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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.

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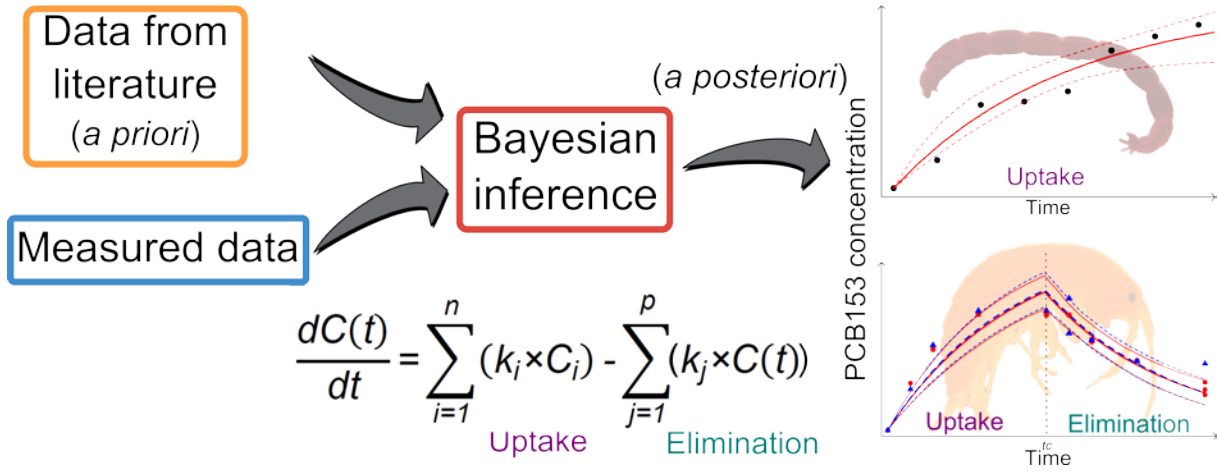
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1 Highlights

- 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 (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 (d^{-1}) and k_g the growth rate (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_{max} is the asymptotic size (mm), L_0 is the
 123 size at birth (mm) and k_g is the growth rate (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 (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⁻¹ 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⁻¹ for water, between 0.003
152 and 0.040 ng g⁻¹ 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⁻¹ 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_{sed} 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_{dw} 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_{PCB153} kg⁻¹_{leaf,ww} with a solution of 5 µg L⁻¹ 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 ± 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 IEEF 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, $5^{\circ}26'22''\text{E}$ -
204 $45^{\circ}95'79''\text{N}$). 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 ± 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 ± 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/ g_{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% H₂SO₄ 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 μ 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 μ 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 μ m). Helium (vector gas, 1.3 mL min⁻¹) and nitrogen
252 (auxiliary gas, 60.0 mL min⁻¹) were used. The injector temperature was set at 280 °C and the
253 detector temperature at 300 °C. The kiln temperature program was: 90 °C for 2 min (80 °C for
254 water samples), 15 °C min⁻¹ to 178 °C (20 °C min⁻¹ to 190°C), 2 °C min⁻¹ to 230 °C (210 °C),
255 30 °C min⁻¹ (15 °C min⁻¹) to 300 °C, 300 °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 α 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 σ 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 σ_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 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 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 d^{-1})
298 according to Péry et al. (2002), corresponding to a growth rate of 1.72 d^{-1} . Priors for L_0 and
299 L_{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 40th 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 ± 0.25 and 83.3 ± 20.8 ng g_{sed}⁻¹ versus 100 ng g_{sed}⁻¹ (dw) expected. At the end of the experiment, the PCB153 concentration in spiked sediment was 89.6 ± 22.4 ng g_{sed}⁻¹ (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 ± 0.009 ng L⁻¹ and 0.230 ± 0.025 ng L⁻¹ in the control and exposed aquaria, respectively, while they were 0.259 ± 0.028 ng L⁻¹ and 3.85 ± 0.420 ng L⁻¹, respectively, at the end of the experiment.

PCB153 concentrations in *C. riparius* exposed to spiked sediment increased from 0.089 ± 0.031 to 142 ± 50.0 ng g_{org}⁻¹ 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 ± 0.110 to 10.9 ± 3.80 and 11.2 ± 3.93 ng g_{org}⁻¹, 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 ± 1.65 and 6.41 ± 2.24 ng g_{org}⁻¹, 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.

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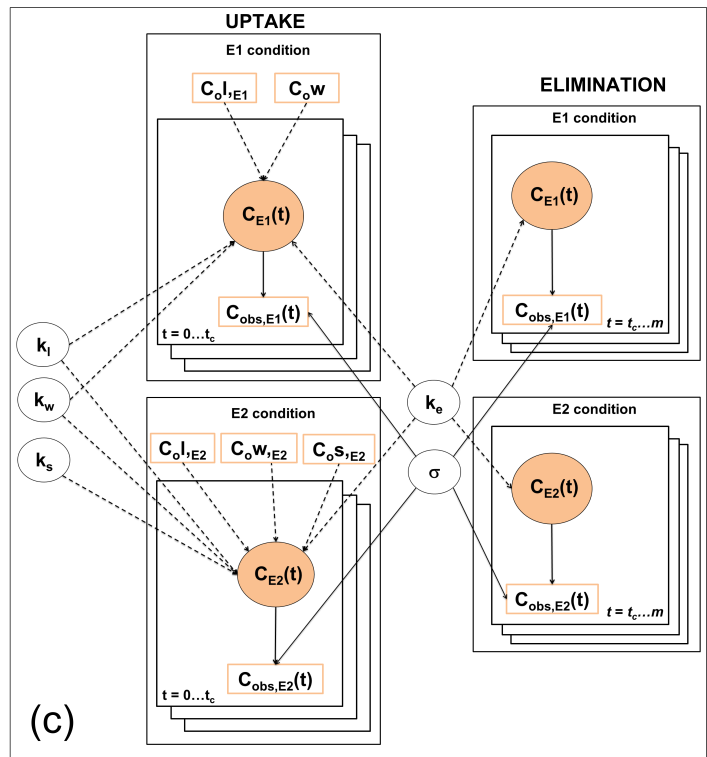
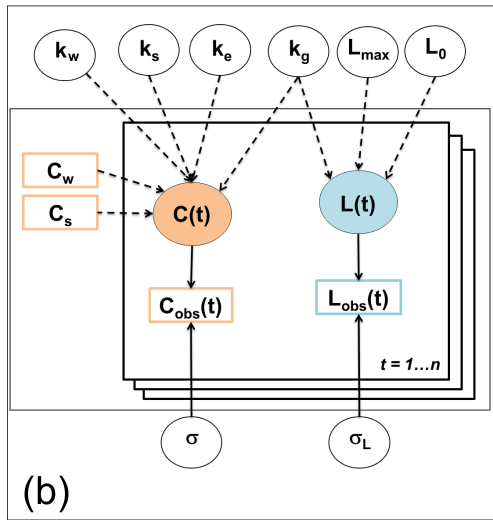
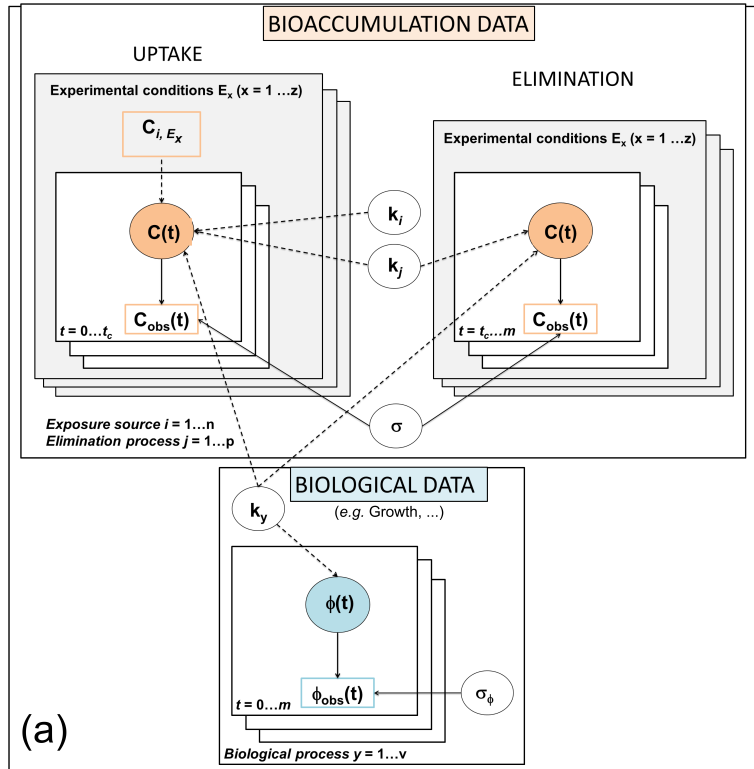


Figure 1. Directed acyclic graph (DAG) for (a) generic (b) chronomid and (c) gammarid models. Observed variables, such as the contaminant concentration in organisms, sediment and leaves, are represented by rectangle nodes. Model parameters and variables are represented by circular nodes. Dotted arrows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

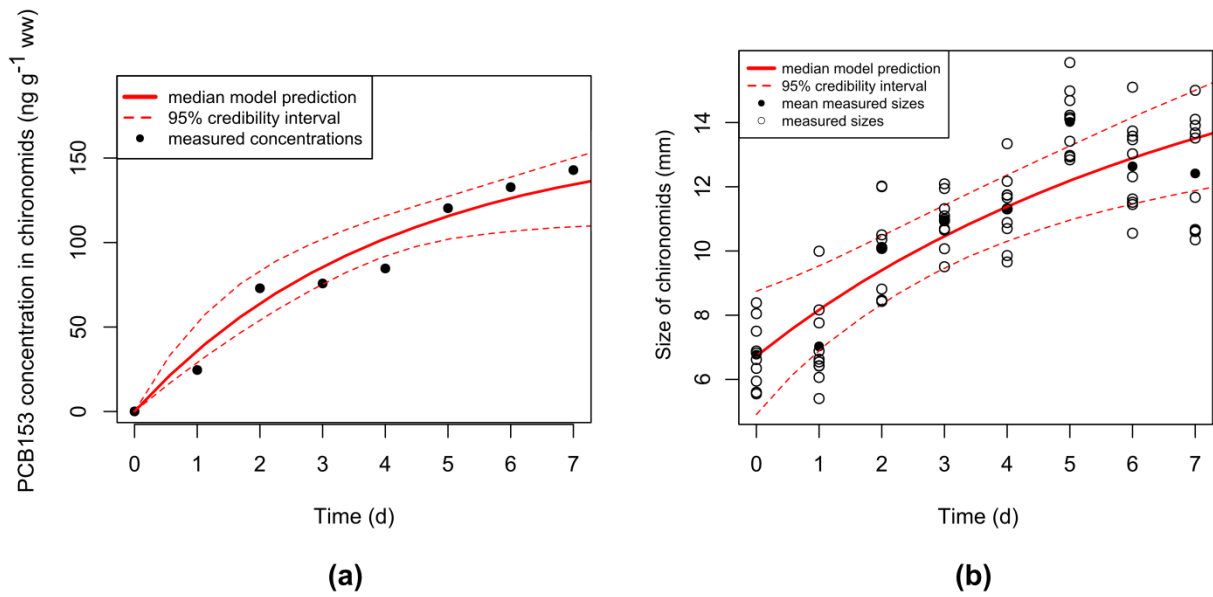


Figure 2. Observed data (dots) and model predictions (solid and dashed lines) for (a) PCB153 concentrations ($\text{ng g}_{\text{org}}^{-1}$) in chironomids (Eq. (6)) and (b) chironomid size (mm) (Eq. (5)) from days 0 to 7. The observed data are single values ($n=1$). On Figure 2-b, mean sizes at each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

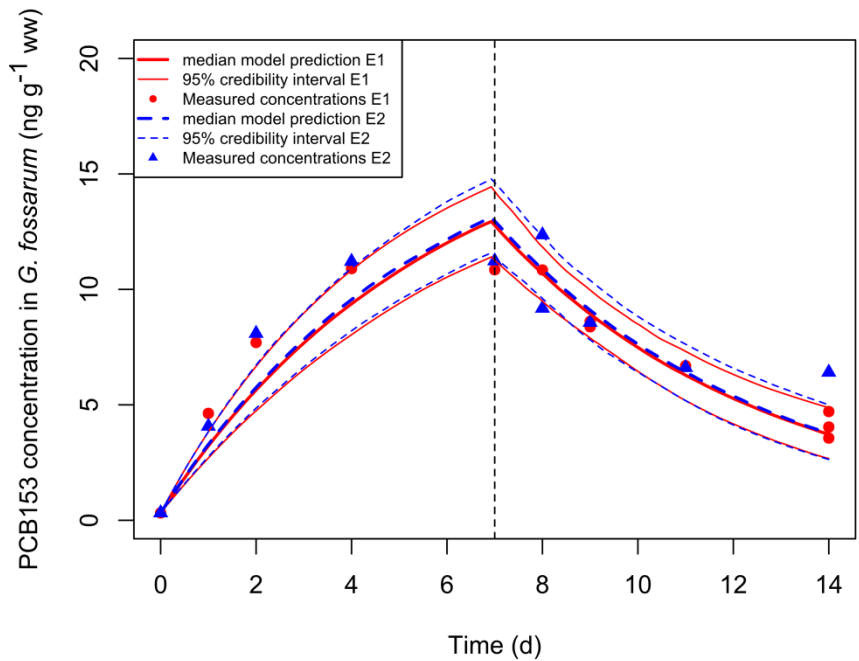


Figure 3. Measured (dots and triangles) and predicted (solid and dashed curves, from Eqs. (12) and (14)) PCB153 concentrations (ng g_{org}⁻¹) in gammarids during the uptake (days 0–7) and the elimination (days 7–14) phases (separated with the black dashed vertical line). The measured data are single 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 ₁₀ -normal (0.113, 5)	0.473	0.359	0.804
	k_e	Log ₁₀ -normal (-0.578, 5)	0.121	0.041	0.392
	k_g	Log ₁₀ -normal (0.236, 2)	0.123	0.029	0.356
	L_0	Log ₁₀ -normal (0.778, 0.64)	6.75	4.85	8.71
	L_{max}	Log ₁₀ -normal (1.056, 0.64)	18.5	13.3	40.7
	σ	Gamma (0.001, 0.001)	11.8	6.53	27.5
	σ_L	Gamma (0.001, 0.001)	1.12	0.672	2.18
<i>Gammarus fossarum</i>	k_l	Log ₁₀ -unif (-5, 2)	0.013	0.010	0.016
	k_s	Log ₁₀ -unif (-5, 2)	0.071	0.057	0.087
	k_e	Log ₁₀ -unif (-5, 2)	0.178	0.131	0.229
	σ	Gamma (0.001, 0.001)	1.48	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 ⁻¹)			Leaves (ng g _{leaves} ⁻¹)			Sediment (ng g _{sed} ⁻¹)		
	0	7	14	0	7	14	0	7	14
Control	0.10	-	0.10	1.46 ± 0.18	1.13	1.09	0.29 ± 0.22	-	0.21 ± 0.11
E1^a	0.30	0.10	0.10	260 ± 6.07	245 ± 30.1	3.86	-	-	-
E2^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
E3^c	0.10	0.80	-	1.46 ± 0.18	5.52 ± 0.87	-	56.6 ± 8.68	45.1 ± 1.44	-

^a spiked leaves; ^b spiked sediment; ^c spiked sediment without gammarids