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Highlights

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

Abstract

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

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

1. Introduction

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

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

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

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

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

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

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

2. Materials and Methods

2.1. Farm model

The farm model, developed using the R freeware (R Development Core Team, 2022), has been partially adapted from previous investigations (Besson et al., 2017, 2016, 2014) to facilitate the acquisition of input values required for conducting a LCA at the farm level. In the present study, the model was customized to simulate daily the production of rainbow trout (*O. mykiss*) in a hypothetical flow-through farm, using actual farm data obtained from surveys conducted in Britany in 2022. The hypothetical farm consisted of 12 concrete raceways 100 m^3 each for the pre-growing phase, and 24 concrete raceways 250 m^3 each for grow-out. Among the 250 m^3 raceways, 50% received first water, meaning that the water entered the tanks directly from the river, while the remaining 50% received second water, supplied solely by the outlet water from the upstream raceways (Figure 1). In addition to the raceways, the farm was equipped with five feed storage silos and two warehouses measuring 60 and 80 m^2 (Figure 1).

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

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

2.1.1. Environmental parameters

The daily temperature (T) was modelled using a sinusoidal function with a period of 365 days. As suggested by Seginer and Halachmi (2008), T_n is given by:

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

where n is a day from 1 to 365, T_m is the mean water temperature (13 °C), T_a is the amplitude of the variation (8°C corresponding to a difference of 2 × 8 = 16 °C between the minimum and maximum daily temperature across the whole year), and ϕ_T is the phase shift (time-delay of 27.36 d) (see Supplementary Material Figure S1A).

Dissolved oxygen concentration ($[O_2]$ in mg L^{-1}; Figure S1B) at day n in surface water was calculated from Mortimer (1956) considering a standard pressure of 1 atm:

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

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

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

2.1.2. Growth

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

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

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

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

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

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

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

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

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

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

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

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

2.1.3. Mortality

This study applied a mortality rate of 15% throughout the production cycle, from 10 to 3000 g. It was assumed that the probability of daily mortality was not linear across the rearing period and is higher for younger individuals (Gåsnes et al., 2021). A Weibull function was considered to model survival, as it is commonly used for survival analysis (Carroll, 2003). So, the hazard function h, which defines the death rate at a given day (n) conditional on survival until time n or later, can be calculated as follows:

$$\mathrm{h(n)} = \frac{\mathrm{f(n)}}{1-\mathrm{F(n)}} \tag{7}$$

considering the Weibull density function $\mathrm{f(n)} = 1 - \exp^{-(\frac{n}{\lambda})^{s}}$ (8)

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

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

2.1.4. Biomass

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

$$\mathrm{BM(n)} = \mathrm{W(n)} \times \mathrm{SN(n)} \tag{10}$$

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

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

2.1.5. Raceways occupation

In our model, the occupancy of the raceways was determined by the densities achieved, which necessitates regular sorting of the fish during rearing. Initially, we assumed that each batch was stocked in a 100-m^3 raceway. As the fish grow, they were periodically redistributed into 2 and then 4 raceways of 100 m^3 before ultimately occupying 4 then 8 raceways of 250 m^3. The maximum density constraints varied depending on the production scenario. In the conventional production scenario, the density limits applied were 50 kg m^{-3} when $W \leq 50$ g, 70 kg m^{-3} when 50 g $< W \leq$ 1000 g and then 90 kg m^{-3} when $W >$ 1000 g (CIPA, 2023). For the organic production scenario, the density limits were as follows according to CIPA (2023): 25 kg m^{-3} when $W \leq 15$ g, 30 kg m^{-3} when 15 g $< W \leq$ 30 g and then 35 kg m^{-3} when $W >$ 30 g. Each rearing structure's occupancy rate was calculated as the sum of the surface used per day divided by the total surface available over a year (expressed as m^2y).

2.1.6. Feeds

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

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

where α is a scaling factor to obtain a realized FCR of 1.30 kg kg^{-1} over the production cycle for each batch in the two production scenarios, assuming that the conventional and organic fish lines have the same feed efficiency (Figure 4). Daily feed intake (DFI, kg d^{-1}) is calculated back from FCR and DWG by:

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

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

2.1.7. Nutrient release

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

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

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

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

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

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

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

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

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

$$N_{suspended}(n) = N_{eaten}(n) \times (1 - Dig_N) \quad (18)$$

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

where Dig_N is the digestibility coefficient set at 94% for proteins and 61% for P (Dalsgaard and Pedersen, 2011).

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

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

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

$$\mathrm{COD(n)} = \left[\left(\mathrm{P}_{feeds}(\mathrm{n}) \times (1 - \mathrm{Dig_P}) \times 1.66\right) + \left(\mathrm{L}_{feeds}(\mathrm{n}) \times (1 - \mathrm{Dig_L}) \times 2.78\right) + \right.$$

$$\mathrm{C}feeds(\mathrm{n}) \times 1 - \mathrm{DigC} \times 1.19 \times \mathrm{DFI(n)} \tag{21}$$

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

2.1.8. Oxygen

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

$$\mathrm{O}_{2_{\mathrm{inlet}}}(\mathrm{n}) = \mathrm{O}_{2_{\mathrm{conc}}}(\mathrm{n}) \times \mathrm{Water_{flow}}(\mathrm{n}) \tag{22}$$

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

$$\mathrm{Water_{flow}}(\mathrm{n}) = \alpha \times \frac{\mathrm{Water_{total}}(\mathrm{n})}{\mathrm{RN}}$$

where $Water_{total}$ is the water flow for the whole fish farm, RN is the number of raceways, and α is a size coefficient (i.e. 0.29 for 100-m^3 raceway and 0.71 for 250-m^3 raceway). Then, O_2 consumption is given by:

$$O_{2_{cons}}(n) = \left[(DFI(n) \times P_{feeds}(n) \times Dig_P) - \left(DFI(n) \times P_{fish}(n)\right)\right] \times \frac{E_P}{Q_{oxP}} + \left[(DFI(n) \times\right.$$

$$Lfeedsn \times DigL - DFIn \times Lfishn \times ELQoxL + DFIn \times Cfeedsn \times DigC - DFIn \times Cfishn \times ECQoxC \quad (23)$$

where Q_{oxP}, Q_{oxL} and Q_{oxC} are the oxy-caloric coefficients of proteins (13.4 MJ kg O_2^{-1}), lipids (13.7 MJ kg O_2^{-1}) and carbohydrates (14.8 MJ kg O_2^{-1}) (Brafield and Solomon, 1972; Elliott and Davison, 1975) and E_P, E_L and E_C are the energy contents of proteins (23.6 MJ kg^{-1}), lipids (39.5 MJ kg^{-1}) and carbohydrates (17.2 MJ kg^{-1}) (Brafield and Llewellyn, 1982).

No oxygenation or aeration was required if the difference between O_2inlet and O_2cons was higher than the 80% saturation O_2 concentration (O_{2}_80% = 0.8 [O_2]). Conversely, a result lower than $O_{2\_80\%}$ indicated the need for O_2 supplementation (O_{2sup}) either through the addition of liquid O_2 or by aeration:

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

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

2.1.9. Energy

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

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

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

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

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

$$E_{oxygen}(n) = 0 \text{ for } \mathrm{O}_{2_{\mathrm{sup}}}(\mathrm{n}) = 0 \quad (29)$$

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

$$E_{oxygen}(n) = \frac{\mathrm{O}_{2_{\mathrm{sup}}}(\mathrm{n})}{1.5} \quad (30)$$

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

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

2.2. Life Cycle Assessment (LCA)

2.2.1. Goal and scope

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

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

2.2.2. Life cycle inventory

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

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

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

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

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

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

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

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

2.2.3. Life cycle impact assessment

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

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

2.2.4. Sensitivity analysis

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

3. Results

3.1. Impact assessment under the two production scenarios

Figure 6 presents the level of the environmental impacts and the contribution of the system components to the impacts in conventional and organic productions of rainbow trout. The impacts are calculated using the ReCiPe method using two different functional units: per tonne of trout (product-based) and per m^2y (surface-based). Among the nine impact categories analyzed, the conventional production system exhibits higher impacts for all categories, except for FEP and WD, when the results are expressed per tonne of trout. For instance, in the conventional production system, one tonne of trout emits 14 kg P eq. and depends on 128,000 m^3 of water, while an equivalent quantity of organic trout emits 19 kg P eq. and depends on 185,000 m^3 of water. When the results are expressed per m^2y, organic production shows lower environmental impacts for all categories, including FEP and WD (Figure 6). Regardless of the functional units chosen (product-based or surface-based), other impact categories also follow similar trends, with the surface-based functional unit leading to a more significant gap between the two production scenarios.

The highest environmental gains observed in the organic system compared to the conventional production were for FETP (35% less in the organic system): the production of one tonne of trout induced 50 kg 1,4-DCB/tonne in the conventional system, but the FETP value of the organic system was noticeably lower with 33 kg 1,4-DCB/tonne. The energy requirements (CED) for producing one tonne of trout were also noticeably different between the two production systems (30% less in the organic system), with values of 68 and 53 GJ/tonne in conventional and organic production, respectively. Terrestrial acidification potential (TAP) was 28% less in the organic system, with 15 and 12 kg SO_2 eq./tonne in conventional and organic production, respectively. Differences between conventional and organic productions were less pronounced for the other impact categories. GWP showed a reduction of 12% in the organic system with, per kg of fish produced: 2602 kg CO_2 eq./tonne were estimated in the conventional system vs. 2319 kg CO_2 eq./tonne for the organic system (Figure 6). The environmental gains through organic production were equal for LU and TETP; both impact categories showed a reduction of 11% in the organic system, while MEP was only diminished by 7% (Figure 6).

3.2. Estimates from literature

The results in Figure 6 cannot be compared with those in previous LCA publications, mainly obtained with the CML baseline method. Thus, the environmental impacts of the production of one tonne of trout were also assessed using the CML baseline methodology. Therefore, we conducted a literature review on existing LCA in rainbow trout farming and compared our findings with those reported in prior studies. As there was no assessment of environmental impacts for organic rainbow trout farming, our comparison was centered on conventional production using tonne of trout as a functional unit (Table 6). Overall, our results are consistent with those found in previous studies. For clarity, the values presented in the subsequent paragraphs are given for the ReCiPe method and per tonne of trout at market size, while the results expressed per m^2y can be found in Figure 6.

3.3. Contribution of the rearing system components

As illustrated in Figure 6, the contributions of the rearing system components (i.e., chemicals, dead biomass, energy, equipment, feeds, fry, and farm functioning) varied according to the impact category and the production system. Overall, the ranking of the different contributors among the seven impact categories remained relatively constant between conventional and organic productions except for chemicals, driven mainly by liquid oxygen, accounting for a non-negligible part of conventional production's environmental impacts, but not in organic production (Figure 6). Results presented in Figure 6 show that, for MEP, FEP, and WD, farm operations contributed the most to the impacts (81-84%, 90-93% and 93-98%, respectively), and the second largest contributors are either feeds for MEP and FEP (15-18% and 5-6%, respectively) or energy for WD (1-5%). For five out of nine impact categories (i.e., LU, TAP, GWP, CED and TETP), exogenous feeds were the main contributors (96-97%, 79-90%, 66-85%, 50-79% and 53%, respectively), whatever the production systems. Equipment and infrastructures plaid a significant role in the FETP and TETP in the two production systems (20-53%), while their role was relatively negligible in the other impact categories. As mentioned earlier, the most remarkable difference in the contributions to environmental impacts of the two production systems concerned the role of chemicals. Indeed, chemicals included antibiotics, other veterinary products and disinfectants, whose quantities remained relatively constant between conventional and organic production (Table 3). On the other hand, the significant difference is related to the use of liquid oxygen only in conventional production, which was included in chemicals (Table 3). Thus, while chemicals represented only <2% of CED, FETP, GWP, TAP, TETP and CED in organic production, they represented between 13% and 44% of the corresponding impacts in conventional production (Figure 6).

3.4. Sensitivity analysis

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

4. Discussion

4.1. Benefits of the modeling approach for environmental impact comparison

Despite the rapid growth of organic agriculture production, organic finfish aquaculture remains relatively new and is still in its early stages (Mente et al., 2011). In Europe, the development of this sector has been hindered by technical challenges, such as the limited availability of organic feed and fry. Additionally, establishing effective communication strategies with clients proves difficult due to competition from other certification schemes, such as the Aquaculture Stewardship Council (ASC) or the Marine Stewardship Council (MSC) (European Commission, 2022). Furthermore, some organic farming systems experience lower yields, and previous research has suggested that using organic feed ingredients may lead to reduced farm eco-efficiency and increased environmental concerns (Pelletier and Tyedmers, 2007). However, peer-reviewed studies comparing the environmental impacts of conventional and organic aquaculture production systems (Biermann and Geist, 2019; Jonell and Henriksson, 2015) are scarce.

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

4.2. LCA literature on conventional production

Before delving into the analysis of environmental impacts between the two studied production systems (conventional and organic), it is crucial to establish a reference point by comparing the results obtained in the conventional production scenario with those from existing literature. This step allows us to compare the results from modelling with those obtained from actual fish farm data. To achieve this, we have used the ReCiPe method and CML baseline method (Guinée, 2002). The latter was commonly used in previous LCA studies focusing on rainbow trout aquaculture, while ReCiPe was only recently used in a rainbow trout aquaculture context, notably in Italy (Maiolo et al., 2021) and Spain (Sanchez-Matos et al., 2023). Overall, the literature comparison corroborated our findings when expressing environmental impact per tonne of trout (Table 6). Indeed, our results showing, for instance, that one tonne of trout emitted 2602 kg CO_2 eq. and 13.4 kg P eq. and required 68 GJ eq. of energy in a conventional scenario are consistent with the literature (Aubin et al., 2009; Boissy et al., 2011; Chen et al., 2015; d'Orbcastel et al., 2009; Dekamin et al., 2015; Maiolo et al., 2021; Papatryphon et al., 2004; Samuel-Fitwi et al., 2013). Nevertheless, the ranges of reported values can be broad. Despite uncertainties related to varying inventory databases and CML assessment method versions, another underlying cause of the differences in environmental impacts among studies is the use of diverse production systems and varying FCRs to produce the same quantity of trout (Philis et al., 2019; Sanchez-Matos et al., 2023).

4.3. Environment impacts under the two production scenarios

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

Overall, our study highlights a significantly lower level of environmental impacts of organic production than conventional production. It is particularly true for TAP, FETP and CED, with 22-35% less impact in the organic scenario. However, when impacts are expressed per tonne of trout, the WD and the FED are higher in the organic system than in the conventional system (Figure 6). Nonetheless, it is important to be cautious when comparing the environmental performance of the two production systems using a product-based functional unit because the production capacity in the organic system is one-third lower. Specifically, organic trout production is limited by lower rearing densities and reduced inputs, such as the absence of liquid oxygen (MAAP, 2010), while conventional intensive systems are managed with high stocking rates and inputs to achieve high productivity (CIPA, 2023). As a result, the larger production volume somewhat dilutes the environmental impacts of the conventional production system. This limitation should be considered when comparing organic and conventional systems using LCA and highlights the need to explore alternative surface-based functional units to gain a more comprehensive understanding of the comparison (van der Werf et al., 2020). Using a surface-based functional unit (m^2y), we find that the FEP and the WD become similar between the two production systems and even slightly lower in the organic system due to the absence of liquid oxygen usage (Figure 6). Our study demonstrates the benefits of organic trout production in terms of overall environmental impacts. However, considering the nuances related to production capacity and LCA functional units is crucial to gaining a well-rounded perspective on the environmental performance of both systems.

The significant importance of liquid oxygen usage in conventional production becomes apparent when conducting a more detailed analysis of the contributions to environmental impacts between the two production systems. Indeed, we found that chemicals, mostly composed by liquid oxygen in the conventional scenario, contributed up to 44% of the environmental impacts, depending on the category, and explained most of the differences we observed between organic and conventional production. Previous studies have also underscored the significance of liquid oxygen in the environmental impacts associated with aquaculture production. For instance, Song et al. (2019) highlighted that liquid oxygen contributed between 5% and 22% to all LCA impact categories. Consequently, such production inputs should not be overlooked in LCA conducted for aquaculture production systems. Likewise, the role of aquafeeds in influencing environmental impacts is fundamental, regardless of whether it is for organic or conventional production. The importance of FCR and aquafeeds, in general, has been emphasized by numerous LCA practitioners. Several studies have already concluded that feed production constitutes a significant source of environmental impact (e.g., Aubin, 2013; Bohnes et al., 2019; Wilfart et al., 2023). Although organic feed helps reduce environmental impacts in many categories, its higher proportion of fishmeal and fish oil, which are rich in P (Oliva-Teles et al., 2015), leads to a greater release of phosphate into the environment in the organic production scenario, resulting in an increased risk of FEP as shown in Figure S2. Indeed, we find that conventional feeds are responsible for 0.53 kg P eq. per tonne of feeds, while this value increased to 0.90 kg P eq. for organic feeds. It is worth noting that while feed formulations cannot be entirely disclosed due to industrial secrecy, efforts have been made to evolve these formulations.

Nonetheless, these results align with the findings of Pelletier and Tyedmers (2007), who reported considerably lower environmental impacts when feeds contained reduced proportions of fish ingredients. One solution to improve the environmental performance of aquafeeds could be to explicitly include environmental performance as a criterion for the inclusion of ingredients in feed formulation. Thus, using a multiobjective formulation process, Wilfart et al. (2023) showed the effectiveness of designing an eco-friendly feed fitting the nutritional requirements of rainbow trout.

4.4. Sensitivity of the results to change in FCR

Given the paramount importance of feeds in determining the environmental impacts of our production systems, we investigated the effects of a change in FCR on impact categories encompassed in LCA. We aimed to shed light on the relationship between FCR and environmental impacts per tonne of trout in our production systems. Here, we established a positive linear correlation between FCR and the environmental impacts observed (Figure 7). Such findings agree with previous LCA studies reporting that all environmental impacts decrease in similar proportions together with the improvement of FCR (d'Orbcastel et al., 2009; Jouannais et al., 2023; Papatryphon et al., 2004). Thus, improving feed efficiency through selective breeding, for instance, improves both the economic and the environmental performances of fish farming (de Verdal et al., 2018; Kause et al., 2022). Indeed, in rainbow trout farming, the feeds represent more than 40% of the total production costs (Kankainen et al., 2016). For these reasons, feed efficiency is now one of the main traits targeted by selection in fish (de Verdal et al., 2018).

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

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

4.5. Challenges and considerations in scaling up organic trout farming

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

5. Conclusion

Our study, through a holistic approach, demonstrated the environmental benefits of organic trout production at the farm level. Thus, we revealed that organic farming significantly reduced environmental impacts per tonne of trout in seven of the nine selected impact categories included in LCA. The only exceptions were freshwater eutrophication and water dependence, where organic production led to higher impacts per tonne of trout. Nonetheless, we found that, when results are expressed by m^2y, organic production leads to less impact in all the categories. Thus, our findings underscore the need for caution when interpreting LCA comparisons of such production systems, as they can be significantly impacted by methodological choices such as the chosen functional unit (product-based vs. surface-based). Our analysis reveals that aquafeeds and liquid oxygen usage are key factors contributing to the environmental impacts of conventional and/or organic trout production systems. By recognizing and addressing the significance of these inputs, we can take further steps towards sustainable finfish aquaculture practices.

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Captions to figures

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

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

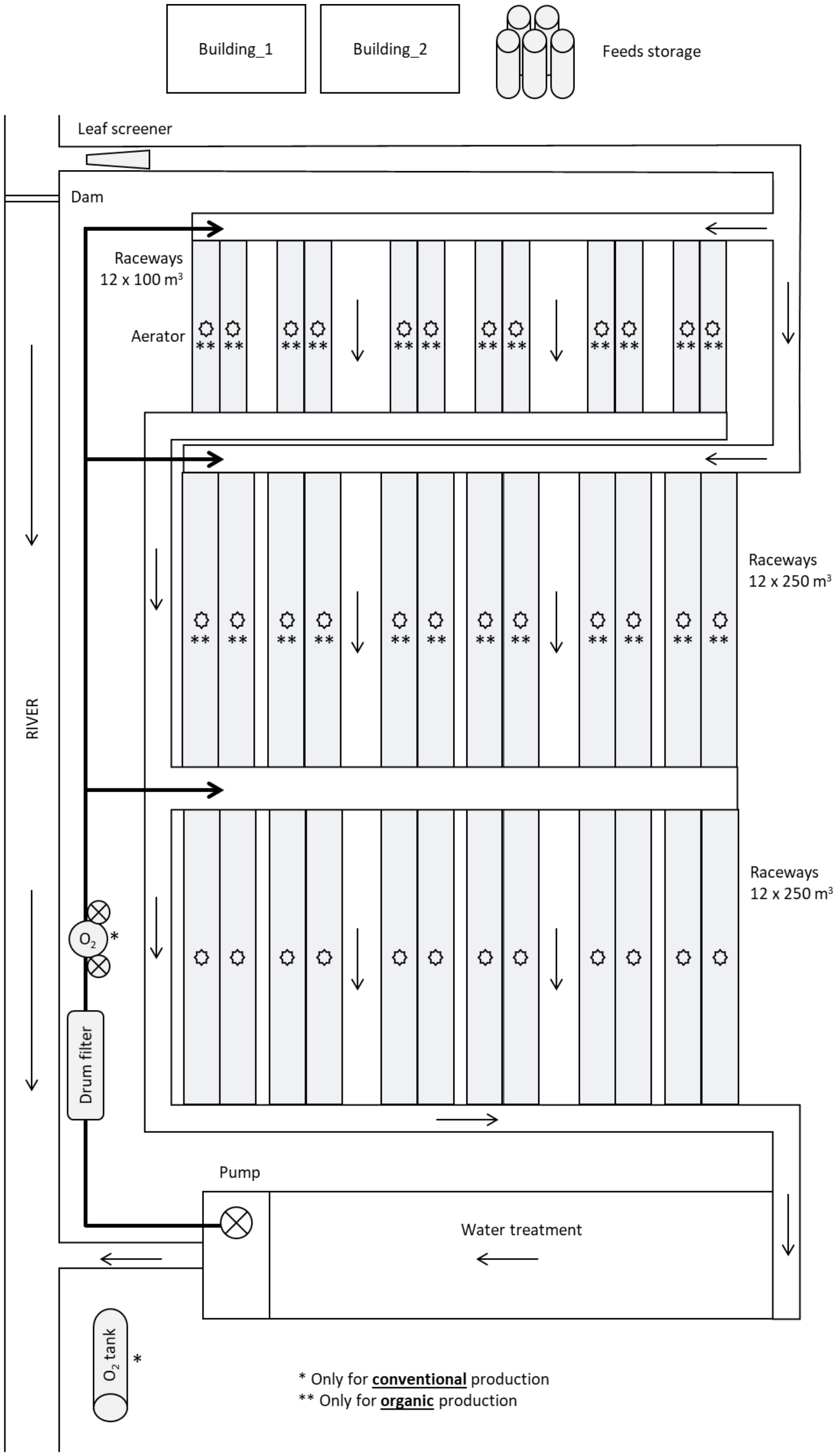
Figure 3. Graphical representations of (A) growth performances, from 0.01 to 3 kg and (B) survival of the three fish batches in conventional (in blue) and organic (in red) production systems.

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

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

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

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


Building_1
Building_2
Feeds storage
Leaf screener
Dam
Raceways
12 x 100 m³
Aerator
Raceways
12 x 250 m³
Raceways
12 x 250 m³
RIVER
O2
*
Drum filter
Pump
Water treatment
O2 tank
*
* Only for conventional production
** Only for organic production

930 Figure 1

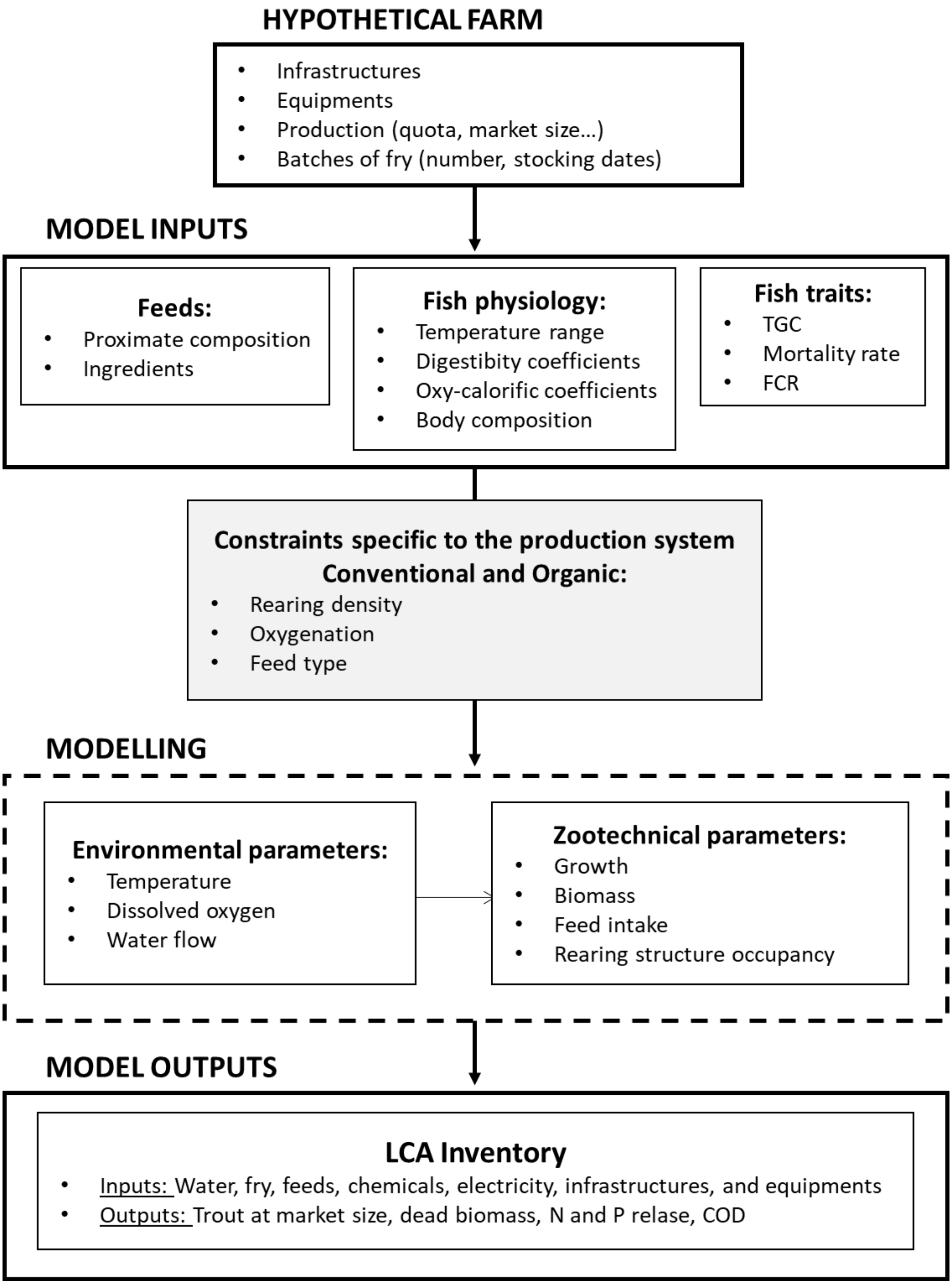
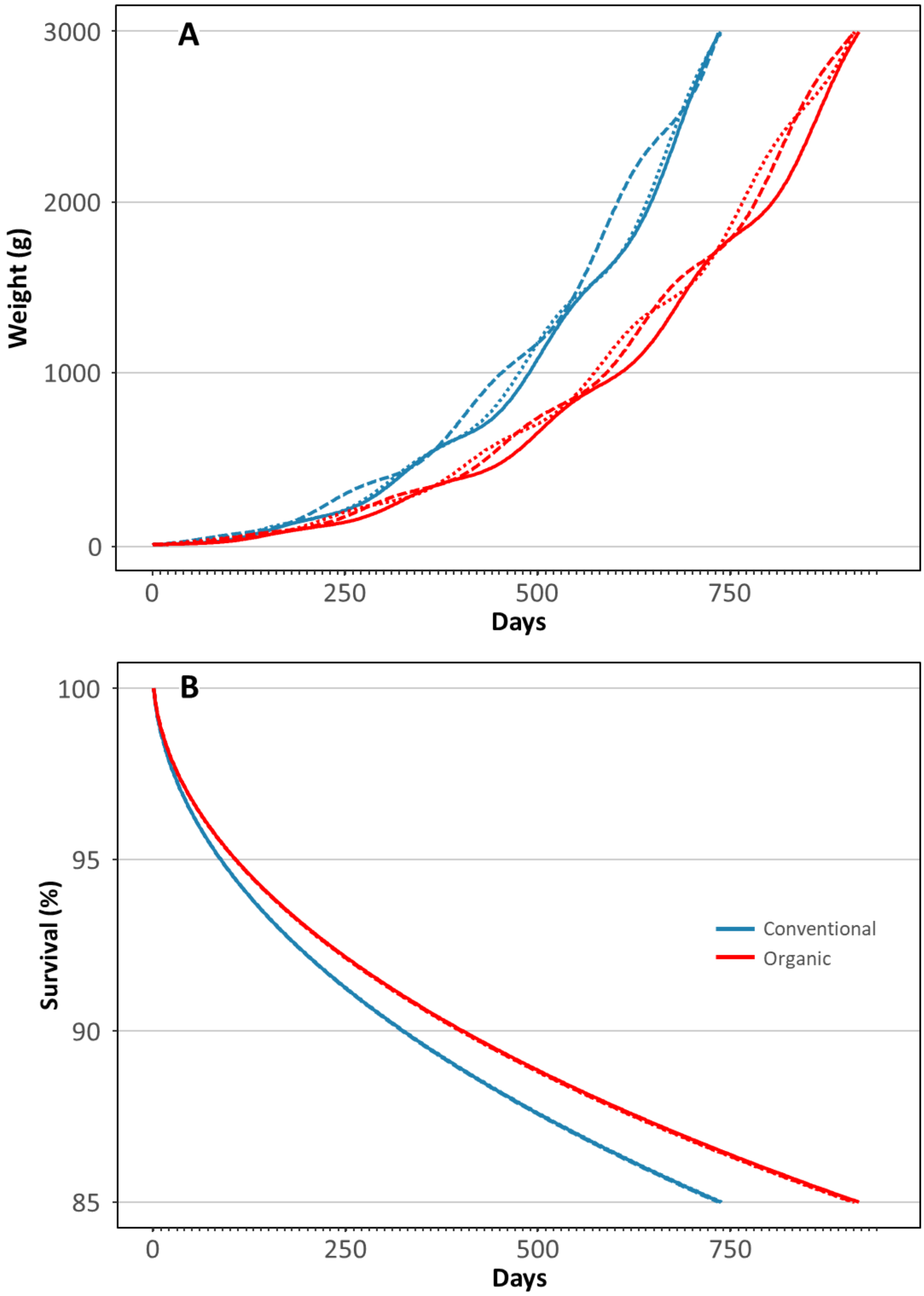

HYPOTHETICAL FARM
• Infrastructures
• Equipments
• Production (quota, market size…)
• Batches of fry (number, stocking dates)
MODEL INPUTS
Feeds:
• Proximate composition
• Ingredients
Fish physiology:
• Temperature range
• Digestibity coefficients
• Oxy-calorific coefficients
• Body composition
Fish traits:
• TGC
• Mortality rate
• FCR
Constraints specific to the production system Conventional and Organic:
• Rearing density
• Oxygenation
• Feed type
MODELLING
Environmental parameters:
• Temperature
• Dissolved oxygen
• Water flow
Zootechnical parameters:
• Growth
• Biomass
• Feed intake
• Rearing structure occupancy
MODEL OUTPUTS
LCA Inventory
• Inputs: Water, fry, feeds, chemicals, electricity, infrastructures, and equipments
• Outputs: Trout at market size, dead biomass, N and P relase, COD

Figure 2


A
Weight (g)
3000
2000
1000
0
0
250
500
750
Days
B
Survival (%)
100
95
90
85
0
250
500
750
Days
Conventional
Organic

932 Figure 3

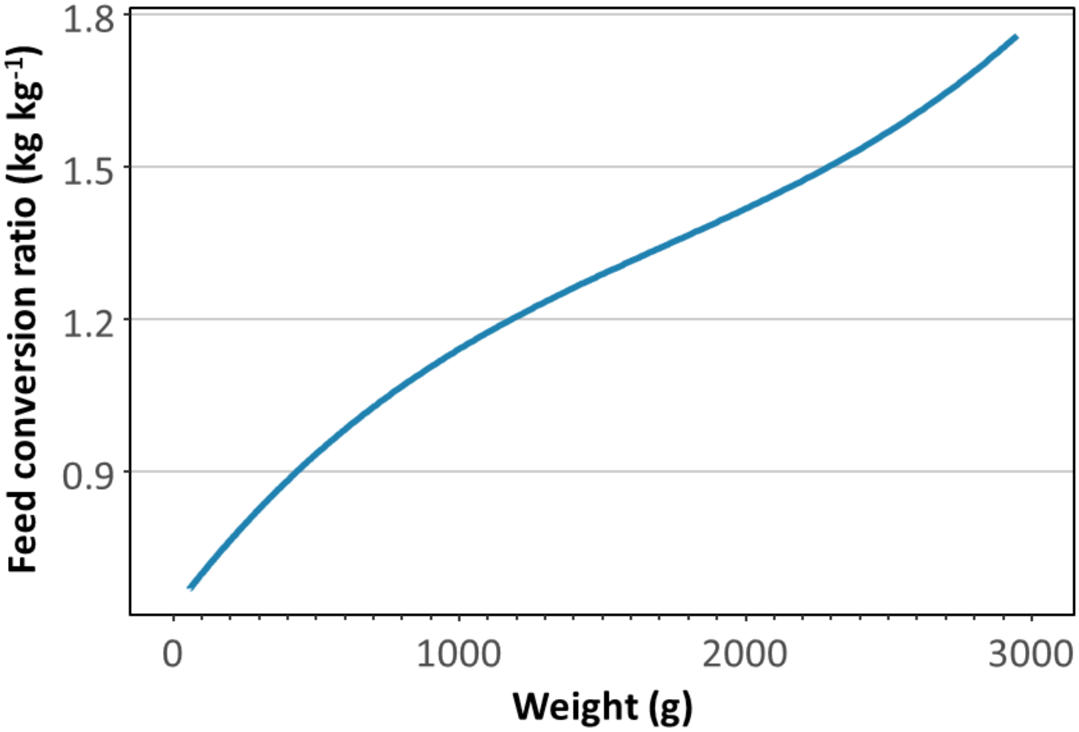

1.8
1.5
1.2
0.9
0
1000
2000
3000
Feed conversion ratio (kg kg^{-1})
Weight (g)

Figure 4

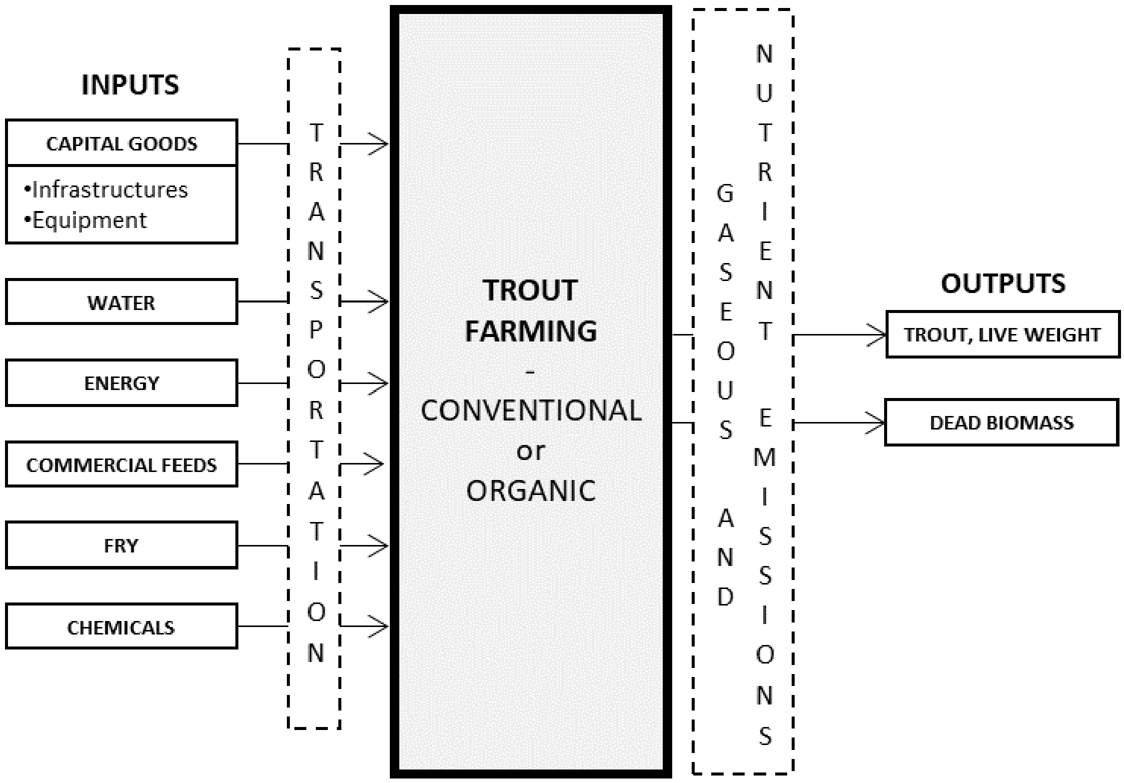

INPUTS
CAPITAL GOODS
•Infrastructures
•Equipment
WATER
ENERGY
COMMERCIAL FEEDS
FRY
CHEMICALS
TRANSPORTATION
TROUT FARMING
-
CONVENTIONAL
or
ORGANIC
GASEOUS AND
NUTRIENT EMISSIONS
OUTPUTS
TROUT, LIVE WEIGHT
DEAD BIOMASS

Figure 5

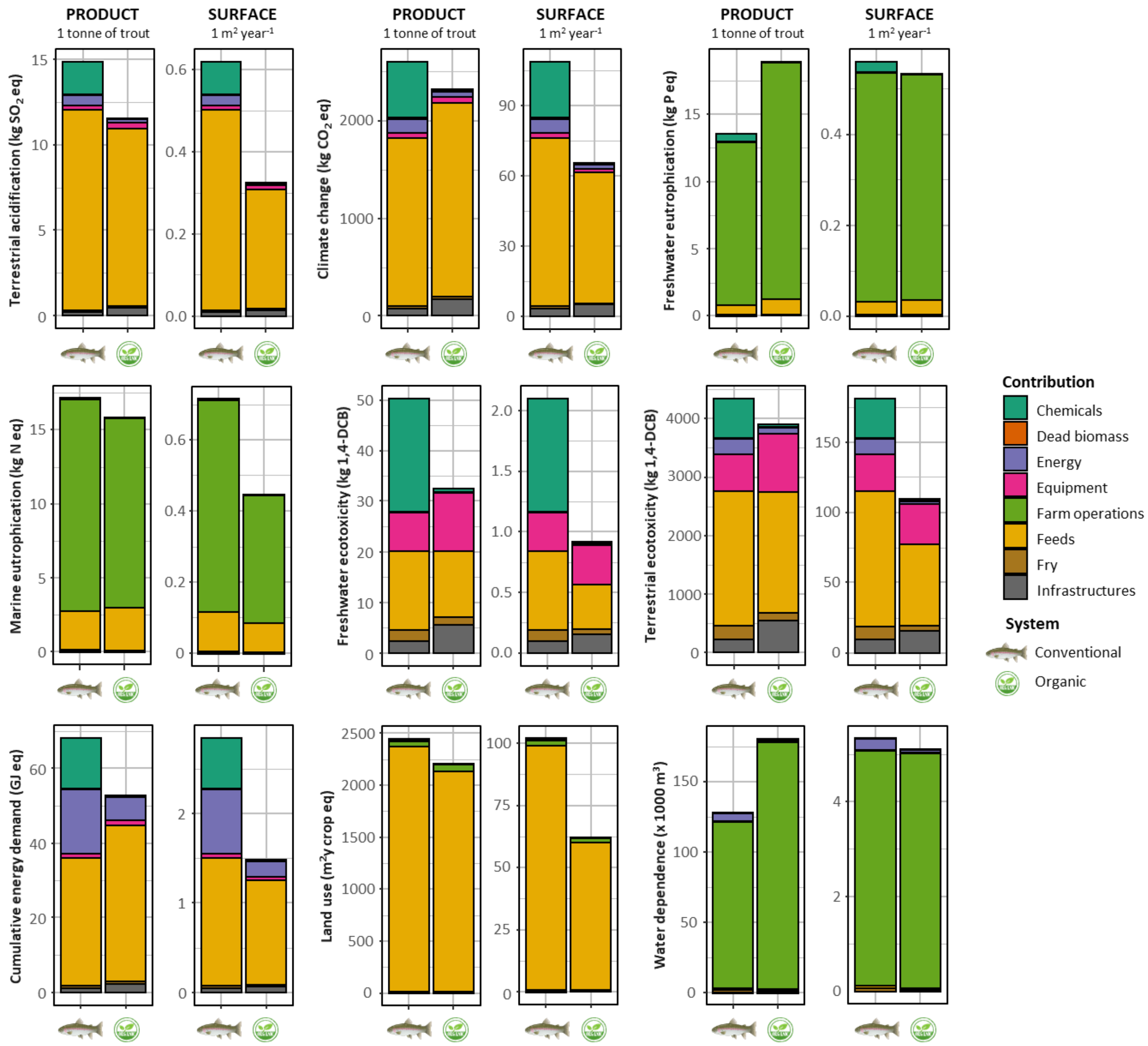
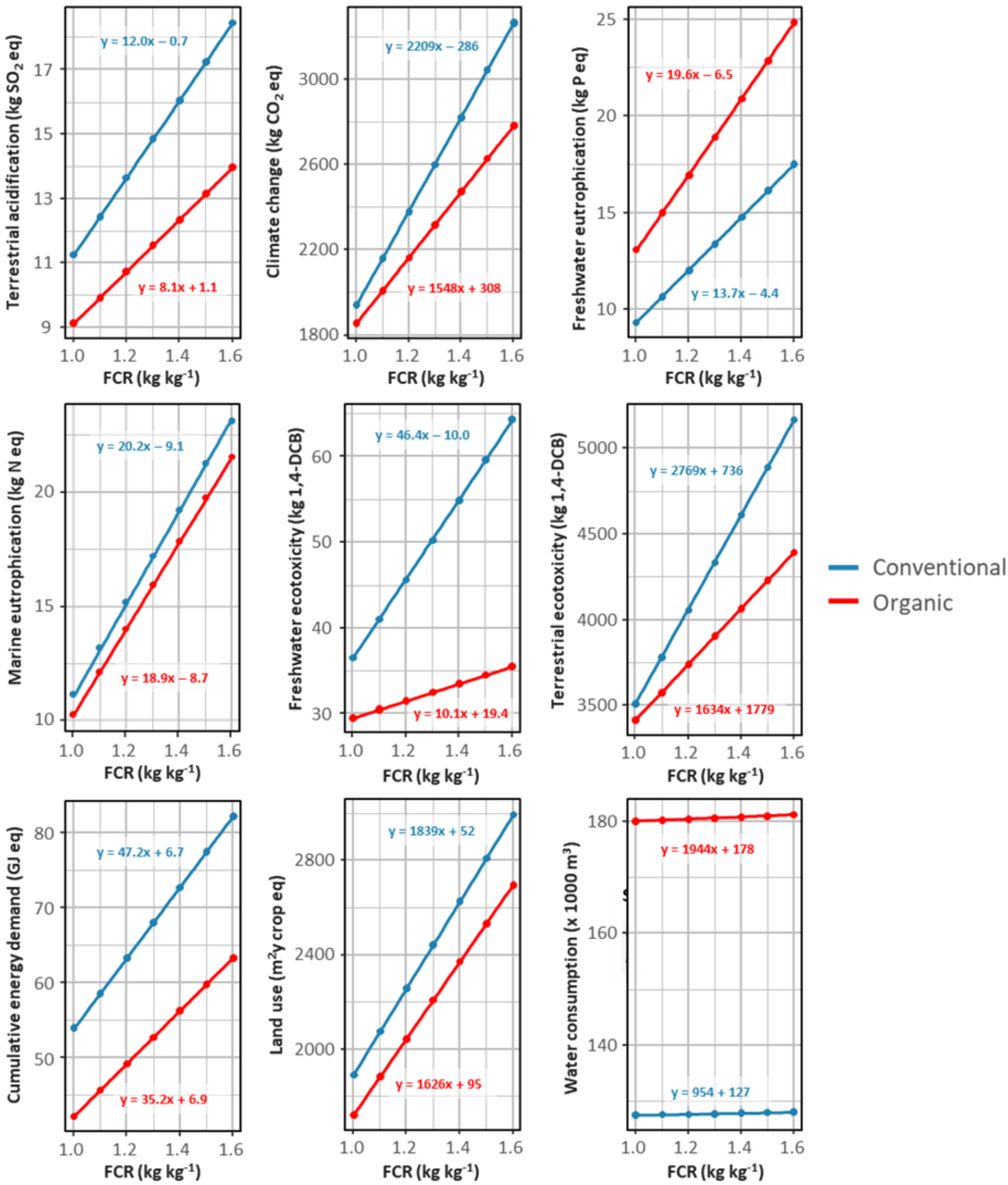

PRODUCT
1 tonne of trout
SURFACE
1 m² year⁻¹
Terrestrial acidification (kg SO2 eq)
Climate change (kg CO2 eq)
Freshwater eutrophication (kg P eq)
Marine eutrophication (kg N eq)
Freshwater ecotoxicity (kg 1,4-DCB)
Terrestrial ecotoxicity (kg 1,4-DCB)
Cumulative energy demand (GJ eq)
Land use (m²y crop eq)
Water dependence (x 1000 m³)
Contribution
Chemicals
Dead biomass
Energy
Equipment
Farm operations
Feeds
Fry
Infrastructures
System
Conventional
Organic

Figure 6



y = 12.0x – 0.7
y = 8.1x + 1.1
Terrestrial acidification (kg SO2 eq)
FCR (kg kg-1)
y = 2209x – 286
y = 1548x + 308
Climate change (kg CO2 eq)
FCR (kg kg-1)
y = 19.6x – 6.5
y = 13.7x – 4.4
Freshwater eutrophication (kg P eq)
FCR (kg kg-1)
y = 20.2x – 9.1
y = 18.9x – 8.7
Marine eutrophication (kg N eq)
FCR (kg kg-1)
y = 46.4x – 10.0
y = 10.1x + 19.4
Freshwater ecotoxicity (kg 1,4-DCB)
FCR (kg kg-1)
y = 2769x + 736
y = 1634x + 1779
Terrestrial ecotoxicity (kg 1,4-DCB)
FCR (kg kg-1)
Conventional
Organic
y = 47.2x + 6.7
y = 35.2x + 6.9
Cumulative energy demand (GJ eq)
FCR (kg kg-1)
y = 1839x + 52
y = 1626x + 95
Land use (m2y crop eq)
FCR (kg kg-1)
y = 1944x + 178
y = 954 + 127
Water consumption (x 1000 m3)
FCR (kg kg-1)

936 Figure 7

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

	Conventional	Organic
Production (t $year^{-1}$)	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

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

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

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

	Item	Unit	Conventional	Organic
INPUTS	Site surface	m^2	16000	16000
	Water	m^3	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^3	277036	-
	Antibiotics	kg	0.24	0.16
	Others	kg	4000	4000
	Electricity	kWh	427512	106440
	Infrastructures			
	60-m^2 building	u	1	1
	80-m^2 building	u	1	1
	100-m^3 raceways	u	12	12
	250-m^3 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^3	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.

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

Table 4. Assumptions made to fill inventory gaps.

	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
Rearing structures occupancy	The occupancy rates of the rearing structures were used as weights for these processes in the LCA: Conventional production: 100-m^3 raceways: 62% 250-m^3 raceways: 45% Organic production: 100-m^3 raceways: 70% 250-m^3 raceways: 87%
Infrastructures weigh	Buildings: Walls: 0.15 m thick. Slab: 0.25 m thick Framework: 40 kg wood m^{-2} Raceways: 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^{-3} Wood density was considered equal to 750 kg m^{-3}
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

Table 5. Characteristics of the selected impact categories.

Impact category	Abbreviation	Unit	Definition
Climate change potential	GWP	kg CO_2 eq. to air	the contribution of greenhouse gases to global warming
Terrestrial acidification potential	TAP	kg SO_2 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_x, NH_3 and SO_2
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^2y crop eq.	the ground surface used directly (land occupied by ponds) and indirectly (land used to grow feed ingredients)
Water dependence	WD	m^3	the water flowing into the production system

Table 6. Comparison of the results assessed with the CML baseline method (Guinée, 2002) and Cumulative Energy Demand indicator (Frischknecht et al., 2007) with literature data on conventional production systems. Impacts are scaled on 1 tonne of trout.

	Global warming (kg CO_2 eq.)	Acidification (kg SO_2 eq.)	Eutrophication (kg PO^{4-} 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**

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

* Maiolo et al. (2021)

**d’Orbcastel et al. (2009)