



HAL
open science

The invasive crayfish *Cherax quadricarinatus* facing chlordecone in Martinique: Bioaccumulation and depuration study

Thomas Baudry, Eric Gismondi, Jean-Pierre Goût, Alexandre Arqué, Juliette Smith-Ravin, Frédéric Grandjean

► To cite this version:

Thomas Baudry, Eric Gismondi, Jean-Pierre Goût, Alexandre Arqué, Juliette Smith-Ravin, et al.. The invasive crayfish *Cherax quadricarinatus* facing chlordecone in Martinique: Bioaccumulation and depuration study. *Chemosphere*, 2022, 286, Part 3, pp.131926. 10.1016/j.chemosphere.2021.131926 . hal-04063266

HAL Id: hal-04063266

<https://hal.science/hal-04063266>

Submitted on 21 Jun 2023

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

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

**The invasive crayfish *Cherax quadricarinatus* facing chlordecone in Martinique:
bioaccumulation and depuration study**

Thomas Baudry¹⁻²⁻³⁻⁴, Eric Gismond⁵, Jean-Pierre Goût¹, Alexandre Arque², Juliette Smith-Ravin⁴ & Frédéric Grandjean³

¹ DEAL Direction de l'Environnement, de l'Aménagement et du Logement

Route de la Pointe de Jaham - BP7212, Schoelcher 97274, Martinique

² ODE Office De l'Eau, 7 Avenue Condorcet, Fort-de-France, Martinique

³ Laboratoire Ecologie et Biologie des Interactions, UMR CNRS 7267 Equipe Ecologie Evolution Symbiose, 5 rue Albert Turpin, Poitiers Cedex, France

⁴ Groupe de recherche BIOSPHERES, Université des Antilles, campus de Schoelcher, Martinique, F.W.I.

⁵ University of Liège, Laboratory of Animal Ecology and Ecotoxicology (LEAE) - Freshwater and Oceanic sciences Unit of reSearch (FOCUS), Chemistry Institute, Bât. B6C, 11 allée du 6 Août, B-4000 Sart-Tilman, Belgium

Corresponding author: Thomas Baudry, thmsbaudry@gmail.com

1 **Abstract**

2 The redclaw crayfish, *Cherax quadricarinatus*, was introduced to Martinique Island
3 for aquaculture purposes at the beginning of the 21st century, in an attempt to revitalize the
4 freshwater crustacean aquaculture sector. Mainly due to its high economical value, it was
5 intentionally released in the wild and was caught and sold by fishermen. Martinican rivers are
6 polluted by chlordecone, considered as one of the worst Persistent Organic Pollutants (POP).
7 Despite its dangerousness, it was used until 1993 in the French West Indies against a banana
8 pest and was always found in the ecosystems. This study aimed to investigate the level of
9 contamination in the muscle of crayfish caught in the wild, as well as the potential of
10 bioconcentration and depuration in the *C. quadricarinatus* muscle. This study could allow us
11 to quantify the risk for consumers but also, to evaluate a depuration process to reduce the risk
12 related to its consumption. Using both *in-vitro* and *in-situ* experiments, results highlighted the
13 importance of the chlordecone concentration in the water and the time of exposure to the
14 pollutant. The bioconcentration seems to be very quick and continuous in crayfish muscle, as
15 chlordecone can be detectable as early as 6 hours of exposure, whatever the concentration
16 tested. Finally, it appears that, even after 20 days of depuration in chlordecone-free water,
17 chlordecone concentrations remained higher to the residual maximum limit (i.e. 20 ng/g wet
18 weight), concluding that the decontamination of the muscle seems not very efficient, and the
19 risk for the Martinican people could be serious.

20

21

22 **Keywords:** *Aquatic ecotoxicology, Decapoda, French West Indies, Invasive species, POP*

23

24

25

26 1. Introduction

27

28 Worldwide aquaculture development is known to be a pathway for the introduction of non-
29 native species to new countries (FAO, 2016). It can lead to the transfer of organisms into new
30 ecosystems facilitated by humans especially when sold alive and at a high price. It can result
31 in serious aquatic invasions threatening native aquatic ecosystems (Filipová et al., 2013).

32 The Australian redclaw crayfish *Cherax quadricarinatus*, is a good example as it was
33 translocated to many territories in tropical and sub-tropical areas (references in Baudry et al.,
34 2020) for aquaculture, due to its high reproductive capacity and rapid growth (Jones, 1990;
35 Patoka et al., 2014; Azofeifa-Solano et al., 2017). In Martinique, a tense ecological situation
36 led to the closure of many aquaculture facilities raising the shrimps *Macrobrachium*
37 *rosenbergii*. In fact, most of the soils where this species was produced became contaminated
38 with chlordecone (CLD). Only one farm was able to survive by exporting its activity to a
39 smaller site, making the production of *M. rosenbergii* unprofitable and forcing the farmer to
40 find an alternative (Baudry et al., 2020). *Cherax quadricarinatus* was finally introduced to the
41 island in the early 2000s, from Cuba, in order to reinvigorate this sector (Baudry et al., 2020).
42 Following intentional releases, crayfish were found in the natural environment and, thanks to
43 their high dispersal capacity and its high tolerance of environmental quality, were able to
44 colonize a large part of the island's river systems (Baudry et al., 2021).

45

46 In the international toxicological context, several studies have shown that invasive
47 species have a higher absorption potential than native ones, particularly in the presence of
48 abiotic stressors (Fedorenkova et al., 2013; Pedro et al., 2017). For example, Pedro et al.
49 (2017) show a higher concentration of POPs in non-native fish species capelin and sandlance
50 (*Mallotus villosus* and *Ammodytes spp.*) than in the native Arctic cod (*Boreogadus saida*).

51 Weir and Salice (2012) show the higher resistance of the snail species *Melanoides*
52 *tuberculatus* to malathion compared to the native *Biomphalaria glabrata*. In bryozoans, large
53 drops in biomass, often with no recovery, are observed in native species exposed to heavy
54 metals, whereas in non-native species, an increase in biomass is sometimes even observed
55 (Crooks et al., 2011).

56

57 French West Indies soils and rivers are contaminated by chlordane (CLD)
58 ($C_{10}Cl_{10}O$; CAS number 143-50-0) (Coat et al., 2006; Lafontaine et al., 2017), considered to
59 be the worst Persistent Organic Pollutant (POP) for these areas. It was widely used to
60 eliminate the banana weevil (*Cosmopolites sordidus*), considered the most important banana
61 crop pest (Serge, 1994; Cabidoche and Lesueur-Jannoyer, 2012). The hazards of this pollutant
62 are known and led to a refusal to register it by the Toxics Commission and the ban of its use
63 in 1977 by the United States (Cannon et al., 1978). Indeed, chlordane is well known to be
64 an endocrine disruptor and could also be responsible for other diseases such as prostate
65 cancer, for example (Multigner et al., 2007, 2010). In France, CLD was only prohibited in
66 1990 and the French West Indies obtained a 3-year derogation, until 1993, waiting to find an
67 alternative to this pesticide (Cabidoche et al., 2009; Cabidoche and Lesueur-Jannoyer, 2012).

68

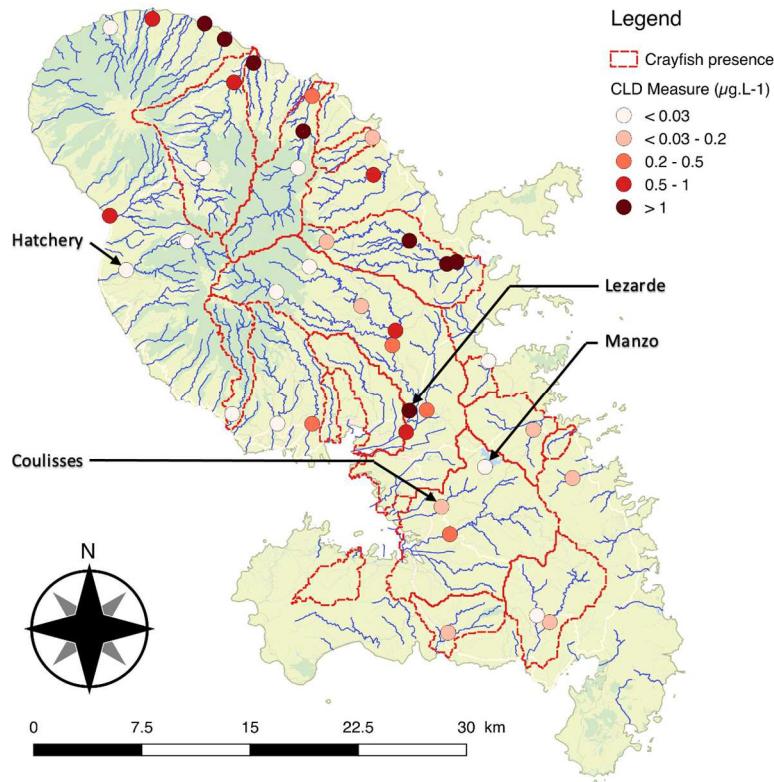
69 Like most pesticides, CLD can be bioaccumulated along trophic chains (Bahner et al.,
70 1977; Hansen et al., 1977). This is rather worrying given that aquatic ecosystems are well
71 known to be the final receptacle of these toxic molecules through soil leaching and/or erosion
72 processes (Snegaroff, 1977; Coat et al., 2006; Cabidoche and Lesueur-Jannoyer, 2012).
73 Because of its very high persistence in soils, CLD is still present today in soils and rivers of
74 Martinique (Merlin et al., 2016) and also in biota. Indeed, several studies highlighted a
75 positive correlation between CLD concentrations in the water and CLD concentrations

76 measured in organisms (Hansen et al., 1977) with potentially very high bioconcentration
77 factors (Bahner et al., 1977). This highlights the dangerousness of the situation in Martinique
78 given that *C. quadricarinatus*, having a preference for slow-moving waters, often located
79 downstream and therefore receiving a large dose of pollutants, is highly likely to accumulate
80 substantial doses of CLD. Highly appreciated by local fishermen, *C. quadricarinatus* is fished
81 in the wild and resold at high prices on the markets or to restaurants, but this could raise a
82 major human health problem in the population, given the high doses of chlordecone found in
83 the water at some sites, especially in the North of Martinique (Figure 1). CLD concentrations
84 in Martinican rivers can be very variable (Figure 1). The figure 1 exposed the variability in
85 terms of CLD concentration in the water. This variability can be explained, for example, by
86 different CLD amount used in the past by each banana plantation owner, known to be largely
87 located in the North. This observation could also be due to the type of soil which plays a role
88 in the CLD retention and depending on the location of the different plantations, from north to
89 south, the soil composition is not the same. Indeed, Cabidoche et al. (2009) modeled the
90 persistence of chlordecone in different type of soil and highlighted that CLD is more
91 persistent in andosol than nitisol. Finally, CLD concentrations measured in rivers are also
92 dependent to the rainfall, responsible for soil leaching, and the North of the island is known to
93 be more exposed to the rainfalls. As the problem is destined to persist over time, given that
94 CLD has a very long persistence in the environment, the French health authorities have set a
95 Maximum Residual Limit (MRL), above which products are considered consumable, which is
96 at 20 ng.g⁻¹ wet weight.

97

98 Therefore, the aims of this study were to (i) investigate the presence of CLD in the
99 abdominal muscle (part consumed by humans) of *C. quadricarinatus* collected from three
100 Martinican rivers according to a contamination gradient (biomonitoring), then (ii) determine

101 the kinetics of bioconcentration of CLD in *C. quadricarinatus* muscle under controlled
102 laboratory conditions to understand the bioconcentration mechanism and finally, (iii)
103 investigate a potential decontamination of the crayfish before its consumption, to assess its
104 risk as a food product.



105
106 Figure 1: Map showing the presence of *Cherax quadricarinatus* in Martinique and the stations
107 where the chlordecone concentration in water is known. Colored points correspond to these
108 stations, darker points indicate more polluted stations. Invaded watersheds by *Cherax*
109 *quadricarinatus* are highlighted by red dotted lines, from data in Baudry et al. (2021).

110

111 2. Material and method

112 2.1. *Is CLD bioaccumulated in wild crayfish?*

113 Three sites, known to be contaminated by chlordecone, and a control site
114 (uncontaminated) were selected to investigate the potential of *C. quadricarinatus* to
115 accumulate CLD.

116 The four sites were selected according to a CLD-gradient of contamination, representing
117 the main CLD concentrations measured in Martinique rivers where crayfish populations were
118 observed (Baudry et al., 2021). These four sites were: a non-contaminated site, corresponding
119 to the hatchery (CLD concentration in the water < limit of detection (LOD)), a low
120 contaminated site “Manzo” (below quantification limit $0.03 \mu\text{g.L}^{-1}$), an intermediate station
121 on the Couliesses stream (annual mean $0.45 \mu\text{g.L}^{-1}$) and a highly contaminated station, on the
122 Lezarde river (annual mean $2.48 \mu\text{g.L}^{-1}$). At each sampling site, five male crayfish were
123 collected, between 10 and 14 cm (i.e. adult stage), weighed and measured. Only males were
124 analyzed to avoid possible bias in analysis due to the reproductive cycle in females
125 (Lafontaine et al., 2017). After sampling, abdominal muscle was dissected in laboratory,
126 weighed and stored at -20°C until analysis.

127

128 *2.2. What are the CLD bioconcentration and depuration kinetics in crayfish in laboratory*
129 *conditions?*

130 *2.2.1. Kinetics of bioconcentration*

131 A total of 180 crayfish males were provided by a hatchery in Carbet (Martinique), a CLD-
132 free area, according to results of ‘Office de l’Eau de Martinique’ (ODE). They were all
133 between 10 and 14 cm body length and between 15 and 59 grams (the most common size
134 range in the Martinique rivers, *Personal observation*) and were transferred to the laboratory of
135 IFREMER, Le Robert, Martinique. As for *in-situ* experiments, only males were used, to avoid
136 bias due to reproduction cycle in females.

137 Thirty 20L aquaria were filled with CLD-free water. Crayfishes were acclimated, for
138 seven days before experiments, under constant oxygenation aeration and a natural light/dark

139 photoperiod. Water temperature and pH were maintained at $27^{\circ}\text{C} \pm 0.11$ and 7.72 ± 0.28
 140 corresponding to optimal values for *C. quadricarinatus* development (de Moor, 2002; Tropea
 141 et al., 2010). Crayfish were fed daily with pieces of carrot.

142 Exposure was carried out using two chlordecone conditions: $0.2 \mu\text{g.L}^{-1}$, $2 \mu\text{g.L}^{-1}$, chosen
 143 for their environmental relevance in rivers of Martinique, and a control (containing 0.01%
 144 acetone, corresponding to the use of prepared stock solutions of CLD in acetone).
 145 Intermediate solutions were prepared with two successive dilutions from CLD (GC purity
 146 area $\geq 98\%$; Sigma-Aldrich, USA): first, a 1 g.L^{-1} CLD stock solution was prepared with
 147 100% acetone and then, a 10 mg.L^{-1} CLD solution was prepared with water (therefore
 148 containing 1% acetone), used in exposure media preparation. The real CLD concentrations of
 149 exposure were measured at the Laboratoire Territorial d'Analyse (LTA) of Martinique (Table
 150 1).

151 Table 1: Comparison of nominal values of the different treatments run during the experiments
 152 with the mean measured values, performed by the Laboratoire Territorial d'Analyses de
 153 Martinique. One water sampling per aquarium, in 3 different aquaria (3 replicates) per
 154 condition was done.

Treatment	Nominal value ($\mu\text{g.L}^{-1}$)	Mean value ($\mu\text{g.L}^{-1}$) \pm Standard deviation
Control	0	$< 0.01^*$
C1	0.2	0.23 ± 0.03
C2	2	2.75 ± 0.21
1 day-depuration	-	0.02 ± 0.003
3 day-depuration	-	0.06 ± 0.02

155 $*0.01 \mu\text{g.L}^{-1}$ representing the Limit of Detection of chlordecone.

156 Ten aquaria per condition were used (six individuals per aquarium). As for the
 157 acclimation period, crayfishes were fed daily with pieces of carrot, small enough (around

158 0.125 cm³) to be eaten quickly and avoid a possible adsorption of pollutant. According to
159 Lafontaine et al. (2017), to keep a constant CLD concentration in water during all the
160 experiments, it is necessary to perform a 7-day period to saturate the aquariums' walls before
161 exposure, and renew the exposure medium every 96h.

162 To analyze the CLD concentration in *C. quadricarinatus*, five individuals per condition
163 were randomly sampled in aquariums after six exposure times: 6h, 1day, 2d, 6d, 12d and 20d.
164 The abdominal muscle was dissected, weighed and conserved at -20°C until analysis.

165 2.2.2. Kinetics of depuration

166 At the end of the bioconcentration experiment (i.e. 20 days), the remaining individuals
167 (from 0.2 and 2 µg.L⁻¹ exposure) were rinsed thoroughly with CLD-free water (to avoid
168 possible traces of CLD adsorbed on the cuticle) and transferred to different aquaria filled with
169 CLD-free water, to analyze a possible decontamination. Experimental conditions in terms of
170 feeding, photoperiod and physico-chemical values were the same as for the bioconcentration
171 kinetics experiment. Moreover, to avoid a possible recontamination of individuals, water was
172 renewed every 3 days. Five crayfishes per condition were collected after 1, 2, 8 and 20 days
173 of depuration. The abdominal muscle was dissected, weighed and stored at -20°C until CLD
174 analysis. Only for aquaria where crayfish initially exposed to 2 µg.L⁻¹ of CLD were placed,
175 the CLD concentration was measured at day 1 and day 3 to quantify a possible CLD release in
176 the depuration water (Table 1).

177 2.3. Chlordecone concentration analysis

178 CLD concentration in *C. quadricarinatus* muscles was analyzed according to the protocol
179 detailed in Lafontaine et al. (2017). Briefly, muscles were freeze-dried and an average of 250
180 mg of lyophilized samples were used to CLD extraction, which was performed in Accelerated

181 Solvent Extraction (ASE) with a mixture of n-hexane:dichloromethane (90:10; v:v; Biosolve-
182 Chimie, France). After fat fraction extraction, samples were evaporated, resuspended in 2 mL
183 hexane, and purified with a volume of 2 mL 98% sulfuric acid, used to remove organic
184 matter. Then, a 3-min centrifugation at 3000 rpm allowed recovery of the organic layer, which
185 was evaporated under gentle nitrogen flow. Finally, the extract was resuspended with 50 μ L
186 n-hexane and 50 μ L of a solution PCB 209 (100 pg/ μ L in n-hexane) as an injection internal
187 injection standard.

188 A procedural blank and a Quality Control were analysed alongside the samples. The
189 procedural blank consisted of ASE extraction without biological matrix, while the Quality
190 Control consisted of ASE extraction with biological matrix (i.e. *Cherax quadricarinatus*
191 muscle, CLD-free) spiked with CLD solution to obtain a final concentration of 2.5 ng.g⁻¹ wet
192 weight.

193 CLD concentrations were analyzed in GC-ECD (Thermo Scientific, USA) using the
194 parameters described in Lafontaine et al. (2017). The limit of detection (LOD) was fixed at
195 0.02 ng/g wet weight (i.e. three times the background noise of chromatogram) and the limit of
196 quantification was 0.06 ng/g wet weight. All CLD concentrations were expressed as ng.g⁻¹
197 wet weight.

198

199 2.4. CLD uptake and depuration rate constants

200 The toxicokinetic model was done according to the OECD 305 Fish Bioconcentration
201 Guidelines (OECD, Test No. 305), with some adaptations because our experimental design
202 does not allow consideration of the fecal and gill elimination constants. Thus, to estimate the
203 uptake rate constant (k_1) and the depuration rate constant (k_2), we assumed that CLD

204 bioconcentration followed a first order kinetics. With this hypothesis, a linear model was
205 obtained, and the depuration rate constant (k_2) correspond to the slope of the equation (1):

$$206 \quad \ln(CLD_{Cq}) = -k_2 \cdot t + c$$

207 with $\ln(CLD_{Cq})$ the natural logarithm of the CLD amount in *C. quadricarinatus* muscle
208 (ng), k_2 the depuration rate constant and c the intercept, corresponding to the natural logarithm
209 of the CLD concentration in *C. quadricarinatus* muscle ($\text{ng}\cdot\text{g}^{-1}$) at the end of the accumulation
210 phase (i.e. start of the depuration phase). As the experiment time was relatively short (40 days
211 in total), we considered no growth variation, and therefore the depuration rate constant was
212 not corrected for organism growth. The uptake rate constant (k_1) can be estimated from k_2
213 with this equation (2):

$$214 \quad C_{Cq} = C_w \cdot \frac{k_1}{k_2} \cdot (1 - e^{-k_2 \cdot t})$$

215 with C_{Cq} the concentration of CLD in the organism ($\text{ng}\cdot\text{g}^{-1}$), C_w the concentration of CLD
216 in the exposure medium ($\mu\text{g}\cdot\text{L}^{-1}$) and t the time in days.

217 Finally, the bioconcentration factor (BCF) can be estimated using the equation (3):

$$218 \quad BCF = \frac{k_1}{k_2}$$

219 Finally, the depuration rate constant (k_2) was used to determine the time of depuration phase
220 necessary to reach (or be) under the MRL (i.e. $20 \text{ ng}\cdot\text{g}^{-1}$) in the muscle of *C. quadricarinatus*.

221 For that, estimation was performed with the equation (4):

$$222 \quad C_{Cq} = C_{Cq0} \cdot e^{-k_2 \cdot t}$$

223 where C_{Cq} equals 20 ng.g^{-1} and C_{Cq0} corresponding to the mean CLD concentration in
224 the muscle tissue at the end of the bioconcentration phase (i.e. $15.91 \pm 7.44 \text{ ng.g}^{-1}$ for the C1
225 treatment and $91.19 \pm 20.73 \text{ ng.g}^{-1}$ for the C2 treatment). Similarly, half-life of CLD in the
226 muscle tissue was calculated, for each condition using the equation (5):

$$227 \quad T_{1/2} = \ln(2)/k_2$$

228 with $T_{1/2}$ corresponding to the half-life of CLD and k_2 the depuration rate constant.

229 From the equation 4 and using k_2 of each exposure conditions, theoretical values of CLD
230 concentration at each sampling time of depuration can be calculated. For that, C_{Cq0} correspond
231 to the mean CLD concentration measured in the crayfish muscle at the end of the
232 accumulation phase (91.19 ng.g^{-1} for $2 \mu\text{g/L}$ exposure and 15.91 ng.g^{-1} for $0.2 \mu\text{g/L}$
233 exposure), k_2 to the depuration rate constant for each condition (0.007 for $0.2 \mu\text{g/L}$ exposure
234 and 0.03 for $2 \mu\text{g/L}$) and t to each sampling time, during the depuration experiment. Thus,
235 these theoretical values can be compared with the measured ones.

236 2.5. Statistical analysis

237 Statistical analyses were all performed using RStudio v1.1.463 (Core Team R
238 Development, 2019). Normality and homogeneity of variance were tested with Shapiro and
239 Bartlett tests ($p > 0.05$). The effect of the exposure time and CLD concentrations of exposure
240 on the concentration measured in *C. quadricarinatus* muscle were analyzed with a two-way
241 ANOVA, with a significant probability less than 0.05. Kruskal-Wallis test was performed on
242 k_1 values, for each treatment, between each exposure time, to visualize a possible significative
243 influence of the exposure time on the uptake rate constant. The same tests were used on the
244 CLD concentration, to highlight the presence of different statistical groups between each
245 treatment, at each exposure time. After that, each group were compared using a posthoc

246 Kruskal test (Dunn's-test). A Generalized Linear Model (GLM) was used to investigate the
247 correlation between the CLD concentration in the water (*in-situ*) and the CLD concentration
248 in the crayfish tissues but also the influence of biotic parameters (body length and total
249 weight) on the CLD bioconcentration in muscle. Student t-test and an Analyze of Variance
250 were utilized to test respectively the difference of BCF between in lab-conditions and *in-situ*
251 conditions.

252

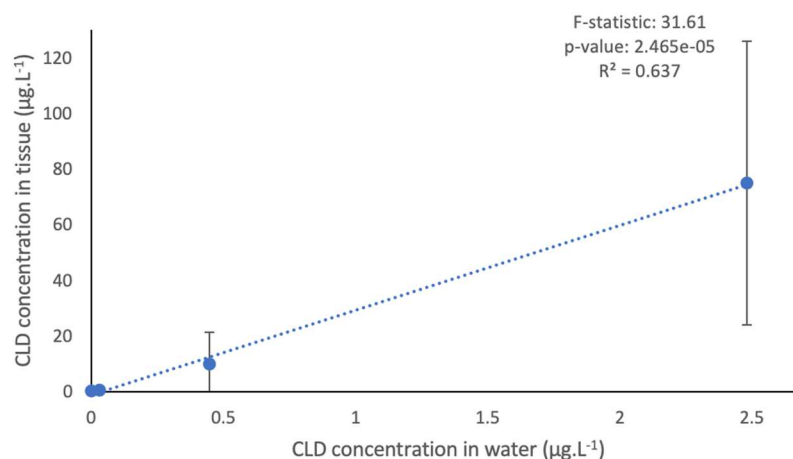
253 **3. Results**

254 *3.1. CLD accumulation in crayfish wild population*

255 A significant positive correlation was observed between the CLD concentration in water
256 and the CLD concentration accumulated in *C. quadricarinatus* muscle of wild populations (R^2
257 = 0.64; p-value < 0.001) (Figure 2).

258 In the hatchery, where no CLD concentration was measured (i.e. below the LOD), the
259 CLD concentration in the *C. quadricarinatus* muscle reached $0.13 \pm 0.12 \text{ ng.g}^{-1}$ while, at the
260 most CLD-polluted station (i.e. Lezarde, with $2.48 \mu\text{g.L}^{-1}$ of CLD concentration all year
261 round), the CLD concentration in the organism muscles reached $74.9 \pm 51.01 \text{ ng.g}^{-1}$ (with a
262 bioconcentration factor of 30.21 ± 20.58). At the intermediate stations, Manzo (below
263 quantification limit $0.03 \mu\text{g.L}^{-1}$) and Coulisses (mean $0.45 \mu\text{g.L}^{-1}$ all year round), the CLD
264 concentrations measured in *C. quadricarinatus* muscles were $0.65 \pm 0.62 \text{ ng.g}^{-1}$ ($21.52 \pm$
265 20.71 of bioconcentration factor) and $9.82 \pm 11.65 \text{ ng.g}^{-1}$ (22 ± 26.12 of bioconcentration
266 factor), respectively (Table 2; Figure 2). No significant difference was observed between the
267 BCF estimated for each of the three contaminated stations ($F = 0.4$; p-value = 0.54).

268



269
 270 Figure 2: Changes in CLD concentration (ng.g^{-1}) in the *Cherax quadricarinatus* muscle
 271 (captured in the wild) depending on CLD concentration measured in the river water ($\mu\text{g.L}^{-1}$).
 272
 273

274 Table 2: Bioconcentration factor (L kg^{-1}) of chlordecone for *in lab* experiments and for *in situ*
 275 conditions, calculated at the end of the 20 days-experiment for the lab conditions (i.e. 0.23
 276 $\mu\text{g.L}^{-1}$ and 2.75 $\mu\text{g.L}^{-1}$) and from individuals (five per site) caught in the four reference
 277 stations in the wild (i.e. Long-term exposure experiment: Hatchery, Manzo, Coulisses and
 278 Lezarde). Letters in the statistical analysis represent the significant difference between each
 279 condition (Student t-test for *in-lab* experiments: $t = 2.59$, $p\text{-value} = 0.04$; and ANOVA for *in*
 280 *situ* experiments: $F = 0.4$; $p\text{-value} = 0.54$).

	<i>In lab</i>		<i>In situ</i>			
			Hatchery	Manzo	Coulisses	Lezarde
Measured CLD concentration	0.23 $\mu\text{g.L}^{-1}$	2.75 $\mu\text{g.L}^{-1}$	< LOD	< 0.03 $\mu\text{g.L}^{-1}$	0.45 $\mu\text{g.L}^{-1}$	2.48 $\mu\text{g.L}^{-1}$
BCF	560.3 ± 464.3 ^A	67.4 ± 29.9 ^B	-	21.52 ± 20.71 ^A	22 ± 26.12 ^A	30.21 ± 20.58 ^A

281

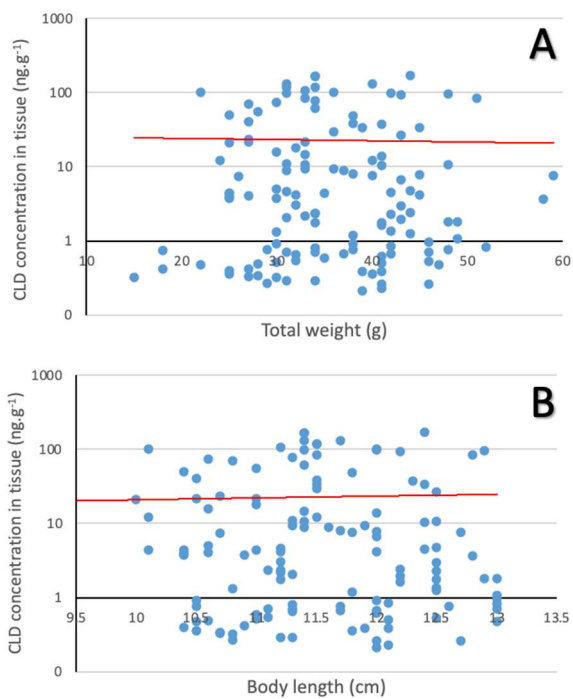
282 3.2. Short term exposure (lab experiment)

283 3.2.1. Kinetics of bioconcentration

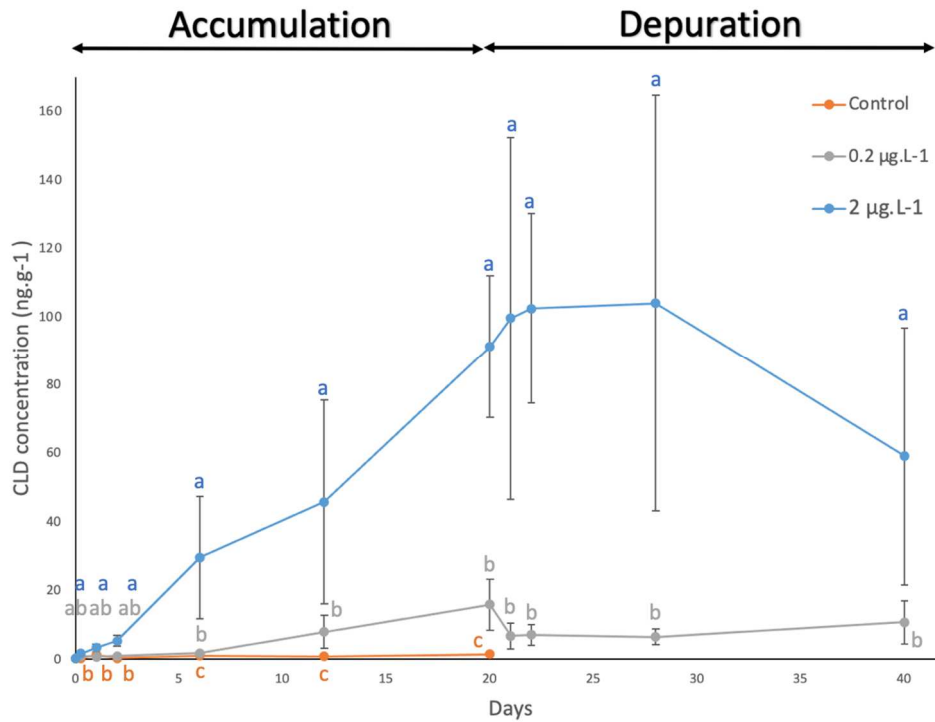
284 The real CLD concentrations, measured in the exposure medium, were close to the
285 nominal values: $0.23 \pm 0.03 \mu\text{g.L}^{-1}$ against $0.2 \mu\text{g.L}^{-1}$ and $2.75 \pm 0.21 \mu\text{g.L}^{-1}$ against $2 \mu\text{g.L}^{-1}$
286 (Table 1). As expected, no CLD was detected in the control condition.

287 The CLD concentration bioaccumulated in *C. quadricarinatus* muscle was significantly
288 influenced by both time of exposure and CLD concentration of exposure (Table 3) but not by
289 body length of crayfish (R^2 : 0.007, p-value: 0.75) nor by crayfish weight (R^2 : -0.007, p-
290 value: 0.84) (Figure 3). At the end of exposure (20 days), the mean CLD concentrations
291 measured were $15.91 \pm 7.44 \text{ ng.g}^{-1}$ for $0.2 \mu\text{g.L}^{-1}$ condition and $91.19 \pm 20.73 \text{ ng.g}^{-1}$ for 2
292 $\mu\text{g.L}^{-1}$ treatment (Figure 4). The CLD concentration in muscle of the crayfish control group
293 reached $1.47 \pm 1 \text{ ng.g}^{-1}$ (Figure 4). Results highlighted also that CLD concentrations
294 measured in muscle were over the LOD and LOQ from 6 hours of exposure, regardless the
295 exposure condition (Figure 4).

296



297
298 Figure 3: Influence of total weight (A) and total body length (B) of the crayfish on CLD
299 concentration measured in the muscle of *Cherax quadricarinatus*, using a Generalized Linear
300 Model (GLM). Neither of the two parameters seems to be significant (respectively for A: R^2 :
301 -0.007, p-value: 0.84 and B: R^2 : 0.007, p-value: 0.75).



302
 303 Figure 4: Accumulation and depuration phases of CLD in the *Cherax quadricarinatus* muscle
 304 for the two exposure conditions tested ($0.2 \mu\text{g.L}^{-1}$ chlordecone exposition condition in orange
 305 and $2 \mu\text{g.L}^{-1}$ in grey). Depuration was done by placing crayfish in CLD-free water during 20
 306 days with a renewal of the water every 3 days. Letters represent the statistical differences
 307 between each condition at each time, following Kruskal-Wallis and posthoc tests.

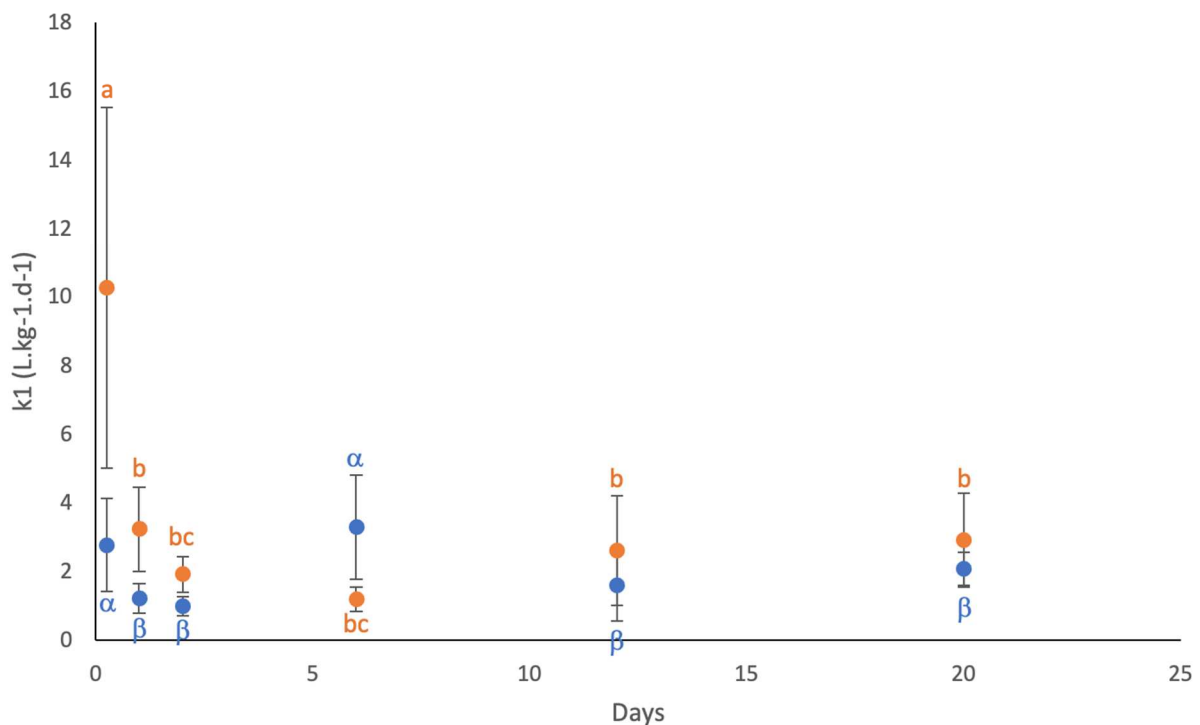
308

309 Table 3: Two-way Analysis of Variance (ANOVA) investigating the effects of exposure time
 310 to CLD (Time) and the CLD concentration of exposure (Medium) on the CLD concentration
 311 bioaccumulated in *Cherax quadricarinatus* muscle.

Source of variation	Sum of square	d.f.	F	p-Values
Time	20472	5	31.88	< 0.001
Medium	10319	1	80.36	< 0.001
Time x Medium	10550	5	16.43	< 0.001
Residuals	5650	44		

312

313 Values of k_1 (uptake rate constant) were calculated at each sampling time for both
 314 exposure conditions (Figure 5). For the $0.2 \mu\text{g.L}^{-1}$ treatment, the maximal k_1 value ($10.27 \pm$
 315 $2.93 \text{ L kg}^{-1} \text{ d}^{-1}$) was observed after 6h of CLD exposure, then it decreased to reach a plateau
 316 around $3 \text{ L kg}^{-1} \text{ d}^{-1}$ until the end of the exposure (Figure 5). For the $2 \mu\text{g.L}^{-1}$ treatment, the
 317 same trend was observed, except that the maximal k_1 value ($3.29 \pm 0.38 \text{ L kg}^{-1} \text{ d}^{-1}$) was
 318 estimated for the 6th day of exposure (Figure 5).



319 Figure 5: Variation of the constant accumulation rate (k_1) for *Cherax quadricarinatus*,
 320 calculated from the constant elimination rate k_2 , over time, for the $0.2 \mu\text{g.L}^{-1}$ CLD exposition
 321 condition (orange dot) and the $2 \mu\text{g.L}^{-1}$ chlordecone exposition condition (blue dot). Letters
 322 represents the different statistical groups between each exposure time, for each condition.
 323
 324

325 BCF values, at the end of the absorption phase, showed that individuals exposed to the
 326 lowest CLD concentration (i.e. $0.2 \mu\text{g.L}^{-1}$) had the highest bioconcentration factor: $560.3 \pm$
 327 464.3 and 67.4 ± 29.9 for the $0.2 \mu\text{g.L}^{-1}$ and the $2 \mu\text{g.L}^{-1}$ treatments, respectively (Table 2)
 328 highlighting a significant difference between the two conditions ($t = 2.59$, $p\text{-value} = 0.04$).

329

330 3.2.2. Depuration

331 Depuration rate constants (k_2) reached respectively 0.007 days^{-1} (for $0.2 \mu\text{g.L}^{-1}$ condition)
332 and 0.03 days^{-1} (for $2 \mu\text{g.L}^{-1}$ condition).

333 After 20 days of depuration, a very low decrease of CLD concentration in muscle was
334 observed for crayfish exposed at $0.2 \mu\text{g.L}^{-1}$: the CLD concentration reached $10.8 \pm 6.26 \text{ ng.g}^{-1}$
335 ¹, corresponding to a 32.12 % loss (CLD concentration at the end of bioconcentration period
336 was $15.91 \pm 7.44 \text{ ng.g}^{-1}$) (Figure 4). This maximum time of depuration tested seemed to
337 reduce the CLD concentration in muscle, even though it was already below the MRL at the
338 end of the exposure time.

339 For *C. quadricarinatus* exposed to $2 \mu\text{g.L}^{-1}$, no decrease in CLD concentration was observed
340 during the six first days. Then, a slow decrease was measured and CLD concentration reached
341 $59.18 \pm 37.47 \text{ ng.g}^{-1}$ at the end of the decontamination period, corresponding to a 35.1 % loss
342 (CLD concentration at the end of the bioconcentration period was $91.19 \pm 20.73 \text{ ng.g}^{-1}$)
343 (Figure 4). This CLD concentration at the end of the depuration phase was always higher than
344 the MRL, however, according to calculations from uptake and elimination constants, 50.57
345 days seems necessary to reach the MRL (20 ng.g^{-1}) in muscle of crayfish exposed for 20 days
346 at $2 \mu\text{g.L}^{-1}$.

347 The half-life of CLD in the muscle tissue is 99 days for the $0.2 \mu\text{g.L}^{-1}$ condition and 23.1 days
348 for the $2 \mu\text{g.L}^{-1}$ condition.

349

350 For the $0.2 \mu\text{g.L}^{-1}$ of CLD exposure condition, the mean measured values of CLD in the
351 crayfish muscle are below the theoretical values at each depuration time, with an increase of
352 the CLD concentration at Day 20 (6.8 ng.g^{-1} at Day 1 and 10.1 ng.g^{-1} at Day 20 for the
353 measured values vs 15.8 ng.g^{-1} at Day 1 and 13.8 ng.g^{-1} at Day 20) (Table 4).

354 For the 2 $\mu\text{g.L}^{-1}$ of CLD exposure condition, the mean measured values raise until day 8 (82.9
 355 ng.g^{-1} at Day 1 to 103.9 ng.g^{-1} at Day 8) and are above the theoretical values for the Day 2 and
 356 Day 8 of depuration. After that the mean CLD measured values decrease until Day 20 and
 357 reach 59.8 ng.g^{-1} , close to the theoretical value calculated (50.1 ng.g^{-1}) (Table 4).

358

359

360 Table 4: Comparison of theoretical values of CLD concentration in crayfish muscle at each
 361 sampling time during the depuration experiment, for each condition, based on calculations
 362 with the equation 4 and the respective k_2 values, with the measured values.

	0.2 $\mu\text{g.L}^{-1}$ of CLD exposure condition		2 $\mu\text{g.L}^{-1}$ of CLD exposure condition	
	Theoretical values (ng.g^{-1})	Measured values (ng.g^{-1})	Theoretical values (ng.g^{-1})	Measured values (ng.g^{-1})
Day 1	15.8	6.8	88.5	103.9
Day 2	15.7	7.1	85.9	82.9
Day 8	15	6.5	71.7	59.8
Day 20	13.8	10.1	50.1	59.8

363

364 Finally, the CLD concentration measurements in water, after 1 and 3 days of
 365 depuration (for crayfish initially exposed to 2 $\mu\text{g.L}^{-1}$ of CLD), revealed an increase of CLD in
 366 the water over time: $0.02 \pm 0.003 \mu\text{g.L}^{-1}$ at Day 1 and $0.06 \pm 0.02 \mu\text{g.L}^{-1}$ at Day 3 (Table 1).

367

368 **4. Discussion**

369 *4.1. CLD in wild crayfish populations*

370 The CLD concentration analysis in muscle of *C. quadricarinatus* wild population
371 highlighted a significant positive correlation between the CLD concentration in river water
372 and the CLD concentration bioaccumulated in organisms. This trend is consistent with several
373 studies showing bioaccumulation of CLD in aquatic organisms like fish or crustaceans. Coat
374 et al. (2006) showed CLD concentrations ranging from 196 ng.g⁻¹ and 386 ng.g⁻¹ in the whole
375 body of *Oreochromis mossambicus* sampled in the Lezarde river in Martinique. Monti (2007)
376 highlighted more impressive CLD bioaccumulation rates in aquatic organisms sampled in the
377 Grande Anse river (Guadeloupe, French West Indies), contaminated at 1.715 µg.L⁻¹ CLD:
378 from 1568 ng.g⁻¹ to 4002 ng.g⁻¹ for the crustaceans *Atya innocuous* and *Xiphocaris elongata*,
379 respectively, and up to 11733 ng.g⁻¹ for the freshwater fish *Eleotris perniger*. These higher
380 CLD bioconcentrations, as compared to the present study, seem to be due to the fact that CLD
381 analysis was carried out in the whole-body organisms and not only using the muscle as in our
382 study. This assumption was confirmed by Lafontaine et al. (2017) who investigated the CLD
383 distribution in the prawn *M. rosenbergii*. The authors observed that the CLD was mainly
384 distributed in the hepatopancreas (i.e. detoxification tissue) (between 50 and 65% of the total
385 CLD amount) and less in the muscle (between 15 and 20% of the total CLD amount).

386 4.2. CLD behaviour under in-vitro conditions

387 The laboratory approach was carried out to better evaluate the CLD toxicokinetic
388 mechanism in the invasive crayfish *C. quadricarinatus*. Results highlighted that although
389 CLD concentration in control conditions was below the LOD, a low concentration of CLD
390 was measured in the muscle (1.47 ± 1 ng g⁻¹), thus confirming a much higher BCF calculated
391 for a lower CLD concentration exposure. The same result was observed in the CLD analysis
392 in wild populations, where *C. quadricarinatus* sampled in Manzo (CLD concentration < 0.03
393 µg/L) were contaminated by measurable CLD concentration. These results confirmed the fact

394 that the analysis of xenobiotic bioconcentration in biota could reinforce the biomonitoring of
395 aquatic ecosystem contaminations. Besides, a rapid CLD absorption in *C. quadricarinatus*
396 muscle happened from the first exposure time (6 hours), whatever the condition (i.e. 0.2 $\mu\text{g.L}^{-1}$
397 ¹ and 2 $\mu\text{g.L}^{-1}$). This is consistent with the fact that the higher k_1 was estimated at the
398 beginning of the bioconcentration phase, as also observed in the prawn *M. rosenbergii*
399 (Lafontaine et al., 2017). Even if these results are in accordance with other studies, they must
400 be considered with cautious due to the fact that the steady-state, recommended for a BCF
401 calculation, was not reached in our study.

402 The kinetic results showed a significant positive influence of exposure time and CLD
403 concentration on the CLD bioaccumulated in the muscle. These findings are in accordance
404 with other studies, such as Lafontaine et al. (2017) who highlighted a quick accumulation of
405 CLD (measurable as early as 6 hours), in the prawn *M. rosenbergii* whole body, and a slow-
406 down after 4 days of exposure: the uptake rate is lower after this time (and so, the
407 accumulation of pollutant) but the saturation (plateau phase) seems to be not reached
408 Lafontaine et al. (2017), as for *C. quadricarinatus* in our study. These results could be
409 explained by the fact that the muscle bioconcentrates CLD slower than other organs like the
410 hepatopancreas. Even if the plateau seems not be reached, we reached CLD concentration
411 close to those measured in the muscle of crayfish caught in the wild, exposed to CLD for
412 longer. This leads us to believe that this accumulation occurs throughout life without reaching
413 stabilization or that this plateau phase would have been reached with a longer exposure.
414 Relatively similar trends were found in terrestrial vertebrates: a great and quick assimilation
415 of CLD was observed in goat and in cows matrices (Lastel et al., 2016; Fournier et al., 2017
416 and references therein). Concerning the depuration of goat matrices studied by Lastel et al.
417 (2016), contaminated with CLD at 1 mg.kg^{-1} body weight, a steady-state was observed after
418 the accumulation phase and so, a delay before an efficient elimination of the CLD, fitting with

419 our results. Interestingly, a relatively close half-life was found between our two experiments
420 (20 days vs 23.1 days). Contrarily, for the muscle of beef, contaminated with CLD at 1 mg.kg⁻¹
421 body weight, the half-life of CLD reach 43 days. The accumulation and depuration were
422 both linked to the physiological status and the metabolic capacities explaining why certain
423 taxonomic groups accumulate more efficiently the CLD through their feeding habits and
424 eliminate more or less efficiently the pollutants than other (via lactation or eggs laying for
425 example) (Lastel et al., 2016 and references therein).

426 Interestingly, results obtained in the laboratory experiment highlighted a similar pattern to
427 that of wild individuals, showing a dose-dependent accumulation of CLD (i.e. higher CLD
428 concentration in muscles of crayfish sampled in rivers highly contaminated) : 15.91 ± 7.44
429 ng.g^{-1} for $0.2 \mu\text{g.L}^{-1}$ treatment against $9.82 \pm 11.65 \text{ ng.g}^{-1}$ for Coulisses river (mean 0.45
430 $\mu\text{g.L}^{-1}$ all year) and $91.19 \pm 20.73 \text{ ng.g}^{-1}$ for $2 \mu\text{g.L}^{-1}$ treatment against $74.9 \pm 51.01 \text{ ng.g}^{-1}$ for
431 Lezarde river ($2.48 \mu\text{g.L}^{-1}$ of CLD concentration all year).

432 Higher levels of CLD contamination were expected in specimens caught *in-situ*. Indeed,
433 absorption via both water and food, contaminated at a lower level, is known to be more
434 efficient (Bahner et al., 1977). This is accentuated in the case of our study, as like the majority
435 of invasive crayfish, *C. quadricarinatus* is omnivorous and opportunistic, capable of
436 predated on macroinvertebrates, and therefore, placed quite high in the trophic chain
437 (Johnston et al., 2011; Jackson et al., 2017) and so, becoming more contaminated than other
438 species at a lower level. Hoekstra et al. (2003) highlighted that large mammals were more
439 contaminated than fish, themselves more contaminated than copepods. Although laboratory
440 experiments highlighting only CLD bioconcentration from water, the CLD concentration
441 measured in crayfish caught *in-situ* seems to be more realistic. This slight difference, less
442 contaminated individuals in the wild, can be explained by the fact that the toxicokinetic

443 constants remains more difficult to estimate in the natural environment because it is necessary
444 to take into account many environmental factors (season, temperature, dissolved oxygen,
445 turbidity, predation, cocktail effect of different pollutants ...) (Rotchell and Ostrander, 2003).

446

447 Some studies highlighted the possibility for invertebrates to eliminate the CLD (more
448 or less quickly) from their own body, partly via their faeces, moulting (Bahner et al., 1977) or
449 other unknown mechanisms (Schimmel et al., 1977; Lafontaine et al., 2017). Here, we
450 highlighted values of CLD concentration in the crayfish just below the MRL for individuals
451 exposed to $0.2 \mu\text{g.L}^{-1}$ for 20 days ($15.91 \pm 7.44 \text{ ng.g}^{-1}$ at the end of the 20 days-absorption
452 phase against 20 ng.g^{-1} for the MRL). But those exposed to higher concentrations were
453 contaminated above this value. Unfortunately, a very slow CLD decontamination was
454 observed in the *C. quadricarinatus* muscle for the $2 \mu\text{g.L}^{-1}$ treatment, or even an absence of
455 decontamination for the $0.2 \mu\text{g.L}^{-1}$ treatment. Based on the depuration constant k_2 determined
456 in this work (0.03 days^{-1}), 50.57 days of depuration seem necessary to reach the MRL in
457 muscle of crayfish exposed for 20 days at $2 \mu\text{g.L}^{-1}$. This can be explained by the fact that the
458 muscle was not very efficient at eliminating the CLD. This time was not possible to be
459 estimated in organisms exposed to $0.2 \mu\text{g.L}^{-1}$ because CLD concentration at the end of the
460 bioconcentration period was already below the MRL. Indeed, previous studies showing CLD
461 depuration, used the entire body, including the hepatopancreas, known to play an important
462 role in the accumulation, detoxification of pollutants in crustaceans (Lafontaine et al., 2017)
463 and their elimination (Sreeram and Menon, 2005; Zeng et al., 2010). The depuration observed
464 in *M. rosenbergii* by Lafontaine et al. (2017) may be mainly due to the detoxification of CLD
465 by the hepatopancreas. However, other detoxification system could explain this observation
466 such as the MXR (e.g. multi-xenobiotics resistance mechanism) pump system, protecting
467 aquatic species against pollutant molecules (Kurelec, 1992), as already investigated in other

468 aquatic invertebrates (i.e. bivalves and crustaceans) (Pain and Parant, 2007; Horion et al.,
469 2015). This MXR system could explain the fact that CLD concentration decreased in the
470 organism and increased in the water over time during the depuration phase, until the renewal
471 at day 3. This release of CLD into the depuration water reinforces the idea that it is not
472 sufficient to simply place the organisms in clean water to allow them to be decontaminated
473 within a defined period, but that it is necessary to renew the medium at least every three days,
474 to avoid possible recontamination of exposed individuals. Another explanation can be a
475 contamination due to traces of pollutant present on the crayfish cuticle after the accumulation
476 period but all the precautions were taken (i.e. thorough washing of crayfish before the transfer
477 in the CLD-free for the depuration period). However, this depuration system seems to be
478 ineffective at quickly reaching the MRL in crayfish muscle. In fact, it cannot be excluded that
479 the presence of CLD in the depuration water could led to a recontamination of the organisms,
480 especially because our results highlight that crayfish didn't reach the steady-state of
481 bioconcentration, and we observed an increase of the CLD muscle concentration measured
482 even during the first time of depuration. This suggests that CLD released in water, even if at a
483 very low concentration, could be uptake again by organisms. This hypothesis is supported by
484 the *in-situ* experiment showing a CLD bioconcentration in crayfishes sampled in the Manzo
485 river, having a similar CLD concentration as the depuration water (i.e. 0.03 ng.L⁻¹). This
486 could therefore also explain the low estimated k_2 after the depuration phase.

487 However, as our results doesn't really show a depuration, these theoretical calculations could
488 be more accurate in the case of a longer depuration which could show a real decrease of CLD
489 in the body. However, the muscle is not a detoxifying organ and so, we are not sur that a
490 longer depuration time would result in a depuration of CLD from the muscle and the CLD
491 concentration could remain the same in the long term.

492

493 *4.3. Risks for consumers*

494 This rapid and high CLD accumulation in combination with an inefficient CLD
495 decontamination of the muscle could pose a human health problem: even if fishing in the
496 Martinican rivers is prohibited, this recreational and ancestral practice remains very difficult
497 to control. Many fishermen put traps in the rivers every day (*Personal observation*) to catch
498 crustaceans for their personal consumption or to sell them on the markets. Moreover, this
499 practice is also very lucrative: *C. quadricarinatus* can be sold for up to 25€ / kg as it is
500 considered a luxury gourmet product. This species is now found and fished in high densities
501 in several martinican rivers (Baudry et al., 2021). The majority of rivers inhabit by *C.*
502 *quadricarinatus* are polluted by CLD beyond the potability thresholds set by the Regional
503 Health Agency (0.1 µg.L⁻¹) or even well above 0.2 µg.L⁻¹, which is an exposure concentration
504 resulting in the MRL being exceeded in crayfish muscle, according to our study (Figure 1).

505 Due to the transfer of CLD along the food chain, CLD can thus reach humans through the
506 consumption of contaminated products. Even if a recent study, based on the analysis of CLD
507 contamination before and after several cooking processes, highlighted a strike reduction of the
508 amount of CLD in beef meat (Martin et al., 2020), exposure to CLD is now chronic and it is
509 detectable in the blood of the majority of the Caribbean population (Multigner et al., 2007).
510 This could generate a health problem since it has been shown that CLD could be responsible
511 to the development of diseases, such as prostate cancer (Multigner et al., 2007). The
512 possibility of cooking products previously decontaminated by a simple depuration period, as
513 studied in this work, could have been an interesting alternative to the consumption of crayfish.
514 But the fact that the muscle of *C. quadricarinatus* is only slowly decontaminated in CLD-free
515 water means that other methods must be tested to allow the safe consumption of these

516 products: a decontamination period longer than 20 days, in a tank, with potential losses of
517 specimens due to cannibalism or inconvenience in an artificial closed environment (leading to
518 mortality), is likely not to be followed by fishermen. At the moment, the most efficient way of
519 stopping the spread of this poison is to inform the population on the dangers of this wild
520 fishery. This social aspect is important as the problem is likely to persist for a very long time
521 as modelling analyses have predicted that CLD in soils would persist for at least 300 years in
522 Martinique (Cabidoche et al., 2009).

523

524 **5. Conclusion**

525 This study aimed to investigate the level of CLD contamination in *C. quadricarinatus*
526 muscle of wild populations and determine the CLD toxicokinetic (i.e. **bioconcentration** and
527 depuration) through *in-vitro* experiments in order to investigate the processes and to
528 communicate about an alternative to consume these fishery products. Here, we showed a clear
529 positive correlation between the CLD concentration in water and the CLD concentration
530 measured in the muscle of crayfish caught in the wild. These results were in accordance with
531 *in-vitro* experiments: a quick (as early as 6 hours of exposure) and efficient **bioconcentration**
532 over time, reaching approximately 100 ng.g⁻¹ of CLD for crayfish exposed to 2 µg.L⁻¹ of
533 CLD and approximately 20 ng.g⁻¹ (corresponding to the MRL) for crayfish exposed to 0.2
534 µg.L⁻¹ was observed. Worryingly, we were not able to decontaminate the muscle of
535 contaminated *C. quadricarinatus*: almost 3 weeks of decontamination did not reduce the
536 concentration of CLD below the MRL for crayfish exposed 2 µg.L⁻¹ CLD. The crayfish
537 resource is inexhaustible in the natural environment as this invasive species seems not to be
538 affected by this abiotic stressor, making the situation alarming: *C. quadricarinatus* is highly
539 prized in Martinique and chlordecone, highly present in a majority of Martinique watersheds,

540 is well known to cause significant health problems. Depuration experiments have shown that
541 the muscle seems to decontaminate too slowly to propose this as an alternative for the
542 consumption of products caught in the Martinique rivers.

543

544 **Acknowledgement**

545 We thank the Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER) for
546 welcoming us into their structure in order to carry out the experiments and the Laboratoire
547 Territorial d'Analyses (LTA) de Martinique for having carried out the river water analyses.
548 We also thank Catherine Adam, technician at the Laboratory of Animal Ecology and
549 Ecotoxicology (LEAE) of the University of Liege, for the CLD analyses in the crayfish
550 tissues.

551 We thank Julian Reynolds for reading the article and correcting the English.

552 This project was funded by the Direction de l'Environnement, de l'Aménagement et du
553 Logement de Martinique (DEAL), the Office de l'Eau de Martinique (ODE) and the
554 Collectivité Territoriale de Martinique (CTM).

555

556 **References**

557 Azofeifa-Solano, J.C., Naranjo-Elizondo, B., Rojas-Carranza, A.H., Cedeño-Fonseca, M.,
558 2017. Presence of the Australian redclaw crayfish *Cherax quadricarinatus* (von Martens,
559 1868) (Parastacidae , Astacoidea) in a freshwater system in the Caribbean drainage of
560 Costa Rica. *BioInvasions Records* 6, 351–355. <https://doi.org/10.3391/bir.2017.6.4.08>
561 Bahner, L.H., Wilson, A.J., Sheppard, J.M., Patrick, J.M., Goodman, L.R., Walsh, G.E., 1977.
562 Kepone® Bioconcentration, Accumulation, Loss, and Transfer Through Estuarine Food

563 Chains. *Chesapeake Science* 18, 299. <https://doi.org/10.2307/1350804>

564 Baudry, T., Becking, T., Goût, J.P., Arqué, A., Gan, H.M., Austin, C.M., Delaunay, C.,
565 Smith-Ravin, J., Roques, J.A.C., Grandjean, F., 2020. Invasion and distribution of the
566 redclaw crayfish, *Cherax quadricarinatus*, in Martinique. *Knowledge and Management*
567 *of Aquatic Ecosystems* 421. <https://doi.org/10.1051/kmae/2020041>

568 Baudry, T., Mauvisseau, Q., Goût, J., Arqué, A., Delaunay, C., Smith-ravin, J., Sweet, M.,
569 2021. Mapping a super-invader in a biodiversity hotspot , an eDNA-based success story.
570 *Ecological Indicators* 126. <https://doi.org/10.1016/j.ecolind.2021.107637>

571 Cabidoche, Y., Achard, R., Cattan, P., Clermont-dauphin, C., Massat, F., Sansoulet, J., 2009.
572 Long-term pollution by chlordecone of tropical volcanic soils in the French West Indies :
573 A simple leaching model accounts for current residue. *Environmental Pollution* 157,
574 1697–1705. <https://doi.org/10.1016/j.envpol.2008.12.015>

575 Cabidoche, Y.M., Lesueur-Jannoyer, M., 2012. Contamination of Harvested Organs in Root
576 Crops Grown on. *Pedosphere: An International Journal* 22, 562–571.
577 [https://doi.org/10.1016/S1002-0160\(12\)60041-1](https://doi.org/10.1016/S1002-0160(12)60041-1)

578 Cannon, S.B., Veazey, James M, Jackson, Robert S, Virlyn, W., Hayes, Carl, Straub, Wesley
579 E, Landrigan, Philip J, Liddle, J.A., Cdc, S.B., Veazey, J M, Jackson, R S, Burse, V.W.,
580 Hayes, C, Straub, W E, Landrigan, P J, Liddle, J.A., 1978. Epidemic kepone poisoning
581 in chemical workers. *American Journal of Epidemiology* 107, 529–537.

582 Coat, S., Bocquené, G., Godard, E., 2006. Contamination of some aquatic species with the
583 organochlorine pesticide chlordecone in Martinique. *Aquatic Living Resources* 19, 181–
584 187. <https://doi.org/10.1051/alr:2006016>

585 Core Team R Development, 2019. R: a language and environment for statistical computing.

586 Crooks, J.A., Chang, A.L., Ruiz, G.M., 2011. Aquatic pollution increases the relative success
587 of invasive species. *Biological Invasions* 13, 165–176. <https://doi.org/10.1007/s10530->

588 010-9799-3

589 De Moor, I., 2002. Potential impacts of alien freshwater crayfish in South Africa. *African*
590 *Journal of Aquatic Science* 125–139. <https://doi.org/10.2989/16085914.2002.9626584>

591 FAO, 2016. Fishery and Aquaculture Statistics 2016.

592 Fedorenkova, A., Vonk, J.A., Breure, A.M., Hendriks, A.J., Leuven, R.S.E.W., 2013.
593 Tolerance of native and non-native fish species to chemical stress: A case study for the
594 River Rhine. *Aquatic Invasions* 8, 231–241. <https://doi.org/10.3391/ai.2013.8.2.10>

595 Filipová, L., Petrusek, A., Matasová, K., Delaunay, C., Grandjean, F., 2013. Prevalence of the
596 crayfish plague pathogen *Aphanomyces astaci* in populations of the signal crayfish
597 *Pacifastacus leniusculus* in France: evaluating the threat to native crayfish. *PloS one* 8,
598 e70157. <https://doi.org/10.1371/journal.pone.0070157>

599 Fournier, A., Feidt, C., Lastel, M.L., Archimede, H., Thome, J.P., Mahieu, M., Rychen, G.,
600 2017. Toxicokinetics of chlordecone in goats: Implications for risk management in
601 French West Indies. *Chemosphere* 171, 564–570.
602 <https://doi.org/10.1016/j.chemosphere.2016.12.054>

603 Hansen, D.J., Goodman, L.R., Wilson, A.J., 1977. Kepone®: Chronic Effects on Embryo,
604 Fry, Juvenile, and Adult Sheepshead Minnows (*Cyprinodon variegatus*). *Chesapeake*
605 *Science* 18, 227. <https://doi.org/10.2307/1350865>

606 Hoekstra, P.F., O'Hara, T.M., Fisk, A.T., Borgå, K., Solomon, K.R., Muir, D.C.G., 2003.
607 Trophic transfer of persistent organochlorine contaminants (OCs) within an Arctic
608 marine food web from the southern Beaufort-Chukchi Seas. *Environmental pollution*
609 (Barking, Essex : 1987) 124, 509–22.

610 Horion, S., Thomé, J.P., Gismondi, É., 2015. Changes in antitoxic defense systems of the
611 freshwater amphipod *Gammarus pulex* exposed to BDE-47 and BDE-99. *Ecotoxicology*
612 24, 959–966. <https://doi.org/10.1007/s10646-015-1438-4>

613 Jackson, M.C., Evangelista, C., Zhao, T., Lecerf, A., Britton, J.R., Cucherousset, J., 2017.
614 Between-lake variation in the trophic ecology of an invasive crayfish. *Freshwater*
615 *Biology* 62, 1501–1510. <https://doi.org/10.1111/fwb.12957>

616 Johnston, K., Robson, B.J., Fairweather, P.G., 2011. Trophic positions of omnivores are not
617 always flexible: Evidence from four species of freshwater crayfish. *Austral Ecology*.
618 <https://doi.org/10.1111/j.1442-9993.2010.02147.x>

619 Jones, C., 1990. The biology and aquaculture potential of the tropical freshwater crayfish,
620 *Cherax quadricarinatus*. *Department of Primary Industries Queensland*.

621 Kurelec, B., 1992. The multixenobiotic resistance mechanism in aquatic organisms. *Critical*
622 *Reviews in Toxicology*. <https://doi.org/10.3109/10408449209145320>

623 Lafontaine, A., Baiwir, D., Joaquim-justo, C., Pauw, E. De, Lemoine, S., Boulangé-lecomte,
624 C., Forget-leray, J., Thomé, J., 2017a. Proteomic response of *Macrobrachium*
625 *rosenbergii* hepatopancreas exposed to chlordecone: Identification of endocrine
626 disruption biomarkers? *Ecotoxicology and Environmental Safety* 141, 306–314.
627 <https://doi.org/10.1016/j.ecoenv.2017.03.043>

628 Lafontaine, A., Gismondi, E., Dodet, N., Caupos, F., Lemoine, S., Lagadic, L., 2017b.
629 Bioaccumulation, distribution and elimination of chlordecone in the giant freshwater
630 prawn *Macrobrachium rosenbergii*: Field and laboratory studies. *Chemosphere* 185,
631 888–898. <https://doi.org/http://dx.doi.org/10.1016/j.chemosphere.2017.07.099>

632 Lastel, M.L., Lerch, S., Fournier, A., Jurjanz, S., Mahieu, M., Archimède, H., Feidt, C.,
633 Rychen, G., 2016. Chlordecone disappearance in tissues of growing goats after a one
634 month decontamination period—effect of body fatness on chlordecone retention.
635 *Environmental Science and Pollution Research* 23, 3176–3183.
636 <https://doi.org/10.1007/s11356-015-5833-4>

637 Martin, D., Lobo, F., Lavison-Bompard, G., Guérin, T., Parinet, J., 2020. Effect of home

638 cooking processes on chlordecone content in beef and investigation of its by-products
639 and metabolites by HPLC-HRMS/MS. *Environment International* 144, 106077.
640 <https://doi.org/10.1016/j.envint.2020.106077>

641 Merlin, C., Devers, M., Béguet, J., Boggio, B., Rouard, N., Martin-Laurent, F., 2016.
642 Evaluation of the ecotoxicological impact of the organochlorine chlordecone on soil
643 microbial community structure, abundance, and function. *Environmental Science and*
644 *Pollution Research* 23, 4185–4198. <https://doi.org/10.1007/s11356-015-4758-2>

645 Monti, D., 2007. Evaluation de la biocontamination en Chlordecone , β -
646 Hexachlorocyclohexane et Cadusaphos de Crustacés et Poissons d'eau douce en
647 Guadeloupe. *Ministère de l'Ecologie, du Développement et de l'Aménagement durables*
648 1–35.

649 Multigner, L., Cordier, S., Kadhel, P., Huc-Terki, F., Blanchet, P., Bataille, H., Janky, E.,
650 2007. Pollution par le chlordécone aux Antilles. Quel impact sur la santé de la
651 population? *Environnement, Risques et Sante* 6, 405–407.
652 <https://doi.org/10.1684/ers.2007.0121>

653 Multigner, L., Ndong, J.R., Giusti, A., Romana, M., Delacroix-Maillard, H., Cordier, S.,
654 Jégou, B., Thome, J.P., Blanchet, P., 2010. Chlordecone exposure and risk of prostate
655 cancer. *Journal of Clinical Oncology* 28, 3457–3462.
656 <https://doi.org/10.1200/JCO.2009.27.2153>

657 Pain, S., Parant, M., 2007. Identification of multixenobiotic defence mechanism (MXR)
658 background activities in the freshwater bivalve *Dreissena polymorpha* as reference
659 values for its use as biomarker in contaminated ecosystems. *Chemosphere* 67, 1258–
660 1263. <https://doi.org/10.1016/j.chemosphere.2006.11.017>

661 Patoka, J., Kalous, L., Kopecký, O., 2014. Risk assessment of the crayfish pet trade based on
662 data from the Czech Republic. *Biological Invasions* 16, 2489–2494.

663 <https://doi.org/10.1007/s10530-014-0682-5>

664 Pedro, S., Fisk, A.T., Tomy, G.T., Ferguson, S.H., Hussey, N.E., Kessel, S.T., McKinney,
665 M.A., 2017. Mercury and persistent organic pollutants in native and invading forage
666 species of the Canadian Arctic: Consequences for food web dynamics. *Environmental*
667 *Pollution* 229, 229–240. <https://doi.org/10.1016/j.envpol.2017.05.085>

668 Rotchell, J.M., Ostrander, G.K., 2003. Molecular markers of endocrine disruption in aquatic
669 organisms. *Journal of Toxicology and Environmental Health - Part B: Critical Reviews*.
670 <https://doi.org/10.1080/10937400306476>

671 Schimmel S.C., Wilson A.J., J., 1977. Acute Toxicity of Kepone to Four Estuarine Animals.
672 *U.S. Environmental Protection Agency Environmental Research Laboratory* 18, 224–
673 227.

674 Serge, S., 1994. La lutte intégrée contre le charançon noir des bananiers, *Cosmopolites*
675 *sordidus*. *Fruits* 49, 151–162.

676 Snegaroff, J., 1977. Les résidus d’insecticides organochlorés dans les sols et les rivières de la
677 région bananière de Guadeloupe. *Phytiatr. Phytopharm.* 26, 251–268.

678 Sreeram, M., Menon, N., 2005. Histopathological changes in the hepatopancreas of the
679 penaeid shrimp *Metapenaeus dobsoni* exposed to petroleum hydrocarbons. *Journal of the*
680 *Marine Biological Association of India*.

681 Tropea, C., Piazza, Y., López, L.S., 2010. Effect of long-term exposure to high temperature
682 on survival, growth and reproductive parameters of the ‘redclaw’ crayfish *Cherax*
683 *quadricarinatus*. *Aquaculture* 302, 49–56.
684 <https://doi.org/10.1016/j.aquaculture.2010.01.027>

685 Weir, S.M., Salice, C.J., 2012. High tolerance to abiotic stressors and invasion success of the
686 slow growing freshwater snail, *Melanoides tuberculatus*. *Biological Invasions* 14, 385–
687 394. <https://doi.org/10.1007/s10530-011-0084-x>

688 Zeng, H., Ye, H., Li, S., Wang, G., Huang, J., 2010. Hepatopancreas cell cultures from mud
689 crab, *Scylla paramamosain*. *In Vitro Cellular and Developmental Biology - Animal*.
690 <https://doi.org/10.1007/s11626-009-9259-5>

691

692