Multiple stressors determine river ecological status at the European scale: Towards an integrated understanding of river status deterioration

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Abstract
The biota of European rivers are affected by a wide range of stressors impairing water quality and hydro-morphology. Only about 40% of Europe’s rivers reach ‘good ecological status’, a target set by the European Water Framework Directive (WFD) and indicated by the biota. It is yet unknown how the different stressors in concert impact ecological status and how the relationship between stressors and status differs between river types. We linked the intensity of seven stressors to recently measured ecological status data for more than 50,000 sub-catchment units (covering almost 80% of Europe’s surface area), which were distributed among 12 broad river types. Stressor data were either derived from remote sensing data (extent of urban and agricultural land use in the riparian zone) or modelled (alteration of mean annual flow and of base flow, total phosphorous load, total nitrogen load and mixture toxic pressure, a composite metric for toxic substances), while data on ecological status were taken from national statutory reporting of the second WFD River Basin Management Plans for the years 2010–2015. We used Boosted Regression Trees to link ecological status to stressor intensities. The stressors explained on average 61% of deviance in ecological status.

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1 | INTRODUCTION

Rivers are among the most threatened ecosystems worldwide (Vörösmarty et al., 2010). They are affected by a wide range of stressors that differ between regions and river types (Reid et al., 2019). In Europe, the ecological status of more than 100,000 river water bodies is regularly analysed in the framework of the world’s most intensive biological monitoring programme (EEA, 2018a). However, there are significant knowledge gaps on how the individual stressors act in concert on the ecological status (including their interactions), which obstruct targeted management actions (Carvalho et al., 2019).

Stressors affecting river biota include diffuse and point source pollution with organic matter, nutrients and toxic substances, hydrological modification due to water abstraction or flow modification, and morphological impairment due to damming, straightening and the disconnection of the river and its floodplain. While the effects of each of these stressors on river biota are well documented (Birk et al., 2012; Hering et al., 2010), it is much harder to assess the effects of two or more stressors that act simultaneously. They may add to each other, may strengthen or weaken each other’s effects, while in other cases the effects of one stressor superimpose the effects of the second stressors (Birk et al., 2020; Schäfer & Piggott, 2018).

Despite an increasing number of studies on multiple stressor effects in aquatic systems (Birk, 2019; Nõges et al., 2016), the state of knowledge remains incomplete. Inconclusive evidence, for instance on the appearance of interactions (Côté et al., 2016; Crain et al., 2008; Jackson et al., 2016), necessitates increased research efforts (Orr et al., 2020). These include elaborating the statistical foundations for multi-stressor analysis (e.g. mechanistically based null model selection; Schäfer & Piggott, 2018), the role of multiple stressors acting at higher levels of biological organization (De Laender, 2018) and strengthening the prediction of the combined effect of stressors (Van den Brink et al., 2016).

Most experiments addressing the effects of multiple stressors have combined two or three stressors in a replicated experimental design and measured the effects on aquatic biota (Beermann et al., 2018; Elbrecht et al., 2016; Jackson et al., 2016). Such experimental results, however, do not necessarily upscale well in space and time since they only are a snapshot of a particular context at a particular time. In addition, most water bodies are oftentimes affected by more than three stressors (Birk, 2019; EEA, 2018a). Field studies at regional scale have delivered contradictory results, even when targeting the same area. While for Central Europe, for instance, stressors acting at the catchment scale are more relevant than those acting at the local scale (e.g. upstream catchment land-use effects superimposing local river habitat quality effects on aquatic biota; Lorenz & Feld, 2013), the role of water quality versus hydro-morphology for the ecological status remains controversial (Berger et al., 2017; Gieswein et al., 2017). The results depend greatly on the selection, spatiotemporal resolution and quality of stressor data included in the analysis.

Even more challenging is the analysis of stressor relevance at a continental scale, considering potential mismatches between monitoring strategies, studied organisms, or temporal and spatial scales of stressors and response (Altermatt et al., 2020). Yet, Europe’s largest biotic monitoring programme of the Water Framework Directive (WFD) offers the opportunity for such continental-scale analysis. The programme includes more than 100,000 river water bodies and determines their ‘ecological status’ (EEA, 2018a). The ecological status is an integrative indicator of stressor effects on ecosystem functioning and structure. This indicator assesses biological attributes of selected aquatic organism groups (e.g. fish species richness, sensitive invertebrate species abundance, total diatom biomass). Slight but not significant deviations of the indicator from the natural, undisturbed conditions are classified as ‘good ecological status’, which is the WFD’s mandatory target for water management (Birk et al., 2012; Nõges et al., 2009). By 2015, only about 40% of the surface water bodies obtained good ecological status (EEA, 2018a) and the achievement of this target by 2027, the deadline set by legislation, is increasingly unlikely (Carvalho et al., 2019). Surprisingly, given the huge monitoring efforts, there is a significant knowledge gap on how the individual stressors, singly or in concert, affect ecological status, hydrology, nutrients, riparian land use, river types, stressor interactions, toxic substances

KEYWORDS
status (Carvalho et al., 2019; Søndergaard & Jeppesen, 2015). Based on data recorded in the first WFD monitoring cycle (2004–2009), Grizzetti et al. (2017) concluded that water quality (in particular nitrogen pollution), hydro-morphology and catchment land use as an overarching driver are the main determinants of ecological status. However, the study did not reveal whether there are differences in stressor–status relationships between regions and river types, left aside stressor interactions and ignored potentially important stressors such as toxic pollution that were ranked as highly relevant by other authors (Malaj et al., 2014).

In this study, we have built river type-specific models linking the intensity of seven stressors to recently reported ecological status data with the aim to disentangle the effects of individual and combined stressors (including interactions) on ecological status at a continental scale. These stressors are as follows: urban and agricultural land use in the riparian zone; alteration of mean annual flow and of base flow; total phosphorous and total nitrogen riverine loads; and mixture toxic pressure. The data on stressors and ecological status were collated for more than 50,000 hydrological sub-catchments with a median size of 60 km². We addressed the following hypotheses:

**Hypothesis 1** We expected that the ecological status of Europe’s rivers is determined by the intensity of multiple individual stressors affecting water and habitat quality, rather than by single, intense stressors (Grizzetti et al., 2017; Reid et al., 2019; Skjelkvåle et al., 2005).

**Hypothesis 2** We expected that stressor hierarchy in determining ecological status depends on the river type (Lyche Solheim et al., 2019). More specifically, we expected nutrient enrichment to be particularly relevant for the ecological status of lowland rivers, whose catchments are often characterized by high agricultural intensities (Hypothesis 2.1; e.g. Lemm & Feld, 2017); hydrological alteration to be particularly relevant for river types with small and

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FIGURE 1 Overview of the locations of 52,847 sub-catchment units and their ecological status class (grey area: no data available)
Hypothesis 3  |  We expected stressor interactions to be relevant for explaining the ecological rivers status at continental scale (Birk et al., 2020), as interactions have frequently been reported between nutrient, toxic, hydrological and morphological stressors in the scientific literature (e.g. Alexander et al., 2016; Chase et al., 2017; Liess et al., 2016; Matthaei et al., 2010; Piggott et al., 2012; Rasmussen et al., 2012; von der Ohe & Goedkoop, 2013; Wagenhoff et al., 2012).

**MATERIALS AND METHODS**

### 2.1  |  Spatial grain and extent

All data were compiled and modelled for sub-catchment units named ‘Functional Elementary Catchments’ (FEC) that were derived from the Catchment Characterisation and Modelling dataset and topologically integrated into the European Catchments and Rivers Network System (ECRINS) database (EEA, 2012a). The model encompasses more than 104,000 FECs (Globevnik et al., 2017): for 52,847 of these, data on all seven stressors and on ecological status could be compiled (Figure 1). The median size of the FECs considered is 60 km² with a minimum of 0.01 km² and a maximum of 2561 km².

### 2.2  |  Broad river types

We classified rivers and streams of the ECRINS river network into 12 broad river types (Figure 2) characterized by size, altitude, catchment geology and region, that is, the main typological factors defined by the WFD. The typology is based on Lyche Solheim et al. (2019), derived from a synthesis of the river typologies of the EU member states. For each FEC, the river type was defined (Table 1); if more than one river type was located in a FEC, the river type at the outlet of the FEC was selected.

### 2.3  |  Stressor data

Data on seven morphological, hydrological and water quality stressors were compiled or modelled (Table 2). They include stressors originating from the presumably most common and relevant pressures affecting the ecological status of Europe’s waters, in particular from hydro-morphological degradation, and pollution from diffuse and point sources (EEA, 2018a).

#### 2.3.1  |  Urban and agricultural land use in the riparian zone

As a proxy for morphological and habitat degradation, we compiled data on the land use in the potentially flood-prone areas as an average of the years 2011–2013. The flood-prone area was derived from two spatial layers: (1) the JRC-flood-hazard-map for Europe with 100-year return period, compiled with the flood model ‘LisFlood’ (Alfieri et al., 2014; Bates & De Roo, 2000) and (2) the Copernicus Potential Riparian Zone layer compiled with data from the Copernicus Land Monitoring Service (CLMS, 2019; EEA, 2015). We used the relative share of agricultural land and of urban areas in the riparian zone of the ECRINS river network within a FEC primarily as a proxy for morphological degradation while the effects of riparian land use on nutrient and pesticide input were covered by the parameters explained below.

#### 2.3.2  |  Alteration of mean annual flow and of base flow

Hydrological stressor data were generated with the global model PCR-GLOBWB (Sutanudjaja et al., 2018). Two datasets of daily time series of river discharges covering the period 2001–2010 were simulated under the same climate (Panagopoulos et al., 2019). The first dataset resulted from a least-disturbed condition scenario, excluding all water uses such as irrigation, abstractions, industry or water management as well as the presence and hydrological impacts from reservoirs. The second dataset represented current conditions including anthropogenic activities affecting run-off and water balances. For both datasets, two FEC-specific indicators were derived: mean annual flow and the mean annual base flow index (i.e. mean long-term ratio of base flow to total river flow), using the ‘Indicators of Hydrologic Alteration’ software package (Richter et al., 1996; The Nature Conservancy, 2009). For both indicators separately, we used the relative deviation of the current anthropogenic from the least-disturbed conditions scenario to quantify the FEC-specific level of hydrological stress. Hydrological alteration and the riparian land use are considered ‘hydro-morphological stressors’ as opposed to the water quality stressors described in the following.

#### 2.3.3  |  Total phosphorous and total nitrogen riverine loading

Based on the run-off data provided by PCR-GLOBWB, riverine nutrient loads were quantified using the process-oriented model...
FIGURE 2 Broad river types of the river sub-catchment units considered (grey area: no data available)

| Broad river type | Number of FECs | Share of FECs in ecological status (%) | Absolute surface area (km²) | Relative surface area (%) |
|------------------|----------------|----------------------------------------|-----------------------------|---------------------------|
| Small lowland rivers, calcareous | 3710 | 2 24 45 21 8 | 286,842 | 8 |
| Small lowland rivers, siliceous | 4514 | 4 30 45 15 6 | 313,756 | 9 |
| Large lowland rivers, calcareous | 7194 | 2 23 51 20 5 | 631,512 | 18 |
| Large lowland rivers, siliceous | 5685 | 5 28 49 15 4 | 480,281 | 14 |
| Small mid-altitude rivers, calcareous | 2879 | 4 42 36 14 3 | 129,279 | 4 |
| Small mid-altitude rivers, siliceous | 4330 | 12 31 42 12 3 | 212,823 | 6 |
| Large mid-altitude rivers, calcareous | 3201 | 2 34 46 15 3 | 182,373 | 5 |
| Large mid-altitude rivers, siliceous | 4692 | 10 29 44 14 3 | 310,759 | 9 |
| Very large rivers | 2635 | 2 18 52 23 6 | 167,197 | 5 |
| Highland/glacial rivers | 4732 | 12 55 27 5 1 | 185,059 | 5 |
| Mediterranean rivers, perennial | 4730 | 5 32 40 17 6 | 326,725 | 9 |
| Mediterranean rivers, intermittent | 4545 | 7 40 36 13 4 | 247,790 | 7 |
| Sum | 52,847 | — — — — — | 3,474,396 | 100 |
The ecotoxicity stressor was derived from Europe-wide integrated exposure and effect modelling, including two components: (1) a spatially and temporally resolved model for emissions and fate-and-transport of chemicals driven by a hydrological model (van Gils et al., 2020), yielding Europe-wide daily predicted environmental concentrations (dissolved part) of 1785 man-made organic chemicals in water bodies to obtain a ‘real-life’ mixture exposure for each FEC (reference year: 2013) and (2) species sensitivity distributions based on effect models considering the acute median effective concentration (EC$_{50}$) of each studied chemical as effect endpoint (Posthuma et al., 2019). Combining (1) and (2), and adding a step of mixture modelling yield the mixture toxic pressure metric, which is expressed as ‘multi-substance Potentially Affected Fraction of species’ (msPAF; De Zwart & Posthuma, 2005). The msPAF ranges between 0 and 1 and estimates the likelihood of direct effects of chemical exposure on growth and reproduction of aquatic organisms (van Gils et al., 2019; Posthuma, van Gils, et al., 2019). In our study, we used the msPAF-EC$_{50}$ based on 95th percentile predicted environmental concentrations of the daily concentration estimates, representing an acute toxic stress level exceeded at 18 days per year. The mixture toxic pressure data were obtained on the spatial level of the hydrologic model E-Hype (Lindström et al., 2010) and then (dis)aggregated at FEC level.

We compared the simulated concentrations of toxic substances to chemical monitoring data for 226 substance/basin combinations which showed that the simulated concentrations were accurate on average. For 65% and 90% of substance/basin combinations, the error was within one and two orders of magnitude, respectively (van Gils et al., 2020), which is relatively low compared to the inter-site concentration variability spanning up to 20 orders of magnitude for the 1785 chemicals present in the analysis (see Supporting Information S-II), and the variability of the species sensitivities, which span nine orders of magnitude when assessed by the medians of the respective Species Sensitivity Distributions (see Supporting Information S-II). Based on both measured and predicted exposures, mixture toxic pressure has been shown to relate to ecological impacts in various datasets (geographies, species groups, chemical mixtures; Posthuma & De Zwart, 2006; Posthuma et al., 2020). This means that the mixture toxic pressure is a metric that meaningfully represents ecological impact magnitudes. Although predicted environmental concentrations have limited precision, the calculated toxic pressure for the mixture is expected to be robust for a high number of substances as evaluated here.

### 2.3.4 Mixture toxic pressure

| Stressor                                                                 | Unit                  | Min   | Max   | Median |
|-------------------------------------------------------------------------|-----------------------|-------|-------|--------|
| Percentage of agricultural land use in the riparian zone                | %                     | 0     | 100   | 12     |
| Percentage of urban land use in the riparian zone                        | %                     | 0     | 100   | 4      |
| Relative deviation of mean annual flow (anthropogenic vs. least-disturbed scenario) | Ratio                | 0     | 100   | 0      |
| Relative deviation of mean annual base flow index (anthropogenic vs. least-disturbed scenario) | Ratio                | 0     | 100   | 0      |
| Total phosphorous riverine load                                         | kg km$^{-2}$ year$^{-1}$ | 0     | 7365  | 26     |
| Total nitrogen riverine load                                            | kg ha$^{-1}$ year$^{-1}$ | 0     | 527   | 7      |
| Mixture toxic pressure (msPAF-EC$_{50}$)                                | Ratio                 | 0.00  | 0.97  | 0.04   |

The ecological status (including ecological potential; Kampa & Hansen, 2004) is an assessment of the quality of the structure and functioning of surface water ecosystems, including rivers (European Commission, 2000). It reflects the influence of pressures (e.g. pollution and habitat degradation) on biological quality elements (BQEs),

### 2.4 Ecological status

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that is, phytoplankton, phytobenthos, macrophytes, benthic invertebrates and fish. Besides BQEs, determination of ecological status is supported by physicochemical and hydro-morphological quality elements. For each BQE, assessment methods are based on indices considering the share of sensitive and tolerant species, diversity, abundance and functional characteristics (Birk et al., 2012). The observed index values of a river to be assessed are compared to expected undisturbed ‘reference conditions’ (Wallin et al., 2003). The resulting ‘Ecological Quality Ratio’ (i.e. the observed value divided by the expected value) is finally classified into one out of five status classes (high, good, moderate, poor and bad). The overall ecological status classification for a water body is determined by the element with the worst status out of all the biological and supporting quality elements (‘one out, all out’ principle; European Communities, 2005).

Data on ecological status were available for 52,847 FECs resulting from the second River Basin Management Plan (RMBP) reports for the years 2010–2015, supplemented by data on the first RBMP reports for missing countries in the second RBMP to increase the coverage. The monitoring was performed by the EU member states and the results were reported to the European Environment Agency (EEA, 2012b, 2018b). If more than one water body with reported ecological status was located in a FEC, the results from the river water body closest to the FEC’s outlet were used, assuming that stressor effects accumulate within a catchment and are strongest at the most downstream part. The classification of ecological status was based on a different number of BQEs considered per FEC, with about 90% of FECs including benthic invertebrates, 66% of FECs including phytobenthos, 50% of FECs including fish and 33% of FECs including macrophytes. For the majority of FECs, two or more BQEs were thus contributing to the status classification, with FECs in less than good status often being classified by two or more BQEs in moderate or worse status.

2.5 Data analysis

Using the FEC as the basic data unit, we calculated multiple-stressor–response relationships with nonlinear Boosted Regression Tree models (BRT; Elith et al., 2008) for each broad river type and for the full dataset. In advance, Spearman’s rank order correlation was analysed for the individual stressors and the ecological response variable, and we tested for multicollinearity using the variance inflation factor (VIF < 5.0; Zuur et al., 2007). This was done to check how strongly the individual parameters are correlated and might therefore indicate an overarching driver.

BRT models were run with untransformed input variables and calibrated using various model attributes: Model complexity was controlled with the help of the ‘learning rate’. Small learning rate values increase the number of trees and decrease the influence of every single tree. We fitted the models using the gbm.step function from the dismo library in R (Hijmans et al., 2017). To achieve comparable results, both learning rate and tree complexity (which fits interactions) were set in a way that each model was based upon a similar number of regression trees (at least N = 1000). Each model run included a k-fold cross-validation using a pre-defined fraction of the data to train the model and the remaining fraction to validate the model. Training and validation data were set to 70% and 30%, respectively, setting the argument bag.fraction = 0.7. The total explained deviance per model was derived by ‘(mean total deviance − mean residual deviance)/mean total deviance * 100’ (Derville et al., 2016).

We compared the results of the final BRT models (including interactions using tree complexity > 1) to simple additive BRT models (using tree complexity = 1), controlling for model overfitting (Elith et al., 2008). The increase in total explained deviance gained by including stressor interactions was expressed as a percentage of the final model’s total explained deviance. We further identified the relative strength of the two most important pairwise interactions per river type (Elith et al., 2008).

### 3 RESULTS

3.1 Spearman’s rank order correlation

For the total population of FECs, Spearman’s rho was highest for the combination of total phosphorous load and total nitrogen load (0.76) and for the combination of altered mean annual flow and altered base flow (0.56; Table 3). These two pairs were also among

| TABLE 3 | Spearman correlations (rho values) for all river types combined (number of FECs = 52,847) |
|----------|---------------------------------|
|          | msPAF  | Base flow | Mean flow | Agricultural land use | Urban land use | Nitrogen load | Phosphorous load |
| Ecological status | 0.26   | 0.16   | 0.17   | 0.23   | 0.18   | 0.17   | 0.11   |
| Phosphorous load   | 0.44   | 0.07   | 0.13   | 0.10   | 0.31   | 0.76   |
| Nitrogen load      | 0.40   | 0.07   | 0.11   | 0.03   | 0.33   |
| Urban land use     | 0.33   | 0.13   | 0.21   | 0.08   |
| Agricultural land use | 0.38   | 0.20   | 0.23   |
| Mean flow          | 0.25   | 0.57   |
| Base flow          | 0.29   |

All rho values are significant with p < 0.001.

Abbreviations: Base flow, altered base flow index; Mean flow, altered mean annual flow; msPAF, mixture toxic pressure.
the most strongly correlated parameters for all individual river types with Spearman’s rho ranging from 0.33 (large calcareous mid-altitude rivers) to 0.91 (large siliceous lowland rivers) for the combination of phosphorus and nitrogen loads, and from 0.43 (large calcareous mid-altitude rivers and permanent Mediterranean rivers) to 0.79 (small calcareous mid-altitude rivers) for the combination of altered mean annual flow and altered base flow (see Supporting Information S-III). The variance inflation factor did not exceed a value of VIF = 3.0.

Although in general stressors were weakly (but always significantly) correlated with ecological status (Spearman’s rho ranging from 0.11 to 0.26), mixture toxic pressure was notably among the most influential stressors for 7 of the 12 river types (mean Spearman’s rho = 0.29) as compared to other river types (mean Spearman’s rho = 0.16; z test, p < 0.001). For very large rivers, ecological status was significantly stronger related to nutrients and the mixture toxic pressure than to land use and hydrological parameters (z test, p < 0.001).

### 3.2 Nonlinear BRT modelling

The joint analysis of stressors using boosted regression tree analyses revealed a different pattern (Figure 3, see also Supporting Information S-IV). Depending on river type, the share of deviance in
ecological status class explained by the seven stressors ranged between 39% (small calcareous lowland streams) and 85% (large calcareous mountain rivers); the average for all river types was 61%. For the total population of 52,847 FECs, the explained deviance was 51%.

For the total population of FECs as well as for the individual river types, each of the seven stressors contributed to the explained deviance (Figure 3). For the total population, the mixture toxic pressure parameter (msPAF-EC50) represented the highest share in relative explained deviance (32.9%), followed by nutrients (phosphorous load: 14.2%; nitrogen load: 16.1%) while the two land uses and hydrological parameters contributed to 20.4% and 16.4% of the explained deviance, respectively. For the 12 individual rivers types, the mixture toxic pressure metric contributed less to the explained deviance (average 26.2%) while the contribution of nutrients (on average 16.4% and 18.0% for phosphorous load and nitrogen load, respectively) and land use/hydrology (on average 23.2% and 16.2%) was in the same order of magnitude. Differences between river types in the relative explained deviance of individual stressors were minor. For all but two river types, the mixture toxic pressure was the individual stressor explaining the highest share of deviance (total nitrogen for small and large calcareous mid-altitude rivers).

For further analysis, we grouped the seven stressors into three categories: 'hydro-morphology' (land use and hydrological stressors), 'nutrients' (phosphorous and nitrogen loads) and mixture toxic pressure (msPAF). On average, hydro-morphological stressors explained 39.4% of the deviance in ecological status (morphology: 23.2%; hydrology: 16.2%) while nutrients explained 34.4% and the mixture toxic pressure 26.2% (mean of all river types). Overall, the explanatory power of the toxic stressor was highest for lowland rivers (on average 29.6% of explained deviance, as compared to 24.6% for the other river types). Nutrient stress tended to be more relevant for mid-altitude rivers (on average 40.1% of explained deviance, as compared to 31.5% for the other river types), while hydro-morphology (including land use) was most relevant for very large rivers (48.1%; morphology: 27.1%; hydrology: 21.0%) and Mediterranean rivers (on average 45.0%; morphology: 27.2%; hydrology: 17.8%), as compared to 38.2% (morphology: 22.3%; hydrology: 15.9%) for the other river types.

The stressor interactions accounted for an increase in the total explained model deviance by 55.2 percent points (median across all river types), meaning that on average more than half of the explained deviance resulted from interaction effects. This increase was particularly pronounced for large calcareous lowland rivers (72.6%) and highland/glacial rivers (71.0%). Stressor interactions were (almost) irrelevant for small mid-altitude siliceous rivers (5.0%) and very large rivers (0%, as only an additive model was run to avoid overfitting; see Supporting Information S-V). Among the most important pairwise interactions, the combination of nutrient and toxic stress was most frequent and strongest (nitrogen load and msPAF, relevant for seven river types), followed by nutrient and morphological stress (relevant for five river types). Comparing the relative strength of all the two most important pairwise interactions per river type, the strongest interaction exceeded the second strongest interaction 1.1–3.2 times across all river types except for small lowland siliceous rivers, for which the interaction of nutrient and toxic stress was 14.5 times stronger than the interaction of hydrological and toxic stress (see Supporting Information S-V).

4 | DISCUSSION

4.1 Multiple stressors acting on riverine biota at continental scale

According to Hypothesis 1, we expected the ecological status of Europe’s rivers to be determined by the intensity of multiple individual stressors, rather than by single, intense stressors (EEA, 2018a). This expectation was generally confirmed, although the seven stressors revealed some modest to strong interrelations. Rank correlation coefficients showed that the two nutrient stressors (phosphorous and nitrogen) as well as the two hydrological stressors (alteration of mean annual flow and of base flow) were correlated with each other. However, correlations of stressors among different categories remained relatively low (including values of the variance inflation factor), and pairwise correlations of ecological status and individual stressors were low, too, thus supporting our abovementioned expectation. If considered in concert, however, the seven stressors explained more than 50% of the deviance in ecological status for the total population of sub-catchment units, and on average more than 60% for the individual river types. This clearly indicates that, at a continental scale, riverine biota are affected by multiple stressors that impose different types of stress and most probably also interact with each other.

On average, the explained deviance was 10% higher for individual river types as compared to the total population of sub-catchment units; this supports the conclusion that there is a river type-specific response of ecological status to the stressors considered. The finding further suggests that the unexplained deviance might be due to natural variation among river types or type-specific confounding factors (including additional stressors) not addressed in this study. Furthermore, each country applies an own set of assessment methods for the different biological quality elements (Birk et al., 2012) that have been intercalibrated between countries (Poikane et al., 2014), but may still be a notable source of variation in the data. In addition, the number of biological quality elements assessed and finally summarized into an ecological status class may differ between sub-catchment units. Despite all these potential sources of variation, the major proportion of variability in ecological status was captured by the considered stressors, supporting the conclusion that hydro-morphological degradation, nutrient enrichment and exposure to mixture of toxic substances together are the main determinants of ecological river status.

In previous studies, the ecological status has mainly been related to a smaller number of stressor types; the respective studies that are often of a regional nature give contrasting results. For instance, it remains controversial if water quality patterns
(Berger et al., 2017) or hydro-morphology and riparian land modification (Gieswein et al., 2017) are the main stressors impacting the ecological status of Central European rivers. Grizzetti et al. (2017) performed a first pan-European analysis with ecological status data and linked those data to a variety of modelled stressors. In contrast to our analysis, the data used by Grizzetti et al. were taken from the first River Basin Management Plans covering the period 2004–2009, which were less complete and quality checked than those provided with the second River Basin Management cycle (2010–2015). The modelled stressors displayed strong gradients, with hydrological stress mainly affecting the Mediterranean and morphological stress prevailing in Central Europe. Overall, the combination of stressors in their study explained Europe-wide ecological status of rivers well, with presence of natural areas in floodplains, nitrogen concentration, infrastructures in floodplains, and urbanization and agriculture in the drained catchment being most relevant. Though toxic substances were not considered as detailed as in the present study, the general results concluding that multiple rather than single stressors were responsible for ecological status are in line with our analysis.

4.2 | Hierarchy of stressor effects: Hydro-morphology > nutrients > toxic substances

According to Hypothesis 2, we expected that multiple stressors, although operating in concert, express some form of river type-specific hierarchy (i.e. some stressors are more influential on ecological status than others); this was only partly confirmed. In general, there are some differences between individual rivers types or between groups of rivers types, but the overall pattern remains the same: The three stressor categories ‘hydro-morphology’ (including riparian land use), ‘nutrients’ and ‘toxic substances’ affect the ecological status of European rivers in roughly a ratio of 1.5 to 1.3 to 1.0. If morphology and hydrology are kept separate, the ratio is 1.3 (nutrients) to 1.0 (toxic substances) to 0.9 (morphology) to 0.6 (hydrology). Our findings are coherent to Grizzetti et al. (2017), who observed N-pollution and various types of structural degradation and catchment land uses as the main stressors; however, we are now able to reveal the additional impact of complex mixture of toxic substances, whose substantial contribution to the multi-stressor effects in Europe’s surface waters is increasingly evidenced (Malaj et al., 2014; Posthuma et al., 2019).

4.3 | Stressor effects across river types

The differences in stressor hierarchies between rivers types observed were partly contrasting to our expectations. We expected nutrient enrichment to be particularly relevant for lowland rivers, which catchments are often characterized by high agricultural intensities (Hypothesis 2.1). The opposite, however, was the case: nutrient enrichment explained a higher share of deviance in mountain rivers than in lowland rivers. A possible explanation for this pattern may relate to a more pronounced gradient of nutrient enrichment in mountainous catchments while large parts of the European lowlands are quite homogeneous in this respect, consistently burdened by intensive human land-use pressures (EEA, 2018a).

We expected hydrological alteration to be particularly relevant for river types in the Mediterranean, which are frequently affected by water extraction for irrigation (Zal et al., 2017; Hypothesis 2.2). However, the share of deviance explained by hydrology in Mediterranean rivers was comparable to the overall population of sub-catchment units. There were differences between permanent Mediterranean rivers (larger share of deviance explained by hydrological parameters) and intermittent rivers (smaller share explained). Biota of intermittent rivers are adapted to regular water shortage and, thus, might be less affected by additional water abstraction by humans (Stubbington et al., 2018).

We expected toxic substances to mainly affect small streams in intensively used agricultural or urban areas (Hypothesis 2.3). However, toxic substances were affecting all river types irrespective of their size, with the exception of very large rivers, for which the impact of toxic substances was relatively smallest. This pattern is likely related to the fact that both smaller and larger rivers may be located in areas with intense pesticide use or affected by waste water treatment plants while the gradient in toxic substance concentration is smallest in very large rivers that show better dilution capacity (Rice & Westerhoff, 2017) and integrate over large and heterogeneous catchment areas (Thorp, 2014).

Furthermore, we expected riparian land use to particularly impact ecological status of small streams, which are likely to be more strongly affected by the immediate surrounding as compared to larger rivers, for which catchment-scale variables (hydrology, nutrient input, toxic substances) are more relevant (Hypothesis 2.4). This was not confirmed, supporting the conclusion that riparian land use still poses a significant effect on biota of larger rivers, for example, by restricting the connection between river and floodplain (Tockner et al., 2010).

4.4 | High relevance of interacting stressors on ecological river status

Finally, we expected stressor interactions to be relevant for explaining the ecological river status at continental scale (Hypothesis 3); this expectation was supported. Interactions substantially contributed to the model performance for almost all river types. This confirms the observation by Birk et al. (2020) of more frequent interaction effects with increasing spatial scale of investigation, presumably driven by longer stressor gradients enhancing the likelihood of interactive stressor effects. As expected, nutrient and toxic stress as well as nutrient and morphological stress interacted most frequently. This underlines the specific multi-stressor challenge for river management, including the increased likelihood of unpredictable ‘ecological surprises’ (Carvalho et al., 2019; Côté et al., 2016).
4.5 Managing Europe’s river status under multiple stress

According to our analysis, the problems affecting Europe’s waters are quite similar, independent from region or river size. Though stress intensity differs greatly between regions (Grizzetti et al., 2017), as also reflected by river ecological status (Figure 1), the same stressor types are relevant across Europe. Almost all European river types are affected, to a varying degree, by riparian land use, hydrological changes, nutrient enrichment and the input of toxic substances including stressor interactions. This does not necessarily mean that each individual river is affected by all stressors, but on larger scales they are all relevant.

Our results suggest that the efforts required to reach the ambitious targets of the European Water Framework Directive can best be made using a holistic concept of water quality assessment and management, that includes the ‘classical’ pressures (such as nutrient pollution by single substances like nitrogen or phosphorous) but also a comprehensive assessment of mixtures of different pollutants. Small-scale and isolated rehabilitation measures, such as physical restoration measures and the treatment of point sources, can certainly contribute to reduce morphological, nutrient and toxic stress while catchment-scale approaches are required to reduce the impact of riparian land use and diffuse pollution with nutrients and pesticides. The improvement of ecological status is, therefore, a long-term objective that requires a comprehensive approach to impact diagnosis and management solutions that acknowledge the key combination of site-specific pressures.

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