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The influence of natural flow regimes on macroinvertebrate assemblages in a semiarid Mediterranean basin.

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Abstract

The investigation of flow-ecology relationships constitutes the basis for the development of environmental flow criteria. The need to understand hydrology-ecology linkages in natural systems has increased due to the prospect of climate change and flow regime management, especially in water-scarce areas such as Mediterranean basins. Our research quantified the macroinvertebrate community response at family, genus and species level to natural flow regime dynamics in freshwater streams of a Mediterranean semiarid basin (Segura River, SE Spain), and identified the flow components that influence the composition and richness of biotic assemblages. Flow stability and minimum flows were the principal hydrological drivers of macroinvertebrate assemblages, whereas the magnitude of average and maximum flows had a limited effect. Perennial stable streams were characterised by flow sensitive lotic taxa (Ephemeroptera, Plecoptera, Tricoptera) and intermittent streams by predominately lentic taxa (Odonata, Coleoptera, Heteroptera and Diptera). Relatively minor biological changes were recorded for intermediate flow regime classes along a gradient of flow stability. Seasonal variation and minimum flows are key hydrological components that need to be considered for river management and environmental flows in the Segura River Basin and other Mediterranean basins. The anthropogenic modification of these parameters, due to both human activities and climate change, would probably lead to significant changes in the structure and composition of communities in perennial stable streams. This would be characterised by a reduction of flow sensitive EPT taxa and an increase in more resilient OCHD taxa.

Key words: natural flow regime, flow stability, minimum flows, macroinvertebrate composition, richness, Segura River Basin, semiarid Mediterranean streams

Introduction

The search for links between instream ecology and hydrology has become one of the fundamental issues in contemporary river science (Vaughan *et al*., 2009). Empirical investigation of regional flow-ecology relationships constitutes the basis for the development of environmental flow (e-flow) criteria (Arthington *et al*., 2006; Poff *et al*., 2010). In addition, the need to understand ecology-hydrology linkages in natural systems has been highlighted by the need to define reference conditions against which modified dynamics can be compared (Tockner *et al*., 2003). These needs are particularly pressing in the light of predicted climate change (European Environment Agency, 2008) and anthropogenic modification of natural flow regimes, especially in water-scarce areas such as Mediterranean basins.

Instream hydrological variability, encapsulating elements of the entire flow regime such as the daily, seasonal and annual patterns of discharge, the frequency, timing, predictability and duration of extreme flows (high and low), rates of change in discharge, and the magnitude of flows, are widely recognised as key ecological organizers in fluvial ecosystems (Richter *et al*., 1996; Poff *et al*., 1997; Hart and Finelli, 1999; Bunn and Arthington, 2002). Spatial variation of these characteristics is determined by variations in climate and mediated by basin geology, topography and vegetation (Winter, 2001). These hydrological and environmental factors influence the physical habitat for aquatic and riparian biota determining the conditions for reproduction and recruitment and affecting the availability of trophic resources, refuges during adverse situations and opportunities for dispersal (Naiman *et al*., 2008). Consequently, flow variability has strong ecological implications which shape the structure and function of riverine ecosystems from the local to regional scales, and from days (ecological effects) to millennia (evolutionary effects) (Lytle and Poff, 2004). It has been hypothesised that sites with similar hydrological characteristics should share similar faunal community composition, traits and ecosystem functioning (Poff and Ward, 1989). Therefore, as Arthington *et al*. (2006) and Poff *et al*. (2010) suggested, ecological responses of flow regimes to a given anthropogenic change should be broadly similar in rivers with similar natural flow regimes.

This hypothesis provides a powerful foundation to predict ecological responses to future flow regime changes, constituting the key element of a new holistic framework for developing scientifically-credible regional environmental flows: the ‘‘Ecological Limits of Hydrologic Alteration’’ (ELOHA) (Arthington *et al*., 2006; Kennard *et al*., 2010; Poff *et al*., 2010). Therefore, identifying and quantifying specific relationships between flow regimes and biological communities in undisturbed river ecosystems are essential steps to ensure sustainable river management (Arthington *et al*., 2006; Jowett and Biggs, 2009). Such relationships have been studied in general at the regional scale, using macroinvertebrates (e.g. Monk *et al*., 2006; Konrad *et al*., 2008; Kennen *et al*., 2010, Armanini *et al.*, 2011), fisheries (e.g Poff and Allan, 1995; Pegg and Pierce, 2002; Kennard *et al.*, 2007; Snelder *et al.*, 2009) or multiple taxonomic groups (e. g. Jowett and Duncan, 1990; Clausen and Biggs, 1997). However, the strength and nature of relationships between the flow regime and the biological assemblage vary depending on the geographical region, the floral or faunal group considered and the taxonomic resolution analysed.

In some areas, such as Mediterranean-climate regions, organisms have to withstand high intra and interannual hydrological variability, together with frequent natural flow extremes (floods and droughts) (Gasith and Resh, 1999). Species may respond over evolutionary time scales by developing morphological, physiological and/or life-history traits to bear such stresses (Poff *et al.*, 1997; Bonada *et al.*, 2007a; Bonada *et al.*, 2007b). Previous studies of Mediterranean streams (e. g. Bonada *et al.*, 2002; Jáimez-Cuéllar *et al.*, 2002; Vivas *et al.*, 2002; Bonada *et al.*, 2004; Mellado, 2005; Sanchez-Montoya *et al.*, 2007; Argyroudi *et al.*, 2009) as well as other semiarid areas (e. g. Boulton and Lake, 2008) have highlighted the importance of flow permanence on the composition and structure of macroinvertebrate communities. A progressive replacement of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa by Odonata, Coleoptera and Heteroptera (OCH) taxa has been reported as flow permanence decreases (Sánchez-Montoya *et al.*, 2007; Argyroudi *et al.*, 2009) or hydrological connectivity is reduced (Bonada *et al.*, 2006); although Diptera have also been associated with river sections with low or no flows and dominate lentic habitats in Southeast Spain (Vivas *et al.* 2002). Consequently, flow stability and hydrological extremes (especially low flows) are expected to be the most important components of Mediterranean flow regimes shaping instream assemblages, although its relative importance is still unclear.

The aim of this study was to quantify the effect of different flow regimes on macroinvertebrate communities. We utilised a dataset containing stream macroinvertebrate records at family, genus and species level across a semiarid Mediterranean region that encompasses a wide gradient of hydrological regimes (Belmar et al 2011) to test these predictions: (1) Flow stability and minimum flows should be the principal hydrological drivers of macroinvertebrate assemblage composition and richness; (2) an increase in the explanatory power of hydrology should occur as taxonomic resolution increases; and (3) a replacement of taxa should take place along a hydrological gradient from permanent streams with stable discharges to streams with high flow intermittence and flow variability. In general, a decrease in the percentage of flow sensitive Ephemeroptera, Plecoptera and Trichoptera families should occur as an increase in the percentage of more resilient Odonata, Coleoptera, Heteroptera and Diptera families takes place.

Methods

Study area

Located in the Southeast of Spain, the Segura River Basin drainage network, including coastal watercourses draining to the Mediterranean Sea, was selected as the study area. The management area of the Segura River Basin, one of the most arid zones of the Mediterranean region, includes watercourses with highly heterogeneous flow regimes. These water-bodies range from perennial rivers, with low seasonal and interannual flow variability, to highly seasonal ephemeral streams (Belmar *et al*., 2011). This variability is due to a strong climatic and altitudinal gradient from NW to SE, despite its relatively small size (18 870 km^2). Climate ranges from wet (>1 000 mm mean annual precipitation) and cold in the high elevation mountains of the NW (>1 000 m.a.s.l.) to semiarid and hot in the SE lowlands (< 350 mm mean annual precipitation). Mean annual temperatures range between 10 and 18 ºC (CHS, 2007). The lithology of the plains is characterised by limestone (karst) and Miocene and Triassic marls, with some small influences of volcanic strata. In contrast, calcites and dolomites dominate the mountainous headwaters. The vegetation is varied and ranges from Mediterranean conifer forests in the NW mountains to arid and semi-arid shrublands in the SE lowlands. This gradient in altitude and climate is coupled with an anthropogenic population density gradient. The river network has low population densities in the forested headwaters, intermediate densities in the agricultural midlands (with major flow regulation) and highly populated cities in the lowlands (Mellado, 2005). Agricultural (52.1%), forest and semi natural (45.2%), and artificial (2.1%) are the dominant landuses in the Segura basin (estimated from Corine Land Cover 2000), making the Segura River Basin one of the most regulated in Europe (Ministerio de Medio Ambiente MMA, 2004). Water resource demands exceed 224% of that available and only 4% of runoff reaches the mouth of the river (Zimmer, 2010). This has resulted in over exploitation of the surface waters, an inter-basin transfer from the Tagus River (a mean of 325 hm^3 yr^{-1}), a mean groundwater extraction of around 478 hm^3/year (over 80% of natural recharge) and a high regulatory capacity of 770 hm^3 (over 90% of the natural input) due to 24 dams over 10 m in height (Grindlay *et al*., 2009; Grindlay *et al*., 2011).

Hydrological data

A drainage network was derived from a 25 m digital elevation model (DEM) developed by the *Instituto Geológico Nacional* (IGN) and layers available from the website of the Spanish Ministry of Environment, using the ArcGIS software (v 9.2) and the ArcHydro extension (v 1.2) (ESRI, Redlands, California, U.S.A.). The network comprises sections that link each network junction or node, and each node was associated with its corresponding watershed (derived from the DEM). The minimum watershed area to define a river section was 10 km^2, resulting a hydrological network with 390 river sections.

The hydrological classification developed for the Segura River Basin in Belmar *et al*. (2011) was used to define distinct natural hydrological regimes. This classification was developed using 73 indices based on the "Indicators of Hydrologic Alteration" (IHA) (Mathews and Richter, 2007). These flow indices represent a wide range of ecologically-relevant flow statistics (Richter *et al*., 1996; Olden and Poff, 2003; Monk *et al*., 2006; Mathews and Richter, 2007; Monk *et al*., 2007) and comprise monthly and annual flow statistics including measures of duration of droughts as well as the central tendency and dispersion of flow magnitude (average, low and high flow conditions). Indices related to the frequency, duration and rate of change of high flow events were not used by Belmar *et al*. (2011) due to the absence of daily flow data. Natural flows were derived from a monthly rainfall-runoff model developed by the Centre for Hydrographic Studies (CEDEX, Ministry of Environment and Public Works, Spain), for the period 1980/81 – 2005/06. The classification of the flow regimes recorded comprised eight flow-regime classes (names are provided throughout to aid interpretation) principally characterised by the magnitude of mean annual flow, the duration of droughts and the interannual variation of flow (Table I). The resulting flow regimes can be placed into four broad hydrological groups: (1) mainstem rivers, with perennial flow thorough the year, low interannual variation and an average annual discharge greater than 10 m^3/s (class 1, *large rivers*) or between 2 and 10 m^3/s (class 2, *medium rivers*); (2) perennial stable streams, which only difference respect to mainstem rivers is their reduced average discharge, between 0.3 and 2 m^3/s (class 3, *creeks*) or lower than 0.3 m^3/s (class 4, *headwater streams*); (3) perennial seasonal streams, which eventually cease flowing (although perennial surface water persists) and with peak discharges in winter (class 5, *winter peak flow seasonal streams*) or spring (class 6, *spring peak flow seasonal streams*); and (4) temporary streams, including *intermittent streams* (class 7), which do not flow for between 20% and 50% of the time, and *ephemeral streams*, that do not experience flow for more than 50% of the time (class 8). Indices and classes were assigned to their corresponding river section.

Macroinvertebrate data

Macroinvertebrate abundance data at family, genus and species level were compiled from the *Biodiversidad* database (*Ecología Acuática* research group, Department of Ecology and Hydrology, University of Murcia, Spain). Species data were available for beetles (Coleoptera), which have been recorded in all kinds of water bodies in the region and have been shown to be good indicators of aquatic biodiversity (Bilton *et al*., 2006; Sanchez-Fernandez *et al*., 2006). Samples had been taken along 100 m stream transects using a kick-net (500 – 1000 μm) and following the multi-habitat protocol (Jáimez-Cuéllar *et al*., 2002). Baseline macroinvertebrate samples were collected between 1980 and 2006.

A minimum of 5 samples per hydrological class were selected, ensuring that they had been collected in freshwater streams (conductivity < 5 000 $\mu S\ cm^{-1}$), above water regulation infrastructures (e.g., dams or weirs) and abstraction areas and in absence of significant evidences of anthropogenic alteration. However, using the criteria above two classes did not have any biological data: *large rivers* (class 1), due to the absence of reference conditions, and *ephemeral streams* (class 8), where no sampling had been undertaken due to their frequent dry status.

Every sample was collected during the spring or early summer from a different sampling site (Figure 1). This time-period is considered the most representative of the annual macroinvertebrate community composition in Mediterranean streams (Bonada *et al*., 2009). Each site was paired with the closest downstream node in the drainage network. In order to avoid pseudoreplication, when there was more than one site (and sample) available for the same node, only the closest to the hydrological node was selected. The final dataset consisted of 35 samples associated with 83 macroinvertebrate families, and 133 genera, and 43 samples associated with 110 Coleoptera species (Appendix A).

Environmental data

Climatic, topographic and geologic variables that were assumed to control hydrological processes (Snelder *et al*., 2005) were derived from different GIS layers available for the watershed. Average annual precipitation and air temperature were derived from 1 km grid maps created by the Spanish Ministry of Environment by means of interpolation using data from the Spanish weather stations network (Estrela *et al*., 1999). Drainage area, mean altitude and slope were calculated using the IGN's digital elevation model (DEM). Geology was characterised by the percentage of karst area in each watershed and derived from the "Spain's Map of Karst" 1:1 000 000 developed by the *Instituto Geológico y Minero de España* (IGME) and, indirectly, through water conductivity (recorded for every biological sample). We hypothesised that the karstic surface would control groundwater storage and baseflow (Snelder and Biggs, 2002) and that higher conductivities would reflect the predominance of sedimentary marls that result in flashy hydrographs that reflect precipitation patterns (Bracken *et al*., 2008).

Data analysis

A Principal Component Factor Analysis (PCFA) (i.e. a Principal Components Analysis (PCA) combined with a Varimax rotation) was used to examine dominant patterns of intercorrelation among the hydrological indices (Belmar *et al*. 2011) and to identify subsets of indices that describe the major sources of variation while minimizing redundancy (i.e. multicollinearity). The Varimax rotation allows obtaining a clearer pattern of loadings (indices clearly marked by high loadings for some axes and low loadings for others) and, therefore, a better interpretation of the meaning of each axis. The hydrological characteristics of each stream in the network were defined through the corresponding PCFA scores (hydrological components) and hydrological class.

Rare taxa (those collected at fewer than 5% of sampling sites) were removed for multivariate analyses. Abundance data were transformed by means of the Beals smoothing function (Beals, 1984; McCune, 1994) to reduce noise by enhancing the pattern of joint occurrences. This function is appropriate in the current investigation because the data consist of a large number of small sample units (Peck *et al*., 1995) and fulfill the requirements established by De Cáceres and Legendre (2008).

For each taxonomic level analysed, we performed a non-metric multidimensional scaling (NMDS) ordination based on Bray-Curtis distances among the sampling sites. The strength of the correlation between the NMDS axes and the environmental variables, as well as the hydrological components, was plotted as vectors. In addition, the individual variables and components were analyzed using Pearson coefficients. Covarying (redundant) environmental variables were removed for subsequent model development since the primary objective of the research was to determine the most important flow components influencing macroinvertebrate assemblages and not to distinguish the independent effect of hydrological and environmental drivers.

Distance based linear models (DistLM) were developed to assess the importance of hydrological components driving taxonomical differences among sites. DistLM calculates a multivariate multiple regression analysis between any symmetric distance matrices, including a permutation test, as described by McArdle and Anderson (2001). The final models were selected following a forward-stepwise procedure. For each taxonomic level, marginal tests determined the variance explained by each flow component and the sequential procedure discarded the variance shared by more than one thereby avoiding the overestimation of their effect on the community.

Similarly, generalised linear models (GLM) were employed to determine how hydrological components (independent variables) affected faunal richness patterns. Models were constructed using log-transformed data following a forward-stepwise procedure, assuming a Gaussian error distribution for the dependent variables. These variables were the richness of Coleoptera species, number of macroinvertebrate genera, number of macroinvertebrate families and the ratio EPT/EPTOCHD (defined by the richness of Ephemeroptera, Plecoptera, Trichoptera, Odonata, Coleoptera, Heteroptera and Diptera families). The latter is based on the EPT/EPTOCH ratio, which is used to characterise temporary and lotic–lentic conditions in Mediterranean-climate regions (Bonada *et al*., 2006).

A non-metric single-factor Analysis of Similarity (ANOSIM) was used to test whether assemblage composition differed among hydrological classes and, therefore, if natural regimes can be used to differentiate distinct groups of invertebrate communities. Global R indicates if assemblages are randomly grouped (i.e., R=0) or not (usually $0 < R \leq 1$, although negative values are possible *sensu* Clarke (1993)). R pairwise values were also obtained for each pair of classes, indicating whether intra-class similarities were greater than inter-class similarities (R value close to 1).

Indicator taxa were defined for each hydrological class using the Indicator Species analysis (IndVal) of Dufrene & Legendre (1997). This analysis generates an indicator value index (IV) for each taxon and class, calculated on the basis of the specifity (maximum when a taxon only occurs in one class) and fidelity (maximum when all sites in a class have the taxon) of each taxon to each class.

All permutation tests (DistLM, ANOSIM and IndVal) were undertaken using 999 permutations. PCFA was undertaken in STATISTICA v 6 (Statsoft, 2001). NMDS and IndVal were conducted using PC-ORD software v 4.42 (McCune and Grace, 2002). ANOSIM and DistLM were undertaken in PRIMER v6 (Clarke and Gorley, 2006). GLM were performed using the R statistical software v 2.12.2 (R Development Core Team, 2008).

Results

Hydrological components

The three first PCFA axes were selected to represent the set of hydrological indices since all of them explained greater than 10% of the variance (46, 28 and 12%, respectively) and the forth axis only explained an additional 4%. The first axis was positively correlated with mean and maximum monthly flows (Table IIa), representing the flow magnitude component of the IHA. The second axis was negatively correlated with the inter-annual coefficients of variation in monthly flows, the intra-annual coefficient of variation in maximum monthly flows and the percentage of time without flows. These variables characterise the inter- and intra-annual variability of the flow regime and as a result this axis was defined as the flow stability component (Table IIb). The third axis, magnitude of minimum flows, was correlated with all the minimum monthly flows and their average value (Table IIc).

These three hydrological components (PCFA axes) displayed significant positive correlations with mean altitude and precipitation in the watershed, and negative correlations with mean temperature (Table III). In addition, karst surface and slope were positively correlated with flow stability and minimum flows, while drainage area was associated with the magnitude of flow. As anticipated, conductivity displayed a negative association with flow magnitude and stability.

Hydrological components determining assemblage composition

The macroinvertebrate NMDS ordinations for different taxonomic resolutions identified similar patterns (Figure 2). Sites were structured along a flow stability gradient from perennial headwater streams (left side, class 4) to intermittent streams (right side, class 7), although some classes were widely dispersed (particularly class 6 - spring peak flow seasonal streams). This gradient was associated with several environmental variables and hydrological components (PCFA axes). Perennial stable streams (classes 3 and 4) were predominately located on karstic rocks and sites in higher altitude areas with steeper slopes, higher flow stability and relatively high minimum flows. In contrast, intermittent streams were associated to low slopes, reduced flow stability and low minimum flows, but higher conductivity and air temperature.

DistLM models indicated that hydrological components accounted for a significant proportion of the variance in the macroinvertebrate community that increased with taxonomic resolution (Table IV): 28% for families, 30% for genus and 38% for Coleoptera species. In all cases, flow stability and minimum flows were the dominant hydrological drivers of taxonomical differences among sites.

Response of taxonomic richness to hydrological components

GLM results showed a moderate effect of hydrological variables on the richness of macroinvertebrate families, genera and species (Table V). However, the model obtained for the EPT/EPTOCHD ratio explained 36 % of the variance using flow magnitude and flow stability as independent variables. Gradual changes to the relative richness of EPT families were observed from perennial to intermittent hydrological classes, decreasing along the flow magnitude gradient, whilst the OCHD families displayed the opposite pattern (Figure 3).

Differences in assemblage composition among hydrological classes

The hydrological classes identified supported significantly different invertebrate assemblages at the family (ANOSIM, R = 0.39; *P*-value < 0.05), genus (ANOSIM, R = 0.34; *P*-value < 0.05) and species taxonomic level (ANOSIM, R = 0.40; *P*-value < 0.05) (Table VI). Pair-wise comparisons revealed significant assemblage differences at all taxonomic resolutions between the extremes of the hydrological gradient, perennial stable streams (creeks and headwaters, classes 3 and 4 respectively) and intermittent streams (class 7). Differences between creek and medium river communities (class 2) as well as between creeks and perennial seasonal streams with peak flows during the winter (class 5) increased with the taxonomic resolution, except for the genus level. However, intermittent streams and perennial seasonal streams, both with winter (class 5) and spring peak flows (class 6), differed at the genus or at the genus and species levels, respectively. No significant differences were found both between creeks and headwater streams or within seasonal streams (winter and spring peak flows) (Table VI).

The IndVal analyses determined indicator families for medium rivers (class 2), headwater streams (class 4), spring peak flow seasonal streams (class 6) and intermittent streams (class 7) (Table VII). Medium rivers were characterised by Polycentropodidae (Trichoptera) and Potamanthidae (Ephemeroptera). Headwater streams were defined by one Ephemeroptera (Leptophlebiidae), five families of Trichoptera (Limnephilinae and Beraeidae showed slightly higher Indicator Values) and one Crustacea (Astacidae). Spring peak flow seasonal streams were characterised by Syrphidae (Diptera), which presented the highest Indicator Value in the Segura Basin. Intermittent streams were defined by the presence of Coenagrionidae and Libellulidae (Odonata), Pleidae (Heteroptera) and Noteridae and Hydrophilidae (Coleoptera).

Indicator genera were found for all classes except creeks (class 3) and winter peak flow seasonal streams (class 5). Medium rivers (class 2) and headwater streams (class 4) were characterised by Ephemeroptera: *Habrophlebia* and *Potamanthus* for the former and *Epeorus* and *Rhithrogena* for the latter. Headwaters were also characterised by seven Coleoptera genera (*Oreodytes*, *Graptodytes*, *Esolus*, *Limnebius*, *Normandia*, *Hydrocyphon* and *Oulimnius*), two Trichoptera (*Rhyacophila* and *Sericostoma*), one

Crustacea (*Austropotamobius*) and two Plecoptera (*Perla* and *Isoperla*). Spring peak flow seasonal streams (Class 6) were characterised by one genus of Coleoptera (*Dytiscus*), Hirudinea (*Helobdella*), Molusca (*Pseudamnicola*) and Odonata (*Platycnemis*), with identical indicator values. Intermittent streams (class 7) highlighted the highest number of indicator genera, with the highest Indicator Values for two Diptera (*Dasyhelea* and *Anopheles*), two Heteroptera (*Heliocorisa* and *Anisops*), two Odonata (*Anax* and *Sympetrum*) and two Coleoptera (*Enochrus* and *Berosus*).

Coleoptera indicator species were detected for all classes except spring peak flow seasonal streams (class 6) (Table VII). Medium rivers (class 2) were primarily characterised by *Hydraena manfredjaechi* and *Normandia nitens*; creeks (class 3) by *Hydraena exasperata*; headwater streams (class 4) by *Helophorus alternans*; winter peak flow seasonal streams (class 5) by *Eretes griseus* and *Ranthus suturalis;* and intermittent streams (class 7) by *Ochthebius delgadoi*.

Discussion

The importance of hydrological components on macroinvertebrate assemblages

The research presented herein supports the general hypothesis that streams with similar flow regimes express greater than random similarity in macroinvertebrate assemblages composition (Resh *et al*., 1988; Poff, 1996). Our results demonstrate relatively strong relationships between community composition and the flow regimes at different taxonomic levels. The strength of these relationships increased with taxonomic resolution suggesting that the species level data yields the strongest relationships and that, where it is available, it should be used in ecohydrological investigations (Monk *et al*., 2012). Flow stability and minimum flows were shown to be the principal hydrological drivers/descriptors of the macroinvertebrate community assemblages in the Segura River Basin. Similar results were reported by Chinnayakanehalli *et al*. (2011) in western USA, where baseflows and seasonality were the main predictors of invertebrate composition. However, these results contrast with studies performed in temperate-maritime regions where the magnitudes of mean flows or high flows were reported to be the best predictors of macroinvertebrate assemblages (Clausen and Biggs, 1997; Monk *et al*., 2006; Monk *et al*., 2008).

Flow stability and minimum flows are major determinants of habitat availability and connectivity that affect aquatic macroinvertebrate assemblages. Flow stability reflects seasonal and interannual patterns of variation, associated with the predictability of flows (Poff, 1996) and the stability of habitat conditions in terms of depth, flow velocity and hydraulic forces (Suen and Herricks, 2009). The variation of stream flow velocity configures stream morphology, water temperature, bed stability and consequently the availability of aquatic habitats for instream organisms (Jowett and Duncan, 1990). Minimum flows represent an extreme of the flow, particularly in the dry season, and reflect the magnitude of seasonal droughts (Smakhtin, 2001). Habitat heterogeneity is reduced under low flow conditions because wetted width, water depth and flow velocity also diminishes (Walters and Post, 2011). In addition, extreme low flows can reduce longitudinal connectivity and increase physical stresses transforming streams into series of isolated pools with higher water temperature and elevated conductivity (Stanley *et al*., 1997). Consequently, droughts have been recognised as an important part of the natural flow regime in intermittent streams (Boulton, 2003; Lake, 2003; Sheldon and Thoms, 2006, Chase, 2007). Species inhabiting intermittent streams must have physiological, behavioural or life-history adaptations to cope with higher conductivities, predation pressures and habitat isolation, such as short life-histories, generalist feeding, aerial respiration or active aerial dispersal (e.g. Bonada *et al*., 2007b). Under these conditions, dispersal abilities and distances between or along water bodies have been found to be primary determinants of community composition (McAbendroth *et al*., 2005), because active movement when the riverbed is dry is limited to a small number of taxa such as dytiscid and hydrophilid beetles (Boulton *et al*., 2006; Larned *et al*., 2010).

Our results indicate a moderately strong relationship between flow regime and faunal richness at the different taxonomic resolutions, weaker than that between flow regime and community composition (especially at species level). Other studies have also reported a moderate effect of minimum flows (Walters and Post, 2011), flow seasonality or the number of days with zero flow (Chinnayakanahalli *et al*., 2011).

In Mediterranean regions, ephemeral and intermittent streams are recognised to be significantly less diverse than perennial streams (Bonada *et al*., 2007b) and to differ in community composition (e.g. Bonada *et al*., 2006; Argyroudi *et al*., 2009). Our results found a strong relationship between flow magnitude, and stability, and the ratio of EPT/EPTOCHD. This supports the findings of Bonada *et al*. (2006) and Sánchez-Montoya *et al*. (2007), who reported a decrease in EPT richness as hydrological isolation and the length of the dry period (temporality) increased. EPT taxa in particular tend to occur in riffles, whereas pools support the majority of OCHD taxa (Vivas *et al*., 2002; Oscoz *et al*., 2011). Therefore, riffle permanence has a strong effect on the structure of benthic assemblages in streams (Feminella, 1996).

Biological significance of hydrological classes

The six hydrological classes examined in this study indicate distinct macroinvertebrate assemblages at all of the taxonomic resolutions considered. Taxonomic differences were greatest between the classes at both extremes of the flow stability gradient, and are similar to results reported by other studies in the Iberian Peninsula (Sanchez-Montoya *et al*., 2007) and in the Segura Basin (Millan *et al*., 2006; Diaz *et al*., 2008; Carbonell *et al*., 2011). However, when the other classes were considered, only minor and gradual biological changes along the gradient were detected. Consequently, a simpler classification with four broad hydrological types (Belmar *et al.* 2011) is more appropriate for management purposes in the Segura River Basin and other semi-arid Mediterranean basins: (1) mainstream rivers (classes 1 and 2), (2) perennial stable streams (classes 3 and 4), (3) perennial seasonal streams (classes 5 and 6) and (4) temporary streams (classes 7 and 8).

We found a clear agreement between the selection of indicator taxa in this study and those from other studies in the Mediterranean region in Spain (e.g., Bonada *et al*., 2004; Mellado, 2005; Sanchez-Montoya *et al*., 2007). Headwater streams were characterised by taxa that inhabit the upper reaches of rivers with colder and oxygen-rich waters, in areas of cobbles and small boulders. These sites supported the greatest presence of Ephemeroptera (Leptophlebiidae) and Trichoptera (e.g. Limnephilinae and Beraidae) families and were also characterised by the presence of typically reophilic Ephemeroptera (*Epeorus* and *Rhitrogena*) and Plecoptera (*Perla* and *Isoperla*) genera. In general, these taxa are considered to have high oxygen requirements and their presence is associated with good water quality (Jacobsen *et al*., 2003). Medium rivers were characterised by Ephemeroptera genera, such as *Potamanthus* and *Habrophlebia,* typical of reaches of large rivers where low to moderate flow velocities, associated with gravel and sand substrates, predominate (Puig *et al*., 1984). Intermittent streams were associated with taxa from shallow standing waters or those with reduced velocities, such as numerous Coleoptera (e.g., *Enochrus, Berosus* and *Noterus*), Odonata (e.g., *Anax, Sympetrum* and *Isnchnura*) and Heteroptera (e.g., *Heliocorisa*, *Anisops* and *Sigara*), with highly mobile adults (Bilton *et al*., 2001) and short life-history development times (Velasco *et al*., 1990; Barahona *et al*., 2005). The importance of Coleoptera in temporary streams highlighted in this study has also been demonstrated in several previous studies (Picazo *et al*., 2012).

Implications to river restoration and conservation

Based on the results presented, the magnitude of monthly minimum flows and the inter- and intra-annual natural variation of flows are two key flow components for the definition of environmental flows in Mediterranean basins. Currently, many historically perennial streams have already become intermittent due to excessive abstraction and impoundment, while others exhibit an inverse seasonal pattern due to water release from reservoirs during the summer months (Belmar *et al*., 2010). Such hydrological modifications could become more intense in the future as a result of climate change (European Environment Agency, 2008), which is expected to intensify supra-seasonal droughts and lead to more anthropogenic water withdrawals. This may lead to the depletion of groundwater in local aquifers and, therefore, flow intermittency in previously perennial streams. Such intermittency could result in significant changes to the faunal community, increasing the risk of local extinctions of drought-sensitive taxa. This effect has already been documented in desert streams (Bogan and Lytle, 2011), where simplified pools composed of the most tolerant and resilient species have been described (*sensu* Cote and Darling, 2010). Therefore, the conservation and, where appropriate, restoration of natural hydrological variability is crucial for the maintenance of riverine ecosystem integrity (i.e., ecosystem structure and function) (Thoms, 2006; Vaughan *et al*., 2009).

Future research should focus on how the degree of hydrological alteration affects aquatic communities and ecosystem functioning. Aquatic macroinvertebrates are ideal candidates for the development of hydro-ecological models to quantify the effects of flow reduction (Castella *et al*., 1995; Niu and Dudgeon, 2011a; Niu and Dudgeon, 2011b). Using the four broad hydrological types stated we will be able to provide a reference framework in the near future to achieve a more sustainable management of ecohydrological resources in the Segura River Basin and other Mediterranean basins, fulfilling the objectives of ELOHA and EU Water Framework Directive criteria.

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Table I. Mean and standard deviation of the mean annual flow (MADIS), time with zero flow (D_L) and coefficient of variation in annual flows (CV_{INTER}) for the natural flow regime classes defined in the Segura River Basin (Belmar *et al*. 2011).

Hydrological class	Number of stream sections	MADIS (m^3/s)	D_L (%)	CV_{INTER}
Class 1: Perennial large size rivers	17	11.30 (± 0.74)	0.00 (± 0.00)	0.52 (± 0.01)
Class 2: Perennial medium size rivers	31	3.76 (± 2.26)	0.00 (± 0.00)	0.50 (± 0.13)
Class 3: Perennial stable creeks	21	1.00 (± 0.45)	0.00 (± 0.00)	0.32 (± 0.09)
Class 4: Perennial stable headwater streams	43	0.18 (± 0.17)	0.00 (± 0.00)	0.26 (± 0.13)
Class 5: Perennial winter peak flow seasonal streams	26	0.37 (± 0.09)	2.31 (± 2.06)	1.39 (± 0.29)
Class 6: Perennial spring peak flow seasonal streams	110	0.06 (± 0.06)	4.46 (± 6.32)	0.81 (± 0.30)
Class 7: Temporary intermittent streams	101	0.04 (± 0.04)	24.88 (± 13.15)	1.71 (± 0.38)
Class 8: Temporary ephemeral streams	41	0.01 (± 0.01)	61.90 (± 20.21)	3.43 (± 0.84)

Table II. Pearson correlation coefficients between the three rotated PCFA axes and the 73 hydrological indices. Coefficients higher than |0.70| are in bold letter. Horizontal lines separate indices associated to the three flow components represented by the axes: (a) magnitude (average and maximum flows), 1st axis (46% of variance); (b) flow stability, 2nd axis (28% of variance); and (c) minimum flows, 3rd axis (12% of variance).

	Variable	Description	PCFA axis 1st	2nd	3rd
(a)	M_A 1	Mean monthly flow (October)	**0.98**	0.13	0.02
	M_A 2	Mean monthly flow (November)	**0.98**	0.13	0.02
	M_A 3	Mean monthly flow (December)	**0.99**	0.12	0.05
	M_A 4	Mean monthly flow (January)	**0.98**	0.14	0.04
	M_A 5	Mean monthly flow (February)	**0.98**	0.14	0.04
	M_A 6	Mean monthly flow (March)	**0.99**	0.12	0.03
	M_A 7	Mean monthly flow (April)	**0.98**	0.14	0.02
	M_A 8	Mean monthly flow (May)	**0.98**	0.15	0.04
	M_A 9	Mean monthly flow (June)	**0.98**	0.15	0.03
	M_A 10	Mean monthly flow (July)	**0.97**	0.16	0.04
	M_A 11	Mean monthly flow (August)	**0.97**	0.16	0.05
	M_A 12	Mean monthly flow (September)	**0.98**	0.13	0.02
	M_A 16	Mean annual flow divided by catchment area	0.18	0.50	0.35
	MEDDIS/A	Median annual discharge divided by catchment area	0.22	0.52	0.35
	M_H1	Mean of the maximum monthly flows (October)	**0.96**	0.08	0.01
	M_H2	Mean of the maximum monthly flows (November)	**0.96**	0.06	0.07
	M_H3	Mean of the maximum monthly flows (December)	**0.91**	0.00	0.05
	M_H4	Mean of the maximum monthly flows (January)	**0.97**	0.14	0.08
	M_H5	Mean of the maximum monthly flows (February)	**0.97**	0.15	0.11
	M_H6	Mean of the maximum monthly flows (March)	**0.94**	0.03	0.02
	M_H7	Mean of the maximum monthly flows (April)	**0.98**	0.10	0.04
	M_H8	Mean of the maximum monthly flows (May)	**0.98**	0.15	0.08
	M_H9	Mean of the maximum monthly flows (June)	**0.98**	0.13	0.00
	M_H10	Mean of the maximum monthly flows (July)	**0.98**	0.13	-0.03
	M_H11	Mean of the maximum monthly flows (August)	**0.98**	0.13	-0.03
	M_H12	Mean of the maximum monthly flows (September)	**0.95**	0.05	-0.04
	M_H13	Mean of the mean maximum flows for all months	**0.98**	0.08	0.04
	MADIS	Mean annual flow for all years	**0.98**	0.14	0.03
	RANGE	Maximum annual discharge minus minimum annual discharge	**0.98**	0.06	-0.05
	Q1	Percentile flow with the annual discharge exceeded 1% of time	**0.99**	0.09	0.01
	Q50	Median annual flow for all years	**0.97**	0.14	0.03
(b)	CV_A 1	Coefficient of variation (October)	-0.08	**-0.83**	-0.30
	CV_A 2	Coefficient of variation (November)	-0.12	**-0.86**	-0.15
	CV_A 3	Coefficient of variation (December)	-0.09	**-0.84**	-0.19
	CV_A 4	Coefficient of variation (January)	-0.19	**-0.88**	-0.21
	CV_A 5	Coefficient of variation (February)	-0.21	**-0.89**	-0.17
	CV_A 6	Coefficient of variation (March)	-0.19	**-0.81**	-0.25
	CV_A 7	Coefficient of variation (April)	-0.26	**-0.90**	-0.20
	CV_A 8	Coefficient of variation (May)	-0.02	**-0.91**	-0.19
	CV_A 9	Coefficient of variation (June)	0.02	**-0.83**	-0.35
	CV_A 10	Coefficient of variation (July)	0.09	**-0.82**	-0.37
	CV_A 11	Coefficient of variation (August)	0.09	**-0.84**	-0.36
	CV_A 12	Coefficient of variation (September)	-0.03	**-0.81**	-0.34
	M_A 13	Range divided by median monthly flow	-0.06	**-0.90**	-0.03
	M_A 14	Interquartile divided by median monthly flow	0.09	**-0.80**	0.05
	CV_{INTRA}	Coefficient of variation in mean monthly flows	0.02	**-0.90**	-0.03
	M_A 15	Mean minus median monthly flow divided by median monthly flow	-0.15	**-0.73**	0.06
	M_A 17	Range divided by median annual flow	-0.22	**-0.93**	-0.10
	M_A 18	Interquartile divided by median annual flow	-0.17	**-0.83**	-0.05
	M_A 19	Mean minus median annual flow divided by median annual flow	-0.17	**-0.84**	0.03
	CV_H	Coefficient of variation in mean maximum monthly flows	-0.27	**-0.79**	-0.08
	D_L	Percentage of months with zero flow	-0.38	**-0.75**	-0.24
	CV_{INTER}	Coefficient of variation in annual flows for all years	-0.21	**-0.92**	-0.25
	Q5/Q50	Q5 divided median monthly flow	-0.23	**-0.88**	-0.08
	Q10/Q50	Q10 divided median monthly flow	-0.21	**-0.87**	-0.06
	STDEV	Standard deviation of annual discharge	**0.99**	0.07	-0.09
	AMAX/Q50	Maximum annual discharge divided by Q50	-0.23	**-0.92**	-0.08
	AMIN/Q50	Minimum annual discharge divided by Q50	-0.25	0.63	0.42
	I_H	Q5 divided mean monthly flow	0.08	-0.04	-0.27
	I_L	Q95 divided mean monthly flow	-0.26	0.60	0.48
(c)	M_L 1	Mean minimum monthly flow (October)	0.02	0.19	**0.92**
	M_L 2	Mean minimum monthly flow (November)	0.04	0.19	**0.92**
	M_L 3	Mean minimum monthly flow (December)	0.03	0.19	**0.92**
	M_L 4	Mean minimum monthly flow (January)	0.11	0.20	**0.77**
	M_L 5	Mean minimum monthly flow (February)	0.08	0.18	**0.88**
	M_L 6	Mean minimum monthly flow (March)	0.04	0.18	**0.93**
	M_L 7	Mean minimum monthly flow (April)	0.10	0.23	**0.78**
	M_L 8	Mean minimum monthly flow (May)	0.03	0.17	**0.93**
	M_L 9	Mean minimum monthly flow (June)	0.00	0.17	**0.90**
	M_L 10	Mean minimum monthly flow (July)	0.01	0.17	**0.90**
	M_L 11	Mean minimum monthly flow (August)	0.04	0.16	**0.89**
	M_L 12	Mean minimum monthly flow (September)	0.05	0.16	**0.88**
	M_L 13	Mean of the mean minimum flows for all months	0.06	0.20	**0.96**

Table III. Pearson correlation coefficients between environmental variables and hydrological components (PCFA axes). Significant correlations ($p<0.05$) are in bold letter.

Environmental variable	Flow magnitude	Flow stability	Minimum flows
Mean precipitation (mm)	**0.26**	**0.64**	**0.39**
Conductivity (µS/cm^2)	**-0.28**	**-0.54**	-0.21
Mean altitude (m)	**0.34**	**0.64**	**0.34**
Mean slope (º)	0.24	**0.37**	**0.27**
Karst surface (%)	0.21	**0.36**	**0.37**
Mean temperature (º C)	**-0.37**	**-0.57**	**-0.27**
Drainage area (km^2)	**0.83**	-0.16	-0.14

Table IV. Results of the DistLM analyses for each taxonomic level. Significance levels are indicated with asterisks (*: p≤0.05; **: p≤0.01; ***: p≤0.001).

	Macroinvertebrate families		Macroinvertebrate genera		Coleoptera species	
Hydrological component	Marginal (%)	Sequential (%)	Marginal (%)	Sequential (%)	Marginal (%)	Sequential (%)
Flow magnitude	7	6*	6	5	4	3*
Flow stability	12***	9**	24***	24***	27***	27***
Minimum flows	13**	13***	11**	6*	16***	8***
Total (%)		**28**		**30**		**38**

Table V. GLM analyses for the different dependent variables, based on richness. Significance levels are indicated with asterisks (*: $p \leq 0.05$; **: $p \leq 0.01$; ***: $p \leq 0.001$).

Dependent variable	Variance explained (%)	Explanatory hydrological components
EPT/EPTOCHD	36	Flow magnitude**, flow stability*
Macroinvertebrate families	21	Minimum flows**
Macroinvertebrate genera	24	Minimum flows**
Coleoptera species	17	Minimum flows**

Table VI. Results of ANOSIM analyses. Significance levels are indicated with asterisks (*: p≤0.05; **: p≤0.01; ***: p≤0.001).

Classes	Macroinv. families	Macroinv. genera	Coleop. species
2, 5	0.22	0.15	0.50**
2, 7	0.59*	0.62*	0.49**
3, 2	0.26*	0.06	0.56**
3, 5	0.25*	0.20	0.76***
3, 6	0.49**	0.30*	0.05
3, 7	0.50**	0.53**	0.86***
4, 2	0.85**	0.67**	0.37**
4, 3	0.00	-0.02	-0.09
4, 5	0.81**	0.80**	0.66***
4, 6	0.53**	0.77**	0.09
4, 7	0.88**	0.86**	0.77***
5, 7	0.29	0.42*	0.17
6, 2	0.27**	0.33**	-0.01
6, 5	0.12	0.02	0.16
6, 7	0.09	0.44*	0.38**
Global R	0.39***	0.34***	0.40***

Class 2: Perennial medium rivers
Class 3: Perennial stable creeks
Class 4: Perennial stable headwater streams
Class 5: Perennial winter peak flow seasonal streams
Class 6: Perennial spring peak flow seasonal streams
Class 7: Temporary intermittent streams

Table VII. Indicator taxa (IV≥25 & p ≤ 0.05) for each hydrological class and taxonomic level.

Hydrological class	Macroinvertebrate families	IV (%)	Macroinvertebrate genera	IV	Coleoptera species	IV (%)
2. Perennial medium rivers	Polycentropodidae	31	*Habrophlebia*	28	*Hydraena manfredjaechi*	47
	Potamanthidae	27	*Potamanthus*	27	*Normandia nitens*	47
					Limnius intermedius	44
					Ochthebius difficilis	34
					Limnius opacus	28
					Pomatinus substriatus	25
3. Perennial stable creeks					*Hydraena exasperata*	55
					Ilybius meridionalis	50
					Ochthebius bellieri	46
					Limnius volckmari	34
					Agabus brunneus	32
					Hydroporus marginatus	30
					Ochthebius bonnairei	30
					Anacaena bipustulata	29
					Deronectes moestus	29
					Hydraena carbonaria	29
					Hydraena capta	27
					Hydraena rufipennis	26
					Stictonectes epipleuricus	26
					Agabus didymus	25
4. Perennial stable headwater streams	Leptophlebiidae	41	*Oreodytes*	45	*Helophorus alternans*	29
	Limnephilinae	29	*Epeorus*	35	*Helophorus brevipalpis*	28
	Beraeidae	29	*Rhyacophila*	31	*Laccobius obscuratus*	28
	Brachycentridae	28	*Graptodytes*	30	*Hydroporus tessellatus*	26
	Rhyacophilidae	27	*Austropotamobius*	30	*Limnebius cordobanus*	26
	Sericostomatidae	26	*Esolus*	29		
	Astacidae	26	*Sericostoma*	29		
			Limnebius	28		
			Normandia	27		
			Hydrocyphon	27		
			Rhithrogena	27		
			Oulimnius	25		
			Perla	25		
			Isoperla	25		
5. Perennial winter peak flow seasonal streams					*Eretes griseus*	76
					Rhantus suturalis	76
					Hydrochus nooreinus	52
					Stictotarsus duodecimpustulatus	52
					Berosus hispanicus	34
					Hydrophilus pistaceus	32
					Laccobius moraguesi	31
					Agabus ramblae	29
6. Perennial spring peak flow seasonal streams	Syrphidae	85	*Dytiscus*	35		
			Helobdella	35		
			Pseudamnicola	35		
			Platycnemis	35		
7. Temporary intermittent streams	Noteridae	35	*Dasyhelea*	63	*Ochthebius delgadoi*	42
	Pleidae	35	*Anopheles*	63	*Enochrus politus*	38
	Coenagrionidae	30	*Heliocorisa*	63	*Helophorus fulgidicollis*	38
	Libellulidae	28	*Anisops*	63	*Laccophilus minutus*	38
	Hydrophilidae	26	*Anax*	52	*Ochthebius auropallens*	38
			Enochrus	48	*Ochthebius grandipennis*	38
			Sympetrum	48	*Ochthebius viridis fallaciosus*	38
			Berosus	45	*Ochthebius jaimei*	35
			Sigara	45	*Helochares lividus*	27
			Plea	45		
			Ischnura	45		
			Noterus	42		
			Potamopyrgus	42		
			Cercion	42		
			Libellula	42		
			Helochares	41		
			Bidessus	40		
			Procambarus	40		
			Limnophora	40		
			Tipula	40		
			Microvelia	40		
			Agabus	36		
			Dryops	32		
			Laccobius	32		
			Orthetrum	32		
			Gerris	30		
			Nebrioporus	27		
			Cloeon	27		
			Micronecta	25		

Appendix A. Taxa collected in the Segura Basin grouped by taxonomic order.

Hirudinea
Erpobdellidae
Dina
Glossiphoniidae
Helobdella
Mollusca
Ancylidae
Ancylus
Ferrissia
Hydrobiidae
Mercuria
Potamopyrgus
Pseudamnicola
Lymnaeidae
Lymnaea
Melanopsidae
Melanopsis
Physidae
Physella
Planorbidae
Gyraulus
Planorbarius
Sphaeriidae
Pisidium
Crustacea
Astacidae
Austropotamobius
Atyiidae
Atyaephyra
Cambaridae
Procambarus
Gammaridae
Echinogammarus
Coleoptera
Dryopidae
Dryops
Dryops gracilis (Karsch, 1881)
Dryops sulcipennis (Costa, 1883)
Pomatinus
Pomatinus substriatus (Müller, 1806)
Dytiscidae
Eretes griseus Motschulsky 1849
Hygrotus confluens (Fabricius, 1787)
Hyphydrus aubei Ganglbauer, 1892
Ilybius meridionalis Aubé, 1836
Meladema coriacea Castelnau, 1834
Rhantus suturalis (McLeay, 1825)
Stictotarsus duodecimpustulatus (Fabricius, 1792)
Agabus
Agabus biguttatus (Olivier, 1795)
Agabus bipustulatus (Linnaeus, 1767)
Agabus brunneus (Fabricius, 1798)
Agabus didymus (Olivier, 1795)
Agabus nebulosus (Forster, 1771)
Agabus nitidus (Fabricius, 1801
Agabus paludosus (Fabricius, 1801)
Agabus ramblae Millán & Ribera, 2001
Bidessus
Bidessus minutissimus (Germar, 1824)
Deronectes
Deronectes depressicollis (Rosenhauer, 1856)
Deronectes fairmairei (Leprieur, 1876)
Deronectes hispanicus (Rosenhauer, 1856)
Deronectes moestus Leprieur, 1876
Dytiscus
Graptodytes
Graptodytes fractus (Sharp, 1880-82)
Graptodytes ignotus (Mulsant, 1861)
Graptodytes varius (Aubé, 1836)
Hydroglyphus
Hydroglyphus geminus (Fabricius, 1792)
Hydroglyphus signatellus (Klug, 1834)
Hydroporus
Hydroporus discretus Fairmaire, 1859
Hydroporus lucasi Reiche, 1866
Hydroporus marginatus (Duftschmid, 1805)
Hydroporus nigrita (Fabricius, 1792)
Hydroporus pubescens (Gyllenhal, 1808)
Hydroporus tessellatus Drapiez, 1819
Laccophilus
Laccophilus hyalinus (De Geer, 1774)
Laccophilus minutus (Linnaeus, 1758)
Nebrioporus
Nebrioporus bucheti cazorlensis (Lagar, Fresneda & Hernando, 1987)
Nebrioporus clarki (Wollaston, 1862)
Oreodytes
Stictonectes
Stictonectes epipleuricus (Seidlitz, 1887)
Stictonectes optatus (Seidlitz, 1887)
Yola
Yola bicarinata (Latreille, 1804)
Elmidae
Elmis
Elmis aenea (Müller, 1806)
Elmis maugetii maugetii Latreille, 1798
Elmis rioloides (Kuwert, 1890)
Esolus
Esolus parallelepipedus (Müller, 1806)
Limnius
Limnius intermedius Fairmaire, 1881
Limnius opacus Müller, 1806
Limnius volckmari (Panzer, 1793)
Normandia
Normandia nitens (Müller, 1817)
Normandia sodalis (Erichson, 1847)
Oulimnius
Oulimnius troglodytes (Gyllenhal, 1827)
Oulimnius tuberculatus perezi Sharp, 1872
Potamophilus
Riolus
Riolus cupreus (Müller, 1806)
Riolus illiesi Steffan, 1958
Gyrinidae
Aulonogyrus
Aulonogyrus striatus (Fabricius, 1792)
Gyrinus
Gyrinus dejeani Brullé, 1832
Orectochilus
Orectochilus villosus (Müller, 1776)
Haliplidae
Peltodytes rotundatus (Aubé, 1836)
Haliplus
Haliplus lineatocollis (Marsham, 1802)
Haliplus mucronatus Stephens, 1832
Helophoridae
Helophorus
Helophorus alternans Gené, 1836
Helophorus brevipalpis Bedel, 1881
Helophorus fulgidicollis Motschuslky, 1860
Helophorus occidentalis Angus, 1983
Helophorus nubilus Fabricius, 1776
Helophorus seidlitzii Kuwert, 1885
Hydraenidae
Hydraena
Hydraena capta Orchymont, 1936
Hydraena carbonaria Kiesenwetter, 1849
Hydraena exasperata Orchymont, 1935
Hydraena hernandoi Fresneda & Lagar, 1990
Hydraena manfredjaechi Delgado & Soler, 1991
Hydraena pygmaea Waterhouse, 1833
Hydraena quilisi Lagar, Fresneda & Hernando, 1987
Hydraena rufipennis Boscá Berga, 1932
Hydraena servilia Orchymont, 1936
Limnebius
Limnebius cordobanus Orchymont, 1938
Limnebius maurus Balfour-Browne, 1978
Limnebius oblongus Rey, 1883
Ochthebius
Ochthebius auropallens Fairmaire, 1879
Ochthebius bellieri Kuwert, 1887
Ochthebius bonnairei Guillebau, 1896
Ochthebius delgadoi Jäch, 1994
Ochthebius difficilis Mulsant, 1844
Ochthebius dilatatus Stephens, 1829
Ochthebius (Enicocerus) exsculptus Germar, 1824
Ochthebius grandipennis Fairmaire, 1879
Ochthebius jaimei Delgado & Jäch, 2007
Ochthebius quadrifoveolatus Wollaston, 1854
Ochthebius tudmirensis Jäch, 1997
Ochthebius viridis fallaciosus Ganglbauer, 1901
Hydrochidae
Hydrochus
Hydrochus grandicollis Kiesenwetter, 1870
Hydrochus nooreinus Henegouven & Sáinz-Cantero, 1992
Hydrophilidae
Anacaena bipustulata (Marsham, 1802)
Anacaena globulus (Paykull, 1798)
Anacaena lutescens (Stephens, 1829)
Coelostoma hispanicum (Küster, 1848)
Hydrophilus pistaceus (Castelnau, 1840)

Appendix A (cont.).

Berosus
Berosus hispanicus Küster, 1847
Enochrus
Enochrus ater (Kuwert, 1888)
Enochrus politus Küster, 1849
Helochares
Helochares lividus (Forster, 1771)
Laccobius
Laccobius bipunctatus (Fabricius, 1775)
Laccobius hispanicus Gentili, 1974
Laccobius gracillis gracillis Motschulsky, 1849
Laccobius moraguesi Régimbart, 1898
Laccobius neapolitanus Rottenberg, 1874
Laccobius obscuratus Rottenberg, 1874
Laccobius sinuatus Motschulsky, 1849
Laccobius ytenensis Sharp, 1910
<u>Noteridae</u>
Noterus
Noterus laevis Sturm, 1834
<u>Scirtidae</u>
Cyphon
Elodes
Hydrocyphon
Diptera
<u>Anthomyiidae</u>
Limnophora
<u>Athericidae</u>
Atrichops
Ibisia
<u>Ceratopogonidae</u>
Dasyhelea
<u>Chironomidae</u>
Chironomini
Corynoneura
Tanytarsini
<u>Culicidae</u>
Anopheles
<u>Diamesinae</u>
<u>Dixidae</u>
<u>Empididae</u>
<u>Ephydridae</u>
<u>Hemerodromiinae</u>
<u>Limoniidae</u>
Eloeophyla
Pseudolimnophila
<u>Orthocladiinae</u>
<u>Simuliidae</u>
<u>Stratiomyidae</u>
Oxycera
<u>Syrphidae</u>
<u>Tabanidae</u>
Tabanus
<u>Tanypodinae</u>
<u>Tipulidae</u>
Tipula
Ephemeroptera
<u>Baetidae</u>
Baetis
Centroptilum
Cloeon
Procloeon
<u>Caenidae</u>
Caenis
<u>Ephemerellidae</u>
Ephemerella
Serratella
Torleya
<u>Ephemeridae</u>
Ephemera
<u>Heptageniidae</u>
Ecdyonurus
Epeorus
Rhithrogena
<u>Leptophlebiidae</u>
Habroleptoides
Habrophlebia
Paraleptophlebia
<u>Polymirtacidae</u>
Ephoron
<u>Potamantidae</u>
Potamanthus
Hemiptera
<u>Aphelocheiridae</u>
Aphelocheirus
<u>Corixidae</u>
Heliocorisa
Micronecta
Sigara
<u>Gerridae</u>
Aquarius
Gerris
<u>Hydrometridae</u>
Hydrometra
<u>Naucoridae</u>
Naucoris
<u>Nepidae</u>
Nepa
<u>Notonectidae</u>
Anisops
Notonecta
<u>Pleidae</u>
Plea
<u>Veliidae</u>
Microvelia
Velia
Odonata
<u>Aeshnidae</u>
Anax
Boyeria
<u>Calopterigydae</u>
Calopteryx
<u>Coenagrionidae</u>
Cercion
Ischnura
Pyrrhosoma
<u>Cordulegastridae</u>
Cordulegaster
<u>Gomphidae</u>
Gomphus
Onychogomphus
<u>Libellulidae</u>
Libellula
Orthetrum
Sympetrum
<u>Platycnemididae</u>
Platycnemis
Plecoptera
<u>Leuctridae</u>
Leuctra
<u>Nemouridae</u>
Nemoura
Protonemura
<u>Perlidae</u>
Dinocras
Eoperla
Perla
<u>Perlodidae</u>
Isoperla
Trichoptera
<u>Beraeidae</u>
<u>Brachycentridae</u>
Micrasema
<u>Drusinae</u>
<u>Hydropsychidae</u>
Cheumatopsyche
Hydropsyche
<u>Hydroptilidae</u>
Agraylea
Hydroptila
<u>Lepidostomatidae</u>
Lasiocephala
<u>Leptoceridae</u>
Athripsodes
<u>Limnephilidae</u>
Allogamus
Halesus
Stenophylax
<u>Limnephilinae</u>
<u>Polycentropodidae</u>
<u>Psychomyiidae</u>
Metalype
Tinodes
<u>Rhyacophilidae</u>
Rhyacophila
<u>Sericostomatidae</u>
Sericostoma

References

Argyroudi A, Chatzinikolaou Y, Poirazidis K, Lazaridou M. 2009. Do intermittent and ephemeral Mediterranean rivers belong to the same river type? *Aquatic Ecology* **43**:465-476.

Armanini DG, Monk WA, Tenenbaum DE, Peters DL, Baird DJ. 2011. Influence of runoff regime type on a macroinvertebrate-based flow index in rivers of British Columbia (Canada). *Ecohydrology*. DOI: 10.1002/eco.234.

Arthington AH, Bunn SE, Poff NL, Naiman RJ. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* **16**:1311-1318.

Barahona J, Millan A, Velasco J. 2005. Population dynamics, growth and production of *Sigara selecta* (Fieber, 1848) (Hemiptera, Corixidae) in a Mediterranean hypersaline stream. *Freshwater Biology* **50**:2101-2113.

Beals EW. 1984. Bray-Curtis ordination: an effective strategy for analysis of multivariate ecological data. *Advances in Ecological Research* **14**:1-55.

Belmar O, Velasco J, Martinez-Capel F. 2011. Hydrological classification of natural flow regimes to support environmental flow assessments in intensively regulated Mediterranean rivers, Segura River basin (Spain). *Environmental Management* **47**:992-1004.

Belmar O, Velasco J, Martinez-Capel F, Marin AA. 2010. Natural flow regime, degre of alteration and environmental flows in the Mula stream (Segura River basin, SE Spain). *Limnetica* **29**:353-368.

Bilton DT, Freeland JR, Okamura B. 2001. Dispersal in freshwater invertebrates. *Annual Review of Ecology and Systematics* **32**:159-181.

Bilton DT, McAbendroth L, Bedford A, Ramsay PM. 2006. How wide to cast the net? Cross-taxon congruence of species richness, community similarity and indicator taxa in ponds. *Freshwater Biology* **51**:578-590.

Bogan MT, Lytle DA. 2011. Severe drought drives novel community trajectories in desert stream pools. *Freshwater Biology* **56**:2070-2081.

Bonada N, Doledec S, Statzner B. 2007a. Taxonomic and biological trait differences of stream macroinvertebrate communities between mediterranean and temperate regions: implications for future climatic scenarios. *Global Change Biology* **13**:1658-1671.

Bonada N, Murria C, Zamora-Munoz C, El Alami M, Poquet JM, Punti T, Moreno JL, Bennas N, Alba-Tercedor J, Ribera C, Prat N. 2009. Using community and population approaches to understand how contemporary and historical factors have shaped species distribution in river ecosystems. *Global Ecology and Biogeography* **18**:202-213.

Bonada N, Prat N, Munne A, Plans M, Sola C, Alvarez-Cabria M, Pardo I, Moya G, Ramon G, Toro M, Robles S, Aviles J, Suarez ML, Vidal-Abarca MR, Mellado A, Moreno JL, Guerrero C, Vivas S, Ortega M, Casas J, Sanchez-Ortega A, Jaimez-Cuellar

P, Alba-Tercedor J. 2002. Intercalibración de la metodología GUADALMED. Selección de un protocolo de muestreo para la determinación del estado ecológico de los ríos mediterráneos. *Limnetica* **21**:13-33.

Bonada N, Rieradevall M, Prat N. 2007b. Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. *Hydrobiologia* **589**:91-106.

Bonada N, Rieradevall M, Prat N, Resh VH. 2006. Benthic macroinvertebrate assemblages and macrohabitat connectivity in Mediterranean-climate streams of northern California. *Journal of the North American Benthological Society* **25**:32-43.

Bonada N, Zamora-Munoz C, Rieradevall M, Prat N. 2004. Ecological profiles of caddisfly larvae in Mediterranean streams: implications for bioassessment methods. *Environmental Pollution* **132**:509-521.

Boulton AJ. 2003. Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages. *Freshwater Biology* **48**:1173-1185.

Boulton AJ, Lake PS. 2008. Effects of drought on stream insects and its ecological consequences. In *Aquatic Insects: Challenges to Populations*, 81-102. Lancaster J, Briers RA (eds). CAB International. Wallingford.

Boulton AJ, Sheldon F, Jenkins KM. 2006. Natural disturbance and aquatic invertebrates in desert rivers. In *Ecology of Desert Rivers*, 103-153. Kingsford RT(ed). Academic Press. New York.

Bracken LJ, Coxi NJ, Shannon J. 2008. The relationship between rainfall inputs and flood generation in south-east Spain. *Hydrological Processes* **22**:683-696.

Bunn SE, Arthington AH. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* **30**:492-507.

Carbonell JA, Gutierrez-Canovas C, Bruno D, Abellan P, Velasco J, Millan A. 2011. Ecological factors determining the distribution and assemblages of the aquatic hemiptera (Gerromorpha & Nepomorpha) in the Segura River Basin (Spain). *Limnetica* **30**:59-70.

Castella E, Bickerton M, Armitage PD, Petts GE. 1995. The effects of water abstractions on invertebrate communities in UK streams. *Hydrobiologia* **308**:167-182.

Chase JM. 2007. Drought mediates the importance of stochastic community assembly. *Proceedings of the National Academy of Sciences of the United States of America* **104**:17430-17434.

Chinnayakanahalli KJ, Hawkins CP, Tarboton DG, Hill RA. 2011. Natural flow regime, temperature and the composition and richness of invertebrate assemblages in streams of the western United States. *Freshwater Biology* **56**:1248-1265.

CHS. 2007. *Estudio general sobre la Demarcación Hidrográfica del Segura*. Confederación Hidrográfica del Segura.

Clarke KR. 1993. Nonparametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**:117-143.

Clarke KR, Gorley RN. 2006. *PRIMER v6: User Manual/Tutorial*. PRIMER-E, Plymouth, United Kingdom.

Clausen B, Biggs BJF. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. *Freshwater Biology* **38**:327-342.

Cote IM, Darling ES. 2010. Rethinking ecosystem resilience in the face of climate change. *Plos Biology* **8**:e1000438.

De Caceres M, Legendre P. 2008. Beals smoothing revisited. *Oecologia* **156**:657-669.

Diaz AM, Alonso MLS, Gutierrez M. 2008. Biological traits of stream macroinvertebrates from a semi-arid catchment: patterns along complex environmental gradients. *Freshwater Biology* **53**:1-21.

Dufrene M, Legendre P. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs* **67**:345-366.

Estrela T, Cabezas F, Estrada F. 1999. La evaluación de los recursos hídricos en el Libro Blanco del Agua en España. *Ingeniería del Agua* **6**:125-138.

European Environment Agency. 2008. *Energy and environment report 2008*. Copenhagen, Denmark.

Feminella JW. 1996. Comparison of benthic macroinvertebrate assemblages in small streams along a gradient of flow permanence. *Journal of the North American Benthological Society* **15**:651-669.

Gasith A, Resh VH. 1999. Streams in Mediterranean climate regions: abiotic influences and biotic responses to predictable seasonal events. *Annual Review of Ecology and Systematics* **30**:51-81.

Grindlay A, Zamorano M, Rodriguez MI, Molero E, Urrea MA. 2009. Territorial transformation and water utilization: hydrological planning scenarios in the Segura river basin. In *Sustainable Development and Planning IV, Vols 1 and 2*. Wit Press. Southampton, 975-+.

Grindlay AL, Zamorano M, Rodriguez MI, Molero E, Urrea MA. 2011. Implementation of the European Water Framework Directive: integration of hydrological and regional planning at the Segura River Basin, southeast Spain. *Land Use Policy* **28**:242-256.

Hart DD, Finelli CM. 1999. Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. *Annual Review of Ecology and Systematics* **30**:363-395.

Jacobsen D, Rostgaard S, Vasconez JJ. 2003. Are macroinvertebrates in high altitude streams affected by oxygen deficiency? *Freshwater Biology* **48**:2025-2032.

Jáimez-Cuéllar P, Vivas S, Bonada N, Robles S, Mellado A, Álvarez M, Avilés J, Casas J, Ortega M, Pardo I, Prat N, Rieradevall M, Sáinz-Cantero CE, Sánchez-Ortega A, Suárez ML, Toro M, Vidal-Abarca MR, Zamora-Munoz C, Alba-Tercedor J. 2002. Protocolo GUADALMED (PRECE). *Limnetica* **21**:187-204.

Jowett IG, Biggs BJF. 2009. Application of the natural flow paradigm in a New Zealand context. *River Research and Applications* **25**:1126-1135.

Jowett IG, Duncan MJ. 1990. Flow variability in New Zealand rivers and its relationship to in-stream habitat and biota. *New Zealand Journal of Marine and Freshwater Research* **24**:305-317.

Kennard MJ, Mackay SJ, Pusey BJ, Olden JD, Marsh N. 2010. Quantifying uncertainty in estimation of hydrologic metrics for ecohydrological studies. *River Research and Applications* **26**:137-156.

Kennard MJ, Olden JD, Arthington AH, Pusey BJ, Poff NL. 2007. Multiscale effects of flow regime and habitat and their interaction on fish assemblage structure in eastern Australia. *Canadian Journal of Fisheries and Aquatic Sciences* **64**:1346-1359.

Kennen JG, Riva-Murray K, Beaulieu KM. 2010. Determining hydrologic factors that influence stream macroinvertebrate assemblages in the northeastern US. *Ecohydrology* **3**:88-106.

Konrad CP, Brasher AMD, May JT. 2008. Assessing streamflow characteristics as limiting factors on benthic invertebrate assemblages in streams across the western United States. *Freshwater Biology* **53**:1983-1998.

Lake PS. 2003. Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology* **48**:1161-1172.

Larned ST, Datry T, Arscott DB, Tockner K. 2010. Emerging concepts in temporary-river ecology. *Freshwater Biology* **55**:717-738.

Lytle DA, Poff NL. 2004. Adaptation to natural flow regimes. *Trends in Ecology & Evolution* **19**:94-100.

Mathews R, Richter BD. 2007. Application of the indicators of hydrologic alteration software in environmental flow setting. *Journal of the American Water Resources Association* **43**:1400-1413.

McAbendroth L, Foggo A, Rundle SD, Bilton DT. 2005. Unravelling nestedness and spatial pattern in pond assemblages. *Journal of Animal Ecology* **74**:41-49.

McArdle BH, Anderson MJ. 2001. Fitting multivariate models to community data: A comment on distance-based redundancy analysis. *Ecology* **82**:290-297.

McCune B. 1994. Improving community analysis with the Beals smoothing function. *Ecoscience* **1**:82-86.

McCune B, Grace JB. 2002. *Analysis of ecological communities*. MjM. Gleneden Beach, OR, USA.

Mellado A. 2005. *Ecología de las comunidades de macroinvertebrados de la cuenca del Río Segura (SE de España)*, Thesis/Dissertation, University of Murcia (Spain).

Millan A, Abellan P, Ribera I, Sanchez-Fernandez D, Velasco J. 2006. The Hydradephaga of the Segura Basin (SE Spain): twenty years studying water beetles (Coleoptera). *Memorie della Società entomologica italiana* **85**:137-158.

Ministerio de Medio Ambiente (MMA). 2004. *Water in Spain*. Secretaría de Estado de Aguas y Costas.

Monk WA, Wood PJ, Hannah DM, Extence CA, Chadd RP, Dunbar MJ. 2012. How does macroinvertebrate taxonomic resolution influence ecohydrological relationships in riverine ecosystems. *Ecohydrology* **5**:36-45.

Monk WA, Wood PJ, Hannah DM, Wilson DA. 2007. Selection of river flow indices for the assessment of hydroecological change. *River Research and Applications* **23**:113-122.

Monk WA, Wood PJ, Hannah DM, Wilson DA. 2008. Macroinvertebrate community response to inter-annual and regional river flow regime dynamics. *River Research and Applications* **24**:988-1001.

Monk WA, Wood PJ, Hannah DM, Wilson DA, Extence CA, Chadd RP. 2006. Flow variability and macroinvertebrate community response within riverine systems. *River Research and Applications* **22**:595-615.

Naiman RJ, Latterell JJ, Pettit NE, Olden JD. 2008. Flow variability and the biophysical vitality of river systems. *Comptes Rendus Geoscience* **340**:629-643.

Niu SQ, Dudgeon D. 2011a. Environmental flow allocations in monsoonal Hong Kong. *Freshwater Biology* **56**:1209-1230.

Niu SQ, Dudgeon D. 2011b. The influence of flow and season upon leaf-litter breakdown in monsoonal Hong Kong streams. *Hydrobiologia* **663**:205-215.

Olden JD, Poff NL. 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications* **19**:101-121.

Oscoz J, Galicia D, Miranda R. 2011. *Identification guide of freshwater macroinvertebrates of Spain*. Springer. Dordrecht, Netherlands.

Peck JE, Hong WS, McCune B. 1995. Diversity of epiphytic bryophytes on 3 host tree species, thermal meadow, hotsprings island, Queen-Charlotte-Islands, Canada. *Bryologist* **98**:123-128.

Pegg MA, Pierce CL. 2002. Fish community structure in the Missouri and lower Yellowstone rivers in relation to flow characteristics. *Hydrobiologia* **479**:155-167.

Picazo F, Bilton DT, Moreno JL, Sanchez-Fernandez D, Millan A. 2012. Water beetle biodiversity in Mediterranean standing waters: assemblage composition, environmental drivers and nestedness patterns. *Insect Conservation and Diversity* **5**:146-158.

Poff NL. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. *Freshwater Biology* **36**:71-91.

Poff NL, Allan JD. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. *Ecology* **76**:606-627.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime. *Bioscience* **47**:769-784.

Poff NL, Richter BD, Arthington AH, Bunn SE, Naiman RJ, Kendy E, Acreman M, Apse C, Bledsoe BP, Freeman MC, Henriksen J, Jacobson RB, Kennen JG, Merritt DM, O'Keeffe JH, Olden JD, Rogers K, Tharme RE, Warner A. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology* **55**:147-170.

Poff NL, Ward JV. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. *Canadian Journal of Fisheries and Aquatic Sciences* **46**:1805-1818.

Puig MA, Gonzalez G, Soriano O. 1984. Introducción al estudio de las comunidades macrobentónicas de los ríos asturianos: efemerópteros, plecópteros, tricópteros, simúlidos y quironómidos. *Limnetica* **1**:187-196.

R Development Core Team. 2008. *R: a language and environment for statistical computing*. Vienna, Austria.

Resh VH, Brown AV, Covich AP, Gurtz ME, Li HW, Minshall GW, Reice SR, Sheldon AL, Wallace JB, Wissmar RC. 1988. The role of disturbance in stream ecology. *Journal of the North American Benthological Society* **7**:433-455.

Richter BD, Baumgartner JV, Powell J, Braun DP. 1996. A method for assessing hydrologic alteration within ecosystems. *Conservation Biology* **10**:1163-1174.

Sanchez-Fernandez D, Abellan P, Mellado A, Velasco J, Millan A. 2006. Are water beetles good indicators of biodiversity in Mediterranean aquatic ecosystems? The case of the segura river basin (SE spain). *Biodiversity and Conservation* **15**:4507-4520.

Sanchez-Montoya MD, Punti T, Suarez ML, Vidal-Abarca MD, Rieradevall M, Poquet JM, Zamora-Munoz C, Robles S, Alvarez M, Alba-Tercedor J, Toro M, Pujante AM, Munne A, Prat N. 2007. Concordance between ecotypes and macroinvertebrate assemblages in Mediterranean streams. *Freshwater Biology* **52**:2240-2255.

Sheldon F, Thoms MC. 2006. Relationships between flow variability and macroinvertebrate assemblage composition: data from four Australian dryland rivers. *River Research and Applications* **22**:219-238.

Smakhtin VU. 2001. Low flow hydrology: a review. *Journal of Hydrology* **240**:147-186.

Snelder TH, Biggs BJF. 2002. Multiscale River Environment Classification for water resources management. *Journal of the American Water Resources Association* **38**:1225-1239.

Snelder TH, Biggs BJF, Woods RA. 2005. Improved eco-hydrological classification of rivers. *River Research and Applications* **21**:609-628.

Snelder TH, Lamouroux N, Leathwick JR, Pella H, Sauquet E, Shankar U. 2009. Predictive mapping of the natural flow regimes of France. *Journal of Hydrology* **373**:57-67.

Stanley EH, Fisher SG, Grimm NB. 1997. Ecosystem expansion and contraction in streams. *Bioscience* **47**:427-435.

StatSoft, Inc. 2001. *STATISTICA (data analysis software system), version 6*. www.statsoft.com.

Suen JP, Herricks EE. 2009. Developing fish community based ecohydrological indicators for water resources management in Taiwan. *Hydrobiologia* **625**:223-234.

Thoms MC. 2006. Variability in riverine ecosystems. *River Research and Applications* **22**:115-121.

Tockner K, Ward JV, Arscott DB, Edwards PJ, Kollmann J, Gurnell AM, Petts GE, Maiolini B. 2003. The Tagliamento River: a model ecosystem of European importance. *Aquatic Sciences* **65**:239-253.

Vaughan IP, Diamond M, Gurnell AM, Hall KA, Jenkins A, Milner NJ, Naylor LA, Sear DA, Woodward G, Ormerod SJ. 2009. Integrating ecology with hydromorphology: a priority for river science and management. *Aquatic Conservation-Marine and Freshwater Ecosystems* **19**:113-125.

Velasco J, Millan A, Nieser N. 1990. Observaciones sobre la colonización y el ciclo de vida de *Heliocorisa vermiculata* (Puton, 1874) (Heteroptera, Corixidae) en pequeños estanques del SE español. *Limnetica* **6**:101-108.

Vivas S, Casas J, Pardo I, Robles S, Bonada N, Mellado A, Prat N, Alba-Tercedor J, Alvarez-Cabria M, Bayo MM, Jaimez-Cuellar P, Suárez ML, Toro M, Vidal-Abarca MR, Zamora-Munoz C, Moyá G. 2002. Aproximación multivariante en la exploración de la tolerancia ambiental de las familias de macroinvertebrados de los ríos mediterráneos del proyecto GUADALMED. *Limnetica* **21**:149-173.

Walters AW, Post DM. 2011. How low can you go? Impacts of a low-flow disturbance on aquatic insect communities. *Ecological Applications* **21**:163-174.

Winter TC. 2001. The concept of hydrologic landscapes. *Journal of the American Water Resources Association* **37**:335-349.

Zimmer A. 2010. New water uses in the Segura Basin: conflicts around gated communities in Murcia. *Water International* **35**:34-48.

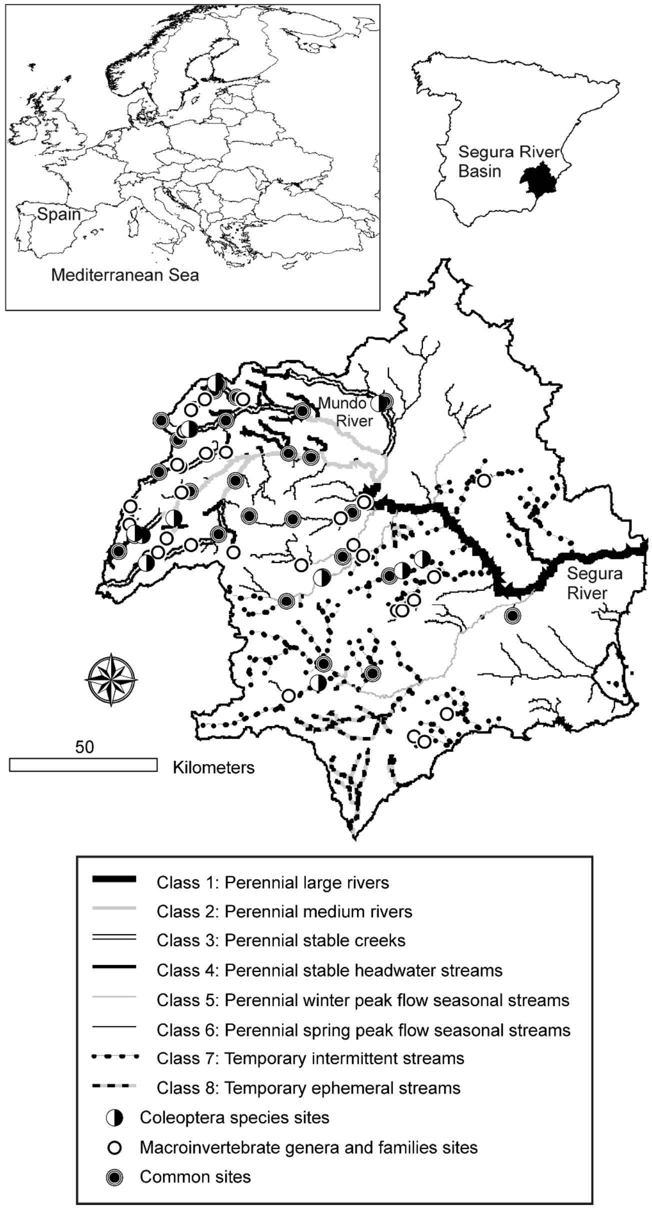

Figure 1. Location of the study area, hydrological classes in the river network and sampling sites.
290x533mm (300 x 300 DPI)

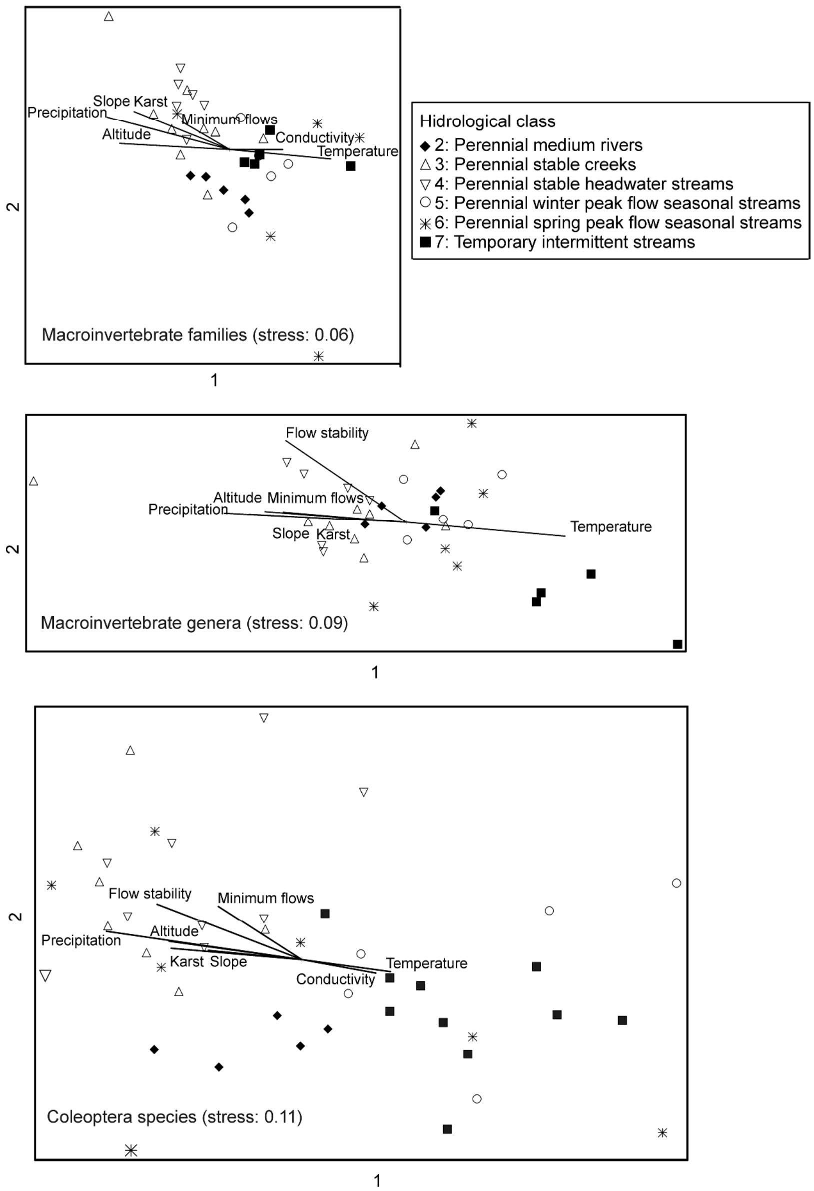

Figure 2. NMDS plots of sites for each taxonomic level. The magnitudes of the correlations between the NDMS axes and the hydrological components as well as the environmental variables are shown as vectors.
289x431mm (300 x 300 DPI)

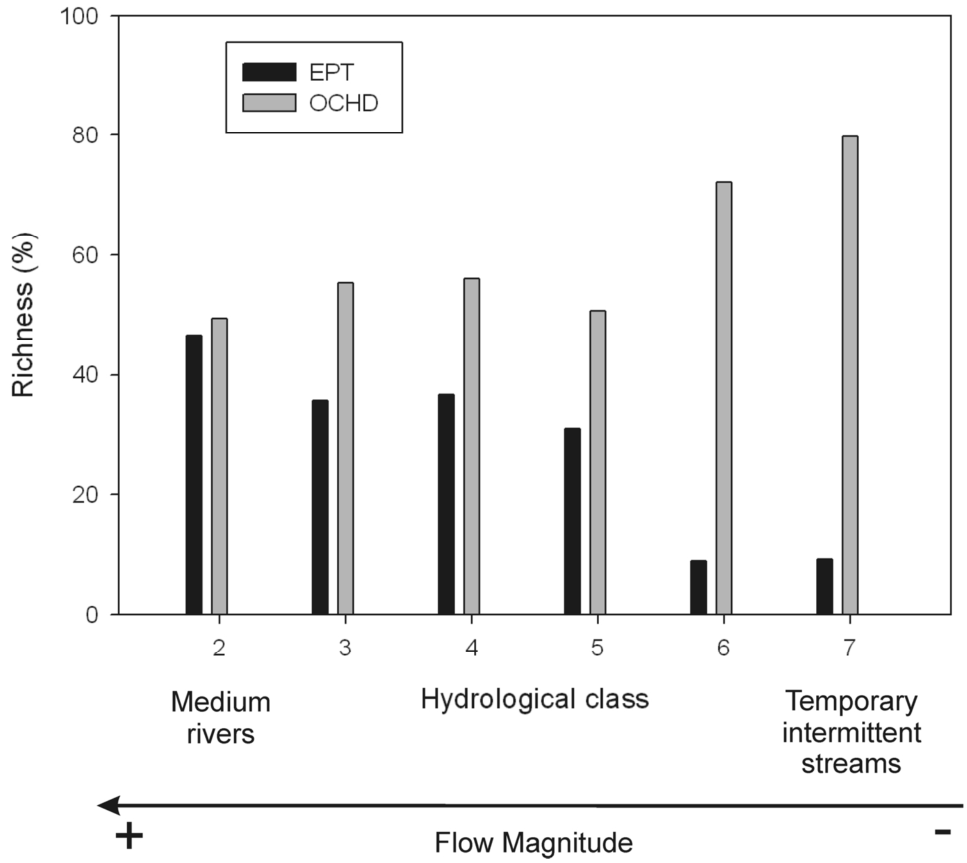

Figure 3. Variation of the percentage of families of the EPT and OCDH groups in the different hydrological classes along the flow magnitude gradient.
140x130mm (300 x 300 DPI)