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Modelling environmental impacts of marine finfish aquaculture

By W. Silvert[1] and J. W. Sowles[2]

[1]*Habitat Ecology Section, Bedford Institute of Oceanography, PO Box* 1006, *Dartmouth, Nova Scotia B2Y* 4*A*2, *Canada*;
[2]*Maine Department of Environmental Protection, Station No.*17, *State House, Augusta, Maine* 04333, *USA*

Summary

Our understanding of the interactions between fish farms and the environment has reached the stage where it is reasonable to expect that quantitative estimates of the possible environmental consequences of aquaculture development can be provided for regulatory and mitigation purposes. This paper summarizes recent developments in this area and provides several case histories of areas where theoretical analyses appear to have direct practical relevance.

Introduction

Aquaculture is a rapidly developing area of food production. There are legitimate grounds for concern about whether the explosive growth in marine fish farming is likely to be detrimental to the environment. Because marine ecosystems are open and contain wild species, some of which have major economic importance as well as farmed organisms, we need to explore ways of evaluating risks and predicting potential consequences before major aquaculture development is undertaken.

Modelling offers a methodology for simulating and predicting the environmental effects of fish farms. Simple dilution models used to predict impacts from conventional waste water treatment plants where both volumes and concentrations of pollutants are easily known are inadequate for estimating impacts from diffuse finfish pen culture wastes. More complex models have not been widely used to date because the field is new and the models are still crude, requiring further development. An additional problem is that complicated models often require more data than resource agencies are currently able to provide. However, even existing models are good enough to provide valuable information to environmental resource managers. Given the relatively poor data available to resource agencies, it may be appropriate to begin using these quantitative methods in the evaluation of development proposals and in the allocation of scarce agency resources to monitor environmental impacts.

We describe some of the more promising modelling approaches to date and illustrate how these can be used to derive practical meaningful management information.

Sources of contaminants

There are two main sources of environmental disturbance associated with fish farming. These are excretion by fish themselves coupled with loss of feed that is not ingested by fish, and both physical and biological disturbance associated with cage structures.

Fish wastes

Fish are a major source of disturbance to the environment, especially in the quantities grown in fish farms. A single cage 10 m across may contain several tonnes of fish growing at rates as high as 2% per day. This means that the fish are adding close to 100 kg of flesh per day, which translates into a flux of about 1 kg m^{-2} d^{-1} into the fish. Even before we begin to translate this figure into nutrient and carbon fluxes, it is clear that we are dealing with values that are orders of magnitude greater than those normally dealt with in marine ecology.

As a first approximation, we can assume that the metabolic budgets of fish remain constant. While this is not strictly true, by basing the budgets on periods when impacts are likely to be greatest the results can be expected to err on the side of caution. Details of how these budgets are calculated are described in a recent paper by Silvert (1994) and will only be summarized here. The budgetary approach is based on the idea that the ratio of various fluxes is constant and that each flux is proportional to the growth rate. For salmon growth data from South West New Brunswick (R. Cook, pers. comm.) the growth rate can be fit well by a temperature corrected allometric growth model of the form $G = aW^{b}e^{QT}$, where Fig. 1 shows the fit with $Q = 6.4$. Although this seems like a large value, it should be kept in mind that the temperature dependence hides other correlated quantities like seasonal adjustments in metabolic rate and changes in photoperiod.

Nutrients

Nutrient enrichment as a result of fish farming has become a primary concern in some areas of the world such as Scandinavia, Japan, and eastern north Pacific, and can be the major determinant of holding capacity. Potential consequences of enrichment include toxic blooms and hypoxia. The dissolved form and ratios of nitrogen and phosphorus (6:1) released from fish farms (Molver et al. 1988) promote phytoplankton growth. Assessment of the potential impacts is complicated by the difficulty of obtaining reliable estimates of the output (mass) of nutrients from farmed fish (Solbé 1988). Some of this can be attributed to continual improvement in feed formulations (Johnsen et al. 1993) and husbandry practices that increase efficiency of feed to flesh conversion thus reducing excess nutrient loading to the water column. The degree of eutrophication depends on hydrographic conditions, particularly flushing rate (Wildish et al. 1993).

Particulates

Particulate wastes include both wasted feed and fish faeces. Larger particulates settle on the bottom and can lead to severe benthic impacts, while finer particulates lead to increased turbidity in the vicinity of fish farms. Findlay and Watling (1994)

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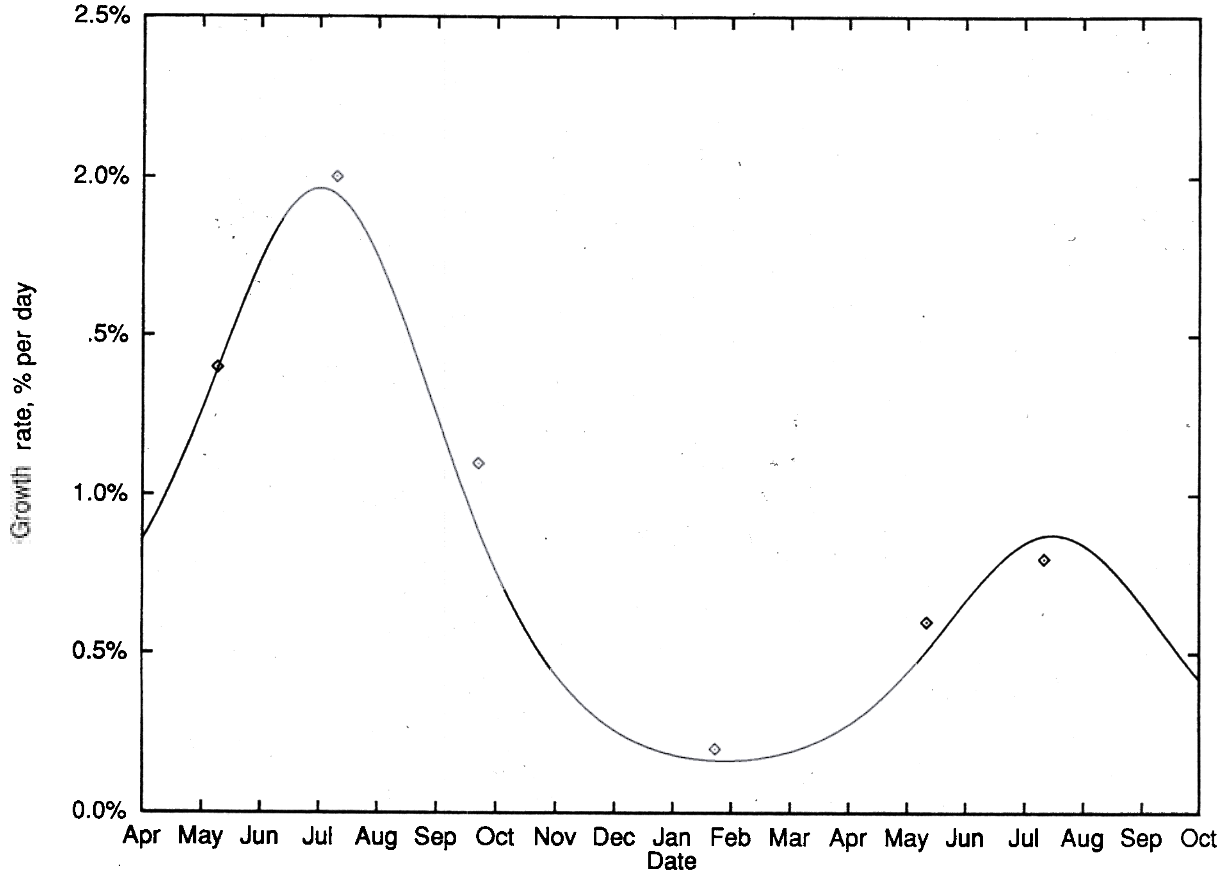

Fig. Growth of salmon plotted against temperature-dependent allometric growth model

report that from 5 to 30% of the feed falls out of the pen system unconsumed. Over a growing season, the weight of combined particulate waste can be as much as 1.36 tonnes per tonne of fish produced (Solbé 1988). Faecal pellets vary greatly in composition and physical characteristics. For salmonids the faeces tend to consist of mucoid strands rather than actual particles, which makes a large difference in their settling characteristics and probably affects their degradation as well. Food pellets may sometimes sink to the bottom intact, although most are probably broken up by sloppy feeding or by physical disturbance as they sink through the cage. Modern feeds tend to decompose faster in water and for this reason, coupled with more efficient feeding practices, it has become increasingly rare for divers to report finding large quantities of intact feed particles on the bottom under cages.

Oxygen demand

Maintenance of adequate oxygen within the water column and sediments is essential for marine life and has become a standard requirement by some regulatory agencies. Obviously, growers are equally interested in this as well if their fish are to thrive and grow. Oxygen demand in the vicinity of fish farms derives from two sources, the fish themselves and the waste particulate organic matter (there may also be increased oxygen demand associated with eutrophication as was mentioned in the section dealing with nutrients). Fish respiration is a function of several conditions including water temperature, fish size, stress, behaviour and satiety. Axler et al. (1993) estimate that 64% of the total oxygen demand associated with pen reared fish comes from respiration. Further, this demand is effectively instantaneous relative to benthic demand, and sometimes becomes a serious problem only for short periods of adverse conditions (Silvert 1992). The remainder of the demand is by microbial decomposition of the organic wastes over time. Unlike flow-through aquaculture systems where there is a continual supply of new water, marine net pen systems are vulnerable to periods of slack tides and low water exchange. Oxygen depletion is often a problem only in the immediate vicinity of the pens, but under conditions of low water exchange it can lead to significant oxygen deficit gradients away from the pen which affect the holding capacity.

Cages

As our understanding of aquaculture impacts increases and our models become more sophisticated, it will become important to take into account the effect that the cages themselves have on the environment, independent of the fish within them. These effects are largely due to the physical presence of the cages, although fouling organisms may also be important to both oxygen demand and nutrient removal.

Physical impacts

There are two main forms of physical interaction between the cages and the environment. One is the ongoing disturbance by the presence of the cages, including mooring systems, which, in addition to directly occupying space on the bottom, can interfere with currents and with fish migration. Another is the episodic disturbance associated with activities such as cleaning the cages. We are unaware of any quantitative measure of these effects.

Growth on cages

The role of epiflora and epifauna has not been extensively studied, but observations by scientists working around fish farms indicate that it may be quite significant. Large masses of *Laminaria* and other epiphytes are frequently observed on cages that have been hauled up for cleaning (B. Hargrave, pers. comm.), but it is not clear that even these quantities of seaweed are sufficient to take up more than a small fraction of the nutrients released by the fish. On the other hand, suspension feeders (mostly *Mytilus*) may remove significant quantities of particulates from the water column. The only study we could find addressing this issue did not find growth of mussels on cages significantly different from growth of mussels located away from the cages (Taylor et al. 1992), although this does not necessarily mean that filtration effects are negligible.

Benthic Impacts

The most detailed modelling work to date has been on the calculation of benthic impacts. This appears to be the result of two factors: benthic impacts are relatively easy to observe and describe, and many of the parameters needed to model benthic deposition are easy to obtain. However, deposition is only part of the impact problem, and further research on the degradation and recovery of areas under fish farms is needed.

Particulate production

Part of the difficulty in modelling benthic impacts is the wide variation in the type of particulates produced by fish farms. As pointed out above, these consist of a variety of different sorts of particulates with different biological properties and settling rates, ranging from intact feed pellets to mucosoid faecal strings.

One reasonable and useful assumption is that benthic impacts are due mostly to carbon loading, and that these impacts are proportional to the amount of carbon and do not depend in any major degree on the form in which the carbon reaches the bottom. This implies that most of the other nutrients released go into the water column, either directly through fish excretion or by leaching from the particulates as they fall through the water column or shortly after they reach the bottom.

Under this assumption we can represent the concentration of particulates produced by a distribution function $X(S)$ such that $X(S)\Delta S$ is the amount of carbon with settling speed between S and $S + \Delta S$. The total carbon production is then $X_{tot} = \int_0^\infty X(S)dS$. If we make the further assumption that the settling speeds are roughly constant during deposition, then this description of the production of particulate carbon is sufficient to permit a detailed computation of the rates of carbon deposition under and near a cage.

Benthic deposition

The calculation of benthic deposition from $X(S)$ is straightforward. If we assume a uniform current $\mathbf{V}$ and depth Z, the time it takes a particle of settling speed S to reach the bottom is $t = Z/S$, and during this time interval the particle will be displaced by the amount $\mathbf{D}_Z(S) = \mathbf{V}t = \mathbf{V}Z/S$. Since we know the distribution function $X(S)$, we can integrate this equation to calculate the distribution of $\mathbf{D}_Z(S)$, which gives the amount of carbon falling at speed S deposited at the point represented by the displacement vector $\mathbf{D}_Z(S)$.

In most cases, such as in tidal inlets, the current $\mathbf{V}$ is variable, and carbon concentrations must be averaged over time. The resulting distribution can be thought of as representing a pile of deposited carbon under each point within the cage. Total deposition is obtained by still another average, this time over all the points within the cage.

As if this were not sufficiently complex, Gowen et al. (1994) addressed the problem that currents are seldom uniform all the way to the bottom. They observed that the correct way to calculate the horizontal displacement in this case is $\mathbf{D}_Z(S) = [\int_0^Z \mathbf{V}(z)dz]/S = \mathbf{V}_{av}/S$, where $\mathbf{V}_{av}$ is the depth-averaged value of $\mathbf{V}$.

A further complication is that the depth is rarely uniform, and thus the current must be averaged over a trajectory like that shown in Fig. 2, corresponding to the actual trajectory of a sinking particle. Gowen et al. (1994) present a rather involved computer algorithm for solving this problem, but conceptually the problem of variable depth is relatively easy to solve. Instead of starting at the surface and following the trajectory down until one reaches the bottom, one starts at the bottom and works upwards to the surface. The advantage of this is that if one starts the trajectory calculation at the surface, the depth at the point at which the particle reaches the bottom is not known until the entire trajectory has been calculated; if one starts from the bottom and backtracks to the surface, the depth is known. Since the horizontal displacement $\mathbf{D}_Z(S)$ is a known function of water depth, and for each point on the bottom the depth Z is known, the value of $-\mathbf{D}_Z(S)$ identifies the point on the surface from which particles which arrive with settling speed S would originate. If this point falls within a cage we can therefore assume that deposition is occurring, and if not we assume that there is no deposition. Although the averaging over settling speed S and over time leads to quite complex mathematical expressions, the algorithm is computationally straightforward.

Much more refined models than this can be developed if sufficient biological and physical data are available. For example, Falconer and Hartnett (1993) describe the use of detailed physical models to evaluate the transport of dissolved and particulate materials around a fish farm in a complex coastal current.

The main limitation to using this type of model is not mathematical, but rather the availability of data and the time and resources needed to do the calculations. Even the type of model calculation represented by Fig. 2 requires detailed bathymetry as well as time-dependent three-dimensional current profiles. In most cases this information is not available, and collecting such detailed data sets is seldom practical. Calculations of this sort are consequently of little value in trying to predict the benthic impacts of mariculture. They are probably more relevant to regulatory policies which restrict the spatial zone of impact to a specified lease area, since in this case what is needed is not the average deposition but the exact boundary of the depositional zone. The backwards algorithm described above is particularly useful in this case, since by working backwards from the bottom at the edges of the lease site one can determine whether any of

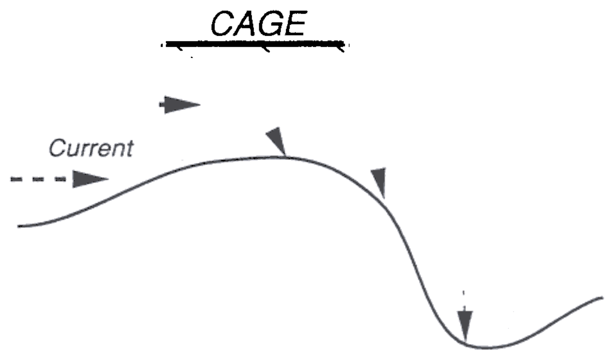

Fig. 2. Trajectory of a particle sinking to an uneven bottom in the presence of a current which varies with depth

the points on the boundary lie within the proposed cage array, and the calculation only need be carried out for the slower settling speeds.

An alternative approach was developed by Silvert (1994) based on a set of simplifying assumptions about the distribution of settling speeds which uses mean current speeds to estimate the horizontal dispersion of particulates. If $<V>$ represents the mean current speed and $<S>$ the mean settling speed, then a rough measure of the displacement of settling particulates is given by $D = <V> <Z>/<S>$, where $<Z>$ is the mean depth. By assuming that the displacement is random, the approximate result $A' = A + \pi D^2$ expresses the area of deposition of particulates, A′, in terms of the area of the cage itself, A. If the cage is circular with radius R, the radius of the depositional area is given by $R' = \sqrt{(R^2 + D^2)}$ and the area is $A' = \pi(R')^2 = \pi R^2 + \pi D^2 = A + \pi D^2$. For other shapes this is not the exact result, but the coefficient is generally close enough to π to make this a useful approximation. Although this is only approximate and involves a number of assumptions, in many cases there is not enough information to hope for more exact results.

The calculation of the deposition rate in this approximation requires only that we know the total output of particulates, X_{tot}, and the area over which they are deposited, A′, giving a mean flux to this area of the bottom of X_{tot}/A'.

It is important to recognize that these calculations are very sensitive to the settling rates used, and thus they are constrained both by the limited data we have on settling rates of particulates under different feeding regimes and by the difficulty of identifying the types of particulates present and their decomposition rates (i.e. the function X(S)). It is also possible that the underlying conceptual models of deposition may be wrong; B. Hargrave (pers. comm.) has suggested that in some cases the mucoid faecal strands from salmonids may settle on the bottom of the cage or on the predator net beneath the cage, and that these are shaken loose during storms and other periods of energetic water movement, which would certainly affect the deposition of these particulates.

Benthic carbon accumulation

Benthic deposition is the forcing function that leads to carbon accumulation on the bottom, but the actual carbon loading is the integrated result of several competing processes which both add and remove carbon; these include not only deposition, but also resuspension, bioturbation, bacterial decomposition and grazing. Each of these processes is complex and difficult to model in detail, but one can try to represent the sum of all of these processes by an uptake-clearance model of the form

$$dC/dt = S - kC \tag{1}$$

where C is the accumulated carbon under the site, S is the depositional rate of particulates, and k is a constant that represents the combined lowest-order effects of removal and degradation processes. Under steady-state conditions C increases asymptotically to a maximum level given by S-k. This seems unreasonable in some respects, since under conditions of constant deposition we might expect to find constant accumulation of buried carbon, but if we interpret C as being the biologically active fraction of total carbon in the benthos, this equation may not be unreasonable.

Benthic deterioration

Accumulated carbon is not itself directly harmful, but its utilization and degradation can have both beneficial and detrimental consequences to the environment. Some of the processes that act on benthic carbon can be beneficial, such as the enrichment of commercially valuable benthic resources like fish and crustaceans. On the other hand, heavy accumulations of organic carbon and other nutrients lead to high bacterial densities and a benthic infauna dominated by a few species tolerant of low oxygen conditions. At extreme depositional fluxes, high bacterial activity leads to anoxia accompanied by the release of H_2S and other toxic gasses, resulting in azoic bottom conditions. In the absence of detailed knowledge about the processes and rates involved in the degradation of organic sediments, the same type of reasoning can be used as in modelling the rate of benthic accumulation; the condition of the bottom is represented by an index B which is determined by a balance between carbon accumulation C and recovery, or

$$dB/dt = wC - rB \tag{2}$$

where w is a constant degradation rate, wC represents a worsening of the benthic conditions proportional to carbon accumulation C, and r is the rate of recovery.

Comparison with field data

The qualitative behaviour of carbon accumulation C and benthic condition B predicted by the above equations is shown in Fig. 3, but the parameter values k, w and r used in the calculations are not easily obtained. This makes it impossible to predict the impacts of fish farms on the environment from first principles, but since the number of parameters is small it is possible to fit the model to field data to evaluate how well it works with the extreme simplifications that have to be made.

Data on 23 sites in the State of Maine have been collected and used to test the model in this way. The data were collected and analysed as part of the state's aquaculture monitoring program (Churchill et al. 1994) and a preliminary analysis was reported in Sowles et al. (1994). Since detrimental benthic impacts at the sites were scored on a basis of 0–4, the benthic condition index B was converted to an index in this range by the transformation $I = 4\sqrt{B}/(1 + \sqrt{B})$ and plotted against the measured scores, as shown in Fig. 4. With the exception of four sites which had anomalously high scores, indicating far worse conditions than predicted by the model, the agreement seems quite good for this stage of model development.

Although there is some scatter on both sides of the line for the lower benthic impacts, at the high end of the scale (benthic scores greater than roughly 1.5) the index predicted by the

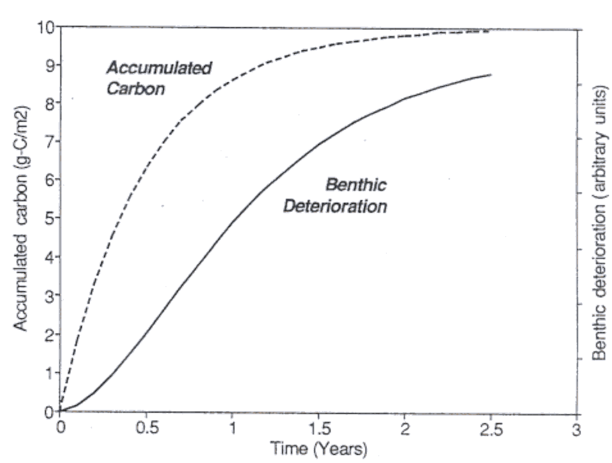

Fig. 3. Carbon accumulation and benthic deterioration under a fish farm, after Sowles et al. (1994)

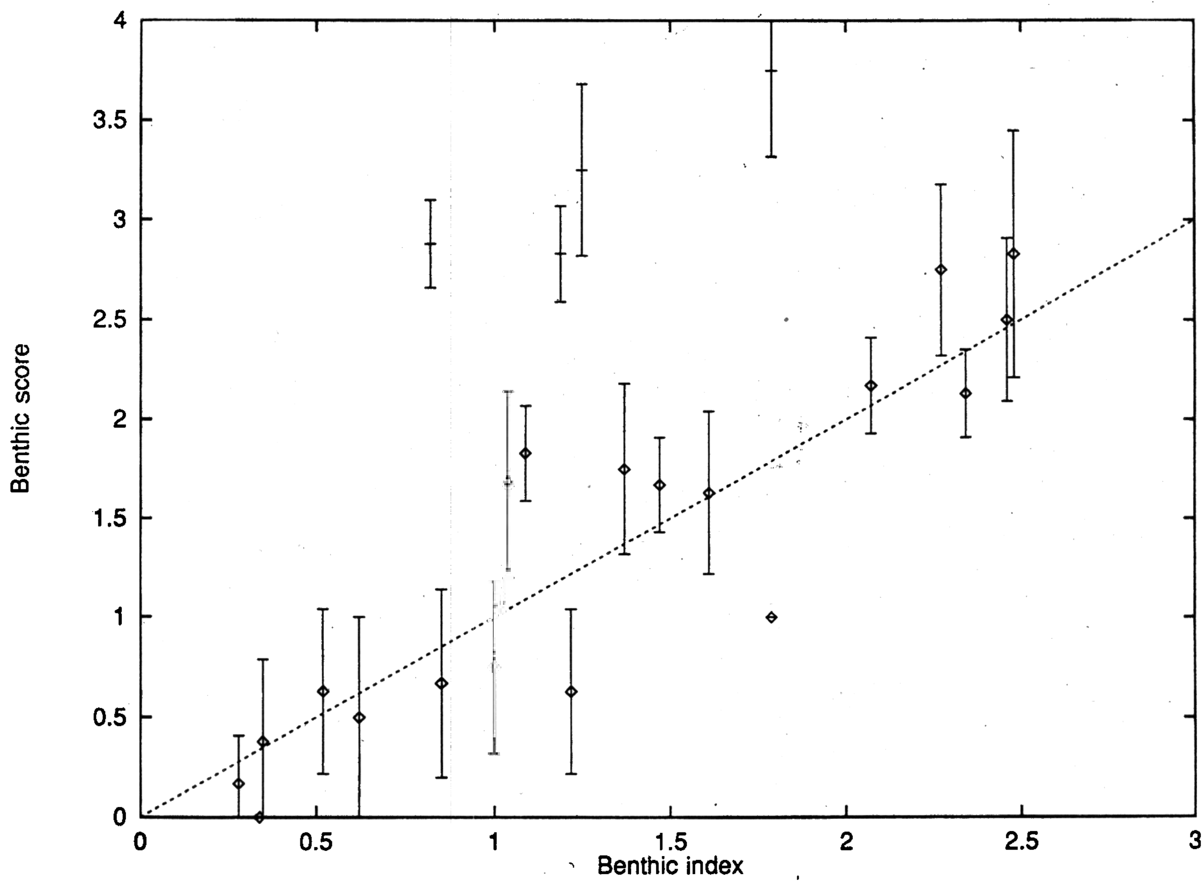

Fig. 4. Observed benthic scores (Churchill et al. 1994) plotted against the benthic index computed from the model

model is comparable to or lower than the observed score. This means that the model as presently formulated is unlikely to indicate greater impacts than actually occur, so use of the model would not lead to predictions that are unduly pessimistic. From the viewpoint of aquaculture development the model can be seen as conservative in this respect. From an environmental management point of view there is a problem in that some sites show far greater benthic deterioration than the model predicts, and these discrepancies are being addressed in ongoing research.

This is a very simple model and involves some drastic assumptions, but it has the advantage that it does not require unrealistic amounts of data to apply. One needs to estimate the particulate output of the fish, which is proportional to growth; the area A occupied by cages is required, as is the settling speed S and depth Z needed for calculation of the time it takes the particulates to reach the bottom. Detailed current data are useful, but a mean current speed V can be used if one assumes that current varies in direction.

Evaluation of NB guidelines

As an illustration of how this model might be used, we have evaluated the proposed guidelines for finfish aquaculture in SW New Brunswick (Anon. 1993) in terms of the type of calculation described above.

The guidelines specify the numbers of fish that can be grown on a site. Without information on fish size or annual production it is difficult to estimate the particulate carbon output, so for the following calculations we have assumed peak waste production of 1 g-C $\cdot$ d^{-1} per fish.

The guidelines relate site potential in numbers of fish to water depth and the area of the lease. The fraction of leased area that can be used for cages is determined by the scope of the mooring lines. Table 1 gives the specified guidelines as well as the benthic carbon loading (BCL) that would be generated, assuming a minimum allowable current of 5 cm $\cdot$ s^{-1}.

The benthic carbon loading (BCL) values are remarkably consistent, indicating that these guidelines will generate roughly the same BCL for farms in different water depths. The BCL level of $\approx$2.5 g-C $\cdot$ m^{-2} $\cdot$ d^{-1} corresponds to an asymptotic benthic score of $\approx$2 on the scale used by Sowles et al. (1994), which appears to be an acceptable level of benthic enrichment (the critical level is about 3), so the guidelines appear consistent with our analysis of the Maine data. Since the guidelines specify that initially the farms should not be allowed to exceed 70% of the site potential, at the present stage of knowledge they appear to incorporate a reasonable assessment of potential benthic impacts.

While these guidelines were developed independently of the model presented here, their consistency with the model indicates

Table 1

Depth (m)	Area (ha)	No. Fish ('000)	BCL
5		120	2.48
10		160	2.47
15		200	2.43
20		240	2.46
25		280	2.45
30		320	2.63

that the model could be used to extend these draft regulatory standards to other regions.

Holding capacity

The calculation of benthic impacts can also be used to assess the allowable size and viability of individual farms, but because the impacts are generally confined to a small area extending no more than a few dozen metres from the edges of the cages, it does not provide a reliable measure of how many fish can be raised in an inlet, since benthic impacts are limiting only on a smaller scale than this (Silvert 1992). The holding capacity, which is a measure of the quantity of fish that can be supported without detriment to either to fish themselves or to the environment, generally depends on effects that occur on a larger spatial scale.

One of the main limitations on the density of fish farms is the specification of a minimum spacing to lessen the chance of disease transmission from one site to another, usually a few hundred metres. Although this is widely used as a regulatory restriction, there appears to be little scientific basis behind the selection of a specific distance.

Another factor of concern is the possibility of nutrient loadings from fish farms affecting primary production, particularly if there is a risk of toxic algal blooms. This is a difficult problem from the viewpoint of predicting how increased nutrients will affect the plankton community, but the calculation of the nutrient changes is not overly difficult and will be described here.

General theory

The nutrient levels within an inlet can be calculated (like almost every other type of loading) by an uptake/clearance type of model

$$dC/dt \doteq N/U - FC \tag{3}$$

where N is the rate of nutrient input, U is the volume, and F is the flushing rate. The nutrient input from fish farming can be calculated on the basis of the model described earlier. Although the model is still at an early stage of development and does not include some important factors, the unfortunate reality is that most regulatory agencies do not possess enough information to exploit even this model fully, and could not make use of more detailed models. Often all they have to work with is an approximate number of fish, whereas a detailed calculation of the wastes would require a breakdown by size and age, along with local temperature data and the like. There are also likely to be inputs from other sources, including wastes from the processing of the fish, so a detailed model of the excretion from individual fish is unlikely to provide a very accurate estimate of the total loadings. However, even estimates of best case (minimum) and worst case (maximum) loadings would be useful knowledge.

Flushing processes

The calculation of nutrient concentrations depends on the rate of flushing, and there are several processes that may need to be taken into account. These processes are roughly additive, so that the total flushing rate can be expressed as $F = F_1 + F_2 + F_3 + \cdots$ where each of the F_i corresponds to a different flushing process. This makes it relatively easy to build generic models since any terms which are not applicable to a particular situation can be omitted without interfering with the evaluation of other flushing mechanisms.

Tidal flushing

Tidal flushing is easily calculated, but there are significant sources of error that must be taken into consideration. A volume of water equal to the tidal prism is exchanged on each tide cycle, so the ratio of the tidal prism to the total volume of the inlet gives an estimate of the amount of flushing per cycle.

This calculation is generally an overestimate of the tidal flushing rate because of incomplete mixing both within the inlet and outside it. Many inlets, particularly those in which the salt water enters beneath a lens of fresh water, are not efficiently flushed by tidal action. If the fish farms are located in brackish water that simply floats up and down on top of the tidal salt water, there may be virtually no tidal flushing. Furthermore, even if the water within the inlet is well-mixed, in many cases the nutrient-laden water that exits on the ebb tide comes back in on the next flood, so the tidal action is not efficient at removing nutrients.

In order to correct the tidal flushing rate for these factors we should multiply it by two mixing terms expressing the fraction of the water which gets mixed both inside and outside the inlet. At present there are few models which can be used to calculate these correction factors, so estimates of tidal flushing must be viewed as overestimates unless there is good reason to believe that the mixing in both regions is close to complete.

Freshwater runoff

Freshwater runoff is relatively easy to measure or calculate but is subject to errors arising from the physical structure of the inlet. If the water is well mixed the flushing rate is simply the amount of runoff divided by the volume of the inlet. However, the situation is quite different if the water is stratified. If a lens of freshwater lies on top of salt water so that the deeper salt water is effectively decoupled from the water surrounding the farms, it will be considerably more efficient at removing nutrients, although nutrients that do penetrate into the deeper saline layer will have a considerably longer residence time.

The situation tends to be very complicated in fjords with sills, since the residence time of water below the level of the sill may be extremely long and diffusion may play an important role in determining nutrient levels at depth.

Since freshwater runoff varies seasonally, it is important to use relevant values and not annual averages. Unfortunately the peak growing season usually coincides with the period of minimum stream discharge which can be up to two orders of magnitude lower than mean annual discharge, so the value of runoff as a mitigating agent for nutrient buildup may be low.

Wind mixing

Flushing by the action of wind-driven wave action is important but difficult to model because of its variable and episodic nature. Wallin and Håkanson (1991) have carried out a statistical analysis of inlets along the Baltic Sea, and their results show that the flushing is related both to exposure (the relationship between the area of the aperture connecting the inlet to the sea and the volume of the inlet) and to geometric factors describing how far away waves can originate and over how wide an angle, which is basically the concept of fetch. It is important to develop a structural model based on the actual mechanisms to incorporate this type of flushing into the model.

Typical calculations

The equilibrium solution of eqn 3 is

$$C = N/FU \tag{4}$$

and can be used to estimate the elevation of nutrient concentrations caused by fish farms (or any other source of nutrient input), given the limitations on the model described above. However, the issue is not what the concentrations are, but whether these are likely to prove environmentally unacceptable.

At present it is very difficult to say how much enrichment is acceptable in the types of inlets in which finfish farming is practiced. In the absence of a clear consensus on this, we have made some calculations on the basis of a reasonable but frankly arbitrary assumption that an increase in nitrogen levels of 1 μmol-N per litre is the maximum acceptable value. Slightly different calculations were made by Silvert (1994) and Cranston (1994) using data from Gregory et al. (1993) with different interpretations of the tidal flushing values.

Some of the results are shown in Table 2. There are three columns of holding capacities (expressed in tonnes of salmon per inlet) listed. The first number is the holding capacity calculated by looking only at tidal exchange under the assumption of complete mixing both inside and outside the inlet (Silvert 1994). The second column, also from Silvert (1994) shows the result of the calculation when flushing due to freshwater runoff is included. The third column shows similar calculations made by Cranston (1994) using a different form of the tidal flushing model, but also assuming perfect mixing. The very close agreement with the first column shows that the models are consistent with each other and indicates that the underlying theory is robust and not sensitive to details of the calculation. In practical terms there is no real difference between the first and third columns.

Summary

Although there remains a great deal of research and development work yet to be done to improve our understanding of the environmental impacts of fish farming in marine waters, models exist that can help us assess these impacts and make reasonable management decisions. It is possible to estimate both the local benthic impacts of a farm and the inlet-wide holding capacity given presently available models. We can hardly expect the exponentially exploding field of mariculture to pause for several years while we refine the models described above. Further research is clearly needed, but to ignore existing scientific methods until they have been perfected and thoroughly tested seems an irresponsible and ill-advised alternative to proceeding with their use while acknowledging their inherent weaknesses.

Table 2

Name of Inlet	Tidal	Runoff	Cranston
Annapolis Basin	30460	42948	30600
Bedford Basin	1873	1953	1876
Blacks Harbour	195	306	192
Canso Harbour	241	271	240
Country Hbr/Isaacs Hbr	2319	2508	2327
Indian Harbour	1082	1158	1081
Letang Harbour and vicinity	7735	9969	7755
Liverpool Bay	797	1081	798
Mahone Bay	23790	24820	23870
Passamaquoddy Bay	37137	43022	36971
Pennant Bay	2669	2818	2667
Pubnico Harbour	2772	3729	2793
Saint John Harbour	10039	18662	10077
Shelburne Harbour	2555	2966	2536
St Croix River	15906	20999	15909
St. Margaret's Bay	16939	17385	16950
Strait of Canso	3132	3252	3132
Sydney Harbour	3543	3776	3545
Wedgeport and vicinity	32285	41910	32343
Yarmouth Harbour	1266	1926	1268

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Author's address: W. Silvert, Habitat Ecology Section, Bedford Institute of Oceanography, PO Box 1006, Dartmouth, Nova Scotia, Canada B2Y 4A2