Scottish executive locational guidelines for fish farming: predicted levels of nutrient enhancement and benthic impact

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SCOTTISH EXECUTIVE LOCATIONAL GUIDELINES FOR FISH FARMING: PREDICTED LEVELS OF NUTRIENT ENHANCEMENT AND BENTHIC IMPACT

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ABSTRACT

The Scottish Executive document 'Locational Guidelines for the Authorisation of Marine Fish Farms in Scottish Waters' provides guidance to the aquaculture industry and regulatory bodies on the environmental suitability of coastal areas for fish farming. This guidance is based on advice from Scottish Natural Heritage and Fisheries Research Services (FRS), relating to the relative environmental sensitivity of local areas supporting aquaculture. The purpose of this document is to describe the predictive modelling approach used to inform the FRS component of this advice, through estimation of the levels of nutrient enhancement and benthic impact arising from current marine finfish cultivation in sea lochs (and similar semi-enclosed water bodies).

A simple box model is used to predict the level of enhancement of soluble nutrient nitrogen from fish farming sources, treating nitrogen as a conservative substance. This model is a function of the flushing rate of a sea loch, the total consented biomass of all the finfish farms in the loch and the nitrogen source rate. A mass balance model was used to estimate the rate of release of nitrogen at 48.2 kgN per tonne of salmon produced per year. These data are used to calculate an equilibrium concentration enhancement (ECE) for nitrogen, expressed in µmol l^{-1}. Non-salmonid species are accounted for in this model, by the application of appropriate 'species factors' to correct for the different rates of soluble nitrogen release from the farming of such species.

Benthic impact is estimated using a modified 'Gowen' model to predict the area of seabed impacted by the deposition of organic matter in the form of solid waste from finfish farms. Using the modified technique described, the along- and across-loch dispersion of solid waste is estimated for each farm in a sea loch. These distances, together with an estimate of the surface area of the farms are used to predict an elliptical area of seabed impacted by organic carbon deposition. An appropriate distribution of particle settling velocities and associated quantities of excreted carbon are applied in order to estimate areas of seabed impacted by different levels of organic carbon deposition within the impacted area. The results of this model are used to estimate the total percentage area of the seabed of a loch impacted by a level of enhanced organic carbon deposition greater than 0.70 kgC m^{-2} y^{-1}. Above this critical value, it has been shown that the infaunal diversity of sediments is reduced, and the seabed can be considered 'degraded'.

The models described here predict the relative levels of nutrient enhancement and percentage areas of seabed degraded by organic carbon deposition for 111 sea lochs. The results of both models are scaled from 0 – 5, and the two scaled values are added together to provide a single combined index. On the basis of this combined index, areas are designated as Category 1, 2 or 3, where Category 1 areas are considered to be the most environmentally sensitive to further fish farming development due to high predicted levels of nutrient enhancement and / or benthic impact.

BACKGROUND

In October 1999, the Scottish Executive published 'Locational Guidelines for the Authorisation of Marine Fish Farms in Scottish Waters'. These guidelines were issued following the announcement in 1997 that authorisation to permit marine fish farm developments would be transferred from the Crown Estate (CEC) to Scottish local authorities. The purpose of the Locational Guidelines was to facilitate this transfer by providing guidance to local authorities, other regulatory bodies and the industry on the future location of marine fish farms. This guidance designated Scottish coastal waters into three Categories, with varying levels of restriction on further fish farming development. Definitions of these Categories and the levels of restriction they impose on further fish farming development are given in the 1999 Locational Guidelines.

The criteria used by the Scottish Executive in designating these areas were based partly on advice from Scottish Natural Heritage (SNH) relating to the location of natural heritage sensitivities and National Scenic Areas. In addition, FRS provided advice on the relative environmental sensitivity of semi-enclosed coastal waters (e.g. sea lochs) as a result of discharges from existing fish farming activity in those areas. This sensitivity was estimated using predictive modelling of the level of soluble nutrient enhancement and benthic impact in semi-enclosed waters that supported fish farming activities.

The Scottish Executive Environment and Rural Affairs Department, Freshwater Fisheries, Aquaculture and Marine Environment division (SEERAD FFAME) requested further advice on the relative environmental sensitivity of sea lochs supporting aquaculture, to inform an update of the Locational Guidelines in 2002. The purpose of this document is to outline the predictive modelling approaches applied by FRS to inform this revision of the Locational Guidelines. Detailed here are the simple computer-based models used to predict nutrient enhancement and benthic impact, together with a description of how the output from these models was used to assign sea lochs (voes etc.) to the different Categories in the revised Locational Guidelines. Details of the data on which these models are based and their assumptions are also given.

THE COMPUTER MODELS

Two computer models are used to compare the environmental impacts of fish farms in different lochs. Both are simple box models and make several simplifying assumptions about the topography, hydrodynamics and water exchange of the sea loch systems. The first model predicts the potential degree of nutrient enrichment in the water column due to the emissions from sea cages. The second model uses the simple Gowen relationship to estimate the area of seabed impacted by deposition of particulate carbon from fish farms. Both models are used to assess the cumulative impact of all consented finfish farms in each sea loch system. To do this, the models require information on each loch, in particular the number of fish farms, the location of each site and the consented maximum biomass of fish present. The models are run routinely for all 111 sea lochs presently included in the FRS sea loch/fish farm environmental database, thereby providing a comparative study of the relative degree of impact in each sea loch system.

Both nutrient and sediment impact models will be described in this report. First, however, the assumptions underpinning both models are explained.

Hydrographic Assumptions

The models represent each sea loch system as a rectangular basin. The physical characteristics of the basin, i.e. length, surface area and volume, have the same values as the real system. The depth of the basin, given by dividing the water volume by the surface area, therefore matches the mean depth of the real system, but does not incorporate details of the natural geomorphology (e.g. sills and deep basins).

The second major simplification concerns the tidal exchange and flushing mechanism of the loch basin. The predominant exchange mechanism is assumed to be the semi-diurnal tide. The flushing time of the loch basin can be calculated by assuming that the water volume is replaced by the volume of water entering and leaving on each tide (the “tidal prism”), giving (see Edwards and Sharples, 1986):

$$T_F = \frac{0.52V}{0.7A.R} \text{ days}$$

where V is the volume of the loch basin (m^3), A is the surface area of the loch (m^2) and R is the tidal range (m). The factor 0.52 is the number of days per tidal cycle (1 tidal cycle = 12.4 hours = 0.52 days), and the factor 0.7 approximates the mean tidal range from the spring tidal range, R (Edwards and Sharples, 1986).

The tidal prism method of calculating flushing times is known to overestimate the exchange of water and therefore underpredict the flushing time (see, for example, Edwards and Sharples, 1986; Gillibrand and Turrell, 1997a; Gillibrand 2001). The exchange rate of sea lochs can also be affected by wind strength and direction, and fluctuations in river flow (Gillibrand, in press). However, these variations are difficult to predict and vary from loch to loch. Tidal exchange is a steady and persistent process, not subject to meteorological fluctuations, and therefore forms the core exchange mechanism of these systems. The tidal prism method, therefore, while not complete, forms the best available method for estimating the flushing of sea lochs.

The Sea Loch/Fish Farm Database

The information contained within the FRS sealoch database acts as the source data for the nutrient, carbon loading and dispersion models.

The database comprises two related worksheets, one containing the hydrographic information on over 100 sea lochs, the other holding information on individual sites within each sea loch, totalling more than 300 finfish sites. Sites in open water locations are not included in the database. The hydrographic information, including the length, area, volume, tidal range and flushing time of each system, is taken either from the Edwards and Sharples catalogue (Edwards and Sharples, 1986) or from digitised bathymetric maps of sea lochs not covered in the catalogue. The number of farms in the loch and the total consented biomass for the loch is also included in this dataset.

The information on individual farm sites within each sea loch includes the maximum biomass as consented by the Scottish Environment Protection Agency (SEPA), the distance of the site from the mouth of the loch, lateral position across the loch, month and year of most recent consent, a database number and the name of the producer or operator. The fish species contained on site is also noted so that differences in waste discharge can be incorporated in to the models (as discussed later).

The sea loch/fish farm database held at FRS is based on the database of consented biomass maintained by SEPA and is updated on a monthly basis from the biomass consent updates sent to FRS by SEPA. The current (July 2002) total consented biomasses for the 111 sea lochs in the database, are summarised in Table 1 below:

Table 1. Distribution of total consented biomass of finfish in 111 Scottish sea lochs, as of July 2002.

Total consented biomass (Tonnes)	**Number of Scottish sea lochs**
> 6000	4
4000 – 5999	6
2000 – 3999	27
1000 – 1999	25
500 – 999	21
< 500	28
Total	**111**

NUTRIENT (NITROGEN) ENHANCEMENT MODEL

The first model used for the categorisation estimates the enhancement of dissolved nitrogen *above background levels* that would occur in the sea lochs due to finfish sites if nitrogen were conserved in the environment and was only removed from lochs by tidal flushing. The model has previously been described in the scientific literature as assessing the equilibrium concentration enhancement (ECE) of nitrogen (Gillibrand and Turrell, 1997a). Similar methods have been described elsewhere in the scientific literature (e.g. Silvert, 1992; Silvert, 1994; Silvert and Sowles, 1996).

The model extracts the core physical parameters for every sea loch from the database. These include the basin volume V (m^3), flushing time T_F (days) and the number of consented farms. From the related site database, the model reads in the consented biomass M_S (tonnes) for each site and the species of fish produced.

Calculation of Flushing Rate

The nutrient enhancement is strongly dependent on the flushing rate, Q ($m^3\ y^{-1}$), of the loch, which is given by

$$Q = 365.V / T_F$$

where the factor 365 converts the units from $m^3\ d^{-1}$ to $m^3\ y^{-1}$. The flushing rate, then, is the total quantity of water that is exchanged over a year.

Calculation of the Nitrogen Source

The nutrient considered by the model is nitrogen, mainly in the form of dissolved ammonia but also accounting for nitrogen emitted as particulate waste and re-dissolving into the water column from the seabed. This is equivalent to considering that seabed conditions at farms are generally not deteriorating with time. The combined source of nitrogen from dissolved and particulate wastes is S = 48.2 kgN per tonne of salmon produced. This value was derived from a mass balance model, used to estimate the release of dissolved and particulate nitrogenous waste from cultivated salmon (Davies, 2000; Appendix 1). Such an estimate is dependent on details such as the stocking, feeding and harvesting strategies employed during cultivation. This information was derived from the records of a major salmon producer in Scotland, averaged over a large number of their on-growing sea cage sites (see Appendix 1). Parameters for the mass balance model were based largely on these real data.

Applying values for feed wastage of 5 % and assuming the diet is 90 % digestible, a farm producing 1000 tonnes of salmon over a 20 month production cycle was found to achieve a food conversion ratio (FCR) of 1.17 (on a wet weight basis). The mean nitrogen content of salmon diets (wet weight) can be estimated at 7.2 % (Nutreco, pers. comm.), and the bulk composition of farmed fish is reported to contain 3.4 % (wet weight) nitrogen (Ackefors & Enell, 1990). Therefore, assuming a total mortality rate over the 20-month production cycle of 10 %, the amount of nitrogen released can be calculated as the difference between the amount input in the feed and that incorporated into fish growth. Using this method, the amount of dissolved nitrogen released is estimated at 35.6 kgN $tonne^{-1}$ production, and the amount in particulate waste (wasted feed plus undigested faecal material) is estimated at 12.6 kgN $tonne^{-1}$ production. The total nitrogen discharge rate is therefore the sum of the dissolved and particulate rates, which is 48.2 kgN $tonne^{-1}$ production. For a more detailed description see Appendix 1.

SEPA regulates the finfish industry by consenting a maximum biomass that may be held on a site at any time. The SEPA database, and therefore the FRS database, consists of consented site biomasses. However, the emission of nutrients to the environment is calculated from the mass of fish produced (i.e. grown). There is no simple relationship relating the consented biomass on site to the production of fish.

For the purposes of this modelling exercise, it is assumed that after salmon smolts are first put into sea cages, with a negligible biomass, the maximum consented biomass is reached during the subsequent twelve months. The mass of fish produced in the first year is therefore equivalent to the consented biomass. During the second year of the production cycle, the maximum consented biomass is sustained whilst fish are being harvested, until eventually the biomass falls away as the site is cleared for fallowing. It is assumed, therefore, that a mass of fish equivalent to the consented biomass is again grown on the site during the second year. In summary, for the purposes of this modelling, the consented biomass is assumed to equate to the annual growth of fish on site (i.e. the total mass of fish produced on a site over a production cycle could potentially be twice the maximum biomass).

Since the species produced on site is recorded in the database, the nutrient source, S, can be modified according to the species factors as discussed below (see Table 2).

Nitrogen discharge by non-salmonid species

There is a marked trend of diversification of the fish species cultured in Scottish waters. The industry is still dominated by Atlantic salmon, but interest in other species is expanding quickly. There have been small quantities of trout and turbot produced for many years, but the main new developments are taking place in halibut, cod and haddock. Halibut production is expected to have been 189 tonnes in 2001, and there are plans to increase production to perhaps 10,000 tonnes in 2010. The cultivation of cod has only recently attained commercial production, although the amount produced is still low. It is likely that the current low level of wild stocks will encourage the establishment of a farmed cod industry, and the industry has plans in place that could lead to production of around 5000 tonnes in 2005.

For the purposes of the Locational Guidelines, it is necessary to assess the potential for the farming of “new” species to release waste to the environment, and to take account of the degree to which the rate of production of waste may differ from that of salmon. The approach that has been taken is similar to that described above for salmon, and to that recommended by OSPAR (Guideline 00/9/2) for quantifying and reporting nitrogen and phosphorus discharges from marine fish farms (OSPAR, 2000). Mass balance calculations have been used to estimate the amounts of nitrogen in dissolved and particulate waste from farms producing “new” species.

The primary variables in such calculations are the Food Conversion Ratio (FCR), the composition of the feed, the bulk composition of the fish, the proportions of feed lost as waste pellets and the digestibility of the feed. Davies and Slaski (Appendix 2) undertook an analysis of this type for halibut farming. They estimated that the loss of nitrogen as dissolved substances was 48 kg $tonne^{-1}$ of fish produced. In addition, a further 15 % of the feed was indigestible and 5 % was wasted as uneaten pellets, both assumed to have the same composition as the feed. This amounts to loss of a further 19.1 kg nitrogen per tonne of production. The total discharge of nitrogen is therefore 67.1 kg tonne^{-1} production.

Similar calculations can be carried out for cod farming. Cod diets differ from those for salmon and halibut in that they contain lower proportions of lipid and higher proportions of protein. Consequently the average nitrogen content of cod diets is around 9.3%. A critical factor is the value of the FCR for cod. Reported FCRs for cod farming vary between 0.8 to 1.5. A reasonable value for the industry mean value for the intensive sea stage of cultivation is 1.1. Calculations similar to those described above indicate losses per tonne of production of 20.4 kg of particulate nitrogen and 51.9 kg of dissolved nitrogen, a total of 72.3 kg tonne^{-1}.

Turbot are cultivated using diets similar to those for cod (9.3 % N), with reported FCR values similar to halibut (1.3 as discussed in Appendix 2). Application of simple mass balance models suggests a high nitrogen discharge rate of 86.9 kgN tonne^{-1}.

Haddock have a similar dietary requirement to cod, and as a first approximation are taken to have the same characteristics of waste discharge. The figures for haddock must be viewed with caution as cultivation in Scotland is only at an experimental phase and reliable values for food conversion efficiency are not yet available.

For ease of application of these data in the regulatory framework, it is convenient to express the relative rates of discharge of waste as ratios to that for salmon, giving species-specific discharge factors for nitrogen (see Table 2).

Table 2. Species-specific levels of nutrient nitrogen discharge, expressed as a 'species factor' relative to salmon.

Species	Total discharge of nutrient nitrogen (kg tonne^{-1} production)	Species factor
Salmon	48.2	1.0
Halibut	67.1	1.4
Turbot	86.9	1.8
Cod	72.3	1.5
Haddock	72.3	1.5

Calculation of the Nutrient Enhancement

The equilibrium concentration enhancement, ECE, is calculated by

$$ECE = S.M / Q$$

where M is the total consented biomass of all the finfish farms in the sea loch (tonnes), Q is the flushing rate ($m^3\ y^{-1}$) and S the Source rate (total discharge of nutrient nitrogen, kg $tonne^{-1}$ production). The ECE units are converted from kg m^{-3} to µmol l^{-1}, since measurements of dissolved nutrients are traditionally presented in these units.

The nutrient enhancement model is run for every sea loch in the database and an ECE value obtained for each loch. The potential degree of nutrient enrichment for each loch is therefore assessed and the systems most at risk of overexploitation can be identified.

CARBON DEPOSITION MODEL

The second model used for the categorisation predicts the cumulative areal extent of impact on the seabed due to the carbon deposited from all the finfish sites in each loch. The model is a modified version of the method known as the "Gowen" model (Gowen and Bradbury, 1987; Gowen *et al*., 1994) for predicting the dispersion of particulate material from a fish cage. Gowen *et al*. (1994) showed that the dispersion distance, D_x (m), is given by

$$D_X = UH / w$$

where U is the water current speed ($m\ s^{-1}$), H is the water depth (m) and w is the settling velocity of the particulate material ($m\ s^{-1}$). The distance D_x (m) is the distance particles travel from the cage before settling on the seabed.

This approach has been modified to:

(i) allow for a range of settling speeds for fish faecal material,
(ii) incorporate, where possible, site-specific water depths,
(iii) incorporate estimated site-specific water current speeds based on tidal exchange of the system,
(iv) calculate both D_x and D_Y, along- and across-loch dispersion distances respectively. This allows an elliptical impacted area to be calculated.
(v) assign a proportion of the total faecal mass from a farm to each settling speed, based on an observed normal distribution of faecal settling speeds. This allows areas where deposition exceeds certain critical values to be calculated.

As with the nutrient enhancement model, the benthic impact model takes core data from the sea loch database, and is run routinely for every sea loch in the database. Similar simplifications and assumptions to those described above apply, namely that each loch is assumed to be a rectangular basin with physical dimensions of length (L m), surface area (A m^2) and volume (V m^3) matching those of the real system. The tidal range (R m) of the real system is also entered. Relevant site data extracted from the database includes the number of farms, the consented biomass of each farm (M_S tonnes), the distance of each farm from the mouth of the loch (X_F m) and, in a limited number of cases, the water depth at the site (H_S m).

Results for each sea loch are ranked and indexed. The model therefore provides an indication of the relative degree of impact among all the lochs in the database, in addition to the indexing and categorisations used for the Locational Guidelines.

Calculation of the Dispersion Distances

For each fish farm site in the database, the model calculates two horizontal dispersion distances, D_X and D_Y, along and across the central axis of the rectangular basin respectively. The along-loch dispersion distance is given by

$$D_X = U_S . H_S / w_i$$

where U_S is the tidal current amplitude at the site (m s^{-1}), H_S is the water depth at the site and w_i is the faecal settling velocity (m s^{-1}). The water depth at the site is either extracted from the database or, if unavailable in the database, the mean water depth in the basin, H, is used, where H=V/A (m). There presently exists a rolling programme of entering the site water depths into the database but at the time of writing this is only partially completed. Where the value H_S is taken as the mean depth, H, the value H_S used by the model is constrained to lie between 15 m < H_S < 30 m, which are typical water depths beneath salmon farms in Scottish sea lochs.

The tidal current amplitude, U_S, is assumed to decrease with distance from the mouth of the loch i.e. tidal currents are weaker towards the head of sea lochs (Fig. 1). The amplitude is therefore calculated by

$$U_S = U_0 .(L - X_F) / L$$

where U_0 is the tidal current amplitude at the mouth of the loch, itself calculated from the tidal prism and cross-sectional area of the rectangular basin (X_A = V/L m^2) i.e.

$$U_0 = \pi . A . R / (X_A . P)$$

where P is the period of the semi-diurnal tide (s), R is the tidal range (m) and A the surface area of loch (m^2). The tidal current amplitude is used to calculate the dispersion distance because it provides the furthest distance that particles can travel from the cages due to tidal currents. It is assumed that particles are spread evenly over the area defined by the dispersion distance. The database holds the mean spring tidal range. The effects of the spring-neap cycle are not considered.

It is recognised that these calculations of tidal current amplitudes do not reflect the variations in tidal current amplitudes observed in real loch systems due to topographic effects. However, these simplifications are adequate and necessary. The aim of these models is not to predict the exact settling distributions of particulate waste, but to provide a scaling estimate of the impacted area.

Once the settling velocity, w_i, is specified, the dispersion distance D_X for a site can be calculated. A range of settling velocities is used in these calculations and will be discussed in more detail later.

The across-loch dispersion distance, D_Y, is calculated in a similar fashion to D_X and is given by

$$D_Y = V_S.H_S / w_i$$

where V_S is the across-loch tidal current amplitude (m s^{-1}). The value of V_S cannot be calculated in the same way as U_S since a peak value in the loch is not known. Instead, the across-loch tidal current amplitude is calculated by scaling the value of the along-loch current amplitude according to the width:length ratio of the loch i.e.

$$V_S = U_S.W / L$$

where W is the width of the rectangular basin given by W = A / L (m). The above calculation results in a short, wide loch having an across-loch tidal current amplitude comparative in magnitude to the along-loch amplitude, whereas a long, narrow loch would have a much weaker across-loch current amplitude and hence smaller across-loch dispersion distance.

The dispersion distances, D_X and D_Y, are then modified to allow for the effect of horizontal diffusion, which is an important process in the horizontal dispersion of particulate waste material (Gillibrand and Turrell, 1997b). Given a horizontal diffusion coefficient, K_H (m^2 s^{-1}), a length scale of dispersion is given by $\sigma_K = \sqrt{2.K_H.t}$, where t (s) is the time scale over which the diffusion acts. In the present application, the time scale is the time that particles spend settling through the water column, given by $t = H_S / w_i$. The dispersion distances were therefore incremented by

$$D_X = D_X + \sqrt{2.K_H.H_S / w_i}$$
$$D_Y = D_Y + \sqrt{2.K_H.H_S / w_i}$$

with K_H set to 0.1 m^2 s^{-1}, which is at the lower end of the range of measured values for Scottish inshore waters (Turrell, 1990).

Calculation of Impacted Seabed Area

Silvert (1994, 1996) presented a calculation of the impacted seabed area, A_I, where

$$A_I = A_F + \pi D^2$$

where A_F is the area of the farm (m^2), and D is a single calculated dispersion distance (m). However, that calculation does not calculate the full extent of the area impacted by particles released from throughout the cage area, and a modified version is used here.

The present model assumes that particles can be excreted by fish at any location within the cage area. The impacted area, therefore, is bounded by particles released at the limits of the farm area (Fig. 2). As illustrated, this calculation requires the approximate dimensions of the farm.

The cage area of the farm, A_C, can be estimated from the consented biomass, M_S (tonnes), by assuming a stocking density, S_D, and net depth, C_D, with

$$A_C = \frac{M_S}{S_D . C_D}$$

For the purposes of providing the Locational Guidelines, a stocking density of S_D = 20 kg m^{-3} and a net depth of C_D = 10 m were assumed. It is recognised that in reality there is considerable variability in these parameters, but is not practical to incorporate such variability into these simple models. The calculation above estimates the total *cage* area. Since cages are grouped together, the farm area can be rather greater than the cage area. Based on a group of 12 cages, in a 6x2 array, with 1m wide walkways around each cage, the farm area, A_F, was increased by a factor of 30% relative to the cage area calculated above i.e.

$$A_F = 1.3 A_C$$

The dimensions of the farm, L_F (as shown in Fig. 2) are then estimated by

$$L_F = 0.5\sqrt{A_F}$$

Again this is a significant simplification, but the variation in cage configurations makes it necessary to adopt a “typical” relationship.

Given the dimensions of the farm, and the across- and along-loch dispersion distances, the impacted seabed area, $A_I(w_i)$, is given by (Fig. 2).

$$A_I(w_i) = \pi.(L_F + D_X).(L_F + D_Y)$$

where A_I is calculated as an ellipse with a major axis ($L_F + D_X$) and minor axis ($L_F + D_Y$).

The area A_I is calculated above for a single faecal settling velocity, w_i. By using a range of settling velocities, a number of seabed areas impacted by different size faecal pellets can be estimated.

Distributions of Faecal Settling Velocities and Carbon Mass

Salmon, and other finfish, excrete faecal pellets of varying size. Pellets of different sizes tend to have different settling speeds. Data on faecal settling velocities collected in order to calibrate the DEPOMOD computer model (Cromey *et al*., 2000) exhibited a range of values from 0.015 m s^{-1} to 0.063 m s^{-1} with a mean and standard deviation of 0.032 m s^{-1} and 0.011 m s^{-1} respectively. The present model used this distribution of settling velocities to predict a corresponding range of dispersion distances and impacted area calculations. By then assigning a proportion of the total excreted faecal carbon mass to each settling velocity, based on the normal distribution described above, a carbon deposition loading was calculated for each predicted impact area. Because the areas reduce in size proportionately as the settling velocities increase, the cumulative loading for each calculated impact area can be estimated by summing loadings from all slower settling particles. This method is presented graphically in Figure 3a.

In the model, nine particle settling velocities were used, increasing linearly from 0.015 m s^{-1} to 0.063 m s^{-1}. For each fish farm site, therefore, an amalgamation of nine elliptical areas, decreasing in size with increasing settling velocity, formed the impacted area (Fig. 3b).

The total mass of carbon excreted by fish in one year is calculated from the consented biomass. As described earlier in this report, the consented biomass is assumed to be equivalent to the mass of fish grown in each 12-month period following smolt input to sea. The mass of carbon excreted, M_C, can be calculated from the Food Conversion Ratio, FCR, the quantity of feed wasted, F_W, the carbon content of the feed, C_C, and the proportion of consumed carbon that is subsequently excreted as particulate waste by the fish, C_E, and is given by

$$M_C = M_S.FCR.(1 - F_W).C_C.C_E$$

The following values were used for the parameters described above:

FCR = 1.17
F_W = 0.05 (5%)
C_C = 0.44 (44%)
C_E = 0.30 (30%)

These values are similar to those used by other authors (e.g. Silvert, 1994)

Analysis of the DEPOMOD data showed that the distribution of faecal settling velocities resembled a normal distribution. Therefore the carbon mass assigned to each settling velocity, $M_C(w_i)$ was given by:

$$M_C(w_i) = \frac{\Delta w.M_C}{\sqrt{2\pi}\sigma} e^{-(w_i - \overline{w_i})^2/2\sigma^2}$$

where $\overline{w_i}$ is the mean settling velocity (= 0.032 m s^{-1}), σ the standard deviation (= 0.011 m s^{-1}) and Δw is the settling velocity increment used in the model (= 0.006 m s^{-1}).

For each particle settling velocity, the associated mass of carbon was assumed to be evenly distributed over the impacted area, giving a carbon loading of

$$C_L(w_i) = M_C(w_i) / A_I(w_i)$$

where $A_I(w_i)$ is the impacted area for each particle size.

The *cumulative loading* over each impacted area, $A_I(w_i)$, was therefore given by incrementing the value $C_L(w_i)$ with the loadings from all smaller size particles i.e. the cumulative carbon loading over an area $A_I(w_i)$ is given by

$$C_L(A_I) = \sum_{j=1}^{i} C_L(w_j)$$

The output of the model for each site, therefore, is a range of nine areas of varying impact and a carbon deposition rate for each of those areas.

Model Output

The aim of this model is to assess the cumulative impact of all the consented finfish farms in a sea loch, and to assess accordingly all the sea lochs in the database.

Two main results are taken from the benthic impact model: (1) the total "impacted" seabed area in the loch and (2) the total "degraded" seabed area in the loch. Both are presented as percentages of the surface area of the loch.

The first output is calculated by summing the areas under each farm in the loch where any deposition is predicted. This is given by the predicted impacted area for the smallest, slowest settling, particles (i.e. $A_I(w_i)$ with i = 1).

The second output, the "degraded" seabed area, is defined as the area where carbon deposition rates exceed 0.70 kgC m^{-2} y^{-1}. This has been found to be a critical value such that in areas where deposition rates exceed this value, the diversity of benthic fauna is significantly reduced (Cromey *et al*., 1998; WRc, 1996). For each farm in the loch, the cumulative carbon loading values for each calculated area, $C_L(A_I)$, are noted until the value exceeds the critical value of 0.70 kgC m^{-2} y^{-1}. The area A_I corresponding to that loading is then recorded as the "degraded" seabed area for that farm. This is repeated for each farm in the loch, and the "degraded" areas are then summed to give the total "degraded" seabed area.

Summary

The total areas of seabed “impacted” and “degraded” by carbon deposition from fish farms in each loch are estimated by applying a modified version of the “Gowen” model of particulate dispersion. This modified model uses the database of physical characteristics of over 100 sea lochs and site details of over 300 finfish farms located in the lochs, making some simplifications and assumptions about the topography and tidal exchange of the lochs.

MODEL RESULTS

The results from the nutrient enhancement model (accurate for July 2002) are summarised in Table 3 below and are presented in Figure 4 as a function of consented biomass (M) and flushing rate (Q). The results from the carbon deposition model are also summarised in Table 4 below. The “degraded” percentage seabed area results are shown as calculated in July 2002.

Table 3. Predicted nutrient enhancement results for Scottish sea lochs (July 2002).

ECE (μmol l^{-1})	Number of Scottish sea lochs
> 10.0	5
3.0 – 10.0	15
1.0 – 3.0	23
0.3 - 1.0	22
< 0.3	46
Total	**111**

Table 4. Predicted “degraded” seabed areas, expressed as a percentage of the total sea loch surface area (July 2002 data).

Predicted “degraded” seabed area (%)	Number of Scottish sea lochs
> 10.0	0
3.0 – 10.0	6
1.0 – 3.0	31
0.3 – 1.0	31
<0.3	43
Total	**111**

CATEGORISATION FOR LOCATIONAL GUIDANCE

In order to interpret the results of both these models in the context of the Locational Guidelines, the predicted ECE values and percentage areas of 'degraded' seabed were combined in a manner which identified the relative potential sensitivity of sea lochs to further fish farming development. The approach adopted involves a semi-logarithmic scaling of ECE values from 0 – 5, such that each sea loch can be assigned an index of nutrient enhancement (see Table 5). In a similar manner, the percentage area of degraded seabed is scaled from 0 – 5, allowing each sea loch to be assigned an index of benthic impact (see Table 6).

Table 5. Index of nutrient enhancement, derived from predicted levels of equilibrium concentration enhancement (ECE) for nitrogen, using the 'nutrient enhancement model' described above.

Predicted ECE for nitrogenous nutrients arising from fish farming (μmol l^{-1})	Nutrient enhancement index
> 10	5
3 – 10	4
1 – 3	3
0.3 – 1	2
< 0.3	1
0	0

Table 6. Index of benthic impact, derived from the percentage area of sea-bed of a loch, predicted to show reduced Infaunal Trophic Indices (ITI) as a result of the deposition of organic matter from fish farms. Percentage areas are derived from the 'carbon deposition model' described above.

Percentage area of sea-bed predicted to be 'degraded' by organic deposition (%)	Benthic impact index
> 10	5
3 – 10	4
1 – 3	3
0.3 – 1	2
< 0.3	1
0	0

These two scaled indices are then added together to give a single combined index for each sea loch. The resultant single index, scaled from 0 – 10, is subsequently used to provide an indication of the relative sensitivity of a sea loch system to further fish farming development. Sea lochs with the highest combined index value are considered most sensitive to the expansion of fish farming operations and as such are considered as Category 1 areas in the Locational Guidelines. All 111 sea lochs modelled are categorised on the basis of the combined index as indicated in Table 7 below.

Table 7. Derivation of Categories 1 - 3 for locational guidance, based on the sum of the nutrient enhancement and benthic impact indices.

Combined 'nutrient enhancement' and 'benthic impact' indices	Category
7 – 10	1
5 – 6	2
0 – 4	3

This derivation of Categories on the basis of a combined index is such that the modelling results for Category 1 sea lochs are towards the top of the scale for either nutrient enhancement or benthic impact. Category 1 areas will necessarily have at least one individual index of 4 or greater (3 - >10 µmol l^{-1} nutrient enhancement or 3 - >10 % degraded sea-bed area). In these areas the most precautious approach to further fish farming development should be adopted. Category 2 areas have at least one individual index value of 3 or greater and a degree of precaution should also be applied to consideration of further fish farming development in these areas.

ACKNOWLEDGMENTS

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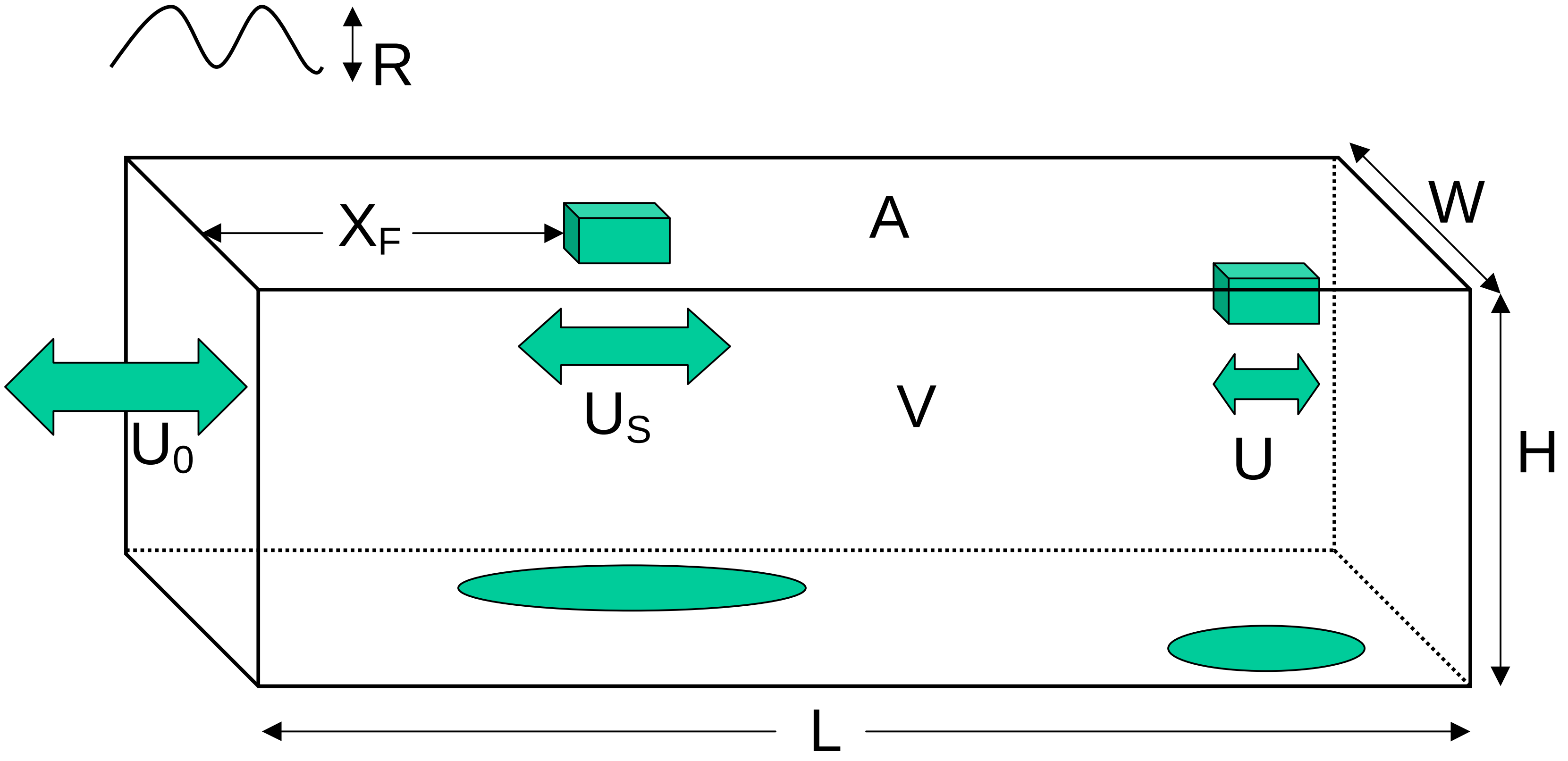

Figure 1. Schematic diagram representing the simplifications used by the carbon depositon model. The sea loch is represented by a rectangular basin with identical physical characteristics to the real system. The tidal current amplitude decreases from U_0 at the mouth to zero at the head, with values at each site, U_S, calculated accordingly. Fish farms are represented by the filled rectangular block, with the areas of impacted seabed denoted by the filled ellipses. All other parameters and variables are described in the text.

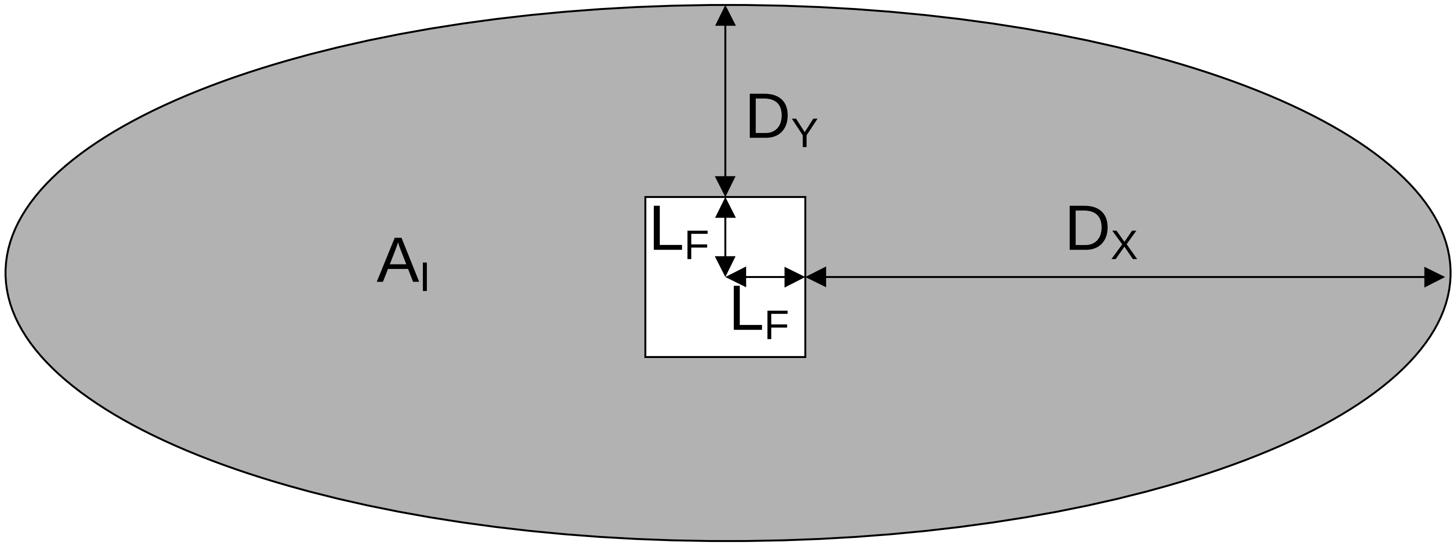

Figure 2. Schematic diagram of the dispersion distances and impacted seabed area beneath a finfish farm. The farm area is considered square. The dispersion distances are maximised by particles released from the edge of the cage area, and the impacted area A_I is calculated as an ellipse by $A_I = \pi\,(L_F + D_X).(L_F + D_Y)$.

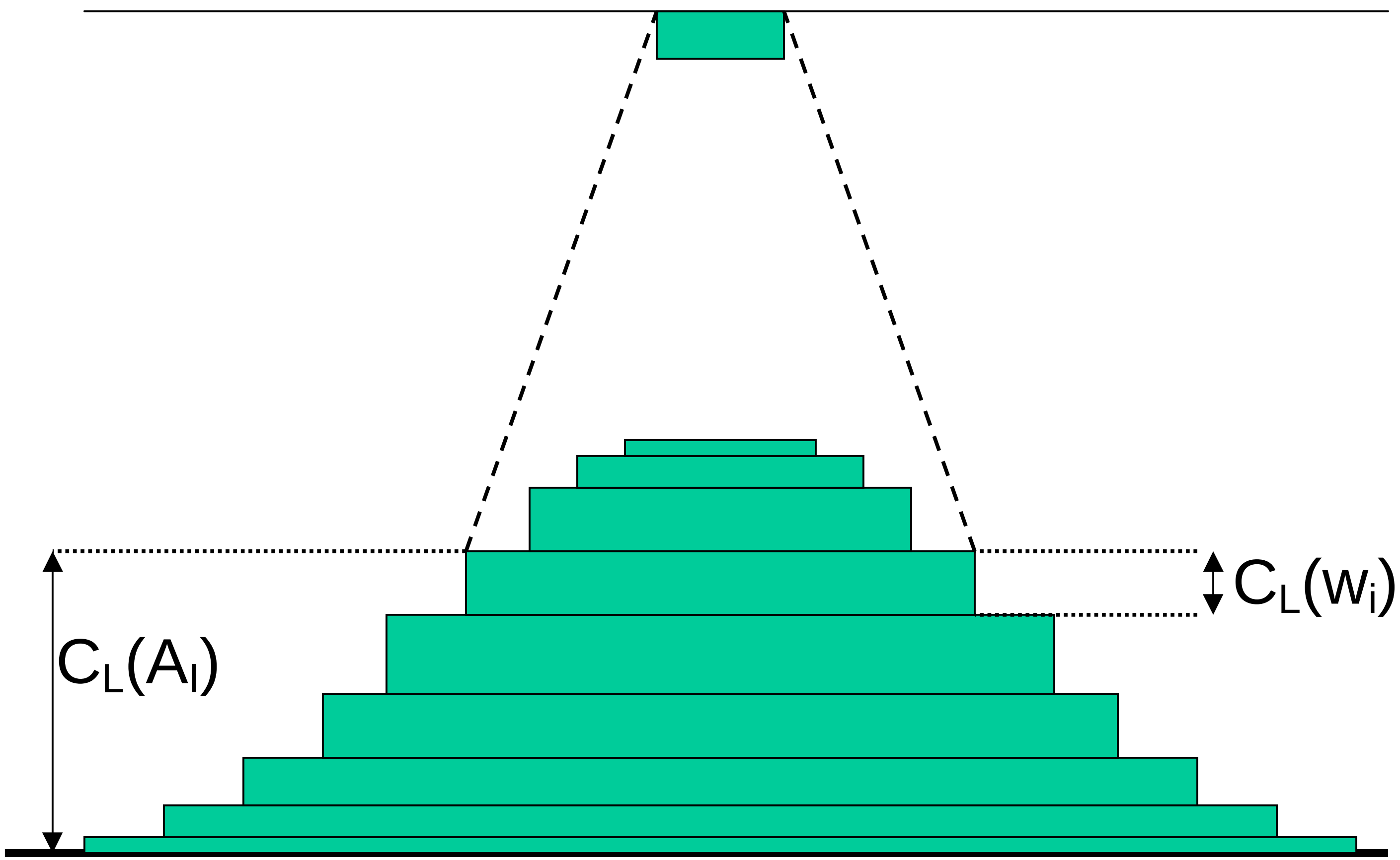

Figure 3a. Schematic diagram showing the accumulation of material of different particle sizes and settling speeds beneath a fish cage. Smaller, slower-settling particles travel further and form the base of the impacted area, with gradually increasing particle sizes and speeds accumulating closer to the cages. The parameter $C_L(w_i)$ is the carbon loading due to particles of settling speed w_i only, which cover an area A_l. The parameter $C_L(A_l)$ is the cumulative loading over area A_l.

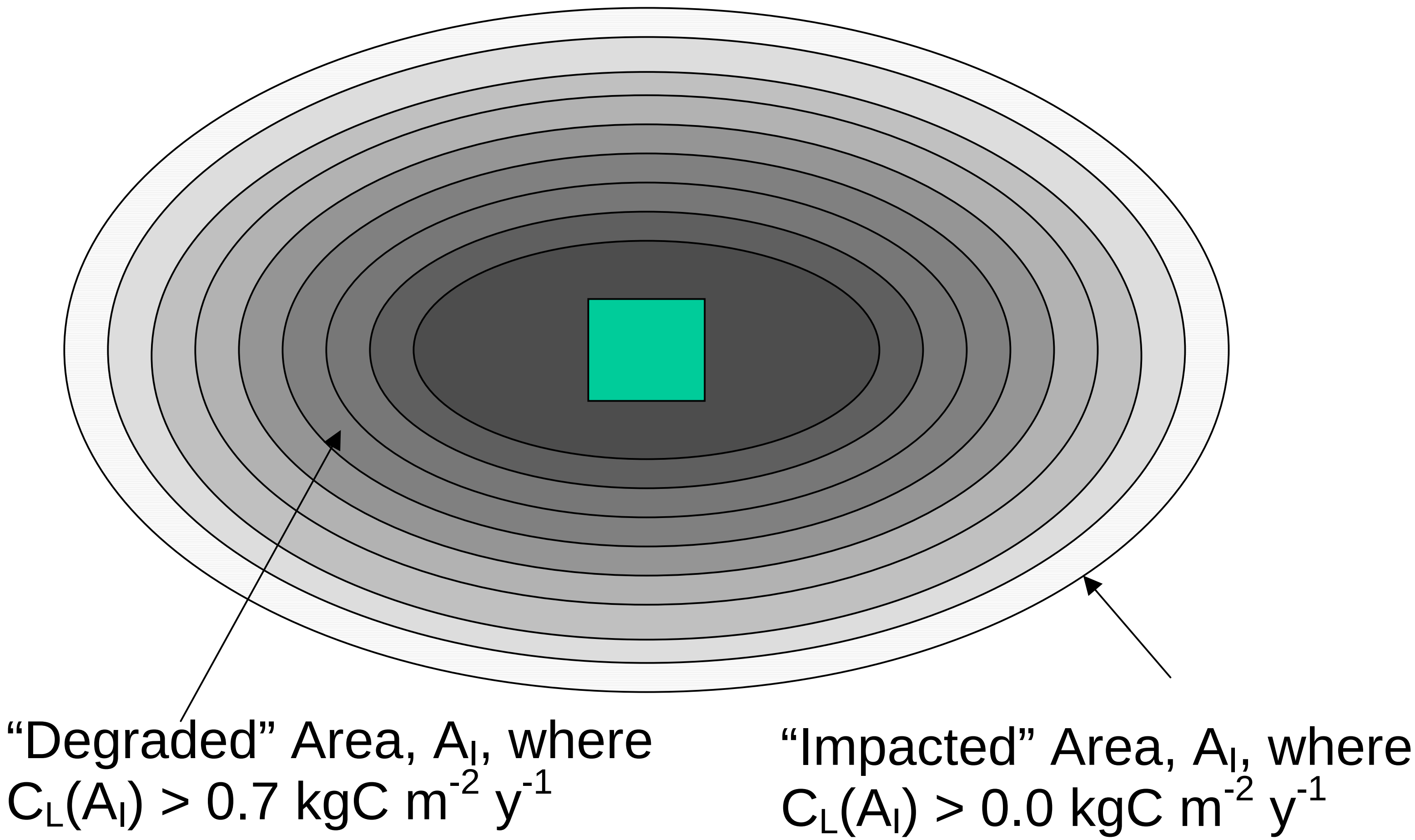

Figure 3b. Schematic diagram showing the elliptical areas calculated for the various particle settling speeds used by the model. The darker shading indicates the increasing cumulative loading, $C_L(A_I)$. The “degraded” seabed area is taken as the largest area, A_I, where $C_L(A_I) > 0.7$ kgC m^{-2} y^{-1}.

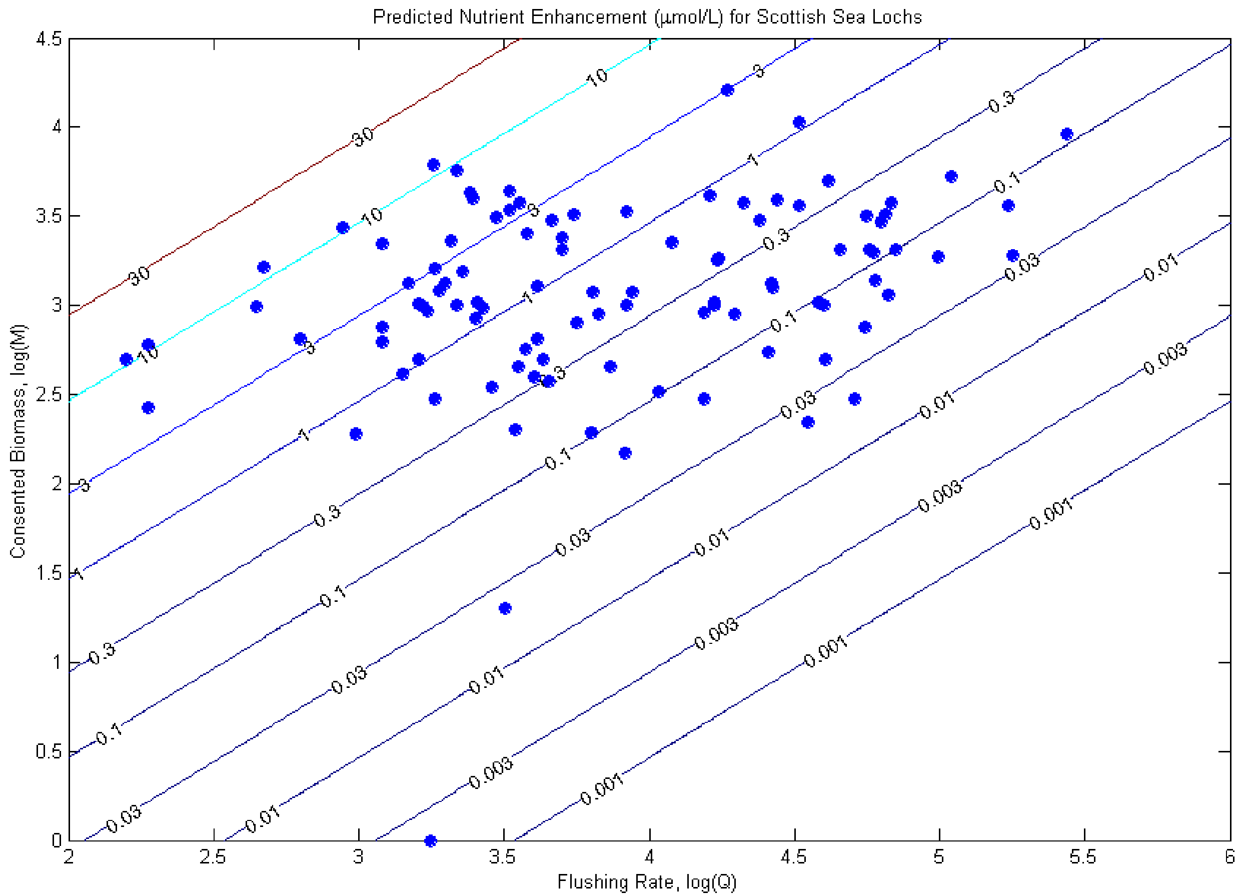

Figure 4. Predicted nutrient enhancement as a function of total consented biomass (M tonnes) and flushing rate (Q m^3 y^{-1}). Contours of ECE (µmol l^{-1}) are plotted and the values for the 111 sea lochs indicated by the filled circles. Logarithmic axes are used for improved clarity.

APPENDIX 1

Not to be cited without prior reference to the authors

International Council for the Exploration of the Sea

CM 2000/O:01
Sustainable Aquaculture Development

WASTE PRODUCTION BY FARMED ATLANTIC SALMON (*SALMO SALAR*) IN SCOTLAND

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ABSTRACT

The assessment of the acceptable level of fish farm development in Scottish coastal areas is strongly dependent on the predicted rate of release of nitrogenous nutrients and particulate organic waste. A simple mass balance model has been used to estimate the rate of production of dissolved and particulate nitrogenous waste by cultivated salmon. The predicted release of dissolved nitrogen is 35-45 kg N/tonne of fish produced, depending upon details of the stocking, feeding and harvesting strategies adopted. Calculations based upon records from a number of Scottish salmon farms indicate that the net waste productions during the cultivation of 'normal' S1 and PP1 (photoperiod modified) smolts are similar.

INTRODUCTION

The most important product of the marine fish cultivation industry in the North-east Atlantic is the Atlantic salmon, *Salmo salar*. Production in coastal waters now exceeds 500,000 tonnes per annum, and the industry continues to make a very significant contribution to the economy of rural areas of countries including Scotland, Norway and Ireland. The growth of the industry has been accompanied by increasingly comprehensive regulatory and monitoring procedures designed to protect the environment. For example, in Scotland, farms are required to obtain Discharge Consent under the Control of Pollution Act, 1974, which commonly limit factors such as the tonnage of fish that may be held (or produced), and the amounts of medicines (such as sea lice treatments) that may be used at each site. The quantities are determined on the basis of a system of environmental quality objectives (EQOs) and standards (EQSs) which have been established to ensure that unacceptable environmental impact does not occur.

An EQO/EQS system can provide a flexible and environmentally sensitive basis for regulation, which takes into account differences (for example in hydrographic characteristics, or conservation interests) between individual sites. However, the equity and effectiveness of the system is dependent upon the quality and reliability of the scientific information used to predict the degree of environmental impact.

Fish farmers in Scotland are required to obtain leases for suitable sites from the Crown Estate Commissioners and permission to discharge wastes (Discharge Consent under the Control of Pollution Act, 1974, as amended by the Environment Act, 1995) from the Scottish Environment Protection Agency. Applications for Leases are now processed through Local Planning Authorities who can take account of a wide range of factors. However, the current Policy Guidance Note issued by the UK Government to assist Local Authorities in this task classifies coastal areas, in terms of their relative potential for further fish farming development. The criteria used to establish these categories include natural heritage conservation aspects, and also the current contributions of fish farming to nitrogenous nutrient levels in the water column, and to organic enrichment of the seabed. Areas where fish farming is already fairly intense in relation to the characteristics of the receiving environment are unlikely to be appropriate for further development.

Similarly, the Scottish Environment Protection Agency, in its considerations of applications for Discharge Consents, takes account of predicted impacts of fish farm developments on nutrient levels and on the sea bed (SEPA, 1998). Measurement of nutrient levels is normally part of the routine environmental monitoring required by SEPA, and the predicted scale of impact on the sea bed is the dominant factor used by SEPA to determine the appropriate tonnage of fish that may be held at each farm site. Waste production rates are therefore central to estimation of the environmental carrying capacity for fish farming, and to the development and regulation of the mariculture sector.

There are relatively few recent studies of the production of waste at salmon farms. At the same time, there have been very considerable developments in feed technology, for example the movement towards high energy feeds with increased proportions of lipid and corresponding lower proportions of protein (and therefore less nitrogen). This has been accompanied by improvements in feed conversion ratios (FCR, the ratio of the weight of feed added to the weight of fish produced) from perhaps 1.6-2.0 to current values of 1.3 or less. As a consequence, the utilisation of nitrogen in feed is now much more efficient.

The most frequently cited values for the rate of excretion of dissolved nitrogen by farmed fish are in the range 75-120 kg N/tonne of fish produced (GESAMP, 1996). A typical salmon farm in Scotland would currently utilise feed containing around 42% protein, with an FCR of around 1.3. This is equivalent to feeding 87 kg N/tonne of fish produced. From this simple calculation it is clear that the higher estimates of the rate of nutrient release (GESAMP, 1996) cannot reflect current farm practice.

The purpose of this paper is to present estimates of the current rates of nitrogen release from marine salmon farms in Scotland. The paper integrates information on typical current (1997-99) feeding, harvesting etc strategies with a mass balance model to provide estimates of waste production resolved into time steps of 4 weeks throughout the marine phase of the production cycle. Comparisons are also made between 'normal' S1 smolts which are transferred to the sea in spring (April-May) and photo-period modified smolts which are placed in sea cages during the autumn (PP1, October-December).

METHODS

A simple mass-balance model has been developed to predict the production of dissolved and particulate waste in salmon farming. In order to undertake the modelling, values have to be selected for a number of basic forcing parameters. These values have been derived initially from a simulation of a salmon farm using values which are considered to be typical of the Scottish industry. The model was then applied to records of a major salmon producer in Scotland, averaged over a large number of their sea sites, using data referring to S1 smolts and PP1 smolts.

The basis of the model is that the dominant source of nitrogen to the fish is the diet, ie that they receive a negligible proportion of their nutrition from natural sources in the surrounding environment of the farm. The feeding rate varies with size of fish and temperature. A proportion of the feed will be lost from the system as uneaten feed pellets. Current salmon farming practice limits wastage to no more than 5% of the added feed, although reliable data on this are rare.

Of the feed which is ingested, a proportion is not digestible by the fish. Feed manufacturers typically estimate the digestibility of feed as in excess of 85%, probably close to 90%. The undigested matter is considered to be excreted as faecal material.

The nitrogen in the digested feed will be partitioned between incorporation into the tissues of the fish as they grow, and released to the environment as dissolved substances (mainly ammonia and dissolved organic compounds). Feed Conversion Ratios (FCR, weight of feed used divided by increase in wet weight of fish) are calculated monthly, from farm records of feed utilised and fish growth indicate. The bulk composition of farmed fish (Ackefors and Enell, 1990) is taken as 3.4% nitrogen (on a wet weight basis), which permits calculation of the amount of nitrogen incorporated in the fish as they grow.

Allowance is made in the model for mortalities of fish. The pattern of mortality reflects common experience, with the highest rates of mortality occurring in the three months immediately after transfer of smolts to sea. The total mortality over the full marine phase is approximately 10%. The harvesting pattern used in the initial simulation again reflects a typical Scottish farm, in which harvesting commences when the mean fish weight reaches approximately 3 kg, and continues for three to five months.

The amount of nitrogen released as dissolved substances is calculated as the difference between the amount input in the feed, and the sum of the amounts in particulate waste (excess pellets plus undigested material) and fish growth.

In the second simulation, real averaged farm records have been used to provide information on the main parameters of the model. These parameters include the staggering of smolt inputs at different sites, the patterns of mortalities, the type of feed used (and therefore its composition), and the feeding rate. The composition of the feed varies as the fish grow; post-smolt diets generally contain more protein than diets fed to larger fish. Allowance is made for harvesting to concentrate on the larger fish in the stocks, and to be spread over a longer time period.

RESULTS

The farm initially simulated by the model produced 1,000 tonnes of salmon from S1 smolts using 1,167 tonnes of feed over a period of 20 months, therefore with an overall FCR (from the transfer to cages to harvest) of 1.17, calculated on total fish weight. Some farms calculate FCR on the weight of fish in harvest condition, after a short period of starvation, and the above FCR is approximately equivalent to an FCR of 1.29 calculated on starved weight of fish. The maximum rate of feed input occurred during the second summer when the fish were in the sea, and ranged from 106-115 tonnes per four week period over 12 weeks (Fig. 1). The peak biomass on the farm occurred over the same period and was 700-800 tonnes (Fig. 2).

The nitrogen input in the feed was calculated to be partitioned as 15% in particulate waste, 40.4% retained in fish growth, 1.6% lost as mortalities and 42.3% lost as dissolved waste. The predicted total amount of dissolved nitrogen released over the full grow-out cycle was 35.6 kgN/t fish produced. The monthly amounts of solid and dissolved waste (Fig. 3) were directly proportional to the amount of feed used, because fixed values were selected for feed wastage, digestibility and FCR.

The sensitivity of the model to variations in the forcing parameters was explored. A decrease of 5% in either the proportion of waste feed pellets or indigestibility decreases the predicted amount of solid waste by 30%, and increases the dissolved nitrogen release by approximately 12%.

Changing the assumed composition of the fish to 3% nitrogen decreased the nitrogen retained by the fish by 12% and increased the dissolved nitrogen loss by 11%.

The model was then applied to real records from a group of salmon farms operated by a major company in the Scottish industry. Calculations were made for S1 smolts and also for PP1 smolts, ie smolts put to sea during the autumn. The results of the calculations are summarised in Table 1. The efficiencies of feed utilisation and the waste production are rather similar for salmon production based upon S1 and PP1 smolts. In both cases, the proportion of feed nitrogen excreted as dissolved waste is greater than that in the initial simulation. This arises from a range of factors, including the relatively poor FCR values experienced in small fish which are contemporaneous with the use of high protein feeds, and the slightly higher rate of losses as mortalities recorded in the real farm data.

The general temporal patterns of release of dissolved nitrogen by S1 and PP1 smolts are rather similar (Fig. 4). Both increase as the rate of feeding increases during the first year at sea. Maximum feed utilisation and dissolved nitrogen excretion occur after the first summer at sea, with a second maximum during the following spring. The second maximum is less marked for the PP1 generation, as by that time the harvest of the PP1 fish is more advanced than that of the S1 fish, and therefore less feed is required. The main differences between the temporal patterns is the greater excretion by the PP1 fish during their first winter and spring at sea, ie before the S1 smolts have gone to sea and during their immediate post-smolt period. Conversely, there is greater excretion by S1 fish in the latter third of their production cycle, mainly due to the earlier harvest of the PP1 fish.

DISCUSSION

The mass balance approach adopted in this model has been shown to provide a robust estimate of the waste production from modern salmon farms in Scotland using high energy feed. The precise values obtained from the model differ between the simulation and the calculations based upon real averaged farm data. Clearly the values obtained are to a degree dependent upon details of the feeding regime, stocking and harvesting patterns. However, taken over a full growing cycle, the release of dissolved nitrogen is very likely to be in the range of 35-45 kg/tonne of production, ie much less than the GESAMP (1996) estimates.

Looking to the future, improvements in husbandry and feed technology are likely to reduce the rate of waste production further. The current main areas of improvement are:

a) There may be some room for improvement in the current digestibility values of feed of around 90%, although the inevitable inclusion to a few percent of ash in the feed will provide a limit to the possible maximum digestibility, unless economic methods are found to reduce the ash content.

b) The wider introduction of automatic feeders responsive to the demands of the fish provide several benefits. For example, the fish will not be fed beyond satiation, and therefore the amount of uneaten feed pellets may be decreased below the 5% assumed in the above calculations. This will reduce the amount of particulate waste, reduce the environmental impact on the sea bed below the farms and make a higher proportion of the feed available for assimilation by the fish.

Also, they will encourage optimum feeding rates throughout the growing cycle. Traditional feeding regimes for S1 smolts greatly reduce the rate of feeding during the second winter at sea. There is increasing evidence that feeding rates can be held at higher values at this time, resulting in continuing rapid growth and improved FCR values.

c) The almost universal use of vaccination for disease control, and the increased availability of a range of sealice treatment chemicals together with strategic approaches to sea lice control, should improve the overall health and performance of the salmon stocks.

As a result of these and other developments, it has been the experience in the Scottish industry that FCR values have improved by approximately 0.05 per growing cycle. It is anticipated that this pattern will continue. If the FCR can be reduced to 1.0, feed wastage to 1%, and indigestibility to 5%, the release rate of dissolved nitrogen would be approximately 33 kg/tonne of production. This may be close to the limit achievable using the methods currently in use in Scotland, and further marked reductions in dissolved nitrogen release may require more innovative approaches.

It is beyond the scope of this paper to discuss in any detail the fate of the nitrogen released from the farmed salmon. The models do not attempt to partition dissolved nitrogen between ammonia, urea and other soluble compounds. The nature of this partitioning will influence the subsequent utilisation of the nitrogen by the algal (micro and macro) and microbial communities.

CONCLUSION

Simple mass balance models can be used to estimate the rate of production of dissolved and particulate waste by cultivated salmon. Calculations based on a single simulated farm, and on data from a multi-site business in Scotland, indicate that the estimates are robust to changes in input parameters within reasonable ranges for the Scottish salmon industry. Comparison with established estimates indicates that current feed technology and husbandry methods result in a comparatively low rate of nitrogen release to the surrounding environment. The models do not attempt to partition dissolved nitrogen between ammonia, urea and other soluble compounds. The nature of this partitioning will influence the subsequent utilisation of the nitrogen by the algal (micro and macro) and microbial communities.

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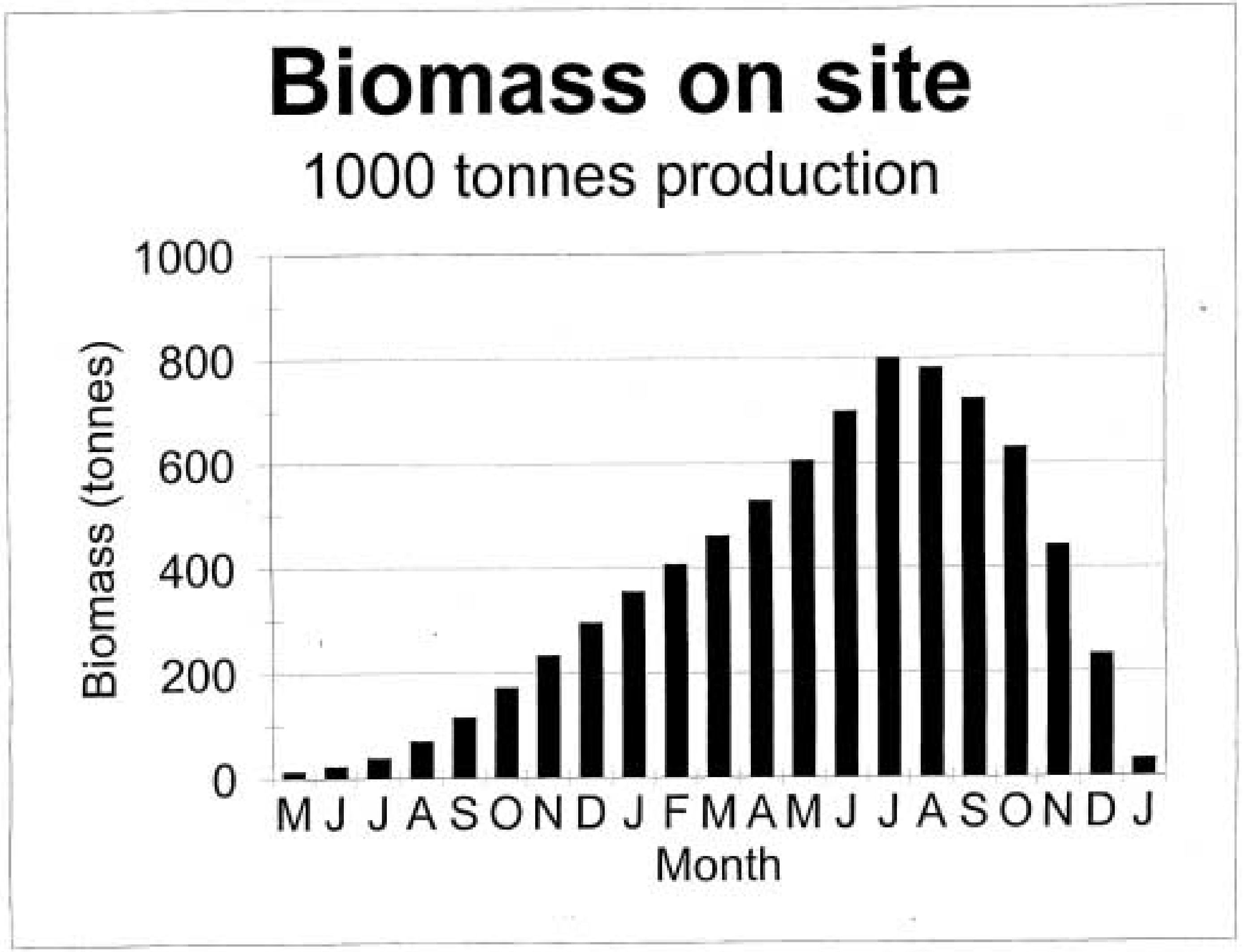
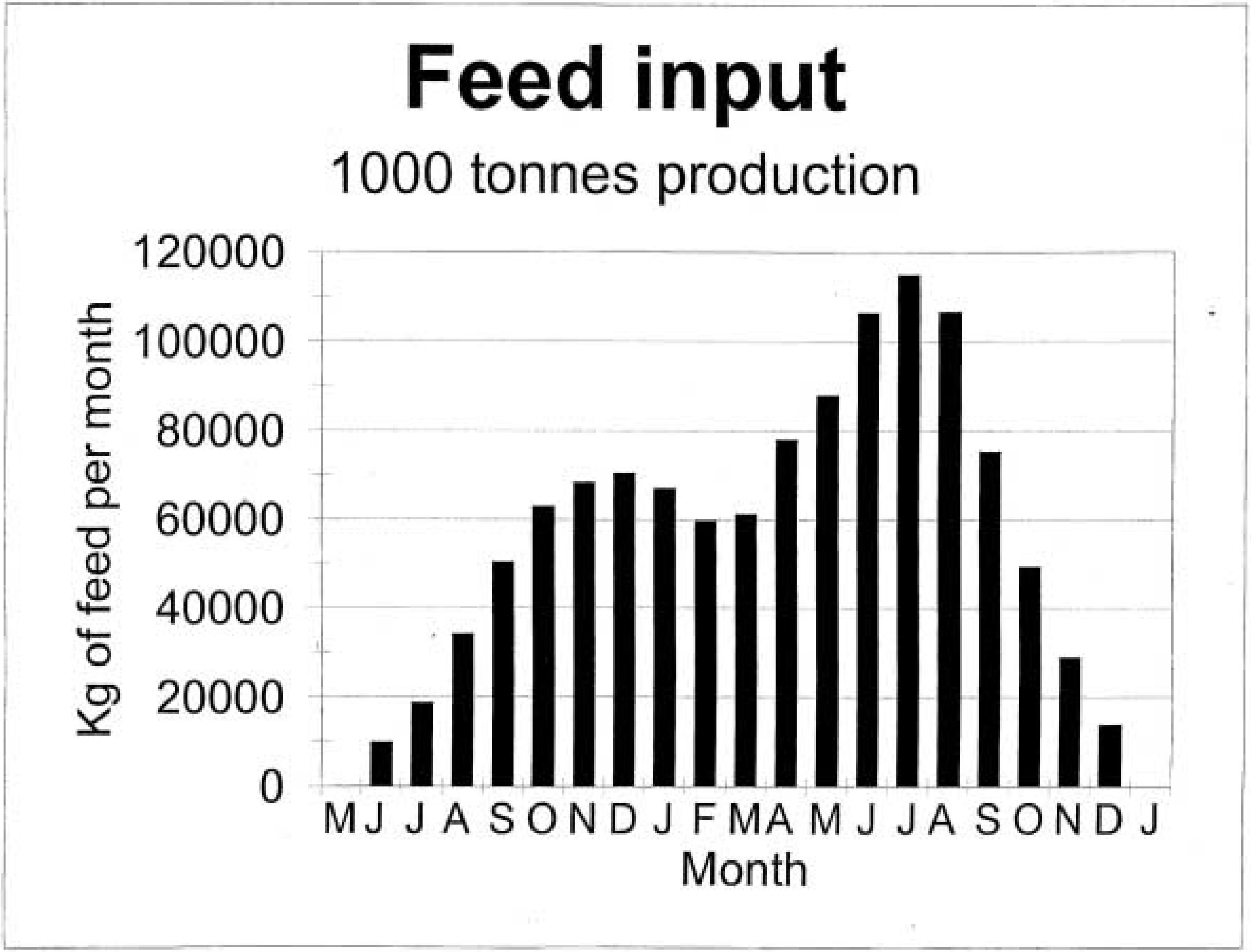
TABLE 1

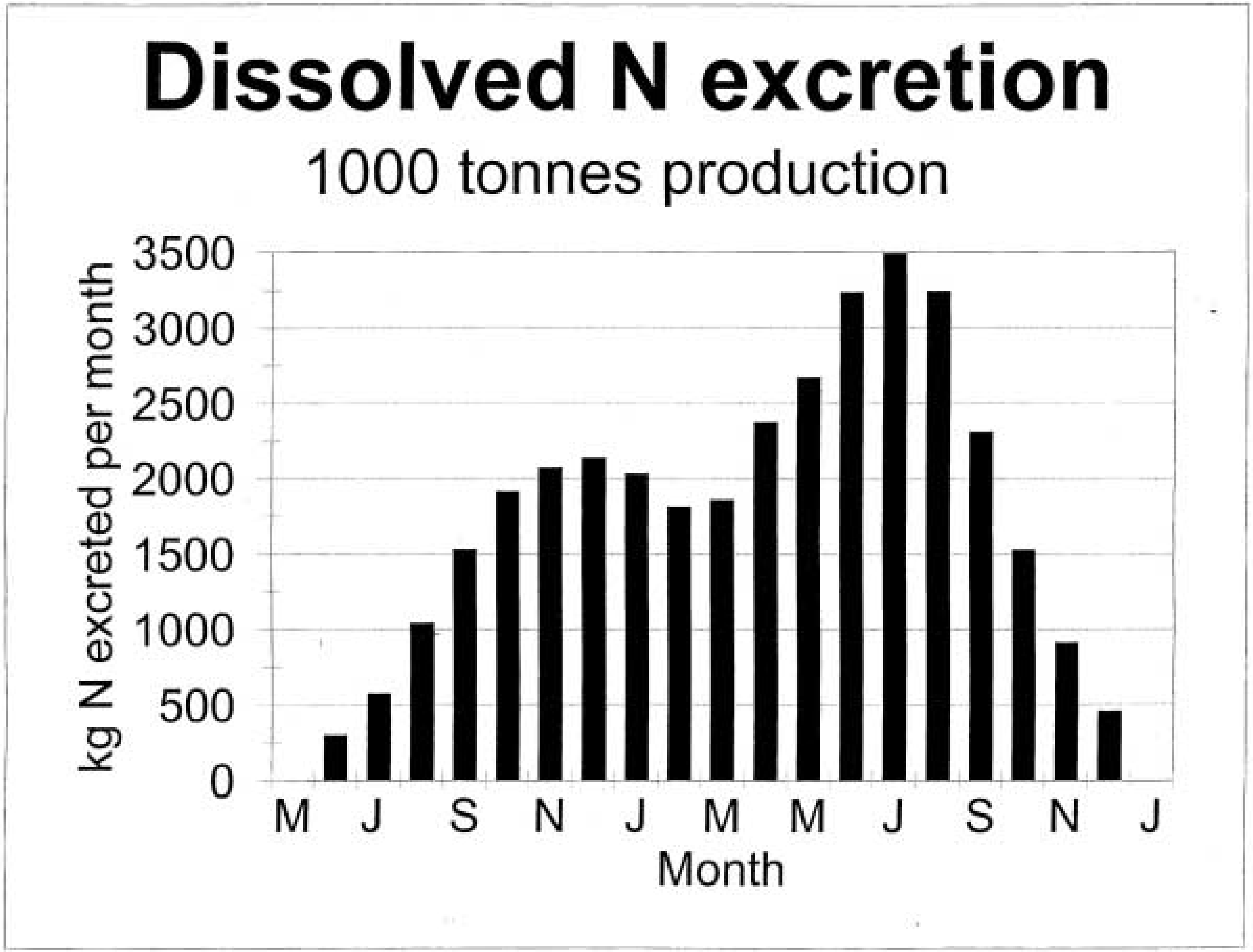
Input parameters and calculated values of waste production during the on-growing of S1 and PP1 salmon smolts.

	S1 smolts	PP1 smolts
Period of transfer to sea	March-May, year 2	Oct-Dec, year 1
Period of harvest	Dec year 2 - Feb year 4	Nov year 2 – Nov year 3
Overall FCR (starved weight)	1.35	1.37
Overall FCR (“live” weight)	1.22	1.23
% N lost as solid waste	15.0	15.0
% N in fish growth	37.9	41.1
% N as dissolved waste	47.1	43.9
Dissolved N lost	42.9 kg/tonne harvest*	44.3 kg/tonne harvest*
Particulate N lost	13.7 kg/tonne harvest*	14.0 kg/tonne harvest*

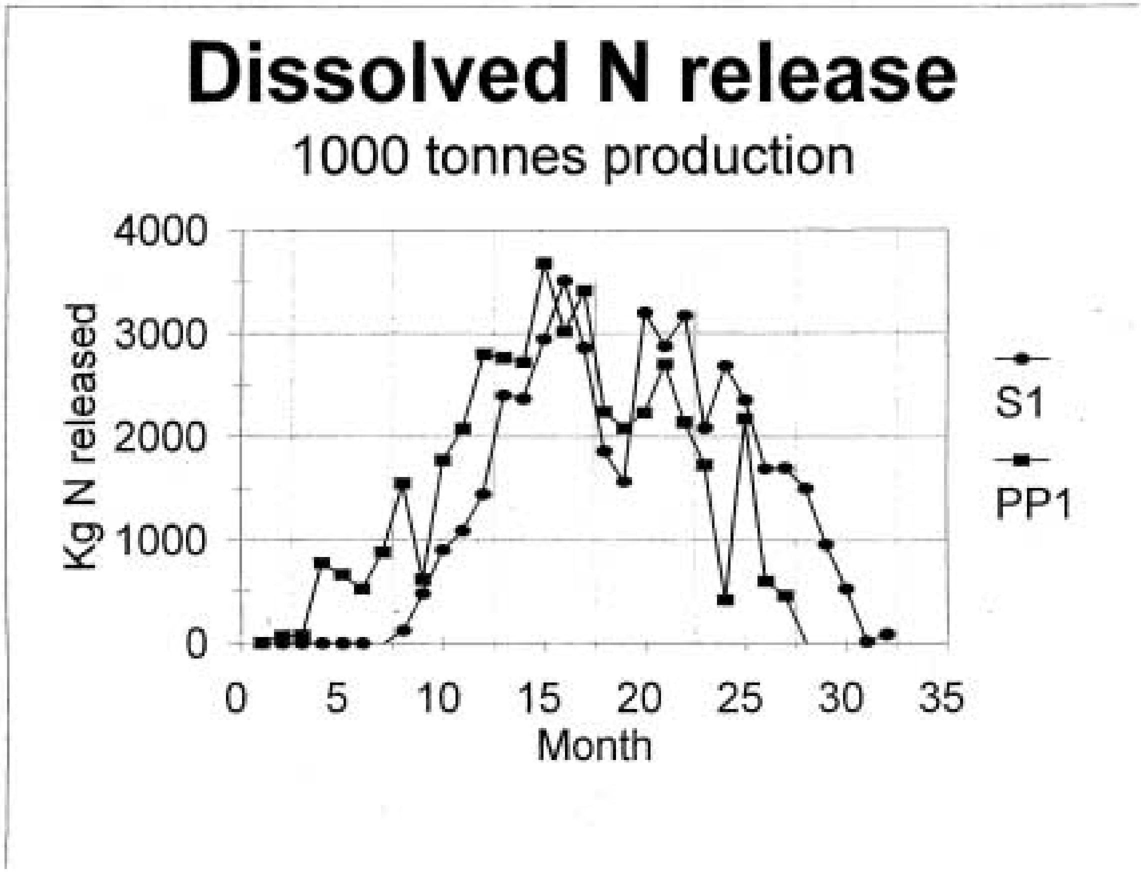
*Based on starved weight, reduce by 10% for ‘live’ weight


Biomass on site
1000 tonnes production
Biomass (tonnes)
1000
800
600
400
200
0
M J J A S O N D J F M A M J J A S O N D J
Month


Feed input
1000 tonnes production
Kg of feed per month
120000
100000
80000
60000
40000
20000
0
M J J A S O N D J F M A M J J A S O N D J
Month



Dissolved N excretion
1000 tonnes production
kg N excreted per month
3500
3000
2500
2000
1500
1000
500
0
M
J
S
N
J
M
M
J
S
N
J
Month


Dissolved N release
1000 tonnes production
Kg N released
0
1000
2000
3000
4000
0
5
10
15
20
25
30
35
Month
S1
PP1

LIST OF FIGURES

APPENDIX 2

Waste production by farmed Atlantic halibut (*Hippoglossus hippoglossus L.*)

Ian M Davies[1*] and Richard J Slaski[2]

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Abstract

The assessment of the acceptable level of fish farm development in Scottish coastal areas is strongly dependent on the predicted rate of release of nitrogenous nutrients and particulate organic waste. A mass balance model of four week resolution has been used to estimate the rate of production of dissolved and particulate waste by cultivated halibut. The predicted release of dissolved nitrogen is 48 kg N/tonne of fish produced and of particulate organic waste is 150-200 kg/tonne production. Modelled biomass-specific rates varied within a range of a factor of about three during the growing cycle, averaging 8.3 mg dissolved N/kg biomass of fish/hr and 28.8 mg/kg/hr of faeces. There is experimental evidence to indicate potential for future developments to lead to significant improvements in feed utilisation by halibut in cages, and an accompanying reduction in waste output. Comparison with experimental data indicate areas where discrepancies between model and measured values suggest that further studies are required in order to more fully inform regulatory processes. These areas include the partitioning of dissolved nitrogen between ammonia, urea and other soluble compounds, and the effects of growth rate and feed utilisation efficiency on waste production.

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1. Introduction

Since its inception, the mainstay of the Scottish marine fish cultivation industry has been the Atlantic salmon, *Salmo salar*. Production in coastal waters now exceeds 100,000 tonnes per annum (FRS, 1998), and has made a very considerable impact on economic activity in remote areas of mainland Scotland and the island groups. The gradual reduction in the price of salmon, limited availability of suitable new sites, and other factors, including regulatory matters, have led the industry to seek opportunities for diversification.

More than 10 years ago, interest began to develop in the cultivation of the Atlantic halibut, *Hippoglossus hippoglossus*. Catches of wild halibut have generally declined, and its high value, good market demand, and potential for rapid growth make it an attractive species for mariculture (Forster, 1999).

Technical and commercial development has been fostered through the British Halibut Association. Although difficulties are still encountered at the first feeding stage of larvae and at metamorphosis from pelagic larvae to flattened benthic fry (Shields *et al*., 1999), it is anticipated that commercial production of around 100 tonnes per annum may be achieved by 2000, and 10,000 tonnes by 2010 (R. Slaski, pers com).

a) Production process

Unlike salmon, halibut is a fully marine species. Hatcheries are required to keep broodfish, and rear juveniles for up to 12 months from eggs through yolk-sac larval, feeding larval, and fry stages to juveniles of 100-200 g which are suitable for transfer to sea cages. The sea cages currently used are converted salmon equipment, with the addition of a taut tarpaulin covering all but the centre of the bottom of the cage. The cages are commonly only around 3 metres deep, and the requirement for particularly sheltered waters to minimise movement of the cage bottom reduces the extent of competition with salmon cultivation for suitable sites. Halibut in sea cages are typically fed on pelleted diets which have been specially formulated for the species. As the industry expands and understanding of the nutritional requirements of halibut continues to improve, it is likely that there will be further development in these formulations. The grow-out period is currently around 30 months (3-4.5 kg fish), compared to 15-22 months for salmon.

b) Regulation

One of the potential constraints on the development of halibut cultivation in Scotland is the need to obtain leases for suitable sites from the Crown Estate Commissioners and permission to discharge wastes (Discharge Consent under the Control of Pollution Act, 1974, as amended by the Environment Act, 1995). Applications for Leases are now processed through Local Planning Authorities who can take account of a wide range of factors. However, the current Policy Guidance Note issued by Government to assist Local Authorities in this task identifies three categories of coastal area, in terms of their relative potential for further fish farming development. The criteria used to establish these categories include natural heritage conservation aspects, and also the current contributions of fish farming to nitrogenous nutrient levels in the water column, and to organic enrichment of the sea bed. Areas where fish farming is already fairly intense in relation to the characteristics of the receiving environment are unlikely to be appropriate for further development.

Similarly, the Scottish Environment Protection Agency (SEPA) in its considerations of applications for Discharge Consents, takes account of predicted impacts of fish farm developments on nutrient levels and on the sea bed (SEPA, 1998). Measurement of nutrient levels in sea water is normally part of the routine environmental monitoring required by SEPA, and the predicted scale of impact on the sea bed is the dominant factor used by SEPA to determine the appropriate tonnage of fish that may be held at each farm site. Waste production rates are therefore central to estimation of the environmental carrying capacity for fish farming, and to the development and regulation of the mariculture sector.

The historical emphasis in Scotland on salmon cultivation has meant that assessments of applications for fish farms, for example in relation to Environmental Quality Standards, are normally carried out using waste discharge amounts and rates as would be predicted for salmon cultivation units. However, the lower feeding rates (normally <1% body weight per day) used in halibut farming than in salmon farming (1-2% body weight per day in smaller fish), and the longer grow-out periods, suggest that waste production rates may be significantly different in halibut cultivation than in salmon cultivation. The purpose of this paper is to present model-based predictions of nitrogenous waste production in halibut cultivation in cages, to make comparisons with the limited published information, and to identify areas where additional research is required.

c) <u>Predicted waste production</u>

A mass-balance model has been developed to predict the production of dissolved and particulate waste in halibut farming. In order to undertake the modelling, values have to be selected for a number of basic forcing parameters.

The basis of the model is that the dominant source of nitrogen to the fish is through the diet, ie they receive a negligible proportion of their nutrition from natural sources in the surrounding environment of the farm. A typical halibut on-growing diet has 48% protein containing 6.25% nitrogen.

A proportion of the feed will be lost from the system as uneaten feed pellets. Current salmon farming practice limits this to no more than 5% of the added feed. The presence of the tarpaulin at the bottom of halibut cages may retain excess pellets longer than open netting, giving the fish more opportunity to ingest the pellets that would otherwise have fallen through the cage. In the absence of published data, 5% loss as uneaten pellets has been assumed.

Of the feed which is ingested, a proportion will not be digestible by the fish. The feed manufacturer (Trouw Aquaculture, pers. comm.) estimates that 15% may be undigested, and this will be excreted as faecal material. Berge *et al.* (1999) measured digestibility coefficients of 84-86% for nitrogen in two types of feed.

The nitrogen in the digested feed will be partitioned between incorporation into the tissues of the fish as they grow, and released to the environment as dissolved substances (mainly ammonia and dissolved organic compounds). The estimated likely Feed Conversion Ratio (FCR, weight of feed used divided by wet weight of fish produced) over a full growth cycle is 1.3, and FCRs of 0.8-1.2 have been measured for halibut in commercial cage cultivation of 100-1,000 g (D. Mitchell, pers comm.). Published data for feed efficiency ratios (wet weight of fish produced divided by the weight of feed used) for halibut of 5-500 g in tanks (Aksnes *et al.,* 1996; Berge *et al*., 1999) range between 1.18 and 1.50 (equivalent to FCR of 0.67-0.85), depending on diet and other factors, but generally group around 1.25-1.30. These data indicate that halibut have the potential, in common with some other flatfish, such as turbot *Scophthalmus maximus* (Nijhof, 1994) to convert feed with very high efficiency under optimum conditions. These data are not inconsistent with data for turbot in commercial production (R. Slaski, unpub) which indicate FCRs of around 1.0 in juveniles and 1.6 in pre-harvest fish (although this will have been influenced by early maturity in males), with a value over the production cycle of 1.3. It was decided to utilise a fixed value for the FCR of 1.3 (the effect of variations of this figure will be discussed later). The bulk composition of farmed fish (Ackefors and Enell, 1990) was taken as 3% nitrogen (on a wet weight basis). Berge *et al.* (1999) report mean whole body protein values of 16.5% in halibut raised in tanks, equivalent to 2.64% nitrogen.

The amount of nitrogen released as dissolved substances is calculated as the difference between the amount input in the feed, and the sum of the amounts in particulate waste (excess pellets plus undigested material) and fish growth.

2. Results

The farm simulated by the model produced 162 tonnes of fish from 202 tonnes of feed over a period of 31 months, with an FCR (from the transfer to cages to harvest) of 1.31. The maximum rate of feed input was 18 tonnes per four week period (Fig. 1), and the peak biomass on the farm was 129 tonnes of fish (Fig. 2). The feeding rate varied seasonally from 0.8% of body weight per day in the first summer, to 0.3% in the second winter (Fig. 3).

The nitrogen input in the feed was calculated to be partitioned as 19.5% in particulate waste, 30% retained in fish growth, and 50.5% lost as dissolved waste. The predicted total amount of dissolved nitrogen released over the full grow-out cycle was 47.8 kgN/t fish produced. The monthly amounts of solid and dissolved waste (Fig. 4) were directly proportional to the amount of feed used, because fixed values were selected for feed wastage, digestibility and FCR.

The sensitivity of the model to variations in the forcing parameters was explored. A decrease of 5% in either the proportion of waste feed pellets or indigestibility decreases the predicted amount of solid waste by 20%, and increases the dissolved nitrogen release by 10%.

Changing the assumed composition of the fish to 2.64% nitrogen (cf Berge *et al.,* 1999), decreased the nitrogen retained by the fish by 12% and increased the dissolved nitrogen loss by 3.5%.

Variation of the FCR between 1.1 and 1.5 changed the predicted production of fish from 188 to 140 tonnes. The predicted release of dissolved N varied between 44% and 54% of the nitrogen added with feed. The mass specific release increased from 36.2 kg N/t to 59.0 kg/t of fish produced.

3. Discussion

a) <u>Dissolved waste production</u>

Skelton (1998) undertook a two month nutrient study of halibut of approximately 250 g initial weight. Fish were held in tanks and fed semi-continuously for 18 hours per day by automatic feeders, which dispensed rations of 0.5% and 3% body weight per day. Feed consumption was very incomplete, and the overall FCR values (2.4 and 6.1 respectively) were high. Consequently, particulate waste dominated the nutrient budgets. However, losses of dissolved nitrogen amounted to 20.5 and 16.0 kgN/t of fish production respectively, 43% and 33% of the rates predicted by the model described above. Similarly, the modelled ratio between dissolved waste nitrogen and nitrogen in fish is 1.66, while Skelton's values are 0.68 and 0.53.

Other workers have generally achieved lower FCR values. Helland and Grisdale-Helland (1998) reported that the retention of nitrogen by halibut in tanks was not affected by a variation of 9% in dietary protein, averaging 41% of the ingested nitrogen. Berge *et al*. (1999) reported that halibut in tanks retained 43-47% of the ingested nitrogen and 51-56% of the digested nitrogen. The comparable modelled values are 30% and 37% respectively. Recalculation of their data indicate losses of 36-39 kg N/t of fish production, which lie between the values measured by Skelton (1998) and the modelled values.

In relation to the potential of dissolved nitrogenous wastes to increase primary production, the proportion of available (inorganic) nitrogen in the effluent is important. Skelton found that ammonia made up 29% and 22% of the dissolved nitrogen, while the rest was not present as nitrate or nitrite, and was probably in organic compounds (which may include urea). In comparison, Bergheim (1984) reported that one third of the dissolved nitrogen excreted by salmon was in the form of ammonia. Dosdat *et al.* (1996) found that 4-6% of the ingested N was excreted as urea by five teleost species. Turbot excreted 20% of the ingested N as ammonia, while the other species excreted 30-38% as ammonia. There is therefore considerable uncertainty in the likely partitioning of excreted dissolved nitrogen between ammonia, urea and other organic compounds, and hence in the availability of this nitrogen to primary producers. The ability of the environment to assimilate (through various processes, including dilution and remineralisation) of nitrogenous waste is dependent on the rate of supply of these materials. The model predicts that dissolved nitrogen release rates of between 4 and 14 mg N/kg biomass of fish/hr, depending upon season (feeding rate), with a mean value of 8.3 mgN/kg/hr (Fig. 5). This is broadly consistent with mean values of 5.4 and 7.5 mgN/kg/hr (fed fish) calculated from Berge *et al.* (1999) and with mean rates of output of ammoniacal nitrogen of 2.3 and 5.1 mgN/kg/hr for starved and fed halibut respectively (Davenport *et al.,* 1990). Similar values have been reported for other flatfish including lemon sole, *Microstomus kitt* (Davenport *et al.,* 1990) and common dab, *Limanda limanda* (Sayer and Davenport, 1987). Dosdat *et al.* (1996) reported values for five species of fish ranging from 10-19 mgN/kg/hr in fish feeding at 1.5% body weight per day, and 3.1-6.3 mgN/kg/hr at 0.5% body weight per day. Turbot excreted significantly less ammonia than the other species studied. The average rates measured by Skelton (1998) of 0.5 and 0.7 mg/kg/hr of ammoniacal nitrogen and 1.8-3.4 mg/kg/hr of total dissolved nitrogen are lower than the modelled rates, and towards the lower end of the general ranges reported from other experimental studies.

There are therefore differences between the measured and modelled rates of dissolved nitrogen release by farmed halibut. The measured values, mostly from experimental systems in which fish are growing rapidly and with high feed efficiency, are generally lower than the modelled rates (47 kg N/t fish produced, 4-12 mg N/kg/hr), but mainly within a factor of two of these values. Approximately one third of this loss is probably in inorganic form (ammonia), with the rest incorporated in dissolved organic compounds. The modelled values may therefore represent possible maximum values.

b) Solid waste production

The proportion of indigestable material in the feed is a forcing parameter of the model, and was set at 15%, in accordance with the feed manufacturer's estimate. From this, the discharge of faecal matter can be estimated from the amount of feed supplied less the wastage of uneaten pellets. This is equivalent to 0.18 tonnes of organic matter per tonne of fish production. The mean rate of production of faeces was 28.8 mg/kg/hr, and ranged seasonally between 15.9 and 46.7 mg/kg/hr (Fig. 6). The inclusion of excess feed pellets at a wastage rate of 5% increases the modelled values for solid waste production by 35%.

There are few published data on the production of faeces by halibut. Grisdale-Helland and Helland (1998) reported digestibility coefficients for protein of 82-86% and of organic matter of 76-86%. Their fish excreted 0.12-0.21 tonnes of organic matter per tonne weight gain of fish (which encompasses the modelled value) and they noted that the use of feeds containing higher concentrations of carbohydrate (e.g. 16-17% starch) resulted in greater production of faecal organic matter.

There is therefore general agreement between experimental and model studies indicating that current feeds contain about 85% of matter digestible by halibut, and that faecal waste organic matter will be excreted at a rate of 0.15-0.20 tonnes per tonne of fish production. The excretion rate will vary markedly between about 15 and 50 mg/kg of fish biomass per hour (mean 30 mg/kg/hr) according to the feeding rate. The very low feed conversion ratios that have been achieved by halibut (and some other flatfish species) under optimised conditions in tank culture suggest that there may be considerable scope for improvement in the performance of halibut in cages beyond that which is currently expected to be achieved in Scotland. Such improvements would be accompanied by reductions in the amount of waste produced, and therefore the rates calculated in this report may approach the maximum rates likely to occur in commercial production.

Conclusion

Data are now available to allow simple mass balance models to be used to estimate the rate of production of dissolved and particulate waste by cultivated halibut. Comparison with experimental data indicate the main areas where discrepancies between model and measured values indicate further studies are required in order to more fully inform regulatory processes. These include the partitioning of dissolved nitrogen between ammonia, urea and other soluble compounds, and the effects of growth rate and feed utilisation efficiency on waste production.

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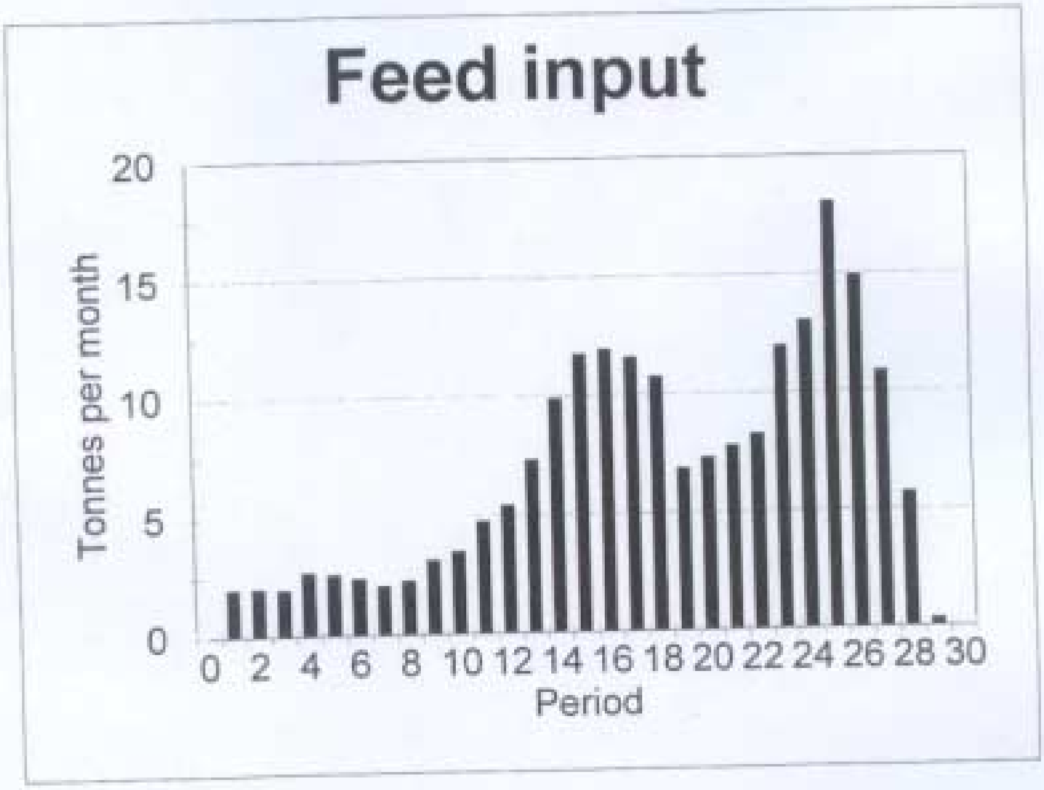
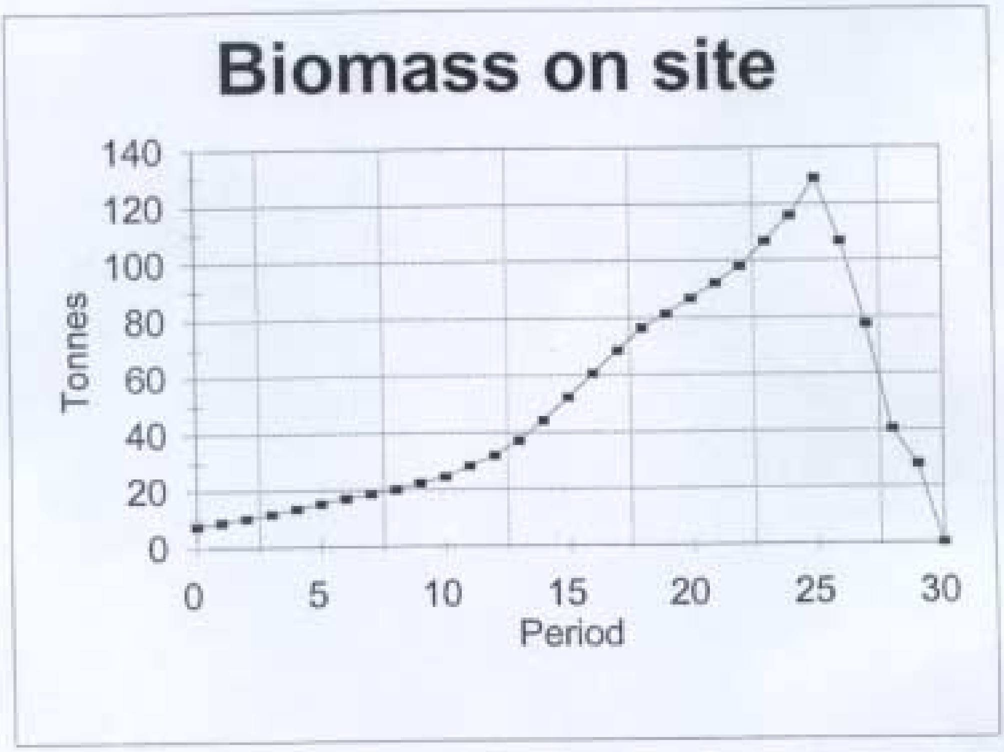
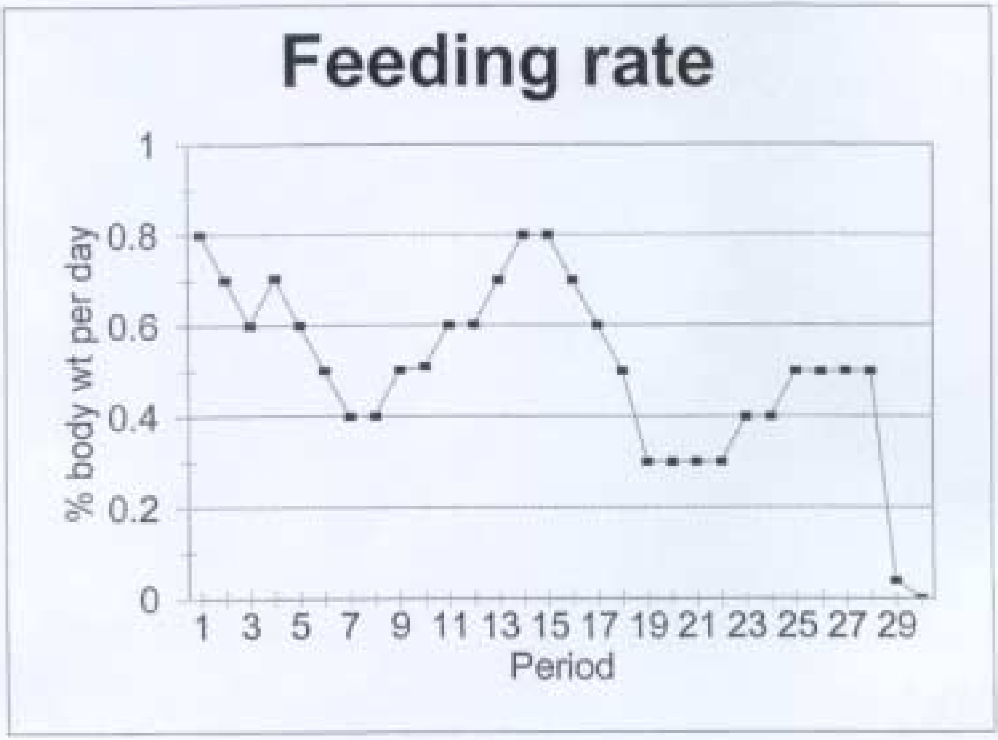
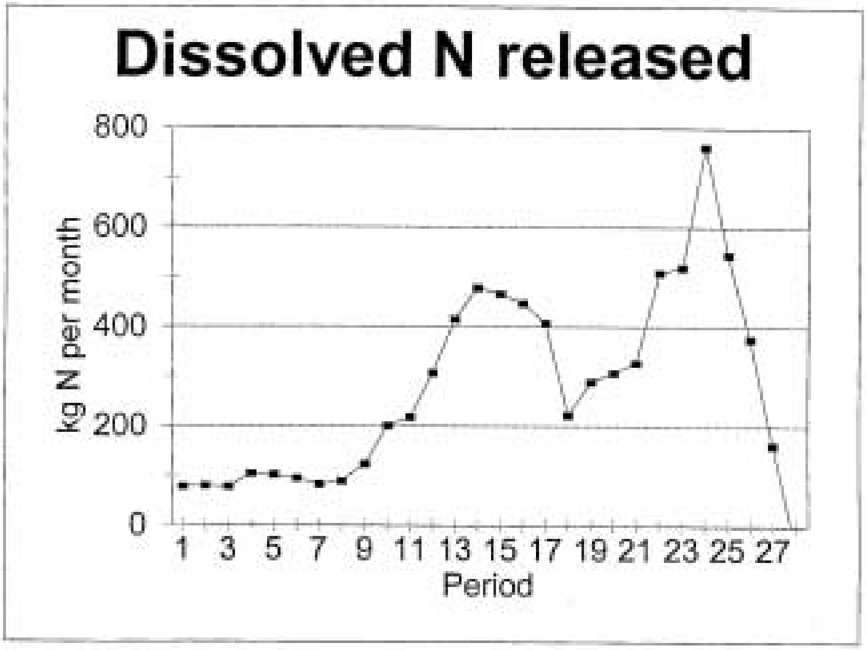
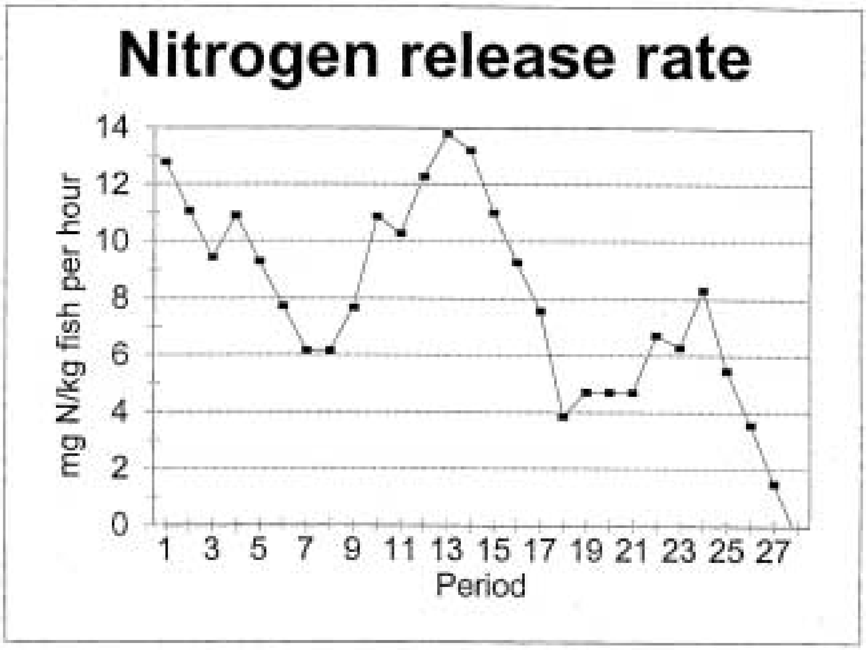
List of Figures

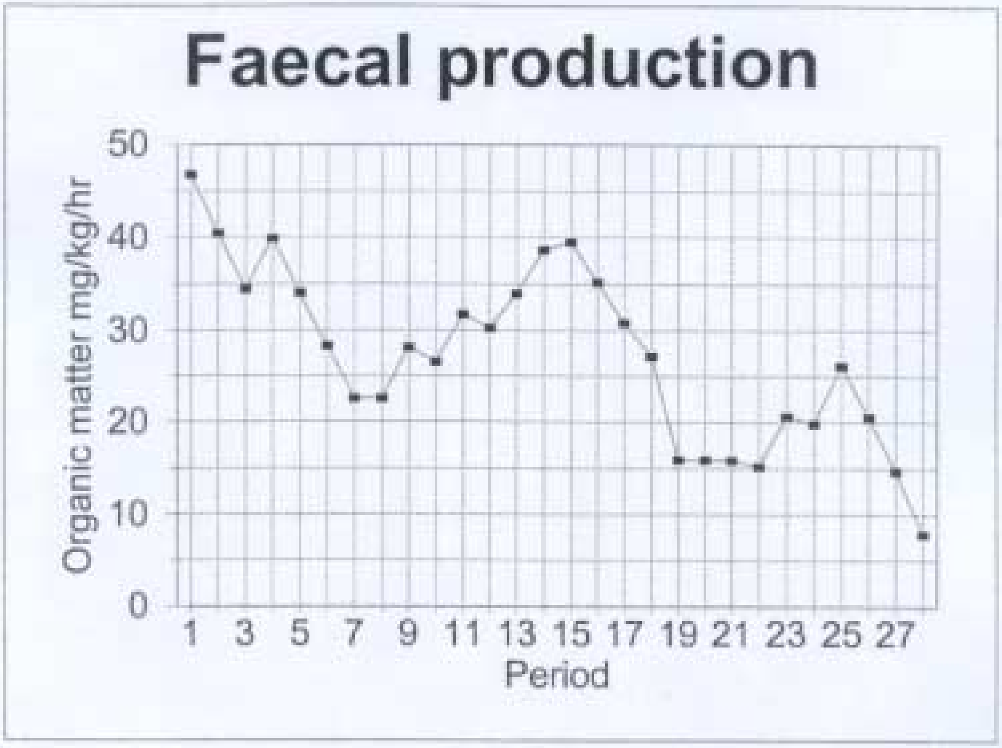

Feed input
Tonnes per month
0
5
10
15
20
0 2 4 6 8 10 12 14 16 18 20 22 24 26 28 30
Period


Biomass on site
Tonnes
140
120
100
80
60
40
20
0
0
5
10
15
20
25
30
Period


Feeding rate
% body wt per day
1
0.8
0.6
0.4
0.2
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29
Period


Dissolved N released
800
600
400
200
0
kg N per month
1 3 5 7 9 11 13 15 17 19 21 23 25 27
Period


Nitrogen release rate
mg N/kg fish per hour
14
12
10
8
6
4
2
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27
Period


Faecal production
Organic matter mg/kg/hr
50
40
30
20
10
0
1 3 5 7 9 11 13 15 17 19 21 23 25 27
Period