Assessing the suitability of a range of benthic indices in the evaluation of environmental impact of fin and shellfish aquaculture located in sites across Europe

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A B S T R A C T

The European Union-funded ECASA project (Ecosystem Approach for Sustainable Aquaculture) studied the impacts from aquaculture on ecosystems from northern Norway to Greece. The objectives of this investigation were to identify quantitative indicators of the effects of aquaculture on marine communities, and to assess their applicability over a range of ecosystems and aquaculture production systems. The study included 6 Mediterranean and 4 Atlantic sites, 7 of which produced finfish (seabream, seabass, tuna, salmon and cod), and 2 bivalve molluscs (oysters, mussels, and clams); one site produced both fish and bivalves. Cultivation methods includedfinfish cages, long-lines and trestles. Similar sampling methodologies were employed at the 10 study sites, obtaining sediment, hydrodynamic, and benthic faunal data. The horizontal impact from organic enrichment extended 50 m from the farms, with contradictory responses in several indicators (individual abundance, biomass) and a more consistent response of the Infaunal Trophic Index (ITI) and AZTI’s Marine Biotic Index (AMBI). By means of Partial Redundancy Analysis, it was demonstrated that the environmental variables explained 53.2% of the variability in the macrofaunal variables (individual abundance, species richness, diversity, AMBI and ITI), whilst the explained variance was partialled out within three groups of variables: (i) ‘hydrography’ (depth, distance to farm, average current speed), which explained 11.5% of the variance; (ii) ‘sediment’ (Eh and percentages of silt and total organic matter), which explained 5.4%; and (iii) ‘cages’ (years of production and annual production), which explained 15.2%. The shared variance explained by interactions among these groups was 21.1%. These results, together with multiple regression analysis, provide an accurate assessment of the degree of impact from aquaculture. In conclusion, the use of several benthic indicators, in assessing farm impacts, together with the investigation of dynamics of the studied location, water depth, years of farm activity, and total annual production, must be included when interpreting the response of benthic communities to organic enrichment from aquaculture.

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1. Introduction

Marine aquaculture continues to expand both globally and within Europe bringing benefits to society, often in fragile coastal communities where traditional employment opportunities are in decline (FAO, 2007). However, there are well-documented cases where aquaculture has a negative impact on the environment (Karakassis et al., 2000; Black, 2001; Buschmann et al., 2006; Kalantzis and Karakassis, 2006; Pergent-Martini et al., 2006; Apostolaki et al., 2007; Giles, 2008; Holmer et al., 2008). These studies have focussed on the effects of waste products (dissolved and particulate nutrients, chemicals and medicines) on benthic and planktonic communities, the transmission of genes, parasites and diseases between wild and cultured species, as well as interactions with both local and oceanic fisheries. There are also well-known conflicts between aquaculture and other coastal uses (Michler-Cieluch and Kodeih, 2008).

Although considerable work has been conducted on this topic, much of the information gathered has yet to be integrated, analysed and...
Table 1

<table>
<thead>
<tr>
<th>Country</th>
<th>Location</th>
<th>Stations (nr)</th>
<th>Depth range (m)</th>
<th>Silt content (%)</th>
<th>Species</th>
<th>Culture method</th>
<th>Annual production (tons)</th>
<th>Years of production (nr)</th>
<th>Averaged current speed (cm s⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Greece</td>
<td>Sounion</td>
<td>6</td>
<td>12.5–17.1</td>
<td>0.1–2.6</td>
<td>Seabream-Seabass</td>
<td>Cages</td>
<td>320</td>
<td>8</td>
<td>6.5</td>
</tr>
<tr>
<td>Greece</td>
<td>Cephalonia</td>
<td>6</td>
<td>19.7–20.6</td>
<td>53.5–94.2</td>
<td>Seabream-Seabass</td>
<td>Cages</td>
<td>1000</td>
<td>22</td>
<td>2.4</td>
</tr>
<tr>
<td>Slovenia</td>
<td>Piran</td>
<td>8</td>
<td>13.0–13.3</td>
<td>100</td>
<td>Seabream-Mussel</td>
<td>Cages-Long-lines</td>
<td>71</td>
<td>14</td>
<td>2.5</td>
</tr>
<tr>
<td>Italy</td>
<td>Biscaglie</td>
<td>4</td>
<td>22.5–25.1</td>
<td>218–58.0</td>
<td>Mussel</td>
<td>Cages</td>
<td>800</td>
<td>15</td>
<td>13.8</td>
</tr>
<tr>
<td>Italy</td>
<td>Chioggia</td>
<td>4</td>
<td>53.0–62.0</td>
<td>11.7–37.0</td>
<td>Tuna</td>
<td>Cages</td>
<td>600</td>
<td>13</td>
<td>12</td>
</tr>
<tr>
<td>Spain</td>
<td>Garrucha</td>
<td>10</td>
<td>2.6 Seabream-Seabass</td>
<td>218–58.0</td>
<td>Mussel</td>
<td>Cages</td>
<td>218</td>
<td>3</td>
<td>14</td>
</tr>
<tr>
<td>France</td>
<td>Baie des Vays</td>
<td>6</td>
<td>Intertidal</td>
<td>–</td>
<td>Oyster-Mussel-Clams</td>
<td>Trestle and pole</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
</tr>
<tr>
<td>United Kingdom</td>
<td>Loch Creran</td>
<td>7</td>
<td>22.0–31.0</td>
<td>33.9–83.1</td>
<td>Salmon</td>
<td>Cages</td>
<td>1500</td>
<td>23</td>
<td>5</td>
</tr>
<tr>
<td>United Kingdom</td>
<td>Shetland</td>
<td>7</td>
<td>16.0–32.0</td>
<td>12.3–34.1</td>
<td>Cod</td>
<td>Cages</td>
<td>1390</td>
<td>21</td>
<td>5.2</td>
</tr>
<tr>
<td>Norway</td>
<td>Abornes</td>
<td>4</td>
<td>54.0–58.0</td>
<td>9.6–17.0</td>
<td>Salmon</td>
<td>Cages</td>
<td>1316</td>
<td>0.5–23</td>
<td>7.2</td>
</tr>
<tr>
<td>TOTAL AND TOTAL RANGE</td>
<td></td>
<td>65</td>
<td>0.0–62.0</td>
<td>0.1–100</td>
<td></td>
<td></td>
<td>71–1516</td>
<td>0.5–23</td>
<td>2.4–14</td>
</tr>
</tbody>
</table>

The ECASA study sites ranged across Europe and covered a wide range of latitudinal locations, water depths, bottom sediment types, and different cultured species (Fig. 1, Table 1). This investigation encompassed 10 study sites, consisting of 6 Mediterranean and 4 Atlantic locations (Table 1). Most of the locations (7) were finfish farms (seabream, seabass, tuna, salmon and cod), 2 were bivalve mollusc farms (oyster, mussel, and clam), and one produced both fish and shellfish.
and bivalves. Farming methods included cages, long-lines and trestles or ‘trays’ (Table 1), being the later an oyster aquaculture method consisting on timber or metal frames (trestles), supporting the mesh oyster bags, and sited on the foreshore in the intertidal area.

At each of the study sites sampling station positions were selected to reflect gradients of impact, where maximum impact was assumed to occur at, or adjacent to, the farm and decrease with distance, dropping to “nil” at the reference station. Where possible station selection was performed by preliminary modelling (using current records, e.g. Garrucha, Loch Creran, Shetland), whilst in other locations (e.g. Bisceglie, Piran) pre-survey testing was done to detect such gradients. To maximise chances of detecting a gradient, and to include distance from the farm as a variable, 4 to 10 stations per site were sampled (at 0 m, 5 m, 10 m, 25 m, 50 m, 100 m, and 200 m from the farm). At least one reference station, at similar depth and sediment type but at a location considered not to be influenced by the farm, in terms of organic matter deposition, was sampled at each location (see Fig. 2, for an example of a sampling scheme). At Creran, Shetland, Garrucha and Bisceglie two opposed transects (starting beneath the cages) within the main current direction were sampled, whereas at Piran three transects were sampled. Samples were taken when impacts were likely to be maximal, i.e. at the end of the production cycle within the farm (normally in summer, except for tuna, which is in March-April). All sampling was carried out in 2006.

At each subtidal station, at least 4 grab (van Veen, 0.1 m²) replicates were taken for macrofauna, and triplicate cores (generally

Fig. 2. Example of the sampling design in Garrucha (Spain), oriented along the main current direction in the area.

Fig. 3. Spatial variability of macrofaunal abundance across the studied transects. The direction is shown between brackets at locations with several transects sampled (S: South, N: North, E: East, W: West). At Bisceglie, Chioggia and Garrucha the macrofaunal biomass is also shown. Distance ‘0’ m is beneath the cages (see Fig. 1).
5 cm of diameter, 30 cm length) for sediment analyses. In Baie des Veys all the stations were located in intertidal areas and sampled directly by hand cores. Hand coring was also performed at the two Greek sites, Sounion and Cephalonia, using SCUBA-divers. Sieves used for extraction of the macrofauna from sediments had a mesh size of 1.0 mm. After sorting out the samples, taxa were identified to species level.

The variables used to characterize sediments were organic matter content (here expressed as percentage of loss on ignition; Total Organic Matter [TOM]), percentage of silt (i.e., sediment fraction

Table 2
Pearson’s correlation coefficients and p-values between the studied variables, taking into account data from sampled sites beneath the cages.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Production</th>
<th>Depth (sr)</th>
<th>Current (log)</th>
<th>Years</th>
<th>Silt</th>
<th>TOM</th>
<th>Eh</th>
<th>Abundance (log)</th>
<th>Richness</th>
<th>Diversity</th>
<th>AMBI</th>
</tr>
</thead>
<tbody>
<tr>
<td>Production</td>
<td>0.77**</td>
<td>0.46</td>
<td>-0.09</td>
<td>-0.10</td>
<td>0.46</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Depth (sr)</td>
<td></td>
<td>0.39</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Current (log)</td>
<td>-0.09</td>
<td></td>
<td>-0.03</td>
<td>0.32</td>
<td>-0.53</td>
<td>-0.49</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Years</td>
<td></td>
<td></td>
<td>-0.34</td>
<td>-0.34</td>
<td>-0.56</td>
<td>-0.44</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Silt</td>
<td>-0.34</td>
<td></td>
<td></td>
<td>-0.49</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TOM</td>
<td>0.06</td>
<td>0.00</td>
<td></td>
<td></td>
<td>0.40</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Eh</td>
<td>0.012</td>
<td>0.12</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Abundance (log)</td>
<td>-0.34</td>
<td>-0.40</td>
<td></td>
<td>-0.20</td>
<td>-0.15</td>
<td>0.40</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Richness</td>
<td></td>
<td></td>
<td>-0.05</td>
<td>-0.13</td>
<td>0.70*</td>
<td>0.49</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diversity</td>
<td>-0.11</td>
<td>-0.23</td>
<td></td>
<td>0.30</td>
<td>0.51</td>
<td>-0.86**</td>
<td>0.09</td>
<td>-0.36</td>
<td>0.02</td>
<td>0.28</td>
<td>0.71*</td>
</tr>
<tr>
<td>AMBI</td>
<td>0.15</td>
<td>-0.37</td>
<td></td>
<td>-0.28</td>
<td>-0.71*</td>
<td>0.78*</td>
<td>0.08</td>
<td>0.51</td>
<td>-0.13</td>
<td>-0.15</td>
<td>-0.58</td>
</tr>
<tr>
<td>ITI</td>
<td>-0.15</td>
<td>-0.41</td>
<td></td>
<td>0.44</td>
<td>-0.75*</td>
<td>-0.86**</td>
<td>-0.19</td>
<td>-0.58</td>
<td>0.16</td>
<td>0.10</td>
<td>0.67*</td>
</tr>
</tbody>
</table>

Key: log — log-transformed; sr — square root transformed. Correlations significant at p < 0.001 (***) , p < 0.01 (**) and p < 0.05 (*). Note: only subtidal locations have been used in the calculation.

Fig. 4. Spatial variability of diversity (H' ) and species richness of macrofauna assemblages across the studied transects. The direction is indicated as in Fig. 2. Distance ‘0’ m is beneath the cages (see Fig. 1).
b.<63 μm: Eleftheriou and Maclntyre, 2005) and redox potential (i.e., Eh, following Langmuir, 1971).

The hydrographic environment of each site is considered an important variable in structuring benthic impacts (Giles, 2008). Hence, current velocity and direction were measured at each site by means of Acoustic Doppler Current Profilers (ADCP) or by conventional current meters (e.g., InterOcean S4). Instruments were deployed for a period of one month or more, and data were recorded using a sampling interval of 10 min. In addition, data on years of farm activity and production volumes were obtained for each study site.

2.2. Biological indicators

The biological data sets, containing individual abundance and species composition, per replicate and station, were checked and standardised (correcting taxonomic and typographical errors), and the key structural parameters (e.g., individual abundance) were standardised per square meter. At several of the locations macrobenthic biomass was derived as dry weight (by drying samples at 80 °C until constant weight). Univariate variables were calculated from this dataset, including Shannon-Wiener’s diversity, ITI, and AMBI. AMBI was calculated using AMBI software (version 4.1, downloadable from www.azti.es) and a species-list established in December 2007. Guidelines specified in Borja and Muxika (2005) were referred to in deriving this index.

2.3. Statistical analyses of data from subtidal areas

Partial ReDundancy Analysis (pRDA) was carried out to study the variation in the macrobenthic univariate summary statistics (individual abundance, diversity, richness, AMBI, ITI) among (i) hydrographical features (average current speed, depth, distance from farms), (ii) farm characteristics or husbandry practices (years of production, annual production in tons), and (iii) sediment characteristics (Eh, percentage of silt, TOM). The pRDA was carried out following Bocard et al. (1992) and Legendre et al. (2005).

Multiple regression models assessing the suitability of predictor responses of the benthic characteristics (individual abundance, diversity, richness, AMBI, ITI) were carried out taking as independent variables the parameters from aquaculture farm and from hydrography (see above).

The statistical analyses were performed by S-Plus 2000, R Commander (Fox, 2005) and CANOCO for Windows (ter Braak and Smilauer, 2002).

Fig. 5. Spatial variability of ITI (Infaunal Trophic Index) and AMBI (AZTI’s Marine Biotic Index) across the studied transects. The direction is indicated as in Fig. 2. Distance ‘0’ m is beneath the cages.
3. Results

3.1. Spatial variability of the benthic characteristics in relation to distance to the farm

Macrofaunal abundance varied along the transects and differed between sites (Fig. 3). At 11 out of the 16 studied transects, the abundance increased and peaked within a radius of 5 to 50 m from the edge of the farm (Fig. 3). In Abornes and Piran (transect E) macrofaunal abundance increased with distance from the cages, whilst at Sounion, Shetland (transect N) and Garrucha (transect N) the opposite pattern was found (Fig. 3). In Garrucha (transect S) the maximum abundance was found at 75 m from the cages (Fig. 3). Macrofaunal biomass was also evaluated along 5 transects. With the exception of Biscaglie, the maximum biomass did not coincide within the maximum abundance (Fig. 3). Taking subtidal data into account from sampling points located beneath the farms, abundance was not significantly correlated with farm husbandry, hydrography or sediment characteristics (Table 2).

Macrofaunal diversity (Shannon-Wiener H') and species richness showed relatively similar patterns within each of the studied transects, but dissimilar patterns among the different studied sites (Fig. 4). Thus, at 10 out of the 16 studied transects the diversity and richness were minimal beneath the farm, whereas at others (e.g. Biscaglie and Garrucha (transect S)), this was not observed (Fig. 4). Considering subtidal data from stations located beneath the farm, species richness was positively and significantly correlated with depth, and both species richness and diversity were negatively correlated with the number of years of production (Table 2). ITI and AMBI showed relatively similar patterns within each studied transect, suggesting an increase in benthic quality with distance from the farms in Sounion, Cephalonia, Shetland, Cerran, Piran and Baie des Veys (Fig. 5). Some differences were apparent among other transects. In Biscaglie for example, AMBI was almost constant with distance, whereas ITI indicated a decrease in benthic quality with distance from the farm (Fig. 5). In Abornes, AMBI was almost constant within the first 100 m, whilst ITI identified better benthic quality at a distance of 50 m from the farm (Fig. 5). It is noteworthy that both ITI and AMBI were significantly (though inversely) correlated with the average current speed and the number of years of production (Table 2). These relationships indicated an increase in the benthic quality with higher current speed and fewer years of production.

3.2. Relationships between benthic characteristics and cages and hydrography in subtidal areas

These relationships are summarized in Table 3. The redundancy analysis shows that environmental variables explain 53.2% of the variability in the macrofauna parameters (abundance, richness, diversity, AMBI and ITI) (F-ratio = 6.68; p-value < 0.001; Monte Carlo permutation test). Species richness, diversity and ITI were positively correlated with Eh, distance from farm, current speed and depth, and negatively correlated with TOM, years of operation and the annual production (Fig. 6a). AMBI was inversely correlated with Eh, distance from farm, current speed and depth, and positively correlated with the TOM content, the years and the annual rate of production (Fig. 6a). The explained variance was partialled out within three groups of variables: (i) ‘hydrography’ (depth, distance to the farm, average current speed), (ii) ‘sediment’ (Eh and percentages of silt and TOM), and (iii) ‘cages’ (years of operation and annual production). Thus, the proportion of the total variation in macrofaunal structure, explained by interactions among these groups (i.e., shared variance), was 21.1%, and the variance explained by ‘cages’, ‘hydrography’, and ‘sediment’ was 15.2%, 11.5% and 5.4%, respectively (Fig. 6b).

Macrofaunal responses to the ‘hydrography’ and ‘cages’ were adjusted by multiple linear regressions (Table 4). The percentage of production (Fig. 6a). AMBI was inversely correlated with Eh, distance from farm, current speed and depth, and positively correlated with the TOM content, the years and the annual rate of production (Fig. 6a). The explained variance was partialled out within three groups of variables: (i) ‘hydrography’ (depth, distance to the farm, average current speed), (ii) ‘sediment’ (Eh and percentages of silt and TOM), and (iii) ‘cages’ (years of operation and annual production). Thus, the proportion of the total variation in macrofaunal structure, explained by interactions among these groups (i.e., shared variance), was 21.1%, and the variance explained by ‘cages’, ‘hydrography’, and ‘sediment’ was 15.2%, 11.5% and 5.4%, respectively (Fig. 6b).

Macrofaunal responses to the ‘hydrography’ and ‘cages’ were adjusted by multiple linear regressions (Table 4). The percentage of

Table 3

| Pearson's correlation coefficients and p-values between the studied variables, taking into account data from all the studied sites. |
|-----------------|-----------------|-----------------|-----------------|-----------------|-----------------|----------------- |
| Latitude        | Production      | Depth (m)       | Current (log)   | Years           | Distance (log x) | Silt    | TOM   | Eh   | Abundance (log) | Richness | Diversity | AMBI     |
| Silt            | -0.14           | -0.20           | -0.50***        | -0.53***       | 0.41***         | 0.10             |
| TOM             | 0.07            | -0.05           | -0.62***        | -0.85***       | 0.46***         | -0.02           | 0.78*** |
| Eh              | 0.20            | 0.27*           | 0.28            | 0.20           | 0.09            | 0.24            | -0.70***| -0.57***|
| Abundance (log) | -0.18           | -0.19*          | -0.38**         | -0.40**        | -0.00           | -0.06           | 0.24   | 0.15  |
| Richness        | 0.04            | 0.01            | 0.48***         | 0.25           | -0.53***        | 0.05             | -0.29* | -0.26*| 0.16  | 0.16  |
| Diversity       | -0.29           | -0.37**         | 0.35            | 0.37**         | 0.62***         | 0.25             | -0.09  | -0.25  | 0.04  | 0.01  | 0.69*** |
| AMBI            | 0.18            | 0.29*           | -0.20           | -0.44          | 0.47***         | -0.38            | 0.06   | 0.30* | 0.11  | -0.42***| -0.77*** |
| ITI             | -0.23           | -0.36**         | 0.56**          | 0.46***        | -0.86***        | 0.19             | -0.17  | -0.40*| 0.13  | -0.23 | 0.50*** | 0.76** |

Key: log — log-transformed, sr — square root transformed, log1 — transformed by log(X + 1). Correlations significant at *p<0.05, **p<0.01, ***p<0.001 and ***p<0.005. Note: only subtidal locations have been used in the calculation.
Table 4
Multiple regression models taking into account data from all the subtidal sampled sites.

<table>
<thead>
<tr>
<th>Dependent variable</th>
<th>Abundance (log(ind. m⁻²))</th>
<th>Richness (nr. of taxa)</th>
<th>Diversity (ind. bit⁻¹)</th>
<th>AMBI</th>
<th>ITI</th>
</tr>
</thead>
<tbody>
<tr>
<td>intercept</td>
<td>4.2703645</td>
<td>−7.524</td>
<td>1.774</td>
<td>4.496</td>
<td>−3.477</td>
</tr>
<tr>
<td>De = Depth (square root (meters))</td>
<td>8.402</td>
<td>0.397</td>
<td>−0.0486</td>
<td>9.371</td>
<td>6.226</td>
</tr>
<tr>
<td>C = Current speed (log(cm s⁻¹))</td>
<td>−1.072</td>
<td>0.593</td>
<td>−0.0104</td>
<td>0.000665</td>
<td>−0.0169</td>
</tr>
<tr>
<td>P = Production (tons yr⁻¹)</td>
<td>−0.000335</td>
<td>0.534</td>
<td>0.491</td>
<td>8.332</td>
<td>0.294</td>
</tr>
<tr>
<td>Di = Distance to cages (log(1+ m))</td>
<td>3.514</td>
<td>0.413</td>
<td>0.001</td>
<td>0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Multiple R-squared</td>
<td>0.226</td>
<td>0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>p-value</td>
<td>&lt;0.001</td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tbody>
</table>

The Pearson and Rosenberg (1978) paradigm predicts that benthic species richness or diversity should decrease with an increase in organic enrichment, above a certain threshold level. However, sometimes, as found at some of the ECASA sites (this study) and elsewhere, the responses were different from what would be expected, as described by Edgar et al. (2005), Sanz-Lázaro and Marín (2006), Aguado-Giménez et al. (2007) and Gibbs (2007). In the specific case of mussel farms, recent studies have shown impact gradients when using benthic indices, but when these were compared with results from the reference sites the differences were not significant (Callier et al., 2008). The wide range in benthic responses to aquaculture enrichment has therefore led to the conclusion that, aside from extreme cases, it is not trivial to anticipate how the benthic community will be affected (Karakassis et al., 2000; Pereira et al., 2004; Buschmann et al., 2006; Klaoudatos et al., 2006).

It is generally accepted that the main factors controlling the extent of benthic organic enrichment are: farm size, husbandry methods and hydrographic conditions (Hartstein and Rowden, 2004; Mente et al., 2006; Giles, 2008). However, very few studies have included a quantitative assessment of the relationships of impact parameters with site and farm characteristics, i.e. an assessment of benthic impacts from site and farm parameters (Giles, 2008). Hence, most of the abovementioned studies, which reported contradictory results in farm impacts, have not included hydrodynamics or husbandry practices in their analyses (e.g. Stenton-Dozey et al., 1999; Danovaro et al., 2004; Pereira et al., 2004; Buschmann et al., 2006), or they have included information which was not used in the analysis (e.g. Gowen and Bradbury, 1987; Aguado-Giménez et al., 2007). Although the data in our study were collected from a wide array of environments, encompassing variance explained by abundance, richness, diversity, AMBI and ITI was 22, 25, 41, 45 and 59%, respectively (Table 4). These regression equations enable us to predict (Figs. 7 and 8) average responses of macrobenthos, in terms of AMBI and ITI, respectively, to different scenarios of farms (production of 100, 800 and 1500 tons yr⁻¹), hydrographical conditions (current speeds of 3, 8, and 14 cm s⁻¹), and water depths (15, 40 and 60 m water depth), at increasing distances from cages. Thus, for example, AMBI values higher than 3.3 (indicating moderate perturbation, see Discussion) are predicted within a radius of 150 m from the farm (Fig. 8).

The outstanding feature of these analyses is that, when considering parameters for site selection (or site suitability, in the case of existing farms), an increase in water depth in these scenarios seems to play a minor role at a given current velocity; however, an increase in current speed seems to play a much bigger role in reducing the AMBI or ITI values, and consequently predicting improved benthic conditions (Figs. 7 and 8).

4. Discussion

Benthic indicators have been used extensively in the assessment of aquaculture impacts (Gowen and Bradbury, 1987; Gylenhammar and Hakanson, 2005; Buschmann et al., 2006; Kalantzi and Karakassis, 2006). When investigating spatial variability of macrobenthos parameters (e.g. abundance, biomass, etc.), in relation to increasing distance from aquaculture farms, a general pattern was found among ECASA sites of increasing abundance, diversity, richness, and ITI, and decreasing biomass and AMBI. However, this pattern was not present at several of the locations (e.g. Abornes, Garrucha, Bisceglie).

Fig. 7. Spatial variability of AMBI (AZTI’s Marine Biotic Index) predicted by the multiple linear regression given in Table 4, for three annual production rates: 100, 800 and 1500 tons yr⁻¹ (indicated with numbers in italics), at different combinations of depth and currents. Key: d — depth, s — average current speed.
a range of latitudes (from 37 to 69° N) and different farmed species, most of the benthic indices examined were significantly correlated with average current speed, farm production level, and number of years of farm activity. It is interesting to note therefore that, although Kalantzis and Karakassis (2006) observed latitudinal differences in farm impacts, this was not found in our study.

The statistical analyses of our data indicate that the major sources of variance are: total production and years of activity (15.2% of the variability explained), and water depth and current velocity (11.5% of the variability explained), together with the interactions between these factors. It is noteworthy that sediment characteristics, such as redox potential and sediment grain size were considerably less important (5.4% of the variability explained). This pattern may explain why some authors do not find strong relationships between benthic parameters and husbandry practices (see, for example, Miron et al., 2005), as they did not account for the “dynamic characteristics” (hydrodynamics) of the studied sites. This may also be the underlying reason behind the observation that despite the differences among locations the physico-chemical and biological structure of the surface sediments surrounding the farms was generally quite similar (see Chamberlain et al., 2001; Gyllenhammar and Hakansson, 2005; Kuti et al., 2007).

The results of our study are in agreement with some of the recent aquaculture reviews that consider different latitudinal, husbandry and hydrographic environments; although 47% of the variability in macrofaunal structure was not explained by the investigated factors. For example, Kalantzis and Karakassis (2006) reviewed 41 papers on the benthic effects of fish farming, and performed a meta-analysis of 120 variables. They found that most biological and geochemical impacts are determined by a combination of distance from the farm, bathymetry and/or latitude. They concluded that the complex interactions between variables and the lack of data, such as current speed, result in difficulties in setting common or uniform environmental quality standards for benthic effects of fish farming. Moreover, they recommend that environmental quality standards should take into account the differences between geographic regions, depth zones and sediment types. Giles (2008) reviewed 64 publications on fish farm benthic impacts and proposed a Bayesian network for the quantitative assessment of the relationships between impact parameters and site and farm characteristics. Furthermore, Giles showed that benthic impact was a function of fish density, farm volume, food conversion ratio, water depth, current strength and sediment mud content. From our study, and the reviews mentioned here, it seems that benthic communities show relatively general patterns of response to aquaculture pressure. Thus benthic indices are useful in environmental impact studies, provided they are combined with other tools (e.g. stable isotopes studies, as recommended in Sarà (2004)) and essential information (e.g. farm husbandry and hydrography).

These findings shed important light on aquaculture management considerations, particularly with regard to site selection and monitoring schemes for existing farms (Borja, 2002). Essential variables that should be considered when selecting a new location for aquaculture, include water depth and current speed. Borja (2002) recommended the following: (i) ‘good’ sites have >30 m water depth and current speed >15 cm s⁻¹; (ii) ‘moderate’ sites have between 15 and 30 m water depth and current speed between 5 and 15 cm s⁻¹; and (iii) ‘bad’ sites have <15 m water depth and current speed <5 cm s⁻¹. In Figs. 7 and 8, it is clear that the conditions prevailing in Borja’s ‘bad’ sites would produce substantially negative impacts in terms of AMBI and ITI values (level of impact determined also by the total annual production of the farm), whilst conditions prevailing in ‘good’ sites are unlikely yield “bad” AMBI or ITI impact scores. Thus, the use of some critical boundaries in biotic indices, such as the limit between ‘slightly’ and ‘moderate’ disturbance in AMBI (established as a value of 3.3, in Borja et al., 2000) or the limit between ‘degraded’ and ‘changed’ in ITI (established as a value of 30, in Word, 1979), allows the prediction of the level of impact when using multiple regression analysis (see Figs. 7 and 8).

This approach can also be used in the assessment and evaluation of cumulative impacts of marine farms (see King and Pushchak, 2008). Current guidelines entail impact assessment on a site-by-site, but this creates an incomplete picture of the range of potential impacts associated with new farms or added production on-site. Gyllenhammar and Hakansson (2005) proposed a general ‘rule of thumb’ whereby a 50 ton fish farm has a footprint (impacted benthos) the size of a ‘football field’; however, Figs. 7 and 8 demonstrate that this rule is rather inaccurate as impacts are a composite result of multiple factors and variables that may also interact among themselves.

Among the key variables that determine aquaculture impacts on the seafloor, water depth and hydrodynamics are positively correlated with dispersion of farm wastes, and inversely correlated with benthic impacts. Nonetheless, Sarà et al. (2006) recently suggested that the relative area of influence of the impacts of fish farms increases proportionally to increasing current velocities. Moreover, they proposed that the distribution of wastes from the cages is likely to be dependent on movements at
the bottom of the water column, suggesting that resuspension (described by Cromey et al., 2002b) is a key factor. Similarly, Kutti et al. (2007) found that, at deepwater sites, organic waste affected the benthic community on a much larger spatial scale than at shallow water sites. Thus, the benthos is affected by hydrodynamic conditions that govern the sediments’ natural assimilative capacity (Macleod et al., 2007).

Finally, as observed by Miron et al. (2005), the wide range of aquaculture impacts reported in the literature may largely be the result of individual use of ecological indices. Certain diversity indices may be misleading if not complemented with other ecological or statistical tools, as shown in our contribution. Hence, although IIT and AMBI have proven to be fairly reliable at indicating impacts in most of the circumstances examined, we have shown that the consideration of additional factors, such as hydrography and husbandry, increases the reliability of the impact assessment or prediction.

5. Conclusions

Assessment of the response of benthic communities to organic enrichment from aquaculture may be improved by using a suite of benthic indicators (rather than a single indicator), and considering variables that are unique to the studied location, e.g. water depth, hydrodynamics, years of farm activity, and total annual production. Assessments that do not consider these factors could lead to an incorrect interpretation of benthic response. Moreover, these factors should be taken into account when studying cumulative effects of existing farms, and when designing monitoring programmes of aquaculture impacts. Lessons learned from such multi-factorial approaches can also be applied to the process of aquaculture site selection.

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