MERAMED – Development of monitoring guidelines and modelling tools for environmental effects from Mediterranean aquaculture

Deliverable 4.2 (WP4.2) – MERAMOD (version 1.4)
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This document contains the following information:

Model summary
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Annex A - Modelling guidelines and data specification (extracted from main guidelines document deliverable 1.2)

MODEL SUMMARY

The capabilities of the MERAMOD model are summarised as follows:

- prediction of flux – the flux or total deposition of waste material (faeces and feed) at the sea bed discharged from mariculture operations
- predict of benthic effects and impact – using semi–empirical relationships established between modelled flux (g solids m^{-2} bed yr^{-1}) and observed descriptors of macrobenthos, values of these descriptors for a particular level of flux can be predicted (species, abundance, biomass, abundance/species (A/S), Shannon Weiner (log_2), Simpson and Eh (4cm)
- relationships between the relative abundance and presence/absence of indicator species and families identified by the project were also established; generalised relationships could not be established but the presented data are useful for model users

- this model has been validated for several sea bream (*Sparus aurata*) and sea bass (*Dicentrarchus labrax*) fish farms in the Eastern Mediterranean; however the use of this model is not restricted to operations of these species, the Eastern Mediterranean environment or fish wastes

A series of newsletters were written by MERAMED partners and disseminated to MERAMED newsletter subscribers during the course of this project. These newsletters included development of the MERAMOD model (August 2003), hydrographic measurements of sites studied in the Eastern Mediterranean (November 2003) and modelling management scenarios (January 2004). These newsletters are a useful supplement to this document.

ETHOS FOR MODEL DEVELOPMENT AND VALIDATION

The flow diagram in Figure 1 was used during this project to ensure the model was developed and validated appropriately. Missing processes from the original model DEPOMOD (Cromey et al. 2002) were identified and coded into the MERAMOD model. Important aspects regarding data input were then identified specific to the environment and species being farmed and measurements undertaken appropriately. Validation of the model was then undertaken with sediment trap data from the MD8 study and from the two detailed project cruises (March, October 2002). Benthic community data measured during the scanning cruise (July 2001) were then used in validation of the benthic response model.

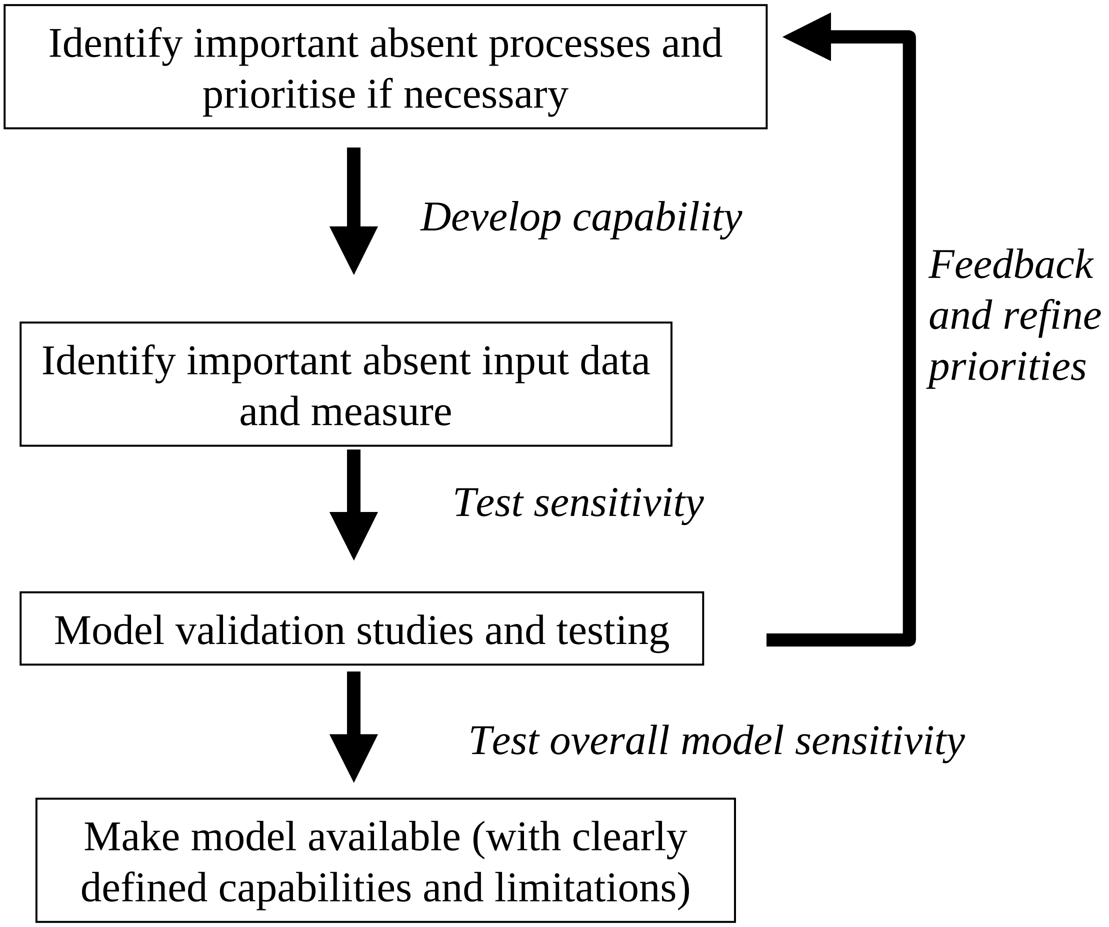


Figure 1. The process of developing an existing model and adapting for a new local environment.

MODEL PROCESS DEVELOPMENTS

Identification and prioritisation of model developments was mainly undertaken with the assistance of sensitivity analyses. These tested the importance of certain input data and model processes, thus ensuring the completed model had been appropriately tested.

1. Wild fish module

The consistent observations of wild fish feeding between sites required the model to have capability to model the effect of removal of these wastes by wild fish. In particular, the removal of uneaten feed pellets arising from the farm in both the water column and the sea bed is an important sink of material arising from the farm. The user may also specify removal of waste faecal material from the water column and sea bed, but this is likely to be a less commonly used component of the module. This module was used in model validation studies using diver observations from MERAMED workpackage 3 for each site, where data were available.

2. Cage specific data (faecal settling rates, species, feed input)

Given the sensitivity of different representations of faecal settling data (Magill et al. In review), some model development was required in the model so that species and thus faecal settling data could be specified for individual cages. Given the mixture of fish size and species within a cage group in many of the Mediterranean farms studied, this further necessitated this capability in the model. Sensitivity testing also showed that when validating model predictions with observations of 24–hour sediment trap data from spring and autumn 2002 cruises, accurate feed input data were required for adjacent cages as well as the experimental cage.

3. Diurnal patterns of feed and faecal release

There is observational evidence that a diurnal pattern exists with the release of feed and faecal material from cages. Although the release of waste feed at the site will be during specific times of feeding which can be detailed, the associated release of faecal material is less easily defined. Little information exists in the literature as to the gut evacuation rates of farmed Sea bass and bream, despite observational evidence suggesting that the majority of faeces are likely to be released during and after feeding. Thus, capability was built into the model so that the user can specify the timing of feeding and defecation events in hourly periods. In addition, the magnitude of the feeding events can be specified as a percentage of the daily feed ration. Similarly, the size of the defecation event is specified as a percentage of the total daily defecation. The user can specify up to five feeding events in a 24 hour period which was sufficient to represent most scenarios. Where an automatic feeding system is in place and many feeding events are in place over the daily cycle, this is sufficiently represented with one continuous feeding event through the working day.

Specifying a diurnal pattern of waste release is likely to be important during short term flux studies (i.e. 24 hrs) and where there is a strong diurnal pattern of wind and hydrodynamic conditions. During the longer term validation study at MD8 (13 days), a diurnal pattern of waste release was found to make little difference to model performance. Some sensitivity of these variables was tested to allow the importance of this capability to be put into perspective.

MODEL INPUT DATA IMPROVEMENTS

A new method was developed for collection and measurement of Sea bass and bream faecal settling rates, identified as being crucial to the accuracy of model predictions. The method included developments on methods of faecal collection, improvement of total particle numbers measured and measurement of particle volume to obtain a settling velocity–volume distribution.

Summary - Settling rates of faecal material from three size categories of cultured gilthead sea bream, *Sparus aurata,* and sea bass, *Dicentrarchus labrax,* were determined. Volume of selected particles was also determined and used to construct a detailed volume distribution across a range of settling velocities for each species. The sensitivity of aquaculture waste dispersal models to different types of data input was also tested. Results indicate that for sea bream, mean settling velocity for all measured particles (all fish sizes) was 0.48 cm s^{-1} with a range of 0.05 – 3.94 cm s^{-1} (*n*=1021). A mean settling velocity of 0.70 cm s^{-1} with a range of 0.10 – 6.27 cm s^{-1} (*n*=1042) was recorded for sea bass. The faecal material largely consisted of very small particles where the mean particles size was 0.71 mm (*n*=151) and 1.12 mm (*n*=150) for *S. aurata* and *D. Labrax* respectively. Percentage volume associated with the observed range of settling velocities indicated that for each of the *S. aurata* size categories and all sizes pooled, more than 50% of the total volume had a settling velocity of less than 2.0 cm s^{-1}. In contrast, for two of the *D. labrax* size categories more than 75 % of the total volume had settling velocities greater than 2.0 cm s^{-1}. Application of the resultant data to the DEPOMOD model firstly as a mean value and then using the volume distribution data, indicated that the model was highly sensitive to the two data types. The predicted seabed flux of organic material (g m^{-2} yr^{-1}) for *D. labrax* using the mean settling velocity at 0 m and 50 m from the cage was 3196 g m^{-2} yr^{-1} and 248 g m^{-2} yr^{-1} respectively. However, using the volume distribution data the predicted flux at 0 m was over three times greater but five times less at 50 m. For *S. aurata*, the predicted flux using the mean settling velocity was 2453 g m^{-2} yr^{-1} and 377 g m^{-2} yr^{-1} at 0 and 50 m respectively. Using the volume distribution data the values were over twice as high at 0 m and but were halved at 50 m. These results indicate that use of a single mean settling velocity value in models of this type does not accurately predict the extent of benthic deposition.

MODEL VALIDATION - SUMMARY

Validation of MERAMOD can be divided into three stages:

(i) validation of the particle tracking model using solids flux (AFDW – ash free dry weight) in 57 sediment traps at MD8 was undertaken. This study was 13 days in length and sampled across a range of flux values (65 – 7535 g AFDW m^{-2} yr^{-1}) from under cage to intermediate field (50 m).

(ii) validation of the particle tracking model using solids deposition (TDS – total dry solids) in a series of six 24 hour sediment trap studies from the spring and autumn 2002 cruises (MD1 (3 experiments), MD5 (2 experiments), MD3 (1 experiment)). These studies concentrated on the high flux zone underneath the cages and included different depths of sediment traps within the same experiment, including traps directly attached to the cage (net) bottom.

(iii) validation of the benthic response model using benthic community data from six sites. This established relationships between modelled flux and numerous descriptors (species (S), abundance (A), biomass, A/S, Shannon Weiner, Simpson, Eh (4 cm)) allowing the model to be used for planning and monitoring scenarios. Useful relationships were also found between modelled flux and relative abundance of indicator species and families

Model capability – The model validation summarised above and detailed further on, resulted in acceptable agreement between observed and modelled variables. This resulted in MERAMOD being satisfactorily validated for predicting flux and benthic response for Eastern Mediterranean mariculture operations. Any reasonable predictive capability in an environment where both sediment trap data and benthic community descriptors vary over such short spatial and temporal scales is acceptable. In addition, a number of tests undertaken in the validation studies showed model performance to increase when using the wild fish module, species–specific faecal settling rates and highly detailed husbandry data. This gives further evidence that identification and subsequent research on these issues by the MERAMED project was justified and has resulted in a progressive model. Importantly, the quality of input data used in the model directly effects its capability and this is particularly true of hydrographic and husbandry data.

Model limitations – The model should be used to predict flux and benthic response with special regard to the model accuracy specified, determined during model validation exercises. The level of accuracy expected also varies on the level of flux predicted. In addition, use of the benthic response module requires care as the reliability differs between the relationships established for each descriptor. Although primarily a data input issue, the detail of husbandry data used in the model effects predictions significantly. Use of monthly summarised husbandry data can be limiting due to the range of fish size and species being farmed within a cage group. This model has not been tested in hard substrate, underwater cliff areas nor does it include a validated resuspension component. It does not include flocculation or dissaggregation behaviour of particles.

MODEL USE

The user should take care to use appropriate settling velocities for the species being modelled. There are important differences between salmon, sea bass and bream faecal settling rates and so the most up to date information in the literature should be sought. The user should also seek detailed husbandry data for the site being modelled. Accurate hydrographic data are also required for this environment (additional detail is given in the modelling guidelines section below).

The model is set up with depth, cage layouts and sampling station locations (GridgenAM.exe). Hydrography, settling characteristics, feed input and wild fish module settings are then input to the model (PartAM.exe). A flux/deposition model (*ResusAM.exe*) then summarises the flux at the sea bed. Degradation of particulate material can also be undertaken with the G-model (Westrich and Berner, 1984). This model does not predict resuspension effects. A contouring package is NOT provided with this model. Example files are given for a hypothetical site (EXAMPLE sub-directories) and an electronic user manual is provided with the model.

MODEL COPYRIGHT AND LICENSING

This model (MERAMOD) can only be used under licence and no unauthorised copying or distribution is allowed. Please contact Chris Cromey.

MODEL INSTALLATION AND SYSTEM REQUIREMENTS

An installation program is provided to deploy this model to a computer. The model should be deployed to the root C directory of the target computer so that the directory system exists as C:\MERAMODv1 etc. This ensures example model runs provided run correctly.

The model runs on a Windows™ 98 platform or later, but it does not run on Windows™ NT. Read/write access is required to the main parent directory (MERAMODv1) and all sub–directories, so administrator rights for Windows™ XP should be set appropriately. Hard disk requirements are small (<50 Mb) as are memory requirements (64 Mb). The model will run on computers below this specification but increased time for simulations should be expected. Any computer less than 3 years old from January 2004 is likely to have the requirements to run this model.

MODEL VALIDATION – DETAIL

Model validation (particle tracking model)

The aim of these validation studies was to measure flux of waste material from an operational fish farm (g m^{-2} yr^{-1}) and compare with modelled predictions. These types of experiments are usually undertaken over short periods (24 hours) in high flux zones (i.e. directly underneath cage groups), but in the validation described here, a longer study incorporating sampling of low flux zones was also undertaken.

All model runs were undertaken with the highest level of detail available for each cage. This included specifying species and feed inputs for individual cages so species–specific faecal settling velocities could be used. Wild fish module settings were varied according to observations and in the case of the MD8 study, it was used to test the effect on model performance.

Current meters and meteorological stations were deployed during each study. In the MD8 study, hourly averaged current data over 13 days were used in the model measured at the surface, mid–water and near bed. In the 24 hour cruise experiments, five minute hydrographical data also collected at these three locations in the water column were used in the model. Detailed information was also obtained on the degree of cage fouling and feeding technique, including for some sites a description of where food pellets entered the cage at the surface. In some of the experiments, this level of detail was included in the modelling. Predictions of deposition (g m^{-2}) were obtained over the modelled period and then standardised by scaling up to obtain flux (g m^{-2} yr^{-1}).

All sediment traps were correctly designed with a length/diameter ratio of at least 5:1 and in most cases, diver deployed. In the 24 hour experiments, traps were deployed either on the sea bed, directly attached to the net or suspended in the water column. Using a variety of methods was useful as not only did this allow development of new methods, but it also allowed testing of model predictions of particle flux at different water column depths. After traps were retrieved, various protocols detailed in WP3 were followed to obtain a mass of AFDW or TDS (dry weight). This could then be scaled accordingly to obtain an observed flux in units of g m^{-2} yr^{-1}, directly comparable with model predictions.

Summary of MD8 sediment trap experiment – To validate a deposition model for marine fish farms in the Eastern Mediterranean, flux predictions of ash free dry weight (AFDW) of waste material arising from the farm were compared with observations of sediment trap data (g m^{-2} yr^{-1}). Model input data were more detailed than usually used in such models, with cage specific data used for food and faecal settling velocities according to feed type and species respectively, as well as feed input. Detailed hydrodynamic data obtained at three depths were also used in the modelling as well as the effect of wild fish feeding on the fate of discharged farm waste.
Diver deployed sediment traps on 8 transects at distances 5, 10, 15, 25, 35 and 50 m from the experimental cage were deployed for a period of 13 days and then analysed for

AFDW. Comparisons between observed and predicted AFDW resulted in a satisfactory regression line when appropriate adjustments were made to observations to account for natural background sedimentation (observed deposition = 1.04 predicted deposition + 82 g m^{-2} yr^{-1}, R^2 = 0.61, n = 57). Accuracy of predictions of AFDW were dependent on the level of deposition with the best accuracy achieved in the mid–range of deposition 501–2500 g m^{-2} yr^{-1} (± 29 %) and reduced model performance at low (0–500 g m^{-2} yr^{-1}) and high (2500+) depositional zones (±111 % and ±35 % respectively). The model performance represents a significant improvement on current models validated for this type of environment and species. The use of Titanium as a marker applied to feed is also discussed.

The model was validated across a range of observed deposition values (65 – 7535 g AFDW m^{-2} yr^{-1}) which is uncommon for models of this type. The study also showed that model performance was improved when species–specific faecal settling data were used. Figure 2 shows the improvement in model performance when including the effect of wild fish, consistent with observations of fish behaviour at the site. These issues resulting in improved model performance are important as this further justifies prioritisation of these research areas made by the MERAMED project.

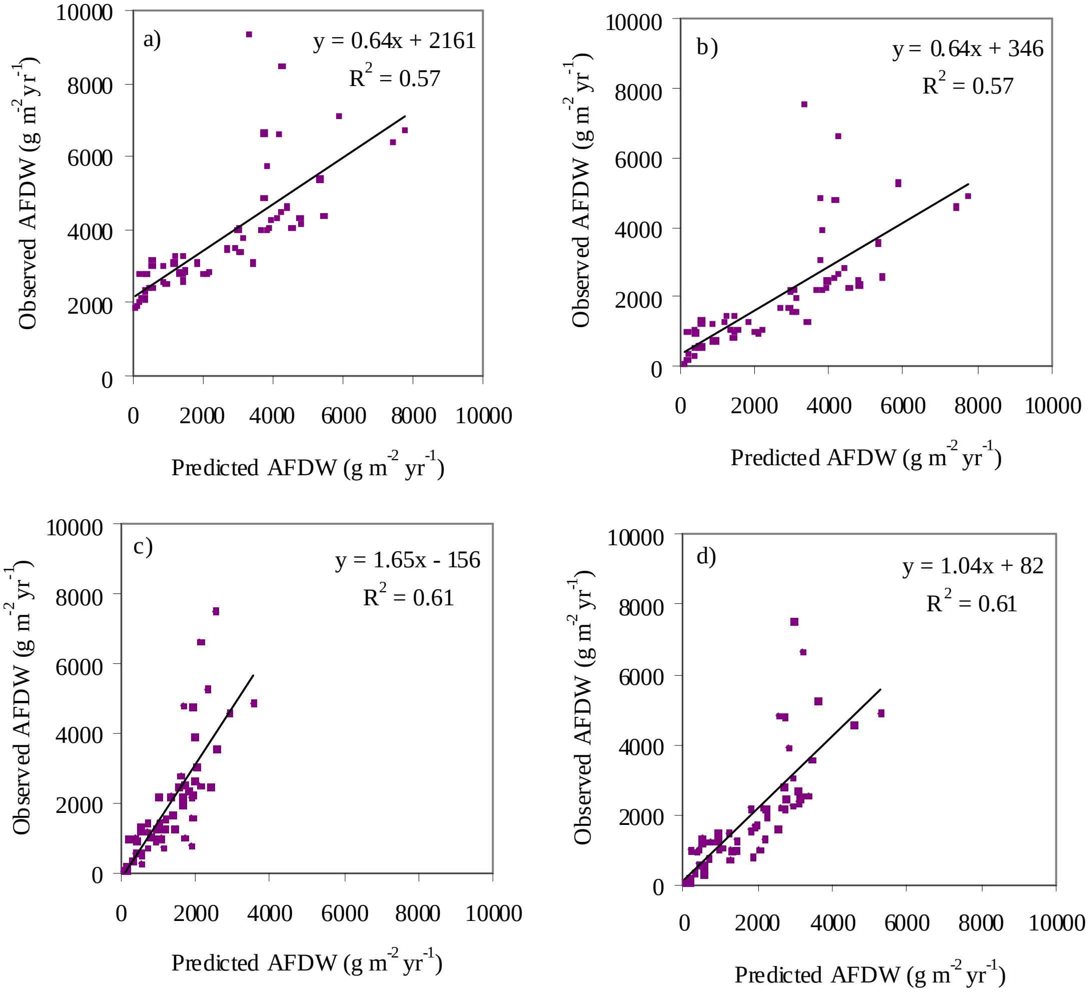


Figure 2. Observed and modelled AFDW (n = 57) showing the improvement in model performance by scenario a) subtracting background sedimentation from observations and variation of the percentage removal of waste feed pellets arising from the farm being eaten by wild fish b) 0 %, c) 100 % and d) 50 % (optimised).

Model validation (particle tracking model) – Summary of cruise sediment trap experiments (MD1, MD5, MD3)

In all of the modelling studies presented in this workpackage, quality control checks are always undertaken by comparing the total amount of predicted solids in the grid area with a calculation undertaken in a spreadsheet. However, as upper traps were used in this series of experiments, basic calculations in a spreadsheet allowed some assessment of the quality of the observed data. In particular, the mechanism for sediment traps gaining mass from natural material (e.g. fall out from nets) is likely to be more dominant than the mechanism for loss (e.g. consumption, resuspension). For this reason, basic checks should allow assessment of the magnitude of observed data. For example, for an observed flux of 30000 g solids m^{-2} yr^{-1}, a basic calculation verifies that for a trap attached to the net bottom, these rates are possible [feed digestibility 85%, food wastage 5%, feed water content 9%, cage bottom area 15m*15m and mean feed input of 95 kg $cage^{-1}$ d^{-1} (=95*1000* ([0.15*0.95*0.91]+[0.05*0.91])/15 /15*365.25 = 27015 g m^{-2} yr^{-1})]. Using this check for each experiment, experiment 4 (MD3, March 2002) was rejected as the observed values of solids in the upper traps could not possibly be obtained from discharge by the farm alone. This was also verified by observations of non–fish farm material made at the time of filtering. As upper traps were often deployed across the width of the net bottom, these basic calculations also revealed whether the waste material was exiting the net evenly across the whole area of the net bottom, or distributed in some other way. This important detail was incorporated into some modelling scenarios.

Given the detail available and the spacing of the individual traps (0.5 to 3 m), this series of experiments was also an opportunity to test whether the model can predict flux to this spatial resolution in mostly high deposition zones underneath cages. Figure 3 shows the modelled and observed averaged data for all experiments in this series. For experiment 1 in this figure, code *1–Upper* is the average of all 6 upper traps, *1–Lower* is the average of all 6 lower traps and *1–All* is the average of all the data in this experiment (i.e. 12 traps). Commonly, observed (and modelled) flux was higher in upper traps compared to lower traps, but this is not always the case. In such a situation, this is usually explained by examining feed input data for the cages. Where feed input in cages adjacent to the experimental cage is much higher, this can result in lower traps being subject to higher flux as material from adjacent cages is the main influence. The upper traps receive less material as these are attached to the net bottom and so are primarily influenced by the experimental cage with a lower feed input. This effect was observed for experiment 1 and husbandry data did show adjacent cages to have a much higher feed input. However, the model did not reproduce this effect.

The model could not predict the excessive variability between traps for this series of experiments, despite using detailed information and using an unusually fine grid resolution of 1 m and particle time step. This is not surprising. The main reason is likely to be that the true exit positions of particles from the cage domain are just not known, fouling and fish activity all having an effect over this small spatial scale. In addition, the current regime between cage bottom and bed may differ slightly to the mooring location. Examination of the standard deviation bars in Figure 3 demonstrates the variation in observations between traps. A similar order of variation was predicted by the model for most experiments.

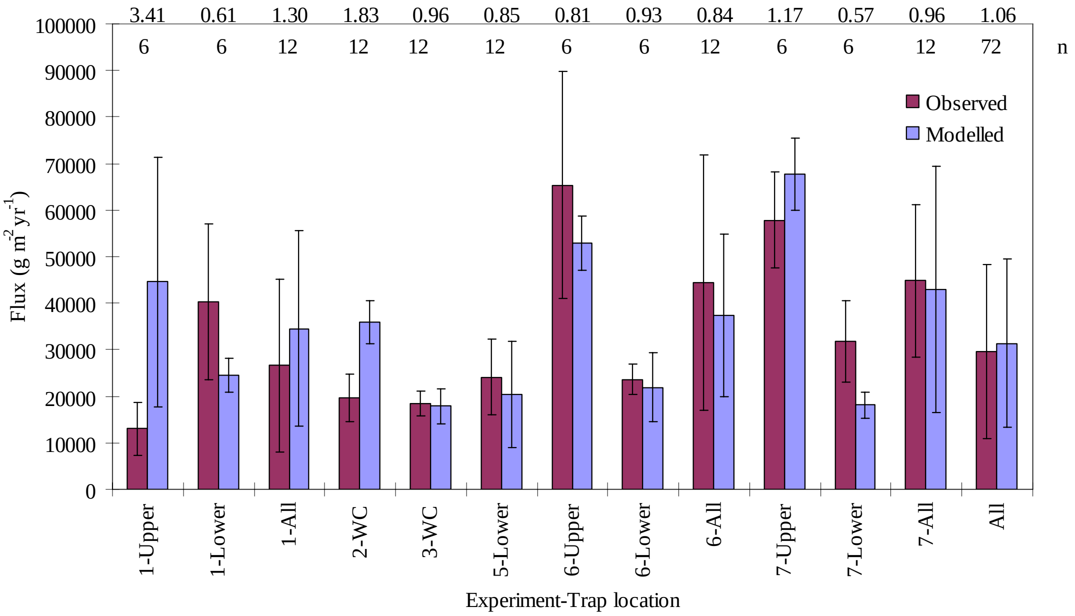


Figure 3. MERAMED cruise sediment trap experiments showing mean observed and predicted flux (total *n* = 72 traps, *n* = 12 traps per experiment). Upper traps are deployed attached to nets, lower traps are on sea bed, WC are traps suspended in the water column. Experiment 1 (2MD1), 2 (2MD1), 3 (2MD5), 5 (3MD1), 6 (3MD5), 7 (3MD3). Standard deviation (error bars), the factor by which the model is different to the observed and the value of *n* for each average is also shown.

Table 1. Summary of model accuracy and the range of observed flux values the model was tested over for all sediment trap experiments. The MD8 experiment was 13 days in length and the cruise experiments were 24 hours in length.

Experiment	Range of observations	Model accuracy	n
MD8 (low flux zone)	65 – 500 g AFDW m^{-2} yr^{-1}	±111 %	9
MD8 (mid flux zone)	501 – 2500 g AFDW m^{-2} yr^{-1}	± 29 %	27
MD8 (high flux zone)	2501 – 7535 g AFDW m^{-2} yr^{-1}	± 35 %	21
MD8 (all data)	65 – 7535 g AFDW m^{-2} yr^{-1}	± 44 %	57
Cruise (upper)	10045 – 111943 g TDS m^{-2} yr^{-1}	± 39 %	18
Cruise (lower)	10314 – 43416 g TDS m^{-2} yr^{-1}	± 66 %	18

Cruise (water column)	12270 – 31133 g TDS m^{-2} yr^{-1}	± 36 %	36
Cruise data (all data)	10045 – 111943 g TDS m^{-2} yr^{-1}	± 49 %	72

Conclusion for all sediment trap experiments

These experiments validate the particle tracking model satisfactorily. In particular, validation across the lower deposition zones at distances up to 50 m from the farm was useful as well as experiments providing total solids and AFDW observations for comparison. Given existing models in the literature, this is likely to a globally important model validation exercise for this type of model.

Model validation (benthic response module)

The aim of this exercise was to establish semi–empirical relationships between modelled flux (g solids m^{-2} yr^{-1}) and benthic community descriptors. Any general relationships which can be established are useful as these allow prediction of benthic effects for different planning and monitoring scenarios.

The majority of mariculture waste deposition models predict the flux of waste material from a discharge, but very few models relate these flux rates to a benthic effect. In this type of modelling, developments which link the physics (flux) to a benthic effect (biological) and/or include a biochemical component are considered progressive. As measurement of benthic community structure is a well established method in assessing the effect of mariculture discharges, it is desirable to have modelling capability to predict changes in this community structure.

Model grids containing bathymetry, cage layouts and sampling stations were set up for each site. The majority of these data were obtained from bathymetric surveys at each of the sites undertaken in the scanning cruise (July 2001).

Of the seven scanning cruise sites, hydrographic data were measured for four sites around the sampling period, two sites had data available for other periods (one site same season, second site different season) and the remaining site had no data available (MD7). Electromagnetic Interocean S4 current meters (Interocean, California) were used at three of these sites and rotary Aanderaa RCM7 current meters (Aanderaa, Bergen, Norway) were used at the other three sites. At five of the sites, surface and near–bed measurements were taken sampled every 10 minutes, whereas mid–water measurements were taken every 20 minutes at the remaining site which was shallow (MD6). Hydrographic data lengths varied in length between 29 and 43 days and subsequently record length for each site determined the length of the modelled period. Horizontal dispersion coefficients were measured for all but one of the seven sites (MD1) during the scanning cruise using six DGPS drifting buoys (accuracy: 57% data ± 1 m, 99% data ± 4 m; fix interval 30 s, sock depth 5 m).

Obtaining representative feed input data for the modelling period and cages near the sampling area has been consistently problematic throughout this project. The reluctance or apathy by fish farmers in sharing detailed husbandry data is a common problem for studies undertaken in all types of environment, not just the Mediterranean areas in this project. As farming practices experienced in this project include different species and sized fish in the same cage group, this forces the requirement for detailed cage by cage husbandry data. Adjacent cages may have feed input different by an order of magnitude as well as different species with differing faecal settling velocities. Monthly summarised data are therefore a limitation and need to be treated with caution. For the majority of sites, an average of the monthly averaged feed input data for the three months prior to benthic sampling was used. These monthly averaged data were typically for the whole farm, but occasionally cage group data were obtained. Using farm summaries can be inaccurate, especially if benthic sampling is concentrated under cages with large fish. These groups containing large fish will have a much higher feed input than inner groups with smaller fish and this detail is not typically contained in monthly summaries. Using a farm summary will result in using feed input lower than in reality, so consequently flux and benthic effects will be under–predicted in the model. To account for these potential inaccuracies at some sites, detailed husbandry data obtained for sediment trap studies in the detailed cruises (March 2002, October 2002) were used to scale farm summaries so that cage groups above benthic sampling transects were represented accordingly.

Sea bass settling velocity–volume particle distributions measured by Magill et al. (In review) (15% of mass < 2 cm s^{-1}) and Sea bream (69% < 2 cm s^{-1}) were used in the model. Where species in individual cages were unknown, a combined distribution of both species was used (22 % < 2 cm s^{-1}). Settling velocity of feed pellets dependent on pellet diameter was specified using data measured by this project.

Observational evidence from workpackage 3 showed the effect wild fish have on the feeding and subsequent removal of uneaten feed pellets arising from the farm. This important loss of waste material has been ignored or not properly represented in existing models. The MERAMOD wild fish module was used in validation of the benthic module. 75% of waste uneaten feed pellets arising from the farm were removed by wild fish before reaching the sea bed. For the flux predictions used in validation of this benthic module, it was noted that the general trends were not overly sensitive to the value of 75 % used. This was due to the low mass of waste uneaten feed compared to faecal mass used in the model, consistent with modern husbandry practices which aim to minimise feed wastage.

Feed digestibility (85%), water content (9%), uneaten feed (5%), vertical dispersion coefficients (0.001 $m^2 s^{-1}$) and particle trajectory evaluation accuracy (60 s) were default for all sites. Waste faecal and feed material was released continuously throughout a 24 hour period and specific times of feeding and defecation events were not modelled. For the length of simulations undertaken here, use of detailed timings of waste release makes no difference to model predictions (see particle tracking validation study sections and MERAMOD management scenario newsletter). In addition, random starting positions of particles were assigned inside the cage volume. In all simulations, total particle numbers were optimised (> $5*10^5$) and increasing particle numbers beyond this value did not result in any change in predictions.

The model was set up and run for the period around benthic sampling in the scanning cruise (July 2001) to obtain predictions of flux at each of the sampling stations. For each of these predictions, observed benthic community descriptors were available (workpackage 2). A series of graphs were then plotted of observed benthic descriptors against modelled flux (g m^{-2} yr^{-1}) to determine any semi–empirical relationships that may exist. It is common to use a logarithmic scale on the x–axis for modelled flux due to the wide range of values expected. Modelled flux + 1 was used so that reference stations or stations where a flux of zero was predicted could be included on the logarithmic scale. On the y–axis, a logarithmic scale was used for abundance and biomass descriptors, by convention.

Relationships between modelled flux and the following benthic descriptors are worthy of presentation here: species, abundance, biomass, abundance/species (A/S), Shannon Weiner (log 2), Simpson (1-lampda) and Eh (4 cm). No useful relationships were formed with the following descriptors: evenness (J), % carbon, % nitrogen, chlorophyll a, and phaeopigments. As expected Margalef (species richness–d) showed a similar relationship as species and so is not presented.

It is difficult to establish general relationships where such a range of variability exists in descriptors. For example, species at the reference stations for all of the sites varies between 55 and 142. Of the six sites tested, two separate groups were evident relating to the change in abundance along the flux gradient. Four of the sites had peaks in abundance at stations receiving high flux near to the cage areas (named group 1). In addition, these sites also had between 96 and 142 species at the reference stations. The two other sites did not show abundance peaks at stations along the gradient (named group 2). One of the sites in group 2 also had the lowest number of reference station species (55). With these two groups separated, general relationships were established for the data in group 1.

Basic trend lines were fitted to the data in group 1 using Excel for Windows™ therefore potentially, albeit cautiously forming the basis for a modelled relationship. Where trend lines were fitted, a value of R^2 is shown. No generalised relationships could be established for group 2 sites. However, there is value in presenting these data as it gives an example of how a site with low species numbers for the reference station might respond to a flux gradient.

Establishing a module which contains some relationships between modelled flux and the abundance, relative abundance and presence/absence of indicator species or families is desirable. As well as testing the difference between use of species or families as indicators, such relationships could be used to determine at what level of modelled flux indicators species or families might be expected. From WP2, indicator species of polluted areas used were: *Apseudes robustus, Capitella capitata, Caulleriella oculata* and *Maldane sarsi* .Indicator species of unpolluted areas used were: *Cirrophorus branchiatus,* *Cossura coasta, Levinsenia gracilis, Magelona alleni* and *Monticellina heterochaeta.* Indicator families of polluted areas were: *Ampelscidae, Lucinidae, Maldanidae, Nemertina* and *Oedicerotidae.* Indicator families of unpolluted areas were: A*mpharetidae, Onuphidae, Opheliidae* and *Semelidae*.

Model validation (benthic response module) – Main highlights and features of each site modelled

There is a gap in model predictions between 1272 and 3140 g m^{-2} yr^{-1} apparent on all graphs (Figures 4, 5, 6). This is caused by predictions of 10 m stations being above 3140 g m^{-2} yr^{-1} and 25 m stations being below 1272 g m^{-2} yr^{-1}. This is an interesting result as this suggests that consistently sampling at 10 and 25 m stations at all sites resulted in absence of sampling a benthic community exposed to flux between 1272 and 3140 g m^{-2} yr^{-1}. This similarity occurred across all sites, despite all sites being modelled with differing environmental and husbandry characteristics.

The relationships in this section use basic, well established indices and do not rely on indices which have a debatable method of determination or underlying principles (e.g. Infaunal Trophic Index in DEPOMOD). For this reason, the MERAMOD benthic module presented here can be seen as a significant improvement on existing benthic response models.

All descriptors on the y–axis are observations, flux values on the x–axis are modelled (semi–empirical relationships).

Species – Species and modelled flux show a promising relationship. 1MD4 had a very low species number at the reference station and so shows a different response to the other sites.

Abundance – Four of the six sites show a peak in abundance along the transect (1MD1, 1MD2, 1MD3, 1MD6). Interestingly, three of the these sites (1MD1, 1MD2, 1MD6) have peaks at approximately the same modelled flux (4545 – 5501 g m^{-2} yr^{-1}); the fourth site (1MD3) shows a peak in abundance at a station subject to much lower flux (1272 g m^{-2} yr^{-1}). Possible explanations for 1MD3 being different may be revealed by the species lists. The main question here is whether the model is inaccurate for this site or whether other reasons such as the station being near the feed barge is an explanation.

Biomass – Peaks in biomass are shown by all sites, but at differing locations on the flux gradient. For 1MD2 and 1MD3, peaks in biomass also coincide with peaks in abundance, but these peaks occur at different locations on the modelled flux gradient (5284 and 1272 g m^{-2} yr^{-1} respectively). For sites 1MD1 and 1MD6, peaks in biomass and peaks in abundance do not occur at the same station. Three of these peaks in biomass (1MD1, 1MD3 and 1MD6) occur at relatively low modelled flux stations (< 1500 g m^{-2} yr^{-1}), the fourth (1MD2) occurring close to high flux areas (5284 g m^{-2} yr^{-1}).

Abundance/Species – This index is low at stations where low flux was predicted, increasing for most sites to a peak at stations in moderate flux areas. In very high flux areas, A/S reduces. Of the sites where a peak in abundance was observed, A/S peaks between 3996 and 5501 g m^{-2} yr^{-1} for three sites (1MD1, 1MD3, 1MD6) and at 7438 g m^{-2} yr^{-1} for the fourth site (1MD2).

Shannon Weiner – This index reduces with increasing modelled flux, as expected.

Simpson – This reduces with increasing modelled flux, as expected but is not as clear as the Shannon Weiner relationship. For example, values of Simpson for 1MD6 are exceedingly low at stations with high modelled flux.

Redox – This reduces with increasing modelled flux, but there is a high degree of data scatter in all predicted flux zones.

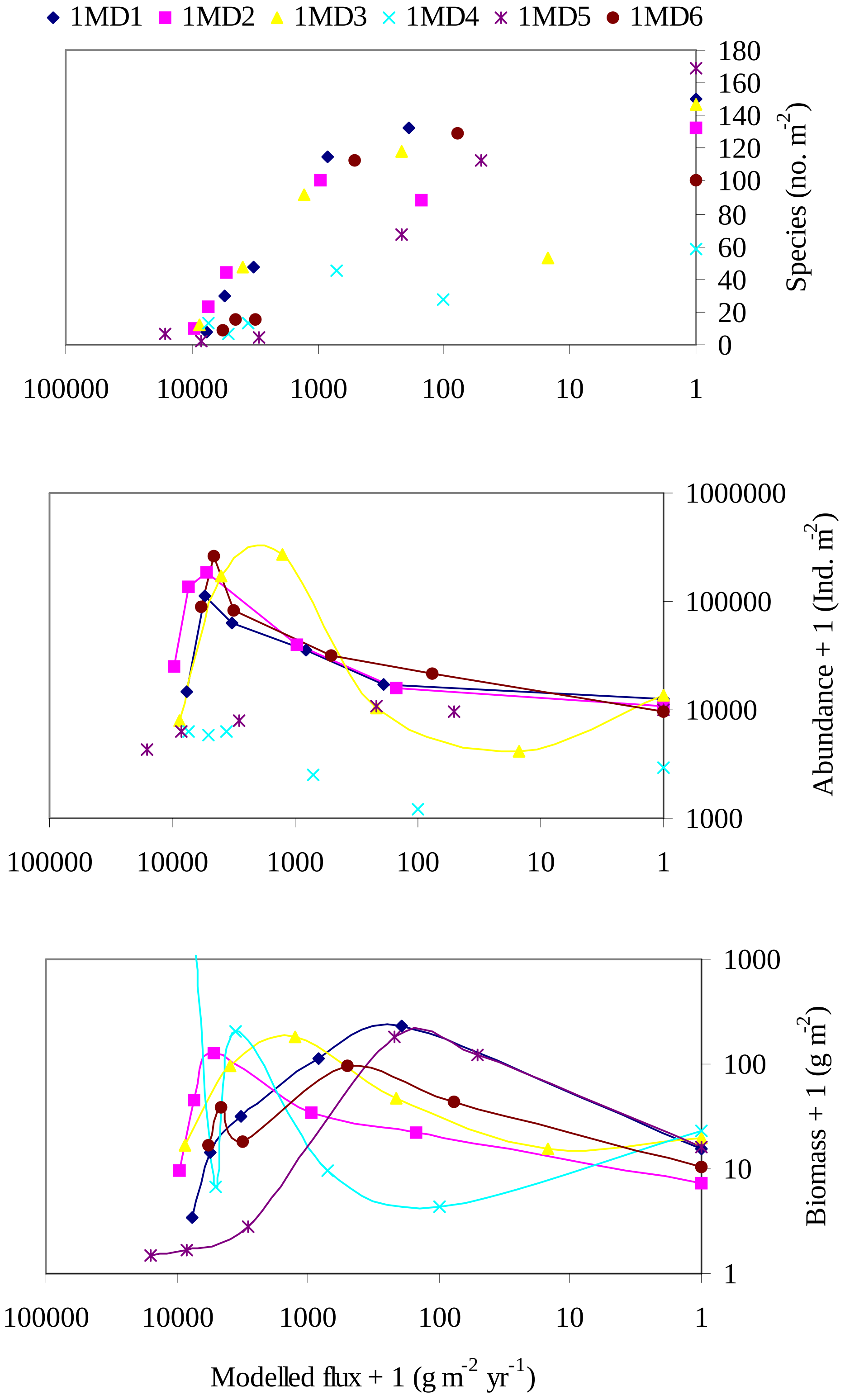


Figure 4. Species, abundance and biomass for each site against modelled flux +1 (g m^{-2} yr^{-1}). The scale on the biomass graph was reduced to eliminate a very high biomass value of 1103 g m^{-2} for 1MD4.

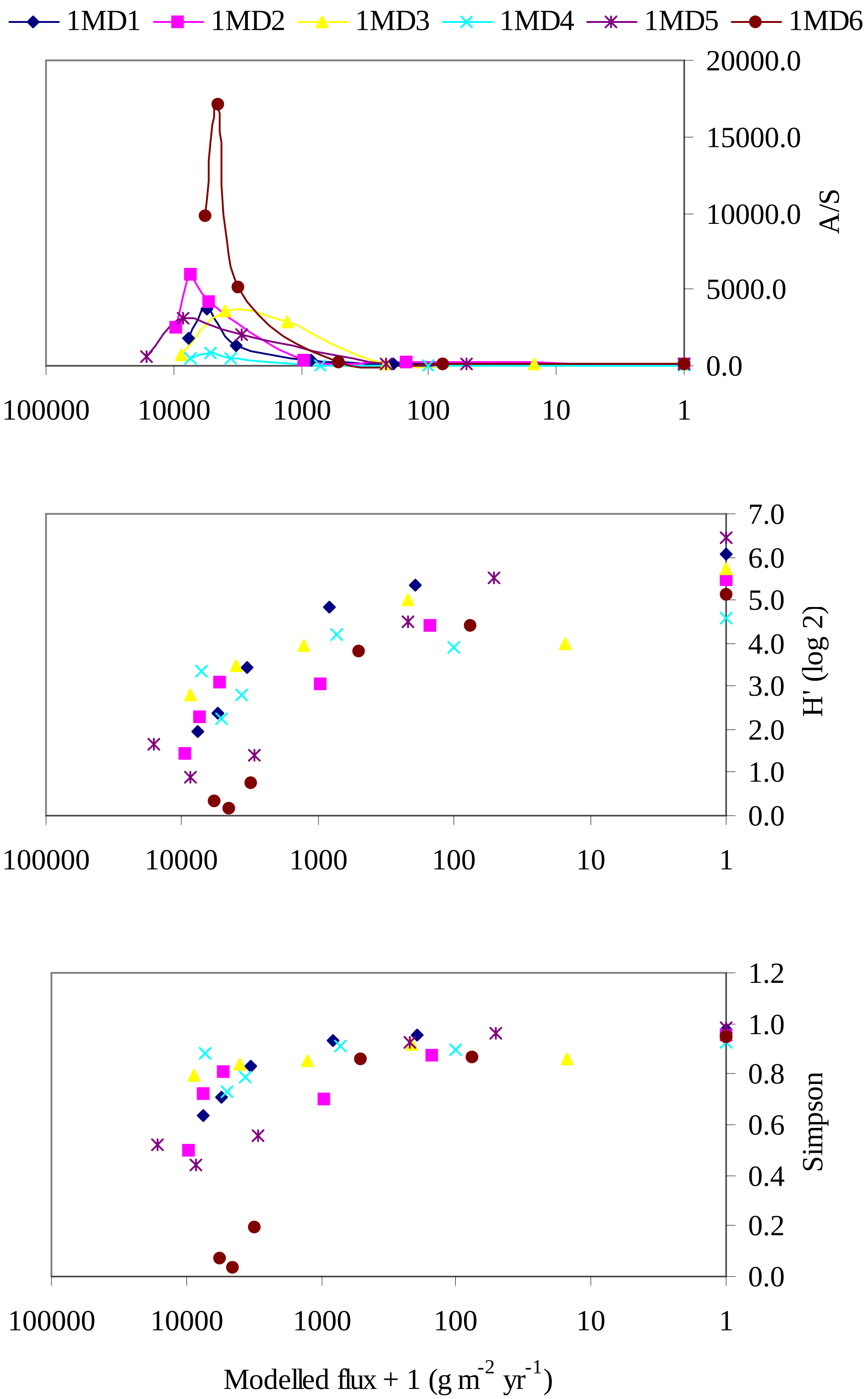


Figure 5. A/S, Shannon Weiner (H’) and Simpson for each site against modelled flux +1 (g m^{-2} yr^{-1}).

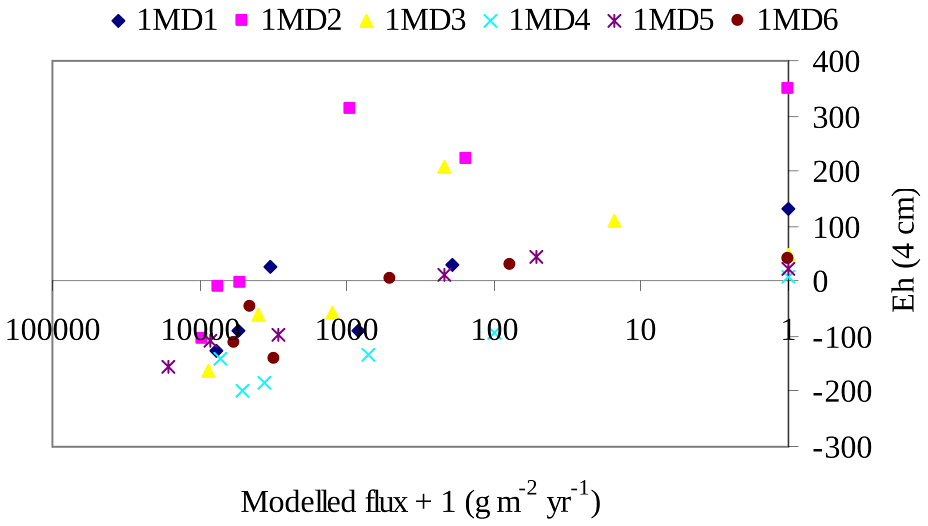


Figure 6. Redox for each site against modelled flux +1 (g m^{-2} yr^{-1}).

Model validation (benthic response module) – Main highlights and features of grouped data and generalised relationships

Without generalised relationships, the benthic module has limited predictive capabilities. Therefore, some generalisation is needed with specification as to what percentage of data are expected to fit within these generalised relationships. In the following series of graphs, generalised relationships are shown by trend lines, but more sophisticated lines of best fit can be generated by other means. Such lines of best fit form a basis for predicting a descriptor from modelled flux.

Group 1 sites exhibit a peak in abundance and show similar characteristics to such an extent that generalised relationships can be fitted to all the descriptors shown (1MD1, 1MD2, 1MD3, 1MD6). The remaining sites forming group 2 (1MD4, 1MD5) do not show a peak in abundance and generalised relationships are not fitted.

Species – This generalised relationship with species and modelled flux for four sites (1MD1, 1MD2, 1MD3 and 1MD6) (Figures 7, 8) is promising with a high R^2. For group 2 sites, the low species number for 1MD4 at the reference station demonstrates this site is different to others. In addition, the low species number at the reference station results in a shallow gradient of the fitted line between 55 and 13 species. Conversely, high species numbers at the reference station for 1MD5 result in a steep gradient of the fitted line between reference and high impact areas.

Abundance – This generalised relationship is also promising as it shows a peak in mid flux zones. However, although the trend line shown peaks at approximately 3000 g m^{-2} yr^{-1}, a more accurately fitted line would most likely peak at approximately 4500 g m^{-2} yr^{-1} more consistent with observations.

Biomass – The generalised relationship shows two peaks of biomass, one larger peak at approximately 1300 g m^{-2} yr^{-1} and the second smaller peak in higher flux areas (6000 g m^{-2} yr^{-1}). However, this should be treated with caution as it may just be an artefact of the trend line fitting algorithm and closer inspection shows very few stations actually on the line itself. However, a more complex fitting curve may also result in a similar shape curve with two peaks.

Abundance/Species – A peak is present at approximately 5500 g m^{-2} yr^{-1} in the generalised relationship coinciding with peaks in abundance observed in the moderate flux zone, as expected.

Shannon Weiner – A useful generalised relationship although appearing to fall between two distinct trends in the data.

Simpson – This index shows similar trends to Shannon Weiner with the generalised relationship falling between two distinct trends in the data.

Redox – A trend line can be fitted to these data, but a high degree of scatter is present either side of the line. Nevertheless, this is a useful relationship.

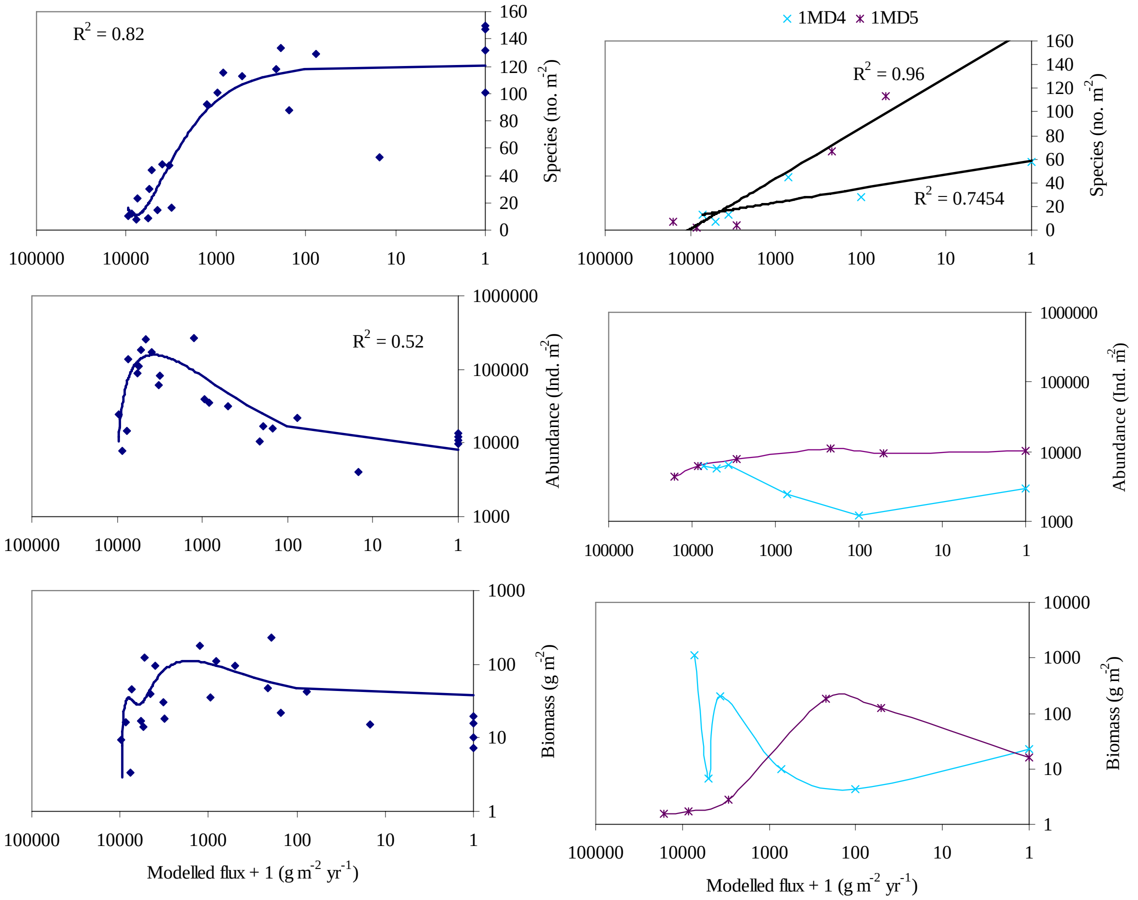


Figure 7. Group 1 generalised relationships (1MD1, 1MD2, 1MD3, 1MD6) and group 2 data (1MD4, 1MD5) for species, abundance and biomass.

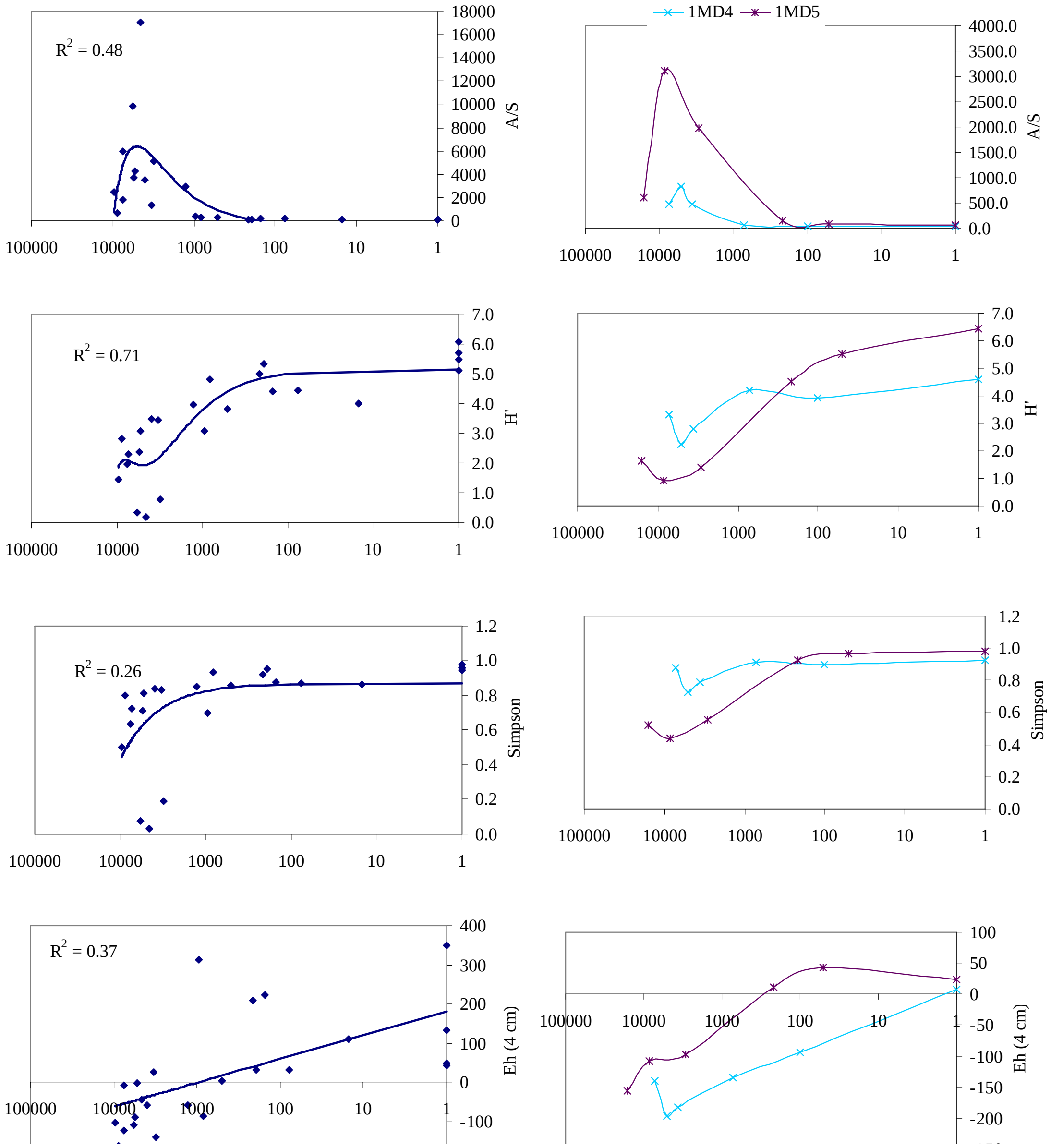


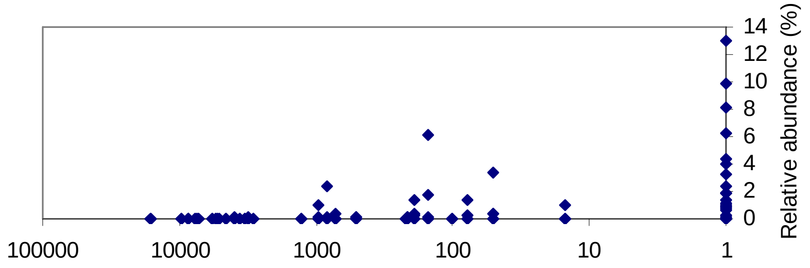
Figure 8. Group 1 generalised relationships (1MD1, 1MD2, 1MD3, 1MD6) and group 2 data (1MD4, 1MD5) for A/S, Shannon Weiner (H'), Simpson and Eh (4cm).

Model validation (benthic response module) – Indicator species and families

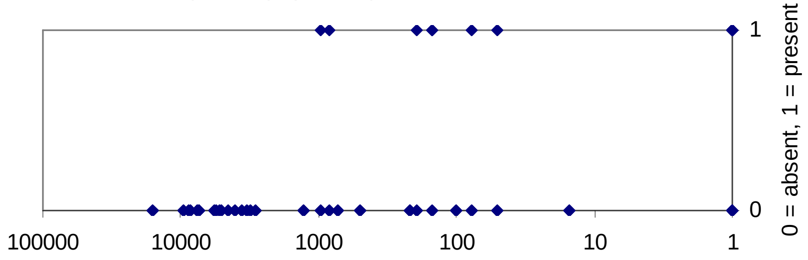
Species – Useful relationships were established between modelled flux and relative abundance and presence/absence of indicator species (Figure 9). For species indicative of unpolluted areas and using absent criterion of relative abundance less than 1%, all of the indicator species are expected to be absent at stations receiving greater than 1500 g m^{-2} yr^{-1}. 100 % of our data fit into this relationship (Envelope of Acceptable Precision (EAP) = 100 %). For species indicative of pollution and using absent criterion of relative abundance less than 5%, the majority of the indicator species are expected to be absent at stations receiving less than 1500 g m^{-2} yr^{-1}. 86 % of our data fit this relationship (EAP - 86%).

Families – The use of indicator families did not result in as many useful relationships compared to indicator species (Figure 10). The indicator families of unpolluted areas were useful in the model as similar trends were observed as for indicator species, with absence and presence of families above and below the 1500 g m^{-2} yr^{-1} level respectively. This was using criteria of absence being less than 1 % relative abundance (EAP = 100%). For the families indicative of pollution, a criterion of 1 % was also used (different to the indicator species for pollution of 5 %). For 75 % of the data, indicator families for pollution were absent at stations receiving less than 1500 g m^{-2} yr^{-1}.

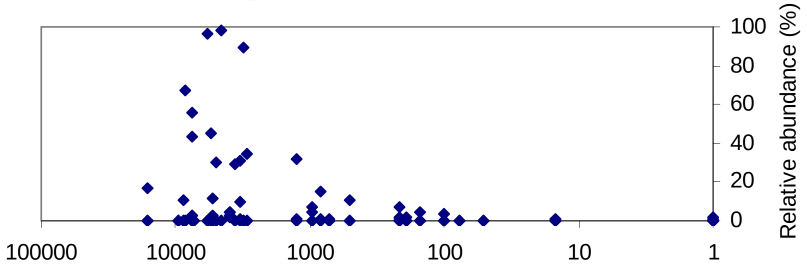
a. indicator species (unpolluted)



b. indicator species (unpolluted)



c. indicator species (polluted)



d. indicator species (polluted)

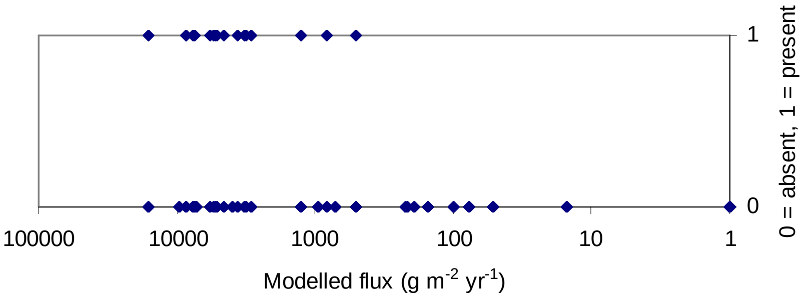
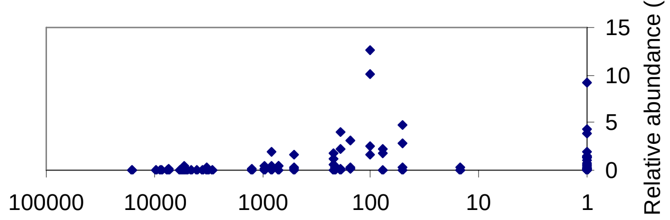
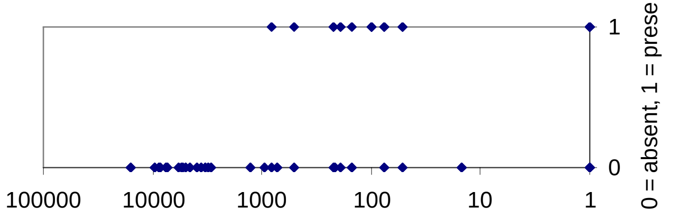


Figure 9. Model relationships between relative abundance and presence/absence of indicator species indicative of polluted and unpolluted areas.

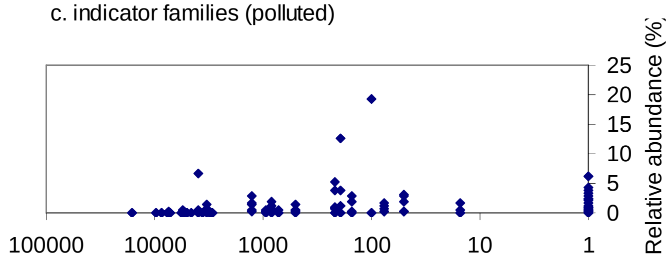
a. indicator families (unpolluted)



b. indicator families (unpolluted)



c. indicator families (polluted)



d. indicator families (polluted)

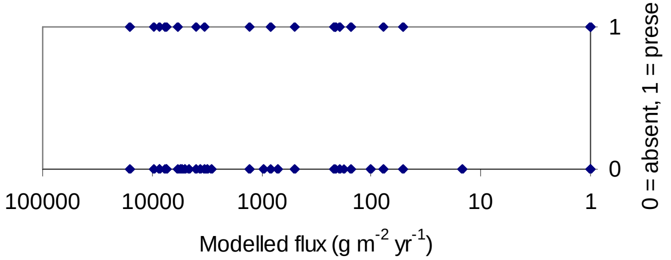


Figure 10. Model relationships between relative abundance and presence/absence of indicator families indicative of polluted and unpolluted areas.

Model validation (benthic response module) – Conclusions

The generalised relationships established between modelled flux and species, abundance, biomass, A/S, Shannon Weiner, Simpson and Redox are a useful advance in benthic effects models of mariculture impacts. In particular, the relationships established for species, abundance and biomass show expected trends for benthic response in low, moderate and high flux (impact) zones.

Relationships between indicators species and families with modelled flux were also established and are likely to be useful. A flux level of 1500 g m^{-2} yr^{-1} appeared to be a boundary, where above and below this there were definite trends in the pollutant tolerant and intolerant species. The use of indicator families indicative of unpolluted areas allowed a relationship to be established, but indictor families for polluted areas were less useful. The relative abundance relationships are a bonus for the benthic module.

Some of these relationships may be improved by using more complex fitting algorithms so that a better fitting relationship will result, incorporating more of the observed data. This forms the basis of a modelled relationship. It is also important to specify an Envelope of Acceptable Precision (EAP). This EAP specifies what percentage of stations is expected to fall within the envelope and is completely transparent in that no complex or potentially misleading statistics are used to describe model performance.

Generalised relationships in this type of modelling are always problematic, given the level of variability in the observed benthic data for replicates at the same station. Minimisation of inaccuracies in model predictions are always sought by using accurate input data but inevitably the level of accuracy will vary between sites. Such input data subject to inaccuracies are monthly summarised feed input data, already discussed. However, despite these occasional inaccuracies this benthic module provides an acceptable level of predictive capability. This is desirable in practical applications as not always the highest level of detail is available for input data.

Annex A. Guidance for using the model MERAMOD

The following data specification for MERAMOD includes details on data required to run the model and also includes some guidelines on data collection. Some information is also provided on which of these data may be standardised between sites and which data are commonly measured for every site modelled. In addition, some guidance is given on the level of detail required for certain data as this will depend on the objectives and level of complexity of the modelling study.

A.1 Summary of MERAMOD input data specification

The following data are desirable for the set up of MERAMOD:

1.1 current velocity data for an area close to the fish farm site (include information on heights of instruments above bed, total depth of water column at mooring, position of mooring, time (e.g. GMT) and direction formats (e.g. degrees true or magnetic)
1.2 some knowledge of the vertical structure of the water column; shear in the water column can be represented in MERAMOD by setting up layers in the model represented by different current velocity records
1.3 horizontal and vertical dispersion coefficients for the area$^+$
1.4 bathymetry of the area of interest either from a site survey or from an Admiralty chart of the area
1.5 number and dimensions (length, width and depth) of cages and the proposed/existing positions of these cages
1.6 feed input data (kg food d^{-1} for the whole farm) and mean fish size for the intended scenarios; information on the proportion of different fish species being farmed is also required as faecal settling rates vary between species
1.7 information on water content and digestibility of the food to be used at the stage of the growing cycle to be modelled; some assessment of feed wasted according to husbandry practice is required*
1.8 food and faecal settling velocity for the fish species being modelled (data from measurements undertaken in the MERAMED project and literature values are available for Sea Bass and Bream)*
1.9 background information on wild fish populations and their behaviour around the farm

*=MERAMOD defaults available
$^+$=MERAMOD recommended values available

The following data would be useful for assessment of site characteristics and interpretation of model results:

1.10 benthic macroinvertebrates present at the study site and the value of some benthic indices (e.g. Species, Total abundance, Biomass, Shannon Weiner, Evennes, etc) for sampling stations at the site

1.11 sediment type and characteristics for the proposed/existing site

General data requirements for modelling of total deposition from a single release of waste material (g m^{-2}):

As for data points 1.1-1.8 above, but with the exception of 1.6 which requires the following:

1.12 feed input data for the whole release period (kg food per unit time)

General data requirements for modelling of total deposition (g m^{-2}) or sediment concentration (g kg^{-1}) of a component adhered to the waste material:

As for data points 1.1-1.9 above and the additional information:

1.13 concentration of component on feed (g component kg^{-1} food)
1.14 excretion of component by fish (e.g. 90% excreted)
1.15 mass and total period of time for component in feed is being used (e.g. 100 kg $cage^{-1}$ over 7 days)

Model Output:

Predictions are given in an ASCII text data file showing predictions for the grid and for sampling stations. Predictions are also given in a x, y, z ASCII file which can be used in a contouring package for visual display. A contouring package is NOT included with MERAMOD.

A.2 Guidelines on model data input issues

A.2.1 Bathymetry

Bathymetry data of the study site is usually provided with an echo sounder interfaced with a GPS positioning system. Other sources of data may be used including bathymetric data from charts, discrete soundings using a hand held echo sounder, SCUBA diver measurements and tape or lead line soundings. In addition to these methods, data may be obtained from digitisation of charts or export of electronic data from vector charts. Accurate information directly around the cages and within the expected deposition footprint is required, with less detail required at distance from the farm. When using charts for information, attention must be paid to copyright issues.

These data are commonly fixed in latitude and longitude, but may be in relation to the cages. Conversion of latitude and longitude to UTM datum is recommended as this is a linear scale and the model uses a linear scale for grid generation. All of these data can be entered into a contouring package (e.g. Surfer for Windows^{TM}, Golden software), contoured and then exported as a grid with equal spacing between nodes. MERAMOD can import these grids via ASCII files in DSAA format.

Resolution of the data will depend on the survey method, but production of a grid with resolution not more than 25 m is suitable for most sites (known as the major grid in MERAMOD). Within the model, a minor grid can zoom in on the area of interest and this should have grid resolution of 5 or 10 m depending on the level of dispersion and sampling station positions. These factors as well as the topography also influence the total size of the grid required for the model, but this in the MERAMED project this was generally not more than 500 m by 500 m for the Mediterranean sites studied.

In the main MERAMED guidelines document, use of sea charts are categorised as visual/descriptive, discrete sounding measurements are defined as semi-quantitative and echo sounder systems profiled with a positioning system are defined as quantitative. Semi-quantitative or quantitative methods are the recommended minimum for obtaining model information for the model but this is site dependent. All methods, including descriptive methods were used in model validation in the MERAMED project.

A.2.2 Cage layout and positioning

It is recommended that cage layouts are obtained during an on-site survey, but the information may also be obtained from the farmer. Cage dimensions of length, width and depth of the net are essential data. Positioning of the cages in relation to the bathymetry data collected must also be known. Typically, positions of the four corners of each cage group are sufficient. Where largely spaced circular cages, the distance between cages should be measured/estimated. It is crucial to obtain the cage numbering system the farmer uses, as detailed husbandry data on a cage by cage basis supplied by the farmer will refer to this numbering system.

A.2.3 Sampling station positioning

Positional data can be obtained and converted to the same datum as the positional data for the cages and bathymetry, or a distance and direction for each station from the nearest cage may be preferable. Either method requires a reasonable degree of accuracy as the transect may be located along a high deposition gradient. Any error in station position in the model grid may result in a difference in predicted flux (g solids deposited m^{-2} bed yr^{-1}) of an order of magnitude.

A.2.4 Husbandry data

The collection of husbandry data is often effectively undertaken during the survey. Obtaining these data post-survey may require requesting the information from the company office, rather than obtaining them from the farmer. Obtaining data from the office may take a lengthy period of time, but does have benefits if data can be provided in an electronic format and contains a high level of detail. It is often the case that obtaining detailed data requires no more effort on the side of the fish farm company than monthly summaries. The level of detail of husbandry data requested depends on the level of complexity of the modelling study and guidelines are given in a later section. A confidentiality agreement may need to be in place before obtaining these data from the farmer.

Information on feed input (kg cage^{-1} d^{-1}) is required to run the model, but information on fish species, numbers, average weight, total biomass and feed diameter should be obtained where available. These data will be required either for the whole farm, individual cage groups or individual cages depending on the study. The time interval of these data will also depend on the study, but is normally on a month by month basis. For short term studies (e.g. 24 hours), it is essential to obtain these data for each cage.

Feeding and defecation events can be set up in the model, so information on the number of feeding events daily should be obtained as well as the feeding method (i.e. hand or automatic). Little information exists in the literature on the evacuation of faecal material by farmed fish in relation to feeding times, so the user may wish to create defecation events over the course of the whole day rather than specific times, particularly for longer term studies.

Additional husbandry observations which may be useful for sediment trap studies include the distribution of feed pellets on the surface whilst hand feeding. In addition, location of feed barrels of loading/unloading points can be noted if near the experimental cage. Recording this level of detail can often give useful clues if particular sediment traps deployed to net bottoms or sides contain elevated levels of waste material.

A.2.5 Feed pellet considerations (wastage as uneaten pellets, digestibility and water content)

The value of uneaten feed as a percentage of feed input is difficult to quantify and few studies exist in the scientific literature. Wastage depends on husbandry feeding method and the level of care taken to prevent overfeeding. Modelling studies currently use between 1 and 5 % of feed input lost as uneaten feed pellets. Feed digestibility and water content can usually be obtained from manufacturers specification sheets and default data are available in the model. Digestibility may well vary with feed pellet type, temperature and fish size. The three variables in this section cannot be varied over time within the period modelled nor between cages in the model. However, different model scenarios can be used to test the effect of varying these parameters.

A.2.6 Hydrodynamic data

Assessment of the quality of hydrographic data is essential and care should be taken in all aspects of data implementation as these sensitive data effect model predictions considerably. The model requires current speed (cm s^{-1}) and direction (° magnetic or ° true) or current vector components u (cm s^{-1} - positive towards east) and v (cm s^{-1} – positive towards north). In addition, the total depth at the location of the current meters and the depth and height of the current meters above the bed are required for input data.

Although the model does not use meteorological data, this is important for interpretation of hydrographic measurements and assessment of general flow patterns in the study area.

Up to five current data records can be implemented into the model to represent change in hydrodynamic conditions with depth. Where current meters are used that sample at a discrete depth, two or three instruments are usually deployed and these data implemented as layers in the model. With profiling instruments, the user should choose three to five bins which adequately describe the water column. Any sampling interval can be used in the model, but 10 minute or 20 minute (unaveraged) or hourly averaged is the most common. Length of current records is normally between 24 hours and 1 month or so. The required level of hydrographic data depends on study complexity and guidelines are given later.

Where limited numbers of instruments are available, priority should be given to obtaining surface and mid-water measurements. Where only one instrument is available, mid-water may be the best deployment depth, particularly if the site is deep. In the case of a deep water site, surface measurements only would tend to result in over-prediction of dispersion by the model as the higher surface flows would be used at all depths. A further advantage in deploying a mid-water current meter is that this is commonly at a similar depth to the bottom of the net. Measurement of the current at this depth is desirable as some of the sediment trap studies in the MERAMED project demonstrated that the exit point of particles at some sites was most likely to be the net bottom and not net sides. Measurements at mid-water depth are likely to be less influenced by the shadowing effect of cages.

In the main MERAMED guidelines document, measurement of hydrographic data is defined as a quantitative method. There is no visual/descriptive or semi-quantitative method suitable for hydrographic measurements that would result in data suitable for use in the model.

A.2.7 Settling rates – faeces

Faecal settling rates of the farmed species are required by the model. The MERAMED project measured settling rates of over 2000 Sea bass and Sea Bream faecal particles and these are available as default data. If another farmed species is being modelled, faecal settling data for this species should be used, if available. Combined data for both bass and bream are also provided by the model, so these may be used to approximate the faecal settling rates of another species. In addition, where species (i.e. bass and bream) are not specified on a cage by cage basis in the model, these combined data best represent both species. Faecal settling rates are not commonly available in the scientific literature for other species, with the exception of Salmo salar (Chen et al., 1999b, 2003; Cromey et al., 2002).

A.2.8 Settling rates – food

A relationship measured between feed pellet diameter and settling velocity is provided with the model so that an appropriate settling velocity can be set. The data for different pellet sizes and types (i.e. pelletised and extruded) is also available in addition to the general relationship. Where pellet diameter data are unknown, the average and standard deviation settling velocity of the whole data set should be used. Feed pellet diameter is constant through the modelled period. Information on feed pellets is more commonly available in the literature, particularly for salmonid feeds. The most likely reason is that feed pellets are easier to obtain for experimentation than faeces (Chen et al., 1999a; Holmer and Kristensen, 1994; Stewart and Grant, 2002).

A.2.9 Wild fish populations

The user can specify the percentage removal of uneaten feed pellets by wild fish in the water column and on the sea bed. In addition, removal of faecal material in the water column and sea bed can be modelled. Background information on wild fish populations at the study site in conjunction with the findings of the MERAMED wild fish workpackage will assist in setting appropriate values in this module. In addition, some of the experimental work detailed in the quality assurance field handbook can be used to measure directly the effect of wild fish populations at the site on the fate of uneaten feed pellets. However, it is recommended that if modelling is being undertaken with a number of sites wild fish module settings should be constant across all sites during initial comparisons (see later section on scenario complexity). As settings of this module directly effect predicted flux at the sea bed and benthic effect, adequate justification of settings used is required.

A.2.10 Dispersion coefficient data

Drifting buoy or dye studies to assess the dispersion characteristics of a water body are not common for areas around fish farms. These studies are more commonly associated with long seas outfalls of domestic sewage, industrial discharges or marine dumping grounds. The MERAMED project undertook a number of drifter surveys using six DGPS drifting buoys at Mediterranean fish farms (fix interval 30 s.; accuracy 57% ± 1 m, 99 % ± 4 m; sock depth 6 m). The main limitation of such studies is that only a snap shot of conditions are obtained during the survey period. In the absence of site specific data, examination of the range of values measured in the MERAMED project may assist in setting an appropriate value. In Scotland, regulatory models apply a standardised horizontal dispersion coefficient (k_x, k_y) of 0.1 $m^2 s^{-1}$ unless site specific data are provided (SEPA, 2003). k_x is resolved for the model x-axis (090° – 270° axis) and k_y for the model y axis (000° – 180° true axis). It is recommended a value of 0.001 $m^2 s^{-1}$ is used for the vertical dispersion coefficient (k_z) in MERAMOD.

A.3 Standardisation of data

MERAMOD model input data generally fall into one of three categories comprising of site specific survey data, site specific data obtained from the farmer and standardised (default) data.

A.3.1 Input data category 1 – Site specific data measured by survey

Hydrographic data (current speed and direction), bathymetry and cage and sampling station layout are necessary for modelling a site and should be given priority during survey planning. Occasionally dispersion coefficients are available from a specifically designed survey. For validation of the model predictions, benthic data and/or sediment trap data from the site can be used to test the model predictions if these are available. The wild fish population and the effects on the fate of wastes are also site specific.

A.3.2 Input data category 2 – Site specific data obtained from the farmer

Husbandry data normally fall into this category and are required for accurate modelling. Occasionally information on cage layout is obtained from the farmer if not obtained during site survey.

A.3.3 Input data category 3 – Standardisation of data (default data)

Percentage of feed input wasted as uneaten pellets, feed digestibility, feed water content, feed and faecal settling velocities are commonly standardised and assigned as default data across sites. This assists comparisons of different scenarios of the same site and between sites. Standardising these data means that differences in predicted flux and benthic effect will be primarily a result of the differences in the model input data of feed input, hydrography and bathymetry. Less commonly, dispersion data and feeding times are standardised.

A.4 Complexity of scenarios

It is good modelling practice to decide on objectives of a modelling study prior to the site survey so that the appropriate data can be collected during the survey. During the modelling exercise it is essential to begin with a simple scenario and increase the level of complexity in stages, with appropriate checks on model output. In addition to building confidence in the model, the effect on model predictions of increasing scenario complexity can be assessed. Increasing the complexity of the modelling scenarios can make little difference to model predictions, depending on the site characteristics and the detail being added. A simple robust model that performs reliably is more desirable than a model requiring extensive data input. Despite this, a simple model should still use good quality input data and where the reliability of these data are uncertain, sensitivity should be tested.

This section is contains recommendations for setting up the model for prediction of solids flux and benthic effects in a continuous release for an operational farm. The first level of complexity allows a quick assessment of the site, particularly important to check that measured hydrodynamic data are accurate (Table 2). The second level of complexity uses accurate data for bathymetry, feed input and species in individual cages (if available). Faecal settling velocities appropriate to the species being farmed are used and feed settling velocity according to pellet diameter. Where the species for each cage is unknown, a combined faecal settling velocity distribution should be used for both species (supplied with the model). The wild fish module should be used at this level with appropriate variables for the site, with justification. Site specific data for dispersion coefficients should also be used, where available. The third and final level of complexity includes modelling the timing of feeding and defecation events and modifying the release position of particles within the cage volume. It is essential appropriate justification is given if the latter variable is modified from default as this has a high level of sensitivity.

It is recommended that the minimum level of complexity for the majority of studies should be 2.

As these levels of complexity are built into the model the following checks are essential:

1. After the model has been run for the first level of complexity, the footprint shape should be checked to establish it exhibits similar features to a scatter plot of current velocity *u* and *v* components. Any features such as a strong residual current in a particular direction should be apparent in both, although the footprint may well be a combination of a number of current velocity data sets.
2. For every level of complexity and for the final scenarios, a spreadsheet is provided with the model which allows the user to check model output with a simple mass balance calculation. This enables the user to verify that the desired wastage rates are being undertaken by the model. This check should be undertaken as necessary throughout scenario development.
3. At complexity level 2 and above, comparisons can be made between the predicted flux and benthic effects and the observed situation for which the model is intended to predict. This may include direct comparisons with model outputs and benthic and sediment trap data, but it may also include use of other measurements to assist in interpretation of agreement between the observed and predicted situation.

Table 2. Model input data guidelines for the levels of modelling complexity for scenarios of flux and benthic effect for a planned or existing farm (continuous release).

	Complexity level		
Input data	1	2	3
Bathymetry	Flat bathymetry using mean depth under cages	Accurate	Accurate
Cage layout and positioning	Approximate	Accurate	Accurate
Sampling station positioning	Approximate	Accurate	Accurate
Husbandry data:			
- Feed input	Approximate	Accurate	Accurate
- Feed/defecation events	Disable	Disable	Accurate
Feed pellet	Use defaults	Use accurate data	Use accurate data

considerations (wastage as uneaten pellets, digestibility and water content)		appropriate to feed being used or standardise across sites	appropriate to feed being used or standardise across sites
Hydrodynamic data	Use site specific data	Use site specific data	Use site specific data
Settling rates – faeces	Use defaults	Accurate for species	Accurate for species
Settling rates – food	Use defaults	Accurate	Accurate
Wild fish populations	Disable (set variables to zero)	Accurate	Accurate
Dispersion coefficient data	Use defaults	Accurate (where available)	Accurate (where available)
Particle starting position in cage domain	Default (random)	Default or accurate (with justification)	Accurate (with justification)
Dispersion coefficient data	Use defaults	Accurate (where available)	Accurate (where available)

For undertaking predictions of flux and benthic effect there are a number of input data which are commonly problematic. These are usually related to the low level of detail of husbandry data available compared to the level of detail which can potentially be used by the model. Typically monthly summaries are provided with biomass, mean fish numbers and weight and feed input. Occasionally, information is provided on feed pellet diameter and the proportion of different species in terms of food ration (model variable = feed input). Care should be taken that it is known whether monthly summaries provided are for the cage group or whole farm.

Monthly information is rarely provided on a cage by cage basis. In these cases, complexity level 2 should still be used but the combined faecal settling velocity distribution for bass and bream should be used. Similarly, the mean feed settling velocity should be used or a settling velocity representative of feed pellets being used in the cages above the benthic sampling transect. In addition, if cage specific feed input data is not provided the monthly value should be used for all cages so that feed input is the same for all cages. Care is required in this case, especially if benthic sampling was undertaken underneath cages containing large fish where the real feed input may be greater than the averaged value used in the model. In this case, the model is likely to underpredict flux and benthic effects at the benthic sampling location and overpredict effects at shallower locations where cages containing smaller fish are located. It is typical for an average feed input of the three months up to the benthic sampling period to be used in the model predictions, but some modifications to this approach can be used if these data are significantly different in these months.

A.5 Typical scenarios

Planning (baseline) - Hourly averaged hydrographic data (ten or twenty minute sampling interval) with a length of at least one month are recommended. Mean and maximum feed inputs for a growing cycle are the usual scenarios undertaken. Typically flux and benthic effect predictions are obtained from these scenarios.

Monitoring - Hourly averaged hydrographic data (ten or twenty minute sampling interval) with a length of at least one month are recommended. Feed input data for the benthic sampling month or for an average of the three preceding months are recommended. Typically flux and benthic effect predictions are obtained from these scenarios.

Experimental field studies and model validation (solids/carbon flux) – The model can be used for comparisons of solids flux with sediment trap data. It is <u>essential</u> that current meter data are collected over the complete duration of the study. For short term studies (e.g. a few days or less), unaveraged hydrographic data (five minute sampling interval) are recommended. Feed input data and species data for individual cages for the experimental period are <u>essential.</u>

Experimental field studies and model validation (tracers/chemicals) – The model can be used for predictions of flux of a chemical or material acting as a tracer. To undertake this, the concentrations of the chemical on feed/faeces must be specified or the concentration on feed and percentage excretion of chemical in the species is required. It is <u>essential</u> that current meter data are collected over the complete duration of the study. These may be hourly averaged ten minute or twenty minute sampled data if the study length is over two weeks or so. Feed input data and species data for individual cages for the experimental period are <u>essential.</u>

Other model applications – MERAMOD has capability to degrade mass via a first order kinetics decay model (G-model – Westrich and Berner, 1984). As the decay constants for the wastes, particularly faeces, are not commonly available this model component is rarely used despite its potential for appropriate application.

In all cases, meteorological data should be collected concurrently and are essential for interpretation of the hydrographic data, although these are not used directly in the model.

A6 Model output and reporting

A contouring package such as Surfer for Windows^{TM} (Golden software) or similar is required for visualisation of output in contour plots of flux and benthic effect. MERAMOD does not provide extensive visual output. Attention should be paid to the minimum contour level used in the contour maps as this should not be set too low. In addition, artefacts of contouring algorithms and the model are often highlighted in these maps in initial scenarios. Action should be taken to eliminate these or at least to highlight them in the figure caption.

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