Journal of Experimental Marine Biology and Ecology
January 2017, Volume 486 Pages 1-12
http://dx.doi.org/10.1016/j.jembe.2016.09.006
http://archimer.ifremer.fr/doc/00352/46364/
© 2016 Elsevier B.V. All rights reserved.

Achimer
http://archimer.ifremer.fr

Impact of contaminated sediment elutriate on coastal phytoplankton community (Thau lagoon, Mediterranean Sea, France)

Ben Othman Hiba [1, 2], Pringault Olivier [3], Louati Héla [1], Sakka Hlaili Asma [1], Leboulanger Christophe [2, *]

[1] Université de Carthage, Zarzouna, 7021 Bizerte, Tunisia
[2] IRD-CNRS-Université Montpellier-Ifremer, Station Marine, 2 rue des Chantiers, 34200 Sète, France
[3] IRD, BP 434, 2 rue des Sports, El Menzah I, 1004 Tunis, Tunisia

* Corresponding author : Christophe Leboulanger, email address : christophe.leboulanger@ird.fr

Abstract :

Effects of sediment-released contaminants and nitrogen were assessed on phytoplankton communities sampled from Thau lagoon (France, Mediterranean Sea) and one close offshore marine station. Phytoplankton was exposed to sediment elutriate (seawater containing a mix of metals, organic chemicals, and nutrients) or to ammonium enrichment for four days using immersed microcosms exposed to natural conditions of light and temperature. Functional (production - respiration balance) and structural (taxonomy and cell densities) responses of the phytoplankton community were assessed. In the lagoon, both treatments stimulated phytoplankton growth, compare to controls. Conversely in the offshore station, the phytoplankton growth was stimulated only with the sediment elutriate addition. In offshore and lagoon stations, both treatments caused a shift in the taxonomic composition of the phytoplankton. Proliferation of potentially toxic diatoms and dinoflagellates resulted from the addition of elutriate. Correspondence analysis determined that phytoplankton from the offshore station was more sensitive to both treatments compared to the lagoon community. According to daily production and respiration balance, lagoon community metabolism remained heterotrophic (P < R) for all treatments, whereas only transient shifts to net autotrophy (P > R) were observed in the offshore community. Direct toxicity of contaminants released from sediment, if any, was therefore masked by nutrient enrichment effects, whereas indirect evidence of contaminant pressure was highlighted by changes in community composition and metabolism.

Please note that this is an author-produced PDF of an article accepted for publication following peer review. The definitive publisher-authenticated version is available on the publisher Web site.

Highlights

► Contaminated sediment elutriates trigger changes in phytoplankton dynamics. ► Phytoplankton community structure and metabolism are affected. ► Elutriates effects are not identical to those of nutrient addition alone. ► Elutriates effects differ depending on the initial community structure.

Keywords : Sediment resuspension, Mediterranean lagoon, Contaminants, Nutrients, Phytoplankton community

Please note that this is an author-produced PDF of an article accepted for publication following peer review. The definitive publisher-authenticated version is available on the publisher Web site.

1. Introduction

Anthropogenic contaminants in marine ecosystems are often eventually trapped by sediments, which act as a sink for pollutants. Resulting contaminated sediments affect mainly the benthic organisms but can also impede pelagic compartments when chemicals are released to the water column (Birch and O'Hea 2007; Jonas and Millward 2010; Roberts, 2012). Disturbance of contaminated sediments by anthropogenic (dredging and disposal of dredged materials, trawling, ship movements and propeller wash) and natural processes (tides, storms and bioturbation) may affect a wide range of species, eventually being incorporated into food webs (Burton and Johnston 2010). Resuspension of contaminated sediments (RCS) has been shown to be an important source of dissolved chemicals in historically contaminated estuaries (Kalnejais *et al*. 2010, Roberts 2012). Dissolved polycyclic aromatic hydrocarbon (PAH) and polychlorinated biphenyl (PCB) concentrations have been shown to increase due to dredged material disposal, even though the levels did not exceed water quality criteria (Cornelissen *et al.,* 2008). For example, in conditions of turbulent mixing in oxygenated seawater, organic (PAH and PCB) and inorganic (trace metals) contaminants were desorbed from sediments (Josefsson *et al.,* 2010; Voie *et al.,* 2002). In anthropized coastal areas, such mixing events also cause nutrients release, with consequent enrichment of the water column (Cantwell *et al.,* 2002; Kalnejais *et al.,* 2010), and ammonia may be released in greater amounts (Jones-Lee and Lee, 2005). A few studies have reported that nutrients and contaminants may have opposite results, with the beneficial effect of the nutrients overcompensating for the detrimental impact of the contaminants (Crain *et al.,* 2008). The effects of RCS on pelagic organisms are still being studied and have not yet been fully incorporated into ecological risk assessment for management of coastal ecosystems (Roberts, 2012). Several studies have shown that marine organisms (such as fish, bivalves, algae and polychaetes) are sensitive to RCS by many pathways (Edge *et al*., 2015; Edge *et al*., 2016, Hill *et al.,* 2009; Tolhurst *et al.,* 2007; Voie *et al.,* 2002).

Phytoplankton feeds pelagic food webs and determines aquatic ecosystem primary production and functioning at higher trophic levels (Field *et al*., 1998). Any change in its community structure and

metabolism may, therefore, trigger a cascade of indirect effects throughout the plankton ecosystem (De Hoop *et al*., 2013). Understanding how phytoplankton responds to contaminants is of primary importance for forecasting the ecological consequences of chemical contamination and for targeting priority management and restoration policies for aquatic systems. There is however no general consensus about the responses of primary producers, in direction and extent, to the simultaneous load of pollutants and nutrients. For example, Riedel *et al.* (2003) reported that phytoplankton growth was inhibited after exposure to a combination of contaminants (mix of trace metals) and nutrients, whereas Lafabrie *et al*. (2013a, 2013b) showed that RCS can stimulate growth and modify phytoplankton community structure, suggesting that the toxic effect of chemicals if any could be offset by nutrients.

Within the marine environment, coastal ecosystems such as lagoons have become particularly impacted by chemical pollutants from anthropogenic inputs (agricultural, urban, and industrial) (Lafabrie *et al*., 2013a; Levin *et al*., 2001). This increased chemical contamination can cause a drastic alteration of the lagoon environment and eventually a threat to the services associated with lagoon environments (fisheries, aquaculture, tourism).This is the case for example of Thau lagoon (France), located on the northern shores of the Western Mediterranean Sea. The Thau Lagoon is one of the largest French Mediterranean Lagoons and is one the most important shellfish farming areas in Europe (Castro-Jimenez *et al*., 2008). It sustains a large population of exploited fish (gilthead seabream *Sparus aurata* and European seabass *Dicentrarchus labrax*) and reared shellfish (mussels, clams and oysters) (Fouilland *et al.,* 2012). This lagoon appears to be under intense anthropogenic pressure characterized by an increase in industrial, agricultural and urban activities (La Jeunesse, 2001). Recent studies in the Thau lagoon have revealed moderate to high chemical contamination in the sediment (Rigollet *et al.,* 2004), and high nutrient enrichment of the water (Anschutz *et al.,* 2007; Fouilland *et al.,* 2012) whereas the combined effects of pollutants and nutrients on the planktonic organisms of the lagoon have retained lesser attention. The Thau lagoon like all shallow marine ecosystems is subject to sediment resuspension that may have important consequences for the

ecosystem functioning, and impact aquaculture activities (changes in the structure and contribution of potentially toxic phytoplankton for example). Among factors that can trigger resuspension (reviewed in Roberts, 2012) in the area, flash floods and windstorms are documented (Fouilland *et al*. 2012), together with dredging activities in the nearby waterways and harbor. This study aimed to assess the effects of elutriates prepared from RCS on the structure and the functioning of the Thau Lagoon phytoplankton community incubated under natural outdoor conditions of temperature and irradiance, compared to near coastal community, as well as on the dynamics and metabolism of autotrophic and heterotrophic compartments. The present work focused on the soluble fraction resulting from sediment resuspension considered as directly bioavailable for plankton communities, taking in mind that suspended sediment itself is prone to have different effects even trough physical damage (Edge *et al*., 2015; Edge *et al*., 2016). The two stations were chosen to compare the responses of a pre-exposed community from a closed lagoon, subjected to local inputs of contaminants to an a priori unexposed community from offshore station, considered as a reference.

2. Materials and Methods

2.1. Study area

The Thau lagoon, one of the largest French Mediterranean Lagoons (43°24'N-3°36'E; Fig. 1) with an area of 70 km^2, is an important shellfishing area. Climate is characterized by autumnal storm events, and a single flood can contribute a quarter of the annual nitrogen supply from watershed (Tournoud *et al.,* 2006). Consequently, dissolved ammonium, especially in the oyster farming areas, can reach a significant concentration (Fouilland *et al.,* 2012; Gilbert *et al.,* 1997). Thau lagoon has also been strongly affected by industrial activities and an intense urbanization since the mid 20^{th} century, which have resulted in a significant concentration of contaminants in sediments including polycyclic aromatic hydrocarbons PAH (440 - 7700µg Kg^{-1}), polychlorobyphenyls PCB (600 - 30,000 ng Kg^{-1}), trace metals (8.9-6098 ng g^{-1}) and pesticides (2921 ng Kg^{-1}) as reported in several studies (Kawakami *et al*., 2008; Léauté, 2008; Rigollet *et al*., 2004).

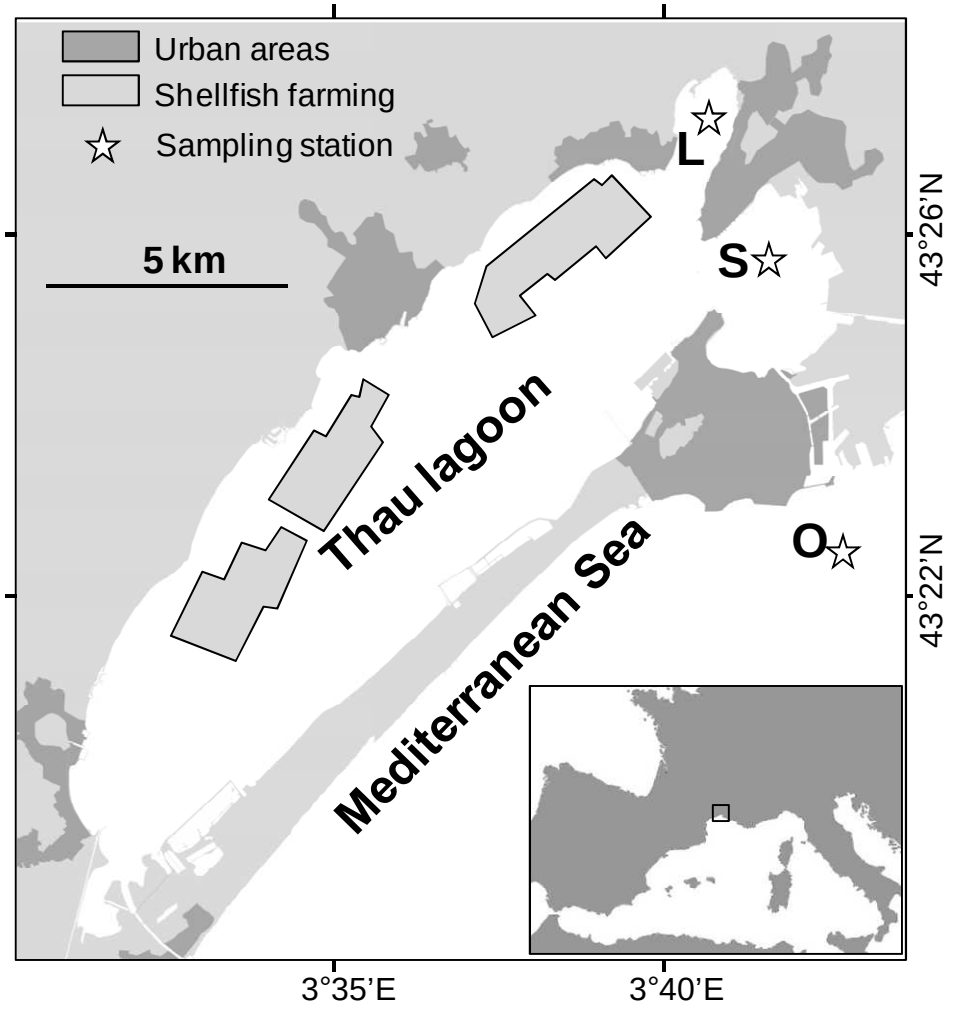


Figure 1. Situation of the Thau Lagoon in Southern France, and location of sampling stations: sediment and water for elutriate preparation(S), lagoon (L) and offshore (O) water inoculum.

2.2. Sediment elutriate preparation and chemical analyses

Water and sediment were sampled in June 2011 in the "*Eaux Blanches"* bay (Station S, Fig. 1). The sediments in this area have previously been characterized as highly contaminated by trace metals, PAH and PCB (Kawakami *et al*., 2008; Léauté, 2008). The sediment was sampled using a Van Veen grab and screened directly through a 2 mm mesh stainless steel sieve to remove stones, macrofauna and plant fragments. Water for elutriate preparation was collected from the surface in a bucket and immediately filtered through a 80 µm mesh to remove the largest organisms and debris. The sediment elutriate was prepared as described by Bonnet *et al.* (2000). Sediment was stirred in water (ratio 1:4 v:v) for 8 h to mimic resuspension. After settling for 8 h, the supernatant was collected and

133 then filtered through 1 and 0.2 µm membranes to remove any native microorganisms and leave only
134 dissolved chemical contaminants. The elutriate was used as contaminated water (CW) for the
135 microcosm experiment.
136 Ammonium concentration in the elutriate was measured spectrophotometrically using indophenol
137 blue complexation in basic medium (Koroleff, 1969), with a detection limit of 0.04 µM.
138 Sub-samples of sediment and elutriates were preserved for further determination of 16 PAHs
139 (naphthalene, acenaphthylene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene,
140 benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene,
141 dibenzo(a,h)anthracene, benzo(ghi)perylene, indeno(1,2,3-cd)pyrene and acenaphthylene) and 7 PCB
142 congeners (180, 28, 52, 101, 118, 138, and 153) by GC-MS method after liquid-liquid extraction for
143 the CW and soxhlet extraction for the sediment. Trace metals (As, Cd, Cr, Cu, Hg, Ni, Pb, Zn, U, V, Mn,
144 Fe, Co and Mo) concentrations in elutriate were measured by ICP-MS, and the NH_4^+ concentration in
145 the CW was determined using fluorometry (Holmes *et al.,* 1999). Analytical methods involved the use
146 of spiked deuterated standards and Canadian National Research Council standard seawater, for PAH
147 and metal analysis respectively as described in Pringault *et al*. (2016).
148 Contaminant concentrations were compared when available to published quality guidelines and
149 effects evaluation such as Effect Range Low and Effect Range Median (ERL and ERM, Long *et al*,.
150 1995), and harmful concentrations for 5% of tested species (HC5, Ben Othman *et al*., 2012, Wang *et*
151 *al*., 2014, Wheeler *et al*., 2002) for sediment and water respectively.

152 **2.3. Outdoor microcosm experiment**

153 Seawater was sampled at station L (“*Lagoon”)*, a highly-contaminated lagoon area characterized by
154 regular occurrences of toxic dinoflagellate blooms (Laabir *et al.,* 2011) and station O ("*Offshore*")
155 chosen as an offshore reference station (Fig. 1). Water samples (50 L), collected from 0 – 2 m depth,
156 was immediately filtered through an 80 µm mesh to remove large grazers and particles, and then
157 stored in the dark in coolers at ambient temperature before processing. Eighty µm-filtered water

158 (FW) was used as plankton inoculum in microcosms, submitted to three different treatments:
159 sediment elutriate addition (E: 3L of FW + 1L of CW), N-nutrient addition (N: 4L of FW + 20 µM of
160 NH_4^+, final concentration, close to the actually measured ammonium concentration in elutriate) and
161 control with no addition (C: 4L of FW). All treatments were performed in triplicate using 5L glass
162 bottles (Schott-Duran) which contained 1L of air to allow gas exchange (de la Broise and Palenik,
163 2007). The E treatment (for elutriate) allowed the effect of both released pollutants and nutrients on
164 the phytoplankton community to be determined, whereas N treatment investigated the impact of
165 NH_4^+ enrichment alone. Ammonium was chosen because it is among the most significant N nutrients
166 in the Thau lagoon prone to have direct effects on planktonic communities (Fouilland *et al.* 2012;
167 Trottet et al. 2016).
168 All microcosms were incubated for 96 h in an outdoor tank exposed to natural sunlight and filled with
169 seawater, which was renewed directly from the lagoon by a pumping-draining system allowing
170 microcosms to support the same temperature as nearby seawater. This simulated the natural
171 conditions of light and temperature of the Thau lagoon surface waters. Each microcosm was sampled
172 at the beginning and at 24, 48, 72 and 96 h of incubation to measure chl *a* and plankton production
173 and respiration. Sub-samples were also taken to determine the initial (at 0 h) and final (at 96 h) cell
174 abundances and taxonomic composition of the phytoplankton communities.

175 **2.4. Chlorophyll *a* analysis**

176 Sub-samples (200 ml) for chl *a* were filtered onto glass fiber filters (GF/F, Whatman), and
177 immediately frozen at -80°C until analysis. Chlorophyll *a* was extracted using 90% acetone for 24 h in
178 the dark at 4°C, followed by ultrasonic grinding in an ice bath. After centrifugation of the extract
179 (3500 rpm for 15 min), pigment concentration in the supernatant was measured by
180 spectrofluorimetry (Neveux and Lantoine, 1993).

181 **2.5. Phytoplankton identification and count**

182 To identify and count the phytoplankton, 250 ml sub-samples were fixed with formaldehyde (4% final
183 concentration). The phytoplankton cells were counted on a settled volume of 100 ml (Utermöhl,

1938), using an upright microscope (Zeiss AX10) with a Sony XCD- U100CR digital camera, or an inverted microscope (Olympus IX70) with a Moticam MoticamPro digital camera. At least 500 cells in each sample were counted to achieve reliable estimates.

2.6. Production and respiration measurement

The net primary production (NPP) and dark respiration (R_{dark}) rates were measured using short time incubations (< 6 h) in gas-tight 30 mL vessels, according to the oxygen light-dark incubation method (Pringault *et al.,* 2007) with a Clark-type oxygen microsensor (Unisense, Denmark). Samples were kept at ambient field temperature and light irradiance during the incubation period. Oxygen concentrations were measured in each flask at one hour intervals, corrected for temperature, salinity and barometric pressure, and oxygen evolution rates calculated from linearized data providing hourly net primary production rates (NPP, O_2 evolution in the light) and dark respiration rates (R_{dark}, O_2 evolution in the dark). The gross primary production (GPP) was then calculated from the NPP and R_{light} values (GPP = $|R_{light}|$ + NPP) considering that R_{light} represented three times R_{dark} following the procedure of Pringault *et al.* (2007). The primary production to community respiration ratios (P:R) were considered from the GPP and R daily rates that were calculated from the GPP R_{dark} and R_{light} hourly rates, respectively. A light period of 12 h was considered for integration of daily GPP rates, and daily respiration was calculated as the sum of hourly rates of R_{light} (*12 hours) and hourly rates of R_{dark} (*12 hours).

2.7. Statistical analyses

Differences in variables between treatments (fixed factor, 3 levels) and sampling times (fixed factor, 5 levels) were tested by two-way factorial analysis of variance (ANOVA). When ANOVA revealed significant differences, Tukey HSD post-hoc comparison test was performed to identify the significant differences. The conditions of distribution normality (Kolmogorov–Smirnov test) and variance homogeneity (Bartlett-Box test) were respected. These analyses were done using SPSS 14.0 statistical software for Windows. The Shannon-Wiener index *H* was calculated using PAST freeware (Hammer and Harper, 2001) to analyze changes in phytoplankton diversity for each treatment.

Correspondence analysis was performed using PAST from the absolute species abundances to detect any phytoplankton composition change and highlight taxa most responsible for community structuring.

3. Results

3.1. Contaminant and nutrient levels

Low levels of PCBs ranging from 0.002 to 0.017 µg g^{-1} were measured in the sediment, whereas greater amounts of PAHs were found (Table 1). The most abundant PAHs in sediment were fluoranthene (1.7 µg g^{-1}), pyrene (1.24 µg g^{-1}), benzo(*a*)anthracene (0.78 µg g^{-1}) and benzo(*a*)pyrene (0.87 µg g^{-1}).

The contaminated water (CW) prepared from these sediment contained high metal levels at concentrations ranging from 0.03 (for Co) to 217.1 µg l^{-1} (for Mo) (Table 1). Apart from these potentially toxic contaminants, a significant concentration of NH_4^+ (21.8 µM) was also measured in CW. Elutriate did not contained any detectable dissolved PAH or PCB.

	Sediment	CW		Sediment	CW		Sediment	CW
PAH			PCB			TME		
Anthracene	0.22	bdl	PCB 101	0.002	bdl	Mo	-	217.1
Benzo(a)anthracene	0.78	bdl	PCB 118	0.012	bdl	Cd	-	0.021
Benzo(a)pyrene	0.87	bdl	PCB 138	0.005	bdl	Pb	-	0.088
Benzo(b)fluoranthene	0.67	bdl	PCB 135	0.007	bdl	U	-	33.3
Benzo(g,h,i)perylene	0.68	bdl	PCB 180	0.017	bdl	V	-	3.91
Benzo(k)fluoranthene	0.5	bdl	PCB 28	0.01	bdl	Cr	-	0.47
Chrysene	0.72	bdl	PCB 52	0.006	bdl	Mn	-	22.67
Fluoranthene	1.7	bdl				Fe	-	1.53
Indenopyrene	0.55	bdl				Co	-	0.03
Naphthalene	0.03	bdl				Ni	-	0.62
Phenanthrene	0.74	bdl				Cu	-	2.49
Pyrene	1.24	bdl				Zn	-	1.75
						As	-	9.56

Table 1. Contaminant levels in the Thau lagoon sediment (µg g^{-1} dry weight) and in the prepared elutriate (CW, µg l^{-1}) (bdl: below detection limit of 0.01 µg l^{-1}; -: not determined). PAH: polycyclic aromatic hydrocarbons; PCB: polychlorobyphenyl congeners; TME: trace metal elements.

3.2. Effect of the treatments on chlorophyll *a* concentrations

In lagoon station L, the concentration of chl *a* was 4.1 µg l^{-1} in the inoculum, and after 24 h, it increased 2 and 3 times compared to the control (5.6 µg l^{-1}) in microcosms E (added CW) and N

(added NH_4^+), respectively (Fig. 2A). Levels remained significantly higher ($p < 0.05$) in E (average of 11.24 µg l^{-1}) and N (average of 12.25 µg l^{-1}) than in the control (average 3.50 µg l^{-1}) until the end of incubation (Fig. 2A) and no significant difference was observed between E and N treatment. By contrast, the concentration of chl *a* was 0.225 µg l^{-1} in the inoculum from the offshore station O. After 48h, it increased 6 and 13 times compared to the control in microcosms E (48h: 6.123 µg l^{-1} in E and 0.63 µg l^{-1} in C; 72h: 13.21 µg l^{-1} in E and 0.7 µg l^{-1} in C) but at the end of the incubation the biomass decreased in microcosms E ($p< 0.05$) (3.21 µg l^{-1}). In general, no significant difference was observed between C and N treatments ($p< 0.05$) for the offshore inoculum all along the experiment, levels remained closely equivalent in C (0.50 µg l^{-1}) and N microcosms (0.70 µg l^{-1}) (Fig. 2B).

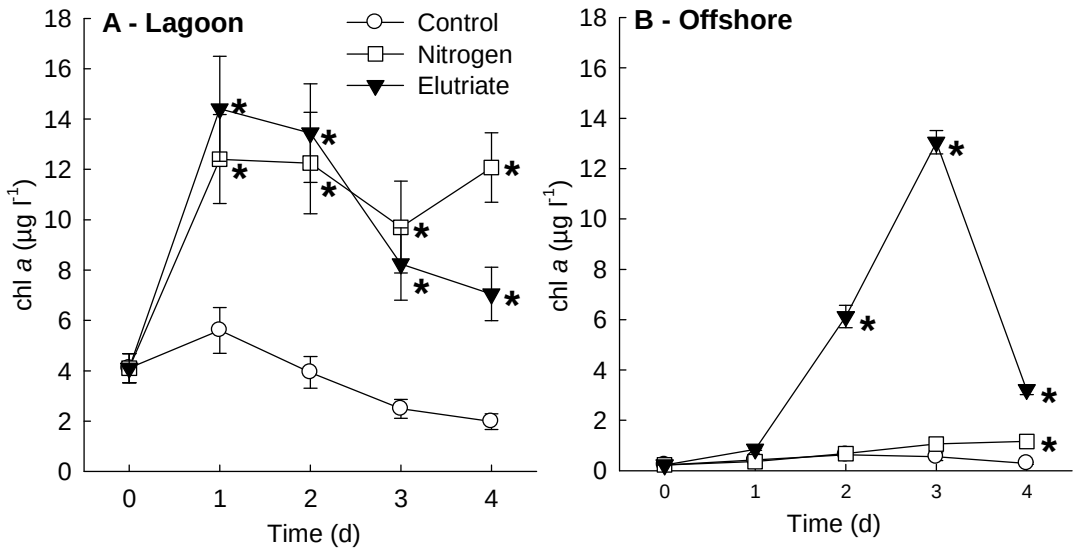


Figure 2. Evolution of chlorophyll *a* concentrations in the microcosms for Lagoon (A) and Offshore experiments (B). Mean ± SE, n = 3.

3.3. Effects of the treatments on phytoplankton abundance and whole community composition

In inoculums, the species diversity of the phytoplankton communities expressed as *H* index was significantly lower in lagoon (1.25 ± 0.05 bits ind^{-1}) than in offshore station (1.45 ± 0.006 bits ind^{-1}).

3.3.1. Lagoon phytoplankton community (station L)

In station L, the phytoplankton inoculum (0.85 10^6 cells l^{-1}) was dominated by chrysophyceae (0.38 10^6 cells l^{-1}), the most abundant species being the loricate chromulinale *Ollicola vangoorii* (known as mixotrophic in Mediterranean waters, Novarino *et al.* 2002) and an unidentified chrysophyceae (50% of total cell counts). The phytoplankton community contained similar proportions (~25%) of dinoflagellates (0.21 10^6 cells l^{-1}) and small unidentified flagellates (0.23 10^6 cells l^{-1}) (Fig. 3A).

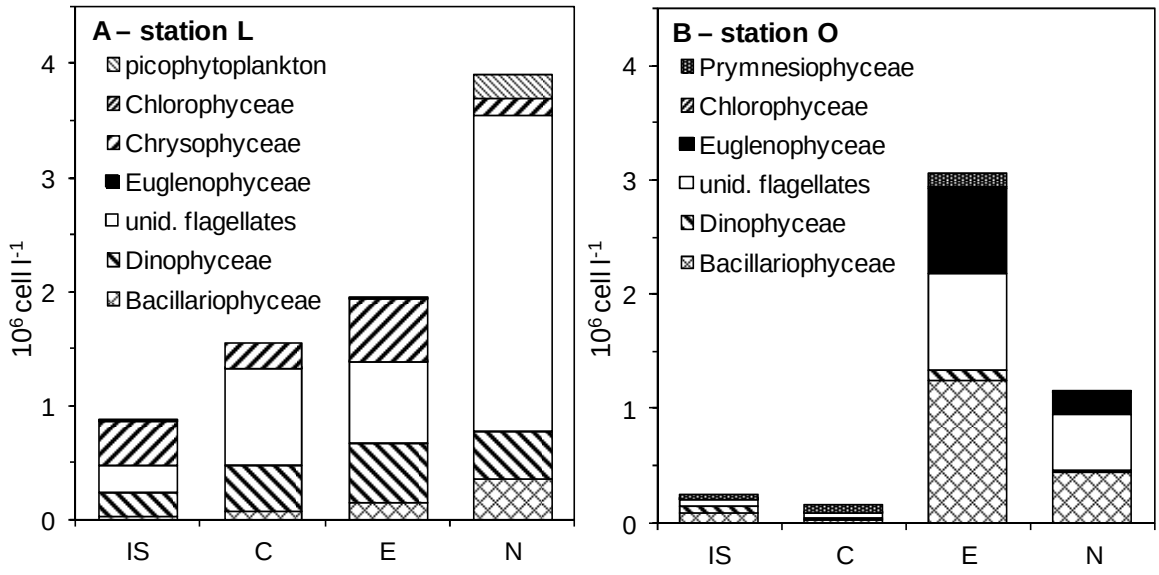


Figure 3. Contribution of the various taxonomic groups to the total phytoplankton abundances in the initial sample community (IS) and after four days in the control (C), sediment elutriate treatment (E), and nitrogen enrichment treatment (N) for Lagoon (Fig. 3A) and Offshore (Fig. 3B) experiments.

In station L, after 96 h of incubation, the phytoplankton density increased in all microcosms, reaching 1.54, 1.93 and 3.90 10^6 cells l^{-1} in the C, E and N microcosms, respectively (Fig. 3A). Diatoms (0.06 10^6 cells l^{-1} in C, 0.13 10^6 cells l^{-1} in S), chrysophycea (*Ollicola vangoorii* and the unidentified chrysophyceae) (0.22 10^6 cells l^{-1}, in C and 0.54 10^6 cells l^{-1} in E), and dinoflagellates (0.40 10^6 in C and 0.51 10^6 cells l^{-1} in E) were stimulated by the addition of elutriate. In the N microcosms, the flagellates density increased significantly (74.9%) followed by the diatoms (9%) and cyanobacteria (5%), the community being dominated by the unidentified flagellates (71%). By comparison, the community in the E microcosms comprised mainly dinoflagellates (11%), chrysophyceae (3.9%) and diatoms (9%). The contribution of diatoms in the N microcosms (9%) seemed to be higher than in the

E microcosms (7.1%). As well as changes in the contributions from the various taxonomic groups to the phytoplankton community, there were changes in the species composition inside taxonomic groups. Phytoplankton diversity increased after 96 h of incubation in all three microcosms compared to the initial samples. The Shannon index *H* was 1.7 ± 0.03; 1.55 ± 0.02 and 1.6 ± 0.01 bits ind^{-1}, respectively in C, E and N microcosms and there was no significant difference between treatments ($p > 0.05$).

3.3.2. Offshore phytoplankton community (station O)

In station O, the initial community of phytoplankton (0.24 10^6 cells l^{-1}) was dominated by diatoms (0.085 10^6 cells l^{-1}) and dinophyceae (0.053 10^6 cells l^{-1}). After 96 h of incubation, the algal density increased in N and E microcosms, respectively 1.25 and 3.04 10^6 cells l^{-1}, compared to the control (0.15 10^6 cells l^{-1}). Diatoms (1.24 10^6 cells l^{-1} in E, 0.014 10^6 cells l^{-1} in C), unidentified flagellates (0.03 10^6 cells l^{-1} in C and 0.85 10^6 cells l^{-1} in E) and chrysophyceae (0 in C and 0.73 10^6 cells l^{-1} in E) were stimulated by the addition of elutriate (Fig. 3B). In the N and E microcosms, the diatoms (40.8 % in E and 37.5 % in N) and chrysophyceae (24.3 % in E and 16.1 % in N) were significantly increased compared to the control (Diatoms: 9.9 %; Chrysophyceae: 0 %). By contrast, dinoflagellates showed a sharp decrease in E (2.8 %) and N (1.4 %) compared to the control (14.1 % of total cell densities). Unidentified flagellates were stimulated in E microcosms (42.6 %) compared to the control and N microcosms (23.7 and 28 % respectively). In the offshore sample experiment, the diversities in C microcosms (1.41± 0.04 bits ind^{-1}) were significantly higher after 96h than in E (1.32± 0.002 bits ind^{-1}) and N microcosms (1.19 ± 0.03 bits ind^{-1}).

3.4. Effect of the treatments on diatom community composition

In station L, the initial diatom community was dominated by three pennate taxa, the potentially toxic *Pseudo-nitzschia* sp. (43%) and *Nitzschia longissima* (43%), followed by *Nitzschia* sp. (14%) (Table 2). After 4 days of incubation, the diatom diversity remained similar in the C and N microcosms, with a large contribution of *Pseudo-nitzschia* sp. (80-90%) and a small contribution of *Nitzschia longissima*

289 (5%) (Table 2). Until the end of incubation time, *Pseudo-nitzschia* spp. were the only diatoms
290 remaining under elutriate addition (E microcosms).

	Lagoon				Offshore			
	IS	C	E	N	IS	C	E	N
Asterionellopsis glacialis					31.4			
Chaetoceros spp.					25.6	11.8	3.5	62.8
Leptocylindrus sp.					4.1			
Nitzschia longissima	42.9	4.8		12.5	3.3		2.7	3.3
Nitzschia sp.	14.3							
Pseudo-nitzschia sp.	42.9	90.5	100	87.5	35.5	76.5	90.3	29.5
Thalassiosira sp.						11.8	3.5	4.4
Unidentified diatoms		4.8						

291 **Table 2.** Diatom species composition (expressed as % of total cell counts in the group) in initial
292 sample community (IS) and after four days in the control (C), sediment elutriate treatment (E), and
293 nitrogen enrichment treatment (N) for lagoon and offshore experiments. Empty cells: taxa not found
294 in the sample.

295 In the station O, the initial community of diatoms (Table 2) was dominated by potentially toxic
296 *Pseudo-nitzschia* spp. (35.5 %), *Asterionellopsis glacialis* (31.4 %) and *Chaetoceros* spp. (25.6 %). After
297 96h exposure, the two species *Asterionellopsis glacialis* and *Leptocylindrus* sp. disappeared in the
298 three treatments. Conversely, the centric diatom *Thalassiosira* sp. appeared in large numbers in C
299 (11.8 %), N (4.4 %) and E (3.5 %) microcosms. The potentially toxic *Pseudo-nitzschia* spp. were the
300 dominant diatom species in C and E microcosms (76.5 % and 90.3 %, respectively), accounting for a
301 smaller contribution in N microcosms (29.5 %) compared to the control (Table 2).

302 **3.5. Effect of the treatments on dinoflagellate community composition**

303 In station L, at the beginning of the experiment, the dinophyceae were dominated by *Alexandrium*
304 sp., which contributed up to 72% to total dinoflagellate community. The contributions of other
305 species varied from 0.9% (*Gonyaulax polygramma, Gonyaulax spinifera, Gyrodinium* sp., *Oblea* sp.,
306 *Prorocentrum lima* and *Protoperidinium* sp.) to 5.2% (*Prorocentrum gracile* and *Gymnodinium* sp.)
307 (Table 3).

308 At the end of incubation, the genus *Alexandrium* was still the dominant one in all three microcosms
309 (54.7 % in C, 56.0% in E and 74.2 % in N). There was also a significant development of *Prorocentrum*

310 species (*P. gracilis, P. micans* and *P. minimum*) in the N microcosm. *Katodinium* sp., which was
311 present in the control, disappeared from the E and N microcosms after the incubation.

	Lagoon				Offshore			
	IS	C	E	N	IS	C	E	N
Alexandrium sp.	72.4	54.7	56.1	74.2				
Gonyaulax polygramma	0.9		0.6	2.2				
Gonyaulax spinifera	0.9				1.9			
Gymnodinium sp.	5.2	13.1	19.5	5.6	52.8	51.6	57.1	16.7
Gyrodinium sp.	0.9	2.2	2.4		17.0			
Katodinium sp.		2.2						
Neoceratium fusus					1.9			
Oblea sp.	0.9	0.7	0.6					
Peridinium quinquecorne	2.6		0.6					
Prorocentrum gracile	5.2	4.4	3.7	6.7				16.7
Prorocentrum lima	0.9							
Prorocentrum micans	3.4		4.3	1.1	3.8	9.7	14.3	
Prorocentrum minimum	2.6	1.5	1.2	5.6		3.2		
Prorocentrum sp.						4.8		
Protoperidinium diabolus		1.5						
Protoperidinium sp.	0.9	0.7	1.2	3.4				
Scrippsiella sp.	2.6	2.9	1.2	1.1	1.9	6.5		
Unidentified dinoflagellates	0.9	16.1	8.5		20.8	24.2	28.6	66.7

312 **Table 3.** Dinoflagellates species composition (expressed as % of total cell counts in the group) in
313 initial sample community (IS) and after four days in the control (C), sediment elutriate treatment (E),
314 and nitrogen enrichment treatment (N) for lagoon and offshore experiments. Empty cells: taxa not
315 found in the sample.

316 In station O, the initial community of dinoflagellates was dominated by *Gymnodinium* spp. with
317 52.83% of relative contribution (Table 3), followed by unidentified dinoflagellates (20.8 %) and
318 *Gyrodinium* sp. (17 %). After 96h of incubation, *Gymnodinium* spp. were dominant in the control C
319 (51.6 %) and E treatment (57.1 %), followed by unidentified dinoflagellates (C: 24.2 %; E: 28.6 %). By
320 contrast, unidentified dinoflagellates were most abundant in the N microcosms (66.7 %) compared to
321 *Gymnodinium* spp. and *Gyrodinium* sp. (16.7 % for both).

322 **3.6. Effect of the treatments on the other phytoplankton community composition**

323 In station L, various flagellates belonging to several classes were present in the inoculum sample
324 dominated by *Ollicola vangoorii* and colored flagellates, together with few cyanobacteria,
325 chrysophyceae, unidentified chrysophyceae and chlorophyceae (Table 4). The chrysophyceae *O.*
326 *vangoorii* was the dominant species (59.3 %). At the end of incubation, the contribution of *O.*
327 *vangoorii* was lower in microcosms C (20.1 %) and N (4.7 %) relative to microcosms E (42.6 %). There

was a higher contribution of other unidentified flagellates in the three microcosm conditions (78.8 % in C, 56.4 % in E and 88.5 % in N) than in the inoculum.

	Lagoon				Offshore			
	IS	C	E	N	IS	C	E	N
cyanobacteria	0.6		0.5	6.6				
chlorophyceae	0.6		0.2		2.2	9.8	0.6	1.8
coccolithophorids					38.8	59.0	6.9	2.1
Eutreptiella braarudii		0.3	0.2					
Ollicola vangoorii	59.3	20.1	42.6	4.7	2.8		43.0	26.3
Unidentified chysophyceae	2.1	0.8		0.2				
Unidentified flagellates	37.4	78.8	56.4	88.5	56.2	31.1	49.6	69.8

Table 4. Chlorophyceae, cyanobacteria and pigmented flagellate species composition (expressed as % of total cell counts in the group) in initial sample community (IS) and after four days in the control (C), sediment elutriate treatment (E), and nitrogen enrichment treatment (N) for lagoon and offshore experiments. Empty cells: taxa not found in the sample.

In station O, the inoculum community was dominated by unidentified flagellates (56.2 %) and coccolithophorids (38.8 %). The remaining part of cell density was constituted by *Ollicola vangoori* and unidentified chlorophycea. After 96h, unidentified flagellates were dominant in the two treatments E and N (E: 49.6 %; N: 69.8 %) compared to the control (31.1 %). The opposite pattern was noticed for coccolithophorids (C: 59.0 %; N: 2.1 %; E: 6.9 %). For the chrysophyceae *Ollicola vangoorii*, almost the same evolution pattern was observed as in the station L with a large increase in cell numbers during the incubation. In fact, this species showed a higher contribution in E microcosms (43.0 %) compared to the N (26.3 %) and C (0 %) ones (Table 4).

3.7. Correspondence analysis

The correspondence analysis, of which the two main axes explained 60.4 % of the observed variances, highlighted the marked changes between phytoplankton composition at the initial and the end of the experiments in the stations L and O (Fig. 4). In lagoon station L, the final community composition in N and E microcosms differed from inoculum and control microcosms. The average treatment effect was more pronounced in offshore station compared to the lagoon station. Results demonstrated a pronounced divergence between the initial samples used as inoculum and the two treatments N and E after incubation (Fig. 4), the control remaining close to the inoculum. The species *Alexandrium* sp., *Chaetoceros* sp*., Pseudo-nitzschia* sp*.,* and *Ollicola vangorii* may have been mainly

responsible for these differences between the different microcosms (inoculum, C N and E). For the lagoon station, cyanobacteria *Spirulina* sp., and *Prorocentrum* species *(P. micans, P. gracile* and *P. minimum*) contributed to the difference between E treatment and the others (inoculum, N and C), whereas *Asterionellopsis glacialis,* coccolithophorids*, Prorocentrum micans, P. gracile, Gyrodinium* sp. and unidentified flagellates may have been mainly responsible for these differences between the different samples (inoculum, C, N and E) in the station O (data not illustrated).

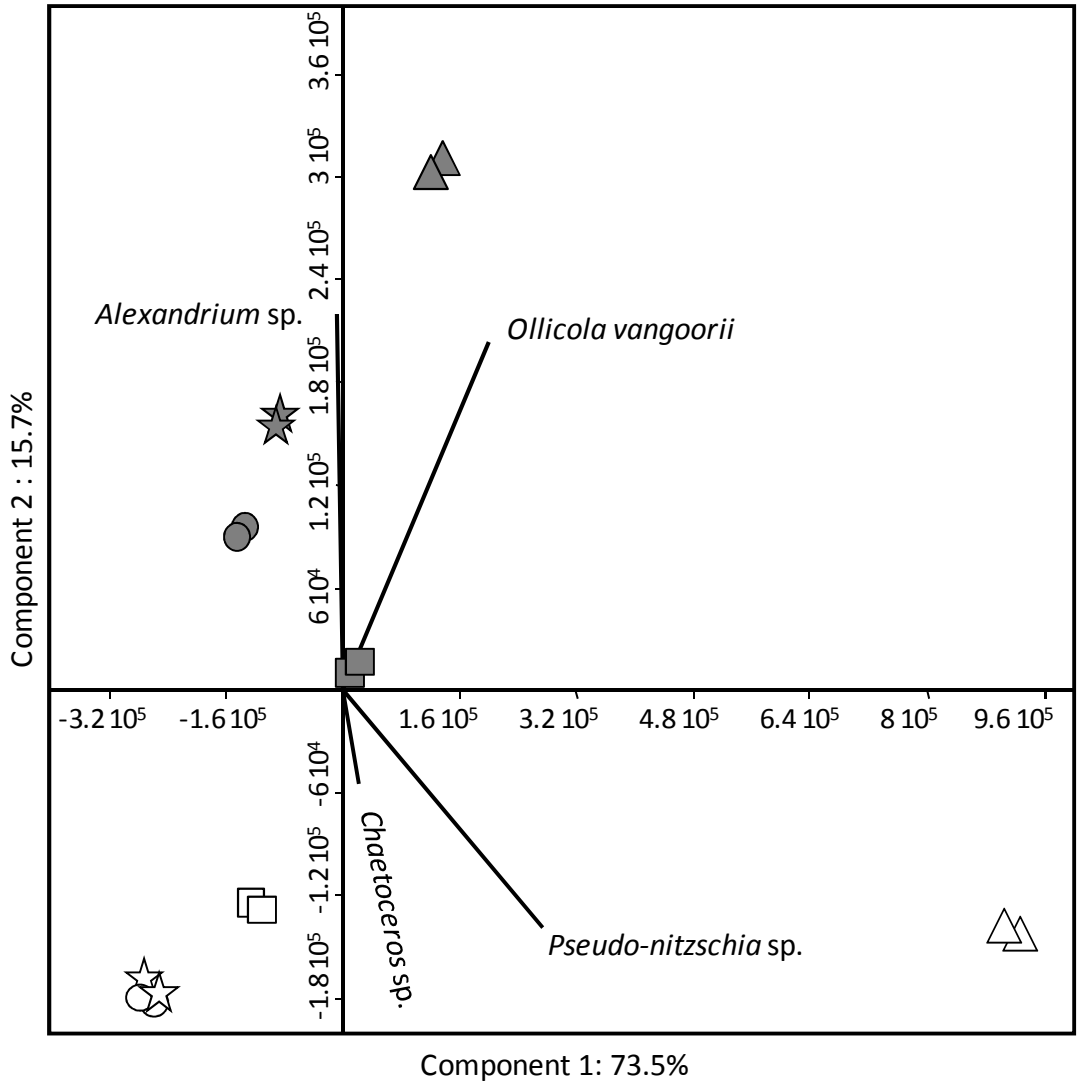


Figure 4. Correspondence analysis scatter plot obtained from the relative contributions of identified phytoplankton taxa for the Lagoon (grey symbols) and Offshore (open symbols) experiments. Stars: inoculum; circles: controls; squares: nitrogen enrichment; triangles: sediment elutriate treatment. Species with loadings >0.5 are shown.

362 **3.8. Effect of the treatments on community metabolism: production and respiration**

363 In the lagoon station L, gross primary production (GPP) rates showed a peak (> 5 µmol O_2 l^{-1} h^{-1}) in
364 nitrogen enriched microcosm relative to control and elutriate treatments after 48h of incubation (Fig.
365 5A). Nevertheless, oxygen evolution rates were almost identical in N and E treatments for all the
366 other incubation times, whereas GPP in controls was lower (below 2 µmol O_2 l^{-1} h^{-1}) and decreased
367 continuously, from start to 96 h. Both incubation time and treatment modified to metabolic pattern
368 (Table 5). In the offshore station, GPP values were significantly stimulated under the three
369 treatments after a short time, from 0.9 to 3.43 µmol O_2 l^{-1} h^{-1} in C, from 0.69 to 2.58 µmol O_2 l^{-1} h^{-1} in
370 N and from 1.32 to 3.91 µmol O_2 l^{-1} h^{-1} in E microcosms (Fig. 5B). Accordingly, during the later part of
371 the incubation period, a decrease in GPP was noticed in the three microcosms. GPP rates were
372 significantly stimulated in E microcosms compared to the C and N ones (Table 5, $p<0.05$). The
373 average GPP rate was 3.03 µmol O_2 l^{-1} h^{-1} in E, almost doubled compared to the C (1.55 µmol O_2 l^{-1} h^{-1})
374 and N microcosms (1.56 µmol O_2 l^{-1} h^{-1}) (Fig. 5B).

	Lagoon						Offshore					
variable	treatment	time	interaction	E-C	N-C	E-N	treatment	time	interaction	E-C	N-C	E-N
chl *a*	0.0000	0.0000	0.0049	*	*	*ns*	0.0000	0.0000	0.0000	*	*	*
GPP	0.0000	0.0000	0.0000	*	*	*	0.0000	0.0000	0.0000	*	*ns*	*
R_{dark}	0.0000	0.0000	0.0005	*	*	ns	0.0000	0.0000	0.0000	*	*ns*	*
NP	0.0004	0.0000	0.0000	*	*	*	*0.7165*	0.0000	0.0496	*ns*	*ns*	*ns*
P:R	0.0000	0.0000	0.0001	*ns*	*	*	0.0038	0.0000	0.0000	*	*	*ns*

375

376 **Table 5.** Output of multifactorial ANOVA on chl *a* concentrations and metabolic rates for the Lagoon
377 and Offshore experiments, depending on treatment or incubation time. GPP: gross primary
378 production; R_{dark}: community respiration in the dark; NP: net production; P:R: production to
379 respiration ratio. P-values are given for each variable; non significant differences are highlighted by
380 italics. Significant differences at 95% confidence levels for treatment comparison (elutriate-control:
381 E-C; nitrogen-control: N-C; elutriate-control: E-C) are marked with an asterisk. ns: non significant.

382

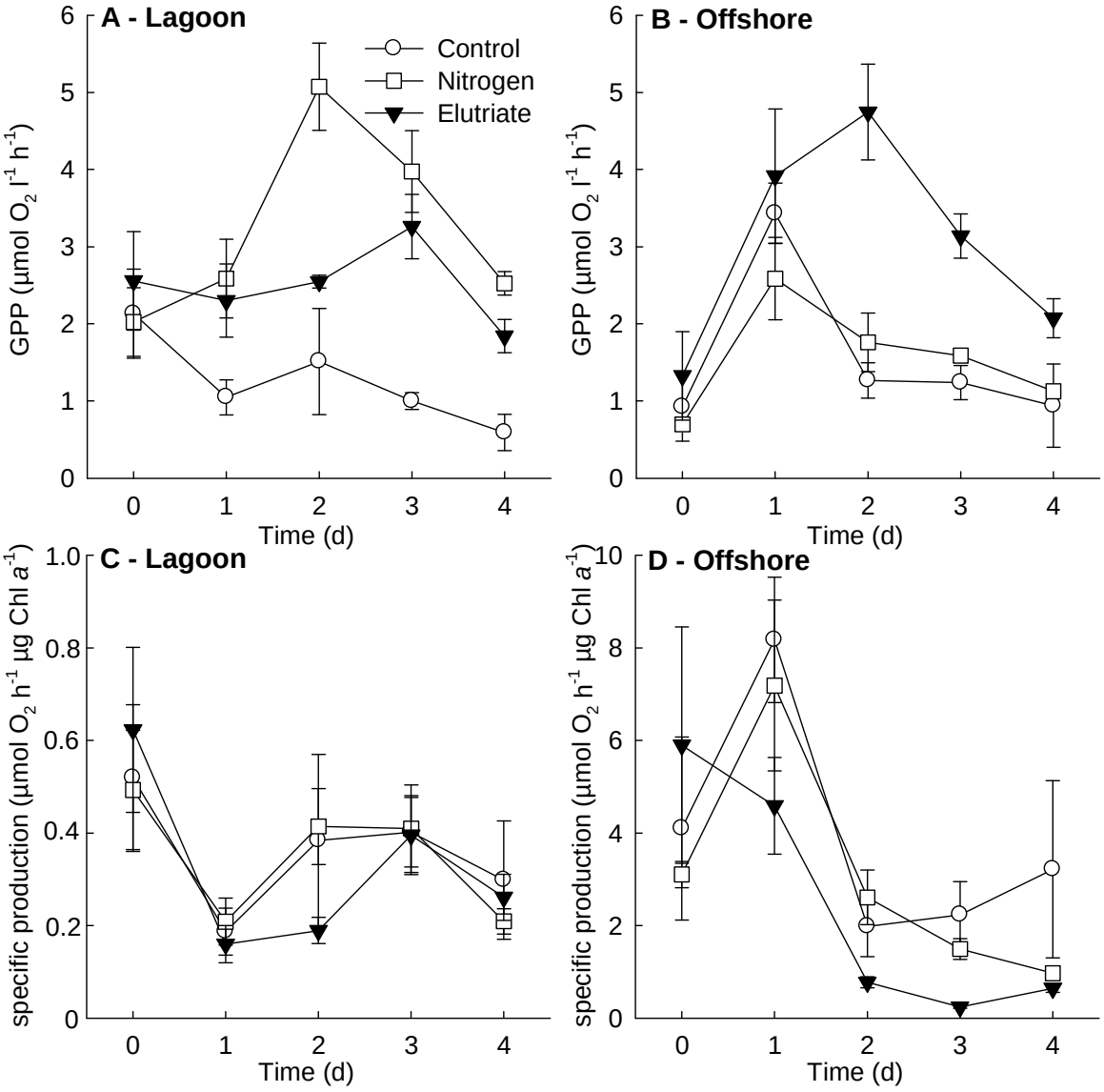
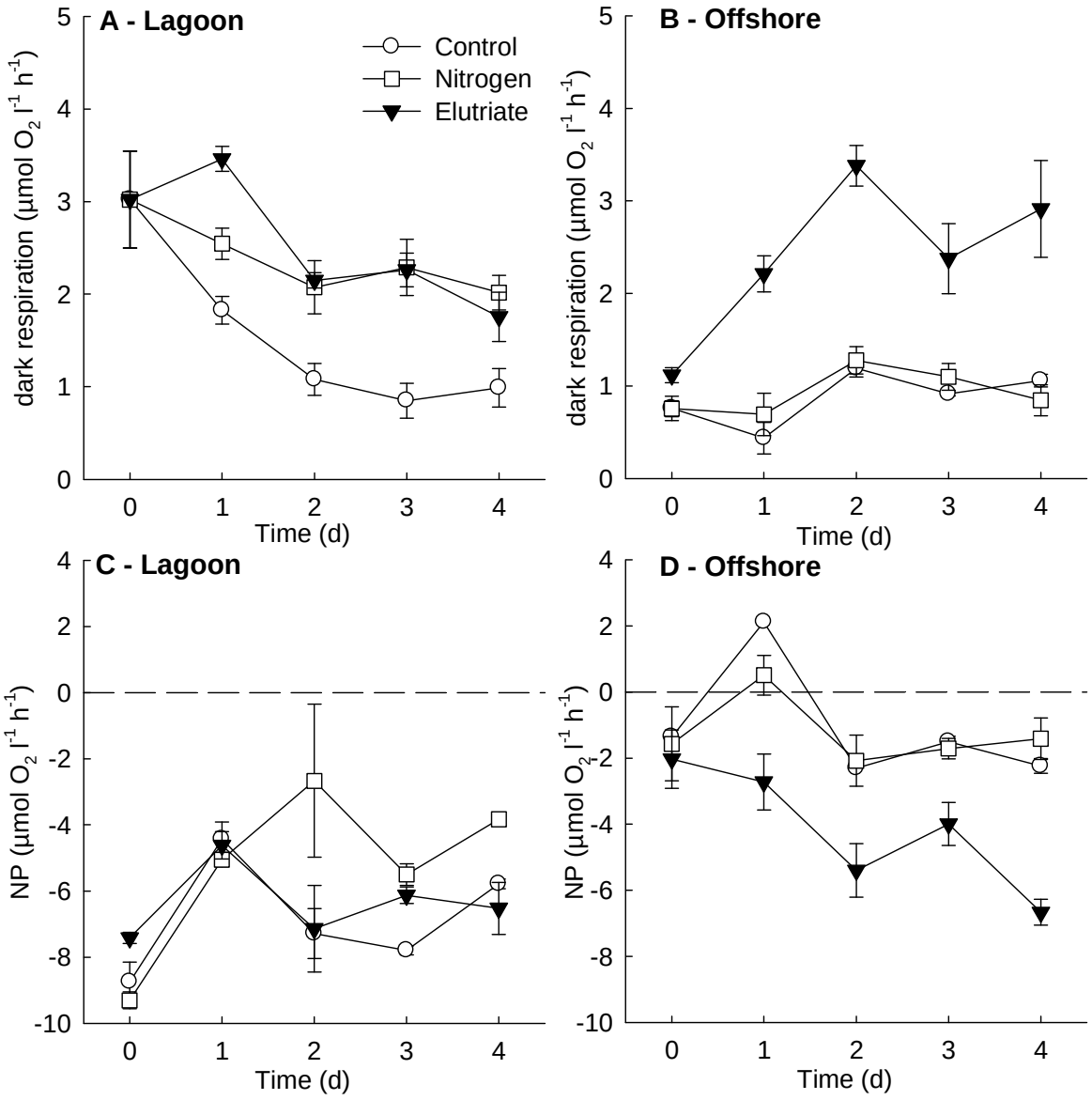


Figure 5. Gross oxygen production rates (GPP) measured in the Lagoon (Fig. 5A) and offshore (5B) experiment during the incubation, and specific oxygen production per unit of chl *a* in the same microcosms (Fig. 5C and 5D). Please note the different y scales for 5C and 5D graphs.

From GPP values and chl *a* concentrations, the specific phytoplankton activity was calculated (specific GPP = GPP / [chl *a*]). A significant decrease was observed in the oxygen productivity in the lagoon experiment over the first 24h with a fall from 0.52 to 0.18 µmol O_2 h^{-1} µg chl a^{-1} in C microcosms, 0.62 to 0.15 µmol O_2 h^{-1} µg chl a^{-1} in E microcosms and 0.49 to 0.2 µmol O_2 h^{-1} µg chl a^{-1} in N microcosms (Fig. 5C). No significant differences arose for specific gross primary production between the three treatments ($p>0.05$). GPP increased over the rest of the incubation period and the average specific

393 phytoplankton activity was 0.36, 0.3 and 0.34 µmol O_2 h^{-1} µg chl a^{-1} respectively for C, E and N
394 microcosms (Fig. 5C), without any significant differences between the three treatments ($p>0.05$). The
395 specific phytoplankton activity in the offshore experiment was marked by a significant increase at
396 24h in C (from 4.1 to 8.2 µmol O_2 h^{-1} µg chl a^{-1}) and N (from 3.1 to 7.2 µmol O_2 h^{-1}µg chl a^{-1})
397 microcosms at 24h but a decrease in E (from 6 to 4.5 µmol O_2 h^{-1} µg chl a^{-1}). For the remnant of the
398 incubation experiment, specific activities decreased regardless of treatment (Fig. 5D). In a general
399 way, specific activities of communities from the offshore station were one order of magnitude higher
400 than the ones measured for lagoon communities.

401 The same patterns were observed for dark respiration measurements compare to GPP. In Lagoon
402 experiment, R_{dark} increased significantly ($p < 0.05$) in the E and N microcosms relative to controls
403 during all the incubation period (Fig. 6A). The average respiration rates measured during the
404 experiment were 1.55; 2.23 and 2.39 µmol O_2 l^{-1} h^{-1} in the C, E and N microcosms, respectively.
405 Throughout the experiment, the net production did not reach positive values in any of the
406 microcosms. Compared to the lagoon station, the respiration in the station O was significantly
407 stimulated only in E microcosms (average rate: 2.39 µmol O_2 l^{-1} h^{-1}) compare to the C (average rate:
408 0.87 µmol O_2 l^{-1} h^{-1}) and N (average rate: 0.93 µmol O_2 l^{-1} h^{-1}) ones (Fig. 6B, Table 5). No significant
409 differences arose between the control and N microcosms ($p>0.05$). In the offshore station, and
410 throughout the experiment, net production was not positive in any of the microcosms, except for a
411 transient state after 24h incubation, where net autotrophy was noticed in C and N microcosms (Fig.
412 6C). On the contrary, in E microcosms the system appeared heterotrophic throughout the whole
413 incubation time (Fig. 6D, Table 5).



414
415 **Figure 6.** Dark respiration rates measured in the Lagoon (Fig. 6A) and Offshore (6B) experiments
416 during the incubation, and net oxygen production (NP) in the same microcosms (Fig. 6D and 6D).
417 Dashed line figures the exact balance between gross production and respiration (NP = 0).

418 # 4. Discussion

419 ## 4.1. Contaminants and nutrients released from the sediments

420 Concentrations of PAHs in Thau lagoon sediments found during this study were significant (ΣPAH16 =
421 8.7 µg g^{-1} dry weight), similar to previous reports from studies in the same area (ΣPAH16 = 0.44 to 7.7
422 µg g^{-1} dry weight, Léauté, 2008). Sediments from *Eaux Blanches* bay and *Crique de l'Angle* creek

(station S and station L respectively in Fig. 1) have been found to be mainly contaminated with PAHs, with total amounts of 8 and 7 µg g^{-1} dry weight, respectively (Léauté, 2008). PAHs concentrations in the sediments of the eastern part of Thau lagoon were similar to those found in other highly contaminated ecosystems, such as in sediments from Ushuaia Bay (7.6 µg g^{-1} dry weight; Massara Paletto *et al*., 2008), but much higher than those found in Daya Bay (South China) (0.04 to 0.158 µg g^{-1} dry weight; Yan *et al*., 2009), Gironde Estuary (up to 2 µg g^{-1} dry weight; Budzinski *et al.,* 1997) and Bizerte lagoon (1.49 µg g^{-1} dry weight; Lafabrie *et al.,* 2013b) for example. Polychlorobiphenyls (PCBs) were also found in significant levels in sediments, at higher concentrations (ΣPCB7 = 0.059 µg g^{-1} dry weight) than previously reported in the same area (ΣPCB7 = 0.0006 to 0.030 µg g^{-1} dry weight; Léauté, 2008). Despite these high levels in sediment, the concentrations of all PAHs and PCBs in the elutriate (CW filtered through 0.2 µm) were below detection limits (Table 1).These PAH and PCB contaminants have hydrophobic properties and a strong affinity for particles in the water column (Luellen and Shea, 2002; Tolosa *et al.,* 1995), and may have been removed by filtration together with particulate matter.

The shift from anoxic (sediment in place) to oxic conditions (suspended sediment) may change the partition and speciation of contaminants, particularly adsorbed trace metals prone to solubilization (Petersen *et al.,* 1996). Metals were not measured in the sediment in the present study, but previous published works reported high level of contamination in the *Eaux Blanches* bay (Kawakami *et al*., 2008). From the top five centimeters layer of a sediment core sampled in the same location (station 5, Fig. 2 in Kawakami *et al.*, 2008), these authors previously found concentrations of 93 µg.g^{-1} for lead, 2.2 µg.g^{-1} for cadmium, 95.3 µg.g^{-1} for copper, and 29.3 µg.g^{-1} for nickel (sediment dry weight). This could explain the high concentrations of trace metals in the elutriate (Table 1). A similar study performed in Biguglia lagoon (Corsica) also reported low levels of PAHs in total elutriates (although not filtered through 0.2 µm) ranging from 0.007 to 0.011 µg l^{-1}, but comparatively high concentrations of trace metals (ranging from 0.07 to 27µg l^{-1}) (Lafabrie *et al*., 2013a). Results

448 presented here show that the resuspension of sediments resulted mostly in the release of metal
449 contaminants and nutrients, especially ammonium (20 µM), in dissolved form in the water column.
450 Sediment quality guidelines and ecological safety values for contaminants in marine waters are still
451 debated, but several studies attempted to summarize toxicity values for individual chemicals in the
452 context of risk assessment. The measured concentration of chemicals and metals in the sediment
453 and elutriates used in the present study were compared to known concentrations with predicted
454 effects on biota (Table 6). Long *et al*. (1995) defined two threshold levels as Effect Range Low (ERL)
455 and Effect Range Median (ERM) derived from effect data distribution on sediment, namely the 10th
456 and 50th percentile respectively, whereas harmful concentration for 5% of the species (HC5) in sea
457 water were derived from species-sensitivity distribution curves (Ben Othman *et al*., 2012, Wang *et*
458 *al*., 2014, Wheeler *et al*., 2002). For all the values but one measured in the present study,
459 contaminant concentrations in sediment were between ERL and ERM, whereas contaminated water
460 values were always below documented HC5. Only naphthalene concentration in sediment was below
461 ERL (Table 6).

	in sediment (ppb = µg kg^{-1} dry weight)			in contaminated water (µg l^{-1})	
organics	measured*	ERL[§]	ERM[§]	measured*	HC5
Anthracene	220	85.3	1100	bdl	
Benzo(a)anthracene	780	261	1600	bdl	
Benzo(a)pyrene	870	430	1600	bdl	0.011[&]
Chrysene	720	384	2800	bdl	
Fluoranthene	1700	600	5100	bdl	21[!]
Naphthalene	30	160	2100	bdl	
Phenanthrene	740	240	1500	bdl	2.33[&]
Pyrene	1240	665	2600	bdl	1.09[&]
$\Sigma_{PCB\ congeners}$	59	22.7	180	bdl	
	(ppm = µg g^{-1} dry weight)			(µg l^{-1})	
metals	measured[#]	ERL[§]	ERM[§]	measured*	HC5[£]
Lead	93	46.3	218	0.088	238.2
Cadmium	2.2	1.2	9.6	0.021	8.22
Copper	95.3	34	270	2.49	22.7
Nickel	29.3	20.9	51.6	0.62	597.7

462 **Table 6.** Comparison of sediment and elutriate contamination levels (*: this study) to documented
463 sediment quality guidelines (ERL and ERM) and predicted ecological effects (HC5) for marine waters.
464 §: data from Long *et al.*, 1995; #: data from Kawakami *et al.*, 2008; &: data from Wang *et al.*, 2014; !:
465 data from Ben Othman *et al.*, 2012; £: data from Wheeler *et al.*, 2002. bdl: below detection limit.

466 **4.2. Effect of contaminated sediment elutriate on phytoplankton growth**

467 Both E and N treatments increased phytoplankton growth in lagoon station, resulting in final chl *a*
468 concentrations that were two and three times higher in the E and N microcosms, respectively, than
469 in the control microcosms. By comparison, in the offshore station, supplementation by ammonium
470 (N microcosms) did not induce increase ($p > 0.05$) in chlorophyll *a* compared to the control (Fig. 2B),
471 whereas E treatment resulted in a ten times increase compare to C and N treatments. These results
472 suggest that the nutrients counteract the possible harmful effects of contaminants on phytoplankton
473 and that nitrogen in the form of NH_4^+ was likely to be the most limiting element for the
474 phytoplankton in the lagoon station. In such a situation, nitrogen alone limits the growth of
475 phytoplankton and elutriate addition can release this limitation. In the offshore community, nitrogen
476 addition alone did not trigger phytoplankton growth, whereas an unknown limiting factor was made
477 available by elutriate addition.

478 Such phytoplankton growth stimulation was also observed for example after a wind-mixing event in a
479 shallow subtropical bay in Florida (Lawrence *et al.,* 2004) and the increases in NH_4^+ and PO_4^{3-}
480 concentrations in the water column were identified as the main causes of this effect. Lafabrie *et al.*
481 (2013b) showed that both elutriate-contaminated and nutrient enriched water increased
482 phytoplankton growth compared to the control in an experiment performed in the Bizerte Lagoon
483 (Tunisia). Other experiments by the same research group in the Biguglia lagoon (Corsica) suggest that
484 brief sediment resuspension events strongly stimulate growth of phytoplankton communities
485 (Lafabrie *et al*., 2013a), even when sediments are contaminated by toxic chemicals. By contrast,
486 other studies have reported a decrease in the total phytoplankton biomass after exposure to a
487 combination of contaminants (mix of trace elements) and nutrients (*e.g.*, Riedel *et al*., 2003). The
488 responses of phytoplankton to a combination of contaminants and nutrients may, therefore, depend
489 on threshold levels above which the contaminants become toxic.

490 There may be other mechanisms that stimulate phytoplankton growth in the present study. The E
491 microcosms were slightly enriched with metallic micronutrients, such as nickel and zinc (Table 1)

492 which are known to be essential for cell metabolism. These metals are involved in the stabilization of
493 protein structure and also facilitate transfer reactions and/or catalysis of the enzymatic reactions
494 (Torres *et al.,* 2008). For example, Zn plays a key role in the photosynthetic electron transport
495 between the thylakoids (Raven *et al.,* 1999). Molybdenum (which was the most abundant trace
496 metal in the E microcosms) Mn and Fe are involved as enzyme cofactors in respiration,
497 photosynthesis and nitrogen metabolisms and may be limiting for natural phytoplankton populations
498 *in situ* (Boyd *et al.,* 1996; Howarth and Cole 1985). Essential metal-micronutrients (Zn, Mo, Mn and
499 Fe) which were abundant in the elutriate could therefore favor algal growth provided that their
500 concentrations do not exceed the tolerance thresholds of organisms. It has been shown that some
501 dissolved metals (Co, Ni and Zn) are able to increase the chlorophyll biomass, but they may also be
502 harmful at higher concentrations (Chakraborty *et al*., 2010).

503 **4.3. Effect of contaminated sediment elutriate on phytoplankton structure**

504 The results of the present study in lagoon station showed that diatoms were stimulated in the N
505 microcosms compared to other treatments. This appeared contradictory with Lafabrie *et al*. (2013a)
506 who showed that the high NH_4^+ levels (100µM) in their experiment resulted in a decrease in diatom
507 abundance. Actually, this concentration was 5 times higher than the NH_4^+ levels (20µM) used in the
508 present study, and can be explained by a potential toxicity of ammonium at high concentrations. In
509 the offshore station, diatoms were stimulated in the E microcosms compare to other treatments.
510 Chikhaoui *et al*. (2008) highlighted the fact that dissolved silicate can be a limiting factor and an
511 essential element to the growth and metabolism of diatoms communities, even in coastal, nutrient-
512 rich waters. Diatoms in the E microcosms were represented exclusively by potentially toxic *Pseudo-*
513 *nitzschia* species in both experiments (inoculum from lagoon and offshore stations). This genus has
514 been shown to produce domoic acid, an amnesic shellfish toxin, which is able to chelate various
515 metals (such as Fe and Cu) and, therefore, can counteract the toxicity of these pollutants (Lelong *et*
516 *al.,* 2012; Rue and Bruland, 2001). It has also been reported that some *Pseudo-nitzschia* species (*P.*
517 *australis* and *P. multiseries*) withstand toxic concentrations of Cu by increasing domoic acid

production (Ladizinsky, 2003; Maldonado *et al.,* 2002). This could suggest that the dominance of *Pseudo-nitzschia* in the E microcosms in the two stations, enriched in trace metals from sediment, may be related to their ability to produce domoic acid. This makes them more resistant to metal toxicity than other species that do not have this ability, such as *Nitzschia longissima* in lagoon station, which disappeared from the E microcosms, and the decrease of the contribution of *Chaetoceros* spp. in offshore station, suggesting that the these species were sensitive to dissolved trace metals.

An increase in the cyanobacteria biomass was observed in lagoon station in the ammonium enriched microcosms (N microcosms). Such a pattern was also reported by Fouilland *et al.* (2012), concluding that inorganic nitrogen limited the growth of prokaryotic picophytoplankton in Thau lagoon. More generally, cyanobacteria are often considered as more tolerant to chemicals, such as reported for example in the study of Stachowski-Haberkorn *et al.* (2009), after exposing marine phytoplankton to a fungicide (Opus at 100µg l^{-1}) which resulted in an increase in *Synechococcus* population density.

Regarding the dinoflagellates group, the persistence of *Alexandrium* sp. and relatively high abundance in the E microcosms (with high level of heavy metals) in the lagoon experiment suggests a relative tolerance of this species to metals. The close relative *Alexandrium catenella* was shown to be able to resist to Cu toxicity by increasing dissolved organic carbon (DOC) release (Herzi *et al*., 2013). For dinoflagellates from offshore station, many species in E and N microcosms were depauperate compare to the control microcosms. These species were *Scrippsiella* sp.; *Prorocentrum* sp. and *Prorocentrum minimum*, appearing therefore not tolerant to the sediment elutriate.

Correspondence analysis highlighted the fact that treatments effects were more pronounced in offshore station compared to lagoon station (Fig. 5), and demonstrated a wide divergence between the initial sample and the three treatments C, N and E microcosms in offshore station. This suggests that phytoplankton sampled from offshore station were more sensitive to the sediment elutriate effects and to the added ammonia compared to the phytoplankton from lagoon station. This can be supported by the observed changes in Shannon index *H* values. At the end of incubations, the phytoplankton diversity in lagoon station did not differ in the E and N treatments compared to the

544 control (*H* index: 1.7 ± 0.03; 1.55 ± 0.02 and 1.6 ± 0.01 bits ind^{-1}, respectively in C, E and N
545 microcosms) whereas in offshore station the diversity in C microcosms (1.41± 0.04 bits ind^{-1}) was
546 significantly higher than in E (1.32± 0.002 bits ind^{-1}) and N microcosms (1.19 ± 0.03 bits ind^{-1}).

547 **4.4. Effect of contaminated sediment elutriate on community production and respiration**

548 In lagoon experiment, community respiration followed the same trend as phytoplankton biomass.
549 Actually, the R_{dark} increased 3 times in N and E microcosms compared to the control ($P < 0.05$). For the
550 offshore experiment, only E treatment resulted in an up to 3-fold increase in dark respiration
551 compared to control microcosms. In this last experiment, R_{dark} did not change compared to the
552 control ($P > 0.05$) under N treatment. This suggests that the respiration in lagoon station was limited
553 by nitrogen, whereas this element was not a limiting factor for respiratory activity at offshore
554 station.

555 The specific phytoplankton activity (GPP:chl *a*) decreased significantly in the control and both
556 treatments (N and E microcosms) during all the incubation time in lagoon station. Almost the same
557 pattern was observed in the three microcosms treatments in offshore station but, only after 24h, a
558 transient increase of GPP:chl *a* was noticed only in C and N treatments compared to E ones. The
559 studied phytoplankton communities in N and C microcosms appeared physiologically more efficient
560 than in E microcosms after 24h incubation in offshore station. Increase in the specific phytoplankton
561 activity, together with slight phytoplankton community structure changes, have been previously
562 reported in a mesocosm experiment for communities exposed to a mix of nutrients with an organic
563 contaminant (pyrene; Hjorth *et al.*, 2008).

564 The net production gives information on the "trophic state" of a system, currently linked to the
565 production:respiration ratio; $P:R > 1$ indicates a net autotrophy state, whereas $P:R < 1$ indicates a net
566 heterotrophy state (Dodds and Cole, 2007). In lagoon station, net production was always negative,
567 highlighting the fact that the three microcosms C, N and E were net heterotrophic during all the
568 incubation time. The systems were also heterotrophic in the offshore station in E, N and C
569 microcosms but for this experiment a transient state appeared for only C and N microcosms, during

570 which the plankton communities were net autotrophic. Previous field-based studies also highlighted
571 the drift of planktonic communities' metabolism towards heterotrophy shortly after sediment
572 resuspension events (Cotner *et al.*, 2000; Lawrence *et al.*, 2004). Similarly, laboratory experiments
573 showed a reduction of the *P*:*R* ratio after exposure to trace metal, reduction of the *P*:*R* ratio that
574 resulted in a bacterially dominated heterotrophic system (Nayar *et al.*, 2004; Rochelle-Newall *et al.*,
575 2008).

576 ## 5. Conclusion

577 This ecotoxicological experiment, conducted on natural phytoplankton sampled from the Thau
578 lagoon and Mediterranean offshore waters, showed that elutriate prepared from sediments
579 stimulated phytoplankton growth despite significant contamination by potentially toxic chemicals.
580 Sediments involved in exposure experiment were considered as significantly contaminated by metals
581 and organics based on quality assessment thresholds. For both phytoplankton communities,
582 significant increases in phytoplankton biomass and cell abundance were observed after elutriate
583 addition. For the lagoon station, the stimulatory effect of elutriate was similar to that of adding NH_4^+
584 alone, without any toxic chemical. It is, therefore, likely that the nutrient present in elutriate
585 (especially nitrogen) caused the proliferation of primary producers. On the other hand, the
586 phytoplankton community from the offshore reference station was clearly beneficial from elutriate
587 addition based on observed growth, since nitrogen alone did not promote community proliferation.
588 This may mask the possible harmful effects of contaminants, making it difficult to detect them. On
589 the contrary, the community in the reference station was not limited by the NH_4^+ addition.
590 The elutriate used in the present study was particularly contaminated with dissolved metals, some of
591 these known to be toxic to phytoplankton (Cd, Pb), while others (Zn, Mo, Mn, Fe) may be essential
592 for cellular metabolism at low concentrations. As a consequence, the presence of essential trace
593 metals in sediment elutriate could be a possible cause of increased phytoplankton growth, provided
594 that their levels remain below the tolerance threshold of these organisms.

595 Contamination also changed the community structure of Thau and offshore phytoplankton. It
596 appears that adding sediment elutriate was most beneficial to species that were potentially toxic.
597 The most significant change concerned the pronounced proliferation of potentially toxic diatoms
598 (*Pseudo-nitzschia* sp.) and dinoflagellates (*Alexandrium* sp. in lagoon station and *Gymnodinium* sp. in
599 offshore station) after the addition of sediment elutriate. Hence, it could be suggested that these
600 microalgae have defense mechanisms (synthesis of toxin excretion of dissolved organic matter, for
601 example) making them more resistant to chemical contamination (especially metallic in the present
602 study). For the phytoplankton in offshore, and in the very short term, the system shifted to
603 autotrophy after sediment elutriate addition compared to the control. By contrast, the lagoon
604 community metabolic balance remained net heterotrophic during all the experiments and for all
605 treatments. Apart from changes in community composition, such a transient shift in the global
606 metabolic balance of plankton community should be considered when addressing the ecological
607 impact of contaminated sediment in coastal marine ecosystems. The present results showed also
608 that the treatments effects were more pronounced in offshore station compared to the lagoon
609 station, suggesting that phytoplankton in offshore station were more sensitive to both elutriate and
610 added ammonia compared to the phytoplankton in lagoon station. Future progress in the risk
611 assessment of contaminated sediment for planktonic organisms will benefit from enhanced
612 knowledge on toxicity of pure chemicals to marine microorganisms (e.g. Ben Othman *et al*., 2012)
613 together with community level evaluation of toxic chemical cocktails taking into consideration the
614 stimulating effects of co-released nutrients (Pringault *et al*., 2016). These approaches will help in
615 discriminating true toxic effects, linked to toxic contamination, from nutrient enrichment effects that
616 accompany sediment resuspension events.

617 **Acknowledgements**

618 This work was partly funded by the BPi France agency and the Région Languedoc-Roussillon, program
619 ECODREDGE-MED grant number F11060012J. H.B.O. benefited from a PhD grant by the French Institute of
620 Research for Development (IRD). Claire Carré is warmly acknowledged for her help during microscopic

phytoplankton determination. The Ecole des Mines d'Alès (Dr. Catherine Gonzalez) and the UMR 5569 HydroSciences (Dr. Françoise Elbaz-Poulichet) provided the analytical facilities for chemical contaminants determination. Experimental work benefited from the logistical and technical support of the MEDIMEER experimental platform (OSU OREME, Sète). English text was edited by Tony Tebby. Anonymous referees are warmly acknowledged for triggering substantial enhancements on the early version of this manuscript.

References

Anschutz, P., Chaillou, G., Lecroart, P., 2007. Phosphorus diagenesis in sediment of the Thau lagoon. Estuar. Coastal Shelf Sci. 72, 447-456.

Ben Othman, H., Leboulanger, C., Le Floc'h, E., Hadj Mabrouk, H., Sakka Hlaili, A., 2012. Toxicity of benz(a)anthracene and fluoranthene to marine phytoplankton in culture: does cell size really matter? J. Hazardous Mater. 243, 204-211.

Birch, G., O'Hea, L., 2007. The chemistry of suspended particulate material in a highly contaminated embayment of Port Jackson (Australia) under quiescent, high-wind and heavy-rainfall conditions. Environ. Geol. 53, 501-516.

Bonnet, C., Babut, M., Férard, J.F., Martel, L., Garric, J., 2000. Assessing the potential toxicity of resuspended sediment. Environ. Toxicol. Chem. 19, 1290-1296.

Boyd, P.W., Muggli, D.L., Varela, D.E., Golblatt, R.H., Chretien, R., Orians, K.J., Harrison, P.J., 1996. *In vitro* iron enrichments in the NE subarctic Pacific. Mar. Ecol. Progr. Ser. 136, 179-193.

Budzinski, H., Jones, I., Bellocq, J., Piérard, C., Garrigues, P., 1997. Evaluation of sediment contamination by polycyclic aromatic hydrocarbons in the Gironde estuary. Mar. Chem. 58, 85-97.

Burton, G.A., Johnston, E.L., 2010. Assessing contaminated sediments in the context of multiple stressors. Environ. Toxicol. Chem. 29, 2625-2643.

Cantwell, M., Burgess, R., Kester, D., 2002. Release and phase partitioning of metals from anoxic estuarine sediments during periods of simulated resuspension. Environ. Sci. Technol. 36, 5328-5334.

Castro-Jiménez, J., Deviller, G., Ghiani, M., Loos, R., Mariani, G., Skejo, H., Umlauf, G., Wollgast, J., Laugier, T., Héas-Moisan, K. , Léauté, F., Munschy, C., Tixier, C., Tronczynski, J., 2008. PCDD/F and PCB multi-media ambient concentrations, congener patterns and occurrence in a Mediterranean coastal lagoon (Etang de Thau, France). Environ. Poll. 156, 123-135.

Chakraborty, P., Babu, P.V.R., Acharyya, T., Bandyopadhyay, D., 2010. Stress and toxicity of biologically important transition metals (Co, Ni, Cu and Zn) on phytoplankton in a tropical freshwater system: An investigation with pigment analysis by HPLC. Chemosphere 80, 548−553.

Chikhaoui, M.A., Sakka Hlaili, A., Hadj Mabrouk, H., 2008. Seasonal phytoplankton responses to N:Si:P enrichment ratio in the Bizerte Lagoon (Southwestern Mediterranean). Comptes Rendus Biologies 331, 389-408.

Cornelissen, G., Arp, H., Pettersen, A., Hauge, A., Breedveld, G., 2008. Assessing PAH and PCB emissions from the relocation of harbour sediments using equilibrium passive samplers. Chemosphere 72, 1581-1587.

Cotner, J.B, Johengen, T.H., Biddanda, B.A., 2000. Intense winter heterotrophic production stimulated by benthic resuspension. Limnol. Oceanogr. 45, 1672-1676.

Crain, C.M., Kroeker, K., Halpern, B.S., 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecol. Lett. 11, 1304-1315.

De Hoop, L., De Troch, M., Hendricks, A.J., De Laender, F., 2013. Modeling toxic stress by atrazine in a marine consumer-resource system. Environ. Toxicol. Chem. 32, 1088-1095.

de la Broise, D., Palenik, B., 2007. Immersed *in situ* microcosms: A tool for the assessment of pollution impact on phytoplankton. J. Exp. Mar. Biol. Ecol. 341, 274-281.

Dodds, W.K., Cole, J.J., 2007. Expanding the concept of trophic state in aquatic ecosystems: It's not just the autotrophs. Aquat. Sci. 69, 427−439.

Edge, K.J., Dafforn, K.A., Simpson, S.L., Ringwood, A.H., Johnston, E.L., 2015. Resuspended contaminated sediments cause sublethal stress to oysters: a biomarker differentiates total suspended solids and contaminant effects. Environ. Toxicol. Chem. 34, 1345-1353.

Edge, K.J., Johnston, E.L., Dafforn, K.A., Simpson, S.L., Tutti, T., Bannister, R.J., 2016. Sub-lethal effects of water-based drilling muds on the deep-water sponge *Geodia barretti*. Environ. Poll. 212, 525-534.

672 Field, C.B., Behrenfeld, M.J., Randerson, J.T., Falkowski, P.G. 1998. Primary production of the biosphere:
673 integrating terrestrial and oceanic components. Science. 281, 237-240.
674 Fouilland, E., Trottet, A., Bancon-Montigny, C., Bouvy, M., Le Floc'h, E., Gonzalez, J.L., Hatey, E., Mas, S.,
675 Mostajir, B., Nouguier, J., Pecqueur, D., Rochelle-Newall, E., Rodier, C., Roques, C., Salles, C., Tournoud,
676 M.G., Vidussi, F., 2012. Impact of a river flash flood on microbial carbon and nitrogen production in a
677 Mediterranean Lagoon (Thau Lagoon, France). Estuar. Coastal Shelf Sci. 113, 192-204.
678 Gilbert, F., Souchu, P., Bianchi, M., Bonin, P., 1997. Influence of shellfish farming activities on nitrification
679 nitrate reduction to ammonium and denitrification at the water -sediment interface of the Thau lagoon
680 France. Mar. Ecol. Progr. Ser. 151, 143-153.
681 Hammer, Ø., Harper, D.A.T., 2001. PAST: paleontological statistics software package for education and data
682 analysis. Paleontologica Electronica 41, 9pp.
683 Herzi, F., Jean, N., Zhao, H.Y., Mounier, S., Hadj Mabrouk, H., Sakka Hlaili, A., 2013. Copper and cadmium
684 effects on growth and extracellular exudation of the marine toxic dinoflagellate *Alexandrium catenella*:
685 3D-fluorescence spectroscopy approach. Chemosphere 93, 1230-1239.
686 Hill, N., King, C., Perrett, L., Johnston, E., 2009. Contaminated suspended sediments toxic to an Antarctic filter
687 feeder: aqueous- and particulate-phase effects. Environ. Toxicol. Chem. 28, 409-417.
688 Hjorth, M., Forbes, V.E., Dahllof, I., 2008. Plankton stress responses from PAH exposure and nutrient
689 enrichment. Mar. Ecol. Progr. Ser. 363, 121–130.
690 Holmes, M.R., Aminot, A., Kérouel, R., Hooker, B.A., Peterson, B.J., 1999. A simple and precise method for
691 measuring ammonium in marine and freshwater ecosystems. Can. J. Fish. Aquat. Sci. 56, 1801-1808.
692 Howarth, R.W., Cole, J.J., 1985. Molybdenum availability, nitrogen limitation, and phytoplankton growth in
693 natural waters. Science 229, 653-655.
694 Jonas, P., Millward, G., 2010. Metals and nutrients in the Severn Estuary and Bristol Channel: contemporary
695 inputs and distributions. Mar. Poll. Bull. 61, 52-67.
696 Jones-Lee, A., Lee, G., 2005. Role of iron chemistry in controlling the release of pollutants from resuspended
697 sediments. Remediation 16, 33-41.
698 Josefsson, S., Leonardsson, K., Gunnarsson, J., Wiberg, K., 2010. Bioturbation-driven release of buried PCBs and
699 PBDEs from different depths in contaminated sediments. Environ. Sci. Technol. 44, 7456-7464.
700 Kalnejais, L., Martin, W., Bothner, M., 2010. The release of dissolved nutrients and metals from coastal
701 sediments due to resuspension. Mar. Chem. 121, 224-235.
702 Kawakami, S.K., Seidel, J.L., Elbaz-Poulichet, F., Achterberg, E.P., 2008. Trace-metal biogeochemistry in the
703 Mediterranean Thau Lagoon, a shellfish farming area. J. Coastal Res. 24, 194-202.
704 Laabir, M., Jauzein, C., Genovesi, B., Masseret, E., Grzebyk, D., Cecchi, P., Vaquer, A., Perrin, Y., Collos, Y., 2011.
705 Influence of temperature, salinity and irradiance on the growth and cell yield of the harmful red tide
706 dinoflagellate *Alexandrium catenella* colonizing Mediterranean waters. J. Plankton Res. 33, 1550-1563.
707 Ladizinsky, N., 2003. The influence of dissolved copper on the production of domoic acid by *Pseudo-nitzschia*
708 species in Monterey Bay, California: laboratory experiments and field observations. California State
709 University. Monterey Bay, CA, USA, 68 p.
710 Lafabrie, C., Garrido, M., Leboulanger, C., Cecchi, P., Grégori, G., Pasqualini, V., Pringault, O., 2013a. Impact of
711 contaminated-sediment resuspension on phytoplankton in the Biguglia lagoon (Corsica, Mediterranean
712 Sea). Estuar. Coastal Shelf Sci. 130, 70-80.
713 Lafabrie, C., Sakka Hlaili, A., Leboulanger, C., Tarhouni, I., Ben Othman, H., Mzoughi, N., Chouba, L., Pringault,
714 O., 2013b. Contaminated sediment resuspension induces shifts in phytoplankton structure and function
715 in a eutrophic Mediterranean lagoon. Knowledge Managt. Aquatic Ecosyst. 410, 05. DOI:
716 10.1051/kmae/2013080
717 Lawrence, D., Dagg, M.J., Liu, H.B., Cummings, S.R., Ortner, P.B., Kelble, C., 2004. Wind events and benthic-
718 pelagic coupling in a shallow subtropical bay in Florida. Mar. Ecol. Progr. Ser. 266, 1-13.
719 Léaute, F., 2008. Biogéochimie des contaminants organiques HAP, PCB et pesticides organochlorés dans les
720 sédiments de l'étang de Thau. [*Biogeochemistry of organic contaminants PAH, PCB and organochlorine*
721 *pesticides in Thau lagoon sediments*]. PhD Thesis, 255 pp., full text: http://tel.archives-
722 ouvertes.fr/docs/00/44/95/16/ PDF/THESE F LEAUTE.pdf
723 Lelong, A., Jolley, F., Soudant, P., Hégaret, H., 2012. Impact of copper exposure on *Pseudo-nitzschia* spp.
724 physiology and domoic acid production. Aquat. Toxicol. 118, 37-47.
725 Levin, L.A., Boesch, D.F., Covich, A., Dahm, C., Erseus, C., Ewel, K.C., Kneib, R.T., Moldenke, A., Palmer, M.A.,
726 Snelgrove, P., Strayer, D., Weslawski, J.M., 2001. The function of marine critical transition zones and the
727 importance of sediment biodiversity. Ecosystems 4, 430-451.

728 Long, E.R., MacDonald, D.D., Smith, S.L., Calder, F.D., 1995. Incidence of adverse biological effects within ranges
729 of chemical concentrations in marine and estuarine sediments. Environ. Managt. 19, 81-97.
730 Luellen, D.R., Shea, D., 2002. Calibration and field verification of semipermeable membrane devices for
731 measuring polycyclic aromatic hydrocarbons in water. Environ. Sci. Technol. 36, 1791-1797.
732 Maldonado, M.T., Hughes, M.P., Rue, E.L., Wells, M.L., 2002. The effect of Fe and Cu on growth and domoic
733 acid production by *Pseudo-nitzschia multiseries* and *Pseudo-nitzschia australis*. Limnol. Oceanogr. 47,
734 515-526.
735 Massara Paletto, V., Commendatore, M.G., Esteves, J.L., 2008. Hydrocarbon levels in sediments and bivalve
736 mollusks from Bahía Nueva. (Patagonia, Argentina): An assessment of probable origin and
737 bioaccumulation factors. Mar. Poll. Bull. 56, 2082-2105.
738 Nayar, S., Goh, B.P., Chou, L.M., 2004. Environmental impact of heavy metals from dredged and resuspended
739 sediments on phytoplankton and bacteria assessed in *in situ* mesocosms. Ecotox. Environ. Safety 59,
740 349–369.
741 Neveux, J., Lantoine, F., 1993. Spectrofluorometric assay of chlorophylls and phaeopigments using the least
742 squares approximation technique. Deep-Sea Res. Part I 40, 1747-1765.
743 Novarino, G., Oliva, E., Pérez-Uz, B., 2002. Nanoplankton protists from the Western Mediterranean Sea. I.
744 Occurrence, ultrastructure, taxonomy, and ecological role of the mixotrophic flagellate *Ollicola vangoorii*
745 (Chrysomonadidae = Chrysophyceae p.p.). Scientia Marina 66, 233-247.
746 Petersen, W., Hong, J., Willamowski, C., Wallmann, K., 1996. The influence of diagenetic processes on the
747 exchange of trace contaminants at the sediment-water interface. Arch. Hydrobiol. SI Advanc. Limnol.
748 47: 295-305.
749 Pringault, O., Lafabrie, C., Avezac, M., Bancon-Montigny, C., Carré, C., Chalghaf, M., Delpoux, S., Duvivier, A.,
750 Elbaz-Poulichet, F., Gonzalez, C., Got, P., Leboulanger, C., Spinelli, S., Sakka Hlaili, A., Bouvy, M., 2016.
751 Consequences of contaminant mixture on the dynamics and functional diversity of bacterioplankton in a
752 southwestern Mediterranean coastal ecosystem. Chemosphere 144, 1060-1073.
753 Pringault, O., Tassas, V., Rochelle-Newall, E., 2007. Consequences of respiration in the light on the
754 determination of production in pelagic systems. Biogeosciences 4, 105-114.
755 Raven, J.A., Evans, M.C.W., Korb, R.E., 1999. The role of trace metals in photosynthetic electron transport in O_2-
756 evolving organisms. Photosynth. Res. 60, 111-149.
757 Riedel, G.F., Sanders, J.G., Breitburg, D.L., 2003. Seasonal variability in response of estuarine phytoplankton
758 communities to stress: Linkages between toxic trace elements and nutrient enrichment. Estuaries 26,
759 323–338.
760 Rigollet, V., Sfriso, A., Marcomini, M., De Casabianca, M.L., 2004. Seasonal evolution of heavy metal
761 concentrations in the surface sediments of two Mediterranean *Zostera marina* L. beds at Thau lagoon
762 (France) and Venice lagoon (Italy). Biores. Technol. 95, 159-167.
763 Roberts, D.A., 2012. Causes and ecological effects of resuspended contaminated sediments (RCS) in marine
764 environments. Environ. Internat. 40, 230-243.
765 Rochelle-Newall, E.J., Delesalle, B., Mari, X., Rouchon, C., Torreton, J.P., Pringault, O., 2008. Zinc induces shifts
766 in microbial carbon flux in tropical coastal environments. Aquatic Microb. Ecol. 52, 57–68.
767 Rue, E., Bruland, K., 2001. Domoic acid binds iron and copper: a possible role for the toxin produced by the
768 marine diatom *Pseudo-nitzschia*. Mar. Chem. 76, 127-134.
769 Stachowski-Haberkorn, S., Quiniou, L., Beker, B., Haberkorn, H., Marie, D., de la Broise, D., 2009. Comparative
770 study of three analysis methods (TGGE, flow cytometry and HPLC) for xenobiotic impact assessment on
771 phytoplankton communities. Ecotoxicology 18, 364-376.
772 Tolhurst, L., Barry, J., Dyer, R., Thomas, K., 2007. The effect of resuspending sediment contaminated with
773 antifouling paint particles containing Irgarol 1051 on the marine macrophyte *Ulva intestinalis*.
774 Chemosphere 68, 1519-1524.
775 Tolosa, I., Bayona, J.M., Albaigés, J., 1995. Spatio temporal distribution, fluxes, and budgets of
776 organochlorinated compounds in Northwest Mediterranean sediments. Environ. Sci. Technol. 29, 2519-
777 2527.
778 Torres, M.A., Barros, M.P., Campos, S.C.G., Pinto, E., Rajamani, S., Sayre, R.T., Colepicolo, P., 2008. Biochemical
779 biomarkers in algae and marine pollution: A review. Ecotox. Environ. Safety 71, 1-15.
780 Tournoud, M.G., Payraudeau, S., Cernesson, F., Salles, C., 2006. Origins and quantification of nitrogen inputs
781 into a coastal lagoon: application to the Thau Lagoon (France). Ecol. Modelling 193, 9-33.
782 Trottet, A., Leboulanger, C., Vidussi, F., Pete, R., Bouvy, M., Fouilland, E., 2016. Heterotrophic bacteria show
783 weak competition for nitrogen in Mediterranean coastal waters (Thau Lagoon) in autumn. Microbial
784 Ecol. 71, 304-314.

785 Utermöhl, H., 1958. Zur Vervollkommnung der quantitativen Phytoplankton-Methodik [For the perfection of
786 quantitative phytoplankton methodology], Mitteilungen internationale Vereinigung Theoretische und
787 Angewandte Limnologie [Communications of international symposium on theoretical and applied
788 limnology] 9: 1−38.
789 Voie, Ø., Johnsen, A., Rossland, H., 2002. Why biota still accumulate high levels of PCB after removal of PCB
790 contaminated sediments in a Norwegian fjord. Chemosphere 46, 1367-1372.
791 Wang, Y., Wang, J., Mu, J., Wang, Z., Yao, Z., Lin, Z., 2014. Aquatic predicted no-effect concentration for three
792 polycyclic aromatic hydrocarbons and probabilistic ecological risk assessment in Liaodong Bay of the
793 Bohai Sea, China. Environ. Sci. Poll. Res. 21, 148-158.
794 Wheeler, J.R., Leung, K.M.Y., Morritt, D., Sorokin, N., Rogers, H., Toy, R., Holt, M., Whitehouse, P., Crane, M.
795 2002. Freshwater to saltwater toxicity extrapolation using species sensitivity distributions. Environ.
796 Toxicol. Assess. 21, 2459-2467.
797 Yan, W., Chi, J., Wang, Z., Huang, W., Zhang, G., 2009. Spatial and temporal distribution of polycyclic aromatic
798 hydrocarbons (PAHs) in sediments from Daya Bay, South China. Environ. Poll. 157, 1823-1830.