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► **To cite this version:**

Javier Castro-Jiménez, Daniela Bănaru, Chia-Ting Chen, Begoña Jiménez, Juan Muñoz-Arnanz, et al.. Persistent Organic Pollutants Burden, Trophic Magnification and Risk in a Pelagic Food Web from Coastal NW Mediterranean Sea. *Environmental Science and Technology*, 2021, 55 (14), pp.9557-9568. 10.1021/acs.est.1c00904 . hal-03177288v2

HAL Id: hal-03177288

<https://amu.hal.science/hal-03177288v2>

Submitted on 12 May 2023

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1 **Persistent organic pollutants burden, trophic magnification and risk in a pelagic**
2 **food web from coastal NW Mediterranean Sea**

3
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23 **Abstract**

24 The storage capacity, trophic magnification and risk of sixty-two POPs have been
25 evaluated in a well-characterized pelagic food web (eleven trophic levels) from an
26 impacted area in NW Mediterranean Sea. Our results show the high capacity of the
27 planktonic compartment for the storage of polybrominated diphenyl ethers (PBDEs) and
28 polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs), consistent with their
29 estimated low trophic magnification factors (TMF) of 0.2-2.0 (PBDEs) and of 0.3-1.1
30 (PCDD/Fs). Σ PBDEs dominated in the zooplankton size-class 200-1000 μ m (~330 ng g⁻¹
31 lw, median), whereas Σ PCDD/Fs accumulated preferentially in phytoplankton size-
32 class 0.7-200 μ m (875 pg g⁻¹ lw, median). In contrast, polychlorinated biphenyls (PCBs)
33 were preferentially bioaccumulated in the higher trophic levels (six fish species and two
34 cephalopods) with TMFs=0.8-3.9, reaching median concentrations of 4270 and 3140 ng
35 g⁻¹ lw (Σ PCBs) in Atlantic bonito (*Sarda sarda*) and chub mackerel (*Scomber colias*),
36 respectively. For these edible species, the estimated weekly intakes of dioxin-like POPs
37 for humans based on national standards overpassed the EU tolerable weekly intake.
38 Moreover, the concentrations of non-dioxin like PCBs in *S. sarda* were above the EU
39 maxim levels in foodstuffs, pointing to a risk. No risk evidence was found due to
40 consumption of all other edible species studied, also for PBDEs. The integrated burden
41 of POPs in the food web reached ~ 18 μ g g⁻¹ lw, representing a dynamic stock of toxic
42 organic chemicals in the study area. We show that the characterized food web could be a
43 useful and comprehensive ‘bioindicator’ of the chemical pollution status of the study area,
44 opening new perspectives for the monitoring of toxic chemicals in NW Mediterranean
45 coastal waters.

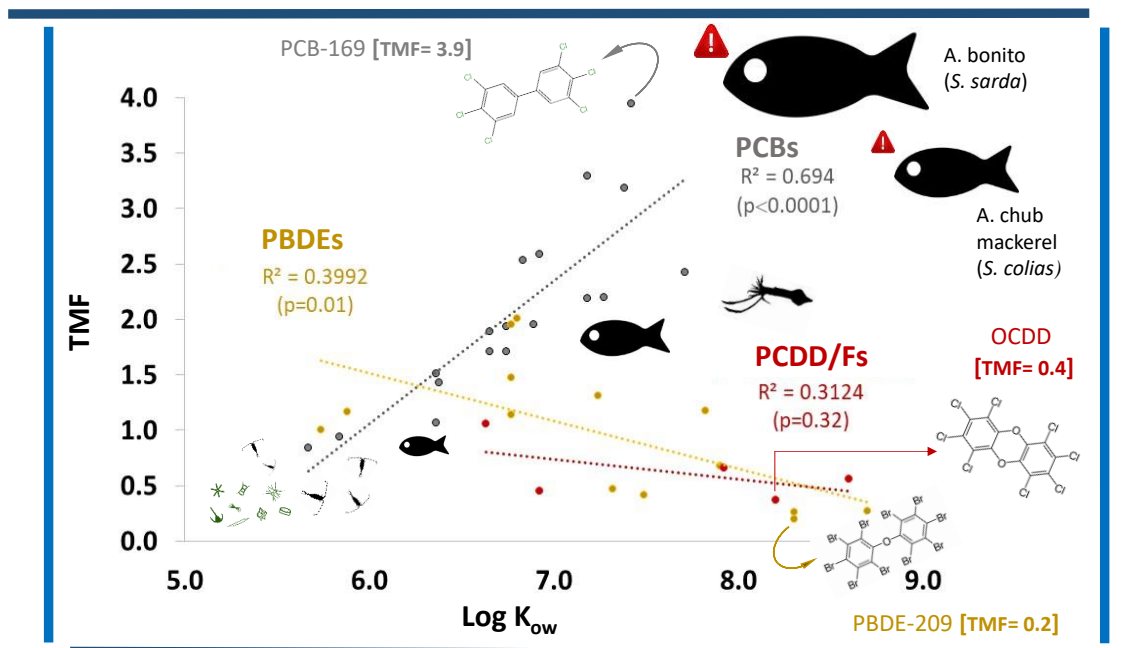
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47 **Keywords:** Environmental exposure, bioaccumulation, flame retardants, dioxins, PCBs,
48 fish, cephalopods, plankton
49
50 **Synopsis:** The integrated burden of POPs in a pelagic food web (eleven trophic levels) in
51 coastal NW Mediterranean Sea reached $\sim 18 \mu\text{g g}^{-1}$ lw. A potential risk for humans mostly
52 due to exposure to dioxin-like POPs *via* consumption of Atlantic bonito and chub
53 mackerel was found.

54
55

56 Abstract Graphic

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63 1. Introduction

64

65 The chemical contamination of the marine environment is a global and transversal issue
66 with important ecological and socio-economic implications, yet not well quantified nor
67 efficiently managed.^{1,2} Populations living in coastal areas, as well as marine organisms,

68 are exposed to multiple sources of chemical contamination, particularly in semi-enclosed
69 environments such as the Mediterranean Sea. The Mediterranean is a sensitive
70 environment with the particularity of being a hotspot of both marine biodiversity (up to
71 18% of the world's marine diversity) and chemical contamination (e.g. organic and
72 inorganic contaminants, plastic debris).³⁻⁵ Among the contaminants impacting this
73 marine environment, persistent organic pollutants (POPs) and related chemicals are well
74 documented in non-biotic compartments⁶⁻¹⁰ and different marine organisms.^{5,8,11-17} POPs
75 are organic contaminants of major concern due to their high toxicity, long-range transport
76 and bioaccumulation potential, and their high resistance to degradation.¹⁸ Even if these
77 toxic chemicals have been banned by international regulations, such as the Stockholm
78 Convention on POPs (SC), their ubiquity in the marine environment many years after
79 their release may still pose potential risks.

80 Polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs), polychlorinated
81 biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) are widespread POPs
82 with different environmental sources. While PCDD/Fs can occur as unintentional by-
83 products in a number of industrial processes and domestic heating^{19,20}, PCBs are legacy
84 industrial chemicals used in many applications (e.g. lubricants, transformers, oil
85 additives).²¹ Major primary sources for these two families of legacy POPs are considered
86 under control in most parts of the world today.²² However, ongoing primary releases from
87 diffuse sources such as open fires and uncontrolled waste incineration, urban/industrial
88 centres, as well as secondary sources such as net re-emissions from environmental
89 reservoirs are still possible.^{21,22} PBDEs were mostly used (and are still in some regions)
90 as flame retardants²³ and, in contrast to PCDD/Fs and PCBs, have been only recently
91 restricted at global scale by the SC.²⁴ These chemicals constitute complex mixtures in the
92 environment to which marine organisms and humans are exposed. Their accurate

93 determination in the biotic compartment represents a very important first step to
94 apprehend potential risks. However, despite the fact that POPs are often measured in
95 organisms from different trophic levels (TLs), these organisms do not always belong to
96 the same food web, making it difficult to properly investigate their trophic magnification,
97 subsequent risks and more generally the environmental status of marine waters, as
98 requested by the European Marine Strategy Framework Directive.²⁵

99

100 Here, we report results for PCDD/Fs, PCBs, and PBDEs in 26 biotic samples consisting
101 of phyto- and zooplankton, six fish species and two cephalopods, all belonging to the
102 same trophic web from an impacted coastal area, the Bay of Marseille, in the NW
103 Mediterranean Sea. The overall aims of this work were: (1) to characterize a pelagic food
104 web representative of the coastal NW Mediterranean, including species consumed by
105 humans, which could help the monitoring and assessment of chemical pollution in the
106 area; (2) to determine the burden and storage capacity of a cocktail of 62 toxic organic
107 chemicals in the selected food web; (3) to assess their potential trophic magnification,
108 exposure to humans and risks *via* consumption of fishery products.

109

110 **2. Materials and Methods**

111

112 *2.1. Study area*

113 Samples were collected in the Bay of Marseille in the south of France (NW Mediterranean
114 Sea) (Figure S1). This bay is highly impacted by multiple contamination sources, such as
115 the Marseille sewage treatment plant, with its outlet (WWTP Cortiou) located around 10
116 km south-east of the sampling site,²⁶ and the port, industry, agriculture and tourism, in
117 particular. The Bay of Marseille is naturally influenced by high solar radiation,²⁷

118 intrusions of the Northern Current on the shelf,²⁸ and occasionally by intrusions of the
119 Rhone river plume and the coastal Huveaune River, giving rise to episodic high inputs of
120 particle and dissolved organic matter.²⁹⁻³¹ Inputs of a wide diversity of POP families and
121 related organic contaminants of emerging concern have been reported in the area coming
122 from the Rhone River,³²⁻³⁶ the Marseille WWTP at Cortiou and the Huveaune,^{35,37-39}
123 and the atmosphere.¹⁰ Marine organisms and pelagic food webs in the Bay of Marseille
124 are certainly exposed to those contaminants (Figure S1).

125

126 2.2. Sampling

127 Phyto- and zooplankton samples were collected at the Solemio station (43.24°N; 5.29°E,
128 Figure S1) on 15th March 2017 on board the RV ANTEDON II. Phytoplankton and
129 suspended particulate matter (mostly microphytoplankton and detritus particles) were
130 obtained by filtering ~60 L of surface water collected with an inner lining Teflon GoFlo
131 12 L bottle through ten pre-combusted 47mm glass fiber filters (GFFs, 0.7 µm) and
132 treated as one sample. Zooplankton was vertically sampled, from 0 to 50 m depth, using
133 a zooplankton net of 150 µm mesh, and then sequentially separated by using stainless
134 steel sieves. Due to the limited amount of sampled material, the various fractions were
135 pooled together into two overall size classes ('Z1': 200-1000µm, and 'Z2': >2000 µm)
136 for POPs analyses. However, all zooplankton size-classes were considered for the food
137 web structure determination (Figure S2). Their groups and species as well as biochemical
138 composition are detailed elsewhere.⁴⁰ Teleosts and cephalopods were sampled by
139 professional fishers in the Bay of Marseille near Solemio and specimens were acquired
140 in the Marseille fish market on the same day. Sampled species were: (1) the planktivorous
141 fish bogue (*Boops boops*, Linnaeus, 1758); (2) the mesopredators *Trachurus trachurus*
142 (Linnaeus, 1758), *Trachurus mediterraneus* (Steindachner, 1868), *Scomber colias*

143 (Gmelin, 1789), the European hake (*Merluccius merluccius*, Linnaeus, 1758), and the
144 cephalopod squid *Illex coindetii* (Vérany, 1839), *Todarodes sagittatus* (Lamarck, 1798),
145 and (3) the top predator Atlantic bonito (*Sarda sarda*, Bloch, 1793). All these fish and
146 cephalopods are commonly caught by local fisheries and consumed in the region. Three
147 individuals of different size were considered by species, except for the cephalopod *T.*
148 *sagittatus* for which only one specimen was available. In the laboratory, the fresh
149 organisms were cleaned, measured and weighed (Table S1). Plankton analyses were made
150 on the entire individuals, while dorsal muscle was sampled for teleosts and mantle for
151 squid. Dissection was done on a clean glass table strictly avoiding contact with plastics
152 or any other material which could be a source of organic matter. All the samples were
153 then frozen at -20°C, freeze-dried and ground into a homogenized fine powder prior to
154 analyses (see text S1 for details). We acknowledge that the overall contaminant transfer
155 in the food web will be dictated by the consumption of the entire prey individuals in the
156 ecosystem. However, the choice of the sampling in this study concerned only the muscle
157 for cephalopods and fishes as generally represents the major contaminant exposure route
158 for humans, therefore enabling a risk assessment (section 3.5). Further analyses in higher
159 lipid-rich organs such as the liver and gonads may reveal a higher total POP burden and
160 a potentially higher biomagnification.

161

162 2.3. Stable isotope and biochemical analyses

163 Three replicates were performed on phytoplankton and zooplankton size classes, while
164 the three individuals of teleosts and cephalopods were considered as replicates for $\delta^{13}\text{C}$,
165 $\delta^{15}\text{N}$, %C, %N, and total lipid analyses. Stable isotope measurements were performed
166 with a continuous-flow isotope-ratio mass spectrometer coupled to an elemental analyzer
167 in LIENSs Laboratory, La Rochelle University (France). Details for total lipid

168 determinations are presented in Text S1. For fish samples, $\delta^{13}\text{C}$ values were corrected for
169 the effects of lipids when the $\text{C/N} > 3.5$.⁴¹ TLs were estimated using the zooplankton 200-
170 300 μm size class (composed mostly of small herbivorous copepods) as 'baseline' with
171 trophic position defined as 2.^{40,42} A classical enrichment factor of 3.4‰ between prey
172 and predator was applied except for the top predator *S. sarda* for which a value of 2.4‰
173 was used⁴³ (See Text S1 for all details).

174

175 2.4. POPs analysis

176 All samples were freeze-dried and zooplankton and fish samples homogenized by using
177 a ball mill mounting stainless steel cells. The corresponding amounts of sample (0.14-
178 0.25 g dw, Table S1) were spiked with ^{13}C -labeled standards of PCDD/Fs, PCBs, and
179 PBDEs prior to Soxhlet extraction (24h) with a n-hexane: DCM (9:1) mixture. Extracts
180 were rota-evaporated and cleaned-up using the automated sample preparation system
181 DEXTech+.¹⁴ The six indicator PCBs (IDPCBs), the eight mono-*ortho* dioxin-like PCBs
182 (DLPCBs) and PBDEs (27 congeners) were collected in a first fraction, while the
183 seventeen 2,3,7,8-substituted PCDD/Fs along with the four non-*ortho* DLPCBs were
184 obtained in a second fraction, summing up a total of 62 POPs. Final extracts were rota-
185 evaporated to ~ 1 mL, transferred to vials, and dried under a gentle nitrogen steam.
186 Fractions were reconstituted in a few microliters of the respective PCDD/F, PCB, and
187 PBDE ^{13}C -labeled injection standards prior to instrumental analysis. Quantification was
188 carried out by isotopic dilution according to 1613 US EPA method⁴⁴ on a gas
189 chromatograph (Trace GC ultra, Thermo Fisher Scientific) coupled to a high-resolution
190 mass spectrometer (DFS, Thermo Fisher Scientific). Details can be found in Text S1.

191 2.5. Quality assurance / quality control (QA/QC)

192 Field blanks for phytoplankton samples (n=2), consisting of baked GFFs which were
193 placed in the freeze-dryer and then processed with the samples, were performed.
194 Analytical laboratory blanks (n=6) accounting for all the sample processing steps were
195 also performed. After sampling, GFFs were wrapped in aluminium foil again and stored
196 in the dark at -20°C. Laboratory blank levels were generally low for the three POP classes
197 compared to their concentrations in the samples, averaging from not detected (n.d) to 25
198 pg (PCBs), from n.d. to 0.13 pg (PCDD/Fs), and from n.d. to 7 pg (PBDEs) depending
199 on the congener and sample (Table S3). Laboratory blanks showed lower or similar levels
200 to field blanks, so no contamination occurred during sampling, storage and analysis.
201 Results were blank-corrected by subtracting the average field and laboratory blanks
202 values for phytoplankton and the rest of samples, respectively. Chromatographic peaks
203 were only considered when the ratio between the two monitored ions was within $\pm 15\%$
204 of the theoretical value, and the signal-to-noise (S/N) ratio > 3 (limit of detection, LOD).
205 Limits of quantification (LOQs) corresponded to $S/N \geq 10$. Calibration curves were daily
206 checked. Median LODs (n=26) ranged from 0.2 to 2.3 pg (PCBs), from 0.1 to 0.8 pg
207 (PCDD/Fs) and from 0.3 to 14.0 pg (PBDEs) depending on the compound and sample
208 (Table S4). Median method recoveries varied from 60 to 111 % (PCBs), from 79 to 102
209 % (PCDD/Fs) and from 61 to 102 % (PBDEs) (Table S5). Results were corrected by
210 recoveries.

211 2.6. *Data analysis*

212 Non-parametric tests (Kruskal–Wallis and Wilcoxon rank-sum) were used in order to
213 investigate significant differences among pollutant levels. The effect of multiple
214 comparisons was considered by applying the false discovery rate method. Non-detected
215 values were imputed to the LOD/2 to run these tests (Text S2 and Table S6). The software
216 employed was STATA/SE 16.1. However, the concentration sums presented in sections

217 3.3.1 and 3.3.2 consider only the detected values, and the concentrations expressed as
218 World Health Organization-toxic equivalents (WHO₀₅-TEQ) presented in section 3.5 for
219 DLPOPs are reported as upper bound values (i.e. the n.d values were substituted by the
220 corresponding LODs before the blank correction) to facilitate their comparison with
221 available threshold values. The sample BB-1 was excluded from the data set for PBDEs
222 due to suspected specific contamination. Trophic magnification factors (TMFs) for
223 individual POPs were calculated using the slope (m) of the linear regression between the
224 \log_{10} of the lipid-normalized contaminant concentration in the food web organisms and
225 their corresponding TLs (calculated as indicated above)^{45, 46} (section 3.4.). For PBDEs
226 and PCDD/Fs, which exhibited more n.d values, only congeners with >30% detection
227 frequency and detected in at least 7 different TLs were considered for the TMF
228 calculation. In order to derive the estimated weekly intake (EWI) of DLPOPs and Σ PBDEs
229 (section 3.5.), the weekly consumption data for France for demersal (*M. merluccius*) and
230 pelagic (rest of species) fish and cephalopod (222, 132 and 9.2 g/week, respectively)⁴⁷
231 were multiplied by the WHO₀₅-TEQ (PCDD/Fs + DLPCBs) and the Σ PBDEs (only
232 detected values) concentration, respectively, measured in the corresponding species and
233 divided by 70 Kg (average human bodyweight, bw), as previously described.¹⁵ Risk
234 quotients (R), were calculated as the concentration in the sample, and its corresponding
235 EWI, to threshold values, legally binding or not, derived by international authorities to
236 protect humans consuming fishery products and aquatic ecosystems.

237

238 **3. Results and discussion**

239

240 **3.1. Food web characterization**

241 The stable isotope ratios of the species and size classes analysed in the Bay of Marseille
242 pelagic food web are presented in Figure S2A and Table S7. Both $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ are

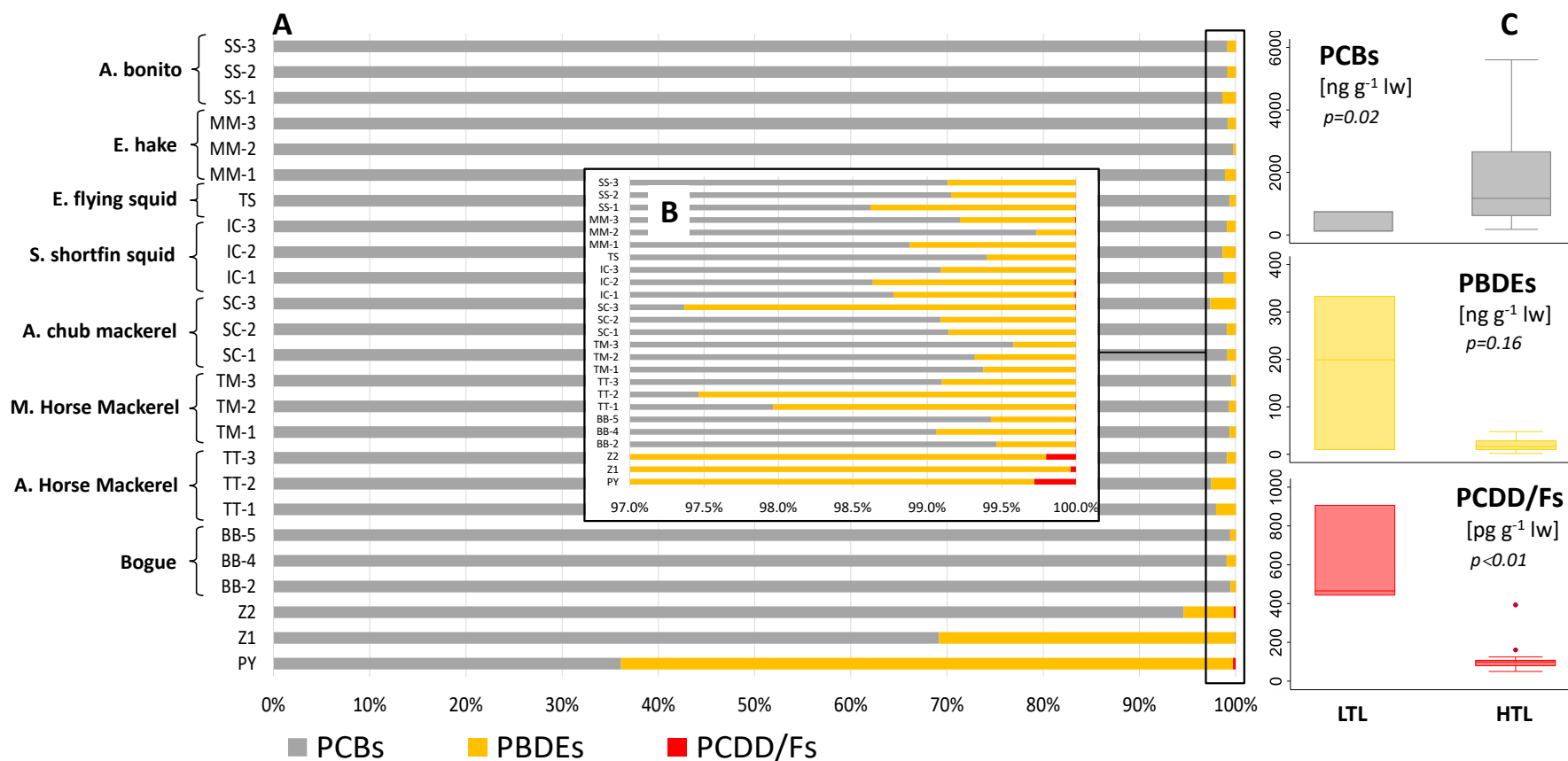
243 significantly different between species and/or size classes (Kruskal-Wallis, $p < 0.0001$).
244 Zooplankton $> 2000 \mu\text{m}$ mainly constituted of gelatinous organisms has the lowest stable
245 isotope ratios related to their filter feeding regime based on small pico ($0.2\text{-}2 \mu\text{m}$) and
246 nano ($2\text{-}20 \mu\text{m}$) phytoplankton. These small size classes of phytoplankton are supposed
247 to be the main source of the zooplankton food web in the North-Western Mediterranean
248 area ⁴⁸ but they could not be sampled for this study in sufficient quantities for POPs
249 analyses. Particles issued from the Cortiou WWTP with low stable isotope ratios are also
250 susceptible to be consumed by the different zooplankton size classes and decrease their
251 stable isotope ratios during particulate rain or hydrodynamic conditions. ^{26,42} Micro-
252 phytoplankton size class may represent the prey of many planktivorous fish such as *B.*
253 *boops*, and all the juveniles of the studied fishes and squid. Herbivorous zooplankton 200-
254 $300 \mu\text{m}$ had also low stable isotope value and were considered as baseline for the
255 estimation of the TL of the other consumer groups. The $\delta^{15}\text{N}$ ratios increased between
256 low TL groups (phyto- and zooplankton) and high TL groups (fish and squid, $\text{TL} > 3$)
257 consistent with their TL position (Figure S2B/Table S7). Predators such as the fish *S.*
258 *sarda* and squid *T. sagittatus* showed the highest stable isotope ratios related to their
259 high TL (5.3 and respectively 4.2).

260

261 3.2. Distribution of POPs and storage capacity in the food web

262 Overall, PCBs exhibited a high detection frequency (DF) of 73-100% for most congeners
263 (Figure S3), and dominated total POP concentrations from zooplankton to top predators,
264 accounting for 69% to around 100% of the ΣPOPs (Figure 1A). PBDEs showed a
265 generally lower DF with most congeners in the range 0-60%, although a certain number
266 of congeners such as BDEs-28, -47, -49, -71, -99, -100 and -154 were highly detected
267 (DF= 84-100%). PBDEs dominated only in phytoplankton, reaching 64% of total POPs

268 concentrations, while accounting for 0.3 – 31% of the total contaminant stock in the rest
269 of the samples. PCDD/Fs levels were below LODs in many cases and DF was mostly in
270 the range 0-40% and only 2,3,7,8-TCDF (88 %) and 1,2,3,4,6,7,8-HpCDD (96%) were
271 highly present. The contribution of PCDD/Fs to the total POPs burden in the studied food
272 web was minor, representing < 0.5% of the total POPs. These contaminants accumulated
273 preferentially in the lowest trophic levels (LTL) like PBDEs (Figure 1B). LTL,
274 considered as the planktonic compartment (i.e. phytoplankton 0.7-200 μm , zooplankton
275 200-1000 μm and zooplankton > 2000 μm), seemed to play a predominant role driving
276 the food web burden and storage capacity of Σ PBDEs and Σ PCDD/Fs exhibiting 12-fold
277 and 5-fold higher median concentration than in higher trophic levels (HTL), respectively
278 (Figure 1C). This trend was followed by individual PBDEs and PCDD/Fs too (Figure S4-
279 S5) showing similar or significantly higher ($p < 0.01-0.04$, Kruskal-Wallis) median
280 concentrations in LTL. In contrast, the Σ PCB burden in the food web may be dominated
281 mostly by HTL showing a significantly 9-fold higher median concentration ($p = 0.02$,
282 Kruskal-Wallis) than the LTL (Figure 1B). Individual PCB congeners generally followed
283 the same trend except for CBs-28 and -52, which exhibited similar or significantly higher
284 median concentrations ($p = 0.03$, Kruskal-Wallis) in LTL as PBDEs and PCDD/Fs (Figure
285 S6). In other words, the lipid-normalized cumulative concentrations of PBDEs and
286 PCDD/Fs in the phytoplankton and the two zooplankton size classes were similar or
287 significantly higher than the cumulative concentrations of the 21 fish and 2 cephalopod
288 individuals studied, pointing to the high capacity of the planktonic compartment for the
289 accumulation and storage of these two POPs families. In contrast, fishes and cephalopods
290 represent efficient biotic storage compartments for PCBs.

292
293

294 **Figure 1.** POPs relative burden in the food web (A) with a zoom showing the higher % of PCDD/Fs found in the planktonic compartment (B), and box plots (C) of the lipid
 295 normalized total concentrations of ΣPCBs , ΣPBDEs ($\text{ng g}^{-1} \text{lw}$) and $\Sigma\text{PCDD/Fs}$ ($\text{pg g}^{-1} \text{lw}$) in the lowest trophic levels (LTL) considered as the planktonic compartment for
 296 the purpose of this comparison (i.e. $\text{PY}=\text{phytoplankton } 0.7\text{-}200 \mu\text{m}$, $\text{Z1}=\text{zooplankton } 200\text{-}1000 \mu\text{m}$ and $\text{Z2}=\text{zooplankton } > 2000 \mu\text{m}$, $n=3$) and the higher trophic levels (HTL)
 297 considered as the other organisms from the food web (fishes and cephalopods, $n=22$) Kruskal-Wallis test p-value. $\text{BB}=\text{B. boops}$; $\text{TT}=\text{T. trachurus}$; $\text{TM}=\text{T. mediterraneus}$;
 298 $\text{SC}=\text{S. colias}$; $\text{IC}=\text{I. coindetii}$; $\text{TS}=\text{T. sagittatus}$; $\text{MM}=\text{M. merluccius}$; $\text{SS}=\text{S. sarda}$.

299 3.3. Environmental concentrations and patterns

300

301 3.3.1. Overall concentrations

302 $\sum_{18}\text{PCB}$ concentrations varied from 3.3 to 496 ng g⁻¹ dw (from 113 to ~5600 ng g⁻¹ lipid
303 weight - lw) in the different individuals from the studied food web. The total PCB load
304 was dominated by $\sum_{10}\text{PCBs}$, accounting for 83-92% of $\sum_{18}\text{PCBs}$ with concentrations
305 ranging from 3.0 to 454 ng g⁻¹ dw (~100 to 5130 ng g⁻¹ lw). $\sum_{\text{DL}}\text{PCBs}$ represented only
306 between 8 and 17% of the total PCBs with levels between 0.3 and 42 ng g⁻¹ dw (13 to 470
307 ng g⁻¹ lw). $\sum_{26}\text{PBDE}$ concentrations varied from 0.08 to 25 ng g⁻¹ dw (1.0 to 333 ng g⁻¹
308 lw) whereas three orders of magnitude lower $\sum_{17}\text{PCDD/F}$ concentrations were measured,
309 ranging from 0.3 to 110 pg g⁻¹ d.w (3.4 to 875 pg g⁻¹ lw). Results for individual PCBs,
310 PBDEs and PCDD/Fs in each sample are presented in Tables S8-10. In addition, in order
311 to facilitate data comparability, POPs concentrations are also expressed on a wet weight
312 (ww) basis (Table S11).

313

314 3.3.2. Levels and patterns in the different organisms

315 Although concentrations of a number of POPs, mostly PCBs and PBDEs, were found
316 significantly different among the different TLs in a first data analysis ($p < 0.0-0.04$,
317 Kruskal-Wallis) (Table S6), when comparing by pairs of TLs (Wilcoxon rank-sum test)
318 and correcting by the effect of multiple comparisons (section 2.6, Text S2), these
319 differences were not statistically significant. These analyses are certainly hampered by
320 the limited amount of data. However, some trends are worth discussing. Although some
321 bioaccumulation was observed in the 200-1000 μm zooplankton for the sum of PCBs, the
322 species exhibiting the highest median $\sum_{18}\text{PCB}$ concentration was the Atlantic bonito (*S.*
323 *sarda*) (4270 ng g⁻¹ lw), presenting the highest position in the food web (i.e TL=5.3),

324 followed by the Atlantic chub mackerel (*S. colias*) (3140 ng g⁻¹ lw). The European flying
 325 squid (*T. sagittatus*) and the Mediterranean horse mackerel (*T. mediterraneus*) were also
 326 among the highest concentrations with 2670 ng g⁻¹ lw (only one sample) and 2280 ng g⁻¹
 327 lw (Table 1). Similar trends were found for individual PCBs, except for CB-28 and -52
 328 which accumulated preferentially in the planktonic compartments with no clear trends in
 329 the other TLs (Figure S7). A consistent PCB pattern was observed in the whole food web,
 330 with PCB-153 as the most abundant congener in all TLs (27-42% of the median
 331 $\sum_{18}\text{PCBs}$), except for the TL 1 (phytoplankton) where PCB-101 and -52 were more
 332 abundant. The predominance of PCBs-101 and -52 has been reported in the dissolved
 333 water phase in other NW Mediterranean coastal areas.⁴⁹ Likely their preferential
 334 bioconcentration at the very first TLs could explain this differential pattern found in
 335 phytoplankton. PCB-118 was the most abundant DLPCBs contributing ~53 – 74% of the
 336 median $\sum_{\text{DL}}\text{PCBs}$ (Figure S8).

337

338 **Table 1.** Median concentrations (only detected values) of indicator, dioxin-like, and total PCBs ($\sum_{\text{ID}}\text{PCBs}$,
 339 $\sum_{\text{DL}}\text{PCBs}$, $\sum_{18}\text{PCBs}$, respectively), total PBDEs ($\sum_{26}\text{PBDEs}$) and total PCDD/Fs ($\sum_{17}\text{PCDD/Fs}$) in the
 340 different trophic levels of the food web under study. Values are expressed in dry, wet and lipid weights
 341 (dw, ww, lw, respectively). Concentrations are in ng g⁻¹ except for PCDD/Fs (pg g⁻¹).
 342

Median concentrations*	$\sum_{\text{ID}}\text{PCB}$			$\sum_{\text{DL}}\text{PCB}$			$\sum_{18}\text{PCB}$			$\sum_{26}\text{PBDEs}$			$\sum_{17}\text{PCDD/Fs}$		
	[ng g ⁻¹]			[ng g ⁻¹]			[ng g ⁻¹]			[ng g ⁻¹]			[pg g ⁻¹]		
Foodweb Species	dw	ww	lw	dw	ww	lw	dw	ww	lw	dw	ww	lw	dw	ww	lw
<i>Phytoplankton (0.7-200 μm)</i>	12.5	2.5	99.2	1.7	0.3	13.8	14.2	2.8	113.0	25.0	5.0	199.0	109.9	22.0	874.6
<i>Zooplankton (200-1000 μm)</i>	39.3	7.9	653.0	5.7	1.1	95.4	45.1	9.0	748.3	20.1	4.0	333.3	23.7	4.7	393.4
<i>Zooplankton (>2000 μm)</i>	3.0	0.6	112.7	0.3	0.1	12.9	3.3	0.7	125.6	0.2	0.04	6.9	7.0	1.4	265.9
<i>Boops boops</i>	38.4	9.2	629.4	6.2	1.5	102.2	44.7	10.7	732.3	0.5	0.1	5.3	2.1	0.5	28.8
<i>Trachurus trachurus</i>	55.7	11.1	858.5	5.6	1.1	79.2	61.3	12.3	937.6	1.0	0.2	19.5	0.7	0.1	9.2
<i>Trachurus mediterraneus</i>	180.2	36.0	2023.4	18.9	3.8	256.0	200.3	40.1	2279.4	0.8	0.2	15.6	0.9	0.2	16.2
<i>Scomber colias</i>	203.0	58.9	2734.2	19.1	5.5	286.4	222.1	64.4	3140.8	1.9	0.6	28.7	2.6	0.8	38.8
<i>Merluccius merluccius</i>	36.8	8.8	538.1	3.9	0.9	45.9	41.1	9.9	584.0	0.3	0.08	4.8	1.6	0.4	23.8
<i>Illex coindetii</i>	69.3	16.6	1229.4	8.1	2.0	155.5	77.5	18.6	1384.9	0.8	0.2	14.9	6.2	1.5	152.2
<i>Todarodes sagittatus</i>	181.8	43.6	2355.8	24.4	5.9	316.3	206.2	49.5	2672.1	1.2	0.3	16.1	6.5	1.6	84.5
<i>Sarda sarda</i>	444.0	106.5	3903.9	41.6	10.0	367.5	485.7	116.6	4271.4	4.2	1.0	37.1	3.9	0.9	43.3

343

*only one sample was available for phytoplankton, the two zooplankton size classes and the cephalopod *Todarodes sagittatus*

344

345 Median Σ_{26} PBDE concentrations clearly dominated in the 200-1000 μ m zooplankton
346 (~ 330 ng g⁻¹ lw) and phytoplankton (199 ng g⁻¹ lw), whereas Σ_{17} PCDD/Fs accumulated
347 preferentially in phytoplankton (875 pg g⁻¹ lw) followed by 200-1000 μ m zooplankton
348 (~ 390 pg g⁻¹ lw). The Σ PBDE predominance in zoo- and phytoplankton was driven by
349 the higher contribution of BDE-209 (70-75% of the detected Σ PBDEs). The high
350 Σ_{17} PCDD/F levels found in the LTL were driven by the higher abundance of
351 octachlorodibenzo-p-dioxin (OCDD) accounting for the 64-80% of the detected
352 Σ PCDD/Fs (Figure S9). Indeed, the predominant of OCDD/Fs in zooplankton has been
353 reported in other marine food webs.⁵⁰ Both BDE-209 and OCDD are highly hydrophobic
354 POPs with log octanol-water partition coefficients (log K_{ow}) of 8.7 and 8.2, respectively,
355 indicating high partition potential on hydrophobic components such as lipids or organic
356 carbon-rich particles. The combination of the following factors may explain this
357 observation: The 200-1000 μ m plankton size classes in the study area, mostly composed
358 of copepods, have been reported to have a high concentration of protein and lipid.⁴⁰
359 Moreover, the composition of smallest size class (0.7–200 μ m) is dominated by
360 microphytoplankton and detritus (mostly from the Marseille WWTP effluents and Rhone
361 inputs), and small copepod stages^{26,40,42}, providing active surfaces for the accumulation of
362 highly hydrophobic contaminants. A previous study showed important atmospheric
363 inputs of particle-bound PBDEs (~ 550 ng m⁻² y⁻¹) and PCDD/Fs (~ 15 ng m⁻² y⁻¹) in the
364 area, also with a clear predominance of BDE-209 and OCDD in the atmospheric particle
365 patterns.¹⁰ This atmospheric loading may constitute a relatively constant and direct source
366 of these congeners to surface waters, facilitating their way to the LTL of the food web.
367 Regarding HTLs, *S. sarda* and *S. colias* presented the highest Σ PBDE median
368 concentrations of 37 and 29 ng g⁻¹ lw, respectively, followed by *T. trachurus*, *T. sagittatus*
369 and, *T. mediterraneus* exhibiting median concentrations in the range 16-20 ng g⁻¹ lw

370 (Table 1). Although individual PBDEs generally followed this trend too (Figure S10),
371 some differences were found in the PBDE patterns. In the group of species showing
372 higher Σ PBDE median concentrations (i.e. *S. sarda*, *S. colias*, *T. trachurus*, *T.*
373 *mediterraneus*, and *T. sagittatus*), the PBDE pattern was characterized mostly by the
374 predominance of BDE-47 and -100 (32-46% and 13-31% of the detected Σ PBDEs,
375 respectively) and a lower contribution of PBDE-209 (4-16% of the detected Σ PBDEs)
376 (Figure S11A). The predominance of these congeners, particularly BDE-47, has been
377 reported for other marine organisms.^{14,17,51} Furthermore, the patterns in *M. merluccius*
378 and *I. coindetii*, two out of the three species showing the lowest median values, showed
379 a more even relative distribution of BDE-47 and BDE-209, accounting for the 24-32 and
380 32-39% of the detected Σ PBDEs, respectively (Figure S11B). These are piscivorous and
381 invertivorous species feeding mainly on pelagic species. Interestingly, the pattern found
382 in *B. boops* showed a similar BDE-209 contribution as *M. merluccius* and *I. coindetii* but
383 lower relative contribution of BDE-47 and higher % of BDE-99, resembling instead that
384 of phytoplankton and 200-1000 μ m zooplankton (Figure S11B,C). The fact that *B. boops*
385 is essentially zooplanktivorous⁴⁰ could partially explain this pattern. The 200-1000 μ m
386 zooplankton have been reported as the zooplankton class-size with the highest energy
387 content in the study area, making it preferable prey for zooplanktivorous fish such as the
388 bogue.⁴⁰ However, a differential metabolic capacity of *B. boops* could also contribute to
389 the observed pattern. Species-specific differences in metabolic rates of PBDE in fish has
390 been already reported.⁵² A higher median BDE-99/BDE-100 ratio (reported as an
391 indicator of fish capacity to metabolize PBDEs)^{49, 53} for *B. boops* (0.9) was observed in
392 comparison to the combined ratio for *M. merluccius* and *I. coindetii* (0.2), suggesting a
393 less efficient capacity of *B. boops* to metabolize PBDEs. For comparison, the Σ PBDE
394 concentrations in HTL organisms in this study (0.02-1.0 ng g⁻¹ww, Table S11B) were

395 similar or slightly lower to those reported in fish from other Mediterranean locations.
396 Values ranging from 0.07 to 3.4 ng g⁻¹ ww were measured in wild fish from the Adriatic,
397 Ionian and Tyrrhenian Seas (E Mediterranean)¹⁷ and average concentrations ranging from
398 ~1.0 to 3.9 ng g⁻¹ ww were reported for fish and seafood collected in Valencian markets
399 (Spain, NW Mediterranean)⁵⁴. The fact that the authors of these studies reported the upper
400 bound concentrations and that other fish species were also studied could explain an
401 important part of the concentration variability.

402
403 Regarding HTL, Σ PCDD/Fs accumulated preferentially in the cephalopods exhibiting
404 median concentrations of 152 and 84 pg g⁻¹ lw in the Southern shortfin squid (*I. coindetii*)
405 and European flying squid (*T. sagittatus*), respectively. The pattern in these organisms
406 was characterized by a predominance of OCDD (around 50% of the detected Σ PCDD/Fs)
407 and was quite similar to that of the planktonic compartment (Figure S12A). Individual
408 congeners followed the same concentrations trend, except for 2,3,7,8-TDCF which also
409 accumulated in *S. colias*, *B. boop*, *S. sarda* and *T. trachurus* (Figures S13), as reflected
410 too in their respective patterns (Figure S12B) with this congener accounting from ~60 to
411 76% (median) of the detected Σ PCDD/Fs. *M. merluccius* and *T. mediterraneus* showed a
412 comparable contribution of 1,2,3,4,6,7,8-HpCDD, 2,3,7,8-TDCF, and OCDFs
413 representing 30-40% of the detected Σ PCDD/Fs depending on the congener and species
414 (Figure S12C).

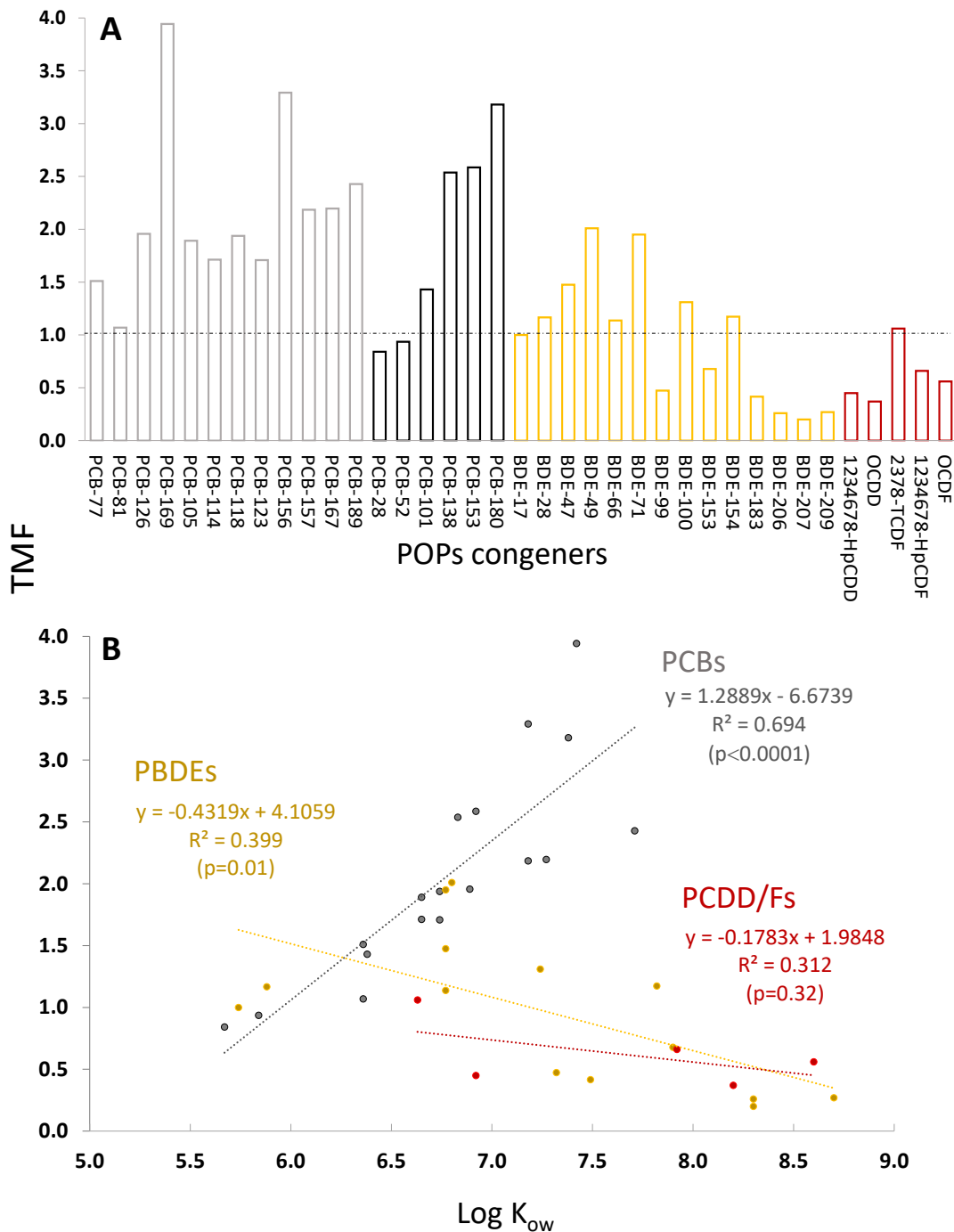
415

416 3.4. Trophic magnification

417

418 TMFs were calculated for the eighteen PCBs, fourteen PBDEs and five PCDD/Fs (Figure
419 S14, Table S12). The majority of POPs (60%) with calculated TMFs bioamplified

420 (TMF>1), 32% biodiluted (TMF<1) and the remaining 8% exhibited no apparent tendency
 421 (TMF~1)⁴⁵ (Figure 2A).



422

423 **Figure 2.** Trophic magnification factors (TMFs) for individual PCBs, PBDEs and PCDD/Fs (A) and their
 424 relationship with the POP's log K_{ow} (B) for the NW Mediterranean Sea food web. Only congeners with DF
 425 > 30% and expanding at least along 7 different TLs were considered. The horizontal dotted line indicates
 426 TMF=1. Light and dark grey, yellow and red indicates DLPCBs, IDPCBs, PBDEs and PCDD/Fs,
 427 respectively. The standard errors and confidence intervals of the slopes used to derive the TMFs can be
 428 consulted in Table S12.

429

430 PCBs showed the highest biomagnification potential (>80% congeners with $TMF > 1$) with
431 $TMFs$ ranging from 1.4 (PCB-101) to 3.9 (PCB-169). Our results indicate that even if
432 PCB-153, one of the most studied PCBs in aquatic organisms, exhibited a lipid-
433 normalized concentration of 4-5 orders of magnitude higher than PCB-169 and PCB-126
434 (Table S8), its estimated trophic magnification potential ($TMF_{CB-153} = 2.6$) is around 35%
435 weaker than that for PCB-169 and only 24% higher than that of PCB-126 ($TMF = 1.9$).
436 PCBs-126 and -169 are indeed the most toxic $DLPCBs$ (toxic equivalent factor, $TEFs = 0.1$
437 and 0.03, respectively).⁵⁵ This may have implications for the appropriate environmental
438 exposure assessment to the most toxic PCBs in the study area, which may be overlooked
439 by only studying the most abundant PCBs, which is often the case, and warrants further
440 research. Only PCB-28 ($TMF = 0.8$) and -52 ($TMF = 0.9$) did not biomagnify, consistent
441 with their higher abundance in the food web LTL (section 3.3.2). A significant positive
442 correlation ($R^2 = 0.69$, $p < 0.0001$) was found between the TMF_{PCBs} and the $\log K_{ow}$ (Figure
443 2B) confirming the general higher trophic magnification for the more hydrophobic PCBs
444 in this Mediterranean food web, as previously reported for other aquatic ecosystems⁴⁶.
445 This biomagnification potential is consistent too with the higher PCB concentrations
446 generally found in the Atlantic bonito (Tables 1, S7 and Figure S7).

447

448 In contrast to PCBs, only 50% of PBDE congeners biomagnified, exhibiting generally
449 lower $TMFs$ ranging from 1.1 (BDE-66) to 2.0 (BDE-49). The other congeners showed a
450 trophic dilution in the food web with $TMFs$ varying from 0.2-0.7 (Figure 2A, Table S12).
451 A negative significant correlation ($R^2 = 0.40$, $p = 0.01$) was found for TMF_{PBDEs} and the \log
452 K_{ow} indicating a lack of biomagnification of the heaviest PBDEs like BDE-206, 207 and
453 209 among others (Figure 2B), consistent with their higher predominance in the
454 planktonic compartment. It is worth mentioning that no or little trophic magnification

455 (0.5 < TMF < 1.5) was found for most of the PBDEs considered in the European Quality
456 Standard for PBDEs (i.e. BDE-28, -47, -99, -100, -153, -154) in fishes (EQS_{biota})⁵⁶.
457 Moreover, a generic biomagnification factor (BMF) of 5 is used for those PBDEs to cover
458 their biomagnification from lower trophic levels to fish species⁵⁷, which is much higher
459 than our field-derived TMFs. PCDD/Fs showed TMFs ranging from 0.4 to 1.1 indicating
460 mostly trophic dilution, with a non-significant negative correlation ($R^2=0.31$, $p=0.32$)
461 between $TMF_{PCDD/Fs}$ and $\log K_{ow}$. This lack of significant correlation with $\log K_{ow}$ has
462 also been reported in other aquatic food webs.⁵⁸ Similarly to PBDEs, a $BMF=10$ is
463 considered for PCDD/Fs and DL -PCBs to account for their biomagnification in aquatic
464 food webs⁵⁹, which is much higher than the TMFs measured in this study. These fact
465 raises the importance of using site and food web specific biomagnification values in order
466 to perform a more reliable site-specific risk assessment instead of applying generic default
467 values.

468

469 Overall, our results indicate that the POPs biomagnification in the studied food web is
470 strongly congener and species dependent. The fact that POPs congeners with similar K_{ow}
471 showed distinct bioaccumulation patterns, particularly the more hydrophobic PCBs,
472 PBDEs and PCD/Fs indicated that lipophilicity was not the only factor explaining their
473 trophic magnification. This differential behaviour has also been reported in other aquatic
474 food webs.⁵⁸ Indeed, the interplay of the specific contaminant bioavailability and
475 metabolic transformation may have driven their final biomagnification. The observed
476 trophic dilution of higher hydrophobic PBDEs in wild fish could be mostly attributed to
477 the metabolic debromination of HMW PBDEs as previously reported.⁵² However, the
478 biodilution of higher hydrophobic PCDD/Fs, exhibiting negligible metabolization rates

479 in most cases, may be better explained by a reduced membrane permeability due to their
480 larger molecular size compared to the lower chlorinated congeners.⁵⁰

481

482 Field-derived TMFs for the studied POPs in marine food webs are scarce in the literature
483 for comparison. In addition, TMFs in marine systems have been reported to be highly
484 variable and a number of factors such as type, structure and length of the food web and
485 the site study specificities and season of the year, may have an important influence on the
486 final TMF value^{60,61}. Therefore, the comparability of TMFs must be carefully interpreted.
487 TMFs of 1.3 and 1.2 were reported for PCB-153 and BDE-47, respectively, in a study
488 conducted in the Gulf of Lion during the period 2004-2006 and in a shorter foodweb⁶⁰.
489 These values are slightly lower than those calculated for the same contaminants in this
490 study ($TMF_{PCB-153} = 2.6$ and $TMF_{BDE-47} = 1.5$). The fact that we studied a longest food
491 web, including a higher top predator (*A. bonito*), could explain our higher values. TMFs
492 within the same range were reported for a pelagic food web in the western Scheldt estuary
493 (Netherlands)⁶² for BDE-28, -47, and -100. This work highlighted too the trophic dilution
494 of BDE-209 with a very similar TMF of 0.2. However, they found BDE-99
495 biomagnifying ($TMF=2.4$) in their studied food web, while a $TMF=0.5$ was found for the
496 same congener in our study. The reported predominance of BDE-99 in the western
497 Scheldt estuary organisms⁶², likely linked to historical local sources, could explain this
498 difference, since this congener was not predominant in our food web. TMFs for PCBs
499 and PBDEs within the same range were reported for a pelagic food web in coastal Atlantic
500 Ocean⁶³. However, our TMFs for these two families of POPs were generally lower than
501 the average (\pm standard deviation) TMFs reported for a global compilation also including
502 fresh water ecosystems, which ranged from 1.7 ± 1.1 to 8.1 ± 6.9 for PCBs and from 0.3
503 ± 0.3 to 3.2 ± 1.9 for PBDEs⁶¹, although a large variability was reported too (Figure S15).

504 To the best of our knowledge, we report here the first field-derived TMFs for PCDD/Fs
 505 in a Mediterranean food web, which are very similar to those reported in other aquatic
 506 ecosystems⁶¹ (Figure S15).

507

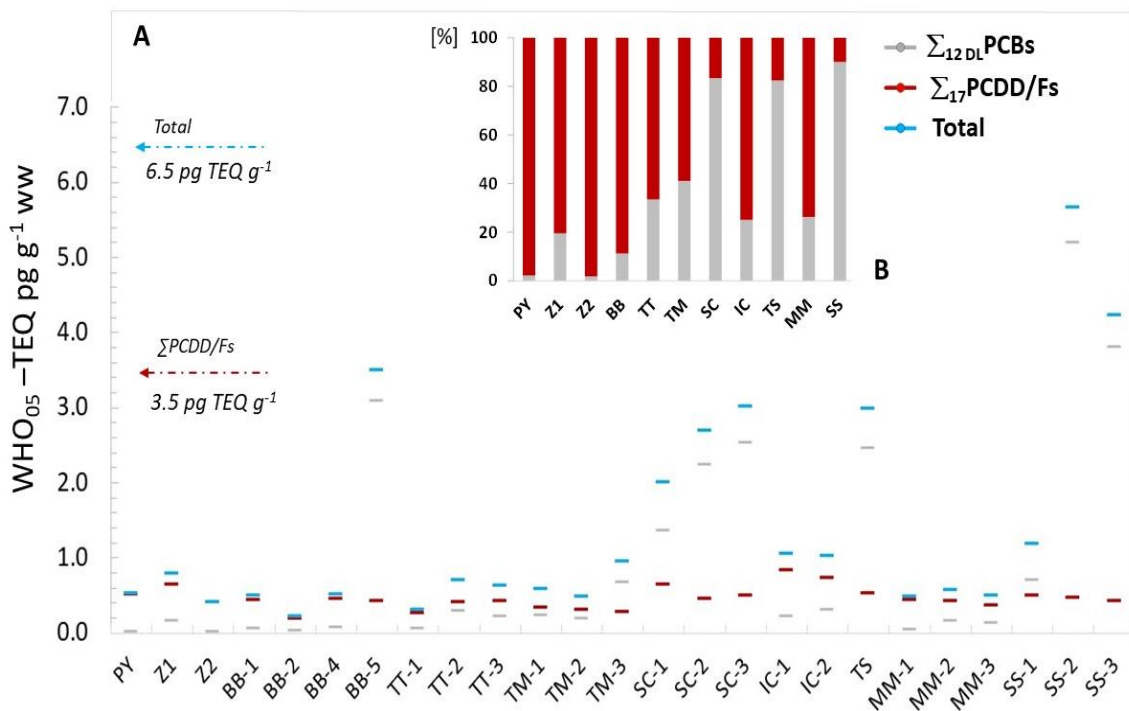
508 3.5.Toxicity potential and risk assessment

509

510 3.5.1. DLPOPs

511 Concentrations expressed as upper bound WHO₀₅-TEQ were derived for DLPCBs and the
 512 2,3,7,8-PCDD/Fs in the food web organisms using the latest available TEFs⁵⁵ in order to
 513 study their toxicity potential against humans. Concentrations ranged 0.008 - 5.2 and 0.2
 514 – 0.8 WHO₀₅ TEQ pg g⁻¹ ww for $\Sigma_{DL}PCBs$ and $\Sigma_{17}PCDD/Fs$, respectively, resulting in
 515 TEQ_{Total} (Σ_{29} dioxin-like POPs) of 0.2-5.7 pg g⁻¹ ww (Figure 3A, Table S13).

516



517

518 **Figure 3.** Calculated WHO₀₅-TEQ (pg g⁻¹ ww) in all samples from the pelagic food web in coastal NW
 519 Mediterranean Sea (A) and median relative contribution (%) of $\Sigma_{12DL}PCBs$ (grey) and $\Sigma_{17}PCDD/Fs$ (red)
 520 to the total WHO₀₅-TEQ (PCBs+PCDD/Fs) in blue (B), except for PY, Z1, Z2 and TS, which are single
 521 values. The horizontal dotted lines show the EU concentration thresholds in sea foodstuffs and wild fish
 522 (see text for details). PY=phytoplankton 0.7-200 μm ; Z1=zooplankton 200-1000 μm ; Z2=zooplankton >
 523 2000 BB=B. boops; TT=T. trachurus; TM=T. mediterraneus; SC=S. colias; IC=I. coindetii; TS=T.
 524 sagittatus; MM=M. merluccius; SS=S. sarda.

525

526 The species *S. sarda*, *T. sagittatus* and *S. colias* exhibited the highest median TEQ_{Total}
527 values of 4.2, 3.0, and 2.7 pg g⁻¹ ww, respectively (Table S13). These TEQs were driven
528 by the predominant contribution of DLPCBs (82-90%) in those species (Figure 3B).
529 Median TEQ_{Total} values in the remaining trophic levels varied from 0.4 to 1.1 pg g⁻¹ ww.
530 PCDD/Fs exhibited a generally higher contribution (53-98%) on these species, except for
531 some isolated samples such as BB-5 and TM-3, which show a higher concentration of
532 PCB-126, increasing the relative contribution of \sum_{DLPCBs} TEQ on those samples. The
533 \sum_{DLPCB} TEQ concentrations measured in the studied food web were within the range or
534 higher than those reported for other fish species in the Mediterranean. For example, levels
535 of 0.4-1.2 (*Sardina pilchardus*), 0.08-0.4 (*Engraulis encrasicolus*) and 0.07-0.27 (*B.*
536 *boops*) TEQ pg g⁻¹ ww have been recently reported.¹⁴ Our concentrations in fish samples,
537 excluding those of the top predator Atlantic bonito, ranged from 0.02 to 3.1 TEQ pg g⁻¹
538 ww for \sum_{DLPCB} , with values varying from 0.05 to 3.1 TEQ pg g⁻¹ ww for *B. boops*. Our
539 \sum_{DLPCB} TEQ values were also higher than those reported for chub mackerel (*Scomber*
540 *japonicas*) (0.002 -0.008 TEQ pg g⁻¹ ww) and horse mackerel (*T. trachurus*) (0.004 –
541 0.016 TEQ pg g⁻¹ ww) caught in the Adriatic Sea.⁶⁴ For comparison, our \sum_{DLPCB} levels
542 in *T. trachurus* varied from 0.05 to 0.3 TEQ pg g⁻¹ ww (Table S13). A recent study
543 measured dioxin-like POPs in Mediterranean bluefin tuna (*Thunnus thynnus*), reporting
544 0.7 and 1.9 TEQ pg g⁻¹ ww for \sum_{DLPCB} and $\sum_{PCDD/Fs}$, respectively.⁶⁵ Our results in
545 the Atlantic bonito ranged from 0.7 to 5.2 and from 0.4 – 0.5 TEQ pg g⁻¹ ww for \sum_{DLPCB}
546 and $\sum_{PCDD/Fs}$, respectively. However, the concentrations reported for the bluefin tuna
547 were lower bound values (non-detected values were considered as zero). These
548 comparisons are only indicative due to the associated variability related to the different
549 species considered, sampling year and season and the quantification strategies adopted in
550 each specific study.

551 At European level, two pieces of legislation, Commission Regulation 1259/2011⁶⁶ and
552 Directive 2013/39/EU⁵⁶, set a maximum level for DLPOPs in seafood and marine
553 organisms aiming at protecting human health *via* consumption of fishery products (but
554 also aquatic top predators from secondary poisoning for the EQS_{biota}). A threshold level
555 of 6.5 pg TEQ g⁻¹ ww for the $\sum_{17}\text{PCDD/Fs} + \sum_{12}\text{DLPCBs}$ in fish muscle and fishery
556 products, and wild fish (EQS_{Biota}) is established. In addition, regulation 1259/2011 also
557 set a threshold value of 3.5 pg TEQ g⁻¹ ww for $\sum_{17}\text{PCDD/Fs}$ only. The concentrations
558 measured in the various species considered did not exceed any of these thresholds,
559 although certain *S. sarda* and *S. colias* individuals (i.e. SS-2-3 and SC-3) approached the
560 6.5 pg TEQ g⁻¹ ww threshold for total DLPOPs (Figure 3A) with risk quotients (R) between
561 0.5 and 0.9 (R>1 indicating risk) (Table S14). The DLPOPs EWIs considering fish and
562 cephalopod consumption data in French population (section 2.6) were calculated for each
563 sample and varied from 0.1 to 11 pg TEQ Kg⁻¹ bw week⁻¹. All individuals of *S. colias* and
564 *S. sarda* exceeded the tolerable weekly intake (TWI) of 2 pg TEQ Kg⁻¹ bw week⁻¹ set by
565 the European Food Safety Authority⁶⁷ with R values ranging from 1.1-5.3 and 1.9-2.8 for
566 *S. sarda* and *S. colias*, respectively (Table S14). The major contributors to the EWI and
567 subsequent risk for humans associated with the consumption of *S. colias* and *S. sarda*
568 were DLPCBs, particularly PCB-126.

569 Moreover, Directive 1259/2011 sets a maximum level (75 ng g⁻¹ ww) for the sum of the
570 six non-dioxin-like PCBs in seafood, also referred as IDPCB in this work. The $\sum_{\text{ID}}\text{PCB}$
571 concentrations in the studied food web were for most samples below that threshold (Table
572 S11), varying from 0.6 to 59 ng g⁻¹ ww. However, some *S. colias* individuals again
573 approached the threshold (R=0.8 for SC-1-2) and two out three samples of the top
574 predator *S. sarda* (i.e. SS-2-3) exceeded it with concentrations between 106-109 ng g⁻¹
575 ww (R=1.4, Table S14) pointing to a risk for humans to non-dioxin-like PCBs due to

576 consumption of this last species. It has been reported that non-dioxin-like PCBs, which
577 are weak aryl hydrocarbon receptor agonists, can modulate the overall toxic potency of
578 2,3,7,8-TCDD and related compounds⁵⁵ and therefore their co-occurrence cannot exclude
579 cumulative effects including synergic harmful effects.

580

581 3.5.2. PBDEs

582 Regarding PBDEs, the EQS_{Biota} of 0.0085 ng g⁻¹ ww (as Σ_6 PBDEs including BDE-28,-
583 47,-99,-100,-153,-154) based on human toxicity⁵⁶ was used for a first risk evaluation.
584 This threshold value was exceeded in every single individual in all TLs (Table S14),
585 indicating a risk for humans consuming the edible species, again pointing to *S. sarda*
586 (R=54-97) and *S. colias* (R=15-65) as the most 'risky' fish. However, some authors argue
587 that the existing EQS_{Biota} for PBDEs is extremely low leading to environmental
588 concentrations exceeding the threshold value in most study cases.^{51,68} Moreover, for
589 OSPAR assessment, there are still questions on the reliability and appropriateness of
590 some of the thresholds set as EQS_{biota}, based on human protection, to support
591 environmental assessment. It has been recently proposed to adopt the Canadian Federal
592 Environmental Quality Guidelines (FEQGs) based only on wildlife protection.⁶⁹
593 Comparing our concentrations with the proposed FEQGs, which have been set for a
594 number of individual PBDE congeners (i.e BDE-28, 47, 99, 100, 153, 154, 183 and 209),
595 there is no indication of apparent risk (Table S15). However, it should be noted that the
596 FEQGs are derived for whole organisms, and PBDE were measured in muscle in this
597 study. The Σ PBDEs EWI in France ranged from 0.02 to 1.9 ng Kg⁻¹ bw week⁻¹, the highest
598 value corresponding to *S. sarda*, followed by *S. colias* (0.7-1.2 ng Kg⁻¹ bw week⁻¹). For
599 comparison, an average estimated intake of 0.14 ng Kg⁻¹ bw d⁻¹ via fish and seafood
600 consumption for adults has been reported for Valencia (Spain) in the NW Mediterranean

601 Sea⁵⁴, corresponding well with our higher end estimate (0.17 ng Kg⁻¹ bw d⁻¹). There is no
602 TWI for Σ PBDE to the best of our knowledge, but the US Agency for Toxic Substances
603 and Disease Registry and the US Environmental Protection Agency have established
604 acute/sub-chronic Minimal Risk Levels⁷⁰ and chronic oral reference doses⁷¹, respectively,
605 for some of the most abundant PBDE congeners (i.e BDE-47, BDE-99, BDE-153, BDE-
606 209). These health standards ranged from 3 to 7000 ng Kg⁻¹ bw d⁻¹ depending on the
607 congener, showing that there is no risk associated with these congeners for humans
608 consuming these Mediterranean fishery products.

609

610 Our study demonstrates that the edible species Atlantic bonito (*S. sarda*) and Atlantic
611 chub mackerel (*S. colias*) bioaccumulated the highest concentrations of POPs which may
612 pose a risk for humans consuming these species caught and /or marketed in the Gulf of
613 Lion (NW Mediterranean Sea) mostly due to DLPOPs exposure. The cephalopods
614 southern shortfin squid (*I. coindetti*) and European flying squid (*T. sagittatus*) showed a
615 high potential for the accumulation of PCDD/Fs, however no indications of risk can be
616 observed according to our evaluation. An in-depth review of available thresholds for
617 PBDEs concentration in biota and the determination of tolerable daily intakes for more
618 congeners are urgently needed to provide a more comprehensive risk assessment. We
619 consider the calculated EWI as a lower-end estimate of their total exposure, since their
620 intake through other food items is not considered and external exposure routes were not
621 accounted for in this evaluation. For example, the reported median outdoor inhalation
622 intake in the study area for DLPOPs (0.028 pg TEQ Kg⁻¹ bw week⁻¹) and Σ PBDEs (0.015
623 ng Kg⁻¹ bw week⁻¹) could range up between 0.4 - 1.3% and 0.8-2.1 % their respective total
624 intakes. These results warrant further research on the overall intake rates of the three POP
625 families by humans and their cumulative risks in this impacted region of NW

626 Mediterranean Sea. Our results would be highly valuable to set up-to-date and comparable
627 concentration thresholds in biota considering trophic level normalization as
628 recommended by EU guidance⁷² and OSPAR.⁷³ Besides, the characterized food web, as
629 a whole, may be a useful and site-specific comprehensive ‘bioindicator’ of the chemical
630 pollution status, representing a dynamic stock of toxic organic chemicals in the study
631 area. A lower-end estimation of its integrated POPs burden in the study season, calculated
632 considering the sum of the median concentrations in all TLs, reached $\sim 18 \mu\text{g g}^{-1} \text{lw}$. This
633 value could be considered a benchmark for further studies using the same food web,
634 opening new perspectives for the monitoring of toxic chemicals in Mediterranean coastal
635 waters.

636

637

638 **Acknowledgements**

639

640 This work was funded by the Labex OT-Med (no. ANR-11-LABX-0061) – MEDPOP,
641 funded by the French Government *Investissements d'Avenir* (ANR) through the
642 A*MIDEX project (no ANR-11-IDEX-0001-02) and contributes to the
643 MERMEX/MISTRALS program. It also received financial support from the PARME
644 CNRS EC2CO project. D. Bănaru had the benefit of leave within the ANR
645 CONTAMPUMP project (no ANR-19-CE34-0001) and C.-T. Chen received PhD
646 funding from French Ministry of National Education, Research and Technology. A
647 financial contribution was obtained too from the Ministry of Agriculture and Fisheries,
648 Food and Environment (MAPAMA), project 15CAES004. J.M-A. acknowledges his
649 contract under projects 15CAES004 and 17CAES004. We thank Bastian Carré for his
650 help for lipid analyses and Christian Re for his assistance in sample preparation.

651

652 **Supporting information available**

653

654 Additional data and details on the sampling and analytical procedures, QA/QC,
655 concentration of individual congeners, TMFs among others are available free of charge
656 via the Internet at <http://pubs.acs.org/>

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658 **References**

659

660 [1] Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F.,
661 Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., De Wit, C.A.,
662 Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin,
663 U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B.,
664 Liverman, D., Richardson, K., Crutzen, P., Foley, J.A. A safe operating space for
665 humanity. *Nature* **2009**, 461, 472-475

666

667 [2] Diamond, M.L., de Wit, C.A., Molander, S., Scheringer, M., Backhaus, T., Lohmann,
668 R., Arvidsson, R., Bergman, Å., Hauschild, M., Holoubek, I., Persson, L., Suzuki,
669 N., Vighi, M., Zetzsch, C. Exploring the planetary boundary for chemical pollution.
670 *Environ. Int.* **2015**, 78, 8-15.

671

672 [3] The MERMEX Group, X. Durrieu de Madron, C. Guieu, R. Sempéré, P. Conan, D.
673 Cossa, F. D'Ortenzio, C. Estournel, F. Gazeau, C. Rabouille, L. Stemmann, S. Bonnet, F.
674 Diaz, P. Koubbi, O. Radakovitch, M. Babin, M. Baklouti, C. Bancon-Montigny, S.
675 Belviso, N. Bensoussan, B. Bonsang, I. Bouloubassi, C. Brunet, J.F. Cadiou, F. Carlotti,
676 M. Chami, S. Charmasson, B. Charrière, J. Dachs, D. Doxaran, J.C. Dutay, F. Elbaz-
677 Poulichet, M. Eléaume, F. Eyrolles, C. Fernandez, S. Fowler, P. Francour, J.C. Gaertner,
678 R. Galzin, S. Gasparini, J.F. Ghiglione, J.L. Gonzalez, C. Goyet, L. Guidi, K. Guizien,
679 L.E. Heimbürger, S.H.M. Jacquet, W.H. Jeffrey, F. Joux, P. Le Hir, K. Leblanc, D.
680 Lefèvre, C. Lejeusne, R. Lemé, M.D. Loÿe-Pilot, M. Mallet, L. Méjanelle, F. Mélin, C.
681 Mellon, B. Mérigot, P.L. Merle, C. Migon, W.L. Miller, L. Mortier, B. Mostajir, L.
682 Mousseau, T. Moutin, J. Para, T. Pérez, A. Petrenko, J.C. Poggiale, L. Prieur, M. Pujo-

683 Pay, V. Pulido, P. Raimbault, A.P. Rees, C. Ridame, J.F. Rontani, D. Ruiz Pino, M.A.
684 Sicre, V. Taillandier, C. Tamburini, T. Tanaka, I. Taupier-Letage, M. Tedetti, P. Testor,
685 H. Thébault, B. Thouvenin, F. Touratier, J. Tronczynski, C. Ulses, F. Van Wambeke, V.
686 Vantrepotte, S. Vaz, R. Verney. Marine ecosystems' responses to climatic and
687 anthropogenic forcings in the Mediterranean. *Prog. Oceanogr.* **2011**, 91, 97-166.

688

689 [4] Cózar, A., Sanz-Martín, M., Martí, E., González-Gordillo, J. I., Ubeda, B., Gálvez,
690 J.A., Irigoien, X., Duarte, C. M. Plastic accumulation in the Mediterranean Sea. *PLoS*
691 *ONE*, **2015**, 10(4). <https://doi.org/10.1371/journal.pone.0121762>

692

693 [5] Marsili, L., Jiménez, B., Borrell, A., 2018. Chapter 7 - persistent organic pollutants in
694 cetaceans living in a hotspot area: the Mediterranean Sea. In: Fossi, M.C., Panti, C. (Eds.),
695 Marine Mammal Ecotoxicology. Academic Press, pp. 185–212.
696 <https://doi.org/10.1016/B978-0-12-812144-3.00007-3>.

697

698 [6] Mandalakis M., Apostolakis M., and Stephanou E.G. Mass budget and dynamics of
699 polychlorinated biphenyls in the eastern Mediterranean Sea. *Global biogeochem. Cy.*
700 **2005**, 19, GB 018, 1-16.

701

702 [7] Gómez-Gutiérrez, A.; Garnacho, E.; Bayona, J.M.; Albaigés J. Assessment of the
703 Mediterranean sediments contamination by persistent organic pollutants. *Environ. Pollut.*
704 **2007**, 148, 396-408.

705

706 [8] Berrojalbiz, N., Dachs, J., Ojeda, M.J., Valle, M.C., Castro-Jiménez J., Wollgast, J.,
707 Ghiani, M., Hanke, G., Zaldivar, J.M. Biogeochemical and physical controls on

708 concentrations of polycyclic aromatic hydrocarbons in water and plankton of the
709 Mediterranean and Black Seas. *Global Biogeochemical Cycles*, **2011**, 25, GB4003,
710 doi:10.1029/2010GB003775.

711

712 [9] Castro-Jiménez, J.; Berrojalbiz, N.; Wollgast, J.; Dachs, J. Polycyclic aromatic
713 hydrocarbons (PAHs) in the Mediterranean Sea: atmospheric occurrence, deposition and
714 decoupling with settling fluxes in the water column. *Environ. Pollut.* **2012**, 166, 40-47.

715

716 [10] Castro-Jiménez, J., Barhoumi, B., Paluselli, A., Tedetti, M., Jiménez, B., Muñoz-
717 Arnanz, J., Wortham, H., Driss, M.R., Sempere, R. Occurrence, loading and exposure of
718 atmospheric particle-bound POPs at the African and European edges of the western
719 Mediterranean Sea. *Environ. Sci. Technol.* **2017**, 51, 13180–13189.

720

721 [11] Bodiguel, X., Loizeau, V., Le Guellec, A-M., Roupsard, F., Philippon, X., Mellon-
722 Duval, C. Influence of sex, maturity and reproduction on PCB and p,p'DDE
723 concentrations and repartitions in the European hake (*Merluccius merluccius*, L.) from
724 the Gulf of Lions (N.W. Mediterranean). *Sci. Total Environ.* **2009**, 408, 304-311

725

726 [12] Berrojalbiz, N., Dachs, J., Ojeda, M.J., Valle, M.C., Del Vento, S., Castro-Jiménez,
727 J., Mariani, G., Wollgast, J., Hanke G. Persistent organic pollutants in Mediterranean
728 seawater and processes affecting their accumulation in plankton. *Environ. Sci. Technol.*
729 **2011**, 45, 4315-4322.

730

731 [13] Cresson, P., Claire Fabri, M., Marco Miralles, F., Dufour, J-L., Elleboode, R., Sevin,
732 K., Mahe, K., Bouchoucha, M. Variability of PCB burden in 5 fish and sharks species of
733 the French Mediterranean continental slope. *Environ. Pollut.* 2016, 212, 374-381.

734

735 [14] Bartalini A., Muñoz-Arnanz J., Marsili L., Mazzariol S., Fossi M.C., Jiménez B.
736 Evaluation of PCDD/Fs, dioxin-like PCBs and PBDEs in sperm whales from the
737 Mediterranean Sea. *Sci. Total Environ*, **2019**, 653, 1417-1425.

738

739 [15] Bartalini A., Muñoz-Arnanz J., Bainsi, M., Panti, C., Galli, M., Giani, D., Fossi M.C.,
740 Jiménez B. Relevance of current PCB concentrations in edible fish species from the
741 Mediterranean Sea. *Sci. Total Environ*, **2020**, 737, 139520.

742

743 [16] Capanni, F., Muñoz-Arnanz J., Marsili L., Fossi M.C., Jiménez B. Assessment of
744 PCDD/Fs, dioxin-like PCBs and PBDEs in Mediterranean striped dolphins. *Mar. Pollut.*
745 *Bull.* **2020**, 156, 111207.

746

747 [17] Miniero, R., Abate, V., Brambilla, G., Davoli, E., De Felip, E., De Filippis, S.P.,
748 Dellatte, E., De Luca, S., Fanelli, R., Fattore, E., Ferri, F., Fochi, I., Rita Fulgenzi, A.,
749 Iacovella, N., Iamiceli, A.L., Lucchetti, D., Melotti, P., Moret, I., Piazza, R., Roncarati,
750 A., Ubaldi, A., Zambon, S., di Domenico, A. Persistent toxic substances in Mediterranean
751 aquatic species. *Sci. Total Environ.* **2014**, 494, 18–27.

752

753 [18] Wania, F., MacKay, D. Tracking the distribution of persistent organic pollutants.
754 *Environ. Sci. Technol.* **1996**, 30, A390–A396

755

756 [19] Harrad, S.J. and Jones, K.C. A source inventory and budget for chlorinated dioxins
757 and furans in the United Kingdom environment. *Sci. Total Environ.* **1992**, 126, 89-107.
758

759 [20] Bruzy, L.P. and Hites, R.A. Global mass balance for polychlorinated dibenzo-p-
760 dioxins and dibenzofurans. *Environ. Sci. Technol.* **1996**, 30, 1797-1804.
761

762 [21] Breivik, K., Sweetman, A., Pacyna, J. M., and Jones, K. C. Towards a global
763 historical emission inventory for selected PCB congeners – a mass balance approach: 2.
764 Emissions, *Sci. Total Environ.* **2002**, 290, 199–224.
765

766 [22] Nizzetto, L., Macleod, M., Borgå, K., Cabrerizo, A., Dachs, J., Di Guardo, A.,
767 Ghirardello, D., Hansen, K. M., Jarvis, A., Lindroth, A., Ludwig, B., Monteith, D.,
768 Perlinger, J. A., Scheringer, M., Schwendenmann, L., Semple, K. T., Wick, L. Y., Zhang,
769 G., Jones, K. C.; Nizzetto, L. Past, Present, and Future Controls on Levels of Persistent
770 Organic Pollutants in the Global Environment. *Environ. Sci. Technol.* **2010**, 44, 6526–
771 6531
772

773 [23] La Guardia, M.J., Hale, C.R., Harvey, E. Detailed Polybrominated Diphenyl Ether
774 (PBDE) Congener Composition of the Widely Used Penta-, Octa-, and Deca-PBDE
775 Technical Flame-retardant Mixtures. *Environ. Sci. Technol.* **2006**, 40, 6247-6254
776

777 [24] United Nations Environment Programme: Stockholm Convention on Persistent
778 Organic Pollutants: Geneva, 2009. Report of the Conference of the Parties of the
779 Stockholm Convention on Persistent Organic Pollutants on the Work of Its Fourth
780 Meeting. p. 112.

781 [25] Directive 2008/56/EC of the European Parliament and of the Council of 17 June
782 2008 establishing a framework for community action in the field of marine environmental
783 policy (Marine Strategy Framework Directive). OJ L 164, 25.6.2008, p. 19–40.
784

785 [26] Millet, B., Pinazo, C., Bănaru, D., Pagès, R., Guiart, P., Pairaud, I. Unexpected
786 spatial impact of treatment plant discharges induced by episodic hydrodynamic events:
787 modelling Lagrangian transport of fine particles by northern current intrusions in the bays
788 of Marseille (France). *PLoS One*, **2018**, 13, e0195257
789

790 [27] Sempéré, R., Para, J., Tedetti, M., Charriere, B., Mallet, M. 2015. Attenuation of
791 UVR and PAR in relation with chromophoric dissolved organic matter in surface coastal
792 waters of the Northwestern Mediterranean Sea. *J Photochem. Photobiol. A: Chem.* **2015**,
793 *91*, 851-861.
794

795 [28] Petrenko, A., Leredde, Y., Marsaleix, P. Circulation in a stratified and wind-forced
796 Gulf of Lions, NW Mediterranean Sea: in situ and modeling data. *Cont. Shelf Res.* **2005**,
797 *25*, 7–27.
798

799 [29] Para, J., Coble, P. G., Charrière, B., Tedetti, M., Fontana, C., Sempéré R.
800 Fluorescence and absorption properties of chromophoric dissolved organic matter
801 (CDOM) in coastal surface waters of the northwestern Mediterranean Sea, influence of
802 the Rhône River. *Biogeosciences*, 2010, 7: 4083-4103.
803

804 [30] Cresson, P., Ruitton, S., Fontaine, M.-F., Harmelin-Vivien, M. Spatio-temporal

805 variation of suspended and sedimentary organic matter quality in the Bay of Marseilles
806 (NW Mediterranean) assessed by biochemical and isotopic analyses. *Mar. Poll. Bull.*
807 **2012**, 64, 1112–1121.

808

809 [31] Fraysse, M., Pairaud, I., Ross, O.N., Faure, V.M., Pinazo, C. Intrusion of Rhone
810 River diluted water into the Bay of Marseille: generation processes and impacts on
811 ecosystem functioning. *J. Geophys. Res.* **2014**, 119, 6535–6556.

812

813 [32] Guigue, C., Tedetti, M., Ferretto, N., Garcia, N., Méjanelle, L., Goutx, M. Spatial
814 and seasonal variabilities of dissolved hydrocarbons in surface waters from the
815 Northwestern Mediterranean Sea: results from one year intensive sampling. *Sci. Total*
816 *Environ.* **2014**, 466–467, 650–662.

817

818 [33] Mourier, B., Desmet, M., Van Metre, P.C., Mahler, B.J., Perrodin, Y., Roux, G.,
819 Bedell, J-P., Lefèvre, I., Babut, M. Historical records, sources, and spatial trends of PCBs
820 along the Rhône River (France). *Sci. Total Environ.* **2014**, 476–477, 568–576.

821

822 [34] Paluselli, A., Aminot, Y., Galgani, F., Net, S., Sempéré, R., 2017. Occurrence of
823 phthalate acid esters (PAEs) in the northwestern Mediterranean Sea and the Rhone River.
824 *Prog. Oceanogr.* **2018**, 163, 221-231.

825

826 [35] Schmidt, N., Castro-Jiménez, J., Fauvelle, V., Ourgaud, M., Sempéré, R. Organic
827 plastic additives in surface waters of the Rhone River (France) – occurrence and temporal
828 variability. *Environ. Pollut.* **2020**, 133637.

829

830 [36] Schmidt, N., Fauvelle, V., Castro-Jiménez, J., Lajaunie-Salla, K., Pinazo, C.,
831 Sempéré, R. Occurrence of perfluoroalkyl substances in the Bay of Marseille (NW
832 Mediterranean Sea) and the Rhône River. *Mar. Pollut. Bull.* **2019**, 149, 110491.
833

834 [37] Schmidt, N., Castro-Jiménez, J., Oursell, B., Sempéré, R. Phthalates and
835 organophosphate esters in surface water, sediments and zooplankton of the NW
836 Mediterranean Sea: exploring links with microplastic abundance and accumulation in the
837 marine food web. *Environ. Pollut.* **2020**, 115970.
838

839 [38] Syakti, A. D., Asia, L., Kanzari, F. Umasangadji, L., Malleret, L., Ternois, Y., Mille,
840 G., Doumenq, P. Distribution of organochlorine pesticides (OCs) and polychlorinated
841 biphenyls (PCBs) in marine sediments directly exposed to wastewater from Cortiou,
842 Marseille. *Environ. Sci. Pollut. Res.* **2012**, 19, 1524–1535.
843

844 [39] Kanzari, F., Syakti, A. D., Asia, L., Malleret, L., Piram, A., Mille, G., Doumenq, P.
845 Distributions and sources of persistent organic pollutants (aliphatic hydrocarbons, PAHs,
846 PCBs and pesticides) in surface sediments of an industrialized urban river (Huveaune),
847 France. *Sci. Total Environ.* **2014**, 478, 141-151.
848

849 [40] Chen, C-T., Bănar, D., Carlotti, F., Faucheux, M., Harmelin-Vivien, M. Seasonal
850 variation in biochemical and energy content of size-fractionated zooplankton in the Bay
851 of Marseille (North-Western Mediterranean Sea). *J. Mar. Syst.* **2019**, 199, 103223.
852
853

- 854 [41] Post, D. M., Layman, C. A., Arrington, D. A., Takimoto, G., Quattrochi, J. and
855 Montan, C. G. Getting to the fat of the matter: models, methods and assumptions for
856 dealing with lipids in stable isotope analyses. *Oecologia*, **2007**, 152,179-189
857
- 858 [42] Bănar, D., Carlotti, F., Barani, A., Grégori, G., Neffati, N., Harmelin-Vivien, M.
859 Seasonal variation of stable isotope ratios of size-fractionated zooplankton in the Bay of
860 Marseille (NW Mediterranean Sea). *J. Plankton Res.* **2014**, 36, 145–156.
861
- 862 [43] Hussey, N. E., Brush, J., McCarthy, I. D., Fisk, A. T. $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ diet
863 discrimination factors for large sharks under semi-controlled conditions. *Compar.*
864 *Biochem. Physiol. B*, **2010**, 155, 455–453
865
- 866 [44] US-EPA, Method 1613. Tetra- through Octa-Chlorinated Dioxins and Furans by
867 Isotope Dilution HRGC/HRMS; U.S. Environmental Protection Agency: Washington,
868 DC, October 1994.
869
- 870 [45] Borga, K., Kidd, K.A., Muir, D.C.G., Berglund, O., Conder, J.M, Gobas, F.A.P.C.,
871 Kucklick, J., Malm, O., Powell, D.E. Trophic magnification factors: Considerations of
872 ecology, ecosystems and study design. *Integr. Environ. Assess. Manag* **2012**, 8, 64–84.
873
- 874 [46] Walters, D.M., Mills, M.A., Cade, B.S., Burkard, L.P. Trophic Magnification of
875 PCBs and Its Relationship to the Octanol-Water Partition Coefficient. *Environ. Sci.*
876 *Technol.* **2011**, 45, 3917-3924
877
- 878 [47] FAOSTAT [WWW Document], 2013. URL <http://www.fao.org/faostat/>

879 en/#data/CL (accessed 04/11/2020).

880

881 [48] Hunt, B. P. V., Carlotti, F. Donoso, K., Pagano, M., D'Ortenzio, F., Taillandier, V.
882 and Conan P. Trophic pathways of phytoplankton size classes through the zooplankton
883 food web over the spring transition period in the north-west Mediterranean Sea. *J*
884 *Geophys Res Oceans*, **2017**, 122, 6309-6324

885

886 [49] Castro-Jiménez, J.; Deviller, G.; Ghiani, M.; Loos, R.; Mariani, G.; Skejo, H.;
887 Umlauf, G.; Wollgast, J.; Laugier, T.; Héas-Moisan, K.; Léauté, F.; Munschy, C.; Tixier,
888 C.; Tronczyński, J. PCDD/F and PCB multi-media ambient concentrations, congener
889 patterns and occurrence in a Mediterranean coastal lagoon (Etang de Thau, France).
890 *Environ. Pollut.* **2008**, 156, 123–135.

891

892 [50] Ruus, A., Berge, J.A., Bergstad, O.A., Knutsen, J.A., Hylland, K. Disposition of
893 polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs)
894 in two Norwegian epibenthic marine food webs. *Chemosphere* **2006**, 62, 1856-1868.

895

896 [51] Munschy, C., Bely, Héas-Moisan, N. K. Olivier, N. Pollono, C., Holland, S., Bodin,
897 N. Tissue-specific bioaccumulation of a wide range of legacy and emerging persistent
898 organic contaminants in swordfish (*Xiphias gladius*) from Seychelles, Western Indian
899 Ocean. *Mar. Pollut. Bull.*, **2020**, 158, 111436.

900

901 [52] Roberts, S.C., Noyes, P.D., Gallagher, E.P., Stapleton, H.M. Species-specific
902 differences and structure- activity relationships in the debromination of PBDE congeners
903 in three fish species. *Environ. Sci. Technol.* **2011**, 45, 1999-2005.

904

905 [53] Voorspoels, S., Covaci, A., Schepens, P. Polybrominated diphenyl ethers in marine
906 species from the Belgian North Sea and the Western Scheldt estuary: levels, profiles, and
907 distribution. *Environ. Sci. Technol.* **2003**, 37, 4348–4357.

908

909 [54] Pardo, O., Beser, M.I., Yusa, V., Beltran, J. Probabilistic risk assessment of the
910 exposure to polybrominated diphenyl ethers via fish and seafood consumption in the
911 Region of Valencia (Spain). *Chemosphere*, **2014**, 104, 7-14.

912

913 [55] Van den Berg, M., Birnbaum, L.S., Denison, M., De Vito, M., Farland, W., Feeley,
914 M., Fiedler, H., Hakansson, H., Hanberg, H., Haws, K.L., Rose, M., Safe, S., Schrenk,
915 D., Tohyama, C., Tritscher, A., Tuomisto, J., Tysklind, M., Walker, N., Peterson, R.E.
916 The 2005 world health organization reevaluation of human and mammalian toxic
917 equivalency factors for dioxins and dioxin-like compounds. *Toxicol. Sci.* **2006**, 93, 223–
918 241.

919

920 [56] Directive 2013/39/EU of the European Parliament and of the Council of 12 August
921 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances
922 in the field of water policy. OJ L 226, 24.8.2013, p. 1–17.

923

924 [57] PolyBDEs EQS dossier 2011. [https://circabc.europa.eu/sd/a/d07ed9f5-0760-4561-](https://circabc.europa.eu/sd/a/d07ed9f5-0760-4561-b642-04bc1e4a580e/PBDE%20EQS%20dossier%202011.pdf)
925 [b642-04bc1e4a580e/PBDE%20EQS%20dossier%202011.pdf](https://circabc.europa.eu/sd/a/d07ed9f5-0760-4561-b642-04bc1e4a580e/PBDE%20EQS%20dossier%202011.pdf) (accessed 31/01/21)

926

927 [58] Khairy, M.A., Weinstein, M.P., Lohmann, R. Trophodynamic Behavior of
928 Hydrophobic organic contaminants in the aquatic food web of a Tidal river. *Environ. Sci.*
929 *Technol.* **2014**, 48, 12533-12542.

930

931 [59] Dioxin and Dioxin-Like PCBs EQS dossier 2011.
932 [https://circabc.europa.eu/d/a/workspace/SpacesStore/f0d90906-c361-4af1-82b1-](https://circabc.europa.eu/d/a/workspace/SpacesStore/f0d90906-c361-4af1-82b1-d2e52f826c14/Dioxins%2520%2526%2520PCBDL%2520EQS%2520dossier%25202011.pdf)
933 [d2e52f826c14/Dioxins%2520%2526%2520PCBDL%2520EQS%2520dossier%252020](https://circabc.europa.eu/d/a/workspace/SpacesStore/f0d90906-c361-4af1-82b1-d2e52f826c14/Dioxins%2520%2526%2520PCBDL%2520EQS%2520dossier%25202011.pdf)
934 [11.pdf](https://circabc.europa.eu/d/a/workspace/SpacesStore/f0d90906-c361-4af1-82b1-d2e52f826c14/Dioxins%2520%2526%2520PCBDL%2520EQS%2520dossier%25202011.pdf) (accessed 31/01/21)

935

936 [60] Harmelin-Vivien, M., Bodiguel, X., Charmasson, S., Loizeau, V., Mellon-Duval, C.,
937 Tronczynski, J., Cossa, D. Differential biomagnification of PCB, PBDE, Hg and
938 Radiocesium in the food web of the European hake from the NW Mediterranean. *Mar.*
939 *Pollut. Bull.* **2012**, 64, 974-983.

940

941 [61] Walters, D.M., Jardine, T.D., Cade, B.S., Kidd, K.A., Muir, D.C.G., Leipzig-Scott,
942 P. Trophic Magnification of Organic Chemicals: A Global Synthesis. *Environ. Sci.*
943 *Technol.* **2016**, 50, 4650–4658.

944

945 [62] Brandsma, S.H., Leonards, P.E.G., Leslie, H.A., de Boer, J., 2015. Tracing
946 organophosphorus and brominated flame retardants and plasticizers in an estuarine food
947 web. *Sci. Total Environ.* **2015**, 505, 22–31.

948

949 [63] Romero-Romero, S., Herrero, L., Fernández, M., Gómara, B., Acuña, J.L.
950 Biomagnification of persistent organic pollutants in a deep-sea, temperate food web. *Sci.*
951 *Total Environ.* **2017**, 605-606, 589-597.

952 [64] Vuković, G., Romanić, S.H., Babić, Z., Mustać, B., Štrbac, M., Deljanin, I.,
953 Antanasijević, D. Persistent organic pollutants (POPs) in edible fish species from
954 different fishing zones of Croatian Adriatic. *Mar. Pollut. Bull.* **2018**, 137, 71-80.
955

956 [65] Barone, G., Storelli, A., Garofalo, R., Mallamaci, R., Quaglia, N.C., Storelli, M.M.
957 PCBs and PCDD/Fs in Bluefin Tuna: Occurrence and Dietary Intake. *Int. J. Environ. Res.*
958 *Public Health.* 2018, 15, 911.
959

960 [66] Commission regulation (EU) no. 1259/2011 of 2 December 2011 amending
961 Regulation (EC) no. 1881/2006 setting maximum levels for certain contaminants in
962 foodstuffs as regards dioxin-like PCBs and non-dioxin-like PCBs. *OJEU* **2011**, 320, 18–
963 23.
964

965 [67] Knutsen, H.K., Alexander, J., Barregård, L., Bignami, M., Brüschweiler, B.,
966 Ceccatelli, S., Cottrill, B., Dinovi, M., Edler, L., Grasl-Kraupp, B. Hogstrand, C., Nebbia,
967 C.S., Oswald, I.P., Petersen, A., Rose, M., Roudot, A.-C., Schwerdtle, T., Vleminckx, C.,
968 Vollmer, G., Wallace, H., Fürst, P., Håkansson, H., Halldorsson, T., Lundebye, A.-K.,
969 Pohjanvirta, R., Rylander, L., Smith, A., Loveren, H. van, Waalkens- Berendsen, I.,
970 Zeilmaker, M., Binaglia, M., Ruiz, J.Á.G., Horváth, Z., Christoph, E., Ciccolallo, L.,
971 Bordajandi, L.R., Steinkellner, H., Hoogenboom, L. (Ron). Risk for animal and human
972 health related to the presence of dioxins and dioxin-like PCBs in feed and food. *EFSA J.*
973 16., 2018. doi:<https://doi.org/10.2903/j.efsa.2018.5333>.
974

975 [68] Fliedner, A., Lohmann, N., Rüdell, H., Teubner, D., Wellmitz, J., Koschorreck, J.
976 Current levels and trends of selected EU Water Framework Directive priority substances

977 in freshwater fish from the German environmental specimen bank. *Environ. Pollut.* **2016**,
978 216, 866–876.

979

980 [69] OSPAR 2020. Background document for Canadian Federal Environmental Quality
981 Guidelines (FEQGs) for Polybrominated Diphenyl Ethers (PBDEs) in sediment and biota.
982 Hazardous Substances and Eutrophication Series. pp1-19.

983

984 [70] ATSDR. Minimal Risk Levels (MRLs) for hazardous substances. Minimal Risk
985 Levels (MRLs) List. October 2020.

986

987 [71] US EPA. Technical Fact Sheet –Polybrominated Diphenyl Ethers (PBDEs)
988 November 2017.

989

990 [72] European Commission. 2014. Guidance Document 604 No. 32 on biota monitoring
991 (the implementation of EQSBIOTA) under the Water Framework Directive. Brussels
992 (BE): European Union. 87 p.

993

994 [73] OSPAR 2019. MIME’s considerations of using EQSbiota for OSPAR assessments,
995 pp-1-10.

996

997

998

999

1000