

1 **Sex-specific changes in gene expression and delayed sex differentiation in response to**
2 **estrogen pollution in grayling (Salmonidae)**

3
4 Oliver M. Selmoni^{1*}, Diane Maitre^{1*}, Julien Roux^{1,2}, Laetitia G. E. Wilkins¹, Lucas Marques
5 da Cunha¹, Etienne L. M. Vermeirssen³, Susanne Knörr⁴, Marc Robinson-Rechavi^{1,2#}, Claus
6 Wedekind^{1#}

7 ¹ Department of Ecology and Evolution, Biophore, University of Lausanne, Lausanne,
8 Switzerland

9 ² Swiss Institute of Bioinformatics, Lausanne, Switzerland

10 ³ Swiss Centre for Applied Ecotoxicology Eawag-EPFL, Dübendorf, Switzerland

11 ⁴ Aquatic Ecology and Toxicology Group, Center of Organismic Studies, University of
12 Heidelberg, Heidelberg, Germany

13 * equal contributors

14 # shared senior authors

15

16 Present addresses:

17 O. Selmoni: Swiss Federal Institute of Technology (EPFL), Bâtiment GC, 1015 Lausanne,
18 Switzerland.

19 L. Wilkins: Department of Environmental Sciences, Policy and Management, 130 Mulford
20 Hall #3114, University of California, Berkeley, CA 94720, USA

21 J. Roux: Department of Biomedicine, University of Basel, Hebelstrasse 20, 4031 Basel,
22 Switzerland

23

24 Correspondence: Claus Wedekind, claus.wedekind@unil.ch, orcid.org/0000-0001-6143-4716

25

26 **Abstract**

27 The synthetic 17 α -ethinylestradiol (EE2) is an estrogenic compound of oral contraceptives
28 and therefore a common pollutant that has been suspected to affect the demography of river-
29 dwelling salmonids. We study a population of European grayling (*Thymallus thymallus*) that
30 suffers from sex ratio distortions. Here we test how ecologically relevant concentrations of
31 EE2 affect sex-specific gene expression around early stages of sex differentiation. We
32 collected gametes from F1s of wild spawners, used them for *in vitro* fertilizations, and raised
33 the resulting embryos singly under experimentally controlled conditions. Embryos were either
34 exposed to 1ng/L EE2 or sham-exposed. RNA was collected from samples taken 10 days
35 before hatching, at the day of hatching, and towards the end of the yolk-sac stage, to study
36 gene expression and relate it to genetic sex (sdY genotype). We found that EE2 affects gene
37 expression of a very large number of genes especially at the day of hatching. The effects of
38 EE2 on gene expression is strongly sex-specific. At the day of hatching, EE2 affected about
39 twice as many genes in females than in males, and towards the end of the yolk-sac larval
40 stage, EE2 effects were nearly exclusively observed in females. Among the many effects was,
41 for example, a surprising EE2-induced molecular masculinization in the females' heads.
42 Histological examination of gonadal development of EE2-treated or sham-exposed juveniles
43 during the first 4.5 months after hatching revealed a delaying effect of EE2 on sex
44 differentiation. Because grayling sex determination goes through an all-male stage (a rare
45 case of undifferentiated gonochorism), the rate of EE2-induced sex reversal could not be
46 unequivocally determined during the observational period. However, two EE2-treated genetic
47 males had ovarian tissues at the end of the study. We conclude that common levels of EE2
48 pollution affect grayling from very early stages on by interfering with male and female gene
49 expression around the onset of sex differentiation, by delaying sex differentiation, and by
50 feminizing some males.

51

52 **Author contribution**

53 MRR and CW initiated the project. OS, DM, LW, LMC, and CW sampled the adult fish, did
54 the experimental *in vitro* fertilizations, and prepared the embryos for experimental rearing in
55 the laboratory. All further manipulations on the embryos and the larvae were done by OS,
56 DM, LW, and LMC. The RNA-seq data were analyzed by OS, JR, and MRR, the histological
57 analyses were done by DM, supervised by SK, the molecular genetic sexing was performed
58 by OS and DM, and EV supervised the EE2 analytics. OS and CW performed the remaining
59 statistical analyses and wrote the first version of the manuscript that was then critically
60 revised by all other authors.

61

62 **1. Introduction**

63 Endocrine-disrupting chemicals are common pollutants that typically enter the environment
64 after wastewater treatment. One of the most potent of these micropollutants is the synthetic
65 17-alpha-ethinylestradiol (EE2) that is used in oral contraceptives and hormone replacement
66 therapies, and that is more stable and persistent than the natural estrogen it mimics [1]. EE2
67 concentrations of 1ng/L and higher have been found in river or lake surface waters [2] in lake
68 sediments [3], and even in groundwater [4].

69 Exposure to 1 or few ng/L EE2 can be damaging to fish at various developmental
70 stages. Embryos and early larvae can suffer from increased mortality, reduced growth, or
71 malformations when exposed to EE2 [5, 6]. In juveniles and adults, exposure to EE2 can
72 affect the response to infection [7], increase the susceptibility to other micropollutants [8],
73 generally reduces growth and fertility [8, 9], and can even induce transgenerational effects on
74 behavior and fertility in F1 [10] and F2 progeny [11]. Studies with experimental populations
75 kept in 1,100 L ponds revealed population declines at concentrations of 1ng/L EE2 [12], and
76 long-term, whole-lake experiments revealed significant ecosystem changes after experimental
77 addition of 5-6 ng/L EE2: local populations of small fish declined (one species nearly got
78 extinct), average body conditions of other fish, including top predators, declined significantly,
79 and the prevalence of some zooplankton and insect species seemed to increase [13].

80 Experimental exposure to EE2, for example of juvenile sticklebacks (*Gasterosteus*
81 *aculeatus*) to 35-40 ng/L, or of juvenile coho salmon (*Oncorhynchus kisutch*) to 2 or 10 ng/L,
82 are associated with significant down- and up-regulations of various physiological pathways
83 [14, 15]. Some of these effects on gene expression may be linked to the toxic effects of EE2
84 observed in juveniles and adults. However, it is likely that EE2 effects on gene expression
85 depend on life history and on the developmental stage of an individual, i.e. on the timing of
86 some physiological pathways in the organism. One important physiological pathway in this
87 context is sex determination and gonad formation.

88 Sex determination is probably best seen as a threshold trait, with processes that occur
89 early in development determining later processes [16]. In amphibians and fishes, these early
90 processes can be very labile, i.e. potentially modifiable by external factors, even if they often
91 have a clear genetic basis [17, 18]. Among these external factors that interfere with these
92 early steps of sex determination are temperature or endocrine disrupting chemicals such as
93 EE2 [17, 19]. When applied at the beginning of sex determination, they can tip the balance
94 and cause a sex reversal, i.e. a mismatch between genotypic and phenotypic sex. Such EE2-
95 induced sex reversals are sometimes but not always observed. If they are observed, the timing
96 of exposure seems more relevant than the concentration: Orn et al. [20] found, for example,
97 nearly complete sex reversal in zebrafish exposure to 5 ng EE2/L during embryo and larval
98 stages, while exposure to 5-20 ng EE2/L at later stages did not seem to induce sex reversal in
99 the same species [21].

100 Apart from gonad development, there can be many other fundamental differences
101 between male and female development. Males and females may differ, for example, in
102 average growth, timing of maturation, habitat use, or susceptibility to various stressors

103 including infections [22]. It is therefore possible that effects of EE2 on the organism crucially
104 depend on whether or not the organism is exposed to EE2 during the early steps of the sex
105 determination cascade, when sex reversal is still possible [23].

106 Exposure to EE2 has been found to induce sex-specific effects in juvenile and adult
107 fish [23]. These sex-specific effects can be large, as expected by the significant differences in
108 male and female physiology. It remains unclear whether large differences should also be
109 expected between the genetic sexes if their gonadal development may have been influenced
110 by the sex-reversing effects of EE2 [24, 25]. Such questions can be studied if reliable sex-
111 linked genetic markers are available for a given study species.

112 Here we study the sex-specific effects of ecologically relevant concentrations of EE2
113 on gene expression and gonad development in grayling (*Thymallus thymallus*), a river-
114 dwelling salmonid fish that may often be exposed to EE2 pollution. Yano et al. [26]
115 established sex-linked genetic markers that could be used to determine the genetic sex of
116 many salmonids, including their sample of grayling taken from a fish farm in France. These
117 markers could be successfully verified in over 100 phenotypically sexed adult grayling
118 sampled from our study population [27]. We therefore use them here to separate effects of
119 EE2 on gene expression in genetic males and females. Maitre *et al.* [27] found large effects of
120 genetic sex on gene expression around the time of hatching from eggs, while gene expression
121 did not seem to differ significantly at late embryogenesis. Their findings thus suggest that the
122 physiological cascade of sex determination starts during embryogenesis and before hatching.
123 We therefore study the interaction between EE2 and genetic sex on gene expression in
124 embryos and larvae. Within-family comparisons are used to minimize potential effects of
125 genetic variation. Possible interactions between EE2 and genetic sex on gonad development
126 are studied histologically on samples taken over a period of several months.

127

128 **2. Methods**

129 *2.1 Experimental breeding and raising*

130 Mature males and females were sampled from a captive breeding stock and stripped for their
131 gametes. These fish are F1 of the natural population described in Wedekind et al. [28]. Their
132 gametes were used in two full-factorial breeding blocks. For each breeding block, 4 females
133 were crossed *in vitro* with 5 males, i.e. 40 (2x4x5) different sibgroups were produced. After
134 egg hardening for 2 hours, the fertilized eggs were transported to the laboratory where they
135 were washed and distributed singly to wells of 24-well plates (Falcon, Becton-Dickinson),
136 following the methods of von Siebenthal et al. [29]. The wells had been filled with 1.8 mL
137 chemically standardized water [30] that had been oxygenated and tempered before use.

138 The embryos were incubated at 7°C and left undisturbed until 14 days post
139 fertilization (dpf), when embryos were exposed either to 1 ng/L EE2 (by adding 0.2 mL water
140 with a concentration of 10 ng/L EE2, see Brazzola *et al.* [6] for details), a strain of
141 *Pseudomonas fluorescens* (“PF”; 10⁶ bacterial cells per well, the low-virulence strain PF1 in
142 Clark et al. [31]), simultaneously to EE2 and *P. fluorescens* (EPF), or sham-treated
143 (“control”, i.e. only adding 0.2 mL standardized water). A sample of 250 live embryos (EE2-
144 treated or sham-treated offspring of one haphazardly chosen female that had been crossed
145 with five males) was separated for analyses of gene expression, such that we had 25 embryos
146 for each of the five sibgroups and the two treatments “EE2” and “controls”. The remaining
147 embryos were raised until 40 dpf, i.e. until several days after hatching, when a subset of each
148 treatment group was distributed to two 200 L tanks each filled with lake water (pumped from
149 Lake Geneva at 40 m depth), fed with live *Artemia* and copepods and later dry food, and
150 sampled in 5 monthly intervals for histological examination of the gonads (see Maitre *et al.*
151 [27] for a description of sex differentiation). For the EE2-treated groups, 200 ng EE2 were
152 dissolved in 200 L tanks each to reach a starting concentration of 1 ng/L. Every 7 days from
153 then on, 40 L (i.e. 20%) were replaced with fresh lake water spiked with 40 mL of a 1 µg/L

154 EE2 stock solution (i.e. 40 L at 1 ng/L EE2). Water samples (100 mL each) were then taken
155 from each of the 4 EE2-treated tanks 1 hour after this weekly water exchange (T_0) and 7 days
156 later, just before the next water exchange (T_7). These water samples were immediately frozen
157 and stored at -20°C protected from light. Four consecutive T_0 and 4 consecutive T_7 samples
158 were each pooled per tank for later determination of EE2 concentrations, i.e. EE2
159 concentrations were determined for the 4-week intervals these pooled samples covered,
160 starting 47 dpf, 75 dpf, 103 dpf, and 131 dpf, respectively.

161 To quantify EE2, the water samples were thawed and filtered over glass fibre filters,
162 their volume was set to 250 mL and the pH to 3. Four ng/L of EE2 D4 was added as internal
163 standard and samples were enriched on LiChrolut EN / LiChrolut RP-C18 cartridges that had
164 been conditioned with hexane, acetone, methanol and finally water (pH 3) [32]. After sample
165 enrichment, cartridges were dried with nitrogen and eluted with acetone and methanol.
166 Subsequently, solvents were changed to hexane/acetone 65:35 and extracts were passed over
167 Chromabond Silica columns [33] and set to a volume of 0.25 mL. LC-MS/MS analysis was
168 performed on an Agilent6495 Triple Quadrupole. An XBridge BEH C18 XP Column, $2.5\ \mu\text{m}$,
169 $2.1\ \text{mm} \times 75\ \text{mm}$ and an acetonitrile / water gradient was used for liquid chromatography
170 followed with post-column addition of ammonium fluoride solution. EE2 was quantified by
171 monitoring the mass transition of 295 to 269, the transition of 295 to 199 served as a qualifier
172 (internal standard was quantified at the following transitions: 299 to 273 and 299 to 147).

173 EE2 concentrations in 24-well plates and uptake by embryos are described in Marques
174 da Cunha et al. (unpublished manuscript). Briefly, plates with newly fertilized embryos were
175 spiked with EE2 to a concentration of 1 ng/L of EE2 in the plate wells. Analogously, plates
176 without embryos were spiked to the same concentration. EE2 concentrations in the 24-well
177 plates with and without embryos were measured in 5 time points across embryo development
178 (i.e. from 1 dpf until shortly before embryo hatching). While EE2 concentrations have
179 remained around 1 ng/L in plates without embryos throughout the 5 time points (indicating no
180 apparent degradation or sorption of EE2 in 24-well plates), in plates with embryos,
181 concentrations fell to below detection limit ($<0.05\ \text{ng/L}$) before the first half of developmental
182 time (indicating rapid embryo uptake).

183 In the 200 L tanks, median EE2 concentration were $0.33\ \text{ng/L}$ at T_0 and $0.11\ \text{ng/L}$ at
184 T_7 , corresponding to a median reduction of 66% of the EE2 dissolved in water over 7 days
185 (see Supplementary Figure S1). We found no significant effects of sampling period on the
186 EE2 measures at T_0 (ANOVA, $F_3 = 1.20$, $p = 0.35$) nor on the weekly reduction of EE2 in the
187 tanks ($F_3 = 1.88$, $p = 0.19$; excluding an unexplained outlier, see Figure S1 for discussion).
188 The median EE2 concentration across T_0 and T_7 samples in the EE2-treated tanks was 0.2
189 ng/L from the first water exchange on, i.e. from 47 dpf on (EE2 was also detected in samples
190 from control aquaria, see Figure S1 for discussion).

191 For the gene expression analyses, the first sampling of 12 embryos per family and
192 treatment took place at 21 dpf, i.e. well before hatching could be expected. Embryos were
193 immediately transferred to RNAlater (Thermo Scientific, Reinach, Switzerland). At 27 and 28
194 dpf, the incubation temperature was raised to 10°C and 11.5°C , respectively, in order to
195 induce and synchronize hatching. The second sampling took place at day of peak hatching,
196 i.e. 31 dpf (8 embryos per family and treatment). The third sampling took place at 52 dpf (5
197 yolk-sac larvae per family and treatment). Hatchlings and yolk-sac larvae were narcotized
198 with $0.5\ \text{mL/L}$ KoiMed (fishmed GmbH, Galmiz, CH) for five minutes and then decapitated.
199 The heads were immediately transferred to RNAlater. All samples were stored at -80°C .

200 RNA was extracted using the QIAgen 96 RNeasy Universal Tissue Kit (QIAGEN,
201 Hombrechtikon, Switzerland). Manufacturer instructions were followed except that
202 centrifugation (Eppendorf 5804 R centrifuge with an A-2-DWP rotor; Eppendorf,
203 Schönenbuch, Switzerland) was done twice as long at half the speed. Because the RNA
204 extraction protocol did not include a DNase treatment, DNA traces inside the RNA samples

205 were amplified to determine the sdY genotype [26] of each individual, using the 18S gene as
206 PCR internal control, either in a multiplex reaction used for samples with high amount of
207 DNA, or after a second PCR amplification in single reactions with half the amounts of the
208 respective primers each for samples with low DNA content (see Maitre *et al.* [27] for a more
209 detailed protocol). Based on the sdY genotype, one female and one male per family and
210 treatment group was haphazardly chosen for further analyses (in 2 of in total 30 combinations
211 of family x treatment x time point, two females or two males, respectively, were used because
212 only one sex could be found in the sample).

213 The RNA extracts were provided for library preparation in a equimolar concentration
214 of 6 ng/ μ L in 100 μ L of volume. 50 μ L (*i.e.* 300 ng of RNA) were each used for library
215 preparation on a robot using the Truseq Stranded RNA protocol (Illumina, Part# 15026495
216 Rev. A), then introduced in the Illumina sequencing platform (HiSeq 2500) for 100 cycles of
217 multiplexed paired-end reads sequencing. The total 60 samples were sequenced in ten lanes
218 (six samples per lane).

219

220 2.2 Bioinformatics pipeline

221 The RNA-seq analysis steps are described in Maitre *et al.* [27]. To summarize, reads were
222 quality trimmed or filtered, and PCR duplicates were removed, resulting in a set of 60 RNA
223 libraries with, on average, 2*40 millions of 80 bp reads each (standard deviation of six
224 million reads). Reads from 24 libraries were assembled using Trinity (version 2.0.3; [34]).
225 The resulting assembly was filtered to remove sequences that did not have a significant match
226 against the UniprotKB/Swiss-Prot database (version of October the 16th 2015) [35] using
227 blastx (version 2.2.26; [36]). Reads from all libraries were pseudo-mapped onto the
228 assembled transcriptome using Kallisto (version 0.42) [37], and the read counts of the
229 different isoforms of each gene were summed up for principal component analysis (PCA) and
230 differential expression analysis at the gene level. PCA was performed on TMM-normalized
231 [38] log₂(count-per-million) values (CPM). Differential expression analysis was performed
232 using the limma-voom Bioconductor package (version 3.26.3) [39, 40], with samples quality
233 weights [41], on CPM values that were additionally cyclic loess normalized. In the linear
234 model we considered developmental stage, sex and treatment as a combined variable (with
235 twelve possible levels) and sib-group as an independent variable. A linear model was then fit
236 for each gene, coefficients and standard errors were computed for all the contrasts of interest.
237 Q-values [42] were calculated for each gene, and a threshold of $q = 0.15$ was used to call
238 differentially expressed genes. Transcripts were annotated by referring to the Gene Ontology
239 (GO) terms (Biological Function) [43] associated to similar genes in zebrafish. An enrichment
240 analysis of GO terms was performed on differentially expressed genes using the goseq
241 Bioconductor package (version 1.22.0; [44])

242 RNA extraction quality, male sex associated locus PCR, RNA-sequencing reads quality,
243 transcriptome assembly, gene expression principal component and differential gene
244 expression within control individuals are shown in Maitre *et al.* [27].

245

246 3. Results

247

248 3.1 Differential gene expression

249 In order to test for sex-specific effects, we compared the changes in gene expression under
250 EE2 treatment for individuals of the same sex at the same developmental stage (Table 1).
251 Under EE2 treatment, at the embryo stage there was an altered expression of several hundred
252 genes in males (Table 1a, Figures S2a, S3, S4, Table S1), but only of a few genes in females
253 (Table 1a, Figure S2b). At hatching day males showed an alteration in the expression of
254 nearly 12,000 genes (Table 1b, Figure S2c, S5, S6, Table S2), while females at the same time
255 point displayed an alteration in the expression of over 24,000 genes (Table 1b, Figure S2d,

256 S7, S8, Table S3). At the first feeding stage only very few genes appeared altered in males
257 (Table 1c, Figure S2e), whereas in female still around 14,000 genes were affected (Table 1c,
258 Figure S2f, S9, S10, Table S4).

259 In Table 2, the sex-specific alterations in gene expression are split according to the
260 direction of the changes. Around hatching, about 5,000 genes were up-regulated in EE2-
261 treated males while down-regulated in EE2-treated females, and about 4,000 were down-
262 regulated in EE2-treated males while up-regulated in EE2-treated females (Table 2). The
263 remaining sex-specific reactions to the EE2 treatment were mainly up- or down-regulation in
264 one sex while there was apparently no change in the other sex (Table 2). See Figures S11 and
265 S12 and Table S5 for EE2 effects on gene expression in both, males and females.

266

267 3.2 Does EE2 treatment feminize males and masculinize females?

268 After focusing on sex-specific gene expression changes induced by EE2 treatment, we
269 compared control males against EE2-treated females and control females against EE2 treated
270 males (Table 1). The aim of this analysis was to investigate whether the EE2 treatment would
271 feminize males, masculinize females, or increase the differences in gene expression between
272 sexes. At embryo stage, we found three genes differentially expressed between EE2 treated
273 males and control females (Table 1a) and 863 genes between control males and EE2 treated
274 females (Table 1a). On hatching day, we found no differences in gene expression levels
275 between control females and males treated with EE2 (Table 1b) and only two genes differing
276 between control males and EE2 treated females (Table 1b). At first feeding stage, EE2 treated
277 males expressed 5,533 genes differently in comparison to control females (Table 1c), while
278 gene expression in control males differed in 42 genes only from the gene expression of EE2-
279 treated females (Table 1c). Overall, there does appear to be transcriptome evidence of
280 feminization of males and of masculinization of females.

281

282 3.3 Gene ontology enrichment analysis

283 Tables S1-S5 show the top 25 Gene Ontology terms enriched among genes differentially
284 expressed in the above-described contrasts. Figures S3-S12 illustrate the Gene Ontology
285 terms significantly enriched, grouped by relatedness. Table 3 shows a summary take-home
286 from these Gene Ontology enrichments. Interestingly, many differences affect developmental
287 processes. In male embryos, we also notice response processes, whether endocrine, xenobiotic
288 or immunological. Finally, in female larvae a wide range of processes are affected, including
289 behaviour and digestion.

290

291 3.4 Sex differentiation

292 Exposure to EE2 delayed the onset of morphological sex differentiation while exposure to *P.*
293 *fluorescens* (PF) showed no effects (Table 4). Figure 1 shows the rates of juveniles without
294 any testis or ovarian tissues over the five monthly samples, separately for fish that were or
295 were not exposed to EE2, to illustrate the delaying effects of EE2. First signs of
296 morphological sex differentiation could be observed at the 2nd sample, but still more than 30%
297 of the fish showed no sign of morphological sex differentiation at the end of the observation
298 period (Figure 1).

299 Among those fish where first stages of sex differentiation could be identified, only
300 testis tissue could be observed at the 2nd sampling (79 dpf), while the rate of ovarian tissue
301 rose quickly to 70.8%, 75.9%, then 79.3% over the 3rd, 4th, and 5th sampling periods,
302 respectively. The rates of ovarian versus testis tissue did not differ between EE2-treated and
303 sham-treated grayling ($\chi^2 = 0.26$, $p = 0.61$).

304 Genetic sexing of all 103 individuals of the 4th and 5th sample (135 dpf and 159-163
305 dpf) revealed a genetic sex ratio of 55.3% males that did not deviate from equal sex ratio (χ^2
306 = 1.2, d.f. = 1, $p = 0.28$). Equal sex ratios can therefore be assumed for all earlier samples. At

307 these two last sampling days, all females except three showed ovarian tissue (ovaries or testis-
308 to-ovaries). The three exceptions were from the sham-treated controls and showed testis
309 tissue, i.e. no genetic female was undifferentiated at that these last sampling dates. In contrast,
310 45 of a total of 57 genetic males were still undifferentiated at that time, 10 showed testis
311 tissue, one showed the testis-to-ovaries phenotype, and one had ovaries. The latter two males
312 with ovarian tissues had both been EE2-treated.

313

314 **4. Discussion**

315 We tested and described the effects of exposure to low, ecologically relevant, concentrations
316 of EE2 on sex-specific gene expression in embryos and larvae of grayling, a river-dwelling
317 salmonid that is often exposed to this type of pollution. From what is known about possible
318 EE2 effects on fish in general, we expected that this common micropollutant may (i) affect
319 sex determination of grayling by influencing the few initial triggers that start the canalized
320 developmental process of gonad formation, and (ii) be toxic to the embryos and larvae
321 because it interferes with different types of physiological processes, especially those that are
322 endocrinologically regulated (see references cited in the Introduction). We therefore expected
323 EE2 to have significant effects on gene expression at various developmental levels, and we
324 indeed found such effects at all the developmental stages we studied here. However, we had
325 no clear *a priori* expectancy about whether EE2 would also affect the genetic males and
326 genetic females differently at any of these stages.

327 We started from the premise that sex in gonochoristic species is a threshold trait, i.e. a
328 canalized developmental process that has one or few initial triggers [16]. In grayling, the
329 initial trigger (or triggers) that determine phenotypic sex happen during embryogenesis well
330 before hatching, because over 25,000 genes are already differentially expressed between
331 genetic males and females at the day of hatching [27]. The 15 genes that Maitre et al. [27]
332 found to be differentially expressed in genetic males and females at the embryo stage 10 days
333 before hatching suggest that sex determination starts around then, i.e. at a time when the
334 embryos had already been exposed to EE2 for several days in the present study.

335 One possible scenario is hence that EE2 could tip the balance at the early steps of sex
336 determination so that all individuals follow the developmental process that leads to the female
337 phenotype regardless of their sdY genotype (i.e. sex reversal of genetic males). If so, EE2
338 would not be expected to show sex-genotype specific effects on gene expression during later
339 stages of sex differentiation. However, we found strong interactions between genetic sex and
340 EE2 on gene expression. These sex-specific reactions to EE2 also depended on the
341 developmental stages we studied. At embryo stage, expression of only few genes seemed
342 biased in genetic females, but gene expression in genetic males was already significantly
343 affected, with about 700 genes up- or down-regulated under the influence of EE2. The
344 outcome was somewhat reverse at the larval stage: now only few genes of genetic males
345 seemed to be affected by EE2, while over 14,000 genes were differentially expressed in
346 genetic females. An even more pronounced effect of EE2 could be seen at the day of
347 hatching: far over 20,000 genes showed differential expression, and about 9,000 of them were
348 either upregulated in genetic females and down-regulated in genetic males or down-regulated
349 in genetic females and upregulated in genetic males.

350 The strong sex-specific responses to EE2 suggest that exposure to ecologically
351 relevant concentrations of EE2 during embryogenesis did not simply tip the balance at early
352 steps of sex determination so that all individuals would become phenotypic females and
353 would show similar patterns of gene expression from then on. Instead, our observations
354 suggest that genetic sex largely determined phenotypic sex, and that EE2 then interfered with
355 sex-specific gene expression, creating the strong sex-specific reactions to EE2. This
356 conclusion is supported by the observation that the low concentrations of EE2 we used are
357 commonly observed in natural rivers and streams while there is little evidence for complete

358 and population-wide sex reversal, even if natural populations sometimes show distorted sex
359 ratios [45].

360 While the interaction between EE2 and genetic sex on gene expression suggested that
361 exposure to EE2 is mostly interfering with the development of a phenotype that would
362 correspond to the genotypic sex, we nevertheless found two genetic males that had developed
363 ovarian tissues when exposed to EE2. It is possible that we missed some sex-reversal and the
364 rate of sex reversal is higher, because we learned only during the course of the study that the
365 grayling is a rare example (and probably even the only one so far) of an undifferentiated
366 gonochorist that goes through an all-male stage before gonads differentiate into testes and
367 ovaries [27]. Testis tissue in early juveniles can therefore not be interpreted as evidence for
368 normal development of a male phenotype, and we probably stopped the study too early to get
369 a reliable estimate of the rate of sex reversals in grayling exposed to EE2. However, by the
370 end of the study, nearly all genetic females had developed ovarian tissue. This suggests that
371 the rate of sex reversals is either low indeed, or that sex reversal would slow down gonad
372 development so much that we would have missed many sex-reversed individuals within our
373 observational window. We know of no examples or arguments in the literature that would
374 support the latter possibility. However, we found that ecologically relevant concentrations of
375 EE2 could affect sex determination in at least some individuals. It would therefore be
376 interesting to learn what makes an early embryo susceptible or resistant against chemically
377 induced sex reversal.

378 Our gene expression analysis suggested that exposure to EE2 induces effects in the
379 transcriptomes of the brain that could be interpreted as partial sex reversal, especially around
380 hatching time when many genes were differentially upregulated in genetic females and
381 downregulated in genetic male, and vice versa, under the influence of EE2. At the day of
382 hatching, genetic males appeared to have a somewhat feminized transcriptome when treated
383 with EE2. This effect seemed to cease before the first feeding stage. Genetic females seemed
384 to experience some form of masculinization of the transcriptome of the brain, at least from
385 hatching day to first feeding stage.

386 Estrogens are known to affect functions of the nervous system, including synapsis
387 homeostasis [46], neurogenesis [47], and sexual differentiation [47-49]. The latter association
388 is particularly interesting here because estrogens have repeatedly been shown to induce, to
389 some degree, a masculinization of the brain [47-49]. This is because androgenic hormones,
390 produced mainly in the male gonads, are transformed by brain cells into estrogen that
391 subsequently act locally via specific receptors to induce typical male development [48].

392 In salmonids, gonadal precursor cells typically differentiate during the weeks that
393 follow hatching [17, 50]. During this period, the emergence of an endogenous synthesis of
394 sexual hormones could explain why we observe a divergent response between sexes,
395 especially if we consider that endocrine active compounds often elicits non-monotonic dose-
396 responses [51, 52]. Such dose-effects could explain why we observed strong sex-specific
397 responses to EE2 at hatching day and why these responses partly declined towards first
398 feeding stage. Apart from the likely effects of EE2 on normal development of male and
399 female phenotypes, exposure to EE2 also affected the expression of genes linked to many
400 other physiological systems, including, for example, various aspects of the immune system,
401 the endocrine system, the development of skeletal muscles, or of the digestive system.

402 In conclusion, exposure to ecologically relevant concentrations of EE2 during early
403 embryogenesis may potentially tip the balance at early steps of sex determination so that all
404 individuals follow the developmental process that leads to the female phenotype regardless of
405 their sdY genotype (i.e. sex reversal of genetic males). If so, and if gene expression is then
406 mainly determined by this gonad development, EE2 would probably not be expected to show
407 strong sex-genotype specific effects on gene expression during later stages of sex
408 differentiation. However, we found strong effects of both, EE2 and genetic sex, on gene

409 expression at three different developmental stages shortly after the onset of sex determination.
410 Such sex-specific effects of EE2 suggest that sex reversal has mostly not happened, although
411 we found two genetic males with gonadal tissues at the end of the study. Our observations
412 support earlier conclusions that ecologically relevant concentrations of EE2 act mostly toxic
413 (i.e. affect growth and viability) but have little effect on sex determination in grayling [53],
414 and that the observed sex ratio distortion may in the population [45] may be due to sex-
415 specific survival instead rather than induced sex reversal.

416

417 **5. Acknowledgements**

418 We thank B. Bracher, U. Gutmann, C. Küng, R. Mani, M. Schmid, and T. Vuille from the
419 Fishery Inspectorate Bern for permissions and provision of fish for experimental
420 fertilizations, the *Pachtverein Thun* for catching the wild fish, the Saint-Sulpice pump station
421 for zooplankton supplies, T. Braunbeck for access to the histology laboratory, the Genomic
422 Technologies Facility of the University of Lausanne for library preparation and sequencing,
423 and C. Berney, T. Bösch, J. Buser, E. Clark, B. des Monstiers, M. dos Santos, J. Kast, C.
424 Luca, Y. Marendaz, D. Nusbaumer, D. Olbrich, E. Pereira Alvarez, M. Pompini, A.-L.
425 Roulin, V. Sentchilo, R. Sermier, B. von Siebenthal, and D. Zeugin for assistance in the field
426 or the laboratory. This project was authorized by the veterinary authorities of the cantons of
427 Bern (BE118/14) and Vaud (VD2710; VD2956) and financially supported by the Swiss
428 Federal Office for the Environment and the Swiss National Science Foundation
429 (31003A_159579).

430

431 **6. References**

- 432 1. Arnold KE, Brown AR, Ankley GT, Sumpter JP. Medicating the environment:
433 assessing risks of pharmaceuticals to wildlife and ecosystems. *Phil Trans R Soc B*
434 2014; 369(1656):20130569.
- 435 2. Tiedeken EJ, Tahar A, McHugh B, Rowan NJ. Monitoring, sources, receptors, and
436 control measures for three European Union watch list substances of emerging concern
437 in receiving waters - A 20 year systematic review. *Sci Total Env* 2017; 574:1140-
438 1163.
- 439 3. Yang YY, Cao XH, Zhang MM, Wang J. Occurrence and distribution of endocrine-
440 disrupting compounds in the Honghu Lake and East Dongting Lake along the Central
441 Yangtze River, China. *Environ Sci Pollut Res* 2015; 22(22):17644-17652.
- 442 4. Vulliet E, Cren-Olive C. Screening of pharmaceuticals and hormones at the regional
443 scale, in surface and groundwaters intended to human consumption. *Environm Pollut*
444 2011; 159(10):2929-2934.
- 445 5. Morthorst JE, Korsgaard B, Bjerregaard P. Severe malformations of eelpout (*Zoarces*
446 *viviparus*) fry are induced by maternal estrogenic exposure during early
447 embryogenesis. *Mar Environ Res* 2016; 113:80-87.
- 448 6. Brazzola G, Chèvre N, Wedekind C. Additive genetic variation for tolerance to
449 estrogen pollution in natural populations of Alpine whitefish (*Coregonus* sp.,
450 Salmonidae). *Evol Appl* 2014; 7(9):1084-1093.
- 451 7. Rodenas MC, Cabas I, Garcia-Alcazar A, Meseguer J, Mulero V, Garcia-Ayala A.
452 Selective estrogen receptor modulators differentially alter the immune response of
453 gilthead seabream juveniles. *Fish Shellfish Immun* 2016; 52:189-197.
- 454 8. Hua JH, Han J, Wang XF, Guo YY, Zhou BS. The binary mixtures of megestrol
455 acetate and 17 alpha-ethynylestradiol adversely affect zebrafish reproduction.
456 *Environm Pollut* 2016; 213:776-784.
- 457 9. Armstrong BM, Lazorchak JM, Jensen KM, Haring HJ, Smith ME, Flick RW, Bencic
458 DC, Biales AD. Reproductive effects in fathead minnows (*Pimphales promelas*)

- 459 following a 21 d exposure to 17 alpha-ethinylestradiol. *Chemosphere* 2016; 144:366-
460 373.
- 461 10. Volkova K, Caspillo NR, Porseryd T, Hallgren S, Dinnetz P, Porsch-Hallstrom I.
462 Developmental exposure of zebrafish (*Danio rerio*) to 17 alpha-ethinylestradiol
463 affects non-reproductive behavior and fertility as adults, and increases anxiety in
464 unexposed progeny. *Horm Behav* 2015; 73:30-38.
- 465 11. Bhandari RK, vom Saal FS, Tillitt DE. Transgenerational effects from early
466 developmental exposures to bisphenol A or 17 alpha-ethinylestradiol in medaka,
467 *Oryzias latipes*. *Sci Rep* 2015; 5.
- 468 12. Schwindt AR, Winkelman DL. Estimating the effects of 17 alpha-ethinylestradiol on
469 stochastic population growth rate of fathead minnows: a population synthesis of
470 empirically derived vital rates. *Ecotoxicology* 2016; 25(7):1364-1375.
- 471 13. Kidd KA, Paterson MJ, Rennie MD, Podemski CL, Findlay DL, Blanchfield PJ, Liber
472 K. Direct and indirect responses of a freshwater food web to a potent synthetic
473 oestrogen. *Phil Trans R Soc B* 2014; 369(1656).
- 474 14. Prokkola JM, Katsiadaki I, Sebire M, Elphinstone-Davis J, Pausio S, Nikinmaa M,
475 Leder EH. Microarray analysis of di-n-butyl phthalate and 17 alpha ethinyl-oestradiol
476 responses in three-spined stickleback testes reveals novel candidate genes for
477 endocrine disruption. *Ecotoxicol Environ Saf* 2016; 124:96-104.
- 478 15. Harding LB, Schultz IR, da Silva DAM, Ylitalo GM, Ragsdale D, Harris SI, Bailey S,
479 Pepich BV, Swanson P. Wastewater treatment plant effluent alters pituitary gland
480 gonadotropin mRNA levels in juvenile coho salmon (*Oncorhynchus kisutch*). *Aquat*
481 *Toxicol* 2016; 178:118-131.
- 482 16. Beukeboom LW, Perrin N: The evolution of sex determination. Oxford: Oxford
483 University Press; 2014.
- 484 17. Devlin RH, Nagahama Y. Sex determination and sex differentiation in fish: an
485 overview of genetic, physiological, and environmental influences. *Aquaculture* 2002;
486 208(3-4):191-364.
- 487 18. Wedekind C. Demographic and genetic consequences of disturbed sex determination.
488 *Phil Trans R Soc B* in press.
- 489 19. Piferrer F. Endocrine sex control strategies for the feminization of teleost fish.
490 *Aquaculture* 2001; 197(1-4):229-281.
- 491 20. Orn S, Holbech H, Norrgren L. Sexual disruption in zebrafish (*Danio rerio*) exposed
492 to mixtures of 17 alpha-ethinylestradiol and 17 beta-trenbolone. *Environ Toxicol*
493 *Pharmacol* 2016; 41:225-231.
- 494 21. Xu N, Chen PY, Liu L, Zeng YQ, Zhou HX, Li S. Effects of combined exposure to 17
495 alpha-ethinylestradiol and dibutyl phthalate on the growth and reproduction of adult
496 male zebrafish (*Danio rerio*). *Ecotoxicol Environ Saf* 2014; 107:61-70.
- 497 22. Soffker M, Tyler CR. Endocrine disrupting chemicals and sexual behaviors in fish - a
498 critical review on effects and possible consequences. *Crit Rev Toxicol* 2012;
499 42(8):653-668.
- 500 23. Scholz S, Kluver N. Effects of endocrine disrupters on sexual, gonadal development in
501 fish. *Sex Dev* 2009; 3(2-3):136-151.
- 502 24. Senior AM, Lim JN, Nakagawa S. The fitness consequences of environmental sex
503 reversal in fish: a quantitative review. *Biol Rev* 2012; 87(4):900-911.
- 504 25. Segner H, Casanova-Nakayama A, Kase R, Tyler CR. Impact of environmental
505 estrogens on fish considering the diversity of estrogen signaling. *Gen Comp*
506 *Endocrinol* 2013; 191:190-201.
- 507 26. Yano A, Nicol B, Jouanno E, Quillet E, Fostier A, Guyomard R, Guiguen Y. The
508 sexually dimorphic on the Y-chromosome gene (sdY) is a conserved male-specific Y-
509 chromosome sequence in many salmonids. *Evol Appl* 2013; 6(3):486-496.

- 510 27. Maitre D, Selmoni OM, Uppal A, Marques da Cunha L, Wilkins LGE, Roux J,
511 Mobley K, Knörr S, Robinson-Rechavi M, Wedekind C. Sex differentiation in
512 grayling (Salmonidae) goes through an all-male stage and is delayed in genetic males
513 who instead grow faster. *Submitted manuscript* 2017.
- 514 28. Wedekind C, Küng C. Shift of spawning season and effects of climate warming on
515 developmental stages of a grayling (Salmonidae). *Cons Biol* 2010; 24(5):1418-1423.
- 516 29. von Siebenthal BA, Jacob A, Wedekind C. Tolerance of whitefish embryos to
517 *Pseudomonas fluorescens* linked to genetic and maternal effects, and reduced by
518 previous exposure. *Fish Shellfish Immunol* 2009; 26(3):531-535.
- 519 30. OECD guideline for the testing of chemicals 203 (fish acute toxicity test), Annex 2. 9.
520 [<http://www.oecd.org/>]
- 521 31. Clark ES, Pompini M, da Cunha LM, Wedekind C. Maternal and paternal
522 contributions to pathogen resistance dependent on development stage in a whitefish
523 (Salmonidae). *Funct Ecol* 2014; 28(3):714-723.
- 524 32. Escher BI, Bramaz N, Quayle P, Rutishauser S, Vermeirssen ELM. Monitoring of the
525 ecotoxicological hazard potential by polar organic micropollutants in sewage
526 treatment plants and surface waters using a mode-of-action based test battery. *J*
527 *Environ Monit* 2008; 10:622-631.
- 528 33. Ternes TA, Stumpf M, Mueller J, Haberer K, Wilken R-D, Servos M. Behavior and
529 occurrence of estrogens in municipal sewage treatment plants -- I. Investigations in
530 Germany, Canada and Brazil. *Sci Total Env* 1999; 225:81-90.
- 531 34. Grabherr MG, Haas BJ, Yassour M, Levin JZ, Thompson DA, Amit I, Adiconis X,
532 Fan L, Raychowdhury R, Zeng Q *et al.* Full-length transcriptome assembly from
533 RNA-Seq data without a reference genome. *Nat Biotechnol* 2011; 29(7):644--652.
- 534 35. Boeckmann B. The SWISS-PROT protein knowledgebase and its supplement
535 TrEMBL in 2003. *Nucleic Acids Res* 2003; 31(1):365--370.
- 536 36. AST® Command Line Applications User Manual.
537 [<https://www.ncbi.nlm.nih.gov/books/NBK279690/>]
- 538 37. Bray N, Pimentel H, Melsted P, Pachter L. Near-optimal probabilistic RNA-seq
539 quantification. *Nat Biotechnol* 2016; 34:525-527.
- 540 38. Robinson MD, Oshlack A. A scaling normalization method for differential expression
541 analysis of RNA-seq data. *Genome Biol* 2010; 11(3):R25.
- 542 39. Law CW, Chen Y, Shi W, Smyth GK. voom: Precision weights unlock linear model
543 analysis tools for RNA-seq read counts. *Genome Biol* 2014; 15(2):R29.
- 544 40. Ritchie ME, Phipson B, Wu D, Hu Y, Law CW, Shi W, Smyth GK. limma powers
545 differential expression analyses for RNA-sequencing and microarray studies. *Nucleic*
546 *Acids Res* 2015; 43(7):e47--.
- 547 41. Liu R, Holik AZ, Su S, Jansz N, Chen K, Leong HS, Blewitt ME, Asselin-Labat M-L,
548 Smyth GK, Ritchie ME. Why weight? Modelling sample and observational level
549 variability improves power in RNA-seq analyses. *Nucleic Acids Res* 2015; 43(15):e97.
- 550 42. Storey JD. The positive false discovery rate: a Bayesian interpretation and the q-value.
551 *The Annals of Statistics* 2003; 31(6):2013--2035.
- 552 43. Ashburner M, Ball CA, Blake JA, Botstein D, Butler H, Cherry JM, Davis AP,
553 Dolinski K, Dwight SS, Eppig JT *et al.* Gene Ontology: tool for the unification of
554 biology. *Nat Genet* 2000; 25(1):25-29.
- 555 44. Young MD, Wakefield MJ, Smyth GK, Oshlack A. Gene ontology analysis for RNA-
556 seq: accounting for selection bias. *Genome Biol* 2010; 11(2).
- 557 45. Wedekind C, Evanno G, Székely T, Pompini M, Darbellay O, Guthruf J. Persistent
558 unequal sex ratio in a population of grayling (Salmonidae) and possible role of
559 temperature increase. *Cons Biol* 2013; 27(1):229-234.

- 560 46. Saldanha CJ, Ramage-Healey L, Schlinger BA. Synaptocrine signaling: steroid
561 synthesis and action at the synapse. *Endocrine Reviews* 2011; 32(4):532-549.
- 562 47. Diotel N, Le Page Y, Mouriec K, Tong SK, Pellegrini E, Valliant C, Anglade I, Brion
563 F, Pakdel F, Chung BC *et al.* Aromatase in the brain of teleost fish: Expression,
564 regulation and putative functions. *Front Neuroendocrinol* 2010; 31(2):172-192.
- 565 48. Wu MV, Manoli DS, Fraser EJ, Coats JK, Tollkuhn J, Honda SI, Harada N, Shah NM.
566 Estrogen masculinizes neural pathways and sex-specific behaviors. *Cell* 2009;
567 139(1):61-72.
- 568 49. Wu MV, Shah NM. Control of masculinization of the brain and behavior. *Current*
569 *Opinion in Neurobiology* 2011; 21(1):116-123.
- 570 50. Krisfalusi M, Cloud JG. Gonadal sex reversal in triploid rainbow trout (*Oncorhynchus*
571 *mykiss*). *Journal of Experimental Zoology* 1999; 284(4):466-472.
- 572 51. Li L, Andersen ME, Heber S, Zhang Q. Non-monotonic dose-response relationship in
573 steroid hormone receptor-mediated gene expression. *Journal of Molecular*
574 *Endocrinology* 2007; 38(5-6):569-585.
- 575 52. Vandenberg LN, Colborn T, Hayes TB, Heindel JJ, Jacobs DR, Lee DH, Shioda T,
576 Soto AM, vom Saal FS, Welshons WV *et al.* Hormones and endocrine-disrupting
577 chemicals: low-dose effects and nonmonotonic dose responses. *Endocrine Reviews*
578 2012; 33(3):378-455.
- 579 53. Pompini M, Buser AM, Thali MR, Von Siebenthal BA, Nusslé S, Guduff S,
580 Wedekind C. Temperature-induced sex reversal is not responsible for sex-ratio
581 distortions in grayling *Thymallus thymallus* or brown trout *Salmo trutta*. *J Fish Biol*
582 2013; 83(2):404-411.
- 583
- 584

585 **Table 1.** Number of genes that are differentially expressed ($q < 0.15$) in males and females
586 tested at (a) embryo stage, (b) hatchling stage, and (c) larval stage at the onset of exogenous
587 feeding, each after they had been exposed to EE2 or sham exposed.
588
589

		sham- exposed females	EE2- exposed males	EE2- exposed females
a) Embryos				
	Sham-exposed males	15 ¹	704	863
	Sham-exposed females		3	32
b) Hatchlings				
	Sham-exposed males	25,372 ¹	11,619	2
	Sham-exposed females		0	24,728
c) Larvae				
	Sham-exposed males	1,110 ¹	4	41
	Sham-exposed females		5,533	14,395

590 ¹ reported in Maitre *et al.* [27]
591

592 Table 2. Number of genes that were upregulated, i.e. had a positive log fold change of
593 expression with $q > 0.15$ (UP), experienced no significant change in expression (NO), or were
594 downregulated (DO) under exposure to EE2.

595
596
597

		Males UP	Males NO	Males DO	Total
a) Embryos					
	Females UP	0	12	0	12
	Females NO	245	34,613	457	35,315
	Females DO	1	18	1	20
	Total	246	34,643	458	35,347
b) Hatchlings					
	Females UP	147	7,959	4,043	12,149
	Females NO	1,341	8,267	1,009	10,617
	Females DO	4,962	7,495	117	12,574
	Total	6,450	23,721	5,169	35,340
c) Larvae					
	Females UP	3	8,589	0	8,592
	Females NO	1	20,949	0	20,950
	Females DO	0	5,803	0	5,803
	Total	4	35,341	0	35,345

598
599

600 Table 3. Summary interpretation of the differential gene expression analysis. The
601 characterization of the biological processes relies on the gene ontology enrichment analysis of
602 differentially expressed genes. Feminization and masculinization represent the situation where
603 few genes (<100) are detected as differentially expressed, under EE2 treatment, in
604 comparison to control female or control male, respectively. See Supplementary Figures S2-
605 S13 and Tables S1-S5 for more detailed information.

606
607

Developmental stage	Sex differences in gene expression ¹	EE2-effects in males	EE2- effects in females
Embryos	Weak	On genes associated with central nervous system development, immune response development, endocrine system and response to xenobiotic stimuli.	Unclear.
Hatchlings	Strong	On genes associated with skeletal muscle function and locomotion, metabolism of fatty acids and carbohydrates, DNA-repair and hindbrain morphogenesis. Possible feminization.	On genes associated with immune system activation, DNA-repair, axonal growth and synaptic transmission. Possible masculinization.
Larvae	Strong	A few genes only.	On genes associated with DNA-repair, signal transduction, immune system activation, carbohydrate metabolism, mechanosensory behavior and digestion. Possible masculinization.

608 ¹ Results from Maitre *et al.* [27]

609

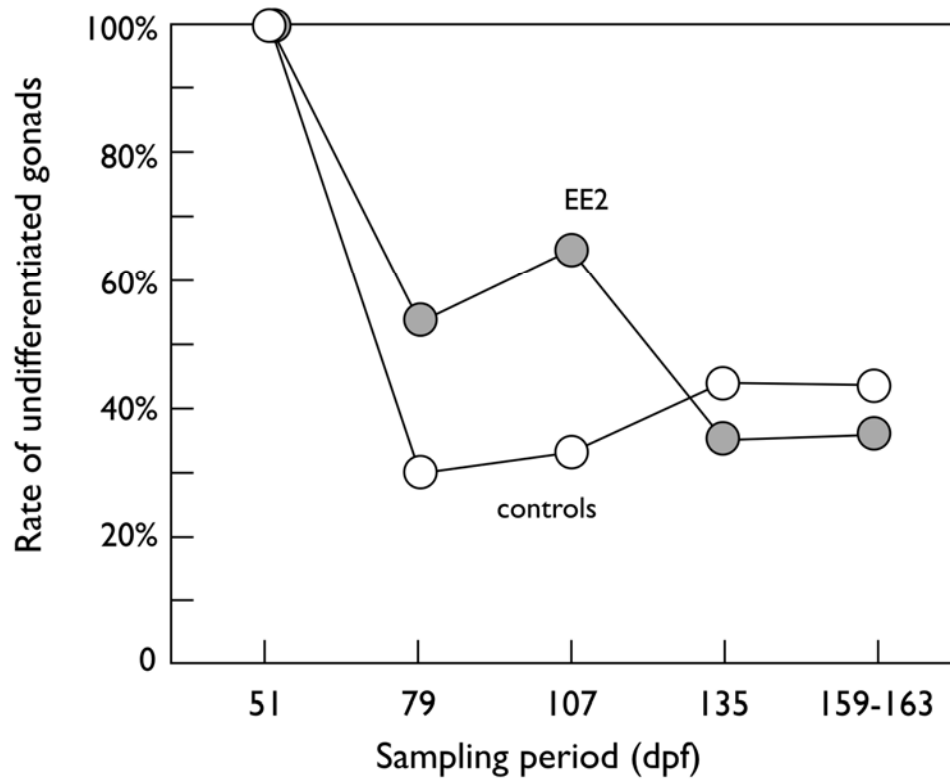
610 Table 4. Likelihood ratio test on rate of undifferentiated gonads explained by time point of
611 sampling and exposure to ethinylestradiol (EE2), the bacterium *P. fluorescens* (PF), or both
612 (EE2 x PF). $N_{\text{total}} = 251$.

613
614

Effect	χ^2	d.f.	P
Time	72.3	4	<0.0001
EE2	<0.1	1	1.0
PF	<0.1	1	1.0
Time x EE2	9.5	4	0.05
Time x PF	0.9	4	0.92
EE2 x PF	<0.1	1	1.0
Time EE2 x PF	4.7	4	0.32

615
616
617

618 **Figure 1**
619 Rates of undifferentiated males and females (i.e. no testis nor ovarian tissues) when exposed
620 to EE2 (closed symbols) or sham exposed (controls, open symbols), with or without
621 additional exposure to *P. fluorescens* during embryogenesis each (exposure to the microbe
622 showed no significant effects on sex differentiation). See text and Table 4 for statistics.
623



624