



**HAL**  
open science

## Impacts of chemical contamination on bacterio-phytoplankton coupling

Olivier Pringault, Marc Bouvy, Claire Carré, Eric Fouilland, Marouan Meddeb, Kaouther Mejri, Christophe Leboulanger, Asma Sakka Hlaili

### ► To cite this version:

Olivier Pringault, Marc Bouvy, Claire Carré, Eric Fouilland, Marouan Meddeb, et al.. Impacts of chemical contamination on bacterio-phytoplankton coupling. *Chemosphere*, 2020, 257, pp.127165. 10.1016/j.chemosphere.2020.127165 . hal-02792811

**HAL Id: hal-02792811**

**<https://hal.science/hal-02792811v1>**

Submitted on 10 Jun 2020

**HAL** is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

# Impacts of chemical contamination on bacterio- phytoplankton coupling

To be submitted to: CHEMOSPHERE

Olivier PRINGAULT<sup>\*1,2,3</sup>, Marc BOUVY<sup>2</sup>, Claire CARRE<sup>2</sup>,  
Eric FOUILLAND<sup>2</sup>, Marouan MEDDEB<sup>3</sup>, Kaouther MEJRI<sup>3</sup>,  
Christophe LÉBOULANGER<sup>2</sup> and Asma SAKKA HLAILI<sup>3,4</sup>

<sup>1</sup>Aix Marseille Univ, Université de Toulon, CNRS, IRD, MIO UM 110, 13288, Marseille, France.

<sup>2</sup>MARBEC Univ Montpellier, IRD, Ifremer, Montpellier, France.

<sup>3</sup>Faculté des Sciences de Bizerte, Université de Carthage, 7021 Zarzouna, Tunisie.

<sup>4</sup>Laboratory of Environmental Sciences, Biology and Physiology of aquatic organisms LR18ES41, University El Manar of Tunis, Faculty of Sciences of Tunis, Tunis, Tunisia.

\*Corresponding author

## 1 Abstract

2 Phytoplankton and bacterioplankton are the key components of the organic matter  
3 cycle in aquatic ecosystems, and their interactions can impact the transfer of carbon  
4 and ecosystem functioning. The aim of this work was to assess the consequences of  
5 chemical contamination on the coupling between phytoplankton and  
6 bacterioplankton in two contrasting marine coastal ecosystems: lagoon waters and  
7 offshore waters. Bacterial carbon demand was sustained by primary carbon  
8 production in the offshore situation, suggesting a tight coupling between both  
9 compartments. In contrast, in lagoon waters, due to a higher nutrient and organic  
10 matter availability, bacteria could rely on allochthonous carbon sources to sustain  
11 their carbon requirements, decreasing so the coupling between both compartments.  
12 Exposure to chemical contaminants, pesticides and metal trace elements, resulted in  
13 a significant inhibition of the metabolic activities (primary production and bacterial  
14 carbon demand) involved in the carbon cycle, especially in offshore waters during  
15 spring and fall, inducing a significant decrease of the coupling between primary  
16 producers and heterotrophs. This coupling loss was even more evident upon  
17 sediment resuspension for both ecosystems due to the important release of nutrients  
18 and organic matter. Resulting enrichment alleviated the toxic effects of contaminants  
19 as indicated by the stimulation of phytoplankton biomass and carbon production,  
20 and modified the composition of the phytoplankton community, impacting so the  
21 interactions between phytoplankton and bacterioplankton.

## 22 Keywords

23 pesticides; trace metallic elements; carbon cycle; primary production; bacterial  
24 carbon demand

25

26

## 27 1 Introduction

28 Bacterioplankton and phytoplankton are the key players in the carbon cycle of  
29 aquatic systems. The two microbial communities are involved in the de novo  
30 production of organic matter by phytoplankton, as well as in its mineralization by  
31 bacterioplankton. Since they represent the first levels of the pelagic marine food web  
32 (Berglund et al., 2007), it is imperative to understand their functioning and their  
33 interactions within the environment to better assess the consequences of natural or  
34 anthropic perturbations for a given aquatic ecosystem. In coastal zone and shallow-  
35 water environments, water column chemistry is closely related to the sediment  
36 interface dynamics (Hochard et al., 2010), thus, the functioning of the  
37 bacterioplankton and phytoplankton can be strongly affected during storm events  
38 resulting in the release of nutrients, microorganisms and contaminants into the water  
39 column (Roberts, 2012).

40 The impact of chemical contamination on the first levels of the trophic chain can be  
41 observed from a structural point of view, with changes observed in the microbial  
42 community, through the selection of species considered tolerant to contaminants  
43 (Dorigo et al., 2010; Lekunberri et al., 2010; Pesce et al., 2009). The impact can also be  
44 observed from a functional point of view, with modifications of the carbon cycle such  
45 as the stimulation of heterotrophy versus autotrophy upon metal spiking (Rochelle-  
46 Newall et al., 2008). Although the extent of the threat of contamination is large and  
47 has been increasing due to demographic patterns and resulting anthropic pressure  
48 on the coastal zone (Schwarzenbach et al., 2006; Small and Nicholls, 2003), pollution  
49 remains one of the least-studied stressors in ecology to date (Johnston et al., 2015). As  
50 a general rule, chemical contaminants are often considered as stressors due to their  
51 toxic potential. In this context, chemical contaminants were demonstrated to induce  
52 marine ecosystem functioning alterations, mainly by decreasing productivity  
53 (primary production) and increasing respiration (Johnston et al., 2015). These  
54 changes in ecosystem functioning by chemical contamination can be concomitant

55 with modification of the biodiversity (alpha and beta) and/or diversity losses  
56 (Johnston and Roberts, 2009), with consequences on interactions between  
57 bacterioplankton and phytoplankton (Goni-Urriza et al., 2018) that may, in turn,  
58 impact the higher trophic levels (Hansson et al., 2013; Hulot et al., 2000).

59 The existence of phytoplankton–bacterioplankton coupling is supported by a  
60 significant correlation between primary production and heterotrophic bacterial  
61 production (Bouvy et al., 1998; Cole et al., 1982), but revisited more recently using a  
62 larger dataset comparing dissolved primary production and bacterial carbon  
63 demand in different aquatic ecosystems (Fouilland and Mostajir, 2011, 2010). In  
64 coastal areas, bacteria production may or not strongly depends on phytoplankton  
65 exudates according to the availability of other sources of carbon such recycled  
66 trophic carbon sources (Fouilland et al., 2014) or external terrestrial sources  
67 (Fouilland et al., 2018; Morán et al., 2002). In these latter cases, the bacterial carbon  
68 demand can largely exceed phytoplankton production (Morán et al., 2002)The  
69 coupling between phytoplankton and bacterioplankton mediates the carbon transfer  
70 to the trophic web and consequently controls the ecosystem functioning.  
71 Anthropogenic perturbations like nutrient loading (Prieto et al., 2015), global  
72 warming (Hoppe et al., 2008) or acidification (Hornick et al., 2017) can severely alter  
73 the coupling between primary producers and heterotrophic bacteria. Nevertheless,  
74 the availability of labile carbon sources is often linked with chemical molecules  
75 whose effects are difficult to discern on the interactions between both compartments.  
76 We suggest that the release of contaminants by sediments in coastal waters might  
77 affect the degree of coupling between phytoplankton and bacteria through two ways:  
78 i) the inhibition of phytoplankton communities linked to the toxicity of chemical  
79 compounds, and ii) the addition of external source of carbon for heterotrophic  
80 bacteria.

81 In this context, the goal of this study was to assess the impact of chemical  
82 contamination on the coupling between phytoplankton and bacterioplankton in two

83 contrasting ecosystems, marine offshore waters and lagoon waters. The two  
84 contrasting ecosystems were sampled to test the hypothesis that the bacterial and  
85 phytoplankton response to chemical contamination would be different in the two  
86 sampling sites, with possible consequences for the coupling between both  
87 compartments, considering the possible selective pressure and the adaptation that  
88 can be triggered by the chemical environment according to the concept of pollution-  
89 induced community tolerance (Blanck, 2002). For that purpose, water incubations in  
90 9L microcosms were performed in different seasons, with additions of a sediment  
91 elutriate or an artificial mixture of contaminants mimicking the main contaminants  
92 present in sediment elutriate (pesticides and metal trace elements). The chemical  
93 changes, along with the experiments with both contaminant exposures, were  
94 previously analyzed (Bancon-Montigny et al., 2019), and the present study focused  
95 on the changes observed for phytoplankton and bacterioplankton activities and the  
96 consequences for the coupling between both compartments.

## 97 2 Materials and Methods

### 98 2.1 Study sites and sampling

99 The study was conducted in southwestern Mediterranean ecosystems, the lagoon  
100 and the bay of Bizerte (Fig. S1 supplementary materials) during four distinct seasons  
101 spring (April 2014), winter (February 2015), fall (November 2015) and summer (July  
102 2016). Like most Mediterranean coastal lagoons (Arvanitidis et al., 2009), the Bizerte  
103 lagoon (lagoon station) in the north of Tunisia is a polluted coastal ecosystem subject  
104 to intense agriculture, urbanization and industrialization pressures, as well as  
105 pressures from naval and commercial shipping harbors (Barhoumi et al., 2014a,  
106 2014b; Yoshida et al., 2002). Consequently, sediments are strongly contaminated by a  
107 wide range of pollutants, including organic contaminants (polycyclic aromatic  
108 hydrocarbons, pesticides and herbicides) and metals (Barhoumi et al., 2014a, 2014b;  
109 Yoshida et al., 2002). The Bizerte bay is less contaminated than the lagoon, although

110 local polycyclic aromatic hydrocarbons (PAH) contamination has been recorded in  
111 the effluent from the oil refinery located on the shore of the bay (Mhadhbi et al., 2019;  
112 Zrafi-Nouira et al., 2009, 2008). The sampling was carried out in an offshore station  
113 (station O, 37° 16' 46.46" N 9° 53' 50.98" E) and a lagoon station (station L, 37° 12'  
114 43.96" N 9° 50' 79.78" E), as shown in Fig. S1 (supplementary materials). Water  
115 samples were collected from a 2 m depth; more details of the sampling procedure can  
116 be found in Pringault et al. (2016). Contaminated water was obtained from the  
117 resuspension of polluted sediment following the protocol described by Bonnet et al.  
118 (2000); this protocol, recommended by the US Environmental Protection Agency (US  
119 EPA), has been successfully used to characterize the contamination level and toxicity  
120 of sediments using nonadapted species (Bonnet et al., 2000; Carr and Chapman,  
121 1995). Polluted sediment was sampled in front of a cement factory in the lagoon  
122 channel (station CH, 37° 15' 40.22" N 9° 51' 30.49" E) as shown in Fig. S1, using a Van  
123 Veen grab. On the same day of sediment sampling, water was also sampled in the  
124 same station CH (see Pringault et al., 2016); this channel zone between lagoon and  
125 marine bay is among the most polluted sites in the Bizerte lagoon (Ben Said et al.,  
126 2010; Yoshida et al., 2002). Just after sampling, the sediment was sieved (2 mm mesh)  
127 to remove large particles and stored in a cool box until its return to the laboratory. In  
128 the laboratory, sediment was mixed with channel water (1:4 w/v ratio), and sediment  
129 was gently stirred for 8 h. After a 12 h settling period, the overlying solution, called  
130 thereafter as "elutriate", was smoothly siphoned off and stored in the dark at 4 °C  
131 until use in microcosm incubations.

## 132 *2.2 Incubation procedure*

133 During the four studied seasons, seawater was incubated for four days in 9 L glass  
134 microcosms (22.5 cm diameter and 23 cm height), covered with a quartz lid to allow  
135 full penetration of the natural sunlight, including UV radiation, according to the  
136 protocol described in Pringault et al. (2016) and briefly detailed below. A series of  
137 three microcosms per treatment was filled with 6 L of sample water (L and O

138 stations; control microcosms, C). Another series of three microcosms was filled with  
139 4.5 L of sample water (L and O stations) completed with 1.5 L of elutriate to get a  
140 final dilution of 25% (contaminated water microcosms; CW). The third series of three  
141 microcosms (artificial contaminated water microcosms; ACW) was filled with 6 L (L  
142 and O stations) and spiked an artificial solution of pesticides and some metabolites  
143 (acetochlore, alachlore, DCPU 1-(3,4-dichlorophenyl)urea, DIA deisopropylatrazine,  
144 diuron, linuron and simazine) and metals (Ni, Cu, Zn, Cd, As, Pb). Final  
145 concentration of pesticides and metals were adjusted to mimic the concentrations  
146 observed in elutriate as well as possible (Table S1, supplementary materials) (see  
147 Bancon-Montigny et al., 2019, for more details). All microcosms, performed in  
148 triplicate, were incubated outside under natural sunlight in a 3 m<sup>3</sup> pool where  
149 seawater was circulating (open system) to maintain *in situ* water temperature. Two  
150 light sensors (Skye, England), measuring incident photosynthetically active radiation  
151 (PAR; 400–700 nm, quantum SKP 215) and ultraviolet A and B radiation (315–380  
152 nm, SKU 420 and 280–315 nm, SKU 430, respectively), monitored the light received at  
153 the surface of microcosms.

### 154 2.3 Chemical analysis

155 Chemical analysis of the nutrients, Chlorophyll *a* (Chl*a*) dissolved organic carbon  
156 (DOC), organic contaminants (pesticides and polycyclic aromatic hydrocarbons) and  
157 metal trace elements (MTE) were performed using the analytical protocols described  
158 in Bancon-Montigny et al. (2019) and Pringault et al. (2016).

### 159 2.4 Phytoplankton counts

160 Water samples were collected from each microcosm (245 mL) at the beginning and  
161 the end of the microcosm incubation, then immediately fixed with buffered  
162 formaldehyde at 2% final concentration. Species identification and counts were  
163 realized with a BX60 Olympus microscope coupled with a Progress C3-cooled  
164 Jenoptik camera. Each species was identified to the highest possible taxonomic level  
165 (class, genus or species) according to Halse et al. (1997), Hoppenrath et al. (2009),

166 Kraberg et al. (2010), Viličić (2014) and the WoRMS database  
167 (<http://www.marinespecies.org>). Counts were realized according to the Utermöhl  
168 technique (1958) and the AFNOR standards (2006). Each taxon was counted  
169 individually, except *Cyanophyceae* spp., *Chaetoceros* spp., *Pseudonitzschia* spp.,  
170 *Prymnesiophyceae* spp., *Euglenoidea* spp. and many flagellates, which were grouped  
171 into general taxa because of species identification difficulties. Two groups of  
172 flagellates were distinguished by size class, <10 µm and 10–15 µm, without  
173 distinction of the taxonomic class.

#### 174 2.5 Metabolic activities

175 Community respiration (CR) and primary production (PP) was measured using an  
176 oxygen microelectrode (Unisense, Denmark) following the protocols described by  
177 Briand et al. (2004) for CR, and by Pringault et al. (2007) for PP. Water samples (1  
178 mL) were placed in gas-tight glass microchambers and immersed in a water bath  
179 with controlled temperature and light (*in situ* temperature and *in situ* light).  
180 Dissolved O<sub>2</sub> concentration was measured a minimum of four times during the 8–12  
181 h of incubation in the vessels. Community respiration and net production (NP) were  
182 deduced from the linear regression established on measurement points in the dark  
183 for CR and in the light for NP. Primary production was then computed as the sum  
184 CR + NP. Bacterial respiration (BR) was expressed in µg C L<sup>-1</sup> h<sup>-1</sup> using a respiratory  
185 quotient of 1 (del Giorgio and Cole, 1998) and assuming that BR represented 30% of  
186 the total community respiration (Aranguren-Gassis et al., 2012). Bacterial respiration  
187 was used to estimate bacterial carbon demand (BCD), which represents the sum BP +  
188 BR, BP representing bacterial production measured by <sup>3</sup>H-thymidine incorporation  
189 (Smith and Azam, 1992). A detailed protocol can be found in Pringault et al. (2016).  
190 Thymidine incorporation rates were converted into carbon production using the  
191 conversion factors of 2.10<sup>18</sup> cells produced per mole of thymidine incorporated and  
192 20 fg C per cell (Lee and Bong, 2008).

## 193 2.6 Statistical analysis

194 Coupling between phytoplankton and bacterioplankton was estimated using the  
195 approach proposed by Moran et al. (2002) by calculating the linear regression  
196 between PP and BCD with a regression significance when  $p < 0.05$ . Analysis of  
197 variance (ANOVA) were performed to test the significance of the differences  
198 observed. Prior to analysis of variance, normality (Shapiro–Wilk test) and  
199 homogeneity of variance (Levene's test) were checked. When these conditions were  
200 not met, differences were evaluated using the nonparametric Kruskal–Wallis  
201 ANOVA test. Posteriori paired multiple-comparisons were then performed using the  
202 Tukey HSD (honestly significant difference) test. ANOVA tests and Tukey's HSD  
203 tests were carried out with the level of significance set at  $p < 0.05$  (Statgraphics  
204 Centurion XV software).

## 205 3 Results

### 206 3.1 In situ conditions

207 The environmental conditions (Table 1) were significantly different depending on the  
208 periods (seasons) and sampling sites (L and O stations). Chl $a$ , nutrients and DOC  
209 were significantly ( $p < 0.05$ ) higher in lagoon waters. Metabolic activities exhibited  
210 also differences according to season and sites. The highest rate of primary production  
211 (PP) were observed during summer for offshore waters whereas community  
212 respiration (CR) was maximal during fall for the lagoon system (Table 1). In the  
213 lagoon ecosystem, CR was always higher than PP (PP:CR  $\leq 0.3$ ), while PP largely  
214 exceeded CR (PP:CR = 1.5–1.6) in winter and summer in offshore waters. Seasonal  
215 variations for environmental conditions were observed for both sites; minimum  
216 values for phytoplankton biomass were observed during winter, whereas maximum  
217 Chl $a$  were observed during spring for offshore and fall for lagoon waters. Metabolic  
218 activities also showed significant variations ( $p < 0.05$ ) according to the season, with  
219 maximum rates during summer and fall and minimum values during winter for both

220 sites. As observed for environmental conditions, the phytoplankton community  
221 structure (expressed as relative abundance of the main phytoplankton groups) also  
222 exhibited significant variations according to the season and sampling site (two  
223 factors ANOVA  $p < 0.05$ ) (Fig. 1). Lagoon and offshore waters were dominated by  
224 Ochrophyta in summer, whereas dinoflagellates were the dominant group in spring  
225 for lagoon, and in fall for offshore waters. Winter was characterized by the  
226 dominance of Haptophyta in offshore waters and flagellates in lagoon waters.  
227 Data regarding the chemical contamination of both sites by pesticides and metal trace  
228 elements (Table S1, supplementary materials) are given in more details in Bancon-  
229 Montigny et al. (2019).

### 230 *3.2 Chlorophyll a, phytoplankton groups and metabolic activities during microcosm* 231 *incubation*

232 Incubation in microcosms during 96 h provoked significant changes for biomass (Fig.  
233 2 and 3) and metabolic activities (Fig. 4 and 5) in all treatments, whatever the season  
234 and the sampling site. As a general rule, incubation with elutriate (CW) provoked a  
235 strong increase in Chl $a$  and bacterial biomass (Fig. 2 and 3) especially in offshore  
236 waters, with values up to 5 times the concentration (Chl $a$ ) observed in controls (C) at  
237 the end of the incubation period. It worth noticing that sediment elutriate (CW)  
238 provoked a significant increase of bacterial biomass relative to control (Student Test,  
239  $p < 0.05$ ), noticed at the beginning of the incubation (Fig. 3). The changes in Chl $a$   
240 concentrations were also accompanied by modifications of the phytoplankton  
241 community composition (relative abundance of the main phytoplankton groups)  
242 with significant differences among treatments (one factor ANOVA  $p < 0.05$ ) (Fig. S2  
243 Supplementary materials).. Marked changes were noticed, especially during winter  
244 and fall, in both ecosystems. As observed for Chl $a$ , the main significant modifications  
245 of phytoplankton composition relative to C microcosm were observed with elutriate  
246 incubation (CW). During summer, offshore and lagoon waters were dominated by  
247 Ochrophyta, representing up to 80% of the phytoplankton relative abundance for L

248 station; this dominance was not modified by the presence of sediment elutriate (CW).  
249 In contrast, in the fall, when the dinoflagellates and flagellates (lagoon and offshore  
250 sites) were the dominant phytoplankton groups, the impact of the elutriate and  
251 artificial contaminant cocktail on phytoplankton structure were more pronounced  
252 with significant ( $p < 0.05$ ) modifications of the relative abundance of the main  
253 phytoplankton groups, relative to the phytoplankton composition observed in the  
254 controls. A similar pattern was also observed during winter when the flagellates  
255 (lagoon) and the Haptophyta (offshore) were the dominant phytoplankton groups.  
256 Upon sediment elutriate, Cryptophyta became the most abundant group in lagoon  
257 waters (in fall and winter), whereas Ochrophyta (winter) and flagellates (winter and  
258 fall) dominated in offshore waters. The large increases in Chl $a$  concentrations in CW  
259 treatments were concomitant with an important stimulation of primary production,  
260 especially in offshore waters, well superior to those observed in lagoon waters (Fig. 4  
261 and 5). The most pronounced effects of elutriate (CW) on Chl $a$  and metabolic  
262 activities were observed during spring and summer for offshore waters and during  
263 spring for lagoon. As a general rule, changes were less marked in ACW microcosms,  
264 where the artificial spiking with a cocktail of pesticides and metals provoked minor  
265 changes regarding phytoplankton biomass as well as in the phytoplankton  
266 composition. The most pronounced significant ( $p < 0.05$ ) inhibition effects were  
267 observed in offshore waters, with final phytoplankton biomass representing less than  
268 50% and 30% of the control values for fall and winter, respectively. For lagoon  
269 waters, inhibition of phytoplankton biomass by ACW treatment was even less  
270 pronounced, with a significant ( $p < 0.05$ ) inhibition (around 50%) only observed  
271 during winter. Surprisingly, during the fall Chl $a$  concentrations were higher in ACW  
272 microcosms relative to control conditions.

273 Regarding the metabolic activities (PP and BCD), spiking with the artificial  
274 contaminant cocktail provoked significant inhibition of PP and to a less extent of  
275 BCD in offshore waters, with values representing on some occasions less than 50% of  
276 the control values. As observed for Chl $a$  concentrations, the impact of artificial

277 contamination (ACW) was significantly less pronounced in lagoon waters for both  
278 PP and BCD (two factors ANOVA  $p < 0.05$ ). The most significant ( $p < 0.05$ ) effects on  
279 both metabolic activities were observed during summer with up to 50% for PP and  
280 30% for BCD of inhibition relative to control conditions.

### 281 3.3 Bacterio–Phytoplankton coupling

282 The metabolic activities (PP and BCD) measured during microcosm incubation were  
283 used to estimate the strength of the phytoplankton–bacterioplankton coupling  
284 depending on the treatments. Data measured at each season were pooled, and the  
285 linear relationship between PP and BCD was estimated according to the  
286 experimental conditions imposed (Fig. 6 and 7). In offshore waters, for the control  
287 conditions, PP and BCD were strongly positively correlated ( $r = 0.806$ ,  $p < 0.0001$ ) with  
288 a slope very close to 1 ( $1.05 \pm 0.10$ ). In contrast, for the C treatment in lagoon waters,  
289 the linear relationship between PP and BCD was weaker ( $r = 0.504$ ) with a significant  
290 decrease of the slope,  $0.35 \pm 0.08$ , relative to the value measured in offshore waters  
291 for the same conditions. Incubation with elutriate (CW) and with an artificial cocktail  
292 of pesticides and MTE (ACW) both provoked similar effects in offshore and lagoon  
293 waters; a significant reduction of the slope and of the correlation coefficient, relative  
294 to the respective values measured in C treatment (Student test,  $p < 0.05$ ). The most  
295 pronounced differences were measured for offshore waters in CW microcosms, with  
296 a slope of  $0.27 \pm 0.07$  ( $r = 0.472$ ,  $p = 0.0007$ ), relative to  $1.05 \pm 0.10$  measured in C  
297 microcosms. In lagoon waters, incubation with elutriate (CW) also resulted in a  
298 significant reduction of the slope with a value of  $0.21 \pm 0.06$  relative to  $0.37 \pm 0.08$   
299 measured in C microcosms. Changes were less marked in ACW microcosm for both  
300 ecosystems, nevertheless, the decrease of the slope relative to the control value  
301 remained significant (Student test,  $p < 0.05$ ).

## 302 4 Discussion

303 The aim of the present study was to assess the consequences of chemical  
304 contamination on the coupling between phytoplankton and bacterioplankton in two  
305 contrasting ecosystems, an open oceanic ecosystem versus a semi-enclosed lagoon  
306 ecosystem. The contaminated sediment used for sediment resuspension was sampled  
307 in the lagoon channel, and according to the water circulation and the wind regime  
308 (Harzallah, 2003), contaminants released by sediment resuspension following strong  
309 gusts or stormy events can affect the pelagic communities in both studied  
310 ecosystems. The two studied Mediterranean ecosystems exhibited contrasting  
311 environmental conditions during the four seasons (Table 1) that were concomitant to  
312 marked differences in chemical contamination (Table S1, supplementary materials);  
313 offshore waters were significantly less contaminated than the lagoon (Bancon-  
314 Montigny et al., 2019). These two contrasted environments allowed testing of the  
315 hypothesis that the bacterial and phytoplankton response to chemical contamination  
316 would be different in the two sampling sites, with possible consequences for the  
317 coupling between both compartments, considering the possible selective pressure  
318 and the adaptation that can be triggered by the chemical environment, according to  
319 the concept of pollution-induced community tolerance (Blanck, 2002).

### 320 4.1 *In situ biomass and metabolic activities*

321 The two Mediterranean ecosystems, lagoon and offshore, were sampled during the  
322 four seasons. The metabolic rates (CR: community respiration and PP: primary  
323 production) and the phytoplankton biomass measured in the present study were in  
324 agreement with previous studies in this region (Meddeb et al., 2018; Sakka Hlaili et  
325 al., 2008) and were comparable to reports from similar ecosystems in the  
326 Mediterranean basin (Ciavatta et al., 2008; Pérez-Ruzafa et al., 2005; Sfriso et al.,  
327 2003), with seasonal variations for both marine and lagoon waters (Salhi et al., 2018).  
328 As a general rule, phytoplankton biomass and metabolic activities involved in the C  
329 cycle (PP, CR and BCD) were significantly higher in lagoon waters relative to

330 offshore waters except for PP in summer were higher rates were observed in offshore  
331 waters relative to lagoon. Similarly, significant differences were observed in the  
332 phytoplankton composition between the two ecosystems, confirming the particular  
333 trophic status played by semi-enclosed ecosystems relative to their counterpart  
334 marine adjacent waters (Newton et al., 2014). Despite a large open channel to coastal  
335 sea, the watershed influence and the very long water residence time (> 200 days)  
336 observed in the Bizerte lagoon (Bejaoui et al., 2017) relative to the offshore adjacent  
337 waters (<20 days; (Sayol et al., 2013), allows accumulation of nutrients, DOC and  
338 chemical contaminants within the lagoon. In addition, temperature and salinity  
339 variations showed distinct patterns with lower temperature and salinity during  
340 winter and spring in lagoon waters (compared to the offshore situation), whereas an  
341 opposite situation was observed during the summer, where temperature and salinity  
342 values were maximal in the lagoon system (Béjaoui et al., 2016). These distinct  
343 environmental conditions favored the development of different planktonic species in  
344 the lagoon environment relative to the marine adjacent waters, as confirmed by the  
345 phytoplankton composition (Fig. 1) but also for the bacterioplankton structure (Goni-  
346 Urriza et al., 2018).

347 Seasonal variations were observed for the phytoplankton diversity and the metabolic  
348 activities involved in the C cycle, suggesting a strong impact on the organic matter  
349 cycle with cascading consequences for the carbon transfer within the trophic web.  
350 Using a modeling approach to estimate the flux and exchanges between the different  
351 components of the pelagic marine trophic web, it was possible to assess the  
352 functioning and stability of the trophic web (Sakka Hlaili et al., 2014). Meddeb et al.  
353 (2019, 2018) reported food-web structure differences between the marine and lagoon  
354 ecosystems in the Bizerte area, with herbivorous and multivorous dominant carbon  
355 pathways in the marine and the lagoon, respectively. The ecological indices, like the  
356 total system throughput (Niquil et al., 2012), used as proxies of the ecosystem  
357 functioning, suggests a more efficient carbon transfer in the open marine ecosystem,  
358 whereas the lagoon environment was more active, but more stressed and less

359 organized (Meddeb et al., 2018). According to the ratio between primary production  
360 and community respiration (PP:CR; Table 1), a strong autotrophic activity occurred  
361 in offshore waters during winter and summer seasons, while the lagoon ecosystem  
362 seemed to be mainly driven by heterotrophic activity (CR>>PP); this supports the  
363 assumption that an intense recycling of organic matter occurred in the lagoon waters  
364 through the bacterial use of organic matter derived from the sediments and the  
365 watershed depending on the bioavailability of OM which strongly affect its  
366 mineralization by bacteria (Blanchet et al., 2017; Decho and Gutierrez, 2017).

#### 367 4.2 Impact of contamination on diversity and function

368 Microbial communities from open (offshore) and lagoon waters were exposed to  
369 sediment elutriate and to a cocktail of chemical contaminants (MTE and pesticides).  
370 The contaminants cocktail was designed from the main compounds identified and  
371 analysed in sediment elutriate (Table S1, supplementary materials) in order to assess  
372 the toxic effects of these contaminants without confounding chemical factors,  
373 nutrients and/or dissolved organic matter that are released during sediment  
374 resuspension (Pringault et al., 2016). Nutrients and organic matter are known to  
375 strongly mitigate the toxicity of pesticides and TME against microorganisms, mainly  
376 attributed to a decrease of contaminant bioavailability by adsorption on dissolved  
377 organic matter (Boyd et al., 2005; Coquillé et al., 2018; Lorenzo et al., 2002) and  
378 changes in the nutrient status, toxicity increasing under nutrient limitation (Chia et  
379 al., 2016; Miao and Wang, 2006). For offshore waters, spiking with pesticides and  
380 MTE resulted in an inhibition of phytoplankton biomass as well as a reduction of  
381 metabolic activities involved in the carbon cycle (PP and BCD). Previous studies have  
382 observed that environmentally relevant concentrations of pesticides, within the range  
383 of ng L<sup>-1</sup> similar to those observed in the artificial contamination (ACW), can severely  
384 affect primary production of microalgae with a strong reduction of photosynthetic  
385 efficiency up to 50% (Devilla et al., 2005). Similarly, MTE can also strongly reduce  
386 phytoplankton biomass and primary production of natural phytoplankton

387 communities (Pérez et al., 2006) at  $\mu\text{g L}^{-1}$  concentrations, in the same order of  
388 magnitude than those used in ACW microcosm for the artificial spiking. As a general  
389 rule, when assessing the impact of pollution on the marine carbon cycle and  
390 ecosystem functioning, primary production is more affected by chemical  
391 contamination than respiration, especially upon herbicide or metal trace  
392 contamination, with a strong reduction of  $\text{O}_2$  production or  $\text{CO}_2$  fixation, whereas  
393 respiration remains less impacted (Johnston et al., 2015). Nevertheless, the inhibition  
394 of metabolic processes or biomass production observed when pelagic communities  
395 are exposed to a single molecule can be alleviated when contaminants are used in  
396 mixture (Johnston et al., 2015), due to the antagonist effects than pollutant can exert  
397 between them (Franklin et al., 2002; Sharma et al., 1999). This hypothesis can be  
398 advanced to explain the absence of clear toxic effects on phytoplankton biomass and  
399 metabolic rates of the artificial mixture from pesticides and metal trace elements  
400 (ACW) for the lagoon waters. In addition, as mentioned before, the lagoon ecosystem  
401 is characterized by a very long water residence time (>200 days), which favors the  
402 accumulation of pollutants and nutrients. This long period can promote the  
403 development of a community resistant to chemical pollution according to the concept  
404 of pollution induced community tolerance (Blanck, 2002) and observed for  
405 meiobentic communities (Boufahja and Semprucci, 2015). Furthermore, the  
406 accumulation of pollutants and nutrients in lagoon waters results in weak  
407 enrichment when exposed to the artificial cocktail of contaminants (Bancon-  
408 Montigny et al., 2019), reducing the potential toxicity of the added contaminants for  
409 the microbial lagoon communities.

410 It is worth noting that the toxic effects observed in offshore waters upon artificial  
411 spiking (ACW) were alleviated when the contaminants present in the sediment  
412 elutriate were accompanied by nutrients and dissolved organic matter (CW). When  
413 offshore waters were exposed to elutriate (CW), phytoplankton biomass and  
414 bacterial abundances strongly increased as well as metabolic activities (primary  
415 production and bacterial carbon demand). The strong stimulation of BCD in CW

416 treatment (Figs. 4) can partly be explained by the external supply of bacteria by  
417 sediment resuspension, as indicated by the significant increase in bacterial abundance  
418 observed at the onset of microcosm incubation in CW relative to control (Fig. 3).  
419 Nevertheless, previous study showed that this increase in bacterial biomass upon  
420 sediment resuspension provoked minor changes on bacterial diversity and  
421 community structure (Goni-Urriza et al., 2018). Stimulation of biomass and carbon  
422 production was most obvious during spring and fall when flagellates and  
423 dinoflagellates dominated the phytoplankton community. In anthropogenic coastal  
424 environment, sediment resuspension results in the release in the water column of a  
425 trail of contaminants, nutrients and dissolved organic matter as well as benthic  
426 microorganisms (Roberts, 2012), leading to enhancement of primary production and  
427 phytoplankton biomass, directly by increasing the number of microalgae cells and  
428 indirectly by stimulating, through nutrients supply, phytoplankton growth (Guallar  
429 and Flos, 2017). In the present study, sediment elutriate was artificially mixed (1/4  
430 final volume) with offshore or lagoon waters. This mixing resulted in important  
431 enrichment in nutrients (especially in dissolved ammonium and silicates), dissolved  
432 organic matter, metal trace elements (Mn, Fe, Mo, Zn and Ni) and pesticides  
433 (simazine, alachlor, deisopropylatrazine and 1-(3,4-dichlorophenyl)urea) (Bancon-  
434 Montigny et al., 2019) and exogenous bacteria by sediment resuspension (Goni-  
435 Urriza et al., 2018). Enrichments in nutrients, DOM and chemical contaminants were  
436 more pronounced in offshore waters relative to lagoon waters, explained by higher  
437 initial concentrations of compounds in the lagoon; thus the addition of sediment  
438 elutriate with their compounds does not change the chemical status of the lagoon  
439 significantly (Bancon-Montigny et al., 2019). For offshore waters, the metabolic and  
440 biomass changes upon elutriate were more pronounced during spring and fall with  
441 *Chla* concentration and primary production up to five times more than observed in  
442 the control. Spring and fall were characterized by important enrichment in nutrients,  
443 iron and manganese upon elutriate exposure (CW) whereas chemical contamination  
444 by Ni, Zn and pesticides were relatively low (Bancon-Montigny et al., 2019). In

445 contrast, during winter, the impact of elutriate was reduced for both phytoplankton  
446 biomass and metabolic activities. This period was characterized by a strong  
447 enrichment in ammonium (33 $\mu$ M relative to *in situ* 0.4  $\mu$ M before mixing)  
448 concomitant with enrichment in pesticides and Ni, with twice the *in situ*  
449 concentration (Bancon-Montigny et al., 2019). It is very likely that this strong  
450 enrichment in ammonium and chemical contaminants did not promote favorable  
451 conditions for phytoplankton growth, as indicated by the low Chl*a* concentration and  
452 PP rates achieved after exposure relative to control values.

#### 453 4.3 Bacterio–phytoplankton coupling and chemical contamination

454 Primary production (PP) and bacterial carbon demand were strongly correlated ( $r =$   
455 0.806,  $p < 0.0001$ ) in control treatments for offshore waters with a slope close to 1  
456 (1.046) suggesting a tight coupling between phytoplankton and bacterioplankton; the  
457 carbon demand of the heterotrophic compartment could be totally sustained by  
458 primary production (Morán et al., 2002). For lagoon waters, despite a significant  
459 positive linear relationship, the slope was strongly lower (0.35) as well as the  
460 correlation coefficient of the slope. This clearly indicated a weaker interdependency  
461 between both compartments, especially during fall and winter. In these two seasons,  
462 lagoon BCD strongly exceeded primary production, consequently, bacterioplankton  
463 must rely on allochthonous organic carbon to sustain its carbon requirements. These  
464 differences between lagoon and offshore situations in the bacterio–phytoplankton  
465 coupling are in agreement with the main observations reported in marine  
466 ecosystems, with tighter coupling and interdependency between bacterioplankton  
467 and phytoplankton increasing from the coast to the open ocean (Fouilland et al.,  
468 2018; Morán et al., 2002).

469 When lagoon or offshore waters were incubated with a mixture of contaminants  
470 (pesticides and MTE, ACW), the linear relationships between both compartments  
471 slightly changed, with, in both cases, a significant decrease of the slope and the  
472 coefficient correlation, suggesting a decrease in the carbon dependency with respect

473 to primary production to sustain the heterotrophic metabolism. These changes on the  
474 carbon dependency were more pronounced during winter and fall in offshore waters  
475 with a strong reduction of primary reduction when Haptophyta or dinoflagellates  
476 dominated the phytoplankton community. Inhibition of primary production were  
477 concomitant to significant changes of the relative abundance of the main  
478 phytoplankton groups. Interactions between phytoplankton and bacteria are species-  
479 dependent (Seymour et al., 2017) and a shift in phytoplankton community structure  
480 can modify bacterial production, bacterial abundance and bacterial community  
481 structure (Camarena-Gómez et al., 2018) with consequently incidences on the  
482 bacterio-phytoplankton coupling. In addition, under chemical contamination,  
483 phytoplankton can release dissolved organic matter as extracellular polymeric  
484 substances (EPS), binding out of the cell, the toxic compounds like metal trace  
485 elements, or pesticides (Decho and Gutierrez, 2017; Naveed et al., 2019). Such defense  
486 mechanisms were reported in phytoplankton cultures (Decho and Gutierrez, 2017;  
487 Herzi et al., 2013a, 2013b) as well as in natural phytoplankton communities  
488 (Rochelle-Newall et al., 2008, 2011) or peryphyton biofilm (Ivorra et al., 2000). The  
489 binding of chemical compounds on the phytoplankton-derived dissolved organic  
490 matter can modify its composition and its physicochemical properties (Naveed et al.,  
491 2019) with incidences on its bioavailability and lability (Enya et al., 2020; Wada and  
492 Suzuki, 2011), decreasing so the potential utilization by the heterotrophic  
493 compartment (Seto et al., 2013), eventually loosening the coupling between both  
494 compartments.

495 When the natural community from offshore and lagoon waters were incubated with  
496 elutriate (CW), the slope and the correlation coefficient of the linear relationships  
497 between BCD and PP strongly decreased relative to their corresponding control  
498 value, especially for offshore waters, with a slope of 0.27 ( $r = 0.472$ ) relative to 1.05 ( $r$   
499  $= 0.806$ ) in the control. For lagoon waters, the slope was even lower (0.18) relative to  
500 0.35 in the control treatment. These very low values, clearly indicated that the  
501 dependency between both compartments was weak when exposed to sediment

502 elutriate. Sediment resuspension released a significant amount of organic matter,  
503 nutrients and bacteria, which resulted in a strong enrichment in DOM, nutrients and  
504 bacterial abundance in offshore and lagoon waters, knowing the low concentrations  
505 of DOM and nutrients measured *in situ* (Bancon-Montigny et al., 2019). The release of  
506 DOM and nutrients consecutive to sediment resuspension is known to stimulate both  
507 phytoplankton and bacterioplankton metabolisms (Roberts, 2012; Uchimiya et al.,  
508 2016) with serious consequences on the biogeochemical cycles (Cotner, 2000)  
509 depending on the bioavailability of OM that strongly affects its recycling (Blanchet et  
510 al., 2017; Decho and Gutierrez, 2017). These conditions favor a loosening of the  
511 coupling between phytoplankton and bacterioplankton; the heterotrophic  
512 compartment may have access to external carbon sources, decreasing its potential  
513 dependency on the phytoplankton exudates. In addition, nutrients released by  
514 sediment can also modify the phytoplankton composition by alleviating nutrient  
515 competition within the community and allowing the growth of microalgae taxa  
516 previously nutrient-limited. A previous study has shown that sediment resuspension  
517 may alter the synchrony between bacterial and phytoplanktonic compartments  
518 (Goni-Urriza et al., 2018). Consequently, the tight coupling observed in offshore  
519 waters between phytoplankton and bacterioplankton can be loosened upon sediment  
520 resuspension by several non exclusive factors i) a modification of the phytoplankton  
521 composition through the release of sediment nutrients, altering so interactions with  
522 the heterotrophic bacteria, ii) the supply of an external source of labile carbon that  
523 can be preferentially used by heterotrophs, iii) the supply of exogenous bacteria that  
524 can enhance the overall bacterial carbon demand.

## 525 5 Conclusion

526 The result of the present study clearly showed that phytoplankton and  
527 bacterioplankton were strongly coupled in offshore waters while the coupling was  
528 less obvious in the lagoon ecosystem. We clearly showed that the presence of  
529 contaminants altered the interactions between phytoplankton and bacterioplankton,

530 with a loosening of the coupling observed in offshore waters. This fact was even  
531 more marked upon sediment resuspension due to the strong supplies of nutrients,  
532 dissolved organic carbon and exogenous bacteria. Considering the climate scenarios  
533 in the Mediterranean basin that predict an increase of storms and paroxysmic events  
534 (Cramer et al., 2018), with consequently higher occurrences of sediment  
535 resuspension, the functioning of coastal ecosystems will be severely impacted by  
536 strong alterations of the interactions between bacterioplankton and phytoplankton  
537 with non-negligible consequences for the carbon cycle and the CO<sub>2</sub> emissions.

## 538 6 Acknowledgments

539 This study was supported by the RISCO project (ANR-13-CESA-0001), funded by the  
540 French National Agency for Research (ANR) and by the Joint International  
541 Laboratory COSYS-Med. English grammar and syntax of the manuscript were  
542 revised by Proof-Reading-Service.com.

## 543 7 References cited

- 544 AFNOR, AFNOR, 2006. Norme guide pour le dénombrement du phytoplancton par  
545 microscopie inverse (méthode Utermöhl). , Comité Technique CEN/TC 230  
546 "Analyse de l'eau."
- 547 Aranguren-Gassis, M., Teira, E., Serret, P., Martinez-Garcia, S., Fernandez, E., 2012.  
548 Potential overestimation of bacterial respiration rates in oligotrophic plankton  
549 communities. *Mar. Ecol. Prog. Ser.* 453, 1–10. <https://doi.org/10.3354/meps09707>
- 550 Arvanitidis, C., Somerfield, P.J., Chatzigeorgiou, G., Reizopoulou, S., Kevrekidis, T.,  
551 Eleftheriou, A., 2009. Do multivariate analyses incorporating changes in pattern  
552 across taxonomic levels reveal anthropogenic stress in Mediterranean lagoons? *J.*  
553 *Exp. Mar. Bio. Ecol.* 369, 100–109. <https://doi.org/10.1016/j.jembe.2008.10.032>
- 554 Bancon-Montigny, C., Gonzalez, C., Delpoux, S., Avenzac, M., Spinelli, S., Mhadhbi,  
555 T., Mejri, K., Hlaili, A.S., Pringault, O., 2019. Seasonal changes of chemical  
556 contamination in coastal waters during sediment resuspension. *Chemosphere*

557 235, 651–661. <https://doi.org/10.1016/j.chemosphere.2019.06.213>

558 Barhoumi, B., LeMenach, K., Devier, M.H., Ben Ameer, W., Etcheber, H., Budzinski,  
559 H., Cachot, J., Driss, M.R., 2014a. Polycyclic aromatic hydrocarbons (PAHs) in  
560 surface sediments from the Bizerte Lagoon, Tunisia: levels, sources, and  
561 toxicological significance. *Environ. Monit. Assess.* 186, 2653–2669.  
562 <https://doi.org/10.1007/s10661-013-3569-5>

563 Barhoumi, B., LeMenach, K., Devier, M.H., El Megdiche, Y., Hammami, B., Ben  
564 Ameer, W., Ben Hassine, S., Cachot, J., Budzinski, H., Driss, M.R., 2014b.  
565 Distribution and ecological risk of polychlorinated biphenyls (PCBs) and  
566 organochlorine pesticides (OCPs) in surface sediments from the Bizerte lagoon,  
567 Tunisia. *Environ. Sci. Pollut. Res.* 21, 6290–6302. [https://doi.org/10.1007/s11356-](https://doi.org/10.1007/s11356-013-1709-7)  
568 [013-1709-7](https://doi.org/10.1007/s11356-013-1709-7)

569 Béjaoui, B., Armi, Z., Ottaviani, E., Barelli, E., Gargouri-Ellouz, E., Chérif, R., Turki,  
570 S., Solidoro, C., Aleya, L., 2016. Random Forest model and TRIX used in  
571 combination to assess and diagnose the trophic status of Bizerte Lagoon,  
572 southern Mediterranean. *Ecol. Indic.* 71, 293–301.  
573 <https://doi.org/10.1016/j.ecolind.2016.07.010>

574 Bejaoui, B., Solidoro, C., Harzallah, A., Chevalier, C., Chapelle, A., Zaaboub, N.,  
575 Aleya, L., 2017. 3D modeling of phytoplankton seasonal variation and nutrient  
576 budget in a southern Mediterranean Lagoon. *Mar. Pollut. Bull.* 114, 962–976.  
577 <https://doi.org/10.1016/j.marpolbul.2016.11.001>

578 Ben Said, O., Goni-Urriza, M., El Bour, M., Aissa, P., Duran, R., 2010. Bacterial  
579 Community Structure of Sediments of the Bizerte Lagoon (Tunisia), a Southern  
580 Mediterranean Coastal Anthropized Lagoon. *Microb. Ecol.* 59, 445–456.

581 Berglund, J., Müren, U., Båmstedt, U., Andersson, A., 2007. Efficiency of a  
582 phytoplankton-based and a bacteria-based food web in a pelagic marine system.  
583 *Limnol. Oceanogr.* 52, 121–131. <https://doi.org/10.4319/lo.2007.52.1.0121>

584 Blanchet, M., Pringault, O., Panagiotopoulos, C., Lefèvre, D., Charrière, B., Ghiglione,  
585 J.-F., Fernandez, C., Aparicio, F.L., Marrasé, C., Catala, P., Oriol, L., Caparros, J.,

586 Joux, F., 2017. When riverine dissolved organic matter (DOM) meets labile DOM  
587 in coastal waters: changes in bacterial community activity and composition.  
588 *Aquat. Sci.* 79, 27–43. <https://doi.org/10.1007/s00027-016-0477-0>

589 Blanck, H., 2002. A critical review of procedures and approaches used for assessing  
590 pollution-induced community tolerance (PICT) in biotic communities. *Hum.*  
591 *Ecol. Risk Assess.* 8, 1003–1034.

592 Bonnet, C., Babut, M., Ferard, J.F., Martel, L., Garric, J., 2000. Assessing the potential  
593 toxicity of resuspended sediment. *Environ. Toxicol. Chem.* 19, 1290–1296.

594 Boufahja, F., Semprucci, F., 2015. Stress-induced selection of a single species from an  
595 entire meiobenthic nematode assemblage: is this possible using iron enrichment  
596 and does pre-exposure affect the ease of the process? *Environ. Sci. Pollut. Res.*  
597 22, 1979–1998. <https://doi.org/10.1007/s11356-014-3479-2>

598 Bouvy, M., Arfi, R., Cecchi, P., Corbin, D., Pagano, M., Saint-Jean, L., Thomas, S.,  
599 1998. Trophic coupling between bacterial and phytoplanktonic compartments in  
600 shallow tropical reservoirs (Ivory Coast, West Africa). *Aquat. Microb. Ecol.* 15,  
601 25–37. <https://doi.org/10.3354/ame015025>

602 Boyd, T.J., Wolgast, D.M., Rivera-Duarte, I., Holm-Hansen, O., Hewes, C.D., Zirino,  
603 A., Chadwick, D.B., 2005. Effects of dissolved and complexed copper on  
604 heterotrophic bacterial production in San Diego Bay. *Microb. Ecol.* 49, 353–366.  
605 <https://doi.org/10.1007/s00248-003-1065-0>

606 Briand, E., Pringault, O., Jacquet, S., Torr ton, J.P., 2004. The use of oxygen  
607 microprobes to measure bacterial respiration for determining bacterioplankton  
608 growth efficiency. *Limnol. Oceanogr. Methods* 2.  
609 <https://doi.org/10.4319/lom.2004.2.406>

610 Camarena-G mez, M.T., Lipsewers, T., Piiparinen, J., Eronen-Rasimus, E., Perez-  
611 Quemali os, D., Hoikkala, L., Sobrino, C., Spilling, K., 2018. Shifts in  
612 phytoplankton community structure modify bacterial production, abundance  
613 and community composition. *Aquat. Microb. Ecol.*  
614 <https://doi.org/10.3354/ame01868>

615 Carr, R.S., Chapman, D.C., 1995. Comparison of Methods for Conducting Marine and  
616 Estuarine Sediment Porewater Toxicity Tests - Extraction, Storage, and Handling  
617 Techniques. *Arch. Environ. Contam. Toxicol.* 28, 69–77.

618 Chia, M.A., Dauda, S., Jibril, T.Z., 2016. Toxicity of atrazine to *Scenedesmus*  
619 *quadricauda* under different nitrogen concentrations. *Environ. Earth Sci.* 75, 960.  
620 <https://doi.org/10.1007/s12665-016-5766-3>

621 Ciavatta, S., Pastres, R., Badetti, C., Ferrari, G., Beck, M.B., 2008. Estimation of  
622 phytoplanktonic production and system respiration from data collected by a  
623 real-time monitoring network in the Lagoon of Venice. *Ecol. Modell.* 212, 28–36.  
624 <https://doi.org/10.1016/j.ecolmodel.2007.10.025>

625 Cole, J.J., Likens, G.E., Strayer, D.L., 1982. Photosynthetically produced dissolved  
626 organic carbon: An important carbon source for planktonic bacteria. *Limnol.*  
627 *Oceanogr.* 27, 1080–1090. <https://doi.org/10.4319/lo.1982.27.6.1080>

628 Coquillé, N., Ménard, D., Rouxel, J., Dupraz, V., Éon, M., Pardon, P., Budzinski, H.,  
629 Morin, S., Parlanti, É., Stachowski-Haberkorn, S., 2018. The influence of natural  
630 dissolved organic matter on herbicide toxicity to marine microalgae is species-  
631 dependent. *Aquat. Toxicol.* 198, 103–117.  
632 <https://doi.org/10.1016/j.aquatox.2018.02.019>

633 Cotner, J.B., 2000. Intense winter heterotrophic production stimulated by benthic  
634 resuspension. *Limnol. Oceanogr.* 45, 1672–1676.

635 Cramer, W., Guiot, J., Fader, M., Garrabou, J., Gattuso, J.P., Iglesias, A., Lange, M.A.,  
636 Lionello, P., Llasat, M.C., Paz, S., Peñuelas, J., Snoussi, M., Toreti, A., Tsimplis,  
637 M.N., Xoplaki, E., 2018. Climate change and interconnected risks to sustainable  
638 development in the Mediterranean. *Nat. Clim. Chang.*  
639 <https://doi.org/10.1038/s41558-018-0299-2>

640 Decho, A.W., Gutierrez, T., 2017. Microbial extracellular polymeric substances (EPSs)  
641 in ocean systems. *Front. Microbiol.* 8, 922.  
642 <https://doi.org/10.3389/fmicb.2017.00922>

643 del Giorgio, P.A., Cole, J.J., 1998. Bacterial growth efficiency in natural aquatic

644 systems. *Annu. Rev. Ecol. Syst.* 29, 503–541.

645 Devilla, R.A., Brown, M.T., Donkin, M., Tarran, G.A., Aiken, J., Readman, J.W., 2005.

646 Impact of antifouling booster biocides on single microalgal species and on a

647 natural marine phytoplankton community. *Mar. Ecol. Prog. Ser.* 286, 1–12.

648 <https://doi.org/10.3354/meps286001>

649 Dorigo, U., Berard, A., Rimet, F., Bouchez, A., Montuelle, B., 2010. In situ assessment

650 of periphyton recovery in a river contaminated by pesticides. *Aquat. Toxicol.* 98,

651 396–406. <https://doi.org/10.1016/j.aquatox.2010.03.011>

652 Enya, O., Heaney, N., Iniama, G., Lin, C., 2020. Effects of heavy metals on organic

653 matter decomposition in inundated soils: Microcosm experiment and field

654 examination. *Sci. Total Environ.* 724, 138223.

655 <https://doi.org/10.1016/j.scitotenv.2020.138223>

656 Fouilland, E., Le Flo'ch, E., Brennan, D., Bell, E.M., Lordsmith, S.L., Mcneill, S.,

657 Mitchell, E., Brand, T.D., Elena García-Martín, E., Leakey, R.J.G.G., Flo'ch, E. Le,

658 Brennan, D., Bell, E.M., Lordsmith, S.L., Mcneill, S., Mitchell, E., Brand, T.D.,

659 García-Martín, E.E., Leakey, R.J.G.G., 2018. Assessment of bacterial dependence

660 on marine primary production along a northern latitudinal gradient. *FEMS*

661 *Microbiol. Ecol.* 94, 150. <https://doi.org/10.1093/femsec/fiy150>

662 Fouilland, E., Mostajir, B., 2011. Complementary support for the new ecological

663 concept of “bacterial independence on contemporary phytoplankton

664 production” in oceanic waters. *FEMS Microbiol. Ecol.*

665 <https://doi.org/10.1111/j.1574-6941.2011.01170.x>

666 Fouilland, E., Mostajir, B., 2010. Revisited phytoplanktonic carbon dependency of

667 heterotrophic bacteria in freshwaters, transitional, coastal and oceanic waters.

668 *FEMS Microbiol. Ecol.* 73, 419–429. [https://doi.org/10.1111/j.1574-](https://doi.org/10.1111/j.1574-6941.2010.00896.x)

669 [6941.2010.00896.x](https://doi.org/10.1111/j.1574-6941.2010.00896.x)

670 Fouilland, E., Tolosa, I., Bonnet, D., Bouvier, C., Bouvier, T., Bouvy, M., Got, P., Le

671 Flo'ch, E., Mostajir, B., Roques, C., Sempéré, R., Sime-Ngando, T., Vidussi, F.,

672 2014. Bacterial carbon dependence on freshly produced phytoplankton exudates

673 under different nutrient availability and grazing pressure conditions in coastal  
674 marine waters. *FEMS Microbiol. Ecol.* 87, 757–769. <https://doi.org/10.1111/1574-6941.12262>

676 Franklin, N.M., Stauber, J.L., Lim, R.P., Petocz, P., 2002. Toxicity of metal mixtures to  
677 a tropical freshwater alga (*Chlorella* sp): The effect of interactions between  
678 copper, cadmium, and zinc on metal cell binding and uptake. *Environ. Toxicol.*  
679 *Chem.* 21, 2412–2422. [https://doi.org/10.1897/1551-5028\(2002\)021<2412:tommmta>2.0.co;2](https://doi.org/10.1897/1551-5028(2002)021<2412:tommmta>2.0.co;2)

681 Goni-Urriza, M., Moussard, H., Lafabrie, C., Carre, C., Bouvy, M., Sakka Hlaili, A.,  
682 Pringault, O., 2018. Consequences of contamination on the interactions between  
683 phytoplankton and bacterioplankton. *Chemosphere* 195.  
684 <https://doi.org/10.1016/j.chemosphere.2017.12.053>

685 Guallar, C., Flos, J., 2017. Enhanced primary production in summer and winter-  
686 spring seasons in a populated NW Mediterranean coastal ecosystem. *J. Sea Res.*  
687 120, 1–12. <https://doi.org/10.1016/j.seares.2016.10.009>

688 Halse, G.R., Syvertsen, E.E., Steidinger, K.A., Tangen, K., Throndsen, J., Heimdal,  
689 B.R., 1997. Identifying marine phytoplankton. Academic Press.

690 Hansson, L.A., Nicolle, A., Graneli, W., Hallgren, P., Kritzberg, E., Persson, A., Bjork,  
691 J., Nilsson, P.A., Bronmark, C., 2013. Food-chain length alters community  
692 responses to global change in aquatic systems. *Nat. Clim. Chang.* 3, 228–233.  
693 <https://doi.org/10.1038/nclimate1689>

694 Harzallah, A., 2003. Transport de polluants dans la lagune de Bizerte simulé par un  
695 modèle de circulation de l'eau. *Bull. l'Institut Natl. des Sci. Technol. la Mer,*  
696 *Salambôo* 1–19.

697 Herzi, F., Jean, N., Zhao, H.Y., Mounier, S., Mabrouk, H.H., Sakka Hlaili, A., 2013a.  
698 Copper and cadmium effects on growth and extracellular exudation of the  
699 marine toxic dinoflagellate *Alexandrium catenella*: 3D-fluorescence spectroscopy  
700 approach. *Chemosphere* 93, 1230–1239.  
701 <https://doi.org/https://doi.org/10.1016/j.chemosphere.2013.06.084>

702 Herzi, F., Sakka Hlaili, A., Le Poupon, C., Mabrouk, H.H., Mounier, S., 2013b.  
703 Characterization of exudates released by the marine diatom *Skeletonema*  
704 *costatum* exposed to copper stress: a 3D-fluorescence spectroscopy approach.  
705 *Biomaterials* 26, 773–781. <https://doi.org/10.1007/s10534-013-9655-x>

706 Hochard, S., Pinazo, C., Grenz, C., Evans, J.L.B., Pringault, O., 2010. Impact of  
707 microphytobenthos on the sediment biogeochemical cycles: A modeling  
708 approach. *Ecol. Modell.* 221. <https://doi.org/10.1016/j.ecolmodel.2010.04.002>

709 Hoppe, H.G., Breithaupt, P., Walther, K., Koppe, R., Bleck, S., Sommer, U., Jürgens,  
710 K., 2008. Climate warming in winter affects the coupling between phytoplankton  
711 and bacteria during the spring bloom: A mesocosm study. *Aquat. Microb. Ecol.*  
712 51, 105–115. <https://doi.org/10.3354/ame01198>

713 Hoppenrath, M., Elbrachter, M., Drebes, G., 2009. Selected Microphytoplankton  
714 Species from the North Sea Around Helgoland and Sylt. *Kleine Senckenberg-*  
715 *Reihe* 49, Stuttgart, Germany.

716 Hornick, T., Bach, L.T., Crawford, K.J., Spilling, K., Achterberg, E.P., Woodhouse,  
717 J.N., Schulz, K.G., Brussaard, C.P.D., Riebesell, U., Grossart, H.P., 2017. Ocean  
718 acidification impacts bacteria-phytoplankton coupling at low-nutrient  
719 conditions. *Biogeosciences* 14, 1–15. <https://doi.org/10.5194/bg-14-1-2017>

720 Hulot, F.D., Lacroix, G., Lescher-Moutoué, F.O., Loreau, M., 2000. Functional  
721 diversity governs ecosystem response to nutrient enrichment. *Nature* 405, 340–  
722 344. <https://doi.org/10.1038/35012591>

723 Ivorra, N., Bremer, S., Guasch, H., Kraak, M.H.S., Admiraal, W., 2000. Differences in  
724 the sensitivity of benthic microalgae to Zn and Cd regarding biofilm  
725 development and exposure history. *Environ. Toxicol. Chem.* 19, 1332–1339.

726 Johnston, E.L., Mayer-Pinto, M., Crowe, T.P., 2015. Chemical contaminant effects on  
727 marine ecosystem functioning. *J. Appl. Ecol.* [https://doi.org/10.1111/1365-](https://doi.org/10.1111/1365-2664.12355)  
728 [2664.12355](https://doi.org/10.1111/1365-2664.12355)

729 Johnston, E.L., Roberts, D.A., 2009. Contaminants reduce the richness and evenness  
730 of marine communities: A review and meta-analysis. *Environ. Pollut.*

731 <https://doi.org/10.1016/j.envpol.2009.02.017>

732 Kraberg, A., Baumann, M., Dürselen, C.-D., 2010. Coastal phytoplankton. Photo  
733 guide for Northern European Seas. Verlag, Munchen Germany.

734 Lee, C.W., Bong, C.W., 2008. Bacterial abundance and production, and their relation  
735 to primary production in tropical coastal waters of Peninsular Malaysia. *Mar.*  
736 *Freshw. Res.* 59, 10–21. <https://doi.org/10.1071/MF07099>

737 Lekunberri, I., Calvo-Diaz, A., Teira, E., Moran, X.A.G., Peters, F., Nieto-Cid, M.,  
738 Espinoza-Gonzalez, O., Teixeira, I.G., Gasol, J.M., 2010. Changes in bacterial  
739 activity and community composition caused by exposure to a simulated oil spill  
740 in microcosm and mesocosm experiments. *Aquat. Microb. Ecol.* 59, 169–183.

741 Lorenzo, J.I., Nieto, O., Beiras, R., 2002. Effect of humic acids on speciation and  
742 toxicity of copper to *Paracentrotus lividus* larvae in seawater. *Aquat. Toxicol.* 58,  
743 27–41. [https://doi.org/10.1016/s0166-445x\(01\)00219-3](https://doi.org/10.1016/s0166-445x(01)00219-3)

744 Meddeb, M., Grami, B., Chaalali, A., Haraldsson, M., Niquil, N., Pringault, O., Sakka  
745 Hlaili, A., 2018. Plankton food-web functioning in anthropogenically impacted  
746 coastal waters (SW Mediterranean Sea): An ecological network analysis. *Prog.*  
747 *Oceanogr.* 162. <https://doi.org/10.1016/j.pocean.2018.02.013>

748 Meddeb, M., Niquil, N., Grami, B., Meiri, K., Haraldsson, M., Chaalali, A., Pringault,  
749 O., Hlaili, A.S., 2019. A new type of plankton food web functioning in coastal  
750 waters revealed by coupling Monte Carlo Markov chain linear inverse method  
751 and ecological network analysis. *Ecol. Indic.* 104.  
752 <https://doi.org/10.1016/j.ecolind.2019.04.077>

753 Mhadhbi, T., Pringault, O., Nouri, H., Spinelli, S., Beyrem, H., Gonzalez, C., 2019.  
754 Evaluating polar pesticide pollution with a combined approach: a survey of  
755 agricultural practices and POCIS passive samplers in a Tunisian lagoon  
756 watershed. *Environ. Sci. Pollut. Res.* 26. [https://doi.org/10.1007/s11356-018-3552-](https://doi.org/10.1007/s11356-018-3552-3)  
757 3

758 Miao, A.J., Wang, W.X., 2006. Cadmium toxicity to two marine phytoplankton under  
759 different nutrient conditions. *Aquat. Toxicol.* 78, 114–126.

- 760 Morán, X.A.G., Estrada, M., Gasol, J.M., Pedros-Alio, C., 2002. Dissolved primary  
761 production and the strength of phytoplankton bacterioplankton coupling in  
762 contrasting marine regions. *Microb. Ecol.* 44, 217–223.
- 763 Naveed, S., Li, C., Lu, X., Chen, S., Yin, B., Zhang, C., Ge, Y., 2019. Microalgal  
764 extracellular polymeric substances and their interactions with metal(loid)s: A  
765 review. *Crit. Rev. Environ. Sci. Technol.* 49, 1769–1802.  
766 <https://doi.org/10.1080/10643389.2019.1583052>
- 767 Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A.C., Colijn, F., Riva, S.D., Gertz,  
768 F., Hansen, J.W., Holmer, M., Ivanova, K., Leppäkoski, E., Canu, D.M., Mocenni,  
769 C., Mudge, S., Murray, N., Pejrup, M., Razinkovas, A., Reizopoulou, S., Pérez-  
770 Ruzafa, A., Schernewski, G., Schubert, H., Carr, L., Solidoro, C., PierluigiViaroli,  
771 Zaldívar, J.-M.M., Viaroli, P., Zaldívar, J.-M.M., 2014. An overview of ecological  
772 status, vulnerability and future perspectives of European large shallow, semi-  
773 enclosed coastal systems, lagoons and transitional waters. *Estuar. Coast. Shelf  
774 Sci.* 140, 95–122. <https://doi.org/10.1016/j.ecss.2013.05.023>
- 775 Niquil, N., Chaumillon, E., Johnson, G.A., Bertin, X., Grami, B., David, V., Bacher, C.,  
776 Asmus, H., Baird, D., Asmus, R., 2012. The effect of physical drivers on  
777 ecosystem indices derived from ecological network analysis: Comparison across  
778 estuarine ecosystems. *Estuar. Coast. Shelf Sci.* 108, 132–143.  
779 <https://doi.org/10.1016/j.ecss.2011.12.031>
- 780 Pérez-Ruzafa, A., Fernández, A.I., Marcos, C., Gilabert, J., Quispe, J.I., García-  
781 Charton, J.A., 2005. Spatial and temporal variations of hydrological conditions,  
782 nutrients and chlorophyll a in a Mediterranean coastal lagoon (Mar Menor,  
783 Spain), in: *Hydrobiologia*. pp. 11–27. <https://doi.org/10.1007/s10750-005-4356-2>
- 784 Pérez, P., Estévez-Blanco, P., Beiras, R., Fernández, E., 2006. EFFECT OF COPPER  
785 ON THE PHOTOCHEMICAL EFFICIENCY, GROWTH, AND CHLOROPHYLL  
786 A BIOMASS OF NATURAL PHYTOPLANKTON ASSEMBLAGES. *Environ.  
787 Toxicol. Chem.* 25, 137. <https://doi.org/10.1897/04-392R1.1>
- 788 Pesce, S., Batisson, I., Bardot, C., Fajon, C., Portelli, C., Montuelle, B., Bohatier, J.,

789 2009. Response of spring and summer riverine microbial communities following  
790 glyphosate exposure. *Ecotoxicol. Environ. Saf.* 72, 1905–1912.

791 Prieto, A., Barber-Lluch, E., Hernández-Ruiz, M., Martínez-García, S., Fernández, E.,  
792 Teira, E., 2015. Assessing the role of phytoplankton-bacterioplankton coupling in  
793 the response of microbial plankton to nutrient additions. *J. Plankton Res.* 38, 55–  
794 63. <https://doi.org/10.1093/plankt/fbv101>

795 Pringault, O., Lafabrie, C., Avezac, M., Bancon-Montigny, C., Carre, C., Chalghaf, M.,  
796 Delpoux, S., Duvivier, A., Elbaz-Poulichet, F., Gonzalez, C., Got, P., Leboulanger,  
797 C., Spinelli, S., Sakka Hlaili, A., Bouvy, M., 2016. Consequences of contaminant  
798 mixture on the dynamics and functional diversity of bacterioplankton in a  
799 southwestern Mediterranean coastal ecosystem. *Chemosphere* 144, 1060–1073.  
800 <https://doi.org/10.1016/j.chemosphere.2015.09.093>

801 Pringault, O., Tassas, V., Rochelle-Newall, E., 2007. Consequences of respiration in  
802 the light on the determination of production in pelagic systems. *Biogeosciences*  
803 4. <https://doi.org/10.5194/bg-4-105-2007>

804 Roberts, D.A., 2012. Causes and ecological effects of resuspended contaminated  
805 sediments (RCS) in marine environments. *Environ. Int.* 40, 230–243.

806 Rochelle-Newall, E., Mari, X., Rouchon, C., Torretton, J.P., Pringault, O., 2008. Zinc  
807 induces shifts in microbial carbon flux. *Aquat. Microb. Ecol.* 52, 57–68.

808 Rochelle-Newall, E.J., Chu, V.T., Pringault, O., Amouroux, D., Arfi, R., Bettarel, Y.,  
809 Bouvier, T., Bouvier, C., Got, P., Nguyen, T.M.H., Mari, X., Navarro, P., Duong,  
810 T.N., Cao, T.T.T., Pham, T.T., Ouillon, S., Torrétton, J.-P., 2011. Phytoplankton  
811 distribution and productivity in a highly turbid, tropical coastal system (Bach  
812 Dang Estuary, Vietnam). *Mar. Pollut. Bull.* 62.  
813 <https://doi.org/10.1016/j.marpolbul.2011.08.044>

814 Sakka Hlaili, A., Grami, B., Niquil, N., Gosselin, M., Hamel, D., Troussellier, M.,  
815 Mabrouk, H.H., 2008. The planktonic food web of the Bizerte lagoon (south-  
816 western Mediterranean) during summer: I. Spatial distribution under different  
817 anthropogenic pressures. *Estuar. Coast. Shelf Sci.* 78, 61–77.

818 <https://doi.org/10.1016/j.ecss.2007.11.010>

819 Sakka Hlaili, A., Niquil, N., Legendre, L., 2014. Planktonic food webs revisited:  
820 Reanalysis of results from the linear inverse approach. *Prog. Oceanogr.* 120, 216–  
821 229. <https://doi.org/http://dx.doi.org/10.1016/j.pocean.2013.09.003>

822 Salhi, N., Triki, H.Z., Molinero, J.C., Laabir, M., Sehli, E., Bellaaj-Zouari, A., Yahia,  
823 N.D., Yahia, O.K.-D., 2018. Seasonal variability of picophytoplankton under  
824 contrasting environments in northern Tunisian coasts, southwestern  
825 Mediterranean Sea. *Mar. Pollut. Bull.* 129, 866–874.  
826 <https://doi.org/10.1016/j.marpolbul.2017.10.029>

827 Sayol, J.M., Orfila, A., Simarro, G., López, C., Renault, L., Galán, A., Conti, D., 2013.  
828 Sea surface transport in the Western mediterranean sea: A lagrangian  
829 perspective. *J. Geophys. Res. Ocean.* 118, 6371–6384.  
830 <https://doi.org/10.1002/2013JC009243>

831 Schwarzenbach, R.P., Escher, B.I., Fenner, K., Hofstetter, T.B., Johnson, C.A., von  
832 Gunten, U., Wehrli, B., 2006. The challenge of micropollutants in aquatic  
833 systems. *Science (80-. )*. 313, 1072–1077.

834 Seto, M., Wada, S., Suzuki, S., 2013. The effect of zinc on aquatic microbial  
835 ecosystems and the degradation of dissolved organic matter. *Chemosphere* 90,  
836 1091–1102. <https://doi.org/10.1016/j.chemosphere.2012.09.014>

837 Seymour, J.R., Amin, S.A., Raina, J.-B., Stocker, R., 2017. Zooming in on the  
838 phycosphere: the ecological interface for phytoplankton-bacteria relationships.  
839 *Nat. Microbiol.* 2, 17065. <https://doi.org/10.1038/nmicrobiol.2017.65>

840 Sfriso, A., Facca, C., Ghetti, P.F., 2003. Temporal and spatial changes of macroalgae  
841 and phytoplankton in a Mediterranean coastal area: The Venice lagoon as a case  
842 study. *Mar. Environ. Res.* 56, 617–636. [https://doi.org/10.1016/S0141-](https://doi.org/10.1016/S0141-1136(03)00046-1)  
843 [1136\(03\)00046-1](https://doi.org/10.1016/S0141-1136(03)00046-1)

844 Sharma, S.S., Schat, H., Vooijs, R., Van Heerwaarden, L.M., 1999. Combination  
845 toxicology of copper, zinc, and cadmium in binary mixtures: Concentration-  
846 dependent antagonistic, nonadditive, and synergistic effects on root growth in

847 *Silene vulgaris*. Environ. Toxicol. Chem. 18, 348–355.

848 Small, C., Nicholls, R.J., 2003. A global analysis of human settlement in coastal zones.  
849 J. Coast. Res. 19, 584–599.

850 Smith, D.C., Azam, F., 1992. A simple, economical method for measuring bacterial  
851 protein synthesis rates in seawater using [3H]-leucine. Mar. Microb. Food Webs  
852 6, 107–114.

853 Uchimiya, M., Motegi, C., Nishino, S., Kawaguchi, Y., Inoue, J., Ogawa, H., Nagata,  
854 T., 2016. Coupled response of bacterial production to a wind-induced fall  
855 phytoplankton bloom and sediment resuspension in the chukchi sea shelf,  
856 Western Arctic Ocean. Front. Mar. Sci. 3.  
857 <https://doi.org/10.3389/fmars.2016.00231>

858 Utermöhl, H., 1958. Zur Vervollkommnung der quantitativen Phytoplankton-  
859 Method. Int. Ver. Theor. Angew. Limnol. 9, 1–39.

860 Viličić, D., 2014. Ecology and Composition of Phytoplankton in the Adriatic Sea.  
861 Koeltz Scientific Books.

862 Wada, S., Suzuki, S., 2011. Inhibitory effect of zinc on the remineralisation of  
863 dissolved organic matter in the coastal environment. Aquat. Microb. Ecol. 63,  
864 47–59. <https://doi.org/10.3354/ame01481>

865 Yoshida, M., Hamdi, H., Nasser, I.A., Jedidi, N., 2002. Contamination of Potentially  
866 Toxic Elements (PTEs) in Bizerte lagoon bottom sediments, surface sediments  
867 and sediement repository, in: Ghrabi, A., Yoshida, M. (Eds.), Study on  
868 Environmental Pollution of Bizerte Lagoon. INRST-JICA, p. 139.

869 Zrafi-Nouira, I., Khedir-Ghenim, Z., Bahri, R., Cheraeif, I., Rouabhia, M., Saidane-  
870 Mosbahi, D., 2009. Hydrocarbons in Seawater and Water Extract of Jarzouna-  
871 Bizerte Coastal of Tunisia (Mediterranean Sea): Petroleum Origin Investigation  
872 Around Refinery Rejection Place. Water Air Soil Pollut. 202, 19–31.

873 Zrafi-Nouira, I., Khedir-Ghenim, Z., Zrafi, F., Bahri, R., Cheraeif, I., Rouabhia, M.,  
874 Saidane-Mosbahi, D., 2008. Hydrocarbon pollution in the sediment from the  
875 Jarzouna-Bizerte coastal area of Tunisia (Mediterranean Sea). Bull. Environ.

876 Contam. Toxicol. 80, 566–572.

877

## Figure legends

Figure 1: Main phytoplankton groups observed in Offshore and Lagoon waters as a function of the season.

Figure 2: Chlorophyll a concentration during microcosm incubation of Offshore and Lagoon waters as a function of treatments for the four sampling seasons. Average and standard deviation (n=3).

Figure 3: Bacterial abundance during microcosm incubation of Offshore and Lagoon waters as a function of treatments for the four sampling seasons. Average and standard deviation (n=3).

Figure 4: Temporal variations of primary production and bacterial carbon demand observed during the microcosm incubation of Offshore waters as a function of treatments for the four sampling seasons. Average and standard deviation (n=3).

Figure 5: Temporal variations of primary production and bacterial carbon demand observed during the microcosm incubation of Lagoon waters as a function of treatments for the four sampling seasons. Average and standard deviation (n=3).

Figure 6: Scatter plot of bacterial carbon demand vs primary production in Offshore waters for the four seasons as a function of treatments. The fitted solid lines represent the linear regression. The dotted line indicates the 1:1 relationship between the two metabolic processes.

Figure 7: Scatter plot of bacterial carbon demand vs primary production in Lagoon waters for the four seasons as a function of treatments. The fitted solid lines

represent the linear regression. The dotted line indicates the 1:1 relationship between the two metabolic processes

Figure1  
[Click here to download high resolution image](#)

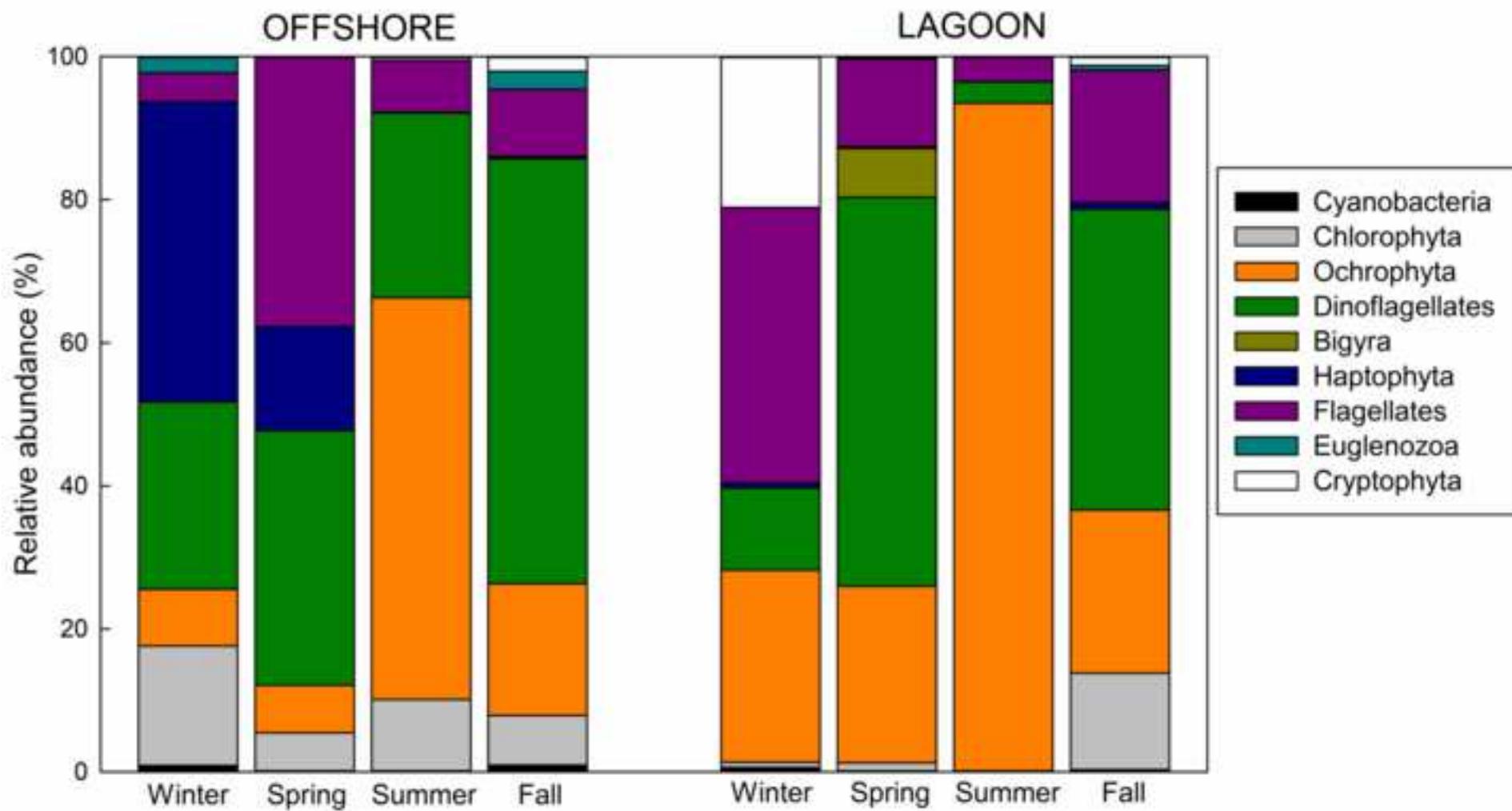


Figure2

[Click here to download high resolution image](#)

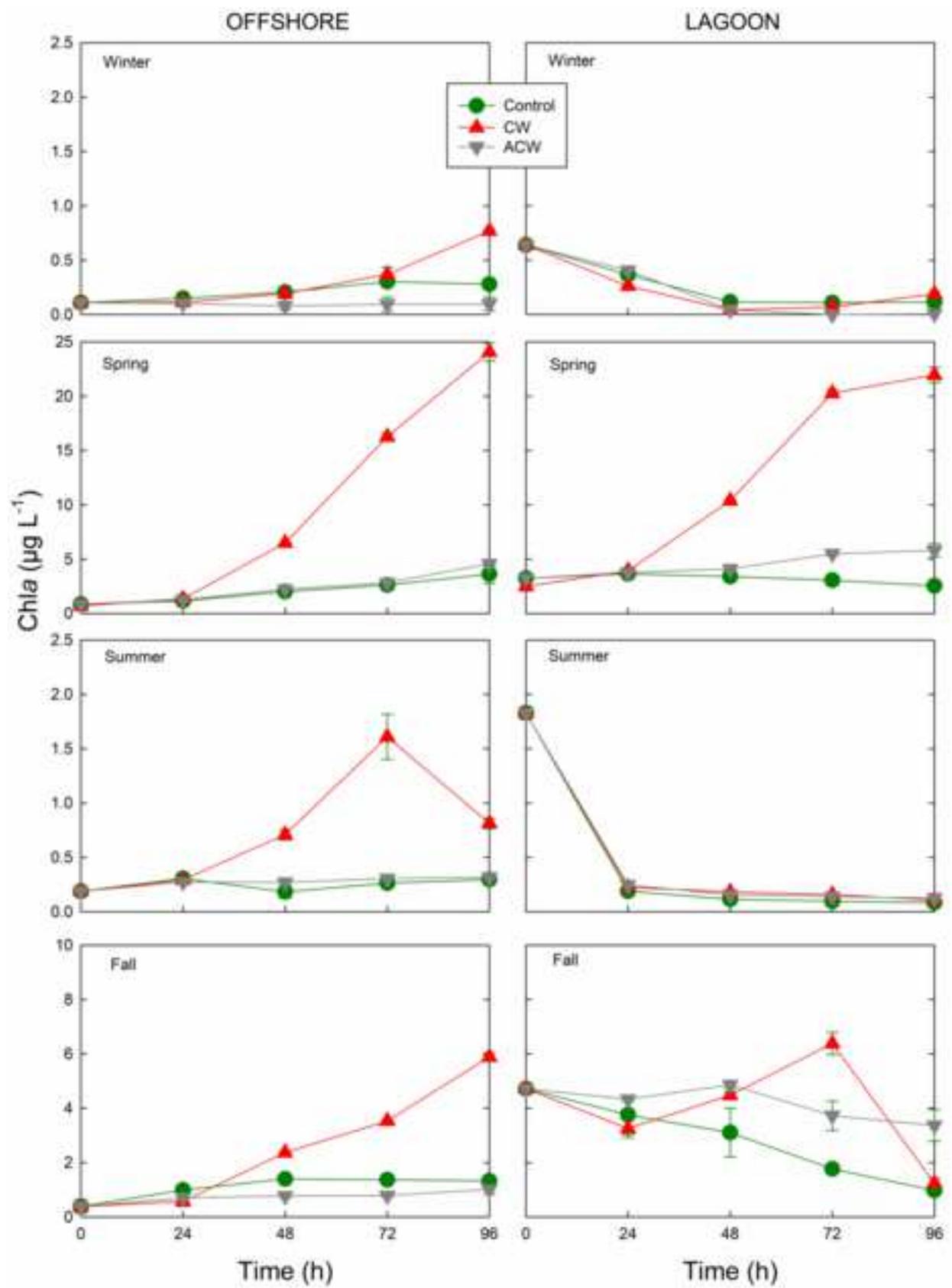


Figure3

[Click here to download high resolution image](#)

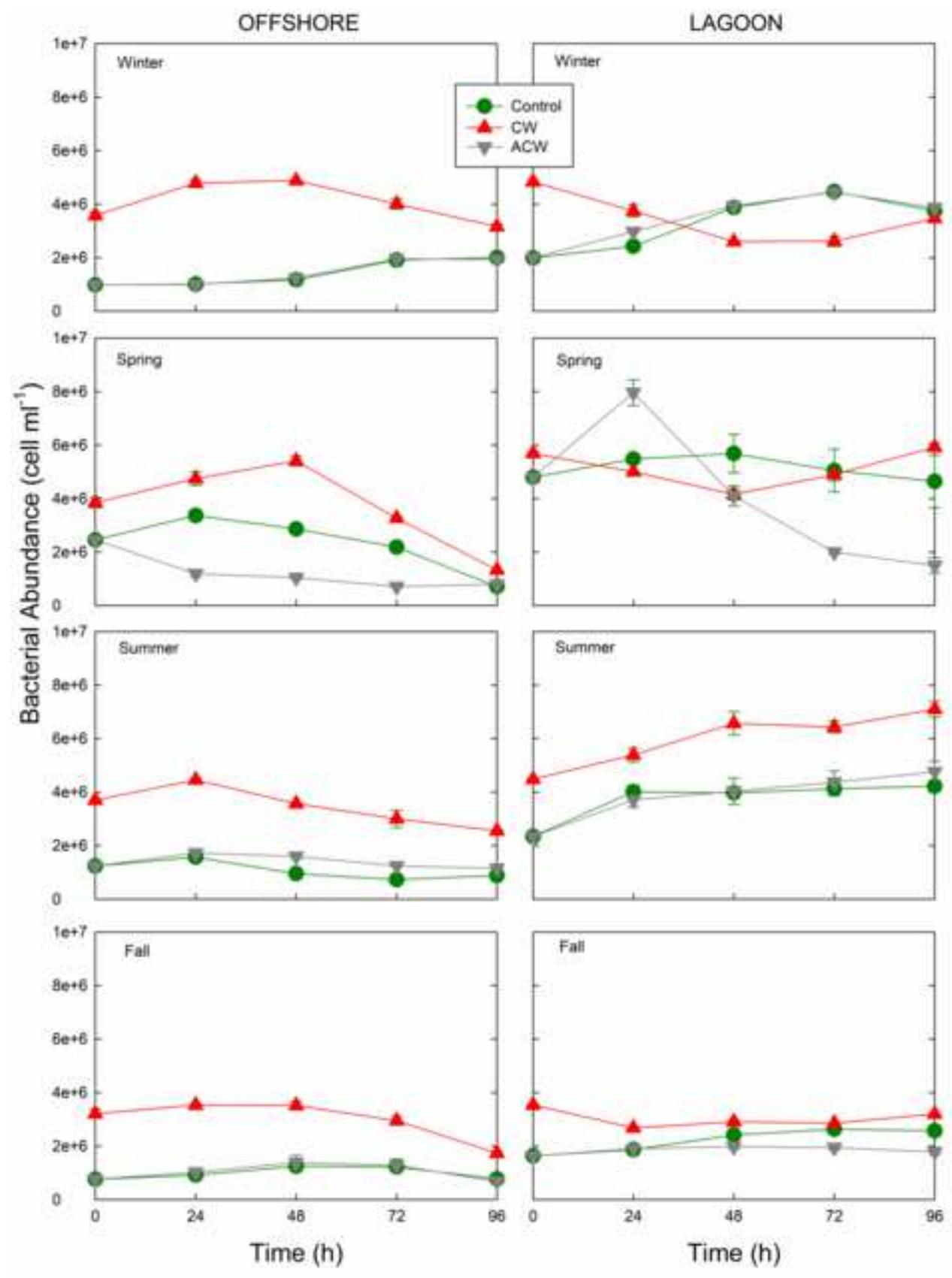


Figure4

[Click here to download high resolution image](#)

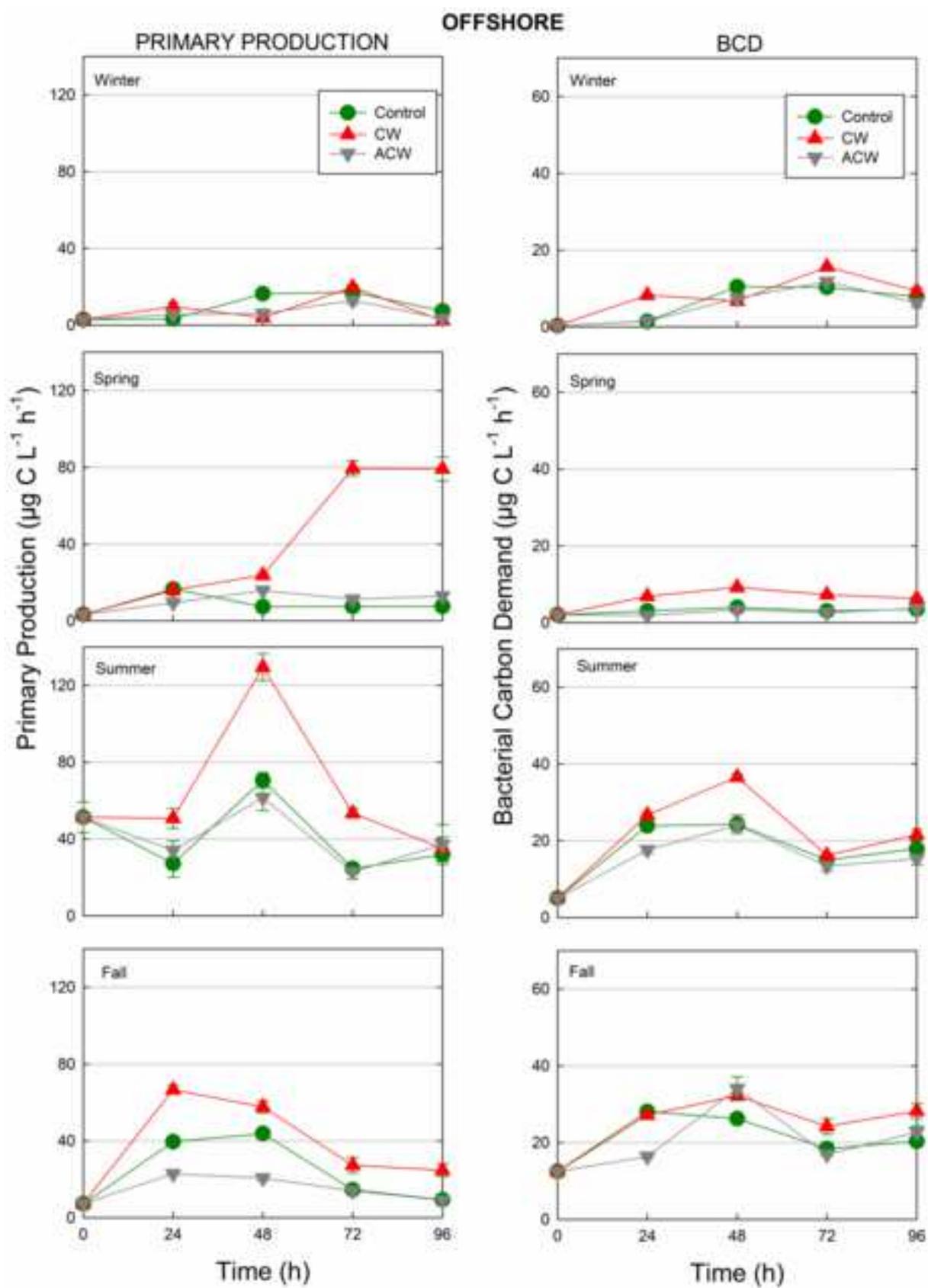


Figure5

[Click here to download high resolution image](#)

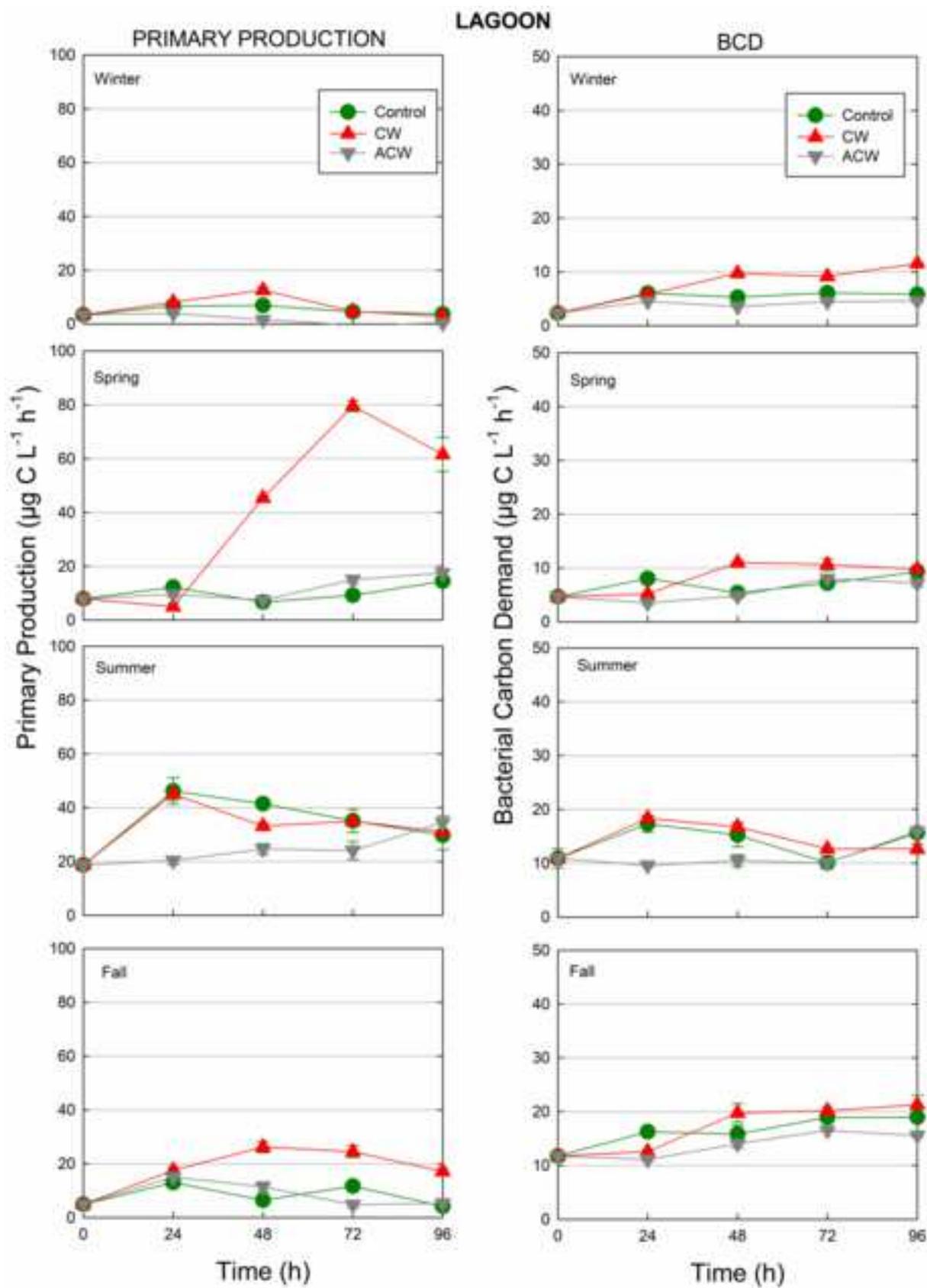


Figure6

[Click here to download high resolution image](#)

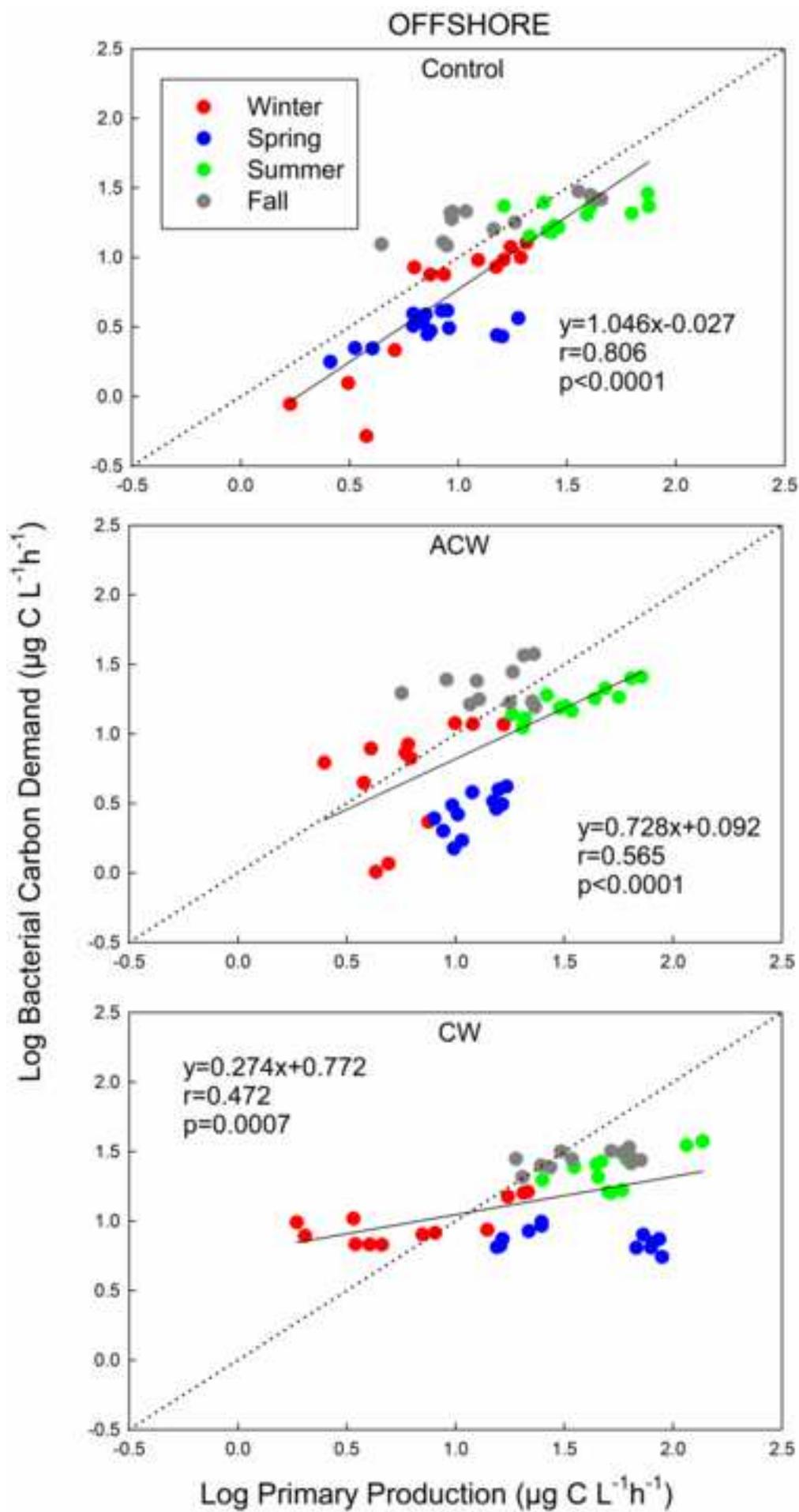
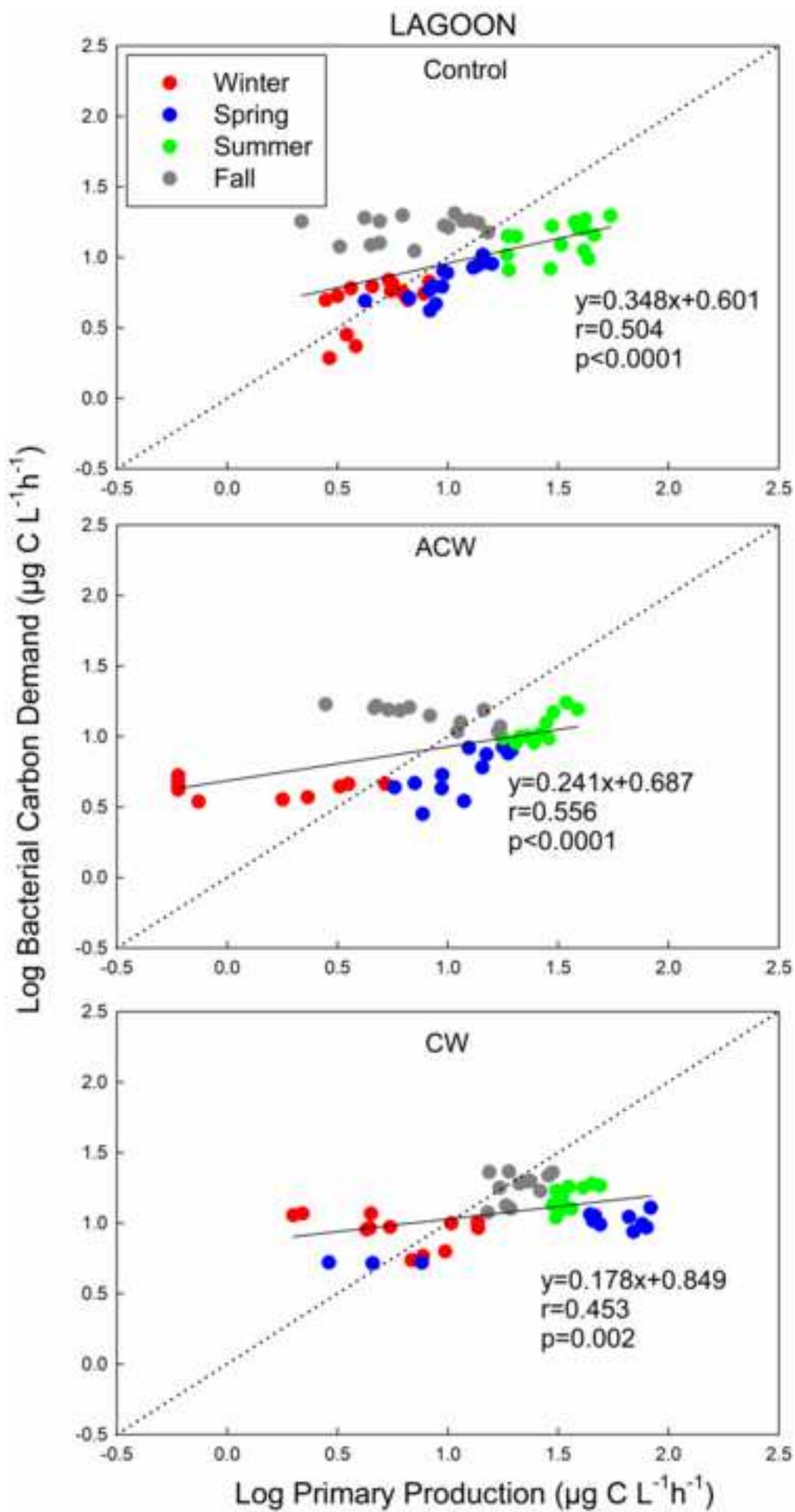


Figure7

[Click here to download high resolution image](#)



**Supplementary Material**

[Click here to download Supplementary Material: Pringault et al Supplementary materials revised version.docx](#)