The environmental effect on the seabed of an offshore marine fish farm in the tropical Pacific

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\textbf{A B S T R A C T}

Marine aquaculture is expanding offshore, where the environmental interactions are not yet fully understood. We performed a benthic environmental assessment of an offshore fish farm on unconsolidated sediment. The physicochemical variables showed marked changes just under the fish farm, although the structure of the community and its bioturbation potential were not influenced. Under no or minimum influence from the fish farm, the physicochemical variables, including acid-volatile sulphides and redox, were notably different to those found in unaffected coastal areas. For this reason, classifications of the environmental status based on physicochemical variables should be adapted to offshore areas. Despite the low degree of impact detected, the organic matter carrying capacity should be carefully determined to avoid environmental drawbacks in terms of fine-grained offshore sediments. Offshore aquaculture could have a lower environmental impact than other types of aquaculture located closer to the coast, but further research is needed to obtain conclusive results.

1. Introduction

The multiple human activities occurring in coastal areas generate a complex scenario of environmental pressures and potential conflict between activities (Primavera, 2006). For this reason, offshore areas have been suggested as potential locations for the expansion of activities like aquaculture (Gentry et al., 2017a; Kapetsky et al., 2013). However, there is a knowledge gap in terms of the environmental interactions of offshore finfish farming (Gentry et al., 2017b).

Organic waste from fish farms can severely impact sediment biogeochemistry, deriving from sediment metabolism stimulation, increased oxygen consumption rates, and a depressed redox potential. Under these conditions, anaerobic metabolic pathways are enhanced, with a consequent increase in the concentrations of their deleterious byproducts in the sediment, such as sulphides and methane. These conditions negatively affect benthic communities, including the macrofauna (Papageorgiou et al., 2009; Sanz-Lázaro and Marín, 2011), which plays a key role in benthic metabolism through bioturbation and bio-irrigation (Meysman et al., 2006). These reducing conditions lead to lower diversity and favour an abundance of small-bodied species, which have a low bioturbation potential (Queirós et al., 2013). Unconsolidated marine sediments play a key role in carbon and nutrient recycling, and organic matter pollution can threaten this important ecosystem function, which translates into ecosystem services, including fueling primary production, which benefit society in terms of food provision and climate change mitigation (Sanz-Lázaro and Marín, 2011).

Offshore aquaculture can be defined as taking place in areas that are at least 3 km from the coastline and in 50 m of water depth (sensu Holmer, 2010). These characteristics favour the dispersion of particulate organic matter exported by fish farms, resulting in less deposition of this additional organic matter on the seabed close to the source (Soto and Wurmann, 2019). Thus, under offshore conditions, we could expect that organic enrichment from fish farming would have a limited environmental impact due to the low input of organic waste to the surrounding seabed. However, since benthic communities in offshore areas receive low inputs of organic matter, any increase in this could still result in a marked deleterious effect, in contrast with the equivalent effect on communities from coastal areas that naturally receive large inputs of organic matter (Goudey, 2006; Holmer, 2010).

This controversy about how vulnerable offshore areas really are to
fish farming has still not been resolved due to the lack of studies on this issue (Gentry et al., 2017b; Price and Morris, 2015). In general, research claiming to study offshore fish farming (Aguado-Giménez et al., 2011; Froehlich et al., 2017; Lin and Bailey-Brock, 2008; Molina Domínguez et al., 2001; Riera et al., 2015) does not comply with the above-mentioned definition of offshore aquaculture (Holmer, 2010). To the best of our knowledge, only one previous study on the benthic impacts of offshore aquaculture has been performed, on hard substrate (Salvo et al., 2017). However, biological communities associated with hard substrates are generally highly sensitive to organic matter pollution (Sanz-Lazaro and Martin, 2008). Hence, future offshore aquaculture facilities are expected to be located in areas of unconsolidated seabed.

The aim of this study was to assess the effect of an offshore fish farm in terms of organic matter pollution on the seabed, and test which variables are the most suitable for detecting this environmental impact under offshore conditions. To do this, the relevant physicochemical and biological variables were measured along a spatial gradient of 900 m from a fish farm located 17 km from the coastline and within a water depth of 65 m. To the best of our knowledge, this fish farm is the most truly offshore conditions in terms of depth and distance from the coastline of any aquaculture facilities studied to date.

2. Methods

2.1. Study area

The study was conducted at an offshore marine fish farm located 17 km from the coast of Manta, (Manabí Province), Ecuador (Eastern Pacific; 0° 47′ 26.10″ S, 80° 42′ 4.68″ W; Fig. S1), producing cobia (Rachycentron canadum). The fish farm consisted of 3 fish cages, each with a diameter of 30 m and the bottom of the cage reaching a depth of 15 m, in a water depth of 65 m. The seabed in the area comprises unconsolidated silt and clay sediments (Wentworth, 1922). The maximum and minimum sea surface temperature throughout the year range from 26.3 to 28.7 °C and from 23.6 to 25.1 °C, respectively. The mean surface current velocity is 0.18 ± 0.02 m s⁻¹ and the maximum is 0.73 ± 0.06 m s⁻¹; no strong deep currents have been detected (Avendaño Villamar and Pazos Noriega, 2018). The wet season in this area is between December–January and April–May, due to the effect of the El Niño current and the Intertropical Convergence Zone. The rest of the year is a dry season due to the prevalence of the Humboldt Current. Nutrient contribution from the nearby Jaramijó River and other ephemeral watercourses could influence the surrounding coastal waters, mainly during the wet season. The fish farm began production in March 2015. The sampling was performed in November 2016. In the sampling year, the fish farm produced 300 tonnes of fish and used 570 tonnes of feed, indicating a relatively high conversion ratio (1.9), mainly due to the use of feed that was not specifically designed for cobia, resulting in a low assimilation efficiency (pers. comm. from fish farmers).

2.2. Sampling

To assess the environmental effects of the fish farm on the seabed, a spatial transect was created “down current” (SW direction; Avendaño Villamar and Pazos Noriega, 2018) of the facilities, with six sampling zones 0, 20, 60, 120, 300, and 900 m distant (Fig. S1). These sampling zones were all at the same depth (64–68 m) and the grain size of the seabed was similar, with silt and clay predominating, followed by very fine and fine sand [mean ± standard error 0.2 ± 0.2, 2.3 ± 0.9, 7.4 ± 2.1, 18 ± 4, and 72 ± 6%; for gravel, very coarse and coarse sand, medium sand, fine and very fine sand and, silt and clay, respectively (Wentworth, 1922)]. In each sampling zone, four replicate samples were taken using a Van-Veen grab (400 cm²).

2.3. Physicochemical analysis

Particulate organic carbon (POC), particulate organic nitrogen (PON), total phosphorus (TP), redox potential, NH₄⁺, PO₄³⁻, and acid-volatile sulphides (AVS) were the sediment physicochemical variables measured. AVS were chosen because as this variable ranks among the most sensitive for detecting benthic impact due to fish farming (Giles, 2008), and it is also a time-integrated indicator of sulphide reduction activity in the sediment. This metabolic pathway is promoted when the sediment organic load increases (Holmer et al., 2005; Jørgensen et al., 2019).

To analyse POC, TON, and TP, the sediment was dried in an oven at 60 °C to a constant weight, before being finely ground. Next, POC (after a pre-treatment consisting of adding 1:1 HCl and PON were determined using a Carlo Erba Inst. EA 1108 Elemental Analyser (Carlo Erba Strumentazione, Milan, Italy). TP was determined following the 4500-FE ascorbic acid method (APHA, 1999). The sediment samples for measuring AVS (5 ml) were stored in ziplock plastic bags containing the least air possible, transported in cool conditions, and frozen in the laboratory. The AVS samples were distilled and quantified following the method proposed by Allen, Fu, & Deng (1993). For the NH₄⁺ and PO₄³⁻ analysis, the samples were also transported to the laboratory under cool conditions, where the porewater was extracted from the top 2–3 cm of the sediment and then frozen until analysis. NH₄⁺ and PO₄³⁻ were analysed using an Automated Wet Chemistry Analyser – Continuous Flow Analyser (Skalar Analytical B.V., Breda, the Netherlands). Redox potential was measured on board the sampling vessel, in the first 2–3 cm of the sediment using a portable data logger and an Orion ORP 91–80 electrode. The redox potential of a sediment sample rapidly increases when it comes into contact with the air. Accordingly, once the grab containing the sediment had reached the boat, the grab was opened above a tray, preserving the original stratification of the sediment, and the redox potential was measured immediately by introducing the redox probe 2 cm into the sediment, which was gently rotated to favour thorough contact between the sediment and the probe, while minimising the penetration of oxygen. Each measurement took approximately 30 s, and the lowest reading was taken. The other variables were then measured or the material was stored appropriately, to be measured in the lab subsequently, as explained above.

2.4. Biological analysis

The macrofaunal community was sampled by sieving the sediment through a 0.5 mm sieve with seawater. The retained material was fixed in a 70% alcohol solution, separated into major faunal groups and stored in a 70% alcohol solution for later identification. Benthic groups were determined to the lowest practical taxonomic level using a binocular dissecting lens. The dry biomass of each single taxon was then determined. In the case of taxa with calcareous exoskeletons, these were removed beforehand, by squashing them gently so the flesh could be extracted.

2.5. Data analysis

The indices of faunal abundance, species richness, and the Shannon-Wiener diversity index (H'; log₂) were calculated from the macrofaunal abundance data. Additionally, since macrofaunal bioturbation plays a key role in ecological functions in terms of sediment biogeochemistry, the community bioturbation potential (BPC) was calculated as a proxy of the bioturbation capacity of the macrofaunal community. The BPC per m² was calculated by combining the macrofaunal abundance and biomass data with information on life traits, specifically of the mobility and reworking capacity of each taxon (Queirós et al., 2013).

To test the effect and range of influence of fish-farming on POC, PON, TP, redox potential, NH₄⁺, PO₄³⁻, AVS, species abundance, species richness, H’, and BPC; the trend of these variables according to distance
from the aquaculture facilities was fitted to first and second-order polynomial, exponential, and inverse regressions. The Akaike information criterion (AIC) was used to choose the best model, making a compromise between the fitting and model complexity (Burnham and Anderson, 2004). Additionally, the SIMPER routine was applied to determine which species contributed the most to the dissimilarity between the sampling zone at 0 m from the fish farm and the other sampling zones.

A principal component analysis (PCA) was performed to explore the relationship between the sampling zones according to the physicochemical variables measured. Prior to the analysis, the variables were normalised (by subtracting the mean and dividing by the standard deviation) so that the scales of the units were comparable. To find significant differences among the sampling zones, SIMPROF was applied using the group-average clustering procedure based on the resemblance matrix calculated using Euclidean distance.

Principal coordinate (PCO) and cluster (group-average) analyses were performed to ordinate and classify the sampling zones based on the macrofaunal assemblage. SIMPROF was used to find significant classification differences among the sampling zones (Clarke and Gorley, 2006). Since large dissimilarities were found in the physicochemical properties, in addition to the abundance data among replicates, the mean values for each sampling zone were plotted to facilitate visualisation.

A DistLM (distance-based linear modelling) procedure was applied to test whether there was a relationship between the changes in the community assemblages and distance from the fish farm (Anderson et al., 2008). The BIOENV routine was employed to determine which combination of environmental variables best explained the macrofaunal assemblage. All the multivariate biological analyses employed the Bray-Curtis similarity matrix based on the untransformed macrofaunal abundance data, since no taxa had an abundance of above 20 individuals per sample. Multivariate analyses were performed using the software package V.6 + PERMONANOVA (Plymouth Marine Laboratory, UK). All the data was reported as the mean ± standard error (SE) and all the statistical tests were conducted with a significance level of α = 0.05.

### 3. Results

The POC values showed a slight variation, ranging from 1.41 ± 0.02 to 1.5 ± 0.04%, at 0 m and 120 m from the fish farm, respectively; and there was no significant trend with distance. The redox levels showed a decreasing trend with distance from the fish farm, ranging from −196 ± 10 to −239 ± 12.1 mV, found at 900 m and 0 m, respectively. The AVS values showed a decreasing trend with distance from the fish farm, ranging from 10.5 ± 4.4 to 126.6 ± 13.3 mg kg⁻¹, recorded at 900 m and 0 m from the fish farm, respectively (Fig. 1; Table 1).

For nutrient levels, PON values showed only a minor variation, although this increase with the distance from the fish farm was actually statistically significant, ranging from 0.21 ± 0.003% to 0.23 ± 0.004%, found at 120 m and 0 m, respectively; NH₄⁺ showed a non-significant minor variation in the trend, ranging from 75.5 ± 1.5 to 77.7 ± 0.9 μM, recorded at 900 m and 0 m from the fish farm, respectively; and TP values showed an increasing trend with distance from the fish farm, ranging from 0.68 ± 0.02 to 0.79 ± 0.1%, found at 300 m and 0 m, respectively. Similarly, PO₄³⁻ values showed an increasing trend with distance from the fish farm, ranging from 5.5 ± 0.3 to 11.6 ± 3.2 μM, recorded at 900 m and 0 m, respectively (Fig. 2; Table 1).

A total of 879 macrofaunal individuals assigned to 59 taxa were identified; these were principally polychaetes (24 families) and molluscs (23 genera; Table S1). The macrofaunal abundance ranged from 693.8 ± 190.8 to 1337.5 ± 134.4 individuals m⁻², found at 0 m and 300 m, respectively, and there was no significant trend with distance from the fish farm (Fig. 3 & Table 1). No taxa presented a high abundance, with absolute numbers always being below 500 individuals m⁻². The SIMPER routine revealed that the species which made the largest contribution to the dissimilarity between the sampling zone at 0 m and the other sampling zones were mainly the taxa Sipuncula, Glyceridae, Lumbrineridae, Spionidae, Goniadidae, Modiolus sp., and Capitellidae. (Table S2). The abundance of the above-mentioned taxa and the family Nereididae were 143.8 ± 50.4, 112.5 ± 52.5, 37.5 ± 23.9, 93.8 ± 41.3, 37.5 ± 23.9, 56.3...
Table 1
Coefficients (mean ± SE) of the significant regression models. Regression models may be inverse \( (a) f(x) = y_0 + \frac{1}{x} \) or second-degree polynomial \( (b) f(x) = y_0 + ax + bx^2 \) according to the AICc (corrected Akaike information criterion). Asterisks indicate if the p value of the coefficient was <0.05 (*), <0.01 (**), or <0.001 (***)..

|       | y0   | a      | b      |
|-------|------|--------|--------|
| Redox | -199.3 ± 7.03*** | -0.04 ± 0.02** | –      |
| AVS   | 19.4 ± 3.99***   | 0.11 ± 0.01***  | –      |
| PON   | 0.22 ± 0.002***  | 0.00001 ± 0.000005* | –      |
| TP    | 0.69 ± 0.02***   | 0.0001 ± 0.00004* | –      |
| PO4 3- | 6.67 ± 0.64***   | 0.01 ± 0.002**   | –      |
| BPC   | 276.7 ± 49.17*** | 1110 ± 461.3*   | -1227 ± 490.1* |

± 12, 43.8 ± 25.8 and 18.8 ± 12, individuals m\(^{-2}\), respectively, at the sampling zone at 0 m from the fish farm; and 212.5 ± 46.2, 50 ± 0.0, 81.3 ± 15.7, 118.8 ± 44.9, 43.8 ± 15.7, 0.0 ± 0.0, 68.8 ± 31.3 and 12.5 ± 7.2 individuals m\(^{-2}\), respectively, in the sampling zone at 900 m from the fish farm (Table S1). The species richness values ranged from 9.5 ± 2.2 to 14.3 ± 1.3 species, recorded at 0 and 20 m, respectively, and there was no significant trend with distance from the fish farm. Similarly, the Shannon-Wiener diversity values ranged from 2.68 ± 0.44 to 3.37 ± 0.09 bits, found at 0 and 20 m, from the fish farm, respectively, with no significant trend with distance.

The taxa that made an important contribution to the community bioturbation potential were biodiffusers \( (R_l = 4) \) with slow, free movement through the sediment matrix \( (M_i = 3) \) and both a large biomass and high abundance in the samples, such as, Sipuncula, Glyceraeidae, and Lumbrineridae. Other taxa with a high bioturbation potential, such as Nereididade, which are biodiffusers \( (R_l = 4) \) with free movement, via burrow systems \( (M_i = 4) \); sensu Queiros et al., 2013, presented a low abundance, therefore only making a minor contribution to the community bioturbation potential. The BPC levels presented a significant humpback trend with distance from the fish farm, ranging from 240 ± 82.4 to 536 ± 82.7, recorded at 0 and 300 m, respectively (Fig. 3; Table 1 and Table S1).

In the PCA analysis, principal components one (PC1) and two (PC2) accounted for 81.3% and 8.9% of the variability, respectively. The SIMPROF analysis significantly separated the sampling zone at 0 m from the fish farm from the other sampling zones. The variables that most notably influenced the configuration of PC1 were AVS, PO4 3-, redox, and TP, their eigenvectors having coefficients of 0.41, 0.41, 0.40, and 0.39, respectively. PC2, on the other hand, was mainly influenced by POC, NH4+, and PON, their eigenvectors having coefficients of 0.66, 0.51, and –0.47, respectively (Fig. 4).

In the PCO analysis, principal coordinates one (PCO1) and two (PCO2) accounted for 44.6% and 25.2% of the variability, respectively. The macrofauna at 20 m and 300 m from the fish farm showed a higher degree of similarity compared to the other sampling zones, despite the

![Fig. 2](image-url) Changes in particulate organic nitrogen, total phosphorus, and NH4+ and PO4 3- in pore water with increasing distance from the fish farm (mean ± SE; n = 4). Solid curves show the regression model, if significant. In these cases, the model fit is shown in addition to its coefficients according to the formula \( f(x) = y_0 + ax \), where \( f \) corresponds to the response variables, \( x \) is the distance from the fish farm, \( y_0 \) is the intercept, and \( a \) is the constant.
Fig. 3. Changes in abundance, species richness, the Shannon-Wiener diversity index, and the community bioturbation potential (BPc) with increasing distance from the fish farm (mean ± SE; n = 4). Solid curves show the regression model, if significant. In this case, the model fit is shown in addition to its coefficients according to the formula \[ f(x) = y_0 + ax + bx^2 \] where \( f \) corresponds to the response variables, \( x \) is the distance from the fish farm, \( y_0 \) is the intercept, and \( a \) and \( b \) are the constants.

Fig. 4. Principal component analysis of the sampled zones (indicated by their corresponding distance from the fish farm in metres) based on the mean values (n = 4) of particulate organic carbon (POC), particulate organic nitrogen (PON), total phosphorus (TP), redox potential, NH\(_4^+\), PO\(_4^{3-}\), and acid-volatile sulphides (AVS). The results are grouped according to the SIMPROF analysis (vertical dotted line; significance level 5%).
fact that the SIMPROF analysis did not significantly group sampling zones. The regression of the DistLM routine showed no significant trend (p = 0.28) and explained a very low fraction of the variance (R² = 0.05). The BIOENV routine indicated that AVS, redox, and TP were the physicochemical variables that showed the best correlation (0.568) with the macrofaunal community pattern, despite not being significant (15%; Fig. 5).

4. Discussion

Our work indicates that the offshore fish farm studied produces a very localised effect on the physicochemical variables of the sediment that did not translate into changes in the structure of the macrofauna.

Of all the physicochemical variables that showed a significant trend with distance from the fish farm, AVS, redox, TP, and PO₄³⁻ showed the most marked change close to the cages and these had the greatest effect on the ordination of the samples. Increased input of organic matter to the sediment promotes sediment metabolism, oxygen consumption and, thus, anaerobic metabolic pathways (Sanz-Lázaro et al., 2011b). Consequently, the environment becomes more reduced, the redox potential is lowered, and there is increased accumulation of by-products derived from anaerobic metabolic pathways, such as sulphides (Holmer et al., 2005).

Iron (Fe) in the first strata of the sediment, where oxic conditions prevail, forms Fe hydroxides, which have the ability to retain PO₄³⁻ in the sediment, creating the so-called: "Fe-lid". When the environment becomes reduced, the sulphides outcompete the hydroxide for the Fe, creating the AVS complexes and causing greater release of PO₄³⁻ to the water column (Rozan et al., 2002). This PO₄³⁻ release can be increased 3-fold under organic matter pollution conditions (Sanz-Lázaro et al., 2015). Since organic matter generally contains phosphorus, organic matter pollution constitutes an input of phosphorus to the sediment, which not only stimulates the release of PO₄³⁻ to the water column, but also favours phosphorus accumulation in the sediment. As in previous assessments of fish farming, (Holmer et al., 2007a,b; Sanz-Lázaro et al., 2011a), TP seems to be a better indicator of fish farm impact than POC or PON. This could be because the proportion of phosphorus in the fish feed could be doubled compared to carbon and nitrogen under the Redfield ratio basis (Sanz-Lázaro et al., 2015).

AVS, redox, and TP, were the most sensitive physicochemical variables in terms of detecting the effect of organic matter pollution in the seabed of the offshore fish farm studied, just as has been recorded in our study, which could be considered a reference zone (i.e., under no or minimum influence from fish farming) are notably different to those in our study, where silt and clay predominate. Sediments with a smaller grain size have lower diffusion rates, limiting the oxygen supply to the sediment, promoting reducing environments where anaerobic metabolic pathways, mainly sulphate reduction, predominate; this is a generalised pattern in the oceanic seabed (Sørgensen et al., 2019). Sediments from offshore areas are therefore expected to have a naturally lower redox potential and higher levels of AVS than sediments that are closer to the coast. Accordingly, the levels of some physicochemical variables, such as redox and AVS, in previous classifications of organic matter pollution in coastal areas may not be suitable for offshore areas. For example, according to those classifications, the redox values found in this study would be considered grossly polluted and highly hypoxic or anoxic (Hargrave et al., 2008; Wildish et al., 2001). These classifications should therefore be adapted to offshore areas, to avoid any possible misinterpretations of the benthic status.

Additionally, in the case of TP in our study, the values at 900 m from the offshore fish farm were comparable to other parts of the Pacific that are eutrophic, such as Chile (Soto and Norambuena, 2004); however, in oligotrophic areas, such as the Mediterranean or areas in the Atlantic, the values reported are one order of magnitude less (M. Holmer et al., 2007a,b; Matijević et al., 2012; Molina Domínguez et al., 2001). Thus, TP is expected to vary among basins with different trophic status, but not be specifically dependent on the distance to the coast (coastal vs. offshore).

In this study, the physicochemical variables only presented noticeable changes under the fish farm itself, indicating that the impact was very localised. However, the biotic variables showed that the structure of the macrofaunal community was not influenced by this offshore fish farm. Capitellidae, Spionidae, Glyceridae, and Nereididae are typical taxa used as indicators of organic matter pollution (Martínez-García et al., 2013). Despite most of these taxa making important contributions to the dissimilarities between the sampling zones, they did not show large abundances under the fish farm, and their abundances did not decrease with distance from the fish farm. Similarly, amphipods, which are considered an organic matter pollution-sensitive taxa (Dawin and Ruellet, 2007), did not present a low abundance under the fish cages and their abundance did not increase with distance from the aquaculture facilities. Neither the species richness, Shannon-Wiener diversity, nor SIMPROF (based on the macrofaunal community) or DistLM routines, identified significant differences in the macrofaunal community structure along distance spatial gradient from the fish farm.

Bioturbation, performed mainly by large benthic fauna, is a key process that enhances the capacity of bacteria to recycle organic matter in the seabed (Meyzman et al., 2006). The results of the community bioturbation potential showed the same trend as the macrofaunal abundance, since this latter parameter directly affects the former. The community bioturbation potential in our study area was within the range of other studies of muddy sediments (Gogina et al., 2017; Zhang et al., 2019). The humpback trend of the community bioturbation potential indicates that the community bioturbation potential is not influenced by the studied fish farm, since the values under the cages and

**Fig. 5.** Principal coordinate analysis of the sampled zones (indicated by their corresponding distance from the fish farm in metres) based on the mean values (n = 4) of macrofaunal community abundance.
in the furthest sampling zone were comparable. Thus, this study suggests that offshore fish farming does not affect the seabed ecosystem functions related to organic matter recycling through bioturbation.

The lack of response of the macrofaunal community and its bioturbation potential, suggest that the environmental effect of the fish farm caused by the input of extra organic matter was notably small, with only slight changes being seen in the environmental parameters of the sediment. These changes were not severe enough to affect the macrofaunal assemblages. The outcomes of this study contrast with previous work on the environmental effects of fish farming on a hard substrate (Salvo et al., 2017). However, fish farming activities are expected to have more deleterious effects on hard substrate habitats than unconsolidated sediments, since the communities that inhabit these are more sensitive to organic matter pollution (Sanz-Lazaro and Marin, 2008).

Following the rationale of Sanz-Lazaro et al. (2011a), the organic matter pollution carrying capacity of the fish farm was not surpassed, since the concentration of POC in the surroundings of the fish farm was comparable to more distant zones, and the macrofaunal structure was not affected. This study suggests that offshore oceanographic conditions may promote organic matter dispersion, diminishing the load that the sediment receives over a specific surface area. Any deleterious effects on the environmental status will consequently be less severe than in fish farms located close to the coast. Nevertheless, if the input of organic matter should go up, due to an increase in fish farm production, the extensive depletion of oxygen deriving from this could have a deleterious effect on the macrofauna, since more fine-grained sediments are expected to be more sensitive to organic matter pollution (Martiñe-García et al., 2015). The carrying capacity related to the organic load that offshore fish farms can receive should, therefore, be carefully calculated to avoid environmental drawbacks.

5. Conclusions

Our results show that the studied fish farm did not generate a notable benthic impact because the organic matter pollution carrying capacity was not surpassed. This suggests that offshore fish farming may have less impact on the benthos than other types of aquaculture that are located closer to the coast. This outcome could be explained by the deep and well-flushed conditions in the area in which the studied fish farm is located. Additionally, this work highlights the fact that physicochemical reference values, including those for redox and AVS, from previous classifications of organic matter pollution based on coastal areas may not be applicable to offshore areas. These classifications should therefore be adapted to offshore areas, to avoid possible misinterpretation of the benthic status. Further research on offshore fish farm facilities with different fish production and environmental conditions is necessary to obtain more generalised results and help establish the organic matter carrying capacity for fish farm wastes in this type of aquaculture.

Credit author statement

Idea: CS, UA and EM; experimental design: CS; sampling and data collection: CS, NC and UA; data analyses: CS, assisted by NC and UA; writing of the manuscript: CS assisted by NC, UA and EM. All authors contributed critically to the drafts and gave final approval for publication.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2021.113712.

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