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# Assessing the environmental impacts of conventional and organic scenarios of rainbow trout farming in France

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1     **Assessing the environmental impacts of conventional and organic scenarios of rainbow**  
2   **trout farming in France**

3

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14 **Highlights**

- 15 • We assessed environmental impacts of conventional and organic trout farming
- 16 • Constraints for conventional and organic farming were applied in a modelled farm
- 17 • Organic farming reduced environmental impacts per tonne in most of the categories
- 18 • Freshwater ecotoxicity and energy per tonne dropped by ~30-35% in organic farming
- 19 • Benefits of organic farming were more marked using a surface-based functional unit

20 **Abstract**

21 In France, rainbow trout farming in flow-through systems raises environmental concerns. To  
22 address this, there is a growing interest in organic aquaculture. In this study, we employed an  
23 attributional life cycle assessment (LCA) to analyze the environmental impacts of rainbow  
24 trout production, comparing conventional and organic practices in a model fish farm. Our life  
25 cycle impact assessment revealed that organic farming significantly reduced environmental  
26 impacts per tonne of trout in seven of the nine selected impact categories. Notably, freshwater  
27 ecotoxicity exhibited the most significant difference, with organic systems showing a 35%  
28 decrease. The only exceptions were freshwater eutrophication and water dependence, where  
29 organic production led to higher impacts per tonne of trout. In conventional farming,  
30 emissions amounted to 14 kg of P eq./tonne, whereas in organic farming, the emissions were  
31 slightly higher (15 kg of P eq./tonne). For water dependence, one tonne of trout production in  
32 the conventional system mobilized  $128 \cdot 10^3 \text{ m}^3$  vs.  $185 \cdot 10^3 \text{ m}^3$  in the organic system. The  
33 environmental benefits of organic production were even more marked when using a surface-  
34 based functional unit ( $\text{m}^2\text{y}$ ). We demonstrated the benefits of organic trout production from an  
35 environmental perspective. However, our findings highlight the caution needed when  
36 interpreting LCA comparisons of such production systems that can be highly influenced by  
37 methodological choices such as the functional unit used.

38 Keywords: Aquaculture systems; Conventional production; Fish; Life cycle assessment;  
39 Organic production

## 40 **1. Introduction**

41 Rainbow trout (*Oncorhynchus mykiss*) is the primary farmed fish species reared in France and  
42 a significant salmonid species in global aquaculture production (953,000 tonnes in 2021;  
43 FAO, 2022). Only ~20% of this production is performed in seawater, as done in Norway and  
44 Chile, while the vast majority comes from freshwater production, as practised in Iran and  
45 Turkey, the two main producing countries (FAO, 2023). Traditionally, freshwater trout  
46 farming relied on flow-through systems with high water exchange. The lack of space for  
47 expansion and new sites (due to competition with other uses and interests), limited freshwater  
48 availability, and concerns over the sustainability of the aquafeeds are considered key obstacles  
49 to further expansion of conventional flow-through aquaculture systems (Albrektsen et al.,  
50 2022; Chen et al., 2015; Maiolo et al., 2021). As consumer demand for sustainable and  
51 environmentally friendly products grows, there is a rising interest in organic aquaculture,  
52 which aims to integrate best environmental practices, natural resource preservation, and high  
53 animal welfare standards (Ahmed et al., 2020).

54 Organic agriculture is often perceived as more sustainable than conventional farming  
55 (Meemken and Qaim, 2018). Despite occupying only 1.6% of global agricultural land and  
56 accounting for less than 10% of retail sales in most countries (Willer et al., 2023), organic  
57 farming is one of the fastest-growing sectors in the food industry. Organizations such as the  
58 International Federation of Organic Agriculture Movement (IFOAM), the Food and  
59 Agriculture Organization (FAO) and the World Health Organization (WHO), through the  
60 *Codex Alimentarius*, are working towards establishing an internationally agreed definition of  
61 organic practices. In essence, organic farming is an agricultural system that prioritizes the  
62 well-being of ecosystems, encompassing soil, plants, animals, and humans. It relies on  
63 ecological processes, biodiversity and cycles adapted to local conditions rather than using  
64 inputs with adverse effects.

65 Moreover, organic farming promotes fair relationships and a good quality of life for all  
66 involved (IFOAM, 2008). The magnitude of the benefits of organic farming can vary  
67 significantly depending on several factors, such as the farm-specific agricultural practices and  
68 management approaches and local environmental conditions (Pépin et al., 2022; Smith et al.,  
69 2019). Thus, while organic farming generally fosters environmentally friendly practices, the  
70 actual environmental benefits can vary case-by-case (Meier et al., 2015). Therefore, a  
71 comprehensive assessment is necessary to accurately evaluate the overall environmental  
72 advantages of organic farming.

73 Different approaches have been employed to compare the environmental impacts of organic  
74 and conventional farming systems, focusing on specific aspects such as biodiversity (e.g.,  
75 Gabriel et al., 2013; Letourneau and Bothwell, 2008), land use (e.g., Badgley et al., 2007;  
76 Connor, 2022; Gibson et al., 2007), or nutrient emissions (e.g., Nowak et al., 2013). However,  
77 these assessments offer a limited perspective on the overall environmental impacts of  
78 agricultural production. To provide a more comprehensive evaluation, efforts have been made  
79 to develop multi-impact methods that can integrate various environmental impact categories,  
80 enabling a holistic assessment. The reference method is the Life Cycle Assessment (LCA),  
81 which examines the material and energy flows throughout a product's life cycle,  
82 encompassing activities like raw material extraction, processing, manufacturing,  
83 transportation, distribution, product use, maintenance, recycling, and waste management.  
84 LCA is recognized as a comprehensive approach by researchers and international standards  
85 (ISO, 2006; JRC, 2010) and enables a thorough examination of the different stages and  
86 impacts associated with a product's life cycle.

87 Tuomisto et al. (2012) and Meier et al. (2015) performed meta-analysis of Life Cycle  
88 Assessment (LCA) studies comparing the environmental impacts of organic and conventional  
89 terrestrial farming. Their findings indicate that organic farming practices generally yield  
90 positive environmental impacts per unit of area, although not necessarily per product unit.

91 Organic production tends to exhibit higher levels of soil organic matter and reduced nutrient  
92 losses (such as nitrogen leaching, nitrous oxide emissions, and ammonia emissions).  
93 However, when measured per product unit, organic systems were found to have higher levels  
94 of nutrient emissions. Additionally, organic systems demonstrated lower energy requirements  
95 but higher land use, eutrophication potential, and acidification potential per product unit.  
96 Nevertheless, this meta-analysis only concerns land-based production. In aquaculture, to the  
97 best of our knowledge, only three case studies have been published in the peer-reviewed  
98 literature: comparisons of conventional and organic production of shrimps (Jonell and  
99 Henriksson, 2015) and carp (Biermann and Geist, 2019) and comparison of ingredient types  
100 in salmon feeds (Pelletier and Tyedmers, 2007).

101 This study compares the environmental impacts of conventional vs. organic rainbow trout  
102 farming. To do that, we modelled a trout farm, practising conventional or organic rearing  
103 rainbow trout production. Our model aims to simulate a production farm in France, in  
104 Brittany, one of the country's main rainbow trout-producing regions.

## 105 **2. Materials and Methods**

### 106 2.1. Farm model

107 The farm model, developed using the R freeware (R Development Core Team, 2022), has  
108 been partially adapted from previous investigations (Besson et al., 2017, 2016, 2014) to  
109 facilitate the acquisition of input values required for conducting a LCA at the farm level. In  
110 the present study, the model was customized to simulate daily the production of rainbow trout  
111 (*O. mykiss*) in a hypothetical flow-through farm, using actual farm data obtained from surveys  
112 conducted in Brittany in 2022. The hypothetical farm consisted of 12 concrete raceways 100  
113 m<sup>3</sup> each for the pre-growing phase, and 24 concrete raceways 250 m<sup>3</sup> each for grow-out.  
114 Among the 250 m<sup>3</sup> raceways, 50% received first water, meaning that the water entered the  
115 tanks directly from the river, while the remaining 50% received second water, supplied solely  
116 by the outlet water from the upstream raceways (Figure 1). In addition to the raceways, the

117 farm was equipped with five feed storage silos and two warehouses measuring 60 and 80 m<sup>2</sup>  
118 (Figure 1).

119 Fish were initially stocked at 10 g and harvested at a fixed weight of 3,000 g, which was  
120 assumed to have a unique market size. The maximal annual production was fixed at 300  
121 tonnes. Three fry batches were stocked throughout the year to stagger the sales period (Table  
122 1). We simulated a production over three years and used the third year as the reference year  
123 for LCA (i.e., the year where the first batches stocked in the first year reached market size).

124 The various parameters used and the constraints imposed, according to conventional and  
125 organic production scenarios, are elaborated in detail below. We incorporated survey data,  
126 scientific literature, and industry specifications for our analysis. Specifically, we used the  
127 French production specifications for large trout provided by the Interprofessional Committee  
128 for Aquaculture Products (CIPA, 2023) and the regulations for the organic production of  
129 aquaculture species established by the French Ministry of Agriculture and Fisheries (MAAP,  
130 2010). A schematic representation of the modelling approach we employed is depicted in  
131 Figure 2.

### 132 2.1.1. Environmental parameters

133 The daily temperature (T) was modelled using a sinusoidal function with a period of 365  
134 days. As suggested by Seginer and Halachmi (2008), T<sub>n</sub> is given by:

$$135 T(n) = T_m - T_a \times \sin\left(2\pi \times \frac{n + \phi_T}{365}\right) \quad (1)$$

136 where n is a day from 1 to 365, T<sub>m</sub> is the mean water temperature (13 °C), T<sub>a</sub> is the amplitude  
137 of the variation (8°C corresponding to a difference of 2 × 8 = 16 °C between the minimum  
138 and maximum daily temperature across the whole year), and φ<sub>T</sub> is the phase shift (time-delay  
139 of 27.36 d) (see Supplementary Material Figure S1A).

140 Dissolved oxygen concentration ([O<sub>2</sub>] in mg L<sup>-1</sup>; Figure S1B) at day n in surface water was  
141 calculated from Mortimer (1956) considering a standard pressure of 1 atm:

$$142 [O_2](n) = \exp^{7.7117 - \ln(T(n) + 45.93)} \quad (2)$$



143 where  $T_n$  is the daily temperature (in °C).

144 The water flow within the fish farm, which experiences fluctuations throughout the year, was  
145 simulated based on actual water flow data obtained from a river in Brittany. Data from 2018  
146 to 2022, specifically from the Aulne River in Brittany, were collected from the reference  
147 HydroPortail database version 3.1.4.3 (HydroPortail, 2023). Two constraints were considered  
148 when calculating the water flows: the inflow into the fish farm could not exceed  $1.5 \text{ m}^3 \text{ s}^{-1}$ ,  
149 and a maximum of 90% of the total river flow could be diverted to the fish farm. To predict  
150 the daily water inflows into the fish farm, a Generalized Additive Model (GAM) was then  
151 employed considering the different constraints (Figure S1C).

### 152 2.1.2. Growth

153 The fish model described the fish's daily weight and weight gain based on the thermal growth  
154 coefficient (TGC). Considering that the relationship between growth rate and water  
155 temperature is non-linear, the TGC formula was corrected for the concave relationship  
156 between growth rate and temperature, using a corrected temperature K (Mallet et al., 1999) as  
157 suggested by Besson et al. (2016):

$$158 \text{ TGC} = \frac{W_f^b - W_i^b}{\sum_{i=1}^d K_i} \times 1000 \quad (3)$$

159 where  $W_f$  represents the final weight at harvest (3000 g),  $W_i$  denotes the initial weight at  
160 stocking (10 g),  $d$  is the rearing time in days, and  $b$  is a weight coefficient set at  $1/3$  for the  
161 overall growing period even if this parameter can vary according to growth (Dumas et al.,  
162 2007).

163 The TGC values were adjusted to 1.80 and 1.45 ( $\text{g}^{1/3} \text{ }^\circ\text{C}^{-1} \text{ d}^{-1}$ ) in the conventional and organic  
164 production scenarios, respectively. We simulated a 24-month production cycle in  
165 conventional production and a 30-month production cycle in organic production (Figure 3).  
166 This rearing time difference corresponds to the expected growth differentials between triploid  
167 monosex trout, primarily used in conventional production, and male and female diploid trout

168 (Aqualande Origins, 2019) used in organic production according to regulatory requirements  
169 (MAAP, 2010).

170 In the conventional production scenario, the storage dates were kept constant throughout the  
171 three years and set at d 30 for the first batch, followed by intervals of 100 days (i.e., d 130 for  
172 batch 2 and d 230 for batch 3) over a year. In the organic production scenario, the frequency  
173 of batch entries was set at 50 days (i.e., d 80 for batch 2 and d 130 for batch 3) to maintain the  
174 same rotation of harvests and stocking (3 entries and 3 harvests per year). This adjustment  
175 was necessary to accommodate the longer rearing duration (i.e. 30 vs 24 months) while  
176 ensuring consistent batch rotation in the organic production system.

177 The corrected temperature (K) at a given day n was calculated as follows:

$$178 \quad K_n = \frac{T_{opt}(T_n - T_{min})(T_n - T_{max})}{(T_n - T_{min})(T_n - T_{max}) - (T_n - T_{opt})^2} \quad (4)$$

179 where  $T_{min} \leq K \leq T_{max}$  and  $K = 0$  for other values. Here,  $T_{min}$  and  $T_{max}$  represent the minimum  
180 and maximum temperatures, respectively, below and above which growth does not occur.  $T_{opt}$   
181 refers to the optimal temperature for growth. Based on extrapolations from Bear et al. (2007),  
182 the values for rainbow trout were set at 3 °C for  $T_{min}$  ( $K = 0$ ), 13 °C for  $T_{opt}$  ( $K = 13$ ), and 24  
183 °C for  $T_{max}$  ( $K = 0$ ). Consequently,  $T_n$  must fall between 3 °C and 24 °C for a positive growth  
184 rate. The daily weight (W) and daily weight gain (DWG;  $g \cdot d^{-1}$ ) can be calculated as follows on  
185 day n:

$$186 \quad W(n) = \left[ W_i^b + \left( \frac{TGC}{1000} \times \sum_{i=1}^n K_i \right) \right]^{\frac{1}{b}} \quad (5)$$

$$187 \quad DWG(n) = W(n) - W(n - 1) \quad (6)$$

188 Growth curves under the two production scenarios are presented in Figure 3A.

### 189 2.1.3. Mortality

190 This study applied a mortality rate of 15% throughout the production cycle, from 10 to 3000  
191 g. It was assumed that the probability of daily mortality was not linear across the rearing  
192 period and is higher for younger individuals (Gåsnes et al., 2021). A Weibull function was

193 considered to model survival, as it is commonly used for survival analysis (Carroll, 2003). So,  
194 the hazard function  $h$ , which defines the death rate at a given day ( $n$ ) conditional on survival  
195 until time  $n$  or later, can be calculated as follows:

$$196 \quad h(n) = \frac{f(n)}{1-F(n)} \quad (7)$$

$$197 \quad \text{considering the Weibull density function } f(n) = 1 - \exp^{-\left(\frac{n}{\lambda}\right)^s} \quad (8)$$

$$198 \quad \text{and the Weibull distribution function } F(n) = \frac{k}{\lambda} \left(\frac{n}{\lambda}\right)^{s-1} \exp^{-\left(\frac{n}{\lambda}\right)^s} \quad (9)$$

199 The Weibull distribution is highly flexible and can model many different patterns depending  
200 on the values of the shape ( $s$ ) and the scale ( $\lambda$ ) parameters. In our model, we kept the  
201 parameter  $s$  fixed at 0.5 while the scale parameter  $\lambda$  was optimized for each fish batch,  
202 ensuring a final mortality rate of 15% across the entire rearing duration. This parameterization  
203 is based on on-farm surveys in Brittany and can be re-adjusted according to the survival  
204 performance of a specific farm.

#### 205 *2.1.4. Biomass*

206 The biomass (BM) at a given day for each batch was determined as follows:

$$207 \quad BM(n) = W(n) \times SN(n) \quad (10)$$

208 Where  $W$  is the individual body weight at a given day  $n$ , and  $SN$  is the number of surviving  
209 fish on this day. In the same way, the dead biomass at day  $n$  was calculated by replacing  $SN$   
210 with the number of dead fish at this day in the equation (10).

211 The total production (in tonnes) was then calculated as the difference between the biomass at  
212 the harvest and the initial biomass at stocking. Harvest occurred at a constant weight of 3000  
213 g in the two production scenarios.

#### 214 *2.1.5. Raceways occupation*

215 In our model, the occupancy of the raceways was determined by the densities achieved, which  
216 necessitates regular sorting of the fish during rearing. Initially, we assumed that each batch

217 was stocked in a 100-m<sup>3</sup> raceway. As the fish grow, they were periodically redistributed into  
218 and then 4 raceways of 100 m<sup>3</sup> before ultimately occupying 4 then 8 raceways of 250 m<sup>3</sup>. The  
219 maximum density constraints varied depending on the production scenario. In the  
220 conventional production scenario, the density limits applied were 50 kg m<sup>-3</sup> when  $W \leq 50$  g,  
221 70 kg m<sup>-3</sup> when  $50 \text{ g} < W \leq 1000 \text{ g}$  and then 90 kg m<sup>-3</sup> when  $W > 1000 \text{ g}$  (CIPA, 2023). For  
222 the organic production scenario, the density limits were as follows according to CIPA (2023):  
223 25 kg m<sup>-3</sup> when  $W \leq 15 \text{ g}$ , 30 kg m<sup>-3</sup> when  $15 \text{ g} < W \leq 30 \text{ g}$  and then 35 kg m<sup>-3</sup> when  $W > 30$   
224 g. Each rearing structure's occupancy rate was calculated as the sum of the surface used per  
225 day divided by the total surface available over a year (expressed as m<sup>2</sup>y).

#### 226 2.1.6. Feeds

227 The feed conversion ratio at a given day (FCR) was modelled by a third-order polynomial  
228 model based on fish body weight ( $W$ ) using an equation extrapolated from Bureau and Hua  
229 (2008):

$$230 \text{FCR}(n) = \alpha \times [(0.051 \times W(n)^3) - (0.261 \times W(n)^2) + (0.688 \times W(n)) + 0.65] \quad (11)$$

231 where  $\alpha$  is a scaling factor to obtain a realized FCR of 1.30 kg kg<sup>-1</sup> over the production cycle  
232 for each batch in the two production scenarios, assuming that the conventional and organic  
233 fish lines have the same feed efficiency (Figure 4). Daily feed intake (DFI, kg d<sup>-1</sup>) is  
234 calculated back from FCR and DWG by:

$$235 \text{DFI}(n) = \text{DWG}(n) \times \text{FCR}(n) \quad (12)$$

236 Our model considered the dynamic nature of fish feed composition, particularly in terms of  
237 protein and lipid content, throughout the rearing period. As a result, four different types of  
238 feed were incorporated based on the weight. Feed 1 was used for fish up to 50 g, feed 2 for  
239 fish up to 500 g, feed 3 for fish up to 1500 g, and feed 4 was used until the harvest weight was  
240 reached ( $W_f$ ). This approach ensures that the nutritional needs of the fish are adequately met,  
241 at each stage of their growth and development. Conventional or certified organic feeds were  
242 used depending on the production scenario.

243 2.1.7. Nutrient release

244 The concentration of nutrients (N and P) and chemical oxygen demand (COD) in effluent  
245 water was determined using a mass-balance approach (Aubin et al., 2011). To model  
246 excretion, the first step involved calculating the total nutrient amount provided by the feeds  
247 ( $N_{\text{feed}}$ ), taking into account two fractions: the portion consumed ( $N_{\text{eaten}}$ ) and the portion  
248 wasted ( $N_{\text{waste}}$ ) on day  $n$ , along with the nutrient fixation by the fish ( $N_{\text{fish}}$ ). It was assumed  
249 that 1% of the distributed feeds remained uneaten (Boujard et al., 1995). The proximate  
250 composition of the different feeds can be found in Table 2.

$$251 N_{\text{feeds}}(n) = N_{\text{content}} \times \text{DFI}(n) \quad (13)$$

$$252 N_{\text{waste}}(n) = N_{\text{feeds}}(n) \times 0.01 \quad (14)$$

$$253 N_{\text{eaten}}(n) = N_{\text{feeds}}(n) - N_{\text{waste}}(n) \quad (15)$$

$$254 N_{\text{fish}}(n) = N_{\text{fish}_{\text{body}}} \times \text{DWG}(n) \times \text{SN}(n) \quad (16)$$

255 where  $N_{\text{fish}_{\text{body}}}$  is the nutrient composition of the fish (in  $\text{kg kg}^{-1}$ ) set at 0.03 for N (Oz and  
256 Dikel, 2015) and 0.004 for P (Kause et al., 2022).

257 The total nutrient excretion ( $N_{\text{excretion}}$ ) was given by:

$$258 N_{\text{excretion}}(n) = N_{\text{eaten}}(n) - N_{\text{fish}}(n) \quad (17)$$

259 Calculation of the suspended ( $N_{\text{suspended}}$ ) and dissolved ( $N_{\text{dissolved}}$ ) was given by:

$$260 N_{\text{suspended}}(n) = N_{\text{eaten}}(n) \times (1 - \text{Dig}_N) \quad (18)$$

$$261 N_{\text{dissolved}}(n) = N_{\text{excretion}}(n) - N_{\text{suspended}}(n) \quad (19)$$

262 where  $\text{Dig}_N$  is the digestibility coefficient set at 94% for proteins and 61% for P (Dalsgaard  
263 and Pedersen, 2011).

264 The final amount of N release was then calculated considering that the sedimentation area  
265 used as water treatment can remove 20% of suspended N (Stewart et al., 2006):

$$266 N_{\text{release}}(n) = 0.8 \times N_{\text{suspended}}(n) + N_{\text{dissolved}}(n) \quad (20)$$

267 COD at a given day n was calculated using feed quantity eaten (DFI) at day n, the proximate  
 268 protein, lipids and carbohydrates contents of the feeds ( $P_{feeds}$ ,  $L_{feeds}$  and  $C_{feeds}$ ) and their  
 269 respective digestibility (Dig) (i.e., 94% for proteins, 91% for lipids and 67% of carbohydrates;  
 270 Dalsgaard and Pedersen, 2011):

$$271 \text{ COD}(n) = [(P_{feeds}(n) \times (1 - \text{Dig}_P) \times 1.66) + (L_{feeds}(n) \times (1 - \text{Dig}_L) \times 2.78) + \\ 272 C_{feeds}(n) \times 1 - \text{Dig}_C \times 1.19 \times \text{DFI}(n)] \quad (21)$$

273 where the coefficients applied for protein, lipids, and carbohydrates came from Meriac et al.  
 274 (2014).

### 275 2.1.8. Oxygen

276 In both production scenarios, the primary constraint for oxygen management was maintaining  
 277 a saturation level of 80% at the outlet of the raceways. However, the approach to  $O_2$   
 278 supplementation differed between the two production scenarios. In conventional production,  
 279 liquid oxygen was used for  $O_2$  supplementation, whereas in organic production, using aerators  
 280 was the only permissible method (MAAP, 2010). In our model, the amount of oxygen added  
 281 was determined based on the difference between the supply of oxygen through the water inlet  
 282 ( $O_{2inlet}$ ), which could come directly from the river or the upstream raceways (Figure 1), and  
 283 the  $O_2$  consumption by the fish ( $O_{2cons}$ ). These two parameters were calculated using the  
 284 following equations:

$$285 O_{2inlet}(n) = O_{2conc}(n) \times \text{Water}_{flow}(n) \quad (22)$$

286 where  $O_{2conc}$  is the  $O_2$  concentration from the water inlet either coming from the river - in this  
 287 case  $O_{2conc} = [O_2](n)$  (see Section 2.1.1) or from the upstream raceway - in this case  
 288  $O_{2conc} = [O_2](n) - O_{2cons}(n)$  of the upstream raceway. Water flow in a given raceway is  
 289 calculated as follows:

$$\text{Water}_{flow}(n) = \alpha \times \frac{\text{Water}_{total}(n)}{RN}$$

290 where  $Water_{total}$  is the water flow for the whole fish farm, RN is the number of raceways, and  
 291  $\alpha$  is a size coefficient (i.e. 0.29 for 100-m<sup>3</sup> raceway and 0.71 for 250-m<sup>3</sup> raceway). Then, O<sub>2</sub>  
 292 consumption is given by:

$$\begin{aligned} 293 \quad O_{2_{cons}}(n) = & [(DFI(n) \times P_{feeds}(n) \times Dig_p) - (DFI(n) \times P_{fish}(n))] \times \frac{E_p}{Q_{oxP}} + [(DFI(n) \times \\ 294 \quad & L_{feedsn} \times Dig_L - DFI_n \times L_{fishn} \times EL_{QoxL} + DFI_n \times C_{feedsn} \times Dig_C - DFI_n \times C_{fishn} \times EC_{QoxC} \\ 295 & \hspace{28em} (23) \end{aligned}$$

296 where  $Q_{oxP}$ ,  $Q_{oxL}$  and  $Q_{oxC}$  are the oxy-caloric coefficients of proteins (13.4 MJ kg O<sub>2</sub><sup>-1</sup>), lipids  
 297 (13.7 MJ kg O<sub>2</sub><sup>-1</sup>) and carbohydrates (14.8 MJ kg O<sub>2</sub><sup>-1</sup>) (Brafield and Solomon, 1972; Elliott  
 298 and Davison, 1975) and  $E_p$ ,  $E_L$  and  $E_C$  are the energy contents of proteins (23.6 MJ kg<sup>-1</sup>), lipids  
 299 (39.5 MJ kg<sup>-1</sup>) and carbohydrates (17.2 MJ kg<sup>-1</sup>) (Brafield and Llewellyn, 1982).

300 No oxygenation or aeration was required if the difference between O<sub>2inlet</sub> and O<sub>2cons</sub> was  
 301 higher than the 80% saturation O<sub>2</sub> concentration ( $O_{2\_80\%} = 0.8 [O_2]$ ). Conversely, a result  
 302 lower than O<sub>2\_80%</sub> indicated the need for O<sub>2</sub> supplementation (O<sub>2sup</sub>) either through the  
 303 addition of liquid O<sub>2</sub> or by aeration:

$$304 \quad O_{2_{sup}}(n) = 0 \text{ when } O_{2_{inlet}}(n) - O_{2_{cons}}(n) > O_{2_{_80\%}}(n) \quad (24)$$

$$305 \quad O_{2_{sup}}(n) = |O_{2_{inlet}}(n) - O_{2_{cons}}(n)| \text{ when } O_{2_{inlet}}(n) - O_{2_{cons}}(n) < O_{2_{_80\%}}(n) \quad (25)$$

### 306 2.1.9. Energy

307 The farm's electricity consumption was modelled considering water filtration, oxygenation  
 308 and recirculation processes. A drum filter (1 kWh) and a recirculation pump (20 kWh)  
 309 operated when the water flow was at its lowest, typically between May and September. They  
 310 aimed to ensure adequate water recirculation under conventional and organic production  
 311 scenarios during this period. Electricity consumption by the filter ( $E_{filter}$ ) and the recirculation  
 312 pump ( $E_{pump}$ ) at a given day  $n$  has been calculated as follows:

$$313 \quad E_{filter}(n) = 1 \times 24 \text{ for May} < n < \text{September and } E_{filter}(n) = 0 \text{ for other dates} \quad (26)$$

$$314 \quad E_{pump}(n) = 20 \times 24 \text{ for May} < n < \text{September and } E_{pump}(n) = 0 \text{ for other dates} \quad (27)$$

315 One key distinction between estimating electricity consumption for conventional and organic  
 316 production lies in the method employed for water oxygenation. In conventional production,  
 317 liquid oxygen was added using an oxygen cone and two pumps with a power consumption of  
 318 20 kWh each. In this case, the electrical consumption at a given day  $n$  was calculated as  
 319 follows:

$$320 \quad E_{oxygen}(n) = 2 \times 20 \times 24 \text{ for } O_{2sup}(n) > 0 \quad (28)$$

$$321 \quad E_{oxygen}(n) = 0 \text{ for } O_{2sup}(n) = 0 \quad (29)$$

322 In the organic production scenario, aerators replaced liquid oxygen and were limited to  
 323 specific uses (such as compensating for increasing temperatures) according to European  
 324 regulations (MAAP, 2010). These aerators enable the addition of 1.5 kg of oxygen per  
 325 kilowatt-hour (kWh) of electricity consumed (Ahmad and Boyd, 1988; Brown et al., 2014).  
 326 Consequently, the calculation for electrical consumption associated with the aerators has been  
 327 calculated as follows:

$$328 \quad E_{oxygen}(n) = \frac{O_{2sup}(n)}{1.5} \quad (30)$$

329 The total electricity consumption ( $E_{total}$ ) was determined by summing the electricity usage for  
 330 water filtration, oxygenation, and recirculation:

$$331 \quad E_{total}(n) = E_{filter}(n) + E_{pump}(n) + [E_{oxygen}(n) \text{ or } E_{aeration}(n)] \quad (31)$$

## 332 2.2. Life Cycle Assessment (LCA)

### 333 2.2.1. Goal and scope

334 An attributional LCA was conducted according to the general requirements of the  
 335 methodology proposed by ILCD standards (JRC, 2010). The methodology was adapted to the  
 336 characteristics of fish farming. The goal and scope of this study was the environmental  
 337 assessment of trout farming in a hypothetical farm producing large rainbow trout following  
 338 either (1) conventional or (2) organic practices in the same infrastructures. The system was  
 339 defined from cradle-to-farm-gate and included five distinct sub-systems (Figure 5): (1)



340 production of purchased feed, including cultivation of ingredients, processing, and  
341 transportation; (2) production of energy expended at farm level (electricity); (3) production of  
342 farming facilities and equipment used; (4) chemicals, including liquid oxygen, veterinary and  
343 disinfection products, and their transportation (5) farm operations, including nutrient  
344 emissions from the biological transformation of feed after on-site treatment of wastewater  
345 (see details in Section 2.2.2). The functional unit in which environmental impacts were  
346 expressed was one tonne of trout produced at the farm level based on one year of routine  
347 production. We also expressed the environmental impacts using a surface-based functional  
348 unit ( $m^2y$ ) as recommended by Van der Werf et al. (2020). Here, we considered only the  
349 surface directly involved in the fish production.

#### 350 2.2.2. Life cycle inventory

351 The life cycle inventory (LCI), presented in Table 3, was conducted by running our farm  
352 model with the specifications for both conventional and organic production scenarios. All the  
353 inputs and outputs were calculated using all the results generated by each batch of fish over  
354 one year of routine production as described in the farm model. The Agribalyse version 3.0  
355 (Koch and Salou, 2022) and Ecoinvent version 3.8 (Wernet et al., 2016) databases were used  
356 to obtain the necessary data for the assessment. Both databases are grounded on the  
357 recommendations in international standards (Wolf et al., 2012).

358 (1) *Production of purchased feed* - Crop-derived ingredients used in fish feed mainly  
359 originated from Brazil and France (e.g. soybean meal from Brazil and wheat bran from  
360 France). Fish-derived ingredients originated from the Peruvian and Norwegian fish milling  
361 industries (e.g., fish meal from Peru and fish meal from fish trimming from Norway). The  
362 feed manufacturer provided the exact composition of the different feeds in ingredients and  
363 nutritional values (Le Gouessant, personal communication). The transport of feed ingredients  
364 to feed manufacturers in France was by trans-oceanic ship and by lorry (>32 t), whereas the  
365 transport of feed from France to the fish farm in Brittany was by lorry (>32 t). Road distances

366 were calculated using Google Maps, and ocean distances were assessed using shiptraffic.net.  
367 Other data required to compute the environmental impact of feed ingredients were based on  
368 the literature (Boissy et al., 2011; Pelletier et al., 2009).

369 (2) *Production of energy expended on the farm* - The electricity used by the farm was coming  
370 from the French energy mix in the Ecoinvent database. Annual on-site consumption from  
371 other energy sources (diesel and gas) was considered negligible.

372 (3) *Production of farming facilities and equipment used* - We considered the construction of  
373 two different buildings with a life span of 30 years. Nevertheless, the life span of each rearing  
374 structure has been adjusted in the LCA inventory according to the rearing structures'  
375 occupancy (Table 4) calculated as described in Section 2.1.5, assuming that the actual life  
376 span of the rearing span is related to their occupancy level. The equipment production (i.e.  
377 pump, tanks) was calculated using data from INRAE.

378 (4) *Chemicals* - This sub-system included veterinary and disinfection products. While these  
379 products vary little between conventional and organic production, the main difference is the  
380 inclusion of the liquid oxygen used only in conventional production in this sub-system. Here,  
381 we considered the production of liquid oxygen from the cryogenic air separation process.

382 (5) *Farm operations* - The farm operation sub-system included using facilities and equipment  
383 and emissions of pollutants from the biological transformation of the feed distributed to the  
384 fish. The amount of nitrogen (N), phosphorus (P) and chemical oxygen demand (COD) of the  
385 dissolved organic matter excreted by the fish in effluent water were calculated through mass  
386 balance (Papatryphon et al., 2005) considering the on-site treatment capacity of the sludge  
387 settling pond. The sludge produced by the farm was used for neighbourhood agricultural  
388 purposes and was not included in the analysis.

389 Gaps in the inventory were filled based on the assumptions reported in Table 4.

### 390 2.2.3. *Life cycle impact assessment*

391 The impact assessment was carried out using ReCiPe 2016 Midpoint (H) version 1.07  
392 (Huijbregts et al., 2017), a methodology based on the Eco-indicator and CML approaches.  
393 According to the European Commission/JRC (2010), ReCiPe is the most up-to-date and  
394 standardized indicator approach for life cycle impact assessment.

395 Table 5 provides a breakdown of the nine selected impact categories from ReCiPe, namely  
396 climate change (GWP), terrestrial acidification (TAP), freshwater eutrophication (FEP),  
397 marine eutrophication (MEP), terrestrial ecotoxicity (TETP), freshwater ecotoxicity (FETP),  
398 land use (LU), water dependence (WD) and the Cumulative Energy Demand method (CED;  
399 Frischknecht et al., 2007). These impact categories have been identified among the most  
400 suitable indicators of aquaculture impacts (Bohnes et al., 2019). The CML baseline (Guinée,  
401 2002) was used as an alternative to the ReCiPe approach to enable comparison with previous  
402 studies on trout production systems. The environmental impacts were calculated using  
403 Simapro version 8.0 software (PRé Consultants, 2014).

### 404 2.2.4. *Sensitivity analysis*

405 Considering that feed use was the major contributor to environmental impacts, a sensitivity  
406 analysis was conducted on FCR. In this study, we ran the model for both conventional and  
407 organic productions to gauge the changes in the different LCA impact categories when FCR  
408 varied from 1.0 to 1.6 in steps of 0.1.

## 409 **3. Results**

### 410 3.1. Impact assessment under the two production scenarios

411 Figure 6 presents the level of the environmental impacts and the contribution of the system  
412 components to the impacts in conventional and organic productions of rainbow trout. The  
413 impacts are calculated using the ReCiPe method using two different functional units: per  
414 tonne of trout (product-based) and per m<sup>2</sup>y (surface-based). Among the nine impact categories  
415 analyzed, the conventional production system exhibits higher impacts for all categories,

416 except for FEP and WD, when the results are expressed per tonne of trout. For instance, in the  
417 conventional production system, one tonne of trout emits 14 kg P eq. and depends on 128,000  
418 m<sup>3</sup> of water, while an equivalent quantity of organic trout emits 19 kg P eq. and depends on  
419 185,000 m<sup>3</sup> of water. When the results are expressed per m<sup>2</sup>y, organic production shows lower  
420 environmental impacts for all categories, including FEP and WD (Figure 6). Regardless of the  
421 functional units chosen (product-based or surface-based), other impact categories also follow  
422 similar trends, with the surface-based functional unit leading to a more significant gap  
423 between the two production scenarios.

424 The highest environmental gains observed in the organic system compared to the  
425 conventional production were for FETP (35% less in the organic system): the production of  
426 one tonne of trout induced 50 kg 1,4-DCB/tonne in the conventional system, but the FETP  
427 value of the organic system was noticeably lower with 33 kg 1,4-DCB/tonne. The energy  
428 requirements (CED) for producing one tonne of trout were also noticeably different between  
429 the two production systems (30% less in the organic system), with values of 68 and 53  
430 GJ/tonne in conventional and organic production, respectively. Terrestrial acidification  
431 potential (TAP) was 28% less in the organic system, with 15 and 12 kg SO<sub>2</sub> eq./tonne in  
432 conventional and organic production, respectively. Differences between conventional and  
433 organic productions were less pronounced for the other impact categories. GWP showed a  
434 reduction of 12% in the organic system with, per kg of fish produced: 2602 kg CO<sub>2</sub> eq./tonne  
435 were estimated in the conventional system vs. 2319 kg CO<sub>2</sub> eq./tonne for the organic system  
436 (Figure 6). The environmental gains through organic production were equal for LU and  
437 TETP; both impact categories showed a reduction of 11% in the organic system, while MEP  
438 was only diminished by 7% (Figure 6).

### 439 3.2. Estimates from literature

440 The results in Figure 6 cannot be compared with those in previous LCA publications, mainly  
441 obtained with the CML baseline method. Thus, the environmental impacts of the production

442 of one tonne of trout were also assessed using the CML baseline methodology. Therefore, we  
443 conducted a literature review on existing LCA in rainbow trout farming and compared our  
444 findings with those reported in prior studies. As there was no assessment of environmental  
445 impacts for organic rainbow trout farming, our comparison was centered on conventional  
446 production using tonne of trout as a functional unit (Table 6). Overall, our results are  
447 consistent with those found in previous studies. For clarity, the values presented in the  
448 subsequent paragraphs are given for the ReCiPe method and per tonne of trout at market size,  
449 while the results expressed per  $m^2y$  can be found in Figure 6.

### 450 3.3. Contribution of the rearing system components

451 As illustrated in Figure 6, the contributions of the rearing system components (i.e., chemicals,  
452 dead biomass, energy, equipment, feeds, fry, and farm functioning) varied according to the  
453 impact category and the production system. Overall, the ranking of the different contributors  
454 among the seven impact categories remained relatively constant between conventional and  
455 organic productions except for chemicals, driven mainly by liquid oxygen, accounting for a  
456 non-negligible part of conventional production's environmental impacts, but not in organic  
457 production (Figure 6). Results presented in Figure 6 show that, for MEP, FEP, and WD, farm  
458 operations contributed the most to the impacts (81-84%, 90-93% and 93-98%, respectively),  
459 and the second largest contributors are either feeds for MEP and FEP (15-18% and 5-6%,  
460 respectively) or energy for WD (1-5%). For five out of nine impact categories (i.e., LU, TAP,  
461 GWP, CED and TETP), exogenous feeds were the main contributors (96-97%, 79-90%, 66-  
462 85%, 50-79% and 53%, respectively), whatever the production systems. Equipment and  
463 infrastructures played a significant role in the FETP and TETP in the two production systems  
464 (20-53%), while their role was relatively negligible in the other impact categories. As  
465 mentioned earlier, the most remarkable difference in the contributions to environmental  
466 impacts of the two production systems concerned the role of chemicals. Indeed, chemicals  
467 included antibiotics, other veterinary products and disinfectants, whose quantities remained

468 relatively constant between conventional and organic production (Table 3). On the other hand,  
469 the significant difference is related to the use of liquid oxygen only in conventional  
470 production, which was included in chemicals (Table 3). Thus, while chemicals represented  
471 only <2% of CED, FETP, GWP, TAP, TETP and CED in organic production, they  
472 represented between 13% and 44% of the corresponding impacts in conventional production  
473 (Figure 6).

#### 474 3.4. Sensitivity analysis

475 The sensitivity analysis results indicated a linear relationship between FCR and the  
476 environmental impacts of rainbow trout farming for the nine impact categories considered in  
477 this study (Figure 7). Across most of the impact categories considered, a reduction of 0.1 kg  
478 kg<sup>-1</sup> in FCR led to a decrease of the environmental impacts by 3 to 12%. Notably, the most  
479 substantial differences were observed for FEP. However, improving feed efficiency had a  
480 negligible effect on WD, mainly linked to the water volume derived from the river and  
481 passing through the rearing structures, reducing it by less than 1%.

## 482 4. Discussion

### 483 4.1. Benefits of the modeling approach for environmental impact comparison

484 Despite the rapid growth of organic agriculture production, organic finfish aquaculture  
485 remains relatively new and is still in its early stages (Mente et al., 2011). In Europe, the  
486 development of this sector has been hindered by technical challenges, such as the limited  
487 availability of organic feed and fry. Additionally, establishing effective communication  
488 strategies with clients proves difficult due to competition from other certification schemes,  
489 such as the Aquaculture Stewardship Council (ASC) or the Marine Stewardship Council  
490 (MSC) (European Commission, 2022). Furthermore, some organic farming systems  
491 experience lower yields, and previous research has suggested that using organic feed  
492 ingredients may lead to reduced farm eco-efficiency and increased environmental concerns  
493 (Pelletier and Tyedmers, 2007). However, peer-reviewed studies comparing the

494 environmental impacts of conventional and organic aquaculture production systems  
495 (Biermann and Geist, 2019; Jonell and Henriksson, 2015) are scarce.

496 The current studies in this field have predominantly followed a field-based approach, wherein  
497 data was directly collected from conventional and organic farms to establish the LCI.  
498 However, employing such an approach may introduce bias, particularly regarding the  
499 distinction between differences arising from the specific production systems (conventional or  
500 organic) and variations inherent to individual farming practices, which can significantly  
501 impact the interpretation of the LCA results (Chen and Corson, 2014). It is especially true in a  
502 context where the representativeness of farming practices is sometimes questioned in the LCA  
503 studies carried out in animal production (Meier et al., 2015). In this study, we employed a  
504 modelling approach associated with LCA to compare the environmental impacts of  
505 conventional and organic rainbow trout production within a hypothetical farm. The farm's  
506 infrastructures and available surface area for production were kept constant in the two  
507 scenarios to determine the differences in environmental impacts between conventional and  
508 organic production in the same infrastructures.

#### 509 4.2. LCA literature on conventional production

510 Before delving into the analysis of environmental impacts between the two studied production  
511 systems (conventional and organic), it is crucial to establish a reference point by comparing  
512 the results obtained in the conventional production scenario with those from existing  
513 literature. This step allows us to compare the results from modelling with those obtained from  
514 actual fish farm data. To achieve this, we have used the ReCiPe method and CML baseline  
515 method (Guinée, 2002). The latter was commonly used in previous LCA studies focusing on  
516 rainbow trout aquaculture, while ReCiPe was only recently used in a rainbow trout  
517 aquaculture context, notably in Italy (Maiolo et al., 2021) and Spain (Sanchez-Matos et al.,  
518 2023). Overall, the literature comparison corroborated our findings when expressing  
519 environmental impact per tonne of trout (Table 6). Indeed, our results showing, for instance,

520 that one tonne of trout emitted 2602 kg CO<sub>2</sub> eq. and 13.4 kg P eq. and required 68 GJ eq. of  
521 energy in a conventional scenario are consistent with the literature (Aubin et al., 2009; Boissy  
522 et al., 2011; Chen et al., 2015; d'Orbcastel et al., 2009; Dekamin et al., 2015; Maiolo et al.,  
523 2021; Papatryphon et al., 2004; Samuel-Fitwi et al., 2013). Nevertheless, the ranges of  
524 reported values can be broad. Despite uncertainties related to varying inventory databases and  
525 CML assessment method versions, another underlying cause of the differences in  
526 environmental impacts among studies is the use of diverse production systems and varying  
527 FCRs to produce the same quantity of trout (Philis et al., 2019; Sanchez-Matos et al., 2023).

#### 528 4.3. Environment impacts under the two production scenarios

529 The choice of functional units in LCA is a crucial point to consider when comparing  
530 production systems, because it influences allocation decisions at the farm gate (Henriksson et  
531 al., 2012). Van der Werf et al. (2020) highlighted the interest in combining product-based and  
532 area-based LCA when comparing conventional and organic production systems. For instance,  
533 although organic animal production generally emits fewer pollutants per unit of land occupied  
534 than conventional agriculture (a surface-based approach), it may have higher impacts per unit  
535 of product (e.g., land occupation, eutrophication and acidification) (Meier et al., 2015). Thus,  
536 while we used one tonne of trout as a first functional unit, we also expressed the  
537 environmental impacts using a surface-based functional unit (m<sup>2</sup>y), an original approach in  
538 LCA aquaculture studies (Bohnes et al., 2019; Pouil et al., 2023).

539 Overall, our study highlights a significantly lower level of environmental impacts of organic  
540 production than conventional production. It is particularly true for TAP, FETP and CED, with  
541 22-35% less impact in the organic scenario. However, when impacts are expressed per tonne  
542 of trout, the WD and the FED are higher in the organic system than in the conventional  
543 system (Figure 6). Nonetheless, it is important to be cautious when comparing the  
544 environmental performance of the two production systems using a product-based functional  
545 unit because the production capacity in the organic system is one-third lower. Specifically,



546 organic trout production is limited by lower rearing densities and reduced inputs, such as the  
547 absence of liquid oxygen (MAAP, 2010), while conventional intensive systems are managed  
548 with high stocking rates and inputs to achieve high productivity (CIPA, 2023). As a result, the  
549 larger production volume somewhat dilutes the environmental impacts of the conventional  
550 production system. This limitation should be considered when comparing organic and  
551 conventional systems using LCA and highlights the need to explore alternative surface-based  
552 functional units to gain a more comprehensive understanding of the comparison (van der Werf  
553 et al., 2020). Using a surface-based functional unit ( $m^2y$ ), we find that the FEP and the WD  
554 become similar between the two production systems and even slightly lower in the organic  
555 system due to the absence of liquid oxygen usage (Figure 6). Our study demonstrates the  
556 benefits of organic trout production in terms of overall environmental impacts. However,  
557 considering the nuances related to production capacity and LCA functional units is crucial to  
558 gaining a well-rounded perspective on the environmental performance of both systems.

559 The significant importance of liquid oxygen usage in conventional production becomes  
560 apparent when conducting a more detailed analysis of the contributions to environmental  
561 impacts between the two production systems. Indeed, we found that chemicals, mostly  
562 composed by liquid oxygen in the conventional scenario, contributed up to 44% of the  
563 environmental impacts, depending on the category, and explained most of the differences we  
564 observed between organic and conventional production. Previous studies have also  
565 underscored the significance of liquid oxygen in the environmental impacts associated with  
566 aquaculture production. For instance, Song et al. (2019) highlighted that liquid oxygen  
567 contributed between 5% and 22% to all LCA impact categories. Consequently, such  
568 production inputs should not be overlooked in LCA conducted for aquaculture production  
569 systems. Likewise, the role of aquafeeds in influencing environmental impacts is  
570 fundamental, regardless of whether it is for organic or conventional production. The  
571 importance of FCR and aquafeeds, in general, has been emphasized by numerous LCA

572 practitioners. Several studies have already concluded that feed production constitutes a  
573 significant source of environmental impact (e.g., Aubin, 2013; Bohnes et al., 2019; Wilfart et  
574 al., 2023). Although organic feed helps reduce environmental impacts in many categories, its  
575 higher proportion of fishmeal and fish oil, which are rich in P (Oliva-Teles et al., 2015), leads  
576 to a greater release of phosphate into the environment in the organic production scenario,  
577 resulting in an increased risk of FEP as shown in Figure S2. Indeed, we find that conventional  
578 feeds are responsible for 0.53 kg P eq. per tonne of feeds, while this value increased to 0.90  
579 kg P eq. for organic feeds. It is worth noting that while feed formulations cannot be entirely  
580 disclosed due to industrial secrecy, efforts have been made to evolve these formulations.  
581 Nonetheless, these results align with the findings of Pelletier and Tyedmers (2007), who  
582 reported considerably lower environmental impacts when feeds contained reduced proportions  
583 of fish ingredients. One solution to improve the environmental performance of aquafeeds  
584 could be to explicitly include environmental performance as a criterion for the inclusion of  
585 ingredients in feed formulation. Thus, using a multiobjective formulation process, Wilfart et  
586 al. (2023) showed the effectiveness of designing an eco-friendly feed fitting the nutritional  
587 requirements of rainbow trout.

#### 588 4.4. Sensitivity of the results to change in FCR

589 Given the paramount importance of feeds in determining the environmental impacts of our  
590 production systems, we investigated the effects of a change in FCR on impact categories  
591 encompassed in LCA. We aimed to shed light on the relationship between FCR and  
592 environmental impacts per tonne of trout in our production systems. Here, we established a  
593 positive linear correlation between FCR and the environmental impacts observed (Figure 7).  
594 Such findings agree with previous LCA studies reporting that all environmental impacts  
595 decrease in similar proportions together with the improvement of FCR (d'Orbcastel et al.,  
596 2009; Jouannais et al., 2023; Papatryphon et al., 2004). Thus, improving feed efficiency  
597 through selective breeding, for instance, improves both the economic and the environmental

598 performances of fish farming (de Verdal et al., 2018; Kause et al., 2022). Indeed, in rainbow  
599 trout farming, the feeds represent more than 40% of the total production costs (Kankainen et  
600 al., 2016). For these reasons, feed efficiency is now one of the main traits targeted by  
601 selection in fish (de Verdal et al., 2018).

602 Our findings align with the conclusions drawn in a meta-analysis conducted by Philis et al.  
603 (2019), revealing a similar positive relationship between FCR and environmental impacts  
604 when comparing the environmental impacts associated with different salmonid production  
605 systems. This observation holds for changes in the FCR within the same production system  
606 and no longer holds across systems (Jouannais et al., 2023). Indeed, while the trend is quite  
607 evident in Recirculating Aquaculture Systems (RAS), it is notably less when considering open  
608 production systems like land-based flow-through systems or open sea cages (Philis et al.,  
609 2019). Such dissimilarity can be attributed to inherent study variations, which become more  
610 pronounced when analyzing flow-through production systems. The RAS, being more  
611 controlled, lend themselves to more straightforward comparability across studies.

612 In contrast, the complexities and diverse factors associated with production in flow-through  
613 systems make it challenging to draw generalizable conclusions. Nevertheless, the emergence  
614 of RAS as an alternative conventional rearing system to flow-through in trout farming has  
615 brought about new challenges, including increased energy consumption, dependence on  
616 equipment like pumps and filters, and potential greenhouse gas emissions and environmental  
617 footprint associated with energy production and waste management (Ahmed and Turchini,  
618 2021; d'Orbcastel et al., 2009). Given the absence of recent comparative LCA available in the  
619 literature (d'Orbcastel et al., 2009; Dekamin et al., 2015; Samuel-Fitwi et al., 2013), it could  
620 be interesting to adapt our model for comparison between flow-through systems and RAS in  
621 trout farming.

622 4.5. Challenges and considerations in scaling up organic trout farming

623 The outcomes of our study provide a foundation for a more comprehensive reflection that  
624 goes beyond the boundaries of the farm. Within the European Union, organic trout production  
625 merely constitutes 2% of the total trout production (EUMOFA, 2022). Although conventional  
626 production is predominant in trout farming, it is interesting to scrutinize the potential  
627 implications associated with a transition toward organic farming. It is crucial to underscore  
628 that the feasibility of scaling up trout production through organic farming is contingent upon a  
629 diverse array of factors. These factors encompass effective institutional policies, market  
630 dynamics, and the accessibility and costs associated with organic inputs (EUMOFA, 2022).  
631 An illustrative example of such considerations is the regulation of stocking density in organic  
632 aquaculture, which inherently constrains productivity (Ahmed et al., 2020). Consequently, the  
633 mandated low intensity of production associated with high production costs, raise significant  
634 concerns regarding organic fish farming development (EUMOFA, 2022). Despite the  
635 environmental benefits of organic aquaculture, it is imperative to acknowledge the necessity  
636 of implementing measures to counterbalance reduced productivity and heightened production  
637 costs. This proactive approach is essential to ensure the economic sustainability and social  
638 viability of the sector, particularly in the context of widespread adoption of organic fish  
639 farming.

## 640 **5. Conclusion**

641 Our study, through a holistic approach, demonstrated the environmental benefits of organic  
642 trout production at the farm level. Thus, we revealed that organic farming significantly  
643 reduced environmental impacts per tonne of trout in seven of the nine selected impact  
644 categories included in LCA. The only exceptions were freshwater eutrophication and water  
645 dependence, where organic production led to higher impacts per tonne of trout. Nonetheless,  
646 we found that, when results are expressed by  $m^2y$ , organic production leads to less impact in  
647 all the categories. Thus, our findings underscore the need for caution when interpreting LCA  
648 comparisons of such production systems, as they can be significantly impacted by

649 methodological choices such as the chosen functional unit (product-based vs. surface-based).  
650 Our analysis reveals that aquafeeds and liquid oxygen usage are key factors contributing to  
651 the environmental impacts of conventional and/or organic trout production systems. By  
652 recognizing and addressing the significance of these inputs, we can take further steps towards  
653 sustainable finfish aquaculture practices.

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908 **Captions to figures**

909

910 Figure 1. Schematic view of the hypothetical rainbow trout farm used to model conventional  
911 and organic production. The equipment specific to the conventional and organic production  
912 systems are annotated with the following symbols: \* and \*\*, respectively. In organic  
913 production, according to the European regulations, the use of aerators is limited to specific  
914 cases (such as to compensate for increasing temperatures; MAAP, 2010).

915 Figure 2. Schematic view of the modelling approach we used. The values for the different  
916 model models and details of the constraints applied for the conventional and organic systems  
917 are detailed in the text.

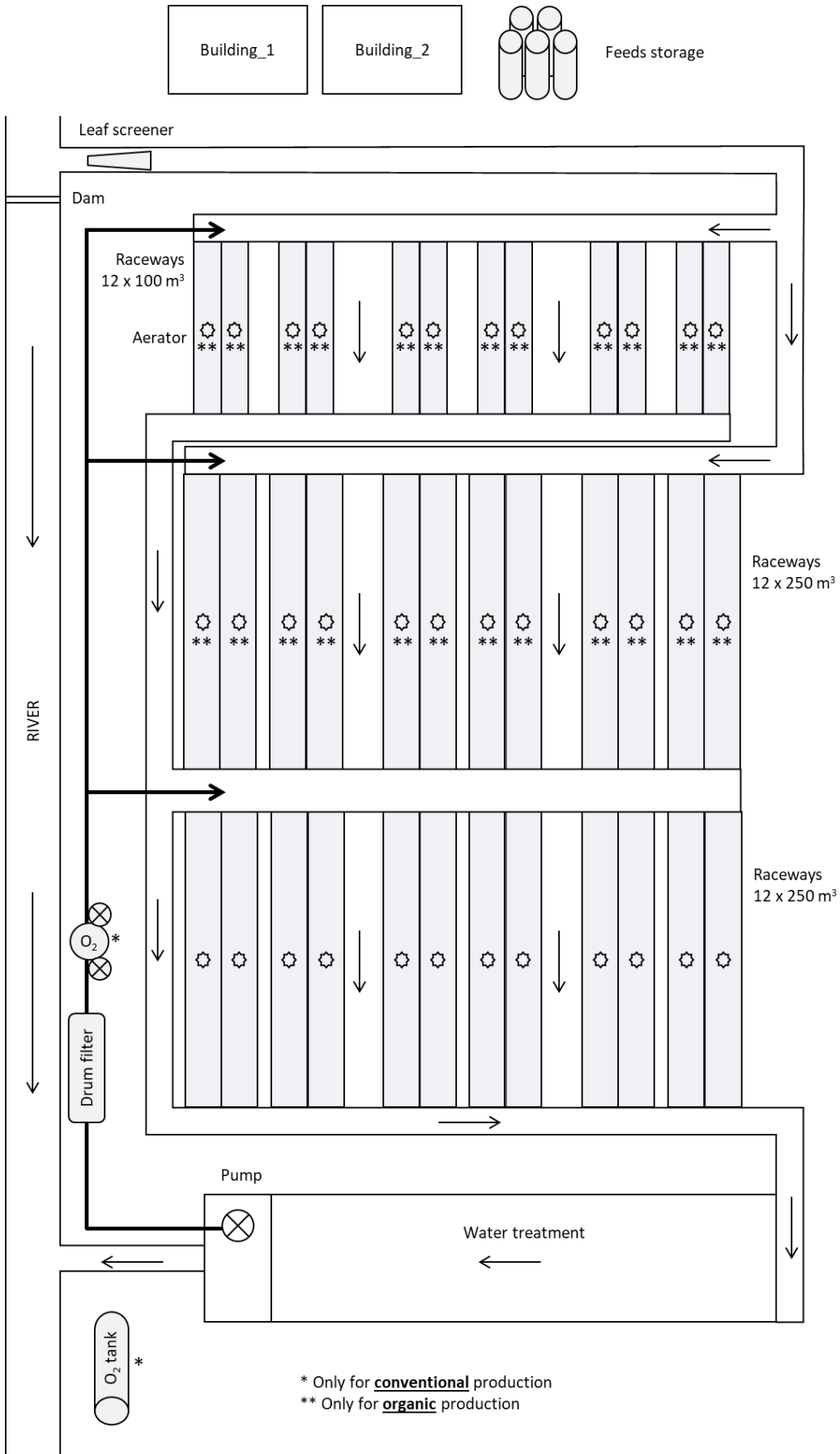
918 Figure 3. Graphical representations of (A) growth performances, from 0.01 to 3 kg and (B)  
919 survival of the three fish batches in conventional (in blue) and organic (in red) production  
920 systems.

921 Figure 4. Estimated FCR of rainbow trout at increasing live weight was extrapolated from  
922 Bureau and Hua (2008).

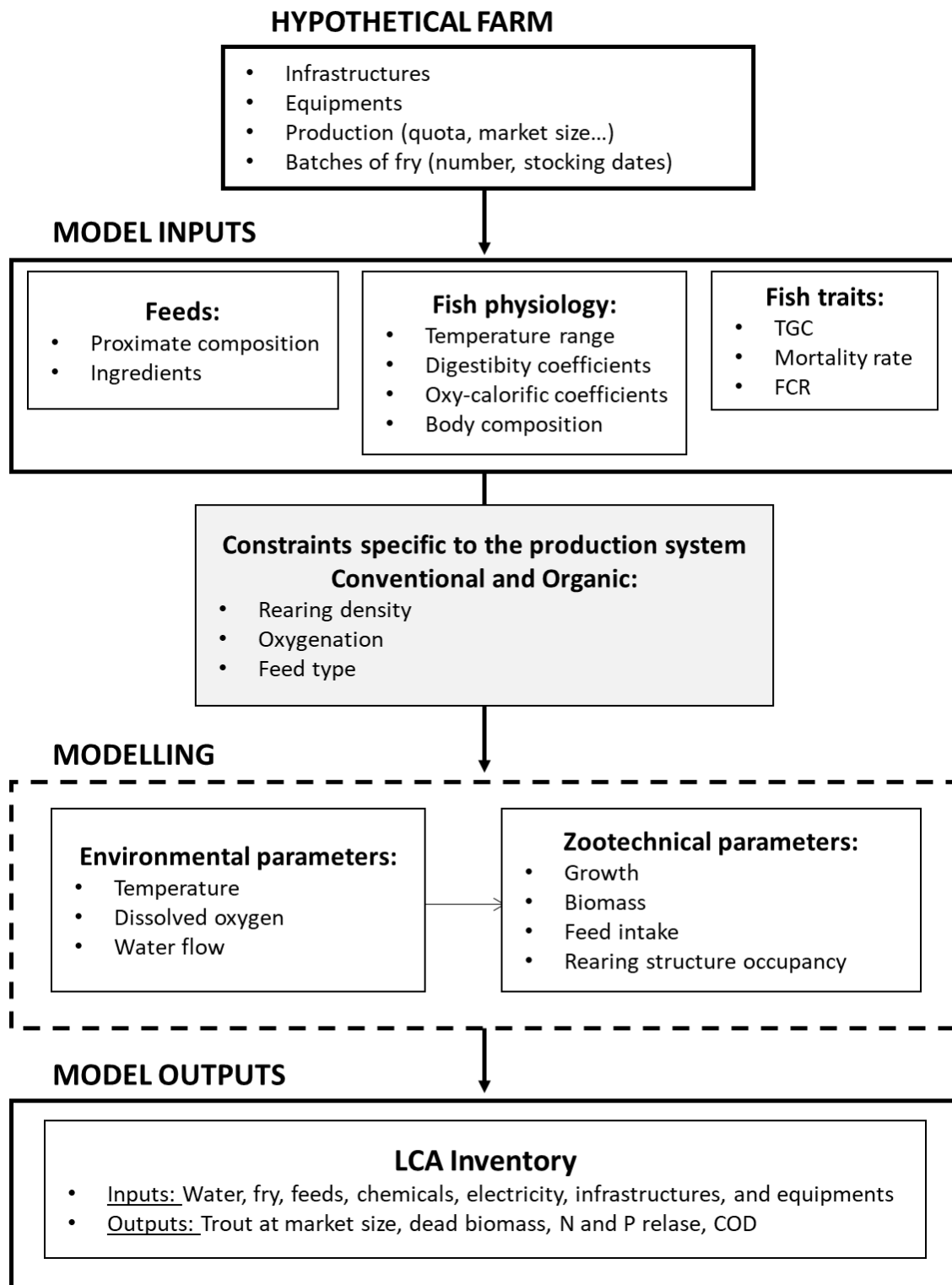
923 Figure 5. System boundaries and flows of rainbow trout *Oncorhynchus mykiss* grow-out  
924 production.

925 Figure 6. Contribution of each input or production step in environmental impacts in  
926 conventional and organic trout production systems. Results are expressed per tonne of trout at  
927 market size (product-based) or per m<sup>2</sup>y (surface-based).

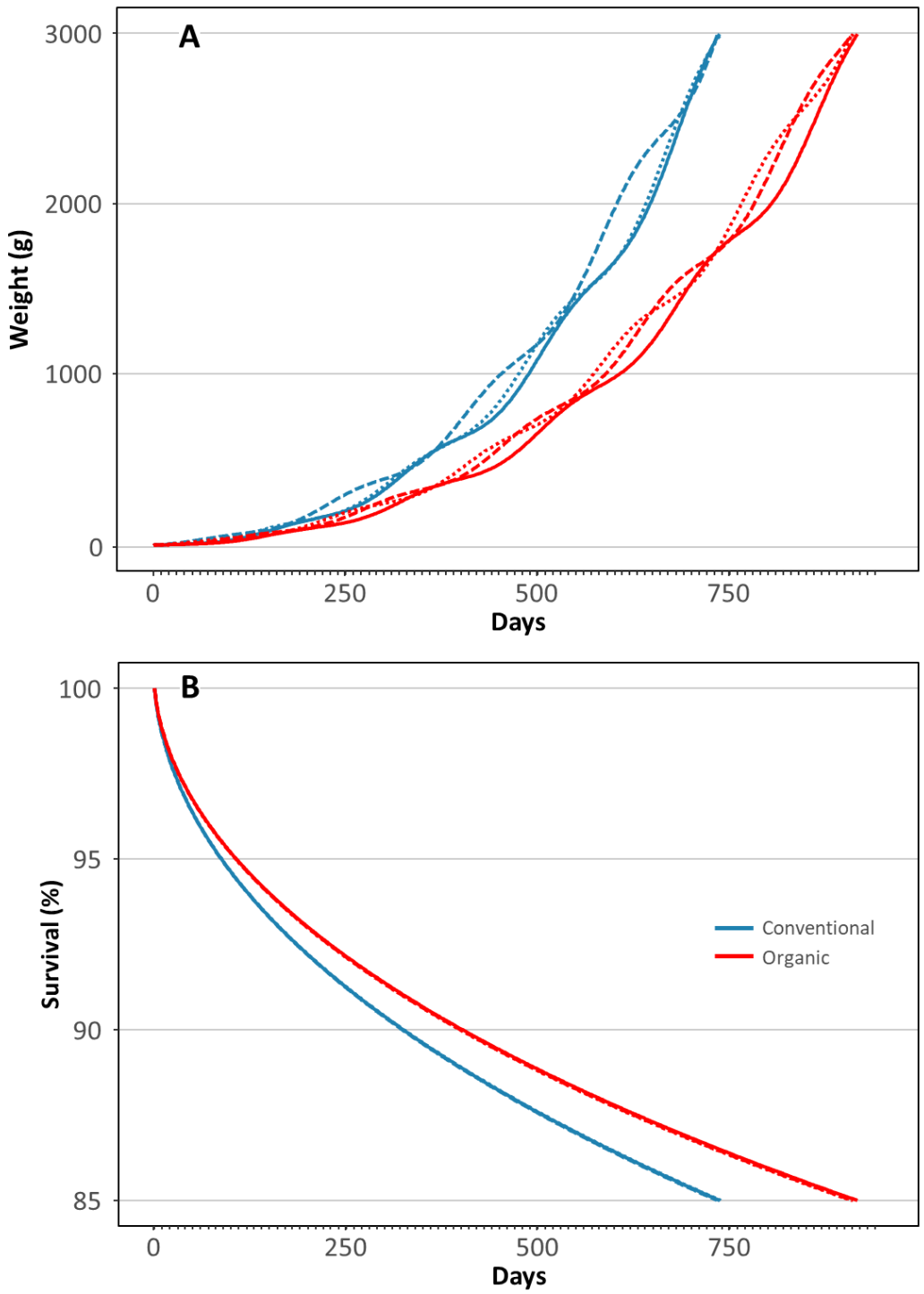
928 Figure 7. Influence of FCR variations in the environmental impacts per tonne of rainbow trout  
929 at market size in conventional (in blue) and organic (in red) production systems.



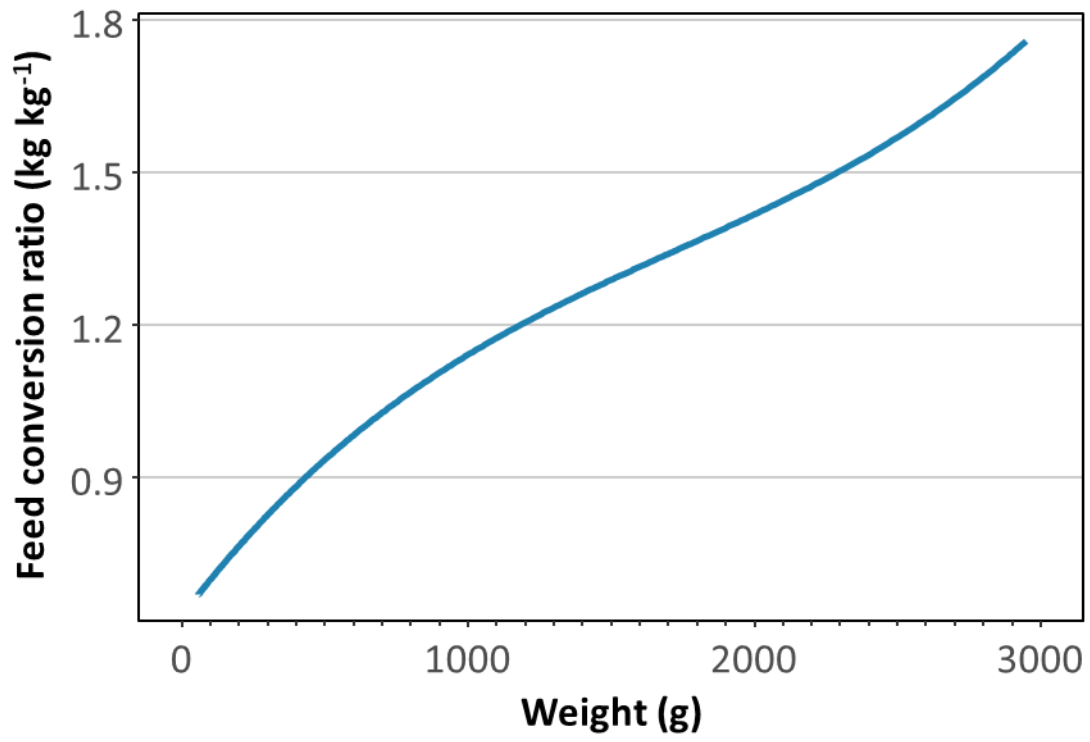
930 Figure 1



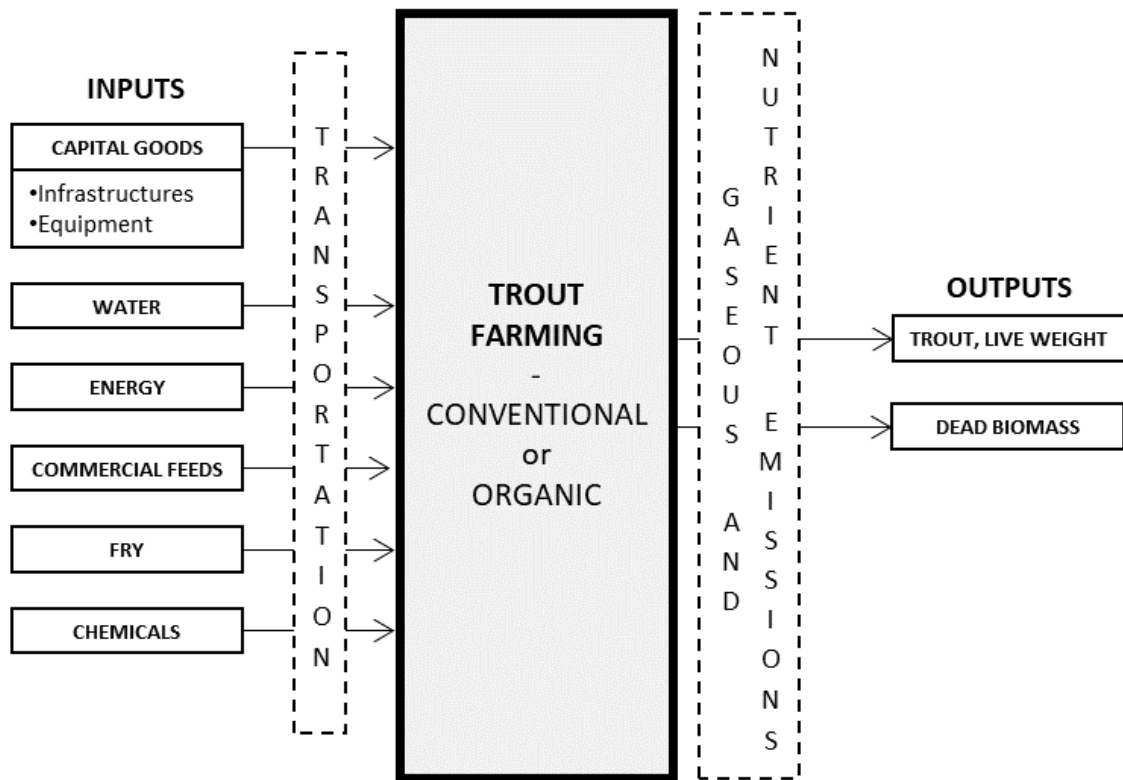
931 Figure 2



932 Figure 3

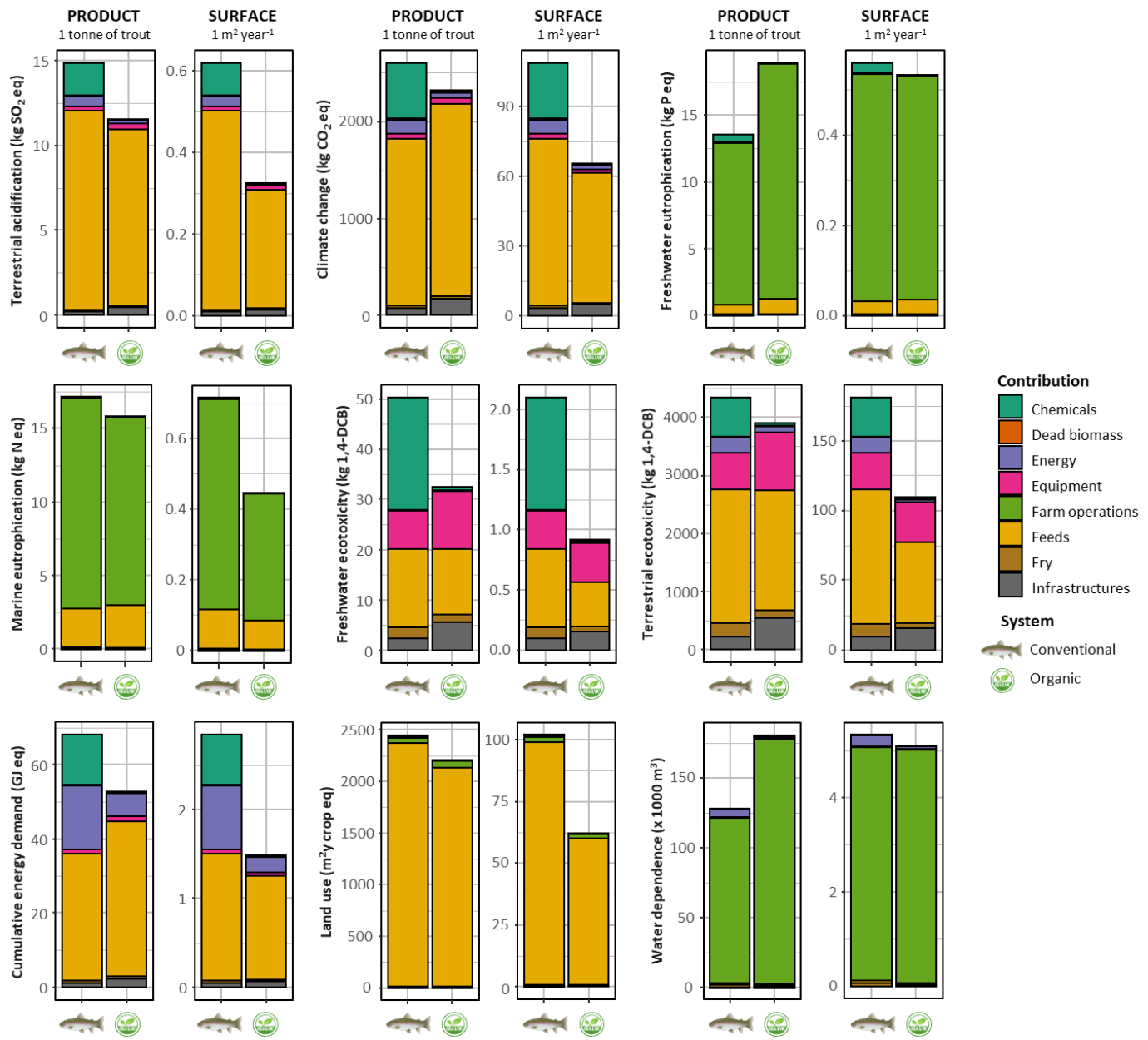


933 Figure 4

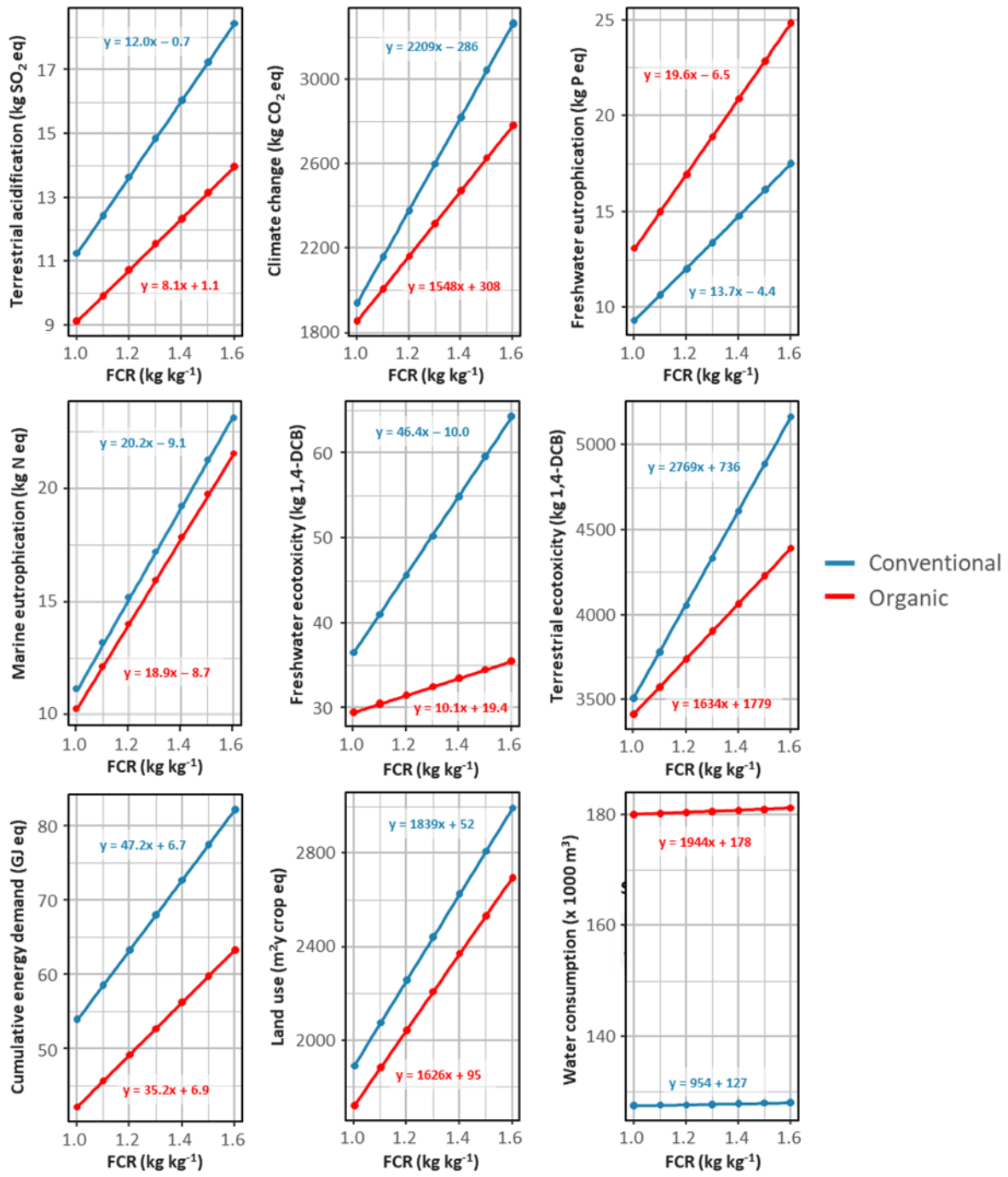


934 Figure 5





935 Figure 6



936 Figure 7

937 Table 1. Type of trout farms considered in the two different scenarios.

	Conventional	Organic
Production (t year <sup>-1</sup> )	300	203
Rearing duration (d)	737 ± 2	913 ± 4
FCR	1.3	1.3
Mortality rate (%)	15	15
Number of batches per year	3	3

938 FCR = Feed Conversion Ratio calculated as the ratio of feed intake to fish weight gain over one cycle

939 of production

940 Table 2. Composition of the feeds used in the two different scenarios.

	Conventional	Organic
<b>Proteins (%)</b>		
Feed 1	45	43
Feed 2	40	39
Feed 3	39	38
Feed 4	38	36
<b>Lipids (%)</b>		
Feed 1	21	21
Feed 2	23	24
Feed 3	27	26
Feed 4	30	28
<b>Carbohydrates (%)</b>		
Feed 1	12.0	13.0
Feed 2	13.9	14.0
Feed 3	12.8	13.6
Feed 4	12.8	11.4
<b>Phosphorus (%)</b>		
Feed 1	0.95	1.70
Feed 2	0.95	1.70
Feed 3	0.90	1.70
Feed 4	0.90	1.60

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	Item	Unit	Conventional	Organic
INPUTS	Site surface	m <sup>2</sup>	16000	16000
	Water	m <sup>3</sup>	35785586	35785586
	Fry (10 g)			
	Triploid trout (♀)	u	120000	-
	Organic trout (♀/♀)	u	-	81000
	Feeds			
	feed_1/ feed_org_1	kg	3127	2120
	feed_2/ feed_org_2	kg	41192	29841
	feed_3/ feed_org_3	kg	120688	69479
	feed_4/ feed_org_4	kg	223868	151370
	Chemicals			
	Liquid oxygen	m <sup>3</sup>	277036	-
	Antibiotics	kg	0.24	0.16
	Others	kg	4000	4000
	Electricity	kWh	427512	106440
	Infrastructures			
	60-m <sup>2</sup> building	u	1	1
	80-m <sup>2</sup> building	u	1	1
	100-m <sup>3</sup> raceways	u	12	12
	250-m <sup>3</sup> raceways	u	24	24
	Equipment			
	Feed storage silo	u	5	5
	Oxygen cone	u	2	-
	Oxygen tank	u	1	-
	Leaf screener	u	1	1
	Fish elevator	u	2	2
	Drum filter	u	1	1
	Electric generator	u	1	1
	Pumps	u	3	1
	Aerators	u	12	36
	PVC pipe	m	1500	1500
OUTPUTS	Trout at market size (3 kg)	kg	300478	202909
	Dead biomass (incinerated)	kg	9158	5990
	Water (back to river)	m <sup>3</sup>	35785586	35785586
	Nitrogen (in river)	kg	14512	8793
	Phosphorus (in river)	kg	2254	2701
	COD (in river)	kg	62435	39631

942 Table 3. Life Cycle Inventory for one year of production.

943 u: unit; COD: Chemical Oxygen Demand; Transportation was included at each step when needed.

944 Table 4. Assumptions made to fill inventory gaps.

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	Assumption(s)
Wastewater treatment	We assumed that a sedimentation area can remove 20% of suspended N and P (Stewart et al., 2006)
Lifespan of infrastructures and equipment	Adoption of the average lifespan (assuming only ordinary maintenance): equipment: 10-15 years; buildings and raceways: 30 years  The occupancy rates of the rearing structures were used as weights for these processes in the LCA:
Rearing structures occupancy	<u>Conventional production:</u> 100-m <sup>3</sup> raceways: 62% 250-m <sup>3</sup> raceways: 45%  <u>Organic production:</u> 100-m <sup>3</sup> raceways: 70% 250-m <sup>3</sup> raceways: 87%
Infrastructures weigh	<u>Buildings:</u> Walls: 0.15 m thick. Slab: 0.25 m thick Framework: 40 kg wood m <sup>-2</sup> <u>Raceways:</u> Walls: 0.15 m thick considering raceways of 1.5 m deep. Slab: 0.25 m thick Concrete density was considered equal to 2150 kg m <sup>-3</sup> Wood density was considered equal to 750 kg m <sup>-3</sup>
Transport distances	Road distances were calculated from Google Maps; ocean distances (transport of aquafeed ingredients from South America to a French harbour) were assessed from shiptraffic.net

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946 Table 5. Characteristics of the selected impact categories.

Impact category	Abbreviation	Unit	Definition
Climate change potential	GWP	kg CO <sub>2</sub> eq. to air	the contribution of greenhouse gases to global warming
Terrestrial acidification potential	TAP	kg SO <sub>2</sub> eq. to air kg	changes in acidity in the soil due to a change in acid deposition, which in turn is a consequence of changes in air emission of NO <sub>x</sub> , NH <sub>3</sub> and SO <sub>2</sub>
Freshwater eutrophication potential	FEP	kg P eq. to freshwater	a change in the levels of P in freshwater caused by emissions of nutrients into water and soil
Marine eutrophication potential	MEP	kg N eq. to freshwater	a change in the levels of N in marine water caused by emissions of nutrients into water and soil
Terrestrial ecotoxicity potential	TETP	kg 1,4-DCB eq. to soil	a change in the levels of toxic chemicals caused by emissions into the soil
Freshwater ecotoxicity potential	FETP	kg 1,4-DCB eq. to freshwater	a change in the levels of toxic chemicals caused by emissions into the water
Cumulative energy demand	CED	GJ eq.	the direct and indirect consumption of energy
Land use	LaU	m <sup>2</sup> y crop eq.	the ground surface used directly (land occupied by ponds) and indirectly (land used to grow feed ingredients)
Water dependence	WD	m <sup>3</sup>	the water flowing into the production system

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948 Table 6. Comparison of the results assessed with the CML baseline method (Guinée, 2002) and Cumulative Energy Demand indicator (Frischknecht et

	Global warming (kg CO <sub>2</sub> eq.)	Acidification (kg SO <sub>2</sub> eq.)	Eutrophication (kg PO <sup>4-</sup> eq.)	Terrestrial ecotoxicity (kg 1,4 DCB eq.)	Freshwater ecotoxicity (kg 1,4 DCB eq.)	Cumulative Energy Demand (GJ)
This study	2571	15	57	114	873	68
Literature						
Flow-through system	1157-3561	10-19	46-75	17-169	1290*	30-78
RAS	2043-13622	13-46	4-21	-	-	63**

949 al., 2007) with literature data on conventional production systems. Impacts are scaled on 1 tonne of trout.

950 Values for flow-through systems were taken from eight studies (Aubin et al., 2009; Boissy et al., 2011; Chen et al., 2015; d'Orbcastel et al., 2009;

951 Dekamin et al., 2015; Maiolo et al., 2021; Papatryphon et al., 2004; Samuel-Fitwi et al., 2013) while values for RAS were taken from three studies

952 (d'Orbcastel et al., 2009; Dekamin et al., 2015; Samuel-Fitwi et al., 2013).

953 \* Maiolo et al. (2021)

954 \*\*d'Orbcastel et al. (2009)