



**HAL**  
open science

## A Bayesian framework for estimating parameters of a generic toxicokinetic model for the bioaccumulation of organic chemicals by benthic invertebrates: Proof of concept with PCB153 and two freshwater species

Aude Ratier, Christelle Lopes, Pierre Labadie, H el ene Budzinski, Nicolas Delorme, Herve Queau, Laurent Peluhet, Olivier Geffard, Marc Babut

### ► To cite this version:

Aude Ratier, Christelle Lopes, Pierre Labadie, H el ene Budzinski, Nicolas Delorme, et al.. A Bayesian framework for estimating parameters of a generic toxicokinetic model for the bioaccumulation of organic chemicals by benthic invertebrates: Proof of concept with PCB153 and two freshwater species. *Ecotoxicology and Environmental Safety*, 2019, 180, pp.33-42. 10.1016/j.ecoenv.2019.04.080 . hal-02145836

**HAL Id: hal-02145836**

**<https://hal.science/hal-02145836v1>**

Submitted on 3 Jun 2019

**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 destin ee au d ep ot et  a la diffusion de documents scientifiques de niveau recherche, publi es ou non,  emanant des  tablissements d'enseignement et de recherche franais ou  trangers, des laboratoires publics ou priv es.

# **A Bayesian framework for estimating parameters of a generic toxicokinetic model for the bioaccumulation of organic chemicals by benthic invertebrates: proof of concept with PCB153 and two freshwater species.**

**Aude Ratier<sup>a</sup>, Christelle Lopes<sup>b,\*</sup>, Pierre Labadie<sup>c</sup>, H el ene Budzinski<sup>c</sup>, Nicolas Delorme<sup>a</sup>, Herv e Qu eau<sup>a</sup>, Laurent Peluhet<sup>c</sup>, Olivier Geffard<sup>a</sup>, Marc Babut<sup>a</sup>.**

<sup>a</sup> Irstea, UR RiverLy, centre de Lyon-Villeurbanne, 5 Avenue de la Doua, CS20244, 69625 Villeurbanne Cedex, France.

<sup>b</sup> Univ Lyon, Universit e Lyon 1, CNRS, Laboratoire de Biom etrie et Biologie Evolutive UMR5558, F-69622 Villeurbanne, France.

<sup>c</sup> CNRS, Environnements et Pal oenvironnements Oc aniques et Continentaux (EPOC), UMR 5805, Laboratoire de Physico- et Toxicochimie de l'environnement (LPTC), Universit e de Bordeaux, 351 cours de la Lib eration, 33405 Talence, France.

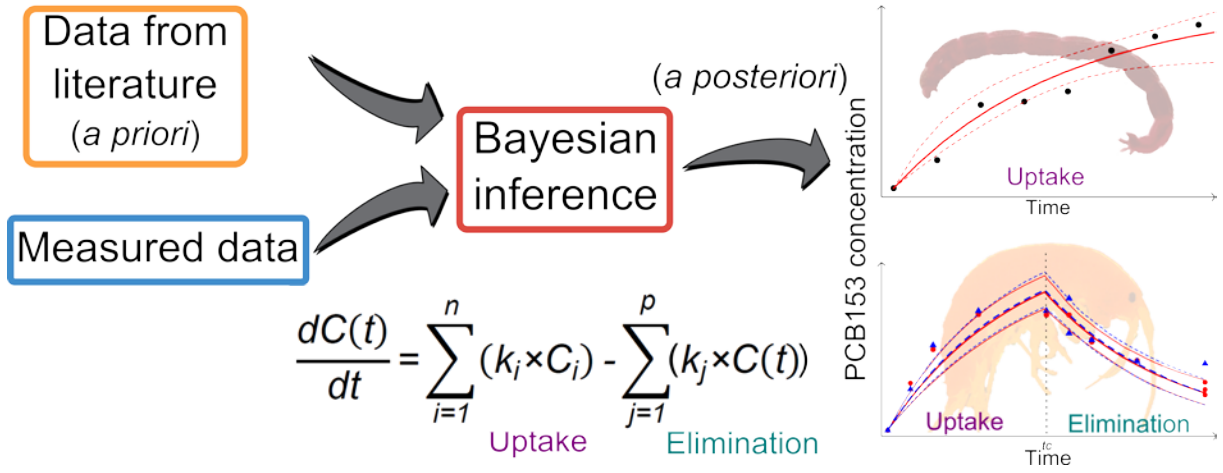
\* Corresponding author: [christelle.lopes@univ-lyon1.fr](mailto:christelle.lopes@univ-lyon1.fr)

DOI: [10.1016/j.ecoenv.2019.04.080](https://doi.org/10.1016/j.ecoenv.2019.04.080)

# 1 Highlights

- 2 • A generic TK model in a Bayesian framework was proposed
- 3 • Each uptake and elimination route was considered as a module in the model
- 4 • Model parameter estimates are relevant regarding species difference

5 Graphical abstract



6

## 7 Abstract

8 Toxicokinetic (TK) models are relevant and widely used to predict chemical concentrations in  
9 biological organisms. The importance of dietary uptake for aquatic invertebrates has been  
10 increasingly assessed in recent years. However, the model parameters are estimated on  
11 limited specific laboratory data sets that are bounded by several uncertainties. The aim of this  
12 study was to implement a Bayesian framework for simultaneously estimating the parameters  
13 of a generic TK model for benthic invertebrate species from all data collected. We illustrate  
14 our approach on the bioaccumulation of PCB153 by two species with different life traits and  
15 therefore exposure routes: *Chironomus riparius* larvae exposed to spiked sediment for 7 days  
16 and *Gammarus fossarum* exposed to spiked sediment and/or leaves for 7 days and then  
17 transferred to a clean media for 7 more days. The TK models assuming first-order kinetics  
18 were fitted to the data using Bayesian inference. The median model predictions and their 95%  
19 credibility intervals showed that the model fit the data well. From a methodological point of  
20 view, this paper illustrates that simultaneously estimating all model parameters from all  
21 available data by Bayesian inference, while considering the correlation between parameters  
22 and different types of data, is a real added value for TK modeling. Moreover, we demonstrated  
23 the ability of a generic TK model considering uptake and elimination routes as modules to add  
24 according to the availability of the data measured. From an ecotoxicological point of view, we  
25 show differences in PCB153 bioaccumulation between chironomids and gammarids,  
26 explained by the different life traits of these two organisms.

27 **Keywords:** Bioaccumulation – Benthic invertebrates – PCB153 – Bayesian inference –  
28 Toxicokinetic model

## 29 1. Introduction

30 In environmental risk assessment (ERA), models based on toxicokinetic (TK)  
31 approaches are widely recognized as providing diagnostics (models for understanding) and  
32 prognostics (models for prediction), sometimes used by decision-makers (e.g., Pavan, 2006;  
33 EPA, 2006; IPCS, 2010). TK models are relevant and widely used to predict chemical  
34 concentrations in biological organisms from those to which they are exposed in their  
35 environment. This process, also called bioaccumulation, depends on environmental conditions  
36 (temperature, light, food availability), contaminant properties (octanol-water partition  
37 coefficient, water solubility, dissociation and volatilization, sorption in sediment) (Mamy, 2015)  
38 and biological characteristics of the species (life traits, diet, lifecycle). TK models describe the  
39 process of bioaccumulation as the net balance between the uptake of contaminants from  
40 different sources (water, diet) and their elimination by different processes (excretion, growth  
41 and/or biotransformation) (MacKay and Fraser, 2000). Different TK models have been  
42 proposed, such as compartmental models and physiologically based toxicokinetic (PBTK)  
43 models (Grech, 2017; Landrum, 1992). Compartmental models describe toxicant fluxes  
44 between compartments, which may or may not have a physiologic or anatomic meaning. A  
45 PBTK model subdivides the body into compartments representing real tissues or organs  
46 connected through a fluid, usually blood (Bois and Brochot, 2016). For aquatic invertebrates,  
47 compartmental models were originally developed for metals and then for some organic  
48 contaminants where the organism is often considered as a single compartment.

49 For soluble contaminants, it is usually assumed that the water column is the main  
50 exposure and uptake route. Nevertheless, dietary uptake is of greater importance for  
51 hydrophobic contaminants due to their high adsorption on organic matter or food (Gross-  
52 Sorokin, 2003). Moreover, it has been shown for several aquatic invertebrates, such as the  
53 insect *Chironomus riparius*, that exposure to sediment cannot be ignored (Leppänen and  
54 Kukkonen, 1998; Sidney, 2016). The importance of dietary uptake for aquatic invertebrates

55 has been increasingly assessed in recent years (Ashauer 2010, Carrasco-Navarro, 2015;  
56 Englert, 2017; Miller, 2017; Rösch, 2017). *Chironomus* sp. and *Gammarus* sp. are freshwater  
57 invertebrates widely used in ecotoxicology, due to their widespread presence throughout the  
58 Northern hemisphere and their capacity to accumulate various organic and inorganic  
59 contaminants (Amiard, 1987; Ashauer, 2012; Bertin, 2014; Lebrun, 2011; López-Doval, 2012;  
60 Lydy, 1999). These organisms are also an important food source for fish, amphibians and birds  
61 (Macneil, 1999), hence the transfer of contaminants within the aquatic food web.

62         Nowadays, TK model parameters are often determined from short-term exposures of  
63 organisms under controlled laboratory conditions (Ashauer, 2010). OECD Guideline 315  
64 (OECD, 2008) suggests two methods to estimate the uptake and elimination rates. The most  
65 frequently used sequential method estimates the elimination rate using nonlinear regression  
66 depuration data which is then fixed to estimate the uptake rate with the uptake data. As  
67 elimination also occurs during the accumulation phase, separately estimating parameters that  
68 are linked does not allow taking into account their correlation on uncertainty. However, the  
69 precision of parameter estimates is a relevant point to strengthen environmental assessments  
70 (Lin, 2004; Richards and Chaloupka, 2009). The simultaneous method estimates both the  
71 uptake and elimination rates together, and is considered a potentially more reliable and more  
72 realistic model. Only recent studies (Ashauer, 2010; Miller, 2016, 2017) have applied  
73 simultaneous methods for parameter approximation. Sequential or simultaneous methods can  
74 be deployed in a frequentist or Bayesian approach. Apart from the problem of the correlation  
75 between all the model's parameters, the frequentist approach (sequential or simultaneous)  
76 cannot simultaneously use different kinds of data (e.g., bioaccumulation and growth data) to  
77 estimate common parameters. Bayesian inference bypasses these limits by estimating all  
78 model parameters from all kinds of data (Gelman, 1995) and thus provides a more  
79 comprehensive approach and a better quantification of uncertainty in parameter estimates as  
80 well as a better consideration of variability in model predictions (Bernillon and Bois, 2000). The

81 application of Bayesian inference to TK models in aquatic invertebrates remains limited (Lin,  
82 2004).

83           The aim of this study was to propose a Bayesian framework to simultaneously estimate  
84 all the parameters of a generic TK model from uptake and elimination data together. As a first  
85 development step, we applied this concept to two aquatic invertebrate species. The resulting  
86 joint posterior distribution giving the probability distribution of all parameters together will  
87 enable a more accurate assessment of uncertainty around estimates and thus TK model  
88 predictions. We illustrate our approach with the bioaccumulation of the well-known  
89 contaminant PCB153 by two freshwater benthic invertebrate species with different life  
90 traits: the Diptera *C. riparius* and the amphipod crustacean *Gammarus fossarum*.



## 91 2. Model

### 92 2.1 Generic TK model

93 A first-order kinetic bioaccumulation model that accounts for different uptake pathways  
94 and elimination processes can be expressed as follows (Eq. (1)):

$$95 \quad \frac{dC(t)}{dt} = \sum_{i=1}^n (k_i \times C_i(t)) - \sum_{j=1}^p (k_j \times C(t)) \quad (1)$$

96 where  $C(t)$  is the contaminant concentration at time  $t$  (days) in the whole organism ( $\text{ng g}_{\text{org}}^{-1}$ )  
97 where the mass of the organism is expressed in wet weight (ww)),  $n$  is the number of  
98 contamination sources,  $k_i$  the uptake rate from the contamination source  $i$ ,  $C_i(t)$  the  
99 contaminant concentration in the contamination source  $i$  at time  $t$  (days),  $p$  the number of  
100 elimination processes and  $k_j$  the elimination rate related to process  $j$ .

101 If we consider that  $C_i(t)$  is constant over time, which is appropriate in laboratory  
102 conditions, Eq. (1) can be analytically integrated by distinguishing uptake (Eq. (2)) from  
103 elimination phases (Eq. (3)):

$$104 \quad \begin{cases} C(t) = \frac{\sum_{i=1}^n (k_i \times C_i)}{\sum_{j=1}^p k_j} + \left( C_0 - \frac{\sum_{i=1}^n (k_i \times C_i)}{\sum_{j=1}^p k_j} \right) \times e^{-\left(\sum_{j=1}^p k_j\right) \times t} & \text{for } 0 < t < t_c \\ C(t) = \frac{\sum_{i=1}^n (k_i \times C_i)}{\sum_{j=1}^p k_j} \times e^{-\left(\sum_{j=1}^p k_j\right) \times (t-t_c)} + \left( C_0 - \frac{\sum_{i=1}^n (k_i \times C_i)}{\sum_{j=1}^p k_j} \right) \times e^{-\left(\sum_{j=1}^p k_j\right) \times t} & \text{for } t > t_c \end{cases} \quad (2)$$

105

106 where  $C_0$  is the contaminant concentration in the whole organism at the beginning of exposure  
107 ( $\text{ng g}_{\text{org}}^{-1}$ ) and  $t_c$  the time at the end of the uptake phase (days).

### 108 2.2 Application of the model to the two species studied

#### 109 2.2.1 Chironomids

110 For chironomids, we consider that the exposure sources are water (respiration) and  
111 sediment (ingestion), while elimination occurs due to excretion and growth dilution; Eq. (1) can  
112 thus be rewritten as follows (Eq. (4)):

113 
$$\frac{dC(t)}{dt} = k_w \times C_w + k_s \times C_s - (k_e + k_g) \times C(t) \quad (4)$$

114 where  $k_w$  is the uptake rate from the water ( $L \text{ g}_{\text{org}}^{-1} \text{ d}^{-1}$ ),  $C_w$  the contaminant concentration in  
 115 water ( $\text{ng L}^{-1}$ ),  $k_s$  the uptake rate from the sediment ( $\text{g}_{\text{sed}} \text{ g}_{\text{org}}^{-1} \text{ d}^{-1}$ ),  $C_s$  the contaminant  
 116 concentration in sediment ( $\text{ng g}_{\text{sed}}^{-1} \text{ dw}$ ),  $k_e$  the elimination rate ( $\text{d}^{-1}$ ) and  $k_g$  the growth rate ( $\text{d}^{-1}$ ).  
 117 1).

118 The chironomid growth rate is obtained from the von Bertalanffy growth equation  
 119 (Eq. (5)), one of the most widely used models for describing the growth of benthic invertebrates  
 120 (von Bertalanffy, 1938; K. Nakamura, 1973):

121 
$$L(t) = L_{\text{max}} - (L_{\text{max}} - L_0) \times e^{(-k_g \times t)} \quad (5)$$

122 where  $L(t)$  is the chironomid size (mm) at time  $t$  (d),  $L_{\text{max}}$  is the asymptotic size (mm),  $L_0$  is the  
 123 size at birth (mm) and  $k_g$  is the growth rate ( $\text{d}^{-1}$ ).

124 Given that, for highly hydrophobic compounds, contamination from water could be  
 125 restricted ( $k_w=0$ ), a sub-model accounting for sediment as the only contamination source was  
 126 also considered:

127 
$$\frac{dC(t)}{dt} = k_s \times C_s - (k_e + k_g) \times C(t) \quad (6)$$

### 128 2.2.2 Gammarids

129 Gammarids feed on detritus such as litter (Forrow and Maltby, 2000). As a  
 130 consequence, exposure to chemicals could occur from water, litter (leaves) and sediment  
 131 consumption, since sediment particles deposit on the surface of leaves when gammarids  
 132 forage (Bertin, 2016). Furthermore, we assumed that gammarids would not grow during the  
 133 experiment, as shown by Galic and Forbes (2017), for the adult size considered here. As a  
 134 result, Eq. (1) can be rewritten as follows (Eq. (7)):

135 
$$\frac{dC(t)}{dt} = k_w \times C_w + k_s \times C_s + k_l \times C_l - k_e \times C(t) \quad (7)$$

136 where  $k_l$  is the uptake rate from the leaves ( $\text{ng g}_{\text{org}}^{-1} \text{d}^{-1}$ ) and  $C_l$  the contaminant concentration  
137 in leaves ( $\text{ng g}^{-1}$ ).

138 According to the experimental conditions, several sub-models could be considered and tested  
139 according to several hypotheses on the exposure routes ( $k_w=0$  and/or  $k_s=0$  and/or  $k_l=0$ , Table  
140 S1).

## 141 3. Materials and methods

### 142 *3.1 Chemicals, reagents and quality control*

143 Solid 2,2', 4,4', 5,5' hexachlorobiphenyl (PCB153) was purchased from Sigma-Aldrich  
144 (St Quentin-Fallavier, France). A working solution was prepared in acetone at 1.01 g L<sup>-1</sup> for the  
145 contamination of sediment and leaves. The native SRM2262 solution and the internal standard  
146 PCB198 (99%) used for the quantification of PCB153 were provided by LGC Promochem  
147 (NIST) and Ultra Scientific, respectively. Native PCB153 recovery was determined using  
148 spiked samples for water (89 ± 1%), technical sand (75 ± 11%), sediment (NIST SRM1941b  
149 Organics in Marine Sediment, 64 ± 8%) and fish (NIST SRM 1947 Lake Michigan Fish Tissue,  
150 65 ± 12%) reference materials. The limit of detection (LoD) was determined as the  
151 concentration with a signal-to-noise ratio of 3 (LoD = 0.003 ng L<sup>-1</sup> for water, between 0.003  
152 and 0.040 ng g<sup>-1</sup> dw for sediment, leaves and organisms). Replicate procedural blanks (n =  
153 13) were analyzed for each series of samples where the PCB153 concentration was always  
154 below the LoD.

### 155 *3.2. Matrix spiking*

#### 156 *3.2.1 Sediment spiking*

157 In January and March 2017, 60 L of natural sediment was collected from a watercourse  
158 in the Miribel-Jonage nature park (Vaulx-en-Velin, eastern central France near Lyon, 4°59'27"E  
159 and 45°47'55"N for the chironomid experiment and 5°00'51"E and 45°79'71"N for the  
160 gammarid experiment). The sediment was collected using a manual dredger, sieved at 2 mm,  
161 pooled in a polypropylene (PP) jar, and stored at 4 ± 2 °C until they were used. For chironomid  
162 and gammarid experiments, respectively, the sediment was characterized by a water content  
163 of 67.9 and 51.0%, a particulate organic carbon content of 11.9 and 2.87% on dry weight  
164 matter, and 0.370 and 0.250% particulate nitrogen content. The sediment was homogenized  
165 and mixed with mechanical action (paint propeller connected to an electric drill) for 20 min.

166 Then 1.2 L of sediment was added in 20 and nine Pyrex bottles (2 L) for the chironomid and  
167 gammarid experiments, respectively. Each bottle was spiked with a solution of PCB153 in an  
168 acetone carrier at a nominal concentration of 100 and 50 ng g<sup>-1</sup> dry weight (dw) for chironomid  
169 and gammarid experiments, respectively. The amount of carrier added to the sediment was  
170 minimal (0.07 and 0.08 µL/g<sub>sed</sub> for chironomid and gammarid experiments, respectively). Then  
171 each bottle was rotated for 24 h (chironomids) to 72 h (gammarids) at 15 revolutions per minute  
172 (rpm) at room temperature (21 °C). After 48 h storage at 4 °C, contaminated sediment in each  
173 bottle was transferred in a polypropylene jar and homogenized and restless with mechanical  
174 action for 20 min before being added to aquaria.

### 175 3.2.2 Leaf spiking

176 Alder leaves (*Alnus glutinosa*) were collected in November 2016 and stored in plastic  
177 boxes. Prior to exposing gammarids to PCB153, the alder leaves were placed in a bucket filled  
178 with several liters of ground water for 7 days at 21 °C. The water was renewed every 2 days  
179 to remove the exudates from the leaves. Several batches of 5 g<sub>dw</sub> leaves were placed in Pyrex  
180 bottles (2 L) containing 1 L of groundwater and were spiked at a nominal concentration of wet-  
181 weight basis 50 ng<sub>PCB153</sub> kg<sup>-1</sup><sub>leaf,ww</sub> with a solution of 5 µg L<sup>-1</sup> of PCB153 in acetone. These  
182 batches were rotated for 72 h at 10 rpm at room temperature (21 °C). Then leaves were rinsed  
183 with ground water for 3 days in Pyrex bottles before being placed in aquaria.

## 184 3.3 Organism exposure to PCB153

### 185 3.3.1 Chironomid exposure

186 A total of seven aquaria (38-20-24.5 cm in polystyrene) were prepared with 3 L of  
187 homogenized spiked sediment and 15 L of groundwater. Each aquarium was allowed to settle  
188 for 10 days before introducing the chironomids. A control aquarium was prepared in the same  
189 way with reagent control sediment (Fig. S1).

190 Benthic invertebrate *C. riparius* were obtained from laboratory cultures carried out  
191 according to standard methods (OECD, 2004; AFNOR, 2010). A total of 400 fourth-instar

192 larvae (7-day-old larvae post-hatching, L4) were added to each aquarium. Chironomids were  
193 exposed to spiked sediment for 7 days at  $21 \pm 0.2$  °C in aerated and static water. A 16:8-h  
194 light:dark cycle was maintained throughout the experiment. Larvae were fed every day with  
195 400 mg commercial food Tetramin® per aquarium. The water quality parameters were  
196 monitored and are presented in Annex S1. Chironomid survival, length and wet weight were  
197 determined at each sampling time. To determine the total length, ten larvae were  
198 photographed using an IEEE 1394 Digital CCD camera (F2, FOculus, Germany) mounted on  
199 an Olympus BX51 light and SZX12 stereo zoom microscopes at low magnifications. The mean  
200 lengths were determined using digital image analysis software (SigmaScan Pro software).

### 201 3.3.2 Gammarid exposure

202 Three weeks before the start of the experiment, about 3,000 male gammarids were  
203 collected with a hand net at a reference site (Saint-Maurice de Rémens, France,  $5^{\circ}26'22''$ E -  
204  $45^{\circ}95'79''$ N). Gammarids were brought to the laboratory and acclimated 3 weeks in aquaria  
205 with continuously renewed groundwater under constant aeration, a 16:8-h light:dark  
206 photoperiod was maintained and the temperature was kept at 12 °C. Organisms were fed *ad*  
207 *libitum* with alder leaves. Only male gammarids ( $11.4 \pm 0.9$  mm) were selected, in order to  
208 eliminate potential biases due to neonate release by females.

209 Gammarid experiments were composed of two phases: uptake and elimination. For the  
210 uptake phase, different exposure routes were tested: gammarids were exposed to spiked  
211 leaves (E1 condition) or to spiked sediment (E2 condition) for 7 days at  $12 \pm 0.2$  °C under a  
212 16:8-h light:dark cycle maintained throughout the experiment. A third condition (E3) was  
213 tested, similar to E2 but without organisms, in order to determine whether there was a  
214 contamination transfer from sediment to leaves (Fig. S1). The overlying water was renewed  
215 daily under constant aeration. A control aquarium for E2 was prepared with homogenized  
216 reagent sediment ( $0.08 \mu\text{L}$  of acetone/ $\text{g}_{\text{sed}}$ ).

217 At the beginning of the experiment, 300 individuals were added per aquarium (test and  
218 control). Gammarids exposed to the contaminated leaves (E1) were distributed in three

219 aquariums each containing 15 L of groundwater and one batch of previously spiked leaves per  
220 aquarium. Organisms exposed to spiked sediment (E2) were distributed in three aquariums  
221 each containing 3 L of spiked sediment, 15 L of groundwater and one batch of previously clean  
222 re-hydrated leaves. After 7 days of exposure, gammarid survival for each condition was  
223 determined and the organisms were transferred to a clean medium: 270 organisms per  
224 aquarium with groundwater for the E1 condition or with clean sediment and groundwater for  
225 the E2 condition. During depuration, gammarids were fed with clean leaves (5 g dw). The water  
226 quality parameters were monitored and are presented in Annex S1. Gammarid survival, length  
227 and wet weight were determined at the end of the uptake and elimination phases.

#### 228 *3.4 Sample collection*

229 The overlying water (OW) was sampled in 1-L polyethylene (PE) bottles. Subsamples  
230 from sediment were deposited in 180-mL PE tubes. Organisms were collected in 50-mL  
231 Falcon® tubes. Every day, chironomids were collected by sieving sediment at 500 µm. The  
232 OW and sediment samples were collected at days 0, 4 and 7. The OW, sediment and organism  
233 control samples were collected at days 0 and 7. At days 0, 1, 2, 4, 7, 8, 9, 11 and 14, 90  
234 gammarids were collected for each condition (E1 and E2). The OW, sediment, leaf and  
235 organism control samples were collected at days 0, 7 and 14. All samples were stored at -21°C  
236 and lyophilized at -65°C (Christ-Alpha 1-4LD, Bioblock Scientific) under a pressure of 0.050  
237 mbar for 48 h for organisms, 72 h for leaves and 7 days for sediment.

#### 238 *3.5 PCB153 analyses*

239 Water samples were filtered using 47 mm GF/F glass microfiber filters (Whatman®),  
240 and approximately 10 mL of filtered samples were extracted using a SPME procedure.  
241 Approximately 0.5 g of sediment samples and 0.2 g of leaves and organisms were extracted  
242 by microwave-assisted extraction (Milestone SRL, Sorisole, Italy) with 12 mL of DCM at 80 °C  
243 for 15 min. The extracts were filtered, concentrated under nitrogen flow at 40 °C and cleaned  
244 up on columns containing activated copper and acidified silica gel (40% H<sub>2</sub>SO<sub>4</sub> w/w) previously

245 conditioned with 5 mL of pentane; after extract loading, PCB153 was eluted with three times 5  
246 mL of a pentane/DCM (90/10, v/v). The eluate was further concentrated, solvent exchanged  
247 to isooctane and taken to a final volume of 100  $\mu$ L.

248 PCB153 was analyzed using 6890N Agilent Technologies gas chromatography  
249 (Massy, France) connected to an electron capture detector (ECD). Analytes were injected  
250 (1  $\mu$ L) in pulsed splitless mode and separated with a J&W HP-5MS column (5% phenyl – 95%  
251 methylpolysiloxane; 60 m  $\times$  0.25 mm  $\times$  0.1  $\mu$ m). Helium (vector gas, 1.3 mL  $\text{min}^{-1}$ ) and nitrogen  
252 (auxiliary gas, 60.0 mL  $\text{min}^{-1}$ ) were used. The injector temperature was set at 280  $^{\circ}\text{C}$  and the  
253 detector temperature at 300  $^{\circ}\text{C}$ . The kiln temperature program was: 90  $^{\circ}\text{C}$  for 2 min (80  $^{\circ}\text{C}$  for  
254 water samples), 15  $^{\circ}\text{C min}^{-1}$  to 178  $^{\circ}\text{C}$  (20  $^{\circ}\text{C min}^{-1}$  to 190 $^{\circ}\text{C}$ ), 2  $^{\circ}\text{C min}^{-1}$  to 230  $^{\circ}\text{C}$  (210  $^{\circ}\text{C}$ ),  
255 30  $^{\circ}\text{C min}^{-1}$  (15  $^{\circ}\text{C min}^{-1}$ ) to 300  $^{\circ}\text{C}$ , 300  $^{\circ}\text{C}$  for 3.8 min (5 min). PCB153 was quantified relative  
256 to internal standard (PCB198, 9.6–11.8 ng).

### 257 *3.6 Data analysis*

258 Significant differences were considered according to the Wilcoxon test with an  $\alpha$  risk of 5%.  
259 Graphical representations were made with the statistical software R (version 3.3.3, R Core  
260 Team, 2017).



## 261 4. Link between model and data: Bayesian inference

### 262 4.1. Stochasticity

263 For both chironomids and gammarids, we assumed a gaussian distribution of the  
264 contaminant concentration in the organism:

$$265 C_{obs}(t) \sim \mathcal{N}(C(t), 1/\sigma^2) \quad (8)$$

266 where N stands for the Normal law,  $C_{obs}(t)$  corresponds to the contaminant concentration in  
267 the organism at time t measured during the experiments,  $C(t)$  is the contaminant concentration  
268 at time t predicted by the model and  $\sigma$  is the standard deviation of contaminant concentration  
269 in the organism.

270 For chironomid size, we also assumed a normal distribution:

$$271 L_{obs}(t) \sim \mathcal{N}(L(t), 1/\sigma_L^2) \quad (9)$$

272 where  $L_{obs}(t)$  is the length measured at time t during the experiments,  $L(t)$  is the length predicted  
273 by the von Bertalanffy model at time t, and  $\sigma_L$  is the standard deviation of organism size.

### 274 4.2. Graphical representation

275 Figure 1 represents the directed acyclic graphs (DAGs) for generic (a) chironomids (b)  
276 and gammarids (c), which symbolize the deterministic links between parameters and variables  
277 for the complete generic (Eq. (1)), chironomids (Eq. (4)) and gammarids (Eq. (7)) models and  
278 the stochastic links between the observed and predicted data.

### 279 4.3. Definition of priors

280 Before conducting an experimental study, a prior distribution is defined for each  
281 parameter according to information available from the literature and/or previous experiments.  
282 Depending on the source and the conditions that the information comes from, informative,  
283 semi-informative or noninformative prior distribution can be used. If a parameter was already  
284 estimated in previous studies or if previous data are available, a normal prior distribution can  
285 be used (with the mean value estimated and the precision with a standard deviation twice the

286 value estimated to take into account the potential differences in experimental conditions).  
287 However, if no information is available but an order of magnitude is (positive only, for example),  
288 it is possible to apply a weakly informative prior, as a uniform distribution. If any information is  
289 available on the order of magnitude of a parameter, its prior can be defined on the decimal  
290 logarithm scale in order to give the same probability to lower or higher estimates. For variance  
291 parameters, we use a noninformative (0.001, 0.001) gamma prior, as is usually done (Lambert,  
292 2005; Richards and Chaloupka, 2009).

293 For chironomids, priors were defined from the values given by Schuler et al. (2003)  
294 where *Chironomus tentans* were exposed to PCB153-spiked sediment. They found a mean  $k_s$   
295 value of  $0.054 \text{ g}_{\text{sed.}} \text{ g}_{\text{org}}^{-1} \cdot \text{h}^{-1}$  and a mean  $k_e$  value of  $0.011 \text{ h}^{-1}$ . We thus assumed a  $\log_{10}$ -normal  
296 (0.113, 5) prior for  $k_s$  (in  $\text{g}_{\text{sed.}} \text{ g}_{\text{org}}^{-1} \text{ d}^{-1}$ ) and a  $\log_{10}$ -normal (-0.578, 5) prior for  $k_e$  (in  $\text{d}^{-1}$ ). For  $k_w$ ,  
297 we assumed a  $\log_{10}$ -uniform (-5, 2) prior. We used a  $\log_{10}$ -normal (0.236, 2) prior for  $k_g$  (in  $\text{d}^{-1}$ )  
298 according to Péry et al. (2002), corresponding to a growth rate of  $1.72 \text{ d}^{-1}$ . Priors for  $L_0$  and  
299  $L_{\text{max}}$  were assumed to follow  $\log_{10}$ -normal distributions and were set, respectively, at (0.778,  
300 0.64; in days) corresponding to an initial size of 6 mm, and (1.056, 0.64; in days) corresponding  
301 to a mean limit size of 11.4 mm (Péry, 2002). We assumed a non-informative (0.001, 0.001)  
302 gamma prior for the precision.

303 For gammarids, little information was available in the literature. As a consequence, we  
304 used a non-informative (-5, 2)  $\log_{10}$ -uniform prior for uptake and elimination rate constants ( $k_s$ ,  
305  $k_i$ ,  $k_w$ ,  $k_e$ ). We also assumed a non-informative (0.001, 0.001) gamma prior for precision.

#### 306 4.4. Implementation of the model – MCMC simulations

307 Model computation was performed with JAGS and R software (R Core Team, 2017;  
308 Plummer, 2016). The models were fitted to bioaccumulation data using Bayesian inference via  
309 Markov Chain Monte Carlo (MCMC) sampling. For each model tested, we started by running  
310 a short sampling (5,000 iterations after a burn-in phase of 10,000 iterations) using the Raftery  
311 and Lewis (1992) method to set the necessary thinning and number of iterations to reach an  
312 accurate estimation of each model parameter.

313 For chironomids, MCMC sampling was based on 150,000 iterations for three chains  
314 after discarding the first 5,000 iterations. Samples from every 40<sup>th</sup> iteration were stored to  
315 reduce autocorrelation. For gammarids, 26,000 iterations were done after discarding the first  
316 10,000 iterations. Samples from every seventh iteration were stored to reduce autocorrelation  
317 in the sample.

318 To monitor the convergence of the chains, we used a visual inspection as well as the  
319 Gelman criterion (Gelman, 1995). The R codes are available in supporting information (Annex  
320 S2).

#### 321 *4.5. Posterior distributions and relevance of model predictions*

322 From the joint posterior distribution, we can obtain the marginal posterior distribution  
323 for each parameter, which can be summarized by the mean or median and standard deviation.  
324 The accuracy of model parameter estimation can be visualized by comparing prior and  
325 posterior distributions: a thin posterior distribution reflects that the data contributed enough  
326 information to precisely estimate parameters.

327 To check the relevance of model predictions, we represent, for each experiment,  
328 observed data superimposed on the model simulated with the median of the posterior  
329 distribution for each parameter and the 95% credibility band of the predicted data considering  
330 parameter uncertainties and stochasticity. To obtain the 95% credibility band, the predicted  
331 data were simulated with the model for each MCMC iteration and the stochastic model  
332 considered for observed data (Eqs. (8) and (9)).

#### 333 *4.6. Model comparisons*

334 Several hypotheses were considered and tested for chironomids (Eqs. (4) and (6)) and  
335 for gammarids (Table S1) according to the experimental data available. To compare the  
336 different sub-models fitted, we analyzed the precision of each parameter estimation and the  
337 relevance of model predictions through the deviance information criterion (DIC), a Bayesian  
338 measurement that weighs the quality of model fit with its complexity. Sub-models with lower

339 DIC values are expected to effectively balance between predictive capacity and complexity  
340 (Spiegelhalter, 2002).

341 For Bayesian inference, JAGS 4.2.0 for Windows and the *rjags* package for R software  
342 were used.

## 5. Results

### 5.1. *Chironomus riparius*

#### 5.1.1. Sediment, water and chironomid contamination

The PCB153 concentrations in control and spiked sediment at day 0 were, respectively,  $1.01 \pm 0.25$  and  $83.3 \pm 20.8$  ng g<sub>sed</sub><sup>-1</sup> versus 100 ng g<sub>sed</sub><sup>-1</sup> (dw) expected. At the end of the experiment, the PCB153 concentration in spiked sediment was  $89.6 \pm 22.4$  ng g<sub>sed</sub><sup>-1</sup> (dw). No significant difference was observed for the concentrations in the sediment monitored during the experiment ( $p$ -value = 0.333).

At the beginning of the experiment, the PCB153 concentrations in water were  $0.077 \pm 0.009$  ng L<sup>-1</sup> and  $0.230 \pm 0.025$  ng L<sup>-1</sup> in the control and exposed aquaria, respectively, while they were  $0.259 \pm 0.028$  ng L<sup>-1</sup> and  $3.85 \pm 0.420$  ng L<sup>-1</sup>, respectively, at the end of the experiment.

PCB153 concentrations in *C. riparius* exposed to spiked sediment increased from  $0.089 \pm 0.031$  to  $142 \pm 50.0$  ng g<sub>org</sub><sup>-1</sup> in 7 days.

#### 5.1.2. Chironomid survival and growth

An acceptable survival rate was observed during the experiment for the test condition (88%) and for the control (76%). No adverse effect of spiked sediment on chironomid growth was observed ( $p$ -value = 0.248).

#### 5.1.3. Parameter estimates

Two models were fitted to uptake data: Equation (4), which considers water and sediment exposure routes, and Equation (6), which considers sediment only. Similar marginal posterior distributions were obtained for each parameter with both models (Table S2, Fig. S2). Since the DICs for both models were similar (87.66 for Eq. (4) and 87.71 for Eq. (6)), we selected the most parsimonious one (with the fewest parameters), Eq. (6), which accounts for exposure from sediment only.

The inference process quickly converged, and thin posterior distributions were obtained for each parameter, meaning that data contribute sufficient information to accurately estimate model parameters (Fig. S2). Median values and 95% credibility intervals for each parameter were estimated and are summarized in Table 1.

#### 5.1.4. Model predictions

The model predictions fit the bioaccumulation and growth data well (Fig. 2). For bioaccumulation (Fig. 2a), two of the eight measurements were outside the 95% credibility band of the predicted data, and for growth 23 of the 80 measurements (Fig. 2b).

## 5.2. *Gammarus fossarum*

### 5.2.1. Leaf, sediment, water and gammarid contamination

PCB153 concentrations in the different matrices (water, leaves and sediment) are summarized in Table 2. There was no difference among PCB153 concentrations in sediment throughout the exposure phase ( $p$ -value = 0.939). Leaves added to spiked sediment without gammarids (E3 condition) were significantly less contaminated than leaves in spiked sediment with gammarids (E2 condition) ( $p$ -value = 0.042).

As shown in Figure 3, we obtained similar PCB153 concentrations in gammarids over time for the two conditions tested (E1 and E2). PCB153 concentrations in *G. fossarum* exposed to spiked leaves (E1) and sediment (E2) increased from an initial concentration of  $0.320 \pm 0.110$  to  $10.9 \pm 3.80$  and  $11.2 \pm 3.93$  ng g<sub>org</sub><sup>-1</sup>, respectively, at the end of uptake phase. When gammarids were transferred into a clean media, the E1 and E2 PCB153 concentration in the organisms decreased to  $4.71 \pm 1.65$  and  $6.41 \pm 2.24$  ng g<sub>org</sub><sup>-1</sup>, respectively, at day 14.

### 5.2.2. Growth and survival

High survival rates were observed during the experiments. Survival after uptake and elimination periods were, respectively, 93 and 91% in control aquaria, 96 and 97% for E1 and 96 and 94% for E2.

### 5.2.3. Model parameters

Several models were fitted according to the exposure routes considered (Table S1). The models with similar lowest DIC values were those corresponding to hypotheses #3, 4, 6 and 7 in Table S1 (DIC values are reported in Table S2 in the SI). Similar model parameter estimations and marginal posterior distributions for each parameter were obtained for hypotheses #3, 4 and 6 (Table S2, Fig. S3). However, when the water exposure route was considered (hypothesis #7), marginal posterior distributions displayed two peaks for uptake rate constants  $k_l$  and  $k_s$  (Fig. S3), and the median values estimated for  $k_l$  and  $k_s$  considerably decreased (Table S2). We concluded accordingly that considering water as a contamination source did not contribute relevant information to accurately estimate model parameters. This conclusion is consistent with the fact that PCB153 concentrations measured in leaves and sediment were around 10,000 times higher than the concentration in water, in accordance with the hydrophobic character of this substance. Consequently, the model corresponding to hypothesis #6 was also eliminated. Between hypotheses #3 and 4, both considering contamination from leaves and sediment, we decided to keep the model corresponding to hypothesis #3. Indeed, this model is simpler than the one corresponding to hypothesis #4, in that it ignores that PCB153 transfers from sediment to leaves, while DICs are similar.

The inference process for the gammarid model corresponding to hypothesis #3 quickly converged, and thin posterior distributions were obtained for all parameters (Fig. S3). Median values and 95% credibility intervals for each parameter were estimated and are summarized in Table 1. The median estimation of  $k_s$  was around five times higher than that of  $k_l$ .

#### 5.2.4. Model predictions

The model predictions fit well with uptake and elimination data (Fig. 3). For bioaccumulation in the E1 condition, two of the 11 measurements were out the 95% credibility band of the predicted data, and two of the 10 measures for the E2 condition.

## 6. Discussion

### *6.1. Robustness of Bayesian inference*

We used Bayesian inference to simultaneously estimate all the model parameters together to enable a more accurate assessment of uncertainty around model predictions. One of the major advantages of the Bayesian analysis is the possibility of using data from different experiments simultaneously and even different types of data (e.g., uptake and elimination data, or bioaccumulation and growth data) to estimate parameters common to the different models. This was illustrated here by estimating the elimination rate from uptake and elimination data for gammarids (Fig. 3, Table 1), as well as by estimating the chironomid growth rate from bioaccumulation and growth data (Fig. 2, Table 1).

To evaluate the model performance, we used four criteria: (i) marginal posterior distributions for each model parameter, which generally provides more information compared to prior distributions; (ii) goodness of fit to experimental data; (iii) DIC value; and (iv) the principle of parsimony.

For both chironomid and gammarid models, thin posterior distributions were obtained, meaning that data provided enough information to estimate parameters precisely (Fig. S2-a and S3-a). The simultaneous estimate of model parameters assesses the uncertainty on parameters considering the correlation between them. This is an improvement compared to the common approach in TK modeling, where model parameters are often estimated sequentially, without accounting for autocorrelation or confounding factors. The existing knowledge on parameter values derived from the literature can also be accounted for, through the definition of prior distributions, which is not the case in the common approaches. Moreover, in the common framework it is not easy to consider the overall uncertainty of predictions, as recommended in the context of environmental risk assessments (Lin, 2004).

For both species, model predictions fit well with experimental data. The measurements out the 95% credibility band of the predicted data for both species could indeed be due to the



low number of data points and their proximity. For chironomids, Schuler et al. (2003) obtained a  $k_s$  mean value of  $1.30 \pm 0.014 \text{ g}_{\text{sed}} \text{ g}_{\text{org}}^{-1} \text{ d}^{-1}$ , and a  $k_e$  mean value of  $0.264 \pm 0.006 \text{ d}^{-1}$  for *Chironomus tentans* exposed to PCB153-spiked sediment. Here, we obtained lower values, especially for  $k_s$ . This could hypothetically be due to (i) the difference between the species tested, (ii) the difference in experimental conditions (shorter exposure duration and lower sediment concentration in Schuler et al. (2003)) and (iii) growth dilution, which was ignored in Schuler et al. (2003). The literature had demonstrated the importance of considering the growth rate for fourth-instar larvae of chironomids (Péry, 2002; Bertin, 2014). In the present study we estimated this model parameter at  $k_g = 0.123 [0.029\text{--}0.356] \text{ d}^{-1}$ ; in the literature, the values vary between 0.355 and 1.72 (Péry, 2002; Bertin, 2014). Furthermore, Watts and Pascoe (2000) observed that fourth-instar larvae of *C. tentans* were much larger than those of *C. riparius*. To our knowledge, this study is the first reporting the bioaccumulation of PCB153 from several potential food sources for gammarids. Previous studies with gammarids examined PCB uptake from water exposure (Sanders and Chandler, 1972; Lynch and Johnson, 1982), whereas accumulation through diet remains unexplored (Pinkney, 1985).

Statistical model selection is commonly based on the parsimony principle, by which hypotheses should be kept as simple as possible. The idea is that by adding parameters to a model we could improve the fit to some degree, but at the same time parameter estimates worsen because there is less information available per parameter. In addition, the computations typically require more time by adding parameters. In this study, we applied this principle when similar DICs were obtained, so the models with the least parameters were selected.

## *6.2. Modeling and biological implications*

One advantage of the generic model developed in this study is that it allows us to consider each uptake or elimination route as a module to add according to the availability of the data measured. The different models tested can then determine which routes are the most important in the accumulation of contaminant by the organism. Sidney et al. (2016) showed

that for hydrophobic PCBs, particle ingestion was the dominant uptake route whatever species was tested (*C. riparius*, *Hyalella azteca*, *Lumbriculus variegatus* and *Sphaerium corneum*). This is consistent with our results for chironomids and gammarids, which confirmed that sediment is the major PCB contamination source.

The model considering the water and sediment exposure route (Eq. (4)) gave similar results to the model considering only the sediment exposure route (Eq. (6)) (i.e., similar marginal posterior distributions for  $k_s$  and  $k_e$ , predicted concentrations and DIC values). As a consequence, we concluded that not considering water exposure in the chironomid model contributes no more information and confirms that sediment is the major exposure route.

For gammarids, several hypotheses were tested, and the most parsimonious model was the one that did not account for the water exposure route (hypothesis #3). Furthermore, considering or not the transfer of PCB153 from sediment to leaves in E2 condition did not change the value and the precision of the parameters (hypothesis #4 and #3, Table S2), demonstrating that these data do not contribute additional valuable information. Indeed, previous experiments with perfluorinated alkyl compounds hypothesized that due to gammarid activity, suspended particles of contaminated sediment were deposited on the surface of leaves and were then ingested by gammarids (Bertin, 2016). These results confirm that sediment is the major contamination source.

### *6.3. Species comparison*

Model parameter estimates are therefore consistent with the life traits of these two species: an uptake rate from sediment approximately ten times higher for chironomids (living in the sediment) than gammarids (living at the surface of sediment) and similar elimination rate constants (Fig S4).

### *6.4. Model limitations and implications for future use*

Two limits could be highlighted in the use of the Bayesian approach in TK models: (i) the complexity of computation and (ii) the choice of prior distributions. Due to its complexity, the use of the Bayesian approach is limited in TK models. However, for invertebrates, TK models

could be considered simpler and its Bayesian computation also stem from the organism being considered a unique compartment. The majority of the problems encountered in the Bayesian approach have resulted from the choice of prior distributions. It is crucial to define prior distributions according to previous data similar to the experiment rather than to consider prior distributions selected in this paper as “generic.” Nevertheless, considering previous data (*a priori*) in the TK model could be useful to limit the cost and the number of experiments.

Further investigations will apply this Bayesian framework to more complex processes, including biotransformation or concentration dependency, where the frequentist approach could have limitations. Moreover, calculating prediction intervals has an advantage in that they can also be calculated around simulations with fluctuating input concentrations, even for scenarios that differ from the calibration experiments, and in that the correlation among parameters is accounted for (Ashauer et al., 2010). In particular for European regulations, this approach could make it possible to predict the concentration in the biota because the concentration in sediment is known, and uncertainties are thus accounted for.

## 7. Conclusions and perspectives

In this paper, we proposed a generic TK model in a Bayesian framework to estimate toxicokinetic parameters. This approach could be useful in order to calculate a more accurate estimation of prediction uncertainty. We demonstrated the ability of Bayesian analysis to simultaneously estimate model parameters considering several exposure routes from a PCB153 environmental exposure experiment under controlled conditions in two invertebrate species, *C. riparius* and *G. fossarum*. From a methodological point of view, this paper illustrates that considering the correlation between parameters and different types of data is a real added value for TK modeling. We demonstrated the ability of a generic TK model to consider uptake and elimination routes as modules that can be added, depending to the availability of measured data. From an ecotoxicological point of view, we showed differences in PCB153 bioaccumulation between chironomids and gammarids, which could be explained by the different life traits of these two organisms. We also confirmed that sediment is the major route of exposure for invertebrates exposed to highly hydrophobic organic contaminants. Further investigations will apply this Bayesian framework to other benthic invertebrate species, and other organic contaminants, so as to address more complex processes, including biotransformation.

## Acknowledgments

This study was supported by the French Agency for Biodiversity. The authors are thankful to Grand Parc Miribel-Jonage for providing the sediment used in experiments. The authors are also grateful to Marie-Laure Delignette-Muller (LBBE, UMR CNRS 5558, France) for Bayesian inference discussions as well as Myriam Arhror and Matthieu Masson (Irstea Lyon, France) for sediment characteristic analysis. The authors are also grateful to anonymous reviewers and to Ms. L. Northrup for editing the manuscript.

## References

- AFNOR. 2010. Determination of the Toxicity of Fresh Water Sediments to Chironomous Riparius - Part 1: Natural Sediments Water Quality.
- Amiard, J. C., C. Amiard-Triquet, B. Berthet, and C. Metayer. 1987. Comparative study of the patterns of bioaccumulation of essential (Cu, Zn) and non-essential (Cd, Pb) trace metals in various estuarine and coastal organisms. *Journal of Experimental Marine Biology and Ecology* **106**:73-89.
- Ashauer, R., A. Hintermeister, I. Caravatti, A. Kretschmann, and B. I. Escher. 2010. Toxicokinetic and toxicodynamic modeling explains carry-over toxicity from exposure to diazinon by slow organism recovery. *Environmental Science and Technology* **44**:3963-3971.
- Bernillon, P. and F. Y. Bois. 2000. Statistical issues in toxicokinetic modeling: A Bayesian perspective. *Environmental Health Perspectives* **108**:883-893.
- Bertin, D., B. J. D. Ferrari, P. Labadie, A. Sapin, J. Garric, H. Budzinski, M. Houde, and M. Babut. 2014. Bioaccumulation of perfluoroalkyl compounds in midge (*Chironomus riparius*) larvae exposed to sediment. *Environmental Pollution* **189**:27-34.
- Bertin, D., P. Labadie, B. J. D. Ferrari, A. Sapin, J. Garric, O. Geffard, H. Budzinski, and M. Babut. 2016. Potential exposure routes and accumulation kinetics for poly- and perfluorinated alkyl compounds for a freshwater amphipod: *Gammarus* spp. (Crustacea). *Chemosphere* **155**:380-387.
- Bois, F. Y. and C. Brochot. 2016. Modeling Pharmacokinetics. Pages 37-62 in E. Benfenati, editor. *In Silico Methods for Predicting Drug Toxicity*. Springer New York, New York, NY.
- Carrasco-Navarro, V., I. Jæger, J. O. Honkanen, J. V. K. Kukkonen, J. Carroll, and L. Camus. 2015. Bioconcentration, biotransformation and elimination of pyrene in the arctic crustacean *Gammarus setosus* (Amphipoda) at two temperatures. *Marine Environmental Research* **110**:101-109.
- Englert D, Zubrod JP, Pietz S, Stefani S, Krauss M, Schulz R, et al. 2017. Relative importance of dietary uptake and waterborne exposure for a leaf-shredding amphipod exposed to thiacloprid-contaminated leaves. *Scientific Reports* 7:16182.
- EPA, E.P.A.U. 2006. Approaches for the Application of Physiologically Based Pharmacokinetic (PBPK) Models and Supporting Data in Risk Assessment (Final Report) (Washington, D.C.).
- Forrow, D. M. and L. Maltby. 2000. Toward a mechanistic understanding of contaminant-induced changes in detritus processing in streams: Direct and indirect effects on detritivore feeding. *Environmental Toxicology and Chemistry* **19**:2100-2106.
- Galic, N. and V. E. Forbes. 2017. Effects of temperature on the performance of a freshwater amphipod. *Hydrobiologia* **785**:35-46.
- Gelman, A., J. Carlin, H. Stern, and D. Rubin. 1995. *Bayesian Data Analysis*, New York, NY, USA.

- Grech, A., C. Brochot, J. L. Dorne, N. Quignot, F. Y. Bois, and R. Beaudouin. 2017. Toxicokinetic models and related tools in environmental risk assessment of chemicals. *Science of the Total Environment* **578**:1-15.
- Gross-Sorokin, M. Y., E. P. M. Grist, M. Cooke, and M. Crane. 2003. Uptake and depuration of 4-nonylphenol by the benthic invertebrate *Gammarus pulex*: How important is feeding rate? *Environmental Science and Technology* **37**:2236-2241.
- IPCS, I.P.o.C.S. 2010. Characterization and Application of physiologically Based Pharmacokinetic Models in Risk Assessment (Geneva, Switzerland).
- Lambert, P. C., A. J. Sutton, P. R. Burton, K. R. Abrams, and D. R. Jones. 2005. How vague is vague? A simulation study of the impact of the use of vague prior distributions in MCMC using WinBUGS. *Statistics in Medicine* **24**:2401-2428.
- Landrum, P. F., M. J. Lydy, and H. Lee. 1992. Toxicokinetics in aquatic systems: Model comparisons and use in hazard assessment. *Environmental Toxicology and Chemistry* **11**:1709-1725.
- Lebrun, J. D., M. Perret, E. Uher, M. H. Tusseau-Vuillemin, and C. Gourlay-Francé. 2011. Waterborne nickel bioaccumulation in *Gammarus pulex*: Comparison of mechanistic models and influence of water cationic composition. *Aquatic Toxicology* **104**:161-167.
- Leppänen, M. T. and J. V. K. Kukkonen. 1998. Relative importance of ingested sediment and pore water as bioaccumulation routes for pyrene to oligochaete (*Lumbriculus variegatus*, Muller). *Environmental Science and Technology* **32**:1503-1508.
- Lin, H. I., D. W. Berzins, L. Myers, W. J. George, A. Abdelghani, and K. H. Watanabe. 2004. A Bayesian approach to parameter estimation for a crayfish (*Procambarus* spp.) bioaccumulation model. *Environmental Toxicology and Chemistry* **23**:2259-2266.
- López-Doval, J. C., J. V. K. Kukkonen, P. Rodrigo, and I. Muñoz. 2012. Effects of indomethacin and propranolol on *Chironomus riparius* and *Physella* (*Costatella*) *acuta*. *Ecotoxicology and Environmental Safety* **78**:110-115.
- Lydy, M. J., J. B. Belden, and M. A. Ternes. 1999. Effects of temperature on the toxicity of M-parathion, chlorpyrifos, and pentachlorobenzene to *Chironomus tentans*. *Archives of Environmental Contamination and Toxicology* **37**:542-547.
- Lynch, T. and H. Johnson. 1982. Availability of a hexachlorobiphenyl isomer to benthic amphipods from experimentally contaminated natural sediments. American Society for Testing and Materials, STP 766, Philadelphia, PA.
- Mackay, D. and A. Fraser. 2000. Bioaccumulation of persistent organic chemicals: Mechanisms and models. *Environmental Pollution* **110**:375-391.
- Macneil, C., J. T. A. Dick, and R. W. Elwood. 1999. The dynamics of predation on *Gammarus* spp. (Crustacea: Amphipoda). *Biological Reviews* **74**:375-395.
- Mamy, L., D. Patureau, E. Barriuso, C. Bedos, F. Bessac, X. Louchart, F. Martin-Laurent, C. Miege, and P. Benoit. 2015. Prediction of the fate of organic compounds in the environment from their molecular properties: A review. *Critical Reviews in Environmental Science and Technology* **45**:1277-1377.

- Miller, T. H., G. L. McEneff, L. C. Stott, S. F. Owen, N. R. Bury, and L. P. Barron. 2016. Assessing the Reliability of Uptake and Elimination Kinetics Modelling Approaches for Estimating Bioconcentration Factors in the Freshwater Invertebrate, *Gammarus Pulex*. *Science of the Total Environment* **547**:396-404.
- Miller, T. H., N. R. Bury, S. F. Owen, and L. P. Barron. 2017. Uptake, biotransformation and elimination of selected pharmaceuticals in a freshwater invertebrate measured using liquid chromatography tandem mass spectrometry. *Chemosphere* **183**:389-400.
- OECD. 2004. Water Chironomid Toxicity Test Using Spiked Sediment. In: OECD Guidelines for the Testing of Chemicals. 1-21.
- OECD. 2008. Test No. 315: Bioaccumulation in Sediment-dwelling Benthic Oligochaetes. In: OECD Guidelines for the Testing of Chemicals. Section 3. 1-33.
- Pavan, M., A.P. Worth, and T.I. Netzeva. 2006. Review of QSAR Models for Bioconcentration. European Commission (Joint Research Centre).
- Péry, A. R. R., R. Mons, P. Flammarion, L. Lagadic, and J. Garric. 2002. A modeling approach to link food availability, growth, emergence, and reproduction for the midge *Chironomus riparius*. *Environmental Toxicology and Chemistry* **21**:2507-2513.
- Pinkney, A. E., G. V. Poje, R. M. Sansur, C. C. Lee, and J. M. O'Connor. 1985. Uptake and Retention of <sup>14</sup>C-Aroclor® 1254 in the Amphipod, *Gammarus tigrinus*, Fed Contaminated Fungus, *Fusarium oxysporum*. *Archives of Environmental Contamination and Toxicology* **14**:59-64.
- Plummer, M. 2016. rjags: Bayesian Graphical Models using MCMC.
- R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
- Raftery, A. E. and S. M. Lewis. 1992. [Practical Markov Chain Monte Carlo]: Comment: One Long Run with Diagnostics: Implementation Strategies for Markov Chain Monte Carlo. *Statist. Sci.* **7**:493-497.
- Richards, R. G. and M. Chaloupka. 2009. Temperature-dependent bioaccumulation of copper in an estuarine oyster. *Science of the Total Environment* **407**:5901-5906.
- Rösch, A., M. Gottardi, C. Vignet, N. Cedergreen, and J. Hollender. 2017. Mechanistic Understanding of the Synergistic Potential of Azole Fungicides in the Aquatic Invertebrate *Gammarus pulex*. *Environmental Science and Technology* **51**:12784-12795.
- Sanders, H. O. and J. H. Chandler. 1972. Biological magnification of a polychlorinated biphenyl (Aroclor® 1254) from water by aquatic invertebrates. *Bulletin of Environmental Contamination and Toxicology* **7**:257-263.
- Schuler, L. J., M. Wheeler, A. J. Bailer, and M. J. Lydy. 2003. Toxicokinetics of sediment-sorbed benzo[a]pyrene and hexachlorobiphenyl using the freshwater invertebrates *Hyalella azteca*, *Chironomus tentans*, and *Lumbriculus variegatus*. *Environmental Toxicology and Chemistry* **22**:439-449.



- Sidney, L. A., N. J. Diepens, X. Guo, and A. A. Koelmans. 2016. Trait-based modelling of bioaccumulation by freshwater benthic invertebrates. *Aquatic Toxicology* **176**:88-96.
- Spiegelhalter, D. J., N. G. Best, B. P. Carlin, and A. V. D. Linde. 2002. Bayesian measures of model complexity and fit. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* **64**:583-639.
- von Bertalanffy, L. 1938. A quantitative theory of organic growth (inquires on growth laws. II). *Human Biology* **10**:181-213.
- Watts, M. M. and D. Pascoe. 2000. A comparative study of *Chironomus riparius* meigen and *Chironomus tentans* fabricius (Diptera:Chironomidae) in aquatic toxicity tests. *Archives of Environmental Contamination and Toxicology* 39:299-306.OCDE. *Test No. 315: Bioaccumulation in Sediment-Dwelling Benthic Oligochaetes*. 2008. doi:doi:<https://doi.org/10.1787/9789264067516-en>.

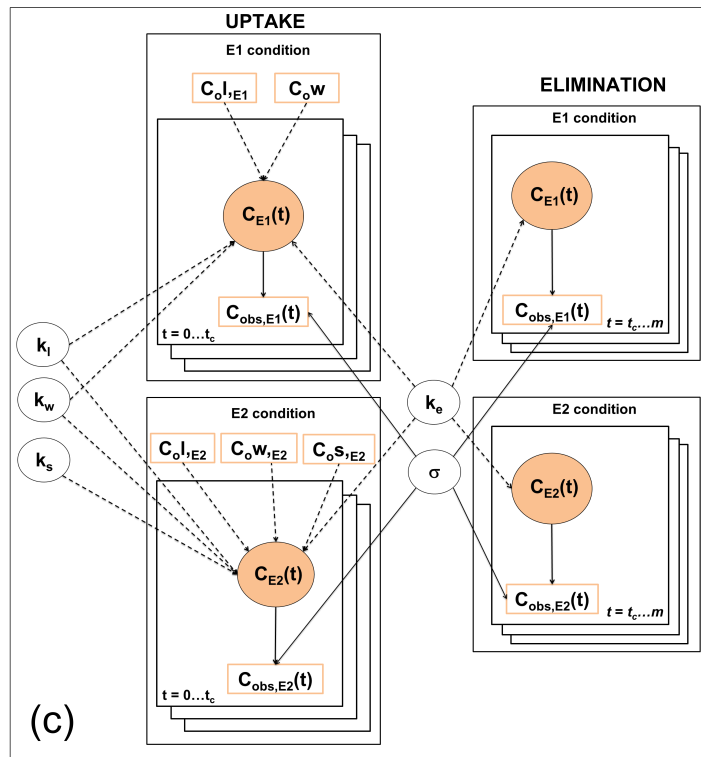
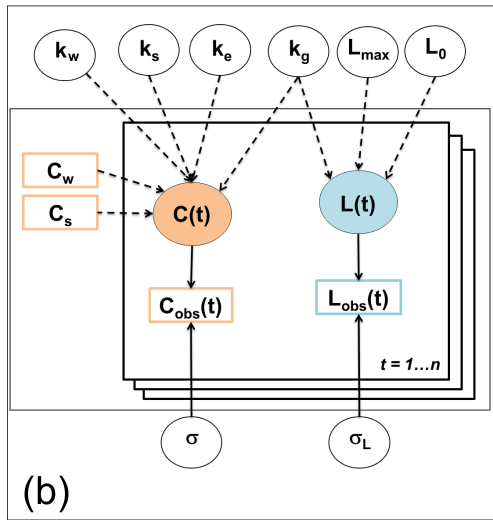
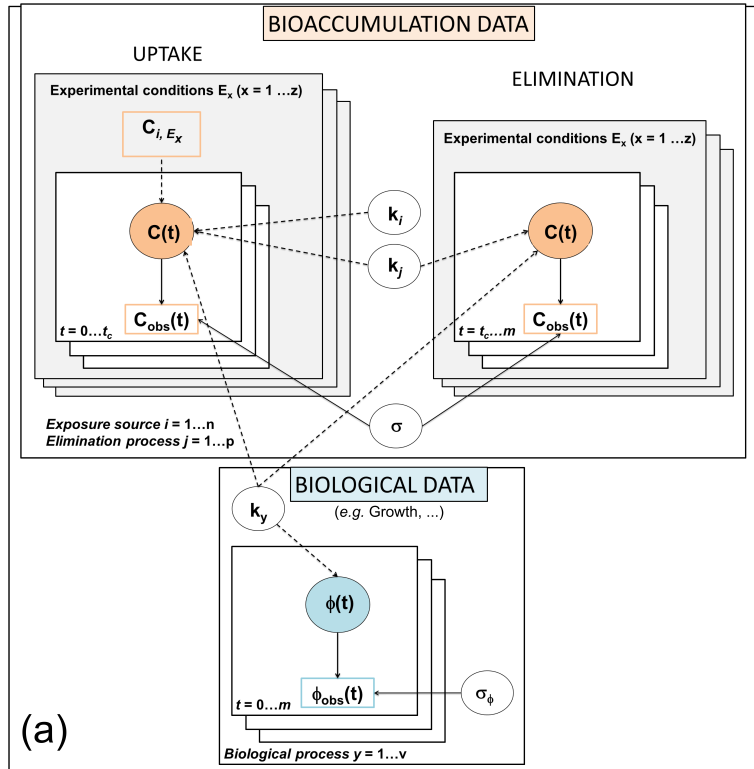
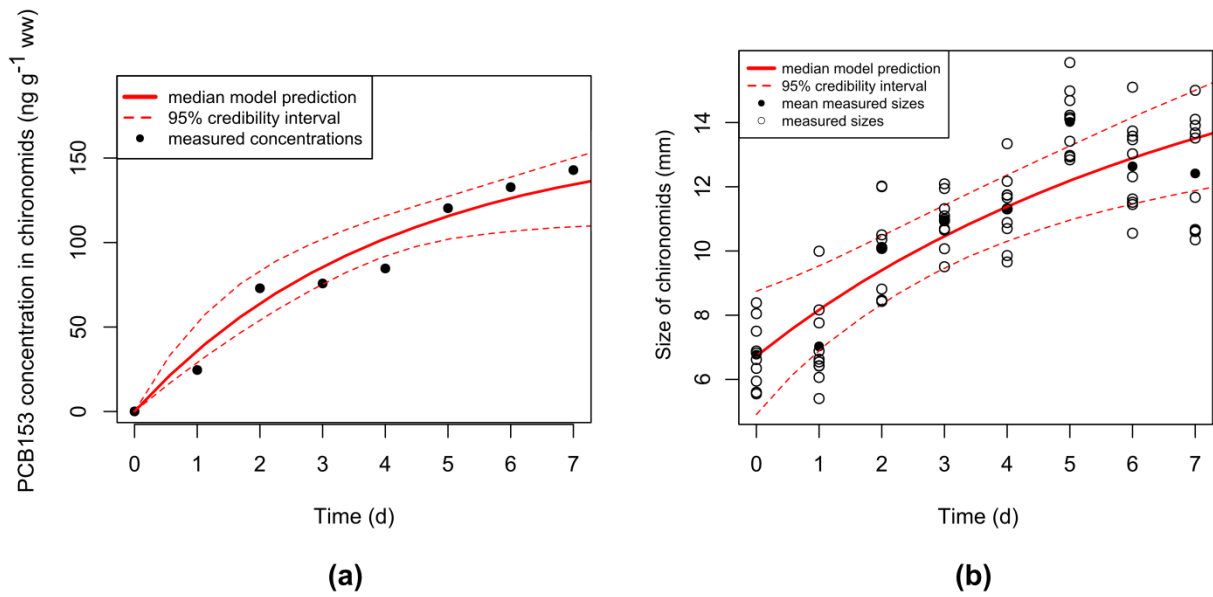
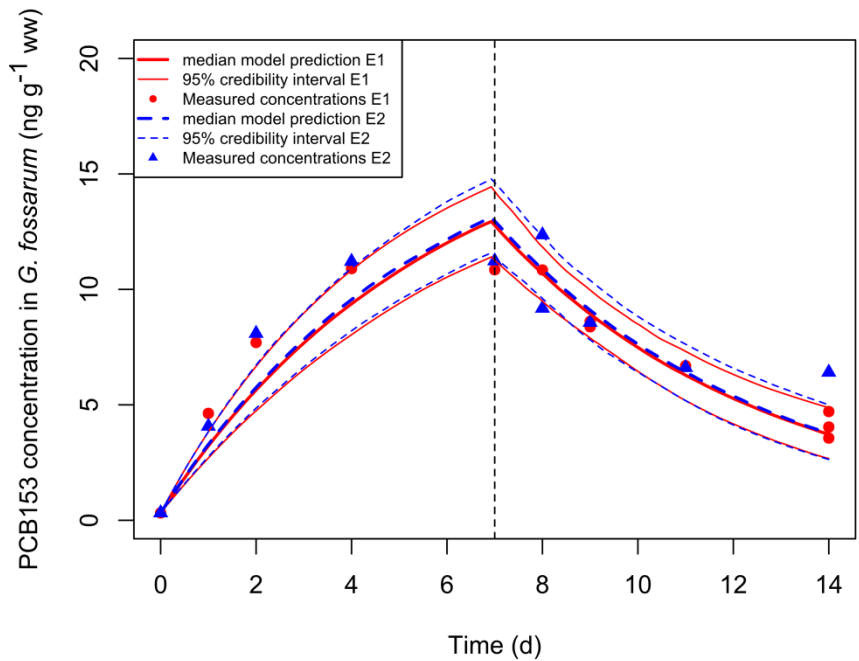


Figure 1. Directed acyclic graph (DAG) for (a) generic (b) chronomid and (c) gammarid models. Observed variables, such as the contaminant concentration in organisms, sediment and leaves, are represented by rectangle nodes. Model parameters and variables are represented by circular nodes. Dotted arrows represent deterministic links (Eqs. (1), (4) and (7)), while solid arrows represent stochastic links between predicted and observed data.



**Figure 2.** Observed data (dots) and model predictions (solid and dashed lines) for (a) PCB153 concentrations ( $\text{ng g}_{\text{org}}^{-1}$ ) in chironomids (Eq. (6)) and (b) chironomid size (mm) (Eq. (5)) from days 0 to 7. The observed data are single values ( $n=1$ ). On Figure 2-b, mean sizes at each day are symbolized by filled dots.



**Figure 3.** Measured (dots and triangles) and predicted (solid and dashed curves, from Eqs. (12) and (14)) PCB153 concentrations (ng g<sub>org</sub><sup>-1</sup>) in gammarids during the uptake (days 0–7) and the elimination (days 7–14) phases (separated with the black dashed vertical line). The measured data are single values (n=1).

**Table 1.** Summary of the toxicokinetic model parameters: priors, median and percentile values determining the 95% credibility interval.

Organisms	Parameters	Priors	Median	Percentiles	
				2.5%	97.5%
<i>Chironomus riparius</i>	$k_s$	Log <sub>10</sub> -normal (0.113, 5)	<b>0.473</b>	0.359	0.804
	$k_e$	Log <sub>10</sub> -normal (-0.578, 5)	<b>0.121</b>	0.041	0.392
	$k_g$	Log <sub>10</sub> -normal (0.236, 2)	<b>0.123</b>	0.029	0.356
	$L_0$	Log <sub>10</sub> -normal (0.778, 0.64)	<b>6.75</b>	4.85	8.71
	$L_{max}$	Log <sub>10</sub> -normal (1.056, 0.64)	<b>18.5</b>	13.3	40.7
	$\sigma$	Gamma (0.001, 0.001)	<b>11.8</b>	6.53	27.5
	$\sigma_L$	Gamma (0.001, 0.001)	<b>1.12</b>	0.672	2.18
<i>Gammarus fossarum</i>	$k_l$	Log <sub>10</sub> -unif (-5, 2)	<b>0.013</b>	0.010	0.016
	$k_s$	Log <sub>10</sub> -unif (-5, 2)	<b>0.071</b>	0.057	0.087
	$k_e$	Log <sub>10</sub> -unif (-5, 2)	<b>0.178</b>	0.131	0.229
	$\sigma$	Gamma (0.001, 0.001)	<b>1.48</b>	1.11	2.12

**Table 2.** Summary of PCB153 concentrations in water, leaves and sediment according to gammarid experiments (control, E1, E2 and E3). Standard deviations (sd) are calculated on analytical replicates (n=2 when sd are given, except for leaves in E3 condition at day 7 and for sediment in E2 condition at day 0 where n=3).

Days	Water (ng L <sup>-1</sup> )			Leaves (ng g <sub>leaves</sub> <sup>-1</sup> )			Sediment (ng g <sub>sed</sub> <sup>-1</sup> )		
	0	7	14	0	7	14	0	7	14
<b>Control</b>	0.10	-	0.10	1.46 ± 0.18	1.13	1.09	0.29 ± 0.22	-	0.21 ± 0.11
<b>E1<sup>a</sup></b>	0.30	0.10	0.10	260 ± 6.07	245 ± 30.1	3.86	-	-	-
<b>E2<sup>b</sup></b>	0.10	0.20	0.00	1.46 ± 0.18	44.9 ± 12.7	1.72 ± 0.01	56.6 ± 8.68	30.0 ± 3.12	0.23 ± 0.12
<b>E3<sup>c</sup></b>	0.10	0.80	-	1.46 ± 0.18	5.52 ± 0.87	-	56.6 ± 8.68	45.1 ± 1.44	-

<sup>a</sup> spiked leaves; <sup>b</sup> spiked sediment; <sup>c</sup> spiked sediment without gammarids