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Solutions for present and future emerging pollutants in land and water resources management

THEME

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Toxicants, environmental pollutants and land and water resources management

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Deliverable

D21.1 Guidance for identification of RBSPs in Mediterranean river basins and list of RBSP including quantification of their ecological impact and modeling–based exposure and risk predictions validated

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[draft 1]

Dissemination Level		
PU	Public	**X**
PP	Restricted to other programme participants (including EC)	
RE	Restricted to a group specified by the consortium (including EC)	
CO	Confidential, only for members of the consortium (including EC)	

1.1 [readable] Summary

Mediterranean rivers are largely different from Northern and Central European rivers in terms of hydrological regime, climate conditions (e.g. ambient temperature, solar irradiation), socio-economics (e.g. land use, tourism, kinds of crops), etc., all of which leads to differences also in the relative importance of the environmental stressors and in the classes, levels and fate of pollutants found. Water scarcity may be critical in affecting water pollution because of lowered dilution capacity of chemicals. These issues were the research focus of a large Spanish project SCARCE. Over 200 organic priority and emerging pollutants were comprehensively monitored in water, sediment and biota from four Iberian river basins (Llobregat, Ebro, Júcar, and Guadalquivir) and subsequently submitted to a prioritization exercise. The prioritization approach applied takes into consideration the frequency of detection and environmental levels found within the SCARCE project and reported acute toxicity values against three different species (Daphnia, algae, and fish) collected from the literature. The pollutants identified as most relevant in this context are pesticides and industrial chemicals. The toxic units (TU) approach was used to assess the risk of individual compounds at a site and then summed for all compounds present (concentration addition model ,CA) to assess the site-specific risk. The link between chemical pollution and aquatic macroinvertebrate communities *in situ* was examined by using four biological indexes: SPEAR ("Species at Risk Index") as the indicator of the decline of sensitive species in relation pollution; and Shannon and Margalef biodiversity indexes. The results suggested that organic chemicals posed a risk of acute effects at 42% of the sampling sites and risk chronic effects at all the sites. Metals posed an acute risk at 44% of the sites. The main drivers of the risk were pesticides and metals.

Land use occupation, physical and chemical stressors, and organic microcontaminants were investigated for single and conjoint effects on the biological communities (biofilms and invertebrates) in a set of impaired rivers. The diversity of invertebrates and the diatom communities were the best descriptors of the distribution patterns of the biological communities against the river stressors. The two biological descriptors decreased analogously according to the progressive site impairment (higher area of

agricultural and urban-industrial activities, high water conductivity, higher dissolved organic carbon (DOC), and dissolved inorganic nitrogen (DIN) concentrations, and a higher concentration of organic microcontaminants (particularly pharmaceutical and industrial compounds). A multivariate analysis of the redundancy (RDA) signaled that the river impairment leads to a general effect on the biological quality, especially in the most industrialized basins. The variance partition analyses of the RDA attributed the major share (10%) of the biological communities' response to the environmental stressors, followed by the land use occupation (6%) and the presence of organic microcontaminants (2%). However, the variance shared by the three groups of descriptors was very high (41%), indicating that their simultaneous effect determined most of the variation in the biological communities. The results indicate that the effects of stressors on biological communities may be synergistic and much higher than those corresponding to the simple addition of stressors. Stressors occurring at multiple spatial and temporal scales define a so-called stressor space with much higher effects than the ones attributed solely to organic microcontaminants or to excess nutrients.

This Deliverable exploits the SCARCE measured concentration data for four Spanish River Basins for the validation of the first parts of the model train that produce predicted environmental concentrations, both with respect to overall levels and with respect to spatial patterns. The specific strength of the SCARCE dataset is its high spatial resolution (next to a large number of chemicals).

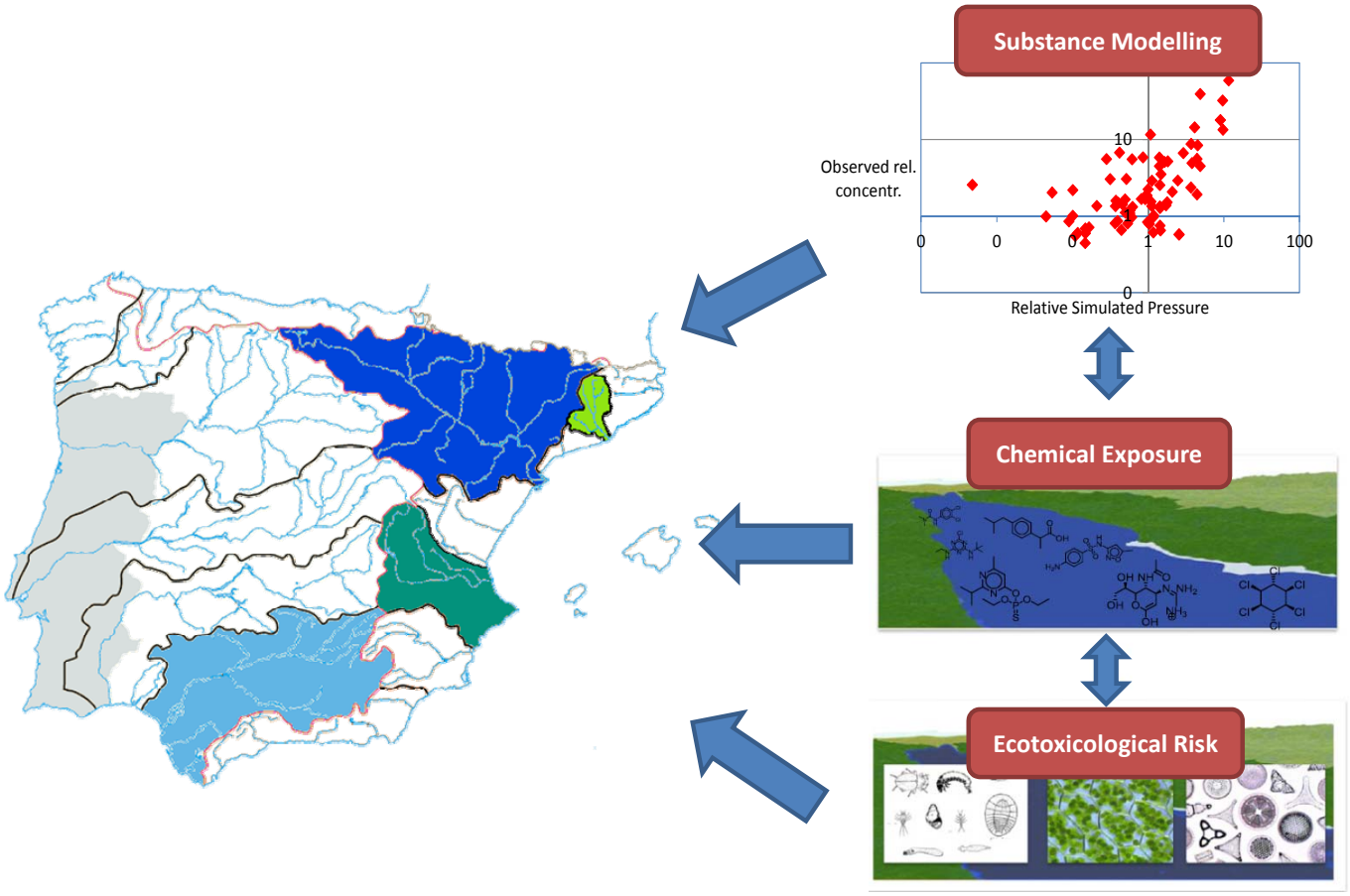
We found that the model train is often able to simulate individual pesticides or pharmaceuticals within one order of magnitude. This conclusion is supported by similar work in other case studies. For REACH chemicals the currently used methodology is insufficient to achieve that target: the hypothesis that adding "use category" information will resolve this still needs to be verified.

The model train produces spatial concentration ranges for individual chemicals that resemble spatial concentration ranges in the field data, and differences in ranges are according to expectations. By comparison of simulated and observed spatial patterns, we concluded that it is reasonable to assume that emissions of REACH registered chemicals and pharmaceuticals follow population density distribution.

We could not confirm that the emissions of pesticides follow agriculture land-use: this is a reason to carefully review our modeling methodology with respect to pesticides.

Finally a parallel prioritization exercise was performed using the NORMAN prioritization method for the water phase of the four Iberian rivers studied. 37 compounds were identified as specific pollutants of concern, which include 3 hormones, 6 industrial compounds, 15 pesticides, 3 personal care products and 10 pharmaceuticals. Top 10 rank compounds are the hormones 17-beta-Estradiol and Estrone, the pesticides Pyriproxyfen, Dichlofenthion, Diazinon, the industrial compounds PFOS and Bisphenol A, and the pharmaceuticals Ibuprofen, Diclofenac and Lorazepam.

1.2 [publishable] Graph



2. List of Contents

3. List of Abbreviations

4DHB	4,4'-Dihydroxybenzophenone
4HB	4- Hydroxybenzophenone
4MBC	4-Methylbenzildiene camphor
anti-DP	Dechlorane Plus Anti
APA	Alkaline phosphatase activity
BDE	Brominated diphenyl ether
BeP	Benzylparaben
BFR	Brominated flame retardant
BLM	Biotic ligand model
BP1	Benzophenone 1
BP3	Benzophenone 3
BPA	Bisphenol A
BT	1H-Benzotriazole
CA	Concentration addition
CAS	Chemical Abstracts Service
CBZ	Carbamazepine
DB	Database
DBDPE	Decabromodiphenyl ether (or BDE-209)
DCA	Detrended correspondence analysis
DDT	Dichlorodiphenyltrichloroethane
Dec 602	Dechlorane 602
Dec 603	Dechlorane 603
DES	Diethylstilbestrol
DIN	Dissolved inorganic nitrogen
DO	Dissolved oxygen
DOC	Dissolved organic carbon
E1	Estrone
E1-3G	Estrone-3-glucuronide
E1-3S	Estrone-3-sulfate
E2	17-β-estradiol
E2-17G	Estradiol-17-glucuronide
E3	Estriol
E3-16G	Estriol-16-glucuronide
E3-3S	Estriol-3-sulfate
EC50	Effective concentration 50%
ECOSAR	Ecological Structure Activity Relationships
EDC	Endocrine disrupting compounds
EE2	Ethinylestradiol
EHMC	2-Ethyl-hexyl-4-trimethoxycinnamate
EQS	Environmental quality standards

EtP	Ethylparaben
FDEA	Perfluorodecyl ethanoic acid
FHEA	Perfluorohexyl ethanoic acid
FOEA	Perfluorooctyl ethanoic acid
GIS	Geographic Information System
HCB	Hexachlorobenzene
HCH	Hexachlorocyclohexane
IOC	Industrial organic compounds
LC50	Lethal concentration 50%
LoD	Limit of detection
LoQ	Limit of quantification
MeP	Methylparaben
NP	Nonylphenol
NP_1EC	Nonylphenol monocarboxylate
NP_1EO	Nonylphenol monoethoxylate
NP_2EO	Nonylphenol diethoxylate
OC	Octocrylene
OD-PABA	Ethylhexyl dimethyl PABA
OP	Octylphenol
OP_1EC	Octylphenol monocarboxylate
OP_1EO	Octylphenol monoethoxylate
OP_2EO	Octylphenol diethoxylate
PABA	4-aminobenzoic acid
PCA	Principal component analysis
PCP	Personal care products
PET	Polyethylene terephthalate
PFBA	Perfluorobutanoic acid
PFBS	Perfluorobutanesulfonate
PFCs	Perfluorinated compounds
PFDA	Perfluorodecanoic acid
PFDoA	Perfluorooctadecanoic acid
PFDS	Perfluorodecanesulfonate
PFHpA	Perfluoroheptanoic acid
PFHxA	Perfluorohexanoic acid
PFHxDA	Perfluorohexadecanoic acid
PFHxS	Perfluorohexanesulfonate
PFNA	Perfluorononanoic acid
PFOA	Perfluorooctanoic acid
PFODA	Perfluorooctadecanoic acid
PFOS	Perfluorooctane sulfonic acid
PFOSA	Perfluorooctane sulfonamide
PFPeA	Perfluoropentanoic acid
PFTeDA	Perfluorotetradecanoic acid

PFTrDA	Perfluorotridecanoic acid
PFUdA	Perfluoroundecanoic acid
PhC	Pharmaceutical compounds
POPs	Persistent organic pollutants
PrP	Propylparaben
PVC	Polyvinyl chloride
RDA	Redundancy analysis
RI	Ranking index
SCI	Science Citation Index
SPEAR	Species at risk index
syn-DP	Dechlorane Plus Syn
TBEP	Tris (butoxyethyl) phosphate
TCC	Triclocarban
TCE	Tris (2-chloroethyl) phosphate
TCPP	Tris (l-chloro-2-propyl) phosphate
TCS	Triclosan
TOC	Total organic carbon
TP	Total phosphorus
TT	Tolytriazol
TU	Toxic units
U.S. EPA	United States Environmental Protection Agency
WFD	Water Framework Directive
WHO	World Health Organization
WWD	Wastewater Discharge

4. Introduction

The present deliverable is aimed at describing the main outcomes of Work Package C3 (WP C3) which is associated with Iberian river basins. Risk assessment under water scarcity is of great relevance to South European countries and was evaluated in several Iberian river basins in the frame of the SCARCE project funded by the Spanish Ministry of Economy and Competitiveness (CSIC). Water scarcity might be critical in affecting water pollution because of lowered dilution capacity of chemicals. The aim of WP C3 is to validate SOLUTIONS approaches and tools under the conditions of water scarcity in close connection with the SCARCE project. To this end specific objectives are:

- Prioritisation of pollutants specific to Iberian (Mediterranean) river basins;
- Assessment of the concurrent effects of chemical stress and hydrological scarcity on aquatic ecosystems;
- Provision of relevant bio-physical aquatic scenarios complementary to those addressed in the other case studies potentially contributing to the validation of the modeling tools developed in SP M.

The present deliverable compiles the outcomes of three tasks undertaken in WP C3, and previously described in the corresponding internal deliverables

- **ID C3.1** Report on the prioritization of pollutants occurring in Iberian Mediterranean basins (Responsible: CSIC)
- **ID C3.2** Report on the relationships between chemical pollution and environmental stressors and ecosystem effects in the Mediterranean river basins and as found in the SCARCE project (Responsible: CSIC)
- **ID C3.3** Report on the application of selected models to Iberian Mediterranean basins (Responsible: DELTARES)

Task C3.1. Priority and emerging compounds as contributors to water pollution in the Iberian Peninsula:

The studies performed so far in the Iberian Peninsula on the occurrence of emerging pollutants have shown contamination levels of, for instance, pharmaceuticals and drugs of abuse, higher in general than in other larger European basins. The main source of most pollutants in the aquatic environment is the discharge of wastewater treatment plants (WWTPs). Under drought situations the WWTP effluents may

represent almost 100% of the total flow of the rivers, showing potential hazardous consequences for the biota (including human) and the ecosystem. This situation is of special concern in the industrialized areas of the Mediterranean region, where scenarios of water scarcity can worsen the existing effects of human pressure. Within this task, a prioritized list of contaminants relevant to the Iberian rivers, elaborated on the basis of the occurrence of more than 200 compounds, including pharmaceuticals and personal care products, illicit drugs, polar pesticides, endocrine disruptors, perfluorinated compounds, polyhalogenated flame retardants, UV-filters, measured in water and biota in the context of SCARCE developed by a collaboration of SOLUTIONS and SCARCE using both approaches.
To facilitate the use of SCARCE project data by the SOLUTION consortium they were transferred to the NORMAN database, and are currently freely available to the project members.

Task C3.2.Ecological status in connection with chemical pollution and hydrological stress:
Results of the study of different biological descriptors associated to different trophic levels (macrophytes, phytoplankton, biofilms, benthic invertebrates, fish community) characterizing both ecosystem function and structure are related to the chemical and hydrological stress data (CSIC) to support or complement the findings of WP T1 and WP T5.

Task C3.3. Modeling of Iberian Rivers:
The SOLUTIONS project is developing a collection of integrated models, to increase our understanding of issues related to emerging chemicals in Europe's river basins. This collection of models is referred to as the "Model Train". The model train consists of four key building blocks: (a) the prediction of substance properties based on their molecular structure, (b) the simulation of emissions, (c) the simulation of fate & transport, and (d) the characterisation of the risk of mixtures of chemicals for human health and aquatic ecosystems. Thus, this task aims at validating the first components of the model train by a comparison between simulated concentrations and observed concentrations in Iberian rivers using the data gathered in the Spanish project SCARCE.
SCARCE data will contribute to check/validate the extension of the model train to Mediterranean River scenarios.

5. Report on the prioritization of pollutants occurring in Iberian Mediterranean basins (Internal Deliverable ID C3.1)

5.1. Introduction

Mediterranean rivers are largely different from Northern and Central European rivers in terms of hydrological regime, climate conditions (e.g. ambient temperature, solar irradiation), socio-economics (e.g. land use, tourism, kinds of crops, etc.), all of which leads to differences also in the relative importance of the environmental stressors, in the classes and levels of the pollutants found and their environmental fate, etc. Water scarcity might be critical in affecting water pollution because of lowered dilution capacity of chemicals. The studies performed so far in the Iberian Peninsula on the environmental occurrence of emerging pollutants have shown contamination levels of, for instance, pharmaceuticals and drugs of abuse, higher in general than in other larger European basins.

The main objective of this deliverable is identifying relevant emerging contaminants under conditions of water scarcity as occurring in South European countries/Iberian (Mediterranean) river basins in close connection with the recently finished SCARCE project funded by the Spanish Ministry of Economy and Competitiveness and coordinated by the CSIC partner (Navarro-Ortega et al., 2012a and 2012b).

The conjoint effects on aquatic ecosystems caused by priority and emerging pollutants together with the expected environmental pressures derived from global change are within the research focus of the SCARCE project. Four river basins, namely, Ebro, Llobregat, Júcar, and Guadalquivir, have been investigated in SCARCE. These rivers are representative of different scenarios from both the biophysical and the socio-economical point of view (Table 1). The Llobregat river basin, for instance, has a relatively small catchment area but it receives extensive urban and industrial wastewater discharges that cannot be diluted by its natural flow, provides water supply to Barcelona metropolitan area (ca. 4 M inhabitants), experiences periodic floods and droughts which lead to frequent morphological variations in the river bed, and waters have a high concentration of pollutants with important effects on the biological communities. In contrast, the Ebro river basin is largely regulated by dams and channels which have altered its hydrological and sedimentary regime, while abstraction of ground and surface water, irrigation, and industrial activities have in addition deteriorated soil and water quality. Special features of the Júcar and Guadalquivir river basins are the intensive agriculture activity that takes place in both basins and the navigable character of the Guadalquivir River in its last part.

Table 5.1. Some characteristics of the four Mediterranean river basins studied.

Basin	**Catchment Area** (km^2)	**River Length** (km)	**Mean Annual Precipitation** (mm)	**Mean Discharge** ($Hm^3\ y^{-1}$)	**Population Density** ($inhab\ km^{-2}$)
Llobregat	4957	165	650	620	545
Ebro	85362	928	672	13408	34
Júcar	21578	512	448	810	207
Gualdalquivir	57071	657	520	7230	69

The main source of most pollutants in the aquatic environment is the discharge of wastewater treatment plants (WWTPs). Moreover, under drought situations such as those frequently occurring in Southern Europe, the WWTP effluents may represent almost 100% of the total flow of the rivers, showing potential hazardous consequences for the biota (including human) and the ecosystem. This situation is of special concern in the industrialized areas of the Mediterranean region, where scenarios of water scarcity can worsen the existing effects of human pressure. Due to the great number of chemical compounds potentially occurring in these environments, there is a need to prioritize them for management optimization purposes. This deliverable addresses this issue by applying different prioritization schemes to monitoring data generated by the SCARCE project. The result is a set of prioritized lists of contaminants relevant to the monitored Iberian rivers, elaborated on the basis of the occurrence of approximately 200 compounds, including pesticides, alkylphenols, pharmaceuticals, drugs of abuse, hormones, personal care products, perfluorinated compounds and various industrial organic chemicals, in water, sediment and biota. This deliverable is linked to Task C3.1. Priority and emerging compounds as contributors to water pollution in the Iberian Peninsula.

5.2. Objectives

The main objectives of this work were (a) to assess the environmental risk associated to about 200 organic micropollutants, including both regulated and emerging contaminants, monitored in water, sediment and fish along four selected rivers located in the Mediterranean side of the Iberian Peninsula; and (b) to prioritize them in each of the investigated compartments taking into account their occurrence and ecotoxicological potential.

This deliverable was in principle expected to cover only the water and biota compartments and the Ebro and Llobregat river basins. However, since data are available also for sediments and for the Guadalquivir and Jucar rivers, these additional scenarios have been also included. The following sections describe briefly the methodologies applied in each of the compartments (water, sediment, and biota) scrutinized and the main results obtained in each case.

5.3. Water

5.3.1. Methodology

5.3.1.1. Target Analytes

The list of target compounds in water included over 200 organic priority and emerging contaminants (Annex I), belonging to the classes of pesticides (49), pharmaceuticals and hormones (90), perfluorinated compounds (PFCs) (22), alkylphenols and other industrial organic compounds (14), drugs of abuse (8) and personal care products (19).

5.3.1.2. Samples

The study area included the four aforementioned Iberian river basins, representative of Mediterranean streams. The main features of the studied rivers such as catchment area, river length, annual precipitation, population density, etc., are described in detail in (Kuzmanović et al. 2015). Grab water samples for chemical characterization were collected at 77 selected locations along the Llobregat (15 sites), Ebro (23 sites), Júcar (15 sites) and Guadalquivir (24 sites) river basins in two monitoring campaigns (Autumn 2010 and 2011). Monitoring sampling sites are shown in Fig. 5.1.

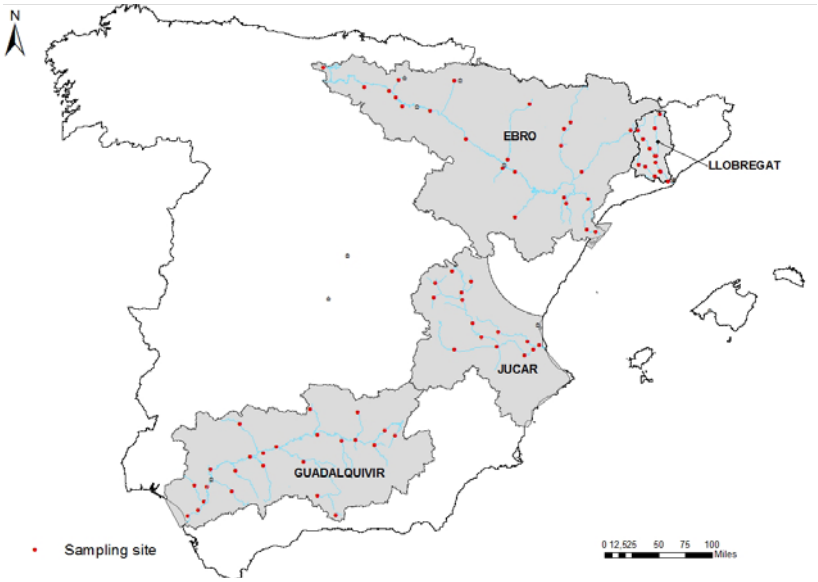


Figure 5.1: Basins under study and location of monitoring stations

5.3.1.3. Prioritization

The prioritization approach applied is based on a ranking index (RI) that considers for each monitored compound both its occurrence (including frequency of detection and measured environmental levels) and its ecotoxicological potential expressed as Toxic Units (TU) (Sprague, 1970):

$$TU_{i\ (algae,\ Daphnia,\ fish)} = c_i / EC50_i \text{ or } LC50_i \qquad \text{(Equation 1)}$$

where TU_i is the toxic unit of a compound i; c_i measured concentration (µg/l) of the compound in the water phase; $EC50_i$ or $LC50_i$ (µg/l) effective or lethal concentration of 50% of individuals when exposed to the substance concerned. The toxicity data of each chemical was collected for three standard test species (green algae *Pseudokirchneriella subcapitata*, invertebrate *Daphnia magna* and fish *Pimephales promelas*) representative of different trophic levels, as recommended by the Water Framework Directive (WFD). Data were collected from peer-reviewed literature and databases (Annex II). Missing toxicity data were estimated by ECOSAR (Annex II).. For prioritization purposes, a 'ranking index' (RI) was developed which is a slight modification of prioritization approach developed by von der Ohe et al. (von der Ohe, Dulio et al. 2011). It is applicable to every compound in a certain area of study (here a river basin) and considers both the toxic units of the compound and their distribution in the area studied. To this end, six logTU ranges or classes were arbitrarily defined as specified in Table 5.2, which cover the typical occurrence values found in environmental samples. Rank frequencies f_x expressed as the fraction of sites (as a percentage) in the river basin where compound's logTU belongs to the specific rank class x are determined by the following Equation:

$$f_x = n_x/N_{total}\ (\%) \qquad \text{(Equation 2)}$$

Where n_x is a number of sites in the river basin falling in rank class x, N_{total} is the total number of sites per river. Sum of all the rank frequencies is equal to 100% as it covers all the sampling sites in the river basin. The compound's ranking index (RI) in the basin under study is defined by summing up the frequencies f_x multiplied by certain arbitrary weights w_x (Equation 3), (Table 5.2):

$$RI = \sum f_x \cdot w_x = (f_1 \times 1) + (f_2 \times 0.5) + (f_3 \times 0.25) + (f_4 \times 0.125) + (f_5 \times 0.0625) + (f_6 \times 0.0) \qquad \text{(Equation 3)}$$

The ranking index is scaled from 0 to 100, where 100 means that compound's log transformed TU is higher than 0 in all sites in the sampled river, and 0 that compound's log TU is not exceeding the value of -4 in any site. Log TU higher than 0 means that the concentration measured exceeds the EC50 value of the compound, which is the threshold for acute effects risk of standard test species concerned. The sixth rank was given the value 0 for those log TU that were less than -4 which stands for 1/10 000 of the EC50 value and it is expected not to cause short term or long term effects in the ecosystem in most of the cases (Liess and Von Der Ohe 2005, Beketov, Foit et al. 2009). Since ranking indexes are related to toxic units, they must be calculated for each test species (algae, Daphnia and fish).
For more details see (Kuzmanović et al. 2015).

Table 5.2. Definition of the six rank classes, their interval ranges and weights used in the calculation of the Rank Index.

Rank class ***x***	**Range** ***Log TU***	**Weight** w_x
1	> 0	1
2	<0,-1>	0.5
3	<-1,-2>	0.25
4	<-2,-3>	0.125
5	<-3,-4>	0.0625
6	<-4	0

5.3.2. Results

Tables 5.3 and 5.4 show the compounds identified as most relevant in the water phase for each of the four Mediterranean basins investigated according to the RI results obtained. Table 5.3 includes the compounds

showing the highest risk indexes (RI ≥ 12.5%) corresponding to high toxic units (as per Table 5.2) in many sampling sites, and Table 5.3 those with high or medium risk indexes (i.e., 12.5% > RI>0) corresponding to high TUs in few sampling sites or medium/low TUs in many sampling sites (as per Table 5.2. Out of the ten compounds found to present very high risk, eight were pesticides and two were pollutants of industrial origin. The group of pesticides included six insecticides (chlorfenvinphos, chlorpyriphos, dichlofenthion, ethion, diazinon, and carbofuran); one fungicide (prochloraz) and one herbicide (diuron), and the two industrial chemicals were nonylphenol (NP) and octylphenol (OP), both breakdown products of polyethoxylated alkylphenol surfactants. *Daphnia* seems the most sensitive species regarding these compounds (Table 5.3). Interestingly, five of these top ten chemicals are within the list of priority pollutants of the WFD (EU Dir, 2013/39), namely, chlorfenvinphos, chlorpyriphos, diuron, nonylphenol, and octylphenol.

Chlorpyriphos and diazinon appeared important in all four river basins, chlorfenvinphos in the Ebro, Júcar and Guadalquivir, dichlofenthion in Ebro and Júcar, prochloraz and ethion in Júcar, and carbofuran, diuron, and octylphenol only in the Llobregat basin.

Table 5.3. Compounds with hig hrisk indexes (RI ≥ 12.5%) in water in the investigated Mediterranean basins.

Compound	Llobregat			Ebro			Júcar			Guadalquivir		
	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish
Chlorfenvinphos					X			X			X	X
Chlorpyriphos		X	X		X			X	X		X	
Dichlofenthion					X	X		X	X			X
Prochloraz							X					
Ethion								X				
Diazinon		X			X			X			X	
Carbofuran		X										
OPs/NPs		X									X	
Diuron	X											

Table 5.4. Compounds with high or medium risk indexes (12.5% > RI>0) in water in the investigated Mediterranean basins.

Compound	Llobregat			Ebro			Júcar			Guadalquivir		
	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish
Sertraline	X	X		X			X					
Triclosan	X			X			X			X		
Parathion-Ethyl					X			X				
Caffeine	X			X			X			X		
Terbutryn	X			X								
Isoproturon	X			X								
Losartan	X			X	X							
Imazalil				X		X	X	X	X			
Tolytriazol	X	X		X						X		
Simazine	X			X						X		
Atrazine				X			X			X		
Azinphos Ethyl		X			X						X	
Malathion		X	X		X			X	X		X	X
Azinphos-Methyl		X			X							
Thiabendazole					X							
Methiocarb		X			X						X	
Venlafaxine	X	X		X	X							
Gemfibrozil			X									X
Pyriproxyfen						X			X			

5.4. Sediment

5.4.1. Methodology

Though not initially considered to be included in this deliverable, sediments, collected in parallel with the water samples discussed above, were also investigated and the most relevant pollutants identified. In this case, prioritization followed the same scheme as for free waters but considering the concentration of the pollutant in the pore-water (C_{PW}) according to the following equation:

$$C_{PW} = C_S/(f_{OC} \times K_{OC})$$

Where C_S is the measured concentration of the pollutant in the sediment, f_{OC} the fraction of organic carbon content of the sediment and the K_{OC} is the partition coefficient between organic carbon-water partition coefficient of the substance (Di Toro et al., 1991).

5.4.2. Results

Tables 5.5 and 5.6 show the compounds identified as most relevant in sediment for each of the four Mediterranean basins investigated according to the results of the prioritization exercise performed. Table 5.5 includes the compounds showing very high-risk indexes and Table 5.6 those with high or medium risk indexes. As can be seen, the first, more risky group of compounds includes five pesticides, a surfactant (nonylphenol) and an antibiotic (ciprofloxacin), i.e., a profile pretty similar to that observed in the aqueous phase.

Table 5.5. Compounds with high risk indexes (12.5% > RI>0) in sediments in the investigated Mediterranean basins.

Compound	Llobregat			Ebro			Júcar			Guadalquivir		
	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish
Chlorpyriphos		X	X		X	X		X	X		X	X
Chlorfenvinphos											X	
Nonylphenol	X	X	X								X	X
Diazinon		X						X			X	
Malathion								X				
Ciprofloxacin										X	X	
Methiocarb								X				

Table 5.6. Compounds with high or medium risk indexes (high toxic units in few sampling sites or low toxic units in many sampling sites) in sediments in the investigated Mediterranean basins.

Compound	Llobregat			Ebro			Júcar			Guadalquivir		
	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish	Algae	Daphnia	Fish
Octylphenol	X	X	X	X	X		X	X				
Dexamethasone	X			X						X		
Acetaminophen		X			X						X	
NP1EC	X	X	X	X	X	X				X	X	X
Ofloxacin		X			X			X			X	
Carbendazim		X	X		X	X						
Dimetridazole							X	X		X	X	X
Triclosan	X	X		X	X	X				X	X	X
Cocaine	X		X								X	X
Sertraline	X	X	X	X			X			X	X	
NP2EO			X			X			X			X
Bisphenol A	X	X	X			X	X	X	X			X
(-)-Δ9-THC	X	X	X			X			X	X	X	X
Metronidazole	X			X			X			X	X	X
Thiabendazole		X	X				X	X	X		X	X
Fluvastatin	X	X	X			X			X	X		X
Propylparaben	X	X		X	X	X				X	X	X
Tebuconazole		X	X					X			X	
TCS	X	X		X	X	X				X	X	X
Imazalil										X	X	X
Tolytriazol	X	X	X	X	X	X	X	X	X	X	X	X
Trazodone	X	X	X	X			X			X		
1h Benzotriazole	X	X	X							X	X	X
Carazolol							X		X			X
Albendazole	X			X						X		
Loratidine	X	X	X							X	X	X
Ketoprofen			X			X			X			X
Diazepam	X	X										
Methadone Hydrochloride										X	X	X
Estradiol	X	X	X									
Trichlorocaraban	X	X		X								
PFBA										X	X	X

5.5. Biota

5.5.1. Methodology

5.5.1.1. Target Analytes

A total of 135 emerging contaminants (see Table 5.7) were analyzed in biota (fish tissue). The list of compounds investigated included 19 endocrine disrupting compounds (EDCs), 21 perfluorinated compounds (PFCs), 51 pesticides (21 organophosphorus, 8 pyrethroids, 4 carbamates, 6 triazines, 2 ureas, 3 chloroacetamides, and 7 other compound), 20 pharmaceuticals, 8 UV filters, 9 brominated diphenyl ethers (BDEs), 3 emerging brominated flame retardants (BFRs) and 4 halogenated norbornenes.

5.5.1.2. Samples

Fish samples (a total of 48) from 14 different species were collected from five selected sampling stations from each of the four Mediterranean rivers investigated during 2010. Figure 5.2 shows the location of the sampling sites within each basin and the fish species collected at each basin.

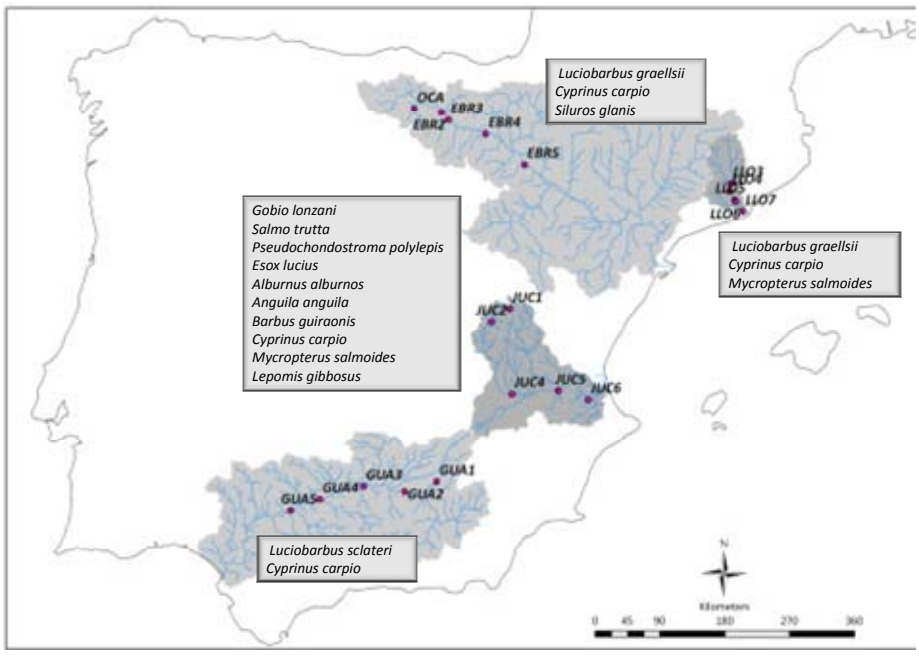
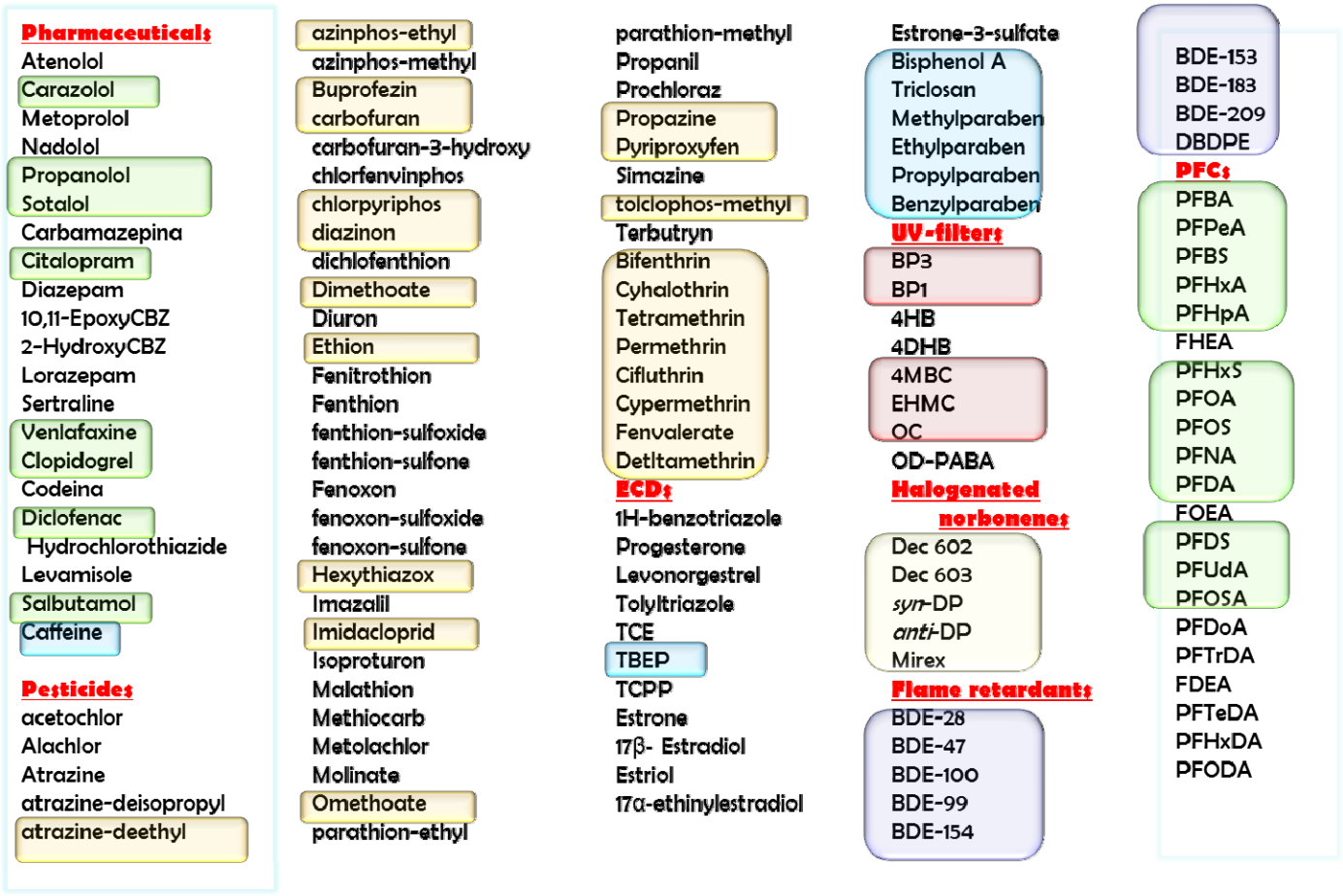


Figure 5.2. Selected sampling sites and fish species collected at each of the four Mediterranean river basins monitored (from North to South: Ebro, Llobregat, Júcar, and Guadalquivir).

Table 5.7. Pollutants (grouped by chemical class) monitored in fish and those detected in at least one sample marked with boxes.



5.5.2. Results

Of the 135 organic pollutants monitored, 54 (those included in Table 5.7 within boxes) appeared in fish tissues. All groups of pollutants were detected.

Figures 5.3 and 5.4 show the forty compounds presenting the highest detection frequencies and maximum concentrations, respectively. The top ten most frequently detected compounds were: perfluorooctane sulfonic acid (PFOS), dechlorane Plus Anti, cis-bifenthrin, cyhalothrin, cypermethrin, tris (butoxyethyl) phosphate (TBEP), dechlorane Plus Syn, BDE-47, dechlorane 603 and fenvalerate, while the top ten detected in highest concentrations case were perfluorohexanoic acid (PFHxA), perfluoropentanoic acid (PFPeA), chlorpyriphos, PFOS, carbofuran, dechlorane 623, BDE-47, dechlorane 602, 2-ethyl-hexyl-4-trimethoxycinnamate (EHMC), and bisphenol A. Thus, overall, PFCs, halogenated flame retardants and pesticides appear to dominate in this matrix. If both detection frequency and maximum/average

concentrations are jointly considered the most relevant compounds in fish are: the regulated perfluorinated compound PFOS, the pyrethroid insecticides cypermethrin (also included in the list of priority pollutants in water) and permethrin, the regulated flame retardant BDE-47, the halogenated flame retardant dechlorane 602, and the personal care product and suspected endocrine disruptor methylparaben.

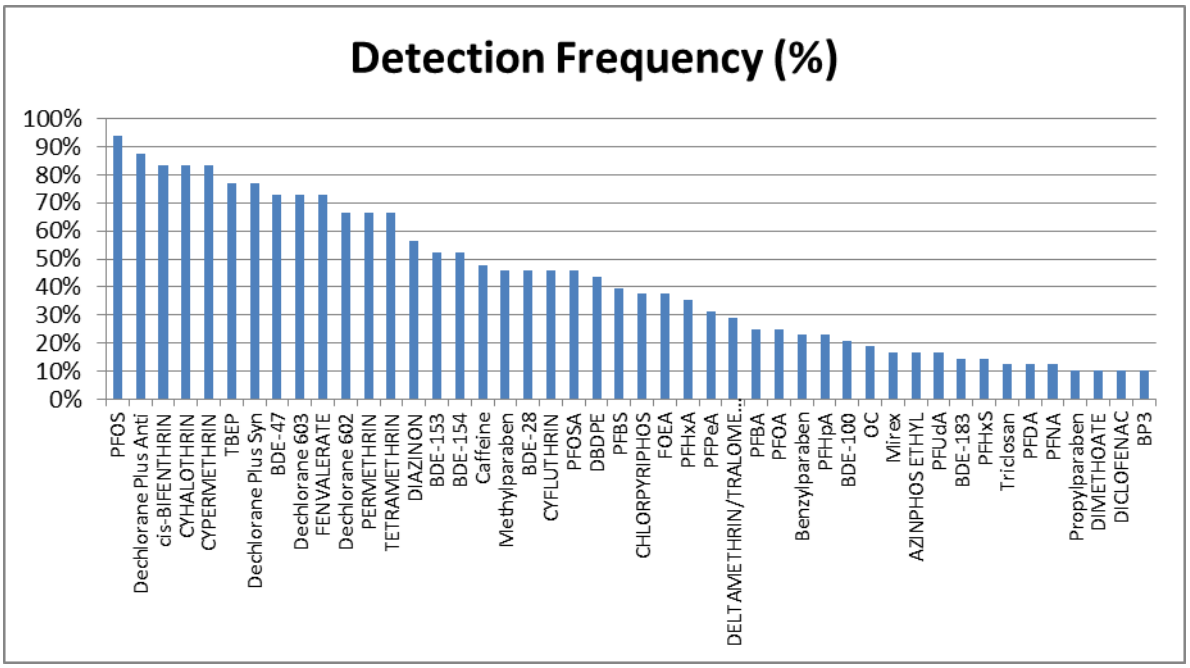


Figure 5.3. Compounds showing the highest detection frequencies in biota.

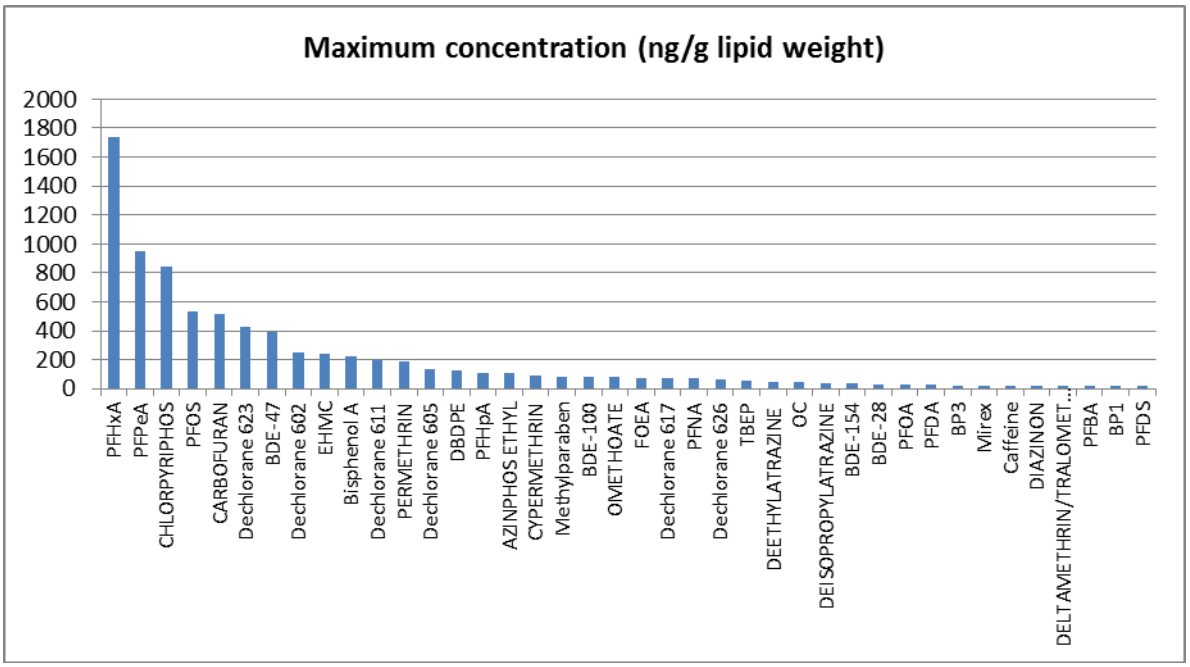


Figure 5.4. Compounds showing the highest maximum concentrations in biota.

Two of these substances, namely, BDEs and PFOS, are subject to environmental quality standards (EQS) in biota (see Table 5.7). The concentrations of BDEs in the SCARCE project are expressed in ng/g lipid weight whereas the established EQS is expressed in ng/g wet weight. If we consider a low fish lipid content of 0.5 % (Geyer et al., 1994) and correct the measured values accordingly, 37 out of the 48 samples analysed within SCARCE would surpass the BDEs EQS of 0.0085 ng/g, while if we consider a lipid content of 20% the number of samples surpassing the EQS increases to 41.
In contrast, in the case of PFOS, the concentrations reported in the SCARCE project are expressed in ng/g dry weight and if we consider an average correction factor of 10% to convert concentrations from dry weight to wet weight, only 6 of the samples analysed would surpass the PFOS EQS of 9.1 ng/g wet weight.

Table 5.7. EQSs established for substances measured in this study in biota according to Directive 2013/39/EC.

Substance	**CAS number**	**EQS (µg/Kg)**
BDEs	32534-81-9	0.0085
PFOS	1763-23-1	9.1

According to the OECD, substances with octanol-water partition coefficients higher than 3 (log Kow > 3) show tendency for bioaccumulation (OECD guidelines for the testing of chemicals, 2008). In the SCARCE study, the majority of the substances most frequently detected in fish followed this 'Kow>3' rule. However, there were also some exceptions whose behavior requires a more in-depth analysis.

In addition to this, a preliminary inspection of the raw SCARCE biota data has shown:

- geographical differences (e.g. UV filters were much more frequently detected in the Guadalquivir (up to 80 %) than in the other rivers (up to 18% of positive samples), while pesticides were mostly detected in the Júcar river; similarly, fish samples taken near the mouth of the rivers commonly presented higher concentrations than those taken at the sources),
- clear relationships between the pollutant levels and their co-occurrence in fish with the land use of the area,
- comparatively higher concentrations in big predators (European catfish) and in bottom-feeding fish (Barbs and Carp) than in other species,
- higher concentrations also in adult individuals than in younger ones.

Nevertheless, all these results are currently being examined in detail and a manuscript with the main overall conclusions is in preparation and should be available soon in the scientific literature.

5.6. Concluding remarks

The pollution status and effects of emerging contaminants are still largely unknown because nowadays the only data frequently available are concentrations of particular contaminants in water. Information on their occurrence in sediments and biota, co-occurrence, synergistic effects and bioaccumulation and biomagnification through the aquatic food web is scant. Within SCARCE, a number of emerging contaminants, including EDCs, pesticides, pharmaceuticals, PFCs and flame retardants have been detected in water, sediment and fish samples from the four most important Spanish Mediterranean river basins. Overall, the most relevant compounds considering the frequency of detection, concentration, and acute toxicity data belong to the classes of pesticides, alkylphenols, perfluorinated compounds and halogenated flame retardants, and their profile varies depending on the river basin and the matrix investigated.

Aquatic organisms may transfer the contaminants that they bioaccumulate from water or sediment to predators that forage on them. The extent to which these contaminants can move through aquatic food webs and thus potentially affect organisms at higher trophic levels is a crucial issue for environmental decision-making. The co-occurrence of a variety of emerging contaminants in some fish demonstrates that aquatic animal life is constantly exposed to low concentrations of biologically active substances, so mixture effects by a plethora of substances have to be scrutinized.

Much work is thus needed on the complex but highly relevant question of contaminant mixtures and multiple stressors and how they jointly act on the aquatic communities. Similarly, additional research is needed to develop ecosystem models that describe and predict both direct and indirect effects of contaminants on a variety of aquatic habitats. Emerging contaminants have arisen as a word wide-scale problem and, as such, much more effort should be devoted to providing global solutions.

5.7. References

Beketov, M. A., K. Foit, R. B. Schäfer, C. A. Schriever, A. Sacchi, E. Capri, J. Biggs, C. Wells and M. Liess (2009). "SPEAR indicates pesticide effects in streams - Comparative use of species- and family-level biomonitoring data." Environmental Pollution 157(6): 1841-1848.

EU Dir. 2013/39, 2013. Directive of the European Parliament and of the Council amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy. Official Journal of the European Union L226/1 (24.8.2013).

Di Toro, D.M., Zarba, C.S., Hansen, D.J. , Berry, W.J., Swartz, R.C., Cowan, C.E., Paulou, S.P. Allen, H.E., Thomas, N.A., Paquin, P.R. 1991. Technical basis for establishing sediment quality criteria for non-iionic organic chemicals using equilibrium partitioning. Environ. Toxicol. Chem. 10(12), 1541-1583

Geyer H.J., Scheunert I., Brueggemann R., Matthies M., Steinberg C.E.W., Zitko V., Kettrup A., Garrison W. 1994. The relevance of aquatic organisms' lipid content to the toxicity of lipophilic chemicals: Toxicity of lindane to different fish species. Ecotox. Environ. Saf. 28: 53-70.

Kuzmanović M., Ginebreda A., Petrović M., Barceló D. 2015. Risk assessment based prioritization of 200 organic micropollutants in 4 Iberian rivers. Sci. Total Environ. 503–504, 289–299.

Liess, M. and P. C. Von Der Ohe (2005). "Analyzing effects of pesticides on invertebrate communities in streams." Environmental Toxicology and Chemistry 24(4): 954-965.

Navarro-Ortega A., Sabater S., Barceló D. 2012a Understanding effects of global change on water quantity and quality in river basins – The SCARCE Project. Environ. Sci. Pollut. Res. 19(4), 915-917.

Navarro-Ortega A., Acuna V., Batalla R.J., Blasco J., Conde C., Elorza F., Elosegi A., Frances F., La-Roca F., Munoz I., Petrovic M., Pico Y., Sabater S., Sanchez-Vila X., Schuhmacher M., Barceló D. 2012b Assessing and forecasting the impacts of global change on Meiterranean rivers. The SCARCE Consolider project on Iberian basins. Environ. Sci. Pollut. Res. 19(4), 918-933.

Sprague, J.B. 1970. Measurement of pollutant toxicity to fish. II. Utilizing and applying bioassay results. Water Res. 4, 3-32.

OECD guidelines for the testing of chemicals. Bioaccumulation in Sediment-dwelling Benthic Oligochaetes. 2008.

Von der Ohe P.C., Dulio V., Slobodnik J., De Deckere E., Kühne R., Ebert R.-U., Ginebreda A., De Cooman W., Schüürmann G., Brack W. 2011. A new risk assessment approach for the prioritization of 500 classical and emerging organic microcontaminants as potential river basin specific pollutants under the European Water Framework Directive. Sci. Total Environ. 409, 2064–2077.

6. Report on the relationships between chemical pollution and environmental stressors and ecosystem effects in Mediterranean river basins as found in the SCARCE project (Internal Deliverable ID C3.2)

6.1. Introduction

Rivers are net receivers of chemical stressors from the anthropogenic origin, including organic matter and inorganic nutrients (phosphorus, nitrogen), and many organic micropollutants (Meybeck 2004) such as pesticides or industrial products. In impaired rivers, other stressors also come to play a role and co-occur with these chemicals, with specific effects and different manifestation in time and space (Stevenson and Sabater 2010). Amongst these stressors, habitat alteration, interruption of flow water regime, or higher water temperature, complicate the survival and life cycle of organisms. This is especially evident in the case of those sensitive organisms and is at the base of local extinctions and the overall decrease of biodiversity (Dudgeon 2010). These alterations perform as additional environmental filters (Poff 1997, Angermeier and Winston 1998, Malmqvist 2002), and condition the composition and relative abundance of species in the riverine biological communities.

Persistent chemical pollution may become a prevalent driver with respect to other stressors in impaired freshwater ecosystems (Malaj et al. 2014). Organic microcontaminants constitute complex mixtures that may differ according to the prevailing land uses, i.e., extensive agriculture, industrial activities, or human conurbations (Posthuma et al. 2008). The composition and concentration of micropollutants also vary between periods of the year depending on their use, and because of higher or lower water discharge (Petrovic et al, 2011). Their concentration may be enhanced or moderated according to the dilution capacity of the receiving river; arid and semiarid basins, but also those subjected to water abstraction (Barceló and Sabater 2010), have low dilution capacity and are candidates to higher effects.

Micropollutant effects not only depend on their concentration but also on the pollutants mixture (Altenburger et al. 2015) and their specific mode of action (Cleuvers 2003). In impaired rivers these organic microcontaminants mix up with nutrients in excess, or with the abundant dissolved organic matter, especially in systems heavily impacted by industrial or urban effluents (Hatt et al. 2004), making up a complex co-occurrence of stressors with effects on biological communities difficult to attribute to any of them (Segner et al. 2014).

The biota inhabiting freshwater ecosystems is the final receptor of this large diversity of influences. It was already shown a long time ago that biological communities were modulated by chemical pressures, such as the dissolved organic matter (evaluated by means of the DOC or TOC), as well as by nutrients in

excess, or by heavy metal pollution (Margalef, 1960 and Goodnight, 1956). These observations can be placed at the basis of the modern use of organisms and communities as indicators of the ecological status of ecosystems, with expressions such as water quality indices (Lecointe et al., 1993 and Armitage et al., 1984) or multimetric approaches (e.g. Fore et al., 1996). Altogether, these applications are based on the evidence that bacteria, algae, invertebrates, or fish, had characteristic ways to respond to the occurring stressors. The specific responses of each group of organisms are related to their particular life cycle and the habitat that they occupy and translate in specific roles in the energy and matter flux in the ecosystem. Shorter life-cycle organisms (bacteria, algae) may respond to rapid changes occurring in the river environment, both physical (temperature, salinity, pH) and chemical (nutrient abundance, organic matter availability) and biological (grazing, predation). The ones occupying the interphase between water and sediments (biofilms) can be the most responsive to short-term changes of this nature (Blanck et al., 1988 and Sabater et al., 2007). On the other hand, longer life-cycle organisms (invertebrates, fish) are able to integrate the long-lasting changes produced in the environment in their physiological status and population dynamics, and may, therefore, be responsive to chemical alterations, but also to physical stressors (hydrological alterations, habitat impairment, altered temperature regime), and as such can be good indicators of persistent stress (Bonada et al., 2006, Boix et al., 2010 and Johnson and Hering, 2009).

The recent awareness that organic anthropogenic substances may enter freshwaters in relatively high concentrations, and that they may affect biological communities (Beketov et al., 2013) has triggered huge efforts to understand their relevance for the ecosystem (Luo et al., 2014). At least some of these substances are able to bioaccumulate and propagate throughout the trophic web (Geyer et al., 2000 and Arnot and Gobas, 2004), and may affect the composition and performance of biological communities (Muñoz et al., 2009, Ricart et al., 2010 and Ginebreda et al., 2014). Their overall relevance when other disturbances also occur (organic matter or inorganic nutrients in excess, high concentrations of solutes such as chloride, hydrological pressures) is unclear, even despite recent indications of the potential relevance of microcontaminants (Liess et al., 2013). An obvious reason for these different perspectives is that matching potential effects to real consequences is not straightforward for the biological communities. Organisms are net receivers of influences at multiple spatial and temporal scales, and their ultimate response defines the carrying capacity of a system (Posthuma et al., 2014). Spatial influences range from basin-scale to reach-scale, that is, from general to local and temporal scales may determine quick or accumulative changes, and translate differently to the organisms in relation to their size and life cycle. This complexity is obvious at the ecosystem level, where multiple vulnerabilities of biological communities co-exist according to their position in the trophic web and evolutionary traits (Segner et al., 2014).

Extensive field studies combining chemical and biological analyses allow for the definition of potential patterns and causes of distribution of the biological communities. Multivariate analyses allow performing joint ordination of several sets of physical, chemical and biological variables assembled from the field, and also to define the distribution patterns of organisms according to the driving pressures in a given set of sites. This is a correlational approach with recognized weaknesses (Legendre and Legendre, 1998), but also sufficiently powerful to define emerging patterns on the structure of ecological data (Legendre and Legendre, 1998). Such an approach may help understand the degree to which the co-occurring stressors affect the community structure of microbial organisms (biofilms, including primary producers and heterotrophs), and invertebrate consumers (herbivores, detritivores, and predators). While biofilms may be more sensitive to inorganic nutrients (Sabater et al., 2000) and to some organic micropollutants (e.g. herbicides, antibiotics; Proia et al., 2013 and Pesce et al., 2011), invertebrates may better respond to habitat alteration as well as to other contaminants such as estrogenic substances, insecticides and even toxic nutrient concentrations (Muñoz et al., 2009, Camargo et al., 2005, De Castro-Català et al., 2013, Azevedo et al., 2015 and Liess et al., 2013). No doubt, the joint response of ones and the others, if produced, may represent the force of evidence of the impact of multiple disturbances on the river ecosystem, and hopefully might define situations and periods when these effects are more obvious. This principle has been recognized in several legislative frameworks (e.g. the European WFD, or the Clean Water Act in the US), where the response of these groups of organisms is considered complementary. The potential patterns derived from field-based exercises may shed light to potential causalities that in subsequent experimental approaches (Sabater et al., 2007 and López-Doval et al., 2010) can be tested for their consistency and mechanisms.

In the present work, sites of four different Mediterranean basins were sampled for their physical (water flow, temperature, land uses), chemical (inorganic nutrients, conductivity, organic micropollutants, metals), and biological (invertebrates, biofilm) descriptors. The patterns of biological communities with respect to the environmental and organic chemical pressures were explored in a variety of situations and multiplicity of scales. The main hypotheses were that, i) biofilms and invertebrates would show corresponding responses to the co-occurring stressors, and ii) that the main effects on the distribution of the biological communities will be produced in the sites where organic micropollutants co-occur with other stressors (nutrients, hydrological disturbances). Finally, we examined the potential relationship between the ecotoxicity associated with local mixtures of pollutants and aquatic macroinvertebrate biological community responses using four different metrics: Shannon and Margalef biodiversity indexes and $SPEAR_{pesticides}$ and $SPEAR_{organic}$

Full details concerning the results presented in this report can be found in Kuzmanovic et al., 2015,

Sabater et al., 2016, and Ponsati et al., 2016.

6.2. Objectives

This deliverable is linked to Task C3.2. "Ecological status in connection to chemical pollution and hydrological stress" and is based on the outcome of the finished SCARCE project funded by the Spanish Ministry of Economy and Competitiveness and coordinated by the CSIC partner (Navarro-Ortega et al., 2012).
Its main objective is to provide results of the study of different biological descriptors associated to different trophic levels (macrophytes, phytoplankton, biofilms, benthic invertebrates, fish community) characterizing ecosystem function and structure in relation to chemical and hydrological stress data.

6.3. Methodology

6.3.1. Study Area

Four Mediterranean river basins of the Iberian Peninsula (Llobregat, Ebro, Júcar, and Guadalquivir) were used in this study. These basins drain a large part of the eastern and the southern Iberian Peninsula and are mainly governed by Mediterranean climate (Sabater et al. 2009) (Table 6.1). A total of 19 sites were selected in the main course of the rivers: 5 in the Ebro (E1, E2, E3, E4 and E5), 5 in the Llobregat (L1, L2, L3, L4 and L5), 5 in the Júcar (J1, J2, J3, J4 and J5) and 4 in the Guadalquivir (G1, G2, G3 and G4) (see location in Fig. 6.1). The sampling of water for chemical analyses and biological communities' characterization was performed at the end of the summer period in two consecutive years (2010 and 2011). The upstream sites in each of the basins were moderately impaired, but the others showed varying degree of impairment by specific inorganic and organic pollution including agricultural, urban and industrial sources, hydrological alterations, and urbanization. The land uses corresponding to the associated basin area in each of the sites was estimated by GIS analysis using the Corine Land Cover database of 2006 (level 1 of classification of 4 classes, namely forested and semi-natural, artificial (urban, industrial), agricultural, and others). The size of the cells was 100x100m. Sub-basins were defined using the extension ArcHydro of ArcGIS

Table 6.1. Some characteristics of the four Mediterranean river basins studied.

Basin	**Catchment Area** (km^2)	**River Length** (km)	**Mean Annual Precipitation** (mm)	**Mean Discharge** ($Hm^3\ y^{-1}$)	**Population Density** ($inhab\ km^{-2}$)
Llobregat	4957	165	650	620	545
Ebro	85362	928	672	13408	34
Júcar	21578	512	448	810	207
Gualdalquivir	57071	657	520	7230	69

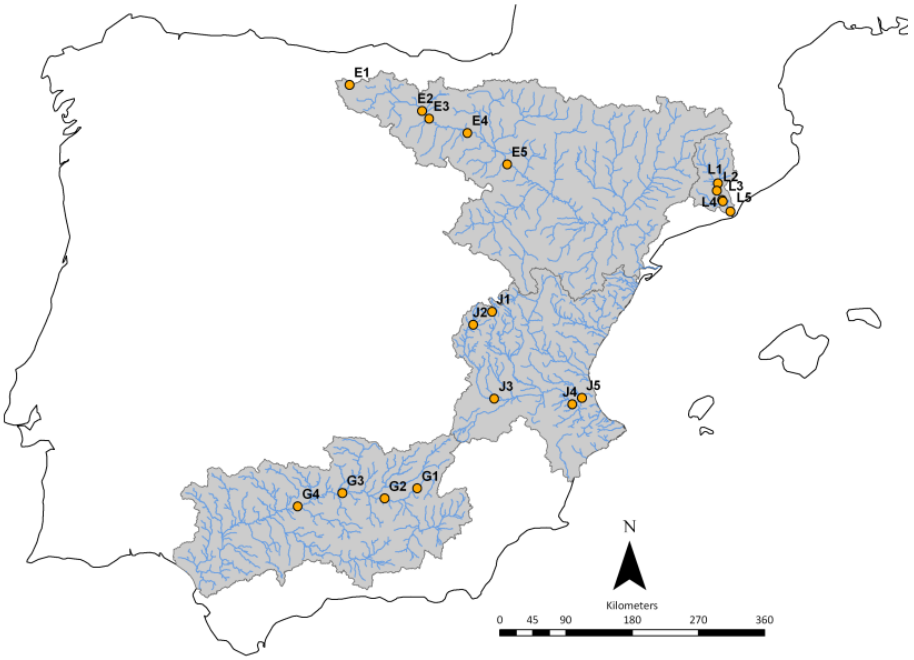


Figure 6.1. Map of the four Mediterranean basins studied (Iberian Peninsula), showing the location of the sites in each of the basins. Llobregat basin: L1, L2, L3, L4, and L5. Ebro basin: E1, E2, E3 E4 and E5. Júcar basin: J1, J2, J3, J4, and J5. Guadalquivir basin: G1, G2, G3, and G4.

6.3.2. Measurements

6.3.2.1. Physical and Chemical Measurements.

Water flow in each sampling site was obtained from daily measurements from the nearest gauging station (data provided by local water agencies) during the 15-day period before the sampling date. In case that no direct measurements were available, the drainage-area ratio of each sub-basin was used to extrapolate the corresponding data. The coefficient of variation (cv) of water flow (Q (cv)) was calculated and used as an estimate of the flow variability in each site throughout the hydrological period. The physical and chemical

variables measured in each site included dissolved oxygen (DO), water temperature (T), pH, and conductivity. Hand probes (WTW multiline 3310, and YSI ProODO, Yellow Springs, OH, USA) were used for the in situ measurements. Water samples for $NH4^+$ and $NO3^-$ analyses (considered jointly as DIN), Dissolved Organic Carbon (DOC), and total phosphorus (TP), were filtered with glass fiber filters (Whatman GF/F) in situ and kept frozen at -20 ºC until analysis. Nitrate was analyzed by ion chromatography (DIONEXIC5000; Dionex Corporation, Sunnyvale, USA) and the concentrations of ammonium and phosphate were determined colorimetrically (Alliance-AMS Smartchem 140 spectrophotometer, Frepillon, France). The DOC concentration was analyzed on a Shimadzu TOC-V CSH (Shimadzu Corporation, Kyoto, Japan).

At each site, grab water samples were taken for chemical analyses of the organic micropollutants and dissolved metals. Concentrations of Cu, Zn and Ni were transformed to bioavailable fraction according to the biotic ligand model (BGM) (Di Toro et al., 2011), and were further used in TU calculations. A total of 157 organic micropollutants were measured using previously published analytical methods based on gas chromatography-tandem mass spectrometry or liquid chromatography-tandem mass spectrometry: (perfluorinated compounds (PFCs) (Onghena et al. 2012), pesticides (Masiá et al 2013), pharmaceuticals (Gros et al, 2009), endocrine disrupting chemicals (EDCs) and related compounds such as hormones, plasticisers, alkylphenolics, parabens, phosphate flame retardants, anticorrosion agents and bactericides (Gorga et al. 2013), and UV filters (Gago-Ferrero et al.2013).

Once every compound was identified and quantified, the products were grouped into several subgroups: herbicides, organophosphate pesticides, fungicides, carbamates, neonicotinoids, and pyrethroids; antibiotics, analgesic and anti-inflammatories, anticoagulants, lipid regulators, histamines, b-blockers, antihypertensives, diuretics, antidiabetics, psychiatric drugs, veterinary pharmaceuticals; alkylphenols, flame retardants, anticorrosives, bisphenol A (BPA); hormones, UV- filters, parabens, and bactericides. This classification was used for the exploration of potential relationships between the microcontaminant classes and biota. Those showing significant correlation with the biological metrics were included in the multivariate analyses.

6.3.2.2. Biological components.

The biofilm and invertebrate communities were used as representative of the main biological components in the sites. We decided to use data on community structure (composition and abundance) of the primary producers of the biofilm (algae) and of the invertebrate community since these are commonly used as biological quality elements in monitoring schemes elsewhere. Data of other groups of organisms could not be used: biofilm bacteria data was restricted to cell density (Ponsati et al., 2016), and data on fish

were not available. Biofilms and invertebrates were collected in the very same reaches, and simultaneously, to the places where the chemical samples and measurements were performed.
Biofilm collection, preparation for diatom examination, and counting followed described standard protocols (Kelly et al. 1998). The diatom community was used as the representative of the algal fraction of the biofilm; diatoms account for the majority of species within the whole algal species set in rivers (Round 1981). Up to 400 valves were counted and determined at the species level 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 1,000x. The number of species (S_D) in each sample and the new variable first component (PC_{1D}) derived from the Principal component analysis (PCA) were used as descriptors of the diatom assemblages in the four basins. Data used in the PCA included the diatom taxa accounting for more than 1% of the relative abundance in at least three samples. In this case, the scores of the first component of the PCA (the first one having the most obvious biological meaning and the higher explained variance) were therefore used as the expression of the whole diatom community structure to the main environmental gradient in the whole set of cases.
Other biofilm measurements were also considered and used as complementary estimates of the biofilm responses. Biofilm material covering cobbles or stones was collected at each site (five replicates), and aliquots used for alkaline phosphatase activity (APA). APA is a measure of the ability of transformation of organic into inorganic phosphorus, mainly by bacteria and cyanobacteria. APA was determined using the substrate analog 4-MUF- phosphatase (from Sigma Aldrich). Samples for chlorophyll analysis were immediately frozen after collection and remained at -20ºC until analysis. Analytical details are given in Ricart et al. (2010) and Proia et al. (2013).
Invertebrate data were obtained from the analysis of sediment samples in each of the sites. Five samples were collected at random and invertebrates were sorted, counted, measured and identified under a dissecting microscope (Leica Stereomicroscope), in order to determine the community composition of the invertebrates (De Castro-Català et al. 2015). The identification was at the species level for nearly all groups of taxa (including Oligochaeta) with the exception of the Chironomids (genus level), and the Nematoda (phylum level). Also, in this case, two variables were selected as descriptors. The number of species present in each sample (S_I), as a richness measure of the invertebrate community, and the first component of a PCA (PC_{1I}) were derived from the invertebrate density data. Complementary variables describing the invertebrate community structure, such as the percentage of chironomids, or the percentage of worms were also estimated.

6.3.3. Risk assessment

The toxic unit (TU) approach (Sprague, 1970; Völker et al., 2013) was used for the ecotoxicological risk assessment of measured concentrations of compounds (Ci). The TU of each compound was based on acute toxicity values i.e. EC50 (50% effective concentration) for algae and invertebrates and LC50 (50% lethal concentration) for fish (Equation 1).

$$TU_{I\,(algae,Daphnia\ sp.,fish)} = \frac{C_i}{EC50i\ or\ LC50i} \qquad \text{(Equation 1)}$$

where TU_i is the toxic unit of a compound i ; c_i measured concentration (µg/l) of the compound in the water phase; EC50i or LC50i (µg/l) effective or lethal concentration of 50% of individuals when exposed to the substance concerned. The toxicity data of each chemical was collected for three standard test species (green algae *Pseudokirchneriella subcapitata*, invertebrate Daphnia magna and fish *Pimephales promelas*) from the literature and the databases when available, mainly ECOTOX (USEPA, 2008) and Pesticides Properties Database (PAN, 2015). Missing toxicity data were estimated by ECOSAR v.1.11. To determine site-specific toxic stress and compare it with biological quality, we used the classical concept of concentration addition (CA). It allows the prediction of the mixture toxicity from concentration and toxicity of constituents of the mixture (Backhaus and Faust, 2012) but without regarding the possible synergistic and antagonistic effects of the different chemicals. Site-specific toxic stress (TUsite) was calculated by summing all the individual TU_i of each detected compound at all of the studied sites. Since different effects in the ecosystem are expected for organics and organic compounds (López-Dovas et al., 2012) toxic units for metals (TU_{metals}) and organics were calculated separately ($TU_{organics}$). Finally, the site-specific risk was expressed as the logarithm of the mixture toxicity for metals, and all the detected organic compounds (Equation 2):

$$TU_{SITE} = \log \sum_{i=1}^{n} TU_i \qquad \text{(Equation 2)}$$

where TU_i is the toxic unit of each of individual compound at the site. Due to large differences in the hazard quotients of the different compounds, along the present article TU associated with each site is expressed in log units. Having in mind the possible different modes of action of the studied compounds, there is a possible overestimation of risk. However, since the modes of action of many studied compounds are still unknown, we used the CA approach which is generally accepted as a first tier approach (Backhaus and Faust, 2012). Additionally, it was shown that the toxicity of the mixture predicted by CA correlated with the SPEAR index (Schäfer et al., 2013) suggesting this is a valid approach for predicting the toxic stress for biological communities *in situ* (McKnight et al., 2015).

6.3.4. Data analysis.

The potential relationships between biofilm and invertebrate metrics with landscape descriptors, physical and chemical parameters, and organic micropollutants grouped in families of products, were first explored using Pearson correlation. Variables were previously inspected for normality, and when necessary accordingly transformed using decimal logarithms. Further, some bivariate relationships were carefully described amongst the ones statistically significant in the correlation analyses. These expressions were used to define the patterns of variation of the biological variables against the pollutant and environmental variables in the whole river dataset. Sigmaplot 11 was used to define the best-fitted regression curves of the biological variables with respect to the significant non-biological.

Redundancy analysis (RDA) was used to detect the ordination of the biological variables with respect to all others. RDA is a direct ordination analysis that selects a set of variables (predictors) that best explains the variance of the biological communities. RDA was performed with CANOCO for Windows (version 4.5, Microcomputer Power, Ithaca, NY, USA). The maximum gradient length for biological data was previously determined using detrended correspondence analysis (DCA). The maximum amount of variation was 1.4 standard deviation units, indicating that linear methods would be appropriate (ter Braak and Smilauer, 2002). To avoid correlation and co-linearity, variables were selected based on the inspection of non-significant correlation and variance inflator factor (VIF<20) (Ter Braak & Verdonschot 1995). This resulted in the selection of water flow (Q), variation coefficient of water flow (Q(cv)), water temperature, DOC, dissolved inorganic nitrogen (DIN), water conductivity, total phosphorus (TP), and the proportion of the different land uses, for physical and chemical variables. Only the families of organic microcontaminants showing significant correlations with the biological variables were selected to participate in the general RDA. Once defined the general RDA, a set of partial RDAs was also performed to understand the fraction of the variance that could be attributed to each of the three groups of non-biological variables (land use patterns, environmental variables, micropollutant variables).

6.4. Results and Discussion

6.4.1. Land uses and hydrological characteristics

Forest was the most prevalent land use (around 60% of the surface area) in the four basins, while agriculture was the second in relevance in the Ebro, Guadalquivir, and Júcar (Table 6.2). The highest proportion of urban and industrial land cover (hereafter referred to as artificial) was in the Llobregat

basin, and the lowest was in the Júcar (Table 6.2). The surface area of artificial land was associated with high water flow, temperature and water conductivity (Pearson correlations, Table 6.3) and higher DIN, and DOC. These areas were also significantly correlated to the higher concentration of several industrial organic compounds (IOCs), personal care products (PCPs), and pharmaceutical products (Table 3). Agricultural land uses were associated with higher water flow and temperature, and DOC, as well as to several microcontaminants (Table 6.3).

Water flow characterized the humid (2010) and dry (2011) periods in all four basins except the Júcar. The Llobregat and Guadalquivir rivers had much higher flows in summer 2010 (Table 2). Water flow was significantly associated (Pearson correlation) with increasing DIN and DOC (r= 0.56, and r=0.59, respectively), as well as with several families of industrial products (e.g. r=0.49 with flame retardants) and of pharmaceutical products (e.g. r=0.46 with diuretics, Table 3). Higher Q (cv) was mainly associated with antibiotics (r=0.40) and with some families of pharmaceuticals (e.g. r=0.52 with psychiatric drugs).

Table 6.2. Land uses and hydrological characteristics of the four different basins included in the study. Land uses are given in percentage of the total of each basin, and hydrological features are provided separately for summer 2010 (wet period) and summer 2011 (dry period). The values of water flow (Q, $m^3 \cdot s^{-1}$) are the mean daily values corresponding to three months before the sampling for each period, and the coefficient of variation (Q(cv), percent) is also indicated.

	LLOBREGAT	EBRO	JUCAR	GUADALQUIVIR
Land uses (%)				
Artificial surfaces	2.9	2.5	1.4	2.2
Agricultural areas	27.5	33.8	35.7	33.7
Forest and semi natural areas	69.4	62.0	62.5	61.8
Others	0.2	1.7	0.4	2.3
Hydrological parameters				
Q_{2010}	18.4	22.5	2.9	36.4
$Q(cv)_{2010}$	97.5	13.4	15.7	28.4
Q_{2011}	5.7	16.9	3.5	23.4
$Q\ (cv)_{2011}$	17.5	2.5	24.3	32.7

Table 6.3. Correlations (Pearson) between land uses, environmental variables and microcontaminants (n=38). The significant results with p values <0.05 are indicated in italics, and those with p values <0.01 are highlighted in bold. Those variables without any significant correlation are not shown.

	Artificial	Agricultural	Water flow	Q(cv)	Cond	Temp.	DIN	TP	DOC
Water flow	**0.61**	**0.48**			**0.46**	*0.4*	**0.56**		**0.59**
Q(cv)					*0.40*		**0.47**		*0.41*
Conductivity	**0.62**	*0.38*	**0.46**	*0.40*		**0.45**	**0.68**	**0.42**	*0.39*
Temperature	**0.77**	**0.61**	*0.40*		*0.45*		**0.48**	*0.36*	*0.38*
DIN	**0.48**		**0.56**	**0.47**	**0.68**	**0.48**		**0.55**	**0.48**
TP	*0.37*				**0.42**	*0.36*	**0.55**		*0.36*
DOC	**0.58**	**0.41**	**0.59**	**0.41**	0.39	0.38	**0.48**	0.36	
Herbicides					**0.45**	*0.34*	**0.63**	**0.62**	
Azoles					*0.34*				**-0.42**
Neonicotinoids	*0.4*				*0.39*	**0.53**	*0.34*	**0.66**	
Miscellaneous pesticides					*-0.32*	**-0.43**			
Antibiotics				*0.40*		*0.33*	*0.4*	**0.45**	
Analgesic and anti-inflammatories	*0.39*					*0.35*		**0.47**	*0.37*
Anticoagulants								**0.45**	
Lipid regulators	**0.6**	*0.35*			**0.59**	**0.5**	*0.4*	**0.62**	**0.42**
Histamines						**0.43**		*0.38*	
B-blockers						*0.4*		**0.7**	*0.41*
Antihypertensive			**0.41**	**0.50**					**0.68**
Diuretic	**0.53**		**0.46**	**0.48**	*0.38*	**0.47**	**0.46**	**0.46**	**0.58**
Psychiatric drugs	*0.4*			**0.52**	*0.39*		*0.39*	**0.51**	**0.61**
Alkylphenols	**0.54**	*0.36*	**0.43**		**0.59**	*0.37*	**0.55**	**0.54**	**0.58**
Flame retardants	**0.61**	*0.39*	**0.49**	*0.40*	**0.65**	**0.46**	**0.68**	**0.61**	**0.58**
Anticorrosives	**0.64**				**0.48**	**0.67**	*0.4*	**0.52**	
Bisphenol A (BPA)	*0.39*				**0.57**		*0.38*	*0.38*	
UV filters	*0.4*				**0.5**		**0.52**	**0.59**	

6.4.2. Chemical characteristics

Water conductivity and nutrient concentrations generally increased in a downstream direction in the four basins. Maximum DIN and TP concentrations occurred in the downstream sites of the Llobregat (11.9 mg

L^{-1} and 2.7 mg L^{-1}, respectively) and Guadalquivir (10.2 mg L^{-1} and 0.6 mg L^{-1}, respectively). Maximum DOC concentrations occurred also in the Llobregat and Guadalquivir (10.2 mg L^{-1} and 9 mg L^{-1}, respectively).

A total of 157 organic compounds were detected in the water samples collected in 2010 and 2011. Forty-two of these were pesticides, including thirteen herbicides, and the rest being insecticides, fungicides, nematicides and bird repellents. Fourteen compounds were industrial-origin products (IOCs), including alkylphenols, flame retardants, bisphenol-A, and anticorrosion compounds. Twenty compounds were personal care products (PCPs), including bactericides, preservatives, and UV-filters. Antibiotics included twelve compounds from eight different families, and the pharmaceutical products (PhCs) gathered sixty-nine compounds including analgesic and anti-inflammatory drugs, anticoagulants, antihypertensive drugs, β-blockers, diuretics, histamine analog compounds, lipid regulators, psychiatric drugs and pharmaceuticals for veterinary use. The most abundant organic compounds in all the studied basins were IOCs, PCPs, and PhCs. Insecticides and herbicides were found in lower concentrations, but occasionally reached presence (particularly carbamates and fungicides) of up to 400 ng L^{-1}. The total PhCs concentrations ranged from 40 to 3000 ng L^{-1}, and those of IOCs from 50 to 2300 ng L^{-1}. Within PhCs, analgesics and anti-inflammatories reached 5 to 510 ng L^{-1}, and diuretics reached concentrations of 100-420 ng L^{-1}. Anti-hypertensive drugs reached maximum concentrations within the range of 180 to 650 ng L^{-1}. Finally, within the group of IOCs, high levels (in the high ng/L up to µg/L range) of certain widely used industrial compounds such as alkylphenols and their ethoxylated derivatives, bisphenol A, trialkyl phosphates or benzotriazoles were detected in all basins.

The organic micropollutants differed among basins and periods. The Llobregat River showed the highest concentrations of organic micropollutants, which ranged from 1000 to 12,000 ng L^{-1}. In the Ebro the range of concentrations was of 500-1800 ng L^{-1}, 600 to 1400 ng L^{-1} in the Júcar and 240-2800 in the Guadalquivir. The maximum concentration of a single group of compounds was that of IOCs in site LLO7 in 2011 (11,000 ng L^{-1}), being alkylphenols, flame retardants and anticorrosives those with the highest concentrations. PhCs were the most common organic compounds in the two periods in the Llobregat, while herbicides, pesticides, and IOCs presented slightly higher concentrations in the drier period (2011). The Llobregat had high levels of almost all PhCs' families. The major organic micropollutants in the Ebro were also IOCs (particularly in site E3), though in lower concentrations than in the Llobregat. Herbicides in the Ebro were in similar concentrations in the two periods, but insecticides, PCPs, antibiotics, and PhCs presented higher concentrations in the wet period. The Júcar had the highest pesticide and the lowest IOC concentrations of the four basins; fungicides, herbicides, and insecticides were detected at high concentrations in this river, especially during the wet period. Antibiotics occurrence was similar in the two periods, but fluoroquinolones (site J5; 109.5 ng L^{-1}) and

nitroimidazoles (site J3; 66.3 ng L^{-1}) were remarkably high in the dry period. The most important organic micropollutants in the Guadalquivir River were the IOCs. These compounds did not show any significant difference between the studied periods, but a particular increase in site G2 in the dry period.

6.4.3. Biological characteristics

Preliminary screening of the available metrics was performed by means of correlation analyses. This discarded the number of diatom species as a suitable descriptor and pointed the PC_{1D}, chlorophyll-*a*, and alkaline phosphatase activity as good biological descriptors of the environmental gradient. The PC_{1D} of the diatom taxa arranged the taxa occurring in less to highly impaired sites along the first axis of the PCA (38% of the total explained variance). Some taxa (*Achnanthidium pyrenaicum*, *Achnanthidium minutissimum*, *Encyonopsis microcephala*) were characteristic of the Júcar, Ebro and Guadalquivir upper reaches, and were opposed in the PCA to *Navicula* and *Nitzschia* species (*Eolimna subminuscula*, *Navicula recens*, *Nitzschia insconspicua*, *Nitzschia palea*, *Nitzschia frustulum*) characteristic of the downstream sites of the Ebro, Llobregat and Guadalquivir. This arrangement reflected the general environmental conditions of all the sites.

The microcontaminants related to the PC_{1D} were the neonicotinoids, lipid regulators, β-blockers, psychiatric drugs, alkylphenols, flame retardants, anticorrosives, and bisphenol A (BPA). In all the cases, the quality pattern defined by the positive values of the PC_{1D} decreased accordingly to the increasing concentration of these products (Table 6.4).

The alkaline phosphatase activity (APA) was also inversely related to several micropollutants (e.g. analgesic and anti-inflammatories, lipid regulators, antihypertensives, diuretics, flame retardants, and parabens) (Table 6.4). There was a coincident negative relationship of several descriptors of the biofilm with microcontaminants occurrence; APA and PC_{1D} showed analogous correlation trends in relation to water flow, lipid regulators, diuretics, and flame retardants (Table 6.4).

Table 6.4. Correlations (Pearson) between land uses, environmental variables and organic microcontaminants with respect to the selected biological variables (the first component of the diatom community analysis PC_{1D}, the alkaline phosphatase activity (APA), and the diversity of the macroinvertebrate community S_I). Significant results with p values <0.05 are indicated in italics, and those with p<0.01 are highlighted in bold. Those variables without any significant correlation are not shown.

	PC_{1D}	APA		S_I
Artificial surface area	**-0.81**	*-0.35*		**-0.69**
Agricultural surface area	**-0.44**			**-0.59**
Conductivity	**-0.71**			**-0.47**
Temperature	**-0.71**			**-0.69**
DIN	**-0.63**	*-0.34*		**-0.51**
TP	**-0.44**			
DOC	**-0.59**	*-0.39*		**-0.47**
Water flow	**-0.65**	**-0.51**		**-0.58**
Herbicides	*-0.35*			
Organophosphates		*0.39*		
Neonicotinoids	**-0.55**			
Antibiotics	-0.31	*-0.34*		
Analgesic and anti-inflammatories	-0.4	**-0.53**		
Lipid regulators	**-0.62**	**-0.47**		
B-blockers	**-0.42**			
Antihypertensives	-0.36	**-0.49**		*-0.41*
Diuretics	**-0.62**	**-0.6**		*-0.44*
Psychiatric drugs	**-0.43**	*-0.4*		
Alkylphenols	**-0.58**	*-0.36*		*-0.46*
Flame retardants	**-0.62**	**-0.5**		*-0.42*
Anticorrosives	**-0.67**			**-0.51**
Bisphenol-A (BPA)	**-0.52**			*-0.44*
UV-filters	**-0.42**			
Parabens		**-0.41**		

6.4.4. Joint data analysis

An RDA was performed using the diatom community composition (PC_{1D}) and the invertebrate community diversity (S_I) as selected metrics of the biological communities' structure (Figure 6.2). These two metrics were used as fixed variables against the physical and chemical variables selected by the analysis. The first axis of the RDA accounted for the 76.4% of the variance and indicated an analogous general response of PC_{1D} and S_I with respect to the environmental gradient. The two biological descriptors were opposed to DIN, increasing water flow, and higher surface area of agricultural lands. PC_{1D} and S_I were also opposed to BPA, lipid regulators, anticorrosives, and artificial (urban and

industrial) land use. The position of the diatoms vector PC_{1D} was the most opposed to all the physical and chemical variables describing multiple stress, particularly those in the downstream sites of the Llobregat, Guadalquivir, and Ebro. The position of S_I was less apparently opposed to this ensemble of variables, and closer to the downstream sites of the Guadalquivir and Júcar (Figure 6.2). The second axis of the RDA (5.1% of the variance) separates the upstream sites of the Guadalquivir, Júcar, and Ebro from the rest.

After this general RDA, a subsequent partition of the variance analysis was performed to discriminate the respective relevance of land-uses, physicochemical, and micropollutant variables with respect to the biological variables. A set of partial RDAs were performed for all the combinations of stressors in order to account for the different interactions and shared variances. The TP, DOC, DIN, water conductivity, water temperature, and water flow were selected for the physicochemical variables, artificial and agricultural land areas were selected for land uses, and bisphenol-A, anticorrosives, and lipid regulators were selected for the group of organic micropollutants. The total explained variance was the 86.3%, where a 2.2% was directly attributed to the organic micropollutants, 5.7% to land uses, and 10.6% of the environmental variables. The shared variance between organic micropollutants and land uses was 4.1%, but the one shared between land uses and physical-chemical variables was 21.2%. The total shared variance of the three groups of variables was 41.3% (Figure 6.3).

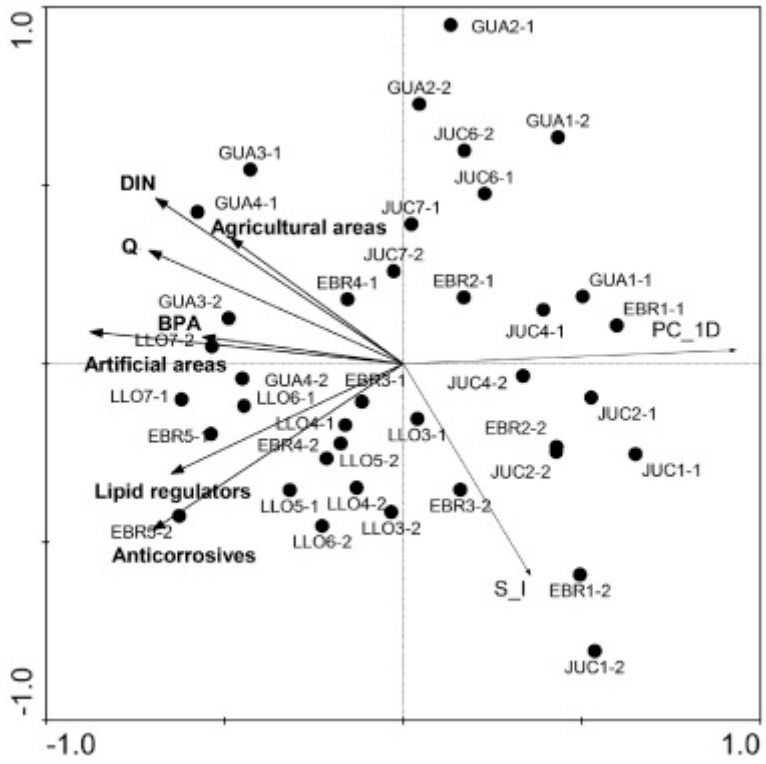


Figure 6.2. RDA performed using the diatom community composition (PC_1D) and the invertebrate community diversity (S_I). The variables participating in the analysis were DIN, water flow, surface areas of agricultural and artificial lands, BPA, lipid regulators, and anticorrosives. The sites of the Llobregat (LLO), Guadalquivir (GUA), Júcar (JUC) and Ebro (EBR) of the 2010 sampling were indicated as −1, and those of the 2011 sampling as −2.

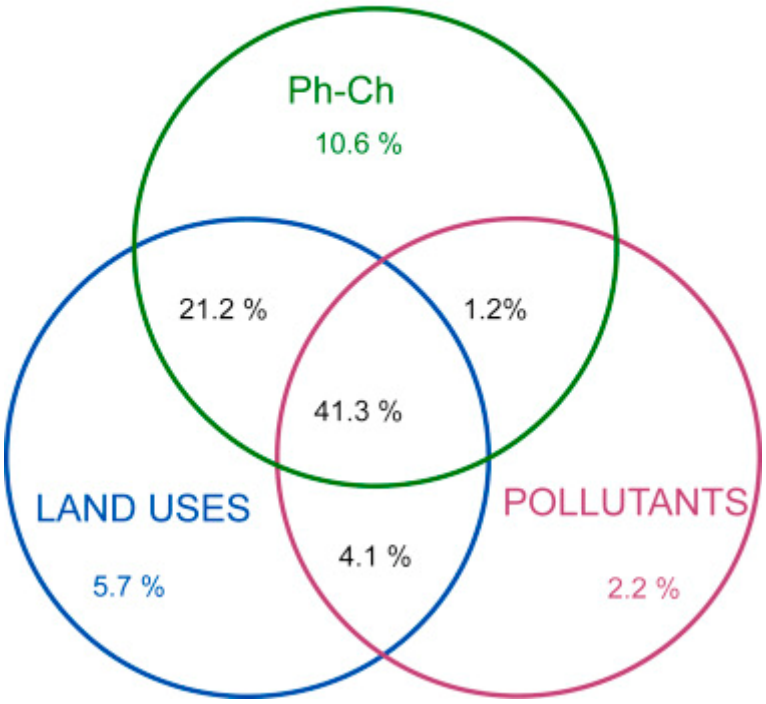


Figure 6.3. Shared variance resulting from the partition of the variance analysis between physical-chemical variables (Ph-Ch), land uses, and organic micropollutants. This analysis was performed by separating land-uses, environmental, and micropollutant variables, and identifying its share with respect to the biological variables included in the study (invertebrates and diatoms).

6.4.5. Ecotoxicological stress and biological status

6.4.5.1. Ecotoxicological risk assessment: acute and chronic risk

To determine the potential effects of chemical pollution on the biological communities in situ we used the effect thresholds as proposed by Malaj et al. (2014). The acute risk threshold was set at the TU ≥-1 (1/10 of EC50 or LC50) for all three test species since the acute effects in the ecosystem are generally expected at that level (Schäfer et al., 2011b; Schäfer et al., 2012; Van Wijngaarden et al., 2005). For the invertebrates, chronic risk threshold value of TU ≥ -3 (1/1000 of EC50) was used. Changes in communities have been observed above that threshold i.e., decrease of sensitive species and shift towards more resistant species assemblages (Beketov et al., 2013; Liess and Von Der Ohe, 2005; Schäfer et al., 2012). However, this threshold is based on the field studies of effects of pesticides on biological communities. Therefore, extrapolating this threshold to other groups of compounds could lead to over or underestimation of the risk for some of the compounds. Also, those studies used the maximum toxic unit (TUmax) in the sample, indicating the minimum estimated toxicity of the mixture as the toxicity of the

most potent compound (Schäfer et al., 2013). In the case when the sum of toxic units is used to represent the mixture toxicity it should be noted that this is a bit more conservative approach but in line with the principle of screening-level risk assessments (McKnight et al., 2015). Due to the absence of studies relating pollution and long-term effects in communities, chronic risk thresholds for algae and fish were based on acute to the chronic ratio (Malaj et al., 2014). For algae, the acute to chronic factor 5 was used and for fish factor 10 (Heger et al. 1995, Länge et al. 1998, Ahlers et al. 2006).

a) Acute effects risk

The toxic units ($TU_{organic}$) indicated that there was a risk of acute effects in biological communities posed by organic compounds at 42% of the sampling sites and risk of chronic effects at all the studied sites. Of the three test species used for risk assessment, invertebrates were the most sensitive group due to the presence of highly toxic insecticides at many sampling sites. Considering the four studied rivers, the total number of sites with exceedance of the acute risk threshold was higher in 2010 (42% for invertebrates, 3% for fish and none for algae), than in 2011 (20% for invertebrates and no exceedance for algae and fish). The highest number of sites exceeding the acute threshold was in the Ebro in 2010 (74% of sites) and in Júcar (67% and 60% in 2010 and 2011, respectively) (Fig. 6.4) mostly due to the presence of the insecticides chlorpyriphos, chlorfenvinphos, and ethion. On the contrary, in 2011 there was no exceedance of acute risk threshold in the Ebro due to relatively lower concentrations of those pesticides. In the Llobregat and Guadalquivir basins, there was exceedance of acute risk threshold at less than 25% of the sites (Fig. 4). In 2011, the only area where the acute risk was increased compared to the previous year was in the lower part of the Llobregat basin. Of all the organic compounds measured in water, the major contributors to the chemical risk were pesticides. The compounds responsible for acute risk in Llobregat were chlorpyriphos and azinphos ethyl and ethion. In Guadalquivir, there was an acute risk at only 4 sites in 2010 and 3 sites in 2011 where high concentrations of chlorpyriphos, ethion and chlorfenvinphos were measured. In general, several pesticides were related with risk of acute effects of which the most important were the insecticides chlorfenvinphos (29% of sites with acute risk exceedance in 2010) and chlorpyriphos (15% sites in 2010). They are both classified by WFD as priority compounds and were identified as the compounds of highest ecotoxicological concern in studied river basins (Kuzmanović et al., 2015). Conversely, in 2011 they were not present in water at such high concentrations and thus the resulting acute risk exceedance was evidently lower, especially in the case of the Ebro where chlorfenvinphos was detected only at one site in that year's sampling campaign. The lower acute risk in 2011 might be an underestimation due to sampling in the dry period with the absence of precipitation which can trigger for the runoff effect of pesticides which were the most toxic compounds

measured. Other pesticides not covered by the WFD, but banned in the European Union, were also detected in water at high toxic units (e.g. ethion up to TU (log units) = − 0.36 in the Júcar).

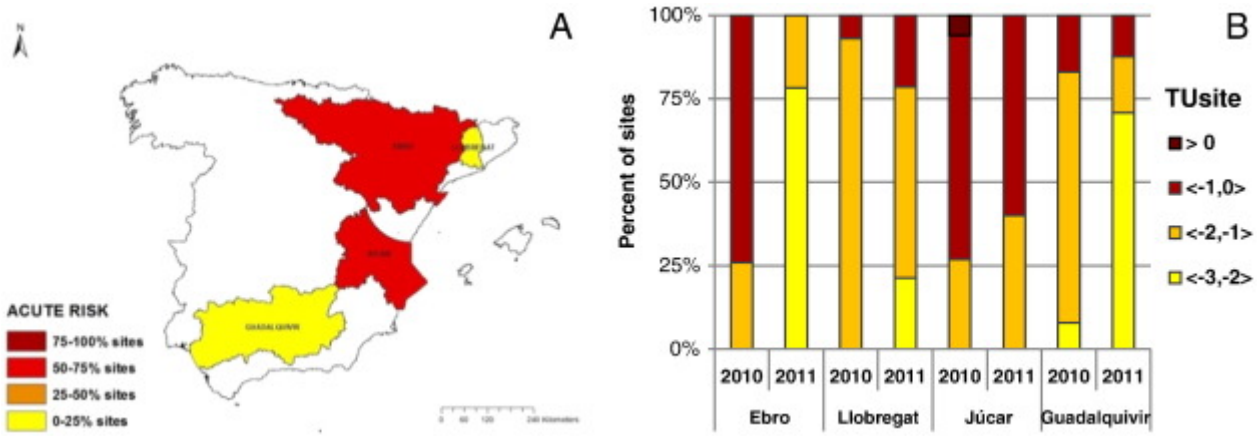


Figure 6.4. Percentage of sampling sites A) with acute risk exceedance and B) with TU_{site} (most sensitive test species) belonging to one of four toxic unit ranges for each of four river basins in 2010 and 2011.

b) Chronic effects risk

The chronic risk threshold was exceeded at all of the sampling sites for at least one of the test species. In 2011, the exceedance was the highest in the Júcar (all sites), the Llobregat (80% of the sites), the Ebro (61% of the sites) and the Guadalquivir (55% of the sites). While only pesticides and metals were responsible for acute risk, all measured compound groups except perfluorinated compounds exceeded the chronic risk threshold for at least one test species. Perfluorinated compounds were in low TU at all the sampling sites. IOCs exceeded the chronic risk threshold at several sampling sites, mostly in the Guadalquivir (54%) and in the lower part of the Llobregat basin (50%). Of that group, the WFD priority compounds alkylphenols and their ethoxylated derivatives were the main contributors to toxic load among compounds detected. Personal care products exceeded algae chronic threshold mostly due to triclosan that was detected around industrial and urban areas (lower part of the Llobregat and the Júcar basins, the northern part of the Ebro basin (Fig. 6.1)). Pharmaceuticals exceeded chronic risk threshold in the Llobregat basin in 2010 with the antidepressant sertraline as the compound most responsible for threshold exceedance. However, in this study, we used acute toxicity data to assess the risk of both acute and chronic effects. Despite the fact that long-term chronic exposure to pollutants is a more realistic scenario (Eggen et al., 2004) there is a paucity of chronic toxicity data, especially for emerging contaminants. As stated by Calow and Forbes (Calow and Forbes, 2003), there is uncertainty in extrapolating results from effects caused after short, high dose exposure to effects caused after long time exposures to low doses of chemicals. There are indications that chronic responses to some chemicals may

be greater than expected from risk assessment procedures similar to the one we followed. The chemicals causing endocrine-disrupting effects at low environmental concentrations are the example for that, and it is reasonable to expect other types of specific chronic effects in the future caused by different compounds (Calow and Forbes, 2003).

6.4.5.2. Relationship between toxic stress and biological status

The only statistically significant correlation (Spearman, $p < 0.05$) between biological community descriptors and toxic stress of organic compounds was between $SPEAR_{organic}$ and $TU_{organic}$ ($r = -0.490$) and $TU_{pesticides}$ ($r = -0.431$) (Table 5). Neither Shannon nor Margalef indexes showed significant correlation with $TU_{organic}$ (Table 5). Moreover, diversity indexes were not correlated with $SPEAR_{pesticides}$ and $SPEAR_{organic}$. It has been reported in several studies, that Shannon and similar biodiversity indexes were not suitable to identify the effects of pesticides at the community level (Ippolito et al., 2012) and are influenced by different natural and anthropogenic factors (Beketov and Liess, 2008). In this study, they were negatively correlated with metals (TU_{metals}) (Table 5). However, only Margalef index was significantly correlated with the metals toxic units TU_{metals} ($r = -0.515$) (Table 5). Metals toxic units were significantly and positively correlated with urban land use type, while Shannon and Margalef indexes were correlated negatively (Table 6.5).

Table 6.5. Correlation matrix based on Spearman rank correlation test between toxic units (TU), land use, Shannon diversity (H'), Margalef richness (d) and species at risk index (SPEAR). (in bold, p< 0,05)

Variables	Urban	Agricultural	Natural	d	H'	SPEAR pesticides	SPEAR organic	TU metals	TU IOC	TU PCP	TU pharmaceuticals	TU pesticides	TU organic
Urban	**1**	-	**-**	-	-	-	-	-	-	-	-	-	-
Agricultural	0,134	**1**	-	-	-	-	-	-	-	-	-	-	-
Natural	**-0,497**	**-0,817**	**1**	-	-	-	-	-	-	-	-	-	-
d	**-0,672**	-0,068	**0,375**	**1**	-	-	-	-	-	-	-	-	-
H'	**-0,436**	0,134	0,140	**0,883**	**1**	-	-	-	-	-	-	-	-
SPEAR pesticides	0,120	0,014	0,028	0,232	0,269	**1**	-	-	-	-	-	-	-
SPEAR organic	0,339	0,337	-0,306	0,088	0,286	**0,481**	**1**	-	-	-	-	-	-
TU metals	**0,600**	0,010	-0,295	**-0,515**	-0,268	0,043	0,330	**1**	-	-	-	-	-
TU IOC	0,045	0,018	-0,063	0,004	-0,004	-0,117	-0,127	0,007	**1**	-	-	-	-
TU PCP	0,248	0,036	-0,151	-0,129	-0,092	0,061	0,105	0,210	**-0,585**	**1**	-	-	-
TU pharmaceuticals	**0,490**	-0,010	-0,243	-0,232	-0,151	0,344	0,303	**0,492**	**-0,389**	**0,674**	**1**	-	-
TU pesticides	**-0,412**	0,160	0,020	0,140	0,156	-0,229	**-0,431**	**-0,405**	0,323	**-0,404**	**-0,606**	**1**	-
TU organic	**-0,394**	-0,012	0,128	0,175	0,155	-0,073	**-0,490**	**-0,459**	/	/	/	/	**1**

That is, we can relate the decrease of macroinvertebrate biodiversity to urban areas. Nevertheless, urban rivers are highly impacted by a variety of stressors and it is known that in some cases, more environmental stressors can interact with the toxicants (Liess et al., 2013). Besides chemical pollution, in urban rivers, there are often present habitat changes, temperature alterations and other stressors (Vörösmarty et al., 2010). Also, the natural gradient of environmental factors along the rivers is one of the most important sources of differences between biological communities (Beketov and Liess, 2008) and each site has its unique combination of natural factors (Schäfer et al., 2007) which should be taken into account when interpreting the macroinvertebrate biodiversity change along the river. The relation between biodiversity indexes and urban land use could be reflecting the response of the community to a variety of stressors present at the urban areas that are acting together along with the pollution. Linear regression between $SPEAR_{organic}$ and total organic stress at site ($TU_{organic}$) was significant with r^2=0.235 ($p < 0.05$) and a relationship between $SPEAR_{pesticides}$ and $TU_{pesticides}$ with $r^2 = 0.104$ ($p < 0.1$). Scatter plots show the relationship between losses of sensitive species and an increase of toxic stress of organic compounds (Fig. 5A) and pesticides (Fig. 5B). All the sites were characterized by medium to high toxic stress (logTU from -2.7 to 0) therefore the gradient of toxicity was relatively low and we could not observe the communities composition in pollution free conditions (i.e., reference conditions). Even though SPEAR index is designed to be a stressor-specific indicator it cannot be excluded that other stressors might have influenced the loss of sensitive species. This could be the case, especially since studied rivers are impacted by a multitude of anthropogenic stressors and some stressors are expected to cause similar changes in trait categories (Rasmussen et al., 2013; Statzner and Bêche, 2010). Besides, different co-occurring stressors (Liess and Beketov, 2011) and their complex relationships with biological communities (Liess et al., 2008) can mask the effects of single toxicant. Naturally, the use of $SPEAR_{pesticides}$ was showing the best results in agricultural streams where pesticides are the predominant stressors (Beketov et al., 2013; Schäfer et al., 2007). However, since only macroinvertebrates in the sediment were sampled in this study, the low values of SPEAR pesticides could be attributed to a relatively large proportion of tolerant species in that habitat (von der Ohe and Goedkoop, 2012; Wolfram et al., 2012) and the starting bias in the data makes any conclusion difficult. However, $SPEAR_{organic}$ as a less specific indicator seems to be more suitable for the multi-chemical polluted rivers. In conclusion, when all four biological indexes used in this study are compared, the most suitable to relate changes in biological communities (i.e. decrease of sensitive species) to organic stress was the $SPEAR_{organic}$ indicator.

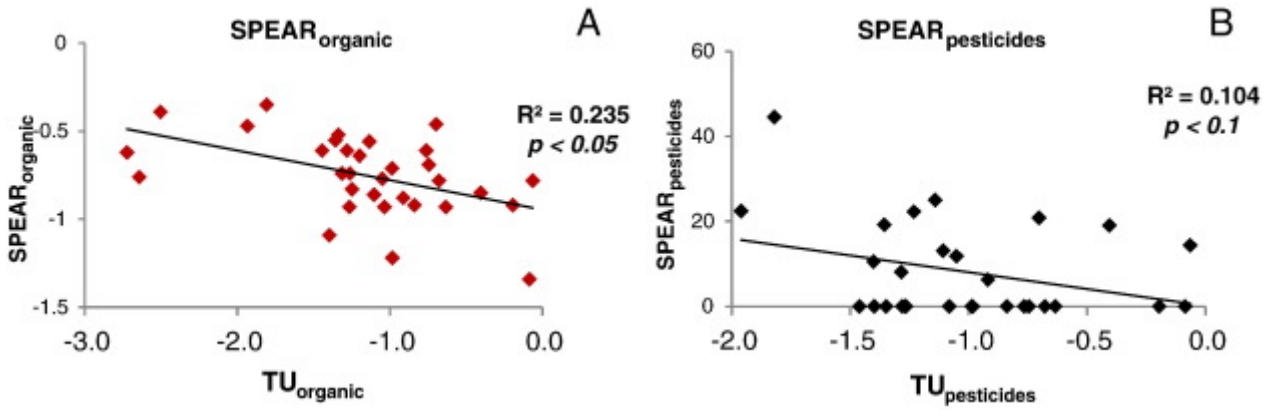


Figure 6.5. Relationship between invertebrate communities in situ and the toxic stress. A) Expressed as $SPEAR_{organic}$ and toxic units of organic compounds ($TU_{organic}$ invertebrates). Linear regression is significant with $r^2 = 0.235$, $p < 0.05$. B) Expressed as $SPEAR_{pesticides}$ and toxic units of pesticides ($TU_{pesticides,}$ invertebrates). Linear regression is significant with $r^2 = 0.104$ at $p < 0.1$

6.5. Concluding remarks

The analysis of the data revealed that the biofilm and the invertebrate community had similar and complementary responses to the stressors occurrence and relevance, with a progressive decrease in biodiversity and associated simplification of the biological structure. Nutrients and DOC in excess, higher abundance of artificial land uses, and higher concentrations of organic microcontaminants accounted, in this order, for the distribution of the two biological communities. However, most of the response of the biological metrics could not be attributed solely to one or the others, but to the joint expression of the different stressors in the sites.

The multivariate analysis (RDA) used in this study attributed a common pattern to the distribution of the algal (biofilm) and invertebrate communities, showing that they were associated to the progressive impairment of the sites. Increasing areas of agricultural, and industrial or urban lands were associated with higher inorganic nutrient concentrations, increasing dissolved organic matter, and increasing concentrations of organic microcontaminants. The general response of the two biological communities to the progressive river impairment was towards a decrease in community diversity and to the higher occurrence of species tolerant to pollution. Even though species replacement naturally occurs along a downstream river gradient, as a response to changes in the river environment (e.g. temperature, habitat, food resources; Margalef, 1983), the ones occurring in our rivers were related to their respective tolerance

to pollution. Both for the algae and invertebrates, the decline in diversity was mainly related to the decline of species non-tolerant to organic pollution, that however occur in analogous but non-polluted systems (Bennett et al., 2011 and Almeida et al., 2014). Biofilms and invertebrates represent two major components of the river trophic webs (Allan and Castillo, 2007): biofilms include primary producers (algae) and heterotrophs (bacteria, fungi) in a highly cooperative consortium (Lock et al., 1984); invertebrate communities include all consumer feeding strategies, from herbivores and detritivores to predators (Anderson and Sedell, 1979). Therefore, biofilms and invertebrates are inclusive of most of the biological elements involved in the transference of energy and matter in the river. Showing a common response can be taken as an indication of the analogous effect caused by the stressors, and as an evidence of the overall effect on river biodiversity.

The physical and chemical variables selected by the multivariate analysis were the ones most relevant for the algal and invertebrate communities. The variables could be considered as stress descriptors acting in the river site. Those variables selected by the analysis were analogous to those affecting the biotic community structure in impaired rivers elsewhere. Artificial and agricultural land uses have been associated to the massive arrival of DIN to the river (Burkart and James, 1999, Nikolaidis et al., 1998 and Poor and McDonnell, 2007), as well as to the continuous inputs of pharmaceutical products and other contaminants (Burkart and Kolpin, 1993 and Allan, 2004). de Zwart et al. (2009) observed significant taxa loss as a result of the highly polluted conditions in the Scheldt River. Effects on species richness of benthic macroinvertebrates have been associated to sediment-bound contaminants (trace metals, polycyclic aromatic hydrocarbons, polychlorinated biphenyls) in the rivers Rhine and Meuse (De Lange et al., 2004). At larger spatial scales, the diversity and composition of algae, macrophytes, invertebrates, and fish in European streams, has been associated with increasing nutrient concentrations (Johnson and Hering, 2009). Pesticide occurrence has been associated to the reduced invertebrate richness in sites when concentrations are close to the legal threshold levels (Stehle and Schulz, 2015), as well as to high local losses in the invertebrate species pools (Beketov et al., 2013). All these field-derived evidences show that several sorts of variables affect the biota in impaired rivers, often coinciding in space and time, and able to produce similar consequences to biodiversity. These consequences follow a rather general mechanism: the most sensitive species are affected, even becoming locally extinct, and tolerant others are favored (Blanck et al. (1998)) up to a certain threshold. This was analogously expressed by the algae and invertebrate communities in our study set: a decrease in diversity and biological communities made up of species tolerant to the new conditions.

The effect of stressors in our study set was not only evidenced by the biodiversity decrease of the algal and macroinvertebrate communities. The alkaline phosphatase activity (APA), an expression of the

transformation of organic phosphorus into inorganic by bacteria, cyanobacteria and some algae (Chróst and Overbeck, 1987), decreased in the sites with higher concentrations of DIN and DOC and higher concentrations of organic contaminants such as analgesics and diuretics. The APA decrease suggests that the biofilm ability to transform organic phosphorus into inorganic (and available) phosphorus could be seriously limited in those polluted areas. Regarding the invertebrate community, De Castro-Català et al. (2015) observed a significant correlation between the activity of the antioxidative enzyme catalase in the invertebrate *Hydropsyche exocellata* and the presence of EDCs and PhCs in the sites. Such a stress response on the invertebrate community has been observed also under different sources of pollution such as heavy metals (Barata et al., 2005).

The correlation analysis and the RDA revealed that effects of environmental stressors such as nutrients in excess and DOC on the distribution of the biological communities were higher than that of organic micropollutants. The partition of the variance showed a low relevance of the measured organic micropollutants on the distribution of diatom and invertebrate communities (2%), while the one corresponding to the environmental factors (nutrients, DOC, water flow) was higher (ca. 10%). Even though the multivariate analysis results need to be used cautiously, the variance of the different stressors expresses the relevance of factors such as irregular flow patterns, high water conductivity, and high DIN and DOC concentrations. Such a result should not be surprising according to the higher potential impact associated to the impairment of river habitat, hydrological patterns, or inorganic nutrients (Elosegi and Sabater, 2013), than the one potentially produced by organic micropollutants.

Our results do not preclude the potential of organic microcontaminants to produce particular effects on the biota. A separate analysis of the associated ecotoxicological risk of contaminants in the four studied basins, based on the toxic units (TU) approach, was performed by Kuzmanovic et al. (2015) using the same sampling scheme. TUs for individual contaminants were calculated using algae (*Scenedesmus*) and invertebrates (*Daphnia*) and then aggregated under the assumption of concentration addition (CA) to derive the site-specific risk. Their risk assessment analysis indicated that organic chemicals were able to pose a risk of acute effects at 42% of the sampling sites, and chronic effects to all of the studied sites, particularly to invertebrates. The higher potential toxicity (TU values of −1.27 to −0.28) was estimated in sites showing the highest concentrations of pesticides. The correlation results of our invertebrate and algal metrics identified some microcontaminants also identified by the risk assessment. Gemfibrozil was identified to show a high risk for the biota (Kuzmanovic et al., 2015), and we also found it to be significantly associated with poorly diverse biofilms communities and low alkaline phosphatase activity. Gemfibrozil is the lipid regulator most abundant in our series of polluted sites and has been described to induce transcriptional responses of several bacterial genes involved in lipid metabolism (Yergeau et al.,

2010), an early indication of potential more severe effects on biofilms. Flame retardants were also identified by the two approaches. These contaminants have been pointed out as disruptors of invertebrate development (Wallstrom et al. 2005), as well as able to produce adverse effects on biofilm algae in locations close to industrial and urban sewage discharges (Cristale et al., 2013). Even though a tight coincidence of the two approaches should not be expected since they provide different perspectives (an *a priori* estimation of chemical effects not necessarily coinciding with real ecosystem effects, and an estimation of the relevance of factors performing as stressors on the biota, respectively), the coincidence highlights the real relevance of these contaminants in the ecosystem.

Overall, our analysis indicates that the organic micropollutants mainly affect the distribution of organisms already affected by other stressors (Allan et al., 2013 and Coors and De Meester, 2008), or the other way round. The partial RDAs show the very high fraction of the variance (nearly half of the total explained) shared between the organic micropollutants and the remaining environmental stressors, pointing to their common relevance for the distribution of the biological communities. Environmental stressors may reinforce the effect of organic micropollutants, or vice-versa (Segner et al., 2014). Stressors occurring at multiple spatial and temporal scales define a so-called “stressor space” where the net receivers are the biological communities, and where synergies could produce much higher effects than the ones attributed solely to organic microcontaminants or to inorganic nutrients. Whatever the causes, it is obvious that the multiple and simultaneous occurrence of multiple stressors challenges the carrying capacity of ecosystems (Posthuma et al., 2014) by affecting their biodiversity and basic functions. Understanding the real risks affecting the biological communities requires quantifying the effects of multiple stressors in impaired systems.

6.6. References

Ahlers, J., C. Riedhammer, M. Vogliano, R. U. Ebert, R. Kühne and G. Schüürmann (2006). "Acute to chronic ratios in aquatic toxicity - Variation across trophic levels and relationship with chemical structure." Environmental Toxicology and Chemistry 25(11): 2937-2945.

Allan, J. D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology and Systematics 35, 257-284.

Allan, J. D.; Castillo, M. A. 2007. Stream ecology: structure and function of running waters. Second edition ed.; Springer: Dordrecht, The Netherlands.

Allan, J. D.; McIntyre, P. B.; Smith, S. D. P.; Halpern, B. S.; Boyer, G. L.; Buchsbaum, A.; Burton, J., G. A. ; Campbell, L. M.; Chadderton, W. L.; Ciborowski, J. J. H.; Doran, P. J.; Ederk, T.; Infante, D. M.;

Johnson, L. B.; Joseph, C. A.; Marino, A. L.; Prusevich, A.; Read, J. G.; Rose, J. B.; Rutherford, E. S.; Sowa, S. P.; Steinman, A. D., Joint analysis of stressors and ecosystem services to enhance restoration effectiveness. PNAS 2013, 110, (1), 372-7.

Almeida, S.F.P., Elias, C., Ferreira, J., Torn_es, E., Puccinelli, C., Delmas, F., D€orflinger, G., Urbani_c, G., Marcheggiani, S., Rosebery, J., Mancini, L., Sabater, S. 2014. Water quality assessment of rivers using diatom metrics across Mediterranean Europe: a methods intercalibration exercise. Sci. Total Environ. 476-477, 768-776.

Altenburger, R., Ait-Aissa, S., Antczak, P., Backhaus, T., Barcel_o, D., Seiler, T.B., Brion, F., Busch, W., Chipman, K., de Alda, M., de Arag~ao Umbuzeiro, G., Escher, B.I., Falciani, F., Faust, M., Focks, A., Hilscherova, K., Hollender, J., Hollert, H., Jager, F., Jahnke, A., Kortenkamp, A., Krauss, M., Lemkine, G., Munthe, J., Neumann, S., Schymanski, E.L., Scrimshaw, M., Segner, H., Slobodnik, J., Smedes, F., Kughathas, S., Teodorovic, I., Tindall, A.J., Tollefsen, K.E., Walz, K.H., Williams, T.D., Van den Brink, P.J., van Gils, J., Vrana, B., Zhang, X., Brack,W. 2015. Future water quality monitoring - Adapting tools to deal with mixtures of pollutants in water resource management. Sci. Total Environ. 512-3, 540-551.

Anderson, N. H., Sedell, J. R. 1979 Detritus processing by macroinvertebrates in stream ecosystems. Ann. Rev. Entomol. 24, 351-377.

Angermeier, P. L., Winston, M. R. 1998. Local vs. regional influences on local diversity in stream fish communities of Virginia. Ecology 79, (3), 911-927.

Armitage, P. D., Moss, D., Wright, J. F., Furse, M. T. 1984. A preliminary classification of running-water sites in Great Britain based on macro-invertebrate species and the prediction of community type using environmental data. Water Res. 17, (3), 333-347.

Arnot, J. A.; Gobas, F. A. P. C.. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. Environ. Toxicol. Chem. 23, (10), 2343-2355.

Azevedo, L.; van Zelm, R.; Leuven, R.; Hendriks, A.; Huijbregts, M. A. J., Combined ecological risks of nitrogen and phosphorus in European freshwaters. Environ Pollut 2015 200, 85-92.

Backhaus T, Faust M. Predictive environmental risk assessment of chemical mixtures: A conceptual framework. Environ. Sci. Technol. 2012; 46: 2564-2573.

Barata, C., Lekumberri, I., Vila-Escal_e, M., Prat, N., Porte, C., 2005. Trace metal concentration, antioxidant enzyme activities and susceptibility to oxidative stress in the tricoptera larvae Hydropsyche exocellata from the Llobregat river basin (NE Spain). Aquat. Toxicol. 74, 3e19.

Barceló, D., Sabater, S. 2010. Water quality and assessment under scarcity. Prospects and challenges in Mediterranean watersheds. J.Hydrol. 383, 1-4.

Beketov, M.A., Liess, M. 2008. An indicator for effects of organic toxicants on lotic invertebrate communities: independence of confounding environmental factors over an extensive river continuum. Environ. Pollut. 156, 980–987.

Beketov, M. A., Kefford, B. J., Schäfer, R. B.; Liess, M. 2013. Pesticides reduce regional biodiversity of stream invertebrates. PNAS 110, (27), 11039-43.

Beketov, M. A.; Liess, M. 2012. Ecotoxicology and macroecology- time for integration. Environ. Pollut. 162, 247-54.

Bennett, C., Owen, R., Birk, S., Buffagni, A., Erba, S., Mengin, N., Murray-Bligh, J.,Ofenback, G., Pardo, I., van de Bund, W. 2011. Bringing European river quality into line: an exercise to intercalibrate macro-invertebrate classification methods. Hydrobiol. 667, 31-48.

Blanck, H.; Wängberg, S.-A.; Molander, S. 1988. Pollution-induced community tolerance - a new ecotoxicological tool. In Functional testing of aquatic biota for estimating hazards of chemicals, Cairns, J., Jr; Pratt, J. R., Eds. ASTM STP 988 American Society for testing materials: Philadelphia, pp 219-230.

Boix, D., García-Berthou, E., Gascón, S., Benejam, L., Tornés, E., Sala, J.; Benito, J., Munné, C., Sabater, S. 2010. Response of community structure to sustained drought in Mediterranean rivers. J. Hydrol. 383, 135-146.

Bonada, N., Prat, N., Resh, V., Statzner, B. 2006. Developments in aquatic insects biomonitoring: a comparative analysis of recent approaches. Ann. Rev. Entomol. 51, 495-523.

Burkart, M. R., James, D. E. 1999. Agricultural-nitrogen contributions to hypoxia in the Gulf of Mexico. J.l Environ. Qual. 28, (3), 850-859.

Burkart, M. R., Kolpin, D. W. 1993. Hydrologic and Land-Use Factors Associated with Herbicides and Nitrate in Near-Surface Aquifers. J. Environ. Qual. 22, (4), 646-656.

Calow, P., Forbes, V.E., 2003. Does ecotoxicology inform ecological risk assessment? Environ. Sci. Technol. 37, 146A–151A.

Camargo, J. A., Alonso, A., Salamanca, A. 2005. Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. Chemosphere 58, 1255-1267.

Chappell, K. R.; Goulder, R. 1994. Enzymes as rivers pollutants and the response of native epilithic extracellular-enzyme activity. Environ. Pollut. 86, 161-169.

Chróst, R. J., Overbeck, J. 1987. Kinetics of alkaline phosphatase activity and phosphorus availability for phytoplankton and bacterioplankton in Lake Plusssee (North German eutrophic lake). Microbial ecology 13, 229-248

Cleuvers, M. 2003. Aquatic ecotoxicity of pharmaceuticals including the assessment of combination effects. Toxicol. Lett. 142, 185-194.

Coors, A., De Meester, L. 2008. Synergistic, antagonistic and additive effects of multiple stressors: predation threat, parasitism and pesticide exposure in Daphnia magna. J. Appl. Ecol. 45, (6), 1820-1828.

Corcoll, N., Bonet, B., Leira, M., Guasch, H. 2011. Chl-a fluorescence parameters as biomarkers of metal toxicity in fluvial biofilms: an experimental study. Hydrobiol. 673, (1), 119-136.

Cristale, J., Katsoyiannis, A., Sweetman, A. J.; Jones, K. C.; Lacorte, S. 2013. Occurrence and risk assessment of organophosphorus and brominated flame retardants in the River Aire (UK). Environ. Pollut. 179, 194-200.

De Castro-Català, N., Muñoz, I., Armendáriz, L., Campos, B., Barceló, D., López-Doval, J., Pérez, S., Petrovic, M., Picó, Y., Riera, J. L. 2015. Invertebrate community responses to emerging water pollutants in Iberian river basins. Sci. Total Environ. 503-504, 142-150.

De Castro-Català , N., López-Doval, J., Gorga, M., Petrovic, M., Muñoz, I. 2013. Is reproduction of the snail *Physella acuta* affected by endocrine disrupting compounds? An in situ bioassay in three Iberian basins. J. Hazard. Mat. 263, Part 1, 248-255.

De Lange, H. J., De Jonge, J., Den Besten, P. J., Oosterbaan, J., Peeters, E. T. H. M. 2004. Sediment pollution and predation affect structure and production of benthic macroinvertebrate communities in the Rhine and Meuse delta, The Netherlands. Journal of the North American Benthological Society 23, (3), 557-579.

De Zwart, D., Posthuma, L., Gevrey, M., Von der Ohe, P. C., de Deckere, E. 2009. Diagnosis of ecosystem impairment in a multiple-stress context: how to formulate effective river basin management plans. Integrated Environmental Assessment and Management 5, (1), 38-49.

Di Toro, D.M., Allen, H.E., Bergman, H.L., Meyer, J.S., Paquin, P.R., Santore, R.C., 2001. Biotic ligand model of the acute toxicity of metals. 1. Technical basis. Environ. Toxicol. Chem. 20, 2383–2396.

Dudgeon, D. 2010. Prospects for sustaining freshwater biodiversity in the 21st century: linking ecosystem structure and function. Current Opinion in Environmental Sustainability 2, 422-30.

Eggen, R.I., Behra, R., Burkhardt-Holm, P., Escher, B.I., Schweigert, N., 2004. Challenges in ecotoxicology. Environ. Sci. Technol. 38, 58A–64A.

Elosegi, A., Sabater, S. 2013. Effects of hydromorphological impacts on river ecosystem functioning: a review and suggestions for assessing ecological impacts. Hydrobiol. 712, 129-143.

Fore, L. S., Karr, J. R., Wisseman, R. W. 1996. Assessing Invertebrate Responses to Human Activities: Evaluating Alternative Approaches. Journal of the North American Benthological Society 15, (2), 212-231.

Gago-Ferrero, P., Maistroianni, N., Diaz-Cruz, S., Barceló, D. 2013. Fully automated determination of nine ultraviolet filters and transformation products in natural waters and wastewaters by on-line solid phase extraction-liquid chromatography-tandem mass spectrometry J. Chromatogr. A 1295, 57-66.

Geyer, H. J., Rimkus, G. G., Scheuner, I., Kaune, A., Schramm, K.-W., Kettrup, A.,Zeeman, M., Muir, D. C. G., Hansen, L. G., Mackay, D. 2000. Bioaccumulation and Occurrence of Endocrine Disrupting Chemicals (EDCs), Persistent Organic Pollutants (POPs), and Other Organic Compounds in Fish and Other Organisms Including Humans. In Bioaccumulation Beek, B., Ed. Springer-Verlag Berlin Heidelberg; Vol. The Handbook of Environmental Chemistry 2, 1.

Ginebreda, A., Kuzmanovic, M., Guasch, H., de Alda, M. L. p., López-Doval, J. C., Muñoz, I.; Ricart, M.; Romaní, A. M.; Sabater, S., Barceló, D. 2014. Assessment of multi-chemical pollution in aquatic ecosystems using toxic units: Compound prioritization, mixture characterization and relationships with biological descriptors. Sci. Total Environ. 468-469, 715-723.

Goodnight, C. J. 1956 The use of aquatic invertebrates as indicators of stream pollution. Transactions of the American Microscopical Society 92, (1), 1-13.

Gorga, M., Petrovic, M., Barceló, D. 2013. Multi-residue analytical method for the determination of endocrine disruptors and related compounds in river and waste water using dual column liquid chromatography switching system coupled to mass spectrometry. J. Chromatogr. A 1295, 57-66.

Gros, M., Petrovic, M., Barceló, D. 2009. Tracing PhACs residues of different therapeutic classes in Environmental Waters by using liquid chromatography/quadrupole-linear ion trap mass spectrometry and automated library searching. Anal. Chem. 81, 898-912.

Hatt, B. E., Fletcher, T. D., Walsh, C. J., Taylor, S. L. 2004. The influence of urban density and drainage infrastructure on the concentration and loads of pollutants in small streams. Environmental Management 34, (1), 112-124.

Heger, W., S. J. Jung, S. Martin and H. Peter (1995). "Acute and prolonged toxicity to aquatic organisms of new and existing chemicals and pesticides. 1. Variability of the acute to prolonged ratio 2. Relation to logPow and Water Solubility." Chemosphere 31(2): 2707-2726.

Ippolito, A., Carolli, M., Varolo, E., Villa, S., Vighi, M. 2012. Evaluating pesticide effects on freshwater invertebrate communities in alpine environment: a model ecosystemexperiment. Ecotoxicology 21, 2051–2067.

Johnson, R. K., Hering, D. 2009. Response of taxonomic groups in streams to gradients in resource and habitat characteristics. J. Appl. Ecol. 46, 175-186.

Kelly, M., Cazaubon, A., Coring, E., Dell'Uomo, A., Ector, L., Goldsmith, B., Guasch, H., Hürlimann, J., Jarlman, A., Kawecka, B. 1998. Recommendations for the routine sampling of diatoms for water quality assessments in Europe. J. Appl. Phycol. 10, (2), 215-224.

Kuzmanovic, M., Ginebreda, A., Petrovic, M., Barceló, D. 2015. Risk assessment based prioritization of 200 organic micropollutants in 4 Iberian rivers. Sci. Total Environ. 503-4, 289-99.

Kuzmanović, M.,López-Doval, J.C., De Castro-Català, N., Guasch, H., Petrović, M., Muñoz, I., Ginebreda, A., Barceló, D. 2015. Ecotoxicological risk assessment of chemical pollution in four Iberian river basins and its relationship with the aquatic macroinvertebrate community status. Sci. Total Environ. 540,324-333

Länge, R., T. H. Hutchinson, N. Scholz and J. Solbé (1998). "Analysis of the ECETOC Aquatic Toxicity (EAT) database. II - Comparison of acute to chronic ratios for various aquatic organisms and chemical substances." Chemosphere 36(1): 115-127.

Lecointe, C., Coste, M., Prygiel, J. 1993. "OMNIDIA" software for taxonomy, calculation of diatom indices and inventories management. Hydrobiol. 269/270, 509-513.

Legendre, P.; Legendre, L.1998. Numerical Ecology. Elsevier B.V.: Amsterdam, The Netherlands, Vol. 20.

Liess, M., Von Der Ohe, P.C. 2005. Analyzing effects of pesticides on invertebrate communities in

streams. Environ. Toxicol. Chem. 24, 954–965.

Liess, M., Schäfer, R.B., Schriever, C.A. 2008. The footprint of pesticide stress in communities—species traits reveal community effects of toxicants. Sci. Total Environ. 406, 484–490.

Liess, M., Beketov, M. 2011. Traits and stress: keys to identify community effects of low levels of toxicants in test systems. Ecotoxicology 20, 1328–1340.

Liess, M.; Foit, K., Becker, A., Hassold, E., Dolciotti, I., Kattwinkel, M., Duquesne, S. 2013. Culmination of low-dose pesticide effects. Environ. Sci. Technol. 47, 8862-8.

Lock, M. A., Wallace, R. R., Costerton, J. W., Ventullo, R. M., Charlton, S. E. 1984. River epilithon: toward a structural-functional model. Oikos 42, 10-22.

López-Doval, J. C., Ricart, M., Guasch, H., Romaní, A. M., Sabater, S., Muñoz, I. 2010. Does grazing pressure change modify diuron toxicity in a biofilm community? . Archiv. Environ. Contamin. Toxicol. 58, 955-62.

López-Doval, J.C., De Castro-Català, N., Andrés-Doménech, I., Blasco, J., Ginebreda, A., Muñoz, I., 2012. Analysis of monitoring programmes and their suitability for ecotoxicological risk assessment in four Spanish basins. Sci. Total Environ. 440, 194–203.

Luo, Y.; Guo, W.; Ngo, H. H.; Nghiem, L. D.; Hai, F. I.; Zhang, J.; Liang, S.; Wang, X. C. 2014. A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. Sci. Total Environ. 473-474, (0), 619-641.

Malaj, E., Von der Ohe, P. C., Groted, M., Kühne, R., Mondy, C. P., Usseglio-Polatera, P., Brack, W.; Schäferb, R. B. 2014. Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. Proc. Natl. Acad. Sci. 111, (26), 9549-9554.

Malmqvist, B. 2002. Aquatic invertebrates in riverine landscapes. Freshwat. Biol. 47, (4), 679-694.

Margalef, R. 1960. Ideas for a synthetic approach to the ecology of running waters. Int. Revue ges. Hydrobiol. 45, (1), 133-153.

Margalef, R., 1983. Limnología. Omega, Barcelona.

Masià, A., Ibañez, M., Blasco, C., Sancho, J. V., Picó, Y., Hernández, F. 2013. Combined use of liquid chromatography triple quadrupole mass spectrometry and liquid chromatography quadrupole time-of-flight mass spectrometry in systematic screening of pesticides and other contaminants in water samples. Anal. Chim. Acta 761, (0), 117-127.

McKnight US, Rasmussen JJ, Kronvang B, Binning PJ, Bjerg PL. 2015. Sources, occurrence and predicted aquatic impact of legacy and contemporary pesticides in streams. Environ. Pollut. 200: 64-76.

Meybeck, M. 2004. The global change of continental aquatic systems: dominant impacts of human activities. Water Sci.e Technol. 49, (7), 73-83.

Navarro-Ortega, A., Acuña, V., Batalla, R.J., Blasco, J., Conde, C., Elorza, F.J., et al. 2012. Assessing and forecasting the impacts of global change on Mediterranean rivers. The SCARCE Consolider project on

Iberian basins. Environ. Sci. Pollut. Res. 19, 918–933.

Muñoz, I., López-Doval, J. C., Ricart, M., Villagrasa, M., Brix, R., Geiszinger, A., Ginebreda, A., Guasch, H., López de Alda, M., Romaní, A. M., Sabater, S., Barceló, D. 2009. Bridging levels of pharmaceuticals in river water with biological community structure in the Llobregat river basin (northeast Spain). Environ. Toxicol. Chem. 28, (12), 2706-2714.

Nikolaidis, N. P., Heng, H., Semagin, R., Clausen, J. C. 1998. Non-linear response of a mixed land use watershed to nitrogen loading. Agriculture Ecosystems And Environment 67, (2-3), 251-265.

Pesce, S., Morin, S., Lissalde, S., Montuelle, B., Mazzella, N. 2011. Combining polar organic chemical integrative samplers (POCIS) with toxicity testing to evaluate pesticide mixture effects on natural phototrophic biofilms. Environ. Pollut. 159, (3), 735-741.

Petrovic, M., Ginebreda, A., Acuña, V., Batalla, R. J., Elosegi, A., Guasch, H., de Alda, M. L., Marce, R., Muñoz, I.,Navarro-Ortega, A., Navarro, E., Vericat, D.,Sabater, S., Barceló, D. 2011. Combined scenarios of chemical and ecological quality under water scarcity in Mediterranean rivers. TrAC Trends in Analytical Chemistry 30, (8), 1269-1278.

Poff, N. L. 1997. Landscape filters and species traits: Towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16, (2), 391-409.

Poor, C. J.; McDonnell, J. J. 2007. The effects of land use on stream nitrate dynamics. J. Hydrol. 332, (1-2), 54-68.

Ponsati L., Corcoll N., Petrovic M., Picó Y., Ginebreda A., Tornés E., Guasch H., Barceló D., Sabater S. 2016. Multiple-stressor effects on river biofilms under different hydrological conditions. Freshwater Biol. 1-14.

Posthuma, L., Bjørn, A., Zijp, M. C., Birkved, M., Diamond, M. L., Hauschild, M. Z., Huijbregts, M. A. J.; Mulder, C.,Van de Meent, D. 2014. Beyond Safe Operating Space: Finding Chemical Footprinting Feasible. Environ. Sci. Technol. 48 6057−6059.

Posthuma, L., Eijsackers, H., Koelmans, J. P. A., Vijver, M. G. 2008. Ecological effects of diffuse mixed pollution are site-specific and require higher-tier risk assessment to improve site management decisions: A discussion paper. Sci. Total Environ. 406, 503-517.

Proia, L., Osorio, V., Soley, S., Köck-Schulmeyer, M., Pérez, S., Barceló, D., Romaní, A., Sabater, S. 2013. Effects of pesticides and pharmaceuticals on biofilms in a highly impacted river. Environ. Pollut. 178, 220-228.

Rasmussen, J.J., McKnight, U.S., Loinaz, M.C., Thomsen, N.I., Olsson, M.E., Bjerg, P.L., et al., 2013. A catchment scale evaluation of multiple stressor effects in headwater streams. Sci. Total Environ. 442, 420–431.

Ricart, M., Guasch, H., Barceló, D., Brix, R.; Conceição, M. H.; Geiszinger, A.; López de Alda, M.; López-Doval, J.; Muñoz, I.; Romaní, A. M.; Villagrassa, M.; Sabater, S. 2010. Primary and complex stressors in polluted Mediterranean rivers: pesticide effects on biological communities. J. Hydrol. 383, 52-61.

Round, F. E. 1981. The ecology of the algae. Cambridge Univ. Press: Cambridge, p 653.

S. Sabater, D. Barceló, N. De Castro-Catalá, A. Ginebreda, M. Kuzmanovic, M. Petrovic, Y. Picó, L. Ponsatí b, E. Torné, I. Muñoz. 2016. Shared effects of organic microcontaminants and environmental stressors on biofilms and invertebrates in impaired rivers. Environ. Pollut. 210, 303-314

Sabater, S., Guasch, H., Ricart, M., Vidal, G., Klünder, C., Schmitt-Jansen, M. 2007. Monitoring the effect of chemicals on biological communities. The biofilm as an interface. Anal. Bioanal. Chem. 387, (4), 1425-1434.

Sabater, S., Guasch, H., Romani, A., Munoz, I. 2000. The effect of biological factors on the efficiency of river biofilms in improving water quality. Hydrobiol. 469, (1-3), 149-156.

Schäfer, R.B., Caquet, T., Siimes, K., Mueller, R., Lagadic, L., Liess, M. 2007. Effects of pesticides on community structure and ecosystem functions in agricultural streams of three biogeographical regions in Europe. Sci. Total Environ. 382: 272-285.

Schäfer, R. B., P. C. Von Der Ohe, R. Kühne, G. Schüürmann and M. Liess (2011). "Occurrence and toxicity of 331 organic pollutants in large rivers of north Germany over a decade (1994 to 2004)." Environmental Science and Technology 45(14): 6167-6174.

Schäfer, R. B., P. C. Von Der Ohe, J. Rasmussen, B. J. Kefford, M. A. Beketov, R. Schulz and M. Liess (2012). "Thresholds for the effects of pesticides on invertebrate communities and leaf breakdown in stream ecosystems." Environmental Science and Technology 46(9): 5134-5142.

Schäfer, R.B., Gerner, N., Kefford, B.J., Rasmussen, J.J., Beketov, M.A., De Zwart, D. 2013.How to characterize chemical exposure to predict ecologic effects on aquatic communities? Environ. Sci. Technol. 47: 7996-8004

Segner, H., Schmitt-Jansen, M., Sabater, S. 2014. Assessing the impact of multiple stressors on aquatic biota: the receptor's size matters. Environ. Sci. Technol. 48, 7690-96.

Statzner, B., Bêche, L.A., 2010. Can biological invertebrate traits resolve effects of multiple stressors on running water ecosystems? Freshwater Biol. 55, 80–119.

Stehle, S., Schulz, R. 2015. Agricultural insecticides threaten surface waters at the global scale. Proc.Nat. Acad. Sci. 112, (18), 5750-5.

Stevenson, R. J., Sabater, S. 2010. Understanding global change in River Ecosystems: Science to support Policy in a Changing World. Hydrobiol. 657. , 3-18.

ter Braak, C. J. F., Smilauer, P. 2002. CANOCO reference manual and CanoDraw for Windows user's guide: software for canonical community ordination (version 4.5). Microcomputer Power: Ithaca, New York, USA.

ter Braak, C. J. F.; Verdonschot, P. F. M. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57, 255-289.

USEPA, 2008. USEPA. ECOTOX 4.0 ecotoxicology database [Accessible]. http://cfpub.epa.gov/ecotox.

Van Wijngaarden, R. P. A., T. C. M. Brock and P. J. Van Den Brink (2005). "Threshold levels for effects of insecticides in freshwater ecosystems: A review." Ecotoxicology 14(3): 355-380.

Von der Ohe, P.C., Goedkoop, W. 2012. Distinguishing the effects of habitat degradation and pesticide stress on benthic invertebrates using stressor-specific metrics. Sci. Total Environ. 444, 480–490.

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., et al., 2010. Global threats to human water security and river biodiversity. Nature 467, 555–561.

Wallstrom, S., Stromberg, E., Karlsson, S. 2005. Microbiological growth testing of polymeric materials: an evaluation of new methods. Polymer Testing 24, (5), 557-563.

Wolfram, G., Höss, S., Orendt, C., Schmitt, C., Adámek, Z., Bandow, N., et al., 2012. Assessing the impact of chemical pollution on benthic invertebrates from three different European rivers using a weight-of-evidence approach. Sci. Total Environ. 438, 498–509.

Yergeau, E.; Lawrence, J. R.; Waiser, M. J.; Korber, D. R.; Greer, C. W. 2010. Metatranscriptomic analysis of the response of river biofilms to pharmaceutical products, using anonymous DNA microarrays. Appl. Environ. Microbiol. 76, 5432-5439.

7. Report on the application of selected models to Iberian Mediterranean basins (Internal Deliverable ID 3.3)

7.1 Introduction

7.1.1. Concentrations of chemicals in Iberian Basins

During the Spanish project SCARCE (Navarro-Ortega et al., 2012a; 2012b), over 200 organic priority and emerging pollutants were comprehensively monitored in water, sediment and biota from four Iberian river basins (Llobregat, Ebro, Júcar, and Guadalquivir, see Figure 7.2). The results of these measurements have been made available to the SOLUTIONS project by partner CSIC, and have been acquired by the authors from project partner EI, who has added these data to the SOLUTIONS database. These data encompass endocrine disruptors, drugs of abuse, perfluorinated compounds, pesticides, pharmaceuticals and UV filter compounds.

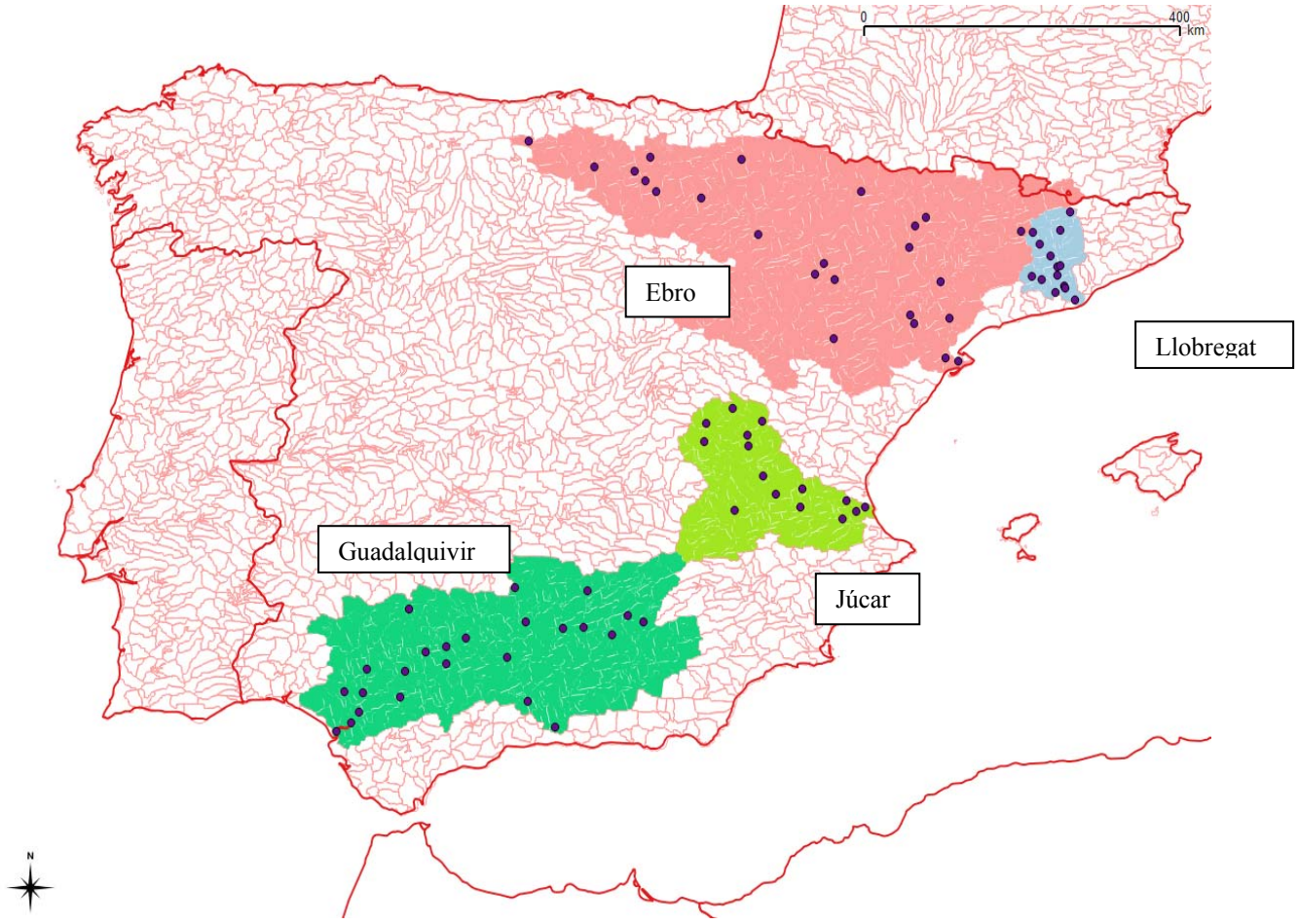


Figure 7.2: Basins under study and location of monitoring stations

Table 7.1: Selected features of the basins under study (López de Alda et al., 2015)

River Basin	Catchment Area (km^2)	River Length (km)	Mean Precipitation (mm/y)	Mean discharge (m^3/s)	Population Density ($/km^2$)
Llobregat	4957	165	650	20	545
Ebro	85362	928	672	425	34
Júcar	21578	512	448	26	307
Guadalquivir	57071	657	520	229	69

The catchment area, mean discharge and population of the four basins are listed in Table 7.1. These Iberian river basins are characterized by drought situations (see Figure 7.2), when WWTP effluents may represent almost 100% of the total flow of the rivers, showing potential hazardous consequences for human health and the ecosystem. This situation is of special concern in the industrialized areas of the Mediterranean region, where water scarcity can worsen the existing effects of human pressure.

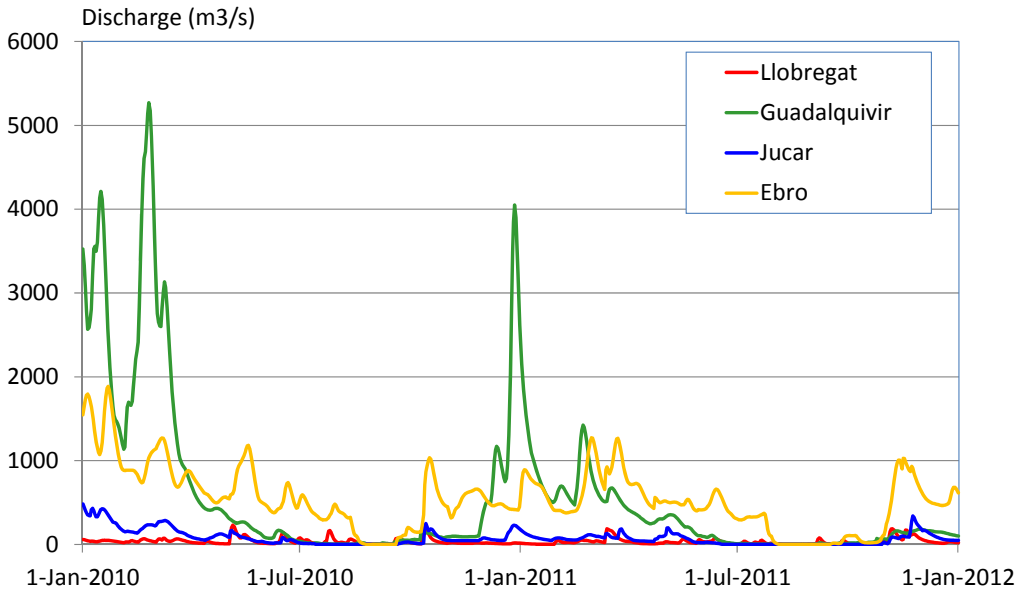


Figure 7.3: Intermittent character of Iberian rivers: E-Hype simulated discharge in m^3/s during 2010-2011 (Donnelly et al., 2013). Drought periods: late August – September 2010; August - October 2011.

7.1.2.The SOLUTIONS Model Train

The SOLUTIONS project is developing a collection of integrated models, to increase our understanding

of issues related to emerging chemicals in Europe's river basins, to support the prioritisation of chemicals and the abatement of the problems they cause and to evaluate future scenarios. This collection of models is referred to as the "Model Train". The model train consists of four key building blocks (see Figure 7.3): (a) the prediction of substance properties based on their molecular structure, (b) the simulation of emissions, (c) the simulation of fate & transport, and (d) the characterisation of the risk of mixtures of chemicals for human health and aquatic ecosystems.

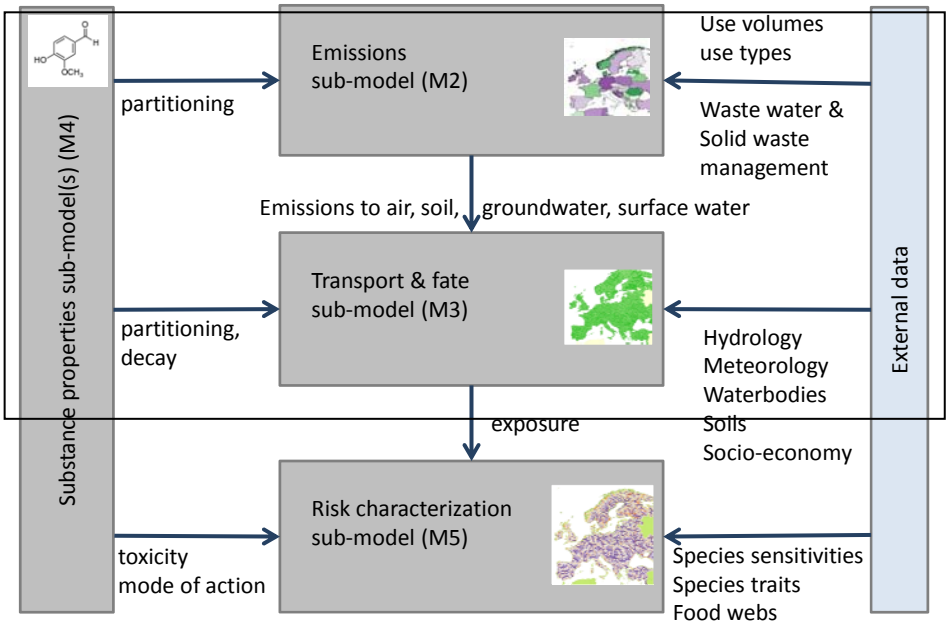


Figure 7.4: schematic overview of the model train.

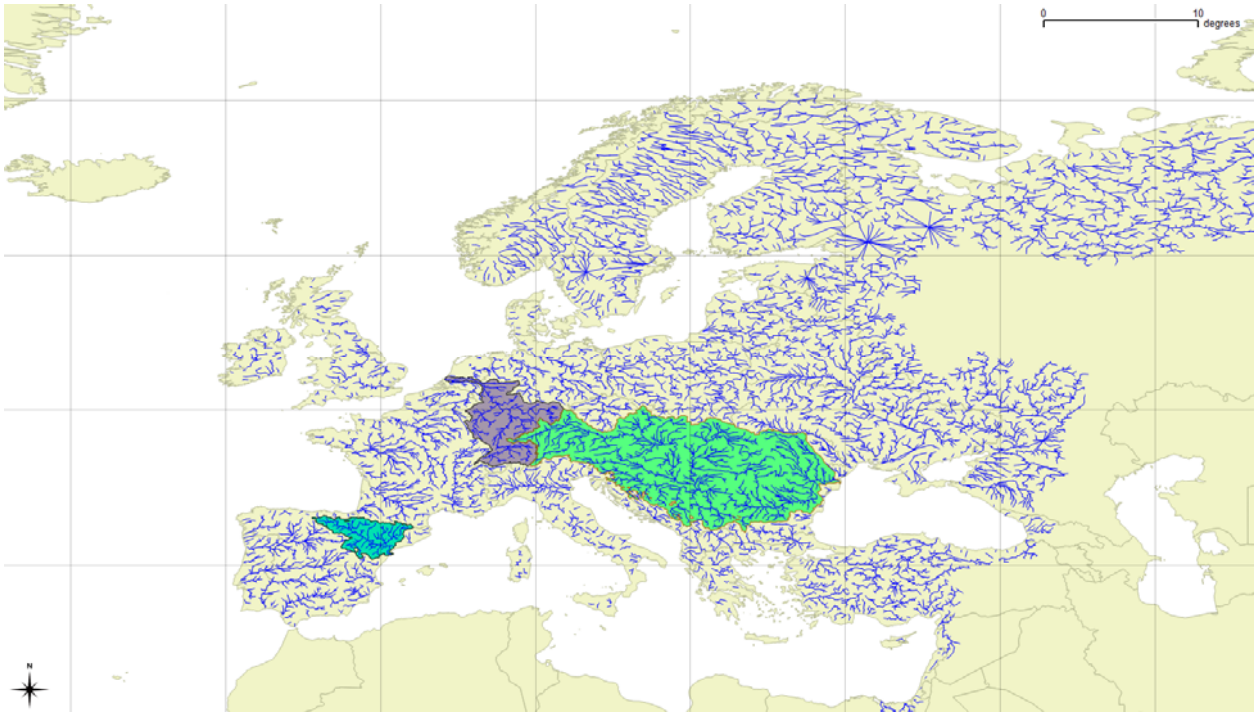


Figure 7.5: Europe-wide schematization of E-Hype model (Donnelly et al., 2013) and case study basins in SOLUTIONS (Rhine, Danube, Ebro).

The model train operates on the scale of Europe as a whole or for one or more individual river basins. The spatial schematization as well as the hydrology, soil, land use and crop cover are derived from the Europe-wide hydrology model E-Hype, developed by SMHI (Donnelly et al., 2013), see Figure 7.4. The model train takes into account the day-to-day variations of the emissions and hydrology, and can, therefore, separate for example between acute short-term effects and chronic effects.

The substances properties models used are all pre-existing models. An overview of these models and the results they produce is available in SOLUTIONS Deliverable 17.2 (Kutsarova et al. 2017). The fate and transport model is called STREAM-EU (Spatially and Temporally Resolved Exposure Assessment Model for European basins). Its description and some relevant applications have been published (Lindim et al., 2016; 2016a; 2017). A tiered assessment framework for human and environmental risk assessment has been laid out by Kortenkamp et al. (2016). It encompasses multi-species environmental risk assessment (ERA) via species sensitivity distributions (SSD) (De Zwart et al. 2006) and population-level environmental risk assessment via individual-based models (IBM) (Baveco and Focks, 2017). The full documentation of the model train is in progress and planned to be completed in November 2017 (van Gils, et al., 2017).

7.1.3. Scope and objectives of the present deliverable

This Deliverable aims at exploiting the abovementioned SCARCE data in the best possible way for the validation of the model train. We focus on the combined use of the substance properties models, the emissions model and the fate & transport model (indicated by the box in Figure 7.3). The validation of the risk characterization models is partly already done in previous projects (for the existing components) and is partly done in other case studies.

Thus, this deliverable aims at validating the abovementioned components of the model train by a comparison between simulated concentrations and observed concentrations in Iberian rivers. First, we look at the overall (average) level of concentrations for a range of chemicals. This is not only done in the Spanish rivers, but also for the other Case Studies in the Rhine and in the Danube, as well as in Swedish rivers (specifically for pesticides) (van Gils et al., 2017). In addition, we look at the spatial variability of the concentrations in the Spanish river basins. The wide range of stations in smaller and bigger streams is a strong feature of the SCARCE dataset that we will try to exploit.

In summary, the objective of the work presented here is:

to evaluate the validity of the first parts of the model train that produce predicted environmental concentrations, both with respect to overall levels and with respect to spatial patterns

The application of the model train in support of the prioritization of chemicals and to evaluate future scenarios is not yet possible, in view of the fact that the train has not been fully finalized. These results will be included in future deliverables, also for the Spanish river basins.

7.2. Materials and Methods

7.2.1. Field data processing

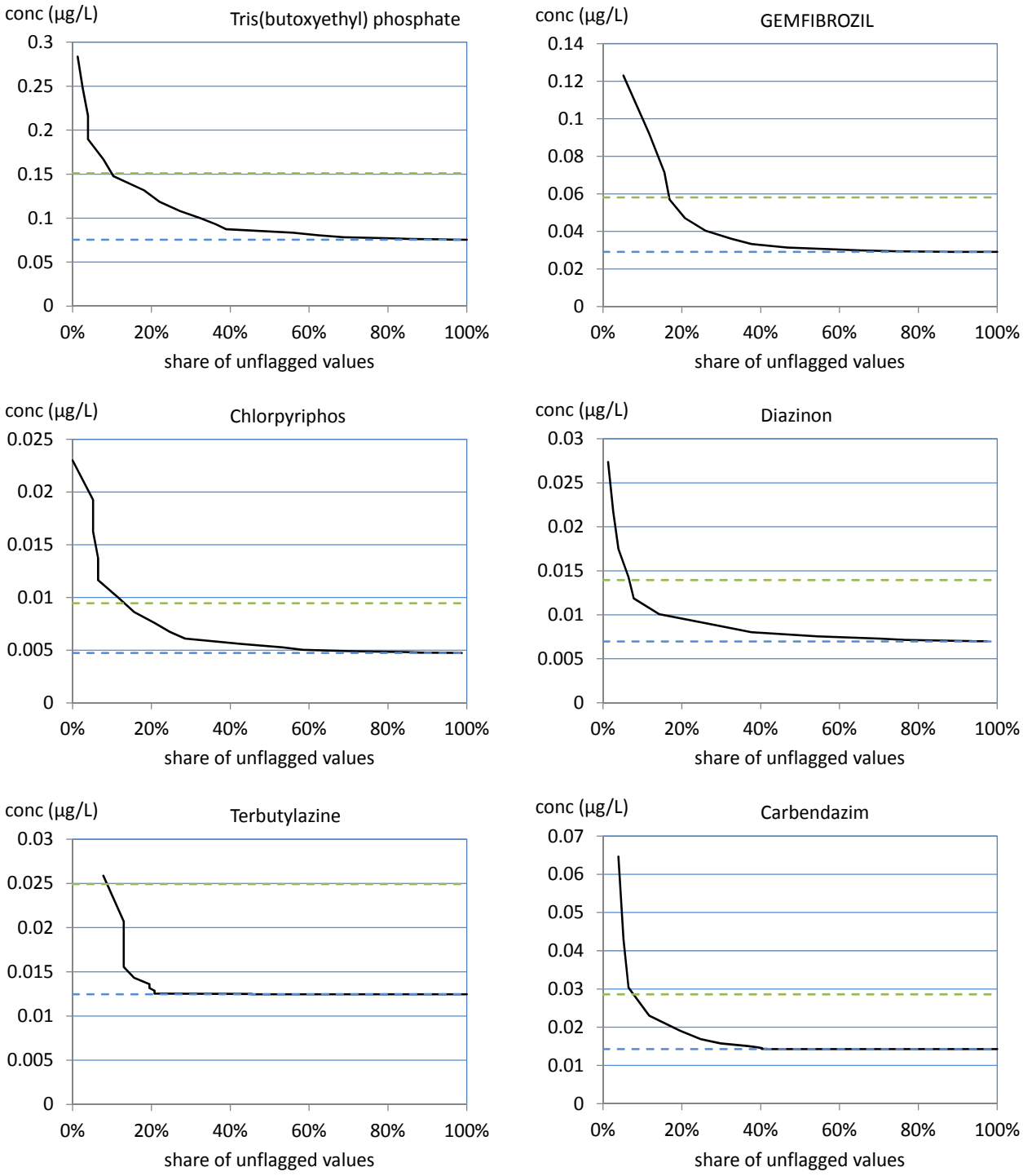
Sampling was performed in September and October of 2010 and in June to November of 2011 (with the majority of samples also from September and October, see Annex III). The monitoring stations, also shown in Figure 7.2, number 24 in the Ebro basin, 24 in the Guadalquivir basin, 15 in the Júcar basin and 14 in the Llobregat basin. The total number of analyses in the dataset amounts to 28,077 records. All values reported as smaller than the Limit of Detection (LoD) or the Limit of Quantification (LoQ) were replaced by these respective limit values, and these values were flagged. In this process, we had to omit the records where the LoD or the LoQ was not specified, which reduced the total size of the database to 27,061 records. We further omitted chemicals without a CAS number and chemicals that were not analysed for all stations, the latter to maintain the homogeneity of the dataset.

The data have been further processed and analysed in two ways (a) for assessment of overall model performance, and (b) for assessment of spatial gradients.

7.2.1.1. Processing for evaluating overall model performance

For the assessment of overall model performance, we calculated the average of all values of a single chemical per station and next to the average of the values of all stations per chemical. We also calculated the percentage of unflagged values (not affected by LoD/LoQ). The result is tabulated in Annex III.
For the assessment of overall model performance, we tested how many unflagged values are necessary to still approach the average value with a reasonable accuracy (within a factor of two). We did that by studying the results for selected chemicals from different substance groups with no or as few as possible flagged values. By applying hypothetical limit values that replace all values below the limit, by counting the number of replacements and by recalculating the average using the limit value instead of the replaced value, we established a relationship between the percentage of unflagged values and the error in the calculated mean value. The results are shown in Figure 7.5. If all analyses are unflagged (fraction 100%), the real average is calculated. When the fraction of unflagged values decreases, the average is not affected a lot, until the fraction reaches values below 0.2. In all cases shown in Figure 7.5, a fraction of unflagged analyses of 20% is sufficient to estimate the mean value with an error less than a factor of 2.

The first two substances in Figure 7.5, tris(butoxyethyl) phosphate and gemfibrozil have 100% and 96% of unaffected values respectively. For those two substances, the approach outlined above is valid. For pesticides however, there are always values affected by LoD/LoQ. The share of unaffected values is 73% for chlorpyriphos, 68% for diazinon, 45% for terbuthylazine and 42% for carbendazim respectively. For these pesticides, the validity of the approach is less obvious. The results however, are robust, so we presume the conclusion valid over the whole range of substances.



Blue dashed lines represent the real average value of all analyses results (obtained with 100% unflagged values). Green dashed lines represent two times that value.

Figure 7.5: Relation between the fraction of unflagged values and the apparent mean concentration.

7.2.1.2. Processing for assessment of spatial gradients

For the assessment of spatial patterns, we aimed to determine which stations show relatively high concentrations and which relatively low concentrations. We first calculated the average of all values per

station and per chemical. Next, we determined for every chemical the median value over all stations, and the ratio to this median per station. The result is a measure of spatial differences that does not depend on the absolute level of the concentration. These values can, therefore, be averaged over groups of substances to obtain a representative picture of the whole group.

This data analysis is again affected by LoD/LoQ issues. Figure 7.6 illustrates this for the case of tris(butoxyethyl) phosphate. The line "Fr = 1" shows the distribution of the ratio C/C_{median} for all 77 stations calculated for the real data, where C is the average concentration of all analyses per station, and C_{median}. is the median of the values of C for individual stations. For this chemical, all stations are unaffected by LoD/LoQ. (Note that an "unaffected station" means that there is at least one unaffected analysis result at that station). The ratio varies from 0.31 to 6.8 and is exactly 1.0 in the middle. Figure 7.6 Figure shows what happens to this result if we create an increasing artificial LoD/LoQ that reduces the share of stations unaffected from 100% to 81%, 62%, 39%, 22% and 10% respectively. While the share of unaffected stations decreases from 100% to 50%, the values of the ratio C/C_{median} below 1.0 on the left side loose accuracy, and finally are all equal to 1.0 when the share of unaffected samples is 50%. This is the direct consequence of the use of a median. When the share of unaffected stations decreases below 50%, the real median value is no longer found and the apparent median value increases. The ratio C/C_{median} for all stations still unaffected decreases, because of the apparent increase of the median. Finally, when the share of unaffected stations approaches 0%, the median value approaches the maximum real value in the dataset and all ratios are equal to 1.0.

In summary, information about the ratios below 1.0 (relatively "clean" stations") gets lost while the share of unaffected samples decreases to 50%. A further decrease will also cause information loss about the ratios above 1.0 (relatively "dirty" stations), where the sensitivity decreases and the ratios decrease while the share of unaffected samples decreases to zero.

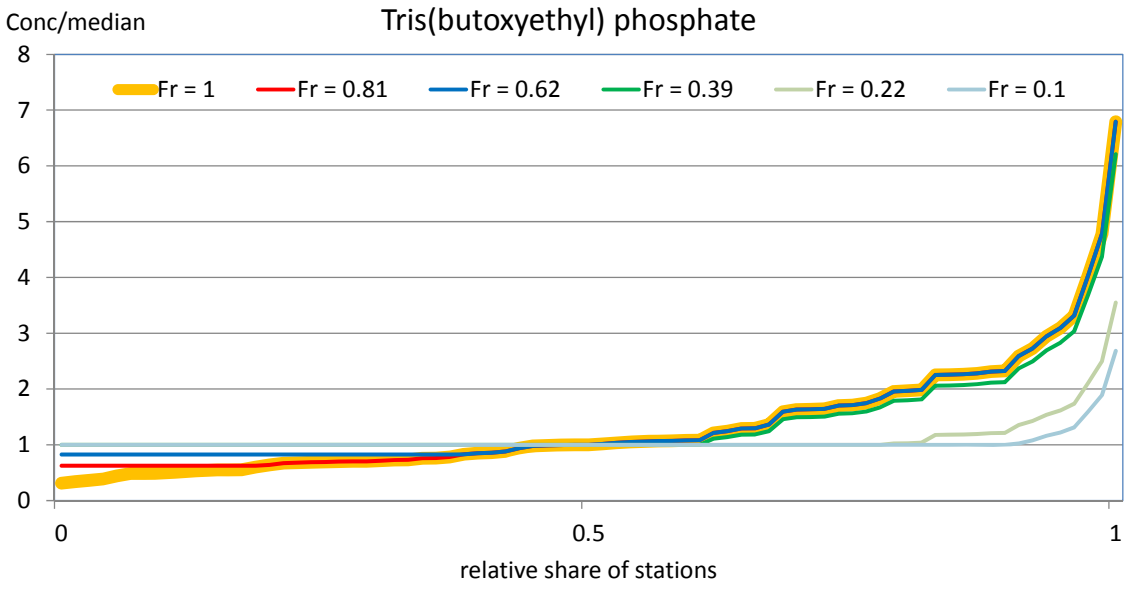


Figure 7.6: Frequency distribution of the ratio of concentration per station and the median of all stations, assuming a variable fraction of unaffected analyses ("Fr").

7.2.2. Model simulations for individual chemicals

The results presented herein are based on simulations for 98 chemicals conducted with the model train as described above. In particular, we combined simulated substances properties with simulated emissions and simulated fate & transport. For details, we refer to the references provided in the Introduction. For the interpretation of the current results, we provide a short overview of the emission estimation methodology (van Gils et al., 2017), see Figure 7.7.

Emission estimates are made for 3 classes of substances. For pharmaceuticals, they are based on sales data, insofar as possible country-specific[1]. The same holds for pesticides. For chemicals registered under REACH, emission estimates are based on Europe-wide so-called "use volumes", the total of production and import minus export. The losses to the environment and to wastewater are estimated on the basis of knowledge and data about the pathways from the place of application to the environment. This is relatively accurate for classes of substances with a well-defined use like pharmaceuticals and pesticides, and less accurate for the diverse group of REACH registered chemicals.

The losses to the environment and to wastewater are spatially distributed on the basis of indicator values (top part of Figure 7.7). Population scaled with Gross Domestic Product (GDP) per country is used to distribute the emissions of pharmaceuticals and of REACH registered chemicals. The rationale behind

[1] In the present simulations we used a preliminary data set for pharmaceuticals that is based only on sales data from Sweden.

this is that a higher standard of living implies a higher use of chemicals. This affects especially the distribution of the emissions of REACH registered chemicals. For pharmaceuticals, the GDP factor is not relevant if country- specific data are available. In the current study, the model domain is situated almost completely in Spain, and the spatial distribution of emissions within the model domain is therefore exclusively determined by the population distribution. The emissions from pesticides follow the distribution of agriculture land use.

After the spatial distribution of the losses to the environment and to wastewater, the model further incorporates the fate of losses to paved areas and to wastewater treatment plants (bottom part of Figure 7.7). This leads to a temporal variation of the emissions to water and soil due to incidental wash off from paved areas. It also leads to a reduction of emissions to water and soil due to treatment and sludge management. The pesticide emissions are further distributed in time, assuming short application periods randomly distributed during the relevant season of cultivation.

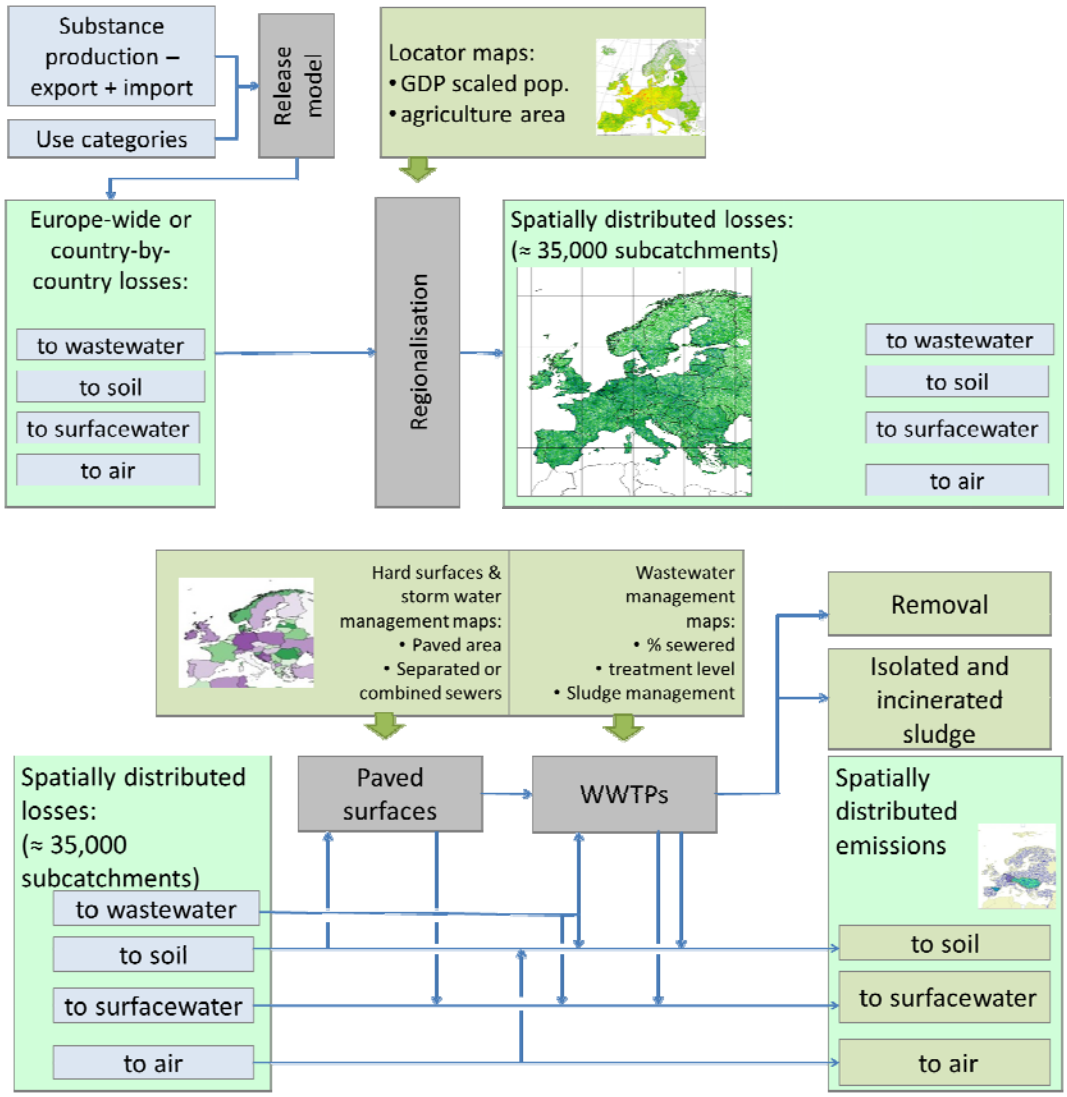


Figure 7.7: Schematic overview of the methodology for the emission estimates.

7.2.3. Model simulations – Assessment of pressures

To support the analysis of the spatial distribution of exposure to chemicals, we conducted separate modelling of "pressures". In the previous section, we explained that population and agricultural land use determine the spatial distribution of emissions. Therefore, we simulated two hypothetical "indicator pollutants", one for population and one for agriculture land use. The model takes into account the temporally and spatially variable hydrology. Per spatial unit ("sub-catchment") we emit a quantity of the two indicator pollutants proportional to the amount of population and agriculture area in the sub-catchment (Figures 7.8 and 7.9). The emitted indicator pollutants are transported downstream and diluted by the river flow. The indicator pollutants are 100% soluble and "passive" or "conservative": they do not undergo any partitioning or removal processes. Thus, the result of these simulations provides a proxy for the accumulated upstream emissions diluted by the available river discharge.

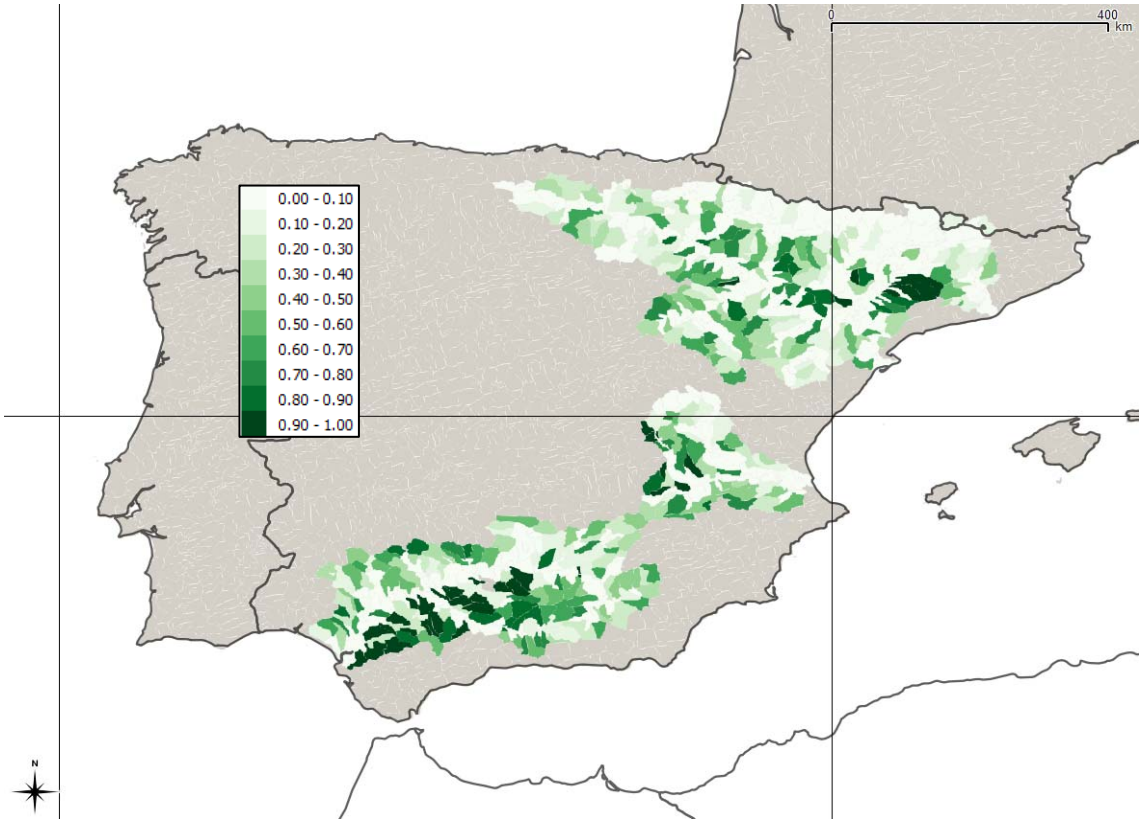


Figure 7.8: Share of agriculture land use (-) in the basins under study (Source: E-Hype model)

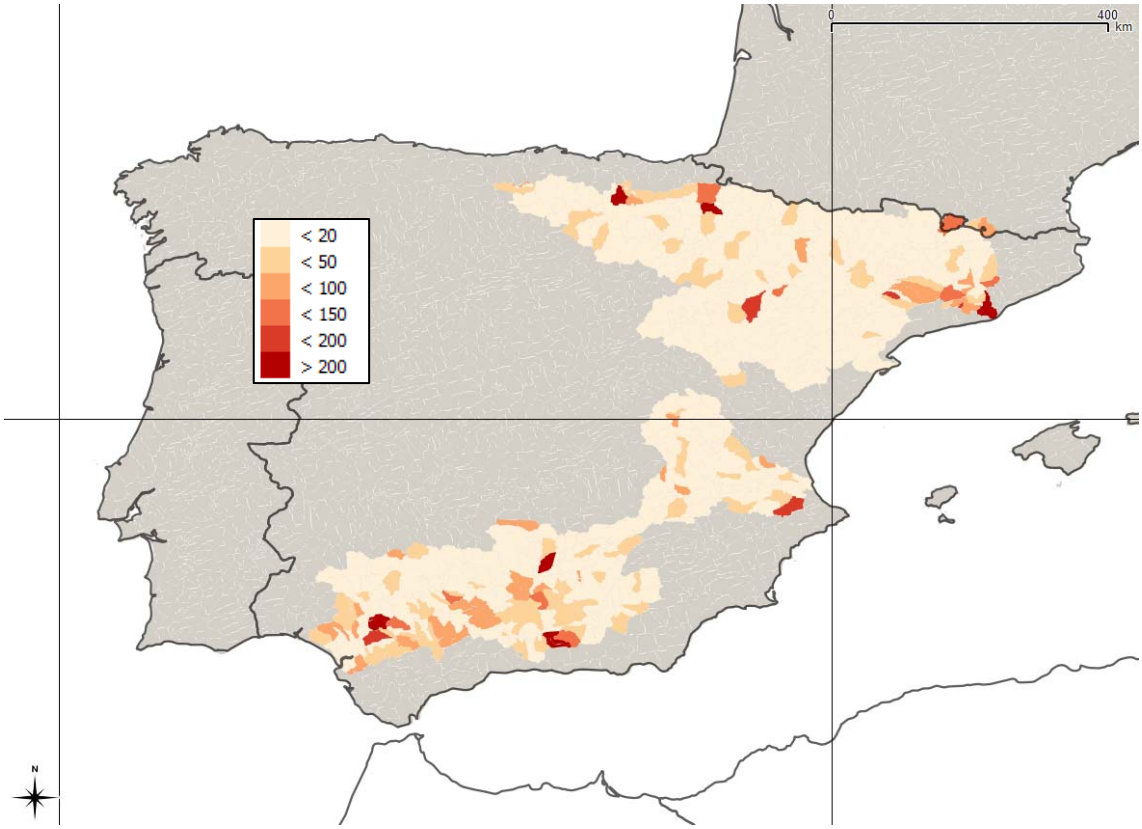


Figure 7.9: Population density (per km^2) in the basins under study (data from 2006).

7.2.4. Combined analysis of field data and simulation results

The simulations of chemicals and indicator pollutants described above are conducted for 2010-2011. Representative results are extracted for the months of September and October only, because these months are drier than average (Figure 7.2 and 7.3) and because the majority of sampling has been done in these months (Annex III). Averages of simulated concentrations are compared to averages of observed concentrations, to assess the overall performance of the models. Note that we use averages here instead of medians because averages are better preserved than medians if the number of unaffected analysis results decreases. This can easily be demonstrated if the assessment presented in Section 7.2.1.1 is repeated for medians. Averages of simulated concentrations are compared to averages of observed concentrations not just for the Spanish rivers discussed here, but also for other SOLUTIONS case studies so that a model evaluation relevant for Europe as a whole is obtained. This will further be reported by van Gils et al. (2017).

This report focuses mostly on the spatial distribution of simulated and observed concentrations, since this a unique feature of the SCARCE data. We will use the ratio C/C_{median} as discussed in Section 7.2.1.1. We will plot it for all stations studied as it emerges from data, from model train simulations, and from indicator pollutant simulations. In this process, chemicals will be clustered in two groups: (a) pesticides, for which emissions are assumed to follow agriculture land use, and (b) all other chemicals, for which emissions are assumed to follow population distribution. By correlation of results derived from field data and results derived from modelling, we will investigate if the methodology for the spatial distribution of emissions is valid.

7.3. Results and Discussion

7.3.1. Average and range of simulated vs. observed concentrations

Figure 7.10 shows the simulated mean concentration of individual chemicals over all stations, plotted against the observed mean concentration at all stations. In this figure, every dot represents one chemical. The figure shows a plot for all chemicals together as well as individual plots per substance group. The number of dots in these plots is fairly limited, since we do not have field data with sufficient analyses unaffected by LoD/LoQ issues for all simulated chemicals.Table 7.2 shows some statistics of these results, for all chemicals together and per substance group. The “bias” is the average value of the logarithm of the ratio of the simulated and the observed mean concentration for all chemicals. The “correlation” is the correlation between the simulated and the observed mean concentration (no logarithm

applied).

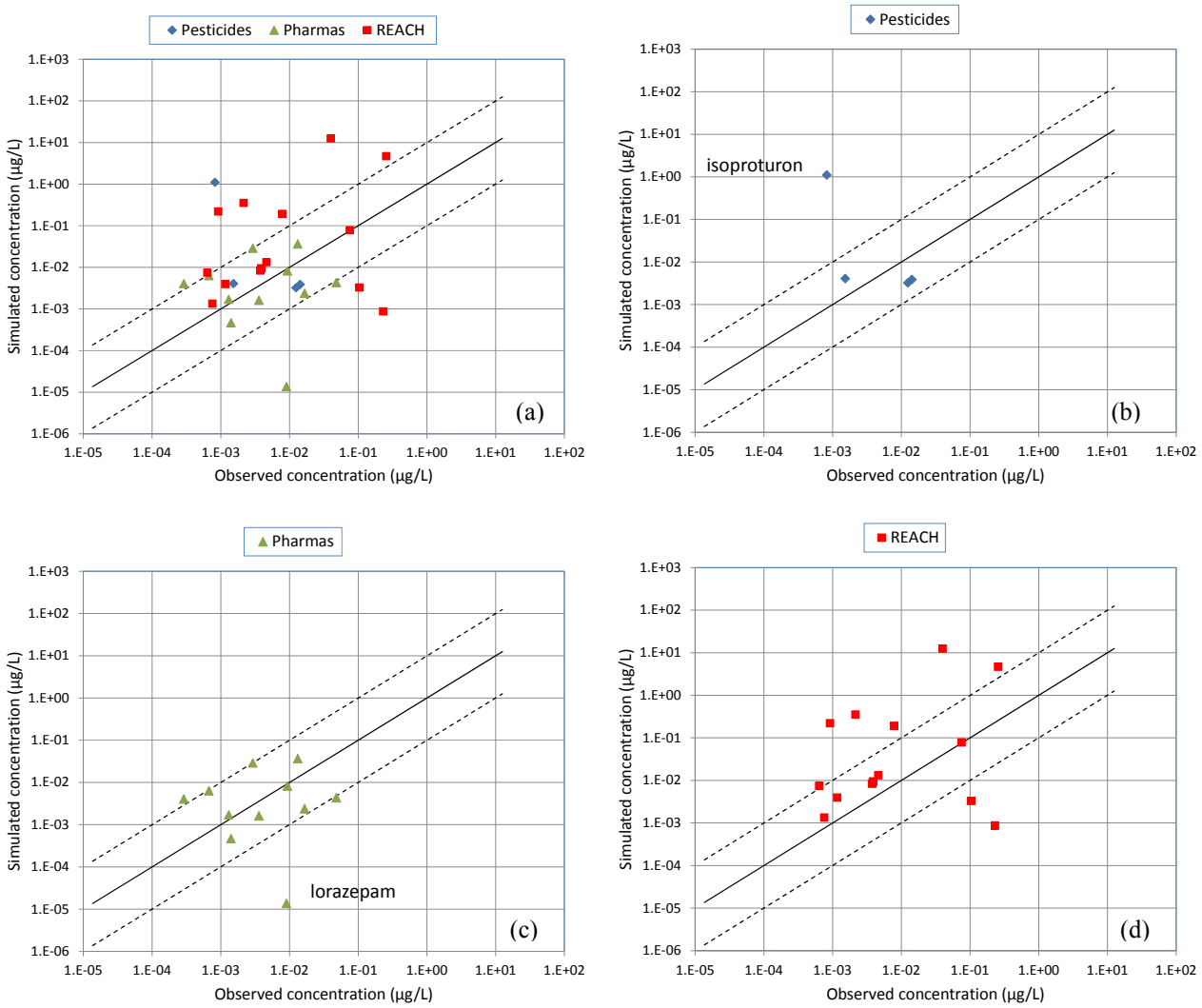


Figure 7.60: Simulated mean of station means versus observed mean of station means for pesticides (b), pharmaceuticals (c), REACH chemicals (d) and all chemicals in one plot (a).

Table 7.2: Summary statistics of comparison of simulated and observed mean of station means.

	All	Pesticides	Pharmas	REACH
N	29	4	11	14
Bias	0.32	0.60	-0.17	0.63
Correlation	0.26	-0.61	-0.01	0.19

Where Figure 7.10 does not show any spatial variability of the observed or simulated concentrations, such information is presented in Figure 7.11. This figure shows the range of simulated station means by a vertical bar and shows the range of the observed station means by three symbols: minimum, average and

mean. The substances are ordered according to the groups they belong to[2] and the substance names are shown along the horizontal axis. Two obvious outliers in the group of pesticides and pharmaceuticals have been marked in Figure 7.10, to facilitate the combined interpretation of the figures. Table 7.3 shows the summary statistics of the observed and simulated ranges of station mean concentrations.

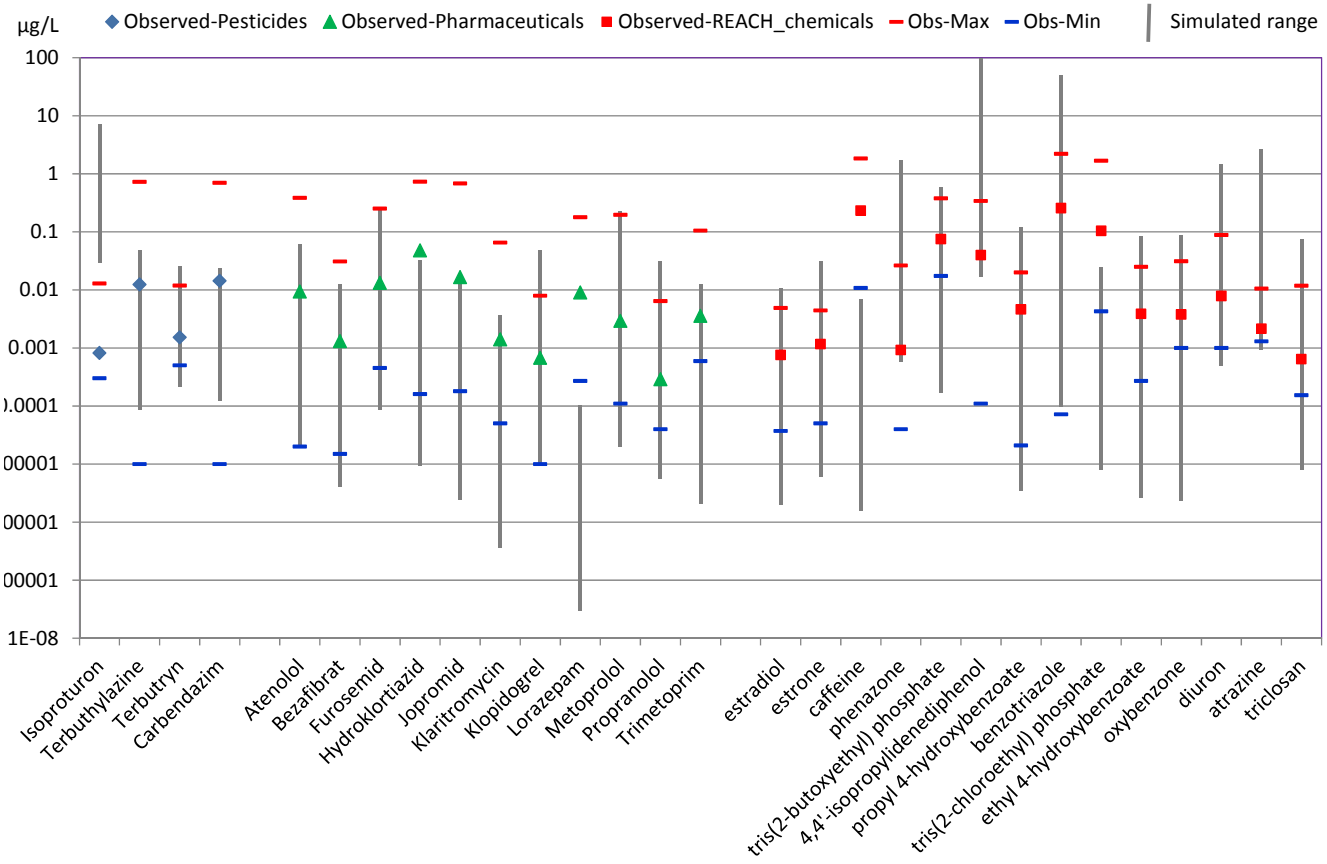


Figure 7.11: Simulated range of station averages (grey bars) vs. average (symbols) and range (red and blue dashes) of observed station averages for pesticides (diamonds), pharmaceuticals (triangles) and REACH registered chemicals (squares).

Table 7.3: Summary statistics of simulated and observed range of station means.

	All	Pesticides	Pharmas	REACH
Count	29	4	11	14
Average Range Data	2.7	3.2	3.1	2.3
Average Range Simulated	3.6	2.4	3.7	4.0

[2] Atrazine and diuron are listed here as part of "REACH chemicals". This grouping stems from the fact that we used the REACH chemicals emission modeling data and algorithm. Whether or not it would be more suitable to use the pesticides related data and algorithm will be discussed by van Gils et al. (2017).

7.3.2. Spatial correlations

In this section we present the results about the relation between simulated and observed spatial patterns. For this purpose, we have aggregated the results for individual substances to two substances groups: pesticides, for which emissions follow agricultural land use, and all other substances, for which emissions follow population distribution. Table 7.4 shows the number of chemicals used for this assessment. We have used the observations in two ways: by using all chemicals where more than 50% of stations have data unaffected by LoD/LoQ issues, and next by using all chemicals where more than 10% of stations have data unaffected by LoD/LoQ issues. As discussed in Section 7.2.1.1., the former brings out all stations with a concentration exceeding the median of all stations, while the latter tends to show only the extremes. In neither case, the field data are expected to give a reliable picture of stations with concentrations below the median of all stations.

Table 7.4: number of chemicals included in spatial analyses.

	Model	Data (> 50% of stations unaffected)	Data (> 10% of stations unaffected)
Other chemicals (N)	79	31	77
Pesticides (N)	19	2	27

The results per station for both substances groups are presented graphically in Figure 7.12 and tabulated in Table 7.5. The stations are clustered per basin and the names of the stations are printed along the horizontal axis.

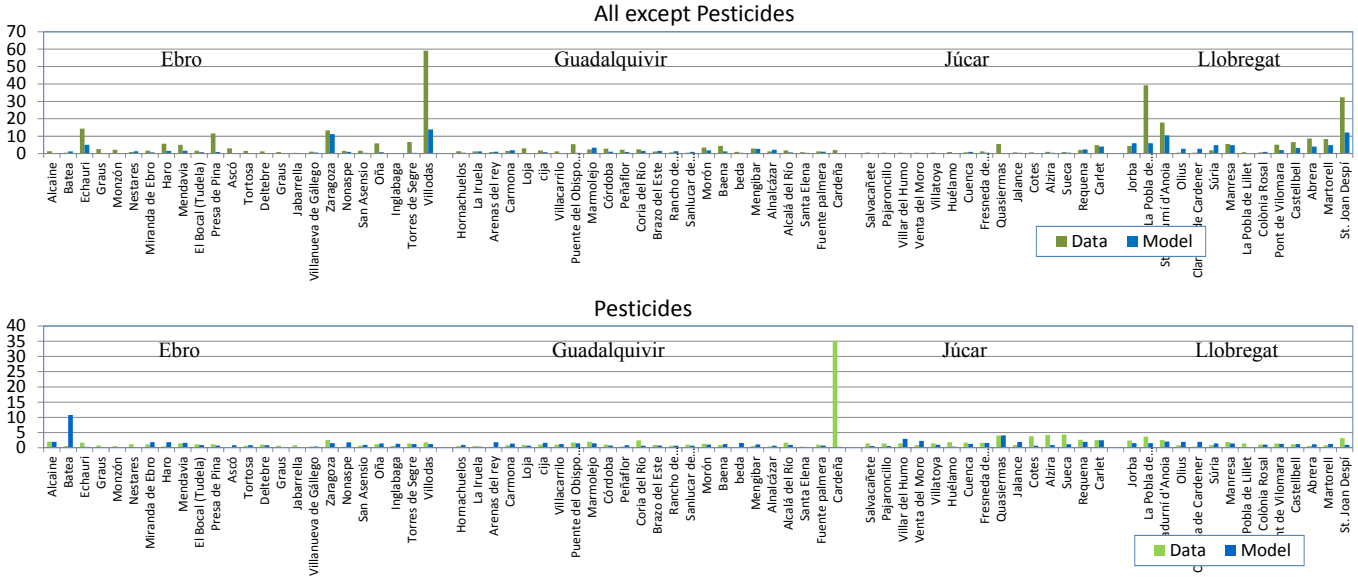


Figure 7.12: Relative concentration, averaged over clusters of chemicals, plotted per station (top: all

except pesticides; bottom: pesticides).

Table 7.5: Observed and simulated relative concentrations for clusters of substances per station.

		All except pesticides		Pesticides	
Station	**Basin**	**Model**	**Data**	**Model**	**Data**
Alcaine	Ebro	0.41	1.49	1.98	2.00
Batea	Ebro	1.32	0.65	10.77	0.60
Echauri	Ebro	5.11	14.41	0.35	1.66
Graus	Ebro	0.00	2.55	0.05	0.76
Monzón	Ebro	0.11	2.19	0.17	0.56
Nestares	Ebro	1.41	1.02	0.16	1.14
Miranda de Ebro	Ebro	0.84	1.66	1.83	1.09
Haro	Ebro	1.54	5.81	1.87	0.51
Mendavia	Ebro	1.63	5.07	1.63	1.42
El Bocal (Tudela)	Ebro	0.90	1.65	0.90	1.13
Presa de Pina	Ebro	1.03	11.55	0.75	1.15
Ascó	Ebro	0.25	3.03	0.84	0.13
Tortosa	Ebro	0.30	1.58	0.85	0.72
Deltebre	Ebro	0.30	1.34	0.82	1.03
Graus	Ebro	0.11	1.00	0.09	0.74
Jabarrella	Ebro	0.14	0.55	0.10	0.85
Villanueva de Gállego	Ebro	0.67	1.18	0.48	0.45
Zaragoza	Ebro	11.22	13.35	1.51	2.56
Nonaspe	Ebro	1.07	1.54	1.77	0.43
San Asensio	Ebro	0.54	1.65	0.91	0.71
Oña	Ebro	0.86	5.84	1.41	1.16
Inglabaga	Ebro	0.16	0.44	1.32	0.68
Torres de Segre	Ebro	0.50	6.72	1.21	1.40
Villodas	Ebro	13.93	59.10	1.19	1.80
Hornachuelos	Guadalquivir	0.56	1.42	0.95	0.57
La Iruela	Guadalquivir	1.35	1.33	0.42	0.63
Arenas del rey	Guadalquivir	1.18	0.97	1.87	0.34
Carmona	Guadalquivir	1.97	1.51	1.39	0.81
Loja	Guadalquivir	0.37	3.05	0.74	0.93
Ecija	Guadalquivir	0.91	1.85	1.61	1.02
Villacarrilo	Guadalquivir	0.24	1.36	1.18	1.01
Puente del Obispo (Baeza)	Guadalquivir	0.54	5.51	1.46	1.74
Marmolejo	Guadalquivir	3.45	2.35	1.46	2.01
Córdoba	Guadalquivir	1.18	2.90	0.79	1.07
Peñaflor	Guadalquivir	0.96	2.23	0.83	0.48
Coria del Río	Guadalquivir	1.57	2.53	0.75	2.45
Brazo del Este	Guadalquivir	1.51	1.27	0.76	0.96
Rancho de Barzoques (Lebrija?)	Guadalquivir	1.45	0.75	0.75	0.74
Sanlucar de Barrameda	Guadalquivir	1.11	0.61	0.70	1.02
Morón	Guadalquivir	1.85	3.52	1.03	1.30

		All except pesticides		Pesticides	
Station	**Basin**	**Model**	**Data**	**Model**	**Data**
Baena	Guadalquivir	1.34	4.49	1.21	0.94
beda	Guadalquivir	0.52	1.11	1.58	0.07
Mengibar	Guadalquivir	2.75	2.91	1.12	0.66
Alnalcázar	Guadalquivir	2.22	1.38	0.71	0.47
Alcalá del Río	Guadalquivir	0.82	1.81	0.93	1.68
Santa Elena	Guadalquivir	0.52	0.86	0.24	0.40
Fuente palmera	Guadalquivir	1.13	1.35	0.79	1.07
Cardeña	Guadalquivir	0.07	2.03	0.35	34.98
Salvacañete	Júcar	0.15	0.68	0.64	1.44
Pajaroncillo	Júcar	0.15	0.60	0.64	1.42
Villar del Humo	Júcar	0.12	0.60	2.96	1.48
Venta del Moro	Júcar	0.34	0.66	2.29	0.80
Villatoya	Júcar	0.14	0.71	1.00	1.46
Huélamo	Júcar	0.05	0.99	0.38	1.82
Cuenca	Júcar	1.08	0.78	1.25	1.70
Fresneda de Altarejos	Júcar	0.61	1.32	1.58	1.63
Quasiermas	Júcar	0.26	5.55	4.03	3.99
Jalance	Júcar	0.48	0.80	1.91	0.87
Cotes	Júcar	0.31	0.80	0.72	3.77
Alzira	Júcar	0.55	1.09	0.91	4.22
Sueca	Júcar	0.58	0.96	1.16	4.39
Requena	Júcar	2.35	2.08	1.95	2.64
Carlet	Júcar	4.13	4.91	2.49	2.55
Jorba	Llobregat	6.00	4.48	1.51	2.40
La Pobla de Claramunt	Llobregat	6.00	39.31	1.51	3.69
St. Sadurní d'Anoia	Llobregat	10.46	17.91	2.06	2.56
Olius	Llobregat	2.80	0.57	1.98	0.82
Clariana de Cardener	Llobregat	2.80	0.57	1.98	0.45
Súria	Llobregat	4.96	1.91	1.41	0.89
Manresa	Llobregat	4.96	5.58	1.41	1.90
La Pobla de Lillet	Llobregat	0.10	0.85	0.30	1.42
Colònia Rosal	Llobregat	1.08	0.83	1.06	1.04
Pont de Vilomara	Llobregat	2.05	5.17	1.24	1.41
Castellbell	Llobregat	3.19	6.62	1.20	1.15
Abrera	Llobregat	4.08	8.72	1.16	0.66
Martorell	Llobregat	4.95	8.37	1.28	0.81
St. Joan Despí	Llobregat	12.14	32.32	0.96	3.19

These results are easier interpreted if the values obtained from the model results and from the field data are plotted against each other. This is done in Figure 7.13 for the two substances groups. This has been repeated in Figure 7.14, but with simulated “pressure” from population and agriculture area respectively, instead of the simulated concentrations of other chemicals and pesticides.

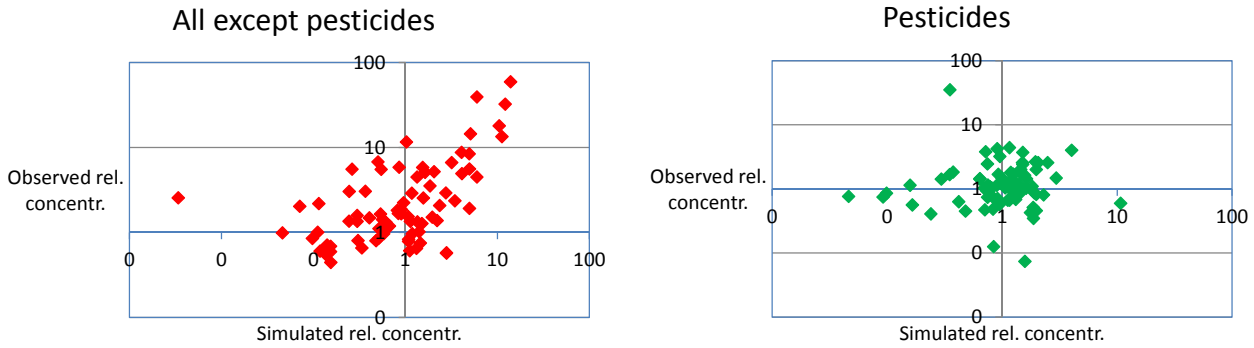


Figure 7.13: Observed relative concentrations vs. simulated relative concentrations; every dot represents a station and reflects relative concentrations of clusters of substances. Left: all chemicals except pesticides, right: pesticides.

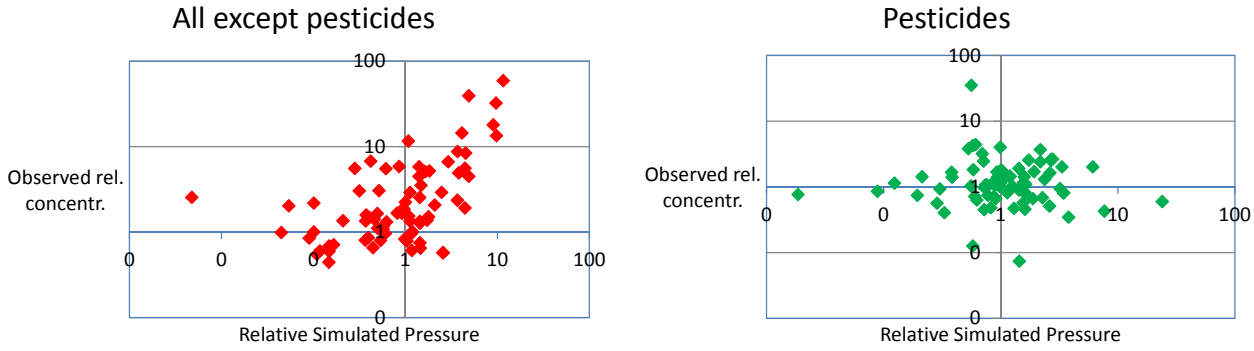


Figure 7.14: Observed relative concentrations vs. simulated relative pressures; every dot represents a station and reflects relative observed concentrations of clusters of substances and simulated pressure (see text for further details). Left: all chemicals except pesticides and population pressure, right: pesticides and agriculture pressure.

We note that the results presented above are all obtained with observation data for chemicals with at least 50% of stations unaffected by LoD/LoQ issues. The results are not significantly different if we use observation data for chemicals with at least 10% of stations unaffected by LoD/LoQ issues. For the sake of readability, these results have not been included.

7.3.3. Discussion

7.3.3.1 Average and range of simulated vs observed concentrations

For pesticides and pharmaceuticals, the simulated basin averages are within one order of magnitude accurate, except for two defined outliers: lorazepam and isoproturon. Similar assessments are being

prepared for other SOLUTIONS case studies, Rhine and Danube, as well. The results obtained there are similar to those presented here (van Gils et al., 2017), and the outliers are the same. Based on these results we tentatively conclude that the modeling methodology is sound. However, the reasons behind the outliers need to be investigated, which will potentially lead to refinements in the methodology, and/or the insight that the methodology cannot be used for certain substances.

For pesticides, we need to point out that concentrations are expected to be highly variable, both because of the episodic/seasonal and spatially variable application rates and because of the episodic nature of the local hydrology. In view of the low number of samples per station (1 or 2, Annex III), the current field data set is not necessarily representative. Our presented model results are all based on averages over the months of September and October of 2010 and 2011, and therefore can be expected to represent mean late summer concentrations. The low time resolution is a common feature of emerging contaminants monitoring data. Only the Rhine Case Study dataset has sufficient time resolution. There we obtain similar results as presented above, which support our tentative conclusion that the modeling methodology is sound. We are in the process of supplementing the SOLUTIONS Case Study data with additional data sets with high time resolution (van Gils et al., 2017).

For REACH registered chemicals, the results presented above are less satisfactory. Though the average bias is low, there is a high scatter in the results and modeling errors for individual chemicals are often more than one order of magnitude. Our current hypothesis is that this is caused by the fact that we do not use information about “use categories” of individual substances, since this information is not available for the wide range of substances we aim to model. Though emission estimates are available for so-called “Specific Environmental Release Categories” (spERC’s), we cannot categorize the individual chemicals (van de Meent et al., 2017). By adding specific use categories for a smaller group of chemicals included in the model validation, we aim to demonstrate that adding such information indeed brings the results for REACH registered chemicals up to the desired level.

The ranges of the simulated concentrations are higher in the model (3.6 orders on average) than in the field data (2.7 orders). The high range in the simulations is no doubt caused by the very low flows occurring during the simulations. The somewhat lower range in the field data could well be related to the low number of samples. In line with the expected high temporal variability, the range in the field data is highest for pesticides, and higher than the modeled range.

7.3.3.2. Spatial correlations

The analysis of spatial correlations shows that the spatial patterns in the model and in the field data correlate for the group of chemicals for which we assume the emissions to be distributed according to

population density: these are all chemicals except pesticides. This is a very encouraging result, which basically illustrates that the modeling methodology is satisfactory. The results for "simulated pressure" from population provide spatial patterns which are very comparable to the spatial patterns in the simulation results for the group of all chemicals. This is in line with expectations. Where for individual chemicals substance-specific behavior (partitioning, degradation) may play a role, the concentration patterns for the complete group of chemicals follows the emission patterns.
Also for pesticides, simulated concentration patterns for the complete group follow the emission patterns. For the pesticides however, we observe no correlation of simulated and observed spatial patterns. A similar conclusion can be drawn from the information presented by Lopez de Alda et al. (2016), where agriculture land use was found a poor predictor for the variability of concentrations. It is possible that this apparent contradiction is caused by a combination of high concentration variability and low amount of samples, which makes the pesticide sampling non-representative. Another option is that the use of pesticides is inhomogeneous in Spain, which means that our country-wide emission estimates are good on average, but wrong in certain places. It is also possible that indeed the spatial distribution of pesticides emissions does not follow agriculture land use. This could be caused by other well-known non-agricultural uses of pesticides such as the application of herbicides on roads, highways, railroads, industrial facilities, parks, golf courses etc., as well as the emissions from WWTPs. In addition, losses during transport and off-field equipment cleaning could play a role. It could also be that our assumed distribution of emissions over water, soil and air is inaccurate or does not consider substance-specific behaviour, possibly in combination with the water scarcity in the present study area. All these factors are under review for the final model train development stage (van Gils et al., 2017).

7.4. Concluding remarks

The work presented here allows the following concluding remarks:

- the model train is often able to simulate individual pesticides or pharmaceuticals within one order of magnitude this conclusion is supported by similar work in other case studies;
- for REACH chemicals the currently used methodology is insufficient to achieve that target: the hypothesis that adding "use category" information will resolve this still needs to be verified;
- the specific strength of the SCARCE dataset is its high spatial resolution (next to the large number of chemicals);
- the model train produces spatial concentration ranges for individual chemicals that resemble spatial concentration ranges in the field data, and differences in ranges are according to

expectations;

- by comparison of simulated and observed spatial patterns we found it is reasonable to assume that emissions of REACH registered chemicals and pharmaceuticals follow population patterns;
- we could not confirm that the emissions of pesticides follow agriculture land-use: this is reason to carefully review our modelling methodology with respect to pesticides.

7.5. References

Baveco, H. and A. Focks, 2017. SOLUTIONS ID M5.3 Progress report on population-level ERA via IBM. Alterra, The Netherlands.

De Zwart, D. and L. Posthuma. 2006. Complex mixture toxicity for single and multiple species: proposed methodologies. Environ. Toxicol. Chem. 24: 2665-2672.

Donnelly, C., Arheimer, B., Capell, R., Dahne, J., Stromqvist, J., July 2013. Regional overview of nutrient load in Europe e challenges when using a large-scale model approach, E-HYPE. Understanding fresh-water quality problems in a changing world. In: Proceedings of IAHS-IAPSO-IASPEI Assembly, Sweden, pp. 49e58 (IAHS Publ. 361, 2013).

Kortenkamp, A. and M. Faust, 2016. SOLUTIONS ID M5.1 A Common Assessment Framework for HRA and ERA for Fish and Drinking Water, Brunel University London, UK.

Kutsarova S., I. Dermen, C. Kuseva, P. Petkov, R. Kuehne and O. Mekenyan, 2017. SOLUTIONS D17.2 Predictions of transformation products, physico-chemical properties, fate and toxicity of candidate compounds from chemical screening, EDA and emission modelling. Laboratory of mathematical chemistry (LMC), Prof. Assen Zlatarov University, Bourgas, Bulgaria.

Lindim C., J. van Gils and I.T. Cousins, 2016. A large-scale model for simulating the fate & transport of organic contaminants in river basins. Chemosphere 144, 803-810.

Lindim, C., van Gils, J., Cousins, I.T., 2016a. Europe-wide estuarine export and surface water concentrations of PFOS and PFOA. Water Res. 103, 124-132.

Lindim C., J. van Gils, I.T. Cousins, R. Kühne, D. Georgieva, S. Kutsarova, O. Mekenyan, 2017. Model-predicted occurrence of multiple pharmaceuticals in Swedish surface waters and their flushing to the Baltic Sea. Environ. Pollut. 1-10

López de Alda, M., A. Ginebreda, D. Barceló, 2015. SOLUTIONS ID C3.1 Report on the prioritization of pollutants occurring in Iberian Mediterranean basins. Agencia Estatal Consejo Superior de Investigaciones Científicas, CSIC, Barcelona, Spain.

López de Alda, M., A. Ginebreda, D. Barceló, 2016. SOLUTIONS ID C3.2 Report on the relationships between chemical pollution and environmental stressors and ecosystem effects in Mediterranean river basins as found in the SCARCE project. Agencia Estatal Consejo Superior de Investigaciones Científicas, CSIC, Barcelona, Spain.

Navarro-Ortega A., Sabater S., Barcelo D. 2012ª. Understanding effects of global change on water quantity and quality in river basins – The SCARCE Project. Environ. Sci. Pollut. Res. 19(4), 915-917

Navarro-Ortega A., Acuna V., Batalla R.J., Blasco J., Conde C., Elorza F., Elosegi A., Frances F., La-Roca F., Munoz I., Petrovic M., Pico Y., Sabater S., Sanchez-Vila X., Schuhmacher M., Barcelo D. 2012b. Assessing and forecasting the impacts of global change on Meiterranean rivers. The SCARCE Consolider project on Iberian basins. Environ.l Sci. Pollut. Res. 19(4), 918-933

Van de Meent, D., J. van Gils, L. Posthuma, D. de Zwart, K. Kramer, M. Rahmberg and A. Focks, 2017. SOLUTIONS Internal Deliverable ID M2.1 Report on use information chemical substances. Mermayde, The Netherlands.

Van Gils, J., A. Focks, H. Baveco, L. Posthuma, I. Cousins, C. Lindim, S. Kutsarova, O. Mekenyan, A. Kortenkamp, J. Greskowiak, R. Kuehne, D. van de Meent, D. de Zwart, 2017. SOLUTIONS Deliverable D14.1 Modelling framework and model-based assessment for substance screening. Deltares, The Netherlands.

Annex I - List of measured compounds with their limits of detection and detection frequencies.

Compound	Compound class	Frequency of detection 2010	Frequency of detection 2011	Limit of detection (ng/L)	Reference
Acetaminophen	Pharmaceutical	36	41	0.04	SCARCE DB
Acridone	Pharmaceutical	56	71	0.03	SCARCE DB
Albendazol	Pharmaceutical	14	23	0.01	SCARCE DB
Alprazolam	Pharmaceutical	6	62	0.02	SCARCE DB
Amlodipine	Pharmaceutical	86	26	0.08	SCARCE DB
Atenolol	Pharmaceutical	64	69	0.02	SCARCE DB
Atorvastatin	Pharmaceutical	52	49	0.005	SCARCE DB
Azaperol	Pharmaceutical	0	0	0.32	SCARCE DB
Azaperone	Pharmaceutical	0	0	0.23	SCARCE DB
Azithromycin	Pharmaceutical	95	81	0.1	SCARCE DB
Bezafibrate	Pharmaceutical	34	50	0.02	SCARCE DB
Carazolol	Pharmaceutical	34	13	0.10	SCARCE DB
Carbamazepine	Pharmaceutical	44	64	0.01	SCARCE DB
Cefalexin	Pharmaceutical	4	3	0.2	SCARCE DB
Cimetidine	Pharmaceutical	49	8	0.1	SCARCE DB
Ciprofloxacin	Pharmaceutical	17	34	0.1	SCARCE DB
Citalopram	Pharmaceutical	45	66	0.02	SCARCE DB
Clarithromycin	Pharmaceutical	14	20	0.1	SCARCE DB
Clopidogrel	Pharmaceutical	66	71	0.01	SCARCE DB
Codeine	Pharmaceutical	70	69	0.02	SCARCE DB
Desloratidine	Pharmaceutical	24	16	0.04	SCARCE DB
Dexamethasone	Pharmaceutical	66	27	0.05	SCARCE DB
Diazepam	Pharmaceutical	18	61	0.05	SCARCE DB
Diclofenac	Pharmaceutical	41	63	0.6	SCARCE DB
Diltiazem	Pharmaceutical	74	48	0.02	SCARCE DB
Dimetridazole	Pharmaceutical	16	2	1.50	SCARCE DB
Enalapril	Pharmaceutical	13	2	0.47	SCARCE DB
Enalaprilat	Pharmaceutical	29	39	1.08	SCARCE DB
Erithromycin	Pharmaceutical	22	16	0.1	SCARCE DB
Famotidine	Pharmaceutical	0	4	0.1	SCARCE DB
Fluoxetine	Pharmaceutical	18	4	0.36	SCARCE DB
Fluvastatin	Pharmaceutical	17	15	0.03	SCARCE DB
Furosemide	Pharmaceutical	44	60	0.45	SCARCE DB
Gemfibrozil	Pharmaceutical	91	100	0.04	SCARCE DB
Glibenclamide	Pharmaceutical	3	1	0.60	SCARCE DB
Hidrochlorothiazide	Pharmaceutical	55	98	0.05	SCARCE DB
Hydrocodone	Pharmaceutical	3	18	0.6	SCARCE DB
Ibuprofen	Pharmaceutical	31	11	1.2	SCARCE DB
Indomethacine	Pharmaceutical	51	52	0.1	SCARCE DB
Iopromide	Pharmaceutical	61	32	0.18	SCARCE DB
Irbesartan	Pharmaceutical	64	81	0.02	SCARCE DB
Ketoprofen	Pharmaceutical	61	100	0.8	SCARCE DB
Levamisol	Pharmaceutical	70	59	0.01	SCARCE DB

Loratidine	Pharmaceutical	40	21	0.1	SCARCE DB
Lorazepam	Pharmaceutical	52	50	0.27	SCARCE DB
Losartan	Pharmaceutical	42	42	0.10	SCARCE DB
Meloxicam	Pharmaceutical	7	32	0.007	SCARCE DB
Metformin	Pharmaceutical	0	0	0.5	SCARCE DB
Metoprolol	Pharmaceutical	9	11	0.1	SCARCE DB
Metronidazole	Pharmaceutical	5	19	0.6	SCARCE DB
Metronidazole-Oh	Pharmaceutical	4	12	0.4	SCARCE DB
Nadolol	Pharmaceutical	12	3	0.06	SCARCE DB
Naproxen	Pharmaceutical	67	77	0.2	SCARCE DB
Norfluoxetine	Pharmaceutical	6	1	0.50	SCARCE DB
Ofloxacin	Pharmaceutical	14	6	0.04	SCARCE DB
Olanzapine	Pharmaceutical	4	5	0.04	SCARCE DB
Oxycodone	Pharmaceutical	40	43	0.1	SCARCE DB
Paroxetine	Pharmaceutical	58	35	0.16	SCARCE DB
Phenazone	Pharmaceutical	27	48	0.04	SCARCE DB
Piroxicam	Pharmaceutical	0	7	0.02	SCARCE DB
Pravastatin	Pharmaceutical	29	27	0.1	SCARCE DB
Propanolol	Pharmaceutical	18	25	0.04	SCARCE DB
Propyphenazone	Pharmaceutical	26	10	0.04	SCARCE DB
Ranitidine	Pharmaceutical	10	9	1.1	SCARCE DB
Ronidazole	Pharmaceutical	0	5	0.83	SCARCE DB
Salbutamol	Pharmaceutical	56	35	0.01	SCARCE DB
Sertraline	Pharmaceutical	3	5	0.63	SCARCE DB
Sotalol	Pharmaceutical	9	7	0.2	SCARCE DB
Sulfamethoxazole	Pharmaceutical	32	27	0.1	SCARCE DB
Tamsulosin	Pharmaceutical	29	5	0.02	SCARCE DB
Tenoxicam	Pharmaceutical	0	9	0.01	SCARCE DB
Tetracycline	Pharmaceutical	3	0	3.5	SCARCE DB
Torasemide	Pharmaceutical	34	48	0.02	SCARCE DB
Trazodone	Pharmaceutical	34	58	0.03	SCARCE DB
Trimethoprim	Pharmaceutical	27	91	0.1	SCARCE DB
Valsartan	Pharmaceutical	92	91	0.05	SCARCE DB
Venlafaxine	Pharmaceutical	49	79	0.02	SCARCE DB
Warfarin	Pharmaceutical	8	6	0.04	SCARCE DB
Xylazine	Pharmaceutical	4	9	0.03	SCARCE DB
Estradiol 17-glucuronide	Hormone	0	4	0.46	[1]
Estriol	Hormone	3	4	0.17	[1]
Estriol 16-glucuronide	Hormone	3	4	0.059	[1]
Estriol 3-sulfate	Hormone	3	17	0.030	[1]
Estrone	Hormone	64	56	0.050	[1]
Estradiol	Hormone	86	8	0.037	[1]
Estrone 3-glucuronide	Hormone	3	5	0.056	[1]
Estrone 3-sulfate	Hormone	3	17	0.0038	[1]
Ethinyl estradiol	Hormone	0	1	0.14	[1]
Diethylstilbestrol	Hormone	1	1	0.043	[1]

Caffeine	Stimulans	84	100	0.021	[1]
Cocaine	Ilicit drug	63	96	0.02	SCARCE DB
Benzoylecgonine	Ilicit drug	81	94	0.02	SCARCE DB
LSD	Ilicit drug	0	0	0.32	SCARCE DB
Cannabidiol	Ilicit drug	0	0	2.27	SCARCE DB
Ephedrine	Ilicit drug	76	83	0.16	SCARCE DB
Methamphetamine	Ilicit drug	4	47	0.045	SCARCE DB
Lorazepam	Ilicit drug	12	34	1.01	SCARCE DB
Morphine	Ilicit drug	13	9	0.3	SCARCE DB
3-Hydroxycarbofuran	Pesticide	4	0	0.2	scarce db
Acethochlor	Pesticide	0	0	2	[2]
Alachlor	Pesticide	0	0	2	[2]
Atrazine	Pesticide	21	4	1.3	[2]
Azinphos ethyl	Pesticide	9	1	0.5	[2]
Azinphos methyl	Pesticide	4	1	0.5	[2]
Burpofezin	Pesticide	80	0	0.5	[2]
Carbendazim	Pesticide	0	41	0.01	SCARCE DB
Carbofuran	Pesticide	21	3	0.2	[2]
Chlorfenvinphos	Pesticide	66	18	0.2	[2]
Chlorpyriphos	Pesticide	99	49	0.2	[2]
Deisopropylatrazine	Pesticide	28	1	2	[2]
Desethylatrazine	Pesticide	21	4	2	[2]
Diazinon	Pesticide	95	43	0.04	[2]
Diclofenthion	Pesticide	45	0	0.5	[2]
Dimetoate	Pesticide	28	0	1	[2]
Diuron	Pesticide	29	17	1	[2]
Ethion	Pesticide	8	22	0.5	[2]
Fenitrothion	Pesticide	1	1	2	[2]
Fenoxon	Pesticide	1	0	0.2	[2]
Fenthion	Pesticide	1	0	0.2	[2]
Fenthion Sulfone	Pesticide	3	1	0.2	[2]
Fenthion sulfoxide	Pesticide	1	0	0.2	[2]
Hexythiazox	Pesticide	78	11	0.2	[2]
Imazalil	Pesticide	62	33	0.3	[2]
Imidacloprid	Pesticide	53	30	0.04	[2]
Isoproturon	Pesticide	16	8	0.3	[2]
Malathion	Pesticide	14	1	0.3	[2]
Methiocarb	Pesticide	4	8	0.3	[2]
Metoalachlor	Pesticide	5	12	0.3	[2]
Molinate	Pesticide	1	0	0.5	[2]
Ometoate	Pesticide	4	1	0.3	[2]
Parathion-ethyl	Pesticide	12	0	2	[2]
Parathion-methyl	Pesticide	0	0	2	[2]
Prochloraz	Pesticide	42	5	0.8	[2]
Propanil	Pesticide	0	0	0.3	[2]

Propazine	Pesticide	8	0	0.3	[2]
Pyriproxyphen	Pesticide	62	1	0.5	[2]
Simazine	Pesticide	4	8	2	[2]
Tebuconazole	Pesticide	/	13	0.13	SCARCE DB
Terbumeton	Pesticide	/	4	0.01	SCARCE DB
Terbumeton-Desethyl	Pesticide	/	14	0.13	SCARCE DB
Terbutilazine	Pesticide	/	22	0.4	SCARCE DB
Terbutilazine-2 Hidroxy	Pesticide	/	29	0.01	SCARCE DB
Terbutryn	Pesticide	8	20	0.5	[2]
Tebutylazine deethyl	Pesticide	/	29	0.4	SCARCE DB
Thiabendazole	Pesticide	/	14	0.02	SCARCE DB
Tolclophos-methyl	Pesticide	14	1	0.5	[2]
1H-Benzotriazole	Industial organic	73	90	0.072	[1]
Tolytriazol	Industrial organic	99	84	0.013	[1]
Nonylphenol monoethoxylate	Industrial organic	0	0	62	[1]
Octylphenol	Industrial organic	96	32	0.14	[1]
Octylphenol diethoxylate	Industrial organic	96	73	0.011	[1]
Octylphenol monocarboxylate	Industrial organic	0	1	0.065	[1]
Octylphenol monoethoxylate	Industrial organic	0	0	17	[1]
Tris(2-chloroethyl) phosphate	Industrial organic	100	97	0.034	[1]
Tris(butoxyethyl) phosphate	Industrial organic	100	88	0.0024	[1]
Tris(chloroisopropyl) phosphate	Industrial organic	100	100	0.0025	[1]
Bisphenol A (BPA)	Industrial organic	68	88	0.11	[1]
Nonylphenol (NP)	Industrial organic	91	42	0.013	[1]
Nonylphenol diethoxylate	Industrial organic	94	96	0.013	[1]
Nonylphenol monocarboxylate	Industrial organic	94	70	0.034	[1]
L-PFOS	Perflourinated compound	26	77	0.004	SCARCE DB
PFBA	Perflourinated compound	77	52	0.04	SCARCE DB
PFOA	Perflourinated compound	52	43	0.04	SCARCE DB
PFNA	Perflourinated compound	14	18	0.4	SCARCE DB
PFDA	Perflourinated compound	13	40	0.04	SCARCE DB
PFUdA	Perflourinated compound	3	9	0.04	SCARCE DB
PFDoA	Perflourinated compound	0	13	0.8	SCARCE DB
L-PFBS	Perflourinated compound	4	52	0.02	SCARCE DB
L-PFDS	Perflourinated compound	0	14	0.004	SCARCE DB
i,p-PFNA	Perflourinated compound	14	19	0.4	SCARCE DB
I,pPFNS	Perflourinated compound	0	13	0.04	SCARCE DB
L-PFHpS	Perflourinated compound	0	3	0.04	SCARCE DB
L-PFHxS	Perflourinated compound	17	27	0.04	SCARCE DB
PFHpA	Perflourinated compound	25	5	0.4	SCARCE DB
PFHxA	Perflourinated compound	13	5	0.4	SCARCE DB
PFHxDA	Perflourinated compound	1	5	0.04	SCARCE DB
PFODA	Perflourinated compound	0	13	0.8	SCARCE DB
PFOSA	Perflourinated compound	0	0	0.2	SCARCE DB
PFPeA	Perflourinated compound	34	48	0.04	SCARCE DB
PFTeDA	Perflourinated compound	4	10	0.02	SCARCE DB

PFTrDA	Perflourinated compound	3	10	0.02	SCARCE DB
4-Methylbenzylidene camphor	Personal care product	18	48	3.5	[3]
Benzophenone-3	Personal care product	14	43	0.7	[3]
Ethylhexyl methoxycinnamate	Personal care product	9	14	0.72	SCARCE DB
Octocrylene	Personal care product	9	0	3	SCARCE DB
2,2'-Dihydroxy-4-methoxybenzophenone	Personal care product	0	0	1	[3]
4,4'-Dihidroxybenzophenone	Personal care product	4	1	1.8	[3]
4-Hydroxybenzophenone	Personal care product	4	5	1.1	[3]
Benzophenone-1	Personal care product	0	22	1	[3]
Benzophenone-2	Personal care product	16	0	1.2	[3]
Ethyl 4-aminobenzoate	Personal care product	0	0	1.5	[3]
Ethylhexyl dimethyl PABA	Personal care product	0	14	0.1	SCARCE DB
Ethylparaben	Personal care product	74	53	0.27	[1]
Methylparaben	Personal care product	90	75	0.20	[1]
Benzylparaben	Personal care product	30	40	0.031	[1]
Propylparaben	Personal care product	99	94	0.021	[1]
Triclorocaraban	Personal care product	0	7	0.036	[1]
Triclosan	Personal care product	23	8	0.17	[1]

SCARCE DB-Scarce Consolider project database-unpublished data

[1] Gorga M., Petrovic M., Barceló D. 2013. Multi-residue analytical method for the determination of endocrine disruptors and related compounds in river and waste water using dual column liquid chromatography switching system coupled to mass spectrometry. J. Chromatogr. A 1295, 57-66.

[2] Masiá A., Ibáñez M., Blasco C., Sancho J.V., Picó Y., Hernández F. 2013.Combined use of liquid chromatography triple quadrupole mass spectrometry and liquid chromatography quadrupole time-of-flight mass spectrometry in systematic screening of pesticides and other contaminants in water samples. Anal. Chim. Acta 761, 117-127.

[3] Gago-Ferrero P., Mastroianni N., Díaz-Cruz M.S., Barceló D. 2013. Fully automated determination of nine ultraviolet filters and transformation products in natural waters and wastewaters by on-line solid phase extraction–liquid chromatography–tandem mass spectrometry. J. Chromatogr. A 1294, 106-116.

Annex II - Toxicological data of studied compounds for algae, Daphnia sp and fish.

Compound	Compound class	EC50 algae (µg/l)	EC50 Daphnia sp.(µg/l)	EC50 fish(µg/l)	Ref.
Acetaminophen	Pharmaceutical	134000	9200	378000	[1]
Acridone	Pharmaceutical	6738	3419	7817	E
Albendazol	Pharmaceutical	174	1225	2282	E
Alprazolam	Pharmaceutical	1064	2845	2499	E
Amlodipine	Pharmaceutical	6883	8479	4754	E
Amoxicilin	Pharmaceutical	/	/	/	/
Atenolol	Pharmaceutical	190000	205000	1096000	ECOTOX
Atorvastatin	Pharmaceutical	/	/	/	/
Azaperol	Pharmaceutical	/	/	/	/
Azaperone	Pharmaceutical	833	1340	9743	E
Azithromycin	Pharmaceutical	1874	3070	1970	E
Bezafibrate	Pharmaceutical	18000	30000	6000	ECOTOX
Carazolol	Pharmaceutical	2660	60000	2500	[2]
Carbamazepine	Pharmaceutical	85000	76300	35400	ECOTOX
Cefalexin	Pharmaceutical	/	/	/	/
Cimetidine	Pharmaceutical	787	379000	80402	E
Ciprofloxacin	Pharmaceutical	2970	60000	100000	
Citalopram	Pharmaceutical	360	652	4467	E
Clarithromycin	Pharmaceutical	46	3307	17364	
Clopidogrel	Pharmaceutical	/	/	/	/
Codeine	Pharmaceutical	1800	23000	16000	[2]
Desloratidine	Pharmaceutical	26981	49307	75054	E
Dexamethasone	Pharmaceutical	983	21438	23910	E
Diazepam	Pharmaceutical	1249	3129	19307	E
Diclofenac	Pharmaceutical	14500	22000	532000	[1]
Diltiazem	Pharmaceutical	/	/	/	/
Dimetridazole	Pharmaceutical	350	4272	25695	E
Enalapril	Pharmaceutical	18695	46266	276429	E
Enalaprilat	Pharmaceutical	2523000	3690000	73000000	[2]
Erithromycin	Pharmaceutical	20	30500	61500	VSDB
Famotidine	Pharmaceutical	478143	314690	3594432	E
Fluoxetine	Pharmaceutical	800	510	1700	E
Fluvastatin	Pharmaceutical	1350	5268	287	E
Furosemide	Pharmaceutical	19797	560033	521136	E
Gemfibrozil	Pharmaceutical	4000	4900	900	ECOTOX
Glibenclamide	Pharmaceutical	/	/	/	/
Hidrochlorothiazide	Pharmaceutical	/	/	/	/
Hydrocodone	Pharmaceutical	4239	5449	44844	E
Ibuprofen	Pharmaceutical	4000	34000	5000	ECOTOX
Indomethacine	Pharmaceutical	18000	26000	3900	[2]
Iopromide	Pharmaceutical	370000000	7660000000	8650000000	[2]
Irbesartan	Pharmaceutical	/	/	/	/
Ketoprofen	Pharmaceutical	164000	248000	32000	[2]

Levamisol	Pharmaceutical	943	1394	175000	E
Loratidine	Pharmaceutical	62	100	115	E
Lorazepam	Pharmaceutical	1683	44712	49067	E
Losartan	Pharmaceutical	180	2100	2151	E
Meloxicam	Pharmaceutical	184	3994	1392	E
Metformin	Pharmaceutical	/	/	/	/
Metoprolol	Pharmaceutical	8305	9383	81557	E
Metronidazole	Pharmaceutical	40400	1000000	1060000	VSDB
Metronidazole-Oh	Pharmaceutical	/	/	/	/
Nadolol	Pharmaceutical	22538	22609	208809	E
Naproxen	Pharmaceutical	137944	121543	193337	E
Norfluoxetine	Pharmaceutical	/	/	/	/
Ofloxacin	Pharmaceutical	2444544	31750	19352000	E
Olanzapine	Pharmaceutical	52515	46786	458553	E
Oxycodone	Pharmaceutical	/	/	/	/
Paroxetine	Pharmaceutical	/	/	/	/
Phenazone	Pharmaceutical	1100	6700	3000	[2]
Piroxicam	Pharmaceutical	289	768	4220	E
Pravastatin	Pharmaceutical	85494	8588	1800	E
Propanolol	Pharmaceutical	/	/	/	/
Propyphenazone	Pharmaceutical	1000	3500	9800	[2]
Ranitidine	Pharmaceutical	66000	63000	1076000	[2]
Ronidazole	Pharmaceutical	1080	19445	242023	E
Salbutamol	Pharmaceutical	/	/	/	/
Sertraline	Pharmaceutical	43	120	408	ECOTOX
Sotalol	Pharmaceutical	/	/	/	/
Sulfamethoxazole	Pharmaceutical	1900	25200	56200	[1]
Tamsulosin	Pharmaceutical	/	/	/	/
Tenoxicam	Pharmaceutical	/	/	/	/
Tetracycline	Pharmaceutical	6000	6000	220000	[1]
Torasemide	Pharmaceutical	/	/	/	/
Trazodone	Pharmaceutical	396	1567	1313	E
Trimethoprim	Pharmaceutical	16000	121000	795000	ECOTOX
Valsartan	Pharmaceutical	3865	44337	88094	E
Venlafaxine	Pharmaceutical	635	1062	7678	E
Warfarin	Pharmaceutical	/	/	/	/
Xylazine	Pharmaceutical	/	/	/	/
Estradiol 17-glucuronide	Hormone	/	/	/	/
Estriol	Hormone	22250	5235	12110	E
Estriol 16-glucuronide	Hormone	/	/	/	/
Estriol 3-sulfate	Hormone	/	/	/	/
Estrone	Hormone	8740	2184	3834	E
Estradiol	Hormone	4299	1129	1578	E
Estrone 3-glucuronide	Hormone	/	/	/	/
Estrone 3-sulfate	Hormone	/	/	/	/
Ethinyl estradiol	Hormone	2000	2500	1610	

Diethylstilbestrol	Hormone	330	180	97	[2]
Caffeine	Stimulans	760	46000	46000	E
Cocaine	Ilicit drug	5482	5482	45092	[3]
Benzoylecgonine	Ilicit drug	12041000	6805000	89593000	E
LSD	Ilicit drug	/	/	/	/
Cannabidiol	Ilicit drug	/	/	/	/
Ephedrine	Ilicit drug	26591	23805	232000	E
Methamphetamine	Ilicit drug	1967	2509	20511	E
Lorazepam	Ilicit drug	1683	44712	49008	E
Morphine	Ilicit drug	43555	32000	257000	E
3-Hydroxycarbofuran	Pesticide	16932	209	15680	E
Acethochlor	Pesticide	0,27	8600	360	PPDB
Alachlor	Pesticide	6	7700	6600	ECOTOX
Atrazine	Pesticide	9,5	35000	4500	ECOTOX
Azinphos ethyl	Pesticide	372	0,2	80	
Azinphos methyl	Pesticide	7150	1,1	20000	E
Burpofezin	Pesticide	330	420	2100	PPDB
CARBENDAZIM	Pesticide	/	/	/	/
Carbofuran	Pesticide	6500	9,4	180	PPDB
Chlorfenvinphos	Pesticide	1360	0,25	1100	PPDB
Chlorpyriphos	Pesticide	480	0,1	1,3	PPDB
Deisopropylatrazine	Pesticide	198	1348	38130	E
Desethylatrazine	Pesticide	2803	1259	68923	E
Diazinon	Pesticide	6400	1	3300	PPDB
Diclofenthion	Pesticide	420	1,1	1,25	PPDB
Dimetoate	Pesticide	30200	560	90400	PPDB
Diuron	Pesticide	2,4	270	6700	ECOTOX
Ethion	Pesticide	326	0,056	500	PPDB
Fenitrothion	Pesticide	1300	8,6	1300	PPDB
Fenoxon	Pesticide	1790	5,7	800	PPDB
Fenthion	Pesticide	/	/	/	/
Fenthion Sulfone	Pesticide	/	/	/	/
Fenthion sulfoxide	Pesticide	/	/	/	/
Hexythiazox	Pesticide	400	470	3200	PPDB
Imazalil	Pesticide	1480	3100	870	PPDB
Imidacloprid	Pesticide	10000	85000	211000	PPDB
Isoproturon	Pesticide	13	580	18000	PPDB
Malathion	Pesticide	13000	0,7	18	PPDB
Methiocarb	Pesticide	2200	8	650	PPDB
Metoalachlor	Pesticide	57100	23500	3900	PPDB
Molinate	Pesticide	500	14900	16000	PPDB
Ometoate	Pesticide	167500	22	9100	PPDB
Parathion-ethyl	Pesticide	500	2,5	1500	PPDB
Parathion-methyl	Pesticide	3000	7,3	2700	PPDB
Prochloraz	Pesticide	5,5	4300	1500	PPDB

Propanil	Pesticide	110	2390	5400	PPDB
Propazine	Pesticide	180	17700	17500	PPDB
Pyriproxyphen	Pesticide	150	400	270	PPDB
Simazine	Pesticide	40	1100	90000	PPDB
Tebuconazole	Pesticide	/	/	/	/
Terbumeton	Pesticide	/	/	/	/
Terbumeton-Desethyl	Pesticide	/	/	/	/
Terbutilazine	Pesticide	/	/	/	/
Terbutilazine-2 Hidroxy	Pesticide	/	/	/	/
Terbutryn	Pesticide	2,4	2060	1100	PPDB
TERBUTYLAZINE DEETHYL	Pesticide	/	/	/	/
THIABENDAZOLE	Pesticide	9000	810	550	PPDB
Tolclophos-methyl	Pesticide	780	/	690	PPDB
1H-Benzotriazole	Industial organic	5904	66766	28321	E
Tolytriazol	Industrial organic	3851	36053	16386	E
Nonylphenol monoethoxylate	Industrial organic	12200	12200	40000	PPDB
Octylphenol	Industrial organic	210	11	7200	PPDB
Octylphenol diethoxylate	Industrial organic	/	/	/	/
Octylphenol monocarboxylate	Industrial organic	/	/	/	/
Octylphenol monoethoxylate	Industrial organic	/	/	/	/
Tris(2-chloroethyl) phosphate	Industrial organic	38000	135300	90000	E
Tris(butoxyethyl) phosphate	Industrial organic	/	/	/	/
Tris(chloroisopropyl) phosphate	Industrial organic	47000	21315	31000	E
Bisphenol A (BPA)	Industrial organic	2700	7750	1284	[2]
Nonylphenol (NP)	Industrial organic	197	140	170	ECOTOX
Nonylphenol diethoxylate	Industrial organic	555	211	274	E
Nonylphenol monocarboxylate	Industrial organic	2250	707	876	E
L-PFOS	Perflourinated compound	23640	37360	3640	[4]
PFBA	Perflourinated compound	262150	177620	273920	[4]
PFOA	Perflourinated compound	748098	207000	260820	[4]
PFNA	Perflourinated compound	481632	92800	120640	[4]
PFDA	Perflourinated compound	437414	77100	35980	[4]
PFUdA	Perflourinated compound	318660	56400	33840	[4]
PFDoA	Perflourinated compound	241916	73680	36840	[4]
L-PFBS	Perflourinated compound	645000	1938000	502000	[4]
L-PFDS	Perflourinated compound	/	4800	/	/
i,p-PFNA	Perflourinated compound	/	/	/	/
I,pPFNS	Perflourinated compound	/	/	/	/
L-PFHpS	Perflourinated compound	/	/	/	/
L-PFHxS	Perflourinated compound	/	/	/	/
PFHpA	Perflourinated compound	/	/	/	/
PFHxA	Perflourinated compound	/	/	/	/
PFHxDA	Perflourinated compound	/	/	/	/
PFODA	Perflourinated compound	/	/	/	/
PFOSA	Perflourinated compound	/	/	/	/
PFPeA	Perflourinated compound	/	/	/	/

PFTeDA	Perflourinated compound	/	/	/	/
PFTrDA	Perflourinated compound	/	/	/	/
4-Methylbenzylidene camphor	Personal care product	/	9900	560	[5]
Benzophenone-3	Personal care product	/	1900	290	[5]
Ethylhexyl methoxycinnamate	Personal care product	/	9870	620	[5]
Octocrylene	Personal care product	/	/	/	/
2,2'-Dihydroxy-4-methoxybenzophenone	Personal care product	/	/	/	/
4,4'-Dihidroxybenzophenone	Personal care product	/	/	/	/
4-Hydroxybenzophenone	Personal care product	/	/	/	/
Benzophenone-1	Personal care product	/	/	/	/
Benzophenone-2	Personal care product	/	/	/	/
Ethyl 4-aminobenzoate	Personal care product	/	/	/	/
Ethylhexyl dimethyl PABA	Personal care product	/	/	/	/
Ethylparaben	Personal care product	20172	18700	34300	[6]
Methylparaben	Personal care product	18092	4600	20432	[6]
Benzylparaben	Personal care product	1735	4000	2300	[6]
Propylparaben	Personal care product	4407	2627	5643	[6]
Triclorocaraban	Personal care product	20	10	120	[6]
Triclosan	Personal care product	0,53	390	270	[6]

E-ECOSAR
ECOTOX - US EPA ECOTOX database http://cfpub.epa.gov/ecotox/
PPDB: Pesticide Properties DataBase, http://sitem.herts.ac.uk/aeru/footprint/index2.htm
VSDB: Veterinary Substances DataBase, http://sitem.herts.ac.uk/aeru/vsdb/index.htm

[1] Grung M., Källqvist T., Sakshaug S., Skurtveit, Thomas K.V. 2008. Environmental assessment of Norwegian priority pharmaceuticals based on the S. EMEA guideline, Ecotoxicology and Environmental Safety, 71, 328-340.
[2] Sanderson H., Johnson D.J., Wilson C.J., Brain R.A., Solomon K.R. 2003. Probabilistic hazard assessment of environmentally occurring pharmaceuticals toxicity to fish, daphnids and algae by ECOSAR screening, Toxicology Letters, 144, 383-395.
[3] MacGillivrauly A. R. 2012. Contaminants of emerging concern in the Delaware River. Pilot monitoring survey 2007-2009. Delaware River Basin Commision.
[4] Durjava M.K., Kolar B., Peijnenburg W. 2012. Case studies on the Development and Application of in-Silico Techniques for Environmental hazard and Risk assessment. National Institute of Public Health and the Environment (RIVM), Laboratory for Ecological Risk Assessment. February 29, 2012.
[5] Fent K., Kunz P.Y., Zenker A., Rapp M. 2010. A tentative environmental risk assessment of the UV-filters 3-(4-methylbenzylidene-camphor), 2-ethyl-hexyl-4-trimethoxycinnamate, benzophenone-3, benzophenone-4 and 3-benzylidene camphor. Marine Environmental Research 69, S4-S6.
[6] Brausch J.M., Rand G.M. 2011. A review of personal care products in the aquatic environment: Environmental concentrations and toxicity. Chemosphere 82, 1518-1532.

Annex III: Statistics of measured concentrations

Column headers:

- "Groups" = Substances groups:
  - ED = endocrine disruptors
  - DRU = drugs of abuse
  - PES = pesticides
  - PHA =pharmaceuticals
  - UV = UV filters
  - PF = perfluorinated compounds
- "stations": nr. of stations with analyses results
- "surveys": average nr. of analyses per station
- "Cmean"" average concentration in µg/L (average of averages per station)
- "Qmean": percentage of analyses unaffected by LoD/LoQ
- "sep-Oct": percentage of analyses with a date in September and October (2010 or 2011)
- "Undef.": percentage of analyses with an undefined date in 2011

CAS	Name	Group	Stations	Surveys	Cmean	Qmean	Sep-Oct	Undef.
58-08-2	Caffeine	ED	77	2	0.231519	100%	86%	0%
78-51-3	Tris(butoxyethyl) phosphate	ED	77	2	0.075495	100%	86%	0%
25812-30-0	GEMFIBROZIL	PHA	77	2	0.029075	96%	88%	0%
115-96-8	Tris(2-chloroethyl) phosphate	ED	77	4	0.104475	94%	86%	0%
94-13-3	Propylparaben	ED	77	2	0.004626	93%	86%	0%
137862-53-4	VALSARTAN	PHA	77	2	0.023404	91%	88%	0%
29385-43-1	Tolytriazol	ED	77	2	0.223954	91%	86%	0%
22071-15-4	KETOPROFEN	PHA	77	2	0.017104	88%	88%	0%
83905-01-5	AZITHROMYCIN	PHA	77	2	0.004519	88%	87%	0%
80-05-7	Bisphenol A	ED	77	2	0.03991	83%	86%	0%
95-14-7	1H-Benzotriazole	ED	77	2	0.256093	82%	86%	0%
99-76-3	Methylparaben	ED	77	2	0.008437	79%	86%	0%
58-93-5	HYDROCHLOROTHIAZIDE	PHA	77	2	0.048029	77%	88%	0%
76-57-3	CODEINE	PHA	77	2	0.002342	77%	87%	0%
578-95-0	ACRIDONE	PHA	77	2	0.002624	77%	87%	0%
138402-11-6	IRBESARTAN	PHA	77	2	0.009274	73%	88%	0%
2921-88-2	Chlorpyriphos	PES	77	2	0.004728	73%	50%	50%
22204-53-1	NAPROXEN	PHA	77	2	0.01465	73%	88%	0%
333-41-5	Diazinon	PES	77	2	0.006976	68%	50%	50%
27193-28-8	Octylphenol	ED	77	2	0.005055	68%	86%	0%
738-70-5	TRIMETHOPRIM	PHA	77	2	0.003591	65%	88%	0%
93413-69-5	VENLAFAXINE	PHA	77	2	0.004307	65%	88%	0%
120-47-8	Ethylparaben	ED	77	2	0.003881	63%	86%	0%
53-16-7	Estrone	ED	77	2	0.001167	62%	86%	0%
113665-84-2	CLOPIDOGREL	PHA	77	2	0.000674	60%	88%	0%
53-86-1	INDOMETHACIN	PHA	77	2	0.004725	59%	88%	0%

CAS	Name	Group	Stations	Surveys	Cmean	Qmean	Sep-Oct	Undef.
14769-73-4	LEVAMISOL	PHA	77	2	0.002202	57%	88%	0%
88150-42-9	AMLODIPINE	PHA	77	2	0.001201	55%	87%	0%
298-46-4	CARBAMAZEPINE	PHA	77	2	0.002904	53%	87%	0%
15307-86-5	DICLOFENAC	PHA	77	2	0.012987	53%	88%	0%
54-31-9	FUROSEMIDE	PHA	77	2	0.013192	52%	88%	0%
846-49-1	LORAZEPAM	PHA	77	2	0.009024	51%	88%	0%
42399-41-7	DILTIAZEM	PHA	77	2	0.001786	48%	88%	0%
50-02-2	DEXAMETHASONE	PHA	77	2	0.000899	47%	88%	0%
35554-44-0	Imazalil	PES	77	2	0.030411	47%	50%	50%
5915-41-3	Terbutylazine	PES	77	1	0.01246	45%	0%	100%
103-90-2	ACETAMINOPHEN	PHA	77	2	0.009635	44%	87%	0%
148-79-8	Thiabendazole	PES	77	3	0.006351	43%	58%	33%
59729-33-8	CITALOPRAM	PHA	77	2	0.001463	42%	88%	0%
10605-21-7	Carbendazim	PES	77	1	0.01429	42%	0%	100%
114798-26-4	LOSARTAN	PHA	77	2	0.00812	42%	88%	0%
50-28-2	Estradiol	ED	77	2	0.000755	42%	86%	0%
29122-68-7	ATENOLOL	PHA	77	2	0.00934	41%	87%	0%
138261-41-3	Imidacloprid	PES	77	2	0.003273	41%	50%	50%
439-14-5	DIAZEPAM	PHA	77	2	0.00085	40%	88%	0%
69327-76-0	Buprofezin	PES	77	2	0.002792	40%	50%	50%
41859-67-0	BEZAFIBRATE	PHA	77	2	0.001309	40%	87%	0%
470-90-6	Chlorfenvinphos	PES	77	2	0.00976	39%	50%	50%
78587-05-0	Hexythiazox	PES	77	2	0.00338	39%	50%	50%
18559-94-9	SALBUTAMOL	PHA	77	2	0.00057	38%	88%	0%
94-18-8	Benzylparaben	ED	77	2	0.000916	38%	86%	0%
66753-07-9	Terbutylazine-2-hydroxy	PES	77	1	0.004087	38%	0%	100%
56211-40-6	TORASEMIDE	PHA	77	2	0.000743	36%	88%	0%
60-80-0	PHENAZONE	PHA	77	2	0.000918	36%	88%	0%
36861-47-9	4-Methylbenzylidene camphor (4MBC)	UV	77	2	0.007986	35%	86%	0%
28981-97-7	ALPRAZOLAM	PHA	77	2	0.000267	34%	87%	0%
19794-93-5	TRAZODONE	PHA	77	2	0.002447	33%	88%	0%
95737-68-1	Pyriproxyphen	PES	77	2	0.012106	31%	50%	50%
36322-90-4	PIROXICAM	PHA	77	2	0.000322	31%	88%	0%
73334-07-3	IOPROMID	PHA	77	2	0.016663	31%	88%	0%
30125-63-4	Terbutylazine-desethyl	PES	77	1	0.004224	30%	0%	100%
76-42-6	OXYCODONE	PHA	77	2	0.000667	30%	88%	0%
134523-00-5	ATORVASTATIN	PHA	77	2	0.000169	29%	88%	0%
61869-08-7	PAROXETINE	PHA	77	2	0.000492	29%	88%	0%
131-57-7	Benzophenone-3 (BP3)	UV	77	2	0.003768	27%	86%	0%
81093-37-0	PRAVASTATIN	PHA	77	2	0.000697	25%	88%	0%

CAS	Name	Group	Stations	Surveys	Cmean	Qmean	Sep-Oct	Undef.
85721-33-1	CIPROFLOXACIN	PHA	77	2	0.000722	25%	87%	0%
525-66-6	PROPRANOLOL	PHA	77	2	0.000291	23%	88%	0%
67747-09-5	Prochloraz	PES	77	2	0.00976	22%	50%	50%
15687-27-1	IBUPROFEN	PHA	77	2	0.018409	21%	88%	0%
723-46-6	SULFAMETHOXAZOLE	PHA	77	2	0.000945	21%	88%	0%
97-17-6	Diclofenthion	PES	77	2	0.006131	21%	50%	50%
479-92-5	PROPYPHENAZONE	PHA	77	2	0.001555	21%	88%	0%
481-97-0	Estrone 3-sulfate	ED	77	2	0.00045	21%	86%	0%
100643-71-8	DESLORATIDINE	PHA	77	2	0.000204	20%	88%	0%
54965-21-8	ALBENDAZOLE	PHA	77	2	0.000124	20%	87%	0%
60-51-5	Dimethoate	PES	77	2	0.004813	20%	50%	50%
114-07-8	ERYTHROMYCIN	PHA	77	2	0.000892	19%	88%	0%
443-48-1	METRONIDAZOLE	PHA	77	2	0.002699	18%	88%	0%
71125-38-7	MELOXICAM	PHA	77	2	8.99E-05	18%	88%	0%
106133-20-4	TAMSULOSIN	PHA	77	2	0.000131	18%	88%	0%
76420-72-9	ENALAPRILAT	PHA	77	2	0.009341	18%	88%	0%
81103-11-9	CLARITHROMYCIN	PHA	77	2	0.00141	18%	87%	0%
30125-64-5	Terbumeton-desethyl	PES	77	1	0.001861	17%	0%	100%
330-54-1	Diuron	PES	77	2	0.007871	16%	50%	50%
3380-34-5	Triclosan	ED	77	2	0.00064	15%	86%	0%
886-50-0	Terbutryn	PES	77	2	0.001528	15%	50%	50%
107534-96-3	Tebuconazole	PES	77	1	0.001099	13%	0%	100%
51481-61-9	CIMETIDINE	PHA	77	2	0.000663	13%	87%	0%
563-12-2	Ethion	PES	77	2	0.001139	13%	50%	50%
6190-65-4	Desethylatrazine	PES	77	2	0.004817	13%	50%	50%
1007-28-9	Desisopropylatrazine	PES	77	2	0.004051	12%	50%	50%
34123-59-6	Isoproturon	PES	77	2	0.000824	12%	50%	50%
93957-54-1	FLUVASTATIN	PHA	77	2	0.000171	12%	88%	0%
131-56-6	Benzophenone-1 (BP1)	UV	77	2	0.002719	11%	86%	0%
54910-89-3	FLUOXETINE	PHA	77	2	0.001121	11%	88%	0%
125-29-1	HYDROCODONE	PHA	77	2	0.001753	10%	88%	0%
1563-66-2	Carbofuran	PES	77	2	0.000526	10%	50%	50%
1912-24-9	Atrazine	PES	77	2	0.002154	10%	50%	50%
37350-58-6	METOPROLOL	PHA	77	2	0.002935	10%	88%	0%
6552-13-2	Fenoxon sulfoxide	PES	77	2	0.001975	10%	50%	50%
79794-75-5	LORATIDINE	PHA	77	2	0.000529	10%	88%	0%
82419-36-1	OFLOXACIN	PHA	77	2	0.002418	10%	88%	0%
551-92-8	DIMETRIDAZOLE	PHA	77	2	0.003262	10%	88%	0%
66357-35-5	RANITIDINE	PHA	77	2	0.003419	10%	88%	0%
3930-20-9	SOTALOL	PHA	77	2	0.002918	8%	88%	0%
51218-45-2	Metolachlor	PES	77	2	0.004296	8%	50%	50%
121-75-5	Malathion	PES	77	2	0.003554	8%	50%	50%

CAS	Name	Group	Stations	Surveys	Cmean	Qmean	Sep-Oct	Undef.
75847-73-3	ENALAPRIL	PHA	77	2	0.001783	8%	88%	0%
57018-04-9	Tolclofos-methyl	PES	77	2	0.001741	8%	50%	50%
21245-02-3	Ethylhexyl dimethyl PABA (OD-PABA)	UV	77	2	0.000242	7%	86%	0%
57775-29-8	CARAZOLOL	PHA	77	2	0.000247	7%	88%	0%
7361-61-7	XYLAZINE	PHA	77	2	0.000105	7%	88%	0%
4812-40-2	METRONIDAZOLE-OH	PHA	77	2	0.000753	6%	88%	0%
59804-37-4	TENOXICAM	PHA	77	2	5.1E-05	6%	88%	0%
10238-21-8	GLIBENCLAMIDE	PHA	77	2	0.001871	6%	88%	0%
2032-65-7	Methiocarb	PES	77	2	0.003057	6%	50%	50%
42200-33-9	NADOLOL	PHA	77	2	0.00013	6%	88%	0%
1137-42-4	4-Hydroxybenzophenone (4HB)	UV	77	2	0.009731	5%	86%	0%
56-38-2	Parathion-ethyl	PES	77	2	0.003388	5%	50%	50%
122-34-9	Simazine	PES	77	2	0.003929	5%	50%	50%
139-40-2	Propazine	PES	77	2	0.000596	5%	50%	50%
50-27-1	Estriol	ED	77	2	0.000336	5%	86%	0%
5466-77-3	Ethylhexyl methoxycinnamate (EHMC)	UV	77	2	0.001606	5%	86%	0%
1649-18-9	AZAPERONE	PHA	77	2	0.000963	4%	87%	0%
2804-05-9	AZAPEROL	PHA	77	2	0.001112	4%	87%	0%
33693-04-8	Terbumeton	PES	77	1	0.000346	4%	0%	100 %
611-99-4	4,4'-Dihidroxybenzophenone (4DHB)	UV	77	2	0.001498	4%	86%	0%
79559-97-0	SERTRALINE	PHA	77	2	0.002453	4%	88%	0%
1113-02-6	Omethoate	PES	77	2	0.000513	3%	50%	50%
132539-06-1	OLANZAPINE	PHA	77	2	0.000126	3%	88%	0%
14086-35-2	Fenoxon sulfone	PES	77	2	0.00026	3%	50%	50%
15686-71-2	CEFALEXIN	PHA	77	2	0.000381	3%	87%	0%
2642-71-9	Azinphos ethyl	PES	77	2	0.000577	3%	50%	50%
56161-73-0	NORFLUOXETINE	PHA	77	2	0.001056	3%	88%	0%
7681-76-7	RONIDAZOLE	PHA	77	2	0.001609	3%	88%	0%
101-20-2	Triclocarban	ED	77	2	7.81E-05	3%	86%	0%
81-81-2	WARFARIN	PHA	77	2	9.21E-05	3%	88%	0%
122-14-5	Fenitrothion	PES	77	2	0.002759	2%	50%	50%
16655-82-6	3-hydroxycarbofuran	PES	77	2	0.00029	2%	50%	50%
481-95-8	Estriol 3-sulfate	ED	77	2	0.000224	2%	86%	0%
56-53-1	Diethylstilbestrol	ED	77	2	7.38E-05	2%	86%	0%
6197-30-4	Octocrylene (OC)	UV	77	2	0.003494	2%	86%	0%
76824-35-6	FAMOTIDINE	PHA	77	2	0.000321	2%	88%	0%
86-50-0	Azinphos methyl	PES	77	2	0.000611	2%	50%	50%
3761-42-0	Fenthion sulfone	PES	77	2	0.000314	1%	50%	50%
60-54-8	TETRACYCLINE	PHA	77	2	0.010836	1%	88%	0%
2212-67-1	Molinate	PES	77	2	0.000556	1%	50%	50%
3761-41-9	Fenthion sulfoxide	PES	77	2	0.000226	1%	50%	50%
55-38-9	Fenthion	PES	77	2	0.000226	1%	50%	50%

CAS	Name	Group	Stations	Surveys	Cmean	Qmean	Sep-Oct	Undef.
57-63-6	Ethinyl estradiol	ED	77	2	0.000153	1%	86%	0%
6552-12-1	Fenoxon	PES	77	2	0.000226	1%	50%	50%
131-53-3	2,2'-Dihydroxy-4-methoxybenzophenone (DHMB)	UV	77	2	0.0011	0%	86%	0%
131-55-5	Benzophenone-2 (BP2)	UV	77	2	0.002	0%	86%	0%
15972-60-8	Alachlor	PES	77	2	0.002	0%	50%	50%
298-00-0	Parathion-methyl	PES	77	2	0.002	0%	50%	50%
34256-82-1	Acetochlor	PES	77	2	0.002	0%	50%	50%
36557-05-8	(±)-11-hydri-Δ9-THC	DRU	77	2	0.00158	0%	86%	0%
709-98-8	Propanil	PES	77	2	0.000308	0%	50%	50%
94-09-7	Ethyl 4-aminobenzoate (Et-PABA)	UV	77	2	0.0005	0%	86%	0%
9002-93-1	Octylphenol diethoxylate	ED	76	2	0.004541	100%	87%	0%
84852-15-3	Nonylphenol	ED	66	1	0.055025	100%	87%	0%
2706-90-3	PFPeA	PF	25	1	0.00012	0%	100%	0%
335-67-1	PFOA	PF	16	1	0.000112	25%	56%	0%
335-76-2	PFDA	PF	14	1	0.000116	7%	86%	0%
375-95-1	PFNA	PF	12	1	0.001182	17%	100%	0%
375-85-9	PFHpA	PF	9	1	0.0012	0%	100%	0%
1763-23-1	PFOS	PF	8	1	0.000012	0%	63%	0%
2479-90-5	Estrone 3-glucuronide	ED	7	1	0.004204	100%	57%	0%
67905-19-5	PFHxDA	PF	4	1	0.000111	25%	75%	0%
375-92-8	PFHpS	PF	4	1	0.00012	0%	100%	0%
16517-11-6	PFODA	PF	3	1	0.001012	100%	100%	0%
376-06-7	PFTeDA	PF	3	1	0.00006	0%	100%	0%
2058-94-8	PFUdA	PF	2	1	0.00012	0%	50%	0%
307-24-4	PFHxA	PF	2	1	0.0012	0%	100%	0%
375-22-4	PFBA	PF	2	1	0.00012	0%	100%	0%
375-73-5	PFBS	PF	2	1	0.00006	0%	100%	0%
72629-94-8	PFTrDA	PF	2	1	0.00006	0%	100%	0%
172155-07-6	i,p-PFNA	PF	1	1	0.0012	0%	100%	0%
355-46-4	PFHxS	PF	1	1	0.000012	0%	0%	0%

Annex IV: Prioritisation of the Mediterranean Iberian Rivers Specific Pollutants using the NORMAN methodology

Overview of the NORMAN Prioritisation methodology

A full description and discussion of the NORMAN prioritization methodology can be found at **External Deliverable 19.4 "Guidance for identification of RBSPs and list of Danube RBSP including quantification of their ecological impact and modeling-based exposure and risk predictions validated with case-study data"** and references [1,2].

The NORMAN prioritisation methodology uses a decision tree that first classifies chemicals into six categories depending on the information available. That allows water managers to focus on the next steps to be taken, e.g. (not exhaustive): (1) derivation of EQS for substances already well investigated with sufficient amount of data on their occurrence and toxicity; (2) improvement of analytical methods for substances monitored whose limits of quantification (LOQs) are higher than PNEC values; (3) additional screening when more occurrence data are needed to confirm a basin wide thread; and, (4) discontinue with monitoring of substances that are already well investigated and proved not to represent a threat to the environment. The priority within each category is then evaluated based on several indicators, including exposure (e.g. frequency of observations above LOQs of used methods, annual usage, use pattern, etc.), hazard (e.g. Persistence, Bioaccumulation, Toxicity (PBT), Endocrine Disruption (ED) and Carcinogenicity, Mutagenicity and Reprotoxicity (CMR) properties) and risk (cf. text below).

Considering the specifics of the JDS3 dataset, no categorisation was run and only two risk indicators were proposed for the prioritisation of target analytes detected in surface water samples, namely the *Frequency of Exceedance (FoE)* and the *Extent of Exceedance (EoE)*, that are subsequently added to a final ranking score (RS between 0 and 2; see Section 3.4.6). The surface water samples from the 68 monitoring sites have been analysed by different laboratories, using various analytical methods. Hence, multiple entries for the same site/compound combination exist. In order to aggregate them to a single measure of exposure for each sampling site, the maximum concentration from all measurements was used. The reason for this was not to bias towards substances, which have been analysed only by one laboratory.

Frequency of Exceedance

The first indicator considers the spatial distribution of potential effects of a certain compound, *i.e.* the frequency of sites with observations above the lowest PNEC. For the calculation of this indicator, the maximum observed concentration at each site (MEC_{site}) is compared to the lowest PNEC. In the JDS3 case, quite often several measurements of a single compound were performed by different laboratories at

the same sample using different methodologies. The maximum concentrations per compound per site were directly used to compare them with the lowest PNEC. Subsequently, the number of sites where the threshold was exceeded was divided by the total number of sites, where the respective compound was measured. Please note that the total number of 68 sites was used for all prioritised substances despite some of the substances were not determined in all samples for some analytical methods (e.g. LVSPE samples for special organic pollutants analysis taken only from 22 sites). The resulting values lie within 0 and 1 and can directly be used as input for the ranking score.

To give an example of the calculation, a hypothetical dataset consists of 20 sites with one sample each. In total, compound A was found 18 times, while compound B was found 12 times. The maximum concentrations of compound A exceeded the lowest PNEC at ten sites, while the maximum concentrations of compound B exceed the lowest PNEC only at 5 sites. The RS for the indicator "*Frequency of Exceedance*" calculates as follows:

Compound A: FoE = 10 sites exceeding lowest PNEC / 20 sites = 0.50

Compound B: FoE = 5 sites exceeding lowest PNEC / 20 sites = 0.25

Hence, compound B has a lower risk as compared to compound A.

Extent of Exceedance

The second indicator considers the extent of local effects. For the calculation of this indicator, again all raw data is used. All concentration data above the LOQ is pooled and used to calculate a MEC_{95}. The MEC_{95} is the 95^{th} percentile of the measured concentrations, separately for each compound. It is recommended to have at least 20 monitoring sites to get a reliable statistical result. For the calculation, the Excel formula "QUANTIL" can be used. The MEC_{95} is then divided by the lowest PNEC to derive the "*Extent of Exceedance*". This value can consist of values below 1 and up to several thousands. RS is assigned as explained in the text above:

- EoE <1 → RS = 0
- 10≥ EoE ≥1 → RS = 0.1
- 100≥ EoE >10 → RS = 0.2
- 1000≥ EoE >100 → RS = 0.5
- EoE >1000 → RS = 1

For the example above, we assume that the MEC_{95} of compound A is 2 µg/l, while the MEC_{95} of compound B is 20 µg/l, due to generally higher concentrations. If the lowest PNEC in this example is 1 µg/L for both substances, the "*Extent of Exceedance*" calculates as follows:

Compound A: EoE = MEC_{95} of 2µg/l / lowest PNEC of 1 µg/l = 2

Compound B: EoE = MEC_{95} of 25µg/l / lowest PNEC of 1 µg/l = 25

The RS score for compound A is then 0.1 (EoE < 10), while compound B has a higher score of 0.2 for the second indicator.

Final Ranking Score

The final ranking score RS is then calculated by simply adding both scores. Please note that the maximum score is therefore a RS value of 2. In our example, the RS calculates as follows:

Compound A: RS 1 of 0.50 + RS 2 of 0.1 = 0.60

Compound B: RS 1 of 0.25 + RS 2 of 0.2 = 0.45

In this example, compound A has a higher priority than compound B, although both compounds had the highest score in one of the two indicators. However, the relatively large distribution of compound A (50% of sites exceeded the lowest PNEC) lead to the overall higher priority.

Results of the prioritization of the Mediterranean Iberian River Basin-Specific Pollutants

The results of the prioritization exercise for the 195 compounds considered, measured at 76 sites along the Iberian rivers (Ebro, Llobregat, Júcar, and Guadalquivir) are summarized in Table IV.1. Compounds yielding a final score (total score) exceeding 0 (37 compounds) were considered relevant and eight of them were selected as River Basin-Specific Pollutants of the Mediterranean Iberian rivers studied (Category 1). The results are presented in Table IV.2 and Figure IV.1. Compounds include 3 hormones, 6 industrial compounds, 15 pesticides, 3 personal care products and 10 pharmaceuticals (Figure IV.2)

Top 10 rank compounds (Score Total ≥0,15) are the hormones 17-beta-Estradiol and Estrone, the pesticides Pyriproxyfen, Dichlofenthion, Diazinon, the industrial compounds PFOS and Bisphenol A, and the pharmaceuticals Ibuprofen, Diclofenac and Lorazepam.

The frequency of exceedance (number of sites where the measured concentration exceeds the PNEC), seems to be the dominating factor in the total score. Thus 8 out of the 10 top compounds exceeding PNEC are coincident with those of highest total score, the other two are the organophosphorus pesticides parathion and malathion.

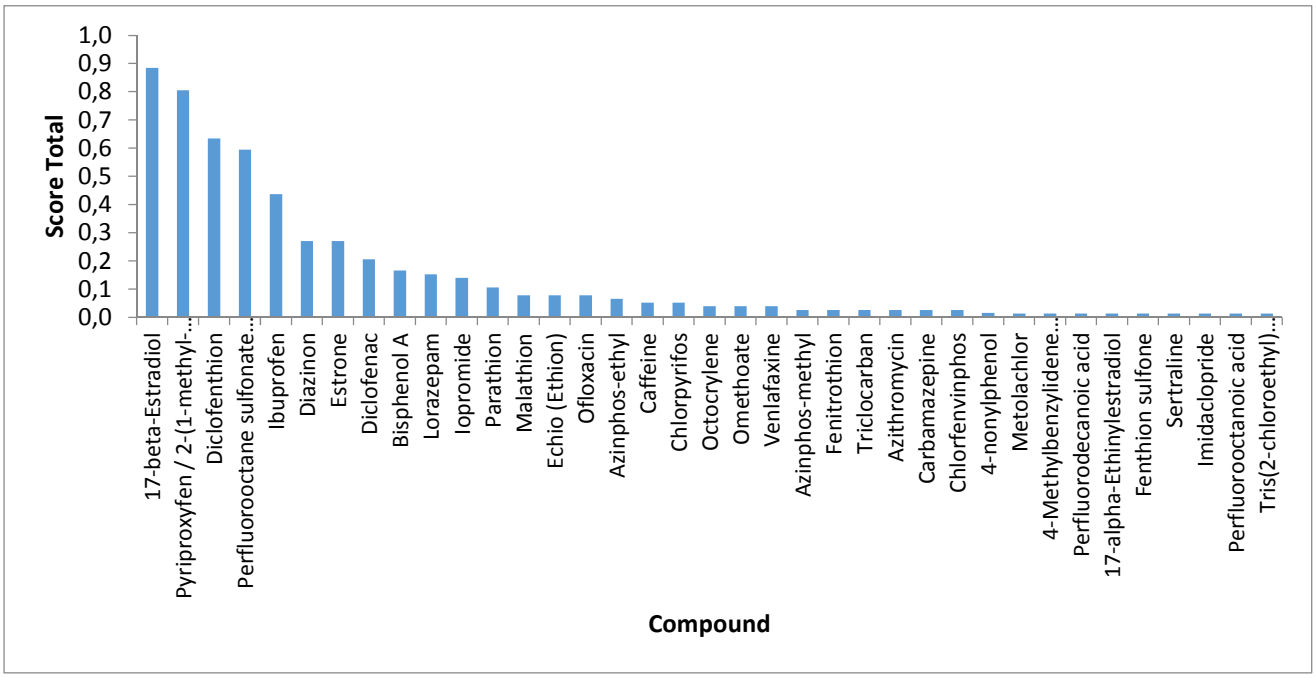


Figure IV.1 River Basin Specific Pollutants for the Iberian Rivers studied, obtained using the Norman prioritization method.

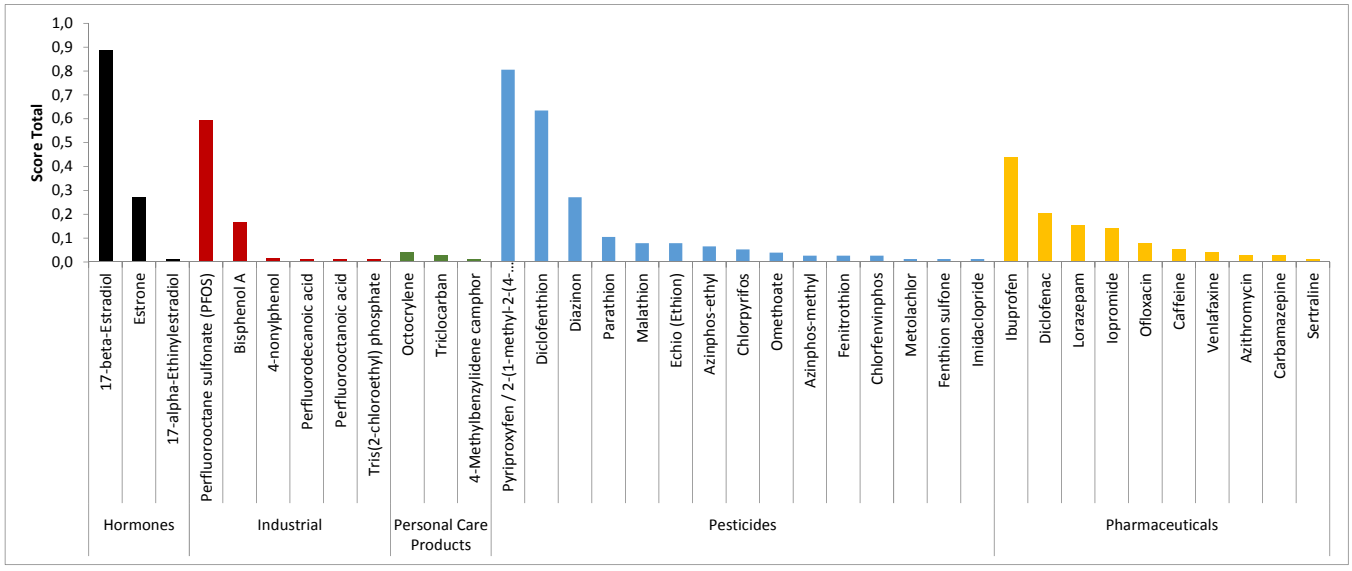


Figure IV.2 River Basin-Specific Pollutants for the Iberian Rivers studied, obtained using the Norman prioritization method and classified per families (Hormones, Industrial, Personal Care Products, Pesticides, and Pharmaceuticals.

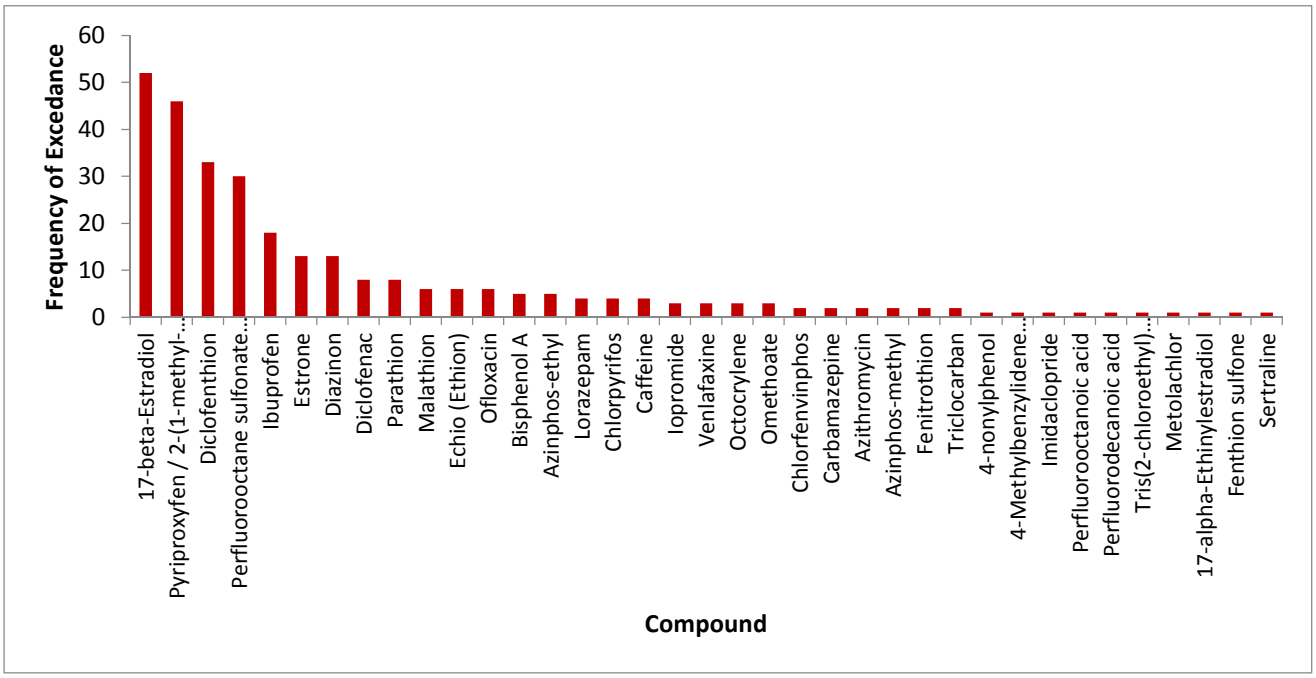


Figure IV.3 River Basin Specific Pollutants for the Iberian Rivers studied, ranked by frequency of exceedance.

References

1. Dulio V. and Von der Ohe P. C. (eds), 2013. NORMAN prioritization framework for emerging substances. NORMAN Association Network of reference laboratories and related organizations for monitoring and bio-monitoring of emerging environmental substances. Working Group on Prioritisation of Emerging Substances NORMAN Association, Verneuil en Halatte, 70 pp. http://www.norman-network.net/sites/default/files/files/Publications/NORMAN_prioritisation_Manual_15%20April2013_final%20for%20website-f.pdf.
2. Dulio V. and Slobodnik J., 2015. In Response: The NORMAN perspectives on prioritization of emerging pollutants. Environ. Toxicol. Chem. 34: 2183–2185; http://onlinelibrary.wiley.com/doi/10.1002/etc.3047/pdf.

Table IV.1. Prioritisation of the Mediterranean Iberian River Basin-Specific Pollutants

Use for prioirty list	Substance	CAS No.	No. of sites (new)	# of sites where MECsite > PNEC (new)	MEC95 (new)	MECsite Max (new)	LoQ min	Cat.	Lowest PNEC	PNEC type	Reference PNEC	Max exceedance	Extent of Exceedence	Score EoE	Score FoE	Score Total	LoQ exceedance
x	17-beta-Estrac	50-28-2	76	52	0,007	0,008	0,0001	1	0,0004	AA-EQS	DIRECTIVE 201	19,433	16,686	0,200	0,684	0,88	0,30
x	Pyriproxyfen /	95737-68-1	76	46	0,090	0,100	0,0015	1	0,0015	PNEC chronic	Footprint (201	66,393	59,713	0,200	0,605	0,81	1,0000
x	Diclofenthion	97-17-6	76	33	0,050	0,055	0,0015	1	0,0041	PNEC acute	Aquire 6797	13,368	12,215	0,200	0,434	0,63	0,37
x	Perfluorooctar	1763-23-1	76	30	0,055	2,709	0,0000	1	0,00065	EQS chronic wa	DIRECTIVE 201	4167,249	85,237	0,200	0,395	0,59	0,02
x	Ibuprofen	15687-27-1	76	18	0,177	0,868	0,0039	1	0,01	EQS chronic wa	EQS DATASHEE	86,782	17,724	0,200	0,237	0,44	0,39
x	Diazinon	333-41-5	76	13	0,013	0,024	0,0002	1	0,01	JD-UQN	UBA (2016) Obe	2,375	1,255	0,100	0,171	0,27	0,02
x	Estrone	53-16-7	76	13	0,006	0,007	0,0002	1	0,0036	EQS-proposal	WL substance	2,040	1,705	0,100	0,171	0,27	0,05
x	Diclofenac	15307-86-5	76	8	0,109	0,280	0,0021	1	0,05	EQS-proposal	WL substance	5,600	2,187	0,100	0,105	0,21	0,04
x	Bisphenol A	80-05-7	76	5	0,250	0,649	0,0004	1	0,2	EQS chronic wa	UBA (2017) EQS	3,247	1,249	0,100	0,066	0,17	0,00
x	Malathion	121-75-5	76	6		0,320	0,0009	2	0,006	AA-QSwater_e	INERIS (2017)	53,392	0,000	0,000	0,079	0,08	0,15
x	Octocrylene	6197-30-4	76	3		0,027	0,0090	2	0,023	PNEC aqua (fre	ECHA DOSSIER	1,174	0,000	0,000	0,039	0,04	0,39
x	Omethoate	1113-02-6	76	3		0,012	0,0009	2	0,004	JD-UQN	UBA (2016) Obe	2,928	0,000	0,000	0,039	0,04	0,23
x	Azinphos-met	86-50-0	76	2		0,009	0,0015	2	0,0065	JG-MKN (totaal	RIVM (2018)	1,337	0,000	0,000	0,026	0,03	0,23
x	Fenitrothion	122-14-5	76	2		0,047	0,0060	2	0,009	JD-UQN	UBA (2016) Obe	5,266	0,000	0,000	0,026	0,03	0,67
x	Triclocarban	101-20-2	76	2		0,003	0,0001	2	0,00112	PNEC chronic	Aquire 90724	3,079	0,000	0,000	0,026	0,03	0,11
x	Metolachlor	51218-45-2	76	1		0,447	0,0009	2	0,2	JD-UQN	UBA (2016) Obe	2,235	0,000	0,000	0,013	0,01	0,00
	2,4-Dihydroxyb	131-56-6	76	0		0,055	0,0032	2	33	PNEC aqua (fre	ECHA DOSSIER	0,002	0,000	0,000	0,000	0,00	0,00
	3-hydroxycarbo	16655-82-6	76	0		0,008	0,0006	2	4,3	Indicatief MTR	RIVM (2018)	0,002	0,000	0,000	0,000	0,00	0,00
	Acetochlor	34256-82-1	76	0			0,0060	2	0,013	AA-QSwater_e	INERIS (2008)	0,000	0,000	0,000	0,000	0,00	0,46
	Alachlor	15972-60-8	76	0			0,0060	2	0,3	EQS chronic wa	DIRECTIVE 201	0,000	0,000	0,000	0,000	0,00	0,02
	Atrazine	1912-24-9	76	0		0,020	0,0039	2	0,6	EQS chronic wa	DIRECTIVE 201	0,033	0,000	0,000	0,000	0,00	0,01
	Carbendazim	10605-21-7	76	0			0,0000	2	0,15	AA-QSwater_e	INERIS (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Carbofuran	1563-66-2	76	0		0,007	0,0006	2	0,016	AA-QSwater_e	INERIS (2008)	0,422	0,000	0,000	0,000	0,00	0,04
	Desethylatrazi	6190-65-4	76	0		0,097	0,0060	2	0,6	PNEC chronic	CIRCA (2008) da	0,162	0,000	0,000	0,000	0,00	0,01
	Desethylterbu	30125-63-4	76	0			0,0000	2	0,25	JG-MKN (totaal	RIVM (2018)	0,000	0,000	0,000	0,000	0,00	0,00
	Diethylstilbes	56-53-1	76	0		0,002	0,0001	2	4,86	Indicatief MTR	RIVM (2018)	0,000	0,000	0,000	0,000	0,00	0,00
	Diuron	330-54-1	76	0		0,151	0,0050	2	0,2	EQS chronic wa	DIRECTIVE 201	0,755	0,000	0,000	0,000	0,00	0,03
	Estriol	50-27-1	76	0		0,006	0,0006	2	0,06	PNEC	Caldwell et al	0,095	0,000	0,000	0,000	0,00	0,01
	Ethyl 4-aminol	94-09-7	76	0			0,0015	2	9,2	PNEC acute	Aquire 182	0,000	0,000	0,000	0,000	0,00	0,00
	Ethylhexyl met	5466-77-3	76	0		0,041	0,0022	2	6	EQS-proposal	WL substance	0,007	0,000	0,000	0,000	0,00	0,00
	Fenthion	55-38-9	76	0		0,003	0,0010	2	0,004	JD-UQN	UBA (2016) Obe	0,660	0,000	0,000	0,000	0,00	0,25
	Fenthion sulfo	3761-41-9	76	0		0,003	0,0010	2	0,24	Indicatief MTR	RIVM (2018)	0,011	0,000	0,000	0,000	0,00	0,00
	Fluoxetine	54910-89-3	76	0		0,017	0,0012	2	0,1	PNEC chronic	Aquire 164093	0,173	0,000	0,000	0,000	0,00	0,01
	Isoproturon /	34123-59-6	76	0		0,025	0,0009	2	0,3	EQS chronic wa	DIRECTIVE 201	0,085	0,000	0,000	0,000	0,00	0,00
	Methiocarb	2032-65-7	76	0		0,003	0,0009	2	0,01	EQS-proposal	WL substance	0,323	0,000	0,000	0,000	0,00	0,09
	Metoprolol	37350-58-6	76	0		0,292	0,0004	2	8,6	AA-EQS	OZ (2016) EQS I	0,034	0,000	0,000	0,000	0,00	0,00
	Molinate	2212-67-1	76	0		0,009	0,0015	2	3,8	PNEC acute	DG-SANCO 200	0,002	0,000	0,000	0,000	0,00	0,00
	Parathion met	298-00-0	76	0			0,0060	2	0,02	JD-UQN	UBA (2016) Obe	0,000	0,000	0,000	0,000	0,00	0,30
	Perfluorohexa	307-24-4	76	0		0,031	0,0012	2	140	AA-EQS	Italian EQS Wo	0,000	0,000	0,000	0,000	0,00	0,00
	Perfluoronona	375-95-1	76	0		0,116	0,0012	2	1	PNEC chronic	Aquire 160553	0,116	0,000	0,000	0,000	0,00	0,00
	Perfluorooctar	754-91-6	76	0			0,0006	2	0,00065	EQS chronic wa	DIRECTIVE 201	0,000	0,000	0,000	0,000	0,00	0,92
	Propanil	709-98-8	76	0			0,0009	2	0,2	AA-QSwater_e	INERIS (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Propazine	139-40-2	76	0		0,013	0,0009	2	0,18	PNEC acute	Footprint (201	0,074	0,000	0,000	0,000	0,00	0,01
	Ranitidine	66357-35-5	76	0		0,050	0,0035	2	3,1	PNEC chronic	Aquire 156179	0,016	0,000	0,000	0,000	0,00	0,00
	Simazine	122-34-9	76	0		0,048	0,0060	2	1	EQS chronic wa	DIRECTIVE 201	0,048	0,000	0,000	0,000	0,00	0,01
	Tebuconazole	107534-96-3	76	0			0,0004	2	0,24	AA-EQS	OZ (2016) EQS I	0,000	0,000	0,000	0,000	0,00	0,00

	Terbumeton	33693-04-8	76	0			0,0000	2	0,023	PNEC acute	Aquire 62304	0,000	0,000	0,000	0,000	0,00	0,00
	Terbuthylazine	5915-41-3	76	0			0,0000	2	0,06	AA-QSwater_e	INERIS (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Terbutryn	886-50-0	76	0		0,015	0,0015	2	0,065	EQS chronic wa	DIRECTIVE 201	0,228	0,000	0,000	0,000	0,00	0,02
	Terbutylazine-	66753-07-9	76	0			0,0000	2	0,0073	PNEC chronic	Aquire 174504	0,000	0,000	0,000	0,000	0,00	0,00
	Tetracycline	60-54-8	76	0		0,027	0,0118	2	236,9947	AA-EQS	OZ (2012) EQS Do	0,000	0,000	0,000	0,000	0,00	0,00
	Tolclofos methyl	57018-04-9	76	0		0,036	0,0015	2	1,2	wettelijk JG-MKN	RIVM (2018)	0,030	0,000	0,000	0,000	0,00	0,00
	Triclosan	3380-34-5	76	0		0,019	0,0006	2	0,02	EQS chronic wate	UBA (2012) EQS D	0,948	0,000	0,000	0,000	0,00	0,03
x	Lorazepam	846-49-1	76	4	0,118	0,306	0,0037	3	0,096	P-PNEC	ToxTram (2017)	3,184	1,230	0,100	0,053	0,15	0,04
x	Iopromide	73334-07-3	76	3	0,327	1,369	0,0006	3	0,143	P-PNEC	ToxTram (2017)	9,572	2,287	0,100	0,039	0,14	0,00
x	4-Methylbenzyli	36861-47-9	76	1	0,116	0,173	0,0034	3	0,171	P-PNEC	Aquire 170704	1,012	0,679	0,000	0,013	0,01	0,02
x	Perfluorodecanc	335-76-2	76	1	0,051	0,213	0,0001	3	0,1655	P-PNEC	ToxTram (2017)	1,287	0,307	0,000	0,013	0,01	0,00
	(±)-Methadone h	-	76	0	0,008	0,020	0,0002	3	0,658	-	-	0,030	0,012	0,000	0,000	0,00	0,00
	1S,2R (+)-Ephedri	-	76	0	0,062	0,144	0,0004	3	38,446	-	-	0,004	0,002	0,000	0,000	0,00	0,00
	ACRIDONE	578-95-0	76	0	0,020	0,043	0,0001	3	0,3886	P-PNEC	ToxTram (2017)	0,110	0,053	0,000	0,000	0,00	0,00
	Albendazole	54965-21-8	76	0	0,002	0,002	0,0001	3	0,2627	P-PNEC	ToxTram (2017)	0,007	0,007	0,000	0,000	0,00	0,00
	Albuterol	18559-94-9	76	0	0,011	0,016	0,0000	3	17,133	P-PNEC	ToxTram (2017)	0,001	0,001	0,000	0,000	0,00	0,00
	Alprazolam	28981-97-7	76	0	0,007	0,008	0,0005	3	0,07684	P-PNEC	ToxTram (2017)	0,105	0,089	0,000	0,000	0,00	0,01
	ATORVASTATIN	134523-00-5	76	0	0,002	0,009	0,0000	3	0,4337	P-PNEC	ToxTram (2017)	0,020	0,004	0,000	0,000	0,00	0,00
	Benzoylecgonine	519-09-5	76	0	0,049	0,651	0,0009	3	2,3338	P-PNEC	ToxTram (2017)	0,279	0,021	0,000	0,000	0,00	0,00
	Benzylparaben	94-18-8	76	0	0,007	0,007	0,0001	3	2,9474	P-PNEC	ToxTram (2017)	0,002	0,002	0,000	0,000	0,00	0,00
	Clopidogrel	113665-84-2	76	0	0,007	0,014	0,0000	3	0,617	P-PNEC	ToxTram (2017)	0,023	0,012	0,000	0,000	0,00	0,00
	Cocaine	50-36-2	76	0	0,014	0,728	0,0003	3	2,456509322	P-PNEC	ToxTram (2017)	0,296	0,006	0,000	0,000	0,00	0,00
	Codeine	76-57-3	76	0	0,012	0,064	0,0001	3	7,185662544	P-PNEC	ToxTram (2017)	0,009	0,002	0,000	0,000	0,00	0,00
	DESLORATIDINE	100643-71-8	76	0	0,002	0,006	0,0001	3	0,3365	P-PNEC	ToxTram (2017)	0,019	0,006	0,000	0,000	0,00	0,00
	Dexamethasone	50-02-2	76	0	0,003	0,005	0,0002	3	19,1327	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	DILTIAZEM	42399-41-7	76	0	0,009	0,043	0,0001	3	0,2252	P-PNEC	ToxTram (2017)	0,189	0,040	0,000	0,000	0,00	0,00
	ENALAPRILAT	76420-72-9	76	0	0,091	0,098	0,0036	3	1,353	P-PNEC	ToxTram (2017)	0,073	0,067	0,000	0,000	0,00	0,00
	Estrone sulphate	481-97-0	76	0	0,005	0,009	0,0000	3	24,606	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Ethyl paraben	120-47-8	76	0	0,017	0,049	0,0009	3	4,599	P-PNEC	ToxTram (2017)	0,011	0,004	0,000	0,000	0,00	0,00
	Furosemide	54-31-9	76	0	0,118	0,296	0,0015	3	0,7067	P-PNEC	ToxTram (2017)	0,420	0,167	0,000	0,000	0,00	0,00
	Hydrochlorothia	58-93-5	76	0	0,326	1,147	0,0002	3	8,381	P-PNEC	ToxTram (2017)	0,137	0,039	0,000	0,000	0,00	0,00
	Ketoprofen	22071-15-4	76	0	0,080	0,357	0,0025	3	2,09574	P-PNEC	ToxTram (2017)	0,170	0,038	0,000	0,000	0,00	0,00
	LEVAMISOL	14769-73-4	76	0	0,036	0,063	0,0000	3	1,8127	P-PNEC	ToxTram (2017)	0,035	0,020	0,000	0,000	0,00	0,00
	Methamphetam	537-46-2	76	0	0,002	0,003	0,0001	3	9,735514	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Methyl paraben	99-76-3	76	0	0,046	0,142	0,0007	3	5	P-PNEC exp.	Aquire 103220	0,028	0,009	0,000	0,000	0,00	0,00
	METRONIDAZOLE	443-48-1	76	0	0,039	0,066	0,0019	3	33,08	P-PNEC	ToxTram (2017)	0,002	0,001	0,000	0,000	0,00	0,00
	Oxybenzone	131-57-7	76	0	0,041	0,044	0,0030	3	1,5411	P-PNEC	ToxTram (2017)	0,029	0,026	0,000	0,000	0,00	0,00
	Oxycodone	76-42-6	76	0	0,007	0,025	0,0002	3	8,03606	P-PNEC	ToxTram (2017)	0,003	0,001	0,000	0,000	0,00	0,00
	Perfluorobutano	375-22-4	76	0	0,552	0,743	0,0001	3	27,753	P-PNEC	ToxTram (2017)	0,027	0,020	0,000	0,000	0,00	0,00
	Perfluorohexane	355-46-4	76	0	0,035	0,089	0,0000	3	190	P-PNEC	ECOSAR v1.11 (20	0,000	0,000	0,000	0,000	0,00	0,00
	Perfluoropentan	2706-90-3	76	0	0,021	0,068	0,0001	3	3,9137	P-PNEC	ToxTram (2017)	0,017	0,005	0,000	0,000	0,00	0,00
	PIROXICAM	36322-90-4	76	0	0,004	0,005	0,0001	3	0,6219	P-PNEC	ToxTram (2017)	0,008	0,007	0,000	0,000	0,00	0,00
	Pravastatin	81093-37-0	76	0	0,008	0,011	0,0004	3	4,5662	P-PNEC	ToxTram (2017)	0,002	0,002	0,000	0,000	0,00	0,00
	Propyl paraben	94-13-3	76	0	0,019	0,026	0,0001	3	12,3	P-PNEC exp.	Aquire 158949	0,002	0,002	0,000	0,000	0,00	0,00
	TAMSULOSIN	106133-20-4	76	0	0,002	0,005	0,0001	3	0,3461	P-PNEC	ToxTram (2017)	0,015	0,006	0,000	0,000	0,00	0,00
	TORASEMIDE	56211-40-6	76	0	0,010	0,020	0,0001	3	0,4898	P-PNEC	ToxTram (2017)	0,042	0,021	0,000	0,000	0,00	0,00

	Trazodone	19794-93-5	76	0	0,032	0,040	0,0001	3	5,2123	P-PNEC	ToxTram (2017)	0,008	0,006	0,000	0,000	0,00	0,00
x	Parathion	56-38-2	76	8		0,042	0,0060	4	0,005	JD-UQN	UBA (2016) Oberl	8,384	0,000	0,000	0,105	0,11	1,20
x	Echio (Ethion)	563-12-2	76	6		0,024	0,0015	4	0,00048	PNEC acute	Aquire 175414	50,583	0,000	0,000	0,079	0,08	3,13
x	Azinphos-ethyl	2642-71-9	76	5		0,003	0,0015	4	0,0011	JG-MKN (opgelos	RIVM (2018)	3,118	0,000	0,000	0,066	0,07	1,36
x	17-alpha-Ethinyl	57-63-6	76	1		0,002	0,0005	4	0,000035	EQS chronic wate	DIRECTIVE 2011/	63,229	0,000	0,000	0,013	0,01	13,43
x	Ofloxacin	82419-36-1	76	6		0,110	0,0001	5	0,021	P-PNEC exp.	Aquire 80421	5,214	0,000	0,000	0,079	0,08	0,01
x	Fenthion sulfone	3761-42-0	76	1		0,014	0,0010	5	0,0095	P-PNEC	ToxTram (2017)	1,439	0,000	0,000	0,013	0,01	0,11
x	Sertraline	79617-96-2	76	1		0,145	0,0021	5	0,0914	P-PNEC exp.	Aquire 107936	1,585	0,000	0,000	0,013	0,01	0,02
	(±)-11-hydro-?9-T	36557-05-8	76	0			0,0043	5	0,2836	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,02
	(±)-11-nor-9-carb	-	76	0		0,017	0,0043	5	0,0715	-	-	0,238	0,000	0,000	0,000	0,00	0,06
	2,2'-Dihydroxy-4	131-53-3	76	0			0,0033	5	0,8588	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	4,4'-Dihidroxybe	611-99-4	76	0		0,153	0,0012	5	6,5636	P-PNEC	ToxTram (2017)	0,023	0,000	0,000	0,000	0,00	0,00
	4-Hydroxybenzo	1137-42-4	76	0		1,458	0,0006	5	2,7727	P-PNEC	ToxTram (2017)	0,526	0,000	0,000	0,000	0,00	0,00
	6-Acetylmorphin	2784-73-8	76	0		0,003	0,0006	5	5,2	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	6-Deisopropylat	1007-28-9	76	0		0,030	0,0060	5	0,4	P-PNEC	ToxTram (2017)	0,078	0,000	0,000	0,000	0,00	0,02
	Amphetamine	300-62-9	76	0		0,007	0,0030	5	24,797	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	AZAPEROL	2804-05-9	76	0		0,004	0,0011	5	0,571	P-PNEC	ToxTram (2017)	0,007	0,000	0,000	0,000	0,00	0,00
	AZAPERONE	1649-18-9	76	0		0,007	0,0008	5	0,974	P-PNEC	ToxTram (2017)	0,007	0,000	0,000	0,000	0,00	0,00
	Benzophenone-2	131-55-5	76	0			0,0060	5	7,2633	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Cannabidiol	-	76	0			0,0092	5	0,1677	-	-	0,000	0,000	0,000	0,000	0,00	0,06
	Cannabinol	-	76	0			0,0096	5	0,0796	-	-	0,000	0,000	0,000	0,000	0,00	0,12
	Carazolol	57775-29-8	76	0		0,003	0,0004	5	0,26549	P-PNEC	ToxTram (2017)	0,010	0,000	0,000	0,000	0,00	0,00
	Cefalexin	15686-71-2	76	0		0,001	0,0008	5	1,4699	P-PNEC	ToxTram (2017)	0,001	0,000	0,000	0,000	0,00	0,00
	CIMETIDINE	51481-61-9	76	0		0,034	0,0003	5	2,3542	P-PNEC	ToxTram (2017)	0,014	0,000	0,000	0,000	0,00	0,00
	DIMETRIDAZOLE	551-92-8	76	0		0,047	0,0049	5	29,5247	P-PNEC	ToxTram (2017)	0,002	0,000	0,000	0,000	0,00	0,00
	Enalapril	75847-73-3	76	0		0,010	0,0016	5	1,5753	P-PNEC	ToxTram (2017)	0,006	0,000	0,000	0,000	0,00	0,00
	Estradiol 17-gluc	NA	4	0		0,007	0,0000	5	31318	-	-	0,000	0,000	0,000	0,000	0,00	0,00
	Estrone 3-glucur	2479-90-5	7	0		0,008	0,0000	5	8,384	P-PNEC	ToxTram (2017)	0,001	0,000	0,000	0,000	0,00	0,00
	Ethylhexyl dimet	21245-02-3	76	0		0,005	0,0003	5	0,3023	P-PNEC	ToxTram (2017)	0,016	0,000	0,000	0,000	0,00	0,00
	Famotidine	76824-35-6	76	0		0,018	0,0003	5	17,319	P-PNEC	ToxTram (2017)	0,001	0,000	0,000	0,000	0,00	0,00
	Fenoxon	6552-12-1	76	0		0,003	0,0010	5	0,198	P-PNEC	ToxTram (2017)	0,013	0,000	0,000	0,000	0,00	0,01
	Fenoxon sulfone	14086-35-2	76	0		0,003	0,0010	5	0,0797	P-PNEC	ToxTram (2017)	0,033	0,000	0,000	0,000	0,00	0,01
	Fenoxon sulfoxic	6552-13-2	76	0		0,051	0,0010	5	0,4821	P-PNEC	ToxTram (2017)	0,105	0,000	0,000	0,000	0,00	0,00
	FLUVASTATIN	93957-54-1	76	0		0,004	0,0001	5	0,1733	P-PNEC	ToxTram (2017)	0,024	0,000	0,000	0,000	0,00	0,00
	Glibenclamide ((	10238-21-8	76	0		0,005	0,0018	5	0,063358995	P-PNEC	ToxTram (2017)	0,073	0,000	0,000	0,000	0,00	0,03
	Heroin	561-27-3	76	0		0,002	0,0017	5	65,47	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Hydrocodone	125-29-1	76	0		0,004	0,0018	5	3,461	P-PNEC	ToxTram (2017)	0,001	0,000	0,000	0,000	0,00	0,00
	i,p-PFNA	172155-07-6	76	0		0,005	0,0012	5	2,222	P-PNEC	ToxTram (2017)	0,002	0,000	0,000	0,000	0,00	0,00
	Loratadine	79794-75-5	76	0		0,010	0,0004	5	0,1185	P-PNEC	ToxTram (2017)	0,088	0,000	0,000	0,000	0,00	0,00
	LSD	-	76	0			0,0018	5	0,3949	-	-	0,000	0,000	0,000	0,000	0,00	0,00
	METRONIDAZOLE	4812-40-2	76	0		0,004	0,0014	5	32,7766	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Morphine	57-27-2	76	0		0,022	0,0009	5	5,38068	P-PNEC	ToxTram (2017)	0,004	0,000	0,000	0,000	0,00	0,00
	Nadolol	42200-33-9	76	0		0,003	0,0002	5	7,2068	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	NORFLUOXETINE	56161-73-0	76	0		0,003	0,0017	5	1,70258	P-PNEC	ToxTram (2017)	0,002	0,000	0,000	0,000	0,00	0,00
	Octylphenol mor	NA	1	0		0,001	0,0000	5	0,7913	-	-	0,002	0,000	0,000	0,000	0,00	0,00
	Olanzapine	132539-06-1	76	0		0,006	0,0002	5	0,0542	P-PNEC	ToxTram (2017)	0,103	0,000	0,000	0,000	0,00	0,00
	Perfluorododeca	307-55-1	76	0		0,010	0,0001	5	0,1149	-	-	0,085	0,000	0,000	0,000	0,00	0,00
	Perfluoroheptan	375-85-9	76	0		0,087	0,0012	5	0,505	P-PNEC	ToxTram (2017)	0,173	0,000	0,000	0,000	0,00	0,00

	Perfluoroheptan	375-85-9	76	0		0,087	0,0012	5	0,505	P-PNEC	ToxTram (2017)	0,173	0,000	0,000	0,000	0,00	0,00
	Perfluoro-n-unde	2058-94-8	76	0		0,004	0,0001	5	0,1257	P-PNEC	ToxTram (2017)	0,029	0,000	0,000	0,000	0,00	0,00
	Perfluorotetrade	376-06-7	76	0		0,018	0,0001	5	0,0828	P-PNEC	ToxTram (2017)	0,212	0,000	0,000	0,000	0,00	0,00
	PFDS	NA	76	0		0,006	0,0001	5	1,443	-	-	0,004	0,000	0,000	0,000	0,00	0,00
	PFHpS	375-92-8	76	0			0,0001	5	0,4806	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	PFHxDA	67905-19-5	76	0		0,005	0,0001	5	0,0777	P-PNEC	ToxTram (2017)	0,067	0,000	0,000	0,000	0,00	0,00
	PFODA	16517-11-6	76	0		0,056	0,0024	5	0,0687	P-PNEC	ToxTram (2017)	0,817	0,000	0,000	0,000	0,00	0,03
	PFTrDA	72629-94-8	76	0		0,010	0,0001	5	0,1032	P-PNEC	ToxTram (2017)	0,094	0,000	0,000	0,000	0,00	0,00
	RONIDAZOLE	7681-76-7	76	0		0,008	0,0028	5	16,7065	P-PNEC	ToxTram (2017)	0,000	0,000	0,000	0,000	0,00	0,00
	Sotalol	3930-20-9	76	0		0,224	0,0008	5	6,5185	P-PNEC	ToxTram (2017)	0,034	0,000	0,000	0,000	0,00	0,00
	TENOXICAM	59804-37-4	76	0		0,002	0,0000	5	0,6714	P-PNEC	ToxTram (2017)	0,002	0,000	0,000	0,000	0,00	0,00
	Terbumeton-des	30125-64-5	76	0			0,0004	5	0,037	P-PNEC	ECOSAR v1.11 (20	0,000	0,000	0,000	0,000	0,00	0,01
	Warfarin / Coum	81-81-2	76	0		0,002	0,0001	5	12	P-PNEC exp.	Aquire 848	0,000	0,000	0,000	0,000	0,00	0,00
	XYLAZINE	7361-61-7	76	0		0,002	0,0001	5	0,6482	P-PNEC	ToxTram (2017)	0,003	0,000	0,000	0,000	0,00	0,00
	1,2,3-Benzotriaz	95-14-7	76	0	1,292	3,185	0,0002	6	19	AA-EQS	OZ (2015) EQS Do	0,168	0,068	0,000	0,000	0,00	0,00
	Acetaminophen	103-90-2	76	0	0,133	0,293	0,0001	6	134	PNEC aqua (fresh	ECHA DOSSIER (03	0,002	0,001	0,000	0,000	0,00	0,00
	AMLODIPINE	88150-42-9	76	0	0,004	0,024	0,0003	6	0,23	PNEC aqua (fresh	ECHA DOSSIER (03	0,102	0,019	0,000	0,000	0,00	0,00
	Atenolol	29122-68-7	76	0	0,085	0,605	0,0001	6	150	AA-EQS	OZ (2010) EQS Do	0,004	0,001	0,000	0,000	0,00	0,00
	Bezafibrate	41859-67-0	76	0	0,022	0,056	0,0001	6	2,3	AA-EQS	OZ (2016) EQS Do	0,024	0,009	0,000	0,000	0,00	0,00
	Buprofezin	69327-76-0	76	0	0,013	0,014	0,0015	6	0,56	Indicatief MTR (o	RIVM (2018)	0,025	0,024	0,000	0,000	0,00	0,00
	Ciprofloxacin	85721-33-1	76	0	0,016	0,020	0,0002	6	0,089	EQS-proposal	WL substance do	0,225	0,181	0,000	0,000	0,00	0,00
	Citalopram	59729-32-7	76	0	0,014	0,032	0,0001	6	10	PNEC chronic	Aquire 167215	0,003	0,001	0,000	0,000	0,00	0,00
	Clarithromycin	81103-11-9	76	0	0,025	0,066	0,0002	6	0,12	EQS-proposal	WL substance do	0,547	0,212	0,000	0,000	0,00	0,00
	Diazepam	439-14-5	76	0	0,006	0,036	0,0004	6	0,291	PNEC chronic	Aquire 167736	0,122	0,022	0,000	0,000	0,00	0,00
	Dimethoate	60-51-5	76	0	0,049	0,069	0,0030	6	0,07	JD-UQN	UBA (2016) Oberf	0,989	0,697	0,000	0,000	0,00	0,04
	Erythromycin	114-07-8	76	0	0,012	0,019	0,0004	6	0,2	EQS-proposal	WL substance do	0,093	0,059	0,000	0,000	0,00	0,00
	Gemfibrozil	25812-30-0	76	0	0,193	0,303	0,0001	6	0,5	PNEC chronic	Aquire 168263	0,605	0,386	0,000	0,000	0,00	0,00
	Hexythiazox	78587-05-0	76	0	0,019	0,024	0,0010	6	0,025	Indicatief MTR (o	RIVM (2018)	0,974	0,753	0,000	0,000	0,00	0,04
	Imazalil / 1-[2-(al	35554-44-0	76	0	0,281	0,683	0,0009	6	0,87	Indicatief MTR (o	RIVM (2018)	0,785	0,323	0,000	0,000	0,00	0,00
	Indomethacin	53-86-1	76	0	0,049	0,137	0,0003	6	1	PNEC chronic	Aquire 164572	0,137	0,049	0,000	0,000	0,00	0,00
	Irbesartan	138402-11-6	76	0	0,068	0,696	0,0001	6	704	AA-EQS	OZ (2013) EQS Do	0,001	0,000	0,000	0,000	0,00	0,00
	Losartan	114798-26-4	76	0	0,124	0,221	0,0003	6	78	PNEC chronic	UBA (2017) EQS d	0,003	0,002	0,000	0,000	0,00	0,00
	Methyl-1H-benz	29385-43-1	76	0	1,508	7,018	0,0000	6	8	PNEC aqua (fresh	ECHA DOSSIER (03	0,877	0,189	0,000	0,000	0,00	0,00
	Naproxen	22204-53-1	76	0	0,087	0,289	0,0006	6	1,7	AA-EQS	OZ (2015) EQS Do	0,170	0,051	0,000	0,000	0,00	0,00
	Octylphenol diet	9002-93-1	75	0	0,016	0,033	0,0000	6	3,5	PNEC acute	Aquire 854	0,009	0,005	0,000	0,000	0,00	0,00
	Paroxetine	61869-08-7	76	0	0,002	0,003	0,0005	6	20	PNEC chronic	Aquire 80408	0,000	0,000	0,000	0,000	0,00	0,00
	Perfluorobutane	375-73-5	76	0	0,025	0,228	0,0001	6	372	AA-EQS	Italian EQS Work	0,001	0,000	0,000	0,000	0,00	0,00
	Phenazone	60-80-0	76	0	0,013	0,041	0,0001	6	1,1	PNEC acute	UBA (2014) EQS d	0,037	0,012	0,000	0,000	0,00	0,00
	Prochloraz	67747-09-5	76	0	0,081	0,084	0,0060	6	0,2	PNEC chronic	Footprint (2018)	0,418	0,406	0,000	0,000	0,00	0,03
	Propranolol	525-66-6	76	0	0,004	0,012	0,0001	6	0,411	PNEC chronic	Aquire 160503	0,029	0,009	0,000	0,000	0,00	0,00

	Propyphenazone	479-92-5	76	0	0,024	0,103	0,0001	6	0,8	PNEC acute	Aquire 40278	0,128	0,030	0,000	0,000	0,00	0,00
	Sulfamethoxazol	723-46-6	76	0	0,019	0,042	0,0003	6	0,6	AA-EQS	OZ (2010) EQS Do	0,069	0,032	0,000	0,000	0,00	0,00
	Thiabendazole	148-79-8	76	0	0,076	0,129	0,0004	6	3,3	MTR (opgelost)	RIVM (2018)	0,039	0,023	0,000	0,000	0,00	0,00
	Trimethoprim	738-70-5	76	0	0,010	0,150	0,0003	6	120	AA-EQS	OZ (2015) EQS Do	0,001	0,000	0,000	0,000	0,00	0,00
	Tris(2-butoxyeth	78-51-3	76	0	0,259	0,659	0,0000	6	24	PNEC aqua (fresh	ECHA DOSSIER (0	0,027	0,011	0,000	0,000	0,00	0,00
	Valsartan	137862-53-4	76	0	0,217	0,699	0,0002	6	560	AA-EQS	OZ (2016) EQS Do	0,001	0,000	0,000	0,000	0,00	0,00
x	Caffeine	58-08-2	76	4	0,990	3,237	0,0001	1/6	1,2	JD-UQN proposal	UBA (2017) Draft	2,697	0,825	0,000	0,053	0,05	0,00
x	Chlorpyrifos	2921-88-2	76	4	0,025	0,041	0,0006	1/6	0,03	EQS chronic wate	DIRECTIVE 2013/	1,374	0,835	0,000	0,053	0,05	0,02
x	Venlafaxine	93413-69-5	76	3	0,030	0,128	0,0001	1/6	0,038	EQS-proposal	WL substance do	3,358	0,782	0,000	0,039	0,04	0,00
x	Azithromycin	83905-01-5	76	2	0,008	0,154	0,0002	1/6	0,019	EQS-proposal	WL substance do	8,091	0,418	0,000	0,026	0,03	0,01
x	Carbamazepine	298-46-4	76	2	0,030	0,065	0,0000	1/6	0,05	PNEC chronic	Aquire 152195	1,309	0,592	0,000	0,026	0,03	0,00
x	Chlorfenvinphos	470-90-6	76	2	0,096	0,107	0,0006	1/6	0,1	EQS chronic wate	DIRECTIVE 2013/	1,069	0,957	0,000	0,026	0,03	0,01
x	4-nonylphenol	84852-15-3	65	1	0,258	0,391	0,0000	1/6	0,3	EQS chronic wate	DIRECTIVE 2011/	1,303	0,858	0,000	0,015	0,02	0,00
x	Imidaclopride	138261-41-3	76	1	0,006	0,052	0,0002	1/6	0,0083	EQS-proposal	WL substance do	6,246	0,671	0,000	0,013	0,01	0,02
x	Perfluorooctano	335-67-1	76	1	0,116	0,189	0,0001	1/6	0,178	PNEC chronic	Aquire 175220	1,060	0,654	0,000	0,013	0,01	0,00
x	Tris(2-chloroeth	115-96-8	76	1	0,778	6,377	0,0001	1/6	4	PNEC chronic	ETOX UBA (2018)	1,594	0,195	0,000	0,013	0,01	0,00

Table IV.2. Finally selected Iberian River Basin Specific Pollutants.

Substance	CAS No.	No. of sites	# of sites where MECsite > PNEC	Max exceedance	Extent of Exceedence	Score EoE	Score FoE	Score Total	Compound Class
17-beta-Estradiol	50-28-2	76	52	19,433	16,686	0,200	0,684	0,88	Horm
Pyriproxyfen / 2-(1-methyl-2-(4-phenoxy-phenoxy)-ethoxy)-pyridine	95737-68-1	76	46	66,393	59,713	0,200	0,605	0,81	Pest
Diclofenthion	97-17-6	76	33	13,368	12,215	0,200	0,434	0,63	Pest
Perfluorooctane sulfonate (PFOS)	1763-23-1	76	30	4167,249	85,237	0,200	0,395	0,59	Ind
Ibuprofen	15687-27-1	76	18	86,782	17,724	0,200	0,237	0,44	Pharm
Diazinon	333-41-5	76	13	2,375	1,255	0,100	0,171	0,27	Pest
Estrone	53-16-7	76	13	2,040	1,705	0,100	0,171	0,27	Horm
Diclofenac	15307-86-5	76	8	5,600	2,187	0,100	0,105	0,21	Pharm
Bisphenol A	80-05-7	76	5	3,247	1,249	0,100	0,066	0,17	Ind
Lorazepam	846-49-1	76	4	3,184	1,230	0,100	0,053	0,15	Pharm
Iopromide	73334-07-3	76	3	9,572	2,287	0,100	0,039	0,14	Pharm
Parathion	56-38-2	76	8	8,384	0,000	0,000	0,105	0,11	Pest
Malathion	121-75-5	76	6	53,392	0,000	0,000	0,079	0,08	Pest
Echio (Ethion)	563-12-2	76	6	50,583	0,000	0,000	0,079	0,08	Pest
Ofloxacin	82419-36-1	76	6	5,214	0,000	0,000	0,079	0,08	Pharm
Azinphos-ethyl	2642-71-9	76	5	3,118	0,000	0,000	0,066	0,07	Pest
Caffeine	58-08-2	76	4	2,697	0,825	0,000	0,053	0,05	Pharm
Chlorpyrifos	2921-88-2	76	4	1,374	0,835	0,000	0,053	0,05	Pest
Octocrylene	6197-30-4	76	3	1,174	0,000	0,000	0,039	0,04	PCP
Omethoate	1113-02-6	76	3	2,928	0,000	0,000	0,039	0,04	Pest
Venlafaxine	93413-69-5	76	3	3,358	0,782	0,000	0,039	0,04	Pharm
Azinphos-methyl	86-50-0	76	2	1,337	0,000	0,000	0,026	0,03	Pest
Fenitrothion	122-14-5	76	2	5,266	0,000	0,000	0,026	0,03	Pest

Triclocarban	101-20-2	76	2	3,079	0,000	0,000	0,026	0,03	PCP
Azithromycin	83905-01-5	76	2	8,091	0,418	0,000	0,026	0,03	Pharm
Carbamazepine	298-46-4	76	2	1,309	0,592	0,000	0,026	0,03	Pharm
Chlorfenvinphos	470-90-6	76	2	1,069	0,957	0,000	0,026	0,03	Pest
4-nonylphenol	84852-15-3	65	1	1,303	0,858	0,000	0,015	0,02	Ind
Metolachlor	51218-45-2	76	1	2,235	0,000	0,000	0,013	0,01	Pest
4-Methylbenzylidene camphor	36861-47-9	76	1	1,012	0,679	0,000	0,013	0,01	PCP
Perfluorodecanoic acid	335-76-2	76	1	1,287	0,307	0,000	0,013	0,01	Ind
17-alpha-Ethinylestradiol	57-63-6	76	1	63,229	0,000	0,000	0,013	0,01	Horm
Fenthion sulfone	3761-42-0	76	1	1,439	0,000	0,000	0,013	0,01	Pest
Sertraline	79617-96-2	76	1	1,585	0,000	0,000	0,013	0,01	Pharm
Imidaclopride	138261-41-3	76	1	6,246	0,671	0,000	0,013	0,01	Pest
Perfluorooctanoic acid	335-67-1	76	1	1,060	0,654	0,000	0,013	0,01	Ind
Tris(2-chloroethyl) phosphate	115-96-8	76	1	1,594	0,195	0,000	0,013	0,01	Ind