Diatom responses to sewage inputs and hydrological alteration in Mediterranean streams

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1. Introduction

Algal assemblages in rivers are potentially affected by multiple stressors, including chemical contamination, irradiance excess and high water temperatures, as well as hydrological alterations. Mediterranean streams are naturally characterized by periods of hydrological stability, only interrupted by autumnal floods and intermittency during the summer period (Lake, 2003; Sabater and Tockner, 2010). Most climate change scenarios predict low-intermittency during the summer period (Lake, 2003; Sabater and Tockner, 2010). The impact sites had high concentrations of ammonium, phosphorus, and pharmaceutical compounds (antibiotics, analgesics, and anti-inflammatory drugs), particularly in those receiving untreated sewage. Impact sites had a higher proportion of teratological forms as well as a prevalence of diatom taxa tolerant to pollution. The differences in the diatom assemblage composition between the paired C and I sites were the largest in the impacted sites that received untreated sewage inputs as well as in the systems with lower dilution capacity. In these sites, the diatom assemblage was composed by a few pollution-tolerant species. Mediterranean river systems facing hydrological stress are highly sensitive to chemical contamination, leading to the homogenization of their diatom assemblages.

We analyzed the conjoint effects of sewage inputs and hydrological alteration on the occurrence of teratological forms and on the assemblage composition of stream benthic diatoms. The study was performed in 11 Mediterranean streams which received treated or untreated urban sewage (Impact sites, I), whose composition and morphological anomalies were compared to upstream unaffected (Control, C) sites. The impact sites had high concentrations of ammonium, phosphorus, and pharmaceutical compounds (antibiotics, analgesics, and anti-inflammatory drugs), particularly in those receiving untreated sewage. Impact sites had a higher proportion of teratological forms as well as a prevalence of diatom taxa tolerant to pollution. The differences in the diatom assemblage composition between the paired C and I sites were the largest in the impacted sites that received untreated sewage inputs as well as in the systems with lower dilution capacity. In these sites, the diatom assemblage was composed by a few pollution-tolerant species. Mediterranean river systems facing hydrological stress are highly sensitive to chemical contamination, leading to the homogenization of their diatom assemblages.
Some of the sites received treated effluents from WWTP while others directly received untreated urban sewage (Table 1). In the streams, we observed a moderate increase to the basal water flow (summer) and higher water flow (spring) conditions. Water depth, velocity, and instant discharge were measured at each sampling campaign with an acoustic Doppler velocity meter (ADV, Flow Tracker, SonTek Handheld-AD™, P-4077). Water pH, dissolved oxygen, and electrical conductivity were measured in situ using hand-held probes at each sampling campaign (WTW, Weilheim, Germany). One water sample for nutrient analyses (nitrate (NO₃, µg N L⁻¹), nitrite (NO₂, µg N L⁻¹), ammonium (NH₄, µg N L⁻¹) and phosphate (PO₄, µg P L⁻¹)), and dissolved organic carbon (DOC, mg L⁻¹) were collected at each site, filtered in 0.7 µm GF/F filters (Whatman Int. Ltd., Maidstone, UK) and kept at −20 °C until analysis. Phosphate concentration was determined colorimetrically using a spectrophotometer (Alliance-AMS Smartchem 140, AMS, Firepillon, France), after Murphy and Riley (Murphy and Riley, 1962). Nitrite, nitrate and ammonium concentrations were determined on a Dionex ICS-5000 ion chromatograph (Dionex Co., Sunnyvale, USA; Hach, 2002). DOC concentrations were determined on a Shimadzu TOC-V CSH coupled to a TNM-1 module (Shimadzu Co., Kyoto, Japan). DOC results were only available for the second sampling campaign, and therefore were not used in the statistical analyses (see below).

Pharmaceutical products were assumed to be the dominant microcontaminants given the urban sources of wastewaters. Their continuous discharge into the aquatic environment makes the PhACs pseudo-persistent contaminants, potentially able to cause adverse effects on living organisms and the environment (Daughton and Ternes, 1999). The PhACs analysis in water samples was conducted following the method developed by Gros et al. (2012). Briefly, the analyses were carried out with an off-line solid phase extraction (SPE) followed by ultra-high-performance liquid chromatography coupled to triple quadrupole linear ion trap tandem mass spectrometry (UHPLC-QqLIT-MS²). Chromatographic separations were carried out with a Waters Acquity Ultra-Performance™ liquid chromatography system, coupled to a 5500 QTRAP hybrid triple quadrupole-linear ion trap mass spectrometer (Applied Biosystems, Foster City, CA, USA) with a turbo ion Spray source. Quantification was carried out by isotope dilution. Finally, all data were acquired and processed using Analyst 1.5.1 software, while quantification was carried out by isotope dilution. Detailed information regarding chemicals and reagents used, as well as method performance parameters of target compounds including limits of detections (LODs), limits of quantifications (LOQs) and recovery rates are described in detail in Mandaric et al. (2018).

### 2.2. Physical and chemical measurements

We conducted sampling surveys during early summer (June) of 2015 and spring (April) of 2016. These two sampling periods respectively covered lower-water flow (summer) and higher-water flow (spring) conditions. Water depth, velocity, and instantaneous discharge were measured at each sampling campaign with an acoustic Doppler velocity meter (ADV, Flow Tracker, SonTek Handheld-AD™, P-4077). Water pH, dissolved oxygen, and electrical conductivity were measured in situ using hand-held probes at each sampling campaign (WTW, Weilheim, Germany). One water sample for nutrient analyses (nitrate (NO₃, µg N L⁻¹), nitrite (NO₂, µg N L⁻¹), ammonium (NH₄, µg N L⁻¹) and phosphate (PO₄, µg P L⁻¹)), and dissolved organic carbon (DOC, mg L⁻¹) were collected at each site, filtered in 0.7 µm GF/F filters (Whatman Int. Ltd., Maidstone, UK) and kept at −20 °C until analysis. Phosphate concentration was determined colorimetrically using a spectrophotometer (Alliance-AMS Smartchem 140, AMS, Firepillon, France), after Murphy and Riley (Murphy and Riley, 1962). Nitrite, nitrate and ammonium concentrations were determined on a Dionex ICS-5000 ion chromatograph (Dionex Co., Sunnyvale, USA; Hach, 2002). DOC concentrations were determined on a Shimadzu TOC-V CSH coupled to a TNM-1 module (Shimadzu Co., Kyoto, Japan). DOC results were only available for the second sampling campaign, and therefore were not used in the statistical analyses (see below).

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and ranked on the following scale: 0.1 (presence), 1 (≤5%), 2 (>5 to ≤25%), 3 (>25 to ≤50%), 4 (>50 to ≤75%) and 5 (>75%). Reference books used for the determination of non-diatom algae and cyanobacteria were those of John et al. (2011) and Wehr and Sheath (2003).

The diatom analysis was performed after cleaning diatom frustules from organic matter. This cleaning was performed in boiling hydrogen peroxide, and cleaned frustules were mounted on permanent slides using Naphrax (r.i. 1.74; Brunel Microscopes Ltd., Chippenham, Wiltshire, UK). Up to 400 valves were counted on each slide by performing random transects under light microscopy (Nikon Eclipse 80i, Tokyo, Japan) using Nomarski differential interference contrast optics at a magnification of 1000×. Diatom taxa were identified according to reference floras (Krammer and Lange-Bertalot 1991a,b), Krammer and Lange-Bertalot (1991a,b, 1997a,b), Hofmann et al., 2011, and complemented through the monographs series of “Diatoms of Europe” and “Bibliotheca Diatomologica”. We inspected the samples for the presence of teratological forms for each diatom taxa. Teratological forms are non-adaptive phenotypic abnormalities usually involving anomalous valve outlines or modification in their striation patterns (Falasco et al., 2009a). The teratological forms were recorded and quantified in both C and I sites, and summarized in a single value when different types of deformities occurred for a given taxon. For all types of teratologies we considered from slight to marked teratologies (Fig. S5) following Cantonati et al. (2014) and Lavoie et al. (2017).

2.4. Data analysis

The chemical variability at the sites (C and I) was explored by means of a Principal Components Analysis (PCA). PCA allows summarizing the variables data provided the Principal Components can be identified as descriptors of a given gradient. Candidate physical and chemical variables were nutrient concentrations, conductivity, dissolved oxygen and PhACs concentrations (Tables 1 and S1). Pearson’s correlation was performed between the candidate variables, and those strongly correlated to each other (correlation coefficient was >0.8) were unselected to avoid multicollinearity. This resulted in the final selection of phosphate, ammonium, nitrate, conductivity (EC), dissolved oxygen (DO), analgesics/anti-inflammatories and antibiotics. Concentrations of

Fig. 1. Map of the sampling locations. Prades (Pr), Bisbal F. (Bi), Poboleda (Po), Maella (Ma), Valderoures (Va), Nonasp (No), Caseres (Ca), Bot Gandesa (BG), Bot Canaleta (BC), Corbera d’Ebre (Co), Prat de Compte (PC).
replaced by a value equal to one-half of the method detection or quantification limit. Before performing the PCA the variables were normalized by subtracting the mean and dividing by the standard deviation.

The order of the diatom assemblage composition was performed using the complete diatom taxa list for all the sites, so forth avoiding arbitrary decisions on removal of rare taxa; downweighting the contributions of rare taxa is an inherent property of the computation of Bray-Curtis similarities (Capone and Kushlan, 1991; Hansen and Ramm, 1994). We analyzed the spatial and temporal patterns of the diatom assemblages’ structure by means of Pearson correlations that specifically highlight linear relationships. Tolerance to pollution and desiccation of the different diatom species was defined using Van Dam et al. (1994).

The diatom assemblages (Bray-Curtis) similarities (derived from the diatom POC) and the chemical (Euclidean) distances (derived from the chemical variables included in the POC) were related to each other. The relationship between the chemical and diatom assemblages in the C and I sites for each location was used to define the degree of change in diatom assemblages composition against the degree of change in the chemical variability. AIC (Akaike Information Index) and RSE (Residual Standard Error) were used to select the best fitted regression curves of the diatoms against the chemical characteristics.

The significance of the occurrence frequency of teratological forms in the I sites was tested using a PERMANOVA. The PERMANOVA was based on the Bray–Curtis similarity scores. The PERMANOVA operates on a resemblance matrix and it is similar to traditional parametric MANOVA (Anderson, 2001). Since C and I pair sites were not independent, we considered in the design the factors location and site (control or impact) nested in location as decreasing relationships of individual taxa across the plot, instead of Pearson correlations that specifically highlight linear relationships. Tolerance to pollution and desiccation of the different diatom species were defined using Van Dam et al. (1994).

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random factors.

The different PCA, PCO, and PERMANOVA (999 permutations) were performed with PRIMER-E 6 v.6.1.11 and PERMANOVA + v.1.0.1 (PRIMER-E Ltd., Plymouth, UK). Analyses were carried out with fourth-root transformed diatom data further converted into a resemblance matrix using Bray–Curtis similarity. Environmental data (except those expressed as ranked variables) were logarithmically transformed \((x + 1)\) before analyses to reduce skewed distributions. Pearson correlations were calculated using the software package IBM SPSS version 21 (IBM Corporation, Armonk, NY, USA). Regressions were conducted using R version 3.4.2 (R Development Core Team, 2013) applying the package nlme.

3. Results

3.1. Physical and chemical characteristics

The sites ranged from 0.1 to 16 m width and between 1 and 400 L/s water flow (Table 1). Discharge was lower during summer 2015 (below annual mean discharge) and higher in spring 2016 (higher than the annual mean for each river). Most of the sites showed high water conductivity (EC), particularly Bot Gandesa, Bot Canaleta, Corbera d’Ebre and Prat de Compte. Oxygen concentrations in the C sites averaged 9.1 ± 2.1 mg/L, and nutrient concentrations in these sites were moderate, particularly regarding ammonium (average 5 ± 6 µg/L) and reactive phosphorus (average 4 ± 4 µg/L), but higher in nitrate (average 2.6 ± 1.6 mg/L). As an exception, the C site of Corbera had high P concentrations in the two sampling occasions (427–704 µg/L). DOC concentrations in the second sampling campaign were low to moderate (0.81–4 mg/L) in the C sites, except in Corbera which also had high DOC concentrations (5.51 mg/L). The I sites had lower oxygen concentration (6.8 ± 2.6 mg/L), and much higher nutrient concentrations (1330 ± 1830 µg/L ammonium, 204 ± 301 µg/L reactive phosphorus, 2.2 ± 2.6 mg/L nitrate). The DOC in the I sites (second sampling campaign) ranged from moderate (Poboleda, 1.67 mg/L) to very high concentrations (10.4 mg/L Bot Gandesa and 11.7 mg/L Corbera d’Ebre).

A total of 76 PhACs, belonging to 15 therapeutic groups (Table S1), were detected in the water samples. The highest concentrations of PhACs were of analgesics and anti-inflammatory drugs, antihypertensives, lipid regulators and cholesterol lowering statin drugs, diuretics, psychiatric drugs and antibiotics. Concentrations of all PhACs moved from 6.53 to 7.92 ng/L in the C sites to up to 151.54 ± 45.98 ng/L for analgesics and anti-inflammatory drugs on treated-sewage WWTP effluents. PhACs in I sites receiving untreated wastewater discharges reached maximum average concentrations of 1.92 ± 1.38 µg/L for analgesics and anti-inflammatory drugs, and 0.14 ± 0.1 µg/L for antihypertensives (Table S1).

The PCA performed with the selected chemical variables (Fig. 2) explained 45.6% of the total variation in its first component. This axis separated C sites (located on the right part) from I sites (expanded from the center towards the left of the graph). Phosphate, ammonium, analgesics/anti-inflammatory drugs and antibiotics were the variables that contributed most to this axis. Some C sites (Corbera d’Ebre) were located closer to the impact sites because of their higher concentrations of phosphates and nitrates. The second axis (17.7% of the total variation) accounted for the relevance of nitrate, EC and DO, the sampling locations of Corbera d’Ebre, Maella, Prat de Compte and Casesers having the highest values, and opposite to the locations of Bishal de Falset, Prades and Poboleda.

3.2. Non-diatom algal and cyanobacterial composition

Up to 37 non-diatom taxa were determined in the studied streams (Table S2). While most of them occurred both in C and I sites, a few were exclusive of ones or the others. Euglena sp. and Phacus sp. were only recorded in I sites. The cyanobacteria Rhabdoderma sp., Rhabdogloea sp. and Rivularia sp. were present in C sites, while the rodophytes Bangia sp., Audouinella sp. and Batrachospermum sp. were recorded in the less illuminated sites of the C sites. Coccoid and filamentous cyanobacteria (Oscillatoria sp., Chroococcus sp., Merismopedia sp., Lyngbya sp., Gloeocapsa sp.) were moderately abundant in C sites, and some of them dominated in the I sites. The zygmenates Mougeotia sp., Zygmena sp. and Spirogyra sp. preferentially occurred in the C sites of the wider river systems (Valderoures, Caseres, Nonasp), but also occurred in the I sites. The filamentous green algae Stigeoclonium and Ulothrix occurred mostly in the I sites. The proteobacteria Sphaerotilus natans was present in some C sites but produced large masses in several I sites.

3.3. Diatom assemblage composition

Diatoms were the most abundant (60–100%) component of the algal assemblage in all the studied locations and sites. A total of 231 diatom taxa were recorded (Table S3). The dominant taxa in the C sites were Achnanthidium minutissimum, Achnanthidium lineare and Encyonopsis minuta, while Amphora pediculus, Navicula veneta and Achnanthidium minutissimum were the most abundant taxa and occurred in 80% of the I sites.

Teratological forms occurred in both C and I sites but were more frequent in the I sites (Fig. 3; PERMANOVA; pseudo-F1,120 = 1.723, \(P = 0.049\)). Five types of Falasco et al. (2009a) teratologies were found in the studied streams (Fig. 4). There was a clear dominance of those affecting valve outline, followed by changes in striaion pattern, raphe canal modifications and valves with more than one type of teratology (mixed type). The taxa Sellaphora nigrig, Achnanthidium minutissimum, Craticula subminucula, Nitzschia frustulum and Planoditium frequentissimum accounted for most of the occurring teratological forms. The presence of the teratological forms could not be attributed to any particular location (PERMANOVA; pseudo-F1,120 = 0.798, \(P = 0.701\)).

The PCO which analyzed the ordination of the diatom assemblages separated those in the C and I sites in the first axis (24.9% of total variation; Fig. 5 and Table S4). Encyonopsis spp., Achnanthidium spp., Comphenoma lateripunctatum and Sellaphora stromiwi were separated from Mayamaea atomus var. permitis, Planoditium frequentissimum, Sellaphora seminulum, Craticula subminucula, Nitzschia inconspicua and Comphenoma parvulum according to their Spearman correlations with the PCO axes. The former were characteristic of the C sites, while the latter were dominant in the I sites. In some of the C sites (Corbera, Poboleda) pollution tolerant diatom taxa were quite abundant, and they were arranged close to their respective I sites (Fig. 5).

The second axis of the PCO (15.1% of the total variance) did not provide any biologically meaningful ordination of the diatom taxa. However, the third axis (8.6% of total variation) separated the taxa for their hydrological preferences. Taxa regularly recorded in temporary streams (Van Dam et al., 1994) (i.e. Amphora indistincta, Amphora ovalis, Nitzschia frustulum, Simonsenia delognei and Navicula tripunctata) were arranged on the upper part of the axis, while Cymbella excisa, Nitzschia fonticola, Cocconeis pediculus, Fusiuliera saprophila and Navicula reichardtiana were arranged on the lower side of the axis and they are taxa mainly occurring in permanent watercourses (Van Dam et al., 1994). The samples from the two sampling campaigns were grouped together, indicating a small effect of the different hydrology of the two periods.

The difference in diatom assemblage composition in the I with respect to the C sites was calculated using the respective pairwise similarities accounted for the Bray–Curtis similarity matrix used in...
the PCO (Fig. 6). The I sites receiving treated effluents (i.e. Bisbal de Falset, Poboleda and Prades) had higher similarity to their respective C sites in the two sampling campaigns. The I sites receiving direct sewage inputs but having higher discharges (Vallderoures and Careses) also presented high similarity to their C sites. The site with the most polluted C samples (Corbera) also showed high similarities to their I samples. The locations with the lowest similarity (and those with the highest impact on their diatom assemblages) were Maella, Bot Gandesa, Nonasp, and Bot Canaleta.

4. Discussion

Overall, the studied set of Mediterranean streams received strong hydrological pressures as well as growing chemical contamination. The impact sites received high concentrations of dissolved organic carbon, ammonium, phosphorus, and pharmaceutical compounds (antibiotics, analgesics, and anti-inflammatories), particularly in those receiving untreated sewage. Concentrations of ammonium and total phosphorus were particularly high in the impact sites (maximum values of 6.64 mg/L and 1.1 mg/L respectively). Pharmaceutical compounds reached maximum concentrations ranging between 2.9 ng/L to 17.5 mg/L (Mandaric et al., 2018). These concentrations were within the range encountered in urban effluents (Gros et al., 2012). The effects of the hydrological and chemical stressors potentially affected the diatom cells as well as their assemblage composition, favoring the contribution of pollution-tolerant taxa.

The diatom taxa were distributed according to their preferences for resistance to desiccation and river size. This was indicated by the third axis of the multivariate ordination analysis (PCO), where taxa occurring in lower discharge (and smaller) sites were categorized as common inhabitants on wet and moist locations (Van Dam et al., 1994), though not as strictly terrestrial taxa. Amongst
these, Simonsenia delognei is frequent in exposed habitats (Witkowski et al., 2014), Amphora indistincta has been observed in spring-fed streams in Majorca (Delgado et al., 2013) and Sardinia (Lai et al., 2016), and Amphora ovalis is common in slow flowing rivers (Levkov, 2009) as well as in spring-fed streams in Majorca (Delgado et al., 2013). On the contrary, the taxa occurring in sites with higher discharges (and larger stream width) are not common in exposed habitats (Van Dam et al., 1994). Cymbella excisa and Nitzschia fonticola have been indicated as abundant in normal-flow and less common under low-flow conditions (Ishева and Ivanов, 2016). Navicula reichardtiana is common in large rivers (Gома et al., 2005) and in moderately hydrological stable habitats (Tornés and Ruhí, 2013). Other taxa are opportunistic, such as Fistulifera saprophila, which is able to thrive under highly to moderately stable hydrological conditions as well as in intermittent sites (Tornés and Ruhí, 2013). This diatom produces mucous films in large quantities (Lange-Bertalot, 2001) and this allows it to adapt to fast current conditions (Wendker, 1992).

Pollution caused, mostly in the small taxa of the genera Sellaphora, Craticula, Achnanthidium, Nitzschia and Planothidium, a higher frequency of deformities in their valves. Teratological diatom forms occur because of physical or chemical environmental stress (Lavoie et al., 2017). Unstable environmental conditions, such as wide changes in temperature, light irradiance, or moisture, favor the occurrence of teratological forms (Falasco et al., 2009b). The hydrological severity in our systems could account for valve deformities in some of the C sites, but chemical contamination made the difference between C and I sites and likely drove the higher proportion of deformities. Teratologic diatoms have been observed under different forms of pollution, such as heavy metal

Fig. 4. Left, type of teratologies and their occurrence in C and I sites. Right, examples of the different types of teratologies observed on: (a) Craticula subminuscula, (b) Planothidium frequentissimum, (c) Nitzschia frustulum, (d) Nitzschia amphibia, (e) Planothidium frequentissimum, (f) Navicula veneta, (g) Nitzschia fonticola, (h) Nitzschia inconspicua, (i) Gomphonema parvulum, (j) Sellaphora nigri. Scale bar = 10 μm.

Fig. 5. Principal coordinates analysis (PCO) using diatom assemblages’ composition of both control and impact sites, including vector overlay. To improve graphic display, overlay was restricted to include only variables with a vector length greater than 0.5 (See Table S3 for all Spearman correlations). The circle is a unit circle, whose relative size and centre is arbitrary with respect to the underlying plot. Each vector begins at the centre of the circle and ends at the coordinates (x,y) consisting of the correlations between that variable and each of the PCO axis respectively. Site codes correspond to those of Fig. 1 and Table 1. Numbers in sample labels represent sampling (1, June 2015; 2, April 2016). Taxa codes are given in Tables S2 and S3.
The effects we described on the structure, composition, and diatom deformities, as well as the accompanying results observed in the other algal groups, stress the importance of chemical pollution and altered discharge patterns on the ecological status of Mediterranean streams. The low water flow of Mediterranean streams enhances the potential ecological risk of chemical pollution. Our study revealed that fluvial systems facing hydrological stress, in the way currently occurring in Mediterranean streams, are highly sensitive to chemical contamination and may experience contamination (Morin et al., 2008). Even though a clear interpretation of the mechanisms and ecological implications of diatom valve deformities is still unclear (Lavoie et al., 2017), our observations suggest that urban pollution (high abundance of nutrients, organic matter, and PhACs) may reinforce the malformations already occurring under hydrological stress.

The multivariate analysis of the chemical variables indicated the relevance of ammonium, phosphate, antibiotics and analgesics/anti-inflammatories over the other chemical stressors. This chemical contamination likely caused the differences in the diatom assemblage composition between the paired C and I sites, since other environmental factors did not differ between impact and control sites. The differences between the paired C-I sites were the largest in those that received untreated sewage inputs. In those sites a few diatom taxa tolerant to organic and chemical pollution (Tornés et al., 2007) accounted for most of assemblage. Fig. 7 shows the relationship that can be built between the similarity of the diatom assemblages and their corresponding chemical change in the paired C-I sites. This relationship follows a negative power curve, where the higher similarity occurs in the sites showing the lowest chemical difference between C and I, including some C sites affected by upstream chemical pollution (which resembled in their composition to those of their I sites). The power expression also suggests a fast decrease in similarity between the paired C and I sites with higher chemical pollution affecting the impact. In fact, the diatom assemblage in the most polluted sites was largely similar irrespectively of their upstream composition, composed by a few pollution-tolerant species. Goldenberg Vilar et al. (2014) showed that eutrophication favored the decrease in species turnover and the increased homogenization of community composition. In our most polluted systems, the assembled chemical contamination of nutrients, organic matter and microcontaminants (PhACs) was driving the homogenization of the diatom assemblages.

The dilution capacities of the receiving watercourses (Petrovic et al., 2011) as well as the quality of the sewage entering the systems were determinants of the degree of effect in the diatom assemblage composition. A low dilution capacity not only favors higher contaminant concentrations, but also affects the architecture and function of benthic biofilms (Ponsati et al., 2016) because of its associated hydrological stability. In our studied set of streams the change in diatom composition between the C and I sites was less pronounced when the latter received treated effluents, and more accentuated when the effluents were untreated (Fig. 8). The potential relation to dilution is suggested by the slightly higher effect (lower similarity) in systems with lower water flow (i.e. lower dilution capacity) (Fig. 8). In the particular set of streams included in this study, where most of them were submitted to water resources exploitation, the extent of the difference between smaller and larger systems is however very limited. We suspect that this difference would be accentuated in rivers not submitted to such strong hydrological pressure than those in our study area.

Our results show that the confluence of chemical pollution (organic matter, nutrients and PhACs) and hydrological alteration enhances the effects of the former on the diatom assemblages. A study performed in temporary streams in Greece showed the prevailing effect of pollution over water stress on diatom assemblage composition (Karaouzas et al., 2018). We also observed that the effects of the two stressors were not as much apparent in non-diatom taxa and cyanobacteria than in diatoms, but some gross effects indeed occurred. Cyanobacteria were the second most abundant taxonomic group in the C sites probably because of their ability to adapt to hydrological alteration using their thick mucilage layer. However, green algae were more common in the I sites, where they showed higher resistance to chemical contamination. An analogous response in the response of algal groups has been observed in a separate experiment under simulated conditions of hydrological stress and pharmaceutical contamination (Serra-Compte et al., 2018).

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homogenization in their biological communities. The high susceptibility of these systems to the environmental impact of organic matter, micropollutants and nutrients stresses the relevance of preventing direct sewage inputs to rivers (Muñoz et al., 2009). Particular attention should be placed on adapting management decisions and existing metrics to co-occurring hydrological alteration and pollution (Karaouzas et al., 2018). Focusing on the combined hydrological alteration and chemical contamination is essential to predict potential ecological problems under future climate change scenarios, as well as an essential step to improve the conservation of these fluvial systems.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.envpol.2018.03.037.

References


