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1 **Interspecific comparison of Cd bioaccumulation in European Pectinidae (*Chlamys***
2 ***varia* and *Pecten maximus*)**

3

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26 ABSTRACT

27 The uptake and loss kinetics of Cd were determined in two species of scallops from the European coasts,
28 the variegated scallop *Chlamys varia* and the king scallop *Pecten maximus*, following exposures via
29 seawater, phytoplankton and sediment using highly sensitive radiotracer techniques (^{109}Cd). Results
30 indicate that, for seawater and dietary pathways, *C. varia* displays higher bioaccumulation capacities in
31 terms of uptake rate from water and fraction absorbed from ingested food (assimilation efficiency) than
32 *Pecten maximus*. Regarding sediment exposure, *P. maximus* displayed low steady-state Cd transfer factor
33 ($\text{TF}_{\text{SS}} < 1$); however, once incorporated, a very large part of Cd transferred from sediment (92 %) was
34 strongly retained within *P. maximus* tissues.

35 Both species showed a high retention capacity for Cd (biological half-life, $T_{b/2} > 4$ months), suggesting
36 efficient mechanisms of detoxification and storage in both species. The digestive gland was found to be
37 the main storage organ of Cd in the two scallops regardless of the exposure pathway. However, Cd was
38 stored differently within this organ according to the species considered: 40 % of the total Cd was found in
39 the soluble cellular fraction in *C. varia* whereas this soluble fraction reached 80 % for *P. maximus*. This
40 suggests that the two species displayed different Cd detoxification/storage mechanisms.

41 Finally, the present study has determined the relative contribution of the different exposure pathways to
42 global Cd bioaccumulation for the two scallop species. Results clearly show that for both species, food
43 constitutes the major accumulation pathway, contributing for > 99 % and 84 % of the global Cd
44 bioaccumulation in *C. varia* and *P. maximus*, respectively. This work confirms the previous assumption,
45 derived from a bibliographic overview, that dietary pathway plays a prevalent role in metal
46 bioaccumulation in Pectinidae.

47

48 KEYWORDS: Cadmium, Metal, Kinetics, Subcellular Distribution, Scallops, Bivalves

49 INTRODUCTION

50 Bivalves usually concentrate efficiently Cd from the surrounded environment (e.g. Eisler 1985). Among
51 them, Pectinidae can display very high concentrations of this non essential metal that is considered as one
52 of the most toxic ones. High levels of Cd in scallop tissues have been reported even for species from
53 pristine and low-contaminated areas such as the Antarctic Ocean or the sub-polar Atlantic Ocean (Mauri
54 et al. 1990, Viarengo et al. 1993, Bustamante & Miramand 2004), suggesting that scallops have evolved a
55 natural capacity to accumulate, detoxify and store this metal in their tissues. Investigations carried out in
56 the field and in the laboratory have revealed the involvement of very efficient detoxification mechanisms.
57 Indeed, the binding of Cd to high-affinity cytosolic proteins, lysosomes, and mineral concretions is well
58 known to result in efficient Cd sequestration in Pectinidae (Carmichael & Fowler 1981, Ballan-
59 Dufrançais et al. 1985, Stone et al. 1986).

60 Even though field investigations have shown that Cd levels are influenced by various factors such as
61 geographical origin, season, size and sexual maturity (Bryan 1973, Evtushenko et al. 1990, Mauri et al.
62 1990, Bustamante & Miramand 2004, 2005a), very little is known on the dynamics of Cd
63 bioaccumulation and retention in this family. To the best of our knowledge, no study has described the Cd
64 accumulation in Pectinidae exposed via different pathways and its depuration using environmentally
65 realistic metal levels. For example the earlier study by Eisler et al. (1972) exposed *Aquipecten irradians*
66 to 10 ppm Cd, a concentration with toxic consequences (Gould et al. 1988) and therefore unlikely to
67 produce a typical accumulation pattern for Cd. In natural conditions, scallops are exposed to metal
68 through seawater and food pathways, sediment potentially contributing to either or both. It is therefore
69 necessary to investigate separately these different exposure pathways to understand their relative
70 contribution in the global accumulation of the metal (Fowler 1982).

71 Seawater has been often considered as the main source of metal intake for marine organisms (e.g.,
72 Janssen & Scholz 1979, Borchardt 1983, Riisgard et al. 1987); however the role of the particulate phase,
73 mainly food, is now recognized to be of primary importance for a large range of taxa (e.g., Warnau et al.

74 1996, 1999, Reinfelder et al. 1998, Wang & Fisher 1999). In the case of Pectinidae, it has been suggested
75 that food could be the major route of Cd intake on the basis of elevated metal concentrations found in the
76 digestive gland (Palmer & Rand 1977, Uthe & Chou 1987, Bustamante & Miramand 2005a). However, it
77 appears necessary to confirm this assumption as the contribution of the dissolved phase could also lead to
78 high metal concentrations in the storage and detoxification organs (e.g., Borchardt 1983).
79 Therefore, the present work investigated uptake and loss kinetics of Cd in two species of scallops,
80 *Chlamys varia* and *Pecten maximus* exposed through seawater, food and/or sediment, depending of their
81 different living habitats - only seawater and food for *C. varia* and all pathways for *P. maximus* which is
82 living buried in the bottom sediment and is able to ingest large particles (Mikulich & Tsikhon-Lukamina
83 1981, Shumway et al. 1987). The use of highly sensitive radiotracer techniques allowed studying
84 bioaccumulation mechanisms at realistic Cd levels encountered in the field. Three levels of biological
85 organization were considered in this study, the whole individual, the different organs and the subcellular
86 fractions of the digestive gland cells, in order to evaluate the biokinetic parameters of the accumulation,
87 the distribution among the body compartments and the cellular forms of storage in the digestive gland,
88 respectively. Finally, we used a bioaccumulation model to determine the relative contribution of the
89 different exposure pathways of Cd for both species.

90

91 MATERIALS AND METHODS

92 **Sampling**

93 In spring 2004 and 2005, one hundred variegated scallops *Chlamys varia* and seventy king scallops
94 *Pecten maximus* were collected on the Atlantic coast (Pertuis Breton, Charente-Maritime) by SCUBA
95 diving. They were carefully transported to IAEA-MEL premises in Monaco and were acclimatized to
96 laboratory conditions for 4 weeks (constantly aerated open circuit aquarium; flux: 50 l h⁻¹; salinity: 36
97 p.s.u.; temperature: 17 ± 0.5°C; pH: 8.0 ± 0.1; light/dark cycle: 12 h/12 h) prior to experimentations.

98 During this period, scallops were fed daily an algal mixed diet (*Isochrysis galbana*, *Skeletonema*
99 *costatum*).

100

101 **Radiotracer and counting**

102 Uptake and loss kinetics of ^{109}Cd in scallop species were determined using a high specific activity
103 radiotracer purchased from Isotope Product Lab (^{109}Cd as CdCl_2 in 0.1M HCl, $T_{1/2} = 426.6$ d). The tracer
104 was counted using a high-resolution γ -spectrometer system composed of three Germanium -N or P type-
105 detectors (EGNC 33-195-R, Intertechnique) connected to a multichannel analyser (Intergamma,
106 Intertechnique). The radioactivity was determined by comparison with standards of known activity and of
107 appropriate geometry. Measurements were corrected for counting efficiency and physical radioactive
108 decay. The counting time was adjusted to obtain a propagated counting error less than 5 %.

109

110 **Seawater exposure**

111 Twenty three *Chlamys varia* and 23 *Pecten maximus* (average weight \pm SD: 30 ± 7 g and 208 ± 46 g,
112 respectively) were placed in a 70-l glass aquarium (constantly aerated closed circuit aquarium; salinity:
113 36 p.s.u.; temperature: $17 \pm 0.5^\circ\text{C}$; pH: 8.0 ± 0.1 ; light/dark cycle: 12 h/12 h) and exposed for 7 d to ^{109}Cd
114 dissolved in seawater (2 kBq l^{-1}). No change in pH was detectable after the tracer addition. Spiked
115 seawater was renewed twice a day the first two days and then daily in order to keep radioactivity in
116 seawater constant. Activity of the ^{109}Cd in seawater was checked before and after each spike renewal,
117 yielding time-integrated activities of $2.1 \pm 0.2 \text{ kBq l}^{-1}$.

118 Nine scallops of each species were collected at different time intervals and were whole-body
119 radioanalyzed alive (same identified individual each time). At the end of the 7-d exposure period, 5
120 scallops of each species were sacrificed and dissected. Shell, digestive gland, kidneys, gills, gonad,
121 mantle, intestine, adductor muscle and the rest of soft tissues were separated and radioanalyzed in order to
122 assess the ^{109}Cd body distribution. The remaining scallops were then placed in non contaminating

123 conditions (constantly aerated open circuit; flux: 50 l h⁻¹; salinity: 38 p.s.u.; temperature: 17 ± 0.5 °C; pH:
124 8.0 ± 0.1; light/dark cycle: 12 h/12 h) for 36 d and nine individuals of each species were regularly
125 radioanalyzed alive in order to follow the loss of ¹⁰⁹Cd from the scallops. Four scallops were collected at
126 the end of the depuration period and dissected into several body compartments as previously described.

127

128 **Food exposure**

129 The prymnesiophycean *Isochrysis galbana* was used to study ¹⁰⁹Cd transfer to scallops through their diet.
130 Phytoplankton cells were exposed to 4.8 kBq l⁻¹ ¹⁰⁹Cd during their growing phase (7 d). After that period,
131 phytoplankton medium was filtrated (1 µm-mesh size; Osmonic filters), and then resuspended in a 70-l
132 aquarium (constantly aerated closed-circuit; salinity: 36 p.s.u.; temperature: 17 ± 0.5°C; pH: 8.0 ± 0.1;
133 light/dark cycle: 12 h/12 h) where six *C. varia* and six *P. maximus* (average weight ± SD: 17 ± 5 g and
134 127 ± 14 g, respectively) were placed for one week before the feeding experiment. The radioactivity of
135 the labelled *I. galbana* was γ-counted before and after the filtration. Scallops were allowed to feed on
136 radiolabelled *I. galbana* for 2 h (cell concentration -5 10⁴ cell ml⁻¹- was selected to avoid pseudofeces
137 production). After the feeding period, all scallops were γ-counted and flowing seawater conditions (50 l h⁻¹
138 l) were restored in the aquarium. Individuals were then whole-body γ-counted alive at different time
139 intervals to follow the loss kinetics of ¹⁰⁹Cd. Four individuals were collected after 16 (*P. maximus*) and 30
140 d (*C. varia*) of depuration, and dissected to determine the ¹⁰⁹Cd tissue distribution among the different
141 body compartments (shell, digestive gland, kidneys, gills, gonad, mantle, intestine, adductor muscle and
142 the rest of soft tissues) and among the subcellular fraction of the digestive gland (see below).

143

144 **Sediment exposure**

145 Since *P. maximus* is living buried into the sediment whereas *C. varia* is fixed on rocks, Cd exposure
146 through sediment was only assayed for *P. maximus*. Sediment was collected in Wimereux (North-Atlantic
147 coast of France). Sediment grain size distribution was measured on a Mastersizer micro and the

148 evaluation of the dry/wet weight ratio was calculated after freeze drying in a LABCONCO Freezone18.
149 Aerated sediment (9 kg) was placed in plastic bottle, exposed to ^{109}Cd (516 kBq) for 6 d with constant
150 agitation, then used to form a homogeneous sediment layer of 4 cm height in a 20-l aquarium. Weakly
151 bound ^{109}Cd was allowed to leach overnight under flowing seawater (50 l h^{-1}) (Warnau et al. 1996). Ten
152 *P. maximus* (average weight \pm SD: $118 \pm 5 \text{ g}$) were then placed for 13 d in the aquarium (constantly
153 aerated open circuit; flux: 50 l h^{-1} ; salinity: 36 p.s.u.; temperature: $17 \pm 0.5^\circ\text{C}$; pH: 8.0 ± 0.1 ; light/dark
154 cycle: 12 h/12 h). Six individuals as well as sediment aliquots were regularly radioanalyzed during the
155 experiment duration. Activity of ^{109}Cd in sediment was constant all along the exposure period (24.2 ± 1.9
156 Bq g^{-1} wet wt). At the end of the uptake period, 4 scallops were collected, dissected (shell, digestive
157 gland, kidneys, gills, gonad, mantle, intestine, adductor muscle and the rest of soft tissues), weighed and
158 γ -counted in order to determine the radiotracer distribution among the body compartments. The remaining
159 individuals were transferred for 49 d to a new 20-l aquarium containing non contaminated sediment with
160 flowing seawater and they were regularly radioanalyzed to follow ^{109}Cd loss kinetics. Also, ^{109}Cd activity
161 in sediment was regularly measured in order to ascertain that no contamination of the clean sediment
162 occurred through ^{109}Cd recycling (for security, the whole sediment layer was renewed anyway after one
163 week). At the end of the loss period, 4 scallops were collected and dissected as described above to
164 determine ^{109}Cd body distribution and its subcellular distribution in the digestive gland.

165

166 **Subcellular distribution**

167 For all the experiments, the digestive gland of both scallop species were considered to assess the
168 partitioning of ^{109}Cd between soluble and insoluble fractions as described by Bustamante & Miramand
169 (2005b). Briefly, part of digestive gland were homogenized individually with a mortar and pestle on ice
170 with 10 ml of 0.02 M Tris-HCl buffer, 0.25 M sucrose, 1 mM phenylmethylsulfonylfluoride (PMSF, as
171 protease inhibitor), at pH 8.6. The homogenates were centrifuged at 80,000 G for 1 h at 5°C in a Sorvall

172 RC28S ultracentrifuge to separate particle-free supernatant (cytosol; soluble fraction) from the pellet
173 (insoluble fraction). Homogenate aliquots, cytosols, and pellets were then radioanalyzed.

174

175 **Data analysis**

176 Uptake of the radioisotope was expressed in term of concentration factors (CF: ratio between the ^{109}Cd
177 activity in scallops – Bq g^{-1} wet wt – and time-integrated activity in the seawater – Bq g^{-1}) over time for
178 the seawater exposure and in term of transfer factors (TF: ratio between the ^{109}Cd activity in scallops – Bq
179 g^{-1} wet wt – and time-integrated activity in the sediment – Bq g^{-1}) over time for the sediment exposure of
180 *P. maximus* (Warnau et al. 1996, 1999). Uptake kinetics of ^{109}Cd in whole-body scallops were fitted using
181 a simple exponential kinetic model (eq. 1) for the sediment exposure (Statistica[®] 6) and using a linear
182 model for the seawater exposure (eq. 2):

$$183 \text{CF}_t = \text{CF}_{\text{ss}} (1 - e^{-k_e t}) \text{ (eq. 1)}$$

$$184 \text{CF}_t = k_u t \text{ (eq. 2)}$$

185 where CF_t and CF_{ss} ($\text{CF}_{\text{ss}} = k_u/k_e$) are the concentration factors at time t (d) and at steady state,
186 respectively; k_u and k_e are the uptake and loss rate constants (d^{-1}), respectively (Whicker & Schultz 1982,
187 Warnau et al. 1996).

188 Depuration of Cd (seawater, food and sediment experiments) was expressed in terms of percentage of
189 remaining radioactivity (radioactivity at time t divided by initial radioactivity measured in scallops at the
190 beginning of the decontamination period * 100). The percentages of remaining activity were plotted
191 against time and loss kinetics were described by a double-component exponential model (eq. 3):

$$192 A_t = A_{0s} e^{-k_{es} t} + A_{0l} e^{-k_{el} t} \text{ (eq. 3)}$$

193 where A_t and A_0 are the remaining activities (%) at time t (d) and 0, respectively; k_e is the depuration rate
194 constant (d^{-1}); 's' and 'l' are the subscripts for the 'short-lived' and 'long-lived' components. For each
195 exponential component (s and l), a biological half-life can be calculated ($T_{b/2s}$ and $T_{b/2l}$) from the

196 corresponding depuration rate constant (k_{es} and k_{el} , respectively) according to the relation $T_{b/2} = \ln 2/k_e$
197 (Warnau et al. 1996). Regarding feeding experiments, the ‘long-lived’ exponential term describes the
198 fraction of the radiotracer ingested with food that is actually absorbed by the organism (Warnau et al.
199 1996). The corresponding A_{01} represents the assimilation efficiency (AE) of the considered radiotracer.
200 The best fitting regression models were selected according to highest determination coefficient and
201 examination of residuals. The level of significance for statistical analysis was always set at $\alpha < 0.05$.

202

203 RESULTS

204 Seawater exposure

205 Uptake of ^{109}Cd in whole-body *C. varia* and *P. maximus* displayed linear kinetics ($r^2 = 0.85$ and 0.66 ,
206 respectively; see Fig. 1). The values estimated for the kinetic parameters and their associated statistics are
207 presented in Table 1. The concentration factors measured at the end of the uptake period (CF_{7d}) of ^{109}Cd
208 were 37 ± 9 in *C. varia* and 18 ± 7 in *P. maximus* (Table 2). Calculated CF_{7d} for the different organs
209 indicated that ^{109}Cd was concentrated selectively in each species, according to the following order:

210 - *C. varia*: kidneys (928 ± 547) > digestive gland (322 ± 175) \approx gills (277 ± 102) \approx foot (265 ± 74) \approx rest
211 of soft tissues (258 ± 56) > gonad, mantle, intestine and adductor muscle ($\leq 53 \pm 11$)

212 - *P. maximus*: kidneys (690 ± 402) \approx digestive gland (659 ± 227) > gills (175 ± 13) > other tissues ($\leq 78 \pm$
213 33).

214 In terms of body distribution, ^{109}Cd was mainly found in the digestive gland and in the gills (~ 30 and 20
215 % of total body load, respectively) for both species. At the end of the uptake experiment, the ^{109}Cd tissue
216 distribution shows a similar pattern ($p_{G\text{-test}} > 0.40$) between *C. varia* and *P. maximus*, with the digestive
217 gland and gills accounting for more than 60 % of the total Cd load (Table 2).

218 After the exposure period, non-contaminating conditions were restored and loss kinetics of ^{109}Cd were
219 followed for 36 d. The whole-body loss kinetics of ^{109}Cd in *C. varia* and *P. maximus* were best described
220 by a two-component exponential model (Fig. 1 and Table 1). The major part of ^{109}Cd was efficiently

221 absorbed in *C. varia* and *P. maximus* ($A_{01} > 77\%$). The estimated loss rate constant of the long-lived
222 components (k_{el}) for *C. varia* was low, i.e. 0.005 ± 0.001 and, consequently, the derived biological half-
223 life reached 145 ± 45 d (Table 1). In the case of *P. maximus*, the loss rate constant was not significantly
224 different from 0 ($p > 0.05$), and the related $T_{b/2l}$ of ^{109}Cd may thus be considered as infinite.

225 After 36 d of depuration, the body distribution of ^{109}Cd displayed a similar pattern than the one observed
226 at the end of the exposure period (Table 2). However, it is striking to note that the ^{109}Cd activity in the
227 digestive gland of *C. varia* and *P. maximus* remained relatively constant throughout the depuration
228 duration within the two species, i.e. from 680 ± 369 Bq g⁻¹ to 549 ± 255 Bq g⁻¹ for *C. varia* and from
229 $1,392 \pm 479$ Bq g⁻¹ to $1,491 \pm 316$ Bq g⁻¹ for *P. maximus*, suggesting either a lack of Cd loss from the
230 digestive gland during this period or a redistribution of the radioisotope from the tissues in contact with
231 seawater towards this storage organ.

232

233 **Dietary exposure**

234 The loss kinetics of ^{109}Cd ingested with food in both *C. varia* and *P. maximus* were best fitted using a
235 double exponential model (Fig. 1 and Table 1). *C. varia* displayed a higher assimilation efficiency ($AE >$
236 86%) than *P. maximus* ($AE > 80\%$). However, in both species, the depuration rate constant, k_{el} , were not
237 significantly different from 0 ($p > 0.39$), and therefore the derived $T_{b/2l}$ were infinite.

238 At the end of the depuration period, the digestive gland contained the main part of ^{109}Cd , i.e. 97% for *C.*
239 *varia* and 82% for *P. maximus* (Table 2).

240

241 **Sediment exposure**

242 Sediment used in the experiment was mainly (95.8%) composed of grains which size ranged from 76 to
243 $302\ \mu\text{m}$ and its dry/wet wt ratio was 0.80.

244 Whole-body uptake kinetics of sediment-bound ^{109}Cd in *P. maximus* was best fitted by a single
245 exponential model (Table 1). TF reached steady-state equilibrium within the 2 weeks of exposure

246 (estimated $TF_{ss} = 0.034 \pm 0.002$). Among the different body compartments, the highest TF_{13d} was found
247 in the digestive gland (3.35 ± 1.68 ; Table 3). This organ also contained the main fraction of the total ^{109}Cd
248 body burden (i.e. 78 %; Table 3). The body compartment containing the second highest proportion was
249 the mantle (14 % of total ^{109}Cd body burden).

250 The ^{109}Cd whole-body loss kinetics could not be described accurately by the exponential models;
251 therefore a linear regression ($Y = aX + b$) was applied in order to estimate the radiotracer retention. The
252 results showed that 92 % of the accumulated ^{109}Cd were efficiently incorporated in *P. maximus* tissues,
253 with a biological half-life not significantly different from infinite (Table 1). At the end of the depuration
254 period (31 d) the body distribution of ^{109}Cd was identical to that at the end of the exposure period (Table
255 3), with the highest proportion of ^{109}Cd located in the digestive gland (≈ 80 %), followed by the mantle (\approx
256 12 - 14 %). In addition, the ^{109}Cd activities were similar in the two latter tissues at the end of exposure
257 and depuration periods, viz. 81 ± 41 and 85 ± 18 Bq g^{-1} in the digestive gland and 1.4 ± 0.4 and 1.5 ± 1.4
258 Bq g^{-1} in the mantle.

259

260 **Subcellular distribution**

261 Examination of subcellular distributions indicated that, whatever the contamination pathway (i.e.,
262 seawater, food or sediment) and the sampling period (i.e., end of uptake or end of loss period), *P.*
263 *maximus* stored the major part of the cellular ^{109}Cd in the soluble fraction (from 70 to 85 %). In contrast,
264 the radiotracer was mainly bound to insoluble compounds in *C. varia* (Fig. 2).

265

266 **DISCUSSION**

267 Pectinidae are an important marine resource which are both fished and cultured for human consumption
268 (Ansell et al. 1991, Waller 1991). Hence, the intake of contaminants such as metals by Man through
269 scallop consumption is a matter of concern. Indeed, Pectinidae are well known for their capacity of
270 accumulating high levels of metals, and especially Cd, in their tissues (Brooks & Rumsby 1965, Bryan

271 1973, Bustamante & Miramand 2004, 2005b). Interestingly, this high bioaccumulation potential for Cd is
272 not specific to anthropogenic contamination since scallops from the Antarctic Ocean have high Cd levels
273 compare to temperate species living in the coastal waters of industrialised countries (Mauri et al. 1990,
274 Viarengo et al. 1993).

275 Several field studies assumed that food would be the main intake pathway of Cd in scallops as high metal
276 levels are always found in the digestive gland (Palmer & Rand 1977, Uthe & Chou 1987, Bustamante &
277 Miramand 2005a). However, the contribution of the dissolved phase is difficult to ascertain in the field as
278 this route can lead to a significant uptake of Cd and to its redistribution towards storage tissues such as
279 the digestive gland. Therefore, there is a need to assess the relative importance of dissolved and
280 particulate Cd pathways in order to better understand their respective contributions, as well as to evaluate
281 the retention mechanisms leading to the high Cd levels measured in scallop tissues.

282 The experimental exposure of *Chlamys varia* and *Pecten maximus* to ¹⁰⁹Cd via seawater confirmed their
283 ability to concentrate Cd from the dissolved phase, as previously shown using elevated exposure levels of
284 stable Cd (Eisler et al. 1972, Carmichael & Fowler 1981). Indeed, after only 7 days of exposure to the
285 dissolved radiotracer, both scallop species exhibited high whole-body concentration factors (CFs), with
286 37 ± 9 for *C. varia* and 18 ± 7 for *P. maximus* whole bodies. This difference in CF between the two
287 species exposed to the same contamination conditions is related (1) to a higher Cd uptake rate (uptake rate
288 constant: 5.4 vs 2.7) and (2) secondarily, to a higher assimilated fraction (87.8 vs 77.1) in *C. varia*
289 compared to *P. maximus* (Table 1). However in the specimens collected from the field, *C. varia* displayed
290 typically lower Cd concentrations than *P. maximus* (Palmer & Rand 1977, Uthe & Chou 1987,
291 Bustamante & Miramand 2005a). This would suggest that *C. varia* has far more limited capacities of Cd storage
292 than *P. maximus*.

293 Considering the tissues separately, the organs involved in respiration (i.e. gills), excretion (i.e. kidneys)
294 and digestion (i.e. digestive gland) displayed higher CFs compared to other body compartments in *P.*
295 *maximus*, whereas the foot and the compartment “rest of the soft tissues” also showed elevated CFs in *C.*

296 *varia* (see Table 2). However, in terms of distribution among tissues and organs, Cd was mainly located
297 in the digestive gland, the gills, the kidney and the mantle in both species, the digestive gland containing
298 more than 30 % of the whole body burden of ¹⁰⁹Cd (Table 2). These results strongly suggest the
299 occurrence of efficient redistribution mechanisms towards the tissues involved in the storage, excretion
300 and detoxification processes, i.e. the kidneys and the digestive gland (e.g., Carmichael & Fowler 1981,
301 Ballan-Dufrançais et al. 1985, Stone et al. 1986). It is also striking to note the difference between both
302 species concerning the Cd CF in the foot that reached elevated values in *C. varia* (Table 2). In the latter
303 species, the foot is well developed and contains a byssal gland which main role is to produce the byssus to
304 stick to rocky substrates whereas *P. maximus* does not produce byssus as it lives buried in the sediment.
305 Byssus is known to play a role in the elimination of metals from bivalves (Szefer et al. 2006), it is
306 therefore likely that some metals are transferred from the soft tissues and concentrated in the byssus
307 rather than merely adsorbed onto its surface from seawater. However, in the case of Cd, previous studies
308 on mussels suggested that this metal is derived mainly from seawater (Coombs & Keller 1981, Nicholson
309 & Szefer 2003). The present study was not designed to address this specific issue and our results do allow
310 supporting internal transfer or waterborne origin of Cd in the byssus. However, further specifically-
311 designed studies using sensitive radiotracer techniques could bring most interesting information on the
312 origin of byssal Cd.

313 It is noteworthy that the Cd distribution pattern among the tissues was similar after 7 d of seawater
314 exposure and after 36 d of depuration for both species (Table 2). Similarly, the subcellular distribution of
315 Cd was identical at both times for *P. maximus*, with more than 80 % in the soluble fraction of the
316 digestive gland cells (Fig. 2). Taking into account the relatively long biological half-life of Cd in *P.*
317 *maximus*, this result indicate that the metal is mainly bound to soluble compounds involved in the storage
318 of this metal. The implication of metallothionein-like proteins in Cd detoxification and storage in the
319 digestive gland is well documented in Pectinidae (e.g., Stone et al. 1986, Evtushenko et al. 1990,
320 Bustamante & Miramand 2005b). However, in *C. varia*, Cd was mainly bound to insoluble compounds

321 (from 59 to 80 %; see Fig. 2), suggesting a time-limited role of the soluble metalloproteins when the
322 metal enters through the dissolved route (as well as via the food as similar results were found for the
323 dietary exposure; see Fig. 2). Such a predominant interaction of Cd with the insoluble cellular fraction in
324 the digestive gland is not a common observation among Pectinidae but has already been shown in some
325 species (e.g., *Adamussium colbecki*; Viarengo et al. 1993) and would be due to the fact that, among
326 insoluble cellular components (i.e., organelles, membranes and granules), the lysosomal system can play
327 a major role in Cd detoxification (by trapping) and excretion (Ballan-Dufrançais et al. 1985, Marigómez
328 et al. 2002).

329 After exposure to sediment-bound Cd, *P. maximus* exhibited very low transfer factors (viz., $TF_{ss} = 0.034$
330 ± 0.009), indicating that direct contamination due to burying into sediment would represent a minor Cd
331 uptake pathway in this species. However, at the end of the exposure period, 80 % of the incorporated
332 metal was found in the digestive gland, which displayed a TF higher than 3 (Table 3). As this organ is not
333 in direct contact with the sediment, it is suggested that either (1) the radiotracer was progressively
334 translocated from the tissues in direct contact with sediment and pore water to the digestive gland and/or
335 (2) *P. maximus* was able to ingest sediment grains. Although sediment grains were never observed in the
336 valves or in the digestive system in the many dissections carried out during this study, this latter
337 hypothesis would be plausible as scallops were reported to be able to ingest particles of a wide size range
338 (particles up to 950 μm have been found in scallop stomachs; Mikulich & Tsikhon-Lukamina 1981,
339 Shumway et al. 1987). Nevertheless, the assimilated Cd in the digestive gland was efficiently retained and
340 was mainly bound to cytosolic compounds in the same proportions as in the food experiment, supporting
341 the hypothesis of ingestion of sediment particles.

342 In the case of dietary exposure, Cd was assimilated to a similar extent in both species, with approx. 80 %
343 of the radiotracer being incorporated in the scallop tissues. Such a high assimilation efficiency (AE) is
344 striking as in other bivalve species, lower values were generally reported, e.g. for the tropical clam
345 *Gafrarium tumidum* (AE = 42 %), the tropical oysters *Isognomon isognomon* and *Malleus regula* (AEs =

346 58 and 51 %, respectively) and the blue mussel *Mytilus edulis* (AE ranging from 8 to 40 %) (e.g., Wang
347 & Fisher 1997; Hédouin 2006). These results suggest that food would be an important source of Cd for
348 Pectinidae. However, inter-specific differences in Cd concentrations in scallops from the field (where *C.*
349 *varia* showed the lowest concentrations) are difficult to explain in regards to the results obtained in our
350 experiments. Indeed, lower depuration rates resulted in calculated biological half-life exceeding 3 years
351 (Table 1), meaning that virtually all the assimilated Cd is readily stored in *C. varia* tissues. In contrast, the
352 biological half-life following food exposure was approx. 4 months for *P. maximus*, indicating a faster
353 turnover of the metal compared to *C. varia*. It is therefore likely that although living in the same areas, *C.*
354 *varia* and *P. maximus* do not share the same food in the marine environment. Indeed, different storage
355 mechanisms in prey can determine Cd bioavailability to higher trophic levels (e.g., Wallace & Lopez
356 1997, Wallace & Luoma 2003). Moreover, the dissolved and sediment pathways should also have a
357 strong importance in *P. maximus* (see above). The use of a bioaccumulation model is therefore a
358 mandatory step to further explore the importance of each exposure pathways (Thomann et al. 1995, Wang
359 & Fisher 1999). When applying such a model, food appears to be the major route of Cd accumulation in
360 *C. varia*, with 99.6 % of the metal being accumulated from phytoplankton. In *P. maximus*, it was not
361 possible to determine accurate data for the model because the kinetic parameters of the post sediment-
362 exposure loss phase were not significant. Therefore, we only considered food and seawater pathways. In
363 such conditions, results indicated that food accounted for 84.0 % of the accumulated Cd in *P. maximus*.
364 Owing to the high assimilation efficiency of sediment-bound Cd ($A_{01} = 92 \%$), it appears necessary to
365 better delineate the sediment contribution to Cd accumulation in order to consider the three different
366 pathways (seawater, food and sediment) on the global Cd bioaccumulation by *P. maximus*.

367

368 CONCLUSION

369 The present work on the bioaccumulation of Cd in two Pectinidae has confirmed the high Cd
370 bioaccumulation potential of *C. varia* and *P. maximus*. The organs accumulating Cd to the highest extent

371 in both species are the digestive gland and the kidneys whatever the exposure pathway was. Comparison
372 of results from laboratory experiments clearly showed that *C. varia* showed higher bioconcentration and
373 bioaccumulation capacities than *P. maximus*. Since field data have reported higher Cd levels in *P.*
374 *maximus* than in *C. varia*, it is suggested that Cd should be bioaccumulated by other uptake pathways
375 than food and seawater. The high assimilation efficiency of Cd ingested through sediment pathway in *P.*
376 *maximus* indicated that the particulate pathway could play an important role in the global Cd
377 bioaccumulation process and studies on sediment as well as on suspended particulate matter should be
378 further investigated to better simulate the different exposure routes of Cd to which Pectinidae are exposed
379 in the field. Nevertheless, differences between field and laboratory observations could be related to
380 different detoxification mechanisms in the two species.

381

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388

389 REFERENCES

- 390 Ansell, A.D., Dao, J-C., Mason, J., 1991. Fisheries and aquaculture: three European scallops: *Pecten*
391 *maximus*, *Chlamys (Aequipecten) opercularis* and *Chlamys varia*. In: Shumway SE (ed) *Scallops:*
392 *biology, ecology and aquaculture*. Elsevier, p 715-752
- 393 Ballan-Dufrançais, C., Jeantet, A.Y., Feghali, C., Halpern, S., 1985. Physiological features of heavy metal
394 storage in bivalve digestive cells and amoebocytes: EPMA and factor analysis of correspondences.
395 *Biol Cell* 53:283-292

- 396 Borchardt, T., 1983. Influence of food quantity on the kinetics of cadmium uptake and loss via food and
397 seawater in *Mytilus edulis*. Mar Biol 76:67-76
- 398 Brooks, R.R., Rumsby, M.G., 1965. The biogeochemistry of trace element uptake by some New Zealand
399 bivalves. Limnol Oceanogr 10:521-527
- 400 Bryan, G.W., 1973. The occurrence and seasonal variation of trace metals in the scallops *Pecten maximus*
401 (L.) and *Chlamys opercularis* (L.). J Mar Biol Assoc UK 53:145-166
- 402 Bustamante, P., Miramand, P., 2004. Interspecific and geographical variations of trace element
403 concentrations in Pectinidae from European waters. Chemosphere 57:1355-1362
- 404 Bustamante, P., Miramand, P., 2005a. Evaluation of the variegated scallop *Chlamys varia* as a biomonitor
405 of temporal trends of Cd, Cu, and Zn in the field. Environ Pollut 138:109-120
- 406 Bustamante, P., Miramand, P., 2005b. Subcellular and body distributions of 17 trace elements in the
407 variegated scallop *Chlamys varia* from the French coast of the Bay of Biscay. Sci Tot Environ
408 337:59-73
- 409 Carmichael, N.G., Fowler, B.A., 1981. Cadmium accumulation and toxicity in the kidney of the bay
410 scallop *Argopecten irradians*. Mar Biol 65:35-43
- 411 Coombs, T.L., Keller, P.J., 1981. *Mytilus* byssal threads as an environmental marker for metals. Aquatic
412 Toxicol 1:291-300
- 413 Eisler, R., 1985. Cadmium hazards to fish, wildlife and invertebrates: a synoptic review. US Fish and
414 Wildlife Service, Laurel, MD, 46 p
- 415 Eisler, R., Zarogian, G.E., Hennekey, R.J., 1972. Cadmium uptake by marine organisms. J Fish Res
416 Board Can 29:1367-1369
- 417 Evtushenko, Z.S., Lukyanova, O.N., Bel'cheva, N.N., 1990. Cadmium bioaccumulation in organs of the
418 scallop *Mizuhopecten yessoensis*. Mar Biol 104:247-250
- 419 Fowler, S.W., 1982. Biological transfert and transport processes. In: G. Kullenberg (ed) Pollutant transfer
420 and transport in the Sea. CRC Press, Inc., Boca Raton, FL, p 2-65

- 421 Gould, E., Thompson, R.J., Buckley, L.J., Rusanowsky, D., Sennefelder, G.R., 1988. Uptake and effects
422 of copper and cadmium in the gonad of the scallop *Placopecten magellanicus*: concurrent metal
423 exposure. *Mar Biol* 97:217-223
- 424 Hédouin, L. 2006. Caractérisation d'espèces bioindicatrices pour la surveillance des activités minières et
425 la gestion de l'environnement en milieu récifal et lagunaire : application au lagon de la Nouvelle-
426 Calédonie. PhD thesis, University of La Rochelle, France, 327 p
- 427 Janssen, H.H., Scholz, N., 1979. Uptake and cellular distribution of cadmium in *Mytilus edulis*. *Mar Biol*
428 55:133-141
- 429 Marigómez, I., Soto, M., Cajaraville, M.P., Angulo, E., Giamberini, L., 2002. Cellular and subcellular
430 distribution of metals in molluscs. *Microsc Res Techniq* 56:358-392
- 431 Mauri, M., Orlando, E., Nigro, M., Regoli, F., 1990. Heavy metals in the Antarctic scallop *Adamussium*
432 *colbecki*. *Mar Ecol Prog Ser* 67:27-33
- 433 Mikulich, L.V., Tsikhon-Lukamina, A., 1981. Food of the scallop. *Oceanology* 21:633-635
- 434 Nicholson, S., Szefer, P., 2003. Accumulation of metals in the soft tissues, byssus and shell of the mytilid
435 mussel *Perna viridis* (Bivalvia: Mytilidae) from polluted and uncontaminated locations in Hong
436 Kong coastal waters. *Mar Pollut Bull* 46:1040-1043
- 437 Palmer, J., Rand, G., 1977. Trace metal concentrations in two shellfish species of commercial importance.
438 *Bull Environ Contam Toxicol* 18:512-520
- 439 Reinfelder, J.R., Fisher, N.S., Luoma, S.N., Nichols, J.W., Wang, W-X., 1998. Trace element trophic
440 transfer in aquatic organisms: A critique of the kinetic model approach. *Sci Tot Environ* 219:117-
441 135
- 442 Riisgard, H.U., Bjornestad, E., Mohlenberg, F., 1987. Accumulation of cadmium in the mussel *Mytilus*
443 *edulis*: kinetics and importance of uptake via food and seawater. *Mar Biol* 96:349-353
- 444 Shumway, S.E., Selvin, R., Schick, D.F., 1987. Food resources related to habitat in the scallop
445 *Placopecten magellanicus* (Gmelin, 1791): A qualitative study. *J Shellfish Res* 6:89-95

- 446 Stone, H.C., Wilson, S.B., Overnell J., 1986. Cadmium binding components of scallop (*Pecten maximus*)
447 digestive gland. Partial purification and characterization. *Comp Biochem Physiol C* 85:259-268
- 448 Szefer, P., Fowler, S.W., Ikuta, K., Paez Osuna, F., Ali, A.A., Kim, B-S., Fernandes, H.M., Belzunce, M-
449 J., Guterstam, B., Kunzendorf, H., Wołowicz, M., Hummell, H., Deslous-Paoli, M. 2006. A
450 comparative assessment of heavy metal accumulation in soft parts and byssus of mussels from
451 subarctic, temperate, subtropical and tropical marine environments. *Environ Pollut* 139:70-78
- 452 Thomann, R.V., Mahony, J.D., Mueller, R., 1995. Steady-state model of biota sediment accumulation
453 factor for metals in two marine bivalves. *Environ Toxicol Chem* 14:1989-1998
- 454 Uthe, J.F., Chou, C.L., 1987. Cadmium in sea scallop (*Plactopecten magellanicus*) tissues from clean and
455 contaminated areas. *Can J Fish Aquat Sci* 44:91-98
- 456 Viarengo, A., Canesi, L., Massu-Cotelli, A., Ponzano, E., Orunesu, M., 1993. Cu, Zn, Cd content in
457 different tissues of the Antarctic scallop *Adamussium colbecki* (Smith 1902): role of
458 metallothionein in the homeostasis and in the detoxification of heavy metals. *Mar Environ Res*
459 35:216-217
- 460 Wallace, W.G., Lopez, G.R., 1997. Bioavailability of biologically sequestered cadmium and the
461 implications of metal detoxification. *Mar Ecol Prog Ser* 147:149-157
- 462 Wallace, W.G., Luoma, S.N., 2003. Subcellular compartmentalization of Cd and Zn in two bivalves. II.
463 Significance of trophically available metal (TAM). *Mar Ecol Prog Ser* 257:125-137
- 464 Waller, T.R., 1991. Evolutionary relationships among commercial scallops (Mollusca: Bivalvia:
465 Pectinidae). In: Shumway SE (ed) *Scallops: biology, ecology and aquaculture*. Elsevier, p 1-74
- 466 Wang, W-X., Fisher, N.S., 1997. Modeling metal bioavailability for marine mussels. *Rev Environ*
467 *Contam Toxicol* 151:39-65
- 468 Wang, W-X., Fisher, N.S., 1999. Delineating metal accumulation pathways for marine invertebrates. *Sci*
469 *Tot Environ* 237/238:459-472

- 470 Warnau, M., Fowler, S.W., Teyssié J.L., 1999. Biokinetics of radiocobalt in the asteroid *Asterias rubens*
471 (Echinodermata): sea water and food exposures. Mar Pollut Bull 39: 159-164
- 472 Warnau, M., Teyssié, J.L., Fowler, S.W. 1996. Biokinetics of selected heavy metals and radionuclides in
473 the common Mediterranean echinoid *Paracentrotus lividus*: sea water and food exposures. Mar.
474 Ecol Prog Ser 141: 83-94
- 475 Whicker, F.W., Schultz, V., 1982. Radioecology: nuclear energy and the environment, Vol. 2. CRC Press,
476 Boca Raton, FL, 320 p

477 Table 1. *Chlamys varia* and *Pecten maximus*. Whole-body uptake and loss kinetic parameters of ¹⁰⁹Cd following different exposure experiments:

478 1) 7-d exposure via seawater (n = 9) followed by 36 d of depuration (n = 9);

479 2) 2-hr feeding on radiolabelled *Isochrysis galbana* followed by a depuration period of 16 d (*P. maximus*, n = 6) or 30 d (*C. varia*, n = 6);

480 3) 13-d exposure of *P. maximus* via the sediments (n = 8) followed by 31 d of depuration (n = 8).

481 Uptake parameters: CF_{ss} / TF_{ss} concentration and transfer factors at steady state; k_u: uptake rate constant (d⁻¹)

482 Depuration parameters: A_{0s} and A_{0l}: activity (%) lost according to the short- and the long-lived exponential component, respectively; T_{b/2} :

483 biological half-life (d). ASE: asymptotic standard error; r²: determination coefficient of the uptake or loss kinetics

Experiment	Species	a. Uptake			b. Loss				
		CF _{ss} / TF _{ss} ± ASE	k _u ± ASE	r ²	A _{0s} ± ASE	T _{b/2s} ± ASE	A _{0l} ± ASE	T _{b/2l} ± ASE	r ²
1) Seawater	<i>C. varia</i>	-	5.4 ± 0.2 ^d	0.85	12.2 ± 3.8 ^b	0.8	87.8 ± 2.4 ^d	145 ± 45 ^b	0.31
	<i>P. maximus</i>	-	2.7 ± 0.1 ^d	0.66	23.4 ± 5.7 ^c	1.1	77.1 ± 4.8 ^d	913	0.49
2) Feeding	<i>C. varia</i>	-	-	-	14.5 ± 4.1 ^c	0.4	85.8 ± 2.1	989	0.21
	<i>P. maximus</i>	-	-	-	20.5 ± 6.1 ^b	0.02	79.5 ± 3.7 ^d	138	0.37
3) Sediment	<i>P. maximus</i>	0.034 ± 0.002 ^d	0.014 ± 0.002 ^d	0.62	NC	NC	92 ^d	NC	-

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485 Probability of the model adjustment: ^a p < 0.05, ^b p < 0.01, ^c p < 0.001, ^d p < 0.0001; NC: not calculated

486

487 Table 2. *Chlamys varia* and *Pecten maximus*. Concentration Factors (mean CF \pm SD) and
 488 tissue distribution (mean % \pm SD) of ^{109}Cd during seawater (end of exposure and depuration
 489 periods) and feeding experiments (16 and 30 d after feeding for *P. maximus* and *C. varia*,
 490 respectively).

Species Compartments	Seawater contamination		Food contamination	
	Uptake (7 d, n=5) Concentration Factor	Distribution (%)	Loss (36 d, n=4) Distribution (%)	Loss (n=5) Distribution (%)
<i>Chlamys varia</i>				
Digestive gland	322 \pm 175	33 \pm 14	41 \pm 18	97 \pm 1
Gills	277 \pm 102	30 \pm 9	23 \pm 6	< 1
Kidneys	928 \pm 547	13 \pm 6	15 \pm 8	< 1
Intestine	23 \pm 7	< 1	1 \pm 1	< 1
Gonad	45 \pm 65	1 \pm 1	1 \pm 1	1 \pm 0
Foot	265 \pm 74	3 \pm 1	2 \pm 0	< 1
Mantle	53 \pm 11	12 \pm 4	10 \pm 6	< 1
Adductor muscle	21 \pm 6	4 \pm 1	5 \pm 3	< 1
Remaining tissues	258 \pm 56	5 \pm 1	2 \pm 0	0 \pm 1
Whole body	37 \pm 9			
<i>Pecten maximus</i>				
Digestive gland	659 \pm 227	38 \pm 10	49 \pm 5	82 \pm 19
Gills	175 \pm 13	28 \pm 11	19 \pm 2	1 \pm 0
Kidneys	690 \pm 402	10 \pm 4	12 \pm 4	6 \pm 12
Intestine	16 \pm 3	< 1	< 1	1 \pm 1
Gonad	18 \pm 10	2 \pm 2	2 \pm 1	9 \pm 17
Foot	13 \pm 5	< 1	< 1	1 \pm 1
Mantle	28 \pm 5	11 \pm 2	10 \pm 7	< 1
Adductor muscle	18 \pm 7	9 \pm 3	7 \pm 1	< 1
Remaining tissues	78 \pm 33	2 \pm 0	1 \pm 0	1 \pm 0
Whole body	18 \pm 7			

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495 Table 3. *Pecten maximus*. Transfer Factors (mean TF \pm SD; n = 4) of ^{109}Cd after a 13-d
 496 exposure via sediment and tissue distribution (mean % \pm SD) of ^{109}Cd at the end of the 13-d
 497 exposure and 31-d depuration period (n= 5).

Compartments	Uptake phase		Loss phase	498
	Transfer Factor	Distribution (%)	Distribution (%)	499
Digestive gland	3.35 \pm 1.68	78 \pm 10	80 \pm 10	501
Gills	0.05 \pm 0.04	4 \pm 3	6 \pm 1	502
Kidneys	0.12 \pm 0.04	1 \pm 1	< 1	503
Intestine	0.09 \pm 0.05	< 1	< 1	504
Gonad	0.06 \pm 0.05	1 \pm 0	< 1	505
Foot	0.03 \pm 0.01	< 1	< 1	506
Mantle	0.06 \pm 0.02	14 \pm 8	12 \pm 10	
Adductor muscle	0.00 \pm 0.00	1 \pm 1	< 1	
Remaining tissues	0.06 \pm 0.05	1 \pm 1	< 1	
Whole body	0.04 \pm 0.01			

507 Caption to figures.

508

509

510 Figure 1. *Chlamys varia* and *Pecten maximus*. Uptake and loss kinetics of ^{109}Cd in scallops
511 exposed for 7 d via seawater (uptake kinetics A1; Concentration Factors -CF-; mean \pm SD; n
512 = 9), then maintained for 36 d in non contaminated conditions (loss kinetics A2; Remaining
513 activity -%-; mean \pm SD; n = 9) and after a 2-hr feeding on radiolabelled phytoplankton
514 *Isochrysis galbana* (loss kinetics B; Remaining activity -%-; mean \pm SD; n = 6 *C. varia* and n
515 = 9 *P. maximus*).

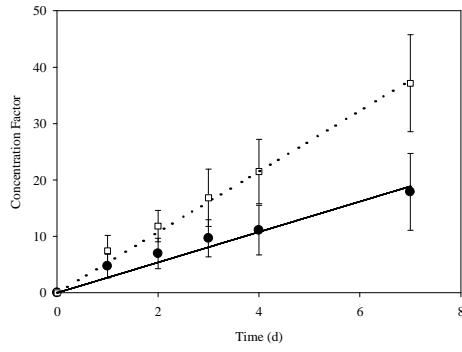
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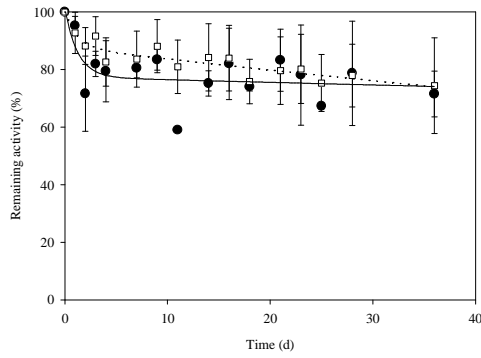
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519 Figure 2. *Chlamys varia* and *Pecten maximus*. Subcellular distribution of ^{109}Cd in the
520 digestive gland cells following different exposure experiments: (1) 7-d exposure via seawater
521 followed by 36 d of depuration; (2) 2-hr feeding on radiolabelled *Isochrysis galbana* followed
522 by a depuration period of 16 d (*P. maximus*) or 30 d (*C. varia*); (3) 13-d exposure of *P.*
523 *maximus* via the sediments followed by 31 d of depuration.

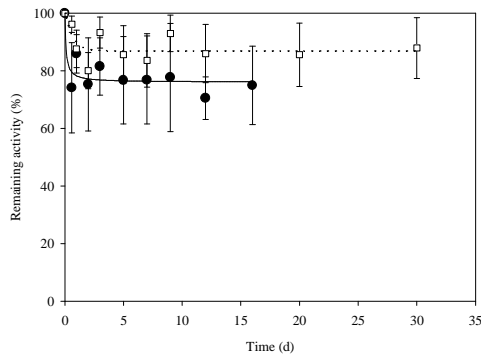
A1. Uptake via seawater



A2. Loss after seawater exposure



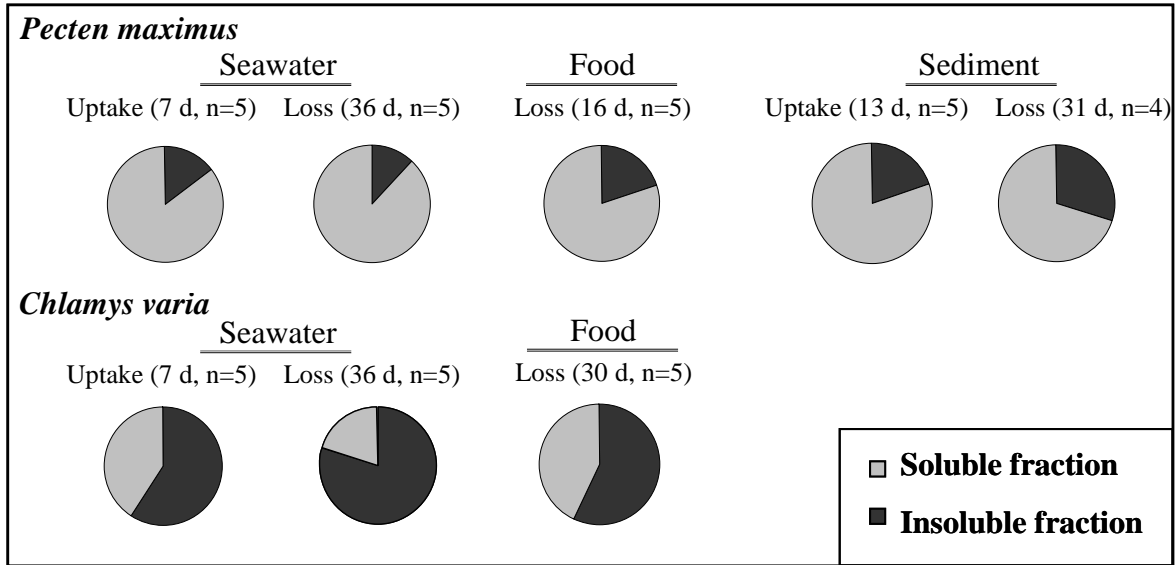
B. Loss after food exposure



.....□..... *Chlamys varia* ● *Pecten maximus*

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Figure 1.



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Figure 2.