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Macrobenthic fauna and the ecological status of a shellfish farm in the Mediterranean Sea (Algeria)

Sonia DILMI[1], Rym BACHETARZI[2] and Chafika REBZANI-ZAHAF[1]

[1] Laboratory of Oceanography Biologic and Marine Environment (LOBEM), Faculty of Biological Sciences (FSB), University of Science and Technology Houari Boumedien (USTHB), PO Box 32, El Alia, Bab Ezzouar, 16111 Algiers, Algeria
[2] Department of Ecology and Environment, Faculty of Natural and Life Sciences and Earth Sciences, University Djilali Bounaama Khemis Meliana (UDBKM), Ain Defla, Algeria

Corresponding author: Sonia DILMI; sdilmi@usthb.dz

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Abstract

A faunistic and ecological study on the benthic macrofauna was carried out in the shellfish farm in Ain Chorb (Ain Taya), central Algeria. This study aims to characterise the composition, structure, and functioning of the benthic community to establish the reference state and ecological quality of the shellfish farm and to monitor its evolution through an adapted sampling and observation methodology adapted and used by the scientific community at the international level. Benthic fauna samples were collected in two different periods (warm and cold season) between February 2020 and June 2021; six grab samples were carried out at three sampling stations. Different indices were evaluated based on the benthic community characteristics (species richness, abundance, density, ecological and trophic groups, and the biotic index) and indicator species. An inventory of the macrobenthic fauna of the farm allowed us to identify 6 phyla, 10 classes, 48 families, 106 genera, and 138 species, with a total of 45960 ind/m^2. The benthic assemblage is characterised by the dominance of the species *Abra alba*, *Salvatoria clavata*, *Caecum* spp., and *Bittium* spp. The results of the four benthic indices (the Shannon-Weaver diversity index [H′], the AZTI Marine Biotic Index [AMBI], the multivariate AZTI Marine Biotic Index [M-AMBI], and the BENTIX) indicated that that the sampling stations have a moderate and good ecological status. These results were confirmed by the abundance/biomass comparison curves and geometric abundance class methods. This study provides the first inventory and represents the reference state of the soft-bottom communities of the aquaculture farm. The findings also indicate that the macrobenthic assemblage is excellent indicator of the ecological status.

Keywords: macrobenthic fauna; biotic index; ecological status; Shellfish farm; Algeria.

Introduction

Aquaculture is an essential food sector throughout the world: it supplies a significant amount of seafood to the global market. It is currently the fastest growing food-production sector, with an annual growth rate of 5.8% between 2001 and 2016 (Tičina *et al.*, 2020). The global aquaculture fish production has reached 82.1 million tons, including 32.4 million tons of algae, 26,000 tons of ornamental clams and pearls, and a total output of 114.5 million tons, a record high. Fish production is dominated by finfish (54.3 million tons), molluscs, mainly mussels (17.7 million tons), and crustaceans (9.4 million tons) (Food and Agriculture Organization, 2020).

Marine aquaculture has both positive and negative essential effects on humans and the environment (Klinger & Naylor, 2012). The rapid expansion in coastal marine areas has led to a general concern for the impact on crucial environmental variables. The potential adverse effects of this growth have received the most attention (Burke *et al*., 2005; Naylor *et al*., 2005; Froehlich *et al*., 2017; Kim *et al*., 2022; Nanou *et al*., 2022; Rios-Fuster *et al*., 2022), including deterioration of water quality (Islam, 2005), reduced income and employment in the capture fisheries sector (Klinger & Naylor, 2012), damage to coastal habitats (Dewalt *et al.*, 1996), disease and genetic pollution (Hindar *et al*., 1991; Johansen *et al*., 2011), and creating conflicts within coastal communities (Barton & Fløysand, 2010).

Parallel to aquaculture industry development, knowledge of its effects (positive or negative) on the surrounding environment is increasing. The environmental sustainability of aquaculture has received much attention in recent decades. Sustainable aquaculture is concerned with mutual interactions of this activity on the receiving environment. Many countries are developing indicators of ecological and environmental quality at the national level as part of their international commitments, such as those under Agenda 21 and the Organization for Economic Co-operation and Development (OECD) Review. Most of the studies focus on chemical parameters but, for several reasons, direct chemical analyses of water and sediments, which are often very sensitive and accurate, do not necessarily reflect the real ecological state (Phillips & Rinbow, 1994).

The most evident effect of this activity on bottom sediments is the accumulation of organic matter, which affects the structure of the benthic communities (Karakassis *et al*., 2000; Hyland *et al*., 2005; Klaoudatos *et al*., 2006; Yucel-Gier *et al*., 2007; Moraitis *et al*., 2013). Analysis of the benthic community structure is widely used in environmental impact assessments (Pearson, 1978; Warwick & Clarke, 1994). Indeed, because of the sedentary nature of the organisms that compose them, benthic communities are considered good indicators of the environmental state (Pearson, 1978; Gray & Mirza, 1979; Hily, 1984; Rosenberg *et al*., 2004). They provide information regarding the quality of the water and sediment, and the relatively long-life cycle of benthic species allows researchers to consider the temporal effects of disturbances. Finally, there is great diversity in the sensitivity of benthic species and, consequently, in their response to disturbance, whether anthropogenic or natural (Pearson, 1978; Gray *et al*., 1988; Dauer, 1993).

Most studies have focused on the environmental effects of marine fish farming (Gray *et al*., 1988; Wu *et al*., 1994; Mazzola *et al*., 2000; Yucel-Gier *et al*., 2007; Neofitou *et al*., 2012; Kucuksezgin *et al*., 2022). Furthermore, shellfish aquaculture appears to be less damaging to the environment (Newell, 2004) because farmed shellfish species are active filter-feeders and are cultured without additional input to the marine ecosystem. They may have a crucial role in the future development of integrated multi-trophic aquaculture practices (Tičina *et al*., 2020). However, there are numerous interactions between shellfish aquaculture and the ecosystem, and organic enrichment can appear under mussel farms. This enrichment is due to the increase in the biodeposition of organic matter in faeces and pseudofaeces (Hatcher *et al*., 1994). Furthermore, the fall of mussels and associated epifauna can modify the substrate's physicochemical characteristics (Christensen *et al*., 2003).

Several studies have been conducted to assess the environmental impact of aquaculture activities (Mazzola *et al*., 2000; Neofitou *et al*., 2010; Moraitis *et al*., 2013; Tičina *et al*., 2020; Sanchis *et al*., 2021). Some studies have evaluated the influence of aquaculture by measuring the chemical characteristics of the sediment (Sutherland *et al*., 2007; Farmaki *et al*., 2014; Gu *et al*., 2021), and others have analysed biological characteristics (Chevillon, 1999; Mazzola *et al*., 2000; Sanchis *et al*., 2021). However, chemical indicators have different response times, so organic enrichment can induce changes in the structure of benthic communities even before a chemical analysis has detected a disturbance (Edgar *et al*., 2005).

For several decades, marine aquaculture has been expanding across the Mediterranean basin, including in Algeria, where a multitude of studies have been conducted. These investigations have primarily centred on the development, innovation, and evolution of this sector. For example, Zerrouki (2012) evaluated environmental parameters and revealed that aquaculture waters are nutrient rich, thus fostering an environment conducive to phytoplankton growth, which is essential for shellfish farming. However, only a handful of studies have delved into the environmental impacts of marine aquaculture. Lounas *et al.* (2020) reported that Algerian mariculture, specifically floating cage systems, does not pose a significant ecological risk to the local aquatic environment in terms of toxic metal pollution. Moreover, Lourguioui *et al.* (2017) investigated impact categories pertaining to acidification and global warming potential by employing life cycle assessment to evaluate the environmental impact of suspended mussel culture in Algeria.

In a world of declining biodiversity, to ensure effective conservation of biodiversity at all levels of biological organisation, the crucial first step is monitoring and assessment (Patrício *et al*., 2016). This information also provides essential knowledge that enriches the database for developing a sustainable aquaculture industry. The objectives of this study were: (a) to study the spatial and temporal variations while characterising the composition and structure of the benthic macrofauna community of the study area, (b) to evaluate the effectiveness of different indices (the Shannon-Weaver diversity index [H′], the AZTI Marine Biotic Index [AMBI], the multivariate AZTI Marine Biotic Index [M-AMBI], and the BENTIX), and (c) to assess the ecological status of the shellfish farm environment to determine the state of ecological health for better management of the farm.

Materials and Methods

Study area

The aquaculture shellfish and oyster farm 'SARL ORCA MARINE', which has been active since the 2000s, is located in Ain Chorb (Surcouf-Ain Taya) (36.79732° N, 3.3151° E.), 27 km north-east of Algiers, in the Bay of Zemmouri. Located 1 km from the coastline, the farm covers an area of 5000 m² (Fig. 1). Mussels and oysters are cultured on longlines (separated by 50 m). Designed to produce approximately 50 tonnes year $^{-1}$, the farm accounted for about 30% of the national production (150 tonnes) in 2013.

The bottom profile of Ain Chorb is divided into two phases: (a) a rocky band that extends over a distance of 100-500 m from the shore, (b) followed by a stretch of coarse sand, up to the 20-30 m isobaths (CAR/ASP - PNUE/PAM, 2015). The sediment below the shellfish farm is coarse sand.

This area, subject to swells and currents and relatively unaffected by urban activity, is sheltered by a bed of marine phanerogams: *Posidonia oceanica* and *Cymodocea nodosa* (Samson-Kechacha, 1992). Operating as an open

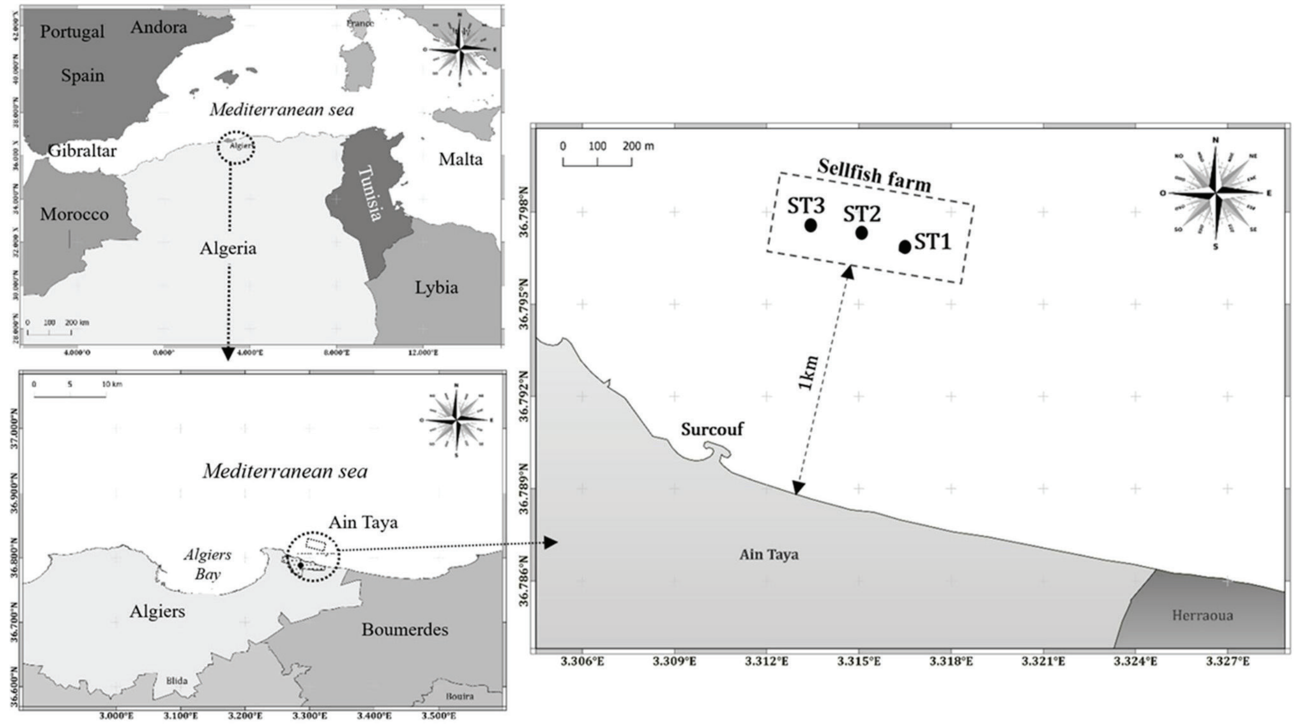

Fig. 1: Location of the shellfish farm, Orca Marine (Ain Chorb, Ain Taya, Algiers), with the positions of the sampling stations.

system, it experiences reduced stress from nutrient inputs, and environmental conditions are favourable, benefiting from a constantly-renewing water current that enhances nutrient accessibility (Zerrouki, 2012).

Sampling strategy

The sampling strategy was limited by the technical availability of the farm. The macrobenthic fauna was sampled in two seasons – in June, representing the warm season, and in February and December, representing the cold season – from 2020 to 2021. Three sampling stations were located at a depth of 23 m (extremities) to 27 m (centre) underneath the mussel bags. Specifically, two stations at both extremities of the farm and one station in the centre, upstream of the main current direction, were sampled to cover the total area of the farm (Table 1). Sampling was carried out with a Van-Veen-type grab sampler that covers a sampling surface of 0.03 m². There were six grab replicates per sampling point for a total surface area of 0.54 m² in each month. The abundances were standardised per square metre. A total of 54 samples were available from two periods. The collected sediment was sieved on a 0.5-mm mesh. The refusal of the sieve containing the benthic fauna was fixed with formaldehyde seawater (5%), then sorted and identified in the laboratory. The biomass was calculated by obtaining the dry weight of each sample of macrobenthic fauna.

Specimens, including juveniles, were meticulously identified to the species level by using a stereomicroscope (ZEISS Stemi DV4) and a trinocular microscope. Each individual was categorised based on several determination keys to guarantee precise identification. The abundances were recorded for both adults and juveniles. Many articles dealing with a single family or species were also used (Santos & da Cunha Lana, 2003; Antit & Azzouna, 2012; Høisæter, 2014; Alvarez-Campos *et al*., 2015; Vannozzi *et al.*, 2015; Middelfart *et al*., 2016; De Souza & Pimenta, 2019; Albano *et al*., 2020; Molina-Acevedo & Idris, 2020). The taxonomic nomenclature of all identified species was verified according to the World Register of Marine Species (WoRMS Editorial Board, 2024).

Table 1. Geographical coordinates and physical characteristics of the studied sites.

Site		Coordinates		Depth	Type of sediment
		N	E	(m)	
ST1	**East side**	36.7966806°	3.3164500°	23	Coarse sand
ST2	**Centre**	36.7971306°	3.3151389°	27	Coarse sand
ST3	**West side**	36.7974806°	3.3134611°	23	Coarse sand

Data analysis

Analytical characteristics provide information on the importance, place, and influence of a species in a community. The results collected during the samplings were processed according to the method developed by Pérès & Picard (1965) and adapted by Rebzani-Zahaf *et al.* (1997); it characterises the community based on its species richness and the number of individuals of each species expressed as the absolute value (abundance, ind/m^2) and the relative value (dominance, %). In order to estimate the qualitative and quantitative richness of the macrobenthic community at each station and to follow the spatial and temporal evolution of this richness, Species frequencies were analysed by utilising the coefficient introduced by Bakalem (1979) and Rebzani-Zahaf (1992, 2003). This coefficient, which reflects the extent of a species presence within a stand, categorises species into four groups based on their frequency in the sample: constant, very common, common, and rare (Table 2). The fauna lists were established for each sample and station of the aquaculture site. For each species in the list, the corresponding zoological, ecological, and trophic groups were determined.

Bio-organisms adapt to their environment's physical, chemical, and biotic components and, consequently, the environmental conditions corresponding to the ecological requirements of each species (Rebzani-Zahaf, 2003). To understand the relationship between the macrobenthic communities and the environmental conditions, the species were classified according to zoological groups (Polychaeta, Mollusca, Crustacea, Echinodermata, and Chordata). The feeding habits of the benthic community must also be considered (e.g., carnivores, deposit feeders, detritivores, suspension feeders, omnivores, and grazers) because the structural organisation is conditioned not only by abiotic factors, but also by the available nutrient resources. This approach allowed us to identify the specific characteristics that may exist during an annual cycle or at a given time.

For more accuracy, the species were also classified according to the five ecological polluosensitivity groups established by Borja *et al.* (2000): sensitive species, indifferent species, tolerant species, opportunistic species of the second rank, and opportunistic species of the first rank.

The benthic indices used in this study can be divided into two categories:

a. This category included indicators based on species diversity and density. H′, with log2 (Shannon and Weaver, 1963), compares unequal species sampling, and the Pielou evenness index (J′) (Pielou, 1966), also called the regularity and equitability index, is the ratio between the diversity measured in a stand and the maximum diversity. These two indices measure and compare the species composition at each station. The biodiversity indices were calculated according to the following formulas:

$$H' = \sum_{i=1}^{s} Pi\ log2\ Pi$$

$$J' = H'/\log_2 S$$

b. This category includes three indices. First, the AMBI developed by Borja *et al.* (2000) evaluates the ecological quality of marine environments. Second, Muxika *et al.* (2007) introduced the multivariate AZTI Marine Biotic Index (M-AMBI) to enhance the AMBI. The M-AMBI incorporates the AMBI, along with the number of species (S) and the Shannon-Wiener diversity index (H′), offering a more comprehensive analysis. In this study, the species list published on the AZTI website on May 2022 was utilised for the analysis. For the AMBI, when the percentage of taxa that are not assigned is high (>20%), the results should be evaluated with care because there may be subsequent problems in the interpretation. Finally, the BENTIX, developed by Simboura & Zenetos (2002), employs a two-group classification (sensitive and tolerant) for ecological assessment.

The index values correspond to five ecological quality status levels: high, good, moderate, poor, and bad, proposed by the Water Framework Directive (WFD). Table 3 lists the class boundaries for the aforementioned indices.

For the statistical analysis, a percentage similarity analysis (SIMPER) was used to determine the contribution of each species to the similarity-dissimilarity between the months, according to the average Bray-Curtis dissimilarity. By using the PRIMER 6 software, non-metric multidimensional scaling (nMDS) ordination based on Bray-Curtis dissimilarity was applied to determine the contribution of each species to the similarity-dissimilarity between the different stations and months. The PAST software (version 3.22) was used to perform the non-parametric Kruskal-Wallis test to determine whether there were significant seasonal and spatial differences in species diversity. This software was also used for linear (Pearson) correlation analysis to examine the relationships between the AMBI, the M-AMBI, BENTIX, and H′, and to explore the degree of potential correlations.

Table 2. Species categories according to frequency.

Species categories	Frequency
Constant	≥75%
Very common	50%-74%
Common	25%-49%
Rare	≤24%

Table 3. Thresholds and ecological status of the community according to the different classifications for the four biotic indices (the Shannon-Weaver diversity index [H′], the AZTI Marine Biotic Index [AMBI], the multivariate AZTI Marine Biotic Index [M-AMBI], and the BENTIX).

Ecological status	AMBI	M-AMBI	H′	BENTIX	Pollution classification
High	0-1.2	1 ≥ BI > 0.77	H′ > 5	4.5 ≤ BENTIX ≤ 6.0	**Unpolluted**
Good	1.2-3.3	0.77 ≥ BI > 0.53	4 < H′ ≤ 5	3.5 ≤ BENTIX ≤ 4.5	**Slightly polluted**
Moderate	3.3-4.3	0.53 ≥ BI > 0.39	3 < H′ ≤ 4	2.5 ≤ BENTIX ≤ 3.5	**Moderately polluted**
Poor	4.3-5.5	0.39 ≥ BI > 0.2	1.5 < H′ ≤ 3	2.0 ≤ BENTIX ≤ 2.5	**Heavily polluted**
Bad	5.5-7.0	0.2 ≥ BI > 0	0 < H′ ≤ 1.5	0	**Extremely polluted**

Ecosystem disturbance was investigated based on the distribution of species in geometric abundance and geometric size classes. The percentage of species was plotted against the number of individuals per species in geometric abundance classes and against the biomass per species in geometric size classes.

Abundance/biomass comparison (ABC) curves, proposed by Warwick (1986) to detect pollution effects on marine zoobenthic communities, were created by using the PRIMER 6.0 software package. The main advantage of this approach is that it provides a direct assessment of the state of the environment without prior knowledge of the site.

By using the PAST software (version 3.22), principal component analysis (PCA) was performed to identify the dominant environmental factors; environmental variables were normalised prior to applying PCA. BIOENV analysis was conducted to explore the correlation between environmental factors and benthic community structure. The environmental data used for PCA and BIOENV were water parameters from Copernicus (for the depths 23-27 m) using the QGIS 3.14 software.

Results

Correlation analysis between abiotic and biotic variables

PCA based on environmental variable, including temperature (T), salinity (S), net primary production (NPPV), dissolved oxygen (OD), and depth (metres) at the sampling stations of the shellfish farm (Fig. 2) showed that the first two principal components (PC1 and PC2) explained 71.7% of the total variability. OD, salinity, and NPPV were the main environmental variables that discriminated the ecological and environmental quality of the sampling stations. In February, the stations were influenced by OD and salinity, while in June, the stations were influenced by higher NPPV values.

The BIOENV analysis results are summarised in Table 4. There was a high correlation (R > 0.8) for all the combinations of the abiotic variables, revealing a homogenised environment. The highest rank correlation between multivariate patterns of abiotic and biotic data were obtained with the combination of the five aforementioned abiotic variables.

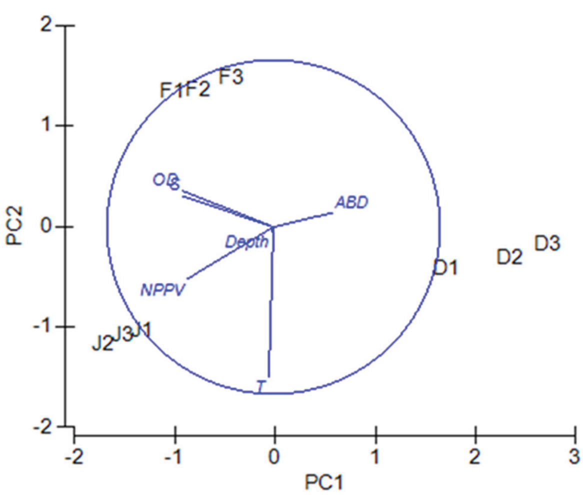

Fig. 2: Two-dimensional principal component analysis (PCA) ordination of the environmental factors of sampling stations. The letters and numbers refer to sampling stations (F = February, D = December, and J = June). Abbreviations: ABD = abundance, NPPV = net primary production, OD = dissolved oxygen, S = salinity, and T = temperature.

Table 4. BIOENV analysis correlating the macrobenthic community with the environmental variables. Abbreviations: NPPV = net primary production, OD = dissolved oxygen, S = salinity, and T = temperature.

Number of variables	Correlation	Selection of variables
5	0.925	T, S, NPPV, OD, depth
4	0.891	S, NPPV, OD, depth
2	0.883	S, NPPV
2	0.883	NPPV, OD
3	0.883	T, S, NPPV
3	0.883	T, NPPV, OD
3	0.883	S, OD, NPPV
4	0.883	T, S, NPPV, OD
3	0.863	NPPV, OD, depth
3	0.850	S, NPPV, depth

Macrobenthic community structure

The taxonomic inventory carried out within the framework of this study allowed to collect a total of 45960 ind/m^2 for the macrobenthos, with 8220 ind/m^2 for D3 and 3495 ind/m^2 for F1. These individuals belong to 138 species divided into four main taxa: Mollusca, Polychaeta, Crustacea, and Echinodermata (Table 5). The minimum number of species occurred in the warm season (61 species at J2), and the maximum number occurred in the cold season (89 species in F1) (Fig. 3).

Quantitative analysis of the systematic structure revealed a particular unbalance in the distribution at the different levels in the cold and warm seasons. Mollusca was the dominant phylum at all stations for the warm and cold seasons, with 1441 individuals for D3 and 241 individuals for D1, followed by Polychaeta and Crustacea (Fig. 4). The qualitative analysis showed the same results, with the dominance of Mollusca with 101 species (72.66%), characterised by the most dominant spe-

Table 5. List of the macrobenthic fauna recorded at the shellfish farm in Ain Chorb (Ain Taya, Algiers). Letters before numbers refer to sampling events (F = February, D = December, and J = June), and the numbers refer to the sampling stations. Abbreviations: Do (%) = dominance; F (%) = frequency; ZG = zoological group (C = Crustacea, CH = Chordata, E = Echinodermata, F = Foraminifera, M = Mollusca, P = Polychaeta); and TG = trophic group (C = carnivores, D = deposit feeders, Dt = detritus feeders, G = grazers, MG = micro- grazers, O = omnivores, and S = suspension feeders).

Species	F1	F2	F3	D1	D2	D3	J1	J2	J3	Do (%)	F (%)	ZG	TG
Abra alba	1	1	1	1	1	1	1	1	1	29.79	100	M	D
Abra tenuis	1	0	1	1	0	0	0	0	0	0.14	33	M	D
Alitta succinea	1	1	1	1	1	1	1	1	1	0.97	100	P	Dt/O
Alvania cancellata	1	0	0	0	1	1	0	0	0	0.14	33	M	G
Alvania dalmatica	0	0	0	0	1	0	0	0	0	0.01	11	M	G
Alvania hispidula	0	0	0	0	0	0	0	0	1	0.02	11	M	G
Alvania pagodula	1	1	1	1	0	1	1	1	0	0.15	78	M	G
Amphipholis sp.	0	0	0	0	0	0	1	0	0	0.01	11	E	
Amphiura chiajei	1	1	0	0	1	0	0	0	0	0.04	33	E	D
Amphiura sp.	0	0	1	0	0	1	0	0	0	0.02	22	E	D
Anomia sp.	0	0	0	0	0	0	1	1	0	0.21	22	M	
Antalis vulgaris	1	1	1	1	1	1	0	0	1	0.27	78	M	D
Arca sp.	0	0	0	0	0	0	1	0	0	0.01	11	M	
Asbjornsenia pygmaea	0	0	0	1	0	0	0	0	0	0.02	11	M	D
Astarte sp.	0	0	0	0	0	0	1	0	0	0.02	11	M	
Astralium armatum	0	0	0	0	0	1	0	0	0	0.01	11	M	
Bittium latreillii	1	1	1	1	1	1	1	1	1	2.25	100	M	
Bittium reticulatum	1	1	1	1	1	1	1	1	1	1.02	100	M	

Continued

Table 5 continued

Species	F1	F2	F3	D1	D2	D3	J1	J2	J3	Do (%)	F (%)	ZG	TG
Bittium submamillatum	1	1	1	1	1	1	1	1	1	3.92	100	M	
Bolinus brandaris	0	0	0	0	0	1	0	1	0	0.02	33	M	D
Brachystomia scalaris	1	1	1	1	0	1	1	1	1	0.52	89	M	
Branchiostoma lanceolatum	1	1	1	1	1	1	1	1	1	1.56	100	CH	S
Caecum glabrum	1	1	1	0	1	1	1	1	1	3.35	89	M	O
Caecum sp. 1	1	1	1	1	1	1	1	1	1	2.30	100	M	O
Caecum sp. 2	1	1	1	1	1	1	1	1	1	4.65	100	M	O
Caecum trachea	1	1	1	1	1	1	1	1	1	1.91	100	M	O
Calliostoma zizyphinum	0	0	0	0	0	0	1	0	0	0.02	11	M	
Calyptraea chinensis	1	1	1	1	1	1	1	1	1	0.78	100	M	G
Capitella capitata	1	0	1	0	0	0	0	0	0	0.02	22	P	D
Caprella linearis	0	1	1	0	0	0	0	0	0	0.13	22	C	
Cerapopsis longipes	1	1	1	0	1	1	0	0	0	0.20	56	C	
Cerithium alucastrum	0	0	0	0	0	0	1	1	1	0.03	33	M	D
Cerithium lividulum	0	0	0	0	0	0	1	0	0	0.01	11	M	D
Chamelea gallina	1	0	1	0	0	1	0	0	0	0.10	33	M	
Chone duneri	1	1	1	1	1	1	0	0	1	0.58	78	P	
Chrysallida excavata	1	1	0	1	0	0	1	1	1	0.11	67	M	
Circulus striatus	1	0	1	1	1	1	0	0	0	0.09	56	M	
Clausinella fasciata	1	0	1	0	0	1	0	0	0	0.04	33	M	
Crisilla semistriata	1	1	1	1	0	1	1	1	1	0.82	89	M	
Crithe alqoensis	0	0	0	1	1	0	0	0	0	0.03	22	M	
Strioturbonilla (*Crysallida*) *segmoidea*	0	0	0	0	0	0	1	0	1	0.02	22	M	
Dentalium sp.	0	0	0	1	0	0	0	0	1	0.04	22	M	D
Derilambrus angulifrons	1	0	1	0	0	0	0	0	0	0.02	22	C	C
Digitaria digitaria	1	0	1	0	1	1	1	1	0	0.17	67	M	
Diodora gibberula	0	0	0	0	0	0	1	0	0	0.01	11	M	
Donax venustus	0	0	1	0	0	0	0	0	0	0.01	11	M	
Dosinia lupinus	0	0	0	1	1	0	1	1	1	0.76	56	M	S
Dosinia sp.	1	1	1	1	0	1	1	1	1	0.53	89	M	S
Ebala pointeli	1	1	1	1	1	1	1	1	1	0.76	100	M	
Elphidium sp.	0	0	0	0	1	0	0	0	0	0.11	11	F	
Eulalia sp.	1	0	1	0	0	1	0	0	0	0.07	33	P	
Eulima sp.	1	1	1	0	1	1	0	0	0	0.25	56	M	
Eulimella scillae	0	0	0	0	0	0	0	0	1	0.02	11	M	
Eupolymnia nebulosa	1	0	1	0	0	0	0	0	0	0.05	22	P	D
Eurydice affinis	1	1	1	1	1	1	1	1	1	0.33	100	C	C
Euspira macilenta	1	0	0	1	1	0	0	1	0	0.11	44	M	C
Exogone naidina	1	1	1	1	1	1	1	1	1	1.22	100	P	C
Gammaropsis sp.	1	1	1	1	1	1	0	0	1	3.61	78	C	
Gari costulata	0	0	0	0	0	1	1	1	1	0.12	44	M	S
Gibberula miliaria	1	1	1	1	1	1	1	1	1	0.25	100	M	
Glycera lapidlium	1	0	0	1	1	1	0	0	0	0.11	44	P	C
Glycymeris glycymeris	0	0	0	0	1	0	0	0	0	0.01	11	M	
Golfingia (*Golfingia*) *elongata*	1	0	1	1	0	0	0	1	1	0.15	56	P	
Goodallia triangularis	1	1	1	1	0	0	1	1	1	0.34	78	M	

Continued

Table 5 continued

Species	F1	F2	F3	D1	D2	D3	J1	J2	J3	Do (%)	F (%)	ZG	TG
Gouldia minima	1	0	0	0	1	0	0	0	0	0.08	22	M	
Haminoea navicular	1	0	1	1	1	0	1	0	0	0.14	56	M	
Haminoea sp.	0	0	0	0	0	0	1	1	1	0.07	33	M	
Hemiliostraca clandestina	0	0	1	0	1	0	0	0	0	0.03	22	M	
Hyalinoecia sp.	0	0	0	0	1	0	0	0	1	0.02	22	P	
Jassa slatteryi	0	0	1	0	1	0	0	0	0	0.03	22	C	
Jassa sp.	1	1	1	1	1	1	0	0	0	0.10	67	C	
Kurtiella bidentata	1	0	0	1	1	1	1	1	1	0.23	78	M	
Lagis koreni	0	0	0	0	0	0	0	0	1	0.07	11	P	D
Lembulus pella	1	0	1	1	1	1	0	1	1	0.12	78	M	D
Leptocheirus guttatus	0	0	0	0	1	0	0	0	0	0.03	11	C	
Leptocheirus longimanus	1	1	1	0	0	1	0	0	0	0.15	44	C	
Leucothoe lilljeborgi	1	0	1	1	1	1	0	0	0	0.22	56	C	
Liocarcinus vernalis	0	0	0	0	0	0	1	0	0	0.02	11	C	C
Loripes orbiculatus	0	0	0	1	0	0	0	0	0	0.01	11	M	S
Lysidice unicornis	1	1	1	0	1	1	1	1	1	0.24	89	P	C
Mangelia payraudeaui	1	1	1	0	1	0	0	0	0	0.07	44	M	
Mangelia unifasciata	1	1	1	0	0	1	1	1	1	0.12	78	M	
Marshallora nigrocincta	0	0	0	0	0	1	0	0	0	0.04	11	M	
Melanella sp.	0	0	0	0	0	0	1	1	1	0.13	33	M	C
Mimachlamys varia	0	0	0	0	0	0	1	0	0	0.01	11	M	
Mitrella svelta	0	0	0	1	0	0	0	0	0	0.01	11	M	
Moerella donacina	1	0	0	0	0	1	1	1	1	0.12	56	M	S
Monophorus perversus	0	0	0	0	0	0	0	0	1	0.03	11	M	
Mystides sp.	1	0	1	1	0	1	0	0	0	0.05	44	P	
Natica sp.	0	0	0	1	0	0	0	0	0	0.01	11	M	
Nucula nitidosa	1	1	1	1	1	1	1	1	1	0.41	100	M	D
Nucula nucleus	1	1	0	1	1	0	0	0	0	0.07	44	M	D
Ocinebrina aciculata	0	0	0	1	1	0	0	0	0	0.02	22	M	
Odostomella doliolum	1	0	1	1	1	1	1	1	1	0.23	89	M	
Odostomia conoidea	0	0	0	0	0	0	1	0	1	0.09	22	M	
Odostomia kromi	1	0	0	1	1	0	0	0	0	0.08	33	M	
Odostomia plicata	1	0	1	0	1	0	0	0	0	0.07	33	M	
Odostomia sp.	1	1	1	0	1	1	1	1	0	0.54	78	M	
Ophiura (*Dictenophiura*) *carnea*	1	1	0	0	0	0	1	1	0	0.09	44	E	D
Ophiura albida	1	1	1	0	0	0	1	0	0	0.07	44	E	D
Paucibranchia bellii	1	1	1	1	1	1	1	1	1	1.14	100	P	C
Papillicardium minimum	1	0	0	1	1	1	0	1	1	0.14	67	M	S
Papyridea soleniformis	0	0	0	0	0	0	1	1	0	0.02	22	M	
Parthenina interstincta	1	1	1	1	1	1	0	0	0	0.88	67	M	
Parthenina terebellum	0	0	0	0	0	0	1	1	1	0.47	33	M	
Parvicardium exiguum	0	0	0	0	0	0	1	0	1	0.05	22	M	S
Parvicardium sp.	1	1	1	0	1	1	1	0	0	0.12	67	M	S
Patella sp.	0	0	0	0	0	0	1	0	0	0.01	11	M	
Phascolion (*Phascolion*) *strombus*	1	1	1	1	1	1	1	1	1	1.27	100	S	D
Phyllodoce sp.	1	1	1	1	0	1	0	0	0	0.13	56	P	C

Continued

Table 5 continued

Species	F1	F2	F3	D1	D2	D3	J1	J2	J3	Do (%)	F (%)	ZG	TG
Pisione remota	1	1	1	1	1	1	1	1	1	0.83	100	P	D
Pitar rudis	0	0	0	0	0	0	0	1	1	0.04	22	M	S
Polyplacophora sp.	1	1	1	0	0	1	1	1	0	0.10	67	M	MG
Protodorvillea keferteini	1	1	1	1	1	1	1	1	1	2.21	100	P	C
Pseudomangelia vauquelini	1	0	0	1	0	0	0	0	0	0.02	22	M	
Pusillina philippi	1	1	1	1	1	1	1	1	1	0.90	100	M	
Pyrgulina stefanisi	1	0	0	0	1	0	0	0	0	0.03	22	M	
Pyrunculus caelatus	1	1	1	1	1	1	1	1	1	1.48	100	M	
Retusa crossei	1	1	1	1	1	1	1	0	1	0.64	89	M	
Retusa mammillata	1	1	1	1	1	1	1	1	1	1.38	100	M	
Retusa miutissima	1	1	1	1	1	1	1	1	1	0.50	100	M	
Ringula conformis	0	1	1	1	1	1	1	1	1	0.15	89	M	
Rissoina sp.	1	1	1	0	0	0	0	0	0	0.08	33	M	
Salvatoria clavata	1	1	1	1	1	1	1	1	1	13.50	100	P	
Scacchia oblonga	0	0	0	0	1	0	0	0	0	0.03	11	M	
Spisula subtruncata	0	0	0	0	0	0	1	1	1	0.39	33	M	S
Stenothoe monoculoides	1	1	1	1	0	1	0	0	0	0.09	56	C	
Syllis lunaris	1	1	0	1	1	1	1	1	1	0.88	89	P	C
Tellimya ferruginosa	1	1	1	1	1	0	1	1	0	0.27	78	M	S
Tricolia gab	1	1	1	0	1	1	0	0	0	0.25	56	M	
Tricolia pullus	1	0	1	1	0	0	0	0	0	0.05	33	M	
Tritia incrassata	1	1	1	1	1	1	1	1	1	0.37	100	M	C
Turbonilla lactea	1	1	1	1	0	1	0	0	0	0.36	56	M	
Turbonilla sp.	1	1	1	0	1	1	0	0	0	0.17	56	M	
Turritella sp.	1	1	1	1	1	1	0	1	1	0.27	89	M	S
Varicorbula gibba	1	1	1	1	1	1	1	1	1	1.46	100	M	S
Vitreolina incurva	1	0	1	0	1	1	0	0	0	0.09	44	M	
Volvulella cylindrica	0	0	0	0	0	1	0	0	0	0.01	11	M	
Total abundance	699	960	1253	701	1319	1644	1051	701	864	**100**			
Species richness	90	65	82	70	75	74	70	61	65				

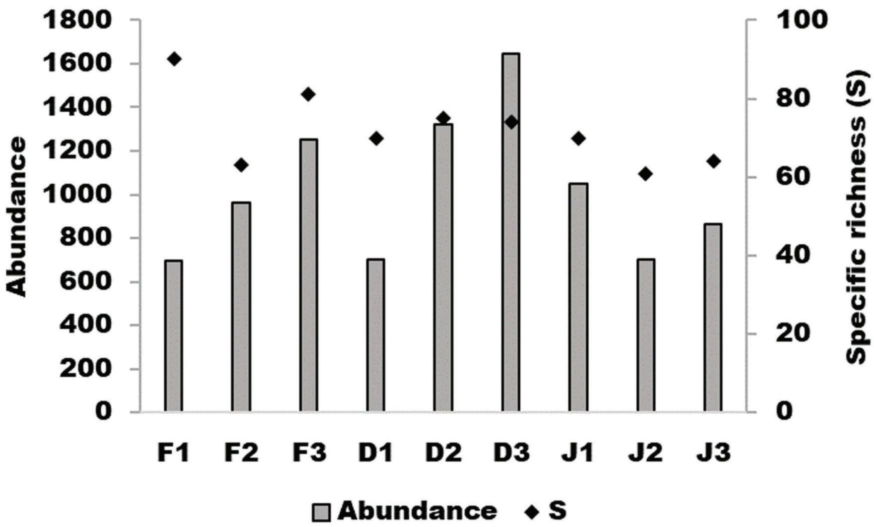

Fig. 3: Temporal and spatial variations in abundance (per 0.2 m^2) and species richness of the macrobenthic fauna at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

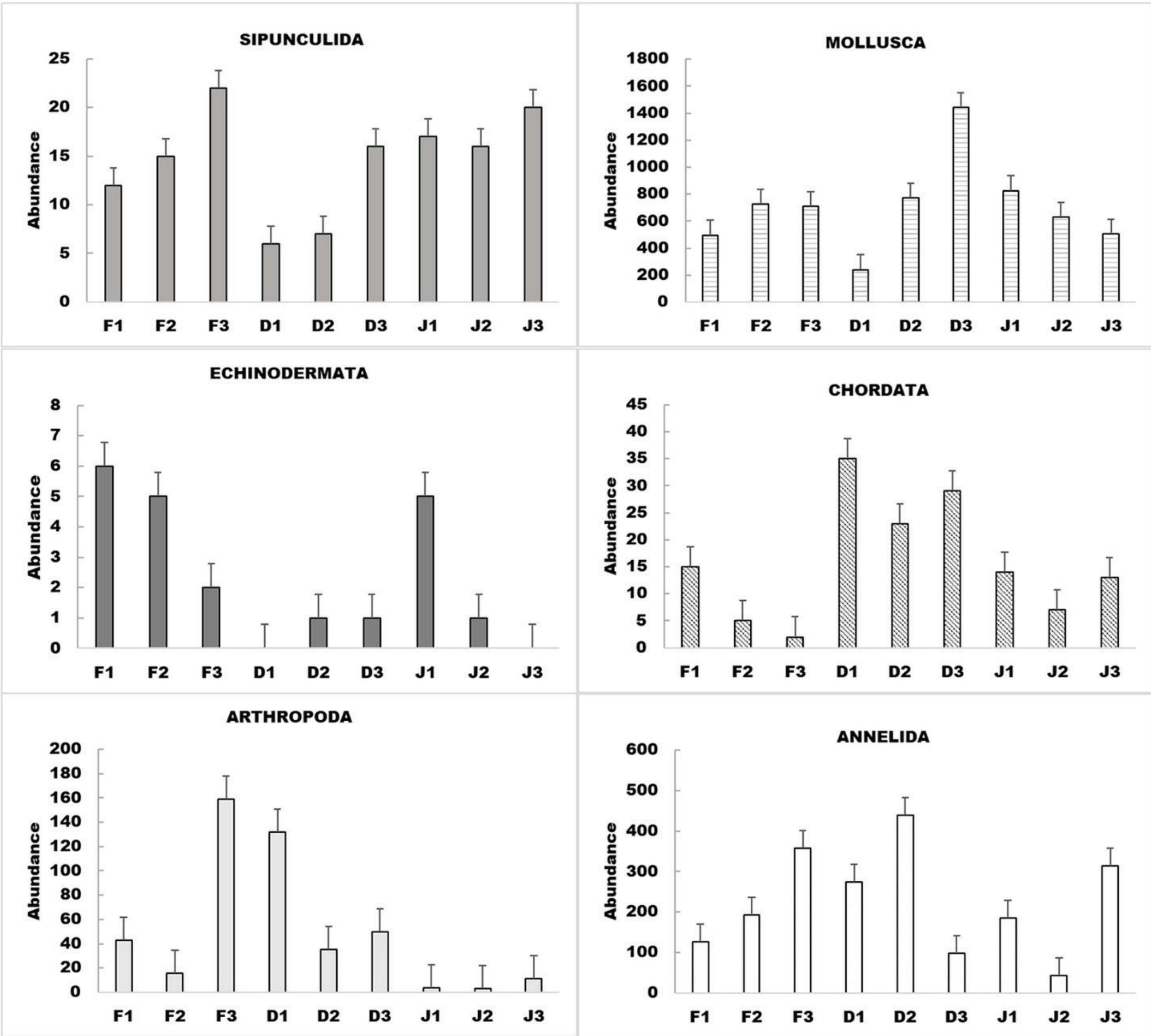

Fig. 4: Temporal and spatial variations in the abundance (per 0.2 m^2) of the macrobenthic fauna at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

cies *Abra alba* (29.79%), followed by Polychaeta with 18 species (12.95%), including the second most dominant species *Salvatoria clavata* (13.50%), and Crustacea with 12 species (8.63%), represented by *Gammaropsis* sp. (3.61%). The remaining groups included eight species (5.76%) (Fig. 5).

Analysis of the species frequencies allowed us to distinguish the distribution of species within the population. The frequent or very frequent species (constant, very common, and common species) included *A. alba*, *S. clavata*, *Caecum* sp. 2, and *Bittium submamillatum*, representing 47.48% of the total diversity. The least frequent species (common and rare) were *Anomia* sp., *Caprella linearis*, *Odostomia conoidea*, and *Gouldia minima*, representing 52% of the total diversity, for a total of 138 species.

For the polluosensitivity ecological groups, following the AMBI scoring list, the benthic community was represented by two major groups for both the cold and warm seasons:

Species very sensitive to organic matter enrichment: this was the dominant group (F1, F3, D1, J1, J2, and J3) represented by >38% of the population. The following molluscs dominated this group: *B. submamillatum*, *Caecum* sp. 2, *Caecum glabrum*, and the amphipod *Gammaropsis* sp.

Species tolerant to organic matter enrichment: this group represented 31.77% (F2, D2, and D3) and was dominated mainly by the mollusc *A. alba*. The tolerant species predominated at station 2 in February and December. The group of indifferent species was in the third position with 27% (Fig. 6).

Three main trophic groups characterised the macrobenthic community of the study area: deposit feeders, dominated by *A. alba*, followed by carnivores and suspension feeders, represented by the polychaetes *Protodorvillea kefersteini* and *Branchiostoma lanceolatum*, respectively. The cold season was characterised by the dominance of deposit feeders (F1, F2, F3, and D1) and carnivores (D2 and D3) (Fig. 7). Quantitative analysis of the trophic structure of the population revealed the preponderance of deposit feeders at all stations (cold and

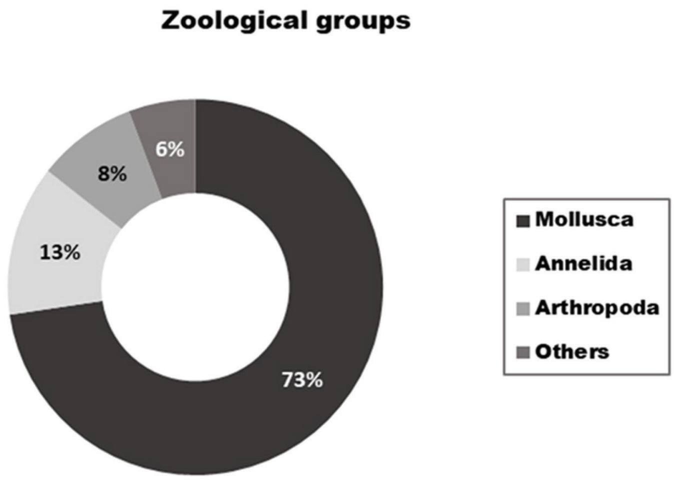

Fig. 5: Dominance of the zoological groups of the macrobenthic fauna at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

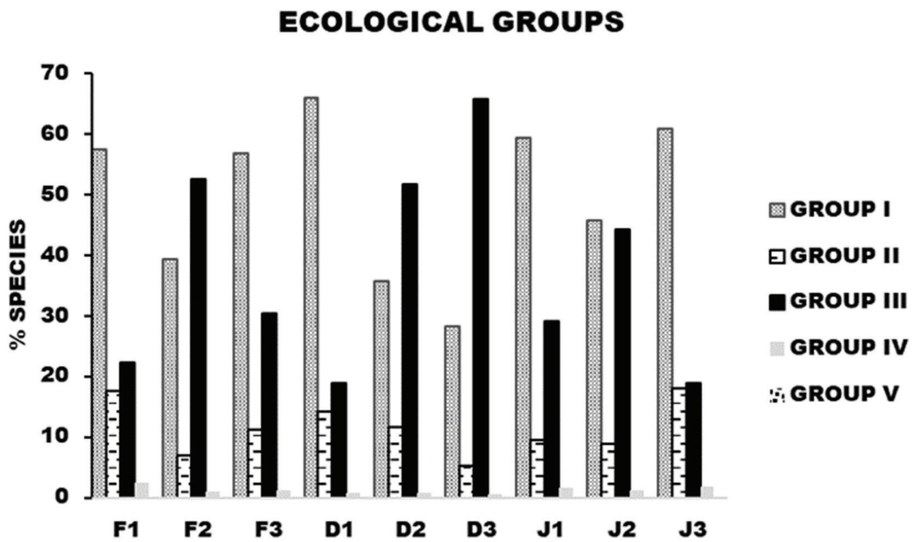

Fig. 6: Temporal and spatial evolution of the ecological groups (I, II, III, IV, and V) according to the AZTI Marine Biotic Index (AMBI) of the soft-bottom macrobenthic fauna at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

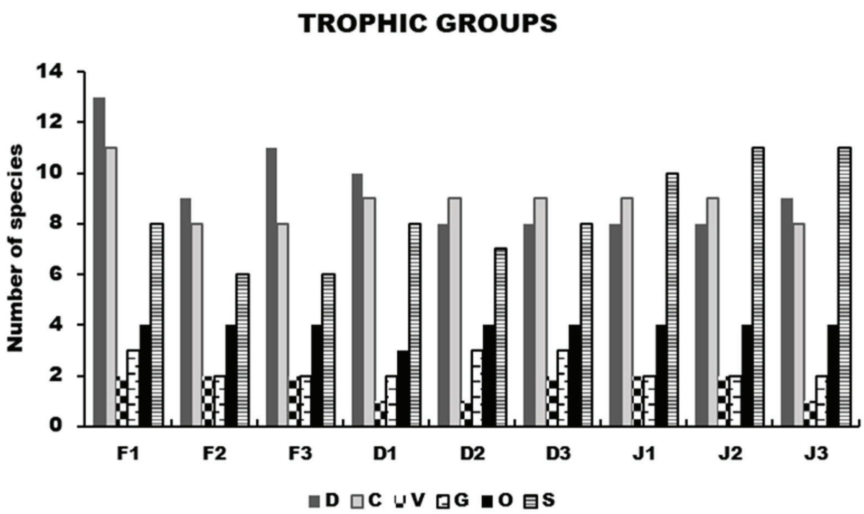

Fig. 7: Temporal and spatial evolution of the trophic groups of the soft-bottom macrobenthic fauna depending on the number of species at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

warm seasons), representing about 61% of the benthic organisms. This result is due to the high predominance of the bivalve *A. alba* (29.79%). This group was followed by omnivores (F1, F2, D2, D3, J1, and J2), dominated by the polychaete *S. clavata* (13.5%), and suspension feeders (F3, D1, and J3) (Fig. 8).

SIMPER analysis showed the six top species lists between the different months for the shellfish farm (the species occurring mainly at each station) according to the average Bray-Curtis dissimilarity (Table 6). The bivalve *A. alba* mainly contributed to the dissimilarities between the months, followed by the polychaete *S. clavata*, occurring mainly in the cold season. The average dissimilarities for samples from February and June were 39.68% and 52.99%, respectively. Bray-Curtis nMDS ordination and cluster analyses of the sites and seasons at the species level showed two different assemblages (A and B) with 70% similarity (Fig. 9). According to SIMPER analysis, *Cerithium alucastrum*,

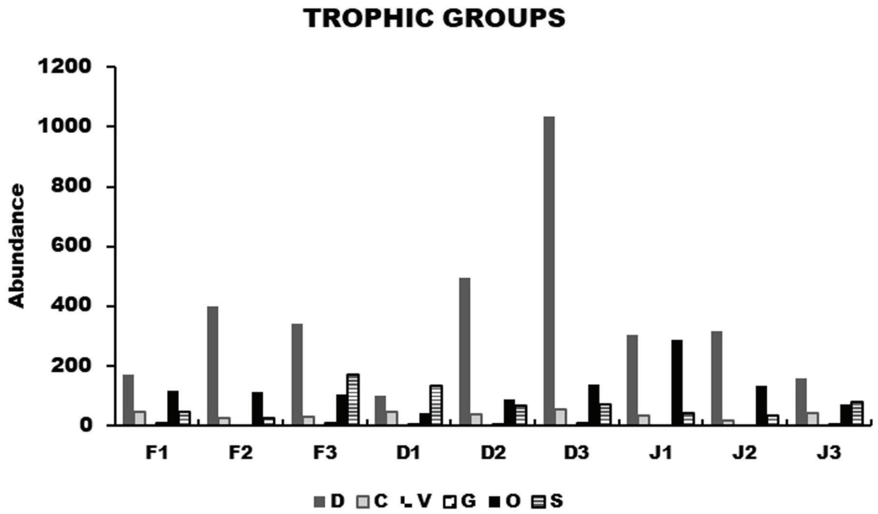

Fig. 8: Temporal and spatial evolution of the trophic groups of the soft-bottom macrobenthic fauna depending on the abundance (per 0.2 m^2) at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

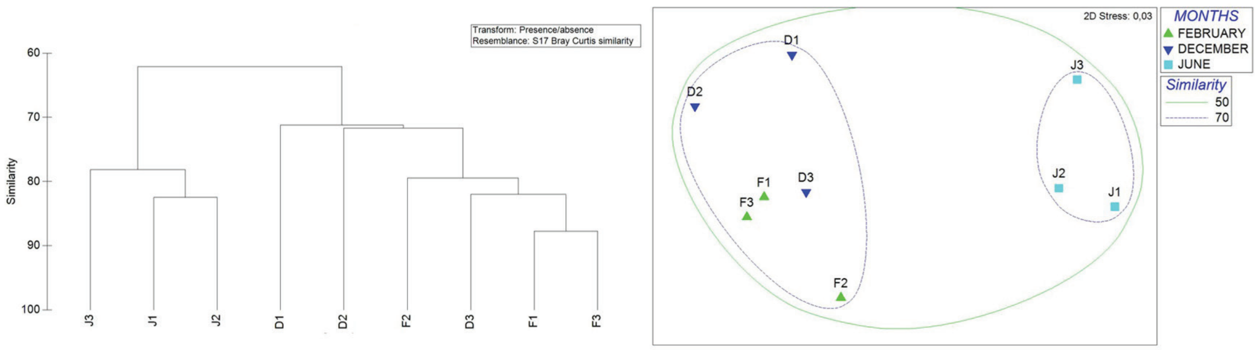

Fig. 9: Bray-Curtis non-metric multidimensional scaling ordination (nMDS) and cluster analyses of the sampling sites.

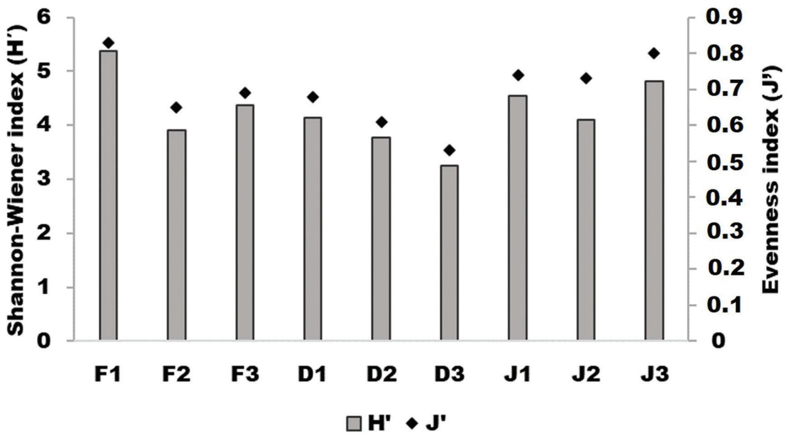

Fig. 10: Temporal and spatial variations in the Shannon-Wiener diversity index (H′) and the evenness index (J’) at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

Table 6. SIMPER analysis showing the species ranked according to the average Bray-Curtis dissimilarity between the different months at the shellfish farm in Ain Chorb (Ain Taya, Algiers). Abbreviations: Abund = abundance, Av. = average, Contrib. = contribution, Cum. = cumulative, Diss. = dissimilarity, and SD = standard deviation.

February & December			**Average diss. = 44.32%**			
	Group February	**Group December**				
Species	**Av. abund.**	**Av. abund.**	**Av. diss.**	**Diss./SD**	**Contrib. (%)**	**Cum. (%)**
Abra alba	247.67	481.33	15.07	1.44	34	34
Salvatoria clavata	136	184.33	7.01	1.32	15.82	49.81
Gammaropsis sp.	51	57	2.84	1.13	6.4	56.22
Caecum sp. 2	50.33	32.33	1.41	1.22	3.18	59.4
Bittium submamillatum	35	48.67	1.17	1.38	2.64	62.04
Branchiostoma lanceolatum	7.33	29	1.04	2.13	2.35	64.39
February & June			**Average diss. = 39.68%**			
	Group February	**Group June**				
Species	**Av. abund.**	**Av. abund**	**Av. diss.**	**Diss./SD**	**Contrib. (%)**	**Cum. (%)**
Abra alba	247.67	183.67	6.48	1.35	16.32	16.32
Salvatoria clavata	136	93.33	5.91	1.39	14.9	31.22
Caecum glabrum	29.33	65	2.65	1.64	6.67	37.9
Gammaropsis sp.	51	2.67	2.42	0.84	6.1	44
Caecum sp. 2	50.33	59.67	1.64	1.49	4.14	48.15
Dosinia lupinus	0	17	0.94	1.05	2.36	50.51
December & June			**Average diss. = 52.99%**			
	Groupe December	**Groupe June**				
Species	**Av. abund.**	**Av. abund.**	**Av. diss.**	**Diss./SD**	**Contrib. (%)**	**Cum. (%)**
Abra alba	481.33	183.67	16.15	1.44	30.48	30.48
Salvatoria clavata	184.33	93.33	7.42	1.31	14.01	44.49
Gammaropsis sp.	57	2.67	3.08	1.05	5.81	50.3
Caecum glabrum	8.33	65	2.88	1.24	5.43	55.74
Caecum sp. 2	32.33	59.67	2.03	1.11	3.83	59.57
Bittium submamillatum	48.67	36.33	1.07	1.46	2.02	61.59

Parthenina interstincta, and *Spisula subtruncata* contributed to this minor dissimilarity (37.93%).

H′ varied between 3.26 and 5.37; The highest H′ value was found in the cold season (F1), and the lowest value was found for D3 (cold season). J′ ranged from 0.53 (December 2020) to 0.83 (February 2020) (Fig. 10). However, these observations relate to trends, as the biodiversity index value for each station of the two periods (cold and warm season) were not significantly different (Kruskal-Wallis test: p = 0.8756). This result indicates that all periods should be regarded similarly in terms of biodiversity.

In February 2020, the mean AMBI value for each sampling station ranged from 1.045 (F1) to 1.725 (F2),

Table 7. Temporal and spatial evolution of the biotic indices (the Shannon-Weaver diversity index [H′], the AZTI Marine Biotic Index [AMBI], the multivariate AZTI Marine Biotic Index [M-AMBI], and the BENTIX) at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

	BENTIX	AMBI	M-AMBI	H'
F1	4.52	1.045	0.99	5.37
F2	3.26	1.725	0.75	3.9
F3	3.55	1.149	0.88	4.37
D1	3.26	0.814	0.85	4.14
D2	2.72	1.769	0.78	3.78
D3	3.34	2.078	0.72	3.26
J1	3.99	1.106	0.86	4.55
J2	3.91	1.515	0.77	4.11
J3	3.57	0.927	0.86	4.81

Table 8. Macrobenthic community parameters at the shellfish farm at the shellfish farm in Ain Chorb (Ain Taya, Algiers). Abbreviations: AMBI = AZTI Marine Biotic Index, H′ = Shannon-Weaver diversity index, J′ = evenness index, and M-AMBI = multivariate AZTI Marine Biotic Index.

	S	N	H′(log) ± SD	Ecological status	J′ ± SD	AMBI ± SD	M-AMBI ± SD	Ecological status	BENTIX ± SD	Ecological status
February	96	2912	4.66 ± 0.75	**Good**	0.71 ± 0.09	1.493 ± 0.366	0.95 ± 0.12	**Good**	3.69 ± 0.66	**Good**
December	107	3664	3.96 ± 0.44	**Moderate**	0.59 ± 0.07	1.782 ± 0.659	0.92 ± 0.06	**Good**	3.10 ± 0.34	**Moderate**
June	86	2617	4.79 ± 0.35	**Good**	0.75 ± 0.05	1.425 ± 0.301	0.93 ± 0.05	**Good**	3.83 ± 0.22	**Good**

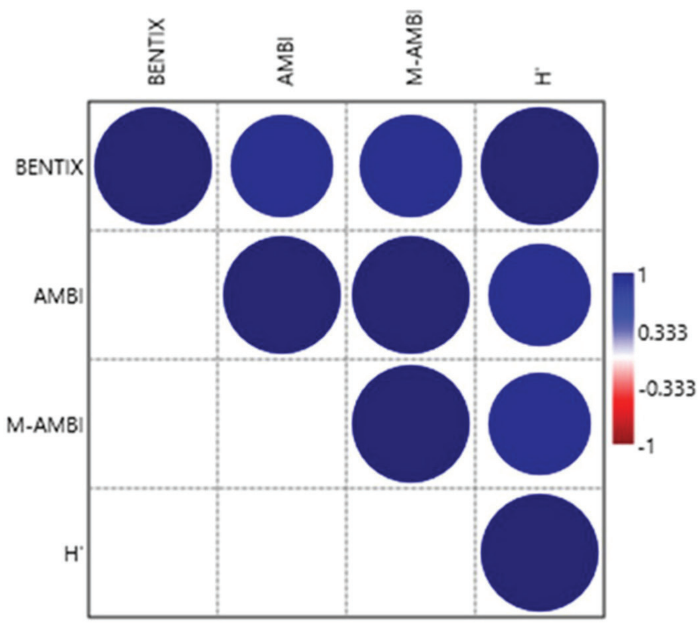

Fig. 11: Correlation matrix for the biotic indices (the Shannon-Weaver diversity index [H′], the AZTI Marine Biotic Index [AMBI], the multivariate AZTI Marine Biotic Index [M-AMBI], and the BENTIX). The blue shaded circles represent significant positive correlations.

with two undisturbed stations (F1 and F3) and only one slightly disturbed station, namely F2 (AMBI = 1.725). The AMBI values show that the benthic environment is undisturbed by human activities.

In December 2020, the mean AMBI values for each sampling station ranged from 0.814 (D1) to 2.078 (D3) (Tables 7 and 8), with two slightly disturbed stations (D2 and D3) and only one undisturbed station D1 (AMBI = 0.814). These data imply that the benthic assemblages had been subjected to only slight disturbances from either environmental changes or human activities.

In June 2021, the mean AMBI values for each sampling station ranged from 0.927 (J3) to 1.515 (J2), with two undisturbed stations (J1 and J3) and only one slightly disturbed station J2 (AMBI = 1.515), implying that the benthic assemblages had been subjected to no or only minor environmental changes or human activities. However, because 67% of the stations had >20% of unassigned species, only 33% of the AMBI results were acceptable. Based on the remaining data, two stations were slightly disturbed and one was undisturbed.

The M-AMBI results revealed 78% of sampling stations with a good ecological status and 22% with a high ecological status (Table 8). The mean M-AMBI values for each month indicated an undisturbed benthic environment.

The BENTIX results varied between sampling month and station. According to this index, the study site was classified from moderate to high. In February and June, the mean BENTIX values varied from 3.69 to 3.83, respectively, which means that the ecological status was good; however, in December all stations had a lower mean BENTIX value of 3.10, indicating a moderate ecological status (Tables 7 and 8). The ecological quality ratio (EQR) values based on the BENTIX showed eight sampling stations with a moderate or good ecological status, and only one station (F1) with a high ecological status.

The correlation matrix (Linear r Pearson) for biotic indices showed a perfect correlation between H′ and the BENTIX ($r = 1$, $p < 0.05$, Fig. 11). The AMBI and the M-AMBI were also significantly correlated to H' and the BENTIX ($r = 0.87$, $p < 0.05$, Fig. 11). In general terms, our results showed that H′, the AMBI, the M-AMBI, and the BENTIX are very close in terms of the diagnosis.

In general, the ABC curves defined the study area as moderately disturbed (Warwick statistic [*W*] = 0.053-0.227). Globally, the abundance curve intersected with the biomass curve except for the month of February (F1 and F3) where the biomass curve was above the abundance curve, classifying the stations as undisturbed (Fig. 12).

The number of geometric abundance classes (Fig. 13) ranged from seven to ten classes throughout all months. In general, the species percentage decreased gradually as the geometric abundance class increased. The curves showed a high number of rare species. There were more species present in low abundance classes than species with high abundance, indicating an unpolluted situation.

Discussion

The variability appeared to be strongly related to the environmental characteristics such as bay exposure to current, water renewal, and sediment type, which seemed to mostly affect trophic behaviour and the quality of aquaculture production. The different analytical methods and biotic indices indicated a moderate to good

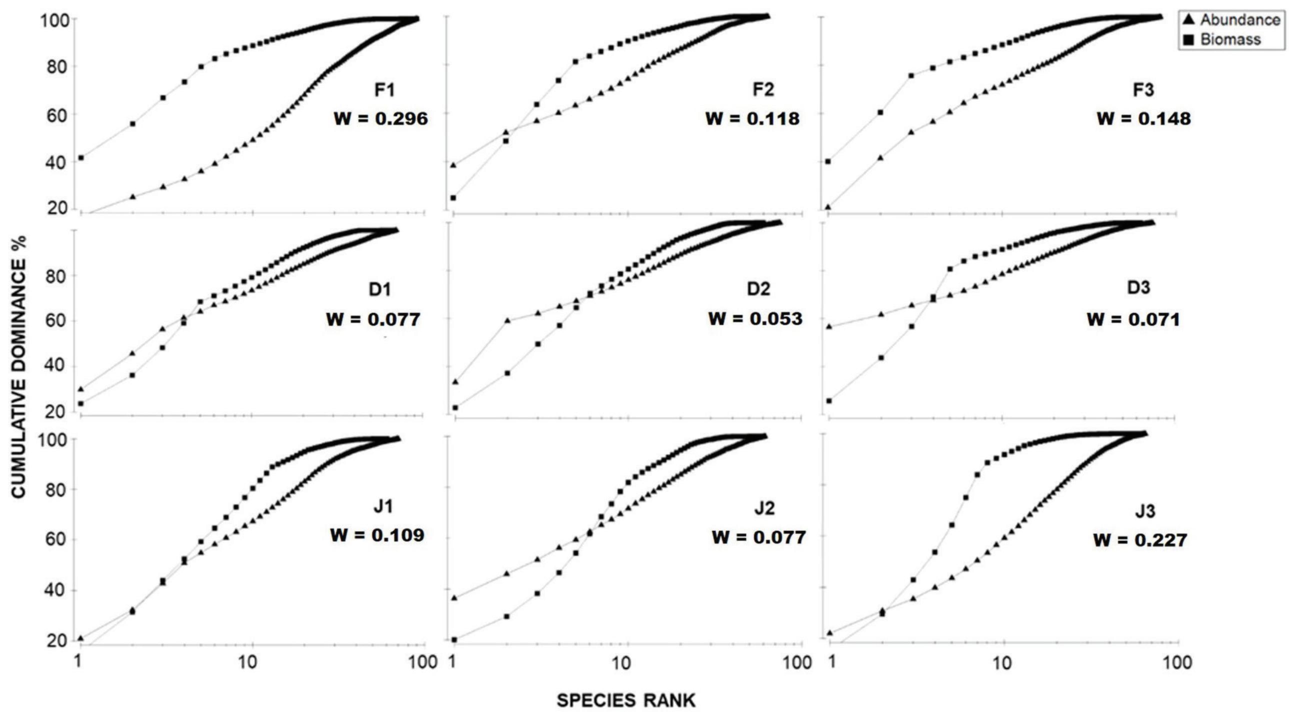

Fig. 12: Abundance/biomass comparison curves for macrobenthic fauna communities analysed at the shellfish farm located in Ain Chorb (Ain Taya, Algiers). *W* is the Warwick statistic.

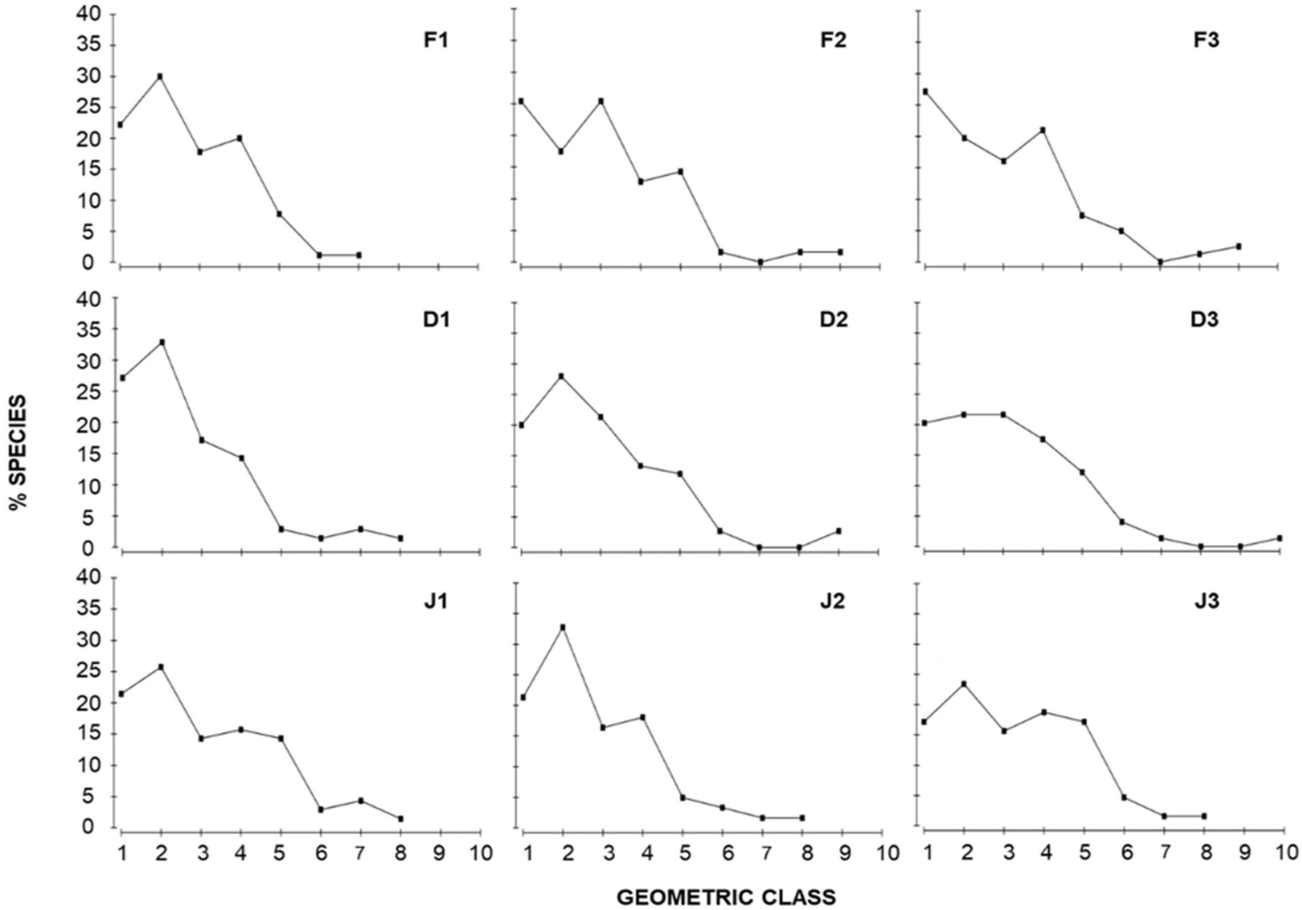

Fig. 13: Distribution of geometric abundance at the shellfish farm in Ain Chorb (Ain Taya, Algiers).

ecological status with a high richness of macrobenthic fauna. The inventory of the macrobenthic fauna of the farm revealed 45960 ind/m^2 divided into four main taxa (Polychaeta, Crustacea, Mollusca, and Echinodermata). The abundance ranged from 3495 to 8220 ind/m^2, which is higher than the abundances that have been observed along the Algerian coast (see Bakalem *et al*., 2009; Belhadj *et al*., 2021; Kerfouf *et al*., 2022), but similar to those observed in the eastern Mediterranean Sea (see Neofitou *et al*., 2012). However, as noted in the literature, this parameter is related to the size of the sampling surface, the relationship between the sampled area and the number of species recorded (Carpentier & Lepretre, 1999), and the study area characteristics (substratum, the influence of anthropogenic discharges, etc.).

During the survey, molluscs and polychaetas were the most abundant taxa (72.66% and 12.95%, respectively), followed by crustaceans (Arthropoda, 8.63%). A previous study on the macrobenthic fauna in the Mediterranean Sea (Bay of Bou-Ismail, Algeria) showed a different structure with the dominance of polychaetes (44.7%), followed by crustaceans, and then echinoderms (7.6%) (Bakalem *et al*., 2020). There were similar findings in the eastern sector of the Gulf of Oran (western Algeria, Mediterranean Sea) (Kerfouf *et al*., 2022). According to Zerrouki (2012), the ecosystem in this area of eastern Algeria, particularly in the infralittoral zone, has traditionally been highly reputed for its abundance of molluscs. The bivalve *A. alba*, one of the main dominant species in our study site, is considered an indicator of instability and of excessive organic matter. This species can be present in normal conditions, but its population is also stimulated by organic enrichment (Rebzani-Zahaf, 2003). Its ability to rapidly colonise the available sandy benthic communities and to exploit the food resources present at the water-sediment interface makes it a pioneer species (Dauvin *et al*., 1993); this ability probably explains the high densities observed in this study site, ranging from 1020 to 2674 ind/m². In addition, according to Dauvin *et al*. (1993), *A. alba* shows remarkable pluriannual variation with strong (500-1000 ind/m²) or very strong (>1000 ind/m²) recruitment from November to December. The high densities of this species in December (2674 ind/m²) confirm his recruitment pathway during the study period.

Analysis of the species frequencies revealed that the qualitative structure of the population of the surveyed zone is formed by a reduced number of frequent species, around which a significant number of less frequent species revolves. We may say that the settlement structure presents a state of imbalance, mainly in favour of rare species. These findings have been confirmed by the geometric abundance class method. The significant number of mollusc species among the uncommon and rare assemblage species highlights that this group plays an important role in the qualitative structure.

Dominant species can characterise a community based on their functional groups, which are defined as species with similar effects on the significant ecosystem processes (Chapin *et al*., 1992). Trophic functional groups play different roles in the benthic ecosystem, including transformation and decomposition of organic and inorganic matter inside the sediment due to their bioturbation – for example, feeding, burrowing, and construction activities (Aller, 1994; 2001). The suspension feeder functional group may induce facilitative interactions and enhance resource consumption (Cardinale *et al*., 2002), which will be beneficial to other groups. Other functional groups (mainly depositional food habit and burrowing behaviour) decompose the detritus, increase the oxygen sediment porewater, and accelerate the decomposition of organic matter (Pearson *et al*., 2001). The top three dominant species at the study site belong to the deposit feeder group, including *A. alba* (also considered a suspension feeder). This group competes with suspension feeders and omnivores (the second most represented group), represented by the polychaete *S. clavata* and the mollusc *C. glabrum*. According to Dewarumez & Blond (1983) and García-Arberas & Rallo (2002), deposit feeders are generally abundant in muddy and muddy sand, low-energy sediments, while suspension feeders prevail in sediments with a low content of fine fractions. The deposit feeders and omnivores were of equal numerical importance: they represented more than half of the macrobenthic population at the shellfish farming stations. An increase in deposit feeders is usually reported in areas subjected to eutrophication and indicates a disturbed environment (Beukema, 1991), but they also play a crucial role in organic matter decomposition and nutrient recycling for the benthic ecosystem (Li *et al*., 2017). However, as outlined by Gray *et al*. (1988) and García-Arberas & Rallo (2002), the utility of feeding group approaches in detecting the effects of pollution is limited because the separation of pollution gradients from natural gradients in estuarine and coastal environments is not evident. The lack of specific knowledge about diets can lead to errors in species grouping. For example, most deposit feeders can also be potential carnivores as indiscriminate predators of juvenile populations or the meiobenthos.

The macrobenthic communities of the studied shellfish farm showed high species richness and were dominated by small size organisms. According to Deslous-Paoli *et al*. (1998), the development of these small individuals is probably due to shellfish farming activity, which acts favourably on the productivity of the ecosystem by reinjecting massive and rapid amounts of nutrients into the water column, even though its predation pressure rapidly eliminates the products. These results can be correlated with a type ‘r’ demographic strategy characterised by short longevity, strong growth, a short development cycle, and several annual egg-laying events. According to the ABC curves, ‘k’ strategy species are progressively replaced by a small number of ‘r’ strategy species, such as *A. alba*, the dominant species in our study (29.79%). This could explain the importance of juveniles in the communities of the aquaculture farm.

Structure analysis indicated the same structure of macrobenthic fauna at all stations for the warm and cold seasons. According to the Kruskal-Wallis test, there were no significant changes ($p > 0.05$). However, we noticed a slightly imbalanced ecological status in December (decrease in H′ and J′) due to the increase in *A. alba* abundance. The ABC curves produced similar results, showing a moderately disturbed and undisturbed pattern.

In this survey, the four benthic and biotic indices (H′, the AMBI, the M-AMBI, and the BENTIX) suggested that the sampling stations of the shellfish farm are of moderate and good quality. Bakalem *et al*. (2009) reported slightly different results for Algiers Bay; the same benthic indices indicated a moderate and high ecological status. The results of the biotic indices confirm their substantial capacity to analyse the ecological quality of shellfish farms and to monitor their impact on the macrobenthic assemblages. The AMBI and the M-AMBI indicated the same ecological status (good) for all sampling periods. On the other hand, H′ and the BENTIX varied among the months, with a good ecological status in February and June and a moderate ecological status in December. The average values of the various biotic indices, based on benthic macrofauna, gave similar values and ecological status (moderate), except for the AMBI and the M-AMBI. The correlation matrix showed significant correlations between the four biotic indices. Finally, the EQR score for the current survey (0.55) was similar to the EQR scores reported by Chabane *et al*. (2018) (0.41) and Bahbah *et al*. (2020) (0.49), who used macroalgal assemblages that are known to be good indicators of water quality (Ballesteros *et al*., 1984; Bishop *et al*., 2002; Arévalo *et al*., 2007; Orlando-Bonaca *et al*., 2008; Lasinio *et al*., 2017).

Conclusion

This preliminary study constitutes the first inventory, representing a reference state, of the shellfish farm located in Ain Chorb (Ain Taya), in the central region of Algeria. The macrobenthic assemblage was dominated by molluscs and polychaetas. The macrobenthic assemblage is an excellent indicator of ecological status; for environmental management, it would be necessary to implement long-term monitoring of the farms located along the Algerian coast. Additional developments considering abiotic and biotic interactions are needed for a more complete environmental evaluation, for better aquaculture production, and the development of an environmentally responsible aquaculture.

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References

Albano, P.G., Azzarone, M., Amati, B., Bogi, C., Sabelli, B. *et al.,* 2020. Low diversity or poorly explored? Mesophotic molluscs highlight undersampling in the Eastern Mediterranean. *Biodiversity and Conservation,* 29(14), 4059-4072.

Aller, R.C., 1994. Bioturbation and remineralization of sedimentary organic matter: effects of redox oscillation. *Chemical Geology*, 114 (3-4), 331-345.

Aller, R.C., 2001. Transport and reactions in the bioirrigated zone. *The Benthic Boundary Layer: Transport Processes and Biogeochemistry*, 269 pp.

Alvarez-Campos, P., Riesgo, A., Hutchings, P., San Martín, G., 2015. The genus Syllis Savigny in Lamarck, 1818 (Annelida, Syllidae) from Australia. Molecular analysis and re-description of some poorly-known species. *Zootaxa*, 4052 (2), 297-331.

Antit, M., Azzouna, A., 2012. Mollusques des milieu littoraux de la baie de Tunis. *Iberus*, 30 (2), 107-133.

Arévalo, R., Pinedo, S., Ballesteros, E., 2007. Changes in the composition and structure of Mediterranean rocky-shore communities following a gradient of nutrient enrichment: descriptive study and test of proposed methods to assess water quality regarding macroalgae. *Marine Pollution Bulletin*, 55 (1-6), 104-113.

Bahbah, L., Bensari, B., Chabane, K., Torras, X., Ballesteros, E. *et al*., 2020. Cartography of littoral rocky-shore communities to assess the ecological status of water bodies through the application of CARLIT method in Algeria (South-Western Mediterranean Sea). *Marine Pollution Bulletin*, 157 (May), 111356.

Bakalem, A., Romano, J.C., 1979. Etude de la dynamique des peuplements benthiques de la Baie d'Alger-Resultats preliminaires.

Bakalem, A., Ruellet, T., Dauvin, J.C., 2009. Benthic indices and ecological quality of shallow Algeria fine sand community. *Ecological Indicators*, 9 (3), 395-408.

Bakalem, A., Hassam, N., Oulmi, Y., Martinez, M., Dauvin, J., 2020. Diversity and geographical distribution of soft-bottom macrobenthos in the bay of Bou Ismail (Algeria, Mediterranean Sea). *Regional Studies in Marine Science*, 33, 100938.

Ballesteros, E., Pérez, M. M., Zabala, M., 1984. Aproximación al conocimiento de las comunidades algales de la zona infralitoral superior en la costa catalana. Collectanea Botanica, 15, 69-100.

Barton, J.R., Fløysand, A., 2010. The political ecology of Chilean salmon aquaculture, 1982-2010: a trajectory from economic development to global sustainability. *Global Environmental Change*, 20 (4), 739-752.

Belhadj, K.T., Baaloudj, A., Kerfouf, A., Benallal, M., 2021. Macrobenthic fauna of the coastal sea beds of Oran's Gulf (Western Algeria). *Ukrainian Journal of Ecology*, 11(4), 1-4.

Beukema, J.J., 1991. Changes in composition of bottom fauna of a tidal-flat area during a period of eutrophication. *Marine Biology*, 111 (2), 293-301.

Bishop, M.J., Underwood, A.J., Archambault, P., 2002. Sewage and environmental impacts on rocky shores: necessity of identifying relevant spatial scales. *Marine Ecology Progress Series*, 236, 121-128.

Borja, A., Franco, J., Pérez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. *Marine Pollution Bulletin*, 40 (12), 1100-1114.

Burke, R.N.M., 2005. Raising tigers of the sea. *Annual Review of Environment and Resources*, 30, 185-218.

Cardinale, B.J., Palmer, M.A., Collins, S.L., 2002. Species diversity enhances ecosystem functioning through interspecific facilitation. *Nature*, 415 (6870), 426-429.

Carpentier, P., Lepretre, A., 1999. Robustness of some diversity indices to sampling. *Océanis*, 25 (3), 435-456.

CAR/ASP - PNUE/PAM, 2015. *Bioecological characterization of the marine area of eastern Algeria (Algeria)*. CAR/ASP - ProjetMedMPAnet, Tunisia, 55 pp.

Chabane, K., Bahbah, L., Seridi, H., 2018. Ecological quality status of the Algiers coastal waters by using macroalgae assemblages as bioindicators (Algeria, Mediterranean Sea). *Mediterranean Marine Science*, 19 (2), 305-315.

Chapin III, F.S., Schulze, E.D., Mooney, H.A., 1992. Biodiversity and ecosystem processes. *Trends in Ecology and Evolution*, 7 (4), 107-108.

Chevillon, C., 1999. Influence of discharges from an aquaculture farm on the coastal environment. "The Peneid of Ouano". *Sedimentology*. Final report.

Christensen, P.B., Glud, R.N., Dalsgaard, T., Gillespie, P., 2003. Impacts of longline mussel farming on oxygen and nitrogen dynamics and biological communities of coastal sediments. *Aquaculture*, 218 (1-4), 567-588.

Dauer, D.M., 1993. Biological criteria, environmental health and estuarine macrobenthic community structure. *Marine Pollution Bulletin*, 26 (5), 249-257.

Dauvin, J., Dewarulez, J., Elkaim, B., Bernado, D., Fromentin, J. *et al*., 1993. Cinetics of *Abra alba* (bivalve mollusc) from 1977 to 1991 in the Channel-North Sea, relationship with climatic factors. *Oceanologca Acta*, 16 (4), 413-422.

Deslous-Paoli, J.M., Souchu, P., Mazouni, N., Juge, C., Dagault, F., 1998. Environment-resource relationships: Impact of shellfish farming on a Mediterranean lagoon environment (Thau). *Oceanologica Acta*, 21 (6), 831-843.

De Souza, L.S., Pimenta, A.D., 2019. Taxonomy of littoral Melanella (Gastropoda: Eulimidae) from Brazil, with comments on the Eulima described by Verrill and Bush (1900). *Marine Biodiversity*, 49, 425-442.

Dewalt, B.R., Vergne, P., Hardin, M., 1996. Shrimp aquaculture development and the environment: people, mangroves and fisheries on the Gulf of Fonseca, Honduras. *World Development*, 24 (7), 1193-1208.

Dewarumez, J.M., Blond, J., 1983. suspended mussel farming in the nord-pas-de-calais region (France): first results. *Journal of Molluscan Studies*, 49 (supp12a), 36-42.

Edgar, G.J., Samson, C.R., Barrett, N.S., 2005. Species extinction in the marine environment: Tasmania as a regional example of overlooked losses in biodiversity. *Conservation Biology*, 19 (4), 1294-1300.

Farmaki, E.G., Thomaidis, N.S., Pasias, I.N., Baulard, C., Papaharisis, L. *et al*., 2014. Environmental impact of intensive aquaculture: Investigation on the accumulation of metals and nutrients in marine sediments of Greece. *Science of the Total Environment*, 485, 554-562.

Food and Agriculture Organization, 2020. *The state of world fisheries and aquaculture 2020. Sustainability in action*. Food and Agriculture Organization, Rome.

Froehlich, H.E., Smith, A., Gentry, R.R., Halpern, B.S., 2017. Offshore aquaculture: I know it when I see it. *Frontiers in Marine Science*, 4, 154.

García-Arberas, L., Rallo, A., 2002. The intertidal soft-bottom infaunal macrobenthos in three Basque estuaries (Gulf of Biscay): a feeding guild approach. p. 457-468. In: *Nutrients and eutrophication in estuaries and coastal waters*. Orive, E., Elliott, M., de Jonge, V.N. (Eds.). Springer, Dordrecht.

Gray, J.S., Aschan, M., Carr, M.R., Clarke, K R., Green, R.H., *et al*., 1988. Analysis of community attributes of the benthic macrofauna of Frierfjord/Langesundfjord and in a mesocosm experiment. *Marine Ecology Progress Series*, 46, 151-165.

Gray, J.S., Mirza, F.B., 1979. A possible method for the detection of pollution-induced disturbance on marine benthic communities. *Marine Pollution Bulletin*, 10 (5), 142-146.

Gu, Y.G., Wang, Y., Ouyang, J., Jordan, R.W., Jiang, S., 2021. Impacts of coastal aquaculture on sedimentary phosphorus speciation and fate: Evidence from a seaweed cultivation area off Nan'ao Island, South China. *Marine Pollution Bulletin*, 171, 112719.

Hatcher, A., Grant, J., Schofield, B., 1994. Effects of suspended mussel culture (*Mytilus spp*.) on sedimentation, benthic respiration and sediment nutrient dynamics in a coastal bay. *Marine Ecology Progress Series*, 115, 219.

Hily, C., 1984. *Variability of the benthic macrofauna in the hyper-trophic environments of the Brest roadstead*. PhD Thesis. University of Western Brittany, France, 696 pp.

Hindar, K., Ryman, N., Utter, F., 1991. Genetic effects of cultured fish on natural fish populations. Canadian *Journal of Fisheries and Aquatic Sciences*, 48 (5), 945-957.

Høisæter, T., 2014. The Pyramidellidae (Gastropoda, Heterobranchia) of Norway and adjacent waters. A taxonomic review. *Fauna Norvegica*, 34, 7-78.

Hyland, J., Balthis, L., Karakassis, I., Magni, P., Petrov, A. *et al*., 2005. Organic carbon content of sediments as an indicator of stress in the marine benthos. *Marine Ecology Progress Series*, 295, 91-103.

Islam, M.S., 2005. Nitrogen and phosphorus budget in coastal and marine cage aquaculture and impacts of effluent loading on ecosystem: review and analysis towards model development. *Marine Pollution Bulletin*, 50 (1), 48-61.

Johansen, L.H., Jensen, I., Mikkelsen, H., Bjørn, P.A., Jansen, P.A. *et al.*, 2011. Disease interaction and pathogens exchange between wild and farmed fish populations with special reference to Norway. *Aquaculture*, 315 (3-4), 167-186.

Karakassis, I., Tsapakis, M., Hatziyanni, E., Papadopoulou, K.N., Plaiti, W., 2000. Impact of cage farming of fish on the seabed in three Mediterranean coastal areas. *ICES Journal of Marine Science*, 57 (5), 1462-1471.

Kerfouf, A. Patricio, R., Tahar, K.B., Denis, F., 2022. Distribution of soft-bottom benthic molluscs in the Gulf of Oran (Western Algeria). *Aquatic Research*, 5 (3), 209-218.

Kim, J., Kim, Y., Park, S.E., Kim, T.H., Kim, B.G. *et al.*, 2022. Impact of aquaculture on distribution of dissolved organic matter in coastal Jeju Island, Korea, based on absorption and fluorescence spectroscopy. *Environmental Science and Pollution Research*, 29 (1), 553-563.

Klaoudatos, S.D., Klaoudatos, D.S., Smith, J., Bogdanos, K., Papageorgiou, E., 2006. Assessment of site specific benthic impact of floating cage farming in the eastern Hios island, Eastern Aegean Sea, Greece. *Journal of Experimental Marine Biology and Ecology*, 338 (1), 96-111.

Klinger, D., Naylor, R., 2012. Searching for solutions in aquaculture: charting a sustainable course. *Annual Review of Environment and Resources*, 37 (1), 247-276.

Kucuksezgin, F., Pazi, I., Gonul, L.T., 2022. Environmental impact of fish farming: assessment of metal contamination and sediment geochemistry at three aquaculture areas from the eastern Aegean coast. *Environmental Monitoring and Assessment*, 194 (4), 313.

Lasinio, G.J., Tullio, M.A., Ventura, D., Ardizzone, G., Abdelahad, N., 2017. Statistical analysis of the distribution of infralittoral Cystoseira populations on pristine coasts of four Tyrrhenian islands: proposed adjustment to the CARLIT index. *Ecological Indicators*, 73, 293-301.

Li, B., Li, X., Bouma, T.J., Soissons, L.M., Cozzoli, F. *et al*., 2017. Analysis of macrobenthic assemblages and ecological health of Yellow River Delta, China, using AMBI & M-AMBI assessment method. *Marine Pollution Bulletin*, 119(2), 23-32.

Lounas, R., Kasmi, H., Chernai, S., Amarni, N., Hamdi, B., 2020. Assessing the contamination and ecological risk of sediments around the mariculture area in northern Algeria using a multivariate approach. *Journal of Applied Aquaculture*, 34 (1), 147-163.

Lourguioui, H., Brigolin, D., Boulahdid, M., Pastres, R., 2017. A perspective for reducing environmental impacts of mussel culture in Algeria. *The International Journal of Life Cycle Assessment*, 22, 1266-1277.

Mazzola, A., Mirto, S., La Rosa, T., Fabiano, M., Danovaro, R., 2000. Fish-farming effects on benthic community structure in coastal sediments: analysis of meiofaunal recovery. *ICES Journal of Marine Science*, 57 (5), 1454-1461.

Middelfart, P.U., Kirkendale, L.A., Wilson, N.G., 2016. Australian tropical marine micromolluscs: an overwhelming bias. *Diversity*, 8 (3), 17.

Molina-Acevedo, I.C., Idris, I., 2020. Reinstatement of species belonging Marphysa sanguinea complex (Annelida: Eunicidae) and description of new species from the mid-Pacific Ocean and the Adriatic Sea. *Zootaxa*, 4816 (1), 1-48.

Moraitis, M., Papageorgiou, N., Dimitriou, P.D., Petrou, A., Karakassis, I., 2013. Effects of offshore tuna farming on benthic assemblages in the eastern mediterranean. *Aquaculture Environment Interactions*, 4 (1), 41-51.

Muxika, I., Borja, A., Bald, J., 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. *Marine Pollution Bulletin*, 55, 16-29

Nanou, G., Peter, B., Elisabeth, D., Klaas, D., Thibaud, M. *et al*., 2022. Nature-based solutions in a sandy foreshore: A biological assessment of a longline mussel aquaculture technique to establish subtidal reefs. *Ecological Engineering*, 185, 106807.

Naylor, R., Hindar, K., Fleming, I.A., Goldburg, R., Williams, S. *et al*., 2005. Fugitive salmon: assessing the risks of escaped fish from net-pen aquaculture. *BioScience*, 55 (5), 427-437.

Neofitou, N., Vafidis, D., Klaoudatos, S., 2010. Spatial and temporal effects of fish farming on benthic community structure in a semi-enclosed gulf of the Eastern Mediterranean. *Aquaculture Environment Interactions*, 1 (2), 95-105.

Newell, R.I.E., 2004. Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: a review. *Journal of Shellfish Research*, 23 (1), 51-62.

Orlando-Bonaca, M., Lipej, L., Orfanidis, S., 2008. Benthic macrophytes as a tool for delineating, monitoring and assessing ecological status: the case of Slovenian coastal waters. *Marine Pollution Bulletin*, 56 (4), 666-676.

Patrício, J., Elliott, M., Mazik, K., Papadopoulou, K.N., Smith, C.J., 2016. DPSIR-two decades of trying to develop a unifying framework for marine environmental management? *Frontiers in Marine Science*, 3, 177.

Pearson, T.H., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology Annual Review*, 16, 229-311.

Pearson, V.K., Sephton, M.A., Bland, P.A., Franchi, I.A., Gilmour, I., 2001. The association between organic matter and clay minerals in carbonaceous chondrites. *Meteoritics and Planetary Science Supplement*, 36, A159.

Pérès, J.M., Picard, J., 1965. Nouveau manuel de bionomie benthique de la Mer Méditerranée. *Station maritime d'Endoume*, 31 (57), 1-147.

Phillips, D.J.H., Rainbow, P.S., 1994. *Biomonitoring of trace aquatic contaminants* (2nd ed.). Chapman and Hall, London, 299 pp.

Pielou, E.C., 1966. The measurement of diversity in different types of biological collections. *Journal of theoretical biology*, 13, 131-144.

Rebzani-Zahaf, C., 1992. *Le peuplement macrobenthique du port d'Alger: impact de la pollution. Hydroécologie Appliquée*, 4 (2), 91-103.

Rebzani-Zahaf, C., 2003. *The macrobenthic populations of the port environments of the Algerian coast: Algiers, Bejaia and Skikda*. PhD Thesis, University of Sciences and Technology, Houari Boumediene, Faculty of Biological Sciences, Algeria, 251 pp. + appendices 72 pp.

Rebzani-Zahaf, C., Bellan, G., Romano, J.C., 1997. Macrobenthic communities in Algiers harbour. *Oceanologica Acta*, 20 (2), 461-103.

Rios-Fuster, B., Alomar, C., Capó, X., Paniagua González, G., Garcinuño Martínez, R.M. *et al*., 2022. Assessment of the impact of aquaculture facilities on transplanted mussels (*Mytilus galloprovincialis*): integrating plasticizers and physiological analyses as a biomonitoring strategy. *Journal of Hazardous Materials*, 424, 127264.

Rosenberg, R., Blomqvist, M., Nilsson, H C., Cederwall, H., Dimming, A., 2004. Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive. *Marine Pollution Bulletin*, 49 (9-10), 728-739.

Samson-Kechacha, F.L., Hallal, O., Helis, L., 1992. Caractéristiques phytoplanctoniques de deux stations côtières de l'Est Algérois. *Hydroécologie Appliquée*, 4 (2), 123-138.

Sanchis, C., Soto, E.H., Quiroga, E., 2021. The importance of a functional approach on benthic communities for aquaculture environmental assessment: trophic groups – a polychaete view. *Marine Pollution Bulletin*, 167 (April), 112309.

Santos, C.S., Lana, P.D.C., 2003. Nereididae (Polychaeta) da costa nordeste do Brasil: III. Gêneros Ceratonereis e Nereis. *Iheringia. Série Zoologia*, 93, 5-22.

Shannon, C.E., Weaver, W., 1963. *A mathematical theory of communication*. Univ. Illi- nois Press, Urbana. 117pp.

Simboura, N., Zenetos, A., 2002. Benthic indicators to use in ecological quality classification of Mediterranean soft bottom marine ecosystems, including a new biotic index. *Mediterranean Marine Science*, 3 (2), 77-111.

Sutherland, T.F., Petersen, S.A., Levings, C.D., Martin, A.J., 2007. Distinguishing between natural and aquaculture-derived sediment concentrations of heavy metals in the Broughton Archipelago, British Columbia. *Marine Pollution Bulletin*, 54 (9), 1451-1460.

Tičina, V., Katavić, I., Grubišić, L., 2020. Marine aquaculture impacts on marine biota in oligotrophic environments of the Mediterranean sea-a review. *Frontiers in Marine Science*, 7, 217.

Vannozzi, A., Pizzini, M., Raines, B., 2015. Revision of South African Caecidae (Mollusca: Gastropoda). *African Invertebrates*, 56 (1), 99-136.

Warwick, R.M., Clarke, K.R., 1994. Relearning the ABC: taxonomic changes and abundance/biomass relationships in disturbed benthic communities. *Marine Biology*, 118 (4), 739-744.

Warwick, R.M., 1986. A new method for detecting pollution effects on marine macrobenthic communities. *Marine biology*, 92, 557-562.

WoRMS Editorial Board, 2023. *World Register of Marine Species*. http://www.marinespecies.org (accessed 6 July 2023).

Wu, R.S.S., Lam, K.S., MacKay, D.W., Lau, T.C., Yam, V., 1994. Impact of marine fish farming on water quality and bottom sediment: a case study in the sub-tropical environment. *Marine Environmental Research*, 38 (2), 115-145.

Yucel-Gier, G., Kucuksezgin, F., Kocak, F., 2007. Effects of fish farming on nutrients and benthic community structure in the Eastern Aegean (Turkey). *Aquaculture Research*, 38 (3), 256-267.

Zerrouki, R., 2012. *Study of environmental parameters and growth of Mytilus galloprovincialis, (Lmk 1819) in extensive rearing at the "Orca Marine" farm in Ain Chorb (ex QSurcouf) Wilaya d'Alger*. Magister Thesis. École nationale superieure des sciences de la mer et de l'aménagement du littoral, Algeria, 60 pp.