Multiple stressor effects on biodiversity and ecosystem functioning in a Mediterranean temporary river


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HIGHLIGHTS
• Testing biodiversity-ecosystem functioning (B-EF) relationship in a temporary river
• Resource use efficiency (RUE) decreased with increased diatom diversity.
• Organic matter breakdown (OMB) increased with increased macroinvertebrate diversity.
• Different mechanisms drive the B-EF relationship in benthic communities.
• RUE and OMB decrease with pollution and intermittency following community structure.

GRAPHICAL ABSTRACT

ABSTRACT
The hydrological and biological complexity of temporary rivers as well as their importance in providing goods and services is increasingly recognized, as much as it is the vulnerability of the biotic communities in view of climate change and increased anthropogenic pressures. However, the effects of flow intermittency (resulting from both seasonal variations and rising hydrological pressure) and pollution on biodiversity and ecosystem functioning have been overlooked in these ecosystems. We explore the way multiple stressors affect biodiversity and ecosystem functioning, as well as the biodiversity-ecosystem functioning (B-EF) relationship in a Mediterranean temporary river. We measured diversity of benthic communities (i.e. diatoms and macroinvertebrates) and related ecosystem processes (i.e. resource use efficiency-RUE and organic matter breakdown-OMB) across a pollution and flow intermittency gradient. Our results showed decreases in macroinvertebrate diversity and the opposite trend in diatom assemblages, whereas ecosystem functioning was negatively affected by both pollution and flow intermittency. The explored B-EF relationships showed contrasting results: RUE decreased with higher diatom diversity, whereas OMB increased with increased macroinvertebrate diversity. The different responses suggest contrasting operating mechanisms, selection effects possibly driving the B-EF relationship in diatoms and complementarity effects driving the B-EF relationship in macroinvertebrates. The understanding of multiple
1. Introduction

Temporary rivers are dynamic and diverse ecosystems, very common in the Mediterranean Basin (McDonough et al., 2011; Acuña et al., 2014). Biodiversity in temporary rivers is lower than in perennial rivers, with native communities able to tolerate natural flow intermittency, but sensitive to anthropogenic disturbances and extreme climatic events (Datry et al., 2014; Skoulikidis et al., 2017). Biodiversity losses because of global change, is a rising threat in these systems, with certain taxonomic groups that are key for ecosystem functioning being more vulnerable (e.g. Tilman et al., 2012; Soria et al., 2017). Even though recent studies have indicated the negative effects of flow intermittency and anthropogenic pollution on biodiversity (e.g. Arenas-Sánchez et al., 2016; Karaozus et al., 2018a), ecosystem functioning (e.g. Acuña et al., 2015) and the supporting goods and services (Datry et al., 2017) separately, there has been less effort in studying potential biodiversity–ecosystem functioning (B-EF) relationships in these ecosystems. The provision of ecosystem services in temporary rivers is being challenged by multiple stressor effects on biodiversity, and this may transmute to ecosystem functioning through the B-EF relationship.

Ecosystem functioning encompasses the processes and properties of a given ecosystem (Fraîre, 2013), and primary production and organic matter breakdown are fundamental components of ecosystem functioning (Lindeman, 1942; Odum, 1956). These processes are driven by abiotic factors such as temperature, hydrology, and light and nutrient availability (von Schiller et al., 2017), but also by biotic ones such as diversity of biological communities (Gessner et al., 2010). These natural environmental factors add to anthropogenic factors such as agriculture and urban sewage, the two being important contributors of organic matter, nutrients and pollutants, including pesticides, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs; Karaozus et al., 2011a; Tzoraki et al., 2015). The intermittent character of temporary Mediterranean rivers allow the accumulation of these pollutants in the sediments (Zoppini et al., 2014, 2016) during periods of zero flow, suggesting that multiple stressors may act together in these systems, affecting ecosystem functioning. As an indication of effects on ecosystem processes, primary production in the biofilm is reduced during non-flow events, particularly in the presence of pollutants (Corcoll et al., 2015). Organic matter breakdown also decreases with water stress (Abril et al., 2016), as well as with increased inputs of toxic chemicals such as heavy metals and pesticides (e.g. Schäfer et al., 2007). In fact, the decrease in leaf litter breakdown has been often associated to the effects of multiple stressors on detritivore macroinvertebrate communities, a key group of macroinvertebrate consumers (e.g. Graça, 2001; Monroy et al., 2016).

Although stream ecology has provided evidence of B-EF relationships (Lecerf and Richardson, 2010), these relationships have focused mainly on benthic macroinvertebrates and their ability to breakdown organic matter (e.g. Chauvet et al., 2016), whereas ecosystem processes related to primary producers in biofilms have been related to the overall metabolism and nutrient use (Cardinale, 2011; Guasch and Sabater, 1995). Biodiversity is related to ecosystem processes not only through the number of species present in a community, but also through their interactions and their specific traits. It is thus critical to consider biodiversity in the context of species richness as well as of community structure and function (McKie et al., 2008; Fraîre et al., 2014), when aiming to detect the mechanisms underlying the B-EF relationship: complementarity effects (i.e. complementary use of resources by the species) can result in a positive B-EF relationship due to differences in species traits, whereas selection effects (i.e. the presence of a species that produces high biomass) could result in a positive B-EF relationship due to increased dominance of the most productive species (e.g. Hillebrand et al., 2008; Truchy et al., 2015).

Our aim was to explore in which manner multiple stressors affected biodiversity and ecosystem functioning as well as their relationship. We performed our research in a Mediterranean temporary basin, focusing on benthic stream communities (diatoms and macroinvertebrates) which were submitted to different degrees of pollution and flow intermittency. We therefore analysed diatom assemblages and macroinvertebrate communities’ composition from four streams in the Evrotas River basin, which covered a pollution and flow intermittency gradient. We further related these communities to resource use efficiency (RUE) of primary producers in biofilms and to organic matter breakdown (OMB) by macroinvertebrates. We tested for (i) the relationships between biodiversity vs. pollution and flow intermittency, (ii) the relationships between the recorded ecosystem processes vs. pollution and flow intermittency, and (iii) the form of the B-EF relationships in the two stream benthic communities as well as the possible mechanisms underpinning them.

2. Materials and methods

2.1. Study area

The Evrotas river basin (South Peloponnese, Greece) is a medium-sized (2.418 km²), mid-altitude (150–600 m) Mediterranean basin, with numerous ephemeral and intermittent streams discharging into the main stem of the river. Climate is typical Mediterranean with hot and dry summers (26.2 °C, average temperature June–August 2009–2016) and cool, wet winters (9.1 °C, average temperature December–February 2009–2016), with annual precipitation varying markedly from year to year; it follows however a predictable seasonal pattern, with most of the rainfall occurring during the months of October through March, with very few rain episodes during summer (Karaozus et al., 2018b). The river basin’s landscape is covered mainly by seminatural areas (61% of the total river basin), with agricultural areas coverage being 38% and urban areas approximately 1%. The dominant anthropogenic pressures are overexploitation of water resources, agro-industrial wastewater discharges, diffuse agrochemical pollution and domestic wastewaters from the municipal treatment plant (Karaozus et al., 2018b). The geographical, geological, hydrological and ecological features of the Evrotsas basin are described in detail elsewhere (Skoulikidis et al., 2011; Kalogianni et al., 2017; Karaozus et al., 2018b).

2.2. Sampling design

We studied 4 streams in the Evrotsas river basin, named here A1 to A4 (Fig. 1), selected because they form a gradient of flow intermittency (i.e. number of days of zero flow). The streams, together with the effect of diffused pollution from anthropogenic stressors, such as agriculture, receive point source pollution from the operation of olive-oil mills. The olive-oil mills operated during a three-month period in winter, causing an acute stress to the biotic communities of the affected sites. Thus, to further account for the effect of the effluents from olive oil mills, in each stream we selected a site upstream from the olive oil mill and another immediately downstream. Each site covered approximately 20 m length and the upstream and downstream sites were located approximately 20 m from the olive oil mill effluent. We sampled all the sites in October 2016, before the onset of olive oil mill...
operation, January 2017, when the mills were under operation, and March 2017, one month after the end of oil mills operation. When sites were dry, no sampling was performed. Flow intermittency was calculated for each site based on the total number of days of zero flow during the study period (October 2016–March 2017), using data from automatic HOBO sensors that continuously recorded conductivity (that was zero at zero flow) and temperature. Details related to flow intermittency are provided in Table S1.

2.3. Chemical analyses

At each site, approximately 8 kg of the uppermost 10 cm sediment layer were collected from 4 to 6 points along the left and right river banks. Sediments were wet-sieved (2 mm mesh) using river water, to remove large debris and animals. Samples were then placed in polyethylene bottles, frozen, and transported on dry ice to the laboratory. In each sampling site, an unsieved subsample was collected for phenol analysis, but was otherwise subjected to the same procedure with the rest of the sediment samples.

Sediment samples were analysed for legacy persistent organic pollutants (POPs), pesticides, phenolic compounds, organic carbon (OC) and total nitrogen (TN). POPs tested included the 16 polycyclic aromatic hydrocarbons regulated by the United States Environmental Protection Agency (Keith, 2015), the polychlorinated biphenyls (PCBs) 18, 28, 31, 52, 44, 101, 149, 118, 153, 138, 180, 180, 170 and 194, and the semi-volatile organochlorine compounds hexachlorobenzene (HCB), gamma hexachlorocyclohexane (g-HCH), DDT and related compounds (DDX). Analysis of the lyophilized sediment samples was performed with a method based on pressurized liquid extraction (PLE), SPE clean-up, and analysis by LC-MS/MS (Köck-Schulmeyer et al., 2013). The list of target pesticides (above 50) included compounds (and a few breakdown products (BPs)) belonging to different chemical classes (phenylureas, triazines, organophosphates, anilides, thiocarbamates, chloroacetanilides, acidic herbicides, benzo- and halo-organodiazines, quinolones, neonicotinoids).

For the determination of phenolic compounds, approximately 20 g of the sediment was sub-sampled and organic matter was separated using ultrasonic extraction. A mixture of methanol and water was used, followed by sonication of the resulted suspension and centrifugation. The clarified extracts were collected, condensed and subsequently extracted with the 4 aminoantipyrine method (Ettinger et al., 1951). Total phenols were measured photometrically, using a Perkin-Elmer 25 Lambda spectrophotometer at a wavelength of 460 nm (Danis and Albanis, 1996; Alonso et al., 1998; Czaplicka, 2001). Organic Carbon (OC) and Total Nitrogen (TN) in the sediment were measured with a CHN analyzer, Thermo Scientific Flash 2000, following Cutter and Radford-Knoery (1991) and Verardo et al. (1990). Water samples were filtered through 0.45 μm membrane filters. Nitrite (NO2, mg L−1) and orthophosphate (PO4, mg L−1) concentrations were determined by a Skalar San++ Continuous Flow Analyzer, ammonia (NH4, mg L−1) concentration was determined using a Skalar Automatic Analyzer, whereas nitrate (NO3, mg L−1) concentration was determined using both Ion Chromatography and a Skalar Automatic Analyzer.

All pollutants detected in the sediments (pesticides, PAHs, PCBs, organochlorides, phenols, OC and TN) were log- or arcsin- (percentages) transformed, and each one was regressed against ecosystem processes (Table S2). Pollutants that significantly affected an ecosystem process (Table S2 in bold) were further used in a Principal Component Analysis (PCA), thus summarized into one variable that determined
the pollution gradient. The pollution gradient was reflected in the first PC axis that summarized the total variability of chemical pollutants (see Results for details). To detect any possible relationship between pollution and intermittency, Spearman correlation analysis was performed between the two stressors.

2.4. Biota

Diatom sampling followed the standard EU protocol (EN 13946, 2003). Submerged cobbles (5–7 per site) were collected from areas with sufficient light and brushed to obtain biofilm material. In the laboratory, samples were digested with hydrogen peroxide to remove organic matter, and cleaned material was prepared (using Naphrax) in permanent slides for microscopical examination. Up to 400 diatom valves were counted and identified at the species or subspecies level in each sample, using light microscopy (Nikon Eclipse 80i, Tokyo, Japan) with Nomarski differential interference contrast optics at 1000× magnification.

Macroinvertebrate collection followed the STAR-AQEM methodology (AQEM Consortium, 2002). Twenty subsamples were collected at each sampling site using a 25 cm × 25 cm square hand net with a 500-μm mesh size nytex screen. Each of the 20 subsamples was taken by positioning the net and disturbing the substrate in an area that equals the square of the frame width upstream of the net (25 × 25 cm). Thus, a total of 1.25 m² (25 × 25 × 20 replicates) was sampled from each sampling site. Subsamples were preserved in ethanol until the experiment. Upon retrieval, the sticks were rinsed with tap water, dried whereas others were on the dry bed for part of the duration of the experiment, some of the sticks remained permanently submerged, whereas others were on the dry bed for part of the duration of the experiment. Upon retrieval, the sticks were rinsed with tap water, dried (70 °C, 72 h) and weighed to determine initial dry mass before deployment in the river. The sticks were incubated at each site for 45–60 days. During the experiment, some of the sticks remained permanently submerged, whereas others were on the dry bed for part of the duration of the experiment. Upon retrieval, the sticks were rinsed with tap water, dried (70 °C, 72 h), weighed and asched (500 °C, 5 h) to determine final ash-free dry mass. Leaching of sticks was simulated in the laboratory, and initial ash content determined to correct initial dry masses. Breakdown rates were calculated according to the negative exponential model (Petersen and Cummins, 1974). This was further expressed in degree-days, to correct for the effect of temperature.

2.6. Statistical analysis

To test the effect of pollution we used the first axis (PC1) of the performed PCA with the chemical organic variables as the summary variable of pollution - see Results for details), and we related it to biodiversity and ecosystem functioning. The flow intermittency (Days of zero flow) was also related to biodiversity and ecosystem functioning. We applied linear mixed-effects models to account for any variation attributed to time of sampling (random effect in the models) on these relationships. The effect of each variable was tested using likelihood-ratio tests (LRTs). Statistical analysis was carried out in R v.3.3.3 (R Core Team, 2017) and package lme4 v.1.1-12 (Bates et al., 2015) was used for mixed effect models.

3. Results

The first components of the PCA performed with the chemical pollutants accounted for 75.6% of the total variance (PC1 = 58.2% and PC2 = 17.4% - Fig. 2). However, only PC1 was found to be significantly correlated to the two ecosystem processes (Table S2) and was thus further used in the analyses. PC2 summarized pollution by a pesticide, but it did not significantly affect any of the two ecosystem processes measured (Table S2). Negative values of PC1 corresponded to high pollution levels (Fig. 2a), and flow intermittency was negatively correlated to PC1 (Spearman coefficient r = −0.71, p < 0.05), indicating that the most intermittent sites were also the most polluted ones (Fig. 2b – pollution is represented as the inverse of PC1 scores, therefore the higher the number in the y-axis, the higher the pollution level).
the case of diatom functional evenness. The diversity indices of the two biotic groups than to functional diversity indices, except in the case of diatom functional evenness. The diversity indices significantly related to RUE in diatoms were Shannon diversity, Pielou’s Evenness, Simpson dominance and Functional Evenness (Table S4). Specifically, RUE decreased with higher diversity, evenness and functional evenness (Fig. 5), whereas it increased with increased Simpson dominance. Structural indices of macroinvertebrates related to OMB were Shannon diversity and Simpson dominance (Table S4). Organic matter breakdown increased with higher macroinvertebrate diversity and decreased with higher values of dominance (Fig. 5). Overall, the two biotic groups tested presented an opposite trend in B-EF relationships, decreasing in the case of diatoms and increasing in the case of macroinvertebrates. The Shannon diversity of macroinvertebrates was positively correlated with the percentage of grazers (Spearman r = 0.60, p < 0.05) and shredders (Spearman r = 0.51, p < 0.05), the two feeding groups that could contribute more to OMB.

4. Discussion

For consistency, only results from Shannon and Simpson indices that explain different aspects of community structure are presented in the figures of the main text and indices that presented significant relationships with stressors and ecosystem processes are discussed. Results from all the applied indices are presented in Table S3.

3.1. Effects of pollution and flow intermittency on biodiversity

Species richness, Shannon diversity and Functional richness of the diatom assemblages were significantly increased with pollution (as summarized by PC1 - Table S3, Fig. 3a, b). Species richness, Shannon diversity and Functional richness of the macroinvertebrate communities significantly decreased, whereas Simpson dominance significantly increased with increased pollution (as summarized by PC1) (Table S3, Fig. 3c, d).

Species richness, Shannon diversity, Pielou’s Evenness and Functional richness of diatoms assemblages significantly increased with increasing days of zero flow (Table S3), while Simpson dominance presented a non-significant decreasing trend with increasing days of zero flow (Table S3, Fig. 3e, f). In contrast, Species richness, Shannon diversity and Functional richness of macroinvertebrate communities significantly decreased with increasing days of zero flow, whereas Simpson dominance presented an increasing, but not significant, trend (Table S3, Fig. 3g, h).

3.2. Effect of use efficiency (RUE) decreased with increased pollution (LRT: χ² = 6.09, DF = 1, p < 0.05, Fig. 4a). This trend was mainly driven by the increased RUE in the P-limited sites A1 and A2. Organic matter breakdown (OMB) also decreased with increased pollution (LRT: χ² = 16.77, DF = 1, p < 0.001, Fig. 4c). RUE decreased with increased number of days of zero flow (LRT: χ² = 9.38, DF = 4, p < 0.01 - Fig. 4b). The two most intermittent sites (A3 and A4) were also N-limited and presented the lowest RUE. OMB also decreased with increased number of days of zero flow (LRT: χ² = 7.31, DF = 4, p < 0.01 - Fig. 4d).

3.3. Biodiversity-ecosystem functioning relationships

Ecosystem processes responded better to structural diversity indices of the two biotic groups than to functional diversity indices, except in the case of diatom functional evenness. The diversity indices significantly related to RUE in diatoms were Shannon diversity, Pielou’s Evenness, Simpson dominance and Functional Evenness (Table S4). Specifically, RUE decreased with higher diversity, evenness and functional evenness (Fig. 5), whereas it increased with increased Simpson dominance. Structural indices of macroinvertebrates related to OMB were Shannon diversity and Simpson dominance (Table S4). Organic matter breakdown increased with higher macroinvertebrate diversity and decreased with higher values of dominance (Fig. 5). Overall, the two biotic groups tested presented an opposite trend in B-EF relationships, decreasing in the case of diatoms and increasing in the case of macroinvertebrates. The Shannon diversity of macroinvertebrates was positively correlated with the percentage of grazers (Spearman r = 0.60, p < 0.05) and shredders (Spearman r = 0.51, p < 0.05), the two feeding groups that could contribute more to OMB.

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4. Discussion

Here we assessed the effect of multiple stressors on the structure of diatom assemblages and macroinvertebrate communities in temporary streams. We also assessed the response of ecosystem processes that are related to these two biotic groups. We observed that even though the biological communities in temporary ecosystems are naturally adapted to a certain level of water stress, they are particularly fragile to anthropogenic pollution and further water reduction due to global change. Disentangling the effects of these co-occurring multiple stressors (i.e. flow intermittency and pollution), is complicated by the accumulation of pollutants in the sediments during dry periods (Zoppini et al., 2014, 2016). Indeed, sediment pollution and flow intermittency were strongly related in our study sites, pollution recording higher levels in the more intermittent sites, where lower dilution, lengthier pollutant transport and lower self-purification processes occurred (Karaouzas et al., 2011b; Mandaric et al., 2018).

The two studied stressors (i.e. sediment pollution and flow intermittency) affected structural and functional diversity of diatoms and macroinvertebrates in a different manner. Effects were negative on macroinvertebrate diversity but positive on diatom diversity. The negative effect of pollution on macroinvertebrate communities has been reported to intensify during low flows, with pollution sensitive taxa being almost extirpated (Kalogianni et al., 2017). The combination of the two stressors could reduce macroinvertebrate diversity and increase dominance at the same time, as community structure changes from rheo- to limnophilic taxa, leading to dominance of functional groups.
associated with these stressors (e.g. Bonada et al., 2007; García-Roger et al., 2013; Karaouzas et al., 2018c). On the other hand, the positive response of diatom assemblages could be related to the reported unimodal relationships between diversity and pollution (e.g. Pandey et al., 2017). Our streams did not range throughout the whole pollution gradient but rather between the low to intermediate pollution parts of the curve. Further, flow intermittency did not affect species richness but rather the structure of the assemblage by favoring the weakly attached species (i.e. high profile species), and thus biofilm evenness (Passy, 2007). Biofilms were thicker, composed by both low and high profile species, which occupied more available niche space (Ponsati et al., 2016; Torresi et al., 2016). The thicker biofilm formation also coincides with higher pollution and high flow intermittency, resulting in increased accumulation of organic matter (Passy, 2007). Thicker biofilms impeded efficient light penetration and nutrient sequestration, resulting in lower photosynthetic activity and RUE (Burns and Ryder, 2001).

The two studied ecosystem processes responded similarly to both pollution and flow intermittency; both stressors reduced RUE and OMB, in agreement to previous studies. Indeed, other experimental and field studies have also shown the decrease of primary productivity with increasing days of zero flow (Acuña et al., 2015; Timoner et al., 2014). Primary production is directly linked to RUE: the presence of primary producers that use more efficiently the resources, converting them to biomass, leads to increased primary production in the biofilm (Ptacnik et al., 2008). Pollution could result in a more pronounced response, probably as a result of the additive effect it produces with water stress on autotrophic processes in biofilms (Corcoll et al., 2015), as well as on macroinvertebrate communities (Karaouzas et al., 2018c). Organic matter breakdown could be related to certain macroinvertebrate groups (i.e. detritivores) but also to the presence of microbes, especially bacteria and fungi (Gessner and Chauvet, 2002; Tank et al., 2010; Chauvet et al., 2016). Microbial activity is an important factor in OMB, especially in wood decomposition (Bilby, 2003); however, it was not accounted for in the present study. Nevertheless, invertebrates have been shown to also play a role in wood decomposition (Collier and Halliday, 2000; Elosegi et al., 2007), and our data (Elosegi, unpublished) shows decomposition of tongue depressors as those used in the present

Fig. 3. Relationship between pollution (PC1) and diversity indices (a–d) and flow intermittency and diversity indices (e–h). a,b,e,f correspond to diatom indices, c,d,g,h correspond to macroinvertebrate indices. Pollution is represented by the inverse of the scores of axis 1 of the PCA (-PC1 score) and flow intermittency by the number of days of zero flow.
study to be more correlated to total than to microbial breakdown. Water stress affects shredders and detritus feeders and could thus result in decreased leaf litter breakdown (Monroy et al., 2016; Karaouzas et al., 2018c). Furthermore, many shredders and detritus feeders are regarded as pollution-sensitive taxa, thus increased pollution levels could also affect OMB (Young et al., 2008; Woodward et al., 2012; Bundschuh and McKie, 2016; Kalogianni et al., 2017).

The ecosystem processes we tested showed significant responses to diversity indices other than species richness, suggesting an important effect of community structure to ecosystem functioning. The relationship between ecosystem functioning and evenness (or dominance) remains equivocal (Hillebrand et al., 2008), and this was reflected in our contrasting results between the two studied biological groups. Although macroinvertebrates showed a positive B-EF relationship (i.e. the more diverse the community, the more intense was OMB), diatoms presented a negative diversity-evenness relationship (i.e. the more species in the assemblage and the more evenly distributed, the less efficient was the resource use). These apparently contradictory results could be explained by either the different drivers that rule each community assembly (e.g. environmental constraints, species interactions, different trophic level, different response to stressors) or by the different mechanisms at play in the two communities (i.e. selection vs. complementarity effect). Different drivers in our study could be the multiple stressors tested and the different response of the two groups, suggesting that the B-EF relationship could be driven by the stressor effects on biodiversity. On the other hand, B-EF theory has proposed two mechanisms to explain the increased productivity of communities following an increase in species richness: selection effect, which occurs when increased species richness leads to increased probability of having a species that yields high biomass (Aarssen, 1997), and complementarity effect.
which occurs when more species increase their productivity through a complementary use of resources (Hector, 1998). These two mechanisms satisfactorily explain contradictions in plant communities, where the evenness (dominance)-ecosystem functioning relationship has been more extensively studied (Polley et al., 2003; Hillebrand et al., 2008).

In the diatom assemblages, high dominance in the least polluted, perennial stream (i.e. A1), resulted in low diversity (Shannon index) and high RUE. This increased RUE could be related to grazing and shredding insects (Jonsson and Malmqvist, 2000; Huryn et al., 2002). Indeed, increased Shannon diversity in our results is consistent with increased percentages of grazers and shredders, suggesting that these groups could be enhancing OM.

5. Conclusions

Multiple stressors acting together in temporary rivers affect both biodiversity and ecosystem functioning. Ecosystem functioning is mainly driven by interactions between species and between species and their environment, the latter being increasingly affected by co-occurring stressors. These ecosystems are very vulnerable to climate change and anthropogenic stressors that could cause the loss of species and thus the loss of ecosystem functioning, with implications on the ecosystem goods and services they provide. This study tested the effect of anthropogenic pollution and flow intermittency on biodiversity and ecosystem functioning in temporary rivers, and established the B-EF relationships in two biotic groups (i.e. diatoms and macroinvertebrates). Even though it is a relatively small-scale study, our results suggest that ecosystem processes are driven by community structure in both groups, positively for macroinvertebrates but negatively for diatoms, suggesting different mechanisms underlying the observed trends. Future work could focus on larger-scale studies and more ecosystem processes to further establish observed relationships that could provide better insight on the protection of these vulnerable ecosystems.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.08.105.

References


