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Harriet Kuranchie-Mensah, Simon Pouil, Jean-Louis Teyssié, François Oberhänsli, Michel Warnau, et al.. Allometric relationship in the bioaccumulation of radionuclides (^{134}Cs & ^{241}Am) and delineation of contamination pathways (food and seawater) in bloody cockle *Anadara senilis* using radiotracer techniques. *Journal of Environmental Radioactivity*, 2018, 192, pp.448-453. 10.1016/j.jenvrad.2018.07.018 . hal-03163215

HAL Id: hal-03163215

<https://hal.science/hal-03163215v1>

Submitted on 5 Sep 2023

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1 **Allometric relationship in the bioaccumulation of radionuclides (^{134}Cs & ^{241}Am) and**
2 **delineation contamination pathways (food and seawater) in bloody cockle *Anadara***
3 ***senilis* using radiotracer techniques**

4

5 Harriet Kuranchie-Mensah^{1,2}, Simon Pouil¹, Jean-Louis Teyssié¹, François Oberhänsli¹,
6 Michel Warnau^{1,3}, Marc Metian¹

7

8

9 1 Radioecology Laboratory, International Atomic Energy Agency - Environment
10 Laboratories (IAEA-EL), 4a Quai Antoine 1er, 98000 Monaco, Principality of
11 Monaco

12 2 Nuclear Chemistry and Environmental Research Centre, National Nuclear Research
13 Institute, Ghana Atomic Energy Commission, P. O. Box LG 80, Legon, Accra,
14 Ghana

15 3 Division for Africa, Department of Technical Cooperation, International Atomic
16 Energy Agency (Vienna International Centre), PO Box 100, 1400 Vienna, Austria |

17

18 * Corresponding author: Marc Metian
19 Radioecology Laboratory
20 IAEA Environment Laboratories
21 4a Quai Antoine 1er
22 MC-98000 Principality of Monaco
23 Telephone: +377 97 97 72 17

24 **Abstract:**

25 The uptake and depuration kinetics of ^{134}Cs and ^{241}Am were investigated in the bloody cockle
26 *Anadara senilis* exposed via seawater and food in controlled conditions, using animals of
27 different weight groups in order to assess how their bioaccumulation is affected by allometry
28 and, hence, the individual's age. This study is one of the few experiments investigating
29 bioaccumulation capacities of radionuclides in a West-African bivalve. Results showed that
30 allometric relationships were mainly dependent on the exposure pathway considered.
31 Significant relationships with body weight of bloody cockles were found during the uptake
32 from dissolved phase for both radionuclides; they followed inverse power functions: smaller
33 cockles concentrated both radionuclides more than larger ones. In contrast, radionuclide
34 absorption and assimilation efficiencies from food did not show any significant relationship
35 with weight: only slight variation was observed between small and large organisms for the
36 retention of ^{241}Am accumulated from food. A bioaccumulation model was used to assess the
37 contribution of each pathway of exposure (food vs. water) in organisms grouped in small and
38 large individuals. We found that, regardless of the size, ^{134}Cs was mainly bioaccumulated
39 through the dietary pathway. In the case of ^{241}Am , the relative contribution of each pathway is
40 weight-dependent: major contribution of dissolved pathway in smaller organisms and the
41 major dietary contribution in larger organisms.

42 Keywords: Ghana, Bloody cockle; Radionuclides; Allometry; Food and water exposure

43 **1. Introduction**

44 Radionuclides of cesium and americium are waste products from nuclear-related activities
45 such as nuclear power generation in nuclear power plants or research reactors (Al-Trabulsy et
46 al., 2011; El Mamoney and Khater, 2004). Although nuclear industries in Western Africa are
47 underdeveloped, Ghana operates a 30-kW miniature neutron source reactor (MNSR) and is
48 currently prospecting for the introduction of nuclear power as supplement to its energy mix to
49 boost industrialization and national development (Dampare et al., 2005) (Diawuo and
50 Kaminski, 2017). At present time, operational safety performances of nuclear facilities
51 worldwide are trustful. However, the occurrence of accidental releases of radionuclides
52 cannot be disregarded since it may impact the public and the environment, including aquatic
53 systems. Such possibilities are part of the preparedness and emergencies scenarios that are
54 examined prior to the installation of any nuclear premise. Prior and during operations, surveys
55 of environmental occurrence of radionuclides are also required to assess initial situation and
56 to allow detecting and identifying any radionuclide release and contamination.

57 Among the common approaches carried out to study environmental contamination, the use of
58 bioindicator species has proven to be a relevant and informative approach (e.g. Nielsen et al.,
59 2007). The criteria for selecting an organism to be used as a bioindicator have been described
60 by many authors (e.g. Jones and Kaly, 1996; Phillips, 1977; Rainbow, 1995). The main
61 criterion is to bioaccumulate the target contaminants. Bioindicators should also be
62 preferentially sessile or sedentary organisms, widely distributed and easy to collect all year
63 round. In addition, after selection of a candidate bioindicator, influence of biotic and abiotic
64 factors (such as temperature, salinity or body size) that are likely to affect its ability to
65 bioaccumulate the target contaminants should be characterized (Ke et al., 2000; Wang and
66 Fisher, 1998; Warnau and Bustamante, 2007). Among these factors, organism body size,

67 which for bivalves is correlated with age and weight, is known to be of primary importance
68 (Boyden et al., 1974; Hédouin et al., 2006; Strong and Luoma, 1981; Tang et al., 2017).

69 Since they fulfill the criteria of relevant bioindicators, bivalves are widely used in
70 biomonitoring programs (Fernández-Tajes et al., 2011; Rainbow, 1995; Yusof et al., 2004;
71 Zuykov et al., 2013). In Ghana, the bloody cockle *Anadara senilis*, a seafood locally
72 consumed due to high nutrient content (Gabriel et al., 2011), is a good candidate bioindicator
73 since it is long-lived, abundant and easy to identify in the field (Mirsadeghi et al., 2013;
74 Sarkar et al., 1994, 2008; Theng et al., 2004). Some studies have already highlighted the
75 ability of this species to bioaccumulate trace elements, including radionuclides such as ^{210}Po
76 (Theng et al., 2004), which can potentially expose human consumers to contamination by
77 ingestion.

78 The main objectives of the present study was to (1) assess the bioaccumulation capacity of the
79 bloody cockles *Anadara senilis* for two radionuclides (^{134}Cs and ^{241}Am) when exposed via
80 seawater and through the food, (2) assess the relative contribution of dissolved vs. trophic
81 exposure pathways to the whole-body radionuclide bioaccumulation, and (3) describe the
82 possible influence of the body weight of the bloody cockles *A. senilis* on the bioaccumulation
83 of these radionuclides in order to select the most appropriate individual weight range to be
84 used in biomonitoring studies.

85

86 **2. Materials and Methods**

87 2.1. Origin and acclimation of bloody cockles

88 Bloody cockles of different sizes and weights (shell length: 21-63 mm, whole-body wet
89 weight: 5-75 g) were hand-picked from the Narkwa lagoon located in the Central region of
90 Ghana. The cockles were transported to the premises of the Environment Laboratories of the
91 IAEA in the Principality of Monaco, where experimental work was carried out. Upon

92 reaching the laboratory, cockles were cleansed of adhering sediments, sorted into different
93 weight batches and held in a 140-L aquarium containing 0.45 μm -filtered seawater. The
94 cockles were then acclimated to experimental conditions in a constantly aerated, open-circuit
95 aquarium (salinity: 38 p.s.u.; water renewal rate: 50 L h⁻¹; temperature: 27 \pm 0.5 °C; pH: 8.0 \pm
96 0.1; light/dark cycle: 12h/12h depicting natural light conditions using fluorescent tubes) for
97 two months prior to experiments. During the acclimation period, the bloody cockles were fed
98 daily a phytoplankton diet (*Isochrysis galbana* and *Skeletonema costatum*). Observed
99 mortality was limited to few individuals.

100 2.2. Experimental procedures

101 2.2.1. Radiotracers and counting

102 Uptake and depuration kinetics of radionuclides in bloody cockles were determined using
103 high-specific activity radionuclides (¹³⁴Cs as chloride in H₂O, t_{1/2} = 2 years; ²⁴¹Am as nitrate in
104 1M HNO₃, t_{1/2} = 433 years) purchased from Czech Metrological Institute and Isotope
105 Products. The radioactivity of Cs and Am in water, cockles and phytoplankton were measured
106 using a high-resolution γ -spectrometry system equipped with five Germanium - N or P type -
107 detectors (EGNC 33-195-R, Canberra® and Eurysis®) connected to a multi-channel analyzer
108 and a computer employing spectral analysis software (Interwinner 6, Intertechnique). To
109 assure quality of radioanalyses, standards were used. These standards were consisting of
110 “phantoms” for each of the types of sample used (liquids and three sizes of bloody cockles),
111 spiked with known radionuclide activities in order to get the appropriate density and
112 geometry (Cresswell et al., 2017). To briefly describe the preparation of the “phantoms” of
113 bloody cockles, three different size standards were done, covering the whole range of sizes,
114 using empty shells filled with imbibed (2M HCL) paper towel, spiked with known
115 concentrations of radiotracers and then closed and placed in a counting box identical to the
116 one used to count the living bloody cockles. The position of “phantoms”, defined for the

117 calibration of our detectors, and living bloody cockles was kept the same during the
118 experiment in order to ensure a constant geometry. Corrections were made for background
119 and physical radioactive decay. The counting time was adjusted to obtain a propagated
120 counting error less than 5% (Rodriguez y Baena et al., 2006).

121 2.2.2. Seawater exposure

122 Bloody cockles of different weights/sizes (7.51-74.17 g, 23-63 mm of length, 16-40 mm of
123 width, 20-52 mm of height; n = 43) were placed in a 70-L aquarium (constantly aerated,
124 closed-circuit; salinity: 38 p.s.u.; temperature: 27 ± 0.5 °C; pH: 8.0 ± 0.1 ; light/dark cycle:
125 12h/12h) and exposed for 28 d to dissolved radiotracers in 0.45-mm filtered seawater (1 Bq
126 $^{134}\text{Cs mL}^{-1}$ and 0.2 Bq $^{241}\text{Am mL}^{-1}$). Seawater and spikes were renewed daily for 5 days, then
127 every second day in order to keep exposure activities as constant as possible. The
128 radioactivity in seawater was checked before and after each spike renewal to yield a time
129 integrated activity of 0.94 ± 0.09 Bq mL^{-1} and 0.13 ± 0.08 Bq mL^{-1} respectively for ^{134}Cs and
130 ^{241}Am . Prior to the renewal of seawater and spike, the cockles were fed for 1 h on a mixed
131 diet of *I. galbana* and *S. costatum* ($5 \cdot 10^4$ cells mL^{-1}) in clean seawater in order to avoid
132 ingestion of radiotracer via the food. Twenty tag-identified bloody cockles from two weight
133 classes were collected at different time intervals. Class “small” had an average weight of 9.8
134 ± 2.2 g (Mean \pm SD; with a length of 26.5 ± 2.9 mm, width of 20.6 ± 2.3 mm, height of $25.0 \pm$
135 2.4 mm, n= 10) and class “big” had an average weight of 36.3 ± 6.7 g (with a length of $43.4 \pm$
136 3.3 mm, width of 31.0 ± 2.6 mm, height of 38.6 ± 2.8 mm; n = 10) were collected at different
137 time intervals and were whole-body radioanalysed alive in order to follow the uptake kinetics
138 of the radiotracers. Prior to radioanalysis, radiolabelled cockles were rinsed in
139 uncontaminated seawater for 2 min to remove loosely bound radionuclides. At the end of the
140 28-d exposure period, all the bloody cockles were radioanalysed and then placed in clean
141 water in the same conditions for 35 d and 10 tag-identified individuals were regularly

142 radioanalysed alive in order to follow the depuration kinetics of the radiotracers. During the
143 35-d depuration period, non-exposed individuals were used as controls of possible tracer
144 recycling through seawater. Bloody cockles were fed daily with non-radiolabeled *I. galbana*
145 and *S. costatum* (5×10^4 cells mL⁻¹).

146

147 2.2.3. Dietary exposure

148 Transfer of radionuclides (¹³⁴Cs and ²⁴¹Am) to bloody cockles through their diet was studied
149 using the Haptophyceae *I. galbana*. Phytoplankton cells were exposed to 10 Bq ¹³⁴Cs mL⁻¹
150 and 2 Bq ²⁴¹Am mL⁻¹ during their exponential growth phase (5 d). After that period, the
151 phytoplankton medium was filtered (1- μ m mesh size; Osmonic® filters) and filters were
152 rinsed with non-contaminated seawater. Since the ingestion rate (IR) is dependent of the
153 weight of the organisms, bloody cockles of two weight classes had been placed into two
154 different 20-L aquaria (closed-circuit; same parameters as previously described), before the
155 dietary experiment. Class “small” had an average weight of 12.20 ± 1.96 g (Mean \pm SD; with
156 a length of 29.8 ± 2.9 mm, width of 22.3 ± 2.1 mm, height of 27.1 ± 2.9 mm; n= 9) and class
157 “big” had an average weight of 28.77 ± 5.07 g (with a length of 40.9 ± 2.3 mm, width of 29.9
158 ± 2.3 mm, height of 38.0 ± 4.9 mm; n = 7). Bloody cockles were single-fed with radiolabeled
159 phytoplankton for 2 h in closed-circuit conditions by resuspension of the filtered cells to a
160 density of 5×10^4 cells mL⁻¹ to avoid pseudofeces production by the bivalves and thus to avoid
161 the loss of a part of the ingested radionuclides prior to the first analysis (Kuranchie-Mensah et
162 al., 2016; Pouil et al., 2015). After the feeding period, the radiolabelled cockles were rinsed in
163 a non-radioactive free flowing seawater aquarium for 2 min to remove possibly adhering
164 radiolabeled phytoplankton, pat dry with absorbent paper and subsequently γ -counted alive.
165 Seawater flow was restored and depuration was followed during 28 d in open circuit
166 conditions. Non-exposed individuals were used as controls for possible tracer recycling.

167 During the 28-d depuration period, bloody cockles were fed daily with non-radiolabeled *I.*
168 *galbana* and *S. costatum* ($5 \cdot 10^4$ cells mL⁻¹).

169

170 2.3. Data analysis

171 2.3.1. Biokinetics

172 Whole-body uptake and depuration kinetics were fitted using nonlinear regression routines
173 and iterative adjustment (Statistica® 7) as described by Warnau et al. (1996). Whole-body
174 uptake kinetics of radiotracers were expressed in terms of changes in bioconcentration factor
175 over time (CF, ratio between activity of the radiotracer in the whole organism or in a body
176 compartment –Bq g⁻¹ wet weight– and time-integrated activity of radiotracer in seawater –Bq
177 g⁻¹–). Radiotracer uptake kinetics were best described using a first-order saturation
178 exponential model (Eq. 1):

$$179 \quad CF_t = CF_{ss}(1 - e^{-k_e t}) \quad (1)$$

180 where CF_t and CF_{ss} are the bioconcentration factors at time t (d) and at steady state,
181 respectively; k_e is the depuration rate constant (d⁻¹).

182 Depuration of radiotracers was expressed as the percentage of remaining radioactivity over
183 time (radioactivity at time t divided by the initial radioactivity measured in the organism at
184 the period $\times 100$). The depuration kinetics for both radiotracers were best described using a
185 two-component exponential model (Eq. 2):

$$186 \quad A_t = A_{0s} \cdot e^{-k_{es}t} + A_{0l} \cdot e^{-k_{el}t} \quad (2)$$

187 where A_t and A_0 are the remaining activities (%) at time t (d) and 0, respectively; k_e is the
188 depuration rate constant (d⁻¹); ‘s’ and ‘l’ are the subscripts for the ‘short-lived’ and ‘long-
189 lived’ components respectively.

190

191 2.3.2. Allometric relationships

192 Concentration Factors and remaining activities measured at the end of uptake and depuration
193 periods, respectively, were plotted against individual whole-body weight (g) and allometric
194 relationships were tested using simple linear, power, hyperbolic or exponential models (Zar,
195 1996). When an allometric relationship was found, the power model was always the best
196 fitting model with the following equation (Eq. 3):

$$197 \quad CF \text{ or } A = aW^b \quad (3)$$

198 where CF = Concentration Factor, A = remaining activity (%), a = constant, W = body weight
199 (g, wet wt) and b = allometric coefficient. Kinetic parameters can be correlated positively ($b >$
200 0), negatively ($b < 0$).

201 Best fitting regression models were selected according to highest determination coefficient
202 and examination of residuals. The level of significance for statistical analyses was always set
203 at $\alpha=0.05$.

204 2.3.3. Bioaccumulation model

205 The relative contribution of dissolved and food pathways was determined using a
206 bioaccumulation model (Landrum et al., 1992; Thomann, 1981; Thomann et al., 1995) as
207 modified by Hédouin et al. (2010b). Briefly, the total activity of radionuclides in the
208 organisms is equal to the sum of each activity resulting of the two different uptake pathways
209 (Eq. 4):

$$210 \quad C_t = C_{w,ss} + C_{f,ss} \quad (4)$$

211 where $C_{w,ss}$ is the radiotracer steady-state activity (Bq g^{-1}) accumulated in bloody cockles
212 from dissolved phase (Eq. 5) and $C_{f,ss}$ is the radiotracer steady-state activity (Bq g^{-1} wet wt)
213 accumulated in cockles via the food (Eq. 6):

$$214 \quad C_{w,ss} = \frac{(A_{0l} k_u C_w)}{k_{el,w}} \quad (5)$$

215 where C_w is the radionuclide activity in seawater (Bq g^{-1} wet wt), A_{0l} is the absorption

216 efficiency (%) of the radionuclide from seawater, k_u is the uptake rate constant (d^{-1}) from
217 seawater and $k_{el,w}$ is the efflux rate constant (d^{-1}) after the exposure through seawater.

$$218 \quad C_{f,ss} = \frac{(AE \ IR \ C_f)}{k_{el,f}} \quad \text{with } C_f = \frac{Kd_f}{C_w} \quad (6)$$

219 where C_f is the radionuclide activity in food ($Bq \ g^{-1}$ wet wt), Kd_f is the partition coefficient for
220 phytoplankton, AE is the assimilation efficiency (%) of the radiotracer ingested with food, IR
221 is the ingestion rate ($g \ g^{-1} \ d^{-1}$, wet wt basis), $k_{el,f}$ is the efflux rate constant (d^{-1}) after the
222 dietary exposure.

223 The relative contribution (%) of each uptake pathway can thus be estimated, using the
224 relation:

$$225 \quad \%food = \frac{C_{f,ss}}{C_{w,ss} + C_{f,ss}} \times 100 \quad (7)$$

$$226 \quad \%seawater = \frac{C_{w,ss}}{C_{w,ss} + C_{f,ss}} \times 100 \quad (8)$$

227 In addition to the parameters obtained from each biokinetic equation, ingestion rate (IR) of
228 bloody cockles of two different weight classes was experimentally assessed (see Metian et al.,
229 2008). On average, IRs were $0.053 \ g \ g^{-1} \ d^{-1}$ and $0.032 \ g \ g^{-1} \ d^{-1}$ wet wt basis for the “small”
230 and the “big” size classes, respectively (Table 1). Kd_f values used in the calculations were 140
231 ± 70 for ^{134}Cs (IAEA, 2014; Pouil et al., 2015) and $2.10^5 \pm 1.10^4$ for ^{241}Am (Fisher et al.,
232 1983; IAEA, 2014).

233

234 **3. Results and Discussion**

235 Body size and body weight are factors that are well known to affect trace element and
236 radionuclides bioconcentrations in organisms (e.g. Boyden, 1974; Hédouin et al., 2006;
237 Strong and Luoma, 1981; Warnau et al., 1995). In the light of this, biomonitoring programs
238 must take into account the size of the organisms used.

239 The present work investigated the effect of body weight on the bioaccumulation of
240 radionuclides (^{134}Cs and ^{241}Am) in the bloody cockles *A. senilis* exposed via dissolved and
241 trophic pathways. The approach followed in this study is somewhat different from the one
242 usually carried out to study allometric effect. Indeed, this latter is usually investigated only for
243 the dissolved pathway, although the trophic pathway can be the major pathway for
244 contaminant bioaccumulation in bivalves (Hédouin et al., 2010b; Metian et al., 2016; Pouil et
245 al., 2015).

246 Significant allometric relationships were found in bloody cockles after a 28-d exposure period
247 to ^{134}Cs and ^{241}Am (Fig. 1A). Inverse relationships between ^{134}Cs and ^{241}Am CFs and bivalve
248 weights displayed quite high determination coefficients ($R^2 = 0.51$ and 0.80 , respectively,
249 Table 1), indicating a decrease in radionuclide uptake efficiency with increasing individual
250 weight. In contrast, no allometric relationship was found in bloody cockles during the
251 depuration period whether after dissolved or trophic exposure (Fig. 1B and 2). However, for
252 ^{241}Am , the remaining activities appeared to be higher in the larger individuals after feeding
253 experiment (Fig. 2), suggesting a retention of assimilated ^{241}Am positively related to the
254 weight of the bivalve. Nevertheless, further investigations using a higher number of
255 individuals are needed to statistically confirm this trend.

256 Bivalves showing greater accumulation of dissolved radionuclides in small than larger
257 individuals have been described for ^{134}Cs (e.g. in the green mussel *Perna viridis*; Ke et al.,
258 2000) as well as for ^{241}Am (e.g. in the Mediterranean mussel *Mytilus galloprovincialis*;
259 Güngör et al., 2001). Nevertheless, in the same mussel species, Güngör et al. (2001) found no
260 allometric relationship for ^{134}Cs . Decreasing bioaccumulation capacity of dissolved
261 radiotracers in larger/older individuals can be related to size-dependent gill surface area to
262 volume ratio, as well as to physiological differences of the organisms such as higher
263 metabolic demands in smaller organism with consequent higher fluid filtration (Strong and

264 Luoma, 1981; Pan and Wang, 2008; McKinley et al., 2012; Tang et al., 2017). In the present
265 study, the degree of allometric coefficient (b values) for the dissolved exposure experiment
266 varied between the two studied radioelements: Cs and Am recorded -0.44 and -0.35,
267 respectively (Table 1). This indicates that the effect of body weight of ^{241}Am on the cockles is
268 not as strong as its effect on ^{134}Cs .

269 Although literature exists on the influence of size, weight or age on the capacity of
270 radionuclide uptake from the dissolved phase, the present study also investigated the
271 influence of the weight of bivalves on their capacity to retain radionuclides after their
272 exposure to ^{134}Cs and ^{241}Am from dissolved and food pathways. Results showed that this
273 capacity did not differ significantly between the two weight classes. This seems similar to
274 what Melinda et al. (2015) observed about the release of Cs by different weights groups of
275 humpback grouper fish *Cromileptes altivelis*. Lee et al. (1998) also reported that efflux rates
276 of Cd, Cr and Zn were poorly influenced by bivalve weight in the species *Potamocorbula*
277 *amurensis* and *Macoma balthica*.

278 There seems to be a wide gap in knowledge about the bioaccumulation of elements
279 (radionuclides or metals) in aquatic species commonly found in coastal African waters, in
280 particular about the incidence of weight-related effect on ^{134}Cs and ^{241}Am bioaccumulation.
281 The relative contribution of each exposure pathway to the whole-body bioaccumulation of
282 ^{134}Cs and ^{241}Am in the bloody cockles of two different classes of weight was calculated using
283 the kinetic parameters determined from the experiments (see Table 2) and other parameters
284 such as the K_{df} of ^{134}Cs and ^{241}Am in phytoplankton ($1.4 \cdot 10^2 \pm 0.7 \cdot 10^2$ and $2.0 \cdot 10^5 \pm 0.1 \cdot 10^5$
285 respectively; Fisher et al., 1983; IAEA, 2014; Pouil et al., 2015) and the ingestion rates ($IR =$
286 $0.053 \pm 0.011 \text{ g g}^{-1} \text{ d}^{-1}$ and $0.032 \pm 0.006 \text{ g g}^{-1} \text{ d}^{-1}$ for the “small” and the “big” cockles
287 respectively; present study).

288 Results of the computations indicate that the food pathway was the major contributor (74-

289 97%) to the whole-body bioaccumulation of ^{134}Cs in the two classes of weights (Fig. 3). Since
290 ^{241}Am is differently taken up (seawater) and assimilated (food) by the cockles of the two
291 different weight classes (Table 2), the relative contribution of each pathway is weight-
292 dependent for this radionuclide although the uncertainties associated with the estimates are
293 high (Fig. 3). Thus, seawater seems to be the major pathway of accumulation of ^{241}Am in the
294 “small” cockles, whereas food dominates the bioaccumulation of ^{241}Am in the “big” cockles
295 (Fig. 3). Ingested particles as the main uptake pathways of Cs in invertebrates is, in many
296 cases, quite well established (e.g. Børretzen and Salbu, 2009; Metian et al., 2016; Pouil et al.,
297 2015). For Am, only few experimental studies were carried out and it is reported that > 80%
298 of the whole-body bioaccumulation is derived from food in the blue mussel *M. edulis* (Wang
299 et al., 1996). The current study, which is the first one that examined the influence of weight
300 on ^{241}Am , showed that the latter relative contribution should be further studied, taking into
301 account weight classes, to ensure it is actually true for bigger organisms, which could have
302 substantial impact on results interpretation of environmental survey campaigns.

303

304 **4. Conclusions**

305 The present study indicates that if the bloody cockle is used as a bioindicator for radionuclide
306 monitoring in coastal marine environments, focus on small individuals would be
307 recommended as their bioconcentration capacity is higher than bigger individuals. However,
308 results would mainly reflect the contamination situation in the food of cockles for Cs, whereas
309 it would inform on dissolved contamination for Am. Using concomitantly larger individuals
310 would allow obtaining information on Am contamination in their food as well.

311

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462 Captions to figures

463

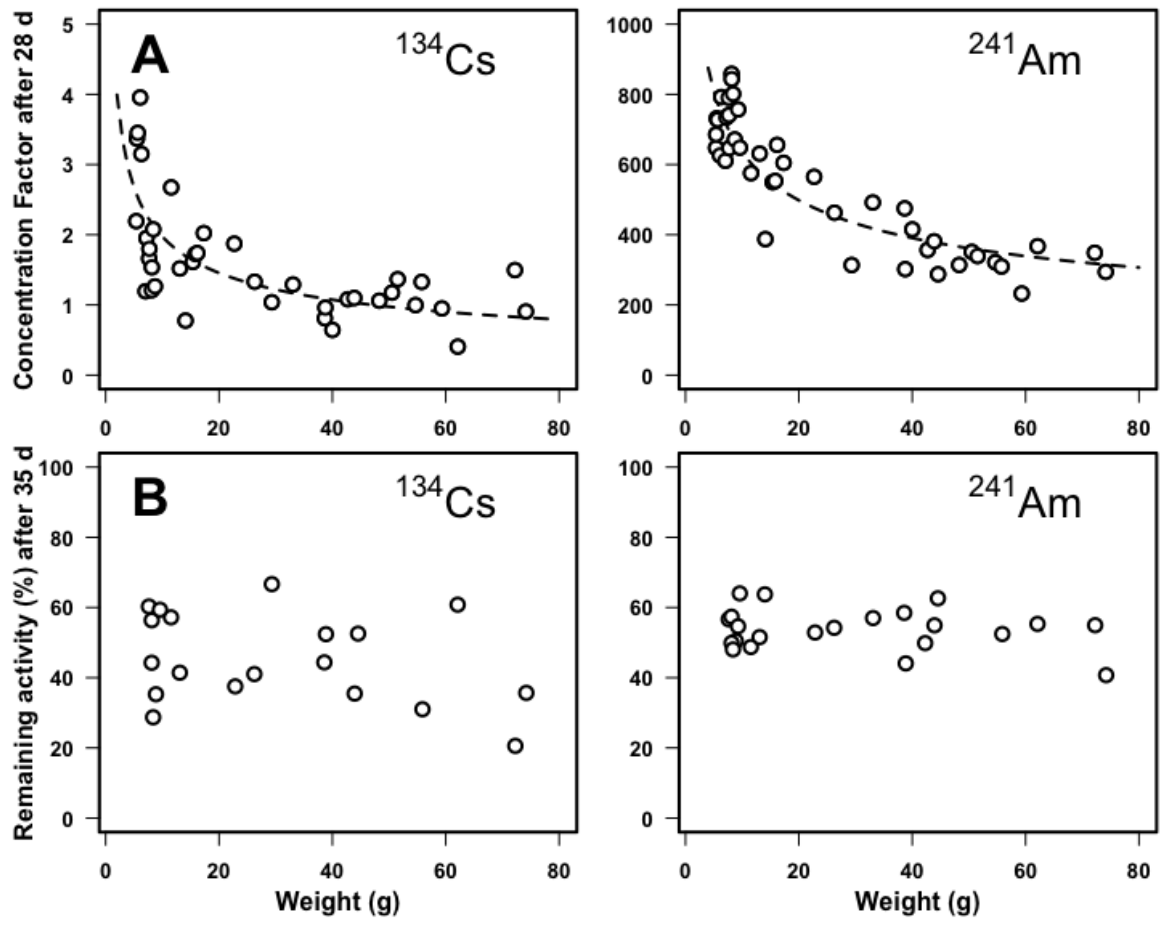
464 Figure 1. Relationships between (A) Concentration Factor (CF), (B) remaining activity and
465 whole-body weight (g) of bloody cockles *A. senilis* after a dissolved exposure to radionuclides
466 (^{134}Cs & ^{241}Am). Each dot represents values measured in one individual (n=19-43). Equations
467 best fitting the relationships are given in Table 1.

468

469 Figure 2. Relationships between remaining activity and whole-body weight (g) of *A. senilis*
470 after a trophic exposure to radionuclides (^{134}Cs & ^{241}Am). Each dot represents values
471 measured in on individual (n=14).

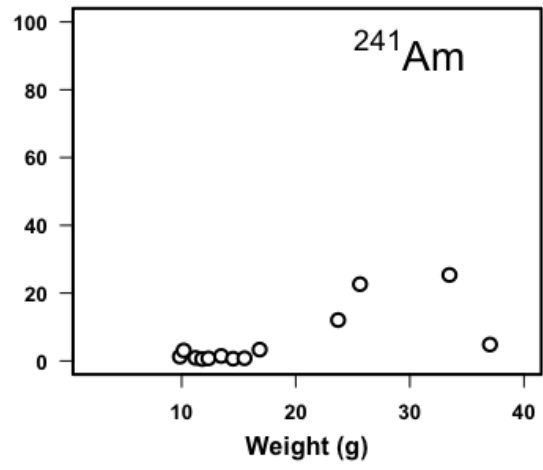
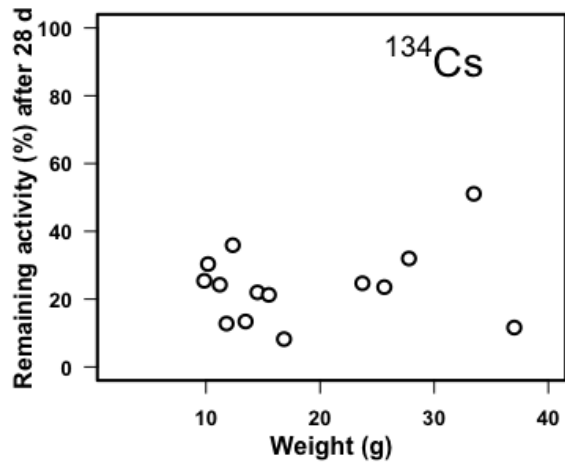
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473 Figure 3. Relative contributions (%) of the uptake pathways (seawater and food) to the total
474 bioaccumulation of radionuclides (^{134}Cs & ^{241}Am) in the bloody cockles of two different
475 classes of weights. Values of parameters used are given in Table 2.



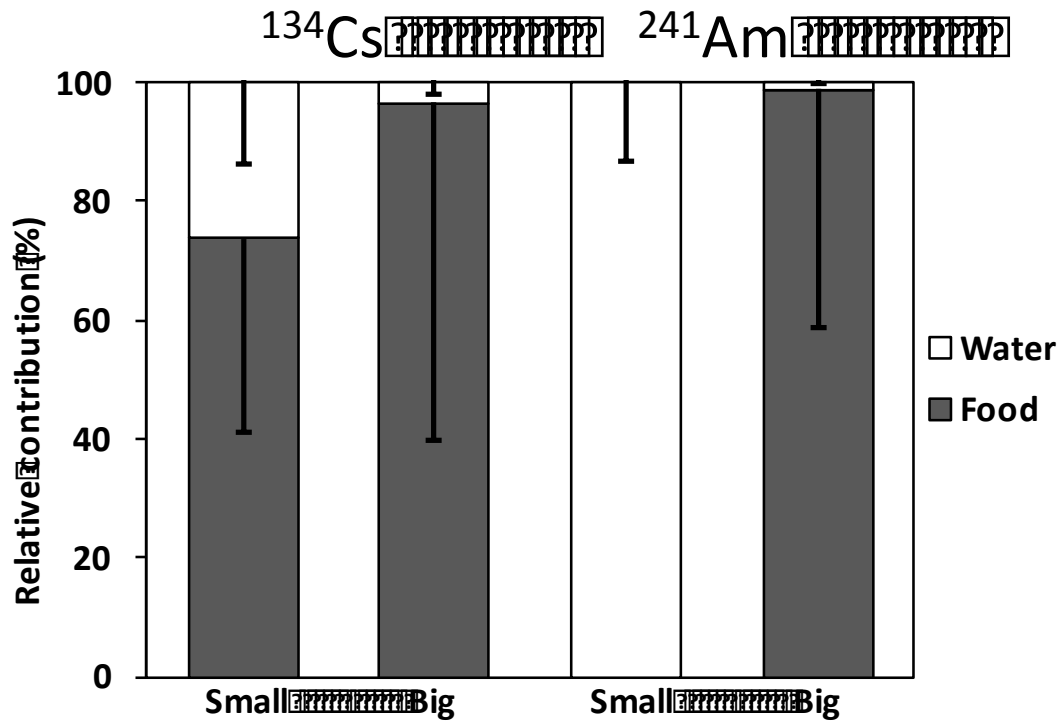
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477 Figure 1



478

479 Figure 2



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481 Figure 3

482

483 Table 1. Parameters and statistics of the power function (CF or Remaining activity =
484 $a \cdot \text{weight}^b$) best describing the relationships between concentration factor (CF) of
485 radionuclides (^{134}Cs and ^{241}Am) and whole-body wet weight (g) in bloody cockle *A. senilis* at
486 the end of the 28-d dissolved exposure period (n=38-43).

Radionuclide	a (Mean \pm ASE)	$p(a)$	b (Mean \pm ASE)	$p(b)$	R^2
^{134}Cs	5.41 \pm 1.11	<0.0001	-0.44 \pm 0.08	<0.0001	0.51
^{241}Am	1422.19 \pm 111.26	<0.0001	-0.35 \pm 0.03	<0.0001	0.80

487

488

489 Table 2. Numeric values of physiological and geochemical factors used in modeling the
 490 radionuclide (^{134}Cs & ^{241}Am) transfer in the bloody cockles of two different classes of
 491 weights.

Dissolved pathway				
	C_w (Bq g ⁻¹ , Mean ± SD)	A_{ol} (%, Mean ± ASE)	k_u (d ⁻¹ , Mean ± ASE)	k_{el} (d ⁻¹ , Mean ± ASE)
“Small” weights (9.8±2.2 g)				
^{134}Cs	0.94 ± 0.05	76.26 ± 4.91	0.59 ± 0.16	0.010 ± 0.003
^{241}Am	0.13 ± 0.01	92.18 ± 0.85	63.75 ± 4.42	0.018 ± 0.001
“Big” weights (36.3±6.7 g)				
^{134}Cs	0.94 ± 0.05	66.83 ± 0.02	0.49 ± 0.14	0.015 ± 0.003
^{241}Am	0.13 ± 0.01	92.50 ± 1.01	34.81 ± 2.55	0.019 ± 0.001
Trophic pathway				
	Kdf (Mean ± ASE)	AE (%, Mean ± ASE)	IR^* (g g ⁻¹ d ⁻¹ , Mean ± SD)	k_{el} (d ⁻¹ , Mean ± ASE)
“Small” weights (12.2±1.9 g)				
^{134}Cs	140 ± 7	36.88 ± 2.70	0.053 ± 0.011	0.021 ± 0.005
^{241}Am	200000 ± 10000	0**	0.032 ± 0.006	0**
“Big” weights (28.8 ± 5.1 g)				
^{134}Cs	140 ± 7	31.10 ± 11.14	0.053 ± 0.011	0.002
^{241}Am	200000 ± 10000	27.75 ± 5.56	0.032 ± 0.006	0.012

492 * IR were experimentally assessed according to Metian et al. (2008)

493 ** Values not significant different from 0 (p>0.05).

494