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DEPOMOD—modelling the deposition and biological effects of waste solids from marine cage farms

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Abstract

To enable better predictive capability of the impact from large marine cage fish farms on the benthos and improved objectivity in the regulatory decision-making process, a computer particle tracking model DEPOMOD was developed. DEPOMOD predicts the solids accumulation on the seabed arising from a fish farm and associated changes in the benthic faunal community. The grid generation module allows the user to set up a grid containing information on depth, cage and sampling station positions for the area of interest. Given the information on wastage rates of fish food and faeces and hydrodynamics of the area, the initial deposition of particles on the seabed can then be predicted with the particle tracking model. The resuspension model then redistributes particles according to near-bed current flow fields to predict the net solids accumulated on the seabed within the grid area. From quantitative relationships between benthic community descriptors and solids accumulation, predictions of the level of benthic community impact can then be made.

The particle tracking model was validated using sediment trap studies. Model predictions of flux (g/m^2/day) generally agreed well with field data with an accuracy of $\pm$ 20% and $\pm$ 13% for a dispersive and depositional site, respectively. Using parameters from the validated resuspension model (rare among models in this field), semi-empirical quantitative relationships between predicted solids accumulation (g/m^2/year) and observed Infaunal Trophic Index (ITI) and total individual abundance were established using data from numerous Scottish marine fish farms. A submodel was also validated for predicting feed input throughout a growing cycle for planning purposes.

DEPOMOD may be used for assessing the potential impact of a farm throughout a growing cycle, or if the biomass consent is increased. It may also be used in the site selection process of a

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new farm to investigate the proposed farm position and biomass levels. Prediction of the dispersion of particulates during use of in-feed medicines may also be undertaken.

Keywords: Modelling; Aquaculture impact; Resuspension; Benthos; Particle tracking; Husbandry; Infaunal Trophic Index

1. Introduction

Applications for fish farm discharge consents for new marine cage sites in Scotland must be accompanied by measurements of current and wind speed and direction along with various data on sediment biology, water-column parameters and bathymetry. Existing sites must present monitoring data on benthic communities, sediment chemistry and water quality, the frequency and degree of sampling of which are governed by the biomass consent (large biomasses in areas of low dispersion requiring more monitoring). The Scottish Environment Protection Agency (SEPA) is currently formulating a common standard for measurement protocols, and the research described in this paper has influenced this (SEPA, 2000a,b).

A general review of modelling approaches to fish farm impacts has been undertaken by Silvert and Cromey (2001). Models developed on fjordic ecosystem dynamics (Ross et al., 1993a,b, 1994) are on a much broader scale than required for this purpose, and provide no information at high resolution. Gowen et al. (1989) described a simple model for predicting carbon deposition rates from marine fish farms based on a current meter record and the production of a site. Although this type of model is a useful starting point, it only provides limited information regarding the deposition of carbon (see Panchang and Richardson, 1992 for a review). It does not include any of the physical and biological parameters which determine the fate of organic material once it has reached the seabed, nor does it include fish husbandry factors affecting variation of input over time. Other approaches include description of an algorithm which calculates particle distributions backwards from the seabed to surface (Gowen et al., 1994; Silvert and Sowles, 1996), while Hevia et al. (1996) used a graphical programming approach to map deposition on the seabed. The use of Geographical Information Systems (GIS) is an increasingly popular method for management of fish farm impacts (Ali et al., 1991; Ross et al., 1993a,b; Ross, 1995; Nath et al., 2000; Pérez et al., in press). More sophisticated fish farm models (e.g. AWATS), which include complex hydrodynamics of tidal/wind-driven current and waves and associated resuspension processes, are less common (Panchang et al., 1997). Validation of current fish farm models rarely include a resuspension component, and the AWATS package represented a significant advance in this field (Dudley et al., 2000). There is continuous debate as to the magnitude of critical thresholds for resuspension. Low thresholds have been used in resuspension models of freshly deposited material, but these do not include fish farm wastes (Sanford et al., 1991; SCCWRP, 1992; Cromey et al., 1998). Validation of the DEPOMOD resuspension model for fish farm wastes resulted in a similarly low critical resuspension speed of 9.5 cm/s (Cromey et al., in press). Conversely, Dudley et al. (2000) report much higher critical thresholds of resuspension between 33 and

66 cm/s for these wastes. However, analysis of near-bed current speed records for 16 Scottish fish farms (minimum 15-day record length) shows maximum hourly averaged speeds ranging between 5 and 26 cm/s. Using a critical resuspension threshold at the higher end of the reported range (e.g. 66 cm/s) would result in no resuspension events being predicted for these sites. This is unrealistic as diver observations, transmissometer evidence (Cromey et al., 2000) and our validation study showed resuspension to be a regular occurrence for these current regimes. The DEPOMOD model described here includes independently validated particle tracking, resuspension and benthic response models essential for the complete impact assessment of fish farms.

Ten years ago, farms with biomass consents of 200 tonnes were considered large, and it was on such sites that much of the basic science regarding impacts to the benthos was established (Hall and Holby, 1986; Weston, 1986; Brown et al., 1987; Kaspar et al., 1988; Ritz et al., 1989). Consent applications being prepared at present typically relate to much larger operations where consents are being sought for biomasses in the range of 1000–2000 tonnes, where recent impact research has focussed (Pereira, 1997; Pearson and Black, 2001). These sites are often located in highly dispersive areas exposed to strong tidal currents, although there is still considerable demand for sites in more sheltered sea lochs. In this project, we made detailed investigations on the hydrography and biological effects of two large sites. The results were then combined with historic and other available data from several other sites for the purposes of developing and validating the model.

This project takes as a starting point the model BenOss developed by Dunstaffnage Marine Laboratory (Cromey et al., 1997, 1998). This model tracks particles of organic solids from source to incorporation and degradation in sediments and predicts indicators (indices) of benthic community response. It has been validated for long sea sewage outfalls and is now in wide use in the water industry. A number of modifications have been made to allow BenOss to be used for fish farms. The level of organic loading on the sediment is approximately two orders of magnitude greater around fish farms compared with long sea sewage outfalls, and consequently, the biological communities present are quite different. In addition, the natural background benthic fauna in sheltered sea lochs is likely to be quite different from the typical fauna found in coastal or estuarine environments where large outfalls are sited. The gradients of organic input comprising the footprint of deposition on the bed underneath cages are extremely steep with very large changes occurring in relatively short distances making sampling station selection critical. The wastes have quite different behaviours in seawater relative to suspended sewage particles and are of different composition. Fish farms are generally sited in more stratified areas; therefore, a greater appreciation of vertical current shear and its effects on particle dispersion are required. Sewage discharges can be generally described in terms of mean flows and concentrations; fish farm output varies dramatically over the growing cycle requiring modelling of typical fish growth for predictive purposes. Food wastage (i.e. uneaten feed pellets) is generally believed to be lower than in the past. Gowen et al. (1989) assumed a 20% loss, but more recently, this has been estimated at 12% from sediment trap studies (Black, Scottish Association for Marine Science, UK, unpublished). With the introduction of video technology underneath cages, which records waste food particles depositing out of the cage bottom, wastage rates below 5% have been reported and are sustainable. In practice, food wastage rates are difficult to quantify due to the large variation in husbandry practices between companies, and it was necessary to standardise this variable across sites. As the industry has developed, the range of food wastage rates between sites is likely to have decreased, as well as the overall mean. This gives further justification for standardising this variable.

The main objective was to produce a model which predicts the solids accumulation and associated benthic effects from fish farms, paying particular attention to resuspension and the benthic response. The modelling package is a useful tool for the regulator and fish farming industry. To help achieve this, the model has a user-friendly interface and requires input data that are generally available for a typical fish farm, thus reducing the need to collect additional information which might restrict its use. With the help of model testing, a set of modelling and field sampling protocols important for both the application of DEPOMOD and more general guidelines for collection of data during fish farm surveys were also developed. Different components of the model have been satisfactorily validated independently (i.e. particle tracking, resuspension, benthic response, fish growth and biomass model), so that the integrated components form an overall model. In this paper, a general description of the model, details of laboratory experiments and validation of the particle tracking and benthic modules are given. A sensitivity analysis is also detailed, and model applications and limitations, such as in areas dominated by wind-wave resuspension, are discussed.

2. Materials and methods

2.1. Description of model

Fig. 1 shows how the individual modules are integrated in DEPOMOD with details given in Appendix. Although the carbon degradation G-model (Westrich and Berner, 1984) is implemented in the model, it was not used in model validation. No literature data were available to allow suitable parameterisation of this model, so total particulate matter rather than carbon was modelled. The grid generation module generates an array used by subsequent modules containing bathymetry, cage and sampling station positions. Fine grid cell resolution is desirable (e.g. 10 m), where the limits of the predicted deposition footprint are expected to be less than 100 m away from the cages and spacing of sampling stations is small. For a larger deposition footprint, cell resolution of 25 m is more appropriate.

The particle tracking model describes transport of particles from the surface to the seabed. Large numbers of particles are used to represent the waste material which are assigned appropriate settling characteristics, although beyond a threshold further increases in particle numbers results in an insignificant change in bed particle distributions. Information on feed input and food to waste conversions (mass/time/cage) allow definition of the solids loading arising from the farm. Particles are subject to settlement through the water column while being advected in two dimensions by hydrodynamic data. These data are often implemented into the model as a number of layers, where each data set represents a layer with different current amplitude and direction. Typically, three data sets are implemented to represent shear in the water column, and this layered method is more

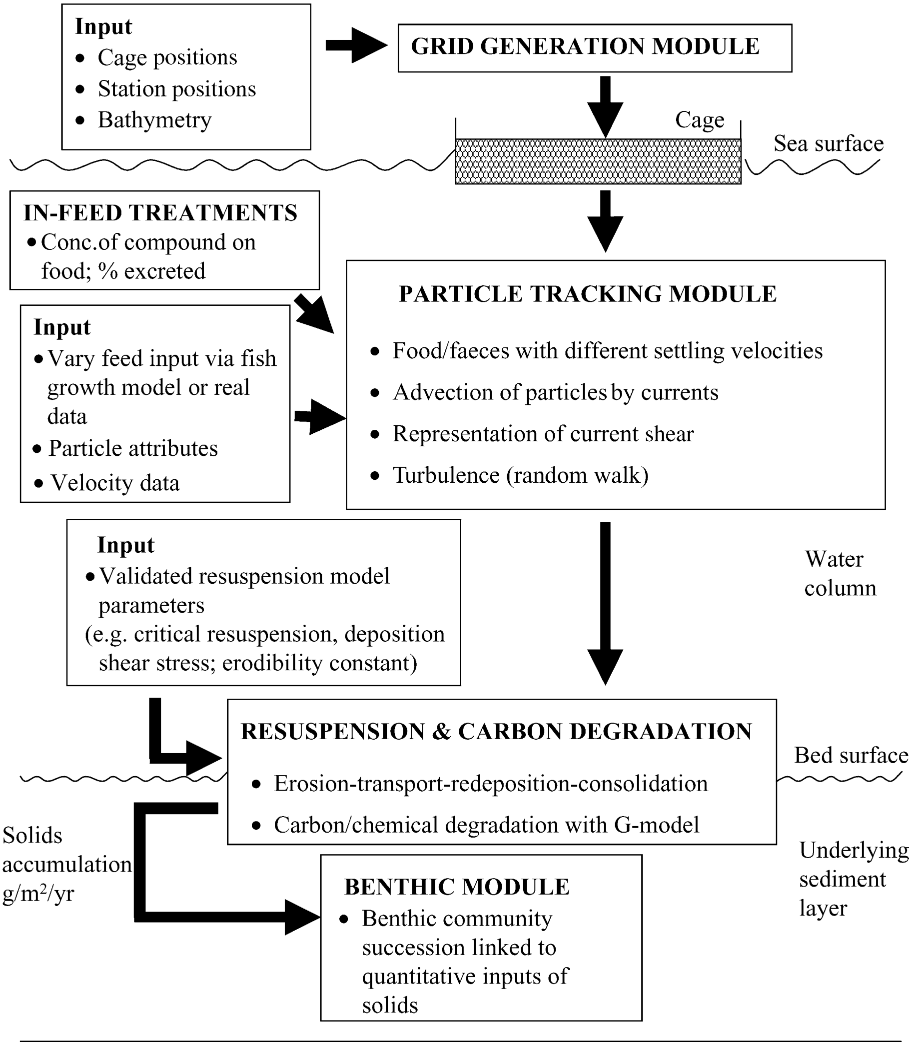

Fig. 1. Integration of the DEPOMOD modules and associated input data used for modelling benthic impacts arising from marine cage fish farms.

accurate at describing the water column than fitting a logarithmic profile to one data set. Such profiles are unsuitable for use in areas around fish farms where shear and stratification in the water column are often significant. For short time scale modelling studies (e.g. 24 h), the use of 10-min current observations has been found to be appropriate, whereas for longer term steady-state solids accumulation predictions, hourly averaged data is desirable. Particles are subject to random walk in three dimensions as a representation of turbulence and, as particles intersect the bed, information is stored for use in further modules.

The resuspension model used is a well-documented compartmentalised model consisting of erosion, transport, deposition and consolidation components. The model uses near-bed current velocity ($z \approx 2$ m) to determine a shear velocity and bed shear stress for each time step. Erosion or deposition events, which occur mutually exclusively, are determined by the magnitude of the shear stress above or below a critical threshold for erosion and deposition, respectively. Erosion rates are also influenced by an erodibility constant. Transport of resuspended particles occurs at ambient current speed, and consolidation of bed particles occurs after a given time period. For a particular level of solids accumulation, the benthic response model gives a prediction for two benthic indices, ITI and total abundance. These relationships were validated using data from several Scottish marine farms.

Model validation was undertaken in four stages. Validation of the particle tracking stage with sediment trap studies, validation of the resuspension model with a tracer study (Cromey et al., in press) and validation of the benthic response model. These separate validation studies served to test the model across a range of scenarios with temporal and spatial differences. Overall model sensitivity analysis, considered part of model validation, was then undertaken to test sensitivity of input data in the fully integrated model.

2.2. Laboratory measurements—feed and faecal pellet settling velocity

Settling characteristics of fish feed and faeces required as model input data are likely to change depending on fish size, feed composition and physical properties of the seawater (Chen et al., 1999a,b). To determine the settling velocity of a range of faecal pellets, fresh faeces were collected by traps no more than 1 h after deposition. Bottom water samples from sediment trap experiments, post-filtration, were used in a calibrated glass cylinder (length = 28.9 cm, internal diameter = 6.7 cm) and settling times measured with a digital stopwatch over the settling column length. No acceleration was observed over the majority of the column. The mean fish weight for the cage group was 3.39 kg being fed with a pellet size of 10.5 mm.

As food wastage percentages as low as 5% contribute to solids deposition, settling velocities of food pellets were also determined in a similar manner to faecal particles. Feed pellet experiments were undertaken in the laboratory using a longer cylinder than in the field (length = 3.1 m, internal diameter = 0.1 m), and pellets were timed from approximately 1 m below the surface. A range of food pellet sizes was used in the experiments with additional data being obtained from Chen et al. (1999a) and BOCM Pauls.

2.3. Particle tracking model validation

Two sites were investigated in detail for the validation study, one dispersive (site A) and one depositional (site B). Hydrographic measurements for site A showed that 60–70% of all the variance in the harmonic analysis could be attributed to tidal activity (Table 1), with similar mean and maximum speeds throughout the water column. Site B showed lower mean and maximum current speeds with only 7–20% of the variance attributable to tidal activity, demonstrating that the hydrodynamics at this site are strongly wind-driven. The

Table 1
Summary hydrographic data for site A (dispersive site) and site B (depositional site) for instruments deployed for 30 and 15 days, respectively

Site	Meter position (height above bed in m)	Mean speed (cm/s)	Max. speed (cm/s)	Residual speed (cm/s)	Residual direction	% variance attributable to tidal activity[a]
A	surface (26)	6.9	32.0	5.1	136	60
	midwater (14)	6.7	32.6	4.3	142	65
	near bed (2)	6.2	30.8	3.2	102	70
B	surface (26)	6.7	30.4	4.2	69	7
	midwater (13)	4.5	19.2	3.8	236	12
	near bed (2)	3.6	24.2	1.2	234	20

[a] Determined from harmonic analysis using a stand standard set of 32 major and 8 related tidal constituents including mean and long period constituents.

surface residual current in site B was strongly correlated with the southwesterly winds measured during the deployment period.

The sediment trap studies were designed to validate the particle tracking model by comparing predicted and observed waste solids deposition (g solids/m^2/day) from a cage group over a 24-h period at both sites. Given the short time scale of the study and the design of the sediment traps, resuspension of the material from the traps was assumed to be negligible. In addition, by situating the traps in the near vicinity of the cage group and encompassing feeding events, deposition from the cages was deemed to be much greater than natural background deposition so that all waste solids were taken to originate from the cage group. As sediment traps may receive solids from a number of cages in the group (in particular the dispersive site), it was necessary to accurately model the whole group rather than a single cage.

The diver-deployed sediment traps were constructed of ABS plastic (height = 55 cm, internal diameter = 11 cm), giving an aspect ratio of 5:1 recommended by many authors (Wassmann et al., 1991). The traps were mounted on flat ABS sheets which were held in place on the seabed by a combination of 2-kg weight and movable rod which divers pushed into the sediment. The traps were located under cages on two perpendicular transects, each with three traps arranged at 4-m intervals where trial model runs indicated high deposition. On retrieval, the tops of the traps were sealed with O-ring screw caps and returned to the surface undisturbed. The contents were filtered through preweighed 150-mm diameter Whatman No. 1 qualitative filter papers. These filtered samples were then transported to the laboratory where they were frozen at -40 °C and freeze-dried for reweighing.

Hydrographic data acquisition followed the recommendations contained in SEPA (2000a). An array of three Interocean S4 electromagnetic recording current meters (Interocean, San Diego, CA) was deployed at each site in a standard u-shaped mooring with subsurface buoyancy at a suitable distance from the cages to avoid any shadowing effect. Although the shadowing effect is known to exist around cages, this has not been quantified by any researchers to such a level which would allow inclusion in this model. The meters were set up and deployed as described in Table 2 and corrected for magnetic variation. Given the short distance of the traps from the mooring location, no spatial

Table 2
Input data used for the grid generation and particle tracking modules for sites A and B for the sediment trap study

Input data	Site A	Site B
Grid total size (m)	500 × 450	195 × 175
Grid cell resolution (m)	10 × 10	5 × 5
Number of cages	2 × 5	2 × 5
Cage dimensions (l × w × d) (m)	16 × 16 × 8	16 × 16 × 8
Cage orientation along length of cage group	NW–SE	NE–SW
Grid bathymetry (chart datum in metres)	4–44	16–30
Depth under cage group (chart datum in metres)	25–35	23
Number of current velocity data sets used	3	3
Height of instruments above sea bed (m)	25, 17, 5	28, 15, 2
Mean depth of water column at mooring (m)	30	32
Length of current velocity record (h)	24	24
Instrument sampling period[a]	2 min in every 10	2 min in every 10
Time step of data used in model (min)	10	10
Approximate farm biomass (tonnes)	500	1300
Feed input (kg/day/cage)	97.1	157.6
Number of particles used	7.0×10^4	1.4×10^4

[a] Sampling at 2 Hz with 2 min of data averaged in every 10-min period.

variation in current was represented in the model domain. Validated three-dimensional hydrodynamic models are still quite rare for Scottish fjordic systems and so our model was not designed to accept spatially varying current. Concomitant meteorological data were obtained from nearby Meteorological Office recording stations. Ideally, on-site meteorological stations would have been used, but as these data were used to assess the wind-driven component of the current for information only, model predictions were not affected.

The RoxAnn™ acoustic bottom discrimination system (Marine MicroSystems, Aberdeen, UK) used consisted of a 50-kHz transducer, with positional information obtained from a NavSymm (Navstar Systems, England) portable differential global positioning system (dGPS). The survey of both sites provided depth and positional information of the area in the immediate vicinity of the cages to allow model grid generation (Table 2). Grid total size and cell resolution was chosen to contain the majority of the predicted footprint, maintaining the required accuracy close to the cages. No adjustment was made to the survey data for tidal elevation during the survey as model predictions were shown to be insensitive to elevation change, especially where the depth greatly exceeds tidal range.

Weekly husbandry data for the study period provided by the farmer were used. Diver observations of the seabed and analysis of the sediment traps showed no waste food pellets for site A, with waste feed pellets being found for one trap at site B. To apply the model consistently across both sites, only waste faecal particles were included in the modelling. Given 100% of the food fed was ingested, food pellet water content and digestibility of 9% and 85%, respectively (Pereira, 1997), 13.65% of the food fed was associated with waste faecal particles. Using a mean settling velocity assigned to all model faecal particles results in an inaccurate representation of the observed data, so a stochastic modelling technique based on a Monte Carlo simulation was used. To take into account the variability observed, every time a new particle was assigned in the model, a settling velocity was taken from the measured normal distribution ($\mu = 3.2$ cm/s, $\sigma = 1.1$ cm/s).

Particle starting positions were at mid-depth in the cage centre and trajectories were calculated every 60 s. Sensitivity tests found a random or mid-cage particle starting position gave similar results, but revealed that surface or cage bottom starting positions were sensitive parameters. The number of particles in the model was increased until no changes in the particle bed distribution were observed. For the random walk model, the horizontal dispersion coefficients (k_x and k_y) were set to 0.1 m^2/s and the vertical dispersion coefficient (k_z) set to 0.001 m^2/s (Gillibrand and Turrell, 1997). As no data were available for both sites, these values were taken as typical for the area and tested in the sensitivity analysis. Model deposition predictions over the 24-h period were obtained for each grid node in terms of g solids/m^2 bed, scaled up to g solids/m^2 bed/year and then compared with sediment trap data. Interpolation for the exact trap position in the grid from surrounding nodes was undertaken where necessary.

2.4. Benthic response model validation

Solids flux predictions were undertaken for sites where sufficient information including benthic data were available to set up the model. These data were obtained from available literature (Pereira, 1997) or by personal communications with fish farmers and SEAS, who were project participants. The objectives of this model were to obtain predictions of solids accumulation (denoted S_{avail}—g solids/m^2/year, see Appendix A.4) for six marine farms and to establish quantitative relationships between S_{avail} and benthic indices. This allowed quantification of a benthic response dependent on the level of impact.

Details of the number of cages, dimensions, orientation and position were obtained for each site modelled for the month when benthic sampling was undertaken. Uniform bathymetry was used at some sites where the depth beneath the cage group was similar to the depth at each sampling station. At all other sites, detailed bathymetry of the area was taken from a chart or acoustic survey of the area using 25-m cell resolution. Where dGPS was used for sampling stations or cage group positions, incorporation into the model grid by converting to Ordnance Survey coordinates was necessary. Where dGPS was not used, distances were incorporated into the grid using geometry, and where sampling stations were duplicated, an average position was used.

Site-specific data were used where possible. Food water content (9%), digestibility (85%), wastage rates (3%), settling velocities of feed (where fish weight was unknown—10.8 cm/s), faeces (mean = 3.2 cm/s, standard deviation = 1.1 cm/s) and horizontal and vertical dispersion coefficients (k_x, k_y = 0.1 m^2/s, k_z = 0.001 m^2/s) were the same across all sites. Justification for standardising these data in this manner is related to the future application of the model, discussed later. All available current data for each site were used in the model to represent shear in the water column, implemented as hourly averages. At three of the six sites, two current data sets were available (surface and near bed), with the remaining sites having three data sets all with a minimum of 15 days in length. Feed input data 3 months prior to the benthic survey date were used if available, otherwise feed input for the benthic survey month was used. Cages in the same group were assumed to have equal feed input with no empty cages. If feed input was varied month to month, feed pellet size and thus settling velocity were also varied according to the mean fish weight for the month being modelled. Using fish weight/feed pellet size tables supplied by BOCM Pauls, a feed pellet settling velocity could be determined using the relationship in Fig. 2. A particle step was evaluated every 60 s, and this optimum value was determined by the fastest settling velocity and the number of current velocity layers used. The number of particles used for each site was variable but always greater than 5×10^5 particles. Either a mean tidal height was added to the lowest astronomical tide depth in the grid or survey depth used with no changes in elevation modelled.

The resuspension model used parameters validated in a tracer study described in Cromey et al. (in press). Critical shear stress for resuspension and deposition were set at 0.0179 and 0.004 N/m^2, respectively ($\approx$ 9.5 and 4.5 cm/s near-bed current speed), with an erodibility constant of 7×10^{-7} kg/m^2/s. Predictions of S_{avail} were obtained at sampling stations for all of the sites studied including resuspension. It is this accumulation of material that is deemed to be available to the benthic community in addition to natural background sedimentation. In the resuspension model, the consolidation time parameter was not validated in the tracer study as no vertical movement of tracer was measured in the sediments. For each particle on the seabed, the time at which a particle becomes unavailable for resuspension is the consolidation time. This parameter represents removal of material in the model by both biological (e.g. burial, consumption) and physical processes effecting the deposited material (e.g. consolidation increasing with sediment age). The consolidation time parameter was varied within acceptable limits (hours to weeks) using these resuspension parameters to obtain the best fit between benthic indices and S_{avail}. The consolidation time parameter was expected to be of the order of several

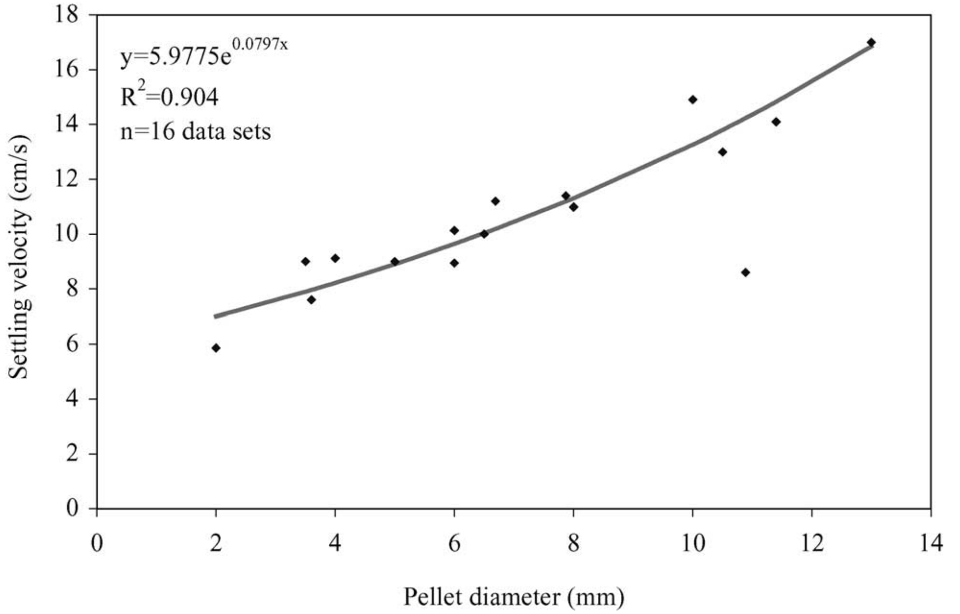

Fig. 2. Relationship between pellet diameter and settling velocity with each point representing the mean of a data set. The mean and standard deviation of all 16 data sets is 10.8 and 2.7 cm/s, respectively.

days for the environment studied here, rather that at the extremes of the acceptable limits used in the validation.

Benthic data were obtained following SEPA (2000b) seabed monitoring guidelines using a 0.1-m^2 van Veen grab. The biota was sorted to species where possible and calculations of the Infaunal Trophic Index (ITI) and total individual abundance were obtained. ITI attempts to describe numerically the dominance of the benthic fauna by dividing them into four groups based on what type of food is eaten, where and how it is obtained. The ITI was developed in California, USA and was first published in 1978 (Word, 1978). Since then, it has been adapted for use in UK waters (Codling and Ashley, 1992), but the principles remain the same (see Appendix A.4, A.5 for details). Much debate surrounds the suitability of the ITI as a benthic descriptor, viz. plasticity of feeding modes, incorrectly assigning groups to species, inclusion or exclusion of nematodes (Maurer et al., 1999). Here, the general suitability of this index is not reviewed, but ITI is included in the modelling as the quantitative relationship between it and a solids accumulation gradient was acceptable. Total abundance refers to the total number of individuals among all species per square meter.

Model predictions, which are lines of best fit through empirical data, were determined for ITI and total individual abundance. An envelope of acceptable precision (EAP) removes the need for use of statistics such as standard deviation and can vary in width depending on the position along the solids accumulation scale (x-axis). The width of the EAP was set to both the natural variability of background fauna at reference stations and between duplicate grab samples.

2.5. Overall model sensitivity analysis

During model validation, sensitivity analysis is particularly important to increase confidence in model performance. In addition, where the accuracy of input data are uncertain, the sensitivity of varying these input data must be tested. Where model parameters such as particle starting position cannot be changed by the end user, its sensitivity must also be established. A number of variable/parameter types were tested: (1) variables readily measured by the user between sites, e.g. current meter accuracy, (2) variables the user may set inappropriately in the model, e.g. grid cell resolution and (3) parameters the user cannot change, e.g. randomness.

In this test of the overall model, both solids accumulation (S_{avail}) and ITI were used to test the effect on resuspension and benthic response model predictions (Table 3). Resuspension model parameters such as critical shear stress for erosion and deposition were tested for sensitivity in Cromey et al. (in press). Predictions were used for site A for seven benthic sampling stations (L1 to L7), four sediment trap positions and three other positions in the model grid at distances of 50, 75, and 75 m from the cages. This total of 14 grid positions had a range of solids accumulation from maximal underneath the cages to just outside the limit of the deposition footprint. Comparing the sensitivity on both S_{avail} and ITI predictions is important, as for a particular station, the significance of the difference for S_{avail} between a test and a control run will be dependent on what end of the scale the S_{avail} predictions occupy. For example, a mean difference of 100 g/m^2/year between a test and a control run will have a greater effect on the ITI prediction if the S_{avail} prediction is between 0 and 500 g/m^2/year (where ITI changes rapidly with S_{avail}) as opposed to if it is between 3000 and

Table 3
Model variables and parameters tested for sensitivity (control conditions are k_x, k_y and k_z of 0.1 and 0.001 m^2/s, respectively, hourly averaged current data with a trajectory evaluated every 60 s, grid cell resolution 25 m, $n = 14$ stations)

Model variable/parameter tested	S_{avail} (g/m²/year)		ITI		
	α	γ	α[a]	γ[a]	% of stations in control EAP
Current meter speed accuracy (over reading)[b]	12869	1621	65.9	3.9	57
Grid cell res. = 50 m	10125	1399	36.0	4.0	57
k_x, k_y increase (0.5 m^2/s)	8634	1341	26.3	4.2	79
Current speed + 15%[c]	6593	1159	27.1	3.4	86
Trajectory evaluation = 600 s	6339	1041	16.8	3.0	86
Current meter dir. accuracy (− 10°)[d]	2808	428	10.6	2.2	93
Current speed − 15%	5883	1097	6.0	1.8	100
Current meter speed accuracy (under reading)[e]	6663	1179	2.7	1.1	100
Random 4[f]	586	282	2.6	1.2	100
k_x, k_y decrease (0.02 m^2/s)	1041	433	2.4	1.2	100
Random 1	708	312	2.1	< 1	100
Grid cell res. = 10 m	1589	493	2.0	< 1	100
k_z increase (0.01 m^2/s)	361	208	1.8	< 1	100
Current meter dir. accuracy (+ 10°)	696	273	1.4	< 1	100
Random 3	488	252	1.2	< 1	100
Random walk model off	1353	488	1.1	< 1	100
Trajectory evaluation = 6 s	234	159	1.1	< 1	100
Random 5	400	239	< 1	< 1	100
Random 2	358	223	< 1	< 1	100
k_z decrease (0.0001 m^2/s)	264	211	< 1	< 1	100

[a] α and γ are statistics which summarise the difference between control and test run predictions defined in Appendix A.5 (α = goodness of fit, γ = mean absolute deviation).
[b] − 5% of actual speed or − 2 cm/s, whichever is greater.
[c] Current scaled up by 15%.
[d] Direction adjusted by subtracting 10°.
[e] + 5% of actual speed or + 2 cm/s, whichever is greater.
[f] Random 1–5 are duplicate runs of the control run, indicating the effect of randomness in the model.

10 000 g/m²/year (where ITI changes little with S_{avail}). The statistics used to determine the sensitivity of the variable are given in Appendix A.5. For each station, a check was also made to see if the predicted ITI for the test run fitted within the EAP of the control run. If this was the case, then the prediction for the test run had not changed significantly.

3. Results

3.1. Laboratory experiments

Data from the faecal settling experiment show a range of settling velocities between 1.5 and 6.3 cm/s with a mean and standard deviation of 3.2 and 1.1 cm/s, respectively. Results from the feed settling experiments and from other sources are shown in Fig. 2, with a curve fitted showing a relationship between pellet size and settling velocity. Given that fish weight and pellet size are more readily known for a fish farm at any point in the growing

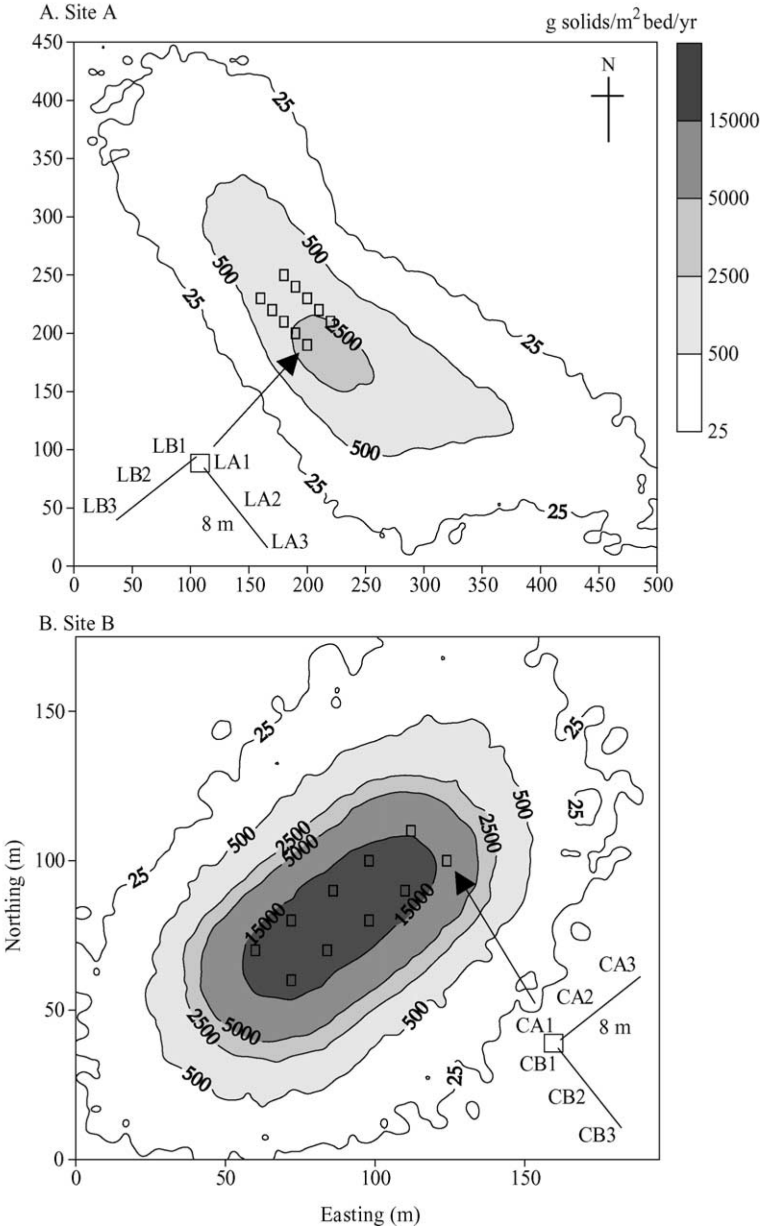

Fig. 3. Modelled solids flux at sites A and B over a 24-h period scaled up to g/m^2/year with the resuspension model turned off (cage centres are shown as □). The six sediment trap positions are shown for each site.

cycle than settling velocity of the feed being administered, this relationship may be used to calculate a settling velocity for use in DEPOMOD.

3.2. Sediment trap validation study

Fig. 3 shows model predictions of solids flux for site A and B. For site A, predictions ranged from 25 g/m^2/year to just below 5000 g/m^2/year at the SE end of the cage group. The sphere of deposition as defined by the 25 g/m^2/year contour extends 220 m to the NW and 320 m to the ESE of the group, a reflection of the observed SE residual current. The direction of this residual current is also shown by the area enclosed by the 2500 g/m^2/year contour displaced to the SE of the cage group. For site B, deposition predictions ranged from 25 to 15 000 g/m^2/year directly underneath the cage group. The sphere of deposition

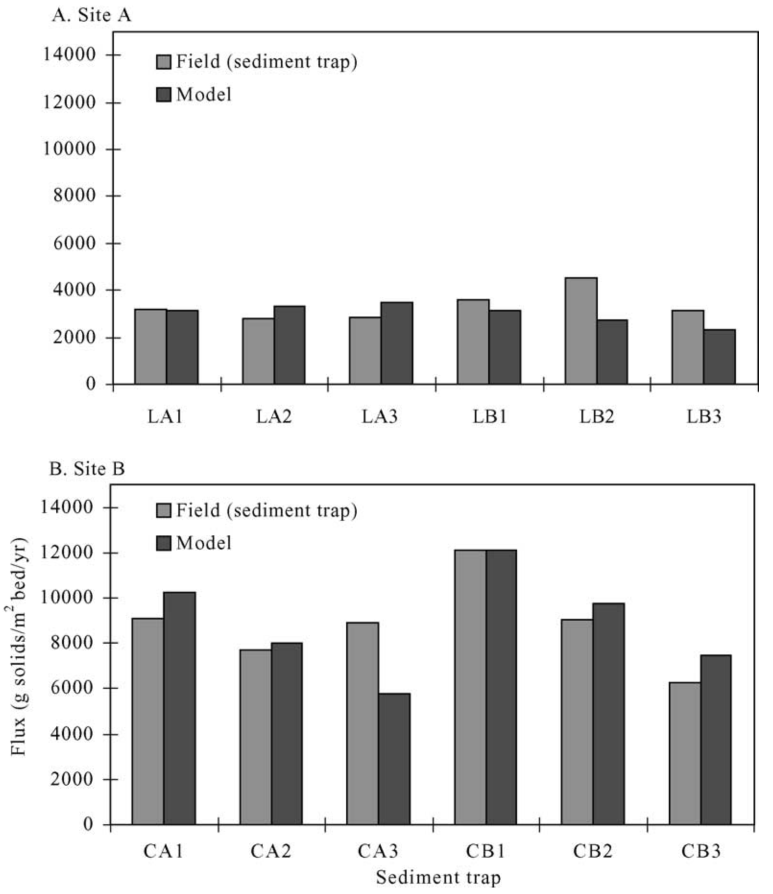

Fig. 4. Comparison between observed and modelled solids flux over a 24-h period for six sediment traps at sites A and B scaled up to g/m^2/year. Modelled data agreed with observed data on average by $\pm$ 20% and $\pm$ 13% for sites A and B, respectively. The range of this percentage error is $\pm$ 3% to $\pm$ 39% and 0% to $\pm$ 35% for the sites, respectively.

defined by the 25 g/m^2/year contour extends 75 m away from the group in all directions at site B. This is not displaced significantly, consistent with the lack of a residual current in any direction.

Comparisons between predictions and observation of deposition for both sites are shown in Fig. 4. Higher deposition was predicted along transect LA caused by the current advecting material along the NW – SE axis in the model, with trap LA3 containing the highest deposition. Transect LB had lower predicted deposition as less material was advected perpendicular to the main NW – SE axis of flow. Summarising these data gave a prediction accuracy of $\pm$ 20% on average for site A. The predicted deposition gradient from the traps at the cage centres (CA1 and CB1) to the traps at the end of the transect (CA3 and CB3) is higher at site B. For site B, the traps at the cage centres have the highest deposition as the lack of any significant residual current results in maximum deposition beneath the cages. Field data for CA3 are higher than CA2 and this has resulted in the model underpredicting for this trap. Summarising these data gives a prediction accuracy of $\pm$ 13% on average, more accurate than for site A.

3.3. Benthic module

Observed relative abundance of each of the ITI groups plotted against predicted S_{avail} is shown in Fig. 5. Group 2 organisms generally had a higher abundance than group 1 organisms for most of the observed data, in particular, where S_{avail}

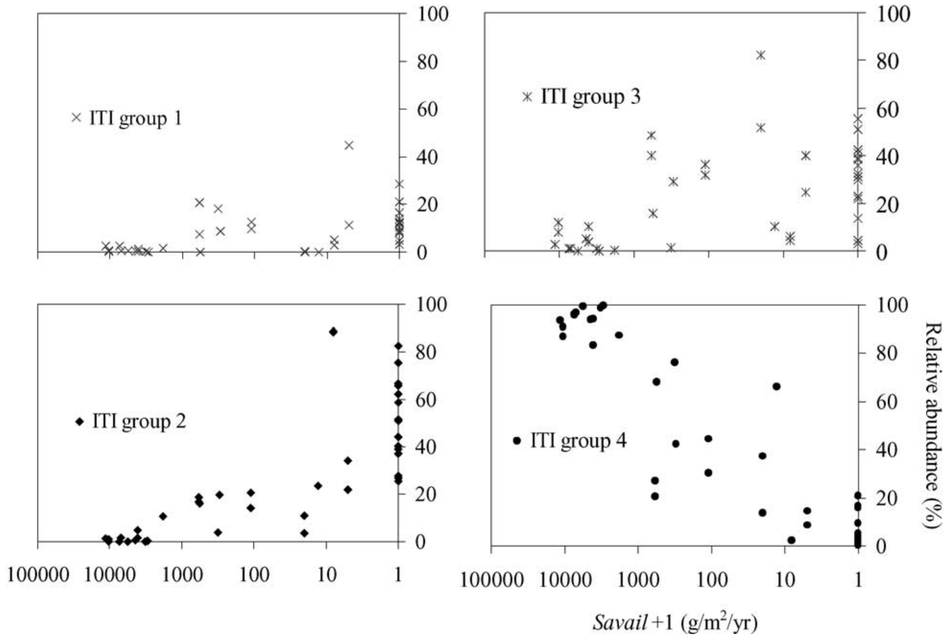

Fig. 5. Modelled solids accumulation (S_{avail}) and observed relative abundance of ITI groups for all sites studied. Each point represents a station sampled with a 0.1-m^2 grab.

predictions were 0 at reference sampling stations and stations beyond the deposition footprints. Beyond 2000 g/m^2/year, relative abundance of groups 1 and 2 is less than 10%. Patterns for the group 3 organisms are less clear, with a wide spread of relative abundance between 0% and 60% for values of accumulation between 1 and 1000 g/m^2/year and less than 15% for above 2000 g/m^2/year. Group 4 organisms mainly comprised of deposit feeders show a reasonably well-defined pattern, with relative abundance above 80% for greater than 1000 g/m^2/year. Some presence of this group was also found at background levels, with relative abundance of less than 20% being observed. Relationships between observed ITI and abundance and predicted solids accumulation for sites investigated are shown in Fig. 6. The ITI curve shows a steep

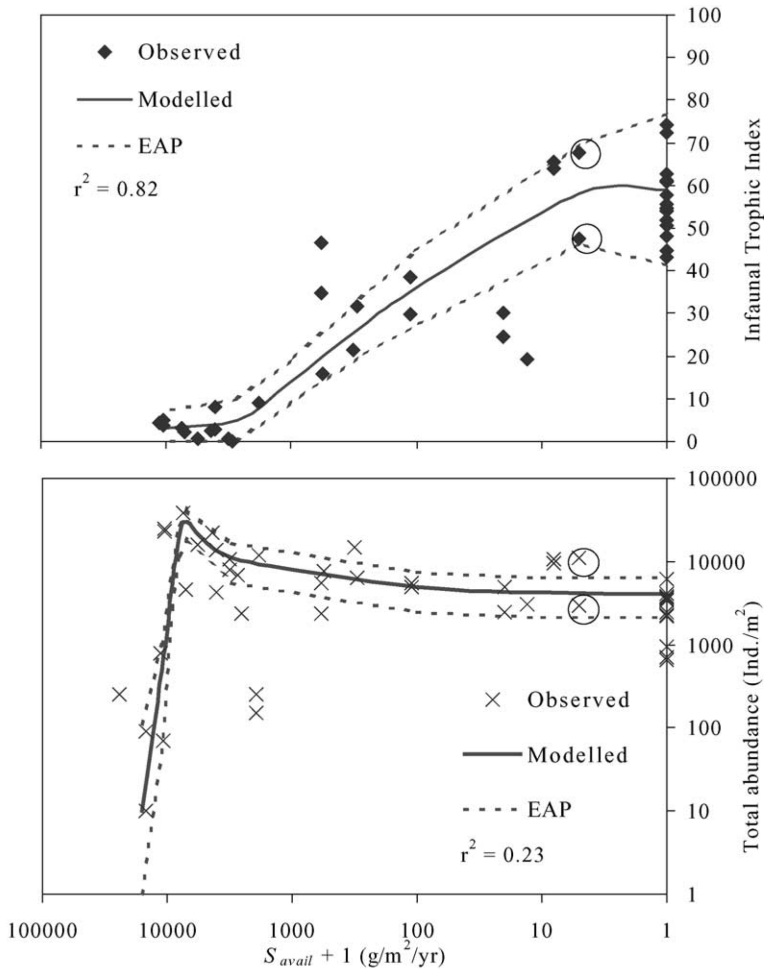

Fig. 6. Modelled relationships for ITI and total abundance with solids accumulation (S_{avail}). Circles exemplify the variation in benthic composition of duplicate grabs, and the width of the envelope of acceptable precision (EAP) is set to this variation (ITI: 88% in EAP, $n = 42$; total abundance: 68% in EAP, $n = 50$).

decrease in ITI between 10 and 1000 g/m^2/year with 88% of the data in the EAP. Although the abundance relationship has only 68% in the EAP, it shows a peak in abundance followed by a sharp decrease in abundance where solids accumulation levels are very high. In both relationships, large variation is shown for reference sampling stations at zero solids accumulation ($S_{avail} = 0$ g/m^2/year). The consolidation time parameter was set at 4 days with minimal sensitivity found for this parameter when varied by ± 2 days.

3.4. Overall model sensitivity analysis

The results of the sensitivity analysis are shown in Table 3. Comparing the results for solids accumulation and ITI shows that current speed measurements (greater than actual measurements), coarse grid cell resolution, horizontal dispersion coefficient increases and trajectory evaluation of 600 s significantly affected predictions. For these variables, a mean absolute deviation of S_{avail} and ITI between test and control runs is greater than 1000 g/m^2/year and 3 ITI, respectively. Current meter measurements, which were reduced from actual measurements, current meter direction accuracy and decrease in horizontal dispersion coefficients showed some sensitivity where mean deviations for ITI were between 1 and 3. Fine grid cell resolution, random walk model off, time step evaluation of 6 s and decrease in vertical dispersion coefficient (k_z) showed little sensitivity in the model with a mean absolute deviation of less than 1 ITI unit.

4. Discussion

4.1. Sediment trap study

Comparisons of predictions and observations from six sediment traps for site A were satisfactory with an accuracy of $\pm 20\%$. For reasonably flat bathymetry, fast settling velocities and hydrodynamics with a range of current amplitudes, model predictions will generally show a gradient from high to low deposition areas without any major patches in between. Significant model patchiness could be caused by variable bathymetry, insufficient particle numbers or where there are similar observed current amplitudes for a period of time with little change between different tidal phases. Trap LB2 which was on a transect perpendicular to the flow had deposition underpredicted by the model. Assuming that no errors were made in the field experiments in terms of trap position or collection and measurement of trap contents, the poor prediction of LB2 can be attributed either to natural patchiness, incorrect grid resolution or absence of a process in the model. Particle numbers and grid resolution were optimised and so these can be eliminated, but cage movement in the current due to slack moorings was not modelled. This movement would most likely displace the cages along the NW – SE axis and not perpendicular to this axis making little difference to predicted deposition at trap LB2. The model assumes constant release of faecal particles over a 24-h period, and this may be a limitation in the model. However, defining a period of release of faecal material would be difficult, and if this was a factor in affecting predictions at LB2, then this would necessitate a large release of faecal material when the current direction was southwesterly. As this current direction was seldom observed during the study at site A, it is unlikely that defining release times of faecal material would improve predictions at LB2. To conclude, the underprediction at LB2 cannot be easily explained from a modelling perspective and is equally as likely to be caused by natural patchiness as by experimental or modelling errors.

Model predictions and sediment trap data for site B compared well with an accuracy of $\pm$ 13%. The least accurate prediction was for trap CA3 which was situated furthest away from the cages in a NE direction. As the current amplitude is low at site B, the random walk model has a significant effect on dispersion. In addition to the potential errors in the modelling study described for site A, the random walk model variables could be a factor causing underprediction in the model at trap CA3. Changing the horizontal and vertical dispersion coefficients would increase dispersion, and a better fit between CA1 and CA3 might be achievable. However, to fit the model to observed data by varying the dispersion coefficients to a number of decimal places would be too empirically based, especially as the resulting values are likely to be higher than the coefficients used at site A which is a much more dynamic site. More likely, inaccuracy in these predictions could be caused by inaccurate positional information on cages and cage movement, as these factors will have more of an effect at site B where deposition gradients are steeper.

4.2. Benthic module

DEPOMOD has predictive capability for the benthic indices ITI and total individual abundance determined from validation of the benthic module at several Scottish fish farms. The EAP method has proved useful as it avoids the use of potentially misleading statistics when specifying the reliability of model predictions. If the observed data fall within the EAP, then the model prediction can be considered to agree with observations.

From our observations and model validation of the ITI, 88% of the benthic data were contained within the EAP, and inaccurate predictions of S_{avail} was the cause of five points on the graph falling outside the EAP (see Fig. 6). One of these stations (duplicate grabs taken) was from a site where the benthic faunal abundance was very patchy along the transect, and this variability could not be predicted by the model. Additionally, it was necessary to place the transect perpendicular to the main axis of flow due to depth considerations. This resulted in the transect being along a very steep enrichment gradient making accurate station positioning crucial and being a potential source of error. The second station (at a different site) which did not fit into the EAP was also sampled by duplicate grabs. This site was heavily impacted with the majority of the near cage stations too heavily impacted to be included in the ITI analysis (number of species ≤ 5). The station at this site, which did not fit into the EAP, was 50 m from the cages, and closer examination of this site revealed that the current measurements undertaken at the site supplied by the farmer did not agree entirely with observations at the cages. The positioning of the current mooring at this site may have been incorrect, even though the 100-m distance of the mooring from the cages was within SEPA (2000a) guidelines. No explanation can be given for the third station as no faults can be found with the data used; thus, the inaccuracy of the model cannot be explained for this station. These anomalies highlight the importance of station positioning and current meter measurements.

The relationship between total individual abundance and S_{avail} has a wide EAP, where most of the predictions have an EAP of $\pm 40\%$ of the prediction. This wide EAP was caused by the wide variability in total abundance observed across all the sites investigated and by the model having less capability for predicting this indicator than for ITI. Despite this limitation and the wide EAP, general features such as the peak in total abundance immediately before near azoic conditions in severely impacted conditions can be predicted. Although only 68% of observations were in the EAP and there are numerous outliers, the inclusion of total individual abundance with solids deposition in DEPOMOD is a useful addition to its predictive capability which has not been achieved in the BenOss model (Cromey et al., 1998) or other currently available fish farm deposition models.

Model performance for both the ITI and abundance relationships including the EAP is shown in Fig. 7. At the impacted end of the scale, the predictive capability of ITI is higher than abundance characterised by the narrow ITI EAP and the amount of data inside the EAP in this zone. Abundance has a narrower EAP at the zero end of the S_{avail} scale than for the impacted zone as the variability of abundance in the impacted zone is much higher and more difficult to predict.

A suite of other benthic diversity indices was tested for predictive capability in the model. Number of species, Shannon diversity, Pielou's evenness and Margalef's richness were not used because it was difficult to describe a relationship with solids accumulation in a single curve, as established for ITI and individual abundance. The number of species, for example, at reference stations was widely varying from 25 to 65. Such variability in the biotic conditions at reference stations means that other indices dependent on number of species were likewise difficult to use predictively. Prediction of these indices versus solids accumulation would require prior knowledge of reference station benthic fauna and a number of curves to describe the benthic response, likely to differ according to pre-farming conditions. Benthic biomass was not used as a predictive index as it was not calculated in many of the data sets available to the authors. The inclusion of the ITI over other traditional indices in the modelling does not imply that the ITI is a superior index.

4.3. Model sensitivity analysis

Accurate current speed measurements were shown to be fundamentally important in obtaining accurate model predictions in both the sediment trap and benthic module validation. In this test, using scaled currents which are greater than the actual currents has a greater effect on model predictions than using scaled currents less than the actual. When using scaled currents greater than the actual, a greater frequency and magnitude of resuspension events were predicted in the resuspension model. For stations at the high end of the S_{avail} scale, the increase in resuspension resulted in a reduction in S_{avail} to less than 3000 $g/m^2/year$ where ITI values change markedly with S_{avail} (Fig. 6). Conversely, where scaled currents lower than actual were used, a decrease in resuspension was predicted. For stations occupying the zone greater than 3000 $g/m^2/year$, this resulted in an increase in S_{avail} but little change in ITI as ITI changes slowly with respect to S_{avail} for predictions greater than 3000 $g/m^2/year$. The shape of the ITI and S_{avail} curve therefore explains the

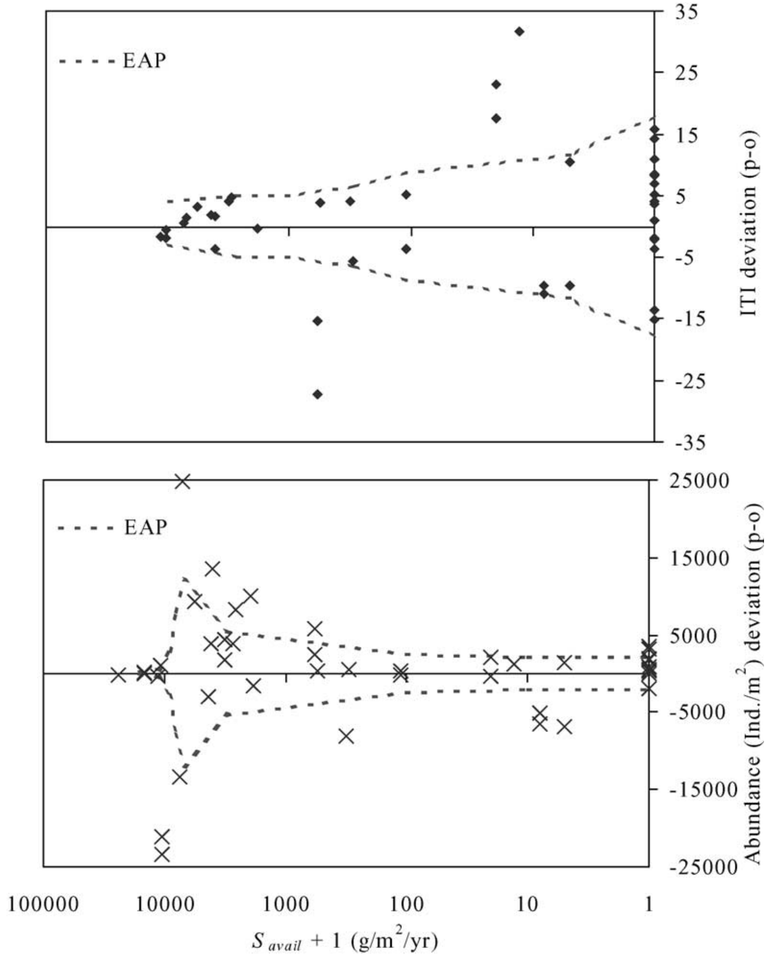

Fig. 7. Model performance shown by deviation of predicted (p) from observed variable (o) plotted against predicted solids accumulation (S_{avail}). Model performance is higher for ITI at the impacted end of the scale (>1000 g/m^2/year), but abundance predictions are less reliable in this zone.

lower level of sensitivity detected using currents less than true observations. The sensitivity of current meter measurements is also likely to vary between sites. For a depositional site such as site B, the effect of increasing the current amplitude by 10% would be minimal, but an increase of 3 cm/s would significantly change the deposition footprint. The accuracy of the current direction measurements is equally important, as errors will compound at increasing distances from the source.

Coarse grid cell resolution was also found to be sensitive in the model as predictions at stations are calculated from interpolation between adjacent grid nodes. Coarse resolution of 50 m decreases model accuracy as this is not sufficiently fine to accurately determine predictions at stations along such a steep deposition gradient. Although using finer grid resolution did not show any sensitivity, the number of particles must be increased when using fine resolution (< 10 m) to avoid model patchiness.

A particle trajectory evaluation time step of 600 s showed some sensitivity which can be explained by examining the settling velocity of the waste material. Food pellets settling at a rate of 9.5 cm/s will sink 57 m in 600 s. If the particle is moved to this vertical distance in one step, any change of current speed and direction with depth implemented in the model to represent current shear will not be applied to the particle as it settles. The appropriate time step that should be used is dependent on the settling velocity of the wastes being modelled, where a time step of 60 s is suitable for these wastes. This results in food pellets sinking 5.7 m every time step, which is adequate to allow accurate representation of the effects of shear on the particle trajectory for most sites. The use of a 6-s time step only increases computational time making little difference to model predictions.

Some sensitivity was seen using large horizontal dispersion coefficients. As dispersion coefficients are increased, the step length in the random walk model becomes larger in relation to the advection step. For example, for current speeds of 5, 10 and 20 cm/s, a particle will be transported 3, 6 and 12 m in 60 s. The length of the random walk model step for a k_x of 0.5 m^2/s is 7.7 m (i.e. $\sqrt{2 \times 0.5 \times 60}$)), and this is large in relation to the advection step for site A, which had a mean current speed of less than 10 cm/s. This value of 0.5 m^2/s, however, is unrealistically too high to use as a general value for any of the sites studied. Although dispersion coefficients are commonly used in dispersion models of all types of discharges, dispersion coefficient data have been more commonly measured for long sea outfalls in coastal environments than for fish farm discharges in sea lochs. Understandably, the reluctance of farmers to allow dye studies near farms has been a factor, but the recent development and use of dGPS drifters has begun to alleviate the situation. Vertical dispersion coefficients were found to be insensitive as the settling velocity of the waste particles is high in relation to the vertical steps calculated by the random walk model.

Testing the sensitivity of particle starting position in the cage was undertaken at an early stage of model development and, as a result, starting positions were assigned as random for benthic module validation. Although some observational evidence suggests that defecation from fish generally occurs at the surface directly after feeding, it is unlikely that any particular spatial distribution in the cage could be modelled accurately. In addition, depending on cage design and biofouling, food pellets may have a starting position close to the centre of the cage bottom due to the pellets rolling towards the centre before leaving the cage. Considering all these factors and the sensitivity of this parameter, it is prudent to assign random starting positions in the cage for continuity across all sites.

Some caution is required when assigning model parameters at the validation stage to obtain a best fit between predicted and observed data which subsequently cannot be changed by the user (e.g. critical threshold for resuspension, consolidation time parameter). Although this implies that these parameters are site-specific, varying of the parameters within limits set according to literature values, subsequent sensitivity analysis and testing at other sites justifies cautious use of the same parameters across all sites. A requirement for the user to determine critical threshold for resuspension, for example, for every site modelled would severely restrict use of the model. Indeed, the literature showed that this has been measured for only a few fish farm sites globally. Other parameters, such as the time consolidation parameter, could be used in a more site-specific manner if required. For example, decreasing its value to describe increased bioturbation would cause particulate material to be removed from the resuspendable fluff layer more quickly. Ideally, this parameter would be varied spatially across the model grid according to the abundance of different types of bioturbators surrounding the farm, but would not easily be validated.

Overall, the most sensitive variables could be defined as those that resulted in predictions being outside the ITI EAP for the control run. This would indicate that these variables have resulted in ITI model predictions differing from the control run by an amount greater than the natural background variation of the ITI (Table 3). Using this definition, the following order of sensitivity was determined on overall model predictions: current speed (over)>coarse grid cell resolution>current speed $(+15\%)$>k_x, k_y increase>time step evaluation (600 s)>current direction (-10°) with all other variables being insensitive.

4.4. Application and limitations of the model

DEPOMOD has been validated for numerous fish farms in Scottish coastal and sea loch systems. The model has been applied across a range of different scenarios in terms of the environmental conditions and farm size. Importantly, the model has also been tested for sites where a range of hydrodynamic conditions and bathymetry were measured.

DEPOMOD has uses in planning and monitoring of existing and proposed fish farms, trial studies and research-orientated applications. A Fish Growth and Biomass model (GaBoM) was developed using fish growth and mortality functions to predict feed input for a growing cycle, given initial data on the farm. Although model validation reproduced some encouraging results, variables such as harvesting are difficult to predict as this often takes place according to market demand. Prior to a survey, DEPOMOD may also be used to predict the deposition footprint, thus enabling confident siting of sampling and reference stations. Post-surveying the existing farms, the model may be used to compare predicted and observed benthic indices. As confidence is established in the model, future impacts can be predicted. Field trials to compare predictions and observations of particulate discharges can also be undertaken. Such studies may be similar to the 24-h sediment trap study described, or may be a comparison of sediment concentrations of a chemical discharged over a longer period. For such studies, it is necessary to obtain accurate input data at a frequency suitable for the period of the study, with a hydrodynamic study over the length of the study often being essential. DEPOMOD is currently being developed to model carbon degradation and oxygen demand of sediments, as well as being validated for the Mediterranean situation.

As a model reaches a satisfactory level of validation, certain limitations will always exist. The presence of extremely steep underwater gradients at some sites causes periodic slumping of material deposited further down the slope (Klaucke et al., 2000). This is likely to result in a complex footprint and associated impacts which is beyond the predictive capability of the model. Shallow exposed sites where the depth underneath the cages is less than 15 m may be subject to a large degree of wind-wave resuspension caused by orbital fluid velocities. DEPOMOD does contain a wind-wave resuspension model validated in BenOss (Cromey et al., 1997), but a different set of parameters would probably be needed for modelling of fish farm waste. Wind-wave resuspension typically results in sediment of high average grain size, compared to low grain size in low current depositional areas. Infaunal animals inhabiting fine sediment areas are adapted to ingesting the organic-rich small sediment particles. As their feeding apparatuses are unable to cope with the larger grain size, different communities than those found in soft sediment areas result (Levinton, 1982). The model therefore has limited scope in areas dominated by wind-wave resuspension. Where areas of redeposition are predicted at distances from the farm, these need to be evaluated as the model does not have a spatially varying hydrodynamic grid. Hydrodynamic conditions in these areas may be quite different from those near the cages. As validated three-dimensional hydrodynamic models become available for Scottish fjordic systems, so will models such as DEPOMOD be developed to interface with such models. However, given these limitations of DEPOMOD, its main virtue is that it was designed from the outset to require data readily available to the target user community, without eliminating fundamental processes. This is often overlooked when developing research-based models for integration into practical applications in industry.

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Appendix A. Description and equations for the principal model

A.1. Grid generation module

The algorithm in the grid generation model interpolates linearly between grid nodes to determine the depth at each zoomed grid node in the area of interest. This method generally generates accurate results ($r^2 = 0.98$, $n = 424$, $\alpha_{2\ \mathrm{tailed}}$, $y = 0.98x$, where y is predicted depth and x is observed depth).

A.2. Particle tracking module

For a given feed input (F), the wastage rates of food and faecal material are:

$$F_c = F(1 - F_w)(1 - F_{\mathrm{wasted}}) \tag{1}$$

$$W_{\mathrm{fd}} = F(1 - F_w)F_{\mathrm{wasted}} \tag{2}$$

$$W_{\mathrm{fae}} = F_c(1 - F_{\mathrm{dig}}) \tag{3}$$

where F_c (kg/day/cage) is the rate of food consumption according to the feed input, the water content of the food (F_w) and the proportion of food wasted (F_{wasted}); W_{fd} (kg/day/cage) is the rate of food loss without consumption and W_{fae} (kg/day/cage) is the rate of faecal material production given the rate of food consumption and digestibility (F_{dig}).

In modelling the horizontal trajectory of a particle, it is convenient to consider the current as a sum of two parts: a slowly varying component relating to tidal or wind forcing plus a more rapidly varying component relating to turbulence with a mean of zero. Given a time step (t) and a position of a particle defined as $P_{(x,\ y,\ t)}$ and velocity components u and v (m/s), then a trajectory can be defined as:

$$P_{(x,y,t+1)} = P_{(x,y,t)} + u_{(z,t+1)}\delta t + rw_{\mathrm{step}(x)} + v_{(z,t+1)}\delta t + rw_{\mathrm{step}(y)} \tag{4}$$

For a settling velocity of vs (m/s), the vertical step can be defined as:

$$P_{(z,t+1)} = P_{(z,t)} + \mathrm{vs}\delta t + rw_{\mathrm{step}(z)} \tag{5}$$

The length step in the random walk model (turbulence) is dependent on the time the particle is in the turbulent field and a dispersion coefficient (Allen, 1982):

$$rw_{\mathrm{step}}(x) = rw_{\mathrm{dir}}\sqrt{2k_x\delta t} \tag{6}$$

where $rw_{\mathrm{step}}(x)$ is the size of the step in the x direction (m), rw_{dir} is the step direction + or $-$ (determined from a random number generator), k_x is a dispersion coefficient (m^2/s) and δt is the time in the turbulent field (s).

A.3. Resuspension module

Resuspension models typically consist of four components: erosion, transport, deposition and consolidation. The erosion function used in DEPOMOD is:

$$M_e = M\left(\frac{\tau_b}{\tau_{ce}} - 1\right) \tag{7}$$

where M_e is the mass resuspended (kg/m^2/s) when the bed shear stress τ_b exceeds the critical shear stress for erosion τ_{ce} and M is an erodibility constant (kg/m^2/s).

Bed shear stress is calculated from the bed shear velocity U_* (m/s) in the following relationship (Bowden, 1983):

$$\tau_b = \rho U_*^2 \tag{8}$$

where ρ is the density of seawater ($\approx$ 1025 kg/m^3). Using a logarithmic profile in the boundary layer (Bowden, 1983) and near bed measured current velocity $U_{(z)}$ (i.e. $z = 2$ m), U_* can be calculated by the following equation (Dyer, 1979):

$$U_* = \frac{\kappa U_{(z)}}{\ln(z/z_o)} \tag{9}$$

where κ is the von Kàrmàn constant (0.4) and z_o is 2×10^{-4} m for a muddy bottom (Soulsby, 1983).

Particles eroded from the bed are transported in the water column until redeposition occurs when the bed shear stress falls below a critical shear stress for deposition:

$$M_d = (\text{vs*pe})\left(1 - \frac{\tau_b}{\tau_{cd}}\right) \tag{10}$$

where M_d is the rate of deposition (kg/m^2/s) and vs and pe are settling velocity (m/s) and particle concentration (kg/m^3), respectively.

Consolidation (F_c) is related to turnover times of sediment (t_{max}) so that for $t' > t_{max}$:

$$F_c = m_p \tag{11}$$

where m_p is the mass of particles (g/m^2) on the bed which have been deposited for t' (days).

A.4. Benthic module

The predictions of solids accumulation (S_{avail}) used in the benthic module are a result of the following mass balance:

$$S_{avail} = M_f + M_d - M_e \tag{12}$$

where S_{avail} = flux of solids arising from fish farm available to the benthic community (g solids/m^2/year), M_f is deposition flux arising directly from the farm, M_d is deposition flux arising from resuspended material originating from the farm and M_e is the resuspension flux. S_{avail} does not include carbon from primary production nor does it take account of carbon degraded by the G-model (Westrich and Berner, 1984).

The calculation of the Infaunal Trophic Index (ITI) (Mearns and Word, 1982) is as follows:

$$\text{ITI} = 100 - 33.3\left(\frac{0n_1 + 1n_2 + 2n_3 + 3n_4}{n_1 + n_2 + n_3 + n_4}\right) \tag{13}$$

where n_i is the abundance in trophic group i.

A.5. ITI groups

Group 1 (suspension feeders): these animals feed on detritus from the water column and usually lack sediment grains in their stomach contents, e.g. *Spio*, *Spiophanes*, *Sabella*, *Ampelisca*, *Corophium*, *Phaxas pellucidus*, *Mya arenaria*, *Ophiothrix fragilis* and *Amphiura filiformis*.

Group 2 (interface/surface detrital feeders): these animals obtain the same types of food as suspension feeders but usually from the upper 0.5 cm of the sediment, e.g. *Nephtys incisa*, *Levinsenia gracilis*, *Polydora*, Cirratulidae, Scalibregmatidae, *Photis*, *Mysella* and *Ophiura*.

Group 3 (deposit feeders): these invertebrates generally feed from the top few centimetres of the sediment and feed on encrusted mineral aggregates, deposit particles or biological remains. While carnivores have been classified as Group 2 by Word (1980), they are included here in Group 3 as in Codling and Ashley (1992), e.g. *Anaitides*, *Goniada maculata*, *Nephtys hombergii*, *Scoloplos armiger*, *Nucula* and *Thyasira*.

Group 4 ("specialised environment" feeders): mobile burrowers that feed on deposited organic material. While exhibiting variable feeding behaviour, they are all adapted to live in highly anaerobic sediment, e.g. *Ophryotrocha*, *Schistomeringos*, *Capitella capitata*, *Notomastus latericeus*, Oligochaeta and *Bittium*.

The following coefficient (α) which is a measure of the goodness of fit and mean absolute deviation (γ) were used to determine the sensitivity of a variable:

$$\alpha = \sum_{i=1}^{n} \frac{(t-c)^2}{c} \tag{14}$$

$$\gamma = \frac{\sum_{i=1}^{n} \mathrm{abs}(t-c)}{n} \tag{15}$$

where c is the predicted S_{avail} or ITI at a station for the control run, t is the prediction for the test run with the variable or parameter changed and n is the total number of stations compared.

A.6. In-feed treatment module

The rate of wastage of the active ingredient of the in-feed medicine in the waste food ($W_{\mathrm{fd(Ing)}}$) is:

$$W_{\mathrm{fd(Ing)}} = W_{\mathrm{fd}}\mathrm{Ing}_{\mathrm{fd}} \tag{16}$$

where $\mathrm{Ing}_{\mathrm{fd}}$ is the concentration of ingredient on the feed (i.e. g ingredient/kg medicated feed). The concentration of the active ingredient of the in-feed medicine which is consumed is $F(1-F_{\mathrm{wasted}})\mathrm{Ing}_{\mathrm{fd}}$, and the amount excreted is this quantity multiplied by $\mathrm{Ing}_{\mathrm{excr}}$, where $\mathrm{Ing}_{\mathrm{excr}}$ is the proportion of consumed ingredient excreted by the fish.

Therefore, for a given medicated feed input (F), the concentration of ingredient on the faeces ($\mathrm{Ing}_{\mathrm{fae}}$) is:

$$\mathrm{Ing}_{\mathrm{fae}} = \frac{F(1 - F_{\mathrm{wasted}})\mathrm{Ing}_{\mathrm{fd}}\mathrm{Ing}_{\mathrm{excr}}}{F(1 - F_{\mathrm{w}})(1 - F_{\mathrm{wasted}})(1 - F_{\mathrm{dig}})} \tag{17}$$

which simplifies to:

$$\mathrm{Ing}_{\mathrm{fae}} = \frac{\mathrm{Ing}_{\mathrm{fd}}\mathrm{Ing}_{\mathrm{excr}}}{(1 - F_{\mathrm{w}})(1 - F_{\mathrm{dig}})} \tag{18}$$

As the concentration of the ingredient on the food and faeces is known, a mass of ingredient associated with each particle can also be assigned.

References

Ali, C.Q., Ross, L.G., Beveridge, M.C.M., 1991. Microcomputer spreadsheets for the implementation of geographic information systems in aquaculture: a case study on carp in Pakistan. Aquaculture 92, 199–205.

Allen, C.M., 1982. Numerical simulation of contaminant dispersion in estuarine flow. Proc. R. Soc. Lond. A 381, 179–194.

Bowden, K.F., 1983. Physical Oceanography of Coastal Waters. Ellis Horwood, Chichester, UK, pp. 26–32.

Brown, J.R., Gowen, R.J., McLusky, D.S., 1987. The effect of salmon farming on the benthos of a Scottish sea loch. J. Exp. Mar. Biol. Ecol. 109, 39–51.

Chen, Y.S., Beveridge, M.C.M., Telfer, T.C., 1999a. Physical characteristics of commercial pelleted Atlantic salmon feeds and consideration of implications for modelling of waste dispersion through sedimentation. Aquac. Int. 7, 89–100.

Chen, Y.S., Beveridge, M.C.M., Telfer, T.C., 1999b. Settling rate characteristics and nutrient content of the faeces of Atlantic Salmon, *Salmo salar* and the implications for modelling of waste dispersion. Aquac. Res. 30, 395–398.

Codling, I.D., Ashley, S.J., 1992. Development of a biotic index for the assessment of pollution status of marine benthic communities. Water Research Council report no SR 2995, Marlow, Bucks SL7 2HD, UK.

Cromey, C.J., Black, K.D., Edwards, A., Jack, I.A., 1997. BenOss version 2 software—Biological Effects and Organic Solids Sedimentation. UK Water Industry, 1 Queen Anne's Gate, London SW1 9BT, UK. ISBN 1 84057 003 2.

Cromey, C.J., Black, K.D., Edwards, A., Jack, I.A., 1998. Modelling the deposition and biological effects of organic carbon from marine sewage discharges. Estuar. Coast. Shelf Sci. 47, 295–308.

Cromey, C.J., Nickell, T.D., Black, K.D., Provost, P.G., Griffiths, C.R., 2000. Observations and modelling of the resuspension of a particulate tracer from a point source for impact prediction. Proceedings of Oceanology International 2000, Brighton, UK, 7th–10th March 2000. Spearhead-PGI Worldwide exhibitions LTD, Surrey, UK, pp. 68–69.

Cromey, C.J., Nickell, T.D., Black, K.D., Provost, P.G., Griffiths, C.R., in press. Validation of a fish farm waste resuspension model by use of a particulate tracer discharged from a point source in a coastal environment. Estuaries.

Dudley, R.W., Panchang, V.G., Newell, C.R., 2000. Application of a comprehensive modeling strategy for the management of net-pen aquaculture waste transport. Aquaculture 187, 319–349.

Dyer, K.R., 1979. Estuarine Hydrology and Sedimentation. Cambridge Univ. Press, Cambridge, UK.

Gillibrand, P.A., Turrell, W.R., 1997. Simulating the dispersion and settling of particulate material and associated substances from salmon farms. Aberdeen Marine Laboratory, Report No 3/97, Aberdeen AB11 9DB, UK.

Gowen, R.J., Bradbury, N.B., Brown, J.R., 1989. The use of simple models in assessing two of the interactions between fish farming and the marine environment. In: Billard, R., de Pauw, N. (Eds.), Aquaculture—a Technology in Progress. European Aquaculture Society, Bredene, Belgium, pp. 189–199.

Gowen, R.J., Smyth, D., Silvert, W., 1994. Modelling the spatial distribution and loading of organic fish farm waste to the seabed. In: Hargrave, B.T. (Ed.), Modelling Benthic Impacts of Organic Enrichment from Marine Aquaculture. Can. Tech. Rep. Fish. Aquat. Sci., vol. 1949, pp. 19–30.

Hall, P., Holby, O., 1986. Environmental impact of a marine fish cage culture. International Council for the Exploration of the Seas, CM-1986/F:46: 20 pp.

Hevia, M., Rosenthal, H., Gowen, R.J., 1996. Modelling benthic deposition under fish cages. J. Appl. Ichthyol. 12, 71–74.

Kaspar, H.F., Hall, G.H., Holland, A.J., 1988. Effects of sea cage salmon farming on sediment nitrification and dissimilatory nitrate reductions. Aquaculture 70, 333–344.

Klaucke, I., Savoye, B., Cochonat, P., 2000. Patterns and processes of sediment dispersal on the continental slope off Nice, SE France. Mar. Geol. 162, 405–422.

Levinton, J.S., 1982. Marine Ecology. Prentice-Hall, Englewood Cliffs, NJ. 526 pp.

Maurer, D., Nguyen, H., Robertson, G., Gerlinger, T., 1999. The Infaunal Trophic Index (ITI): its suitability for marine environmental monitoring. Ecol. Appl. 9 (2), 699–713.

Mearns, A.J., Word, J.Q., 1982. Forecasting effects of sewage solids on marine benthic communities. In: Mayer, G.F. (Ed.), Ecological Stress and the New York Bight: Science and Management. Columbia S Carolina Estuarine Research Federation, Columbia, S Carolina, US, pp. 495–512.

Nath, S.S., Bolte, J.P., Ross, L.G., Aguilar-Manjarrez, J., 2000. Application of geographic information systems (GIS) for spatial decision support in aquaculture. Aquac. Eng. 23, 233–278.

Panchang, V., Richardson, J., 1992. A review of mathematical models used in assessing environmental impacts of salmonid net-pen culture. J. Shellfish Res. 11, 204–205.

Panchang, V., Cheng, G., Newell, C., 1997. Modelling hydrodynamics and aquaculture waste transport in coastal Maine. Estuaries 20, 14–41.

Pearson, T.H., Black, K.D., 2001. The environmental impacts of marine fish cage culture. In: Black, K.D. (Ed.), Environmental Impacts of Aquaculture. Sheffield Academic Press, Sheffield, UK, pp. 1–31. ISBN 0-8493-0501-2.

Pereira, P.M.F., 1997. Macrobenthic succession and changes in sediment biogeochemistry following marine fish farming, PhD thesis, University of Stirling. 220 pp.

Pérez, O.M., Telfer, T.C., Beveridge, M.C.M., Ross, L.G., in press. Geographical Information Systems (GIS) as a simple tool for modelling waste distribution under marine cages. Estuar. Coast. Shelf Sci.

Ritz, D.A., Lewis, M.E., Shen, M., 1989. Response to organic enrichment of infaunal macrobenthic communities under salmonid sea cages. Mar. Biol. 103, 211–214.

Ross, L.G., 1995. Environmental modelling and management of complex datasets for aquaculture: progress in GIS. Aquac. News, 21–24.

Ross, A.H., Gurney, W.S.C., Heath, M.R., Hay, S.J., Henderson, E.W., 1993a. A strategic simulation model of a fjord ecosystem. Limnol. Oceanogr. 38, 128–153.

Ross, L.G., Mendoza, Q.M.E.A., Beveridge, M.C.M., 1993b. The application of geographical information systems to site selection for coastal aquaculture: an example based on salmonid cage culture. Aquaculture 112, 165–178.

Ross, A.H., Gurney, W.S.C., Heath, M.R., 1994. A comparative study of the ecosystem dynamics of four fjords. Limnol. Oceanogr. 39, 318–343.

Sanford, L.P., Panageotou, W., Halka, J.P., 1991. Tidal resuspension of sediments in northern Chesapeake Bay. Mar. Geol. 97, 87–103.

SCCWRP, 1992. Modification and verification of sediment deposition models. Southern Californian Coastal Water Research Project Authority, California, US, Final report Sep. 1992. 331 pp.

SEPA, 2000a. Hydrographic data requirements for application to discharge from marine cage fish farmsAttachment VIII-Regulation and Monitoring of Marine Cage Fish Farming in Scotland—a Procedures Manual. Scottish Environment Protection Agency, Stirling, Scotland. 13 pp.

SEPA, 2000b. Seabed Monitoring. Annex F-Regulation and Monitoring of Marine Cage Fish Farming in Scotland—a Procedures Manual Scottish Environment Protection Agency, Stirling, Scotland. 14 pp.

Silvert, W., Cromey, C.J., 2001. The environmental impact of marine fish cage culture. In: Black, K.D. (Ed.), Environmental Impacts of Aquaculture. Sheffield Academic Press, Sheffield, UK, pp. 154–181. ISBN 0-8493-0501-2.

Silvert, W., Sowles, J.W., 1996. Modelling environmental impacts of marine finfish aquaculture. J. Appl. Ichthyol. 12, 75–81.

Soulsby, R.L., 1983. The bottom boundary layer of shelf seas. In: Johns, B. (Ed.), Physical Oceanography of Coastal and Shelf Seas. Elsevier, New York, US, pp. 189–266.

Wassmann, P., Heiskanen, A.-S., Lindahl, O., 1991. Sediment trap studies in the Nordic countries. Proceedings of the Symposium on Sediment Traps in Marine Ecological Research and Monitoring. Kristineberg Marine Biological Station, Fiskebäckskil, Sweden, pp. 269–272. 309 pp.

Weston, D.P., 1986. The environmental effects of floating mariculture in Puget Sound. Special Report, Washington Dept. of Fisheries and Ecology, Seattle USA.

Westrich, J.T., Berner, R.A., 1984. The role of sedimentary organic matter in bacterial sulphate reduction: the G-model tested. Limnol. Oceanogr. 29, 236–249.

Word, J.Q., 1978. The Infaunal Trophic Index. Annual Report 1978. Coastal Water Research Project, El Segundo, CA, USA, pp. 19–39.

Word, J.Q., 1980. Classification of benthic invertebrates into Infaunal Trophic Index feeding groups. In, Coastal Water Research Project Biennial Report 1979–1980. SCCWRP, Long Beach, California, USA, pp. 103–121.