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1 **Delineation of Pb contamination pathways in two Pectinidae: the variegated scallop**

2 ***Chlamys varia* and the king scallop *Pecten maximus***

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28 **Abstract:** Bioaccumulation of Pb was determined in *Chlamys varia* and *Pecten maximus*
29 exposed to ^{210}Pb via seawater, food and sediment. Both scallops readily concentrated
30 dissolved Pb with whole-body 7-d concentration factors of 250 ± 40 and 170 ± 70 ,
31 respectively. In both species, more than 70% of Pb taken up from seawater was strongly
32 retained within tissues (biological half-life > 1.5 month) whereas Pb ingested with
33 phytoplankton was poorly assimilated ($< 20\%$). As *P. maximus* lives buried in the sediment,
34 this exposure pathway was assessed and showed low bioaccumulation efficiency for
35 sediment-bound Pb (transfer factor < 0.015). Despite the poor transfer efficiency of Pb from
36 food and sediment, the use of a global bioaccumulation model indicated that the particulate
37 pathway (food and/or sediment) constituted the major bioaccumulation route of Pb in both
38 scallops. Whatever the exposure pathway, the digestive gland and kidneys always played a
39 major role in Pb accumulation. In scallop tissues, Pb was predominantly associated with the
40 insoluble subcellular fraction, suggesting a low bioavailability of Pb for scallop consumers.

41
42 **Keywords:** Metal; Uptake kinetics; Retention; Bioavailability; Mollusc

43

44 **1. Introduction**

45

46 Lead (Pb) is widely reported as a contaminant of concern in the marine environment (Laws,
47 2000; Clark, 2001). Marine contamination by this non-essential metal is mainly due to
48 releases from anthropogenic activities and the atmosphere constitutes the principal transport
49 vector towards the Oceans (Laws, 2000). Pb is bioaccumulated by living organisms and more
50 particularly by marine invertebrates (e.g., Temara et al., 1998; Neff, 2002).

51 Among filter-feeders, scallops have demonstrated their ability to concentrate elevated levels
52 of various metals in their tissues (e.g., Bryan, 1973; Bustamante and Miramand 2005a,b;
53 Metian et al., 2007, 2008a,b), even in areas far from anthropogenic sources such as the
54 Antarctic Ocean (e.g., Berkman and Nigro, 1992). In particular, scallops were reported to
55 display higher bioaccumulation capacity for Pb than other filter-feeders such as oysters and
56 mussels occurring in the same areas (Brooks and Rumsby, 1965; Segar et al., 1971).
57 Moreover, Pb concentrations in *Chlamys varia* tissues were shown to reflect the
58 contamination level of their environment (Bustamante and Miramand, 2005a). However, little
59 is known about the metabolism of Pb in pectinids and available information is mainly
60 restricted to baseline Pb concentrations in tissues and organs of scallops from different
61 geographic areas, viz. the Antarctic Ocean (Berkman and Nigro, 1992), the Greenland waters
62 (Johansen et al., 2000) and the coastal waters of France, United Kingdom and Canada (Bryan,
63 1973; Ray et al., 1984; Bustamante and Miramand, 2005b).

64 Therefore, the aim of the present work was to investigate the pathway-specific
65 bioaccumulation of Pb in pectinids (viz., water, food and/or sediment pathways). Two
66 pectinid species were considered (the variegated scallop *Chlamys varia* and the king scallop
67 *Pecten maximus*) and bioaccumulation was investigated using highly sensitive radiotracer
68 techniques (^{210}Pb) (Warnau and Bustamante, 2007). The bioaccumulation of ^{210}Pb was

69 studied at the levels of 1) whole individual, 2) tissues and organs and 3) cells of the different
70 tissues, in order to determine the biokinetic parameters of Pb uptake and retention, and the
71 body and subcellular distribution of Pb. In addition, the relative contribution of each exposure
72 pathway to the global Pb bioaccumulation was assessed for both species using a dynamic
73 bioaccumulation model (Landrum et al., 1992; Thomann et al., 1995; Wang et al., 1996;
74 Metian et al., 2008b).

75

76 **2. Materials and methods**

77

78 *2.1. Sampling*

79 During spring 2005, one hundred variegated scallops, *Chlamys varia*, and seventy king
80 scallops, *Pecten maximus*, were collected on the French Atlantic coast (Pertuis Breton,
81 Charente-Maritime) by SCUBA diving. They were cautiously transported to IAEA-MEL
82 premises in Monaco and were acclimated to laboratory conditions (constantly aerated open
83 circuit aquarium; flux: 50 l h⁻¹, salinity: 36 p.s.u.; temperature: 17 ± 0.5 °C; pH: 8.0 ± 0.1;
84 light/dark cycle: 12 h/12 h) for 6 weeks prior to experiments. During this period, scallops
85 were fed daily an algal mixed diet (5 10⁴ cell ml⁻¹) composed of *Isochrysis galbana* and
86 *Skeletonema costatum*.

87

88 *2.2. Radiotracer and counting*

89 The ²¹⁰Pb radiotracer of high specific activity was purchased from CERCA LEA, France
90 (²¹⁰Pb as Pb(NO₃)₂ in 1M HNO₃, T_{1/2} = 22.3 years). The tracer was γ-counted (46.5 KeV;
91 4.05% intensity) using two well-type NaI detectors connected to a multi-channel analyzer and
92 a computer equipped with a spectra analysis software (Interwinner[®] 6). The absolute detection
93 efficiency was 3.4 ± 0.2% under the conditions selected. The radioactivity was determined by

94 comparison with standards of known activity and of appropriate geometries. Measurements
95 were corrected for counting efficiency and physical radioactive decay. Activities of ^{210}Pb in
96 the samples were much over the detection limits but the counting time was adjusted to obtain
97 a propagated counting error less than 5% (Rodriguez y Baena et al., 2006a).

98

99 2.3. Seawater exposure

100 Fifteen *C. varia* and 15 *P. maximus* (average weight \pm SD: 29 ± 6 g and 73 ± 6 g,
101 respectively) were placed in a 70-l glass aquarium (0.45- μm filtered seawater, constantly
102 aerated closed circuit aquarium; salinity: 36 p.s.u.; temperature: 17 ± 0.5 °C; pH: 8.0 ± 0.1 ;
103 light/dark cycle: 12 h/12 h) and exposed for 7 d to ^{210}Pb dissolved in seawater (0.5 kBq l^{-1}).
104 No change in pH was detectable after the tracer addition. ^{210}Pb spike and seawater were
105 renewed twice a day the first two days and then daily in order to keep radioactivity constant in
106 seawater. During each renewal of seawater and spike, the scallops were fed briefly (30 min) *S.*
107 *costatum* and *I. galbana* ($5 \cdot 10^4$ cells ml^{-1}) in clean seawater. ^{210}Pb activity in seawater was
108 checked before and after each spike renewal, yielding a time-integrated activity of ^{210}Pb (0.33
109 $\pm 0.14 \text{ kBq l}^{-1}$) (Rodriguez y Baena et al., 2006b).

110 Nine tag-identified scallops of each species were removed at different time intervals to be
111 radioanalyzed alive and then returned to their aquarium. At the end of the 7-d exposure
112 period, 5 individuals of *C. varia* and *P. maximus* (not belonging to the tag-identified batches)
113 were sacrificed and dissected. Shell, digestive gland, kidneys, gills, gonad, mantle, intestine,
114 adductor muscle and the remaining soft tissues were separated and radioanalyzed to assess
115 ^{210}Pb body distribution. The 10 remaining scallops were placed in non-contaminating
116 conditions (constantly aerated open circuit; flux: 50 l h^{-1} ; salinity: 36 p.s.u.; temperature: $17 \pm$
117 0.5 °C; pH: 8.0 ± 0.1 ; light/dark cycle: 12 h/12 h; daily feeding on *S. costatum* and *I. galbana*,
118 $5 \cdot 10^4$ cells ml^{-1}) for 36 d (*P. maximus*) or 91 d (*C. varia*). The 9 tag-identified individuals of

119 each species were regularly radioanalyzed to determine the depuration kinetics of ^{210}Pb . At
120 the end of the depuration period, 4 individuals of *C. varia* and *P. maximus* were collected and
121 dissected into several body compartments as previously described.

122 When dissections were carried out, subcellular distribution of ^{210}Pb between soluble and
123 insoluble fractions was also investigated, according to the method described in Bustamante
124 and Miramand (2005b). Briefly, a part of digestive gland, gills and adductor muscle was
125 homogenized individually with a mortar and pestle on ice with 10 ml of 0.02 M Tris-HCl
126 buffer, 0.25 M sucrose, 1 mM phenylmethylsulfonylfluoride (PMSF, as protease inhibitor), at
127 pH 8.6 and in presence of dithiothreitol as antioxidant agent. The homogenates were
128 centrifuged at 80,000 G for 1 h at 5°C in a Sorvall RC28S ultracentrifuge to separate particle-
129 free supernatant (cytosol; soluble fraction) from the pellet (insoluble fraction). Homogenate
130 aliquots, cytosols, and pellets were then γ -counted.

131

132 2.4. Food exposure

133 The Haptophyceae *Isochrysis galbana* was used to study ^{210}Pb dietary transfer to scallops.
134 Phytoplankton cells were exposed to 1.5 kBq l⁻¹ of ^{210}Pb during their exponential growth
135 phase (7 d). The cells were then filtrated (1- μm mesh size Osmonic filters) when
136 haemocytometer counting determined a concentration of 10⁷ cell ml⁻¹. Phytoplankton cells
137 were then resuspended in a 70-l aquarium where 6 individuals of both *C. varia* and *P.*
138 *maximus* (average weight \pm SD: 22 \pm 4 g and 91 \pm 4 g, respectively) had been placed
139 previously for one week. The ^{210}Pb content of the radiolabelled *I. galbana* and exposure
140 medium was γ -counted before and after the filtration. Scallops were allowed feeding for 2 h
141 on the radiolabelled *I. galbana* (average concentration of 5 10⁴ cell ml⁻¹ among this time).
142 After the feeding period, all scallops were γ -counted and flowing seawater conditions (50 l h⁻¹
143 l) were restored in the aquarium, with daily feeding on *S. costatum* and *I. galbana* (5 10⁴ cells

144 ml⁻¹). Over the next 8 d, all individuals were regularly radioanalyzed to follow the whole-
145 body depuration kinetics of ²¹⁰Pb. All individuals were collected after 8 d of depuration and
146 sacrificed to determine the body distribution of the remaining ²¹⁰Pb (shell, digestive gland,
147 kidneys, gills, gonad, mantle, intestine, adductor muscle and the remaining soft tissues).

148

149 2.5. Sediment exposure

150 Sediment was collected in Wimereux (North-Atlantic coast of France). Sediment grain size
151 distribution was determined using a Mastersizer micro (Malvern) and the dry/wet wt ratio was
152 calculated after freeze-drying using a LABCONCO Freezone18. Aerated sediment (8 kg) was
153 dispatched in two 5-l plastic bottles, spiked with ²¹⁰Pb (275 kBq in each bottle) and then
154 constantly agitated for 6 d according to the method described in Danis et al. (2003) and
155 adapted by Metian et al. (2007). After this period, part of the sediment was laid in a 20-l
156 aquarium to form a continuous, 4-cm height layer. Weakly bound ²¹⁰Pb was allowed to leach
157 overnight under flowing seawater (50 l h⁻¹). Twelve *P. maximus* (average weight ± SD: 143 ±
158 3 g) were then placed in the aquarium for 7 d (open circuit, seawater flux: 50 l h⁻¹), during
159 which 8 tag-identified individuals were regularly radioanalyzed alive. During the exposure
160 period, sediment samples were regularly collected and γ-counted to verify that the ²¹⁰Pb
161 activity in sediment remained constant. At the end of the uptake period, 4 *P. maximus* were
162 collected, weighed and dissected. Separated tissues and organs (shell, digestive gland,
163 kidneys, gills, gonad, adductor muscle and remaining soft tissues) were radioanalyzed to
164 determine the body distribution of ²¹⁰Pb. The remaining individuals were transferred for 16 d
165 to a new aquarium (non contaminating conditions; open circuit, seawater flux: 50 l h⁻¹) to
166 follow the depuration kinetics of ²¹⁰Pb. At the end of the depuration period, 4 scallops were
167 collected and dissected as described above to determine the ²¹⁰Pb body distribution. The 4

168 remaining organisms were dissected grossly (separation of the whole soft parts from the
169 shells) and γ -counted.

170

171 2.6. Data analysis

172 Uptake and depuration kinetics for all exposure pathways were fitted using kinetic models and
173 statistical methods as described by Warnau et al. (1996a,b) and Metian et al. (2007). The level
174 of significance for statistics and modelling was always set at $\alpha = 0.05$.

175

176 3. Results

177

178 3.1. Seawater exposure

179 Uptake of ^{210}Pb in whole-body *C. varia* and *P. maximus* followed linear kinetics ($R^2 = 0.81$
180 and 0.93, respectively; Fig. 1A). ^{210}Pb was accumulated more rapidly in *C. varia* than in *P.*
181 *maximus* ($k_u = 36$ vs. 24 d^{-1} , respectively; Fig. 1A). Consequently, the whole-body
182 concentration factor (CF) of ^{210}Pb measured at the end of the uptake period ($\text{CF}_{7\text{d}}$) was higher
183 in *C. varia* (250 ± 40) than in *P. maximus* (170 ± 70). The $\text{CF}_{7\text{d}}$ calculated for the different
184 organs and tissues (Table 1) clearly show that ^{210}Pb was concentrated differently by the body
185 compartments of each species. For *C. varia*, the highest $\text{CF}_{7\text{d}}$ were found in the digestive
186 gland and kidneys reached ($1,490 \pm 240$ and $1,780 \pm 770$, respectively). These $\text{CF}_{7\text{d}}$ were not
187 significantly different from each other ($p > 0.05$) but were higher than those of the other body
188 compartments. In the case of *P. maximus*, the kidneys displayed the highest $\text{CF}_{7\text{d}}$ ($1,890 \pm$
189 $1,050$).

190 In terms of body load, ^{210}Pb distribution among soft tissues also differed between the two
191 species (Table 1). In *C. varia*, ^{210}Pb was mainly distributed in the digestive gland (50%) and
192 only 15% of the total ^{210}Pb was found in the kidneys. In contrast, ^{210}Pb was distributed

193 similarly ($p_{\text{Tukey after arcsin transformation}} > 0.05$) among the different organs of *P. maximus* at the
194 end of the exposure period: digestive gland, kidneys, gills, gonad, mantle, intestine, adductor
195 muscle all accounted for ~12 to ~30% of the total ^{210}Pb body load.

196 After the 7-d exposure experiment, non-contaminating conditions were restored and
197 depuration kinetics were followed for 91 and 36 d for *C. varia* and *P. maximus*, respectively.
198 The whole-body depuration kinetics of ^{210}Pb were best described by a double exponential
199 model ($R^2 \geq 0.85$; Fig. 1) displaying similar ^{210}Pb depuration kinetic parameters for both
200 species. The major part of ^{210}Pb ($A_{01} > 70\%$) was efficiently retained ($T_{b/21} > 1.5$ month; Fig.
201 1A).

202 At the end of the depuration period, the ^{210}Pb body distribution differed from that observed at
203 the end of the exposure period (Table 1). This difference was limited in *C. varia*: the digestive
204 gland remained the organ containing the highest ^{210}Pb proportion ($54 \pm 14\%$ of the total body
205 load), and the only slight but significant difference ($p = 0.028$) was found in the gonad (7% at
206 the end of exposure period vs. 4% at the end of the depuration period). In contrast, difference
207 in ^{210}Pb body distribution between exposure and depuration periods was more pronounced in
208 *P. maximus*: the percentage in the kidney increased from $15 \pm 5\%$ at the end of the uptake
209 period to $70 \pm 19\%$ at the end the depuration period. This increase was due to (1) a significant
210 increase in ^{210}Pb activity in the kidneys between the two periods (from 620 ± 350 up to $1,390$
211 $\pm 1,110$ Bq g^{-1} wet wt); and (2) a simultaneous decrease in ^{210}Pb activity in the other tissues
212 (i.e., 76 to 96% according to the body compartment; data not shown).

213 In Figure 2, we show the subcellular distribution of ^{210}Pb in the digestive gland, gills and
214 adductor muscle of both species at the end of the uptake and depuration periods of the
215 seawater experiment. ^{210}Pb was always mainly present in the insoluble fraction of the cells of
216 the considered tissues ($> 50\%$) and increased between the two sampling times (end of uptake
217 and end depuration period), except in the adductor muscle of *C. varia*.

218

219 3.2. Dietary exposure

220 The depuration kinetics of ^{210}Pb ingested with food (the Haptophyceae phytoplankton
221 *Isochrysis galbana*) were best fitted using a double exponential model in both *C. varia* and *P.*
222 *maximus* ($R^2 = 0.94$ and 0.99 , respectively; Fig. 1 B). Both species were characterized by a
223 rather low assimilation efficiency for ^{210}Pb ingested with food ($\text{AE} = 6 - 18\%$). However,
224 once incorporated, ^{210}Pb was very strongly retained in the tissues of both species, with
225 biological half-lives that were virtually infinite (depuration rate constants k_{el} not significantly
226 different from 0; Fig 1B).

227

228 3.3. Sediment exposure

229 Sediment used in our experiment was mainly composed (95.8%) of grains with a diameter
230 ranging from 76 to 302 μm and was characterized by a dry/wet wt ratio of approximately
231 0.80. The ^{210}Pb activity in spiked sediment remained constant all along the exposure period
232 (viz., $47 \pm 3 \text{ Bq g}^{-1}$ wet wt, $n = 21$).

233 The whole-body uptake kinetics of sediment-bound ^{210}Pb in *P. maximus* was best fitted by a
234 first-order saturation model ($R^2 = 0.62$; Fig. 3) that rapidly reached a plateau (estimated $\text{TF}_{ss} =$
235 0.011 ± 0.001).

236 Among the different body compartments of *P. maximus*, the kidneys displayed the highest
237 TF_{7d} which values were characterized by a rather high inter-individual variation ($\text{TF}_{7d \text{ kidneys}} =$
238 0.20 ± 0.20 ; Table 2). The digestive gland showed the second highest TF_{7d} (0.17 ± 0.07) and
239 contained the highest proportion of the total ^{210}Pb body load (i.e. 46%) at the end of the
240 exposure period (Table 2).

241 The ^{210}Pb whole-body depuration kinetics after exposure via the sediment were best described
242 by a bi-exponential model ($R^2 = 0.54$; Fig. 3). A large fraction of the ^{210}Pb taken up ($A_{0s} =$

243 47%) was rapidly lost whereas the remaining ^{210}Pb fraction ($A_{01} = 53\%$) was strongly retained
244 in *P. maximus* tissues, with an estimated $T_{b1/2}$ of 43 d (Fig. 3).

245 When the different tissues were considered separately, the activity levels of ^{210}Pb were near or
246 under the detection limit. Nevertheless, ^{210}Pb activity could be measured accurately in the
247 whole soft tissues and compared to that determined at the end of the uptake period. A
248 significant decrease of ^{210}Pb activity was observed at the end of the depuration period (from
249 0.88 ± 0.19 down to 0.14 ± 0.02 Bq g^{-1} wet wt). After 16 d in non contaminated conditions,
250 only 15% of the initial ^{210}Pb activity remained associated with the whole soft parts (data not
251 shown).

252

253 **4. Discussion**

254 Pectinids are well known to accumulate high levels of metals in their tissues (e.g., Brooks and
255 Rumsby, 1965; Bryan, 1973; Bustamante and Miramand, 2005b; Metian et al., 2007,
256 2008a,b). However, only few works have focused on Pb levels in scallops (see Table 3).
257 These studies have shown that the concentrations of this metal in the whole soft parts ranged
258 between 0.65 and 16 $\mu\text{g g}^{-1}$ dry wt, which are generally higher than those found in other filter-
259 feeders such as oysters and mussels inhabiting the same area (e.g., Brooks and Rumsby, 1965;
260 Segar et al., 1971). It has also been showed that Pb body concentrations in *Chlamys varia*
261 were directly related to the degree of Pb contamination in the surrounding environment
262 (Bustamante and Miramand, 2005b) and that, in a given area, Pb concentrations varied
263 according to the scallop species considered (Bryan, 1973). Therefore it appeared important to
264 investigate Pb uptake and depuration kinetics in scallops exposed via different contamination
265 pathways in order to better understand/characterize the processes driving Pb bioaccumulation
266 in scallops.

267 When exposed to dissolved Pb, both the variegated scallop *C. varia* and the king scallop
268 *Pecten maximus* displayed efficient bioconcentration capacities. *C. varia* bioconcentrated Pb
269 ca. 50% faster than *P. maximus*. After 7 days of exposure, uptake kinetics were still in a linear
270 phase and reached relatively elevated concentration factors (CF) for both species ($CF_{7d} = 250$
271 ± 40 for *C. varia* and 170 ± 70 for *P. maximus*) compared to other organisms such as the
272 mussel *Mytilus galloprovincialis* ($CF_{7d}: 150 \pm 20$; Boudjenoun et al., 2007). The high
273 bioconcentration capacity of the scallops is probably related to their biological characteristics
274 as they have a more efficient filtration system and/or a larger gill surface area than mussels
275 (Beninger and St-Jean, 1997, Ward et al., 1998). Indeed, gills are generally considered as the
276 main entry of dissolved metals in marine filter-feeder organisms (Marigómez et al., 2002).
277 Once taken up from the dissolved phase, most of the Pb ($A_{01} > 70\%$) was strongly retained
278 within the tissues of both species ($T_{b/2l} > 1.5$ month). These observations are in accordance
279 with previous studies that reported elevated absorption efficiencies ($A_{01} \geq 60\%$; Fisher et al.,
280 1996) and retention ($T_{b/2l} \geq 23$ d; Boudjenoun et al., 2007) of dissolved Pb in mussels. Such
281 efficient bioconcentration and retention of Pb in *C. varia* and *P. maximus* suggests that
282 seawater could play an important role in the global Pb bioaccumulation in scallops.
283 When scallops were exposed to Pb via the food pathway, the metal was poorly assimilated by
284 the two species ($AE \leq 18\%$). The major part of the Pb ingested with phytoplankton ($> 80\%$)
285 was lost within less than 2 days. The Pb AE determined in scallops is much lower than in the
286 mussel *Mytilus galloprovincialis* fed the phytoplankton *Thalassiosira pseudonana* ($AE =$
287 50% , Fisher et al., 1996). Metian et al. (2008b) have shown that the AE of Ag ingested with
288 food was influenced substantially by the phytoplankton species used as food. However,
289 according to Fisher et al. (1983; 1987), this should not be the case for Pb as these authors
290 have shown that Pb bioaccumulation is rather non specific and invariant with physiological
291 condition of the phytoplankton cells. Therefore the low Pb AE found for both scallop species

292 would not be due to the particular phytoplanktonic species used as food but rather to some
293 specificities of the scallops (e.g., differences in digestive processes). Interestingly, even
294 though AE ranged between only 6 and 18%, we found out that dietary transfer may play a
295 major role in the global accumulation of Pb in the scallops (see below).

296 In contrast to *C. varia* which is living on rocky substrata, *P. maximus* lives buried into the
297 sediments. When exposed to radiolabelled sediment, uptake of Pb by *P. maximus* was quite
298 poor ($TF_{ss} = 0.011$). However the assimilated fraction of the sediment-bound metal was
299 retained with a similar biological half-life ($T_{b/2l} = 43$ d) than when taken up via the food (52
300 d) or seawater (47 d). The granulometry of the sediment used in our study (mostly composed
301 of grain of size from 76 to 302 μm) allows assuming that no ingestion of particles did occur,
302 as the gills, the peribuccal system or the stomach were reported to be efficient in selecting the
303 size of the particles that are actually ingested by the scallops (e.g., Beninger et al., 1991;
304 Brillant et al., 2000). If this hypothesis was to be true, this would suggest that sediment-
305 related contamination would have occurred through Pb bioaccumulation from porewater.
306 However, two facts are supporting the counter hypothesis that particles could actually have
307 been ingested. First, some studies have reported the occurrence of large particles (up to 950
308 μm) in the stomachs of scallops (Mikulich and Tsikhon-Lukanina, 1981; Shumway et al.,
309 1987) and, second, the digestive gland contained 45% of the whole Pb load when scallops
310 were exposed to contaminated sediment. As a porewater exposure pathway should be
311 equivalent to a seawater exposure, one would have expected to find a rather more
312 homogeneous distribution among digestive gland, gills and kidneys, as was observed at the
313 end of the exposure period in our seawater experiment (see Table 1). Hence, although
314 requiring confirmation, ingestion of sediment grains could actually represent a source of Pb
315 exposure for *P. maximus*.

316 In order to assess the contribution of the different routes of exposure in global Pb
317 bioaccumulation, the biokinetic parameters determined from our experiments were used along
318 with parameters taken from the literature (distribution coefficient in sediment, concentration
319 factor in phytoplankton, scallops ingestion rate; IAEA, 2004; Metian et al., 2008b) to feed the
320 global bioaccumulation model originally described Thomann (1981). This model was further
321 developed and commonly used by several authors (e.g., Thomann et al., 1995; Wang et al.,
322 1996; Reinfelder et al., 1998; Metian et al., 2008b). Assuming that Pb is distributed in the
323 different compartments of the marine environment (seawater, phytoplankton and sediment)
324 according to the distribution coefficient in sediment ($K_{d,Pb} = 10^5$; ocean margin sediments)
325 and the range of concentration factors in phytoplankton ($CF_{phyto,Pb} = 10^4 - 10^5$) available from
326 the literature (IAEA, 2004), the model computations indicated that the relative contributions
327 of each exposure route were substantially influenced by the $CF_{phyto,Pb}$ values. In particular, in
328 *C. varia* for which only seawater and food exposure pathways were considered, the dietary
329 pathway was found to contribute for 86% of the global Pb bioaccumulation when considering
330 the recommended $CF_{phyto,Pb}$ value (10^5 ; IAEA, 2004), whereas the dissolved pathway was the
331 dominant pathway (62%) when considering the lowest $CF_{phyto,Pb}$ value (10^4). Lower food
332 contribution when $CF_{phyto,Pb}$ increase could be related to a stronger retention of Pb on the
333 phytoplanktonic cell, thus reducing the bioavailability of the metal. For the sediment-dweller
334 *P. maximus*, three different exposure pathways were taken into account (seawater, food and
335 sediment). Model computations showed that the particulate pathway (food and sediment) was
336 always the major source (95 - 97%) of Pb bioaccumulation for *P. maximus*. However,
337 depending on the $CF_{phyto,Pb}$ value, the contributions of food vs sediment pathways shifted from
338 5 vs 90% (for $CF_{phyto,Pb} = 10^4$) to 33 vs 64% (for $CF_{phyto,Pb} = 10^5$). As available values for
339 $CF_{phyto,Pb}$ and $K_{d,Pb}$ parameters are essentially average, non-specific values published by the
340 IAEA (2004), it is recommended to further refine these two parameters for environmental

341 conditions close to those prevailing in the scallop habitats before further interpreting and
342 extrapolating the results presented here.

343 The kidneys and the digestive gland appeared to play an important role in the uptake and
344 retention of Pb in the tissues of both scallop species. However, these organs were involved
345 differently according to the exposure pathway and the species considered. In scallops
346 collected in the field, the highest Pb concentrations are always reported in the kidneys, when
347 those organs were analysed separately (Table 3). In addition, the kidneys contained more than
348 50% of the total Pb body burden in *Aequipecten opercularis*, *C. varia* and *P. maximus* (e.g.,
349 Bryan, 1973; Bustamante and Miramand, 2005b). As concentrations recorded from the field
350 are supposed to represent a steady-state situation, kidneys may be reasonably considered as
351 organs playing a key role in detoxification and storage of Pb. This is also supported by our
352 experimental observations in *P. maximus*. Indeed, differences in body distribution observed
353 between the end of the seawater uptake and depuration periods suggest that translocation of
354 Pb from different tissues and organs towards the kidneys and/or its storage in these organs
355 occurred in *P. maximus* (which contained 70% of total whole soft parts Pb body burden one
356 month after the exposure period). In contrast, if any translocations of Pb in *C. varia*, target
357 organ(s) of those processes would be different than in *P. maximus* as Pb remained mainly
358 present in the digestive gland all along both exposure and depuration periods. It is known that
359 bivalves take up dissolved Pb mainly through the gills and that the metal is then translocated
360 towards the different organs and tissues via blood granulocytes and high molecular weight
361 plasma ligands (Marigómez et al., 2002). Interspecific variations in Pb concentrations
362 observed in scallops in the field (see Table 3) could thus be related to interspecific differences
363 in translocation mechanisms. In this respect, waterborne Pb would be mainly translocated
364 towards the kidneys (followed by storage) in *P. maximus* whereas it would be mainly
365 translocated towards the digestive gland in *C. varia*. It is noteworthy that in field studies,

366 renal Pb levels are always much higher in *P. maximus* than in *C. varia*, even for organisms
367 displaying similar whole soft parts Pb concentrations (see Table 3). Nevertheless, it would be
368 worth developing studies on translocation routes to and storage mechanisms in scallop
369 digestive gland and kidneys as it would help greatly in better understanding how Pb as well as
370 other metals are handled and detoxified by these organisms. The results of the subcellular
371 fractioning, in both *C. varia* and *P. maximus*, indicated that Pb was mainly associated with the
372 insoluble cellular fraction, which is consistent with similar fractioning carried out with *C.*
373 *varia* from the field (Bustamante and Miramand, 2005b). As this insoluble subcellular
374 fraction includes the haemocyte lysosomes which are well-known to play an important role in
375 metal detoxification and storage in bivalves (e.g., Ballan-Dufrançais et al., 1985), further
376 studies should not neglect the role of these free-circulating cells in addressing Pb
377 translocation targetting.

378

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387

388 **References**

389 Ballan-Dufrançais, C., Jeantet, A.Y., Feghali, C., Halpern, S. Physiological features of heavy
390 metal storage in bivalve digestive cells and amoebocytes: EPMA and factor analysis
391 of correspondences. *Biol Cell* 1985;53:283-292.

392 Beninger, P.G., Le Pennec, M. Functional anatomy of scallops. In: Shumway, S.E. (ed)
393 Scallops: biology, ecology and aquaculture. Elsevier, pp. 133-224; 1991

394 Beninger, P.G., St-Jean, S.D. Particle-processing on the labial palps of *Mytilus edulis* and
395 *Placopecten magellanicus* (Mollusca: Bivalvia). *Mar Ecol Prog Ser* 1997;147:117-
396 127.

397 Berkman, P.A., Nigro, M. Trace metal concentrations in scallops around Antarctica:
398 Extending the Mussel Watch Programme to the Southern Ocean. *Mar Pollut Bull*
399 1992;24:322-323.

400 Boudjenoun, R., Teyssié, J.L., Rodriguez y Baena, A., Fowler, S.W., Warnau, M. Lead
401 bioaccumulation in the Mediterranean mussel *Mytilus galloprovincialis*. *Rapp Com*
402 *Intern Mer Méd* 2007;38:237.

403 Brooks, R.R., Rumsby, M.G. The biogeochemistry of trace element uptake by some New
404 Zealand bivalves. *Limnol Oceanogr* 1965;10:521-527.

405 Bryan, G.W. The occurrence and seasonal variation of trace metals in the scallops *Pecten*
406 *maximus* (L.) and *Chlamys opercularis* (L.). *J Mar Biol Assoc UK* 1973;53:145-166.

407 Brilliant, M.G.S., MacDonald, B.A. Postingestive selection in the sea scallop, *Placopecten*
408 *magellanicus* (Gmelin): the role of particle size and density. *J Exp Mar Biol Ecol*
409 2000;253:211-227.

410 Bustamante, P., Miramand, P. Evaluation of the variegated scallop *Chlamys varia* as a
411 biomonitor of temporal trends of Cd, Cu, and Zn in the field. *Environ Pollut*
412 2005a;138:109-120.

413 Bustamante, P., Miramand, P. Subcellular and body distributions of 17 trace elements in the
414 variegated scallop *Chlamys varia* from the French coast of the Bay of Biscay. *Sci Tot*
415 *Environ* 2005b;337:59-73.

416 Clark, R.B. *Marine Pollution*. Fifth edition. Oxford University Press, 184 p; 2001

417 Cossa, D., Auger, D., Averty, B., Lucon, M., Masselin, P., Noel, J., Sanjuan, J. Niveaux de
418 concentration en métaux, métalloïdes et composés organochlorés dans les produits de
419 la pêche côtière française, IFREMER, Plouzane, France, 24 p ; 1990

420 Danis, B., Cotret, O., Teyssié, J.L., Fowler, S.W., Bustamante, P., Warnau M. Delineation of
421 PCB uptake pathways in a benthic sea star using a radiolabelled congener. *Mar Ecol*
422 *Prog Ser* 2003;253:155-163.

423 Fang, Z.Q., Cheung, R.Y., Wong, M.H. Heavy metal concentrations in edible bivalves and
424 gastropods available in major markets of the Pearl River Delta. *J Environ Sci (China)*
425 2001;13:210-217.

426 Fisher, N.S., Burns, K.A., Cherry, R.D., Heyraud, M. Accumulation and cellular distribution
427 of ²⁴¹Am, ²¹⁰Po, and ²¹⁰Pb in two marine algae. *Mar Ecol Prog Ser* 1983;11:233-237.

428 Fisher, N.S., Teyssié, J.-L., Fowler, S.W., Wang, W.-X. Accumulation and retention of metals
429 in mussels from food and water: A comparison under field and laboratory conditions.
430 *Environ Sci Technol* 1996;30:3232-3242.

431 Fisher, N.S., Teyssié, J.-L., Krishnaswami, S., Baskaran, M. Accumulation of Th, Pb, U, and
432 Ra in marine phytoplankton and its geochemical significance. *Limnol Oceanogr*
433 1987;32:131-142.

434 Greig, R.A., Wenzloff, D.R., Mackenzie, C.L., Merrill, A.S., Zdanowicz, V.S. Trace metals in
435 sea scallops *Placopecten magellanicus*, from eastern United States. *Bull Environ*
436 *Contam Toxicol* 1978;19:326-333.

437 IAEA. Sediments distribution coefficients and concentration factors for biota in the marine
438 environment. Technical Report Series (TRS) No. 422. International Atomic Energy
439 Agency, Vienna, Austria, 95 p; 2004

440 Johansen, P., Pars, T., Bjerregaard, P. Lead, cadmium and selenium intake by Greenlanders
441 from local marine food. *Sci Tot Environ* 2000;245:187-194.

442 Julshamn, K., Duinker, A., Frantzen, S., Torkildsen, L., Maage, A. Organ distribution and
443 food safety aspects of cadmium and lead in great scallops, *Pecten maximus* L., and
444 horse mussels, *Modiolus modiolus* L., from Norwegian Waters. *Bull Environ Contam*
445 *Toxicol* 2008;80:385-389.

446 Landrum, P.F., Lee, H., Lydy, M.J. Toxicokinetics in aquatic systems: model comparisons
447 and use in hazard assessment. *Environ Toxicol Chem* 1992;11:1709-1725.

448 Laws, .E.A. Metals. In: *Aquatic Pollution: An Introductory Text*, 3rd ed. John Wiley & Sons,
449 New York, 639 p; 2000

450 Marigómez, I., Soto, M., Cajaraville, M.P., Angulo, E., Giamberini, L. Cellular and
451 subcellular distribution of metals in molluscs. *Microscop Res Tech* 2002;56:358-392.

452 Metian, M., Warnau, M., Oberhansli, F., Teyssié, J.L., Bustamante, P. Interspecific
453 comparison of Cd bioaccumulation in European Pectinidae (*Chlamys varia* & *Pecten*
454 *maximus*). *J Exp Mar Biol Ecol* 2007;353:58-67.

455 Metian, M., Bustamante, P., Hédouin, L., Warnau, M. Accumulation of trace elements in the
456 tropical scallop *Comptopallium radula* from coral reefs in New Caledonia. *Environ*
457 *Pollut* 2008a;152:543-552.

458 Metian, M., Bustamante, P., Cosson, R.P., Hédouin, L., Warnau, M. Investigation of Ag in
459 the king scallop *Pecten maximus* using field and laboratory approaches. *J Exp Mar*
460 *Biol Ecol* 2008b;367:53-60.

461 Mikulich, L.V., Tsikhon-Lukamina, A. Food of the scallop. *Oceanol* 1981;21:633-635.

462 Neff, J.M. Bioaccumulation in marine organisms: effect of contaminants from oil well
463 produced water, Elsevier Science, Oxford, UK, 460 p; 2002

464 Nielsen, S.A., Nathan, A. Heavy Metal Levels in New Zealand Molluscs. New Zealand J Mar
465 Freshwat Res 1975;9:467-481.

466 Pesch, G.G., Renolds, B., Rogerson, P. Trace metals in scallops from within and around two
467 ocean disposal sites. Mar Pollut Bull 1977;8:224-228.

468 Ray, S., Woodside, M., Jerome, V.E., Akagi, H. Copper, zinc, cadmium and lead in scallop
469 (*Placopecten magellanicus*) from Georges and Browns Banks. Chemosphere.
470 1984;13:1247-1254.

471 Reinfelder, J.R., Fisher, N.S., Luoma, S.N., Nichols, J.W., Wang, W.X. Trace element trophic
472 transfer in aquatic organisms: A critique of the kinetic model approach. Sci Tot
473 Environ 1998;219:117-135.

474 Rodriguez y Baena, A.M., Miquel, J.C., Masqué, P., Povinec, P., La Rosa, J. A single vs.
475 double spike approach to improve the accuracy of ^{234}Th measurements in small-volume
476 seawater samples. Mar Chem 2006a;100:269-281.

477 Rodriguez y Baena, A.M., Metian, M., Teyssié, J.L., De Broyer, C., Warnau, M.
478 Experimental evidence for ^{234}Th bioaccumulation in three Antarctic crustaceans:
479 potential implications in particle flux studies. Mar Chem 2006b;100:354-365.

480 Segar, D.A., Collins, J.D., Riley, J.P. The distribution of the major and some minor elements
481 in marine animals. Part II. Molluscs. J Mar Biol Assoc UK 1971;51:131-136.

482 Shumway, S.E., Selvin, R., Schick, D.F. Food resources related to habitat in the scallop
483 *Placopecten magellanicus* (Gmelin, 1791): A qualitative study. J Shellfish Res
484 1987;6:89-95.

485 Storelli, M.M., Marcotrigiano, G.O. Consumption of bivalve molluscs in Italy: estimated
486 intake of cadmium and lead. Food Additives Contam 2001;18:303-307.

487 Temara, A., Aboutboul, P., Warnau, M., Jangoux, M., Dubois, P. Uptake and fate of lead in
488 the common asteroid *Asterias rubens* (Echinoderm). Water, Air, Soil Pollut
489 1998;102:201-208.

490 Thomann, R.V. Equilibrium model of fate of microcontaminants in diverse aquatic food
491 chains. Can J Fish Aquatic Sci 1981;38:280-296.

492 Thomann, R.V., Mahony, J.D., Mueller, R. Steady-state model of biota sediment
493 accumulation factor for metals in two marine bivalves. Environ Toxicol Chem
494 1995;14:1989-1998.

495 Topping, G. Heavy metals in shellfish from Scottish waters. Aquaculture 1973;1:379-384.

496 Wang, W.-X., Fisher, N.S., Luoma, S.N. Kinetic determinations of trace element
497 bioaccumulation in the mussel *Mytilus edulis*. Mar Ecol Prog Ser 1996;140:91-113.

498 Ward, J.E., Levinton, J.S., Shumway, S.E., Cucci, T. Particle sorting in bivalves: *in vivo*
499 determination of the pallial organs of selection. Mar Biol 1998;131:283-292.

500 Warnau, M., Bustamante, P. Radiotracer techniques: a unique tool in marine ecotoxicological
501 studies. Environ Bioindicators 2007;2:217-218.

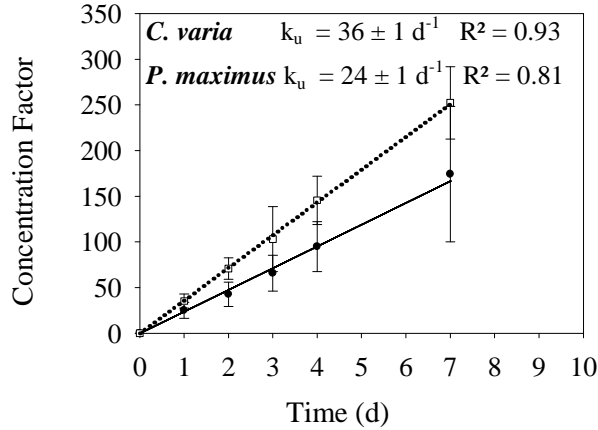
502 Warnau, M., Fowler, S.W., Teysse, J.L. Biokinetics of selected heavy metals and
503 radionuclides in two marine macrophytes: the seagrass *Posidonia oceanica* and the
504 alga *Caulerpa taxifolia*. Mar Environ Res 1996a;41:343-362.

505 Warnau, M., Teysse, J.L., Fowler, S.W. Biokinetics of selected heavy metals and
506 radionuclides in the common Mediterranean echinoid *Paracentrotus lividus*: sea water
507 and food exposures. Mar Ecol Prog Ser 1996b;141:83-94.

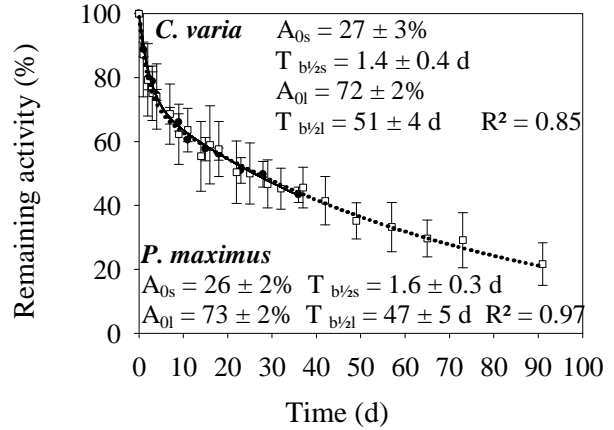
508 Wu, R., Groves, A. Cadmium and lead in tissues of scallops from Port Phillip Bay, Australia.
509 Wat Sci Technol 1995;31:473-483.

A. Seawater experiment

A1. Uptake kinetics



A2. Depuration kinetics



B. Food experiment

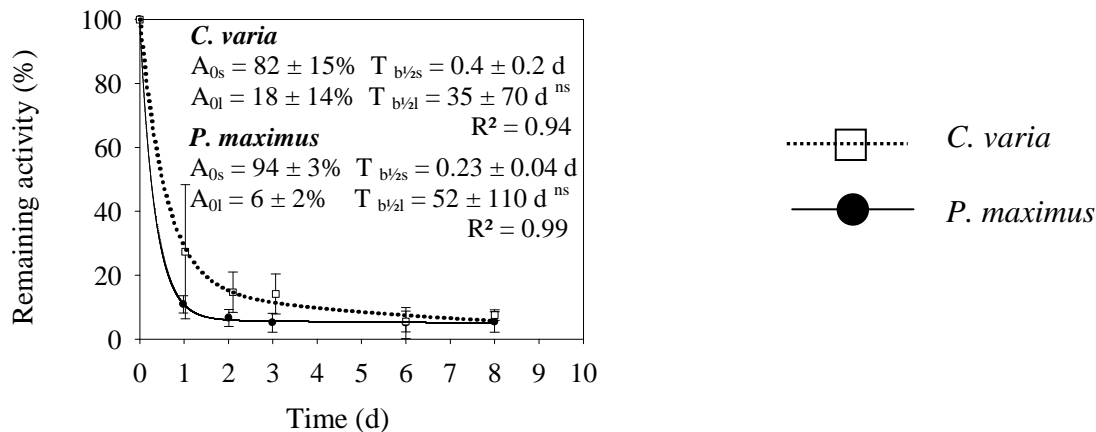
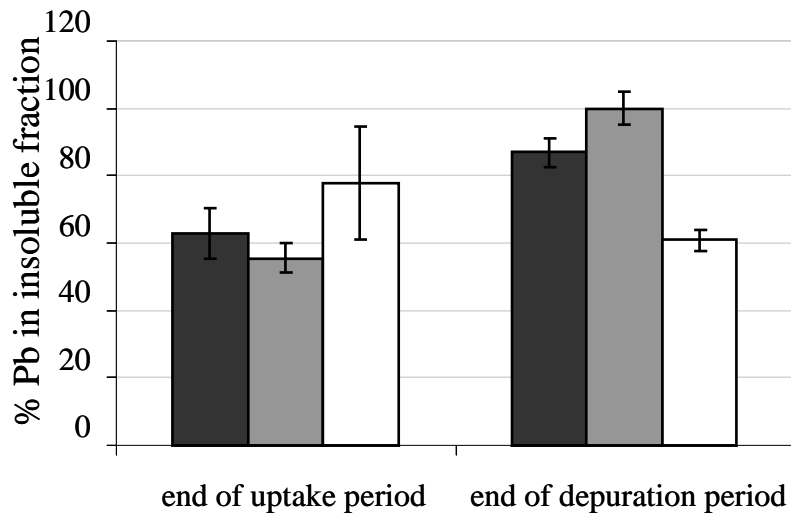


Figure 1. Uptake and depuration biokinetics of ^{210}Pb in the scallops *Chlamys varia* and *Pecten maximus* (A) exposed for 7 d to the radiotracer via seawater (A1. Concentration Factors; mean \pm SD; $n = 9$), then maintained for 36 or 91 d in non contaminated conditions (A2. Remaining activity, %; mean \pm SD; $n = 9$) and (B) fed for 2 h radiolabelled *Isochrysis galbana* (Remaining activity, %; mean \pm SD; $n = 6$).

Estimated parameters. k_u : uptake rate constant (d^{-1}); A_{0s} and A_{0l} : activity (%) lost according to the short- and the long-lived exponential component, respectively; $T_{b/2}$: biological half-life (d). ns: parameter not significantly different ($p > 0.05$) from 0 (k_{el}) or from the infinite ($T_{b/2l}$).

A. *Chlamys varia*



B. *Pecten maximus*

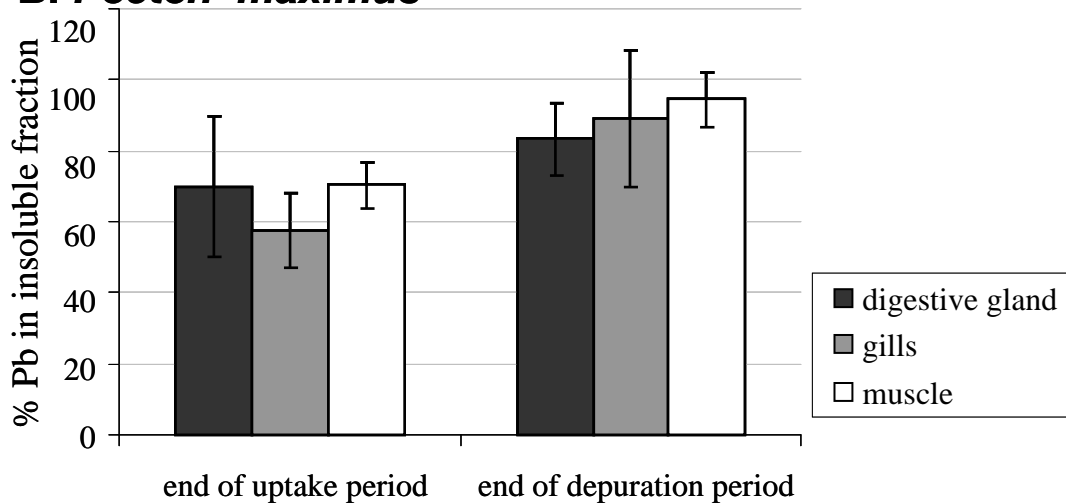
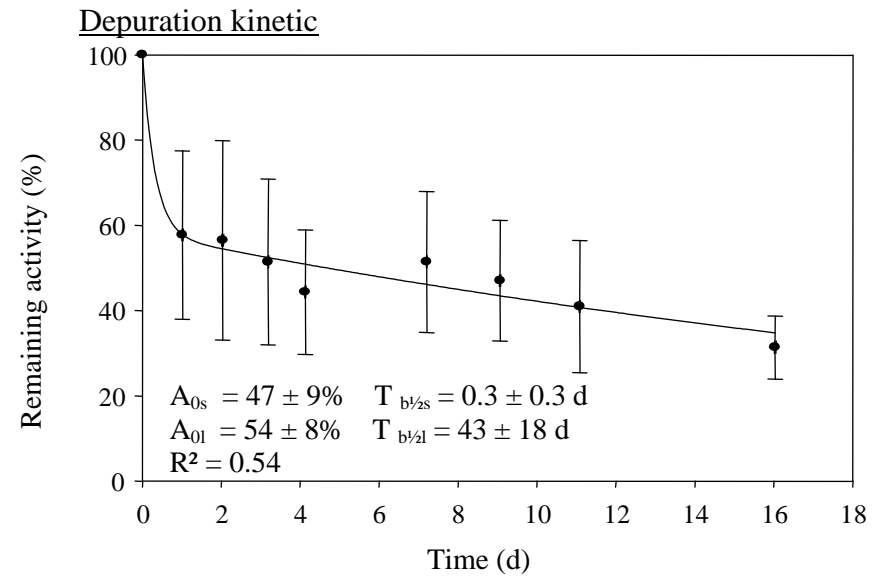
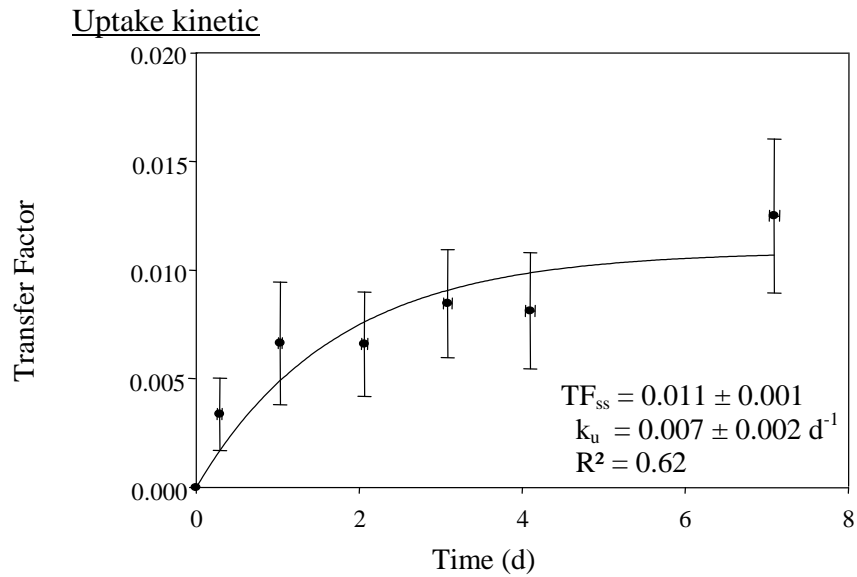


Figure 2. Subcellular distribution (%) of ^{210}Pb in selected tissues (digestive gland, gills and muscle) of *Chlamys varia* (A) and *Pecten maximus* (B) at the end of the 7-d seawater exposure (n = 4 for both species) and of the depuration period (36 d, n = 4 for *P. maximus* and 91 d, n = 4 for *C. varia*) periods.

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2

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4 Figure 3. Uptake and depuration biokinetics of ^{210}Pb in the scallop *Pecten maximus* exposed for 7 d via the sediment (Transfer Factor ; mean \pm
5 SD; n = 8), then maintained for 16 d in non contaminated conditions (Remaining activity, %; mean \pm SD; n = 8).

6 Estimated parameters. TF_{ss} : transfer factor at steady state; other parameters as in Fig. 1.

7

Table 1. Concentration Factors (mean \pm SD) and body distribution among soft parts (mean % \pm SD) of ^{210}Pb in the scallops *Chlamys varia* and *Pecten maximus* exposed to the metal via seawater.

Compartment	Seawater exposure		
	End of 7-d uptake period (n = 5)	End of depuration period (91 d for <i>C. varia</i> , 36 d for <i>P. maximus</i> , n = 4)	
	Concentration Factor	Distribution (%)	Distribution (%)
<i>Chlamys varia</i>			
Digestive gland	1490 \pm 240	50 \pm 6	54 \pm 14
Gills	250 \pm 240	10 \pm 3	4 \pm 3
Kidneys	1780 \pm 550	15 \pm 6	17 \pm 16
Intestine	490 \pm 680	1 \pm 0	3 \pm 4
Gonad	410 \pm 140	7 \pm 2	4 \pm 1
Foot	550 \pm 360	3 \pm 3	3 \pm 4
Mantle	45 \pm 11	5 \pm 1	9 \pm 8
Adductor muscle	56 \pm 26	8 \pm 3	4 \pm 2
Remaining	230 \pm 80	3 \pm 2	2 \pm 1
Whole body	250 \pm 40		
<i>Pecten maximus</i>			
Digestive gland	570 \pm 250	28 \pm 5	12 \pm 8
Gills	110 \pm 30	22 \pm 7	7 \pm 6
Kidneys	1890 \pm 1050	15 \pm 5	70 \pm 19
Intestine	340 \pm 130	2 \pm 1	< 1
Gonad	300 \pm 230	4 \pm 2	5 \pm 4
Foot	150 \pm 60	1 \pm 1	1 \pm 0
Mantle	35 \pm 8	14 \pm 4	1 \pm 1
Adductor muscle	48 \pm 35	12 \pm 5	4 \pm 2
Remaining	170 \pm 40	4 \pm 2	1 \pm 0
Whole body	170 \pm 70		

Table 2. Transfer Factors (mean \pm SD, n = 4) and body distribution (mean % \pm SD, n = 4) of ^{210}Pb in *Pecten maximus* after 7 d of exposure via the sediment.

Compartment	Transfer Factor	Distribution (%)
Digestive gland	0.17 \pm 0.07	46 \pm 4
Gills	0.008 \pm 0.006	5 \pm 5
Kidneys	0.20 \pm 0.20	15 \pm 13
Gonad	0.05 \pm 0.06	7 \pm 7
Adductor muscle	0.003 \pm 0.003	8 \pm 6
Remaining tissues	0.009 \pm 0.005	18 \pm 5
Whole soft parts	0.017 \pm 0.004	100

Table 3. Concentration of Pb ($\mu\text{g g}^{-1}$ dry wt) in different scallop species from various locations.

Scallop species	Location	Whole soft parts concentration ($\mu\text{g g}^{-1}$ dry wt)	Organ displaying highest concentration ($\mu\text{g g}^{-1}$ dry wt)	References
<i>Adamussium colbecki</i>	Antarctica (Explorers cove)	2.6 \pm 0.4	-	Berkman and Nigro, 1992
<i>A. colbecki</i>	Antarctica (Swoya station)	from 1.0 to 3.2	-	Berkman and Nigro, 1992
<i>Aequipecten opercularis</i>	U.K. (English channel)	12.0 \pm 5.2	kidneys (827 \pm 236)	Bryan, 1973
<i>Chlamys islandica</i>	Greenland	0.65 \pm 0.12 ^a	-	Johansen et al., 2000
<i>C. nobilis</i>	Hong Kong markets	2.05 \pm 1.35 ^a	-	Fang et al., 2001
<i>C. nobilis</i>	Guangdong markets	1.0 \pm 0.2 ^a	-	Fang et al., 2001
<i>C. varia</i>	W France (Ré island)	1.91 \pm 0.07	kidneys (28.4 \pm 7.0)	Bustamante and Miramand, 2005b
<i>C. varia</i>	W France (La Rochelle)	2.66 \pm 0.21	kidneys (41.6 \pm 3.7)	Bustamante and Miramand, 2005b
<i>Pecten alba</i>	Australia (Port Phillip bay)	14.06 \pm 0.72*	gonad (18.8 \pm 3.9) **	Wu and Groves, 1995
<i>P. jacobaeus</i>	Italy (Middle Adriatic Sea)	0.75 \pm 0.30 ^a	-	Storelli and Marcotrigiano, 2001
<i>P. novae-zelandiae</i>	New Zealand (Tasman bay)	16	kidneys (137)	Brooks and Rumsby, 1965
<i>P. novae-zelandiae</i>	New Zealand	6.70 ^a	stomach (7 ^a)	Nielsen and Nathan, 1975
<i>P. maximus</i>	NW France (W. English channel)	-	0.57 \pm 0.14 ^{†††}	Cossa et al., 1990
<i>P. maximus</i>	NW France (E. English channel)	-	0.69 \pm 0.38 ^{†††}	Cossa et al., 1990
<i>P. maximus</i>	U.K. (English channel)	2.0 \pm 0.2	kidneys (159 \pm 38)	Bryan, 1973
<i>P. maximus</i>	Scotland (different locations)	from < 0.5 to 5.0 ^a	gonad (from < 1.0 to 8.5 ^a) **	Topping, 1973
<i>P. maximus</i>	Irish Sea	8.3	gonad (31) **	Segar et al., 1971
<i>P. maximus</i>	Norway (Frøya)	0.6 \pm 0.1 ^a	kidneys (41 \pm 13 ^a)	Julshamn et al., 2008
<i>Placopecten magellanicus</i>	Canada (George & Browns Banks)	2.0 [†]	gills (2.1) ^{††}	Ray et al., 1984
<i>P. magellanicus</i>	U.S.A. (mid-Atlantic coast)	3.59	-	Pesch et al., 1977
<i>P. magellanicus</i>	U.S.A. (Atlantic coast)	-	gonad (< 19.5 ^a) **	Greig et al., 1978

^a Data on a wet wt basis were converted on a dry wt basis, using a conversion factor of 5 (Greig et al., 1978).

* whole soft parts without gonad and adductor muscle.

** only adductor muscle and gonad were analyzed separately.

† whole soft parts without adductor muscle, mantle and gills.

†† only adductor muscle, mantle, gills and viscera were analyzed separately.

††† only adductor muscle were analyzed separately.