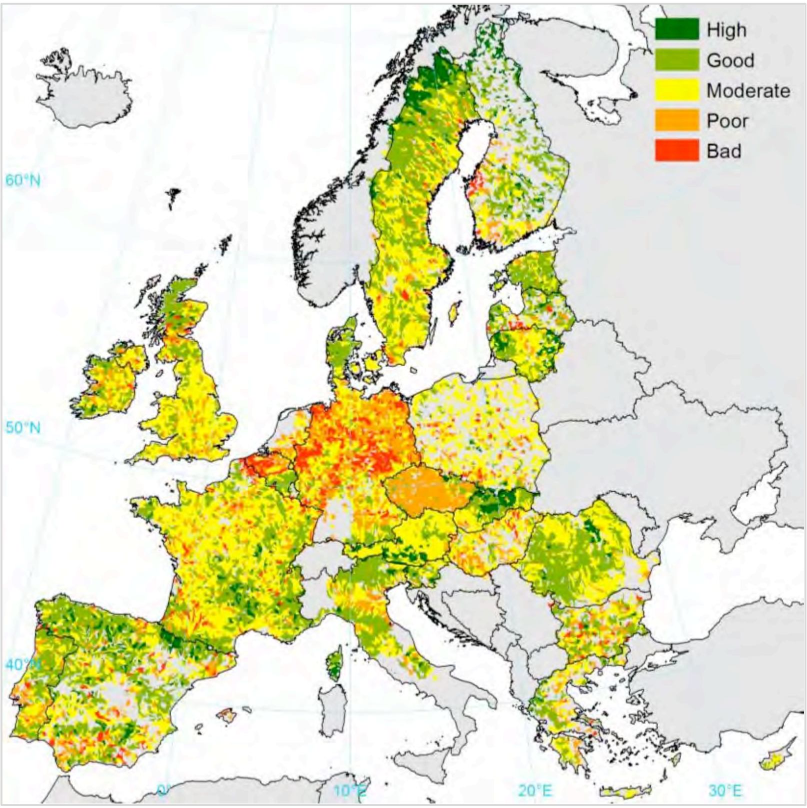
Figure 1.1. Maps of pressures on European rivers. a. nitrogen concentration; b. phosphorus concentration; c. pollution from urban runoff; d. water demand; e. preservation of low flow at 25th percentile; f. preservation of low flow at 10th percentile, g. infrastructures in floodplains; h. natural areas in floodplains; i. urban areas in floodplains; j. agricultural areas in floodplains; k. artificial land cover in catchment area; l. agricultural land cover in catchment area. Details of the pressures indicators are provided in Table 1.1.





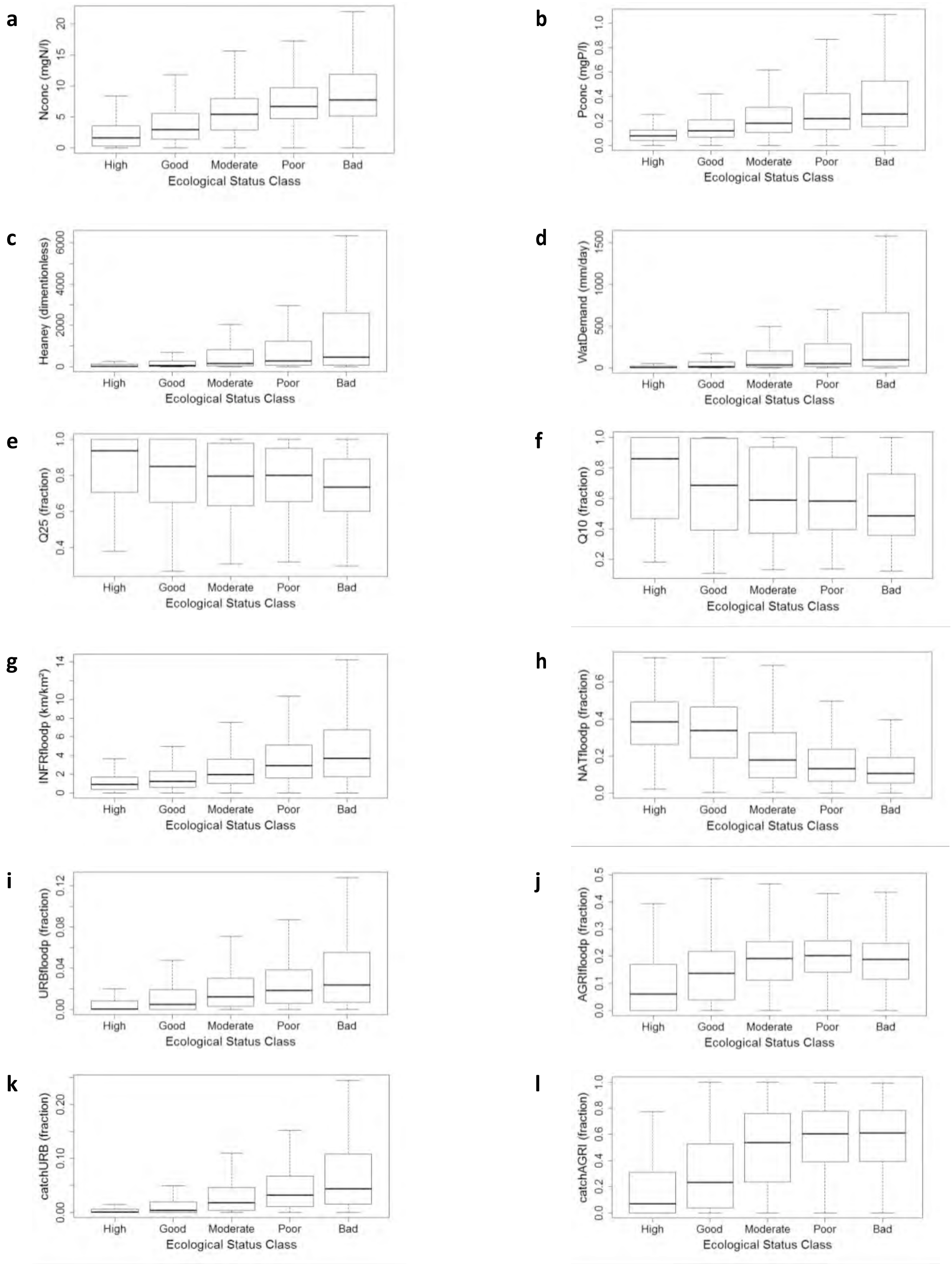
In parallel, we computed a proxy indicator of the ecological status of rivers (at the same spatial resolution of pressures), based on the data reported by EU Member States (Figure 1.2). The ecological status is an integrative evaluation of aquatic ecosystem health, designed to reflect changes in community structure and ecosystem functioning in response to anthropogenic pressures (Heiskanen et al. 2004). It is expressed in five classes — high, good, moderate, poor and bad — and its assessment is carried out by EU Member States (per single water body), using biological assessment methods. The national classification scales are harmonised by intercalibration to assure their consistency at the EU level. The target set by EU water policy is to reach a good ecological status for all rivers (by 2015 or 2027). Our proxy indicator for the ecological status of European rivers covers 77% of the EU's surface. Out of this area, 38% is estimated to be in good or high ecological status, 42% in a moderate state and the rest in poor or bad status.

Figure 1.2. Proxy of ecological status. Classes indicate the dominant ecological status class of measurements for rivers reported by Member States within the catchment (average size of the catchments is 180 km^2). The analysis refers to the period 2004-2009, for which data on the ecological status were reported.



When looking at the distribution of individual pressures per class of ecological status, we observe significant correlations and trends in the expected direction (Figure 1.3). For all indicators of pressures medians significantly differ per class of ecological status (Kruskall–Wallis test, $p<0.05$). Nitrogen and phosphorus concentrations increase towards poor and bad ecological classes, and the same happens for the indicators of hydromorphological alterations in floodplains. Also, pressures related to urban and agricultural land in the drained area take higher values in poor and bad classes, while greater maintenance of natural low flow and the presence of natural riparian areas are related to good and high ecological status.

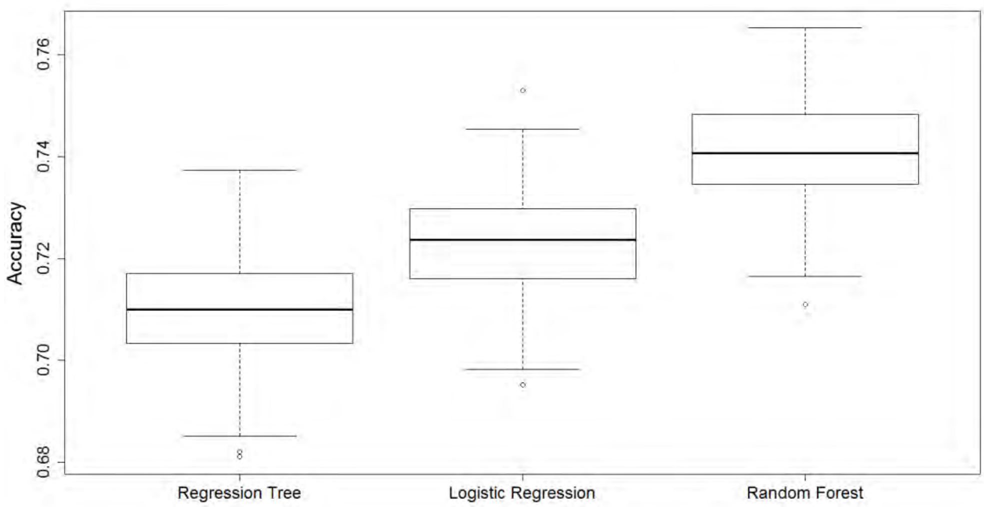
Figure 1.3. Relationship between the indicators of pressures and the proxy of the ecological status. a. nitrogen concentration; b. phosphorus concentration; c. pollution from urban runoff; d. water demand; e. preservation of low flow (at 25th percentile); f. preservation of low flow (at 10th percentile); g. infrastructures in floodplains; h. natural areas in floodplains; i. urban areas in floodplains; j. agricultural areas in floodplains; k. artificial land cover in catchment area; l. agricultural land cover in catchment area. The indicators of pressures are described in Table 1.1.



We explored the combined effects of multiple pressures on the achievement of good ecological status of rivers, applying statistical classification methods (notably, regression tree (RT), logistic regression (LR) and random forest (RF)). The accuracy of the models' predictions was up to 0.74 (0.70 for RT, 0.72 for LR and 0.74 for RF respectively, Figure 1.4a). The results of the models showed that the good ecological status of rivers is explained by a combination of pressures, and the most important predictors are the presence of natural areas in floodplains, nutrient concentration (especially nitrogen), infrastructures in floodplains and urbanisation and agriculture in the drained catchment (Figure 1.4b).

Figure 1.4. Model results. a. accuracy of classification using the regression tree (RT), logistic regression (LR) and random forest (RF) models. b. importance of the variables in the classification of the random forest method computed by the mean decrease Gini index[48,49]. The analysis refers to the period 2004-2009, for which data on the ecological status were reported and most of the pressures indicators were available.

a



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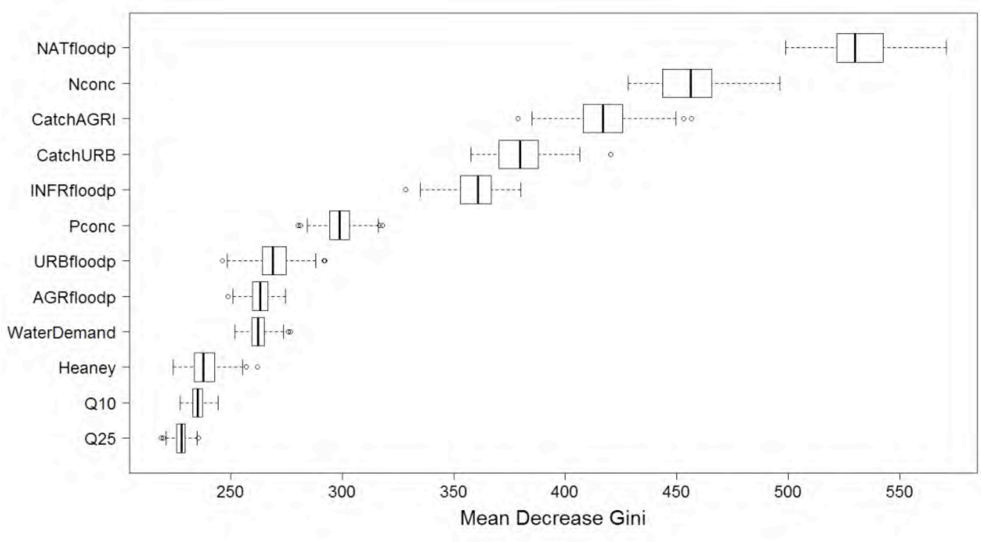
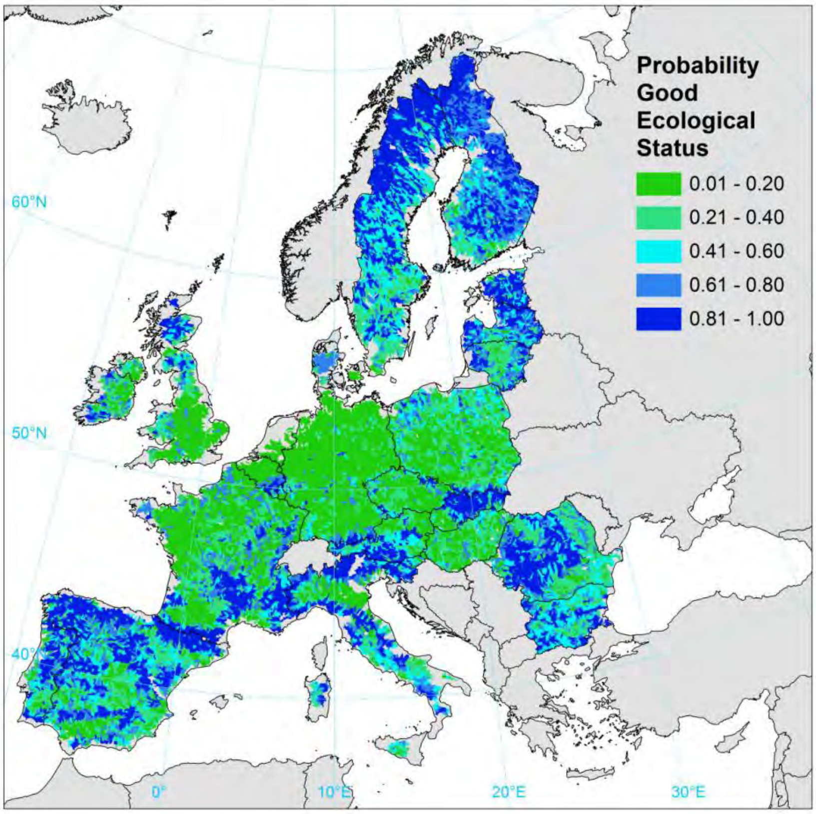


To what extent has the EU water policy target of good ecological status been achieved?

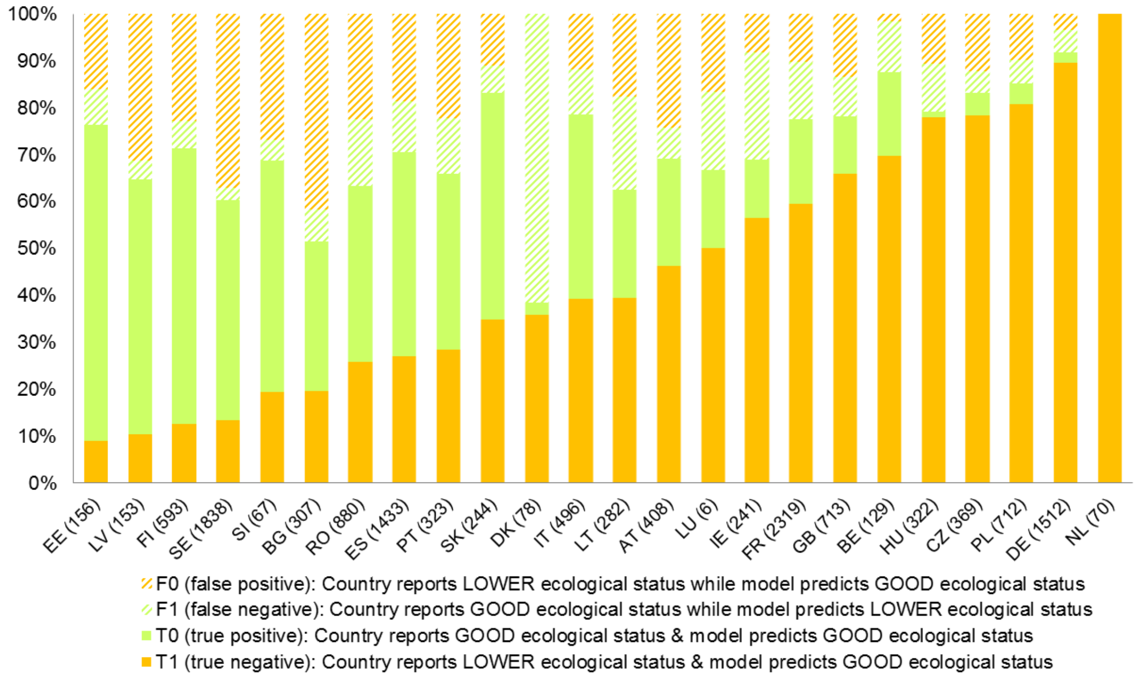
To examine this second question, we estimated the level of achievement of the EU water policy objective, using the relationship established by modelling (RF). We estimated the probability of meeting the policy target of good ecological status for all EU rivers in catchments with complete data on pressures (89% of the EU's surface), therefore, also in areas where direct measurements of ecological status were not available. According to our estimations, the proportion of the EU surface where rivers meet the water policy target, with a probability of at least 70%, is 32% (Figure 1.5).

Figure 1.5. Probability of good ecological status of rivers. Values estimated by the random forest method applied to all catchments with complete data on pressures (89% of EU).



The distribution of the model's accuracy and error type per country can provide more insights (Figure 1.6). False negatives (9%, the country reports meeting the target while the model predicts not meeting the target) could indicate where pressures are overestimated by the European assessment or local measures are not taken into account. For example, this could be the case of Denmark, where substantial investments have been made in the restoration of wetlands (Hoffmann and Baattrup-Pedersen 2007). On the other hand, false positives (17%, the country reports not meeting the target while the model predicts meeting the target) could suggest where pressures are underestimates or not captured by the current indicators. This could be the case of Sweden, where local water flow modifications could be the reason for not achieving the good ecological status (Renöfält et al. 2010). Among errors, dominance of false positives could characterise countries that adopt stricter rules or more conservative reference status in the implementation of the WFD, compared to the average of EU countries. Contrarily, dominance of false negatives might occur for countries that have slightly lower standards or consider a partially impacted ecological status as reference conditions for the water bodies.

Figure 1.6. Distribution of model accuracy and errors per country. The values within brackets indicate the number of catchments with available data. Results are based on the random forest method. The analysis refers to the period 2004-2009, for which data on the ecological status were reported and most of the pressures indicators were available.



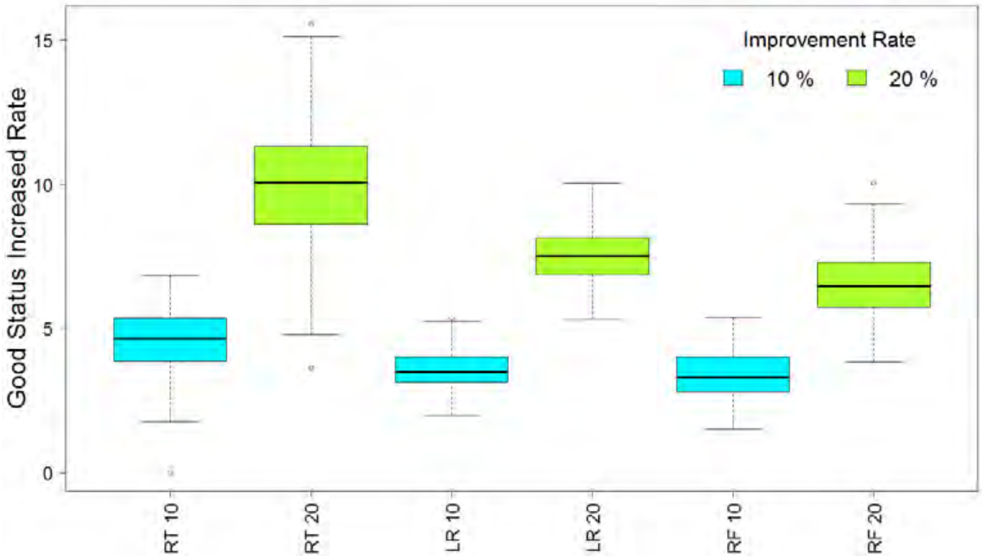
Besides misrepresentation of pressures and local measures, or difference in reference status among the national assessments, another reason that could explain the model errors is a different interaction of multiple pressures according to river typology or ecological regions. However, overall, discrepancies between model predictions and the ecological status reported by the countries are spread homogeneously across the study area, indicating no particular bias in the assessments by Member States. This is an encouraging signal considering the large effort spent on the national assessments and on the intercalibration of methods among Member States.

How and where would measures to improve the ecological status of rivers be effective?

To shed light on this third question, we examined the effects of measures to improve the ecological status of rivers through scenario analysis. We tested the scenario of concurrently reducing nitrogen pollution and increasing natural areas in floodplains (using RT, LR and RF models, Figure 1.7), as these pressures were among the most significant variables explaining the good ecological status (according to the results of the RF, Figure 1.4). The analysis showed that 4% of EU catchments with degraded rivers would achieve a good ecological status by reducing nitrogen pollution and increasing natural areas in floodplains by 10%, and up to 8% of catchments could meet the policy target if the same measures were raised to 20%. However, this is a conservative estimation, as the methods adopted do not include the effect of improving the ecological quality in one catchment on the downstream area.

Yet the scenario analysis helps us understand how addressing a combination of pressures can affect the ecological status compared to changes in single pressures (which are presented in Figures 1.8, 1.9 and 1.10) and where measures are likely to yield good ecological status. According to our results, the predicted increase in good ecological status by simultaneously reducing nitrogen concentration in rivers and enhancing natural areas in floodplains is slightly higher than the sum of the predicted increase by changing the two pressures independently, showing a synergistic effect.

Figure 1.7. Scenarios of measures for improvement of river ecological status. The scenarios are simulated by the three classification methods: regression tree (RT), logistic regression (LR) and random forest (RF). The scenarios 'measures for improvement' estimate the effects of contemporary reduction of nitrogen concentration in rivers and the increase of natural areas in floodplains, considering improvement rates of 10% and 20%.



Scenario	Model	Increased rate of good ecological status (*) (median)	Average
Improvement 10 %	RT10	4.7	3.8
	LR10	3.5	
	RF10	3.3	
Improvement 20 %	RT20	10.1	8.0
	LR20	7.5	
	RF20	6.5	

(*) The increased rate is calculated as the ratio of catchments in less than good ecological status (in the baseline) that under the scenario are predicted to pass to good ecological status (see 'Methods').

Figure 1.8. Expected effect of changing one pressure at a time on meeting the good ecological status, simulated by the regression tree (RT) method. a. changing single variable by ± 10%. b. changing single variable by ± 20%. The good status increased rate is calculated as the ratio of catchments in less than good ecological status (in the baseline) that under the scenario are predicted to pass to good ecological status.

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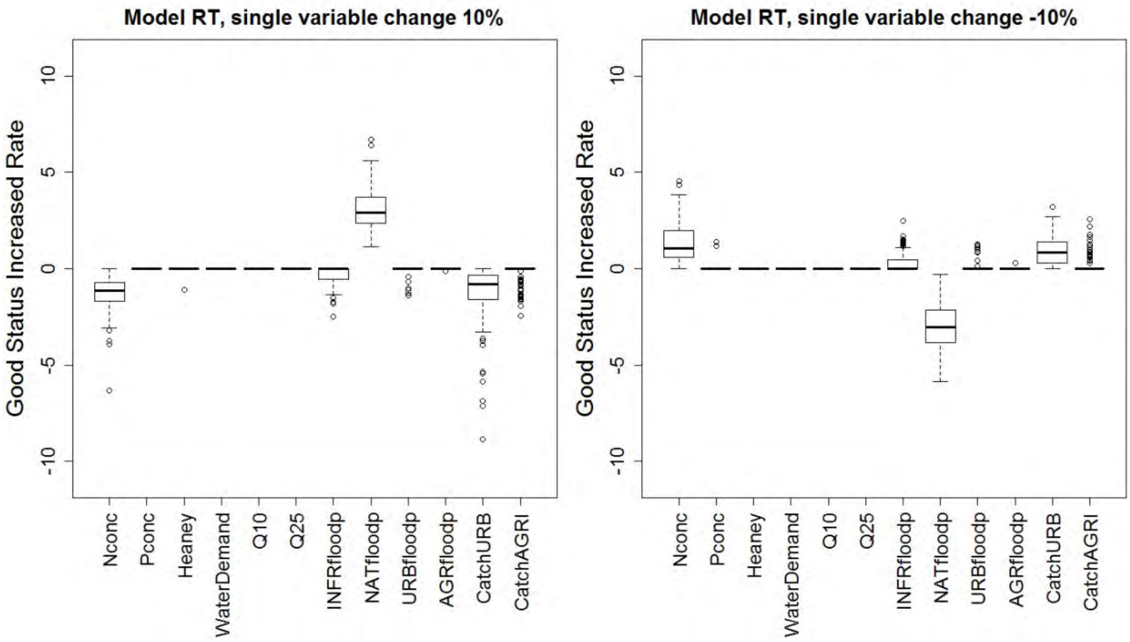

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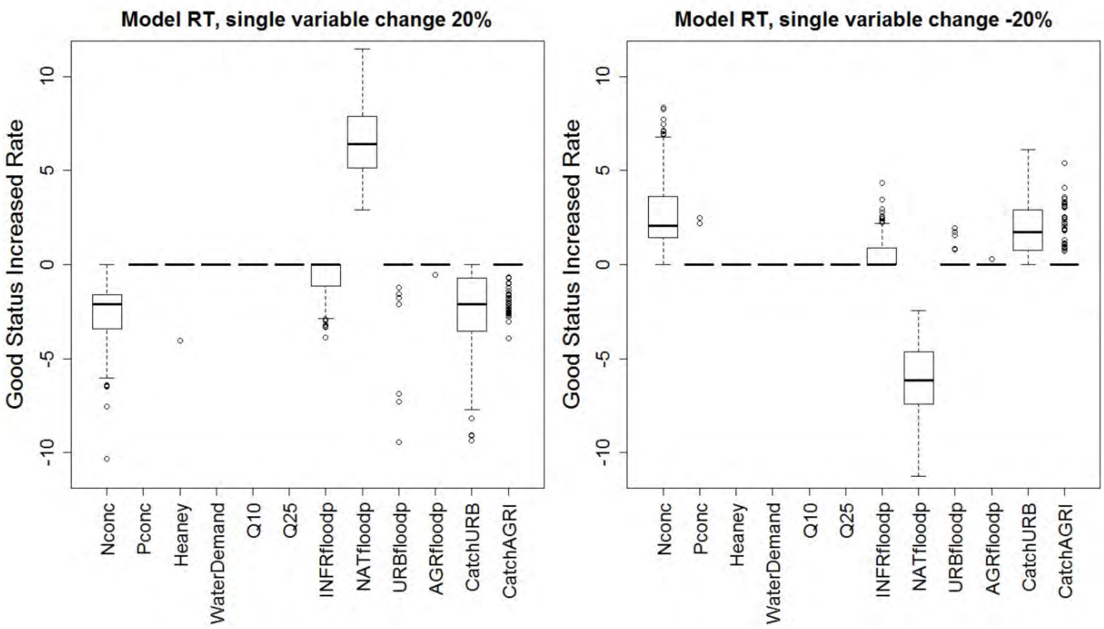
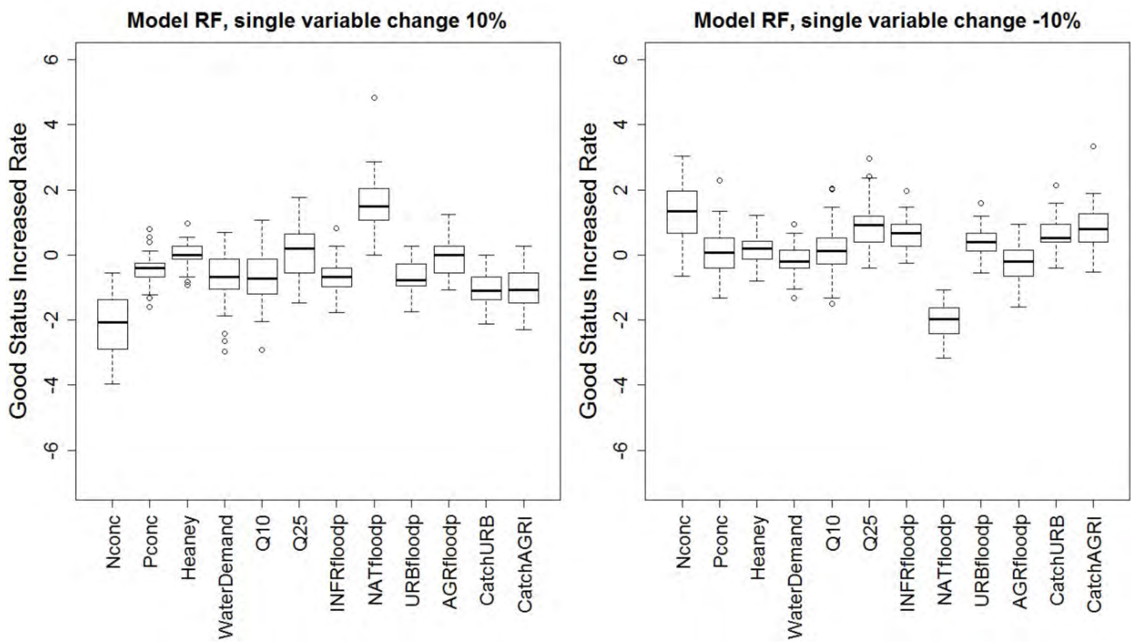


Figure 1.9. Expected effect of changing one pressure at a time on meeting the good ecological status, simulated by the random forest (RF) method. a. changing single variable by ± 10%. b. changing single variable by ± 20%. The good status increased rate is calculated as the ratio of catchments in less than good ecological status (in the baseline) that under the scenario are predicted to pass to good ecological status.

a



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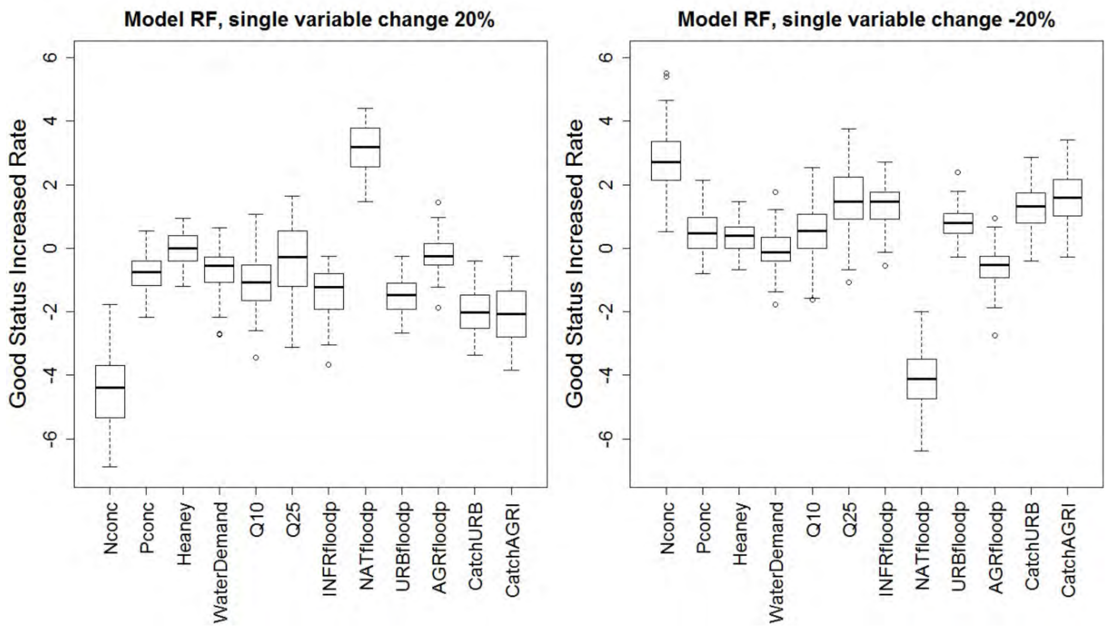
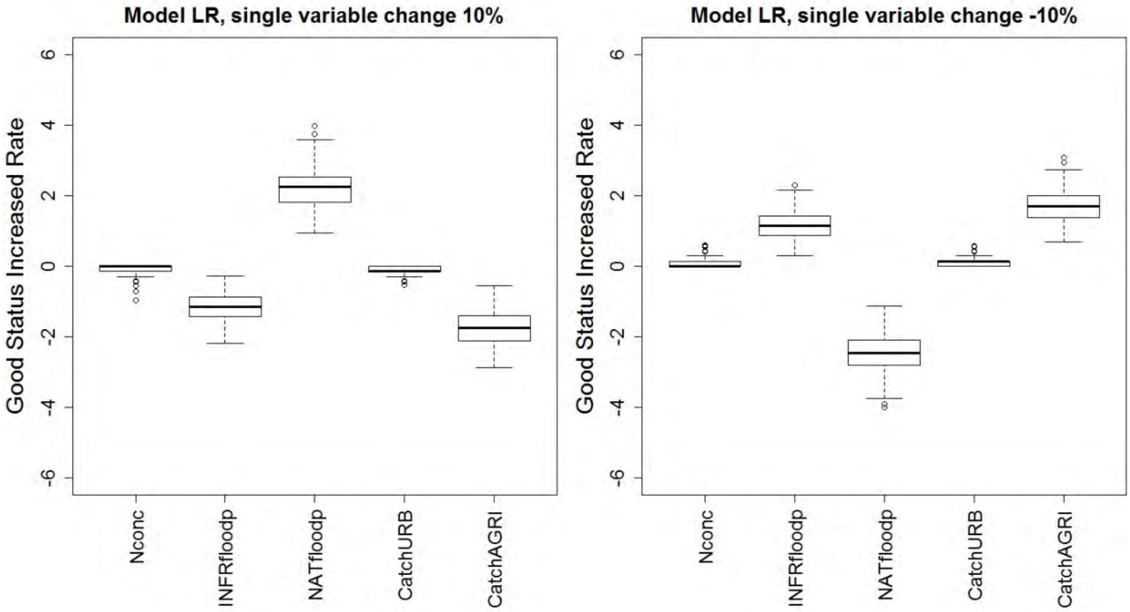


Figure 1.10. Expected effect of changing one pressure at a time on meeting the good ecological status, simulated by the logistic regression (LR) method. a. changing single variable by ± 10%. b. changing single variable by ± 20%. The good status increased rate is calculated as the ratio of catchments in less than good ecological status (in the baseline) that under the scenario are predicted to pass to good ecological status.

a



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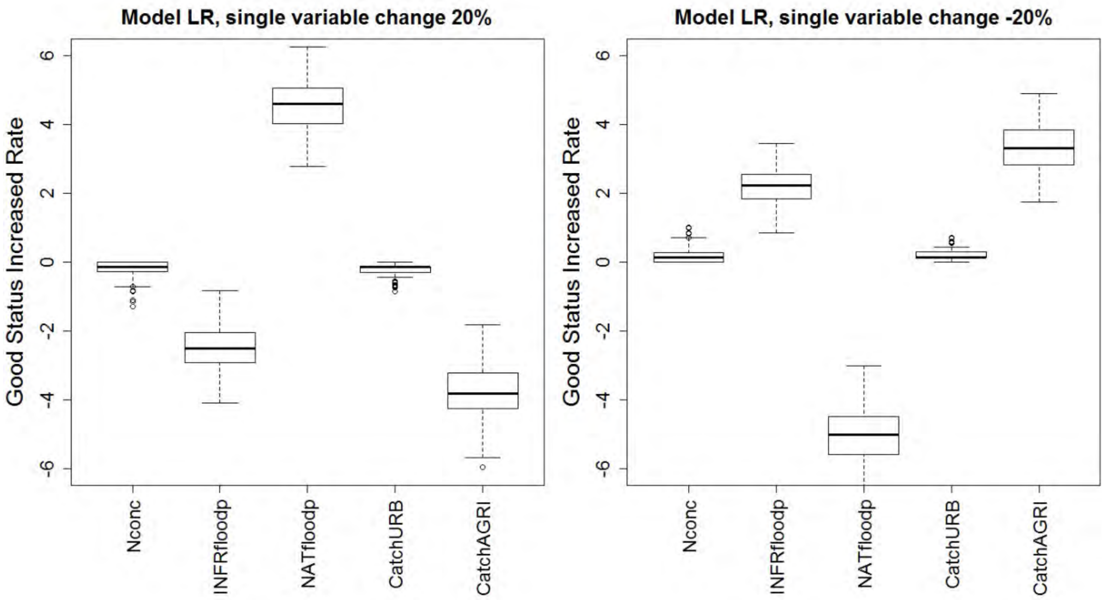


Discussion

Statistical classification models, as adopted here, cannot bring strong evidence of a causal relationship between the pressures and the ecological status, but they can unveil patterns. Our results show that the ecological status of European rivers can be explained by multiple pressures, and in particular by a combination of local pressures (i.e. hydromorphological alterations) and catchment pressures (i.e. nutrient pollution and land use). Measures to improve the ecological quality of rivers should consider these two dimensions, as well as synergistic effects of acting simultaneously on more pressures.

In our analysis, flow regime alteration and water abstractions appeared less significant. They were probably not completely represented by selected indicators or spatial information. At the same time, it is currently under debate whether the present assessment of the ecological status sufficiently accounts for hydrological alterations of river ecosystems (European Commission 2015b). Other pressures not included in this study might also be relevant to explaining the ecological status, such as the disruption of upstream-downstream connectivity, historical impacts having legacy effects and the introduction of invasive species. In addition, the river typology could explain the different impact of similar pressures combination.

The joint effort of EU Member States in monitoring the ecological status remains crucial to ensuring that effective measures for protecting and restoring aquatic ecosystems are deployed, considering the panoply of vital ecosystem services they provide (Allan et al. 2013; Grizzetti et al. 2016). Similarly, models and remote sensing data represent useful tools to assess multiple pressures across Europe, especially in less data intensive areas.

Our results indicate that maintaining natural floodplains and limiting nitrogen pollution can be key measures to improve the ecological status of rivers and achieve water policy goals, producing synergetic effects. They also suggest that preserving natural land cover as opposed to urban sprawling, which erodes the capacity of the ecosystem to buffer pressures (Pistocchi et al. 2017), should be seen an investment in ecosystem resilience. Overall, our results confirm evidence of the need to halt urban land take, curb nutrient pollution and preserve natural areas along water courses, in order to protect the ecological quality of rivers and ensure future benefits for humans.

Methods

Spatial extent and resolution

The area covered by the study is the European Union (EU). As at January 2017 the EU is composed of 28 Member States (notably Belgium, Bulgaria, Czech Republic, Denmark, Germany, Estonia, Ireland, Greece, Spain, France, Croatia, Italy, Cyprus, Latvia, Lithuania, Luxembourg, Hungary, Malta, Netherlands, Austria, Poland, Portugal, Romania, Slovenia, Slovakia, Finland, Sweden, United Kingdom). We based the spatial analysis on a consistent hydrological geodatabase covering Europe, with elementary catchments of 180 km^2 on average, called the HydroEurope database (Bouraoui et al. 2011). For inland waters the EU is divided into 23 187 catchments, corresponding to 4 098 757 km^2. In the study, we referred to this area as reference for the EU, although it is slightly less (7%) than the EU surface, as small coastal catchments are not included in the database.

Multiple pressures

The anthropogenic pressures on aquatic ecosystems were identified based on the Common Implementation Strategy (CIS) and the first River Basin Management Plans (RBMPs) submitted by the EU Member States (European Commission 2015a; Pistocchi et al. 2015). The main types of pressures reported for river ecosystems were nutrient and chemical pollution, hydrological alterations and morphological modifications. We proposed a set of 12 indicators that could inform on the quantitative presence of these pressures and could be computed consistently across Europe, using already established models or available spatial data, considering the best available data for the period 2004-2009. The indicators of pressures proposed in this study are summarised in Table 1.1, including the available reference year and the spatial coverage. For pollution, nitrogen and phosphorus concentration in surface waters were considered, based on the nutrient loads estimated by the GREEN model combined with water flow estimated by a simple hydrological model based on a Budyko framework (Grizzetti et al. 2012). In addition, load from urban runoff was estimated by an indicator accounting for urban population and rainfall, derived from the loading function proposed by Heaney et al. (1976), as described in Pistocchi et al. (2015). For hydrological alteration, the total water demand was derived from the European maps at 5 km resolution used as input by the LISFLOOD hydrological model (De Roo et al. 2012). These include water demand for irrigation, public supply, industry (including energy production) and livestock. The indicators of flow regime alteration were computed as the number of days in which the actual stream flow is below the 10th and 25th natural flow percentile, normalised by the corresponding natural duration (i.e. 36.5 and 91.25 days respectively). The actual and natural flow duration curves were estimated using the LISFLOOD model under the 2006 baseline conditions, in presence and in absence of water abstractions respectively (Pistocchi et al. 2015; De Roo et al. 2012). A series of (proxy) indicators of hydromorphological pressures were considered to reflect the conditions of floodplains, including the share of the floodplain occupied by agricultural land, by artificial areas and by natural areas (riparian functional areas), and the density of infrastructures (roads and railways) in the floodplain. Floodplains were identified through the data set described by Clerici et al. (2013). Agricultural and artificial land cover shares were estimated on the basis of the CORINE Land Cover 2006 map (European Environmental Agency, 2014). Infrastructures were extracted from the freely accessible OpenStreetMap data set (OpenStreetMaps 2014). The presence of riparian functional areas was calculated as the average riparian vegetation buffer width divided by the floodplain width, where the average riparian vegetation buffer width was derived by aggregation of the vegetation maps developed by Weissteiner et al. (2016). All variables relating to floodplains were aggregated at 1 km resolution across the stream network. Finally, the fraction of the drained catchment occupied by urban areas and by agricultural land were considered as two additional integrated indicators of pressures on rivers related to the land use in the catchment. All pressures indicators were computed or aggregated at the spatial resolution of catchments of the HydroEurope database (Bouraoui et al. 2011) (Figure 1.1).

Ecological status

The ecological status is a synthetic judgement that represents the condition of water bodies as high, good, moderate, poor or bad, based on assessment methods for biological quality elements (BQEs, that are phytoplankton, flora, invertebrate fauna and fish fauna), combined with information on physico-chemical and hydromorphological conditions. The ecological status is defined in general terms by the WFD, which is the EU water law; then each individual Member State develops national assessment methods. Depending on the Member State, the assessment of the ecological status was based on full BQEs, pressure assessments, expert judgement or combinations of the above. This variability in approaches limits the methodological consistency across the EU. However, classification scales for the biological classification methods have been intercalibrated across EU Member States (Birk et al. 2012; Poikane et al. 2015; Poikane et al. 2016).

For this study, we used ecological status data from River Basin Management Plans reported according to Article 13 of the WFD, extracted from the WISE2 database, compiled by the European Environment Agency (European Environmental Agency, 2012), including data from 2004 to 2009. For each monitored river stretch the data set reports the class of ecological status or potential and the length of the stretch. A river stretch is defined as a water body in the WFD. Only the coordinates of the centroid of each water body were available for this study, while the geographic delineation of the stretch was not available at the European scale. To overcome this lack of information and the different spatial density of monitoring across the EU, we developed a proxy indicator of the ecological status of rivers that could be representative at the scale of HydroEurope catchments, the same spatial unit at which pressure indicators were aggregated. For each catchment, we considered the ecological status assigned to all centroids of water bodies falling in that catchment, yielding valid and usable data for 79 630 water bodies across the EU. Then, for each catchment, we computed the percentage of monitored river length under each class of ecological status (with 1 = High, 2 = Good, 3 = Moderate, 4 = Poor, 5 = Bad) and the dominant class CMODE (corresponding to the mode), i.e. the class that appears most often in the total monitored length of the observations. CMODE takes values between 1 and 5, corresponding to the five classes of ecological status (Figure 1.2). We also considered a simple Boolean variable called TARGET to indicate if the good ecological status is met or not. TARGET takes value 0 when the sum of percentages of monitored river length in high and good ecological status is higher than the sum of percentages in moderate, poor and bad status, and takes value 1 otherwise. Therefore, TARGET is a proxy indicator of meeting the WFD target of good ecological status.

Data sample

The spatial extent covered by the 12 pressures indicators varies depending on the input data used in each pressure assessment (see the specific extent covered in Table 1.1). We did not have complete information on pressures for four countries — Greece, Croatia, Cyprus and Malta — whose surface represents about 5% of the EU. We could develop a completed data set of pressures for 89% of the EU's surface (85% of catchments). Data on rivers' ecological status were available for 15 052 catchments of HydroEurope (65% of EU catchments, 77% of the EU's surface). In total, there were 13 651 catchments with complete indicators of pressures and complete data on ecological status that we used for the models' calibration. This represents 59% of the catchments and 71% of the EU's surface. The temporal extent of the analysis refers to the period 2004-2009, for which data on the ecological status were reported and most of the pressures indicators were available.

Analysis

We explored the data distribution and correlation, and we performed a factor analysis. We analysed the distribution of values of each indicator of pressures per class of ecological status, using the most frequent status class reported per catchment CMODE as proxy for the ecological status. We assessed for all indicators of pressures that the medians per class of ecological status were significantly different ($p<0.05$) by a Kruskall–Wallis test (Figure 1.3).

We applied statistical methods to investigate how multiple pressures can explain the ecological status in rivers, using the variable TARGET as indicator of meeting the policy objective in each catchment. Specifically, we considered three types of classification techniques: regression trees (RT) (Breiman et al. 1984), logistic regression (LR) (McCullagh and Nelder 1983) and random forest (RF) (Breiman 2001). These methods establish a classification of catchments using the information embedded in the data. We applied the three methods using the complete data set on pressures and ecological status. This means that the temporal extent of the analysis does not refer to a specific year but is centred on the period 2004-2009.

For the analysis the three classification methods (RT, LR and RF) were applied 200 times using random samples (without replacement) extracted from the data set. Each iteration included three steps: 1. randomly balance the data set (as the number of catchments with TARGET=1 exceeded those with TARGET=0); 2. randomly select, out of the balanced data set, a training sample (80% of data) and a testing sample (the 20% remaining); and 3. run the three models (RT, LR and RF) using the training sample (model calibration). Then the accuracy of the models was measured using the testing sample (model verification), as the ratio of samples (catchments) whose value (TARGET) is correctly predicted over the total number of samples (Figure 1.4). The overall accuracy of each method was reported as the median of the 200 model runs.

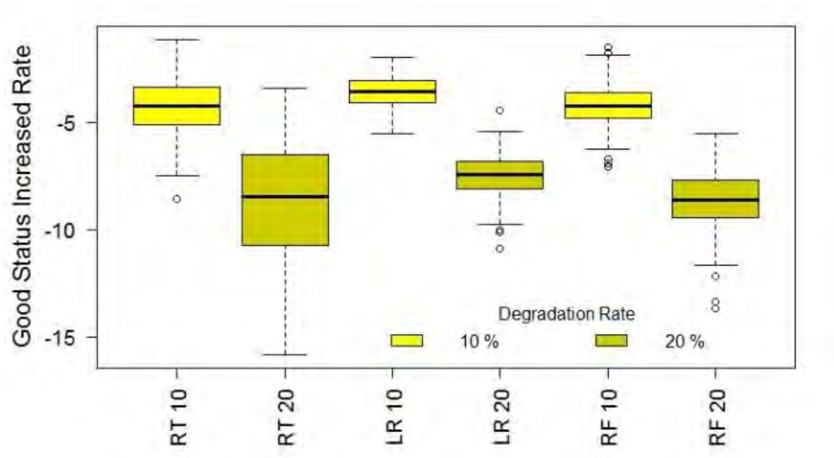
The RT and RF models were set including all 12 pressure indicators as explanatory variables. The LR model was run firstly with 12 pressures and then including only the significant variables ($p<0.1$ two-sided) and with sign coherent with the expected physical effect on ecological status. The importance of the variables in the classification of the random forest method was computed by the mean decrease Gini index (Breiman 2001; Liaw and Wiener 2015).

We used the RF method (and the variable TARGET) to predict the probability of meeting the policy target of good ecological status in all EU catchments for which we had complete pressures indicators (89% of the EU's surface) (Figure 1.5). For reporting the EU's area meeting the policy target we considered a probability threshold of 0.7.

Similarly, we based the analysis of predictions' accuracy and errors per EU country on the RF method (Figure 1.6), showing where modelled and reported ecological status were in agreement on meeting (T0) or non-meeting (T1) the policy target of good ecological status, and the frequency of false positive (F0, the model predicts meeting the target while the reported data indicate lower ecological status) and false negative (F1, the model predicts not meeting the target while the reported data indicate at least good ecological status).

Finally, we simulated two types of scenarios: the effect of measures for improvement of the ecological status (Figure 1.7) and the effect of further degradation (Figure 1.11), using the three methods, RT, LR and RF (and the variable TARGET). In the scenario 'measures for improvement' we tested the concurrent reduction of nitrogen concentration in rivers (– 10% and – 20%) and increase of natural areas in floodplains (+ 10% and + 20%), while in the scenario 'further degradation' we simulated the simultaneous increase of nitrogen concentration in rivers (+ 10% and + 20%) and reduction of natural areas in floodplains (– 10% and – 20%). The effects of the changes were quantified as the increase rate of catchments predicted in good ecological status (meeting the target of the water policy) compared to the baseline. For the scenarios, the models were run according to the three-step iteration presented above, and the effects tested on the catchments correctly classified by the models. We reported the overall expected effect of the scenarios as the average of the medians of the three models' predictions. We also simulated a variation of ± 10% and ± 20% of one pressure at a time, using the three methods (RT, LR and RF). The results are shown in the Figure 1.8, 1.9 and 1.10.

Figure 1.11. Scenarios of further degradation of river ecological status. The scenarios are simulated by the three classification methods: regression tree (RT), logistic regression (LR) and random forest (RF). The scenarios 'further degradation' estimate the effects of contemporary increase of nitrogen concentration in rivers and the decrease of natural areas in floodplains, considering degradation rates of 10% and 20%.



Scenario	Model	Increased rate of good ecological status (*) (median)	Average
Degradation 10%	RT10	-4.3	-4.0
	LR10	-3.6	
	RF10	-4.3	
Degradation 20%	RT20	-8.5	-8.2
	LR20	-7.5	
	RF20	-8.6	

(*) The increased rate is calculated as the ratio of catchments in less than good ecological status (in the baseline) that under the scenario are predicted to pass to good ecological status (see 'Methods').

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Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

Deliverable 5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis

D5.1-1 Part 2: Analysis of pressure - response relations: classification of multiple pressures on broad river types

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Due date of deliverable: **Month 36**
Actual submission date: **Month 36**

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)	

Content

List of Annexes

Non-technical summary

For this deliverable a unique and comprehensive collation of input data were derived. Information from different data sources, in varying formats, spatial resolution, comprising information on hydrology, physico-chemical water quality, geo-morphological characteristics, ecological status and other, were harmonized and merged to an extended database. The data were derived for about 100,000 sub-catchments (FECs) covering Europe, EFTA states and further, hydrologically connected areas to the east.

From this database pressure indicators were deduced and statistically compared to the ecological status reported by the EU-countries. An important and novel indicator is the impact of hydrological alteration on major flow characteristic like base flow, floods or duration of low flows. These were derived by comparing modelled flows for current conditions and for semi-natural conditions. The goal was identifying the most explanatory pressure indicators impeding a good ecological status.

First, the general statistics on the distribution of all pressure indicators were conducted, secondly, the pressure indicators were compared to the ecological status as assessed in 1st River Basin Management Plan (RBMP).

Importance of pressures for supporting good ecological state varies a lot among river types and regions in Europe. On large rivers, chemical stressors, percentage of broad leaved forest and share of agricultural land in floodplain are three most important pressures. On lowland, medium to large rivers, high flow hydrological characteristics become very important also. Share of coniferous forest in floodplain is important pressure in mid altitude rivers, whereas base flow and oxygen demanding substances are important for highland rivers.

Our results also suggest, that diffuse pollution of nutrients and decrease of riparian vegetation at present do not support good ecological status mainly in the Mediterranean and Atlantic regions. In the Central and Baltic region, the most important cause for a deterioration of ecological status is the combination of diffuse pollution of nutrients and hydrological alterations. In the Eastern Continental region all three types of pressures, namely, hydrological, morphological and chemical are equally important.

Classification of multiple pressures on European broad rive types presented here is closely related to JRC work (Grizzetti et al., 2017) and NTUA work (MARS, 2017) in the same work package of the MARS project. JRC has unveiled patterns between human pressures and ecological status of European rivers in non - stratified manner. NTUA has analysed relation of low flows and ecological flows (E-flows) to ecological status and contributed data for hydrological pressures. This contribution is very important, since in our study we indeed show that hydrological pressures are very important, and in some regions and river types even prevail over morphological pressures.

Our results serve as an input to scenario analysis tool at the European scale (namely work package 7.4) and will be expanded with additional data and expert knowledge.

1 Introduction

We analysed which pressures or combinations of pressures impact ecological status of European rivers. We distinguished between pressures that are caused primarily by human activities and can be managed (e.g. pollution of nutrients, hydro-morphological alterations, changes of land use in flood plain) and pressures due to differences of natural factors that also impact ecological status (e.g. natural characteristics of upper, middle and lower course of river; altitude; longitude, latitude). The same pressure or the same natural factor can have a different impact on ecological status as different biological communities may be involved. On a pan-European scale, analyses are almost impossible to be conducted on biological community or species basis. Therefore the analyses were stratified by river types and regions since they represent natural and geographic characteristics of rivers, and potentially group cognate ecological habitats.

For the purpose of this study, we used empirical and distribution-free methods. We obtained empirical data from national monitoring programmes reported by national institutions to the European Commission or other European institutions. Since we wanted to capitalise on chance, we chose distribution-free statistical methods. The selected approach belongs to the field of machine learning and predictive modelling. This approach was also used by regional observational studies for ecological casual assessments when analysis were hampered by high natural variability and the influence of confounding factors (Gerritsen et al, 2015). When addressing large geographical and diverse areas like Europe, casual assessments of relations between multiple pressures and ecological response becomes very challenging, may be obstructed by limited and spatially heterogeneous data.

Multiple pressures are acting concurrently on water bodies, but their combined effect is poorly understood. Nevertheless, understanding relations between multiple pressures and ecological status is a prerequisite to plan effective polices and management measures (Hering et al, 2015; Teichert et al, 2016). Being aware of these challenges, we aimed to identify important pressures on European waters, their spatial distribution and combinations as well as their effects on the ecological status of rivers. We argue that, combined or single pressure, as long as they do not exceed given values (threshold values), most likely do not cause a significant deterioration of ecological status.

The findings can support management decision spatially by identifying hot-spot regions and eclectic by identifying types and combinations of important pressures. We expect that our results will generate new and additional questions and needs for explanations on how multiple pressures in Europe impact river ecological quality.

2 Data and methods

2.1 Data sources

Pressures, relevant for surface waters, are according to EC (2014), grouped into five categories (WP2, MARS Terminology, June 2015):

- Point pressures (urban waste water, combined sewer storm overflows, IED1 and non IED plants, contaminated/abandoned industrial sites, waste disposal sites, mine waters, aquaculture, and others).
- Diffuse pressures (urban runoff, agriculture, forestry, transport, contaminated/abandoned industrial sites, discharges not connected to sewerage network, atmospheric deposition, mining, aquaculture, and others).
- Abstraction/flow diversion of surface waters (for purpose of agriculture, public water supply, industry, cooling water, fish farms, and others).
- Physical and hydrological alteration (physical alteration of channel/bed/riparian area/shore of water body; dams, barriers and locks; and all other hydro-morphological alterations impacting the flow regime (e.g. disconnecting floodplains, river canalisation or straightening).
- Other pressures (introduced species and diseases; exploitation/removal of animals/plants; litter/fly-tipping; change in thermal, mixing or ice regime2).

The majority of data used for our analyses was collected in national monitoring programmes and reported to the European Commission or other European institutions under various water and environment protection policies (Table 1). For our analyses pressure indicators were derived from this data set. Further, we included physico-chemical parameters reported as State of the Environment (WISE) for selected stations (SoE stations) in Europe. Besides national monitored and reported data we included modelled pressure indicator on nutrient emissions and modelled hydrological parameters (description below).

As response indicator we used is the ecological state of rivers as reported in 1st RBMP[3]. All data therefore represent period 2005-2010. Analyses of relations between pressure and ecological status (pressure - response relations) were performed for SoE stations, therefore spatial relations between SoE stations and pressure indicators derived from data sources given in Table 1 were build.

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[1] Directive on Industrial Emissions, committing European Union member states to control and reduce the impact of industrial emissions on the environment.

[2] Change in thermal, mixing or ice regime are added by MARS project.

[3] 1st Cycle River Basin Management Plans

Table 1: Available datasets, used for pressure indicators and methods for their linkage with SoE stations.

Data owner	Database name	Spatial object to relate	Content used in analysis	Methods for linkage of data to WISE SoE station
EU directives	WFD	river water body centroids and vectors	ecological status, naturalness category (natural/artificial/heavily modified)	Directly from WB vectors and indirectly through WB code applied to river segment
	UWWTD	discharge points (locations of emissions from UWWTP) and agglomerations	yearly loads of BOD5, total phosphorus and total nitrogen	locations linked and shifted to river segments
EEA	WISE SoE (SoE)	locations of monitoring	physico-chemical parameters, 2005-2010 averages; EQR values, EQR normalised values	SoE stations linked to river segments; each SoE station lying on main drain river in FEC were assigned also hinterland data; checking if SoE station is representative for FEC (the most downstream SoE station on main drain)
	EEA Corine Land Cover 2006 and Globe Corine 2009	territory of EEA members (polygons)	level 1 and 2 categories	share of each land use category on FEC, its hinterland and river buffer
	Copernicus LC/LU data	buffer strips along rivers	all	share of each land use category in river buffer (also named floodplain)
EUROSTAT	nutrients	polygons representing EU MS	data on fertilizer application and livestock excreta	NUTS data disaggregated to agricultural area and recalculated to FECs
EC	E-EPRTR	emission point of large industrial and communal installations	release data	locations linked and shifted to river segments
JRC	population density	EU MS territory (raster format)		grid values recalculated to FEC and hinterland; SoE stations on main drain are related to data of hinterland; other stations only data from FEC
IGB	Diffuse pollution of nutrients	MONERIS modelling, prepared for each FEC	nitrogen balance on agricultural land (tonnes/year), phosphorus degree of saturation	values given on FEC level
DELTARES	runoff without abstractions and runoff with water abstraction	raster 5 arc minutes		disaggregated to FEC, WISE SoE station representative of FEC gets this info

2.2 Spatial data and integration of datasets

The collated data (Table 1) were pre-processed, quality checked, harmonized and compiled into an integrated MARS spatial database (MARSgeoDB). Following its spatial resolution, all data considered for this task were processed and aggregated on Functional Elementary Catchments (FECs) level, as given by the EEA Catchments and Rivers Network System (ECRINS) (EEA, 2012). For our purpose we selected FECs that belong to European river basins and modified some due to spatial inconsistencies. Incorrect routes of some rivers and incorrect (sub)catchment division between Black Sea and Adriatic Sea catchments were corrected by changing inflow-outflow FEC relations stored in "Code_Arbo" attribute. For modelling of nutrient emissions further spatial data was used (to be described in deliverable D7.2). A detailed technical description of the correction and modification of FECs spatial features is given in Annex 1.

A corresponding hinterland was assigned to each FEC, i.e. catchment/drainage area, composed by all FECs upstream of a selected FEC (including the selected FEC). Mean or sum values of all natural characteristics and pressures from all contributing FECs were calculated for the hinterlands.

Each FEC contains numerous river segments (linear spatial features), representing the FEC's main drain as well as its tributaries. Point information of pressures (explained in later chapter as well as water body codes reported in 2010 were linked to these river segments. By default, ECRINS (v1.1) does not contain link between ECRINS features and corresponding WFD WB. Such links were established in a separated exercise by EEA (2015, personal communication). As result of this exercise WFD WB codes were assigned to the majority of main drain river segments. More details are given in Annex 2.

All pressure indicator data were linked to the river segment representing the main drain of a corresponding FEC. European regional statistics, reported for administrative regions (NUTS)[4] were disaggregated to FECs by calculating shares of each administrative region to a FEC in the first step. In the second step, data available in absolute values were transferred to area specific values (per km^2 of region). In the third step, the density values were assigned to FECs by multiplying area specific values with area shares of each administrative region on a FEC. In the final step, absolute values per FEC were calculated by multiplying the mean area specific values with the FEC area.

Raster data with different resolutions (e.g. altitude and population density) were first re-projected to ETRS 1989 LAEA projection which is used for all features in MARSgeoDB. Data were further aggregated to FEC using ESRI's Zonal Statistics tool. To FECs smaller than grid cells, grid values corresponding to the FEC's centroid was assigned.

[4] See http://ec.europa.eu/eurostat/web/nuts/overview

Data available as geographical points (e.g. UWWTD discharge points and E-PRTR release data) were linked to the FECs using the ESRI's spatial join tool. For UWWTP or E-PRTR releases located in the same FEC with an SoE station, it is important to identify their spatial relation (upstream, downstream or on different branch of a river). This information was obtained by linking points of pollution and SoE to river segments inside the FEC. In the first step, the points were linked to the closest river segment by allocable river names. Remaining points were linked to the closest river segment within 1000 m search radius.

WFD river water body defined in 1st RBMP were by obligation reported as points, though some countries provided also polylines. Where possible, linkage were established between SoE stations and WFD river water body polylines, otherwise, WFD water body points were linked to SoE stations via the already established link between SoE stations and ECRINS river segments provided by EEA. Details on the procedure gives Annex 2.

Out of 731,112 ECRINS river segments, 32 % were assigned to corresponding WFD WB elements. The low coverage can be explained by the fact that small tributaries, representing the larger share of ECRINS river segments, were not included in the WFD WB codes reported by the EU countries. Further, no WFD WB codes were available for all river segments in FECs located in non-EU countries.

WFD database gives information on ecological status and naturalness of water body. The latter is classified into three descriptive categories: natural water body, heavily modified water body and artificial water body. The ecological status for natural water bodies is classified into five descriptive categories: high, good, moderate, poor and bad. Figure 1 and Figure 2 show the distribution of points representing natural and heavily modified / artifical water bodies, respectively. The colour of the point indicate the reported ecological stauts, visualy revealing a higher share of of good and high ecological status for natural water bodies than for heavily modified / artifical water bodies.

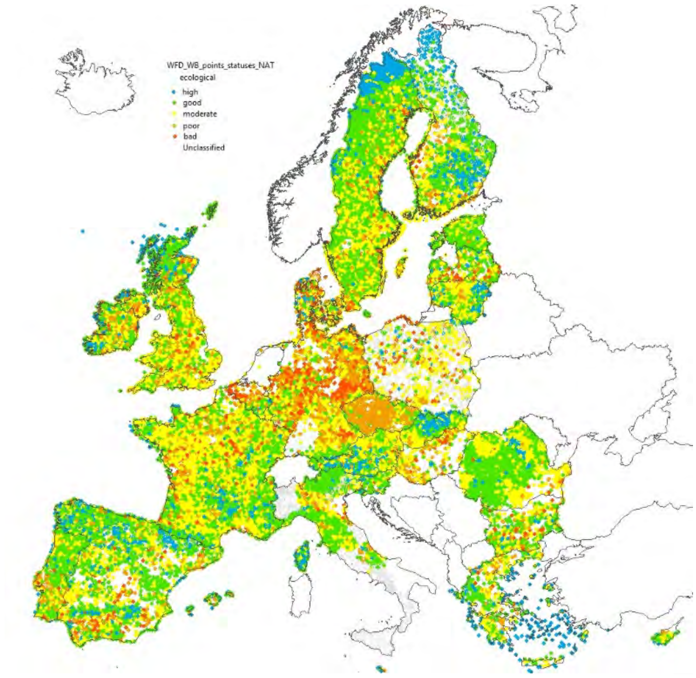

Figure 1: WFD points of natural water bodies coloured according to the reported ecological status. Red and orange points represent rivers water bodies in bad and poor status respectively; green and blue points represent river water bodies in good and high ecological status respectively.

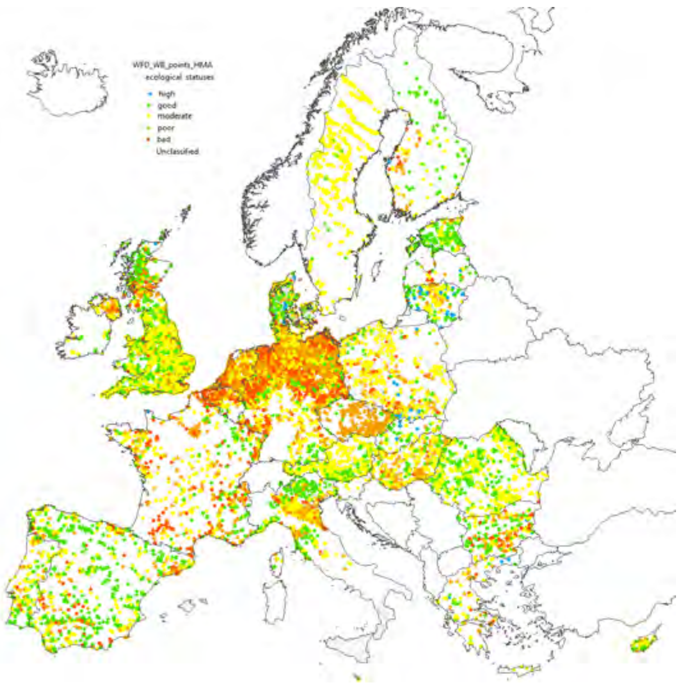

Figure 2: WFD points of heavily modified and artificial water bodies according to the reported ecological status. Red and orange points represent rivers water bodies in bad and poor status respectively; green and blue points represent river water bodies in good and high ecological (potential) status respectively.

For the pressure - response analyses, all information reported under WFD were assigned to the SoE stations. A detailed description of this working step is given in Annex 2. For 11,714 out of 16,129 SoE stations, WFD WB information could be assigned (Figure 3). Figure 4 gives the distribution of ecological status categories assigned to the SoE stations. 8,798 SoE stations are classified as natural, whereas 2,916 are attributed to heavily modified or artificial water bodies. The distributions of status categories is significantly different between the two water body groups. In both water body groups moderate ecological status is dominant (40 % of natural and 48 % of SoE heavily modified / artificial water bodies). However, in natural water bodies the share of good and high ecological status is higher (41 %) than in heavily modified or artificial water bodies (17 %). Only 19 % of the natural water bodies where classified poor or bad, whereas 35 % of the heavily modified and artifical water bodies were attributed to this status category.

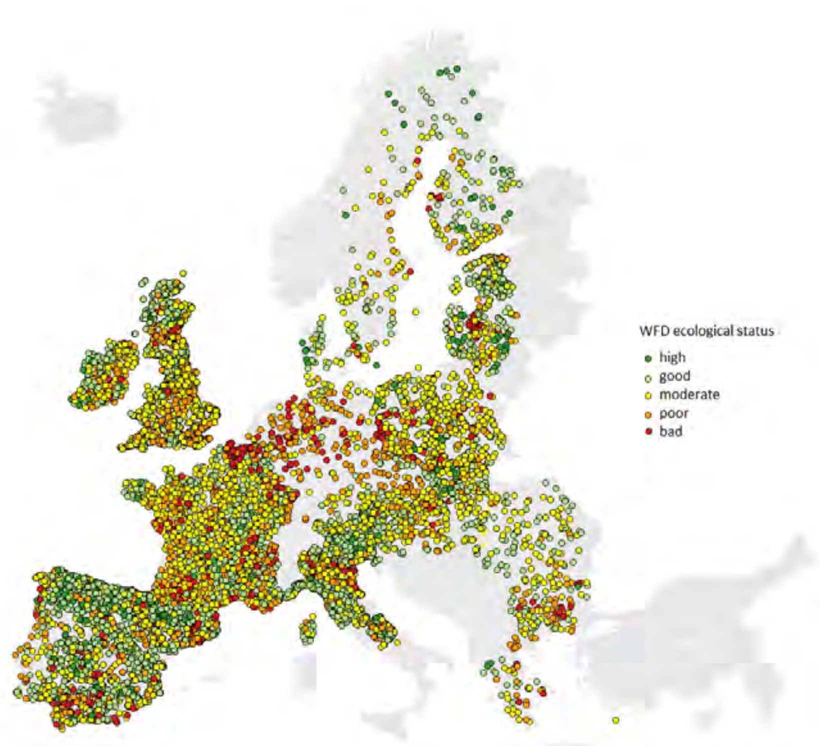

Figure 3: SoE stations with ecological status assigned from WFD river water bodies. Red and orange points represent rivers water bodies in bad and poor status respectively; green and blue points represent river water bodies in good and high ecological (potential) status respectively.

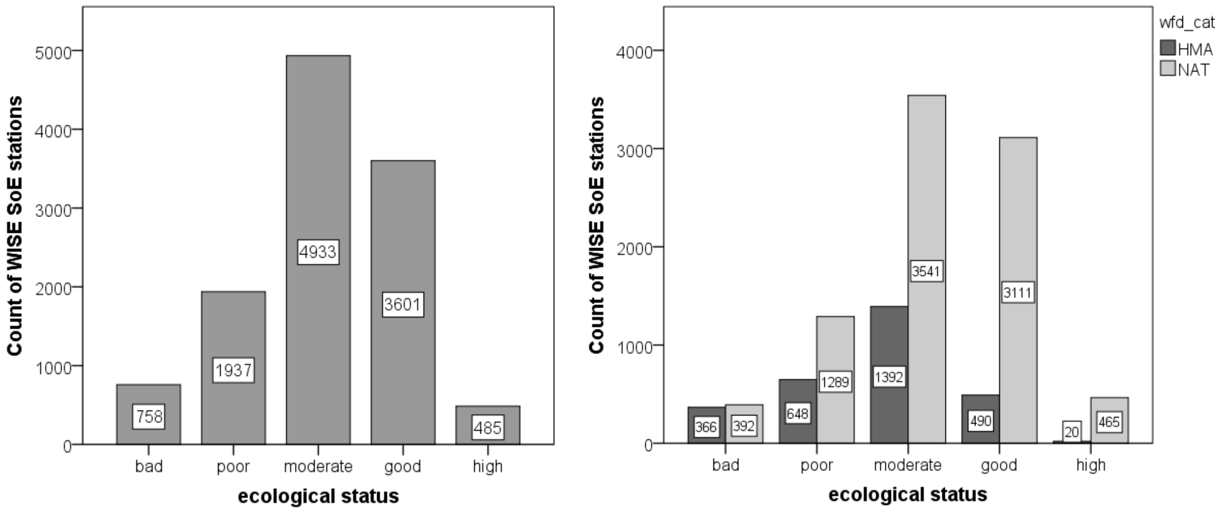

Figure 4: Number of SoE stations in each ecological status (or ecological potential) class; on left all SoE stations together (in total 11714) and on right separated by natural (NAT) (8798 in total) and heavily modified or artificial river segments (HMA) (2916 in total or 26% of all).

2.3 Broad river typology

National types of rivers and lakes have been defined in each EU Member State (MS) and Norway according to the WFD, including a variety of typology factors, (altitude, size, geology, distance from river source, mean water depth and slope, river flow category, air temperature and precipitation, etc.). The typology is important for establishing reference conditions for one or more quality elements, used to assess the ecological status. The analysis of the first WFD River Basin Management Plan (RBMPs) reported in 2010 showed that MSs have reported altogether 1,599 river types and 673 lake types (Lyche Solheim et al. 2012, Nixon et al. 2012). The typology factors, most often used for rivers, are catchment size, altitude and geology (ETC/ICM, 2015). For allowing a comparison of WFD river types and types of rivers defined under the Habitat Directive (HD), a set of broad rivers types were derived by EEA and ECOSTAT in 2015. A combination of cluster analysis and iterative dialogues with Member States, through the WFDCIS-WG ECOSTAT was applied to assess the national type similarity based on the most commonly used typology factors. These factors are altitude, geology and catchment area for rivers, and altitude, geology (alkalinity and colour), surface area and mean depth for lakes. Many national WFD types have high similarity and could be aggregated to 20 broad river types and 15 broad lake types based on altitude, size and geology (and mean depth for lakes), including most EU MS and Norway (ETC/ICM, 2015). The broad river type categories are described in Table 2.

Table 2: European broad river types (ETC, Anne Lyche Solheim, 2015).

Broad river type code	Broad river type name	Altitude (m a.s.l.)	Catchment area (km2)	Geology
1	Very large rivers (all Europe)	any	>10 000	any (usually mixed)
2	Lowland, Siliceous, Medium-Large	≤200	100 - 10 000	Siliceous
3	Lowland, Siliceous, Very small-Small	≤200	≤100	Siliceous
4	Lowland, Calcareous or Mixed, Medium-Large	≤200	100 - 10 000	Calcareous/Mixed
5	Lowland, Calcareous or Mixed, Very small-Small	≤200	≤100	Calcareous/Mixed
6	Lowland, Organic and Siliceous	≤200	<10 000	Organic and Siliceous
7	Lowland, Organic and Calcareous/Mixed	≤200	<10 000	Organic and Calcareous/Mixed
8	Mid altitude, Siliceous, Medium-Large	200 - 800	100 - 10 000	Siliceous
9	Mid altitude, Siliceous, Very small-Small	200 - 800	≤100	Siliceous
10	Mid altitude, Calcareous or Mixed, Medium-Large	200 - 800	100 - 10 000	Calcareous/Mixed
11	Mid altitude, Calcareous or Mixed, Very small-Small	200 - 800	≤100	Calcareous/Mixed
12	Mid-altitude, Organic and siliceous	200 - 800	<10 000	Organic and Siliceous
13	Mid-altitude, Organic and Calcareous/Mixed	200 - 800	<10 000	Organic and Calcareous/Mixed
14	Highland (all Europe), Siliceous, incl. Organic (humic)	>800	<10 000	Siliceous
15	Highland (all Europe), Calcareous/Mixed	>800	<10 000	Calcareous/Mixed
16	Glacial rivers (all Europe)	> 200	<10 000	any
17	Mediterranean, Lowland, Medium-Large, perennial	≤200	100 - 10 000	any
18	Mediterranean, Mid altitude, Medium-Large, perennial	200 - 800	100 - 10 000	any
19	Mediterranean, Very small-Small, perennial	< 800	≤100	any
20	Mediterranean, Temporary/Intermittent streams	any	<1 000	any

For assigning broad river typologies to river segments in the spatial model we used a geology map and information on altitude and catchment size from the MARSgeoDB. A detailed description of the geology map processing and broad river typology assignment is given in Annex 2

To map multi-stressor conditions, ECRINS main drain river typology information were applied to associated FECs[5]. Main drains in a majority of FECs (82 %) are attribute to only one single broad river type. In the remaining FECs, where more than one broad type was assigned to main drain river segments, the broad type of the outflow river segment was selected as a broad type representative for the whole FEC. Figure 5 shows FECs coloured by broad river type. The largest share of FECs (10.1 % or more than 10,000 FECs) belongs to low land-siliceous-small type rivers (type 3, Table 3). Second largest share (9.8 % or 9812 FECs) are Mediterranean small perennial

[5] This operation was conducted only for FECs that are not fully covered by lake(s).

rivers (type 19). Third in line is type 9 (mid altitude, siliceous, small rivers) with 9.2 % (almost 9300) FECs. The low land, medium to large size and calcareous or mixed river type (type number 4) is representative for 8.7 % FECs. Figure 6 shows the distribution of broad river types allocated the SoE stations. 2,802 out of 16,110 SoE stations belong to broad river type 4, followed by broad river type 18 and 2 with 1,673 and 1,475 SoE stations, respectively.

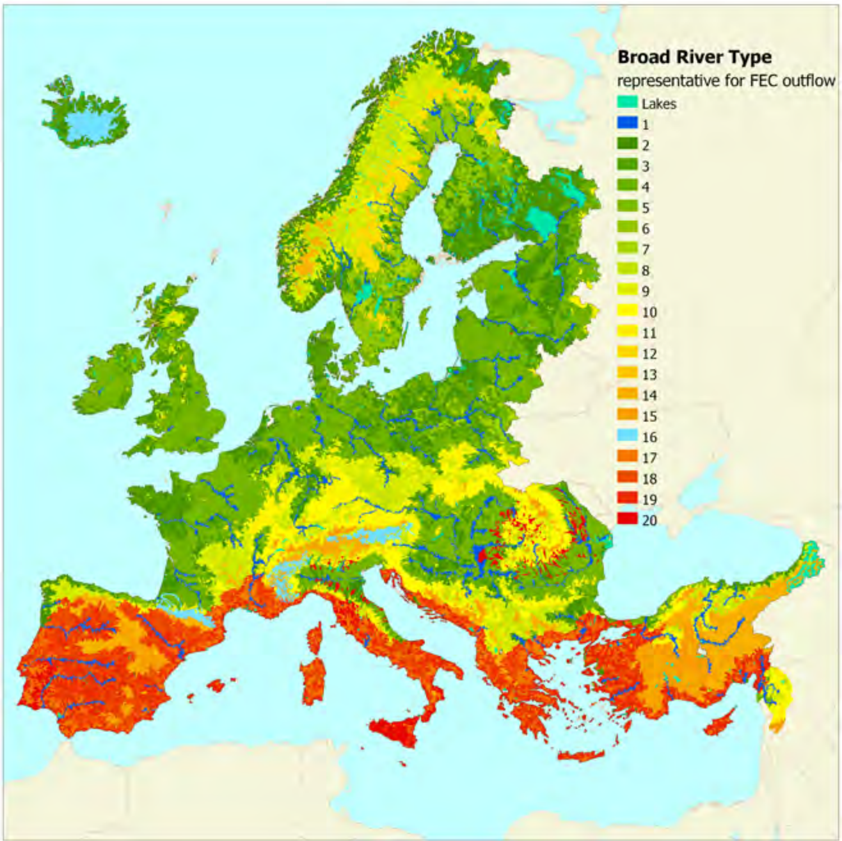

Figure 5: River broad types applied to FECs. Each FEC has been assigned one representative river broad type (delegated from a river segment that represents FEC outflow).Number is legend represents river broad type as described in Table 2.

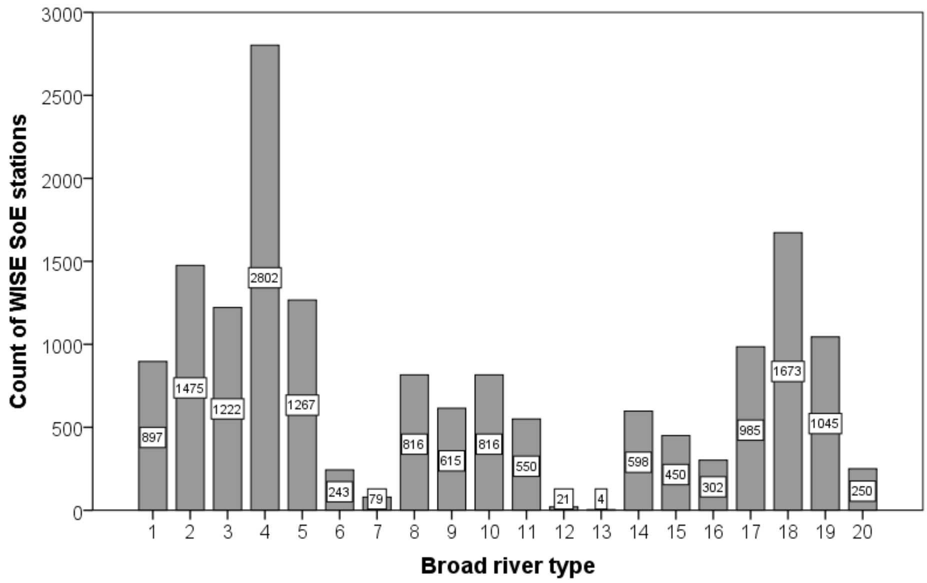

Figure 6: Number of SoE stations linked to ECRINS rivers aggregated on river broad type.

Table 3: Number and percentage of FECs belonging to each river broad type.

Broad river type (longer description in Table 2)	Number of FECs belonging to river broad type	Percentage of FECs belonging to river broad type
1: Large rivers	4091	4.10%
2: Lowland, Sil, MedLarg	7057	7.00%
3: Lowland, Sil, Small	10114	10.10%
4: Lowland, Calc, MedLarg	8712	8.70%
5: Lowland, Calc, Small	7079	7.00%
6: Lowland, OrgSil	2975	3.00%
7: Lowland, OrgCalc	499	0.50%
8: MidAlt, Sil, MedLarg	6373	6.30%
9: MidAlt, Sil, Small	9299	9.20%
10: MidAlt, Calc, MedLarg	4326	4.30%
11: MidAlt, Calc, Small	5311	5.30%
12: MidAlt, OrgSil	1708	1.70%
13: MidAlt, OrgCal	44	0.00%
14: Higland, Sil	7561	7.50%
15: Highland, Calc	5827	5.80%
16: Glacial	1873	1.90%
17: MED, Lowland, MedLarg	2759	2.70%
18: MED, MidAlt, MedLarg	3849	3.80%
19: MED, Small, Peren	9812	9.80%
20: MED, Temp	1284	1.30%
total	100553	100%

2.4 Broad hydro regions

Next to the broad types, rivers have been classified according to their bio-geophysical type. Broad hydro regions were designated on European scale, based on expert knowledge from the WFD inter-calibration exercise (Poikane et al., 2014).

They are derived from the Natura 2000 biogeographical regions[6] as compiled, processed and published by EEA. The biogeographical regions dataset contains the official delineations used in the Habitats Directive (92/43/EEC) and for the EMERALD Network set up under the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention) (EEA, 2016).

There are five broad hydroregions (Figure 7): Nordic (NOR), Eastern Continental (EC), Alpine (ALP), Mediterranean (MED) and Central and Baltic (CB), the latest further divided into three hydro sub-regions: Atlantic (CB-ATL), Mediterranean (CB-MED) and Continental (CB-CON).

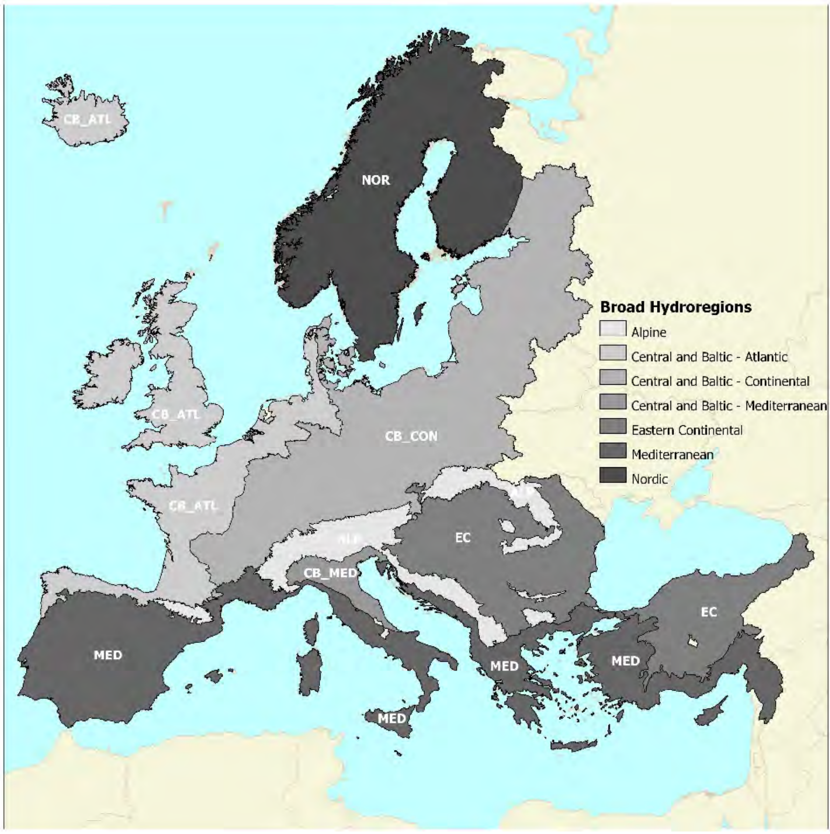

Figure 7: MARS broad hydroregions.

In order to create broad hydro regions, based on geographical intercalibration groups (GIG), EEA biogeographical regions were re-categorised as follows. The Atlantic and Continental biogeographical regions were re-categorised into hydro sub-regions and belong to one broad

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[6] See http://ec.europa.eu/environment/nature/natura2000/biogeog_regions/index_en.htm

hydro region named Central - Baltic (CB). South-Eastern part of the Boreal biogeographical region was merged with the Continental (CB-CON) hydro sub-region. The shares of the Atlantic and Alpine biogeographical regions in Norway were merged with the Boreal biogeographical region and named Nordic (NOR) broad hydro region. The Pannonian biogeographical region was merged with the Continental biogeographical region in the South-East Europe and named Eastern Continental (EC) hydro region. This region also contains remaining areas of the Eastern Turkey. The territory of the Po valley was assigned to the Central-Baltic (CB) broad hydro region and represents its Mediterranean hydro sub-region (CB-MED). All broad hydro regions were extended to cover the spatial model of the MARS project.

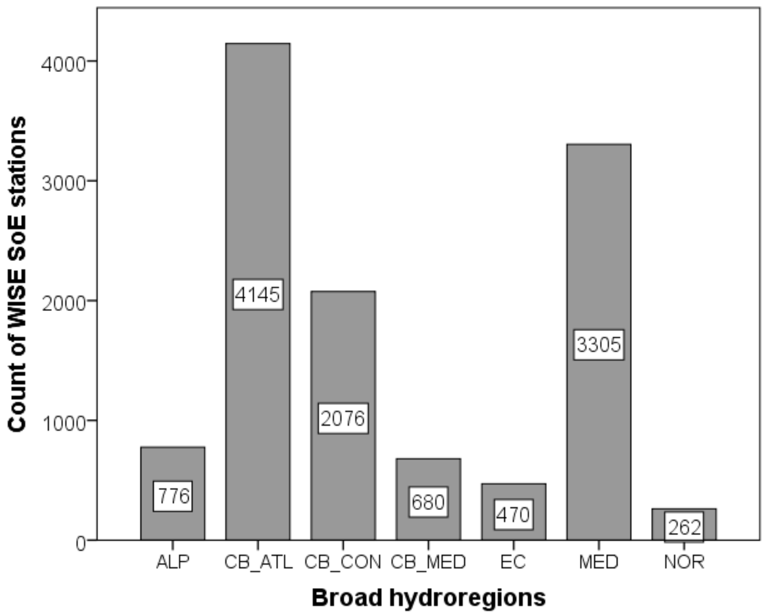


Figure 8: Count of SoE stations on rivers classified to different broad hydro regions.

Table 4: Density of SoE stations per 10000 km^2 of broad hydroregion

Region	**Subregion**	**Abbreviation**	**Area [km^2]**	**Number of SoE stations**	**Density [number of SoE stations per 1000 km^2]**
Alpine		ALP	375578	776	20.7
Central and Baltic	Atlantic	CB-ATL	877212	4145	47.3
Central and Baltic	Continental	CB-CON	1518795	2076	13.7
Central and Baltic	Mediterranean	CB-MED	89821	680	75.7
Eastern Continental		EC	899058	470	5.2
Mediterranean		MED	1210164	3305	27.3
Nordic		NOR	1152937	262	2.3

Although the areas of broad hydro regions are quite similar (Central and Baltic – Continental being the largest and Central and Baltic – Mediterranean being the smallest), SoE stations are very unequally distributed among them (Figure 8). The largest number of SoE stations (4145) is in Central Baltic – Atlantic hydro sub-region, and the second largest in Mediterranean hydro region. The least SoE stations is in Nordic hydro region (262), although it is the third largest hydro region. However, the density of SoE stations is the highest in the smallest hydro sub-region Central and Baltic – Mediterranean and amounts to 75.7 SoE stations per 10000 km^2 of region. On the other hand, the lowest density is in Nordic hydro region, namely, 2.3 SoE station per 10000 km^2 of region (Table 4).

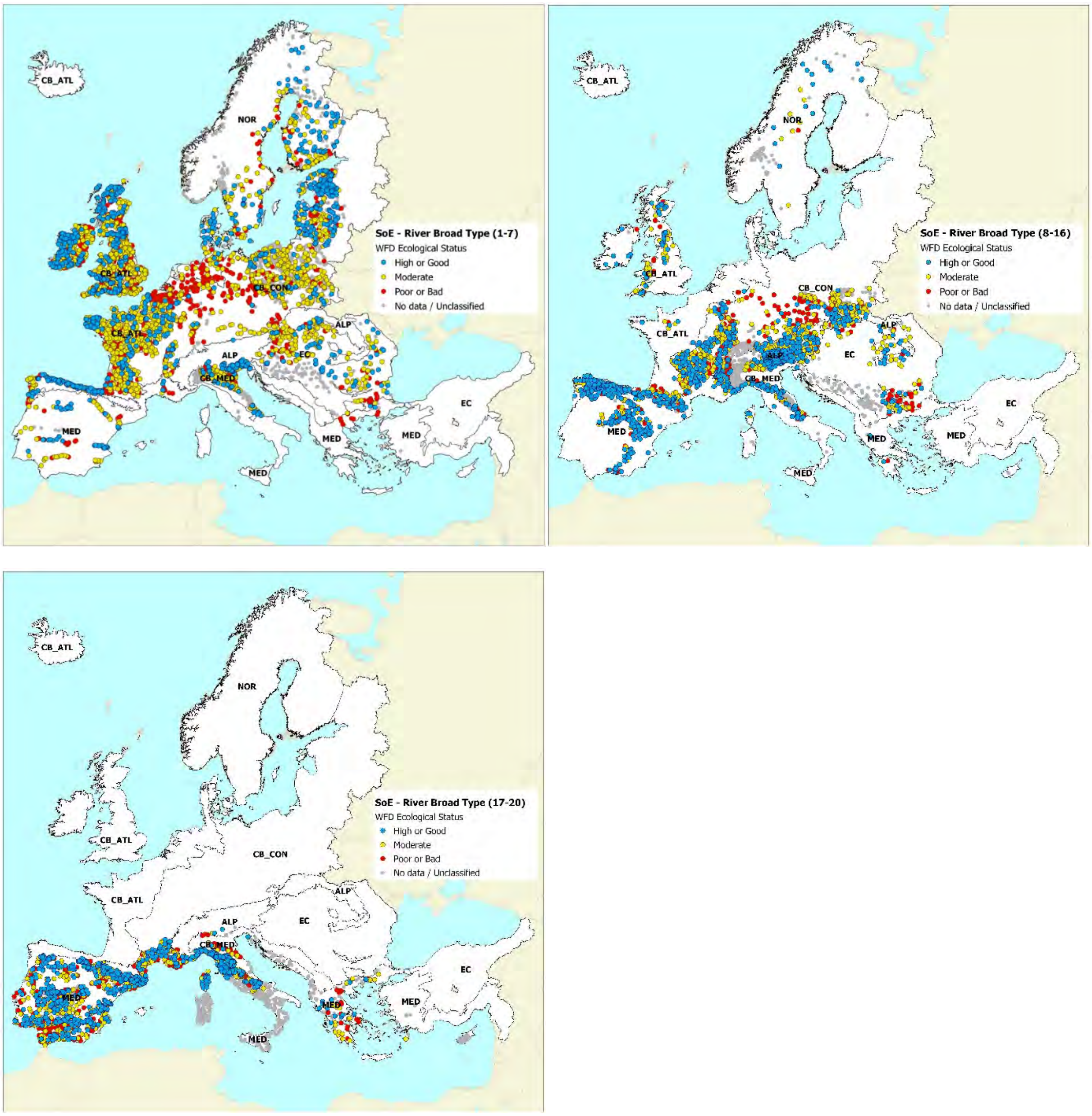


Figure 9: Distributions of SoE stations among broad hydroregions. Upper left: SoE stations on rivers with broad river type from 1 to 7 (lowland), upper right: SoE stations on rivers with broad river type from 8 to 16 (mid-altitude and highland), bottom: SoE stations on rivers with broad river type from 17 to 20 (Medditeranean). SoE stations are coloured by ecological status.

The largest number of SoE stations (9226) belong to lowland broad types, it means that they lie under 200 m above sea level. On highland and mid-altitude (broad river types from 8 to 16) there is 2724 SoE stations and almost the same number (2751 SoE stations) in Mediterranean region (broad river types from 17 to 20). As can be seen from Figure 9 and Table 5, SoE stations on lowland broad types have the smallest share of high or good ecological status, namely, 25 %, whereas SoE stations on mid-altitude and highland and SoE stations in Mediterranean region have 48 % and 44 % of good and high status respectively. Similarly SoE stations on lowland broad river types have the largest share of poor and bad ecological status (27 %). The smallest share have SoE stations on mid-altitude and highland (16 %).

Table 5: Ecological status of SoE stations by groups of broad river types

Broad river type	**Number of SoE stations**	**Percentage of SoE stations in each ecological status**		
		High or good	**Moderate**	**Poor or bad**
1-7 (lowland)	6226	25	48	27
8 - 16 (mid altitude and highland)	2724	48	36	16
17 - 20 (Mediterranean)	2751	44	35	21

2.5 Development of pressure indicators

From a datasets presented in Table 1 we prepared pressures indicators and various descriptive variables related to SoE stations. In the first step we considered all indicators and descriptive variables to be potentially relevant for the pressure – response analyses. Pressure indicators were grouped by type of information as described in Table 4. In the following the methods applied for determining indicator on hydrological alteration, diffuse pressure and proxies of hydromorphological alteration are described in detail.

In a subsequent second level quality check some pressure indicators and descriptive variables were gradually excluded due to incomplete spatial coverage, weak representativeness and redundancy. Concluding, we decided to only consider pressures for our analyses that can be managed. Information on altitude and slope were therefore excluded from the analysis. The final set of pressure indicators is a result of data exploration and screening by classification methods used in pressure – response analysis (explained in the following sub-chapter).

Table 6: Description of pressure indicators and other descriptive variables used in pressure – response analysis

Data Group	Description
general	river broad type (see chapter 2.3)
	broad hydroregion (see chapter 2.4)
	altitude in meters gradient (slope) of river in m/1000m
	WFD category: natural, artificial, heavy modified

	river fish assemblage type (FAT): HWS, MGR, LLR, MED (see Trautwein et al., 2011)
land use in FEC and hinterland	% of urban area (Corine Land Cover) in FEC
	% of agricultural area (Corine Land Cover) in FEC
	% of forest area (Corine Land Cover) in FEC
	% of urban area (Corine Land Cover) in hinterland
	% of agricultural area (Corine Land Cover) in hinterland
	% of forest area (Corine Land Cover) in hinterland
land use in buffer zone model 3 (long buffer upstream and downstream of WISE SoE location)	% of urban area (Corine Land Cover) in buffer zone
	% of agricultural area (Corine Land Cover) in buffer zone
	% of forest area (Corine Land Cover) in buffer zone
	% of broad-leaved forest area (Corine Land Cover) in buffer zone
	% of coniferous forest area (Corine Land Cover) in buffer zone
	% of transitional woodland and scrubs area (Corine Land Cover) in buffer zone
	% of forest area (Corine Land Cover) in buffer zone
	% of woodland and forest area (Corine Land Cover) in buffer zone
	% of woodland area (Corine Land Cover) in buffer zone
	% of natural grassland area (Corine Land Cover) in buffer zone
	% of area with little or no vegetation (Corine Land Cover) in buffer zone
hydrological alterations	alteration of mean annual flow
	alteration of base flow index
	alteration of low pulse threshold
	alteration of high pulse threshold
	alteration of extreme low flow duration
	alteration of high flow duration
	alteration of small floods duration
	alteration of high flow pulses
proxy for hydro-morphological alterations in potential riparian zone	% of natural MAES* habitats in potential riparian zone**
	% of natural water bodies in potential riparian zone
	% of actual riparian zone in potential riparian zone
	% of natural MAES habitats
proxy for hydro-morphological alterations in intermediate buffer, tested with fish metrics	share of MAES categories 3.4 – transitional woodland and scrub
	share of MAES categories 3.1 – broadleaved forest
	share of MAES categories 3.1.3 – broadleaved forest (T.C.D. > 30 - 50%)
	share of MAES categories 3.2 – coniferous forest
	share of MAES categories 3.2.1 - coniferous forest (T.C.D. > 80%)
	share of MAES categories 3 – woodland and forest
	share of MAES categories 9.2 – lakes and reservoirs
	share of MAES categories 4.2.2 – natural grassland without trees and scrubs
physico-chemical state of water	ammonium (mg/l N)
	biological oxygen demand (mg/l O_2)
	chemical oxygen demand (mg/l O_2)

	dissolved organic carbon (mg/l C)
	nitrate (mg/l N)
	orthophosphate (mg/l P)
	oxygen saturation (%)
	water temperature (°C)
	total organic carbon (mg/l C)
	total phosphorus (mg/l P)
diffuse pressures of nutrients: nitrogen and phosphorus input in FEC and hinterland	load (tonnes) of nitrogen per year in a FEC
	specific load (tonnes/km2) of nitrogen per year in a FEC
	load (tonnes) of phosphorus per year in a FEC
	specific load (tonnes/km^2) of phosphorus per year in a FEC
	load (tonnes) of nitrogen per year in hinterland
	specific load (tonnes/km^2) of nitrogen per year in hinterland
	load (tonnes) of phosphorus per year in hinterland
	specific load (tonnes/km^2) of phosphorus per year in hinterland
diffuse pressure of nutrients: degree of phosphorus saturation and N_surplus in agricultural area	degree of phosphorus saturation in agricultural soils in % in a FEC
	nitrogen surplus on agricultural areas (tonnes/km^2/yr) in a FEC
point pollution UWWTP	number of urban waste water treatment plants upstream 10 km
	number of urban waste water treatment plants upstream 50 km

* MAES ecosystems types are used under Target 2 of the EU 2020 Biodiversity Strategy and presents land use and land cover categories in Copernicus LU/LC data base (see explanation below); **Spatial layer in Copernicus LU/LC dataset (EEA, 215) and explained in Annex 4.

Indicators of hydrological alterations

Time series on daily run-off modelled with the global water balance model PCR-GLOBWB were provided by Deltares as gridded (spatial resolution: 5° minutes) accumulated run-off. NTUA allocated the outlet of a FEC to corresponding grid cells and derived daily time series on run-off under current conditions and for semi-natural conditions. Semi natural conditions mean that water abstraction and effects of reservoirs operations are eliminated. For more details see Annex 3.

Time series for a period 2001 – 2010 on both conditions were analysed using the IHA software (Nature Conservancy, 2009) providing statistical approach for identifying major hydrological parameters. Out of 67 parameters calculated with this software, eight are selected to calculate indicators of hydrological alterations for pressures- response analysis:

- Mean annual flow: average of all daily flows in period.
- Baseflow index: seven (7) consecutive days with a minimum flow in a year divided by a mean yearly flow.

- Low pulse threshold: flows lower than or equal to defined low flow threshold are a low flow events. The low flow threshold is defined as 50th percentile of all daily flows in selected period.
- High pulse threshold: flows greater than defined threshold are high flow events. The high flow threshold is defined as 75th percentile of daily flows in selected period.
- Extreme low flow duration: days with the flow less than or equal to 10th percentile of all daily low flows in selected period.
- High flow duration: days with discharges higher than 75th percentile of all daily flows in selected period Small floods duration: number of days with flow greater or equal to two (2) years return period flood and lower than 10 year return period flood.
- High flow pulses: a number of events in a year when a daily discharge (named “run-off” in PCR-GLOBEWB model) is greater than or less than a specified discharge threshold. High flow pulse is an event when discharge (named as “run-off” in PCR-GLOBWB model) reaches 75th percentile of all daily discharges in selected period.

Indicators of hydrological alterations are defined as ratio between model run under current conditions (with abstractions) and model run for semi-natural conditions (without abstractions) for each of hydrological parameters described above. If the indicator of hydrological alteration is higher than 1, it means that hydrological parameter for current conditions is higher than hydrological parameter for semi-natural conditions. If the indicator of hydrological alteration is lower than 1, it means that hydrological parameter for current conditions is lower than hydrological parameter for semi-natural conditions. If the indicator of hydrological alteration is exactly 1, it mean that hydrologic parameter for both conditions are the same.

Proxy for hydromorphological alterations

As a proxy for hydromorphological alterations Copernicus land cover / land use data (LC/LU), actual and potential riparian zones (RZ) were used (EEA, 2015). For land use assessment Copernicus LC/LU categories are used (Annex 4). All these three spatial datasets were overlaid with buffer zones around SoE stations to calculate different indicators of changes of hydro morphology. They are defined as the share of different land cover / land use categories in buffers zones. We used buffer zones of intermediate scale regarding Juen (2011). These are long strips along river at SoE locations, with width of 500 m to each side of ECRINS rive line, and length differing from 5 km to 25 km according to Strahler order of the river where SoE station is located: 5 km up- and downstream for Strahler order 3 and 4, 12.5 km for Strahler order 5 and 6 and 25 km for Strahler order 7, 8 and 9. Proxy for hydromorphologic alterations were calculated as share of all Copernicus LC/LU categories for all 4 levels. Afterwards those with the most significant response of EFI+ index were selected (in collaboration with BOKU).

All proxies for hydromorphologic alterations described above were calculated also from Corine Land Cover level 1 categories (artificial surfaces, agricultural areas, forest and semi-natural areas) and some higher level categories corresponding to Copernicus LC/LU categories, selected for final pressure-response analysis. Detailed description of applying riparian zone land cover / land use data can be found in Annex 4.

Indicators of diffuse nutrients pressure

Nutrient pressure data have been compiled from EUROSTAT (2016) and then further processed using two distinct methods. The first method use statistical data on fertilizer application and livestock excreta and method calculates total nitrogen and phosphorus inputs on agricultural land. As data in EUROSTAT are given per countries, we have disaggregated them to the FEC level, considering the share of agricultural land (i.e. arable land, permanent crops, pastures and heterogeneous agricultural areas as CLC categories). Afterwards we have recalculated N and P releases also per hinterland. Results are given as density (per km^2), as well as an absolute number per FEC and per hinterland. Detailed description of calculating total nitrogen and total phosphorus releases is given in Annex 5.

The second method calculates nitrogen balance and degree of phosphorus saturation with MONERIS model. N-balances were collated on national level for 41 countries in Europe and area specific values corrected by the areas of land in agricultural use given by Corine Land Cover (CLC) 2012 and Glob-Corine (ESA 2010 and UCLouvain). In countries for which no N-balance information was available, values from countries with assumed similar agro-economic conditions were used. Shares of organic and mineral fertilizer application as well as the share of atmospheric deposition on the N-balances were derived from the national fertilizer consumption statistics and the given total amount of N applied to agricultural land. Here, it was assumed that N fertilizers from these three considered sources are equally well and fast consumed by plants. This allows using the fertilizer application shares as N-balances shares. N-Balance shares originating from mineral fertilizer were corrected by the spatial distribution of summer precipitation (May to September).

The P-accumulation was calculated as the accumulative sum of P-balances over the years (1950-2014). As atmospheric deposition is less relevant as it is for nitrogen balances the P accumulation was distributed following the approach described above for nitrogen, without taking atmospheric deposition into account. DPS was estimated considering the soil type information and the bulk density by the LUCAS physical top soil information map (Ballabio, Panagos, & Monatanarella, 2016) and considering transformation function according to Pöthig, Behrendt, Opitz, & Furrer (2010). Detailed description on deriving nitrogen balances and degree of phosphorus saturation is given in Annex 6.

2.6 Method for multi pressure - response relationship analyses for rivers

We used empirical and distribution-free method for pressure - response relations analysis. Methods evolve within machine learning field. The two most commonly addressed tasks in machine learning are classification and regression. They are concerned with predicting the value of one field from the values of other fields. The target field is called the class (dependent variable in statistical terminology), we call it **"response variable".** The other fields are called attributes (independent variables in statistical terminology), we call it **"explanatory variables".** The common term **predictive modelling** refers to both classification and regression (Džeroski , 2008). Classification (if the response variable is categorical) and regression (if response variable is continuous) are an alternative to multiple regression methods, in that multiple independent (explanatory) variables are used to predict or explain a single response variable. Explanatory variable can as well be called "predictive variable".

Classification and regression approach

A classification or regression approach to learning from a set of independent variables leads to induction of classification or regression tree (CART), called also an induction of decision trees. There are various algorithms for the induction of decision trees, the most popular of which is the C4.5 (Quinlan, 1993). The J48 algorithm is a Java re-implementation of C4.5. The algorithms for the induction of decision trees are based on the Top-Down Induction of Decision Trees (TDIDT) principle (Quinlan, 1986, 1993). The algorithm repeatedly partitions the data set into subsets, as homogeneous as possible (in terms of number of cases) with respect to the response variable. Thus, the major task of the algorithm is to find the optimal splitting values of the measured variable and to give the most accurate prediction of the response variable. The principle follows two basic steps. The homogeneity of the subsets is defined by impurity, a measure with a value of 0 for completely homogeneous subsets, and higher values for less homogeneous subsets regarding the value of the target variable. Thus, the variable that splits the data into subsets with lowest impurity is selected as the **"best variable".** The split value is called **"decision boundary**". There are several measures to calculate impurity, most commonly used are the Information (entropy) index and the Gini index (Breiman et al., 1984; Quinlan, 1993; De'ath & Fabricius, 2000). Nevertheless, there is no best variable selection method (or variable ranking method) and the selection of ranking methods with different learning techniques, either statistical or entropy-based, may give quite different results for balanced accuracy of the model (Novaković et al, 2011).

The induction of decision tree is repeated until all examples are correctly classified. However, this can result in a big tree with many branches, which is difficult to interpret, and also causes the so-called problem of overfitting, that is, when tested on new data sets the model fails in its predictions. **Pruning** is a powerful technique to cope with tree complexity and overfitting. It improves the transparency of the induced trees by reducing their size, as well as enhances their classification accuracy by eliminating errors that are present due to noise in the data (Bratko, 1989).

After the tree is constructed from the training (learning) data set, it is necessary to assess the model quality, that is, the **accuracy of prediction**. This can be done by simulating the model on a testing data set and comparing the predicted values of the target with the actual values. The model accuracy is expressed as a percentage of correctly classified examples. Another option is to employ cross-validation, where a given (training) data set is partitioned into a chosen number of folds (N). In turn, each fold is used for testing, while the remaining folds are used for training. The final error is averaged error of all models.

We can induce a predictive model on a part of data set (called the training set) and evaluate the predictive performance of the learned model on the remaining part (on "unseen data"). This part is the testing or validation set. More reliable estimates of performance are obtained by using **cross-validation,** which partitions the entire data available into N (typically set to 10) subsets of roughly equal size. Each of these subsets is in turn used as a testing set, with all of the remaining data used as a training set. The performance figures for each of the testing sets are averaged to obtain an overall estimate of the performance of the learned model on unseen data.

Random forest

New methods based on fitting an ensemble of regression trees have improved the robustness of the method. Random forests (Breiman, 2001) as one of them are among the most popular machine learning methods. Random forest consists of a number of decision trees based on random selection of data and random selection of variables. Every node in the decision trees is a condition on a single variable, designed to split the dataset into two so that similar response values end up in the same set. There are two major beliefs about random forest (Breiman, 2001). First is that most of the tree can provide correct prediction of class for most part of the data. The second is that tree are making mistakes at different places, therefore we conduct voting for each data of the observation and then decide about the class of the observation. Based on poll results it is expected to be more close to the correct classification.

Random forest also provide two straightforward methods for feature selection: **average (mean) decrease** impurity and mean decrease accuracy. The measure based on which the (locally) optimal condition is chosen is called impurity. For classification, it is typically either Gini impurity or information gain/entropy and for regression trees it is variance. Thus when training a tree, it can be computed how much each variable decreases the weighted impurity in a tree. For a forest, the impurity decrease from each variable can be averaged and the variable are ranked according to this measure. The Gini impurity, like entropy measures how heterogeneous distributed some value is over a set. Decision trees chooses criteria to split a set into more homogenous subsets. Gini impurity or information gain/entropy are used to describe this difference that's being maximized. In the discrete case, if pi are the relative frequencies of values in a set, then entropy is - $\sum pi$ logpi and **Gini impurity is (1-** $\boldsymbol{\sum pi\, pi}$ **).** Another variable selection method, the mean decrease accuracy, is a direct measurement of an impact of each variable on accuracy of the model.

The general idea is to permute the values of each variable and measure how much the permutation decreases the accuracy of the model. This means that for unimportant variables, the permutation should have little to no effect on model accuracy, while permuting **important variables** should significantly decrease it. Random forest require little variable tuning, nevertheless, there might be traps in data interpretation. With correlated variables, strong ones can end up with low scores and the method can be biased towards correlated predictor variables (Strobl et al. 2008).

Lately, a gradient forest (GF) method, a novel approach in aquatic ecology have been applied to investigate biodiversity response to physical parameters by Ellis et al. (2012) and Wagenhoff et al. (2016). The method works on multiple response variables, builds random forest models for each and uses cross-validation accuracy prediction model. Wagenhoff et al. (2016) derived macroinvertebrate, periphyton and bacterial assemblages thresholds as response to nutrient and fine sediments effects. Investigation of biological response with the gradient forest method as developed by Ellis et al. (2012) uses three measures: the goodness-of–fit measure R2 (proportion of the variance of response variable explained by the RF model as derived), accuracy importance of explanatory variable and the "raw" importance of an explanatory variable at a split (or decision boundary). The analogue of later is the decrease of Gini impurity. Ellis et al. (2016) points out that assessment of the importance of an explanatory variable with the gradient forest method can be viewed as a supplement to other statistical method commonly used (variance partitioning, correlation measures, Akaike weighting etc.) and show where along the gradient of explanatory variable values important changes occur.

Accuracy of the predictive model

There are many statistical measures available for evaluation of model accuracy. Classification accuracy and presentation of the modelled (predicted) results of a response variable with a confusion matrix (contingency table) are straightforward. The error between the actual and the modelled response values can also be calculated by following measures: precision (a measure of exactness), recall (the number of positive predictions divided by the number of positive response values; also called sensitivity), F1 score (or F measure that conveys the balance between the precision and the recall), root mean-squared error, mean absolute error, root relative squared error, relative absolute error, correlation coefficient, kappa statistics etc. The (Cohen's) kappa statistics that tests reliability across multiple explanatory variables also corrects agreement that occurs by chance and perform well with unbalanced data of a response variable.

Method used for classification of pressures on European river broad types

We determine important pressures that effect ecological status of European rivers with the random forest (RF) classification method. It is a part of the machine-learning package WEKA (Witten, Frank & Hall, 2011). Classification is done with 15 trees with depth of four branches and pruning with 5, 10 or 15 items (5 in small data sets and 15 in large). We use 10-fold cross-validation. The result satisfy condition that an interrater reliability as given by Kappa statistic is larger than 0.2 (0.2-0.6 present fair to good agreement) and average classifying accuracy is larger than 60%.

In the first step we worked with all pressure indicators and descriptive variables listed in Table 6. We tested performance accuracy and explored data with many classification algorithms (e.c. CART, regression rules) and variable ranking methods (Information gain, Gain ratio and RelieF evaluation) included in WEKA package. Result supported our decision to classify pressures by river broad types, broad hydroregions and use data only relevant for natural water bodies. The RF chose catchment land uses, namely share of urban and agricultural, as the "best" explanatory variable for classification of pressure. Model accuracy was usually higher than 85%. Nevertheless, we excluded them due to assumption that only "manageable" pressure should be selected as explanatory variable for classification. At the end, 21 pressure indicators (explanatory variables) were selected for the final classification process (Table 7). For river broad type 2 we added five (5) physical parameters.

We use data on ecological status as a response variable and re-classify five ecological classes into two classes. Good and high ecological statuses are in first class and the rest in the second ("moderate", "bad" and "poor" ecological class). Ecological status represents conditions of rivers for the period 2005-2010 and has been reported by EU Member States in 1st River management Plan (1st RBMP).

We calculated importance of explanatory variables with two methods, mean impurity decrease and Gini index. The first is an entropy based method as explained above and is integrated in WEKA software. Gini indexes are calculated for explanatory variables that were selected after interpretation of all regression trees in a single model. Since trees are built with pruning (3 depth and minimum 5-20 objects in a leaf), it is possible rather easy to observe combinations of "best" explanatory variables and corresponding decision boundaries that lead to final decision on class. In spite of the fact that random forest builds trees by chance, some variables appear more frequently and in various combinations. Furthermore, corresponding decision boundaries are very similar if not the same in all combinations. When observing final node of a tree, we get information on precision (or recall), what gives additional criteria for selection of explanatory variables and decision boundaries. Gini index of a variable was calculated in relation to its decision boundary. Variables with the smallest Gini index are then determined as the most important, since they have the largest explanatory power. Consequently, they are the most important pressures for distinct river type. We argue that combined or single pressure, as long as they not exceed given threshold value (decision boundary from a classification model), most likely do not cause deterioration of ecological status. Furthermore, if the value exceeded, there is a high chance for not achieving good ecological status. In the case of proxy hydromorphological indicators that do not represent pressure (and causing deterioration of ecological status at increase), but rather a support (e.g. percentage of natural vegetation in flood area), threshold values should be exceeded to prevent deterioration of ecological status.

For each river type, we use decision boundaries of four most important pressures as threshold values. For each FEC we tested real values of important pressures against their threshold and counted number of pressures that may, individually or in combination, cause deterioration of ecological status. Situation when more than one pressure may cause deterioration of ecological status is characterised as multi-pressures situation. We detected all such FECs in Europe and produced European multi-pressure maps.

Table 7: List of pressure indicators used in pressure - response analysis (explanatory variables in classification process).

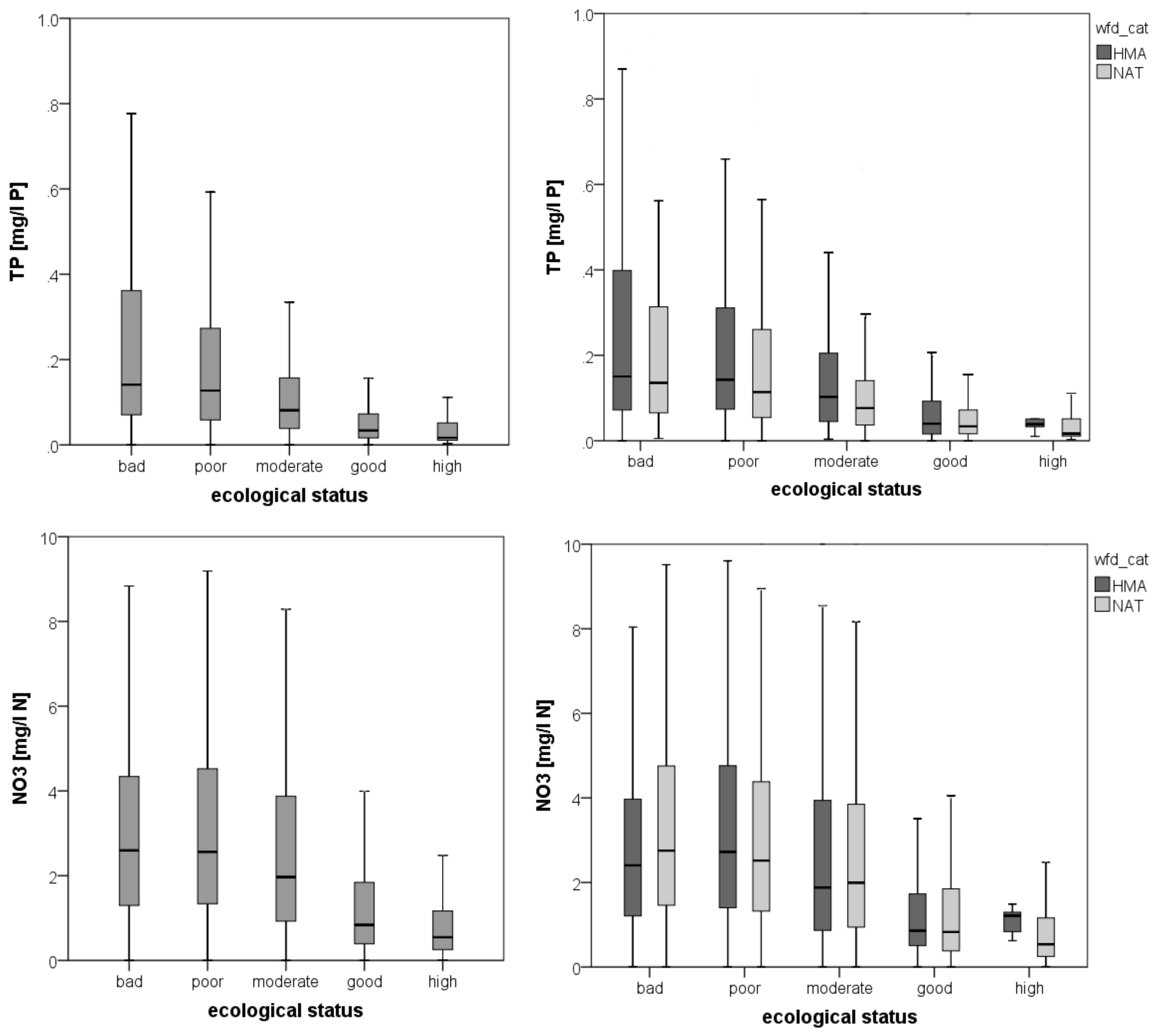
Label	Deskription
URB	% of urban area in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (Corine Land Cover)
AGR	% of agricultural area in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (Corine Land Cover)
FT	% of transitional forest area in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (Copernicus LC/LU)
FB	% of broad leaved forest area in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (Copernicus LC/LU)
FC	% of coniferous forest area in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (Copernicus LC/LU)
WF	% of forest and woodland (transitional forest and scrub) in floodplain (survey area is buffer 5 to 25 km long and 1000 km wide) –
GRS	% of natural grassland in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (Copernicus LC/LU)
LAK	% of lakes and water surface area in reservoirs in floodplain (survey area is buffer 5 to 25 km long and 1 km wide) (ECRINS lakes)
BaseF	alteration of base flow index (PCR-GLOBWB, IHA)
HLT	alteration of low pulse threshold (PCR-GLOBWB, IHA)
HHT	alteration of high pulse threshold (PCR-GLOBWB, IHA)
HLD	alteration of extreme low flow duration (PCR-GLOBWB, IHA)
HHD	alteration of high flow duration (PCR-GLOBWB, IHA)
HSFL	alteration of small flood duration (PCR-GLOBWB, IHA)
HHP	alteration of high flow pulses (PCR-GLOBWB, IHA)
BOD5	bod5 - biochemical oxygen demand in 5 days (mg/l O2); average for period 2005 – 2010 (WISE SOE)
NO3	nitrate (mg/l N), average for period 2005 – 2010 (WISE SOE)
PO4	orthophosphates (mg/l P), average for period 2005 – 2010 (WISE SOE)
TP	total phosphorus (mg/l P), average for period 2005 – 2010 (WISE SOE)
NH4	ammonium (mg/l N), average for period 2005 – 2010 (WISE SOE)
SPM*	suspended particulate matter (mg/l), average for period 2005 - 2010 (WISE SoE)
TMP*	water temperature (°C), average for period 2005 – 2010 (WISE SoE)
TOC*	total organic carbon (mg/l C), average for period 2005 – 2010 (WISE SoE)
DO*	dissolved oxygen (mg/l O2), average for period 2005 – 2010 (WISE SoE)
EC*	electrical conductivity (μS/cm), average for period 2005 – 2010 (WISE SoE)
OS*	oxygen saturation (%),average for period 2005 – 2010 (WISE SoE)
NBAL	nitrogen surplus per FEC in tonne per year per km^2 of FEC (MONERIS)
PSAT	degree of phosphorus saturation on agricultural land (%) in FEC (MONERIS)

*Only used for classification of river broad type 2.

3 Results and discussion

3.1 Physico-chemical parameters in rivers at SoE locations

Figure 10 shows the relationship between different chemical parameters in rivers and the reported ecological status for all SoE stations and separately for heavily modified water bodies (HMA) and natural water bodies (NAT). For all studied chemical parameters the mean concentration declined with improving ecological status, which is more obvious if we consider only SoE stations on natural water bodies. Further, for the same ecological status mean parameter concentrations in natural water bodies were lower than in corresponding heavily modified water bodies. However, the variability in concentrations of all parameters was high and differences between adjoining status classes were not significant. Because of significant differences between HMA and NAT sites for most parameters and because NAT sites were showing better response to ecological status, only SoE stations on natural WBs were taken into further pressure – response analysis.



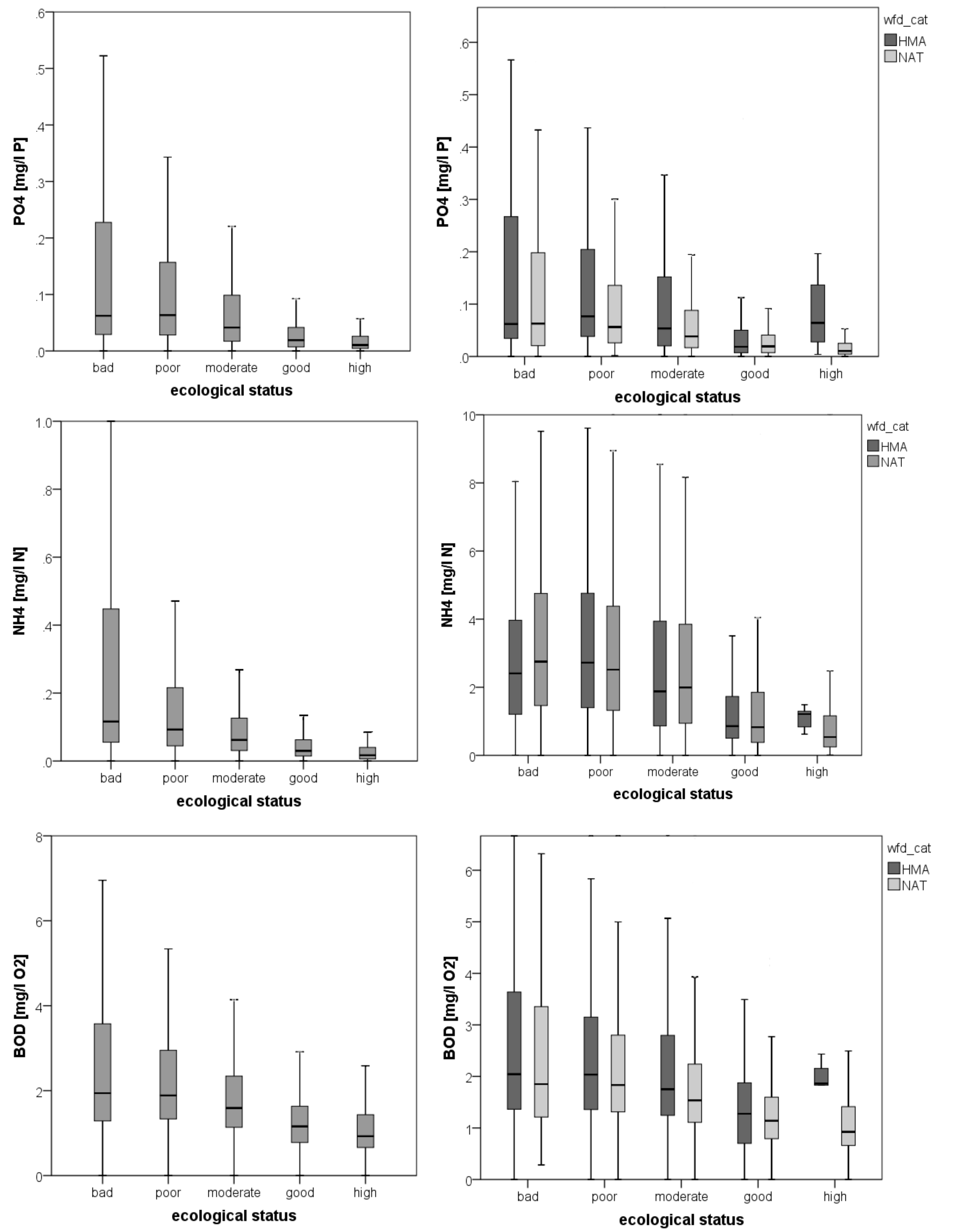
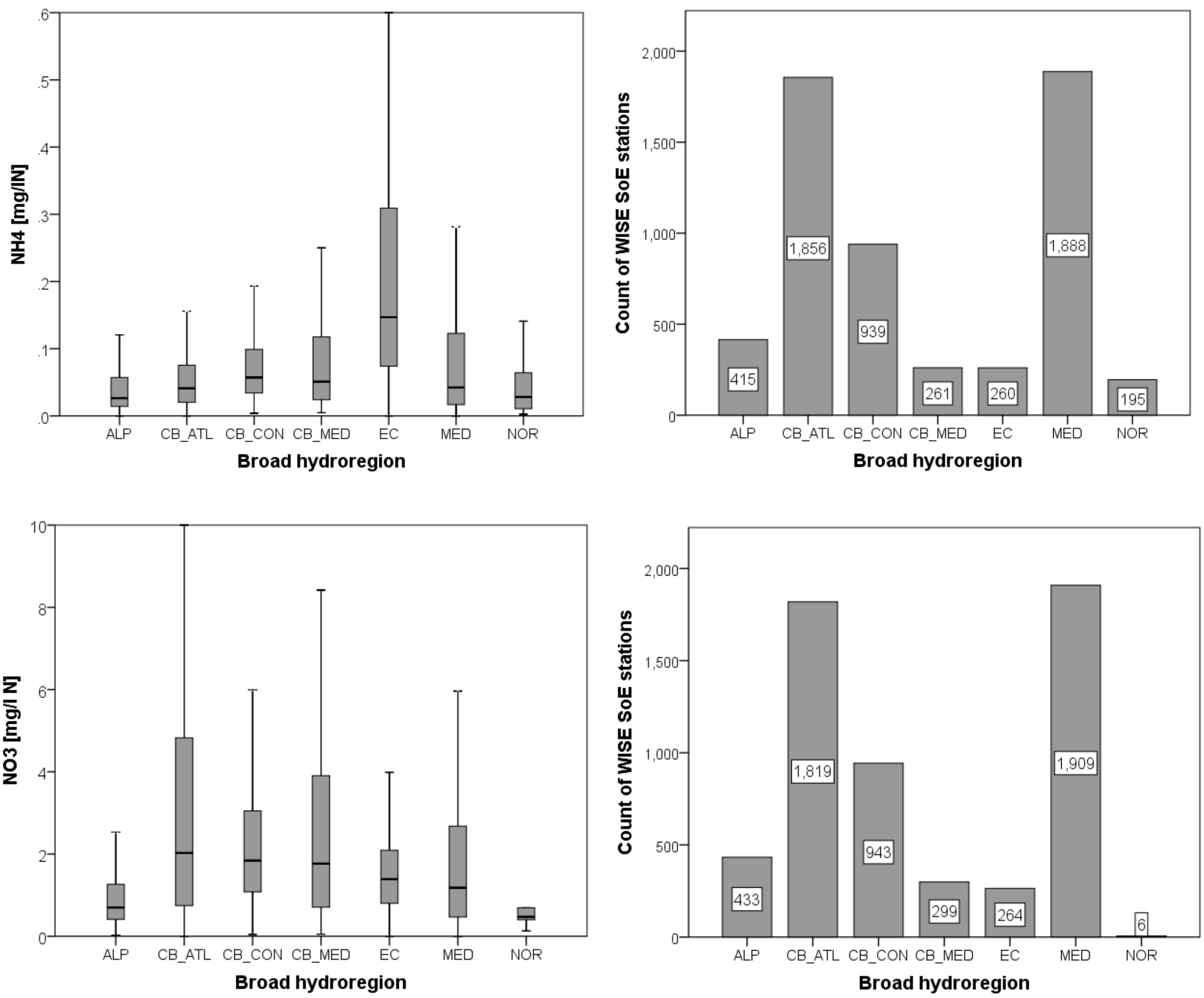


Figure 10: Distribution of values of different physico-chemical parameters on WISE SoE stations in regard to ecological status. On the left side there are graphs with all WISE SoE stations included, whereas on the right side there are WISE SoE stations separated into those located on heavily modified or artificial river reaches from those on natural river reaches. In order to improve transparency of graphs outliers are not shown.

There were significant differences in observed values of different physico-chemical parameters between some broad hydro-regions (Figure 11) and also between broad river types (results not shown). Eastern continental region stands out in high values in ammonium (NH4), total phosphorus (TP) and biological oxygen demand (BOD5). Lowest concentrations of all selected chemical parameters are expectedly observed at Alpine and Nordic region, what reflects low degree of urban and agricultural area in these regions.



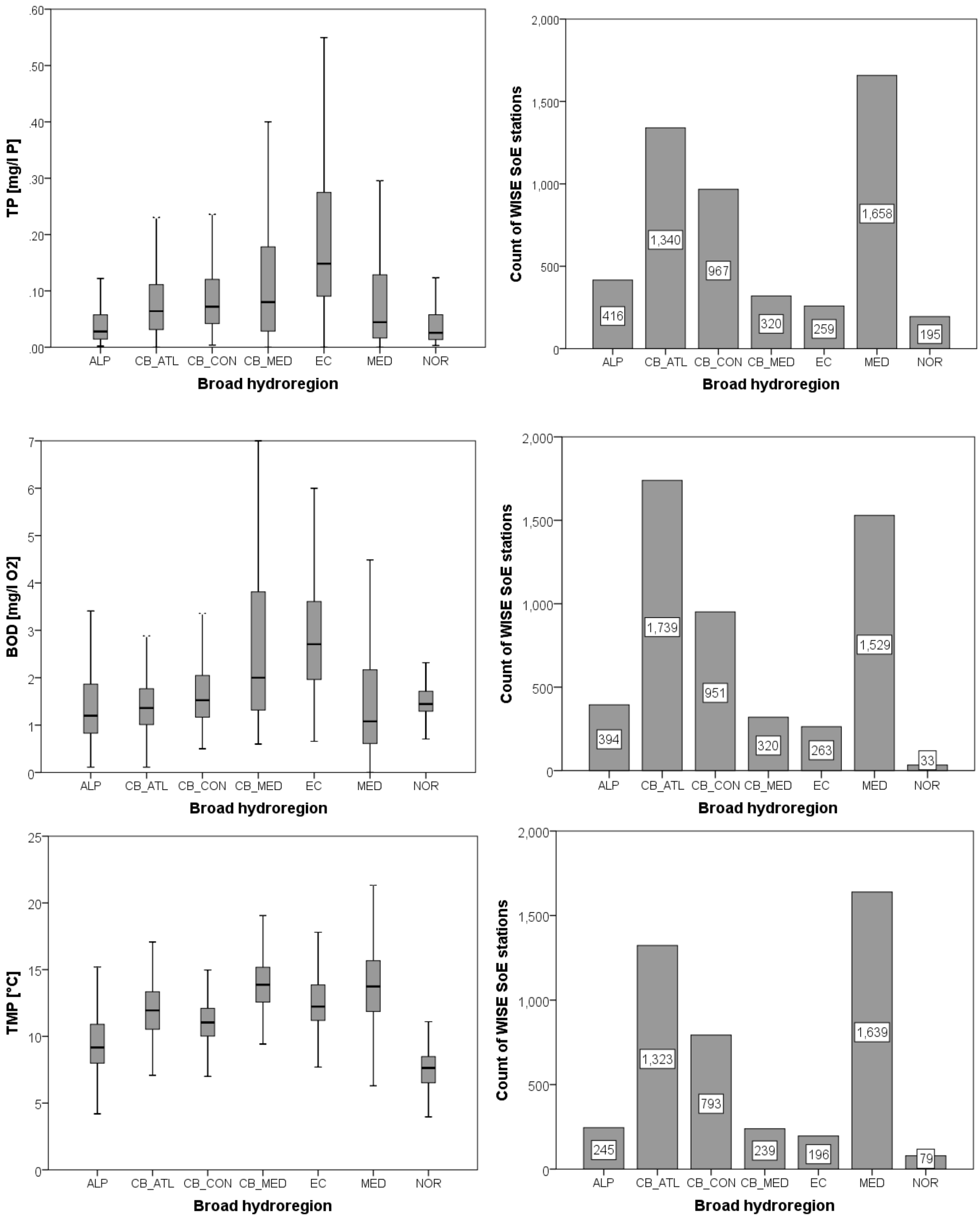


Figure 11: Distribution of values of different physico-chemical parameters on WISE SoE stations in regard to broad hydro regions. Note that WISE SoE station with data on selected chemical parameter are unevenly distributed among broad hydro regions (graphs on the right side). In order to improve transparency of graphs outliers are not shown.

3.2 Basic statistics of pressure indicators

A standard statistical analysis was conducted for all pressure indicators, belonging to three groups:

- Hydrological alterations of flow in a main drain of each FEC: it is a rate of change between current conditions (altered flow with abstractions upstream) and flow without abstractions in each FEC (semi-natural conditions);
- Proxy for hydro-morphological alterations are shown as percentage of selected riparian land cover in floodplain within each FEC;
- Yearly nitrogen surplus (in tonnes N per km^2) and degree of phosphorous saturation (in %) as modelled by MONERIS and presented as mean for each FEC;
- Physico-chemical state of water.

Floodplain areas associated with SoE stations (in total 11,714) where in average covered to 55 % by agricultural land and to 10 % by urban areas. Forest and woodland (transitional forest and scrub) cover 26 % of the floodplains, but only 3.4 % are grasslands and 2.6 % reservoirs or lakes. Five percent of floodplains along SoE stations have less than 3 % of forest and woodland or more than 65 % (5 and 95 percentile, respectively).

In average, all indicators of hydrological alteration (ratios between current and semi-natural hydrological flow conditions) have mean value above one, i.e. hydrological parameter for current conditions have larger values than for semi-natural conditions. However, this statistical result cannot be generalized, as the variance of observed ratios is large and individual site conditions have to be considered. Hydrological alteration indicators (ratios of hydrological parameters between current and semi-natural conditions) cannot be below 0, but are not limited upwards. An increase of a hydrological parameters (of current conditions in comparison to semi-natural conditions) can be larger than 10 folds, meanwhile a decrease ranges from 0 to 0.99, and therefore the average does not show the real picture. When looking at flow (discharges), it has decreased on more stations compared to stations with an increase. Mean annual flow (not shown in here) has increased on 3,473 stations (30 %) and decreased on 4,846 (41 %) stations. Low and high flow thresholds have decreased on 4380 and 4343 (37 %) stations and increased on 3790 (32 %) and 4060 (35 %) stations respectively. Even larger decreases than increases of hydrological alterations are observed on European territory (see next chapter). In general, we can observe large alteration of natural flows through the Europe.

The average value of BOD5, indicating the level of oxygen demanding substance in water, is 57.5 mg O_2 /l (with median 2.14). Average nitrate and orthophosphate concentration are 63.3 mg N/l (with median 2.48) and 64.6 mg P/l (with median 0.11). The average concentration of total phosphorus is 64.6 mg P/l (with median 0.25). All values are positively skewed as indicated by the median.

Nitrogen balance (surplus) in FEC area with SoE station is in average 53.8 kg/ ha/ year, and degree of phosphorus saturation in agricultural soil a little less than 63 %.

A lumped statistical description of all pressure indicators used for the pressure – response analysis derived for the SoE stations is given in Table 8. The results of the statistical analysis according to broad river types are collected in Annex 7.

Table 8. Basic statistics of pressure indicators (explanatory variables in classification process) for analysed SoE stations (in total 11714).

Label of pressure indicator	Data availability [% of SoE]	Mean	Median	Std. Deviation	Variance	Skewness	Percentiles	
							5	95
URB [%]	89.0	10.4	5.4	14.0	196.3	2.43	0.0	38.5
AGR [%]	89.0	54.8	57.3	26.7	713.5	-0.27	7.6	93.7
FT [%]	66.4	3.0	1.2	4.9	24.2	3.81	0.0	11.9
FB [%]	88.9	13.7	9.0	13.4	179.9	1.66	0.1	42.1
FC [%]	88.9	5.4	0.9	10.1	101.8	2.92	0.0	27.8
WF [%]	88.9	26.0	20.4	20.0	398.6	0.89	2.5	65.7
GRS [%]	66.4	3.4	1.2	5.9	35.2	3.83	0.0	15.0
LAK [%]	100.0	2.6	0.2	8.6	74.1	5.17	0.0	15.8
BaseF [-]	92.3	1.01	1.00	0.12	0.01	-1.61	0.91	1.19
HLT [-]	91.0	1.87	1.00	24.11	581.19	80.56	0.68	2.40
HHT [-]	92.1	1.43	1.00	10.45	109.10	53.00	0.82	1.38
HLD [-]	92.3	1.13	1.00	1.77	3.12	21.04	0.65	1.58
HHD [-]	91.1	1.15	1.00	2.73	7.45	23.00	0.75	1.25
HSFL [-]	92.3	1.02	1.00	0.15	0.02	0.53	0.91	1.24
HHP [-]	92.2	1.04	1.00	5.79	33.47	103.51	0.63	1.23
BOD5 [mg/l O2]	57.5	2.14	1.47	4.07	16.56	16.39	0.34	5.01
NO3 [mg/l N]	63.3	2.48	1.58	2.83	7.99	4.49	0.17	7.58
PO4 [mg/l P]	64.6	0.11	0.03	0.27	0.07	7.96	0.00	0.42
TP [mg/l P]	56.2	1.74	0.07	25.27	638.49	24.48	0.01	0.66
NH4 [mg/l N]	64.4	0.25	0.05	1.13	1.28	14.50	0.01	0.80
SPM [mg/l]	3.2	24.7	7.0	109.5	11985.1	17.3	0.5	91.3
TMP [°C]	48.9	13.0	12.4	9.7	93.7	36.80	8.2	17.8
TOC [mg/l C]	22.4	33.41	4.42	367.45	135019.64	16.26	1.10	17.64
DO [mg/l O2]	20.6	10.45	10.60	2.78	7.72	33.76	7.98	12.07
EC [µS/cm]	17.1	447.6	362.9	686.2	470882.4	15.5	53.5	964.2
OS [%]	60.0	92.7	94.9	12.7	161.6	-1.86	71.4	107.2
NBAL [tonne/year/km^2]	97.9	5.38	4.50	3.30	10.90	3.60	2.16	11.10
PSAT [%]	86.2	62.66	64.34	11.61	134.76	-1.63	46.44	77.08

3.3 Spatial distribution of pressure indicators

Values of all analysed pressure indicators are mapped on FEC level.

The upper map of Figure 12 hydrological alteration of mean annual flow are shown in three categories: increase, decrease and no changes in mean annual flow. The overall data coverage accounts for 98 % of FECs. Out of these FECs, 41 % no significant changes in mean annual flow was found, 25 % show an increase and 34 % a decrease of mean annual flow. Increases are observed mainly along larger rivers, whereas decreases are observed at their tributaries. Map on the left bottom side of Figure 12 shows only FECs with a decrease in mean annual flow, coloured by intensity of decrease. Furthermore, right map shows only FECs with increase in mean annual flow, coloured by intensity of increase. 3.5 % of FECs show an increase of mean annual flow for more than 50 % regarding to semi-natural conditions, while 1.2 % of FECs show decrease for more than 50 %.

Figure 13 shows the hydrological alteration of the base flow index as ratio between current conditions (with water abstractions) and semi-natural conditions (without water abstractions). The overall data coverage accounts for 98 % of FECs. Out of these FECs, 54 % have no changes in baseflow index, 25 % show an increase and 21 % a decrease of base flow index. In general, for the base flow index differences between current conditions and semi-natural conditions are small compared to the other hydrological indicators. Further, in contrast to alterations of mean annual flows, no difference was observed for large and small rivers. As a consequence, for more than half of the FECs no changes in baseflow index were found and the shares of FECs with more than 50 % decrease (0.6 % of FECs) or 50 % increase (0.8 % of FECs) of base flow index are low.

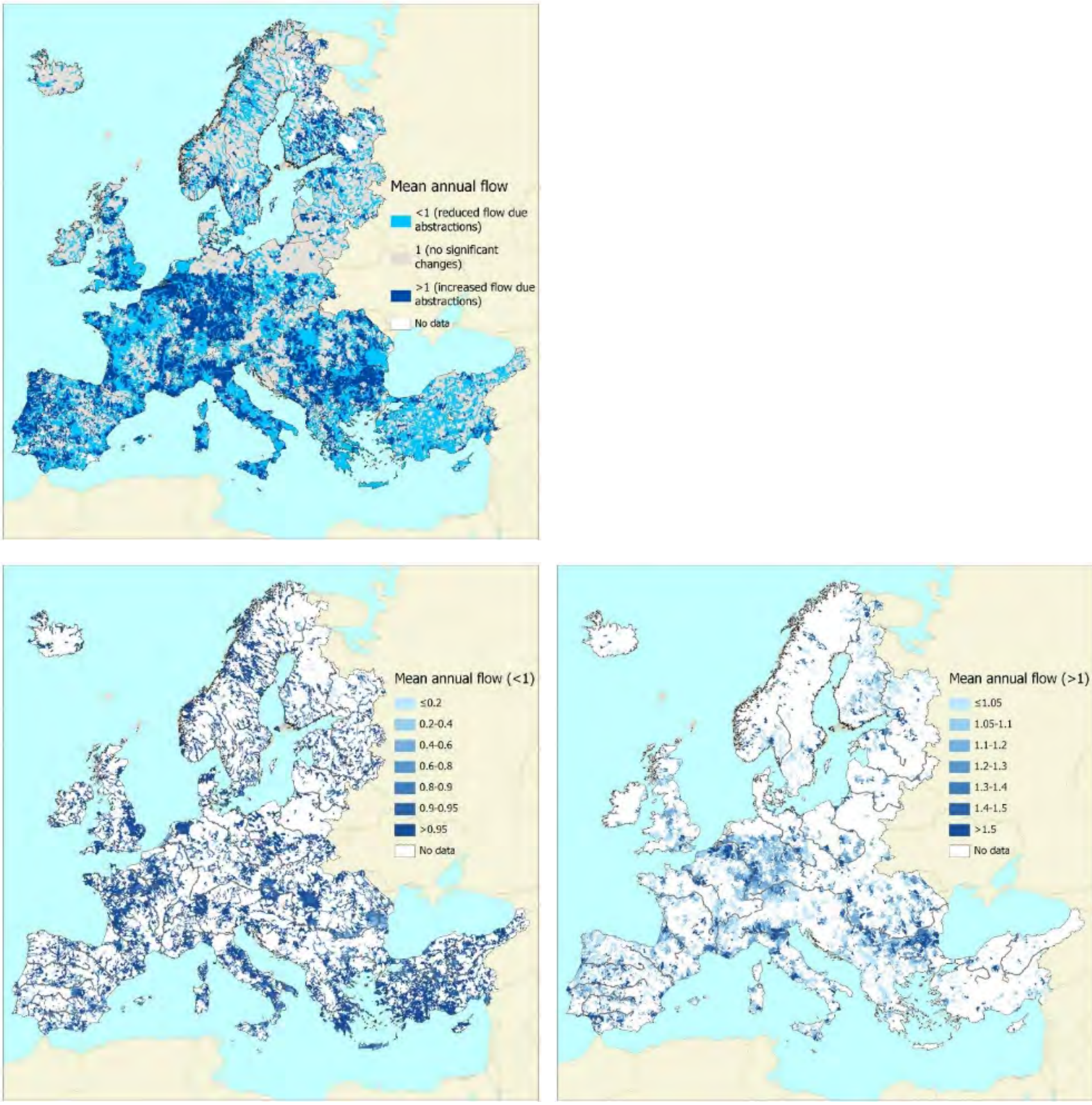


Figure 12: Mean annual flow rate of change across Europe. Left upper map shows both, a decrease (rate smaller than 1) and increase (rate larger than 1) of mean annual flow due to abstraction. The map on the bottom shows the intensity of decrease (left) and of increase (right).

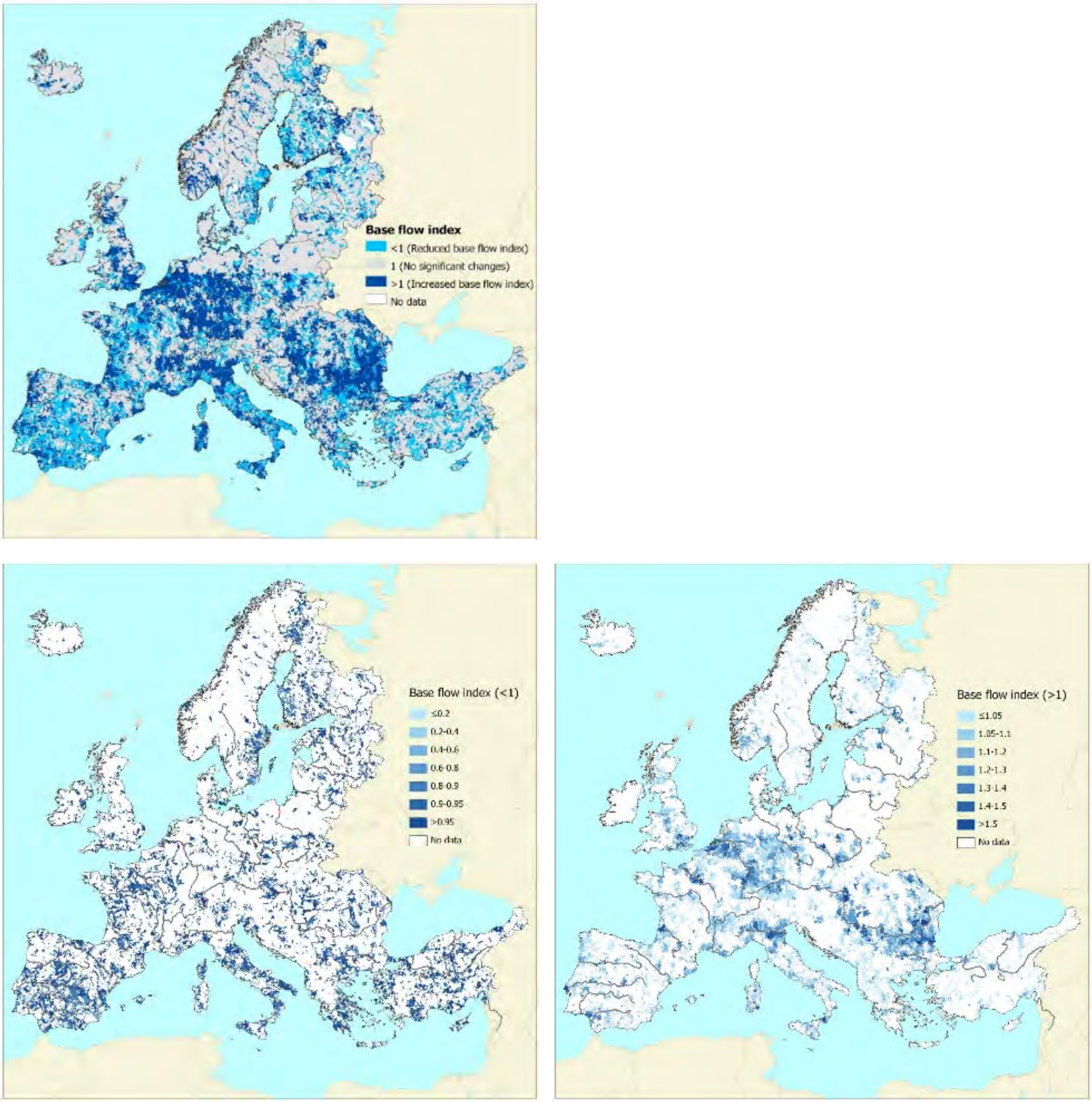


Figure 13: Ratio in change of the base flow index across Europe under current and semi-natural conditions. The upper map shows both, the decrease (ratio smaller than 1) and increase (ratio larger than 1) of base flow index. The left bottom map shows the intensity of decrease (left) of increase (right) by categories.

Hydrological alteration in extreme low flow duration between current conditions (with water abstractions) and semi-natural conditions (without water abstractions) are presented in Figure 14. The overall data coverage accounts for 98 % of FECs. Out of these FECs, 49 % have no changes in extreme low flow duration. Although the share of FECs with increased extreme low flow duration (27 %) is lower than share of FECs with decreases in mean annual flow (34 %), the share of FECs with large increase (> 50 %) of extreme low flow duration is considerably larger (5.6 %)

than large increase in mean annual flow. A decrease of extreme low flow duration by more than 50 % was found for 2.4 % of the FECs.

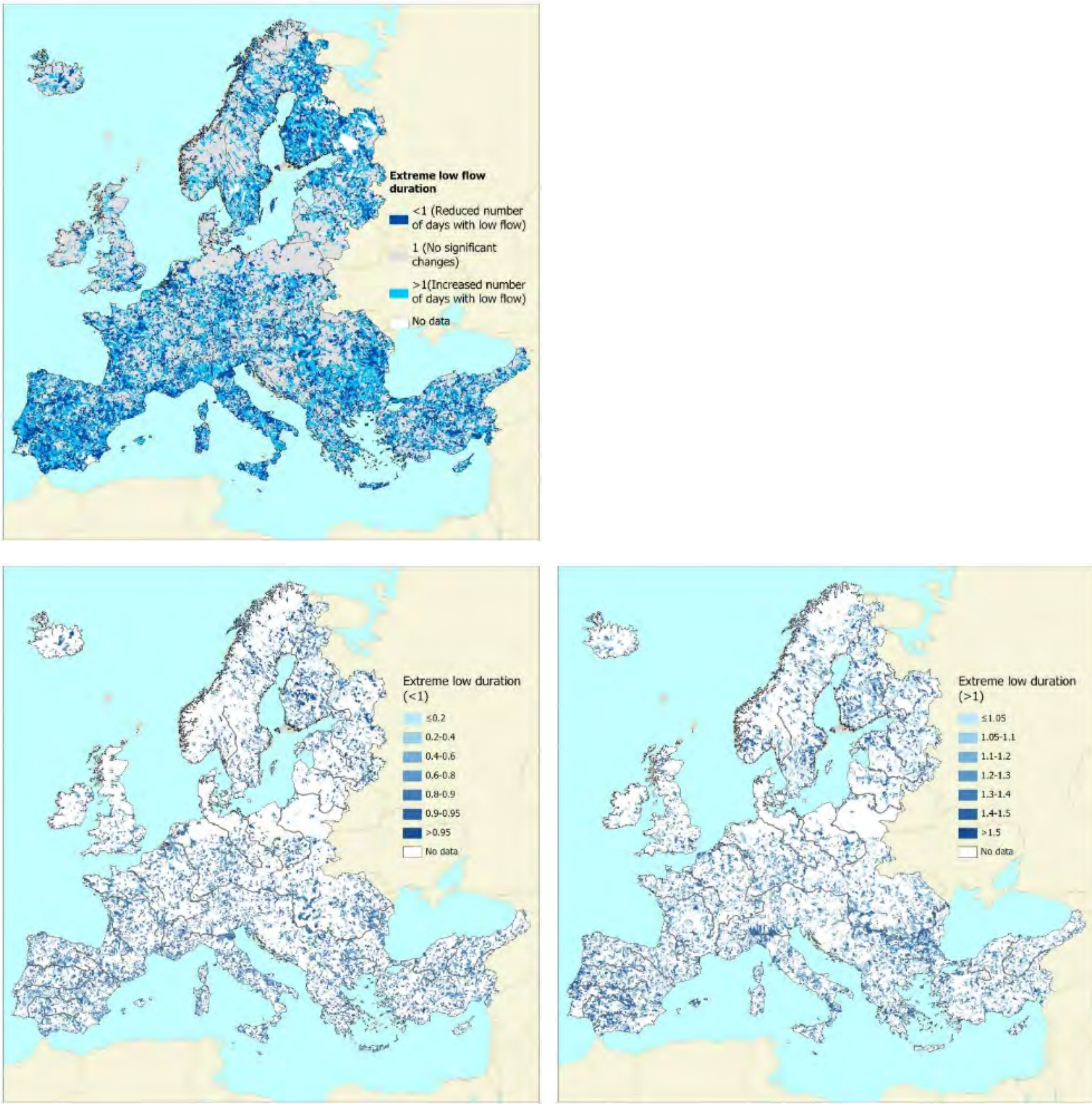


Figure 14: Extreme low flow duration rate of change across Europe. The upper map shows both, a decrease (rate smaller than 1) and increase (rate larger than 1) of extreme low flow duration due abstraction. The lower maps show the intensity of decrease (left) and the intensity of increase (right).

More than 40 % of FECs s have no changes in high pulse thresholds (Figure 15), 29 % show an increase and 30 % a decrease of high pulse thresholds. Similarly to the mean annual flow, increases are observed mainly along larger rivers, whereas decreases are observed at their tributaries (Figure 15). Even the share of FEC with large increases and large decreases in high pulse thresholds is similar to those observed at mean annual flow: 3.8 % of FECs show an increase by more than 50 % compared to semi-natural conditions, while 1.2 % of FECs show a decrease by more than 50 %.

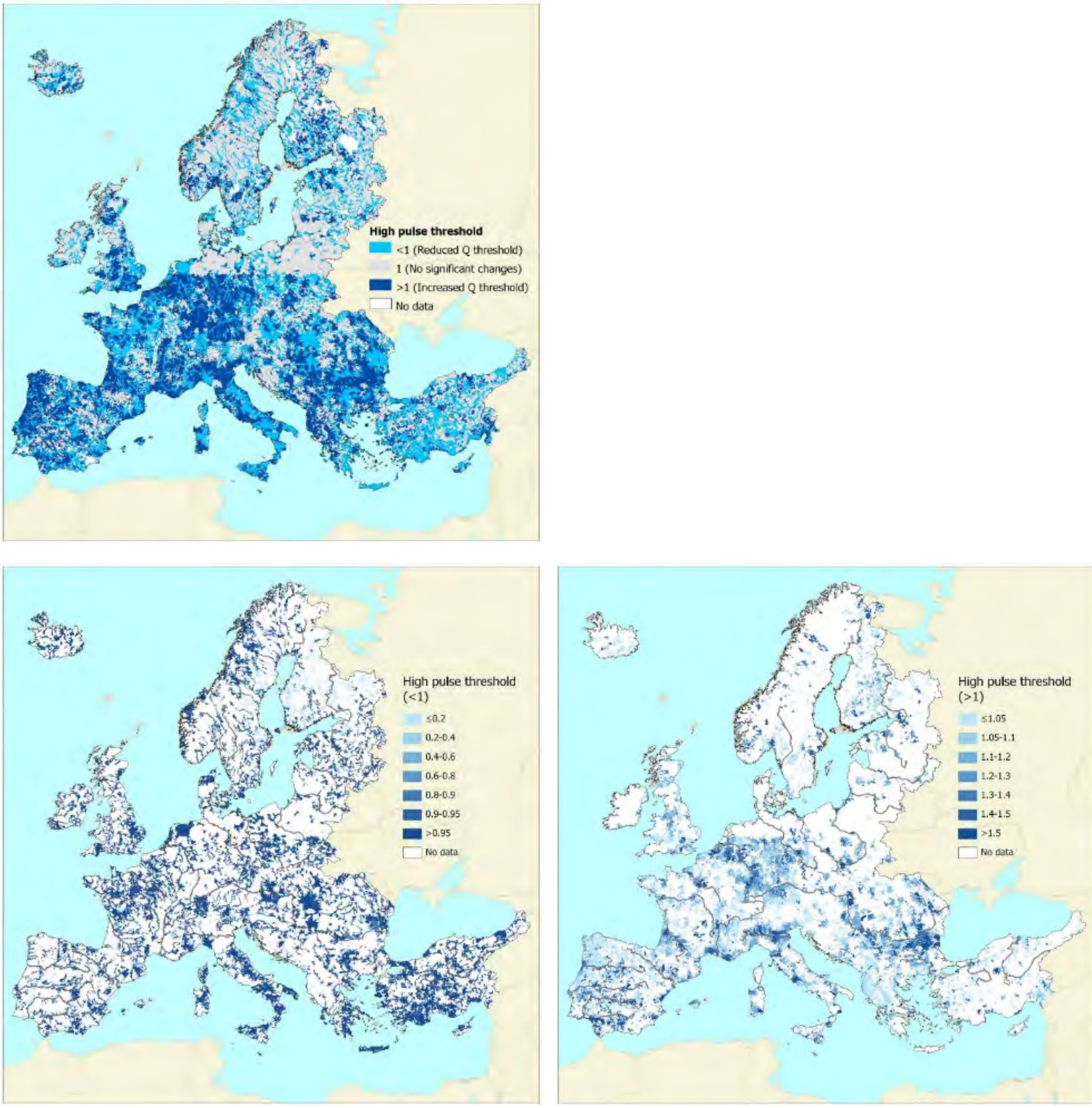


Figure 15: High pulse threshold rate of change across Europe. Upper map shows both, a decrease (rate smaller than 1) and increase (rate larger than 1) of high pulse threshold due abstraction. The map on the bottom show the intensity of decrease (left) and the intensity of increase (right) of a high pulse thresholds.

Figure 16 shows hydrological alterations in high flow pulses between current conditions (with water abstractions) and semi-natural conditions (without water abstractions). In comparison to indicators of hydrological alteration described before, for a significantly larger share of FECs changes in high flow pulses between current and semi-natural conditions were derived. Only 19 %

of FECs have no changes in high flow pulses, whereas 42 % show an increase and 39 % a decrease of high flow pulses. 1.7 % of FECs show an increase of high flow pulses for more than 50 % regarding to semi-natural conditions, while 3.3 % of FECs show decrease for more than 50 %.

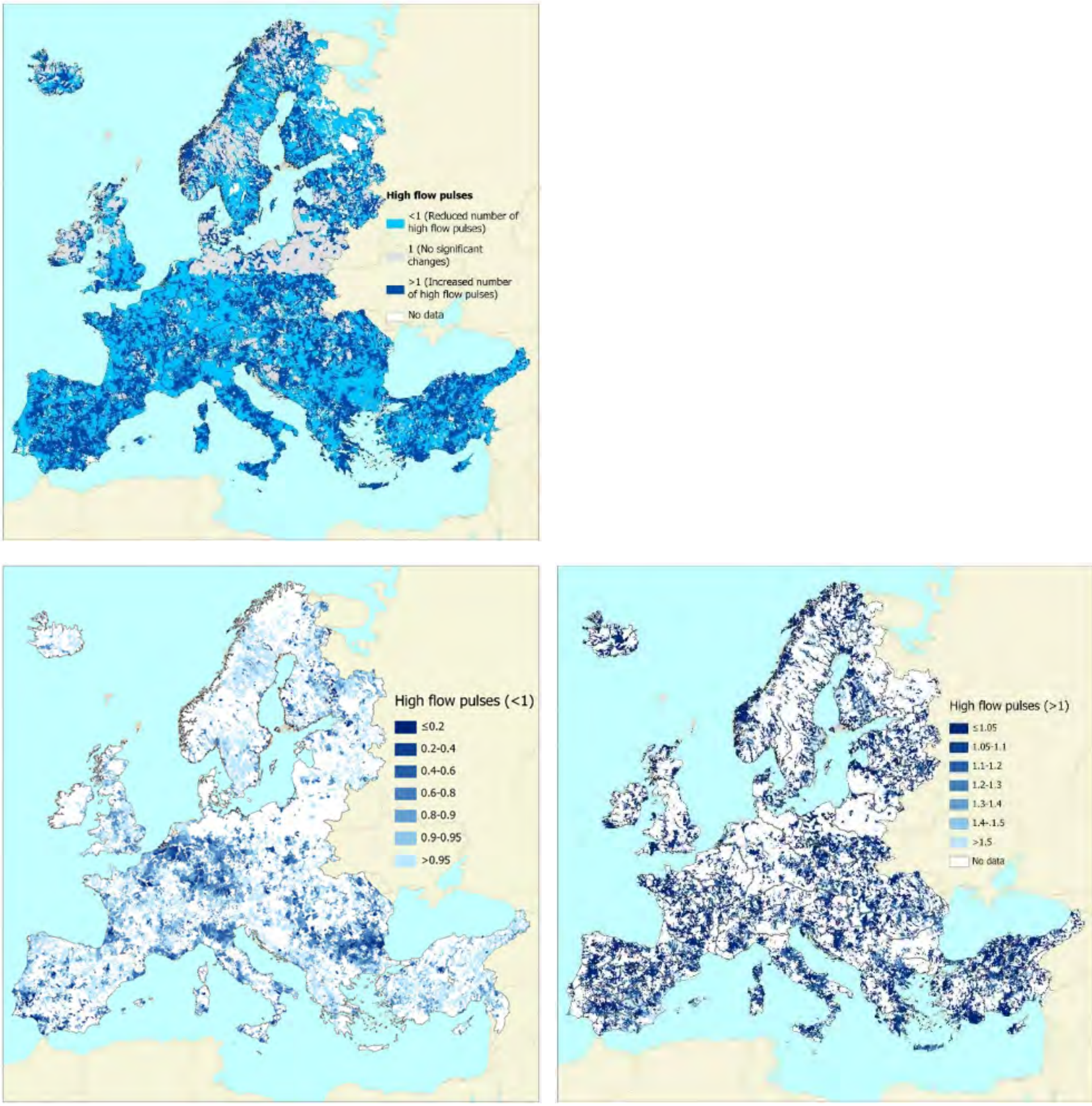


Figure 16: High flow pulses rate of change across Europe. Upper map shows both, a decrease (rate smaller than 1) and increase (rate larger than 1) of high flow pulses due to abstraction. The map on the bottom show the intensity of decrease (left) and the intensity of increase (right) of high flow pulses.

Hydrological alteration in small flood duration are mapped in Figure 17. For more than 53 % of all FECs no changes in small flood duration could be derived. For the remaining half of the FECs small flood duration increased (23 %) and decreased (22 %) to equal shares. Increases were observed for some large European rivers: Danube, Rhine and Po. Shares of FECs with very large differences between current and semi-natural conditions are quite small: 1.5 % of FECs show an increase and only 0.6 % of FECs show decrease by more than 50 %.

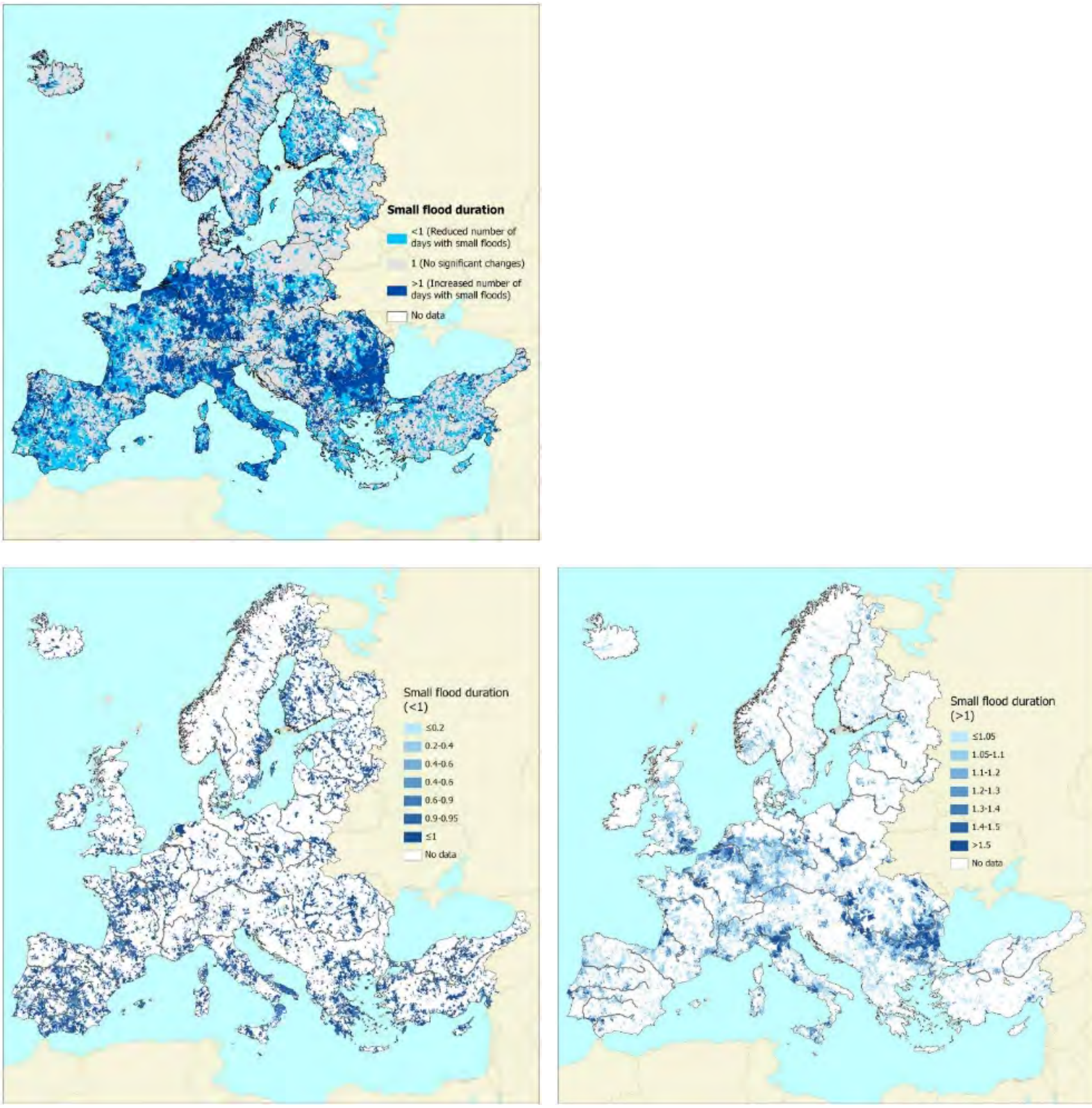


Figure 17: Small flood duration rate of change across Europe. Upper map shows decrease (rate smaller than 1) and increase (rate larger than 1) of small flood duration due to abstraction. The map on the bottom show the intensity of decrease (left) and the intensity of increase (right) of small flood duration.

For the hydrological alteration of low pulse thresholds, again the differences between large rivers (decrease) and their tributaries (increase) was found (Figure 18). Out of all FECs, 42 % have no changes in low pulse thresholds, 29 % show an increase and 29 % a decrease of low pulse thresholds. For a significant share of FECs (8.4 %) low pulse thresholds increase by more than 50 % is, whereas for only 1.9 % of the FECs decrease by more than 50 % could be found.

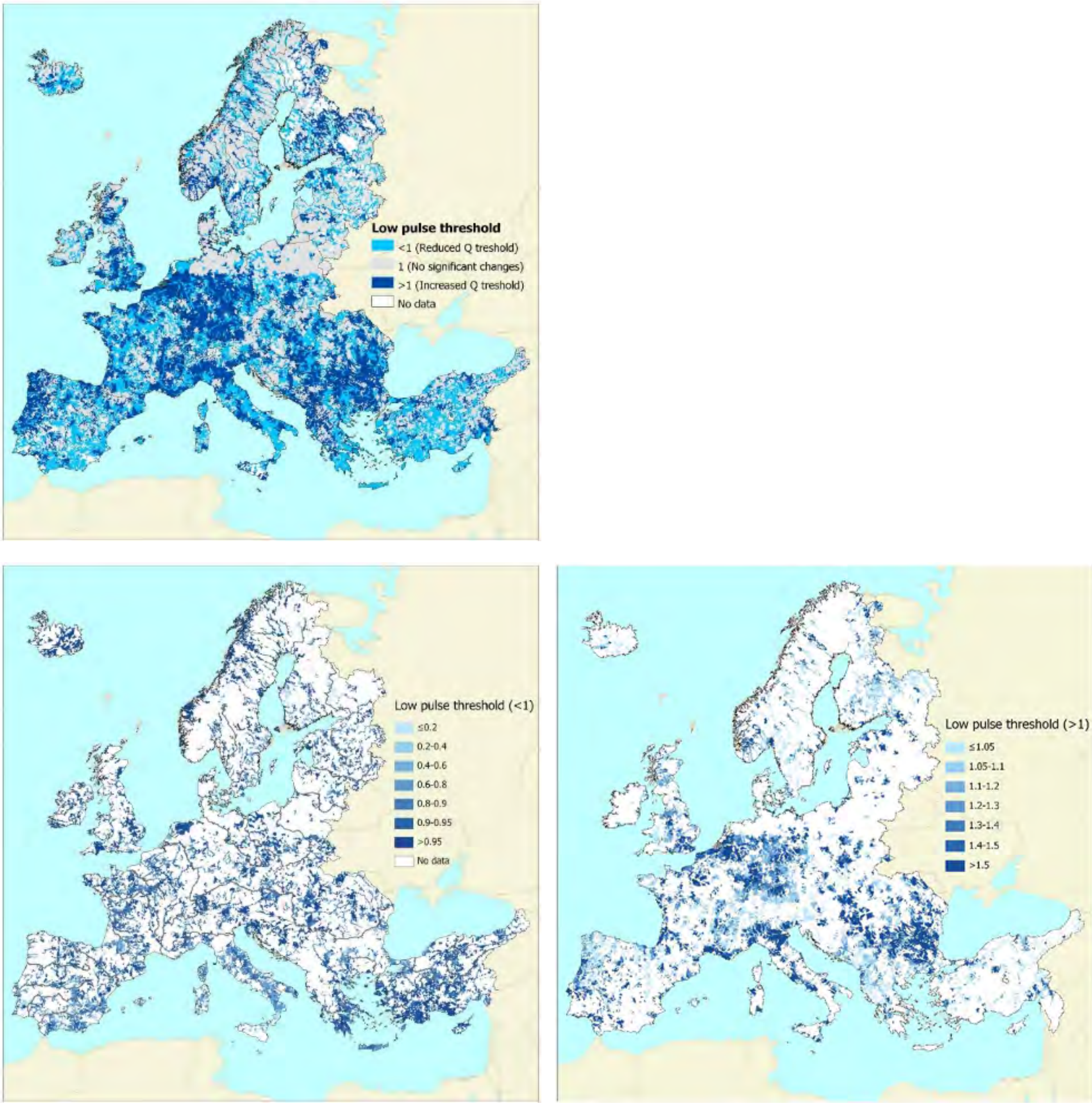


Figure 18: Low pulse threshold rate of change across Europe. Upper left map shows both, a decrease (rate smaller than 1) and increase (rate larger than 1) of low pulse threshold due to abstraction. The map on the bottom show the intensity of decrease (left) and the intensity of increase (right) of low pulse threshold.

More than 8 % of floodplains are covered by urban areas (Figure 19). Urban areas are beside residential, industrial and commercial facilities including also transport infrastructure, sport and leisure facilities as well as mineral extraction sites. In more than 8000 FECs urban areas are covering more than 30 % of floodplain. A vast majority of such FECs are lying in Central and

Baltic - Atlantic (CB-ATL) broad hydroregion. Shares of urban land in FEC's floods are the smallest in Nordic and Mediterranean broad hydroregion.

Agricultural areas are covering more than 30 % of all floodplains. In almost 20 % of all FECs, more than 50 % of flood plains are covered with agricultural land. High density of such areas can be found in Mediterranean broad hydro region and Central and Baltic – Mediterranean broad hydro subregion.

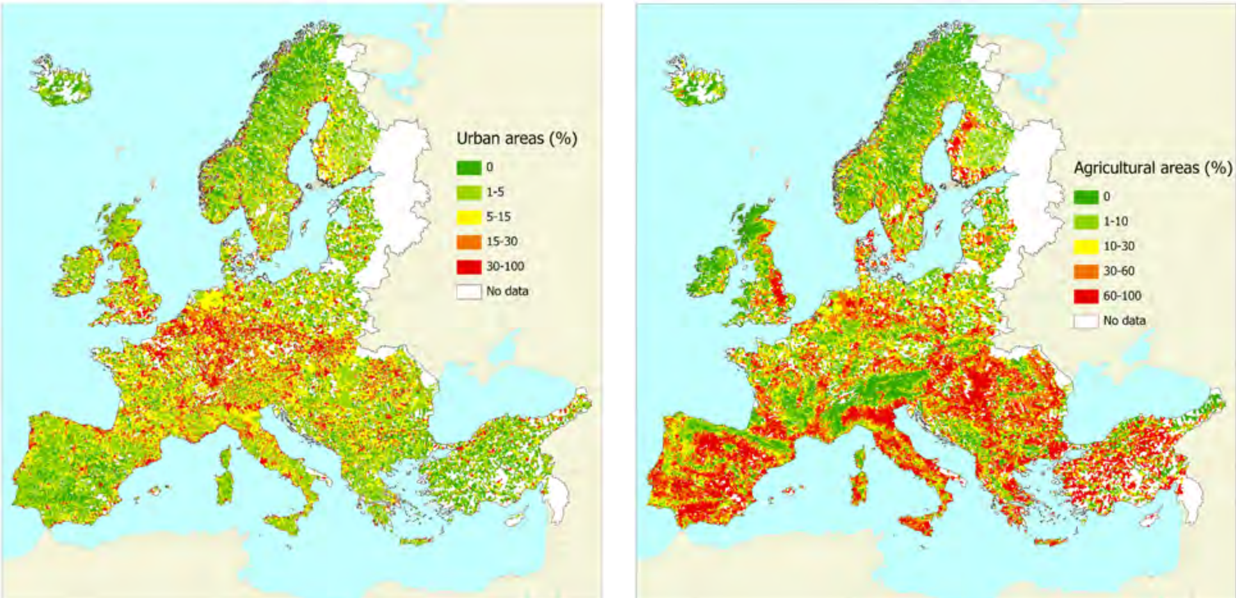


Figure 19: Share of urban (left) and agricultural (right) area in FEC's floodplains, coloured by classed

Woodlands and forests (transitional forest, broadleaved forest and coniferous forest) are covering 22 % of floodplains (Figure 20). In almost 20 % of these FECs, forests are covering more than half of the corresponding floodplain. The highest densities of coniferous forest within floodplains are significant for Nordic and Alpine broad hydroregions whereas higher densities of broadleaved forests can be found in Mediterranean as well as Central and Baltic broad hydro regions.

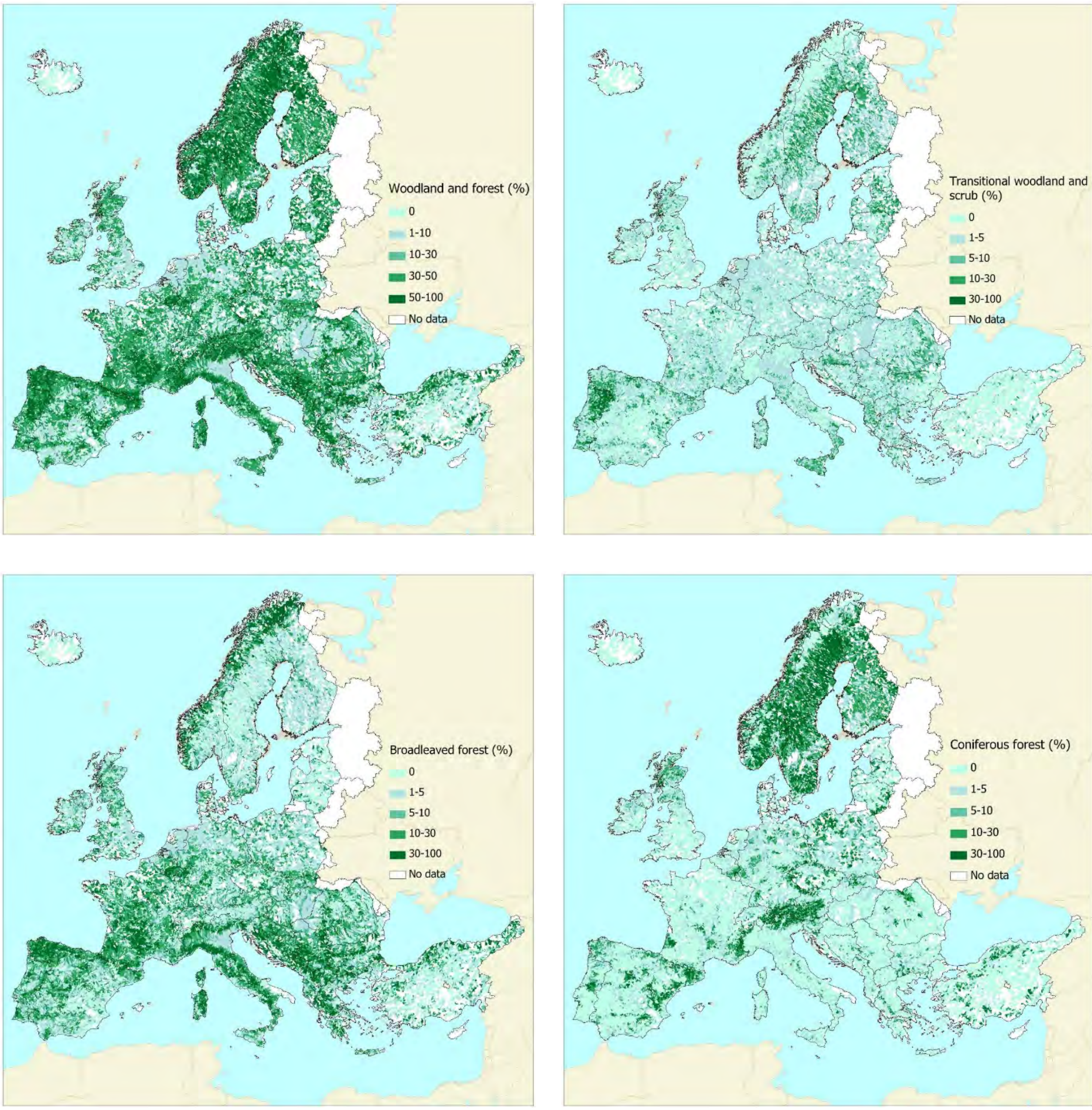


Figure 20: Percentage of area covered with woodland and forest (upper left map), area with only transitional forest (upper right map), and areas with only broadleaved (bottom left) or coniferous forest (bottom right) in FEC's floodplains. Woodland and forest coverage (upper left) is a sum of other tree categories, transitional, broadleaved and coniferous forest.

High percentage of lakes and reservoirs in floodplain area is significant for Nordic broad hydroregions, whereas small coverage with no significant differences between other broad hydroregions is observed (Figure 21). High percentage of natural grassland in floodplain are is observed at Eastern Continental region and Mediterranean hydroregions and in part of Central and Baltic – Atlantic sub-region.

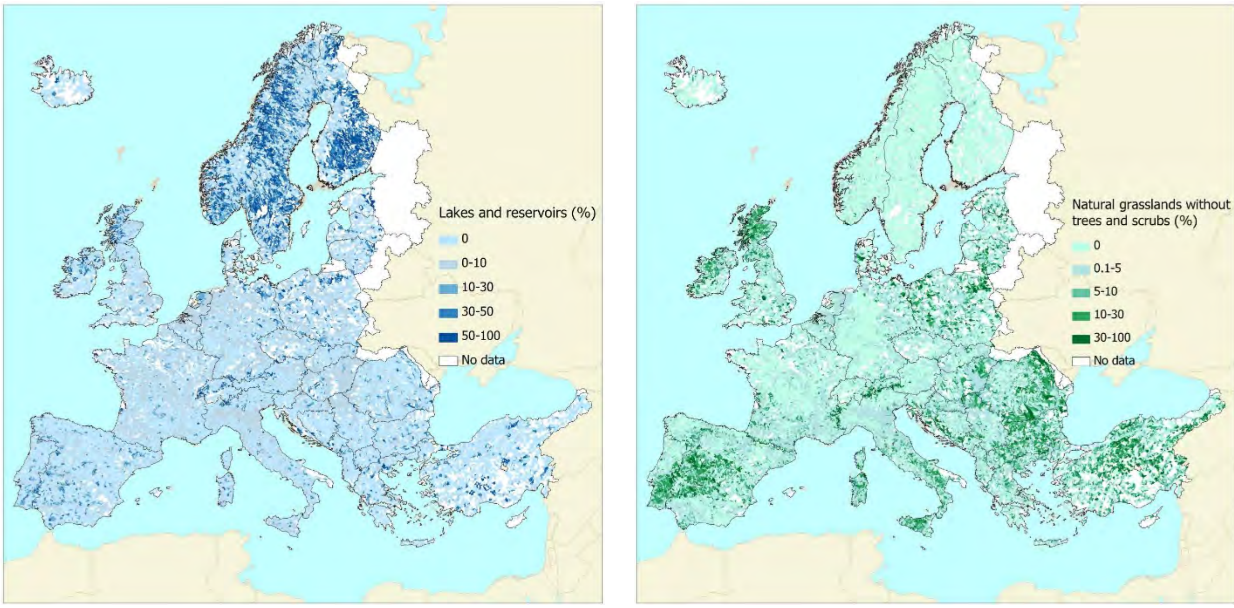


Figure 21: Percentage of area covered with lakes and reservoirs (left) and natural grassland (right).

Phosphorus intake per FEC calculated from EUROSTAT shows very high values for Central Europe and the lowest values for Nordic and Alpine broad hydroregions (Figure 22). On the other hand, degree of phosphorus saturation on agricultural land, modelled by MONERIS show quite high values for Nordic broad hydroregion, especially Finland. Low values are observed for most of Southern and South-Eastern Europe. Both parameters give the highest value for lowland areas in Belgium, Netherlands, Northern Germany and Denmark. In case of phosphorus intake (EUROSTAT), this area of highest value is extended also to United Kingdom, Ireland and France.

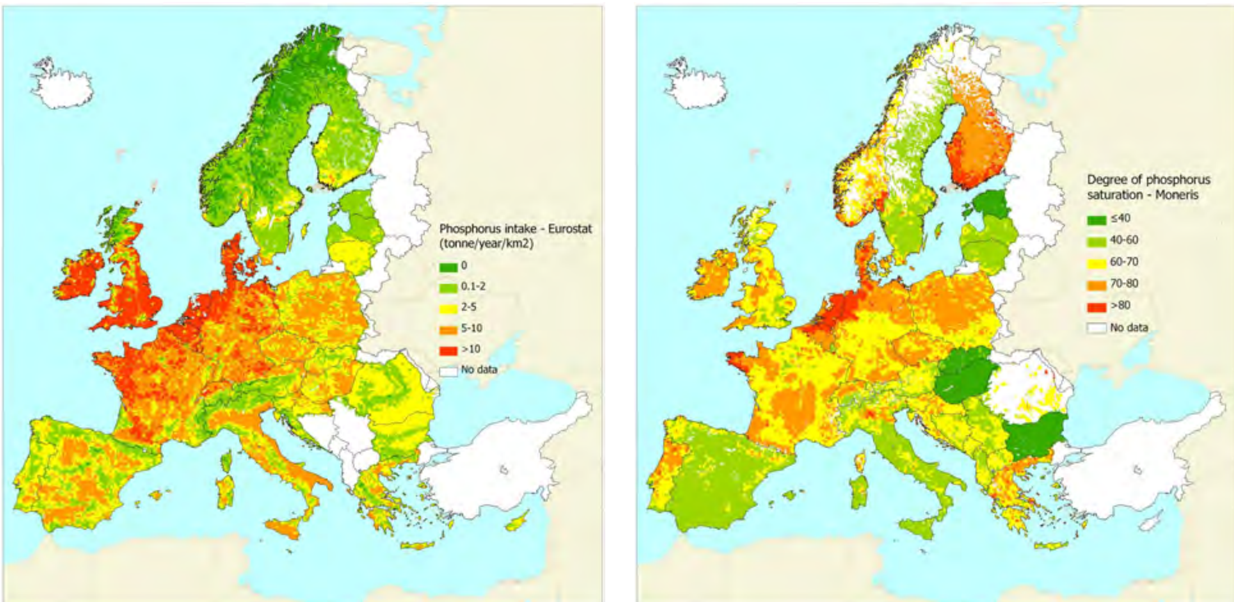
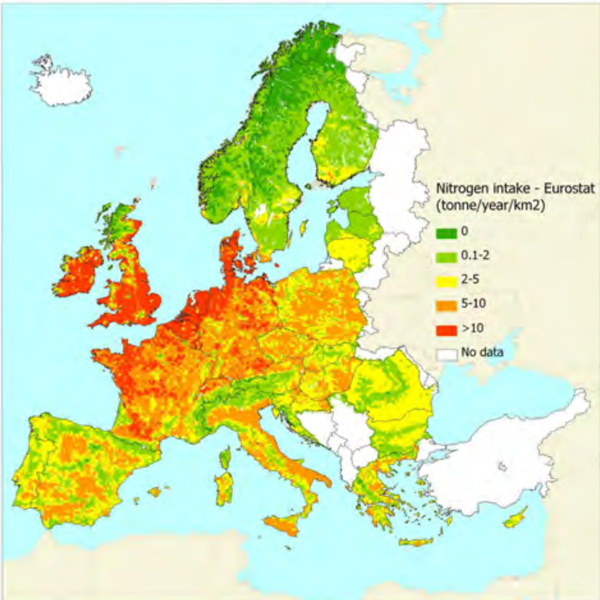


Figure 22: Phosphorus intake (EUROSTAT) (left), degree of phosphorus saturation in agricultural area (MONERIS) (right).

Nitrogen intake in FEC calculated by two methods are shown on Figure 23. Both methods show high values in Central lowland Europe and lower values in Northern Europe, but for some regions quite obvious differences can be observed. In Alpine broad hydroregion there are significantly lower values derived from EUROSTAT than from MONERIS. On the other hand, MONERIS gives lower values for majority of Central and Southern Europe. In pressure – response analysis MONERIS data were used.



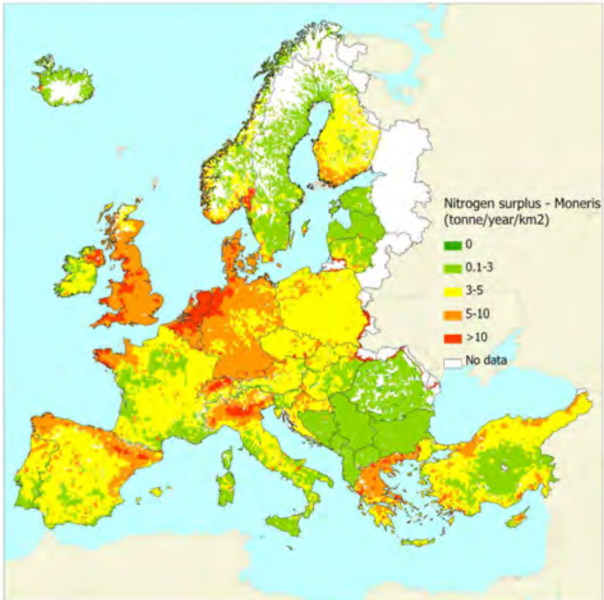


Figure 23: Nitrogen intake (EUROSTAT) (left), nitrogen surplus (MONERIS) (right).

3.4 Results of multipressure – response analyses for river broad types

We classified multi-pressures on 17 river broad types in Europe, five of them also by broad hydro (sub)regions. The total number of dataset is 28. There was not enough data for the classification of pressure for river types 7 ("Lowland, Organic and Calcareous/Mixed), 12 (Mid-altitude, Organic and siliceous) and 13 (Mid-altitude, Organic and Calcareous/Mixed). On the last two, 12 and 3, namely, there are only nine and two SoE stations.

The average classification accuracy of 28 models is 65 %, ranging from 51 % to 88% (Table 9). Average kappa statistics is 0.37, ranging from 0.2 to 0.78. Large classification accuracies have model on type 1, 2 4 in the Mediterranean regions and 14 on the Alpine region. Accuracy lower than 60%, have classification of pressures on river broad types 2 and 10 in the sub-region "CB CON " (Central and Baltic - Continental) and river broad type 11 in region "CB" (Central and Baltic). The classification accuracy in river broad type 2 was even lower (30-40%, Kappa from -0.05 to 0.1) when we build the model with the selected set of 28 explanatory variables. After we added data on total organic carbon concentration and physical parameters measured in water (temperature, electrical conductivity, oxygen demand, oxygen saturation, and suspended particles matter) the results improved to 54%. Nevertheless, the classification models of multiple pressure as obtained for low land siliceous and mid altitude calcareous river in the Continental sub-region have the lowest reliability among all models. For these rivers one should add additional explanatory variables or improve dataset coverage. Indeed, on more than 1000 SoE stations on river broad type 2 in this region, not more than quarter of them have data on above mention physico-chemical parameters. The good data coverage of the Mediterranean is also the main reason for good performance of models as described above.

Combination of pressures indicators, selected from random forest trees by river types are presented in Table 10. As explained in the methodological part, we can observe that decision boundary values are almost same if not similar for all selected explanatory variable. On broad river type 1 (large river) in the Eastern Continental hydroregion we selected eight pressures. As interpreted from regression forest, the ecological good status is highly supported if percentage of urban area smaller than 4.8%, percentage of agricultural area smaller than 26.5%, percentage of coniferous forest smaller than 0.37% (not existing), BOD5 smaller than 1.69 mg O2/l and NO3 smaller than 1.63 mg N/l. The base flow index is better not be higher than 3% in comparison to semi-natural flow regime. We should not conclude that all listed conditions should be satisfied, here as well can exist also "or" rule. But we need deeper "ecological" look at the rule, that good ecological status is supported if high flow peaks (high flow pulses threshold indicator) are reduced for more than 5%.

Nevertheless, here we do not produce rules for reaching good ecological status on this river type, but rather investigate what are four most important pressures. It may as well be, that this hydrological indicator is not recognised as important pressure. If this is a case, we may conclude, that random forest has made a mistake (it builds random trees anyway) or that we modelled pressures – response relations with less reliable data.

Table 9: Description of data sets for which we build classification models (label, number of cases in each class (response variable) and performance of classification models expressed with Kappa statistics and classification accuracy.

River broad type, hydroregion	RT1, CB	RT1, MED	RT1, EC	RT 2, BC-CON	RT 2, BC-ATL	RT2, CB-MED	RT3	RT 4, CB-MED	RT4, CB-CON	RT 4, CB-ATL	RT5	RT6	RT8	RT9	
"pbm"	132	86	59	42	193	48	287	129	363	623	127	37	154	162	
"gh"	23	72	15	10	102	20	284	42	213	179	57	35	131	163	
number of all cases	155	158	74	52	295	68	571	171	576	802	184	72	285		
Kappa stats	0.27	0.6	0.29	0.23	0.32	0.5	0.23	0.54	0.38	0.32	0.39	0.78	0.43	0.48	
Accuracy	63%	80%	77%	54%	60%	74%	59%	71%	61%	61%	61%	88%	72%	74%	
	RT10, ALP	RT10, CON	RT11	RT14, ALP	RT 14, MED	RT 14 CB	RT16, ALP and MED	RT16, CB-ATL	RT17	RT18	RT19	RT20	Average RT1-RT20	Min RT1-RT20	Max RT1-RT20
"pbm"	81	130	158	23	55	20	40	21	319	559	208	65	159	20	623
"gh"	42	49	99	111	130	46	40	10	95	595	266	118	113	10	595
number of all cases	123	179	257	134	185	66	80	31	414	1154	474	183	270	31	1154
Kappa stats	0.22	0.25	0.21	0.26	0.39	0.22	0.48	0.52	0.2	0.42	0.4	0.32	0.32	0.2	0.52
Accuracy	55%	51%	51%	0.85	77%	71%	55%	71%	59%	71%	71%	74%	67%	51%	88%

*"pbm": poor or bad or moderate ecological status; "gh": good or high ecological status

From the analyses of 28 subsets of pressures, we identified (low concentrations of) total phosphorus and nitrate as the most frequent pressure indicators for good ecological status (in 65% and 54% cases respectively).

High explanatory power for ecological status were found for ammonium, orthophosphate, share of woodland in floodplain, nitrogen surpluses and phosphorus saturation in FEC. They explain the ecological status in almost half of cases. The share of broadleaved forest has the same explanatory power as BOD5**,** both appear in one third of cases.

Importance of hydrological alterations as indicated with high flow pulses threshold, base flow index and duration of two year return period floods are also recognized as more important pressures then share of agricultural and urban area in floodplain.

Results of selection of important pressures by Gini index are presented Table 10. On the river type 1 in the Eastern Continental hydroregion four most important pressures as determined with Gini index are base flow index, high flow pulses threshold, BOD 5 and share of coniferous forest. In spite the fact, that these findings open questions either about method and data, nor on ecological aspects (including classification methods) we may conclude, that hydrological alteration pressures are equally if not even more important for support of good ecological status.

Table 10: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Part 1:

Proxy hydromorphological alteration indicators								Chemical stressors in water					Nutrient diffuse pollution indicators		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	$[t/km^2/Y]$	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 1: Very large rivers; Broad hydro region: CB (Central and Baltic) Kappa statistic: 0.27; Classification accuracy: 63%																					
													<1.86								
			<3.4																		<1.09
								<2.37	<0.78												
	<32.5																				
River type 1: Very large rivers; Broad hydro region: EC (Eastern Continental)Kappa statistic: 0.29; Kappa statistic: 0.29; Classification accuracy: 77%																					
				<0.37				<1.69													
	<26.5														<1.03		<0.95				
									<1.63		>0.05						<0.93				
<4.8																					
River type 1: Very large rivers; Broad hydro region: CB MEDITERRANEAN Kappa statistic: 0.60; Classification accuracy: 80%																					
										<0.02											
			<11.5																		<1.03
			<11.5							>0.01		<0.22									
															>0.96						<1.03
					>37.5			<0.64									<0.8				

Part 2: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Proxy hydromorphological alteration indicators								Chemical stressors in water								Nutrient Diffuse poll. Ind.		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	PO4	TP	SPM	TMP	TOC	DO	EC	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l]	[°C]	[mg/l C]	[mg/l O2]	[µS/cm]	[t/km²/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 2: Lowland Siliceous, Medium – Large (100-10000 km2); Broad hydroregion: CENTRAL AND BALTIC _ MEDITERRANEAN; Kappa: 0.54; Accuracy: 75%																								
					>14.9			<1.85																
																	<53.6					<0.95		
								<1.85															0.96-1.02	
			>8.95																	>0.98		<0.95		
									<0.26					OS: >87.6 %									<1.02	
River type 2: Lowland Siliceous, Medium – Large (100-10000 km2); Broad hydroregion: CENTRAL AND BALTIC _ CONTINENTAL; Kappa 0.23, Accuracy: 54%																								
								>0.79				<11.9		>118										
										<0.09	>15.5				<273.6									
										<0.15			<3.16	>9.72										
										<0.06		<11.9		>9.72										
																				>0.95				
River type 2: Lowland Siliceous, Medium – Large (100-10000 km2); Broad hydroregion: CENTRAL AND BALTIC _ ATLANTIC, Kappa 0.32; Accuracy: 60%																								
									< 0.07		NO3: <1.4								<1.6					
					>28.5		< 25.3													<1.08				
								<1.02		<0.07			>5.9											
< 1.5									< 0.01		NO3: <0.51													< 1.02

Part 3: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Proxy hydromorphological alteration indicators								Chemical stressors in water					Nutrient diffuse poll.in.		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km^2/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 3: Low land siliceous, Very small - Small (<100 km2) Kappa: 0.23; Classification accuracy 59%																					
<7.5													<7.24							<1.06	
	<55.6								<2.99						<1.05						
	<55.8													<65.3						<1.07	
											<0.05									<1.03	
River broad type 4: Low land, Calcareous or Mixed, Medium – Large Broad hydroregion: CENTRAL AND BALTIC - CONTINENTAL; Kappa: 0.38; Classification accuracy: 61%																					
					>24.1				<3.21		<0.1										
												<0.09				>0.81				>0.99	
											<0.11		<2.42								
										<0.06				<62.7		<1					
<2.9																					
River broad type 4: Low land, Calcareous or Mixed, Medium – Large Broad hydroregion: CENTRAL AND BALTIC – MEDITERRANEAN Kappa: 54, Classification accuracy: 71%																					
<32.3							<27.4									<1.42					
																<1.39					0.84-1.02
											<0.09				0.99-1.14						
										<0.03							<1				>0.70
River broad type 4: Low land, Calcareous or Mixed, Medium - Large , Broad hydroregion: CENTRAL AND BALTIC – ATLANTIC* Kappa: 0.32; Classification accuracy: 61%																					
													<6.57	<75	<0.97						
													<6.57	<73.4	<0.96						

*target class not balanced, therefore results are very biased;

Part 4: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Proxy hydromorphological alteration indicators								Chemical parameters in water					Nutrient diffuse p.		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km^2/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 5: Low land calcareous, Very small - Small (<100 km2) Kappa: 0.39; Accuracy: 61%																					
											<0.07										
											<0.07									<1.15	
														<60.7							
					>31%																>0.96
									<3		<0.08										
River type 6: Low land organic and siliceous, lower than 10 000 km2; Broad hyd. CENTRAL and BALTIC Kappa: 0.78; Accuracy: 88%																					
													<7.56			<1.03					
					>8.14									<66.3							
										<0.03			<7.56				<1.08				
								<1.83													
																				<1.04	
River type 6: Low land organic and siliceous, lower than 10 000 km2 ; Broad hydroregion: NORDIC Kappa: 0.27; Accuracy: 65%																					
				>7.13								<0.02									
							>0.03				<0.03										
				<26.4							<0.01										
	<14.0									<0.01							<1.04				
													<3.72			>0.85					
				>7.13								<0.02									

Part 5: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Proxy hydromorphological alteration indicators								Chemical parameters in water					Nutrient diffuse p.		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km^2/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 8: Mid altitude, Siliceous, Medium to large rivers (100-10000 km2) Kappa: 0.43; Accuracy: 72%																					
												<0.08	>4.47								
											<0.06		>4.4								
									<1.34				>4.48	<72.4							
					>21.5						<0.2		>4.47								
															<1.01						>0.94
River type 9: Mid altitude, Siliceous, Very small - Small (<100 km2) Kappa: 0.48; Accuracy: 74%																					
						<11.2					<0.08										
						<11.2						<0.03									
						<11.7					<0.08	>0.01									
				<2.2							<0.06										
			>18.2					<0.88				<0.04					<1.08				
River type 10: Mid altitude, Calcareous or mixed, Medium to large; Broad hydroregion: ALPINE Kappa: 0.22; Accuracy: 55%																					
	<58.7											<0.04									
											<0.03	<0.04									
			<27.4								<0.08						1-1.02				
									<1.07			<0.17		<69.1							

Part 6: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Proxy hydromorphological alteration indicators								Chemical parameters in water					Nutrient diffuse p.		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km^2/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 10, Mid altitude, Calcareous or mixed, Medium to large Broad hydroregion: CB -CONTINENTAL Kappa: 0.25; Accuracy: 51%																					
			>16.8						<2.9		<0.16				<1.02						
							<23.1		<0.87												
					>25				<0.97												
			>20.9									<0.04			<1.02						
<7.1																					
River type 11, Mid altitude, Calcareous or Mixed, Very small – Small (< 100 km2) Kappa: 0.21; Accuracy: 51%																					
										<0.06		<0.04								>0.98	
										<0.05							<1.02	<1.14			
			>27%																		>0.98
															>0.98						
									<1.19												
											<0.05										
River type 14: Highland, Siliceous or Mixed, Humic; Broad hydroregion: MEDITERRANEAN Kappa: 0.26; Accuracy: 85%																					
	<66.2								<2.07						>0.87						
					>36.6						<0.04										
							>0.07						<5.08								
									<0.37								<1.02				
					>36.				<0.37		<0.04		<5.08		>0.87		<1.02				

Part 7: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.																					
Proxy hydromorphological alteration indicators								Chemical parameters in water					Nutrient diffuse p.		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km^2/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 14: Highland, Siliceous or Mixed, Humic; Broad hydroregion: ALPINE; Kappa: 0.26; Accuracy: 85%																					
									<0.74	<0.01											
								<1.71				<0.04									
												<0.03	<6.48								
			>3.5	<46									<7.54								
								<1.27	>0.29			<0.03									
			>3.48	<46																	
											<0.04			<62.8							
River type 15, Highland, Calcareous/Mixed Kappa 0.26; Accuracy: 76%																					
													<8.97			>0.98				<1.03	
									<2.62		<0.02					>0.99					
																>0.98					>0.99
				>5.44		<2.41														>0.99	
River type 16, Glacial; Hydroregion: MEDITERRANEAN and ALPINE Kappa 0.48; Accuracy: 55%																					
										<0.02											<1.12
<12.4				<36.4																	
											<0.05									>0.97	
													<2.64								

Part 8: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.																					
Proxy hydromorphological alteration indicators								Chemical parameters in water					Nutrient diffuse pollution indicator		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km²/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 16 (Glacial Rivers); CENTRAL and BALTIC - ATLANTIC Kappa 0.52; Accuracy: 71%																					
												<0.04		>69.4							
													<3.67	>69.4							
									<1.21					>69.4							
			>23.5							<0.04											
											<0.07										
					>33.2																
River type 17: Mediterranean low land, medium-large, perennial Kappa: 0.2; Accuracy: 59%																					
<17.1	<80.8																<1.03				
			>22.9												<1.2						
					>13.3						<0.03	<0.3									
								<1.58					<2.37		>1.06						
River type 18: Mediterranean, Mid altitude, Medium-Large, perennial Kappa 0.42; Accuracy:71%																					
<3					>18.9												<1.02				
									<1.16		<0.14										

Part 9: Boundary conditions for pressure indicators, selected as most probable important explanatory variables (predictors) for good ecological status.

Proxy hydromorphological alteration indicators								Chemical parameters in water					Nutrient diffuse pollution indicators		Hydrological alteration indicators						
URB	AGR	FT	FB	FC[	WF	GRS	LAK	BOD5	NO3	PO4	TP	NH4	NBAL	PSAT	BaseF	HLT	HHT	HLD	HHD	HSFL	HHP
[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	[mg/l O2]	[mg/l N]	[mg/l P]	[mg/l P]	[mg/l N]	[t/km^2/Y]	[%]	[-]	[-]	[-]	[-]	[-]	[-]	[-]
River type 19: Mediterranean, Very small, perennial Kappa 0.4; Accuracy:71%																					
					>18.6					<0.14			<9.44								
											<0.14	<0.07									
								<1.24		<0.11											
									<1.79	<0.14	<0.02										
															>0.83						>0.71
River type 20: Mediterranean, temporary/Intermittent streams Kappa 0.4; Accuracy: 71%																					
									<3.36					<67.5							
											<0.17		<5.89								
								<3.17					<5.94								
					>32.1								<7.03	<67.5							

Table 11: Four most important pressures indicators for each river type and broad hydroregion derived by Gini index method.

Broad river type and broad hydroregion		pressure indicators			
RT1	CB_ATL,CB_CON	nitrogen surplus in FEC	% broad leaved forest	nitrate	high flow pulses
RT1	EC	base flow index	flow high pulses threshold	BOD 5	forestcon
RT1	MED	% of broad leaved forestorest	flowhighpul	orthophophate	ammonium
RT2	CB_ATL	BOD 5	total phosphorus	nitrate	% woodland and forest
RT2	CB_CON	total organic carbon	dissolved oxygen	electrical conductivity	suspended particles matter
RT2	CB_MED	% broad leaved forest	% woodland and forest	phopshorus saturation	small flood duration
RT3		% agricultural land	phopshorus saturation	nitrate	base flow index
RT4	CB_ATL	base flow index	phopshorus saturation	nitorgen surplus in FEC	
RT4	CB_CON	total phosphorus	orthophophate	nitorgen surplus in FEC	phopshorus saturation
RT4	CB_MED	low flow pulses threshold	total phosphorus	orthophophate	base flow index
RT5		% woodland and forest	total phosphorus	phopshorus saturation	high flow pulses
RT6	CB_ATL,CB_CON,CB_MED	phopshorus saturation	nitrogen surplus in FEC	% agricultural land	orthophophate
RT6	NOR	ammonium	% of agricultural land	orthophophate	flow high pulses threshold
RT8		nitrogen surplus in FEC	nitrate	ammonium	total phosphorus
RT9		% grassland	BOD 5	forestcon	ammonium
RT10	ALP	ammonium	nitrate	total phosphorus	% broad leaved forestorest
RT10	CB_CON	ammonium	total phosphorus	nitrate	base flow index
RT11		ammonium	small flood duration	flow high pulses threshold	flowlowdur
RT14	ALP	ammonium	BOD 5	nitrogen surplus in FEC	nitrate
RT14	CB_ATL,CB_CON,CB_MED	phopshorus saturation	BOD 5	total phosphorus	nitrate
RT14	MED	% woodland and forest	nitrate	total phosphorus	flow high pulses threshold
RT15		nitrate	flowlowth	total phosphorus	nitrogen surplus in FEC
RT16	ALP,MED	% urban land	forestcon	flow high pulses	orthophophate
RT16	CB_ATL	nitrate	phopshorus saturation	total phosphorus	orthophophate
RT17		% woodland and forest	% of urban land	total phosphorus	ammonium
RT18		% woodland and forest	nitrate	% of urban land	total phosphorus
RT19		% woodland and forest	orthophophate	nitrate	total phosphorus
RT20		% woodland and forest	total phosphorus	nitrogen surplus in FEC	nitrate

Table 12: Mean impurity decrease of four most important pressure indicators for each river type and broad hydroregion derived by random forest

	RT1, CB	RT1, MED	RT1, EC	RT 2, CB-CON	RT 2, CB-ATL	RT2, CB-MED	RT3	RT 4, CB-MED	RT4, CB-CON	RT 4, CB-ATL	RT5	RT6, CB	RT6, NOR	RT8
ammonium			0.38					0.28				0.25	0.25	0.15
BOD 5	0.35	0.26					0.14		0.23				0.33	
nitrogen balance		0.21		0.34			0.15	0.31	0.17	0.21	0.18	0.32	0.23	0.15
phosp saturation				0.37	0.26		0.15				0.17	0.28		0.15
PO4					0.19									
Base flow	0.35	0.21				0.45		0.29			0.15			
Flood	0.34					0.34	0.18		0.15	0.21				
Hhighf DUR				0.42		0.38			0.24	0.17				
Highf PUL		0.23		0.43		0.34								
Highf TH			0.35											
Lowf DUR														
% agricu land	0.34		0.36					0.29		0.2		0.26	0.24	
% forest broad			0.36		0.25									0.17
% grassland					0.26									
% forest conif											0.13			

	RT9	RT10, ALP	RT10, CON	RT11	RT14, ALP	RT 14, MED	RT 14 CB	RT 15	RT16, ALP and MED	RT16, CB-ATL	RT17	RT18	RT19	RT20
ammonium	0.17	0.25			0.15		0.14	0.25		0.52				0.16
BOD 5	0.18	0.27	0.28	0.18	0.18	0.17	0.17	0.28	0.32			0.08		0.12
nitrogen balance			0.24	0.19	0.19		0.13				0.19		0.13	0.15
phosp saturation						0.19	0.14			0.67		0.11	0.12	
PO4				0.21										
Base flow		0.4						0.31			0.2			
Flood	0.14					0.18					0.2	0.11		
Hhighf DUR									0.36	0.88				
Highf PUL						0.17						0.09		
Highf TH			0.24						0.4				0.11	
Lowf DUR					0.17									
% agricu land	0.22	0.34	0.26					0.27	0.32	0.65	0.2		0.1	0.13
% forest broad														
% grassland														
% forest conif				0.51										

There are 15 important pressure indicators recognised across all river types in Europe with mean impurity decrease method: five (5) pollution indicators (BOD5, PO4, NH4 and diffuse pollution of nitrogen as indicated by nitrogen balance and phosphorus saturation in FEC), six (6) hydrological alteration pressures (high flow pulses threshold, high flow pulses duration, high flow duration, base flow index and small floods duration) and four (4) proxy hydromorphological alteration indicators (percentage of agricultural land, broad leaved forest, coniferous forest and grassland in floodplain) (Table 12). Large number of impurity (horizontal axes, range between 0 and 1) means larger importance of pressure due larger explanatory power.

Nitrogen balance, BOD5 and percentage of agricultural share in floodplain are three most frequent important pressures (present in 16 out of 28 tested groups of river broad type – broad hydroregion combinations). Ammonium and phosphorus saturation are next most frequent (present in 12 out of 28 river broad-regions groups), followed by three hydrological alteration pressures (base flow index, floods and high flow duration.

The most important pressure in Europe, as comes out of averaging mean impurity decrease measures is a percentage of forest coniferous in flood plain, followed by high flow duration and base flow index (Figure 24). Percentage of agricultural land only comes as seventh most important pressure and BOD5 on ninth place. These relations changes when we exclude broad river type 6 in Atlantic sub-region (Figure 25), since the mean decrease measure here are much larger than others. In this case, the most important pressure becomes high flow duration, followed by forest coniferous share in flood plain and base flow index. Percentage of agricultural land is fourth most important in Europe. It is interesting that by mean impurity decrease measure method for selection of pressure explanatory variable, total phosphorus is not recognised as one of the four most important pressure on any of analysed broad river- hydroregion groups.

Ranking of pressures by river broad types and broad regions show a lot of difference (Table 12). As an example, on large rivers (river type 1) ammonium, percentage of broad leaved forest and agricultural land in floodplain are three most important pressures, followed by high flow pulse indicator (Figure 26). On lowland, siliceous and medium to large river (river type 2, Figure 27) hydrological alterations far overcome all the other pressures. Percentage of coniferous forest in floodplain is far most important pressure in mid altitude rivers (river broad types from 8 to 13). Base flow index, percentage of arable land and BOD 5 are relevant for highland rivers. On Mediterranean perennial rivers, the highest importance is given to base flow index, ammonium and nitrogen balance.

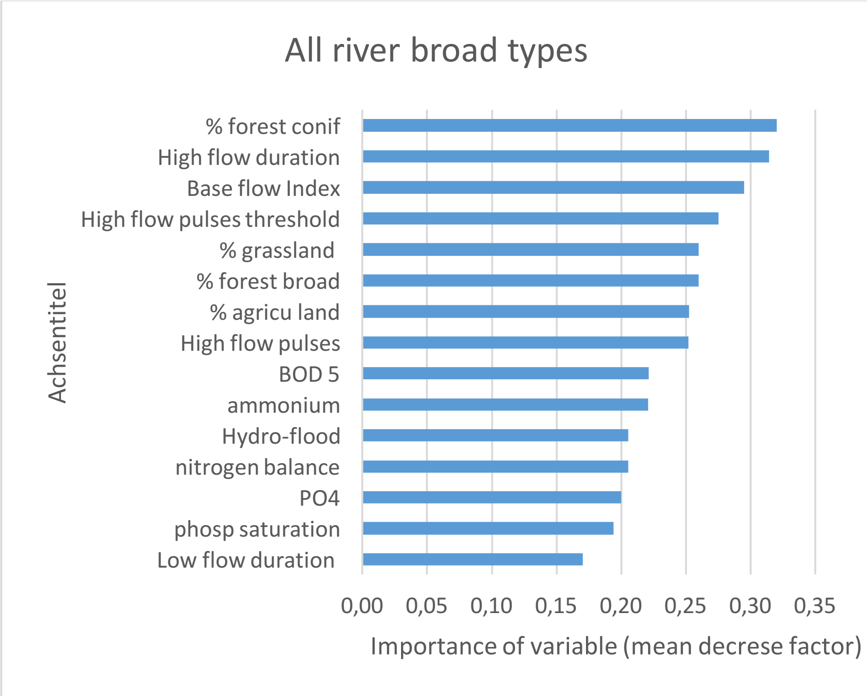


Figure 24: Importance of pressures - average over all river broad types and broad regions In Europe

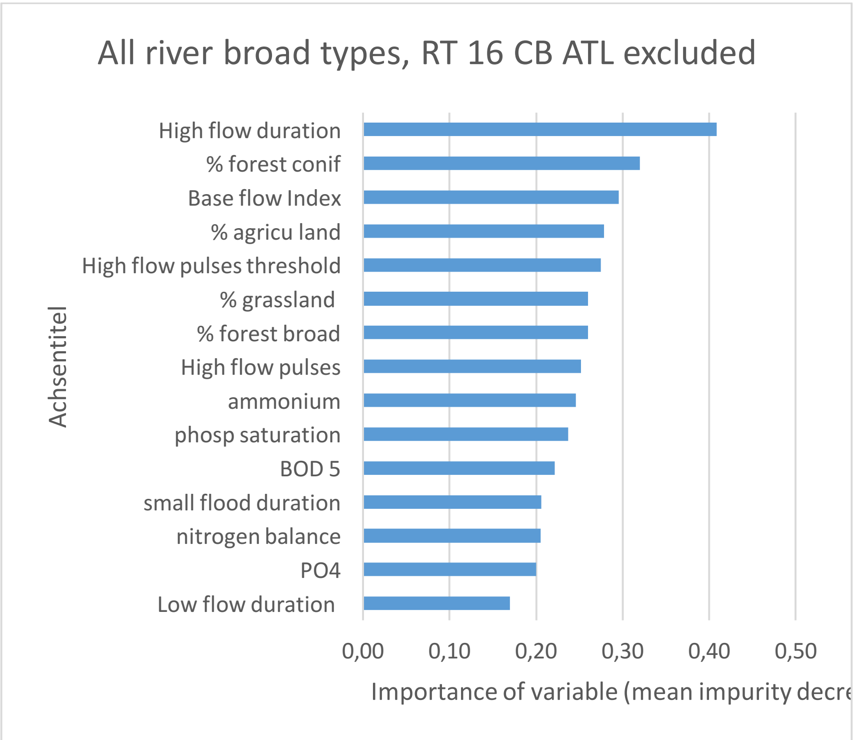


Figure 25: Importance of pressures - average over all river broad types and broad regions In Europe except over river broad type 16 in Atlantic subregion.

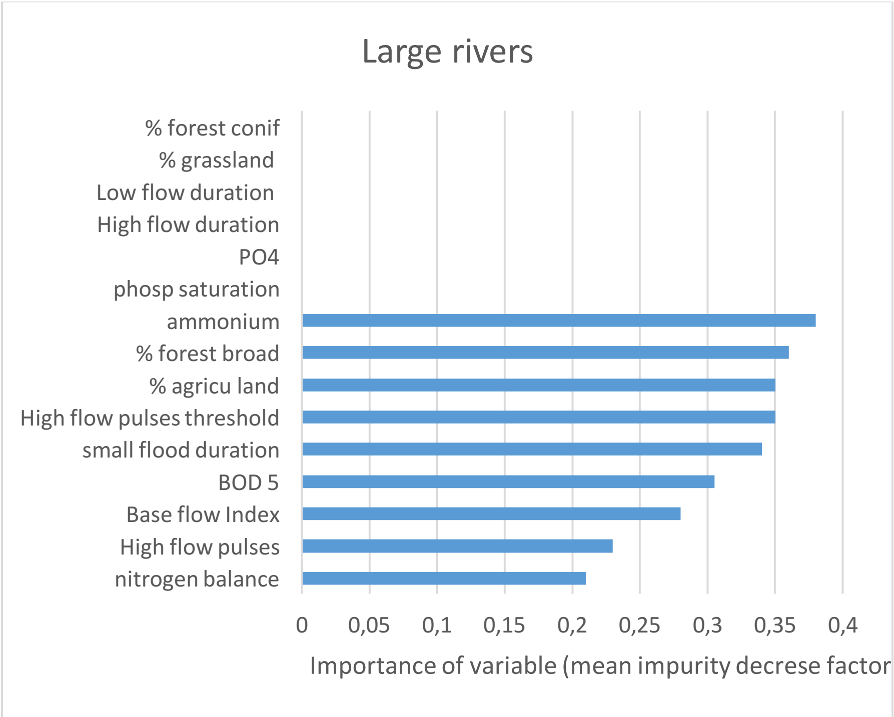


Figure 26: Importance of pressures on large rivers in Europe (river broad type 1).

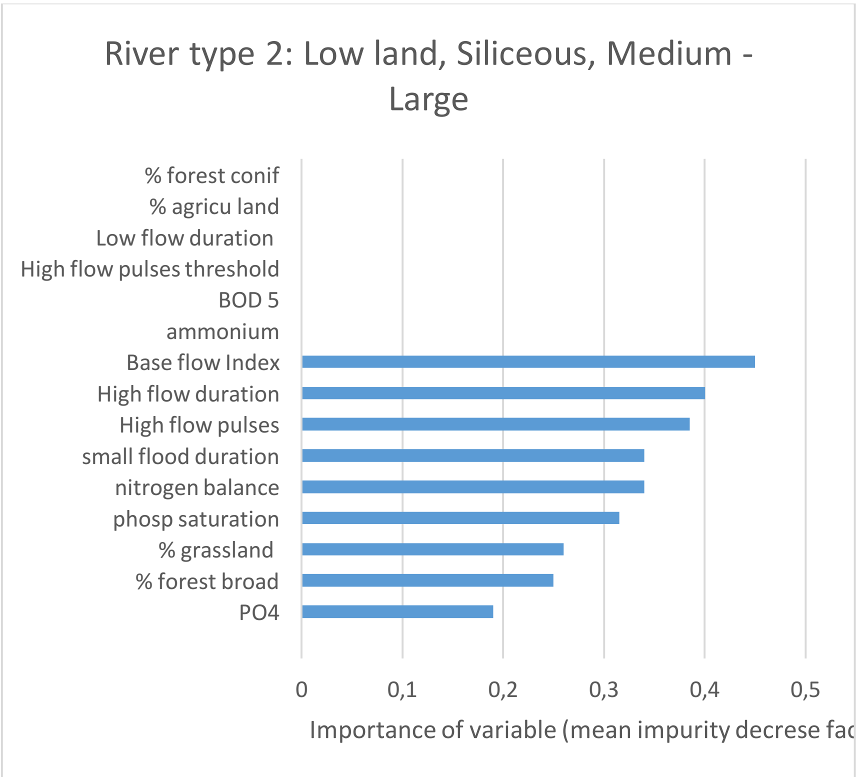


Figure 27: Importance of pressures on low land, siliceous medium large rivers in Europe (river broad type2).

Results of selection of important pressure by Gini index and mean entropy impurity decrease are similar. They both recognised ammonium, base flow index, high flow pulses threshold, BOD5 and share of coniferous forest as very important pressures. In spite the fact, that these findings may open questions about the method and data, when trying to understand un-expecting findings (e.g. decrease of high flow pulses threshold on large rivers in Eastern Continental hydroregion may not cause deterioration of ecological status), we conclude that hydrological alteration pressures are equally if not even more important in supporting good ecological status.

The mean impurity decrease measure method, neither Gini index method for selection of important explanatory variables, do not give any information about the character of common or simultaneous multi-pressures effects on the classification of response variable (ecological status). Nevertheless, we interpreted boundary decision values chosen by random forest (and then used for calculation of Gini index), as threshold for deterioration of ecological status.

As a final result we produce European multi-pressure maps. For each river type, we use decision boundaries of four most important pressures as threshold values. For each FEC we tested real values of important pressures against their threshold and counted number of pressures that may, individually or in combination, cause deterioration of ecological status. Situation when more than one pressure may cause deterioration of ecological status is characterised as multi-pressures situation. We detected all such FECs in Europe and produced European multi-pressure maps (Figure 28, Figure 29 and Figure 30).

In almost 60 % of all FECs (Figure 29) included in this analysis[7], at least one pressure (Table 13) does not satisfy its boundary conditions given in Table 14. The highest number of such FECs (15714 FECs or 30 % respectively of all analysed) are situated in Mediterranean broad hydroregion, 18 % in Central and Baltic-Atlantic and 21% Central and Baltic-Continental broad hydroregions.

There are 5% of all FECs, where at least three pressure do not satisfy boundary conditions (Figure 28). The majority of such FECs are situated in Central and Baltic-Atlantic (1743 FECs, or 2 % of all FECs included in this analyses) and Central and Baltic-Continental broad sub-hydroregion (1247 FECs or 1.4 % of all FECs included in this analyses).

[7]Out of 101957 FECs, 88500 have full data coverage for this analysis.

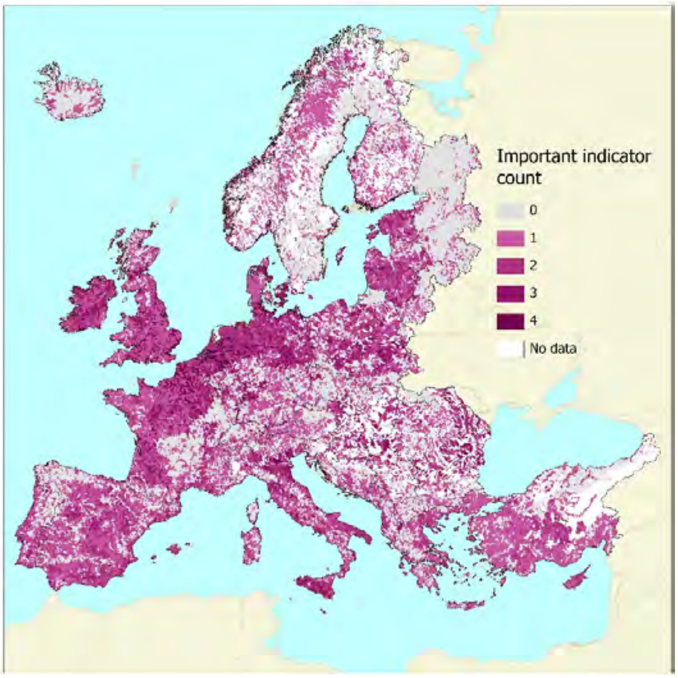


Figure 28: Multi-pressure map: Number of important pressures in each FEC that not satisfy boundary conditions for good ecological status; conditions are different for individual river broad type and hydro region.

There are 30% of FECs, where one or both diffuse pressure indicators (nutrient indicator) do not satisfy boundary conditions and 11 % where is at least one of hydrological alteration indicators outside given boundary conditions (Figure 29). Similar percentage, 32% of FECs do not satisfy boundary conditions for land cover or land use characteristics of floodplains (river morphology proxy indicators).

Morphological boundary conditions are mainly not satisfied in Mediterranean broad hydro regions, whereas boundary conditions of nutrient pressures are exceeded more frequently in Central and Baltic broad hydro region. The highest number of FECs where hydrological boundary conditions are not satisfied in Central and Baltic Continental broad sub-region and Eastern Continental broad hydroregion.

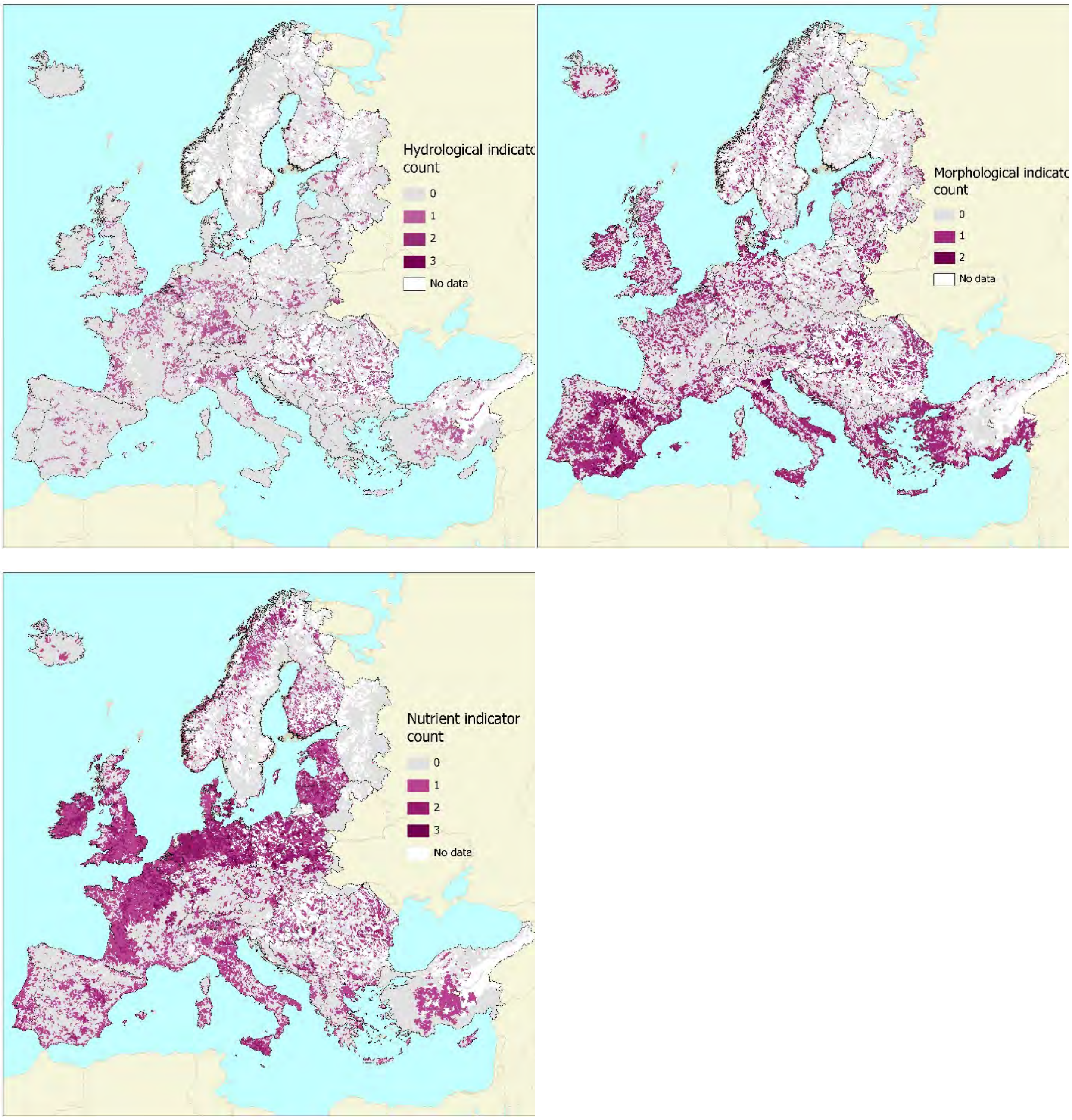


Figure 29: Maps of single pressure presents a number (count) of pressures from a same group that not satisfy boundary conditions for good ecological status; conditions are different for individual river broad type and hydro region):

The maps below (Figure 30) show 12000 FECs (more than 13 % of FECs) where combination of at least two of pressure (at least one pressure from hydrological alterations, and/or one from proxy morphological alteration or/and one from nutrient pressure indicators) do not satisfy boundary conditions for good ecological status.

The combination of morphological and nutrient pressures was assessed as the most common pressure combination occurring on 9065 FECs (75 % of FECs). A vast majority of such FECs are situated in Mediterranean and Atlantic (sub) hydroregions.

Hydrological and nutrient pressures is the second most common pressure combination occurring on more than 4000 FECs (4.7 % of FECs). Such combination is significant especially for Central and Baltic broad hydroregion.

The combination of hydrological and morphological pressures occurs on 3 % of FECs, whereas combination of three pressures (nutrient, hydrological and morphological pressures) occurs on more than 2 % of all FECs. These multiple pressure combinations are most significant for the Central and Baltic and Eastern Continental broad hydroregions.

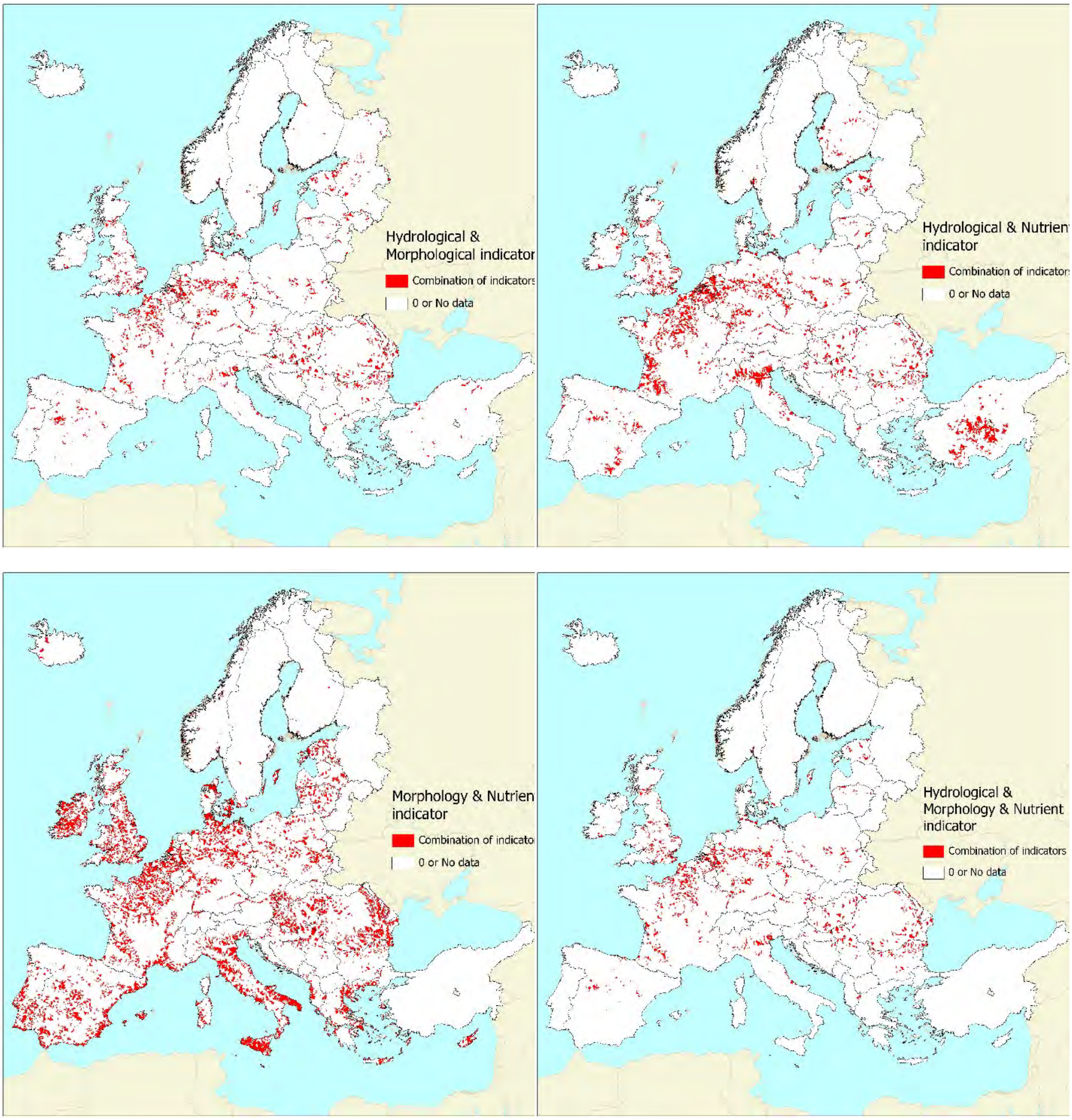


Figure 30: Maps of FECs with combination of two and three different types of pressures (hydrological alteration, diffuse pollution, proxy indicator for morphological pressures) that not satisfy boundary conditions for good ecological status; conditions are different for individual river broad type and hydro region). In the legend: "0" means that boundary conditions are satisfied.

4 Conclusion

Casual assessments of relations between multiple pressures and ecological response is very challenging but possible with stratified approach. We have partitioned European rivers into types and Europe into hydro-regions and worked on functional elementary catchments (FEC) as basic spatial units that build hydrological network and contributing areas. Regionalisation and typology of rivers were indeed found very important when identifying the most explanatory pressure indicators impeding a good ecological status. To compute pressure-response analysis, random forest classification machine learning method has been recognized as the most adequate for this task.

There are 15 important pressure indicators recognised across all river types in Europe with mean impurity decrease method: five pollution indicators (BOD5, PO4, NH4 and diffuse pollution of nitrogen as indicated by nitrogen balance and phosphorus saturation in FEC), six hydrological alteration pressures (high flow pulses threshold, high flow pulses duration, high flow duration, base flow index and small floods duration) and four (4) proxy hydromorphological alteration indicators (percentage of agricultural land, broad leaved forest, coniferous forest and natural grassland in floodplain).

High explanatory power for ecological status were found for ammonium (NH4), orthophosphate (PO4), share of woodland in floodplain, nitrogen surpluses and phosphorus saturation in FEC. They explain the ecological status in almost half of cases. The share of broadleaved forest has the same explanatory power as BOD5; both appear in one third of cases. NH4, PO4 and BOD5 chemical variables have higher explanatory power than TP and NO3. The reasons behind can be higher dynamics, bio availability etc. BOD5 is for example directly related to oxygen availability, thus directly impacting oxygen sensitive organisms that usually define good ecological status such as insects and salmonids. PO4 is on the other hand readily available for autotrophic organisms, thus probably more important that TP.

Importance of hydrological alterations as indicated with high flow pulses threshold, base flow index and duration of two year return period floods are also recognized as more important pressures as than share of agricultural and urban area in floodplain. In some regions, hydrological alteration of high flows explain even more variability than structural elements of vegetation such as percentage of forests or natural grasslands.

Ranking of pressures by river broad types and broad regions shows a lot of difference. On large rivers, ammonium, percentage of broad leaved forest, agricultural land in floodplain and high flow pulse indicator were recognized as four most important pressures. On lowland, siliceous and medium to large river, hydrological alterations far overcome all the other pressures. Percentage of coniferous forest in floodplain is far most important pressure in mid altitude rivers. Base flow index, percentage of arable land and BOD5 are relevant for highland rivers. On Mediterranean perennial rivers, the highest importance is given to base flow index, ammonium and nitrogen balance.

We are aware that findings provided by these analyses are not necessarily reflection of ecological processes, but rather a reflection of limited data availability, data reliability, methods and approaches for ecological status classification. The reliability of assessment could be improved if better data coverage would be available especially for physical data such as sediments, temperature, saturation, pH etc. Such data are now very scarce in the WISE SoE database which was otherwise found very useful as it covers almost whole assessed region and includes a considerable amount of parameters.

There is also a lack of European consistent data on river morphology and water uses (e.g. floods, energy, water abstractions due to irrigations). Since we miss regional data on river structural elements, results might also be biased towards "Copernicus" LC/LU (used as proxy for morphological alteration of floodplain areas) and hydrological alteration data. Moreover, the data on ecological status are from the first assessment cycle and reported in 1st RBMP (in 2010/2011), when the assessment has not been very systematic and comprehensive and not based on advanced intercalibration terms and conditions (Poikanea et al., 2014).

On the other hand, modelled data that cover all European territory (e.g. Nitrogen balance, Phosphorus saturation, hydrological parameters, Copernicus land use/land cover data) have been found very useful and often offer good basis for explanation of findings. They are also very informative for communication of problems and results via maps and graphs. Basic sub catchment units (FECs) were during the process assessed as to small for comprehensive analyses but large enough for visualization of pressures on European level.

Classification of multiple pressures on European broad rive types presented here is closely related to JRC and NTUA work in the same work package of the MARS project. JRC has unveiled patterns between human pressures and ecological status of European rivers in non- stratified manner. NTUA has analysed relation of low flows and ecological flows (E-flows) to ecological status and contributed data for hydrological pressures. This contribution is very important, since in our study we indeed show that hydrological pressures are very important, and in some regions and river types even prevail over morphological pressures. Our results serve as an input to scenario analysis tool at the European scale (namely work package 7.4) and will be expanded with additional data and expert knowledge.

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Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

Deliverable 5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis.

D5.1-1 Part 3: Multiple stressors and groundwater status analysis and statistical modelling at the European scale

Lead beneficiary: **Vit Kodes (CHMI)**
Contributors: **Ben Marchant (NERC), John Bloomfield (NERC), Lidija Globevnik (UL), Sian Loveless (NERC) and Tereza Hajkova (CHMI), Yiannis Panagopoulos (NTUA)**

Due date of deliverable: **Month 36**
Actual submission date: Month 36

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)	

Content

Non-technical summary

The aim of the work is to analyse groundwater status and stressors (pressures) relevant for groundwater using available data at European scale reported by European countries (WISE-WFD and WISE-SoE datasets managed by the EEA). In particular, a definition of spatial extent of ground waters in poor status, acting single stressors (pollution, abstraction, saltwater intrusion) and stressor combinations including an identification of prevailing pollutants causing failure of good groundwater status. The aim of the statistical analysis is to use simple statistical models to investigate the large-scale pressures on the chemical status and quantitative status of groundwater reported by European Union Member states. In particular, to see if it is possible to use these models to investigate and understand any interactions between different pressures on groundwater status.

The analysis of stressors and status shows that prevailing stressor causing failure of good groundwater status is pollution, followed by groundwater abstraction. Pollution in combination with groundwater abstraction appears to be most common stressor combination in Europe. Salt water intrusion is almost always associated with groundwater abstraction or/and pollution, but it does not take place in all coastal areas. The most common type of groundwater pollutants are agrochemicals (nutrients and pesticides) affecting whole Europe and especially agricultural areas. When assessing pesticide pollution at European scale, one must take into account a bias induced by various monitoring strategies used by countries, there is lack of comparable data on pesticide metabolites that may occur more frequently and in higher concentrations than parent pesticides. EU WFD common implementation strategy does not assure sufficient harmonization of monitoring strategies among EU member states preventing comparable pan-European assessments.

The study demonstrated how 'data-led' methods, such as stepwise regression, can be used to suggest and estimate models of groundwater status. However, we note that they should be used with caution as such approaches can include spurious relationships which result from not accounting for multiple hypothesis tests. Only limited interactions have been investigated to date, however, there is some evidence for a synergistic interaction between arable farming and winter precipitation (when the regression does not include country as a random effect) on the chemical status of groundwater. There is, however, less confidence in the results of models of groundwater quantitative status which appears, as may be expected, to be largely driven by weather variables.

2.1 Multi-stressors and groundwater status analysis at the European scale

2.1.1 Introduction

Groundwaters are impacted by various stressors leading to either depletion of groundwater quantity or/and quality, including groundwater dependent ecosystems. The review on multi-stressor effects in groundwater identified climate and its change, water abstraction, salt water intrusion and pollutants the major stressors for the groundwater. A general review of the mechanisms and significance of three facets of aquifer degradation (depletion of aquifer storage and its effects on groundwater availability and dependent ecosystems; groundwater salinization arising from various different processes; and vulnerability of aquifers to pollution) is given for example by Foster and Chilton (2003).

Climate

Regarding natural stressors, the groundwater recharge as a key process securing a replenishment of groundwater is directly influenced by climate and hydrogeological settings. Since hydrogeological settings do not change in time, the climate influences the changes of groundwater recharge predominantly. Climate processes influence groundwater patterns in a complex way, with a number of direct and indirect effects. Climatic variables influence hydrological processes, so any change in precipitation, evapotranspiration, snow accumulation and snow melt will influence recharge and groundwater formation (Kløve et al., 2014). Substantial reductions in potential groundwater recharge are uniformly projected in southern whereas increases are consistently projected in northern Europe (Taylor et al., 2013). Changes in recharge rates and mechanisms may also increase the mobilisation of pesticides and other pollutants in the unsaturated zone and reduce groundwater quality (Kløve et al., 2014). Comprehensive reviews of impacts of climate change on groundwater were compiled by Taylor et al. (2013) and Kløve et al. (2014). As concluded by Kløve et al. (2014), generalising the effects of climate on groundwater quantity and quality for any particular region is challenging and subject to considerable uncertainty, therefore climate was not included into the analysis.

Water abstraction

Water consumption in intensively irrigated regions was estimated at 1364 km3/year globally and up to about half of the global water withdrawal for irrigation was found to draw from non-renewable and/or nonlocal water resources (Rost et al., 2008). A global analysis of climate change impacts on irrigation demand suggests that two thirds of the irrigated area in 1995 will be subjected to increased water requirements for irrigation by 2070 (Taylor et al., 2013). Excessive groundwater abstraction is a worldwide problem especially in regions of dry climate. Barron et al. (2012) addressed drying climate in south-western Australia resulting in significant impacts on surface water and groundwater resources as well as the ecosystems dependent on them. Such climate change is likely to cause drying conditions in the region, similarly to other regions with a Mediterranean climate type worldwide. Increasing water demand, which in a drying climate is likely to be accompanied by high rates of groundwater abstraction, may pose a further risk to groundwater-dependent ecosystem. Wriedt et al. (2009) identified an irrigation is a key driver of water use in Southern Europe. The study estimated various irrigation strategies requirements for whole Europe by crop growth modelling, the most water demanding strategy requires up 1220 mm/year of water in the Mediterranean, which ought to be covered by groundwater resources mainly. The absolute net irrigation requirements were estimated highest in the Mediterranean and lowest in the boreal crop region reflecting the climatic conditions. Hydrological modelling for various climate scenarios was used to assess an impact of climate change on groundwater resources in the Mediterranean (Turkey) by Ertürk et al. (2014), showing the decrease of precipitation and subsequently the recharge and increase of evaporation. Almost 60% of precipitation is lost via evapotranspiration in the region. Similarly Candela et al. (2009) modelled climate change impact on coastal aquifer system and dependent ecosystems on Mallorca Island. Simulations for 2025 showed that predicted climate change will affect the entire hydrogeological system i.e., components of the water budget as well as dependent ecosystems including increase of seawater intrusion and a need for reduction of water abstraction. Menció and Mas-Pla (2010) showed that a particular pressure is caused by agricultural uses with intense withdrawal rates during the irrigation, resulting in very fast decline of stored groundwater resources in the alluvial formations during summer, leading to an immediate effect on stream discharge within a study area in Spain. A lack of an appropriate base flow is even responsible for a loss of ecological quality, as the necessary discharge to dilute waste water is not provided. An impact of climate change on groundwaters in an aquifer system in Belgium was studied by Goderniaux et al. (2009). The groundwater levels are projected to significantly decrease for all considered climate change scenarios, even in the temperate climate. Drier summers will also likely cause increases in exploitation rate of groundwater. Intensification of irrigation practices by groundwater extraction will also induce an additional water volume leaving the system by evapotranspiration. Additionally, problems of contaminant accumulation (e.g. salts, pesticides, fertilizers) could also appear in this aquifer because of the circulation in a closed system. Cases of severe land subsidence caused by excessive groundwater abstraction were addressed by Tomás et al. (2005) and Giambastiani et al. (2007).

Salt water intrusion

A comprehensive review of salt water intrusion processes and current state of salt water intrusion research is provided by Werner et al. (2013). Salt water intrusion occurs in coastal areas worldwide. According to Solheim et al. (2012) 9 EU countries have identified salt water intrusion as a reason for failure to achieve good chemical status groundwater bodies. The limited number of studies conducted to date on groundwater quality have primarily addressed salt water intrusion into coastal aquifers, and some studies indicate that groundwater pumping is expected to have more of an effect than climate change and sea level rise on seawater intrusion in some coastal aquifers (Kløve et al., 2014). Global sea-level rise of 1.8 mm/year over the second half of the twentieth century is expected to have induced fresh-saline water interfaces to move inland. The extent of seawater intrusion into coastal aquifers depends on a variety of factors including coastal topography, recharge, and critically groundwater abstraction from coastal aquifers (Taylor et al., 2013). The phenomenon of salt water intrusion occurs mainly in arid and semi-arid areas i.e.in case of Europe in the Mediterranean. Groundwater salt water intrusion is a common environmental problem in Greece. Depletion of groundwater, deterioration of its quality caused by overexploitation of aquifers of Peloponnesus and Crete, a coastal aquifer in Northeastern Korinthia, aquifers in Thrace resulting in a salt water intrusion into the aquifers was studied by Lambrakis and Kallegris (2001), Vodouris (2006) and Petalas and Lambrakis (2006), respectively. Petalas et al. (2009) documented similar salt water intrusion in the coastal Rhodope aquifer system due to its overexploitation predominantly for irrigation needs. Additionally irrigation water was identified as the major transport medium for agriculture contaminants such as nitrates. A coastal aquifer seawater intrusion in Turkey caused by excessive pumping was described by Demirel (2004). Salt water intrusions into coastal aquifers in the Cecina and Ravenna areas in Italy were described by Grassi et al. (2007) and Giambastiani et al. (2007), respectively. Groundwater pumping, exceeding the recharge, has caused severe seawater intrusion in the Cecina aquifer system additionally polluted by boron mainly from industrial activities and nitrate pollution mostly from agriculture transported to the aquifer system by natural recharge processes, whereas irrigation plays an important role locally. In addition to the salt water intrusion into Ravenna aquifer, a strong land subsidence induced by an overexploitation of the aquifer was documented.

Pollution

Changes in recharge rates and mechanisms may also increase the mobilisation of pesticides and other pollutants in the unsaturated zone and reduce groundwater quality (Kløve et al., 2014). In addition to recharge, land use and management practices play an important role in contamination of groundwater especially in case of agricultural activities such as irrigation, fertilization, pesticide application or even urban settings (Kampbell et al., 2003; Carlson et al., 2011; Menció et al. 2011; Stuart et al., 2011; Wick et al., 2012; Köck-Schulmeyer et al., 2014). Environmental properties of pollutants that influence their leaching must be taken into account in addition to the above (Baran et al., 2008; Köck-Schulmeyer et al., 2014). Nitrate was identified as the most widespread groundwater pollutant in the EU by Solheim et al. (2012) as well as pesticides as representatives of synthetic organic pollutants. Masseti et al. (2008) and Menció et al. (2011) showed positive correlation between the increase of groundwater recharge and the occurrence of high nitrate concentration in groundwater, similar effect for pesticides was described by Baran et al. (2008). On the contrary, Kampbell et al. (2003), Wick et al. (2012) and Pasini et al. (2012) demonstrated opposite relationship i.e. higher nitrate concentrations during dry periods. Higher temperatures may also lead to lower nitrate pollution of groundwater, possibly as a result of increased evapotranspiration (Wick et al., 2012). The implications for nitrate leaching to groundwater as a result of climate changes are not yet well enough understood to be able to make useful predictions without more site specific data (Stuart et al., 2011). Complex chemical mixtures of various natural and anthropogenic contaminants occur in groundwater, but little is known about their potential health effects (Toccalino et al., 2012). Concurrent occurrence of various contaminants at many sites in Europe and in U.S. was documented by Loos et al. (2010) and Toccalino et al., 2012, respectively. Bloomfield et al. (2006) and Stuart et al, (2011) both conclude that the indirect effects of climate-induced changes in demand for water and other natural and agricultural resources and changes in land use may have a greater effect on fate and transport of pesticides and on nitrate concentrations respectively than direct effects of climate change itself.

2.1.2 Methods

2.1.2.1 Data sources

WISE-WFD

All the WFD attribute data were compiled from the EEA WISE-WFD database as reported by EU member states under the 2000/60/EC directive (Water Framework Directive, WFD) in 2009 (http://www.eea.europa.eu/data-and-maps/data/wise_wfd). The database contains status, pressures and supplementary data for 27 EU countries (AT, BE, BG, CY, CZ, DE, DK, EE, ES, FI, FR, GR, HU, IE, IT, LT, LU, LV, MT, NL, PL, PT, RO, SE, SI, SK, UK) in total for 13309 groundwater bodies.

The groundwater GIS reference layer (EEA, 2013) containing 13345 groundwater bodies (GWBs) throughout the EU (Duscher, 2013) was used for spatial analysis. Groundwater bodies in the dataset, consistent with the Guidance document No. 9 on implementing the Geographical information elements (GIS) of the WFD (EC, 2003), are layered in 7 horizons representing distinct vertical layers of groundwater bodies. Individual horizons were rasterized from polygons into grids. The grids of individual horizons were merged into one 1 km grid using map algebra and cell statistics based on the simplifying assumption that the ID of the uppermost groundwater body within vertical sequence of groundwater bodies lying on top of each other determines the groundwater body ID of a respective grid cell. Processed attribute WISE-WFD data were then joined with the grid cells using groundwater body ID. Resulting grid contains 96% of GWBs with linked WFD info and 4% of GWBs with no WFD info, see Table 1. Number of groundwater bodies with missing WFD info within respective countries is given in Table 2. The grid represents 11139 uppermost groundwater bodies in 27 countries (AT, BE, BG, CY, CZ, DE, DK, EE, ES, FI, FR, GR, HU, IE, IT, LT, LU, LV, MT, NL, PL, PT, RO, SE, SI, SK, UK) from 6 horizons (Fig. 1).

Table 1: Number of groundwater bodies in GWB grid

Total GWBs in a grid	GWBs with WFD info	GWBs with no WFD info
11139	10714	425

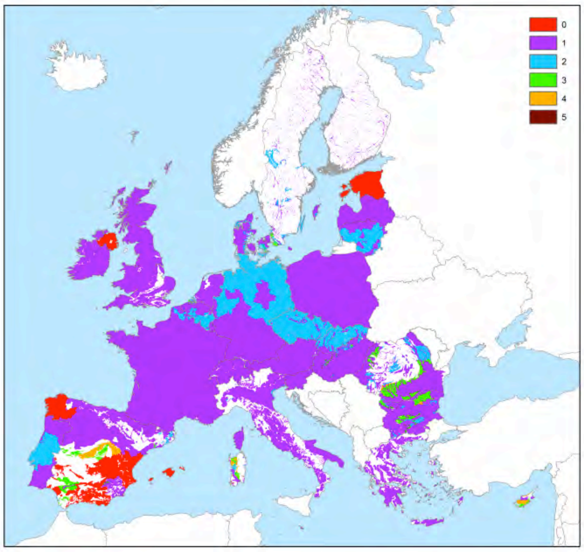


Figure 1: Groundwater body horizon assignment

Table 2: Number of groundwater bodies with missing WFD info within respective countries

Country	No. of GWBs with no WFD info	No. of GWBs with WFD info
IT	155	355
SE	1	2278
ES	77	632
EE*	22	0
BE	34	22
CH**	124	0
FI	12	2979

* *GWB IDs in GIS reference layer and WISE-WFD database do not match*

** *Switzerland does not report under the WFD*

WISE-SoE

All the State of the Environment (SoE) data were compiled from a EEA database that contains concentration data from groundwater monitoring networks reported to the EEA under the WISE-SoE dataflow in a period of 2008-2012 (http://rod.eionet.europa.eu/obligations/30). In total, data from 16385 SoE stations from 36 countries (AL, AT, BA, BE, BG, CY, CZ, DE, DK, EE, FR, GB, GR, HR, CH, IE, IS, IT, LI, LT, LU, LV, ME, MK, NL, NO, PL, PT, RO, RS, SE, SI, SK, TR, UK, XK) were used. Data were aggregated as annual mean concentrations at individual stations prior provision for the MARS project. The highest annual mean concentrations from the 2008-2012 period were used and compared with drinking water standards pursuant to the Directive 98/83/EC on the quality of water intended for human consumption for the assessment. The Directive 2006/118/EC on the protection of groundwater against pollution and deterioration sets the standards just for nitrates and pesticides (identical with drinking water standards), the threshold values (TVs) for other pollutants were established by Member States and used for chemical status assessment. In accordance with the 2006/118/EC directive the TVs may be set individually at the national, RBD or even groundwater body level. The WISE-WFD database does not contain adequate info on TVs used for the status assessment by countries thus EU wide drinking water standards were used instead TVs (Tab. 3).

All (WISE-WFD, WISE-SoE) attribute data were stored and processed in RDBMs Oracle 12c Enterprise Edition (http://www.oracle.com/database). GIS analyses and maps were produced with ArcGIS 10.2.2 for desktop and its Spatial analyst extension (http://www.esri.com/software/arcgis).

Table3: 98/83/EC directive drinking water standards

Substance	Limit	Unit
1,1,2,2-Tetrachloroethene	10*	µg/l
1,1,2-Trichloroethene	10*	µg/l
1,2-Dichloroethane	3	µg/l
Aluminium	200	µg/l
Ammonium	0.5	mg/l
Antimony	5	µg/l
Arsenic	10	µg/l
Benzene	1	µg/l
Benzo(a)pyrene	0.01	µg/l
Boron	1000	µg/l
Cadmium	5	µg/l
Chromium	50	µg/l
Chloride	250	mg/l
Chloroethene	0.5	µg/l
Copper	2000	µg/l
Cyanide	50	µg/l
Fluoride	1500	µg/l
Lead	10	µg/l
Mercury	1	µg/l
Nickel	20	µg/l
Nitrate	50	mg/l
Nitrite	0.5	mg/l
Pesticide	0.1	µg/l
Sulphate	200	mg/l

Substance	Limit	Unit
Selenium	10	µg/l

* *Drinking water standard for the sum of 1,1,2,2,-Tetrachloroethene and 1,1,2-Trichloroethene*

Literature review

There is a lack of data on emerging pollutants in both, WISE-WFD and WISE-SoE databases. In order to complete the datasets, articles on various groundwater stressors were collected and respective study areas were georeferenced in order to visualize a spatial distribution of stressors and their combinations studied in literature (Fig. 2). In total, 119 articles focusing mainly on groundwater abstraction, saltwater intrusion and groundwater pollution at 645 sites throughout the Europe were gathered. Some articles documented an occurrence of emerging pollutants such as pharmaceuticals in groundwater. The list of articles is provided in Supplement A of references.

Figure 2: Study areas of stressors and their combinations documented in literature

2.1.2.2 Stressors assessed

WFD pressures as pollution, groundwater abstraction and salt water intrusion were assessed as key stressors induced by anthropogenic activities, where pollution and abstraction are stressors acting throughout whole Europe while saltwater intrusion is a problem mainly in coastal areas. A spatial extent of single forenamed stressors identified by the EU member states and their combinations was analyzed. Although chemical, quantitative and groundwater status was reported by 26 member states, the pressure information for Denmark and Wallonia region of Belgium were not available in the WFD-WISE database. On the other hand pressure data for Greece were available in the database despite the fact that there was no status information reported by Greece.

2.1.3 Results & discussion

2.1.3.1 Status

The area and count of groundwater bodies in poor chemical status (Fig. 3, Fig. 6) are much larger than groundwater bodies in poor quantitative status (Fig. 4, Fig. 6) hence the groundwater status reflect mainly poor chemical status of groundwater bodies (Fig. 5, Fig. 6). There was no information on status reported for Greece and Wallonia region of Belgium.

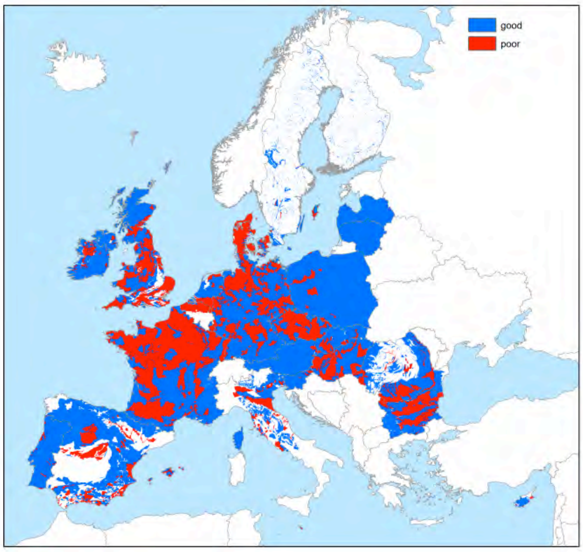


Figure 3: Chemical status

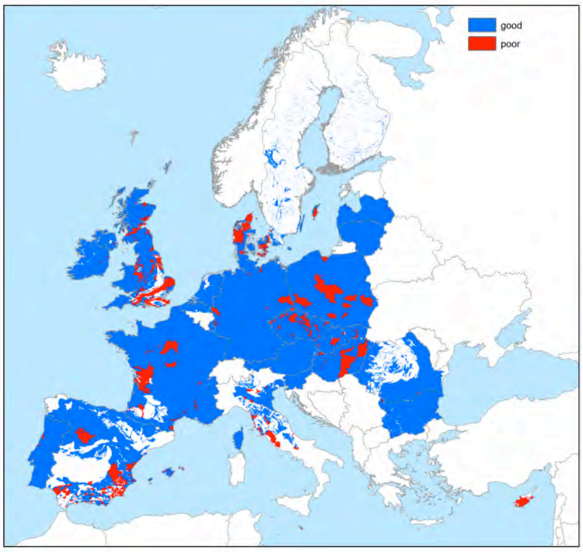


Figure 4: Quantitative status

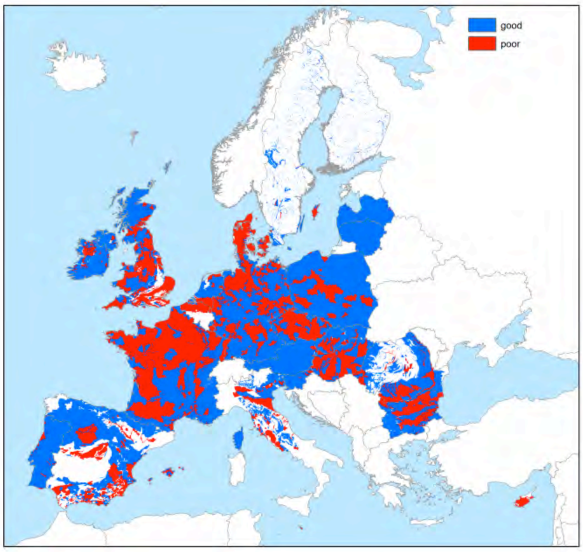


Figure 5: Groundwater status

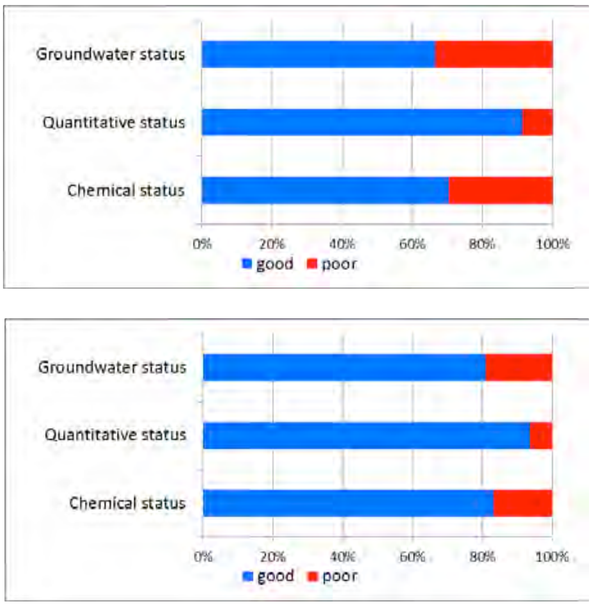


Figure 6: GWB area (top), GWB count (bottom)

2.1.3.2 Single pressures

Pollution

Pollution has been identified by EU member states as the major pressure in more than 85 % of groundwater bodies area (Fig. 7). Around 97 % of groundwater bodies area in poor chemical status (Fig. 8), 75 % of groundwater bodies area in poor quantitative status (Fig. 9) and around 90 % of groundwater bodies area in poor groundwater state (Fig.10) are influenced by pollution, predominantly from diffuse sources i.e. agriculture.

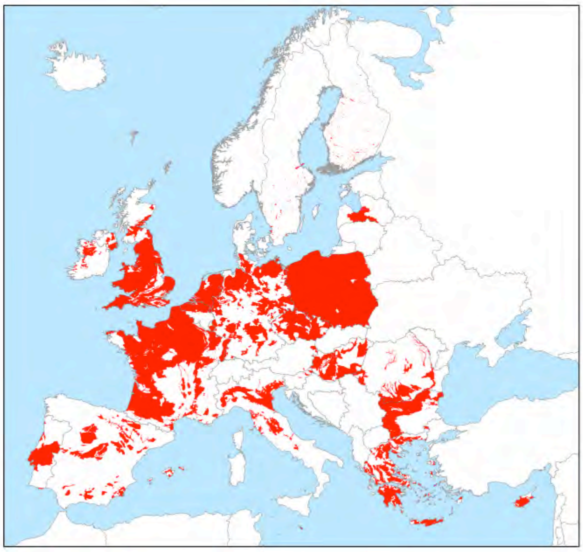


Figure 7: All GWBs

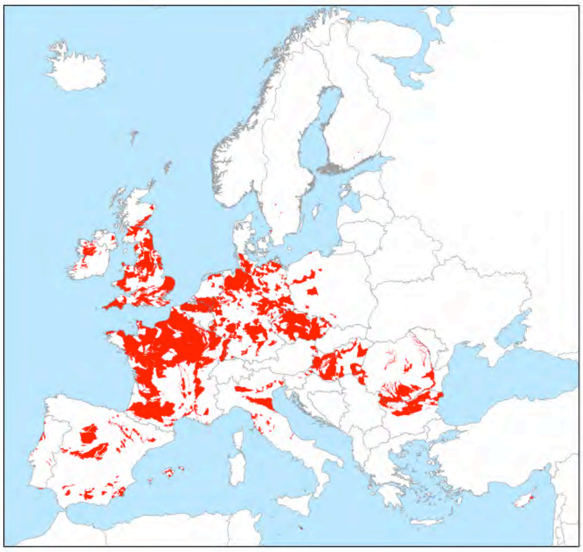


Figure 8: GWBs in poor chemical status

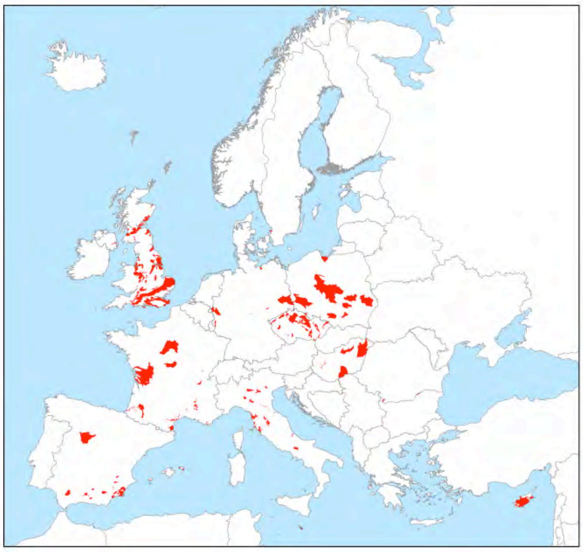


Figure 9: GWBs in poor quantitative status

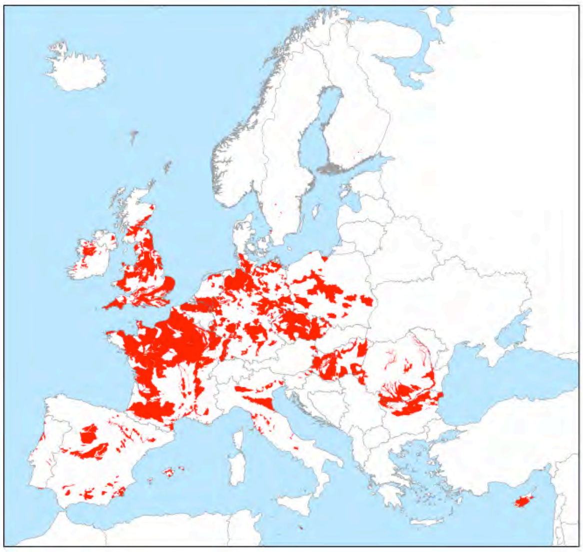


Figure 10: GWBs in poor groundwater status

Abstraction

Groundwater abstraction has been identified by EU member states as a second important pressure after pollution. More than 60 % of groundwater bodies area (Fig. 11), around 40 % of groundwater bodies area in poor chemical status (Fig. 12), 92 % of groundwater bodies area in poor quantitative status (Fig. 13) and around 48 % of groundwater bodies area in poor groundwater state (Fig.14) are influenced by groundwater abstraction.

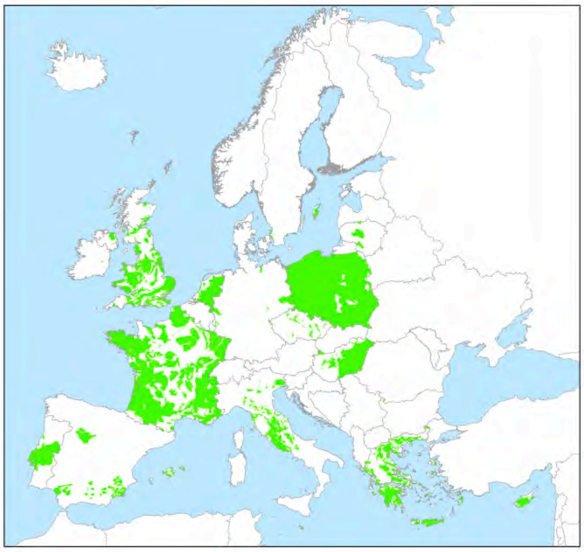


Figure 11: All GWBs

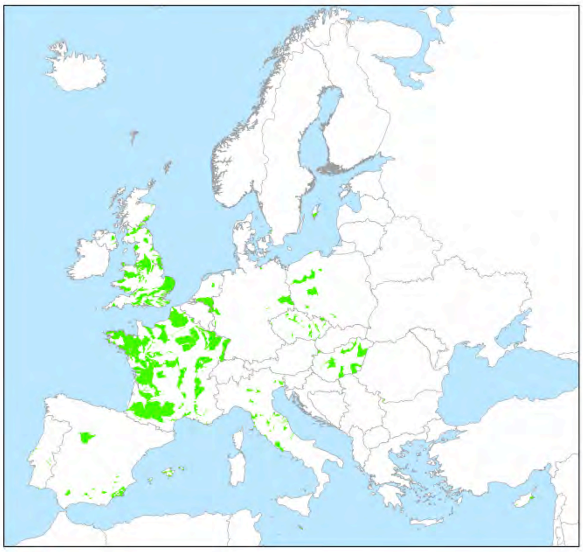


Figure 12: GWBs in poor chemical status

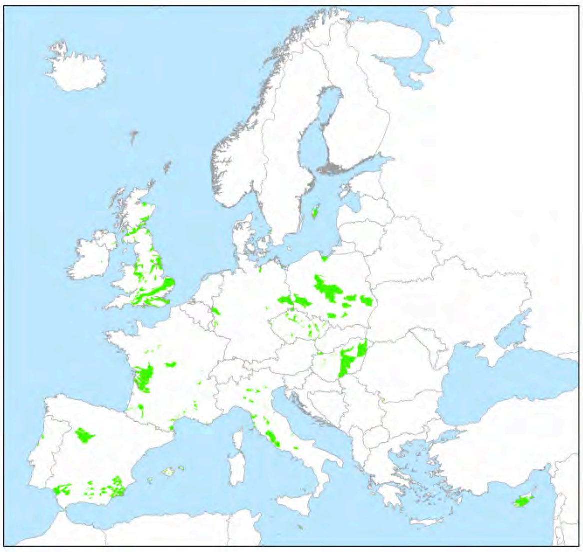


Figure 13: GWBs in poor quantitative status

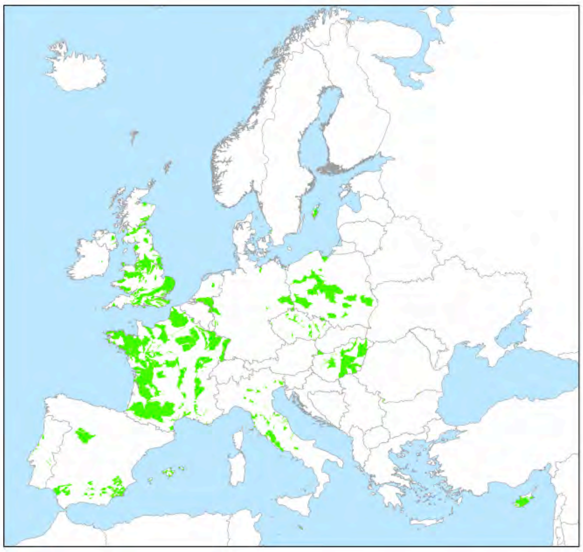


Figure 14: GWBs in poor groundwater status

Salt water intrusion

Salt water intrusion has been identified by EU member states as an important pressure by countries with coastal aquifers. In addition aquifer prone to salt water intrusion were delineated also in the International Hydrogeological Map of Europe – IHME1500 (BGR, 2013), see Fig. 15. Less than 4 % of groundwater bodies area (Fig. 15), around 3 % of groundwater bodies area in poor chemical status (Fig. 16), 10 % of groundwater bodies area in poor quantitative status (Fig. 17) and around 3 % of groundwater bodies area in poor groundwater state (Fig.18) are influenced by salt water intrusion mainly in Mediterranean region and at coast of the United Kingdom.

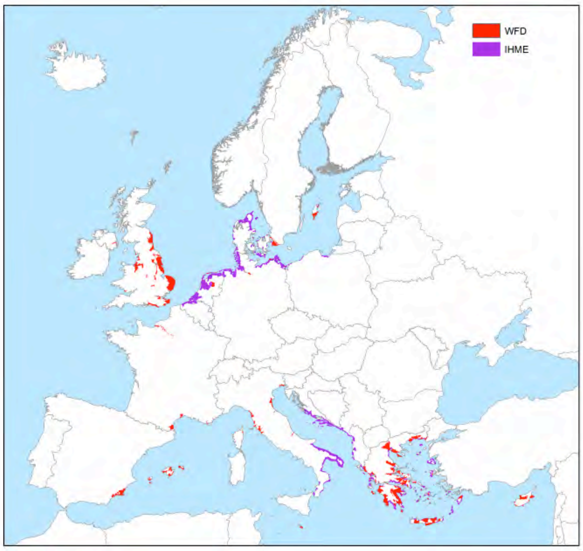


Figure 15: All GWBs

Figure 16: GWBs in poor chemical status

Figure 17: GWBs in poor quantitative status

Figure 18: GWBs in poor groundwater status

2.1.3.3 Pressure combinations

Single pressures have been identified all over the Europe (Fig. 19, Fig. 20) while combinations of 2 pressures have been reported by France, United Kingdom, Netherlands, Hungary, Italy, Greece, Spain, Portugal, Cyprus, Germany, Czech Republic, Sweden and Finland (Fig. 21). Groundwater bodies in poor groundwater status with combinations of 2 pressures were reported by France, United Kingdom, Netherlands, Hungary, Italy, Spain, Portugal, Germany, Czech Republic and in few GWBs by Cyprus, Sweden and Finland (Fig. 22). Combination of all three pressures was reported by France, United Kingdom, Netherlands, Spain, Greece and Cyprus (Fig. 23) while groundwater bodies in poor status with three pressures were reported by France, United Kingdom, Netherlands, Spain, and Cyprus (Fig. 24).

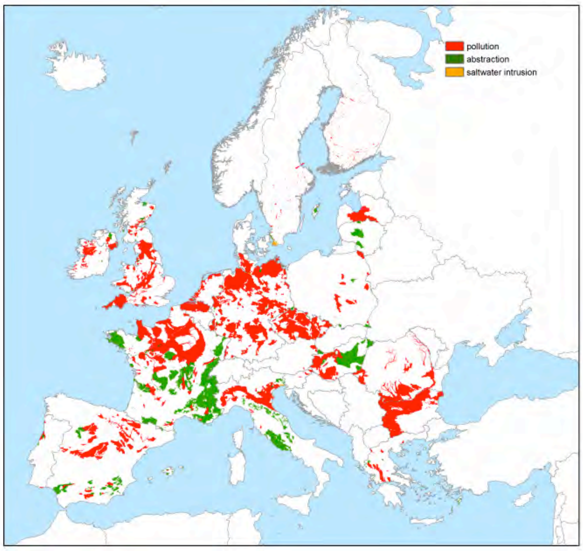


Figure 19: All GWBs – 1 pressure

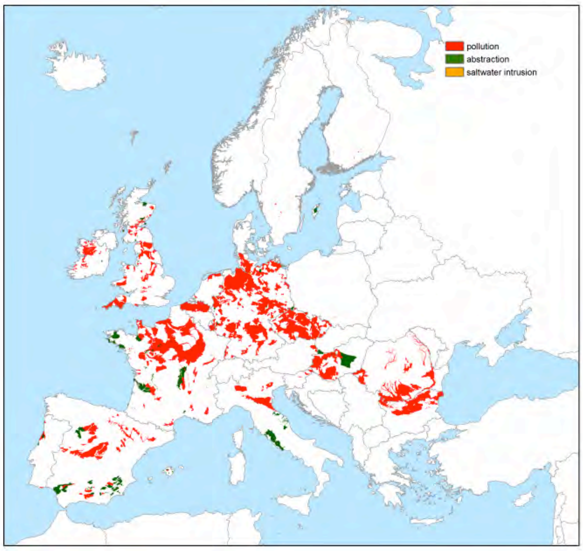


Figure 20: Poor groundwater status – 1 pressure

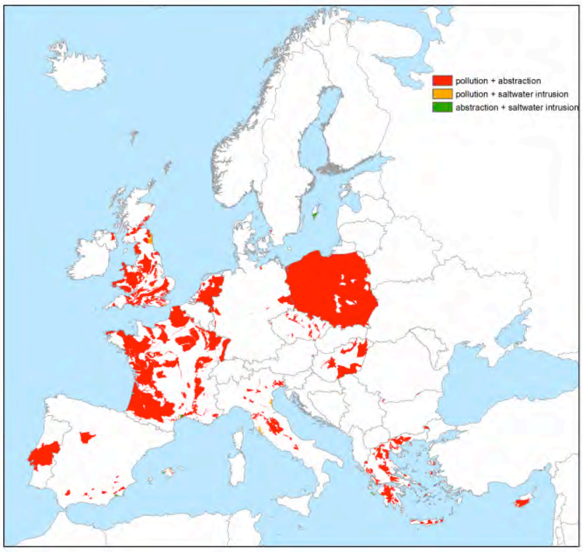


Figure 21: All GWBs – 2 pressures

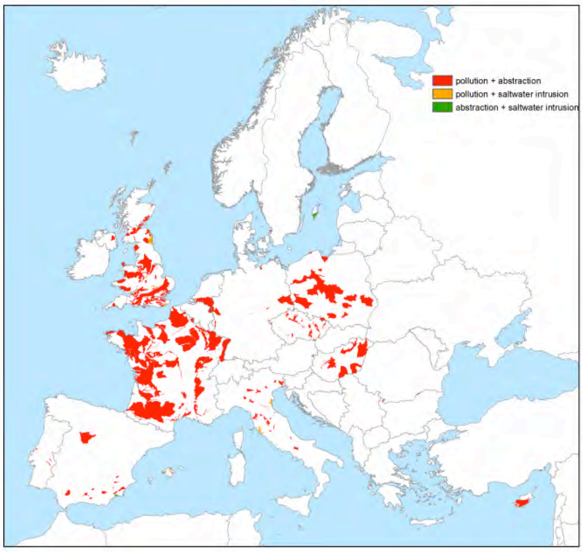


Figure 22: Poor groundwater status – 2 pressures

Spatial distribution of all pressures and their combinations in all groundwater bodies is shown in Figures 25, 26. Pressures and their combinations relevant for groundwater bodies in poor status are shown in Figures 28, 29. The most common combination of two pressures i.e. pollution and abstraction was identified in around 43 % of groundwater bodies area, 36 % of groundwater bodies area in poor chemical status, 62 % of groundwater bodies area in poor quantitative status and 38 % of groundwater bodies area in poor groundwater status. Combination of all three pressures was reported for around 3 % of groundwater bodies' area, 2 % of groundwater bodies' area in poor chemical status, 9 % of groundwater bodies' area in poor quantitative status and 2 % of groundwater bodies' area in poor groundwater status (Fig. 27).

Figure 23: All GWBs – 3 pressures

Figure 24: Poor groundwater status – 3 pressures

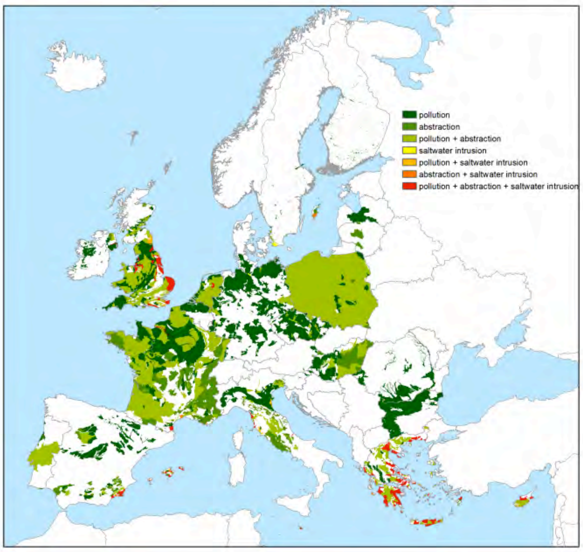


Figure 25: All GWBs

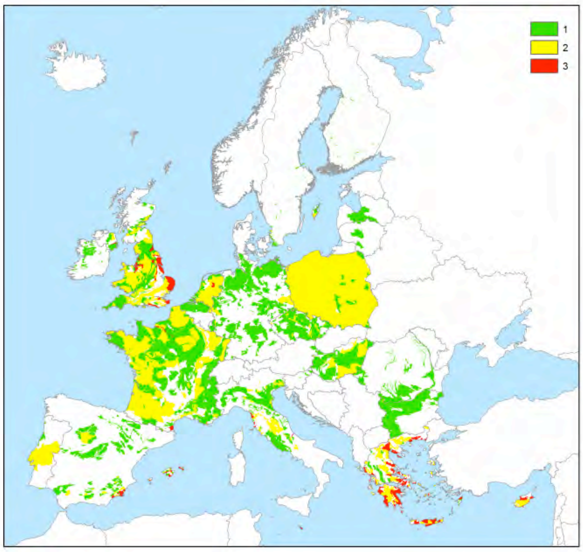


Figure 26: All GWBs – no. of pressures

If proportions are expressed by groundwater body count and not by area, the situation does not change too much, the pressure combination pollution - abstraction still prevails (Fig. 30).

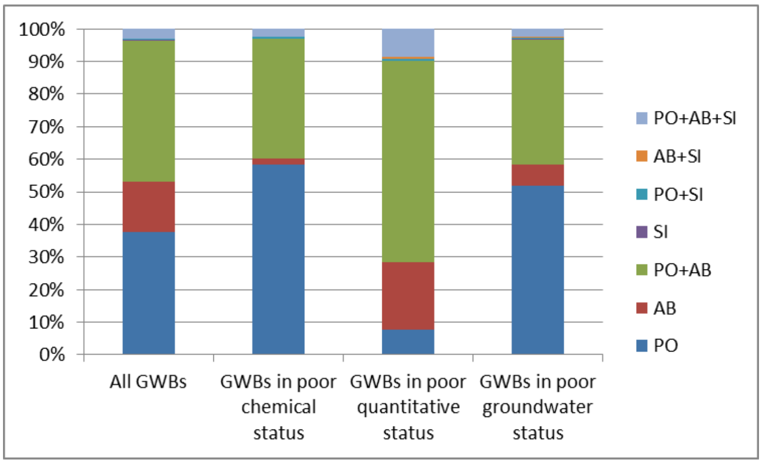


Figure 27: Pressures by GWB area (PO - pollution, AB - abstraction, SI - saltwater intrusion)

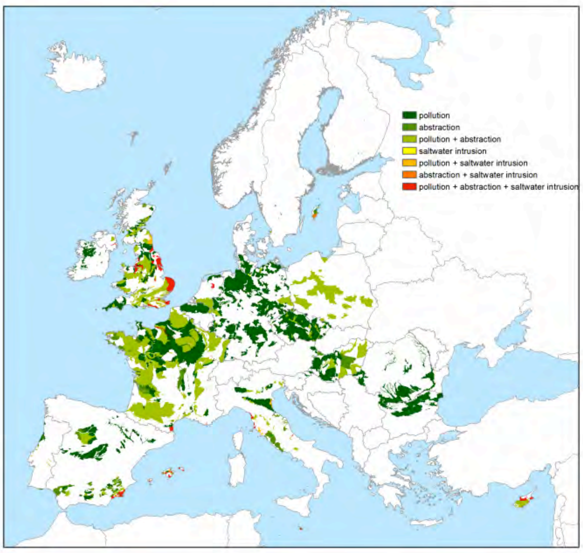


Figure 28: GWBs in poor groundwater status

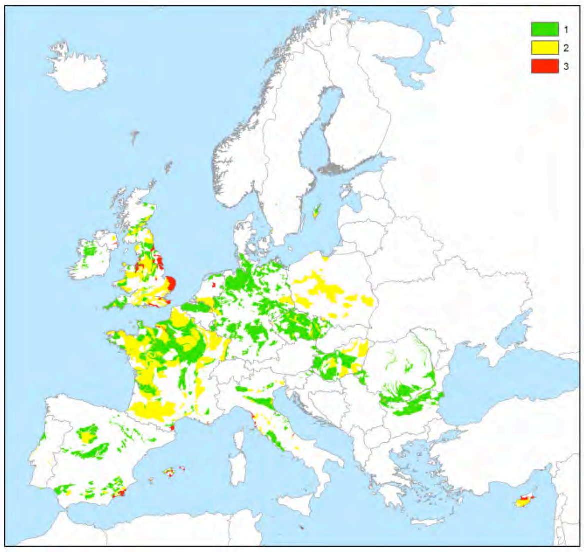


Figure 29: GWBs in poor groundwater status – no. of pressures

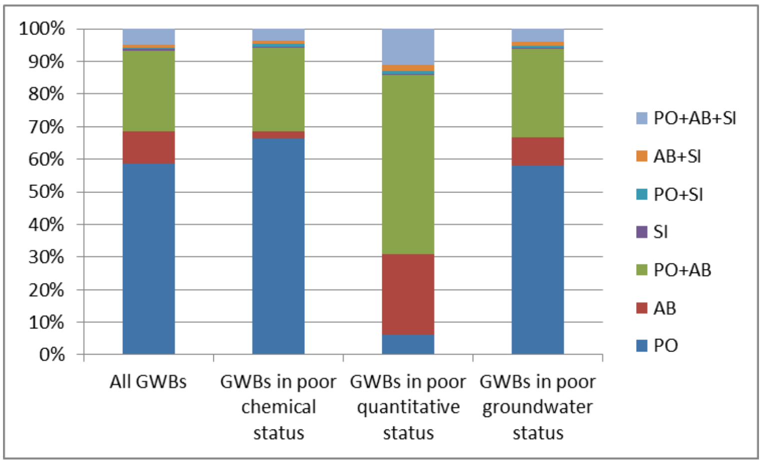


Figure 30: Pressures by GWB count (PO - pollution, AB - abstraction, SI - saltwater intrusion)

2.1.3.4 Groundwater pollutants

Groundwater pollutants can be with respect to an origin divided into two main groups:

- agricultural pollutants e.g. nutrients (mainly nitrates and ammonium) and pesticides
- other pollutants such as industrial pollutants e.g. PAHs, volatile organic compounds (VOCs) and other chlorinated solvents, metals, other organics e.g. alkyl phenols, chlorophenols, surfactants etc. and other inorganics e.g. chlorides, sulphates, electrical conductivity, other macro components etc.

The most common type of groundwater pollutants are agrochemicals such as nutrients and pesticides that are usually, unlike other pollutants, applied on soil in large areas and they leach into groundwater if the groundwater is vulnerable. On the other hand organic micro pollutants and metals are typically related with point source contamination (industrial sites, landfills) of relatively small areas compare to agrochemicals resulting in local groundwater contamination. There are 148 pollutants causing failure of good chemical status reported by EU member states (Table 4, 5), eight determinands thereof are not substance specific and may represent number of various pollutants (marked in grey in Table 5). The highest number of pollutants causing poor chemical status of single groundwater body was 21 (Fig. 31).

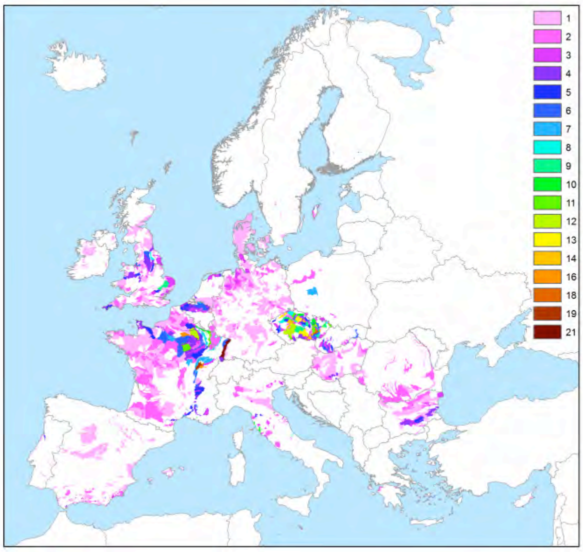


Figure 31: No. of pollutants causing failure of good chemical status in individual groundwater bodies

Table 4: Pollutants groups causing poor chemical status

Pollutant group	Subgroup	Number of pollutants
Agrochemicals	nutrients	4
	pesticides	73
Industrial pollutants	metals	16
	PAHs	8
	VOCs	19
Other organics		13
Other inorganics		15

Table 5: Pollutants causing poor chemical status

Pollutant	Group	Subgroup
Ammonium	Agrochemicals	nutrients
Nitrate	Agrochemicals	nutrients
Nitrite	Agrochemicals	nutrients
Phosphate	Agrochemicals	nutrients
Phosphorus	Agrochemicals	nutrients
2,6-Dichlorobenzamide	Agrochemicals	pesticides
Alachlor	Agrochemicals	pesticides
Aldrin	Agrochemicals	pesticides
Aminotriazole	Agrochemicals	pesticides
AMPA	Agrochemicals	pesticides
Atrazine	Agrochemicals	pesticides
Atrazine 2-hydroxy	Agrochemicals	pesticides
Atrazine desethyl	Agrochemicals	pesticides
Atrazine desethyl desisopropyl	Agrochemicals	pesticides
Atrazine desisopropyl	Agrochemicals	pesticides

Pollutant	Group	Subgroup
Benalaxyl	Agrochemicals	pesticides
Bentazone	Agrochemicals	pesticides
Bromacil	Agrochemicals	pesticides
Bromoxynil	Agrochemicals	pesticides
Carbendazim	Agrochemicals	pesticides
Carbofuran	Agrochemicals	pesticides
Chlordecone	Agrochemicals	pesticides
Chlorotoluron	Agrochemicals	pesticides
DDT-p,p’	Agrochemicals	pesticides
Diazinon	Agrochemicals	pesticides
Dieldrin	Agrochemicals	pesticides
Diflufenican	Agrochemicals	pesticides
Dichlorprop	Agrochemicals	pesticides
Dimethachlor	Agrochemicals	pesticides
Dinoterb	Agrochemicals	pesticides
Diquat	Agrochemicals	pesticides
Diuron	Agrochemicals	pesticides
Endosulfan	Agrochemicals	pesticides
Epoxiconazole	Agrochemicals	pesticides
Ethidimuron	Agrochemicals	pesticides
Fenitrothion	Agrochemicals	pesticides
Flusilazole	Agrochemicals	pesticides
Glufosinate	Agrochemicals	pesticides
Glyphosate	Agrochemicals	pesticides
Heptachlor	Agrochemicals	pesticides
Heptachlor epoxide	Agrochemicals	pesticides
Hexazinone	Agrochemicals	pesticides
HCH-alpha	Agrochemicals	pesticides
HCH-delta	Agrochemicals	pesticides
HCH-gamma	Agrochemicals	pesticides
Imidacloprid	Agrochemicals	pesticides
Isoproturon	Agrochemicals	pesticides
Linuron	Agrochemicals	pesticides
Malathion	Agrochemicals	pesticides
MCPA	Agrochemicals	pesticides
Mecoprop	Agrochemicals	pesticides
Metalaxyl	Agrochemicals	pesticides
Metazachlor	Agrochemicals	pesticides
Methomyl	Agrochemicals	pesticides
Metolachlor	Agrochemicals	pesticides
Monolinuron	Agrochemicals	pesticides
Monuron	Agrochemicals	pesticides
Nicosulfuron	Agrochemicals	pesticides
Norflurazon desmethyl	Agrochemicals	pesticides
Oxadiazon	Agrochemicals	pesticides
Oxadixyl	Agrochemicals	pesticides
Parathion	Agrochemicals	pesticides
Parathion methyl	Agrochemicals	pesticides
Pendimethalin	Agrochemicals	pesticides
Pesticides	Agrochemicals	pesticides

Pollutant	Group	Subgroup
Piperonil	Agrochemicals	pesticides
Procymidone	Agrochemicals	pesticides
Secbumeton	Agrochemicals	pesticides
Simazine	Agrochemicals	pesticides
Simazine hydroxy	Agrochemicals	pesticides
Sulcotrione	Agrochemicals	pesticides
Terbumeton	Agrochemicals	pesticides
Terbumeton desethyl	Agrochemicals	pesticides
Terbuthylazine	Agrochemicals	pesticides
Terbuthylazine 2-hydroxy	Agrochemicals	pesticides
Terbuthylazine desethyl	Agrochemicals	pesticides
Terbutryn	Agrochemicals	pesticides
Trifluralin	Agrochemicals	pesticides
Ag	Ind. pollutants	metals
Al	Ind. pollutants	metals
As	Ind. pollutants	metals
B	Ind. pollutants	metals
Cd	Ind. pollutants	metals
Co	Ind. pollutants	metals
Cr	Ind. pollutants	metals
Cu	Ind. pollutants	metals
Fe	Ind. pollutants	metals
Hg	Ind. pollutants	metals
Mn	Ind. pollutants	metals
Ni	Ind. pollutants	metals
Pb	Ind. pollutants	metals
Sb	Ind. pollutants	metals
Se	Ind. pollutants	metals
Zn	Ind. pollutants	metals
Benzo(a)pyrene	Ind. pollutants	PAHs
Benzo(b)fluoranthene	Ind. pollutants	PAHs
Benzo(g,h,i)perylene	Ind. pollutants	PAHs
Benzo(k)fluoranthene	Ind. pollutants	PAHs
Fluoranthene	Ind. pollutants	PAHs
Indeno(1,2,3-cd)pyrene	Ind. pollutants	PAHs
Naphthalene	Ind. pollutants	PAHs
PAH sum	Ind. pollutants	PAHs
1,1,1-Trichloroethane	Ind. pollutants	VOCs
1,1,2,2-Tetrachloroethene	Ind. pollutants	VOCs
1,1,2-Trichloroethene	Ind. pollutants	VOCs
1,1,2-Trichloroethene + 1,1,2,2-Tetrachloroethene	Ind. pollutants	VOCs
1,1-Dichloroethane	Ind. pollutants	VOCs
1,1-Dichloroethene	Ind. pollutants	VOCs
1,2-Dichloroethane	Ind. pollutants	VOCs
1,2-Dichloroethene	Ind. pollutants	VOCs
1,2-Dichloropropane	Ind. pollutants	VOCs
Benzene	Ind. pollutants	VOCs
Chloroethene	Ind. pollutants	VOCs
Bromodichloromethane	Ind. pollutants	VOCs
Dibromochloromethane	Ind. pollutants	VOCs

Pollutant	Group	Subgroup
Ethylbenzene	Ind. pollutants	VOCs
Toluene	Ind. pollutants	VOCs
Trichloromethane	Ind. pollutants	VOCs
VOC	Ind. pollutants	VOCs
Xylene-o,m,p	Ind. pollutants	VOCs
Xylene-p	Ind. pollutants	VOCs
Nitrate + pesticide	Other inorganics	
Annex II pollutants*	Other organics	
C10-40	Other organics	
Cyanide	Other organics	
Dichlorophenol	Other organics	
Hexachlorobenzene	Other organics	
MTBE	Other organics	
Naphta	Other organics	
Other	Other organics	
Pentachlorophenol	Other organics	
Pentochlorobenzene	Other organics	
Phenol	Other organics	
TAME	Other organics	
Tri, tetra, penta chlorophenols	Other organics	
Acid neutralisation capacity	Other inorganics	
Br	Other inorganics	
Ca	Other inorganics	
COD-Mn	Other inorganics	
Conductivity	Other inorganics	
Chloride	Other inorganics	
F	Other inorganics	
F + V	Other inorganics	
Hardness	Other inorganics	
K	Other inorganics	
Mg	Other inorganics	
Na	Other inorganics	
Sulphate	Other inorganics	

** Annex II pollutants are: As, Cd, Pb, Hg, ammonium, chloride, sulphate, 1,1,2-trichloroethene, 1,1,2,2-tetrachloroethene and conductivity*

In total, 109 pollutants exceeded drinking water standards in 5430 stations within the WISE-SoE monitoring network. Numbers of pollutants within pollutant groups and list of pollutants exceeding the drinking water standards are provided in Tables 6 and 7.

Table 6: Pollutants groups exceeding drinking water standards in SoE stations

Pollutant group	Subgroup	Number of pollutants
Agrochemicals	nutrients	3
	pesticides	77
Industrial pollutants	metals	19
	PAHs	1
	VOCs	5
Other organics		1
Other inorganics		1

Table 7: Pollutants exceeding drinking water standards in SoE stations

Pollutant	Group	Subgroup	No. of stations > standard	No. of stations <= standard
Ammonium	Agrochemical	nutrient	1550	6700
Nitrate	Agrochemical	nutrient	1472	11923
Nitrite	Agrochemical	nutrient	198	5677
2-(4-chlorophenoxy)propionic acid	Agrochemical	pesticide	7	12
2,4,5-T	Agrochemical	pesticide	3	65
2,4-D	Agrochemical	pesticide	12	282
2,4-DB	Agrochemical	pesticide	2	101
2,6-Dichlorobenzamide	Agrochemical	pesticide	96	320
Acetochlor	Agrochemical	pesticide	1	49
Alachlor	Agrochemical	pesticide	4	679
Aldrin	Agrochemical	pesticide	42	209
Alpha-Endosulfan	Agrochemical	pesticide	42	124
Alpha-HCH	Agrochemical	pesticide	40	73
AMPA	Agrochemical	pesticide	23	33
Atrazine	Agrochemical	pesticide	175	2658
Atrazine desethyl	Agrochemical	pesticide	371	2265
Atrazine desethyl deisopropyl	Agrochemical	pesticide	54	113
Atrazine desisopropyl	Agrochemical	pesticide	76	687
Atrazine hydroxy	Agrochemical	pesticide	5	100
Bentazone	Agrochemical	pesticide	152	966
Beta-Endosulfan	Agrochemical	pesticide	42	90
Beta-HCH	Agrochemical	pesticide	42	63
Bromacil	Agrochemical	pesticide	11	133
Bromoxynil	Agrochemical	pesticide	17	181
Clopyralid	Agrochemical	pesticide	28	182
Cyanazine	Agrochemical	pesticide	20	205
Dalapon	Agrochemical	pesticide	1	54
DDT + DDE + DDD	Agrochemical	pesticide	1	5
DDT, o,p'	Agrochemical	pesticide	2	216
DDT, p,p'	Agrochemical	pesticide	2	920
Desmedipham	Agrochemical	pesticide	4	4
Dicamba	Agrochemical	pesticide	23	216
Dieldrin	Agrochemical	pesticide	45	249
Dichlobenil	Agrochemical	pesticide	45	122
Dichlorprop	Agrochemical	pesticide	14	284
Dichlorvos	Agrochemical	pesticide	2	110
Diketo-metribuzin	Agrochemical	pesticide	5	19
Dimethoate	Agrochemical	pesticide	2	112
Diuron	Agrochemical	pesticide	37	638
Endrin	Agrochemical	pesticide	42	165

Pollutant	Group	Subgroup	No. of stations > standard	No. of stations <= standard
Epoxiconazole	Agrochemical	pesticide	23	98
Epsilon-HCH	Agrochemical	pesticide	1	48
Ethofumesate	Agrochemical	pesticide	13	164
Fenitrothion	Agrochemical	pesticide	2	80
Fenoprop	Agrochemical	pesticide	3	131
Fenpropimorph	Agrochemical	pesticide	1	135
Gamma-HCH	Agrochemical	pesticide	42	294
Glyphosate	Agrochemical	pesticide	31	125
Heptachlor epoxide	Agrochemical	pesticide	40	1294
Hexachlorocyclohexane (HCH)	Agrochemical	pesticide	2	70
Hexazinone	Agrochemical	pesticide	4	76
Chloridazon	Agrochemical	pesticide	4	117
Chlorpyrifos	Agrochemical	pesticide	2	197
Ioxynil	Agrochemical	pesticide	3	185
Isodrin	Agrochemical	pesticide	40	112
Isoproturon	Agrochemical	pesticide	39	528
Lenacil	Agrochemical	pesticide	2	27
Linuron	Agrochemical	pesticide	8	319
MCPA	Agrochemical	pesticide	23	409
MCPB	Agrochemical	pesticide	2	230
Mecoprop	Agrochemical	pesticide	53	417
Metalaxyl	Agrochemical	pesticide	5	61
Metamitron	Agrochemical	pesticide	4	47
Metazachlor	Agrochemical	pesticide	36	270
Methoxychlor	Agrochemical	pesticide	2	103
Metolachlor	Agrochemical	pesticide	32	224
Metribuzin	Agrochemical	pesticide	4	24
Metribuzin desamino-diketo	Agrochemical	pesticide	13	21
Metsulfuron methyl	Agrochemical	pesticide	2	54
Pendimethalin	Agrochemical	pesticide	7	196
Pirimicarb	Agrochemical	pesticide	15	201
Prometryn	Agrochemical	pesticide	29	288
Propazine	Agrochemical	pesticide	2	553
Propiconazole	Agrochemical	pesticide	3	81
Simazine	Agrochemical	pesticide	41	1846
Terbumeton	Agrochemical	pesticide	1	4
Terbuthylazine	Agrochemical	pesticide	31	375
Terbuthylazine desethyl	Agrochemical	pesticide	15	222
Terbutryn	Agrochemical	pesticide	24	327
Trifluralin	Agrochemical	pesticide	5	284
Aluminium	Industrial	metals	358	5562
Arsenic	Industrial	metals	537	4692

Pollutant	Group	Subgroup	No. of stations > standard	No. of stations <= standard
Arsenic dissolved	Industrial	metals	63	909
Boron	Industrial	metals	77	5743
Cadmium	Industrial	metals	34	3694
Cadmium dissolved	Industrial	metals	15	1299
Copper	Industrial	metals	9	7283
Copper dissolved	Industrial	metals	3	4928
Chromium	Industrial	metals	25	4729
Chromium 3^+	Industrial	metals	1	47
Chromium 6^+	Industrial	metals	2	191
Chromium dissolved	Industrial	metals	3	1352
Lead	Industrial	metals	430	4654
Lead dissolved	Industrial	metals	46	1639
Mercury	Industrial	metals	27	2353
Mercury dissolved	Industrial	metals	7	199
Nickel	Industrial	metals	421	6762
Nickel dissolved	Industrial	metals	205	3201
Selenium	Industrial	metals	16	895
Benzo(a)pyrene	Industrial	PAH	93	302
1,1,2,2-Tetrachloroethene	Industrial	VOC	129	1247
1,1,2-Trichloroethene	Industrial	VOC	70	905
1,2-Dichloroethane	Industrial	VOC	7	814
Benzene	Industrial	VOC	15	438
Chloroethene	Industrial	VOC	6	96
Chloride	Other inorganics		269	8248
Hexachlorobenzene	Other organics		42	117

Agrochemical pollutants

Nitrates, nitrites and ammonium as forms of nitrogen occurring in groundwater and phosphorus were considered nutrients in this study. Nutrients and pesticides as commonly EU wide used agrochemicals were assessed together as group of agrochemical pollutants.

Ammonium

Ammonium caused poor chemical status in the United Kingdom, Belgium, France, Spain, Portugal, Italy, Czech Republic, Poland, Romania, Bulgaria and Hungary (Fig. 32) while drinking water standards were exceeded in almost every country monitoring ammonium (Fig. 33). Interestingly high concentrations of ammonium in Po valley in Italy were not reflected in chemical status assessment.

Figure 32: Poor chemical status due to ammonium

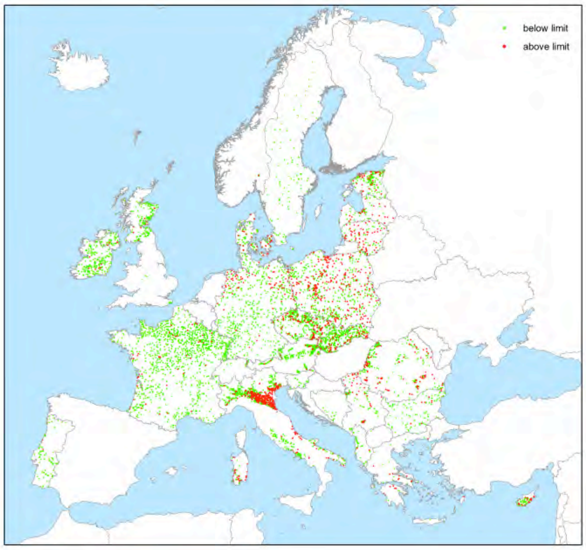


Figure 33: Ammonium concentrations (SoE)

Nitrates

Nitrate is the most common groundwater pollutant worldwide. Nitrate caused poor chemical status in every single EU member state except Baltic and Scandinavian countries (Fig. 34). SoE network shows widespread pollution by nitrate all over the Europe except Sweden, Albania and Bosnia.

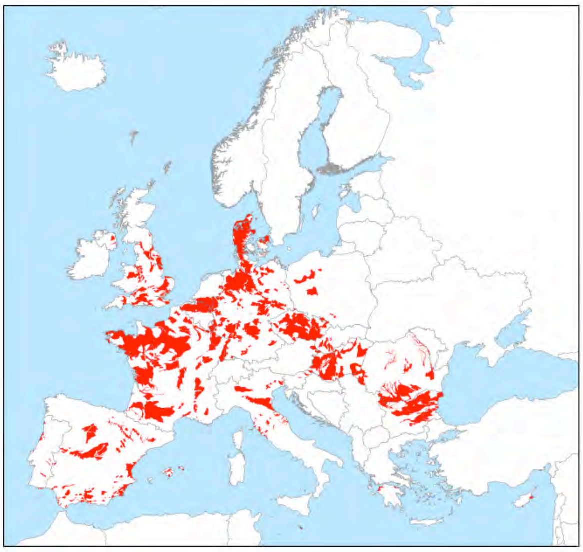


Figure 34: Poor chemical status due to nitrates

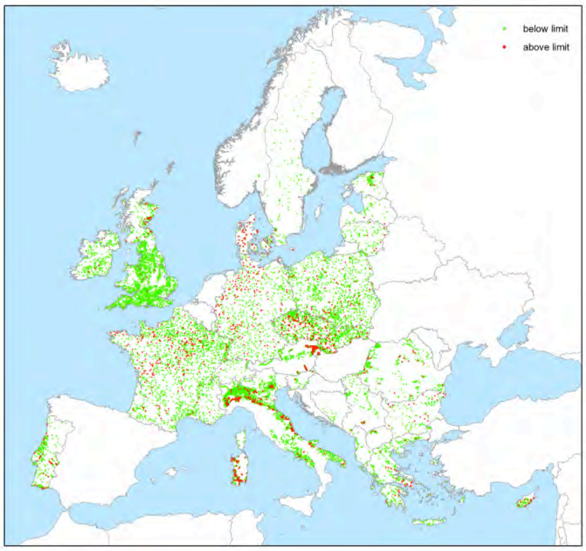


Figure 35: Nitrate concentrations (SoE)

Nitrites

Nitrite as the least frequent form of nitrogen is rarely monitored in high concentrations. Poor chemical status due to nitrite was recorded in the Czech Republic and Italy only (Fig. 36). Concentrations exceeding drinking water standard were found mainly in Italy and in less extent in Czech Republic, Germany, United Kingdom, Ireland, France, Denmark, Poland, Lithuania, Greece and Austria (Fig. 37).

Figure 36: Poor chemical status due to nitrites

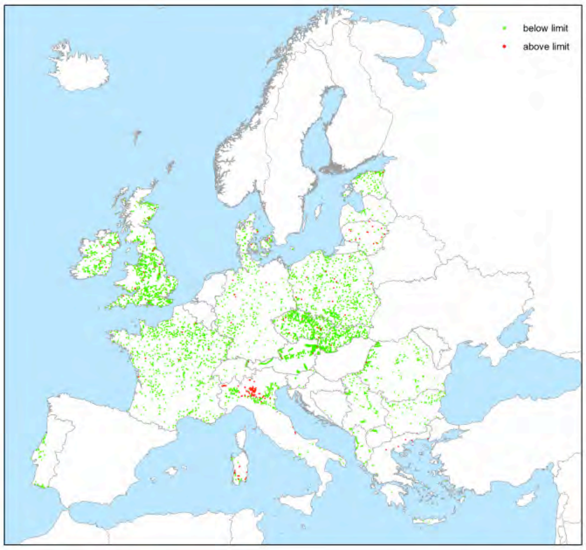


Figure 37: Nitrite concentrations (SoE)

Nutrients

Nutrients (in descending orders nitrates, ammonium, phosphorus and nitrites) are causing poor chemical status (Fig. 38) of 20% of groundwater bodies area (cca 670 000 km^2, cca 15 % of EU area). Groundwater quality regarding nutrients is thus fully dominated by nitrates; spatial distribution of concentration levels of nutrients in Europe is shown in figure 39.

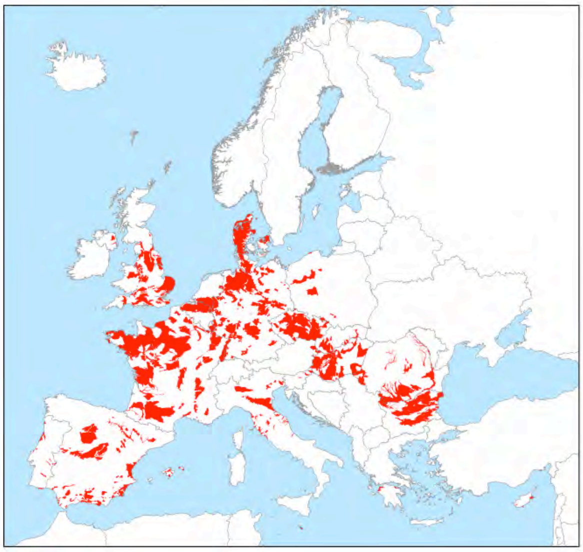


Figure 38: Poor chemical status due to nutrients

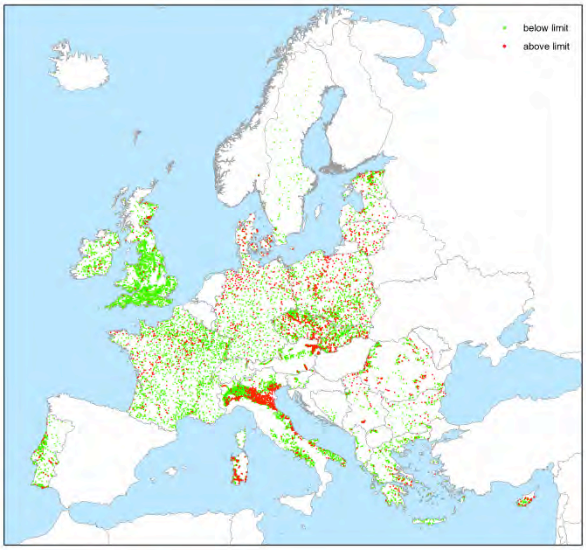


Figure 39: Nutrients concentrations (SoE)

Pesticides

Pesticides caused poor chemical mainly in western and central Europe. This might be caused by the fact that pesticide monitoring is costly, demanding state of art analytical instrumentation and thus may be limited in certain countries. Large groundwater areas in poor chemical status due to pesticides were reported by FR, BE, UK, DE and CZ. Smaller areas were reported by DK, IT, HU, SK, ES, S and FI (Fig. 40). In total 73 pesticides (Tab. 5, 6) caused poor chemical status in the EU. Contamination by pesticides in the SoE network is shown on Figure 41. The figure confirms contamination by in total 77 pesticides (Tab. 7, 8) in western and central Europe, namely in France, United Kingdom, Belgium, Denmark, Germany, Italy, Austria, Czech Republic and Slovakia. 10 % of groundwater bodies' area (Fig. 45) and 4 % of GWBs (Fig. 46) are in poor chemical status due to pesticides. Atrazine and its metabolites occur in high concentrations most frequently.

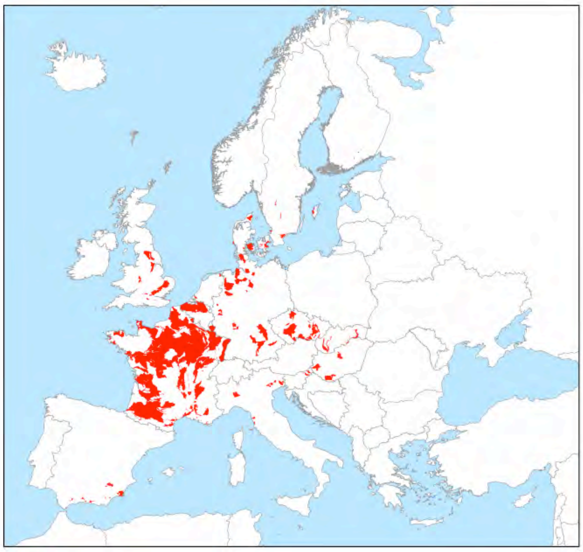


Figure 40: Poor chemical status due to pesticides

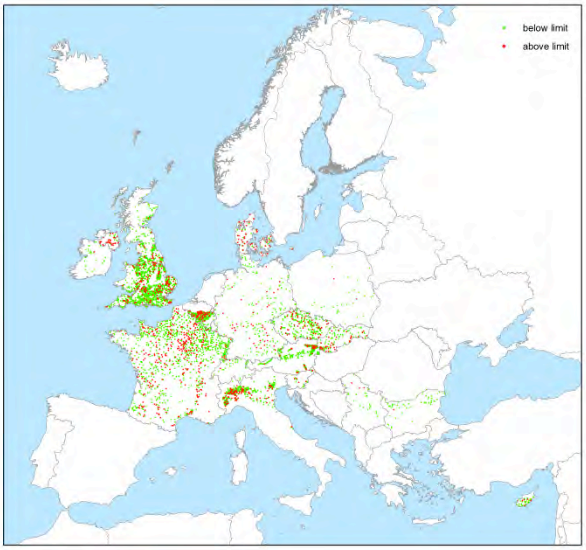


Figure 41: Pesticides concentrations (SoE)

Agrochemicals

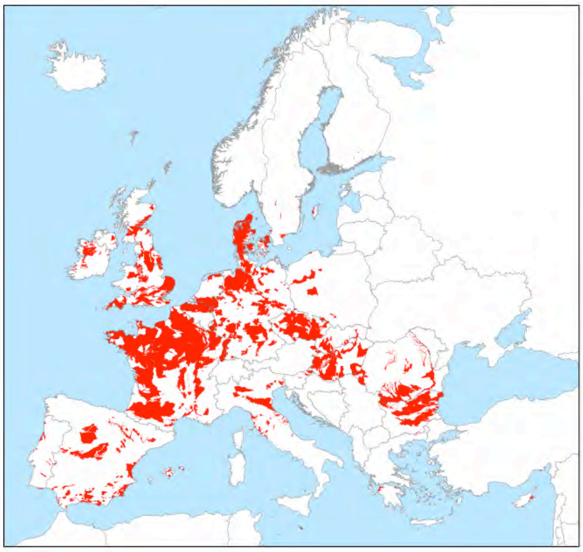


Figure 42: Poor chemical status due to agrochemicals

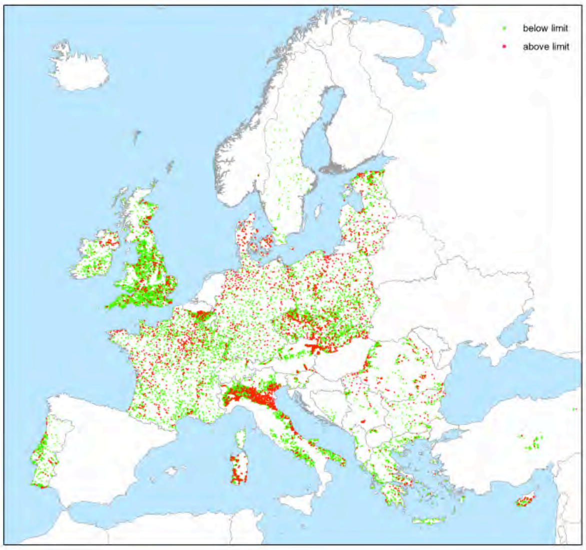


Figure 43: Agrochemical concentrations (SoE)

An overview of poor chemical status due to agrochemicals and spatial distribution of increased concentrations of agrochemicals throughout the Europe is shown on Figures 42 and 43. The highest number of agrochemicals causing poor chemical status of individual groundwater body is 20 (Fig. 44). 26 % of groundwater bodies' area (Fig. 45) and 12 % of GWBs (Fig. 46) are in poor chemical status due to agrochemicals.

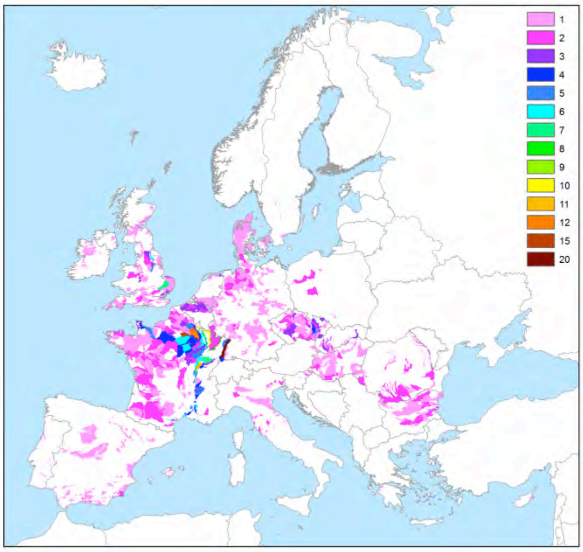


Figure 44: No. of agrochemicals causing failure of good chemical status of individual GWBs

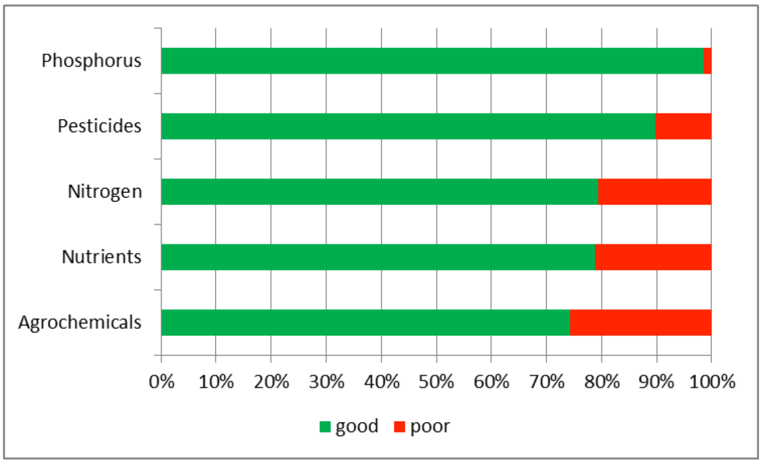


Figure 45: Proportion of area of GWBs in good/poor chemical status due to agrochemicals

The assessment especially concerning pesticides may be biased by various monitoring strategies used by Member States; there is lack of comparable reported data on pesticide metabolites that may occur frequently and in higher concentrations than parent compounds. If the harmonized monitoring strategies throughout the EU were used, results of this assessment on agrochemicals would be probably even less favorable, but they would reflect a reality more precisely.

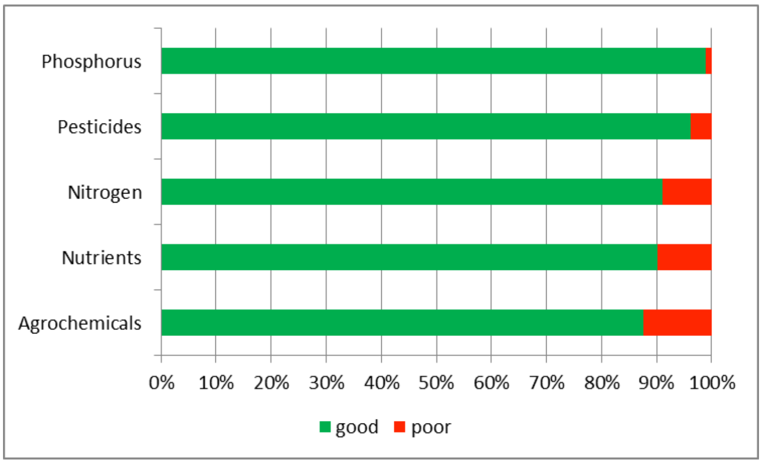


Figure 46: Proportion of number of GWBs in good/poor chemical status due to agrochemicals

Industrial and other pollutants

This group of pollutants contrary to the agrochemicals may be relevant just for certain regions in Europe or even individual sites since these pollutants are generally related to point sources or sources of relatively small areas such as industrial sites, landfills, mines etc. Figure 47 shows a spatial extent of groundwater bodies in poor chemical status and number of pollutants causing poor chemical status of individual groundwater bodies. Increased concentrations of industrial pollutants in SoE network is provided on Figure 48. Percentage of groundwater bodies in poor chemical status due to metals, VOCs, other industrial pollutants and PAHs in descending order is shown on Figure 49.

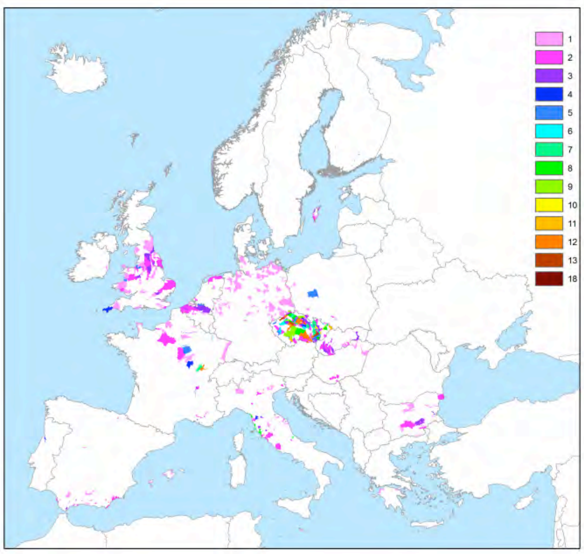


Figure 47: No. of industrial pollutants causing failure of good chemical status of individual GWBs

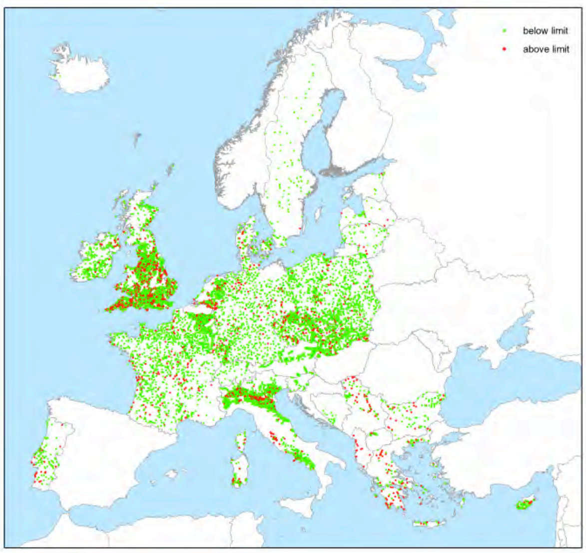


Figure 48: Industrial pollutants concentrations

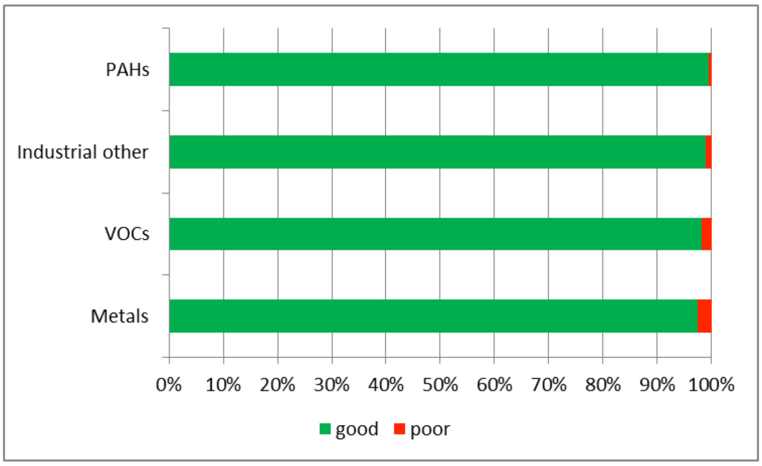
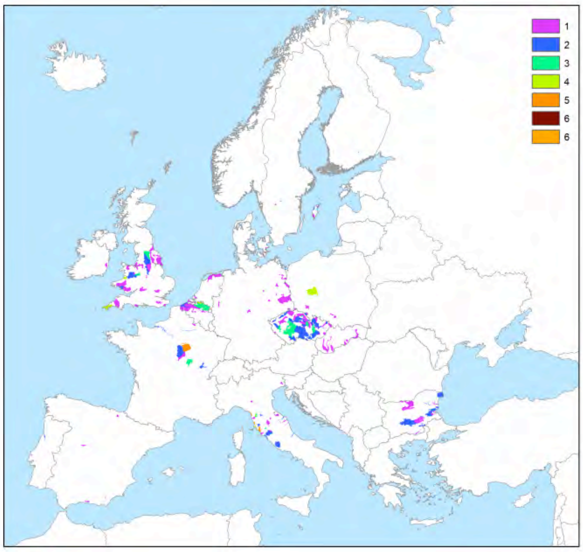


Figure 49: Percentage of number of GWBs in good/poor chemical status due to pollutants other than agrochemicals

Less than 3 % of groundwater bodies fail good chemical status due to metals. Up to 6 metals cause poor chemical status of individual groundwater bodies mainly in western and central Europe (Fig.50). Elevated concentrations of metals occur mainly in the United Kingdom and Po valley in Italy. Aluminum, arsenic, lead and nickel cause poor chemical status and exceed the drinking water standard most often.



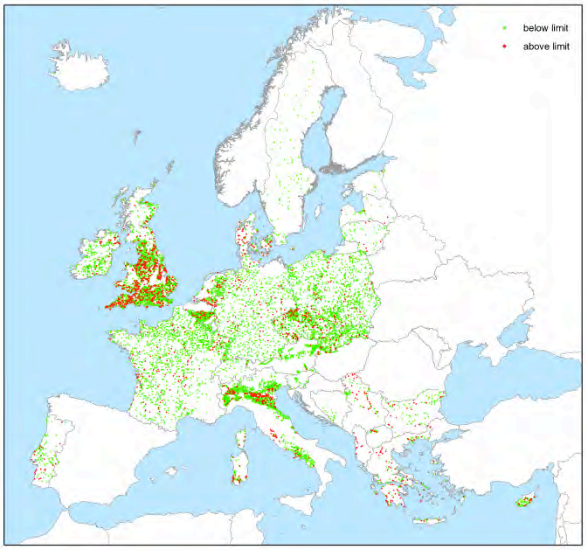


Figure 50: Poor chemical status due to metals *Figure 51: Metal concentrations (SoE)*

Less than 2 % of groundwater bodies fail good chemical status due to VOCs. Up to 8 VOCs cause poor chemical status of individual groundwater bodies mainly in western and central Europe (Fig.52). Elevated concentrations of VOCs occur mainly in the Czech Republic, France, United Kingdom and Italy. Chlorinated solvents (1,1,2,2-tetrachloroethene, 1,1,2-trichloroethene) and benzene cause poor chemical status and exceed the drinking water standard most often.

Only 0.5 % of groundwater bodies fail good chemical status due to PAHS. Up to 9 PAHs cause poor chemical status of individual groundwater bodies in the Czech Republic only (Fig.54) although elevated concentrations of PAHs were also found elsewhere (Fig.55).

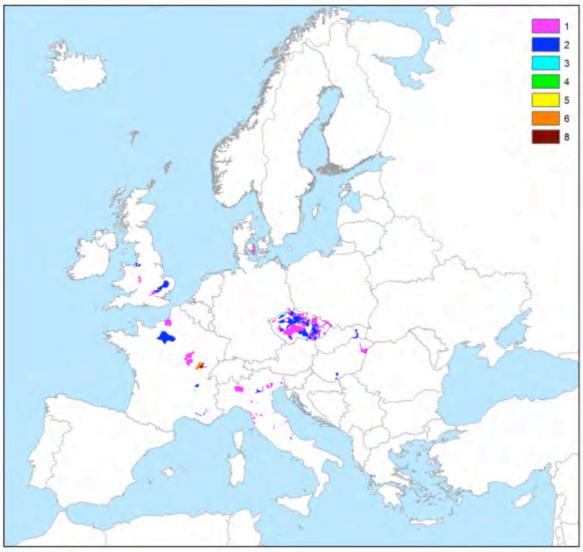


Figure 52: Poor chemical status due to VOCs

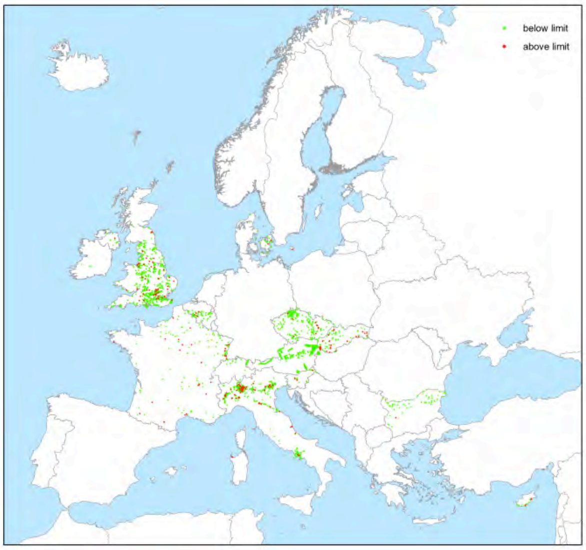


Figure 53: VOC concentrations (SoE)

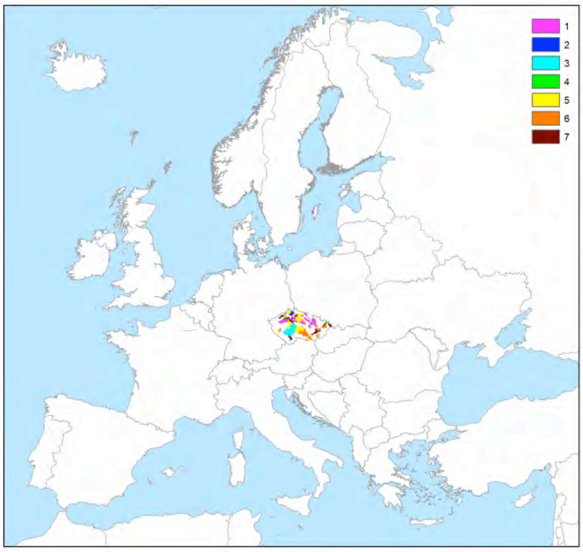


Figure 54: Poor chemical status due to PAHs

Figure 55: PAH concentrations (SoE)

The other pollutants are mainly represented by the Annex II group of pollutants causing failure of good chemical status reported namely by Germany, Netherlands and United Kingdom with no further clarification of individual pollutants. The group Annex II consists according to the Directive 2006/118/EC on the protection of groundwater against pollution and deterioration of following pollutants: arsenic, cadmium, lead, mercury , ammonium, chloride, sulphate, 1,1,2-

trichloroethene, 1,1,2,2-tetrachloroethene, and conductivity. Spatial extent of groundwater bodies in poor chemical status due to other pollutants and number of pollutants causing failure of individual groundwater bodies is shown on Figure 56.

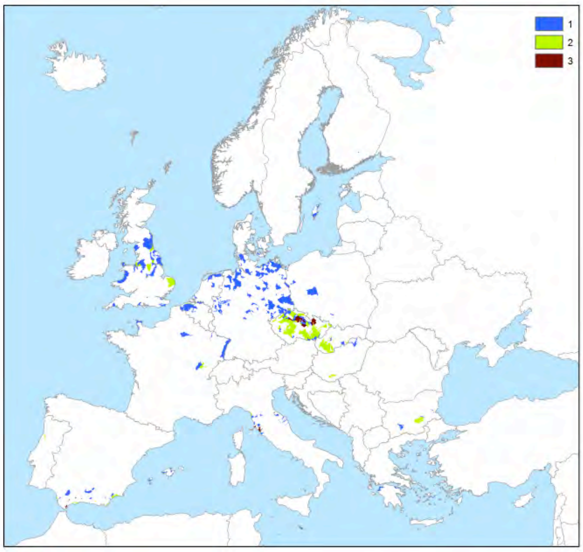


Figure 56: Poor chemical status due to other pollutants

Pollutants - summary

Agrochemicals contribute to the poor chemical status by far the most and industrial pollutant the least (Fig. 57). Group of other pollutants very probably represents partly industrial pollutants and also partly agrochemicals as it was clarified hereinbefore.

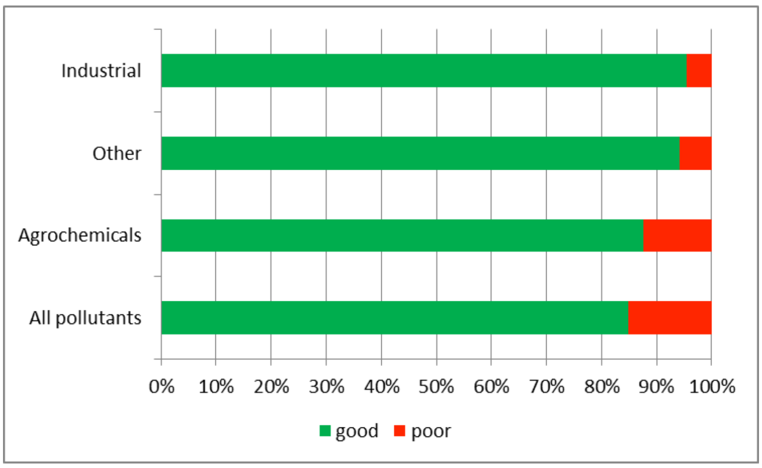


Figure 57: Percentage of number of GWBs in good/poor chemical status due to all pollutants

As information complementary to the WISE-WFD and WISE-SoE data, published articles (Supplement A of references) focusing on selected stressors and their combinations were processed and mapped. Number of articles concerning respective stressors is shown on Figure 58. Number of articles concerning various pollutant groups is provided in Figure 59.

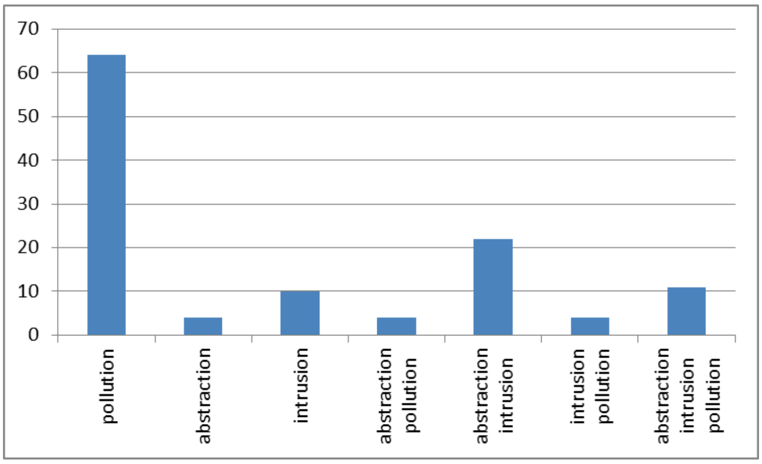


Figure 58: Number of articles concerning respective stressors

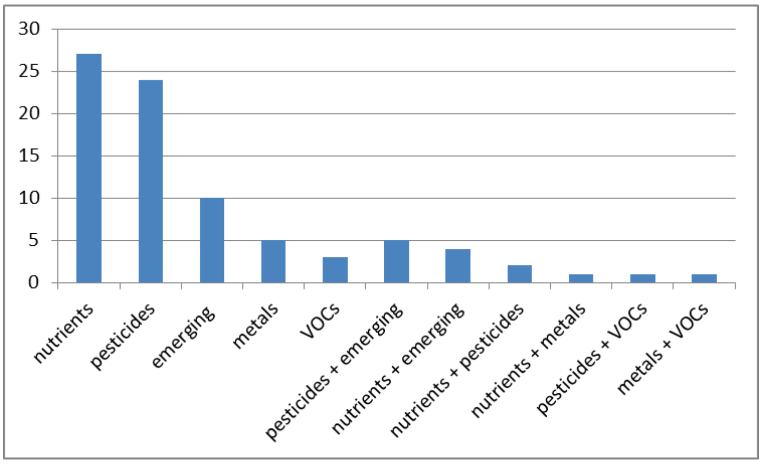


Figure 59: Number of articles concerning various pollutant groups

A spatial distribution of studied stressors and their combinations in Europe is given on Figure 60. Majority of sites are located in United Kingdom, Spain, Italy and Austria. Majority of articles focused on nutrients and pesticides, just 19 dealt with emerging pollutants namely pharmaceuticals, surfactants, artificial sweeteners, UV filters and other personal care products, hormones, (Fig. 59) at sites located in Austria (34 sites), Spain (5), Germany (6), Denmark (1) and Sweden (1) (Fig. 61). Results indicated an occurrence of such emerging pollutants in groundwater.

Still the agrochemicals and mainly nutrients remain a major problem concerning groundwater quality EU wide. Extensity of the problem is shown on Fig. 62, where data from all three utilized data sources are put together whereas other pollutants may be of interest just in certain areas (Fig. 63) with exception for metals that can be present in groundwater in elevated concentrations due to high natural background concentrations.

Since two different datasets were used in the study coming from two different assessments and having various spatiotemporal resolution and scale, both datasets provide a bit different results in some areas possibly leading to different conclusions. On the other hand combination of such two datasets can increase understanding of the subject and provide comprehensive outcomes.

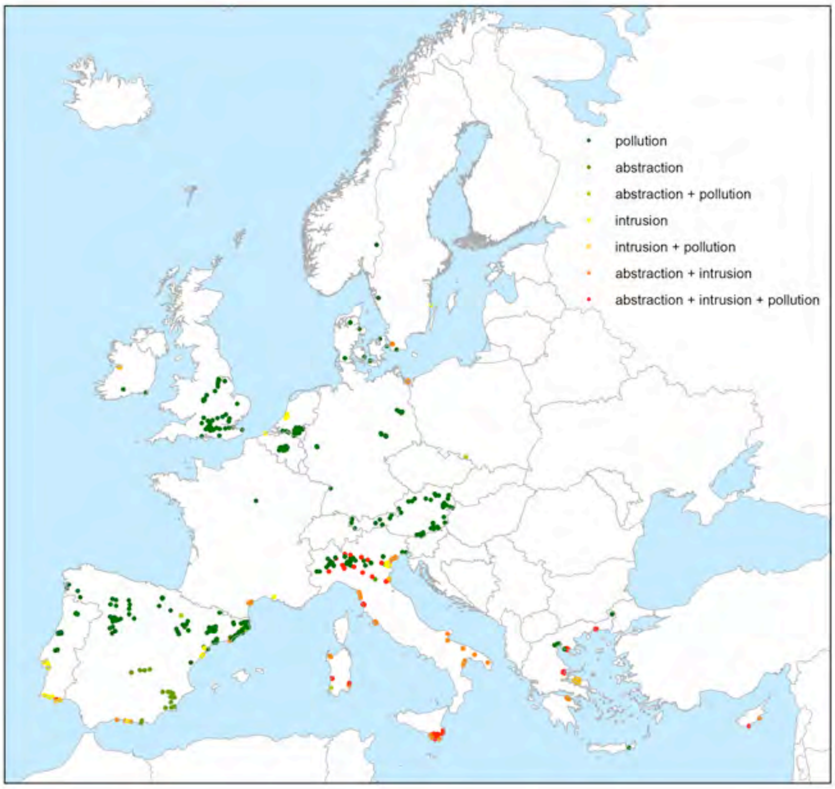


Figure 60: Stressors and their combinations documented in literature

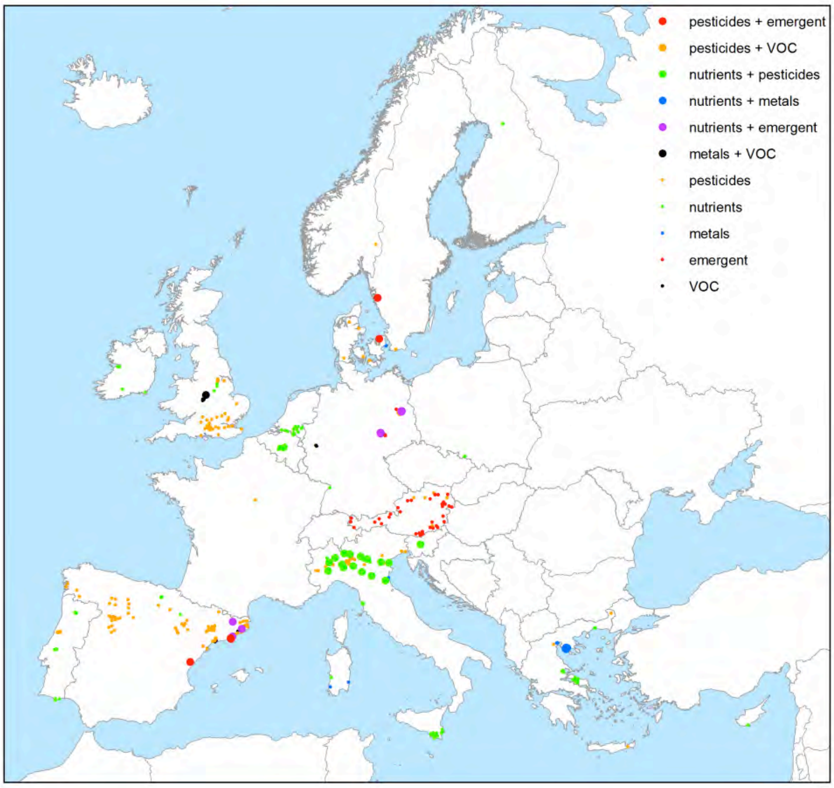


Figure 61: Pollutant groups documented in literature

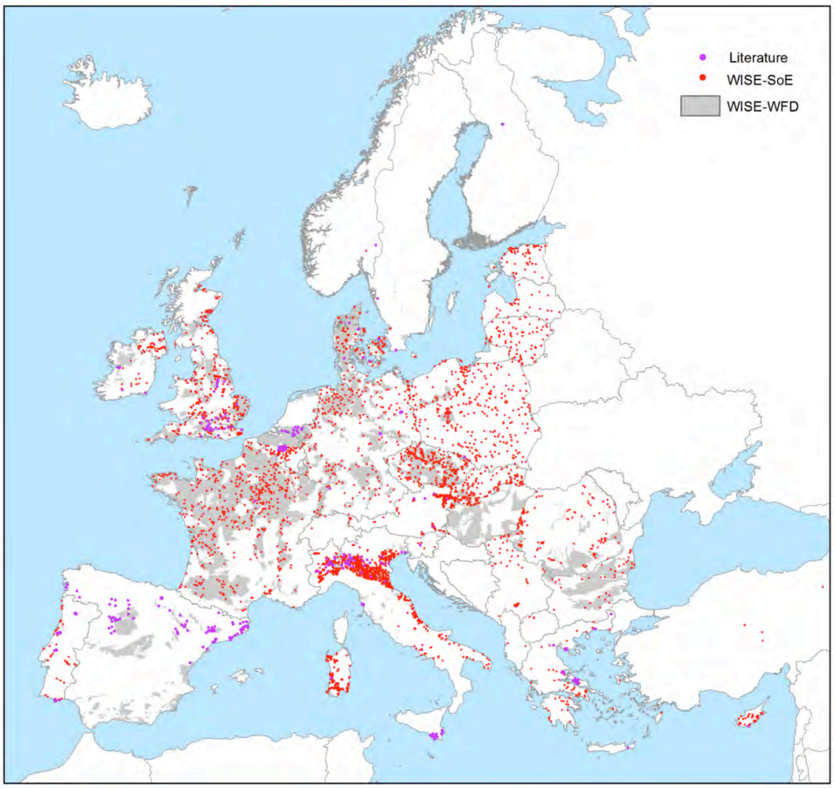


Figure 62: Areas of increased concentrations of agrochemicals in groundwater

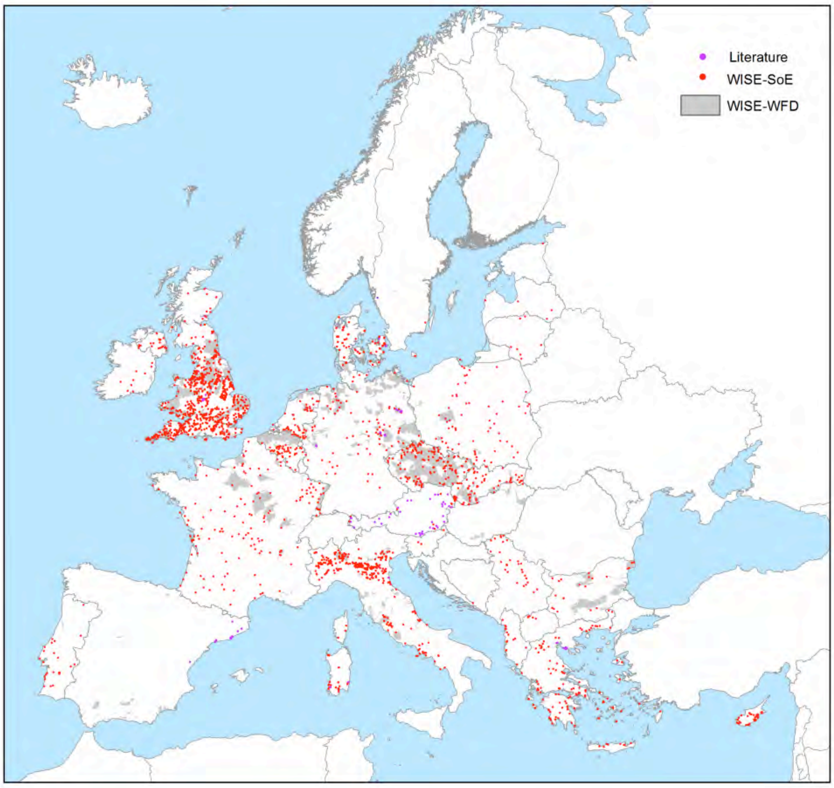


Figure 63: Areas of increased concentrations of other pollutants (metals, VOCs) in groundwater

Figure 64: Areas of increased concentrations of metals in groundwater

2.2 Statistical modelling of groundwater status at the European scale

2.2.1 Aims of the statistical modelling

The aim of the modelling exercise was to estimate statistical models of the groundwater body chemical status (Figure 1a) and quantitative status (Figure 1b) and to use these models to identity pressures on groundwater bodies at the European-scale. Furthermore, we sought to better understand the interactions between different pressures on groundwater status.

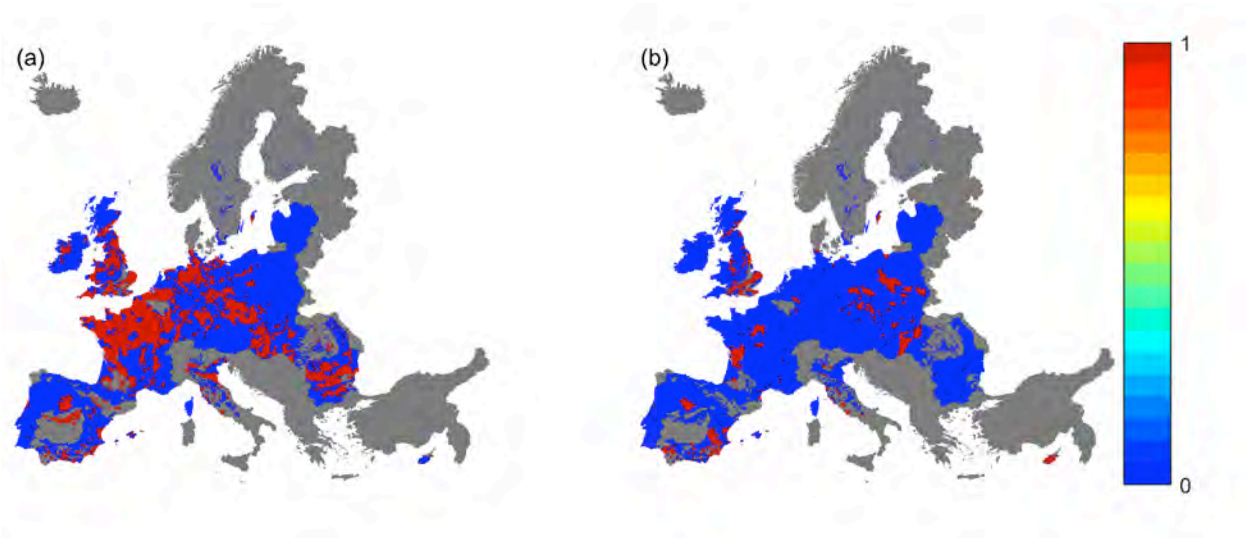


Figure 1: Recorded groundwater body chemical status (a) and groundwater body quantitative status (b) across the European Union (EU). Poor status is designated '1' (red) and good status is designated '0' (blue). Areas where no status information is available and that have not been modelled are shown in grey.

2.2.2 Data description

2.2.2.1 Response variables - Groundwater chemical and quantitative status

The chemical and quantitative status of groundwater across Europe as reported by EU member states under the 2000/60/EC directive (Water Framework Directive, WFD) in 2009 is shown in Figs 1a and 1b respectively.

These maps show the status on a 1km raster and have been derived from the groundwater GIS reference layer (EEA, 2013) containing 13345 groundwater bodies throughout the EU (Duscher, 2013). Groundwater bodies in the dataset, consistent with the Guidance document No. 9 on implementing the Geographical information elements (GIS) of the WFD (EC, 2003), are layered in 7 horizons representing distinct vertical layers of groundwater bodies. Information on chemical and quantitative status is held for each groundwater body within a horizon. Individual horizons were rasterized from polygons into grids. The grids of individual horizons were merged into one grid using map algebra and cell statistics based on the simplifying assumption that the status of the uppermost groundwater body within vertical sequence of groundwater bodies lying on top of each other (if there is one) determines the status of a respective grid cell. Additional information on total area of groundwater bodies, horizon and groundwater body ID were added to a grid value attribute table as well. The resulting grid represents 11139 groundwater bodies from 6 horizons (Table 1).

Table 1: Number of groundwater bodies within each horizon represented in groundwater bodies status grid (horizon 0 – no horizon assigned by EU member state, horizon 1 – shallowest, horizon 5 – deepest)

WFD - Groundwater Horizon	Number of Groundwater Bodies
0	438
1	9434
2	1121
3	111
4	31
5	4
Total	11139

2.2.2.2 Predictor variables

Fifteen predictor variables have been used in the models with an additional data set, European state boundaries, used as a fixed effect in one of the models. The data sets used are as follows:

- summer temperature
- annual precipitation, winter precipitation and summer precipitation
- proportion of arable, grassland, forest
- population and total abstraction
- modelled P and N, and
- distance to coast.

The predictor datasets have been taken from the MARS GeoDatbase.

2.2.3 Methods

The statistical models used the various 1-km gridded datasets described above as predictors of the two groundwater body status response variables. Prior to estimating these models, each gridded predictor dataset was standardised to a Gaussian variable with zero mean and unit variance to ensure that any unusual distribution of predictor values did not unduly influence the resultant model.

Within the earth science community, machine learning techniques are often used to form predictive relationships between gridded-datasets such as these (McBratney et al., 2016). The model might take the form of a classification tree in which the response category for the grid cells is predicted based on a series of subdivisions of the set of cells based on the values of the different predictor variables. Efficient algorithms exist to determine the optimal subdivisions that lead to accurate predictions (Michie et al, 1994). Such methods have proved to be particularly effective in producing accurate maps of soil properties at the national, continental and global scales (McBratney, et al., 2016). However, although these approaches lead to accurate maps and predictions, this does not necessarily mean that the various terms within the model reflect causal relationships between the properties of interest and the various predictors. They could instead result from correlations between variables that occur by chance.

There are a number of reasons why non-causal relationships might be included within a classification tree or any complex statistical model. Some form of hypothesis test is usually applied to decide whether a particular term should be included in the model. The test might indicate, for example, that there is a less than 1 in 20 probability that the observed correlation could have occurred by chance. If a single predictor variable is being considered, then such a test is a reasonable safeguard against including spurious relationships. However, when estimating a more complex model many hundreds or thousands of potential relationships might be considered. Therefore, one would expect that even if none of the relationships reflected a causal link that a sizeable number of them would pass the test. Lark et al. (2007) discuss the strategies that can be applied to account for multiple hypothesis tests. Also, the hypothesis tests are often based on the assumption that the available observations are independently sampled when in reality there is likely to be substantial auto-correlation amongst observations of a gridded variable. There could also be correlations between the various predictor variables which mean that even when a causal relationship is present, it can be difficult to determine which of the variables is the driver of the relationship.

We took steps to guard against such non-causal relationships appearing in our models. Rather than using the complete set of more than 2.5 million grid cells to estimate the models, we formed a calibration data set of 10,000 cells by random sampling with replacement. We treated this calibration data as if they were independent observations. We also restricted the number of hypothesis tests that were made and only considered the relationships that an expert considered to be likely combinations of drivers of groundwater stuatus. We avoid the inclusion of two highly correlated predictor variables within the model. Finally, rather than basing our models on flexible but complex classification trees we consider simpler but more tractable linear relationships between variables.

We predicted the groundwater status at n sites using a logistic mixed effects model:

$$\ln\left(\frac{\mathbf{p}}{1-\mathbf{p}}\right) = \mathbf{M\beta} + \mathbf{Zu}.$$

Here, **p** is a length n vector containing the probability of poor groundwater status at each of the sites. These probabilities are a function of the fixed effects, $\mathbf{M\beta}$, and the random effects $\mathbf{Zu}$. The fixed effects describe the causal links between the p proposed predictor variables and the status. The $n \times p$ design matrix **M** contains the value of each (standardised) predictor variable at each of the sites. It is possible to include interactions between two variables in the fixed effects by assigning the product of the two variables to the elements of column of **M**. The length n vector, **β**, contains the coefficients of the linear relationships. The random effects account for other factors that might influence the observed status data but do not reflect drivers of groundwater status. For example, we might believe that different countries have different approaches to designating status and therefore 'country' could be included as a random effect. The $n \times q$ design matrix **Z** contains the value of the q variables that are thought to influence the observed status at each of the n sites.

We proposed different formulations of the **M** and **Z** matrices based on the hypotheses proposed by the groundwater expert. Each model was then compared in terms of quality of predictions of status and consistency with our understanding of the groundwater system. We estimated the coefficients of each logistic mixed effects model by maximum likelihood (Dobson, 2001) using the Matlab (Mathworks, 2014) command *fitglme*. The likelihood is the probability that the observed data would have arisen from a specified model. The maximum likelihood estimator uses a numerical optimizer to find the parameter values which lead to the largest likelihood value. It yields an estimate of each element of **β** and a corresponding estimate variance which we used to determine whether, under a Gaussian assumption, each element was significantly different to zero and could hence be considered to be a significant predictor of groundwater status. We compared the quality of fit of different models using the Akaike Information Criterion (AIC; Akaike, 1973):

$$\mathrm{AIC} = 2k - 2L,$$

where k is the number of estimated parameters in the model and L is the natural logarithm of the maximised likelihood. The model with the lowest AIC is considered to be the best compromise between complexity (number of parameters) and fit to the data (likelihood). We use the AIC to compare model structures that reflect minor variations in the hypotheses of which predictor variables should be included in the fixed effects.

It is also possible to estimate logistic mixed effects models for every combination of inclusion or exclusion of each of the gridded datasets and their interactions in the fixed effects design matrix. The resultant model with the lowest AIC is likely to yield accurate predictions of the groundwater status. Alternatively, we might employ stepwise regression (Draper & Smith, 1998) to select the variables to include in the fixed effects. In this approach, the fixed effects initially consist of a constant. Then a series of models are estimated to discover which predictor variable, when added to this constant, will lead to the largest increase in the likelihood (or any other fitting criterion). If this improvement in the fit is significant (e.g. if the AIC decreases) then this variable is added to the fixed effects design matrix. The process continues iteratively considering at each step whether further predictors should be added to the fixed effects. It is possible that in some instances that the addition of a predictor variable means that a variable that is already included is now superfluous. It is possible to test for this by experimenting with removing each variable in turn and seeing if the fit of the model is significantly worse. If the removal of a variable does not lead to a significantly worse fit then it is removed from the fixed effects. The process continues until no variables are either added or removed from the fixed effects design matrix. These approaches to modelling are analogous to the machine learning approaches described above in that they are purely controlled by the correlations amongst the observed data rather than our understanding of the groundwater system. Therefore, there is the same potential for spurious processes appearing in the model.

Having estimated the model parameters, the Matlab command predict was used to predict the probability of groundwater status for each grid cell where predictor variables were available. Models of both groundwater body chemical status and quantitative status were prepared, although the main focus of the work was on modelling groundwater body chemical status.

2.2.3.1 Models of groundwater body chemical status

Groundwater chemical status is sensitive to a wide range of potentially contaminating activities are the land surface, including diffuse pollution from agriculture and more focussed pollution from industry and leaking sewer systems. Other factors and processes, such as salinization of groundwater, may be locally significant. Given the available predictor variables our initial hypothesis was that the primary drivers of groundwater body chemical status across Europe as a whole are arable farming, precipitation and pollution from the local population. Arable farming requires the addition of nutrients and pesticides to the soil which may either be prone to leaching into the groundwater and or to dilution dependent on the timing and intensity of precipitation, while the presence of high population densities may be associated with contaminant loading of groundwater. Therefore, we estimated a logistic model where the random effects were zeros and the fixed effects were a linear function of the following standardised gridded covariates: (i) proportion of arable land use (Figure 2a), (ii) population (Figure 2b), (iii) winter precipitation (Figure 2c), and (iv) summer precipitation (Figure 2d). This model (referred to as C2 in Tables 1 and 2) had a lower AIC than a logistic model with constant fixed effects (C1) as shown in Table 1. We see in Table 2 that each of the regression coefficients for C2 are significantly different to zero and that the signs are consistent with a hypothesis that increases in arable land use, population and winter precipitation (leaching effect) are associated with increased likelihood of poor groundwater status while increases in summer precipitation (dilution effect) are associated with a reduced likelihood of poor groundwater status. Note that factors which lead to poor chemical status will have a positive sign and those that improve groundwater status will have a negative sign. The predictions that result from C2 are shown in Figure 3b.

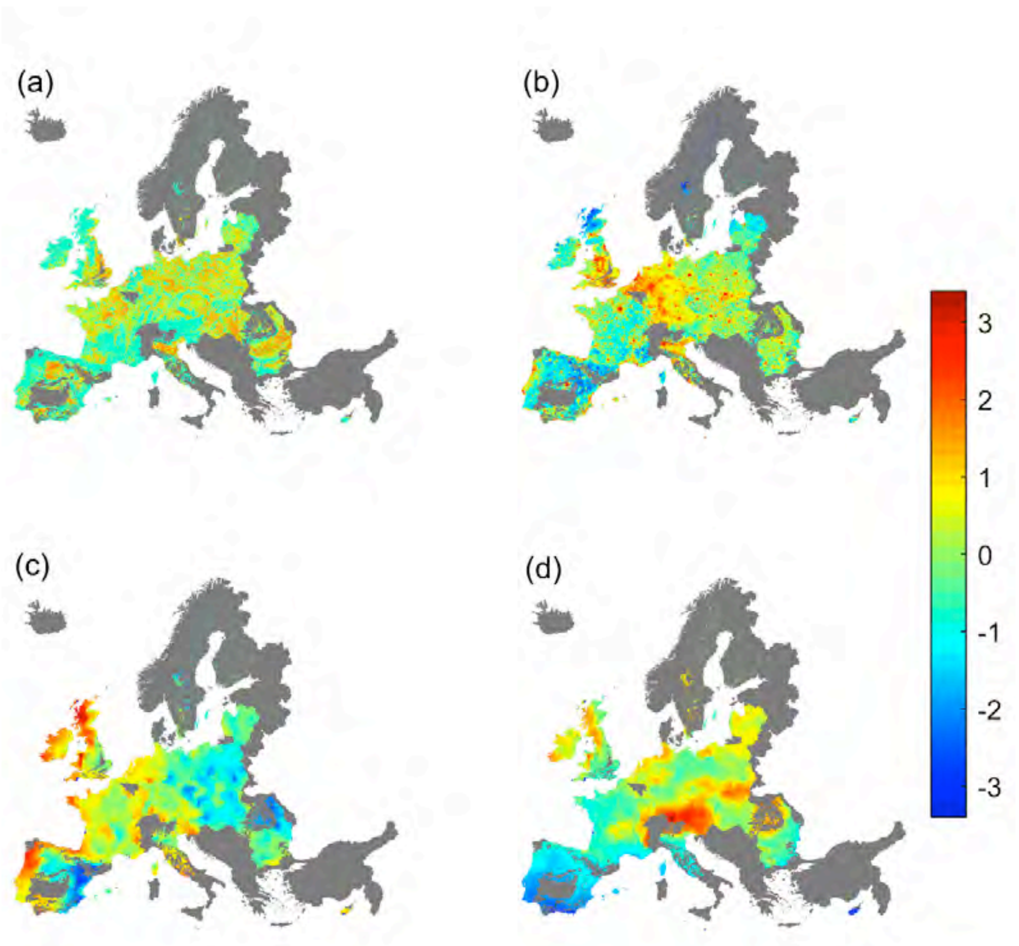


Figure 2: Proposed standardised covariates for inclusion in the C2 model of chemical status. (a) proportion or arable land use, (b) population, (c) winter precipitation and (d) summer precipitation.

It is apparent that for some states, such as in eastern Europe, and for some regions, for example in the Iberian Peninsula, that more poor groundwater chemical status is predicted by model C2 (Figure 3b) than was recorded (Figure 3a). This might be because these countries and regions manage their land in a manner which improved groundwater chemical status or because approaches to recording groundwater chemical status vary between states in Europe. We consider whether such behaviours might be included in the model by adding a random effect which varies according to the country in which the grid cell is situated (model C3). Figures 4 and 5 illustrate how the variation in groundwater chemical status is divided between the fixed and random effects. This model, model C3, has a lower AIC than C2 and appears to better match the recorded chemical status (compare Figure 3a and Figure 5b). However, contrary to the results of C2, winter precipitation as well as summer precipitation are both associated with a decreased likelihood of poor groundwater status (Table 2), from which it could be inferred that precipitation at any season has a generally diluting effect on nutrient loads. Given the change in sign of the winter precipitation explanatory variable, there may be a concern that some of the drivers of groundwater chemical status might have been confounded with the random effects. Consequently, we set the random effects to zero in all subsequent models.

Table 1: Maximised log-likelihood (L), AIC, variance explained in calibration data (VE cal) and variance explained in validation data for the models estimated for chemical groundwater body status. Predictor variables are proportion of arable (Ar), population (Pop), winter precipitation (Wp), summer precipitation (Sp), modelled P input (P) and modelled N input (N).

Model	Fixed effects	Random effects	L	AIC	VE cal	VE val
C1	1	0	-6246.64	12495.28	0.00	0.00
C2	1, ar, pop, wp, sp	0	-5584.82	11179.64	0.12	0.12
C3	1, ar, pop, wp, sp	Country	-4468.65	8949.29	0.33	0.31
C4	1, ar, pop, wp, sp, ar×wp	0	-5547.40	11106.79	0.13	0.13
C5	1, P, pop, wp, sp	0	-5368.87	10747.74	0.16	0.28
C6	1, N, pop, wp, sp	0	-5581.38	11172.76	0.12	0.12
C7	1, P, pop, sp	0	-5372.44	10752.88	0.16	0.15
C8	Stepwise	0	-4623.91	9283.82	0.29	0.28

Table 2: Sign of fixed effect coefficients for models listed in Table 1. Values that are not significantly different to zero at the 0.05 level are written in brackets.

Model	Ar	Pop	Wp	Sp	Ar×Wp	P	N
C1							
C2	+	+	+	-			
C3	+	+	-	-			
C4	+	+	+	-	+		
C5		+	-	-		+	
C6		+	(+)	-			+
C7		+		-		+	

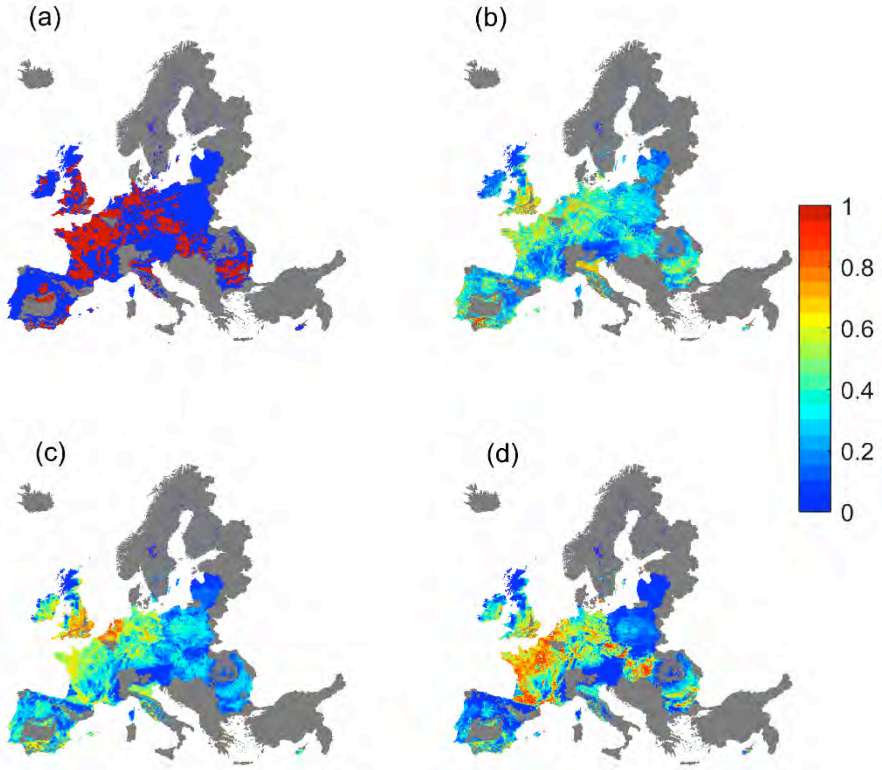


Figure 3: (a) Recorded groundwater chemical status, (b) predicted groundwater chemical status according to C2, (c) predicted groundwater chemical status according to C7: three variable logistic model and (d) predicted groundwater chemical status according to C8: logistic model formed by stepwise regression.

Next we test whether, as we hypothesise, there is a significant interaction between arable farming and winter rainfall by adding an interaction term to the fixed effects (model C4). This model has a lower AIC than C2, but not as low as C3 the model with random (country) effects (Table 1), and the coefficients for the two variables and their interaction are all positive indicating a synergistic relationship leading to increased likelihood of poor chemical status.

Other gridded datasets were available which reflected the addition of nutrients to the soil by agriculture. These included the modelled P inputs (Figure 6b) and the modelled N inputs (Figure 6c). The proportion of arable farming (Figure 6a) and the two modelled nutrient inputs were all derived using the Corine landcover classification (European Environment Agency, 2010). Therefore, we did not include all three of these variables in the same model. We experimented with replacing the arable term in model C2 with the P inputs (model C5) and the N inputs (model C6). Both of these changes led to a decrease in AIC with the P inputs leading to the largest decrease (Table 1). However, these changes also led to the coefficient for winter rainfall being either negative or not significant. This might reflect that the different rates of winter rainfall have been accounted for in the nutrient modelling. In model C7, the potential contamination of groundwater by agriculture is represented by the modelled P inputs and winter precipitation is removed from the model. The resultant predictions are shown in Figure 3c.

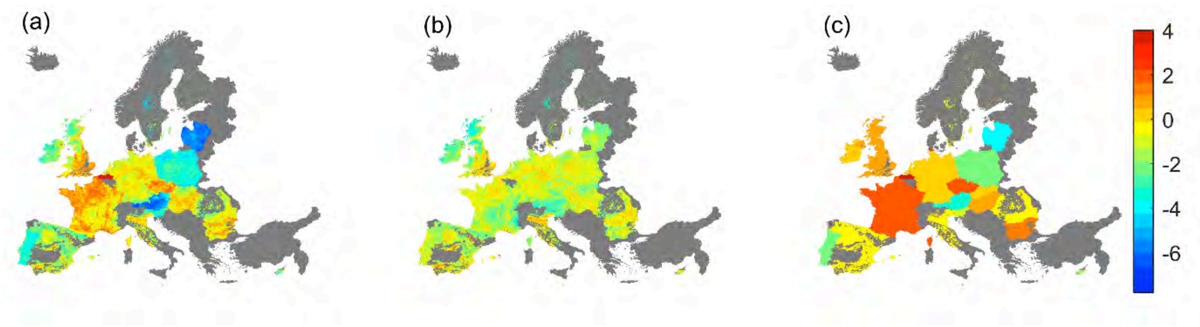


Figure 4: (a) Predicted values of $ln\left(\frac{p}{1-p}\right)$ for the generalised linear mixed effects model; (b) fixed effects contribution to prediction of $ln\left(\frac{p}{1-p}\right)$ and (c) random effects contribution to prediction of $ln\left(\frac{p}{1-p}\right)$.

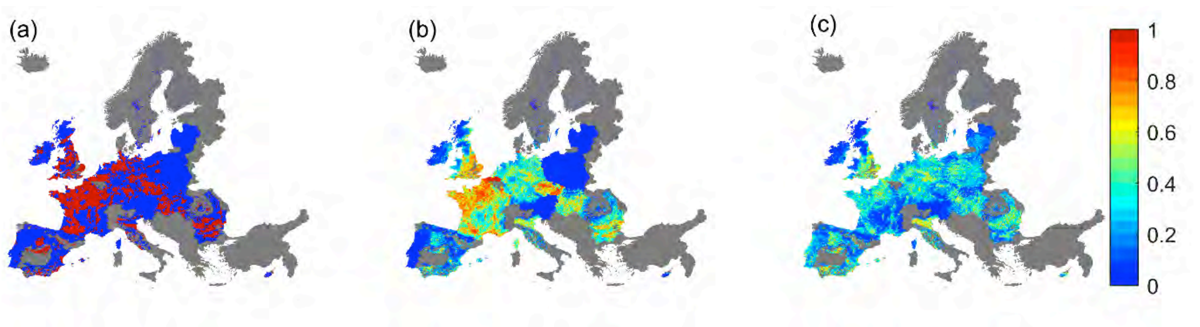


Figure 5: (a) Recorded groundwater chemical status, (b) predicted groundwater chemical status according to four variable generalised linear mixed effects model, and (c) random effects for the generalised linear mixed effects model.

Finally, we estimated a model that could potentially include all of the gridded datasets in the fixed effects using a stepwise approach. The variables included in the fixed effects and the signs of their coefficients were (in order of inclusion in the model): modelled P (+), modelled N (+), summer temperature (+), proportion of arable (+), summer precipitation (-), population (+), total abstraction (+), proportion of grassland (-), proportion of forest (-), aquifer recharge (-), autumn precipitation (-), annual precipitation (+), distance to coast, winter precipitation (+). This model had the lowest AIC and explained the largest proportion of the variance in groundwater chemical status with the exception of the model, C3, containing the random (country effects). The signs of the majority of terms included in the model are consistent with our understanding of the groundwater system. The predictions from this model are shown in Figure 3d.

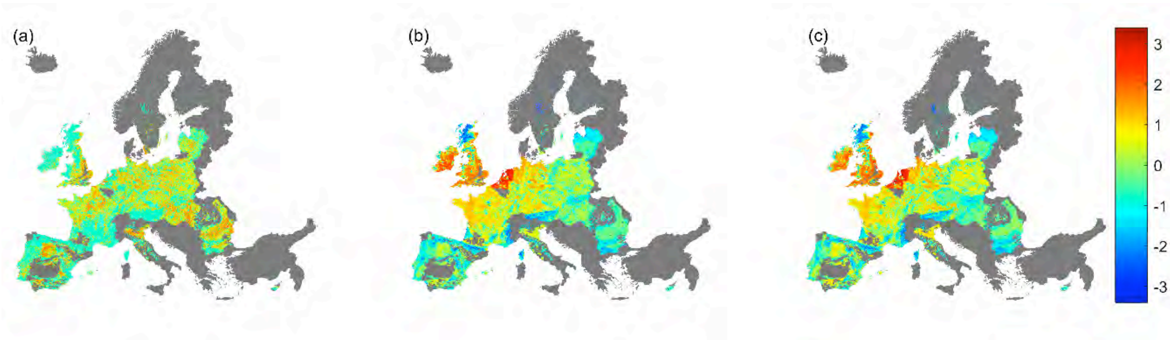


Figure 6: (a) standardised proportion of arable land (b) standardised modelled P input and (c) standardised modelled N input.

Another issue with regards to determining relationships between environmental properties is the scale at which the different variables are presented. For example, we have been exploring the relationships between groundwater status and predictor variables both at the 1-km scale. However, groundwater status within a 1-km grid cell might be caused by factors from a wider area. We illustrate this idea by considering whether there is a stronger relationship between groundwater status and the proportion of arable land use if the arable information is amalgamated over a wider area. We use a wavelet transform to upscale the arable gridded data to the 2, 4, 8,…, 128, 256 km scale (Figure 7) and estimate a series of logistic models where the only predictor is the arable information at a particular scale. We see in Figure 8, that the largest likelihood is achieved if the arable information is presented at the 16-km scale.

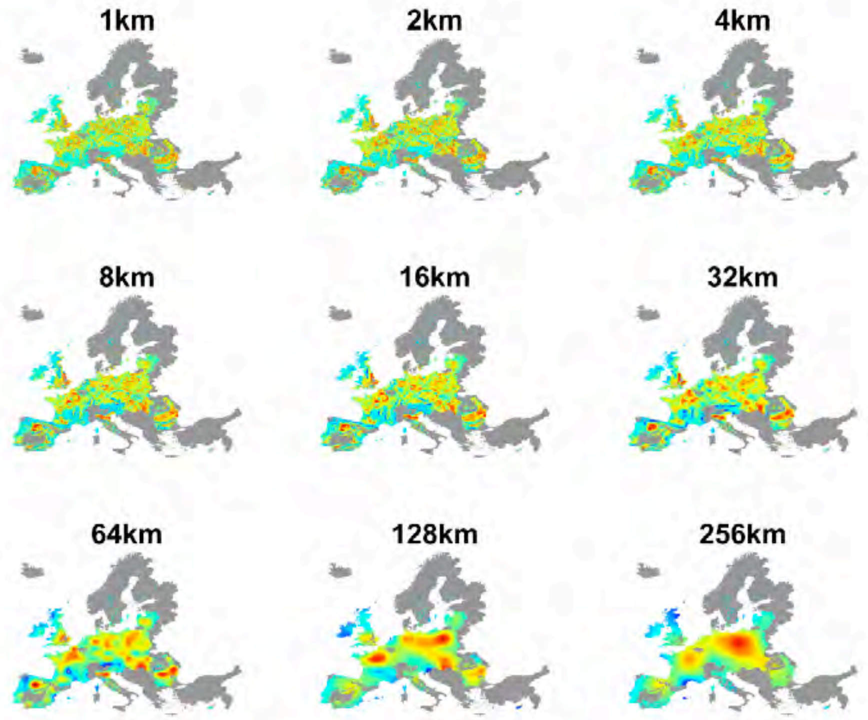


Figure 7: The proportion of arable land presented at different spatial scales.

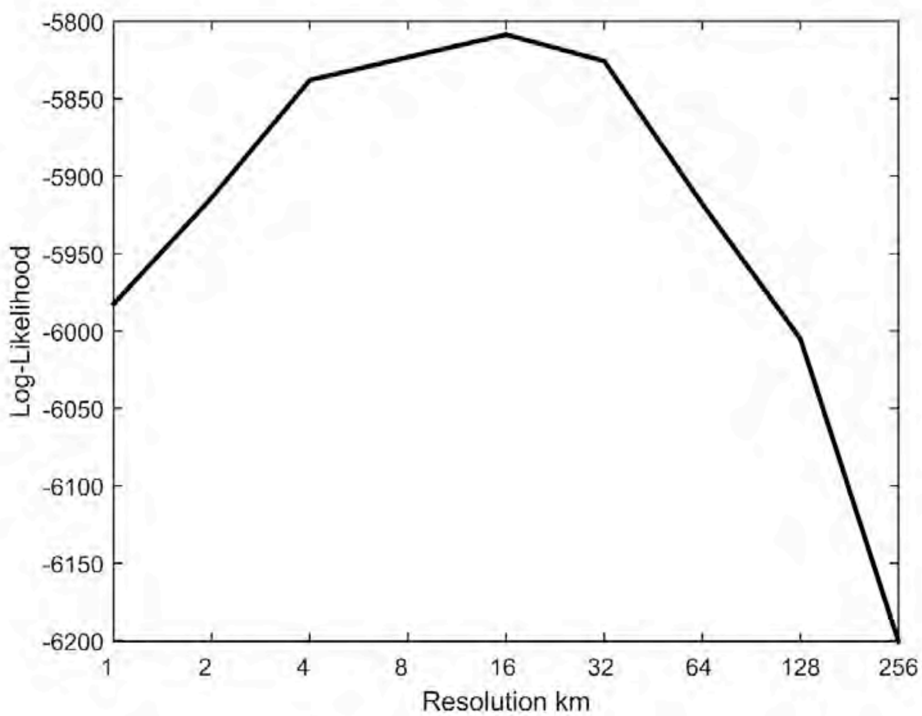


Figure 8: Variation in the maximised likelihood according to the scale of the arable information for a model of groundwater chemical status where the only predictor is the (scaled) proportion of arable land.

2.2.3.2 Models of groundwater body quantitative status

Our initial hypothesis was that the quantitative groundwater status is driven by population (a surrogate for abstraction and groundwater use), winter precipitation (with good quantitative status associated with high winter recharge) and high summer temperatures (with poor quantitative status associated with hot summers leading to high evapotranspiration and increased demand). These variables (Figure 9) were therefore included in the fixed effects of a logistic model (Q2). The model had a lower AIC than the constant fixed effects model (Table 3) and all of the fixed effects coefficients were significantly different to zero (Table 4). Note that factors which lead to poor quantitative status will have a positive sign and those that improve the quantitative status will have a negative sign consequently the signs in Table 4 are consistent with our hypothesis regarding quantitative status.

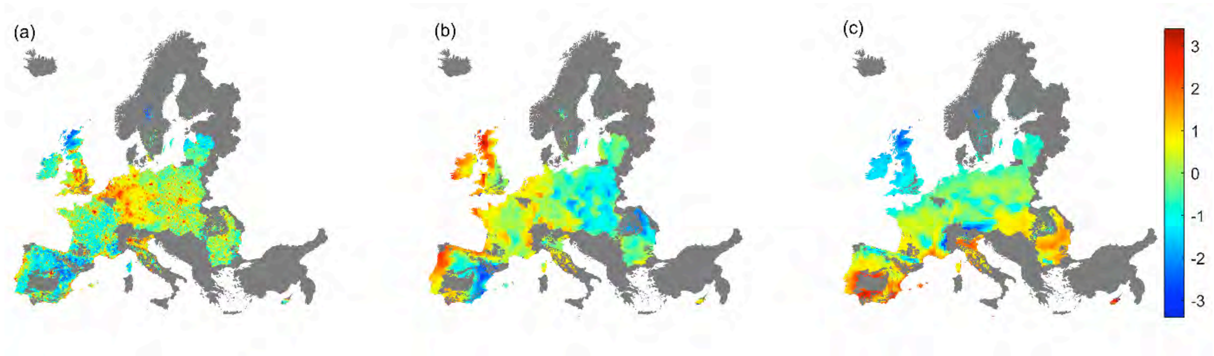


Figure 9: Standardised covariates for inclusion of groundwater body quantitative status. (a) population (b) winter precipitation and (c) summer temperature.

We further hypothesised that there was an interaction between the effects of the population and the summer temperature. We tested this hypothesis by including such a term in Q3. The resultant coefficient was significant but negative, indicating that the combined effects of hot summer temperatures and a large population is less than the sum of their individual effects.

Table 3: Maximised log-likelihood (L), AIC, variance explained in calibration data (VE cal) and variance explained in validation data for the models estimated for quantitative groundwater body status. Predictor variables are population (pop), winter precipitation (wp), summer temperature (st).

Model	Fixed effects	L	AIC	VE cal	VE val
Q1	1	-3004.51	6011.01	0.00	0.00
Q2	1, pop, wp, st	-2851.92	5711.85	0.03	0.03
Q3	1, pop, wp, st, pop×st	-2823.66	5657.33	0.03	0.03
Q4	Stepwise	-2481.91	4995.83	0.14	0.11

Table 4: Sign of fixed effect coefficients for models listed in Table 3. Values that are not significantly different to zero at the 0.05 level are written in brackets.

Model	pop	wp	st	pop×st
Q2	+	-	+	
Q3	+	-	+	-

Finally, we estimated a logistic model which could potentially include all of the gridded datasets by stepwise regression. The variables included in the fixed effects and the signs of their coefficients were (in order of inclusion in the model): mean precipitation (-), modelled P input (+), summer rain (-), population (+), autumn precipitation (-), spring precipitation (-), proportion or arable (+), depth to water table (-), altitude (+), forest (+), groundwater recharge (-) and distance to coast (+). These findings are largely consistent with the hypothesis that weather variables are the primary drivers of quantitative status although the modelled P input also features highly. The stepwise model only explains around 14% of the variation in groundwater status and the simpler models only explain 3% (Table 3 and Figure 10). This is considerably less than the variation of chemical status that was explained (Table 1).

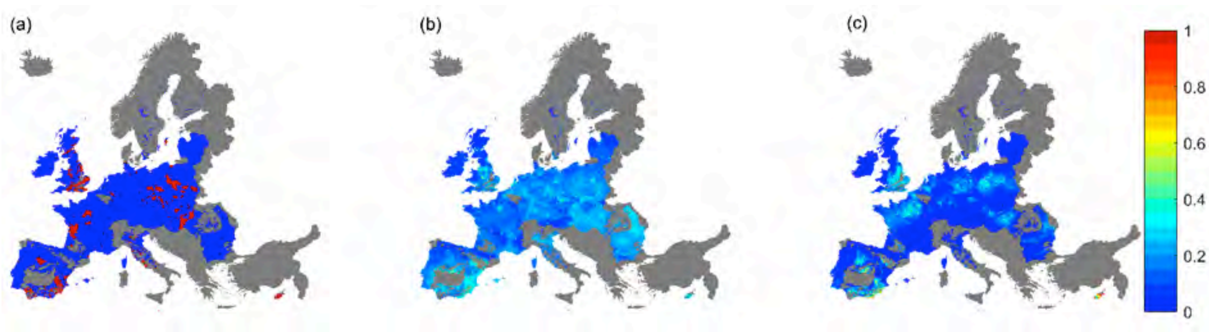


Figure 10: Recorded groundwater quantitative status (a) predicted groundwater quantitative status according to four variable generalised model (b) and predicted groundwater quantitative status according to model determined by stepwise regression.

2.2.4 Discussion

2.2.4.1 Generic observations regarding the modelling approach

We have demonstrated how ‘data-led’ methods, such as stepwise regression, can be used to suggest and estimate models of groundwater status. However, we note that they should be used with caution as such approaches can include spurious relationships which result from not accounting for multiple hypothesis tests. For example, in the present study, although a stepwise of groundwater chemical status had a relatively low AIC and explained a relatively large proportion of the variance compared with other regression models, it is not a simple task to interpret the resulting model in terms of a process-based understanding of the system. We also note that where interactions or higher order terms are of interest, if they are used in such models, then their effects can quickly become intractable.

Alternatively, we suggest that simpler models based on prior knowledge of the system may be more appropriate since they enable specific hypotheses to be tested. However, although more tractable, such models have less predictive accuracy. We also note that the strengths of relationships between variables can vary according to the scale at which they are analysed. In the present study we assessed the effect of the support scale for one of the predictive variables, agricultural land coverage, on a simple regression model of groundwater chemical status and found that the model goodness -of-fit and modelled predictive accuracy could be improved by optimising the support scale. Such an approach should ideally be taken with all predictive variables used in the regression models.

2.2.4.2 Specific observations regarding the model results

Comparing Tables 1 and 3 it can be seen that generally more of the variation in groundwater chemical status can be explained than the variation in groundwater quantitative status using the simple regression models or the respective step-wise regression models. This may in part be due to how the groundwater status (response variable) mapping was prepared. Although the groundwater status information has been reported by member states either associated with no depth horizon or associated with one of a number of horizons of increasing depth, to optimise the number of groundwater bodies that could be used in a consistent model of groundwater status a single, EU-wide status layer was prepared where the status for a given 1km by 1km grid cell was based on the simplifying assumption that the status of the uppermost groundwater body within a vertical sequence of groundwater bodies lying on top of each other (if there is one) determines the status of a respective grid cell. This simplifying assumption is conceptually suited to representing groundwater contamination scenarios associated with movement of contaminants from the land surface vertically through an unsaturated zone to an underlying groundwater body. It is likely to be less suited to processes of saturated flow where movement has a significant lateral component. Consequently, it may be expected given the need to produce a single coherent groundwater chemical status mapping and single quantitative status mapping for groundwater that the models explain more of the variation in groundwater chemical status.

As hypothesised, groundwater chemical status appears to be driven by inputs from agriculture and population. The model that explains most of the observed variance in the calibration and validation data includes country as a random effect. In this model, winter and summer precipitation are both associated with a decreased likelihood of poor groundwater status (Table 2). Such a model would be consistent with a process of either diluting effect on nutrient loads in recharge to groundwater or a more general loss of nutrients to surface waters. However, we also note that the sign of the effect of winter precipitation in a similar regression that does not include country as a random effect is opposite and that there is the possibility that the drivers of groundwater chemical status might have been confounded with the random effects.

Only limited interactions have been investigated to date, however, there is some evidence for a synergistic interaction between arable farming and winter precipitation (when the regression does not include country as a random effect).

There is less confidence in the results of models of groundwater quantitative status (as perhaps may be expected given the manner in which the groundwater status has been estimated, see earlier comments in the discussion), however, it appears to be largely driven by weather variables.

2.2.4.3 Recommendations for additional modelling

- Given that use of an appropriate support scale for one of the predictive variables (agricultural land cover) appears to improve model fit and calibration, we recommend that future modelling seeks to optimise the appropriate support scale for all predictive variables.
- Further modelling should be undertaken to characterise interaction effects in regression models where country is a random effect. It is known that different states in the EU have used quite different processes to assess groundwater chemical and quantitative status so it is reasonable to expect that country should be a random effect.
- A new single mapping of groundwater quantitative status, taking in to account the layered nature of the reporting of groundwater quantitative status, should be investigated and developed for testing within the regression modelling scheme. The revised mapping should be based on the assumption that quantitative status is a function of sub-horizontal flows, and the aim should be to try and develop models with a similar predictive capability to that currently available for the groundwater chemical status.

2.3 Conclusions

- Prevailing stressor causing failure of good groundwater status is pollution, followed by groundwater abstraction.
- Pollution in combination with groundwater abstraction appears to be most common stressor combination in Europe
- Salt water intrusion is almost always associated with groundwater abstraction or/and pollution.
- Interaction of all three studied stressors (pollution, abstraction, salt water intrusion) does not take place in all coastal areas of Europe
- The WFD and also SoE data clearly show that the most common type of groundwater pollutants are agrochemicals (nutrients and pesticides) affecting whole Europe especially in agricultural areas
- An assessment of pesticides may be biased by various monitoring strategies used by countries, there is lack of comparable data on pesticide metabolites that may occur more frequently and in higher concentrations than parent compounds

- EU WFD common implementation strategy does not assure sufficient harmonization of monitoring strategies among EU member states preventing comparable whole European assessments
- Emerging pollutants of various origin may occur in the groundwater
- Statistical modelling can be used for groundwater pressure-response analysis
- Groundwater chemical status appears to be mainly driven by inputs from agriculture and population
- Statistical modelling provided some evidence of synergistic interaction between arable farming and winter precipitation leading to poor chemical status of groundwater in Europe
- There is less confidence in the results of statistical modelling of groundwater quantitative status, however, it appears to be largely driven by weather

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Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

Deliverable 5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis.

D5.1-2: Relation of low flows, E-flows, and Ecological Status

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Due date of deliverable: **Month 36**
Actual submission date: **Month 36**

Dissemination Level		
PU	Public	X
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Contents

Non-technical summary

The present report 'Relation of low flows, E-flows, and Ecological Status' presents a European scale analysis of hydrologic data at the resolution of the Functional Elementary Catchment (FEC). Simulated daily time-series of river flows from the PCR-GLOBWB global model were used based on a hypothetic near-natural scenario where water abstractions from water bodies do not exist and an anthropogenic scenario with water abstractions occurring.The latter practically represents the reality. Many hydrologic indicators expressing the characteristics of the rivers' hydrologic regime were calculated for all FECs with the Indicators of Hydrologic Alteration (IHA) methodology and software package and the deviations of the indicators' values between the two scenarios were used as proxy metrics of hydrologic alteration or hydrologic stress of rivers. Regressions between indicators with the rather limited dataset of EQR values of two BQEs (macroinvertebrates and phytobenthos) showed insignificant or very weak relationships when processed with the entire dataset for Europe or separately for each of the 20 Broad River Types (BRTs). However, by conducting two examples at smaller scales (catchment or region) with better ecological response datasets clearer relationships were found.

Hydrologic alteration metrics were averaged per BRT without reference to any ecological response not showing remarkable hydrologic stress in certain BRTs or considerable differences in the degree of alteration among the various BRTs. Clearer results could be indicated by mapping the hydrologic alteration on Europe's geographical background. The mapped indicators, especially some of those connected with low flow conditions were the most informative showing that Southern Europe is more hydrologically stressed due to groundwater abstractions for irrigation. In the rest of Europe hydrologic conditions change less frequently within a single year and a multi-year period.

The determination of a minimum ecological flow connected with good ecological status needs further research with updated datasets, but the water community can already take advantage of the results produced herein to obtain a view of hydrologic stress in Europe, identify significant hydrologic stress on a local basis and try to interpret the impacts of this stress on river's ecology with the use of appropriate response data.

1. INTRODUCTION

The present report is part of the MARS Deliverable 5.1 (Reports on stressor classification and effects at the European scale), entitled D5.1-2: 'Relation of low flows, E-flows, and ecological status'. The work and findings described here are associated to the respective MARS WP5 subtask 5.1.3: 'Relation of low flows and ecological flows (E-flows) to ecological status'.

The MARS 'Description of Work' (DoW) document (MARS, 2013) sets the scope and objectives of subtask 5.1.3 as follows:

We will carry out a large scale analysis on the relation of low water flow (as estimated by statistical analysis) to class of ecological status and determine the resulting minimum ecological flow. Time series of daily streamflow data available in the WISE-SoE WQ and the EEA Water Accounts database will be used for the analysis, jointly with data from the WFD. We will analyse the effect of water quantity (and climate variability) on Ecological Status measured with different BQEs and the suitability of low flows and E-Flows as a diagnostic tools in underpinning the cause-effect relations between water quantity and Ecological Status.

The DoW also mentions the expected impacts and potential users of the related work: as major expected impact a *better and more coherent planning of environmental flows* is stated, while as main end-users to be facilitated by the project's findings, the document considers *hydropower companies and environmental protection agencies* (MARS, 2013 – Table 3.C.).

The work of the 'e-flow' task, summarized in the present deliverable, has been conducted at the large (European) scale using river flow time-series all across the continent. In line with the DoW, the general purpose was to identify a clear relationship between hydrology and ecology by analyzing hydrologic data and data of ecological status. In particular, our objective was to relate characteristics of the hydrologic regime of rivers (preferably of low flow conditions) with different classes of reported ecological status (Good, Bad etc.) or with numerical ecological indicators such as Biological Quality Elements (BQEs), which are used to determine the class of the ecological status.

A first shortcoming in carrying out this work was the lack of enough and homogeneously observed/reported data for both river flows and ecological status across Europe. The coverage of the large study area was small as regards officially reported information on measured daily river flows and BQEs. Alternative sources of hydrologic data (simulated river flow time-series) and identification of an appropriate dataset with BQE information assisted us to overcome the problems. However, from our analysis (and available BQE data), it was clear that no significant relationship could be observed between any characteristics of river flows with ecological status at the European scale. This could imply that ecological conditions of water bodies are not directly and strongly connected with water quantity characteristics. Certainly, in this report, we do not conclude that hydrology has a relatively weak or secondary role in determining ecological status. Besides, the Good Ecological Status target can never be reached without attention for sustainable quantitative management of water. As demonstrated in the report through a national and a catchment scale example, this can be shown by focusing on smaller scales. Nevertheless, we recognize the need for providing stronger scientific proofs in the future regarding the relationship of hydrology with the ecological response and the ecological status of rivers.

Based on the above, this report and the respective work do not determine a *resulting minimum ecological flow* (from MARS DoW – see above) for each European stream or river for the preservation of its good ecological status, nor they demonstrate the *suitability of low flows and E-Flows as a diagnostic tools in underpinning the cause-effect relations between water quantity and Ecological Status* (MARS DoW – see above). Instead, towards the need for a useful and comprehensive hydrologic stress analysis in MARS, we provide a more descriptive report on the hydrological alteration in today's European rivers. Hydrologic alteration is attributed to water abstractions for the satisfaction of urban, industrial and agricultural needs and expresses the hydrologic stress of rivers and streams, namely the disturbance or deviation of their hydrologic regime (water availability and temporal variation of flows) from the ideal undisturbed or natural conditions.

The report provides general information on the concept of ecological flows and the existing methods for evaluating them. It then describes the Indicators of Hydrologic Alteration (IHA) approach (Richter et al., 1996), which was efficiently used to address hydrologic stress in Europe at the Functional Elementary Catchment (FEC) level (http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network). A detailed presentation of the methodology follows describing how we associated simulated hydrologic data to the thousands of FECs across Europe. The daily discharge data used refer to a 10-y period (climate of 2001-2010) and were simulated by the global water balance model PCR-GLOBWB (Van Beek et al., 2011, Sutanudjaja et al., 2014) for both the baseline or anthropogenic scenario (real conditions with water abstractions occurring across Europe) and the near-natural scenario representing near-natural conditions with no abstractions from water bodies. We examine possible relations between recently reported ecological data (BQEs) for rivers and the magnitude of the resulted hydrologic change in the anthropogenic scenario (alteration) expressed through a large number of hydrologic indicators. We further investigate the connection of the hydrological alteration with riverine ecology through the presentation of case studies that utilize more detailed ecological data sets on national (Germany) and catchment scale (Pinios river, Greece). The hydrologic results derived for the entire Europe are presented on the basis of the variation of the hydrologic indicators at the anthropogenic scenario (baseline) and their change from the near-natural scenario across Europe (maps), as well as, they are aggregated among various Broad River Types (Solheim et al., 2015) (graphs).

Where possible, we try to indicate how the hydrologic indicators, and in particular the indicators associated with low flows, vary across Europe and to assign key hydrologic characteristics to the Broad River Types. Although no clear relationship can be shown with the ecological status of rivers, the analysis forms a standalone hydrologic work that can inform environmental protection agencies and other interested parties across Europe about the current hydrologic stress.

1.1. Definition of ecological flows for WFD implementation

The concept of ecological flow is not a very novel concept and there are already a lot of relevant publications addressing it. Most of them share a common definition of the ecological flow (e-flow) that is related to the amount of water that is left in an aquatic ecosystem or provided to it with the purpose of maintaining ecological components, functions and services (e.g. Arthington and Pusey, 2003; Arthington et al., 2006; Brown and King, 2003).

However, only recently the ecological flows were considered to be integrated into the Water Framework Directive (WFD). The WFD CIS Guidance Document No 31 (CIS, 2015): *Ecological Flows in the Implementation of the Water Directive*, aims to provide an understanding of the e-flow concept and how to use it in the RBMPs including information on the methodologies, monitoring, evaluation and measures.

First of all, the guidance document offers a working definition of the e-flows in the context of the WFD. Ecological flows are considered as “an hydrological regime consistent with the achievement of the environmental objectives of the WFD in natural surface water bodies as mentioned in Article 4(1)”: a) non-deterioration of the existing status, b) achievement of good ecological status, c) compliance with standards and objectives for protected areas, including the ones designated for the protection of habitats and species where the maintenance or improvement of the status of water is an important factor for their protection, including relevant Natura 2000 sites designated under the Birds and Habitats Directives (BHD).

1.2. Methodologies for assessing eflows

A global review of the present status of e-flow methodologies revealed the existence of some 207 individual methodologies, recorded for 44 countries within six world regions. These could be differentiated into hydrological, hydraulic rating, habitat simulation and holistic methodologies (Tharme, 2003). In order to monitor, evaluate and assess e-flows the guidance document also proposes categories based on scale, complexity and volume of required data (CIS, 2015). All the proposed methodologies however, are grouped into three broader categories. The table below summarizes the pros and cons of each of the three methodology categories.

Table 1.1 – Information regarding the three methodological categories for assessment of e-flows (taken from CIS, 2015).

<table>
<tr><th>Methodology category</th><th>Scale</th><th>Duration of assessment (months)</th><th>Relative frequency of use</th><th>Information required</th></tr>
<tr><td>Hydrological</td><td>Whole rivers, applicable for regional assessments</td><td>1-6</td><td>+++</td><td>Consistent and spatially distributed hydrological data (at least 15 years of continuous measures)

Reliable hydrological models to extrapolate streamflow time series to ungauged sites

Literature review of the linkages between flow regime and key riverine processes</td></tr>
<tr><td>Hydraulic-Habitat</td><td>Applied at a study site / river segment scale, upscaling to whole river basin based on the assumption of “representative” site conditions</td><td>6-18</td><td>++</td><td rowspan="2">Collecting new data, basic ecological modelling and economic assessment methods

Synthesis of information and articulation of expert judgement into e-flows recommendations occurs within the framework of a flow workshop with diverse participants</td></tr>
<tr><td>Holistic</td><td>Whole rivers, applicable for regional or river specific scales</td><td>12-36</td><td>+(increasing)</td></tr>
</table>

The simplest, typically desktop e-flow methodologies are the hydrological methodologies, which rely on the use of hydrological data, usually in the form of naturalized, historical monthly or daily flow records, for making environmental flow recommendations. Hydrological methodologies are based on the analysis of historic (existing or simulated) streamflow data and make use of the assumption that the full range of natural variability in the hydrological regime is necessary to conserve river ecosystems. Natural flow regimes display variability at a range of time scales including seasonal and inter-annual, and native aquatic and riparian biota are adapted to this variability. For this reason, the magnitude, frequency, duration, timing and rate of change of the natural flow regime are generally agreed to be the key elements central to sustaining and conserving native species and ecological integrity (Bunn & Arthington, 2002; Poff et al., 1997).

Therefore, the hydrological methodologies consider the *hydrological regime that is needed to maintain the whole system's morphological and ecological processes* (Richter et al., 2011). Their biggest advantage is their ease of use as they just require time series of gauged or simulated flow data, thus, they currently represent the most commonly used approaches for assessing e-flows. On the contrary, hydraulic and holistic approaches require detailed information that can be obtained usually through fieldwork and application of hydraulic and ecological modelling techniques. They have the advantage of considering directly the ecological and morphological information of the ecosystem.

Provided that there is a suitable daily flow data series, hydrological methods may be a reasonable approach to cover not only a basin but even larger scales (regional, national, continental). What remains challenging however is the definition of flow alteration - ecological response relationships, which – if clear and robust – may facilitate the environmental flow-setting processes towards protection and restoration. In an effort to develop quantitative relationships between various kinds of flow alteration and ecological responses, the article of Poff et al. (2010) reviewed 165 papers published over the last four decades. Their aim was to determine if general relationships could be drawn from disparate case studies in the literature that might inform environmental flows science and management. Ecological responses were characterized according to taxonomic identity (macroinvertebrates, fish, riparian vegetation) and type of response (abundance, diversity, demographic parameters). Many of the studies documented strong ecological responses to specific types of flow alteration and showed that the risk of ecological change increases with increasing magnitude of flow alteration. However, the article concludes that no general, transferable quantitative relationships between flow alteration and ecological response can be developed. In this regard, we expected that identifying flow - ecology relationships at the large scale (e.g. Europe) may not be very likely.

1.3. Indicators of Hydrologic Alteration (IHA) method and e-flow components (EFCs)

The Indicators of Hydrologic Alteration (IHA) was originally proposed by Richter et al. (1996, 1997, 1998; Poff et al., 1997) to assess the degree of hydrologic alteration caused by human intervention on rivers. The method is based on the calculation of 33 hydrologic parameters that characterize the intra- and inter-annual variability in water conditions (Table 1.2), including the magnitude, frequency, duration, timing and rate of change of flows or water levels (Richter et al., 1996). Apart from their ability to reflect human-induced changes, the parameters have ecological relevance (Richter et al., 1997). Other researchers propose a smaller set of hydrologic parameters after identifying those that are redundant and inadequate (Table 1.2). The calculation of the hydrologic parameters is computed with the use of a free software tool developed by The Nature Conservancy, called the Indicators of Hydrologic Alteration (IHA). The IHA method was updated with a new set of hydrologic parameters called 'E-Flow Components' (EFCs). The rationale behind this is described thoroughly in the article of Mathews & Richter (2007). The basic idea was that simpler and more effective ways to evaluate the flow conditions were needed that could be translated into e-flow recommendations. A set of 34 new parameters (EFCs) were added to the IHA software (IHA, 2009). These new parameters represent crucial relationships between flow and ecological functions and are categorized in five major components of flow, all considered as ecologically important. These five major components are:

- **Low flows** are related to the amount of water that is available for most of the year. Therefore, they determine the "general" characteristics (temperature, connectivity, flow velocity) of the habitats.
- **Extreme low flows** occurring during times of drought are related with changes in water chemistry, dissolved oxygen, water temperature and concentration of species. They can reduce habitat connectivity affecting the movement of aquatic organisms.
- **High flow pulses** are related to events of rainstorms and snowmelt. These brief changes in the water level can have a beneficial effect on aquatic organisms when they occur after periods of low-flow conditions.
- **Small floods** happen when water level overtops the main channel banks and are events that occur frequently (every 2-10 years). They affect the mobility of aquatic organisms providing access to habitats for refuge, spawning and feeding.
- **Large floods** occur rarely and modify the habitat conditions (move woody debris, vegetation, organic matter and sediments) affecting the aquatic organisms.

Table 1.2 – List of hydrologic indicators used in the IHA method (IHA, 2009). Short lists modified by Acreman et al. (2009) and UK TAG (2008).

IHA full List (Richter *et al.*, 1996)	IHA short list for UK (Acreman *et al.*, 2009)	IHA short list for UK (UK TAG, 2008)
December flow (m^3/s)		
January flow (m^3/s)	Mean January flow (m^3/s)	Mean January flow (m^3/s)
February flow (m^3/s)		
March flow (m^3/s)		
April flow (m^3/s)	Mean April flow (m^3/s)	Mean April flow (m^3/s)
May flow (m^3/s)		
Jun flow (m^3/s)		
July flow (m^3/s)	Mean July flow (m^3/s)	Mean July flow (m^3/s)
August flow (m^3/s)		
September flow (m^3/s)		
October flow (m^3/s)	Mean October flow (m^3/s)	Mean October flow (m^3/s)
November flow (m^3/s)		
1 day minimum flow (m^3/s)		
3 day minimum flow (m^3/s)		
7 day minimum flow (m^3/s)	Mean of annual minimum 7 day flow (m^3/s)	Q95[16]
30 day minimum flow (m^3/s)		
90 day minimum flow (m^3/s)		
1 day maximum flow (m^3/s)		
3 day maximum flow (m^3/s)		
7 day maximum flow (m^3/s)	Mean of annual maximum 7 day flow (m^3/s)	Q5
30 day maximum flow		
90 day maximum flow		
Mean Julian day of minimum flow		
Mean Julian day of maximum flow		
Number of times flow rate rises above Q25	Mean number of times per year flow exceeds Q25 (1)	Estimates based on the ratio of Q50:Q95
Number of times flow rates drops above Q75	Mean number of times per year flow is less than Q75	
Mean fall rate		
Mean duration of high pulses	Mean number of times of flow rises	
Mean duration of low pulses		
Number of low rises		
Number of flow falls		
Mean rise rate	Mean fall rate-mean different between falling flows (m^3/s per day)	

The IHA software categorizes each daily flow value into one of the five EFC major components, and then calculates their magnitude, frequency, duration, timing and rate of rise and fall. As a result a set of 34 parameters that reflect the timing, magnitude, duration, and frequency of these events and the rate of change between these events and the low-flow baseline is generated. The EFC parameters are summarized in Table 1.3.

The hydrological methods do not take into account any ecological information. In order to identify relationships between hydrological parameters and the ecological status, we thus have to use statistical approaches to define the minimum flow conditions required for good ecological status. Basically, what is proposed in this document is the implementation of statistical analysis (e.g. regression techniques) that will attempt to identify thresholds of hydrologic indicators where GOOD Ecological Status is met for a particular Biological Quality Element (BQE).

On top of the 67 hydrologic indicators (33 IHA + 34 EFCs), the method and associated software package (IHA, 2009) export some other general hydrologic indices from the analysis of flow time-series. Two interesting ones to our work are the baseflow index and the flood free season, which are further explained later in this report. It should be also mentioned that some of the above mentioned hydrologic parameters are inter-correlated, resulting in considerable information redundancy (Gao et al., 2009). To increase the efficiency of the analysis at a large-scale (multiple sites with flow time-series) there is a need to reduce the number of indices to be used to those which are adequate to provide an accurate overall determination of the hydrologic alteration and its impact on ecological status. There is a hope that regressions may indicate the most important hydrologic parameters.

Table 1.3 – The set of Ecological Flow Components (EFCs) calculated by the IHA software package (IHA, 2009).

Major components	Parameters	Ecological role
1.Low flows	Mean or median values of low flows during each calendar month *Subtotal: 12 parameters*	· Provide adequate habitat for aquatic organisms · Maintain suitable water temperatures, dissolved oxygen, and water chemistry · Maintain water table levels in floodplain, soil moisture for plants · Provide drinking water for terrestrial animals · Keep fish and amphibian eggs suspended · Enable fish to move to feeding and spawning areas
2.Extreme low flows	Mean or median values of extreme low flow event: · Duration (days) · Peak flow (minimum flow during event) · Timing (Julian date of peak flow) · Frequency *Subtotal: 4 parameters*	· Enable recruitment of certain floodplain plant species · Purge invasive, introduced species from aquatic and riparian communities · Concentrate prey into limited areas to benefit predators
3.High flow pulses	Mean or median values of high flow pulse event: · Duration (days) · Peak flow (maximum flow during event) · Timing (Julian date of peak flow) · Rise and fall rates · Frequency *Subtotal: 6 parameters*	· Shape physical character of river channel, including pools, riffles · Determine size of streambed substrates (sand, gravel, cobble) · Prevent riparian vegetation from encroaching into channel · Restore normal water quality conditions after prolonged low flows, flushing away waste products and pollutants
4 & 5.Small Floods & Large floods	Mean or median values of flood event: · Duration (days) · Peak flow (maximum flow during event) · Timing (Julian date of peak flow) · Rise and fall rates · Frequency *Subtotal: 12 parameters*	· Provide migration and spawning cues for fish · Trigger new phase in life cycle (i.e. insects) · Provide new feeding opportunities for fish, waterfowl · Control distribution and abundance of plants on floodplain · Maintain balance of species in aquatic and riparian communities · Deposit gravel and cobbles in spawning areas · Flush organis materials (food) and woody debris (habitat structures) into channel · Drive lateral movement of river channel, forming new habitats

2. METHODS

2.1. Hydrologic data at European scale

The implementation of the proposed IHA method requires time series of daily flows for at least 10-15 years. This means that a detailed dataset covering the whole Europe and containing the needed information is crucial for the objectives of this work. Data from gauged sites that meet the above requirements are rare. The most complete database regarding flow data is the Waterbase – Water Quantity database which is freely available from (http://www.eea.europa.eu/data-and-maps/data/waterbase-water-quantity-7). The database contains various data on water quantity collected from EEA member countries through the WISE-SoE data collection process. Although the information is based on monitoring data from national databases, the coverage of the monitoring stations and the time step of the data series are not appropriate for applying the IHA methodology. Figure 2.1 shows the spatial distribution of the monitoring stations among the member countries where the uneven coverage of gauged sites can be noted. Moreover, many of these sites contain data only on monthly basis that cannot be used effectively for assessing the alteration of the hydrological regime.

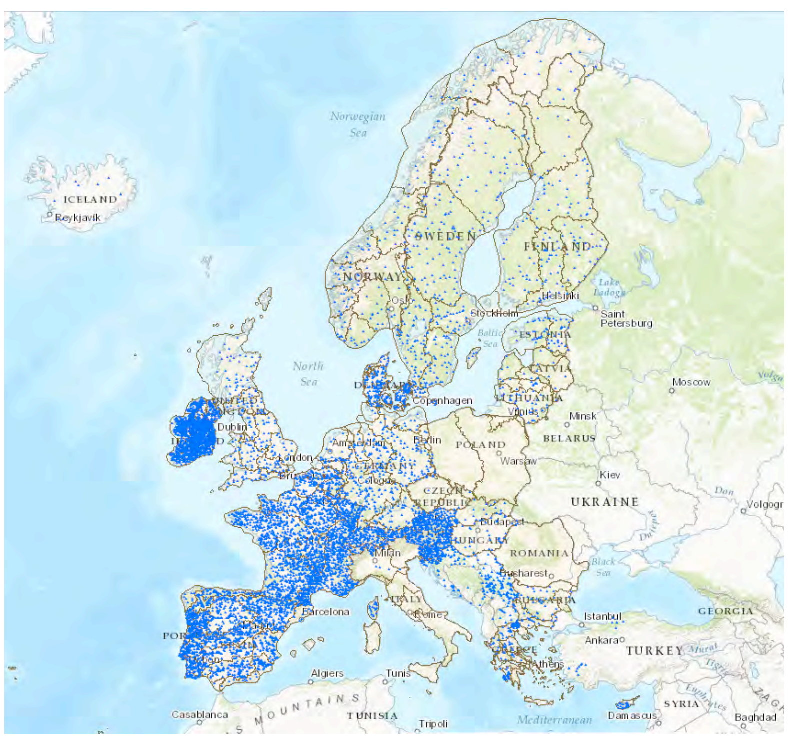


Figure 2.1. Distribution of streamflow flow gauging stations across Europe. Data source WISE SoE 2013 reporting period.

In order to overcome the data scarcity problem it was agreed to take advantage of modelled hydrology data simulated by the large scale hydrology model PCR-GLOBWB (Van Beek et al., 2011, Sutanudjaja et al., 2014). The model includes an online water demand scheme to estimate irrigation water requirement. Briefly, this scheme separately parameterizes two different irrigated crop groups: paddy and non-paddy, aggregated from 26 crop classes from the MIRCA2000 dataset (Portmann et al., 2010) that accounts for various growing season lengths under different regional practices and climatic conditions. The crop vegetation phenology and rooting depths were based on the Global Crop Model (Siebert and Döll, 2010). Calculation of the irrigation water requirement followed crop specific calendars that ensure optimal crop growth. Principally, this irrigation water demand scheme aims to maintain certain soil moisture levels in order to provide optimal crop transpiration, but still takes into account soil water availability, interception, bare soil evaporation, as well as open water evaporation over inundated paddy fields. Over daily time steps, irrigation water demand is calculated by considering the deficit of readily available water in the soil moisture layers (thickness $\leq$ 1.2 m) to their total storage capacities (Wada et al., 2014). The dynamic irrigation scheme in PCR-GLOBWB also considers historical growth of irrigated areas based on FAOSTAT (http://faostat.fao.org/).

Other sectoral water demands, including those from livestock, industry and household, were compiled from several sources, e.g. Wint & Robinson (2007) and FAOSTAT (http://faostat.fao.org/). The development of these historical sectoral water demand databases in PCR-GLOBWB is mainly based on the algorithm developed by Wada et al. (2011) that considers many factors, including past change in population, socio-economic and technological development. Livestock water demand was calculated by multiplying the number of livestock in a grid cell with its corresponding drinking water requirement, which is a function of air temperature. The gridded global livestock densities of cattle, buffalo, sheep, goats, pigs and poultry and their corresponding drinking water requirements were obtained from FAO (2007), Steinfeld et al. (2006) and FAOSTAT (http://faostat.fao.org/).

Historical and gridded industrial demand data were obtained from several sources (e.g. Vörösmarty et al., 2005). The algorithm of Wada et al. (2011) calculates country-specific economic development based on four socioeconomic variables: gross domestic product (GDP), electricity production, energy consumption, and household consumption. Associated technological development per country was then approximated by energy consumption per unit electricity production, which accounts for industrial restructuring or improved water use efficiency. Household or domestic water demand was estimated by multiplying the number of population in a cell with the country-specific per capita domestic withdrawals. The country domestic withdrawals were mainly taken from the FAO AQUASTAT (http://www.fao.org/nr/water/aquastat/main/index.stm). Economic and technological developments were taken into account and seasonality of household/domestic water demand was also considered using air temperature as a proxy. Here available gridded global population maps per decade were used to downscale the country-scale map to produce the gridded water demand data.

PCR-GLOBWB simulations were performed for the period 1960-2010. As for the meteorological forcing, the monthly precipitation, monthly temperature and reference potential evaporation of CRU TS3.21 (Harris et al., 2014) were used. As PCR-GLOBWB runs at daily resolution, the monthly fields of CRU TS3.21 forcing data were downscaled to daily values using the products from the European Centre for Medium-Range Weather Forecasts (ECMWF) and the ECMWF ERA-Interim re-analysis. For downscaling the forcing period 1960-1978, ERA-40 (available for the period 1957-2001) was used, while ERA-Interim (available since 1979) was used for the period 1979-2010. An extensive explanation on the downscaling methodology can be found on van Beek & Bierkens (2009) and Sutanudjaja et al. (2011).

Two scenarios were performed: a naturalized (no abstraction) run and an anthropogenic run (with human influence).

Near-natural (no abstraction) *scenario*:

- A grid-cell in this scenario constitutes up to three land cover classes: short vegetation, tall vegetation and surface water bodies.
- Basically, the parameters for the first two land cover classes are based on the Global Land Cover Characteristics Data Base Version 2.0 (GLCC 2.0, http://edc2.usgs.gov/glcc/globeint.php). More detailed explanation about this can be found on van Beek & Bierkens (2009) and Sutanudjaja et al. (2011).
- Fractions of land cover classes are assumed to be fixed throughout the entire model simulation (e.g. no deforestation), i.e. there is no land use/cover change. For this scenario, only natural surface water bodies, e.g. rivers, wetland and lakes, are considered. Reservoirs (dam constructions) are not simulated.
- No water demand was simulated, and, therefore, no water abstraction.

Anthropogenic scenario:

- A grid-cell in this scenario constitutes up to five land cover classes: short natural vegetation, tall natural vegetation, surface water bodies (including reservoirs), as well as two classes of irrigated crop types: paddy and non-paddy types.
- The parameters for the first two classes (natural, non-irrigated land types) are basically based on the GLCC 2.0 (see the aforementioned explanation about the near-natural scenario).
- For this scenario, areal extents of fractions of all land cover classes change on yearly basis, particularly due to expansion of irrigated areas and progressive construction of dams/reservoirs. Therefore, land use/cover change is simulated.
- To parameterize the reservoirs, the GRanD dataset was used.
- Water demand is simulated and, therefore, water abstraction is also simulated.

The result of the two scenarios was two data sets of daily discharges for a ten year period (2001-2010). One dataset represents the daily discharges for the anthropogenic scenario (baseline), while the second dataset simulates daily discharges under the near-natural (no abstraction) scenario. The purpose of the 2nd dataset is to simulate the hydrologic conditions in Europe under a status of minimal anthropogenic pressures on water arising from abstractions and land use modifications. By implementing the IHA method for analyzing these two hydrologic datasets we can compare the results and derive a degree of alteration between the baseline conditions and the near-natural scenario expressing proxies of pressures related to hydrologic alteration.

2.2. Connecting modeled data to FECs

The next crucial step of the implemented methodology was to assign efficiently the gridded hydrologic data produced by the PCR-GLOBWB model to the Functional Elementary Catchment (FEC) level. The GIS procedure described below, resulted to a hydrologic dataset comprised by daily data for a 10-year period for all the 104,334 FECs included in the MARS GeoDatabase (http://mars-project.eu/downloads/geodatabase/MARS_geodatabase_20150930/MARS_Geodatabase_20150930.gdb.7z).

At first, the centroids of the FEC polygons were calculated and a new shapefile was produced as shown in Figure 2.2. The share of the upstream area that corresponds to each FEC was obtained from the Ecrins (http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network, last modified 22 March 2016) and was matched with the FEC centroid objects. Next, a new shapefile of the centroids of the modelled raster cells was created adding the upstream area for each cell (Figure 2.3).

Figure 2.2. An example figure showing calculated centroids of FECs polygons.

Figure 2.3. An example figure showing calculated centroids of PCR-GLOBWB model raster cells.

For each FEC's centroid a buffer with a 15 km radius was created (Figure 2.4) and then intersected with the PCR-GLOBWB centroid shapefile to identify which cell centroids fall within the buffer area of each FEC's centroid.

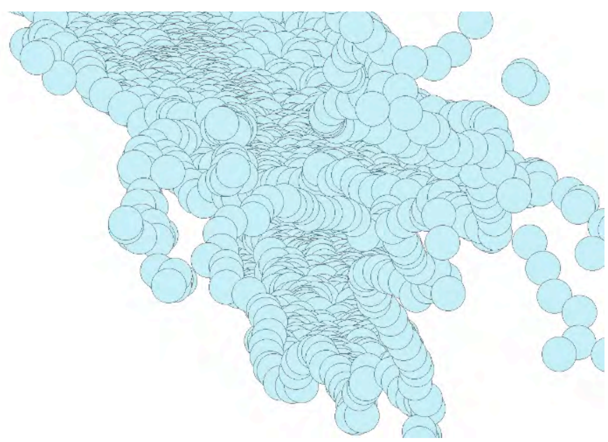


Figure 2.4. An example figure showing buffer zones created based upon the FEC centroids.

This resulted into having several grid points in one FEC buffer (Figure 2.5). Then for each case (FEC buffer) we selected the cell centroid for which the absolute difference between the FEC's upstream area and Grid cell's upstream area was the minimum. This allowed us to minimize the number of cases where a grid cell with a large upstream area was wrongly assigned to a FEC with minimal influence from the upstream area (e.g. a small tributary).

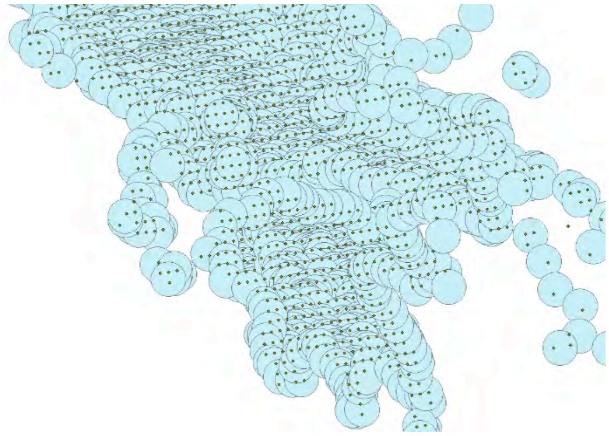


Figure 2.5. An example figure showing the PCR-GLOBWB centroid cells that fall within the created buffer zones.

In this way the upstream area of the FEC and the upstream area of the grid cell were strongly correlated as shown in Figure 2.6.

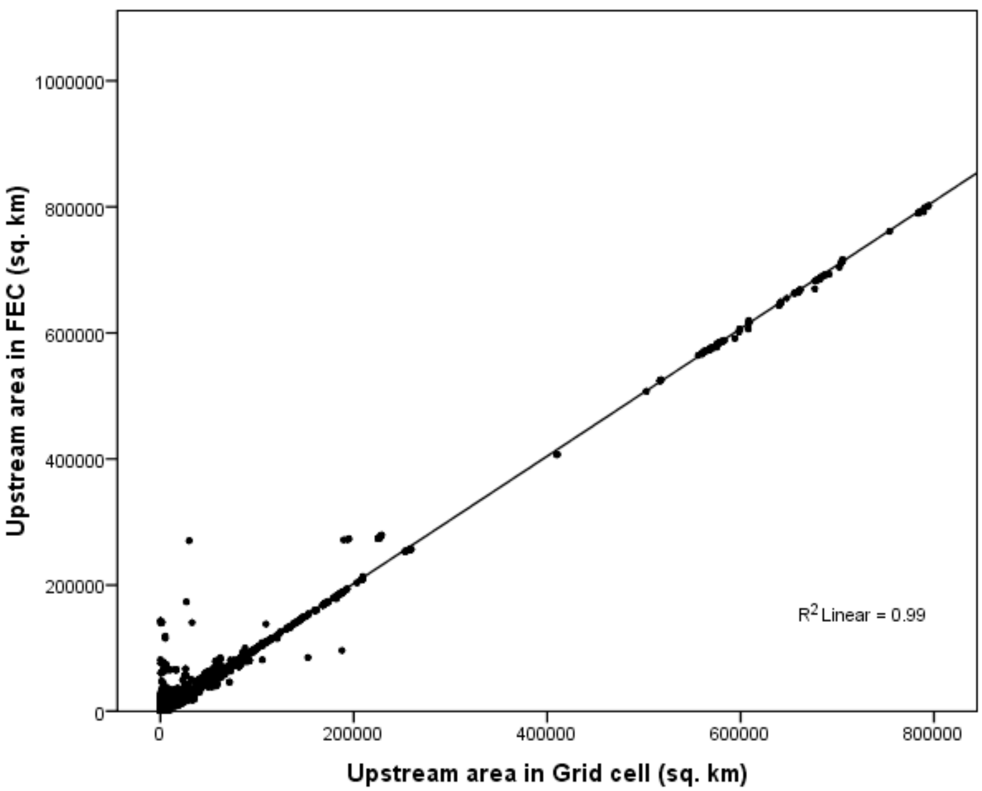


Figure 2.6. Scatter plot between upstream areas of the FECs and the respective grid cells.

2.3. Calculation of Indicators of Hydrologic Alteration

The next step, after assigning hydrologic information (daily flows) to each FEC was the calculation of a series of indicators of hydrologic alteration proposed originally by the methodological framework of Richter et al. (1996) (Tables 1.2 and 1.3). The calculation was achieved with the use of the IHA software (IHA, 2009). As already described, the IHA calculates a total of 67 statistical parameters and some additional more general ones, for the entire simulation period. These parameters are subdivided into two groups, the IHA parameters (33) and the EFC parameters (34). Parameters can be calculated using parametric (mean/standard deviation) or nonparametric (percentile) statistics. However, because of the skewed nature of most hydrologic datasets, non parametric statistics were selected as the best option for the calculation of the IHA parameters in this task (IHA, 2009). In the next paragraphs we provide a brief description of the algorithm used for the calculation of the IHA and EFC parameters.

For the calculation of the IHA parameters the 3-, 7-, 30-, and 90-day minimums and maximums were taken from moving averages of the appropriate length calculated for every possible period that is completely within the water year. The zero flow days and base flow index parameters were also calculated here. Also, reversals were calculated by dividing the hydrologic record into "rising" and "falling" periods, which correspond to periods in which daily changes in flows are either positive or negative, respectively. A rising or falling period is not ended by a pair of days with constant flow, only by a change of sign in the rate of change. The number of reversals is the number of times that flow switches from one type of period to another (IHA, 2009).

For the calculation of the EFC parameters the IHA software uses an algorithm that distinguishes between high flows and low flows. Specifically, this is achieved with the use of four parameters:

High flow threshold: All flows greater than this threshold are classified as high flows. This parameter can be specified as a percentile of all daily flows or as a flow value. The default value is the **75th** percentile of daily flows and was used in this study.

Low flow threshold: All flows less than or equal to this threshold are classified as low flow events. This parameter must always be less than the *high flow threshold.* This parameter can be specified as a percentile of all daily flows or as a flow value. The default value is the **50th** percentile of daily flows and was used in this study.

High flow start rate threshold: When flows are between the *high flow* and *low flow thresholds*, this parameter controls the start of high flow events. It also controls whether the ascending limb of an event is restarted from the descending limb. The default value is **25%** and was also used in this study.

High flow end rate threshold: When flows are between the *high flow* and *low flow thresholds*, this parameter is used to end high flow events during their descending limb. It also controls the transition between the ascending and descending limb of an event. The default value, also used here, is **10%**.

Then the algorithm further distinguishes the high flow events to small floods and large floods according to the below criteria:

Small flood minimum peak flow: All high flow events that have a peak flow greater than or equal to this value (and less than the peak flow value for large floods, if there are three flow classes) will be assigned to the small flood class. All events with a peak flow less than this value will be assigned to the high flow pulse class. The user has the option to enter this as either a return interval, a flow value, or a percentile of all daily flows. We used the default **2-y** return interval.

Large flood minimum peak flow: All high flow events that have a peak flow greater than or equal to this value will be assigned to the large flood class. All events with a peak flow less than this value will be assigned to the high flow pulse class or the small flood class, depending on whether there are two or three high flow classes. The user has the option to enter this as either a return interval, a flow value, or a percentile of all daily flows. We used the default **10-y** return interval. It should be noted that the length of the available time-series in this study (10 years) is rather limited for the calculation of representative flood indicators.

Finally, all low flow days with a flow value less than or equal to the extreme low flow threshold will be classified as extreme low flows. The default value is the **10th** percentile of the daily flows.

An example of categorization of daily flows into the five major ecological flow components is shown in Figure 2.7.

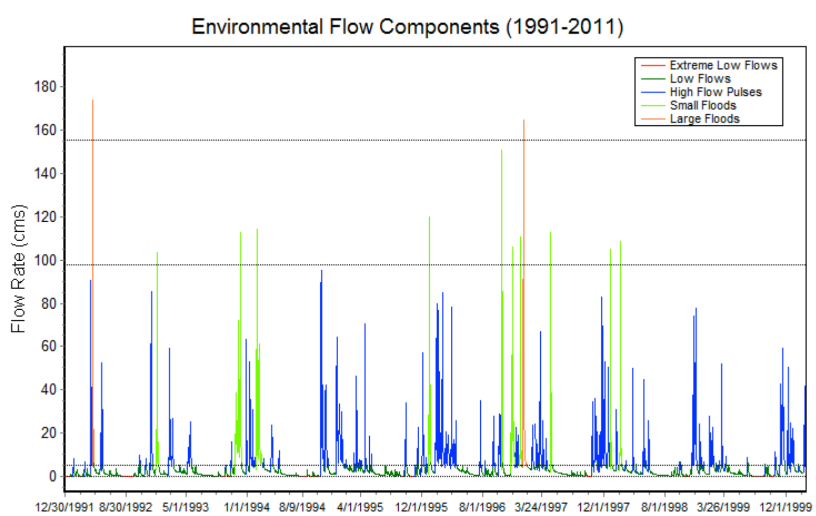


Figure 2.7: Example of daily flows categorized in five major flow components in a subbasin of Pinios catchment in Greece.

The calculation of IHA and EFCs provide the option to generate spatial maps depicting their variability across Europe. The thematic maps can be a first tool to assess variability and identify important hydrologic indicators able to express hydrologic alteration and potentially its connection with the ecological status.

2.4. Broad river typology

To investigate the variation of certain hydrologic indicators among areas and regions of Europe, the existing broad river typology was applied where FECs are classified into 20 Broad River Types (BRTs) according to the criteria described in Solheim et al. (2015) (Table 2.1). The BRTs were defined after taking into consideration a combination of ecological characteristics, feedback from the countries and the need to limit their number for meaningful EU-level assessments of status and pressures. Figure 2.8 depicts the 20 BRTs on a map of Europe.

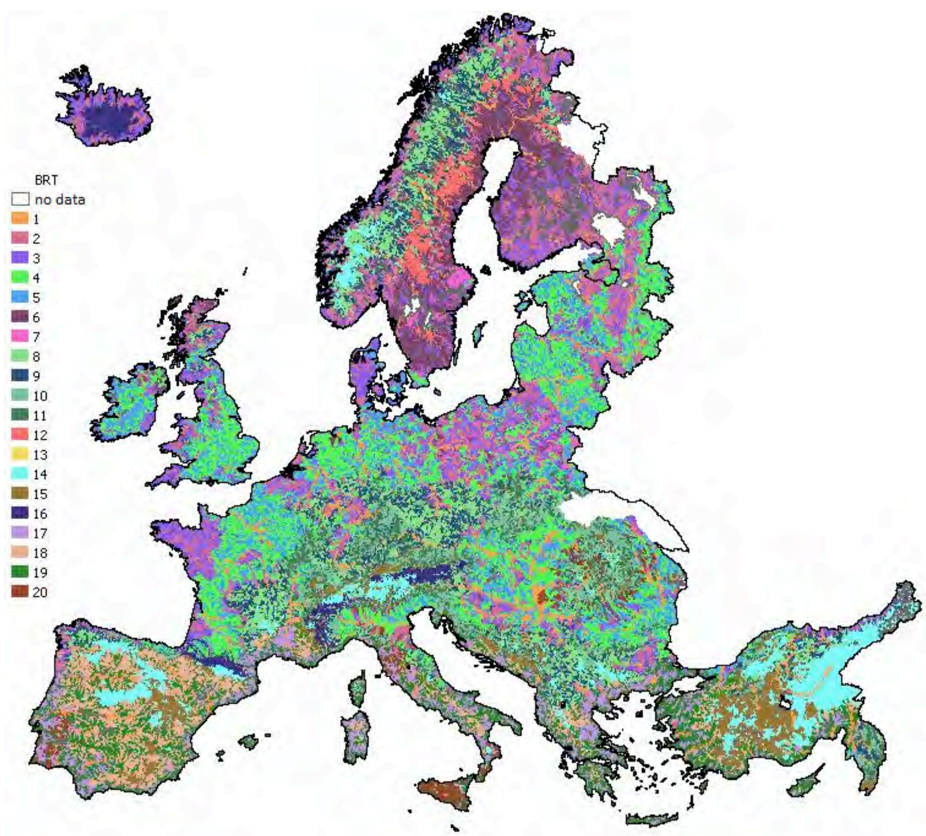


Figure 2.8: The 20 Broad River Types of Europe (explanations in Table 2.1 and in Solheim et al., 2015).

Table 2.1. Broad river typology (from Solheim et al., 2015).

Broad river type name	Code	Altitude (masl)	Catchment area (km^2)	Geology	National types	WBs	% of WBs	No of FECs
Very Large Rivers (all Europe)	1	Any	>10000	Any (usually mixed)	54	827	1.0	4529
Lowland, Siliceous, Medium-Large	2	<200	100-10000	Siliceous	24	1139	1.4	8732
Lowland, Siliceous, Very small-Small	3	<200	<100	Siliceous	29	7285	8.8	11534
Lowland Calcareous or Mixed, Medium-Large	4	<200	100-10000	Calcareous/Mixed	68	2873	3.5	9306
Lowland Calcareous or Mixed, Very small-Small	5	<200	<100	Calcareous/Mixed	47	14137	17.1	7483
Lowland, Organic and Siliceous	6	<200	<10000	Organic and Siliceous	18	6193	7.5	3402
Lowland, Organic and Calcareous/Mixed	7	<200	<10000	Organic and Calcareous/Mixed	10	353	0.4	528
Mid altitude, Siliceous, Medium-Large	8	200-800	100-10000	Siliceous	41	3051	3.7	6641
Mid altitude, Siliceous, Very small-Small	9	200-800	<100	Siliceous	37	8627	10.5	8641
Mid altitude, Calcareous or Mixed, Medium-Large	10	200-800	100-10000	Calcareous/Mixed	60	1796	2.2	4464
Mid altitude, Calcareous or Mixed, Very small-Small	11	200-800	<100	Calcareous/Mixed	48	7663	9.3	5316
Mid altitude, Organic and siliceous	12	200-800	<10000	Organic and Siliceous	8	3290	4.0	1497
Mid altitude, Organic and Calcareous/Mixed	13	200-800	<10000	Organic and Calcareous/Mixed	6	154	0.2	40
Highland (all Europe), Siliceous, incl. Organic (humic)	14	>800	<10000	Siliceous	16	1525	1.8	5820
Highland (all Europe), Calcareous/Mixed	15	>800	<10000	Calcareous/Mixed	17	2227	2.7	4235
Glacial rivers (all Europe)	16	>200	<10000	any	16	3251	3.9	1804
Mediterranean, Lowland, Medium-Large, perennial	17	<200	100-10000	any	16	941	1.1	3116
Mediterranean, Mid altitude, Medium-Large, perennial	18	200-800	100-10000	any	13	615	0.7	3935
Mediterranean, Very small-Small, perennial	19	<800	<100	any	21	1942	2.4	10623
Mediterranean, Temporary/Intermittent streams	20	any	<1000	any	26	3549	4.3	1340
Total					*575*	*71438*	*86.6%*	

The result of the broad river typology led to a classification of 575 national river types from 26 countries into 20 broad types. The proportion of river water bodies included in these national types comprises 87% of all river water bodies in the countries that could be included in the analysis, including natural rivers, as well as heavily modified and artificial water bodies. Broad Type 3 (Lowland, Siliceous, Very small-Small) and Broad Type 19 (Mediterranean, Very small-Small, perennial) are represented by 11,534 and 10,623 FECs, respectively, which means that 21.2% of the FECs correspond to two BRTs. The least representative BRT is type 13 (Mid altitude, Organic and Calcareous/Mixed) with 40 FECs.

2.5. European datasets with biological/ecological data

In order to investigate potential relationships between the hydrology and the ecology at the European scale we used ecological data that are available from the WISE SoE River database (http://www.eea.europa.eu/data-and-maps/data/waterbase-rivers-10). The WISE SoE River databases contains EQR values and ecological status information for two BQEs (Macroinvertebrates and Phytobenthos, 5,200 and 2,600 samples approximately) (Figure 2.9 and Figure 2.10).

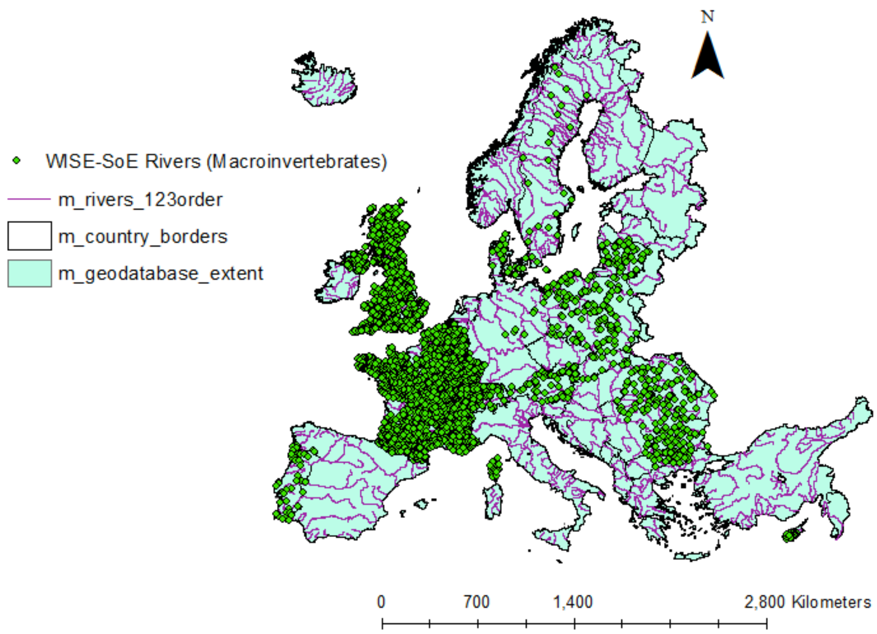


Figure 2.9: Sites from the WISE-SoE Rivers database that contain EQR data and ecological status based on macroinvertebrate assessment.

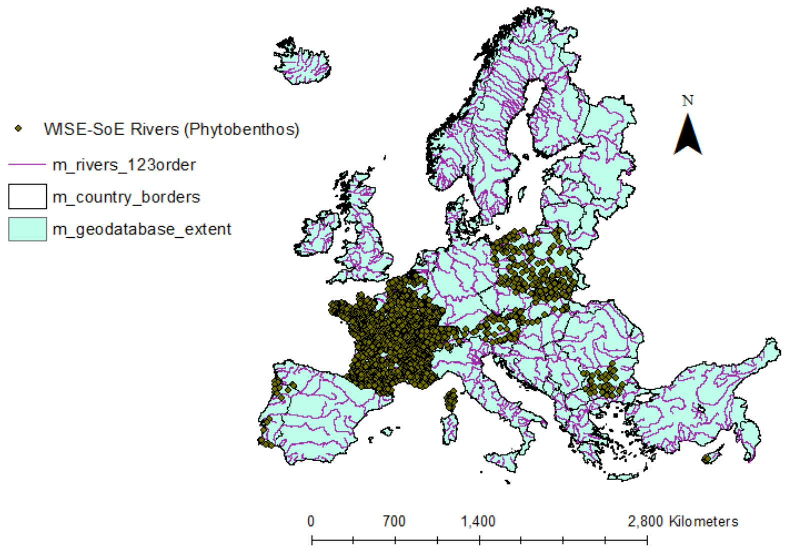


Figure 2.10. Sites from the WISE-SoE Rivers database that contain EQR data and ecological status based on phytobenthos assessment.

The assignment of the WISE SoE stations to the FECs was based on the distance between the SoE station and the FEC outflow point (the closest station to the FEC outflow was selected). This procedure resulted to a dataset that contains EQR data corresponding to 1604 different FECs for the BQE of macroinvertebrates and 1277 FECs for the BQE of phytobenthos (Figure 2.11). These data are unevenly distributed among the 20 Broad River Types and for almost half of the types the available SoE data correspond to less than 50 cases (FECs). Moreover almost 86% of the SoE data on macroinvertebrates and 60% of SoE data on phytobenthos correspond to Good or High status meaning that the data are unevenly distributed among the ecological classes and there are very few occasions where Ecological Status is classified as Poor or Bad (Figure 2.12). These issues could possibly affect the analysis of the relations between the EQR data and the IHA/EFC parameters, particularly for the BRTs that the SoE data are scarce.

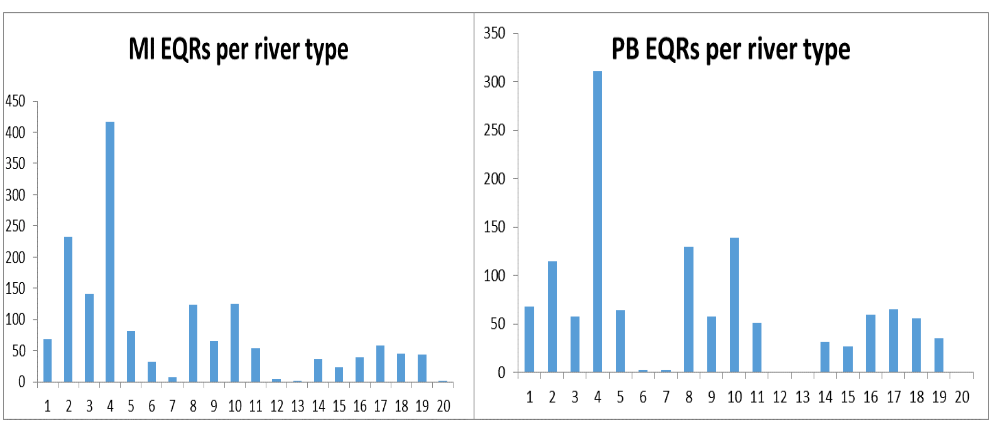


Figure 2.11. Bars show the number of FECs per Broad River Type where macroinvertebrate (MI) and phytobenthos (PB) EQR data are available.

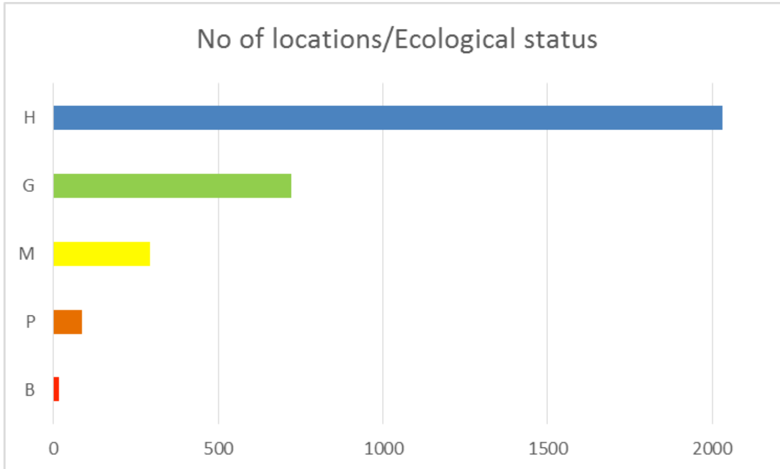


Figure 2.12. Horizontal bars show the number of locations per ecological class assessed based on macroinvertebrates.

2.6. Data and mapping analysis

Data Analysis at Pan-European scale

As described in the previous section, the Indicators of Hydrologic Alteration (IHA) and Environmental Flow Components (EFCs) have been calculated for two datasets of simulated daily discharges. One dataset is modelled assuming zero water abstractions and natural type of land uses (near-natural scenario), and the other dataset is obtained through baseline model runs (anthropogenic scenario) of the PCR-GLOBWB model. In order to assess the deviation of the baseline hydrologic conditions from the near-natural scenario we calculated both the differences and the ratios between the values of the indicators for the near-natural scenario and the values of the indicators for the anthropogenic scenario (near-natural scenario minus anthropogenic scenario and near-natural scenario over anthropogenic scenario). Focusing on ratios, if the value for a certain indicator is 1, it means that there is no alteration between the "anthropogenic" model run and the "near-natural" model run. If the ratio is above 1, then the value of the hydrologic indicator for the anthropogenic is lower than the near-natural scenario . Focusing on differences, the critical value showing no alteration is zero.

The values of the IHA and EFCs for the anthropogenic scenario and the values of the ratios were compared among the 20 different BRTs with the use of non-parametric Kruskal Wallis test. Pairwise comparisons for each indicator were further examined with the use of Dunn-Bonferroni post hoc tests (SPSS v 23). These are presented in Tables II-III-IV of the Annex.

In order to investigate significant relationships between the ratios of the hydrologic indicators and the EQRs using macroinvertebrates and phytobenthos, Spearman correlations were run for the whole Pan-European dataset and for each Broad River Type dataset separately. Redundant hydrologic indicators were excluded after testing for collinearity according to to the Variation Inflation Factor (VIF) criterion (Feld et al., 2016). Following a stepwise procedure, variables with VIF value larger than seven were removed. The remaining variables (Table 2.2) were tested for significant Spearman correlations with the EQR values. The same procedure was applied for each separate BRT dataset.

Table 2.2: Remaining IHA and EFC parameters after eliminating redundant ones (VIF>7)

Parameter	VIF
Annual CV	5.213
Constancy/predictability	3.967
No of floods in a 60d period	1.211
Flood free season	1.337
30 day minimum flow	6.103
Low pulse count	2.673
Low pulse duration	1.734
High pulse count	3.314
High pulse duration	1.563
Rise rate	1.755
Fall rate	2.044
Number of reversals	2.062
April Low Flow	5.769
Extreme low duration	1.233
Extreme low freq.	1.262
High flow duration	1.505
High flow frequency	4.209
High flow fall rate	5.065
Small Flood duration	1.481
Small Flood freq.	1.466
Small Flood rise rate	2.038
Small Flood fall rate	1.535
Large flood peak	3.102
Large flood duration	1.388
Large flood rise rate	1.242
Large flood fall rate	1.567
Total No Days of extreme low flows	1.587
Total No Days of Low flows	3.708
Total No Days of High flow pulses	4.036
Total No Days of Small Floods	1.795

Methods and data analysis at catchment and regional scale

The possible effects of certain hydrologic indicators on the river ecology was further examined at catchment and regional scales by using two additional datasets. The first dataset contains macroinvertebrate community data obtained from 30 samples taken from 30 sites in the Greek Pinios catchment. The second dataset is comprised of macroinvertebrate data obtained from 192 samples taken from 92 sites distributed among two federal states in Germany (North Rhine-Westphalia and Saxony Anhalt).

Pinios catchment

The Pinios catchment covers almost entirely the River Basin District of Thessaly in Central Greece (Figure 2.13). The mean annual Pinios river flow at the outlet (Aegean sea) is reported close to 80 m^3/s and the mean annual precipitation of the catchment around 700 mm (Panagopoulos et al., 2014; Stefanidis et al., 2016). The latter is characterized by an uneven temporal distribution with mean monthly precipitation in winter exceeding 80 mm but being around only 20 mm in summer. The Pinios flows permanently to the outlet, even with some small water quantities during the driest periods of the year occurring due to the continuous baseflow contribution of the mountaneous areas in the northern and western part of the catchment.

The macroinvertebrate community dataset of Pinios contains qualitative information at family level from a total of 30 sites across the catchment. These sampling sites were selected due to their proximity to the outflow point of the FEC that are located within. The data were obtained from previously published sources (Chatzinikolaou, 2007; Chatzinikolaou et al. 2010) and the Greek National Monitoring programme. From the available data we calculated the BMWP (Biological Monitoring Working Party) and the ASPT (Average Score Per Taxon) score according to the BMWP system (Armitage et al., 1983), the number of Ephemeroptera, Plecoptera and Trichoptera families (EPT) and the total number of taxa (richness).

Collinearity among the IHA and EFC variables was assessed by stepwise removal of the variables that exceeded a VIF (variation inflation factor) value of 7. Spearman correlations were run between the remaining variables and the macroinvertebrate metrics in order to detect significant relationships.

Relationships between IHA and EFC parameters and macroinvertebrate community data were analysed with a Canonical Correspondence Analysis (CANOCO 4.5). The “significance” of the remaining variables was examined with the use of Monte Carlo random test (Table 2.3)

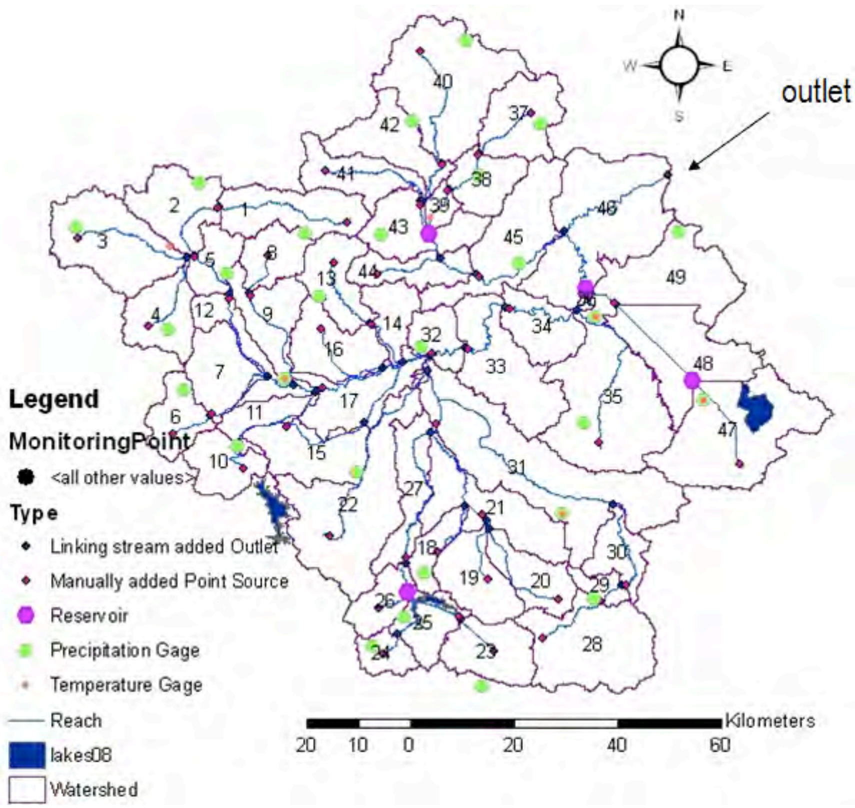


Figure 2.13: Catchment of Pinios and the subbasins with their outlets

Table 2.3: IHA and EFC parameters included in the CCA.

Variable	Monte carlo sig.	F
Small Flood Peak	***0.004***	2.09
Low Pulse Threshold	***0.002***	2.31
High Pulse duration	***0.004***	2
30 day min.	***0.004***	1.7
High Flow duration	***0.016***	1.69
November Low Flow	***0.046***	1.49
Extreme Low Flow Duration	0.106	1.36
Extreme Low peak	0.068	1.39
October Low Flow	0.152	1.28
Fall Rate	0.172	1.22
January Low Flow	0.284	1.16
Base Flow Index	0.288	1.15
High Flow Rise Rate	0.362	1.07
Small Flood fall rate	0.41	1.05
Rise Rate	0.418	1.02

German WFD data

The German WFD database contains data for 92 sites of European BRT 4 (Lowland Calcareous or Mixed, Medium-Large) according to ETC/ICM (2015), which cover 40 different medium to large lowland rivers (Figure 2.14). This broad type corresponds to the German types 15, 15G and 17 (AQEM Consortium, 2004). Macroinvertebrate data were obtained from 192 samples taken from 2004 to 2013. The data were collected from February to November using a multi-habitat sampling approach (Hering et al., 2004). For each sample, 20 representative sampling units were taken that cover all important microhabitat types (at least 5% of the sample reach) using a kick-net with 25×25cm frame and a mesh size of 500µm. The highest taxonomic level was identified which was not always as high as the species level. Invertebrate metrics were calculated with the Software ASTERICS V. 4.0.4 based on the operational taxa list (AQEM Consortium, 2004).

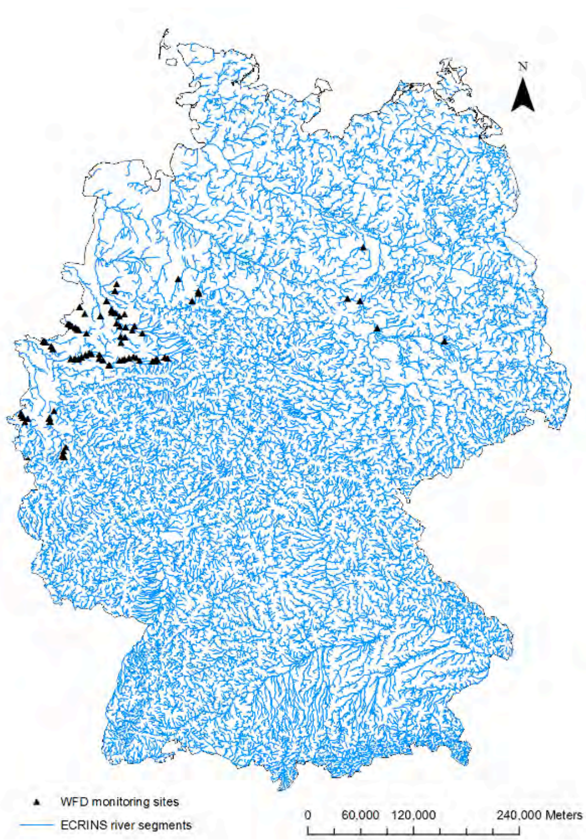


Figure 2.14. Distribution of 92 sites with WFD macroinvertebrate monitoring data in Germany.

Preliminary correlation analyses were run in order to find possible relationships between the macroinvertebrate metrics and the IHA and EFC parameters. Further data analysis included a Principal Component Analysis for the total of the hydrologic variables in order to reduce the number of the explanatory variables into principal components. Spearman correlations were run between the principal components and the macroinvertebrate metrics.

3. RESULTS

3.1. Anthropogenic scenario

According to the results of Kruskal-Wallis test all the IHA and EFC parameters for the anthropogenic scenario were significantly different ($p < 0.05$) among the 20 BRTs. Figures 3.1, 3.2, 3.3, 3.4 and 3.5 illustrate the mean value of selected IHA and EFC parameters among the BRTs. These figures facilitate the identification of notable variations of selected IHA and EFC parameters among the 20 BRTs. For example Figure 3.1 shows that BRT 1 (Very Large Rivers) is clearly characterized by the very high annual discharge. Regarding the Base Flow Index, it seems that types 4, 5 (Lowland calcareous or mixed) and 20 (Mediterranean Temporary) present the lowest values and especially the BRT 20. As far as other IHA and EFC parameters are concerned, BRTs 6 and 12 (Lowland and Mid-altitude organic and siliceous) appear to differentiate from other types as exhibit the highest duration in Extreme Low Flow, High Flow, Small Flood and Large Flood events and the lowest frequency of High Flow and Extreme Low Flow events.

3.2 Deviation from the near-natural scenario

According to the results of Kruskal-Wallis test all the IHA and EFC parameters were significantly different ($p < 0.05$) among the 20 BRTs. The mean ratios of selected IHA and EFC parameters at the near-natural" to anthropogenic scenario are graphically presented through Figures 3.1 to 3.5. The graphs help to identify which BRTs show the largest "change" of the IHA between the anthropogenicscenario and the near-natural scenario. The higher the value of the ratio, the higher the deviation from the "natural" conditions.

For example, BRT 1 (Very Large Rivers) showed the highest ratio values for the mean annual discharge, suggesting that the anthropogenic pressures in the catchments belonging to this Type have a very large effect on hydrology. The variation of the Base Flow Index ratio among the BRTs showed remarkable higher values in Type 20 (Mediterranean Temporary) than the other Types, suggesting that the base flow in the catchments of this type is heavily influenced by the water abstractions. The variation of the ratios of low and high pulse thresholds among the BRTs is similar to the variation of the annual discharge ratio since thresholds definitions are standard (the 50^{th} and the 75^{th} percentiles of the daily flows). Bars that are exceeding the value of one (dashed line) indicate that under the naturalized conditions the indicator value is higher than the anthropogenic conditions. Depending on the indicator, a ratio > 1 could suggest a negative impact of anthropogenic induced pressures on hydrology (e.g lower annual discharge). Ratios very close to one indicate no alteration between the two scenarios, while ratios lower than one would imply that the value of the indicator under the anthropogenic conditions scenario is higher than the "naturalized" conditions, with this implying positive or negative impacts on hydrology depending on the indicator. Ratios regarding the small and large flood duration exceed the value of one for 18 out of 20 BRTs for the small flood duration and for 19 out of 20 BRTs for the large flood duration.

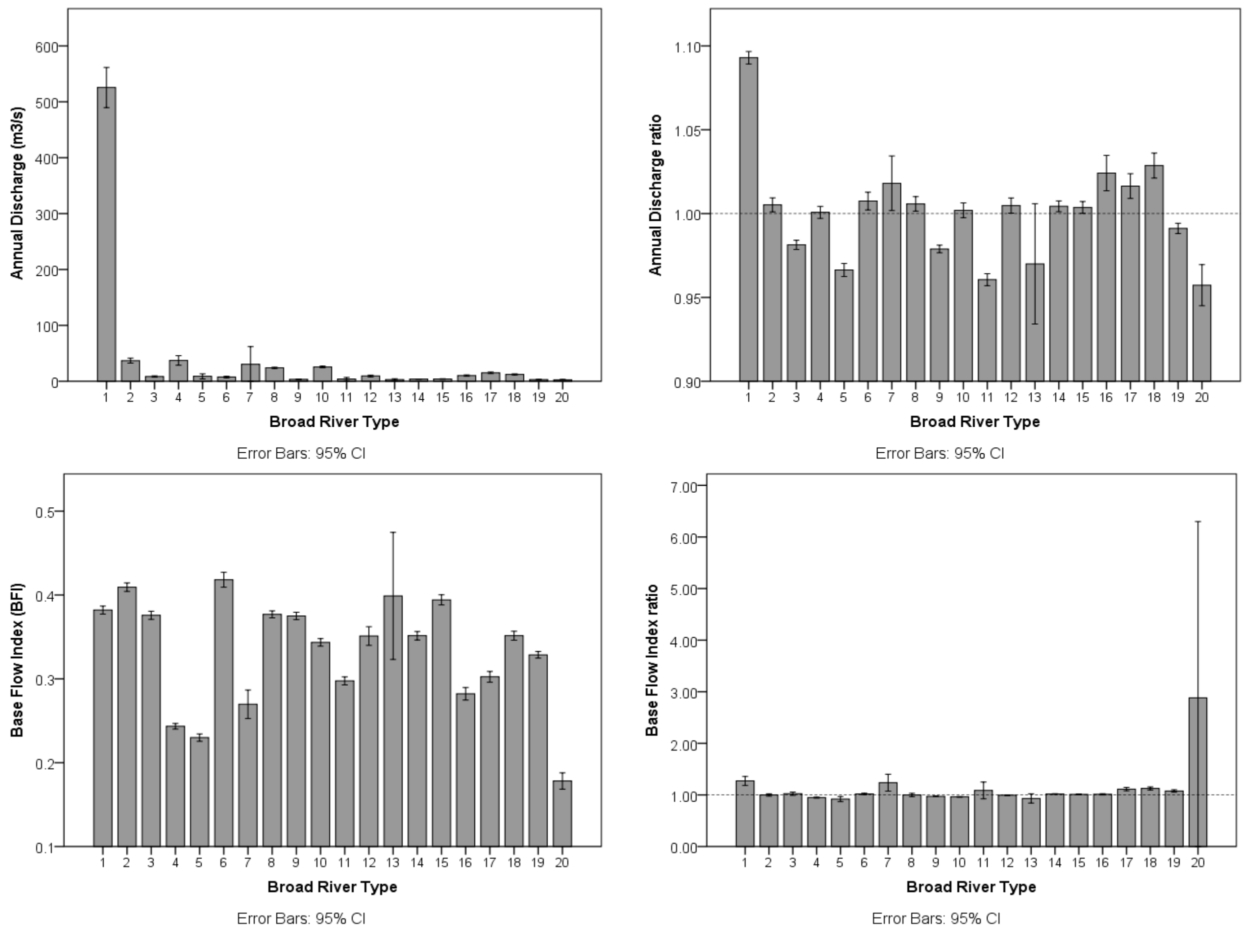


Figure 3.1. Bars showing mean annual discharge and base flow index (BFI) per Broad River Type for the anthropogenic scenario (top left and bottom left graphs) and the mean ratio of annual discharge and base flow index per BRT (top right and bottom right graphs). Error bars indicate 95% CL.

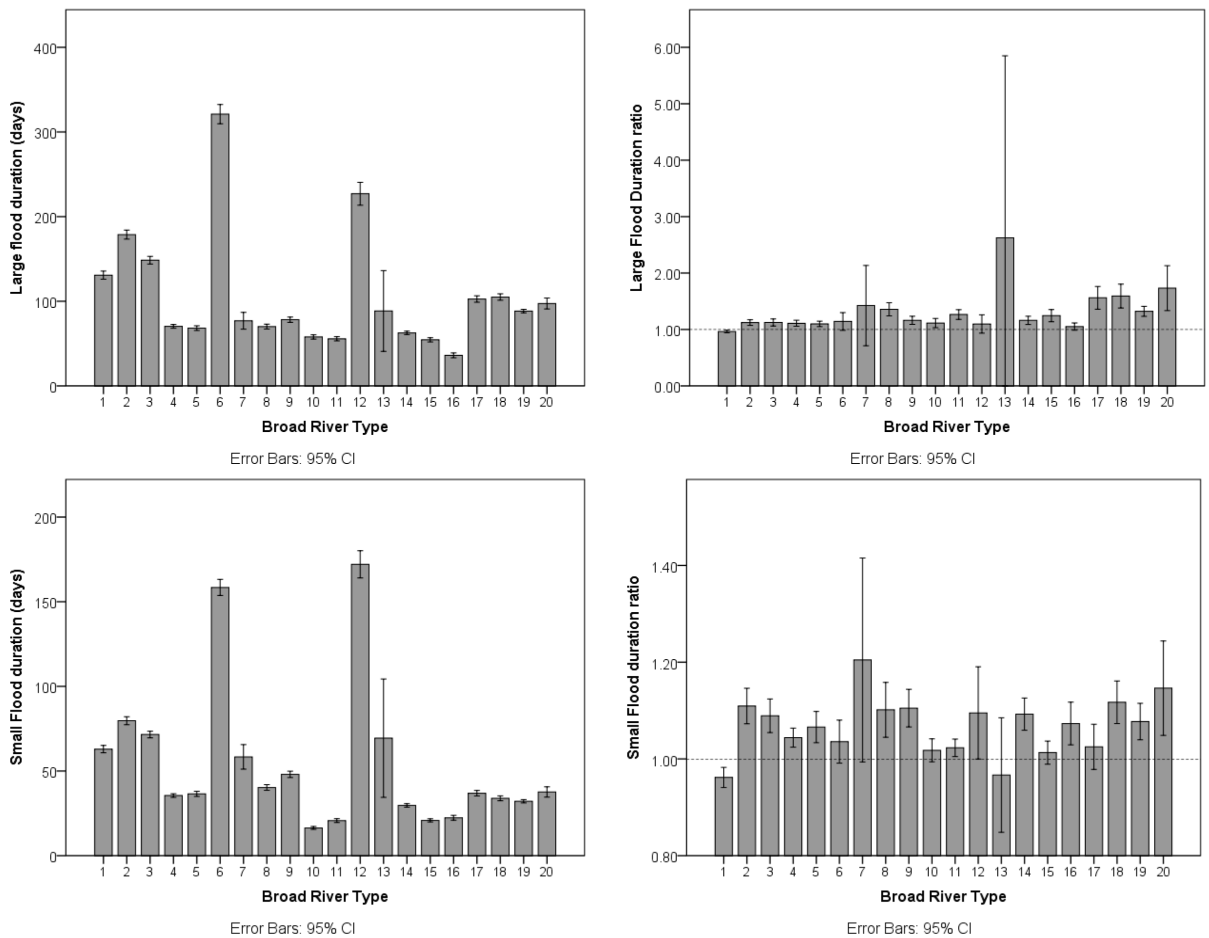


Figure 3.2. Bars showing mean large flood and small flood duration per Broad River Type for the anthropogenic scenario (top left and bottom left graphs) and the mean ratio of large flood and small flood duration per BRT (top right and bottom right graphs). Error bars indicate 95% CL.

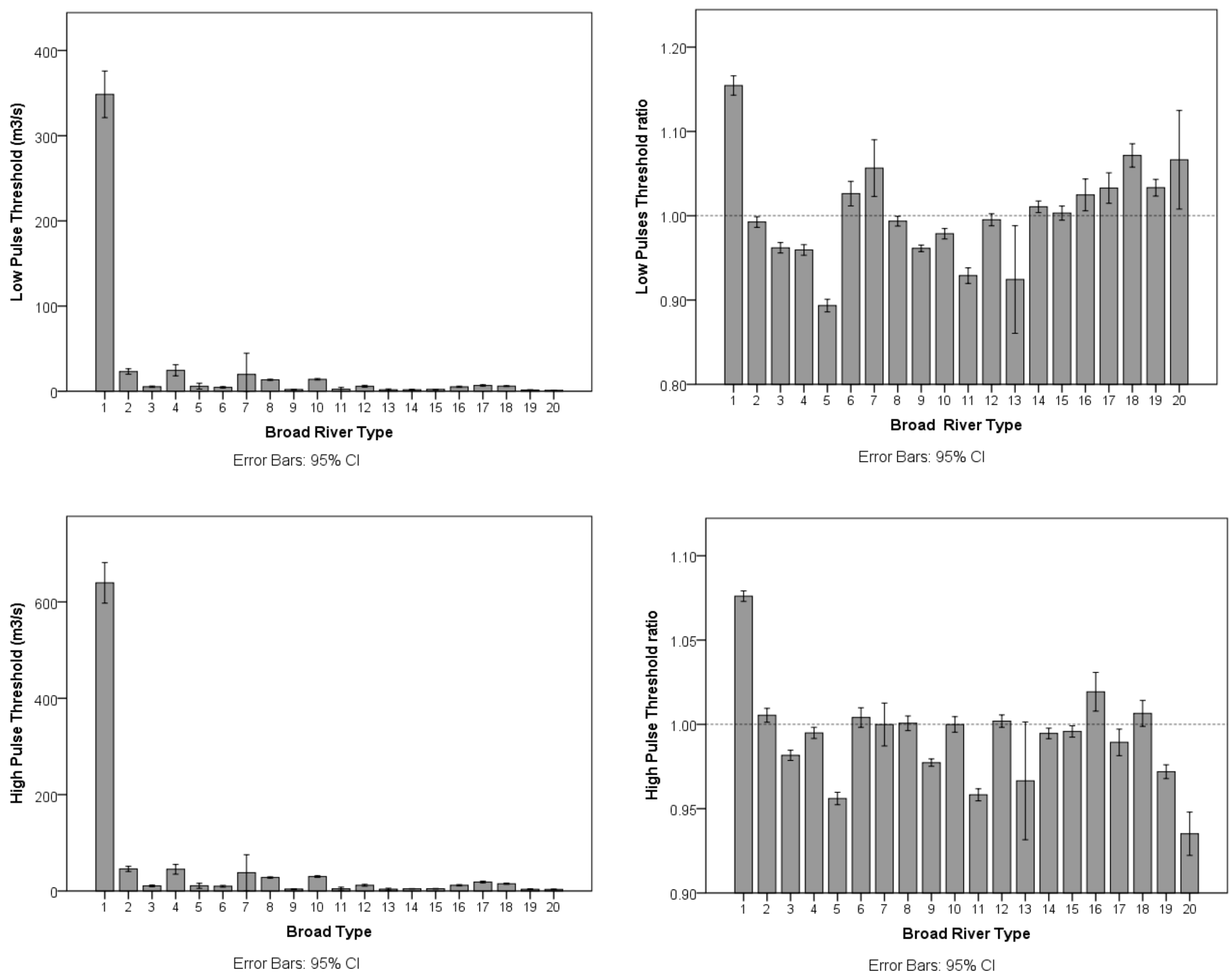


Figure 3.3. Bars showing mean low pulse and high pulse threshold per Broad River Type for the anthropogenic scenario (top left and bottom left graphs) and the mean ratio of low pulse and high pulse per BRT (top right and bottom right graphs). Error bars indicate 95% CL.

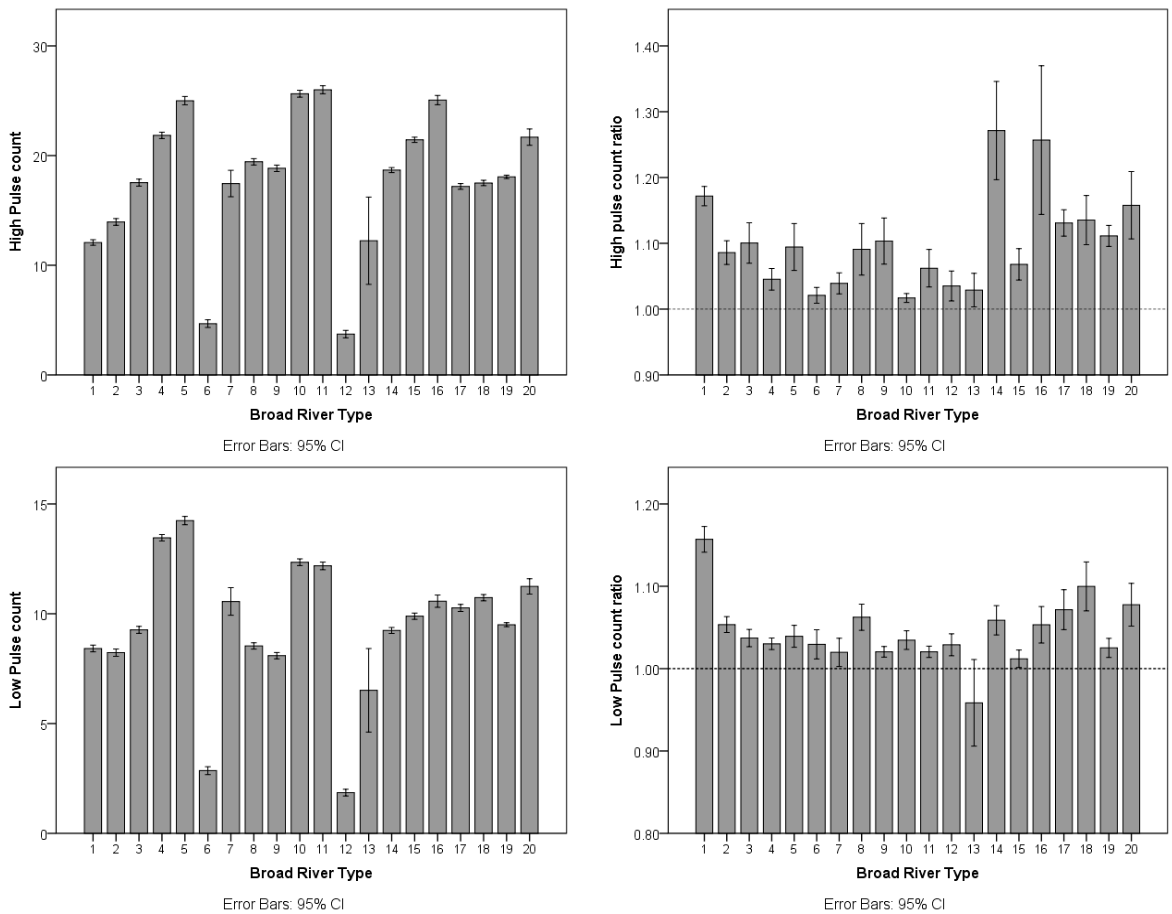


Figure 3.4. Bars showing mean high pulse and low pulse count per Broad River Type for the anthropogenic scenario (top left and bottom left graphs) and the mean ratio of high pulse and low pulse per BRT (top right and bottom right graphs). Error bars indicate 95% CL.

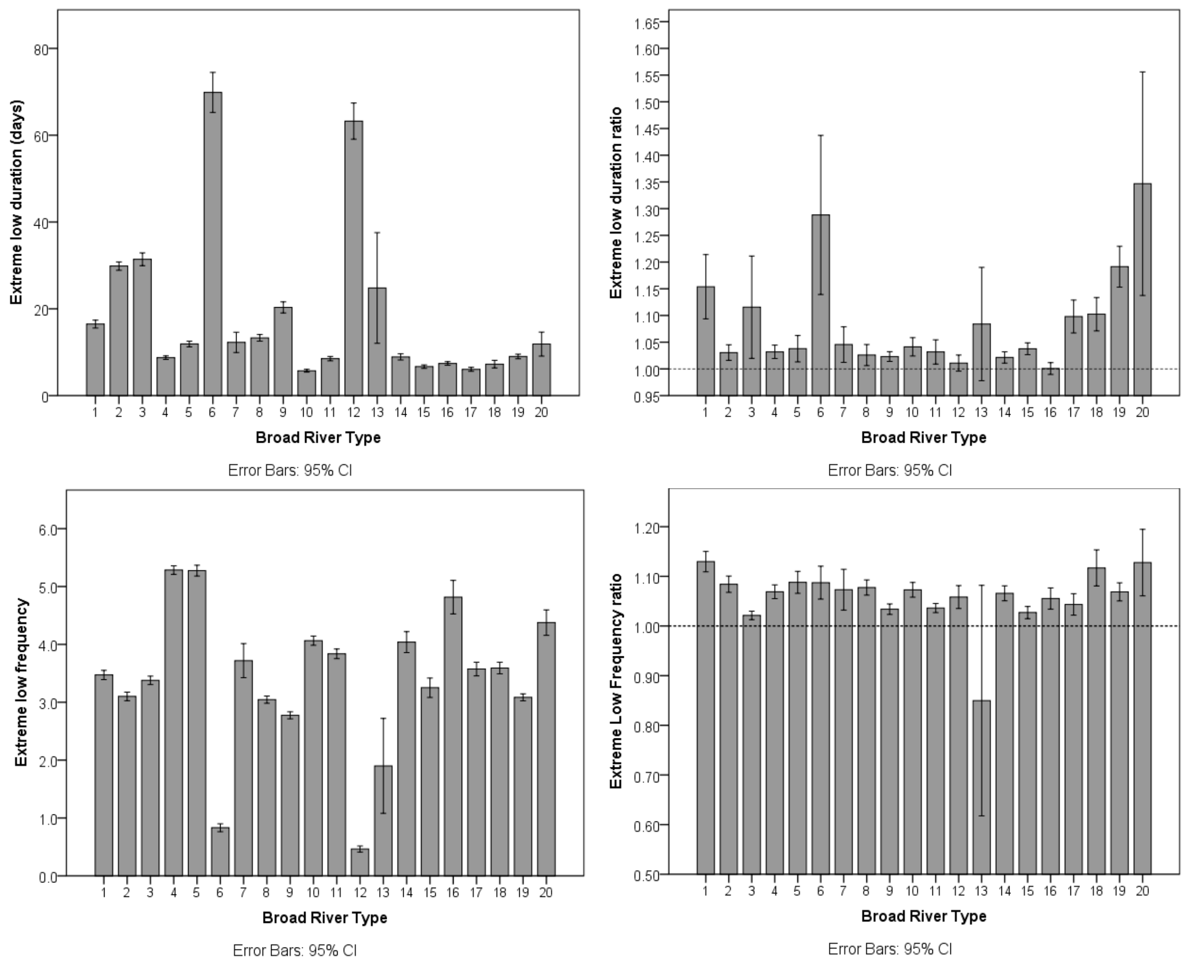


Figure 3.5. Bars showing mean extreme low duration and extreme low frequency per Broad River Type for the anthropogenic scenario (top left and bottom left graphs) and the mean ratio of extreme low duration and extreme low frequency per BRT (top right and bottom right graphs). Error bars indicate 95% CL.

3.3 Relations of IHA and EFCs with EQRs at pan-European scale

The results overall showed very weak correlations between the IHA and EFC parameters and the EQRs ($p>0.05$), suggesting that there is no connection between key hydrologic variables and EQRs based on macroinvertebrates and phytobenthos regardless of the BRT. When examining correlations per BRT the results showed that for certain types there were few significant correlations ($p< 0.05$). For example for BRT 1 (Very Large Rivers) the Spearman correlations revealed significant results ($p< 0.05$) between EQR based on macroinvertebrates and several hydrologic indicators such as 30 day minimum flow, April low flow, December low flow, Number of reversals, small flood fall rate and small flood rise rate (Table 3.2). For other Types, such as BRT 3 (Lowland siliceous very small), no significant correlations were identified.

*Table 3.2. Most notable Spearman correlations between IHA and EFC parameters and EQR (r coefficients) for MI among the BRTs 6,7,10,12,13,19 and 20 are not listed due to SoE data scarcity. * Indicates statistically significant relationship (p<0.05), ** Indicates statistically significant relationship (p< 0.001).*

	BT 1	BT 2	BT 3	BT 4	BT 5	BT 8	BT 9	BT 11	BT 14	BT 15	BT 16	BT 17	BT 18
	r coef.	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*	*r coef.*
30 day minimum	0.239*												
Annual C.V						0.341**							
April Low flow	0.257*												
December Low flow	0.316*												
Extreme low duration												-0.319*	
Extreme low Peak							0.328*					0.286*	
Fall rate		0.064*				-0.198*							
February low flow													
Flood free season													
Flow Predictability													
High flow duration													
High flow fall rate													
High Flow Freq.													
High flow pulses													
High pulse duration		-0.054*										-0.203*	
Large flood riserate													
LargeFloods													
Low Flows													
Low pulse count				0.047*									
No of reversal	-0.289*												
Rise rate	0.307*												
Small flood fall rate	0.273*						-0.293*						
Small flood rise rate	0.264*												
Small Floods							-0.306*				0.361*		
Extreme LowFlows							0.295*					-0.331*	

3.4. Relation of IHA and EFCs to ecology on catchment and regional scale

Spearman correlations for the Pinios dataset did not reveal significant results with the exception of those between the Small Flood Peak or Extreme Low Duration and the macroinvertebrate metrics BMWP and ASPT ($r = -0.529$, $p < 0.005$ and $r = -0.382$, $p < 0.05$) (Fig. 3.5).

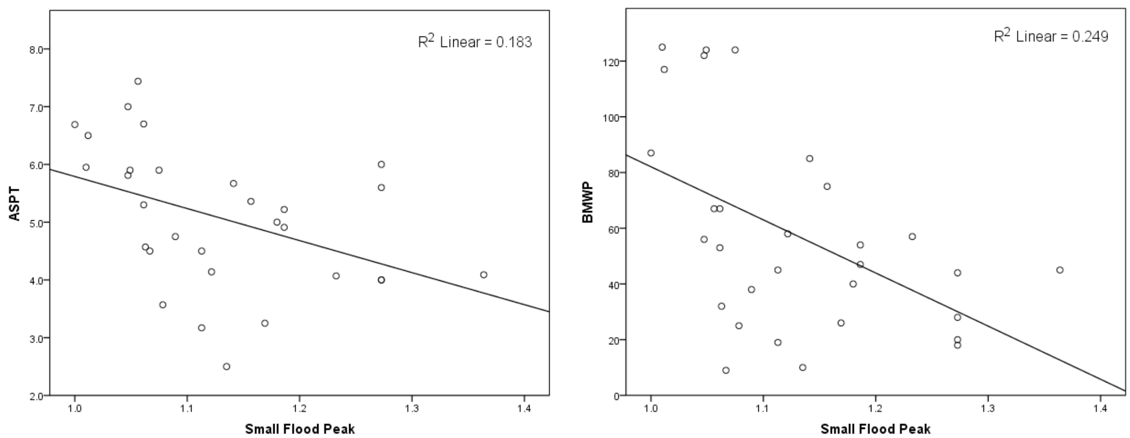


Figure 3.5. Scatter plots between BMWP, ASPT and Small Flood Peak indicator in Pinios catchment.

A Canonical Correspondence Analysis (CCA) was also run between the macroinvertebrate community data and the dataset of the hydrologic indicators (IHA and EFCs) in order to explore possible relationships between taxa and hydrologic parameters. According to the results of CCA the first axis explains the 13.9 % of data variance and 22 % of taxa – hydrology relations variance (Table 3.3). Specifically, the results indicated that Small Flood Peak was highly correlated with Axis 1 while Low Pulse threshold presented the highest correlation with Axis 2 (Table 3.4). The ordination diagram in Figure 3.6 depicts the relationships between taxa and hydrologic variables. A negative response for the majority of macroinvertebrate families to Small Flood Peak can be shown that agrees with the significant negative correlation between Small Flood Peak and BMWP and ASPT. Additionally, what can be shown from the ordination diagram is that almost all the EPT taxa (such as Ephemeridae, Heptageniidae, Perlidae, Perlodidae, Rhyacophilidae and others) are positioned at the left part of the ordination diagram where they appear to relate positively to increased ratios of duration of high flow pulses and negatively with increased ratios of peaks of floods. On the right part of the diagram several taxa are scattered along the 2nd axis showing positive correlations with ratios of flood peaks, base flow index, October and November low flow.

Table 3.3. Variance of the community data attributed to each axis

Axes	1	2	3	4	Total inertia
Eigenvalues	0.555	0.340	0.240	0.197	3.993
taxa hydrology correlations	0.982	0.915	0.887	0.939	
Cumulative percentage variance					
of taxa data	13.9	22.4	28.4	33.4	
of taxa-hydrology relation	22.0	35.4	44.9	52.7	
Sum of all eigenvalues					3.993
Sum of all canonical eigenvalues					2.529

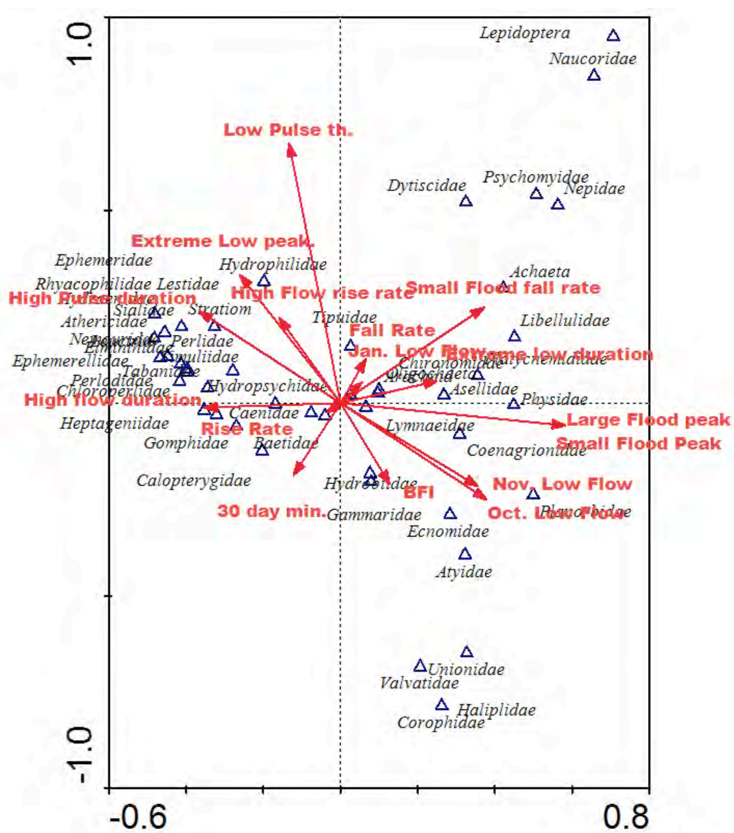


Figure 3.6. Canonical Correspondence Analysis (CCA) biplot showing relations between macroinvertebrate taxa and IHA variables.

Regarding the results obtained from the analysis for the German dataset, several significant correlations were identified between certain macroinvertebrate metrics and the scores of the first principal component derived by the Principal Component Analysis. Specifically, the most notable relationships included positive correlations between PC1 and number of hyporhithral taxa, Coleoptera and EPTCBO, and negative correlations between PC1 and hololimnic taxa, share of alien species and ratio r-selected/K-selected taxa (Figure 3.7).

Table 3.4. Bi-plot scores for constraining variables.

	AXIS 1	AXIS 2	30 day min.	BFI	High pulse duration	Low pulse threshold	Rise Rate	Fall Rate	Jan. Low Flow	Oct. Low Flow	Nov. Low Flow	Extreme Low Flow peak	Extreme Low Flow duration	High Flow duration	High Flow rise rate	Small Flood Peak	Small Flood fall rate
AXIS 1	1																
AXIS 2	-0.0217	1															
30 day min.	-0.1197	-0.1713	1														
BFI	0.1247	-0.1925	-0.2561	1													
High pulse duration	-0.3605	0.2167	-0.18	0.1547	1												
Low pulse threshold	-0.131	**0.6154**	-0.2307	0.0231	0.7286	1											
Rise Rate	-0.0357	-0.0222	0.0848	0.1385	-0.2124	-0.1405	1										
Fall Rate	0.0642	0.1042	-0.0464	-0.1699	0.0029	0.0357	-0.2821	1									
Jan. Low Flow	0.0551	0.0495	0.0571	-0.0081	-0.0699	-0.1461	0.1767	-0.08	1								
Oct. Low Flow	0.3715	-0.229	-0.0381	0.2067	0.3274	0.1625	-0.0727	-0.0972	-0.1732	1							
Nov. Low Flow	0.3489	-0.1973	0.0707	-0.0064	-0.4149	-0.2645	-0.2651	-0.2464	-0.2489	-0.018	1						
Extreme Low Flow peak	-0.2558	0.3033	0.389	-0.7411	-0.1294	0.1335	-0.0778	0.0777	0.0108	-0.2093	0.1519	1					
Extreme Low Flow duration	0.2458	0.0523	0.4286	-0.1102	0.0525	-0.1368	0.0991	-0.3698	0.2278	-0.0384	0.1389	-0.0268	1				
High Flow duration	-0.3481	-0.0115	-0.1133	-0.4262	-0.3409	-0.178	-0.1223	-0.0307	0.1106	-0.4873	0.051	0.38	-0.2963	1			
High Flow rise rate	-0.1562	0.201	0.4096	-0.0144	0.1608	0.1144	-0.2472	0.0836	-0.1414	-0.098	0.0853	0.034	0.2945	-0.2633	1		
Small Flood Peak	**0.572**	-0.0532	-0.0263	0.098	0.2762	0.2789	-0.2271	-0.0385	-0.2322	0.4745	0.2361	-0.1523	0.3566	-0.5457	0.0551	1	
Small Flood fall rate	0.3648	0.2263	0.0985	0.1754	0.2086	0.475	-0.1604	0.12	-0.1336	0.2078	-0.0224	-0.1753	0.0354	-0.3751	0.4032	0.4884	1

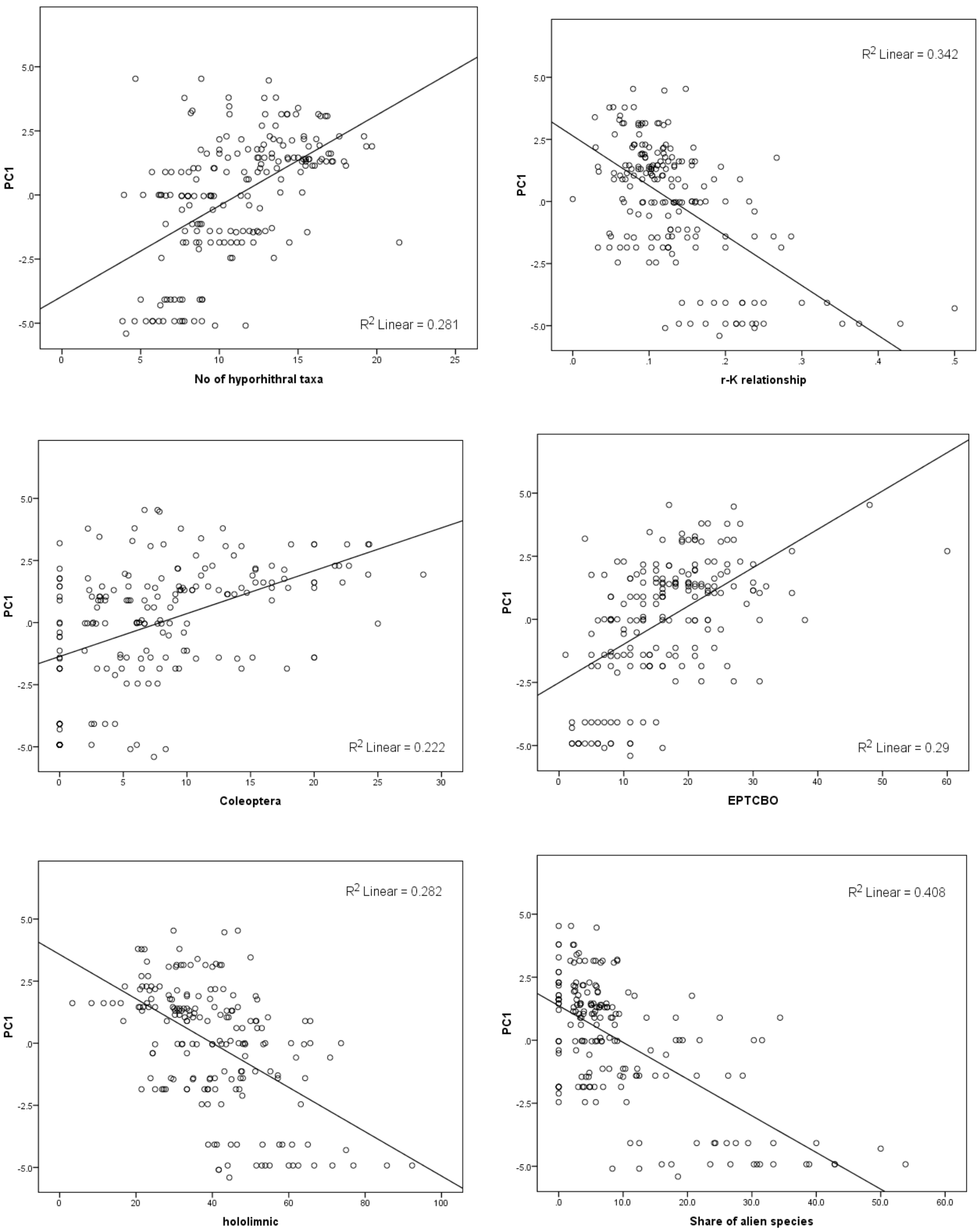


Figure 3.7. Scatter plots between PC1 scores and macroinvertebrate metrics for 191 German samples.

3.5. Pan-European mapping results

This section depicts the hydrologic alteration results at the European scale based on selected indicators. In the absence of strong connection between specific hydrologic indicators and BQEs, we present several hydrologic indicators, trying to explain the characteristics of the hydrologic stress across Europe without reference to any ecologic response. The maps consist of 104,334 FECs for which all hydrologic indicators have been calculated under both the anthropogenic (baseline) and near-natural scenario. Each pair of maps depicts the baseline indicator on the left and its alteration on the right. Alteration is calculated as the ratio *near-natural / anthropogenic (baseline)* or the difference *near-natural – anthropogenic* of the respective indicators. Alteration is presented by using the more informative metric for each of the indicators, either the ratio or the difference. In the first case (ratio), a number above unity shows that the denominator is smaller, thus the metric is higher under near-natural conditions and is reduced due to abstractions (anthropogenic scenario). Ratios below unity show the opposite, while ratios equal to unity imply no alteration between the two scenarios. In the case of the calculated difference, positive values show reduction of the indicator in the baseline and negative values the opposite, with zero values showing no alteration. It should be noted that the 'pressure' on water is not expressed consistently by above unity and positive numbers (or the opposite) but depends on the nature of the indicator analyzed. We have to keep in mind that this descriptive presentation of the European results, is influenced by the rather short time-series length (10-y), which is close to the minimum requirements of the hydrologic (IHA) method. Technical constraints due to the large size of the dataset in combination with a limited timeframe did not allow us to use a longer period of discharge data for the 104,334 FECs of the analysis at this stage. From the following mapping results, indicators related to small and large floods should be interpreted with more caution than the others.

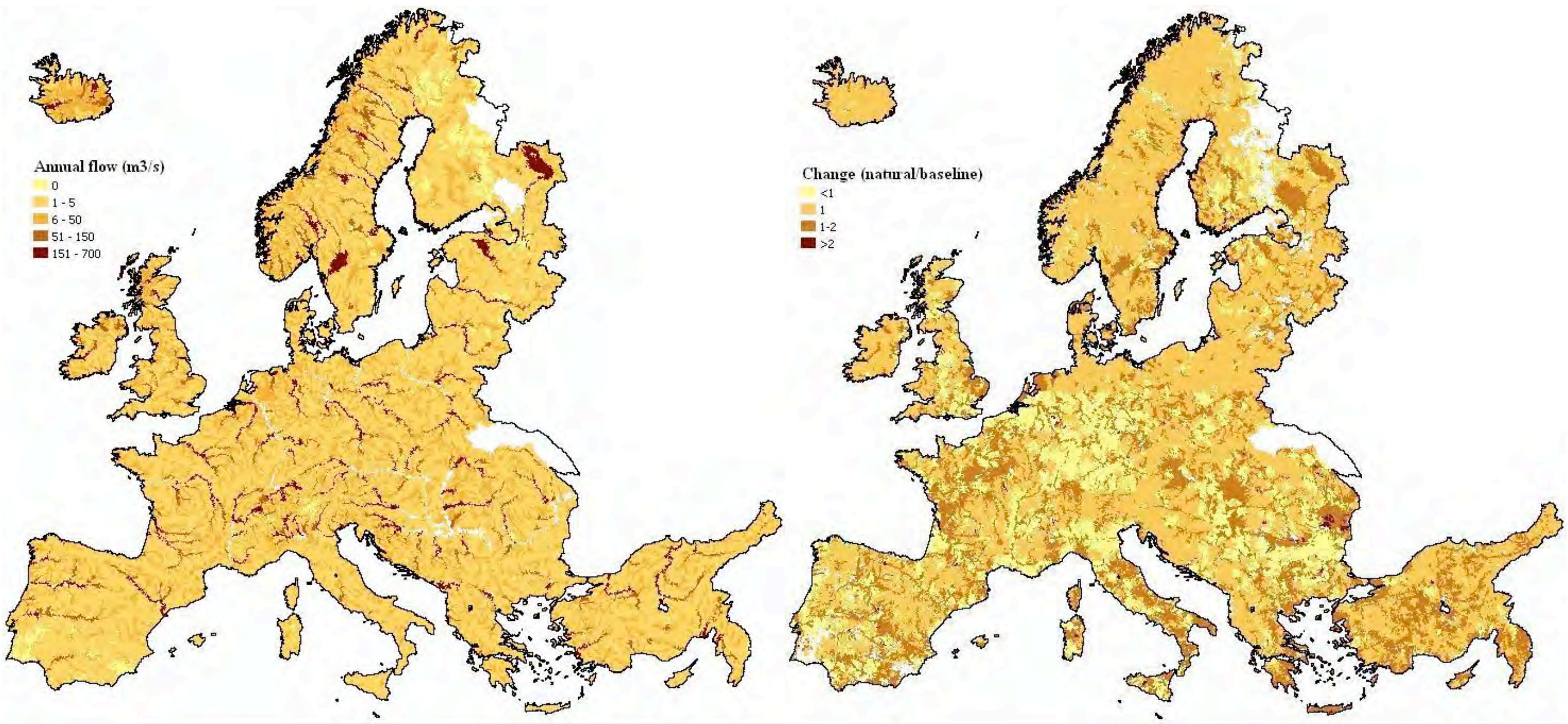


Figure 3.8. Left: Median annual flow (m^3/s) of the 10-y period 2001-2010 under the anthropogenic (baseline) scenario (water abstractions). Right: Alteration of the median annual flow from the natural conditions, expressed as median flow (natural) / median flow (baseline). Ratios below unity indicate increase in flow due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. Reduction is depicted on the right map with deeper colours and is observed in parts of the Mediterranean countries. Increase in flows (ratio <1) is found in Central Europe, with very large parts all across the continent and especially in the North remaining unaltered with respect to annual river flows.

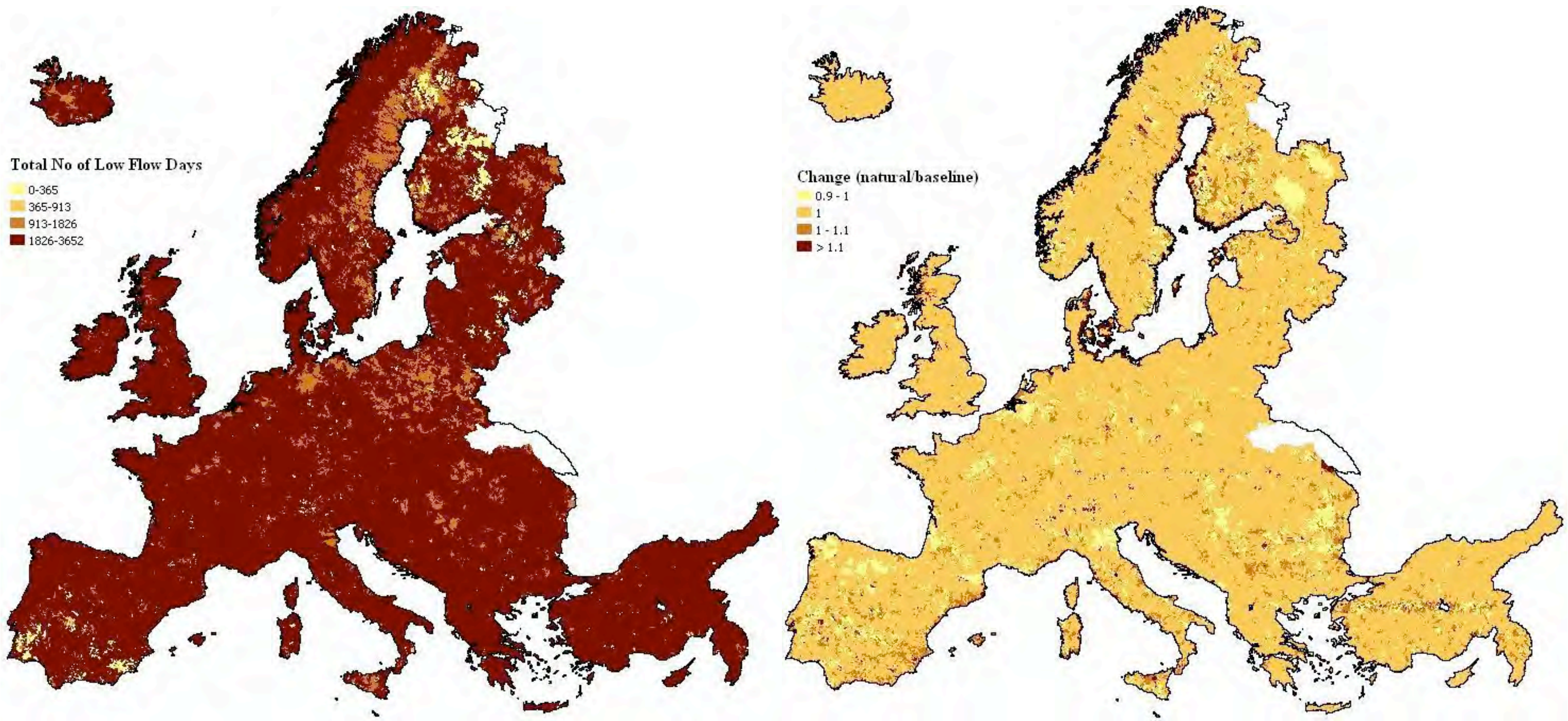


Figure 3.9. Left: Total number of low flow days within the 10-y period 2001-2010 (3652 days) under the baseline scenario (water abstractions). Right: Alteration of the total number of low flow days from the natural conditions, expressed as Low flow days (natural) / Low flow days (baseline). Ratios below unity indicate increase in low flow days due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. The total number of low flow days within the 10-y simulated period remains unaltered for the majority of the European area, representing more than half of this period's length.

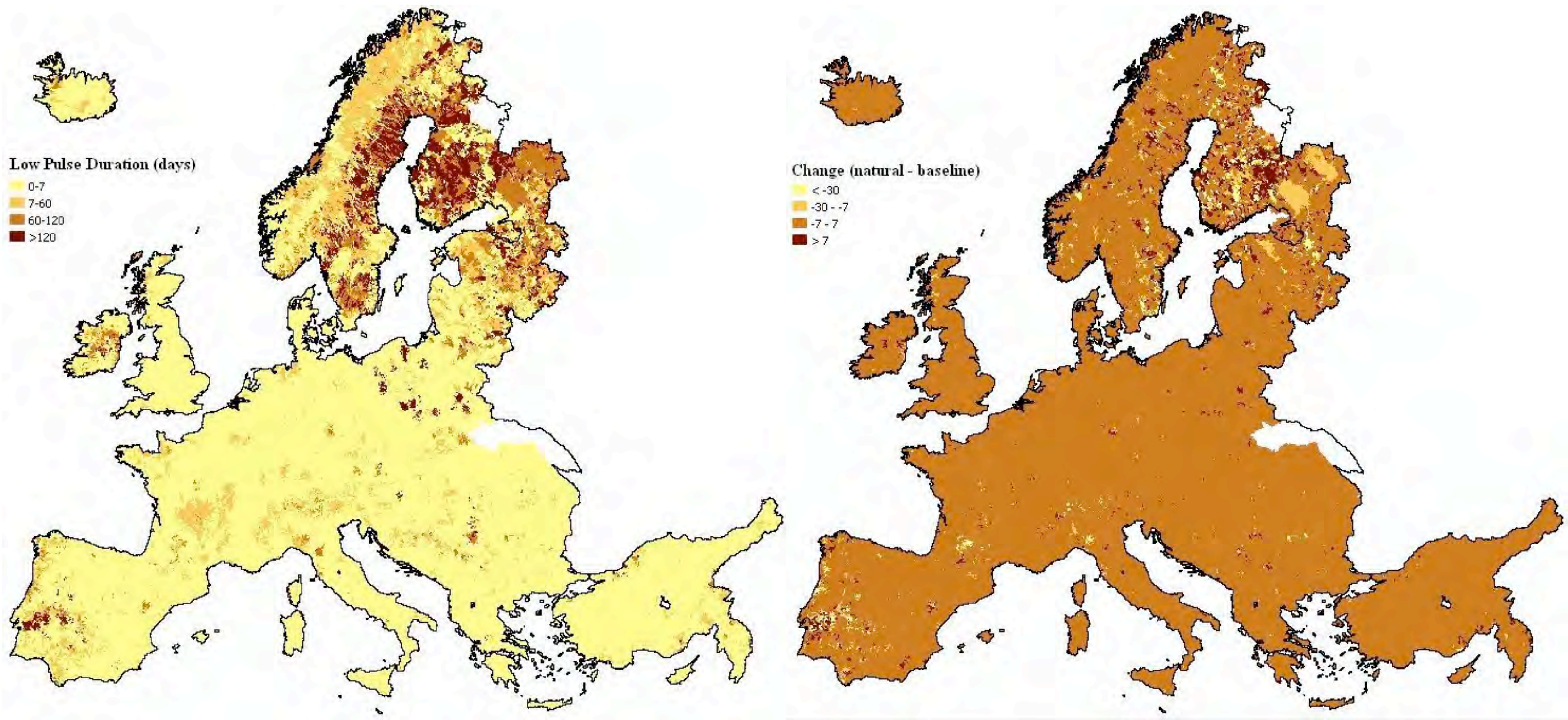


Figure 3.10. Left: Median duration of low flow events within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration of the median duration of low flow events from the natural conditions, expressed as Low flow duration (natural) – Low flow duration (baseline). Differences below zero indicate increase in duration (in days) due to abstractions, values around zero show no alteration and values above zero show reduction from the natural conditions. The duration of low flow events is high in Northern and North-eastern Europe but very small in the rest of Europe. The average duration of low flow events is altered less than a week almost everywhere across Europe due to human activities (abstractions).

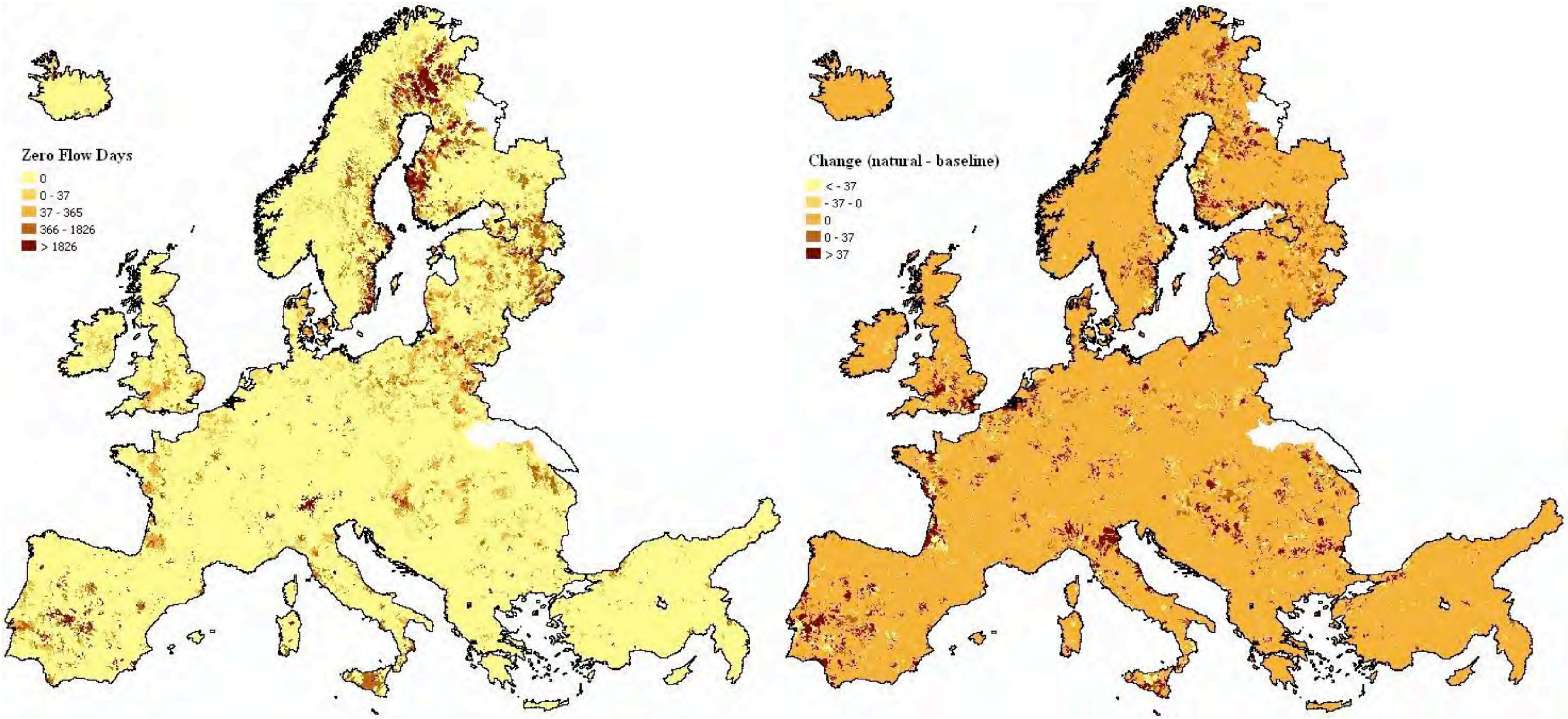


Figure 3.11. Left: Total number of zero flow days within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration in the total number of zero flow days from the natural conditions, expressed as Zero flow days (natural) – Zero flow days (baseline). Differences below zero indicate increase in zero flow days due to abstractions, values around zero show no alteration and values above zero show reduction from the natural conditions. The deep red on the right map shows that for scattered areas all across Europe the anthropogenic scenario decreased zero flow days by more than 1% (37 days) of the total period of analysis (3652 days). Abstractions in the anthropogenic scenario may have contributed with significant return flows (irrigation runoff returns or point sources releasing used water to streams) to the rivers of specific FECs with rather poor hydrology in the near-natural scenario. The vast majority of Europe doesn't meet any change with respect to the zero flow days.

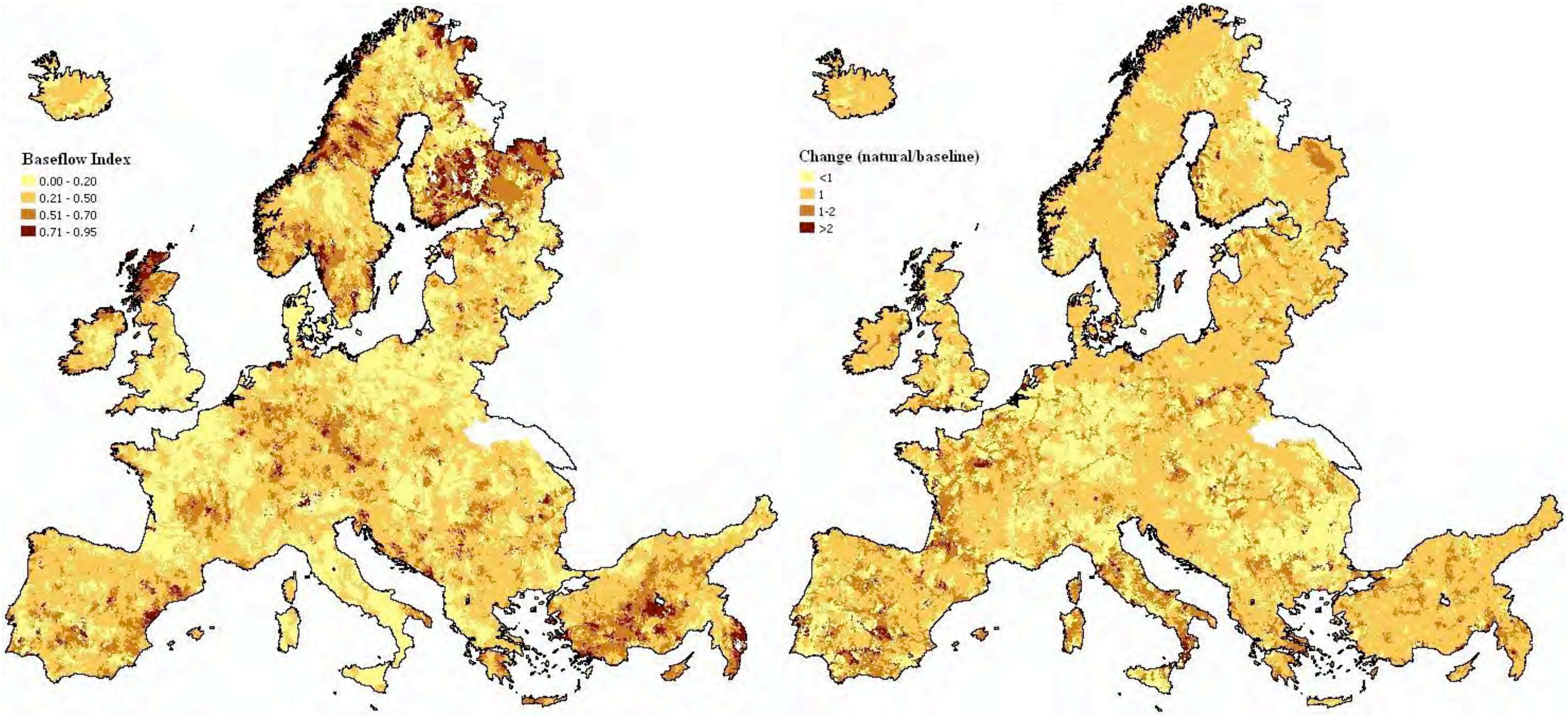


Figure 3.12. Left: Baseflow index (BFI) within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration of the BFI from the natural conditions, expressed as BFI (natural) / BFI (baseline). Ratios below unity indicate increase in BFI due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. Baseflow index is defined as: 7-day minimum flow / mean flow for year. The higher the BFI is, the greater the groundwater contribution to streamflow. The highest BFI values are observed in parts of Central Europe and Scandinavia (left). The BFI has not been altered due to abstractions (right) for large parts of Europe but has been decreased only in parts of the Mediterranean countries (ratio >1) due to abstractions. This is possibly connected to groundwater abstractions for irrigation which reduce the groundwater levels and the capability of aquifers to contribute with water to adjacent streams and rivers.

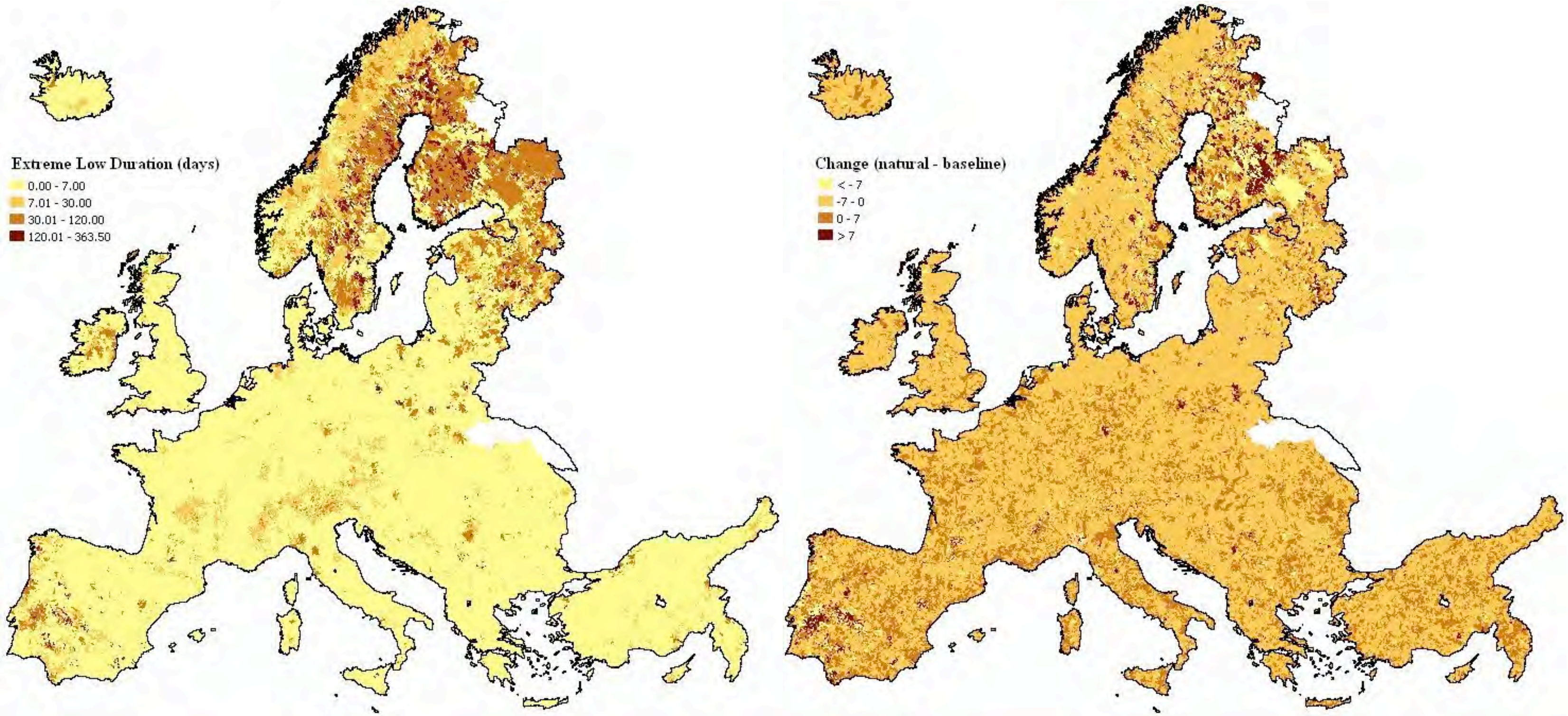
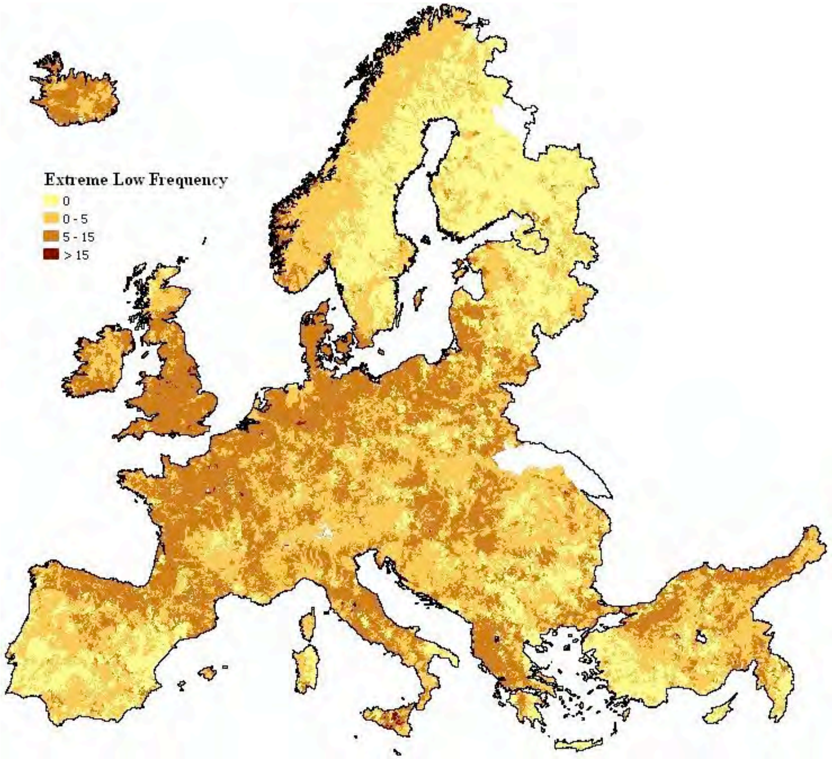


Figure 3.13. Left: Median duration of extreme low flow events within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration in the median duration of extreme low flow events from the natural conditions, expressed as extreme low flow duration (natural) – extreme low flow duration (baseline). Differences below zero indicate increase in extreme low flow duration due to abstractions, values around zero show no alteration and values above zero show reduction from the natural conditions. Long extreme low events are observed in Scandinavia where precipitation is evenly distributed in time enhancing in general the prolongation of low flow conditions in rivers (left). In the greatest part of Europe an increase up to a week is observed in the duration of extreme low flow events (right), which is the typical maximum duration in the baseline (left). For scattered areas all across Europe there is also decrease in the extreme low flow duration due to abstractions. This is attributed to local factors, which cannot be identified at this scale of analysis.



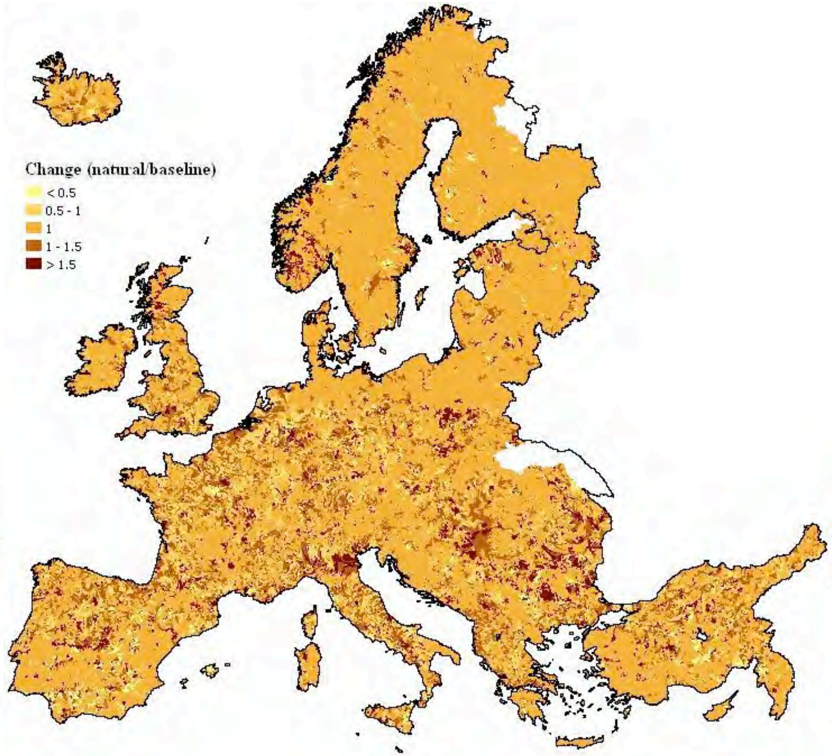


Figure 3.14. Left: Median frequency of extreme low flow events per year within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration in the median frequency of extreme low flow events from the natural conditions, expressed as extreme low flow frequency (natural) / extreme low flow frequency (baseline). Ratios below zero indicate increase in the frequency of extreme low flow events due to abstractions, values around unity show no alteration and values above unity show reduction from the natural conditions. Where alterations are observed, these mostly correspond to positive ratios (right), thus, to decreased frequencies of extreme low flow events from the natural conditions.

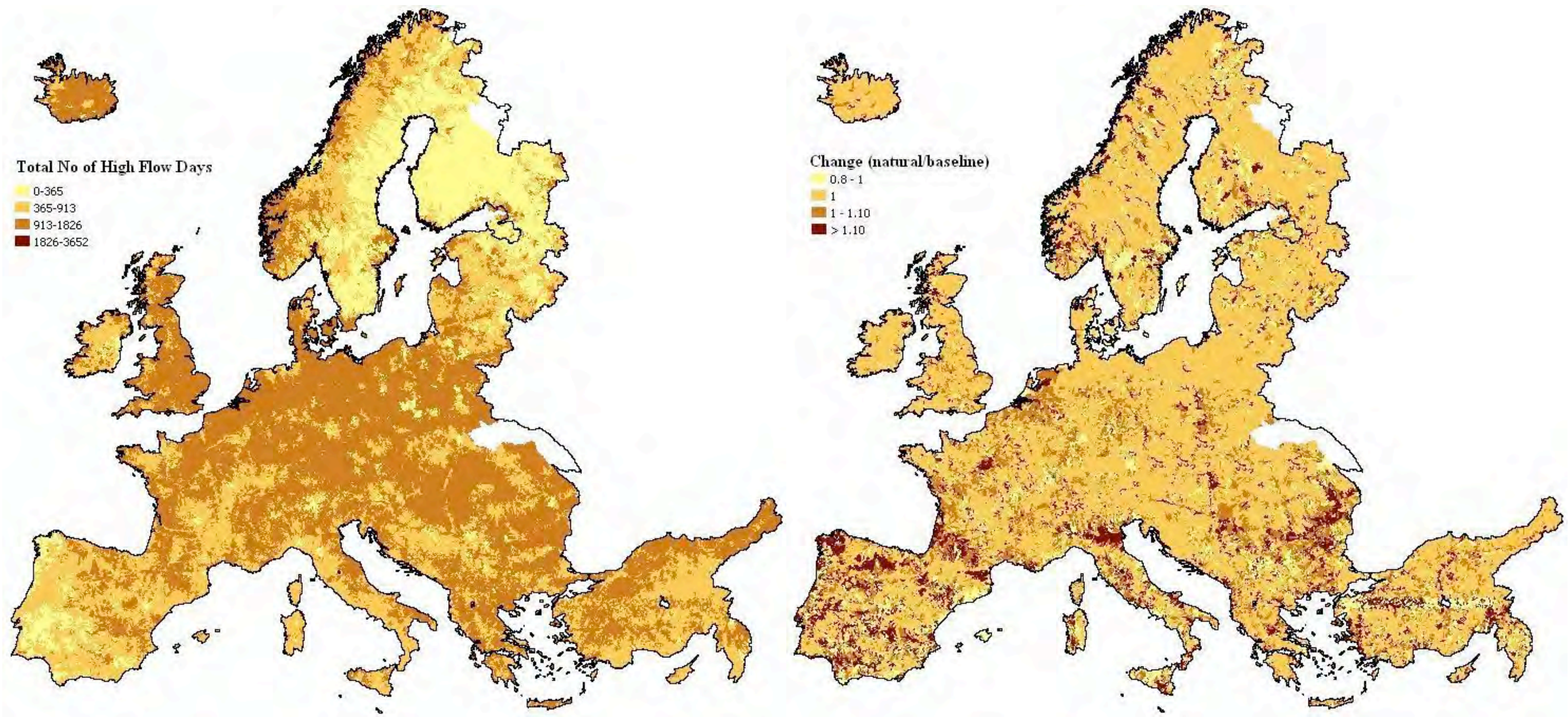


Figure 3.15. Left: Total number of high flow days within the 10-y period 2001-2010 under the baseline scenario (water abstractions)
Right: Alteration of the total number of high flow days from the natural conditions, expressed as High flow days (natural) / High flow days (baseline). Ratios below unity indicate increase in high flow days due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. The total number of high flow days within the 10-y simulated period represents a 10%-50% (365-1826 days) of the total period's length (3652 days) and remains unaltered for the largest part of the European area. High flow days in Scandinavia are < 10% (365 days) of the total time in the baseline (left). Moreover, abstractions have resulted in a reduction of high flow days in areas scattered across Europe but mostly in the Mediterranean and Balkan countries. This is attributed mainly to agricultural abstractions, which reduce water availability for discharge.

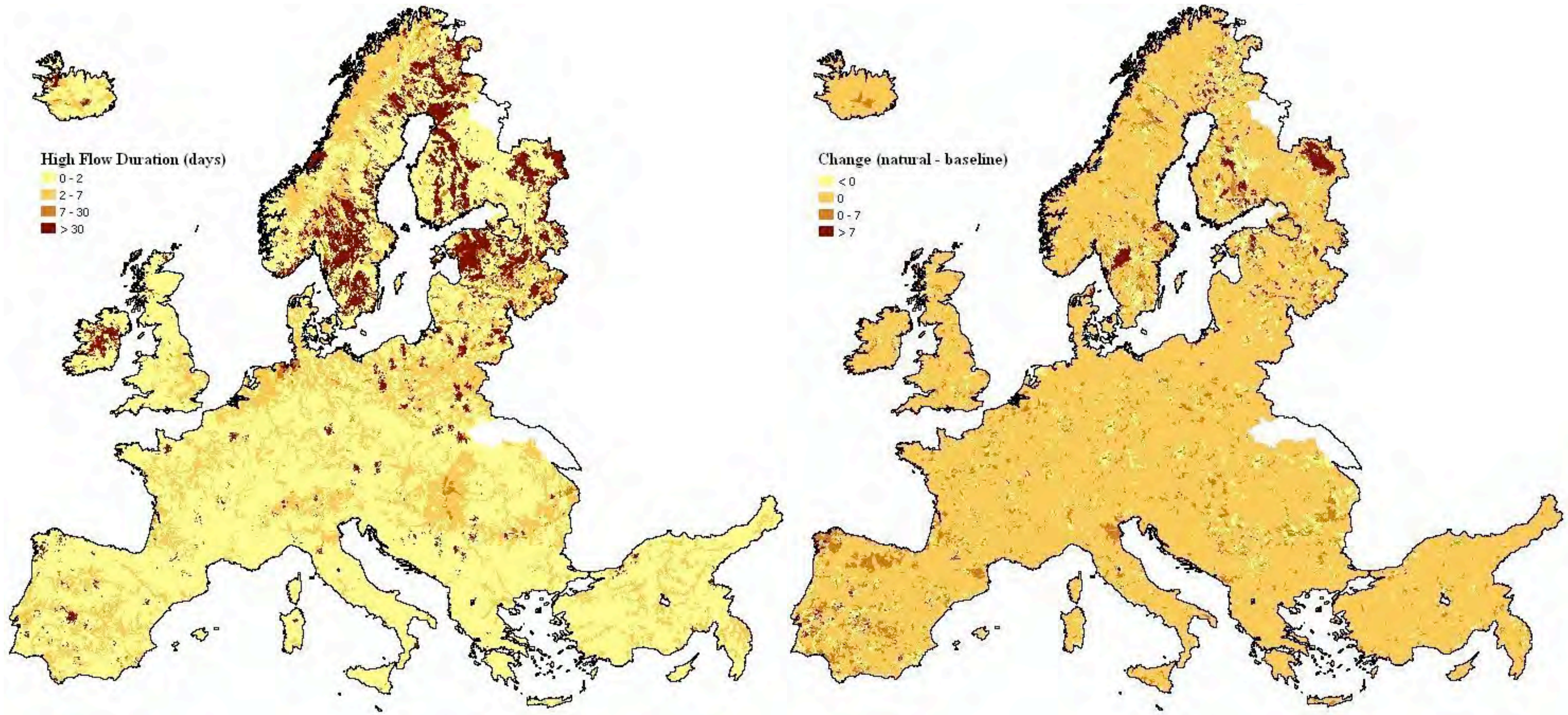


Figure 3.16. Left: Median duration of high flow events within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration in the median duration of high flow events from the natural conditions, expressed as high flow duration (natural) – high flow duration (baseline). Differences below zero indicate increase in high flow duration due to abstractions, values around zero show no alteration and values above zero show reduction from the natural conditions. In the greatest part of Europe high flow events typically last up to a week (left) and no alteration is observed from the natural conditions (right). The most prolonged events are observed in parts of North-eastern Europe supporting the previous finding that a certain hydrologic regime of rivers there does not change very frequently. A decrease up to a week (0-7 days) and more than a week (> 7 days) can only be observed locally (right).

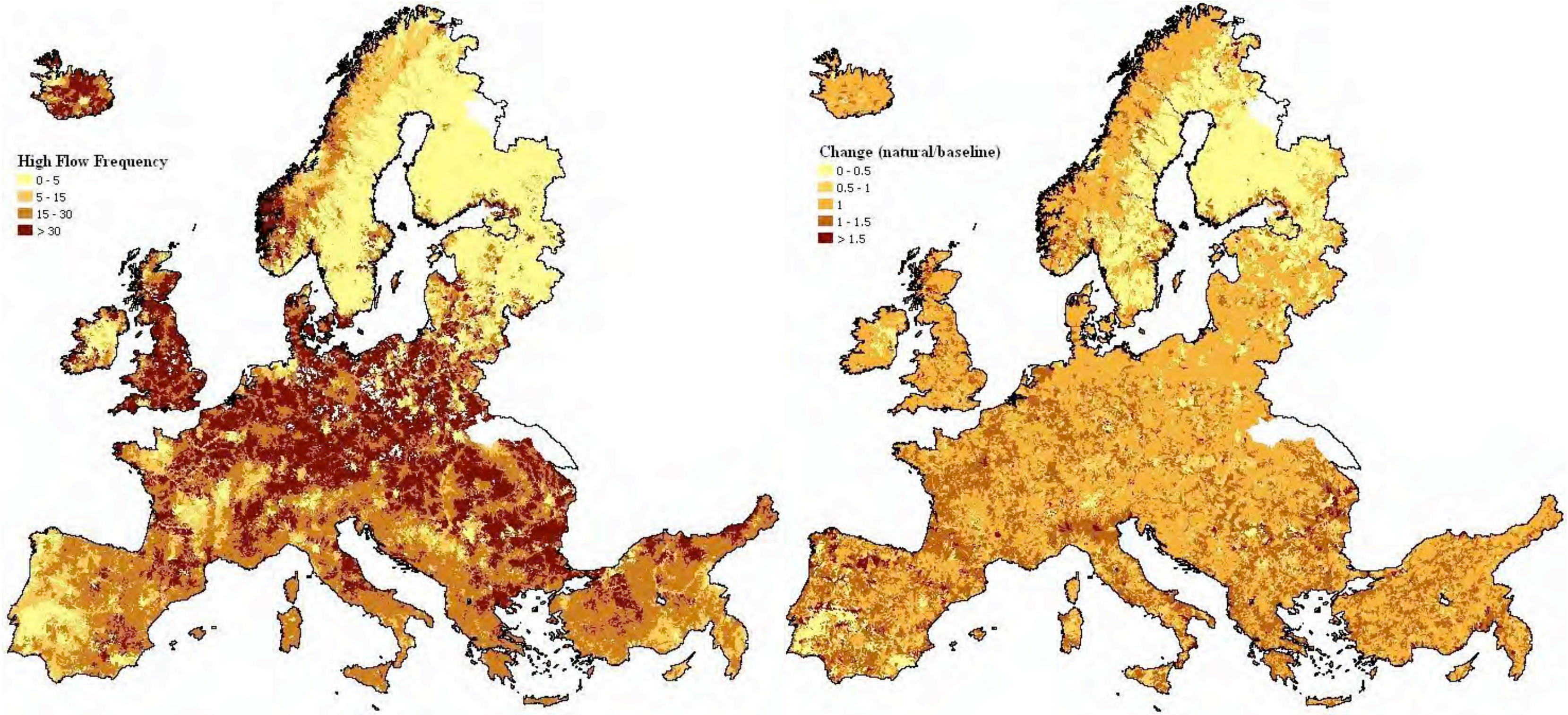


Figure 3.17. Left: Median frequency of high flow events per year within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: alteration in the median frequency of high flow events from the natural conditions, expressed as high flow frequency (natural) / high flow frequency (baseline). Ratios below unity indicate increase in the frequency of high flow events due to abstractions, values around unity show no alteration and values above unity show reduction from the natural conditions. Reduced frequency of high flows is observed in scattered areas of the Mediterranean countries (ratios > 1) and increase (ratios < 1) in the Scandinavian region, where however, the mangitude of the baseline frequencies was already low (left) and this has severe mathematical influence on the resulted ratios (right).

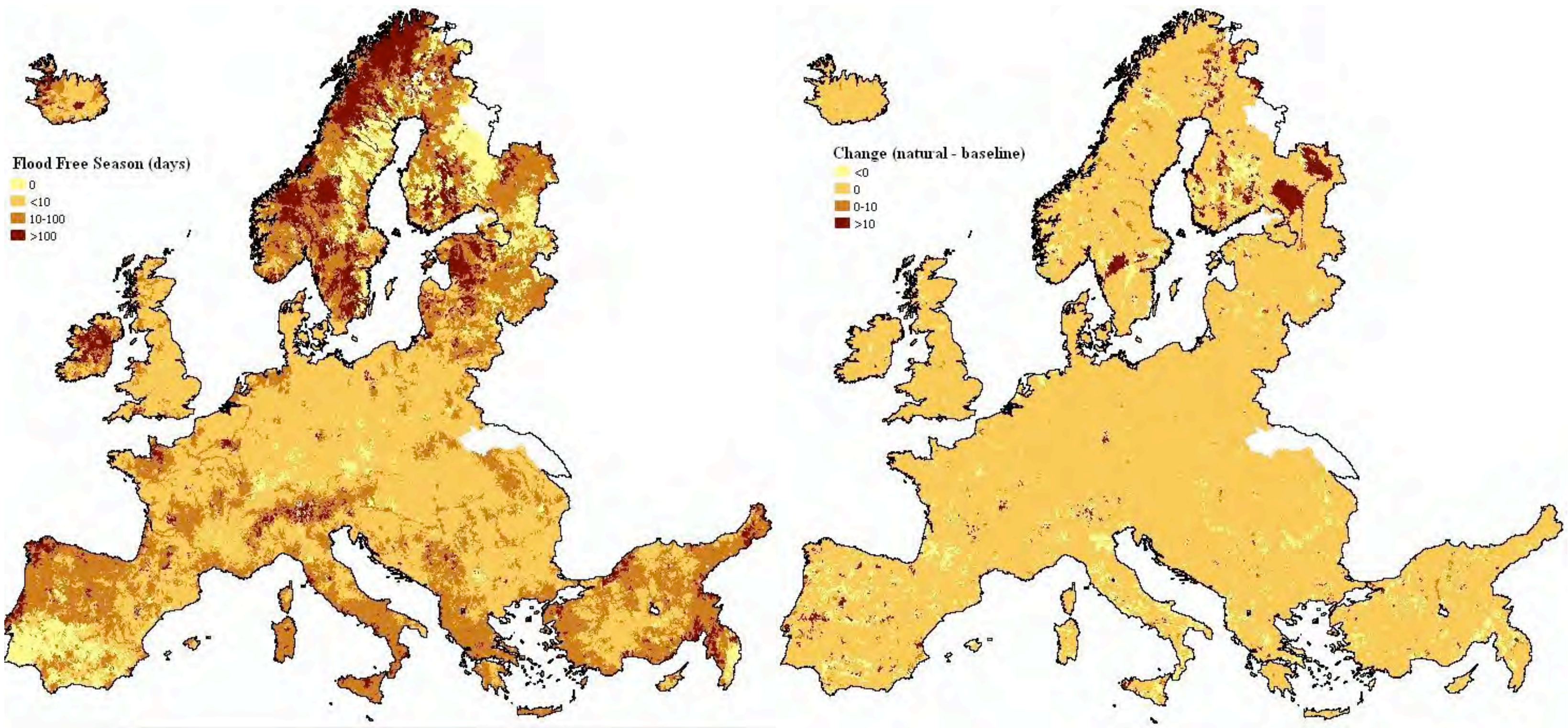


Figure 3.18. Left: Length of flood free season (FFS) within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration in the length of flood free season from the natural conditions, expressed as Flood Free Season (natural) – Flood Free Season (baseline). Differences below zero indicate increase in FFS (days) due to abstractions, values around zero show no alteration and values above zero show reduction from the natural conditions. The FFS is defined as: the length in days of the longest period common to all water years where flows are at or below the high pulse threshold in every year. The indicator is connected to low flow days and low flow duration in particular. Thus, considerable FFS occur in the South region but even longer in the North and North-eastern Europe (left). Alteration is not significant in this indicator because of abstractions (right).

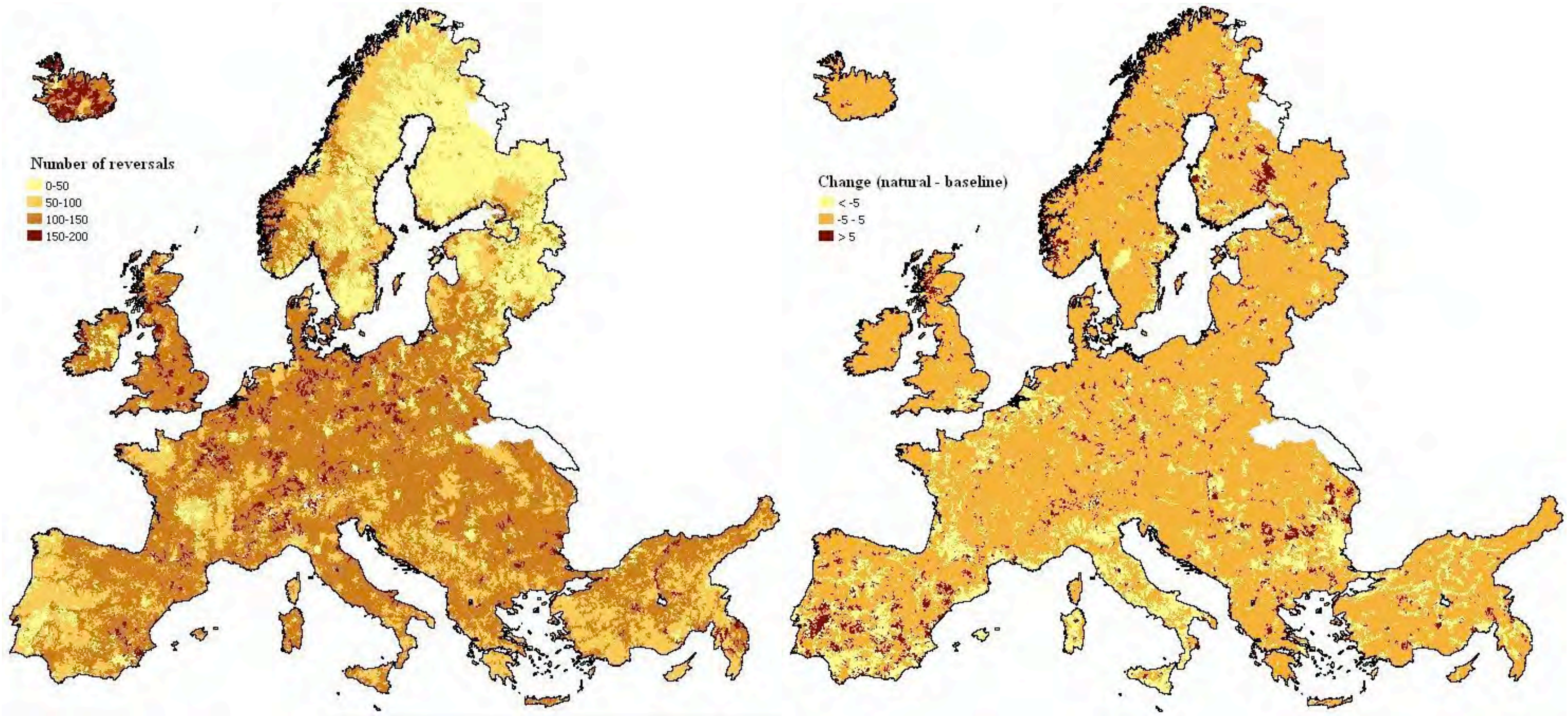


Figure 3.19. Left: Median number of reversals (NoR) per year within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration in the median number of reversals from the natural conditions, expressed as Number of Reversals (natural) – Number of Reversals (baseline). Differences below zero indicate increase in NoR due to abstractions, values around zero show no alteration and values above zero show reduction from the natural conditions. The NoR is defined as: the number of times flow switches from one type of period to another.The small numbers (left) of the Northern part of Europe agree with the more stable flow conditions shown from other indicators previously. Alteration (times) is mostly insignificant in this indicator because of abstractions (right) except parts within some Mediterranean and Balkan countries where reversals are enhanced due to abstractions (< -5).

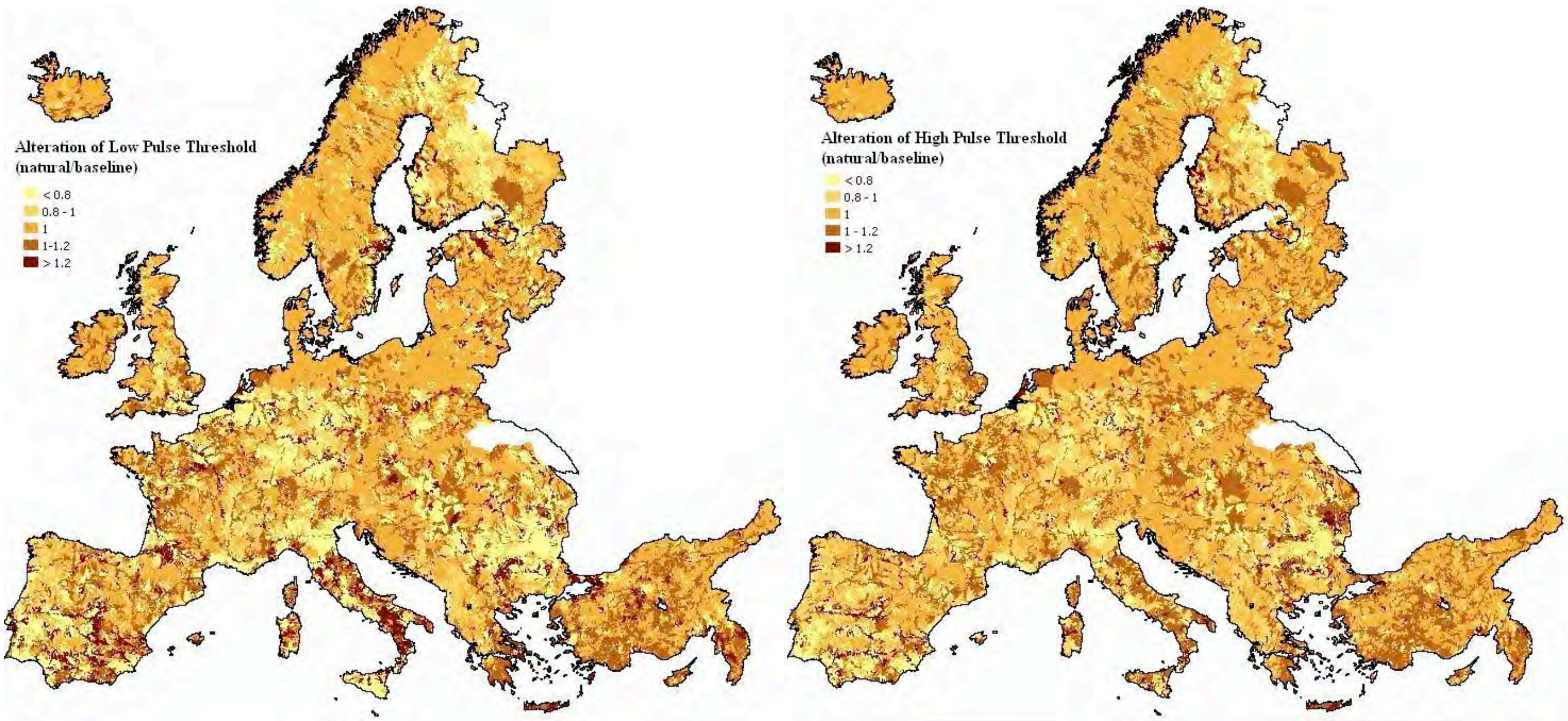


Figure 3.20. Left: Alteration of low pulse threshold (LPT) from the natural conditions, expressed as LPT (natural) / LPT (baseline). Right: Alteration of high pulse threshold (HPT) from the natural conditions, expressed as HPT (natural) / HPT (baseline). Ratios below unity indicate increase in thresholds due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. The LFT within the 10-y simulated period remains mostly unaltered or increases slightly in Central and Northern Europe (<1 left map) and decreases mostly in parts of Southern Europe (>1 left map) under abstractions. The alterations of the HFT within the 10-y simulated period occur locally all across Europe with more concentrated areas in the South. It seems that local rules govern most its alteration compared to LFT. .

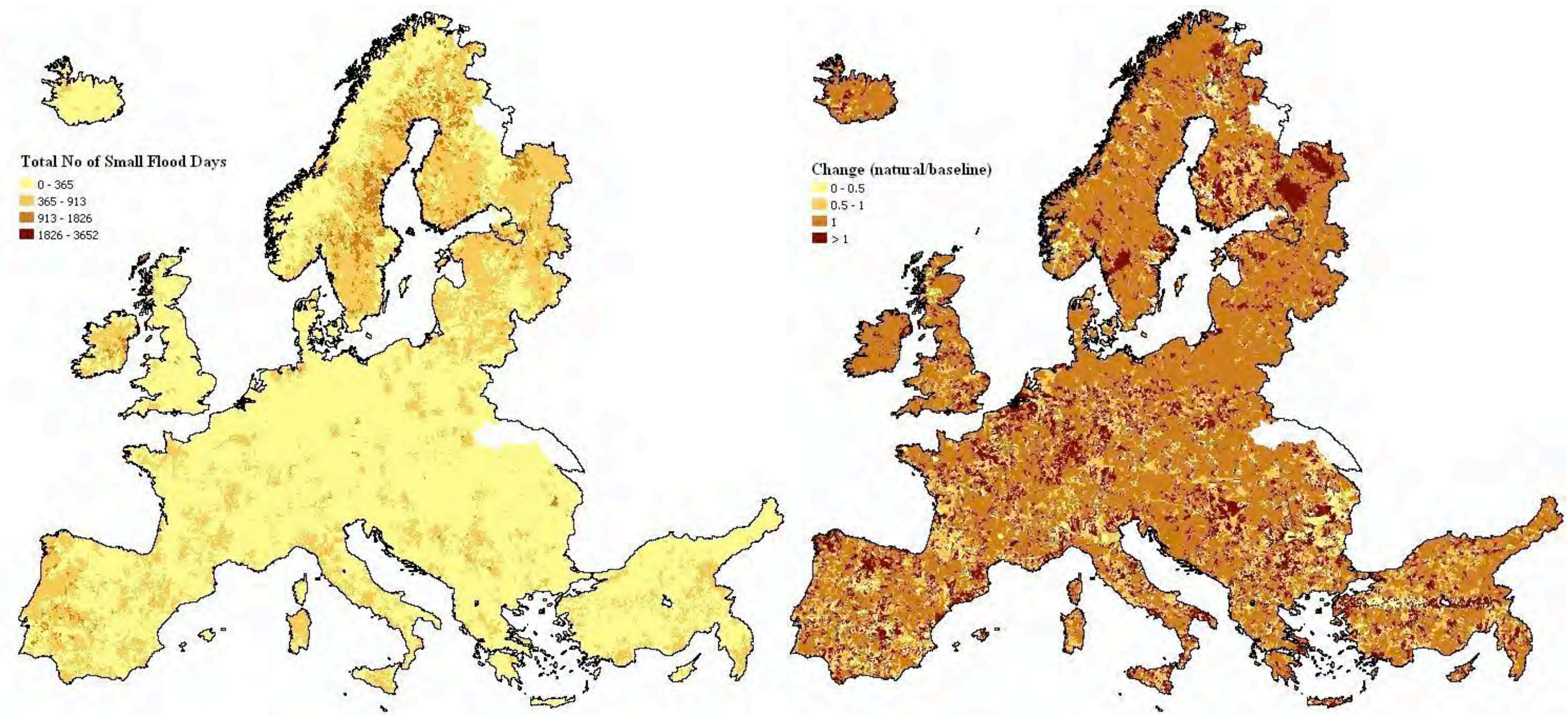


Figure 3.21. Left: Total number of small flood days within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration of the total number of small flood days from the natural conditions, expressed as Small Flood days (natural) / Small Flood days (baseline). Ratios below unity indicate increase in small flood days due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. The alteration in the number of small flood days is observed only on a local basis across Europe due to abstractions and would need exploration of the local conditions to be explained. However, due to the small number of available years in our time-series flood indicators are not quite representative and may lead to misinterpretations.

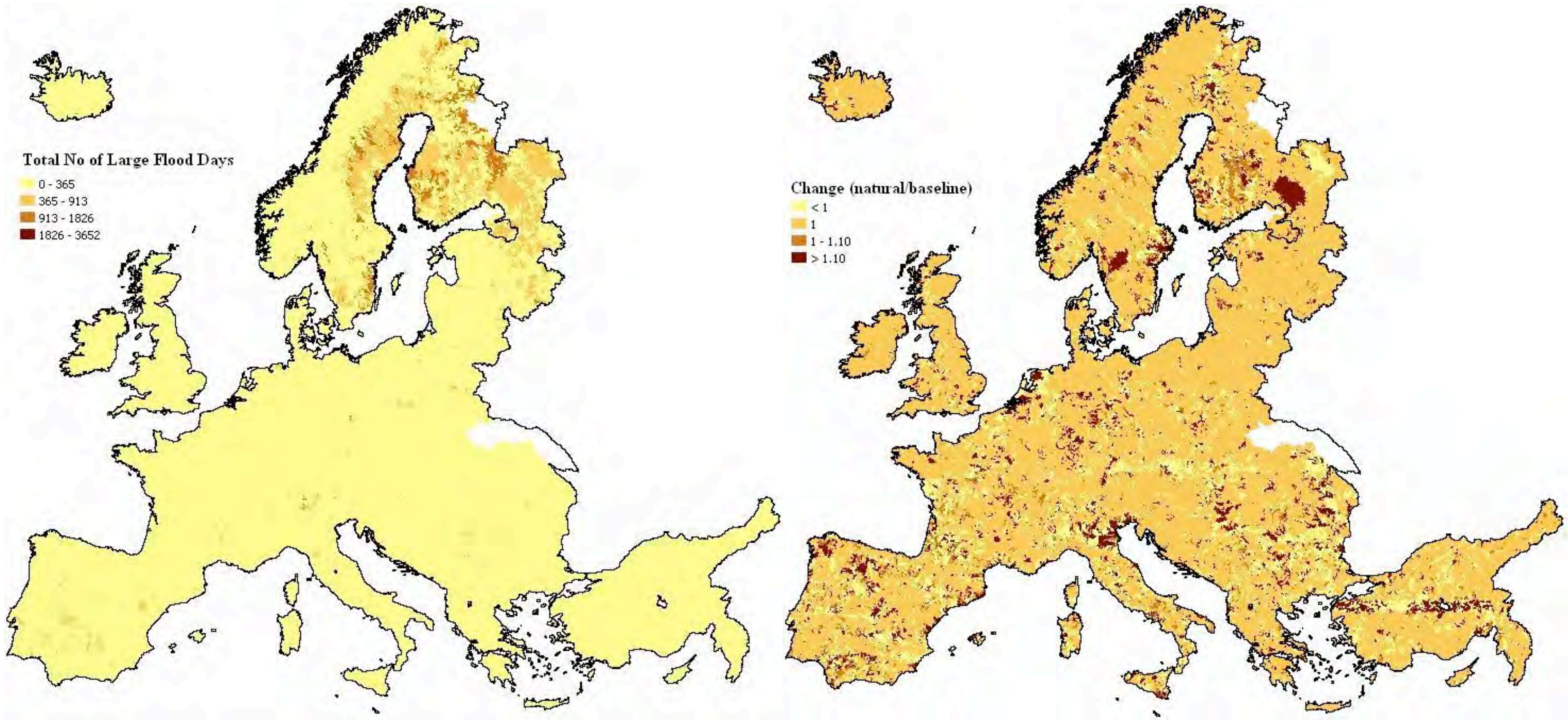


Figure 3.22. Left: Total number of large flood days within the 10-y period 2001-2010 under the baseline scenario (water abstractions). Right: Alteration of the total number of large flood days from the natural conditions, expressed as Large Flood days (natural) / Large Flood days (baseline). Ratios below unity indicate increase in large flood days due to abstractions, values equal to unity show no alteration and values above unity show reduction from the natural conditions. The alteration in the number of large flood days is observed only on a local basis across Europe due to abstractions and would need exploration of the local conditions to be explained. However, due to the small number of available years in our time-series flood indicators are not quite representative and may lead to misinterpretations.

4. DISCUSSION and KEY MESSAGES

4.1. Relations of hydrology and ecology

This study clearly showed that there are no significant relationships between the hydrologic alteration (stress) and the EQR values of two BQEs (macroinvertebrates and phytobenthos), by using either the pan-European EQR dataset or separate datasets per Broad River Type (BRT). This is attributed mostly to the poor quantity and limited representativeness of the BQE metrics available in this work. However, by focusing on two case studies (Pinios catchment and Germany) it was concluded that by investigating the responses of specific biological metrics to hydrologic alteration metrics, significant relationships can be found. Particularly, in the German case study it was shown that when a principal component analysis was applied to generate a "proxy" parameter (principal component 1) that accounts for most of the variance in the original variables, there were significant correlations between the principal component and several macroinvertebrate metrics.

First, the use of selected metrics (e.g. number of specific taxa) as biological responses to hydrologic alteration variables is more likely to result to significant relationships than when using solely EQR as a response. The explanation for this probably lies in the fact that macroinvertebrate EQRs are based upon individual metrics that are more sensitive to organic and nutrient pollution, or oxygen depletion. On the other hand, individual metrics, such as specific taxa richness, can be affected by changes in hydrology such as dilution effects of water compounds caused by changes to water volume. There are several studies that have shown various taxa-specific responses to flow alterations and merely agree with some of the findings presented in this report. For example Dewson et al. (2007) showed that sensitive EPT disappeared from streams with depleted flows, whereas overall macroinvertebrate abundance remained unchanged. Moreover there was indication that EPT richness declined with increased severity of inflated baseflows. Carlisle et al. (2014) also showed that reduced richness of sensitive aquatic insect taxa (Ephemeroptera, Plecoptera and Trichoptera) as well as taxonomic completeness was strongly related to increased severity of baseflow depletion.

From the Pinios case study, EPT taxa showed to relate negatively with highly altered indicators of flood peaks, baseflow index and October, November low flows whereas they appeared to prefer conditions where the duration of high flow events is smaller than under the near-natural conditions. These results could be interpreted as a preference of these EPT taxa for hydrologic conditions with lower peaks of flood events, lower duration of high flow pulses and lower baseflows. In contrast, Carlisle et al. (2014) have shown that higher total macroinvertebrate abundance related to increasingly inflated high flows, although this response was highly variable. Additionally, from the findings from Buchanan et al. (2013) suggested that macroinvertebrate responses vary in strength depending on the flow metric. Buchanan et al. (2013) found that taxa specific macroinvertebrate metrics, such as % scrapers, demonstrated a strong decline with specific hydrologic alteration metrics like the duration, frequency and change of flow metrics. The increased sensitivity of % scrapers was explained due to the group's reliance on periphyton food, which is washed downstream by high flows. Weak responses of EPT could be explained due to replacement of flow-specialist taxa by more tolerant, flow-generalist taxa. For example, free-living EPT taxa are more susceptible to dislodgement due to increased flows and therefore are expected to get replaced by taxa that are able to construct firm refugia cemented to bottom substrate. Undoubtedly, the structure of freshwater macroinvertebrate communities is critically influenced by high flow events, even of moderate magnitude. As shown by Theodoropoulos et al. (2017), the abundance of benthic macroinvertebrates, diversity, taxonomic richness and EPT richness significantly decreased, while the composition of functional feeding groups was altered after high flow events. These results suggest that macroinvertebrate responses to hydrologic alterations are indirectly "controlled" by changes in habitat conditions due to flow alterations. For example, a decrease in base flows would affect habitat diversity, connectivity and channel morphology that in turn would influence taxa with slow colonization rates and low tolerance to low oxygen concentration (Graeber et al., 2013; Scholl et al., 2016). These conditions would favor fast growing gatherers (such as Chrinomidae).

The conclusion that can be drawn from the results of the two case studies (German monitoring data and Pinios) is that the macroinvertebrate responses to long-term flow alterations are most likely taxa-specific that reflect responses of taxa-related traits such as life history characteristics, mode of feeding, sensitivity to impaired environments and mobility (Carlisle et al., 2013; Scholl et al., 2016). Therefore, it is not always possible to infer that certain relationships between macroinvertebrate related "broad" metrics/indicators (e.g EQR) and several hydrologic alteration metrics exist. On the contrary, the results presented here seem to agree with findings from other published studies concluding that in order to assess the impact of flow alteration on aquatic communities, habitat changes should also be considered (Buchanan et al., 2013; Leigh, 2013; Nichols et al., 2016; Holzapfel et al., 2017) and that it would be optimal to focus on responses of specific metrics that can best reflect the degree of the hydrological alteration (Konrad et al., 2008; Buchanan et al., 2013; Nichols et al., 2016).

Another interesting finding is that the use of data reduction methods, such as principal component analysis, revealed stronger relationships between the principal components and the ecological responses than the relationship between certain hydrologic variables and ecological indicators. Interestingly, Visser et al. (2017) showed very recently that in few occasions the use of principal components as explanatory variables in multistep regression models may improve the models and increase the predictive power. However, there are concerns regarding this approach, since it is very likely that principal components may not capture all the ecologically relevant variables (Monk et al., 2007; Visser et al., 2017).

4.2. Hydrologic alteration among Broad River Types

Hydrologic alteration in Europe's rivers was investigated through the analysis of two 10-y time-series of daily flows for the anthropogenic (baseline) scenario and a near-natural scenario where water abstractions were removed. We calculated the ratio and difference of a large number of hydrologic indicators for each FEC to express the degree of hydrologic alteration from the natural conditions. The average ratios of natural versus anthropogenic hydrologic indicators were calculated per BRT and were presented without any reference to ecological response. This calculation has the disadvantage that ratios that deviate significantly from unity (expressing severe alterations) on the local basis cannot be indicated. The purpose was to explore possible remarkable differences in the degree of alteration between various BRTs. The averaged ratios from numerous FECs within each BRT have not shown clear results (average ratios quite far from unity) that could characterize certain BRTs differently than others. Hydrologic alteration of rivers due to water abstraction is expected to be important, even severe in local areas, but the impact of the large number of FECs without alteration in a BRT offsets the remarkable impact of specific FECs within it. This is enhanced by the widespread extent of many BRTs across the continent that doesn't allow regional trends to be clearly indicated.

For certain hydrologic indicators the ratios can be larger or lower than 1 depending on the BRT, but only slightly (0.95 < ratio < 1.05). For some occasions, even these slight deviations from unity could imply that the anthropogenic abstractions have the opposite effect than expected. For example, the annual discharge ratio is lower than 1 for the BRT 20, "Mediterranean Temporary/Intermittent streams", meaning that when water abstractions are present (anthropogenic scenario) the annual discharge is higher than the "near-natural scenario". Considering the very low values of the Base Flow Index (BFI) under the anthropogenic scenario and the extremely high ratio of alteration for the BFI in BRT 20, we can presume that there is a very large negative effect of groundwater abstractions on baseflow. But in certain cases, returned water through surface runoff losses of irrigation may have increased river discharge of temporary/intermittent rivers, that would have negligible flow otherwise. Or simulated point source discharge in the anthropogenic scenario may have increased water flowing in intermittent rivers. On the other hand, for most of the BRTs, absence of water abstractions (near-natural scenario) incur a slight increase of mean annual discharge as expected. The BRT 20, "Mediterranean Temporary/Intermittent streams" paradigm can show that the interpretation of hydrologic alterations caused by anthropogenic interventions (abstractions) is facilitated for BRTs of a particular hydrologic regime (Temporary/Intermittent streams) and of a concentrated geographical presence (Mediterranean).

4.3. European mapping of hydrologic alteration

To further investigate hydrologic stress in Europe we depicted the alteration on maps consisted of the 104,334 FECs of the analysis. The alteration was calculated here either as the ratio or as the difference of the value of a hydrologic indicator at the anthropogenic scenario from the near-natural, and the more comprehensive was selected for each individual indicator.

A clear reduction of annual river flows was indicated for parts of all the Mediterranean countries (Greece, Turkey, Spain, France, Italy) due to water abstractions occurring in the anthropogenic scenario. However, the average flow conditions are not influenced for the rest of Europe.

The analysis shows that the total number of low flow days within the 10-y simulated period remained also unaltered for the majority of the European area, with low flow events continuing to be longer in Northern and North-eastern Europe under water abstractions and shorter in all other areas. A comprehensive baseflow index has indicated that aquifers play a key role in the streams' total flow in central and northern Europe, enhancing long low flow events. This situation is almost identical with the respective one under the ideal natural flow conditions. This is not the case for Southern Europe where, in large parts, groundwater abstractions for irrigation have severely reduced aquifers' water contribution to adjacent rivers and streams. Similarly to low flows, extreme low flow events are important in the North (particularly in the Scandinavian region) due to climate characteristics which enhance the prolongation of low river flow conditions. The anthropogenic scenario has slightly increased the duration of extreme low flow events in areas scattered across Europe.

On the other hand, as low flows predominate, the total number of high flow days always covers less than the 50% of the time period and was found to remain unaltered for the largest part of the European area due to water abstractions. In contrast, alterations occur in scattered areas mostly within the Mediterranean and Balkan countries where also the frequencies of high flow events are reduced. This is also attributed to irrigation, which reduces water availability for river flow. High flow events do not last more than a week on average in both Southern and Central Europe but are much longer in Northern Europe. Alteration is not found here except at the very local basis.

A series of other hydrologic indicators and their ratios across Europe have supported the above general findings. Alteration of rivers' hydrology from the natural conditions due to anthropogenic activities is generally found to be clearer in Southern Europe where agricultural water use predominates. The FEC level calculations allow to investigate possible high alterations all across Europe and to this end, the produced GIS alteration layers are valuable. We have to mention that the assessment of hydrologic alteration in this work was based on simulated data from a global model with modelling uncertainties and simplifications which are unambiguously transferred to the present results. The short length of the simulated flow time-series does not allow us to focus more on flood events and their hydrologic characteristics as small and large floods are defined based on a 2-y and a 10-y interval respectively, certainly not in line with the 10-y length of available data.

In the absence of strong connection between specific hydrologic indicators and BQEs, the hydrologic stress on European rivers was investigated through a descriptive work in this report. The determination of a minimum ecological flow for each river and stream able to ensure good ecological status or the cause-effect relations between water quantity and ecological status could not be achieved. Mainly this is because of the poor ecological response data available and their reduced representativeness for the large area under study and its high resolution (FEC level). We hope that even the hydrologic alteration approach in this work can be helpful for all interested parties across Europe to make general assessments of the hydrologic stress and associate it with ecological response.

4.4. Key messages

- Regressions of hydrologic alteration indicators with the EQR values of two BQEs (macroinvertebrates and phytobenthos) showed insignificant or very weak relationships when applied both for all sites with available BQEs across Europe and separately for each Broad River Type.
- The lack of clear relationship is attributed mostly to the limited BQE information and representativeness (the vast majority shows high ecological status).
- By focusing on smaller scales (catchment or region) and implementing more sophisticated analyses (canonical correspondence analysis or principal component analysis), there were significant correlations of hydrologic alteration metrics with the structure of the macroinvertebrate community or selected macroinvertebrate metrics.
- Hydrologic stress of European rivers can be comprehensively expressed through the calculation of the ratio or difference (alteration) of numerous hydrologic indicators derived from time-series of daily river discharge occurring in a near-natural scenario without water abstractions, and time-series of discharge occurring in an anthropogenic (baseline) scenario with the typical water abstractions to cover needs.
- The calculated ratios from the 104,334 FECs of the analysis can be averaged per the FECs belonging to each of the 20 BRTs of Europe for exploring possible remarkable differences in the degree of alteration between various BRTs. This task did not reveal a high variation of hydrologic stress among BRTs.
- Mapping the hydrologic alteration on Europe’s geographic background, clearer trends in hydrologic stress were indicated across a North-South gradient.
- From the hydrologic indicators mapped, some of those, connected to low flow regimes (baseflow index, flood free season, low flow threshold), were the most informative showing that Southern Europe is more hydrologically stressed than the rest of Europe.
- Specifically, the magnitude of river flows, high flow days and the frequency of high flow events decrease in parts of the Mediterranean countries in the anthropogenic scenario, mostly because baseflow is significantly reduced due to abstractions.
- In the rest of Europe, especially the Northern part, flow temporal variations are much less pronounced, and the natural hydrologic conditions are preserved in the anthropogenic scenario.

- Alteration of rivers' hydrology from the natural conditions due to anthropogenic activities is generally found to be clearer in Southern Europe where agricultural water use predominates.
- However, the FEC level calculations within the GIS alteration layers allow the identification of possible significant hydrologic stress for local areas scattered all across Europe.
- The determination of a minimum ecological flow required to ensure rivers' good ecological status or the cause-effect relations between water quantity and ecological status were not shown in this work and need further research with the use of appropriate datasets. However, the water community can use effectively the outputs of this work to make assessments of the hydrologic stress and associate it with known ecological response across Europe.

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6. ANNEX

Table I. A list of calculated hydrologic parameters (included IHA and EFCs) with a short description

IHA and EFC parameter	Short Description
Mean annual flow	Mean annual flow in cubic meters per second
Annual C. V.	the standard deviation of all the daily flow values, divided by the mean annual flow
Flow predictability	Predictability ranges in value from 0 to 1 and is composed of two additive components: constancy (C), a measure of temporal invariance, and contingency (M), a measure of periodicity. The predictability of a stream with very constant flow will be mostly due to C, while the predictability of a stream with highly variable flow with a fixed periodicity will be mostly due to M.
Constancy/predictability	Flow constancy / flow predictability. C / (C+M).
% of floods in 60d period	Maximum proportion of floods that occur during any common 60 day period in all years during the period of record. Floods are defined as any flows above the high pulse threshold.
Flood-free season	Length of flood-free season. This is the length in days of the longest period common to all water years where flows are at or below the high pulse threshold in every year

January **February** **March** **April** **May** **June** **July** **August** **September** **October** **November** **December**	50th percentiles (medians)
1-day minimum **3-day minimum** **7-day minimum** **30-day minimum** **90-day minimum** **1-day maximum** **3-day maximum** **7-day maximum** **30-day maximum** **90-day maximum**	3-, 7-, 30-, and 90-day minimums and maximums are taken from moving averages of the appropriate length calculated for every possible period that is completely within the water year
Number of zero days **Base flow index**	The zero-flow days and base flow index parameters in group 2 are modeled after the suite of flow parameters described by Poff and Ward (1989).
Date of minimum	Julian date of each annual 1-day minimum
Date of maximum	Julian date of each annual 1-day maximum

Low pulse count **Low pulse duration** **High pulse count** **High pulse duration** **Low Pulse Threshold** **High Pulse Threshold**	A day is classified as a pulse (low or high) if it is greater than or less than a specified threshold (Low Pulse and High Pulse threshold). Count refers to number within each year and duration to median duration of pulses (in days).
Rise rate	median of all positive differences between consecutive daily values
Fall rate	median of all negative differences between consecutive daily values
Number of reversals	Reversals are calculated by dividing the hydrologic record into "rising" and "falling" periods, which correspond to periods in which daily changes in flows are either positive or negative, respectively
January Low Flow **February Low Flow** **March Low Flow** **April Low Flow** **May Low Flow** **June Low Flow** **July Low Flow** **August Low Flow** **September Low Flow** **October Low Flow** **November Low Flow** **December Low Flow**	median values of low flows during each calendar month
Extreme low peak	minimum flow during extreme flow event
Extreme low duration	median value of duration

Extreme low timing	Julian date of peak flow
Extreme low freq.	Frequency of extreme low flows during each water year or season
High flow peak	maximum flow during high flow event
High flow duration	median value of duration
High flow timing	Julian date of peak flow
High flow frequency	Frequency of high flows during each water year or season
High flow rise rate **High flow fall rate**	rise and fall rates
Small Flood peak	maximum flow during small flood event
Small Flood duration	median value of duration
Small Flood timing	Julian date of peak flow
Small Flood freq.	Frequency of small floods during each water year or season
Small Flood riserate **Small Flood fallrate**	rise and fall rates
Large flood peak	maximum flow during large flood event
Large flood duration	median value of duration
Large flood timing	Julian date of peak flow
Large flood freq.	Frequency of large floods during each water year or season
Large flood riserate **Large flood fallrate**	rise and fall rates
Zero flow days	Total number of zero flow days within the entire simulation period
Xlowflows	Total number of extreme flow days within the entire simulation period
Lowflows	Total number of low flow days within the entire simulation period
Highflowpulses	Total number of high flow days within the entire simulation period
SmallFloods	Total number of small flood days within the entire simulation period
LargeFloods	Total number of large flood days within the entire simulation period

Table II: Significant (p<0.05) pairwise comparisons (Dunn-Bonferroni test) for the ratio of annual discharge, Base Flow Index, Low Pulse Threshold and High Pulse Threshold

Annual Discharge			Base Flow Index			Low Pulse Threshold			High Pulse Threshold	
Adj. Sig.			Adj. Sig.			Adj. Sig.			Adj. Sig.	
1- all types	0.000		1- all types	0.000		1- all types	0.000		1- all types	0.000
2-3	0.000		2-3	0.000		2-3	0.000		2-3	0.000
2-5	0.000		2-5	0.000		2-5	0.000		2-5	0.000
2-9	0.000		2-9	0.012		2-9	0.000		2-8	0.000
2-11	0.000		2-10	0.000		2-11	0.000		2-9	0.000
2-20	0.000		2-11	0.000		3-5	0.000		2-11	0.000
3-5	0.001		3-5	0.000		3-9	0.022		2-14	0.000
3-9	0.000		3-10	0.000		3-11	0.000		2-17	0.000
3-11	0.000		3-11	0.000		4-3	0.000		2-18	0.000
4-2	0.000		4-5	0.000		4-5	0.000		2-19	0.000
4-3	0.000		4-10	0.000		4-9	0.000		2-20	0.000
4-5	0.000		4-11	0.000		4-11	0.000		3-5	0.000
4-6	0.000		6-2	0.032		6-2	0.040		3-9	0.000
4-9	0.000		6-3	0.000		6-3	0.000		3-11	0.000
4-11	0.000		6-4	0.000		6-5	0.000		3-20	0.000
4-12	0.000		6-5	0.000		6-8	0.000		4-3	0.000
4-20	0.000		6-8	0.000		6-9	0.000		4-5	0.000
5-11	0.000		6-9	0.000		6-10	0.032		4-8	0.000
6-3	0.000		6-10	0.000		6-11	0.000		4-9	0.000
6-5	0.000		6-11	0.000		6-12	0.000		4-11	0.000
6-9	0.000		6-12	0.001		6-20	0.001		4-14	0.028
6-11	0.000		7-3	0.000		7-3	0.000		4-17	0.000
6-20	0.000		7-4	0.004		7-5	0.000		4-18	0.000
7-3	0.000		7-5	0.000		7-8	0.016		4-19	0.000
7-5	0.000		7-8	0.001		7-9	0.000		4-20	0.000
7-9	0.000		7-9	0.000		7-11	0.000		5-11	0.000
7-11	0.000		7-10	0.000		7-12	0.002		6-3	0.000
7-20	0.000		7-11	0.000		7-20	0.002		6-5	0.000
8-3	0.000		7-12	0.002		8-3	0.000		6-9	0.000
8-5	0.000		8-5	0.000		8-5	0.000		6-11	0.000
8-9	0.000		8-10	0.000		8-9	0.000		6-17	0.000
8-11	0.000		8-11	0.000		8-11	0.000		6-18	0.000
8-20	0.000		9-5	0.000		9-5	0.000		6-19	0.000
9-11	0.000		9-10	0.000		9-11	0.000		6-20	0.000
10-3	0.000		9-11	0.000		10-3	0.000		7-3	0.001

10-5	0.000		10-11	0.000		10-5	0.000		7-5	0.000
10-9	0.000		12-5	0.001		10-9	0.000		7-9	0.000
10-11	0.000		12-10	0.342		10-11	0.000		7-11	0.000
10-20	0.000		12-11	0.000		12-5	0.000		7-17	0.012
12-3	0.000		14-10	0.000		12-11	0.000		7-19	0.000
12-5	0.000		14-11	0.000		14-10	0.000		7-20	0.000
12-9	0.000		14-12	0.000		14-11	0.000		8-3	0.000
12-11	0.000		14-2	0.000		14-12	0.000		8-5	0.000
14-11	0.000		14-3	0.000		14-2	0.000		8-9	0.000
14-12	0.007		14-4	0.000		14-20	0.000		8-11	0.000
14-20	0.000		14-5	0.000		14-3	0.000		8-17	0.012
14-3	0.000		14-8	0.000		14-4	0.000		8-19	0.000
14-5	0.000		14-9	0.000		14-5	0.000		8-20	0.000
14-6	0.002		15-10	0.000		14-8	0.000		9-5	0.002
14-9	0.000		15-11	0.000		14-9	0.000		9-11	0.000
15-11	0.000		15-12	0.001		15-10	0.000		10-3	0.000
15-12	0.000		15-2	0.023		15-11	0.000		10-5	0.000
15-2	0.000		15-3	0.000		15-12	0.000		10-9	0.000
15-20	0.000		15-4	0.000		15-2	0.000		10-11	0.000
15-3	0.000		15-5	0.000		15-20	0.000		10-17	0.000
15-5	0.000		15-8	0.000		15-3	0.000		10-18	0.000
15-6	0.000		15-9	0.000		15-4	0.000		10-19	0.000
15-9	0.000		16-10	0.000		15-5	0.000		10-20	0.000
16-11	0.000		16-11	0.000		15-8	0.000		12-3	0.000
16-20	0.000		16-3	0.000		15-9	0.000		12-5	0.000
16-3	0.000		16-5	0.000		16-10	0.014		12-9	0.000
16-5	0.000		16-8	0.014		16-11	0.000		12-11	0.000
16-9	0.000		16-9	0.001		16-12	0.000		12-17	0.001
17-10	0.000		17-10	0.000		16-2	0.019		12-18	0.035
17-11	0.000		17-11	0.000		16-20	0.000		12-19	0.000
17-12	0.000		17-12	0.000		16-3	0.000		12-20	0.000
17-14	0.000		17-14	0.000		16-5	0.000		14-11	0.000
17-15	0.031		17-15	0.000		16-8	0.000		14-17	0.001
17-16	0.000		17-16	0.000		16-9	0.000		14-19	0.000
17-19	0.000		17-2	0.000		17-10	0.000		14-20	0.000
17-2	0.000		17-20	0.000		17-11	0.000		14-3	0.000
17-20	0.000		17-3	0.000		17-12	0.000		14-5	0.000
17-3	0.000		17-4	0.000		17-2	0.000		14-9	0.000
17-4	0.000		17-5	0.000		17-20	0.000		15-11	0.000
17-5	0.000		17-6	0.000		17-3	0.000		15-17	0.000
17-6	0.000		17-8	0.000		17-4	0.000		15-18	0.001
17-8	0.000		17-9	0.000		17-5	0.000		15-19	0.000
17-9	0.000		18-10	0.000		17-8	0.000		15-20	0.000

18-10	0.001	18-11	0.000	17-9	0.000	15-3	0.000
18-11	0.000	18-12	0.000	18-10	0.000	15-5	0.000
18-12	0.000	18-14	0.000	18-11	0.000	15-9	0.000
18-16	0.009	18-15	0.000	18-12	0.000	16-11	0.000
18-19	0.000	18-16	0.000	18-14	0.000	16-19	0.000
18-2	0.000	18-2	0.000	18-16	0.015	16-20	0.000
18-20	0.000	18-20	0.000	18-2	0.000	16-3	0.000
18-3	0.000	18-3	0.000	18-20	0.000	16-5	0.000
18-5	0.000	18-4	0.000	18-3	0.000	16-9	0.000
18-6	0.000	18-5	0.000	18-4	0.000	17-11	0.000
18-8	0.000	18-6	0.000	18-5	0.000	17-20	0.000
18-9	0.000	18-7	0.012	18-6	0.000	17-5	0.000
19-11	0.000	18-8	0.000	18-8	0.000	17-9	0.000
19-20	0.000	18-9	0.000	18-9	0.000	18-11	0.000
19-3	0.000	19-10	0.000	19-10	0.000	18-20	0.000
19-5	0.000	19-11	0.000	19-11	0.000	18-5	0.000
19-9	0.000	19-12	0.000	19-12	0.000	18-9	0.000
20-11	0.000	19-14	0.000	19-2	0.000	19-11	0.000
		19-15	0.000	19-20	0.000	19-20	0.001
		19-16	0.000	19-3	0.000	19-5	0.000
		19-2	0.000	19-4	0.000	19-9	0.000
		19-20	0.000	19-5	0.000		
		19-3	0.000	19-6	0.006		
		19-4	0.000	19-8	0.000		
		19-5	0.000	19-9	0.000		
		19-6	0.000	20-11	0.000		
		19-8	0.000	20-5	0.000		
		19-9	0.000				
		20-10	0.000				
		20-11	0.000				
		20-12	0.034				
		20-3	0.000				
		20-5	0.000				
		20-8	0.011				
		20-9	0.001				

Table III: Significant (p<0.05) pairwise comparisons (Dunn-Bonferroni test) for the ratio of Large Flood duration, Small Flood duration, Extreme low duration and Extreme low frequency.

Large Flood duration		Small flood duration		Extreme low durarion		Extreme Low Freq.	
	Adj. Sig.		Adj. Sig.		Adj. Sig.		Adj.Sig
2-1	0.000	2-1	0.000	3-1	.000	1-2	.000
2-3	0.000	2-4	0.000	3-2	.000	1-3	.000
2-4	0.000	2-8	0.011	3-4	.000	1-4	.000
2-5	0.000	2-10	0.000	3-8	.000	1-5	.000
2-8	0.000	2-15	0.000	5-2	.046	1-8	.000
2-9	0.000	2-17	0.000	9-1	.000	1-9	.000
2-10	0.000	2-18	0.000	9-2	.000	1-10	.000
2-11	0.000	2-19	0.000	9-4	.000	1-11	.000
2-14	0.000	2-20	0.003	9-8	.000	1-14	.000
2-15	0.000	3-1	0.000	10-1	.016	1-15	.000
2-16	0.000	3-4	0.000	10-2	.000	1-16	.000
2-17	0.000	3-5	0.000	10-8	.049	1-17	.000
2-18	0.000	3-8	0.000	11-2	.003	1-18	.000
2-19	0.000	3-10	0.000	14-2	.006	1-19	.000
2-20	0.000	3-11	0.000	15-1	.000	2-3	.000
3-1	0.000	3-14	0.000	15-2	.000	2-9	.000
3-4	0.000	3-15	0.000	15-4	.001	2-10	.031
3-5	0.000	3-17	0.000	15-8	.001	2-14	.000
3-8	0.000	3-18	0.000	17-1	.008	2-15	.000
3-9	0.000	3-19	0.000	17-2	.000	2-16	.000
3-10	0.000	3-20	0.000	17-4	.027	2-17	.019
3-11	0.000	4-1	0.000	17-8	.024	2-19	.000
3-14	0.000	4-10	0.000	18-1	.000	4-3	.000
3-15	0.000	4-17	0.000	18-2	.000	4-9	.000
3-16	0.000	4-18	0.000	18-4	.000	4-15	.003
3-17	0.000	4-19	0.000	18-5	.003	5-3	.000
3-18	0.000	5-1	0.000	18-8	.000	5-9	.000
3-19	0.000	5-10	0.000	19-1	.000	5-15	.002
3-20	0.000	5-17	0.000	19-2	.000	6-3	.000
4-1	0.000	5-18	0.000	19-4	.000	6-4	.021
4-17	0.000	6-1	0.000	19-5	.001	6-9	.000
4-19	0.037	6-4	0.002	19-8	.000	6-10	.004
5-1	0.000	6-10	0.000	20-1	.000	6-11	.049
5-17	0.000	6-15	0.007	20-11	.022	6-14	.000
6-1	0.000	6-17	0.000	20-14	.009	6-15	.000
6-2	0.000	6-18	0.000	20-2	.000	6-16	.000
6-3	0.000	6-19	0.000	20-4	.000	6-17	.002

Large Flood duration		Small flood duration		Extreme low durarion		Extreme Low Freq.	
	Adj. Sig.		Adj. Sig.		Adj. Sig.		Adj.Sig
6-4	0.000	6-20	0.021	20-5	.001	6-19	.000
6-5	0.000	7-1	0.000	20-6	.017	8-3	.000
6-7	0.000	7-4	0.008	20-8	.000	8-9	.000
6-8	0.000	7-10	0.000			8-15	.001
6-9	0.000	7-15	0.007			8-16	.020
6-10	0.000	7-17	0.000			11-3	.004
6-11	0.000	7-18	0.000			11-9	.003
6-12	0.000	7-19	0.000			11-15	.011
6-14	0.000	7-20	0.004			20-10	.000
6-15	0.000	8-1	0.000			20-11	.003
6-16	0.000	8-10	0.000			20-14	.000
6-17	0.000	8-17	0.000			20-15	.000
6-18	0.000	8-18	0.000			20-16	.000
6-19	0.000	8-19	0.000			20-17	.000
6-20	0.000	9-1	0.000			20-18	.006
7-1	0.000	9-4	0.000			20-19	.000
7-10	0.018	9-8	0.014			20-3	.000
7-17	0.000	9-10	0.000			20-4	.001
7-18	0.002	9-15	0.000			20-5	.003
7-19	0.003	9-17	0.000			20-8	.009
7-20	0.020	9-18	0.000			20-9	.000
8-1	0.000	9-19	0.000				
8-10	0.019	9-20	0.003				
8-17	0.000	10-1	0.000				
8-18	0.000	10-17	0.000				
8-19	0.000	11-1	0.000				
9-1	0.000	11-10	0.000				
9-4	0.000	11-17	0.000				
9-5	0.000	11-18	0.000				
9-10	0.000	11-19	0.000				
9-14	0.000	12-1	0.000				
9-15	0.008	12-10	0.000				
9-17	0.000	12-17	0.000				
9-18	0.000	12-18	0.000				
9-19	0.000	12-19	0.000				
9-20	0.000	14-1	0.000				
10-1	0.001	14-10	0.000				
10-17	0.000	14-17	0.000				
11-1	0.000	14-18	0.000				
11-10	0.005	14-19	0.000				
11-17	0.000	15-1	0.000				

Large Flood duration		Small flood duration		Extreme low durarion		Extreme Low Freq.	
	Adj. Sig.		Adj. Sig.		Adj. Sig.		Adj.Sig
11-18	0.000	15-10	0.021				
11-19	0.000	15-17	0.000				
12-1	0.000	15-18	0.001				
12-4	0.000	16-1	0.000				
12-5	0.000	16-10	0.000				
12-10	0.000	16-17	0.000				
12-14	0.000	16-18	0.000				
12-15	0.004	16-19	0.000				
12-17	0.000	18-1	0.000				
12-18	0.000	18-17	0.000				
12-19	0.000	19-1	0.000				
12-20	0.000	19-17	0.000				
14-1	0.000	20-1	0.000				
14-17	0.000	20-17	0.000				
15-1	0.000						
15-17	0.000						
16-1	0.000						
16-17	0.000						
18-17	0.003						
19-1	0.000						
19-17	0.000						

Table IV: Significant (p<0.05) pairwise comparisons (Dunn-Bonferroni test) for the ratio of High Pulse count and Low Pulse count

High Pulse Count	Adj. Sig.		Low Pulse count	Adj. Sig.
1- all types	0.000		1-2	0.000
2-3	0.000		1-3	0.000
2-6	0.000		1-4	0.000
2-9	0.000		1-5	0.000
2-10	0.000		1-6	0.000
2-11	0.000		1-7	0.000
2-12	0.001		1-8	0.000
3-9	0.013		1-9	0.000
4-3	0.000		1-10	0.000
4-6	0.000		1-11	0.000
4-9	0.000		1-12	0.000
4-10	0.000		1-13	0.000
4-11	0.000		1-14	0.000
4-12	0.000		1-15	0.000
5-3	0.000		1-16	0.000
5-6	0.000		1-17	0.000
5-9	0.000		1-18	0.000
5-10	0.000		1-19	0.000
5-11	0.000		2-3	0.000
5-12	0.004		2-6	0.072
7-6	0.045		2-9	0.001
7-9	0.037		2-12	0.006
8-3	0.000		2-15	0.000
8-6	0.000		2-16	0.001
8-9	0.000		2-19	0.020
8-10	0.000		4-3	0.000
8-11	0.000		4-6	0.014
8-12	0.000		4-9	0.000
14-10	0.000		4-12	0.002
14-11	0.000		4-15	0.000
14-12	0.000		4-16	0.000
14-15	0.015		4-19	0.001
14-16	0.000		5-3	0.000
14-2	0.000		5-9	0.014
14-3	0.000		5-12	0.022
14-4	0.000		5-15	0.000
14-5	0.000		5-16	0.008
14-6	0.000		8-3	0.000

High Pulse Count Adj. Sig.			Low Pulse count Adj. Sig.	
14-8	0.001		8-6	0.009
14-9	0.000		8-9	0.000
15-10	0.000		8-12	0.001
15-11	0.000		8-15	0.000
15-12	0.001		8-16	0.000
15-3	0.000		8-19	0.001
15-6	0.000		10-2	0.000
15-9	0.000		10-3	0.000
16-9	0.017		10-4	0.000
17-10	0.000		10-5	0.000
17-11	0.000		10-6	0.000
17-12	0.000		10-8	0.005
17-14	0.000		10-9	0.000
17-15	0.000		10-11	0.000
17-16	0.000		10-12	0.000
17-18	0.000		10-14	0.000
17-19	0.000		10-15	0.000
17-2	0.000		10-16	0.000
17-20	0.000		10-19	0.000
17-3	0.000		11-3	0.001
17-4	0.000		11-15	0.000
17-5	0.000		14-15	0.002
17-6	0.000		17-0	0.000
17-7	0.000		17-10	0.001
17-8	0.000		17-11	0.000
17-9	0.000		17-12	0.000
18-10	0.000		17-13	0.031
18-11	0.000		17-14	0.000
18-12	0.000		17-15	0.000
18-14	0.000		17-16	0.000
18-15	0.000		17-19	0.000
18-16	0.000		17-2	0.000
18-19	0.000		17-3	0.000
18-2	0.000		17-4	0.000
18-20	0.001		17-5	0.000
18-3	0.000		17-6	0.000
18-4	0.000		17-7	0.000
18-5	0.000		17-8	0.000
18-6	0.000		17-9	0.000
18-7	0.000		18-0	0.000
18-8	0.000		18-11	0.000

High Pulse Count			Low Pulse count	
Adj. Sig.			Adj. Sig.	
18-9	0.000		18-12	0.000
19-10	0.000		18-14	0.000
19-11	0.000		18-15	0.000
19-12	0.000		18-16	0.000
19-14	0.000		18-19	0.000
19-15	0.000		18-2	0.000
19-16	0.000		18-3	0.000
19-2	0.000		18-4	0.000
19-3	0.000		18-5	0.000
19-4	0.000		18-6	0.000
19-5	0.000		18-8	0.000
19-6	0.000		18-9	0.000
19-7	0.000		19-15	0.015
19-8	0.000		20-0	0.000
19-9	0.000		20-10	0.000
20-10	0.000		20-11	0.000
20-11	0.000		20-12	0.000
20-12	0.000		20-13	0.003
20-14	0.000		20-14	0.000
20-15	0.000		20-15	0.000
20-16	0.000		20-16	0.000
20-2	0.000		20-18	0.000
20-3	0.000		20-19	0.000
20-4	0.000		20-2	0.000
20-5	0.000		20-3	0.000
20-6	0.000		20-4	0.000
20-7	0.000		20-5	0.000
20-8	0.000		20-6	0.000
20-9	0.000		20-7	0.000
			20-8	0.000
			20-9	0.000

Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

MARS PROJECT

Managing Aquatic ecosystems and water Resources under multiple Stress

Deliverable D5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis.

D5.1-3: Impact of multi-stressors on ecosystem services and their monetary value

Lead beneficiary: **Joint Research Centre (JRC)**

Contributors: **Bruna Grizzetti (JRC), Camino Liquete (JRC), Alberto Pistocchi (JRC), Olga Vigiak (JRC), Arnaud Reynaud (JRC), Denis Lanzanova (JRC), Cosimo Brogi (JRC), Ana Cristina Cardoso (JRC), Grazia Zulian (JRC), Marta Faneca Sanchez (Deltares)**

Due date of deliverable: **Month 36**

Actual submission date: **Month 36**

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)	

Content

Non-technical summary

Which are the ecosystem services (i.e. the contribution of nature to human well-being) provided by European rivers, lakes, and coastal waters? Can we map and quantify them? Do enhanced ecosystem conditions and biodiversity support higher benefits for people? These are the questions addressed in this research.

We quantify the main ecosystem services provided by aquatic ecosystems at the European scale, including fish provisioning, water provisioning, water purification, erosion prevention, flood protection, coastal protection, and recreation. These services are provided by aquatic ecosystems, such as lakes, rivers, groundwater and coastal waters. We show European maps of ecosystem service capacity, flow (actual use), sustainability or efficiency and, when possible, benefit.

Our results indicate that the ecosystem services are mostly positively correlated with the ecological status of European water bodies (that is a measure of the ecosystem integrity and biodiversity), except for water provisioning, which strongly depends on the climatic and hydrographic characteristics of river basins. We also highlight how provisioning services can act as pressures on the aquatic ecosystems. Based on the relationship between ecosystem status and delivery of services, we explore qualitatively the expected changes of ecosystem services under scenarios of increase in different pressures.

Finally, we perform an economic valuation of the ecosystem services provided by European lakes, considering the current conditions and scenarios of improvement of the ecological status. Using a benefit transfer approach, we estimate that the average economic value of ecosystem services delivered by a European lake is 2.92 million EUR per year. We also demonstrated that the ecological status of lake has an impact on valuation. The expected benefit from restoring all European lakes into at least a moderate ecological status is estimated to be 5.9 billion EUR per year, which corresponds to 11.7 EUR per person and per year.

Quantifying and valuing ecosystem services helps to recognise all the benefits that humans receive from nature, offering stronger arguments to protect and restore ecosystems and thus fostering the implementation of the European water policy. This study offers scientific evidence to this aim.

1. Introduction

The EU FP7 project MARS analyses the impacts of multiple stressors on the ecological status of European aquatic ecosystems (rivers, lakes, groundwater and coastal water) and on their provisioning of ecosystem services at different spatial scales (water body, river basin, European scale), with the overall objective to support the implementation of the Water Framework Directive (WFD) and the sustainable management of water resources (Hering et al. 2014).

Why it is relevant for the EU water policy to assess and value ecosystem services provided by aquatic ecosystems? Ecosystem services are the direct and indirect contributions of ecosystems to human well-being (TEEB 2010). Quantifying and valuing ecosystem services helps to recognise all the benefits that humans receive from nature, offering stronger arguments to protect and restore ecosystems (Guerry et al. 2015). To this purpose the ecosystem service approach considers all the benefits that people receive from aquatic ecosystems, highlighting also hidden benefits that are often not accounted in cost-benefits analysis (Liquete et al. 2016a). Including all ecosystem services provided by aquatic ecosystems could justify the cost of their conservation and restoration. This aspect is of interest for the implementation of the water policy in the EU (WFD), since the application of River Basin Management Plans for the protection of water resources and aquatic ecosystems involves substantial costs. (For an analysis of the risks and benefits of applying the ecosystem service concepts in the implementation of the WDF see Grizzetti et al. 2016a).

In Europe the MAES (Mapping and Assessment of Ecosystems and their Services) Working Group was established to support the implementation of the EU Biodiversity Strategy to 2020 (European Commission, 2011) and the EU Green Infrastructure Strategy (European Commission, 2013), with the aim of developing common methodologies in Europe on the mapping and assessment of ecosystem services (Maes et al. 2016). In addition several EU funded research projects have been working on the mapping and the operationalisation of the ecosystem services concepts for improved management of water, land and urban areas (OpenNESS 2017, OPERA 2017).

These projects have produced valuable mapping of terrestrial ecosystem services (Maes et al. 2015) and analysis of tools and real case applications (Dick et al. submitted). However the assessment of ecosystem services provided in specific by aquatic ecosystems and their economic valuation at the European scale has not been developed so far. In addition, what is missing is a systematic analysis of the relationship between the ecological status and the delivery of ecosystem services at the large scale. Understanding this relationship, and more generally the links between pressures, ecological status and ecosystem services remains crucial to develop sustainable management of water resources and aquatic ecosystems and could shed light on the effects of future scenario of change of multiple stressors.

The research presented in this report addresses these challenges. The **objective** of this study was to quantify and map the biophysical and economic values of ecosystem services delivered by aquatic ecosystems at the European scale, analyse their relationship with the ecological status and explore their changes under multi-stressors scenarios. Assessing the impacts of multi-stressors on ecosystem services requires understanding the links between pressures, status and ecosystem services.

The analysis is based on the **methodology** and literature review developed in the project MARS Deliverable D2.1 (Part 2) and Grizzetti et al. 2016b. The ecosystem services of interest for the water management are those related to the aquatic ecosystems and to the interaction of water and land in different ecosystems, such as forests, agricultural lands, riparian areas, wetlands, and water bodies. In this study we indicate all these services as 'water ecosystem services' (Grizzetti et al. 2016b).

This report is organised in three sessions. The first part presents the **assessment of ecosystem services** related to aquatic ecosystems at the European scale, showing European maps of the services (Section 2). The second part analyses their **relationship with the ecological status**, and their expected changes under scenarios of multiple pressures (Section 3). Finally, the third part describes the **economic valuation** of the ecosystem services provided by European lakes, considering the current conditions and scenarios of improvement of the ecological status (Section 4).

2. Ecosystem services provided by European aquatic ecosystems

2.1 Assessment of ecosystem services

We quantified major ecosystem services provided by aquatic ecosystems at the European scale, including fish provisioning, water provisioning for different uses, water purification, erosion prevention, flood protection, coastal protection, and recreation (Table 2.1). These services are provided by aquatic ecosystems, such as lakes, rivers, groundwater, coastal waters, or by ecosystems at the interface, strongly affected by the interaction of land and water, such as riparian areas, floodplains and wetlands (Table 2.2).

Table 2.1. Ecosystem services considered in the study.

	Ecosystem services
Provisioning	Food provisioning (fisheries and aquaculture)
	Water provisioning for drinking and economic use
Regulation & Maintenance	Water purification
	Erosion prevention
	Flood protection
	Coastal protection
Cultural	Recreation

Table 2.2. Relevance/presence of ecosystem services per ecosystem type.

	Ecosystem services	Rivers	Lakes	Coastal and transitional water	Groundwater	Floodplains	Riparian areas	Wetlands	Soil-groundwater
Provisioning	Food provisioning	√	√	√					
	Water provisioning for drinking and economic use	√	√		√				
Regulation & Maintenance	Water purification	√	√	√	√	√	√	√	√
	Erosion prevention					√	√	√	
	Flood protection		√			√	√	√	√
	Coastal protection			√				√	
Cultural	Recreation	√	√	√		√	√		

For each ecosystem service we quantified proxies/indicators that are able to describe the different aspects of the service, considering indicators of the service capacity, flow, sustainability, efficiency, and when possible of the benefits, according to the conceptual framework presented in Grizzetti et al. (2016b) and indicators presented in Liquete et al. (2013a) (Figure 2.1). The *capacity* refers to the potential of the ecosystem to provide the service. The *flow* is the actual use of the service. *Benefits* are associated with human well-being and value system. Indicators of *sustainability* of the service informs on the sustainable use of the service, considering capacity and flow together. But as the capacity for some services is unknown or unaccountable, indicators of service *efficiency* can inform on the efficiency of the process responsible of the service (Figure 2.1).

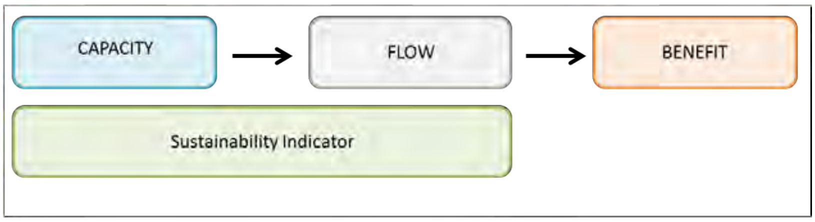


Figure 2.1. Conceptual framework to classify indicators of water ecosystem services (source Grizzetti et al. 2016b; MARS Deliverable D2.1 Part 2).

Distinguishing between the different typologies of indicators allows to correctly identify the information provided by each indicator, supporting the analysis of the relationship between pressures, ecological status and ecosystem services delivery that will be discussed in Section 3.

In this study the quantification of ecosystem services indicators was performed at the spatial resolution of small catchments (average size 180 km^2) for rivers, water body for lakes and areas units for coastal water (average coastal length 30 km), which are spatial units relevant for the River Basin Management Plans.

The proposed proxies/indicators to quantify water ecosystem services at the European scale are presented in Table 2.3. Their choice was based on the review and analysis of previous international studies, EU projects and the MAES working group activities, as presented in Grizzetti et al. (2016b) and MARS Deliverable D2.1 (Part 2). The ecosystem services are estimated using European data and models, according best data available. The following part describes the ecosystem services indicators (considering capacity, flow, sustainability or efficiency, benefits) and their main limitations, providing key messages for each service.

Table 2.3. Proposed proxies/indicators to quantify ecosystem services at the European scale.

Ecosystem services	Natural capacity	Service flow	Sustainability or efficiency	Benefit
Food provisioning (fisheries and aquaculture)	• "Composition, abundance and age structure of fish fauna"	• Fish catch • Aquaculture production	• Trend of the inland wild captures	
Water provisioning (for drinking and non-drinking)	• Total renewable water	• Water demand	• Water Exploitation Index	
Water purification	• Natural areas in floodplains	• Nitrogen retention	• Ratio of nitrogen retained vs. total input to water body	• Value of nitrogen retention
Erosion mitigation	• Density of vegetated riparian land	• Sediment retention in riparian land	• Ratio sediment retention in riparian land vs. total input to water body	
Flood protection	• Natural areas in floodplains	• Water volume retained for a flood with 200 years return time		
Coastal protection	• Protection capacity of natural systems	• Protection supply		• Human demand for coastal protection
Recreation and tourism	• Recreation potential	• Recreation opportunity spectrum		

2.1.1 Food provisioning

The service

Food provision is the delivery of biomass for human consumption and the conditions to grow it. Regarding fresh water habitats, it relates to inland fishing activities. In Europe, we can neglect subsistence fishing and recreational fishing as relevant sources of food, so we concentrate on industrial fisheries. They can rely on wild captures and on aquaculture (mainly fish harvesting).

Selected indicators

Natural capacity

The ecological assessment of fish populations (diversity, stock, species condition, etc.) can be used to characterise the natural capacity to supply fish for human consumption. The first of these ecological assessments, applied and intercalibrated at European scale, is one of the biological quality elements (BQE) under the WFD: "Composition, abundance and age structure of fish fauna". This is a sub-indicator of the overall ecological status that should be available for most European surface water bodies (Figs. 2.2 and 2.3).

Other possible indicators that could be explored in a later stage of the MARS project are (1) the conservation status of fresh water fish species of community interest under the Habitats Directive, but this information is less explicit in geographical terms and the latest reporting (of 2013) is still not available as a European database; (2) an ecological assessment comparable to the previous one developed by Freyhof and Brooks (2011), still lacking detailed geographical representation; or (3) specific stock assessments of commercial species, usually developed at regional level and not publicly available.

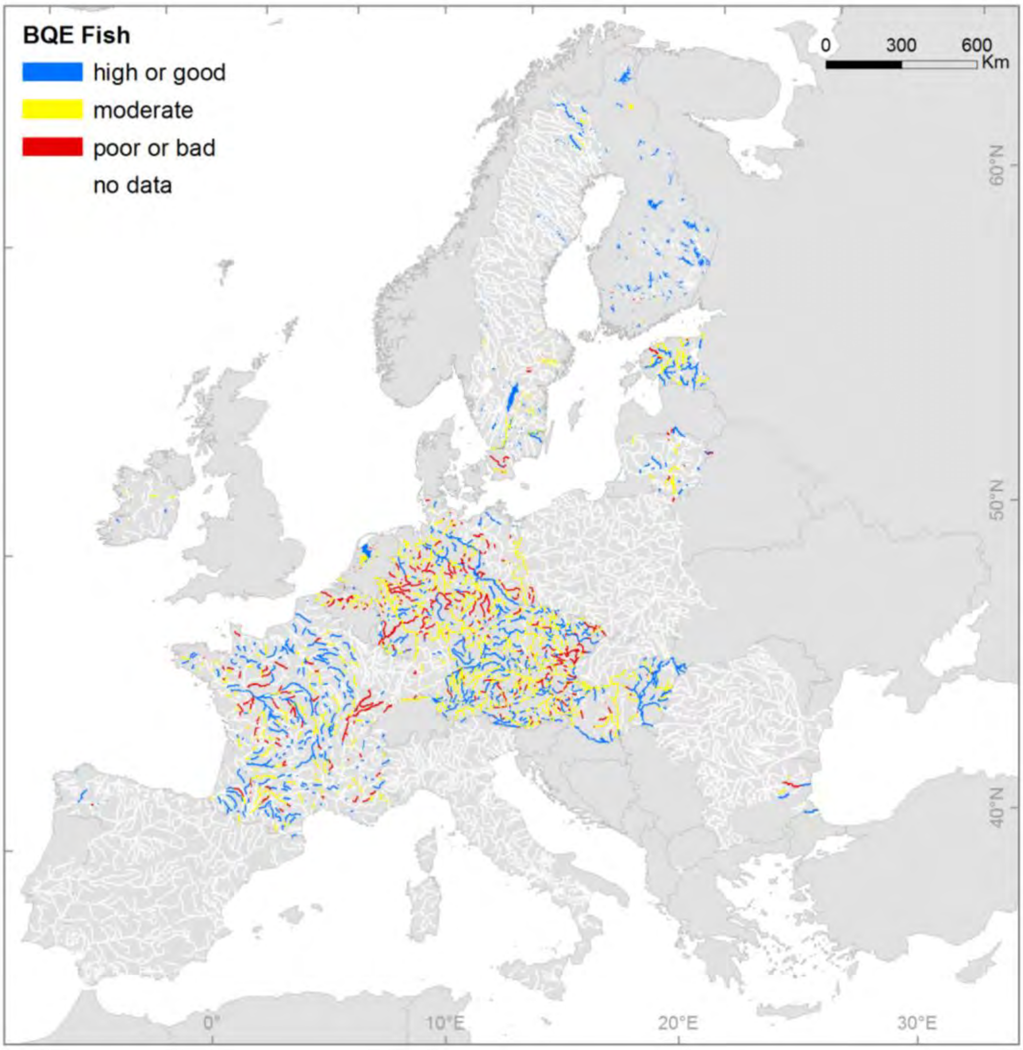


Figure 2.2. Fish ecological status (sub-indicator Fish of the WFD ecological status) reflecting the composition, abundance and age structure of fish per surface water body. This is selected as an indicator of the natural capacity to provide fish.

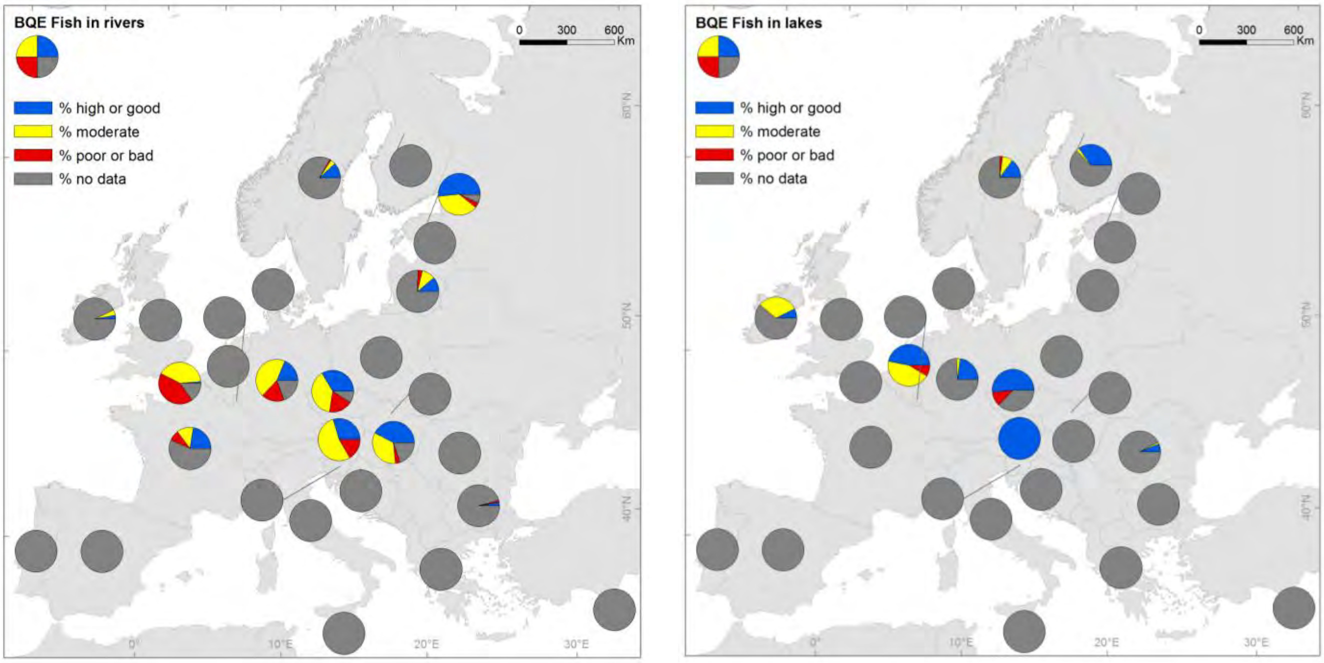


Figure 2.3. Fish ecological status reflecting the composition, abundance and age structure of fish per country. This is selected as an indicator of the capacity of freshwater ecosystems to provide food.

Service flow

The total production from inland fisheries (catches), differentiating between wild captures and aquaculture, represents the flow or delivery of food provision. The best source of information for a European scale analysis is the EUROSTAT Fisheries database[1] which holds data on inland fisheries production from catches and aquaculture at national level, but with a variable time coverage between 1998 and 2011 (Fig. 2.4). The FAO global fisheries database (FishStatJ[2]) compiles similar data from 1950 to 2010, making it better for long-term analyses. Additional sources of information could be Mitchell et al. (2012) for national statistics or the FishBase[3] for species details.

[1] http://epp.eurostat.ec.europa.eu/portal/page/portal/fisheries/data/database
[2] http://www.fao.org/fishery/statistics/software/fishstatj/en
[3] http://www.fishbase.org/search.php

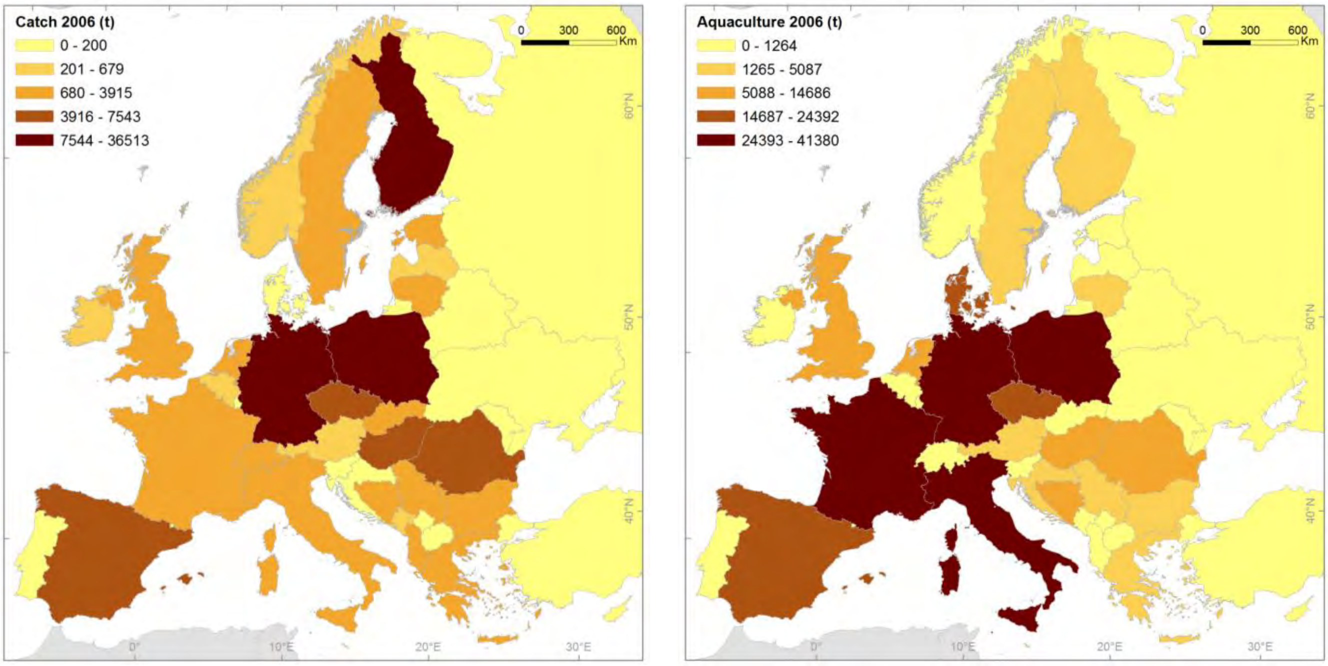


Figure 2.4. Wild captures (catch) and aquaculture production from inland fisheries (in tonnes live weight for the year 2006) reported by Member States and collected in the EUROSTAT Fisheries database. One value per country is the maximum resolution at which these data are available.

Sustainability or efficiency

An ideal sustainability indicator of food provision would compare the wild captures (flow) *vs.* quantity of adult, healthy commercial fish (capacity). However, the lack of coverage of our capacity indicator and the lack of spatial resolution of our flow indicator makes it impractical to compare both proxies.

As an alternative, we analyse the temporal trend of the inland fish catches focusing on the evolution of wild captures, since aquaculture production depends more on human inputs and industrial decisions than on natural capacity. The trend analysis of freshwater fish captures in EU, candidate and Balkan countries (the 34 countries represented in Fig. 2.5) was based on 20 long-term series (with data since the 50s or 60s) and 14 shorter data series from the FishStatJ database. The most common trend, observed in 24 countries, is a continuous rise in captures until reaching a maximum peak, usually occurring during the 80s, followed by a more or less sudden drop until today (Fig. 2.6). Following this general trend, the total European catches decreased from 182.5 to 109 thousand tonnes between 1980 and 2012; and the actual magnitude of this drop is probably larger since in 1980 only 19 of the 34 countries were reporting to FAO. Among the other 10 countries showing increasing trends, there are four states with long-term data series showing a continuous growth in captures (BG, NO, PL, UK) and other six states with poorer data and a shorter or more instable history (BA, HR, CZ, MK, RS, SK).

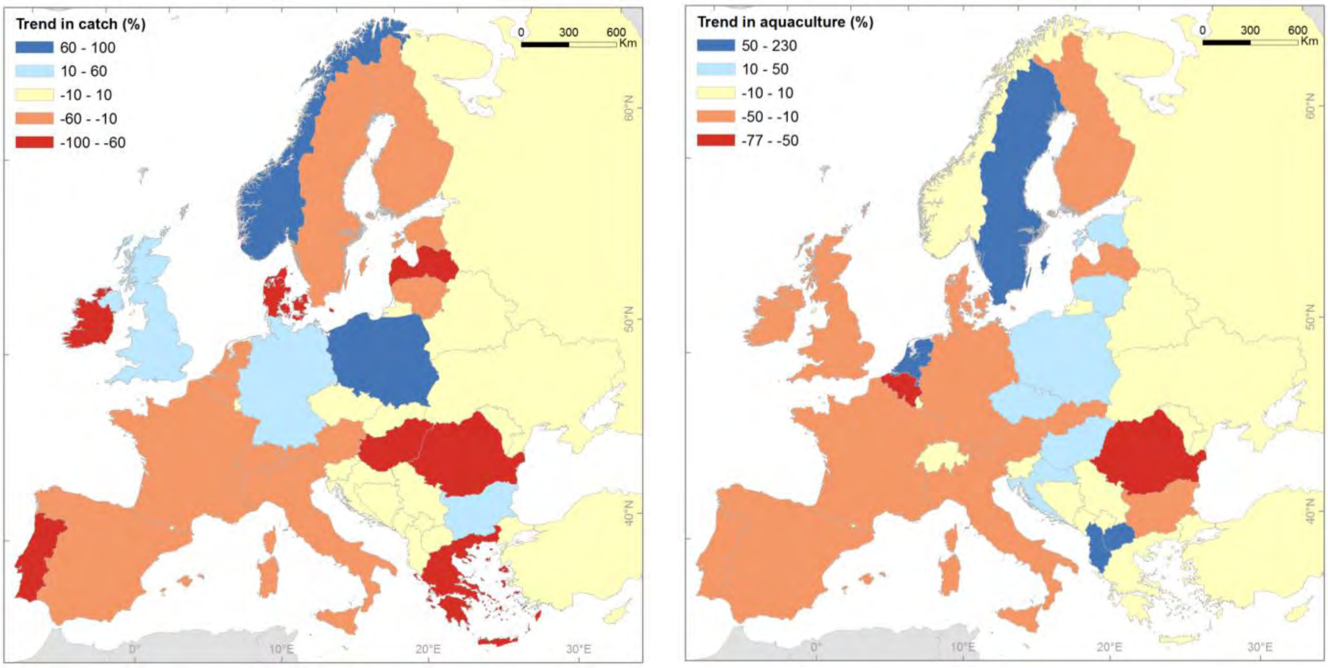


Figure 2.5. Trend in wild captures or catch (from the 1980s to today) and in aquaculture production (from the 1990s to today) from inland fisheries as reported by FAO.

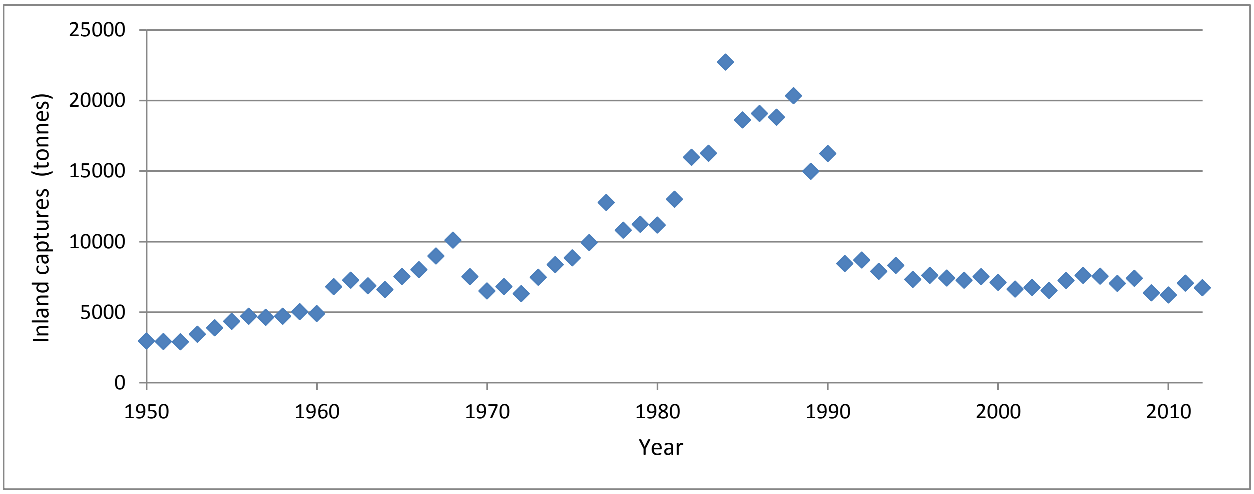


Figure 2.6. Representation of the most common trend of inland wild fish captures in Europe, in this case with data from Hungary. It shows the evolution of the delivery of food by fresh water ecosystems.

Benefit

We can propose some suitable indicators to value this service such as the market value of the fish catch (differentiating between public and private investments, or industrial benefits and subsidies, which can be particularly important in this sector); the employment generated by the fishing activities; or the analysis of diet composition as an indicator of human demand for fish. For monetary values, possible data sources could be the EUMOFA[4] database, the GLOBEFISH[5] market reports, or specific publications (e.g. Tveterås et al. 2012).

Main limitations

These results must be considered preliminary. The low spatial resolution of the public data available (e.g. fisheries information only at country level) and the number of data gaps found (e.g. lack of monitoring or reporting of the Good Ecological Status' sub-indicators in many water bodies) hamper the usefulness of these results. However, the proposed metrics could be applied and could generate significant results at a lower scale, provided that more detailed information is available.

The indicator "Composition, abundance and age structure of fish fauna" is quite ambiguous and qualitative. More specific, quantitative metrics coming from monitoring networks or modelling approaches would be preferred.

The analysis of fishing trends is shown as a proxy of the state of fish stocks but it also relates to multiple natural and economic factors, so it cannot be interpreted as a direct sustainability index. An optimal alternative could be for instance the ratio between wild captures and total population biomass.

Key message

The capacity of European freshwaters to support fisheries is largely unknown at the scale of this analysis and varies among countries, although fish in lakes seem to be in better conditions. The decrease of fish catch is evident in most European countries at least since the 1980s, most probably due to the health and biomass decline of fish populations. Aquaculture production is increasing only in countries where its relative importance (total production) is still low.

[4] http://ec.europa.eu/fisheries/market-observatory/home
[5] http://www.globefish.org/homepage.html

2.1.2 Water provisioning

The service

Fresh water is a fundamental service that nature provides to humans. Groundwater, rivers and lakes can be sources of clean water for drinking purposes and domestic uses, and they provide water for economic activities, such as industry, energy production, irrigation and livestock. Water provisioning refers to the water that is abstracted from the water bodies, and can eventually be released back to the water system.

Selected indicators

Natural capacity

The total renewable water (m^3/yr, long term average of the stream flow plus net groundwater recharge) that is naturally produced by a river basin indicates the capacity of the system to provide water. It depends on climate, geology, topography, soil and vegetation characteristics of the river basin. Several hydrological models are available to estimate the amount of water produced by a river basin, ranging from simple Budyko approaches (Pike, 1964), considering the difference between precipitation and actual evapotranspiration, to more sophisticated models, including equations representing the physical processes occurring in the basins. Examples of the latters applied at the European scale are PCR-GLOBWB (Sperna Weiland et al. 2010) and LISFLOOD (De Roo et al. 2000, 2012).

European maps of total renewable water estimated by Budyko approach (Pistocchi et al. in prep.a), is presented in Figure 2.7.

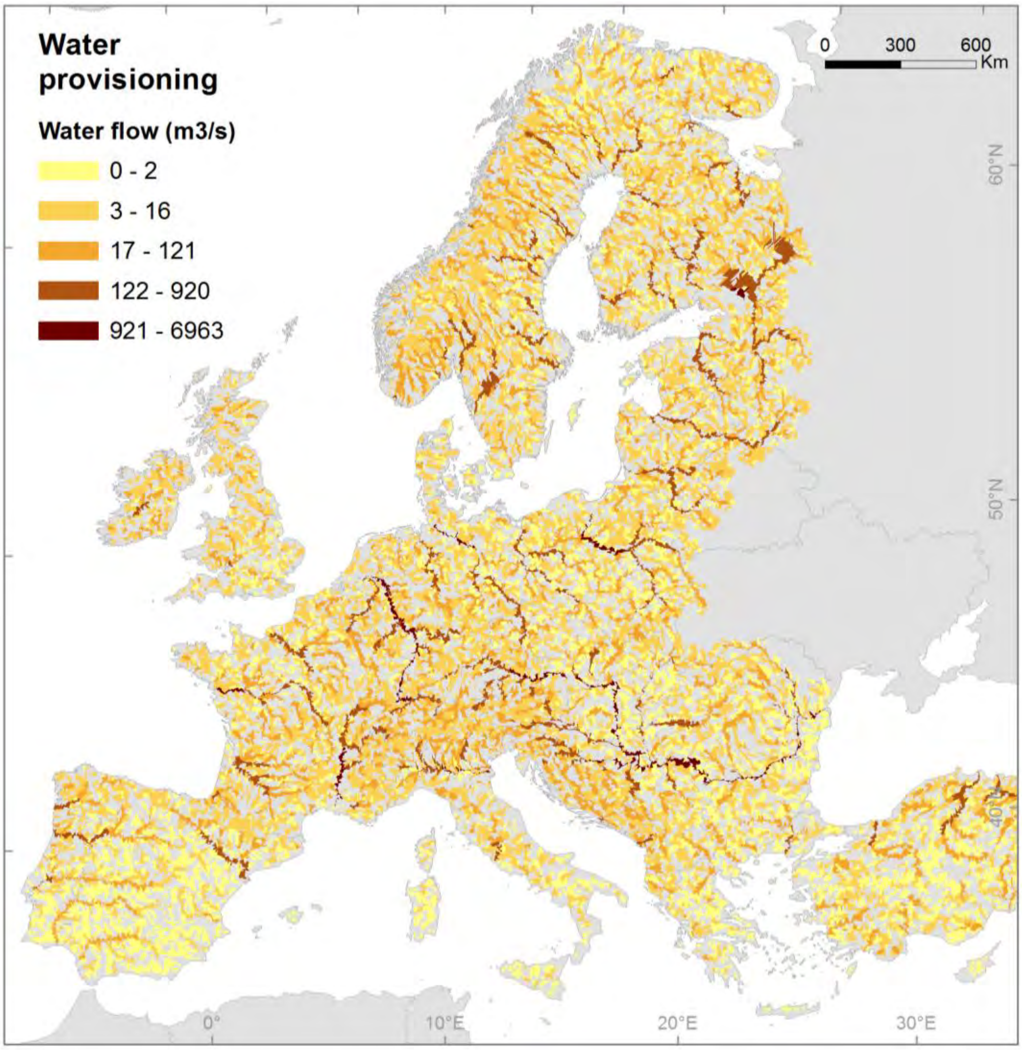


Figure 2.7 Total renewable water estimated by Budyko approach.

Service flow

The actual use of freshwater by humans is quantified by the annual water abstractions for different uses (m^3/yr), which include: drinking purposes, domestic use, industry, energy production, irrigation and livestock. Each use has specific requirements on the quality of water and the temporal availability. We assumed that water demand is a good proxy for water abstractions. The quantification and mapping of water abstractions is based on statistics reported by countries to EUROSTAT and FAO (Vandecasteele et al. 2014; Mubareka et al. 2013; De Roo et al. 2012).

A European map of total water demand based on the data of De Roo et al. (2012) is shown in Figure 2.8.

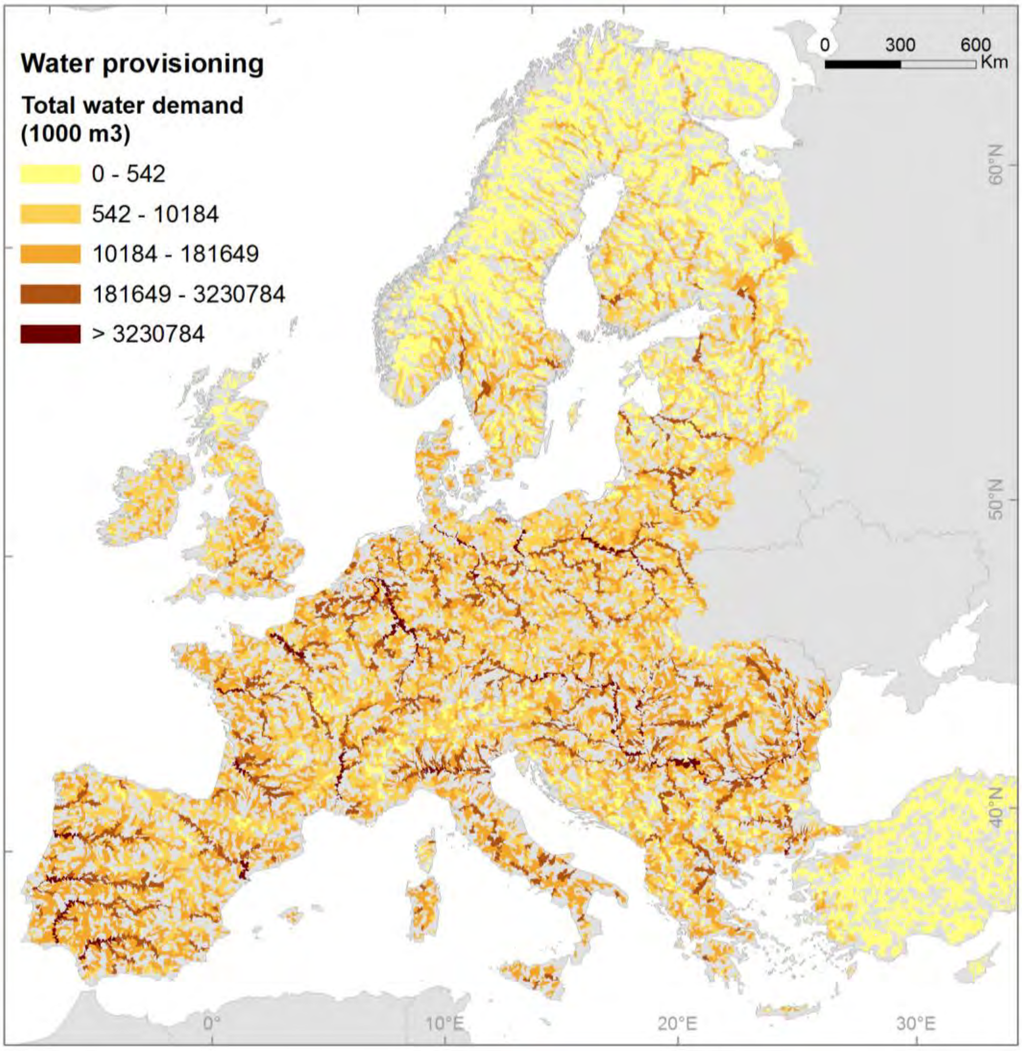


Figure 2.8 Total water demand (De Roo et al. 2012).

Sustainability or efficiency

The sustainability of water provisioning can be assessed by indicators of water scarcity that combine the natural water availability with the amount of human abstractions in a river basin. The Water Exploitation Index (WEI) (EEA 2010) has been applied in studies on water scarcity at the European scale, in particular in the Blueprint for Europe's Waters (De Roo et al. 2012). It is computed as the ratio between total water abstractions (m^3, considering all uses), and the total available water (m^3) (Eq. 1). WEI is expressed as a fraction.

WEI = Total water abstractions / Total available water (Eq. 1)

A European map of WEI computed considering the total available water estimated by Budyko approach (Pistocchi et al. in prep.a) is presented in Figure 2.9.

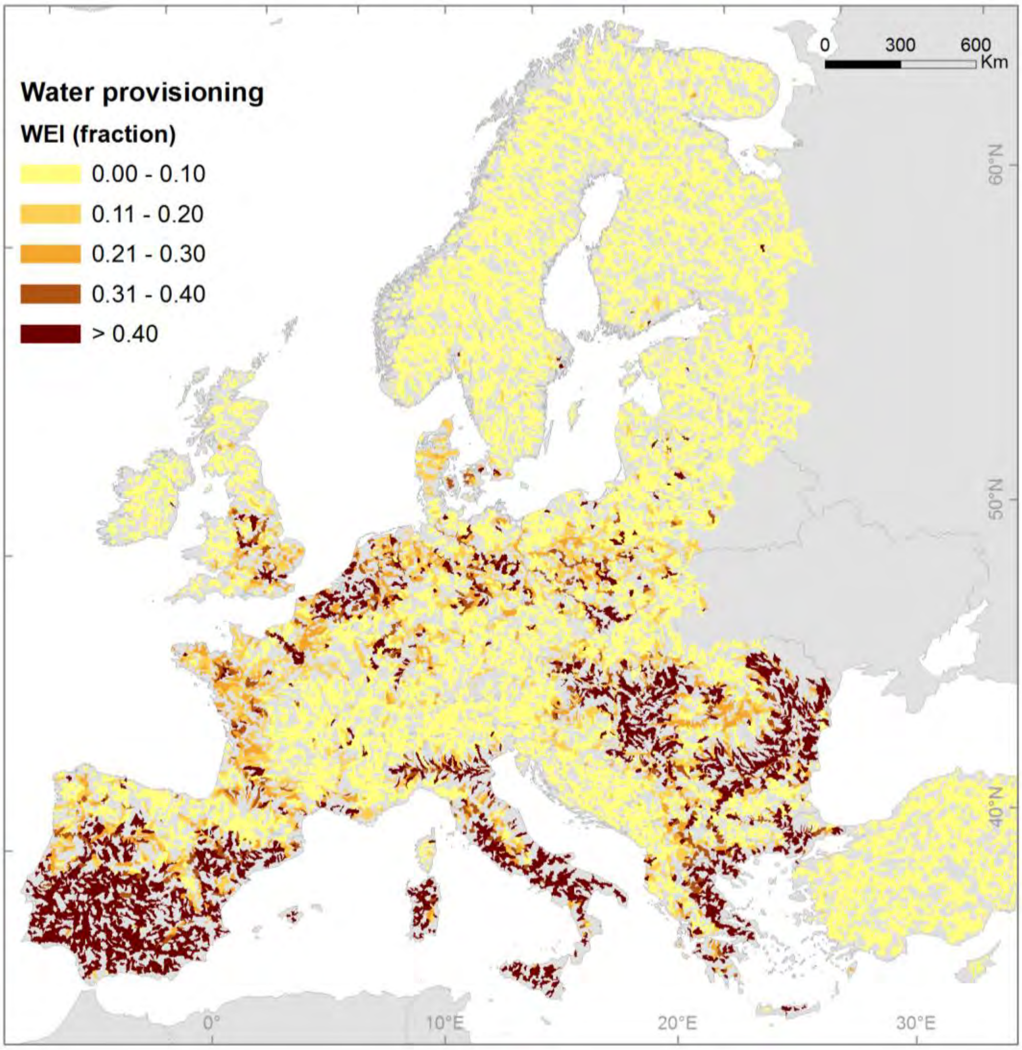


Figure 2.9 Water Exploitation Index (WEI), computed as ratio between water abstraction and water availability.

Main limitations

In Europe several hydrological models are available at the continental and the river basin scale that can provide reliable estimation of the available water resources. However these approaches rely on the quality of information for their set up and calibration. In particular they need to be validated against time series measurements of water flow. While the capacity of the system to provide water can be estimated by modelling, the quantification and spatial resolution of the water withdrawals for different uses entirely depends on statistics reported by countries. The latter often do not specify the source of abstractions, whether it is groundwater or surface waters, and generally do not provide the exact location of the abstractions, but are reported by administrative units that differ from the river basins, which are the hydrological functional units. This implies some assumptions when spatializing the information on water abstractions.

The WEI is a widely used indicator for water scarcity, whose formulation and computation is straightforward. Obviously its accuracy depends on the quality of the estimation of water availability and water abstractions. WEI indicates the regions concerned but higher water stress or scarcity, considering both natural conditions and human consumption of water resources. In absence of reliable data on water abstractions other indicators can be used, such as the Falkenmark indicator that considering the population living in the river basin instead of the water abstractions (Falkenmark 1989).

Key messages

Europe is naturally rich in water resources, especially in Northern, Western and Central part, with some limitation in the Mediterranean region, due to climatic conditions. Water abstractions are quite intense across the continent, they related to the high population density is some areas, the energy production and economic activities, and the irrigation of crops in water scarce regions that are also intensive agricultural areas. As a result of the water abstractions, even water rich areas are suffering water stress (WEI values between 20 and 30), such as in the Northern Europe and in the Po valley, or even water scarcity (WEI values >30), such as in Southern European regions.

2.1.3 Water purification

The service

Water purification indicates the removal of pollutants from water that is mediated by microorganisms. In large scale assessments the nitrogen retention has been suggested as proxy for water purification service (Liquete et al. 2015; Sharp et al. 2015; La Notte et al. 2015).

Selected indicators

Natural capacity

In aquatic ecosystems nitrogen retention can be temporal, related mainly to algae and plant uptake, or permanent when nitrogen is lost to the atmosphere by the process of denitrification operated by bacteria. Denitrification takes place in anoxic conditions where nitrate and electron donors are simultaneously available. Besides in soils and wetlands, these conditions occur in groundwater, hyporheic zones, riparian sediments, bottom waters and sediments of lakes and estuaries, and in the water column of suboxic river reaches (Seitzinger et al. 2006). Overall, the interface between land and water are very actives zones for biogeochemical processes and denitrification.

The actual capacity of the ecosystem to remove nitrogen, or pollutants, cannot be measured. For large scale assessments, spatial data on area occupied by wetlands, riparian vegetation, rivers and lakes can be used as proxies to map the presence of the service (Fig. 2.10).

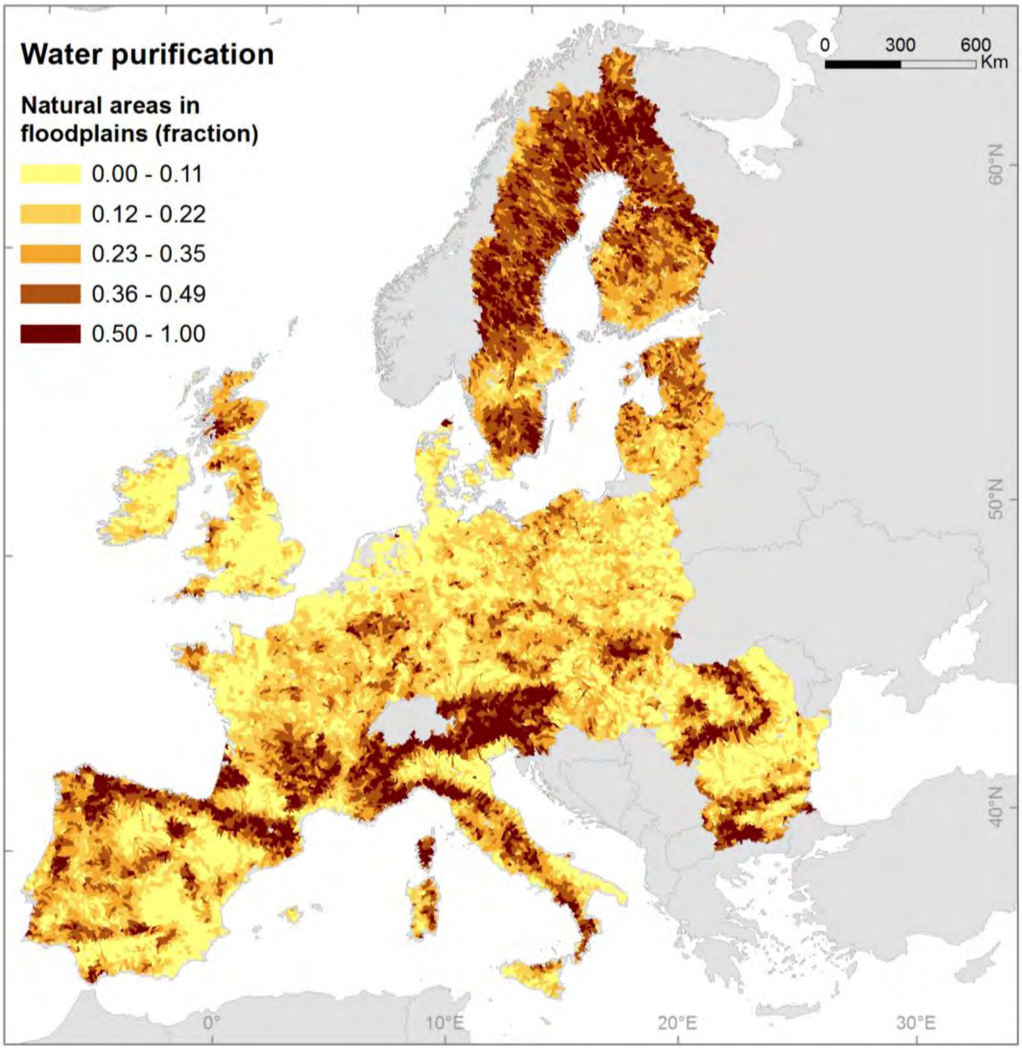


Figure 2.10 Natural areas in floodplains (Pistocchi et al. 2015).

Service flow

Generally the amount of nitrogen removed by water ecosystems is estimated as the difference between input and measured output, or by means of biogeochemical models that consider the water cycle and nutrient processes. Models can have different complexity in the way they represent the sources and pathways of nitrogen and the hydrological processes in the river basin (Bouraoui and Grizzetti 2014).

A European map of nitrogen retention in surface waters estimated by the GREEN model (Grizzetti et al. 2012) is presented in Figure 2.11. In the project MARS also the model MONERIS will be used (Venohr et al. 2011).

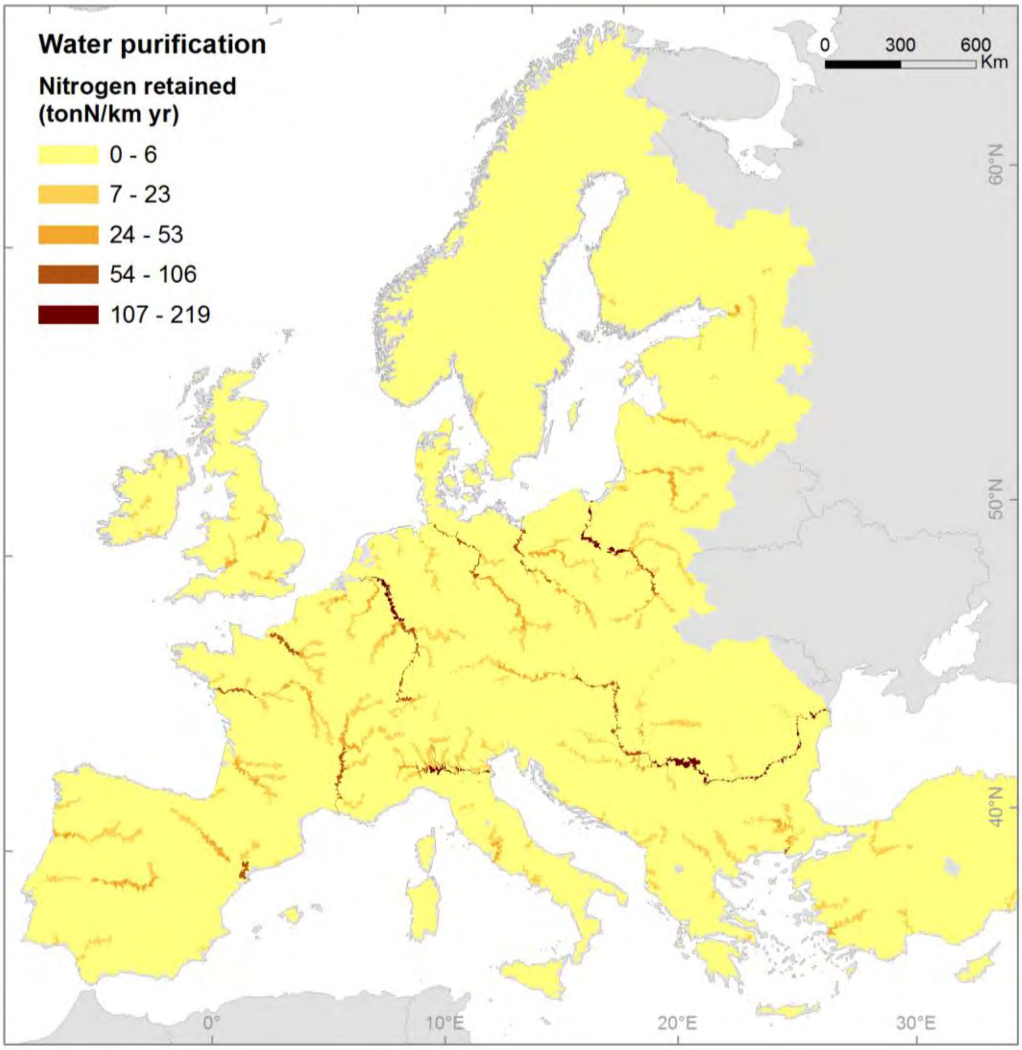


Figure 2.11 Nitrogen retention in surface waters estimated by the GREEN model.

Sustainability or efficiency

Estimations of nitrogen retention per river length or per lake area provide information on the ration between service flow and service presence. But as the latter is only a proxy of the capacity of the ecosystem, the nitrogen retention per river length is not a proper indicator of the sustainability of the service. In addition, scientific studies have shown that increasing the amount of nitrogen discharged in the water body lowers the efficiency of nitrogen removal (Mulholland et al. 2008), indicating that the efficiency of the service flow under higher pollution load can decline. Furthermore, increasing nitrogen load to water bodies can foster the process of eutrophication with a consequent reduction of habitat and biodiversity. This in turn can reduce the ability of the ecosystem to uptake nitrogen (Cardinale et al. 2011).

To take these aspects into consideration, nitrogen retention efficiency (fraction) can be suggested as an indicator of efficiency of the ecosystem service in surface waters (Fig. 2.12). It is calculated as the quantity of nitrogen removed by the water system divided by the total amount of nitrogen that reaches the water system. The computation is performed by spatial units (catchments).

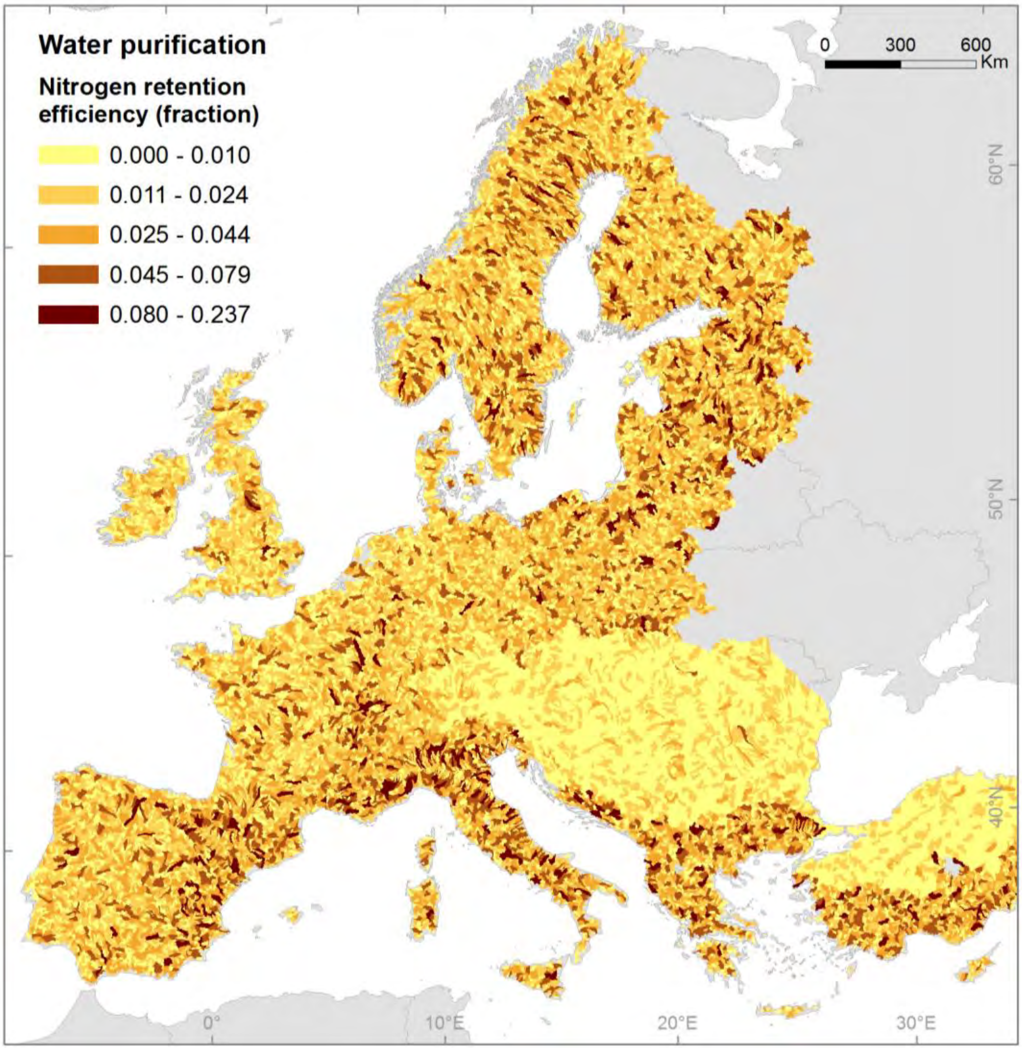


Figure 2.12 Nitrogen retention efficiency based on the results of the model GREEN.

Benefit

The economic valuation of regulation & maintenance services is particularly difficult, as these services do not have markets (Adams 2014). Alternative methodologies, such as replacement cost and contingent valuation, have been developed for these ecosystem services for which market prices are not available (Grizzetti et al. 2016b). La Notte et al. (2017) have estimated the economic value of nitrogen retention at the European scale, using the estimations of nitrogen retention provided by the GREEN model and the replacement cost methodology (Fig. 2.13). In particular, this approach takes into account the sustainability of the nitrogen retention service, introducing a threshold, based on nitrogen concentration in waters, beyond which the value of further retention starts declining.

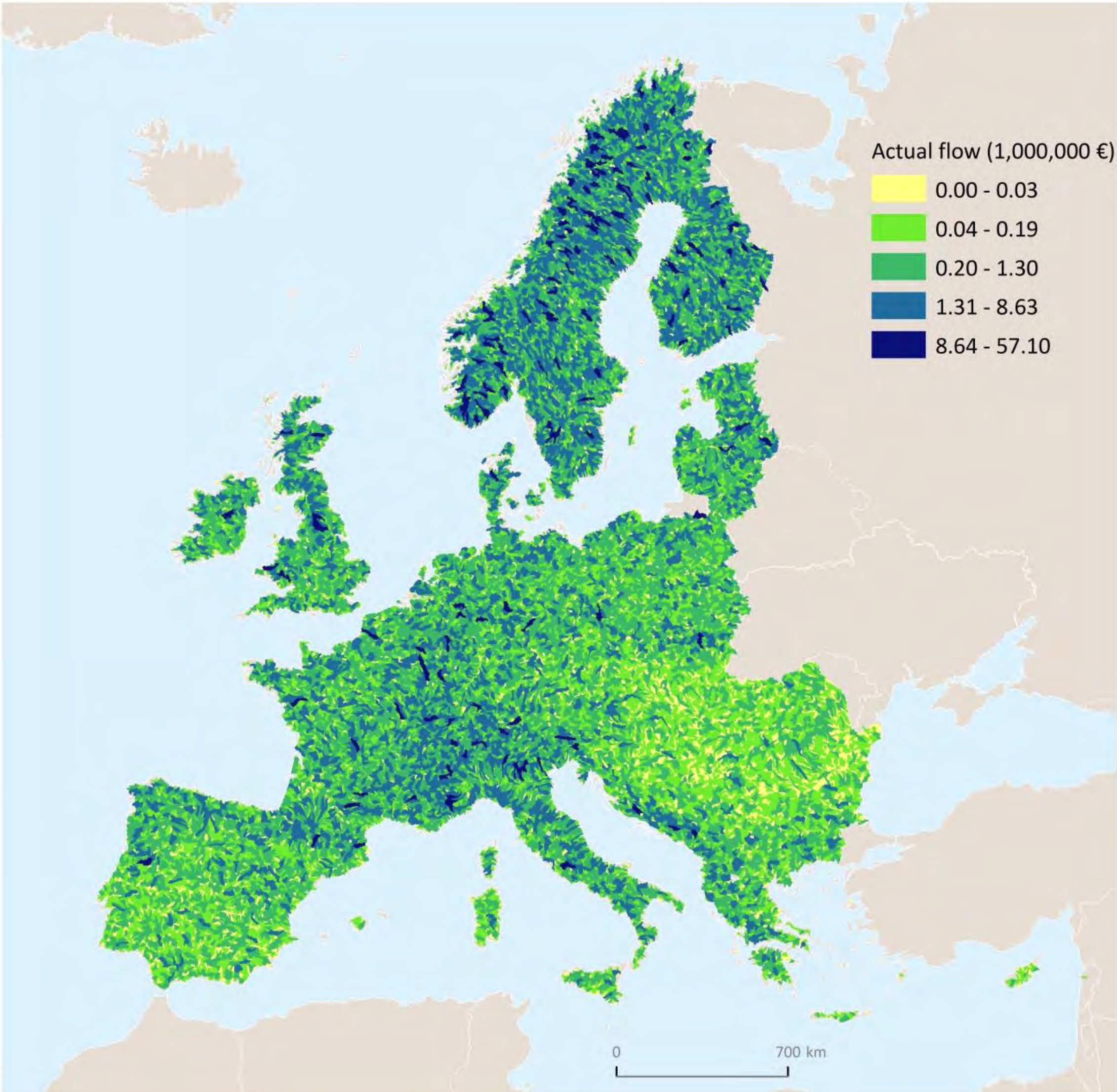


Figure 2.13 Value of nitrogen retention (map source: La Notte et al. 2017).

Main limitations

A thorough quantification of the capacity of the aquatic ecosystems to purify water is not possible, as this is operated by microorganisms and strongly depends on local physic-chemical conditions varying in time. For nitrogen, the estimation of nitrogen retention (both temporal and permanent) in river basins is challenging, for the difficulty in quantifying nitrogen diffuse and point sources with sufficient spatial resolution, as well as the uncertainty in assessing nitrogen removal taking place in the different water bodies in the river basin, i.e. aquifers, riparian areas, rivers, lakes and estuaries. In these systems the processes of nitrogen denitrification, burial in sediments, immobilization, transformation and transport take place (Billen et al. 1991; Bouwman et al. 2013). However, many of these processes are represented only by simply retention coefficients in the models, or require detail information for the quantification. Furthermore, as the residence time of the different water bodies in the river basin can vary widely, from hundreds years in aquifers to months in rivers, there is a time lag between the input of nitrogen sources in the aquatic system and their impacts, which makes the computation of retention more complex. (For a discussion on the challenges in assessing nitrogen retention in river basins see Grizzetti et al. 2015).

A real indicator of sustainability of nitrogen retention cannot be computed. The assessment of the efficiency of nitrogen retention in surface waters is affected by the uncertainty in the estimation of nitrogen sources and removal. In addition, when modelling nitrogen retention there is a risk of compensation between the estimation of nitrogen retention in soils and aquifers, before entering the river system, and the estimation of nitrogen retention in riparian areas, rivers and lakes, as direct measurements on nitrogen diffuse emission to surface waters are not possible at the river basin scale (Hejzlar et al. 2009, Kronvang et al. 2009; Grizzetti et al. 2015).

Finally, it is important to stress that the economic value associated to the nitrogen retention of European rivers (Fig. 2.13) does not represent the exact value of the service, rather it was developed to raise awareness among decision makers on the relevance of the service and the benefit that the ecosystem provide to the society (La Notte et al. 2017).

Key messages

Nitrogen retention is a good proxy of the purification mediated by microorganism, especially considering the relevance that nitrogen pollution has assumed at the global scale (Rockström et al. 2009). Estimations of nitrogen retention in surface water are available for Europe, based on modelling tools. However, while nitrogen retention increases with the input of nitrogen to water, the efficiency of the process may decline, as shown by scientific evidence (Mulholland et al. 2008) and large scale assessments. Indeed nitrogen retention is higher in large rivers and lakes with longer residence time (Fig. 2.11), but the efficiency decrease from upstream to downstream rivers (Fig. 2.12).

2.1.4 Erosion prevention

The service

Sediment retention is the service provided by vegetation mitigating the adverse impact of incoming sediments on the freshwater body. Riparian areas are active zones for biogeochemical processes and hydrological connectivity. They are crucial to many ecosystem services. Besides nursery habitat and pollution retention, riparian areas reduce sediment fluxes in the freshwater systems, by trapping sediments generated in the basin and stabilizing the stream banks (Stutter et al. 2012; Dosskey et al. 2012).

Selected indicators

Natural capacity

Riparian land has been mapped at the European scale by Clerici et al. (2013) and more recently by COPERNICUS (Weissteiner et al., 2016), based on the analysis of satellite images and topographic information. Natural capacity can be expressed in terms of riparian land area for km of reach (km^2/km). For example, based on Clerici et al. (2013), riparian land density was estimated for the Danube river basin (Fig. 2.14; Vigiak et al. 2016).

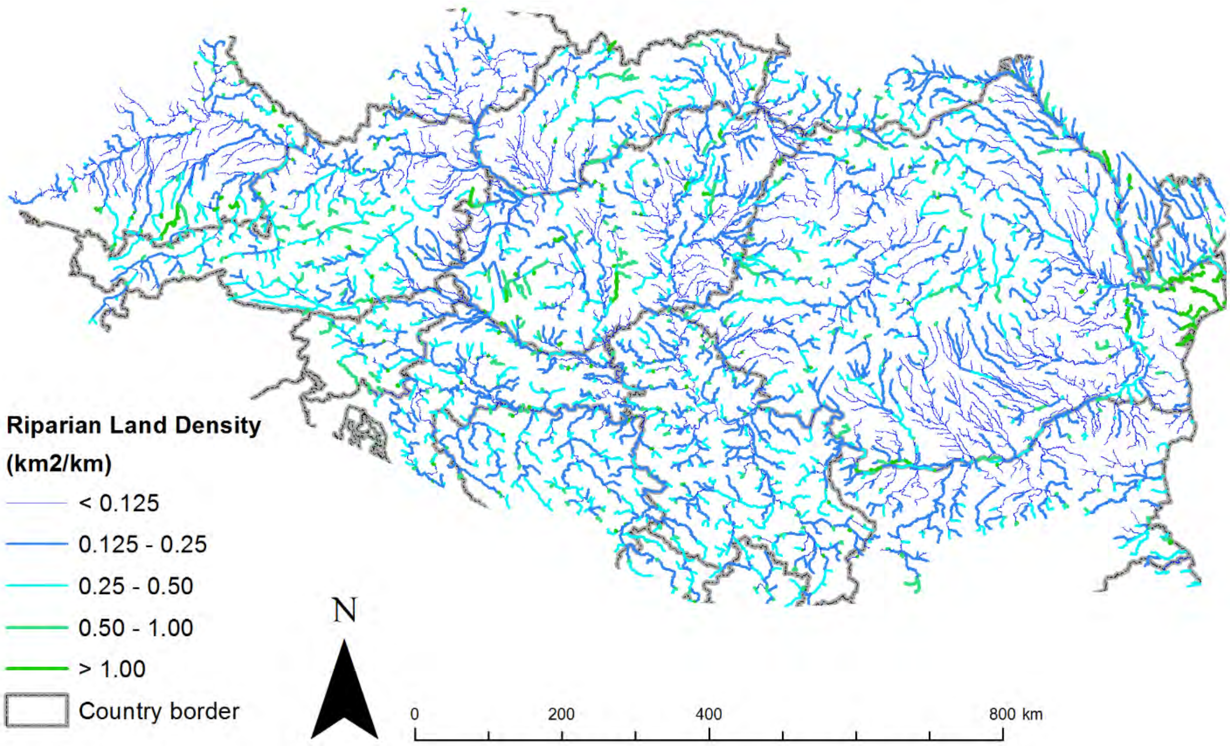


Figure 2.14 Riparian land density (riparian area per km of reach, km2/km) in the Danube river basin (from Vigiak et al. 2016 based on pan-European riparian land map of Clerici et al. 2013).

Service flow

The service flow of sediment retention can be expressed as sediment load removal afforded by riparian land, namely as the difference of mean annual sediment yields (t/km^2/y) that cross any given reach in the absence and in the presence of riparian land . Sediment yields can be estimated by process-based models that simulate sediment fluxes in the landscape, e.g. hillslope erosion, the sediment trapping in the riparian areas before reaching the river network, sediment transport in rivers, and streambank erosion in reaches. This approach considers both the process of trapping of sediment loads generated by hillslope erosion and prevention of streambank erosion. These models combine spatial information on climate, topography, soil, land use and cover, vegetation characteristics and farming practices.

The hydrological model SWAT (Arnold et al. 2012) has been used in spatial analysis of sediment transport in river basins (Gassman et al., 2014). An assessment of sediments removal by riparian land in the Danube river basin based on the SWAT model (Vigiak et al. 2016) is presented in Figure 2.15.

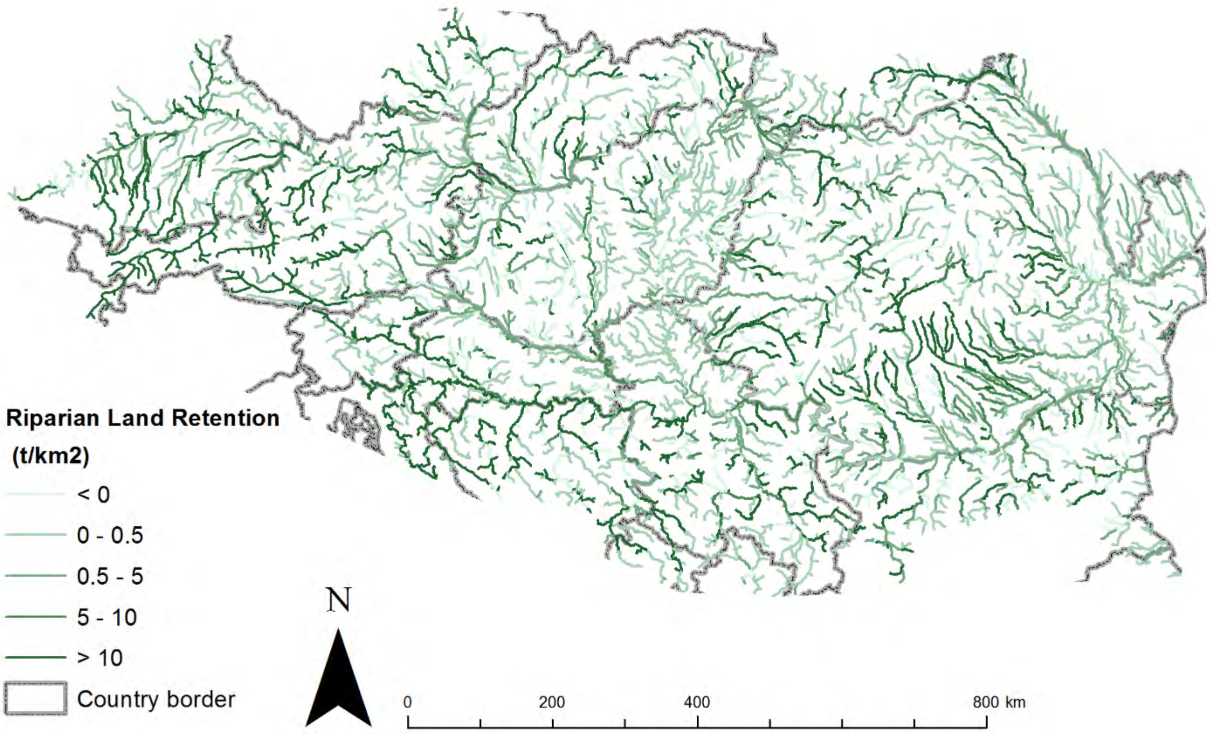


Figure 2.15 Sediment removal (t/km^2/yr) by riparian land in the Danube river basin based on the SWAT model (Vigiak et al. 2016). Data refers to annual means for the 15-year simulation period 1995-2009.

Sustainability or efficiency

Under high sediments load riparian land can saturate and its capacity to trap sediments can decline progressively (Dosskey et al. 2010). At the large scale the sustainability of the erosion prevention service cannot be assessed with field measurements. However, the efficiency of sediments removal by riparian land can be estimated by modelling outputs, as the ratio between the sediments retained by the riparian land (service flow) and the total amount of incoming sediments in the absence of riparian land.

Figure 2.16 shows the estimation of the riparian land sediment removal efficiency (fraction) in the Danube river basin based on the SWAT model (Vigiak et al. 2016).

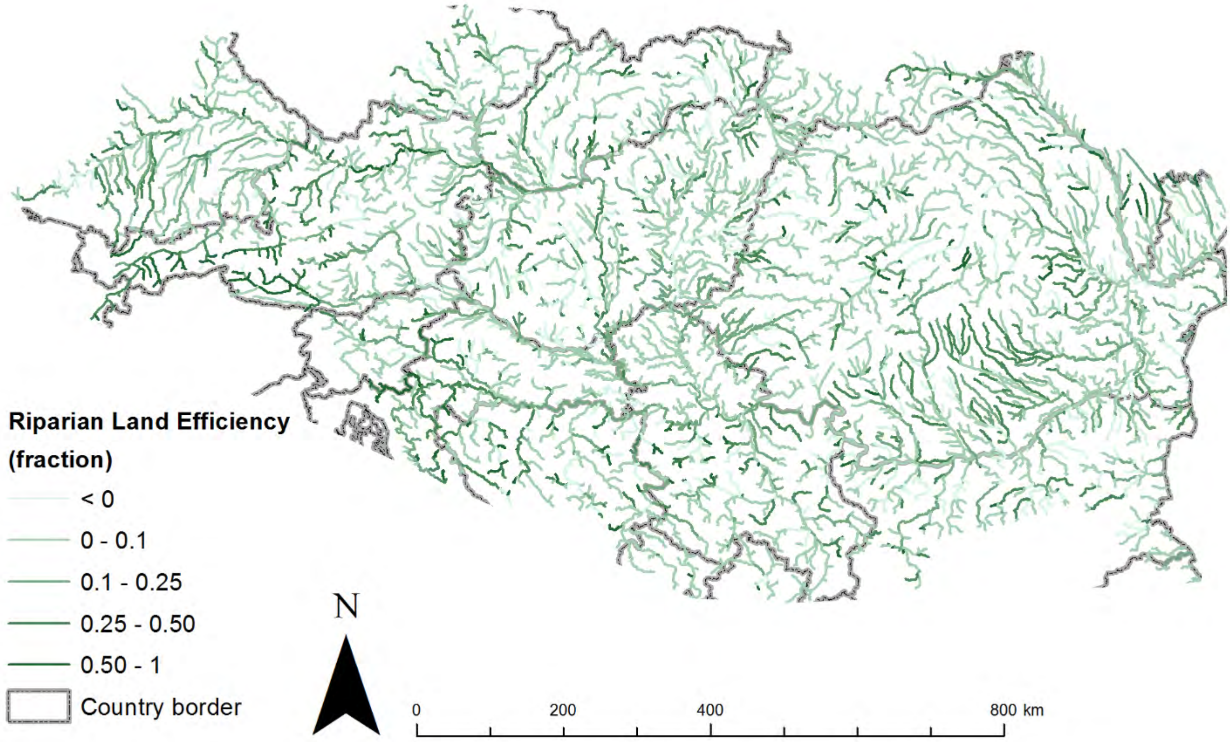


Figure 2.16 Efficiency of sediments removal (fraction) by riparian land in the Danube river basin based on the SWAT model (Vigiak et al. 2016). Data refers to annual means for the 15-year simulation period 1995-2009.

Main limitations

The availability of spatial data on the location and type of riparian vegetation is crucial to estimate the erosion mitigation in riparian areas. The recently published data from the project COPERNICUS at the European scale represent a major step forward; however these data do not cover smaller river stretches. A limitation of the proposed indicators of service flow and efficiency is related to the modelling of the erosion and hydrological processes, which is data and time demanding. There is a certain level of uncertainty in the representation of the processes of filtering and sediment transport in the models like SWAT. In addition, measurements of sediment needed for model calibration are often scarce, which increase the uncertainty of the predictions.

Key messages

Riparian lands provide multiple ecosystem services, including nursery habitat, pollution and sediment trapping, which contribute to the good ecological status of surface water. Maps of riparian land at the European scale have been recently made available by the project COPERNICUS. An analysis for the Danube river basin based on the SWAT model (Vigiak et al. 2016) has estimated that on average natural riparian lands retain 0.86 ton/km^2/yr of sediments. Sediment retention increases from lower to higher Strahler's order reaches (median values from 0.002 to 2.80 ton/km^2/yr) (Fig.

2.15). The filtering process was more efficient in lower Strahler's order reaches; median filtering efficiency decreased from 17% in reaches of Strahler's order 1 to 5% in reaches of Strahler's order larger than 3. At the same time, streambank protection service was important in reaches of lower Strahler order of 3 or more and in regions of high stream power, like in the Alps and in the Sava areas. The combination of filtering and streambank protection was such that riparian efficiency was basically independent of Strahler's order with an average efficiency of 8% reduction of sediments reaching the river system (Fig. 2.16).

2.1.5 Flood protection

The service

Flood protection is the service provided by floodplains that can store and slow down the water flow during floods events. Also coastal natural areas can act as buffer to protect against inundation from marine storms.

Selected indicators

Natural capacity

The capacity of the ecosystem to protect against floods is represented by the connected natural areas in floodplains, where water can expand and be stored, slowing down the high flow peaks.

A European assessment of the fraction of natural areas in floodplains is presented in Figure 2.10. The indicator is taken from the study of Pistocchi et al. (2015), which used the floodplains delineated by Weissteiner (2016) combined with the land cover data of the CLC 2012.

Service flow

The protection against floods is represented by the reduction of flood peak discharges. In a catchment, if the whole floodplain were left natural, the flood peak reduction can be quantified at a screening level on the basis of a simple model following Marone, 1971:

$$\eta = \frac{Qout}{Qin} = \max\left(0, 1 - \frac{Ah}{W}\right) \qquad \text{(Eq. 2)}$$

where Qout is the peak flood discharge of the hydrograph downstream of the floodplain and Qin the discharge of the hydrograph upstream the floodplain, A the area of the floodplain and h the average depth of water in the floodplain at the peak of discharge, while W is the flood hydrograph volume. We present here a tentative quantification of this indicator for Europe, using the results of flooding simulations from Alfieri et al. (2014). In particular, we use the flood hydrograph volume W as defined therein, as well as the flooded area extent A and the flood peak discharge Qout. The flood extent is mapped through Boolean map as grids with a resolution of 1 ha, where 1 means presence of flooding and 0 its absence. We assume that, in the absence of flooding, the flood peak discharge estimated by the authors would be higher and equal to Qin.

In order to compute the average depth h, we make use of Manning's formula in the form:

$$h = \left(\frac{n\ Qout}{B\sqrt{J}}\right)^{0.6} \qquad \text{(Eq. 3)}$$

where n is the roughness coefficient of the floodplain (that we assume equal to 0.1 s $m^{-1/3}$), B is the width of the floodplain orthogonal to the flood propagation direction, and J the slope of the floodplain.

We compute the indicator η with reference to 5-km long stretches of the main European stream network (corresponding to rivers with a drainage area of 500 km^2 or more). For each of these stretches, estimates of Qout are available following Alfieri et al. (2014). For this exercise, we carried out the analysis for floods with a return period of 200 years. We compute Thiessen polygons (nearest neighbour regions) for each 5-km stretch, and we consider the extent of flooded area within each Thiessen polygon, yielding parameter A in Equation 2. Dividing A by the length of the stretch (5 km) yields an estimate of the average width, B. Slope J is assigned to the river stretch as the average of values estimated at 1 km resolution for Europe, as described in Pistocchi et al. (2007). With these assumptions, we obtain the indicator η from Equation 2.

The flow peak attenuation (m^3/s) per catchment is computed as:

$$Attenuation = Q_{out} * \frac{1-\eta}{\eta} \qquad \text{(Eq. 4)}$$

This reduction of flood peak discharges would represent the maximum theoretical attenuation of the floodplain if all area were inundated. However, in Europe large parts of floodplains have been occupied by agricultural lands and urban areas. Flooding of a natural floodplain, in principle, does not entail any damage as this is an area structurally fit to allow this hydrological process. But in many floodplains, in order to protect settlements and, less frequently, agricultural land, flooding is prevented by man-made defences.

The actual service flow for flood protection can be described as the flood attenuation allowed only by floodplains in natural conditions in presence of a flood event. A first order quantification of this attenuation can be computed by using Equation 2 and Equation 4, where the total flooded area A is computed as the total areas of the floodplain minus the urban areas and part of agricultural areas (Attenuation URB). We used the Corine Land Cover (CLC)[6] (year 2012) to identify urban land and agricultural land in floodplains, considering the CLC classes "Artificial surfaces" (class 1) and "Agricultural areas" (class 2) respectively. We overlaid the information on land cover with the flood extent maps produced by Alfieri et al. (2014), considering floods with return time of 200 years and

[6] See http://land.copernicus.eu/pan-european/corine-land-cover

artificial flood defence of agricultural areas of 50%. With lower A, application of Equation 2 yields higher values of η, i.e. lower attenuation of the flood peaks (Fig. 2.17).

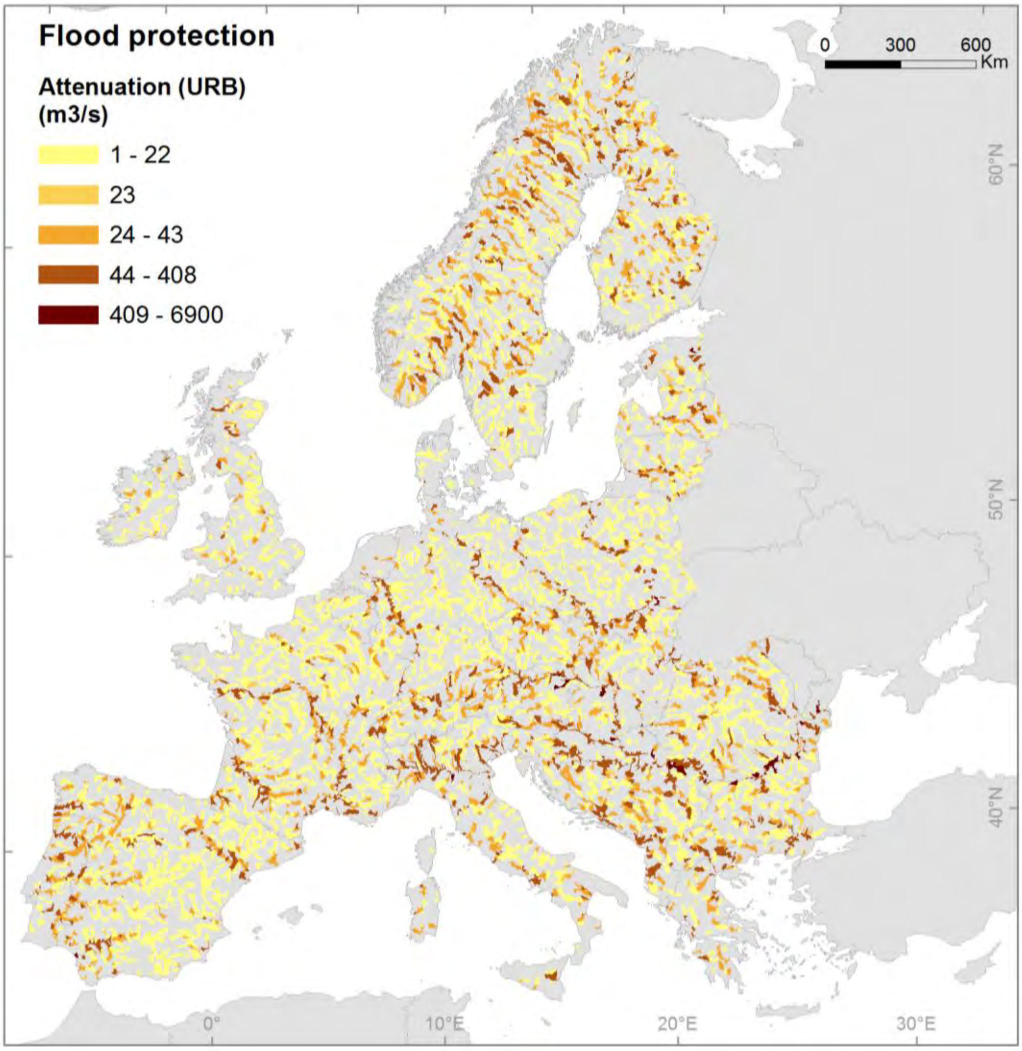


Figure 2.17 Flood attenuation considering that urban areas and half of agricultural areas are protected (Pistocchi et al. in prep. b).

Sustainability or efficiency

As an indicator of the efficiency of the ecosystem service of flood attenuation we computed the ration between the flood attenuation and the maximum flow peak (Fig. 2.18).

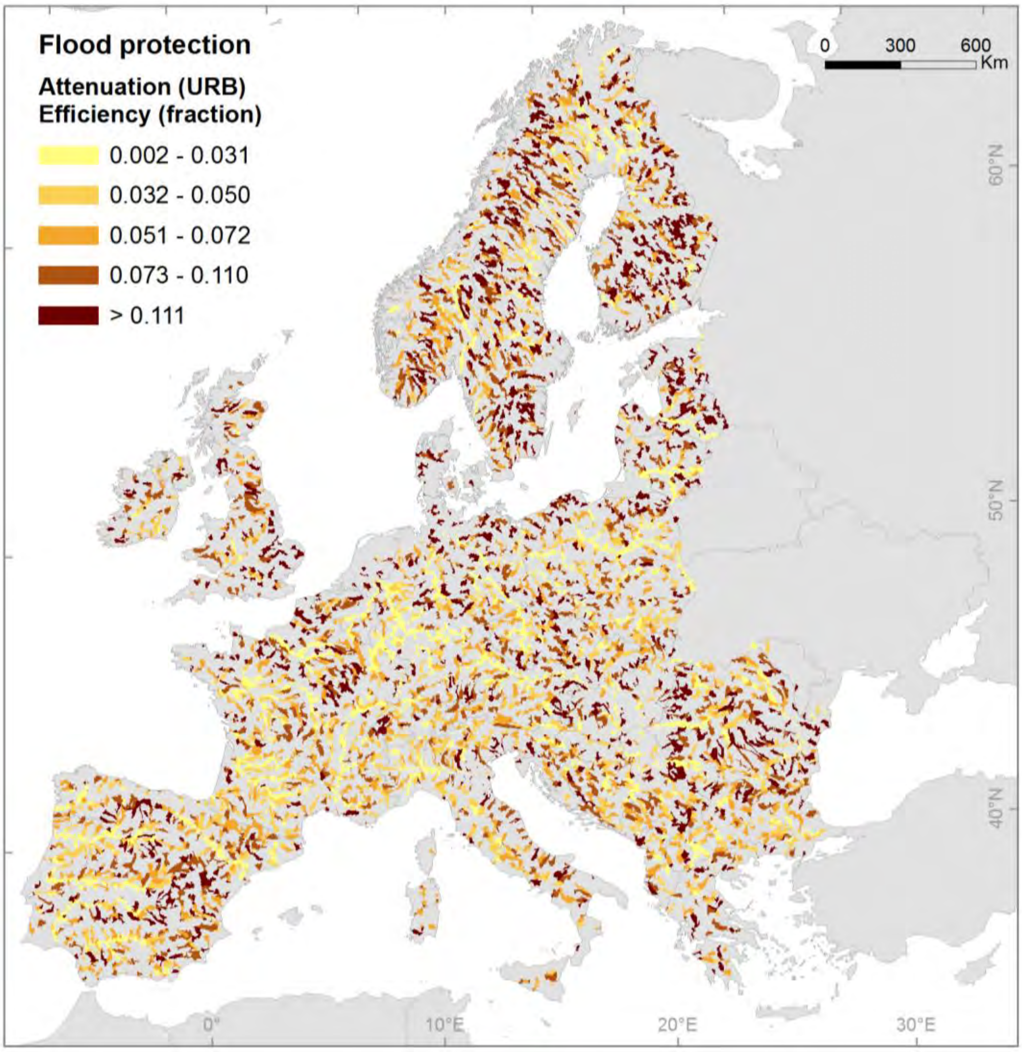


Figure 2.18 Efficiency of flood attenuation considering that urban areas and half of agricultural areas are protected.

Main limitations

The calculation is to be regarded as preliminary and indicative. The model of Equation (2) was developed by Marone (1971) with reference to reservoirs for the attenuation of floods. Although the author developed the model through numerical experiments that may be referred also to the case of floodplains, the assumptions made by the author are extremely simplified and are not necessarily realistic for many real world situations.

Key messages

The indicators of flood protection service proposed in this study, the attenuation of flow peak by natural areas of floodplains (Fig. 2.17), combines information on the floodplain capacity to store water and the volume of flood with a certain return time, taking into account the land cover change (specifically, the presence of artificial and agricultural lands). The assessment of flood protection highlights the widespread presence of this service across Europe, with higher rates in floodplains where flood risk is higher and natural vegetation prevails on urbanization.

2.1.6 Coastal protection

The service

Coastal protection is the role that ecosystems play in reducing the impacts of coastal hazards such as inundation and erosion from waves, storms surge or sea level rise. The service includes all habitat types but excludes human-made structures which cannot be considered as an ecosystem service.

Selected indicators

Liquete et al. 2013b developed specific indicators for the capacity, flow and benefit of this service in Europe, and those indicators where improved in Liquete et al. (2016b) and tested for the Euro-Mediterranean zone. Here, we are applying for the first time the last update of the coastal protection indicators for all EU-28. Data sources include hydrodynamic models or observations, habitats and land-use maps, and geographical and sociological characteristics. The study area is the coastal zone potentially affected by extreme hydrodynamic conditions (as defined in Liquete et al. 2013b). More information about these indicators can be found in the supplementary information of Liquete et al. (2016b).

Natural capacity

The indicator of coastal protection capacity (CPcap) represents the natural capacity that coastal ecosystems possess to attenuate waves and currents or to harden coasts (Fig. 2.19). The methodology estimates a protection score (i.e. the level of protection provided by each natural feature) of each morpho-sedimentological feature, seabed habitat and land cover type present in the coastal zone. CPcap integrates quantitatively data about coastal geomorphology, slope, and presence and distribution of both emerged and submerged habitats.

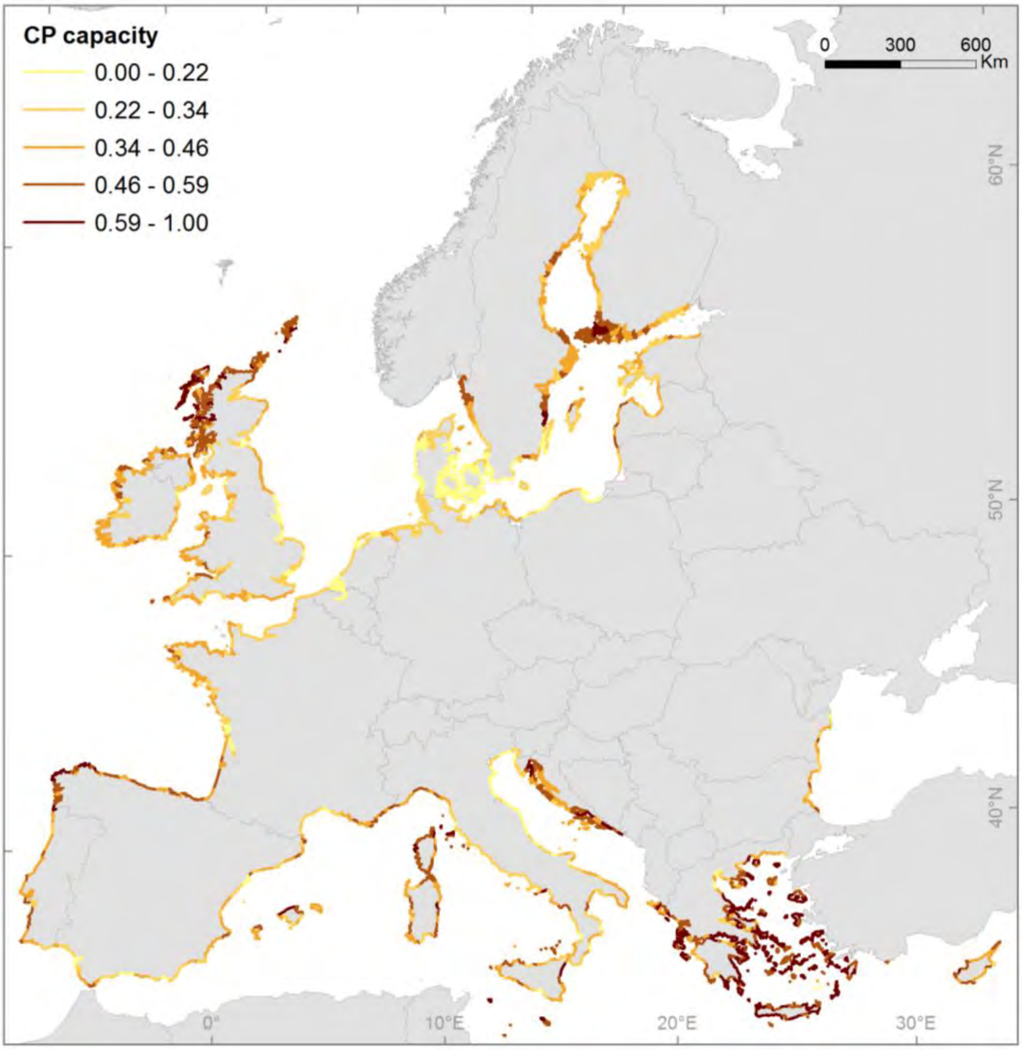


Figure 2.19 Coastal protection capacity, indicator of the natural capacity to provide protection for the year 2010.

Service flow

The level of supply of coastal protection (CPsup) integrates the abovementioned natural capacity with an indicator of exposure, estimating the excess of capacity over exposure (Fig. 2.20). The natural exposure of a coastal zone is based on the oceanographic conditions, namely wave regime, storm surge, tide and relative sea level rise. Values close to -1 point to deficient natural capacity for the existing oceanographic conditions while values close to +1 indicate enough capacity to deal with the natural exposure.

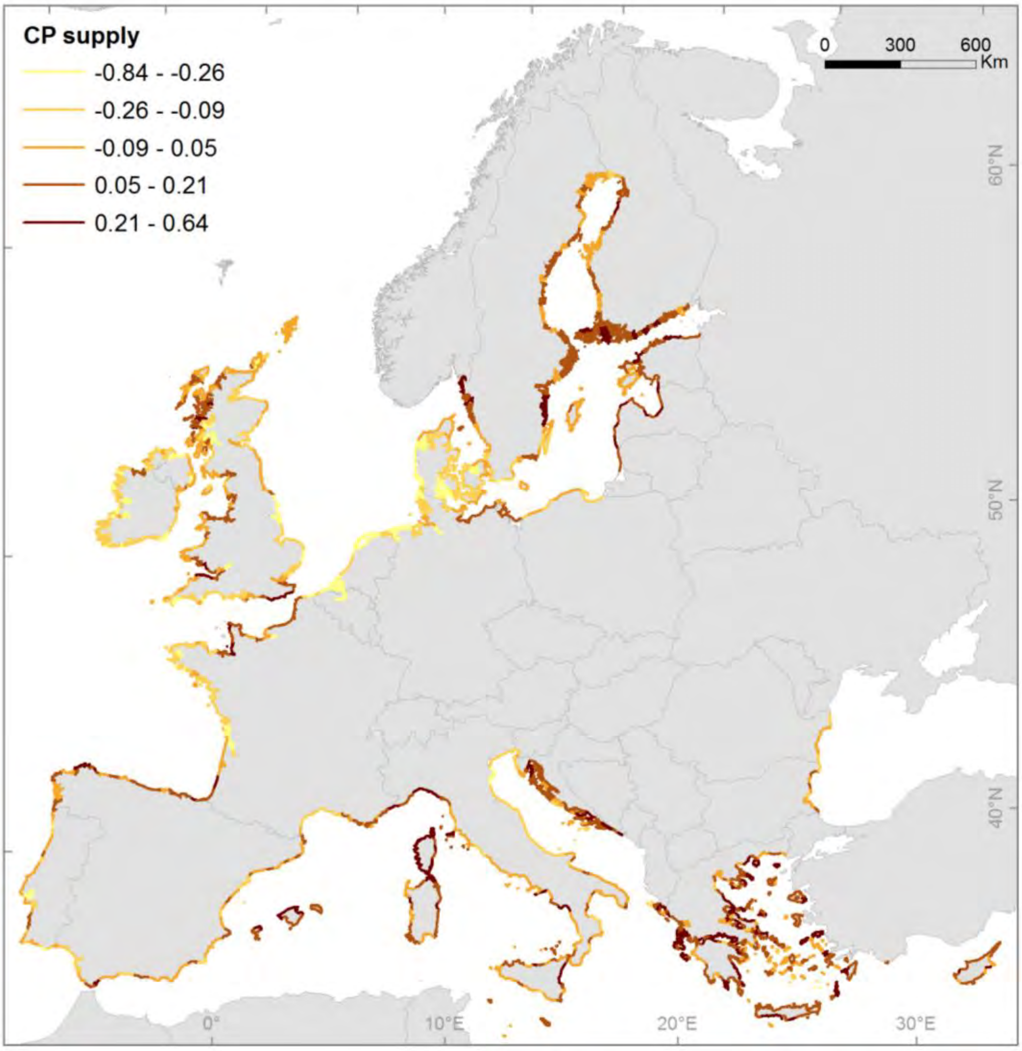


Figure 2.20 Coastal protection supply, indicator of the flow of coastal protection for the year 2010.

Benefit

The social benefit of this service can be reflected by the estimated human demand for protection in the coastal area (CPdem) (Fig. 2.21). This indicator is based on the presence of residents and assets in the coastal zone, in particular census of population, artificial surface and cultural sites (the latter with slightly less importance in the final calculation).

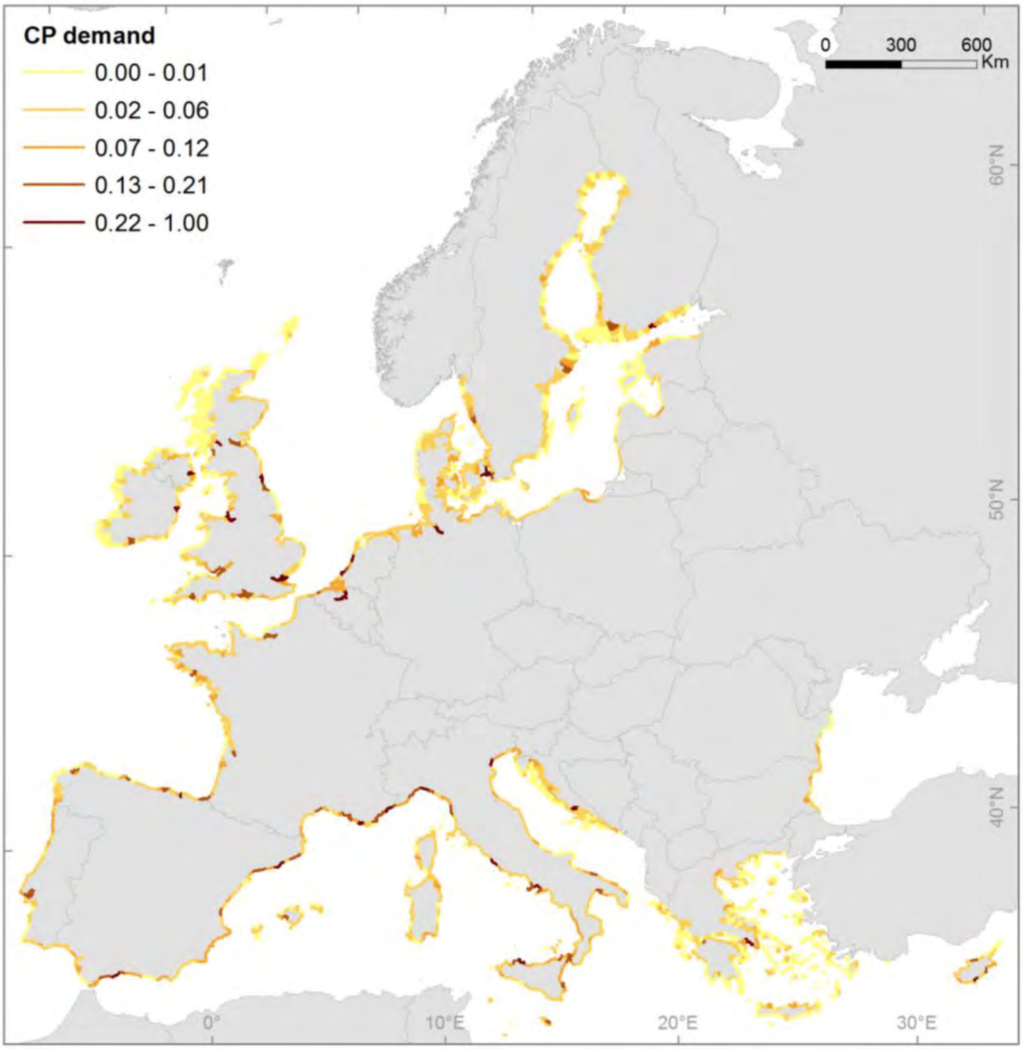


Figure 2.21 Coastal protection demand, indicator of the benefit from coastal protection for the year 2010.

Main limitations

The model-derived indicators about coastal protection show relative values, thus dependent on the study area. Although the input parameters have physical units, the final indicators must be interpreted as a ranking of coastal zones.

There are large data gaps to compute these indicators, especially in the aquatic systems (e.g. appropriate seabed habitat maps from the Central and Eastern Mediterranean Sea). Also, the information about some of the input parameters is static (e.g. again, seabed habitat maps), leading to temporal assumptions.

The magnitude and effects that different ecosystems have protecting the shoreline are highly context dependent. However, this large (continental) scale analysis cannot account for the local processes, namely local sediment budget (sand availability, beach stability, etc.); subsidence; main direction of morphologic features with respect to the wave action; coastal development and management; the local change of the relative weight of the variables; detailed and dynamic habitat mapping with specific non-linear responses; or dynamic adaptation capability of a coastal area.

Key message

The most negative values of CPsup, pointing to a potential unsustainable situation between the ecosystem service capacity and the exposure, concentrate in around the North Sea, NE Atlantic and N and W Adriatic Sea. We remind that human-made protection works (e.g. hard defence structures, designed flooding areas), specially developed in the shores of the North Sea, are not reflected this analysis.

The maximum average values of CPcap are present in Malta and Greece while those of CPsup are found in Latvia. Belgium shows both the maximum exposure and the maximum CPdem values, pointing to one of the riskiest contexts.

The capacity of natural habitats to reduce the impacts of coastal hazards should be analysed through time since it tends to decrease in recent decades driven by land use and shoreline changes (Liquete et al. 2016b). The possible decline of CPcap and CPsup combined with an expected growth of CPdem should be of concern for coastal communities.

2.1.7 Recreation

The service

Nature-based outdoor recreation concerns outdoor activities generating benefits in daily life, spanning from having a walk in the closest green urban area, to a short bike ride in a local natural park, to a day trip (<100 km travelling distance) with the sole purpose to experience nature. All ecosystems are considered to be potential providers of the recreation service, irrespective from their conservation status, though the range of provision changes according to ecosystem characteristics and people’s preferences and behaviour.

Selected indicators

Zulian et al. (2013) and Paracchini et al. (2014) developed a model to assess nature-based outdoor recreation in Europe. A recent update of such model (including improved parametrization and data sources) has been used here to extract the following indicators.

Natural capacity

The Recreation Potential Indicator (RPI) estimates the capacity of ecosystems and natural features to support nature-based recreation activities (Fig. 2.22). RPI integrates four components: suitability of land to support recreation (land use/land cover classes scored for recreation); natural features (presence and typology of natural protected areas, presence of grassland in agricultural areas); water (distance from water bodies and coast, presence of natural riparian areas); and presence of green urban areas.

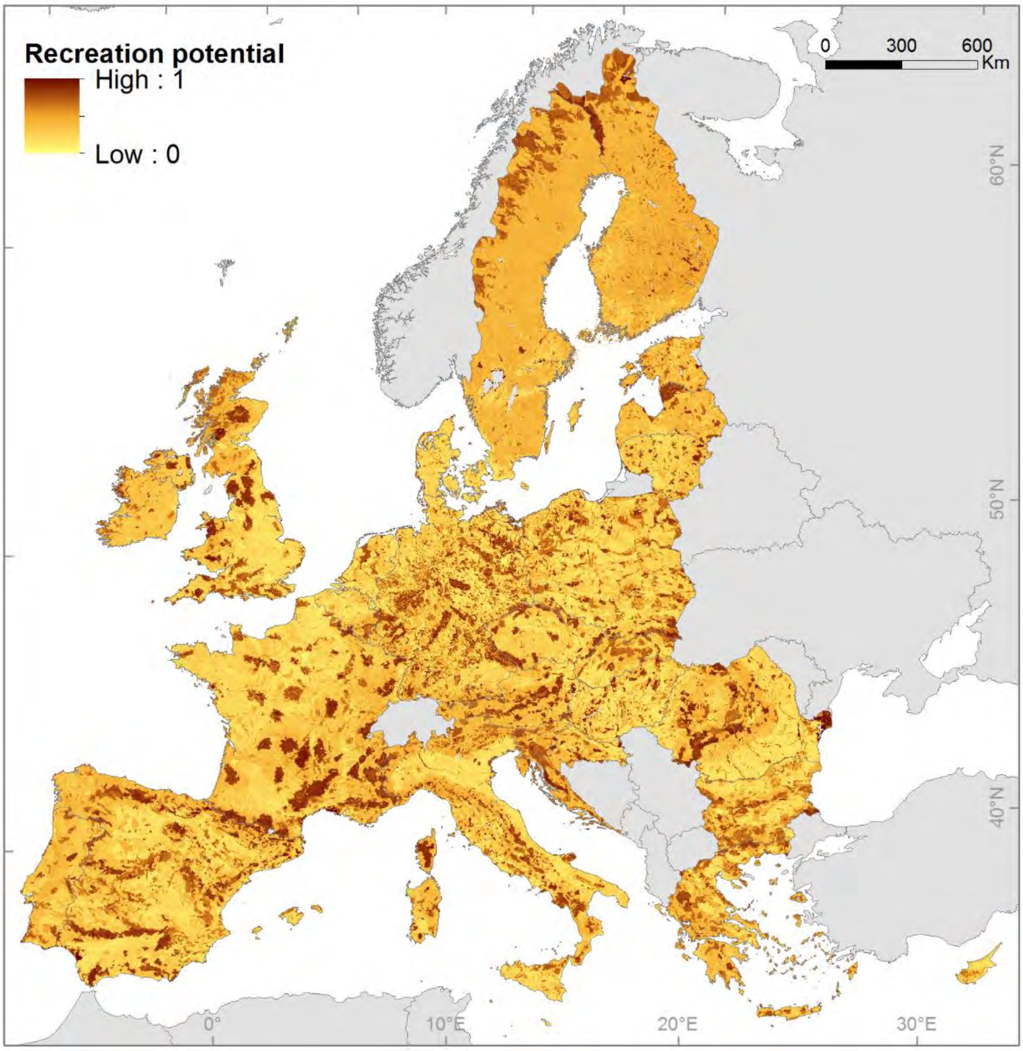


Figure 2.22 Recreation Potential, indicator of the capacity of ecosystems to provided nature-based recreation.

Service flow

The Recreation Opportunity Spectrum (ROS) integrates qualitatively RPI with a zoning map of Europe in terms of remoteness (distance from residential areas) and proximity (distance from roads) (Fig. 2.23). Still, this indicator is not able to account for the actual visitors' flow, since that information is not available at continental scale. ROS values are divided into 9 qualitative classes (from 1 to 9) which combine different levels of service provision and remoteness.

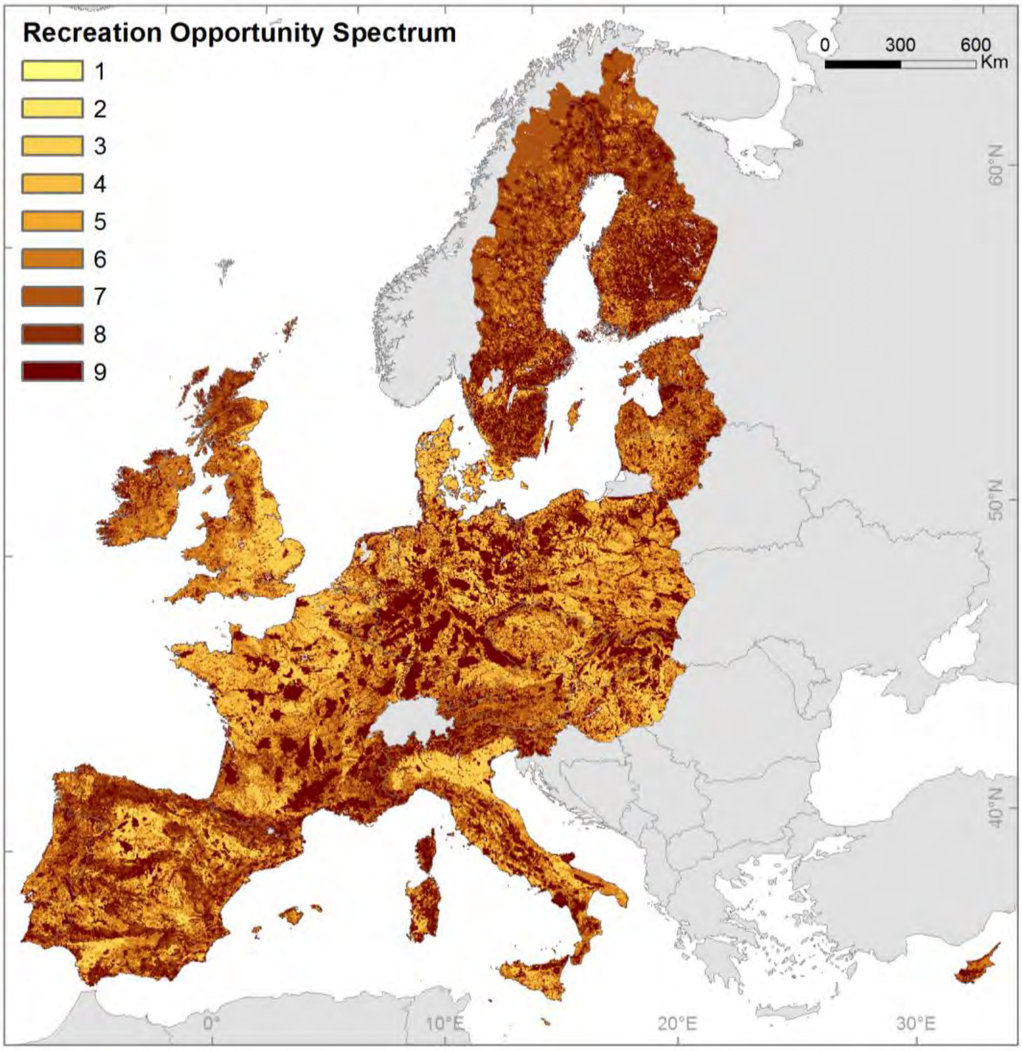


Figure 2.23 Recreation Opportunity Spectrum, indicative of the actual flow of nature-based recreation. ROS values: 1= Low provision not easily accessible, 2= Low provision accessible, 3= Low provision easily accessible, 4= Medium provision not easily accessible, 5= Medium provision accessible, 6= Medium provision easily accessible, 7= High provision not accessible, 8= High provision accessible, 9= High provision easily accessible.

Benefit

As a proxy of demand, Zulian et al. (2013) and Paracchini et al. (2014) propose the potential trips to areas with different levels of service. Hence, the distribution of potential demand for local recreation estimates the share of population that can theoretically access the different ROS zones, according to a spatial impedance function.

Main limitations

The indicator of natural capacity to support recreation activities does not include the ecosystem condition, e.g. people is supposed to be attracted by the presence of a lake, but the water quality is not included as an influential factor.

In this case, the indicator of service flow estimates a potential use of the service (i.e. potential flow of visitors). The actual number of visitors is not available at the scale of this analysis.

Key message

The outdoor recreation is measured here in terms of extent and quality of citizens' access to nature, considering all ecosystems as potential providers of the service but highlighting the attraction of water bodies for recreation. Mapping of the Recreation Potential illustrates that the service capacity in Europe is relatively high. According to Paracchini et al. (2014), almost half of the territory is classified in the highest classes of recreation provision, but the spatial distribution of such potential is uneven. Based on the new results presented in this report, all EU countries show an average RP below 0.4, with Slovenia and Croatia getting the maximum values.

The natural capacity transforms into a service flow when people can reach sites for outdoor recreation, reflected by ROS. Again, the service flow in Europe is relatively high, with 33% of the territory under "High provision easily accessible" and 10% under "High provision accessible". We must note that only 5% of Europe is not easily accessible in this analysis.

3. Relationship between ecosystem services and ecological status

Since 2011, the Convention on Biological Diversity (through the Aichi biodiversity targets) and the EU Biodiversity Strategy have adopted the ecosystem services approach to protect biodiversity. However, understanding the relationship between the ecosystem functioning, integrity, biodiversity, and the delivery of ecosystem services is still an impellent research question (Liquete et al. 2016c). Indeed, although there are numerous evidences supporting a positive relationship between biodiversity, ecosystem functions, and the delivery of ecosystem services (Egoh et al. 2009, Cardinale 2011, Isbell et al. 2011, Mace et al. 2012, Harrison et al. 2014), there is not much consensus on what the links are and how they operate (Loreau et al. 2001, Harrison et al. 2014). In particular, studies at the large scale are not available.

In this study we explored these links for European aquatic ecosystems, analysing whether a better ecological status supports higher delivery of ecosystem services and when ecosystem services instead coincide with pressures.

3.1 Working hypothesis

According to the CICES classification (v4.3) followed in this study, ecosystem services are classified into three broad types: provisioning, regulating and cultural services. For aquatic ecosystems we might expect provisioning ecosystem services to act as pressures, since they involve the extraction of products like water or fish from the ecosystem (i.e. water provisioning involves water abstraction, fish provisioning entails fish catch), implying that the higher is the provision of the service the higher is the impact on the ecosystem. On the contrary we might expect regulating services, such as climate regulation, water purification and pollination, to be enhanced in healthy ecosystems, with more service level provided by good ecosystem functioning. In the case of cultural services the relationship between ecosystem services and conditions may not be straightforward. For example, the service of recreation is supported by the beauty of the natural landscape or the quality of bathing waters, but also by the presence of infrastructures and the site accessibility, and at high rates the service use contributes to the degradation of the ecosystem, due to pollution or habitat destruction.

If we consider the ecological status as an indicator of ecosystem functioning, integrity and biodiversity for aquatic ecosystems, the relationship between ecosystem services and ecological status might be the following:

- *Provisioning* services are expected to have a negative relation with the ecological status;
- *Regulating* services are expected to have a positive relation with the ecological status;
- *Cultural* services are expected to have a positive relation with the ecological status but only to a certain limit;

The expected relationship between ecosystem services and ecological status in aquatic ecosystem is shown in Figure 3.1. This relationship might hold when considering indicators of the *flow* of the services (the actual use of the service).

Differently, for indicators of *capacity* and *efficiency* of the services (the potential of the ecosystem to provide the service and the efficiency of the process, respectively) we expect a positive relationship with the ecological status, to indicate that good ecosystem functioning, high level of integrity and biodiversity support the capacity and the efficiency of the ecosystem to provide services. On the contrary, proxies of service demand or *benefit* should be linked to densely populated areas and thus to more degraded ecosystems.

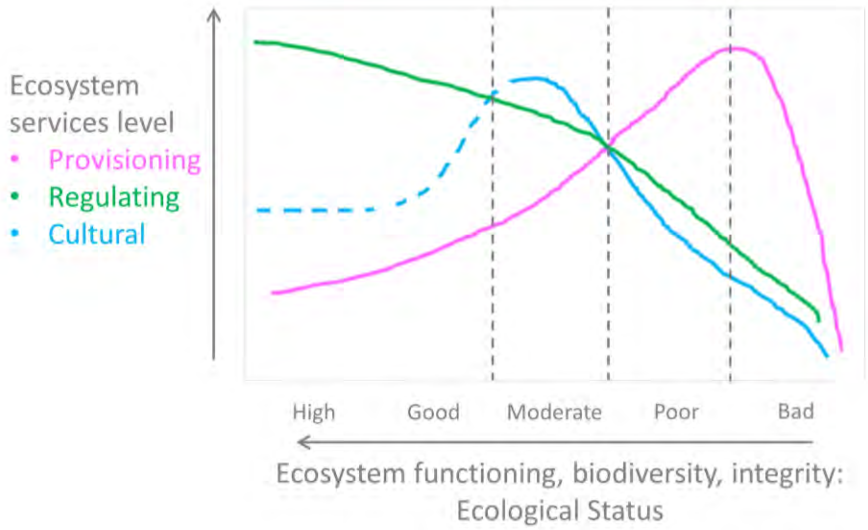


Figure 3.1 Expected relationship between the level of ecosystem services (flow) and ecological status in aquatic ecosystems. Modified from Kandziora et al. (2013).

3.2 Analysis

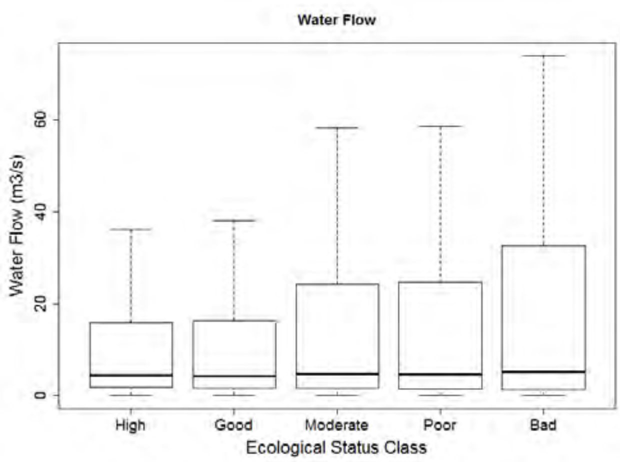
We tested this hypothesis at the European scale, using the ecosystem services quantified in this study (presented in Section 2) and the data on ecological status reported by EU Member States under the Water Framework Directive first reporting period (2004-2009).

The ecological status is an integrative measure of the condition of the water body based on assessment methods for biological quality elements (BQEs, that are phytoplankton, flora, invertebrate fauna and fish fauna), combined with information on physico-chemical and hydromorphological conditions. The ecological status is defined in five categories: high, good, moderate, poor and bad.

For this study, only the coordinates of the centroid of each water body were available, but not the water body geographical delineation at the European scale. To overcome this lack of information, we used a proxy indicator of the ecological status that could be representative at the scale of the assessment (Grizzetti et al. submitted). We computed the most frequent class (mode) of ecological status for catchments (average size 180 km^2), coastal spatial units (average coastal length 30 km) and lakes (geographical information from Ecrins).

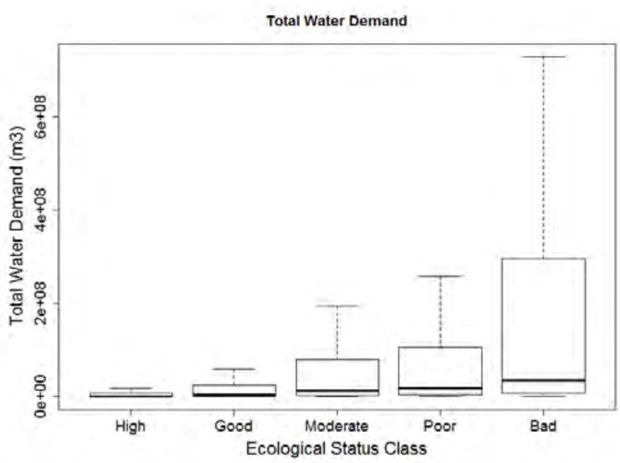
We considered the services: water provisioning, water purification, erosion mitigation, flood protection, coastal protection and recreation (Section 2). Importantly, for each service we included the indicators of capacity, flow, sustainability/efficiency, and benefit, to distinguish the information provided. We could not analyse food provisioning as data on this service were available only at the national scale.

3.2.1 Water provisioning



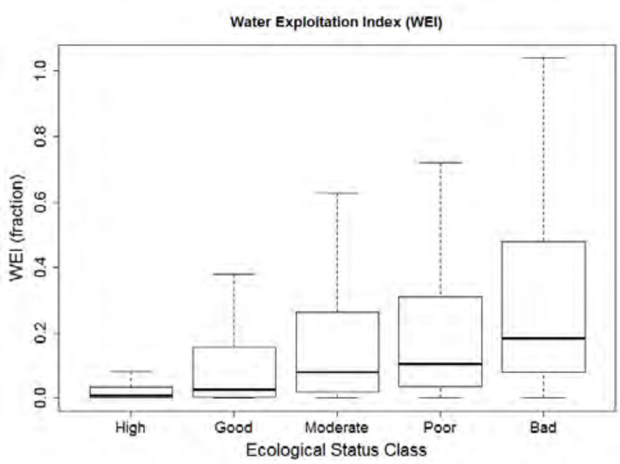
a. Water provisioning CAPACITY

(Kruskal-Wallis rank test $p<0.05$)



b. Water provisioning FLOW

(Kruskal-Wallis rank test $p<0.05$)

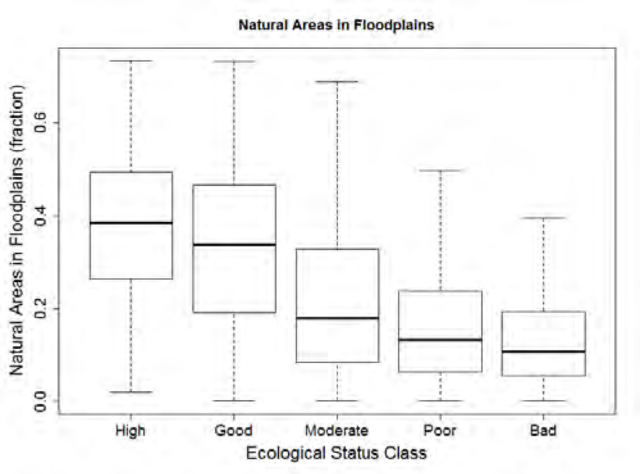


c. Water provisioning SUSTAINABILIY

(Kruskal-Wallis rank test $p<0.05$)

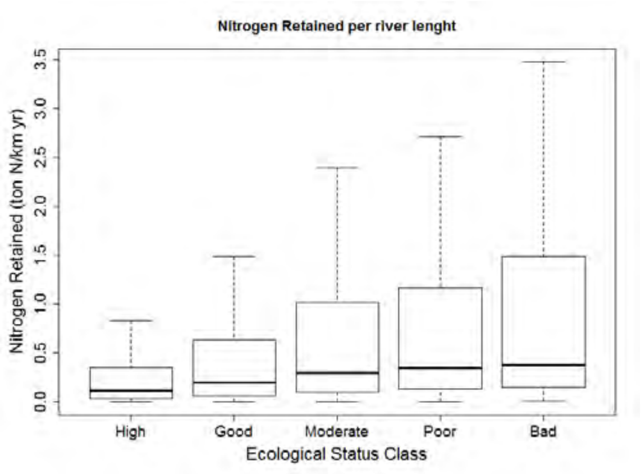
Figure 3.2 Relationship between the indicators of ecosystem service water provisioning and the proxy of the ecological status for European rivers.

3.2.2 Water purification



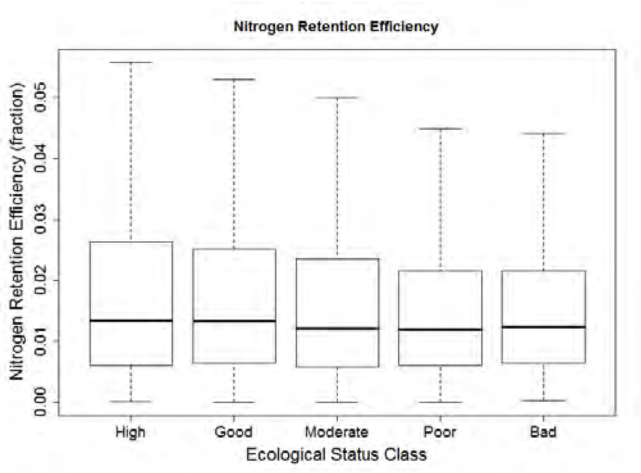
a. Water purification CAPACITY

(Kruskal-Wallis rank test p<0.05)



b. Water purification FLOW

(Kruskal-Wallis rank test p<0.05)

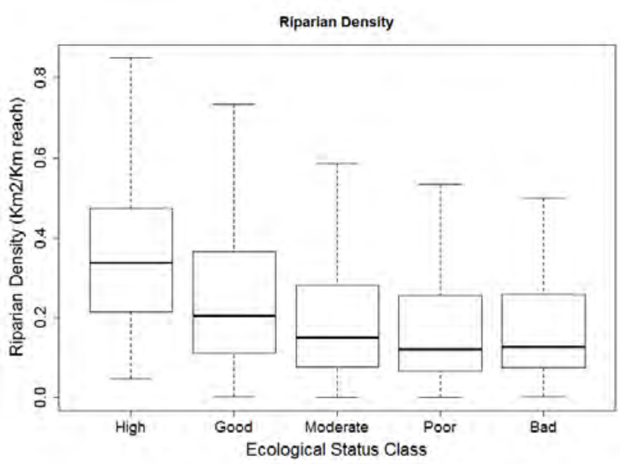


c. Water purification EFFICIENCY

(Kruskal-Wallis rank test p<0.05)

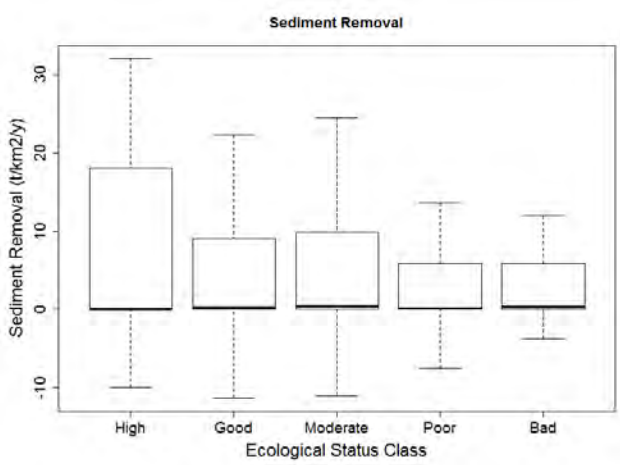
Figure 3.3. Relationship between the indicators of ecosystem service water purification and the proxy of the ecological status for European rivers.

3.2.3 Erosion prevention



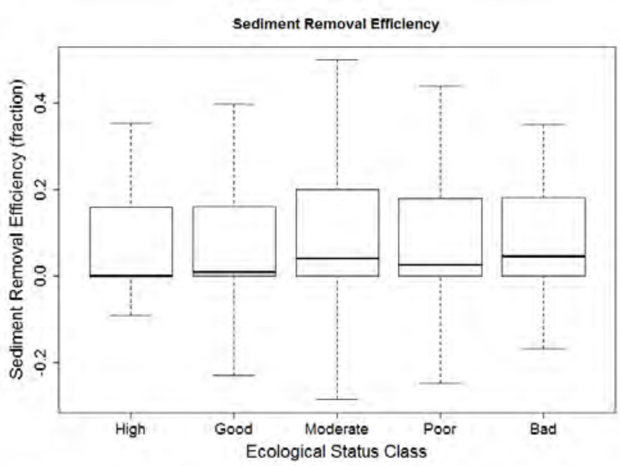
a. Sediment mitigation CAPACITY

(Kruskal-Wallis rank test $p<0.05$)



b. Sediment mitigation FLOW

(Kruskal-Wallis rank test $p=0.1054$)

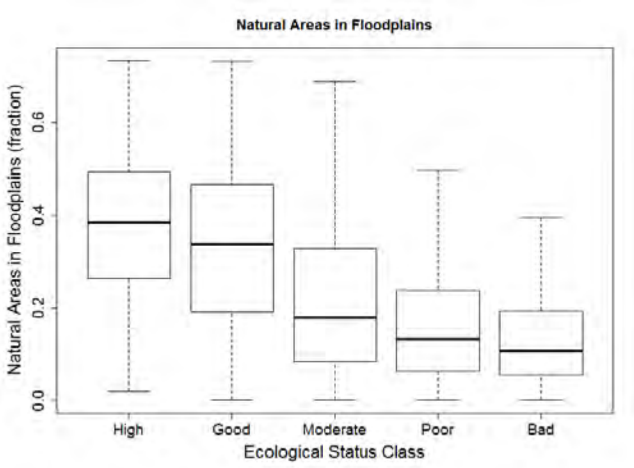


c. Sediment mitigation EFFICIENCY

(Kruskal-Wallis rank test $p=0.1692$)

Figure 3.4 Relationship between the indicators of ecosystem service sediment mitigation and the proxy of the ecological status for rivers in the Danube river basin.

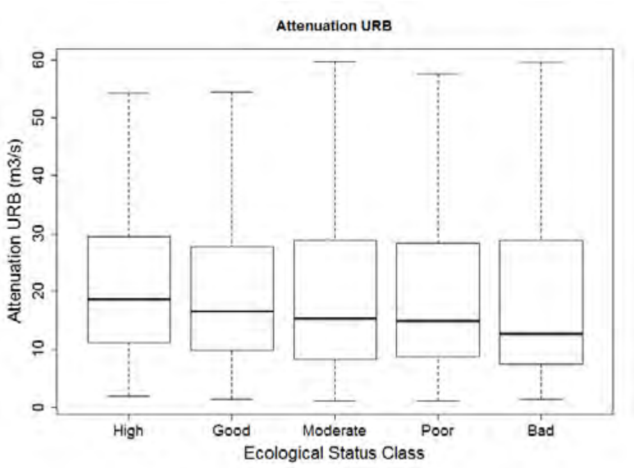
3.2.4 Flood protection



a. Flood protection CAPACITY

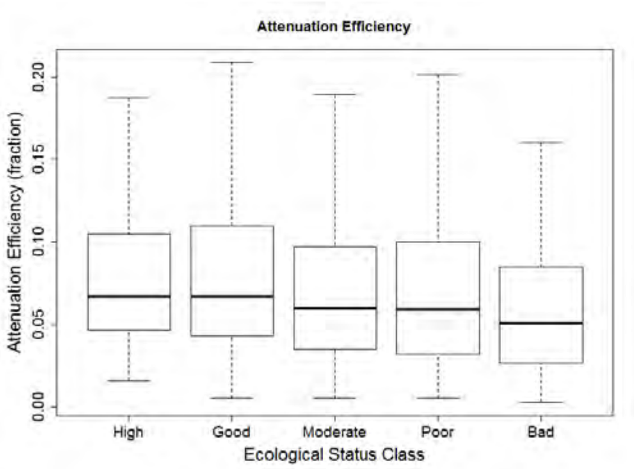
(Kruskal-Wallis rank test $p<0.05$)

**Same indicator as in Figure 3.3a*



b. Flood protection FLOW

(Kruskal-Wallis rank test $p<0.05$)

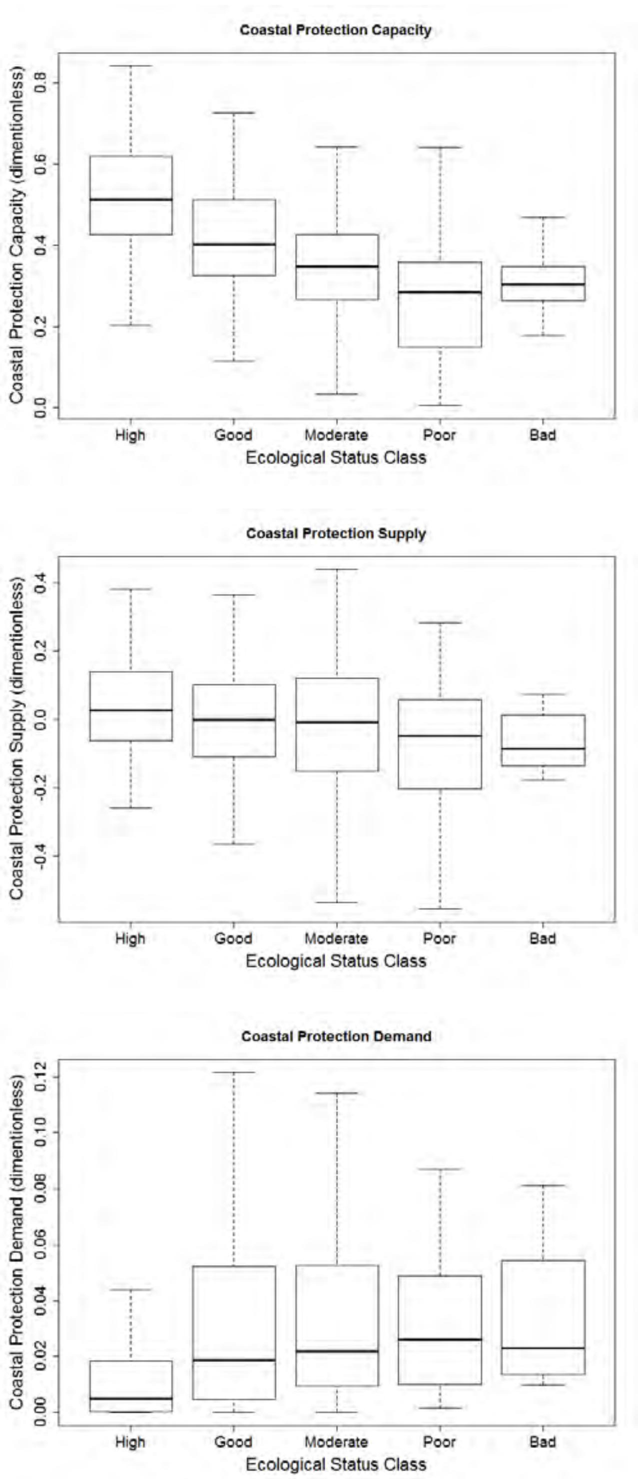


c. Flood protection EFFICIENCY

(Kruskal-Wallis rank test $p<0.05$)

Figure 3.5 Relationship between the indicators of ecosystem service flood protection and the proxy of the ecological status for European rivers.

3.2.5 Coastal protection



a. Coastal protection CAPACITY

(Kruskal-Wallis rank test $p<0.05$)

b. Coastal protection FLOW

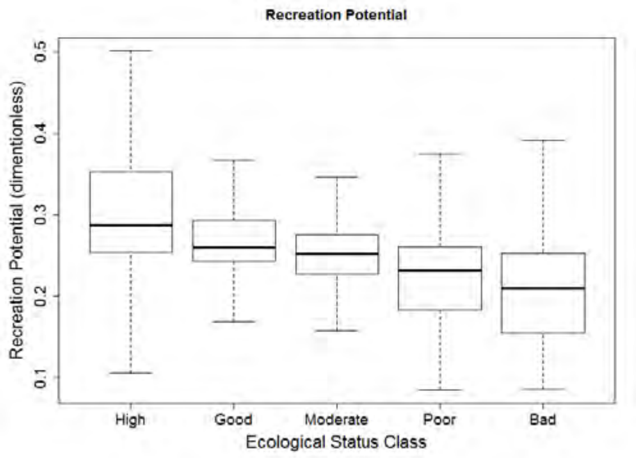
(Kruskal-Wallis rank test $p<0.05$)

c. Coastal protection BENEFIT

(Kruskal-Wallis rank test $p<0.05$)

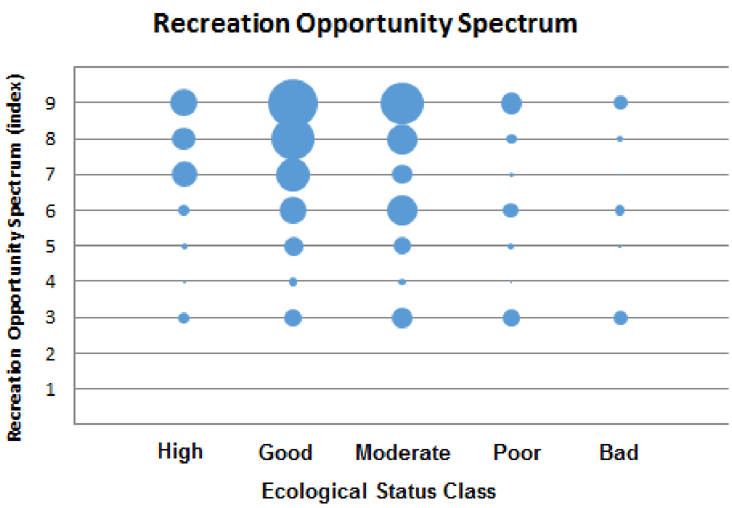
Figure 3.6 Relationship between the indicators of ecosystem service coastal protection and the proxy of the ecological status for European coastal waters.

3.2.6 Recreation



a. Recreation CAPACITY

(Kruskal-Wallis rank test $p<0.05$)



b. Recreation FLOW

Figure 3.7 Relationship between the indicators of ecosystem service recreation and the proxy of the ecological status for European lakes.

3.2.7 Key messages

We performed a coherent analysis of the relationship between ecological status and ecosystem services of aquatic ecosystem at the European scale, considering indicators of ecosystem services that distinguish between the capacity, flow, efficiency/sustainability and benefit of the services. The expected and observed relationships are summarised in Table 3.1.

The results showed that the ecosystem capacity to provide the services is always positively correlated to the ecological status, except for water provisioning, which however strongly depends on the climatic and hydrographic characteristics of the river basin, more than on the conditions of water bodies. Indeed, water provisioning is less correlated to biodiversity compared to other ecosystem services (Harrison et al. 2014). In addition, in the present analysis we only considered the water quantity without taking into account the water quality (which is also mediated by ecosystem processes) that is required by the different uses.

From the analysis we observe that provisioning services (flow indicator) act as pressures for the aquatic ecosystems, i.e. their increase degrades the ecosystem functioning. Similarly, benefits/demand decrease with improved ecological status, mainly due to the population density (see for example coastal protection). On the contrary regulating services increase with better ecological status, both their actual use (flow indicator) and their efficiency (efficiency indicator). The flood protection and coastal protection are clear examples. However, in the case of water purification the indicator nitrogen retention is related to human input of nitrogen pollution to rivers and lakes. The more nitrogen is discharged to the water bodies the more nitrogen is removed by the ecosystem, but the efficiency of the service (efficiency indicator) decreases with lower ecosystem conditions. For cultural services, our analysis shows that recreation is higher where lakes are in better ecological status, with a change in behaviour already starting for lakes in moderate status. (This is in line with the findings by the economic assessment presented in Section 4).

Overall, our results indicate that the ecosystem services are mostly positively correlated with the ecological status of European water bodies. We also shed light on the role of provisioning services, distinguishing between indicators that describe their action as pressures (flow), and those describing the capacity and sustainability of the service.

Table 3.1 Summary of relationships between ecosystem services provided by European aquatic ecosystems (rivers, lakes and coastal waters) and their ecological status observed in this study. In blue within brackets we report the expected type of relationship (see explanation in Section 3.1). Legend: (↗) indicates a positive relationship; (↘) indicates a negative relationship; (*) indicates that the relationship was not significant.

	Ecosystem Service Indicators			
	Capacity	Flow	Efficiency or Sustainability	Benefit
Provisioning				
Water provisioning	(↗) ↘	(↘) ↘	(↗)↘	
Regulating				
Water purification	(↗) ↗	(↗) ↘	(↗) ↗	
Sediment mitigation	(↗) ↗	(↗) *	(↗) *	
Flood protection	(↗) ↗	(↗) ↗	(↗) ↗	
Coastal protection	(↗) ↗	(↗) ↗		(↘) ↘
Cultural				
Recreation	(↗) ↗	(↗) ↗		(↘)

3.3 Impact of pressures on ecosystem services

3.3.1 Expected effect of changes in pressures

Considering the relationship between pressures and status (described in Grizzetti et al. submitted and MARS Deliverable D5.1-1), and the relationship between ecosystem services and ecological status (presented in Section 3.2), we can infer on the expected effects of changes in pressures on the delivery of ecosystem services. The possible effects are described in Table 3.2.

*Table 3.2 Expected effect of an increase of the main pressures on ecosystem services of aquatic ecosystems considered in this study. The main pressures were described in Table 2.1 of the MARS deliverable D2.1-2. Legend: (↗) indicates an increase; (↘) indicates a decrease. *(urbanization, channels, reservoirs, changes in habitat)*

	Ecosystem services	Increase in water abstractions	Increase in water pollution	Increase in morphological changes*	Increase in fishing and alien species
Provisioning	Food provisioning (fisheries and aquaculture)	↘	↘	↘	↗↘
	Water provisioning for drinking	↗	↘	↗	
	Water provisioning for non-drinking	↗		↗	
Regulation	Water purification		↗	↘	
	Erosion prevention			↘	
	Flood protection			↘	
	Coastal protection			↘	
Cultural	Recreation		↘	↘	↗↘

3.3.2 MARS scenarios

The present analysis is based on the data available in the MARS project consortium at 1st January 2017. The complete assessment of scenarios by modelling will be developed by the Work Package 7 by the end of the project. In that context the indicators proposed in this report for water provisioning and water purification can be computed.

The project MARS has developed three scenarios storylines. Starting from the indicators of ecosystem services presented in this study for Europe, we considered their expected change under the three MARS scenarios (Table 3.3), on the basis of the description of the relative changes of relevant elements described in the MARS stakeholders workshop on scenario (MARS Deliverable 2.1 Part 4, Feneca Sanchez et al. 2015).

*Table 3.3 Expected effects on ecosystem services (marked by arrows within brackets) under the three MARS scenarios, considering the relative changes of relevant elements (marked as + and - including different intensities) selected from Feneca Sanchez et al. 2015 (MARS Deliverable 2.1 Part 4 Table 6). Legend: (↑) indicates an increase; (↓) indicates a decrease. *Values reported in the workshop that seem partially contradictory.*

Ecosystem service	**Element change in the scenario** Legend: + increase; - decrease;	**Expected change in the ecosystem services** Legend: (↑) increase; (↓) decrease; (−) no change		
		Techno World MARS ad hoc World	**Consensus World MARS World**	**Survival of the fittest No MARS World**
Coastal protection (flow)	Protection of coastal zones	+ (↑)	+++ (↑)	--- (↓)
Flood protection (capacity)	Preservation of natural areas	+ (↑)	+++ (↑)	--- (↓)
Flood protection (flow)	Natural flood retention	+* (↑)	++ (↑)	--- (↓)
Water purification (capacity) & erosion prevention (capacity)	Restoration of riparian zones	- (↓)	++ (↑)	--- (↓)
Water purification (capacity) & erosion prevention (capacity)	Loss of riparian zones in favor of touristic areas, agriculture, etc.	+ (↓)	0 (−)	+++ (↓)
Fish provisioning (capacity)	Habitat loss	++ (↓)	+ (↓)	+++ (↓)
Flood protection (flow)	Urbanisation	+++ (↓)	++ (↓)	+++ (↓)
	Deforestation	++	+	+++
	Agricultural areas for crops	-	0	--
Erosion prevention (efficiency)	Sediments in water due to erosion	++ (↓)	+ (↓)	+++ (↓)
Erosion prevention (efficiency)	Use of crops to prevent erosion	0 (↓)	++ (↑)	--- (↓)
Water purification (efficiency)	Nutrient load	++ (↓)	+ (↓)	+++ (↓)
Nitrogen retention (efficiency)	Use of fertilisers	+*(↓)	++ *(↓)	+++ (↓)
	Water treatment plants	++	++	+
Water provisioning for non-drinking (flow)	Efficient irrigation	++ (↑)	++ (↑)	---
Fish provisioning (capacity)	Environmental flow needs covered	+ (↑)	++ (↑)	--- (↓)
Water provisioning (flow)	Overexplotation of water resources	++ (↑)	+ (↑)	+++ (↑)

4. Economic valuation of ecosystem services provided by European lakes

Environmental economists have been often unable to provide ecosystem services values at a large scale and with a high resolution. The main obstacle is that these values have to be measured in small scale and usually time-consuming case studies. The value of ecosystem services provided by lakes is examined in this chapter using a meta-analysis on a database including 35 lakes from 12 different European countries. Based on that database, we conduct a benefit transfer to value ecosystem services provided by lakes in Europe and we produce several maps of economic value of lakes at the European scale. We also discuss how the European scale mapping may help public authorities in charge of water management. There are two categories of benefit transfer: (a) value transfer, which applies a single value from a similar and previously valued site (study site) for sites for which values are to be estimated (policy sites); and (b) function transfer, which uses a valuation function to calibrate the value being transferred from the study sites according to specific physical and demographic characteristics of the policy sites. Here, we use the second approach.

The aggregated benefits derived from European lakes are relatively high and variable spatially. We estimate both the biophysical potential of a lake to deliver ecosystem services, and the flow of services consumed and valued by the potential beneficiaries of these services. Our approach is also driven by a specific policy question: will people in EU benefit from an improvement of the ecological status? Thus, we consider ecological status in our approach, first, because ecosystem integrity is supposed to underpin most of the ecological functions that drive ecosystem services' supply, and second, because ecological status is of primary interest for the EU Water Framework Directive. We demonstrate that the ecological status of lake has an impact on valuation. Using this benefit transfer approach, the expected benefit from restoring all European lakes into at least a moderate ecological status are estimated to be 5.9 billion EUR per year, which corresponds to 11.7 EUR per person and per year.

4.1 Developing a meta-model to value ecosystem services provided by lakes

4.1.1 Meta-database

We performed a systematic review selecting all the scientific references that included the keywords "Lake" AND ("Valuation" OR "Value" OR "Willingness to pay" OR "WTP" OR "Stated preferences") on:

- academic search engines (namely Scopus, Science Direct, Wiley, Web of knowledge, RepEc and AgEconSearch);
- databases specialized in environmental valuation (namely Environmental Valuation Reference Inventory, the Nordic Environmental Valuation Database and the Greek Environmental Valuation Database); and

- general search engines to look for "grey literature" (namely Google Scholar and Science.gov).

We limited our search to lakes located in European countries with reported or estimated ecological status. The reported ecological status from EU Member States come from the WISE2 database[7] and corresponds to article 13 of the EU Water Framework Directive (WFD, Directive 2000/60/EC). According to the WFD, the ecological status is ranked in five classes: High-Good-Moderate-Poor-Bad, based on biological quality elements, physico-chemical characteristics, hydrological and morphological conditions. For European countries outside EU or without a reported ecological status, an estimated ecological status was derived from scientific literature with support from the original authors of the publications.

The resulting meta-database has 107 observations from 16 publications related to 35 lakes from 12 European countries[8] (see Fig. 4.1 and Table 4.1). Each observation corresponds to an estimated economic value linked to a lake ecosystem service. The list of ecosystem services provided by lakes has been derived from the reviewed publications. Compared with other established classifications of ecosystem services (such as TEEB[9] or CICES[10]), our list details more the recreational activities. Thus, the meta-database gathers economic values for 8 ecosystem services provided by lakes: 1 provisioning (drinking water), 1 regulating (maintaining populations and habitats) and 6 cultural services (boating, sightseeing, recreational fishing, swimming, unspecified recreational and symbolic appreciation/cultural heritage).

Different valuation methods were used in the scientific references, including contingent valuation (CV), choice experiment (CE), hedonic prices (HP) and travel costs (TC). This raised the issue of consistency in welfare measures across primary studies (Johnston and Rosenberger 2010) which requires that welfare measures represent the same theoretical concept (Smith and Pattanayak 2002). As discussed in Kuminoff and Pope (2014), HP studies measure values capitalized in property values. These are very difficult to relate to utility-theoretic welfare measures without making strong assumptions on market structures (e.g. homeowner flexibility to move), which are hardly supported when working at large scales. For welfare consistency reasons we excluded three HP studies from the meta-analysis.

[7] http://www.eea.europa.eu/data-and-maps/data/wise_wfd
[8] Even if the number of observations may appear relatively low, it is similar to those found in previous meta-analysis on valuation of ecosystem services (e.g. Brouwer et al. 1999).
[9] http://www.teebweb.org/resources/ecosystem-services/
[10] http://cices.eu/

Table 4.1. Extract from the meta-database.

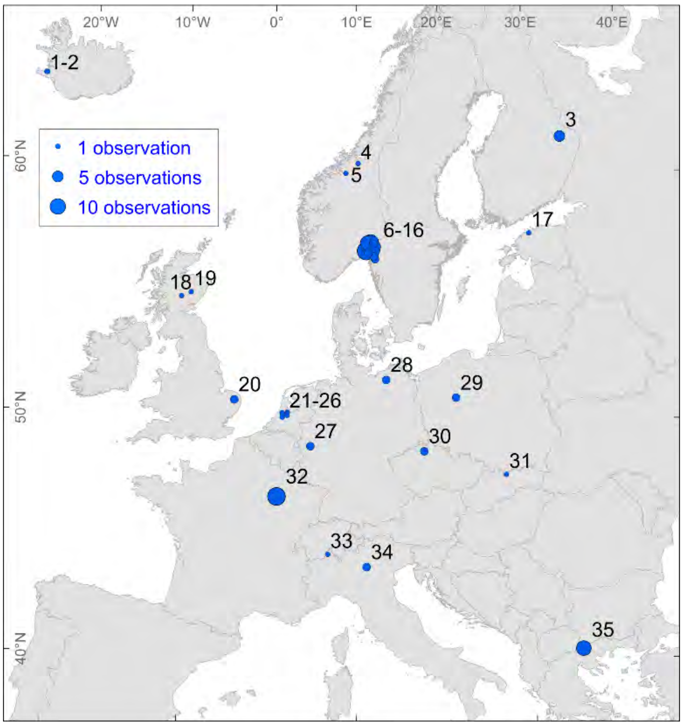
- *Reference: 1=Magnussen (1997), 2=Oglethorpe and Miliadou (2000), 3=Scherrer (2003), 4=Cooper et al., (2004), 5=Muthke and Holm-mueller (2004), 6=Groom et al. (2007), 7=Barton et al. (2009), 8=Spash et al. (2009), 9=Czajkowski and Ščasnỳ (2010), 10=Notaro and De Salvo (2010), 11=Jóhannesdóttir (2010), 12=Vojáček and Pecáková (2010), 13=Pädam and Ehrlich (2011), 14=Schaafsma et al. (2012), 15=Lehtoranta et al. (2013), 16=Söderberg and Barton (2014).*
- *Valuation method: CV=contingent valuation, CE=choice experiment, TC=travel costs.*
- *Value type: WTP=Willingness to pay or willingness to accept, CVar=Compensating variation, CSur=Consumer surplus, MIP=Marginal implicit price.*
- *Ecological Status: 1=High, 2=Good, 3=Moderate, 4=Poor, 5=Bad.*

Label in Fig.4.1	Lake name	Country	Area (km2)	No. of observations	Reference	Valuation method	Value type	Drinking water	Populations and habitats	Recreational fishing	Swimming	Boating	Sightseeing	Recreation (unspec.)	Symbolic/Cultural heritage	Ecological Status	Precipitation (mm)	Temperature (degC)	Elevation (m)	Density of lakes in 10 km	Visible area (km2)	GDP in 10 km (EUR/person)
1	Lake Vifilsstaðavatn	Iceland	0.3	1	11	TC	CSur			X						3	992	4.3	45	1.2	68.3	25953
2	Lake Elliðavatn	Iceland	1.9	1	11	TC	CSur			X						2	1136	4.4	79	1.1	147.8	25953
3	Lake Pielinen	Finland	962.9	5	15	CV	WTP			X	X	X	X	X		2	634	2.0	91	4.5	2371.8	24482
4	Anoya	Norway	10.6	1	1	CV	WTP					X	X	X		3	912	4.4	149	4.2	123.2	52151
5	Ovre Neadalsvatnet	Norway	0.4	1	6	CV	WTP		X							2	1080	1.1	778	3.0	43.5	53841
6	Bjorkelangen	Norway	3.4	2	7	CV, CE	WTP, MIP		X	X	X	X	X	X		3	749	5.2	126	5.1	82.0	47983
7	Oyeren	Norway	84.3	8	7	CV, CE	WTP, MIP		X	X	X	X	X	X		2	799	5.6	101	3.7	305.1	46488
8	Langen	Norway	2.6	7	1, 7	CV, CE	WTP, MIP		X	X	X	X	X	X		3	846	5.4	151	5.3	42.7	50846
9	Hemnessjoen	Norway	12.4	2	7	CV, CE	WTP, MIP		X	X	X	X	X	X		3	799	5.4	135	7.0	113.3	44277
10	Rodenessjoen	Norway	18.6	4	7	CV, CE	WTP, MIP		X	X	X	X	X	X		4	822	5.6	122	6.2	139.9	40684
11	Oymarksjoen	Norway	15.1	2	7	CV, CE	WTP, MIP		X	X	X	X	X	X		4	842	5.8	124	10.4	149.9	38627
12	Glomma	Norway	30.4	12	7	CV, CE	WTP, MIP		X	X	X	X	X	X		3	850	6.5	24	3.5	256.2	41426
13	Vansjo	Norway	36.9	11	7, 16	CV, CE	WTP, MIP		X	X	X	X	X	X		3	859	6.7	30	2.0	238.9	41904
14	Visterflo	Norway	3.2	2	7	CV, CE	WTP, MIP		X	X	X	X	X	X		2	809	7.0	21	5.8	26.0	41294
15	Femsjoen	Norway	11.4	2	7	CV, CE	WTP, MIP		X	X	X	X	X	X		3	862	6.3	79	5.4	64.5	40706
16	Orsjoen	Norway	6.4	2	7	CV, CE	WTP, MIP		X	X	X	X	X	X		2	870	6.0	144	7.7	42.9	39380
17	Lake Ülemiste	Estonia	9.2	1	13	CV	WTP							X		3	647	5.2	41	1.3	110.9	14894
18	Loch Tummel	UK	6.1	1	8	CV	WTP	X	X							2	1024	7.3	168	0.8	85.7	24322
19	Lochnagar	UK	0.1	1	6	CV	WTP		X				X			2	868	2.9	846	0.9	26.2	32060
20	Lake University	UK	0.1	3	4	CV	WTP		X				X			4	632	10.1	11	0.4	3.0	20301

	East Anglia																					
21	Westeinder Plassen	Netherlands	9.3	1	14	CE	WTP		X	X	X	X	X	X		5	825	9.3	1	3.5	142.0	42167
22	Nieuwkoopse Plassen	Netherlands	1.6	1	14	CE	WTP		X	X	X	X	X	X		4	831	9.4	1	4.2	52.3	33412
23	Reeuwijkse Plassen	Netherlands	7.6	1	14	CE	WTP		X	X	X	X	X	X		4	831	9.4	1	1.4	41.1	33251
24	Ankeveense Plassen	Netherlands	1.3	1	14	CE	WTP		X	X	X	X	X	X		3	832	9.2	1	14.3	38.3	35952
25	Loosdrechtse Plassen	Netherlands	9.3	1	14	CE	WTP		X	X	X	X	X	X		4	832	9.2	1	7.6	71.6	37862
26	Maarsseveense Plassen	Netherlands	0.6	1	14	CE	WTP		X	X	X	X	X	X		3	823	9.2	1	7.3	55.2	38805
27	Ville–Seen	Germany	0.2	2	5	CV	CVar			X	X					2	717	9.7	104	1.7	15.9	28149
28	Guestrower–Seen	Germany	2.9	2	5	CV	CVar			X	X					1	596	8.4	12	2.4	44.5	18801
29	Legowskie Lake	Poland	0.6	2	9	CV	WTP						X			3	528	7.9	70	2.3	24.1	6674
30	Mácha Lake	Czech Republic	2.8	3	9, 12	CV, CE	WTP				X	X	X	X		4	709	8.0	252	0.7	64.7	9517
31	Dlugi Staw Gasienicowy	Poland	0.01	1	6	CV	WTP						X			2	1180	0.4	1720	0.8	5.4	5358
32	Lac du Der	France	40.2	10	3	CV, TC	WTP, CVar		X		X		X			2	801	10.1	135	2.8	371.4	27128
33	Lago Paione Inferiore	Italy	0.02	1	6	CV	WTP			X			X			2	1148	1.4	2171	0.1	49.6	25674
34	Lake Garda	Italy	367.4	2	10	CV	WTP		X						X	2	978	12.9	81	0.3	802.7	30094
35	Lake Kerkini	Greece	64.2	9	2	CV	WTP		X	X				X	X	4	543	15.1	25	0.0	637.6	12864

A minimum level of consistency of the dependent variable across observations is required to ensure the validity of a meta-analysis approach (Smith and Pattanayak 2002, Nelson and Kennedy 2009). In this case, the commodity consistency is satisfied since the analysis collects values of goods and services similar across studies (i.e. ecosystem services offered by lakes located in Europe). The ecosystem services analysed are mostly recreational activities, which can be pooled in a single meta-analysis (Moeltner et al. 2007).

Figure 4.1. Location of the 35 lakes present in the meta-database. The size of the symbols is proportional to the number of observations per lake, as illustrated in the legend. The labels correspond to the lake ID shown in Table 4.1.



We standardized the observations found in the literature by converting all values from primary studies in monetary units per respondent per year (Germandhi et al. 2010, Ghermandi and Nunes 2013). Differences in currencies and in purchasing power among countries have been accounted for by using the Purchasing Power Parity (PPP) index provided by the PennWorld Table[11]. Price levels have been converted to a 2010 baseline using the national customer price indexes provided by the International Monetary Fund (World Economic Outlook 2014[12]). As a result, all ecosystem services' values are expressed in 2010 EUR PPP per person per year.

The meta-database was completed with key biophysical variables that are assumed to influence the delivery and the value of ecosystem services, namely temperature, precipitation, elevation, lake density in the lake's surrounding area (i.e. presence of substitutes) and sight from the lake (see Table 4.1). These variables were extracted from international data sets using Geographic Information Systems. GDP per capita derived from Eurostat statistics was also added to the

[11] http://cid.econ.ucdavis.edu/pwt.html
[12] https://www.imf.org/external/pubs/ft/weo/2014/02/weodata/index.aspx

meta-database. Combining information from primary studies with external spatial data from geographic information system (GIS) data layers has become a common practice in recent meta-analyses (Ghermandi and Nunes 2013, Ghermandi 2015, Johnston and al. 2016). Table 4.2 explains the data sources and processing techniques for each variable. Other biophysical variables were explored (i.e. nitrogen and phosphorous concentration, percentage of urban area surrounding the lake, land use type around the lake, slope of the visible area) but later they were omitted due to autocorrelation or to lack of significance in the econometric model.

Table 4.2. Calculation of geographical variables including data sources and GIS processing.

Variable	AREA
Definition	Surface area of the lake in km^2
Data source	The new Ecrins data set which is an updated beta version (v1.5) of Ecrins (European catchments and rivers network system, http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network). It includes 388,264 lakes across Europe ranging from 0.4 ha to 1.7 million ha. We selected 12,590 lakes that had available information about their ecological status.
Processing	We calculate the area of each lake polygon. The two smallest lakes of our meta-database (lakes no. 31 and 33 from Table 4.1) were not present in the new Ecrins data set and had to be digitized based on satellite imagery.
Variable	POPULATION
Definition	Number of persons at their usual place of residence per square kilometre for the Census reference year 2011
Data source	The GEOSTAT 2011 grid dataset with population data (http://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/population-distribution-demography)
Processing	We treat the duplicate cells in the border between countries and transform the data.
Variable	PRECIPITATION
Definition	Mean annual precipitation in mm, i.e. sum of the monthly average precipitation in the lake for the period 1951-2000
Data source	"Mean monthly precipitation totals" from the new Global Precipitation Climatology version 2015 (Meyer-Christoffer et al. 2015, Global Precipitation Climatology Centre http://gpcc.dwd.de). It is based on gauge data from ca. 75,000 stations, it has at 0.25° x 0.25° resolution and it focuses on the period 1951-2000. The.
Processing	We converted the data from netcdf, extracted the information for Europe, estimated the annual total as a sum of the monthly averages, and extracted the values for each lake.
Variable	TEMPERATURE
Definition	Average annual temperature in Celsius degrees for the period 1950-2000
Data source	WorldClim version 1.4 (Hijmans et al. 2005, http://www.worldclim.org/current). WorldClim is a set of global climate layers with a spatial resolution of about 1 km^2. The selected parameter was "average monthly mean temperature" at 30 arc-seconds resolution.
Processing	We estimated the annual average temperature for Europe and we extracted the data for each single lake.
Variable	ELEVATION
Definition	Altitude above sea level in m
Data source	GTOPO30 (https://lta.cr.usgs.gov/GTOPO30) is a global digital elevation model with a horizontal grid spacing of 30 arc-seconds (approximately 1 km). Data is available from the U.S. Geological

	Survey (http://eros.usgs.gov/find-data).
Processing	We transformed (adapted) the original data and extracted the values for each individual lake. Some areas from the Netherlands that lay below the sea level were data gaps in the digital elevation model. We applied value=1m for the lakes in that situation.
Variable	ECOLOGICAL STATUS
Definition	According to the EU Water Framework Directive, the ecological status of each water body is ranked in five classes (High-Good-Moderate-Poor-Bad) based on biological quality elements, physico-chemical characteristics, hydrological and morphological conditions.
Data source	The WISE2 databases (http://www.eea.europa.eu/data-and-maps/data/wise_wfd) are compiled by the European Environmental Agency and contain information from River Basin Management Plans reported by EU Member States according to article 13 of the Water Framework Directive. Not all the data are made available for public download since they are being updated. The ecological status of the Norwegian lakes found in the meta-analysis was extracted from the Norwegian Water Information System (http://vann-nett.no/portal/Default.aspx) which follows the same metrics.
Processing	We used the tables related to surface water bodies to extract information about the ecological status or potential of EU lakes. We located the centroid of each lake with valid data and overlapped them with the new Ecrins data set, getting 12590 lakes with ecological status. In some cases, we had to select the dominant value from different water bodies coinciding with single Ecrins lakes. Most of the lakes included in the meta-database showed a perfect match between WISE2 and Ecrins data sets, but for 9 lakes we had to attributed the ecological status of the surface water bodies found in their functional elementary catchments (as defined in Ecrins). For lakes no. 1, 2 and 33 the ecological status was assigned by expert opinion based on scientific publications.
Variable	DENSITY OF LAKES
Definition	Percentage of surface area covered by other lakes within a buffer of 10 km around each lake
Data source	The new Ecrins data set which is an updated beta version (v1.5) of Ecrins (European catchments and rivers network system, http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network). It includes 388,264 lakes across Europe ranging from 0.4 ha to 1.7 million ha.
Processing	We estimated the percentage of area covered by alternative lakes in a 10 km external buffer around each of the 12,590 selected EU lakes. The analyzed lake area keeps out of the calculation. A value of 0% represents a lake with no alternative lakes in the buffer area.
Variable	GDP
Definition	Average GDP per capita in a buffer of 10 km around each lake
Data source	Gross domestic product (GDP) at current market prices by NUTS 3 regions from Eurostat (http://ec.europa.eu/eurostat/statistics-explained/index.php/GDP_at_regional_level), in particular data on GDP per capita for the year 2011.
Processing	We created a buffer around each lake's shores and intersected it with the GDP data. We used a weighted arithmetic mean: the values of each GDP area intersecting the 10 km buffer were weighted and compared to the total buffer area.
Variable	AREA VISIBLE FROM LAKE
Definition	Surface area visible from the lake shores within a buffer of 20 km
Data source	Shuttle Radar Topographic Mission (SRTM) Digital Elevation Model with 3-arc second (ca. 100m resolution) provided by the USGS (http://srtm.usgs.gov/index.php)
Processing	For each lake we created a full buffer (i.e. inside and outside the lake) and extracted a specific DEM from the data source. Using the lake outline and a visibility tool, we defined the cells that are visible from the lake shores and estimated their area coverage.

4.1.2 Meta-analysis

We explored several multiple regression models based on the meta-database, keeping the natural logarithm of lake values in EUR2010 (ln y) as the dependent variable. The explanatory variables are grouped in different matrices that include a vector of dummy variables describing the ecosystem services provided by the lake (denoted by ES), the study characteristics (X^s, i.e. survey method, payment vehicle, elicitation format), the lake characteristics (X^b, i.e. size of the lake, ecological status), and context-specific explanatory variables (X^c, i.e. GDP, lake density, temperature, etc.). We use a panel data model to tackle the potential *within-study* autocorrelation that could arise from using multiple lake values from the same primary study (Rosenberger and Loomis 2000). Our meta-analytical regression model is specified as a semi-logarithm form:

$$\ln y_{ij} = \gamma ES_{ij} + \beta^b X^b_{ij} + \beta^s X^s_{ij} + \beta^c X^c_{ij} + \mu_i + \varepsilon_{ij} \qquad (1)$$

where the subscript *i* takes values from 1 to the number of studies and subscript *j* takes values from 1 to the number of observations, μ_i is an error term at the second (study) level, ε_{ij} is an error term at the first (observation) level and the vectors $(\beta^b, \beta^s, \beta^c, \gamma)$ contain coefficients to be estimated for the associated explanatory variables. We assume that μ_i and ε_{ij} follow a normal distribution with means equal to zero and variances σ^2_μ and σ^2_ε respectively.

We estimated several empirical specifications of Equation (1), all using random-effect generalized least-squares regression. Following Stanley and Doucouliagos (2012), we applied a general-to-specific modelling strategy by estimating two regression models, one *full model* including all the selected explanatory variables and a *parsimonious model* considering only the significant explanatory variables ($p<0.1$).

Validity checks

We performed three validity checks of our meta-regression models. First, we plotted for each observation the point estimate and 95% confidence interval of the predicted lake value.

The second approach to assess the validity of the model was to compute a measure of the transfer error rate. Following previous literature (Enjolras and Boisson 2010, Chaikumbung et al. 2016), we used the Mean Absolute Percentage Error (MAPE) defined as:

$$MAPE = \left|\frac{predicted\ value - observed\ value}{observed\ value}\right| \times 100 \qquad (2)$$

where predicted value stands for the lake value predicted by the model and observed value corresponds to the lake value reported in the primary study.

The third validity control of our model is based on out-of-sample predictions. We followed Brander et al. (2007) and Chaikumbung et al. (2016) by running n-1 bootstrap out-of-sample regressions, i.e. we removed one lake observation at a time, re-run the meta-regression model and then used the associated benefit transfer to estimate the value for the omitted lake.

4.1.3 Benefit transfer and spatial aggregation of lake benefits

Our objective is to use a benefit transfer approach to value ecosystem services delivered by lakes in Europe. We need to make the distinction between the *biophysical potential* of European lakes to provide services to people and the *effective delivery* of these services to people. Some authors refer to these two valuation categories as supply and delivery, respectively (Karp et al. 2015). Both valuation types are relevant for decision-making. The assessment of service biophysical potential is crucial for conservation and sustainability studies, where decision-makers should seek to maintain the long-term capacity of the ecosystems to deliver services. The assessment of service delivery, on the other hand, reflects societal needs and benefits, which are the main goals of most policies. Proving a monetary valuation for these two dimensions offers a more holistic view of the socio-ecological systems and supports a better informed policy making. Our benefit transfer approach will then require two steps.

The first step of the benefit transfer approach consists in applying the estimated benefit transfer function described to a sample of European policy lakes. To make operational the benefit transfer to policy lakes, all independent variables in Equation (1) have to be computed for each lake. The outcome of this first step is to value the *biophysical potential* of European lakes to provide services to people, irrespective of whether people actually benefit from them. The valuation provided, even if it is based on people's willingness to pay, indeed corresponds to the potential or "function" of the service as defined in the conceptual framework for linking ecosystems and human well-being of TEEB (2010). Such value is based on each lake biophysical context and the assumed delivery of the three ecosystem services of the parsimonious model.

The second step of the benefit transfer approach consists in valuing the delivery or flow of lake ecosystem services to people. It is then needed to consider the beneficiaries of these services. Passing from an economic value for the lake biophysical potential to an economic valuation of ecosystem services delivered by lakes raises three technical issues related to the way individual benefits are spatially aggregated.

The first issue consists in defining the *pertinent market* that is the area on which individual benefits will be aggregated. This market delineation is known to be one of the most controversial questions in environmental valuation (Bateman et al. 2006). Following the existing literature, we will assume that the population located beyond a certain distance to the lake does not benefit from ecosystem services delivered by the lake (Hanley et al. 2003). The rationale behind using a certain radius for aggregating individual values comes from the observation that for certain types of use value, it is reasonable to consider that the willingness to pay declines with distance to the site, in particular because the use of an environmental resource, such as for recreation, is likely to be lower for people who live further away from it. Thus, above a certain distance to a lake, we can expect the willingness to pay for ecosystem system services provided by a lake to be negligible.

The second issue is the parametric specification of the *distance decay function* that represents how distance to the lake affects individual willingness to pay for ecosystem services. The parametric form of the spatial distribution of values has been analysed in the literature with a wide range of approaches. In its simplest form, it is assumed that values decrease linearly with distance from the policy site (e.g. Pate and Loomis 1997, Hanley et al. 2003, Söderberg and Barton 2014). Non-linear distance decay functions have also been considered but there are only a few studies that provide guidance on how to really specify such a function (Hanley et al. 2003, Schaafsma et al. 2013).

The third issue is related to the sensitivity of the willingness to pay to changes in the ecosystem service provision by lakes. In our context, we face situations where a person can be located at less than 30 km from at least two lakes. In this case, how the willingness to pay for ecosystem services delivered by each lake should be aggregated remains highly debated. This is the *scope effect* analysed in abundant environmental valuation literature (Whitehead 2016). One of the major criticisms levelled at the environmental valuation methods is their insensitivity to the size or amount of the good being valued. A famous example can be found in Bateman et al. (2005) who obtain an insignificant difference in WTP estimates between the valuations given for protecting 4 lakes and protecting 400 lakes.

To identify the beneficiaries for each lake we use the GEOSTAT 2011 grid dataset with population data (1km×1km) (see Table 4.2). For each lake, we select all cells within the pertinent market. The total value for ecosystem services delivered by lake *i* (EUR 2010/yr) is:

$$TV_i = \sum_{k \in M(i)} Pop_k \times \hat{V}_i \times \mathrm{D}(\mathrm{d}_{i,k}) \qquad (3)$$

where *k* indexes grid cells, Pop_k is the population in cell *k*, *M(i)* is the set of grid cells belonging to the pertinent market of lake *i*, $\hat{V}_i$ is the predicted economic value for the biophysical potential of this lake (EUR 2010/person/year) and $\mathrm{D}(\mathrm{d}_{i,k})$ is the linear distance decay function applied to the distance between the centroid of lake *i* and the centroid of grid cell *k* (for example, for a 30 km radius of the pertinent market, D(30)=0 and D(0)=1).

We compute two additional indicators of economic value for each lake. First, the lake value per capita (EUR 2010/person/yr) is defined as:

$$VCAP_i = \frac{TV_i}{\sum_{k \in M(i)} Pop_k} \qquad (4)$$

and second, the lake value per area of lake (EUR 2010/ha/yr) is defined as:

$$VAREA_i = \frac{TV_i}{VAREA_i} \qquad (5)$$

where $VAREA_i$ is the area in ha of lake *i*.

4.2 Valuation and mapping of lake ecosystem services at the European scale

4.2.1 Analysis of available economic values

Based on the 107 observations of the meta-database, the mean value for lake's ecosystem services in Europe is 87.21 EUR 2010/person/year while the median value is 46.27 EUR 2010/person/year, showing that the distribution of values is skewed with a long tail of low values and a few very high values. On average, these values are relatively similar to the ones reported by Brouwer et al. (1999) for wetlands' ecosystem services, i.e. 108 EUR 2010/respondent/year.

In Table 4.3, we have broken down the mean value of ecosystem services delivered by lakes according to several potential explanatory variables. Mean lake values have been calculated by (1) countries, (2) ecosystem service present in the primary valuation study, (3) valuation method used in the primary study, (4) publication status of the primary study, (5) lake's ecological status, and (5) class of GDP.

Large variations in lake values across countries are documented. The lowest values are found for Italy (4.41 EUR 2010/person/year) and for Estonia (5.87 EUR 2010/person/year), but these values have been obtained on a limited number of observation for this two countries. The mean value of ecosystem services delivered by lakes is greater than 100 EUR 2010/person/year only in two countries, Norway (121.80 EUR 2010/person/year) and the Netherlands (140.85 EUR 2010/person/year).

Next we have split our sample depending upon the fact that one of the 8 ecosystem services was offered or not by each lake. In that case, the number of observations in Table 4.1 does not sum up to 107 since one lake value reported in a primary study may correspond to a bundle of ecosystem services. Indeed, among our 107 observations, 85 lake's values include the *sightseeing* service, 81 the *maintaining populations and habitats* service, and 72 both services. So values reported in Tables 4.1 and 4.3 cannot be directly interpreted as the mean value of each particular ecosystem service but as the mean value of all ecosystem services given that the ecosystem service mentioned is provided. In general, low values are found for drinking water (32.62 EUR 2010/person/year) and symbolic appreciation/cultural heritage (12.57 EUR 2010/person/year). The value associated with recreational services varies around 100 EUR 2010/person/year, from 103.66 EUR 2010/person/year for the sightseeing service to 116.39 for the boating service.

Table 4.3. Descriptive statistics on the annual value for ecosystem services delivered in the lakes (2010 EUR PPP per capita) reported in the meta-database.

		Value of ecosystem services delivered by lakes (Per capita annual value in 2010 EUR PPP)			
	Obs.	Mean	Std. Dev.	Min.	Max.
Full sample	107	87.21	96.06	0.61	406.58
By country					
Czech Republic	3	28.04	16.32	11.27	43.87
Estonia	1	5.87	.	5.87	5.87
Finland	5	13.23	6.03	5.38	20.11
France	10	82.17	91.57	1.95	253.43
Germany	4	38.17	9.62	28.99	50.53
Greece	9	11.95	5.71	4.78	24.10
Iceland	2	96.95	23.18	80.56	113.35
Italy	3	4.41	0.62	3.80	5.04
Netherlands	6	140.85	9.51	127.94	153.15
Norway	56	121.80	109.37	0.61	406.58
Poland	3	10.30	7.81	1.61	16.70
United Kingdom	5	37.95	23.57	6.84	70.27
By ecosystem service					
Drinking	1	32.63	.	32.63	32.63
Boating	67	116.39	103.31	5.38	406.58
Sightseeing	85	103.66	100.61	1.61	406.58
Maintaining populations and habitats	81	104.64	102.54	0.61	406.58
Fishing	70	110.95	103.38	3.80	406.58
Swimming	76	111.43	102.18	2.92	406.58
Unspecified recreational	70	110.92	104.13	4.78	406.58
Symbolic appreciation	4	12.57	9.56	4.37	24.10
By valuation method					
Choice experiment	33	176.02	113.84	11.27	406.58
Travel cost	6	152.16	64.55	80.56	253.43
Contingent valuation	68	38.38	37.64	0.61	161.57
By publication status					
Peer reviewed	40	37.42	46.69	0.61	153.15
Not peer reviewed	67	116.94	105.51	1.95	406.58
By ecological status					
High	2	30.66	2.37	28.99	32.34
Good	38	54.91	69.63	0.61	277.18
Moderate	42	143.64	112.88	5.87	406.58
Poor	24	42.63	45.33	4.78	153.15
Bad	1	127.94	.	127.94	127.94
By GDP per capita					
Low (<26,000 EUR)	29	25.27	25.37	1.61	113.35
Medium (≥26,000, ≤40,000)	32	72.56	67.63	1.95	253.43
High (>40,000 EUR)	46	136.46	114.38	0.61	406.58

Column 'Obs.' gives the number of observations on which the mean values are computed

The type of valuation method leads to significant different estimates for the value of ecosystem services delivered by lakes, with a quite low value for contingent valuation studies (38.38 EUR 2010/person/year). This suggests that methodological heterogeneity in the primary studies may influence the regression results.

Peer-reviewed studies report lower values than non-peer reviewed studies, consistent with the findings of Enjolras and Boisson (2010) and Chaikumbung et al. (2016), and perhaps due to the fact that the peer review tends to select more conservative valuations.

The relationship between lake ecological status and economic value of ecosystem services delivered by lakes appears to non-monotonic but interpretation is made difficult due to the very limited number of observation in the two extreme classes (high and bad ecological status). If we exclude these two extreme classes, we find on average a lower value for lake in poor ecological status in comparison with lakes in moderate or good ecological status.

The results regarding the GDP indicate that income may play a role in explaining ecosystem service values. Ecosystem services values tend to be significantly higher for lake surrounded by a population having a high GDP per capita. This suggests that lakes may be viewed as normal goods.

4.2.2 Estimation of the meta-analysis regression

In Table 4.4, we report the result of the two estimated models. The *full model* includes as determinant of lake economic value eight dummy variables for the eight ecosystem services analysed. The second model is a more parsimonious one where non-significant ecosystem services have been excluded.

Table 4.4. Meta-regression of economic valuations of ecosystem services provided by lakes in Europe

	Full model		**Parsimonious model**	
	Coeff.	**Std. err.**	**Coeff.**	**Std. err.**
Ecosystem services (ES)				
Drinking water	2.0022**	0.8659	1.7722**	0.8627
Fishing	0.3288	0.3036		
Swimming	-0.0057	0.306		
Boating	1.5761***	0.5032	1.6132***	0.4300
Sightseeing	0.9753***	0.3374	0.6178*	0.3634
Unspecified recreational	-0.5483	0.392		
Maintenance of populations and habitats	-0.8858***	0.3017		
Symbolic appreciation	-0.1816	0.5201		
Study characteristics (X^s)				
Choice Experiment	0.9779***	0.1978	0.9268***	0.1758
Travel Cost	2.3063***	0.4319	2.8267***	0.3849

Lake characteristics (X^S)				
Poor or bad ecological status	-0.3544	0.2488	-0.5083*	0.2831
Area visible from lake	-0.0008***	0.0002	-0.0008***	0.0002
Context-specific variables (X^C)				
Temperature (degrees)	0.1299**	0.0508	0.0925*	0.0491
Precipitation (mm)	-0.0013	0.0012	-0.0024*	0.0013
Elevation (m)	-0.0009*	0.0005	-0.0004	0.0005
Lake density 10km	-0.1048***	0.0406	-0.1007**	0.0479
GDP capita 10 km (in ln)	0.7242**	0.3289	0.3118	0.3120
Constant	-3.9253	3.1708	0.8572	2.8522
R-square	0.7457		0.7134	
Number of studies	35		35	
Number of observations	107		107	

Note: Panel data random-effects models estimated by using generalized least squares with STATA software. ***, **, * for significant at the 1, 5, 10 percent level, respectively.

The full meta-regression model provides some indications on how people value lake's ecosystem services. Four of the eight ecosystem services appear to be significant: drinking water, boating, sightseeing and maintenance of populations and habitats.[13] Lakes providing drinking water, boating and sightseeing are more highly valued. In contrast lakes providing the maintenance of populations and habitats service are less valued possibly because the provision of this service is not directly observed by people.

As expected, the type of valuation method matters: studies using a choice experiment method or a travel cost approach report a higher value for the ecosystem services delivered by lakes compared to contingent valuation studies (the reference category). The methodological heterogeneity in the primary studies then influences the regression results, maybe because the different methods do not rely on the same theoretical construct.

The full meta-regression model shows that the value people attribute to lakes' ecosystem services is positively associated to temperature (i.e. warm lakes are more valued) and GDP per capita (i.e. people with larger income can consume and value more goods and services, especially recreational services). On the contrary, lakes' value seems to be (slightly) negatively correlated to precipitation (i.e. people can profit more of lakes in regions where precipitation are less frequent), to lake density (i.e. the presence of nearby substitutes can decrease the value attributed to single lakes), and to the surface area visible from the lake shores (probably because this variable is linked to the lake surface area, so larger lakes have slightly lower values).

The original WFD categories for the ecological status (from 1=high to 5=low) were reclassified to find the possible correlation with lakes value in the meta-regression model. The dummy

[13] In their recent meta-analysis of coastal recreation values in developing countries, Chaikumbung et al. (2016) report that only 3 ecosystem services (among the 12 considered) appear to be significant.

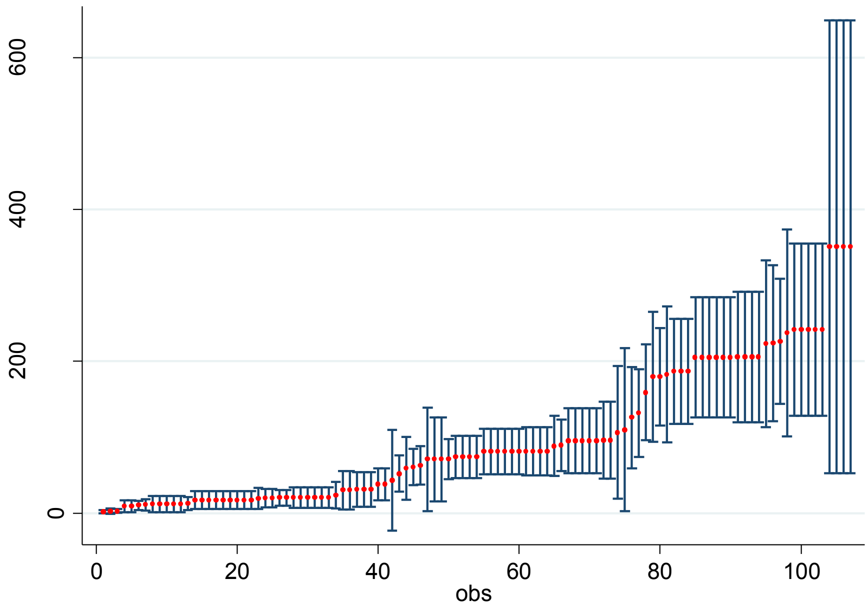
variable that showed the most significant relation is 0=moderate to high ecological status, 1=poor or bad status. At scientific level, there are established and intercalibrated methodologies to characterise the ecological status of water bodies across the EU (Poikane et al. 2015). However, the appreciation of the status by people probably depends on the visible conditions of the lake and the uses that are permitted. Thus, only poor or bad conditions related to algal blooms or reduced transparency that may limit fishing or recreational uses can be clearly perceived by people. This could explain the results of our modelling, where the variable "poor and bad" ecological status shows a significant negative effect on the lakes value, while the other categories (moderate, good or high) were not significant.

The parsimonious meta-regression model keeps only three variables for ecosystem services, namely drinking water, boating and sightseeing, all of them being significant and consistent with the full model. In the same way, the estimated coefficients for study characteristics variables, lake characteristics variables, and context-specific variables are very consistent.

Validity checks

Before moving toward the benefit transfer to European lakes, some validity checks have been conducted using the parsimonious model. First, in Fig. 4.2 we have plotted for each observation the point estimate and the 95% confidence interval of the predicted value for ecosystem service delivered by each lake.

Figure 4.2. Point estimate and 95% confidence interval for predicted lake values (in EUR 2010/person/year) with the parsimonious model.



Second, in-sample and out-sample MAPE (transfer error rate) have been computed. The average MAPE for our parsimonious model is about 115%, which can be considered rather high, but this is mostly driven by a small number of "aberrant" transfer values. By excluding the six observations having the highest MAPE, the average MAPE based on the remaining 101 observations drops to 65.9%. Indeed, the median MAPE of the parsimonious model is 44.6% and one-fourth of the observations have a MAPE lower than 22.2%. These results are comparable to other forecast errors for valuation studies on wetlands that report MAPE values of 58% (Brander et al. 2006), 85% (Brouwer 2009) or 186% (Brander et al. 2007), or for valuation studies of coastal lagoons with a MAPE value of 87% (Enjolras and Boisson 2010).

The out-of-sample average MAPE for the parsimonious model is about 285%, a figure largely greater that the in-sample MAPE reported above. But again, the high error transfer is driven by a few outliers. The out-of-sample median MAPE of the parsimonious model is 61.73% to be compared to 44.59% for in-sample predictions. Thus, the predictive power of the model slightly decreases when passing from in-sample to out-of-sample predictions, but we consider that our estimated benefit transfer function can be robust enough to extrapolate to our policy sample, i.e. to estimate the value of ecosystem services provided by lakes at the European level.

4.2.3 Benefit transfer to European lakes

In this section we propose a benefit transfer approach to assess the economic value of ecosystem services delivered by European lakes. The first step of the procedure consists in identifying the policy lakes on which the benefit transfer will be realized. In the second step, we select the most appropriate transfer function, which is then applied on each policy lake in order to get an estimate of the biophysical potential of this lake to deliver ecosystem services. In the third step, we spatially aggregate individual benefits in order to get an estimate of the economic value of ecosystem services used by populations surrounded each lake.

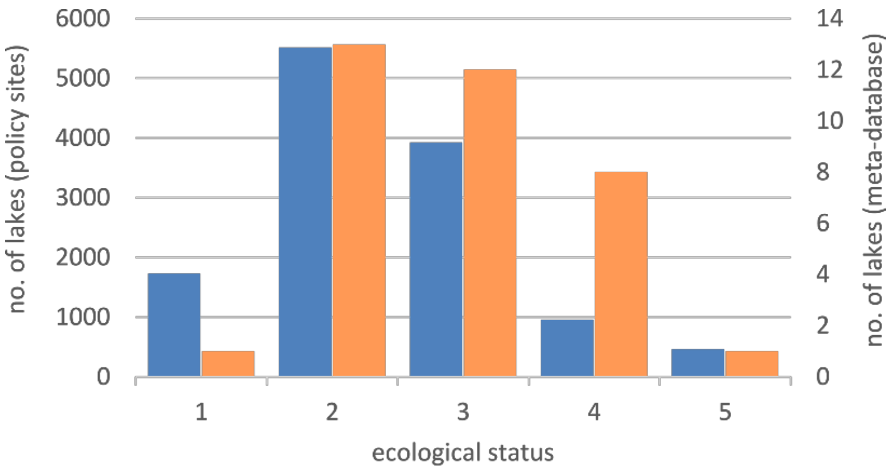
Description of the policy lakes

We apply the estimated benefit transfer function to a sample of 12,590 lakes located in EU Member States (hereafter the 12590 EU lakes are also called the policy sample or policy sites). We describe here the main characteristics of the policy lakes, and we check to what extend they differ from the 35 study lakes used for estimating the meta-regression.

The policy lakes have been selected from the ECRINS data set (European Catchments and RIvers Network System) based on the availability of data to cover the biophysical covariates of the meta-database, being the ecological status the most limiting variable. Table 4.2 explains the data sources and processing techniques.

The policy lakes are located between the sea level (typical from the Netherlands) and nearly 3,000 m high (found in some Alpine sites). Average annual precipitation is relatively high, over 700 mm, and temperature relatively low, below 5 Celsius degrees (Table 4.5), due to the predominance of northern lakes (more than 8200 lakes or 65% of the policy sites are located in Sweden and Finland). The most common situation is that each individual lake has no substitutes in a 10 km radius, although in average 6% of the area surrounding each lake is covered by other lake substitutes. The GDP per capita around the European lakes (within a 20 km radius) averages almost 26,000 EUR, although the minimum and maximum values (present in the border between Romania and Moldavia and near Copenhagen, respectively) differ in more than one order of magnitude. The area visible from each lake shores has an average of 57 km^2, but it is extremely variable due to the different landscapes and lakes' size.

Figure 4.3. Ecological status reported in the study sites included in the meta-database (in orange, right hand side) and in the policy sites (in blue, left hand side). 1=High, 2=Good, 3=Moderate, 4=Poor, 5=Bad.



In the policy sample, 13.7% of the lakes have a high ecological status, 43.8% a good ecological status, 31.2% a moderate ecological status, 7.6% a poor ecological status, and the remaining 3.6% have a bad ecological status (Fig. 4.3). Thus, the average ecological status is between moderate and good, while the mode is good. This distribution is quite similar to that of the study lakes (see Fig. 4.3). European lakes with high ecological status are mostly located in the Baltic region (Sweden, Finland, Lithuania, northern Germany and Denmark), Scotland and Ireland.

In Table 4.5 we more formally compare the lake's characteristics in study sites and in policy sites using two-sample mean-comparison tests. It appears that both samples significantly differ in terms of precipitations, temperature, lake density. However, no significant difference is documented for elevation, poor of bad ecological status and visible area from the lake.

Table 4.5. Average characteristics and comparison of the lakes included in the meta-database and in the policy sites.

	Meta-database (35 lakes)		*Policy lake database (12 590 lakes)*		*Two-sample mean-comparison test*	
	Mean	*St. dev.*	*Mean*	*St. dev.*	*Mean*	*St. dev.*
Precipitation (mm)	834.4	158.5	727.4	272.3	-107.1***	(-3.98)
Temperature (°C)	6.8	3.2	4.4	3.7	-2.368***	(-4.35)
Elevation (m)	224.0	470.8	259.7	264.8	35.71	(-0.45)
Ecological status (1 to 5)	2.9	0.9	2.4	0.9	-0.471**	(-2.86)
Ecological status poor or bad (0,1)	0.3	0.4	0.1	0.3	-0.145	(-1.93)
Lakes density (%)	3.7	3.2	6.2	5.6	2.521***	(-4.64)
GDP/capita (€)	32397.4	12856.6	25946.8	7641.7	-6450.6**	(-2.97)
Visible area (km^2)	195.9	415.3	57.5	90.5	-138.4	(-1.97)

t statistics in parentheses

* $p<0.05$, ** $p<0.01$, *** $p<0.001$

Valuing the potential of lakes to provide ecosystem services in Europe

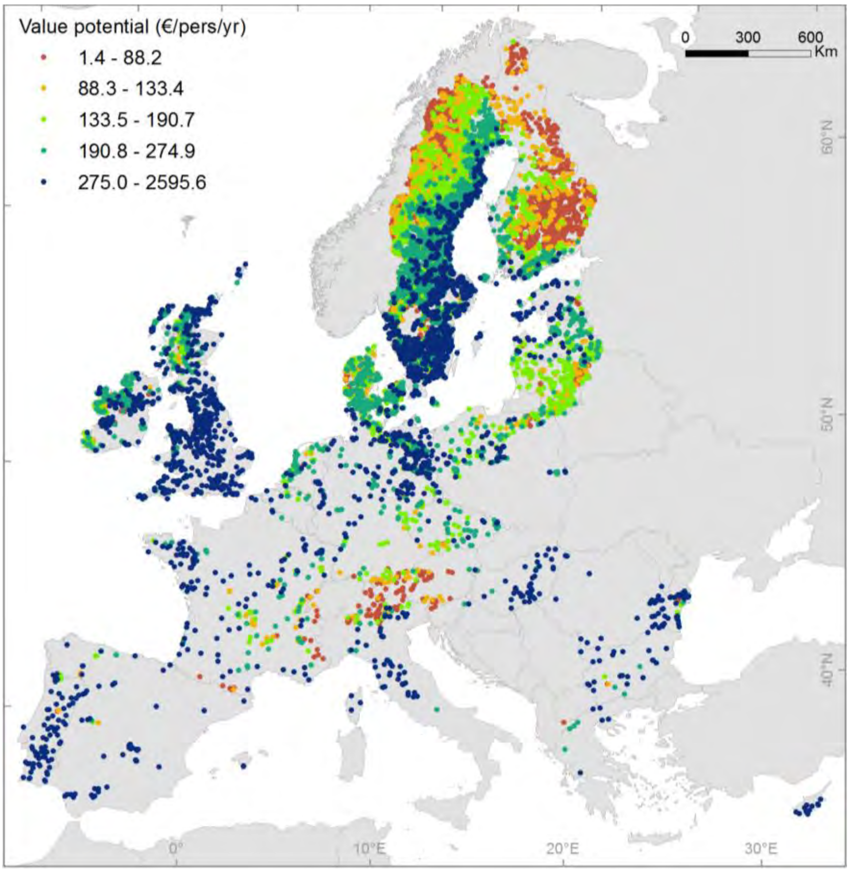
Here we apply the estimated benefit transfer function (*parsimonious model*) to predict the economic value for the biophysical potential of European lakes to provide ecosystem services to people. To make operational the benefit transfer to the 12,590 policy lakes, we must estimate all independent variables in Equation (1) for each lake. Data is available for study characteristics, lake characteristics and context-specific explanatory variables (see Table 4.2). However, data availability was a constraint for the dummy variables related to the presence/absence of the three ecosystem services appearing in our parsimonious model (the provision of drinking water, boating and sightseeing)[14]. This is a classical problem of large-scale benefit transfer for ecosystem service values (Ghermandi and Nunes 2013). For the provision of drinking water, we estimated the presence of the service by the national share of drinking water that is taken from surface water sources, using Eurostat data[15] on abstraction for the public supply system per country (reference year 2010). For the countries where data were not available (Ireland, Greece, Italy and Portugal) we used the average value of EU countries (33%). For boating and sightseeing, we assumed that the services were always offered by lakes.

Applying the parsimonious benefit transfer function, we provide for each policy lake an estimate of economic value for its biophysical potential to deliver the ecosystem services considered (provision of drinking water, boating and sightseeing) (Fig. 4.4). All values are expressed in EUR 2010 PPP per capita per year. Thus, based on our policy sample of 12,590 lakes, the estimated average economic value of a European lake is 203 EUR/person/year, with a median equal to 177 EUR/person/year. For 75% of our sample, the estimated average economic value of the lake biophysical potential is lower than 250 EUR/person/year.

[14] The European-scale benefit transfer will be conducted using the *parsimonious model* in which the vector ES includes dummy variables for the provision of drinking water, for the boating service and for the sightseeing services. Other ecosystem services are not individually identified but are accounted for in the value transfer.
[15] http://ec.europa.eu/eurostat/statistics-explained/index.php/Water_statistics

Figure 4.4. Economic value of the biophysical potential of lakes to provide ecosystem services in Europe (in EUR 2010/person/year).

(A) Individual data per lake in the policy sample.



(B) Data grouped (averaged) at country level.

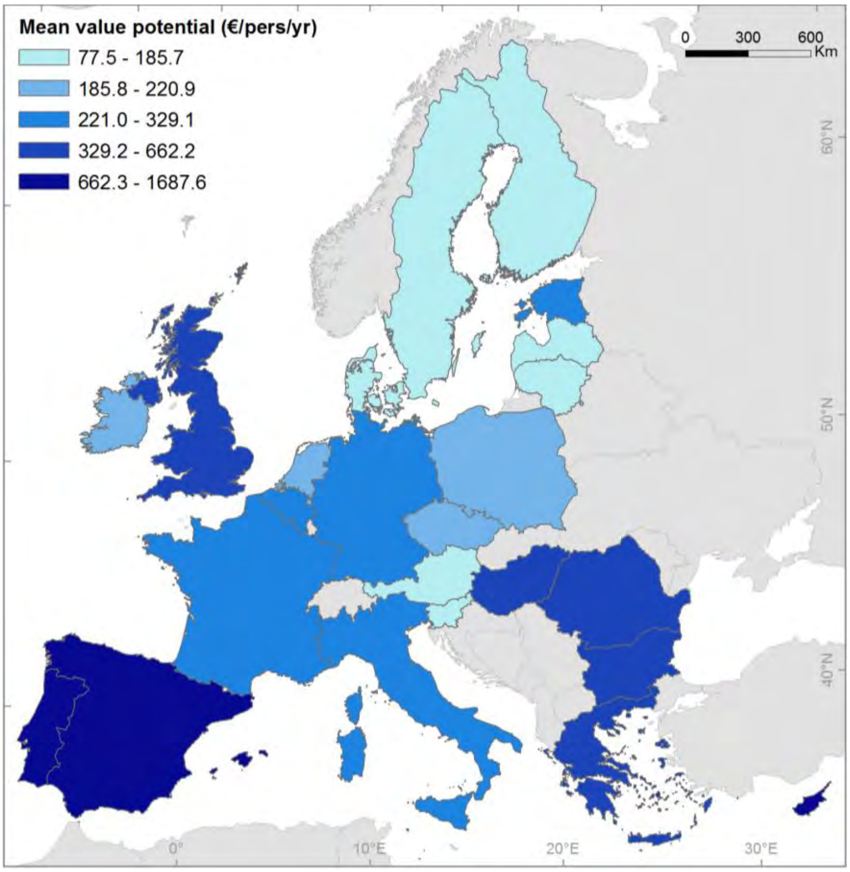


Table 4.6. Summary statistics of the economic value attributed to the biophysical potential of lakes to provide ecosystem services in the EU-27 (in EUR 2010/person/year).

	No. of lakes considered	Mean	Standard Deviation	Minimum	Maximum
Austria	57	77.53	62.34	14.48	406.49
Belgium	12	222.16	71.62	99.94	308.04
Bulgaria	29	439.48	206.25	67.01	879.81
Cyprus	12	1687.55	675.47	312.98	2533.47
Czech Republic	60	206.39	66.56	86.28	437.10
Denmark	630	172.89	61.71	14.02	291.71
Estonia	76	259.11	75.72	3.44	367.74
Finland	1515	92.52	63.88	4.10	354.86
France	224	272.64	156.00	11.25	720.77
Germany	520	242.65	109.61	14.28	462.85
Greece	10	409.39	292.89	58.42	981.85
Hungary	45	491.14	113.78	259.35	639.85
Ireland	597	189.44	74.18	6.04	521.40
Italy	106	256.04	219.38	15.12	925.78
Latvia	227	173.16	72.35	43.94	353.78
Lithuania	322	110.66	45.69	27.12	265.23
Netherlands	68	206.71	80.24	72.84	499.46
Poland	156	213.98	83.91	24.96	414.66
Portugal	75	818.01	340.22	86.53	1312.51
Romania	82	488.47	288.63	41.25	1192.44
Slovenia	4	166.31	102.14	74.01	283.57
Spain	75	908.92	606.31	94.20	2595.56
Sweden	6727	174.64	106.69	1.40	881.16

There are significant differences in lake value across countries (see Table 4.6). The lowest national average values are found in Austria and Finland (with 78 and 93 EUR/person/year, respectively), while the largest ones appear in southern countries like Cyprus, Spain and Portugal (with 1688, 909 and 818 EUR/person/year, respectively).

Regarding individual data, lakes with relatively high values (over 250 EUR/person/year) are found all across Europe, finding the maximum values (over 2500 EUR/person/year) in Spain and Cyprus (Table 4.6). Moderate values (120-250 EUR/person/year) concentrate mainly in south-east France, Czech Republic, the Netherlands, Ireland, Scotland, Denmark, northern Poland, Lithuania and mid-northern Sweden. Relatively low values (below 120 EUR/person/year) occur in the most remote lakes, i.e. Alpine regions, eastern Lithuania and Latvia, and across Finland and Sweden. Specifically, the minimum lake values (below 5 EUR/person/year) are documented in Sweden, Estonia and Finland.

Locations with warm temperature, scarce precipitation and limited lake substitutes (like Cyprus and Spain) receive higher values than colder, wetter places with numerous lakes (like mountainous and Scandinavian regions). These latter locations are usually remote and more pristine and, thus, tend to be in a better ecological status. At the same time, lakes in warmer locations, attracting more people and activities, are often under higher human pressures. This may explain why at the European scale the spatial distribution of potential lake values could seem opposite to the distribution of lakes' ecological status. Still, the relationship between lakes value and poor or bad ecological status is negative, stressing people's preference for good environmental conditions.

Figure 4.5 shows a measure of the confidence level of our analysis that applies to all the European results in this paper. This is expressed as the percentage of total lake's area represented by our European policy sample (based on the ECRINS data set, see Table 4.2) and, thus, taken into account in our model. It is observed, for example, that some regions from Spain or Bulgaria show a relatively low representativeness of the total lakes, while other areas from Austria or Ireland are relatively well represented. This is driven by data availability, especially by the reporting of the ecological status of water bodies.

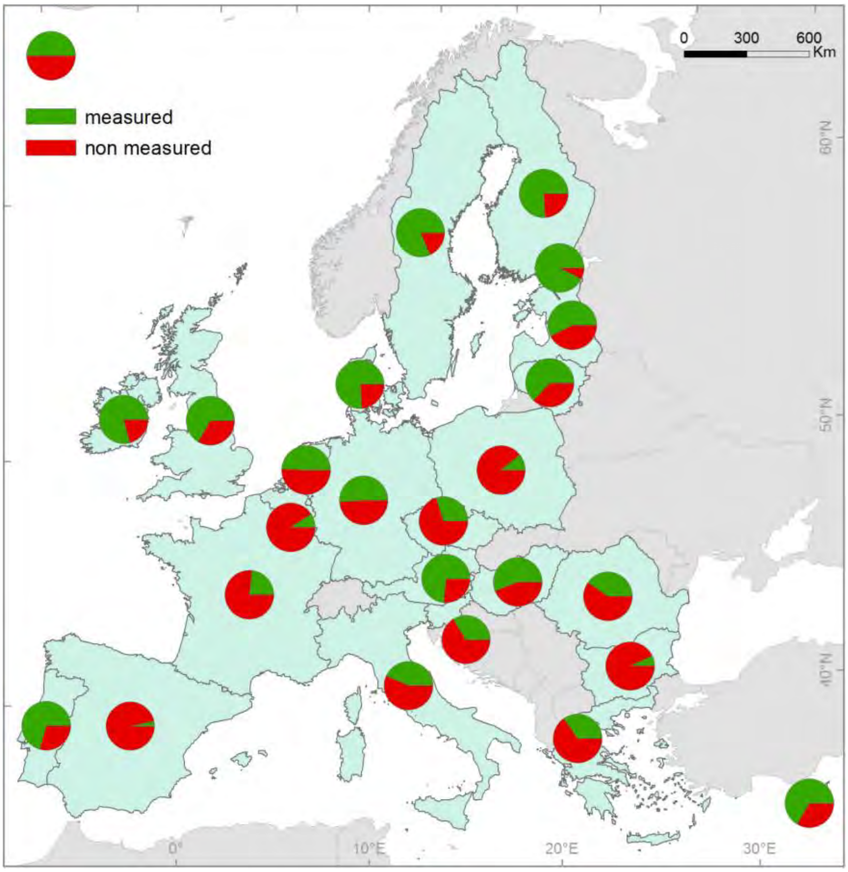


Figure 4.5. Relative coverage of our policy sample (in surface area of lakes) with respect to the total lakes of Europe, as reported in the ECRINS data set.

Valuing the delivery of ecosystem services provided by lakes in Europe

This section presents the economic valuation of lake services delivered to the local populations. Valuing the delivery of lake's ecosystem services requires to spatially aggregate the individual lake values (reported in the previous section) based on the population who effectively benefit from these services (population data are explained in Table 4.2).

As discussed in section 4.1.3, the first issue to tackle in this exercise is to define the *pertinent market*, i.e. the area on which individual benefits must be aggregated. Determining this maximal distance has been highly debated and its value depends upon the type of ecosystem considered. Here we use a threshold of 30 km (although we also conducted some spatial robustness checks by using 10 km and 20 km thresholds, see Table 4.7)[16]. The second issue is the parametric specification of the *distance decay* function which represents how distance to the lake affects individual willingness to pay for ecosystem services. Following previous works (e.g. Pate and Loomis 1997, Hanley et al. 2003, Söderberg and Barton 2014) we use a linear decay function. As a robustness check we replicate our analysis in Table 4.7 without considering any distance decay function. Lastly, we must consider the scope effect. Here we propose to use a weighted sum of the willingness to pay for ecosystem services delivered by each lake. The weights are based on the distance to each lake. The intuition is that a person considers all lakes located at less than 30 km but with a higher weight put on lakes located at a smaller distance. Thus, the results shown in this section show the benefits aggregated in a 30km pertinent market with a linear decay function and scope effect modelled with a weighted sum of WTP based on distance to lakes. Other alternatives of aggregating benefits are presented in Table 4.7.

[16] Hanley et al. (2003) is one of the first articles providing an in-depth analysis of deciding on the relevant population for aggregating individual benefits. They estimate the willingness to pay (WTP) of households for limiting low water flow in the Mimram river in Southern England. Whereas the WTP for household located on the shoreline of the river is estimated to be more than 8 pounds per respondent, it decreases to less than 3 and 2 pounds per respondent for households located respectively 4 km and 20 km away from the river. Their results suggest a WTP almost negligible above 30 km.

Table 4.7. Spatial aggregation robustness checks. Average economic value of ecosystem services delivery per lake in the policy sample (million EUR 2010/year) using different ways for aggregating economic values for the lake biophysical potential.

	Aggregation of population within 10 km to the lake		*Aggregation of population within 20 km to the lake*		*Aggregation of population within 30 km to the lake*	
	Linear decay function	*No decay function*	*Linear decay function*	*No decay function*	*Linear decay function*	*No decay function*
Economic value of ecosystem services delivery (million EUR 2010/year) with scope effect						
- Baseline scenario	1.29	3.66	2.31	6.54	2.92	8.29
- Remediation scenario	1.47	4.16	2.67	7.51	3.39	9.59
- % change	14.0%	13.7%	15.6%	14.8%	16.1%	15.7%
Economic value of ecosystem services delivery (million EUR 2010/year) without scope effect						
- Baseline scenario	3.39	9.54	12.14	34.1	25.66	72.54
- Remediation scenario	3.82	10.7	13.59	38.11	28.61	80.67
- % change	12.7%	12.2%	11.9%	11.8%	11.5%	11.2%

Table 4.8 presents the general results at continental level aggregated either per lake, per capita or per area of lake. These results have different units (and slightly different estimation methods) to serve various purposes in awareness raising or policy-making support. For instance, a cost-benefit analysis for the WFD could make use of the results aggregated per an individual lake; a regional policy maker could be interested in the lakes' value distributed by local inhabitants; and a land/water manager or a researcher willing to apply a value transfer approach would be looking for the data aggregated per area of lake. Figures 4.6, 4.7 and 4.8 illustrate the distribution of these results.

Table 4.8. Summary statistics of the results of value aggregation, i.e. the valuation of ecosystem services delivered by European lakes to local population.

	Mean	*Median*	*Standard Deviation*	*Kurtosis*	*Skewness*	*Minimum*	*Maximum*
Lake value (EUR/yr)	2,929,551	38,625	14,933,574	457	16	0	687,219,323
Lake value per capita (EUR/person/yr)	7.9	1.3	27.8	131	10	0	678
Lake value per area of lake (EUR/ha/yr)	180,328	334	1,744,025	864	24	0	92,323,432

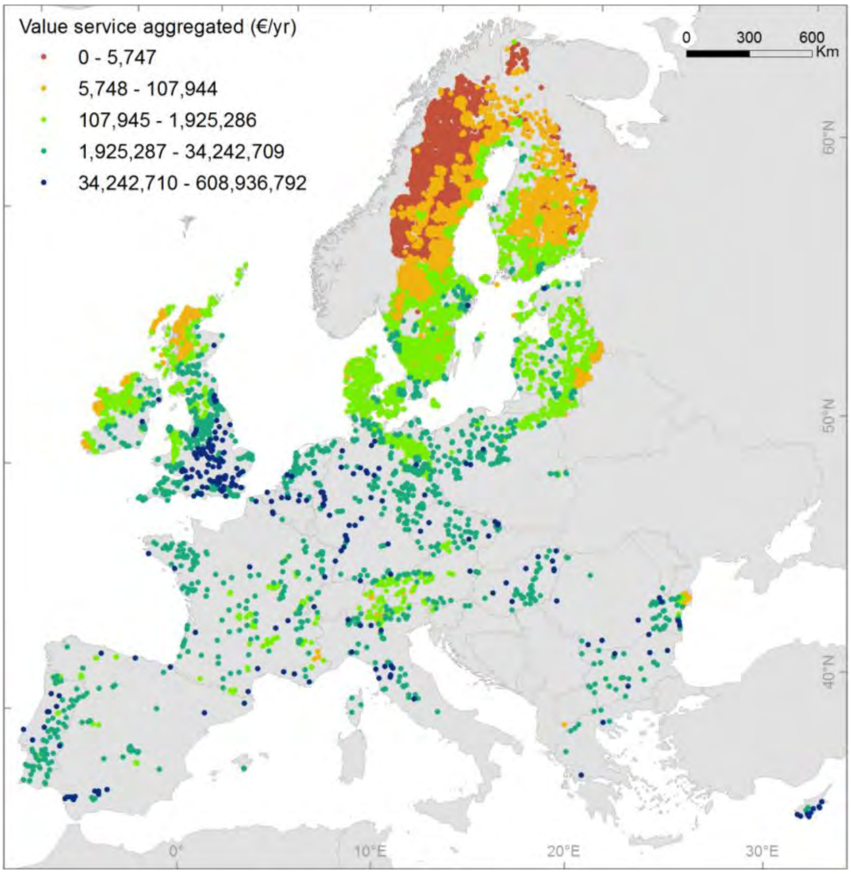
Table 4.9. Summary statistics of the economic value attributed to the actual delivery of ecosystem services aggregated per lake (value in thousands of EUR 2010/person/year).

	No. of lakes considered	*Mean*	*Standard Deviation*	*Minimum*	*Maximum*
Austria	57	2479.30	6884.42	70.51	43335.00
Belgium	12	66296.05	50603.06	13420.86	193003.66
Bulgaria	29	18016.41	17982.93	1014.69	77038.39
Cyprus	12	49178.31	25147.72	2528.56	86868.48
Czech Republic	60	12376.83	14090.32	883.55	67878.53
Denmark	630	593.42	707.05	29.72	5034.28
Estonia	76	2006.99	6170.65	1.79	37503.57
Finland	1515	191.05	976.63	0.04	26030.38
France	224	14872.77	34136.38	37.55	267716.17
Germany	520	10265.79	28542.53	12.94	391286.78
Greece	10	45214.38	80440.44	91.74	263068.98
Hungary	45	20660.31	22336.92	2639.73	85375.46
Ireland	597	641.63	3034.52	1.54	44796.01
Italy	106	22302.94	68742.22	28.42	687219.32
Latvia	227	630.52	1658.91	16.85	13409.26
Lithuania	322	478.13	1062.27	9.46	9751.36
Netherlands	68	13969.78	10490.07	2884.58	66189.10
Poland	156	4964.06	6788.42	144.39	50679.75
Portugal	75	15299.14	26006.68	479.38	176542.90
Romania	82	13920.11	26256.02	21.33	145514.91
Slovenia	4	15133.59	15110.51	1226.56	30427.04
Spain	75	24303.66	47626.63	171.19	267305.92
Sweden	6727	155.28	1097.14	0.00	37985.17

In general, lakes with very high values (over 1.2 M EUR/yr) are distributed all across continental Europe, Cyprus and England. The largest average national value is reported in Belgium (66.3 M EUR/yr) while the most valued individual lake (687.2 M EUR/yr) is the *Idroscalo* in Milan, Italy, probably linked to its very densely populated surroundings. Relative high values, roughly above the median (from 30,000 to 1.2 M EUR/yr), are mostly present in the Danube delta, Alpine region, the Mecklenburg-Vorpommern Lander of Germany, northwest UK, and around the Baltic Sea (Denmark, Sweden, Finland, Estonia, Latvia, Lithuania). Relatively low values (below 30,000 EUR/yr) concentrate in the most remote zones of Finland, Sweden, Scotland and Ireland. Minimum results are found in Sweden and Finland, with national average values of 155,300 and 191,000 EUR/yr and individual minimum values of 0 and 44 EUR/yr, respectively (see Table 4.9). Despite higher GDP per capita, this is explained in the model by the colder temperature, low population density and large availability of substitute lakes in the Baltic region. Despite higher GDP per capita, this is explained in the model by the colder temperature, low population density and large availability of substitute lakes in the Baltic region.

Figure 4.6. *Maps with lake values linked to the actual delivery of ecosystem services. Values are aggregated per lake.*

(A) Individual data per lake in the policy sample.



(B) Data grouped (averaged) at country level.

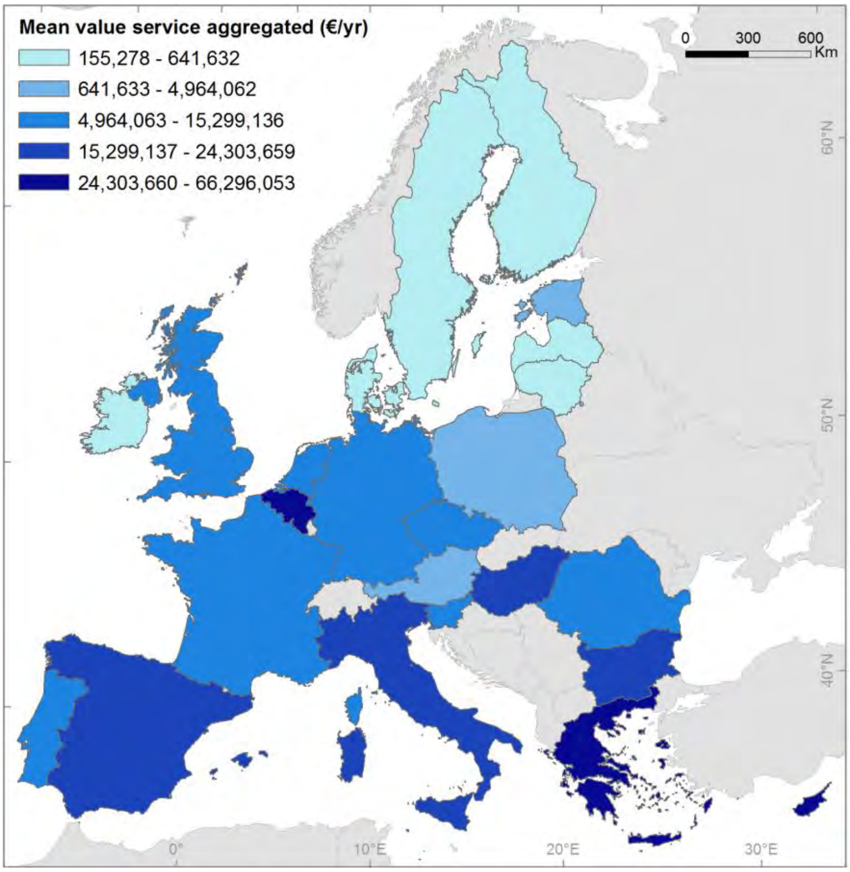
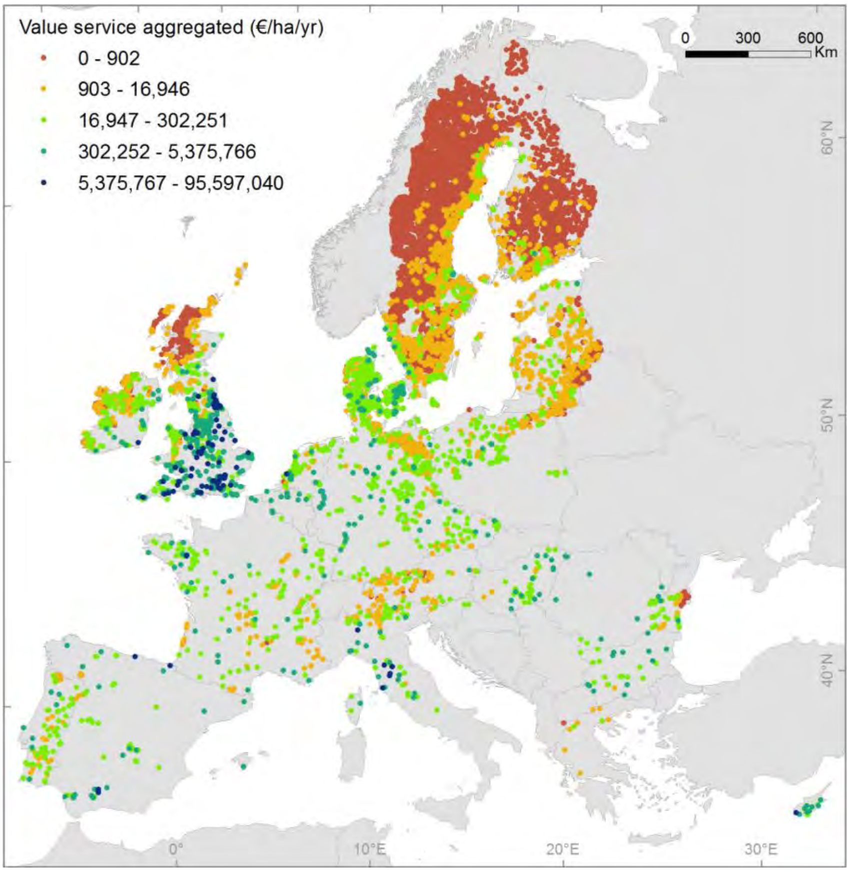


Figure 4.7. Maps with lake values linked to the actual delivery of ecosystem services. Values are aggregated per area.

(A) Individual data per lake in the policy sample.



(B) Data grouped (averaged) at country level.

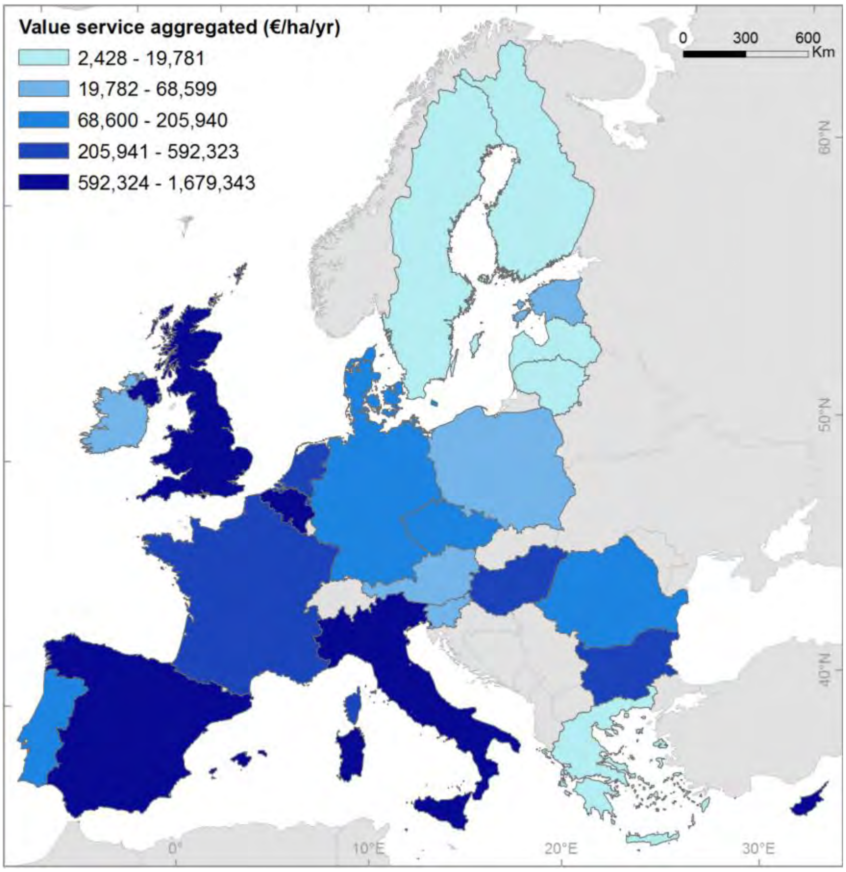
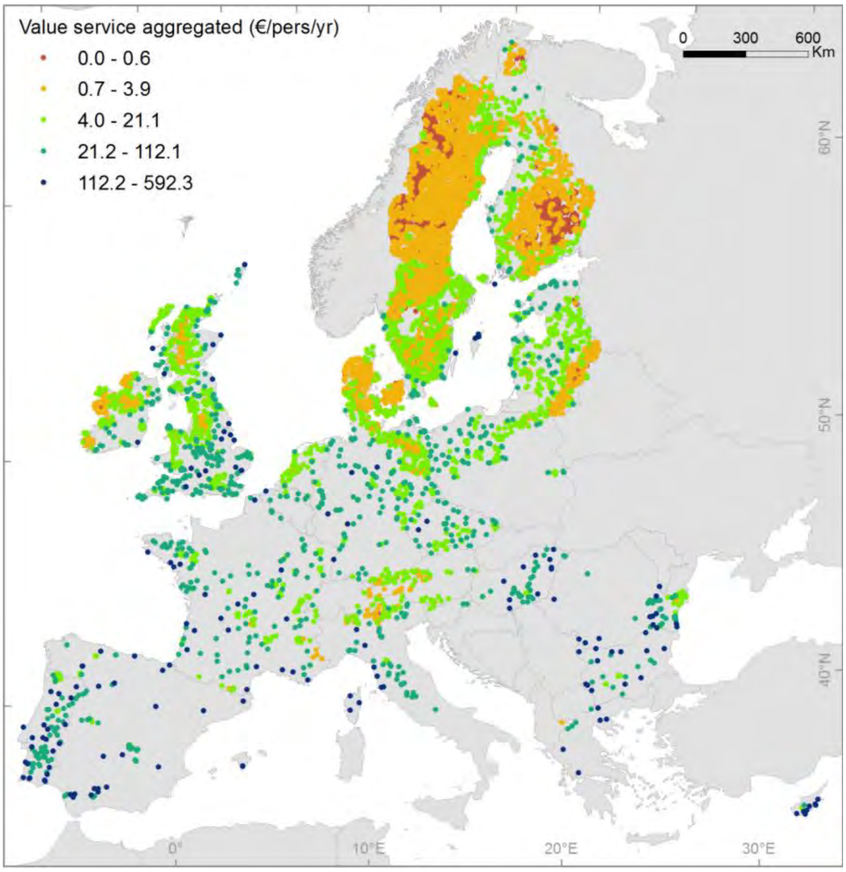
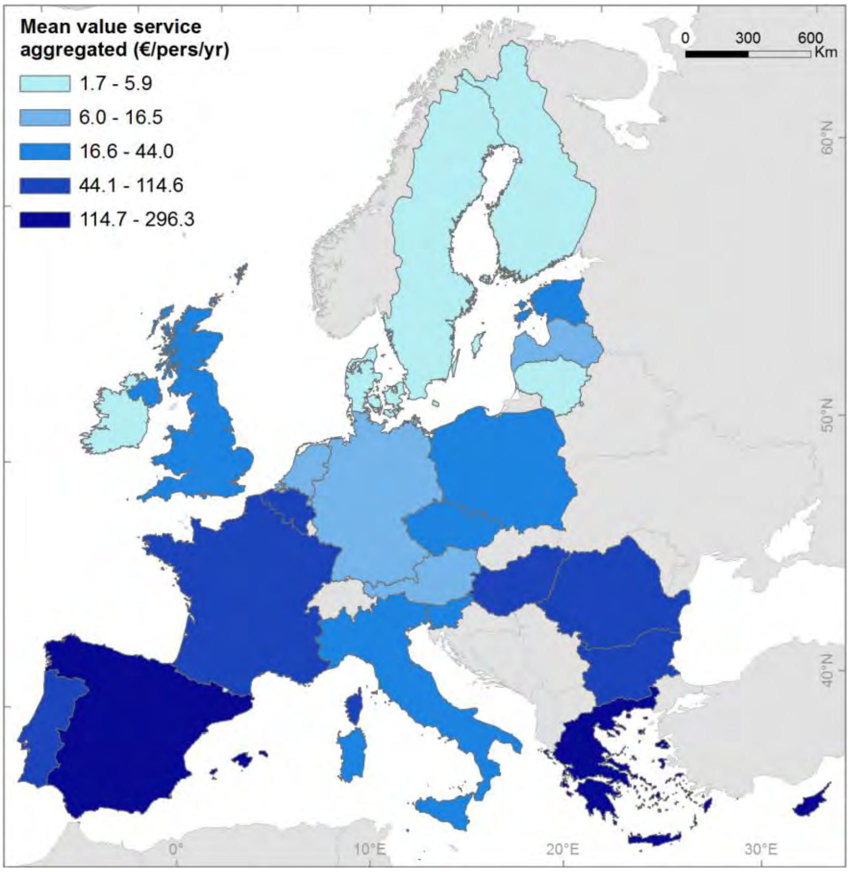


Figure 4.8. Maps with lake values linked to the actual delivery of ecosystem services. Values are aggregated per capita.

(A) Individual data per lake in the policy sample.



(B) Data grouped (averaged) at country level.



4.3 Scenario analysis

In the EU lake policy sample, we have 1416 lakes in bad or poor ecological status (Fig. 4.3). In this section, we test some scenarios to value a potential improvement in the ecological status of European lakes. In particular, we analyse the economic benefit from passing all the lakes in poor or bad ecological status into at least moderate ecological status. The aim is to estimate the value for a potential restoration programme that would improve the environmental status of all those lakes.

A particularity of lakes in poor or bad ecological status is that they are located in areas with high population density. On average in Europe, 24 678 people live within a 10 km distance from a lake, 90 504 at less than 20 km and 192 372 at less than 30 km. There in a negative monotonic relationship between the ecological status and the population living near the lake. For instance, the population living within a radius of 10 km from a lake is continuously increasing from ca. 8 000 inhabitants if the lake is in high ecological status to ca. 62 500 if it is in bad ecological status. Similar results hold with a 20 km and a 30 km radius. This relationship is expected to affect the pressures acting over water bodies. Indeed, the presence of urban areas and nutrient pollution are among the major pressures affecting the ecological status of surface waters in European river basins (Grizzetti et al. submitted).

This population distribution is a crucial aspect for the valuation of the scenarios of change. Indeed, a restoration programme that promotes all lakes in a poor or bad ecological status into at least a moderate ecological status will improve the welfare of people (and the associated economic valuation) through two different canals. First, it will increase the economic value of the biophysical potential to provide services by these lakes (negative sign in the meta-regression for lakes being in a poor or bad ecological status). Secondly, it will increase the economic value for the delivery of services in particular due to the high population density around those lakes.

In Table 4.10 we present the economic assessment of the scenario of restoration of European lakes in poor or bad ecological status. We distinguish economic values for the biophysical potential and for the delivery of ecosystem services. The “baseline scenario” corresponds to the existing ecological status and values, as reported in the previous sections. The “restoration scenario” represents the value attributed to lakes when all of them are at least in moderate ecological status. Considering the biophysical potential of the full lake policy sample to deliver services, the average lake economic value increases from 203 to 219 EUR/person/yr (an increase of 7.7%) from the baseline to the restoration scenario. Focusing on the lakes with poor or bad ecological status, we observe that the mean economic value for lake ecosystem services’ potential is significantly lower than the one obtained on the full sample (171 instead of 203 EUR/person/yr). This implies that people in Europe value less the lakes in poor or bad ecological status. The impact of the restoration scenario in this case is very important, increasing the economic value by 81.5%. Similarly, the change from the baseline to the restoration scenario of the lakes presently in poor or bad ecological status causes an increase of 81.6% in the economic value of the actual delivery of services, from 5.1 to 9.3 M EUR/person/yr. Such an increase reaches 16.1% (from 2.9 to 3.4 M EUR/person/yr) considering the full policy sample, highlighting not only the rise in potential values but also the influence of the population concentrated mostly around the restored lakes.

Table 4.10. Economic assessment of the restoration scenario of European lakes in poor or bad ecological status.

	All policy lakes (12 590 obs.)				*Lakes in poor or bad ecological status (1 416 obs.)*			
	Mean	*Median*	*Min*	*Max*	*Mean*	*Median*	*Min*	*Max*
Economic value of lake biophysical potential (EUR 2010 per person per year)								
- Baseline scenario	203.1	159.1	1.4	2 595.5	171.2	136.8	5.3	1 435.3
- Restoration scenario	218.8	177.0	1.4	2 605.1	310.7	248.3	9.6	2 605.1
- Value change (%)	+7.7%	+11.3%	+0.0%	+0.4%	+81.5%	+81.5%	+81.1%	+81.5%
Economic value of ecosystem services delivery (million EUR 2010 per year)								
- Baseline scenario	2.92	0.038	0.00	687.21	5.11	0.322	0.00	267.71
- Restoration scenario	3.39	0.040	0.00	687.21	9.28	0.585	0.00	485.93
- Value change (%)	+16.1%	+5.3%	0.0%	0.0%	+81.6%	+81.7%	0.0%	+81.5%

Note: The valuation of ecosystem services delivery is obtained by aggregating benefits of the population located up to 30 km from the considered lake. We use a linear distance decay function assuming that the value of ecosystem services for a person located at the shoreline of a lake corresponds to the total value of the potential to provide services by that lake. For a person located at more than 30 km from the lake, the lake value is equal to zero. Scope effect is included.

4.4 Conclusions

This chapter shows an attempt to value the ecosystem services provided by European lakes. The methodology proposed is based on the best available data, but it has some limitations like the relatively small number of case studies (valuations), the partial coverage of ecosystem services, the integration of different valuation methods, or the assumptions of the meta-regression model. Still, even if we acknowledge that this kind of monetary assessments captures only partially the total importance of natural ecosystems (in this case, lakes) and their ecosystem services, they can be crucial to take into account the so-called externalities in economic accounting and in decision-making affecting ecosystems (TEEB 2010).

Under current conditions and the assumptions of this study, the average economic value of ecosystem services delivered by a European lake is 2.92 million EUR per year. This aggregated value corresponds to the average number of persons living at less than 30 km of a lake, which is 192,372. The median economic value of ecosystem services delivered by a European lake is 0.038 million EUR per year. The distribution of values is highly skewed with a long tail of low values and a few very high values. This is related to the very uneven distribution of the population over space and across lakes. In fact, in our lake policy sample, for one-quarter of the lakes, the population located at less than 30 km is smaller or equal to 4,400 persons. At the other extreme, for the 50 lakes located in areas with the highest population density (almost all of them being located in the UK) the aggregated population is greater than 3.5 million of inhabitants.

Still considering the full lake policy sample, we find a high impact of the restoration scenario on the economic value of ecosystem services delivery: the average economic value passes from 2.92 to 3.39 million EUR per year, which represents an increase of 16.1%.

It is also interesting to focus specifically on lakes in a poor or bad ecological status in the baseline scenario. The average economic value of ecosystem services delivery by a lake in a poor or bad ecological status is 5.11 million EUR per year, a figure 75% higher than the one obtained for all European lakes whatever their ecological status. This result is driven by the fact that lakes in a poor or a bad ecological status are located in areas with high population density. Again, we notice a highly skewed distribution of values with a long tail of low values and a few very high values. Hence, there is a very high impact of the remediation scenario for lakes in a poor or a bad ecological status, where the average economic value of ecosystem services delivery passes from 5.11 to 9.28 million EUR per year (an increase of 81.6%).

Finally, even if these numbers can be highly speculative, we can estimate the total gain from passing all lakes in a poor or bad ecological status (for the full lake policy sample) to at least a moderate ecological status. Then the average gain per lake represents 0.47 million EUR per year, which translates into an aggregated value of 5.9 billion EUR per year for our 12,590 European lakes (which represent 67% of the total European lakes' surface). Based on a EU28 population of 506 million inhabitants in 2014, the benefit to be expected from passing all European lakes in a poor or bad ecological status into at least a moderate ecological status corresponds to 11.7 EUR per person and per year.

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Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

Deliverable 5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis.

D5.1-4: Effects of multiple stressors on ecosystem structure and services of phytoplankton and macrophytes in European lakes

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Due date of deliverable: **Month 36**
Actual submission date: **Month 36**

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)	

Content

Non-technical summary

The aim of this deliverable was to assess the impacts of multiple stressors on lake ecosystems at the European scale. We have examined ecological responses of two main biological groups (quality elements), namely algae (phytoplankton) and other aquatic plants (macrophytes), to a range of stressor combinations in large populations of lakes. Moreover, the impacts of future multiple stressor scenarios - future climate and nutrient concentrations - have been assessed for a phytoplankton community index.

While nutrients are a key stressor in all regions of Europe, MARS also focuses on the following key environmental changes for specific regions: water scarcity and flow alterations (Southern Europe); changes in hydrology and morphology (Central Europe); and changes in hydrology and temperature (Northern Europe). More specifically, the stressors that have been investigated in this report are related to increased air temperature and precipitation, hydropower and water abstraction for irrigation and public water supply, hydrological changes (flushing or water level changes), salinisation, or increase in humic substances ("brownification"). We have analysed effects on ecological status (ecological quality ratio values), and in addition a set of indicators of environmental stressors for both biological quality elements. For phytoplankton, the main indicators analysed were chl-a, abundance of cyanobacteria (a group of potentially harmful algae) and PTI (phytoplankton trophic index). For macrophytes, the main indicators were the water-drawdown index (WIc) for regulated lakes, a proportion of macrophyte coverage (%PVI), and other indices based on specific species or species groups. Interactions within the lake community, including also zooplankton (small crustaceans), were addressed by analysis of data from mesocosms across Europe. Potential effects of multiple stressors on ecosystem services (e.g. nutrient retention, nutritional value of fish, and cultural services to lake visitors) have also been investigated by case studies and national datasets. The main large-scale data sources used in our studies include the European Environment Agency's WISE-SoE datasets (Waterbase), data compiled during previous EU projects (WISER), and national monitoring data. Moreover, information on lake and catchment characteristics (such as land use) was obtained from the MARS geodatabase. The natural characteristics of lakes (such altitude, surface area, mean depth, alkalinity and humic level) were explicitly considered in most of the studies, either as co-variables or as determinands of lake types.

The analysis of EEA's water quality data in combination with land use data showed that, not surprisingly, total phosphorus (P) concentration in lakes clearly increased and Secchi depth (transparency) generally decreased with increasing proportion of arable and pasture lands in lake catchments across Europe. Total P was the stressor that correlated best with ecological status of phytoplankton, while Secchi depth better explained the ecological status of macrophytes. Climatic variables such as air temperature and precipitation, in contrast, had apparently no effect on the ecological status. This result does not contradict that climate change may cause additional stress for lake ecosystem. Instead, the space-for-time approach (using geographic variation in climate as a substitute for temporal variation) in these analyses may not be the most appropriate for detecting real effects of climate change. For the individual phytoplankton indicators

(cyanobacteria and PTI), interactions between effects of nutrients and climatic stressors (temperature and/or precipitations) were found for some of the lakes or lake types. For example, the analysis of time series indicated that cyanobacteria are most favoured by nutrient stress in lakes of low nutrient status and sensitive to summer rainfall in short residence time lakes. However, the studies also revealed large variation in the combined stressor effects among the different lakes types. It was therefore difficult to generalise such results across lake types. For the PTI index (Northern Europe), the strongest interaction between nutrients and temperature stress was found for lowland siliceous lakes. We used this empirical relationship to predict the future PTI scores for this lake type under the MARS future climate scenarios. According to our model, increased temperature and precipitation will result in higher PTI scores, indicating impaired ecological status. In the short term (2030), however, climate-induced changes in PTI will probably not be sufficient to change the ecological status class of lakes (e.g., from Good to Moderate).

The analysis of Mediterranean (Turkish) lakes suggest that warming together with expected changes in land use in this regions may result in higher salinisation and eutrophication with more frequent cyanobacteria blooms and loss of biodiversity. Consequently, under such conditions, the ecosystem services potential (e.g. drinking and irrigation water, biodiversity etc.) are likely to be deteriorated if not lost completely. To counteract, stricter control of nutrients emissions and human use of water is urgently needed.

The interactions between nutrients and climatic stressors could most clearly be interpreted from the experimental data based on former mesocosm experiments. For example, these experimental results indicate that global climate warming might favour growth of macrophytes at moderate water level decrease southern regions, even under relatively eutrophic conditions. However, if the water level decrease becomes so extreme that macrophytes are directly negatively affected, and longer and intense drought periods become more common, the combined effects of eutrophication and extreme water level reductions may adversely affect the development of macrophytes. In contrast, warmer temperatures in northern regions may hamper macrophyte growth due to increased precipitation and, thus increased water levels and nutrient loading.

The MARS project have resulted in much new information on the combined effect of eutrophication and climate change and their interactions on trophic structure and dynamics - showing that combined effects through a series of cascading events can lead to deterioration in water quality and ecological status - there are still some knowledge gaps to be filled. Knowledge on differences in interactions along altitude, latitude and other biogeographical gradients are needed before firm and safe conclusions relevant for managers and WFD can be drawn. We also need more knowledge on the resilience of lake community structure and dynamics to extreme climatic events such as heat waves, drought, and heavy rainfall, since we can expect an increase of such events.

1. Introduction

1.1. Background: multiple stressors for lakes in Europe

The aim of MARS task 5.3 is to analyse the impacts of multiple stressors on lake ecosystems over large spatial scales. In particular, we will examine ecological responses of primary producers in large populations of lakes, assess the impacts of future multiple stressor scenarios, and use the results to inform land-use policy and lake management at a European or regional scale.

Europe's lakes are impacted by multiple stressors, which affect ecological and chemical status, water quantity and ecosystem functions and services. According to Europe's first River Basin Management Plans (RBMPs), 44% of lakes failed to achieve the good status targets of the Water Framework Directive (EC (European Commission), 2000) (European Environment Agency, 2012). There are, however, strong regional differences, and the reasons are manifold. The EEA (European Environment Agency, 2012) report lists the most important pressures impacting individual water categories. Two pressures prevail: diffuse pollution and hydromorphological degradation (both >30% of lakes). Viewed in more detail, both diffuse pollution and hydromorphological degradation are composed of several individual components with complex interactions. While single stressors such as strong organic pollution or acidification are declining, European lakes and water resources are now affected by a complex mixture of stressors resulting from urban and agricultural land use, water power generation and climate change (Hering et al., 2015).

The occurrence of multiple stressors in water bodies and their abiotic and biotic responses were quantified as items of ecological evidence in a recent literature review (Nõges et al., 2016), based on the MARS deliverable no. 2.1-1. Despite the existence of a huge conceptual knowledge base in aquatic ecology, few studies actually provided quantitative evidence on multi-stress effects. For lakes, the main drivers were found to be agriculture + aquaculture (87 evidence items) and climate warming (72). The main stressors for lakes were nutrients stress (119 evidence items), hydrological stress (61) and thermal/optical stress (47). Considering multi-stress combinations, the three most studied 2-stressor combinations included nutrients as one of the stressors. Models that included more than one stressor could generally explain the responses better (higher R^2) than single-stressor models. Stressors interactions were quantified in only 15% of the studies: in those cases, synergistic interactions (6%) were more common than antagonistic interactions (3%).

A comparison of the biological indicators to multiple stressors showed that fish and macrophytes were the favourite indicators to study multiple stressor effects in lakes (both ~ ⅓) followed by zooplankton (Nõges et al., 2016). Phytoplankton studies formed only 10%; possibly because phytoplankton typically shows a strong response to a single stressor (total P). This implies that our assessment of multiple stressors in lakes at European scale should pay attention to all of these biological quality elements. However, the Pan-European datasets available for our assessment were dominated by phytoplankton. Therefore, the responses of other important biological indicators (macrophytes and to some degree zooplankton) will be analysed for more limited geographic regions. The responses of fish are reported in another deliverable (no. 5.1-5).

1.2. Approaches for analysing multiple stressors at the European scale

General approach

The conceptual model for MARS (Figure 1) is a framework that explicitly links the assessment of environmental risk (left column), ecological status (middle column) and ecosystem services (right column). Our report focusses on multiple stressor effects on indicators of ecological status (sensu WFD) for phytoplankton and macrophytes, and therefore follow the DPSIR scheme. Drivers (D, e.g. land use or climate change) affect pressures (P, e.g. increased nutrient loads), which in turn affect the lake state (S) of both abiotic and biotic elements. In the MARS terminology (http://www.freshwaterplatform.eu/index.php/glossary.html), "stressor" can represent both drivers, pressures and abiotic state.

Key stressor combinations have been identified for different geographic regions of Europe (Hering et al., 2015). While nutrients are a key stressor in all regions, we will also focus on the following key stressors: water scarcity and flow alterations (Southern Europe); hydrology and morphology (Central Europe); and hydrology and temperature stress (Northern Europe).

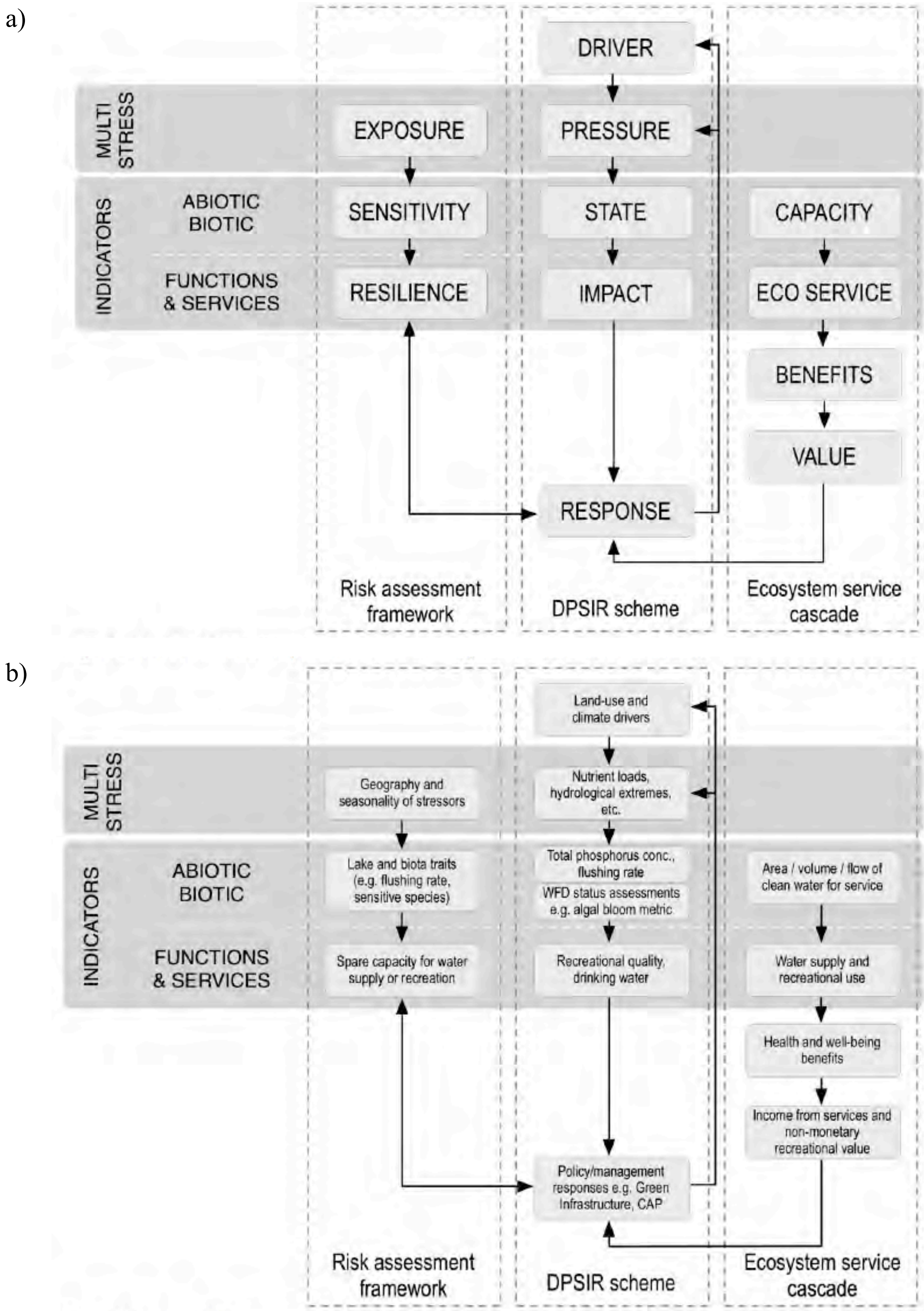

Figure 1. The MARS conceptual model for an integrated assessment framework (a), exemplified for a lake affected by intense agriculture and climate change (b).

Common benchmark indicators (BInd) have been defined for MARS (Deliverable 2.1-3) to enable comparison of responses to multiple stressors across different scales. In this report, the following benchmark indicators are analysed.

- **BInd01: Ecological status of surface water body**. Ecological status of phytoplankton and macrophytes, respectively, are response variables in Chapter 2.1.
- **BInd02: Total phosphorus concentration in the water column**. Total P is a key predictor variable in most of the chapters.
- **BInd03: Total nitrogen concentration in the water column**. Total N is a predictor variable in several chapters: 2.1, 5.1, 4.4 and 6.1. N:P ratio is used in Chapter 2.2.
- **BInd06: Annual water-level fluctuations**. Water level changes and hydrological stress are key predictor variables in Chapter 4 (caused by regulation of lakes) and Chapter 5 and 6 (representing both climate-relaterd drought and water abstraction for other purposes).
- **BInd08: Growing season mean of water column chlorophyll-a concentration**. Chl-a is a response variable in Chapters 2.1, 2.3 and 6.1.
- **BInd10: Biovolume of toxic/nuisance phytoplankton species**. The concentration of cyanobacteria is a response variable in Chapters 2.2 and 2.3.
- **BInd11: Abundance of submerged, emergent and floating-leafed macrophytic vegetation.** The abundance of macrophytes or macrophyte groups are analysed are response variables in Chapters 5.1, 4.1, 4.2, 4.3 and 5.1.

The effects of multiple stressors on ecosystem services is a complex issue, which needs to be understood at the scale of individual lakes or smaller regions. A methodology for evaluation of ecosystem services is described in MARS Deliverable 2.1-2. We have not attempted to analyse effects of multiple stressors on ecosystem services for whole regions of Europe, but will give examples for one individual lake (Lake Võrtsjärv in Estonia, Chapter 4.4) and for a country (ca. 700 lakes in Finland, Chapter 3.1)

The importance of natural characteristics of lakes, such as altitude, surface area, mean depth, alkalinity and humic level, is addressed in most of the chapters. The differences among lakes are accounted for both by using typology factors as individual predictor variables (e.g. Chapter 2.1) and by grouping the lakes into types (e.g. Chapter 2.2). A new set of broad types for rivers and lakes that was recently been published by the EEA (ETC/ICM, 2015) has been applied for this purpose. The broad lake types are described in Table 1. These lake types were assigned to all lakes in the MARS geodatabase (see below). The broad lake types are by default clear, unless they are described as Organic (humic).

Table 1. Broad lake types based on the most commonly used typology factors for WFD national types (ETC/ICM, 2015). Number of WBs refer to the WFD RBMP database.

Broad lake type name	Broad Lake type code	Altitude (masl)	Lake area (km^2)	Geology	Mean depth (m)	Stratification	Number of national types	Number of WBs	% of WBs
Very large lakes, shallow or deep and stratified (all Europe)	1	any	>100	any	> 3	stratified	6	126	0.7 %
Lowland, Siliceous	2	<200	<100	Siliceous	>3	stratified	34	2059	12.0 %
Lowland, Stratified, Calcareous/Mixed	3	<200	<100	Calcareous/Mixed	>3	stratified	41	1721	10.1 %
Lowland, Calcareous/Mixed, Very shallow/unstratified	4	<200	<100	Calcareous/Mixed	≤3	unstratified	39	1045	6.1 %
Lowland Organic (humic) and Siliceous	5	<200	<100	Organic (humic) and Siliceous	> 3	stratified	23	2275	13.3 %
Lowland Organic (humic) and Calcareous/Mixed	6	<200	<100	Organic (humic) and Calcareous/Mixed	>3	stratified	13	130	0.8 %
Mid altitude, Siliceous	7	200 - 800	<100	Siliceous	>3	stratified	43	2673	15.6 %
Mid altitude, Calcareous/Mixed	8	200 - 800	<100	Calcareous/Mixed	>3	stratified	27	281	1.6 %
Mid-altitude, Organic (humic) and Siliceous	9	200 - 800	<100	Organic (humic) and Siliceous	>3	stratified	11	1381	8.1 %
Mid-altitude, Organic (humic) and Calcareous/Mixed	10	200 - 800	<100	Organic (humic) and Calcareous/Mixed	>3	stratified	4	24	0.1 %
Highland, Siliceous (all Europe), incl. Organic (humic)	11	>800	<100	Siliceous	>3	stratified	15	539	3.1 %
Highland, Calcareous/Mixed (all Europe), incl. Organic (humic)	12	>800	<100	Calcareous/Mixed	>3	stratified	10	48	0.3 %
Mediterranean, small-large, siliceous	13	< 800	0.5-100	Siliceous	any	any	11	129	0.8 %
Mediterranean, small-large, Calcareous/Mixed	14	< 800	0.5-100	Calcareous/Mixed	any	any	13	121	0.7 %
Mediterranean, Very small	15	< 800	<0.5	any	<15	any	0	0	0.0 %
					Total		**290**	**12552**	**73.3 %**

Notes: WBs is waterbodies, "% of WBs" is % of WBs in all Member States included in the analysis of national WFD types. Many large lakes are split into smaller water bodies, and thus do not appear as large lakes

Main data sources

The fundament for our work has been large-scale European datasets from lakes that were already compiled during previous projects. This decision was made because compilation of large-scale biological and environmental data is very resource-demanding, and not possible within the frame of this task. While certain relevant pan-European datasets were available, such datasets typically have lower information (e.g. temporal and taxonomic resolution) than datasets on national or smaller scale (Figure 2). The more detailed datasets were needed to analyse certain types of responses, e.g., community structures and ecosystem services, although on smaller geographic scale. In order to analyse different types of ecological responses, we therefore made a selection of datasets ranging from individual lakes to European scale. The main data sources are described in the following.

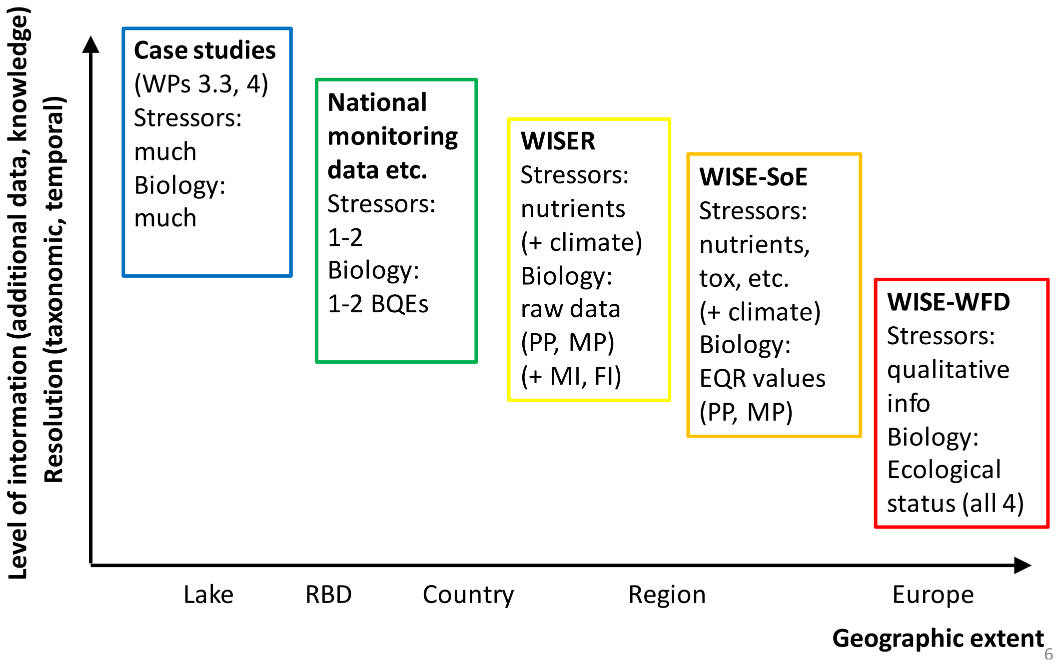

Figure 2. Types of datasets used in MARS Task 5.3 for analysing multiple stressors in lakes at different geographic scales. PP = phytoplankton, MP = macrophytes, MI = macroinvertebrates, FI = fish. EQR = ecological quality ratio. WISE = Water Information System for Europe. SoE = State of Environment. WISER = a former EU project. BQE = biological quality element.

Abiotic stressors and biological indicators

Individual lakes. Data from a single lake (Võrtsjärv) was used as a case study in for ecosystem services (Chapter 4.4).

Country scale. Data from national monitoring or covering a large part a country were used for analysing macrophytes in Norway, Finland and Ireland (Chapter 4.2-4.3) and for whole lake communities in Turkey (Chapter 6.1).

Gradient across Europe. Data from mesocosms in 6 countries representing a North-South gradient was used to analyse responses of macrophytes and zooplankton (Chapter 5.1).

Regions of Europe. Time series from 26 individual lakes have been combined from 8 countries for analysis of phytoplankton (cyanobacteria) (Chapter 2.3). Data from the former EU project WISER covering large regions of Europe have been used for analysing different phytoplankton indices WISER (Chapters 2.2, 2.5, 2.2). Data on ecological status for phytoplankton and macrophytes from 18 countries across Europe was obtained from the European Environment Agency's WISE-SOE database (Water Information System for Europe - State of the Environment).

Pressures and additional data

The individual datasets used are described in each chapter. In addition, the following data were made available to the whole task.

Pressure data: Categorical pressure information for individual lakes was obtained from WFD database (see Chapter 2.1).

Climate data: Data on air temperature, precipitation and wind for all of Europe were obtained from the Agri4Cast Data portal (http://agri4cast.jrc.ec.europa.eu/DataPortal) of the Joint Research Centre (JRC). The CGMS (Crop Growth Monitoring System) database contains meteorological parameters from weather stations interpolated on a 25x25 km grid. Meteorological data are available on a daily basis from 1975 to the last calendar year completed, covering the EU Member States, neighbouring European countries, and the Mediterranean countries. These data are used in Chapters 2.1, 2.2, 2.5 and 2.2.

Lake characteristics*:* extensive data on lake and catchment characteristics were available from the MARS geodatabase (available from http://www3.fgg.uni-lj.si/~/mars/MARSgeoDB/), as documented in the report "Catalogue of MARS geodatabase vector features, version 2" (available from the same URL).

Lake types*:* broad lake types (Table 1) were obtained from the MARS geodatabase. The procedure documented in the report "Matching codes from WFD SWB database and WISE SoE data base to MARS rivers and lakes and determining broad types for rivers and lakes", which is published as an appendix to MARS deliverable no. 5.1-1.

Land use: The percentages of land cover types for the catchment of each lake was calculated from Corine Land Cover data (100 m x 100 m grid) and stored in the MARS geodatabase (used in Chapter 2.1 and 2.2). The full procedure is described in Appendix 1.

MARS storylines and future scenarios

Within MARS, future climatic and socio-economic scenarios have been developed (deliverable no. 2.1-4) (Faneca Sanchez, 2015). These scenarios provide both a qualitative framework and, where possible, quantitative data for modellers to run simulations. A selection of scenarios have been used to define the three MARS storylines, "Techno world", "Consensus world" and "Fragmented world", which are described in the fact sheet "MARS scenarios and storylines" (http://mars-project.eu/files/download/fact_sheets/MARS_fact_sheet03_storylines.pdf).

In task 5.3, we have made use of the future climate data provided by the MARS scenarios. The future climate data contain daily values of air temperature, precipitation, wind and other variables on a 0.5 ° x 0.5 ° grid, for the period 2006-2095. The future climate data are obtained from two different climate scenarios, the Representative Concentration Pathways (RCP) 4.5 and 8.5. These climate scenarios are used in the "Consensus world" and "Fragmented world" storylines, respectively. We have used future climate scenarios to predict the response of a phytoplankton community index to future climate in Northern Europe, using an empirical model that relates the response of this index to combined nutrient and climatic stress (Chapter 2.5).

2. Stressor combination 1: Nutrients, temperature and precipitation

2.1. Effects on ecological status of phytoplankton and macrophytes (Europe)

Contributors: Jukka Arovita, Peeter Nõges, Tiina Nõges, Jannicke Moe, Anne Lyche Solheim

Summary

We analysed the relationships between multiple stressors (nutrients, climate, land use) and ecological status of phytoplankton and macrophytes in lakes, using a pan-European dataset (WISE-SoE) from the European Environment Agency. Ecological status is a measure of the deviance from the reference (undisturbed) condition of a biological index, for a given lake type and geographic region. Random Forest models were used to identify the pressures and abiotic stressors with highest effect on the normalised EQR (ecological quality ratio) values of these biological quality elements, as well as the importance of natural lake characteristics (typology variables). For phytoplankton EQR, the main stressor was total P, while Macrophyte EQR values increased most strongly with Secchi depth and was less reduced by total P. The EQR of both biological quality elements increased with mean depth and decreased somewhat with alkalinity. Both total P and Secchi depth was strongly affected by the land use of the catchment; mostly to the percentage of arable land and to some degree by the percentage of pastures. In addition, the mean depth and the altitude affected the total P concentrations and the Secchi depth. Climatic variables had little impact on the ecological status. This is reasonable since the climate data in this dataset represent geographic variation (rather than temporal variation), to which the local biota is adapted.

Introduction

The WFD requires monitoring and assessment of ecological status of water bodies using biological quality elements (BQEs) and supporting abiotic quality elements. Ecological status is a measure of the deviance from the reference (undisturbed) condition of a biological index, for a given lake type and geographic region. The common management target across Europe is good ecological status of all BQEs. An important aim of MARS Task 5.3 is therefore to analyse the ecological status of phytoplankton and macrophytes in relation to concentrations of nutrients and other stressors for large regions of Europe. To align with the conceptual model of MARS (Figure 1), we have analysed the stressor-response relationships in two steps.

1) Pressure - Abiotic state: Effects of land use, climate etc. on nutrient concentrations and other physico-chemical elements (total P, total N, Secchi depth etc.)
2) Abiotic state - biotic state: Effects of nutrients in combination with other stressors on ecological status of phytoplankton and macrophytes

Our analysis also considers the importance of lake typology factors, such as altitude, depth, alkalinity and water colour level (Table 1).

Data

The main data source was the Waterbase - Lakes published by EEA in 2014 (http://www.eea.europa.eu/data-and-maps/data/waterbase-lakes-10). This database contains the WISE-SoE (State of the Environment) data reported to EEA by its member states, including biological data (ecological status and ecological quality ratios) for phytoplankton and macrophytes. The reported data are yearly aggregated values for individual lake monitoring stations. These data are also published in WISE maps:

http://www.eea.europa.eu/data-and-maps/explore-interactive-maps/phytoplankton-in-lakes

http://www.eea.europa.eu/data-and-maps/explore-interactive-maps/macrophytes-in-lakes

We extracted all available biological data, as well as the relevant data on nutrients and other physico-chemical variables and lake characteristics. The biological data were aggregated to water body (lake) before they were linked to the abiotic data from the same lake and year. The initial dataset consisted of 1554 records (lake-years) from 803 lakes (Figure 3).

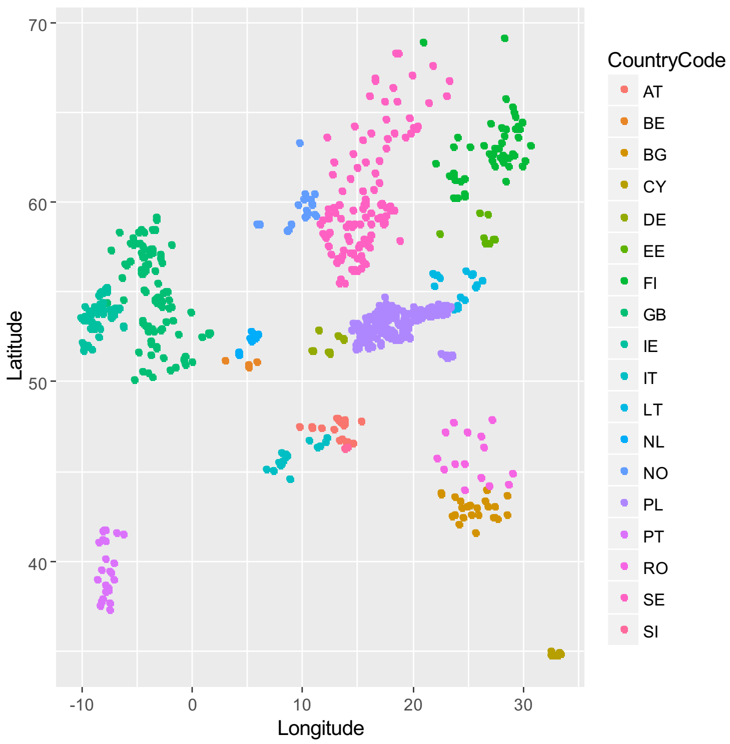

Figure 3. Location of the 803 lakes from the SoE dataset across Europe.

Lake characteristics

Natural characteristics of the lakes were mainly retrieved from the Lakes Stations table of Waterbase. Information on broad types were obtained from the MARS geodatabase. The individual typology variables were also supplemented with values from the MARS geodatabase, where possible (altitude, geology and surface area). A detailed description of the natural characteristics of the lakes is given in Appendix 2.

Biological data: ecological status and ecological quality ratios

The biological data contain the categorical ecological status classes (High, Good, Moderate, Poor, Bad) for all records. In addition, the continuous ecological quality ratios (EQR values) are available for the majority of the records. EQR values are reported by the member states to EEA as national EQR values; based on these, EEA calculates the normalised EQR values (nEQR) which can be compared across countries (for illustration, see: http://dd.eionet.europa.eu/visuals/Biology_20110617.jpg). Normalised EQR values correspond to ecological status as follows: Bad = 0-0.2, Poor = 0.2-0.4, Moderate = 0.4-0.6, Good = 0.6-0.8, and High = 0.8-1.0. In the following, "EQR" will refer to the normalised EQR values. We selected the biological determinands that represented the impact type "eutrophication", alternatively "general degradation". More details on the biological data can be found in the data dictionary (http://dd.eionet.europa.eu/datasets/latest/WISE-SoE_WaterQuality/tables/BiologyEQRData) and in the report (Moe and Solheim, 2013).

Not all lakes had all variables measured. For our study we selected those lakes where Ecological Quality Ratio (EQR) value for either of the quality elements were reported and supplemented with the dataset with national phytoplankton EQRs from Finland and Norway (Figure 4). The dataset consisted of 432 lake water bodies (795 lake-year records) assessed with phytoplankton EQR (Figure 2), and 441 lake water bodies (601 lake-year records) assessed with macrophyte EQR. In total 212 lake water bodies had an EQR for both quality elements.

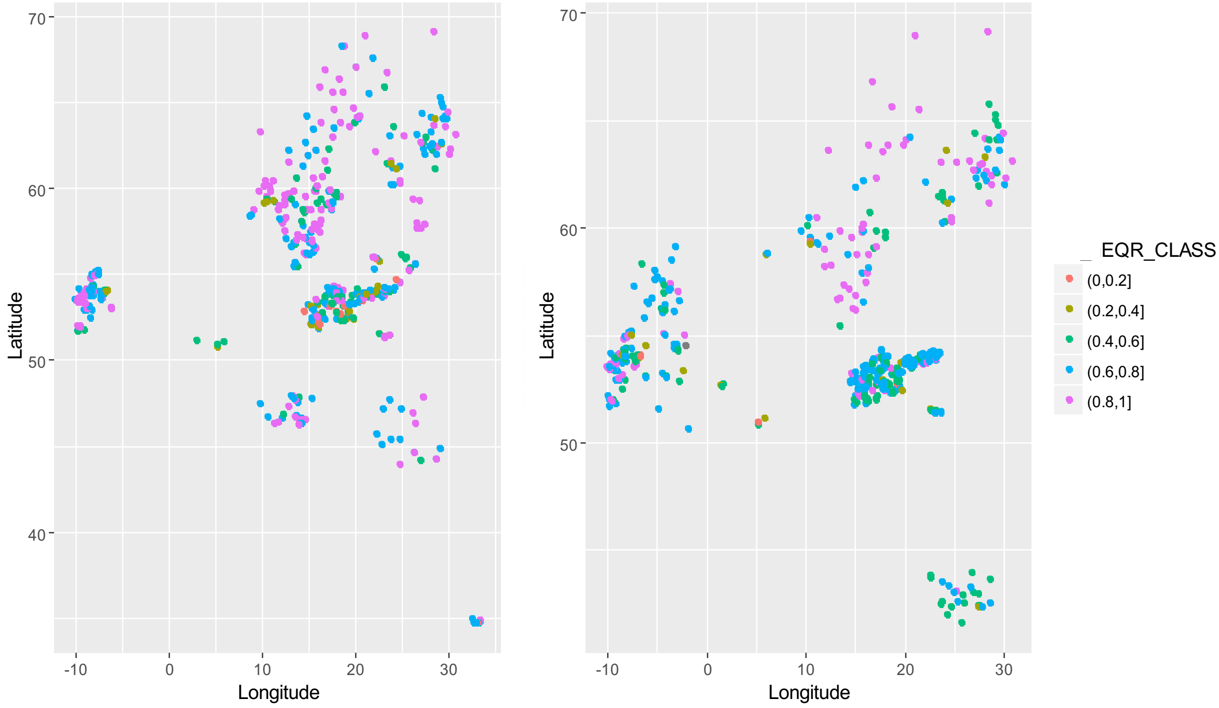

Figure 4. Location of the lakes assessed by phytoplankton (left, N = 795 from 432 lake water bodies) and by macrophytes (right, N = 601 from 441 lake water bodies) in Europe. Only lakes with a provided EQR-value are shown. The colours show intervals (status classes) of the EQR-values. Maybe further add here info of TOTP-analyses data and mark somehow data used for final analyses (TOTP: 978 samples from 537 lake water bodies, PP: 459 samples from 218 lake water bodies, MP: 423 samples from 299 lake water bodies).

Availability of the lake characteristic, water chemistry and climate data (see below) further reduced the data used in final analyses. The final dataset used in the final Random Forest models (see below) for explaining total phosphorus concentration consisted of 978 samples from 537 lake water bodies, for phytoplankton of 459 samples from 218 lake water bodies, and for macrophytes of 423 samples from 299 lake water bodies.

Water chemistry

Data on water chemistry and other physico-chemical variables were obtained from the Waterbase Lakes tables on nutrients and supporting determinands. The variables included in our analysis are: total P, total N, ammonia, alkalinity, water colour and Secchi depth. Of these variables Total P has the best coverage, while others have more missing values (e.g. 187 records lacked total N values). The water chemistry of the lakes is described in Appendix 2.

Climate

The climate data were obtained from the Agri4Cast Data portal of JRC, as described in Chapter 1. The variables included in this analysis were mean spring temperature, mean summer temperature and max summer temperature. Descriptive statistics of the climate data are provided in Appendix 2 (Table 14).

Land Cover

The Corine Land Cover data were from MARS geodatabase, as described in Chapter 1. The following variables were included in this analysis: Arable land (21) and Pastures (23).

Qualitative pressure data (WFD)

The qualitative pressure data from the WFD RBMP database were included in the dataset and quality-checked (Table 14). The links from the WFD water body codes to the SOE stations were obtained from the MARS geodatabase. For each lake, pressures are reported as by categories (e.g. "point source" or "diffuse source"). A weakness with this data source is many lakes had no pressure information; this could be interpreted either as "no pressure" or "no information reported". Because of this uncertainty, the pressure data were not included in further analysis.

Methods

We explored the importance of natural lake characteristics, climate and land use on the lake status variables. We explicitly aimed to quantify the effect of natural variable on the lake status variables because across large geographical scales nutrient levels may show marked natural variation among lakes (e.g. (Cardoso et al., 2007; Nõges, 2009; Olson and Hawkins, 2013)). We first explored the importance of natural lake characteristics, climate and land use on total phosphorus concentration (Total P) and on Secchi depth. Total P is the main nutrient stressor variable whereas Secchi depth is a widely available measure depicting decreased light conditions related to eutrophication. We then explored the importance of nutrients and climate in explaining the phytoplankton status (EQR) and chlorophyll a concentration and macrophyte status (EQR) of the lakes.

We used Random Forest (RF) models to explore the importance of the predictor variables. RF models are a useful and powerful tool for the purpose as the effects of many predictor variables can be examined simultaneously, and moreover, the models account for the effects of predictor variables so that unique effect of each predictor can be examined separately after effects of all other predictor variables are accounted for (Hastie et al., 2001).

We developed the RF models iteratively so that first all potential predictors were included (see e.g. (Hill et al., 2013)). A stepwise removal of the least important predictors was done until performance was maximized (as measured by % variance explained). We then used partial dependence plots (De'ath, 2007) to show the relationships between lake status and each of the most important predictor variables The partial dependence plots characterize the average unique effect of each predictor on the response variable.

Results

General gradients and patterns

Across the whole SoE data, gradient in % of arable in the lake catchments ranged from 0% to 90% (mean 18%), Secchi depth from 0.19 m to 13.3 m (mean 2.56 m) and Total P from 1.40 µg/L to 1003 µg/L (mean 45 µg/L). Secchi depth generally decreased and total phosphorus concentration (Total P) clearly increased with increasing % of arable in the lake catchments (Figure 5). By contrast, EQRs of phytoplankton (PP) and macrophytes (MP) did not show a clear trend with increasing % of agriculture across the whole SoE data, indicating difference in the biological responses to same level of pressure in different lakes and regions. PP EQR was generally higher in lakes with large Secchi depth (large water transparency), whereas in lakes with small Secchi depth, PP EQR was variable (Figure 6). PP EQR decreased clearly with increasing Total P. MP EQR showed a stronger relationship to Secchi depth increasing in larger Secchi (Figure 6). MP EQR also decreased with increasing Total P, but the relationship was weaker than with PP EQR.

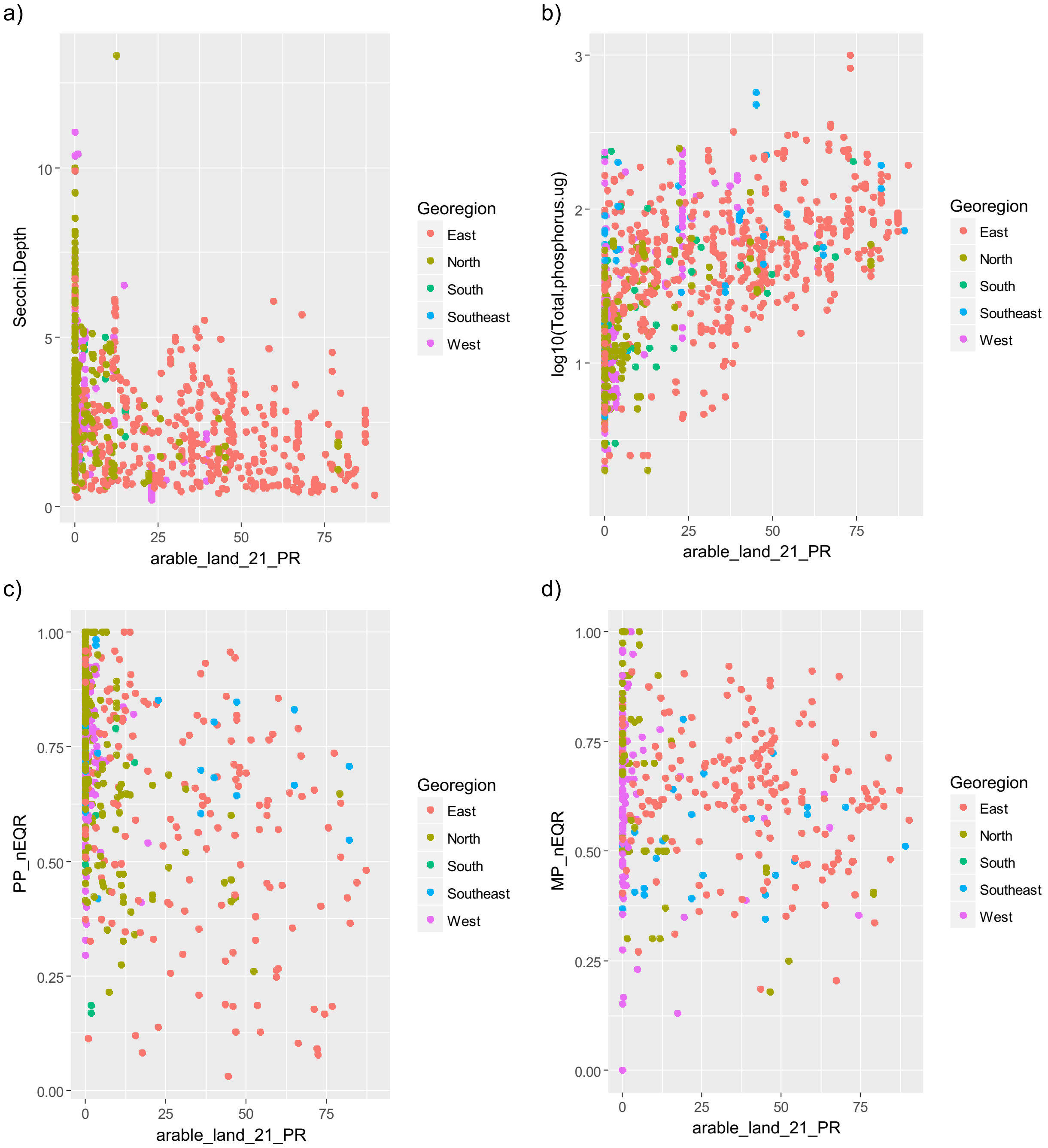

Figure 5. Relationship of Secchi depth (m), total phosphorus concentration (µg/L) and EQRs of phytoplankton and macrophytes to % of arable land in the lake catchments in the SoE dataset across five geographical regions in Europe.

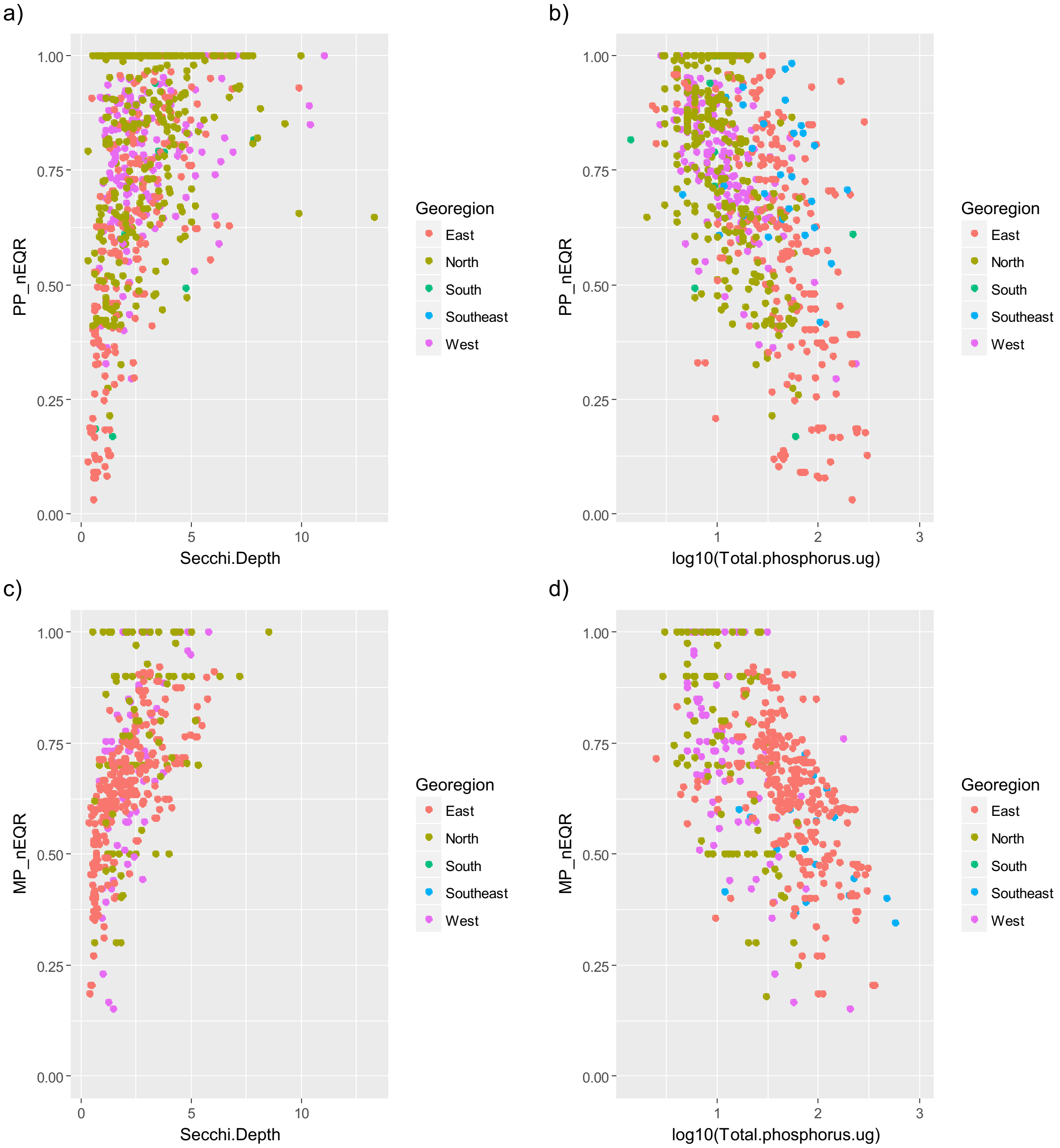

Figure 6. Relationship between EQRs of phytoplankton (upper panel) and macrophytes (lower panel) to Secchi depth (left) and total phosphorus concentration (right) in the SoE lake dataset across five European regions.

Predictors of total P concentrations

Two land use variables and two natural lake characteristics were important predictors of total phosphorus concentrations, accounting altogether 70% of variance (Table 2, Figure 7). Total P increased particularly with increasing % of arable land, but also with % of pastures in the catchment. On average, Total P concentration increased uniformly from about 30 µg/L to more than 100 µg/L as % of arable land in the catchment increased from 0 to 70% (Figure 8a). Also % pastures increase from 0 to ~10% had a unique effect on Total P concentration which then increased from ~ 40 µg/L to 60 µg/L (Figure 8b). Total P had also natural variation that was not related to land use (Figure 8c and d). Total P decreased from 70 µg/L to ~ 40 µg/L when lake mean depth increased from 1 m to ~7 m. Also lake altitude had a negative effect on Total P so that when altitude increased from 0 m to ~ 200 m, Total P decreased from 60 µg/L to ~ 40 µg/L (Figure 8d). The stepwise removal of predictors showed that other variables, including all climate variables, did not show marked unique importance in explaining Total P and thus did not increase the % variance explained.

Table 2. Summary of Random Forest (RF) models predicting total phosphorus concentration, Secchi depth, Chlorophyll a concentration, phytoplankton (PP) EQR and macrophyte (MP) EQR in the SoE lakes. The water chemistry stressor models were built with natural and agricultural land use pressure predictor variables. EQR values were predicted with natural and water chemistry stressor variables. Predictors are listed in order of decreasing importance in the model. Signs indicate direction of the variable in response to predictor. Total P and chlorophyll a increase with agricultural pressure, other variables decrease. The column "% var" shows the percentage of variation explained by the model.

Variable	% var	Predictors
Total P	70	% of arable land in the catchment (+), % of pastures in the catchment (+), lake mean depth (-), lake altitude (-)
Secchi depth	71	lake mean depth (+), % of arable land in the catchment (-), % of pastures in the catchment (-), lake altitude (+)
Chlorophyll a	65	total P concentration (+), lake mean depth (-), alkalinity (+)
PP EQR (Complete data model)	59	total P concentration (-), lake mean depth (+), alkalinity (-), lake altitude (+/-)
PP EQR (All variables model)	70	total P concentration (-), alkalinity (-), total N concentration (-), lake mean depth (+), lake altitude (+), TempAirMaxSum (+)
MP EQR	65	Secchi depth (+), alkalinity (-), mean depth (+), total P concentration (-), spring air temperature (-)

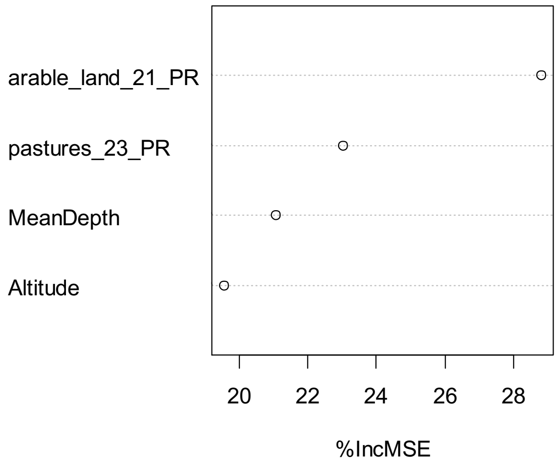

Figure 7. Predictor importance in the Random Forest model for Total P concentration. %IncMSE is the Mean decrease in accuracy (%) of each predictor. The most important predictors rank highest.

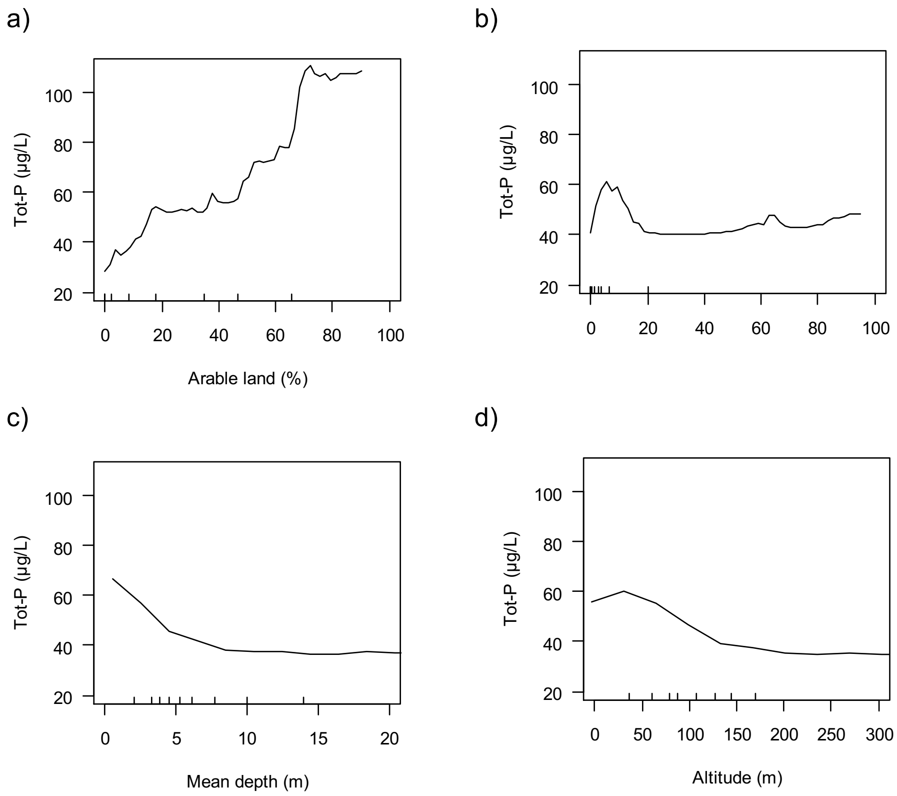

Figure 8. Partial dependence plots showing the effect of land use pressure and natural predictor variables on total phosphorus concentration (Total P) in the Random Forest model from the SoE dataset on European lakes (N = 978). The model describes how Total P on average varies along each of the gradients. The plots characterize the average unique effect of each predictor variable after the effects of all other predictor variables is accounted for. The y-axis values do not thus represent the raw data. Ticks on the x-axis indicate distribution (deciles) of the raw data.

Predictors of Secchi depth

Mean depth was the most important predictor of Secchi depth, followed by % arable land and pastures and altitude (

Figure 9). Secchi depth increased from 1.7 m to 3.2 m as lake mean depth increased from 1 m to 15 m (Figure 10). Increase in agriculture resulted in decrease in Secchi depth, whereas location of lake at higher altitudes increased Secchi depth.

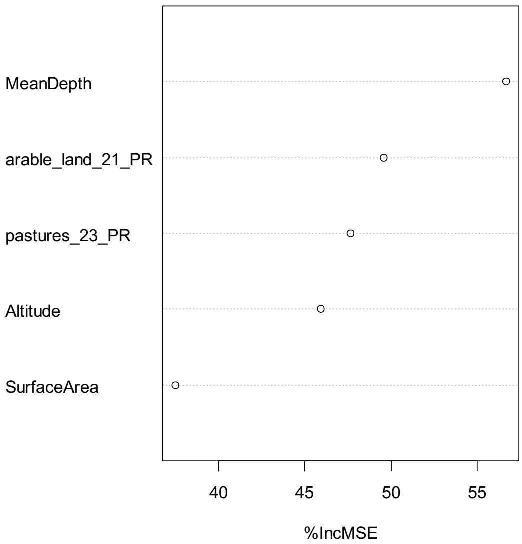

Figure 9. Predictor importance in the Random Forest model for Secchi depth. %IncMSE is the Mean decrease in accuracy (%) of each predictor. The most important predictors rank highest.

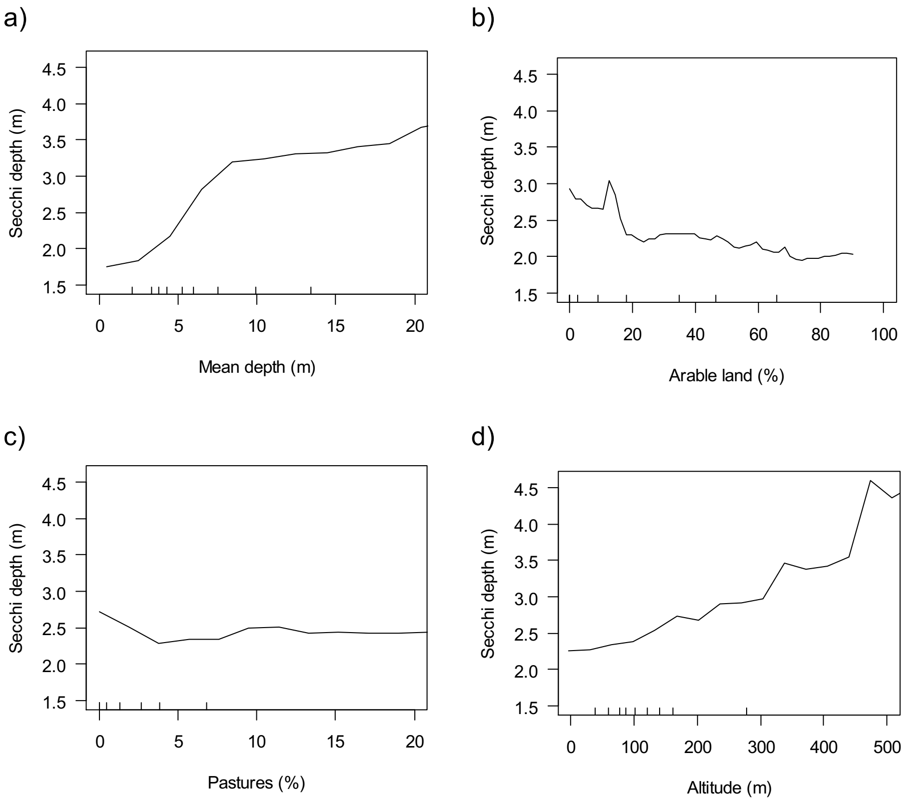

Figure 10. Partial dependence plots showing the effect of land use pressure and natural predictor variables on Secchi depth in the Random Forest model. N = 921. See Figure 8 for details of the partial plots.

Predictors of phytoplankton status

Chlorophyll a. Total phosphorus concentration was the most important predictor of chlorophyll a concentration, followed by lake mean depth and alkalinity (Figure 11). Chlorophyll a increased with increasing nutrient concentrations and alkalinity, whereas it decreased with increasing lake mean depth (Figure 12). On average, an increase in Total P to about 150 µg/L resulted in chlorophyll a increase from ~7 to ~35 µg/L. Higher Total P concentrations did not have unique effect on chlorophyll a. An initial modelling with smaller dataset indicated that chlorophyll a increased also with increase in Tot-N to up to 2 mg/L (average effect in chlorophyll a from ~10 to ~25 µg/L), but higher Tot-N concentrations did not have a unique effect on chlorophyll a.

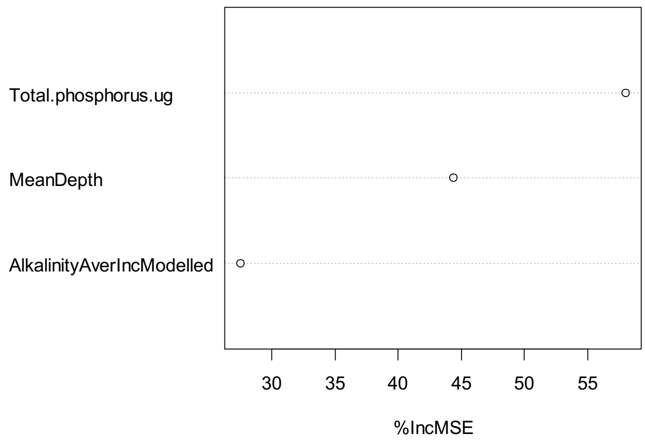

Figure 11. Predictor importance in the Random Forest model for Chlorophyll a. %IncMSE is the Mean decrease in accuracy (%) of each predictor. The most important predictors rank highest.

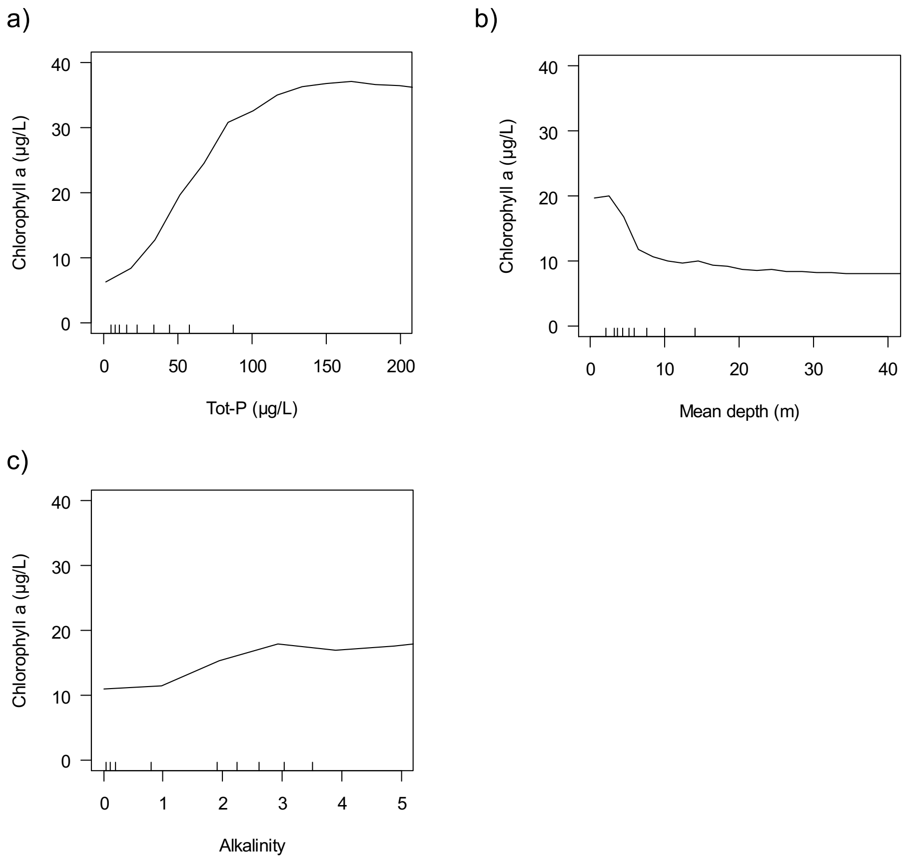

Figure 12. Partial dependence plots showing the effect of water chemistry stressors and natural predictors on chlorophyll a concentration in the Random Forest model. N = 900. See Figure 8 for details of the partial plots.

Phytoplankton EQR. Total phosphorus concentration was by far the most important predictor of PP EQR (Figure 13). Including other variables only little increased the percentage of variance accounted for (Table 2). Other variables were lake mean depth, alkalinity and maximum summer air temperature. The importance of temperature was very small, as it accounted for only about 1% increase in the variance explained in PP EQR.

PP EQR decreased with increasing nutrient concentrations and increased with lake altitude (Figure 14). When Total P increased from ~2 to ~100 µg/L, the average decrease of PP EQR was from ~ 0.85 to ~0.55. However, PP EQR did not show further unique response to Total P at concentrations higher than ~100 µg/L. An initial modelling with smaller dataset indicated that PP EQR decreased similarly also with increase in Tot-N to up to 2 mg/L, but higher Tot-N concentrations did not have a unique effect on PP EQR.

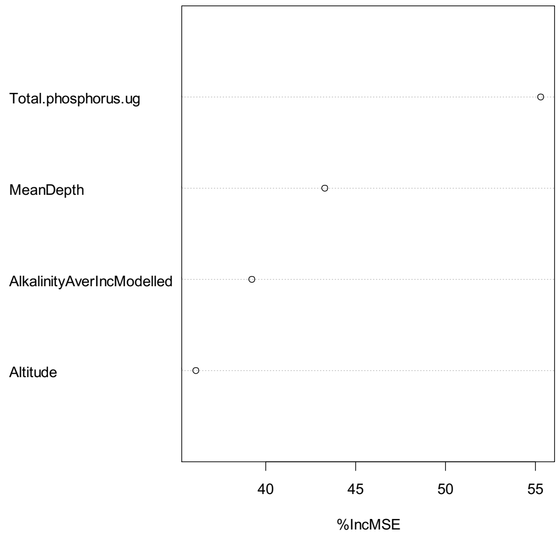

Figure 13. Predictor importance in the Random Forest model for phytoplankton EQR. %IncMSE is the Mean decrease in accuracy (%) of each predictor. The most important predictors rank highest.

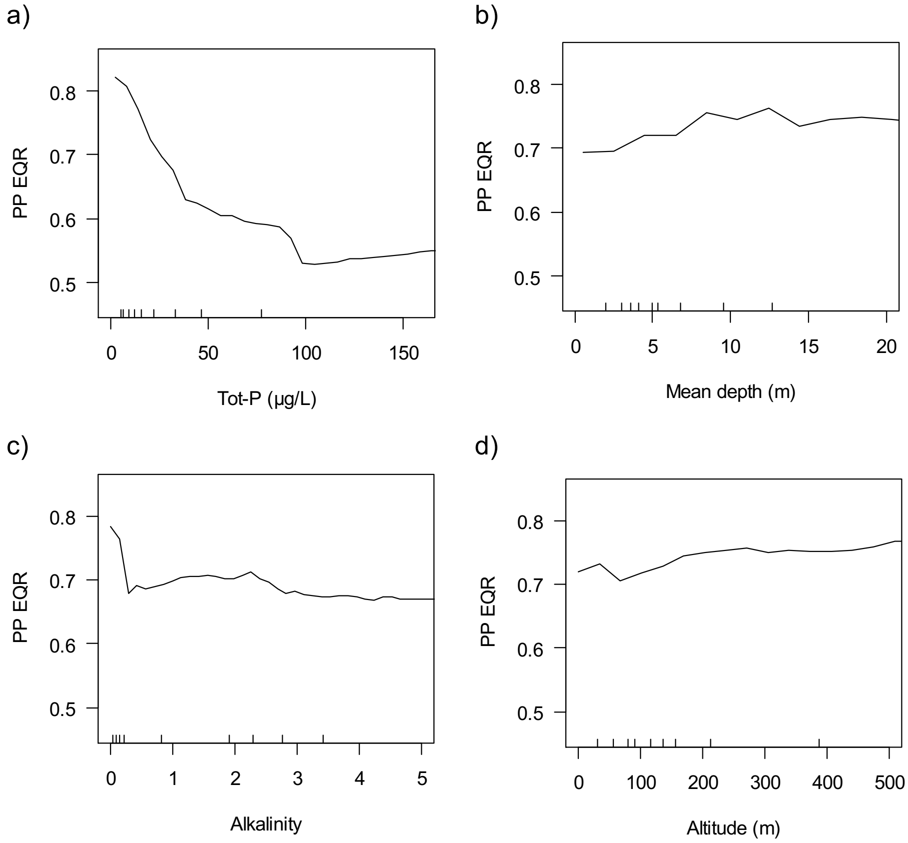

Figure 14. Partial dependence plots showing the effect of water chemistry stressors and natural predictors on phytoplankton status (EQR) in the Random Forest model. N = 714. See Figure 8 for details of the partial plots.

Predictors of macrophyte status

Macrophyte EQR. Secchi depth was by far the most important predictor of MP EQR (Figure 15). Including alkalinity and Total P increased the percentage of variance accounted for by the model. Further including lake mean depth, and air temperature (either spring or summer max or mean gave similar outcome) increased the percentage of variance accounted for (Table 2). The stepwise removal of predictors showed that other variables did not show marked unique importance in explaining PP EQR and thus did not increase the % variance explained.

MP EQR decreased with decreasing Secchi depth, increasing alkalinity and Total P EQR (Figure 16). Secchi depth had the strongest unique effect on MP EQR so that the EQR decreased from ~ 0.75 to ~0.55 as Secchi depth decreased from ~8 m to ~0.5 m. Lake mean depth also had a minor unique effect to MP EQR so that the EQR slightly increased with increasing depth. As well, air temperature had a unique effect on MP EQR so that the EQR decreased when air temperature increased from 2 to 12 °C.

Initial RF modelling with a smaller dataset indicated that the importance of nitrogen concentration was similar as that of Total P: MP EQR decreased with increasing Tot-N up to ~2 mg/L, but showed little further response to Tot-N at concentrations > 2 mg/L. As well with electrical conductivity that had missing values, MP EQR decreased as conductivity increased from 0 to 600 but did not show a unique response to conductivity values > 600.

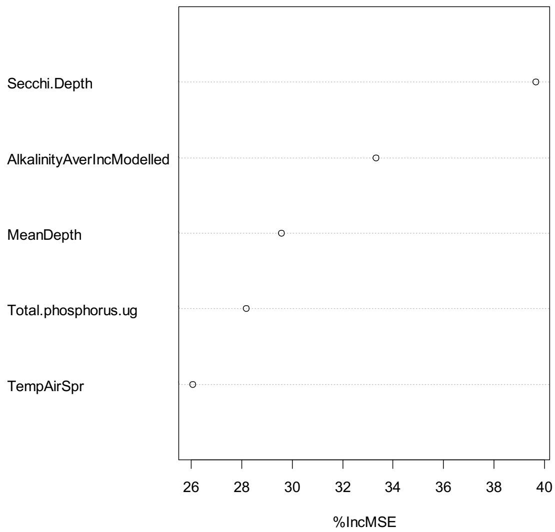

Figure 15. Predictor importance in the Random Forest model for macrophyte EQR. %IncMSE is the Mean decrease in accuracy (%) of each predictor. The most important predictors rank highest. N = 447.

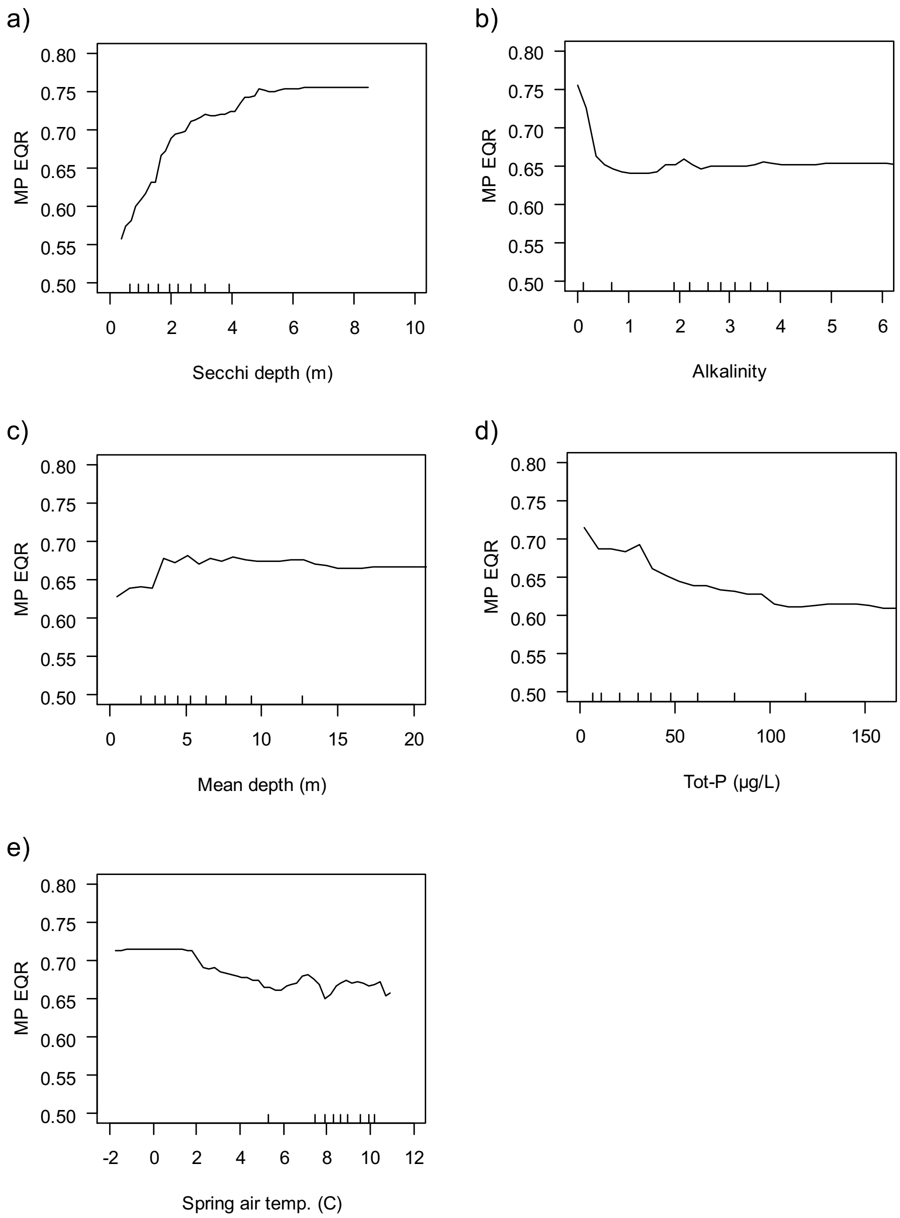

Figure 16. Partial dependence plots showing the effect of water chemistry stressors and natural predictors on macrophyte status (EQR) in the Random Forest model. N = 714. See Figure 8 for details of the partial plots.

Discussion

Agriculture as the dominant eutrophication pressure

Agricultural lands including both arable lands and pastures occupy globally about 40% of the land surface (Foley et al., 2005) and have become one of the largest terrestrial biomes on the planet, expanding mostly on the account of shrinking forests and other natural areas. Several factors deriving from intensive agriculture in the catchment areas of surface water bodies such as increased loadings of nutrients (Granlund et al., 2005), suspended solids (Edwards and Withers, 2008), dissolved organic carbon (DOC; (Wilson and Xenopoulos, 2009)), and the alkalinity export (Barnes and Raymond, 2009) affect water quality, one of the strongest pressures being the nutrient loading from fertilizer use. Intensely used agricultural areas and areas where animal slurry is used as fertilizer, are being polluted with excess phosphorus (Peñuelas et al., 2013). Phosphorus as a fertilizer continues to be overused in croplands of developed countries, causing eutrophication of downstream ecosystems (van der Velde et al., 2013). In many cases, the increases in livestock and crop production could be largely responsible for eutrophication of water bodies and initiate sudden state changes in their ecosystems (Bunting et al., 2016). Due to reduced loads from municipal and industrial point sources resulting from improved waste water treatment in the EU, agriculture has often remained the main contributor to phosphorus losses to surface waters from rural areas (European Environment Agency, 2005).

Our analysis of SoE data demonstrated a clear response of the status of lake water bodies to the proportion of arable land and pastures in the catchment both increasing the concentration of Total P (and nitrogen) in lakes. Increased phosphorus loading from agriculture enhances phytoplankton production. Our analysis showed that with the effect of other variables partialled out, an increase in Total P to about 150 µg/L resulted in a mean chlorophyll a increase of about 35 µg/L. Increase in water turbidity is one of the most visible consequences of eutrophication. Phytoplankton turbidity resulting from increased nutrient loading is obviously the main mechanism linking the agricultural land use with the water transparency (Secchi depth) although loadings of suspended solids and DOC may also have contributed to it. However, the proportion of crop and pasture lands is by far not the only factor affecting Total P and Secchi depth and thus the rather broad scatter of the relationships (see Figure 5) was anticipated. So, for instance, besides the proportions of land use, surface water quality is closely associated with the configurations of urban, agricultural, and forest areas within the watershed (Lee et al., 2009), used agricultural practices, soil types, and the natural hydrologic setting, as was evidenced in the SoE data.

The importance of natural variation for the stressors

Several studies have shown the importance of lake morphology on the water quality (Cardille et al., 2004) and that the links between lake water quality and morphometry become stronger in larger and shallower lakes. An earlier study based of the EEA database (Nõges, 2009) showed that in deeper lakes the water is more transparent and the concentrations of chlorophyll, organic matter and nutrients are generally lower than in shallower lakes. It was also shown that along the decreasing gradients of latitude, altitude and relative depth, the present phosphorus concentration and its deviation from the reference concentration increase (Nõges, 2009). Our Random Forest models with the SoE dataset indicated that when effect of agriculture as the most important anthropogenic pressure was partialled out, lake mean depth and altitude both still showed a unique negative effect on phosphorus concentration, i.e. deeper lakes and lakes at higher altitudes had lower Total P due to natural setting. The importance of natural setting was even more important for Secchi depth, which was most strongly related to lake mean depth and only secondly to agriculture.

Climate variables independently did not explain any variability in Total P in the lakes. The low sensitivity of Total P to climate indices in European lakes is concurrent with the recent results in North America (Lake Winnipeg) where climate predictors alone explained only 22.0% of changes in lake sediment fossil eutrophication indices during the 20th century, whereas similar analysis using either catchment crop or livestock variables explained threefold more variation in pigments and geochemistry (Bunting et al., 2016).

The importance of the stressors to the lake status

Agricultural land use, lake morphometry and transparency form a complex interplay with changing importance of effects on individual lake status variables. Basically agricultural land use increases nutrient concentrations, which increase phytoplankton and other primary production and thus decrease water transparency, as also evidenced in the SoE data analyses. The analyses also indicated that, independent of the effect that agricultural land has on the Total P concentration, Total P and chlorophyll a decreased, and water transparency increased with increasing mean depth. Whereas Total P was the most important stressor degrading the status (EQR) of phytoplankton, water transparency was the most important predictor for macrophyte status. When other factors were taken into account, Total P was only the fourth most important variable explaining variation of the macrophyte status. The fact that Total P is always partially measured from phytoplankton cells in the same water samples may partly explain its stronger relationship with phytoplankton than with macrophytes. Both EQRs were only to a small extent uniquely related to the mean depth, indicating that the natural setting of lake depth gradient is taken into account in national assessment techniques.

Water transparency has been proved to be a good predictor of lake water quality, especially in shallow lakes (Peeters et al., 2009). Sufficient water transparency is also a prerequisite for the presence of submerged aquatic vegetation that, in its turn, stabilizes clear-water state by preventing sediment resuspension, nutrient release and phytoplankton growth (Scheffer et al., 1993), a fact that likely also explains the gradient of macrophyte EQRs in the SoE data. In contrast, if submerged aquatic vegetation is scarce or lost entirely, the same lakes may have high concentrations of nutrients, phytoplankton blooms, high turbidity and poor water quality. Accordingly, a stable state shift theory has been developed to describe the shift between macrophyte-dominated clear water states and phytoplankton-dominated turbid water states in shallow lakes (Scheffer et al., 1993).

Lake depth, that in the SoE data was the most powerful factor determining water transparency affects sediment resuspension, internal loading of nutrients, and underwater light conditions. The larger and shallower is the lake the stronger is the wind action that causes bottom sediment resuspension and decreases water transparency (Nõges, 2009). In deep lakes, eutrophication caused by higher nutrient loading from the catchment is assumed to be the most important factor increasing phytoplankton biomass and lowering water transparency which may favour the development of potentially toxic cyanobacteria (Romarheim et al., 2015).

The low importance of climate

The WFD Intercalibration process aimed at Europe-wide consistency and comparability of the classification results of the monitoring systems operated by each EU Member State for the biological quality elements, within broad regions and within broad common water body types. The criteria to define regions and types included climatic, geographic, geological, morphometric and other natural differences. The process envisaged agreeing on reference conditions and rules for deriving high-good and good-moderate class boundaries for biological quality element consistent with the normative definitions given in the WFD. As the main outcome of the process, the virtually unimpacted sites should fall into "high" status class whereas slight and moderate deviation from the reference status should result, correspondingly, in "good" and "moderate" status of the water body in whatever climatic region in EU. In this way the potential effect of climatic differences on water body status across Europe, as measured by EQRs, should be eliminated and the climatic gradients cannot and should not be used in a space-for-time-substitution approach to study the effect of e.g. temperature changes because in each region the local biota is best adapted to the local climate. In that sense, the large scale SoE data, although including data on water temperature, is not suitable for studying climate warming as a stressor unless the climate data are compared with the local long-term averages and converted into climatic anomalies which then could be related to the biological indicators. Indeed, the best adaptation of biota to local climate explains why temperature variables accounted at most for only few percentages of the variance in the PP or MP EQRs.

Still it has been recurrently shown that several metabolic rates in aquatic ecosystems have a strong temperature dependence and thus the rate of processes such as nutrient or food uptake, growth rate, production, respiration, reproduction, decomposition of organic matter etc. may have large regional differences implying that also the sensitivity of ecosystems to stress may differ between climate zones. The study by (Staehr and Sand-Jensen, 2006) although finding distinct responses to relatively small temperature increases, suggested that the interaction between nutrient availability and ambient temperature was responsible for most of the observed variability in phytoplankton growth, photosynthesis and respiration.

Splitting the SoE lakes into cool and warm relative to their summer air temperatures below or above 17.5 °C revealed higher Chl a vs. Total P slopes in warmer lakes although the ranges of both variables were almost the same. However, the different slopes cannot be attributed entirely to temperature differences because the cool and warm lakes differed significantly also by water colour and mean depth. Humic matter giving the brownish colour to surface waters can be a strong confounding factor in temperature effect studies as it acts as a sink of P, due to its tendency to chelate Fe and P and adsorb organic P forms (Christophoridis and Fytianos, 2006). Also the higher mean depth of the warm group of lakes may have certain impacts Total P conversion efficiency to algal biomass. An earlier study examining the Total P – Chl a relationship using data from over 1,000 European lakes (Phillips et al., 2008) found no significant effect of geographic region or humic content on this relationship. However, contrary to our results, the authors found the lowest yield of chlorophyll per unit of nutrient in deep lakes and the highest yield in low and moderate alkalinity shallow lakes.

Comparability of the EQRs in the SoE data

If in the Total P – Chl *a* relationship both variables depend on direct lab analyses data, all attempts to relate environmental stress factors with the normalised EQR values of either phytoplankton or macrophytes include subjective uncertainties related to the Member State specific definition of reference conditions and class boundaries. Despite indisputable success of the WFD Intercalibration process in harmonising the classification methods among Member States, our analysis revealed large country-specific differences. Both PP and MP EQRs showed a clear general decreasing trends with increasing Total P levels and decreasing Secchi depth, indicating that nutrient stress has been the main degradation gradient when member states have set the criteria for PP and MP. Although the best models based on all data described 65-70% of the ecological status differences, the scatter was still extremely broad so that some of the country specific clouds of the PP EQR – Total P relationship had even no overlap.

The differences in Member State water quality scales became especially obvious with dividing the lakes into cool and warm groups. At a given Total P level, the Nordic countries placed their lakes generally into a lower quality class compared to e.g. Poland, Lithuania and Romania (see Figure 6), in principle due to differences in geology. As Romania has not intercalibrated their classification system, a different scale of their data is understandable. Still the Nordic countries with lakes falling in the cooler group seem to have more stringent water quality criteria compared to the regions of the warmer lakes.

Finally, after removing some obvious outliers that were likely errors (see Appendix 2), the SoE data proved generally as a useful and reliable data source for analysing multiple stressor impacts on state variables of lakes (e.g. Chl a, Secchi depth) in Europe. Apparently country-specific differences in reference conditions and/or class boundary setting compromise to a certain extent the usefulness of the EQR data for the purpose of European-scale analysis in MARS. As local climate and its variability are considered in reference conditions, climatic gradients in Europe cannot be used for testing climate change impacts on the status of lakes. For these purposes either long-term time series from single locations, climatic anomalies in extreme years or the coherence studies of changes over extended areas would be the appropriate approaches.

Key messages

- Using SoE lake data complemented with catchment land use and climate variables, the best Random Forest models explained 59-71% of the variability in lake state variables such as Total P, Chl *a*, Secchi depth, and the ecological quality ratios (EQR) of phytoplankton and macrophytes.
- With increasing proportion of arable and pasture lands in lake catchments across Europe, total phosphorus concentration in lakes clearly increased and Secchi depth generally decreased.
- PP EQR decreased clearly with increasing Total P whereas MP EQR showed a stronger relationship to Secchi depth increasing with water transparency.
- Among natural parameters, lake altitude and mean depth had the strongest unique effect on both pressures and status assessments. Excluding the effect of the % agricultural land

use, Total P tended to be higher in lowland and shallow lakes. Increasing mean depth was even the strongest factor causing an increase in water transparency although transparency increased also with higher location and less agricultural lands within the catchment.

- As local biota can be considered best adapted to local climate, the climatic gradients across Europe should not affect the results of ecological status assessment. As our analysis showed, temperature had virtually no effect on the EQRs of phytoplankton and macrophytes. Still, as several metabolic rates in aquatic ecosystems have a strong temperature dependence, implying that also the sensitivity of ecosystems to stress may differ between climate zones. Indeed, our analysis revealed higher Chl a vs. Total P slopes in warmer lakes (summer mean temperature > 17.5°C) although it could not be attributed entirely to temperature differences because the cool and warm lakes differed significantly also by water colour and mean depth.
- Despite indisputable success of the WFD Intercalibration process in harmonising the classification methods among Member States, our analysis revealed country-specific differences in relationships between stressor levels and ecological status assessments that could likely be attributed to differences in established reference conditions and class boundaries.

2.2. Effects on abundance of cyanobacteria (large-scale data)

Contributors: Jessica Richardson, Laurence Carvalho, Stephen Maberly, Jannicke Moe, Phil Taylor

Summary

Cyanobacteria blooms are becoming an ever increasing threat to global water security. What role climate change has in this threat, alone and in combination with other freshwater stressors, needs to be better understood. There is particular concern that cyanobacteria blooms will increase with higher temperatures and that this response will only be exacerbated by nutrient enrichment. It is also expected that changes in rainfall patterns could have a role in this relationship, changing the delivery of nutrients and the duration of periods where conditions within the water column are favourable for cyanobacteria dominance. Using data from 779 natural and 96 artificial European lakes, we aimed to detect and quantify the response of cyanobacteria to the combined effects of these stressors: nutrient enrichment in the form of total phosphorus and climate change in the form of increased air temperature and altered summer rainfall. Based on evidence that cyanobacteria biovolume varies with lake characteristics such as depth, colour and alkalinity and that responses to stressors acting alone can be modulated by these types, we expected the response of cyanobacteria to multiple stressors to vary among different lake types.

We grouped lakes into risk types based on the distribution of cyanobacteria biovolume across levels of lake characteristics: shallow (≤5 m) vs deep, high alkalinity vs low alkalinity and low humic vs medium/high humic. In total we defined eight risk categories: high, medium type 1, medium type 2 and low in shallow and deep lakes. For each risk category cyanobacteria biovolume was higher in shallow lakes than deep lakes in the sample, and so overall had a higher incidence of exceeding WHO low risk health thresholds. The response of cyanobacteria to the three stressors was then modelled within each type. Where the size effect of stressors could be compared (in additive models), we found that in both shallow and deep lakes, the effect of temperature was weaker than the effect of TP. The effect of TP in deep lakes was larger in high-risk types and in more nutrient-limited lakes. We found that it was not possible to generalise the response of cyanobacteria to stressors acting alone or in combination across all lakes, because the significance of interactions depended on the length of the nutrient gradient and on the lake type. In general, we found that the interactions detected depend on how the gradient was defined, e.g. when comparing oligotrophic to oligo-mesotrophic, and that the effect size of the interaction varied between lake types. The variance explained by the effects of stressors was often low compared to the variance explained by the natural characteristics of the lakes. More specifically, we found that there was a higher prevalence of interactions (50%) in shallow lakes compared to deep lakes (41%). However, the proportion of which were synergisms was higher in deep lakes (60%) than in shallow lakes (36%).

As hypothesised, synergisms were found between temperature and nutrients, however this interaction was not consistently positive nor prevalent between risk types. These result suggest that the response of cyanobacteria to multiple stressors will depend on the gradient of the stressor and environmental context, but also that the overall effect will depend on the natural variability between lakes. This analysis highlights some of the issues with large datasets that are not compiled for question-specific research; the availability of biological and environmental data limited the analysis to temperate climatic gradients whilst the resolution of the stressor data (in terms of how informative it is as an explanatory variable) incorporated uncertainty.

For future work in assessing the response of cyanobacteria to multiple stressors, data which cover longer stressor gradients, more ecologically relevant measures of stressors and over a better balance of lake types and other confounding variable are needed. Despite these uncertainties, this work highlights that when managing multiple stressors, the characteristics of the waterbody, the gradient of the stressor and also system-specific variation need to be considered.

Introduction

Cyanobacteria blooms are becoming an ever increasing threat to global water security. What role climate change has in this threat, alone and in combination with other freshwater stressors, needs to be better understood.

As phototrophic organisms cyanobacteria require nutrients, light and appropriate temperature for growth. Whilst these are prerequisites, dominance is driven by a balance of the physical and chemical environment which suits the adaptive traits of many cyanobacteria taxa e.g. higher temperature growth optima, the ability to fix nitrogen and also the ability to regulate buoyancy (Carey et al., 2012). Through anthropogenic activities we are not only enhancing but also combining some of the optimal conditions for cyanobacteria dominance. Firstly, despite remediation efforts, nutrient enrichment is hardly abating as the demand for intensive agriculture continues, whilst internal cycling of nutrients within lakes retains a legacy of past activities (Nürnberg, 2009). Secondly, and at the forefront of this discussion, is the issue of climate change; over the past three decades water temperatures, on average, have increased by 0.34 °C a decade (O'Reilly et al., 2015) with an increase in the onset (McCormick, 1990; Winder and Schindler, 2004), and duration (Wagner and Adrian, 2009) of thermal stratification. Furthermore it is predicted that there will be changes in the timing and intensity of rainfall events (Milly et al., 2005) which could drastically alter the timing and intensity of flow and nutrients entering lakes.

It is predicted that cyanobacteria blooms will increase with higher temperatures and that this response will only be exacerbated by nutrient loading (Paerl and Huisman, 2008). It is also expected that an increases in extreme rainfall events could further facilitate this synergistic relationship as extreme rainfall events followed by periods of droughts will provide both an increase in nutrients and the required physical conditions for dominance. Here it is hypothesised that these predicted interactions between stressors may only be relevant in certain lake types, under certain conditions. Studies have shown that environmental context is important in determining the response or the strength of the response to drivers of biomass, for example nutrients are more important in shallow, polymictic lakes and nutrient limited lakes whilst temperature is more important in dimictic lakes (Rigosi et al., 2014; Taranu et al., 2012). Other studies have shown that even within similar lake types the response to stressors can be very different because the effect of local events (Huber et al., 2012), highlighting that interactions between stressors could not only be lake type specific but also potentially lake specific.

Recently, many large freshwater datasets have become available and are being used to explore ecological questions, expanding our knowledge from single systems to local, regional and continental scales. Other studies have already used some of these data to explore the role of nutrient stress and climate in explaining variance in cyanobacteria and phytoplankton biomass, both indices of water quality (Carvalho et al., 2013; Phillips et al., 2008). To our knowledge only one study has used a large dataset to specifically explore the response to nutrients and temperature in combination (Rigosi et al., 2014). The aim of this work is to extend from these studies, to identify multiple stressor interactions between identified drivers of cyanobacteria – nutrients, flow/retention time and temperature - and to determine if and how they change depending on the characteristics of the lake. The focus of this analysis is to contribute relevant evidence suitable for the management of our freshwaters in the face of global change.

Data

Data on cyanobacteria biovolume (mm^3 L^{-1}), nutrients (total phosphorus, µg L^{-1}) and lake type variables (depth, alkalinity, colour, altitude and surface area) were extracted from the WISER database (Moe et al., 2013) and summarised as monthly means for July, August and September. These months were chosen on the basis that this is when cyanobacteria blooms are most reported in temperate, northern latitudes and when biological sampling is most intense thereby retaining the highest number of lakes possible. We chose only to select data after the year 2000 as sampling methods from this period are most standardised. As typological variables were an important consideration in the analysis, only lakes which included all variables (altitude type, surface area type, depth type, humic type and alkalinity type) were included. In total 779 natural lakes and 96 artificial lakes matched these criteria, the locations of these lakes are shown in Figure 17.

Climatic data was obtained from the Joint Research Council (JRC) at a resolution of 25 km^2 grid squares. The JRC grid which mapped to the grid-reference of the lake's sampling point was used as the match. Data was summarised as monthly values: mean monthly temperatures (°C, mean of the daily mean) and total summer precipitation (mm, months: June, July and August). Air temperature was used as a proxy for water temperature and rainfall as a measure of relative variations in the flushing rate of the lake.

Figure 17. Locations of lakes included in the analysis. Blue points are the locations of natural lakes, black points the location of artificial lakes (heavily modified waterbodies).

Methods

Data was explored using pairwise plots, conditional plotting and boosted regression tree analysis to identify covariation between the response, stressors and lake type variables, both as pairwise covariation and multiple, conditional covariation. The preliminary analysis highlighted that cyanobacteria varied with stressors (Figure 22) but also lake characteristics (

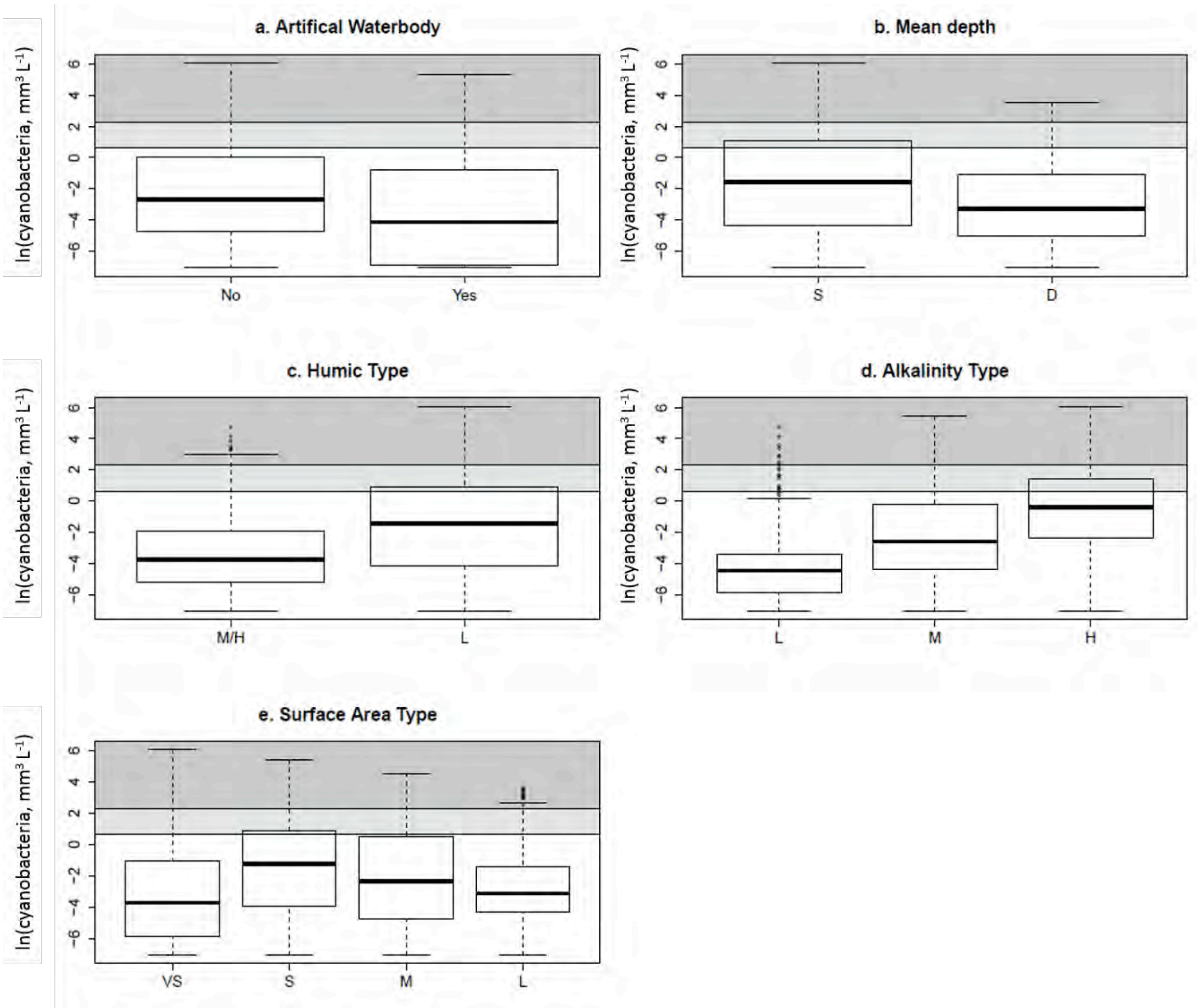

Figure 18) and that the response to stressors changed with these lake attributes. To account for these interactions, levels of each influential lake characteristic were combined and defined as 'risk types' i.e. for each lake characteristic the level with the highest average cyanobacteria biovolume were combined, and so on (Table 3). The lake characteristics which defined these types were: depth (shallow, ≤5 m and deep, >5 m), alkalinity (high and low/medium) and humic type (low humic and medium/high humic). Altitude co-varied with TP, depth and temperature and so its influence was retained through these variables. In total eight risk types were defined - high, medium type one, medium type two, and low for shallow and deep lakes. The term 'risk' is used to reflect that the higher risk categories have a higher percentage of lake months which exceed the WHO low risk threshold (equivalent biovolumes as defined in (Carvalho et al., 2013) as shown in Figure 21b. As such, these are the lakes of most concern when considering future scenarios of multiple stressor combinations as synergisms could increase the incidence of exceeding these health thresholds. Table 3 gives a description of each risk type and the number of lakes in each type and Figure 21 a shows the spatial distribution of these lakes.

Table 3. Description of types and the number of natural lakes within each type and depth category.

Shallow (≤ 5 m)		N
	High risk	218
	Low humic, high alkalinity	
	Medium risk type 1	71
	Medium-high humic, high alkalinity	
	Medium risk type 2	37
	Low humic, low-medium alkalinity	
	Low risk	83
	Medium,-high humic, low—medium alkalinity.	
Deep (>5 m)		
	High risk	162
	Low humic, high alkalinity	
	Medium risk type 1	20
	Medium-high humic, high alkalinity	
	Medium risk type 2	107
	Low humic, low-medium alkalinity	
	Low risk	81
	Medium-high humic, low-medium alkalinity	
Total		779

The response of cyanobacteria biovolume to multiple stressors was then modelled in these types along the nutrient gradient. Experimental studies have shown that interactions can change along the stressor gradient when the response to single stressors are non-linear (Piggott et al., 2015). There is evidence, from the literature (Carvalho et al., 2013) and from the exploratory analysis (Figure 23) that the response of cyanobacteria to TP in these lakes is non-linear, with the highest response to TP seen in lakes which are nutrient limited. To account for potential changes in interactions along the gradient the response to TP was modelled within TP intervals which were chosen *a priori* based on nutrient derived eutrophication levels (Caspers, 1984) – oligotrophic as < 12 μg L^{-1} , mesotrophic as 12-24 μg L^{-1}, eutrophic as 24-96 μg L^{-1} and hypertrophic as > 96 μg L^{-1}. The response was also modelled over the full TP gradient (global) for comparison.

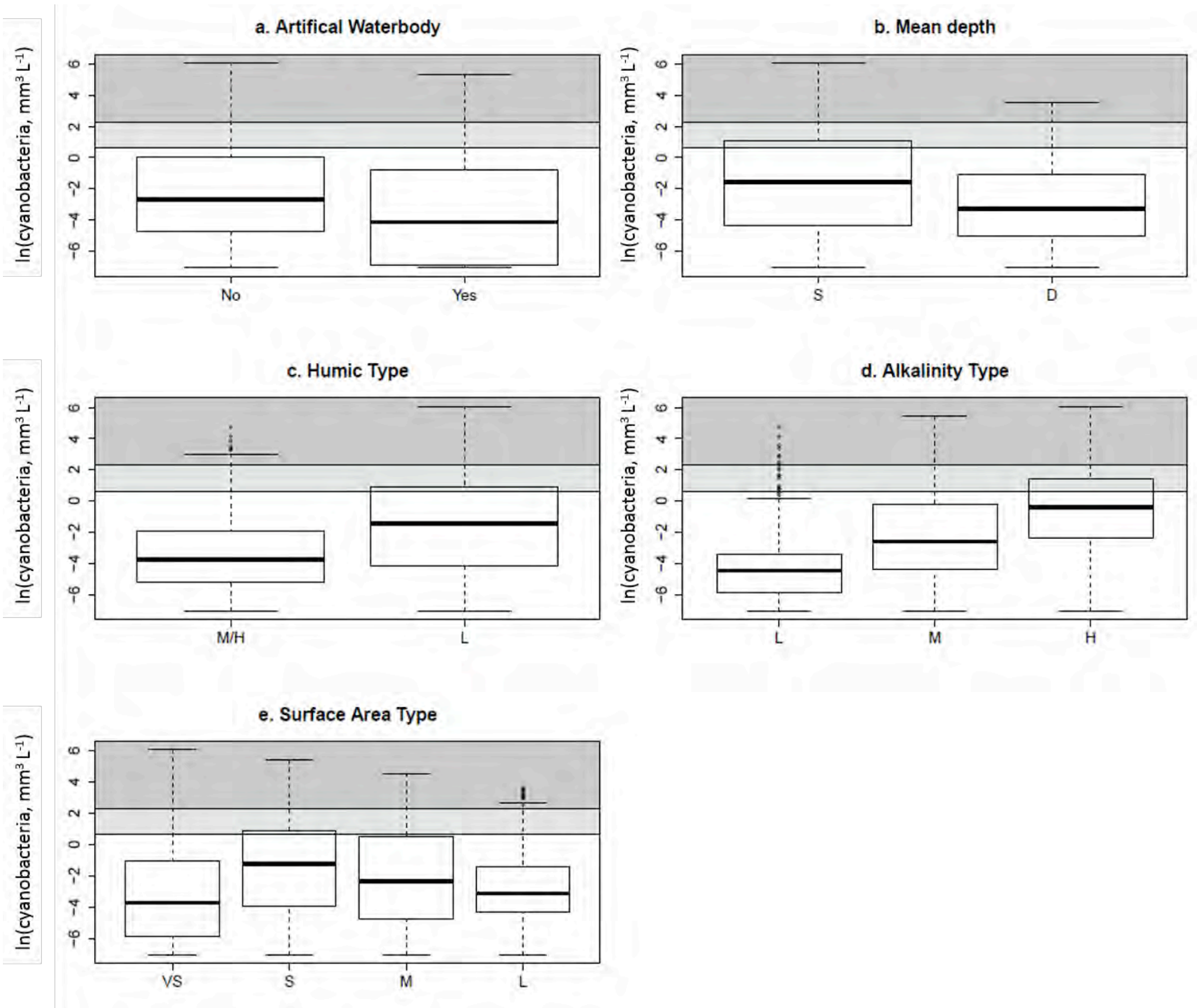

Figure 18. Box plots showing the distribution of ln(cyanobacteria biovolume, mm^3 L^{-1}) given different lake characteristics. Boxplots display the values of the 25th, 50th (black line) and 75th percentiles. Whiskers extend to the most extreme data point less than 1.5 times the interquartile range. Shaded areas represented WHO guideline thresholds; light grey is exceedance of low threshold (>ln 0.64, >1.9 mm^3 L^{-1}), dark grey is exceedance of medium threshold (>ln 2.3, >9.9 mm^3 L^{-1}). a. waterbody type, No = natural lakes, Yes = artificial waterbody; b. average mean depth: D = deep (>5 m), S = shallow (≤ 5 m); c. humic type: M/H = medium to high (> …), L = Low (< …); d. alkalinity type: L= Low( ), M = medium ( ), H= High ( ); e. surface area type: VS = very small (), S = small (), M = medium (), L = large()

The response was also modelled over the full TP gradient (global) for comparison.

Artificial vs natural lakes

Exploratory analysis highlighted differences in the response of cyanobacteria between natural and artificial waterbodies (heavily modified) which could confound the response. On this basis, as well as because of functional differences between these systems, the analysis was split by natural and artificial waterbodies. The same risk types were assigned to these lakes based on depth, alkalinity and humic type, however a bias in the data meant only low humic, high alkalinity lakes were retained (high risk types). The exploratory analysis also showed that depth was confounding the relationship between cyanobacteria and stressors and so as with natural lakes the analysis was split by mean depth type.

Statistical analysis

All model were fitted using linear mixed models, using the lme4 package (Bates et al., 2015) for R statistical software (R Core Team, 2016). To meet the assumptions of normality and homoscedasticity, cyanobacteria biovolume and TP were log (natural) transformed. All stressor variables were then standardised (mean centred with a standard deviation of one) so that the size effect of stressors could be compared between models and between stressors (when no interaction terms were present). For each 'risk type' the following model was fitted:

$$Y = \beta_0 + \beta_1 X_{TP} + \beta_2 X_{temp} + \beta_3 X_{rain} + \beta_4 X_{TP\,x\,temp} + \beta_5 X_{TP\,x\,rain} + \beta_6 X_{temp\,x\,rain} + \beta_7 X_{TP\,x\,temp\,x\,rain} + \delta_{SurfaceAreaType} + \gamma_{lakeID} + \varepsilon,$$
$$\gamma \sim (0, {\sigma_l}^2), \varepsilon \sim (0, {\sigma_r}^2)$$

where Y is the log cyanobacteria biovolume response of interest, β_0 is the intercept term, β_1, β_2, and β_3 are model parameters for the TP term, temperature term and summer rainfall term, respectively. β_4, β_5, β_6 and β_7 are model parameters for the interaction between TP and temperature, TP and summer rainfall, temperature and summer rainfall and TP, temperature and summer rainfall, respectively and δ is the model parameter for the surface area type term, for which there was four levels: very small, small, medium and large – the reference level in all models was 'large'. γ is the random effect term for lake ID which allows the response to vary on the intercept for individual lakes and ε is the overall error term, both with a mean of zero and unknown variance. Initially year and month were also incorporated into the model as random terms to account for sampling within lakes over multiple months and years but this did not explain additional variance so were removed from the final models for parsimony. This model was then simplified by removing higher order interaction terms in turn, comparing simplified and more complex models using AIC and BIC, favouring simpler models when retaining more complex terms did not improve the model. As degrees of freedom and therefore p values can only be approximated for mixed effect models significance here at the 5% level has been interpreted as a t value of +/- 2. The variance explained by the model is reported as marginal R^2 which describes the proportion of variance explained by the fixed factor(s) alone and conditional R^2 which describes the proportion of variance explained by both the fixed and random factors (Nakagawa and Schielzeth, 2013).

Results

Artificial lakes

Stressor gradients.

Most deep artificial lakes (n= 62) were reservoirs located in Spain, contributing 79% of lake month data; the remainder were lakes from the Netherlands (13%), Belgium (6%), UK (2%) and Denmark (<1%). Whilst shallow artificial lakes (n=35) were mainly distributed in central latitudes with 78% of lake month data contributed from the Netherlands, 11% from Belgium, 9% from Denmark, and 2% from Hungary. This spatial distribution of lakes resulted in differences in climatic stressors between deep and shallow artificial waterbodies, with higher air temperatures and lower summer rainfall seen for deep lakes which were predominantly located in a Mediterranean climate (Table 4). There were also differences in the range of TP, reflecting the spatial distribution of TP and land use within Europe (Figure 22, Table 7), however both deep lakes and shallow lakes spanned all trophic states (oligotrophic – hypertrophic) although there was a lower TP maxima in deep artificial waterbodies (510 μg L^{-1}) compared to shallow artificial waterbodies (1590 μg L^{-1}).

Table 4. Response (cyanobacteria biovolume mm^3L^{-1}) mean, min, max and stressor gradients in artificial lakes.

	Cyanobacteria biovolume mm^3L^{-1}		TP μg L^{-1}		Mean monthly temp (°C)		Total summer rainfall (mm)	
Risk	Mean	Min-Max	Min	Max	Min	Max	Min	Max
Shallow	7.5	0-203	39	1590	14	23	15	138
Deep	0.7	0-54	3	510	12	27	0	109

Response

In deep reservoirs (mean depth >5 m) cyanobacteria biovolume ranged between 0 – 3.9 mm^3 L^{-1} whilst chlorophyll *a* biomass ranged between 0.1 – 110 μgL^{-1}. Across the full TP gradient, there was a significant positive effect of rainfall (estimate = 0.4, se = 0.2, t value = 2.2, Table 5 and Table 10), however this effect explained a negligible amount of variance (R^2 = 0.04). Modelling the data by trophic state improved the variance explained, identifying significant interactions in eutrophic lakes and additional main effects of stressors in hypertrophic lakes (there was insufficient data to model oligotrophic and mesotrophic lakes). In hypertrophic lakes there was a significant positive effect of rainfall (estimate = 1.3, se = 0.3, t value = 4.6) but also a negative effect of TP (estimate = -3.6, se=1.2. t value =-2.9); the effect of TP was greater than the effect of summer rainfall. The variance explained by the fixed effects of TP and rainfall (R^2 = 0.43) was considerably more than explained by the effect of rainfall alone in the global model. Furthermore, the size effect of rainfall was greater when modelling the response within hypertrophic lakes. In eutrophic lakes a positive interaction was identified between TP, temperature and rainfall (R2 = 0.25), this three-way interaction can be visualised in Figure 19. The plots show that at temperatures below 16.4 °C (Figure 19 a.) there is a negative interaction between summer rainfall and TP, whilst at temperatures above 16.4°C the interaction changes to be positive (Figure 19 b-c). The length of the temperature gradient and rainfall gradient between the global model, eutrophic model and hypertrophic model are very similar which suggests the differences between models are because of differences in the response along the nutrient gradient. Alternatively, interactions may have been present but non-detectable in hypertrophic lakes because of reduced power (N=40 in hypertrophic lakes compared to N=67 in eutrophic lakes). In all deep artificial lakes, surface area type is not influential in explaining cyanobacteria biovolume.

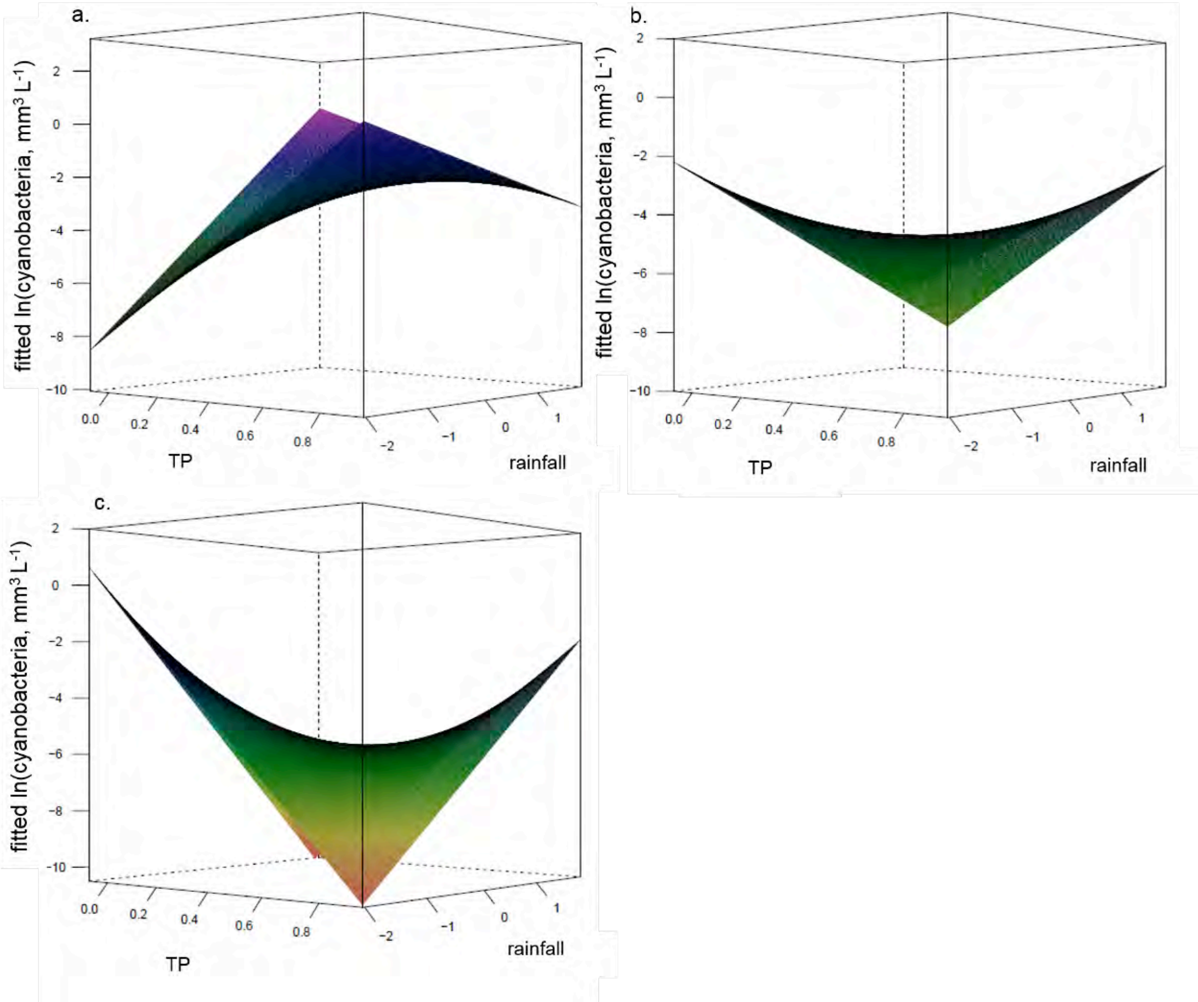

Figure 19. Positive interaction between TP, summer rainfall and temperature in deep, eutrophic artificial lakes. The fitted response of ln(cyanobacteria, mm^{3}L^{-1}) to the interaction between standardised ln(TP, µgL^{-1}), and standardised total summer rainfall (mm) is fitted for three intervals of standardised mean monthly temperature (°C), Temp.st, corresponding to a. ≤16.4°C, b. 16.5 - 23.5°C, and c. 23.5 – 26.3°C. At temperatures below 16.4°C there is a negative interaction between summer rainfall and TP, whilst at temperatures above 16.4°C the interaction changes to be positive.

Contrasting to deep lakes, the response of cyanobacteria and chlorophyll *a* were very similar in shallow (≤ 5 m) artificial lakes (maxima of 203 and 204 µg L^{-1}, respectively), but, similar to deep lakes, the response to stressor combinations depends on the trophic gradient analysed, with more interactions seen in the global response. In the global model there was a significant negative interactions between TP and temperature and TP and rainfall, however only a negative interaction between TP and rainfall was seen in eutrophic lakes and a negative interaction between TP and temperature in hypertrophic lakes (Table 5, Figure 20 a and b, respectively). This could be a result of trophic state specific responses or because of lower power in each state (N= 35 for eutrophic and N=47 for hypertrophic) compared to the global dataset (N=82). Again, similar to deep artificial lakes, modelling the response by trophic state improved the variance explained, especially in eutrophic lakes where the marginal R^2 was 0.75 compared to an R^2 of 0.5 for the global model, despite the global model containing more parameters.

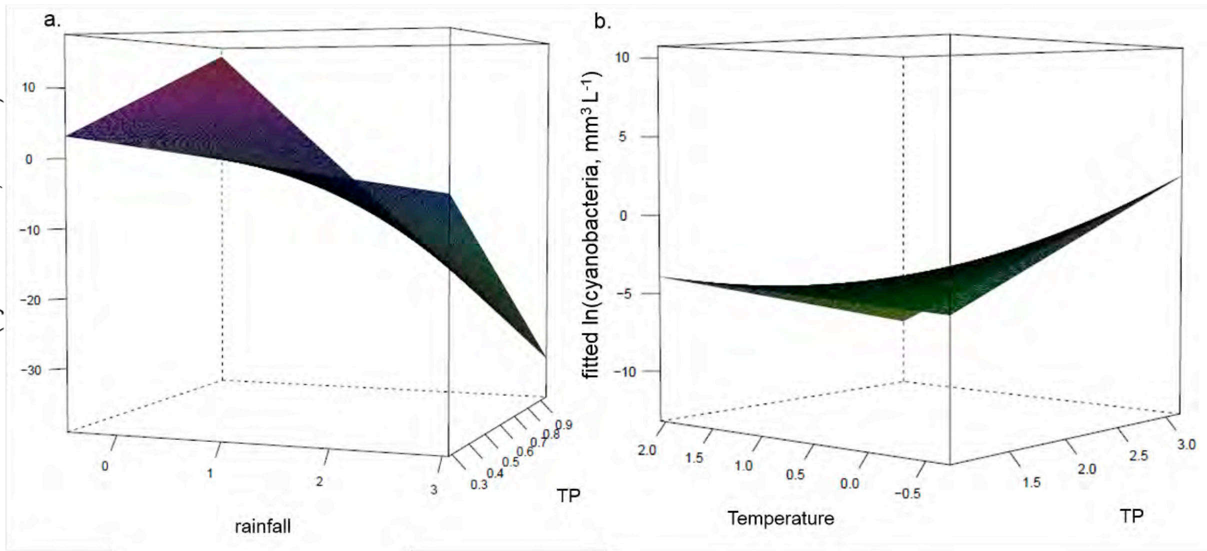

Figure 20. Interaction between stressors in shallow, artificial lakes. The plots show the fitted response of ln(cyanobacteria biovolume, mm^3 L^{-1}) to pairwise stressor combinations in a. eutrophic (negative TP x rainfall interaction) and b. hypetrophic lakes (negative TP x temperture interaction). All stressors are standardised (scaled to have a mean of zero and a standard deviation of one).

Table 5. Overview of interactions and main effects of stressors in artificial lakes of different depths. Dots represent terms contained in the model and the sign indicates whether the coefficients are positive or negative. Grey colours are for coefficients with a t value of <+/- 2, interpreted as non-significant. Green are for significant negative single terms or interaction terms and red for significant positive single terms or interaction terms. Significance at the 5% level is interpreted as being when the t value is ≥± 2. N = lake months. R^2_M = marginal R^2, R^2_C = conditional R^2

	N	R^2_M	R^2_C	T - values									
				TP	°C	rain	VS	S	M	TP * °C	TP* rain	°C *rain	TP * °C *rain
Deep	146	0.04	0.57			🔴+							
Eutrophic	67	0.25	0.78	⚫+	⚫-	🔴+				⚫-	⚫-	🟢-	🔴+
Hypertrophic	40	0.43	0.56	🟢-		🔴+							
Shallow	82	0.50	0.84	⚫+	🔴+	⚫+	🟢-	⚫-	⚫+	🟢-	🟢-	⚫-	⚫+
Eutrophic	35	0.75	0.79	⚫-	⚫+	🔴+	🟢-	🟢-	🔴+		🟢-		
Hypertrophic	47	0.51	0.78	⚫+	⚫+	🟢-	🟢-	⚫-	⚫-	🟢-			

Natural lakes

Stressor gradients

Natural lakes from this dataset are predominantly located in central and northern latitudes (Figure 21). In both deep and shallow categories, the lowest 'risk type' lakes are located at more northerly latitudes and higher 'risk type' lakes within central latitudes. This spatial distribution of types co-varies with the spatial distribution of TP but also geology, alkalinity and land use (Table 6, Table 7 and Figure 22). High TP lakes are located in calcareous, arable catchments at central latitudes and low TP lakes are located in siliceous, forested catchments at higher latitudes. This results in different TP gradient lengths between types, with higher 'risk types' having longer gradients of TP as each type is partly defined by a level of alkalinity type. This covariation between TP and type could confound the response so care should be taken when comparing global models, however between trophic states comparisons are appropriate where this co-variation between type and TP has been removed. Across types mean monthly temperature reached a minimum of 5 °C and a maximum of 24 °C with a variation of 4 °C between type maximum mean temperatures. The highest air temperatures were seen in shallow and deep high 'risk type' lakes. Total summer rainfall also varied by type, although as most of the lakes are located in the temperate climatic zone, there was no evidence of clear extremes.

a.

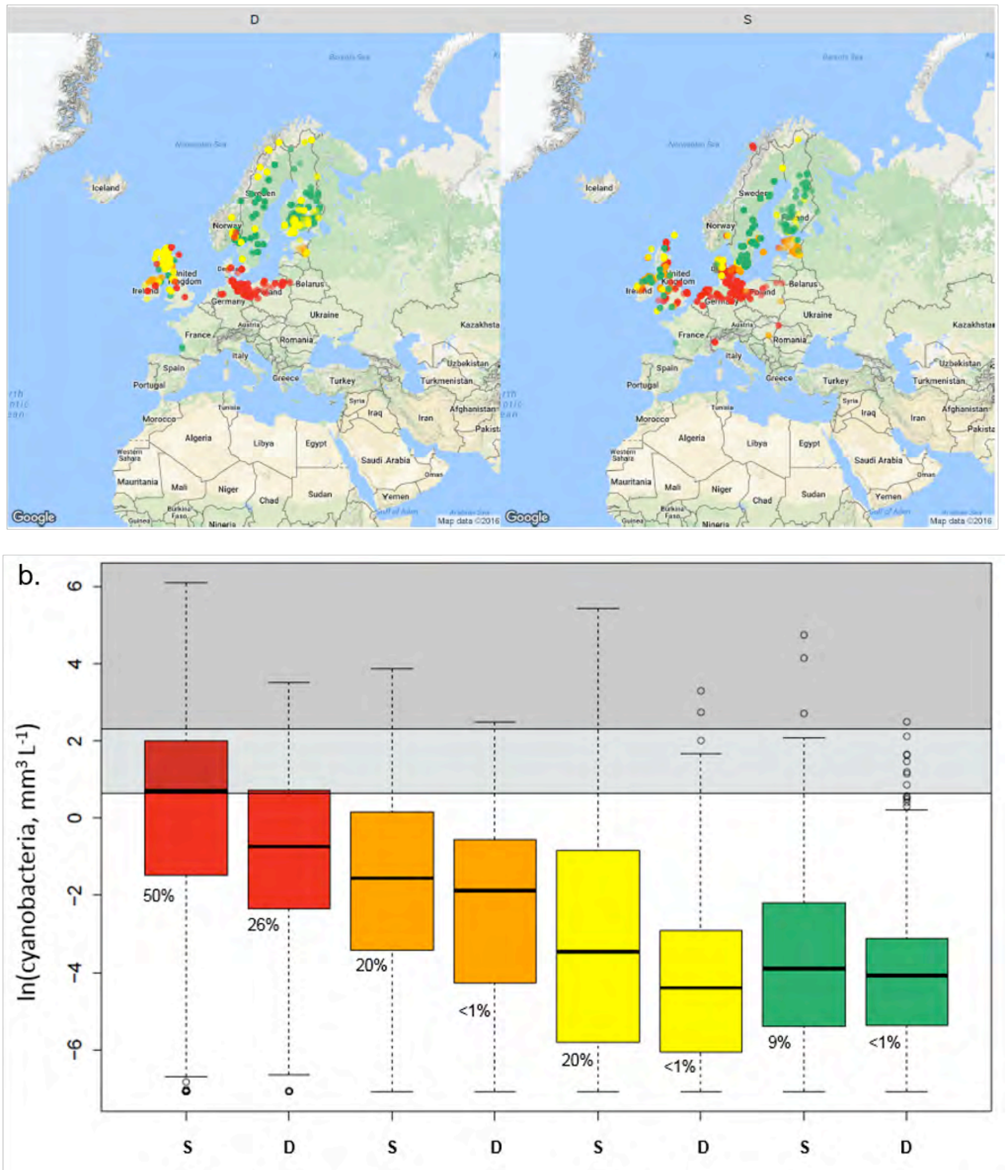

Figure 21. (a) Spatial distribution of deep (D) and shallow (S) natural lakes types: high (red), medium type 1 (yellow), medium type 2 (orange) and low (green) – see Table 3 for descriptions of these types. (b) The distribution of log monthly mean cyanobacteria biovolume ($mm^3\ L^{-1}$) in each type (S = shallow and D = deep). Shaded areas represent WHO risk thresholds, the lighter grey for exceedance of the low risk threshold and the dark grey for exceedance of the medium risk threshold. The percentages under each distribution indicates the occurrence of summer mean cyanobacteria biovolume ($mm^3\ L^{-1}$) which exceeds the low risk WHO threshold in each defined risk category.

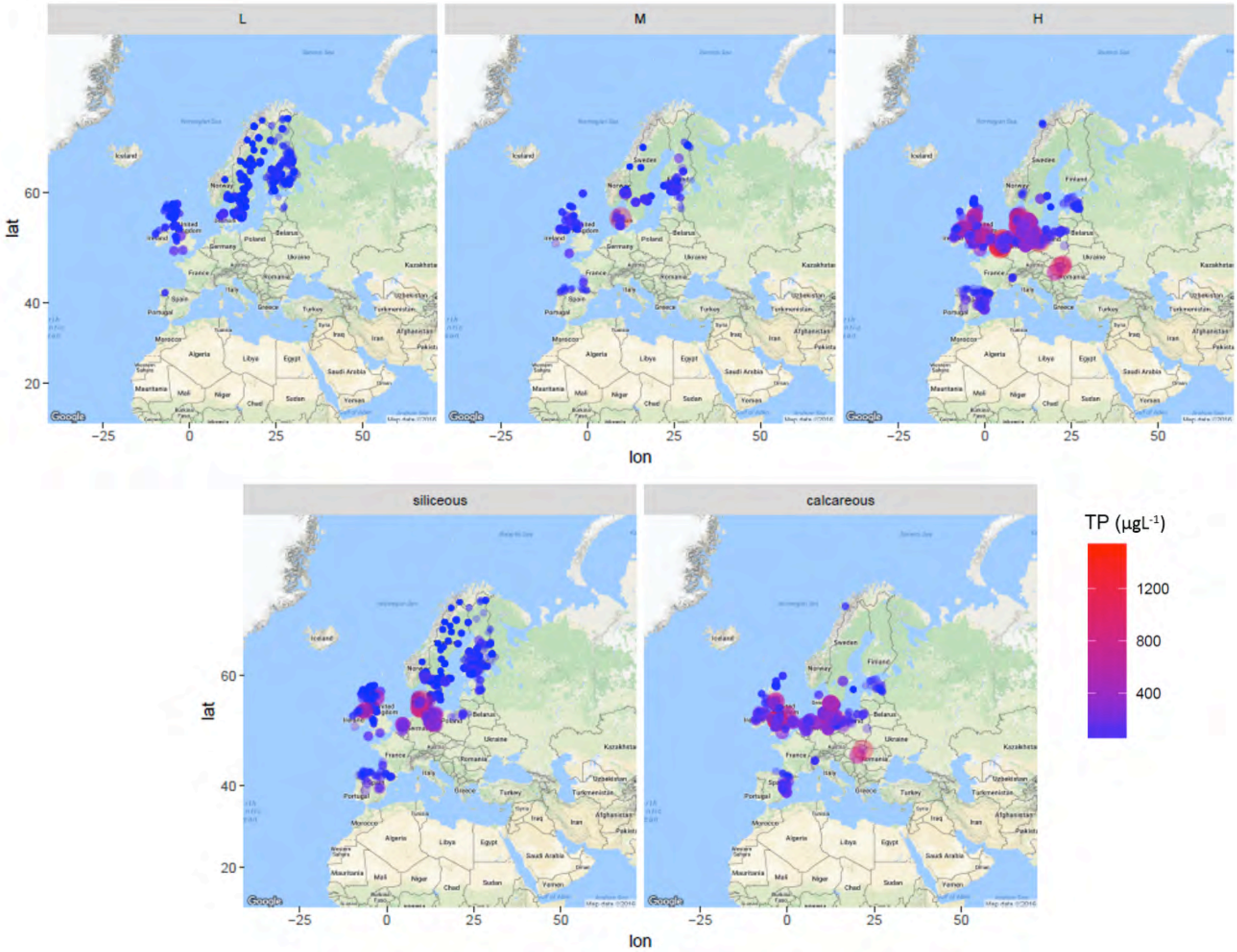

Figure 22. (a) Map showing the covariation between TP and alkalinity types – low (L), medium (M) and high (H). Lake TP concentration µg L^{-1} is represented on a scale of colour and size of the point (small to large size points and blue to red indicating small to large concetnrations of TP). (b). Map showing the covariation between TP and the geology type of the lake catchment (siliceous and calcarous).

Table 6. Response (cyanobacteria biovolume mm^3 L^{-1}) mean, min, max and stressor gradients in natural lakes.

		Cyanobacteria biovolume mm^3 L^{-1}		TP µg L^{-1}		Mean monthly temp (°C)		Total summer rainfall (mm)	
	Risk	Mean	Min-Max	Min	Max	Min	Max	Min	Max
Shallow									
	High	8.2	0-438	7	1600	9	24	1	177
	Medium.1	2.2	0-47	2	840	12	23	15	169
	Medium.2	4.1	0-224	1.5	1555	10	21	13	137
	Low	0.9	0-114	2	181	5	21	4	181
Deep									
	High	1.8	0-34	3	1570	11	24	20	116
	Medium.1	0.9	0-12	2	195	12	20	41	123
	Meidum.2	0.3	0-27	1	90	5	20	13	175
	Low	0.2	0-12	2	97	6	21	13	198

Table 7. Pearson's R correlation coefficient between TP, alkalinity, latitude, % of arable land in the lake catchment and % of forested land in the lake catchment.

	ln(TP, µg L^{-1})	ln(Alkalinity, meq L^{-1})	Latitude	ln(Arable, %)	Forested (%)
ln(TP, µg L^{-1})	1				
ln(Alkalinity, meq L^{-1})	0.58***	1			
Latitude	-0.52***	0.35***	1		
ln(Arable, %)	0.54***	0.43***	-0.39***	1	
Forested (%)	-0.33***	-0.41***	0.35***	-0.10***	1

Global responses to stressors

The overall response of cyanobacteria to ln(TP, µg L^{-1}), mean monthly temperature (°C) and total summer rainfall (mm) are shown in Figure 23. There is a significant positive relationship between cyanobacteria and TP (R=0.59) and a positive but weaker relationship between cyanobacteria and mean temperature (R=0.35), whilst the response of cyanobacteria to rainfall is weakly negative (R=-0.12). There is evidence that the response to TP and also potentially rainfall is non-linear. The non-linear response to TP is accounted for by the *a priori* decision to model the response by trophic state, and potential non-linearity in the response to rainfall are checked in model residuals. These relationships show the response across all lakes and so do not account for potential interactions between stressors nor the effects of lake type on this response.

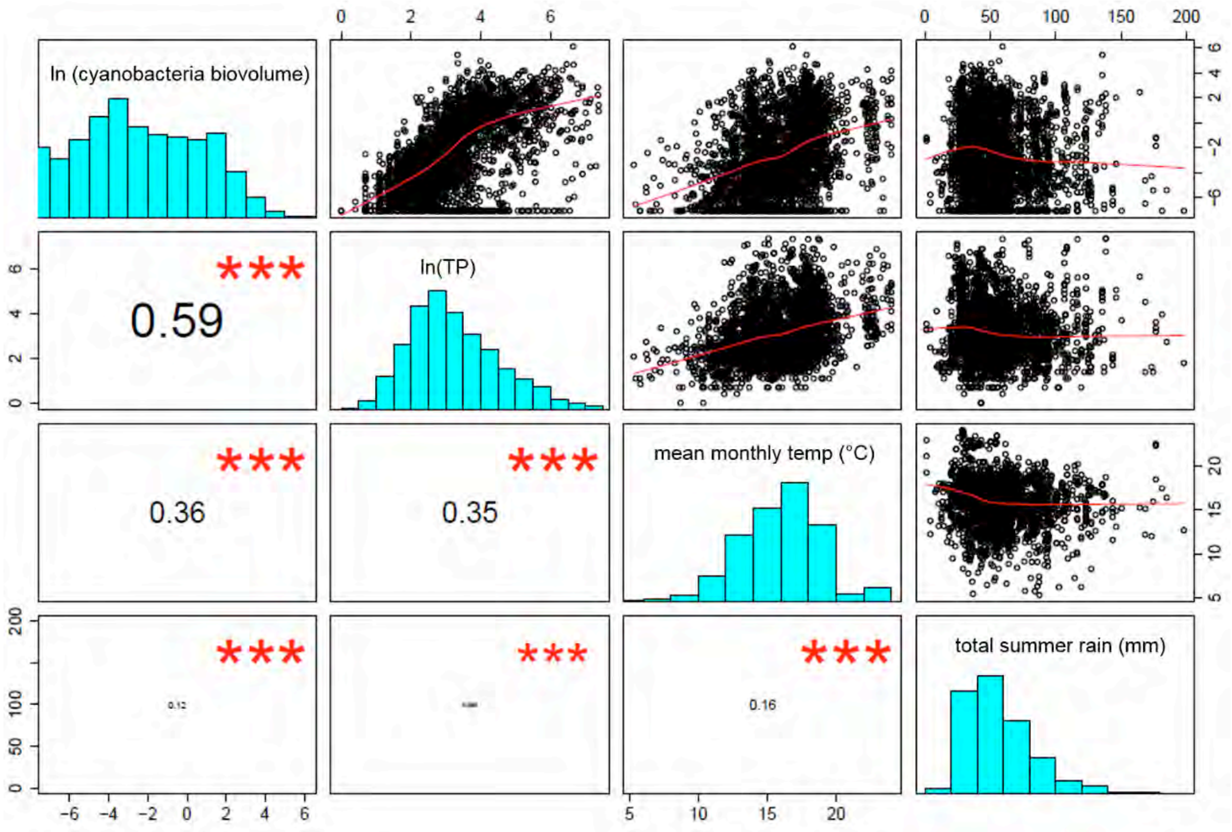

*Figure 23. Relationship between ln(cyanobacteria biovolume, mm^3 L^{-1}) and stressors: ln(TP, µg L^{-1}), monthly mean temperature (°C) and total summer rainfall (mm). The top right diagonal panel shows the data with a locally weighted regression line fitted to identify any non-linear relationships, the middle diagonal panels show the distribution of each variable and the lower diagonal panels show Pearson's correlation coefficients, the size of the text representing the relative magnitude of r and *** indicating that the pairwise relationship is significant at the <0.001 level.*

Shallow lakes (Table 8)

High risk. In high risk lakes (high alkalinity and low humic types) cyanobacteria biovolume is significantly lower in very small lakes and has a significant positive relationship with TP (estimate = 0.7, se = 0.2, t value = 4). These effects explained 15% of the variance in cyanobacteria biovolume.

However, modelling the response by trophic state showed dissimilarities from the global model. In all models, with exception to hypertrophic lakes there was significantly less biovolume in very small lakes, in hypertrophic lakes there was no effect of surface area although biovolume is lower in very small lakes (Figure 25 a, the bottom layer is the average response in very small lakes). The effect of stressors alone and in combination varied depending on trophic state. Only eutrophic lakes had the same model construct as the global model with a significant effect of TP (estimate = 1.3, se = 0.6, t value = 2.1) and lower biomass in very small lakes (R^2 =0.19). The size effect of TP was higher in eutrophic lakes (estimate of 1.3) than when considering the full nutrient gradient (estimate of 0.7). In mesotrophic (R^2 = 0.39) and hypertrophic lakes (R^2 =0.09) there was a significant interaction between TP and summer rainfall, however in mesotrophic lakes this interaction also depended on temperature. Figure 24 shows the how the negative interaction between TP and rainfall increases as temperature increases and Figure 25 a shows the interaction between TP and rain in hypertrophic lakes; the different layers showing a non-significant effect of surface areas but an influence of this variable in model selection. As there was insufficient data to model the response in oligotrophic nutrient gradient, the response to stressors within oligotrophic-mesotrophic TP interval was modelled. In these lakes biomass was significantly lower in small and very small lakes but variance in cyanobacteria biovolume was not explained by any stressor ($R^2 = 0.33$).

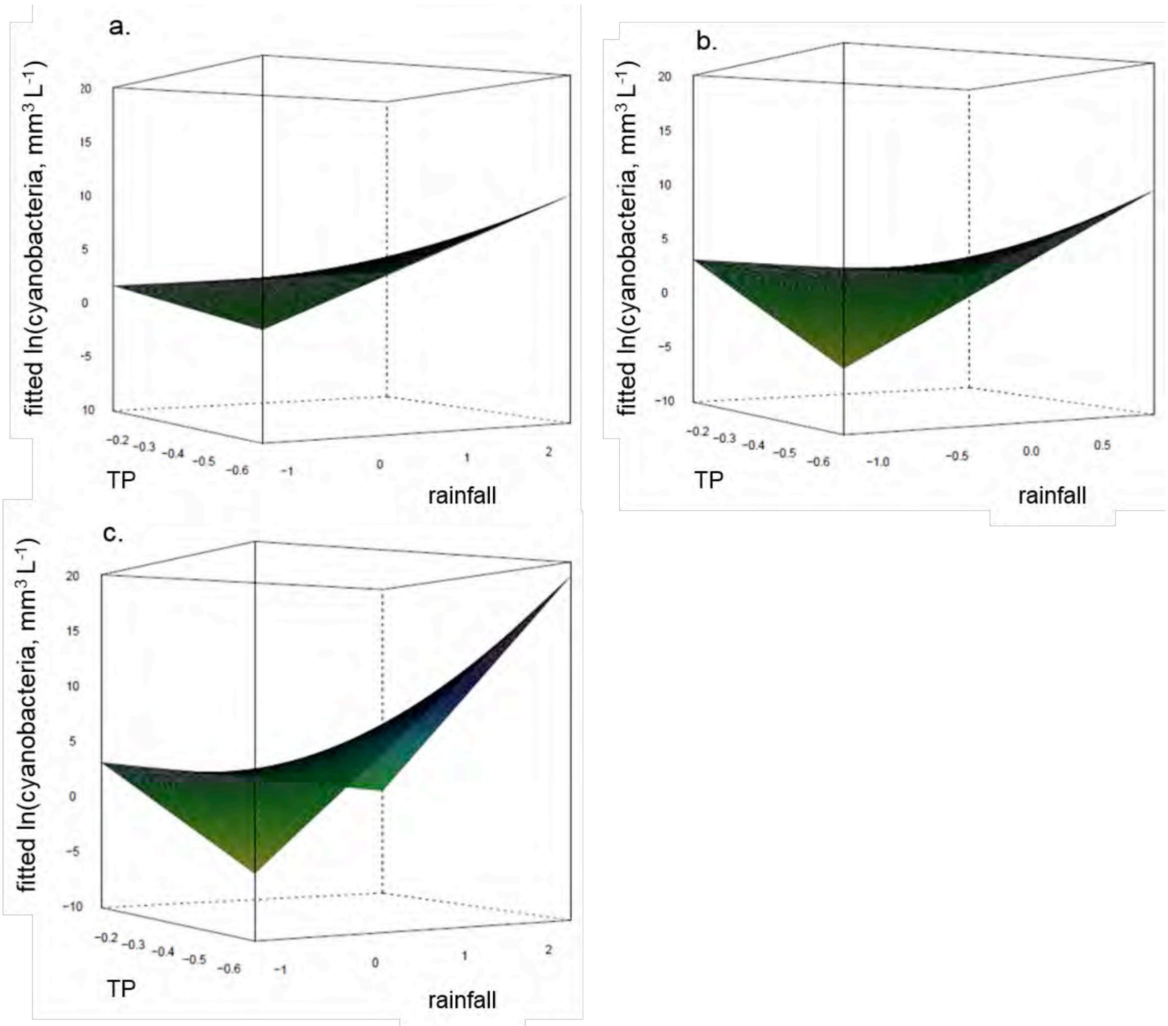

Figure 24. Negative interaction between TP, summer rain and temperature in shallow, mesotrophic high risk lakes. Plots a – c shows how the relationship between TP and rain becomes more negative with an increase in temperature: a. <16 °C, b. 16.1 – 19.4 °C, c. 19.5 – 23 °C. All stressors are standardised (scaled to have a mean of zero and a standard deviation of one).

Medium risk, Type 1. In medium risk type 1 lakes (high alkalinity and med-high humic types) cyanobacteria biovolume was explained by a negative interaction between TP, summer rainfall and temperature which explained 7% of the variance. Modelling by trophic state resulted in higher explained variance in hypertrophic lakes (R^2 = 0.37); in these lakes the response to each stressor was negatively influence by another stressor i.e. significant negative interactions between TP and temperature, TP and total summer rainfall and temperature and summer rainfall. In oligotrophic-mesotrophic lakes there was a significantly positive interaction between temperature and summer rainfall (R^2 = 0.08) whilst in eutrophic lakes there was a significant positive interaction between TP and temperature (R^2 = 0.08). In mesotrophic lakes cyanobacteria biovolume was not explained by any stressor alone or in combination, the most parsimonious model containing the random intercept term i.e. between lake variance (R^2m = 0, R^2c=0.59). There was no effect of surface areas in any medium risk type 1 lakes, irrespective of the nutrient gradient.

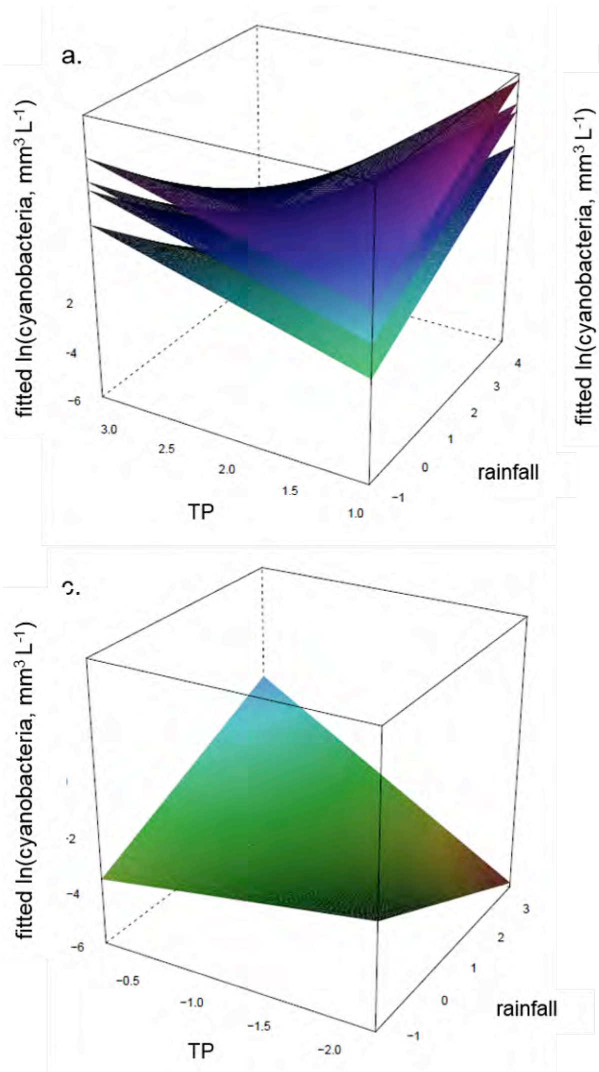

Figure 25. Interactions between total summer rainfall and TP. Negative interaction in a. high risk, shallow, hypertrophic lakes and b. low risk, shallow, hypertrophic lakes. c) shows a positive interaction in medium risk type 1, shallow, oligotrophic-mesotrophic lakes. The different layers in a. shows how the average response varies by surface area type, the bottom most layer is the response in very small lakes. The effect of surface area is not shown for all plots. All stressors are standardised (scaled to have a mean of zero and a standard deviation of one).

Table 8. Overview of interactions and main effects of stressors in shallow (≤5 m) lakes of different cyanobacteria risk. Dots represent terms contained in the model and the sign indicates whether the coefficients are positive or negative. Grey colours are for coefficients with a T-value of <+/- 2, interpreted as non-significant. Green are for significant negative single terms or interaction terms and red for significant positive single terms or interaction terms. Significance at the 5% level is interpreted as being when the t value is ≥± 2. Models which do not contain significant interaction terms are underlined and coefficients, se and t values are reported in Table 10.

Risk category	N	R^2_M	R^2_C	T - values									
				TP	°C	rain	VS	S	M	TP * °C	TP* rain	°C *rain	TP * rain °C
High	567	0.15	0.67	🔴+			🟢-						
Oligo-meso	71	0.33	0.86				🟢-	🟢-	⚫-				
Mesotrophic	61	0.39	0.84	⚫-	⚫+	⚫-	🟢-	⚫-	⚫-	⚫-	🟢-	⚫-	🟢-
Eutrophic	214	0.19	0.67	🔴+			🟢-	⚫-	⚫-				
Hypertrophic	282	0.09	0.61	⚫+		🔴+	⚫+	⚫-	⚫-		🟢-		
Medium Type 1	157	0.07	0.67	⚫+	⚫+	⚫-				⚫-	⚫+	⚫+	🟢-
Oligo-meso	42	0.08	0.81		⚫-	⚫-						🔴+	
Mesotrophic	33	0	0.59										
Eutrophic	74	0.08	0.61	⚫+	⚫-					🔴+			
Hypertrophic	41	0.37	0.67	⚫+	🔴+	🔴+				🟢-	🟢-	🟢-	
Medium Type 2	132	0.06	0.91	🔴+									
Oligotrophic	64	0	0.8										
Oligo-meso	87	0.31	0.84	⚫+	🔴+		🟢-	⚫+	⚫-		🔴+		
Mesotrophic	23	0.61	0.99	🟢-	🔴+	🔴+	🟢-	⚫+	⚫-	🔴+	🔴+	🟢-	🟢-
Eutrophic	38	0.04	0.9	⚫+	⚫+							🟢-	
Low	422	0.17	0.75	🔴+	🔴+	🟢-	🟢-	⚫-	⚫-	🔴+			
Oligotrophic	144	0	0.61										
Oligo-meso	261	0.07	0.8				🟢-	⚫-	⚫-				
Mesotrophic	117	0.08	0.84				🟢-	⚫-	⚫-				
Eutrophic	140	0.3	0.68		🔴+		🟢-	⚫-	⚫-				
Hypertrophic	21	0.53	0.94	🟢-	🟢-	🔴+	🟢-	⚫-	n.a		🟢-		

Medium risk, Type 2. In medium risk type 2 lakes (low alkalinity and low humic types) cyanobacteria biovolume was explained by a significant positive effect of TP (estimate=0.9, se = 0.3, t value = 2.8, R^2=0.06). In oligotrophic lakes cyanobacteria biovolume was not explained by any stressor (R^2m = 0, R2c = 0.8) whilst in oligotrophic-mesotrophic, mesotrophic and eutrophic lakes there were significant interactions between stressors (there was insufficient data to model the response in hypertrophic lakes). Most variance was explained in oligo-mesotrophic and mesotrophic lakes (R2 = 0.31 and R^2 = 0.61, respectively) whilst negligible variance was explained by the model in eutrophic lakes (R^2 = 0.04). The interactions identified depended on the interval of the nutrient gradient, when combining oligo-trophic and mesotrophic lakes a significant positive interaction between TP and summer rainfall was seen (Figure 25 c) whilst modelling the response within a mesotrophic lakes identified more complex interactions between all stressors (negative interaction between TP, summer rainfall and temperature).

Low risk. In low risk lakes (low alkalinity and med-high humic types) cyanobacteria biovolume was explained by a significant negative effect of summer rainfall and surface area type and a significant positive interaction between TP and temperature (R^2 = 0.17). As with other risk types, the response of cyanobacteria to stressors depends on the trophic state. In oligotrophic lakes cyanobacteria biovolume was not explained by any stressor (R^2m = 0, R^2c = 0.61) whilst in oligotrophic-mesotrophic and mesotrophic lakes variance was explained by surface area type, with biovolume being significantly lower in very small lakes (R^2=0.07 and R2=0.08, respectively).

From a total of 21 shallow lake models there was an effect of stressors on cyanobacteria in two-thirds of the models, in the remaining either variance was explained in part by surface area or no variance was explained by fixed terms in the model. Antagonisms (50% of models) were more common than synergisms (29% of models) and additive effects (21% of models). All significant interactions between TP, summer rain and temperature were negative whilst lower order interactions showed less consistency, although generally interactions between TP and temperature were positive, between TP and summer rainfall were negative and between temperature and summer rainfall were negative (Table 8). Because of the presence of interactions in most cases and differing TP gradients between ‘risk types’ it is only possible to compare the size effects of stressors in two risk types: high risk eutrophic lakes and low risk eutrophic lakes. In high risk lakes there is a positive effect of TP (estimate = 1.3) whilst in low risk lakes there is no effect of TP but a weaker but also positive effect of temperature (estimate = 0.7) suggesting that is highly likely that high risk lakes are more sensitive to nutrient enrichment.

Deep lakes (Table 9)

High risk. In high risk lakes (high alkalinity and low humic types) there was a significant positive effect of TP (estimate = 0.8, se = 0.2, t value = 4.6) but a significant negative effect of temperature (estimate = -0.3, se = 0.1, t value = -2.9) and summer rainfall (estimate = -0.35, se = 0.1, t value = -2.3), R^2 = 0.07. Generally modelling the response within trophic states does not improve the variance explained by fixed effects (mesotrophic R^2 = 0.03, eutrophic R^2 = 0.09 and hypertrophic R^2 = 0) apart from when oligotrophic and mesotrophic lakes are combined (R^2=0.14). In these lakes variance is explained by the significant positive effect of TP (estimate

= 2.3, se = 0.6, t value = 3.7) and effect of surface area (significantly lower biovolume in very small lakes). The response is similar in mesotrophic lakes with a significant positive effect of TP (estimate = 2.5, se = 1.1, t value = 2.2) but no effect of surface area. In eutrophic lakes there was a significant positive interaction between TP and temperature and biomass was lower in very small lakes (Figure 26 a.). The size effect of TP on cyanobacteria in high risk lakes is similar between oligo-mesotrophic lakes and mesotrophic lakes.

Medium risk, Type 1. In medium risk type 1 lakes (high alkalinity and med-high humic types) cyanobacteria biovolume was not explained by any stressor ($R^2m = 0$, $R^2c = 0.69$). There was insufficient data to model the response within trophic gradients.

Medium risk, Type 2. In medium risk type 2 lakes (low alkalinity and low humic types) cyanobacteria biovolume was most explained by a significant positive effect of TP (estimate=0.4, se = 0.2, t value =) and temperature (estimate = 0.21, se = 0.1, t value), however despite the significance of these terms, overall this only explained 0.3% of the variance. The same model construct was selected for oligotrophic and oligo-mesotrophic lakes which explained 0.6% and 0.3% of the variance, respectively. The size effect of TP in oligotrophic lakes was higher (estimate = 0.7, se = 0.3, t value = 2.5) than in oligo-mesotrophic lakes (estimate = 0.4, se = 0.2, t value = 2.2) whilst the effect of temperature was similar in both, and had less effect than TP (estimate = 0.3, se = 0.1, t value = 2.7 and estimate = 0.2, se = 0.1, t value = 2.5, respectively). In mesotrophic lakes there was a negative interaction between TP and rainfall (R2 = 0.07). There was insufficient data to model the response in eutrophic and hypertrophic lakes.

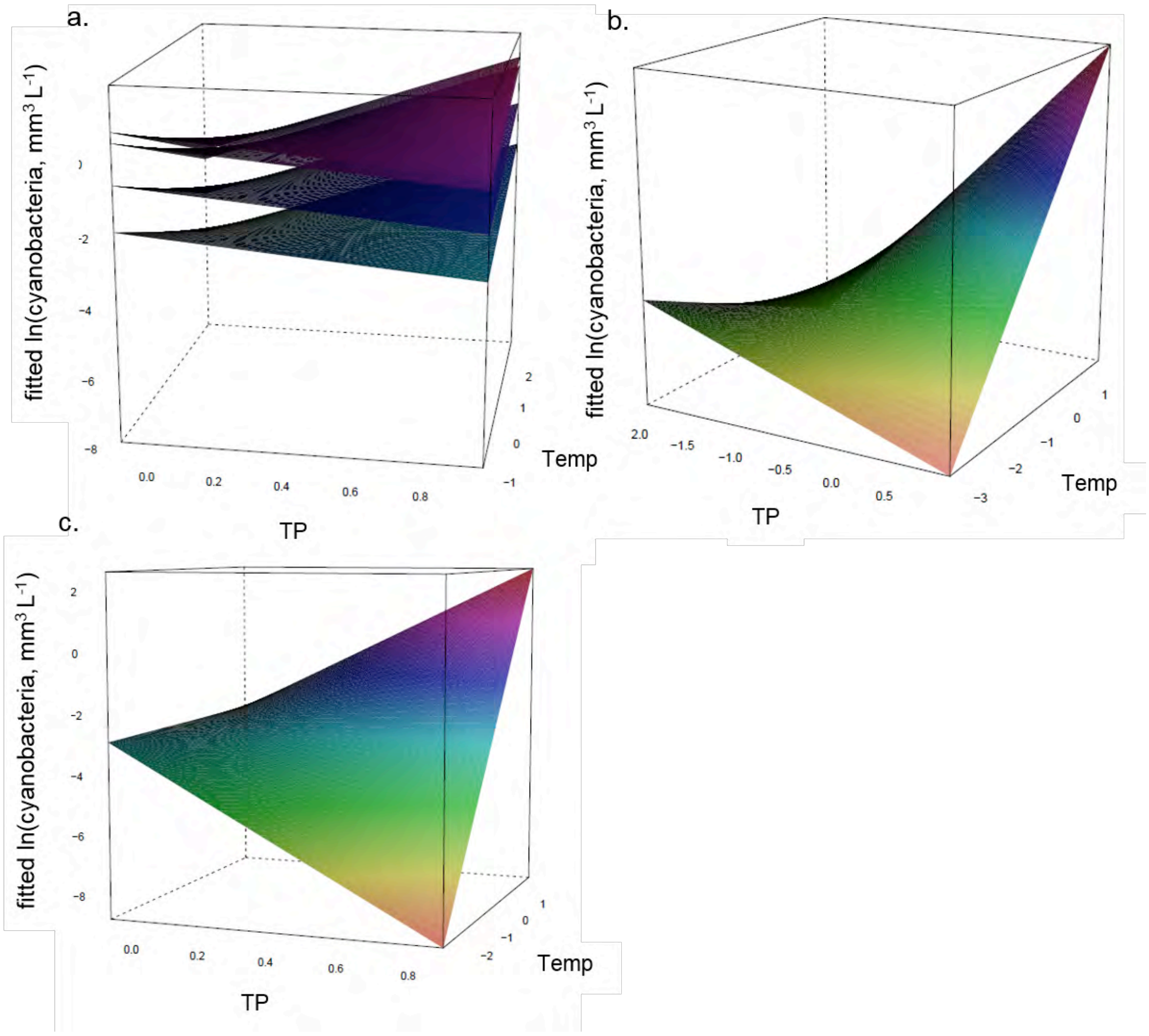

Figure 26. Positive interactions between mean temperature and TP in a) high risk, deep, eutrophic lakes (temperature gradient length: 12-24 °C) b) low risk, deep lakes and c) low risk, deep, eutrophic lakes (temperature gradient length: 7-21 °C). All stressors are standardised (scaled to have a mean of zero and a standard deviation of one).

Table 9. Overview of interactions and main effects of stressors in <u>deep</u> (>5 m) lakes of different cyanobacteria risk. Dots represent terms contained in the model and the sign indicates whether the coefficients are positive or negative. Grey colours are for coefficients with a T-value of <+/- 2, interpreted as non-significant. Green are for significant negative single terms or interaction terms and red for significant positive single terms or interaction terms. Significance at the 5% level is interpreted as being when the t value is ≥± 2. Models which do not contain significant interaction terms are underlined and coefficients, se and t values are reported in Table 10.

Risk category	N	R^2_M	R^2_C	T - values									
				TP	°C	rain	VS	S	M	TP * °C	TP* rain	°C *rain	TP * °C *rain
High	458	0.07	0.56	●+	●-	●-							
Mesotrophic	129	0.03	0.39	●+									
Eutrophic	243	0.09	0.74	●+	●-		●-	●+	●+	●+			
Hypertrophic	65	0	0.7										
Oligo-meso	150	0.14	0.36	●+	●-		●-	●+	●+				
Medium type 1	36	0	0.69										
Medium type 2	414	0.03	0.68	●+	●+								
Oligotrophic	269	0.06	0.67	●+	●+								
Mesotrophic	125	0.07	0.72	●+		●-					●-		
Oligo-meso	394	0.03	0.66	●+	●+								
Low	416	0.07	0.69	●+	●+					●+			
Oligotrophic	244	0.01	0.66	●+	●+	●+				●+	●+	●+	●+
Mesotrophic	102	0.32	0.68	●+	●-		●-	●-	●-	●-			
Eutrophic	69	0.14	0.68	●+	●+					●+			
Oligo-meso	346	0.12	0.72				●-	●-	●-				

Low risk. In low risk lakes (low alkalinity and med-high humic types) cyanobacteria biovolume was most explained by a significant positive interaction between TP and temperature (R2 = 0.07). The response to stressors then varies by trophic gradient. In oligotrophic lakes cyanobacteria biovolume was not significantly explained by any stressor (R^2m = 0.1, R^2c = 0.66) although the best model contained all terms and interactions. Unlike the global model, no interactions were found in mesotrophic and oligo-meso trophic lakes (R2 = 0.32 and R2 = 0.12, respectively whilst like the global model there was a positive interaction between temperature and TP in eutrophic lakes (R2=0.14). Figure 26 shows the how the strength of the interaction between temperature and nutrients varies by type and trophic level; the size effect is clearly larger in low risk lakes (plot c, estimate = 2.8) than in high risk lakes (plot a, estimate = 1.1). The difference in the effect size between the global model (plot b, estimate = 0.4) and eutrophic model (plot c, estimate = 2.8) highlights the importance of assessing both the type and size of the interaction along the nutrient gradient.

Table 10. Estimates plus standard errors and t value for TP, temperature and rainfall model parameters for the response of cyanobacteria in lake types where interactions are absent. Each stressor has been standardised (mean of zero and standard deviation of one) so are directly comparable between stressors and between models.

WB type	Risk	TP			Temp			Rain		
		est	se	t	est	se	t	est	se	t
Artificial	**Deep**									
	Global							0.4	0.2	2.2
	Hypertrophic	-3.6	1.2	-2.9				1.3	0.3	4.6
Natural	**Shallow**									
	High, global	0.7	0.2	4.0						
	High, eutrophic	1.3	0.6	2.1						
	Medium 2, global	0.9	0.3	2.8						
	Low, eutrophic				0.7	0.2	3.2			
	Deep									
	High, global	0.8	0.2	4.6	-0.3	0.1	-2.9	-0.3	0.1	-2.3
	High, mesotrophic	2.5	1.2	2.2						
	High, oligo-mesotrophic	2.3	0.6	3.7						
	Medium 2, global	0.4	0.2	2.4	0.2	0.1	2.3			
	Medium 2, oligotrophic	0.7	0.3	2.5	0.3	0.1	2.7			
	Medium 2, oligo-mesotrophic	0.4	0.2	2.2	0.2	0.1	2.5			

From a total of 15 shallow lake models there were stressor effects in 73% of the models, in the remaining either variance was explained in part by surface area or no variance was explained by fixed terms in the model. Additive effects (58% of models) were more common than synergisms (25% of models) and antagonisms (17% of models). All positive interactions were between TP and temperature whilst negative interactions were between TP and rainfall and TP and temperature. The main effect of TP and temperature (from additive models) varied by risk type and also by trophic gradient. The effect of temperature was always weaker than TP and was fairly consistent between trophic gradients within risk types, however the effect was much higher in high risk mesotrophic lakes (estimate = 1.1) than in medium risk type 2 oligotrophic and oligo-mesotrophic lakes (estimate = 0.3 and 0.2, respectively) – see Table 10. The effect of TP on the other hand varied depending on risk type and trophic type. The highest response to TP was in high risk mesotrophic lakes (estimate = 2.52) whilst the lowest response to TP was in medium risk type 2 oligo-mesotrophic lakes (estimate = 0.4). There appears to be a difference in the effect size when comparing oligotrophic lakes to oligo-mesotrophic, with the combination dampening the size effect in medium risk type 2 lakes whilst there is no clear difference when comparing the size effects between oligo-mesotrophic and mesotrophic in high risk lakes (Table 3).

The response between corresponding risk types in shallow and deep lakes showed little consistency when comparing interactions. However, the effect of random effects was large for all types and trophic states, reflecting that the response of individual lakes to stressors within each subset contributed considerably to the overall variance.

Discussion

Our results highlight that it is not possible to generalise across all lakes how cyanobacteria respond to stressors acting alone or in combination. Instead we found that the significance and direction of interactions among anthropogenic stressors depends on environmental context, defined here as combinations of lake attributes which explain variation in cyanobacteria, the gradient of the nutrient stressor and also natural variation between individual lakes.

Our results show that the impact of multiple stressors on cyanobacteria are strongly affected by the characteristics of the lake, specifically: depth, water colour, alkalinity and whether the waterbody is natural or heavily modified. This is in agreement with other studies which have highlighted the importance of incorporating lake type in large scale analyses, identifying water colour and alkalinity particularly as being important variables to consider (Carvalho et al., 2011; Moe et al., 2014). We also found that the response varies depending on the gradient of the nutrient stressor. This *a priori* decision was based on experimental work in streams (Piggott et al., 2015) and was supported by our results as well as from another study which found that the response changed depending on the nutrient gradient (Rigosi et al., 2014). However, it has been highlighted in a recent methodology paper that for stressor interactions to be detected, 75% of the stressor gradient is needed (Feld et al., 2016). We found that significant interactions could be detected within approximate quartiles of the gradient, even when the number of observations were low (<100). This could be because TP is a strong driver of cyanobacteria, for stressors with weaker effects longer gradients may be needed. In this analysis, the exploration of interactions along climatic gradients were not considered appropriate or necessary as data was limited to fairly narrow gradients and there was little evidence for non-linearity in the response of cyanobacteria to air temperature or summer rainfall. Our results also highlight the need to incorporate the structure of the data into the model when using large, nested datasets so that the effects of explanatory variables can be disentangled from natural variation at the lake level. In many of our models, despite significant effects of stressors or combinations of stressors, a low proportion of the variance was explained by these effects whilst among-lake variance remained high. The analysis of time series data (Chapter 2.3), which explores the response of cyanobacteria to the same stressor combinations, also found that within lake types there is variation in the average response but also that the response (i.e. the slope) can vary among lakes.

We did not find any clear patterns in the significance and direction of interactions between types and gradients. Although Rigosi et al. (2014) supports our result that interactions vary by eutrophication state (identifying synergisms in eutrophic and hypertrophic lakes), it is not possible to comment on whether the interactions in our study are consistent with theirs as their models did not account for any confounding effects of lake typology, additional stressors or between lake variation. Of particular interest for managers is the incidence of synergistic interactions - overall there was a higher prevalence of interactions (50%) in shallow lakes compared to deep lakes (38%), however the proportion of which were synergisms was higher in deep (lower risk) lakes (60%) than shallow lakes (36%). In the risk type of most concern (shallow, high risk) we did not find any evidence of synergistic interactions. As hypothesised, synergisms were found between temperature and nutrients, but this interaction was not prevalent between risk types and the magnitude of the effect of the interaction. We found that the interactions detected depend on how the gradient is defined, e.g. when comparing oligotrophic to oligo-mesotrophic and that the effect size of the interaction varies between types. These results highlight the importance of assessing both the type and size of the interaction at the lake level.

This work makes an important extension of previous multiple stressor research, incorporating lake type as an informative component of the response whilst also using more advanced statistical methods to improve the estimation of the effects of stressors. However, conclusions from this analysis should be carefully drawn. Firstly, the availability of biological and environmental data was limited to central and northern regions; as a result, the analysis lacked strong climatic gradients, and so inferences from these models are limited to similar climatic gradients. Weaker effects of climatic variables and low ability to detect interactions may be a result of short gradients or that the expression of these stressors could be non-informative in the context of the response of cyanobacteria. Without information about the thermal profile of the lake, air temperature alone may not be informative enough in explaining cyanobacteria variance. This is especially pertinent in shallower, polymictic lakes which respond more dynamically to changes in temperature but also other local climatic factors (Huber et al., 2012; Jöhnk et al., 2008). Similarly, changes in flushing rates would be more translatable to shifts in cyanobacteria ecology than measurements of direct rainfall (Elliott, 2010). There was also an imbalance in the number of lakes represented within each risk type and nutrient gradient, incorporating uncertainty and limiting the comparison between types/gradients where data is insufficient or missing. In particular, there was a clear bias in the spatial distribution of artificial deep and shallow lakes which likely co-varies with operational use; most deep lakes were located in Spain where artificial waterbodies are used for drinking water and irrigation whilst most shallow lakes were in the Netherlands where waterbodies are modified to impound water and suffer problems with cyanobacteria blooms in the summer months because longer retention times (Jagtman et al., 1992; Verspagen et al., 2006). This bias means that inferences cannot be made about the artificial deep and shallow lakes in general. These points highlight some of the issues with large datasets, which are invariably not compiled for question specific research; for future work in assessing the response to multiple stressor, data which cover stronger stressor gradients, more ecologically relevant measures of stressors and over a better balance of lake types and other confounding variable are needed. However, despite these uncertainties, this work highlights that when managing multiple stressors, the characteristics of the waterbody, the gradient of the stressor and also system specific variation needs to be considered.

Key messages

- The response of cyanobacteria to multiple nutrient and climatic stressors cannot be generalised across all lakes.
- The typology of the lake (specifically depth, water colour and alkalinity) as well as the nutrient gradient considered can alter the interactions between stressors.

- The average response can vary considerably between individual lakes within the same type and stressor gradient, resulting in uncertainty of the magnitude of the response at the individual lake level.
- We found some evidence for a synergistic interaction between temperature and nutrients, but this interaction was not prevalent and varied in the magnitude of the effect.
- For management of the multiple stressors effect on cyanobacteria, both the typology of the lake and the gradient of the stressor at the individual lake level need to be considered.

2.3. Effects on abundance of cyanobacteria (time series)

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Summary

The responses of two lake phytoplankton metrics used in WFD status assessment, summer chlorophyll concentrations and cyanobacterial biovolume, were examined in relation to three stressors: nutrients (spring total phosphorus), summer rainfall and summer temperatures, acting alone or in combinations of two stressors. The analysis was based on 705 lake-years of data from 26 lakes, with chlorophyll and cyanobacteria data for 677 and 596 lake years respectively. The data span 1964-2014 and cover wide environmental gradients. The analysis methods followed MARS work package 6 guidance (Chapman et al. April 2016). The following results only consider the cyanobacteria response.

A highly significant relationship was observed with spring TP, explaining about 7% of variation in cyanobacteria biovolume. The response to TP was often weak in individual lakes, but generally positive. The strength of the individual lake response depended on the lake's TP gradient and the lake's position on the TP gradient. A separate analysis of Loch Leven time-series dataset alone indicated that the relationship with TP varied considerably with the season: from a negative to a positive relationship going from winter to summer TP.

The response to spring TP varied significantly by eutrophication level (and alkalinity type), with lower levels of cyanobacteria in oligo-mesotrophic lakes compared with eutrophic lakes, as expected. Similarly, as expected, cyanobacteria in oligo-mesotrophic lakes were more sensitive to increasing spring TP compared to eutrophic lakes.

The response to summer rainfall was very weak when the global dataset was examined. Individually, lakes with relatively short residence times (<0.5 years) showed a strong negative relationship between cyanobacteria and summer rainfall, with a significant effect explaining 15% of the total variation in cyanobacteria. Lakes with longer residence times had more varied or flat responses.

The general response to summer temperature was also very weak. There was little difference in the mean cyanobacteria biovolume in summer between cool (<15 °C), warm (15-17 °C) and hot (>17 °C) summers, although much higher values were observed in hot years (i.e. significant response in upper percentiles).

There were no significant interactions between Spring TP and Summer rainfall in the global dataset. In lakes with short residence time, effects of rainfall and TP generally appeared to be additive. There was a significant antagonistic interaction between Spring TP and Summer temperature: the positive effect of Spring TP did not remain significant at high temperatures. Other algal groups may have been favoured by this combination of stressors.

The analysis shows that it is difficult to generalise how cyanobacteria respond to these stressors and how they interact. Individual lake responses are based on a combination of lake characteristics and the position of lake along the stressor gradient. Generic lake type relationships are more apparent and these indicate that cyanobacteria are most sensitive to nutrient stress in lakes of low nutrient status and sensitive to summer rainfall in lakes with short residence time.

Introduction

The responses of two lake phytoplankton metrics used in WFD status assessment (summer chlorophyll concentrations and cyanobacterial biovolume) have been examined in relation to three stressors: nutrients (spring total phosphorus), summer rainfall and summer temperatures, acting alone or in combinations of two stressors. The analysis is based on 705 lake-years of data from 26 lakes (Figure 27), with chlorophyll and cyanobacteria data for 677 and 596 lake years respectively. Data span 1964-2014 and cover wide environmental gradients.

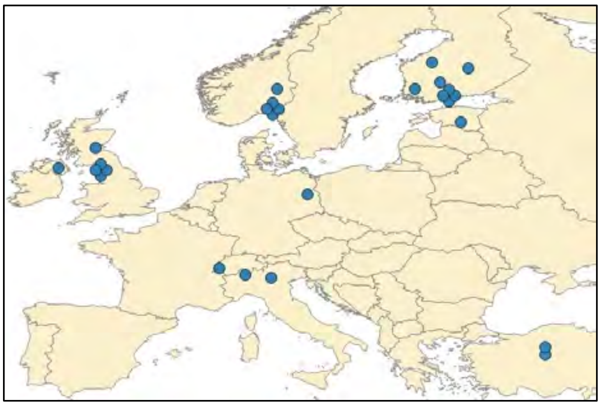

Figure 27. Distribution of 26 lakes in 8 countries used in the time-series analysis.

Methods

The analysis followed a Generalized Linear Mixed Effects Modelling framework using the mgcv package in R. Lake and year were considered as random effects in the models with slopes and intercepts allowed to vary by lake. Lake types were considered as fixed categorical effects: e.g. Trophic Type (oligo-meso, eutrophic), Residence Type (short, medium, long residence times) or Mixing Type (mixed or stratifying lake). All data were transformed (Box-Cox) and centred. The methods followed the guidance from MARS WP6 guidance (deliverable 6.1-1).

Results

The following results only consider the response of cyanobacteria.

Response to nutrient stress

Examining the whole dataset, a highly significant relationship was observed with spring TP, explaining about 7% of variation in cyanobacteria biovolume (Figure 28).

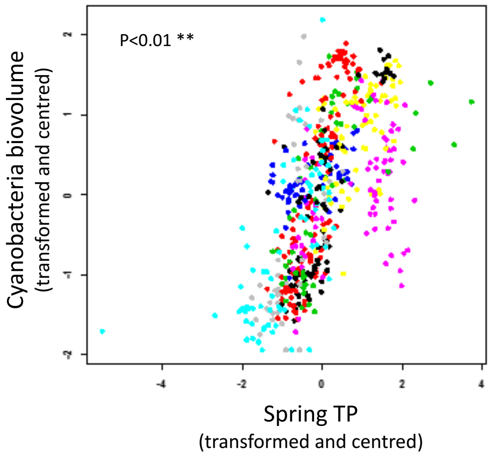

Figure 28. Summer cyanobacteria response to Spring Total Phosphorus (TP) in global dataset of 26 European & Turkish lake time-series. Colours represent individual lakes.

The response to TP was often weak in individual lakes, but generally positive (Figure 29). The strength of the individual lake response depended on the breadth of the lake's TP gradient in the time series and where the lake was generally placed on the TP gradient. A separate analysis of Loch Leven time-series dataset alone indicates that the relationship with TP varies considerably depending on the season considered in the model: from a negative to a positive relationship going from winter to summer TP

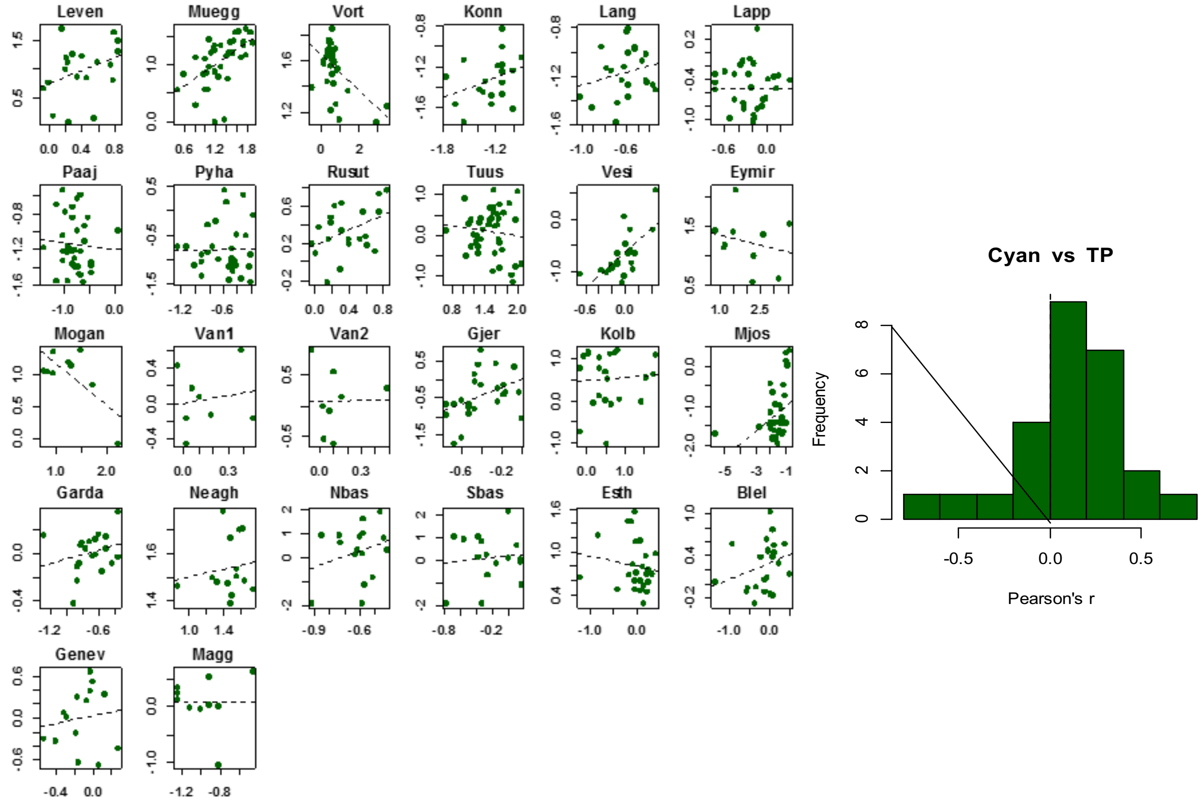

Figure 29. Summer cyanobacteria response to Spring Total Phosphorus (TP) (data centred and standardised) in 26 individual lake time-series, including a histogram of Pearson's correlation coefficients in the 26 lakes.

The spring TP – cyanobacteria relationship varied significantly by trophic type (and alkalinity type), with lower levels of cyanobacteria in oligo-mesotrophic lakes compared with eutrophic lakes, as expected. Similarly, as expected, cyanobacteria in oligo-mesotrophic lakes were most sensitive to increasing spring TP (steeper slope) compared with a relatively flat response in eutrophic lakes.

Response to hydrological stress

The summer rainfall – cyanobacteria relationship was very weak when the global dataset was examined (Figure 30). Examining individual lake responses, it appears that lakes with relatively short residence times (<0.5 years) show a strong negative relationship between cyanobacteria and summer rainfall, with a significant effect explaining 15% of the total variation in cyanobacteria (Figure 31). Lakes with longer residence times have more varied or flat responses.

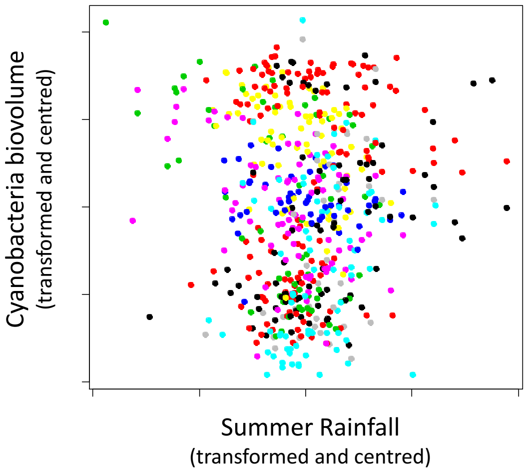

Figure 30. Summer cyanobacteria response to total Summer rainfall in global dataset of 26 lake time-series. Colours represent individual lakes.

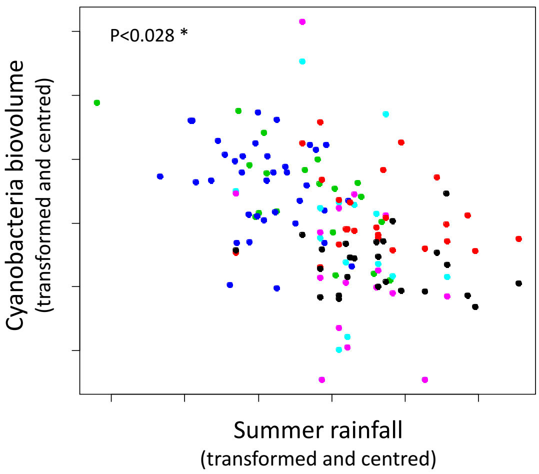

Figure 31. Summer cyanobacteria response to total Summer rainfall in short-residence time lakes. Colours represent individual lakes.

Response to temperature stress

The summer temperature – cyanobacteria relationship is very weak when the global dataset is examined, explaining <1% of the total variation in cyanobacteria (Figure 32). Further exploratory analysis showed that there was little difference in the mean cyanobacteria biovolume in summer between cool (<15 °C), warm (15-17 °C) and hot (>17 °C) summers, but generally much higher values were observed in hot years (i.e. significant response in upper percentiles, not mean) (Figure 33). No significant patterns were observed examining individual lake or lake type responses.

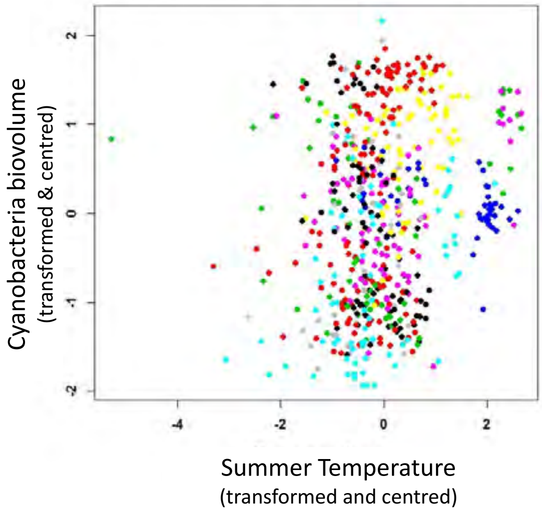

Figure 32. Summer cyanobacteria response to mean Summer temperature in global dataset of 26 European & Turkish lake time-series. Colours represent individual lakes.

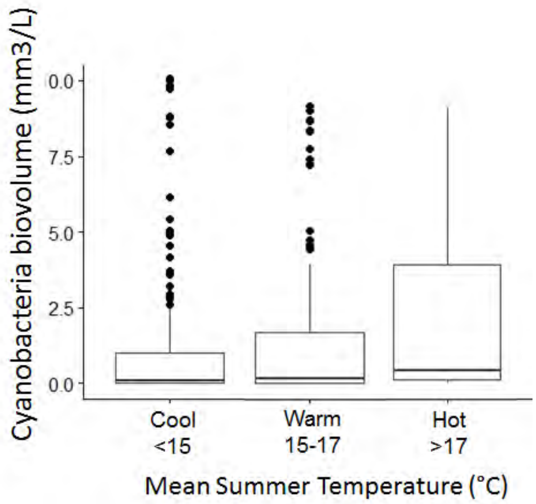

Figure 33. Boxplot of Summer cyanobacteria biovolume response to mean Summer temperatures in 26 European & Turkish lake time-series, with data grouped by years of cool, warm or hot summers.

Stressor Interactions

There were no significant interactions in the cyanobacteria response to both Spring TP and Summer rainfall in the global dataset. In short residence time lakes, rainfall and TP appeared to be additive (i.e. low summer rainfall and high spring TP increased cyanobacteria) (Figure 34).

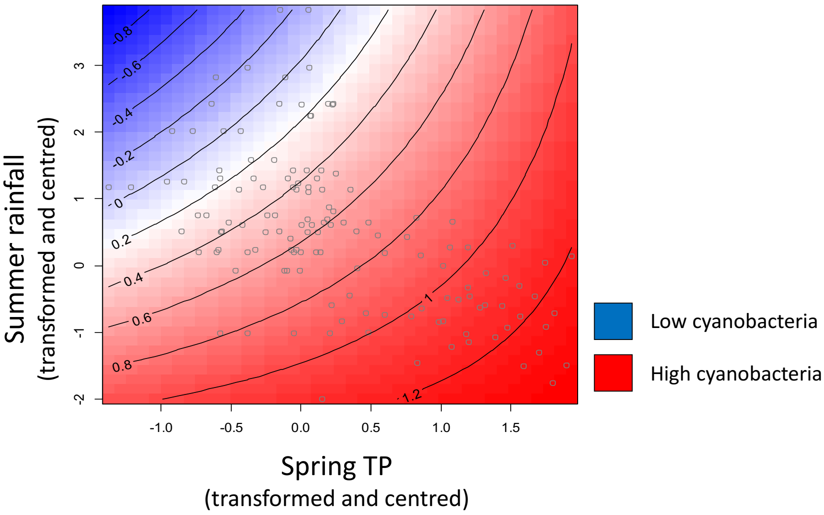

Figure 34. Interaction plot of Summer cyanobacteria response to mean Summer temperature and Spring total phosphorus (TP) in short residence time lakes. Colour indicates cyanobacteria biovolume: blue colour indicates below average and red colour indicates above average centred and standardised values.

There was a significant antagonistic interaction between Spring TP and Summer temperature in the global dataset, where the significant positive effect of Spring TP did not hold at high temperatures (Figure 35). Presumably other algal groups are favoured by this combination of stressors (based on evidence from MARS mesocosm experiment; see Deliverable 3.2).

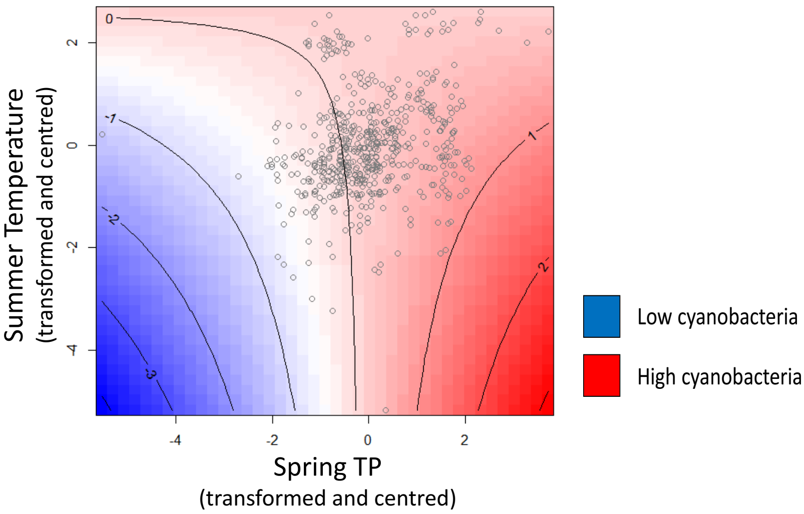

Figure 35. Interaction plot indicating Summer cyanobacteria biovolume response to mean Summer temperature and Spring total phosphorus (TP) in global dataset. Colour indicates cyanobacteria biovolume: blue colour indicates below average and red colour indicates above average centred and standardised values.

Conclusions and key messages

The analysis shows that it is difficult to generalise how cyanobacteria respond to these drivers and how they interact. Individual lake responses are based on a combination of lake characteristics and where a lake sits on the stressor gradient.

Generic lake type relationships are more apparent and these aid predictability of the response of cyanobacteria to individual and multiple stressors. The analysis indicates that cyanobacteria are most favoured by nutrient stress in lakes of low nutrient status and sensitive to summer rainfall in short residence time lakes.

Comparisons across study scales

Studies of the response of cyanobacteria to multiple stressors of nutrients, temperature and rainfall have been carried out at >770 lakes at the European scale (Chapter 2.2) and across a smaller dataset of 26 European lakes with long time-series data (Chapter 2.3). Both these studies indicate that the response of cyanobacteria to these multiple stressors, acting individually or in combination, cannot be generalised across all European lakes. In both studies the response of cyanobacteria varied by both lake typology factors as well as the nutrient gradient considered in the analysis. The typology classes in the two studies were not comparable and so detailed results cannot be compared. However, in both the spatial European analysis and the temporal analysis of lake time series, nutrients, in the form of total phosphorus (TP), had the strongest effect of all the stressors, with significant positive effects observed in global datasets and lake type datasets. The strongest effect of TP was observed in lakes at the lower end of the nutrient gradient, with little effect seen at hypertrophic conditions. Temperature appeared to interact with nutrients, although the form of this interaction was not consistent, with sometimes a synergistic interaction and sometimes an antagonistic interaction. Rainfall showed little effect in the European study but and similarly in the time-series study, but individual lakes with short residence times were sensitive to summer rainfall. In the time-series work, short residence time lakes, appeared to show an additive effect of nutrients and low rainfall.

Despite large uncertainties observed in both individual lakes and populations of lakes within a type. It is clear that two factors help predict the response of cyanobacteria to multiple stressors: typology and stress gradients. Firstly, the environmental context (or typology) of the lake, particularly depth, alkalinity, humic type and flushing type, can greatly influence the sensitivity of a lake to these stressors. Secondly, the shape of the interaction between stressors depends greatly on the gradient of the stressor in a lake over time or across a population of lakes. These studies help to locate lakes within this “response landscape” allowing managers better understanding of how their lake(s) may respond to future changes in climate and nutrient status.

2.4. Effects on community composition of phytoplankton (Northern Europe)

Contributors: Niina Kotamäki, Jannicke Moe, Anne Lyche Solheim, Birger Skjelbred, Hege Gundersen, Marko Järvinen, Olli Malve, Jessica Richardson

Summary

The phytoplankton trophic index (PTI) is one of the indices used for assessing the ecological status of lakes in Northern Europe. We analysed the response of PTI to nutrients (total P and P:N ratio) in combination with climatic variables (air temperature and precipitation), using data from 940 lakes compiled during the former EU project WISER. We used a 3-level hierarchical regression model to account for differences in stressor-response relationships among broad lake types, as well as differences in PTI level among lakes. Our results showed that PTI increased with Total P for all lake types, but the relationship varied among the lake types. The positive effect of TP was most prominent in the lowland siliceous lakes, which tend to have lower PTI inherently. The effects of climatic variables also varied considerably among the lake types. The interaction between TP and temperature was also most positive (gave higher PTI values) for the four siliceous lakes types, which tend to have lower PTI inherently. This implies that siliceous lakes are more sensitive to combined effects of TP and temperature, than the calcareous lakes.

Introduction

In the previous chapter, the responses of phytoplankton to multiple stressors were analysed for a large region of Europe, using EQR and ecological status as common response variables. The analysis revealed large-scale relationships between e.g. land use, nutrients and EQR, but this yearly aggregated ecological response variable was not particularly sensitive to climatic variables.

In this chapter we will focus on one biological index for phytoplankton, the Phytoplankton Trophic Index (PTI) (Phillips et al., 2013; Ptacnik et al., 2009). This index is one out the four indices used for assessment of ecological status of phytoplankton in the Northern Geographical Intercalibration Group (N-GIG; Ireland, UK, Norway, Sweden and Finland). The three other indices are chl-a, total phytoplankton biomass and concentration of cyanobacteria. For these three indices, the effects of nutrients in combination with temperature increase has already been studied thoroughly (e.g. (Carvalho et al., 2011; Elliott, 2010; Jeppesen et al., 2009)). The response of PTI to multiple stressors has so far not been analysed as extensively. However, the index has proved sensitive to climatic variables such as air temperature and precipitation, as well as to lake typology variables (Phillips et al., 2013). In this study we used the genus-level version of PTI, which is based on the total P optimum of each phytoplankton genus, weighted by the genus' contribution to the total biomass of the sample.

To analyse the effects of nutrients and climatic stressors in more detail, we used monthly aggregation (rather than yearly (Phillips et al., 2013)) and included the lake types in the model (Table 1). The purpose of this model is twofold: to estimate the effects of multiple stressors on PTI, and to predict the changes in PTI based under scenarios of changed stressor levels.

Data

The phytoplankton and chemistry data were obtained from the WISER central database (Moe et al., 2013) (see Chapter 1). Data from UK, Norway, Sweden and Finland were used (Figure 36). Only the records with all selected stressor variables and lake type were information were included in the analysis. The dataset used in this analysis covers 3984 records of PTI and predictor variables from 940 lakes (Table 11, Table 12). Of these lakes, 27 UK lakes belong to the Central-Baltic GIG rather than N-GIG (because of higher alkalinity level); these lakes were anyway kept in the analysis since removing them did not affect the results significantly. The samples are selected from month June to September and from year 1988 to 2009 (see Appendix 3 for more details).

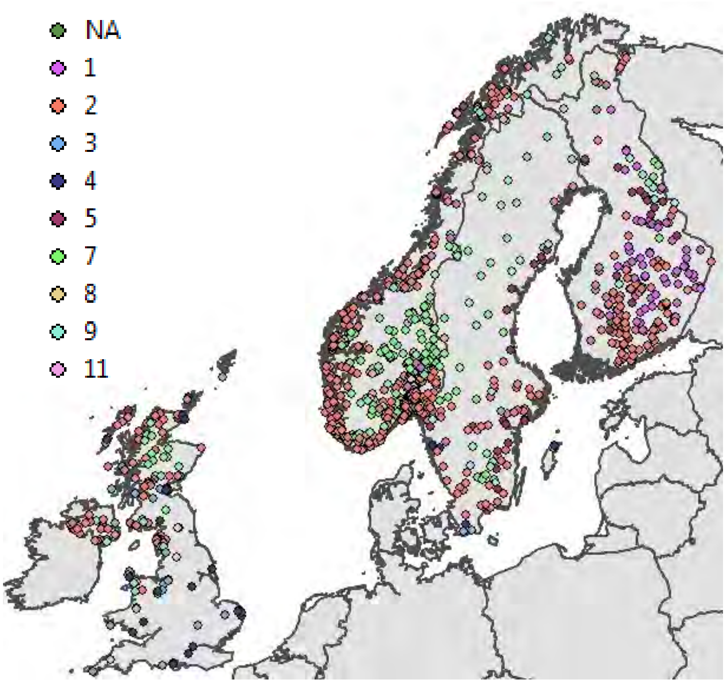

Figure 36. Map of lakes included in the analysis of PTI. Colour codes represent broad lake types (see Table 11 and Table 12).

The broad lake types are more thoroughly described in Chapter 1 (Table 1). In this dataset, the dominating lake type was type 2 (Lowland, Siliceous), followed by types 7 (Mid-altitude, Siliceous), 1 (Very large lakes) and 5 (Lowland Organic and Siliceous)

Table 11. Number of waterbodies per country and per lake type.

Lake type	FI	NO	SE	UK	Tota
1 Very large lakes, shallow or deep and stratified	76	13			89
2 Lowland, Siliceous	96	259	54	30	439
3 Lowland, Stratified, Calcareous/Mixed		8	6	11	25
4 Lowland, Calcareous/Mixed, Very shallow/unstratified		9	4	14	27
5 Lowland Organic (humic) and Siliceous	27	2	50	1	80
7 Mid altitude, Siliceous	10	130	48	20	208
8 Mid altitude, Calcareous/Mixed		8		3	11
9 Mid-altitude, Organic (humic) and Siliceous	10	18	20		48
11 Highland, Siliceous, incl. Organic (humic)		8	5		13
Total	**219**	**455**	**187**	**79**	**940**

Table 12. Number of PTI samples per country and per lake type.

Lake type	FI	NO	SE	UK	Total
1 Very large lakes, shallow or deep and stratified	349	61			410
2 Lowland, Siliceous	238	1412	282	92	2024
3 Lowland, Stratified, Calcareous/Mixed		60	19	30	109
4 Lowland, Calcareous/Mixed, Very shallow/unstratified		17	15	48	80
5 Lowland Organic (humic) and Siliceous	28	13	273	2	316
7 Mid altitude, Siliceous	15	464	255	73	807
8 Mid altitude, Calcareous/Mixed		39		8	47
9 Mid-altitude, Organic (humic) and Siliceous	10	20	115		145
11 Highland, Siliceous, incl. Organic (humic)		23	23		46
Total	**640**	**2109**	**982**	**253**	**3984**

Phytoplankton and water chemistry

The PTI index was calculated for each phytoplankton genus, based on its TP optimum score (CCA Axis 1 values in Figure 4 of (Phillips et al., 2013). For each sample, the TP optimum score (s_j) of each genus was weighted by its biomass (aj) and summed using the following formula:

$$PTI = \frac{\sum_{j=1}^{n} a_j s_j}{\sum_{j=1}^{n} a_j}$$

The PTI values of individual sample were first aggregated temporally by month (within each station), and then spatially by lake.

Likewise, total phosphorous and total nitrogen values were aggregated by month and by lake, then linked to the PTI records from the same lake-month.

Climate

The climate data were obtained from the Agri4Cast Data portal of JRC, as described in Chapter 1. The variables included in this analysis were mean summer (June - August) temperature and total summer precipitation.

Methods

The steps in the statistical analysis follow the recommendations of the MARS WP6 guidance for data analysis ([WP6]), as well the recommendations from WP4 (Feld et al., 2016). The initial exploratory data analysis (inspection of outliers, distributions etc.) is described in Appendix 3. To normalize the distributions and to linearize the relationships the variables TP, TN and TP:TN ratio were ln-transformed (base e). PTI, air temperature and precipitation were kept in the original scale. Hereafter we mean the ln-transformed variables when we refer to TP, TN and TP:TN. All explanatory variables were also standardized (z-transformation, mean=0, sd=1) to ensure comparable numerical range.

The subsequent data analysis has the following steps. First, correlations between all variables were inspected, including typology variables. Second, the effects individual stressors (nutrients and climate) on PTI for different lake types were analysed by hierarchical linear regression (mixed-effect model). Third, we inspected the combined effects of the explanatory variables following the methods recommended by WP4 (Feld et al., 2016). The impact of the different stressors on PTI were ranked by Random Forest analysis (Ishwaran and Kogalur, 2015). Furthermore, the potential interactions between predictors were ranked by the R function find.interaction. Fourth, a final hierarchical regression model for PTI was selected best on the outcome of the previous steps.

Results and Discussion

Correlations between variables

For the whole dataset the response variable PTI had highest overall correlation with TP (0.71), TN (0.57), N:P ratio (-0.44) and summer time air temperature (0.35). PTI had the lowest correlation with precipitation (0.16). The signs of the correlations make sense: higher TP and TN led to higher PTI; temperature also had positive effect on PTI. The nutrient ratio had negative overall effect: when TN:TP ratio decreases, the PTI level increases.

Not surprisingly, there was also quite high correlation between the stressor variables, indicating possible collinearity (Figure 37). The nutrient ratio TN:TP was highly correlated with TP (-0.73), but not with TN (0.0058). The correlation between TP and TN was 0.68.

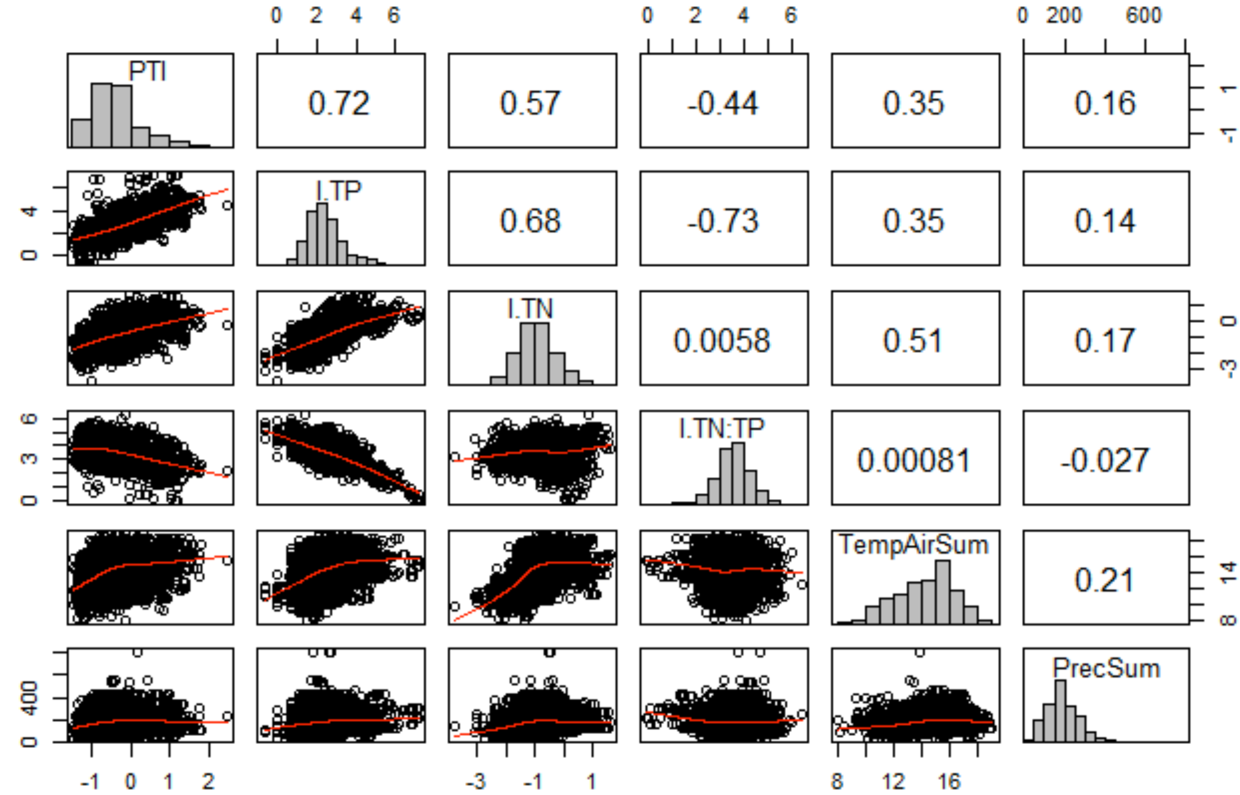

Figure 37. Pair plots showing correlations between PTI and all stressor variables included in the analysis. The histogram for each variable is displayed on the diagonal panel. The lower triangle panel shows scatterplot of data and a non-parametric regression curve (LOESS). The upper triangle panel shows the correlation coefficient. PTI = phytoplankton trophic index; l.TP = ln-transformed total phosphorus (µg); l.TN = ln-transformed total nitrogen (mg); l.TN:TP = ln-transformed ratio TN:TP; TempAirSum = mean summer air temperature; PrecSum = total summer precipitation.

PTI was also affected by the typology variables, on which the broad lake types are based (Figure 38). For the whole dataset the correlation between PTI and the mean depth was -0.42, thus the deeper lakes tend to have a lower PTI level. Alkalinity also had a clear effect on PTI: for higher-alkalinity lakes the PTI level is higher, indicating that these are more eutrophied lakes. PTI had rather strong negative correlation with altitude (-0.29) and latitude (-0.19). Pair plots (such as Figure 37) for all variables are given in Appendix 3.

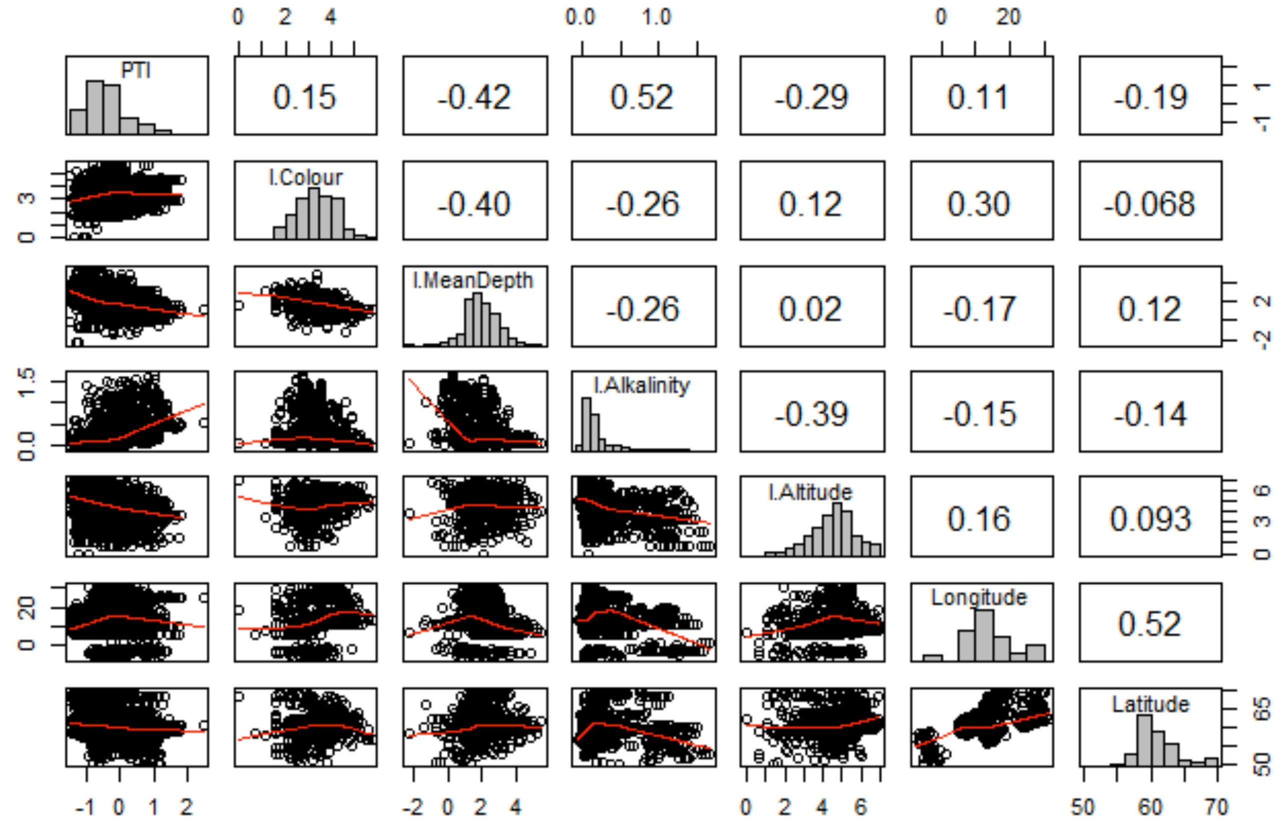

Figure 38. Pair plots showing correlations between PTI and all lake typology variables included in the analysis. For more information, see Figure 37.

Even though TP and TN:TP ratio were highly correlated (see Figure 37), the variance inflator factors (VIFs) were all under 3, indicating there was no severe collinearity. Moreover, when using the model for predictions, the multicollinearity is usually not considered a problem anyway (for more details, see Appendix 3).

Both the level and the spread of PTI values differed among the lake types Figure 39: the highest PTI values were found in lowland calcareous lakes, while the lowest values were from highland, siliceous lakes. This inherent variation in PTI among lake types, as well as the variation among individual lakes of within a lake type, are both accounted for in the hierarchical statistical model.

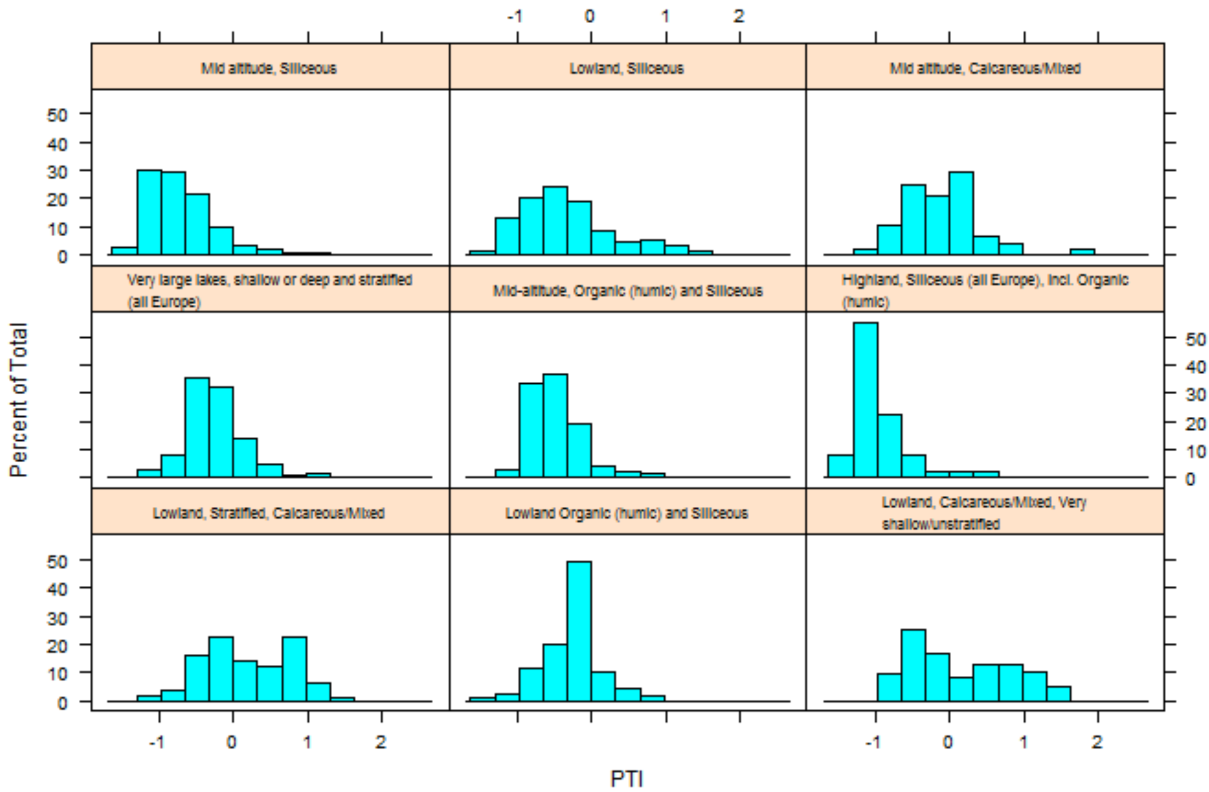

Figure 39. Histogram showing the distribution of PTI for each broad lake type.

Effect of individual stressors on PTI

All type-specific relationships between stressors and PTI are shown in Appendix 3. Below, we show only the type-specific effects of the stressor variables that were eventually selected for the final hierarchical model (Figure 40 - Figure 43).

Total P alone had a strong positive effect on PTI in general (all p<0.05), but the slope varied among the lake types (Figure 40). The steepest slopes were found for lowland lakes (Lowland siliceous and Lowland, Stratified, Calcareous/Mixed).

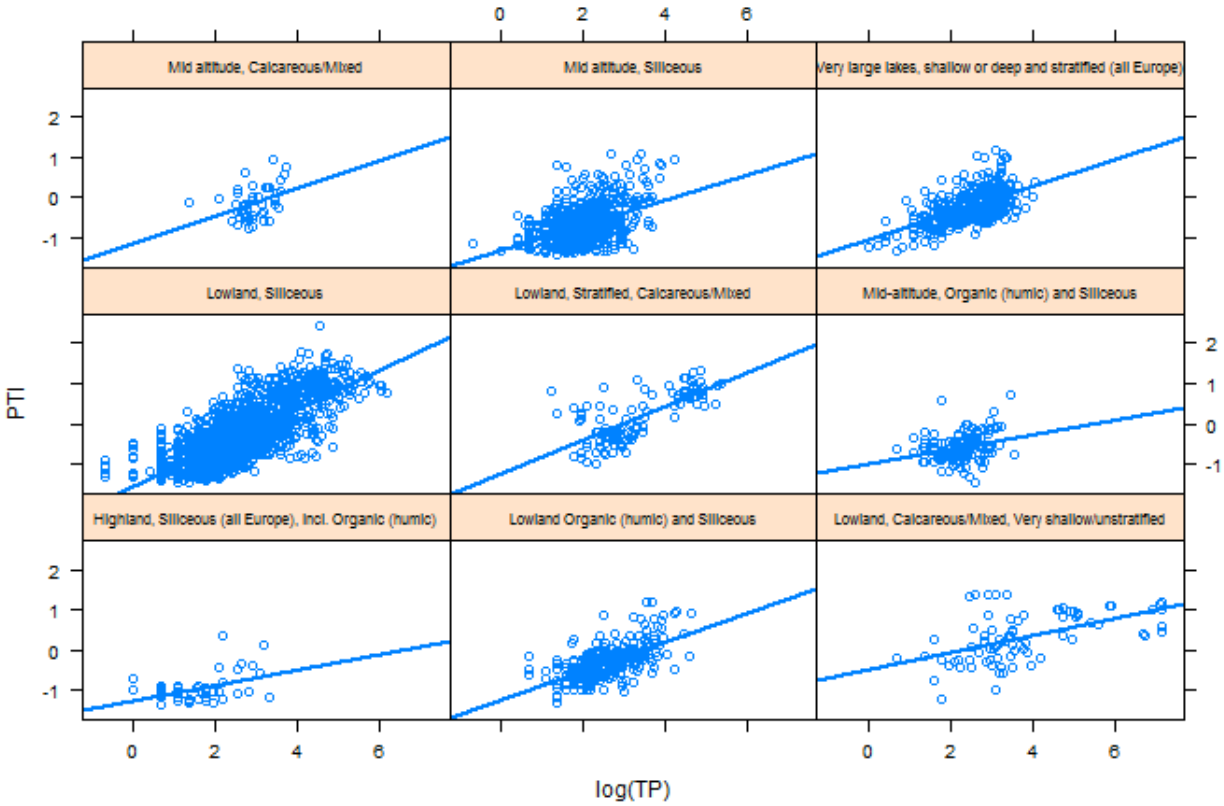

Figure 40. Type-specific relationships between total P (µg/l, ln-transformed) and PTI.

The ratio TN:TP alone apparently has a negative effect on PTI (Figure 41). However, this effect is probably an artefact caused by a stronger positive effect of TP than of TN. The effect of this predictor variable should therefore be considered only in combination with total P.

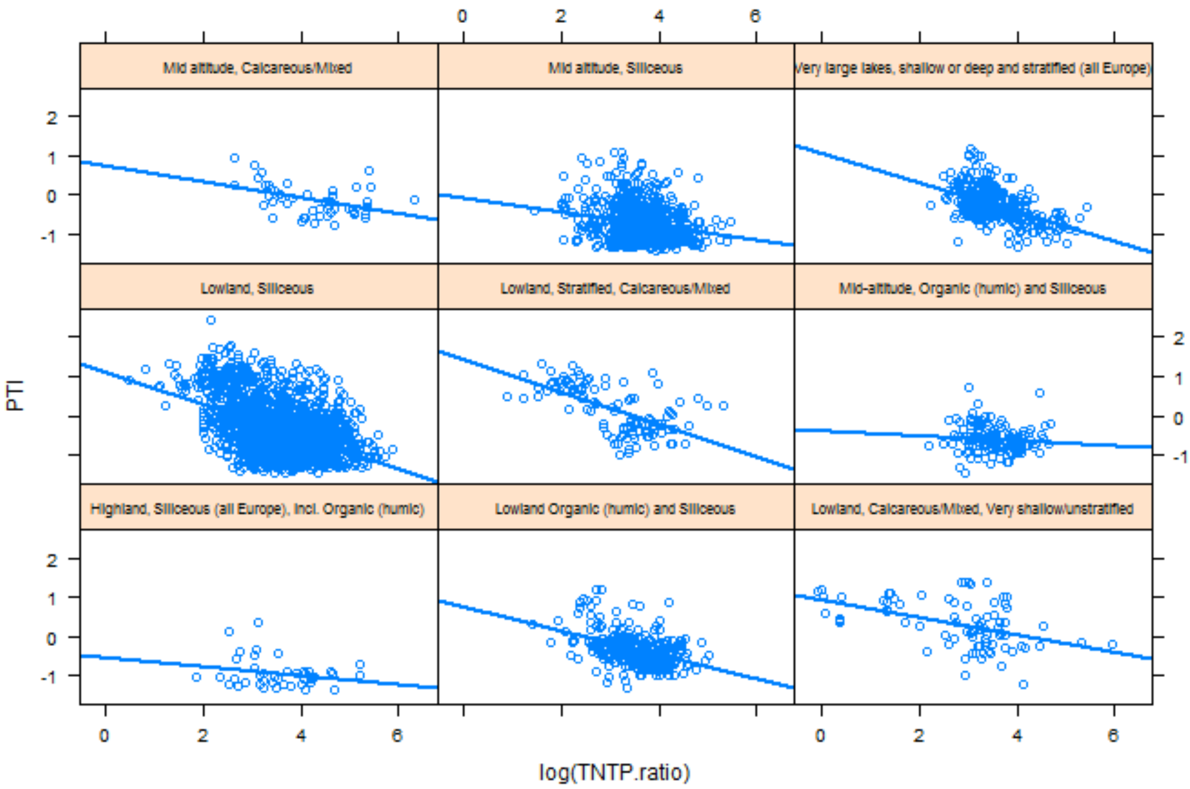

Figure 41. Type-specific relationships between TN:TP (mg:µg, ln-transformed) and PTI.

Summer air temperature was positively related to PTI for four of the lake types (Figure 42) lowland siliceous, lowland calcareous, mid-altitude siliceous and very large lakes. The lowland lakes had the steepest slopes.

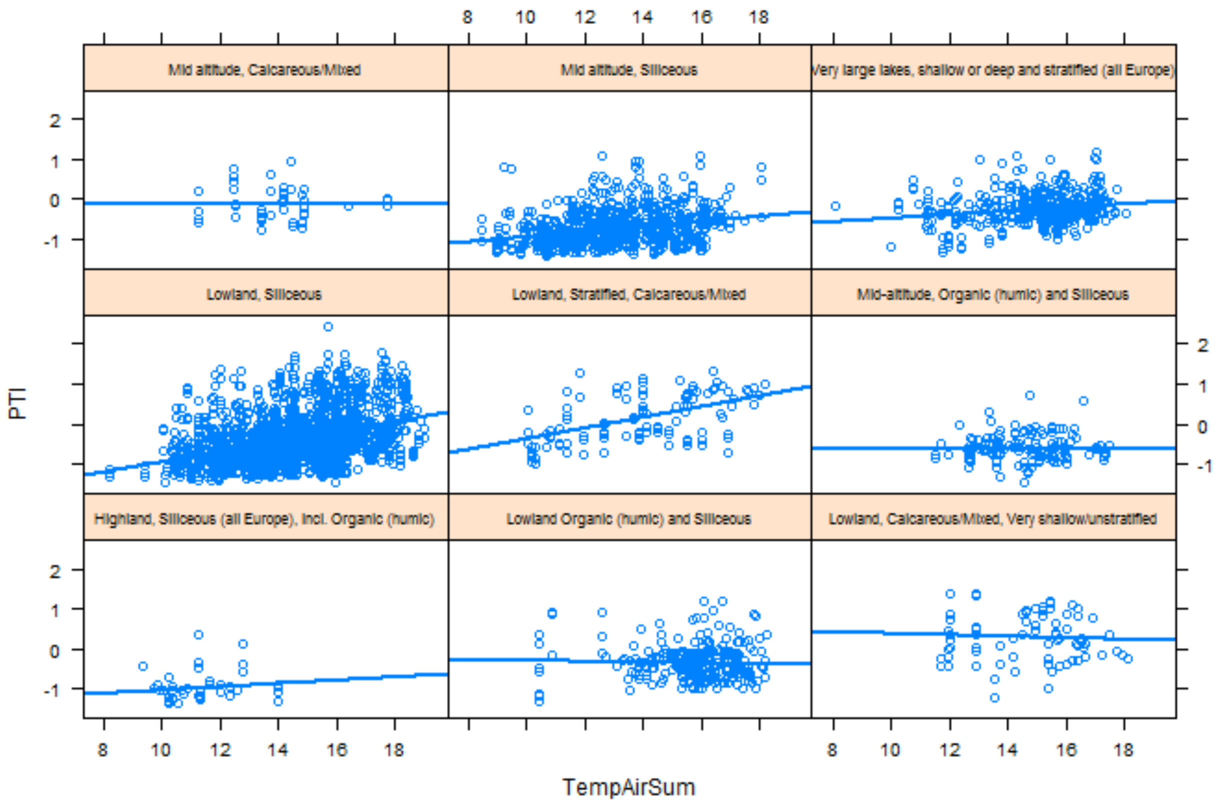

Figure 42. Type-specific relationships between mean summer air temperature (°C, June - August) and PTI.

The total summer precipitation had a positive relationship with PTI in four of the lake types (Figure 43): lowland siliceous, lowland calcareous, mid-altitude siliceous and mid-altitude calcareous. Thus, the PTI of large lakes were related to temperature, but not to precipitation. The opposite was the case for mid-altitude calcareous lakes (but here the number of samples was low and therefore the results were more uncertain). A negative relationship between precipitation and PTI was found for highland lakes only. A positive effect of precipitation on PTI in the low- and mid-altitude lakes may be related to increased nutrients loading from the catchments.

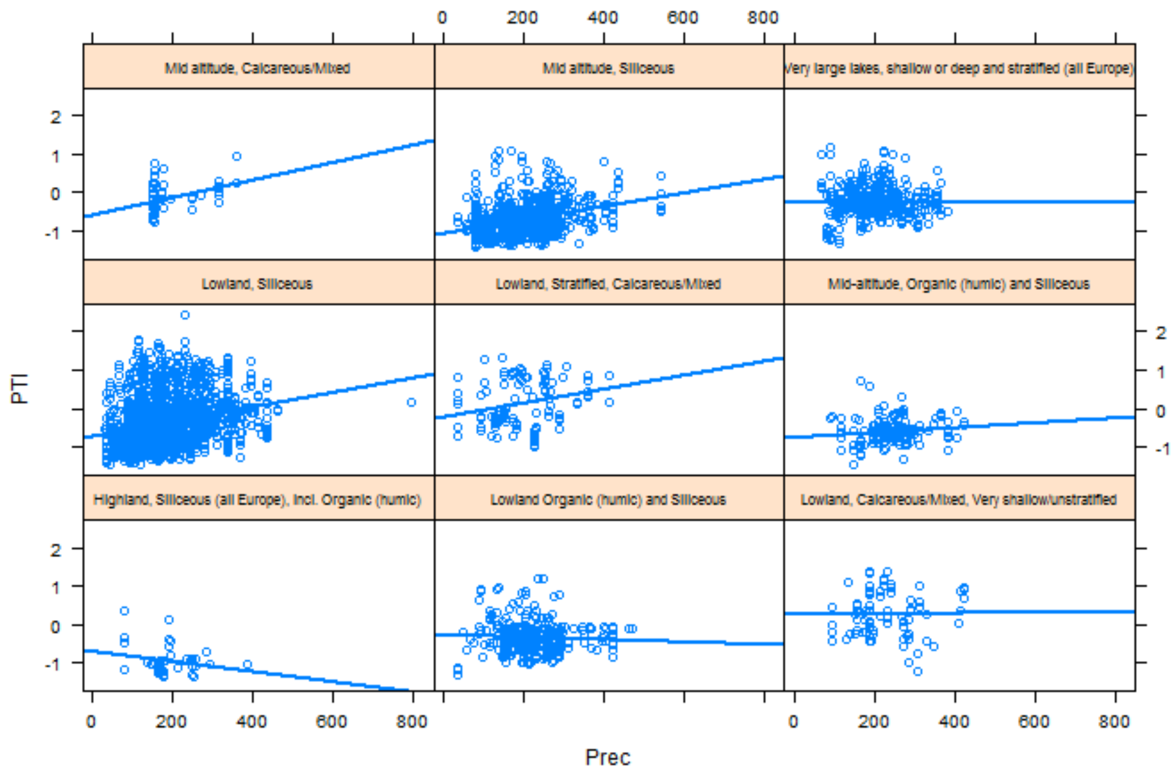

Figure 43. Type-specific relationships between total summer precipitation (mm, June - August) and PTI

Ranking of predictor variables and potential interactions

In this step the selected predictor variables and their interactions were ranked by their effect on PTI, by the Random Forest method. The analysis included the nutrients (TP, TN and TN:TP), the climatic variables (air temperature and precipitation) and a selection of the typology variables (altitude, mean depth and water colour). Surface area was excluded because of the high correlation with mean depth. The ranking of individual predictor variables by the (Figure 44) showed that total P was by far the most important stressor, followed by total N and the TN:TP ratio. Of the climatic variables, summer air temperature was more important than precipitation. The typology variables also got relatively high score, which supports our decision of including lake typology in the final regression model (next section).

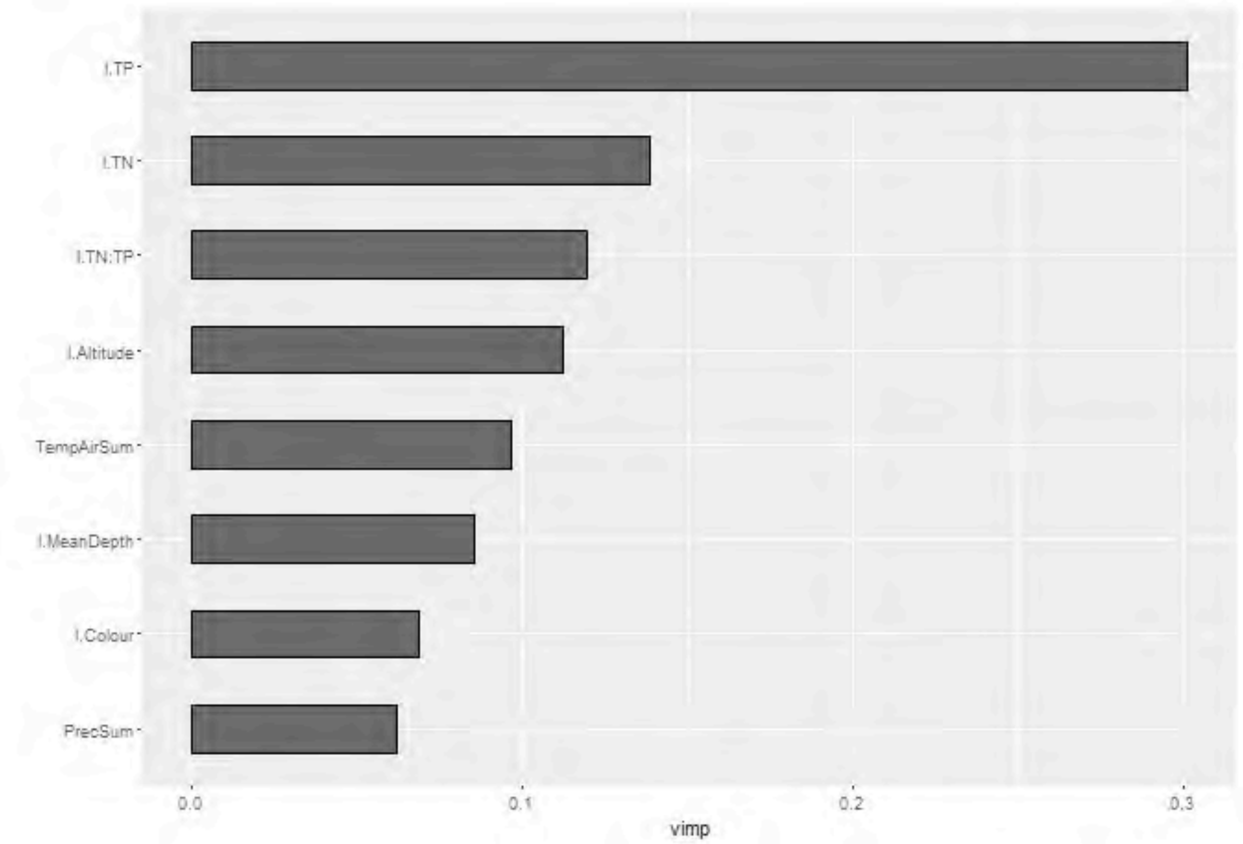

Figure 44. Ranked predictor variable importance (vimp), estimated by the random forest model.

The importance of the interactions between the selected predictor variables were also investigated by the random forest model. The importance of the interaction is indicated by the column Difference (Table 13), which gives the difference between the additive effect and the paired effect for each combination of variables A large positive or negative difference between 'Paired' and 'Additive' indicates an association worth pursuing if the univariate VIMP for each of the paired-variables is reasonably large (Ishwaran and Kogalur, 2015). The table shows that typology variables (altitude, mean depth and colour) are included in the top-ranked interactions. Furthermore, Total P had interactions with most of the typology variables, as well as with temperature.

Table 13. Ranked list of interactions between predictor variables for PTI. The column "Additive" shows the sum of the variable importance calculated separately (see Figure 44), while the column "Paired" shows their calculated paired variable importance. The "Difference" shows the difference between the additive effect and the combined effect. Only the top 10 interactions are shown.

Ranked No.	Variables	Additive	Paired	Difference
1	l.Altitude:TempAirSum	0.2082	0.2413	-0.0331
2	l.Altitude:l.MeanDepth	0.1976	0.2302	-0.0326
3	l.TP:l.Colour	0.3699	0.4011	-0.0312
4	l.TP:l.MeanDepth	0.3877	0.4174	-0.0297
5	l.TP:TempAirSum	0.3984	0.4255	-0.0271
6	l.TP:l.Altitude	0.4134	0.4402	-0.0268
7	l.TN:TP:l.Altitude	0.2309	0.2577	-0.0268
8	TempAirSum:l.MeanDepth	0.1821	0.2084	-0.0263
9	l.TP:PrecSum	0.3641	0.3896	-0.0255
10	l.Altitude:l.Colour	0.18	0.2053	-0.0253

The hierarchical PTI model

Hierarchical structure. As we have seen from the exploratory analysis, the response of PTI to stressor variables (nutrients and climate) vary among the different lake types (Figure 40 - Figure 43). These differences are due to natural lake characteristics such as altitude, mean depth, alkalinity and humic level. Our main interest is in the stressor-response relationship, that is, the response of PTI to nutrients and climate variables. Hence, the variables TP, TN:TP, air temperature and precipitation were fixed effects in our model. Furthermore, we did not want to estimate an effect of each lake type, but to account for the similarities and differences in responses among the lake types. We therefore included the lake types as groups.

The data structure had one more hierarchical level: for each lake, there could be more than one record (lake-month). This two-level hierarchy in the data was accounted for using linear mixed effects modelling (also called hierarchical modelling). Mixed effects models have both fixed effects and random effects in the model structure. Fixed effects represent the systematic, overall variation in PTI, and random (group-specific) effects show the random variation in PTI (covariance structure).

Selection of main variables and interactions. In the final model selection all models had PTI as the response variable and TP, TN:TP, temperature and precipitation as the stressor variables. We wanted to limit the number interaction terms to avoid making the model too complicated. The overview of interactions (Table 13) shows that typology variables were involved in potential interactions. However, since the effects of typology variables would largely be accounted for by the groups (random effects), these were not included as predictor variables in the final model. We therefore selected only the TP x temperature interaction (row no. 5) for the final hierarchical PTI model.

These predictors and their interactions were selected based on extensive preliminary analysis that included also other potential stressor variables and interactions. Also multiple different covariance structures were tested, including temporal effects. The possible temporal variation within lakes was dealt by including the lake as a random effect (intercept) in the model. More details can be found in Appendix 3. The final model was selected based on AIC values and model residuals as well as expert judgment. All effects of interest for predictive purposes were kept in the model even if their global effect was not significant, because we were interested in the type-specific effects.

The final model. The final selected model for PTI is a linear mixed-effects model with global (fixed) intercept and slopes, type-specific (random) intercepts and slopes, and lake-specific (random) intercepts:

PTI ~ TP + TN:TP + TempAirSum + PrecSum + TP:TempAirSum
+ (1 + TP + TN:TP + TempAirSum + PrecSum + TP:TempAirSum | lake type)
+ (1 | lake)

The estimated global (fixed) effects are shown in Table 14. The effect of TP is highly significant and positive, which means that an increase in TP concentration will increase the PTI level. Air temperature has also positive global effect. The global effects of TN:TP, precipitation and the interaction TP x temperature, but these variables are still of interest because they differ among lake types (Figure 45).

Table 14. Parameter estimates for the global (fixed effects) part of the hierarchical regression model for PTI.

Variable	Estimate	Std. Error	df	t value	Pr(>|t|)
(Intercept)	-0.359691	0.078486	7.896	-4.583	0.00186
TP	0.241111	0.036552	9.61	6.596	0.00007
TN:TP	0.052294	0.029378	6.224	1.78	0.12359
TempAirSum	0.039415	0.014871	5.223	2.65	0.04349
PrecSum	0.015334	0.014173	6.172	1.082	0.31974
TP:TempAirSum	0.009264	0.01353	4.247	0.685	0.52907

The overview of type-specific effects estimated by the hierarchical model is illustrated in Figure 45. These results of this model may differ from the analysis of the individual predictor variables (Figure 40 - Figure 43), since this model also accounts for correlations between predictor variables. First, we see that the intercept (PTI level) differ among lake types: it was significantly lower than average in most of the siliceous lake types (except for the lowland humic type) and significantly higher than average in the two lowland calcareous types. The positive effect of TP is most prominent in the lowland siliceous lakes, while the mid-altitude siliceous have a weaker TP effect than the average. Highland lakes and very large lakes also have weaker positive effects of TP (but not significantly different from the average).

The type-specific effects of the climate variables are generally opposite of the PTI levels: lake types with lowest PTI intercept have the strongest positive effect of the climate variables, and vice versa. This pattern is most clear for precipitation, where the lowland calcareous types have negative effect of precipitation while the mid-altitude siliceous and highland lakes have positive effect.

The positive effect of precipitation is strongest for the lowland and mid-altitude siliceous types, and weakest for the lowland calcareous types. The effects of precipitation on cyanobacteria, which generally contribute to higher PTI values in a lake, are analysed in more detail in Chapter 2.2.

The interaction between TP and temperature was most positive (gave higher PTI values) for the four siliceous lakes types, which tend to have lower PTI inherently. This implies that siliceous lakes are more sensitive to combined effects of TP and temperature, than the calcareous lakes. Only one of the siliceous lake types (the lowland humic) did not have higher interaction term then the average. This is consistent with the fact that humic conditions are not optimal for cyanobacteria, which tend to rise the PTI score. The very large lakes had lower TP x temperature effect than the average; these large lakes may be less sensitive to changes in air temperature due to their large volume.

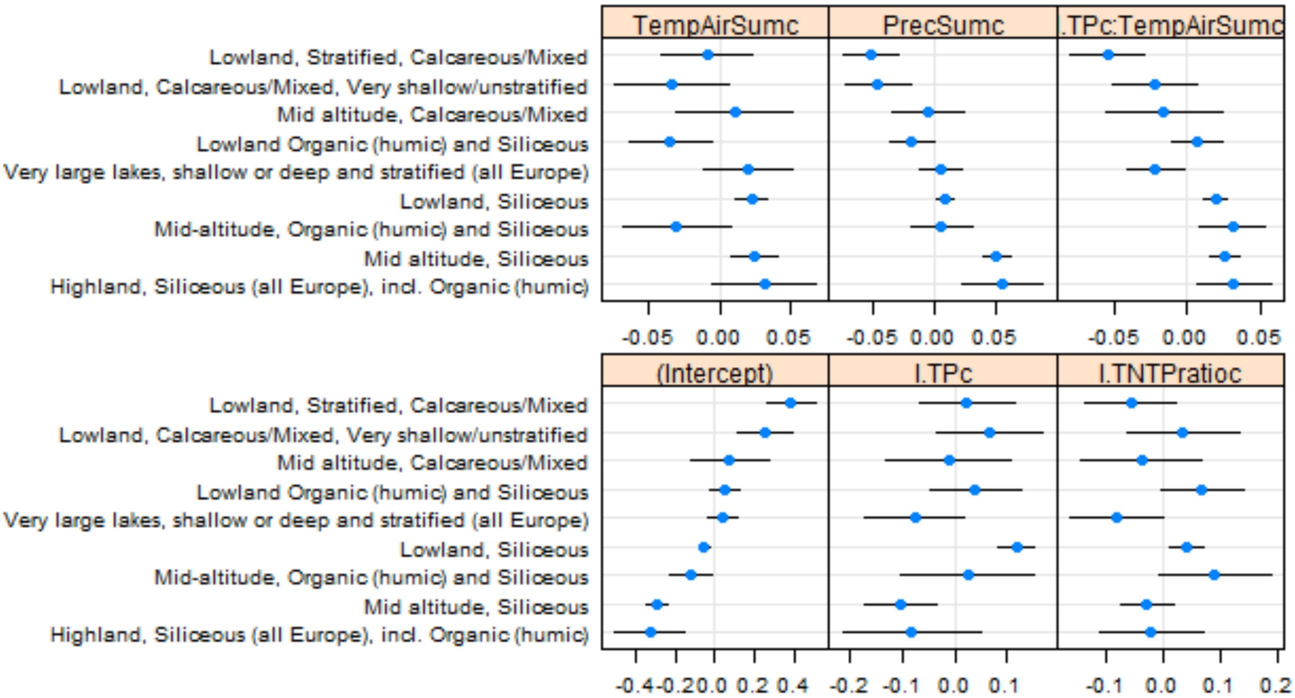

Figure 45. Parameter estimates for the random (type-specific) part of the hierarchical regression model for PTI. The blue dots indicated the deviation of each lake type from the global parameter estimate (given in Table 14). Vertical lines indicate the confidence intervals. A type-specific estimate different from zero means that type-specific effect is significantly different from the global effect. Note that the lake types are by increasing PTI intercept.

By using hierarchical modelling, we obtained more accurate estimates for the individual lake types compared to a single global model (cf. the blue dots vs. the vertical line in each plot of Figure 45). Moreover, the type-specific estimates were more precise (smaller confidence intervals) than if the data were analysed for each lake type separately, because of the higher number of observations. More details on the model performance and predictions are given in Appendix 3. Improved accuracy and precision are important for assessing the effects of combined stressors on indicators of lake status for different lake types, especially for the types where the number of observations is limited. This modelling approach also gives higher predictive power, which is particularly important when trying to predict the responses of lakes to future scenarios of stressor combinations (Chapter 2.5).

Key messages

- The combined effects of stressors (nutrients and temperature) on PTI varied considerably among the broad lake types.
- By using hierarchical modelling (type-specific stressor-response slopes and lake-specific PTI levels), we obtained more accurate estimates for the individual lake types compared to a single global model.
- The siliceous lake types were more sensitive (in terms of PTI) to an increase in total P.
- The siliceous lake types were also more sensitive to the interaction between total P and air temperature.

2.5. Projected effects of future climate on phytoplankton communities

Contributors: Jannicke Moe, Niina Kotamäki, Anne Lyche Solheim, Birger Skjelbred, Raoul-Marie Couture, James Sample.

Summary

The aim of this study was to predict the effects of nutrients (total P and N) in combination with current future climatic variables (air temperature and precipitation) on an index of the community composition of phytoplankton, the phytoplankton trophic index (PTI). We used an hierarchical regression model as a predictive model by replacing some of the predictor variables (air temperature, precipitation and/or TP) by projected future values for these variables. The projected future climate data were based on the MARS storylines, "Consensus world" and "Fragmented world", for the short-term future (year 2030) and long-term future (2090). Our predictions focused on the direct effect of increased temperature and/or precipitation on PTI, even under constant concentrations of TP. The predicted effects on PTI differed much between the two future scenarios, as well as for the different lake types. Under the best-case future climate scenario (Consensus World), PTI increased significantly in the long-term future (2090) for only three of the lake types, and was most significant for types 2 and 4 (lowland siliceous and lowland calcareous/mixed very shallow, respectively). In the worst-case climate scenario (Fragmented World), PTI increased significantly in the long-term future for all of the lake types except type 11 (Highland siliceous).

For the most common lake type in the dataset (type 2, lowland siliceous), we did a more thorough assessment of the combined effects of changed TP concentrations and future climate. According to our model, even if current TP concentrations would remain, warmer climate will increase PTI and may thereby reduce the ecological status of lakes in the long run (2090). If TP concentrations were 50% higher, temperature-induced increase in PTI could be expected also in the short run (2030). If TP concentrations were 50% lower (roughly corresponding to the WFD targets for TP), temperature-induced increase in PTI could still be expected in the worst-case CC scenario. However, temperature-induced change in PTI probably not sufficient to reduce ecological status class of lakes.

Introduction

The aim of this study was to assess the impact of nutrients in combination with climatic stressors on lakes in Northern Europe based on phytoplankton indicators, under future climatic conditions. More specifically, we wanted to predict the effects of nutrients (total P and N) in combination with current future climatic variables (air temperature and precipitation) on an index of the community composition of phytoplankton, the phytoplankton trophic index (PTI) (Phillips et al., 2013). For this purpose, we used a space-for-time approach in two steps. First we developed a hierarchical regression model for estimating the effects of all predictor variables on PTI for each broad lake type (Chapter 2.2). Secondly, we used this empirical model as a predictive model by replacing some of the predictor variables (air temperature, precipitation and/or TP) by projected future values for these variables. The projected future climate data are based on the two climate scenarios that are used in the MARS storylines, "Consensus world" and "Fragmented world" (MARS deliverable D.2.1-4).

Data

Lake data

The dataset used for estimation of the relationships between stressors and PTI is described in Chapter 2.2 (e.g. Table 12). In brief, the dataset contained almost 4000 phytoplankton samples from 940 lakes in the UK, Norway, Sweden and Finland, from the WISER central database (Moe et al., 2013). The PTI index was calculated at the genus level for each sample, and aggregated to lake level (in case of multiple stations within a lake) and to month (in case of multiple samples within a month). The predictor variables included in the final model were:

- total P and the ratio total N:total P from the same month as the phytoplankton samples (from the WISER dataset)
- mean air temperature and total precipitation for the summer months (June - August) from the same year as the phytoplankton samples (from JRC)
- broad lake types (ETC/ICM, 2015) (from the MARS geodatabase) .

Future climate scenarios and data

The generation of the future climate data by a set of five climate models (global circulation models) is described in MARS deliverable D.2.1-4 (Faneca Sanchez, 2015). We used the outcome from the climate model IPSL-CM5A-LR, which was found to best represent Northern Europe (Couture et al., 2016). Daily values for air temperature, precipitation and wind data were generated on a 0.5 x 0.5 degrees grid for all of Europe, for the two climate scenarios RCP (representative concentration pathway) 4.5 and 8.5. The MARS story lines "Consensus World" and "Fragmented World" are based on these two climate scenarios, respectively (in combination with socio-economic scenarios that are not considered here). Each lake in our dataset was spatially joined to a grid cell, and linked to the future climate time-series this grid cell. For prediction of PTI values we selected three periods: current (2006-2015), short-term future (2026-2035) and long-term future (2086-2095). In the following, each period is labelled by the year in the middle of this period (2010, 2030 and 2090).

Future nutrient scenarios

While "future climate data" were available in the project, corresponding future projected data for nutrient concentrations were not available. Future projected P and N loads (the pressure; Figure 1) might be available for large parts of Europe. However, nutrient loads data would first need to be linked to the in-lake concentrations (the abiotic state), which was beyond the scope of our study. Instead, we developed simple "what-if" scenarios for the nutrient concentrations. Focusing on TP as the main stressor, made three scenarios: (1) "Current TP" (100% TP, corresponding to the data used in the analysis); (2) "Low TP" (50% reduction compared to the current TP); and (3) "High TP" (50% increase compared to the current TP). For the most common broad lake type (type 2), a 50% reduction in TP corresponds roughly to the reduction that would be needed to obtain the management target (i.e., the High/Good status class boundary for TP) for lakes belonging to type 2 in our dataset (Phillips and Pitt, 2015).

Modelling

The predictive model corresponds to the hierarchical (mixed effects) regression model from (Chapter 2.2):

PTI = TP + TN:TP + Temperature + Precipitation + TP:Temperature
+ (1 + TP + TN:TP + Temperature + Precipitation + TP:Temperature | type)
+ (1 | lake)

In the equation above, the first line contains the global effects as well as an interaction between TP and Temperature. Here, each predictor variable has a common estimate for all lakes. The second line represents the type-specific effects for each predictor variable as well as a type-specific intercept and TP x temperature interaction. The third line represents the lake-specific intercept.

For predicting the effects of future climate on PTI, the observed temperature and precipitation were replaced by the future projected variables for the selected climate scenarios and periods. Likewise, for predicting the effects of TP scenarios on PTI, the observed TP values were replaced by the Low TP and High TP scenario values. All input variables were centered and scaled in the same way as the original predictor variables (see Appendix 3). The PTI values were obtained by the predict() function in R.

Results and Discussion

Climate

The projected future summer mean air temperature and precipitation was calculated for each individual lake and for each year (as illustrated in Figure 1). When averaged across the lakes, the future summer air temperature generally increased from the current period (2010) to the short-term future (2030) and to the long-term future (2060) (Table 15). The temperature increase was higher for the Fragmented World scenario than for the Consensus World scenario. Likewise, the predicted summer precipitation increased during both periods, but only slightly more in the Fragmented scenario.

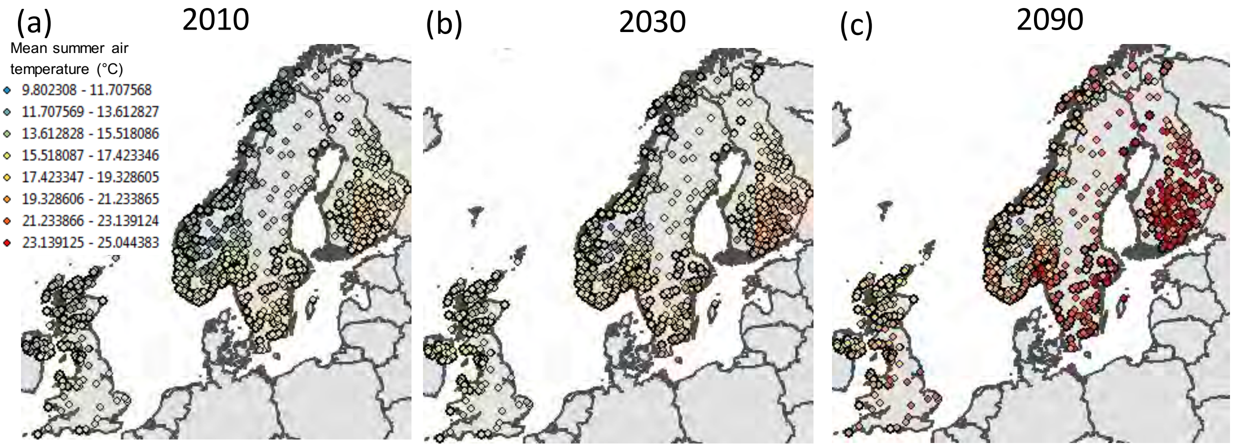

Figure 46. Projected future mean summer air temperature for individual lakes, for years 2010 (a), 2030 (b) and 2090 (c).

Table 15. Projected future air temperature (mean) and precipitation (sum) for the summer months (June - August). The values are averaged across all lakes, and the standard error represent the variation among lakes. Each year represent a decade (e.g. 2010 represent 2006-2015). Climate scenarios correspond to the MARS scenarios "Consensus world" (RCP 4.5) and "Fragmented world" (RCP 8.5).

Climate scenario	Year	Temperature, mean (°C)	Temperature, std. error	Precipitation, sum (mm)	Precipitation, std. error
Consensus World	2010	15.8	0.040	270	1.78
Consensus World	2030	17.1	0.036	286	1.83
Consensus World	2090	18.5	0.041	327	2.13
Fragmented World	2010	15.8	0.040	286	2.22
Fragmented World	2030	17.3	0.041	289	1.90
Fragmented World	2090	21.3	0.044	347	2.21

Phytoplankton Trophic Index

Effects of climate change (all lake types)

The changes in PTI under the future climate illustrated here (Figure 47) represent the direct effect of increased temperature and/or precipitation on PTI, and does not account for potential indirect effects of climate on e.g. nutrient loads, or interactions between climatic variables and nutrient concentrations.

Under the best-case future climate scenario (Consensus World, Figure 47 a), PTI increased significantly in the long-term future (2090) for only three of the lake types (types 1, 2 and 4, as well as the unknown types). There was no significant increase in the short-term future (2030). The increase was most significant for types 2 and 4 (lowland siliceous and lowland calcareous/mixed very shallow, respectively). In the worst-case climate scenario (Fragmented World, Figure 47 b), PTI increased significantly in the long-term future for all of the lake types except type 11 (Highland siliceous). The types with most significant long-term increase were 1 (very large), 2, 4, 5 (lowland organic siliceous) and 7 (mid-altitude siliceous). In addition, PTI increased significantly also in the short-term future (2030) for lake type 4.

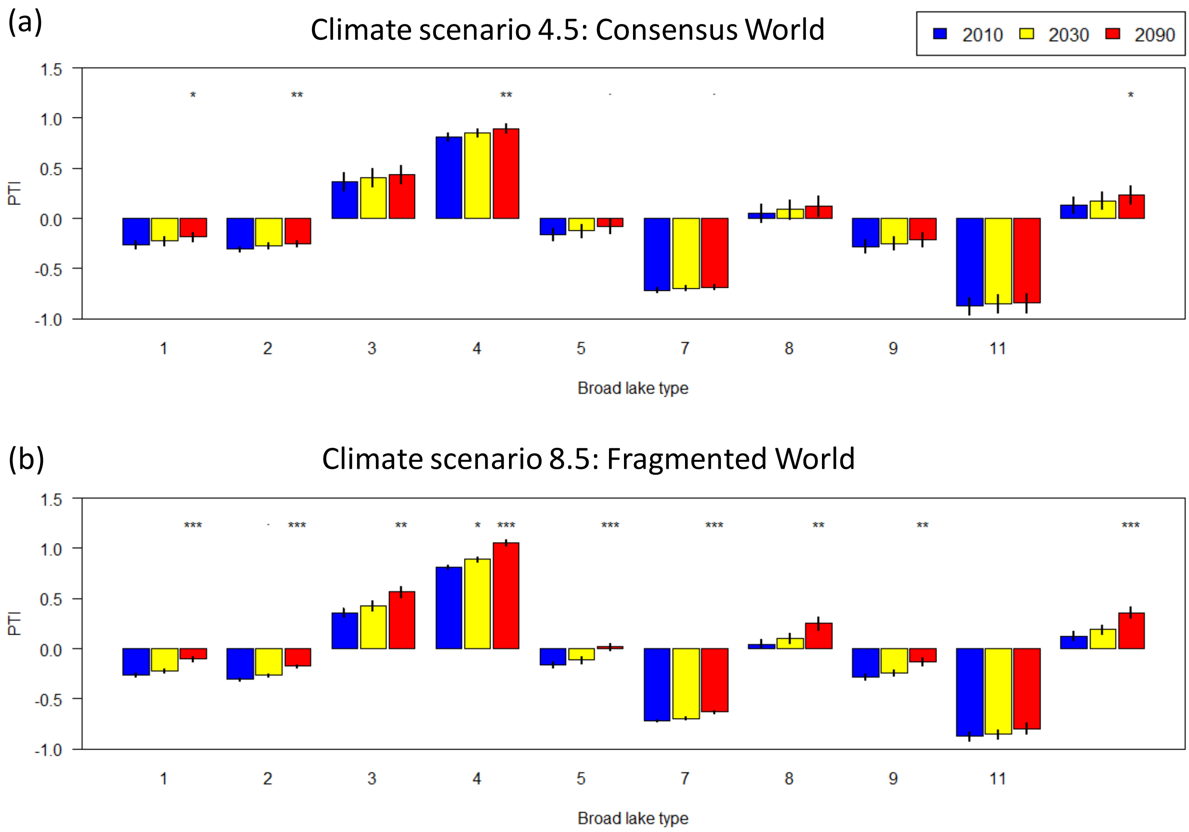

*Figure 47. Predicted PTI values for the three periods (2010, 2030 and 2090) and for each lake type, under the two climate scenarios Consensus World (a) and Fragmented World (b). Asterisks indicate significant increase in PTI between the years within each lake type (p values: *** < 0.005 < ** 0.01 < * < 0.05). The lake types are described in Table 1 (no label means unknown lake type).*

These result indicate that the phytoplankton communities the lowland lakes (types 2-5) are more likely to obtain change towards higher PTI (i.e. worse ecological status) than those in mid-altitude lakes (types 7-9), due to direct effects of higher temperature and/or more precipitation in the future.

Climate and nutrient scenarios (broad lake type 2)

The combined effect of future climate and nutrient stressors (TP scenarios) on PTI are illustrated in Figure 48. Here we have focused on lake type 2 (lowland siliceous), which is by far the most common lake type in our dataset (34% of the lakes and 51% of the samples). Moreover, this lake type showed a stronger response to all of the predictor variables than the average, including the TP x temperature interaction (Figure 45).

The current TP scenario ("100%") corresponds to the actual dataset (cf. type 2 in Figure 47). The low TP scenario ("-50%") represents 50% lower TP concentration compared to the observed data (applied to all three years), in combination with the climate scenarios for the three given years. Higher TP concentrations always result in higher PTI values (compare bars of the same colour in e.g. Figure 48 a and b; the difference is not analysed statistically), as expected. Within a given nutrient scenario, the difference between the bars represents the effect of climate chance alone. As described above, under the current TP scenario, climate change will result in increased PTI only in the long-term future (2090). For the low TP scenario, PTI will increase in the long-term future only under the worst-case climate scenario (Figure 48 d). Under the high TP scenario, however, PTI will increase already in the short-term future (2030) (Figure 48 b and e). This result is significant for both climate scenarios, but more significant in the Fragmented World scenario (Figure 48 e).

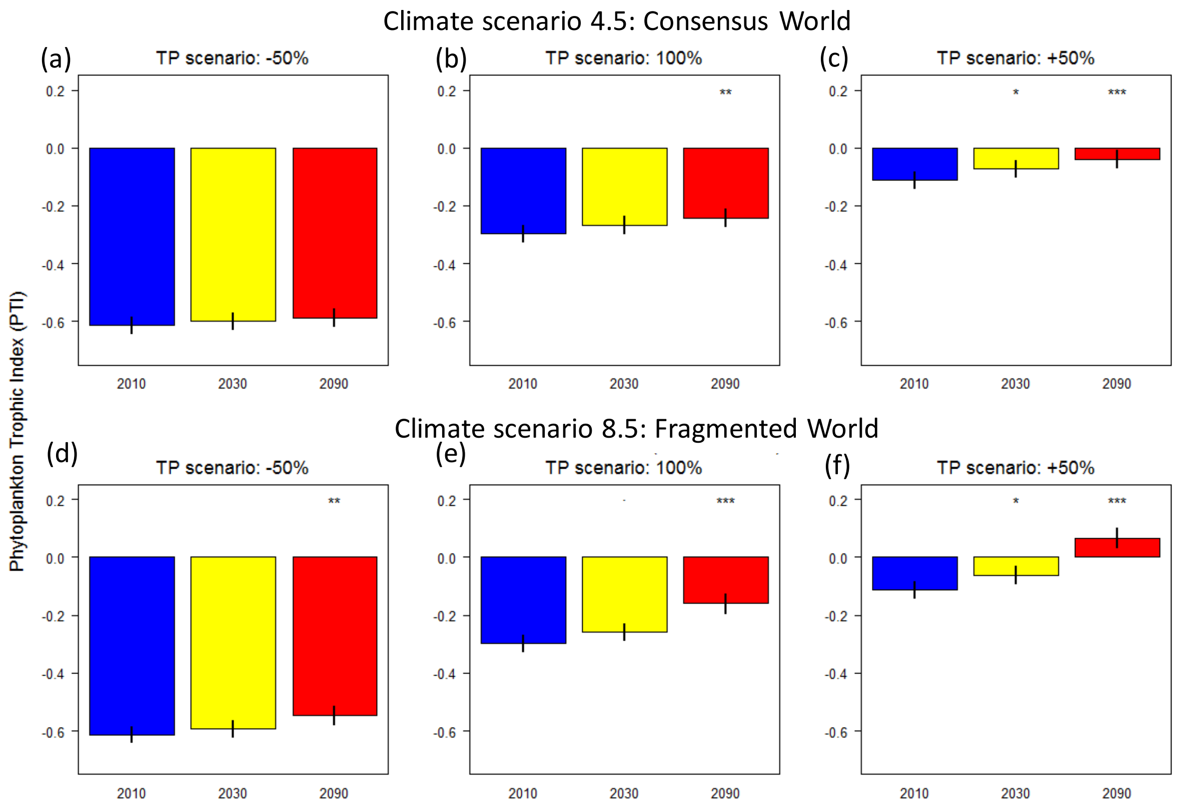

Figure 48. Predicted PTI values for lake type 2 (Lowland siliceous) the three periods (2010, 2030 and 2090), under the two climate scenarios Consensus World (a-c) and Fragmented World (d-f). The three vertical panels represent different scenarios of Total P concentrations: current TP (100%) (b, e); Low TP (-50%) (a, d); and High TP (+50%) (c,f).

Individual lakes (broad lake type 2)

So far we have presented the predicted change in PTI averaged for all lakes within a lake type. An analysis of the change of PTI for individual lakes will give more information on the risk of to a lower status class. The PTI value cannot be linked directly to ecological status (which depends on the national lake type as well as on other indices), but a larger increase in PTI is more likely to imply a drop in ecological status class. As a rule of thumb, we suggested that an increase in PTI by 0.5 units corresponds to one ecological status class (since the PTI scale ranges 2.5 units). An increase of PTI by more than 0.25 units is thus likely to imply a drop in ecological status class for this index.

In the analysis below (Figure 49), we only consider the direct effect of climate change on PTI under different nutrient scenarios (but not the effect of changes in nutrients *per se*). The plots illustrate the predicted change in PTI for each individual lake for the whole period 2010-2090. In the best-case climate scenario (Figure 49 a-c), increases in PTI caused by climate change alone are generally below 0.25 units, regardless of the TP scenario; this means climate change alone is not likely to cause worsened ecological status class. For the worst-case climate scenario, (Figure 49 d-f), in contrast, increases in PTI exceeding 0.25 units are more frequent: ranging from 5% in the low-TP scenario (Figure 49 d) to 23% in the high-TP scenario (Figure 49 f). According to these model predictions, we can expect that ca. 10-20% of lakes will have reduced ecological status indicated by PTI, due to long-term future climate change.

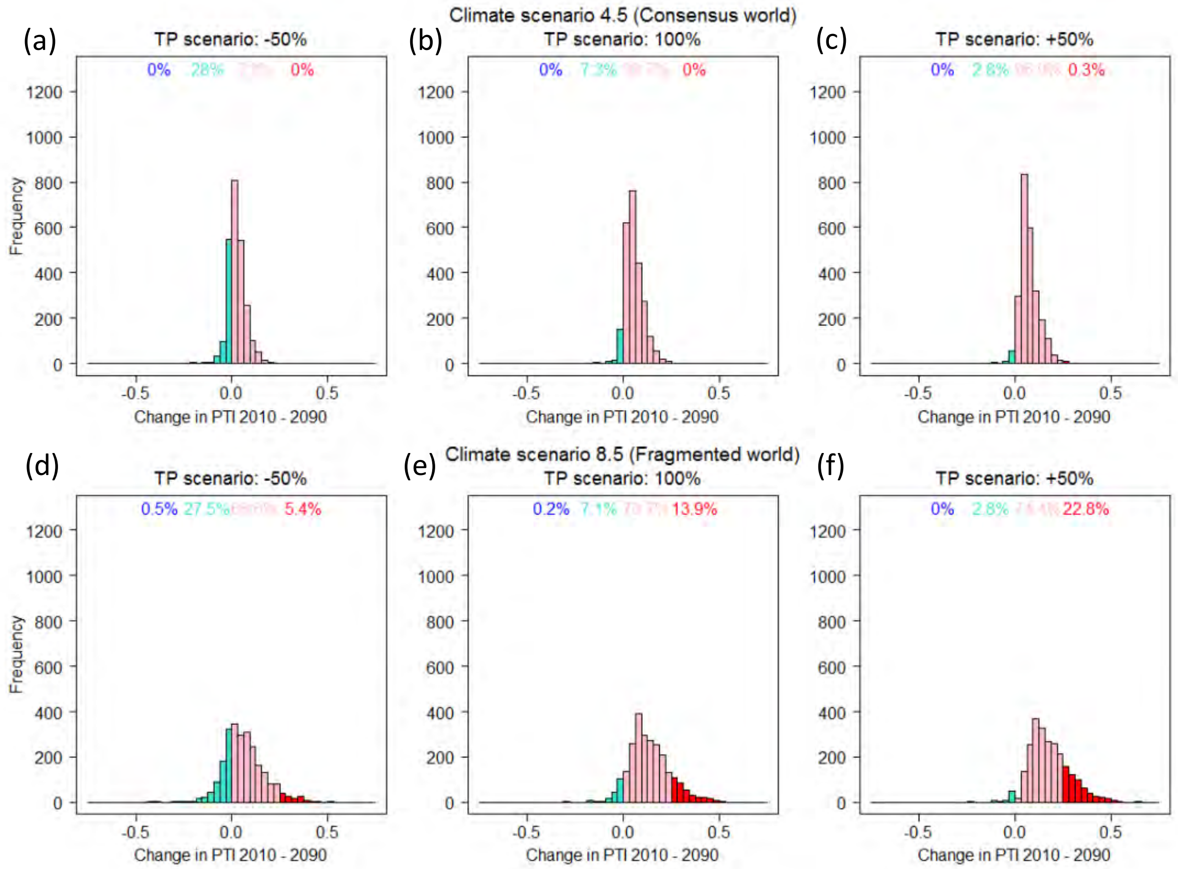

Figure 49. Predicted change in PTI values for individual lakes of lake type 2 (Lowland siliceous) the three periods (2010, 2030 and 2090), under the two climate scenarios Consensus World (a-c) and Fragmented World (d-f). The three vertical panels represent different scenarios of Total P concentrations: current TP concentrations (b, e); 50% reduction of TP (a, d); and 50% increase of TP (c,f). The numbers inside the plot indicate the percentage of lakes with a change of PTI within the given intervals: reduction by >0.25 units (dark blue), reduction by 0-0.25 units (turquoise), increase by 0-0.25 units (pink), and increase by >0.25 units (red).

Conclusion

Our assessment has demonstrated that large-scale data sets with high taxonomic resolution are valuable for assessing effects of multiple stressors on lake ecosystems. The large differences in responses for different lake types show that it is important to consider the lake type in analysis of multiple stressors and assessments of future risks. Considering the risk of future climate change, our assessment show that result may depend strongly on the selected climate scenario, therefore several climate scenarios should be included.

Our large-scale modelling approach depend on many assumptions, such as the space-for-time substitution and the linear relationship between predictor values and PTI. Such large-scale analysis should be supplemented with in-depth analyses of individual lakes, where the mechanisms behind empirical relationships can be better understood (cf. Chapter 2.3).

Our further work with this dataset will expand the assessment to other broad lake types and include a comparison of responses of PTI to future multiple stressor scenarios across the most common broad lake types.

Key messages

The key messages are based on the detailed analysis of broad lake type 2 (Lowland siliceous).

- Under the current levels of TP concentrations, warmer climate will increase PTI and may thereby contribute to reduced ecological status of lakes in the long run (2090).
- If TP concentrations were 50% higher, temperature-induced increase in PTI could be expected also in the short run (2030).
- If TP concentrations were 50% lower (roughly corresponding to the WFD targets of good status for TP), temperature-induced increase in PTI could still be expected in the worst-case CC scenario.
- Temperature-induced change in PTI are probably not sufficient to reduce ecological status lakes by a whole class, such as from Good to Moderate status.

3. Stressor combination 2: Nutrients and organic carbon

3.1. Effects on phytoplankton and food quality of fish (Finland)

Contributors: Marko Järvinen (contributors outside the MARS consortium are listed in Appendix 4).

Summary

The pressures eutrophication (increase of phosphorus, TP) and brownification (increase of dissolved organic carbon, DOC) can change phytoplankton community structure and decrease production of essential omega-3 fatty acids in lakes. This influence is transferred through the food web: perch growing in oligotrophic clear-water lakes contain 1.5-1.9 times more essential omega-3 fatty acids than those grown in eutrophic and brown-water lakes. In this regard, both eutrophication and brownification can contribute to impaired ecosystem services in lakes, in terms of the nutritional values of fish. The results were published by (Taipale et al., 2016).

Introduction

Fish in lakes provide an ecosystem service as high-quality food for humans, because of their high content of polyunsaturated fatty acids (FA). The most important and best known polyunsaturated omega-3 FAs are EPA (eicosapentaenoic acid) and DHA (docosahexaenoic acid). However, fish are dependent on algae producing the essential omega-3 fatty acids, which are then transferred in the food chain via zooplankton to fish. Not all algal groups are capable to synthetize EPA and DHA.

Potential responses of phytoplankton to eutrophication and to some degree to brownification of water have been described in other parts of this report (Chapter 2). Here, we report on a study of how the EPA and DHA content of phytoplankton changed along with eutrophication and brownification of lakes, and whether this change is detectable in a provisioning ecosystem service of lakes: the quality of a large piscivorous fish (European perch, *Perca fluviatilis*), a widely distributed species used for human consumption (Taipale et al., 2016).

The study was done by the research consortium TERLA funded by the Academy of Finland, with contributions from MARS (see Appendix 4).

Methods

Phytoplankton community data from 713 lakes in Finland (June - September 1985-2011, 2547 quantitative phytoplankton samples, collected by SYKE) were combined with the information of fatty acid content of 40 different algal taxa that were cultured in the laboratory of University of Jyväskylä (Figure 50). The studied lakes represented oligo-mesotrophic, eutrophic and dystrophic lakes (TP 3-180 µg l^{-1}, DOC 4-31 mg C l^{-1}). European perch (*Perca fluviatilis*) were collected from 14 lakes with different TP and DOC by researches from Universities of Joensuu, Helsinki and Jyväskylä.

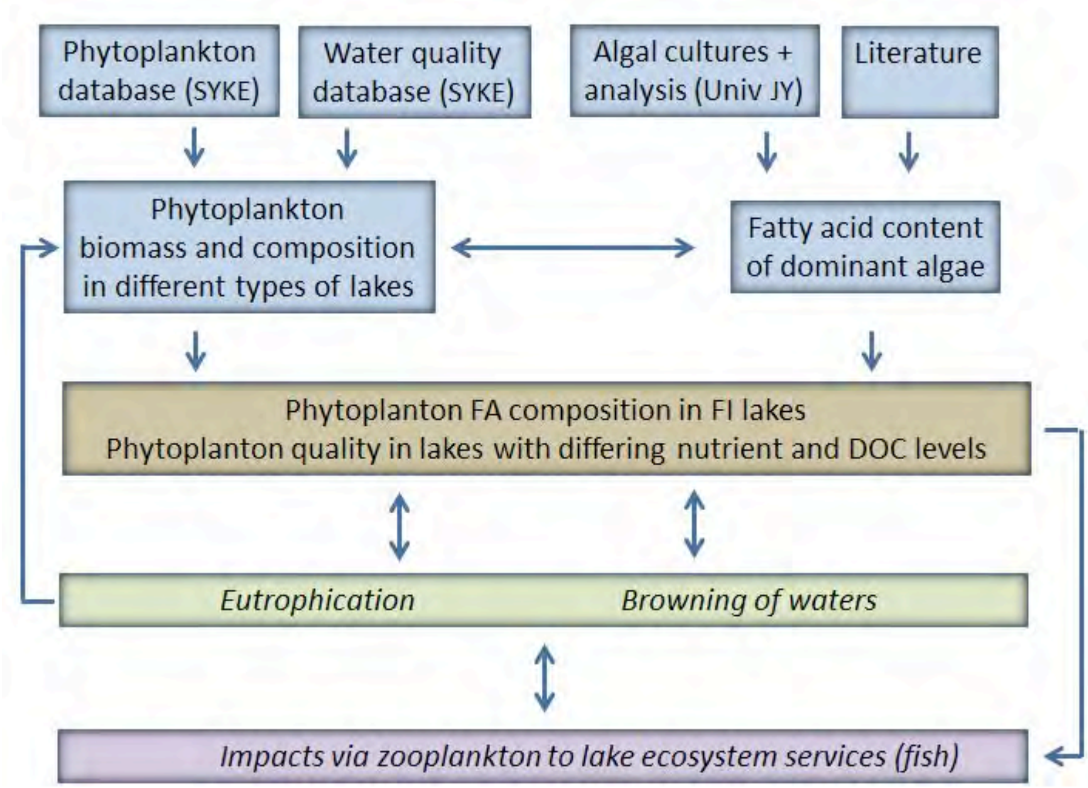

Figure 50. Schematic description of information sources and the work flow.

Results and Discussion

The results (Taipale et al., 2016) show that eutrophication and brownification change lake phytoplankton community composition. The proportion of EPA and DHA producing algae tends to decline, thus leading to decreased production of essential omega-3 fatty acids in lakes. Especially cyanobacteria and green algae, which are poor in omega-3 fatty acids, benefit from eutrophication. Cryptophytes, diatoms, chrysophytes and dinophytes are the most important algal groups producing these essential fatty acids. These algal groups predominate phytoplankton communities of oligotrophic clear-water lakes. The influence is transferred through the food web and impairs the ecosystem services of eutrophic and humic lakes, because perch growing in oligotrophic clear-water lakes contain 1.5-1.9 times more essential omega-3 fatty acids than those grown in eutrophic and brown-water lakes (Figure 51). This was the first study to demonistrate that the fatty acid content of phytoplankton impacts the food chain up to predatory fish, and thus an ecosystem service.

TP and DOC concentration influence also zooplankton and fish biomasses and community structure. In boreal eutrophic lakes the percid fish biomass can be 1.9-fold greater than that in oligo-mesotrophic lakes (see (Olin et al., 2002)), but the EPA and DHA content in individual perch is highest in the oligo-mesotrophic lakes. Our study suggests that a person should eat between 1.5-1.9 times more perch from eutrophic and dystrophic lakes, respectively, compared with those from oligotrophic lakes, to achieve the daily recommended intake of EPA and DHA.

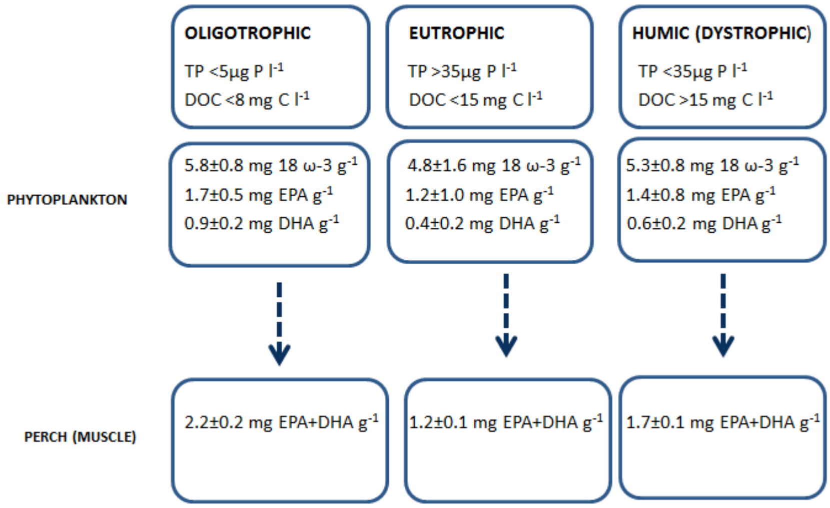

Figure 51. Schematic approximation of the routes of ω-3 fatty acids across lake food webs in oligotrophic (incl. mesotrophic), eutrophic and dystrophic lakes (modified from Taipale et al. 2016). TP and DOC influence phytoplankton biomass and composition. Algal biomass can be 5-fold higher in eutrophic lakes than in oligo-mesotrophic or dystrophic lakes, but due to the high contribution of non-EPA and non-DHA synthesizing taxa, the phytoplankton 18ω-3 FA (ALA+SDA), EPA and DHA content is lower in eutrophic lakes.

Key messages

Eutrophication is still a great problem in lakes in Europe. During the last decades, brownification of lakes has been detected in the whole northern hemisphere. EPA and DHA content of phytoplankton decreases along with eutrophication and brownification of the lakes. Accordingly, the EPA and DHA content decreased in the muscle of large piscivorous perch. Thus, eutrophication and brownification pressures significantly worsen the quality of perch used in human diets, and thus ecosystem services.

4. Stressor combination 3: Nutrients and water level changes

4.1. Effects on macrophytes: background and conceptual model (Europe)

Contributors: Seppo Hellsten (SYKE), Matthew O'Hare (CEH), Marit Mjelde (NIVA)

Introduction

Lacustrine macrophyte communities are altered by anthropogenic pressures such as eutrophication caused by point source and diffuse pollution (Figure 52). Macrophytes are limited by reduced light climate and sedimentation, but also due to different pathways to utilize macronutrients and carbon. Changes in hydrological conditions cause fluctuating water levels which also has significant effect on macrophyte stands (Figure 52). The Water Framework Directive describes hydromorphological quality elements of lakes as hydrological regime and morphological conditions. High hydromorphological status refers totally or nearly totally undisturbed conditions, but in good and moderate status the values are only consistent with the achievement of the values specified above for the biological quality elements. Hydrological regimes of lakes are disturbed by human activities related to their function as water stores for hydropower generation and water supply, general water regulation for flood defence and navigation activities and also in some cases for recreational use (Wantzen et al., 2008). Hydropower effects are typical in northern and high altitude lakes, usually without any other pressures, whereas regulation for navigation and recreation purposes are often seen in lowlands lakes situated in densely populated areas. Additionally, also lakes and especially reservoirs are regulated for drinking water and irrigation purposes.

Morphological alterations, dams and weirs for example, can also effect continuity of rivers situated downstream from regulated lakes. Flood protection works and the drainage of floodplains that produced embankments in particular can significantly change morphology of lowland lakes. Such lakes, which are often artificial or created by damming of coastal shallow basins, are common in some regions, in the Netherlands for example. Generally large scale morphological alterations are more common in small lakes surrounded by agricultural areas and population centres.

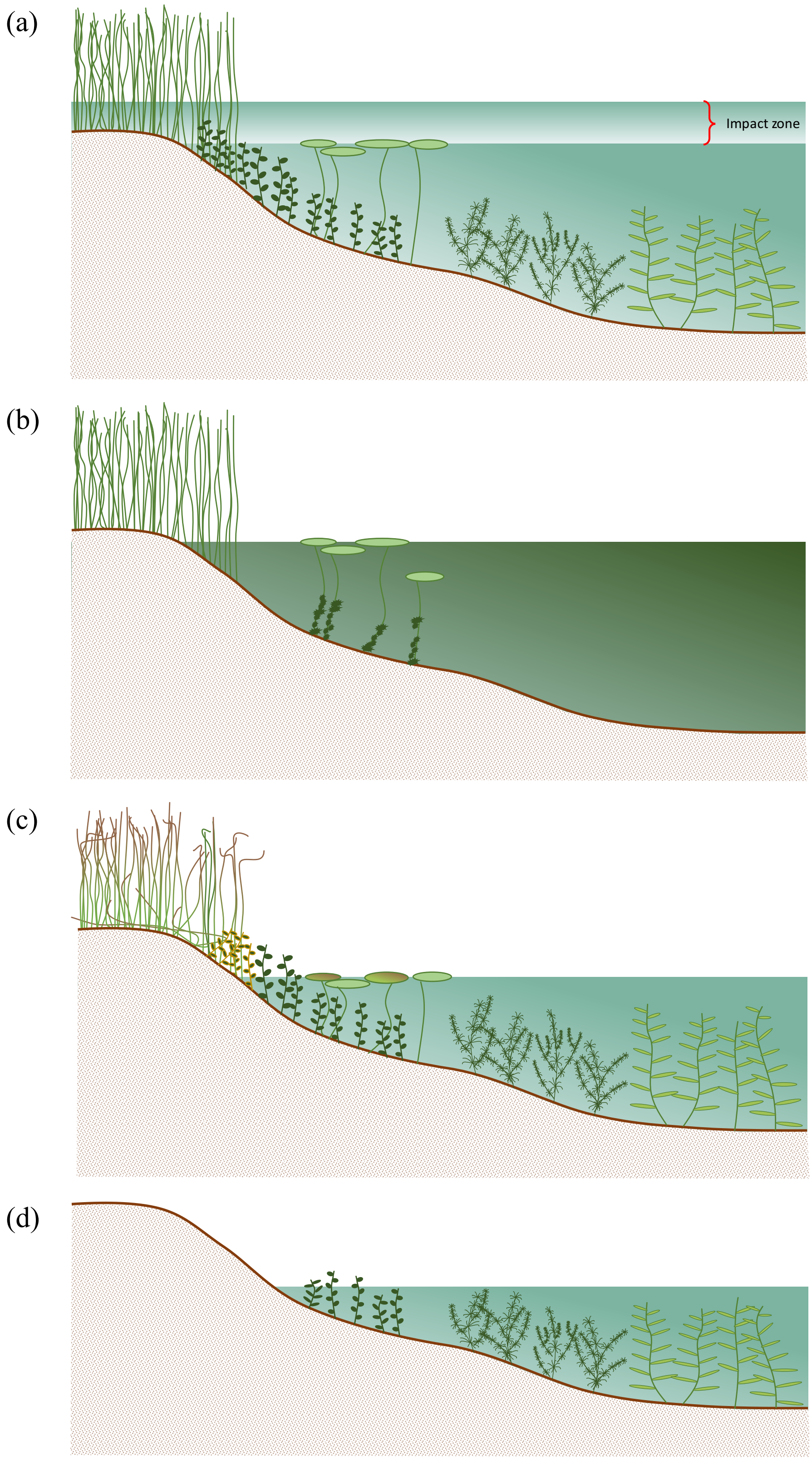

Figure 52. (a) Healthy lake with impact zone illustrated. (b) Eutrophication. (c) Moderate water level impacts. (d) Significant water level impacts.

Morphological and hydrological changes mainly affect the uppermost littoral zone although changes in retention time can indirectly also change the trophic status of whole lakes too. Also the general lowering of water level can significantly affect resuspension of bottom sediment. As it is the littoral zone which is most heavily impacted by physical alterations it is the organisms that reside in that zone which are the quality elements which are most sensitive to those alterations. Littoral zone macrophytes, are one of the key indicators of the hydro-morphological changes, such as water level fluctuations of lakes. Macrophytes exhibit a vertical zonation pattern in the littoral; helophytes grow in the uppermost zone whereas isoetids, elodeids and charids occupy deeper water. This general pattern is present in lakes throughout Europe. Even small changes in dynamics of water level fluctuation can affect the distribution and elevation of zones (Keto et al., in prep.). Morphological changes to the littoral caused for example by dredging or embankments are deeply disturbing vegetation development.

Hydromorphological pressures interact with other significant pressures on lake macrophytes. Two key stressor interactions are described here, for eutrophication, which is a long standing issue, and brownification which is a new emerging issue linked to climate change. The combination of eutrophication and hydromorphological pressures is typically present in lowlands lakes and reservoirs under intensive use. The combined impact of the two stressors can be both independent of one another and they can also interact; hydromorphological changes can alter eutrophication processes for example changes in residence time can rapidly affect the nutrient status of a water body. Macrophytes occupying the littoral zone are sensitive to changes in light climate. Therefore, changes in depth affect resuspension of bottom sediment and the balance between different macrophyte species causing harmful impacts for the ecosystem. Changes in retention time may affect, via general water quality, phytoplankton growth and macrophytes. The combination of hydromorphological and eutrophication pressures are, in principle, easy to handle by manipulating water level and retention time. In practice it can be difficult to implement where the use of shore areas is intensive.

Nutrient enrichment may also compensate for degradation caused by fluctuating water level, because it can compensate the loss of fine grained nutrient rich sediment. On the other hand, highly modified barren littoral is often more sensitive to resist eutrophication compared to vegetated well-developed littoral zone. Brownification of lakes means an increase in the brown substances of lake and stream water, caused mainly by dissolved humic matter of terrestrial and wetland origin, which absorbs solar radiation strongly in the short wavelength part of the visible spectrum (Evans et al., 2005). Increase of humic substances mean also diminished growing zone of aquatic macrophytes due to lack of light.

In short, a complex picture is emerging where hydromorphological pressures can potentially impact macrophytes in a variety of ways and there are a number of mechanism by which they could in theory interact with two other common pressures, eutrophication and brownification. Below we review the evidence that water level fluctuations, the most straight-forward hydromorphological pressure, are having an impact and describe the conditions which influence that impact. We introduce a conceptual model which aims to encompass the key interactions between stressors. The aim is that the model is a useful starting point for both managers and researchers to tackle standing waters under multiple stress.

Review of effects of water level fluctuation in lacustrine ecosystem

Rørslett (1988) defined hydrolake as a water body where the water levels are operated for generating hydro-electric power (HEP). He also suggested a classification of the hydrolakes and natural lakes dividing lakes to five groups. Hydrolakes were divided to oscillating ones (H1) with very short residence time high wintertime water level, intermediate reservoirs with short residence time, small water level fluctuation (2-4 m) and high winter level (H2) and storage reservoirs (H3) with a long residence time, significant water level fluctuation (more than 4 metres) and clear winter drawdown. Further he divided natural lakes to river-run (N1) with short and to others (N2) with long residence times.

In general water level fluctuation leads to a decrease in macrophyte diversity. Rørslett (1989) showed in his analysis of 17 Norwegian hydropower lakes that the species richness (S) followed equation:

$$S = 16.4 - 1.34\ \Delta W - 0.013\ H + 0.085\ A,$$

where ΔW = mean annual range of water level (m), H = lake altitude (m a.s.l.) and A = lake area (km^2).

Further in his analysis of 641 lakes from Norway, Sweden, Denmark and Finland, he found that lake area was the best predictor of species diversity (Rørslett, 1991), which is linked to available habitat diversity. A stepwise prediction model also included hydromorphology related variables such as water level range and lake lowering with water conductivity and lake elevation values.

Rørslett (1989) found lower diversity of macrophytes in Norwegian lakes with extended water level fluctuation. Hellsten (Hellsten, 2001) showed similar trend in Finnish regulated lakes. Hill et al. (Hill et al., 1998) demonstrated lowered diversity in lakes with fluctuating water level in Canada. Nilsson et al. (Nilsson et al., 1997) found that biodiversity was much lower in Swedish river reservoirs compared to free flowing sites. This relationship, however, is not linear. Extensive surveys of Scandinavian lakes showed that general biodiversity correlated mainly with draw-down of water level, but regulation amplitude between 1 and 3 m supported the highest biological diversity (Rørslett, 1991). In the Netherlands too, the natural water level fluctuation during the year before regulations was in general approximately 1 meter and supported high biodiversity. Today with the water level fluctuations strictly managed, absent or 'reversed' (higher levels in summer than in winter), the biodiversity of especially the helophyte community has decrease as a result of that change (Geest et al., 2005).

A slight increase in disturbance could even create suitable habitats for aquatic macrophytes as noted by Murphy et al. (Murphy et al., 1990). Similar phenomenon was found in hydrolakes of

New Zealand, where increasing monthly water level fluctuation range even increased biodiversity (Riis and Hawes, 2002).

Water level fluctuation is also reported to have a strong influence on vegetation succession. Disturbances caused by drawdown events may prevent competitive exclusion of desiccation-sensitive species, or may stimulate germination of species (Bonis and Grillas, 2002). Amphibious species show a clear association with alternating exposed and reflooded sites (Partridge, 2001). Temporary lake drawdown creates a window of opportunity for establishment of species that produce large numbers of propagules (Bonis and Grillas, 2002). Early successional species have easily dispersed propagules, such as oospores, plant fragments and turions (Haag, 1983). The increased emergence of *Chara* as a result of drawdown may partly explain the cyclic pattern of abundance in floodplains and dominance of benthic filamentous macro-algae, were positively related to the proportion of drawdown area in the lakes (Geest et al., 2005). Late-successional species often possess heavy or large propagules that are transported over relatively short distances, making them poor colonizer; e.g. for *Nuphar lutea* several years without drawdown are needed to allow establishment in lakes, due to the high vulnerability of both seeds and juvenile plants to desiccation (Smits et al., 1989).

Depth variations are usually related to an artificial increase or decrease of water level. Water levels are increased to extend storage capacity of reservoirs or regulated lakes. A sudden increase of water level will initiate erosion processes, which lower biodiversity (Hellsten et al., 1996; Nilsson, 1981). It should be noted that taxonomic composition is a poor indicator of water level increase, because most of the species are still present after water level increase, although abundance may differ significantly (Hellsten et al., 1996; Nilsson and Keddy, 1988). Effects of raised water level also depend on ageing; after inundation shock of Swedish reservoirs species diversity was highest 30-40 years subsequent to the initiation of the regulation (Nilsson et al., 1997). In most cases diversity is slightly increased after inundation due to stabilization of the shoreline.

In general, lowering of water level will lead to increased diversity, as found in several studies (Lohammar, 1949; Rørslett, 1991; Toivonen and Nybom, 1989). The main reason for increased diversity is that a newly exposed littoral zone or general shallowness allows the sublittoral zone to cover the entire water body. Several shallow water lake studies have demonstrated a sensitive balance between different species groups (Best, 1987; Van den Berg et al., 1998).

Several studies indicate that abundance is a much more sensitive indicator for hydrological change than species composition (Coops et al., 1999; Hellsten et al., 1996; Hellsten, 2001; Nilsson and Keddy, 1988). Generally, water level fluctuation affects zonation patterns, which are a function of the relative abundances of different species with different degrees of adaptation to stress caused by depth and drying. Therefore, changes in the amount of water level fluctuation are reflected by changes in distribution of species. As this is mainly due to the bathymetry of a lake, this is a lake-specific reaction and with small variations in water level large areas may come available in lakes where the slope of the bottom is small. Full lake surveys are necessary to monitor these changes accurately.

In addition to the range of water level fluctuation, the dynamics of the fluctuation significantly affects the abundance of macrophytes. For example, the timing and range of the spring flood clearly affects the zonation of sedge species in northern areas (Hellsten, 2001; Sjöberg and Danell, 1983; Walker and Wehrhahn, 1971). The generally observed increase of common reed (*Phragmites australis*) abundance in Scandinavia may be related to lowered early spring water level (Partanen and Hellsten, 2005; Partanen et al., 2006; Rintanen, 1996). Reeds also benefit from stabilized water levels and growth periods (Coops et al., 1999; Coops and Van Der Velde, 1995). A general decline of reed beds in Middle-Europe seems, however, to be related to changes in sediment due to eutrophication (Clevering, 1999; Weisner, 1991). Lowering of the water level while a lake is ice-covered will have significant effects, especially on large sized isoetids such as *Isoetes lacustris* and *Lobelia dortmanna*. Reports of their decline cover northern Scandinavia (Hellsten, 2002; Quennerstedt, 1958; Rintanen, 1996; Rørslett, 1984) and Scotland (Murphy et al., 1990; Smith et al., 1987). Additional to the effect of freezing, changes in sediment quality will also significantly affect their distribution (Murphy, 2002).

Apart from this new development, there are few classification schemes related to relationships between seasonally distributed hydro-morphological factors and macrophytes. The direct response of *Isoetes lacustris* to ice penetration enables its distribution to be used for classification purposes (Hellsten, 2001; Rørslett, 1989; Rørslett and Johansen, 1996). The deepest growing areas of *I. lacustris* are also sharply limited by lack of light and therefore their growing niche is easy to predict (Rørslett, 1988). The distribution of other large isoetids such as *Isoetes echinospora, Lobelia dortmanna* and *Littorella uniflora* can also be used for classification purposes, because they are all relatively weak against ice erosion and changes in sediment structure (Murphy, 2002; Rørslett, 1989).

A classification based on strategy analysis and the division species into Stress-tolerating, Ruderal and Competitive categories has been effective in classifying regulated lakes in Norway (Rørslett, 1989) and in other areas (Murphy et al., 1990). Ruderal species, with high resistance to disturbance, were typical in shallow water communities of regulated lakes, whereas Stress-tolerating species prevailed in deeper areas.

The effects of depth changes have been generally used in simple calculation procedures to describe the available growth area for macrophytes. Known relationships between deepest growth limits of bottom-rooted helophytes have produced a large number of different applications for Finnish lakes (Hellsten, 2002). Hudon (Hudon, 1997) developed similar relationships between average water level scenarios and areas dominated by different vegetation types in floodplain lakes of St Lawrence River.

Conceptual model of macrophyte response on multiple pressures

The response of aquatic macrophytes to hydrological alteration can be described by a conceptual model (Figure 53), consistent with the DPSIR scheme of the MARS conceptual model (Figure 1). Macrophyte species composition is driven by distributional factors which are further sorted by water quality especially related to alkalinity and main nutrients. Further water quality effects on attenuation of light via increased production and humic compounds; especially latter has increased due to brownification phenomenon observed all over northern hemisphere. Submerged elodeids and isoetids are vulnerable for changes in light climate and partly also for increased sedimentation.

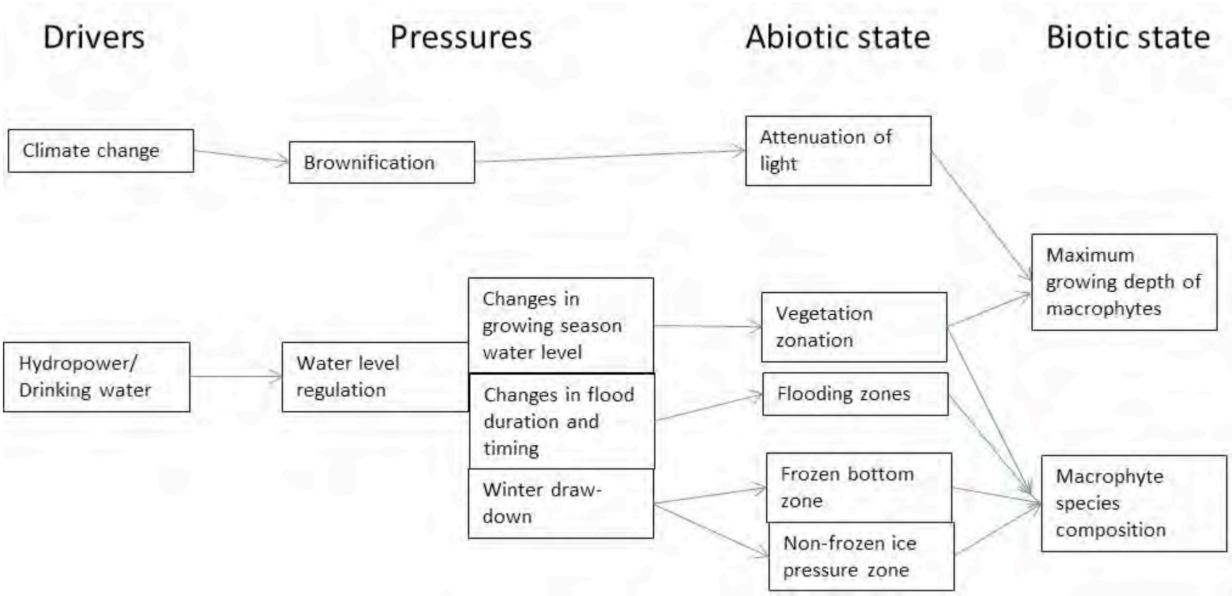

Figure 53. Conceptual model of factors affecting on aquatic macrophytes.

Water level regulation includes a need to increase storage capacity of lake or reservoir by raising the water level. This leads to changes in light climate and starts significant erosion processes at the shoreline. Further water level regulation includes drawdown period depending on use of regulated water body. In northern ice covered lakes regulated for hydropower production water level is lowered during winter causing massive effect by ice pressure. Especially perennial species such as large isoetids disappear on the uppermost zone of frozen bottom sediment whereas lower zone of penetrating ice has also negative effect on these plants.

Changes in winter time water level effects also on magnitude and duration of spring flood which is essential for flood dependent littoral plant species such as sedges. Further changes of water level will always effect on distribution of helophytes such as *Phragmites* and *Schoenoplectus* species. Changes in helophytes are a major factor affecting on macrophytes in lakes regulated for other purposes like navigation or irrigation. Larger regulation amplitude during growing season usually means larger helophyte beds, whereas fluctuation outside summer may lead to enhanced erosion limiting distribution of helophytes.

4.2. Effects of water level fluctuation due to hydroelectric power production on macrophyte communities in Nordic lakes

Contributors: Marit Mjelde (NIVA), Seppo Hellsten (SYKE), Mattie O'Hare (CEH)

Summary

The Nordic macrophyte-based water level index WIc index was tested on 55 Finnish and Norwegian lakes, including lakes with different regulation purpose, and both clear and humic lakes. The index showed good relationship with water level drawdown in storage lakes (H3), but not in other regulated lakes. Clear lakes are more resistant to winter drawdown than humic lakes due to better light climate.

In clear water lakes, *Isoetes lacustris* is common or make stands in lakes with winter drawdown down to 3.5 m, whereas in humic lakes the stands are limited to the depth of 1.5 m.

Introduction

The aim of this work was to test the response of the Nordic macrophyte-based water level index (WIc) (Mjelde et al., 2013) to water level stress caused by hydroelectric power production. We used new data from lakes representing different regulation purpose and different lake types. In this analysis we focus on the difference between clear and humic lakes, since brownification (higher humic level) of lakes resulting from climate change may also represent a stressor for the macrophyte communities.

Data

A total of 55 lakes are included; 20 from Finland and 35 from Norway. including 14 new lakes from Norway (Table 16). Several of the new lakes are moderate alkalinity lakes.

The Finnish dataset includes low alkalinity, both clear and humic, lakes. The colour in the humic lakes varied between 45 and 140 mg Pt/l. Annual water level fluctuation varied between 0.1 and 4.7 m. The Norwegian dataset consists mainly of clear water, both low alkalinity lakes and moderate alkalinity lakes, with annual water level fluctuations between 0.1 and 6.2 m. All lakes in the dataset are oligotrophic to slightly mesotrophic lakes, expecting eutrophication effects on macrophytes to be negligible. Some of the lakes have short time variations through the year, in addition to winter drawdown.

Table 16. Analysed lakes.

Reg. type	NGIG	country	Lake name	altitude	lake area	draw-down	year	Secchi	colour	No. of species	WIc
[1]				m	km2	m		m	mg Pt/l		
H2	102	FI	Hyrynjärvi			1.30		2.08	70	14	42.86
H2	102	FI	Iijärvi			1.19		2.08	70	19	21.05
H2	102	FI	Unnukka			0.12		2.63	45	24	45.83
H3	102	FI	Irnijärvi			3.24		2.63	70	6	-33.33
H3	102	FI	Iso-Pyhäntä			3.50		1.92	85	18	-44.44
H3	102	FI	Kemijärvi			2.38		1.97	80	15	-20.00
H3	102	FI	Kiantajärvi			3.12		2.22	60	14	-35.71
H3	102	FI	Kiimanen			1.43		2.32	54	22	31.82
H3	102	FI	Koitere			1.76		2.08	70	13	7.69
H3	102	FI	Kostonjärvi			4.02		2.63	40	11	-45.45
H3	103	FI	Loitimo			1.37		1.66	120	12	25.00
H3	102	FI	Nuasjärvi			1.52		2.13	60	9	33.33
H3	103	FI	Onkivesi			0.79		1.79	130	25	32.00
H3	102	FI	Ontojärvi			3.51		2.22	60	9	11.11
H3	102	FI	Oulujärvi			1.54		2.15	57	33	33.33
H3	102	FI	Oulujärvi10			1.54		2.15	57	30	30.00
H3	103	FI	Porovesi			0.66		1.56	140	18	44.44
H3	102	FI	Suolijärvi			2.27		2.63	40	16	-25.00
H3	103	FI	Syväri			1.51		1.97	100	15	33.33
H3	102	FI	Vuokkijärvi			4.71		2.08	70	15	-40.00
H2	101	NO	Breisjøen	248	0.2	1.32	1998	8	<30	11	-27.27
H2	101	NO	Farris	22	21.1	0.94	1992	5.7	<30	19	0.0
H2	201	NO	Fiskumvatn	19	3	0.13	2001	4.7	<30	30	30.0
H2	101	NO	Nisser	247	76.1	0.85	2015	7.1	14	9	11.1
H2	201	NO	Norsjø	15	55.2	0.29	2015	5.1	18	24	4.2
H2	001	NO	Salsvatn	8	45	0.76	2016	8.7	16	15	-6.67
H2	102	NO	Snåsavatn	24	118	0.47	2016	4.9	34	21	-4.76
H2	101	NO	Vaggatem	51	26.7	0.63	1993		(<30)	23	-4.3
H2	202	NO	Vansjø, Vanemfj.	25	35	0.44	2004		>30	21	19.05
H2b	101	NO	Kilefjorden	167	7.35	0.86	1982	10.5	<30	16	-31.3
H2b	001	NO	Suldalsvatn	69	28.7	0.42	1988	10	<30	9	-22.22
H2b	101	NO	Venneslafjorden	38	1.7	1.18	1986	9	(<30)	17	-23.5
H3	201	NO	Aursunden	689	44	5.41	1982	8	(<30)	14	-28.57
H3	101	NO	Bjørnsjøen	337	1.6	3.3	1941	5	-	16	-25.00
H3	201	NO	Eikeren	19	27.7	0.79	2015	6.9	14	17	23.5
H3	101	NO	Hakkloa	372	2.0	6.2	1941	5	-	8	-75.00
H3	001	NO	Hartevatn	757	5.8	5.71	1976	10	<30	8	-75
H3	101	NO	Hurdalsjøen	176	33.2	2.8	2016	6.1	22	20	10.00
H3	101	NO	Katnosa	464	2.3	4.9	1941	5.2	-	10	-30.00
H3	101	NO	Limingen	418	95.7	6.08	2016	11.1	8	11	-36.40
H3	201	NO	Mjøsa	123	365	3.25	2014	7.9	10	32	9.4
H3	102	NO	Osensjøen	435	45.2	5.49	1982	4	>30	8	-87.50
H3	201	NO	Randsfjorden	134	136.9	2.57	2015	7	<30	31	22.6
H3	201	NO	Randsfjorden	134	136.9	2.49	1982	6.5	<30	22	18.18
H3	201	NO	Røssvatn	374	190	4.9	2016	13.8	8	8	-37.5
H3	201	NO	Savalen	707	15.4	4.7	2012	9.3	11	12	0.00
H3	101	NO	Selbusjøen	157	57.5	4.7	2016	6.6	18	20	-10.00
H3	101	NO	Store Sandungen	390	2.41	3.1	1941	5	-	16	-6.25
H3	201	NO	Storsjøen i Rendalen	251	47.9	3.56	19891	6	<30	7	0.00
H3	101	NO	Tinnsjøen	191	51.6	2.76	2015	9.5	10	15	-20
H3	201	NO	Tyrifjorden	63	121.3	0.62	2015	5.4	18	24	16.7
H3	201	NO	Tyrifjorden	63	121.3	0.52	1981	7.5	<30	34	14.71
H3	101	NO	Vesle Sandungen	390	1.4	3.1	1941	5	-	10	-10.00
H3	201	NO	Øyeren	101	86.7	2.08	1997	3.2	<30	47	14.9
H3b	001	NO	Byglandsfjord	202	33.5	3.02	1982	10.5	<30	13	-53.85

[1] The river-run lakes are indicated as H2b and H3b. The H3b lake are excluded from the H3-analysis; it acts different from the other H3 lakes.

Methods

We have classified the lakes (Table 16) into normal (a) and river-run (b) , using a definition slightly modified from (Rørslett, 1988), see Table 17.

Table 17. Hydrolakes - definitions.

Type	Lake types	Purpose	Level range	Water levels
H3	Storage reservoirs	Hydroelectric power (HEP)	Medium – very large	Drawdown in late winter (often to a lower level than original level). Stabilized high water level through summer and early autumn.
H2	Small reservoirs	drinking water reservoirs; lakes effected by HEP regulation upstream; other reasons	Small - medium	Stabilized water level throughout the year, reduced water level in spring and some higher water level in summer. Small short time variations.

The lakes are typified according to the typology used in European intercalibration for the implementation of the Water Framework Directive (Poikane et al., 2011); where type 001 and 101: very low and low alkalinity clear water lakes, 102: low alkalinity humic lakes, 201: medium alkalinity clear water lakes, and 202: medium alkalinity humic lakes. Low alkalinity implies less than 0.2 meq l^{-1} and medium alkalinity implies between 0.2 and 1.0 meq l^{-1}. Clear water lakes have colour less than 30 and humic lakes more than 30 mg Pt l^{-1}. We have also indicated very humic lakes as more than 90 mg Pt l^{-1} (type 103).

The daily water level data were collected from the Hertta database (SYKE) in Finland and the NVE database in Norway. In Finland water level data from 1980-1999 were used for all lakes, whereas Norwegian data included the last 5 or 10 years prior to the macrophyte survey.

We used winter drawdown as an indicator of water level regulation amplitude (see (Hellsten, 2001; Keto et al., 2006; Keto et al., 2008)). Winter drawdown was calculated as the average difference between the highest water level during the period October-December and the lowest level during the following period April-May. The same calculations where used also for the H2 lakes even though they do not have winter drawdown.

The WIc-index is based on the ratio between sensitive and tolerant macrophyte species. The sensitive and tolerant species are identified based on a percentile approach, analysing species presence or absence along the winter drawdown range. In the present work, we have used the same species list used by (Mjelde et al., 2013), see Table 18.

Table 18. Tolerant and sensitive species.

Group	Tolerant species	Sensitive species
ISOETIDS	*Eleocharis acicularis* *Limosella aquatica* *Ranunculus reptans* *Subularia aquatica*	*Elatine hydropiper* *Isoetes lacustris* *Littorella uniflora* *Lobelia dortmanna*
ELODEIDS	*Callitriche hamulata* *Callitriche hermaphroditica* *Callitriche palustris* *Hippuris vulgaris* *Juncus bulbosus* *Utricularia vulgaris*	*Callitriche cophocarpa* *Elodea canadensis* *Myriophyllum alterniflorum* *Myriophyllum verticillatum* *Potamogeton alpinus* *Potamogeton berchtoldii* *Potamogeton obtusifolius* *Ranunculus peltatus*
NYMPHAEIDS	*Sparganium angustifolium* *Sparganium hyperboreum*	*Nuphar lutea* *Nuphar pumila* *Nymphaea alba* *Persicaria amphibia* *Potamogeton natans* *Sagittaria natans* *Sagittaria sagittifolia* *Sparganium emersum* *Sparganium natans*
LEMNIDS		*Lemna minor*

Results

Figure 54 shows the correlation between winter drawdown and WIc index for the whole dataset, including all lakes from Finland and Norway, and both H3 and H2 types.

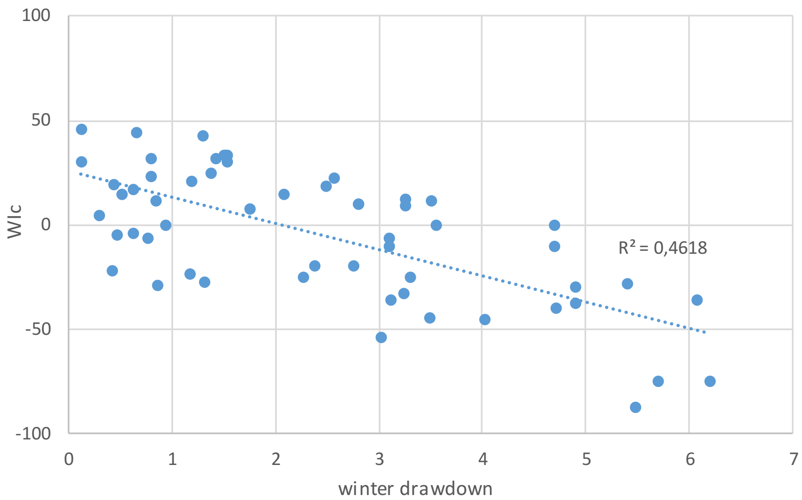

Figure 54. WIc-drawdown correlation for all 56 lakes.

The H3 lakes have winter drawdown in spring and stabilised water level through summer, while the H2 lakes have more or less stabilised water level throughout the year. These two regulation types affect the macrophyte community in different way (e.g. (Mjelde et al., 2013)). Splitting in two groups improves the correlation for H3 lakes, while, as expected, there are no correlation between WIc and winter drawdown for H2-lakes (Figure 55).

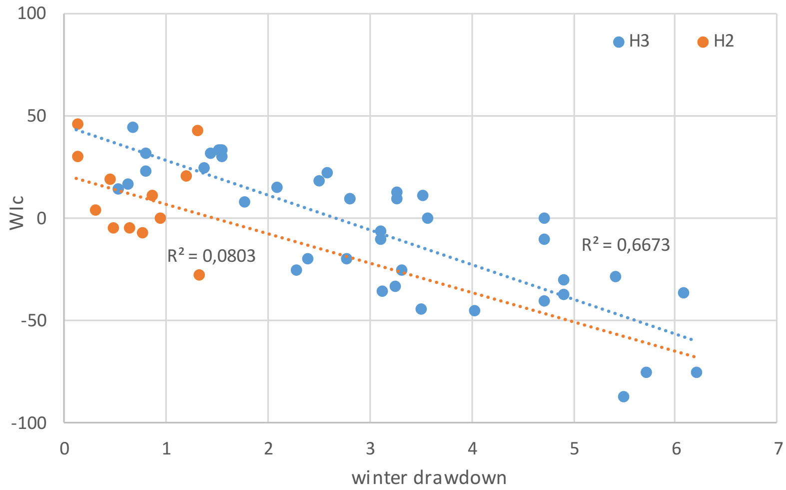

Figure 55. WIc-drawdown correlation for H3 versus H2 lakes.

We continued the analysis with H3 lakes only. The correlation between WIc and winter drawdown seems to differ between Finnish and Norwegian lakes (Figure 56), however, we believe that the reason for this difference is the inclusion of moderate alkalinity lakes in the Norwegian dataset (Figure 57 and Figure 58).

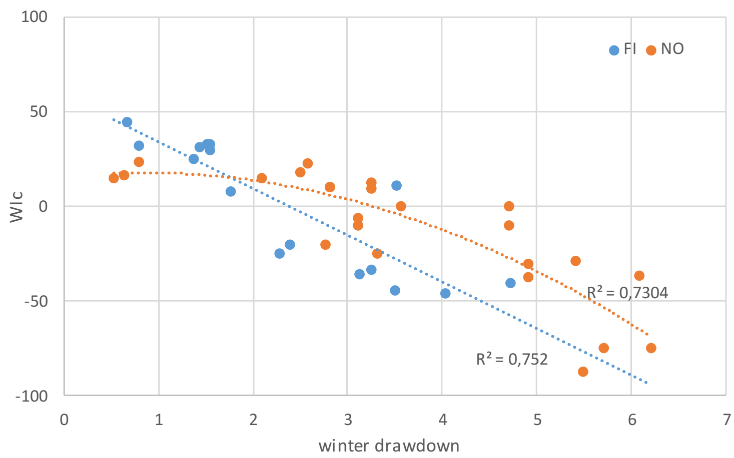

Figure 56. WIc-drawdown correlation for FI versus NO lakes (only H3).

Figure 57 includes only low alkalinity lakes in the two countries, while Figure 58 shows the difference between moderate (only NO lakes) and low alkalinity lakes (FI and NO lakes merged).

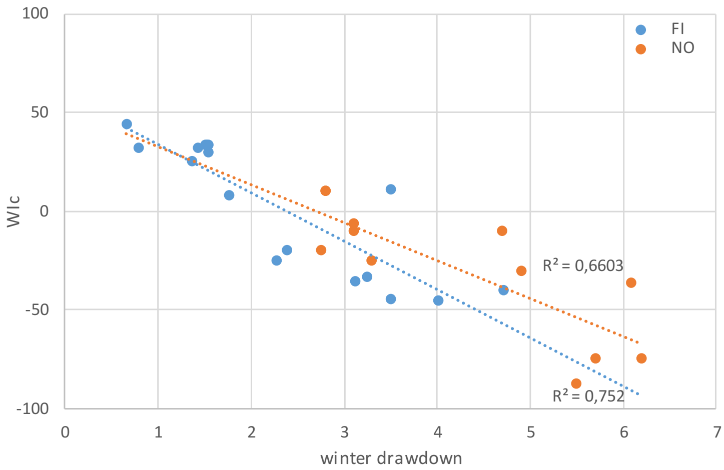

Figure 57. WIc-drawdown correlation for low alkalinity lakes, FI versus NO

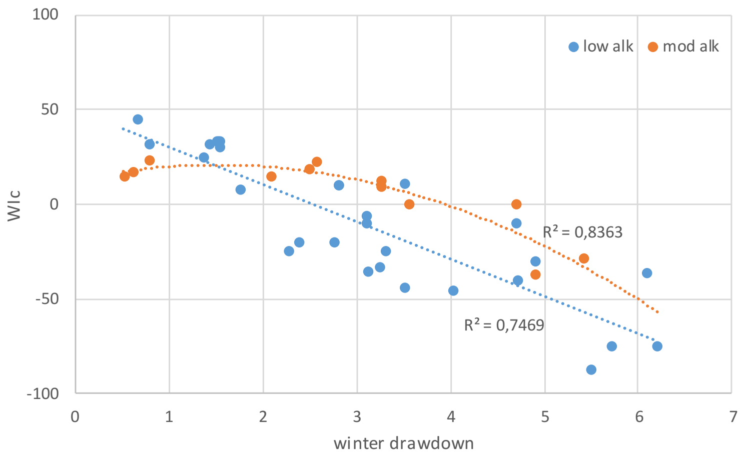

Figure 58. WIc-drawdown correlation for low alkalinity lakes (FI & NO) versus moderate alkalinity lakes (NO).

The diversity in low alkalinity lakes is often lower than in moderate alkalinity lakes, which could be the reason for the moderate lakes seem more resistant to water level regulation. However, diversity depends also of the lake area and elevation, which differs between countries ((Rørslett, 1991).

Figure 59 shows the correlation between WIc and winter drawdown for clear water versus humic low alkalinity lakes. It seems that the clear water lakes are more resistant for winter drawdown due to better light climate. The diversity in humic lakes is not lower than in the clear water lakes. There are only three very humic lakes with relatively limited winter drawdown, but obviously these lakes are even less resistant than humic lakes.

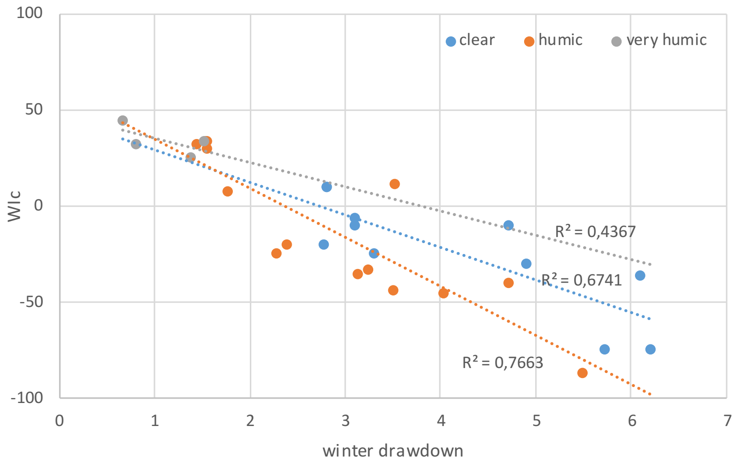

Figure 59. WIc-drawdown relationship in clear water, humic and very humic low alkalinity lakes.

Additionally, we investigated the use of variations in the abundance of *Isoetes lacustris* as an indicator. In clear water lakes, abundance of *Isoetes* (> 3 at the semi-quantitative scale) can be found in lakes with winter drawdown down to 3.5 m, whereas in humic lakes dense stands are limited to the depth of 1.5 m (Figure 60).

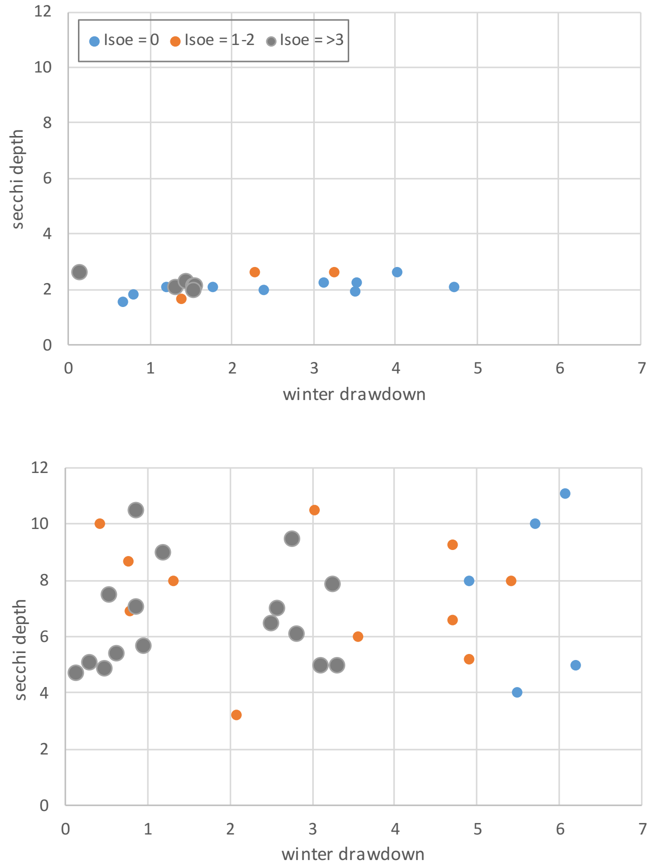

Figure 60. The upper graph shows the abundance of Isoetes lacustris *in Finnish humic lakes, while the lower graph illustrates the stands of Isoetes lacustris in clear water Norwegian lakes.*

Isoetes lacustris is one of the key ecosystem engineers of lake littoral providing growing surfaces for benthic algae and feeding habitats for fishes. Depth distribution of *I. lacustris* is limited by light attenuation and as a perennial ever green plant it cannot resist penetrating ice. Additionally, it is suffering of sedimentation and therefore cannot grow on silty bottom. Abundance indicator shows clearly the effect of increased humic substances on *I. lacustris*; dense stands cannot survive in reduced light climate and increased water lever drawdown.

Key messages

- The WIc index was tested on Scandinavian lakes with different regulation purpose. The index showed good relationship with water level drawdown in storage lakes (H3), but not in other regulated lakes.
- Clear lakes seem to be more resistant to winter drawdown than humic lakes.
- *Isoetes lacustris* is one of the key ecosystem engineers of lake littoral, and may be an indicator for multiple stress from water level regulation and brownification.

4.3. Effects of water level fluctuations on macrophytes in reservoirs subject to eutrophication: an example from Ireland

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Introduction

The aim of this work was to determine the impacts of water level and other stressors on lake macrophytes assemblages. The previous chapter describes a conceptual framework for our understanding of stressor interactions on this community. In this chapter we examine the potential development of a macrophyte metric, diagnostically sensitive to water level fluctuations.

The River Basin Management Plans reported to the EU by member states indicate that many water bodies are potentially subject to both nutrient and water level fluctuation stresses. This has created some initial research interest but has required more detailed research to provide practical answers (Leira and Cantonati, 2008). In response to that management challenge, the call text of Task 5.3 specifies that 'new [macrophyte] community-based indices more sensitive to water level stress, will be analysed'. A first step therefore was to develop a new water level index using macrophytes.

Previous, successful attempts have been made to develop a metric sensitive to water level fluctuations. A metric does exist for Nordic lakes (Mjelde et al., 2013). It is calculated from macrophyte assemblage data and is sensitive to seasonal fluctuations in water level. In Fenno-Scandian countries water levels drop in reservoirs and lakes, exposing overwintering macrophytes to the elements. Freezing and drying out are considered to be the main mechanisms by which macrophytes are damaged in these areas. The reason water level drops are that reservoirs continue to supply water throughout the winter but it is not replenished by rainfall, rather precipitation is in the form of snow, and it is only in spring that water levels re-establish.

While the Fenno-Scandian metric exists, a Pan–European index does not. The question needed to be asked, 'does it make sense to have a single index for Europe?'. While a single index is desirable as it would circumvent the need for inter-calibration across member states it was concluded that this was impractical and that a regional approach was more realistic, see box 1. Like other forms of hydromorphological degradation alterations to water level fluctuation effect macrophytes through different mechanisms and these exhibit regional differences in prevalence. This is the fundamental reason for a regional approach.

The theoretical understanding of lake macrophytes responses to both water level fluctuations and eutrophication stresses, and their interactions are described in detail in the previous chapter. They indicate that direct effects of water level fluctuation should be most pronounced in the upper littoral zone and focusing on the flora of this zone gives the best potential for an indicator which is sensitive to and diagnostic for water level fluctuations. This approach has proven successful in the past (Hawes et al., 2003; Riis and Hawes, 2002).

The Fenno-Scandian metric was used as the starting point for the development of a new metric from temperate Northern Europe. The geographic region was determined by the availability of suitable data to the project. To construct the original metric, indicator species were selected and given a weighting factor, based on their occurrence in systems subject to water level fluctuation. A weighted average of the lake macrophyte abundance was calculated and tested by correlating it against a seasonal water level fluctuation index.

For the new data a similar approach was taken:

1. Indicator species were chosen
2. A weighted lake macrophytes abundance was calculated
3. It was correlated against seasonal water level fluctuations.

We tested the data to see if it was possible to directly or indirectly copy the Scandinavian approach. We also tested other types of response metrics based on community assemblages and we looked for evidence of the effects of both eutrophication and water level fluctuation.

Box 1 Constraints on a generic water level response index for European macrophytes

Contrasting seasonal patterns in water level

The Nordic macrophyte metric works because the plants suffer during periods of winter drawdown. Elsewhere along the western sea board of Europe this is a relatively localised possibility limited to lakes feed from spring snow melt. In the more temperate areas, subject to Atlantic frontal weather patterns, rainfall is higher on average in winter than summer. This temporal pattern can be observed in annual lake level patterns. The pattern of water use is such that demand is heaviest in the summer period. Water managers indicate that this leads to an extenuation of the seasonal patterns with water reserves built up in winter for consumption in summer leading to higher levels in winter, lower levels in summer.

Regional differences in species composition

While Europe as a whole has a limited number of macrophyte species they do differ across the region making it difficult to directly transfer metrics based on species responses, for example (Mjelde et al., 2013) identified macrophytes as either sensitive (22 species) or tolerant (12 species) to winter drawdown and used these to calculate a winter drawdown index (WI_c) while Ireland has 16 of the sensitive species and 7 of the tolerant species.

Data

While the implementation of the Water Framework Directive has encouraged the survey of macrophytes and such data are readily available, it is uncommon to have associated water level fluctuation data of sufficient temporal frequency to facilitate metric development. Ireland is exceptional, the Irish Environment Protection Agency, hold both long term water level fluctuation data, nutrient stressor data and macrophytes survey data. The macrophyte surveys are recorded by depth, facilitating detailed analysis. Of the 48 lakes for which data was supplied 22 were managed primarily as drinking water reservoirs, 1 for drinking water and hydropower (Pollaphuca reservoir) and of the remaining 25 lakes many are managed as sport fisheries, with some possibly subject to minor water level regulation and have near natural water level fluctuations (Table 19).

Water level, macrophyte and water chemistry data were supplied by the Irish Environment Protection Agency. Daily or hourly water level data were available for all the study sites and or most lakes at least 10 years' data were available. The macrophyte data is recoded for a number of sampling stations on each lake. At each station transect are run from the shore out to the maximum growing depth of the vegetation with the vegetation samples at a series of fixed intervals from the shore (0, 2.5, 5, 7.5, 10, 25, 50, 75 and 100 m). Data on the lateral extend of reed beds was also recorded, as was the substrate type and exposure of the shoreline.

*Table 19. A list of 30 lakes and reservoirs with water level data and macrophytes data. *Indicates the system is used for hydropower too.*

Lake	Area	Used for water supply?	Species richness	Macrophyte Status	Seasonal Water level Index
Acorrymore	13.87	yes	7	Moderate	0.518
Allen	3331.9	no	17	Poor	1.24
Anure	132.56	no	30	Good	0.9365
Carra	1557.88	yes	35	High	0.68275
Carrowmore	911.19	yes	30	Good	0.9
Corrib Lower	5042.05	yes	41	Good	0.80053
Corrib Upper	11519.92	yes	44	High	0.842
Cutra	382	no	25	Good	1.126
Derg	858.92	yes	15	Good	0.56
Drumore	60.47	no	18	Poor	1.278
Easky	118.68	yes	14	High	0.442
Emy	52.39	yes	9	Bad	1.03
Ennell	1151.45	no	33	Moderate	0.749
Eske	385.22	no	22	Good	0.88
Fad	40.2	yes	17	High	0.581
Feeagh	393.09	no	25	Good	0.998
Gill	1375.33	yes	16	Moderate	0.91
Inchiquin	107.26	yes	30	Moderate	1.342
Leane	1944.29	no	41	Good	1.548
Lene	414.55	yes	18	High	0.374
Mask	7796.76	yes	37	Good	1.864
Mourne	66.28	yes	13	Good	1.336
Muckno (Blayney)	354.34	no	17	Bad	1.0995
Oughter	658.21	no	44	Moderate	3.019
Owel	1017.64	yes	16	High	0.391
Pollaphuca Reservoir*	1946.16	yes	26	Moderate	1.04
Scur	113.24	no	23	Poor	0.8211
Sheelin	1808.23	no	34	Moderate	0.6302
Shindilla	65.34	no	22	High	0.957
Upper Lough Skeagh	61.04	yes	17	Poor	0.806

Methods

Hydrological analysis of water level. Rørslett (Rørslett, 1988) categorised lakes by their water level fluctuations. Using the categories of water level fluctuation, the time period of which is not specified but assumed to be long term annual averages; 47 lakes have 'small fluctuations', ≤ 2 m and 3 lakes have 'medium fluctuations' 2 – 4 m. (Mjelde et al., 2013) provide examples of the patterns in annual water level fluctuations for natural lakes, drinking water reservoirs and storage reservoirs.

Seasonal depth Index. A seasonal water level fluctuation index (SWL) was calculated as follows for each study site: as the median difference between the highest water level during the period October–February and the lowest level during the following period April–September, calculated for the 5-year period preceding macrophyte sampling. This was based on a similar Scandinavian metric.

Macrophyte data processing. Before analysis the Irish data was reviewed and a standardised list of Operational Taxonomic Units (OTU) was developed, as not all specimens were identified to species level, some were typically identified to genus level only. As the number of transects differed between lakes all macrophyte data was standardised by sampling effort by dividing the abundance of the species at a lake by the total number of transects sampled.

Testing of the existing Scandinavian macrophytes metrics sensitive to water level fluctuation on Irish data. The potential application of the Nordic WIc metric was tested by calculating it with Irish macrophytes data and testing for a relationship with the SWL using linear regression analysis. We examined the overlap in species that are used to generate the metric between Scandinavia and Ireland, the rank order of the species with depth

Screening for new metrics and indicator species. We tested the relationship of individual species with the season depth index. We also looked at summary metrics, total abundance and species richness, and the key groups, obligate hydrophytes and helophytes. Regression analysis and fitted line plots were used to test for relationships. The research was focused on species which occurred in the upper littoral, here defined as 0.5 m depth or less.

Stressor interactions. To test for stressor interactions we used ordination analysis. Canonical Correspondence Analysis was used to reveal the effect of nutrient gradients, the water level index and natural drivers of aquatic plant assemblages. The plots produced by CCA provide a visual representation of environmental variables, their relationship with plant assemblages and their interactions.

Statistical Analysis. All analyses were carried out in Minitab 16.1 or Canoco. Data were normalised as necessary.

Results

Hydrological analysis of water level

The Irish lakes did not exhibit large fluctuations of between 4 and 8 m. 47 lakes have ‘small fluctuations’, ≤ 2 m and 3 lakes have ‘medium fluctuations’ 2 – 4m. Reservoirs and ‘natural’ lakes could not be distinguished on the magnitude of their water level fluctuations.

Both natural lakes and reservoirs exhibit a strong seasonal pattern with higher water levels in winter and lower water levels in summer, examples are given in Figure 61 and Figure 62. Acoorymore is unusual showing little fluctuation throughout the year.

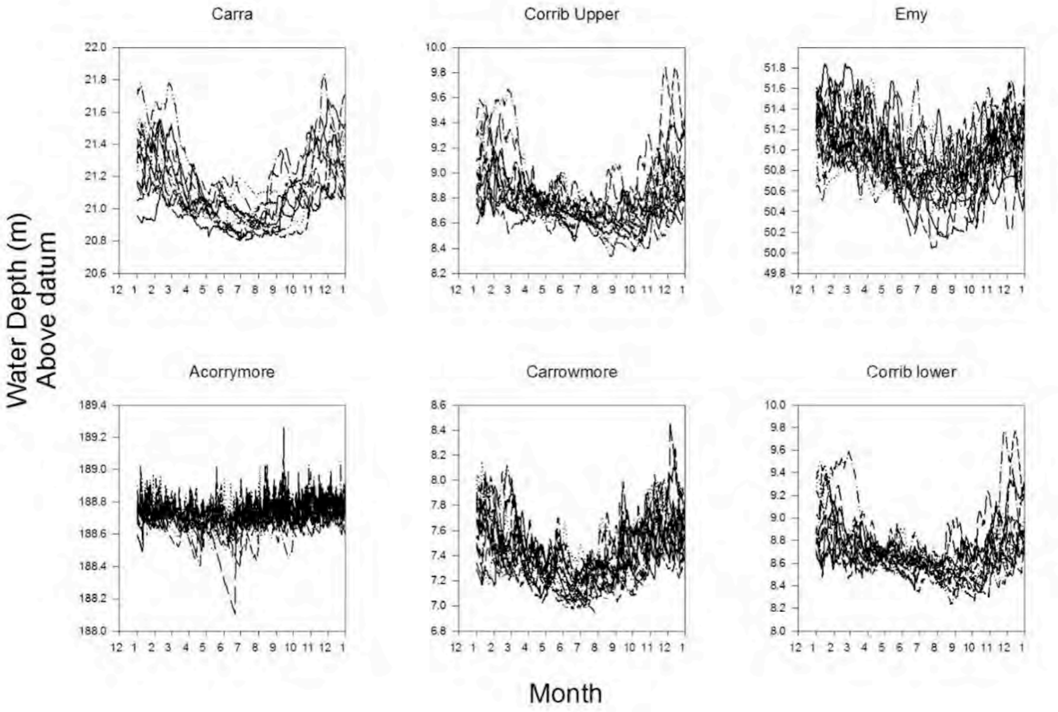

Figure 61. Six lakes in Ireland which act as water supply reservoirs. Data are daily values for a period of 10 years preceding 2012.

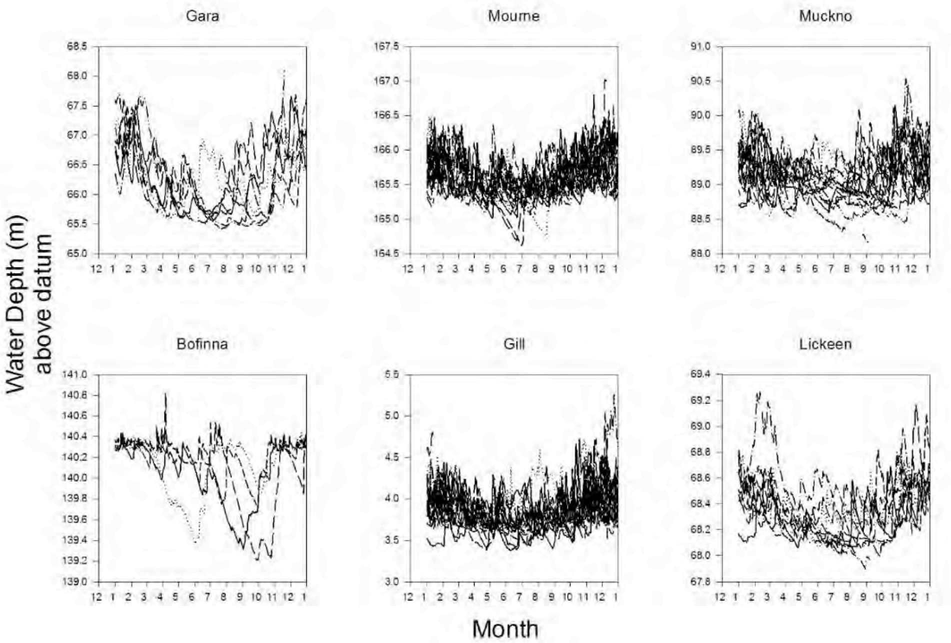

Figure 62. Six lakes in Ireland where no or only minor water level regulation is in place. Data are daily values for a period of 10 years preceding 2012.

Seasonal depth Index

The seasonal depth index, which is the 5-year median of annual fluctuations in depth between winter and summer, ranged from 0.37 m at Lough Lene to 3 m at Lough Oughter (Figure 63).

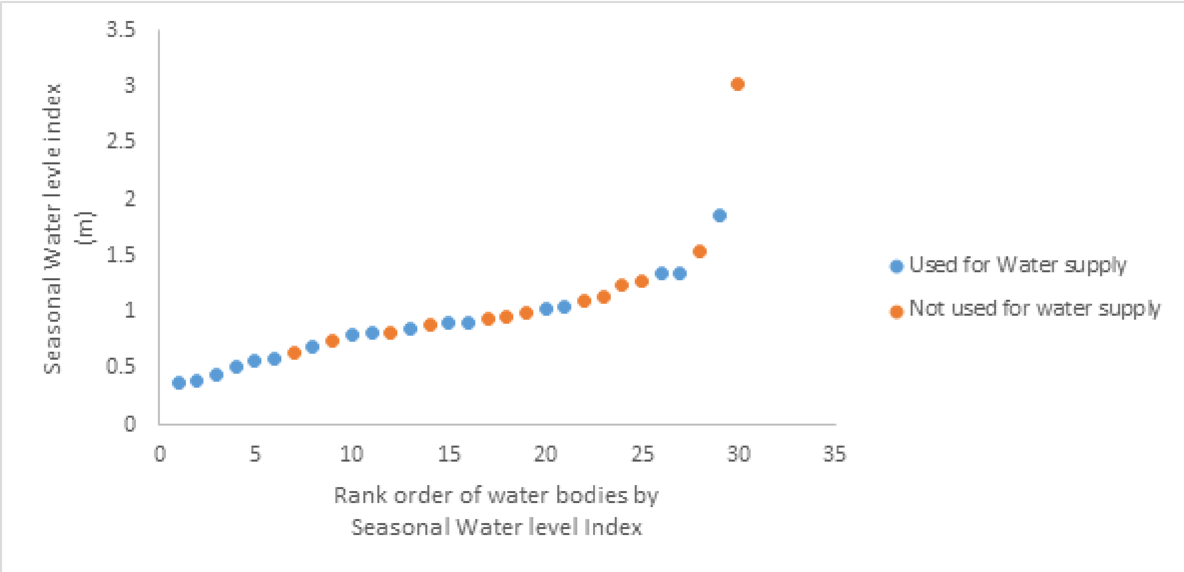

Figure 63. A plot of the sites with both water level and macrophyte data illustrating no significant difference in the SWI between lakes and reservoirs used for those that are not used for water supply.

Macrophytes – basic ecology

Species composition and number. In total 72 OTU (operational taxonomic units) of macrophytes were recorded in the Irish study lakes. The dominant aquatic macrophyte species were *Littorella uniflora*, *Ranunculus flammula*, *Mentha aquatica*, *Fontinalis antipyretica* and *Hydrocotyle vulgaris*.

Depth zonation. The macrophytes vegetation demonstrated strong depth preferences across lakes (Figure 64). The macrophytes show a classic pattern of zonation with depth. Emergent species are replaced by amphibious marginal species and then obligated submerged species. While the mean depths for individual species do not exceed 2 m their ranges could be considerably more. The maximum growing depth of the macrophytes could extend to 8.4 m (Lough Lene) and maximum growing depths. The mean maximum growing depth was 3.46 m.

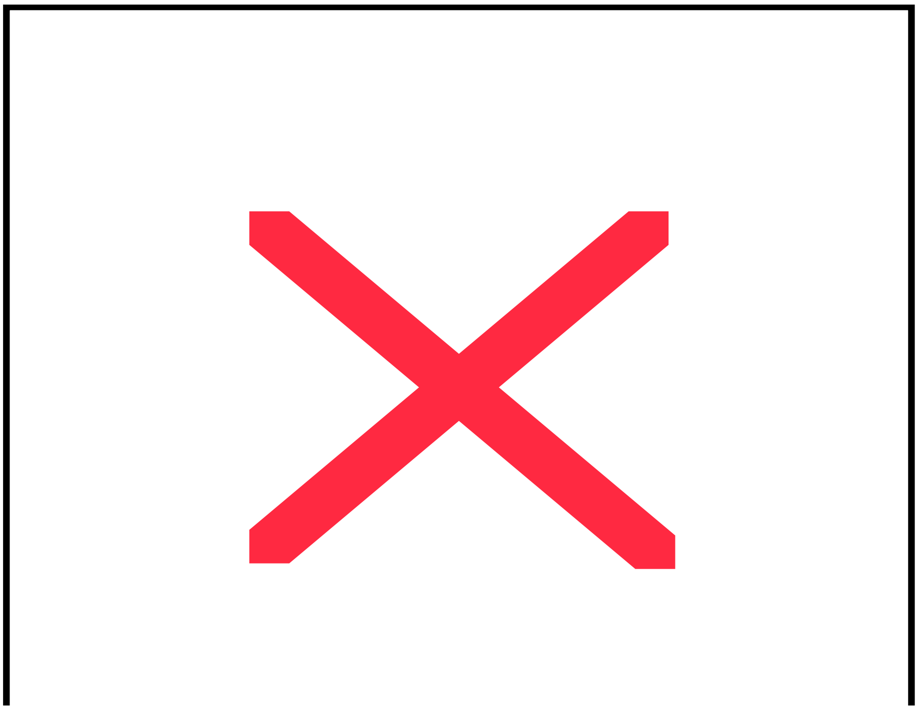

Figure 64. A rank order plot of species and OTUs (operational taxonomic units) by depth. Depth data was collected from all water bodies sampled. Error bars indicate standard error. A list of the species and OTUs are given in Table 20.

Table 20. The species and OTUs (operational taxonomic units) ranked by their mean depth preference across all water bodies sampled.

Species & OTU	Rank order	Species & OTU	Rank order
Juncus articulatus	1	*Potamogeton filiformis*	37
Juncus effusus	2	*Myriophyllum alterniflorum*	38
Hydrocotyle vulgaris	3	*Potamogeton crispus*	39
Lythrum salicaria	4	*Schoenoplectus lacustris* submerged	40
Iris pseudacorus	5	*Juncus bulbosus var. fluitans*	41
Mentha aquatica	6	*Eriocaulon aquaticum*	42
Ranunculus flammula	7	*Fontinalis antipyretica*	43
Filipendula ulmaria	8	*Apium inundatum*	44
Caltha palustris	9	*Potamogeton natans*	45
Equisetum arvense	10	*Myriophyllum verticillatum*	46
Juncus acutiflorus	11	*Phragmites australis*	47
Baldellia ranunculoides	12	*Schoenoplectus lacustris* emergent	48
Eleocharis palustris	13	*Fontinalis squamosa*	49
Didyomosphenia	14	*Elodea nuttallii*	50
Phalaris arundinacea	15	*Alisma lanceolatum*	51
Oenanthe crocata	16	*Callitriche hamulata*	52
Persicaria hydropiper	17	*Nuphar lutea*	53
Carex rostrata	18	*Myriophyllum spicatum*	54
Sparganium erectum	19	*Oenanthe aquatica*	55
Alisma plantago-aquatica	20	*Sparganium emersum*	56
Persicaria amphibia	21	*Lemna trisulca*	57
Glyceria maxima	22	*Chara spp.*	58
Chaetophora	23	*Potamogeton alpinus*	59
Hippuris vulgaris	24	*Potamogeton obtusifolius*	60
Juncus bulbosus	25	*Potamogeton gramineus*	61
Littorella uniflora	26	*Nymphaea alba*	62
Hildenbrandia	27	*Utricularia spp.*	63
Equisetum fluviatile	28	*Isoetes lacustris*	64
Menyanthes trifoliata	29	*Elodea canadensis*	65
Potamogeton pectinatus	30	*Potamogeton perfoliatus*	66
Ranunculus circinatus	31	*Potamogeton friesii*	67
Spirodela polyrhiza	32	*Potamogeton lucens*	68
Zannichellia palustris	33	*Potamogeton pusillus*	69
Lemna minor	34	*Nitella spp.*	70
Ceratophyllum demersum	35	*Potamogeton berchtoldii*	71
Lobelia dortmanna	36	*Lagarosiphon major*	72

Testing of the existing Scandinavian macrophytes metrics sensitive to water level fluctuation on Irish data. We examined the potential for using the seasonal macrophyte based metric developed in Scandinavia in Ireland. The macrophytes index for Scandinavia is sensitive to winter drawdown, a condition which does not appear in Ireland where drawdown occurs in summer. However, the macrophytes could still show sensitivity to both impacts. Mjelde et al. 2013 identified macrophytes as either sensitive (22 species) or tolerant (12 species) to winter drawdown and used these to calculate a winter drawdown index (WIc). The Irish dataset has only 16 of the sensitive species and 7 of the tolerant species. A version of the WIc index calculated using the reduced list of indicator species proved insensitive to summer drawdown (Figure 65).

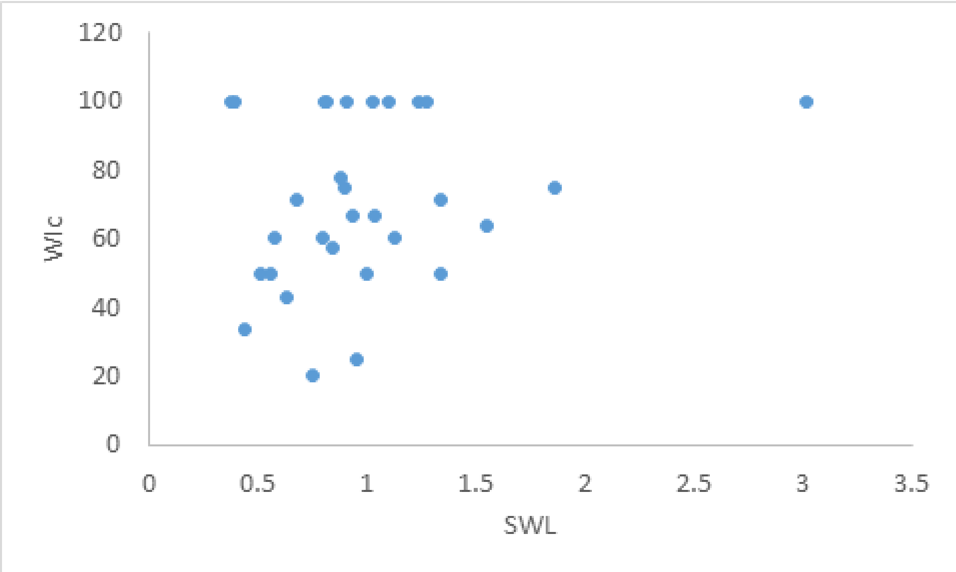

Figure 65. A scatter plot of the seasonal Water Level Index and the Scandinavian WIc index, calculated using Irish macrophyte data.

Screening for new metrics and indicator species

Species richness. There was no statistically significant relationship between species richness and the 5-year seasonal water level index, for all lakes or water supply reservoirs or natural lakes alone (Figure 66).

Individual species (OTU) responses. The following species (OTU) had notable individual responses to SWL; *Chara spp*. (R sq adjusted 20%, P 0.055 n = 14), *Elodea canadensis* (R sq adjusted 49%, P 0.003 n = 13), *Juncus bulbosus var. fluitans* (R sq adjusted 43%, P 0.00.017 n = 10), *Litorella uniflora* (R sq adjusted 8.3%, P 0.084 n = 25), *Persicaria spp.* (R sq adjusted 85%, P 0.001 n = 7).

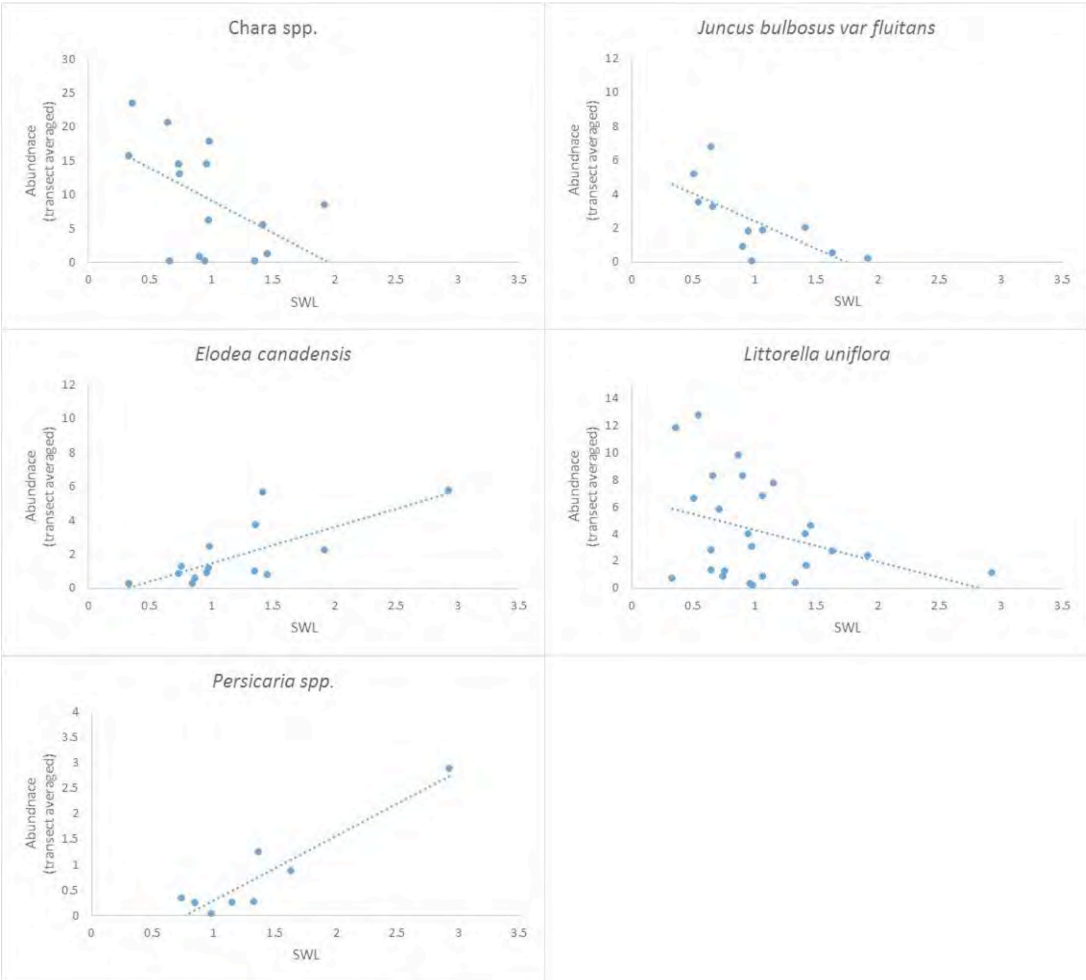

Figure 66. Species and OTUs with notable linear relationships with the Seasonal Water Level Index.

Macrophyte groups in the upper littoral. The ratio between emergent and obligate submerged species shifts to fewer submerged species/ more emergent species with increased seasonal variance, (Rsq adj 17.7% F = 7.02, n = 28, P < 0.05).

Reed bed extent. The reed bed extent in sheltered shores has a weak but significant relationship with the 5-year index. The extent of the beds decreases with increasing seasonal fluctuation, (Rsq adj 24.6% F = 4.91, n = 12, P < 0.05).

Multi-stressor interactions. SWI did not represent a strong environmental gradient across Irish standing water plant assemblages (Figure 67). The strongest gradient across sites were those represented by alkalinity/conductivity and a second gradient represented by total phosphorus (TP).

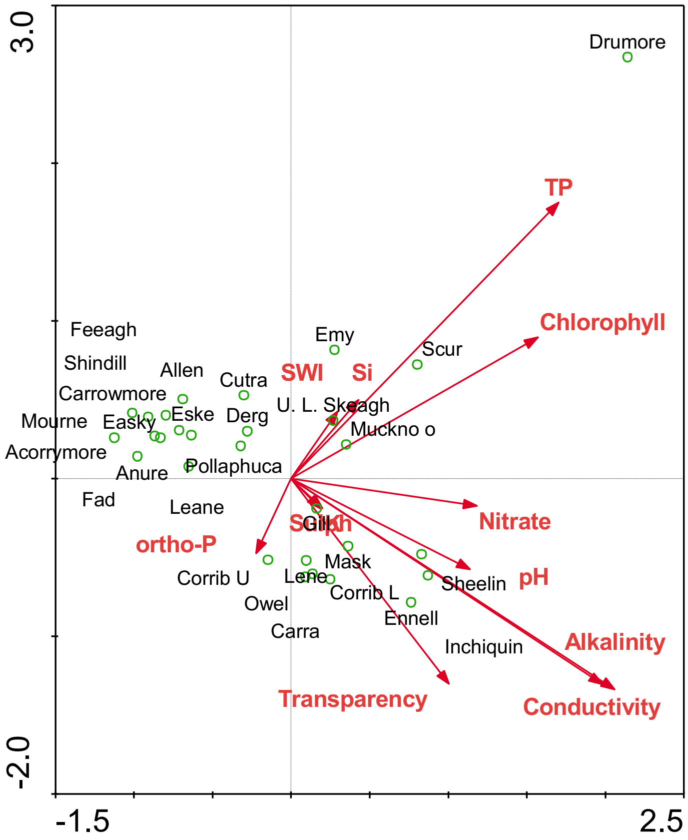

Figure 67. A biplot of environmental variables and Irish standing waters. Environmental variables are represented by arrows. Arrows pointing in the same direction explain similar trends in the data, the length of the arrows indicate their relative importance and how closely arrows align indicates how correlated they are.

Forward selection of environmental variables in the CCA did not indicate either a strong marginal or conditional effect of seasonal water level SWI (Table 21 and Table 22).

Table 21. The marginal (independent) effects of environmental variables on the macrophytes assemblages across Irish standing waters.

	Marginal Effects	
Variable	**Var.N**	**Lambda1**
Alkalinity	1	0.49
TP	10	0.48
Conductivity	3	0.47
Chlorophyll	2	0.35
Transparency	11	0.32
Nitrate	4	0.28
pH	6	0.26
SWI	12	0.16
Si	8	0.13
Ortho-P	5	0.12
K	7	0.1
Sulphate	9	0.1

Table 22. The conditional (partial) effects of environmental variables on the macrophyte assemblages across Irish standing waters.

	Conditional Effects			
Variable	**Var.N**	**LambdaA**	**P**	**F**
Alkalinity	1	0.49	0.002	3.56
TP	10	0.48	0.002	3.79
Nitrate	4	0.21	0.008	1.74
Chlorophyll	2	0.19	0.024	1.6
pH	6	0.16	0.112	1.38
Conductivity	3	0.16	0.082	1.36
Transparency	11	0.14	0.192	1.21
Sulphate	9	0.13	0.298	1.14
Si	8	0.14	0.134	1.28
SWI	12	0.13	0.302	1.14
ortho-P	5	0.1	0.562	0.91
K	7	0.08	0.722	0.76

Discussion

Water level responses & potential for a macrophyte metric in a climatic context

The magnitude of the water level fluctuations in Ireland reflects the use of the lakes, primarily for drinking water where demand is not as severe as elsewhere and rainfall is common throughout the year. That there is no obvious distinction between reservoirs and natural lakes, based on water level is in agreement with another recent study of Irish reservoirs and lakes which examined a range of seasonal measures using Bayesian belief modelling (Molinos and Donohue, 2011).

However, the current situation may not prevail if the demands for water increase in Ireland and reservoirs exhibit more intense water level management. Water use globally is increasing and independently from population growth in developed countries, where consumption per individual is on the increase. Equally and of direct relevance to MARS, climatic change scenarios may lead to more distinct fluctuations in precipitation patterns which in turn will affect water levels in both natural lakes and reservoirs. It has been demonstrated that coupling of lake water level dynamics and climatic parameters (e.g. the North Atlantic Oscillation) is strongly lake-specific (Haghighi and Klove, 2015; Molinos and Donohue, 2014). For these reasons it made sense to pursue the investigation of potential macrophyte responses to water level, despite the lack of distinction in water level between natural regimes and manipulated ones.

The unsuitability of applying the Nordic macrophytes metric (WIc) to Ireland is not surprising in light of the significant differences in the drawdown periods between the two regions, the differential responses to water depth by the macrophytes and the differences in the aquatic flora. The differences are informative in the European context however as both the Nordic region and Ireland represent two distinctly different precipitation patterns, and by inference water level patterns, which are both wide spread across Europe. Ireland is representative of the 'Atlantic north' climatic zone, sensu (Metzger et al., 2005), and similar precipitation patterns can be seen in the 'Atlantic central and 'Lusitenean' areas. Across these western seaboard areas of Europe water inflow to lakes will be primarily driven by frontal rainfall which follows similar patterns to that in Ireland, albeit with an increasing influence of convectional rainfall as one moves inward over the continent. Snow is relatively unimportant in these areas, expect where lake inflow is strongly influenced by mountains. The Nordic seasonal patterns are representative of 'Alpine North' and 'Boreal' regions and have some seasonal comparability with the 'Alpine south' and continental' areas of Europe, areas where snow locks up precipitation during winter. By inference a similar impact of the different precipitation patterns is likely to underpin any response by macrophytes in these areas.

The level of water level fluctuation in Ireland is moderate and the response of the majority of the macrophyte species can be described as muted. This is in accordance with other studies which suggest larger fluctuations are necessary before severe, visually obvious denudation of the upper littoral is visible (Bornette and Puijalon, 2011; Mjelde et al., 2013; Smith et al., 1987). In those other studies inter-annual variation of 3m or more typified standing waters with significant reductions in macrophytes, while in Ireland the major of standing waters had inter-annual fluctuations of 2 m or less. It was therefore concluded that developing a new metric based on the few species which showed a significant response in Ireland would be relatively uninformative in the European context. The response of the individual species, is useful as it does indicate future avenues of research for the development of new metrics. As expected a number of taxa found in the upper littoral or lake margins showed a strong relationship with summer drawdown. Species such as those of the genus *Persicaria* are ruderal, sensu (Grime, 2002), and can spread quickly on damp exposed shorelines and are tolerant of inundation and are likely to increase with greater seasonal fluctuations. A similar process would explain the greater size of reed beds and the subtle increase in emergent/marginal species relative to obligate submerged species in the upper littoral. More surprisingly is the increase in Canadian pondweed with increasing water level fluctuation. It is an invasive, aggressive species which may be able to respond quickly when other submerged species are compromised by large seasonal fluctuations.

The finding that broad groups of macrophytes can indicate impact to hydromorphological impact has been observed in river restoration studies. There individual species rarely occur across sufficient numbers of sites to act as effective indicators but once grouped together they can indicate impact.

Stressor interactions

Previous research has demonstrated that the most obvious stress on Irish lakes is nutrient enrichment (DeNicola et al., 2004; Leira et al., 2006). Our findings confirm the importance of eutrophication with associated parameters (chlorophyll a and total phosphorus) both exhibiting strong gradients in the ordination analysis of the standing water macrophyte assemblages. So too alkalinity is known to represent an especially strong gradient in Ireland with lakes found on hard calcareous geologies and others in acidic conditions.

Against the background of these very strong environmental drivers, the seasonal water level index exhibited little influence and it can be concluded that for the Irish situation it is of secondary importance to eutrophication. There is some indication of a weak correlations between SWL and eutrophication. This is not likely to be a causal interaction across the Irish data and was not statistically significant. In general, Irish lakes which are subject to water level fluctuation as sources of human drinking water, tend to have high water quality and support good populations of macrophytes.

Conclusions

- Water level indices should be developed with regard to the biogeography of macrophytes and precipitation patterns across Europe as represented by European climatic areas.
- Data collected to date suggest water level fluctuations of 2 m or less have limited impact on macrophytes communities although reeds and marginal vegetation do respond.
- Multi-stressor interactions between water level management and eutrophication are possible but depend on the magnitude of both stresses.

Key messages

- Both eutrophication and artificially exaggerated water level fluctuations in lakes can damage aquatic plant communities.

- Evidence from existing studies and work carried in Scandinavia and Ireland during MARS, indicate water level fluctuations damage established plants living near the edge of the water and those in the shallows and can encourage marginal weedy species. There is some evidence submerged species are also effected.
- The intensity of impact from water level fluctuations is related to the seasonal difference in water level, with a fluctuation between winter and summer of 3 m considered to cause significant damage.
- Macrophytes metrics sensitive to water level fluctuations are possible to develop but need to be regional, reflecting the differences in natural water level regimes, which are primarily driven by climate. As different components of the flora are more heavily impacted by eutrophication and water level it should be possible, with more data, to develop metrics which can distinguish between the two stresses.
- In the datasets examined either eutrophication or water level fluctuations were a dominant pressure and interaction strength was weak or undetectable. This finding is specific to the data and the evidence from the literature indicates the stressors would have a significant cumulative impact on macrophytes community if both operated at the same water body.
- Brownification is an emerging stressor whereby the colour of water is becoming more heavily stained as tannins are released with greater frequency during mild winters and springs in Scandinavia and elsewhere. The colour alters the transmission of light and this in turn is considered to impact the aquatic plant community. Like eutrophication, which impact mechanism is similar, there is potential for a combined impact of water level fluctuation and brownification.

4.4. Effects on ecosystem services (case study)

Contributors: Sirje Vilbaste, Ain Järvalt, Kristel Kalpus, Tiina Nõges, Peeter Pall, Kai Piirsoo, Lea Tuvikene, Peeter Nõges (all EMU)

This study (Vilbaste et al., 2016) is the first attempt in Europe to cover a wide spectrum of ecosystem services (ESS) of a large lake. The published paper is provided in Appendix 4. The aims of the study were to make an inventory of the ESS provided by Võrtsjärv and to analyse how these ESS are affected by natural variability and the ecological status. According to earlier studies, the major pressures on the ecosystem of Võrtsjärv are the large fluctuation of water level due to regulation, as well as eutrophication. The most important ESS provided by Võrtsjärv and the estimates of their current state, trends, and main anthropogenic pressures are identified (Table 23).

*Table 23. ESS provided by Võrtsjärv. Associated key species habitats and processes as well as current state, trend and main anthrophogenic pressure are identified. Question marks identifys there is insufficient information to make a judgement; * indicates potential ecosystem service*

Type of service	ESS	Species, community or process of interest	Current state	Trend	Main anthropogenic pressure
Provisioning	Fishery	Eel	Moderate	Decreasing	Fishing
		Pike-perch	Good	Increasing	Fishing
		Pike	Good	Labile	Fishing
		Bream	Good	Static	Fishing
		Burbot	?	Labile	Fishing
		Perch	Bad	Decreasing	Fishing
		Total catch	Good	Stable	Fishing
	Reed manufacture	Common reed	Good	Increasing	Harvesting
	Drinking water*	None	Moderate to good ecological state according WFD; not used	None	Consumption
	Sapropel*	Sedimentation	200 x 10^6 m^3; not used	None	Extraction
Regulating	Maintaining populations and habitats	Mainly native species	Good	?	Eutrophication
	Water purification	Retention of N, P, C	Good	Static	Eutrophication
	Water flow regulation	Flood control	Flow is not regulated	None	Modified hydrological regime
Cultural	Recreation	Contact recreation	Accommodation for less than 1000 person at 29 guest houses	Increasing	Climate change
		Navigation	26 harbours	Increasing	Modified hydrological regime
	Education	Ecosystem functions	Good	Increasing	Various
	Science	Ecosystem functions	Good	Increasing	Various

We analysed a set of ESS indicators against the annual mean values of environmental parameters for 2006-2013. To characterize the provisioning services, fish catches and reed harvesting were investigated. For characterizing regulating services, we used retention of total nitrogen (TN), total phosphorus (TP), and dissolved organic carbon (DOC) in the lake. The importance of the lake for maintaining biodiversity and habitats was estimated on the basis of relevant literature data. For characterizing cultural services, we used the number of visitors of the Lake Museum and the Visiting Centre drawn from the Lake Võrtsjärv Foundation's homepage. According to the principal component analysis, the eutrophication-related and hydrology-related factors explained about 70% of the environmental variability of the lake and showed strong relationships with some of the ESS. Among the provisioning ESS, the annual eel catch and the total fish catch were positively related to different eutrophication indicators while the catches of pike, bream and burbot depended more on hydrological factors. Reed harvesting efficiency was related to the lake's water level. The lake retained 40-80% of the total nitrogen load (Figure 68 a) and 55-74% of the total phosphorus load (Figure 68 b). In the years when the lake accumulated water, it also accumulated carbon, while in the years with a negative water budget, it leaked carbon (Figure 68 c). The indicators of the regulating, maintenance and cultural ESS showed very high variability in different years. The number of visitors depends on many factors, among which socio-economic factors tend to play a more important part compared to the environmental factors. Still, hydrology strongly affects the conditions for recreation at the lake. In years with extremely low water level, the shoreline recedes in some places by up to 0.5 km, leaving the reed belt and mud flats on dry land. In other places, the area of sandy beach may enlarge, but people have to walk hundreds of metres to reach the water for swimming or bathing. Boating is restricted as shallow areas become overgrown by submerged macrophytes, stony lake bottom poses danger, and water depth in ports decreases below critical level. We discovered numerous trade-offs between ESS benefitting from higher trophic state or regulated water level of the lake and the goals of good ecological status of the lake. Our analysis showed a clear need for rules prioritizing life-supporting regulatory services against other ESS.

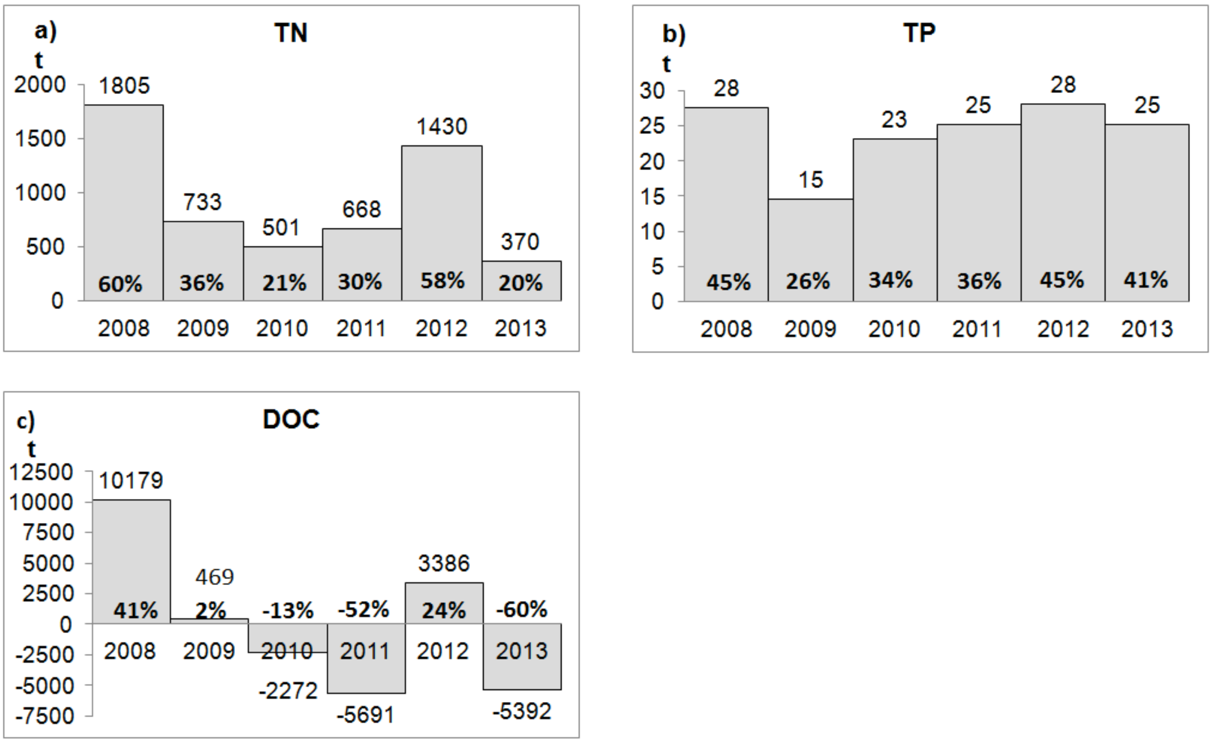

Figure 68. Absolute (t) and relative (%) retention of TN (a), TP (b), DOC (c) by Võrtsjärv in 2008-2013

This study is the first attempt to bring together a holistic picture of real and potential ESS, provided by a large shallow temperate lake, and a description of the main factors affecting the use of these ESS. The poor availability of comparable data made the complex research in this field difficult. This fact, particularly when applying to one of the best studied lakes in Europe, indicates a more general gap in our knowledge about ecosystems. This gap needs to be filled in the nearest future by adjusting the state monitoring programme and by elaborating new indicators for which information should be collected at the national level. A simple analysis carried out in the present paper showed that the environmental factors supporting ESS were not consistent with those needed for achieving good ecological status of water bodies but were rather linked to the resource gathering and encouraged for regulating the naturally fluctuating water level and maintaining the high trophic state beneficial for lake fisheries and reed growth.

Key messages

The ecosystem services concept as an anthropocentric approach is necessary and useful for instructing urbanized human beings who have lost their immediate contact with nature about values of ecosystems not visible at the first glance. However, as it does not establish the priority rules for the variety of ESS provided, the approach includes a risk of becoming a consumerist tool for managers for selecting first-hand management options with the aim of maximising the short term economic benefit, but fails to avoid trade-offs between short-term social and economic goals and long-term goals of environmental sustainability.

5. Stressor combination 4: Nutrients, water level changes and temperature

5.1. Effects on macrophytes and zooplankton (mesocosms)

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Summary

Mesocosm experiments are commonly used in hypothesis testing of shallow lake ecosystem dynamics. This approach enables replicable controlled experiments under natural conditions. In this chapter, the results from four different studies are presented, focusing on different aspects of lake ecosystem using data from the mesocosm experiments of the former EU project REFRESH. More information on the four manuscripts are given in Appendix 6. In Manuscripts 1-3, the same experimental setup was used along latitudinal gradient of Europe in 6 countries, testing the impacts of multiple stressors of temperature, water levels and nutrients. **Manuscript 1** (Coppens et al., 2016) focused on the impact of multiple stressors of nutrients, water level and temperature on retentions of nutrients (TP, TN, DIN, SRP). **Manuscript 2** (Ersoy et al., in prep.) focused on macrophytes growth. **Manuscript 3** (Tavşanoğlu et al., in revision) focused on zooplankton community structure and biodiversity. **Manuscript 4** (Özen et al., 2014) is in-situ mesocosm study from the Mediterranean region, which focused on microbial loop related to water level and fish predation in eutrophic systems.

Impact of multiple stressors of nutrients, water level and temperature on retentions of nutrients (Manuscript 1)

Nutrient retention is one of the key ecosystem services of the shallow lake ecosystems. Nutrient loading and temperature are important drivers determining the nutrient retention capacity of shallow lakes. In this study, controlled experimental mesocosms were used with a space-for-time substitution approach to study nitrogen and phosphorus loss at different depths and nutrient concentrations. The experiments were conducted along latitudinal and temperature gradient from Sweden to Greece between May and November 2011. The experiments had a 2×2×4 factorial design with two water level treatments (shallow: 1 m, deep: 2 m), two nutrient levels (high: 200 µg L^{-1} total phosphorus (TP) and 2.0 mg L^{-1} total nitrogen (TN), low: 25 µg L^{-1} TP and 0.5 mg L^{-1} TN), representing eutrophic and mesotrophic conditions, and four replicates for each treatment. The results showed external nutrient loading to be of key importance for N and P loss in all countries. Significant proportion of all dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) were lost or taken up in biomass in all mesocosms, indicating the high nutrient uptake capacity of shallow lake ecosystems. In the shallow mesocosms, warmer temperature had a positive effect on TN and TP loss, most likely related to macrophyte growth. Increasing N loss could also be attributed to increased denitrification under warmer conditions due to lower availability of DO. However, at the high water levels warmer temperature had a negative effect on TN and TP loss and no effect on DIN or SRP loss, indicating higher algal production of organic N and P in warmer systems causing higher organic N and P accumulation in the system.

(The full manuscript is available in Appendix 6; (Coppens et al., 2016))

Impact of multiple stressors of nutrients, water level and temperature on macrophyte growth (Manuscript 2)

Using the same highly standardised, controlled Pan-European mesocosm experiment as in paper 1, with two water levels (shallow and deep) and two nutrient levels (low and high) in six countries along a latitudinal temperature gradient from Sweden to Greece to evaluate the effect of contrasting depths, nutrient levels and climate on macrophyte growth. This study allowed us to combine space for time substitute approach, using latitude as a substitute for time, with controlled mesocosm experiments, minimizing undesirable effects of local biogeographic and confounding factors and better elucidating complex responses to climate change in natural ecosystems.

We hypothesised that due to higher light availability in the shallow mesocosms at both low and high nutrient levels, macrophyte growth would be higher in shallow than in deep mesocosms. We also expected that the potential and indirect negative effects of nutrients would be weaker in the shallow mesocosms in the warmer southern countries where the expected water level decrease is larger. We anticipated that higher temperatures and low nutrient conditions would promote macrophyte growth and the effects of temperature would be less pronounced in the deep mesocosms due to less apparent effect of water level decline.

ANCOVA revealed significant effects of depth alone and depth-nutrient interactions on %PVI (the percentage of the water filled with plant), whereas nutrients alone had no effect. Mean PVI% reached over 40% in the SL mesocosms of Turkey and Greece. The SL mesocosms of Germany and the Czech Republic exhibited a PVI% of approximately 20%, whereas PVI% was less than 15% in Sweden and Estonia. The greatest growth in the SH mesocosms was observed in Turkey, followed by Sweden and Greece. Macrophyte growth in the DL mesocosms (at most 21% PVI%) was mainly observed in Greece and Germany. However, negligible growth was recorded in the DL treatments of other countries. Only the mesocosms in Greece and Turkey exhibited macrophyte growth in the DH treatment (Figure 69). PVI% increased significantly with temperature under low nutrient conditions demonstrating a pattern of increase from Sweden to Greece. However, the effect of temperature and nutrient interactions was less distinct in the deep mesocosms. PVI% in the SH mesocosms in Greece and Estonia decreased after August, while the enclosures in Turkey retained high PVI% (Figure 69). PVI% in the SL mesocosms peaked in September in Estonia and Greece, in October in Sweden, Germany and Turkey and late October in the Czech Republic. The highest PVI% in DL mesocosms occurred in October in the Czech Republic and Germany, in September in Turkey and late October in Greece. After this peak, macrophyte growth started to decline.

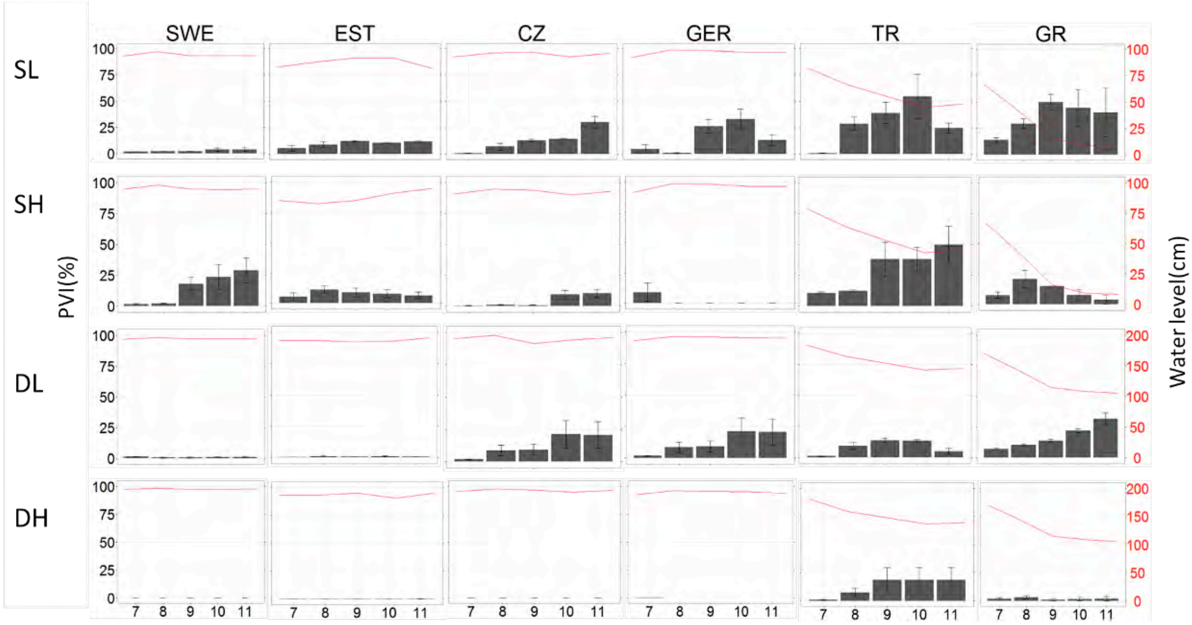

Figure 69. Change in PVI% (columns) and water level (red line) throughout the experiment (%) for SL, SH, DL and DH treatments. SWE: Sweden, EST: Estonia, CZ: Czech Republic, GER: Germany, TR: Turkey, GR: Greece.

The GLM analysis corroborated the results of the ANCOVA analysis and revealed a significant positive effect of mean air temperature on PVI% ($p<0.005$), and a significant negative effect of K_d and water depth ($p<0.001$ for both). Moreover, macrophyte biomass was negatively related to K_d and water depth. Insignificant macrophyte growth was recorded in most of the mesocosms with high light attenuation and high water depth.

In summary the results showed single and combined effects of nutrient, water level and temperature on submerged macrophytes varying between northern and southern countries. Specifically, our experiment revealed i) strong combined effects of temperature and nutrients on macrophyte growth leading to higher PVI% with increasing average temperatures under low nutrient conditions; ii) strong effects of depth-nutrient interactions, PVI% being higher in the shallow mesocosms, both at high and especially at the low nutrient concentrations; iii) negative effects of extreme water level reduction on macrophyte growth.

The results therefore indicate that global climate warming might favour growth of macrophytes with a moderate water level decrease, even under relatively eutrophic conditions in some southern regions. However, if the water level decrease becomes so extreme that macrophytes are directly negatively affected, and longer and intense drought periods become more common, the combined effects of eutrophication and extreme water level reductions may adversely affect the development of macrophytes. In contrast, warmer temperatures in northern regions may not be adequate to induce high macrophyte growth due to increased precipitation and, thus increased water levels and nutrients.

Impact of multiple stressors of nutrients, water level and temperature on zooplankton community structure and biodiversity (Manuscript 3)

The main objectives of our study were to assess the combined effects of changes in nutrient concentrations and water levels on zooplankton community and size structure in shallow lake ecosystems in different climate zones using a space-for-time substitution experimental approach. A highly standardised controlled mesocosm experiments were conducted along a latitudinal gradient ranging from Sweden to Greece (the same as in Manuscripts 1 and 2) and covering an average water temperature from 14.6 to 23.4 °C for the period May to November. We hypothesised that temperature variation and water level change across a latitudinal gradient would have a profound impact on the abundance, biomass and species composition of zooplankton communities. Based on generally accepted predictions, we expected a shift towards small-sized zooplankton accompanied by reduced abundance of large cladocerans in southern countries, not least at the higher nutrient levels. The slope of size spectra is expected to decrease with increasing temperature. Accordingly, we also expected a decrease in richness and diversity in southern countries, size diversity would be narrower reflecting low size variation in community when the lakes get warmer.

Difference in community composition between countries

Non-metric multidimensional scaling demonstrated clear differences in zooplankton communities among the countries (Figure 70), and ANOSIM showed significant differences in zooplankton species composition. In all countries except Greece, the zooplankton community was dominated by cladocerans (~50% of biomass) with small-sized taxa contributing most to total cladoceran biomass (Figure 71). Copepod abundance was similar in all countries except Greece which exhibited the lowest nauplii:Copepoda ratio. *Chydorus* and *Bosmina* (0.3-0.5 mm) were found in all countries along the temperature gradient. *Keratella*, *Lecane*, *Polyarthra*, *Lepadella* and *Trichocerca* were the most common rotifer genera in all countries.

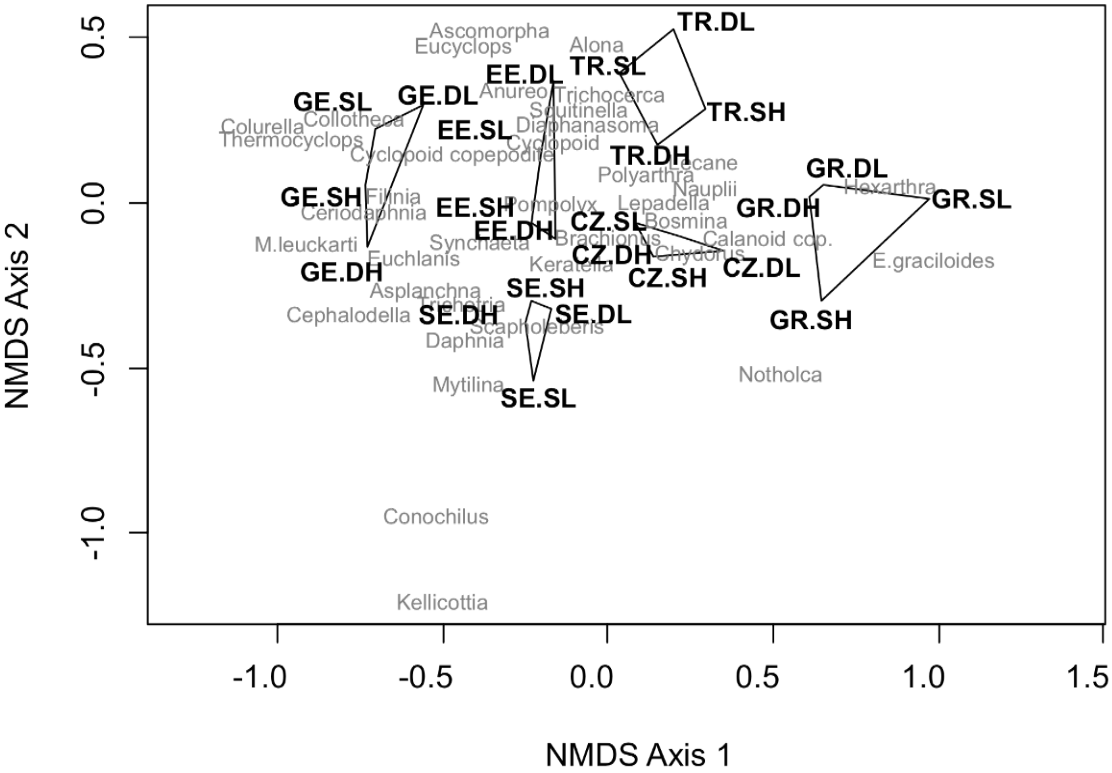

Figure 70. Results of non-metric multidimensional scaling based on the biomasses of zooplankton genera (µg DW L^{-1}). DH: Deep with High nutrients; DL: Deep Low; SH: Shallow High; SL: Shallow Low; SE: Sweden; EE: Estonia; CZ: Czech Republic; GE: Germany; TR: Turkey; GR: Greece.

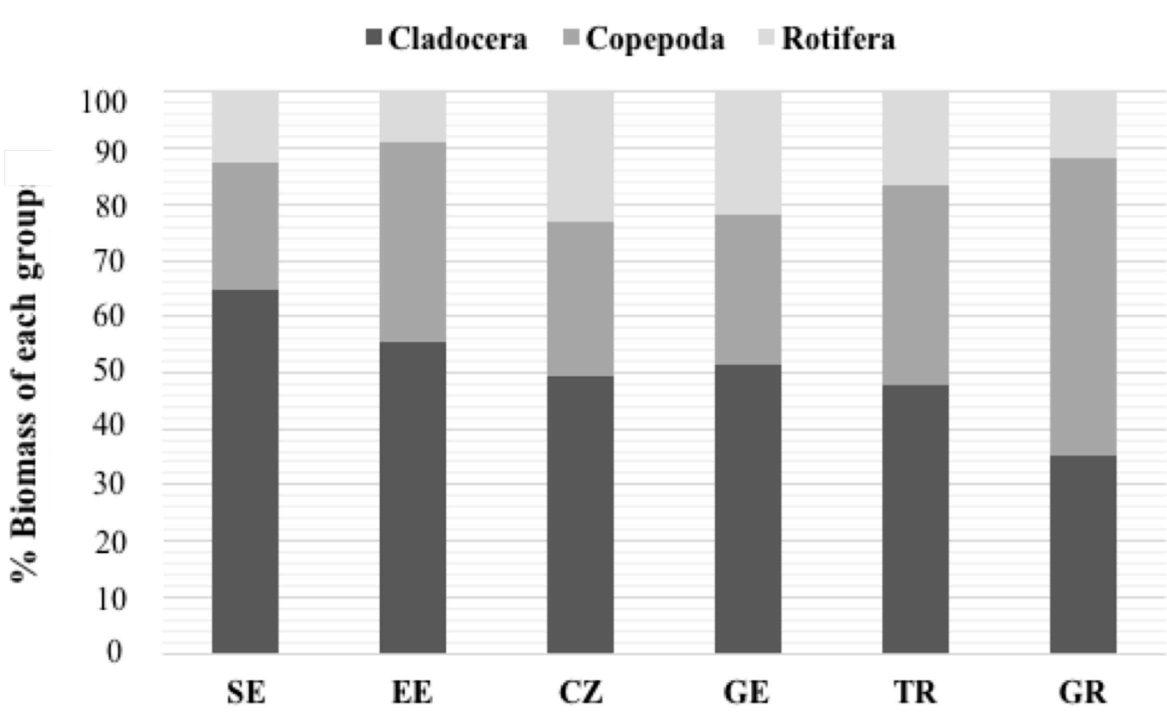

Figure 71. Percentage of zooplankton groups in each country based on biomass. Black columns: Cladocera, light grey columns: Copepoda (including copepodites and nauplii), dark grey columns: Rotifera.

Temperature and treatment effects on biodiversity

The highest genus richness (*S*) was observed in Germany (*S*=36) followed by Estonia (*S*=35), Sweden and the Czech Republic (*S*=35), and the lowest richness was recorded in Greece (*S*=20) followed by Turkey (*S*=24). ANCOVA results confirmed the occurrence of a significant decrease in genus richness with increasing temperature, while neither nutrients nor depth effects were significant. Despite the fact that the overall genus diversity showed no clear pattern, the genus diversity of cladocerans and rotifers decreased with increasing temperature. Accordingly, the dominance of a single copepod species in Greece meant that the Greek mesocosms had the lowest copepod diversity of all the experimental mesocosms. Pielou's evenness (*J*) showed positive interactions in shallow high nutrient mesocosms, while in the deep mesocosms no significant effect appeared with increasing temperature.

Temperature and treatment effects on biomass

ANCOVA revealed no significant depth and nutrient effects on the biomass of cladocerans and copepods, while interaction between temperature and nutrient level had a negative effect on rotifer biomass. In the colder countries (e.g. Sweden), large-bodied cladocerans, such as *Daphnia*, and the large predatory rotifer *Asplanchna* were recorded, while in the warmer countries (Turkey and Greece) *Daphnia* was absent and *Asplanchna* occurred only in low abundances; *Daphnia* did occur at the initiation of the experiment, but in low abundances, in both Greece and Turkey (3.0±2.4 µg DW/L and 0.5±0.3 µg DW/L, respectively). The ANCOVA revealed no significant results for *Daphnia* and nauplii biomass. Interaction between temperature and depth had a significant positive effect on the small:total Cladocera ratio in the shallow mesocosms. Depth had a significant positive effect on the small:total Cladocera ratio in both high nutrient and low nutrient mesocosms. Copepods occurred in all mesocosms in all countries (range 8 to 109 µg DW/L), the highest abundance being observed in Greece where the calanoid *Eudiaptomus* dominated the zooplankton biomass.

Temperature and treatment effect on size structure

Taxonomic diversity was positively related to size diversity, but did not show significant differences among treatments, though size diversity tended to be slightly higher in the low nutrient mesocosms in most countries. The normalised size spectrum (NSS) slope significantly increased with temperature but did not differ among depths or nutrient levels. The slope of NSS was steeper in Greece than in the other countries. The NSS intercept also significantly increased with temperature and was on average highest in Greece.

Conclusions

In summary, the study provided information of the effects of warming on size, biomass and taxonomic structure of zooplankton community. We found that neither taxonomic nor size diversity showed any difference along the temperature gradient. Zooplankton richness, however, was lower in the warmer mesocosms where water loss was higher. Furthermore, the biomass of large-sized cladocerans such as *Daphnia* was notably lower in the warmer countries. A temperature increase would therefore entail a lower biomass of larger crustaceans, resulting in less effective grazing upon phytoplankton. In arid and semi-arid regions, frequency and magnitude of prolonged drought events is expected to increase as a result of global warming. Persistent drying may threaten the local survival of species in the food web.

Water level and fish-mediated cascading effects on the microbial community in eutrophic warm shallow lakes (Manuscript 4)

This study was conducted as an *in situ* mesocosm experiment focusing on effects of water level and fish on microbial community in Mediterranean lakes. Since Mediterranean region is exposed to inter-intra annual water level fluctuations, information on the effects of water level changes on lake ecosystem dynamics are vital. Furthermore, information on how microbial planktonic communities are affected by water level is limited. In this study, we hypothesized that fish predation through top-down control may alter microbial community by changing zooplankton grazing, and water level may have an indirect effect on microbial loop and phytoplankton community by influencing development of submerged macrophytes. Water level and fish interaction was also evaluated since top down regulation of fish could be weaker in shallow mesocosms due to higher abundance of macrophytes. Mesocosm experiment was conducted in Lake Eymir, Turkey, which was run 4-month (June to September 2009). Mesocosms were cylindrical shaped with 0.8-m- (low-water-level) and 1.6-m-deep (high-water-level) and was designed to be open to atmosphere and sediment. To elucidate the roles of fish predation on microbial loop, fish were added to half of the mesocosms, while the rest were kept fishless. The results demonstrated strong top down effect of fish on planktonic community, however, this effect was weakened down through the food chain. The direct effect of water level on microbial community was also found to be comparatively minor. The effect of fish was more prominent in the shallowest mesocosms as lower zooplankton biomass and lower zooplankton:ciliate and HNF:bacteria biomass ratios was observed. Higher bacteria and lower phytoplankton biomasses were found in the shallow mesocosms indicating indirect effects of submerged macrophyte dominance in shallow mesocosms. On the contrary, in the somewhat deeper mesocosms with fish, lowest bacteria, total microbial and HNF biomass was observed due to higher abundance of phytoplankton and absence of macrophytes. Overall the results suggest that water level decrease in warm shallow lakes may enhance the roles of the microbial community at the expense of phytoplankton, likely reflecting higher density of submerged macrophytes; the effect will be less pronounced in the presence of fish.

(The full manuscript is available in Appendix 6; (Özen et al., 2014))

Key messages

- The results showed external nutrient loading to be of key importance for N and P loss in all countries. Significant proportion of all dissolved inorganic nitrogen and soluble reactive phosphorus were lost or taken up in biomass in all experimental mesocosms, indicating the high nutrient uptake capacity of shallow lake ecosystems. In the shallow mesocosms, warmer temperature had a positive effect on TN and TP loss, most likely related to macrophyte growth

- The experiment revealed i) strong combined effects of temperature and nutrients on macrophyte growth leading to higher PVI% with increasing average temperatures under low nutrient conditions; ii) strong effects of depth-nutrient interactions, PVI% being higher in the shallow mesocosms, both at high and especially at the low nutrient concentrations; iii) negative effects of extreme water level reduction on macrophyte growth. The results therefore indicate that global climate warming might favour growth of macrophytes with a moderate water level decrease, even under relatively eutrophic conditions in some southern regions. However, if the water level decrease becomes so extreme the combined effects of eutrophication may adversely affect the development of macrophytes
- Neither taxonomic nor size diversity showed any difference along the temperature gradient. Zooplankton richness, however, was lower in the warmer mesocosms where water loss was higher. Furthermore, the biomass of large-sized cladocerans such as *Daphnia* was notably lower in the warmer countries. A temperature increase would therefore entail a lower biomass of larger crustaceans, resulting in less effective grazing upon phytoplankton. In arid and semi-arid regions, the frequency and magnitude of prolonged drought events is expected to increase as a result of global warming. Persistent drying may threaten the local survival of species in the food web.
- Water level decrease in warm shallow lakes may enhance the roles of the microbial community at the expense of phytoplankton, likely reflecting higher density of submerged macrophytes; the effect will be less pronounced in the presence of fish.

6. Stressor combination 5: Nutrients, temperature, water level changes and salinity

6.1. Effects on phytoplankton, zooplankton, macrophytes and fish (Turkey)

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Summary

In this chapter, effects of stressor combinations of nutrients, temperature, water level and salinity on lake communities in Mediterranean region were demonstrated with 3 manuscripts (see Appendix 7). In Manuscript 1 (Beklioğlu et al., in prep.), data of thirty-one lakes from Turkey covering a wide array of climatic and land use intensity, was analyzed in order to elucidate the response of lake communities and biodiversity to warming and nutrients constraints in Mediterranean region. In Manuscript 2 (Erdoğan et al., in prep.), the effects of major abiotic and biotic drivers on phytoplankton cell/unit size and variance in size structure were investigated in Mediterranean lakes that they were largely controlled by fish, salinity and TP. In Manuscript 3 (Brucet et al., in review), size distribution of phytoplankton, zooplankton and fish communities in Mediterranean lakes were analyzed using data of 30 lakes from Turkey, in which both biotic and abiotic interactions were taken into account.

Together these studies revealed strong variation in shallow lakes ecosystem structure and functions through nutrient level, trophic structure and climate (e.g. temperature) and hydrology to various extent.

Influences of climate and nutrient enrichment on the ecology of mediterranean lakes: a space-for-time substitution approach (Manuscript 1)

In this study, a space-for-time substitution approach was used to assess the response of trophic and community structures and biodiversity to temperature and hydrological constraints in lakes of Mediterranean region that have already been declared as the most sensitive in the world. The data of thirty-one lakes displaying a wide array of climatic features and land use intensity in the western Anatolian plateau of Turkey were analyzed. The warmer southern lakes were found to be more saline and eutrophic (higher proportion of small fish and higher nutrient and Chl *a* concentrations with cyanobacteria dominance) with lower species diversity of most organism groups than the northern lakes as a result of hydrological constraints at similar agricultural intensity. This occurred despite similar land-use conditions in the two regions. Strong top-down control of fish on zooplankton was also traced in southern lowland lakes. The Chl *a*:TP ratio, omnivorous fish biomass and the fish:zooplankton biomass ratio were also higher, whereas the zooplankton:phytoplankton biomass ratio and macrophyte coverage were lower, suggesting a high top-down control of fish on zooplankton. The warmest lakes (southern lowlands) were the most eutrophic and saline, and had a lower species diversity of most organism groups. On a contrary lakes located in the northern highlands, with the lowest agricultural activity and temperatures, had low nutrients and chlorophyll *a* (Chl *a*), low proportions of small fish, large proportions of piscivores, dominance of large-bodied cladocerans and calanoid copepods, all of which indicate low top-down control of fish. Our results indicate that climate warming together with land use in Mediterranean lakes will result in higher salinisation and eutrophication with more frequent cyanobacteria blooms and loss of biodiversity. Consequently, under such conditions ecosystem services (e.g. drinking and irrigation water, biodiversity etc.) are likely to be deteriorated if not lost completely. To counteract, strict control of nutrients and human use of water is urgently needed.

The lakes, sampled using snap shot sampling protocol, were selected in the current study are located over a wide range of latitude and altitude in the Western Anatolian Plateau, spanning five latitudes (41°52´N to 37°06´N) and altitudes from 1 m to 1328 m. The lakes, however, are clumped in two altitude groups, 0-50 and 700-1328 m.a.s.l (Figure 72).

The thirty-one study lakes were largely shallow and small with varying salinity and alkalinity. Total and dissolved nutrient concentrations varied widely (mesotrophic to eutrophic conditions). The visibility index and Chl *a* varied markedly between clear to turbid water. All lakes had fish assemblages largely dominated by small omnivores and piscivores occurred only in a limited number of the lakes. Non-metric Multidimensional Scaling analyses (nMDS) using the bioclimatic and environmental variables differed along latitudinal and altitudinal gradients and generated four distinct lake groups: northern and southern highland and lowland lakes (NH, SH, NL and SL, respectively) (Figure 72). The four groups differed significantly when running a global ANOSIM test (*P*:0.001, R:0.796) followed by post-hoc pairwise comparisons (with a range of *P*:0.006-0.001, R:0.8-1). Lowland lakes differed from the highland lakes by having milder winter conditions (high MTCQ), higher mean summer air temperatures, higher TP, TN and Chl *a* concentrations and higher proportions of small fish (Figure 72). Furthermore, the SL lakes differed from the NL lakes by having the lowest de Martonne aridity index value, indicating the highest aridity, as well as higher salinity and net evaporation (Figure 72). Conversely, lakes at high altitudes differed from lowland lakes by having a higher visibility index and higher PVI%, especially in the northern highland lakes. Moreover, for the SH lakes, precipitation seasonality, net evaporation, salinity and aridity were higher than in the northern highland lakes. Finally, for a few SH lakes nutrients and Chl *a* were higher (Figure 72). Some of the key bioclimatic and environmental variables, critical for separating the lake groups, are given in Figure 73, including mean summer air temperature, net evaporation, salinity, TP, Chl *a* concentrations and the proportion of small fish.

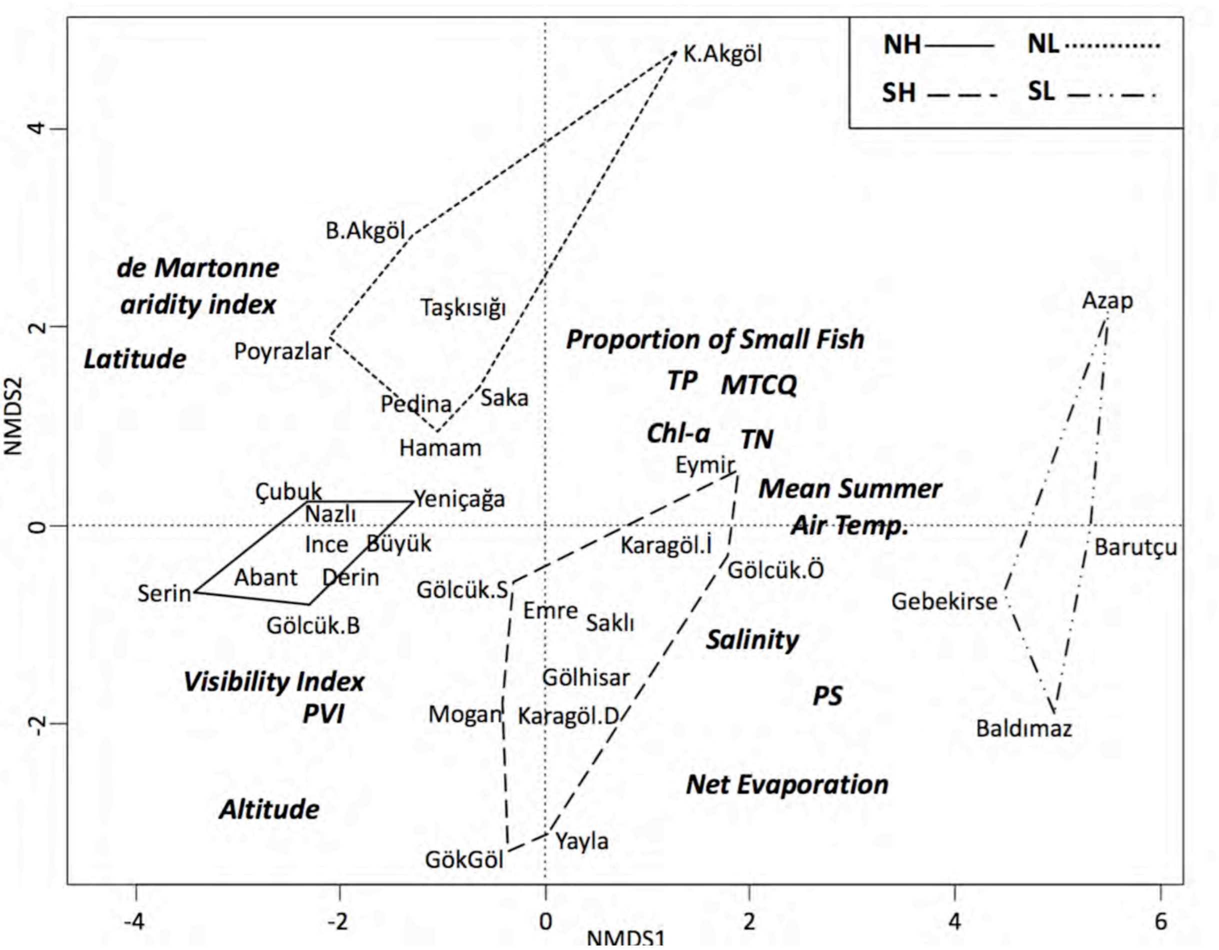

Figure 72. Classification of 31 lakes using non-metric multidimensional scaling analysis creating four groups (NH: northern highland with solid line; NL: northern lowland with dotted line; SH: southern highland with dashed line; SL: southern lowland with long dashed-dotted line) based on bioclimatic variables (net evaporation, altitude, latitude, mean summer air temp: mean summer air temperature, PS: precipitation seasonality and MTCQ: mean temperature of coldest quarter) and environmental variables (visibility index, salinity, proportion of small fish, Chl a: chlorophyll a, TN: total nitrogen, TP: total phosphorus, PVI: percent plant volume infested).

The results from the study lakes representing different latitudes, altitudes and land uses indicate that climate factors such as temperature, as well as salinity and eutrophication, are key parameters determining the trophic structure and community composition of Turkish Mediterranean shallow lakes. As we studied both northern and southern lowland and highland lakes deviating in bioclimatic and environmental factors and land use, we are to a certain extent able to disentangle the effects of eutrophication and climate factors.

Lakes located in the northern highlands (NH), with the lowest agricultural impact and lowest temperatures, had low concentrations of nutrients and Chl *a*, which is associated with more clear water conditions with high abundance of especially short-growing plants (Figure 73). The fish assemblages had a low proportion of small fish and large proportions of piscivores, indicating low top-down control by fish on the zooplankton, which was also evident from the dominance of large-bodied cladocerans and calanoid copepods and a relatively high zooplankton:

phytoplankton biomass ratio. Such conditions are characteristic of north temperate, relatively nutrient-poor lakes (e.g. (Jeppesen et al., 2000)).

In contrast, northern lowland (NL) lakes subjected to higher temperatures and a higher agricultural impact than the NH lakes had high proportions of small fish, higher nutrient and Chl *a* concentrations, a higher phytoplankton biomass and stronger omnivorous fish dominance. The southern lowland (SL) lakes, with an agricultural impact similar to the NL lakes, also showed high proportions of small fish and dominance of omnivorous fish, but were more eutrophic (Chl *a* and phytoplankton biomass). SL lakes also had higher salinity with higher temperatures, precipitation seasonality, drought and net evaporation, leading to higher nutrient and Chl *a* concentrations as nutrients are concentrated in less water where internal loading is higher (Özen et al., 2010), and they therefore become sensitive to excessive water use for irrigation (Beklioğlu et al., 2011). Warmer and drier conditions with higher precipitation seasonality, net evaporation and higher aridity in the southern lakes (Figure 73) seem to enhance not only eutrophication but also salinisation. Enhanced salinity and temperature along with eutrophication are apparently the key factors determining the trophic structure and community composition and diversity of organisms in Turkish shallow lakes. The southern lakes were more saline and eutrophic than the northern lakes as a result of hydrological constraints at similar livestock density and fertilisation levels, particularly in the lowlands, and they overall had lower species diversity of most organism groups.

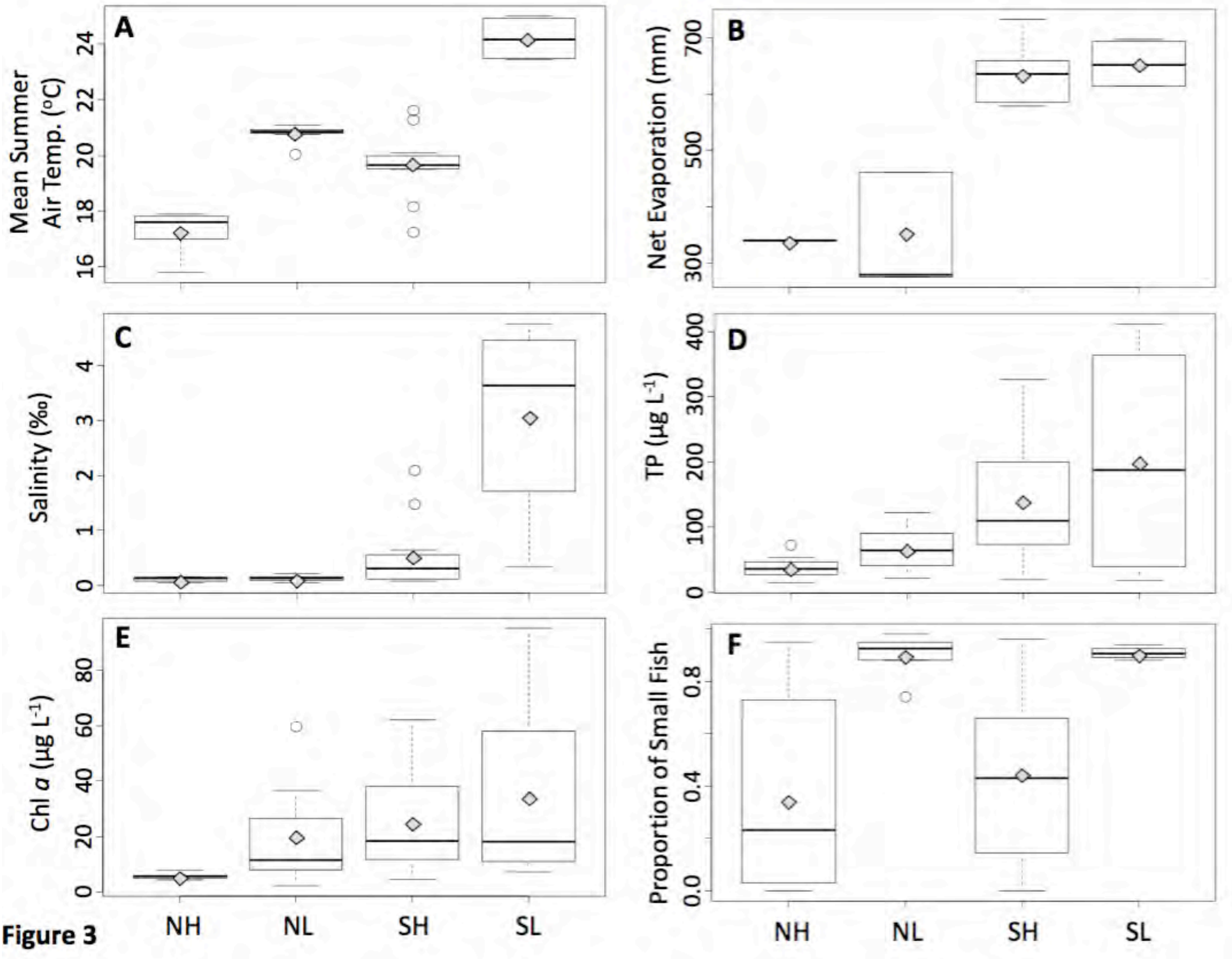

Figure 73. Box-plot of bioclimatic and environmental variables in relation to lake groups (nh: northern highland; nl: northern lowland; sh: southern highland; sl: southern lowland) (a) mean summer air temperature (mean summer air temp.), (b) net evaporatıon, (c) salinity, (d) total phosphoshorus (TP), (e) chlorophyll a (chl a), (f) proportion of small fish.

Determinants of phytoplankton size structure in warm, shallow lakes (Manuscript 2)

In this study, the effects of major abiotic and biotic drivers on phytoplankton cell/unit size and variance in size structure were investigated in Mediterranean lakes. Forty-six mostly shallow and permanent lakes located in Western Anatolian Plateau of Turkey were sampled for various biotic and abiotic variables. Structural Equation Modelling (SEM), was used to detect interaction pathways and direct and indirect effects of numerous biotic and abiotic variables on phytoplankton size structure and their variance. The SEM results showed that only rotifers had direct positive significant effect on phytoplankton unit size. Owing to their small size, rotifers generally are not considered as a potential phytoplankton biomass regulator, however when present in high densities rotifers have a strong grazing impact on phytoplankton biomass as found most of the study lakes. Moreover, instead of direct effect of TP on phytoplankton size, we found indirect of TP effect via zooplanktivorous fish and through rotifer Variance in size also increased with eutrophication probably owing to increase of colonial or filamentous cyanobacteria species. Top-down control was also found to be more pronounced in eutrophic and hypereutrophic lakes than in oligotrophic lakes. Effect of planktivorous fish on phytoplankton was strong, probably due to enhanced predation on zooplankton. The sensitivity of the size structure to biotic and abiotic interactions suggest that trait-based approaches, cell size in particular, can be used as a tool to assess ecological responses to climate change in aquatic ecosystems.

Here, we investigated the effects of major abiotic and biotic drivers, in 46 Turkish lakes, to further our understanding of drivers that control phytoplankton cell/unit size and variance structure in warm regions. In the present study, we used direct microscopic measurements, which allowed us to make more accurate predictions. Additionally, we sampled most of the biotic, physical and chemical parameters and these detailed ecosystem data allowed us to explore driver interactions and effects of them on phytoplankton size structure. We hypothesized that:

- While bottom-up control on phytoplankton mean size is more pronounced under low nutrient conditions, top down control on phytoplankton mean size is stronger under eutrophic and hyper-eutrophic conditions.
- Mean phytoplankton cell/unit size in a community is smaller under high temperature conditions.
- Variance in phytoplankton unit size is highest in eutrophic and hypereutrophic lakes.

structural equation modelling (SEM), a multivariate statistical analysis method that detects interaction pathways, also direct and indirect effects among numerous variables (Anderson and Gerbing, 1988; Grace, 2006) was conducted with TP, temperature, zooplankton and zooplanktivorous fish. Mean unit size analysis included TP, salinity, zooplanktivorous fish and rotifer as independent variables (Figure 74). SEM did not reveal direct effects of TP, salinity and zooplanktivorous fish on phytoplankton size, however their interaction with rotifer abundance and the rotifer biomass direct effect on phytoplankton size structure were statistically significant. Overall SEM results explained 20% of total variance (RMSEA 95% CI = (0, 0.223), $X^2 = 0.310$, df = 5) (Figure 74).

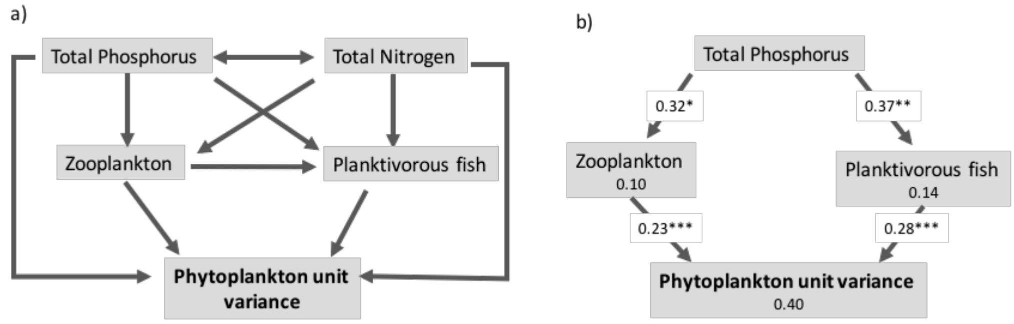

Figure 74. Variance in unit size SEM analysis results a) SEM diagram, b) final SEM results. Arrows represent casual positive relationship, coefficients and significance values were presented on arrow lines. R2 values were given under variable names. p<0.05; 0.01**; 0.001****

The SEM analysis of variance in unit size included TP, TN, zooplankton and zooplanktivorous fish data as predictors (Figure 75). While zooplankton and zooplanktivorous fish had a direct effect on phytoplankton unit size variance, TP had an indirect effect. Overall SEM result explained 40% of total in phytoplankton unit size variance (RMSEA 95% CI = (0, 0.335), X^2 = 0.203, df = 5 2).

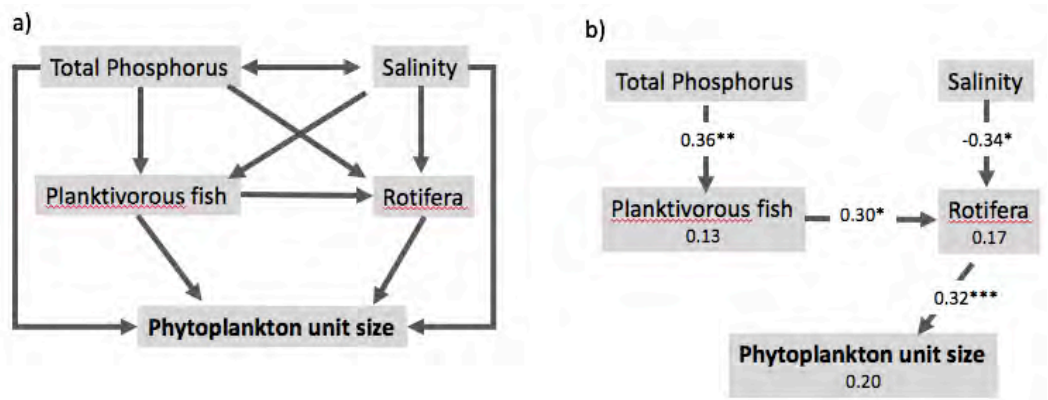

Figure 75. Phytoplankton unit size SEM analysis results a) initial SEM diagram, b) final SEM results. Arrows represent casual positive relationship, coefficients and significance values were presented on arrow lines. R2 values were given under variable names. p<0.05; 0.01**; 0.001****

According to SEM analysis, only rotifer biomass had direct positive significant effect on phytoplankton unit size and no significant relationship was found for cell size (Figure 74). The reason was probably due to the selective rotifer predation pressure on small size phytoplankton species (Figure 74). Owing to their small size, rotifers generally are not considered as a potential phytoplankton biomass regulator, however when present in high densities rotifers have a strong grazing impact on phytoplankton biomass (Lionard et al., 2005). Since most of our lakes were eutrophic and hypereutrophic this result was in accordance with our first hypothesis which stated that top down regulation on phytoplankton size would be more pronounced under eutrophic and hypereutrophic conditions. Moreover, instead of direct effect of TP on phytoplankton size, we found indirect of TP effect via zooplanktivorous fish and through rotifer (Figure 74). Selective fish predation on large zooplankton species (e.g. *Daphnia*) could be the main reason of positive correlation between zooplanktivorous fish and rotifer. Accordingly, total rotifer biomass was positively correlated with eutrophication in our lakes and while it was highest in hypereutrophic lakes, mesotrophic lakes had the lowest biomass. This is also in accordance with previous studies as high fish predation leads to small sized grazers to predominate (Strecker et al., 2004).

Temperature differed considerably among the study lakes, and it was hypothesized (second hypothesis) to be an important driver of phytoplankton size, nevertheless we did not observe a significant temperature effect. Moreover, (Rüger and Sommer, 2012) found only one out of seven phytoplankton species, showing significant size shrinkage in response to temperature increase. On the other hand, many studies support species replacement hypothesis for the mean phytoplankton community size decrease (Daufresne et al., 2009; Winder et al., 2009).

In summary, our results highlight the sensitivity of cell size structure to biotic and abiotic variables, like nutrient increase and zooplankton grazing. Moreover, our results suggest that trait-based approaches, cell size in particular, can be used as a tool to assess ecological responses to climate change in aquatic ecosystems (Litchman and Klausmeier, 2008; Yvon-Durocher et al., 2011). We found that the top-down regulation is more pronounced in eutrophic and hypereutrophic lakes than in oligotrophic lakes. Planktivorous fish had an especially strong effect on phytoplankton, probably due to enhanced predation on zooplankton. Climate change scenarios predict increased drought periods, higher evaporation rate such conditions likely to lead to intensified irrigation, and consequently, salinization as well as intensifies eutrophication of already nutrient rich lakes in semi-arid to arid Mediterranean (Christensen et al., 2013; Jeppesen et al., 2009). To better understand lake ecosystem responses to environmental parameters, shallow lake ecosystems should be monitored regularly. However, taxonomic identification of phytoplankton is time consuming and requires expertise, but size data collection is simpler and does not require much taxonomic background and according to our results can give insights about ecosystem functioning, however more detailed researches are needed to clarify main mechanisms.

Size-based interactions across trophic levels of the food web in shallow mediterranean lakes (Manuscript 3)

In this study, both biotic interactions and environmental factors were considered as a determinant of the size distribution of aquatic communities in Mediterranean lakes. Potential predation effect of size-structured predators (i.e. predation by individuals of different sizes) on prey size structure using data from 30 shallow Turkish lakes spanning over five latitudes were evaluated. Confounding effects of temperature and resource availability were also considered in the analysis. The results demonstrated corresponding size structures between the two interacting trophic levels of the planktonic food web; thus, highly size diverse fish assemblages were associated with highly size diverse zooplankton assemblages. Temperature had negative effects on the size evenness of fish while the effect was positive for phytoplankton. In addition, resource availability was the only predictor of the phytoplankton size diversity, though the effect was week. Overall the results suggest that the size structure at adjacent trophic levels may also control size structure within a trophic group in addition to temperature and resource availability. The positive relationship between the size diversity of adjacent trophic levels (fish and zooplankton) suggest that higher diversity of the resources drives a higher size diversity of consumers or vice versa. Additionally, the results pointed out that variation in temperature and resource availability should also be taken into account when studying size based trophic interactions.

Here, we assessed the potential predation effect by size-structured predators on prey size structure by searching for relationships between size diversity and size evenness of predator and prey across the tri-trophic planktonic food web (fish, zooplankton, and phytoplankton). We hypothesized that the enhanced strength of top-down control at increasing predator size diversity (García-Comas et al., 2016; Ye et al., 2013) will lead to a negative relationship between size diversity and size evenness of predators and prey (i.e. negative relationship between fish and zooplankton size diversity or between zooplankton and phytoplankton size diversity). A simultaneous comparison of size diversities of predators and prey communities across several lakes is not yet available, but a negative relationship has been found between zooplankton and phytoplankton size diversities in marine systems (García-Comas et al., 2016). We further evaluated the relationship between the size diversity of the prey and the log biomass ratio between adjacent trophic levels as a measure of classic top-down control (i.e. when assessing factors determining phytoplankton size diversity, we added the log zooplankton:phytoplankton biomass ratio as an additional predictor). We expected to find a negative relationship indicating that increased density of predators reduces prey size diversity (Quintana et al., 2015).

Thirty shallow lakes spanning over almost five latitudes, from the warm temperate north (41°52´N, 27°58´E) to the semiarid south (37°06´N, 29°36´E) of the Western Anatolian Plateau of Turkey, and with an altitude range of 1 to 1328 m, were selected. The lakes included two distinct climates, the semiarid region located in mid to south-west Turkey and the warm temperate sub-humid region located in north-west Turkey, exhibiting average annual (1980–2010) temperatures and precipitation of 14.5 and 12.0ºC and 545.4 and 632.3 mm, and net evaporation of 616.3 and 338.8 mm, respectively (Turkish State Meteorological Service; www.mgm.gov.tr). The lakes also covered wide gradients of nutrient concentrations, conductivity, and lake area.

Fish size diversity (µfish) was significantly positively related to µzooplankton, explaining 35.5% of the variation in the data (Figure 76). As judged from the significant positive relationship between µfish and µzooplankton, fish size distributions with a wide size range and more similar proportions of the different sizes were associated with zooplankton size distributions with similar characteristics. High µzooplankton reflected the presence of large-sized Cladocera or Copepoda in similar proportions as small-sized rotifers and nauplii, causing a bimodal size distribution with a second dome, corresponding to large sizes beginning around size class -0.7 (log2 µg dry weight). However, when only few sizes of fish dominated (low µfish,), µzooplankton was low, and the zooplankton size distribution had a unimodal shape and a narrower size range, with dominance of small sizes mainly represented by rotifers. When only non-piscivorous fish were analyzed, their size diversity was also positively related only to µzooplankton.

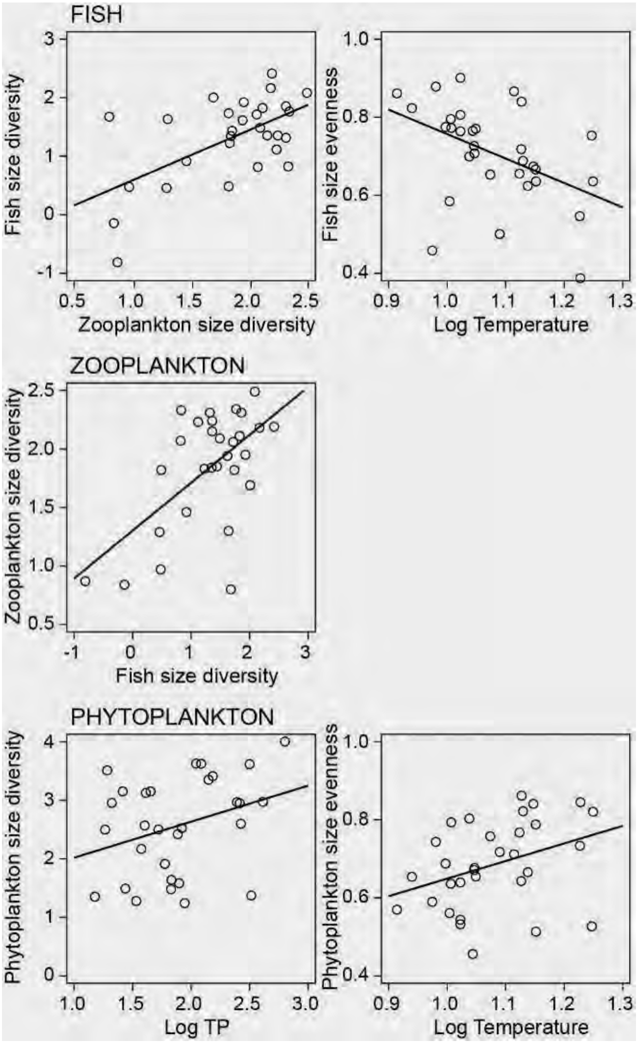

Figure 76. Relationship between size metrics of different organism groups (fish, zooplankton, and phytoplankton) and the independent variables. TP, Total Phosphorus.

In contrast to what we hypothesized, our results showed correspondence of size structures between interacting trophic levels of the planktonic food web. Thus, highly size diverse fish assemblages were associated with highly size diverse zooplankton assemblages, a relationship that was not violated by variation in temperature and resource availability (TP). The correspondence between fish and zooplankton size diversity agrees with the correspondence found in the size distributions of piscivorous and non-piscivorous fish in European lakes (Mehner et al., 2015). A potential explanation is that higher diversity of resources drives higher consumer size diversity. Albeit our focus is size diversity, the underlying mechanism would be similar to that proposed for the positive relationship between species diversity of adjacent trophic groups in terrestrial systems (e.g. (Haddad et al., 2009)): a prey community (zooplankton) highly diverse in sizes could promote size diversity at the higher trophic levels (fish) via productivity effects or by enabling niche partitioning (Currie, 1991; Tilman, 1982). In contrast, low zooplankton size diversity (e.g. lower abundance of large body sizes) could create energetic bottlenecks in fish, potentially explaining the low size diversity values. Evidence that a diversity of prey sizes may favour a size diverse predator community has previously been found in laboratory and field experiments, though the signal was weak (Rudolf, 2012). An alternative explanation may also be possible: higher diversity of sizes in consumers could also promote diversification of resources by size. Thus, high size diversity in fish assemblages may create more chances for resource partitioning in terms of prey size (e.g. zooplankton, macroinvertebrates) (Woodward and Hildrew, 2002), likely resulting in a reduced predation pressure on large-bodied zooplankton (Jansson et al., 2007; Persson et al., 2003) and thus an increase in zooplankton size diversity. Conversely, a community of predators with similar-sized individuals (e.g. dominance of small size fish) occupying similar niches may result in a prey community less diverse in size because some prey sizes would be disproportionally predated over the rest. In conclusion, our results suggest that, in Turkish lakes, size structure within a trophic group may be controlled by the size structure in other trophic groups, as well as by temperature, resource availability, and taxonomic diversity. The positive relationship between the size diversity of fish and zooplankton suggests that higher diversity of prey may drive a higher size diversity of predators, as earlier suggested in studies of species diversity, or vice versa, and these effects are beyond those mediated by taxonomic diversity. In contrast, the size diversity and size evenness of phytoplankton are mainly influenced by physical factors. Additionally, our results suggest that it is important to take variation in temperature and resource availability into account when studying trophic interactions in size-structured predator–prey systems.

Key messages

- Climate warming may enhance salinization and eutrophication with more frequent cyanobacteria blooms and loss of biodiversity in Mediterranean lakes.
- To compensate the effects of climate change, reduction in nutrient loading and water use are required.
- In addition to the temperature and resources availability, the size structure in aquatic ecosystems is also affected by the size structure in adjacent trophical levels also effect. The higher the diversity in resources implies higher diversity in consumers.
- Not only large-sized zooplankton, but also small-sized rotifers can have strong grazing potential if they are present in high densities.
- Taxonomic identification of phytoplankton and zooplankton can be time-consuming, hence trait-based approaches (like cell size) are promising for examining the response of biological organisms to environmental variables in aquatic ecosystems.

7. Synthesis and conclusions

Contributors: Anne Lyche Solheim, Erik Jeppesen, Jannicke Moe

7.1. Overview of stressor combinations and response indicators

The purpose of this report is to assess the response to multiple stressors for indicators of ecological status of lakes, using large-scale European datasets and case studies. The common stressor present in all the analyses is nutrient enrichment causing eutrophication. The main driver responsible for nutrient enrichment is agriculture, and especially arable land.

The additional stressors that have been investigated in this report are related to climate change, hydropower and water abstraction for irrigation and public water supply: temperature increase, hydrological changes (flushing or water level changes), salinisation or increase in humic substances ("brownification"). The latter is also related to reduction of acid precipitation during the last couple of decades. These additional stressors often co-occur with nutrient enrichment.

The main response indicators analysed in this report are phytoplankton and macrophytes, but we have also addressed potential effects on food web interactions and ecosystem services. For phytoplankton, the specific indicators included are (i) ecological status for the whole quality element, expressed as normalized EQRs, (ii) taxonomic composition, expressed as the index PTI, (iii) biomass of cyanobacteria, (iv) concentration of chl-a, and (v) content of fatty acids. For macrophytes, the specific indicators are (i) ecological status mainly based on the trophic index (TIc), expressed as normalized EQRs, (ii) the water level fluctuation index WIc, and (iii) %PVI (plant volume infested; the percentage of the water filled with plant). For food web interactions, the main focus is on interactions between phytoplankton, zooplankton, macrophytes and fish (Turkey). The ecosystem services included are fisheries and reed harvesting (Estonia) and the quality of fish as a food resource (Finland).

An overview of the contents in terms of the different stressor combinations and response indicators is presented in Table 24.

Table 24. Overview of stressor combinations and ecological response indicators. Numbers in parentheses are no. of lakes.

Response indicator / Stressor combination	Phytoplankton	Macrophytes	Community structure and trophic interactions	Ecosystem services	Chapter
Nutrients + temperature + precipitation	Europe (432), Ecological status: nEQR	Europe (441), Ecological status: nEQR			2.1
	Europe (779), Cyanobacteria biomass			Water quality for consumption and bathing	2.2
	Europe (26), Cyanobacteria biomass			Water quality for consumption and bathing	2.3
	Northern Europe (940), Taxonomic composition: PTI				2.4, 2.5
Nutrients + organic carbon	Finland (713), taxonomic composition, content of fatty acids			Finland (713), food quality of fish (content of fatty acids)	3.1
Nutrients + water level		Fennoscandia (56) + Ireland (48): Taxonomic composition, WIc, abundance of single species		Estonia (1), Fisheries and reed harvesting	4.1-4.4
Nutrients + water level+ temperature			Europe (6 mesocosms), Phytoplankton biomass, macrophytes, zooplankton, fish, microbial loop	Nutrient retention; drinking and irrigation water	5.1
Nutrients + water level + temperature + salinity	Turkey (30), Chlorophyll a	Turkey (30), %PVI	Turkey (30), phytoplankton, zooplankton, macrophytes, fish		6.1

7.2. Main results and conclusions

7.2.1. Phytoplankton

Nutrient enrichment and temperature effects on chlorophyll a and ecological status (EQR) for phytoplankton

The total phosphorus, chlorophyll and phytoplankton EQR data available in the EEA WISE-SoE database were combined with temperature data from JRC to analyse the response of chlorophyll a and phytoplankton EQR to total phosphorus and temperature in 218 lakes. As expected, both chlorophyll a and phytoplankton EQR responded clearly to total phosphorus. Temperature only affected chlorophyll a, as revealed by higher chlorophyll a versus total phosphorus slopes in warmer lakes. This response, however, could not be entirely attributed to temperature, because the warmer lakes also have better light conditions, as they have less water colour, and are more shallow than the colder lakes. The lack of temperature response of phytoplankton EQR is probably due to the adaptation of the classification systems to local climatic conditions, causing the deviation from reference conditions to be more or less independent of temperature differences across Europe. This does not mean that future warming of lakes will not affect the ecological status of phytoplankton (see next paragraph).

Nutrient enrichment and climate change effects on taxonomic composition (PTI)

The phytoplankton trophic index (PTI) is one of the indices used for assessing the ecological status of lakes in Northern Europe. We analysed the response of PTI to nutrients (total P and P:N ratio) in combination with climatic variables (air temperature and precipitation), using data from 940 lakes compiled during the former EU project WISER. We used a 3-level hierarchical regression model to account for differences in stressor-response relationships among broad lake types, as well as differences in PTI level among lakes.

The key results showed that PTI increased with Total P for all lake types, but the increase was most prominent in the lowland siliceous lakes, which tend to have lower PTI reference values. The effects of climatic variables will increase PTI for most lake types, suggesting that climate change may worsen the ecological status of lakes in the long run. The interaction between TP and temperature was most positive (gave higher PTI values) for the four siliceous lakes types, implying that siliceous lakes are more sensitive to combined effects of TP and temperature, than the calcareous lakes. In the worst-case climate scenario, however, PTI increased significantly also for very shallow, calcareous, lowland lakes, even in the short-term (2030). If total phosphorus concentration increases, temperature-induced increase of PTI could be expected also for other lowland lake types in the short run. Nutrient reduction measures can partly compensate for the climate-induced increase of PTI.

This study has demonstrated that large-scale data sets with high taxonomic resolution are valuable for assessing effects of multiple stressors on lake ecosystems. The large differences in responses for different lake types show that it is important to consider the lake type in analysis of multiple stressors and assessments of future risks.

Nutrient enrichment and climate change effects on cyanobacteria biomass

Cyanobacteria biomass and total phosphorus data available in the WISER database from roughly 800 lakes across Europe were combined with climate data from JRC (mean monthly air temperature and total summer precipitation) to assess the combined impacts of total phosphorus, temperature and rainfall on cyanobacteria biomass. Various statistical analysis (e.g. pairwise plots, conditional plotting and boosted regression trees) were used to reveal the main response patterns both for the overall dataset, as well as for separate lake types.

The overall response shows a clear increase of cyanobacteria with total phosphorus, a smaller increase with temperature and a weak decrease with summer rainfall, the latter probably due to increased flushing. The relatively weak impact of the climate variables may be due to short climatic gradients in the data, which were strongly dominated by lakes in central and northern regions.

The analysis also provide evidence that the interactions between the nutrient and climate stressors vary depending on lake type, especially concerning the typology factors: depth, water colour and alkalinity.

7.2.2. Macrophytes

Factors affecting EQR of macrophytes at the European scale

Macrophytes in lakes are subjected to numerous stressors. The importance of the different stressors in a European context can in part be achieved by analysing monitoring data and reported EQR (ecological quality ratio) for macrophytes (MP). In MARS, information of MP EQR was assessed from State of the Environment (SoE) reporting of EU member states (601 lake-years from 441 lake water bodies). The analysis revealed that Secchi depth was the by far most important predictor of MP EQR (Figure 15), accounting for 31% of the variance. Including alkalinity, total P, lake mean depth and air temperature increased this figure to 65% (Table 2). Secchi depth is affected by many stressor including eutrophication and brownification. Moreover, changes in water level will influence the depth to which plants can reach at a given Secchi depth, which, in turn, also through a number of feed-back mechanisms affect the Secchi depth. The weak effect of temperature indicate that EQR values will not vary systematically from North to South in Europe, for a given level of Secchi depth and nutrients.

Effects of variation in hydrological regime determined by water regulation

In some lakes the hydrological regimes are heavily disturbed by human activities in terms of water storage for hydropower generation and water supply, general water regulation for flood defence and navigation activities and also in some cases for recreational use. Hydropower effects are typical in northern and high altitude lakes, usually without any other pressures, whereas regulation for navigation and recreation purposes are often seen in lowlands lakes situated in densely populated areas. Morphological and hydrological changes mainly affect the uppermost littoral zone although changes in retention time can indirectly also change the trophic status of whole lakes. Littoral zone macrophytes are one of the key indicators of the hydro-morphological changes in lakes. Macrophytes exhibit a vertical zonation pattern in the littoral, and even small changes in dynamics of water level fluctuation can affect the distribution and elevation of zones.

a) A conceptual model. Hydromorphological stressors can interact with other significant stressors on lake macrophytes. In this report, two key stressor interactions with water level are discussed: eutrophication and brownification, the latter being an emerging issue linked to climate change. Both stressors mean diminished light conditions for the plants. We introduced a conceptual model which aims to encompass the key interactions between stressors (Figure 53). Macrophyte species composition is driven by distributional factors which are further sorted by water quality especially related to alkalinity and main nutrients. Further water quality has effects on attenuation of light via increased production and humic compounds; especially the latter has increased due to brownification. Submerged elodeids and isoetids are vulnerable for changes in light climate and partly also for increased sedimentation.

Water level regulation includes a need to increase storage capacity of lake or reservoir by raising the water level. This leads to changes in light climate and starts significant erosion processes at the shoreline. Further water level regulation includes drawdown period depending on use of regulated water body. In northern ice covered lakes regulated for hydropower production, water level is lowered during winter causing massive effect by ice pressure. Especially perennial species such as large isoetids disappear on the uppermost zone of frozen bottom sediment whereas lower zone of penetrating ice has also negative effect on these plants.

b) Test of the Fenno-Scandian macrophyte index (WIc) on Finnish and Norwegian lakes. The existing Fenno-Scandian water level-drawdown index (the WIc-index), based on sensitive and tolerant species using a percentile approach, was tested on 55 lakes from Norway and Finland. A good relationship was found between the WIc index and the degree of water level regulation (Figure 54), expressed as the average difference between the highest water level during the period October-December and the lowest level during the following period April-May. The correlation was further improved when the dataset included only lakes characterised by winter drawdown in spring and stabilised water level during summer improved the correlation further.

The results further indicated that clear-water lakes are more resistant to winter drawdown than humic lakes, due to better light climate for the macrophytes. Additionally, we found that variations in the abundance of *Isoetes lacustris* could be a useful indicator. In clear-water lakes, abundance of *Isoetes* (> 3 at the semi-quantitative scale) could be found in lakes with winter drawdown down to 3.5 m, whereas in humic lakes dense stands are limited to the depth of 1.5 m, indicating that this widespread species might be a sensitive indicator of multiple stress.

Isoetes lacustris is one of the key ecosystem engineers of the lake littoral, providing growing surfaces for benthic algae and feeding habitats for fishes.

Freezing and drying out are considered to be the main mechanisms by which macrophytes are damaged in the Fenno-Scandian areas. The water level drops because reservoirs continue to supply water throughout the winter but it is not replenished by precipitation in the form of snow, and it is only in spring that water levels re-establish. Our next aim was to investigate to what extent the results from the Fenno-Scandian lakes be used on a European scale.

c) Application of the WIc index on regulated lakes in Ireland. A test of the WIc index was performed on data from 48 lakes in Ireland, of which 22 were managed primarily as drinking water reservoirs, 1 for drinking water and hydropower and of the remaining 25 lakes many are managed as sport fisheries and have near natural water level fluctuations. Besides the Fenno-Scandian WIc index, we also used a Seasonal Water Level fluctuation index (SWL) expressed as the median difference between the highest water level during the period October–February and the lowest level during the following period April–September during a 5-year period preceding macrophyte sampling.

In total 72 species or OTU (operational taxonomic units) of macrophytes were recorded. There was no statistically significant relationship between species richness and the 5 year SWL index, for all lakes or water supply reservoirs or natural lakes alone (Figure 65). However, several taxa had notable individual responses to SWL (Figure 66).

The ratio between emergent and obligate submerged species shifted in disfavour of submerged species and the reed bed extent in sheltered shores had a weak but significant relationship with the 5 year SWL index. The extent of the beds decreased with increasing seasonal fluctuation.

The low applicability of the WIc for the Irish lakes was not surprising in light of the significant differences in the drawdown periods between the two regions, the differential responses to water depth by the macrophytes and the differences in the aquatic flora. The water level fluctuations in the lakes and the response of the majority of the macrophyte species can be described as muted. This is in accordance with other studies which suggest larger fluctuations are necessary before severe, visually obvious denudation of the upper littoral is visible. It was therefore concluded that developing a new metric based on the few species which showed a significant response in Ireland would be relatively uninformative at the European context. However, broad groups of macrophytes can indicate impact to hydromorphological impact as also demonstrated in river restoration studies.

The finding in this study confirms the importance of eutrophication with associated parameters (chlorophyll a and total phosphorus). Alkalinity is also known to represent an especially strong gradient in Ireland. Against the background of these very strong environmental drivers, the SWL index exhibited little influence and it can be concluded that for the Irish situation it is of secondary importance to eutrophication.

Water level effect: a Pan-European mesocosm experiment

To investigate the effect of changes in water level and climate on macrophytes, we analysed data from the REFRESH mesocosm experiment along a latitudinal and temperature gradient from Sweden to Greece (Chapter 5.1). The experiments had two water level treatments (Shallow/Deep) and two nutrient levels (High/Low, representing eutrophic and mesotrophic conditions). It was hypothesised that due to higher light availability in the shallow mesocosms at both low and high nutrient levels, macrophyte growth would be higher in shallow than in deep mesocosms. It was also expected that the potential and indirect negative effects of nutrients would be weaker in the shallow mesocosms in the warmer southern countries where the expected water level decrease is larger. It was further anticipated that higher temperatures and low nutrient conditions would promote macrophyte growth, and that the effects of temperature would be less pronounced in the deep mesocosms due to less apparent effect of water level decline.

The study revealed significant effects of depth alone and depth-nutrient interactions on %PVI (the percentage of the water filled with plant), whereas nutrients alone had no effect. Only the southernmost mesocosms in Greece and Turkey exhibited macrophyte growth in the DH treatment (Figure 69). PVI% increased significantly with temperature under low nutrient conditions, demonstrating a pattern of increase from Sweden to Greece. However, the effect of temperature and nutrient interactions was less distinct in the deep mesocosms.

In summary, the results showed that single and combined effects of nutrient, water level and temperature on submerged macrophytes varied between northern and southern countries. Specifically, the experiment revealed i) strong combined effects of temperature and nutrients on macrophyte growth leading to higher PVI% with increasing average temperatures under low nutrient conditions; ii) strong effects of depth-nutrient interactions, PVI% being higher in the shallow mesocosms, both at high and especially at the low nutrient concentrations; iii) negative effects of extreme water level reduction on macrophyte growth.

The results therefore indicate that global climate warming might favour growth of macrophytes in lakes with a moderate water level decrease, even under relatively eutrophic conditions in some southern regions. However, if the water level decrease becomes so extreme that macrophytes are directly negatively affected, and longer and intense drought periods become more common, the combined effects of eutrophication and extreme water level reductions may adversely affect the development of macrophytes. In contrast, warmer temperatures in northern regions may not be sufficient to induce high macrophyte growth due to increased precipitation and, thus increased water levels and nutrients.

7.2.3. Community structures and trophic interactions

It is well-established that the trophic structure and interactions in lakes are highly affected by and sensitive to stressors. While the effects of enhanced nutrient loading from catchments and cities have been extensively studied and also included as a key element in WFD, much less are known about other stressors, not least climate variation (and change) and associated changes in nutrient concentrations, salinity and water level. Even less so are known about the combination of climate and eutrophication effects. Especially the Mediterranean region will face such combined stressors. To elucidate the effects on community structure and trophic interactions, we analysed results from field studies, field experiments and controlled mesocosm experiments.

Cross-system analysis of Turkish lakes with contrasting latitudes and altitude

A space-for-time substitution approach was used to assess the response of trophic and community structures and biodiversity to temperature and hydrological constraints in lakes of Mediterranean region. The data of thirty-one lakes studied in the western Anatolian plateau of Turkey spanning five latitudes and a large altitude gradient displayed a wide array of climatic features and land use intensity The lakes fall into two altitude clusters (Figure 73).

The results indicate that climate factors such as temperature, as well as salinity and eutrophication, are key factors determining the trophic structure and community composition of Turkish Mediterranean shallow lakes. The warmer southern lakes were more saline and eutrophic with higher proportion of small fish, higher nutrient and Chl *a* concentrations and higher cyanobacteria dominance than the northern lakes, despite similar agricultural intensity. They also had lower species diversity of most organism groups. This differences were both seen in lowland and highland lakes, though the differences between northern and southern lakes were overall highest in lowland lakes. A strong top-down control of fish on zooplankton was traced in southern lowland lakes. The Chl *a*:TP ratio, omnivorous fish biomass and the fish:zooplankton biomass ratio were higher, whereas the zooplankton:phytoplankton biomass ratio and macrophyte coverage were lower, suggesting a high top-down control of fish on zooplankton. On the contrary. lakes located in the northern highlands, with the lowest agricultural activity (similar though to southern highland lakes) and temperatures, had low nutrients and chlorophyll *a* (Chl *a*), low proportions of small fish, large proportions of piscivores, dominance of large-bodied zooplankton, all of which indicate low top-down control of fish.

The results indicate that climate warming together with expected changes in land use in Mediterranean lakes will result in higher salinisation and eutrophication with more frequent cyanobacteria blooms and loss of biodiversity. Consequently, under such conditions, ecosystem services (e.g. drinking and irrigation water, biodiversity etc.) are likely to be deteriorated if not lost completely. To counteract, strict control of nutrients and human use of water is urgently needed.

In a subset of these lakes, size distribution of selected aquatic communities was analysed in order to identify indicators of changes in trophic interactions (Figure 76). The results demonstrated corresponding size structures between the two interacting trophic levels of the planktonic food web; thus, highly size diverse fish assemblages were associated with highly size diverse zooplankton assemblages. Temperature had negative effects on the size evenness of fish while the effect was positive for phytoplankton. In addition, resource availability was the only predictor of the phytoplankton size diversity, though the effect was week. The positive relationship between the size diversity of adjacent trophic levels (fish and zooplankton) suggests that higher diversity of the resources drives higher size diversity of consumers and vice versa. Additionally, the results pointed out that variation in temperature and resource availability should also be taken into account when studying size based trophic interactions.

Effect of changes in water level and predation in Turkish Mediterranean lakes: a mesocosm experiment

Mediterranean lakes are strongly influenced also by year-to-year variation in water level and more long term climate change induced changes. To elucidate the effect of water level in shallow lakes an *in situ* mesocosm experiment elucidating the effects of water level and fish on microbial community was conducted in a Mediterranean lake. Furthermore, information on how microbial planktonic communities are affected by water level is limited. It was hypothesized that fish predation through top-down control may alter microbial community by changing zooplankton grazing, and water level may have an indirect effect on microbial loop and phytoplankton community by influencing development of submerged macrophytes. The mesocosms had low or high water level. To elucidate the roles of fish predation on microbial loop, fish were added to half of the mesocosms. The results demonstrated strong top down effect of fish on planktonic community, however, this effect was weakened down through the food chain. The direct effect of water level on microbial community was minor. Overall the results suggest that water level decrease in warm shallow lakes may enhance the role of the microbial community at the expense of phytoplankton, likely reflecting higher density of submerged macrophytes; the effect will be less pronounced in the presence of fish.

Effect of changes in water level and nutrients: a cross-European mesocosm experiment

Data from the REFRESH mesocosms (Chapter 1325.1) were further analysed to elucidate changes in trophic interactions at contrasting water level and nutrients at a European scale. Non-metric multidimensional scaling demonstrated clear differences in zooplankton communities among the countries (Figure 70), and ANOSIM showed significant differences in zooplankton species composition, as well as a decrease in proportion of cladocerans and an increase in proportion of copepods from colder to warmer lakes (Figure 71).

The results revealed a significant decrease in genus richness with increasing temperature, while neither nutrients nor depth effects were significant. Although the overall genus diversity showed no clear pattern, the genus diversity of cladocerans and rotifers decreased with increasing temperature.

Taxonomic diversity was positively related to size diversity, but did not show significant differences among treatments, though size diversity tended to be slightly higher in the low nutrient mesocosms in most countries. The normalised size spectrum (NSS) slope significantly increased with temperature but did not differ among depths or nutrient levels. The slope of NSS against temperature was steeper in Greece than in the other countries. The NSS intercept also significantly increased with temperature and was on average highest in Greece.

We found that neither taxonomic nor size diversity showed any difference along the temperature gradient. Zooplankton richness, however, was lower in the warmer mesocosms where water loss was higher. Furthermore, the biomass of large-sized cladocerans such as *Daphnia* was notably lower in the warmer countries. A temperature increase would therefore entail a lower biomass of larger crustaceans, resulting in less effective grazing upon phytoplankton. In arid and semi-arid regions, frequency and magnitude of prolonged drought events is expected to increase as a result of global warming. Persistent drying may threaten the local survival of species in the food web.

7.2.4. Ecosystem services

Although effects ecosystem services were not the main focus of this report, we highlight some findings.

Nutrient retention by plants (i.e. nutrient loss from the water) is an important regulating ecosystem service of shallow lake ecosystems. The mesocosms experiment (Chapter 5.1) demonstrated an interaction between water level and temperature on the nutrient loss of total N and total P. In the shallow mesocosms warmer temperature had a positive effect on TN and TP loss, most likely related to macrophyte growth. However, at the high water levels, warmer temperature had a negative effect on TN and TP loss, indicating higher algal production of organic N and P in warmer systems causing higher organic N and P accumulation in the system.

The quality of perch as food for humans was analysed in relation to total phosphorus and dissolved organic carbon in Finnish lakes (Chapter 3.1). These two stressors changed the phytoplankton community structure and decrease production of essential omega-3 fatty acids in lakes. This influence was transferred through the food web: perch growing in oligotrophic clear-water lakes contain 1.5-1.9 times more essential omega-3 fatty acids than those grown in eutrophic and brown-water lakes. Thus, this type of provisioning ecosystem service was impaired by both eutrophication and brownification.

The case study of Lake Võrtsjärv (Chapter 4.4) was the first attempt to bring together a holistic picture of real and potential ecosystem services (provisioning, regulating and cultural) provided by a large shallow temperate lake, and a description of the main factors affecting the use of these ecosystem services. The environmental factors supporting ecosystem were not consistent with those needed for achieving good ecological status of lakes, but were rather linked to the resource gathering and encouraged for regulating the naturally fluctuating water level and maintaining the high trophic state beneficial for lake fisheries and reed growth. The poor availability of comparable data made the complex research difficult even for this well-studied lake, and indicates a more general gap in our knowledge about ecosystem services of lakes.

7.3. Key messages

Key messages

Ecological status of phytoplankton and macrophytes

- With increasing proportion of arable and pasture lands in lake catchments across Europe, total phosphorus concentration in lakes clearly increased and Secchi depth generally decreased. By contrast, the ecological status of lakes assessed using phytoplankton or macrophytes did not show a clear trend with increasing agricultural land use.
- Phytoplankton EQR (ecological quality ratio) decreased clearly with increasing Total P whereas macrophytes EQR showed a stronger relationship to Secchi depth (water transparency).
- Among natural parameters, lake altitude and mean depth had the strongest unique effect on both pressures and status assessments. Total P tended to be higher in lowland and shallow lakes. Transparency also increased with altitude and less agricultural lands within the catchment.
- As local biota can be considered best adapted to local climate, the climatic gradients across Europe should not affect the results of ecological status assessment. As our analysis showed, temperature had virtually no effect on the EQRs of phytoplankton and macrophytes. Still, as several metabolic rates in aquatic ecosystems have a strong temperature dependence, implying that also the sensitivity of ecosystems to stress may differ between climate zones.
- Despite the success of the WFD Intercalibration process in harmonising the classification methods among Member States, our analysis revealed country-specific differences in relationships between stressor levels and ecological status assessments that could likely be attributed to differences in established reference conditions and class boundaries.

Phytoplankton indices

- The combined effects of stressors (nutrients and temperature) on PTI varied considerably among the broad lake types. The siliceous lake types seem to more sensitive (in terms of PTI) to an increase in total P, and to the interaction between total P and air temperature.
- For lowland siliceous lakes, model predictions suggest that even if current TP concentrations would remain, warmer climate will increase PTI and may thereby reduce the ecological status of lakes in the long run (2090). If TP concentrations were 50% higher, temperature-induced increase in PTI could be expected also in the short run (2030). However, the temperature-induced change in PTI will probably not sufficient to reduce ecological status class of lakes.
- The analysis of individual lake time series indicates that cyanobacteria are most sensitive to nutrient stress in lakes of low nutrient status and sensitive to summer rainfall in short residence time lakes. The large-scale analysis suggests a synergistic interaction between temperature and nutrients, but this interaction was not prevalent.
- It was difficult to generalise the response of cyanobacteria to multiple nutrient and climatic stressors across lakes and lake types. These findings highlight that management of multiple stressors need to consider both the lake type and the stressor gradient of the individual lake.

Macrophyte indices

- Good relationship was found between the WIc index and the degree of water level for Finnish and Norwegian lakes. Clear lakes are more resistant to winter drawdown of water level than humic lakes due to better light climate; brownification of lakes due to climate change may therefore impair their resistance to winter draw down.
- The test of the Fenno-Scandian WIc-index on 48 Irish lakes subjected to different precipitation pattern and water use was unsuccessful and it is concluded that specific water level effect indices need to be developed across Europe, taken into account variation biogeography of macrophytes and precipitation pattern prevailing.
- Data from Irish lakes suggest that water level fluctuations of 2 m or less in deep lakes have limited impact on submerged macrophytes communities, but may affect reeds and marginal vegetation.
- For the Irish lakes a number of species and assemblages of species were, however, well related to a Seasonal Water Level index (SWL). Such relationships are encouraging and needs further development at a European scale.
- The cross-European mesocosm experiment indicates that global climate warming might favour growth of macrophytes at moderate water level decrease southern regions, even under relatively eutrophic conditions. However, if the water level decrease becomes so extreme that macrophytes are directly negatively affected, and longer and intense drought periods become more common, the combined effects of eutrophication and extreme water level reductions may adversely affect the development of macrophytes. In contrast, warmer temperatures in northern regions may hamper macrophyte growth due to increased precipitation and, thus increased water levels and nutrient loading.

Community structure and trophic interactions

- Climate factors such as temperature, as well as salinity and eutrophication, are key parameters determining the trophic structure and community composition of Turkish Mediterranean shallow lakes.
- The warmer southern lakes were more saline and eutrophic, had higher proportion of small fish, higher nutrient and Chl *a* concentrations and higher cyanobacteria dominance than the northern lakes, despite similar land-use effects. They also had lower species diversity of most organism groups. Strong top-down control of fish on zooplankton was also traced in southern lowland lakes. The Chl *a*:TP ratio, omnivorous fish biomass and the fish:zooplankton biomass ratio were higher, whereas the zooplankton:phytoplankton biomass ratio and macrophyte coverage were lower, suggesting a higher top-down control of fish on zooplankton.
- Climate warming together with expected changes in land use in Mediterranean lakes may result in higher salinisation and eutrophication with more frequent cyanobacteria blooms and loss of biodiversity. Consequently, under such conditions ecosystem services (e.g. drinking and irrigation water, biodiversity etc.) are likely to be deteriorated if not lost completely. To counteract, strict control of nutrients and human use of water is urgently needed.
- Zooplankton richness was lower in the warmer mesocosms where water loss was higher. Furthermore, the biomass of large-sized cladocerans such as *Daphnia* was notably lower

in the warmer countries. A temperature increase would therefore entail a lower biomass of larger crustaceans, resulting in less effective grazing upon phytoplankton. In arid and semi-arid regions, the frequency and magnitude of prolonged drought events is expected to increase as a result of global warming. Persistent drying may threaten the local survival of species in the food web.

- Water level decrease in warm shallow lakes may enhance the roles of the microbial community at the expense of phytoplankton, likely reflecting higher density of submerged macrophytes; the effect will be less pronounced in the presence of fish.

8. Appendices

List of appendices:

Appendix 1. (Chapter 1): Land cover data for lake catchments. MARS WP 5.1.1 Internal report. UL 31.05.2016.

Appendix 2. (Chapter 2.1): Structure and quality check of the compiled database of SOE lakes and WFD pressures.

Appendix 3. (Chapter 2.2): Statistical analysis of the WISER PTI data.

Appendix 4. (Chapter 3.1): (Taipale et al., 2016). Lake eutrophication and brownification downgrade availability and transfer of essential fatty acids for human consumption. Environment International 96: 156-166. Full text URL: http://dx.doi.org/10.1016/j.envint.2016.08.018.

Appendix 5. (Chapter 4.4): (Vilbaste et al., 2016). Ecosystem services of Lake Võrtsjärv under multiple stress: a case study. Hydrobiologia: 780:145-159. URL: http://dx.doi.org/10.1007/s10750-016-2871-y

Appendix 6. (Chapter 5.1): Four manuscripts based the REFRESH mesocosm experiments.
(1) (Coppens et al., 2016): Full text URL:
https://www.researchgate.net/publication/283285600_The_influence_of_nutrient_loading_climate_and_water_depth_on_nitrogen_and_phosphorus_loss_in_shallow_lakes_a_pan-_European_mesocosm_experiment
(2) (Ersoy et al., in prep.): abstract.
(3) (Tavşanoğlu et al., in revision): abstract.
(4) (Özen et al., 2014): Full text URL:
https://www.researchgate.net/publication/263659951_Water_level_and_fish-mediated_cascading_effects_on_the_microbial_community_in_eutrophic_warm_shallow_lakes_A_mesocosm_experiment

Appendix 7. (Chapter 6.1): Three manuscripts based on data from lakes in Turkey.
(1) (Beklioğlu et al., in prep.): abstract.
(2) (Erdoğan et al., in prep.): abstract.
(3) (Brucet et al., in review): abstract.

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Funded by the European Union within the
7th Framework Programme, Grant Agreement 603378.
Duration: February 1st, 2014 – January 31th, 2018

Deliverable 5.1: Reports on stressor classification and effects at the European scale: EU-wide multi-stressors classification and large scale causal analysis.

D5.1-5: New functional diversity indices allowing assessing vulnerability in abiotic multi-stressor context

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Due date of deliverable: **Month 36**
Actual submission date: **Month 36**

Dissemination Level	
PU	Public
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)

Content

Non-technical summary

A community hosted by an ecosystem composed of species sharing the same characteristics i.e. species showing the same response to the environment and/or species with the same impact on their environment, can be defined as a community with high functional redundancy. Such community is supposed to be less vulnerable to species loss and the ecosystem functioning is also supposed to be less impacted than when communities are composed of species with different functional characteristics.

In this work, we first described the fish communities of lakes, rivers and estuaries of France, Spain and Portugal using species richness and functional diversity. Functional diversity was a measure of the extent of complementary among species considering five characteristics previously define by different sources (literature, available database): fish size, vertical position in the water body, spawning habitat, trophic group, and swimming mode. For the three aquatic systems, the number of species and functional diversity was generally higher in northern and western France than in the Mediterranean areas; this geographical pattern was explained by historical events (recolonization after the last glacial period). Higher functional diversity found in estuaries compared to lakes and rivers was explained by the importance of the connectivity between adjacent environments.

Analysing correlations between functional redundancy and species richness, results suggest that higher taxonomic richness in freshwater ecosystems is likely to increase the stability and resilience of fish assemblages after environmental disturbance because of higher species redundancy whereas it is not the case in estuaries.

Studying the impact of species loss following different scenarios, we also demonstrated that, in rivers and estuaries, rare species support singular ecological functions not shared by dominant species. Our results suggest also that functional diversity of fish assemblages in rivers can be more affected by environmental disturbances than in lakes and estuaries.

Finally, using functional redundancy and taxonomic vulnerability, we proposed a composite index of functional vulnerability, minimised for highly redundant assemblages composed of species with low extinction risk. Fish communities of estuarine ecosystems appear less vulnerable to species loss in comparison with assemblages of lakes and rivers. Although these latter systems obtained comparable scores, the functional vulnerability was not influenced by the same component. Fish assemblages in lakes are often redundant but composed of a large part of vulnerable species, whereas river assemblages are in general poorly redundant but composed of species with low intrinsic vulnerability. This new score is proposed to be used in conservation perspective to define management priorities.

D 5.1.-5 - Reports on stressor classification and effects at the European scale: New functional diversity indices allowing assessing vulnerability in abiotic multi-stressor context

Functional redundancy and vulnerability of fish assemblages in rivers, lakes and estuarine systems

Introduction

The worldwide ecological footprint of human activities entailed rapid decline of biodiversity over the past decades (Butchart et al., 2010), which affects functions and services delivered by ecosystems (Cardinale et al., 2012). The concern about acceleration of species' extinction risk has led growing research initiatives in conservation science to evaluate the role of biodiversity in ecological resilience and stability of ecosystems (Mori et al., 2013). To this end, ecologists investigate the functional structure of communities through indices (functional richness, functional evenness, functional divergence) reflecting different components of the functional traits supported by coexisting species, gathered under the term of functional diversity (Barbault, 1995; Díaz and Cabido, 2001; Mason et al., 2005; Schleuter et al., 2010). The functional traits are directly linked to ecosystem processes (effect traits) and/or related to the performances of organisms in a changing environment (response traits) (Hooper et al., 2005). According to the insurance hypothesis of biodiversity, the maintenance of high functional diversity and redundancy increase the stability of biological communities and their associated ecological processes (Yachi and Loreau, 1999). The concept rests on evidences that the variability of responses to disturbances among species that share similar functions, i.e. portfolio effect (Figge, 2004), insures ecosystem recovery after disturbance by compensating the loss of functionally redundant species (Elmqvist et al., 2003). Preventing the decline of both response diversity and functional redundancy is thus a crucial issue in environmental management to ensure the long-term stability of ecosystems subjected to human-induced disturbance (Mori et al., 2013; Rosenfeld, 2002). However, the development of management strategies requires quantitative criteria to evaluate the functional vulnerability of biological systems and to determine conservation priorities in a context of limited resources (Mouillot et al., 2014; Parravicini et al., 2014).

Species respond in different ways against environmental disturbance, so that extinction or population decline at local, regional or global scales are not necessary random processes (Zavaleta et al., 2009). The species extinction risk is consistently related to intrinsic components driving specific response to disturbance, e.g. life-history traits, habitat requirement, population size (Olden et al., 2007), but also depends on extrinsic factors, such as the intensity of threats (Tracy and George, 1992). The combination of these factors influences the order of species' extinction within a community, and consequently leads to a predictable, i.e. non-random, pattern of functional diversity loss (Pimm et al., 2014). Due to this selective loss of vulnerable species in response to environmental disturbances, functional alterations can be more pronounced than non-selective random patterns of extinction (Rosenfeld, 2002). The shape of this relationship between species richness and functional diversity is a key component to evaluate the impact of species loss on ecosystem functioning (Petchey and Gaston, 2002a). It is expectedly linear when all species of a community support singular ecosystem functions (i.e. non-redundant species), meaning that the loss of any species will produce a sharp decrease in functional diversity (Micheli and Halpern, 2005). On the contrary, functionally redundant assemblages will display curvilinear relationships, i.e. saturation trends, as species differ in their contribution to functional diversity. In such cases, the subtraction of one species does not always cause functional alteration. This property can be used to evaluate the functional redundancy of assemblages and by extension the vulnerability of the ecosystem functioning, depending on the mechanisms of species extinctions i.e. randomly or not, if the first loss concerns species with particular traits for example (Fonseca and Ganade, 2001; Petchey and Gaston, 2002a).

Aquatic ecosystems largely contribute to maintaining overall environmental health and provide goods and services for human populations, such as aquatic resources for food or nutrient regulation (Martin-Ortega et al., 2015; Van den Belt and Costanza, 2012). However, these ecosystems are globally threatened by anthropogenic activities that cause habitat degradation, water pollution, resource overexploitation, or invasion of alien species, which result in a rapid decline of aquatic biodiversity (Halpern et al., 2008; Helfman, 2007; Vörösmarty et al., 2010). In addition, European aquatic ecosystems are often affected by complex mixtures of stressors (Hering et al., 2015; Schinegger et al., 2012). About 2,251 species (41%) of the 5,435 animals present in the 2000 Red list of the International Union for Conservation of Nature (IUCN) are living in aquatic environments (IUCN, 2011). Effective management and conservation strategies are thus required to maintain high level of biodiversity and ensure the long-term sustainability of ecosystem functioning, resilience, and delivered services (Abson et al., 2014; Bennett et al., 2015; Dudgeon et al., 2006). Because of their implication in food web dynamics, nutrient cycling, or redistribution of bottom sediment, fish communities may influence aquatic ecosystem processes (Holmlund and Hammer, 1999; McIntyre et al., 2007). Fish are mobile fauna and have a major impact on the distribution and abundance of other organisms through trophic and competitive interactions. The decline of fish diversity resulting from environmental disturbances can thus produce drastic changes in ecosystem functioning. Since several decades, monitoring programs conducted in aquatic ecosystems take advantage of the key role of fish communities to assess ecosystem health on the basis of quantitative indicators (Birk et al., 2012; Hering et al., 2006; Karr, 1981; Reyjol et al., 2014). Ecological indicators are often composed of several metrics describing specific components of the taxonomic and functional structure of fish assemblage, which are influenced by the intensity of anthropogenic threats (Borja and Dauer, 2008; Pont et al., 2007). Ecological assessments are thereafter used to define conservation and restoration priorities to prevent any further degradation of aquatic environment. However, the current methods do not provide quantitative information to estimate the vulnerability of fish assemblages and the impact of species loss on ecosystem functioning. To this aim, assessment of the functional relationship and complementarity between species is essential to guide conservation efforts toward preservation of the more vulnerable fish assemblages (Rosenfeld, 2002).

Hence, in this contribution, we investigated the consequence of species loss on functional diversity across three aquatic systems, i.e. lakes, rivers, and estuaries. Particularly, we described the geographical patterns within the functional structure of fish communities in France, Spain, and Portugal. The relationship between species and functional richness and, on the basis of scenarios of species loss, the functional redundancy of fish communities was then investigated. Finally, using this community characteristic and taxonomic vulnerability, we proposed a composite index of functional vulnerability, minimised for highly redundant assemblages composed of species with low extinction risk. The characteristics of this new index are discussed in a conservation perspective to define management priorities.

Methods

Available data

We used various data sources collected within monitoring programs related to the EU Water Framework Directive (WFD; 2000/60/EC) and other sources to obtain estimates of fish abundance in 49 estuaries, 302 lakes, and 1403 river reaches distributed throughout the southern Western Europe. The WFD requirements ensure the availability of relatively homogeneous fish dataset for each aquatic system in term of standardization of sampling efforts and fishing techniques (Birk et al., 2012; Perez-Dominguez et al., 2012). For estuarine systems, fish abundances were estimated on the basis of trawl surveys conducted between 2005 and 2013. Briefly, the protocol consists of performing several hauls distributed across the whole salinity gradient by using a beam trawl (from 1.5 to 3 m large, 8-20 mm mesh size). Trawling was performed against the current during 5 to 20 min, at a speed ranging from 1 to 3.5 knots. The number of hauls (from 5 to 71 hauls per estuary) was defined according to the system size to ensure the sampling representativeness. Abundances of all taxa were expressed in density by dividing the number of individuals by the sampled surface (number of individuals per 1000 m²).

For lakes, fish data were obtained in application of the Norden gillnet standardised protocol (C.E.N., 2005). Benthic multi-mesh gillnets (12 different panels with mesh sizes ranging between 5 mm to 55 mm, following a geometric series) and pelagic gillnets (11 different panels with mesh sizes ranging between 6.25 mm to 55 mm) were set in different depth strata during the summer period and the sampling effort (gillnet-nights) depended on lake depth and area. Nets were set before dusk and lifted after dawn in order to cover the activity peaks of all the fish species.

For rivers, fish data were extracted from an extensive database (EFI+Consortium, 2007) containing fish surveys conducted by several academic institutions and environmental agencies across Europe. Sites were sampled by electrofishing (wading) during low flow periods using European standards (C.E.N., 2003). To minimise the risk of false absences, we included only sites where fished areas were greater than $100m^2$ with more than 50 individuals caught. Abundances were expressed in number of individuals per m^2. Sites were associated with four fish assemblage types (FATs, i.e. Headwater streams, Medium gradient rivers, Lowland rivers and Mediterranean streams) based on fish community and environmental characteristics (Schinegger et al., 2013).

The fish assemblages were determined in term of species occurrence and abundance by gathering the available samplings of each system.

Fish functional traits

The functional niches of fish were described based on five complementary traits, which are commonly used in studies examining functional diversity in fish assemblages (Buisson et al., 2013; Eros et al., 2009; Guillemot et al., 2011; Mouillot et al., 2014; Parravicini et al., 2014; Pont et al., 2006; Pool et al., 2014): fish size, vertical position, spawning habitat, trophic group, and swimming mode. They reflect different ecological functions of species in ecosystems, focusing on key elements determining species habitat preferences and their position in the food web. They are considered as both effect and response traits because of their implication in ecosystem processes and their expected response to environmental disturbance (Table 1). We used coarse categorical traits, as the detail level of ecological information is highly heterogeneous between species and did not allow accounting for possible ontogenetic shifts in functional traits. Fish size corresponds to the maximum total length reported in literature and was coded using six ordered categories: 0-8, 8.1-15, 15.1-30, 30.1-50, 50.1-80, and >80.1 cm. Position in the water column characterizes the habitat usually used by fish for living and feeding, and was coded using two categories: benthic, and non-benthic. Species were assigned in seven trophic categories according to the dominant food item in the diet: herbivorous, omnivorous, piscivorous, planktivorous, parasitic, insectivorous, and detritivorous. Spawning habitat denotes the preference of species for specific reproductive conditions, and was coded using six categories: lithophilic, pelagophilic, phytophilic, polyphilic, nest builder, internal brooder. Swimming mode reflects the body-shape and swimming factor commonly used to describe locomotive performances of fish. It was coded using eight categories: carangiform, sub-carangiform, diodontiform, anguilliform, labriform, balistiform, amiiform, and rajiform. Information about the five functional traits were obtained from FishBase (Froese and Pauly,

2000) and consortium researches on fish assemblages in rivers (EFI+Consortium, 2009) and lakes (Caussé et al., 2011).

Table 1: Ecological relevance of the five categorical traits selected to describe the functional niches of fish in aquatic systems. The implication of traits in ecosystem functioning is detailed (as effect traits), as well as their expected responses to environmental disturbances (as response traits).

Trait	Effects on ecosystem process	Responses to disturbance	References
fish size	highly related to food-web structure, trophic levels, and energy flow in ecosystem	correlated with demographic performances of species and tolerance to environmental stress	(Costa, 2009; Emmerson and Raffaelli, 2004; Jennings et al., 2001; Logez and Pont, 2011; Wilson, 1975; Winemiller and Rose, 1992)
vertical position	influence species interactions and the set of available prey and bentho-pelagic energy flow	influenced by availability and quality of habitats along to the water column (e.g. sediments pollution, hydro-morphologic alteration)	(Bellwood et al., 2006; Vander Zanden and Vadeboncoeur, 2002)
spawning habitat	influence the redistribution of bottom sediment, competition for ground habitats and mobility of early stages	influenced by availability and quality of essential habitats for reproduction (e.g. clogging, habitat loss)	(Ciannelli et al., 2015; Holmlund and Hammer, 1999)
trophic group	drive the trophic interactions with other ecosystem components and regulate food web dynamic	influenced by resource availability (e.g. depletion of primary diet)	(Power, 1990; Vander Zanden et al., 1999)
swimming mode	influence mobility, food acquisition and ability to escape from predation	influenced by availability and quality of habitats (e.g. hydro-morphologic alteration)	(Helfman et al., 2009; Sfakiotakis et al., 1999)

Functional diversity

The functional diversity (FD) of fish assemblages was described using a dendrogram-based measure (Petchey and Gaston, 2002b), which reflects the richness component of functional diversity (Mouchet et al., 2010). This index captures the extent of complementary among species of a local community by measuring the total branch length of a dendrogram summarizing the functional distances between species in trait space (Petchey and Gaston, 2006). Such measure of functional richness was preferred to indices calculated from a multidimensional functional space (Laliberte and Legendre, 2010; Villeger et al., 2008) because calculations can be achieved for poor-species assemblages, e.g. less than four species, without substantial loss of information. Although functional diversity is influenced by species richness, it does not increase by the addition of a species functionally identical to a species already in the set (Ricotta, 2005). Larger values of functional diversity therefore indicate greater functional differences between species within assemblages. We calculated a distance matrix between all pairs of species using the Gower dissimilarity index (Gower, 1971), which handle categorical and ordered variables (Pavoine et al., 2009). An Unweighted Pair Group Method with Arithmetic Mean (UPGMA) clustering algorithm was then applied to produce a functional dendrogram, as it provided the highest cophenetic correlation coefficient with the initial distance matrix (c = 0.74) compared to others clustering methods (Mouchet et al., 2008).

The relationship between species richness and functional diversity was examined using multiple linear regressions for each aquatic system. We tested for the saturation effect of the relationship by integrating a quadratic term of species richness as predictor variable. Significant outcome indicates redundancy in contributions of species to functional diversity. The robustness of our results to the exclusion of one trait (do we observe the same relationship when considering only 4 traits in place of five?) was assessed by calculating the functional richness of all combinations of four out of five traits. We evaluated the influence of traits combinations on the slope of the relationship between species richness and functional diversity on the basis of an analysis of covariance (ANCOVA).

Functional redundancy

The functional sensitivity to species loss is closely related to the proportion of species that support singular ecological functions, so that functionally redundant assemblages are less sensitive to taxonomic erosion (Mouillot et al., 2014; Parravicini et al., 2014). Hence, the amount of functional redundancy within fish assemblages was assessed by investigating the relationship between species richness and functional diversity on the basis of scenarios of species extinction (Petchey and Gaston, 2002a). The effect of species loss on functional diversity depends on the assemblage composition and the order in which species are lost. Therefore, we simulated five extinction scenarios that differed in their trajectories: (i) a random scenario, (ii) a best-case scenario, (iii) a worst-case scenario, (iv) an abundance-based scenario, and (v) a trait-based scenario. Simulation of random extinctions consisted to remove species sequentially in a random order and re-calculate the FD index after each species lost. The extirpation process assumed an equal probability of extinction between species and was conducted until functional diversity was equal to zero, i.e. only one species remaining. We simulated 999 random trajectories for each assemblage. The best- and worst-case scenarios reflect extreme trajectories where the order of species extinction minimise or maximise the loss of functional diversity at each species loss. In the abundance-based scenario, species extinction occurs in inverse order of their abundance within assemblages, reflecting a plausible scenario where rare species disappeared first. In the trait-based scenario, the extinction order was defined by the intrinsic vulnerability score (range from 0 to 100) proposed by Cheung et al. (2005) and available from FishBase (Froese and Pauly, 2000). This composite score is calculated on the basis of life-history traits and ecological attributes that influence the resilience abilities of local populations (Cheung et al., 2007). Contrary to the IUCN assessment criteria, the scoring process does not consider the abundance and distribution range of species but reflect their intrinsic recovery abilities when facing threats, which is relevant in large-scale studies (Strona, 2014).

The impact of species loss on functional diversity was assessed for each trajectory by calculating the Area Under the Curve (AUC) defined by the residual proportion of functional diversity against the proportion of species lost (Fig. 1). Lower AUC values denote high functional sensitivity to species loss due to the limited redundancy of fish assemblage or the early loss of singular species along the extinction trajectory. On the contrary, higher AUC values suggest greater functional redundancy and trajectories where redundant species tend to disappear first. The AUC values obtained from the abundance- and trait-based scenarios were compared to those calculated from random trajectories to determine whether the functional diversity was more strongly affected by these plausible scenarios than by random extinctions. To this end, we calculated standardised effect size (SES) according to the formula,

$SES = (Obs – Mean_{rand})/sd_{rand}$,

where Obs was the AUC value of the directional scenario, $Mean_{rand}$ and sd_{rand} were respectively the mean and standard deviation of AUC values obtained from the 999 random trajectories (Gotelli and McCabe, 2002).

Negative SES AUC values indicated communities where functional diversity was affected more strongly by the impact of directional scenarios than random expectation. For each aquatic system, we tested whether the SES AUC values of directional scenarios were significantly lower than zero based on unilateral Wilcoxon rank tests. Finally, the AUC values derived from the random trajectories were averaged to obtain an index of functional redundancy, which reflects the overall compensatory potential of fish assemblages.

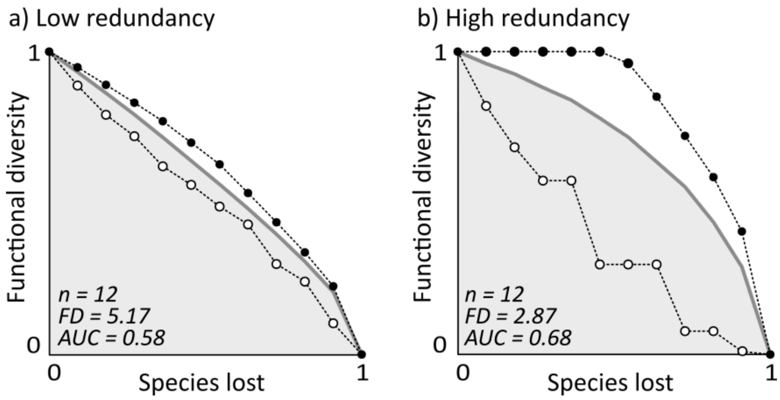

Figure 1: Effect of species loss on functional diversity (FD) simulated from extinction scenarios across two river fish assemblages composed of twelve species with a) poorly redundant functional traits, and b) highly redundant functional traits. The solid grey lines represent the mean of 999 random trajectories of species extinction, which are comprised between the dashed lines representing the best-case (full circle) and worst-case (open circle) scenarios. The grey sections represent the Area Under the Curve (AUC) of the mean random trajectories.

Assemblage vulnerability

In the present study, we proposed a composite index of functional vulnerability considering the taxonomic vulnerability to species loss and the compensatory potential of fish assemblages (Fig. 2). A multi-criteria decision analysis method was used to express the assumption that vulnerability is minimised for highly redundant assemblages composed of species with low extinction risk. The Technique for Order Preference by Similarity to Ideal Solution –TOPSIS- (Shih et al., 2007), was used to provide a vulnerability score for each assemblage on the basis of the relative distance to a positive ideal solution (Parravicini et al., 2014). The ideal solution was defined as an assemblage showing the maximum value of functional redundancy (i.e. mean AUC values of random trajectories) and the minimum proportion of vulnerable species. We used the intrinsic vulnerability score extracted from FishBase (Froese and Pauly, 2000). This score reflects the propensity of species to become locally extinct after environmental disturbances (Cheung et al., 2005). The proportion of vulnerable species was thus determined from the number of species within an assemblage classified from 'high' to 'very high' categories of vulnerability (i.e. vulnerability score over 50). The criteria values were standardized using a linear scale transformation before performing the TOPSIS analysis. The vulnerability scores range from 0 to 1, where higher values denote communities composed of vulnerable and non-redundant species. Differences in taxonomic sensitivity and functional vulnerability between the three aquatic systems, i.e. lakes, rivers, and estuaries, were tested using non-parametric Kruskal-Wallis rank sum test, and completed by Nemenyi-test for calculating pairwise multiple comparisons.

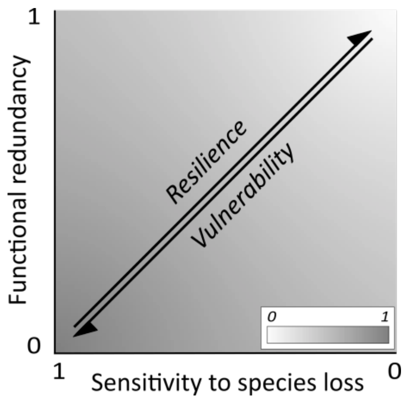

Figure 2: Conceptual scheme expressing the assemblage vulnerability as a composite result of both, taxonomic sensitivity to species loss and functional redundancy. Low proportion of sensitive species and high compensatory potential in fish assemblages decrease vulnerability and provide resilience for ecosystems reorganization following disturbances. The grey gradient represents the vulnerability score ranging between 0 and 1.

Results

Species and functional richness

We described the functional traits of 295 species from 63 families occurring in the studied aquatic systems. The species richness was overall higher among fish assemblages of estuaries (mean = 26.7; range = 4-57) than those of rivers (mean = 8.1; range = 3-31) and lakes (mean = 8.9; range = 3-19). For the three aquatic systems, the number of species was generally higher in northern and western France (Fig. 3). Species richness decreased across fish communities located in the Mediterranean region and south of the Pyrenees Mountains, i.e. Spain and Portugal. Overall, functional richness followed similar geographical patterns than species richness, but tended to increase in southern Portugal (Fig. 3).

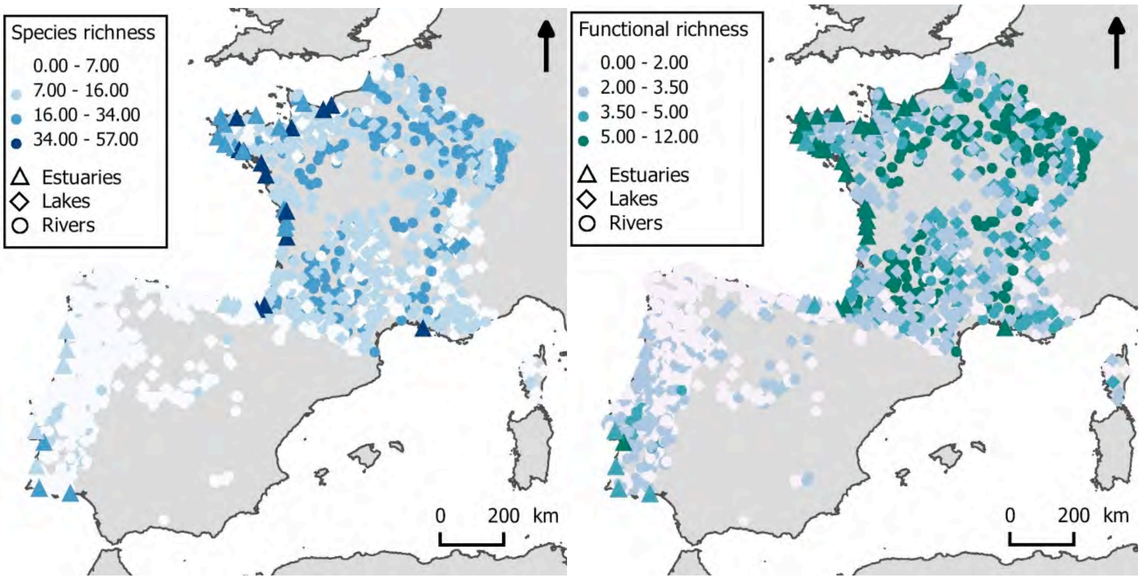

Figure 3: Species richness (on the right) and functional richness (on the left) of fish communities across lakes, rivers and estuaries in France, Spain and Portugal.

The geographical pattern of species and functional richness in rivers reflected differences among the sampled FAT's (Fig. 4 a and b). Sites in Headwater streams (HWS), essentially located in the Northern and Central System of the Iberian Peninsula, and sites in Mediterranean Streams (MES), concentrated on the western Iberian Peninsula (Portugal), showed the lowest species and functional richness values (richness, HWS, mean = 4.4 ; richness, MES, mean = 5.1; functional richness, HWS, mean = 1.6; functional richness, MES, mean = 2.0). Sites located in Medium Gradient Rivers (MGR) showed a high variability in species and functional richness (richness, mean = 12.2; functional richness, mean = 3.9) and sites in Lowland Rivers (LLR) showed the highest values in these two parameters (richness, mean = 15.3; functional richness, mean = 4.8).

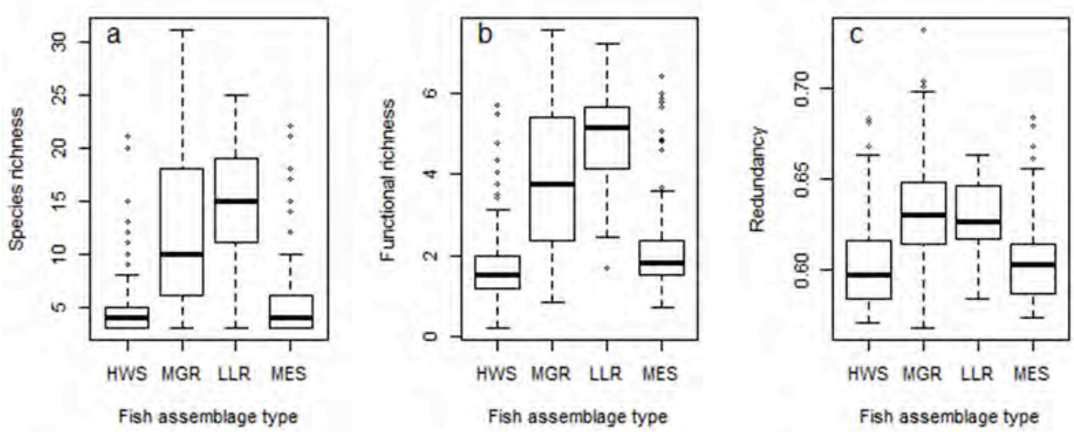

Figure 4: Differences among river types in species richness (a), functional richness (b) and redundancy (c). HWS = Headwater Streams, MGR = Medium Gradient Rivers, LLR = Lowland Rivers, MES = Mediterranean Streams.

The functional richness was closely related to species richness of fish communities in lakes, rivers and estuaries (Fig. 5). The quadratic terms of species richness integrated in regression models were significant for lakes and rivers ($P < 0.001$), which indicate a saturation effect in these relationships (lakes, $r^2 = 0.77$; rivers, $r^2 = 0.94$). On the other hand, the effect of the quadratic term was not significant for estuaries ($P = 0.42$) revealing that the linear model was a more parsimonious way to describe the relationship ($r^2 = 0.93$). The amount of variation explained by the relationship remained high and comparable for the different combination of functional traits used to test the robustness of results in lakes (r^2 ranged between 0.61 and 0.77), rivers (r^2 ranged between 0.89 and 0.94), and estuaries (r^2 ranged between 0.90 and 0.93). For the three systems, the slope of the relationship between species richness and functional diversity was significantly influenced by the different combinations of traits (ANCOVA, all $P < 0.001$). Nevertheless, the percentage of variation explained by the interactive term between species richness and the combination of traits was relatively low for each system (lakes, 2.9%; rivers, 3.6%; estuaries, 2.1%). These results suggest a minor effect of the combination of functional traits chosen to describe the functional diversity of fish assemblages.

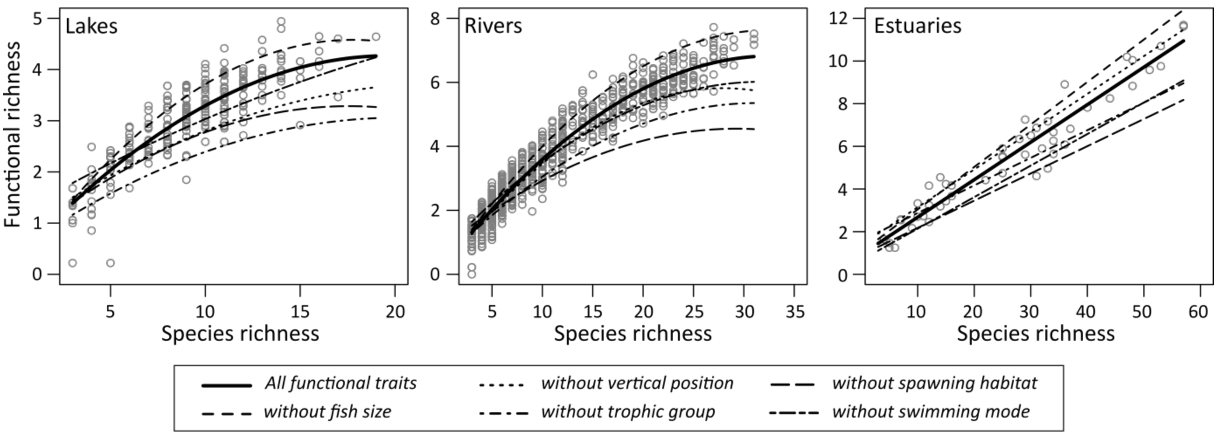

Figure 5: Relationship between species richness and functional diversity across fish assemblages of lakes, rivers and estuaries. The grey points represent the results obtained from the combination of the five functional traits, i.e. fish size, vertical position, trophic group, spawning habitat, and swimming mode. The solid lines show the modelled relationships using all the five traits and the dashed lines represent the relationships obtained from the different combinations of four traits.

Functional redundancy and extinction scenarios

The mean of AUC values calculated from the random extirpation trajectories was used to assess the functional redundancy of fish communities. Redundancy values were higher for estuarine assemblages (mean = 0.65, range = 0.59-0.71) than for those of lakes (mean = 0.63, range = 0.57-0.69) and rivers (mean = 0.61, range = 0.56-0.73), but differences are low. Overall, the functional redundancy of fish communities was higher across lakes and rivers located in northern France than aquatic systems of Spain and Portugal (Fig. 6). Nevertheless, functional redundancy tended to increase from northern Spain to southern Portugal. This pattern is reflected by the differences found across the fish assemblage types, with HWS and MES showing much lower functional redundancy than MGR and LLR (Fig. 4). No clear geographical pattern of redundancy was highlighted for fish assemblages in estuaries.

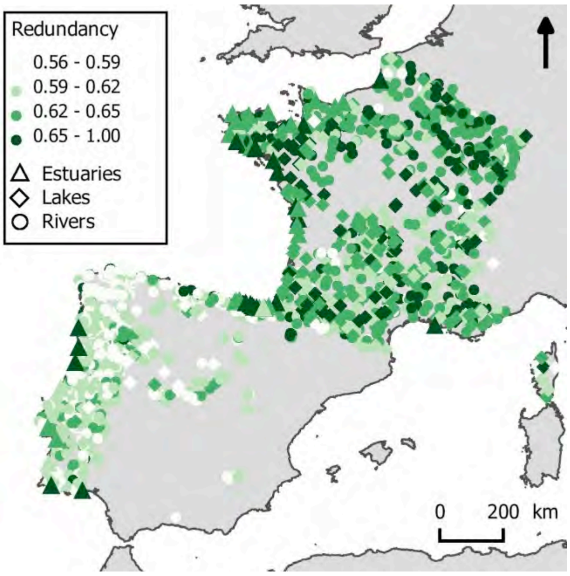

Figure 6: Functional redundancy of fish communities across lakes, rivers and estuaries in France, Spain and Portugal.

Functional redundancy was positively correlated with the species richness of fish communities in lakes and rivers (lake, r = 0.60, *P* < 0.001; river, r = 0.50, *P* < 0.001), but it was not influenced by the number of species in estuaries (r = 0.09, *P* = 0.51; Fig. 6).

Similar trends were highlighted for AUC values calculated from the best-case scenarios (lake, r = 0.67, *P* < 0.001; river, r = 0.58, *P* < 0.001; estuaries, r = 0.20, *P* = 0.16), whereas the worst-case scenarios were uncorrelated with species richness of assemblages in river and estuaries (rivers, r = -0.04, *P* = 0.07; estuaries, r = 0.08, *P* = 0.55) and showed a slight negative correlation for lakes (lake, r = -0.12, *P* = 0.03).

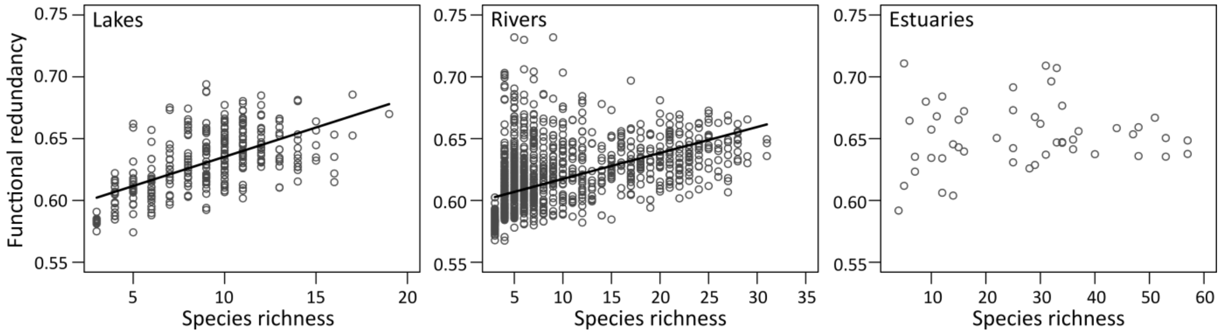

Figure 6: Relationship between species richness and functional redundancy across fish assemblages of lakes, rivers and estuaries. The functional redundancy of each assemblage was assessed from 999 random trajectories of species extinctions by calculating the average Area Under the Curve (AUC) defined by the proportion of species lost against the residual proportion of functional diversity. The solid lines show the significant linear relationships between species richness and functional redundancy using a combination of five traits, i.e. fish size, vertical position, trophic group, spawning habitat, and swimming mode.

In rivers, the functional richness was significant positively correlated with species richness for all FAT (Pearson correlation, P < 0.05; Fig. 7). This relationship was stronger in HWS and MES. For these FATs, the distribution of species richness along sites was very left skewed, where a few sites with high species richness strongly influenced the slope of the linear relationship (Fig. 7).

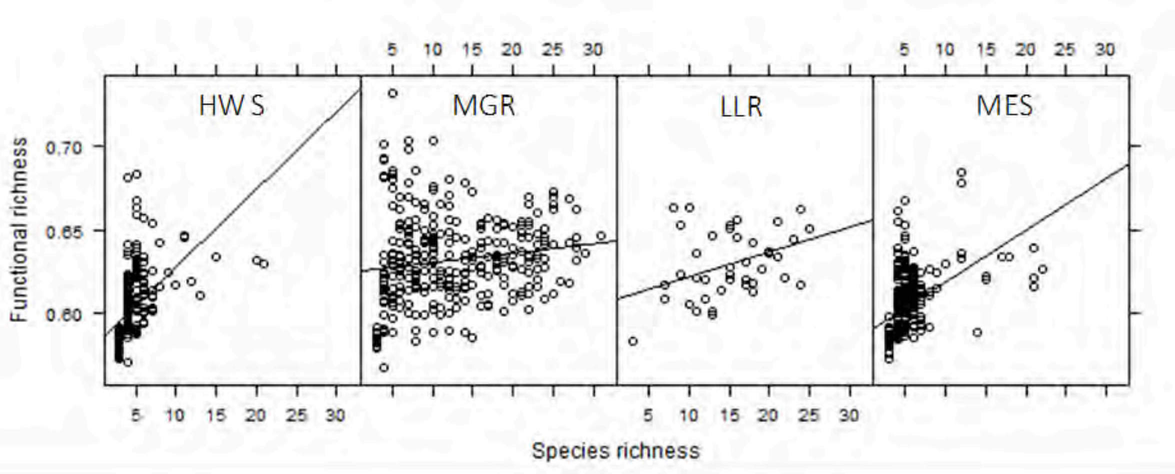

Figure 7: Relationship between species richness and functional redundancy across Fish Assemblage Types in rivers.

The standardised effect size of AUC obtained from the abundance-based scenarios were significantly lower than zero for river and estuarine communities (all P < 0.001), which reflects a substantial contribution of non-abundant species to the functional diversity of these systems (Fig. 8). The trait-based scenario simulated from the river assemblages caused greater functional alterations than random patterns of species extinction ($P < 0.001$; Fig. 8), whereas the standardised effect size of AUC were not significantly lower than zero for lake and estuarine communities (lakes, $P = 0.99$; estuaries, $P = 0.98$).

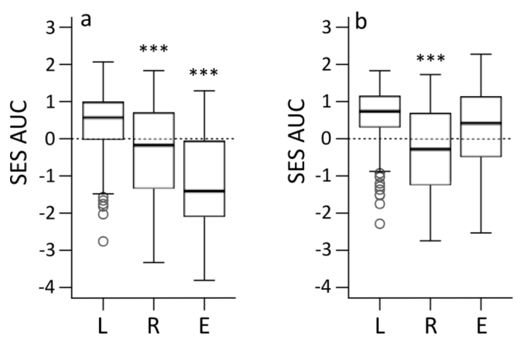

Figure 8: Standardised effect size (SES) of Area Under the Curve (AUC) calculated from a) the abundance-based and b) the trait-based scenarios of species extinction within fish assemblages of lakes (L), rivers (R), and estuaries (E). Negative SES AUC values indicate communities in which the functional diversity is affected more strongly by directional scenarios compared to random trajectories of species extinction. The asterisks designate the aquatic systems in which SES values were significantly lesser than zero based on Wilcoxon signed rank tests.

For rivers, the standardised effect size of AUC obtained from both the abundance-based and trait-based scenarios (Fig. 9) were significantly lower than zero across all FAT (Wilcoxon signed rank tests, $P < 0.001$, except for HWS and MGR in the trait-based scenario, with $P = 0.03$ and $P = 0.002$, respectively), i.e., causing greater functional alterations than random patterns of species extinction. Still, the SES of AUC in the case of LLR and MES differed more markedly from zero in comparison to HWS and MGR (Fig. 9).

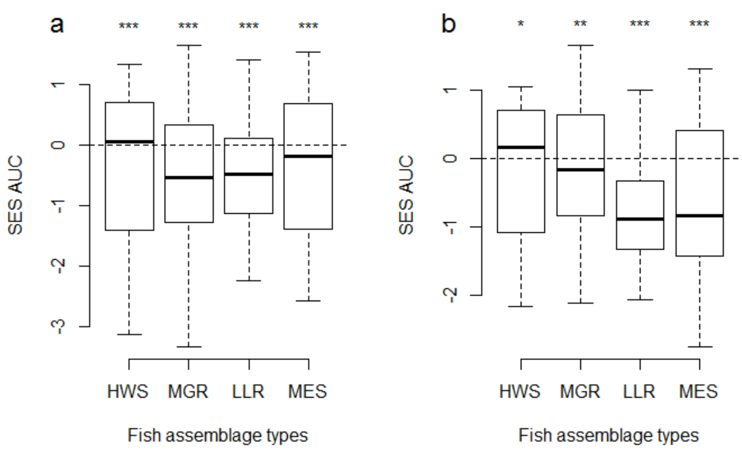

Figure 9: Standardised effect size (SES) of Area Under the Curve (AUC) calculated from a) the abundance-based and b) the trait-based scenarios of species extinction across river fish assemblages

*types. * - P < 0.05; ** - P < 0.01; *** - P < 0.001. HWS = Headwater Streams, MGR = Medium Gradient Rivers, LLR = Lowland Rivers, MES = Mediterranean Streams.*

Vulnerable species

The proportion of vulnerable species (i.e. categories 'high' to 'very high' of intrinsic vulnerability) varied significantly between the three aquatic systems ($P < 0.001$). The taxonomic vulnerability of fish assemblages was higher in lakes (mean = 0.52, range = 0-1) than in rivers (mean = 0.43, range = 0-1) and estuaries (mean = 0.18, range = 0.0-0.4). The highest proportion of vulnerable species among fish assemblages of rivers was observed in northern Spain and Portugal (Fig. 8). Conversely, taxonomic vulnerability tended to be higher in fish communities of lakes located in France than those of Spain and Portugal. No clear geographical pattern of taxonomic vulnerability was highlighted for fish assemblages of estuaries.

Functional vulnerability

The scores of functional vulnerability were significantly lower for estuarine communities ($P < 0.001$) since they are composed of a higher proportion of redundant and non-vulnerable species in comparison with fish assemblages of lakes and rivers (Fig. 10). Despite a slight significant difference ($P = 0.011$), the vulnerability scores estimated for lakes and river were almost comparable, but vulnerability was mainly driven by taxonomic sensitivity for lake communities while estimates of rivers were mainly due to low functional redundancy (Fig. 11). No clear geographical pattern of functional vulnerability was observed for fish assemblages in lakes and estuaries. On the contrary, the functional vulnerability of river fish assemblages tended to increase in the southwestern France, and was maximal in the north-western part of Spain and Portugal (Fig. 10).

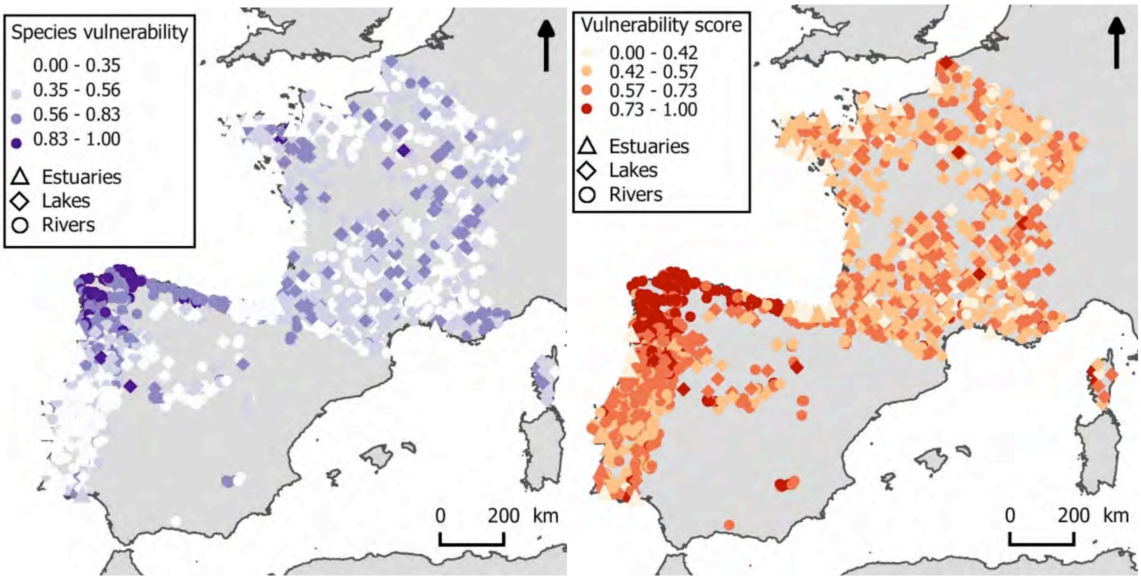

Figure 10: The proportion of vulnerable species (on the right) and functional vulnerability (on the left) of fish communities across lakes, rivers and estuaries in France, Spain and Portugal. Vulnerable species were determined on the basis of the intrinsic score of vulnerability, i.e. from 'high' to 'very high' categories of vulnerability. The functional vulnerability is a composite score which is minimised for highly redundant assemblages composed of species with low extinction risk.

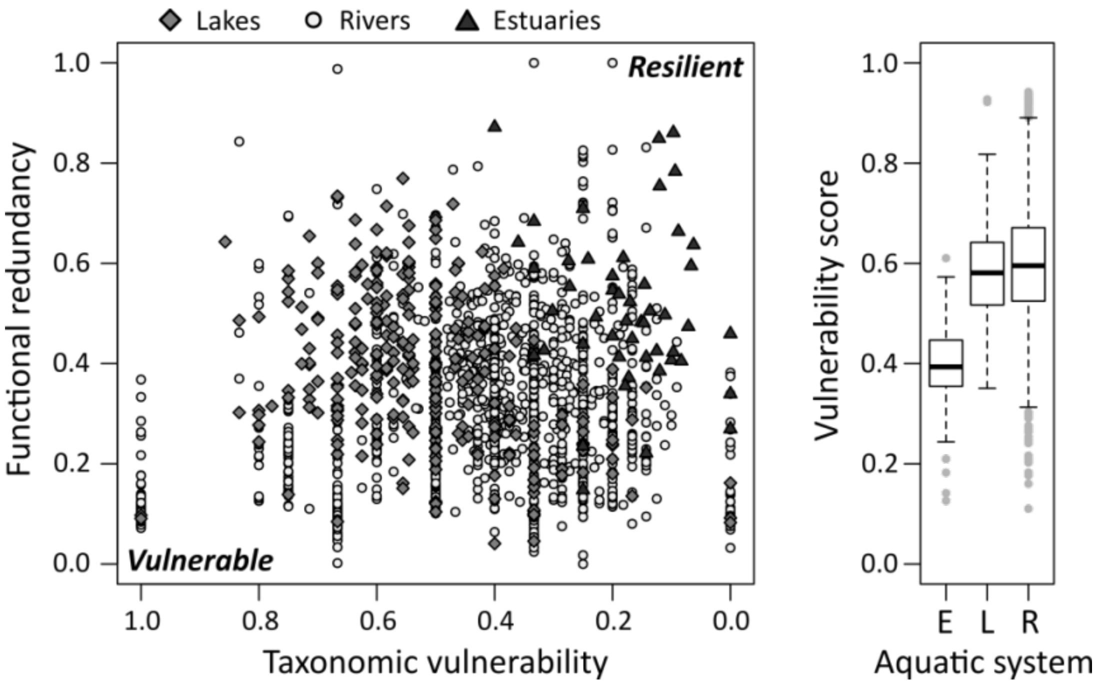

Figure 11: Vulnerability gradient of fish assemblages expressed as function of taxonomic vulnerability and functional redundancy for lakes, rivers, and estuaries. Fish assemblages located at the bottom left of the graph (left) are assumed most vulnerable when facing threats due to high proportion of sensitive species and low compensatory potential. The boxplots (right) show the vulnerability score estimated for fish communities of the three aquatic systems, i.e. estuary (E), lake (L), and river (R).

Discussion

Species extinction is a crucial concern in conservation science, especially regarding the consequences of biodiversity loss on ecosystem stability and functioning (Connolly et al., 2013; Loreau et al., 2001). Most previous studies reported a positive relationship between species richness and functional diversity in natural assemblages, which suggests that species extinction is often associated with severe decline in ecological functions (Bihn et al., 2010; Boyer and Jetz, 2014; Petchey and Gaston, 2002a). We also found a strong positive link between species and functional richness among fish communities of the three aquatic systems. Species richness and functional diversity of estuarine communities were commonly higher than those of rivers and lakes. Both factors were decreasing gradually from estuaries to lakes, due to different degrees of connectivity within the three adjacent systems (much higher in estuaries with contiguous freshwater and coastal assemblages). This reflects also the importance of connectivity among these systems in maintaining functional diversity (USEPA, 2015) and in recovering from degraded situations (Verdonschot et al., 2013).

Functional redundancy in aquatic systems

Estuarine communities are composed of marine, estuarine, diadromous and freshwater species that occupy various ecological niches and spread over a large range of ecological gradients, e.g. salinity (Potter et al., 2015). In estuaries, the functional richness was linearly related to the number of species without evidence of a saturation effect, which indicates a proportional contribution of supplemental species to functional diversity. The non-significant relationship between species richness and functional redundancy among estuarine communities was confirmed by our index of functional redundancy. This result emphasizes that species-rich assemblages of estuaries are not necessarily more functionally redundant than species-poor assemblages. Such pattern was already observed in diversified communities of invertebrates in tropical forests (Bihn et al., 2010) or coastal assemblages of temperate areas (Micheli and Halpern, 2005). Similarly, Guillemot et al. (2011) demonstrated that the functional-species relationship of coral reef fish communities displayed a non-asymptotic curve, so that singular species and functions commonly occur in species-rich marine ecosystems (Micheli et al., 2014).

In Europe, in comparison to other regions of the globe, fish of freshwater ecosystems are relatively species-poor due to the intensity of the last glacial period and patterns of species richness vary greatly across biogeographical regions (Reyjol et al., 2007). For lakes and rivers, we found a positive correlation between species richness and functional redundancy, which was also reflected by a saturation effect in the functional-species curve. These findings suggest that higher taxonomic richness in freshwater ecosystems is likely to increase the stability and resilience of fish assemblages after environmental disturbance because of higher species redundancy. The high level of functional redundancy highlighted for the most species-rich communities can provide resilience against the loss of ecological function by means of a compensatory effect. Under this assumption, the decline of sensitive species can be compensated by the increase of less sensitive species that have similar ecological roles (Naeem, 1998). Nevertheless, we have to note the negative relationship between functional redundancy and species richness of lakes shown in the worst-case scenarios. This suggests that, in lakes, even if the compensatory potential tends to increase with the number of species, some combinations of functional traits are supported by non-redundant species, so that their loss among first positions induces a large impact on functional diversity.

Although functional redundancy tends to increase with species diversity in freshwater ecosystems, redundancy levels were highly variable among fish assemblages composed of the same number of species. This result supports the assumption that different assemblage processes can operate to shape fish communities in response to biotic and abiotic environmental conditions (França et al., 2011; Nicolas et al., 2010). Indices of functional redundancy reflecting the amount of species complementarity within assemblages is thus essential for management purposes, namely to assess the impact of biodiversity decline on functional diversity at a local scale.

Impact of species loss on functional diversity

The impact of species loss on functional diversity depends on the amount of trait similarity between species in a community, but it is also influenced by the order in which species are lost. Random extinction is a standard assumption to provide a conservative estimate of functional vulnerability, but extinction scenarios mediated by specific ecological features can affect the rate of functional diversity loss (Fonseca and Ganade, 2001; Petchey and Gaston, 2002a).

Our abundance-based scenarios showed a decrease in functional richness when non-abundant species were early removed from river and estuarine communities. These results demonstrate that rare species of these ecosystems support singular ecological functions, which were not shared by dominant species. Similarly, Mouillot et al. (2013) showed that rare species among communities of coral reef fishes, alpine plants, and tropical trees, frequently support vulnerable functions and significantly increase the level of functional diversity. Extinction of rare species can produce an important alteration of ecosystem functioning since they promote different functions and ecosystem services, which in turn sustains local ecosystem properties (Isbell et al., 2011). Considering this general tendency for rare species to support specific ecological functions, their lower importance in lakes is surprising. We can therefore suspect a bias due to the selective and passive sampling protocol providing fish samples composed of (the most abundant species) ou (species less rare than in the two others waterbody types). We can therefore suspect a bias of the fishing method specifically undersampling rare species. A comparison of species occurences in the different systems would allow to verufy this hypothesis.

Because rare species support high diversity of trait values, they also can play a major role in ecosystem stability and recovery when environmental changes occur through compensatory dynamics (Elmqvist et al., 2003; Flöder et al., 2010). In effect, this is the reason why rare species are very often key elements in indices to assess the biological integrity of aquatic systems (Wan et al., 2010). Conservation strategies should thus consider the importance of non-abundant species and prevent their extinction to ensure long-term stability and functioning of river and estuarine ecosystems.

We also considered a trait-based scenario where the extinction order was defined by the intrinsic vulnerability score of species (Cheung et al., 2005), which reflects an overall propensity of species to become locally extinct after environmental disturbance. This scenario caused greater functional alteration in river communities, especially in Lowland Rivers and Mediterranean Streams, but was not more hurtful than random patterns of extinction across fish assemblages of lakes and estuaries. In other words, this result suggests that functional diversity of fish assemblages in rivers can be more affected by environmental disturbances causing species loss than in lakes and estuaries.

The intrinsic score of species vulnerability is measured on the basis of demographic traits related to extinction risk, such as growth rate, life span or maturity age, but does not consider the species response to specific threats. Further studies should investigate the impact of specific environmental disturbances on functional diversity by simulating species loss according to their sensitivity when facing particular threats. For example, hydro-morphological or water quality alterations are widespread in aquatic systems and could result in a selective loss of functional diversity when sensitive species support similar ecological functions. Identifying the impact of specific threats on functional diversity is thus a curtail challenge in management activities for predicting shift in ecosystem functioning.

Functional vulnerability of fish assemblages

A recent study demonstrated a global mismatch between species richness and vulnerability of fish assemblages on tropical reefs (Parravicini et al., 2014). This finding suggests that species-poor assemblages require more management efforts to prevent the loss of ecological function, due to their low functional redundancy (Boyer and Jetz, 2014). Similar patterns can be observed for fish communities in lakes and rivers, but our results demonstrated that functional redundancy is not always related to species richness among aquatic ecosystems. For example, the species-poor assemblages of small estuaries located in northern Spain are often more functionally redundant and less vulnerable than some species-rich assemblages of large estuarine systems, such as Gironde or Loire estuaries in France. The measure of species richness is thus insufficient to reflect the functional diversity and redundancy of fish assemblages in order to assess their vulnerability when facing environmental disturbances. In this study, the functional vulnerability of fish assemblages was assessed by a composite measure integrating two components, i.e. the taxonomic vulnerability which reflects the propensity for species extinction within assemblage and the functional redundancy which reflects the ability to compensate species loss. Overall, our results highlighted that fish assemblages in estuaries were more redundant than in rivers and lakes, and mainly composed of species with low intrinsic vulnerability. As a result, fish communities of estuarine ecosystems appear less vulnerable to species loss in comparison with assemblages of lakes and rivers. Although these latter systems obtained comparable scores, the functional vulnerability was not influenced by the same component. Fish assemblages in lakes are often redundant but composed of a large part of vulnerable species, whereas river assemblages are in general poorly redundant but composed of species with low intrinsic vulnerability. Beyond this average trend, a large variability was emphasized between fish assemblages and revealed distinct geographical patterns of vulnerability, particularly for river communities, possibly related with different fish compositions in different river types. Among river fish assemblages, a hotspot of functional vulnerability was highlighted in northwestern Spain, where rivers are largely inhabited by non-redundant and vulnerable species, such as *Pseudochondrostoma duriense* or *Salmo trutta fario*. It is known that this area, among others in Europe, has been described as a glacial refuge for some species, either in coastal waters (Campo et al., 2010), transitional (Finnegan et al., 2013) or in continental systems (Schmitt, 2007), which can explain this differentiation. In terms of species decline, the identification of vulnerable communities and habitats is a primary concern, especially for prioritizing conservation efforts to preserve ecosystem function (Franca et al., 2012).

Management implications

Resources allocated for protecting biodiversity are often limited, so that environmental managers have to define priority areas for conservation investment (Wilson et al., 2006). Species richness and endemic species richness are often used to define conservation strategies, but previous studies and our findings underlined the weakness of these taxonomic diversity criteria to assess the impact of threats on ecosystem functioning (Cardinale et al., 2012; Mouillot et al., 2014). When facing environmental disturbances, our index of vulnerability aims to assess propensity of assemblages to functional alteration caused by species loss. It does not consider the biodiversity value of communities, and thus provides complementary information for conservation purpose by focusing on ecosystem processes. Our findings can be used to prevent alteration or shift in ecosystem functioning by taking measures of protection for preserving the more vulnerable assemblages. Parravicini et al. (2014) highlighted the importance of coupling measures of functional vulnerability with the distribution of potential threats in order to identify vulnerable communities impacted by human-induced disturbance. In this framework, allocations in conservation effort can be ordered in favor of vulnerable communities subject to high intensity of threats.

Key messages

- For rivers, lakes and estuaries, species richness and functional diversity of fish communities were generally higher in northern and western France than in the Mediterranean areas (Spain and Portugal). This geographical pattern was explained by historical events (recolonization after the last glacial period).

- Higher functional diversity was found in estuaries compared to lakes and rivers, probably due to the importance of the connectivity between adjacent environments and habitat heterogeneity.

- Higher taxonomic richness measured in freshwater ecosystems is likely to increase the stability and resilience of fish assemblages after environmental disturbance because of higher species redundancy whereas it is not the case in estuaries.

- Studying the impact of species loss following different scenarios, we demonstrated that, in rivers and estuaries, rare species support singular ecological functions not shared by dominant species. Our results suggest also that functional diversity of fish assemblages in rivers can be more affected by environmental disturbances than in lakes and estuaries.

- Using functional redundancy and taxonomic vulnerability, we proposed a composite index of functional vulnerability that can be used for management purposes in conservation perspective. This index is minimised for highly redundant assemblages composed of species with low extinction risk.

- This index allowed to show that:
    - Fish communities of estuarine ecosystems were less vulnerable to species loss in comparison with assemblages of lakes and rivers.
    - Fish assemblages in lakes are often redundant but composed of a large part of vulnerable species,
    - River fish assemblages were in general poorly redundant but composed of species with low intrinsic vulnerability.

- From a management perspective, the effort should be put into maintaining the functional diversity of estuarine fish, the natural richness and the rare species of riverine fish and the vulnerable native species existing in lake ecosystems.

Acknowledgements

We want to thank the Basque Water Agency (URA) and the Basque Government, which supported the study through a Convention with AZTI.

Special thanks are due to all partners and contributing institutions of the EFI+ project in Europe for providing fish data. Moreover, Andreas Melcher provided helpful comments on the taxonomic classification of riverine fish assemblages and Nathalie Reynaud helpful supports on the lake database. We are grateful to Marie-Hélène Lizée for fruitful comments on a draft version of this document.

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