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The suitability of DEPOMOD for use in the management of finfish aquaculture sites, with particular reference to Pacific Region

La pertinence de DEPOMOD comme outil de gestion de la pisciculture, plus particulièrement pour la Région du Pacifique

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ABSTRACT

We have applied the Scottish aquaculture waste model DEPOMOD to one British Columbia finfish farm as part of a project to test and validate this model for use in the Pacific Region. The site modeled had an extensive suite of field observations at multiple time points during a growth cycle. Detailed farm productions and configuration information were complete for the corresponding period. Since several parameter settings used in the model are not well known, we explored the effect of this uncertainty on the predicted carbon flux by running a series of simulations using a range of values representing best- to worse-case estimates. The resulting model predictions covered a broad range of outputs whose extremes represented the most optimistic and pessimistic scenarios in terms of benthic loading.

The simulation of resuspension processes within the model resulted in predictions where virtually all of the applied material was exported from the model domain and precluded meaningful comparisons of model fluxes with field data. Model predictions, with no resuspension, showed the expected steep gradient in carbon flux with distance from the edge of the net cages.

Significant relationships were demonstrated between predicted carbon flux (no resuspension) and several measures of benthic impact, namely sediment sulphide concentration, species diversity, infaunal trophic index (ITI) and faunal abundance. The sediment chemistry and biology showed a clear effect from the deposition of wastes from the finfish farm.

We discuss key limitations of the model, uncertainty surrounding model parameter settings and simulation of resuspension processes, and make several recommendations for further research and testing of the model.

RÉSUMÉ

Le modèle écossais DEPOMOD, sur les déchets d'aquaculture, a été appliqué à un site piscicole de la Colombie-Britannique dans le cadre d'un projet visant à tester et valider ce modèle dans la Région du Pacifique. Une vaste série de données de terrain était disponible pour le site modélisé à différents moments d'un cycle de croissance. Des informations complémentaires détaillées sur la production et la configuration de la ferme ont été récoltées pour la période correspondante. Étant donné que les valeurs de certains paramètres utilisés dans le modèle étaient mal connues, nous avons exploré l'effet de cette incertitude sur l'estimation du flux de carbone. Pour ce faire, une série de simulations utilisant une gamme de valeurs, des meilleurs aux pires cas estimés, a été réalisée. Les prévisions du modèle couvrent une large gamme de réponses où les extrêmes représentent les scénarios les plus optimistes et pessimistes en terme de charge benthique.

La simulation des processus de resuspension a produit des prévisions où la presque totalité des particules a été exportée hors de la grille du modèle, empêchant ainsi la comparaison entre les flux modélisés et les données de terrain. Les prévisions du modèle, sans resuspension, ont montré un fort gradient attendu de flux de carbone, en fonction de la distance du bord des cages.

Des relations significatives ont été démontrées entre le flux de carbone prédit (sans resuspension) et plusieurs mesures d'impact benthique, comme la concentration de sulfides dans le sédiment, la diversité spécifique, l'indice d'intégrité biotique et l'abondance faunique. Les paramètres chimiques et biologiques mesurés dans les sédiments ont nettement été affectés par la déposition de déchets piscicole.

Les limites clés du modèle, l'incertitude associée à la paramétrisation et la simulation des processus de resuspension sont discutées. Des recommandations pour des études et des tests futurs du modèle sont présentées.

INTRODUCTION

Concomitant with the growth of aquaculture operations worldwide has been a rising level of concern regarding the environmental consequences and sustainability of the industry. There are a range of potential impacts resulting from open net cage finfish farming on the marine ecosystem. The most frequently reported and best characterized of these are the effects of increased sedimentation to the proximal benthic environment. Indeed, many reports and scientific papers have been published that describe and quantify changes to the near-field benthic environment resulting from fish farms (Ackefors and Enell, 1990; Bergheim *et al.*, 1991; Braaten, 1991; Silvert, 1992; Folke *et al.*, 1994; Gowen *et al.*, 1994; Hansen, 1994; Findlay *et al.*, 1995; Rosenthal *et al.*, 1995; Bergheim and Åsgård, 1996; GESAMP, 1996; Burd, 1997; Findlay and Watling, 1997; Dudley *et al.*, 2000; Mazzola *et al.*, 2000; Morrisey *et al.*, 2000; Nash, 2001; Pearson and Black, 2001; Pohle *et al.*, 2001; Buschmann, 2002; SECRU, 2002; Brooks and Mahnken, 2003; Carroll *et al.*, 2003; Wildish *et al.*, 2004).

Wildish *et al.* (2004) provided a review of knowledge to date on the near field effects of marine cage fish farming operations. A combination of factors including production levels, feed characteristics (including ingredient composition and digestibility as well as physical characteristics such as pellet size etc), feeding efficiency, bathymetry, circulation and the assimilative capacity of the benthic environment will all influence the degree of impact to the benthic community and fish habitat. Levings *et al.* (2002, Appendix 1) used 12 criteria to demonstrate effects of organic enrichment on fish habitat productivity at an abandoned fish farm site in British Columbia. “Area affected”, which is a key result from some of the models cited above, was one of the criteria.

Minimizing the impact to the benthic environment is a central element in the monitoring and regulation of marine aquaculture operations in many countries. Generally, this is implemented through monitoring indicators of benthic health in the vicinity of farm sites, at regular intervals or specific time points during the production cycle. Monitoring data are used to determine the scale and extent of any benthic effect that has occurred, but collection of field data is time consuming and costly. Over recent years, there has been an increased interest by regulators and operators alike to have a predictive capability in assessing the assimilative capacity of the benthos to a range of potential aquaculture scenarios. Models that predict the potential benthic impact of aquaculture operations can also be used in the planning phase of the development to assess appropriate size of farms and thereby influence considerations of the economic viability of individual farms. Because models map both the shape and extent of the depositional field of the farm wastes, model outputs may be used to target and/or direct monitoring programs, thereby reducing the costs of high resolution monitoring schedules. Moreover, the modeling of waste deposition and benthic impacts of aquaculture operations are increasingly being recognized as important components of the management process (Ervik *et al.*, 1997; Henderson *et al.*, 2001; Cromey *et al.*, 2002a; Pérez *et al.*, 2002; SEPA, 2003).

To be effective, the overall model must adequately represent all the important processes (e.g. advection, deposition, resuspension etc.) that lead to and cause benthic impact. In addition, the model requires extensive knowledge of the environment (spatial and temporal), the farm configuration and production, and the characteristics of the particles being modeled. The accuracy of model predictions will be determined by the suitability of the model to the test environment, how the model is configured and the quality of the data used.

Inherent in the model are the inaccuracies that result from the simplification of the key processes that the model attempts to simulate (e.g. hydrodynamics, particle deposition, etc.). In addition, there is uncertainty regarding a number of key parameters used in aquaculture impacts models and the consequent effect these have on the model outputs. Finally, errors in the input data (current measurements, feeding rates, etc.) will also lead to inaccuracies in the model outputs. Although some models have been validated against certain specific environmental conditions, the extrapolation of these results to other sites with differing conditions should be approached cautiously.

The overall objective of this paper is to test the suitability of DEPOMOD – a model of finfish farm solid wastes – for use in the management of finfish farms in British Columbia. We do not use the benthic impact module of DEPOMOD since it was only validated for the near shore areas around Scotland where the model was initially developed. Instead, our approach is to determine if a significant relationship exists between predicted waste fluxes and various measures of benthic impacts as measured at a B.C. salmon farm. We begin with a general overview of aquaculture impact models followed by a more detailed description of the particle tracking model DEPOMOD and its constituent sub-models or modules. The next section discusses the processes that are modeled in the individual DEPOMOD modules, the data input requirements and the range of settings for the various model parameters. The section on model validation describes the application of the model to one BC finfish farm – Site A – and the comparison of model outputs with the field measurements. Limitations in the model and uncertainties in model parameter settings are discussed. We end with the conclusions and recommendations for further work.

INTRODUCTION TO MODELS OF AQUACULTURE IMPACT

A general review of modeling approaches to fish farm impacts was undertaken by Silvert and Cromey (2000). A number of sedimentation models have been developed which predict the magnitude and spatial extent of the deposition of particulate matter from finfish farms. These models typically attempt to predict the trajectory of particles of waste (waste feed pellets and/or faecal material) as they fall through the water column and are deposited on the seabed (Hevia *et al.*, 1996).

The fundamental forcing parameters used in these models were initially reported by Gowen *et al.* (1989) as the hydrographic regime, depth of water column, and the settling velocity of the waste material (Figure 1). Over time, increasingly complex models have been developed, improving the use of these fundamentals with the incorporation of spatially varying flow fields and detailed bathymetric grids. The use of Lagrangian particle tracking algorithms to describe the trajectory of individual particles from a defined point in the water column to their intersection with the seabed (e.g. Panchang and Richardson, 1992; Cromey *et al.*, 2002a&b) was another significant advance in techniques.

Information on farm configuration (net pen dimensions, layout and positions), fish production (species, biomass, size) and feed input to the site are important necessary data requirements. Data on quantity and quality of waste material produced from the farm are critical in determining the nature and scale of effect on the benthos.

Once wastes settle onto the bottom, the currents, if sufficiently strong, may transport wastes through resuspension and saltation processes. Physical removal and transport of material away from a point source through resuspension processes depend on a number of key factors (Clarke and Elliot, 1998; Cromey *et al.*, 1998) and often result in a reduction of material available to the benthic community proximal to the farm.

The final component or step in the modeling process is to predict some measure of change in the benthic community and/or sediment quality as a result of increased flux or accumulation of waste material. A number of semi-empirical models have been developed that predict measures of benthic impact such as the benthic enrichment index in sediments (Hargrave, 1994) or indices of benthic diversity (Cromey *et al.*, 2002a). When such relationships can be demonstrated to be significant, model predictions of the degree and spatial extent of benthic impact may be made at other locations having similar substrates, oceanographic and hydrodynamic conditions.

DEPOMOD

The most well developed and often published model of the processes leading to, and the biological consequences of, the deposition upon the sea bed of particulate wastes from marine cage fish farms is the DEPOMOD model (Cromey *et al.*, 1999, 2000, 2002a&b). It is notable among fish farm impact models in that a number of the processes it models have been validated against field measurements. DEPOMOD was developed at the Scottish Association for Marine Science, Dunstaffnage Marine Laboratory in Oban, Scotland. DEPOMOD is used as a regulatory tool in Scotland for discharge consents of in-feed chemotherapeutants (SEPA, 2003) and is currently in the developmental stages for use in setting biomass limits (SEPA, 2005). Elsewhere, the model has a distribution of licensed users worldwide in Europe, Canada, Australia, Chile and South Korea, where, for the most part, the model is currently used as a research and/or site planning tool.

DEPOMOD predicts particle deposition on the seabed arising from finfish farms and associated changes in the benthic community structure. Using the sinking characteristics for the wastes, together with information about the local hydrographic conditions and quantity of wastes (faeces and waste food particles), the model then maps the accumulation or sedimentation rate (flux) of wastes on the model grid.

DEPOMOD consists of a series of modules that address components of the processes involved. The main modules are

- Grid generation
- Particle Tracking
- Resuspension
- Benthic impact

Figure 2 (amended from Cromey *et al.*, 2000) illustrates how the different modules run sequentially and the type of information required by each of the individual modules.

The grid generation module takes user defined input data of bathymetry, cage and sample station positions and generates a sea bed array which is used by the particle tracking and resuspension modules. A major grid is first created covering the overall area of interest. With experience, a reasonable estimation of the potential area of impact can be deduced through examination of the site hydrographic and bathymetric data. A minor grid with finer resolution is then defined on areas where deposition is likely to occur. It is possible that the major and minor grids may be virtually the same size.

A Lagrangian particle tracking model is used to simulate the settling of particles and their movement within the water column. The model applies a large number of particles to simulate the waste material and requires information characterizing the particles' attributes (e.g. settling velocity, mass, % carbon content). The particles' start positions are randomly defined within the defined cage structures, after which they are advected in two dimensions by the applied hydrographic flows. Particles are also subjected to a random walk (Allen, 1982) in a horizontal and vertical direction as a representation of turbulence. Particles are tracked as they pass into different grid nodes or hydrodynamic layers, thereby allowing a change in trajectory as particles settle through the water column until they intersect with the bathymetric grid. The output of the particle tracking module describes the initial deposition footprint on the seabed of the particles ejected from the farm site.

A resuspension model (Cromey *et al.*, 2002b) addresses the observed though poorly understood resuspension process and further advection of this deposited material. Particles are subject to resuspension when the applied near-bed velocity exceeds a pre-defined critical shear stress. The quantity of material available for resuspension is determined by the initial deposition patch and the consolidation time of material on the model grid. The proportion of material that is resuspended from the seabed is defined by a relationship between the near-bed current velocity and a defined 'erodibility function'. Resuspended particles are advected according to the applied flow field until the current flow falls below a velocity equivalent to a pre-defined critical shear stress for deposition, after which they are redeposited onto the model grid.

The final module within DEPOMOD predicts the benthic community response to the modeled solids accumulation on the seabed. As stated previously, this module is not applied in this present study as it has been validated for Scotland only. One of the aims of this project is to explore the relationships between predicted carbon flux and measures of benthic impact at B.C. salmon farm sites. The following description of the DEPOMOD benthic impact module is presented for information purposes only.

The model uses empirical relationships between modeled solids flux and observed changes in the benthic community in terms of total abundance (number of individuals m^{-2}) and the Infaunal Trophic Index (ITI) (Word, 1990). The purpose of the ITI is to describe the feeding behaviour of soft bottom benthic communities in terms of a single understandable parameter. These animals fall into four groups; they are either suspension or deposit feeders that feed above, on or below the sediment surface. The feeding modes employed by the benthic community has been shown to correlate with the degree of disturbance to the benthos. It is important to note that Cromey *et al.* (2002a) advise that the inclusion of the

ITI over other traditional indices in their modeling does not imply that the ITI is a superior index. Indeed, a multitude of indices have been developed over the years to describe the status of the benthic community structure, each with different strengths and weaknesses. At present, there appears to be no consensus on which measure is most accurate or appropriate for discerning perturbations on benthic community structure from aquaculture operations.

DEPOMOD METHODS AND SETTINGS

The input data, methods and parameter settings required to set up and run DEPOMOD for a particular finfish site are discussed in this section. As discussed earlier, DEPOMOD is not a single model but a collection of sub-model or modules. The key modules for our paper are the grid generation module, particle tracking module and the resuspension module. The output of interest is the prediction of the waste fluxes or footprint of the farm. We have chosen total carbon as the key waste parameter of interest as it is presently being used as a threshold for delineation of areas potentially subject to a Fisheries Act Section 35(2) Authorization by Fisheries & Oceans Canada for British Columbia finfish farms.

The default settings used within DEPOMOD were formulated from Cromey *et al.*, (2000; 2002a&b) and from the Scottish Environment Protection Agency's Fish Farm Manual (SEPA, 2003). Stucchi and Chamberlain (2004) specified methods and settings for DEPOMOD simulations for B.C. finfish farms. However, the latter is an active document as it is acknowledged that some of the parameter settings will require updating from time to time as new research findings become available, as new diet formulations are developed, and as technological changes are implemented in the operation of the farms.

BATHYMETRY/MODEL GRID RESOLUTION

The level of detail required to create a model grid that is representative of site conditions is determined by the complexity of the local topography. If the area is relatively flat with little change in depth across the domain, then data collected at a low resolution may be suitable. In areas of more complex bathymetry, a low spatial resolution data set along with a standard linear interpolation technique may not result in an accurate representation of site conditions. Emphasis should be directed on producing higher quality bathymetric data in those areas where the majority of particles will be deposited on the grid. Less importance may be placed on data representing areas that are not well used within the model.

Bathymetric data may be obtained from a variety of sources. The accuracy of the model grid will be determined by the spatial resolution of the sampling points and the topography of the seabed. The results of an acoustic survey at the site should produce excellent resolution data. Canadian Hydrographic Service (CHS) field sheets of the area of interest generally provide good resolution, whereas CHS navigational charts may present only sparse data points. Appropriate model grid resolution will depend on the expected dispersal and spatial sampling frequency of field survey data (typically 5 to 25 m).

HYDROGRAPHIC INFORMATION

The use of high quality hydrographic data that are representative of site conditions is essential for confidence in model outputs. Sensitivity analysis has shown that hydrographic data are an important factor in particle tracking model predictions (Silvert and Cromey, 2000; Cromey and Black, 2005). Early models used summary statistics of current flow that was constant both spatially and with depth (Gowen *et al.*, 1989). More recently, the most common approach uses depth varying, horizontally homogenous flow fields. This provides sufficient detail close to the cages and current meter mooring position, but decreases with increasing distance from the source of measurement. An approach to overcome this reduction in data quality has been to couple aquaculture impact and hydrodynamic models which may apply flow fields that vary horizontally across the model grid (Panchang *et al.*, 1997, Doglioli *et al.*, 2004) which is particularly important where circulation patterns are complex. However, the availability of these models in areas where aquaculture operations are ongoing is rare because of the cost and time necessary to develop and test them. Additionally, these models are normally pseudo 3-D as generalized vertical profiles are fitted to the predicted flow fields. Another disadvantage to the use of 3-D hydrodynamic models is the coarse scale which is at best 50 m, and in reality, does not provide any more information than a single point mooring near the farm site with respect to the area where the majority of particles settle.

The effect of different hydrographic data records, collected from the same location, on model outputs was investigated by Cromey and Black (2005). They examined the application of multiple contiguous 15 day long hydrographic records from a 206 day long data (a 15 day data set is the minimum record length requirement in Scotland (SEPA, 2003) and is taken to represent one spring-neap cycle). Significant differences in the model outputs were obtained when using the different hydrographic data sets and short 15 records were considered unlikely to be representative of the site under all conditions. Cromey and Black (2005) cautioned that any decision based on model outputs will be completely dependant on the conditions during the measurement period and the reliability of predictions would be increased with longer current records.

In British Columbia, the Provincial regulatory requirement for hydrographic data at aquaculture sites is for a continuous 30 day data set (BCMWLAP, 2002). We consider this should provide greater information on site conditions than the Scottish standard. However, it should be noted that the data still may not reflect ‘normal’ site conditions and the effect this may have on the model output is an important issue.

PARTICLE INFORMATION

For a given feed input, there will be a proportional output of waste feed or uneaten feed pellets, and faecal material that will settle onto the bottom. Accurately specifying the quantities and the physical and chemical properties of these particles is critical to the modeling process and uncertainties will result in a corresponding uncertainty in model outputs. For the present modeling study, a range of values were used to quantify the parameters defining the particles’ characteristics. These values were identified from the literature as encompassing the range of measurements and were considered to represent a reasonable range from best- to worse-case estimates. A centre 'median' value was also used to represent average conditions.

Feed Particles

The feeding rates input to the model for a particular finfish farm may be obtained from a variety of sources with differing levels of detail. The level of detail required and the method used to determine the daily feed inputs will depend on the objectives of the modeling study. There are some detailed models of fish growth that are used to optimize feeding schedules. An alternate source of feed input information is from the farm operator who maintain detailed feeding records and may be able to make projections of feed budgets based from past experience at a particular site or area. Feed input data may be available in detailed form (e.g. kg feed cage^{-1} day^{-1}) or as the total feed budget for the production cycle (e.g. 2,400 tonnes feed over 18 month grow-out period). Both of these are suitable for modeling localized impact predictions. If the purpose of the study is to simulate the footprint of a farm site at, for example, peak biomass, then total feed input for the specified period is probably sufficient.

Waste Feed Particles

The amount of feed that is uneaten or wasted is a major source of uncertainty in all studies modeling the fate of solid wastes from finfish farms. Few studies on feed wastage exist in the literature and generalizations are difficult given the variation in husbandry and feeding practices. Improvements in feed monitoring systems, such as through the use of underwater cameras, have resulted in significant reductions in the amount of waste feed. Gowen *et al.* (1989) estimated a 20% loss, but the value is now generally considered to be much less. Findlay and Watling (1994) calculated feed wastage rates of 5% and 11% from observations, whilst Pearson and Black (2001) reported this value to be in the 5% to 15% range. Brooks and Mahnken (2003) in their review paper indicate that present day feed wastage rates are 5% or less.

Interestingly, UK regulatory agencies propose to use a wastage rate of 3% to represent 'worse case scenario' (SEPA, 2005) in their modeling simulations. Stucchi *et al.* (2005) used 8%, the average of the Findlay and Watling (1994) range, for their modeling study. For DEPOMOD simulations in B.C., Stucchi and Chamberlain (2004) have specified a value of 3% based on feedback from farm operators in British Columbia. An additional consideration is that although waste feed is generally believed to be in the form of uneaten feed pellets, Brooks and Mahnken (2003) reported that a small percentage (<0.5 %) may result in the abrasion of uneaten feed pellets as they travel through automated feeding systems.

Given the different setting for this parameter reported in the literature a range of values (0%, 5% and 10%) will be used in this modeling study.

The calculation of rate of solids deposited (dry weight basis) on the bottom from the wasted feed is straightforward given the concentrations of water F_w in the feed, the feeding rate F (kg d^{-1}) and the feed wastage rate F_{wasted} (Eq. 1).

$$W_{feed} = F\,(1 - F_w)\,F_{wasted} \tag{1}$$

The rate of carbon deposition from the waste feed is then simply the product of the rate of solids deposition from the waste feed W_{feed} and the concentration of carbon in the feed pellets.

McGhie *et al.* (2000) and Sutherland *et al.* (2001) reported similar total carbon concentrations of around 57% in the food provided to cultured salmonids. In an earlier study, Hall *et al.* (1990) measured total carbon concentrations in various diets for trout to be in the range 50% to 52% (dry weight). Findlay and Watling (1994) reported that the organic carbon content of feed pellets was 46% ± 6% of the weight of food (wet) and ranged from 32% - 53% carbon for dry weight of food. In their DEPOMOD Methods and Settings document, Stucchi and Chamberlain (2004) specify a value of 57% to be used in model simulations.

Given the range of values reported for the carbon content of various commercial diets we tested the effect of varying this parameter on the model output by setting the carbon concentration to values of 45%, 55% and 65%.

<u>Faecal Particles</u>

The quantity, and physical and biochemical properties of the faecal material will vary with fish species, size of fish, health of the fish and composition of the diet. Tlusty *et al.,* (2000) found that different faeces may be produced by the same type of fish, which they termed 'granular' and 'cohesive', which would result in different structure, carbon content and settling velocity (see below).

Following Cromey *et al.* (2002), the calculation of the production rate of faecal solids or dry matter W_{faeces} is based on the amount of dry food consumed or ingested per day, $F_{consumed}$, and the digestibility F_{dig} of the food (Eq. 2). The dry food consumed per day is calculated by correcting the total daily feeding rate F for the concentration of moisture F_w in the food and for the proportion that is wasted of uneaten F_{wasted} (Eq. 3).

$$W_{faeces} = F_{consumed} (1 - F_{dig}) \tag{2}$$

$$F_{consumed} = F (1 - F_w) (1 - F_{wasted}) \tag{3}$$

The rate of carbon deposition from the faecal material is then calculated from the product of the rate of faecal solids production W_{faeces} and the concentration of carbon in the faecal dry matter.

The literature on the carbon concentration in faeces of cultured salmonids, usually Atlantic salmon (*Salmo salar*), is variable and based on relatively few samples. Findlay and Watling (1994) measured the organic carbon content of faecal pellets to be 19-25% (dry weight). Panchang *et al.* (1997) reported that the faeces of salmonid smolts contained 28-35% carbon (dry weight). Chen *et al.* (1999) reported that the faeces of 0.7 to 1.0 kg Atlantic salmon (*Salmo salar*) contained 32% to 35% total carbon (dry weight), whilst McGhie *et al.* (2000) measured 40.3% and 31.8% in faecal samples from trout (*Oncorhynchus mykisss*) and Atlantic salmon (*Salmo salar*) respectively.

Given the range of values reported for the carbon content of faecal material we tested the effect of varying this parameter on the model output by setting the carbon concentration to values of 30%, 40% and 50%.

Settling Velocity

All particle tracking models require characterization of the settling velocity of particles in order to calculate their horizontal displacement and sedimentation rate. Within models, the settling velocities may either be assigned single mean value (Gowen *et al.*, 1989) or, more realistically, be treated as a probability distribution with defined mean and standard deviation (Hagino, 1977).

Several studies have been carried out in the laboratory to investigate sinking rates of feed pellets. Cromey *et al.* (2002a) reported a relationship between sinking rate and feed pellet diameter and determined a mean sinking rate of 10.8 cm s^{-1} ± 2.7 cm s^{-1} for a pellet diameter of 7mm. A similar mean settling velocity of 10 cm s^{-1} was reported by Panchang *et al.* (1997). Findlay and Watling (1994) reported sinking rates between 5 - 15 cm s^{-1} for a range of pellet diameters.

Stucchi *et al.* (2005) used a sinking rate of 10 cm s^{-1} in their modeling study, but caution that the sinking rate of feed pellets will be affected by both size and composition of the feed pellet as well as the pelleting condition used during manufacture.

The determination of sinking rates of faecal material is not as well studied, probably because it is not as straightforward as for feed pellets. The particles are not inert spheres as assumed in Stokes Law calculations but rather dynamic organic particles. Faecal material is generally in the form of a gelatinous mass or mucoid string that may disintegrate whilst in the water column.

Chen *et al.* (1999) reported settling velocities of faecal material within the range of 4 to 6 cm s^{-1} with a mean value of 5.3 cm s^{-1}. Similar velocities were reported by Panchang *et al.* (1997) with a mean settling velocity of 3.2 cm s^{-1} (although they used a value of 4 cm s^{-1} during modeling) with 70% of their observations within the range of 2 to 4 cm s^{-1} from smolts (approx 0.11 kg). Further, Cromey *et al.* (2002a) reported faecal settling velocity of 3.2 cm s^{-1} ± 1.1 cm s^{-1}.

In summary, the reported sinking rates of waste feed pellets lie within a wide range of values from 5 to 15 cm s^{-1} which depend upon pellet size. The reported settling rates of faecal material cluster around a mean value of around 3 cm s^{-1}. Therefore, for the present modeling study, settling velocity values of 11.0 cm s^{-1} and 3.2 cm s^{-1} ± 1.1 cm s^{-1} were used for feed pellet and faecal particles respectively.

RANDOM WALK

Representation of a turbulent flow field may be simulated in particle tracking models by the application of a random walk model (Allen, 1982). Dispersion coefficients used to parameterize these models may be calculated from drifter or dye surveys. Site specific data are generally not available and are of limited usefulness as they only represent conditions of the area whilst surveying was being carried out. To overcome this limitation, regulatory models applied in Scotland use standardized horizontal and vertical dispersion coefficients determined by Gillibrand and Turrell (1997) of 0.1 m^2 s^{-1} for k_x and k_y, and 0.001 m^2 s^{-1} for k_z. In the absence of dispersion data at the site modeled in this report, the above values were applied within the model.

RESUSPENSION

Resuspension models typically attempt to simulate the effect of near-bed flow fields on the erosion and transport of recently deposited material on the seabed. It is generally acknowledged that this is a complex process that is difficult to quantify accurately and may vary substantially between substrate types. The fluctuation of near bed current speed and the degree of "stickiness" of the freshly deposited material, among others, will affect the potential for resuspension. However, there are a number of models that have achieved notable levels of validation (Panchang *et al.*, 1997; Cromey *et al.*, 2002b).

These models are usually parameterized by three main variables:

1. critical threshold velocity for resuspension – the near bottom current velocity at which particles are resuspended off the seabed.
2. erodibility function – defining the quantity or proportion of material resuspended as a function of the applied near bed current velocity.
3. consolidation time – the period of time that a particle remains on the seabed before it is assumed to be part of the sediment and no longer available to resuspension processes.

The critical threshold velocities for resuspension reported by Cromey *et al.* (2002b) and Panchang *et al.* (1997) were very different and would result in contrasting predictions of resuspension effects, despite both having been validated in areas containing aquaculture operations. Not only would differences in solids accumulation beneath the cages be different, but this would also affect the predictions of net mass export from the areas. In an examination of the thresholds used in different reports, Cromey and Black (2005) noted that the type of particulate material, environment and method employed during studies varied quite significantly which may go some way to explaining the starkly contrasting threshold values (see below).

Typically, freshly deposited material, so called 'fluff', is considered to have a much lower resuspension threshold than consolidated bed material, with values close to 10 cm s^{-1} reported by numerous researchers (Burt and Turner, 1983; de Jonge and van de Bergs, 1987; Sanford *et al.*, 1991; Washburn *et al.*, 1991; Lund-Hansen, 1997). Recent unpublished work has provided evidence of waste feed particles saltating along the seabed, however, no quantitative values for shear stress to induce such movement are currently available.

Cromey *et al.* (2002b) carried out a validation study using a UV fluorescent particle tracer that had similar diameter, specific gravity and settling velocity to fish faecal material. The particles were deployed on the seabed and the resuspension and redistribution processes quantified through an intensive sampling program. They calculated that a critical resuspension velocity of 9.5 cm s^{-1} resulted in the best agreement between observed and predicted mass budget analysis. This value is similar to other erosion thresholds of freshly deposited material (see above references).

Panchang *et al.* (1997) used a range of critical erosion thresholds from 0 to 30 cm s^{-1}. Dudley *et al.* (2000) used annular flumes deployed on the seabed to investigate erosion events by measuring turbidity within the flume at different current velocities. Erosion thresholds were calculated between 33 and 66 cm s^{-1}, however, the authors advise that this method would measure the erosion of both the fish farm material and natural sediment. Consequently, if realistic impact predictions are to be made, further investigation of these processes is required.

In DEPOMOD, the critical resuspension velocity is "hard-coded" to 9.5 cm s^{-1} and cannot be changed by the user. Consequently, for this modeling exercise, we investigated the effect of resuspension processes by either enabling the resuspension process – turned ON - or disabling the resuspension process – turned OFF- in the model.

VALIDATION

In this section we examine the application of the model at one farm site in British Columbia – from hereon referred to as Site A - for which there was high quality field survey, site and husbandry information available for multiple time periods during a production cycle covering 2000 and 2001. High quality/resolution data are essential during testing model validation and may not be quite as stringent for general application of the model once variables have been quantified and parameterized.

The model was applied using these data whilst testing the effect of different parameter settings for:

- Percentage waste feed component
- Carbon concentration of feed particles
- Carbon concentration of faecal material

within the ranges described previously. Silvert and Cromey (2000) emphasize the importance of sensitivity analysis in determining how much uncertainty in different parameter values contribute to the overall performance within the model. The model outputs will provide useful information on the uncertainties associated with these parameters and overall model suitability.

SITE DESCRIPTION

Bathymetric Data

Detailed information on the bathymetry and coastline around Site A was obtained from the Canadian Hydrographic Service field sheet of the area. The resolution of sounding points was sufficient to allow a model grid to be produced that was a good representation of the area. The seabed relatively flat to the north and east of the farm site (Figure 3) with a gentle upward slope leading towards the shoreline to the south and west of the cage structure.

Hydrographic Information

High quality hydrographic data were collected at the site using an RD Instruments Workhorse Sentinel 300 kHz acoustic Doppler current profiler (ADCP) mounted near the seabed on a small aluminum tripod. The ADCP was deployed towards the west end of the cage group, approximately 30 m north of the pens in 35 m depth (Figure 3). Current velocities were measured in 2 metre bins from the near bottom to the near surface with the centre of the first bin approximately 5 m above the seabed. Data were recorded for over 30 days starting on 1 September 2000. The complete description of methods, data and analyses are reported in Cross *et al.* (2001).

Three of the data bins were selected for use within the model with bin centers located at 5 m, 17 m and 29 m above the seabed or at depths of 6 m, 18 m and 30 m below mean sea level. The data from the 3 depths were considered to be representative of the current velocity profile of the site as the velocity profile was relatively uniform over the water column.

Speed and direction measurements were made every 20 minutes which were subsequently hourly averaged for use input to the model. Summary statistics of these data are presented in Table 1 and presented graphically in Figure 4. The currents at this site were predominantly tidal with reasonably energetic speeds. Maximum speeds were in the 30 to 40 cm s^{-1} range and mean speeds in the 8 to 11 cm s^{-1} range. The principle direction of the flow was NW-SE (Fig. 4). In the top half of the water column, there was a residual flow to the northwest of approximately 5 cm s^{-1} which then diminished to zero near the bottom. It is interesting to note that both the mean and maximum current speeds were greater in the mid-water data than near the surface, which may indicate some form of shading from the cage structure.

Farm Configuration

The farm configuration during the period of interest for this study is shown in Figure 3. The farm comprised of ten 30 x 30 m cages (15 m depth) in a single line, with two additional 15 x 15 m cages at the west end of the group. Farm records indicate that there were additional cages attached to the site at periods during the grow-out cycle but these did not contain stock. The operator numbered the cages 1 to 12, with the 15 x 15 m cages being numbers 1 (north) and 2 (south) and then from 3 to 12 in order in an easterly direction. We have maintained this numbering system throughout this report.

Fish were routinely transferred and split between pens during the production cycle. The feed inputs to the different cages during the period of interest for this study are presented in Table 2.

FIELD SURVEY DATA

Brooks (2001a) carried out an extensive study – The Focus Study- of a number of aquaculture operations in British Columbia during the summer and autumn of 2000. To quote Brooks (2001a) “The purpose of this study was to evaluate the predictability of biological responses to organic loading using surrogate physicochemical endpoints and to determine the spatial extent of physicochemical and biological changes in sediments adjacent to British Columbia salmon farms". Field data collected at one of these farm sites – also referred to as Site A in the Brooks (2001a) report – was the focus of our interest for this particular modeling study.

A wide range of benthic parameters were recorded during surveys carried out in June, August and October 2000, including sediment sulphide, redox, total volatile solid (organic matter), particle size analysis and benthic community structure. All samples were collected using a 0.1 m^2 modified van Veen grab, with subsamples for physical-chemical analysis being taken prior to benthic macrofauna samples being sorted through a 1 mm mesh. Full methods for sample processing and analysis are given in Brooks (2001a). Additional analysis of the benthic community structure was carried out by Brooks (2001b) using the Infaunal Trophic Index (ITI) to examine changes to the benthic community structure.

On each survey, a 225 metre long transect was taken in a NNW direction away from the cage structure along the 30 to 35 m depth contour (Figure 3). Sample stations were spaced at 15 m intervals from the cage edge to 105 metres and 20 m intervals from that point to the end of the transect. Sample station locations were determined using DGPS (Differential Global Positioning System) and were observed to vary by as much as 15 m between each survey. All model outputs were extracted from the known position of sampling rather than distance from cage edge.

Reference stations were located approximately 750 m to the north west of the cage group in an area of similar depth and bottom type (Figure 3). Three reference samples were collected during each of the survey and identical parameters measured as those along the transect.

Single grab samples for analysis were obtained from each of the sample stations on each survey. The results of all analyses are given in Brooks (2001a&b) and are summarized in Table 3. Additional samples from the site were collected by AES (Aquatic Environmental Sciences) during August 2001 to comply with Provincial monitoring requirements. Three replicate sediment samples were collected from stations located at 0, 30 and 100 metres from the cage structure along a similar transect to Brooks (2000). Sediment analysis on this occasion was for sediment sulphide concentration only. Again, sample station locations were recorded using DGPS and the corresponding position on the model domain was calculated and model outputs extracted from these points. The results from this survey are summarized in Table 4.

MODEL INPUT DATA AND PARAMETER SETTINGS

As discussed above, setting up the model requires information on both site specific data input (feed values, cage size and location, hydrographic data, bathymetric data) and variable parameterization (e.g. waste feed settings, carbon concentrations, random walk, settling velocities).

Detailed information on feed and distribution of biomass amongst the net cages were obtained from the site operator. Monthly feed input to the different cages was obtained for the entire grow-out cycle. Dimensions of the net cages and their positions were also provided by the site operator. The feed input data used in this modeling study are presented in Table 2. The information used to parameterize the model is presented in Table 5.

Model simulations were run applying the data presented in Table 2 together with the described range of values for waste feed and carbon content of food and faecal material to examine the effects on predicted flux to the seabed.

Additionally, the resuspension model was either turned on or off – there could be no in between as discussed previously. This allowed testing of the above parameters with and without the deposited material being shifted around the model domain.

MODEL OUTPUTS

DEPOMOD outputs are in the form of an *x,y,z* array which describe the predicted flux (*z*) at specific locations (*x,y*) on the model domain. Information from this grid may be run through interpolation algorithms to produce 'footprints of deposition' which are typical of benthic impact models and provide a general overview of the dispersion of waste material. More detailed information on predicted flux at specific points on the model grid can be calculated by extracting data from a single or small cluster of nodes at the position of interest.

Modeling of all the different scenarios resulted in 54 outputs for each time point/feed input value. Each of the model footprints was examined to assess the predicted location of the depositional patch on the model grid. From this evaluation, it became apparent that model simulations that included resuspension processes resulted in predictions where almost all (98%) of the applied material was transported outwith the model grid. Conversely, when resuspension processes were not simulated, all the material was retained within the model domain, with the area of greatest predicted flux located directly beneath the farm structure, reducing in size concentrically with distance from the cage group. Clearly, the simulation of resuspension processes has a significant effect on the resulting model predictions in terms of both location and magnitude of predicted flux around the farm site.

The field survey data clearly showed appreciable alterations to the benthic community structure and sediment chemistry around the farm site, which Brooks (2001a) attributed to increased flux of material depositing onto the sea bed. These alterations were not reflected in the model predictions when resuspension processes were simulated.

Consequently, further discussion of the model outputs will focus solely simulations where resuspension was NOT applied and issues surrounding the resuspension model settings and application are reviewed in the discussion section.

The predicted carbon flux at the field survey sample station locations, as determined from the DGPS and not by distance from cage edge, were extracted from the model grids. Although using a distance measure is not specifically correct for labeling of sample stations or for comparisons between surveys, it is useful to plot predicted fluxes vs. pseudo-distance to illustrate the gradient of flux away from the farm site. These are presented in Figures 5, 6 and 7a.

For each of the different parameter setting scenarios that were tested in the model at each time step, there was a corresponding data output describing the predicted flux rate distribution. Variability and magnitude in the model outputs was greatest close to the cage structure and dramatically decreased with distance from the farm site.

As expected, different parameter settings resulted in a range of values for predicted fluxes. The median predicted flux from all the model scenarios at each sample location are presented in Table 6 and as the solid black line in Figures 5, 6, 7a. The range in predicted flux is illustrated by the dashed lines on these figures representing the highest and lowest predicted values. Thus all the predicted flux values are enclosed within these high/low envelopes. The model outputs resulting in the highest and lowest predicted values were those parameterized by the worse- and best-case estimates of particle characteristics. These high/low outputs are taken to represent the most pessimistic/optimistic model scenarios.

Predicted fluxes were lowest during June 2000 when the feed input to the farm was the smallest at an average of 575 kg d^{-1} for the whole farm. The median predicted flux close to the farm (0 m) during June 2000 was around 0.47 g C m^{-2} d^{-1}. Predicted flux ranges for June 2000 were 0.2 to 0.72 g C m^{-2} d^{-1} at the cage edge reducing to 0.1 to 0.22 g C m^{-2} d^{-1} at 60 m from the cage structure.

The increased feed input in August 2000 (average 1,182 kg d^{-1} for whole farm) was reflected in the higher predicted flux values for this sample period (Figure 6). Variability close to the cage structure was very large, with a range of 0.81 to 4.19 g C m^{-2} d^{-1}. The median predicted flux at this point was 2.29 g C m^{-2} d^{-1}. The range of variability rapidly decreased with distance from the farm, with a predicted range of 0.44 to 1.02 g C m^{-2} d^{-1} at 60 m from the cage.

An increase in flux was not predicted at the sampling locations in October (Table 6) despite the feed input more than doubling from August to October (c.f. 1,182 kg d^{-1} to 2,548 kg d^{-1}). Examination of the predicted footprint for October 2000 (Figure 8a) revealed that although there was an increase in the mass of material deposited on the grid equivalent to the increased feed input, the distribution of the wastes had shifted because of the corresponding change in biomass distribution within the cage system. The feed input and husbandry data provided by the operator of the site indicated that by the end of October 2000, a high biomass of fish had been moved from the cages that had been stocked up to that point into alternate cages within the structure (Table 2). The redistribution of fish in the farm net cages resulted in a shift of the peak waste fluxes away from the sample stations - slightly towards the east of the transect (Figure 8a).

The seemingly contradictory October decline in predicted fluxes at the sampling stations illustrates the ambiguity that results from input data that are not representative of the entire modeling time interval. We do not have information on precisely when in October the fish were redistributed in the farm cages. If the fish had been redistributed at the beginning of the month then we would have some confidence that the predicted fluxes would be representative for October. However, if the fish had been redistributed towards the end of the October 2000, then feed input to the cages would be considerably different (Table 7), and the extracted model outputs using these data were much higher, as shown in Figures 7b and 8b and in the Alt. October column of Table 6.

Because we do not know when the fish were moved in October, the model predictions are ambiguous. Consequently we present two model simulations for the October time period. One assuming the biomass redistribution as provided in the farm records (‘October’) and the other, assuming no redistribution of fish (‘Alt. October’).

From the results of the Alt. October model outputs, it can be seen that the median predicted flux at cage edge was 4.26 g C m^{-2} d^{-1} within a range between 1.62 and 7.52 g C m^{-2} d^{-1}. A similar rapid decrease in predicted flux and variability with distance from the farm site was observed.

The median predicted flux from the feed input values for August 2001 (mean feed input 10,423 kg d^{-1}) (Table 6) at the 0 m sample station location was 6.72 g C m^{-2} d^{-1} with a range from 2.55 to 11.82 g C m^{-2} d^{-1}. A dramatic reduction in predicted flux was observed at the 30 m sample station location with a median predicted flux of 4.05 g C m^{-2} d^{-1} and at 100 m this value was 1.12 g C m^{-2} d^{-1}.

MODEL COMPARISON WITH FIELD DATA

The effect of increased sedimentation from finfish farming operations on the proximal benthic environment is commonly observed as a gradient of organic enrichment, highest near the farm site and decreasing with distance. The effect of organic enrichment is observed both in the sediment geochemisty and benthic community structure.

In this present study, we compared the predicted organic enrichment gradient, as described by the median carbon flux values at the different sample station locations, with indicators of benthic health as measured by Brooks (2001a&b) and WALP (unpublished data). The different indicators used were sediment sulphide concentration, benthic community structure, macrofauna species diversity, the infaunal trophic index and macrofaunal abundance.

Data are presented using both the October and Alt. October model outputs. The general effect of these different data sets on the presented analyses is there are a greater number of data points in the higher predicted flux scale when applying the Alt. October data.

Sediment Sulphide Concentration

There is a well established relationship between organic enrichment processes and concentration of sulphide within the sediment pore water (Wildish *et al.*, 2004). The sediment sulphide concentrations measured at the sample station locations are presented in Tables 3 and 4. Values were greatest at stations closest to the farm site through all sampling periods, and increased in magnitude as the production cycle progressed. A significant relationship between predicted median fluxes (both October and Alt. October data) and sediment sulphide concentration was observed (Figure 9a&b).

There was a near linear relationship between predicted carbon flux and sediment sulphide concentration up to 4 g C m^{-2} d^{-1}, afterwhich a dramatic increase in sulphide concentration was observed. All sulphide concentration samples in excess of 6000 µM were collected at the cage edge during the August 2001 survey.

Benthic Community Structure

Relative abundance of benthic macrofauna species changed markedly between sample stations with distance from the farm site (Table 8). The relative abundance of the polychaete *Capitella capitata* decreased steeply with the increase of distance from the net-pen, while conversely, the polychaete *Lumbrineris luti* and the amphipod *Ischyrocerus* sp. increased in relative numbers. High dominance of *Capitella capitata* is generally indicative of organically enriched conditions in the sediment and an unhealthy community of macrobenthic infauna.

The relationship between the benthic community structure and predicted flux (Alt. October only) for the October 2000 survey is illustrated in Figure 10. The non-metric multi-dimensional scaling (MDS) plot of 4^{th} root Bray Curtis similarities shows a clear separation between sample stations close to the farm site and those at a greater distance. The predicted flux values at the sample station locations are show on a relative basis where the circle diameters are scaled to the maximum predicted median flux. Similar, albeit less well defined separation of the sample stations proximal to the cage structure compared to those further afield was also observed in the June and August data sets.

Macrofaunal Diversity

The relationship between observed species diversity (Shannon Index - H') of macrobenthic infauna and predicted median carbon flux is shown in Figure 11a&b. Correlation between the data is highly significant for both October and Alt. October data sets and the curve shows a steep decrease in diversity between predicted flux rates of 0.5 and 2 g C m^{-2} d^{-1}.

When predicted carbon flux was < 1 g C m^{-2} d^{-1}, the diversity of the samples was > 3 indicating a healthy and diverse benthic community. A reduced diversity of between 1.5 and 2.5 was more commonly observed when the predicted flux was > 1 g C m^{-2} d^{-1}, although there remained a number of samples with H' diversity around 3.

Infaunal Trophic Index (ITI)

The infaunal trophic index (ITI) is a numerical representation of the distribution of dominant feeding groups of benthic infauna, which may be used to quantitatively model benthic community response to disturbance of the benthos (typically organic enrichment).

Brooks (2001b) calculated the ITI score of the macrobenthic samples collected at Site A for each of the surveys (Table 9). The methodology used was consistent with Word *et al.* (1980) whereby the data used were reduced to include those taxa, representing, in aggregate ≥ 70% of the total abundance of each survey. In addition, all taxa representing > 5% of each survey were included. The different species' ITI group assignations from Brooks (2001b) are presented in Table 10.

We recalculated the different ITI scores for each of the sampling stations using a revised list of feeding group allocations. This revised list (Table 10) was determined by examination of the feeding mode of the different species observed, as described by numerous authors (Day, 1967; Fauchald and Jumars, 1979; Word, 1990). There were a small number of differences between the species allocation of Brooks (2001b) and our revised list. Our recalculation of ITI scores generally resulted in a lower value than that of Brooks (2001b).

The relationship between both Brooks (2001b) and our revised ITI scores, and the predicted median carbon flux is illustrated in Figure (12a&b). It is apparent that both ITI data sets exhibit similar trends with a steep decrease in ITI between a predicted median flux of 0.5 and 2 g C m^{-2} d^{-1}. The correlation between both Brooks (2001b) and our ITI calculations and predicted carbon flux was highly significant ($p< 0.0001$).

Similar to the diversity index results, in both October and Alt. October analyses, where the predicted carbon flux was <1 g C m^{-2} d^{-1}, the ITI scores were generally high (> 50) indicating a healthy 'unimpacted' benthic faunal community. A rapid decline in ITI was observed in most samples where predicted carbon flux was > 1 g C m^{-2} d^{-1}, suggesting that the flux of material to the seabed had exceed the tolerance of the majority of species.

Macrofaunal Abundance

Spearman correlation analysis (Table 11) indicated a significantly negative relationship between predicted carbon flux (Alt. October data only) and observed abundances of total individuals ($p< 0.01$) and three main groups (polychaetes ($p< 0.05$), mollusks ($p< 0.01$) and crustaceans ($p< 0.01$)).

DISCUSSION

With the continuing expansion and evolution of the aquaculture industry worldwide, environment agencies responsible for regulating the effects of fish farm activities have had to modify and develop their management strategies to ensure sustainable development and stewardship of the near-shore marine ecosystem. This paper proposes one such approach in the use of models that allow *a priori* assessments of the nature and scale of effect of individual aquaculture operations on their near-field benthic environment.

We ran the particle tracking model DEPOMOD, configured for an open-net pen fish farm on the British Columbia coast, simulating four specific time points during the production cycle for which detailed feed and stocking information were available. For each time point, a number of scenarios were tested, characterized by varying key parameters within the model through reasonable ranges whose extremes reflected the best- and worse-case estimates. The different model outputs were used to examine how the simulations compared with observed field measurements of both sediment chemistry and benthic community structure.

The effect of simulating resuspension processes on the model outputs was tested by enabling and disabling the resuspension module within DEPOMOD. Simulations in which resuspension was enabled resulted in almost all (98%) of the wastes being exported from the model domain and precluded meaningful comparisons with observed benthic effects. Consequently, the following discussions focus on the model outputs where resuspension processes were not simulated. These are followed by an analysis of the parameters used within the resuspension model and the processes that may affect the model outputs.

Overall, the model performed well, simulating a gradient of particle deposition with distance from the farm site. Significant relationships were calculated between predicted carbon fluxes and observed changes to the benthos at each of the different time periods examined. As the production cycle progressed and increasing quantities of feed load were applied to the farm site, the predicted carbon flux to the seabed also increased. This increase was reflected in the observed changes to the sediment chemistry and benthic community structure with an increasingly 'altered' environment being recorded with time.

Alterations to the sediment chemistry were observed in the sediment sulphide concentrations. This measure increased dramatically throughout the grow-out period as increasing quantities of material from the farm were deposited on the seabed. These changes were reflected in the model outputs with a significant correlation being observed between sediment sulphide and predicted carbon fluxes.

Multivariate analysis of the benthic community structure, as calculated by multi-dimensional scaling, illustrated delineation between samples taken close to the cage structure and those collected further away from the farm site. This separation of sample stations was shown to correlate well with the predicted carbon fluxes at these locations.

Significant correlations between predicted carbon fluxes and changes to the benthic macrofauna were observed. Changes to the benthic community structure as measured by univariate analyses (Shannon diversity index and macrofaunal abundance), illustrated significant trends in the data with predicted carbon fluxes.

Alterations to the distribution of benthic macrofauna, classified by the dominant feeding behaviour in the ITI, were compared with model predictions. There has been much debate over the use of this index (Maurer *et al.,* 1999) and with differences over the assignation of feeding modes for some species. Levington (1991) found that opportunistic feeding by many benthic species can blur the distinction between feeding categories. Dauer (1984) indicated that many polychaete families contain more than one feeding mode and deciding which mode to use for a particular species may be arbitrary. This uncertainty in allocation of species to feeding groups is illustrated in our recalculation of ITI values for the benthic community data at Site A compared to those of Brooks (2001b) (Table 10). However, despite these differences, similar significant correlations between ITI and predicted carbon flux were calculated. Importantly, differences between the two ITI calculations occurred primarily at the high end of the index (ITI > 50) which Mearns and Word (1982) classify as unimpacted and likely to exhibit a high degree of natural variability. Conversely, there was little difference between the two ITI calculations towards the low end of the scale (ITI < 30) where both approaches indicated significant degradation of the benthic faunal communities. Consequently, it may be argued that subtle differences in the assignation of feeding modes for some benthic species do not affect the ability of the index to differentiate between degraded and non-impacted conditions.

Throughout all these analyses, appreciable changes in indicators of benthic health were observed when the predicted carbon fluxes were in the range of 0.5 to 2 g C m^{-2} d^{-1}. This range straddles the approximate 1 g C m^{-2} d^{-1} threshold between oxic and anoxic sediments determined by the carbon flux to the sediments (Hargrave, 1994).

The analyses and statistics described above are all based upon the median predicted carbon flux values extracted from the model outputs. Because of uncertainty associated with some of the parameter settings used within the model, we configured the model using settings that covered the wide range of potential values. This resulted in a range of model outputs within an envelope of uncertainty whose extremes were considered to represent the most optimistic and pessimistic model output scenarios.

It is important to make the distinction between uncertainty and variability with respect to the model outputs. Uncertainty is introduced into the modeling through acknowledgement that we presently do not have robust and/or defensible information for some of the parameters used that are critical in determining the predicted fluxes Consequently, we tested the model using a range of values that are considered within reasonable bounds whose extremes reflected the best- and worse-case estimates. As better information becomes available for these variables, the uncertainty in model predictions will be dramatically reduced. This is exemplified by the waste feed parameter tested within the model. Values for this parameter used within the model were 0, 5 and 10% of the applied feed load. The extremes in the predicted model outputs close to the cage edge, where the greatest range in outputs was observed, were primarily driven by the highest and lowest waste feed values. If modeling had been carried out using a single value or small range (e.g. ± 1-2%) for this parameter, the range of uncertainty in model outputs would have been significantly reduced. However, we do not feel there is a single definitive value that can be used at present that could be justified for this parameter. The range in predicted carbon flux as a result of the uncertainty in the waste feed parameter seen in these results illustrates the importance of this value to the overall flux of carbon to the seabed and should be used to encourage good feed management practices at farm sites.

Variability in the model output is introduced through the model data inputs such as hydrographic data, particle and bathymetric information, and feed quantity and distribution.

Within the range of values bounded by the envelope of uncertainty and variability, it is difficult to justify the selection of one model output over another when analyzing trends in the data and correlating with observed field measurements. However, to obtain some meaningful information from the model runs, and carry out analysis, data had to be selected. We decided that the use of the median value was most appropriate as we considered that the ranges used in the variables tested, centered on reasonable estimates and were within defendable scales. Additionally, we consider that similar trends in the observed field data with model predictions would be maintained irrespective of how the model was parameterized within the envelope.

Resuspension

As described previously, the simulation of resuspension processes at this site, through the application of the RESUS module within DEPOMOD, resulted in virtually all the material that was initially deposited on the model grid being resuspended and exported from the model domain. However, observed alterations to the benthic environment around the fish farm that were indicative of organic enrichment suggest that a significant proportion of the material from the farm site is actually depositing on the sediment and causing an effect on the benthos.

We have reported on the results using no resuspension in this paper. However, we acknowledge that resuspension processes do actually occur and can significantly affect the fate of waste material from the farm sites. This leads us to suggest two possible explanations for this apparent disparity in comparisons between model results when resuspension is applied and the field data.

One explanation is that the resuspension model parameters as set by Cromey *et al.* (2002b) result in a significant over-estimation of the resuspension and transport of material away from the site. The critical erosion threshold applied in the model is at the 'low end' of the reported values discussed in the DEPOMOD Methods and Settings section above, and a higher value would most probably have resulted in a greater quantity of material remaining within the model domain. At present, it is unclear why such a wide range of values have been reported and what factors are potentially contributing to these seemingly disparate results. One possibility is, as previously noted, the field experiments undertaken by Cromey *et al.* (2002b) to calibrate and validate the resuspension model used tracer particles whose characteristics were similar to those of faecal material (particle diameter = 2 - 6 mm, settling velocity $\approx$ 3.4 cm s^{-1}, after Chen *et al.* (1999) and Cromey *et al.* (1999)). Consequently, when resuspension processes are applied to the particles on the model grid, all particles are characterized as 'faecal' material. This could potentially result in inaccuracies being introduced in the model predictions, as a significant proportion of the deposited material on the seabed may be in the form of waste feed pellets. Because of their larger size and mass, these feed pellets will take a number of days to breakdown (Stewart and Grant, 2002), intuitively have a greater resuspension threshold than faecal material, and may remain on the seabed when the model has predicted resuspension and transport. The use of a single critical resuspension value to represent all deposited waste material is probably an oversimplification and different fractions of the faecal and waste feed material will have very different resuspensive characteristics encompassing a wide range of velocities. A second potential factor is that the DEPOMOD resuspension model was validated at a reasonably quiescent site (mean flow 4.3 cm s^{-1}; maximum flow 23 cm s^{-1}; Cromey *et al.* 2002b) and extrapolation of the results to represent resuspension processes in all environmental conditions is questionable. Indeed, in a similar study to the one presented in this paper, a recent investigation on the effect of finfish farms sited in strongly tidal areas above maerl beds (loose lying coralline algae) revealed significant accumulations of farm derived material directly beneath and around the sites, whereas DEPOMOD simulations (including resuspension processes) predicted virtually no deposition (J. Hall Spencer, pers comm.).

Another explanation is that the resuspension model parameters are correct and the deposited material is indeed resuspended and transported away from the farm site. However, due to the highly reactive/labile nature of the waste material, it will cause the observed changes to the benthos during the period that it remains on the sea bed (prior to resuspension). This hypothesis regarding the rapid remineralization of labile farm waste is supported by findings of Strain and Hargrave (2005). Using dissolved oxygen uptake measurements in surface sediments containing high concentrations of feed pellets and faeces, Strain and Hargrave (2005) estimated that about half of the available carbon was oxidized within 5 days.

What is actually occurring in the sediments is likely to be a combination of the two explanations provided above. Some of the farm wastes are remaining on the sediments and some are being resuspended and transported away from the farm site. Clearly, this is a very complicated process and one that will be difficult to overcome. Further work is required to better quantify the effects of resuspension on the fate of aquaculture wastes.

Time scale of model simulations

The time scale or period utilized to simulate input of material to the seabed, as characterized by feed input data is another factor for consideration in model studies. We used feed input data from one month prior to sampling to simulate flux of waste material. Any additional input to the seabed from outwith this period will not be captured in the model outputs. Use of a longer time step may have provided greater information on inputs to the seabed. This difficulty was observed during the October 2000 modeling. Cromey and Black (2005) discuss this issue and note that the length of time step required depends on the length and complexity of the modeling study. The optimum time step used should capture all important feeding events that are relevant to the aims of the modeling exercise. However, use of any length of time step is difficult to justify. Additionally, there is little information regarding the breakdown and degradation of waste materials over time once they have deposited on the seabed.

Calculation of Carbon flux in faecal wastes

In nutritional studies, digestibility of a nutrient is determined from that fraction of the nutrient in the ingested food that is not excreted in the faeces (Forester 1999, NRC 1981 and 1993). Generally, the nutrients of interest are protein, lipids, carbohydrates and organic matter. In freshwater experiments or culture, the determination of the digestibility of dry matter is possible because the only source of dry matter is in the foodstuff. However, in marine culture of fish, dry matter digestibility is not used because the fish may take up elements from the seawater they drink (Thodesen *et al.*, 1999). The meaning or interpretation of the digestibility coefficients is important to the calculation of waste fluxes.

The interpretation of the digestibility coefficient used by Cromey *et al.* (2002a) is different from that used by Stucchi *et al.* (2005). In DEPOMOD the digestibility coefficient F_{dig} is for dry matter (Eq. 2), while in Stucchi *et al.* (2005) the apparent digestibility coefficient (ADC) is for the organic matter. The calculation of the production of faecal organic matter OM_{faeces} (Eq. 4) is calculated using an equation identical to equation 2, but the variables refer to organic matter not dry matter.

$$OM_{faeces} \quad = \quad OM_{consumed} \cdot (1\text{-}ADC) \qquad (4)$$

Organic matter digestibility coefficients for high energy extruded diets for salmon range from 81% to 90% (D. Higgs personal communications). Cromey *et al.* (2002a) used a digestibility coefficient of 85% in their comparison of DEPOMOD predicted solid fluxes to observation of solid fluxes collected with sediment traps. Other than Cromey *et al.* (2002a) we could not find any reference to digestibility coefficients for dry matter of foodstuffs determined in marine culture of salmonids.

The computation of faecal carbon fluxes using equation 2 is problematic because digestibility coefficients for dry matter are not strictly applicable in the marine culture of salmonids. Furthermore, the digestibility coefficient of 85% used by Cromey *et al.* (2002a) results in the voiding of 15% (100%-85%) of the ingested dry matter, a fraction which is substantially lower than the rough estimates of faecal dry matter production in the 20 to 25% range (Dave Higgs personal communications).

An alternate approach to determine the faecal carbon flux is to use the ADC for organic matter of salmonid foodstuffs together with the carbon concentration in the organic matter fraction of the faeces. Unfortunately, we could find no published data on the carbon content of the organic matter fraction of faecal wastes from salmonid culture. Analyses of faecal samples are presently being undertaken to determine their carbon content. Given our concern about the method used to calculate faecal carbon fluxes and lack of data on carbon concentration in faecal organic matter we have chosen several carbon concentrations in the model settings to represent the uncertainty in the calculation of this critical component of the model predictions.

Benthic Sampling

It is generally acknowledged that an individual grab sample may not provide sufficient information to adequately describe the benthic community structure as infaunal animals often exhibit clumpy or patchy distributions. One 0.1 m^2 grab is usually inadequate to represent the macrobenthic infauna at a sample site. Ideally a minimum surface area of 0.3 - 1.0 m^2 should be sampled randomly at each location, equivalent to 3 - 10 grab samples, for the purposes of examining spatial-temporal relationships, distributions of common species, or benthic communities as a whole (Swartz, 1978; Štirn, 1981; Eleftheriou and Holme, 1984).

The overall purpose of carrying out benthic sampling along a transect is generally two-fold. Firstly, it is to adequately describe the benthic community structure at each sample locations. With this information, the second purpose is to capture the signal of environmental change, if present, within the noise of natural variability.

The individual samples collected at Site A display significant differences between those taken close to the farm cages and further afield. This was fortunate, as we were able to illustrate a change in environmental conditions with distance from the farm site (fulfilling the second purpose described above), without necessarily having sufficient information to confidently describe the community structure.

This may not be the case at other locations and consideration of sample size should be a high priority when planning field surveys.

Additional Considerations.

Models have inherent errors and limitations because they simplify the processes that they attempt to explain and reproduce. Several researchers have discussed the limitations of particle tracking models used for aquaculture wastes (Silvert and Cromey, 2000, Cromey and Black, 2005 and Stucchi *et al.*, 2005). In this section, we discuss several model limitations which may be of particular importance to the site we have modeled.

The assumption of a spatially homogenous horizontal velocity field is potentially a large source of error in the model predictions. The importance of this factor will be defined on a site specific basis. The magnitude of the effect that the assumption of a horizontally uniform velocity field may have on model outputs will be determined by the topographic complexity of the site. In a hypothetical scenario where a farm is situated in an open water environment with a relatively flat bathymetry and straight shoreline, then a spatially homogenous current flow field around the model domain may accurately reflect actual conditions. However, with an increasing complexity of coastline features and varying bathymetry, the assumption that the model flow field will be representative of locations in the model grid may not be valid. Determining a threshold of complexity, beyond which the assumption of a homogenous flow field is not valid, is difficult. This approach becomes increasingly more complex when resuspension processes are considered.

At Site A, the applied flow field was measured from approximately 30 m to the north of the cage group and the recorded major axis of current flow was NW-SE. Intuitively, one would expect the current flow to the eastern end of the cage group to exhibit a major current axis in a WNW-ESE direction, and the flow to change direction at a similar angle to the coastline south of the farm group. This expected change in current flow direction is not captured in the modeling presented and the effect this would have on the model outputs is unknown.

An additional potentially complicating factor with regard to the hydrographic flow is the attenuation of the natural flow or shading effect caused by the physical structure of the farm net pens. This is also not taken into account in the model and its effect would not only be site specific but also dependant on the degree of biofouling on the nets, the size and orientation of the cages, and proximity of the current meter to the farm cages. Interestingly, the hydrographic data for Site A was collected when the farm structure was in place and the reduced current velocities measured near the surface compared to the mid-water record may be as a result of this effect. However, hydrographic data are often measured prior to net pens being put in place and this effect is not commonly observed. The obstruction of the flow by the net cages generates secondary flows around and under the net pens which may differ significantly from the primary flow field measures. Preliminary investigations into quantifying this effect suggest reductions in current velocity of up to 20 % are possible (N. Hartstein, pers comm.).

The movement of the cages caused by winds and currents may also be an important factor in this modeling study that cannot easily be accounted for in the model set-up. Cromey and Black (2005) examined the effect of cage movement on DEPOMOD model outputs. A small amount of cage movement was measured during an overnight sampling period, with the majority of positional fixes (from DGPS) within 5 m of starting position. Little correlation was found between current direction and cage movement from which they concluded that the cage groups were primarily wind-rode throughout the sampling period. They tested the effect of the observed cage movement on model outputs and calculated that it had minimal effect, showing a low sensitivity. Unfortunately, Cromey and Black (2005) do not report the depth of site or mooring design of the farm under investigation. We consider that these factors could significantly affect the degree of movement of cage structures. The potential importance of this effect in our study is because the field data used for comparisons with model outputs is very tightly spaced (~15 m), especially close to the cages. If the cage movement was only slightly greater than that measured by Cromey and Black (2005) then there is the possibility that sample stations separated by 15 m could potentially overlap on different sampling occasions. This could confound both model validation and field monitoring exercises.

CONCLUSIONS AND RECOMMENDATIONS

The application of DEPOMOD at Site A showed significant correlations and relationships between predicted carbon flux and a range of indicators of benthic impact. These trends were observed in both the benthic community structure (diversity, ITI, abundance) and sediment chemistry (sulphide concentration). However, all model outputs used to establish these relationships were calculated using model runs where resuspension processes were not simulated. The resuspension model within DEPOMOD that was validated by Cromey *et al.* (2002b) was not applied at this site, as the model predicted that virtually all the material would be exported from the model domain, whereas field observations suggested that waste deposition was occurring. The resuspension model and more specifically, the parameters, detailed by Cromey *et al.* (2002b) provide an excellent framework for the examination of these processes. However, the threshold derived from the validation exercise carried out at a relatively quiescent site may not be applicable in areas subject to higher energy hydrographic regimes, such as Site A. As the model currently does not permit the user to alter the critical resuspension velocity we could not explore the effect of changing this important setting. It should be noted that as we do not have a suitable understanding of the resuspension processes, the setting of an alternative critical resuspension velocity would be somewhat arbitrary.

The wide range of uncertainty predicted in the model outputs as a result of varying the applied values for feed wastage rate, and carbon concentration in feed and fecal material, illustrates the requirement for further research to be undertaken to obtain accurate estimates of these parameter settings. We expect that ongoing research on these parameter settings will result in a significant reductions in the uncertainty of the model predicted fluxes.

The requirement for accurate farm input data (total biomass, feeding rate and distribution, cage location and dimensions) is critical during validation exercises. This was shown in the October 2000 model outputs at Site A where the redistribution of fish on site had a significant effect on the location of predicted footprint. Once the model has achieved an acceptable level of validation, the relaxation in the information requirements for general application of the model can be explored through sensitivity analysis.

We have presented model outputs for multiple time periods at one site only. Further testing of the model at additional locations in a range of environmental conditions is necessary for an acceptable level of validation to be achieved and general trends identified.

From the work carried out to date on the DEPOMOD validation project, we make the following recommendations:

- Further model testing is required at a number of farm sites in a range of environmental conditions.

- At present, the model should be applied conservatively and the results quality audited prior to any management decisions being made using the outputs. The quality audit should include an option to not accept the modeling results, because of uncertainties or unknowns in the model parameterization or poor data input quality.
- Model outputs should always be considered in concert with other information from the site on benthic deposition.
- Research on the resuspension characteristics of waste feed and faecal particles is a high priority for further model testing and development.
- Further studies to determine waste feed rates in order to better constrain the large range of uncertainty in model outputs produced by this parameter setting.
- Similarly, analysis of the carbon concentrations of feed and faecal material, and inclusion of these data in the model parameterization will result in greater confidence in model predictions.

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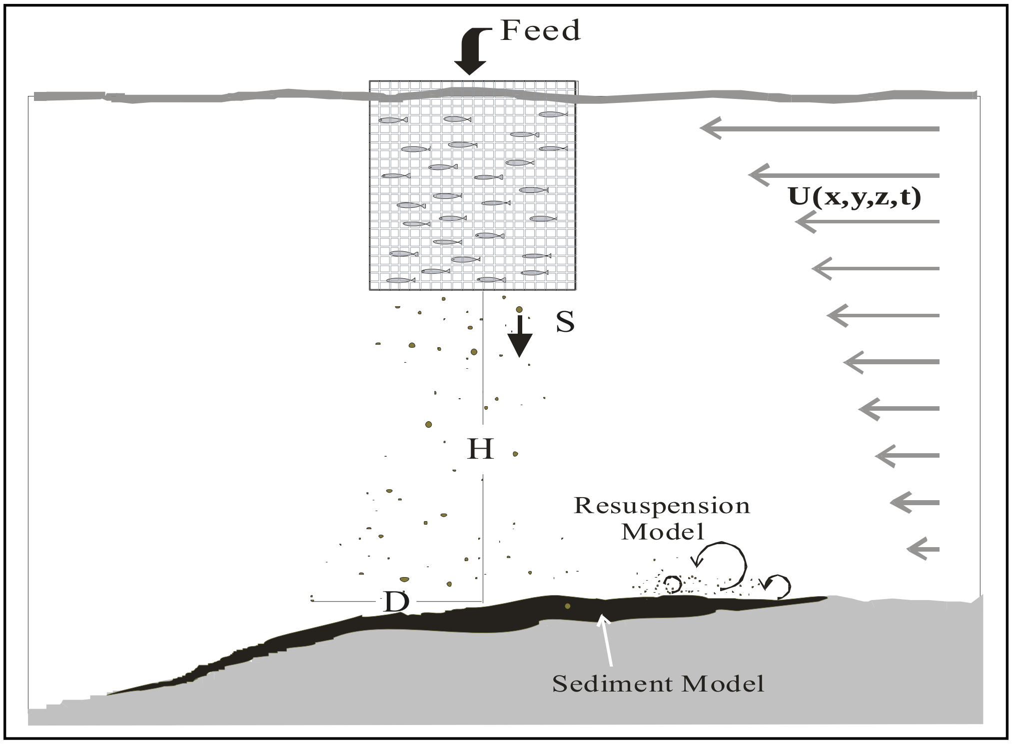


Figure 1 Schematic diagram showing the components and main processes relating to the dispersion and transport of solids wastes from open net cage finfish farms. The capital letters, S, H, and U respectively refer to the sinking rate and vertical fall distance for the particles, and the spatially and temporally varying horizontal current velocity.

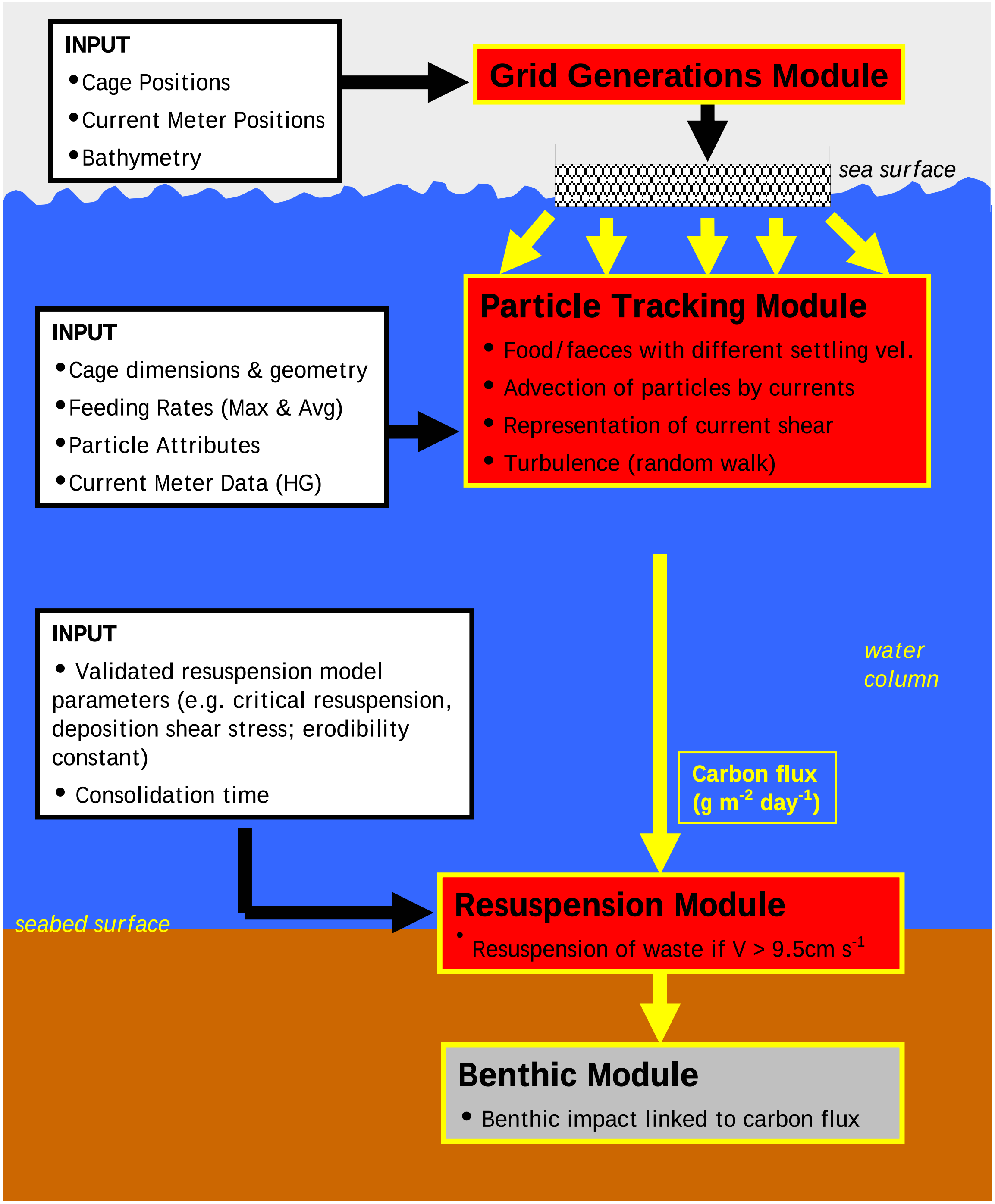


Figure 2 Schematic showing the integration of the DEPOMOD modules and associated input data used for modelling benthic impacts resulting from marine finfish farms (adapted from Cromey *et al.*,(2002a).

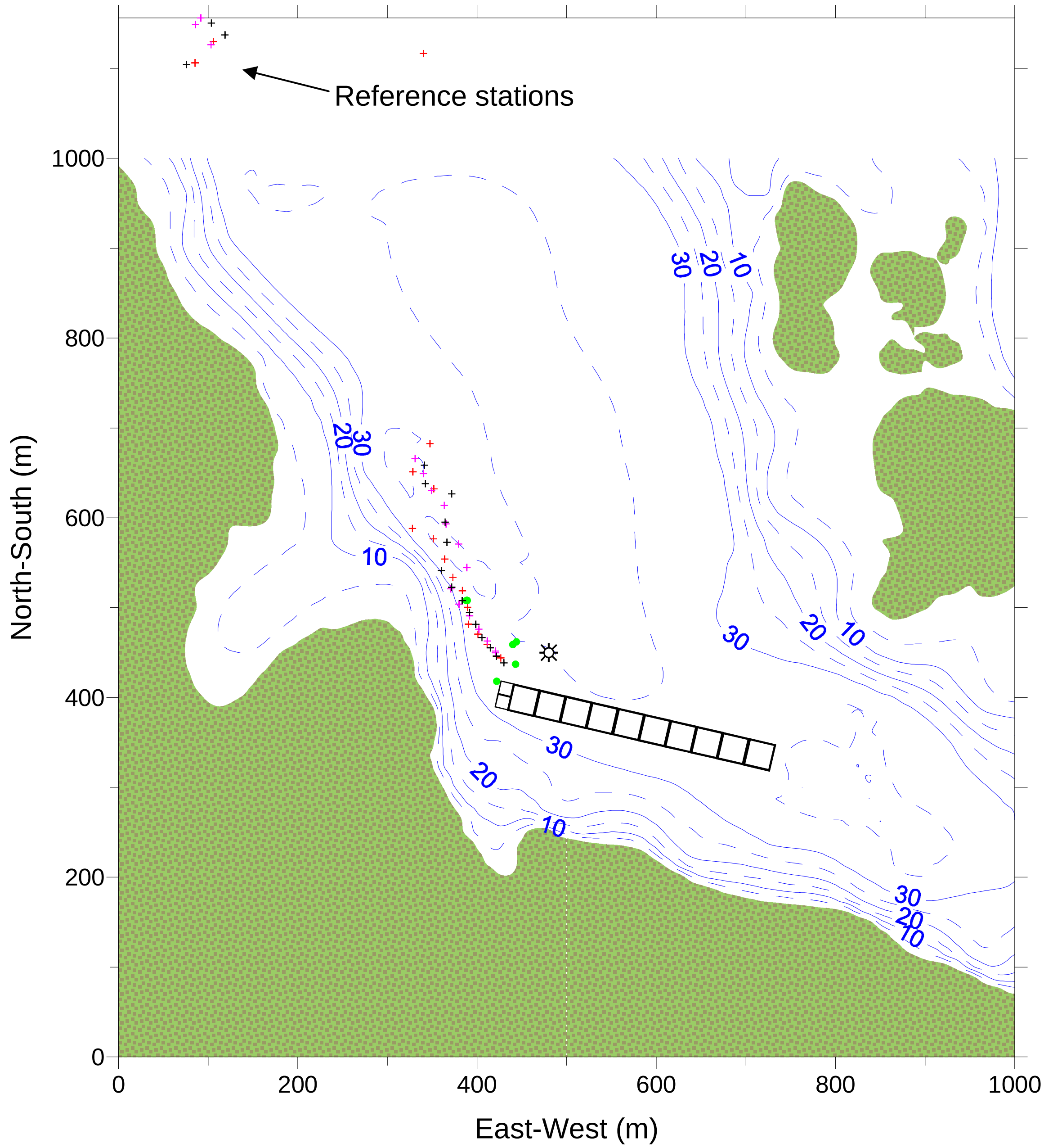


Figure 3 Map of Site A showing coastline, bathymetry, net cages, current meter mooring (☼) and location of all benthic sampling stations (+ and •).

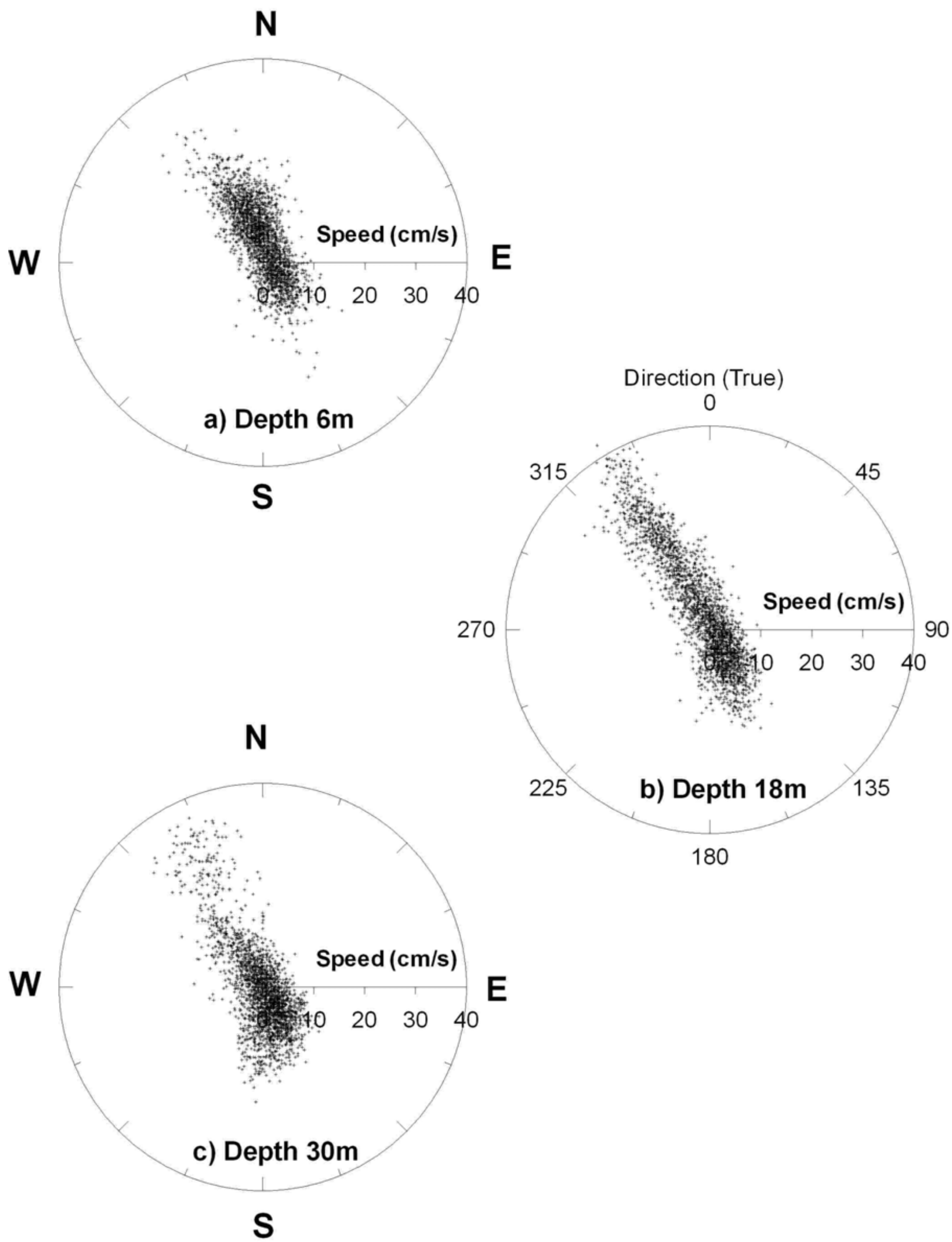


Figure 4 Polar plots of current velocity vectors from depths of a) 6 m, b) 18 m and c) 30 m. Tail of current vector is at the origin and head is the + sign.

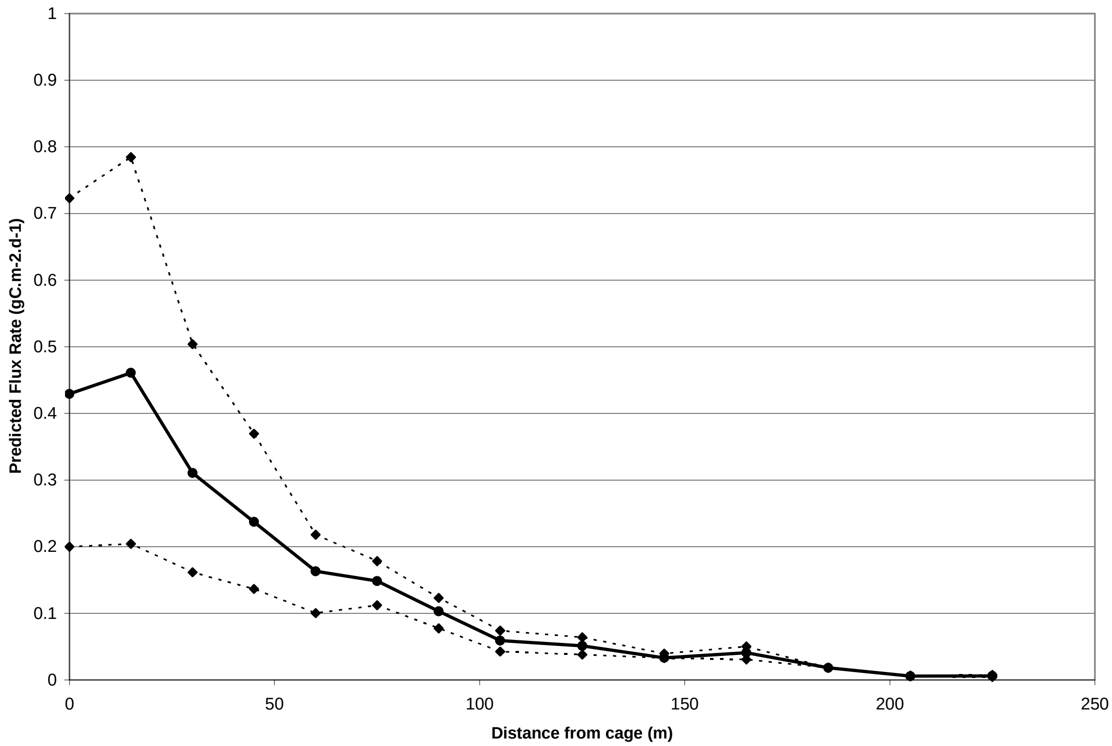


Figure 5 Graph showing predicted carbon fluxes (g C m^{-2} d^{-1}) for June 2000 at various distances from farm along transect. Solid line represents the median predicted fluxes while dashed lines show range of uncertainty.

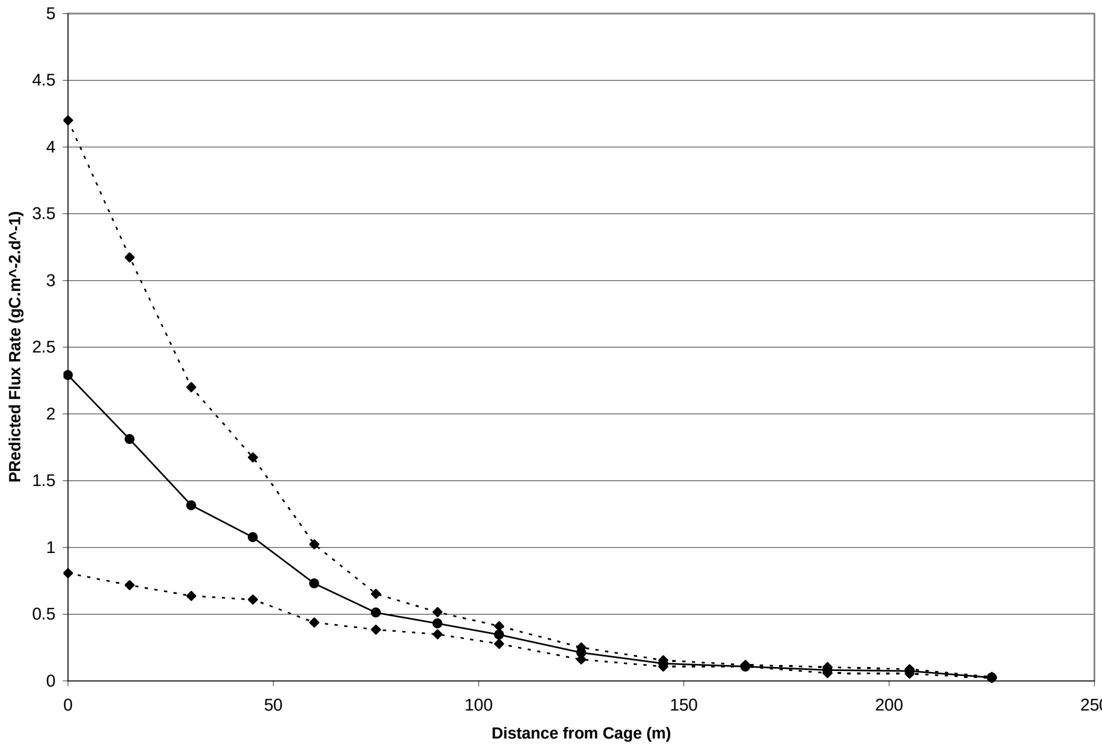
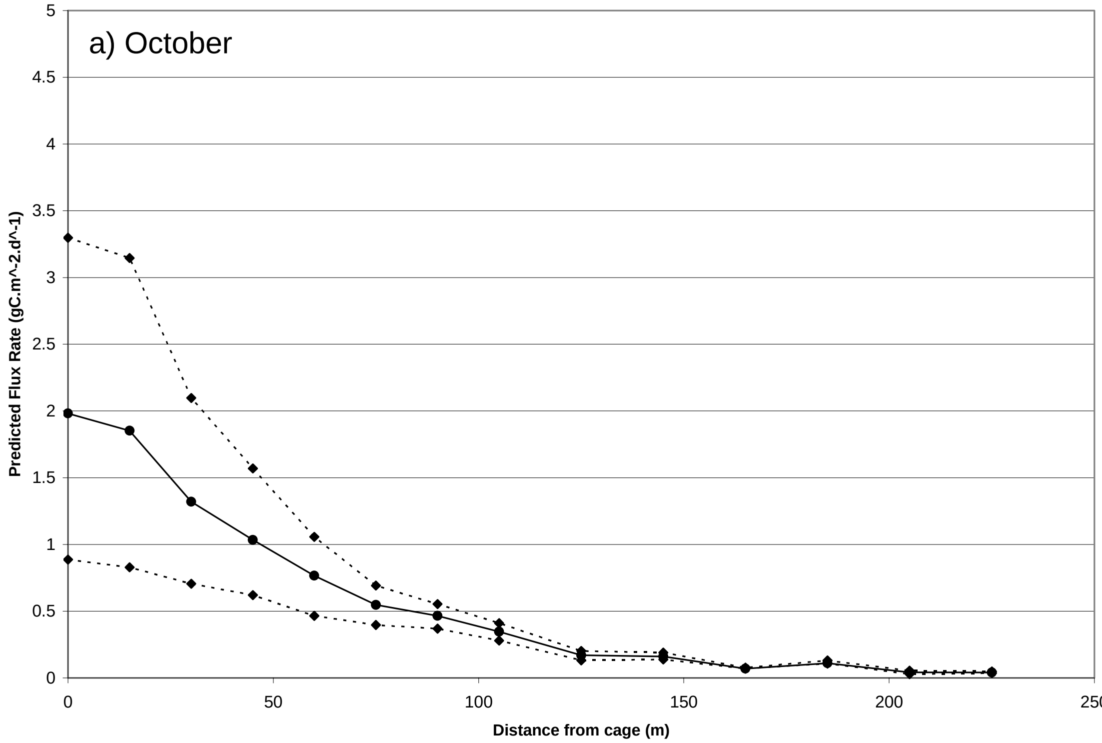


Figure 6 Graph showing predicted carbon fluxes (g C m^{-2} d^{-1}) for August 2000 at various distances from farm along transect. Solid line represents the median predicted fluxes while dashed lines show range of uncertainty.



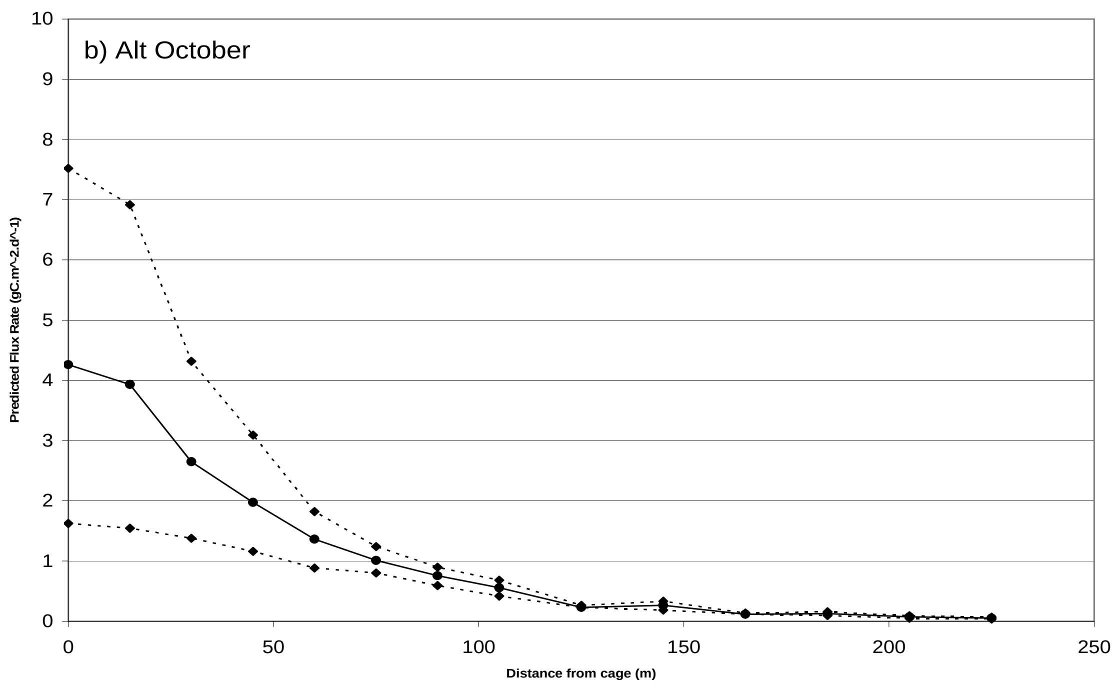
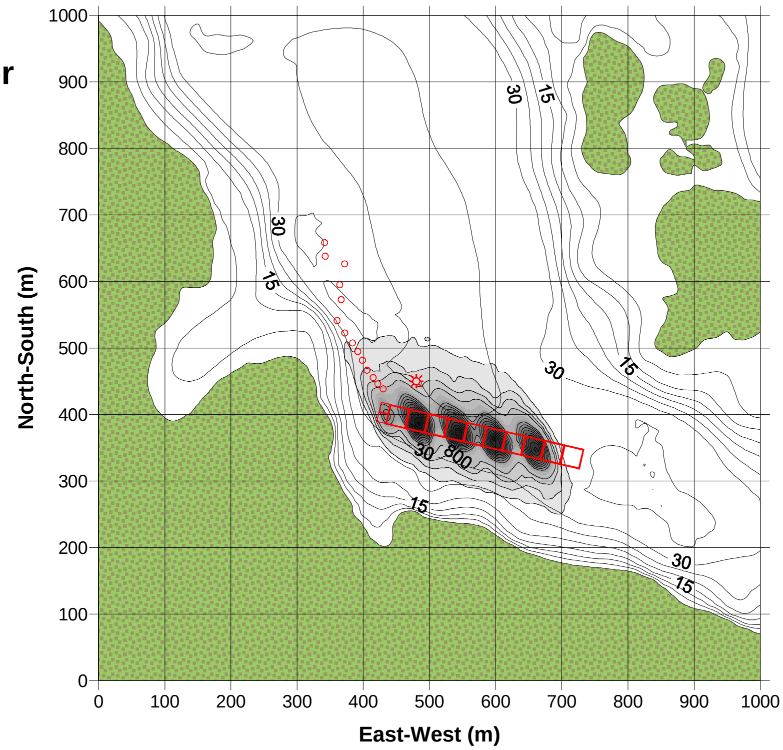


Figure 7 Graph showing predicted carbon fluxes (g C m^{-2} d^{-1}) for a) October 2000 and b) alternate October (Alt October) at various distances from farm along transect. Solid line represents the median predicted fluxes while dashed lines show range of uncertainty.

a) October



b) Alt October

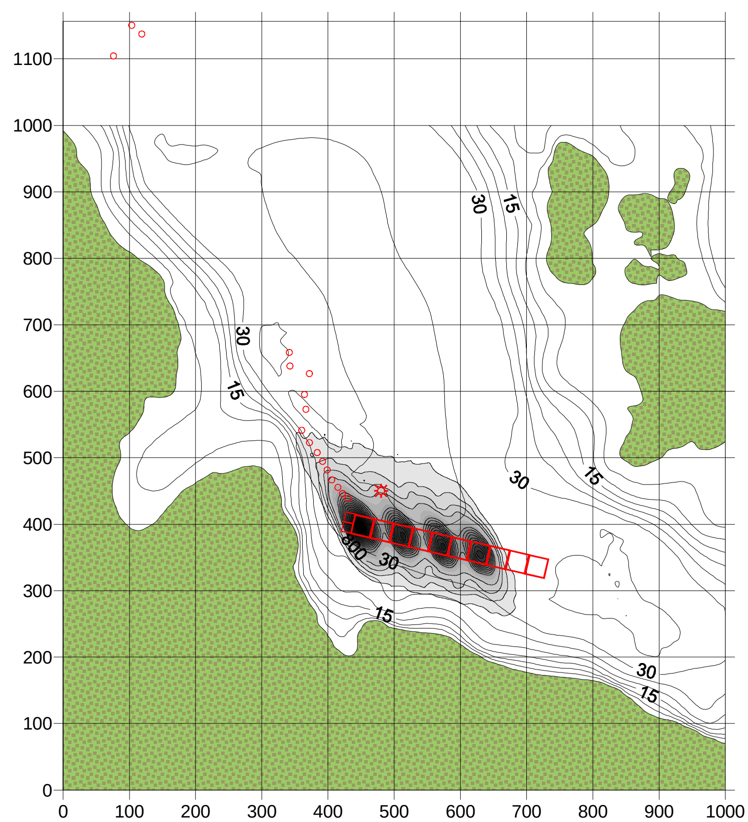
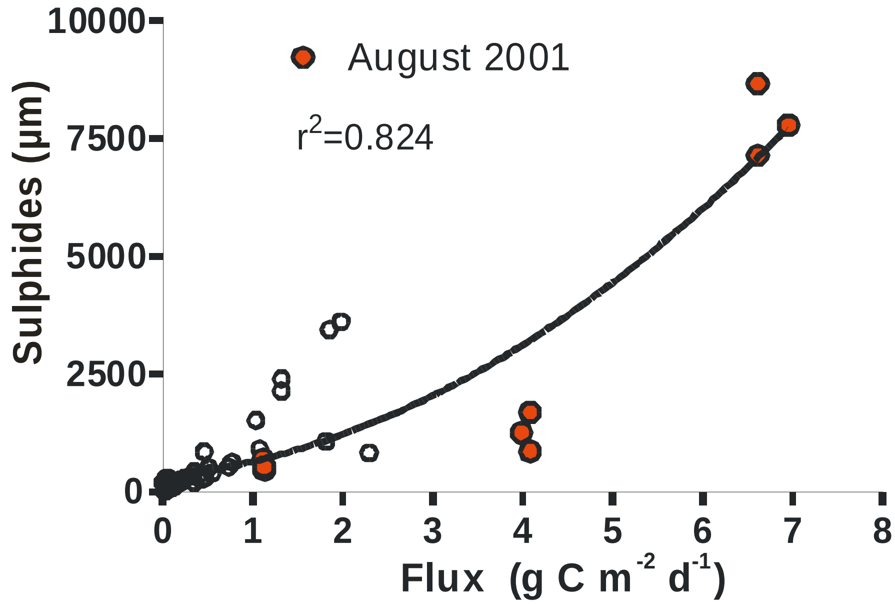


Figure 8 Map of Site A showing contours of constant carbon fluxes (g C m^{-2} d^{-1}) for a) October 2000 and b) alternate October (Alt. October) feed input data.



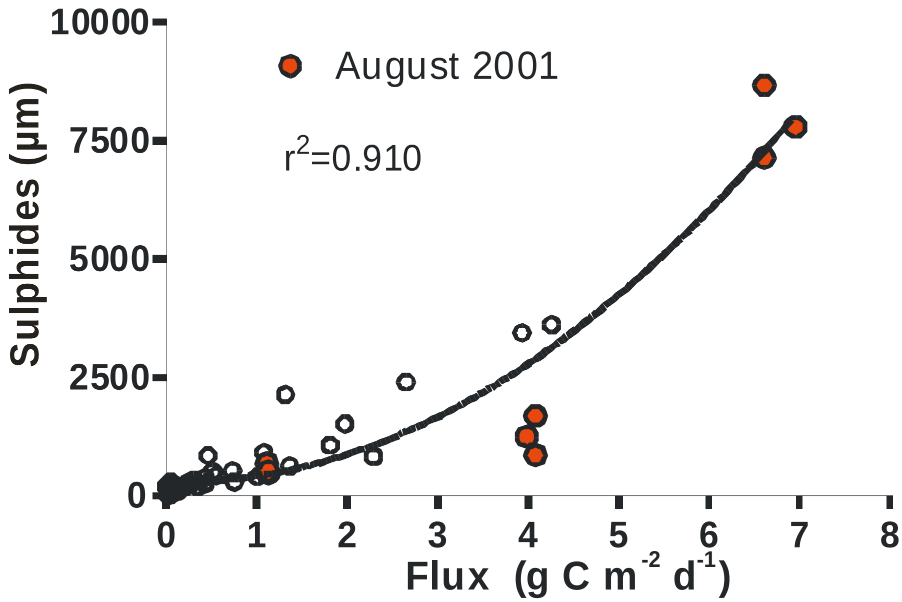


Figure 9 Graphs showing measured sulphides plotted against predicted carbon fluxes with least squares fit curve with r^2 coefficient for (top) original October predictions and (bottom) alternate October predictions (Alt October). Data from August 2001 sample period are filled red circles.

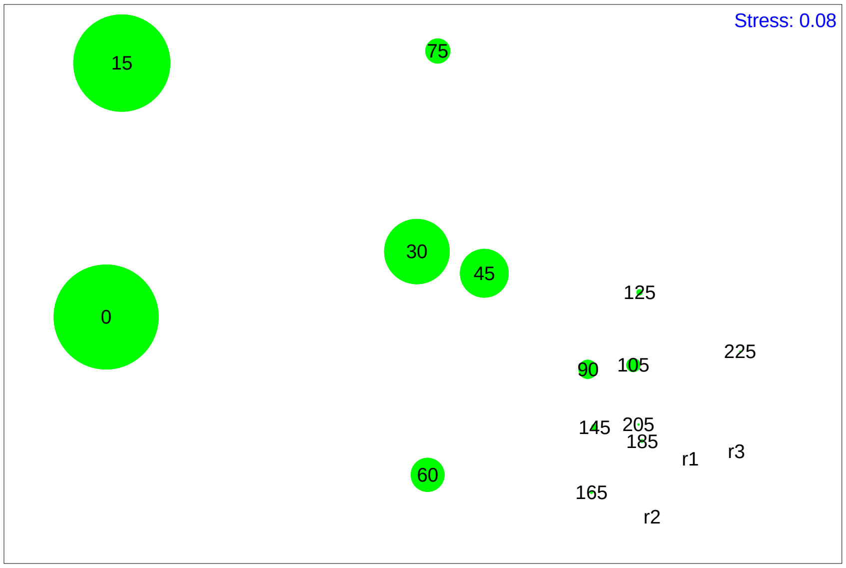
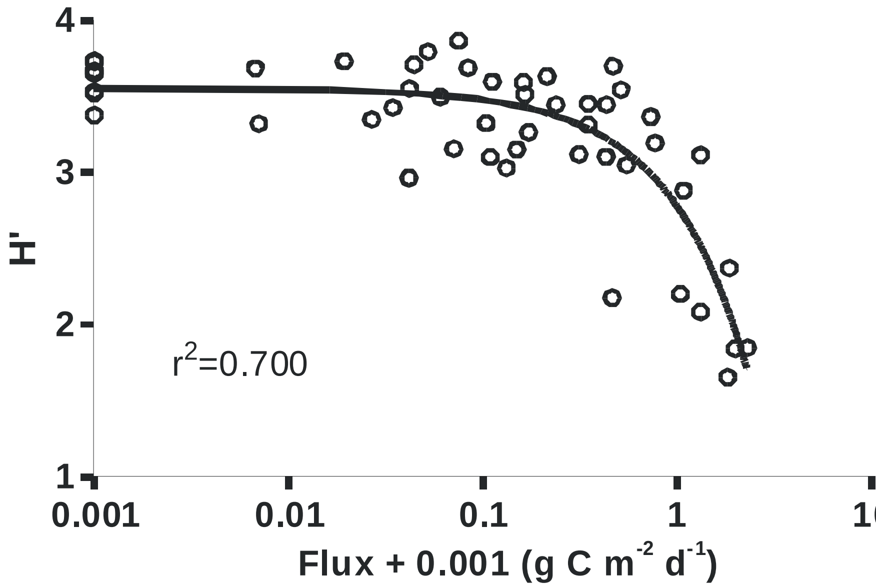


Figure 10 MDS ordination of macrobenthic infauna (using Bray-Curtis similarities on √√-transformed abundance) along the transect at Farm A in October 2000 and superimposed circles of predicted median carbon flux. Numbers denote distance (m) from the perimeter of the net-pen and r1-r3 denotes reference stations. The circle diameters are scaled to the maximum flux.



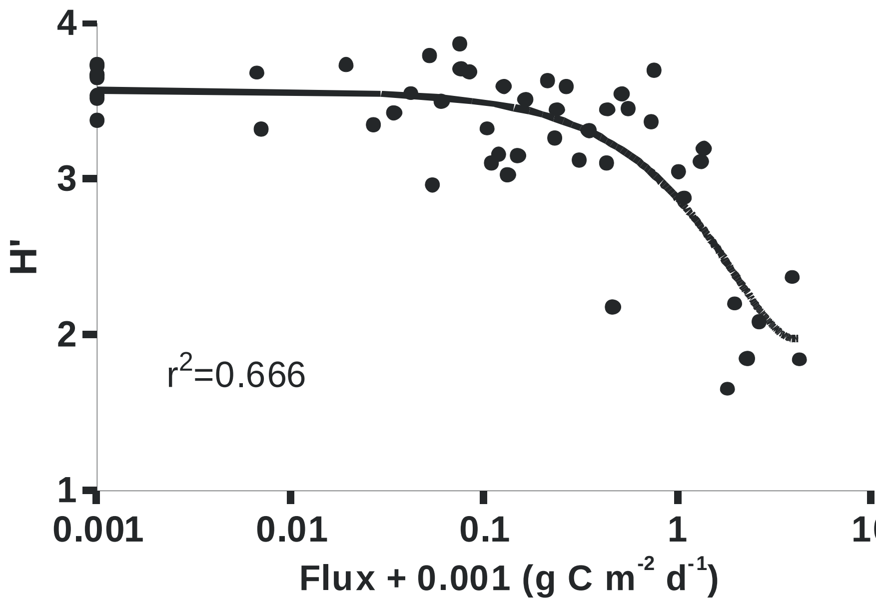
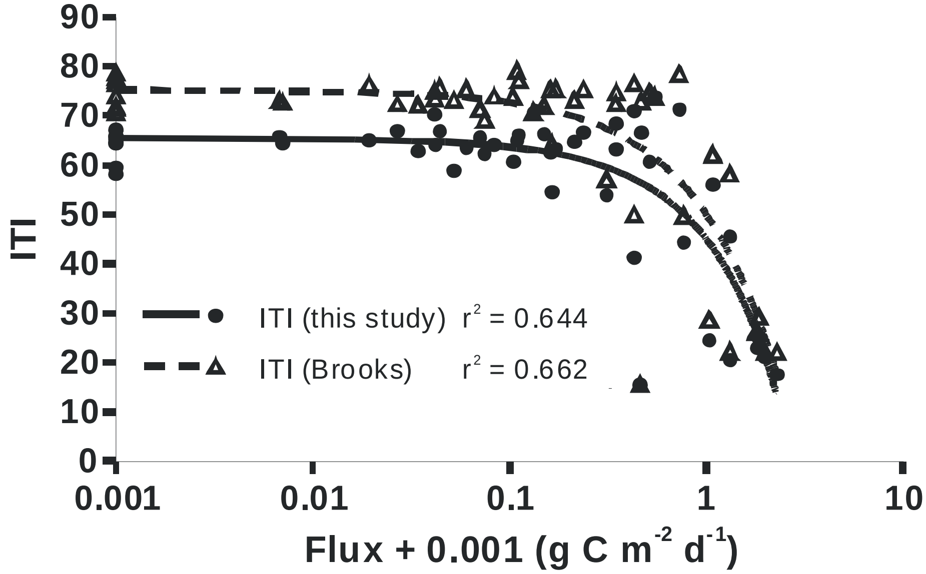


Figure 11 Graphs showing computed Shannon diversity indices (H') plotted against predicted carbon fluxes with least squares fit curve with r^2 coefficient for (top) original October predictions and (bottom) alternate October predictions (Alt October).



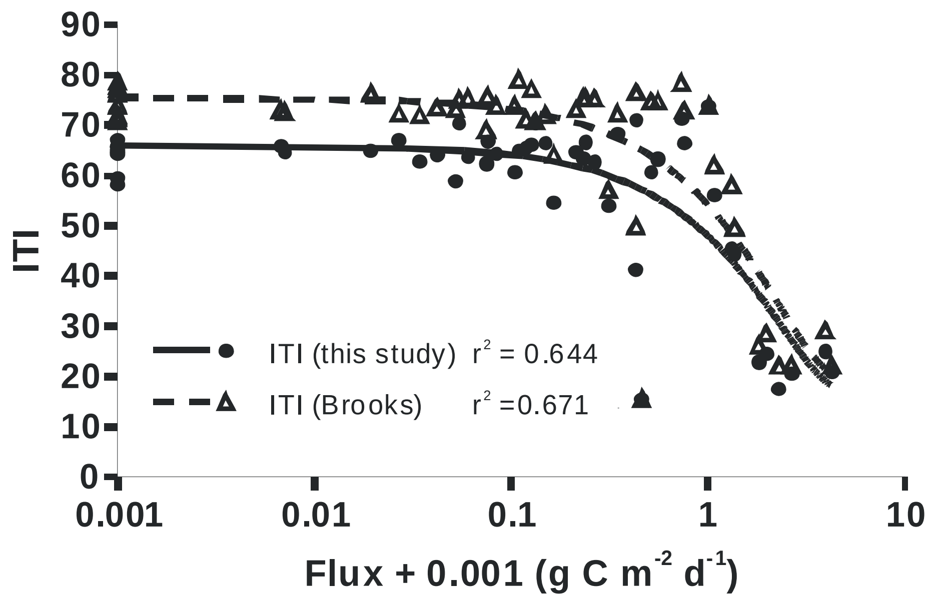


Figure 12 Graphs showing two separate calculated ITI indices (△ Brooks, 2001b and • this study) plotted against predicted carbon fluxes with least squares fit curve with r^2 coefficient for each for (top) original October predictions and (bottom) alternate October predictions (Alt October).

Table 1 Current meter statistics for the top, middle and bottom layers used in the model

	Near-surface	Mid-water	Near-bottom
Height of bin above seabed (m)	29	17	5
Mean Speed (cm s^{-1})	8.1	10.9	7.9
Maximum Velocity (cm s^{-1})	30.3	42.4	36.2
Residual Speed (cm s^{-1})	4.6	4.4	0.6
Residual Direction (°T)	354	332	206
Major current flow axis (°T)	345	335	155

Table 2 Daily feeding rate allocated to farm cages for June, August and October 2000 and August 2001.

SITE A		Feed input (kg $cage^{-1}$)							
Cage Number	Dimension (LxWxD) (m)	June-00		August-00		October-00		August-01	
		$Month^{-1}$	Day^{-1}	$Month^{-1}$	Day^{-1}	$Month^{-1}$	Day^{-1}	$Month^{-1}$	Day^{-1}
1	15x15x15	635	22.6	1455	52.0	3625	129.5	-	-
2	15x15x15	610	21.8	1396	50.0	3275	116.9	-	-
3	30x30x15	3770	134.6	7921	282.9	-	-	20695	689.8
4	30x30x15	-	-	-	-	16275	581.3	16470	549.0
5	30x30x15	3900	139.3	8458	302.0			55254	1841.8
6	30x30x15	-	-	-	-	16250	580.3	37135	1237.8
7	30x30x15	3550	126.8	6875	245.5	-	-	57118	1903.9
8	30x30x15	-	-	-	-	16150	576.8	40947	1364.9
9	30x30x15	3635	129.8	7004	250.1	-	-	40997	1366.5
10	30x30x15	-	-	-	-	15775	563.4	44360	1478.6
11	30x30x15	-	-	-	-	-	-	-	-
12	30x30x15	-	-	-	-	-	-	-	-
	Total	16100	574.9	33109	1182.5	71350	2548.2	312976	10423.3

Table 3 Field survey results for June, August and October 2000 (data from Brooks, 2001a&b)

Distance from cage group (m)	June				August				October			
	Sulphide	Abundance	Diversity	Infaunal Trophic Index	Sulphide	Abundance	Diversity	Infaunal Trophic Index	Sulphide	Abundance	Diversity	Infaunal Trophic Index
0.0	388	81	3.10	50.0	831	106	1.85	22.1	3615	105	1.84	22.1
15.0	852	87	2.17	15.5	1070	509	1.65	26.3	3445	38	2.37	29.1
30.0	335	70	3.12	57.0	2140	76	3.11	58.1	2400	165	2.08	22.2
45.0	286	82	3.44	75.3	916	113	2.88	62.1	1520	180	2.20	28.5
60.0	202	180	3.51	64.2	531	208	3.37	78.4	632	129	3.19	49.6
75.0	186	161	3.15	72.0	514	182	3.54	74.6	405	54	3.05	73.8
90.0	153	138	3.32	73.8	254	227	3.44	76.4	299	221	3.70	72.8
105.0	232	120	3.49	75.3	193	165	3.31	72.4	426	204	3.45	74.7
125.0	149	137	3.79	73.1	208	187	3.63	73.1	241	232	3.26	75.3
145.0	223	112	3.42	72.0	128	136	3.03	70.7	240	240	3.59	75.2
165.0	299	147	3.55	73.4	127	71	3.10	79.1	123	192	3.15	71.2
185.0	116	199	3.73	76.3	256	229	3.69	73.9	97	229	3.59	77.1
205.0	198	196	3.68	72.9	126	170	3.86	69.0	52	258	3.71	75.6
225.0	206	257	3.32	72.7	10	123	3.35	72.4	30	130	2.96	75.1
Ref1	77	196	3.67	71.7	7	134	3.52	76.3	58	240	3.65	71.4
Ref2	69	221	3.73	70.7	14	248	3.67	77.6	24	234	3.37	71.8
Ref3	136	231	3.66	74.0	2	259	3.72	78.6	74	339	3.53	77.0

Table 4 Sediment sulphide concentrations from field survey results for August 2001 (WALP, unpublished data)

Distance from cage group (m)	August 2001 (3 replicates)		
	1	2	3
0	7790	7150	8670
30	1690	1250	860
100	684	473	521

Table 5 Model parameter settings and input data

Input Data	June-00	August-00	October-00	August-01
Grid total size (m)	1000 x 1000			
Major grid cell resolution (m)	25 x 25			
Minor grid cell resolution (m)	10 x 10			
Bathymetric Data Source	CHS Field Sheet			
Total number of cages (number in use)	12 (6)	12 (6)	12 (6)	12 (8)
Water content (%)	10			
Digestibility (%)	90			
Food wasted as % of food fed	Varied in model between 0 – 5 – 10			
Carbon as % of Food Pellet (Dry weight)	Varied in model between 45 – 55 – 65			
Carbon as % of Faeces (Dry weight)	Varied in model between 30 – 40 – 50			
Settling velocity of Food pellet (cm s^{-1})	11.0			
Settling velocity of faecal particles (cm s^{-1})	3.2 ± 1.1 (Normal Distribution)			
Hyrdographic Data Source	ADCP			
Depth at mooring (m)	35			
Height of surface bin above seabed (m)	29			
Height of middle bin above seabed (m)	17			
Height of bottom bin above seabed (m)	5			
Time step of hydrographic data (s)	3600			
Length of current data velocity data (time steps)	720			
Mean tidal height	2.5			
Turbulence model K_x $m^2.s^{-1}$	0.1			
K_y $m^2.s^{-1}$	0.1			
K_z $m^2.s^{-1}$	0.001			
Resuspesion model ON	Consolidation time = 4 days			
Loops to run model for	2			
Resuspension model OFF	Consolidation time = N/A			
Output	Flux g C $m^{-2}yr^{-1}$			

Table 6 Model predicted carbon fluxes (gC m^{-2} d^{-1}) for June, August, October and Alternate October 2000 and August 2001 simulations at the benthic sampling station locations (* 100m sample station marker).

Distance (m)	2000				2001
	June	August	October	Alt October	August
0	0.43	2.29	1.98	4.26	6.72
15	0.46	1.81	1.85	3.93	
30	0.31	1.32	1.32	2.65	4.05
45	0.24	1.08	1.04	1.97	
60	0.16	0.73	0.77	1.36	
75	0.15	0.51	0.55	1.01	
90	0.10	0.43	0.47	0.75	
105	0.06	0.35	0.35	0.56	1.12*
125	0.05	0.21	0.17	0.23	
145	0.03	0.13	0.16	0.26	
165	0.04	0.11	0.07	0.12	
185	0.02	0.08	0.11	0.13	
205	0.01	0.07	0.04	0.07	
225	0.01	0.03	0.04	0.05	

Table 7 Daily feeding rate allocated to farm cages for alternate October 2000 model set-up.

SITE A	Feed input (kg $cage^{-1}$)	
Cage Number	October-00 (alternate)	
	$Month^{-1}$	Day^{-1}
1	3625	129.5
2	3275	116.9
3	16275	581.3
4	-	-
5	16250	580.3
6	-	-
7	16150	576.8
8	-	-
9	15775	563.4
10	-	-
11	-	-
12	-	-
Total	71350	2548.2

Table 8	Mean relative abundance (%) of 42 common species of macrobenthos at Farm A in 2000. R denotes reference stations.

Species	Distance (m) from the perimeter of the net-pen					
	0	30	60	125	205	R
POLYCHAETA						
Capitella capitata	45.45	26.36	5.61	1	0.54	0.04
Chaetozone spinosa		0.64	0.84	2.1	1.36	2.02
Chaetozone setosa		0.48		1.03	2.16	1.85
Decamastus gracillis			0.26	0.25	1.62	1.1
Eteone longa		0.2				0.21
Euclemene zonata		1.08	2.44	2.24	1.62	2.15
Eunoe depressa	0.32		1.59	2.15	1.35	0.24
Eusillis sp.				0.18	0.72	2.76
Exogone molesta		0.4	0.68	0.86	1.69	2.18
Harmothoe imbricata				0.18		0.05
Leitoscoloplos pugettensis	3.24	5.09	2.04	3.76	2.54	3.66
Lepidasthenia longicirratta	2.84	1.73	3.69	5.71	5.64	2.32
Laonice cirrata	1.86			0.49	0.17	0.62
Lumbrineris bicirrata	0.31		0.35	1.71	1.82	0.98
Lumbrineris luti	2.37	4.73	4.43	4.7	8.55	9.21
Nephtys ferruginea	0.82	0.48	1.19	1.16	1.05	2.06
Nicomache lumbricalis					0.26	2.5
Onuphis iridescens		2.63	0.52	1.3	0.82	1.06
Ophelina breviata	0.41	0.48	0.16	1.41	1.75	0.81
Phyllodoce sp.	1.14	2.69	1.71	0.39	1.44	0.25
Platynereis bicanaliculata	0.72	0.92	2.72	4.36	1.76	2.36
Polydora sp.	1.23	0.48			0.39	0.3
Prionospio sp.	1.46		0.19			0.16
Scalibregma inflatum	1.23		1.62	2.29	0.82	0.65
Schistomeringos sp.	0.73	0.92	0.53		0.17	0.05
Spio cirrifera	0.31		0.74	1.78	1.84	1.02
Spiochaetopterus costarum	0.63	1.75	0.37		0.39	0.1
Syllis elongata		0.44	1.3	0.78	1.81	1.65
Terrebellides stroemi			0.52	0.36	0.33	0.49
MOLLUSCA						
Alia gaussipata	1.35	3.64	4.36	0.78	1.31	1.8
Alvania sp.	2.54	0.88	3.02	1.98	0.69	0.6
Axinopsida serricata	0.72	1.01	4.71	2.48	2.15	0.84
Glycymeris subobsoleta	0.32		0.26	0.14	0.26	0.38
Lucinoma annulata					0.17	0.54
Mysella tumida	0.32	0.69	1.02	1.31	1.27	0.24
CRUSTACEA						
Aoroides sp.			0.19	2.11		
Ischyrocerus sp.	0.31	0.81	1.76	3.95	3.84	3.76
Megaluropsus sp.			0.9		0.69	0.85
Monoculoides sp.		4.55	0.7	0.43	2.14	1.64
Orchomene sp.	2.06	1.65	5.82	2.13	4.11	1.8
Rhepoxynius variatus			0.16	1.75	3.73	1.93
Westwoodilla caecula		0.48	2.49	1.06	2.54	1.09

Table 9 Comparison between Brooks' and our ITI calculation at different distance from the perimeter of the net-pen in different months at Farm A.

Distance	June 2000		August 2000		October 2000	
(m)	Brooks	This study	Brooks	This study	Brooks	This study
0	50.00	41.33	22.10	17.60	22.10	21.01
15	15.56	15.56	26.32	22.88	29.17	25.00
30	57.04	54.07	58.16	45.39	22.22	20.53
45	75.36	66.67	62.12	56.06	28.51	24.56
60	64.27	54.67	78.40	71.40	49.65	44.44
75	72.01	66.41	74.64	60.63	73.87	73.87
90	73.87	60.61	76.49	71.10	72.85	66.47
105	75.38	63.64	72.41	68.39	74.73	63.27
125	73.18	59.00	73.19	64.73	75.38	63.33
145	72.09	62.79	70.79	70.45	75.26	62.63
165	73.46	64.078	79.10	64.97	71.24	65.58
185	76.38	65.09	73.90	64.26	77.17	66.12
205	72.99	65.80	69.09	62.37	75.68	66.83
225	72.75	64.51	72.40	67.03	75.14	70.34
Ref 1	71.76	67.13	76.36	64.73	71.48	64.35
Ref 2	70.74	58.27	77.66	65.78	71.86	64.52
Ref 3	74.00	59.56	78.65	65.85	77.03	65.22

Table 10 Comparison of the different feeding node assignation and allocation of species to ITI group between Brooks (2001b) and present study (* The species are grouped based on Day (1967), Fauchald & Jumars (1979) and Word (1990)).

Species	Group (Brooks, 2001b)	Group* (This Study)	Comments
Amage anops	1	2	Surface detritus feeder
Euclemene reticulata	1	3	Subsurface deposit feeder
Euclemene zonata	1	3	Subsurface deposit feeder
Exogone molesta	3	2	Deposit feeder or carnivore
Kefersteinia cirrata	3	2	Carnivore
Laonice cirrata or pugettensis	1	2	Surface detritus feeder
Leitoscoloplos pugettensis	2	3	Non-selective deposit feeder
Lumbrineris bicirrata	1	2	Carnivore or scavenger
Lumbrineris luti	2	3	Carnivore or scavenger
Nicomache lumbricalis	1	3	Surface deposit feeder
Polydora sp.	1	2	Detritus feeder or partially suspension feeder
Praxillella affinis	1	3	Surface deposit feeder
Spio cirrifera	2	1	Surface detritus feeder
Terrebellides stroemi	1	2	Surface detritus feeder
Heterofoxus oculatus	1	2	Carnivore or subsurface detritvore
Ischyrocerus sp.	2	1	Suspension feeder and surface deposit feeder
Megaluropsus sp.	2	1	Suspension feeder
Monoculoides sp.	3	2	Carnivore or subsurface detritivore
Westwoodilla caecula	1	2	Surface deposit detritivore or carnivore

Table 11 Spearman correlation coefficient (r) between predicted carbon fluxes and observed abundances of total macrobenthos, polychaetes, mollusks and crustaceans at Farm A (*: $p < 0.05$; **: $p < 0.01$)

Total macrobenthos	Polychaetes	Mollusks	Crustaceans
-0.425**	-0.294*	-0.435**	-0.397**