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FOAM, a new simple benthic degradative module for the LAMP3D model: an application to a Mediterranean fish farm

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Abstract

The modelling framework already introduced by Doglioli, Magaldi, Vezzulli and Tucci to predict the potential impact of a marine fish farm is improved following different directions, namely (1) real historic current-metre data are used to force the simulations, (2) settling velocity values specifically targeting Mediterranean fish species are used, and (3) a new benthic degradative module, the Finite Organic Accumulation Module, is added to the modelling framework. The Finite Organic Accumulation Module uses the output of the other functional units of the modelling framework to calculate the organic load on the seabed. The Finite Organic Accumulation Module considers the natural capability of the seafloor in absorbing part of the organic load. Different remineralization rates reflect the sediment stress level according to the work of Findlay and Watling. Organic degradation for both uneaten feed and faeces is evaluated by changing the release modality (continuous and periodical) and by varying the settling velocities. It is found that the maximum impact on the benthic community is observed either for quickly sinking uneaten feed released twice a day, or for less intense near-bottom current conditions. If both the above-mentioned scenarios coexist, a high stress level is established in the sediment. The model also suggests that the use of self-feeders in cages can reduce farm impacts significantly. These results show how the new and more complete modelling framework presented here is able to improve the objectivity in the decision-making processes and how it may be successfully used for planning and monitoring purposes.

Keywords: biodegradation modelling, aquaculture impact, Mediterranean Sea, husbandry, net-pen, organic matter

Introduction

The increase in global fish consumption and the decrease in wild fish stocks are the main reasons behind the continuous development of marine aquaculture (FAO Fisheries Division, 2006, http://www.fao.org/docrep/009/a0874e/a0874e00.htm).
The worldwide expansion of marine fish farms, however, has always generated concern regarding the possible impacts on coastal ecosystems. Already in 1995, the Food and Agriculture Organization (FAO) of the United Nations adopted a Code of Conduct for Responsible Fisheries. The Code provided the necessary framework for national and international efforts to ensure sustainable exploitation of aquatic living resources. Particular attention was paid to the aquaculture growth in accord with the sustainable and integrated use of the environment, taking into

account the fragility of coastal ecosystems, the finite nature of their natural resources and the needs of coastal communities. In 2001, following the same direction, the European Union started to set up a strategy for sustainable aquaculture development with the Biodiversity Action Plan for Fisheries (COM 2001a) and the European Strategy for Sustainable Development (COM 2001b). These two documents led to the more recent and specific Strategy for the Sustainable Development of European Aquaculture (COM 2002).

Marine aquaculture operations are still very expensive, and the only means by which profitability can be sustained is to intensify fish production. Unfortunately, this intensification increases the already existing concerns about reaching and surpassing the natural capability of the environment. The scientific literature has identified the main environmental impact from fish farms to be the release of particulate waste products (Hall, Anderson, Holby, Kollberg & Samuelsson 1990; Holmer & Kristensen 1992; Karakassis, Tsapakis, Hatziyanni, Papadopoulou & Plaiti 2000). Particulate wastes increase the organic load on the benthic environment and might determine changes in the community structure and in the biodiversity of the benthic assemblages (Tsutsumi, Kikuchi, Tanaka, Higashi, Imasaka & Miyazaki 1991; Wu, Lam, MacKay, Lau & Yam 1994; Vezzulli, Chelossi, Riccardi & Fabiano 2002; Vezzulli, Marrale, Moreno & Fabiano 2003). Therefore, we are in need of predictive tools able to assess whether or not the establishment of a new farm (or the permission for an increase in production of an already existing one) can result in a potential impact on the surrounding environment.

Numerical models can be used to perform environmental impact predictions and test different scenarios. The interest in tracking aquaculture wastes with mathematical models has been rapidly increasing in time as a consequence (Henderson, Gamito, Karakassis, Pederson & Smaal 2001). In the past, we have moved from using analytical models describing oversimplified dispersion patterns in a constant flow in time and space (Gowen, Bradbury & Brown 1989), to implementing equations with too many simplifying assumptions about hydrodynamics (Gillibrand & Turrell 1997). Others have developed particle-tracking models using hydrographic data and were therefore limited in their simulations by the sparse data in time and in space (Cromey, Nickell & Black 2002). Ocean dynamics, instead, are usually very complex, and ocean ecosystems are likely to experience current reversals and flow variability. Pioneering numerical studies used circulation models focusing on strong tidally driven systems. In this case, the flow could have been considered obeying two-dimensional (2D) vertically averaged dynamics (Panchang, Cheng & Newell 1997; Dudley, Panchang & Newell 2000). Unfortunately, the 2D approximation can be inappropriate in more complex and dynamical systems where vertical phenomena affect the dispersion of different particles. Keeping this in mind, some of us were the first ones to directly take into account the three-dimensional (3D) ocean circulation and its variability in tracking different aquaculture wastes (Doglioli, Magaldi, Vezzulli & Tucci 2004; hereinafter referred to as DMVT04). Nevertheless, to our knowledge, DMVT04 still represents the only application of a 3D hydrodynamical model for aquaculture purposes.

The present study takes place following the continuous effort in improving the framework initially set up by DMVT04. The improvements and the assessment of their relative importance have been carried out mainly in three areas and represent the core and the original intent of this work. Namely, in this study, we (a) improved our hydrodynamics using real historic current-metre data to force the simulations, (b) improved our dispersion using a larger number of particles and updating the settling velocity values specifically for Mediterranean fish species and for their feed, and (c) added a new coupled benthic module to consider the environmental response to the organic load from the cages.

In DMVT04, some of us used idealized winds to force simulations. The choice of the winds was based on a statistical treatment of 34 years of wind data, and it allowed us to carry out a complete 12-day hydrodynamic simulation during which wind direction and speed were changing according to a typical local meteorological sequence. In a later paper, however, some of us successfully used historical current-metre data to study the hydrodynamic characteristics of the area under examination (Doglioli, Griffa & Magaldi 2004). Because the focus of this study is to move towards a more realistic scenario, we decided to implement the already validated forcing set-up used in Doglioli, Griffa *et al.* (2004). This mainly implies that the open boundary conditions and the forcing evaluation are improved by applying realistic current measurements.

On the other hand, settling velocity values for uneaten feed and faeces represent key parameters for aquaculture waste dispersion models. Due to the lack of values specifically targeting Mediterranean fish and their feeds, DMVT04 had to use the only values available in the literature, i.e. the ones measured for salmonids (Chen, Beveridge & Telfer 1999a, b). However, two recent works filled this important gap. On the one hand, some of us, in Vassallo, Doglioli, Rinaldi and Beiso (2006), presented the settling velocities of a feed usually utilized in Mediterranean farms (the 'Marico Seabass and Seabream' pellets produced by Coppens International), while on the other, Magill, Thetmeyer and Cromey (2006) measured the settling velocities of gilthead sea bream and sea bass faecal particles collected in sediment cores in a Greek fish farm. The second original aspect of this study is that it uses these new values, paying particular attention to the role they play in the overall results.

Finally, we recognize that following only the fate of the particles, as we did in DMVT04, is not sufficient to correctly assess the organic load on the sea bottom. The modelling effort should consider the natural capability of the benthic environment in reacting and absorbing fluctuations in the organic load. Our model framework is integrated with an additional new numerical benthic degradative module, the Finite Organic Accumulation Module (FOAM). Finite Organic Accumulation Module is mainly based on the ideas expressed in the work of Findlay and Watling (1997) (hereinafter referred to as FW97). They proposed an index of impact based on the ratio between the quantity of oxygen supplied to the sediment and the quantity of oxygen required by the sediment. The oxygen supply is a function of the near-bottom flow velocities and is calculated using the empirical relation put forth by FW97. The oxygen demand is based on the organic load from the cages, and it is strongly related to the microbial benthic metabolism rate. As pointed out by the same authors, the equations proposed by FW97 can be easily exploited by numerical modellers, because the only needed input variables are the bottom flow velocities and the organic flux towards the seabed. Because our model does consider the vertical dimension, it is also able to provide these important, required data. Furthermore, its intrinsic Lagrangian nature allows a simple numerical implementation of the ideas proposed in FW97.

The rest of this paper is organized as follows: in 'Methods', a description of the study area and the details of the modelling effort are provided. The results of the numerical experiments are presented in 'Results' and discussed in 'Discussion'. Finally, the conclusions are given in 'Conclusions'.

Methods

The simulations are carried out for the offshore fish farm located in the Ligurian Sea already described in DMVT04 (Fig. 1). The sea cages are located at about 1.5 km from the coast, and they cover an area of 0.2 km^2. The bottom depth ranges between 38 and 41 m. The farm is composed of eight fish cages with a capacity of 2000 m^3 each. The reared biomass is 20 kg m^{-3} for an annual mean production of about 200 tonne year^{-1}. The fish in the cages are gilthead sea bream (*Sparus aurata*) and sea bass (*Dicentrarchus labrax*).

The modelling framework consists of different models that are coupled together into a single functional unit (Fig. 2). The hydrodynamic model is the Princeton Ocean Model (POM), and it is used to derive space and time information of the circulation of the coastal area. Princeton Ocean Model is coupled online with the three-dimensional Lagrangian Assessment for Marine Pollution Model (LAMP3D). Three-dimensional Lagrangian Assessment for Marine Pollution Model is used to track the particle positions in time and space. The Finite Organic Accumulation Model represents the biochemical component of the modelling system, and it uses POM and LAMP3D outputs to estimate the potential environmental impact due to the organic load from the cages. Princeton Ocean Model and LAMP3D calculate the bottom velocities and the particle fluxes to the bottom, and these values are then used by FOAM to calculate the final organic load in each mesh of its numerical domain.

The following part of this section gives a more detailed description of the entire modelling framework.

The advective and dispersive models: POM and LAMP3D

Some historical measurements of the coastal current in the area are available in terms of current-metre time series and hydrographic surveys, covering a total of 10 months during 1978–1979 (Astraldi & Manzella 1983). Data are archived in the SIAM database (http://www.santateresa.enea.it/wwwste/banchedati/bd.ambmar.html), and they have been kindly provided to us by the Italian National Agency for New Technologies, Energy and Environment (ENEA) and the National Research Council (CNR). In this study, we concentrate on the winter–spring period, when the currents are stronger and better defined and the stratification is being formed. We select

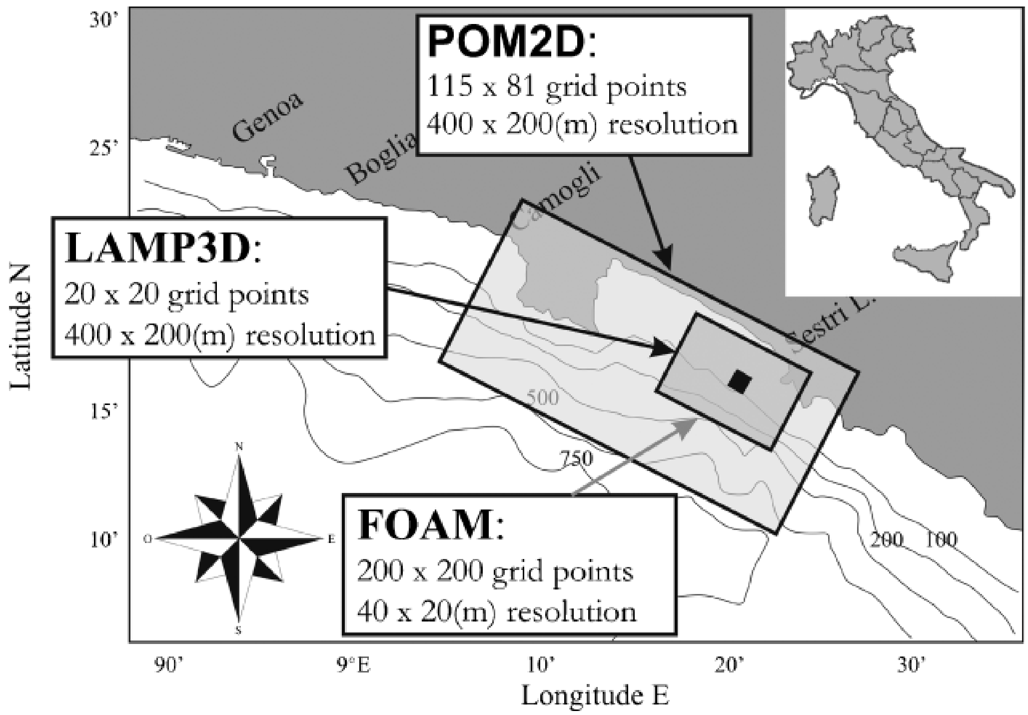

Figure 1 Study area and position and dimensions of the three different grids. The lines represent the isobaths in metres, while the black square represents the fish farm position.

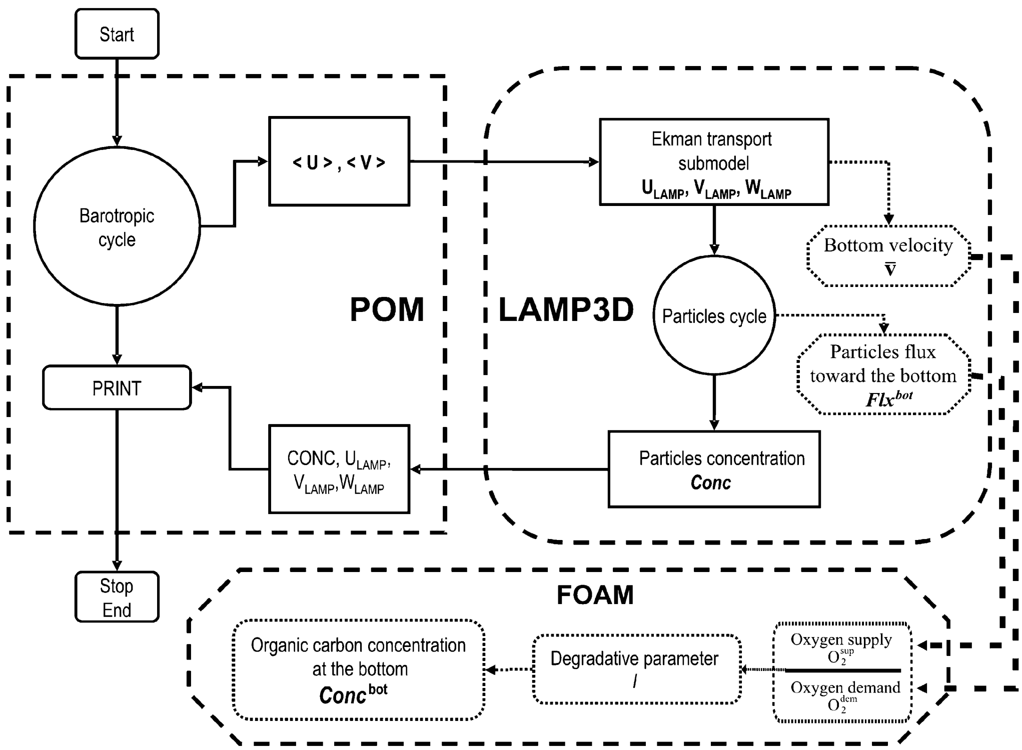

Figure 2 Model flow chart. The thick solid arrows represent on-line connections, while the thick dashed arrows represent off-line connections. Finite Organic Accumulation Module reads the bottom velocity and particle flux and computes oxygen supply and demand. The two parameters provide information on the sediment and on the benthic stress.

the period from 8 February 1979 to 30 June 1979, and we force the model on the eastern boundary. At the western boundary, a radiation condition is imposed. The set-up described is the same as that used and validated in Doglioli, Griffa *et al.* (2004). The reader is referred to this paper for a more detailed description of the boundary conditions and for their validation. Here, complete 4-month simulations are carried out, obtaining current data necessary for the dispersion–degradation runs. The first velocity value ($U = -0.19\ \mathrm{m\,s^{-1}}$), measured on 8 February 1979, is provided on the whole domain as an initial condition for all the simulations. Because the objective of this work is simulating longer time periods, we cyclically repeat the real current-metre data as boundary conditions to force the runs. Consequently, the organic matter accumulation on the seabed can be estimated for longer time periods and the dependence of the model results on the initial condition can be reduced.

Moreover, because the primary focus of this study is the organic load on the seafloor and not the fate of the dissolved nutrients, we adopt a new set-up with respect to that used in DMVT04 for the hydro-dynamical model POM and the dispersive model LAMP3D (see Fig. 1). The POM grid has 115 × 80 meshes with a spatial resolution of 400 m along the *x*-direction and 200 m along the *y*-direction. This resolution reflects the best available bathymetric data in the area. The LAMP3D numerical domain, instead, is smaller (8 km × 4 km), and it is nested in the POM grid with the same spatial resolution (400 m × 200 m).

The other dispersive parameters are unchanged with respect to DMVT04, with the exception of the Lagrangian particle number, which was increased to 620 000 for greater precision and better rendering. DMVT04's assumptions on the organic carbon concentration in feed and faecal waste are adopted. In particular, the value of 5% for the feed loss was recently confirmed by results of the MERAMED project (http://www.meramed.com). Nevertheless, because the number of particles is increased with respect to DMVT04, we calculate new conversion factors for uneaten feed and faecal waste (Table 1). We also keep calculating nitrogen-loading rates for validation purposes (see 'Results') using the same conversion factors as that used in DMVT04 for nitrogen.

Due to the lack of data for Mediterranean species, in DMVT04, we used the values proposed by Chen *et al.* (1999a, b) for salmonids. Recently, Magill *et al.* (2006) have measured the settling rates of faecal material of gilthead sea bream and sea bass, while, under laboratory conditions reproducing the Mediterranean seawater, Vassallo *et al.* (2006) have provided the settling velocity values of a typical growing sequence of feed pellets for the same species. We therefore used the values of these recent works in our simulations.

Table 1 Input parameters for POM–LAMP3D and FOAM

	Value
POM–LAMP3D parameters	
POM physical domain (km)	46 × 16
LAMP3D physical domain (km)	8 × 4
Horizontal resolution (m)	400 × 200
Vertical resolution (m)	10
Barotropic cycle time step (s)	1
Smagorinsky diffusivity coefficient	0.1
Asselin filter coefficient	0.05
Ekman depth, ΔE (m)	50
Wind drag coefficient, C_d	0.001
Horizontal standard deviation, σ (m)	3.46
Particle cycle time step (s)	60
Number of particles	620 000
Feed conversion factor for organic carbon, $w_{\mathrm{feed\,C}}$ ($\mathrm{mmol\,C\,particle^{-1}}$)	308.6
Faeces conversion factor for organic carbon, $w_{\mathrm{faeces\,C}}$ ($\mathrm{mmol\,C\,particle^{-1}}$)	5.8
Feed conversion factor for nitrogen, $w_{\mathrm{feed\,N}}$ ($\mathrm{mmol\,N\,particle^{-1}}$)	167.8
Faeces conversion factor for nitrogen, $w_{\mathrm{faeces\,N}}$ ($\mathrm{mmol\,N\,particle^{-1}}$)	66.4
FOAM parameters	
Physical domain (km)	8 × 4
Horizontal resolution (m)	40 × 20
O_2 supply parameter A ($\mathrm{mmol\,O_2\,m^{-2}\,day^{-1}}$)	736.3
O_2 supply parameter B ($\mathrm{mmol\,O_2\,s^{-1}\,m^{-3}\,day^{-1}}$)	672.5
O_2 demand parameter C ($\mathrm{mmol\,O_2\,mmol\,C^{-1}}$)	1.07
O_2 demand parameter D ($\mathrm{mmol\,O_2\,m^{-2}\,day^{-1}}$)	−32.6
Settled matter non-stress, N_s ($\mathrm{mmol\,C\,m^{-2}\,day^{-1}}$)	27.53
Settled matter intermediate-stress, I_s ($\mathrm{mmol\,C\,m^{-2}\,day^{-1}}$)	57.50
Settled matter hyper-stress, H_s ($\mathrm{mmol\,C\,m^{-2}\,day^{-1}}$)	30.59
Range amplitude parameter, Δ_{fw}	0.5

FOAM, Finite Organic Accumulation Module; LAMP3D, three-dimensional Lagrangian Assessment for Marine Pollution Model; POM, Princeton Ocean Model.

All the parameters used in the hydrodynamic and dispersive models are summarized in the upper part of Table 1, while the different settling velocity values are reported in Table 2.

The benthic module: FOAM

A new bottom boundary condition is implemented in our model. When a numerical particle touches the seabed, it is considered to be biodegradable settled matter and is treated by the benthic module FOAM.

Table 2 Settling velocity values for feed (Vassallo *et al.* 2006) and for faecal pellets (Magill *et al.* 2006)

Feed pellets		**Faecal pellets**	
Diameter (mm)	V_{sed} **(m s^{-1})**	**Fish species [size (g)]**	V_{sed} **(m s^{-1})**
3	0.087 ↓	*Sparus aurata* (380)	0.004 ↓
3.5	0.118	*S. aurata* (60)	0.005
4.5	0.103	*Dicentrarchus labrax* (280)	0.006
5	0.144 ↓ ↓	*D. labrax* (80)	0.007 ↓ ↓
6	0.088		

Arrows indicate the values used in the experiments: slowly (↓) and quickly (↓ ↓) sinking particles.

Finite Organic Accumulation Module covers the same area of the dispersive model, but its resolution is 10 times higher, namely 40 m × 20 m (Fig. 1). This resolution adequately represents the known processes of degradation and is acceptable in terms of computational time. In the case of FOAM, a higher resolution is feasible, because its calculations are performed off-line.

According to FW97, the organic accumulation at the bottom leads to different rates of mineralization in relation to the level of stress that the seabed is exposed to. In order to simulate the biological reaction of the microbial benthic community to the variations in the organic enrichment, we assign the status of the sediment in each grid mesh according to the ratio between the benthic oxygen supply and the demand.

In FOAM, the same equations and constants proposed by FW97 are used. The oxygen supply is a function of the near-bottom velocities and can be calculated using simple Fickian diffusion arguments and expressed by the empirical relation

$$O_2^{\mathrm{sup}} = A + B\,\log(\bar{v}) \qquad (1)$$

where A and B are constants (see Table 1) and $\bar{v}$ is the time-averaged current velocity taken at 1 m from the bottom. It is important to note that $\bar{v}$ is just the numerical value of the bottom flow velocity when it is expressed in centimetre per second. In our model, this value is obtained by linear interpolation of the velocity in the deepest vertical grid cell and by using an average time interval of $\Delta t = 2$ h. This choice was already made by FW97 to describe oxygen supply to the benthos. Moreover, this time interval seems to be critical, because a 2-h exposure to reduced oxygen and elevated hydrogen sulphide concentrations causes permanent damage to the gill tissues of sensitive infauna (Theede, Ponat, Hiroki & Schlieper 1969). The same choice of $\Delta t = 2$ h is also supported by the more recent work by Morrisey, Gibbs, Pickmere and Cole (2000).

The oxygen demand, instead, is a function of the organic carbon flux towards the sea bottom Flx^{Bot} according to the relation

$$O_2^{\mathrm{dem}} = CFlx^{\mathrm{Bot}} + D \qquad (2)$$

where, again, C and D are just constants (Table 1; for more details refer to Figs 2 and 3 in FW97). If i and j are the grid mesh indexes in the x and y directions, respectively, the carbon flux Flx^{Bot} in each grid mesh (i, j) at the instant k is calculated on the basis of the number of particles reaching the bottom, $n_{i,j}^{\mathrm{Bot}}$, during an integration time interval dt, i.e.

$$Flx_{\mathrm{k}}^{\mathrm{Bot}} = \frac{n_{i,j}^{\mathrm{Bot}} w^{\mathrm{C}}}{\mathrm{d}t\Delta x\Delta y} \qquad (3)$$

In Eqn (3), Δx and Δy are the horizontal grid sizes, while w^{C} stands for the adopted organic carbon conversion factor. The value of w^{C} varies if we consider feed or faeces, and the different values are listed in Table 1.

Once the model provides O_2^{sup} and O_2^{dem} for each grid mesh, we can calculate the index of impact I as suggested by FW97 as

$$I_{i,j} = \frac{O_2^{\mathrm{sup}}}{O_2^{\mathrm{dem}}} \qquad (4)$$

Based on I, we can identify three different levels of stress: non-stressed sediments, intermediately stressed sediments and highly stressed sediments. FW97 suggested that when $I > 1$, the supply of oxygen is in excess of the demand and therefore the impact is minimal. When $I \approx 1$, the impact can be moderate, while when $I < 1$, the sediment exhibits the azoic sediment endpoint and the resulting impact is high. In our model, the discrete FW97 criterion for the different levels becomes

no stress,	if	$I_{i,j} > 1 + \Delta_{\mathrm{fw}}$;
medium stress,	if	$(1 - \Delta_{\mathrm{fw}}) <; I_{i,j} <; (1 + \Delta_{\mathrm{fw}})$;
high stress,	if	$I_{i,j} <; 1 - \Delta_{\mathrm{fw}}$.

Sensitivity tests on the Δ_{fw} parameter are performed in a range varying from $\Delta_{\mathrm{fw}} = 0.05$ to 0.5. We observe no meaningful differences, and hence, for precautionary reasons, the value of $\Delta_{\mathrm{fw}} = 0.5$ is adopted.

When the level of stress is decided according to the value of $I_{i,j}$, different rates of mineralization are used in each grid mesh. In our code, this is obtained by subtracting different quantities from the already calculated organic carbon concentration fluxes. The subtracted amounts are the same as FW97 and are given in the lower part of Table 1. On the basis of the obtained fluxes, the organic carbon concentration $\mathrm{Conc}^{\mathrm{Bot}}$ in each grid mesh (i, j) is calculated as

$$\mathrm{Conc}^{\mathrm{Bot}} = \sum_{k=1}^{NT} Flx_{\mathrm{k}}^{\mathrm{Bot}} \mathrm{d}t \qquad (5)$$

where NT is the number of time intervals of the simulation.

All the parameters used in the benthic module are summarized in the lower part of Table 1.

Results

The modelled water circulation is in agreement with the past literature in the area: the simulations show the presence of the observed westward transport (Astraldi & Manzella 1983; Astraldi, Gasparini & Manzella 1990), persisting for almost the entire simulated period (winter–spring). The obtained general circulation also agrees with other numerical experiments, such as Baldi, Marri and Schirone (1997) and DMVT04. It is also possible to observe current separation and eddy formation behind the Portofino Promontory as in Doglioli, Griffa *et al.* (2004). For a more quantitative hydrodynamic validation, we use the same approach as in DMVT04. Current data simulated by the model are compared with data collected by one current metre, C1, located 2 km to the west of the farm. Current speed and direction were sampled every hour at 20 m depth from February 1993 to March 1994. Table 3 shows current data from C1 and the model outputs. When we use only one cycle at the eastern boundary and we prescribe the first velocity value on the whole domain as an initial condition, the seasonal averages from the observations are systematically lower than the model ones (see the values for the first cycle). When we cyclically repeat the boundary conditions to force the runs and we use the last velocity field of the previous cycle as the initial condition, the comparison with the C1 data improves (see the values for the fifth cycle). We speculate that the larger discrepancy observed in the first cycle is due to the artificial highly energetic initial condition. Therefore, we decide to run five cycles of linked simulations, and we subsequently neglect the first two in order to reduce the sensitivity to the initial conditions. The three linked cycles account for a total of 430 simulated days, and their averages are also reported in Table 3. In this case, the data are very close to the values calculated using the model. Current direction agrees with the observed along-shore water movement. Sporadic current reversals are also simulated, thanks to the inversions of the direction of the velocity at the inflow boundary condition.

At the same time, we can use sediment observations around the cages to validate our dispersive runs. A Van Veen grab was used to collect sediment samples in three repetitions in each of the four stations surrounding the fish farm (see Fig. 7 of DMVT04 for the exact location of the sampling stations). With respect to DMVT04, additional recent data were collected in the same stations. All the samples were analysed for total nitrogen and total phosphorus. A comparison between absolute values of these data and the model outputs is not possible because, in order to express both of them in the same units, we would need to make strong assumptions on the sediment density as well as on the sampling methodology. We therefore use the same approach as that used in DMVT04. Figure 3 shows the agreement between the field and modelled data. In particular, field sediment nutrients are the highest in station S2 and the lowest in station S4, which agrees with the

Table 3 Comparison between current measurements and model outputs at 20 m depth

	Winter average (SD)	Spring average (SD)	Summer average (SD)	Autumn average (SD)	Annual average (SD)
Observations (m s^{-1})					
C1	0.066 (0.057)	0.075 (0.065)	0.063 (0.052)	0.070 (0.052)	0.069 (0.057)
	Winter average (SD)	**Spring average (SD)**	**Summer average (SD)**	**Autumn average (SD)**	**Overall average (SD)**
Model output (m s^{-1})					
1st cycle	0.076 (0.051)	0.103 (0.084)	–	–	0.088 (0.047)
5th cycle	0.059 (0.034)	0.082 (0.066)	–	–	0.057 (0.034)
3rd → 5th cycles	0.064 (0.042)	0.078 (0.050)	–	–	0.061 (0.034)

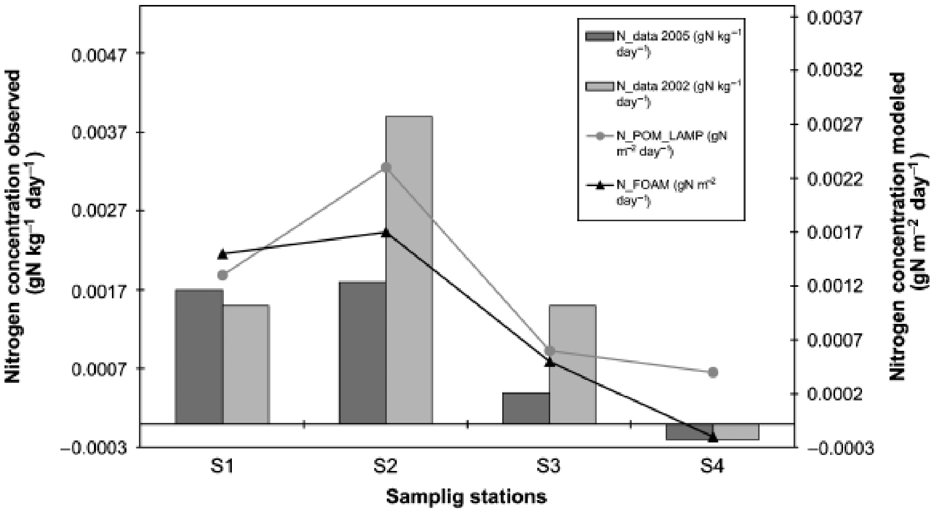

Figure 3 Daily nitrogen loading rate observation in g N kg^{-1} day^{-1} (the dark grey bar represents the 2000–2005 data, while the light grey bar represents the 2000–2002 data) and modelled in g N m^{-2} day^{-1} (the black line represents FOAM outputs, while the light grey line represents DMVT04 ones) in the four sampling stations around the fish farm. FOAM, Finite Organic Accumulation Module; DMVT04, Doglioli, Magaldi, Vezzulli and Tucci (2004).

model output for total nitrogen under westward transport. To facilitate the comparison of results for the reader, in the same Fig. 3, we also show the performance of the old set-up adopted in DMVT04 and the field data as they were at that time.

The above comparison with the only data available in the area allows us to focus on the dispersion model and on the benthic-modelled impact. The time series of the dispersion model output are also referred to the three last cycles of linked simulations. In order to explore the differences in the runs by varying the waste typology, release condition and settling velocity, we set up the following experiments:

(A1) slowly sinking feed in continuous release;
(A2) quickly sinking feed in continuous release;
(B1) slowly sinking feed in periodical release;
(B2) quickly sinking feed in periodical release;
(C1) slowly sinking faeces (continuous release); and
(C2) quickly sinking faeces (continuous release).

Note that for periodical release, we mean that the feed is supplied twice a day, and slowly and quickly sinking are referred to the minimum and maximum values listed in Table 2 for the two different waste typologies. In Table 2, the slowly sinking values are indicated by a single arrow pointing towards the bottom (↓), while the quickly sinking values are indicated by a double one (↓ ↓).

Results from the benthic module are presented in relation to (i) the extension of the impacted area, (ii) the position of this area in terms of its barycentre, (iii) the benthic trophic conditions and (iv) the predicted organic concentration at the barycentre. The choice of these parameters allows the simple and objective estimation of the degree and the location of the potential impact.

The impacted surface S is the sum of the areas of the grid meshes, where particles are still present even after the benthic degradation activity. The position of the barycentre (x_b, y_b) of this area is basically a position weighted by the number of particles left in each cell after the degradation. It is simply expressed as

$$x_b = \frac{\sum_{j=1}^{N} \sum_{i=1}^{M} i \times n_{i,j}^{\text{left}}}{n_{\text{Tot}}^{\text{left}}} \tag{6}$$

$$y_b = \frac{\sum_{j=1}^{N} \sum_{i=1}^{M} j \times n_{i,j}^{\text{left}}}{n_{\text{Tot}}^{\text{left}}} \tag{7}$$

where M and N are the numbers of meshes in the x and y directions, $n_{i,j}^{\text{left}}$ is the number of particles left at the bottom in the mesh (i, j) and $n_{\text{Tot}}^{\text{left}} = \sum_{j=1}^{N} \sum_{i=1}^{M} n_{i,j}^{\text{left}}$ is the total number of particles left at the bottom after the degradation. The benthic trophic condition and the predicted organic concentration at the barycentre are simply given by the parameter I and Conc$^{\text{Bot}}$ in the grid mesh corresponding to the barycentre position.

We initially describe the effects on the extension of the impacted area. Figure 4 shows the time series of the calculated extensions in the different

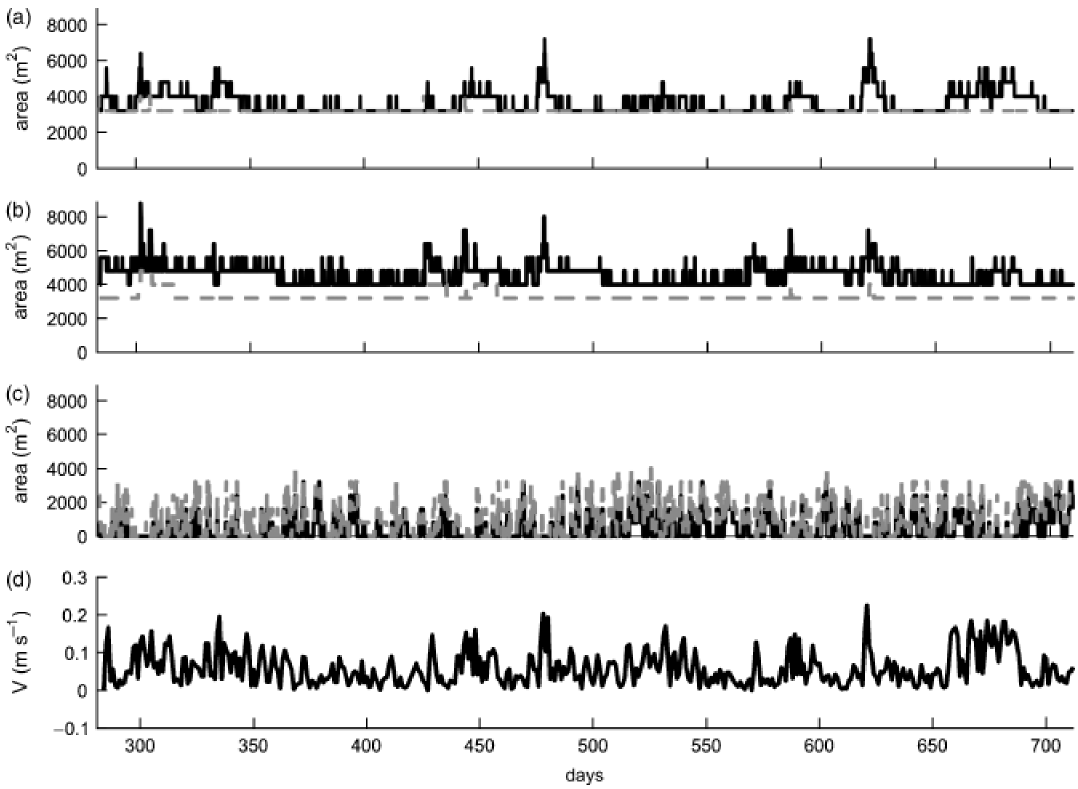

Figure 4 Time series of impact area for (a) continuously released feed pellets, (b) periodically released feed pellets, (c) faeces and (d) computed current velocity near cages. The solid black line represents slowly sinking wastes, while the dotted grey line represents quickly sinking ones.

experiments and the temporal variations of the modelled current velocity near the cages (Fig. 4d). The slowly sinking feed particles released continuously (Fig. 4a; solid black line) are dispersed by the current slightly more than the quickly sinking ones (dashed grey line). This is also confirmed by the time averages and the standard deviations for experiments (A1) and (A2) (see Table 4). The situation changes when we consider periodical release (Fig. 4b). In this case, both slowly and quickly sinking particles are dispersed on a larger area than that in the continuous case. However, while for slowly sinking particles this area is much larger and less variable in time than that in the continuous release, for quickly sinking ones, the area is just slightly larger and more variable (see Table 4). The variability of the dispersion is therefore associated with the current velocity, and it increases both with periodical release and with decreasing settling velocity values. For faecal pellets (Fig. 4c), the impacted area is smaller than that in the previous cases. Moreover, faecal wastes show greater time variability than the uneaten feed, no matter what the feed release is. The slowly sinking faecal particles impact smaller areas with respect to the quickly sinking ones, and also the variability is smaller than the quickly sinking ones (Table 4).

Figure 5 gives a better visualization of what has been stated so far and, at the same time, it shows the position of the barycentre of the impacted area. In this figure, we schematize the extension of the impacted area with a circle centred in the barycentre and having an area equivalent to the one already calculated. The grey scale represents the time evolution of the results every 60 days, while a circle is drawn every 10 days. In the case of feed, for both continuous and periodical releases, the barycentre of the impacted area is found at approximately 25 m south-westward from the cages. For the same simulations, a less significant time variability is observed (Fig. 5a1, a2, b1 and b2), and this means that the impacted area is always larger and that higher stress levels are expected. In the case of faecal wastes, instead, the barycentre shows greater time variability, according to changes in current direction and intensity (Fig. 5c1

Table 4 Time-averaged impacted area, benthic trophic conditions and organic concentrations at the barycentre position for the different experiments

Experiment	Simulation typology (release)	*S*, impacted area (mean ± SD; m^2)	Parameter *I*: No stress (% days)	Medium stress (% days)	High stress (% days)	Organic concentration (mean ± SD; g C m^{-2})
A1	Slow feed (continuous)	3576 ± 582	74	22	4	1450 ± 404
A2	Quick feed (continuous)	3202 ± 41	71	27	2	1490 ± 453
B1	Slow feed (periodical)	4513 ± 563	87	4	9	895 ± 380
B2	Quick feed (periodical)	3277 ± 266	88	4	8	1590 ± 387
C1	Slow faeces	377 ± 656	99	0	1	<1
C2	Quick faeces	941 ± 962	99	0	1	<1

and c2). This variability results in a dispersion of the faecal particles in different areas and therefore lower stress levels are expected.

To better emphasize these results, we can look at the scatter diagram of the parameter *I* at the barycentre position in time (Fig. 6). For clarity, all values >2 are artificially assigned to 2 in this figure. For feed particles continuously released (Fig. 6a), *I* mainly remains in the non-stressed range (i.e. $I>1.5$) and sometimes goes up to the intermediately stressed range, but the highly stressed level is rarely reached. There is a slight tendency for quickly sinking particles to remain more in the intermediate regime than the slow ones (Table 4). For periodical release (Fig. 6b), *1* is often in the non-stressed range, very rarely in the intermediately stressed range, but it reaches the highly stressed level more frequently than before. An easy and quick check shows that the highly stressed values are registered, in this case, in the period going from 2 to 4 h after the release. No significant difference can be observed between slowly and quickly sinking particles. For faecal wastes (Fig. 6c), the parameter *I* is practically always >2 (for this reason, in the plot, all dots are squeezed in the top) for both slowly and quickly sinking particles.

Finally, the mean values of the computed organic matter concentration $\mathrm{Conc}^{\mathrm{Bot}}$ remaining on the seabed at the barycentre position after the degradation are reported in the last column of Table 4. The organic carbon amount due to feed waste increases with time almost linearly, and the maximum values are reached in the case of the most quickly sinking feed particles in periodical release. The faecal waste instead seems to be completely degraded, and it does not contribute to organic carbon concentration at the bottom.

All the results are summarized in Table 4.

Discussion

The new model set-up is shown to better reproduce both the hydrodynamics and the dispersion in the investigated area. This is mainly due to the new forcing that is based on current-metre data and leads to more realistic results. As already remarked in previous works, current direction and intensity strongly influence the position of the impacted area and the degradation of the settled matter (FW97; Morrisey *et al.* 2000). Nevertheless, the processes involved are strongly non-linear and it is difficult to assess the role played by each parameter in the model.

When particles sink relatively quickly (settling velocities are one order of magnitude less than the current velocity, as a lower limit), the impacted area does not vary and the barycentre position depends on the main direction of the current. Instead, for relatively slowly sinking particles, the variability of the current starts to play a major role. This different behaviour explains why the barycentre of the uneaten feed remains practically motionless, while the one for the faeces is very mobile.

At the same time, current intensity reduces bottom degradation, thanks to two different processes. On the one hand, a stronger current brings more oxygen to the sediment and makes degradation more efficient. On the other hand, the same stronger current increases waste dispersion, resulting in a wider impacted area and in lower waste concentrations at the bottom. Faeces do not contain much organic carbon and the strong degradation is able to remove almost all the settled matter. Uneaten feed contains more organic carbon and sinks more rapidly than faeces. As a result, much more carbon accumulates on the sea floor, and it is only partially degraded even in the presence of strong currents. This also explains

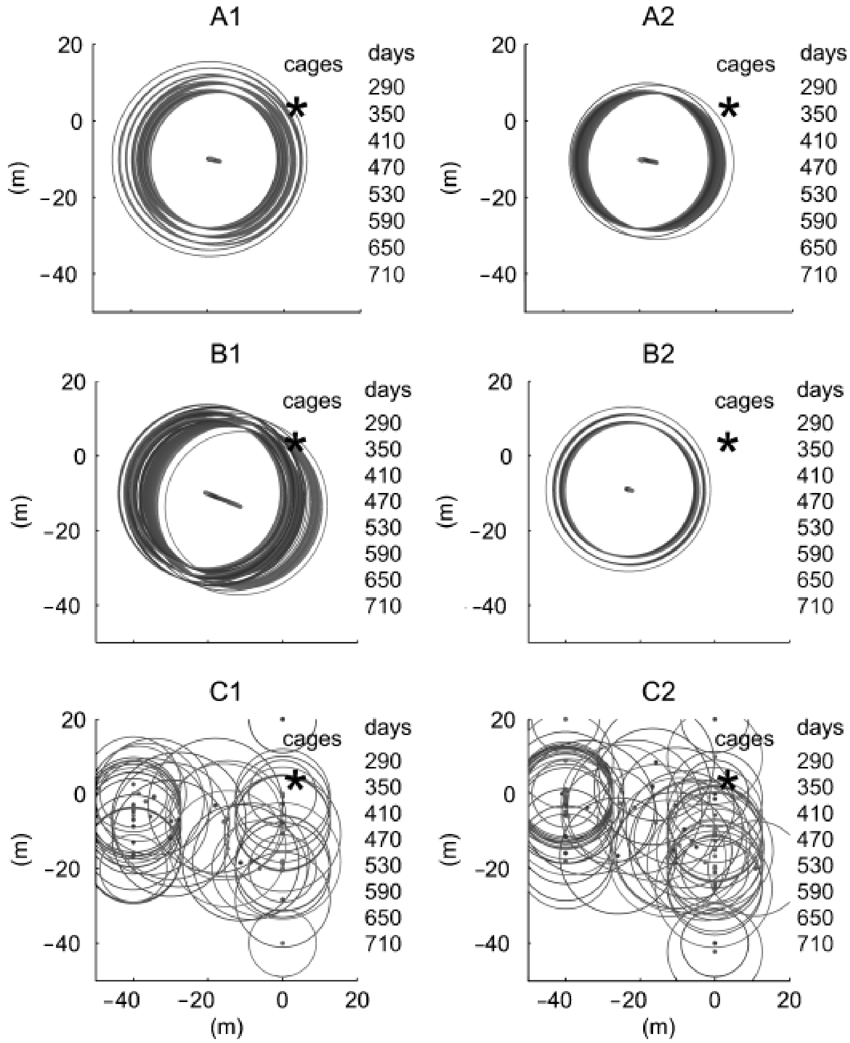

Figure 5 Time trend of impact area (circles) and barycentre (dots): (a) continuously released feed pellets, (b) periodically released feed pellets and (c) faeces. The left column shows the slowest particles and the right column shows the fastest particles of each kind of waste. In the schematization adopted, the area of circles is equivalent to the impact area.

the observed small variability in the size and position of the impacted area.

Because we use the settling velocity values for faeces measured by Magill *et al.* (2006), it is particularly interesting to compare our results with theirs. In order to do this, we calculated the accumulation rates in the barycentre for different sinking faeces. We obtained values of 11 and 19 g faeces m^{-2} $year^{-1}$ for slowly and quickly sinking particles respectively. These values are about two orders of magnitude smaller than the ones reported in Magill *et al.* (2006). Two arguments can be provided to explain this discrepancy. Firstly, the total fish biomass in the cages is not reported in Magill *et al.* (2006), and this does not allow for a correct quantitative comparison. The policy of the fish farm studied in this work is to maintain a low biomass per cage (F. Roberto Co', AQUA s.r.l., pers. comm.). It is likely that the Greek fish farm studied by Magill *et al.* (2006) has a high biomass value per cage. Secondly, with the new module FOAM, we introduced the degradation of the settled organic carbon, which is not considered instead by Magill *et al.* (2006). However, because we adopted the degradation rates proposed by FW97 for Atlantic fish farm, it is also possible that these values are too high with respect to the Mediterranean ones. Unfortunately, to the best of our knowledge, no value is available in the literature to check whether this is

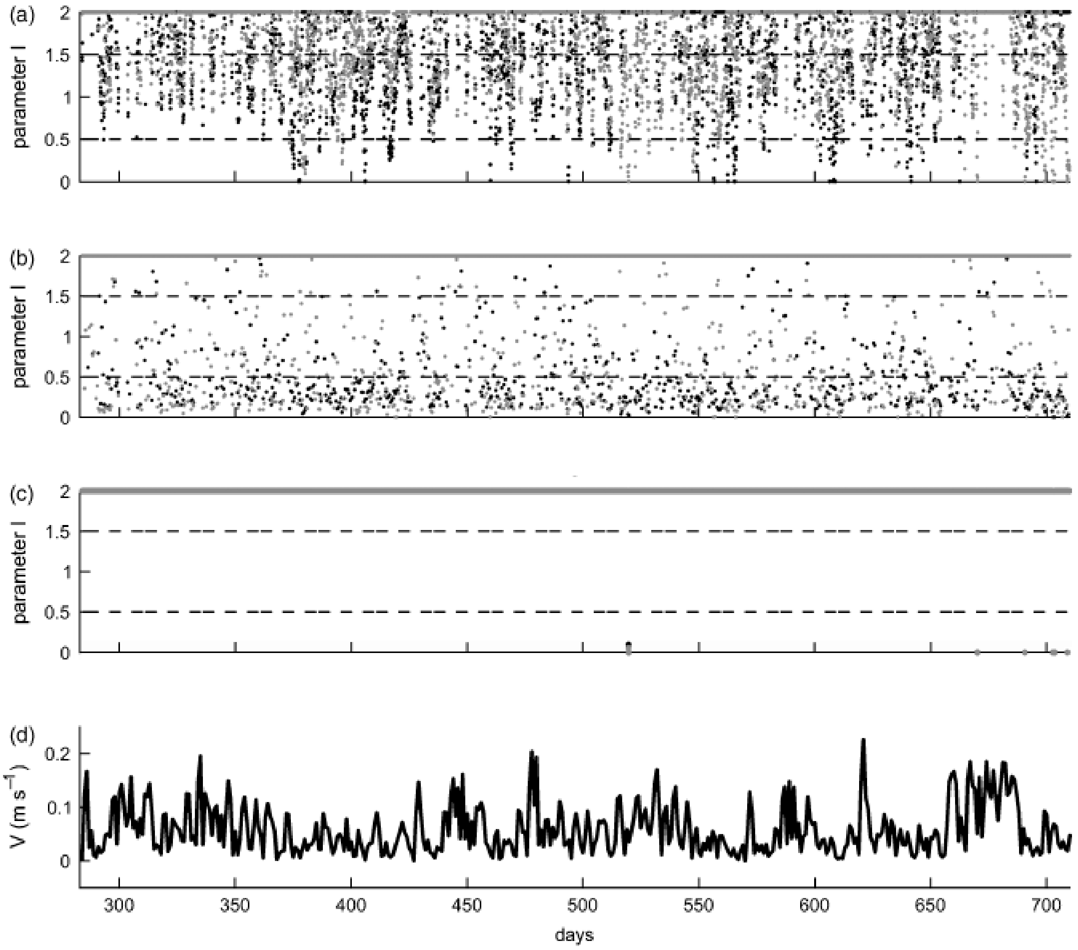

Figure 6 Spread diagram of the parameter *I* at the barycentre position. All values >2 are fixed to 2: (a) continuously released feed pellets, (b) periodically released feed pellets, (c) faeces and (d) current velocity near cages. The black dots represent the slowest particles, while the grey ones represent the fastest particles of each kind of waste. The dashed lines represent the stress-level thresholds.

really the case. On the other hand, our results agree with the work of Magill *et al.* (2006) in predicting a greater impact for the faeces of *D. labrax* respect to *S. aurata* ones near the cages. The same results also agree with previous studies and confirm the uneaten feed to be the primary cause of ecological impact on the benthos community (Beveridge, Phillips & Clarke 1991; Vezzulli *et al.* 2003). For this reason, we studied the feed release conditions in more detail. We found that a release occurring twice a day results in (i) more frequent conditions of highly stressed sediments and (ii) larger impacted areas than a continuous release. These results support the idea already proposed in previous studies of using self-feeders to reduce the uneaten feed loss without affecting fish growth rates (Azzaydi, Madrid, Zamora, Snchez-vzquez & Martnez 1998; and references therein).

Conclusions

Aquaculture is the food-related activity with the most rapid growth in the world. Because this growth produces an immediate concern, it is necessary to develop tools to predict the environmental impacts arising from intensive marine fish farms.

In this study, we improved the capability of the POM–LAMP3D model already proposed in a previous work (DMVT04). We developed both a more realistic advection–dispersion setup and a new benthic model, FOAM. Using the empirical relations put forth by FW97, FOAM calculates the organic carbon degradation for three different levels of sediment stress. We performed several runs to simulate different scenarios by varying the waste typology (faecal or feed), the settling velocity of particles (on the basis of feed dimensions, fish size and reared species) and the release conditions of feed (periodical or continuous). At the same time, the same runs allowed us to test the stability of the model that appears to be very satisfactory.

We obtained more satisfactory results for the hydrodynamics and dispersion than that in DMVT04. Moreover, FOAM revealed its ability to simulate different scenarios by switching suitable parameters.

The results presented in relation to the extension of the impacted area and the position of its barycentre show that the continuously released feed settles within a narrow area near the cages (impact area maximum 6500 m^2, barycentre shifting amplitude 10 m, cages maximum distance 25 m), while the feed released twice a day spreads on a larger area centred near the cages (maximum area 8500 m^2, barycentre shifting amplitude 15 m, cages maximum distance 25 m). Faecal pellets accumulate on a smaller area within a greater and more variable range from the cages (maximum area 4000 m^2, barycentre shifting amplitude 100 m, cage's maximum distance more than 50 m) with respect to uneaten feed. Maximum impacts, in terms of both stress parameter I and organic carbon concentration, are due to the quickly settling feed, released in periodical mode and during slow current periods. Some mitigation of the impact is observed if the feed is released continuously. The use of self-feeders has therefore been suggested to the farmers.

Further investigations may be necessary to verify the impact of combined feed and faeces settling, while mineralization rates for Mediterranean conditions and validation with specific *in situ* measurements are required.

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