Stressors affecting the ecological status of temporary rivers in the Mediterranean region

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Abstract

Temporary rivers are widespread in the Mediterranean Basin area and impose a challenge for the implementation of the Water Framework Directive (WFD) and other environmental regulations. Surprisingly, an overarching analysis of their ecological status and the stressors affecting them is yet missing. We compiled data on the ecological status of 1504 temporary rivers in seven European Mediterranean Basin countries and related their ecological status (1) to publicly available data on pressures from the European WISE-WFD dataset, and (2) to seven more specific stressors modelled on a sub-catchment scale. More than 50% of the temporary water bodies in the Mediterranean countries reached good or even high ecological status. In general, status classes derived from phytobenthos and macrophyte assessment were higher than those derived from the assessment of benthic invertebrates or fish. Of the more generally defined pressures reported to the WISE-WFD database, the most relevant for temporary rivers were ‘diffuse agricultural’ and ‘point urban waste water’. Of the modelled more specific stressors, agricultural land use best explained overall ecological status, followed by total nitrogen load, and urban land use, while toxic substances, total phosphorus load and hydrological stressors were less relevant. However, stressors differed in relevance, with total nitrogen being most important for macrophytes, and agricultural land use for phytobenthos, benthic invertebrates and fish. For macrophytes, ecological quality increases with stressor intensity. The results underline the overarching effect of land use intensity for the ecological status of temporary water bodies. However, assessment results do not sufficiently reflect hydrological stress, most likely as the biological indicators used to evaluate these systems were designed for perennial water bodies and thus mainly target land use and nutrient impacts. We conclude that biomonitoring systems need

to be updapted or newly developed to better account for the specific situation of temporary water bodies.

Keywords: phytobenthos, macrophytes, benthic invertebrates, fish, intermittent streams

1 Introduction

Freshwater life in the Mediterranean Basin area is highly diverse, but at the same time subjected to multiple natural and anthropogenic disturbances (Skoulikidis et al., 2017). The hot and dry climate leads to low flows and rivers frequently dry up in summer. This is a natural phenomenon and temporary rivers are thus widespread in the Mediterranean area (Messager et al., 2021). On top of desiccation in summer, temporary rivers can be subject to extreme floods in autumn and winter. Therefore, biota inhabiting temporary waters need to be adapted to desiccation and generally to unstable and highly seasonal conditions (Lopez-Rodriguez et al., 2009; Garcia et al., 2014; Cid et al., 2017; Tonkin et al., 2017). Species adapted accordingly are an integral part of the natural heritage of Mediterranean ecosystems.

Human demands like water abstraction for agriculture and drinking water, along with climate change, reduce the discharge of both perennial and temporary water bodies particularly in the Mediterranean. As a result, many formerly permanent water bodies become temporary, while the flow phase of temporary water bodies is shortened and the dry phase extended. This has unpredictable consequences for their biota. Besides these hydrological stressors, temporary water bodies are exposed to the same range of stressors that impact most rivers in Europe, such as hydromorphological degradation and pollution from point and diffuse sources (Arenas-Sanchez et al., 2016). However, the resilience of temporary waters may differ as compared to permanent streams. Under low flow conditions, the concentration of nutrients, pesticides and organic waste increases (Sanchez-Montoya et al., 2011), making temporary water bodies particularly susceptible to pollution (Arenas-Sanchez et al., 2016). Effluents of wastewater treatment plants and runoff from irrigation cropfields, however, can also change formerly temporary to perennial streams. Temporary water bodies are, therefore, exposed to a complex mix of stressors that affect them differently, and likely more severely, than their perennial counterparts (Bonada and Resh, 2013).

European water bodies are subjected to regular ecological status assessment following the guidelines laid down in the Water Framework Directive (WFD, EC 2000). For the ecological status assessment, Biological Quality Elements (BQEs) are monitored (Birk et al., 2012); in the case of small to medium size rivers, these are fish, benthic invertebrates, macrophytes and phytobenthos. All these BQEs are also applied to the ecological status assessment of temporary rivers, albeit they are monitored to a different degree in the individual EU Member States. In most countries, except Spain (Gallart et al., 2017), there are no specific assessment systems for temporary rivers, and generally the methods used for perennial rivers are applied. However, an overview of the ecological status of temporary water bodies in the Mediterranean is yet missing.

In addition to the data on their ecological status or potential (in case of Heavily Modified Water bodies), the EU Member States are recording a variety of supporting data on each water body. This includes information on size, category, chemical status, and pressures and impacts acting on the water bodies that might be responsible for deteriorating the ecological quality. These data are centrally stored in the “WISE-WFD database”. Lyche Solheim et al. (2019) used these data to establish a broad typology for all streams, rivers and lakes, which includes temporary rivers as a separate type. In general, the pressures and impact assessment compiled in the WISE-WFD database provides important background information to analyse the causes of ecological status deterioration. However, there are also more specific data for this purpose available. Lemm et al. (2021), for instance, modelled the intensity of seven main stressors, i.e. total phosphorous, total nitrogen, urban and agricultural land use in the riparian zone, potentially toxic substances and two parameters of hydrological alteration, for more than 100,000 “Functional Elementary Catchments” (FECs). Relating the intensity of these stressors to the ecological status of rivers throughout Europe highlights that about 50% of the variability in ecological status is captured by these stressors, out of which hydrological and morphological stressors explain on average 40%, followed by nutrient pollution (35%) and exposure to potentially toxic substances (25%) (Lemm et al., 2021). A more specific analysis on temporary water bodies, however, has not yet been performed.

Against this background, this manuscript aims to close some important knowledge gaps on the ecological status of temporary water bodies in the Mediterranean, addressing the questions: 1.) What is the ecological status of temporary Mediterranean streams and rivers in the individual EU Member States? 2.) What are the main pressures and stressors impacting them? 3.) How do these impacts relate to ecological status?

2 Methods

We limited our analysis to the EU Member States that belong (at least partly) to the Mediterranean basin area and provided stream assessment results to the WISE-WFD database. We compiled data on the overall ecological status (and ecological potential in case of Heavily Modified Water Bodies) of temporary water bodies as well as status assessment results that are based on individual BQEs from the WISE-WFD database. Furthermore, national experts particularly from Portugal, Spain, Greece and Cyprus provided additional BQE data for specific water bodies. For each of the water bodies, for which status results were available, we compiled data on the pressures according to the WISE-WFD database and on stressors of the respective catchment from the dataset generated by Lemm et al. (2021). Status and pressure / stressor data were related to each other.

While the term “pressure” is used for the Water Framework Directive implementation, the term “stressor” is much more commonly applied in the scientific literature. Here we use both terms as synonyms, applying “pressure” when referring to the WISE-WFD dataset and “stressor” when referring to Lemm et al. (2021).

2.1 Datasets

EU Member States report the overall ecological status of their water bodies and partly the assessment results of the individual BQEs and the pressures acting on the water bodies to the European Union, which collates them in the WISE-WFD database. Here we used the data from the 2nd River Basin Management Plans downloaded from the WISE-WFD database. Data include the overall ecological status of the water body, as well as the status assessed for each individual BQE. The ecological status assessment is based on site specific samplings of the different organism groups. The surveys are organised by the individual EU Member States according to country-specific official methods. The samplings were performed in the aquatic flowing phase partly varying in each hydrological year, using the time window between winter flood events and late spring low flow. It should be noted that in many cases not all BQEs (diatoms, macrophytes, invertebrates and fish) were assessed, thus resulting in different numbers of observations per BQE. However, the final ecological status of each water body is based on the “One Out–All Out” principle representing the lowest ecological status assessment of each BQE investigated.

The pressure data are based on estimations by local water managers made in the years 2010-2016. The data indicate the presence or absence of pre-defined pressures without specifying their intensity. In addition, we used a dataset generated by Lemm et al. (2021) that includes both remotely sensed and modelled stressors for about 100,000 Functional Elementary Catchments (FECs): (1-2) Urban and agricultural land use in the riparian area calculated as the averages of the years 2011-2013, (3-4) Alterations of mean annual and base flow derived from a global PCR-GLOBWB on the daily bases of timely recorded discharges from 2001-2010, (5-6) TP (Total Phosphorus) and TN (Total Nitrogen) modelled according to Venohr et al. (2011) for the period 2001-2010 and (7) a toxic pressure indicator (msPAF, De Zwart & Posthuma, 2005), which accounts for the potentially affected fraction of aquatic species by contaminant mixtures that vary spatially based on species sensitivity distributions. (Further details on the modelled stressors can be found in the supplementary information.)

2.2 Data preparation

We first matched the FEC level information of the Lemm et al. dataset to the water body IDs of the EU WISE data base. Then we assigned all water bodies to its respective broad type according to Lyche Solheim et al. (2019) and selected all temporary Mediterranean water bodies. This resulted in a list of Mediterranean temporary water bodies with all available information supplied by the EU WISE-WFD database and with the stressor information modelled by Lemm et al. (2021). In an additional filtering step, we excluded artificial water bodies and all water bodies, for which no ecological status class for at least one BQE was reported. If water bodies were part of several FECs, the stressor information for the involved FEC was summarised by weighted average (weighted by the length of the section in each FEC).

2.3 Data analysis

An overview of ecological status was generated by displaying the fraction of the five classes (“high”, “good”, “moderate”, “poor” and “bad”) in stacked bar plots per member state, where the total is expressed as 100%. This was performed for each of the seven Mediterranean EU Member States that reported temporary streams, and for both the overall status and the status of individual BQEs.

The seven stressors modelled by Lemm et al. (2021) were used as predictor variables (independent variables) in ordinal regression (Bürkner and Vuorre, 2019) using the brms package in R (Bürkner, 2017) predicting the five ecological status classes (“high”, “good”, “moderate”, “poor” and “bad”) as dependent variables. We selected a cumulative ordinal regression with logit link as the underlying model. The prior of the intercepts was set as normally distributed with a mean of 0 and standard deviation of 10 [N(0, 10)]. However, the prior for the regression coefficients was suggested to have a skewed normal distribution with a mean of -1, standard deviation of 1 (see supplementary information for rationale). We selected each country as a random effect in the model, but only by the intercept. The number of chains was set to five with 3000 iterations, the first 1000 as burn-in. The minimum of the effective sample size was allowed to be not <3000. The chains were checked with trace and density plots; Rhat was not allowed at maximum > 1.00 to assess acceptable convergence plus a posterior predictive check. Each dependent variable was cube root transformed as this smoothed the fit of the model to the data and allows zeros.

The ordinal models with a logit link return log odds as regression coefficients having a similar interpretation as the regression coefficients (parameters) in a binomial logistic regression. Thus, they are expressed on the “link” scale. The link function (“logit” link) links the linear component to the expected value, in the ordinal model expressed as $E(\log[\mu/(1-\mu)])=\beta_0-\beta_i X_i$. The estimated coefficients (β), when predicting, needs to be inversed as: $E(\mu)=\exp(\beta_0-\beta_i X_i)/(1+\exp(\beta_0-\beta_i X_i))$. Within the link function the linear component i.e., $\beta_i=-0.35$ indicates a decrease of 0.35 units of the expected value with one increase of the unit X_i on the link scale. A negative value indicates a negative association of predictor to target variable and vice versa for positive values. We expressed the regression coefficients of with 95% credibility intervals.

The results of the ordinal regression were displayed in barplots, where the log odds (regression coefficients) and the credibility intervals at 95% were given. The expected value was displayed as Partial Effect (PE) plot and its uncertainty in Hypothetical Outcome Plots (HOPS, Kale et al., 2019), displaying 500 simulated values (regression lines) from the posterior. The skewed normal distribution was assessed using the fGarch package (Wuertz et al., 2022). All calculations were performed in R (R Core Team, 2021), all figures were plotted with the ggplot2 and cowplot packages (Wickham, 2009; Wilke, 2019).

3 Results

3.1 Quantity and quality of pressures

Applying the typology of Lyche Solheim et al. (2019), 2444 water bodies in the Mediterranean EU Member States were assigned as “temporary” (Fig. 1). Particularly in Cyprus, Italy and Portugal, many water bodies were classified as temporary, while

Slovenia, Croatia and Greece reported only small numbers (Table 1). France did not report any temporary water bodies, i.e. the status of temporary rivers was not assessed. After removing artificial water bodies and all those water bodies, for which no ecological status classes for individual BQEs were reported, 1504 water bodies remained for further analysis.

The mean number of pressures per water body (as reported to the WISE-WFD database) was highest in Spain and Croatia and lowest in Portugal and Greece. In the latter countries, the pressures were often unknown or result from ‘diffuse sewage discharges’ or ‘urban point sources’. Likewise, in the other countries ‘diffuse sources’, ‘urban point sources’ and particularly the overarching pressure ‘diffuse agriculture’ prevail.

3.2 Overview of ecological status classes

In total, 46.1% of all temporary water bodies were reported to be in a good or high ecological status and another 34.1% in moderate status. Spain and Greece reported the most water bodies with poor or bad status, while Cyprus and Portugal assessed their water bodies predominantly either as good or moderate (Fig. 2). Slovenia and Croatia reported assessment results for only five temporary water bodies. In Croatia, the overall ecological status was reported for 197 temporary water bodies, but BQE status was only reported for four of these; we restricted our analysis to the latter.

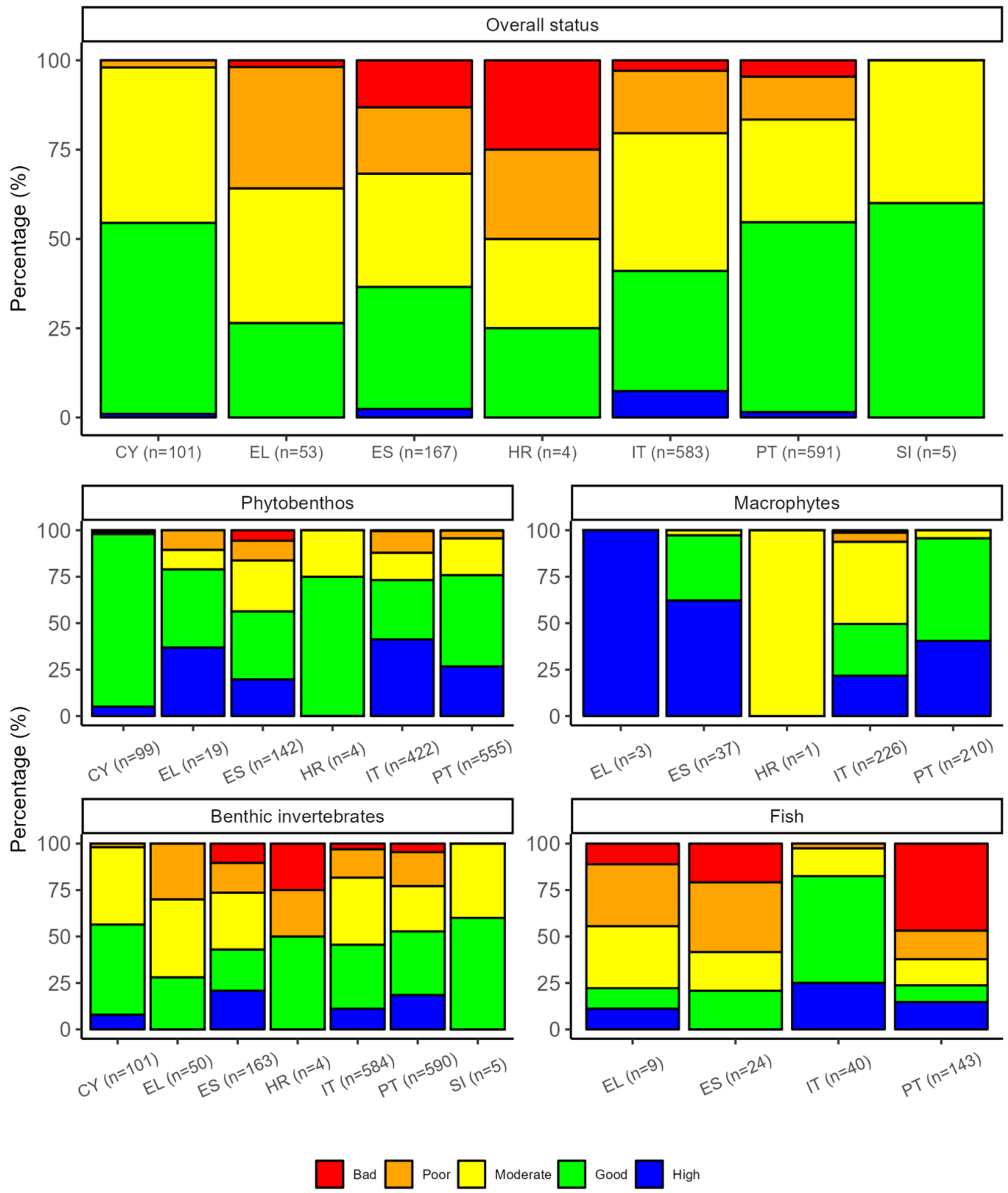

Figure 2. Overall status and status of the four BQEs in temporary Mediterranean basin water bodies per EU Member State. In brackets the number of water bodies.

The ecological status reported for benthic invertebrates and fish is generally lower than that of phytobenthos and macrophytes, and thus decisive for the overall ecological status (Fig. 2). Phytobenthos and macrophyte assessment results were on average ‘good’ or even ‘high’ in the vast majority of water bodies and in all Member States, except for the macrophytes assessment in Italy and the phytobenthos assessment in Spain. On the contrary, fish assessment on average was ‘moderate’ to ‘bad’, especially in Portugal, Spain and Greece.

3.3 Stressor intensities

Stressor intensities in the Functional Elementary Catchments of temporary water bodies showed marked differences between countries (Table S1). Water bodies in Italy and Greece were impacted by twice as high mean total nitrogen loads compared to water bodies in Spain. For total phosphate, highest mean loads were modelled for water bodies in Cyprus and Greece, while Spain again showed lowest values. The results for the Slovenian and Croatian water bodies had only limited diagnostic value due to the very low number of temporary water bodies reported (< 6). In Portugal and Italy, about 50% of the floodplains of temporary streams are used by agriculture, while the lowest shares are found for Croatia with about 18% (but the number of water bodies reported is limited in Croatia). Share of urban land use is on average much lower and rarely exceeds 10%. Spanish and Cyprus’ temporary water bodies are most affected by urban land use.

3.4 Stressor association to the overall ecological status and BQEs

The ordinal model captured around 20-40% of the variability in ecological status by the seven stressors, but with different magnitudes, variability, and directions (Fig. 3 and Tab S2 for the estimates). Agriculture had the strongest negative association with the overall ecological status and with the status of all four BQEs. The likelihood of high status declined strongly already at a low percentage of agricultural land cover, while the likelihood of moderate to bad status classes increased (exception: macrophytes, Fig. 3). Urban land use was the second stressor with strong negative associtions to overall status, benthic invertebrates and also phytobenthos. In contrast, TN was positively associated with macrophytes (Fig. S1), i.e. high amounts of TN were positively related to the ecological status as well with benthic invertebrates and overall ecological status. However, in case of the macrophytes and fish, uncertainty of the independent variables was large, as reflected by the credibility intervals (Fig. 3) and the hypothetical outcome plots (Fig. 4D-E). Surprisingly, TP was positively, albeit weakly, associated to the quality classes of the individual BQEs (Fig. 3) and also the boxplots showed this weak and uncertain association (Fig. S2).

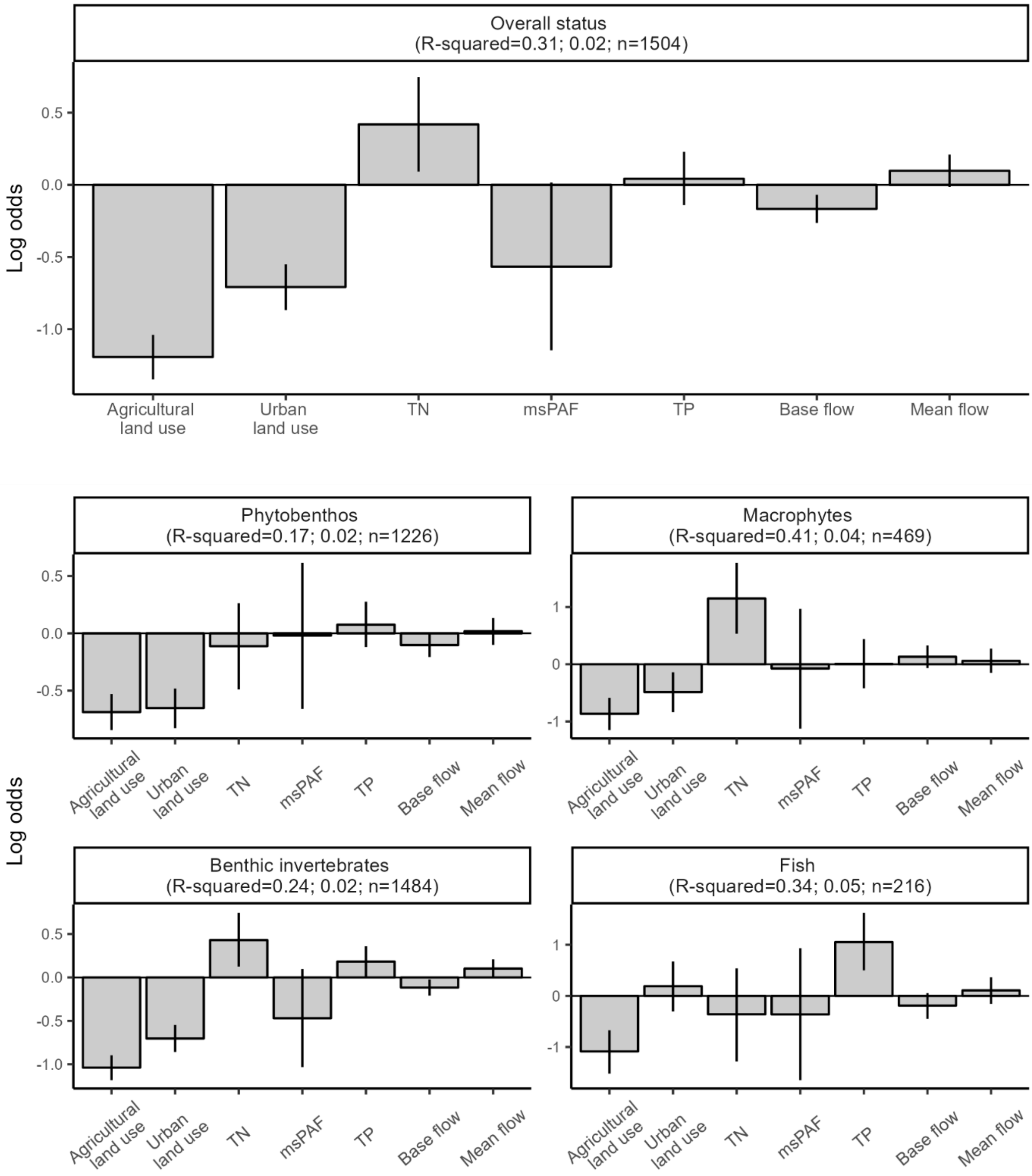

Figure 3. Log odds (regression coefficients) of each ordinal model for the overall ecological status and the status for the individual BQEs. The vertical bars represent the credibility intervals at 95%.

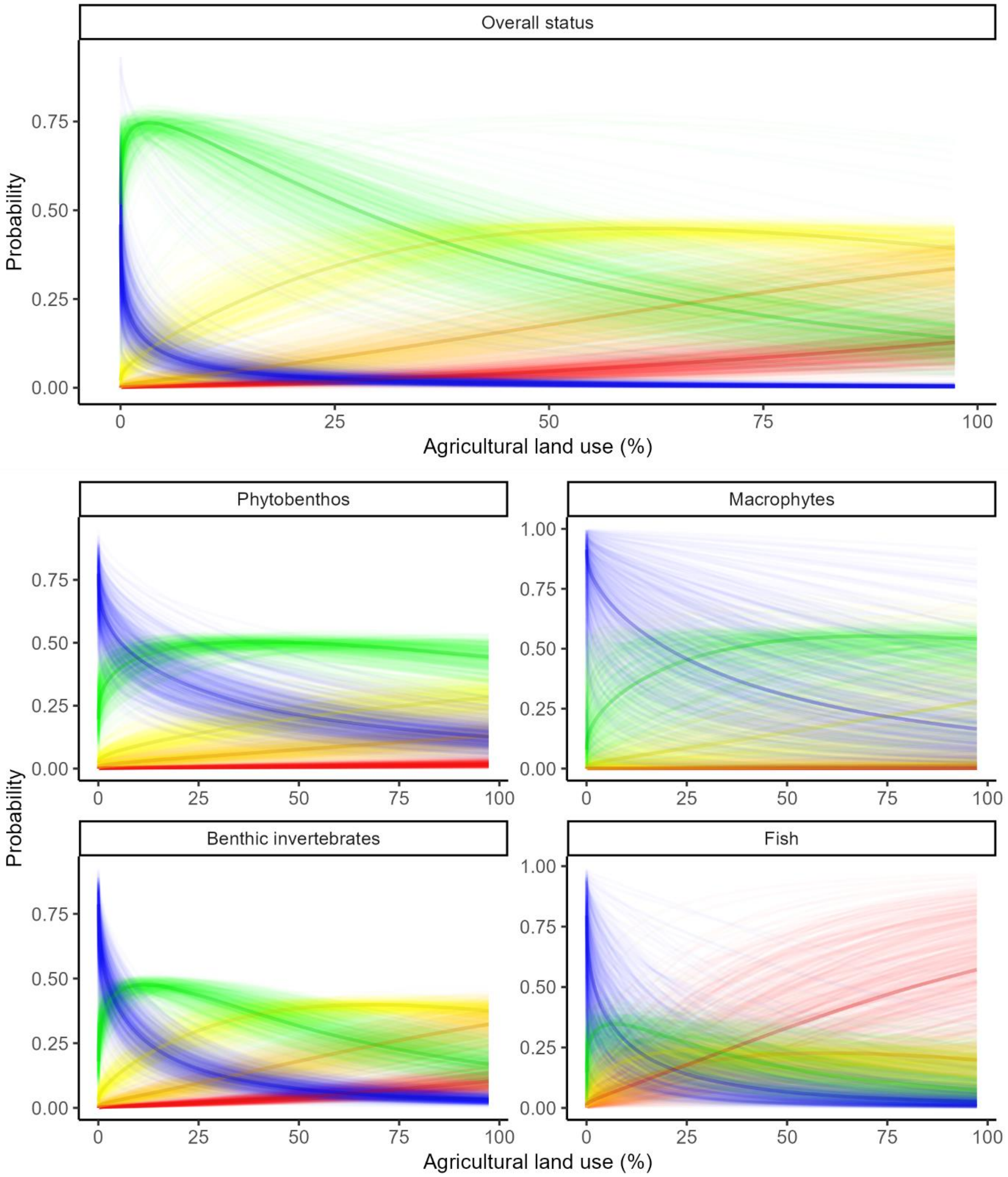

Figure 4. Hypothetical outcome plots for the overall ecological status and for the individual BQEs along the gradient of agricultural land use. The colours indicate the different ecological status classes as given in Fig. 2.

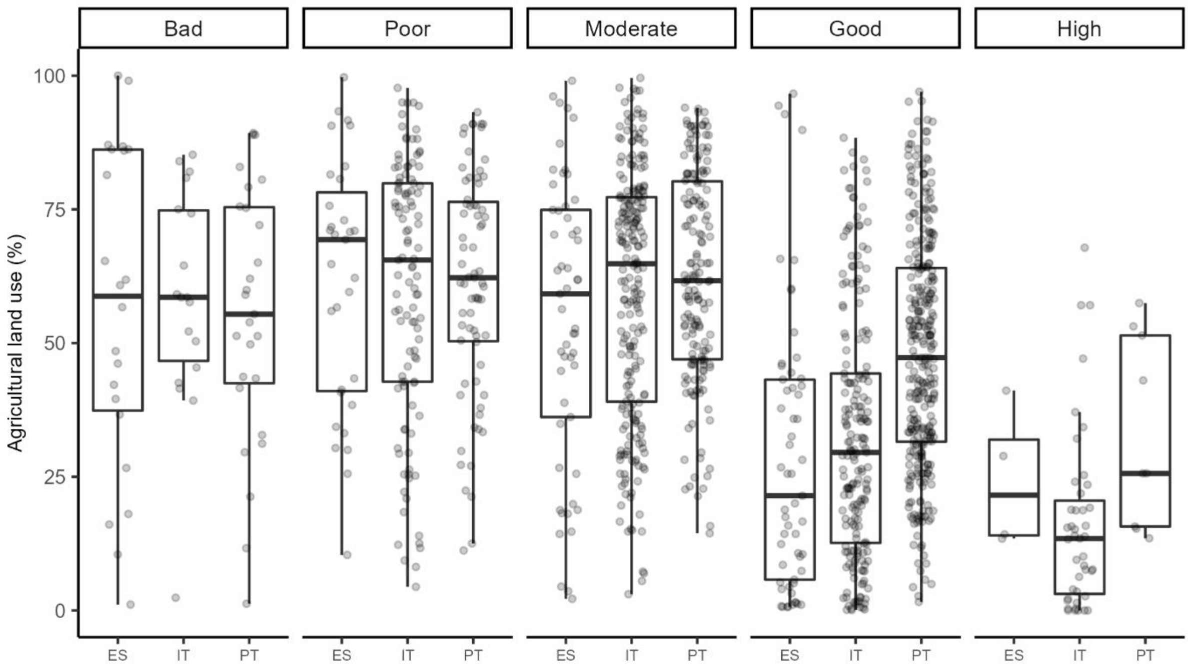

Figure 5. Overall ecological status of temporary water bodies in relation to the percentage of agricultural land use in the three countries with more than 150 assessed water bodies. The horizontal line in the box indicates the median.

The detailed country-specific analysis of status classes in relation to agricultural land use in the floodplain (Fig. 5) revealed an average share of 25% (Spain, Italy) to 50% (Portugal) associated with good overall ecological status, while for moderate to bad status the average percentages of agricultural land were well above 50%. This result is mirrored by the status derived with phytobenthos (Fig. 6) and benthic invertebrates (Fig. 8). Also here, Spanish and Italian water bodies showed distinctly lower percentages of agricultural land use for water bodies with high or good BQE status as compared to Portuguese water bodies. Nonetheless, with increasing agricultural land use the status classes of benthic invertebrates and phytobenthos deteriorated, which is particularly obvious in Italy and Spain. For macrophytes, TN was the most important stressor (Fig. 7), showing moderate to bad ecological status for higher levels of TN in Spain. Suprisingly, in Portugal it displayed an unexpected decreasing

TN trend with increasing deterioration. The average TN levels showed a wide range between countries and status classes. For fish (Fig. S3), the most important stressor was agricultural land use, which displayed a slight but continuous degradation with increasing agricultural land use in all three countries.

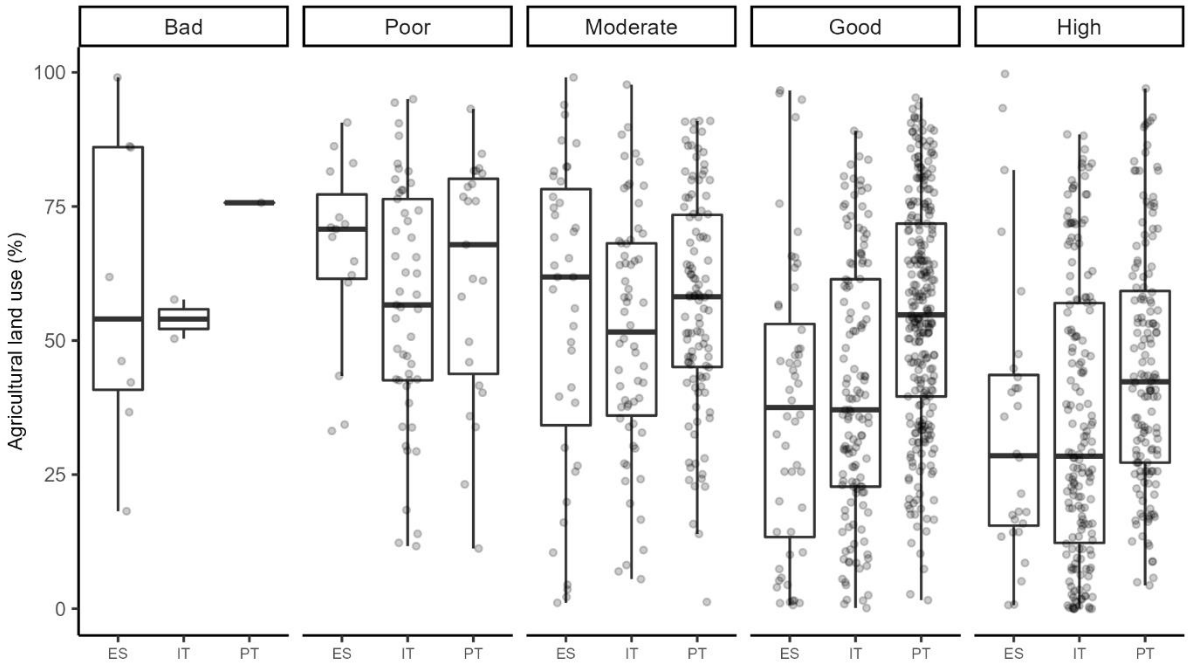

Figure 6. Ecological status of temporary water bodies derived with phytobenthos in relation to the percentage of agricultural land use, in the three countries with more than 150 assessed water bodies. The horizontal line in the boxes indicates the median.

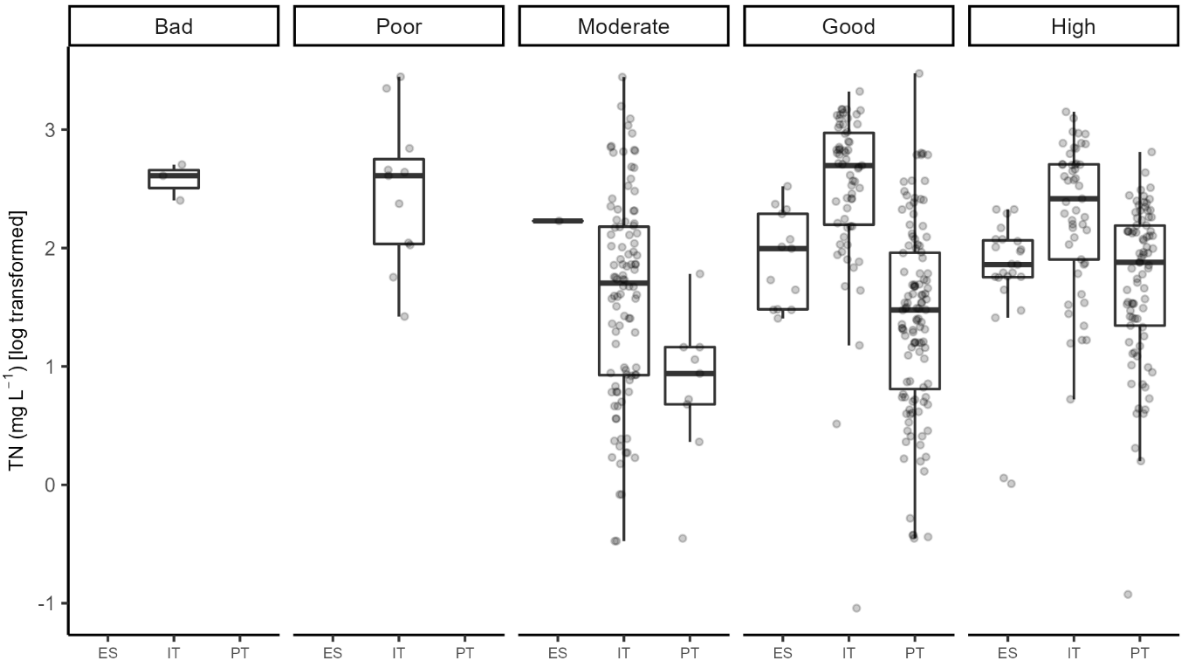

Figure 7. Ecological status of temporary water bodies derived with macrophytes in relation to TN, in the three countries with more than 150 assessed water bodies. The horizontal line in the boxes indicates the median.

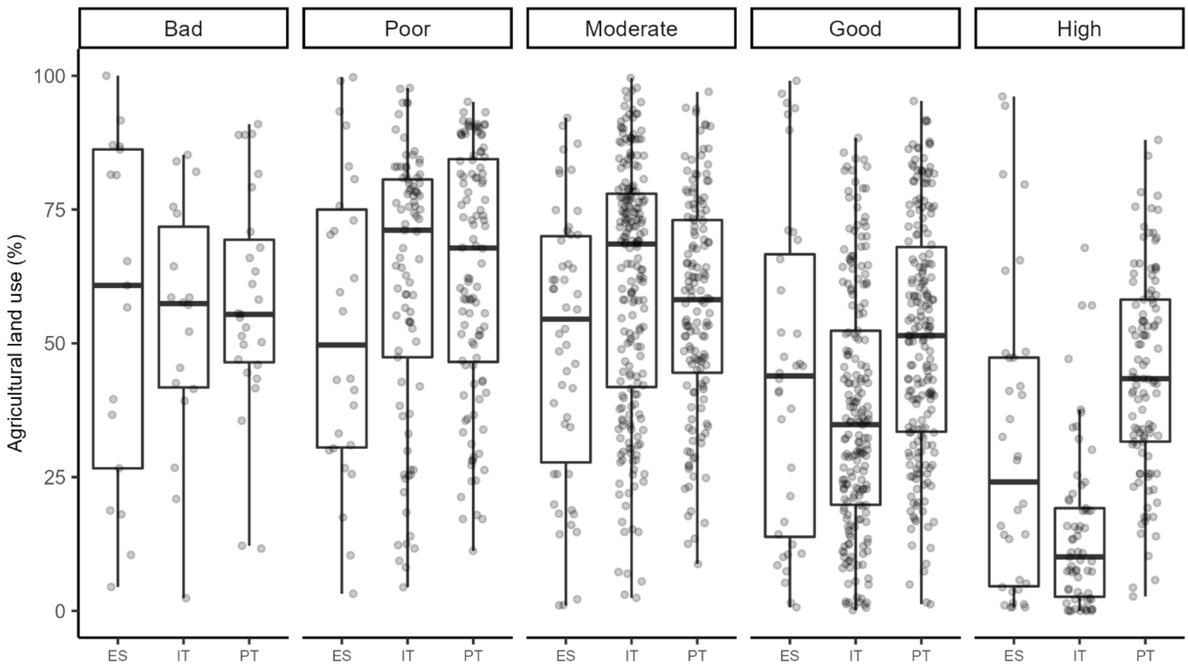

Figure 8. Ecological status of temporary water bodies derived with benthic invertebrates in relation to the percentage of agricultural land use, in the three countries with more than 150 assessed water bodies. The horizontal line in the boxes indicates the median.

4 Discussion

4.1 Status and distribution of temporary streams in the Mediterranean basin area

Our analysis revealed that 46% of the temporary water bodies in the Mediterranean basin EU Member States were reported to be in a good or high ecological status, while another third (34%) was assessed as moderate. This is comparable to the distribution of status classes of permanent stream types in the respective countries (EEA, 2018).

Our analysis was based on 1504 water bodies, for which the ecological status class for at least one BQE was reported. However, the assessment of many of these water bodies was based on a single or on two BQEs only, most often benthic invertebrates. Accordingly, the overall ecological status and the status derived from benthic invertebrates were very similar, both in terms of the distribution of classes and in their relation to stressors (Fig. 3). However, there were striking differences between BQEs and between countries. Based on phytobenthos and macrophytes, the vast majority of sites was in a high (29% phytobenthos, 34% macrophytes) or good ecological status (46% phytobenthos, 41% macrophytes), while the percentage was lower for benthic invertebrates (14% high, 34% good) and fish (15% high, 19% good). In general, it is easier for plants to survive times of desiccation and to re-establish populations once the flow has re-started again, while this is most difficult for fish and hololimnic benthic invertebrates. This might be a reason for the assessment differences between BQEs. With the exception of the Spanish region of Catalonia (Munne et al., 2021), none of the countries used assessment systems specifically designed for temporary water bodies, but simply used the methods applied for perennial systems. Class boundaries, however, were different: for temporary rivers, the assessment systems for benthic invertebrates and phytobenthos were intercalibrated (intercalibration type R-M5), leading to status class boundaries different from those for perennial Mediterranean rivers. Nevertheless, several authors question assessment systems designed for perennial waters to be appropriate for temporary water bodies (Soria et al., 2020; Prat et al., 2014; Sánchez-Montoya et al., 2011). Obviously, drying acts as an overarching stressor on the biota of perennial streams, while it is a natural disturbance in temporary water bodies, to which the biota is adapted. Applying methods developed for perennial water bodies to temporary rivers may therefore harvest misleading results (Soria et al., 2020). Furthermore, it has been shown that assessment results may differ greatly depending on the timing of sampling. We assume that this is most pronounced in case of benthic invertebrates and fish. Fish, in particular, depend obviously on the permanent presence of water, which makes fish diversity in temporary streams extremely variable through space and time.

The overall status of temporary water bodies in Spain, Greece and Croatia was generally lower than in the other countries. The rationale for these differences remains speculative, but may be related to three causes. First, these regions are particularly vulnerable to water scarcity and extreme weather events (Gao et al., 2006). Second, the average number of pressures reported to the EU by the Member States was distinctively higher in Spain and Croatia posing multiple threats to the different BQEs. Third, Spain, Italy and Portugal assessed a comparatively large number of temporary water bodies with several BQEs. Therefore, the “one out-all out principle” may have affected the assessment most strongly, as many streams have also been sampled for fish, which were generally assessed most negatively. The assessment results of temporary rivers for benthic invertebrates and phytobenthos, however, have been intercalibrated between countries, so the class boundaries should be comparable (Poikane et al., 2014). Nonetheless, e.g. in the case of diatoms ecoregional-specific indices could refine status assessment and determine more accurately ecological status (Çelekli et al., 2022).

Besides the lack of assessment systems specifically designed for temporary rivers and the low number of BQEs sampled in most water bodies, it is obvious that only a small fraction of the temporary water bodies in the Mediterranean basin was included into the monitoring programmes. This has diverse causes, such as the historic focus on perennial water bodies and that intermittent rivers and streams were not considered “real” water bodies (Datry et al., 2014; Garcia et al., 2014). For example, France does not monitor temporary water bodies at all, although in the French Mediterranean coastal area many streams and rivers are temporary (Datry, 2012). The situation is even more difficult in ephemeral streams, which pose challenges for sampling, due to the short aquatic time spans and which are often only hydrogeomorphologically assessed, e.g. in Spain (Munne et al., 2021). In general, this reflects that each EU Member State has its own river typology which may or may not take into account perennial/temporary distinction, which in turn impairs in parts the comparability of data and results between Member States.

4.2 Pressures and stressors associated with temporary streams in the Mediterranean area

We used two data sources on pressures and stressors: The opinion-based pressure reporting of local water managers to the WISE-WFD database, and model results of seven stressors on the level of Functional Elementary Catchments (Lemm et al., 2021). Both data sources led to comparable results. The main stressors according to Lemm et al. (2021) associated with the overall ecological status and the different

BQEs were agricultural land use (Erba et al., 2015) and TN, followed by urban land use. This corresponds to the main pressures reported by experts to the WISE-WFD database that list diffuse agriculture sources, other diffuse sources and urban point sources of wastewater as the main pressures. The causal linkages between the reported pressures and the modelled stressors are obvious. The pressure 'diffuse agricultural pollution' is reflected by the modelled main stressors 'agricultural land use' and 'total nitrogen'. The other main reported pressure, 'point source pollution by urban waste water', is contributing to the modelled stressor 'total phosphorus' and reflects additionally the stressor 'urban land use'. Thus, opinion-based pressure reporting is well mirrored by modelled stressor intensity.

4.3 Relationship between stressors and ecological status

The analysis of pressures and stressors leads to the conclusion that urban stressors, such as waste water run-off, and stressors related to agriculture, such as fertilizers, are the main causes for status deterioration of temporary Mediterranean rivers, while hydrological stress and toxic substances seemed to have a lower impact. The question remains if the assessment systems are well reflecting the latter types of stressors in these temporary water bodies (Prat et al., 2014; Sanchez-Montoya et al., 2011; Sanchez-Montoya et al., 2009). Nonetheless, at least for benthic invertebrates, the stressor msPAF (toxic pressure indicator; De Zwart & Posthuma, 2005) had a clear impact on the assessment results. Temporary water bodies are characterized by distinct hydrological features, in particular regular drying, while drying would be a severe stressor effect in perennial water bodies. Biota of temporary water bodies may be better adapted to the drying-associated physico-chemical and hydrological conditions than those of perennial water bodies, thus explaining the low influence of the modelled stressors ‘alteration of base flow’ and ‘alteration of low flow’. Again, it needs to be underlined that, with the exception of the indices intercalibrated under intercalibration type R-M5, the assessment systems may have used incorrect benchmarks that are suited for perennial, but less suited for temporary water bodies. The specific situation of ceasing flow and increasing concentration of ions, nutrients and toxic substances is less relevant for perennial systems (Sánchez-Montoya et al., 2011). Furthermore, complete desiccation of river beds is a major problem for hololimnic species (fish and many benthic invertebrates). Thus, indicator taxa of assessment systems which rely on constant water availability, cannot necessarily survive in temporary rivers. Their absence may influence assessment results for temporary rivers, when perennial river assessment system indicators are applied (Arias-Real et al., 2022; Soria et al., 2020; Acuña et al., 2005).

Not all pressures were negatively associated with the ecological status for the individual BQEs. Surprisingly, the status for macrophytes was higher with increasing TN. As nitrogen (and phosphorus) support plant growth, medium levels of these nutrients might enhance plant abundance and diversity and thus ecological status. Some national assessment systems seem to account for this nutrient driven increase in diversity.

5 Conclusion

The results of status assessment inadequately reflect the deterioration of temporary water bodies in the Mediterranean region. First, temporary water bodies are poorly represented in the monitoring programmes. An Europe-wide consistent definition, cartographical delineation, standardized sampling and reporting is absolutely necessary. Second, the application of assessment systems that were developed for perennial water bodies to temporary water bodies has various shortcomings, as different taxa, particularly invertebrates and fish, occur in temporary water bodies are not adequately considered in the assessment systems. Thus, recent advancements in stress indication and the development of new specific metrics like functional redundancy and response diversity (Soria et al., 2020) or trait-based approaches focussing on drying-resistent taxa (Arias-Real et al., 2022) or their dispersal and recovery capacities (Arenas-Sanchez et al., 2021) should be urgently incorporated into assessment systems. Additionally, metacommunity approaches (Cid et al., 2020) could also aid assessment purposes. This finally should be incorporated into an intercalibration exercise for all BQEs in temporary streams and rivers. Third, the hydrologic peculiarity (i.e. extrem low flows, complete drying, or even artificial flow permanency caused by waste water effluents) impact the biota of temporary water bodies in many ways causing additional stress, such as increases in concentrations of nutrients or toxic substances (Sabater et al., 2018). Therefore, adequate ecoregional specific metrics for each organism group (e.g. Çelekli et al., 2022; Tornés et al., 2021) should be developed to determine the ecological status of water bodies. This finally would aid and guide the management and conservation of this common stream type in the Mediterranean basin (Munné et al. 2021).

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Tables

Table 1. Number of temporary water bodies and prevailing pressures in Mediterranean EU Member States as reported to the EU WISE data base.

Country code	**EU member state**	**No. temporary water bodies**	**Mean number of pressures**	**Main pressures**
CY	Cyprus	101	1.51	Diffuse others, diffuse agriculture, abstraction agriculture
EL	Greece	53	1.32	Diffuse agriculture, point source non IED plants
ES	Spain	167	5.73	Diffuse agriculture, point urban wastewater, abstraction
HR	Croatia	4	6.25	Physical alteration of channel bed riparian area, diffuse agriculture
IT	Italy	583	2.57	Diffuse agriculture, diffuse others, diffuse urban, point contaminated sites
PT	Portugal	591	1.28	Diffuse agriculture, diffuse others, point source urban wastewater
SL	Slovenia	5	2.20	Anthropogenic pressure unknown, point urban waste water
Total		**1504**	**2.31**	**Diffuse agriculture, diffuse others, point urban waste water**

Figure captions

Figure 1. Distribution of temporary water bodies in the Mediterranean basin as reported to the WISE-WFD data base of the 2nd river basin management plan.

Figure 2. Overall status and status of the four BQEs in temporary Mediterranean basin water bodies per EU Member State. In brackets the number of water bodies.

Figure 3. Log odds (regression coefficients) of each ordinal model for the overall ecological status and the status for the individual BQEs. The vertical bars represent the credibility intervals at 95%.

Figure 4. Hypothetical outcome plots for the overall ecological status and for the individual BQEs along the gradient of agricultural land use. The colours indicate the different ecological status classes as given in Fig 2.

Figure 5. Overall ecological status of temporary water bodies in relation to the percentage of agricultural land use, in the three countries with more than 150 assessed water bodies. The horizontal line in the box indicate the median.

Figure 6. Ecological status of temporary water bodies derived with phytobenthos in relation to the percentage of agricultural land use, in the three countries with more than 150 assessed water bodies. The horizontal lines in the boxes indicate the median.

Figure 7. Ecological status of temporary water bodies derived with macrophytes in relation to TN, in the three countries with more than 150 assessed water bodies. The horizontal lines in the boxes indicate the median.

Figure 8. Ecological status of temporary water bodies derived with benthic invertebrates in relation to the percentage of agricultural land use, in the three countries with more than 150 assessed water bodies. The horizontal lines in the boxes indicate the median.