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## **Life-history phenology strongly influences population vulnerability to toxicants: a case study with the mudsnail *Potamopyrgus antipodarum*.**

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1 *Life-history phenology strongly influences population vulnerability to toxicants: a case*  
2 *study with the mudsnail Potamopyrgus antipodarum*

3

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12 **Abstract**

13           One of the main objectives of ecological risk assessment is to evaluate the effects of  
14 toxicants on ecologically relevant biological systems such as populations or communities.  
15 However, the effects of toxicants are commonly measured on selected sub-individual or  
16 individual endpoints due to their specificity against chemical stressors. Introducing these  
17 effects into population models is a promising way to predict impacts on populations. Yet  
18 currently employed models are very simplistic and their environmental relevance needs to be  
19 improved to establish the ecological relevance of hazard assessment. This study with the  
20 gastropod *Potamopyrgus antipodarum* combines a field experimental approach with a  
21 modelling framework. It clarifies the role played by seasonal variability of life-history traits in  
22 the population's vulnerability to the alteration of individual performance, potentially caused  
23 by toxic stress. The study comprised three steps: (i) characterization of the seasonal variability  
24 of the life-history traits of a local population over 1 year with *in situ* experiments on caged  
25 snails, coupled with a demographic follow-up, (ii) development of a periodic matrix  
26 population model which visualizes the monthly variability of population dynamics, and (iii)  
27 simulation of the demographic consequences of an alteration of life-history traits (*i.e.*,  
28 fertility, juvenile and adult survival). The results revealed that demographic impacts strongly  
29 depend on the season when alterations of individual performance occur. Model analysis  
30 showed that this seasonal variability of population vulnerability is strongly related to the  
31 phenology of the population. We underline that improving the realism of population models is  
32 a major objective for ecological risk assessment, and that taking into account species  
33 phenology in modelling approaches should be a priority.

34

35 Keywords (limit 5 keywords):

36 Phenology, life-history, *in situ* caging, ecological risk assessment, population modelling

## 37 **1. Introduction**

38 Biological effects of toxicants are most frequently assessed in terms of alteration of  
39 sub-individual or individual performance (*e.g.*, biomarkers) by means of bioassays or field  
40 monitoring. Nevertheless, for ecological risk assessment, these impacts on populations,  
41 communities and ecosystems are of primary concern [1-8]. Unfortunately, supplying an  
42 ecologically relevant assessment by measuring toxic impacts directly on these complex  
43 integrated systems is challenging, particularly because distinguishing toxicant impacts from  
44 the effects of other environmental factors or anthropogenic stressors can be difficult. One  
45 alternative methodology links the effects observed on individual-level endpoints measured in  
46 toxicity tests (in the laboratory or during *in situ* experiments) to impacts on populations [9-  
47 11]. However, a major problem is the complex relationship between the effects measured at  
48 the individual level and outcomes occurring at the population level [12-14]. Introducing the  
49 effects of toxicants on demographic parameters (*i.e.*, related to life-history traits) into  
50 population models is one way to investigate this relation and to anticipate impacts at the  
51 population level [10, 15-16]. In fact, such mechanistic ecological models provide a  
52 quantitative way of integrating multiple individual-level endpoints, including survival, growth  
53 and reproduction, in projections of population-level consequences such as changes in  
54 population abundance or growth rate [17].

55 A large number of studies have demonstrated the value of using demographic models  
56 in an ecotoxicological context (reviewed in Galic et al. 2010). However, despite attempts to  
57 use more elaborate modelling approaches that integrate environmental complexity [18-23],  
58 the ecological relevance of the population models involved is generally low. In fact, in the  
59 great majority of studies, demographic models are based on laboratory assays with species  
60 that are not representative of the ecosystems of interest (*e.g.* tropical fishes or daphnids for  
61 studies on lotic and temperate systems). These models are popular and useful tools for

62 isolating the effects of a toxicant on the population growth rate [7], but they could fail to  
63 understand what occurs in the field on local populations [24]. In fact, the determinants of  
64 demographic sensitivity of populations are subject to strong variability (*e.g.*, interspecies and  
65 interpopulation variability of life cycle). This variability cannot be taken into account with  
66 overly simple demographic models [6, 18, 25-26]. Notably, these models do not capture the  
67 seasonal variability of population dynamics, even though several studies have underlined a  
68 seasonal variability of population vulnerability. A study on the crustacean amphipod  
69 *Corophium volutator* [27] showed that the same level of mortality impacts the population  
70 differently depending on the season at which mortality occurs. Similarly, in field studies on  
71 the amphipod *Leptocheirus plumulosus*, McGee and Spencer [28-29] highlighted a strong  
72 monthly variability of the sensitivity of the population growth rate to the different life-history  
73 traits. Thus, we need to develop population models which can integrate the seasonal  
74 variability of population dynamics to propose a more ecologically relevant assessment of the  
75 impact of toxicants on freshwater populations.

76 In the present study, we illustrate the strength of a field experimental approach (*in situ*  
77 caging and demographic follow-up) with a modelling framework (periodic matrix model) to  
78 decipher the role of seasonality in the vulnerability of populations to toxicants. The model  
79 organism is the widespread mudsnail *Potamopyrgus antipodarum* (Gray). We selected this  
80 species due to its sensitivity to a large range of chemicals for ecotoxicological tests in the  
81 laboratory [30-34] or in the field [35-36]. Notably, *P. antipodarum* is proposed to the  
82 Organization for Economic Cooperation and Development (OECD) as a relevant test species  
83 to assess the impacts of endocrine disruptors on freshwater molluscs [37]. In the first step we  
84 characterized the seasonal variability of the life-history traits of a population of *P.*  
85 *antipodarum* over 1 year, with experiments on caged snails and a demographic follow-up of  
86 the population. In the second step, we developed periodic matrix population models which

87 allowed us to describe the demographic changes of this population and to picture the monthly  
88 variability of its dynamics. In the third step, we simulated the demographic consequences for  
89 this population of an alteration of life-history traits (*e.g.*, reduction in fertility, juvenile or  
90 adult survival), in order to illustrate the importance of the seasonal variability of population  
91 dynamics for ecological risk assessment of chemicals.

92

## 93 **2. Material and methods**

### 94 2.1. Biological data

95 *P. antipodarum* is a deposit-feeding gastropod which in Europe reproduces mainly by  
96 parthenogenesis of female populations [38]. We conducted a battery of experiments on a  
97 population on the Upper Rhône, in the Villebois reservoir (05°27'55.3 E; 45°46'30.7 N,  
98 Rhône, France). This site was selected because it contains durable populations of freshwater  
99 molluscs monitored for more than one decade [39] and because it was accessible all year  
100 round (the water-level fluctuations do not exceed 0.50 m during the year). Temperature was  
101 continuously recorded every 2 h using the Tinytag Aquatic 2<sup>®</sup> temperature logger.

102 To characterize the different life-history traits and the population dynamics of *P.*  
103 *antipodarum*, we used an *in situ* approach. Firstly, we conducted experiments with snails  
104 caged on the study site at different seasons for life-history trait quantification: fertility (*i.e.*,  
105 number of neonates produced per reproductive female per day) and growth (*i.e.*, increase of  
106 shell length (SL) of juveniles and adults). Secondly, we carried out a demographic follow-up  
107 based on a monthly population census to estimate the time-course in population  
108 characteristics: densities, SL structure and fecundity (*i.e.*, number of embryos in the brood  
109 pouch).

110

#### 111 2.1.1. *In situ* caging experiments

112 *In situ* caging experiments were conducted from October 2009 to November 2010  
113 during 10 campaigns lasting 21 days. We measured contrasted environmental conditions  
114 between the different campaigns. In this way, mean water temperature, which is known to  
115 strongly influence *P. antipodarum* life-history traits [40-43], varied from 4.6 to 22.7 °C. We  
116 focused our experiments on measuring fertility and growth. During the year, we were able to  
117 study fertility during eight campaigns (insufficient numbers of adults during winter) and  
118 growth during seven campaigns (insufficient numbers of juveniles during three campaigns in  
119 summer and autumn). Snails were sampled and calibrated (*i.e.*, size selection) directly at the  
120 study site. Initial SL was measured each time with a sample of 30 snails. SL was measured  
121 between the apex and the distal extremity of the aperture under the binocular microscope,  
122 which corresponded to the maximal height of the shell. Initial SL varied from 2.28 ( $\pm$  0.16)  
123 mm to 2.85 ( $\pm$  0.34) mm for juveniles and from 4.38 ( $\pm$  0.22) to 4.68 ( $\pm$  0.34) for adults  
124 between the different campaigns.

125 Four replicates of 30 juveniles and four replicates of 50 adults were used for growth  
126 and fertility measurement. Snails were placed in polypropylene cylindrical containers  
127 (diameter, 10 cm; length, 12 cm) with pieces of net (mesh size, 100  $\mu$ m) on perforations to  
128 allow water flow. Because *P. antipodarum* lives in the upper layers of the sediments [41], a 2-  
129 cm layer of sediment removed from the study site and sieved at 315  $\mu$ m (*i.e.*, keeping out  
130 autochthonous snails) was added to the containers. Two containers with sediment but without  
131 snails were deployed for each campaign as control of the absence of autochthonous snails.  
132 Finally, containers were placed in a perforated protective case in polyvinylchloride (PVC)  
133 with fixing elements for the containers. The containers were placed on the bottom of the river,  
134 in close proximity to the site used for the demographic follow-up (see below) at a depth about  
135 1.2 m. After the 21-day exposure period, juveniles and adults were fixed with 20% alcohol  
136 and measured in the laboratory in order to estimate the snails' daily growth rates. Neonates

137 laid during the period were also fixed with 20% alcohol and counted under a binocular  
138 microscope. Then we estimated the fertility rates  $b$  (*i.e.*, number of neonates produced by  
139 snail per day) as follows:

$$140 \quad b_i = \frac{n_i}{((l_{i,0} + l_{i,t}) / 2) \times t} \quad (1)$$

141 where  $b_i$  corresponds to the fertility rate of the replicate  $i$ ;  $n_i$  to the number of neonates laid in  
142 the replicate  $i$ ;  $t$  is the duration in days of the experiment (here  $t = 21$  days for all assays);  $l_{i,0}$   
143 and  $l_{i,t}$  are the number of living snails at the start and at the end of the experiment (here  $l_{i,0} =$   
144 50 for all assays).

145

#### 146 2.1.2. Demographic follow-up

147 A monthly demographic follow-up was performed from October 2009 to November  
148 2010. For each month, snails were sampled in four stations along a 300-m transect at a depth  
149 from 0.50 to 1.5 m using a rectangular hand-net (25 × 18 cm); the total area sampled was 1  
150 m<sup>2</sup>. Samples were fixed on-site in 20% alcohol. Then we measured the SL of the snails  
151 present in a sub-sample corresponding to 9 out of 25 of the total sample in order to estimate  
152 monthly population densities and SL distributions. Considering 5% percentiles in SL  
153 distribution of juveniles, reproductive individuals and total individuals, we also determined,  
154 for each month, SL at birth, SL at maturity and maximum SL to provide guidance in the  
155 choice of the model's size classes. To estimate SL at maturity, we dissected 30 individuals  
156 covering a large range of sizes, and we counted the number of embryos in the brood pouch  
157 (*i.e.*, fecundity) according to the methodology described in Duft et al. [44]. This measurement  
158 also allowed us to calculate a relationship between SL and fecundity and to estimate the  
159 percentage of reproductive individuals for each adult class defined in the model.

160



161 2.2. Modelling framework and demographic analysis

162

163 2.2.1. Definition of the population model

164 We used a periodic Lefkovitch matrix population model with five size classes [15, 45]  
165 to capture the dynamics of the *P. antipodarum* population. Periodic matrix models [46-47] are  
166 often used to study cyclical temporal variation (*e.g.*, seasonal or interannual) operated within  
167 a single projection interval. The models take the form of periodic matrix products. We used  
168 size class models, in contrast to most ecotoxicological studies in which age class (Leslie  
169 models) or stage class models are employed [16]. We explain this choice with the following  
170 arguments: (*i*) a valid method does not exist to determine the age of snails in field populations  
171 for this species, (*ii*) we observed a strong correlation between SL and the life-history traits of  
172 *P. antipodarum* (growth rate, maturity, fecundity) and (*iii*) in highly variable environments  
173 (*e.g.*, contrasted seasonality), the life-history of individuals in such short-living species  
174 strongly depends on their date of birth, which makes age a very weak predictor of biological  
175 features. This model thus distinguishes two classes of juveniles (J1 and J2) and three classes  
176 of adults (A1, A2 and A3). It integrates the heterogeneity of vital rates (survival, growth and  
177 fecundity) between size classes throughout the year.

178 Let  $n_i(k)$  be the number of individuals of size class  $i$  ( $i = 1$  for J1,  $i = 2$  for J2,  $i = 3$  for  
179 A1,  $i = 4$  for A2 and  $i = 5$  for A3) at the beginning of month  $k$ . The five  $n_i(k)$  can be gathered  
180 in a population vector  $\mathbf{n}(k)$ . Then we can define 12 monthly matrices  $\mathbf{M}_k$  which link up the  
181 population vectors  $\mathbf{n}(k)$  between months  $k$  and  $k+1$  as follows:

182 
$$\mathbf{n}(k+1) = \mathbf{M}_k \mathbf{n}(k) \tag{2}$$

183 with:

$$\mathbf{M}_k = \begin{bmatrix} s_1(k) \left(1 - \sum_{j>1} g_{1,j}(k)\right) & 0 & f_3(k) \sqrt{s_1(k)} \sqrt{s_3(k)} & f_4(k) \sqrt{s_1(k)} \sqrt{s_4(k)} & f_5(k) \sqrt{s_1(k)} \sqrt{s_5(k)} \\ s_1(k) g_{1,2}(k) & s_2(k) \left(1 - \sum_{j>2} g_{2,j}(k)\right) & 0 & 0 & 0 \\ s_1(k) g_{1,3}(k) & s_2(k) g_{2,3}(k) & s_3(k) \left(1 - \sum_{j>3} g_{3,j}(k)\right) & 0 & 0 \\ s_1(k) g_{1,4}(k) & s_2(k) g_{2,4}(k) & s_3(k) g_{3,4}(k) & s_4(k) (1 - g_{4,5}(k)) & 0 \\ s_1(k) g_{1,5}(k) & s_2(k) g_{2,5}(k) & s_3(k) g_{3,5}(k) & s_4(k) g_{4,5}(k) & s_5(k) \end{bmatrix} \quad (3)$$

185 where  $s_i(k)$  is the survival rate of the size class  $i$  during month  $k$ ,  $g_{i,j}(k)$  the transition rate  
 186 between the size classes  $i$  and  $j$  during month  $k$ , and  $f_i(k)$  the reproductive rate of the size  
 187 class  $i$  during month  $k$ . The product of the 12 monthly matrices  $\mathbf{M}_k$  leads to an annual  
 188 periodic matrix  $\mathbf{L}$ , which links the population vector from year  $t$  to year  $t+1$  as follows:

$$\mathbf{n}(t+1) = \left( \prod_{k=1}^{12} \mathbf{M}_k \right) \mathbf{n}(t) = \mathbf{L} \mathbf{n}(t) \quad (4)$$

190 To assess the seasonal variability of the demographic sensitivity of the population, we also  
 191 defined four seasonal periodic matrices:  $\mathbf{L}_A$  for autumn,  $\mathbf{L}_W$  for winter,  $\mathbf{L}_{SP}$  for spring and  
 192  $\mathbf{L}_{SU}$  for summer. These matrices correspond to the product of the three monthly matrices  $\mathbf{M}_k$   
 193 corresponding to each season (*i.e.*, September, October and November in autumn; December,  
 194 January and February in winter; March, April and May in spring; and June, July and August  
 195 in summer).

196

### 197 2.2.2. Parameter estimation

198 We estimated the reproductive rates  $f_i(k)$  for the three size classes of adults ( $i = 3, 4$   
 199 and 5) as follows:

$$f_i(k) = b_i(k) \rho_i(k) \Delta t(k) \quad (5)$$

201 where  $b_i(k)$  corresponds to fertility (*i.e.*, number of neonates produced by reproductive female  
 202 per day) for class  $i$  during month  $k$ ,  $\rho_i(k)$  to the percentage of females in reproduction in class  
 203  $i$  during month  $k$ , and  $\Delta t(k)$  to the number of days of month  $k$ .  $\rho_i(k)$  was estimated with data

204 from the demographic follow-up (see above), and  $b_i(k)$  was predicted with the mean monthly  
205 water temperature, according to the relationship between mean temperature and fertility  
206 established from outcomes of caging experiments. Because only one size class of adults was  
207 used for *in situ* caging, we controlled the size effect between classes in the calculation of  $b_i(k)$   
208 using the relationship between SL and fecundity (*i.e.*, number of embryos in the brood  
209 pouch), which is established from the demographic follow-up. Similarly, we calculated the  
210 transition rates  $g_{i,j}(k)$  using the relationship between growth and temperature estimated in *in*  
211 *situ* caging experiments. Using the mean water temperature recorded during month  $k$ , we  
212 predicted the growth of individuals with size at the limits of each class  $i$  between months  $k$   
213 and  $k+1$ , and then we estimated for each size class the proportion of individuals which stayed  
214 in size class  $i$ , or which attained the larger size classes.

215         It is not possible to estimate the survival rates directly from field experiments. In fact,  
216 mark and recapture methodologies, which are currently used for larger organisms (*e.g.*, fish,  
217 mammals) are not easy to set up for *P. antipodarum* due to the difficulty marking the snails  
218 durably. Alternatively, the survival rates observed with the caged snails were not ecologically  
219 relevant (*e.g.*, lack of predation, competition). Therefore, to estimate the survival rates of  
220 individuals for a month  $k$ , we compared the densities observed during the demographic  
221 follow-up in month  $k$  to the theoretical densities predicted by the observed densities of month  
222  $k-1$  and the growth and reproductive rates of the individuals estimated for month  $k-1$ .

223

### 224 2.2.3. Model outcomes, elasticity analyses and simulations

225         The Lefkovitch matrix  $\mathbf{L}$  is a primitive matrix and can be processed analytically as a  
226 Leslie matrix. It presents a first dominant eigenvalue  $\lambda$ , corresponding to the asymptotic  
227 population growth rate [15 , 48]. The right eigenvector  $w$  associated with this first eigenvalue  
228 gives the asymptotic stable size structure. According to the first matrix used in the matricial

229 product of the 12 monthly matrices  $\mathbf{M}_k$ , we can obtain the different SL structures at the end  
230 of each month of the year. The demographic elasticities were analyzed by simulation with the  
231 application of 10% reduction in each life-history trait successively (*i.e.*, survival of each class,  
232 fertility and growth) in order to estimate the subsequent relative reduction in the asymptotic  
233 population growth rate  $\lambda$ . To examine the between-season variability of population  
234 vulnerability, we also simulated the demographic consequences of different levels of  
235 alteration of life-history traits at different dates in the year. To accomplish this, reductions  
236 from 0% to 100% on fertility, juvenile survival and adult survival were applied to the three  
237 months corresponding to each season.

238

### 239 2.3. Statistical analyses

240 Statistical procedures and population models were all implemented with the R  
241 software [49]. Before using parametric analysis (ANOVA procedure), normality and  
242 homoscedasticity were checked using the Shapiro-Wilk test and the Bartlett test, respectively.  
243 To quantify growth of *P. antipodarum* snails in *in situ* caging, we fitted, independently for  
244 each caging experiment, a logistic model on SL data using the *nls* function, simultaneously  
245 considering the two categories of caged snails (juveniles and adults) as follows:

$$246 \quad L(t) = \frac{Lmax}{1 + \left( \frac{Lmax}{Linit} - 1 \right) e^{(-r t)}} \quad (6)$$

247 where  $L(t)$  corresponds to the SL of snails at time  $t$ ,  $Lmax$  to the maximal SL of snails  
248 observed in the population,  $Linit$  to the SL of the two categories of caged snails at the  
249 beginning of the experiment,  $r$  to the daily growth coefficient of the logistic model, and  $t$  to  
250 the time. We fixed  $Lmax$  at 5.5 mm (maximum value observed during the demographic  
251 follow-up). No replicate effect was considered when fitting the logistic models (one per  
252 campaign), since no significant difference in SL of juveniles and adults were detected

253 between replicates at the end of the test for each one of the seven campaigns. Concerning the  
254 seasonal variability of fertility and daily growth, we fitted Gaussian relationships between  
255 these life-history traits and water temperature using the *nls* function. For the demographic  
256 follow-up, the influences of SL and month on fecundity were tested using linear modelling  
257 including the interaction terms (ANOVA procedure).

258

### 259 3. Results

260

#### 261 3.1. Seasonal variability of life-history traits

262 By means of *in situ* caging experiments, we recorded strong seasonal variability in the  
263 production of *P. antipodarum* neonates, which varied from 0 to 0.8 neonates per female per  
264 day. This seasonal variability in fertility was highly correlated with water temperature (Figure  
265 3A). Thus, we fitted a Gaussian curve to describe fertility  $b_i$  (production of neonates per day  
266 per reproductive female) as a function of mean water temperature  $\theta$  (in °C) as follows:

$$267 \quad b_i(\theta) = 6.92 \times e^{\left(\frac{17.91 - \theta}{3.85}\right)^2} \quad (7)$$

268 Fertility is optimal at a temperature of 17.91 °C. Concerning growth, here again we detected  
269 strong seasonal variability of individual SL gains, in relation with the mean water temperature  
270 (Figure 3B). We used nonlinear regression to fit a Gaussian relationship between growth  
271 coefficient  $r$  and mean water temperature  $\theta$  (in °C) as follows:

$$272 \quad r(\theta) = 0.006 + 0.21 \times e^{\left(\frac{18.61 - \theta}{3.72}\right)^2} \quad (8)$$

273 In comparison with fertility, we added a constant parameter in order to take into account a  
274 minimal daily growth rate different from 0. In fact, contrary to fertility, which was null during

275 the caging with the lower temperature (Figure 3A), we observed that individuals grow even at  
276 low temperatures (Figure 3B). Growth is optimal at 18.61 °C.

277         Concerning the monthly demographic follow-up, the evolution of the population's SL  
278 structure is presented in Figure 1. The highest densities were observed in autumn (*e.g.*, more  
279 than 10,000 individuals per square metre in October and November) and the lowest in spring.  
280 Juveniles (*i.e.*, SL < 3.5 mm, size classes J1 and J2 of the model) are present throughout the  
281 year and account for the major part of the population, while adult densities (*i.e.*, SL > 3.5 mm,  
282 size classes A1, A2 and A3) show highly variable frequencies during the year. In fact, in  
283 winter and spring, the population comprises mainly juveniles, while in summer and autumn  
284 adults appear in the population. We estimated a SL at birth of 0.5 (0.1) mm, a SL at maturity  
285 of 3.5 (0.2) mm and a maximum SL of 5.2 (0.3) mm. According to these very weak standard  
286 deviations, we did not consider monthly differences in the SL at birth, at maturity and the  
287 maximum SL for the parameterisation of the population model. Concerning the percentage of  
288 individuals in reproduction (*i.e.*, females bearing embryos)  $\rho_i(k)$ , for all dates of the year,  
289 more than 90% of the individuals with a SL greater than 4.2 mm were reproductive.  
290 Individuals between 3.5 and 4.2 mm presented a seasonal variability: 50% of individuals were  
291 reproductive between November and May and 80% between June and October. We observe a  
292 strong positive relationship between SL and fecundity (Figure 2), with no seasonal effect  
293 (ANOVA test: interaction terms,  $p > 0.1$ ; seasonal effect,  $p > 0.81$ ; SL effect,  $p < 10^{-15}$ ). In  
294 this way, we note that females of the model's A1 size class have a mean fecundity of 12.2  
295 embryos; size class A2 females a mean fecundity of 24.3 embryos and size class A3 females a  
296 mean fecundity of 37.6 embryos.

297

298 3.2. Population model analysis

299           Regarding parameter estimation, we report a strong between-class and between-month  
300    variability of the reproductive rates  $f_i(k)$ . Between November and March, reproductive rates  
301    were very low for all classes due to low water temperatures, while all size classes reproduced  
302    between April and October. Depending on the size class and the month, reproductive rates  
303    varied from 0.01 to 35.05 neonates per month per individual. Regarding the transition rates  
304     $g_{i,j}(k)$ , the majority of individuals stayed in their initial size class from month to month  
305    during winter and spring, in contrast to summer and autumn, where growth was faster and a  
306    great majority of individuals changed one or two size classes over 2 months. Adult survival  
307    showed high monthly variability with very low survival rates between January and May and  
308    higher survival rates in summer and autumn. Juvenile survival (for size classes J1 and J2) was  
309    generally higher than adult survival, particularly in winter and spring. For some months, we  
310    calculated survival rates higher than 1, in particular when densities of individuals were low  
311    (uncertainty increased with small size samples). For the parameterization of the matrix model,  
312    we tested two possibilities: (i) we applied the survival rates keeping the values higher than 1  
313    (apparent survivals), or alternatively (ii) we fixed a ceiling level of 1 for maximum survival  
314    rates. Despite differences in the absolute value of the asymptotic population growth rate  $\lambda$ , the  
315    stable size distribution and elasticity pattern were very similar in both cases. For the following  
316    results on model analysis, we chose to use survival rates capped at 1, but conclusions on the  
317    demographic behaviour of the population remain unchanged with apparent survival rates.

318           In a first time, we calculated the asymptotic population growth rate  $\lambda$  with the annual  
319    periodic matrix  $L$ . We found a value of  $\lambda = 1.17$ . We also computed the stable size  
320    distribution for the different seasons that we compared to the population structure observed  
321    during the demographic follow-up (Figure 4). We noted good coherence between the model  
322    and the field data. Thus, it appears that our mechanistic modelling framework can identify the  
323    dynamics of the *P. antipodarum* population throughout the year. In a second time, we

324 characterized the seasonal variability of the demographic fitness of the population by  
325 calculating the asymptotic population growth rate of the four periodic seasonal matrices ( $L_A$ ,  
326  $L_W$ ,  $L_{SP}$  and  $L_{SU}$ ). We found 3-month  $\lambda$  values equal to 1.02 in autumn, 0.17 in winter, 0.16  
327 in spring and 2.31 in summer. For comparison to the annual matrix  $L$ , the standardization of  
328  $\lambda$  to a 3-month time step supplies a value of 1.04. Thus, we underline a strong seasonal  
329 variability of snail population dynamics, with a potential growth of the population mainly in  
330 summer and autumn.

331 The elasticity analysis on the annual matrix  $L$  showed that the asymptotic population  
332 growth rate  $\lambda$  was more sensitive to relative changes in juvenile survival (S1 and S2) than to  
333 changes in the other life-history traits (Figure 5). The life-history trait corresponding to the  
334 second highest elasticity was adult survival (cumulative elasticities of S3, S4 and S5)  
335 followed by fertility. Concerning growth (Figure 5), we observe that the population growth  
336 rate was not sensitive to relative changes in this life-history trait. On the contrary, we note that  
337 a reduction of the daily growth rates strikingly increases the asymptotic population growth  
338 rate  $\lambda$ . We also performed elasticity analysis on the four seasonal matrices ( $L_A$ ,  $L_W$ ,  $L_{SP}$  and  
339  $L_{SU}$ ) (Figure 6). These analyses reveal two contrasted patterns. On one hand, in spring and  
340 winter, the population growth rate was very sensitive to juvenile survival alteration but  
341 remained unchanged by reduction in adult survival or reproduction rates. Concerning growth,  
342 we observe, as for the annual model, that the reduction of this life-history trait increased the  
343 asymptotic population growth rate. On the other hand, in summer and autumn, the population  
344 growth rate was sensitive to changes in juvenile survival but also to changes in adult survival,  
345 fertility and growth.

346 Strong variability of population impacts (percentage of reduction of the population  
347 growth rate  $\lambda$ ) can be seen when alterations of life-history traits at different seasons were  
348 integrated into the model (Figure 7). In fact, except for juvenile survival for which we noted



349 that the population growth rate was highly affected for all seasons, concerning reproduction  
350 and adult survival, between-season differences were substantial. Indeed, the population was  
351 very sensitive to impacts on fertility in summer and particularly in autumn, but showed very  
352 low sensitivity in spring and winter, even for substantial fertility inhibitions. Furthermore, the  
353 population was very sensitive to impacts on adult survival in summer, even for very small  
354 inhibitions, but quite insensitive in the other seasons even for considerable inhibitions.

355

#### 356 **4. Discussion**

357

##### 358 4.1. Seasonal variability of life-history traits of *P. antipodarum*

359 We observed fluctuating densities during the year. The high-density pattern at the end  
360 of summer and autumn and the low-density pattern in winter and spring agree with previous  
361 data obtained during demographic follow-up conducted for *P. antipodarum* [41, 50].  
362 Furthermore, these density variations during the year are consistent with the observations of a  
363 long-term follow-up of mollusc populations conducted in this study site (data not published  
364 for *P. antipodarum*). However, Schreiber et al. [51] observed the highest densities in spring  
365 and summer. This contrast with the present study can be explained by the delaying effect of  
366 temperature rise due to a large snow-melt upstream of the Rhône watershed. For Richards and  
367 Shin [52], these fluctuating densities are driven by density-dependent processes. Nevertheless,  
368 in populations with small temperature fluctuations, Quinn et al. [53] observed that densities  
369 are very stable during the year, and it is widely accepted that *P. antipodarum* population  
370 densities are generally strongly correlated with water temperature [50], consistent with our  
371 observations. The population in our case was primarily composed of juveniles, particularly in  
372 winter and spring, which agrees with previous data [51, 54]. In this way, the persistence of the  
373 population in winter is ensured by the survival of a reserve of juveniles. In summer and

374 autumn, the population is composed of two cohorts, one with small juveniles and one with  
375 large adults. The size at birth, size at maturity and maximum size values are consistent with  
376 previous reports for this species [30, 51, 54-57]. Similar to Schreiber et al. [51], who observed  
377 65% reproductive females in their population, we observed that the majority of females  
378 carried embryos. During dissections, we never observed males, as in other studies which  
379 stated that the vast majority of European populations of *P. antipodarum* are made up of  
380 parthenogenetic females [38] with a few exceptional males [58]. This explains why we did  
381 not integrate the sex ratio into our model.

382         The methodology of *in situ* caging implemented here throughout the year provides a  
383 useful tool to estimate life-history traits in the field (*e.g.*, realistic exposure conditions, good  
384 reproducibility of the assays) [59-60]. Until now, only a few studies have conducted *in situ*  
385 experiments with *P. antipodarum* [35-36, 61]. Here, we chose this methodology rather than a  
386 laboratory approach in order to obtain environmentally relevant data for the calibration of the  
387 population model. In fact, several factors are difficult to control in laboratory conditions, in  
388 particular the diet of *P. antipodarum*. Here, we report a strong seasonal variability in neonate  
389 production and growth rates. Several studies have shown that *P. antipodarum* life-history  
390 traits are strongly correlated with water temperature [40, 42]. The quality of the fits observed  
391 in Figure 3 confirms this pattern. The maximum fertility value (0.80 juveniles per day  
392 recorded during caging at a mean temperature of 17.5 °C) is higher than the values reported in  
393 other studies for this temperature range [30-31]. Nevertheless, all the studies which measured  
394 fertility directly (*i.e.*, counting production of neonates) were conducted during laboratory  
395 experiments, which probably do not offer optimal conditions for the reproduction of this  
396 species. To our knowledge, our study reports for the first time a direct *in situ* measurement of  
397 fertility. In fact, in *in situ* assays, fertility is usually estimated indirectly from fecundity (*i.e.*,  
398 counting of the number of embryos in the brood pouch) [35-36, 61]. This measure can

399 provide useful information but it remains difficult to predict the realized reproduction of  
400 individuals without information on laying dynamics. We estimated an optimal temperature of  
401 17.9 °C for fertility. This value is consistent with previous laboratory studies [40, 42]. Schmitt  
402 et al. [36] did not observe a relationship between temperature and fertility during different *in*  
403 *situ* experiments, but the temperature range was very narrow compared to our experiments.  
404 Concerning growth, we estimated an optimal value of 18.7 °C close to the fertility value.  
405 Thus, water temperature between 17 and 19 °C appeared to provide optimal conditions for the  
406 development of *P. antipodarum*.

407

#### 408 4.2. Demographic insights from the population model analysis

409 The adequacy of the stable size distribution computed for each season with the  
410 population size structure observed during the demographic follow-up (Figure 4) illustrates  
411 that our modelling framework is able to reliably describe the dynamics of this *P. antipodarum*  
412 population taking into account its particular phenology. The analysis of seasonal matrices  
413 ( $L_A$ ,  $L_W$ ,  $L_{SP}$  and  $L_{SU}$ ), which simulate population dynamics under hypothetical scenarios of  
414 eternal autumn, winter, spring or summer, allows us to underline substantial seasonal  
415 variability in the demographic fitness of the population (quantified by the asymptotic growth  
416 rate  $\lambda$ ). On one hand, winter and spring are seasons with a potential population decrease, and  
417 on the other hand, summer and autumn seasons have a high potential for a population  
418 increase. This is explained by reproductive rates that are nearly zero in spring and winter  
419 along with very low survival rates, in particular for adults.

420 With the annual model, we observed that the population dynamics is particularly  
421 sensitive to changes in juvenile survival (Figure 5). This is consistent with the conclusions of  
422 several demographic studies on *P. antipodarum* [30, 34, 62]. Pedersen et al. [30], when  
423 studying the effects of the polycyclic musk HHCb on individual and population-level

424 endpoints, observed that the asymptotic population growth rate of *P. antipodarum* is  
425 approximately four times more sensitive to juvenile survival alteration than to adult survival,  
426 and ten times more sensitive to juvenile survival alteration than to fertility inhibition.  
427 Surprisingly, we noted that a reduction of the daily growth rate increased the asymptotic  
428 population growth rate  $\lambda$ . This unexpected demographic outcome can be explained by the very  
429 low survival rates of the last classes of adults for several months in spring and winter. Thus,  
430 when the daily growth rate is decreased, individuals stay for more months in the first class of  
431 adults with higher survival rates, which increases their cumulative reproductive value for the  
432 population. This advantage for organisms with a low growth rate is similar to observations on  
433 fish populations under fishing pressure, in which higher mortality rates lead to the selection of  
434 individuals with lower growth ability [63-64]. With the seasonal models, we show an  
435 important seasonal variability of the pattern of elasticities (Figure 6). In fact, in the same  
436 manner as for the population dynamics, we can identify two contrasted periods. In spring and  
437 winter, the population is mostly sensitive to the reduction in juvenile survival, while in  
438 summer and autumn, the demographic sensitivity to juvenile survival reduction is  
439 considerably reduced, and the population demography becomes more sensitive to the  
440 alteration of other traits. This strong seasonal variability of population sensitivity is explained  
441 by the phenology of the population: in winter and spring, maintenance of the population is  
442 ensured by a stock of juveniles (no reproduction, high adult mortality), whereas in summer  
443 and autumn, juveniles grow up, mature as adults, and therefore reproduce. Seasonal  
444 variability of population sensitivity is also observed in the amphipod *Leptocheirus plumulosus*  
445 using a field-based periodic matrix population model [28-29]. Thus, analyzing the elasticity of  
446 periodic matrix models can provide valuable insights into the relative importance of the  
447 demographic rates at different periods of year.

448

#### 449 4.3. Seasonal variability and field-based population models in ecological risk assessment

450 Despite the substantial seasonal variability of population demographic sensitivity, to  
451 our knowledge, only a few ecotoxicological studies have addressed the temporal variability of  
452 effects on population dynamics [27-29]. We roughly simulated reductions in fertility, juvenile  
453 survival and adult survival at different seasons. Population impacts strongly depend on the  
454 season at which toxic effects on individual performance occur (Figure 7). For instance, a time  
455 window of high population vulnerability to reproductive alteration is focused in summer and  
456 autumn, in contrast to winter and spring, for juvenile mortality. Thus, the development of  
457 population models that integrate seasonality is a relevant way to increase our ability to project  
458 toxic effects on individual performance into population demographic impacts. As an  
459 illustration, when studying HHCB effects on *P. antipodarum*, Pedersen et al. [30] observed  
460 significant effects on offspring production up to 42% reduction) but stated that such  
461 inhibitions will not give rise to significant impacts on population dynamics. In spring and  
462 winter, our simulations agree with this conclusion. However, in summer and particularly in  
463 autumn, we anticipate considerable population consequences of such levels of fertility  
464 inhibition. In fact, in autumn, we observed that a 40% reduction in fertility means a 30%  
465 decrease of the asymptotic population growth rate (Figure 7). Similar to the great majority of  
466 studies addressing population extrapolation, the life cycle is roughly parameterized from  
467 laboratory data. In fact, life-history traits are often estimated in the controls of the  
468 experiments and are not representative of life-history of local populations. Thus, with this  
469 laboratory approach, models do not provide valuable information about potential  
470 demographic consequences of toxicant impacts in field populations [30]. Laboratory  
471 conditions are too favourable (*e.g.*, no predation or competition, water temperature is  
472 constant) or in contrast, they fail to provide optimal conditions or complexity for all abiotic  
473 and biotic parameters (*e.g.*, oxygenation, water flow, temperature, food, crowding effect). One

474 way currently used to address such concerns about the ecological relevance of population  
475 extrapolation consists in exploring a range of values of life-history traits and showing that  
476 changes in the population growth rate are low. By this means, authors aim to assess the  
477 robustness of their conclusions against divergence in life histories between laboratory and  
478 wild populations. For example, Pedersen et al. [30] validated the robustness of their  
479 population model by testing three scenarios: decrease in juvenile survival, decrease in  
480 juvenile and adult survival, and decrease in juvenile survival, adult survival and fertility. But  
481 the range of tested fluctuations is not at all “field-realistic” regarding our field observations:  
482 (i) only 50% for fertility, whereas we showed in the *P. antipodarum* population that fertility is  
483 very low during a great part of the year with greater reduction than in this study; and (ii) only  
484 a 20% reduction in adult survival and a 90% reduction in juvenile survival, while we observed  
485 that juvenile mortality is generally lower than adult mortality in the field. Thus, it appears that  
486 a field-based approach can be of great interest to guarantee the relevance of hazard  
487 assessment at the population level, and to provide realistic scenarios for exploring its  
488 soundness with respect to between-population variability of life-histories.

489 Improving the ecological realism of population models is a major concern for  
490 ecological risk assessment. Incorporating species phenologies is feasible with modelling  
491 approaches and should be one priority when seeking to put the “eco into ecotoxicology” [2].

492

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494

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497 **References**

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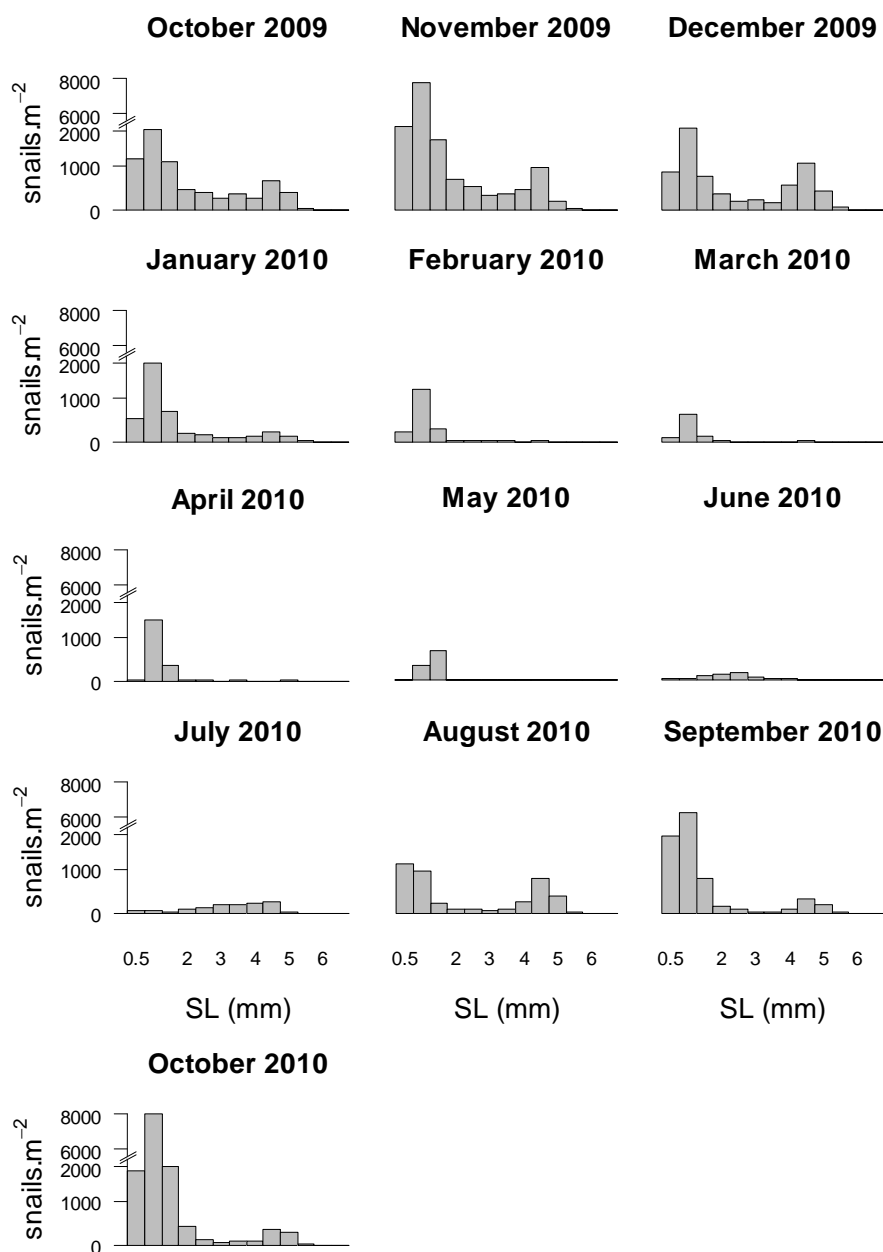
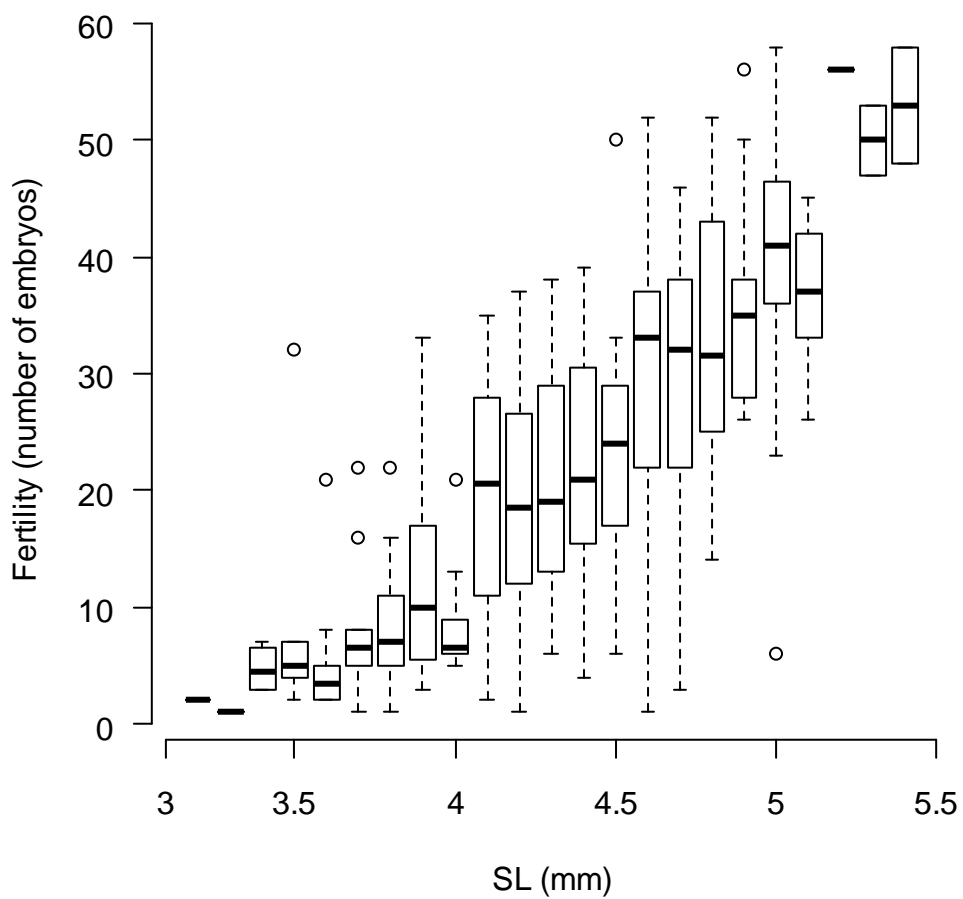
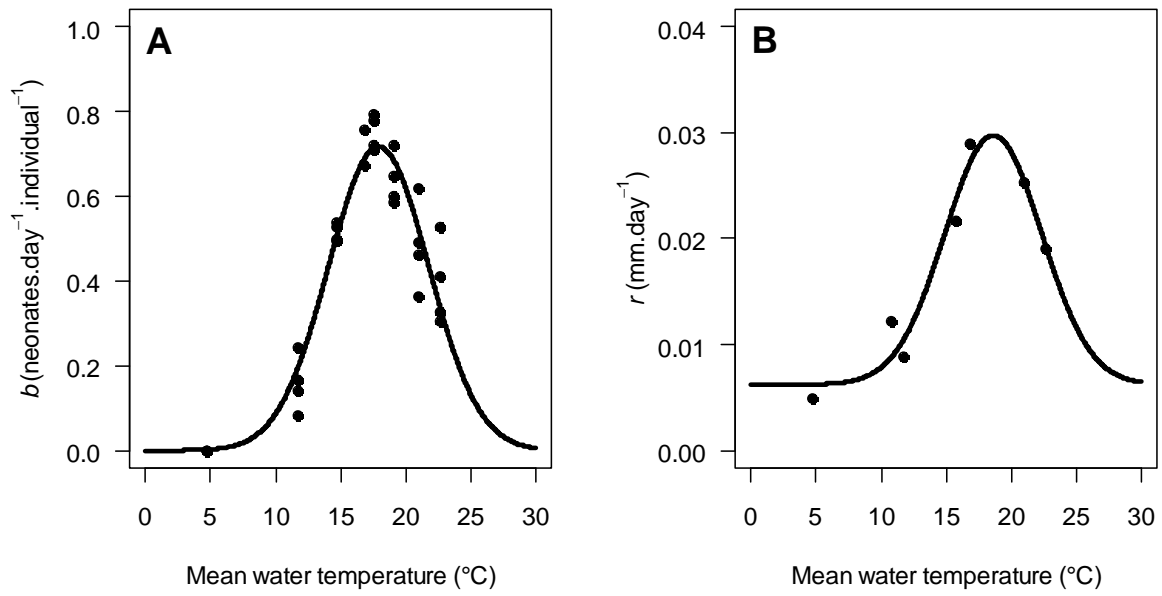


Figure 1.

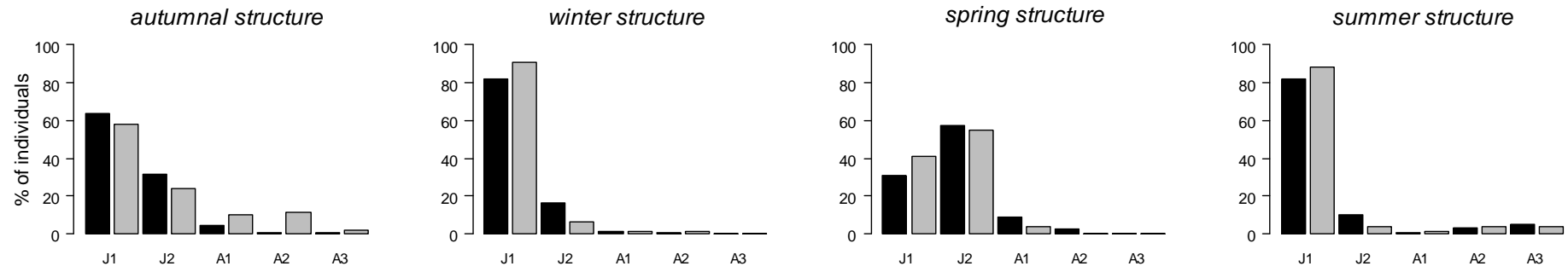


**Figure 2.**



**Figure 3.**





**Figure 4.**

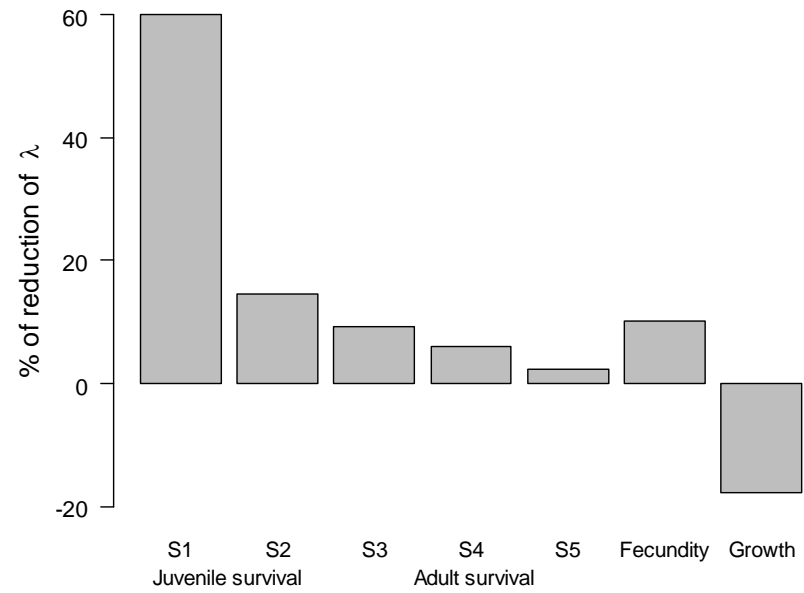


Figure 5.

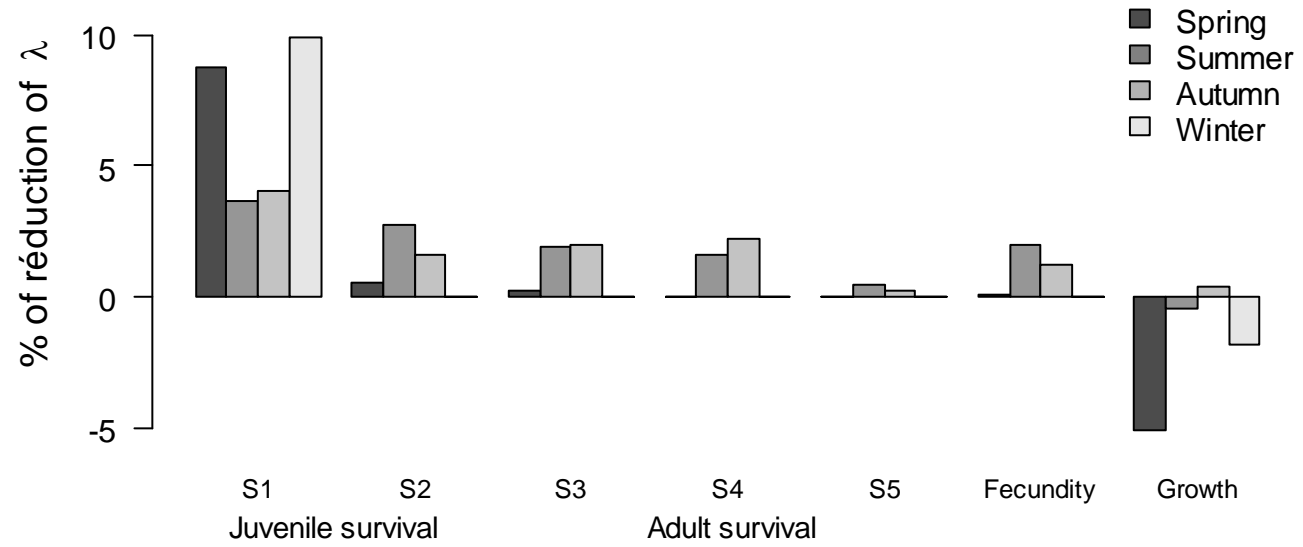
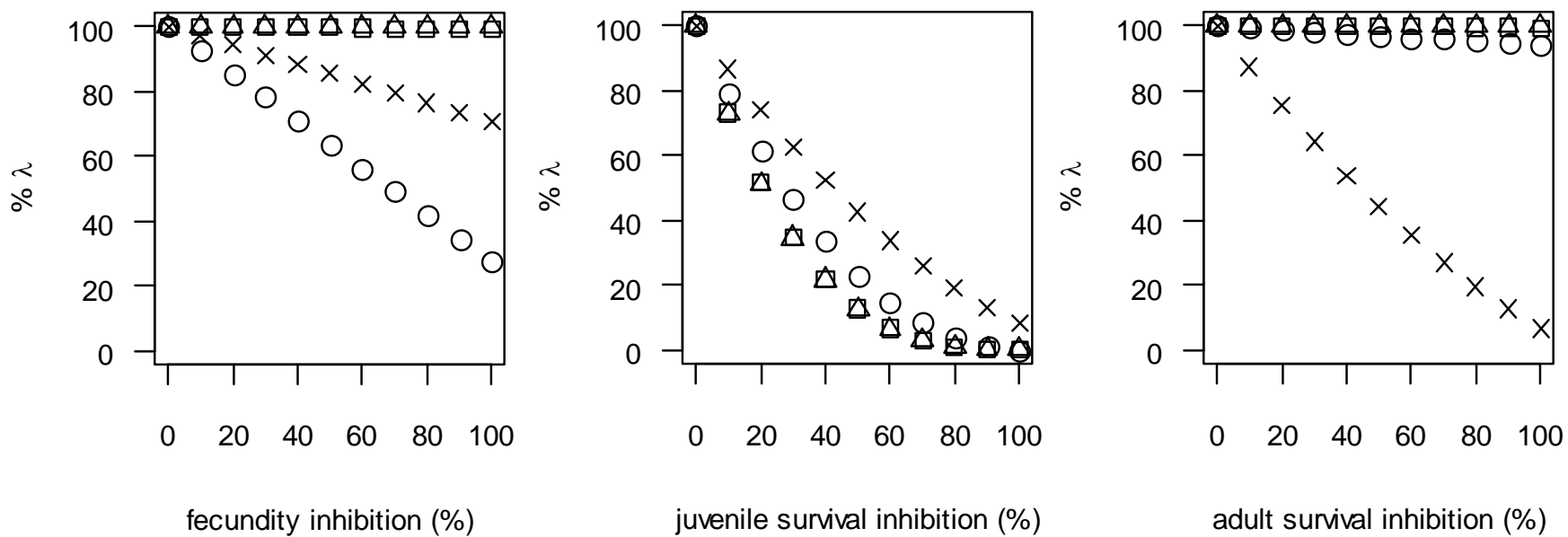


Figure 6.







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4 **Figure 7.**