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 6.1 Synthesis of stressor interactions and indicators

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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

Overview

The Deliverable 6.1 is composed of 3 reports.

Summary D6.1-1

Currently, practical management of water bodies focuses on the control of single stressors which are assumed to be dominant. Work by the MARS project and others using ecosystem scale and experimental observations has demonstrated that the relationships between primary stressors and ecological response indicators can be confounded through interactions with secondary pressures, giving rise to a potentially novel approach in the management of water bodies to achieve ecological recovery: the *multi-stressor-response approach*.

Little is known about the commonality of multiple stressor effects on ecological and ecosystem service indicators and whether these vary in space and time. This information is needed to support large scale multiple-stressor management approaches, for example, to off-set the effects of climate change through abatement of nutrient stressors.

This report addressed knowledge gaps in this field by developing a standard quantitative assessment of data analysis across MARS experiments, long-term monitoring, and river basin spatial monitoring case studies, providing a comprehensive comparison of responses from mesocosm to Europe in scale.

The common analysis approach allowed for quantification of interaction strength and forms between three widespread stressor combinations found across Europe: (1) nutrients and high temperatures, (2) nutrients and low flow, and (3) nutrients and high flow.

To synthesise multi-stressor interactions, we developed a simple and standardised analysis workflow to quantify the interacting effects of nutrient stress and one other stressor on a range of ecological responses in a consistent manner. We requested that individual analysts across the project team complete the analysis on their own datasets and report the results for syntheses. Therefore, the analysis workflow was designed to be as simple as possible and to generalise across a range of potential response variable types and study designs. Some analysts sub-setted their data set to explore the sensitivity of the relationships between multiple stressor and indicators across different spatial and - temporal scales.

In total we obtained results from 47 analyses completed within 12 separate studies. Of these, 43 originated in northern and central Europe while just four analyses were from Southern Europe. In addition to nutrient stress as the most common primary stressor, the most common secondary stressors examined were high temperature and high flow. Indicators of phytoplankton responses were most commonly reported, followed by fish, with few studies of invertebrates and macrophytes.

Given the large number of studies examining the abundance of cyanobacteria, and to a lesser extent fish abundance, results were also considered in relation to the ecosystem services of water supply, recreational value and fisheries.

Statistically significant responses to nutrient stress (i.e. the primary stressors) were found in 85% of analyses (40 out of 47) while significant responses to the secondary stressors were apparent in only 38% of analyses (18 of 47). Responses to the secondary stressors were most commonly detected in analyses of temperature and morphology, but were rarely detected in analyses of high or low flow. 4

Classifying overall interaction types based on both stressor effects resulted in roughly equivalent tallies among the three types of interaction considered: 18 antagonistic effects, 13 opposing effects and 16 synergistic effects across all analyses. For phytoplankton only, the results were 10 antagonistic effects, 11 opposing effects and 11 synergistic effects. There was little sign of clear differences in interaction types among the types of secondary stressor or across ecosystem type. However, this comparison was hampered by the relatively small sample size and the fact that a relatively low proportion of analyses yielded statistically significant interaction terms. Of the 47 modelled interactions, only 8 (17%) achieved statistical significance at $P < 0.05$ and these did not show a clear tendency towards any one of the interaction types.

Our study highlights that experimental approaches often provide the clearest signal of stressor interactions. They do not, however, provide a comprehensive understanding of how stressors interact in the real-world, over varying sites and stressor gradients; for this monitoring data are more relevant.

The range of responses in stressor interactions across all our case-studies highlight that it is often difficult to predict how two stressors may interact at a given site and both synergistic and antagonistic responses may be possible for the same stressor combination at sites with different characteristics or different levels of stress. Sometimes the significance of stressor effects, both acting singly or in combination, may be masked by other covariates either in different seasons or years (e.g. effects of nutrients may be masked by high flow in rivers) or at sites of differing typology (e.g. deep lakes may differ in sensitivity from shallow lakes). We offer recommendations on improving the analytical approach to detect the effects of interacting stressors in this context.

An improved understanding of the impact of stressor reduction is vital to evaluate the success of potential management options and underpin practical MARS guidance on river basin management planning (RBMP). Unfortunately, the data collated here do not allow examination of stressor abatement responses. We do, however, review evidence on stressor abatement and offer some considerations with respect to developing recovery concepts within future work.

Summary D6.1-2

Aim of this study was to analyse a large set of bioassessment metrics to identify and quantify stressor-specific metric responses reacting to one group of stressors but not to another.

We hypothesise that stressor-specific responses occur when the individual stressors show independent 'modes of action' (i.e. the specific stress-induced changes of environmental factors that modify the ecological niches of the species constituting the biological community).

The data used comprised three biological groups (macrophytes, benthic invertebrates, fish) covering three broad river types in Western and Central Germany. The stressor groups under investigation were physico-chemical, hydromorphological and hydrological stress.

We performed linear variation partitioning to reduce the large set of metrics to a set of candidates for further non-linear analyses using a combination of boosted regression tree modelling and variation partitioning.

The linear analyses revealed 16 candidate metrics that met our criteria, most of them for the medium to large lowland rivers. Macrophyte- and fish-based metrics were most relevant. In a geographically and methodologically more precise data subset, invertebrate metrics revealed more promising models than in the broader data set.

Subsequent non-linear modelling resulted in two truly stressor-specific metrics, both based on invertebrate data: The *Index of Biocoenotic Region* (specifically indicating hydromorphological stress) and the *Share of alien species* (specifically indicating physico-chemical stress).

We concluded that the biological community generally responds to stressors in rather an integrative than a specific way, but stressor-specific metrics can be identified. Future research on diagnostic metrics should focus on quantifying those stressor parameters that represent individual 'modes of action'.

Summary 6.1-3

This report is the Deliverable presenting the results for the 4th aim in relation to the work package task 6.4 entitled *'Integrated River basin management: evaluation of the MARS conceptual model'*.

Within task 6.4 an evaluation was made on the current river basin management practises for dealing with multiple stressors and how existing river basin management plans can be improved by incorporating elements of the MARS conceptual model and the MARS Tools (WP7). We reflect on these current practises and evaluate the MARS conceptual model and MARS Tools as an aid to daily water management on a local level. In particular, we focused on two key European policy/management questions: the benefits of sustaining ecological flows and the value of green infrastructure for natural water retention measures (flood regulation and drought

mitigation) in relation to other water management questions, strategies and practises. These two topics are seen only as examples, as many other aspects of RBMPs could also be assessed.

Using a structured questionnaire and a workshop with WP4 case-study partners, their associated river basin managers, and a wider group of river basin managers from our applied partners and elsewhere in Europe, we were able to obtain an overview of the current practises in setting river basin management plans and selection of measures in relation to the multiple stressor challenges throughout Europe. The main aim of the questionnaire was to get a better understanding of the following questions:

- How does daily water management practice deal with the selection of cost-effective measures, for water bodies exposed to multiple stressors?
- Is knowledge on pressure interactions and biological response taken into account when selecting and prioritizing the measures?
- How can MARS best contribute to a potential gap in knowledge and tools from the perspective of the stakeholders?

We specifically challenged the workshop participants to link their current practises to the topics relevant within the MARS project and linked this to the potential need and usage of tools that could help identify the role of multi-stressor challenges within their daily management practises. With this information we defined how the conceptual model could be used in practice and what gaps in indicators or tools are currently hampering daily practise.

Funded by the European Union within the
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Deliverable 6.1 Synthesis of stressor interactions and indicators

D6.1-1: Stressor interactions and most sensitive indicators of impact at the species, ecosystem and land- scape scale that are relevant to sustaining eco- system services and human benefits

Lead beneficiary: Natural Environment Research Council (NERC)
Authors: Daniel Chapman, Bryan Spears, Lindsay Banin, Sebastian Birk, Alexander Gieswein, Jenica Hanganu, Niina Kotamäki, Shenglan Lu, Sarai Pousa, Christel Prudhomme, Jessica Richardson, Susanne Schneider, Stephen Thackeray, Jarno Turunen, Laurence Carvalho
Contributing Partners: CEH (Chapman, Spears, Banin, Prudhomme, Richardson, Thackeray, Carvalho); UDE (Birk; Gieswein); DDNI (Hanganu); SYKE (Kotamäki); AU (Lu); AZTI (Pousa); NIVA (Schneider); ENVIRONMENT-FINLAND (Turunen)

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

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Contents

Summary

1. Currently, practical management of water bodies focuses on the control of single stressors which are assumed to be dominant. Work by the MARS project and others using ecosystem scale and experimental observations has demonstrated that the relationships between primary stressors and ecological response indicators can be confounded through interactions with secondary pressures, giving rise to a potentially novel approach in the management of water bodies to achieve ecological recovery: the *multi-stressor-response approach*.
2. Little is known about the commonality of multiple stressor effects on ecological and ecosystem service indicators and whether these vary in space and time. This information is needed to support large scale multiple-stressor management approaches, for example, to off-set the effects of climate change through abatement of nutrient stressors.
3. This report addressed knowledge gaps in this field by developing a standard quantitative assessment of data analysis across MARS experiments, long-term monitoring, and river basin spatial monitoring case studies, providing a comprehensive comparison of responses from mesocosm to Europe in scale.
4. The common analysis approach allowed for quantification of interaction strength and forms between three widespread stressor combinations found across Europe: (1) nutrients and high temperatures, (2) nutrients and low flow, and (3) nutrients and high flow.
5. To synthesise multi-stressor interactions, we developed a simple and standardised analysis workflow to quantify the interacting effects of nutrient stress and one other stressor on a range of ecological responses in a consistent manner. We requested that individual analysts across the project team complete the analysis on their own datasets and report the results for syntheses. Therefore, the analysis workflow was designed to be as simple as possible and to generalise across a range of potential response variable types and study designs. Some analysts sub-setted their data set to explore the sensitivity of the relationships between multiple stressor and indicators across different spatial and - temporal scales.
6. In total we obtained results from 47 analyses completed within 12 separate studies. Of these, 43 originated in northern and central Europe while just four analyses were from Southern Europe. In addition to nutrient stress as the most common primary stressor, the most common secondary stressors examined were high temperature and high flow. Indicators of phytoplankton responses were most commonly reported, followed by fish, with few studies of invertebrates and macrophytes.
7. Given the large number of studies examining the abundance of cyanobacteria, and to a lesser extent fish abundance, results were also considered in relation to the ecosystem services of water supply, recreational value and fisheries.
8. Statistically significant responses to nutrient stress (i.e. the primary stressors) were found in 85% of analyses (40 out of 47) while significant responses to the secondary stressors were apparent in only 38% of analyses (18 of 47). Responses to the secondary stressors were most commonly detected in analyses of temperature and morphology, but were rarely detected in analyses of high or low flow.

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10. Our study highlights that experimental approaches often provide the clearest signal of stressor interactions. They do not, however, provide a comprehensive understanding of how stressors interact in the real-world, over varying sites and stressor gradients; for this monitoring data are more relevant.
11. The range of responses in stressor interactions across all our case-studies highlight that it is often difficult to predict how two stressors may interact at a given site and both synergistic and antagonistic responses may be possible for the same stressor combination at sites with different characteristics or different levels of stress. Sometimes the significance of stressor effects, both acting singly or in combination, may be masked by other covariates either in different seasons or years (e.g. effects of nutrients may be masked by high flow in rivers) or at sites of differing typology (e.g. deep lakes may differ in sensitivity from shallow lakes). We offer recommendations on improving the analytical approach to detect the effects of interacting stressors in this context.
12. An improved understanding of the impact of stressor reduction is vital to evaluate the success of potential management options and underpin practical MARS guidance on river basin management planning (RBMP). Unfortunately, the data collated here do not allow examination of stressor abatement responses. We do, however, review evidence on stressor abatement and offer some considerations with respect to developing recovery concepts within future work.

Introduction

Ecosystems can respond through changes in their ecological structure and function to multiple and interacting stressors resulting in a wide range of response forms that are often difficult to predict. Generally, scientific studies have focused on the impact of one primary stressor on aquatic ecosystems. In recent years, however, there has been a move to understand how simultaneously multiple stressors may affect ecosystems, since this reflects more realistic conditions (Vinebrooke et al. 2004; Figure 1 and 2). By failing to analyze a complete suite of stressors, results could be confounded, erroneous and misleading (Tockner et al. 2010) and this may lead to incorrect decisions taken in ecosystem management. It is therefore important to identify the most prominent of stressors acting on our ecosystems and to examine whether this prominence varies in space and time.

Currently, practical management of water bodies focusses on the control of single stressors which are assumed to be dominant. This approach is attractive in that it meets the practical needs of water managers in that it offers a simple conceptual model; reduce the primary pressure and the ecosystem will recover. In a recent analysis of recovery case studies, Verdonschot et al. (2011) confirmed that most reports of river, lake, estuarine and coastal waters in the literature consider responses from single pressure abatement, only. However, the efficacy of the *primary-stressor-response* approach is questionable where ecological responses to primary pressures are known to vary depending on the magnitude or frequency of the stressor. For example, in lakes, the form and timing of ecological responses in primary producers to increasing, compared with decreasing, nutrient concentrations (phosphorus and nitrogen) can vary markedly depending upon the physical, biological, and meteorological characteristics of the lake (Scheffer et al. 2001).

The selection of the most appropriate stressors to manage may not necessarily be based on site-specific understanding of stressor-response relationships but instead is often influenced by the weight of evidence on common primary stressors operating at the meta-system scale. For example, the need to control phosphorus across eutrophic lakes for ecological restoration has been argued (Schindler et al. 2016) and clearly this is a sensible approach where meta-lake relationships indicate P loading to be the primary driver of phytoplankton biomass at this scale. However, when we consider any single lake on these meta-lake plots it is apparent that a large proportion of lakes do not conform to this general relationship (Spears et al. 2013). These departures from general relationships can be influenced by the effects of secondary pressures which drag individual lakes from the regression lines. Practically speaking, this means that an interaction between two stressors has caused the lake to behave in a manner that is unexpected when considering the *primary stressor approach*. Many authors have demonstrated that secondary pressures, i.e. those acting to alter the primary stressor-response relationships, can act at the meta-lake scale. For example, Weyhenmeyer et al. (2007) and Moss et al.

(2011) have demonstrated the widespread effects of climate change (e.g. temperature) on the relationships between nutrients and ecological structure and function in lakes, giving rise to a potentially novel approach in the management of water bodies to achieve ecological recovery: the *multi-stressor-response approach*. However, both primary and secondary pressures can operate across scales although the extent to which interactions occur across scales and ecosystem types is a major knowledge gap in this field which limits practical application of the *multi-stressor-response approach*.

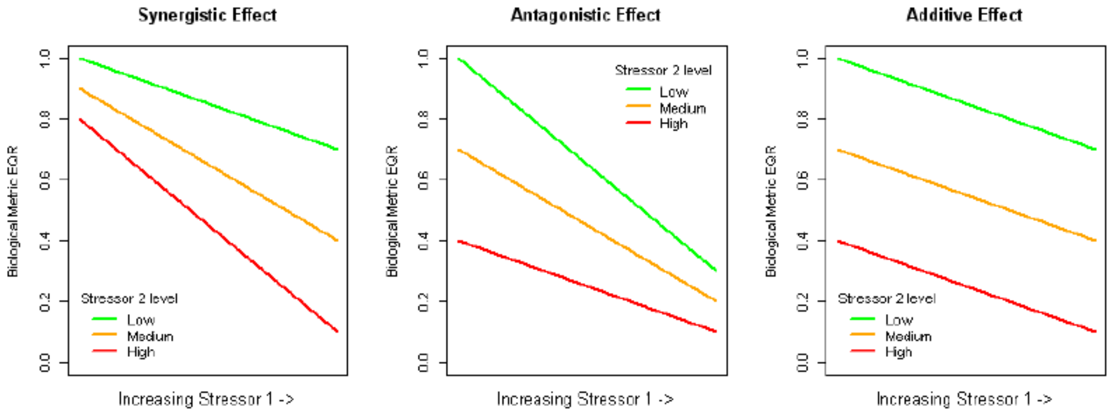


Figure 1. Schematic diagram of the types of interaction (synergistic, antagonistic and additive) between two stressors, where stressor 1 is continuous, stressor 2 is categorical with three levels. (Taken from Dunbar 2013)

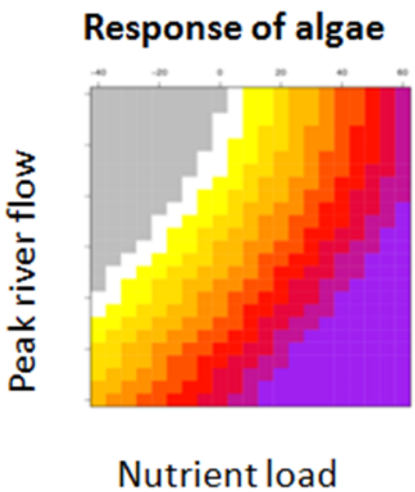


Figure 2. Interaction between two continuous stressors (e.g. peak flow and nutrient load) with colour intensity representing varying degree of response (e.g. abundance of algae). The direction of the colour shift indicates the nature of the interaction.

Work by the MARS project and others using ecosystem scale and experimental observations has demonstrated that the relationships between primary stressors and ecological response indicators can be confounded through interactions with secondary pressures. Piggott et al. (2015) presented a conceptual framework within which these interaction forms can be categorised broadly into *additive* (the effect of one stressor is not dependent on the level of another stressor), *synergistic* (increase in one stressor amplifies the negative effect of another) or *antagonistic* (increase in one stressor reduces the negative effect of another) forms.

Nõges et al. (2015) reviewed 219 peer reviewed publications to gather evidence on the occurrence of common stressor combinations, the magnitude of the combined impacts on response indicators, the reliability of the evidence base, and the forms of the interactions across rivers, lakes, and transitional and coastal waters. The main conclusion from this study was that, despite the fact that the conceptual evidence base underpinning the multi-stressor-response approach is well advanced, very few studies actually quantify interactions between multiple stressors (from 15% in lakes to 65% in rivers report interaction forms), instead simply reporting the net effects of all stressors. Nõges et al. (2015) report that nutrient stressors were commonly considered as primary stressors across all ecosystem types and that hydro-morphological stressors were the most common secondary stressor reported. Dual stressors were the most commonly reported multi-stressor scenario (43% of reports). Finally, the predictive power of stressor-response models improved only for lakes when *multi-stressor-response* models were constructed compared with *primary stressor-response* models. Similarly, models to predict the stressor-response effects on fish across all ecosystem types were improved through construction of *multi-stressor-response* models when compared to *primary-stressor-response* models. When reported, synergism was most common in groundwaters, additive effects were most common in transitional and coastal waters and no dominant form was reported for lakes. All interaction forms were reported to occur in all ecosystem types.

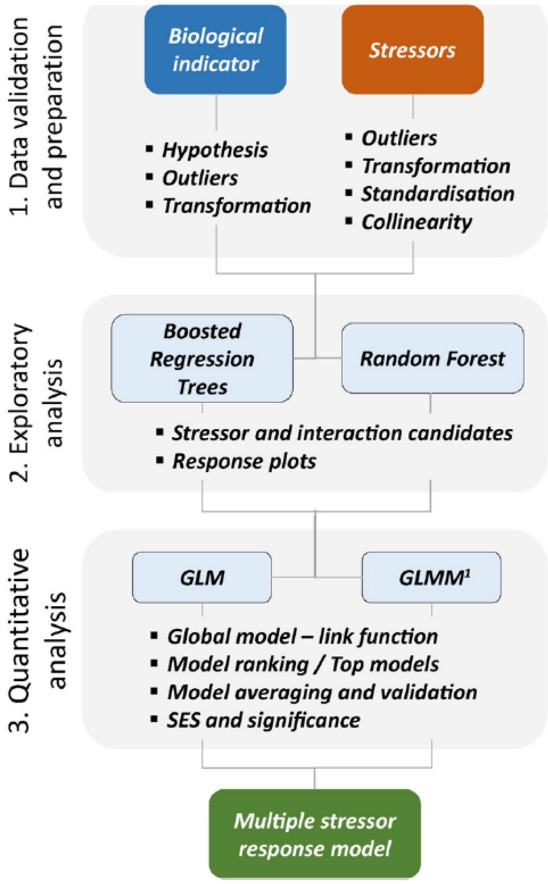


Figure 3. Analytical framework for the identification of multi-stressor impacts in aquatic ecosystems using biomonitoring data, from Feld et al. (2016).

To improve the quality of multi-stressor-response evidence necessary to support the development of practical management approaches in this context, it is important first to construct a common analytical approach that can be applied to a range of data types. Feld et al. (2016) considered this challenge and produced a 'Cook Book' of statistical approaches for the detection of the impact of multiple stressors using aquatic biomonitoring data. This approach provides a framework for the development of multiple-stressor-response models. When applied at the meta-system scale it allows comparison of stressor scenarios, magnitude of independent and interaction effects, sensitivity of indicators, interaction types, and improvements on primary-stressor-response models through inclusion of secondary stressors. The approach also acknowledges the need to examine multi-stressor effects on response indicators using time-series, spatio-temporal and experimental data sources.

We aim here to test the hypotheses raised by Nõges et al. (2015), that nutrients are the primary stressors acting across all aquatic ecosystem types considered by the MARS project. This was achieved by developing a standard quantitative approach to be adopted across the MARS project (following aspects of the general approach outlined in Feld et al. 2016). This approach was applied to data from experiments, long-term monitoring, and river basin spatial monitoring case studies. The common analysis approach allowed for quantification of interaction strength and forms between dual stressors providing a comprehensive comparison of responses from mesocosm to Europe in scale. The results from these standardised quantitative studies were then systematically collated and examined using a qualitative meta-analysis approach. The analysis was supplemented with additional data derived from specific case studies where necessary.

The case-studies examined focused on three multiple-stressor situations commonly found across Europe:

1. The ecological response to nutrient stress and high temperatures.
2. The ecological response to nutrient stress and conditions of low flow.
3. The ecological response to nutrient stress and conditions of high flow.

Aims and objectives of the report

Our aim is to synthesise the results of standardised analyses performed on a large number of individual datasets. We have, therefore, designed the statistical analysis to be as simple as possible and in a way that generalises across a range of potential response variable types and study designs. In practice, this means selecting the appropriate model from a range of linear, generalised and mixed models (see subsequent sections for details).

We recognise that the proposed analysis may not utilise the 'best' model to explain variation in each dataset. Instead each individual analysis should deliver a standardised measure of ecological responses to stressors, following the three common stressor combinations described above. We will combine these individual results to synthesise general patterns across systems. We are particularly interested in the interactions between the two stressors mentioned in the three common stressor combinations, and so have designed the analyses to quantify the nature and relative importance of these interactions.

The primary aims of this report are to describe the methodology developed towards a common analysis of multiple stressor evidence across the MARS project and to synthesize these results to address the following **objectives**:

1. To identify indicators that respond to widespread multiple stressor combinations across multiple ecosystem types.
2. To test general patterns in interaction types among multiple stressors across ecosystem types.
3. To identify water-body types that are sensitive or tolerant to specific multiple stressor conditions.
4. To make recommendations on best–practice analytical and monitoring approaches to detect multiple stressors interactions.

Methods

To synthesise multi-stressor interactions, we developed a simple and standardised analysis workflow to quantify the interacting effects of nutrient stress and one other stressor on a range of ecological responses in a consistent manner. We requested that individual analysts across the project team complete the analysis on their own datasets and report the results for syntheses. Therefore, the analysis workflow was designed to be as simple as possible and to generalise across a range of potential response variable types and study designs. Some analysts sub-setted their data set to explore the sensitivity of the relationships between multiple stressor and indicators across different spatial and -temporal scales.

To communicate the analysis workflow, we produced a detailed guidance document, presented this at the project mid-term meeting (February 2016) and through a web meeting (June 2016). Full details of the analysis workflow, including R code, can be found in Appendix 1. Below the major steps in the analysis workflow are summarised.

Model construction

Choice of response variables

Individual analysts were free to choose their own response variable, though we requested that response variable choices would prioritise the MARS benchmark indicators or responses (MARS Deliverable 2.1).

Choice of stressor variables

Based on the common questions, each analysis considered responses to two stressors, which were two of nutrient stress, high temperature, low flow, high flow or morphological change. We requested that the individual analysts used their knowledge of their system to decide upon the most relevant measures of the stressors for their own particular analyses.

Variable transformations

All continuous variables (responses and stressor variables) were transformed prior to analysis to improve their conformance to normal distributions and standardised to zero mean and unit variance. This aids statistical model convergence and reduces model heteroscedasticity. In most cases a version of the Box-Cox transformation was used, including an offset to ensure strict positivity (all values > 0). However, some analysts used log transformations.

Choice of statistical model

The type of statistical model fitted depended on two major considerations:

- The type of system (which determined whether a mixed model with random effects was needed).
- The type of response data (which determined whether a generalised model was needed).

The first decision was to determine whether a standard linear model was sufficient, or whether a mixed effects model that accounts for 'random effects' was needed (Table 1). Mixed effects models were required when the study structure included grouping factors, such as experimental block, site or year (Table 1). In most cases the analysts included random effects in the standard way as random intercept terms. However, we also allowed them to use more advanced random slope models if they felt this was more appropriate.

The second decision was to choose the appropriate linear or mixed effects model based on the type of response variable being modelled (Table 2). We allowed for response variables taking one of four types:

- Continuous – can take any value (possibly within a range);
- Binary – can only take one of two categories, e.g. 0/1, presence/absence, A/B;
- Count – integers from 0 to ∞, e.g. population sizes of a species (If you have very high count values you may wish to treat the data as a continuous variable);
  Ordered categories – a discrete scale that ranks data, e.g. bad/poor/moderate/good/high status, plant cover-abundance scales.

Following selection of the appropriate model, the analysts fitted the models to their data, specifying main effects of both stressors and an interaction term.

Table 1: Criteria for determining whether a mixed effects model with random effects was required.

System	Mixed effects model required?
Mesocosm experiment	Possibly, depending on experimental design. Grouping factors such as block or measurement period were included as random effects.
Single-site, multi-year (temporal)	No
Multi-site, multi-year (spatio-temporal)	Yes. Random effects of site and year were included.
Multi-site, single-year (spatial)	No

Table 2: Criteria for determining the type of linear or mixed effects model to use.

Response variable type	Linear model	Mixed effects model
Continuous	Generalised linear model (GLM)	Generalised linear mixed model (GLMM)
Binary	Generalised linear model (GLM)	Generalised linear mixed model (GLMM)
Count	Generalised linear model (GLM)	Generalised linear mixed model (GLMM)
Ordered categories	Cumulative link model (CLM)	Cumulative link mixed model (CLMM)

Residual autocorrelation

Where necessary we tested whether the model residuals contained strong spatial or temporal autocorrelation. This was only necessary when the analysis used linear models without random effects, since the random effects in the mixed effects models should account for grouping in space and time. Autocorrelation means that the residuals of observations close in space or time are more similar than expected by chance. This indicates that model assumptions have been violated and can cause the statistical significance of model terms to be exaggerated. Autocorrelation in space or time was identified with Moran tests on the model response residuals and where substantial autocorrelation was detected, 'trend surfaces' generated using smoothing splines or polynomial functions were included in the models.

Model evaluation

To evaluate the models, residuals were examined for correlation to the fitted values and deviation from the normal distribution. Model fits were evaluated as their marginal R^2, i.e. the proportion of variance explained by the model fixed effects, ignoring the contribution of any random effects (Nakagawa and Schielzeth 2013). Model fixed effects (main effects of both stressors and their interactions) were evaluated as a z-score (estimated coefficient divided by its standard error). These indicate both the direction of the effect (positive or negative) and its statistical significance (high absolute z-scores are more significant) in a standardised way.

Importance of the interaction term

Three simple methods to estimate the importance of the interaction term were used:

- Z-score of the interaction term
- Change in Akaike Information Criteria (AIC) when the interaction term was dropped from the model.
- Change in marginal R^2 when the interaction term was dropped from the model.

Interaction visualisation

Predicted heat maps similar to Figure 2 were produced from each fitted model in order to visualise the form of the interaction fitted by the model.

Interaction classification

The type of interaction was characterised from the fitted model fixed effect coefficients, ignoring statistical significance. We applied a simple classification scheme to the full model, based on both stressors' main effects and the interaction term (Table 3). This was based on the direction of the interaction effect, relative to the directions of the main effects of both stressors. We also classified interaction types with respect to each individual stressor (Table 4). If the interaction indicated that the focal stressor had a greater effect when the second stressor was present we termed the interaction synergistic. If the converse was true we termed the interaction antagonistic.

Advanced model selection

As well as the simple model developed above, we also requested that analysts consider fitting models that include additional stressor variables or habitat characteristics that they think may be important drivers of the response in their system. We proposed a forward stepwise addition of additional variables to the basic model based on minimising AIC. These fuller models were analysed and evaluated as described above.

Reporting

An Excel spreadsheet containing a form for reporting meta-data about the study and details of the analysis was provided.

Table 3: Overall interaction types considering both stressors in the model

Type of interaction	Characterisation
Synergistic	Model coefficients for both stressors and their interaction all have the same sign (i.e. all positive or all negative)
Antagonistic	Model coefficients for both stressors have the same sign, but their interaction has the opposite sign
Opposing	Model coefficients for both stressors differ, sign of the interaction term not important

Table 4: Partial interaction types considering an individual stressor in the model

Type of interaction	Characterisation
Synergistic	Interaction term has the same sign as the stressor main effect coefficient.
Antagonistic	Interaction term has the opposite sign as the stressor main effect coefficient.

Results

Description of data used in meta-analysis

In total we obtained results from 47 analyses completed within 12 separate studies (see Table A1 in Appendix 2). Of these, 43 originated in northern and central Europe while just four analyses were from Southern Europe. In addition to nutrient stress, the most common second stressors were high temperature (n=21) and high flow (n=18) (Figure 4). Monitoring studies were more common than experimental ones (n=8 for mesocosms vs. 22 spatial monitoring analyses and 17 spatio-temporal monitoring studies) (Figure 4). The most common response variable types related to phytoplankton (Figure 5a) and included measures of chlorophyll a and cyanobacterial biomass. Consequently, the most strongly potentially affected ecosystem services were water quality and recreational value (Figure 5b). Because phytoplankton was the most common response, we analysed the results both across all studies and for the 32 phytoplankton analyses separately.

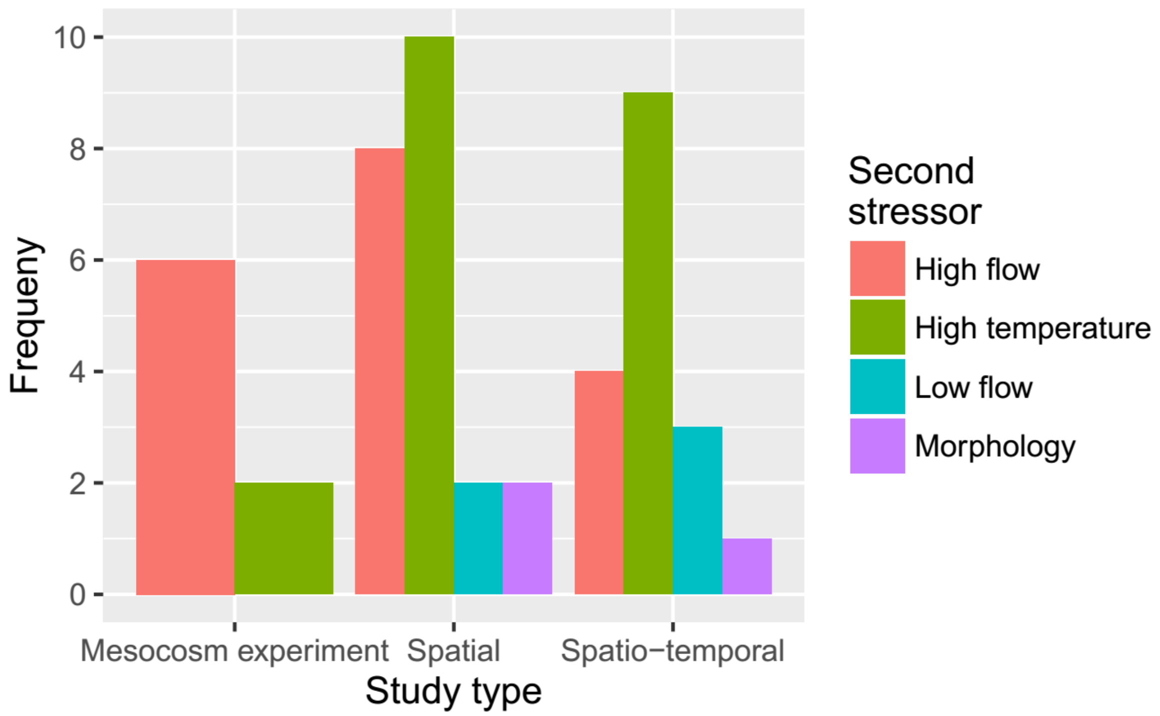


Figure 4. Frequency of analyses grouped by the type of secondary stressor, additional to nutrient stress, and the type of study.

(a)

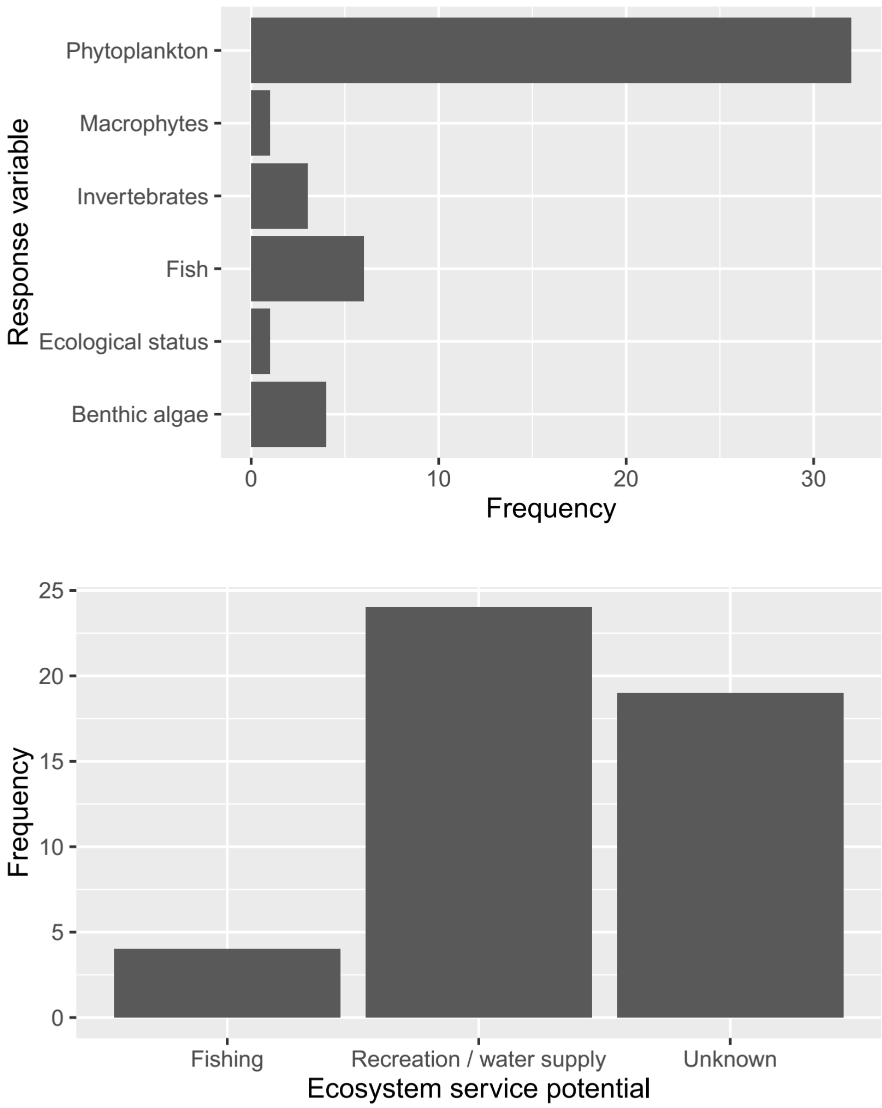


Figure 5. Frequencies of (a) response variable types and (b) their potentially related ecosystem services among the analyses in the synthesis.

Indicator responses to widespread multiple stressor combinations across ecosystems

The fitted models had a median marginal R^2 of 0.134, i.e. explained an average of 13% of the variation, and this had no clear pattern of variation among the second stressors (Figure 6). Statistically significant responses to nutrient stress (classified as present if the nutrient main effect or its interaction term had $P < 0.05$) were found in 85% of analyses (40 out of 47). By contrast, the equivalent statistically significant responses to the second stressors were only apparent in 38% of analyses (18 of 47). Responses to the second stressors were most commonly detected in analyses of temperature and morphology, but were rarely detected in analyses of high or low flow (Figure 7). Very similar results were obtained for the analyses of only phytoplankton responses.

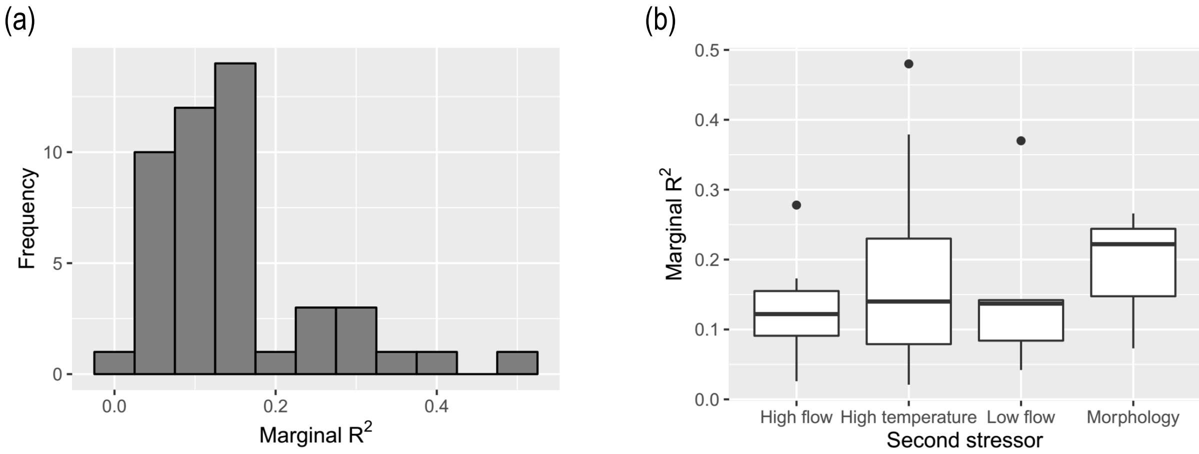


Figure 6. (a) Histogram of model marginal R^2 (proportion of variation explained by the fixed effects) and (b) boxplot showing the variation in marginal R^2 among analyses with different second stressors.

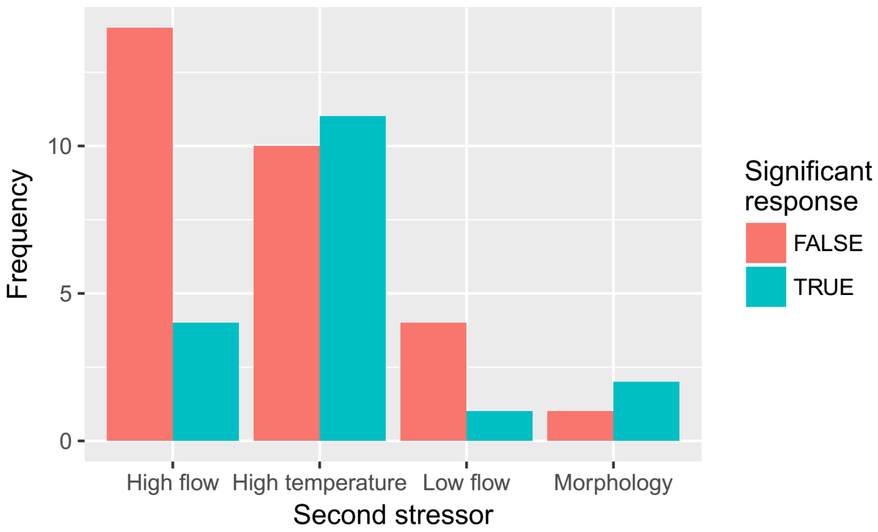
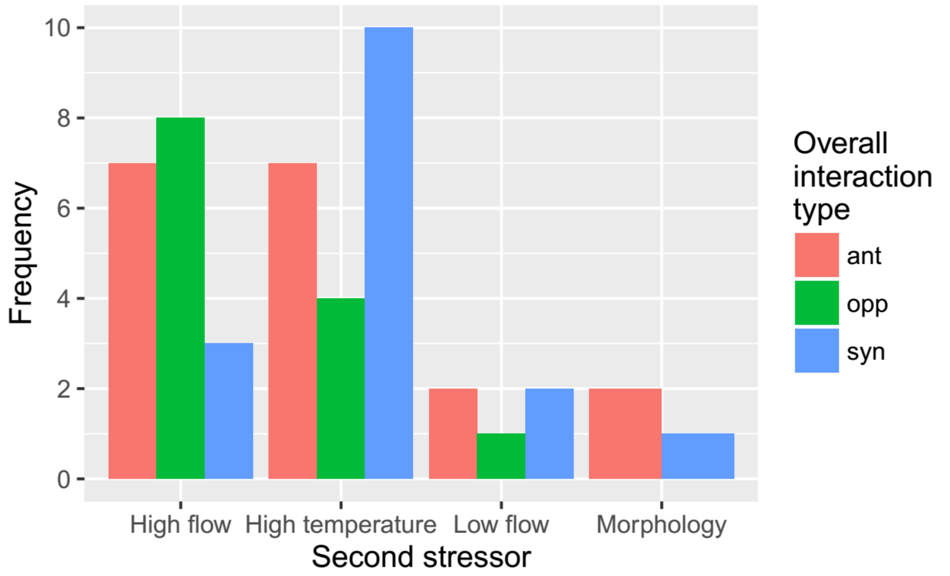


Figure 7. Frequencies of statistically significant responses to the different types of secondary stressor. Stressors were classified as having a significant response if either their main effect or interaction with nutrients achieved significance.

Patterns in interaction types among multiple stressors across ecosystem types

Classifying overall interaction types based on both stressor fixed effects resulted in roughly equivalent tallies among the three types of interaction considered: 18 antagonistic effects, 13 opposing effects and 16 synergistic effects (exact multinomial test $P = 0.694$) across all analyses and 10 antagonistic effects, 11 opposing effects and 11 synergistic effects ($P > 0.999$) for the analyses of phytoplankton responses. There was little sign of clear differences in interaction types among the types of secondary stressor (Figure 8a). However, this comparison was hampered by the relatively small sample size and the fact that a relatively low proportion of analyses yielded statistically significant interaction terms. Of the 47 modelled interactions, only 8 (17%) achieved statistical significance at $P < 0.05$ and these did not show a clear tendency towards any one of the interaction types (Figure 8b).

(a)



(b)

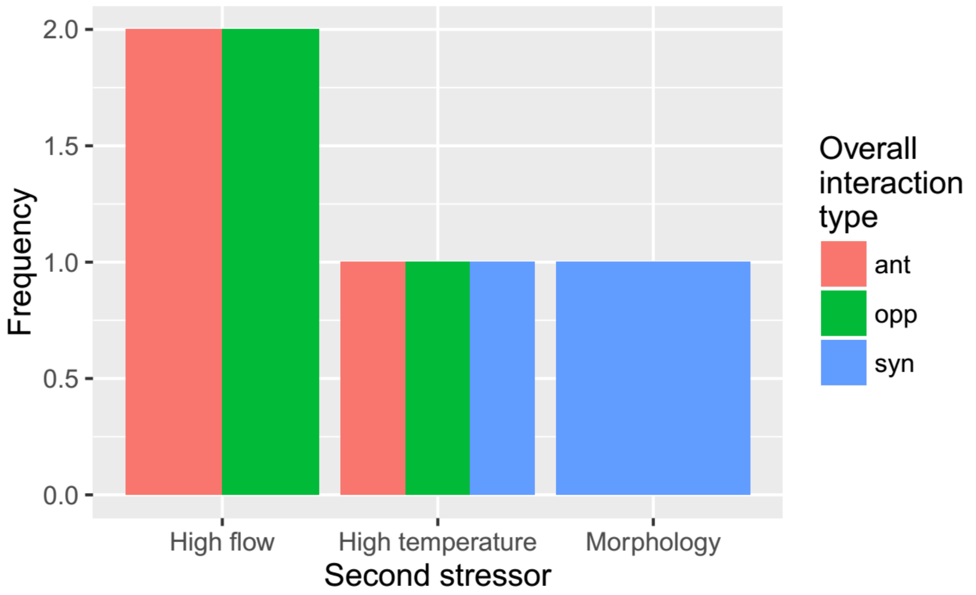
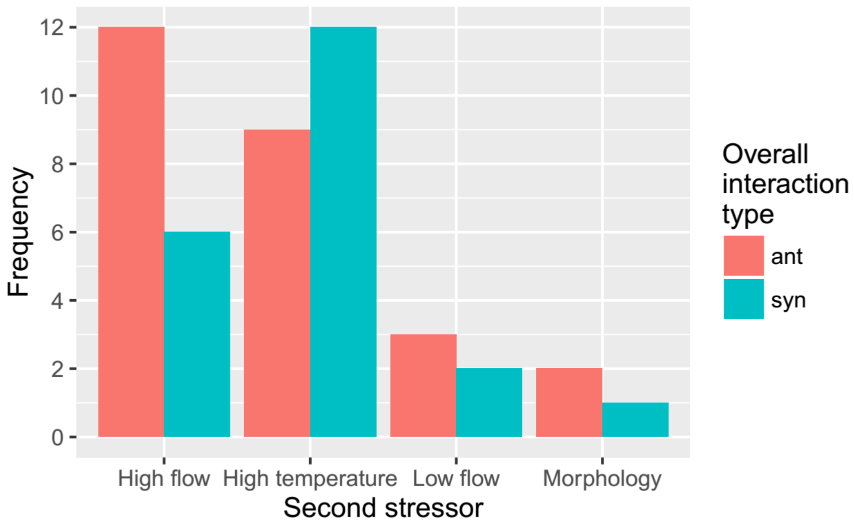


Figure 8. The frequencies of overall model interaction types for each second stressor type, for (a) all analyses and (b) analyses yielding a statistically significant interaction term.

(a)



(b)

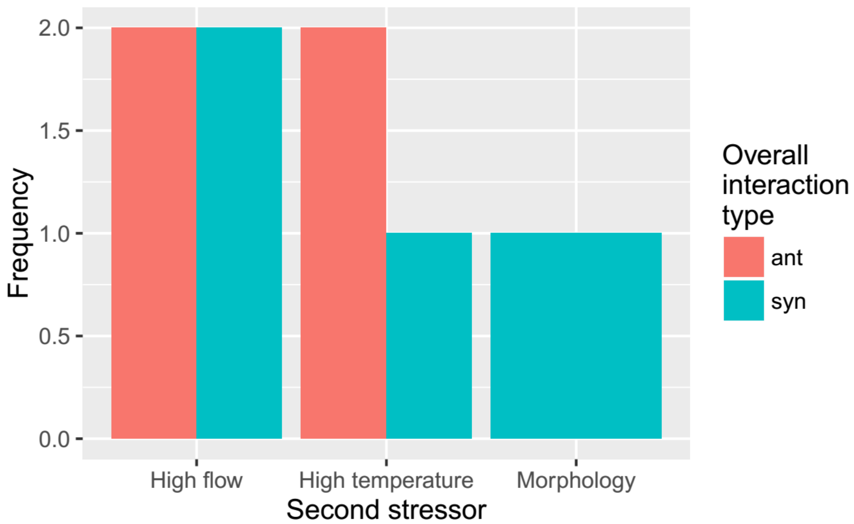


Figure 9. The frequencies of model interaction types with respect to nutrient stress for each second stressor type, for (a) all analyses and (b) analyses yielding a statistically significant interaction term.

Classifying interaction types with respect to nutrient stress only also gave roughly equivalent numbers of antagonistic and synergistic interactions (26 antagonistic, 21 synergistic, exact binomial test P = 0.560). There were equivalent results for just the analyses of phytoplankton (17 antagonistic, 15 synergistic, exact binomial test P = 0.860). There were apparent tendencies in the data for interactions between nutrients and high flow to be antagonistic and for interactions between nutrients and high temperature to be synergistic (Figure 9). However, these were not statistically significant departures from equal frequencies (exact binomial test, $P > 0.2$ in both cases for all analyses and analyses of phytoplankton).

A key factor in determining whether the analyses detected statistically significant stressor interactions was the sample size for analysis (Figure 10). This was most apparent for the observational studies, for which (log) sample size correlated negatively to the interaction term *P*-value (*r* = -0.404, d.f. = 37, *P* = 0.011 for all analyses; *r* = -0.476, d.f. = 26, *P* = 0.010 for analyses of phytoplankton).

Synthesizing the analyses of phytoplankton responses by examining the distribution of z-scores, there were consistently positive main effects of nutrients, high temperature and low flow on phytoplankton indicators (e.g. chlorophyll a, cyanobacterial biomass, PTI), although there were only two analyses of low flow (Figure 11a). However, the median effect of high flow was not significantly different to zero, indicating a general pattern for weak and inconsistent effects (Figure 11a). The z-scores for the stressor interactions also followed this pattern (Figure 11b). Their median effects were not significantly different to zero indicating that interactions with nutrients were weak and exhibited no clear trend towards antagonistic or synergistic effects.

Although the sample sizes are lower, the equivalent analysis for fish response indicators revealed a consistently negative effect of nutrients and low flow (but for very low sample size) but no overall trend for their interactions (Figure 12). Other response indicator types were considered too rare to evaluate separately.

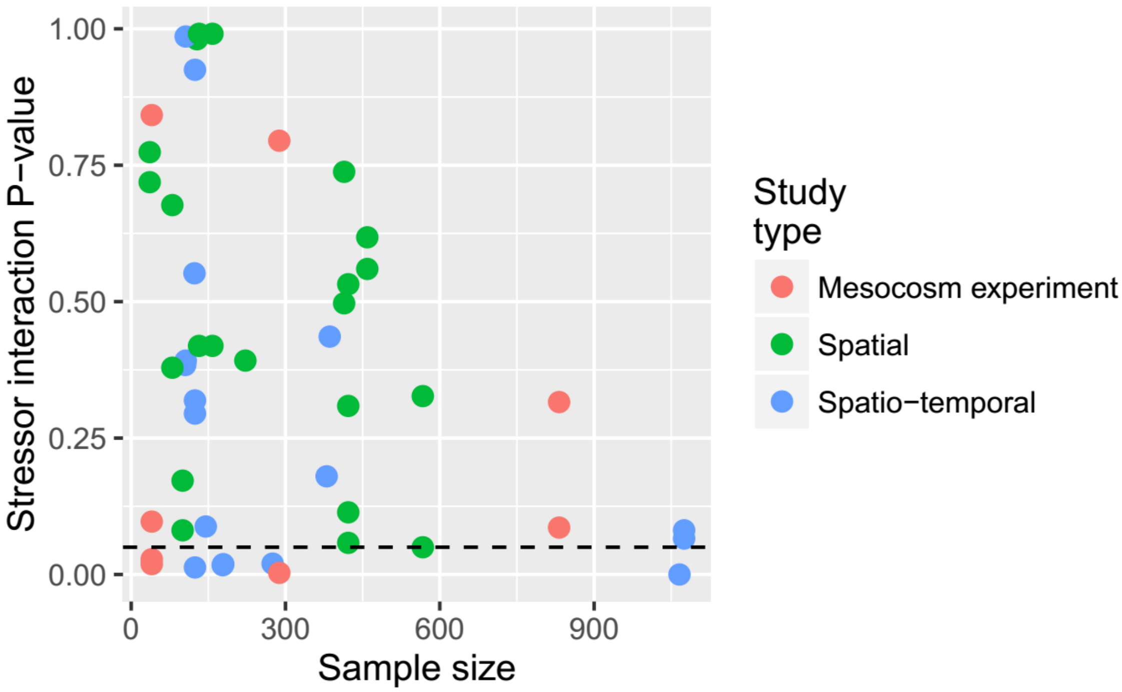


Figure 10. Relationship between study sample size and statistical significance (*P*-value) of the interaction term between nutrients and second stressor.

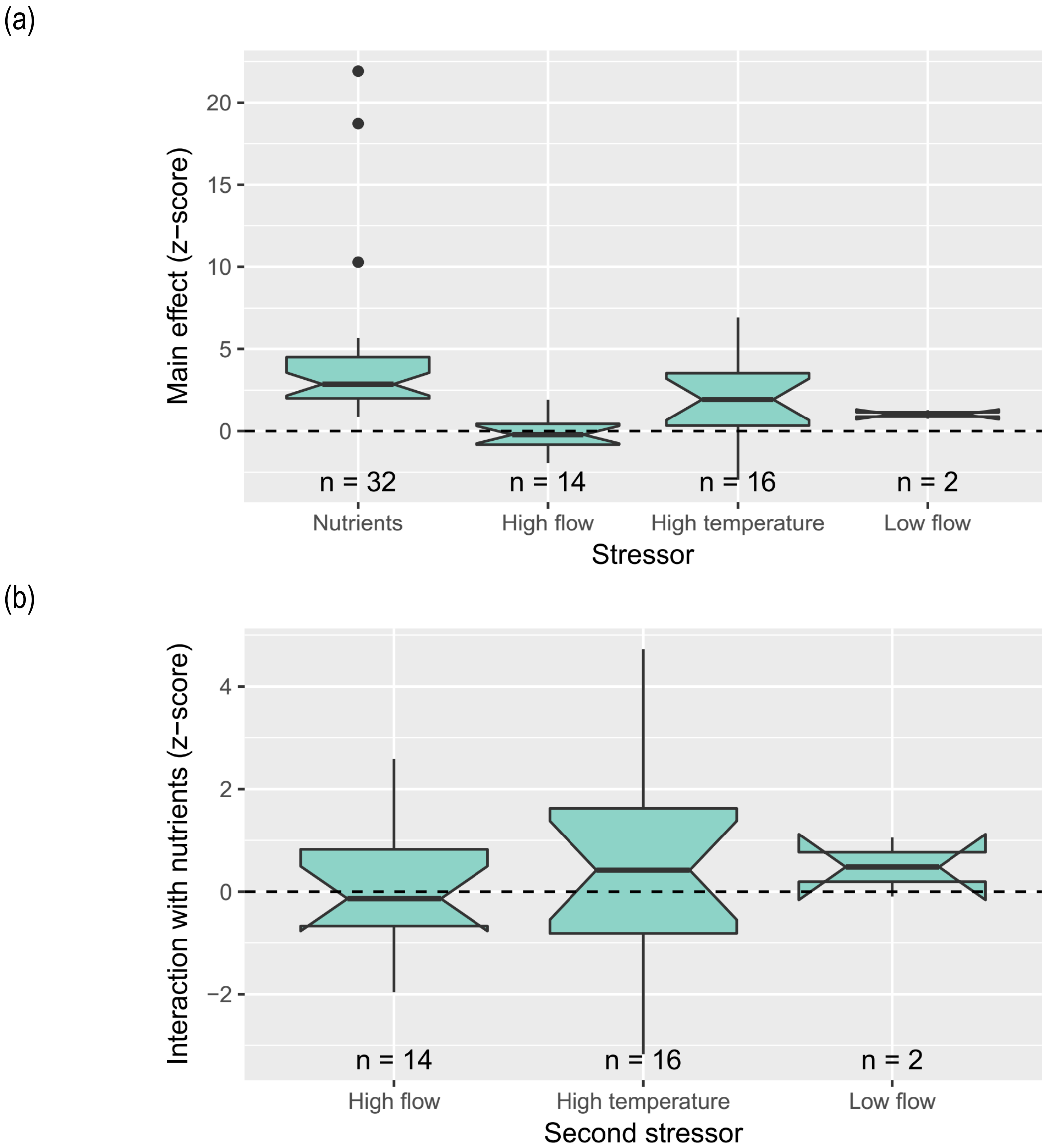


Figure 11. Notched boxplots synthesizing the (a) main effects and (b) stressor interactions for analyses of phytoplankton responses based on nutrients and one other stressor. Effect sizes are presented in a standardized way as z-scores. Boxplot notches extend to 1.58 * inter-quartile range / √sample size, which approximates a 95% confidence interval for the median. Where the notch does not overlap with zero, the median is significantly different to zero (no effect) across all the analyses.

(a)

(b)

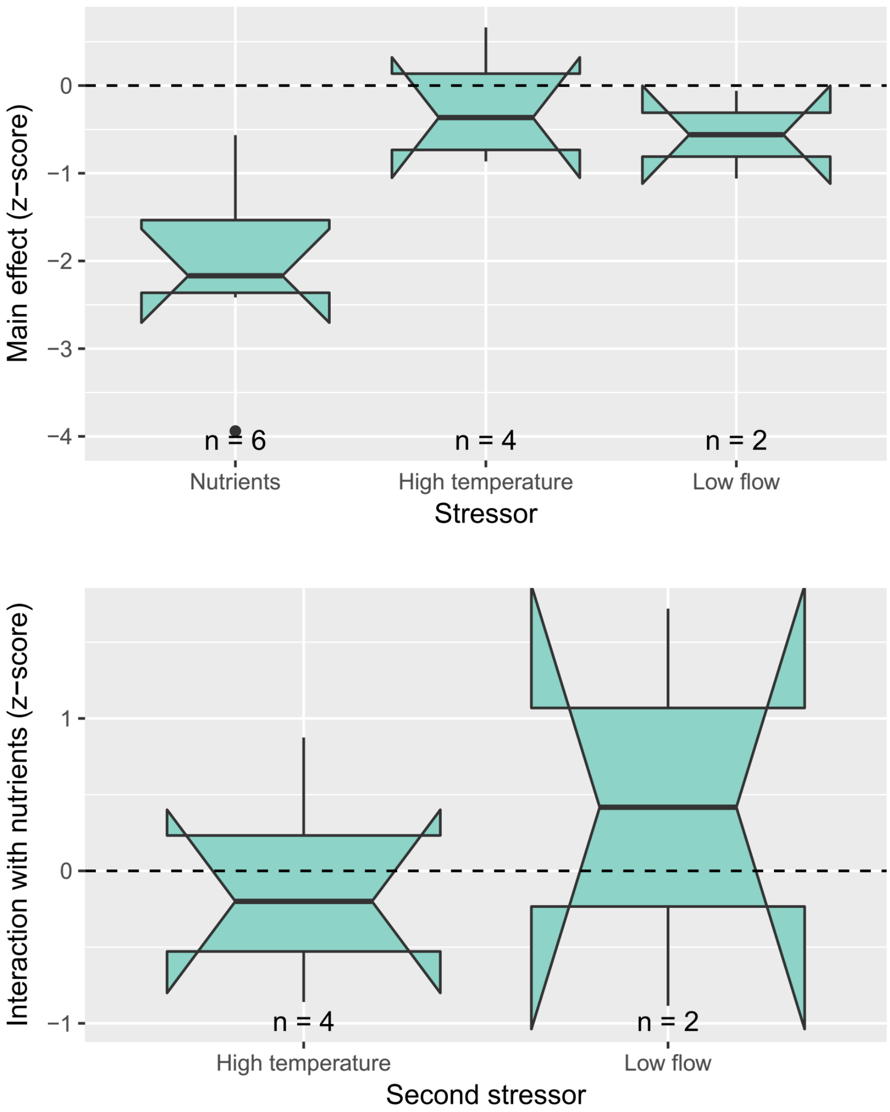


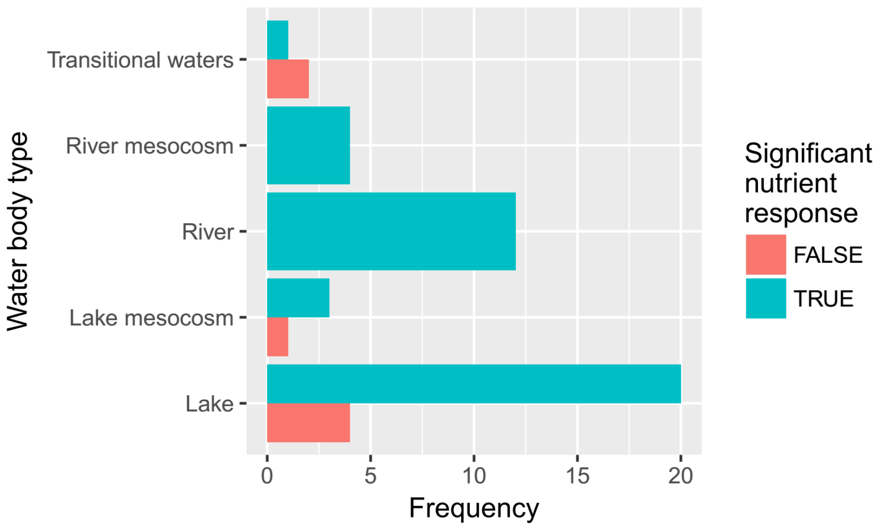
Figure 12. Equivalent of Figure 11, but for analyses of fish responses.

Assessment of water-body types that are sensitive or tolerant to specific multiple stressor scenarios

Across all analyses, there was a significant effect of water body type on the probability of finding a significant response to nutrients either as a main effect or an interaction (ANOVA on binomial GLM, P = 0.047). From Figure 13a it appears that analyses of lakes and transitional waters had a lower proportion of significant nutrient responses than rivers. However, the sample sizes are low, especially for transitional waters, and the same effect was not apparent in the analyses of just to phytoplankton responses (ANOVA on binomial GLM, P = 0.371).

Differences in the frequencies of significant stressor interactions among water body types were not statistically significant (ANOVA on binomial GLM, P = 0.414 for all analyses and P = 0.809 for phytoplankton analyses) (Figure 13b).

(a)



(b)

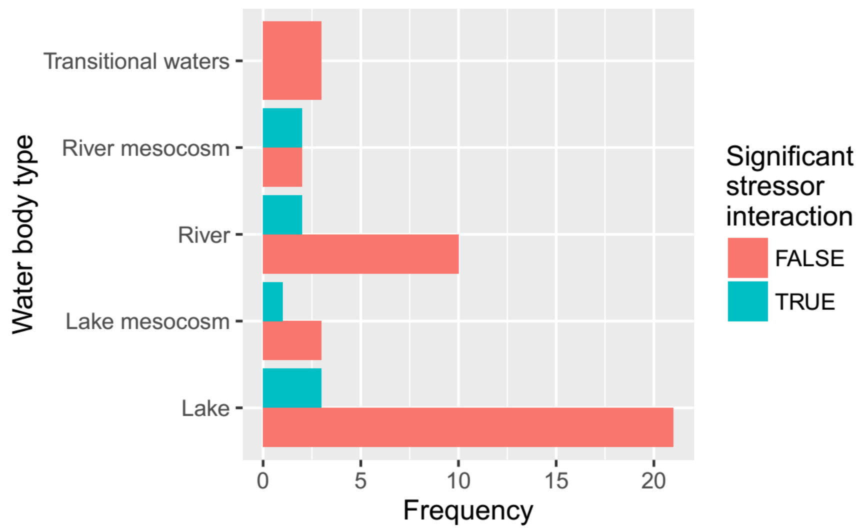


Figure 13. Variation among water body types in the frequencies of statistically significant (a) nutrient responses (main effects or interaction) and (b) interactions between nutrients and a second stressor.

Assessing sensitivity of multiple stressor effects across spatio-temporal scales

The synthesis results highlight a large number of studies where only the primary stressor (nutrients) is significant and where the secondary stressor and stressor interactions are not significant. Three of the studies subset their data to examine whether significant secondary stressor effects and interactions are more apparent under certain ecological conditions or in particular water bodies.

In the River Thames case-study, Bowes et al. (2016) demonstrate temporal patterns in responses due to thresholds in physical factors (flow, temperature and sunshine) that affect the river phytoplankton response to nutrient concentrations (which were in excess). Prolific algal blooms only occurred in the river during warm, sunny periods (>19 °C) when flow rates were <30 m^3 s^{-1} (Figure 14).

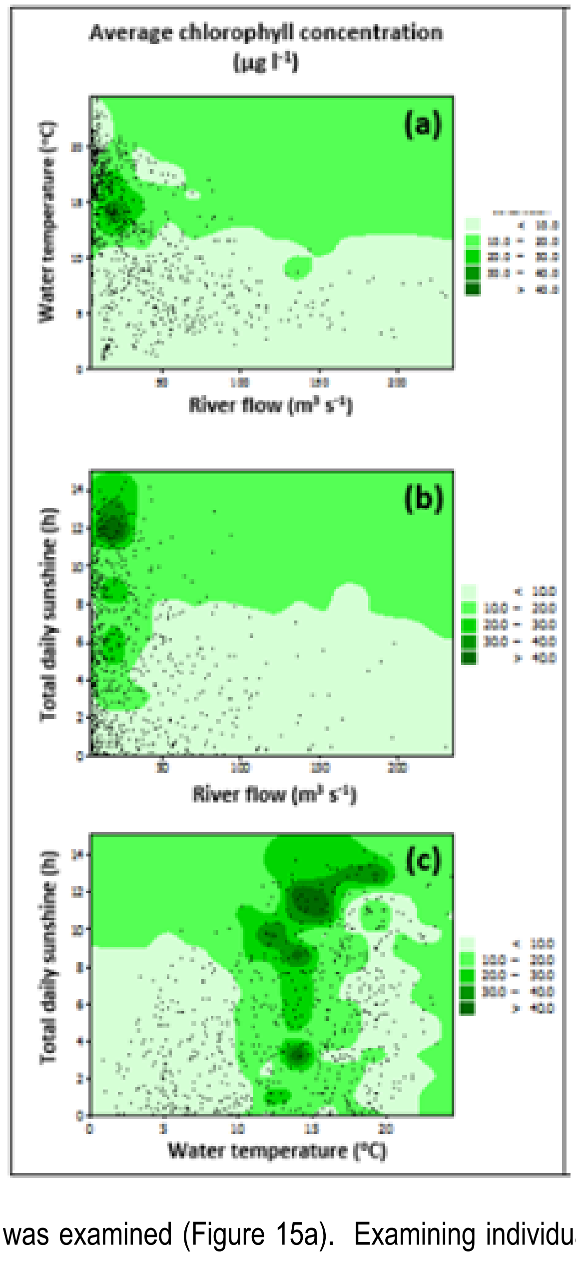


Figure. 14. Multiple stressor relationships between phytoplankton growth (daily chlorophyll concentrations) and daily water temperature, hours of sunshine and flow rate in the River Thames at Reading, UK (Taken from Bowes et al. 2016).

The importance of considering water body typologies, or "sensitivity factors", that are most sensitive to the stressors was examined in two of the case studies. These two studies both examined the response of cyanobacteria to multiple stressors of nutrients, temperature and rainfall and were carried out in a large dataset of >750 lakes at the European scale (spatial study) and across a smaller dataset of 26 European lakes with long time-series data (spatio-temporal study). A more detailed analysis is reported in MARS Deliverable 5.1-4.

No general pattern of the response of cyanobacteria to these multiple stressors, acting individually, or in combination, was observed in the study of 26 long time-series. The summer rainfall – cyanobacteria relationship was very weak when the global dataset was examined (Figure 15a). 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 15b). Lakes with longer residence times had more varied or flat responses.

Where a water-body lies on a stressor gradient may also affect its sensitivity to stressors. The cyanobacteria response to nutrients showed a range of relationships from strongly negative to strongly positive, with the majority of lakes showing a weak positive effect (Figure 16). In the 26 lake time-series, pooling the data from all lakes together revealed a highly significant relationship with nutrients (spring TP), explaining about 7% of variation in the abundance of cyanobacteria over the summer. Sub-setting the whole dataset by trophic type only showed a highly significant spring TP – summer cyanobacteria relationship in oligo-mesotrophic lakes compared to a weak response in eutrophic lakes.

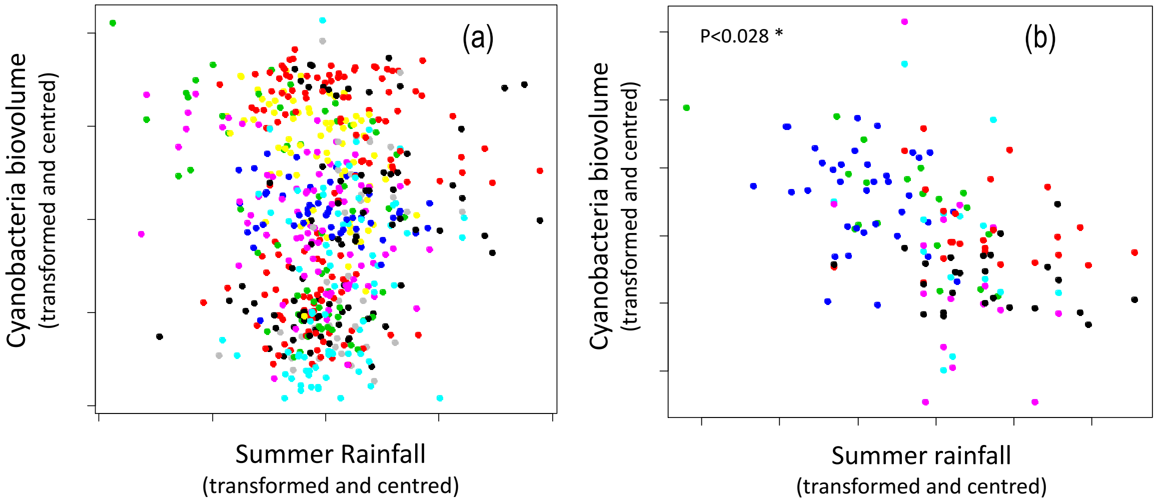


Figure 15. Summer cyanobacteria response to total Summer rainfall in (a) a dataset of 26 lake time-series and (b) a dataset of 6 lakes with short residence times. Colours represent individual lakes.

Finally, surprisingly, there were no significant relationships between summer cyanobacteria and summer temperatures in the European lake studies, with temperature explaining <1% of the total variation in cyanobacteria in the 26 lake time-series dataset and no significant patterns observed in individual lakes or lake types. 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 there were generally much higher values observed in hot years (Figure 17).

The typology classes in the two European lake studies were not comparable and so detailed results cannot be compared. However, in both the spatial European analysis and the spatio-temporal analysis of lake time series, nutrients, in the form of total phosphorus (TP), had the strongest effect of all the stressors. Temperature appeared to interact with nutrients, although the form of this interaction was not consistent, with sometimes a synergistic interaction (e.g. Deep (>5 meters) eutrophic, low humic & high alkalinity lakes) and sometimes an antagonistic interaction (e.g. deep, mesotrophic, high humic and low alkalinity European lakes).

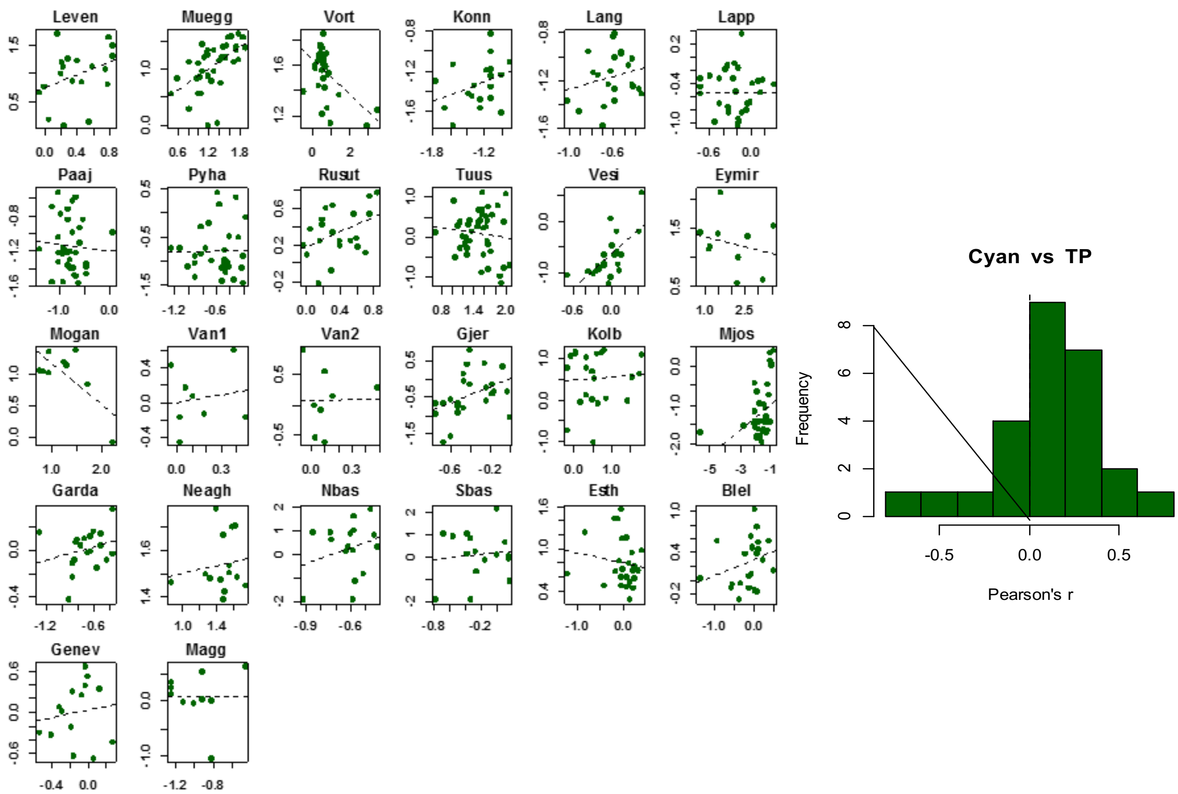


Figure. 16. Summer cyanobacteria response to Spring Total Phosphorus (TP) (data centred and standardised) in 26 individual lake time-series, including a histogram of Pearson's correlation coefficients in the 26 lakes.

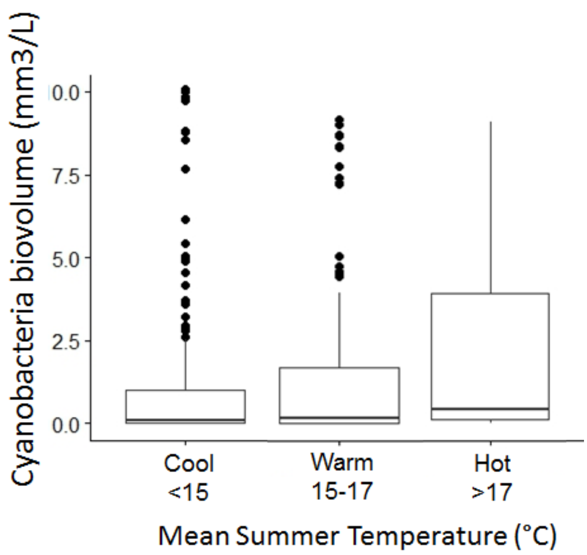


Figure 17. Boxplot of Summer cyanobacteria biovolume response to mean Summer temperatures in 26 European lake time-series, with data grouped by years of cool, warm or hot summers.

Discussion

General description of meta-analysis and limitations of the data

Our meta-analysis was designed to allow an assessment of stressor interactions across indicators and ecosystems. In total we received data from 47 case studies, 8 of which were from the MARS lake (4) and river (4) experiments. Of the others, the majority reported on lakes (24) with lower number of data returns from river (12) and transitional waters (2). As such, our capacity to compare responses across ecosystem types were limited to qualitative assessments, as outlined below. The data returned represented case studies with data covering 3438 discrete sample sites producing a collective 370 years of monitoring data across 12 case studies spanning mesocosm to European region in scale.

For stressors data, we invited the data providers to consider stressor combinations that they expected to be important in their case studies. As such, our analysis is based, in part, on the assumption that the data providers had a comprehensive understanding of their case studies, and not simply, that they reported data on indicators that were available. In keeping with the Nõges et al. (2016) hypothesis, our data providers listed nutrients as the primary driver for all case studies. Nitrogen was classified as the primary stressor in 6 analyses and phosphorus in 33. In 4 of the analyses (from lake mesocosm experiments) nutrient stressors were classified as combined enrichment with N + P additions. However, the measure of the intensity of the stressors, here reported as enrichment with nitrogen, phosphorus, or a combination of both, varied between case studies. In total, 8 different stressor indicators for nutrients were reported. This presents a potential problem when comparing the effects of indicators of nutrient enrichment as a stressor. For example, we expect that phytoplankton responses to total P calculated as a growing season mean will differ when compared to total P calculated as an annual mean. A similar scenario was observed for secondary stressors, where a lack of standard methodology for reporting stressor intensity was apparent. Turunen et al. (2016) demonstrate well the issues associated with this approach. Firstly, the ecological response to any given stressor can vary along the stressor gradient where different, and at times opposing, responses can occur. Secondly, responses are expected to vary with the duration, or form, of the stressor where high intensity pulse pressures may elicit a different response when compared with long-term moderate intensity stressors. Similar consideration should be given to the indicators used to represent responses in the main organism groups. Indicators of phytoplankton responses were most commonly returned from our case studies, although more than 9 distinct indicators were included within this response category. Our analysis does not consider these artefacts of multiple stressor interactions or responses. Although this meta-analysis is the most comprehensive conducted to date with which commonalities in stressor-response relationships across multiple case studies can be examined, we acknowledge the limitations of the approach in that comparisons are generally qualitative in nature.

Similar to Nõges et al. (2016), hydromorphological stressors were commonly reported secondary stressor returned across the 47 case studies. Specifically, high flow (n=18), low flow (n=5) or morphology (n=3). High temperature (n=21) was, however, the single most commonly reported secondary stressor in our case studies. Our results are therefore biased towards the combination of nutrients as primary stressor with either temperature or high flow acting as secondary stressors. As such, our analysis will provide a relatively high degree of confidence in the occurrence of interactions between these stressors.

Indicator responses to common stressors across ecosystems

The synthesis suggested that the strong effects of nutrients on freshwater ecosystems exceed the effects of the second stressors considered here (high temperature, high flow, low flow and morphological alteration). Statistically significant responses to nutrients were detected in the great majority of analyses. By contrast, significant responses to the second stressors were only found in around 50% of the 21 analyses for high temperatures, in 2 out of 3 analyses for morphological alteration and very rarely in any of the 23 analyses of hydrological stress (extreme high or low flow).

The relative strength of nutrient effects may reflect a true dominant effect of nutrients or could be due to statistical biases common across studies. One possible reason for statistical bias is that direct measurement of nutrient concentrations in the water may have more precisely quantified the nutrient stress than commonly used measures of the second stressors. For example, measures of the second stressors such as air temperatures and precipitation amounts were often used to capture thermal or hydrological (flow) stress, but these might be relatively crude estimates of the stress to which freshwater organisms are actually exposed to in the water body. Another explanation for the dominance of nutrient effects may relate to the gradient lengths considered (Feld et al. 2016). For example, rainfall or flow rates naturally occurring in the observational studies may not have been sufficiently extreme to strongly affect ecosystem responses. This might also explain the apparent success of the studies in detecting significant effects of morphological stress. Although this stress was estimated in relatively crude indices of channelization or bed quality, the magnitude of extreme morphological alterations in the data may represent severe disturbance and so lead to a clear ecosystem response.

Among indicator types, we found consistently positive effects of nutrients, high temperature and low flow on phytoplankton responses. Most of the phytoplankton responses were measures of abundance of the whole community (chlorophyll a) or of cyanobacteria (chlorophyll-a or biovolume), this direction of response is, therefore, as expected with increasing nutrients and temperature and reduced flushing all leading to higher population growth rates. Conversely, fish, which were also most frequently abundance measures, generally responded negatively to these stressors (though not significantly so for high temperatures), potentially as a response to secondary effects of nutrients and temperature associated with increased phytoplankton abundance (e.g. high levels of phytoplankton respiration at night leading to deoxygenation, or higher temperatures reducing dissolved oxygen) and also direct and indirect effects of low flows. This consensus for these two biological groups across multiple studies suggests that even though statistically significant effects of high temperature and low flow were relatively rare in individual analyses, synthesizing data from multiple studies can reveal responses not otherwise detectable. The implication of this general pattern of responses is that increases in nutrient pollution, ongoing climate warming and droughts will all contribute to declines in fish and allow greater concentrations of phytoplankton, including cyanobacteria, to develop. Such shifts are likely to have negative implications for ecosystem services such as negative impacts on water supply and recreation due to harmful algal blooms and reduced quality of fisheries.

Surprisingly however, we found no consistent effect of high flow on phytoplankton, although the study of 26 lake time series did reveal highly significant negative effects of high flow on cyanobacteria abundance in short residence time lakes, highlighting that wetter summers will reduce the incidence of harmful algal blooms, and that increasing flushing in this lake type could be a management option.

Patterns in interaction types among multiple stressors across ecosystem types

Across the analyses included in the synthesis, statistically significant interactions between nutrients and the other stressors were relatively rare (8 of 47 studies) and exhibited no clear trend towards antagonistic or synergistic effects. This was apparent across all studies, when looking at each second stressor separately, for individual water body types and for different types of ecological response indicators. Although those results are contingent on the relatively small number of studies we were able to use for the synthesis, they suggest that interactions between nutrients and other stressors across European freshwater ecosystems are generally weak and that the general pattern is for multi-stressor effects to be additive. The implication of this is that management to ameliorate particular stressors should have beneficial outcomes that will not be impeded by strong antagonistic interactions with other stressors. However, a lack of synergistic interactions means amelioration of one stressor is unlikely to have better-than-expected outcomes on ecological status.

Previous meta-analyses have suggested that multi-stressor interactions are more common than we found here, and are dominated by antagonistic effects (Jackson et al. 2016). Though our study did not generally concur with that conclusion, we did find a pronounced but non-significant tendency for antagonistic effects of high flow on responses to nutrients (12 antagonistic vs 6 synergistic). A potential explanation for the difference between our findings and those of Jackson et al. (2016) is that their meta-analysis was based on paired comparisons of stressed vs unstressed conditions in predominantly controlled experimental conditions. Indeed, within our results antagonistic interactions dominated in the experimental studies (7 out of 8 analyses, though only 3 being statistically significant). However, the majority of our analyses exploited observational monitoring data collected across environmental gradients. This necessitated the use of statistical models based on regression to try to identify stressor effects and interactions. This approach may be less powerful for detecting stressor interactions than controlled experiments, where the data are less noisy, as responses may be influenced by several other factors not measured or included in the analyses. Different monitoring studies may also cover varying stressor gradients, which may lead to very different biological communities of varying sensitivities to stressors at higher or lower ends of the stressor gradients. For example, cold-water salmonid and white fish communities in oligotrophic waters in Northern Europe vs nutrient and temperature tolerant coarse fish communities in Central and Southern Europe. Our studies were also dominated by the response of phytoplankton, which may be particularly sensitive to nutrient effects, dominating all other stressor effects unless specific waterbody characteristics are taken into account. Nevertheless, if antagonistic stressor interactions were much more common than other interaction types, as suggested by Jackson et al. (2016), then it is surprising that this was not reflected in our analyses.

Assessment of water-body types that are sensitive or tolerant to specific multiple stressor scenarios

Due to the limited number of case-studies we are unable to offer comprehensive advice as to which water-body types are most sensitive or tolerant to specific stressor combinations. The sub-set analyses carried out in three of the studies highlighted three approaches worth considering to split datasets, based on background ecological understanding from exploratory analyses, to identify multiple stressor "typologies" more finely:

1) Restrict temporal scale of the analysis to periods of strongest response to stressors.

2) Focus on water body typologies, or "sensitivity factors", that are most sensitive to the stressors.

3) Restrict analyses to sites that show the greatest stress response (e.g. subset sites along the primary (dominant) stressor gradient, such as sites with low and high nutrient concentrations.

Temporal sensitivity to stressors is particularly important to consider in rivers, where flow may often have an over-arching effect on the biological response (Bowes et al. 2016). In the River Thames study, as in other published river studies, flow rates need to be low to allow dense phytoplankton populations to persist, so there is little value in including times of the year when flow limits population development. Similarly in lakes, it is often best to consider biological responses during periods of the year that are biologically active; particularly important in Northern and Central Europe when temperature and day-length may limit any biological response to stressors during winter time.

The importance of considering water body typologies, or “sensitivity factors”, that are most sensitive to the stressors was highlighted in the results from two of the European scale lake case studies examining the response of cyanobacteria to nutrients, temperature and rainfall (D5.1-4 ref). 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, but require consideration of which water-body types are most sensitive to stressors and where a site lies on the stressor gradients.

Recommendations on best–practice analytical and monitoring approaches to detect multiple stressors interactions in the future

Our study highlights that experimental approaches often provide the clearest signal of stressor interactions. They do not, however, provide a comprehensive understanding of how stressors interact in the real-world, over varying sites and stressor gradients. The range of responses in stressor interactions across all our case-studies highlight that it is often difficult to predict how two stressors may interact at a given site and both synergistic and antagonistic responses may be possible for the same stressor combination at sites with different characteristics or different levels of stress. Sometimes the significance of stressor effects, both acting singly or in combination, may be masked by other covariates either in different seasons or years (e.g. effects of nutrients may be masked by high flow in rivers) or at sites of differing typology (e.g. deep lakes may differ in sensitivity from shallow lakes). To overcome these challenges and understand how stressors interact in the real-world we recommend a number of approaches:

- Understand the environmental context (or typology) of the water body, and how factors such as flow/flushing, depth, alkalinity and humic type can greatly influence the sensitivity of the water body to stressors.
- Recognise that the shape of the interaction between stressors depends greatly on the gradient of the stressor in the water body over time or across a population of sites. This understanding can help to locate your water body within a “response landscape” allowing managers to better understanding how a site may respond to future changes in stressor levels.
- To gain further understanding, pool monitoring data from many sites and over time, i.e. carry out spatio-temporal studies. Combination of large spatial gradients and time series approaches to characterize within-site responses will give the most powerful way to use monitoring data to detect interactions.

- Account for variation in interactions along stressor gradients of other environmental gradients by data-subsetting, allowing random site slopes or using additional covariates and their interactions in any analysis of stressor interactions.
- Produce more sophisticated measures of local stress, especially for spatial studies. E.g. average temperature differences among sites is most likely not the best measure of temperature stress if communities or species are locally adapted. Instead, local temperature anomalies would be a better stress measure.
- Similarly communities may respond to extreme levels of stress, not average conditions over the season, and particularly physical stressors such as flow or temperature extremes. The cyanobacterial response to temperature in the lake time-series study highlighted the importance of considering the response to extreme temperatures, i.e. upper percentiles of response variable, not the mean response). The use of quantile regression to examine cyanobacteria responses to nutrient stress has also been demonstrated previously by Carvalho et al. (2013).
- Constrain statistical models by biological knowledge of response shapes and trajectories. For example, to ensure consistency between case-studies we did not include quadratic terms for curvilinear responses, but for an individual study more flexibility in model structure is possible. Responses may also be contingent on historical stress because of hysteresis. For example, responses to increasing nutrients may not be the same as responses to reducing nutrients. Therefore, knowledge of water body stress histories should be accounted for in statistical models, using data subsetting or including explicit effects of local stress history.
- Combine experiments with monitoring data. Experiments were more powerful at detecting stressor interactions. They can inform monitoring scheme designs and statistical model development.

Detecting recovery following abatement of multiple-stressors

Our understanding of the effects on ecological indicators of relieving single stressors when multiple stressors are known to be operating is poor. An improved understanding of the impact of stressor reduction is vital to evaluate the success of potential management options and underpin practical MARS guidance on river basin management planning (RBMP), as demonstrated for the Otra River, Norway, by Wright et al. (2017). Unfortunately, the data collated here were insufficient to allow examination of even primary stressor abatement and associated ecological responses, let alone an assessment of combined stressor abatement. A thorough review of recovery is outside the scope of this report. However, we do offer some considerations with respect to concepts for future work within MARS or elsewhere.

Our understanding of recovery processes across ecosystems lags behind our understanding of deterioration processes (Hering et al. 2013; Verdonschot et al. 2013). The meta-analysis presented in this report described the effects of multiple stressors on a suite of indicators but does not provide evidence of recovery or impact responses, necessarily. What is clear from the literature is that recovery is not necessarily the mirror image of degradation and that restoration (Jeppesen et al. 2005) efforts commonly fail as a result of poor understanding of the processes driving recovery (Feld et al. 2011). Few studies have attempted to assess commonalities in stressor abatement-response relationships within or across ecosystem types (Feld et al. 2011) and fewer still have assessed how recovery processes vary across large spatial scales in response to even single stressor abatement scenarios (e.g. Jeppesen et al. 2005). In order to determine the likelihood of recovery following multiple stressor abatement it is important that we use a suite of complementary approaches including experiments, field observations and modelling. Examples are available in the literature of stressor abatement effects confirming that ecological responses can be dependent upon interactions between multiple stressors. Baho et al. (2015) report on variable responses in zooplankton communities following changes in water level and nutrient loading that are dependent on both of these stressors and also on the occurrence of drought conditions. Guterrez et al. (2016) and Rolighed et al. (2016) confirmed using field data from Lake Søbygaard, Denmark, that recovery in large bodied zooplankton, following nutrient stressor abatement, was dependent upon climate warming. However, in an assessment of plankton responses to nutrient management and warming across 17 Danish lakes, Őzkan et al. (2016) reported that climate warming effects were weak relative to nutrient abatement effects at the regional scale. Specifically, the increase in zooplankton body size expected to occur following nutrient abatement was off-set as a result of increased abundance of small fish, and increased grazing of large bodied zooplankton, caused by increased temperatures. Novel model ensembles have been produced allowing new insights into the effects of multiple stressors and abatement strategies in lakes (Hu et al. 2016).

We reviewed the literature (e.g. Elliott et al. 2007; Feld et al. 2009, Kelly 1990; Mumby et al. 2013; Neimi et al. 2004) to identify suitable determinants of recovery that can be used to perform a comparative analysis across ecosystem types and temporal and spatial scales. In general, the concepts discussed here are well established in ecology and are reviewed by Grimm et al. (1992). Inherent to the success of this approach is a comprehensive understanding of the stressors acting on each case study system and the timing and magnitude of these stressors, both before and after abatement. There is an opportunity to consider the types of stressor scenarios occurring within the MARS case studies in this context. For example, an extreme weather event (pulse stress) in the presence of continuously high phosphorus loads (press stress) in rivers, as was demonstrated in the river experiment conducted by Bondar-Kunze et al. (2016). In this case the 'pulse' stressor could represent stressor abatement following the flood event although it is likely that changes in one stressor will alter the magnitude of the other as described above (Molinos and Donohue, 2010; Zhang et al. 2016). The analysis presented here offers a standard approach for assessing the impact of degradation and can also be used to compare pre- and post-stressor abatement effects on stressor-response relationships and interaction forms, where sufficient data are available. Similarly, standard techniques can, or have already been developed with which the recovery process can be quantified and compared across ecosystem types and response indicators. The maximum difference between pre- and post-abatement conditions can be used as a measure of recovery as can the period of time taken for no further significant change to occur in the reported response indicators (Spears et al. 2013). A novel analysis would combine these three approaches, where the frequency and intensity of multiple stressors were quantified, the forms of interactions between them identified and used to define the intensity of impact, and, following stressor abatement, the period of recovery and recovery end points are quantified.

References

Adrian, R., E. Papastergiadou, P. Zingel, M. Søndergaard, E. Jeppesen, D.G. Angeler. 2016. Sustainability, 7, 1142-1160.

Baho, D.L., Ü.N. Tavşanoğlu, M. Šorf, K. Stefanidis, S. Drakare, U. Scharfenberger, H. Agasild, M. Beklioğlu, J. Hejzlar,

Bondar-Kunze, E., S. Maier, D. Schönauer, N. Bahl., T. Hein. 2016. Antagonistic and synergistic effects on a stream periphyton community under the influence of pulsed flow velocity increase and nutrient enrichment. Science of the Total Environment. 573, 594-602.

Bowes, M.J, Loewenthal M, Read D.S., Hutchins M.G. Prudhomme C., Armstrong L.K., Harman S.A., Wickham H.D., Gozzard E. and Carvalho, L. (2016) Identifying multiple stressor controls on phytoplankton dynamics in the River Thames (UK) using high-frequency water quality data. Science of the Total Environment, 569: 1489-1499.

Carvalho L, McDonald C, de Hoyos C, Mischke U, Phillips G, Borics G, Poikane S., Skjelbred B, Lyche Solheim A, Van Wichelen J. & Cardoso A.C. 2013. Sustaining recreational quality of European lakes: minimising the health risks from algal blooms through phosphorus control. Journal of Applied Ecology, 50, 315-323.

Dunbar, M. 2013. Multiple hydro-ecological stressor interactions assessed using statistical models. PhD Thesis, University of Reading.

Elliott M., Burdon, D., Hemingway, K.L., Apitz, S.E. 2007. Estuarine, coastal and marine ecosystem restoration: Confusing management and science: A revision of concepts. Estuarine, Coastal and Shelf Science. 74, 349-366.

Feld, C., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Pedersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P., Friberg, N. 2011. From natural to degraded rivers and back again: a test of restoration ecology, theory and practice. Ecosystems in a Human-modified Landscape: A European Perspective, Ed: Guy Woodward, 2011, Academic Press, 374 pages.

Feld, C.K., da Silva, P.M., Sousa, J.P., de Bello, F., Bugter, R., Grandin, U., Hering, D., Lavorel, S., Mountford, O., Pardo, I., Partel, M., Rombke, J., Sandin, L., Jones K.B., Harrison, P. 2009. Indicators of biodiversity and ecosystem services: a synthesis across ecosystems and spatial scales. Oikos 118: 1862-1871

Feld, C.K., Segurado, P., Gutierrez-Canovas, C. 2016. Analysing the impact of multiple stressors in aquatic biomonitoring data: a 'cookbook' with applications in R. Science of the Total Environment. 573, 1320-1339.

Grimm, V., E. Schmidt, C. Wissel. 1992. On the application of stability concepts in ecology. Ecological Modelling, 63, 143-161.

Gutierrez, M.F., M. Devercelli, S. Brucet, T.L. Lauridsen, M. Søndergaard, E. Jeppesen. 2016. Is recovery of large-bodied zooplankton after nutrient loading reduction hampered by climate warming? A long-term study of shallow hypertrophic lake Søbygaard, Denmark, Water, 8, 341: doi:10.3390/w8080341

Hering, D., Borja, A., Carvalho, L., Feld, C. 2013. Assessment and recovery of European water bodies: key messages from the WISER project. Hydrobiologia, 704, 1-9.

Hu, F., K. Bolding, J. Bruggeman, E. Jeppesen, M.R. Flindt, L. van Gerven, J.H. Janse, A.B.G. Janssen, J.J. Kuiper, W.M. Mooij, D. Trolle. FABM_PCLake – linking aquatic ecology with hydrodynamics. Geosciences Model Development, 9, 2271-2278.

Jackson, M. C., Loewen, C. J. G., Vinebrooke, R. D., Chimimba, C. T., 2016. Net effects of multiple stressors in freshwater ecosystems: a meta-analysis. Global Change Biology 22, 180–189.

Jeppesen, E., Meerhoff, M., Jacobsen, B.A., Hansen, R.S., Sondergaard, M., Jensen, J.P., Lauridsen, T.L., Mazzeo, N., Branco, C.W.C., 2007. Restoration of shallow lakes by nutrient control and biomanipulation-the successful strategy varies with lake size and climate. Hydrobiologia 581, 269-285.

Molinos, J.G., Donohue, I. 2010. Interactions among temporal patterns determine the effects of multiple stressors Ecological Applications, 20, 1794-1800.

Moss, B., Kosten, S., Meerhoff, M., Battarbee, R.W., Jeppesen, E., Mazzeo, N., Havens, K., Lacerot, G., Lie, Z., De Meester, L., Paerl, H., Scheffer, M. Allied Attack: climate change and eutrophication. Inland Waters. 1, 101-105.

Mumby, P. J., Chollett, I., Bozec, Y., Wolff, N.H. 2013. Ecological resilience, robustness and vulnerability: how do these concepts benefit ecosystem management? Current Opinion in Environmental Sustainability, 7, 22–27.

Nakagawa, S., Schielzeth, H. 2013. A general and simple method for obtaining R2 from generalized linear mixed-effects models. Methods in Ecology and Evolution. 4(2): 133-142.

Niemi, G., Wardrop, D., Brooks, R., Anderson, S., Brady, V., Paerl, H., Rakocinski, C., Brouwer, M., Levinson, B., McDonald, M. 2004. Rationale for a New Generation of Indicators for Coastal Waters. Environmental Health Perspectives, 112, 979-986.

Nõges, P., Argillier, C., Borja, A., Garmendia, J.M., Hanganu, J., Kodes, V., Pletterbauer, F., Saguois, A,. Birk, S. Quantified biotic and abiotic responses to multiple stress in freshwater, marine and ground waters. Science of the total Environment. 540, 43-52.

Őzkan, K., E. Jeppesen, T.A. Davidson, R. Bjerring, L.S. Johansson, M. Sondergaard, T.L. Lauridsen, J.-C. Svenning. 2016. Long-term trends and temporal synchrony in planktonic richness, diversity and biomass driver by re-oligotrophication and climate across 17 Danish lakes. Water, 8, 427: doi:10.3390/w8100427

Piggott, J.J., Townsend, C.R., Matthaei, C.D. 2015. Reconceptualising synergism and antagonism among multiple stressors. Ecology and Evolution. 5, 1538-1547.

Rolighed, J., E. Jeppesen, M. Søndergaard, R. Bjerring, J.H. Janse, W.M. Mooij, D. Trolle. 2016. Climate Change Will Make Recovery from Eutrophication More Difficult in Shallow Danish Lake Søbygaard. Water, 8, 459, doi:10.3390/w8100459

Scheffer, M, Carpenter, S., Foley, J.A., Foke, C., Walker, B. 2001. Catastrophic shifts in ecosystems. Nature. 413, 591-596

Schindler, D.W., Carpenter, S.R., Chapra, S.C., Hecky, R., Orihel, D.M. 2016. Reducing phosphorus to curb lake eutrophication is a success. Environmental Science and Technology. 50, 8923-8929.

Spears, B.M., Carvalho, L., Dudley, B., May, L. 2013. Variation in chlorophyll a to total phosphorus ratio across 94 UK and Irish lakes: implications for lake management. Journal of Environmental Management. 115, 287-294.

Spears, B.M., M. Lurling, S. Yasseri, A.T Castro-Castellon, M. Gibbs, S. Meis, C. McDonald, J. McIntosh, D. Sleep, F. Van Oosterhout. 2013. Lake responses following lanthanum-modified bentonite clay (Phoslock®) application: an analysis of water column lanthanum data from 16 case study lakes. Water Research, 47(15):5930-42.

Tockner, K., Pusch, M., Borchardt, D., Lorang, M.S. 2010. Multiple stressors in coupled river-floodplain ecosystems. Freshwater Biology. 55, 135-151.

Turunen, J., T. Muotka, K.-M. Vuori, S.M. Karjalainen, J. Rääpysjärvi, T. Sutela, J. Aroviita. 2016. Disentangling the responses of boreal stream assemblages to low stressor levels of diffuse pollution and altered channel morphology. Science of the Total Environment, 544, 954-962.

Verdonschot, P.F.M., Spears, B.M., Feld, C.K., Brucet, S., Keizer-Vlek, H., Borja, A., Elliott, M., Kernan, M., Johnson R.K. 2013. A comparative review of recovery processes in rivers, lakes, estuarine and coastal waters. Hydrobiologia. 704, 453-474.

Vinebrooke, R. D., Cottingham, K. L., Norberg, J., Scheffer, M. J., Dodson, S. I, Maberly, S. C. and Sommer, U. 2004. Impacts of multiple stressors on biodiversity and ecosystem functioning: the role of species co-tolerance. Oikos, 104, 451–457.

Wright, R.F., R.-M. Couture, A.B. Christiansen, J.-L. Guerrero, Ø. Kaste. B.T. Barlaup. 2017. Effects of multiple stresses hydropower, acid deposition and climate change on water chemistry and salmon populations in the River Otra, Norway. Science of the Total Environment, 574, 128-138.

Zhang Z., Y. Caio, E. Jeppesen, W. Li. 2016. The response of Vallisneria spinulosa and plankton to pulse addition of organic nitrogen with different loading patterns. Hydrobiologia, 767, 175-184.

Appendix 1. Full details of the statistical analysis workflow, including R code

Model construction

Choice of response variables

To address the common questions, each analysis will quantify and test responses of the MARS benchmark indicators or responses from individual species abundance and the abundance of functional groups. Responses being investigated by WPs are collated in the spreadsheet "Feedback_WP345_stressors_responses.xlsx". The form of the response varies in that some are continuous variables (e.g. chlorophyll-*a*) and some are categorical (e.g. status class). This form will affect the type of analysis available (see later).

Choice of stressor variables

Based on the common questions, each analysis will consider responses to two of the following stressors:

1. High temperature (Q1)
2. Low flow (Q2)
3. High flow (Q3)
4. High nutrients (Q1-4)
5. Morphological change (Q4)

We require individual analysts to use their knowledge of their system to decide upon (and fully document) the measures of the relevant stressors that they use for their own particular analyses. The choice of stressor variables is one of the most important steps in the analysis. To choose stressors, we encourage analysts to draw on their understanding of ecological processes in their study system and also carry out preliminary exploratory analyses such as plotting and correlation analyses.

WP6 requires, however, some standardisation, so please make sure that high values of the chosen measure indicate high stress – i.e. the system being pushed outside of its normal limits. Some measures of exposure to stress that you may wish to consider are:

- The average or overall value of the stressor during a biologically-relevant time period (e.g. total phosphorus in the growing season).
- The maximum value of the stressor during a biologically-relevant time period (e.g. maximum temperature in July).
- The total time when the stressor is above a threshold (e.g. a historically high value, biologically-determined critical value) during a biologically-relevant time period (e.g. number of days in summer below the historical 10th percentile low flow threshold).
- Measures of cumulative exposure to stressful conditions beyond a threshold (e.g. growing degree days with base temperature reflecting a critical temperature above which the system is under stress).

For spatial data (multi-site but no temporal component), long term average measures of stress are likely to be most useful. For multi-year data (single or multiple site), to provide comparative analyses for WP6, each year should provide one data point for stress per site. If your data are at higher temporal resolution, for WP6 you will need to pre-process it to an annual level.

If any stressor variable is categorical it is important that the variable has **only two levels** representing unstressed vs. stressed conditions. Please ensure these are correctly specified in R as factors whose first recognised level relates to the unstressed condition. For example, if the stressor variable x1 has two levels labelled in the data as 'control' and 'stress.treatment' use:

```
x = factor(x, levels=c("control", "stress.treatment"))
```

Variable transformations

To aid model convergence and reduce model heteroscedasticity, all **continuous variables** (responses and stressor variables) should be transformed to improve their conformance to normal distributions. Any non-continuous response variables (e.g. binary, counts, categories) or categorical stressor variables (factors) should not be transformed, but instead will require generalised modelling techniques (see below).

We suggest using a version of the Box-Cox transformation offset to ensure strict positivity (i.e. all values > 0). An advantage of the Box-Cox transform is that if the data are already close to normally distributed then little transformation will be applied. Further, we can back-transform from the transformed variable to the original scale of the data, provided the parameters are saved.

Please check the results of the transformation are roughly normally distributed by plotting histograms. If the data exhibit extreme skew because of extreme outliers, the transform may not work well and you might want to consider excluding the outliers from the analysis. If this occurs, please contact us.

Following Box-Cox transformation, each transformed variable will be centred and scaled, so they have a mean of zero and standard deviation/variance of one. This is principally needed for the continuous stressor variables and should aid model convergence.

R functions to estimate and apply the variable transformation (after removing missing values, see Section 2):

```
estimateBC = function(x){
# function to estimate transformation parameters for continuous variable x
  require(car)
  gamma = min(x, na.rm=T) - 0.001 # offset (min value minus a small number)
  x = x - gamma # subtract gamma from x, so that it is strictly positive
  lambda = powerTransform(x~1, family="bcPower")$lambda # estimate lambda
of Box-Cox transformation...
  xT = bcPower(x, lambda=lambda) # apply box-cox transform
  xT.mean = mean(xT) # mean of transformed values, for centring
  xT.sd = sd(xT) # sd of transformed values, for scaling
  # return the transformation parameters
  return(c(gamma=gamma, lambda=lambda, xT.mean=xT.mean, xT.sd=xT.sd))
}

applyBC = function(x, P=estimateBC(x)){
```

```
# function to transform continuous variable x using transformation
parameters P
  require(car)
  gamma = P[1]
  lambda = P[2]
  xT.mean = P[3]
  xT.sd = P[4]
  xT = bcPower(x-gamma, lambda) # apply box-cox transform
  xT = (xT-xT.mean)/xT.sd # centre and scale
  return(xT)
}
```

To transform the continuous response variable (y) and the continuous stressor variables (x1 and x2):

```
library(car)
P.y = estimateBC(y)
yT = applyBC(y, P.y)

P.x1 = estimateBC(x1)
x1T = applyBC(x1, P.x1)

P.x2 = estimateBC(x2)
x2T = applyBC(x2, P.x2)
```

Please ensure you have installed the latest version of the R 'car' package (2.1-1 at the time of writing, use R function `update.packages` if necessary). If you encounter any errors please contact us (Dan Chapman dcha@ceh.ac.uk).

Choose the type of model to use

The type of statistical model you fit will depend on two major considerations:

- The type of system (which determines whether a mixed model with random effects is needed).
- The type of response data (which determines whether a generalised model is needed).

For each type of system, the table below shows whether a mixed model with random effects is required.

Table 3: Summary of model choice criteria

System	Is a mixed model (with random effects) needed?
Mesocosm experiment	Possibly, depending on experimental design. Grouping factors such as block or measurement period should be included as random effects.
Single-site, multi-year (temporal)	No
Multi-site, multi-year (spatio-temporal)	Yes. Random effects of site and year should be included.
Multi-site, single-year (spatial)	No

The response variables will most likely take one of four types, which will determine the type of model to use:

- Continuous – can take any value (possibly within a range);
- Binary – can only take one of two categories, e.g. 0/1, presence/absence, A/B;
- Count – integers from 0 to ∞, e.g. population sizes of a species (If you have very high count values you may wish to treat the data as a continuous variable);

Ordered categories – a discrete scale that ranks data, e.g. bad/poor/moderate/good/high status, plant cover-abundance scales.

If the response variable is ordered categorical, please ensure it is represented as an ordered factor in R. This can be done using R code similar to:

```
y=factor(y, levels=c("bad","poor","moderate","good","high"), ordered=TRUE)
```

Please contact us if your response variables are of a different type (Dan Chapman dcha@ceh.ac.uk).

For each response variable type, the table below indicates the type of model to use. Generic R code is given to fit the models. See the footnote for explanation of the symbols in the code.

Response variable type	**Linear model (no random effects)**	**Mixed effects model (with random effects)**

Continuous	Generalised linear model (GLM)
	`M = glm(yT ~ x1T*x2T)`
		Generalised linear mixed model (GLMM)
		`library(lmerTest)`
		`M = lmer(yT ~ x1T*x2T + (1|RE1) + (1|RE2), REML=FALSE)`

Binary	Generalised linear model (GLM)
	`M = glm(y ~ x1T*x2T, family="binomial")`
		Generalised linear mixed model (GLMM)
		`library(lme4)`
		`M = glmer(y ~ x1T*x2T + (1|RE1) + (1|RE2), family="binomial")`

Count	Generalised linear model (GLM)
	`M = glm(y ~ x1T*x2T, family="poisson")`
		Generalised linear mixed model (GLMM)
		`library(lme4)`
		`M = glmer(y ~ x1T*x2T + (1|RE1) + (1|RE2), family="poisson")`

Ordered categories	Cumulative link model (CLM)
	`library(ordinal)`
	`M = clm(y ~ x1T*x2T)`
		Cumulative link mixed model (CLMM)
		`library(ordinal)`
		`M = clmm(y ~ x1T*x2T + (1|RE1) + (1|RE2))`

`M` = the fitted model

`y` = the response variable

`yT` = transformed values of the response variable

`x1T, x2T` = transformed values of the stressor variables

`RE1, RE2` = random effects (grouping factors, e.g. block, site, year)

For the mixed effects models (lmer, glmer, clmm) the code in the table above specifies random intercept terms for each level of the random effects. Random intercepts allow mean values of the response to vary among levels of the associated grouping factor (e.g. site or year).

It is also possible to model random slopes, which allow the relationship between response and stressor to vary among levels of the associated grouping factor i.e. the slope of a relationship is allowed to be steeper in some lakes than others if a random slope with respect to lake is included. If you are confident with mixed effects models you may also wish to consider including random slopes for one or more of the random effects. For example, if you use the formula y ~ x1T*x2T + (1|RE1) + (1+x1T*x2T|RE2) then the model will estimate random intercepts for every level of factor RE1 and random intercepts and slopes for every level of factor RE2. Model AIC can be used to compare and identify optimal random effect specifications.

Test and correct for residual autocorrelation

Next, you should test whether the model residuals contain strong spatial or temporal autocorrelation. Autocorrelation means that the residuals of observations close in space or time are more similar than expected by chance. This indicates that model assumptions have been violated and can cause the statistical significance of model terms to be exaggerated.

Testing for autocorrelation is only necessary for linear models without random effects, i.e. for analysis of single-site, multi-year data (temporal autocorrelation) and multi-site, single-year data (spatial autocorrelation). For the mixed models, random effects should account for grouping in space and time.

System	Do you need to test for residual autocorrelation?
Mesocosm experiment	No. Good experimental design and/or random effects should account for grouping in space and time.
Single-site, multi-year	Yes (temporal autocorrelation).
Multi-site, multi-year	No. Random effects should account for grouping in space and time.
Multi-site, single-year	Yes (spatial autocorrelation).

To test for residual autocorrelation, you first need to extract residuals from the fitted model. For consistency across models, we will use response residuals (the observed response minus the fitted response).

Response variable type	Linear model (no random effects)	How to calculate response residuals (r)
Continuous	`M = glm(yT ~ x1T*x2T)`	`r = residuals(M, type="response")`
Binary	`M = glm(y ~ x1T*x2T, family="binomial")`	`r = residuals(M, type="response")`
Count	`M = glm(y ~ x1T*x2T, family="poisson")`	`r = residuals(M, type="response")`
Ordered	`M = clm(y ~ x1T*x2T)`	`classProbs = predict(M,`

categories		newdata=data.frame(x1T,x2T))$fit # matrix of fitted probabilities for each category classObs = sapply(levels(y), function(x) { as.numeric(x==y) }) # binary dummy matrix of observed categories r = classObs - classProbs # response residuals for each category

To test for residual autocorrelation you will use the Moran test, based on the *I* statistic. Note that for models of categorical data, separate residuals are obtained for each category. Therefore you will need to perform a separate autocorrelation test for each category.

To test for **temporal autocorrelation** in the residuals:

1. Create inverse weights from the differences in years between each pair of observations:

```
w = 1/as.matrix(dist(year))
diag(w) = 0
```

2. Perform Moran's test on the residuals (r):

```
library(ape)
Moran.I(x=r, weight=w)
```

To test for **spatial autocorrelation** in the residuals:

1. Calculate the distances between each pair of sites, using their decimal longitudes and latitudes:

```
library(sp)
d = spDists(cbind(lon,lat), longlat=T)
```

2. Convert the distance matrix into inverse weights:

```
w = 1/d
diag(w) = 0
```

3. Perform Moran's test on the residuals (r):

```
library(ape)
Moran.I(x=r, weight=w)
```

For large datasets, very small levels of autocorrelation can be statistically significant, but are unlikely to affect the model conclusions. Therefore we suggest substantive problems will be indicated by Moran's *I* > 0.1.

If strong autocorrelation is detected, you should attempt to correct for it using a trend surface. A trend surface is a flexible function of either year or space that can capture trends in the response variable not explained by the model explanatory variables.

For non-categorical response variables (continuous, binary or count), trend surfaces will be generated using smoothing splines available from the *mgcv* R package. It is not possible to do this for models of categorical responses, so in this case we will include a polynomial function of year or space.

To implement the **temporal** trend surface linear models:

Response variable type	How to estimate the temporal trend surface of the linear model	How to update the linear model with the temporal trend surface

Continuous	`library(mgcv)`
	`G = gam(yT ~ x1T*x2T + s(year))`
	`trendSurface = predict(G, newdata=data.frame(x1T=0,x2T=0,year)) - coef(G)[1]`
		`M = update(M, offset=trendSurface)`

Binary	`library(mgcv)`
	`G = gam(y ~ x1T*x2T + s(year), family="binomial")`
	`trendSurface = predict(G, newdata=data.frame(x1T=0,x2T=0,year)) - coef(G)[1]`
		`M = update(M, offset=trendSurface)`

Count	`library(mgcv)`
	`G = gam(y ~ x1T*x2T + s(year), family="poisson")`
	`trendSurface = predict(G, newdata=data.frame(x1T=0,x2T=0,year)) - coef(G)[1]`
		`M = update(M, offset=trendSurface)`

Ordered categories	Not needed	`M = clm(y ~ x1T*x2T + poly(year,3))`

To implement the **spatial** trend surface linear models:

Response variable type	How to estimate the spatial trend surface of the linear model	How to update the linear model with the spatial trend surface

Continuous	`library(mgcv)`
	`G = gam(yT ~ x1T*x2T +s(lon,lat))`
	`trendSurface = predict(G, newdata=data.frame(x1T=0,x2T=0, lon,lat)) - coef(G)[1]`
		`M = update(M, offset=trendSurface)`

Binary	`library(mgcv)`
	`G = gam(y ~ x1T*x2T + s(lon,lat), family="binomial")`
	`trendSurface = predict(G, newdata=data.frame(x1T=0,x2T=0, lon,lat)) - coef(G)[1]`
		`M = update(M, offset=trendSurface)`

Count	`library(mgcv)`
	`G = gam(y ~ x1T*x2T + s(lon,lat), family="poisson")`
	`trendSurface = predict(G, newdata=data.frame(x1T=0,x2T=0, lon,lat)) - coef(G)[1]`
		`M = update(M, offset=trendSurface)`

Ordered categories	Not needed	`M = clm(y ~ x1T*x2T + poly(lon,3)*poly(lat,3))`

After inclusion of the trend surface, Please re-calculate the residual autocorrelation of the updated models to confirm that the problems have been reduced.

Model evaluation

To evaluate the final model, please report the following:

1. The marginal and conditional model R^2. Marginal R^2 uses only the fixed effects, while the conditional R^2 also includes the random effects (Note this is not possible for models of categorical data):

```
library(MuMIn)
r.squaredGLMM(M)
```

2. The Pearson's product-moment correlation between the model response residuals (r) and the fitted values. Code for extracting residuals from most model types is given in the sections on autocorrelation. For GLMMs, you can use `r = residuals(M, type="response")`. For the linear mixed model

of ordered categorical responses (CLMM), residuals cannot be extracted so no test can be performed. To test the correlation:

```
cor.test(r, fitted(M))
```

For the linear model of ordered categorical responses (CLM), where there are residuals for each category calculated as in the tables above, use:

```
for(i in 1:ncol(r)) print(cor.test(r[,i], classProbs[,i]))
```

3. The Shapiro-Wilk test for normality of residuals (r). Remember that models for categorical responses have residuals for each different category and require separate testing. The test should be non-significant, unless the dataset is very large in which case very small deviations from normality are statistically significant:

```
shapiro.test(r)
```

For the linear model of ordered categorical responses (CLM), where there are residuals for each category, use:

```
apply(r, 1, shapiro.test)
```

For the linear mixed model of ordered categorical responses (CLMM), residuals cannot be extracted so no test can be performed.

Determine importance of the interaction term

Three simple methods to estimate the importance of the interaction term will be used:

1. The Z-score for the interaction term (large absolute values indicate an important interaction):

```
(summary(M)$coef[,1]/summary(M)$coef[,2])["x1T:x2T"]
```

2. The change in model AIC if the interaction is removed (large positive values indicate an important interaction):

```
AIC(update(M, ~.-x1T:x2T)) - AIC(M)
```

3. The drop in marginal and conditional R^2 if the interaction is removed (large negative values indicate an important interaction):

```
r.squaredGLMM(update(M, ~.-x1T:x2T)) - r.squaredGLMM(M)
```

Note that R^2 cannot be calculated for models of categorical responses.

Visualise the interaction

The fitted model equation can be used to plot the response surface for the two main stressors, x1T and x2T.

First, you will need to extract the model fixed effect coefficients:

Model function used	Fitted fixed effect for model object M
glm	`B = coef(M)`
lmer	`B = fixef(M)`
glmer	`B = fixef(M)`

clm	`B = c(coef(M)[1], coef(M)[c("x1T","x2T","x1T:x2T")])`†
clmm	`B = c(coef(M)[1], coef(M)[c("x1T","x2T","x1T:x2T")])`†

† This code plots the transition between the first two ordered categories of the response variable. Change [1] to [2], [3], etc. to plot subsequent transitions.

We have written a function `interactionPlot()` to plot the response from a fitted model:

```
interactionPlot = function(B, X1, X2, Y,
      TP=list(P.x1=if(exists("P.x1")) P.x1 else NA,
              P.x2=if(exists("P.x2")) P.x2 else NA,
              P.y=if(exists("P.y")) P.y else NA,
              family="gaussian"),
      responseLab="z", x1Lab="x1", x2Lab="x2") {
# Function to plot interactions from fitted models.
# B = vector of model fixed effect coeffcients
# X1, X2 = vectors with values of the stressors (not transformed)
# Y = vector with values of the response (not transformed)
# TP = list of transformation parameters containing the following elements;
#        P.x1 = output from estimateBC() for x1, or NA if no transformation
applied
#        P.x2 = output from estimateBC() for x2, or NA if no transformation
applied
#         P.y = output from estimateBC() for y, or NA if no transformation
applied
#       family = family of the generalised model. It should be one of
#               "gaussian" (for continuous response),
#               "poisson" (for count response),
#               "binomial" (for binary or ordered categorical response), or
#                 NA (if you want to plot the response on the linear model
scale)
# responseLab = label for the response variables
# x1Lab, x2Lab = labels for the stressor variables

  require(emdbook)
  if(is.numeric(X1) & is.numeric(X2)){ # X1 and X2 are both continuous
    myF <<- function(X1=X1, X2=X2, transPar=TP) {
      if(sum(is.na(transPar$P.x1))==0) X1 = applyBC(X1, transPar$P.x1)
      if(sum(is.na(transPar$P.x2))==0) X2 = applyBC(X2, transPar$P.x2)
      z = B[1] + B[2]*X1 + B[3]*X2 + B[4]*X1*X2
      if(!is.na(transPar$family)){
        if(transPar$family=="gaussian") z = backBC(z, transPar$P.y)
        if(transPar$family=="poisson") z = exp(z)
        if(transPar$family=="binomial") z = 1/(1+exp(-z))
      }
      return(z)
    }
    curve3d(myF, from=c(min(X1),min(X2)), to=c(max(X1),max(X2)),
            sys3d="image",
col=colorRampPalette(c("blue","white","red"))(64),
            xlab=x1Lab,            ylab=x2Lab,            main=responseLab,
varnames=c("X1","X2"))
    points(X1, X2, pch=21, bg="grey50")
    curve3d(myF,      from=c(min(X1),min(X2)),       to=c(max(X1),max(X2)),
sys3d="contour",
            add=T, labcex=1, varnames=c("X1","X2"))
    box()
  }
```

```
  if(is.numeric(X1) & is.factor(X2)){ # X1 continuous, X2 is a factor
    myF <<- function(X1, X2=0, transPar=TP) {
      if(sum(is.na(TP$P.x1))==0) X1 = applyBC(X1, TP$P.x1)
      z = B[1] + B[2]*X1 + B[3]*X2 + B[4]*X1*X2
      if(!is.na(TP$family)){
        if(TP$family=="gaussian") z = backBC(z, TP$P.y)
        if(TP$family=="poisson") z = exp(z)
        if(TP$family=="binomial") z = 1/(1+exp(-z))
      }
      return(z)
    }
    curve(myF(X1=x, X2=0), from=min(X1), to=max(X1), col="blue", lwd=2,
xlab=x1Lab, ylab=responseLab,
          ylim=if(is.numeric(y) & !is.na(TP$family)) range(y) else NULL)
    curve(myF(X1=x, X2=1), add=T, col="red", lwd=2, lty=2)
    rug(x1)
    legend("topright", legend=levels(X2), lty=1:2, lwd=2,
col=c("blue","red"), title=x2Lab)
  }

  if(is.factor(X1) & is.numeric(X2)){ # X1 is a factor, X2 continuous
    myF <<- function(X1=0, X2, transPar=TP) {
      if(sum(is.na(TP$P.x2))==0) X2 = applyBC(X2, TP$P.x2)
      z = B[1] + B[2]*X1 + B[3]*X2 + B[4]*X1*X2
      if(!is.na(TP$family)){
        if(TP$family=="gaussian") z = backBC(z, TP$P.y)
        if(TP$family=="poisson") z = exp(z)
        if(TP$family=="binomial") z = 1/(1+exp(-z))
      }
      return(z)
    }
    curve(myF(X1=0, X2=x), from=min(X2), to=max(X2), col="blue", lwd=2,
xlab=x2Lab,
          ylab=responseLab, ylim=if(is.numeric(y) & !is.na(TP$family))
range(y) else NULL)
    curve(myF(X1=1, X2=x), add=T, col="red", lwd=2, lty=2)
    rug(x2)
    legend("topright", legend=levels(X1), lty=1:2, lwd=2,
col=c("blue","red"),
           title=x1Lab)
  }

  if(is.factor(X1) & is.factor(X2)){ # X1 is a factor, X2 is a factor
    z = c(B[1], B[1]+B[2], B[1]+B[3], B[1]+B[2]+B[3], sum(B))
    names(z) = c(paste(levels(X1)[1], levels(X2)[1], sep=" / "),
      paste(levels(X1)[2], levels(X2)[1], sep=" / "),
      paste(levels(X1)[1], levels(X2)[2], sep=" / "),
      paste("E(", paste(levels(X1)[2], levels(X2)[2], sep=" + "), ")",
sep=""),
      paste(levels(X1)[2], levels(X2)[2], sep=" x "))
    #names(z) = paste(rep(levels(X1),2), rep(levels(X2),each=2), sep=" / ")
    if(!is.na(TP$family)){
      if(TP$family=="gaussian") z = backBC(z, TP$P.y)
      if(TP$family=="poisson") z = exp(z)
      if(TP$family=="binomial") z = 1/(1+exp(-z))
    }
    z2 = z - z[1] # z values minus the control
    par(mfrow=c(2,1))
    barplot(z, ylab=responseLab, xlab=paste(x1Lab,x2Lab,sep=" / "))
    abline(h=0)
    barplot(z2, ylab=paste(responseLab,"- control"),
xlab=paste(x1Lab,x2Lab,sep=" / "))
```

```
    abline(h=0)
    par(mfrow=c(1,1))
  }

  return(NULL)
}
```

To use `interactionPlot()` you first need to create a list that holds the variable transformation parameters (outputs of `estimateBC()` if continuous variables or NA for factors where no transformation is applied) and the family of model, which determines how to transform the predicted response variables ('gaussian', 'poisson' or 'binomial' to plot responses on the scale of the raw data, or NA to plot responses on the scale of the model linear function). For example, if both stressors and response are continuous, so a Gaussian model was used:

```
myTP = list(P.x1=P.x1, P.x2=P.x2, P.y=P.y, family="gaussian")

print(myTP)

library(emdbook)

interactionPlot(B=B, X1=x1, X2=x2, Y=y, TP=myTP, responseLab="z (actual
values)", x1Lab="x1", x2Lab="x2")
```

To make the equivalent plot on the linear scale of the model:

```
myTP = list(P.x1=P.x1, P.x2=P.x2, P.y=P.y, family=NA)

interactionPlot(B=B, X1=x1, X2=x2, Y=y, TP=myTP, responseLab="z (model
scale)", x1Lab="x1", x2Lab="x2")
```

Interaction classification

The type of interaction will be characterised from the fitted model fixed effect coefficients. We are still deciding on the final classification scheme following discussions in Fulda. Although we may apply more complicated classification schemes later in the synthesis, at this stage you may wish to assign the interaction to one of the following 3 categories:

Type of interaction	Characterisation
Synergistic	Regression slopes for x1T, x2T and their interaction all have the same sign (i.e. all positive or all negative)
Antagonistic	Regression slopes for x1T and x2T have the same sign, but their interaction has the opposite sign
Opposing	Regression slopes for x1T and x2T differ, sign of the interaction term not important

Advanced model selection

As well as the model developed above, we also request that analysts consider fitting models that include additional stressor variables or habitat characteristics that they think may be important drivers of the response in their system. In this case, please report models for both:

1. **The basic model:** Only the two key stressors and their interaction (the model developed above)
2. **A fuller model:** The key stressors and their interaction, as well as fixed effects of other explanatory variables representing key local drivers (model selection will be needed)

Please note that additional explanatory variables added to the analysis cannot also add missing data, or the comparison of models will not be valid. So, to run these models, analysts should construct a data set of response, stressor and additional explanatory variables and any random effects factors with no missing values prior to analysis.

For model selection in the second analysis, start with the basic model and use a forward stepwise procedure as follows:

1. For each candidate explanatory variable, add it to the current model (e.g. using R's `update` function) and calculate the change in AIC (R's `AIC` function).
2. If at least one variable addition reduced model AIC, permanently update the current model to include the candidate explanatory variable that most reduced AIC, and return to step 1.
3. If no variable additions reduced AIC, stop the stepwise addition and retain the current model.

The fuller model should be analysed and reported in the same way as the basic model.

Reporting

An Excel spreadsheet containing a form for reporting meta-data about the study and details of the analysis has been provided (MARS_WP6_Synthesis_reporting_form_15042016.xlsx).

The information needed to fill in the form can be copied and pasted from the R console output using the computer mouse, or the R command `write.table(x, "clipboard", sep="\t", row.names=F)`.

In some cases, the same analyst may perform more than one analysis. For example, if you:

1. Address more than one of the common questions with your dataset,
2. Analyse more than one response variable for the same common question, or
3. Fit both a basic model (fixed effects of the two main stressors only) and a fuller model (fixed effects of the two main stressors as well as other important explanatory variables) for the same common question and response variable.

In this case, we request a separate reporting form is completed for each individual analysis. These can be saved as multiple sheets in the same Excel file. Please email your completed reporting spreadsheets to Dan Chapman dcha@ceh.ac.uk.

We would also be grateful if you could save your R workspace for each analysis as an .Rdata file, for example using the R function `save.image()` and then email that file to us alongside the Excel form. This will allow us to run additional model checks and clarify any queries we have about the reporting form. Please note that the .Rdata file would by default contain your data. This would be treated in confidence and not used for any purpose outside WP6.

Appendix 2. Summary of the studies used in the synthesis

Table A1. Summary of the analyses used in the synthesis. Model effects are summarized by the z-score indicating the direction and significance of the effect. Statistically significant z scores (*P* < 0.05) are in bold. Field codings as follows. Region: C = central Europe, N = northern Europe, S = southern Europe. Study type: S = spatial, S-T = spatio-temporal, ME = mesocosm experiment. Site type: R = river, L = lake, TW = transitional waters, FM = flume mesocosm, LM = lake mesocosm. Second stressor: T = high temperature, HF = high flow, LF = low flow, M = morphology. Model: GLM = generalized linear model, GLMM = generalized linear mixed model, (G) = Gaussian errors, (P) = Poisson errors. Interaction types: A = antagonism, S = synergy, O = opposing.

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor effect (z)	Stressor interaction (z)	Interaction type	Interaction wrt nutrients
1	Ruhr Basin	C	S	R	Ecological status	Ecological Quality Ratio	1	Total N (mg/l)	M	Quality of the bed structure index	222	No	GLM (G)	0.222	**-3.51**	**-6.26**	0.86	A	A
2	Thames Basin	N	S-T	R	Phytoplankton	Chla growing season	8	Total P over growing season	T	Water degree days above 9 °C	123	No	GLMM (G)	0.325	**2.24**	**4.53**	0.60	S	S
2	Thames Basin	N	S-T	R	Phytoplankton	Chla growing season	8	Total P over growing season	HF	Number of high flow pulses	124	No	GLMM (G)	0.173	**3.60**	-0.34	**2.51**	O	S
2	Thames Basin	N	S-T	R	Phytoplankton	Chla growing season	8	Total P over growing season	HF	High flow pulses duration	124	No	GLMM (G)	0.170	**3.47**	1.34	1.00	S	S
2	Thames Basin	N	S-T	R	Phytoplankton	Chla growing season	8	Total P over growing season	LF	Number of low flow pulses	124	No	GLMM (G)	0.142	**2.95**	0.75	-0.09	A	A

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor	Stressor interaction	Interaction type	Interaction wrt nutrients
2	Thames Basin	N	S-T	R	Phytoplankton	Chla growing season	8	Total P over growing season	LF	Low flow pulses duration	124	No	GLMM (G)	0.137	**2.86**	1.29	1.05	S	S
3	Finnish Lakes	N	S	R	Macrophytes	Ecological Quality Ratio (EQR) of macrophytes	1	Average total P in water during ice-free period	M	Channelization intensity index (0-2)	128	No	GLM (G)	0.073	**-2.88**	-1.20	0.02	A	A
4	Danube Delta lakes	S	S-T	L	Fish	Fish biomass (g/m2 dry)	15	Total N (mg/l)	LF	Residence time ratio (month residence time / year residence time)	145	No	GLMM (G)	0.042	**-2.42**	-0.06	1.72	A	A
5	Multiple stressors in lakes at European scale	N	S-T	L	Phytoplankton	Cholorophyll a (ug/l)	8	Total P (µg/l)	T	Air temperature (°C)	1066	No	GLMM (G)	0.480	**21.92**	**6.78**	**4.73**	S	S
5	Multiple stressors in lakes at European scale	N	S-T	L	Phytoplankton	PTI	Null	Total P (µg/l)	T	Air temperature (°C)	1075	No	GLMM (G)	0.379	**18.71**	**5.19**	1.75	S	S
5	Multiple stressors in lakes at European scale	N	S-T	L	Phytoplankton	Cyanobacteria biomass (mg/l)	10	Total P (µg/l)	T	Air temperature (°C)	1075	No	GLMM (G)	0.230	**10.29**	**6.91**	1.84	S	S
6	Nervion estuary	S	S-T	TW	Fish	AFI	Null	NH_3 at the bottom (µmol/l)	T	Water bottom temperature (°C)	106	No	GLMM (G)	0.163	**-3.94**	0.66	-0.86	O	S

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor	Stressor interaction	Interaction type	Interaction wrt nutrients
6	Nervion estuary	S	S-T	TW	Fish	Fish richness	Null	NH_3 at the bottom (µmol/l)	T	Water bottom temperature (°C)	106	No	GLMM (P)	0.034	-1.33	-0.69	0.02	A	A
6	Nervion estuary	S	S-T	TW	Fish	Fish abundance (ind/ha)	15	NH_3 at the bottom (µmol/l)	T	Water bottom temperature (°C)	105	No	GLMM (G)	0.021	-0.57	-0.87	0.88	A	A
7	Multiple stress acting on very large rivers in Europe	C	S-T	R	Invertebrates	Proportion of invertebrates individuals in Ephemeroptera, Plecoptera and Trichoptera	c.f. 12	NO_3-N in water column (mg/L)	M	Commercial navigation intensity index	275	No	GLMM (G)	0.266	**-4.26**	**-3.01**	**-2.35**	S	S
8	Odense	C	S	R	Invertebrates	BInd12	12	Mean total P (mg/l)	LF	Minimum flow (m^3/s)	100	No	GLM (G)	0.370	**4.29**	**-6.39**	-1.38	O	A
8	Odense	C	S	R	Invertebrates	BInd12	12	Mean total P (mg/l)	T	Maximum water temperature (°C)	100	No	GLM (G)	0.270	**-2.33**	**-3.83**	-1.76	S	S
8	Odense	C	S	R	Fish	BInd15	15	Mean total P (mg/l)	LF	Minimum flow (m^3/s)	80	No	GLM (G)	0.084	**-2.20**	-1.06	-0.89	S	S
8	Odense	C	S	R	Fish	BInd15	15	Mean total P (mg/l)	T	Maximum water temperature (°C)	80	No	GLM (G)	0.075	**-2.14**	-0.04	-0.42	S	S
9	Norway flume experiments	N	ME	FM	Benthic algae	log green algae	Null	Nutrient treatment	HF	Flood treatment	40	No	GLMM (G)	0.159	**3.93**	0.05	0.21	S	S

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor effect (z)	Stressor interaction (z)	Interaction type	Interaction wrt nutrients
9	Norway flume experiments	N	ME	FM	Benthic algae	log diatoms	Null	Nutrient treatment	HF	Flood treatment	40	No	GLMM (G)	0.134	**5.28**	2.14	**-2.93**	A	A
9	Norway flume experiments	N	ME	FM	Benthic algae	log ChlA	8	Nutrient treatment	HF	Flood treatment	40	No	GLMM (G)	0.108	**5.04**	1.67	-1.98	A	A
9	Norway flume experiments	N	ME	FM	Benthic algae	log cyanobacteria	10	Nutrient treatment	HF	Flood treatment	40	No	GLMM (G)	0.103	**5.00**	1.93	**-3.15**	A	A
10	Northern European lakes (eutrophic)	N	S-T	L	Phytoplankton	Mean cyanobacteria	10	Spring total P	T	Summer mean temperature (°C)	178	No	GLMM (G)	0.068	1.04	-0.85	**-2.52**	O	A
10	Northern European lakes (oligo-mesotrophic)	N	S-T	L	Phytoplankton	Mean cyanobacteria	10	Spring total P	T	Summer mean temperature (°C)	386	No	GLMM (G)	0.095	**4.35**	-0.22	-0.78	O	A
10	Northern European lakes (eutrophic)	N	S-T	L	Phytoplankton	Mean cyanobacteria	10	Spring total P	HF	Summer precipitation (mm)	179	No	GLMM (G)	0.050	0.87	-0.87	**2.59**	O	S
10	Northern European lakes (oligo-mesotrophic)	N	S-T	L	Phytoplankton	Mean cyanobacteria	10	Spring total P	HF	Summer precipitation(mm)	380	No	GLMM (G)	0.087	**4.07**	-0.14	1.34	O	S
11	Shallow Lake Mesocosm Experiment	N	ME	LM	Phytoplankton	Chlorophyll a	8	Nutrient treatment ($NaNO_3$ + $NaPO_4$)	T	Heating treatment (+4 °C)	832	No	GLMM (G)	0.276	**5.66**	1.68	-1.77	A	A

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor effect (z)	Stressor interaction (z)	Interaction type	Interaction wrt nutrients
11	Shallow Lake Mesocosm Experiment	N	ME	LM	Phytoplankton	Chlorophyll a	8	Nutrient treatment ($NaNO_3$ + $NaPO_4$)	HF	Flushing treatment (every 12 weeks)	832	No	GLMM (G)	0.278	**5.15**	1.91	-1.02	A	A
11	Shallow Lake Mesocosm Experiment	N	ME	LM	Phytoplankton	Cyanobacteria chlorophyll a	10	Nutrient treatment ($NaNO_3$ + $NaPO_4$)	T	Heating treatment (+4 °C)	288	No	GLMM (G)	0.145	**3.35**	**3.20**	**-3.17**	A	A
11	Shallow Lake Mesocosm Experiment	N	ME	LM	Phytoplankton	Cyanobacteria chlorophyll a	10	Nutrient treatment ($NaNO_3$ + $NaPO_4$)	HF	Flushing treatment (every 12 weeks)	288	No	GLMM (G)	0.026	1.13	0.33	-0.26	A	A
12	European lakes (shallow and high risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	567	Yes	GLMM (G)	0.140	**4.50**	0.78	0.98	S	S
12	European lakes (shallow and high risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	567	Yes	GLMM (G)	0.140	**4.52**	-0.69	-1.96	O	A
12	European lakes (shallow and medium risk group 1)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	158	Yes	GLMM (G)	0.110	**2.15**	0.33	-0.81	A	A
12	European lakes (shallow and medium risk group 1)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	158	Yes	GLMM (G)	0.120	**2.00**	-1.20	-0.01	O	A

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor effect (z)	Stressor interaction (z)	Interaction type	Interaction wrt nutrients
12	European lakes (shallow and medium risk group 2)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	132	Yes	GLMM (G)	0.110	**2.15**	0.33	-0.81	A	A
12	European lakes (shallow and medium risk group 2)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	132	Yes	GLMM (G)	0.120	**2.00**	-1.20	-0.01	O	A
12	European lakes (shallow and low risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	422	Yes	GLMM (G)	0.171	1.53	**3.17**	1.90	S	S
12	European lakes (shallow and low risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	422	Yes	GLMM (G)	0.166	1.72	-1.94	-0.63	O	A
12	European lakes (deep and high risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	459	Yes	GLMM (G)	0.079	**4.90**	**-2.95**	0.50	O	S
12	European lakes (deep and high risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	459	Yes	GLMM (G)	0.074	**5.00**	-0.31	-0.58	O	A

Study ID	Study name	Region	Study type	Site type	Response category	Response variable	BInd	Nutrient stress	Second stressor	Second stressor variable	Sample size	Additional covariates	Model	R^2	Nutrient effect (z)	Second stressor effect (z)	Stressor interaction (z)	Interaction type	Interaction wrt nutrients
12	European lakes (deep and medium risk group 1)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	36	Yes	GLMM (G)	0.081	1.57	0.38	-0.36	A	A
12	European lakes (deep and medium risk group 1)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	36	Yes	GLMM (G)	0.124	**2.07**	1.16	0.29	S	S
12	European lakes (deep and medium risk group 2)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	414	Yes	GLMM (G)	0.064	**2.31**	**2.92**	0.34	S	S
12	European lakes (deep and medium risk group 2)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	414	Yes	GLMM (G)	0.036	1.86	0.46	-0.68	A	A
12	European lakes (deep and low risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	T	Mean air temperature (°C)	422	Yes	GLMM (G)	0.145	**2.87**	**2.19**	1.58	S	S
12	European lakes (deep and low risk)	C+N	S	L	Phytoplankton	Cyanobacteria biomass (mm^3/l)	10	Total P (µg/l)	HF	Summer precipitation (mm)	422	Yes	GLMM (G)	0.143	**2.75**	0.39	-1.02	A	A

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 6.1 Synthesis of stressor interactions and indicators

D6.1-2 Manuscript on evaluation of methods for diagnosing cause of deteriorations in ecological status

Lead beneficiary: University of Duisburg-Essen (UDE)
Authors: Jan U. Lemm, Christian K. Feld, Sebastian Birk

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)	

Table of contents

Non-technical summary

1. Aim of this study was to analyse a large set of bioassessment metrics to identify and quantify stressor-specific metric responses reacting to one group of stressors but not to another.
2. We hypothesise that stressor-specific responses occur when the individual stressors show independent ‘modes of action’ (i.e. the specific stress-induced changes of environmental factors that modify the ecological niches of the species constituting the biological community).
3. The data used comprised three biological groups (macrophytes, benthic invertebrates, fish) covering three broad river types in Western and Central Germany. The stressor groups under investigation were physico-chemical, hydromorphological and hydrological stress.
4. We performed linear variation partitioning to reduce the large set of metrics to a set of candidates for further non-linear analyses using a combination of boosted regression tree modelling and variation partitioning.
5. The linear analyses revealed 16 candidate metrics that met our criteria, most of them for the medium to large lowland rivers. Macrophyte- and fish-based metrics were most relevant. In a geographically and methodologically more precise data subset, invertebrate metrics revealed more promising models than in the broader data set.
6. Subsequent non-linear modelling resulted in two truly stressor-specific metrics, both based on invertebrate data: The *Index of Biocoenotic Region* (specifically indicating hydromorphological stress) and the *Share of alien species* (specifically indicating physico-chemical stress).
7. We concluded that the biological community generally responds to stressors in rather an integrative than a specific way, but stressor-specific metrics can be identified. Future research on diagnostic metrics should focus on quantifying those stressor parameters that represent individual ‘modes of action’.

Introduction

Aquatic ecosystems are impacted by multiple human pressures (Hering et al. 2015), such as point source pollution from urban areas and diffuse pollution from agricultural land use. Together with severe hydrological and morphological modifications, these pressures are widespread in Europe and continue to impact aquatic biodiversity in lakes, rivers, estuaries and coastal waters (EEA 2012a). The anthropogenic environmental impact of the pressures is quantified by numerous stressors, i.e. measurable environmental factors that exceed the range of natural variation and thus cause biological deterioration (Odum 1985, Underwood 1989). Eutrophication and more specifically the load of rivers and lakes with nitrogen and phosphorus is quantified by the concentrations of respective nutrient compounds, for example, ammonia, nitrite, nitrate or soluble reactive phosphorus. This kind of stressor data is usually monitored in parallel with biological monitoring schemes.

At present, comprehensive monitoring data on biology and environmental stressors is available for about 120,000 water bodies in Europe (EEA 2012a). This data provides the basis for ecological status assessment and subsequently for the derivation of appropriate management and restoration options to improve ecological status according to the European Water Framework Directive (WFD). However, while biological assessment is relatively straightforward in Europe (Birk et al. 2012), the derivation of suitable management options is not. Often, ecological status assessment combines multiple stressors effects into one or several biological metrics that together form a multi-metric index (Karr and Chu 1999). In brief, current multi-metric bioassessment systems usually integrate stressors effects (e.g. pollution, hydrological and morphological degradation) across different spatial scales (e.g. catchment, stream segment, reach, site). The commonly used percentage of Ephemeroptera, Plecoptera and Trichoptera taxa (%EPT) is an example of such an integrative metric. The metric accounts for sensitive taxa of the three insect orders, however many taxa are not specifically sensitive to one stressor, which is why %EPT taxa can be found in numerous studies addressing evenly numerous combinations of stressors at different spatial scales (e.g. Böhmer et al. 2004, Hering et al. 2006a, Collier 2013).

From a purely ecological viewpoint, the holistic evaluation of ecosystem status neither expects nor desires stressor-specific biological response (Verdonschot 2000). On the other hand, multi-metric bioassessment refers to the concept of stressor-specific bioindication (Hering et al. 2006b); yet concrete empirical evidence is largely pending. Furthermore, using integrative assessment systems or integrative composite metrics, it is likely to be impossible to distinguish individual stressors effects (Gieswein et al., submitted). This bears a serious challenge for river basin management, because to improve ecological status water managers need to be able to distinguish the stressors importance (hierarchy) to be able to derive appropriate management options (and their hierarchy).

Our mechanistic understanding of stressor-specific response is based on general concepts of multi-stress effects in biological communities (Breitburg et al. 1998, Vinebrooke et al. 2004):

Stressor effects are determined by their 'modes of action' (Escher and Hermens 2002), i.e. the specific stress-induced changes of environmental factors that modify the ecological niches of the species constituting the biological community (Hutchinson 1957). These changes in niche factors affect the species adapted to these niches, i.e. these species are sensitive to the stressor and will ultimately disappear from the stressed biological community. We hypothesise that in a multi-stressed environment stressor-specific biological response can be observed if:

- The individual stressors show independent 'modes of action', and
- these 'modes of action' affect different features of the biological community (according to the concept of 'negatively correlated species co-tolerances' *sensu* Vinebrooke et al. 2004).

For instance, organic pollution (causing oxygen depletion stress) and pesticide contamination (causing toxic stress) feature independent 'modes of action' that affect particular species with specific life-history and physiological traits within the biological community.

Inspired by the few studies that demonstrated successful applications of the concept of stressor-specific biological response (e.g. Baattrup-Pedersen et al. 2016, Statzner and Bêche 2010), here we analyse an extensive monitoring dataset to systematically scrutinise the response patterns. The different community features mentioned above are represented by 782 bioassessment metrics computed from the sampling data of three biological organism groups (macrophytes, benthic invertebrates, fish). These metrics quantify the presence and abundance of taxa, and the proportion of individuals within the biological community that share the same ecological and biological traits (e.g. taxonomic affiliations; life-history, physiological or morphological traits; habitat preferences). Against this plethora of possible community response gradients, we investigated into single stressor effects across up to three co-acting stressor groups. These groups comprised physico-chemical, hydromorphological and hydrological stressors. Aim of our study was to identify stressor-specific bioassessment metrics that react to a single group of stressors while not responding to any other co-acting stressor group.

Material and Methods

Database

We obtained WFD monitoring data from three German federal states (North Rhine-Westphalia, Saxony-Anhalt, Hesse), covering 769 sites at three European broad river types (BT; ETC/ICM, 2015) (Figure 1): Medium to large lowland rivers of calcareous or mixed catchment geology (BT4, n = 92 sites; corresponding to German river types 15, 15_g, 17 according to Pottgiesser and Sommerhäuser 2008), small lowland rivers of calcareous or mixed catchment geology (BT5, n = 392 sites; corresponding to German river types 14, 16, 18), and small mid-altitude rivers of siliceous catchment geology (BT9, n = 285 sites; corresponding to German river types

5 and 5.1). Biological data covered three biological organism groups (macrophytes, benthic invertebrates and fish) and non-biological data on river physico-chemistry, hydromorphology, hydrology (only for BT4), catchment size and altitude at sampling sites.

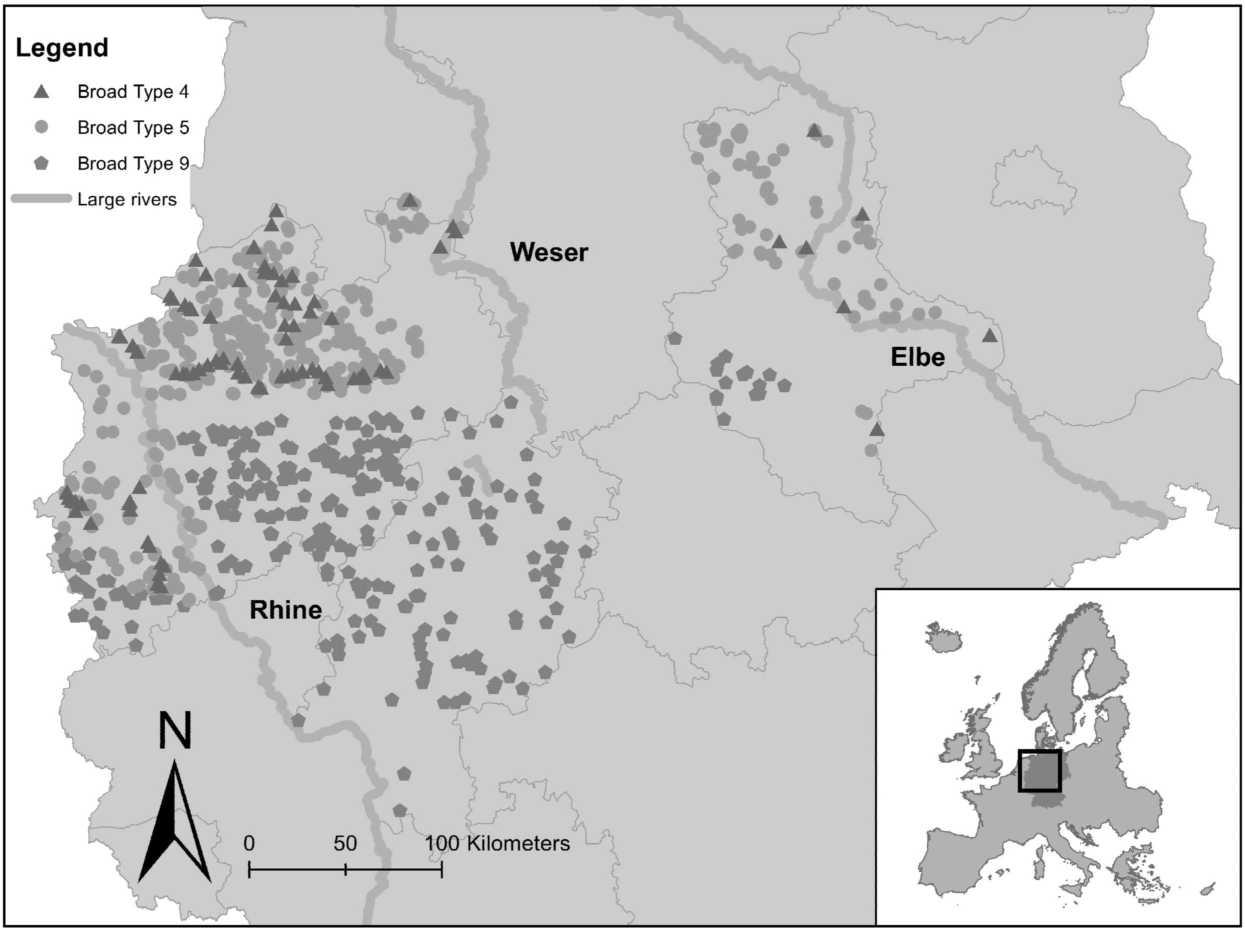


Figure 1: Location of the 769 sampling sites used in this study, covering macrophyte, benthic invertebrate and fish data at three broad river types in Western and Central Germany.

Biological data

Macrophyte data was available for 505 sites including 846 surveys conducted in the years 2005 to 2010. Species composition and abundance of aquatic plants (macroalgae, bryophytes and angiosperms) were recorded by visually inspecting representative river stretches of 100 m length during the growing season (May to September; Schaumburg et al. 2004).

Data on taxonomic composition and abundance of benthic invertebrates was available for 615 sites including 874 samples taken in the years 2004 to 2013. Benthic invertebrates were collected during spring and summer, respectively, following a multi-habitat sampling protocol (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 x 25 cm^2 frame and a mesh size of 500 μm. Benthic invertebrates were identified to species level where possible.

Data on species composition and abundance of fishes was available for 275 sites including 352 samples taken in the years 2004 to 2009. The sampling procedure followed CEN (2003) using electrofishing along river stretches of several 100 m in length. All sampled specimens were identified to species level, counted, measured in length and released after sampling (Dahm et al. 2013).

On the basis of the biological data, we calculated a total number of 782 bioassessment metrics for the three organism groups (see Annex 1). The metrics covered a wide range of categories like richness, abundance, diversity, sensitivity and functional traits (Karr 1981, Birk et al. 2012). Three hundred twenty-three macrophyte metrics were computed referring to ecological classification methods (Birk and Willby 2010), growth form types (Wiegleb 1991), ecological attribute groups (Willby et al. 2000) and further trait information (Baattrup-Pedersen et al. 2016). Three hundred ninety-two benthic invertebrate metrics we calculated with the Software ASTERICS Version 4.0.4 (Meier et al. 2006). Data on functional invertebrate traits were acquired from Schmidt-Kloiber and Hering (2015), including information of Chevene et al. (1994) and Tachet et al. (2010). We computed the relative abundance scores of invertebrate traits according to Dolédec et al. (2011). Two hundred eighty fish metrics were calculated using the autecological information of Holzer (2008) collated in the European research projects EFI+ (EFI+ Consortium 2009) and FAME (Kestemont and Goffaux 2002).

Non-biological data

Non-biological data was available for three stressor groups, i.e. physico-chemical, hydromorphological and hydrological parameters (Annex 2). Data on physico-chemical water parameters was acquired from routine quality monitoring programmes of the federal states (UBA 2014) and spatio-temporally matched to the biological samples. For the macrophyte samples, we selected annual average values of the parameters *Water temperature*, *Oxygen concentration*, *Chloride*, *Total nitrogen* and *Total phosphorus*. The same parameters were selected for the benthic invertebrate samples, except for *Total nitrogen* that was replaced by *Nitrate*. For the fish samples, we collated the average annual records of Water temperature, *Oxygen concentration*, *Conductivity*, *Chloride* and *Total phosphorus*.

Hydromorphological data on selected physical habitat quality features was available from standardised field surveys of 100 m river stretches (LAWA 2000, LUA 2001), aggregated to 500 m reaches upstream of the biological sampling site. Ten different instream, riparian and floodplain quality features were evaluated by scores ranging from 1 (= near-natural) to 7 (= totally impaired) (see Dahm et al. 2013, Annex 2). For broad type 4, data on hydrological alteration was acquired from the hydrological model PCR-GLOBWB (van Beek and Bierkens 2008). Score differences of 81 indicators of hydrological alteration (IHA; Richter et al. 1996) derived from two model scenarios (i. altered hydrology including water abstraction and ii. near-natural hydrology) were calculated on the level of the Functional Elementary Catchments (FEC) of the European catchments and rivers network system (EEA 2012b, Globevnik et al. 2017, Panagopoulos et al. 2017). We assigned the FEC-specific IHA score differences to each biological sampling site located in the respective FEC.

Further environmental data used in our study comprised altitude and catchment size at the site of biological sampling (i.e. 'natural variables').

Data preparation

Histograms of all non-biological variables were visually inspected for normal distribution and, where necessary, we transformed the variables using yj-power transformation (Yeo and Johnson 2000). Separately for each of the nine datasets collated (i.e. three biological organism groups X three broad river types), we minimised collinearity between the variables by a stepwise reduction of variables showing a variance inflation factor > 7. The remaining collinearity was checked by Spearman rank correlation. All non-biological variables were then centered and scaled.

We additionally compiled a subset of benthic invertebrate and respective non-biological data that covered 161 samples at 107 sites of the German river type 5 (small coarse substrate-dominated siliceous highland rivers) sampled during a single season. With this geographically and methodologically more homogeneous data subset within BT9 we intended to study the effects of data quality on the analytical outcomes.

Data analysis

We used variance partitioning analysis (*Varpart*) to investigate into stressor-specific biological response (Borcard et al. 1992, Peres-Neto et al. 2006). *Varpart* uses regression analysis of groups of predictor variables against a response variable to quantify the variance fractions (given as adjusted R^2 values) explained by each predictor group alone (individual fraction) or in combination (joint fraction) (Figure 2). The stressor groups (physico-chemical, hydromorphological, hydrological) and both environmental variables together (altitude, catchment size) were each treated as single groups of predictor variables. The bioassessment metrics were used as response variables.

To calculate all single fractions [a], [b], [c] and [d] as shown in Figure 2, single and combined regression models of the predictor groups against the biological response variables were calculated. We performed linear *Varpart* in R using the *vegan* package (Oksanen et al. 2008) for all 782 bioassessment metrics computed from the biological data. To compare linear and non-linear patterns of stressor-specific response, we ran *Varpart* based on non-linear modeling via Boosted Regression Tree analysis in R (BRT; Elith and Leathwick 2017) using the *dismo* (Hijmans et al. 2016) and *gbm* (Ridgeway 2015) packages. This was done only for a limited number of metrics due to the considerable time expenditure involved in this modeling. In both analyses, negative variance fractions were replaced by zero values according to Borcard et al. (2011).

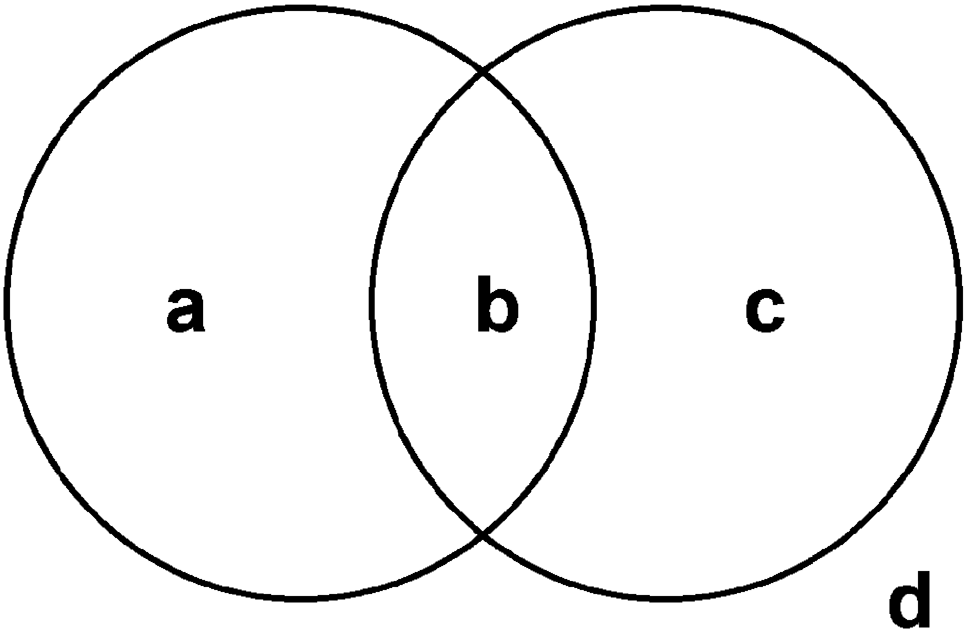


Figure 2: Schematic Venn Diagram resulting from variance partitioning of two different stressor groups. The explained variance of stressor 1 is represented by fraction [a+b]. The explained variance of stressor 2 is represented by fraction [b+c]. [a] and [c] represent the individual fractions of variance explained by stressor 1 and stressor 2, respectively. [b] is the joint fraction, and [d] is the residual variance.

Varpart using linear regression

We automated the *Varpart* using linear regression by applying a parameterised R markdown document (Allaire et al. 2016). The groups of physico-chemical and hydromorphological stressors, plus the group of environmental variables, were included in the analysis of all nine datasets. For the data of BT4, the hydrological stressors were additionally used as a separate predictor group. In order to balance the number of explanatory variables in the linear *Varpart*, the parameters within each stressor group were reduced to the first three components using Principal Components Analysis (Annex 3).

We used the following criteria to identify stressor-specific bioassessment metrics based on the linear *Varpart*: (i) The explained variance of the full model was at least 25 %, (ii) the highest individual fraction was more than twice the second highest individual fraction, (iii) the highest individual fraction was larger than the joint fraction, and (iv) the share of the highest individual fraction to the total explained variance was larger than one-third. For highly correlated metrics (Spearman's $R \geq 0.7$), we removed those with lower explained variance of the full model or the highest individual fraction.

Varpart using non-linear regression

BRT models were run for 100 metrics, including those meeting the above-mentioned criteria, and the explained variances of the individual models were fed into the *Varpart* scheme (see

Peres-Neto et al. 2006). To gain a stronger focus on the stressor effects, we excluded the group of environmental variables from this analysis. Stressor-specific biological response metrics were identified using the same criteria as above. For the biological metrics fulfilling these criteria, we checked for significant correlations with the single parameters included in the respective stressor group. Interaction effects between stressor groups were scrutinised using the *gbm.interaction* function in the R package *dismo* (Hijmans et al. 2016).

Results

Varpart using linear regression

Table 1 shows the stressor-specific metrics resulting from the linear *Varpart*. We identified a total number of 16 metrics meeting our criteria, from which twelve metrics yielded stressor-specific responses for the medium to large lowland rivers (BT4). Especially macrophyte- and fish-based metrics were relevant for this river type, with most of the metrics specifically responding to hydromorphological stress. We found only three invertebrate-based metrics, with many others failing to meet *criterion iii* (i.e. the highest individual fraction was larger than the joint fraction). This was different for the geographically and methodologically more homogeneous data subset that we analysed in addition: Eight metrics responding specifically to physico-chemical stressors were identified (Table 2).

On average, the stressor-specific response (i.e. the highest individual fraction) amounted to 18 % across all river types and organism groups, irrespective of the stressor group. Hydromorphological stressors yielded a specific response of 19 % of explained variance. The two metrics responding to hydrological stress showed an average of 14 %, while the physico-chemical stressors resulted in 13 %. The latter raised to an average of 22 % for the BT9 invertebrate data subset. Average stressor-specific explained variances were 16 % (macrophytes), 18 % (fish) and 20 % (benthic invertebrates, incl. data subset).

The explained variances of the full models increased from macrophytes (31 %) to fishes (34 %) to benthic invertebrates (37 %, incl. data subset). The joint fractions, indicating a non-stressor-specific response, were low for macrophytes and fishes (6 % and 5 %, respectively) and high for benthic invertebrates (incl. data subset, 14 %). The environmental variables (altitude, catchment size) explained an average variance of 4 % (macrophytes), 1 % (benthic invertebrates, incl. data subset) and 12 % (fishes).

The three stressor-specific metrics performing best in the linear *Varpart* were: *German Saprobic Index* for the BT9 data subset (physico-chemical stressors, individual fraction: R^2 = 27 %), *Relative abundance of eurytopic fish species* for BT4 (hydromorphological stressors, individual fraction: R^2 = 26 %) and *Relative abundance of species tolerant to water quality deterioration* (hydromorphological stressors, individual fraction: R^2 = 25 %).

Table 1: Overview of explained variances (given as adjusted R²) per broad type (BT) and biological organism group gained from linear Varpart. N = number of samples, 90th = 90th percentile

Biological group	BT	N	Statistical descriptors	Full model	Individual fractions				Joint fraction
					Physico-chemistry	Hydromorphology	Hydrology	Environm. variables	
Macrophytes	4	314	Median	0.103	0.010	0.010	0.013	0.008	0.024
			90th	0.266	0.067	0.095	0.055	0.043	0.116
			Max	0.396	0.175	0.199	0.086	0.099	0.272
Benthic invertebrates	4	391	Median	0.167	0.002	0.016	0.026	0.024	0.052
			90th	0.355	0.026	0.051	0.085	0.137	0.179
			Max	0.556	0.107	0.126	0.136	0.402	0.368
Fish	4	280	Median	0.017	0.036	0.033	0.062	-0.008	0.171
			90th	0.126	0.146	0.102	0.213	0.062	0.350
			Max	0.169	0.259	0.177	0.463	0.209	0.514
Macrophytes	5	322	Median	0.045	0.013	0.003	-	0.008	0.006
			90th	0.154	0.065	0.018	-	0.069	0.029
			Max	0.341	0.128	0.048	-	0.214	0.074
Benthic invertebrates	5	389	Median	0.098	0.015	0.023	-	0.027	0.015
			90th	0.264	0.061	0.075	-	0.108	0.064
			Max	0.356	0.124	0.135	-	0.194	0.124
Fish	5	280	Median	0.127	0.015	0.012	-	0.075	0.000
			90th	0.283	0.065	0.052	-	0.271	0.037
			Max	0.380	0.136	0.128	-	0.313	0.087

Table 1 (cont.): Overview of explained variances (given as adjusted R^2) per broad type (BT) and biological organism group gained from linear Varpart. N = number of samples, 90th = 90^{th} percentile

Biological group	BT	N	Statistical descriptors	Full model	Individual fractions				Joint fraction
					Physico-chemistry	Hydromorphology	Hydrology	Environm. variables	
Macrophytes	9	203	Median	0.041	0.004	0.002	-	0.010	0.004
			90th	0.130	0.033	0.002	-	0.109	0.029
			Max	0.182	0.109	0.057	-	0.141	0.047
Benthic Invertebrates	9	392	Median	0.155	0.048	0.010	-	0.012	0.069
			90th	0.380	0.125	0.036	-	0.043	0.210
			Max	0.544	0.187	0.083	-	0.124	0.323
Fish	9	187	Median	0.217	0.022	0.016	-	0.078	0.049
			90th	0.321	0.085	0.073	-	0.229	0.098
			Max	0.397	0.153	0.166	-	0.250	0.140

Table 2: Results of Varpart using linear regression. The table specifies the adjusted R^2 values of the different fractions and the full model resulting from Varpart.

Organism group	Broad type	Metric	Individual fractions				Joint fraction	Full Model
			Physico-chemistry	Hydro-morphology	Hydrology	Environm. variables		
Macrophytes	4	Total abundance of taxa with plant life-form: Hemicryptophytes	0.146	0.062	0.027	0.011	0.111	0.357
		Total abundance of taxa with plant life-form: Geophytes	0.032	0.160	0.070	0.047	0.038	0.347
		Total number of taxa with aerenchyma	0.030	0.155	0.042	0.050	0.050	0.327
		Total abundance of free-floating taxa	0.004	0.164	0.059	0.052	0.075	0.269
		Total number of taxa with growth form: Hydrocharids	0.025	0.157	0.023	0.046	0.003	0.254
Benthic invertebrates	4	German Fauna Index (D03)	0.008	0.036	0.119	0.032	0.071	0.266
	5	Croatian Saprobic Index	0.124	0.062	-	0.026	0.104	0.316
	9	Relative abundance of taxa with reproduction by cemented isolated eggs	0.165	0.000	-	0.020	0.139	0.324
	9 (subset)	German Saprobic Index	0.276	0.007	-	0.003	0.225	0.511
		Relative abundance of EPT taxa	0.243	0.018	-	0.002	0.206	0.469
		SPEAR index	0.235	0.000	-	0.014	0.168	0.417
		Average Score Per Taxon	0.216	0.006	-	0.007	0.177	0.406
		German Fauna Index (German type 5)	0.194	0.012	-	0.001	0.182	0.389
		Rheoindex	0.202	0.008	-	0.000	0.153	0.363
		Relative abundance of taxa with locomotion type swimming or diving	0.164	0.003	-	0.031	0.137	0.335
		Relative abundance of alien species	0.205	0.000	-	0.023	0.024	0.252

Table 2 (cont.): *Results of Varpart using linear regression. The table specifies the adjusted R^2 values of the different fractions and the full model resulting from Varpart.*

Organism group	Broad type	Metric	Individual fraction				Joint fraction	Full model
			Physico-chemistry	Hydro-morphology	Hydrology	Environm. variables		
Fish	4	Relative abundance of species with life span > 15 years	0.020	0.117	0.054	0.209	0.064	0.352
		Relative abundance of species intolerant to habitat degradation	0.000	0.202	0.025	0.119	0.087	0.335
		Relative abundance of species tolerant to water quality deterioration	0.029	0.249	0.030	0.011	0.008	0.327
		Relative abundance of species with females maturing before/at age of 4 or 5 years	0.017	0.019	0.154	0.138	0.030	0.358
		Relative abundance of species tolerant to habitat degradation	0.000	0.245	0.049	0.048	0.104	0.446
	5	Average relative abundance of species fecundity scores	0.136	0.051	-	0.157	0.004	0.279
	9	Relative abundance of species intolerant to habitat degradation	0.131	0.019	-	0.082	0.088	0.354
		Shannon Wiener Index	0.097	0.000	-	0.189	0.037	0.272

Varpart using nonlinear regression

The 100 nonlinear *Varparts* that we analysed covered 23 macrophyte-based metrics, 48 invertebrate-based metrics and 29 fish-based metrics, respectively (see Annex 4). The overall mean explained variance of the full models was 77 %. The highest individual fractions amounted to an average of 15 %, while the average joint fraction was 61 %. None of the metrics identified in the linear *Varpart* could be confirmed in the nonlinear *Varpart*, except for the *Croatian Saprobic Index*. This metric, however, differed in stressor-specificity (physico-chemical versus hydromorphological stressors). For all other metrics, the joint fraction notably exceeded all individual fractions, rendering these responses non-stressor-specific. Only three metrics (applied to four datasets) met our selection criteria established for the linear *Varpart*, featuring an average explained variance of remarkable 43 % for the highest individual fractions (Table 3).

The *Index of Biocoenotic Region* showed significant Spearman correlations with the hydromorphological parameters *Flow variation* (R = -0.20) and *Substrate diversity* (R = -0.23). The *Share of alien species* revealed significant correlations with the physico-chemical parameters *Oxygen concentration* (R = -0.26), *Chloride* (R = 0.36), *Water temperature* (R = 0.34) and *Total phosphorus* (R = 0.26). The hybrid-response of the *Croatian Saprobic Index* was characterised by significant correlations with physico-chemical and hydromorphological parameters: *Chloride* (R = 0.27), *Nitrate* (R = 0.21), *Oxygen concentration* (R = -0.31), *Total phosphorus* (R = 0.31), *Water temperature* (R = 0.31), *Width variance* (R = -0.33), *Riparian zone* (R = -0.34), *Longitudinal banks* (R = -0.28), *Curvature/Bends* (R = -0.28), *Transverse structures* (R = -0.15), *Flow variation* (R = -0.22), *Substrate diversity* (R = -0.19) and *Special bank features* (R = -0.41). We did not observe strong interaction effects between stressor groups in the full BRT models.

Table 3: Results of the non-linear Varpart. The table specifies the adjusted R² values of the different fractions and the full model resulting from Varpart.

Biological group	Broad type	Metric	Individual fractions			Joint fraction	Full model
			Physico-chemistry	Hydromorphology	Hydrology		
Benthic invertebrates	4	Index of Biocoenotic Region	0.026	0.594	0.000	0.225	0.845
	5	Croatian Saprobic Index	0.114	0.341	-	0.325	0.780
	9	Share of alien species	0.329	0.029	-	0.073	0.431
	9 (data subset)	Share of alien species	0.448	0.000	-	0.000	0.448

Discussion

Aim of our study was to identify bioassessment metrics that respond specifically to different groups of stressors. We used *Varpart* as the analytical tool to detect stressor-specific response. A stressor-specific response was defined as the fraction of metric variance that (i) is explained individually by a single stressor group, and (ii) exceeds all other (individual or joint) fractions in explanatory power. Based on our criteria for selecting stressor-specific bioassessment metrics, 25 and three metrics could be identified from the linear and nonlinear *Varparts*, respectively.

The results gained from both analyses were hardly comparable. Only one metric identified by the linear *Varpart* was retrieved in the nonlinear *Varpart*, but its stressor response was ambiguous. This points at fundamental differences between the two analytical techniques. The power of the nonlinear *Varpart* to explain the full model and joint fraction variances was up to ten times higher than for the linear *Varpart*. However, the mean explanatory power of the individual fractions was comparable among analytical techniques. This suggests that the highly efficient BRT algorithm to establish strong relationships between explanatory and response variables inflates the explanatory power of all stressor groups included, overruling the signal of individual stressor effects. We observed this pattern for most of the metrics except for those three presented above (see Table 3).

We consider two of the three metrics identified in the nonlinear *Varpart* truly stressor-specific:

1. The *Index of Biocoenotic Region*, which represents the invertebrate community related to the longitudinal river zonation (AQEM consortium 2002), responds almost exclusively to hydromorphological stressors in the medium to large lowland rivers (BT4). Its negative relationship with *Flow variation* and *Substrate diversity* demonstrates the community shift due to potamalisation effects (Jungwirth et al. 1995).
2. The *Share of alien species* seems an ideal stressor-specific metric for the small mid-altitude rivers (BT9): With a single stressor fraction being high and the other fractions being (almost) zero, this metric responds exclusively to physico-chemical impairment. This is in line with Früh et al. (2012a, 2016) who, similar to our findings, described the important role of water quality parameters (esp. chloride, oxygen concentration and water temperature) to explain the presence of alien species. For medium to large rivers, however, the occurrence of aliens also relates to hydromorphological stressors (Früh et al. 2012b). This highlights that the ability of bioassessment metrics to provide stressor-specific responses differs among river types. Moreover, stressor-specific bioassessment benefits from a stringent river type definition, as shown by the generally better performance for the BT9 data subset.

The stressor-specific response of the third metric resulting from the nonlinear *Varpart*, the Croatian Saprobic Index (Wegl 1983, Birk and Schmedtje 2005), is more unclear. Though meeting our selection criteria, the values for the different fractions demonstrate strong effects of all stressor groups on the metric variance. This is confirmed by the significant correlations with the single physico-chemical and hydromorphological parameters. Designed to indicate water quality deterioration, the Saprobic Index also responds to hydromorphological stressors because (i) these stressors often entail issues in water quality (e.g. Shields et al. 2010) and (ii) the Index refers to taxa sensitive to oxygen depletion (like Ephemeroptera, Plecoptera and Trichoptera) that are affected by other 'modes of action' as well.

The Croatian Saprobic Index exemplifies the challenge of unveiling stressor-specific bioassessment: We almost always found high joint fractions in our *Varpart* models, suggesting that the biological community responds to stressors in rather an integrative than a specific way. Two main reasons may provide explanations for our findings:

1. **Metric design:** All metrics are calculated on the basis of biological community data. The metrics either directly process the information on the taxonomical affiliation of the sampled organisms, or refer to autecological information assigned to each taxonomic unit identified from the sample. In both cases, species (or lower taxonomic ranks) form the basic 'entities' for bioassessment. The traits of these entities that imply specific stressor sensitivity, however, are not independent features, but co-occur and associate with other traits (Verberk et al. 2013). This suggests that species always react to different stressors even if the metric only accounts for single trait features (but see Mondy et al. 2016).
2. **Stressor data:** Empirically testing our concept of stressor-specific response related to independent 'modes of action' would require the modeling of single direct stressors (e.g. chloride concentration and water temperature). From this perspective, the stressor groups that we defined represent heterogeneous amalgamations of different environmental factors modified by anthropogenic activities. Especially the group of hydromorphological stressors comprise a multitude of factors, each with several 'modes of action' affecting different aspects of the ecological niches (e.g. decreasing flow variation entails changes in hydraulic stress and oxygen supply).

Conclusions

Stressor-specific bioassessment metrics hold a promise for diagnosing single stressor effects of multi-stressor conditions, but it is hard to deliver on. Our findings offer very few suitable metrics relevant only for specific river types. Future research on diagnostic metrics primarily needs to focus on quantifying those stressor parameters that represent individual 'modes of action'.

Acknowledgments

We greatly acknowledge Jürgen Böhmer for calculating the fish metrics and Daniel Gebler for computing the macrophyte metrics. We also thank Fabian Bolik for supporting us in multiple ways while preparing this deliverable. This work is part of the MARS project (Managing Aquatic ecosystems and water Resources under multiple Stress) funded under the EU Seventh Framework Programme, Theme 6 (Environment including Climate Change), Contract No.: 603378 (http://www.mars-project.eu).

References

Allaire, J.J., Cheng, J., Xie, Y.; McPherson, J., Chang, W., Allen, J., Wickham, H., Atkins, A. and Hyndman, R. 2016. rmarkdown: Dynamic documents for R. R package version 1.3. Available online at: https://CRAN.R-project.org/package=rmarkdown.

AQEM Consortium 2002. Manual for the Application of the AQEM System. A Comprehensive Method to Assess European Streams Using Benthic Macroinvertebrates. Developed for the Purpose of the Water Framework Directive. Version 1.0. 202 pp.
Available online at: www.aqem.de.

Baattrup-Pedersen, A., Göthe, E., Riis, T. and O'Hare, M. T. 2016. Functional trait composition of aquatic plants can serve to disentangle multiple interacting stressors in lowland streams. Science of the Total Environment 543: 230–238.

Birk, S., Bonne, W., Borja, A., Brucet, S., Courrat, A., Poikane, S., Solimini, A., van de Bund, W., Zampoukas, N. and Hering, D. 2012. Three hundred ways to assess Europe's surface waters: An almost complete overview of biological methods to implement the Water Framework Directive. Ecological Indicators 18: 31–41.

Birk, S. and Schmedtje, U. 2005. Towards harmonization of water quality classification in the Danube River Basin: Overview of biological assessment methods for running waters. Archiv für Hydrobiologie Suppl. Large Rivers 16: 171–196.

Birk, S. and Willby, N. 2010. Towards harmonization of ecological quality classification: establishing common grounds in European macrophyte assessment for rivers. Hydrobiologia 652: 149-163.

Böhmer, J., Rawer-Jost, C., Zenker, A., Meier, C., Feld, C. K., Biss, R., and Hering, D. 2004. Assessing streams in Germany with benthic invertebrates: Development of a multimetric invertebrate based assessment system. Limnologica 34: 416-432.

Borcard, D., Legendre, P. and Drapeau, P. 1992. Partialling out the spatial component of ecological variation. Ecology 73: 1045–1055.

Borcard, D., Gillet, F. and Legendre, P. 2011. Numerical Ecology with R. Springer, New York, 306 pp.

Breitburg, D. L., Baxter, J.W., Hatfield, C. A., Howarth, R. W., Jones, C. G., Lovett, G. M. and Wigand, C. 1998. Understanding effects of multiple stressors: Ideas and challenges. In: Pace, M. L. and Groffman, P. M. (eds): Successes, Limitations and Frontiers in Ecosystem Sciences. Springer, New York, pp. 416–431.

CEN 2003. Water quality-sampling of fish with electricity. European Standard-EN 14011:2003. European Commitee for Standardization. Brussels, 18 pp.

Chevene, F., Doleadec, S. and Chessel, D. 1994. A fuzzy coding approach for the analysis of long-term ecological data. Freshwater Biology 31: 295–309.

Collier, K. J. 2014. Wood decay rates and macroinvertebrate community structure along contrasting human pressure gradients (Waikato, New Zealand). New Zealand Journal of Marine and Freshwater Research 48: 97–111.

Dahm, V., Hering, D., Nemitz, D., Graf, W., Schmidt-Kloiber, A., Leitner, P., Melcher, A. and Feld, C. K. 2013. Effects of physico-chemistry, land use and hydromorphology on three riverine organism groups: A comparative analysis with monitoring data from Germany and Austria. Hydrobiologia 704: 389–415.

Dolédec, S., Phillips, N. and Townsend, C. 2011. Invertebrate community responses to land use at a broad spatial scale: Trait and taxonomic measures compared in New Zealand rivers. Freshwater Biology 56: 1670–1688.

EFI+ Consortium 2009. Manual for the application of the new European Fish Index-EFI+. A Fish-based method to assess the ecological status of European running waters in support of the Water Framework Directive. Vienna, 45 pp.

Elith, J. and Leathwick, J. 2017. Boosted Regression Trees for ecological modeling. Online Tutorial. 22 pp.
Available online at: https://cran.r-project.org/web/packages/dismo/vignettes/brt.pdf.

Escher, B. I. and Hermens, J. L. M. 2002. Modes of action in ecotoxicology: Their role in body burdens, species sensitivity, QSARs, and mixture effects. Environmental Science & Technology 36: 4201–4217.

European Environment Agency 2012a. European Waters — Assessment of status and pressures. European Environment Agency. EEA Report No 8. Copenhagen, 96 pp.

European Environment Agency 2012b. European catchments and Rivers network system (Ecrins). Version 1.
Available online at: http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network.

ETC/ICM 2015. European freshwater ecosystem assessment: Cross-walk between the Water Framework Directive and habitats directive types, status and pressures. ETC/ICM Technical Report 2/2015, European Topic Centre on inland, coastal and marine waters. Magdeburg, 95 pp.

Früh, D., Haase, P. and Stoll, S. 2016. Temperature drives asymmetric competition between alien and indigenous freshwater snail species, *Physa acuta* and *Physa fontinalis*. Aquatic Sciences 79: 187–195.

Früh, D., Stoll, S. and Haase, P. 2012a. Physico-chemical variables determining the invasion risk of freshwater habitats by alien mollusks and crustaceans. Ecology and Evolution 2: 2843–2853.

Früh, D., Stoll, S. and Haase, P. 2012b. Physicochemical and morphological degradation of stream and river habitats increases invasion risk. Biological Invasions 14: 2243:2253.

Gieswein, A., Hering, D. and Feld, C. K. 2017. Additive effects prevail: The response of biota to multiple stressors in an intensively monitored watershed. Science of the Total Environment. submitted.

Globevnik, J., Birk, S., Koprivsek, M., Mahnkopf, J., Panagopoulos, Y., Pucher, M., Schinegger, R., Sanchez, M. F., Snoj, L., Stefanidis, K., Venohr, M. 2017. Analysis of pressure – response relationships: classification of multiple pressures on broad river types. In: MARS project EU Deliverable 5.1. Five Reports on stressor classification and effects at the European scale. pp. 38-110.

Hering, D., Johnson, R. K., Kramm, S., Schmutz, S., Szoszkiewicz, K. and Verdonschot, P. F. M. 2006a. Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: A comparative metric-based analysis of organism response to stress. Freshwater Biology 51: 1757–1785.

Hering, D., Moog, O., Ofenböck, T. and Feld, C. K. 2006b. Cook book for the development of a Multimetric Index for biological condition of aquatic ecosystems: experiences from the European AQEM and STAR projects and related initiatives. Hydrobiologia 566: 311-324.

Hering, D., Carvalho, L., Argillier, C., Beklioglu, M., Borja, A., Cardoso, A. C., Duel, H., Ferreira, T., Globevnik, L., Hanganu, J., Hellsten, S., Jeppesen, E., Kodes, V., Solheim, A. L., Nõges, T., Ormerod, S., Panagopoulos, Y., Schmutz, S., Venohr, M. and Birk, S. 2015. Managing aquatic ecosystems and water resources under multiple stress - An introduction to the MARS project. Science of the Total Environment 503–504: 10–21.

Hering, D., Moog, O., Sandin, L. and Verdonschot, P. F. M. 2004. Overview and application of the AQEM assessment system. Hydrobiologia 516: 1–20.

Holzer, S. 2008. European Fish Species : Taxa and guilds classification regarding fish-based assessment methods. Thesis. Vienna, 195 pp.

Hijmans, R. J., Phillips, S., Leathwick, J. and Elith, J. 2016. dismo: Species distribution modeling. R package version 1.1-1.
Available online at: https://CRAN.R-project.org/package=dismo.

Hutchinson, G. E. 1957. Concluding remarks. Cold Spring Harbor Symposia on Quantitative Biology 22: 415–427.

Jungwirth, M., Muhar, S. and S. Schmutz, S. 1995. The effects of re- created instream and ecotone structures on the fish fauna of an epipotamal river. Hydrobiologia 303: 195–206.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6: 21–27.

Karr, J. R. and Chu., E. W. 1997. Biological monitoring and assessment: using multimetric indexes effectively. EPA 235-R97-001. University of Washington. Seattle, 149 pp.

Kestemont, P. and Goffaux, D. 2002. Metric selection and sampling procedures for FAME (D 4 - 6). Final report: development, evaluation & implementation of a standardised fish-based assessment method for the ecological status of European rivers - A Contribution to the Water Framework Directive (FAME): 90 pp.

Länderarbeitsgemeinschaft Wasser (LAWA) 2000. Gewässerstrukturgütekartierung in der Bundesrepublik Deutschland. Verfahren für kleine und mittelgroße Fliessgewässer. Schwerin, 190 pp.

LUA 2001. Merkblatt 26 Gewässerstrukturgüte in Nordrhein-Westfalen. Anleitung für die Kartierung mittelgroßer bis großer Fließgewässer. Landesumweltamt Nordtrhein-Westfalen. Essen, 153 pp.

Meier, C., Haase, P., Rolauffs, P., Schindehuette, K., Schoell, F., Sundermann, A. and Hering, D. 2006. Methodisches Handbuch Fließgewaesserbewertung: Handbuch zur Untersuchung und Bewertung von Fließgewaessern auf der Basis des Makrozoobenthos vor dem Hintergrund der EG-Wasserrahmenrichtlinie, 110 pp.
Available online at: http://www.fliessgewaesserbewertung.de.

Mondy, C. P., Muñoz, I. and Dolédec, S. 2016. Life-history strategies constrain invertebrate community tolerance to multiple stressors: A case study in the Ebro basin. Science of The Total Environment 572: 196–206.

Odum, E. P. 1985. Trends expected in stressed ecosystems. BioScience 35: 419-422.

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P. R., O'Hara, R. B., Simpson, G. L., Solymos, P., Stevens, M. H. H., Szoecs, E. and Wagner, H. 2016. vegan: Community ecology package. R package version 2.4-1.
Available online at: https://CRAN.R-project.org/package=vegan.

Panagopoulos, Y., Stefanidis, K., Birk., S., Globevnik, L., Zachos, A., Lemm, J., Sanchez, M. F., Snoj, L., Koprivsek, M., Mimikou, M. 2017. Relation of low flows, E-flows, and Ecological Status. In: MARS project EU Deliverable 5.1. Five Reports on stressor classification and effects at the European scale. pp. 177-254.

Peres-Neto, P. R., Legendre, P., Dray, S. and Borcard, D. 2006. Variation partitioning of species data matrices: Estimation and comparison of fractions. Ecology 87: 2614–2625.

Pottgiesser, T. and Sommerhäuser, M. 2008. Beschreibung und Bewertung der deutschen Fließgewässertypen - Steckbriefe und Anhang. Essen, 29 pp.

Richter, B. D., Baumgartner, J. V., Powell, J. and Braun, D. P. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10: 1163–1174.

Ridgeway, G. 2013. gbm: generalized boosted regression models. R package version 2.1. R Project for Statistical Computing, Vienna, Austria.
Available online at: http://cran.r-project.org/web /packages/gbm/.

Schaumburg, J., Schranz, C., Foerster, J., Gutowski, A., Hofmann, G., Meilinger, P., Schneider, S. and Schmedtje, U. 2004. Ecological classification of macrophytes and phytobenthos for rivers in germany according to the water framework directive. Limnologica 34: 283–301.

Shields, F. D., Lizotte, R. E., Knight, S. S., Cooper, C. M. and Wilcox, D. 2010. The stream channel incision syndrome and water quality. Ecological Engineering 36: 78–90.

Schmidt-Kloiber, A. and Hering, D. 2015. www.freshwaterecology.info - An online tool that unifies, standardises and codifies more than 20,000 European freshwater organisms and their ecological preferences. Ecological Indicators 53: 271–282.

Statzner, B. and Bêche, L. A. 2010. Can biological invertebrate traits resolve effects of multiple stressors on running water ecosystems? Freshwater Biology 55: 80–119.

Tachet, H., Richoux, P., Bournaud, M. and Usseglio-Polatera, P. 2010. Invertébrés d'eau douce: systématique, biologie et écologie. CNRS Editions, Paris, 600 pp.

UBA 2014. Strategien zur Optimierung von Fließgewässerrenaturierungsmaßnahmen und ihrer Erfolgskontrolle. Umweltbundesamt. Dessau, 178 pp.

Underwood, A. J. 1989. The analysis of stress in natural populations. Biological Journal of the Linnean Society 37: 51-78.

Van Beek, L. P. H. and Bierkens, M. F. P. 2008. The Global Hydrological Model PCR-GLOBWB: Conceptualization, Parameterization and Verification. Report. Department of Physical Geography. Utrecht University. Utrecht, 53 pp.
Available online at: http://vanbeek.geo.uu.nl/suppinfo/vanbeekbierkens2009.pdf.

Verberk, W. C. E. P., van Noordwijk, C. G. E. and Hildrew, A. G. 2013. Delivering on a promise: integrating species traits to transform descriptive community ecology into a predictive science. Freshwater Science 32: 531–547.

Verdonschot, P.F.M. 2000. Integrated ecological assessment methods as a basis for sustainable catchment management. Hydrobiologia 422-423: 389–412.

Vinebrooke, RD, and Cottingham, K. L. 2004. Impacts of multiple stressors on biodiversity and ecosystem functioning: The role of species co-tolerance. Oikos 104: 451–457.

Wegl, R. 1983. Index für die Limnosaprobität. - In: Bundesanstalt für Wassergüte, Bundesministerium für Land- und Forstwirtschaft, Wasser und Abwasser (ed.): Beiträge zur Gewässerforschung. Bundesanst. f. Wassergüte d. Bundesmin. f. Land- u. Forstwirtschaft, Wien.

Wiegleb, G. 1991. Lebens-und Wuchsformen der makrophytischen Wasserpflanzen und deren Beziehungen zur Ökologie, Verbreitung und Vergesellschaftung der Arten. Tuexenia 11: 135–147.

Willby, N. J., Abernethy, V. J. and Demars, B. O. L. 2000. Attribute-based classification of European hydrophytes and its relationship to habitat utilization. Freshwater Biology 43: 43-74.

Yeo, I. and Johnson, R. A. 2000. A New family of power transformations to improve normality or symmetry. Biometrika 87: 954–59.

Annex 1

Table A1: The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Macrophytes	ALG	Algae
Macrophytes	BRH	Bryophytes - liverworts
Macrophytes	BRM	Bryophytes - moss
Macrophytes	HET	Heterotrophic
Macrophytes	LIC	Lichens
Macrophytes	LIG	Trees/shrubs
Macrophytes	PHE	Helophytes
Macrophytes	PHG	Higrophytes
Macrophytes	PHX	Other forms (wetland taxa)
Macrophytes	PHY	Hydrophytes
Macrophytes	PTE	Tracheophytes
Macrophytes	AL	Alien species
Macrophytes	Bry	Aquatic mosses
Macrophytes	B	Batrachid
Macrophytes	Cer	Ceratophyllid
Macrophytes	Cha	Charid
Macrophytes	Falg	Filamentous algae
Macrophytes	I	Isoetid
Macrophytes	Lmon	Large monocot
Macrophytes	Lple	Large pleustophyte
Macrophytes	Le	Lemnid
Macrophytes	Mag	Magnopotamid
Macrophytes	N	Nymphaeid
Macrophytes	Pep	Peplid
Macrophytes	R	Riccielid
Macrophytes	Rot	Rooting caulescent hydrophyte
Macrophytes	Hel	Small to medium-sized helophyte
Macrophytes	V	Vallisnerid
Macrophytes	E	Elodeide
Macrophytes	Eq	Equisetide
Macrophytes	G	Graminoide
Macrophytes	Herb	Herbide
Macrophytes	Hych	Hydrocharide
Macrophytes	Ju	Juncide
Macrophytes	M	Myriophyllide
Macrophytes	Ppot	Parvopotamide
Macrophytes	S	Stratiotide
Macrophytes	RoteList	Red list Germany
Macrophytes	pCCA_sil-Index	pCCA_sil-Index
Macrophytes	pCCA_cal-Index	pCCA_cal-Index
Macrophytes	pCCA_Lall-Index	pCCA_Lall-Index

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Macrophytes	pCCA_Lrhi-Index	pCCA_Lrhi-Index
Macrophytes	pCCA_Lpot-Index	pCCA_Lpot-Index
Macrophytes	NE	Ellenberg Nitrogen
Macrophytes	Meris-ma	Meristem single apical growth point
Macrophytes	Meris-sb	Meristem single basal growth point
Macrophytes	Meris-sa	Meristem multiple apical growth point
Macrophytes	Meris-sa-ma	Meristem single-multiple apical growth point
Macrophytes	Morph-ind	Morphology index
Macrophytes	Leaf-area	Leaf area
Macrophytes	Seeds	Reproduction by seeds
Macrophytes	Rhizome	Reproduction by rhizome
Macrophytes	Frag	Reproduction by fragmentation
Macrophytes	Rep-org	Number of reproductive organs per year and individual
Macrophytes	Overw-org	Overwintering organs
Macrophytes	ConUK	Conservation in UK & IRL
Macrophytes	RS-n	Rarity status in Britain RS - Present, not rare or scarce
Macrophytes	RS-r	Rarity status in Britain RS - Rare (1-15 10-km squares in Britain, 1987-1999)
Macrophytes	RS-s	Rarity status in Britain RS - Scarce (16-100 10-km squares in Britain, 1987-1999)
Macrophytes	Change	Change between 1930-1960 and 1987-1999 Chg
Macrophytes	Hg	Height
Macrophytes	Lg	Length
Macrophytes	Pa	Perennation - Annual
Macrophytes	Pb	Perennation - Biennial, including monocarpic perennials
Macrophytes	Pp	Perennation - Perennial
Macrophytes	Ch	Life form - Chamaephyte
Macrophytes	Gb	Life form - Bulbous geophyte
Macrophytes	Gn	Life form - Non-bulbous geophyte (rhizome, corm or tuber)
Macrophytes	Hc	Life form - Hemicryptophyte
Macrophytes	PHy1	Life form - Perennial hydrophyte (perennial water plant)
Macrophytes	Hz	Life form - Annual hydrophyte (aquatic therophyte)
Macrophytes	Ph	Life form - Mega-, meso- and microphanerophyte
Macrophytes	Pn	Life form - Nanophanerophyte
Macrophytes	Th	Life form - Therophyte (annual land plant)
Macrophytes	H	Herbaceous plant
Macrophytes	W	Woody plant
Macrophytes	E1	Biogeographic element, major biome E1
Macrophytes	E2	Biogeographic element, eastern limit categoryE2
Macrophytes	C	Continentality

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Macrophytes	TJan	UK & IRL January mean temperature (°C)
Macrophytes	TJul	UK & IRL July mean temperature (°C)
Macrophytes	Pre	UK & IRL Annual precipitation (mm)
Macrophytes	UK-Hab-1	UK Broad habitat preferences - Broadleaved, mixed and yew woodland
Macrophytes	UK-Hab-3	UK Broad habitat preferences - Boundary and linear features (eg hedges, roadsides, walls)
Macrophytes	UK-Hab-4	UK Broad habitat preferences - Arable and horticultural (includes orchards, excludes domestic gardens)
Macrophytes	UK-Hab-5	UK Broad habitat preferences - Improved grassland
Macrophytes	UK-Hab-6	UK Broad habitat preferences - Neutral grassland (includes coarse Arrhenatherum grassland)
Macrophytes	UK-Hab-8	UK Broad habitat preferences - Acid grassland (includes non-calcareous sandy grassland)
Macrophytes	UK-Hab-11	UK Broad habitat preferences - Fen, marsh and swamp (not wooded; includes flushes, rush-pastures, springs and mud communities)
Macrophytes	UK-Hab-12	UK Broad habitat preferences - Bog (on deep peat; includes bog pools as well as acid lowland valley mires on slightly shallower peat)
Macrophytes	UK-Hab-13	UK Broad habitat preferences - Standing water and canals
Macrophytes	UK-Hab-14	UK Broad habitat preferences - Rivers and streams
Macrophytes	UK-Hab-15	UK Broad habitat preferences - Montane habitats (acid grassland and heath with montane species)
Macrophytes	UK-Hab-16	UK Broad habitat preferences - Inland rock (heterogeneous - includes quarries, limestone pavement, cliffs, screes and skeletal soils over rock)
Macrophytes	UK-Hab-17	UK Broad habitat preferences - Built-up areas and gardens
Macrophytes	UK-Hab-19	UK Broad habitat preferences - Supralittoral sediment (strandlines, shingle, coastal dunes)
Macrophytes	UK-Hab-21	UK Broad habitat preferences - Littoral sediment (includes saltmarsh and saltmarsh pools)
Macrophytes	Ellenberg-L	Ellenberg light number (present in PLANATT)
Macrophytes	Ellenberg-F	Ellenberg moisture number (present in PLANATT)
Macrophytes	Ellenberg-R	Ellenberg reaction number (present in PLANATT)
Macrophytes	Ellenberg-N	Ellenberg nitrogen number (present in PLANATT)
Macrophytes	Ellenberg-S	Ellenberg salinity number (present in PLANATT)
Macrophytes	Ellenberg-Veg	Ellenberg vegetation type (present in PLANATT)
Macrophytes	Ell-L	Ellenberg light (from European databases - original)
Macrophytes	Ell-N	Ellenberg nitrogen number (from European databases - original)
Macrophytes	Ell-T	Ellenberg temperature number (from European databases - original)
Macrophytes	Ell-K	Ell continentality number (from European databases - original)
Macrophytes	Ell-F	Ellenberg moisture number (from European databases - original)

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Macrophytes	Ell-R	Ellenberg reaction number (from European databases - original)
Macrophytes	Ell-S	Ellenberg salinity number (from European databases - original)
Macrophytes	EllVeg-Ae	Ellenberg vegetation types - Aerophytes
Macrophytes	EllVeg-Ch	Ellenberg vegetation types - Chamaephytes
Macrophytes	EllVeg-Ge	Ellenberg vegetation types - Geophytes
Macrophytes	EllVeg-H	Ellenberg vegetation types - H
Macrophytes	EllVeg-Nx	Ellenberg vegetation types - N
Macrophytes	EllVeg-Ph	Ellenberg vegetation types - Phanerophytes
Macrophytes	EllVeg-Th	Ellenberg vegetation types - Therophytes
Macrophytes	RipVeg	Riparian vegetation
Macrophytes	MarVeg	Marginal (hygrophytes species)
Macrophytes	Bry	Bryophytes
Macrophytes	Emerg-Broad-leaved	Emergent broad-leaved herbs
Macrophytes	Emerg-Reed-sedge	Emergent reeds/sedges/rushes/analogous
Macrophytes	FloatLeaved-rooted	Floating leaved (rooted)
Macrophytes	FreeFloat	Free-floating
Macrophytes	Subm-Rosette	Submerged rosette
Macrophytes	Subm-Broad-leaved	Submerged broad-leaved
Macrophytes	Subm-Linear-leaved	Submerged linear leaved
Macrophytes	Subm-Fine-leaved	Submerged fine leaved
Macrophytes	Subm-Ran	Submerged ranunculus
Macrophytes	Subm-Myr	Submerged Myriophlloid
Macrophytes	Subm-Elod	Submerged Elodeoid
Macrophytes	Fil-Algae	Filamentous algae
Macrophytes	TerestLeaves	Submerged plants which produce terrestrial leaves
Macrophytes	Clone1	Type of reproduction 1
Macrophytes	Clone2	Type of reproduction 2
Macrophytes	Frflsr	Free floating surface
Macrophytes	Frflsb	Free floating submerged
Macrophytes	Anflle	Anchored, floating leaves
Macrophytes	Ansule	Anchored, submerged leaves
Macrophytes	Anemle	Anchored, emergent leaves
Macrophytes	Anhete	Anchored, heterophylly
Macrophytes	Emerg	Emergents
Macrophytes	Lemrg	Linear emergents
Macrophytes	BRemerg	Branched emergents
Macrophytes	Subm	Submerged
Macrophytes	Psubm	Patch-submerged
Macrophytes	Lsubm	Linear-submerged
Macrophytes	Moss	Mosses
Macrophytes	STOR	Störzeiger (5-19)
Macrophytes	GUTE	Gütezeiger (9-19)

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Macrophytes	RI-EQR	Reference Index
Macrophytes	ITEM	ITEM
Macrophytes	NITEM	Number of species used to calulated ITEM
Macrophytes	IBMR	Macrophytical Biological Index for Rivers (French index)
Macrophytes	NIBMR	Number of species used to calulated IBMR
Macrophytes	MIR	Macrophyte Index for Rivers (Polish method)
Macrophytes	mICM_low1	Intercalibration Common Metric lowland1
Macrophytes	NmICM_low1	Number of species used to calulated mICMlow1
Macrophytes	mICM_low2	Intercalibration Common Metric lowland2
Macrophytes	NmICM_low2	Number of species used to calulated mICMlow2
Macrophytes	mICM_mount	Intercalibration Common Metric mountain
Macrophytes	NmICM_mount	Number of species used to calulated mICMmount
Macrophytes	RMHI	River Macrophyte Hydraulic Index (British method)
Macrophytes	NRMHI	Number of species used to calulated RMHI
Macrophytes	RMNI	River Macrophyte Nutrient Index (British method)
Macrophytes	NRMNI	Number of species used to calulated RMNI
Benthic invertebrates	sin	German Saprobic Index (new version) saprobic score
Benthic invertebrates	sgn	German Saprobic Index (new version) weighting factor
Benthic invertebrates	sio	German Saprobic Index (old version) saprobic score
Benthic invertebrates	sgo	German Saprobic Index (old version) weighting factor
Benthic invertebrates	szx	Zelinka & Marvan: saprobic valence (xenosaprob)
Benthic invertebrates	szo	Zelinka & Marvan: saprobic valence (oligosaprob)
Benthic invertebrates	szb	Zelinka & Marvan: saprobic valence (beta-metasaprob)
Benthic invertebrates	sza	Zelinka & Marvan: saprobic valence (alpha-metasaprob)
Benthic invertebrates	szp	Zelinka & Marvan: saprobic valence (polysaprob)
Benthic invertebrates	szs	Zelinka & Marvan: saprobic value
Benthic invertebrates	szg	Zelinka & Marvan: weighting factor
Benthic invertebrates	czx	Czech Saprobic Index: saprobic valence (xenosaprob)
Benthic invertebrates	czo	Czech Saprobic Index: saprobic valence (oligosaprob)
Benthic invertebrates	czb	Czech Saprobic Index: saprobic valence (beta-metasaprob)
Benthic invertebrates	cza	Czech Saprobic Index: saprobic valence (alpha-metasaprob)
Benthic invertebrates	czp	Czech Saprobic Index: saprobic valence (polysaprob)
Benthic invertebrates	czsi	Czech Saprobic Index: saprobic value
Benthic invertebrates	czv	Czech Saprobic Index: weighting factor
Benthic invertebrates	slszx	Slowakian Saprobic Index: saprobic valence (xenosaprob)
Benthic invertebrates	slszo	Slowakian Saprobic Index: saprobic valence (oligosaprob)
Benthic invertebrates	slszb	Slowakian Saprobic Index: saprobic valence (beta-mesosaprob)
Benthic invertebrates	slsza	Slowakian Saprobic Index: saprobic valence (alpha-mesosaprob)

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Benthic invertebrates	slszp	Slowakian Saprobic Index: saprobic valence (polysaprob)
Benthic invertebrates	slszs	Slowakian Saprobic Index: saprobic value
Benthic invertebrates	slszg	Slowakian Saprobic Index: weighting factor
Benthic invertebrates	sihrHRIS	Croatian Saprobic Index (new)
Benthic invertebrates	sihrWEGL	Croatian Saprobic Index (old)
Benthic invertebrates	siRo	Romanian Saprobic Index
Benthic invertebrates	NSX	Netherland Saprobic valence xenosaprob
Benthic invertebrates	NSO	Netherland Saprobic valence oligosaprob
Benthic invertebrates	NSB	Netherland Saprobic valence beta-mesosaprob
Benthic invertebrates	NSA	Netherland Saprobic valence alpha-mesosaprob
Benthic invertebrates	NSP	Netherland Saprobic valence polysaprob
Benthic invertebrates	IVD01	Fauna index (AQEM): stream type 14
Benthic invertebrates	IVD02	Fauna index (AQEM): stream type 11
Benthic invertebrates	IVD03	Fauna index (AQEM): stream type 15
Benthic invertebrates	IVD04	Fauna index (AQEM): stream type 5
Benthic invertebrates	IVD05	Fauna index (AQEM): stream type 9
Benthic invertebrates	FI011	Fauna index: stream type 1.1
Benthic invertebrates	FI012	Fauna index: stream type 1.2
Benthic invertebrates	FI021	Fauna index: stream type 2.1
Benthic invertebrates	FI022	Fauna index: stream type 2.2
Benthic invertebrates	FI031	Fauna index: stream type 3.1
Benthic invertebrates	FI032	Fauna index: stream type 3.2
Benthic invertebrates	FI04	Fauna index: stream type 4
Benthic invertebrates	FI05	Fauna index: stream types 5+5.1 + 6+6K + 7
Benthic invertebrates	FI09	Fauna index: stream type 9
Benthic invertebrates	FI091	Fauna index: stream type 9.1
Benthic invertebrates	FI091_K	Fauna index: stream type 9.1K
Benthic invertebrates	FI092	Fauna index: stream type 9.2
Benthic invertebrates	FI11_12	Fauna index: stream types 11+12
Benthic invertebrates	FI14_16	Fauna index: stream types 14+16+18
Benthic invertebrates	FI15_17	Fauna index: stream types 15+17
Benthic invertebrates	FI152	Fauna index: stream type 15.2
Benthic invertebrates	FI19	Fauna index: stream type 19
Benthic invertebrates	PTI	Potamon Typie Index (Schöll & Haybach, 2003)
Benthic invertebrates	rst	Reproduction strategy (1=r strategy or 2= k strategy) (Schoell & Haybach)
Benthic invertebrates	Neozoa_D	Neozoa
Benthic invertebrates	wNeozoa	Thermophilic neozoa
Benthic invertebrates	zeu	Preference for crenal (spring)
Benthic invertebrates	zhy	Preference for hypocrenal (spring-brook)
Benthic invertebrates	zer	Preference for epirithral (upper-trout region)
Benthic invertebrates	zmr	Preference for metarhithral (lower-trout region)
Benthic invertebrates	zhr	Preference for hyporhithral (greyling region)
Benthic invertebrates	zep	Preference for epipotamal (barbel region)
Benthic invertebrates	zmp	Preference for metapotamal (brass region)
Benthic invertebrates	zhp	Preference for hypopotamal (brackish water)
Benthic invertebrates	zli	Preference for Litoral

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Benthic invertebrates	zpr	Preference for Profundal
Benthic invertebrates	RTI	Rhithron-Typie-Index
Benthic invertebrates	hpe	Preference for microhabitat Pelal
Benthic invertebrates	har	Preference for microhabitat Argyllal
Benthic invertebrates	hps	Preference for microhabitat Psammal
Benthic invertebrates	hak	Preference for microhabitat Akal
Benthic invertebrates	hli	Preference for microhabitat Lithal
Benthic invertebrates	hph	Preference for microhabitat Phytal
Benthic invertebrates	hpo	Preference for microhabitat POM
Benthic invertebrates	hot	Preference for other microhabitats
Benthic invertebrates	AHT1	Stone dwelling taxa score (Braukmann (Steinbesiedler))
Benthic invertebrates	cup	Current preference
Benthic invertebrates	RIB	Rheoindex according to Banning (calculated according to Illies)
Benthic invertebrates	fgr	Feeding type: grazers and scrapers
Benthic invertebrates	fmi	Feeding type: miners
Benthic invertebrates	fxy	Feeding type: xylophagous
Benthic invertebrates	fsh	Feeding type: shredders
Benthic invertebrates	fga	Feeding type: gatherers/collectors
Benthic invertebrates	faf	Feeding type: active filterer
Benthic invertebrates	fpf	Feeding type: passive filterer
Benthic invertebrates	fpr	Feeding type: predators
Benthic invertebrates	fpa	Feeding type: parasites
Benthic invertebrates	fot	Feeding type: other
Benthic invertebrates	fspez	Feeding type: flag
Benthic invertebrates	fspez_typ	Feeding type: main group
Benthic invertebrates	lss	Locomotion type: swimming/scating
Benthic invertebrates	lsd	Locomotion type: swimming/diving
Benthic invertebrates	lbb	Locomotion type: burrowing/boring
Benthic invertebrates	lsw	Locomotion type: sprawling/walking
Benthic invertebrates	lse	Locomotion type: (semi)sessil
Benthic invertebrates	lot	Locomotion type: other
Benthic invertebrates	acidclass new	Acid class (new) according Braukmann & Biss 2004
Benthic invertebrates	acidclass	Acid class (old) according to Braukmann 2000
Benthic invertebrates	AcidScore	Acid Score (Hendrikson & Medin)
Benthic invertebrates	SAI	Swedish acid index
Benthic invertebrates	ltiv	Lake outlet index (LTI): LP value
Benthic invertebrates	ltig	Lake outlet index (LTI): weighting factor
Benthic invertebrates	SPEAR_org	SPEAR organic
Benthic invertebrates	SPEAR_pest	SPEAR pesticides
Benthic invertebrates	sorg	SPEAR organic
Benthic invertebrates	SPEAR_art	SPEAR toxic (taxon level)
Benthic invertebrates	SPEAR_fam	SPEAR toxic (family level)
Benthic invertebrates	salfr	Salinity preference < 0.5
Benthic invertebrates	salol	Salinity preference 0.5 - < 5
Benthic invertebrates	salme	Salinity preference 5 - < 18
Benthic invertebrates	salpo	Salinity preference 18 - 30

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Benthic invertebrates	saleu	Salinity preference > 30
Benthic invertebrates	ll1	Length of life < 1 year
Benthic invertebrates	ll2	Length of life > 1 year
Benthic invertebrates	glow1	Generations per year: < 1
Benthic invertebrates	gequ1	Generations per year: 1
Benthic invertebrates	ghig1	Generations per year: > 1
Benthic invertebrates	masg	Mayfly average score: group
Benthic invertebrates	mass	Mayfly average score: value
Benthic invertebrates	masl	Mayfly average score: value (large rivers)
Benthic invertebrates	masgl	Mayfly average score: group (large rivers)
Benthic invertebrates	Mod1	Austrian Sensitive Taxa Score (number of sensitive taxa)
Benthic invertebrates	dsfis	Danish stream fauna index (DSFI): Family
Benthic invertebrates	dsfi1	Danish stream fauna index (DSFI): Indicator- group 1
Benthic invertebrates	dsf2	Danish stream fauna index (DSFI): Indicator- group 2
Benthic invertebrates	dsf3	Danish stream fauna index (DSFI): Indicator- group 3
Benthic invertebrates	dsf4	Danish stream fauna index (DSFI): Indicator- group 4
Benthic invertebrates	dsf5	Danish stream fauna index (DSFI): Indicator- group 5
Benthic invertebrates	dsf6	Danish stream fauna index (DSFI): Indicator-group 6
Benthic invertebrates	DSFI_Score	Danish stream fauna index (DSFI): Score
Benthic invertebrates	DSFI_Group	Danish stream fauna index (DSFI): diversity group
Benthic invertebrates	DSFI_IG	Danish stream fauna index (DSFI): Indicator group (IG)
Benthic invertebrates	ibef	Indice Biotico Esteso (IBE): family
Benthic invertebrates	ibeg	Indice Biotico Esteso (IBE): indicator group
Benthic invertebrates	ibell	Indice Biotico Esteso (IBE): limit (low)
Benthic invertebrates	ibelh	Indice Biotico Esteso (IBE): limit (high)
Benthic invertebrates	bbif	Belgian biotic index (BBI): family
Benthic invertebrates	bbig	Belgian biotic index (BBI): Indicator group
Benthic invertebrates	bmwp	Biological Monitoring Working Party: Score (German version)
Benthic invertebrates	bmwpf	Biological Monitoring Working Party: Familie (German version)
Benthic invertebrates	bmwpe	Biological Monitoring Working Party: Score (Spanish version)
Benthic invertebrates	bmwpef	Biological Monitoring Working Party: Familie (Spanish version)
Benthic invertebrates	bmwppl	Biological Monitoring Working Party: Score (Polish Version)
Benthic invertebrates	bmwpfpl	Biological Monitoring Working Party: Familie (Polish Version)
Benthic invertebrates	bmwphu	Biological Monitoring Working Party: Score (Hungarian Version)
Benthic invertebrates	bmwpfhu	Biological Monitoring Working Party: Familie (Hungarian Version)
Benthic invertebrates	bmwpgr1	Biological Monitoring Working Party: Score (abundances <= 1%, Greek version)
Benthic invertebrates	bmwpgr2	Biological Monitoring Working Party: Score (abundances 1% - 10%, Greek Version)

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Benthic invertebrates	bmwpgr3	Biological Monitoring Working Party: Score (abundances > 10%, Greek Version)
Benthic invertebrates	bmwpfgr	Biological Monitoring Working Party: Group (Greek Version)
Benthic invertebrates	bmwpcz	Biological Monitoring Working Party: Score (Czech Version)
Benthic invertebrates	bmwpfcz	Biological Monitoring Working Party: Familie (Czech Version)
Benthic invertebrates	LIFE	British LIFE-Index
Benthic invertebrates	awic	British AWIC-Index
Benthic invertebrates	asdi	Austrian structure index
Benthic invertebrates	sel_EPTD	Intercalibration common metric
Fish	dia	Diadromous species (migration)
Fish	Pot	Potamodromous species: migrating within streams (freshwater)
Fish	nom	No migration
Fish	oce	Oceanodromous species: migrating between fresh and salt waters
Fish	ldm	Long distance (migration)
Fish	mc	Migration-classified individuals: number of taxa or ind. With indicator values
Fish	pel	Pelagic: open water area above the bottom
Fish	ben	Benthopelagic: area near the bottom
Fish	dem	Demersal: area just above the benthic zone (forms a layer of the larger profundal zone)
Fish	hc	Habitat-classified individuals/taxa: number of taxa or ind. With indicator values
Fish	rhe	Rheophilic: fish prefer to live in a habitat with high flow conditions and clear water
Fish	lim	Limnophilic: fish prefer to live, feed and reproduce in a habitat with slow flowing to stagnant conditions
Fish	eur	Eurytopic: fish that exhibit a wide tolerance of flow conditions, although are generally not considered to be to be rheophilic
Fish	rc	Rheophily-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	benf	Benthivorous: habitat of fish that feed on bottom-dwelling organisms
Fish	wat	Water column: habitat of fish that feed in the open water zone
Fish	fh	Feeding habitat-classified individuals/taxa
Fish	phy	Phytophilic: reproduction habitat of plant spawner
Fish	lit	Lithophilic: reproduction habitat of rock and gravel spawners with benthic larvae
Fish	pli	Phyto-lithophilic: reproduction habitat of non-obligatory plant spawner
Fish	psa	Psammophilic: reproduction habitat of sand spawner
Fish	oth	Other reproduction habitats
Fish	re	Reproduction-classified individuals/taxa: number of taxa or ind. with indicator value
Fish	fre	Freshwater (salinity)

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Fish	fbr	Freshwater-brackish (salinity)
Fish	brm	Brackish-marine (salinity)
Fish	fbm	Freshwater-brackish-marine (salinity)
Fish	fma	Freshwater-marine (salinity)
Fish	sc	Salinity-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	inv	Invertivorous: feeding on invertebrates
Fish	pis	Piscivorous: feeding on fish
Fish	fphy	Phytophagous: feeding on plants
Fish	omn	Omnivorous: feeding on all kinds of foos indiscriminately
Fish	car	Carnivorous: feeding on animals
Fish	foth	Other (feeding diet)
Fish	fc	Feeding diet-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	ls1	Lifespan is less than 8 years (life span 1)
Fish	ls2	Lifespan is between 8 to 15 years (life span 2)
Fish	ls3	Lifespan is more than 15 years (life span 3)
Fish	lsm	Averaged species life span
Fish	lc	Number of taxa or ind. with indicator values
Fish	bl1	Body length is less than or equal 20 cm (body length 1)
Fish	bl2	Body length is between 20 to 39 cm (body length 2)
Fish	bl3	Body length is more than or equal 39 cm (body length 3)
Fish	blm	Averaged species body length
Fish	bc	Body length-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	sh1	Ratio is less than or equal 4,35 (shape factor 1)
Fish	sh2	Ratio is between 4,35 and 4,78 (shape factor 2)
Fish	sh3	Ratio is between 4,78 and 5,6 (shape factor 3)
Fish	sh4	Ratio is more than 5,6 (shape factor 4)
Fish	shm	Averaged species shape factor
Fish	sh	Body length-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	sw1	Ratio is less than or equal 0,38 (swimming factor 1)
Fish	sw2	Ratio is between 0,38 and 0,43 (swimming factor 2)
Fish	sw3	Ratio is more than 0,43 (swimming factor 3)
Fish	swm	Averaged species swimming factor
Fish	sw	Swimming factor-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	ma1	Females mature before or equal 2 years old for the first time (maturity 1)
Fish	ma2	Females mature between 2 and 3 years old for the first time (maturity 2)
Fish	ma3	Females mature before or equal 3 and 4 years old for the first time (maturity 3)

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Fish	ma4	Females mature before or equal 4 and 5 years old for the first time (maturity 4)
Fish	ma5	Females mature after or equal 5 years old for the first time (maturity 5)
Fish	mam	Averaged species female maturity
Fish	ma	Female maturity-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	st1	Fish spwan in winter time (spawning time 1)
Fish	st2	Fish spwan in summer time (spawning time 2)
Fish	st	Female maturity-classified individuals/taxa
Fish	ip1	Incubation time is less than or equal 7 days (incubation period 1)
Fish	ip2	Incubation time is between 7 and 14 days (incubation period 2)
Fish	ip3	Incubation time is more than 14 days (incubation period 3)
Fish	ipm	Averaged species incubation period
Fish	ip	Incubation period-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	fe1	Number of oocytes is less than or equal 55000 (fecundity 1)
Fish	fe2	Number of oocytes is between 55000 and 60000 (fecundity 2)
Fish	fe3	Number of oocytes is more than 60000 (fecundity 3)
Fish	fem	Averaged species fecundity
Fish	fe	Fecundity-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	fr1	Relative fecundity is less than or equal 57 (fecundity relation 1)
Fish	fr2	Relative fecundity is between 57 and 200 (fecundity relation 2)
Fish	fr3	Relative fecundity is more than 200 (fecundity relation 3)
Fish	frm	Averaged species relative fecundity
Fish	fr	Relative fecundity-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	ed1	Egg diameter is less than 1,35 mm (egg diameter 1)
Fish	ed2	Egg diameter is between 1,35 and 2 mm (egg diameter 2)
Fish	ed3	Egg diameter is more than 2 mm (egg diameter 3)
Fish	edm	Averaged species egg diameter
Fish	ed	Relative fecundity-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	ll1	Length of larval fish is less than or equal 4,2 cm (larval length 1)
Fish	ll2	Length of larval fish is between 4,2 and 6,3 cm (larval length 2)
Fish	ll3	Length of larval fish is more than 6,3 cm (larval length 3)
Fish	llm	Averaged species larval length

Table A1 (cont.): The table presents the main characteristics of the metrics of the three organism groups. Calculation of metrics comprises additional values on the basis of number of taxa, relative abundance, and or the logarithm of relative abundance.

BQE	Abbreviation	Description
Fish	ll	Larval length-classified individuals/taxa: number of taxa or ind. with indicator values
Fish	nop	No protection (parental care)
Fish	pnh	Protection with nester or eggs hiders (parental care)
Fish	nnh	No protection with nester or eggs hiders (parental care)
Fish	ld1	Larval stage is less than 12 days (larval duration 1)
Fish	ld2	Larval stage is between 12 and 25 days (larval duration 2)
Fish	ld3	Larval stage is more than 25 days (larval duration 3)
Fish	tol	In general tolerant to to usual (national) water quality parameters.
Fish	im	In general tolerant/intolerant to usual (national) water quality parameters.
Fish	intol	In general intolerant to usual (national) water quality parameters.
Fish	O2tol	In general tolerant to to usual (national) water quality parameters.
Fish	O2im	In general tolerant/intolerant to usual (national) water quality parameters.
Fish	O2intol	In general intolerant to usual (national) water quality parameters.
Fish	TOXtol	In general tolerant to to usual (national) water quality parameters.
Fish	TOXim	In general tolerant/intolerant to usual (national) water quality parameters.
Fish	TOXintol	In general intolerant to usual (national) water quality parameters.
Fish	htol	Tolerant to habitat degradation
Fish	him	Tolerant/intolerant to habitat degradation
Fish	hintol	Intolerant to habitat degradation
Fish	euther	Eurytherm (Temperature)
Fish	ShW	Shannon-Wiener Index (Diversity)
Fish	FRI	Fish Region Index (River zonation)

Annex 2

Table A2: Selected statistical descriptors of abiotic and biotic parameters per broad type. The numbers in brackets refer to the benthic invertebrate data subset (see text for details).

Variable group	Parameter	Unit	Broad type 4			Broad type 5			Broad type 9		
			Min	Median	Max	Min	Median	Max	Min	Median	Max
Physico-chemistry	Chloride	[mg/l]	29.15	88.08	451.50	5.22	45.58	730.63	2.50 (2.50)	23.78 (22.05)	1417.10 (1417.10)
	Conductivity	[µS/m]	45.00	88.53	197.25	17.83	70.85	304.50	3.80	32.26	406.50
	Total nitrogen	[mg/l]	0.02	4.25	10.38	0.00	4.99	31.81	0.00	2.70	10.43
	Nitrate	[mg/l]	0.01	2.85	9.56	0.00	2.93	17.95	0.00 (0.00)	2.04 (1.92)	10.45 (7.75)
	Oxygen concentration	[mg/l]	5.64	9.47	13.90	1.45	9.54	18.00	6.50 (6.50)	10.56 (10.79)	12.80 (12.80)
	Total phosphorus	[mg/l]	0.02	0.13	0.40	0.01	0.15	1.20	0.00 (0.00)	0.05 (0.05)	0.82 (0.82)
	Water temperature	[°C]	3.80	13.82	24.00	0.60	12.06	25.60	4.80 (5.45)	10.87 (10.40)	21.70 (16.01)
Hydromorphology	Variation of width	—(rank)	1.17	5.79	7.00	0.00	4.11	7.00	0.00 (0.00)	4.98 (4.99)	7.00 (7.00)
	Riparian zone	—(rank)	1.00	5.72	7.00	0.00	5.50	7.00	0.00 (0.00)	5.42 (5.41)	7.00 (7.00)
	Longitudinal bars	—(rank)	1.00	5.45	7.00	0.00	5.64	7.00	0.00 (0.00)	4.42 (4.87)	7.00 (7.00)
	Curvature/Bends	—(rank)	1.80	6.00	7.00	0.00	5.94	7.00	0.00 (0.00)	4.99 (5.10)	7.00 (7.00)
	Transverse structures	—(rank)	3.38	6.57	7.00	0.00	6.65	7.00	0.00 (0.00)	4.00 (3.98)	7.00 (7.00)
	Special bed features	—(rank)	1.00	5.38	7.00	0.00	5.14	7.00	0.00 (0.00)	3.86 (4.27)	7.00 (7.00)
	Flow variation	—(rank)	2.11	5.00	7.00	0.00	4.92	7.00	0.00 (0.00)	4.00 (4.09)	6.75 (6.75)
	Substrate diversity	—(rank)	2.00	4.13	7.00	0.00	3.76	7.00	0.00 (0.00)	4.04 (4.11)	6.76 (6.76)
	Bank vegetation	—(rank)	1.11	4.00	7.00	0.00	4.09	7.00	0.00 (0.00)	3.89 (3.84)	7.00 (6.63)
	Special bank features	—(rank)	1.11	5.00	7.00	0.00	5.84	7.00	0.00 (0.00)	4.36 (4.55)	7.00 (6.98)
Hydrology	% of floods in 60d period	-	0.00	0.01	7.00	-	-	-	-	-	-
	Base flow index	-	0.00	0.03	0.11	-	-	-	-	-	-

Table A2 (cont.): Selected statistical descriptors of abiotic and biotic parameters per broad type. The numbers in brackets refer to the benthic invertebrate data subset (see text for details).

Variable group	Parameter	Unit	BT4			BT5			BT9		
			Min	Median	Max	Min	Median	Max	Min	Median	Max
Hydrology	Low pulse count	-	0.00	0.34	0.86						
	Low pulse duration	-	0.00	0.16	0.86	-	-	-	-	-	-
	High pulse count	-	0.00	0.33	0.86	-	-	-	-	-	-
	High pulse duration	-	0.00	0.00	1.00	-	-	-	-	-	-
	Rise rate	-	0.00	0.23	4.74	-	-	-	-	-	-
	Fall rate	-	0.00	0.24	8.30	-	-	-	-	-	-
	Number of reversals	-	0.00	0.76	2.99	-	-	-	-	-	-
	Extreme low duration	-	0.00	0.28	1.57	-	-	-	-	-	-
	Extreme low timing	-	0.00	0.98	2.37	-	-	-	-	-	-
	Extreme low frequency	-	0.00	0.52	2.37	-	-	-	-	-	-
	High flow duration	-	0.00	0.00	0.89	-	-	-	-	-	-
	High flow timing	-	0.00	1.19	2.34	-	-	-	-	-	-
	High flow frequency	-	0.00	0.63	2.34	-	-	-	-	-	-
	Small Flood duration	-	0.00	0.50	5.50	-	-	-	-	-	-
	Large flood duration	-	0.00	0.00	12.00	-	-	-	-	-	-
	Large flood timing	-	0.00	0.00	126.00	-	-	-	-	-	-
	Zero flow days	-	0.00	0.00	148.00	-	-	-	-	-	-
	Small floods	-	0.00	2.00	148.00	-	-	-	-	-	-
Natural variables	Catchment	[km²]	0.00	326.81	4964.21	0.93	22.64	493.85	3.12 (10.17)	24.70 (39.31)	390.82 (390.82)
	Height	[m]	16.11	55.35	4464.95	15.29	54.94	190.47	44.88 (67.64)	237.80 (228.64)	599.81 (479.56)

Annex 3

Table A3: Three top variable loadings of the first three PCA Axes. PCA was performed for every broad type and every organism group for every stressor group. The variables in brackets for BT9 are corresponding to the subset with German river type 5 only.

Biological group	BT	Stressor group	PCA Axis	% Variance explained	Factor loading (top three)		
Fish	4	Physico-chemistry	1	41.72	Conductivity (-0.63)	Chloride (-0.57)	Total phosphorus (-0.43)
			2	25.00	Oxygen concentration (0.71)	Water temperature (-0.58)	Chloride (0.32)
			3	19.50	Total phosphorus (0.68)	Water temperature (-0.61)	Chloride (0.33)
		Hydromorphology	1	48.97	Substrate diversity (-0.36)	Special bank features (-0.35)	Special bed features (-0.35)
			2	11.82	Curvature/Bends (0.60)	Transverse structures (-0.42)	Bank vegetation (-0.41)
			3	9.25	Transverse structures (-0.71)	Riparian zone (0.40)	Bank vegetation (0.33)
		Hydrology	1	34.30	High pulse count (0.32)	Low pulse count (0.28)	High flow frequency (0.28)
			2	13.47	Baseflow index (0.49)	Extreme low frequency (0.43)	Fall rate (0.37)
			3	9.43	Large flood timing (-0.54)	Large flood duration (-0.37)	High flow duration (0.36)
Benthic Invertebrates	4	Physico-chemistry	1	36.13	Chloride (0.63)	Conductivity (0.61)	Water temperature (0.34)
			2	25.01	Oxygen concentration (0-0.58)	Water temperature (0.57)	Nitrate (-0.50)
			3	17.03	Total phosphorus (-0.96)	Water temperature (0.19)	Oxygen concentration (0.13)
		Hydromorphology	1	44.85	Special bed features (-0.40)	Variation of width (-0.39)	Longitudinal bars (-0.36)
			2	13.52	Bank vegetation (0.54)	Transverse structures (-0.52)	Substrate diversity (0.41)
			3	10.25	Transverse structures (0.58)	Curvature/Bends (-0.52)	Bank vegetation (0.39)
		Hydrology	1	29.66	High pulse count (-0.32)	Low pulse duration (-0.29)	Fall rate (-0.28)
			2	13.76	Baseflow index (-0.47)	Extreme low frequency (-0.42)	Rise rate (0.33)
			3	9.78	Large flood timing (0.48)	Large flood duration (0.41)	% of floods in 60d period (0.25)
Macrophytes	4	Physico-chemistry	1	35.94	Chloride (-0.64)	Conductivity (0.63)	Total nitrogen (-0.33)
			2	27.81	Oxygen concentration (-0.66)	Water temperature (0.57)	Total nitrogen (-0.45)
			3	16.00	Total phosphorus (0.95)	Oxygen concentration (-0.24)	Chloride (-0.19)
		Hydromorphology	1	42.44	Special bed features (-0.40)	Flow variation (-0.38)	Variation of width (-0.35)
			2	12.71	Bank vegetation (-0.50)	Transverse structures (0.49)	Special bed features (0.40)
			3	12.15	Curvature/Bends (-0.55)	Transverse structures (0.50)	Bank vegetation (0.43)
		Hydrology	1	35.60	High pulse count (0.31)	% of floods in 60d period (0.29)	High flow frequency (0.27)
			2	13.74	Baseflow index (0.45)	Extreme low frequency (0.39)	Large flood timing (-0.34)
			3	10.83	Large flood timing (-0.47)	Large flood duration (-0.42)	Baseflow index (-0.33)

Table A3 (cont.): Three top variable loadings of the first three PCA Axes. PCA was performed for every broad type and every organism group for every stressor group. The variables in brackets for BT9 are corresponding to the subset with German river type 5 only.

Biological group	BT	Stressor group	PCA Axis	% Variance explained	Factor loading (top three)		
Fish	5	Physico-chemistry	1	40.09	Chloride (0.59)	Conductivity (0.59)	Water temperature (0.44)
			2	22.76	Oxygen concentration (-0.78)	Water temperature (0.37)	Conductivity (-0.32)
			3	17.81	Total phosphorus (-0.90)	Oxygen concentration (-0.31)	Water temperature (0.23)
		Hydromorphology	1	65.71	Special bank features (0.35)	Special bed features (0.33)	Curvature/Bends (0.33)
			2	8.27	Bank vegetation (-0.67)	Riparian zone (-0.41)	Transverse structures (0.39)
			3	6.11	Flow variation (0.50)	Substrate diversity (0.48)	Transverse structures (0.37)
Benthic invertebrates	5	Physico-chemistry	1	35.37	Chloride (0.56)	Conductivity (0.56)	Total phosphorus (0.40)
			2	26.11	Nitrate (-0.57)	Oxygen concentration (-0.52)	Water temperature (0.33)
			3	13.06	Total phosphorus (-0.85)	Water temperature (0.38)	Nitrate (0.24)
		Hydromorphology	1	59.11	Special bank features (-0.36)	Transverse structures (-0.35)	Curvature/Bends (-0.34)
			2	9.45	Bank vegetation (0.68)	Special bed features (-0.36)	Transverse structures (-0.32)
			3	6.73	Riparian zone (0.66)	Flow variation (-0.53)	Substrate diversity (-0.41)
Macrophytes	5	Physico-chemistry	1	36.56	Chloride (-0.58)	Conductivity (-0.58)	Total phosphorus (-0.40)
			2	23.51	Oxygen concentration (0.65)	Water temperature (-0.58)	Total nitrogen (0.43)
			3	13.69	Total nitrogen (0.83)	Water temperature (0.37)	Conductivity (-0.29)
		Hydromorphology	1	55.48	Special bank features (0.36)	Longitudinal bars (0.35)	Variation of width (0.34)
			2	10.45	Bank vegetation (0.68)	Substrate diversity (0.33)	Special bed features (-0.32)
			3	6.93	Variation of width (-0.52)	Transverse structures (0.50)	Bank vegetation (0.37)

Table A3 (cont.): Three top variable loadings of the first three PCA Axes. PCA was performed for every broad type and every organism group for every stressor group. The variables in brackets for BT9 are corresponding to the subset with German river type 5 only.

Biological group	BT	Stressor group	PCA Axis	% Variance explained	Factor loading (top three)		
Fish	9	Physico-chemistry	1	44.96	Conductivity (0.59)	Chloride (0.56)	Total phosphorus (0.42)
			2	30.36	Water temperature (0.69)	Oxygen concentration (-0.60)	Total phosphorus (-0.29)
			3	13.13	Total phosphorus (-0.84)	Chloride (0.45)	Conductivity (0.22)
		Hydromorphology	1	63.57	Special bed features (0.36)	Special bank features (0.35)	Transverse structures (0.34)
			2	9.63	Riparian zone (-0.57)	Bank vegetation (-0.46)	Substrate diversity (0.46)
			3	7.73	Bank vegetation (-0.42)	Curvature/Bends (0.41)	Riparian zone (-0.40)
Benthic invertebrates	9	Physico-chemistry	1	44.58	Conductivity (-0.55) (Oxygen conentration (-0.52))	Chloride (-0.53) (Total phosphorus (-0.51))	Total phosphorus (-0.47) (Chloride (-0.50))
			2	26.72	Water temperature (-0.69) (Water temperature (0.62))	Oxygen concentration (0.62) (Nitrate (-0.46))	Nitrate (0.31) (Oxygen conentration (-0.43))
			3	12.18	Nitrate (0.88) (Nitrate (0.82))	Chloride (-0.34) (Chloride (-0.42))	Oxygen concentration (-0.18) (Total phosphorus (-0.30))
		Hydromorphology	1	70.89	Special bed features (0.34) (Flow variation (0.36))	Flow variation (0.34) (Special bank features (0.35))	Special bank features (0.33) (Variation of width (0.35))
			2	7.53	Riparian zone (0.73) (Riparian Zone (0.63))	Bank vegetation (0.51) (Bank vegetation (0.60))	Substrate diversity (-0.27) (Substrate diversity (-0.28))
			3	4.82	Substrate diversity (0.57) (Longitudinal bars (0.63))	Special bank features (-0.46) (Substrate diversity (-0.44))	Flow variation (0.38) (Curvature/Bends (-0.38))
Macrophytes	9	Physico-chemistry	1	40.23	Conductivity (-0.54)	Chloride (-0.53)	Total phosphorus (-0.43)
			2	24.93	Oxygen concentration (0.65)	Water temperature (-0.64)	Total nitrogen (0.25)
			3	13.92	Total nitrogen (0.91)	Total phosphorus (-0.31)	Chloride (-0.21)
		Hydromorphology	1	68.18	Special bed features (-0.35)	Substrate diversity (-0.34)	Special bank features (-0.34)
			2	8.76	Riparian zone (-0.70)	Bank vegetation (-0.53)	Substrate diversity (0.28)
			3	5.83	Substrate diversity (0.50)	Riparian zone (0.48)	Special bank features (-0.40)

Annex 4

Table A4: Results of the Varpart based on non linear regression explained variances.

Biological group	BT	Biological metric	Individual fractions			Full Model	Joint fraction
			Physico-chemistry	Hydromorphology	Hydrology		
Macrophytes	4	Total abundance of taxa submerged and broad leaved	0.150	0.273	0.000	0.922	0.503
		Heterophylly anchored taxa	0.080	0.205	0.015	0.911	0.611
		Relative number of taxa submerged myriophillid	0.206	0.048	0.003	0.897	0.640
		Ellenberg light index	0.110	0.110	0.095	0.873	0.559
		Relative number of taxa submerged and fine leaved	0.066	0.219	0.115	0.857	0.457
		Total number of taxa with meristem single basal growth point	0.042	0.057	0.020	0.850	0.730
		Relative abundance of taxa rooting caulescent hydrophyte	0.034	0.098	0.050	0.833	0.651
		Relative number of taxa rooting caulescent hydrophyte	0.125	0.057	0.052	0.791	0.556
		Number of reproductive organs per year and individual	0.163	0.046	0.005	0.784	0.569
		Ellenberg nitrogen number	0.020	0.183	0.057	0.771	0.510
		Morphology index	0.059	0.107	0.014	0.765	0.585
		Total number of taxa that are vallisnerid	0.051	0.142	0.032	0.743	0.518
		Overwintering organs	0.098	0.208	0.029	0.706	0.372
		Total abundance of magnopotamid taxa	0.000	0.067	0.000	0.701	0.921
		Relative number of myriophyllide taxa	0.143	0.046	0.000	0.684	0.700
		Reproduction by seeds	0.070	0.026	0.000	0.683	0.624
		Total number of ceratophyllid taxa	0.134	0.009	0.030	0.680	0.507
		Ellenberg salinity number	0.136	0.000	0.253	0.668	0.305
		Total number of meristem single basal growth point	0.000	0.066	0.000	0.547	0.632

Table A4 (cont.): Results of the Varpart based on non linear regression explained variances.

Biological group	BT	Biological metric	Individual fractions			Full Model	Joint fraction
			Physico-chemistry	Hydromorphology	Hydrology		
Macrophytes	4	Total abundance of taxa with plant life-form: Hemicryptophytes	0.052	0.029	0.024	0.732	0.627
		Total abundance of taxa with plant life-form: Geophytes	0.000	0.034	0.058	0.544	0.521
		Total number of taxa with aerenchyma	0.065	0.085	0.063	0.798	0.585
		Total abundance of free-floating taxa	0.034	0.000	0.081	0.616	0.615
Benthic invertebrates	4	Rhitron Typie Index	0.038	0.200	0.000	0.942	0.727
		Relative abundance of taxa preferring hyporhithral	0.035	0.055	0.000	0.842	0.772
		Relative number of taxa preferring hyporhithral	0.022	0.062	0.000	0.818	0.777
		Relative abundance	0.068	0.139	0.000	0.806	0.721
		Biocoenotic region index	0.026	0.594	0.000	0.845	0.245
		Relative number of taxa with microhabitat preference to lit	0.025	0.113	0.000	0.756	0.771
			0.000	0.095	0.000	0.698	0.648
		Relative number of grazers and scrapers	0.051	0.048	0.000	0.745	0.713
		Relative number of grazers, scrapers, gatherers and filterers	0.077	0.032	0.000	0.746	0.639
		Relatuve number of selected nonEP taxa	0.050	0.092	0.000	0.804	0.683
		Relative number of taxa preffering type RP	0.033	0.135	0.000	0.814	0.692
		Relative number of oligotrophic taxa	0.029	0.141	0.000	0.762	0.596
		Relative number of oligotrophic taxa (based on abundance classes)	0.089	0.153	0.000	0.811	0.577
		RETI	0.059	0.087	0.000	0.780	0.706
		SPEAR index	0.143	0.000	0.098	0.718	0.601
		German Fauna Index (type D 03)	0.060	0.107	0.018	0.818	0.633
	5	Relative abundance of selected EPTD taxa	0.005	0.163	-	0.696	0.528
		Average score per taxon (ASPT)	0.074	0.268	-	0.817	0.475
		SPEAR pesticides index	0.000	0.155	-	0.665	0.616

Table A4 (cont.): Results of the Varpart based on non linear regression explained variances.

Biological group	BT	Biological metric	Individual fraction			Full Model	Joint fraction
			Physico-chemistry	Hydromorphology	Hydrology		
Benthic invertebrates	5	Total number of Plecoptera and Trichoptera taxa	0.000	0.181	-	0.774	0.599
		Relative abundance of Plecoptera taxa	0.048	0.062	-	0.878	0.768
		Croation Saprobic Index (WEGL Method)	0.114	0.341	-	0.780	0.325
	9	German saprobic index (new version)	0.012	0.108	-	0.793	0.673
		Total number of EPT taxa	0.000	0.173	-	0.800	0.696
		SPEAR pesticides index	0.000	0.111	-	0.739	0.745
		Portuguese index	0.106	0.157	-	0.778	0.515
		Relative abundance of Crustacea taxa	0.000	0.184	-	0.741	0.600
		Rheo index	0.109	0.116	-	0.744	0.518
		Relative abundance of EP taxa	0.000	0.130	-	0.701	0.653
		SPEAR index	0.000	0.122	-	0.703	0.695
		Relative abundance of metarhithral taxa	0.088	0.153	-	0.779	0.538
		Share of alien species	0.329	0.029	-	0.431	0.072
		Relative abundance of taxa with reproduction by cemented isolated eggs	0.000	0.133	-	0.616	0.580
		German saprobic index (new version)	0.059	0.176	-	0.922	0.686
	9 subset	Total number of EPT taxa	0.187	0.144	-	0.558	0.227
		SPEAR pesticides index	0.000	0.037	-	0.671	0.790
		Portuguese Iindex	0.003	0.228	-	0.808	0.578
		Relative abundance of Crustacea taxa	0.000	0.285	-	0.810	0.590
		Rheoindex (Banning with abundance classes)	0.000	0.158	-	0.737	0.579
		Relative abundance of EP taxa	0.000	0.096	-	0.576	0.600
		Relative abundance of metarhithral taxa	0.173	0.089	-	0.779	0.516
		German saprobic index (new version)	0.039	0.196		0.882	0.647
		Relative abundance of EPT taxa	0.000	0.128		0.740	0.707
		SPEAR index	0.002	0.239		0.823	0.582
		Average score per taxon (ASPT)	0.011	0.235		0.836	0.591

Table A4 (cont.): Results of the Varpart based on non linear regression explained variances.

Biological group	BT	Biological metric	Individual fractions			Full Model	Joint fraction
			Physico-Chemistry	Hydromorphology	Hydrology		
Benthic Invertebrates	9 subset	German Fauna Index (type 5)	0.000	0.060		0.621	0.766
		Rheoindex (Banning with abundance)	0.000	0.105		0.684	0.684
		Share of alien species	0.448	0.000		0.448	0.029
		Relative abundance of eurythermic species (log)	0.022	0.040	0.023	0.995	0.909
Fish	4	Relative abundance of eurythermic species	0.020	0.026	0.002	0.993	0.944
		Relative number of species with number of oocytes > 60,000	0.077	0.171	0.027	0.925	0.650
		Relative number of species with shape factor 1	0.151	0.130	0.003	0.922	0.638
		Relative number of species averaged life span	0.049	0.081	0.055	0.913	0.729
		Relative abundance of species tolerant to habitat degradation(log)	0.022	0.003	0.130	0.908	0.754
		Relative number of species lifespan > 15 years	0.053	0.098	0.017	0.905	0.737
		Relative abundance of species lifespan < 8 years (log)	0.070	0.113	0.001	0.893	0.709
		Relative number of species body length less or equal 20 cm	0.320	0.139	0.198	0.891	0.235
		Relative abundance of invertivorous species (log)	0.050	0.066	0.000	0.883	0.801
		Average relative abundance of species shape factor (log)	0.026	0.078	0.101	0.878	0.672
		Relative abundance of omnivorous species (log)	0.056	0.047	0.008	0.877	0.766
		Relative abundance of species length of larval fish between 4.2 and 6.3 cm	0.056	0.090	0.000	0.863	0.737
		Relative number of benthopelagic species	0.033	0.023	0.368	0.857	0.433
		Relative number of phytophilic species	0.031	0.000	0.022	0.853	0.813
		Relative number of demersal species	0.040	0.000	0.368	0.853	0.458
		Relative number of species with number of oocytes =< 55,000	0.112	0.050	0.184	0.836	0.490
		Relative abundance of psammophilic species (log)	0.093	0.201	0.137	0.825	0.394

Table A4 (cont.): Results of the Varpart based on non linear regression explained variances.

Biological group	BT	Biological metric	Individual fractions			Full Model	Joint fraction
			Physico-Chemistry	Hydromorphology	Hydrology		
Fish	4	Average relative number of species fecundity	0.032	0.048	0.173	0.822	0.569
		Relative number of species with egg diameter between 1.35 and 2 mm	0.074	0.000	0.000	0.821	0.838
		Relative number of species body length between 20 to 39 cm	0.069	0.078	0.000	0.772	0.736
		Relative abundance of species with salinity preference to freshwater(log)	0.000	0.001	0.000	0.763	0.909
		Relative abundance of species with egg diameter between 1,35 and 2mm (log)	0.084	0.254	0.021	0.758	0.399
		Relative abundance of incubation period-classified species	0.123	0.124	0.073	0.634	0.313
		Relative number of species with salinity preference to freshwater-brackish	0.000	0.000	0.000	0.595	0.792
		Relative abundance of migration-classified species	0.096	0.081	0.039	0.575	0.359
		Shannon Wiener Index	0.005	0.000	0.000	0.386	0.722
		Relative abundance of species intolerant to habitat degradation	0.000	0.012	0.024	0.939	0.904
		Relative abundance of species tolerant to habitat degradation	0.022	0.003	0.130	0.908	0.754

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 6.1 Synthesis of stressor interactions and indicators
D6.1-3: Recommendations for more integrated river basin management and gaps in tools

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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

1. Introduction

1.1 About this deliverable

Within the MARS project various activities have been defined to further improve the understanding of multiple stressors in aquatic ecosystems and the response of the aquatic ecosystem to these stressors. The results from these activities are synthesised within WP6, which aims to bring together the results from the MARS WPs 3-5 in the framework of the concepts and approaches out-lined in WP2. The four main objectives of work package 6 are to:

1. Enhance understanding of stressor interactions and stressor-response relationships across scales, including the sensitivity of particular species, water-body types, or ecosystem services to common stress combinations and identify the species, habitat or landscape traits that make them sensitive;
2. Evaluate indicators that can diagnose changes in the ecological, quantitative and chemical status of water bodies across scales, and identify the principal stressors responsible for their deterioration;
3. Develop integrated risk assessment frameworks for European waters that link physico-chemical state and ecological status to socio-economic assessment through a greater understanding of exposure and sensitivity (response strength) of ecological status and key ecosystem services to multiple stressors;
4. Identify indicator and tool gaps for improving Integrated river basin management across Europe

This report is the Deliverable presenting the results for the 4th aim in relation to the work package task 6.4 entitled *'Integrated River basin management: evaluation of the MARS conceptual model'*.

Within task 6.4 an evaluation was made on the current river basin management practises for dealing with multiple stressors and how existing river basin management plans can be improved by incorporating elements of the MARS conceptual model and the MARS Tools (WP7). We reflect on these current practises and evaluate the MARS conceptual model and MARS Tools as an aid to daily water management on a local level. In particular, we focused on two key European policy/management questions: the benefits of sustaining ecological flows and the value of green infrastructure for natural water retention measures (flood regulation and drought mitigation) in relation to other water management questions, strategies and practises. These two topics are seen only as examples, as many other aspects of RBMPs could also be assessed.

Using a structured questionnaire and a workshop with WP4 case-study partners, their associated river basin managers, and a wider group of river basin managers from our applied partners and elsewhere in Europe, we were able to obtain an overview of the current practises in setting river basin management plans and selection of measures in relation to the multiple stressor challenges throughout Europe. The main aim of the questionnaire was to get a better understanding of the following questions:

- How does daily water management practice deal with the selection of cost-effective measures, for water bodies exposed to multiple stressors?
- Is knowledge on pressure interactions and biological response taken into account when selecting and prioritizing the measures?
- How can MARS best contribute to a potential gap in knowledge and tools from the perspective of the stakeholders?

We specifically challenged the workshop participants to link their current practises to the topics relevant within the MARS project and linked this to the potential need and usage of tools that could help identify the role of multi-stressor challenges within their daily management practises. With this information we defined how the conceptual model could be used in practice and what gaps in indicators or tools are currently hampering daily practise.

1.2 MARS Conceptual model

The MARS conceptual modelling framework presented in Figure 2 aims explicitly at combining the risk-assessment framework (left column), the driver-pressure-state-impact-response (DPSIR, middle column) framework and the ecosystem service cascade (right column) in a joint modelling framework that enables the investigation of the impacts of multiple stressors on biotic/abiotic state and on ecosystem services.

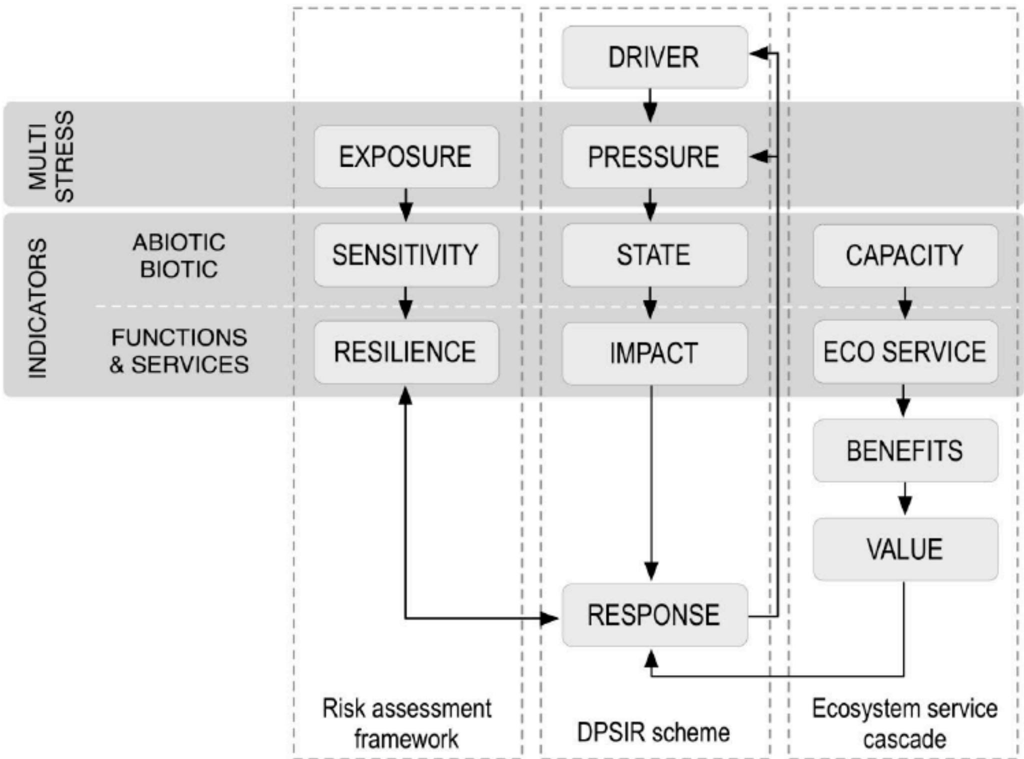

Figure 1 The MARS conceptual model for an integrated assessment framework (Hering et al, 2014).

Risk assessment combines the magnitude of a stressor (or a combination of stressors) with the consequences of exposure to it. The consequences are based on the sensitivity of the targeted indicators, e.g. species, habitats and ecosystem processes and services. The WFD status assessment follows 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 (Figure 2). The ecosystem service cascade model quantifies the capacity (i.e. their structures, processes and functions), links it to the flow of a specific service used by humans (assessed using socio-economic data), and finally translates into benefits of ecosystem services (Grizzetti B. et al, 2015).

There are obvious linkages between these three frameworks (Figure 2) through indicators of a water body's sensitivity or resilience to stressors, its status and its capacity to provide services. Further, management decisions ("response") are not only based on the state-impact chain through the DPSIR model, but also must consider ecosystem service values, too.

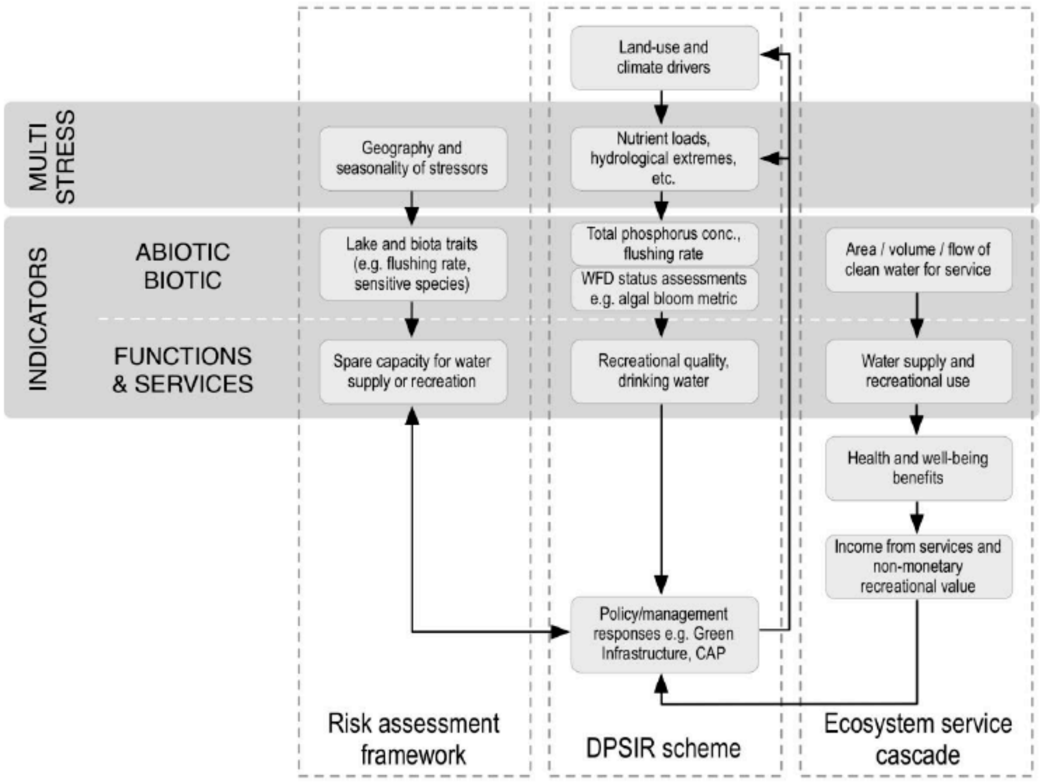

Figure 2 The MARS conceptual model for an integrated assessment framework exemplified for a lake affected by intense agriculture and climate change (Hering et al, 2014).

In the MARS project a total of 16 river basins covering the main European regions were selected to address representative stressor combinations. For each river basin, multiple stress effects on indicators are modelled by linking the outcome of "abiotic" models (including groundwater and surface water hydrology) to the biota and to ecosystem processes and services, either empirically or using process-based models (Ferreira et al, 2016).

The term *stressor* is not used in the terminology of the DPSIR framework. A stressor is change in environmental conditions that places stress on the health and functioning of an organism, population and/or ecosystem. Stressors can be either natural or anthropogenic in origin, and either direct (e.g. oxygen deficiencies) or indirect (e.g. lack of food availability due to stresses on prey species) in their effects on an ecosystem.

The vulnerability of different species to stressor effects can depend on factors such as life stage, habitat preference, reproductive cycle and community structure. Stressor effects are detected in ecosystems through measurable response variables, such as changes in ecological status or ecosystem service provision. Stressors can belong to either the pressure or state category in the driver-pressure-state-impact-response (DPSIR) causal framework used for describing the interactions between society and the environment (Birk et al, 2015).

1.3 MARS Tools

Within the MARS project we are developing a family of tools diagnosing and predicting multiple stressors in water resource management to support river basin management planning. In the following paragraph each tool is briefly described.

MARS Web-based Information System

The Information System is designed for water and river basin managers and allows access to information about multiple stressors as well as practical tools and models generated in MARS and other projects. The main components of this web-based information system are (1) an information library focussing on the DPSIR framework, common stressor combinations and their potential future impacts on aquatic ecosystems and the entire river basins. The implications of multiple stressors at a local scale are illustrated by the MARS case studies (2). A model selection tool (3) allows the user to find suitable modelling tools to be applied in river basin management planning based on different benchmark criteria. Guidance documents for river basin management planning and MARS key results (4) will be accessible on the MARS Information System as well.

MARS Diagnostic Tool

The MARS Diagnostic Tools aims to aid the diagnosis of the causes of deterioration at the water body scale. The tool will allow the user to identify major causes and its hierarchy of importance, both of which can inform the derivation of appropriate management and restoration measures to reduce the causes, or to mitigate its ecological effects. The tool requires the user to enter selected biological metric and index values commonly applied in European monitoring systems (e.g. % macro-invertebrate EPT taxa). A link to the information system will provide context-specific background information at the water body-scale. In addition, information on a broader geographic scale will be provided through the Scenario Analysis Tool (see below). The diagnosis will be based on Bayesian Networks, which combine the conditional probabilities of causes (stressors) and effects (biological metrics, indices) in a cause-effect network. The outcome summarises the ranked relatedness of causes to particular metrics/indices, which then informs the end user about the rank of importance of the potential causes of deterioration. Two prototypes on benthic invertebrate diagnosis in a common European mid-sized lowland river type and a phytoplankton diagnosis in a large lowland river type will be presented.

MARS Framework for combining abiotic and biotic models

MARS Framework for combining abiotic and biotic models The MARS project aims to develop a modelling framework for an integrated assessment of the effectiveness of restoration and mitigation measures in order to improve the ecological status of Europe’s rivers and lakes. European water managers are currently using a wide range of process-based (mechanistic) abiotic tools in river basin management, but adequate linkages to ecological response models are lacking. Through a probabilistic Bayesian modelling framework that is currently developed within the MARS project, abiotic and biotic models are combined for river basin management planning. This framework represents a set of variables and causal relationships based on observed data, expert judgement and assumptions. For all biological quality elements of the WFD, schemes with causal relationships are set up which link biotic status indicators (metrics, indices) to abiotic variables. These relationships are built in Bayesian belief networks, and can be used by water managers as a starting point for development of a specific Bayesian network approach for their catchment. Subsequently, for a selected number of MARS case studies, a Bayesian modelling approach is currently developed to predict the effects of measures to reduce to the impacts of multiple stressors on the ecological status. These case studies can also be used as examples for the use of BBN to assess the effectiveness of measures to improve the ecological status of Europe's rivers and lakes.

MARS Scenario Analysis Tool

The Scenario Analysis Tool is a web-based mapping system to evaluate and analyse multi-stressor conditions at the level of sub-catchment, river basin, region and Europe. Backbone of this scenario analysis system is a combined approach of the models GlobWB (water quantity) and MONERIS (water quality). The Scenario Analysis Tool will base on European wide information at a scale of appr. 50 km^2, comprising information on land use, nutrient emissions, water balances, groundwater and surface water flow, water quality, and information on hydro-morphology. Further, the tool will incorporate current conditions and future scenarios for selected pressures. The pressure indicator assessment will base on the pressure response relation ships derived in MARS (WP5) to identify potential or active pressure impeding to attain a good ecological status.

1.4 Aim of this report

The aim of this report is to give an overview of the current daily practises of river basin management in both defining river basin management plans and selection of measures and in which ways multiple pressures play a role in their decision making. We discuss the future need for tools and guidelines to fill gaps in indicators and tools and evaluate the use of the MARS conceptual model for two specific measures that were defined in the onset of the full project, being environmental flows and green infrastructures.

In line with the reporting of the European Environment Agency (EEA), we used the more common terms "pressures" rather than "stressors", "river basin" rather than "system" and "ecological status" rather than "ecosystem functioning" in the questionnaire sent to river basin managers.

2. Methods

2.1 Questionnaire for river basin managers

A multiple choice questionnaire with questions on the current daily practise on how measures are currently selected (part 1.) and the potential usefulness of the MARS-project results (specifically tools from WP7, part 2) was sent out to river basin managers to understand the response of their system to multiple pressures.

In total (approximately) 120 people were contacted. Most of these represent regional water managers, directly responsible for a given river basin, or set of water bodies within a larger river basin. Additionally, some water managers on national and international level were contacted (e.g. from the larger transnational river basins such as the Danube, Elbe and Rhine) and some water managers from local cities.

The full questionnaire can be found in Appendix 1. Questions focussed on identifying the responsible partners for defining RBMPs and the entities responsible for implementing the measures. The methods for selecting and evaluating (cost-effective) measures in relation to the socio-economic context were identified, including the most common types of measures applied. We also assessed the views of the respondents on multi pressures in their water systems and how they currently perceived the response of their waters to these multiple pressures. This was linked to assessment whether they would be interested in using new tools to further increase their understanding of their water systems with respect to this subject. In line with the reporting of the European Environment Agency (EEA), we used the more common terms “pressures” rather than “stressors”, "river basin" rather than "system" and "ecological status" rather than "ecosystem functioning" in the questionnaire sent to river basin managers.

The results of the questionnaire were discussed during the MARS stakeholder meeting organized in October 2016 in Den Helder, the Netherlands, which was attended by 9 stakeholders and 19 MARS partners. A list of attendees can be found in Appendix 3. The related discussion is used for input to the discussion chapter in this report.

2.2 Evaluation of the MARS conceptual model

During the Den Helder meeting in October 2016, a discussion was held within the MARS consortium on the usefulness of the MARS conceptual model (Hering et al, 2015) for river basin managers. The MARS conceptual model was used to describe the pressures and impacts on biological indicators in the various case studies within MARS in a harmonized way (Ferreira et al, 2016). For the two measures ‘green infrastructure’ and ‘environmental flows’ it was specifically checked if the conceptual model would offer a way to evaluate the effectiveness of these measures on the aquatic ecosystem.

3. Results

3.1 Results of the Questionnaire

A total of 38 responses were received on the questionnaire. These were coming from 23 countries, with a large reply from Dutch water managers (n=9) and from German water managers (n=4, including Elbe and Ruhr River Basin Responsible Managers) and the ICPD, ICPR and ISRBC. Respondents come from various governmental layers, ranging from international river basin committees to local and regional water managers.

In this questionnaire, respondents had the possibility to explain or detail their response in the category ‘other’ to enable a good representation of answers that were more complex than the predefined set of multiple-choice answers. In all cases, multiple scores per question were possible. These detailed answers are also summarized below the graphs per question. Although all questionnaires are based on the reply of 38 persons, the total sum of numbers might differ per question depending on the amount of options selected by individual respondents.

Part 1. – Current working approach for definition of RBMPs and selection of measures

1. Responsibilities of defining RBMPs and their implementation.

The responsible governmental bodies are not always the same as those that will implement the measures (Figure 3). It is often a higher governmental level that decides the measures (e.g. a ministry) than those carrying out the measures (a regional or local body). Thus it can also be that the technical assessment of the selection of measures is done by a different body than the one administratively responsible for the selection of the measures/reporting to the EU. For example, in Sweden the generic management plans were indicated as ‘too coarse and inaccurate to be used directly by the municipalities that are responsible for the implementation’ and needed to be further detailed to become workable per individual water body. Also the implementation of measures is often done in cooperation with a larger group of stakeholders or responsible authorities than the group responsible for defining the plans. For example, in Slovenia the ministry of environment is responsible for the overall proposal of plans (in cooperation with governmental institutions that provide technical information), but the responsible institutions for carrying out the measures depends on the type of measure, the property and administrative management. It is unclear how the communication between the parties responsible for selecting measures and implementing is arranged in detail, but often the miniseries or governmental departments are responsible for the monitoring of the measures after implementation.

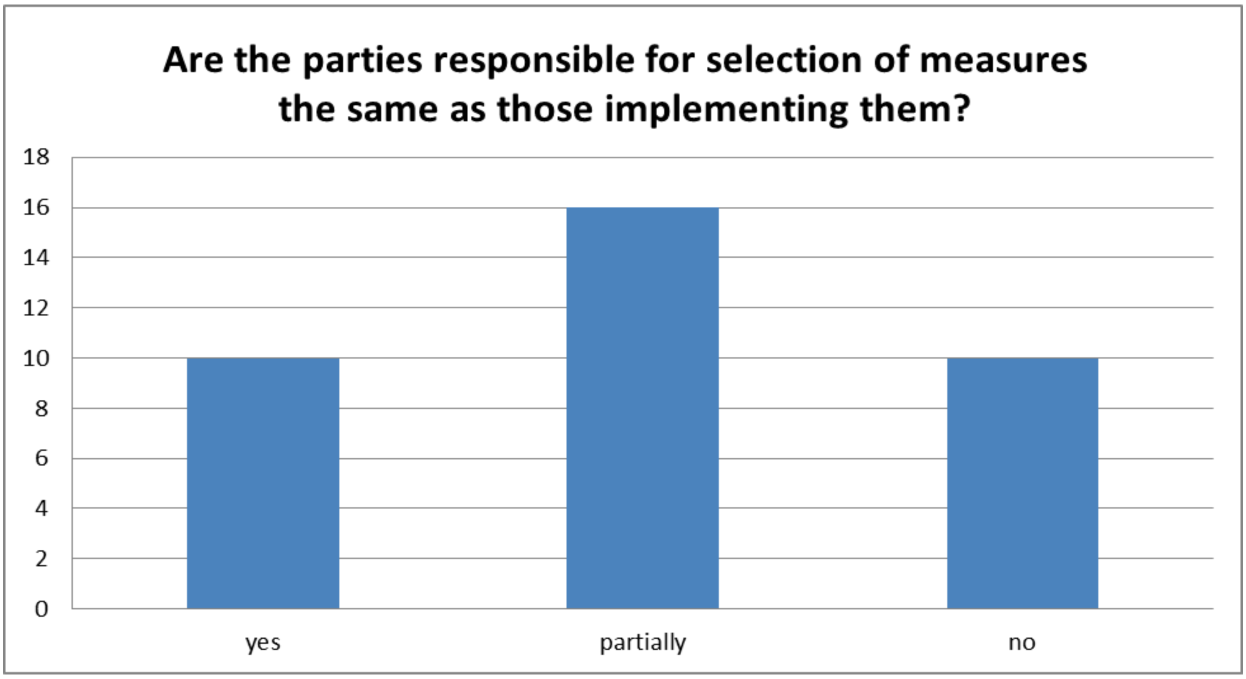

Figure 3 Overview of responses per category for the question 'Are the parties responsible for selection of measures the same as those implementing them?'

2. Methodologies to select measures (multiple answers are possible)

The methodologies to select measures can be based on multiple approaches, depending on the available time and data for doing so (Figure 4).

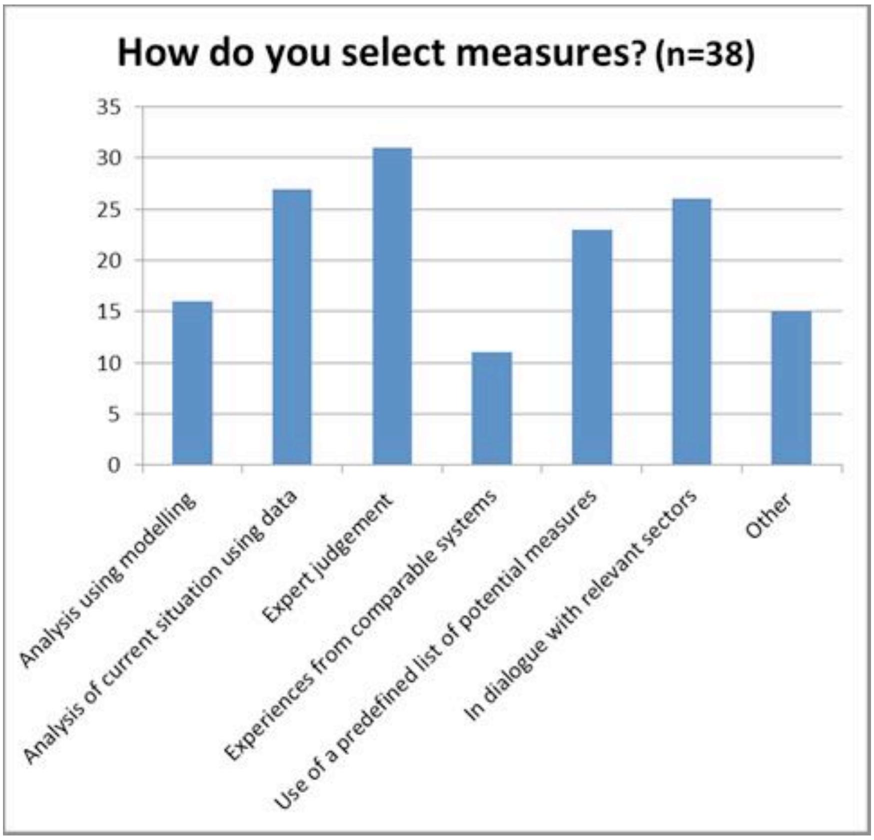

Figure 4 Overview of responses per category for the question 'How do you select measures?'

It is often a mix of 3 or 4 methodologies in which expert judgement and stakeholder involvement is often important. Many respondents indicate that the lack of sufficient resources makes that the selection might be less optimal than desired.

3. Prioritization of measures (multiple answers are possible)

The societal impact of measures is a clear reason for prioritization of measures (
Figure 5). Next to that there is a lot of uncertainty in how best to prioritize the measures in a multi-pressure situation. For example, in Portugal it is recognized that the lack of a system to weigh the importance of multiple pressures and the effect of multiple measures makes it more difficult to prioritize them. Although technical reasons (e.g. lack of tools) exist that make it difficult to perform quantitative analyses, the responses indicate that these technical aspects are playing a minor role in the prioritization of the measures. Using expert judgement is part of the process of prioritizing measures, and in this there is a search for measures that have impact on many pressures simultaneously. At the same time some countries are stricter in their methods for prioritizing measures than others. For example, in the German Federal State 'North-Rhine Westphalia' there exists no prioritization and all measures have to be realized and in Slovakia the EU regulations arising from the requirements of various EU Directives are followed very strictly taking into account mainly technical feasibility within the required time period. At the same time, in other countries the local stakeholders might have a say in the prioritization.

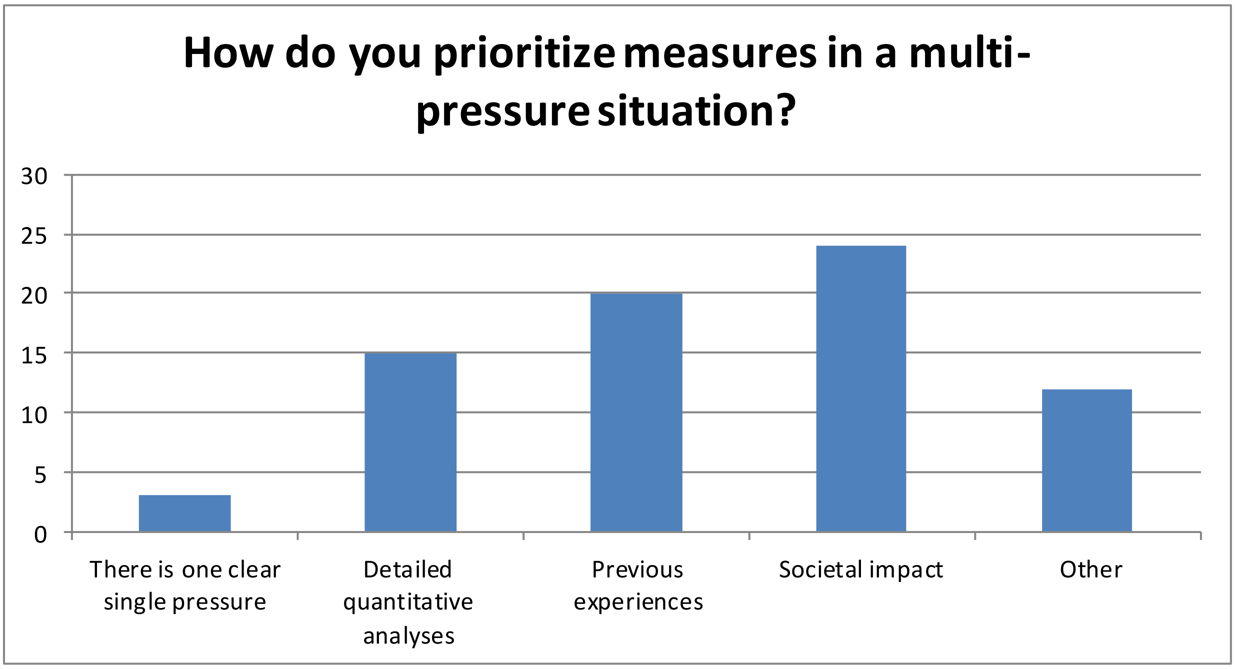

Figure 5 Overview of responses per category for the question 'How do you prioritize measures in a multipressure situation?'

4. Economic aspects of selecting measures

In most cases the economic aspects of measure prioritization are taken into account and cost-effectiveness is an important part of selecting measures (Figure 6). Only in 10 cases, all listed measures must be implemented. The political implications of measure selection are therefore an important aspect. The overall available budget to implement measures is an important boundary condition, but it is not clear if specific methods are used to define cost-effectiveness. This definition of ‘cost-effectiveness’ seems to be largely based on expert judgement (see also question 2 and 3). At the same time small scale no regret measures are easy to implement and will go through without much discussion. For example, a regional water authority in the Netherlands indicates that it is easier to take measures on terrains that are physically owned by the water authority, so they are often the most cost-effective ones to implement. The larger scale issues and impacts of measures are only considered after those easy measures are implemented.

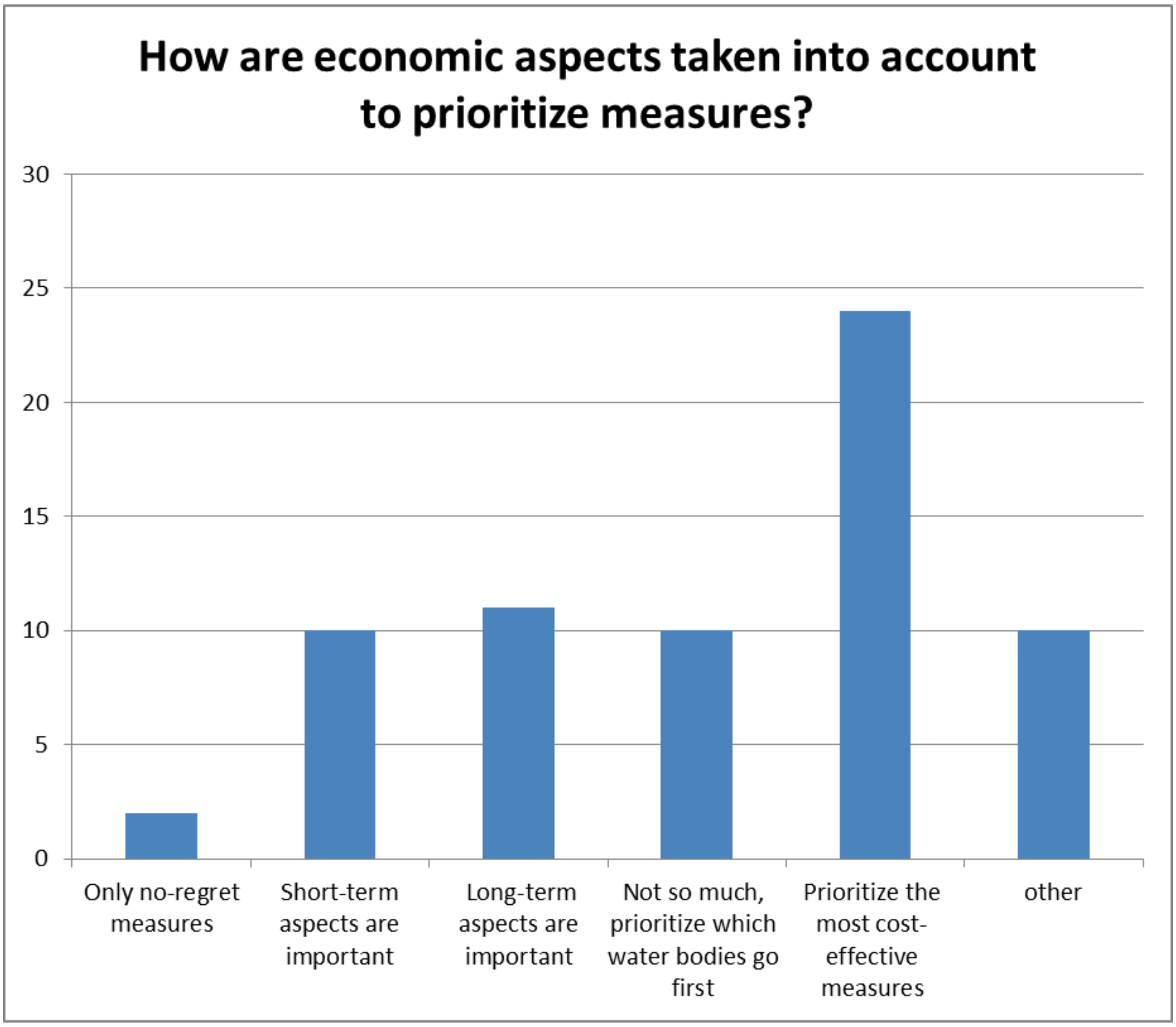

Figure 6 *Overview of responses per category for the question ‘How are economic aspects taken into account to prioritize measures?’*

5. Evaluation of measures

Judging the effect of measures can be done by using evaluation criteria that define their successfulness and also help understand the mechanisms by which the measure influences the water system. Monitoring and evaluating the impact of measures is a valuable but often expensive aspect of water management. The evaluation of measures is therefore most often done following the standard 'national' WFD ecological monitoring approach, and is often not specifically focussed on the true effect of an individual measure (Figure 7). For fish passages there might be specific monitoring of the effect of the fish passage, as this is such a prominent and visible measure also for the general public. It was pointed out by one respondent from the UK that if multiple measures are carried out within one water body at the same time (as is often the case), the monitoring may not be able to distinguish the individual impact of each single measure, making it difficult to score the effectiveness per measure. Many respondents indicate that there is too little budget for a proper evaluation of the functional success of a measure and that they regret not having the option to do this in a better manner.

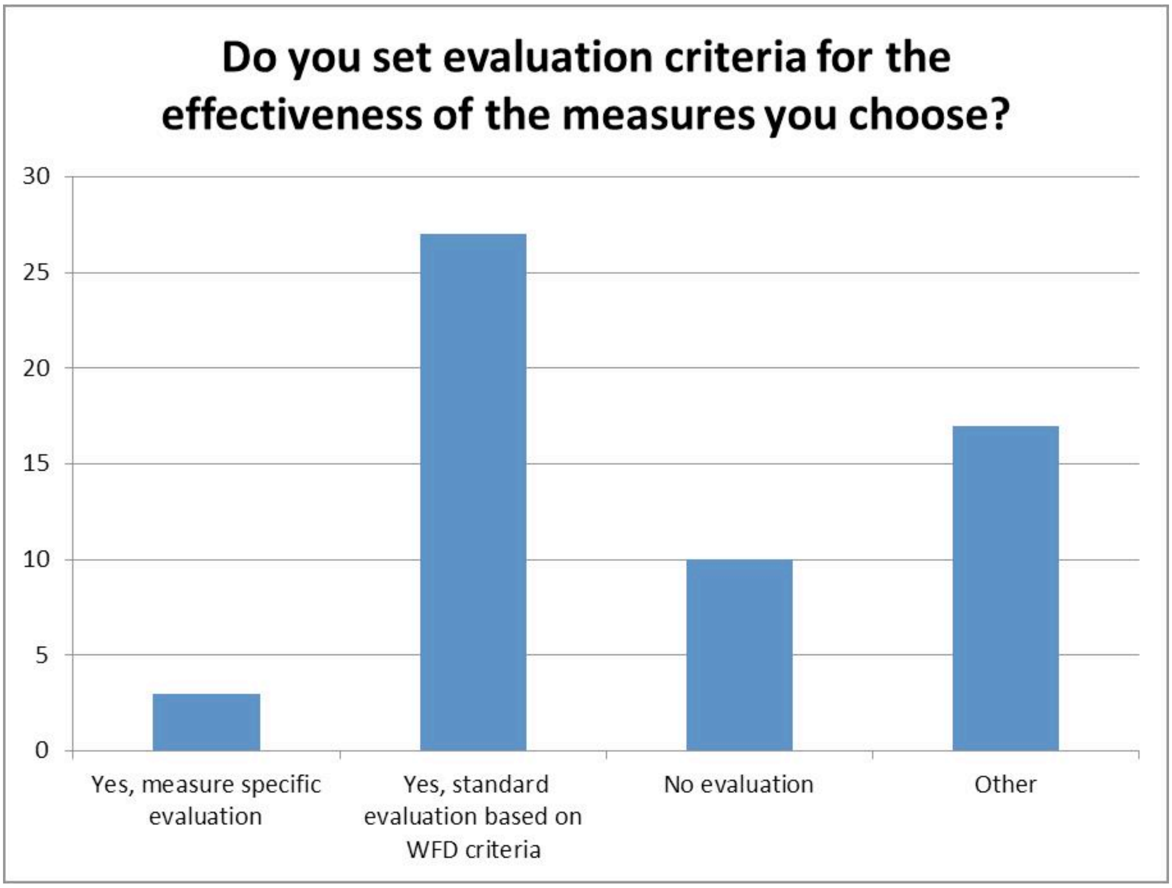

Figure 7 Overview of responses per category for the question 'Do you set evaluation criteria for the effectiveness of the measures you choose?'

6. Most common types of measures

The most common types of measures are the implementation of fish passages, and nutrient – related measures, being both related to point sources and diffuse sources (Figure 8). Some changes in bank morphology are carried out, but measures with regards to flow and groundwater regulation are less commonly carried out from an ecological perspective, rather they are being carried out for water quantity management. Although green infrastructures are listed regularly it is not always clear on what basis these are selected and whether this is for water quantity, water quality, ecological objectives or if they serve multiple purposes. For example some respondents indicate that these are predominantly chosen to target flood-related objectives and not ecological objectives.

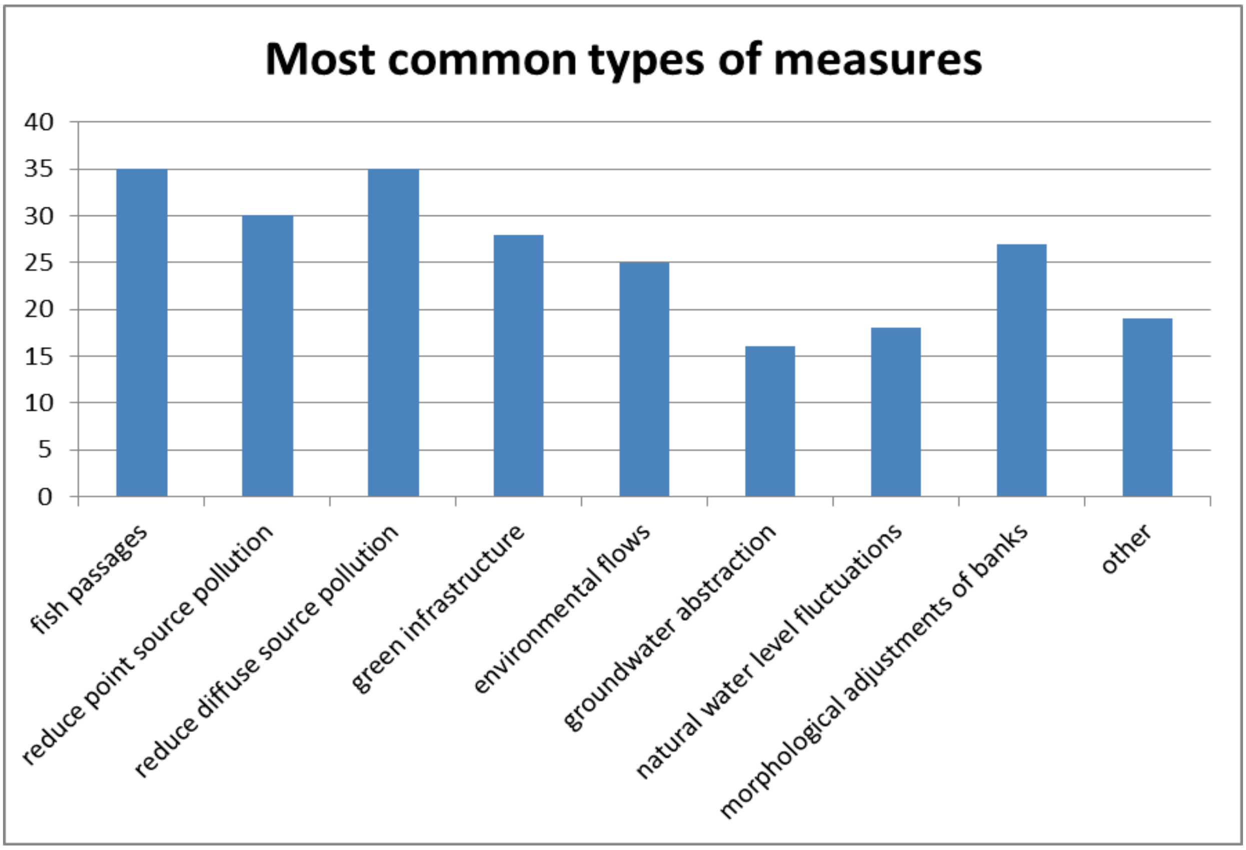

Figure 8 Overview of responses per category for the question 'What are the most common types of measures'

Some region specific measures were not listed in the full list of predefined measures (e.g. related to peat-soil acidification problems, micro-pollution and effects of specific land uses). Next to these predefined measures that 'alter' the current system (e.g. by implementing a fish passage where a fish barrier exists) there are also measures to improve the regular maintenance of shorelines and waterways. For example, in the Netherlands new methods are being assessed in which removal of excess aquatic vegetation is done in a more ecologically friendly way, using only partial removal rather than completely clearing the waterway from vegetation (Hendriks et al, 2016). In these cases the budget normally remains the same each year, but changing the working procedure might have benefits for the ecological functioning of the system.

Other measures that were also mentioned were measures related to changes in society, e.g. through legal functioning or educational measures in Poland (*Eds:* presumably for the general public), fishery regulation in Hungary and forestry measures in Finland.

7. Tools used to define RBMPs

Most often tools used to derive a RBMP are focussed on data handling and analysis (Figure 9). Available databases on local, national and international scales are often used to derive RBMPs in combination with spatially explicit data using GIS platforms. Specific process-based models are used less frequently, both those developed for individual users and open sources/commercially available tools. Note that the type of tools depends on the responsibilities of the respondent. Some respondents are responsible for the implementation of the RBMPs and therefore do not make use of tools. For example, it might be that some tools e.g. hydrological/engineering software for water quantity studies are used by others within the organisation, that inform ecologists in the selection of measures, while these are not considered as tools 'used' by those deriving the RBMPs. Yet, these tools are then still being used by the greater 'organization'.

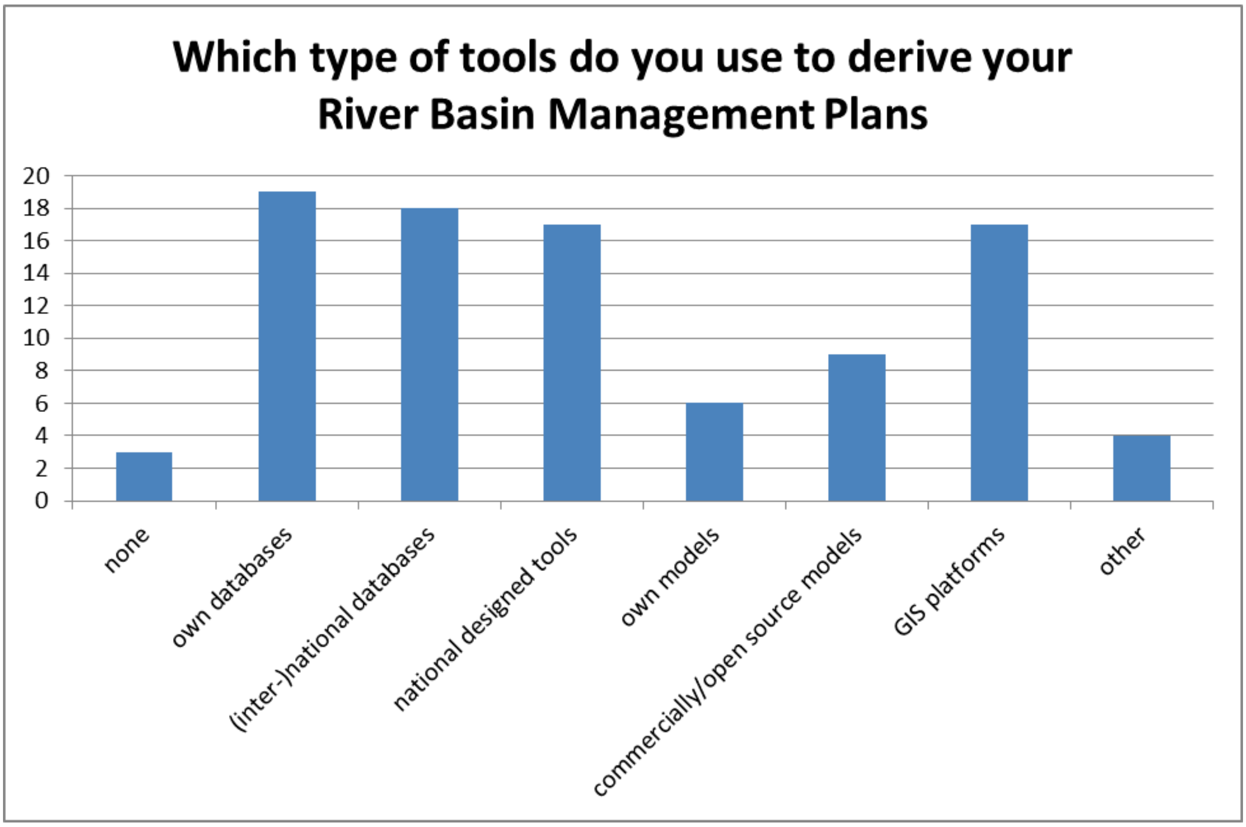

Figure 9 Overview of responses per category for the question 'Which type of tools do you use to derive your river basin management plans?'

8. Insight in current functioning of the water system

Many respondents indicate they are still struggling to gain a full understanding of the functioning of their water systems (being either a single water body or a full river basin), due to a lack of data or general knowledge e.g. on multiple pressure interaction (Figure 10). For example, a respondent from Czech Republic indicated that there are extensive data sets for flow, nutrients and chemical pollution in the water bodies themselves, but a lack of data for atmospheric deposition from agriculture and only expert judgement information of the effects of hydro-morphological changes. In general such disparity between the types of information sources creates a feeling of uncertainty in selecting the right measures. However, this does vary between individual water bodies within a management region, depending on the information available per water body. In general, many water authorities will have a given number of 'focus' water bodies, that either due to historic reasons (e.g. the settings or given challenges in these systems), or due to financial constraints, these focus water bodies act as demonstration sites to gather a great understanding and check the effectiveness of measures. Such focus water bodies are often better documented and better understood than other water bodies within the same authorities range.

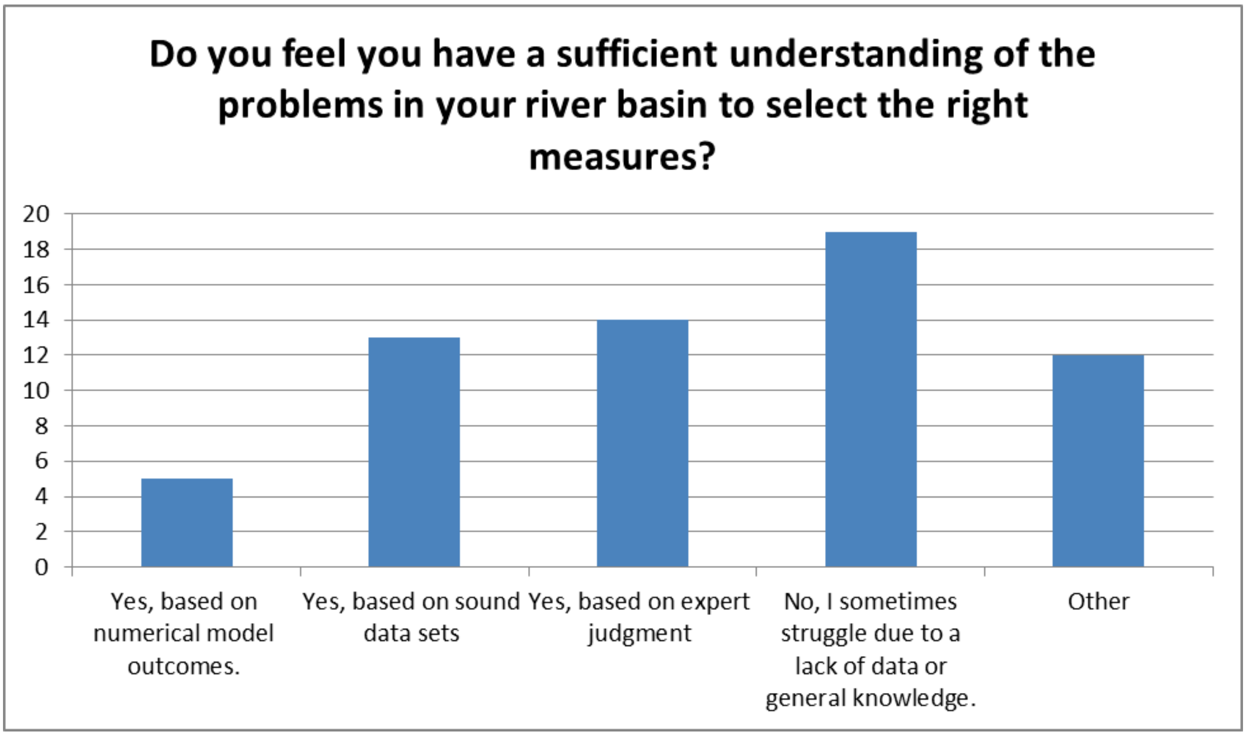

Figure 10 Overview of responses per category for the question 'Do you feel you have a sufficient understanding of the problems in your river basin to select the right measures?'

Part 2 – Usability of MARS results

In part 2 we asked questions related to multiple pressures and how these affect the daily practises of water managers.

2.1 Most common set of multiple pressures

The MARS project identified four major pressure combination options. These respondents often identified the pressure combination ‘morphological change and nutrients’ as the most important one in their system. Extreme temperature was most often listed as a problem in Dutch systems (7 times listed) and in Portugal and Greece (but strikingly was not mentioned in countries neighbouring the Netherlands with a similar climatic regime, or changes therein due to climate change). Organic pollution, changes in pH due to acidity in combination with nutrient stress and priority substances are sometimes mentioned as problematic, but not in many cases.

Often hydro-morphological pressure is considered ‘one’ type of pressure. However, as indicated by the Austrian respondent, splitting hydro-morphological pressures into morphological pressures, hydrological pressures and continuity disruptions is necessary as they all have different impacts on the ecosystem response to these pressures. In the questionnaire we specifically used this separation between the flow and morphological change, but not the continuity as a separate aspect.

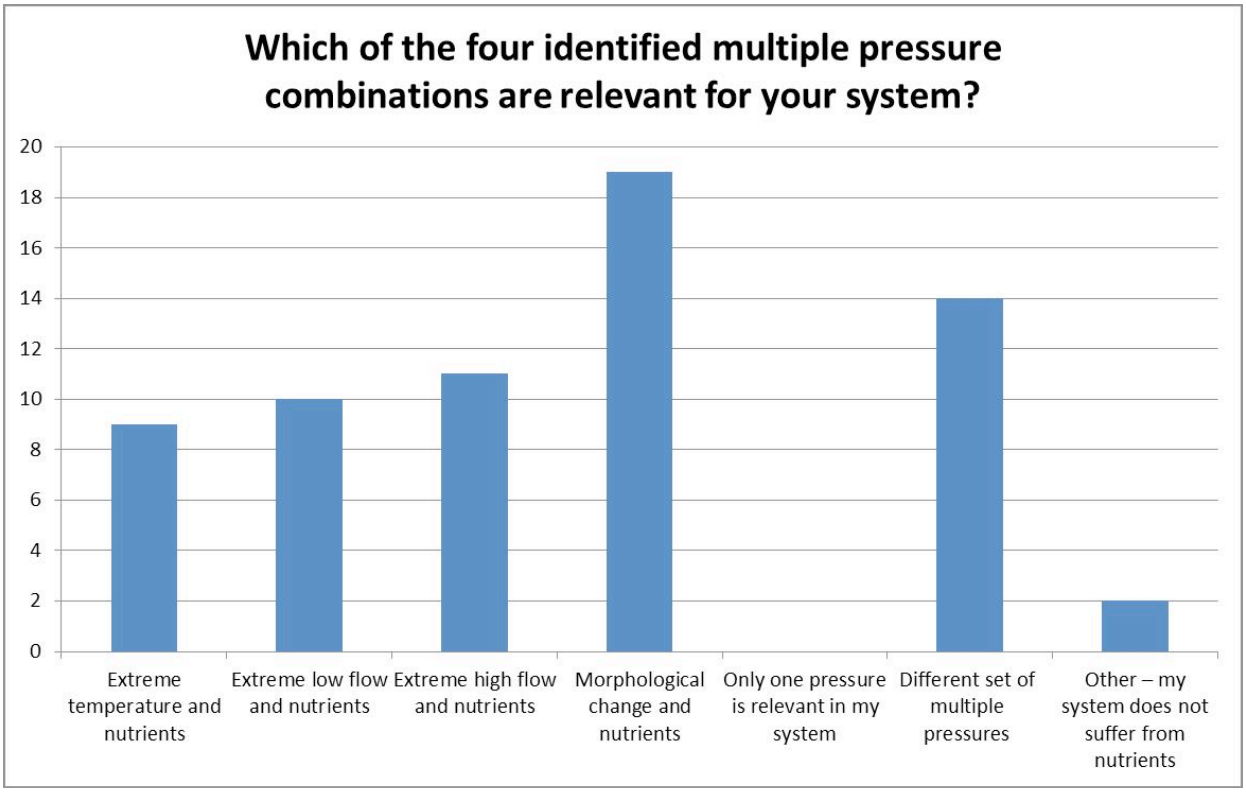

Figure 11 Overview of responses per category for the question ‘Which of the four identified multiple pressure combinations are relevant for your system?’

Some respondents feel that combining only two pressures is not enough to represent the multiple stress conditions they experience in their systems, especially in the denser populated areas of Europe, or as stated by the Belgium respondent: 'our country is a multiple-pressure environment with only a little bit of room for water'.

2.2 Understanding of the response of the aquatic ecosystem to multiple pressures

Many respondents indicate that they have a good understanding of the impact of multiple pressures on the aquatic ecosystem (or at least for a given water body, while they may not have this for all of their water bodies). Yet most respondents feel that they do not fully understand the impact of multiple pressures and whether the combined effect is worse than the single pressures, or less than the single pressures (Figure 12). For example, one respondent from the Netherlands indicates that the nutrient levels in his area are so high that they are dominating the potential impact of other pressures that he knows are also present in system. Also, an example from Germany shows that the lack of shoreline vegetation and straightened channels, also limits the culmination of excessive amounts of organic sediments on the river bed. At the same time it is generally accepted that shoreline vegetation is beneficial to stream ecosystems and that straightening of channels is also considered as an undesired change in hydro-morphology. So, in some cases a single pressure might overpower the effects of other pressures to such an extent that the overall result is less negative than the two single pressures would be if just simply added.

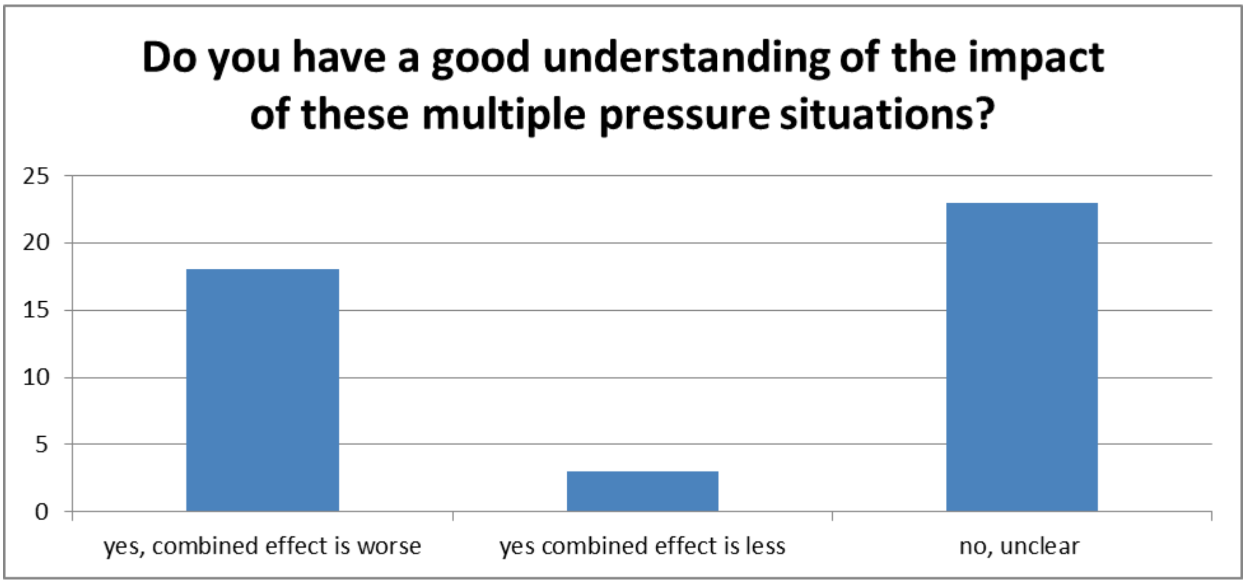

Figure 12 Overview of responses per category for the question 'Do you have a good understanding of the impact of these multiple pressure situations?'

2.3 Understanding of the consequences of multiple pressures on end users and delivery of ecosystem services

There was a more or less even split between respondents on their understanding of the impacts of multiple pressures on the delivery of ecosystem goods and services to end users (Figure 13). Commonly mentioned end users and related ecosystem services are drinking water supply, irrigation for agriculture, recreation, navigation, tourism and fisheries.

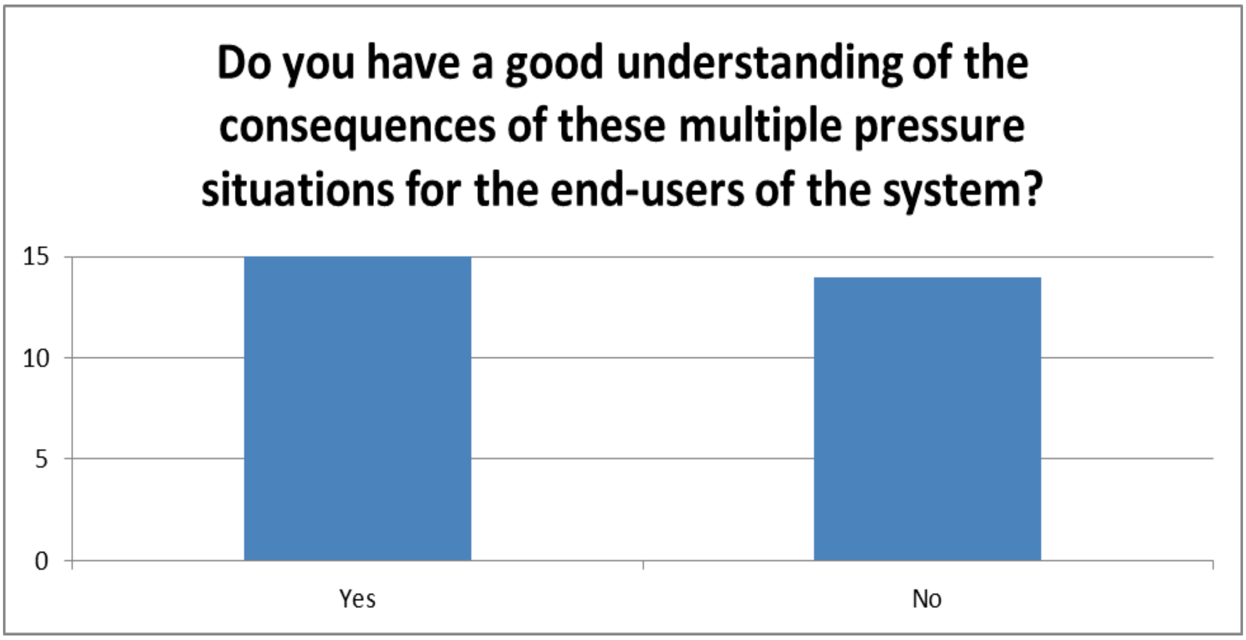

Figure 13 Overview of responses per category for the question 'Do you have a good understanding of the consequences of these multiple pressure situations for end-users of the system?'

3.1 Taking long-term developments into account

Long-term changes, such as even further increasing effects of climate change or societal changes affecting land use (e.g. 30 - 50 year horizon) are mainly taken into account for some specific measures that are costly (e.g. upgrading wastewater treatment plants or large scale re-designing of river systems for river restoration or flood protection). However, the day-to-day challenges in water management might make the long term effects, especially from climate change, less important (Figure 14).

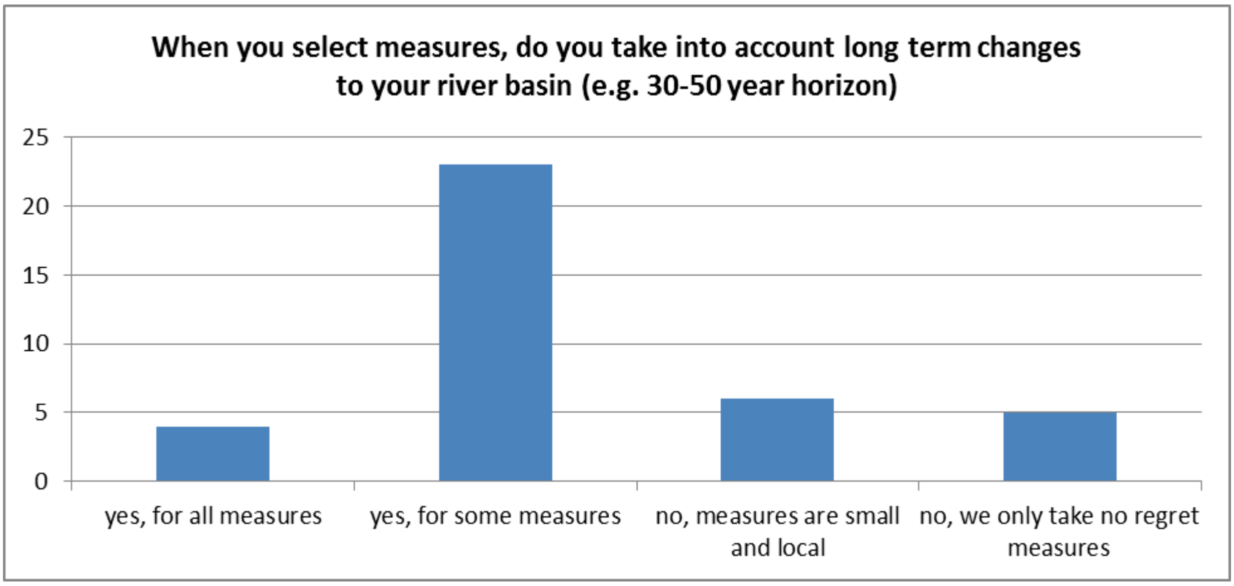

Figure 14 Overview of responses per category for the question 'When you select measures, do you take into account long term changes to your river basin (e.g. 30-50 year horizon)

3.2 Use of MARS scenarios for more insight in long-term changes

Most respondents feel that their current way of working would change if they had a better understanding of the response of their river basin or water body over a longer period of time (Figure 15). The MARS scenarios can be a first generic step that would help with this understanding. However, these scenarios are broad and based on general knowledge and still need to be translated to local scale (e.g. of a specific single water body) to become really useful for water managers having to implement this in their own water systems.

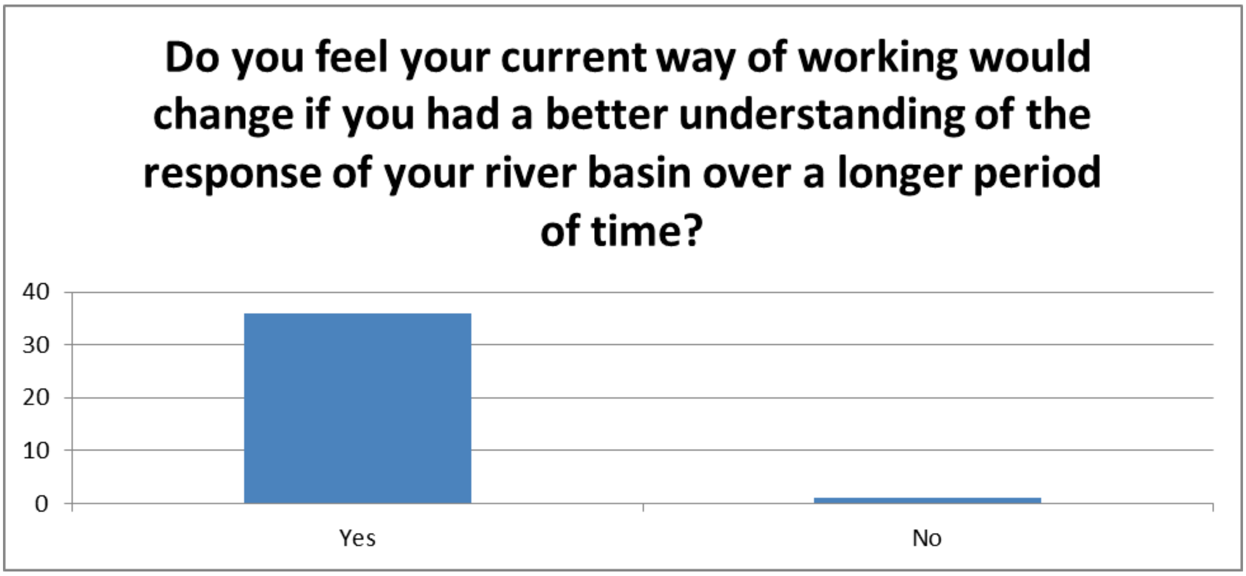

Figure 15 Overview of responses per category for the question 'Do you feel your current way of working would change if you had a better understanding of the response in your river basin over a longer period of time.'

4. Potential usability of the MARS Diagnostic tool

The MARS Diagnostic Tool (see paragraph 1.3) is perceived as a useful additional tool for the diagnosis of the response of the aquatic ecosystem to multiple pressures (Figure 16). Many respondents see it as a beneficial supplement tool in line with the already used tools and methodologies. Some wonder if the tool will work on the small and fine resolution of a single specific water body and state that there is a need for a decision support system under multiple pressures that uses easy to understand standards and understandings with simple guidelines and in the national language of the practitioners. (Note: At the time of the questionnaire the final version of the MARS diagnostic tool was not yet available, and some people found it difficult to give a clear answer to this question as a result of that situation).

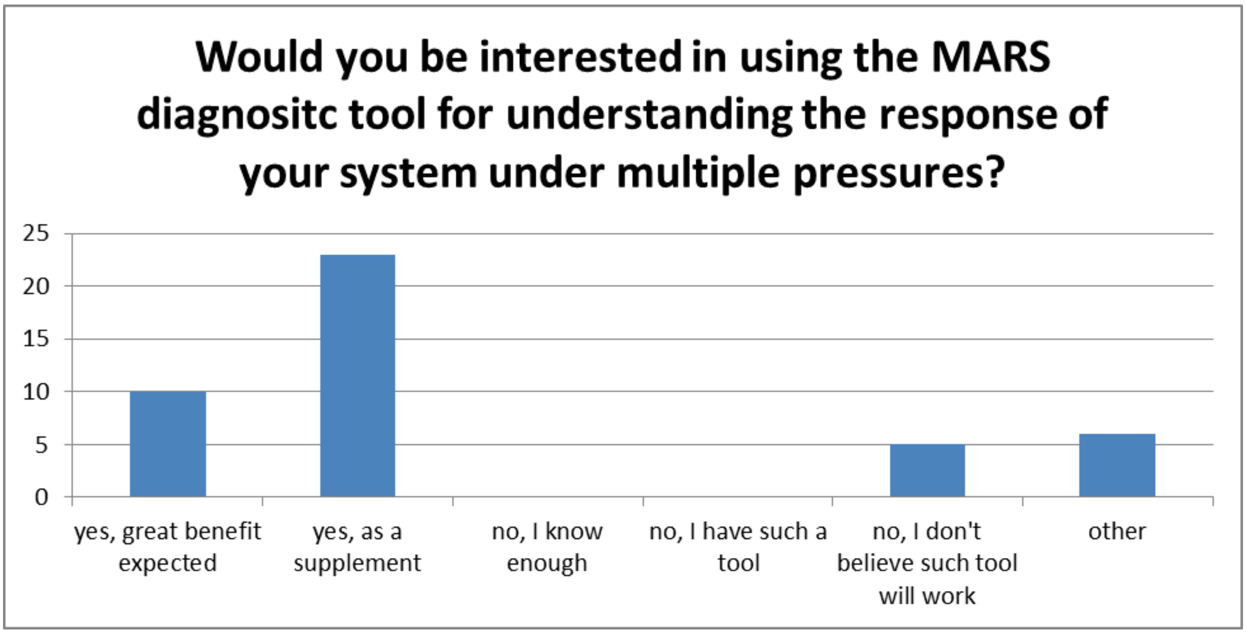

Figure 16 Overview of responses per category for the question 'Would you be interested in using the MARS diagnostic tool for understanding the response of your system under multiple pressures?'

3.2 Results of the stakeholder workshop discussion

At the end of October 2016, a combined MARS workshop of WP 6.4 and WP7 was organised in Den Helder, The Netherlands. The aim of the workshop was to discuss the first applications of the MARS WP7 tools to assess multiple pressures in river basins and design measures to mitigate the impacts of multiple pressures. In total 9 stakeholders and applied partners from The Netherlands, Austria, Germany, Portugal, Romania and the UK attended the workshop (Appendix 3).

The workshop started with an introduction round of stakeholders and MARS scientists. The results of the questionnaire were presented to the stakeholders and used as a starting point for the discussion. The invited stakeholders were asked beforehand to fill in the questionnaire to be familiar with the questions. In general the stakeholders agreed with the general results of the questionnaire. In the first part of the meeting the following key questions were addressed:

- How do water managers deal with the selection of cost-effective measures, for water bodies exposed to multiple pressures?
- Is knowledge on pressure interactions and biological response taken into account when selecting and prioritizing the measures?
- How can MARS best contribute to a potential gap in knowledge and tools from the perspective of the stakeholders?

MARS scientists and invited stakeholders discussed the challenges and bottlenecks of river basin management planning from a water managers' point of view. It was pointed out that in the process of setting up river basin management plans there has been a change in perception from solely focusing on water body to catchments and a "sea to source" view; e.g. for cost-effectiveness estimates and impact of measures, however mostly based on expert knowledge. Some stakeholders stated that they are well aware of pressure interactions in their river basins. However this is still not addressed when it comes to implementation of measures. The stakeholders shared their experiences and addressed challenges when it comes to prioritization and evaluation of the effectiveness of measures.

Many water managers make use of a catalogue of measures at a national level and prioritize measures. Measures are selected based on cost-effectiveness and societal impact. However, water managers are still in the need of identifying costs-effectiveness of such measures. Currently, this is mostly done individually by expert judgement. Some stakeholders stated that prioritization of measures is mainly based on limited land availability.

It is pointed out that the time lag before the status of a water body improves is part of the challenges to evaluate the effectiveness of measures. Monitoring, preferably for a longer period of time, is needed to understand and verify improvement and the proper investment of funding in the right measures. However, many stakeholders stated that the funding is not sufficient to implement targeted monitoring to show the success of measures. Decisions are still often made based on commonly accepted measures for short-term success. As many measures are expected to only be proven successful in the long term, it will be useful to have commonly accepted tools indicating the effect of measures with a sound scientific basis. Pressure specific indicators and both short and long term response indicators are lacking.

The stakeholders stressed that the MARS tools (see paragraph 1.3) would also be helpful in the discussion with the public to increase public awareness and visualize the likelihood of changes essential for public activities such as fishing, sailing, swimming etc.

In smaller groups stakeholders and tool developers discussed the MARS tools focussing on the following questions:

- Are the tools meeting the river basin manager's needs and requirements for diagnosing and assessing the impacts of multiple pressures and designing relevant management measures?
- How to adapt the MARS tools and applications to meet the needs and requirements of river basin managers?

Freshwater Information System

The Freshwater Information System will provide information on interaction of pressures and ecosystem responses, management options for the mitigation of multiple pressures, a selection of models to be used in river basin management, selected case studies and guidelines to support the 3rd cycle of RBMP. The stakeholders had the opportunity to make use of the online prototype (Figure 17) and were asked for feedback. Several stakeholders stressed that they have a broad spectrum of work and are in need of compiled and concise information about specific topics. The Freshwater Information System should provide short summaries of the main MARS results with access to related deliverables and scientific articles in case more in depth reading is needed. Furthermore, clear visualizations with brief descriptions are preferred. Stakeholders are looking for information that is generally applied and scientifically proven. This can be very useful for writing proposals addressing commonly applied measures described in a concise way with references included. The Freshwater Information System will be accessible on the Freshwater Information Platform (FIP; http://www.freshwaterplatform.eu/) of which many stakeholders who hadn't been to earlier MARS workshops were not aware. Stakeholders agree that such a platform is a good medium to compile freshwater related information. However, promotion of the existence of this platform and Freshwater Information System is needed. Stakeholders favour the presentation of case studies but they quickly need to get an overview the most important aspects and do not want to spend too much time searching for relevant information. Stakeholders were very interested in a selection of models for RBMP based on specific pressures and a link to river basins where models are applied.

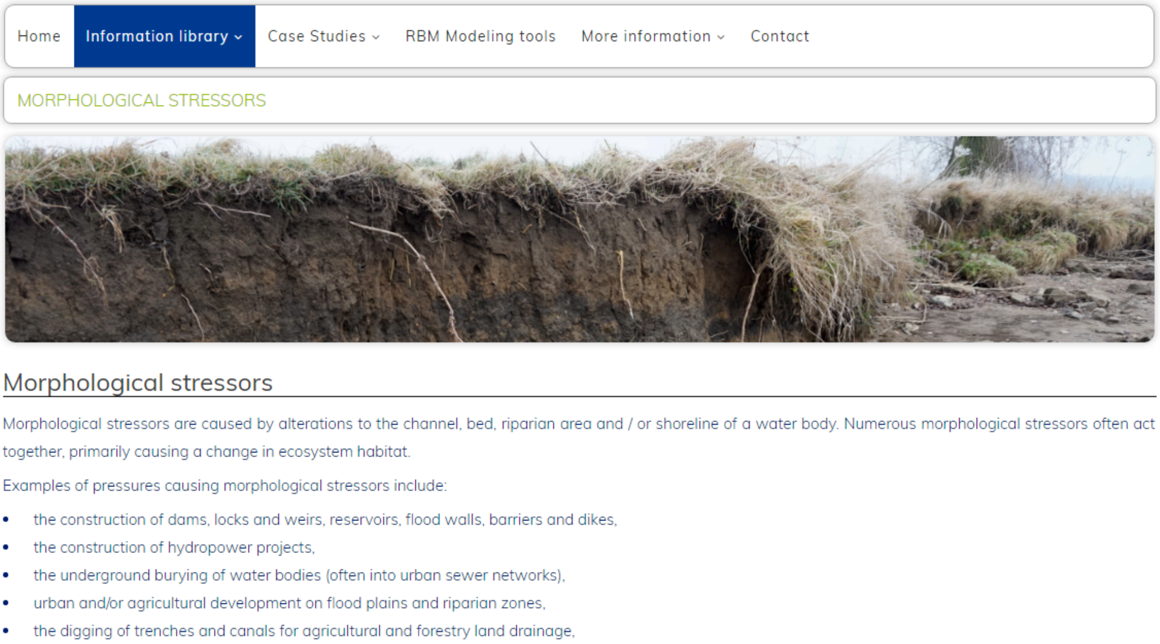

Figure 17 Prototype of the Freshwater Information System with a description of morphological stressors

Diagnostic tool

During the workshop an online demonstration of the Phytoplankton Diagnostic Tool of the river Elbe was given (Figure 18). The stakeholders agreed that this online demonstration helped to get a better understanding of the aim and the function of the tools. The Diagnostic Tool aims at detecting problems within a given water body and upstream and tries to identify potential causes of deterioration. Adaptive measures, showcases and a manual to implement a diagnostic tool tailored for the specific user's requirements will be provided. Stakeholders stressed that more explanation of variables is needed and that the time scale and units of the indicator variables clearly have to be defined. Stakeholders and tool developer agreed that more testing is needed.

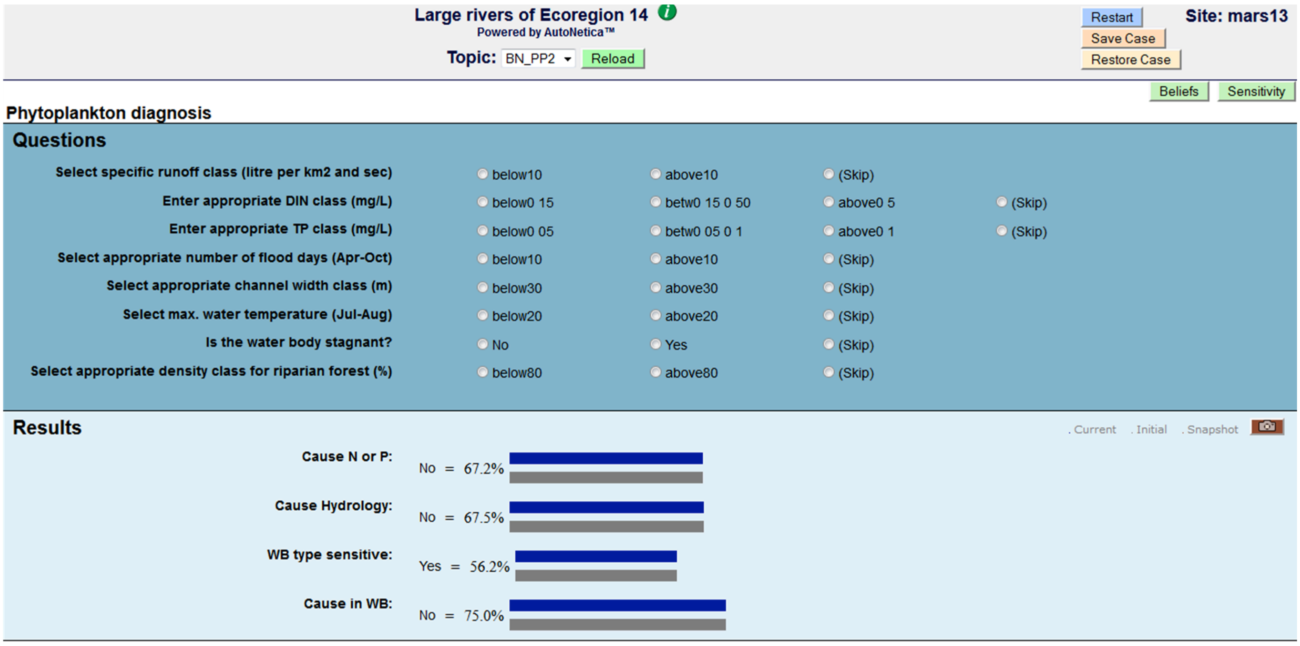

Figure 18 First prototype of the phytoplankton diagnostic tool for the river Elbe

Scenario Analysis Tool

In general the stakeholders agreed that this tool would be useful to get an idea of general future trends (Figure 19). When asked if stakeholders would use the tool for planning purposes they stated that reliable results on water bodies are preferred. However, it might be interesting to compare catchments in Europe with similar characteristics and similar targets. Stakeholders questioned the quality of results from EU data and pointed out that this tool seems to be more relevant for pressures from up-stream areas focussing on nutrients and is less relevant for (hydro-morphological) pressures on a local level. The scale addressed is of relevance for the stakeholders and especially the transboundary issues are of special interest. The Scenario Analysis Tool would be more valuable if stakeholders can run their own scenarios and their own data. Stakeholders and tool developer agreed that more testing is needed.

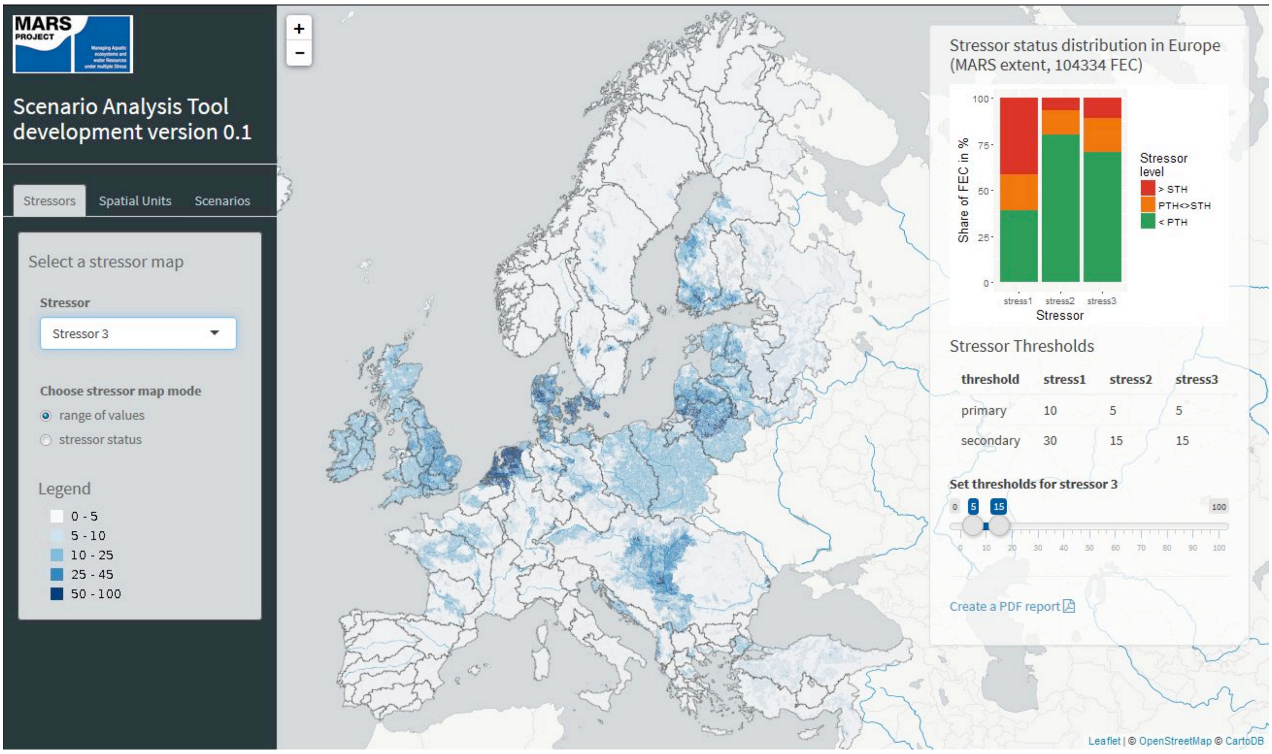

Figure 19 Development outline of the Scenario Analysis Tool as presented at the workshop, content and presentation will be further developed

Bayesian Neural Networks (BNN)

In general it is assumed that the most dominant pressures and ecological status are known. The assessment is based very much on expert judgment but prognosis is important for acceptance of measures by public and policy makers. Stakeholders stated that the need of a tool depends on the 'maturity' of the case:

- some countries do already have modelling tools in use;
- some countries have decided on measures to be applied and therefore do not need an additional new tool;

- some countries rely on monitoring;
- some countries need tools to quantify the effect of input on ecology;
- some countries indicate that a BBN tool can be especially useful for visualisation, communication and risk assessment (and not for choosing the appropriate measures);

Stakeholders agreed that a Bayesian Belief Network is potentially useful tool for prognosis and the implementation of appropriate measures. They support the user to consider several factors and execute a sensitivity analysis of the system. According to the stakeholders Bayesian Belief Networks offer opportunities to be used as a communication tool between water managers and policy makers or public to combine data and expert judgment for visualization and discussion. The uncertainties of the results need to be better explained, because there is a fear that if these are too large the usability of the tool becomes limited.

During the workshop it was concluded that stakeholders and MARS scientists will be further engaged in testing the currently developing tools and discuss their applicability in water management. A blog article about the MARS workshop in Den Helder is available at the "Freshwater Blog":
(https://freshwaterblog.net/2016/11/03/tools-for-managing-multiple-pressures-workshop-collaborations-between-mars-and-ecostat/)

4.3 Evaluation of the MARS conceptual model

During the Den Helder meeting October 2016, a discussion was held within the MARS consortium on the usefulness of the MARS conceptual model (Hering et al 2014) for the MARS case studies and the river basin managers. Being mostly familiar with the DPSIR framework of the MARS model and to harmonize the modelling strategy, MARS case studies used the DPSIR framework as their starting point to set up a first conceptual model (Figure 20). The river basin teams mainly focused on collecting data, getting all models up and running.

Key elements in the modelling of the interactions between and impacts of multiple stressors have been identification of:

- The relevant basin-specific stressors;
- Appropriate indicators of system status and environmental impact; and,
- Key ecosystem services to be included in the modelling.

The modelling strategy followed by each basin varied slightly and was a result of the needs raised by each basin conceptual model. Fourteen different process-based models were employed, overall, with Swat and PERSiST being the most widespread, in terms of use among the several basins (Ferreira et al, 2016). The empirical modelling framework was more similar between basins and closely followed the deliverable "Cookbook on data analysis" (Feld et al, 2016).

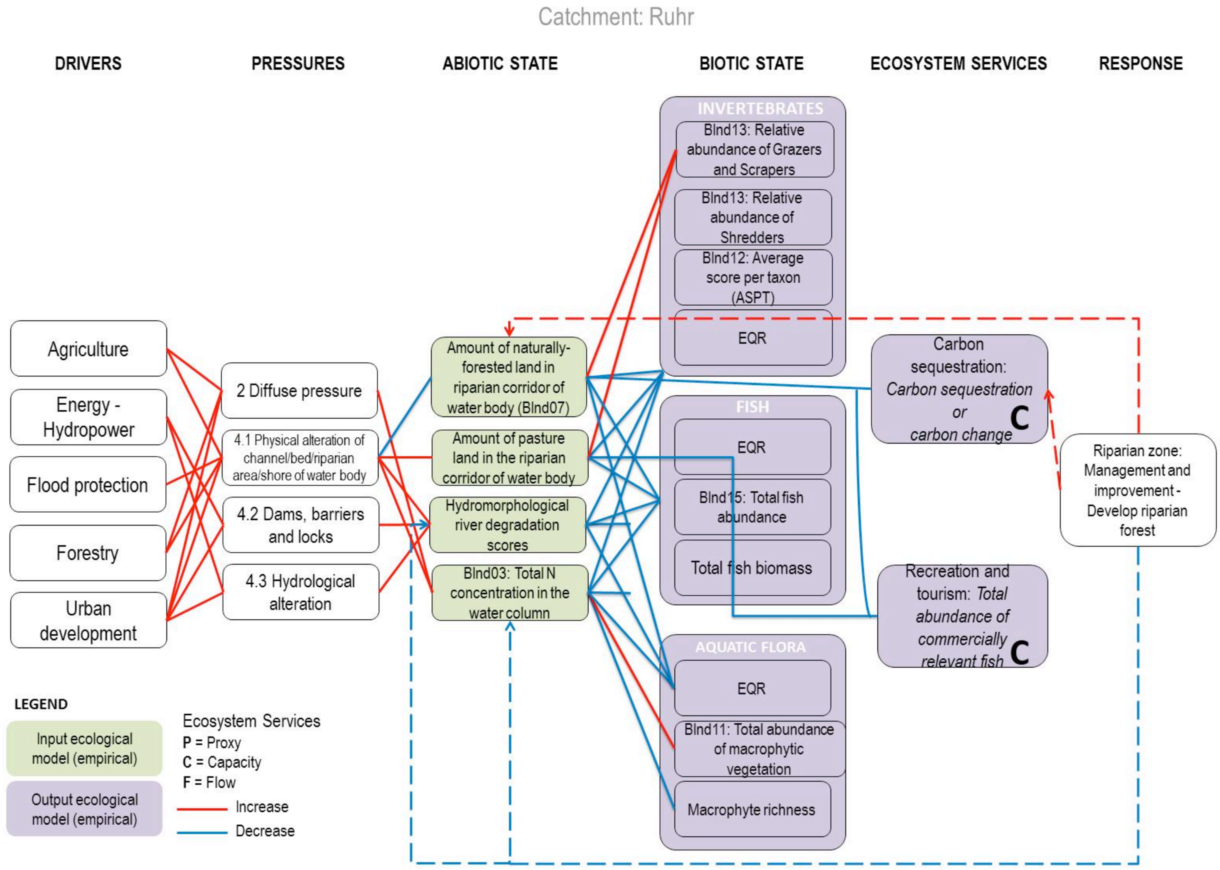

Figure 20 DPSIR framework and ecosystem service capacities of the Ruhr basin (Bloomfield et al, 2016)

The MARS conceptual model, explicitly the DPSIR framework, has been specifically useful within the MARS-consortium to create a common language for discussing pressures and status and service response. However, the quality and completeness of the conceptual model varied among the case studies, mostly depending on the data availability and number of investigated multiple stressor combinations.

The risk assessment part of the model is not well known and many participants of the workshop had no experience with this risk based thinking, nor did they apply this in daily practise. Also, the synthesis report of the MARS case studies reveals that only for the river basin Ruhr up the entire MARS conceptual model was filled in (Figure 21). Also, the ecosystem service cascade has not been often implemented in the modelling workflow applied by the case studies and was only expressed as 'Impact' of the DPSIR framework. Six case studies did not apply specific modelling to investigate multi-stressor effects on ecosystem serviced (Ferreira et al, 2016). Furthermore, not all relationships included in the DPSIR framework are data-driven and the question remains what the evidence is based on in case of process-based models (e.g. for parameter settings in process-based models).

For the two measures 'green infrastructure' and 'environmental flows' it was specifically checked if the conceptual model would offer a way to evaluate the effectiveness of these measures on the aquatic ecosystem.

Based on the responses to the questionnaire both green infrastructures and environmental flows are frequently applied as measures throughout Europe (28 and 25 times respectively).

For green infrastructures there was a subdivision in different types of green infrastructures based on their size being 'narrow buffer strips (<4m wide)', 'restoration of riparian zones (~4-20m wide)' and 'wider flood plain areas', depending on the type of water body and space available for implementing such measures. The size of the responses per sub-category was not always indicated by respondents and as a result the responses per subcategory are too small and uncertain to make a good evaluation on whether a given size is more common than another size. **Fehler! Verweisquelle konnte nicht gefunden werden.** gives a good example on how the conceptual model was used to relate the existing pressures to the conclusion that the enhancement of a riparian buffer zone might be a good measure for the case study of the Ruhr as described within the conceptual model. In the MARS case studies management and improvements of the riparian zone, are carried out in 6 cases, of which 4 in Central Europe, 1 in Northern Europe and 1 in Southern Europe. Improvement of lateral connectivity and wetland improvement in floodplains was carried out in 5 northern and 7 central cases (Kuijper et al in prep.). Environmental flows are also frequently applied measures, and in the case studies 8 southern, 7 central and 2 northern cases indicated this measure to be of interest. Unfortunately, little information is available on whether these two measures were selected to serve multiple goals, for example the reduction of flood risk and the improvement of the ecological state.

Summarizing, the full MARS conceptual model itself seems to be rather complex which limits the applicability by stakeholders for daily river basin management. The DPSIR-part of the model appeared to be the most useful part of the framework to be used in river basin management planning.

The MARS consortium, however, concludes that synergies between the EU Water Framework Directive and the EU Flood Directive could result in win-win measures which 'cover it all', such as measures to increase ecosystem services, support EU FD and improve the ecological status of the water body, e.g. by Green Infrastructure. The MARS conceptual model would be useful to combine different aspects of both directives.

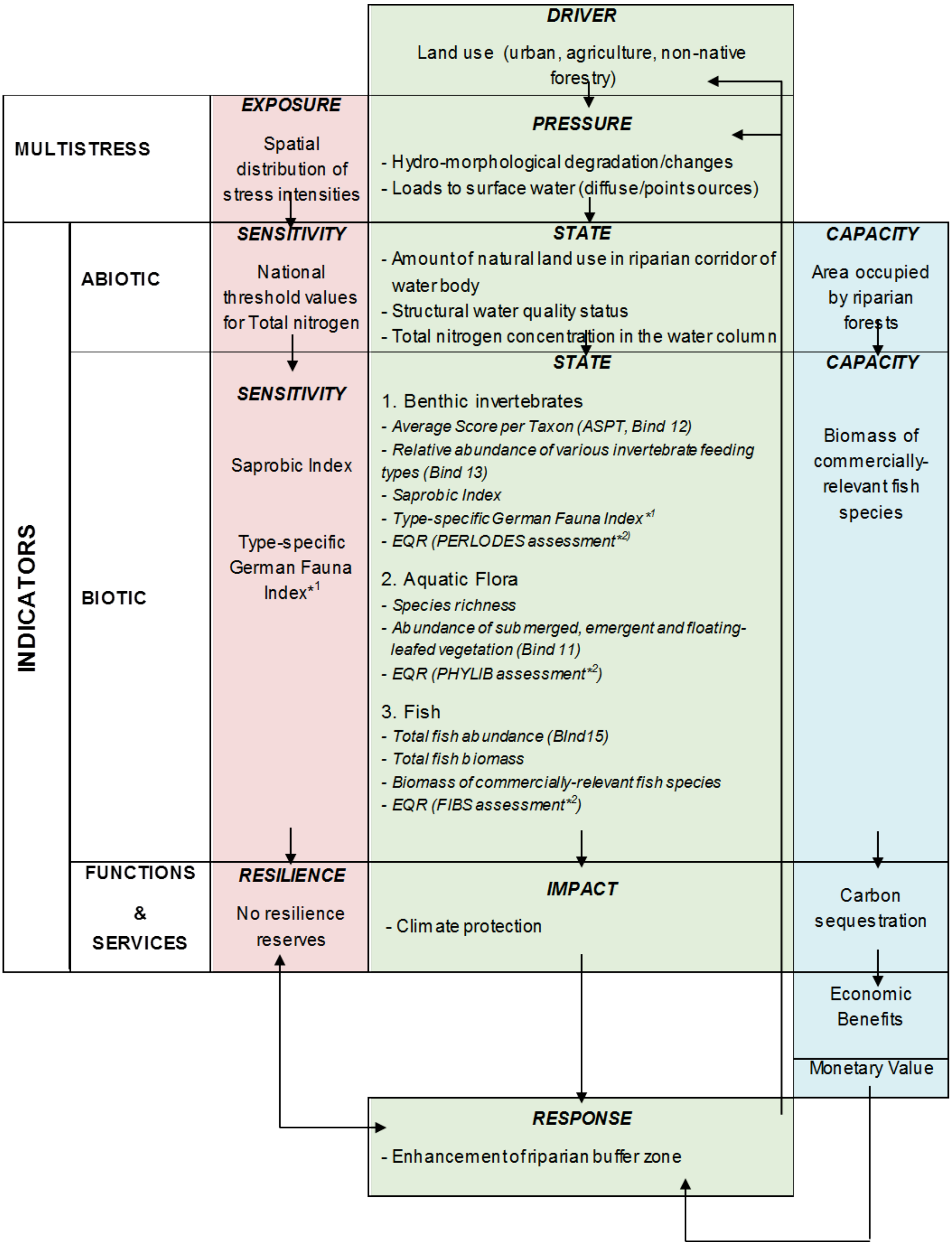

*Figure 21 MARS conceptual model for the Ruhr basin. The three columns represent the risk assessment framework (left column green), the DPSIR framework (middle column, red) and the ecosystem service cascade (right column, blue) (Bloomfield et al, 2016). (*1 German Fauna Index is a multimetric Index for stream assessment, which is mainly focussed on the impact of hydromorphological degradation on the macroinvertebrate fauna. *2 PERLODES, PHYLIB and FIBS are the official WFD compliant assessment systems for rivers and streams in Germany).*

4. Discussion

4.1 Daily river basin management practice

Although daily river basin management practices vary throughout Europe, some general conclusions can be drawn from the MARS questionnaire and discussions with water managers during the workshop held in October 2016. For example, in many countries there is a difference between the authorities designing the overall river basin management plans and the entities responsible for implementing measures in the field. For overall RBMP design, some countries (e.g. Slovakia, Romania) and the international river basin district committees mention they use EU provided tools and guidelines, while other member states have a distinct 'own' approach, which might even differ between different river basin districts within a country or between federal states (e.g. the Netherlands, Finland, UK and Germany) (Schinegger et al 2016). For example, in the Netherlands there is a significant difference between approaches in the planning of measure between different regional water management authorities, with some using extensive quantification of system understanding and potential impact of measures, while others rely heavily on expert judgement. This is also the case in other countries, such as Germany, where the approaches differ per federal state, in the UK where approaches and systems differ between devolved administrations (e.g. Scotland vs. England & Wales), and in Finland, where there are large differences per region.

The analysis of 'technical' underlying data (including the use of modelling tools) is often not conducted by those designing the RBMPs or selecting the measures. As this is a different group of people than the ones filling in the questionnaire, the response to the questions regarding tools might be misleading, suggesting that these tools are not frequently used. However, the limited replies from the current group of respondents on use of tools show that there might be a gap between those who use models, and those who would be able to make use of the results of these models to define measures. We would suggest that further development of operational mechanisms to help translate results from models and tools into actions and measures in the field should be considered across Member States. This could help the selection of measures being less reliant on expert judgement and, ultimately, should encourage more cost-effective measures.

Many respondents regret not being able to evaluate measures better and more specifically, as well as not having the time or resources to monitor every water body individually. As a result, there is a tendency to first take no regret measures, often based on expert judgement, that do not require much proof of their effectiveness locally. Many measures are taken at the scale of water bodies rather than taking a full catchment approach. This hampers a good understanding of the upstream and downstream effects of such measures, especially for measures focusing on connectivity such as fish migration and environmental flows, as well as the need for reduction of nutrient pollution upstream to protect downstream water bodies (including coastal waters).

Similarly, fundamental processes such as changes in sedimentation balances, at full catchment scale and the impact of changes therein on the ecosystem are neglected due to the constraints in time and spatial scale considered for taking measures.

4.2 Knowledge on multiple stressors, long-term effects of climate changes and ways to assess their impact

At present, many water managers undertake measures without/or a clear awareness of the potential interaction between stressors, and the long-term effects of changing societies, land use and climate change. Often it is assumed that climate change will have synergistic effects, whereas the literature (Noges et al. 2015) and the MARS synthesis (D6.1-1 – Chapman et al, this report) indicate that many stressor interactions are dominated by one stressor (often nutrients) or are antagonistic. At the same time, many respondents indicate they would appreciate more information and insight into how to include long- term effects of climate change and pressure interaction in their prediction tools.

Interestingly, some of the multiple stressor combinations appear to be focused on national practices rather than based on a geographic or climatic region. For example, almost all respondents from the Netherlands indicate they experience the stressor combination ‘extreme temperature and nutrient stress’, while neighboring countries such as Germany and Belgium do not indicate this stressor combination as a problem. This is most likely due to personal biases in the respondents from that country, as stressor combinations are unlikely to differ greatly between neighbouring countries (EEA, 2012).

At the same time, some countries where we expected a given pressure combination to be important did not list this pressure combination. For example, in Austria there is a strong focus on the impact of morphological changes, while nutrient stress is not perceived as problematic, while surrounding countries do list nutrient pollution as a problem.

Some respondents indicate that it is unclear if measures will work out ‘differently’ under climate change (compared to no climate change), see also e.g. Pletterbauer et al (2015), and as a result, they do not take climate change into account in the selection of measures. It was also stated that it is very difficult fully unravel certain effects of climate change, especially those affecting average (water) temperature. For example, in Norway a study showed changes in competition between Arctic char and brown trout as a result of these temperature effects to be strongly interlinked with changes in lake productivity gradients (Finstad et al, 2011). Also, the impact climate change might have via changes in the likelihood of invasive species to successfully establish and thus alter the biotic community is not easy to solve by taking ‘standard’ measures (Rahel and Olden, 2008).

One important aspect of visualizing the knowledge on multiple stressors and assessing their impact is to have a good overview of the spatial up- and downstream interdependency between pressures and the changes therein over time. An example on how this might be done in a schematic way is given in **Fehler! Verweisquelle konnte nicht gefunden werden.**, in which a river basin is divided into 7 water bodies along an upstream – downstream gradient, using the 'ecological key factors' approach (STOWA 2014). Two upstream stretches, one suffering from diffuse pollution (pressure1), and one clean stream, merge into a main flow. This main stretch is blocked by a dam (pressure 4) and as a result suffers from physical (pressure 2) and hydrological alterations (pressure 3). Two measures are available: fish passage (red) or dam removal (blue). The checks in the different water bodies show how many water bodies will benefit from the given measure for the given pressure: by installing a fish passage all upstream water bodies will reduce the impact from P4, but P3 remains. By removing the dam the water bodies directly up- and downstream of the dam will also benefit the reduction of P3. Assessing the impact of multiple stressors and potential impact of different sets of measures can be helped by providing such overviews.

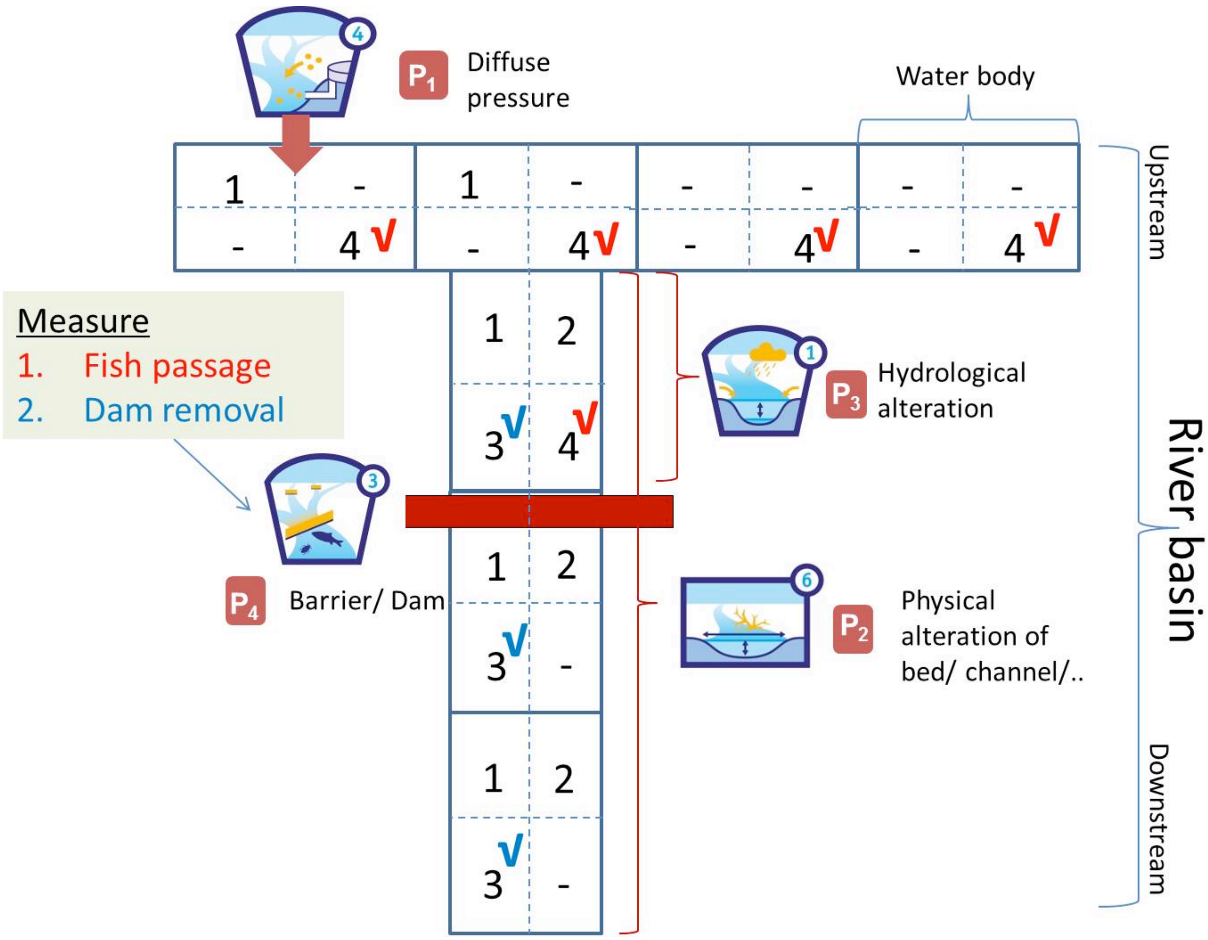

Figure 22 A schematic representation of a river basin which compromises 7 water bodies. Each water body is impacted by one or more pressures. Impacts of pressures are on the spot i.e. in the water body or distant (upstream or downstream). Restoration or mitigation measures address all or subset of the pressures. The improvement of ecological status depends not only on the implemented measures, but also on the impact of the remaining pressures. P1 – P4 = pressures. √ indicates the water bodies benefitting from implemented measures (red = fish passage; blue = dam removal.)

4.3 Use of tools and the MARS conceptual model

The most commonly used tools for selecting measures are tools that support data-analysis and visualisation of relevant data using GIS platforms. In contrast, analytical process-based tools, such as numerical ecological- or hydrological models are almost never used by the respondents. The interpretation of monitoring data to derive measures is mainly done with expert knowledge, and not with (semi)automatic tools. Nevertheless, for analysis of the impact of pressures on abiotic and biotic states, specifically for unravelling the combined effects of multiple pressures and for predictions of future states, these numerical models may be required. Most respondents indicate that they are interested in using some additional tools to give more insight into multiple pressure effects within their system. However, some also indicate that they fear such general tools will not be able to represent the specific situation within their catchments or water bodies, or that there is not enough input data to feed such a model. This recognition - that individual water bodies are to some extent unique in their response to multiple pressures and data required for decision-making are often limited - may be the reason that expert judgement is often used to decide measures. Evidence from the MARS synthesis (D6.1-1) that some generalised response of water bodies may be predictable given a water-bodies specific typology and information on stressor gradients, may help water managers taking expert judgements based on this greater understanding in the future.

During discussions on the MARS conceptual model at the Den Helder workshop, it was concluded that the model has proven specifically useful within the MARS-consortium to create a common language for discussing pressures and status and service response, but that it is not a practical tool for daily water management. Especially the part dealing with risk assessment was unfamiliar to many ecologists participating in the discussions on the evaluation of the model. Although many tools are available to quantify responses to restoration measures, especially abiotic processes and responses (e.g. nutrient loads, flow), the link with ecological status and services is often not made. The ecosystem service aspect of the conceptual model is indirectly strongly interlinked in daily management practises, as many respondents indicate that stakeholder involvement and opinions are highly valued in the prioritization of measures. The MARS-conceptual model makes this interaction more explicitly visible and can help in the dialogue with these stakeholders.

The intention of the MARS conceptual model initially was to bring water quality and water quantity practitioners together and to create a common terminology to better harmonize different water management efforts. Currently, however, we notice that the link between measures selected for flood risk management and how these might affect ecological status and water quality is still weak, or little specified. Many member states do take green infrastructure or environmental flow measures, but from our results it is not clear how/if these are linked with ‘standard’ flood- and drought management.

The awareness for long-term changes is generally lacking in daily water management, a fact that needs to be improved. Better knowledge on statistical year-to-year variation in relation to climate change is needed, also at regional level water authorities, as they are often responsible for the implementation of measures. Because climate change might well cause the ecosystem to exceed a threshold or tipping point of the ecosystem (e.g. Lyche Solheim et al 2009), data analysis of the current situation might not be able to provide clear insight on when exactly such a tipping point will be reached. The MARS Scenario analysis tools can help raising the needed awareness about this issue.

In order to better deal with multiple stressors, a diagnostic tool is being designed within the MARS project. This includes the use of Bayesian network techniques to assess the impact of multiple stressors. This approach appears useful for analysing monitoring data. Other tools that use similar techniques (including e.g. knowledge rules derived by neural network analysis) have already been successfully applied in the WFD Explorer, which is a Dutch tool that allows water managers to get a quantified estimate of the response of all BQEs on measures taken in a catchment (Harezlak and Meijers, 2014). It is a good example to show that step-by-step, such tools are being transferred from the academic realm into daily water management practise.

4.4. Gaps in tools and indicators - future steps and recommendations, incl. general remarks on the questionnaire and working approach

The questionnaire was sent to water managers with a role in the implementation of the WFD. Most water managers are ecologists or environmental scientists by education and need to work in an interdisciplinary team to conduct their work. Thus, we can't estimate how the results of the questionnaire are affected by disciplines (e.g. whether it was someone with an ecology or hydrology background who had filled in the questionnaire). A water manager has many different objectives to pursue, of which the WFD objective is only one. More important for many water authorities is the management of water quantity, especially floods and droughts, as these pose direct risks to the general public and stakeholders using the water for e.g. agriculture, drinking water, energy production, cooling water, navigation and recreation. Often, taking balanced decisions therefore is not only considering multiple stressors but also on water uses and risks imposed by water bodies. However, combining these different objectives can produce fruitful synergies, e.g. benefits of natural water retention measures also for the ecological status when planning a flood risk reduction measure. Some natural flood management measures, such as re-meandering rivers to slow flow, and green infrastructure projects often have a dual purpose of reducing flood risk and restoring rivers to achieve the good ecological status (e.g. The Eddleston Water Project in Scotland http://www.tweedforum.org/projects/current-projects/eddleston). Also, in Belgium, the Sigmaplan (http://www.sigmaplan.be/) was specifically designed to work towards a safe, natural, and economically viable and attractive Scheldt region, in which multiple objectives are merged in one single integral plan for improvements.

Water managers tend to use comparatively simple approaches in deriving measures and in the interpretation of monitoring data, while scientists prefer the more sophisticated ways. Closing this gap requires close communication. Using tools requires an initial investment in time and training, as no single tool will work 'automatically' as a plug and play device. Yet, tool-developers should also keep user friendliness at the forefront of their mind while developing such tools, because many water managers indicate they are interested in trying, but then fail due the technical complications of learning to use such a tool. As such, the perceived 'gaps in tools' might also be 'the feeling of lack of urgency to use such a tool', or 'a perception that such a tool will definitely not work for a single basin'. Without a feeling of urgency to better quantify the system processes at a catchment scale, a good understanding of the timing and choice of measures will not be achieved.

At the same time, some examples of improvement in ways to link stressors, state and measures can also help to prioritize measures. For example, in the Netherlands the Ecological Key Factors concepts and tools (STOWA, 2014) are used both for communication and prioritization of measures in relation to stressors, taking a water system analysis approach as a basic foundation for the resulting actions. The catchment-wide approach that is advocated in the Ecological Key Factors also helps to take into account long term changes (both societal and climatic) and to provide a better link between local measures and overall river basin management plan strategies for the full catchment.

Tools to connect people

Tools help to give insight in the functioning of an aquatic ecosystem and can help in searching for the best ways to manage such systems, especially in cases of problems arising from multiple stress situations. In some MARS case studies (Thames, Odense, Regge & Dinkel), tools have been used to quantify both, water quantity and water quality responses, but this is not a EU-wide common practise yet. The WP7.3 Bayesian modelling tool will add to bridging this gap between abiotic models and ecological responses, and thus the dialog between different stakeholders. Also, the MARS conceptual model has created a common language that has helped with the comparative analysis of different case studies by applying the 'DPSIR framework'. Because of a general lack of data and knowledge on the integration of ecosystem services, it was not possible to further quantify this part of the model. Yet, we see from the reasons for prioritization of measures, that ecosystem services are often an important issue in water management as they represent the interests of different stakeholder groups. However, the common understanding/definition of (aquatic) ecosystem services and the integration of this concept into WFD implementation and water management issues is unclear.

In some European regions, in daily water management, there is still a separation between disciplines, such as between catchment modellers, hydrologists, ecologists and socioeconomists, which hampers an overall integrated understanding and management of the full system. Some examples of initiatives to bring together these people in larger integrated plans exist (see paragraph 4.1) and there are opportunities to use tools to help in this process. For example, the advanced modelling tools that are used in flood forecasting such as the 'Flood Early Warning Systems 'FEWS', can be linked to water quality related questions for forecasting of water quality related issues (e.g. risk of harmful algal blooms).

We therefore propose a set of simple steps to help water managers making decisions:

1. Develop a conceptual model of your water body/catchment of concern: This can be a simple schematic drawing in which you indicate the most prominent pressures and problems (water quantity hazards, pressures and status and services affected). The MARS conceptual model can be a useful staring point (e.g *Figure 21)*

2. Assess which tools are currently available, both within your organisation and using the MARS Model Selection Tool for different sections of the conceptual model (note: Discuss this matter also interdisciplinary, e.g. with hydrologists, morphologists, economists etc.) and also discuss with your neighbouring water management authorities. They might have tools that are relevant to your system.

3. Assess if relevant tools can be adjusted to benefit both water quantity and water quality and ecological objectives. Make sure that during such discussions there are different disciplines at the table: let hydrologists, morphologists and ecologists get together to define one commonly shared technical approach, but also check with the social scientists and economist on the likely acceptance and cost-effectiveness of the options: how do stakeholders respond to the suggested approach and what are the financial consequences.

4. See if the MARS multi-stressor toolbox (chapter 1.3) can give you additional tools to link abiotic and ecological goals and assessments if the currently existing tools are not sufficiently satisfying.

Finally, we question whether water managers around Europe feel the urgency to use relatively complex analysis tools. The current situation appears already challenging for most water managers, so learning the application of new tools might be too hard, which implies that the gap between tool design and daily water management remains. Also, social-economic aspects are important here: as it seems, there is currently no strong need to provide more complex tools for ecological assessment, compared to the specific ones already used to assess flood risks and related social damage, as the societal impact of 'ecological failure' is not well noticed by the general public and especially by politicians (maybe with the exception of sever algal blooms in lakes used for drinking water and recreation). However, if tools can also be promoted to raise awareness about the current and potential future state of the aquatic ecosystem, this might be very beneficial. Moreover, by being able to communicate well the impact of measures through quantified visualisations will also help water managers justify their daily choices.

5. Acknowledgements

We are grateful to all respondents of the questionnaire and their contributions in the various discussions that were held during the MARS project on this topic.

References

Birk S, Bloomfield J, Carvalho L, Feld C, Ferreira T, Globevnik L, Hering D, Kuijper M, Schmutz S, Schinegger R, Lyche-Solheim A, 2015. Terminology. MARS project.

European Environmental Agency, 2012. European waters – assessment of status and pressures. EEA report 8/2012. Copenhagen. ISSN 1725-9177

Feld CK, Birk S, Eme D, Gerisch M, Hering D, Kernan M,... & Poikane, S. (2016). Disentangling the effects of land use and geo-climatic factors on diversity in European freshwater ecosystems. Ecological Indicators, 60, 71-83.

Feld CK, Segurado P, Gutiérrez-Cánovas C. 2016. Analysing the impact of multiple stressors in aquatic biomonitoring data: A 'cookbook' with applications in R. Science of The Total Environment. http://dx.doi.org/10.1016/j.scitotenv.2016.06.243

Ferreira T, Panagopoulos Y, Bloomfield J, Couture RM, Ormerod S. 2016. D4.1 Case study synthesis - Final Report.

Finstad AG, Forseth T, Jonsson B, Bellier E, Hesthagen T, Jensen AJ, ... & Foldvik A. (2011). Competitive exclusion along climate gradients: energy efficiency influences the distribution of two salmonid fishes. Global Change Biology, 17(4), 1703-1711.

Grizzetti B, Lanzanova D, Liquete C, Reynaud A. (2015) Cook-book for ecosystem service assessment and valuation. JRC Science and Policy Report EUR 27141 EN. Luxembourg Publication Office of the European Union. 136 pp.

Harezlak V, Meijers E 2014. WFD Explorer. An interactive water quality tool. Deltares report 1000006-002. Delft The Netherlands.

Hering D, Carvalho L, Argillier C, Beklioglu M, Borja A, Cardoso AC, Duel H, Ferreira T, Globevnik L, Hanganu J, Hellsten S, Jeppesen E, Kodeš V, Lyche Solheim A, Nõges T, Ormerod S, Panagopoulos Y, Schmutz S, Venohr M, Birk S, (2015) Managing aquatic ecosystems and water resources under multiple stress - an introduction to the MARS project, Science of The Total Environment 503-504, 10-21, http://dx.doi.org/10.1016/j.scitotenv.2014.06.106

Hendriks, P., Schollema, PP,, Pot, R., Ottens, H.J., Querner, E., Verdonschot, R. 2016. Ruimte voor natuur bij onderhoud aan watergangen (in Dutch: room for nature during maintenance of water ways). H2O-online 15-febr. 2016. 11 p.

Kuijper M, Birk S, Bloomfield J., Andersen HE, Gieswein A, Schinegger R, Mischke U, Hutchins M, Laize C, Ascott M, Prudhomme C, Rankinen K, Couture R, Cremona F, Gutierrez-Canovas T, Bucak T, Beklioglu M, Hanganu J, Chifflet M, Stefanidis K, Segurado P. (in prep.) MARS conceptual models – Coordination for case studies.

Moe J (ed.), Alahuhta J, Aroviita J , Beklioğlu M, Bucak T, Carvalho L,…. Et al., 2017. MARS Deliverable 5.1-4: Reports on stressor classification and effects at the European scale: Effects of multiple stressors on ecosystem structure and services of phytoplankton and macrophytes in European lakes.

Lyche-Solheim A, Rekolainen S, Moe SJ, Carvalho L, Philips G, Ptacnik R, Penning WE, Toth LG, O'Tool C, Schartau AKL, Hesthagen T (2008) Ecological threshold responses in European lakes and their applicability for the Water Framework Directive (WFD) implementation: synthesis of lakes results from the REBECCA project. Aquat. Ecol. 42: 317-334. DOI 10.1007/s10452-008-9188-5

Nõges P, Argillier C, Borja A, Garmendia JM, Hanganu J, Kodeš V, Pletterbauer F, Sagouis A, Birk S 2016. Quantified biotic and abiotic responses to multiple stress in freshwater, marine and ground waters, Science of The Total Environment, 540:43-52 http://dx.doi.org/10.1016/j.scitotenv.2015.06.045.

Pletterbauer F, Melcher AH, Ferreira T, & Schmutz S. (2015). Impact of climate change on the structure of fish assemblages in European rivers. Hydrobiologia, 744(1), 235-254.

Rahel FJ, Olden JD (2008) Assessing the effect of climate change on aquatic invasive species. Conservation Biology 22(3): 521-533

Schinegger R. Schülting, L., Schmutz S, Lyche-Solheim A, Hering D, Feld C, Birk S. 2016. Workshop on multiple pressures in River Basin Management: Evidence, diagnosis and management at water body and catchment scales. Summary report of a workshop organized under the MARS project april 7th and 8th Vienna, Austria.

STOWA 2014. Ecologische Sleutelfactoren. Stowa-report number 2014-19, Amersfoort, the Netherlands. http://www.stowa.nl/publicaties/publicaties/ecologische_sleutelfactoren__begrip_van_het_watersysteem_als_basis_voor_beslissingen

APPENDIX 1 Questionnaire 'Multiple Pressures in River basin management '

Name:
Organisation:
Email:

Introduction to the questionnaire:

A large proportion of European water bodies are exposed to a range of different pressures, such as nutrient pollution, hydromorphological alterations and climate change. The cumulative impact of these multiple pressures on the overall ecological status of the water bodies is often not well understood. For water bodies in moderate, poor or bad ecological status measures are needed to improve the ecological status and to achieve good status, which is the objective of the Water Framework Directive. But how are these measures selected, if there are multiple pressures influencing the ecological status at the same time? And how can scientific results, such as those from the MARS project support this selection of measures?

This questionnaire is being sent to river basin managers at the local level who are responsible for single river basins within larger river basin districts throughout Europe. The purpose of the questionnaire is to prepare for a MARS stakeholder workshop in October 2016, where the relevance and usefulness of MARS tools will be discussed and adapted to fit the needs of the local river basin managers in their work towards the 3rd cycle of river basin management plans. To enable the MARS scientists to streamline the tools towards the managers' needs we need to know how daily water management practice deals with the selection of cost-effective measures for water bodies exposed to multiple pressures, and whether knowledge on pressure interactions and biological response are taken into account when selecting and prioritizing the measures.

The questionnaire is structured into two sets of questions: the first set of questions deals with the current methods for water management and the choice of measures, while the second set of questions concerns tool development within the MARS project.

With all questions there is a multiple choice answer to help EU-wide comparison of the answers, while at the same time leaving additional space for comments of further clarifications of your answer.

The time required to fill in the answers is estimated to approximately 30 minutes.

We thank you very much for your cooperation,
On behalf of the MARS consortium,

Marijn Kuijper, Clara Chrzanowski and Ellis Penning
Deltares

PART 1 – General questions with regards to selection of measures

1. Identification of target group for MARS tools:
   a) Who in your river basin/river basin district/country was responsible for selecting the measures included in the Programme of Measures for the 2nd RBMP?
   Name of the organization/function: ________________________________
   Comment:
   ____

   b) Who in your catchment/country are responsible for implementing the measures?

   Name of the organization/function: ________________________________
   Comment:________________________________

2. How do you select the measures? (multiple answers are possible)
   - Based on detailed analysis and understanding of the river basin and its water bodies, using modelling to predict the effect of the measures
   - Based on detailed analysis and understanding of the current situation and the distance to the good ecological status target, using monitoring data analysis
   - Based on expert judgement of the pressures and status of the water bodies in the river basin and how to achieve good status
   - Based on experiences from comparable river basins/water bodies (national/international)
   - Based on a predefined list of potential measures
   - Based on a dialogue with relevant sectors responsible for the pressures
   - Other (please describe)
     ____

   Comments:
   ____

3. If multiple pressures are relevant for your river basin, what is the basis for prioritization of the measures? (multiple answers are possible)
   - Question not relevant, there is a clear single pressure causing the degraded ecological status
   - We use detailed quantitative analyses to show the most important pressures affecting the ecological status, e.g. pressure accounts and ecological responses to the combined pressures
   - We rely on previous experiences in comparable situations where some measures have shown to be more successful than others
   - Our choice is often driven by the societal impact of a measure (stakeholders wishes/ sector interests /political priorities are an important driver of our choices)
   - Other:
     ____
     ____

4. How are economic aspects taken into account in the prioritization of measures?
   - We only take no-regret measures
   - Short-term economic aspects for implementation of measures plays an important role
   - Long-term (multi-year) economic aspects for implementation of measures plays an important role
   - Not so much, but we do prioritize which water bodies should be improved first and which ones we leave for later

- o We prioritize the most cost-effective measures needed to achieve good ecological status for each water body
- o Other: ____

5. Do you set evaluation criteria for the effectiveness of the measures you choose?
   - o Yes, each measure is always presented in the programme of measures together with an evaluation text and a relevant monitoring effort to test if the measure is effective, i.e. fulfills the initially expected outcome
   - o Yes, we evaluate the effect of the measures via the regular monitoring programme in the area according to WFD standards
   - o No, the effect of the measures are not specifically evaluated for effectiveness after implementation

   Comment: ____

6. What is the most common type of measure (combination) implemented in your catchment or region? At which spatial scale are most measures implemented: Individual water bodies or river basin wide? (multiple answers are possible)

Measures	**Scale (please specify)**
o Fish passages to improve connectivity	o Water bodies o River basin o Combination of both
o Nutrient reduction of point source pollution (e.g. improved collection and treatment of waste water)	o Water bodies o River basin o Combination of both
o Nutrient reduction of diffuse source pollution via agricultural measures (e.g. reduced fertilization, reduced autumn tillage)	o Water bodies o River basin o Combination of both
Establishment of green infrastructure areas for multiple benefits: flood mitigation, natural water retention areas, pollution reduction, habitat improvements to enhance biodiversity via riparian zone restoration and management and buffer strips: o Narrow buffer strips (<4m) o Restoration of riparian zones (~ 4-20m) o Wider floodplain areas	o Water bodies o River basin o Combination of both
o Flow adaptation to meet environmental flow requirements	o Water bodies o River basin o Combination of both
o Groundwater abstraction measures being:________________________________	o Water bodies o River basin o Combination of both
o Water level adjustments to meet more natural fluctuation of water levels	o Water bodies o River basin Combination of both
o Morphological adjustments of river	o Water bodies

banks/shore lines of lakes and transitional waters	○ River basin ○ Combination of both
○ Other measure:	○ Water bodies ○ River basin ○ Combination of both
○ Other measure:	○ Water bodies ○ River basin ○ Combination of both
○	○

7. Which type of tools do you use to derive your River basin management Plans?
   - None
   - Our own local or regional databases for the river basin or river basin district. Specify: ____
   - National databases for and/or transboundary data (for transboundary river basin districts). Specify: ____
   - National designed tools. Specify: ____
   - Our own data processing models. Specify: ____
   - Commercially/open source software for data analysis. Specify: ____
   - GIS-platforms. Specify: ____
   - Other ____

8. Do you feel you have a sufficient understanding of the problems in your river basin to select the right measures?
   - Yes, I have sufficient overview of the pressures, status and functioning of my river basin, which is strongly based on analysis of numerical model outcomes.
   - Yes, I have sufficient overview of the pressures and status of water bodies in my river basin, based on analysis of long term, well managed data sets.
   - Yes, I have sufficient overview of the pressures and status of water bodies in my river basin, mainly based on expert judgment (based on own field experience)
   - No, I sometimes struggle to understand the origin of the problems and their combined impacts on the ecological status of the water bodies in the river basin, due to a lack of data or general knowledge.
   - Other: ____
     ____

PART 2 – questions about the MARS project
The Mars Project aims to develop methods and tools to aid the assessment of the impact of multiple pressures on aquatic ecosystems. If there are multiple pressures causing the deterioration, the total impact might be obscured by interactions between these pressures. Understanding these interactions and their combined impact might be necessary in order to identify the most effective measures.

Topic 1 – Commonly occurring pressure combinations
The commonly occurring multiple pressure combinations identified from the EU Member States reporting of the first RBMPs, are combinations of nutrient stress, hydromophological changes, and climate change. In the MARS project these have been further specified into four common pressure combinations and linked to a most likely set of consequences for end users. These consequences are given in parenthesis below:

1. Extreme temperature and nutrient stress, causing algal blooms and oxygen depletion. Consequences for drinking water supply and recreation (bathing & fishing)
2. Extreme low flow (droughts) and nutrient stress (mainly relevant in rivers and groundwater). Consequences for drinking water supply, recreation (fishing) and irrigation/agriculture
3. Extreme high flow (floods) and nutrient stress. Consequences for drinking water supply (sewage overflow), the population (security) and agriculture (flooding of fields)
4. Changes in morphology (e.g. barriers) & nutrient stress (mainly relevant in rivers). Consequences for fishing

1.1. Which of the four identified multiple pressure combinations are relevant for your system? (multiple answers are possible)

- o Extreme temperature and nutrient stress
- o Extreme low flow and nutrient stress
- o Extreme high flow and nutrient stress
- o Changes in morphology an nutrient stress
- o Only one pressure is relevant in my system. Specify:______________________________________
- o Other – my system has a different set of multiple pressures being: ____
- o Other – my system does not suffer from nutrient stress. Specify:____

1.2. Do you have a good understanding of the impact of these multiple pressures situations?

- Yes, I know that the multiple pressures occurring in my river basin cause an even stronger negative response of the aquatic ecosystem than the expected effect of each of the single pressures. Specify which pressure combinations: ____
- Yes, I know that the multiple pressures occurring in my river basin cause a less negative response in the aquatic ecosystem than the expected effect of each of the single pressures. Specify which pressure combinations: ____
- No, I do not fully understand if the multiple pressures occurring in my river basin enhance each other or reduce each other's effect

1.3. Do you have a good understanding of the consequences of these multiple pressures situations for the end-users of the system?

- Yes, I know that these pressures not only impact the aquatic ecosystem, but also have the following negative consequences for users: ____
- No, I don't know how the multiple pressures affect users in a different way than the single pressures

Topic 2 – Effect of long-term changes in society on ecosystem response
Within MARS the combination of pressures is assessed not only for the current situation, but also in combination with three storylines (scenarios) for climate and societal change in the coming 50 years. These storylines are used to model the expected change in response to pressures over a longer time frame.

2.1. When you select measures, do you take into account long term changes to your river basin, such as even further increasing effects of climate change or societal change affecting land use (e.g. 30-50 year horizon)
- Yes, all the measures are so costly/have so much impact that we need to take these longer term changes into account
- Yes, some of the measures are so costly/have so much impact that we need to take these longer-term changes into account, but not for all the measures. The measures for which long-term changes are considered are: ____
- No, the measures are mainly local and small or reversible; therefore we do not take long term changes into account.
- No, the effects of climate change are so uncertain that we only take no-regret measures that are already relevant on the short term

2.2. Do you feel your current way of working would change if you had a better understanding of the response of your river basin over such a longer period of time?

- Yes, I would appreciate more insight in these long-term effects
- No, I do not need to have these insights now; managing the current situation is already challenging enough

Topic 3 – Diagnostic tools

Within the MARS project we try to provide a better understanding of the role of multiple pressures on aquatic systems. One of the outputs is a diagnostic tool that is currently under development. It will provide a quantified system (based on your own data and expert judgment applying Bayesian network analysis) for better diagnosis of the underlying pressures that can interplay within a river basin (or water body?). The tool will be available via the MARS information platform

Would you be interested in using such a tool for diagnosis of the response of your system to multiple pressures?

- Yes, I would greatly benefit from such a tool as I currently have no good information on multiple pressures at all
- Yes, I would benefit from such a tool in combination with my own good understanding of the river basin and its water bodies
- No, I already know my river basin well enough
- No, I already have such a tool available. Specify: ____
- No, I do not believe the response of my river basin can be sufficiently represented by such a diagnostic tool

APPENDIX 2 – Detailed responses in the category 'other' per question

This appendix gives the detailed responses as written down by individual respondents under the category 'other' as specifications of their multiple choice answers.

Part 1

1. Identification of target group for MARS tools:

2. How do you select the measures?

Detailed answered under the category 'others':

- Workshop approach with local water management unit officers
- Also „Handlungsanleitung Aufstellen der Maßnahmenprogramme" http://www.lfu.bayern.de/wasser/wrrl/bewirtschaftungsplaene_1621/hintergrunddokumente/index.htm
- The basic measures under EU legislation have been took into consideration as complying obligations (i.e UWWTD, Nitrate Directive, IED, SEVESO, etc.) and research studies for supplementary measures
- The knowledge about pressures and amount of monitoring data varies a lot between different waterbodies
- It is a mix of water system analysis (not all information is sufficiently available, models, expert judgement and social and political interests.
- The lack of good water system analysis makes it difficult to make well grounded selection of measures
- Measures were designed with regard of analysis on the compliance of individual EU directives (Art. 11 paragraph. 3) WFD).
- Method of selection varies from pressure to pressure
- The 2nd Danube River basin management Plan (Update 2015) includes measures of basin-wide importance oriented towards the agreed visions and management objectives for 2021. It is based on the national programmes of measures elaborated by Danube countries, which shall be made operational by December 2018, and describes the expected improvements in water status by 2021.
- Different methods for different pressures and sectors. Our measures are structured in a library with predefined estimations of costs. The effects were modelled on pilot areas where we investigated the cost effectiveness and feasibility of the measure type linked to a certain pressure. Not all measures were available in this model why those were planned using best available knowledge and expert judgement.
- We select the measures based on a mix of knowledge based on monitoring data, detailed modeling of the catchment including nutrient and pollutant loading via mostly storm water. The aim of our Local programmes of measure is to give us a local perspective of what measures that are needed to meet the objectives. Our goal is to achieve a "library" of potential measures that together meets and exceeds the objectives given by the River Basin Authorities. This gives us a larger degree of freedom when choosing what measures to be carried out and when as long as the objectives are met. The measures include information about who is responsible, when the measure should be carried out, the cost of the measure and the effect. The actual measures to be carried out are selected from their cost efficiency and their synergetic effects.
- Measures based on state river development plan (Gewässerentwicklungskonzept des Landes)
- We choose measures that get approval from land owners, support from local community and can be funded adequately from various sources
- First we predefined a list of potential measures. A cost-effectiveness analysis (with the tool MKM MaatregelKostenModule) defined the more relevant measures. Budget is a limiting factor. Therefore, we selected some areas where we take first action to improve the ecological status. Afterwards, we screened with our local experts at water body level the most relevant measures. At this level, the choice is driven by more practical and societal aspects (total cost, timing of different actions, the responsible organization who must initiate the measure)

3. If multiple pressures are relevant for your river basin, what is the basis for prioritization of the measures?

Detailed answers under 'other':

- If nutrient pressures are dominant and hymo pressures are present, hymo-measures can be postponed/prioritized backward
- Also important is the feasibility of measures. For the water board, it lies within its reach to take measures at sewage treatment plants: so we took measures at the sewage treatment plants

- Also important is the estimation of efficacy of measures by national advisory organs (e.g. PBL planbureau voor de leefomgeving). PBL stated that measures pertaining to the physical restoration of brooks and rivers would have the biggest impact on ecology, more so than measures aimed at water quality for instance. So we have restored, and will restore our rivers and brooks
- Our choice is often driven by the possibilities of implementing the measure through available fields
- Water system analyses give an impression of the pressures (but this is not always quantified)
- Partly based on expert-judgement, no regret measures and cooperation with other parties, very little is based on research and analyses of existing data.
- Setting of priorities was based on an analysis of fulfilment of Slovakian obligations to the EU, arising from the requirements of various EU Directives, in particular Directive 91/271/EEC, Directive 91/676/EEC and Directive 2010/75/EU and with regard to technical feasibility of the measures within the required time period
- An "Ecological prioritisation approach for river and habitat continuity restoration in the Danube River Basin" was used. In order to enable a sound estimation of where to target measures most effectively at the basin-wide scale, an ecological prioritisation of measures to restore river and habitat continuity in the DRBD was carried out for the 1st DRBM Plan and updated for the 2nd DRBMP Plan. The elaborated approach provided indications on the step-wise and efficient implementation of restoration measures at the basin-wide scale. It provided useful information on the estimated effects of national measures in relation to their ecological effectiveness at the basin-wide scale and served as a supportive tool for a number of countries in the implementation of measures. Therefore, it also supports feedback from international to national level and vice versa. The ecological prioritisation approach for continuity restoration is addressing all reported river continuity interruptions in the DRBD.
- There is the need of prioritization of pressures before selecting measures
- Within the RBM for the federal state North-Rhine Westphalia there exists no prioritization; all measures in the list have to be realized
- We lack a system where we can weigh the importance/effects of multiple measures on multiple pressures. However we have a quite clear understanding on which pressures are causing most harm and what measures are feasible to implement. These two do not always meet of course. We hope for a tool to use for real planning and prioritization but it needs to be transparent and based on stakeholder acceptance.
- A expert analyses of the cost/effectiveness/feasibility of the measures applied to each water body
- Try to find measures that have impact on many pressures simultaneously

4. How are economic aspects taken into account in the prioritization of measures?

Detailed answers under 'other':

- We prioritize the most cost-effective measures needed to achieve good ecological status at the level of subbasin scale
- The overall budget for WFD measures is also taken into account
- We do not take economic aspects into account in the prioritization of individual measures. We are only restricted to a total amount of (limited) resources
- It is important that the measures do not lead to disproportionally high costs. The measures should not result in an extreme increase in water-taxes and are therefore also partially chosen based on political decisions (*Eds*: water management related taxes are billed separately in the Netherlands and differ per regional water management authority)
- First predominantly no-regret measures and measures that can be carried out on land that we own ourselves such as natural shorelines and fish passages. Only after that we combine with other measures and policies.
- Funds availability (EU + national budget + own resources of the subject which is implementing the specific measure).
- It is a mixture of what is cost beneficial and what is affordable over the shorter term
- In the second WFD management cycle, the cost-effectiveness analysis (CEA) was an issue addressed at national level, but not on basin-wide scale. No basin-wide CEA was performed for the DRBM Plan – Update 2015. However, the planning period until 2021 could be used to "pave the way" for a possible use of CEA in the third management cycle, when supplementary measures will gain importance for reaching WFD objectives for certain SWMIs (such as nutrient pollution).
- Within the RBM for the federal state North-Rhine Westphalia there exists no prioritization; all measures in the list have to be realized

- We strive to identify and present the most cost effective measure combinations for catchments but the final decision of which measures are implemented is in the hands on the implementing stakeholder and political decisions.

5. Do you set evaluation criteria for the effectiveness of the measures you choose?

Detailed responses for 'other'

- For some specific measures (like fish passages) we do test the facility (fish friendly pumping station, fish ladder
- We often face situations where existing monitoring data are insufficient to evaluate the effect of measures (limited monitoring due to financial reasons)
- Some measures are evaluated individually on effectiveness
- We do not monitor measures separately, only the state of a water body is monitored (the sum of multiple measures in a single water body). Occasionally individual measures are monitored, but this is very infrequent.
- We occasionally monitor individual measures and most often this is not done in a 'scientific' way, without a proper analysis of the results that include also aspects such as cost-effectiveness and life cycle costs, including the costs for adjusted maintenance (e.g. natural shores require a different management regime than traditional shores).
- Data collection for the evaluation of the effectiveness of measures (hydro-morphology, waste water treatment, management in agriculture and management of sediment) is carried out under the operational monitoring of water that is part of the monitoring plan in accordance with Article 8 of the WFD. However the effect of the measures has not yet been evaluated specifically.
- If multi measures are present then our monitoring may not be able to distinguish the individual impact of each measure on a water body.
- Expert judgement, the only evaluation criteria are the overall quality ratio
- The 2nd Danube River basin management Plan (Update 2015) includes measures of basin-wide importance oriented towards the agreed visions and management objectives for 2021. It is based on the national programmes of measures elaborated by Danube countries, which shall be made operational by December 2018, and describes the expected improvements in water status by 2021.
- We conduct also modelling to evaluate the effectiveness of measures
- Until now the effectiveness of measures has not really been monitored but it is foreseen to include a monitoring "part" in the implementation of certain measures in order to evaluate their effectiveness after implementation
- Neither of the options above. We evaluate the total effect of measures at the end of the planning period when the planning for the third period starts. The evaluation is based on the ecological classification and we lack a tool to investigate and evaluate the single measures implemented (as we do not always know the exact implementation rate and exact geographical target of measures for example for forestry and agriculture). We evaluate the success of measures by looking at i) the ecological response on large scale ii) the stakeholder willingness to implement the measure. The latter is of great importance and a detailed analysis of the effects of a single measure should not overrun a stakeholder's decision or response to the suggestions in the PoM.
- For each measure a set of indicators are selected in order to assess its degree of implementation
- Some of the measures are however evaluated specifically. For example some of our storm water dams are targets for specific monitoring
- Monitoring only partly covers the implementation of measures. This is mainly due to cost coverage.
- We evaluate the effects of measures when we have the resources for it, but not always
- Mainly for point source pollution. It is difficult to evaluate effect of the measures for diffuse sources or morphology because of the small amount of input data

6. What is the most common type of measure (combination) implemented in your catchment or region? At which spatial scale are most measures implemented: Individual water bodies or river basin wide?

Detailed answers under the category 'others' often imply that other measures were also implemented.

- Improvement of sediment transport/bed load regime
- Organisational legal and educational measures
- Reduction of micropollutants
- More ecological friendly maintenance

- Dredging of organic material in canals and ditches
- Ecological mowing regimes of water ways: http://hhnk.acceptatie.webgispublisher.nl/Viewer.aspx?map=Overbreedte-in-beeld
- Re-connection of oxbow with the flow, restoration of flow in the arms;
- Revitalization of the original riverbed.
- Forestry measures
- Measures connected to acid sulphate soils
- Improvement of waste water treatment
- Multiple plants to remove phosphorus from inlet water
- Regulation of fishery management, implementation of good practice to reduce nutrient load from fish pounds and regulation of fishing in natural water bodies
- Removing nutrient rich bottom deposits
- Morphological adjustments of the river bed
- Treatment of polluted storm water in end of pipe-dams as well as measures in the catchment (green roofs and other local storm water adaption)
- Flood protection measures
- Appropriate controls regarding abstraction of fresh surface water and groundwater and impoundment of fresh surface waters (including a register or registers of water abstractions) have to be put in place as well as the requirements for prior authorization of such abstraction and impoundment. In line with the WFD, it must be ensured that the available groundwater resource is not exceeded by the long-term annual average rate of abstraction
- For green infrastructure the main focus comes from flood risk management plans

7. Which type of tools do you use to derive your River basin management Plans?

Specific replies for 'Local or regional databases':

- MySQL+PHP Oracle
- Regional water management authorities databases, environmental inspection databases
- GeoWeb - Sobek – MIPWA/MODFLOW – GIS-ratio - … etcetera
- WasserBLIcK provided by the German Federal Institute of Hydrology (BfG)
- Data spreadsheets and GIS in combination with local knowledge/expert review
- Special database for data entries, data research, and visualization for the state of Bavaria ("Gewässeratlas")
- There are local and basin thematic databases in which primary data are kept and maintained. The available database in electronic format consists in Office (Excel, Word, and PowerPoint), scanned documents, e-mail storage. For the WISE reporting process databases at sub-units level have been developed according to WISE Access database structure which contains thematic data and information required in a specified/pre-defined format
- Water quality database Rijkswaterstaat and our own database
- Water quality monitoring databases maintained by LEGMC; database on waste water pollution load maintained by LEGMC
- Our own water quantity- and water quality models
- For whole country - Environmental information system http://register.keskkonnainfo.ee/envreg/main#HTTPuXIofziLMZmzcNYJVZgrbolhq11q9Y
- OKIR, MAHAB , Biological databases of Hungarian Academy of Sciences, Centre of Ecological Research
- Local data for the city of Stockholm http://dataportalen.stockholm.se/dataportalen/ http://miljobarometern.stockholm.se/
- Flemish (VMM) Datawarehouse with monitoring results about surface water, emissions, groundwater, hydromorphology
- Monitoring programme

Specific answers for 'National databases or transboundary data bases' used:

- Data bases of Slovenian Environment Agency, ICPDR and ISRBC
- National Access Database (WISE); ICPDR DANUBEGIS
- Water quality database Rijkswaterstaat and our own database
- National forest database maintained by State Forest Service; exchange of water quality monitoring data with neighbouring countries – Lithuania and Belarus; IPPC modelling results on pollution air transfer

- Emissieregistratie, stone
- Catchment planning system
- Transnational Monitoring Network (TNMN), national water information systems, basin-wide inventories for emission sources, hydromorphological alterations and accident risk hot-spots.
- In the SAVA RBMP only official data from the countries were used
- For whole country - Environmental information system http://register.keskkonnainfo.ee/envreg/main#HTTPuXIofziLMZmzcNYJVZgrbolhq11q9Y
- National databases coming from the WFD monitoring
- Water quality an hydrological monitoring network (http://snirh.pt/), - Permits data base (https://siliamb.apambiente.pt/login.jsp, - Large dams data base (http://www.apambiente.pt/index.php?ref=77&subref=839)
- http://viss.lansstyrelsen.se/
- NATIONAL NETWORK MONITORING
- Within the National Monitoring programme we collect all the data, but additionally there are included also the data from bilateral/cross-boundary common surveys (samplings) and harmonised evaluation and assessment of status - all in the Framework of each Transboundary Commission legislation - (Slovakia - Hungary/Austria/Poland/Ukraine/Czech Republic
- Protected landscape areas

Specific answers for 'National designed tools'

- GROWA-SI
- Modelling of nutrient fluxes (Moneris/MoRE) and N-entry into groundwater (GWNBW)
- Moneris
- VannNett, Vannmiljø – national databases
- WFD explorer
- Ecological Key Factors (STOWA)
- Software for data handling – evaluation of water status
- Pressure specific models, classification tools
- EELIS system, designed for officials – data and GIS tools
- LuxMaPro[1], internally developed access databases which cover specific needs
- ELWAS WEB (www.elwasims.nrw.de)
- National RBM-Tool": SQL-Server Database including all relevant data for development of RBMPs (waterbody delineation, typology, pressures, monitoring results, risk assessment, status assessment, HMWB-designation, set and planned measures). The tool allows a (partly) automated risk- and status assessment using pressures- and monitoring data.
- National tools developed by the administration or research projects
- Ecological Key Factors (STOWA), PC lake PC ditch
- MONERIS, River water quality model
- Pattern for river basin management plans

Specific answers for 'Own data processing models'

- WAQ (WAter Quality) applies yearly nutrients load balance equation taking into account point, not-point sources and natural background loads for each scenario
- Hydrological models within IBRAHYM
- Modelling tools ESTMODEL
- GLM, Regression Tree
- Stormtac
- MKM MaatregelKostenModule, Pegase model (agriculture)

Specific answers for 'Commercially/open source software for data analysis'

- PC-Lake, PC-Ditch
- Swedish Mass Balance model for diffuse pollution load estimation
- Statistical methods to evaluate trends, non-parametric tests (Mann - Kendall test)
- Nutrient emission modelling tool MONERIS
- Software for the management of monitoring data (WISKI, LIMS)
- STATISTICA, Matlab, RStudio, Canoco

- Microsoft Excel (Access, Excel)

Specific answers for 'GIS platforms'
- Mainly ArcGIS (*Eds.:*often listed)
- http://geoportal.kzgw.gov.pl/gptkzgw/
- DanubeGIS - Danube countries cooperate and provide information for basin-wide scale on DanubeGIS
- All data from the Sava RBM Plan are available in Sava GIS geoportal (http://savagis.org)
- National basic GIS tools provided by our administration, used with creativity and experience
- http://sniamb.apambiente.pt/Home/Default.htm
- Internal GIS-platforms via the City of Stockholm
- GIS were used in development of typology, WBs designation and categorisation, ES assessment, ROM selection\

8. Do you feel you have a sufficient understanding of the problems in your river basin to select the right measures?

Detailed responses in 'others'
- In some cases we are using expert judgement technique, and in some cases pretty developed knowledge and robust methods.
- more knowledge is needed in relation to the correlation between pressures, especially multiple pressures and biological responses and effectiveness of associated measures
- This depends. I am convinced that all of us yet have to learn a lot about the complexities of our water systems and the complexity of the ecology in these systems. I think nobody has complete understanding (or something bordering complete understanding) of causes of problems and effects of measures. At the other hand, we have to work with what we have got. With what we know now. It may not be correct, but we can, and have to take decisions
- We work to gain more insight in our water systems using detailed quantified analysis of both water quantity and water quality in order to advise for useful measures and adjustments of targets to be met.
- The 2nd Danube River basin management Plan (Update 2015) includes measures of basin-wide importance oriented towards the agreed visions and management objectives for 2021. It is based on the national programmes of measures elaborated by Danube countries, which shall be made operational by December 2018, and describes the expected improvements in water status by 2021.
- When there are multiple pressures then there is uncertainty and further investigation is needed, i.e. additional monitoring, pressure analysis etc.
- Time is also a problem. It takes a lot of time to do a proper analysis of a water system. At Noorderzijlvest we did not have sufficient time to analyze every system on time.
- Yes, I have general overview of the pressures, status and functioning of my river basin to select measures, but because of combined impacts of multiple stressors on the ecological status need models/other methods for prioritization of different measures
- We combine model output with long term data sets
- Difficult to say in general. It depends on the type of pressure and on the particular water body. For example we have a lot of data sets for flow, nutrient, metal of chemical pollution in the water bodies. Solving problems in the hydromorphology and biodiversity is mainly based on expert judgment. But there is a lack of data for atmospheric deposition or nutrients from agricultural areas (diffuse sources in general

Part 2

1.1 Which of the four identified multiple pressure combinations are relevant for your system?

Specific additional comments for 'different sets of multiple pressures':
- Changes in morphology (disruption of river continuity, modifications of river banks, river bed), nutrient stress, changes of (sediment) flow, heat load by thermal discharge, (in the future changes in flow and water temperature related to climate change (low flow, high flow)
- Combinations of morphology (e.g. river regulation), nutrient stress and resulting sediment colmation
- Organic substances and nutrients
- Sometimes human induced changes in a water system have impact on many aspects of the system.
- Organic pollution and nutrients. Priority substances and specific substances and nutrients

- Hazardous substances emissions represent an additional pressure in potential combination with others (e.g. low flow conditions)
- Changes in morphology – lack of biological / ecological continuity – sediment accumulation – nutrient stress – other pollutant stress – low flow combined with nutrient stress
- Extreme low PH due to drainage of acid sulphate soils and nutrient stress
- Changes in morphology, nutrient stress and stress from environmental pollutants (River basin specific pollutants as well as priority substances)
- Little water availability compared to water use of landowners
- Slow eutrophication and habitat deterioration
- Point and non-point sources of pollution
- In Flanders, there is a multi-pressure environment with only a little bit of room for water

1.2 Do you have a good understanding of the impact of these multiple pressures situations?
Specifications of the answer for the response '*Yes, I know that the multiple pressures occurring in my river basin cause an even stronger negative response of the aquatic ecosystem than the expected effect of each of the single pressures*'.

- Organic substances and nutrients
- Flood and nutrient loss
- Nutrient loading (agriculture) Nutrient release from sediments, floods and sewer overflow (meanly in city water systems) and high oxygen consumption by the outlets of the sewer overflow (Organic loads (oxygen dynamics) and loading of the sediments. This causes multiple problems in my aquatic ecosystem
- Low flow and nutrient stress, changes in morphology and nutrient stress
- Changes in morphology and nutrient stress, organic pollution and nutrients, priority substances and specific substances and nutrients
- Habitat alternation, nutrient morphological structure
- Nutrients, changes in morphology, low flow (lower as usual), temperature (long term changes, climate change)
- Low (summer) flow, high temperature, nutrients, hydromorphological alterations
- E.g.: weirs for hydropower generation impounding rivers + no dense vegetation / trees on the river banks due to agricultural land use or settlements => higher water temperature due to sunlight exposure (no shade) + higher retention time (impoundment) + higher sediment and nutrient (phosphorus) intake due to surface runoff (missing buffer zone) => increasing trophic status, eutrophication, higher than in single stressor situation
- Acidity and nutrient stress/extreme high flow and nutrient stress
- Yes, but I cannot quantify it
- Low flow and nutrient stress
- Changes in morphology, nutrient stress and stress from environmental pollutants (River basin specific pollutants as well as priority substances) as well as high flows.
- Extreme low flow situations (Niedrigwassersituation) + nutrient stress; extreme low flow situations (Niedrigwassersituation)+ water level regulation and damming of watercourses ("Stauhaltung")
- Ee mainly understand the combined effect but the result varies from site tot site and species to species
- Dams/ impoundment + temperature + organic pollution/ eutrophication

Specifications of the answer: '*Yes, I know that the multiple pressures occurring in my river basin cause a less negative response in the aquatic ecosystem than the expected effect of each of the single pressures.*'

- Nutrients are dominating, lakes and low flow urban waters are having the biggest problems
- No dense vegetation / trees on the river banks due to agricultural land use or settlements + increased flow velocity due to straightening of the river course or due to flash flooding from hydropower works => higher flow velocity reduces by erosion/ sediment transport the negative effect of sedimentation / colmation (filling) of the interstitial room due to missing buffer zones on the banks. Mainly in mountainous rivers the interstitial room is the main habitat for invertebrates and the spawning ground for most fish species. Colmation of the interstitial room therefore is reducing the habitat condition und decreasing fish reproduction with in total reducing the ecological status. Artificially increased flow velocity may reduce these negative effects. Rock ramps and weirs with the water falling down in cascades (interrupting the continuum for aquatic organisms and sediment transport) have high reaeration rates reducing the effects of saprobic pollution.

- yes but I cannot quantify it

1.3 Do you have a good understanding of the consequences of these multiple pressures situations for the end-users of the system?

- Waterworks, navigation (high and low flows), agriculture, fisheries
- The pressures described in 1.1 have a dominant impact on the aquatic system.
- Drinking water supply, recreation (fishing) and irrigation/agriculture
- Reduced quality of drinking water, irrigation, recreation
- There are negative consequences for different users and this is depending on the needs of the users. For example swimmers want healthy water without bacteriological contamination. Bird watcher like to see birds. Birds can contaminate the water.
- Risk of flooding of fields
- Civilians, tourists
- Users as – fishing, recreation etc.
- E.g. decreased water quality for drinking water supply, increased danger of flooding, decreased quality for recreation, higher costs for river maintenance, reduced attractiveness for game fishing reducing the earnings for fishing licenses, adverse effects for agriculture due to mass development of water plants in draining channels or due to deepening of the riverbed by erosion causing problems of draught in the fields (that was the reason for building still existing artificial irrigation channels also in the Ruhr catchment ("Wiesenbewässserungsgräben")
- Recreation, fishing, water abstraction, flooding, agriculture and land use, security
- It depends, I am aware of some of the negatives effects from reduction of low flow and nutrients increasing and high temperatures on the treatment level of the water to supply to the populations and on the level of treatment of waste water.
- Recreation irrigation
- The pressures impacts the end users by affecting fishing, bathing and swimming, drinking water and floods due to high waters flows.
- Water availability in the floodplains for land purposes, extensive growth of vegetation hampers flow in periods of flooding
- Drinking water scarcity, irrigation

2.1 When you select measures, do you take into account long-term changes to your river basin, such as even further increasing effects of climate change or societal change affecting land use (e.g. 30-50 year horizon)
Yes some of the measures are so costly:

- Re-design of the larger river systems and measures in the cities
- Applying the Code for Good Agriculture Practices
- Sewage treatment, reduce rain water run-off and floods
- River restoration
- Water inlets and new pumping stations and fish passages and the effects of climate change
- o Morphological adjustments of river banks and Establishment of green infrastructure
- o overdimensioneren van watergangen, maatregelen gericht op verduurzaming van landbouw, aanleg vispassages, centralisatie en optimalisatie van rwzi's.
- Climate changes adaptation measures, measures focused on water retention and accumulation as well as the artificial regulation of the surface water outflow regime, measures to protect against harmful effects of water, measures for water regulation in catchment areas
- Water industry, flood risk management
- It was done based on the WFD art. 5 analysis
- Measures related to urban waste water treatment (e.g. societal changes), hydromorphological measures
- Restoration of small river systems, large water storage systems.
- Management of the large reservoirs in the Ruhr catchment (low flow management to guarantee water supply for drinking water and industrial uses (e.g. model to predict the vulnerability of the reservoir system due to climate change conditions); flood management, accident-caused pollution management) , some of the optimizing measures in our larger WWTP and sewer systems (e.g. considering decline in population density).

- Removal of fish migration barriers and other hydromorphological measures
- Water retention in the flood plains by replacing dams with rock ramps (“Sohlgleiten”); decreasing flow cross section for flow diversification (“Verringerung des Abflußquerschnittes zur Diversifizierung der Strömung”)
- Fish passages, nutrient reduction measures
- Mostly measures on water quantity

2.2 Do you feel your current way of working would change if you had a better understanding of the response of your river basin over such a longer period of time?

3 Would you be interested in using such a tool for diagnosis of the response of your system to multiple pressures?

- The planned resolution of the diagnostic tool must work on a very small scale to be useful for a decision support in a specific water body; for that, the effort of a modelling system is too much; what we clearly need for a decision support under multiple pressures are short and easy to understand standards and understandings, best prepared in “cooking recipes” and in the national language of the practitioners.
- Description doesn’t give sufficient information. Before evaluating necessity, we would like to learn more about this tool.
- Yes, I would benefit from such a tool if it provides information specific for the system I am working on. Since I do not know the tool, this question is hard to answer. There are tools that might look interesting from a scientific or theoretical point of view, but are hard to use in the daily life of water management.
- I don’t believe, that such a tool can sufficiently contain enough information for a valid diagnosis (especially with regard to the list of input data presented in the workshop in Delft September 2014)
- The results of the MARS project are only applicable (to us) when they can be applied to small scale water systems, in many cases smaller than water bodies (Ecological Analyses Areas). Par example: we need to know the interaction between the effects of water depth, nutrient load, hydrological loading and sediment depth on aquatic vegetation and fauna in a ditch, which goes further than PCDitch. And we would like to know the multiple pressures that determine macrofauna in small lakes.
- The results of the MARS project should be applicable to individual waterbodies, which are all very different. Results that are similar for all shallow peaty lakes are not useful, as all our shallow peaty lakes have very different characteristics and pressures. All modelling has to be done on a n=1 scale
- I suggest using the “ecological key factor” approach as a framework.
- Depending on the accuracy and scale (water body!) of such a tool.

APPENDIX 3 - List of attendees to the Stakeholder workshop October 2016, Den Helder

	Name	Country	Institute
1	Bernie ter Steege	The Netherlands	Waterboard Vechtstromen
2	Bert Knol	The Netherlands	Waterboard Vechtstromen
3	Cristian Rusu	Romania	National Administration “Romanian Waters"
4	Helena Alves	Portugal	Tagus and West River Basin District Administration
5	Helena Mühlmann	Austria	Federal Ministry of Agriculture, Forestry, Environment and Water Management
6	JoAnne Pitt	UK	Environment Agency UK
7	Mariana Pedras	Portugal	Tagus and West River Basin District Administration
8	Petra Podraza	Germany	Ruhr Waterboard
9	Steven Verbeek	The Netherlands	STOWA, Foundation for Applied Water Research