Geographical Information Systems (GIS) as a Simple Tool to Aid Modelling of Particulate Waste Distribution at Marine Fish Cage Sites

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Received 26 May 2000 and accepted in revised form 20 November 2000

Deposition of particulate organic waste from marine fish farm cages on to sea-bed sediments can cause major changes to the benthic ecosystem. Validated spatial models are considered as the most cost-effective tools for predicting environmental impacts. An improved version of an existing predictive particulate waste distribution model for farmed Atlantic salmon (Salmo salar L.) is presented, which uses Geographic Information Systems (GIS) combined with a spreadsheet. The model presented uses existing distribution algorithms but also incorporates functions to calculate feed loading for all the cages within a pontoon independently, spreads the input load over the whole cage area and simulates post-depositional distribution of the carbon. The model uses approximate estimates of feed and faecal waste derived from dietary considerations (mass balance model) and separate, unique settling velocities for waste feed and faecal particles. The model incorporates values of current speed and direction recorded over spring and neap tides. Output from the model is in the form of a contour plot of organic carbon (g C m⁻²), showing distribution of the particulate organic carbon material as deposited on the sea-bed. During this study using hydrographic data collected from near a fish farm, the model predicted a smooth gradient of sediment carbon concentrations which decreased with distance from the cages. Model performance was validated using measured levels of sediment carbon, and showed a significant correlation between predicted and actual sediment loading (R=0·7; P<0·01). The differences between predicted and measured quantities of carbon found at some sampling stations are likely to be due to processes not included in the model, such as small differences in bathymetry, differences in bottom type which may have increased or decreased the carbon distribution through saltation, or natural variation in the sediment composition.

Keywords: distribution modelling; particulate-waste-deposition; GIS; cage-aquaculture

Introduction

The rapid expansion and development of the aquaculture industry has increased environmental concerns and questions about possible ecological impacts. Environmental managers, and especially regulators, have pointed out the necessity of minimizing environmental impacts if productivity in the new industry is to be sustainable (e.g. Scottish Executive, 1999). However, a complete reduction to zero of wastes discharged from marine fish cages is not possible for present culture systems, from either a technological or an economical point of view. There are many forms of wastes produced as a consequence of the transformation of any natural resource into a marketable product, and this is as true for marine fish aquaculture as other forms of resource utilization. However, of all the possible wastes released by marine fish farming to the environment, particulate organic waste in the form of uneaten feed and faeces are usually the most significant fraction (Beveridge, 1996). This material, which generally settles on the seabed near to the cages, provides a net input of organic carbon and nitrogen to the sediments, thus the accumulation of waste can cause major changes in the benthic community and may exceed the environment’s capacity to bioprocess this material (Findlay & Watling, 1994; GESAMP, 1996; Gowen & Bradbury, 1987; Gowen et al., 1988; Hargrave, 1994). Moreover, environmental deterioration due to high organic matter concentrations in the sediments may affect the health of farmed fishes and hence profitability (Beveridge, 1996; GESAMP, 1991b). Modelling of input and distribution of wastes and discharges is a cost-effective tool that can assist in the prediction of future impacts and aid decision-makers.

It is not possible to describe, explain or predict ecosystem behaviour without knowing how ecosystem components are distributed in time, space or
with respect to each other (‘what is where?’) and understanding the relationships and processes that explain their distribution and behaviour (‘why is it there?’). As well as requiring knowledge of spatial distribution and relationships, the ability to make reliable predictions (‘what happens if?’) often demands knowledge about temporal trends. GIS are powerful tools that can be used to organize and present spatial data in a way that allows effective environmental management planning. Spatial modelling using GIS is now under development with encouraging results (Corwin & Wagenet, 1996; Sunday-Tim, 1996). However, further investigation is required before this can be fully integrated into environmental planning, particularly for coastal zone management. The success of GIS for modelling purposes stems from their capacity for fast image generation and manipulation, the flexibility to run alternative scenarios, statistical analysis of the image and generation of sophisticated output which helps visual interpretation of results. Distribution models for organic waste from aquaculture can be used to predict possible impacts on the environment, helping environmental regulators to make informed decisions when licensing new marine fish farm developments and granting consents to discharge waste. Some models have been developed to forecast loading and distribution of particulate waste carbon from fish farms (Fox, 1990; Gowen et al., 1989; Hevia et al., 1996; Silvert, 1992, 1994; Telfer, 1995), but only DEPOMOD has undergone substantial validation (C. Cromey, pers. comm.).

This paper describes the development of GIS spatial modelling techniques with a pre-existing particulate distribution model for farmed Atlantic salmon (Salmo salar L.) (Telfer, 1995; V. Walls, pers. comm.), which is based on distribution equations developed by Gowen et al. (1989). The GIS based model was validated using field data and fish production information from a salmon farm in Scotland.

**Model development**

There are three main stages within this model; quantification of the waste material (uneaten feed and faces) using mass balance techniques, calculation of the distribution of the waste components using Gowen’s formula (Gowen et al., 1989), and calculation and generation of the final contour distribution diagrams. The first two submodels are run in a spreadsheet and the third is carried out using GIS software.

**Mass balance**

The quantities of waste released to the environment, particulate organic carbon in form of uneaten feed and faecal material, are calculated using a mass balance model (Figure 1). The expected fish production during a set period of time is multiplied by the expected food conversion ratio (FCR) for that period. In the present model, the percentage of water in the feed is assumed to be 8% (Findlay, 1994) while that of carbon in the feed is assumed to be 46% (Penczak et al., 1982). From the feed given to the fish, 90% is consumed and the remaining 10% is lost as uneaten feed (Hargrave, 1994). It is assumed here that 50% of the consumed carbon is respired (Gowen et al., 1988) and that 14% is incorporated into body tissues (Chen, 2000), although it is important to bear in mind that mass balance quantities for organic carbon flux for cage salmonid production vary from author to author (e.g., Gowen et al., 1989; Hevia et al., 1996; Silvert, 1994; Telfer, 1995). Carbon in faecal material is calculated as the difference between carbon consumed and carbon used for respiration and growth.

**Distribution equation**

The horizontal distribution of a particle (X and Y components for each cage of the farm) is calculated using the equations of Gowen et al. (1989) (see Equations 1 and 2). The depth under the cage (d), the mean current speed (V), current direction (θ), settling velocity (u) and position of each cage (x, y) are site specific measured quantities. For modelling purposes, the farm is located in the middle of a 500 by 500 cell array, each cell representing 1 m².

\[
X = \frac{d \cdot V \sin \theta}{u} + x \tag{1}
\]

\[
Y = \frac{d \cdot V \cos \theta}{u} + y \tag{2}
\]

Because wastes fall more or less vertically through the cage (Inoue, 1972), the source of distribution was assumed to be from the depth corresponding to the bottom of cage. The same equation is used to calculate the distribution of both uneaten feed and faces, but different settling velocities are assigned to each; 0.128 m s⁻¹ and 0.04 m s⁻¹ respectively (Warrr-Hansen, 1982; Chen et al., 1999).

**Geographical Information Systems (IDRISI32)**

The carbon deposition co-ordinates calculated with Gowen’s formula (Gowen et al., 1989) and their
associated carbon values are exported to IDRISI32 GIS software. Interpolation between values is then undertaken by the GIS program to generate a complete surface. This technique is a particular strength of using GIS for modelling purposes, in that small or sparse data sets can be used to produce a complete map. However, the nature of the interpolation process means that the carbon quantities are over-estimated which requires correction later in the model. The reason for sparse number of data points used for modelling, and the need to interpolate to smooth the data, is that the current readings, speed \((V)\) and direction \((\theta)\), are taken at regular intervals over a defined time period (e.g. 1 h). During that period \(t_0\) all the carbon produced is assumed to settle in one location \((X_0, Y_0)\) and during the subsequent hour \(t_1\) in another location \((X_1, Y_1)\). In reality, the process of moving from one location to another is done gradually and the carbon is dispersed over all locations, not only at the initial and end locations, as illustrated in Figure 2(a).

Most models assume a single point as the source of carbon output from a cage (feed and faeces), usually the cage centre. To eliminate this assumption,
predicted carbon values allocated on the seabed are spread over an area equal to the cage area by applying a filter within IDRISI [Figure 2(b)].

Variations in initial distribution of waste and post-deposition changes in carbon are considered by using a second filtering technique within IDRISI, which redistributes the amount of carbon from each cell into the eight surrounding cells by a predetermined percentage, which differs between the relatively dense feed and lighter faeces. Each cell represents 1 m², hence the final area affected from each of the initial cells is 9 m² [Figure 3(a–c)]. The filter structure is based upon theoretical work (Chen, 2000). Pellets and faecal matter travel along the sediment by saltation (rolling, sliding or hopping) when current speeds attain a critical value, defined here as the velocity that causes 50% of the pellets to move. Distance travelled and number of re-suspended particles are site specific. Although this final GIS filter has minimal impact on modelled carbon distribution, as shown by sensitivity analysis with and without the filter, it is included to give a more realistic picture of the processes involved in the final carbon distribution.

Finally, it is necessary to correct the carbon quantities in the resultant image, which are over-estimated due to the interpolation process. Interpolation generates additional data between a set of known values. However, it does not reduce the original carbon concentrations to compensate for this, which means that the model assumes that there is considerably more carbon entering the sediment than there really is. The correction is achieved by multiplying the resultant output by a correction factor (CF), which is calculated by dividing the total predicted waste carbon from the mass balance (feed and faeces) by the total carbon in the resultant image (Equation 3).

\[
CF = \frac{\text{Total predicted waste carbon (kg)}}{\text{Waste carbon in the image (kg)}}
\]  

### Model validation

#### Introduction

A mathematical model is an approximation of the real world and thus its predictions are inherently uncertain. The uncertainty results from lack of knowledge of natural processes, lack of quantity and quality of data, and also from the assumptions within the model. All are potential sources of error, which decrease the accuracy of prediction. Models, therefore, require validation in order to establish agreement between predictions and observations. Model validation is accepted as being achieved when the model output compares favourably with ‘real’ environmental data (e.g. measured carbon, biological data) from an independent location (GESAMP, 1991).

#### Sample site

A set of sediment samples and tidal current data were obtained from an Atlantic salmon (Salmo salar L.) sea cage site within a sheltered sea loch on the west coast of Scotland. There were no fish farming activities at the site prior to the validation study. Fish were reared according to current Scottish salmon farming practices and using standard commercial pelleted feeds. The validation was based on a four months operational period with a maximum standing biomass of 172.5 t fish and an average FCR over the production period of 1.2. Actual monthly biomass values were used in the model validation.

#### Field measurements

Sediment samples were taken twice approximately 4 months after production began by the farm operator at the stations shown in Figure 4. Samples included those from two reference stations at a distance of 5%
>500 m from the farm but in the same depth and a similar sediment type. Carbon content of sediment grab samples were determined by an ignition technique (Holme & McIntyre, 1971). Current velocity and direction were measured adjacent to the cage in mid-water, as surface currents are attenuated by the cage structure and the bottom currents are of greater importance in re-suspension and the final allocation of the wastes than in the initial horizontal distribution. Current velocity and direction were measured using Valeport BFM308 current meters over a 12-h neap and 12-h spring tide. Recordings were made every 10 min over a 1-min averaging period.

**Statistical comparison method**

To assess how predicted carbon compared with actual values measured in the field, a Pearson correlation test (two-tailed) was performed between output from the model and measured levels of carbon for all sampling stations along the four transects.

**Results**

Water current data, speed and direction at neap and spring tides recorded at the farm location are shown in Figure 5(a and b). Both, neap and spring tide currents flowed predominantly in north-east and south-west directions, with the neap tide currents being the faster. Table 1 shows the minimum, maximum and mean current speeds for neap and spring current readings.

The final output from the model is a contour map showing the distribution of particulate organic carbon.
deposited on the seabed. **Figure 6** shows the carbon distribution predicted by the model to which the measured background carbon value has been added (1.353 kg C m\(^{-2}\)). The farm comprised 18 cages located in two rows of nine. The depth under the cages was 15 m. From the mass balance calculations 35.4 t of carbon were wasted, 10.3 and 25.0 t due to uneaten feed and faeces respectively. The carbon was distributed mainly in NE and SW directions. The highest concentrations were located under the cage, reaching values of 12 kg C m\(^{2}\). The concentration of carbon decreased as the distance from the cage increased.

The Pearson correlation test (R = 0.7; \(P < 0.01\)) indicates a strong relationship between modelled and real carbon values. On average, model predictions were accurate to ±18%, 0% to 40% differences being observed at some stations. Modelled and actual carbon values from the field survey along the two transects (A–C and B–C in **Figure 4**) are plotted in **Figure 7**.

**Discussion**

The results indicate that the modelled outputs of distribution concur with sediment measurements (**Figure 7**). The model predicted a smooth gradient of sediment carbon concentrations which decrease with distance from the cages, as did the benthic enrichment model of Hevia *et al.* (1996) for salmon cages in Norway. The differences between predicted and measured quantities of carbon at some sampling stations may have been due to processes which were not included in the model, such as small differences in bathymetry, differences in bottom type which may have increased or decreased the carbon distribution through saltation, or unpredictable natural variations in sediment composition. Predictions from the model reflect a smoother, idealized gradient that allows only for input from fish wastes and a general background carbon level. Although, model predictions generally agree with field data the model would benefit from further development and validation under different hydrodynamic and fish farm production scenarios.

Models are based upon assumptions based on both a knowledge of the process involved and personal judgement of their importance, their relation to each other and with the environment. Because assumptions carry an inherent risk of inaccuracy, it is desirable both to reduce their use to a minimum and to identify their impact on overall predictions in order to understand and interpret model outputs. Possible sources of
inaccuracy in the present model are the lack of consideration of environmental capacity and the assumption that all carbon wastes from the cages settle on the seabed, where none is degraded or consumed by invertebrates and wild fish. Although it is likely that this will vary with location, the present GIS model considers post-deposition movement of carbon, and has eliminated some assumptions from previously used modelling techniques. Furthermore, few data are needed to run the model, so reducing costs of field surveys.

Although the model has only been developed for particulate carbon, the distribution of any other solid material from a fish farm, such as N, P or in-feed chemotherapeutants, can also be modelled, providing accurate data on inputs that are available.

Potential applications for the model are within Environmental Impact Assessment (EIA), design of monitoring programmes, site selection, farm management and rapid generation of ‘what if?’ scenarios. An environmental management plan should include an EIA and this often requires the use of predictive models to quantify significant potential impacts and design a monitoring programme (GESAMP, 1996). Numerical models, such as described here, have the potential to generate quantitative predictions and are

**Figure 7.** Comparison between predicted carbon quantities corrected for background value and actual field measurements along the longest transect A to C (a) and transect B to D (b). ● Sediment △ Model.
therefore a useful tool in quantifying impacts of cage aquaculture wastes. Moreover, Jørgensen (1991) suggested that the use of validated models to predict environmental impacts was the most cost-effective approach. Models can also be used to allocate development areas (zonation) or for comparison of outputs at different sites by modelling the waste deposition pattern for any given production level. Similarly, the maximum desirable production at a site can be established. However, any modelling-based predictions must be tested by monitoring (GESAMP, 1991a; Silvert, 1992; Sowles et al., 1994).

Environmental models remain insufficiently developed to be used as the sole tool to predict impacts in the environment. They are complementary to field surveys and risk assessment. It is important to be aware of the influence of site characteristics on the interpretation of model outputs. Natural processes are site specific and carbon inputs that may cause problems at one site may not pose a problem at another (Findlay & Watling, 1994).

Acknowledgements

We thank the Scottish Environment Protection Agency for the use of their field data in the model validation. This work was carried out as part of a training project financed by the EC under the FAIR program (GT 973516). TCT and MCMB acknowledge the assistance of the CRAFT FAIR project CT-98-9201. The authors would like to acknowledge V. Walls for her contribution to this research.

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