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MODELLING BENTHIC IMPACTS OF ORGANIC ENRICHMENT FROM MARINE AQUACULTURE

by

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ABSTRACT

Hargrave, B.T. [ed.]. 1994. Modelling benthic impacts of organic enrichment from marine aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

This report consists of six chapters that present models, calculations, and field observations of organic matter loading and benthic impacts due to salmon net-pen aquaculture. The contributions by different authors arose from a workshop held at the St. Andrews Biological Station, Department of Fisheries and Oceans, St. Andrews, N.B., on May 13, 1993, to discuss data obtained from observations and modelling of environmental conditions at sites used to culture Atlantic salmon in net-pens in coastal areas of New Brunswick and Maine. Subjects considered include evaluating the carbon budgets for different kinds of feed and different food conversion ratios, elemental composition of food and faeces and faecal pellet sinking rates, development of a model to evaluate the amount of carbon accumulation and benthic deterioration under fish farms, direct measurements of sedimentation and model estimates of the benthic deposition of organic particulate matter under and near fish cages, modelling benthic oxygen supply and consumption in relation to organic matter input, empirical models of changes in sediment geochemical and pore water variables as a result of increased organic matter loading, and estimates of the oxygen demand due to fish farming and the total nutrient loadings in an inlet based on total fish production. Detailed examples are included from salmon net-pen aquaculture sites in Maine. The information provides a review of current approaches to modelling the environmental impacts of fish farms. The model results and empirical observations may be used to provide a standardized basis for predicting potential benthic impacts at different sites where aquaculture facilities presently exist or may be established.

RÉSUMÉ

Hargrave, B.T. [ed.]. 1994. Modelling benthic impacts of organic enrichment from marine aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

Le présent rapport se compose de six chapitres décrivant des modèles, des calculs et des observations de terrain concernant l'accumulation de matières organiques et les effets benthiques occasionnés par l'élevage du saumon en cages de filets. Les contributions des divers auteurs découlent d'un atelier tenu le 13 mai 1993 à la Station de biologie du ministère des Pêches et des Océans de St. Andrews (N.-B.) pour discuter de données provenant de l'observation et de la modélisation des conditions des milieux d'élevage du saumon en cages de filet dans des secteurs côtiers du Nouveau-Brunswick et du Maine. Il y est notamment question de l'évaluation des bilans du carbone dans diverses sortes d'aliments et à divers taux de conversion des aliments, de la composition élémentaire des aliments et fèces ainsi que des taux de plongée des boulettes fécales, de l'élaboration d'un modèle d'évaluation de l'accumulation de carbone et de la détérioration du benthos sous les exploitations piscicoles, des mesures directes de la sédimentation et de modèles d'estimation du dépôt de particules organiques dans le benthos, sous les cages à poisson et aux alentours de celles-ci; on y traite aussi de la modélisation de l'approvisionnement et de la consommation du benthos en oxygène par comparaison avec l'apport de matières organiques, de modèles empiriques des changements qui surviennent dans les variables concernant la géochimie sédimentaire et les eaux interstitielles par suite de l'accumulation de matières organiques, d'estimations de la demande en oxygène imputable à la pisciculture et de l'accumulation totale de matières nutritives dans une baie par rapport à la production totale de poisson. On présente des exemples détaillés provenant d'exploitations de salmoniculture en cage du Maine. L'information fournie permet un survol des méthodes actuelles de modélisation des impacts environnementaux des piscicultures. Les résultats de la modélisation et les observations empiriques peuvent fournir une base standard de prédiction des impacts benthiques possibles en différents endroits où sont ou seront implantés des établissements aquicoles.

PREFACE

Mariculture of shellfish and finfish in coastal environments offers the promise of increasing supplies of food from marine ecosystems which cannot be expected through expansion of traditional harvest fisheries. However, recent studies have shown that these areas have a finite ability to support sustained aquaculture yield (Gowen and Bradbury 1987; Folke and Kautsky 1989). There is a potential for eutrophication associated with the release of soluble nutrients and particulate food and faecal matter from aquaculture activities in enclosed coastal embayments. Nutrient additions from aquaculture operations must be considered in the context of the entire aquatic ecosystem, water and sediments, to assess the relative magnitude of nutrient supplies from various sources (Håkanson and Wallin 1991; Makinen 1991). Environmental information is also needed for optimum siting and decisions concerning possible impacts of expansion of existing aquaculture operations in coastal areas (Pillay 1992). Physical conditions such as maximum and minimum water temperature, water depth, and current speed and biological factors such as the potential for the spread of pathogens, impacts on wild stocks, and stimulation of toxic plankton blooms are now recognized as important environmental variables that can limit aquaculture activities in coastal waters.

Observations of environmental factors necessary to ensure the continued development of marine shellfish and finfish aquaculture have recently been summarized (Håkanson et al. 1988; Hoffman 1991; Pillay 1992). Of several possible negative effects, deposition of particulate matter as egested particles from suspension-feeding molluscs and as unconsumed food and faecal matter from finfish pen culture have been identified as having potential long-term negative environmental impacts. Enhanced organic matter accumulation under fish pens can lead to anoxic sedimentary environments with associated depletion of benthic macrofauna (Weston 1990) and increased benthic fluxes of dissolved oxygen and inorganic nutrients (Hargrave et al. 1993). Altered geochemical conditions, such as lower oxidation reduction potentials and associated accumulation of sedimentary sulfides at culture sites, would also affect sediment-water exchanges of trace metals, although studies to examine these impacts have not been reported. Additives such as antibiotics, food supplements, and antifouling agents are also often used in aquaculture of some finfish species; and the deposition/accumulation of these compounds below aquaculture sites could alter the structure and metabolic activity of sediment microbial communities. Although, in general, organic enrichment through discharges of domestic sewage and industrial wastes to coastal waters and sediments is more widespread than the release of waste products from aquaculture facilities, the latter may be a significant source in small, less-populated embayments.

Mass balance studies of carbon, nitrogen, and phosphorus fluxes through salmonid fish farms (Håkanson et al. 1988; Hall et al. 1990; Gowen et al. 1991; Holby and Hall 1991; Hall et al. 1992) have shown that from 20% to 70% of these elements supplied as food to caged salmon can be deposited as unconsumed food pellets and faeces under the pens. Consumption of food pellets by wild stocks of fish and invertebrates can reduce the amount of organic matter that accumulates in underlying sediments. Benthic microbial metabolism and anaerobic sulfate reduction also increase with high rates of organic matter supply under marine fish cages to further reduce organic matter accumulation (Holmer and Kristensen 1992). However, these biological processes are not always sufficient to limit benthic organic matter accumulation, especially in areas where hydrographic conditions and/or low current speeds result in low rates of oxygen supply to the sediment surface. Recent observations in Maine coastal embayments show that the benthic responses to organic enrichment through enhanced sedimentation from salmon net-pen aquaculture is site specific, spatially limited, and highly dependent on physical factors such as water current speed and seasonal storm-related resuspension (Findlay et al. 1994).

Although biogeochemical effects of benthic organic enrichment arising from the aquaculture activities cited above have been identified in site-specific studies, only recently have attempts been made to derive general or generic models to link numerous physical, chemical, and biological factors that are altered by increased organic matter supply to sediments (Lumb 1989; Stewart et al. 1990; Silvert 1992). Recently, on May 13, 1993, a workshop was convened at the St. Andrews Biological Station, Department of Fisheries and Oceans, St. Andrews, N.B., where some recently developed models of benthic impacts due to salmon net-pen aquaculture were reviewed to determine their generality for prediction of effects of organic loading under various aquaculture sites. This report summarizes information presented at the workshop supplemented by contributions made afterward. The aim in publishing the report is to present the information in a timely way for use in developing guidelines that might serve as a basis for management decisions concerning the suitability of a coastal site for new or expanded salmon aquaculture development.

The report consists of six chapters that discuss quantitative approaches to modelling benthic impacts of salmon net-pen aquaculture. Acronyms and abbreviations used commonly throughout all chapters are described in the table below. The chapters are organized first to present theoretical approaches to modelling benthic impacts of organic enrichment followed by site-specific studies to summarize observations as empirical statistical models. The first two chapters describe generalized approaches to model benthic deposition and impacts of organic matter loading. Chapter 1 presents a general model for calculating Benthic Carbon Loading (BCL) from a single net-pen with discussion of the cumulative effects of several farms and comparison of calculations of oxygen demand by farmed fish and sediments under a pen, as well as a detailed treatment of the calculation of holding capacity. Chapter 2 reviews conceptual approaches used to model the horizontal displacement of sedimenting particles arising from deposition of uneaten food and fish faeces and develops an improved model which takes into account changes in current speed with depth, variable bathymetry, and a range of particle settling velocities.

Chapters 3 and 4 provide a transition from generalized model formulation to testing model prediction with empirical data. A Benthic Index (BI) based on impacts due to BCL and benthic deterioration due to accumulated organic carbon over time is described in Chapter 3. The performance of BI in predicting benthic conditions at 23 net-pen systems in Maine is evaluated by comparison of BI values with a semi-quantitative benthic score reflecting increasing levels of benthic enrichment. Chapter 4 presents results from studies within three salmon net-pen facilities in Maine coastal waters to determine directly quantitative relationships between salmon production and waste generation, sedimentation of waste food pellets and faeces, and the ability of an underlying benthic community to assimilate deposited organic matter.

The final two chapters use empirical data to compare particulate organic carbon sedimentation, sediment accumulation, and organic carbon burial rates with geochemical variables in sediments and porewater. In Chapter 5 a Benthic Enrichment Index (BEI) as the product of sediment oxidation reduction potentials and organic carbon content in surface sediment is shown to be correlated with organic carbon sedimentation. Chapter 6 presents empirical data that correlate organic carbon burial derived from sediment accumulation rates and organic carbon content with depth gradients of ammonium and sulfate in sediment pore waters. Both chapters compare data from present-day aquaculture sites with coastal, continental shelf, and open-ocean regions not subject to high rates of organic carbon loading.

This presentation of models, calculations, and empirical observations of benthic impacts of finfish aquaculture provides a summary of modelling approaches and data. The information may be useful for answering management questions concerning site selection and expansion of current facilities. Many of the calculations provide a standardized basis for modelling the environmental impacts of organic matter and dissolved nutrient loading associated with aquaculture development.

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TABLE OF ACRONYMS AND ABBREVIATIONS

Acronym/Abbreviation	Description	Chapter
A_{fp}	bottom area onto which waste food/ faecal pellets settle	4
AC	accumulated organic carbon (g C m^{-2})	3
BCL	benthic organic carbon loading (g C m^{-2} d^{-1})	1
BD	benthic deterioration index (arbitrary units)	3
BEI	benthic enrichment index (Eh in mv x mol C m^{-2} in upper 1 cm)	5
BI	BD (scaled 0 to 4)	3
BS	benthic score (scaled 0 to 4)	3
C	current speed (cm s^{-1})	1
CBR	organic carbon burial rate in sediment column (g C m^{-2} d^{-1})	6
D	water depth below bottom of a pen (m)	4
C_{fd}	organic carbon in food added pen^{-1} (g C)	4
C_{fc}	faeces produced pen^{-1} (g C)	4
C_{wf}	waste food produced pen^{-1} (g C)	4
F	weight of food added pen^{-1} (g)	4
$FLUX_{wf}$	predicted rate of organic carbon deposited as waste food (g C m^{-2} d^{-1})	4
$FLUX_{fc}$	predicted weight of organic carbon deposited as faeces (g C m^{-2} d^{-1}))	4
G	fish growth rate (g $fish^{-1}$ d^{-1})	1

TABLE OF ACRONYMS AND ABBREVIATIONS (Cont.)

Acronym/Abbreviation	Description	Chapter
GC	fish growth rate as organic carbon ($g\ C\ fish^{-1}\ d^{-1}$)	1
IC	ingested organic carbon converted to growth by fish ($g\ C\ fish^{-1}\ d^{-1}$)	1
IF	fraction of food ingested by fish d^{-1}	1
MLW	mean water depth at low tide (m)	4
% WF	percent of waste food	4
S	settling speed ($m\ s^{-1}$)	1
SF	surplus food (TFxIF)	1
SOC	sediment organic carbon ($mol\ cm^{-2}$)	5
SR	organic carbon sedimentation rate from the water column ($g\ C\ m^{-1}\ d^{-1}$)	5
SR_{fp}	settling rate of food and faecal pellets ($cm\ s^{-1}$)	4
TF	total feeding rate ($g\ fish^{-1}\ d^{-1}$)	1
V	average current velocity ($cm\ s^{-1}$)	4

CHAPTER 1

MODELLING BENTHIC DEPOSITION AND IMPACTS OF ORGANIC MATTER LOADING

by

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ABSTRACT

Silvert, W. 1994. Modelling benthic deposition and impacts of organic matter loading, p. 1-18. *In* B.T. Hargrave [ed.]. Modelling Benthic Impacts of Organic Enrichment from Marine Aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

Detailed descriptions are provided for evaluating models of the environmental impacts of organic loadings from fish farms. These include evaluating the carbon budgets for different kinds of feed and different food conversion ratios, estimating the benthic deposition of organic particulate matter under and near fish cages, estimating the total nutrient loadings in an inlet on the basis of total fish production, and determining the oxygen demand due to fish farming. Detailed examples are included. These calculations should provide a standardized basis for modelling the environmental impacts of fish farms.

INTRODUCTION

Calculation of the benthic impacts of aquaculture involves both determining the loadings and understanding the biological processes which arise from these changes in the benthic environment. This chapter presents the basic theory of how loadings are calculated, including a general theory of benthic deposition that requires only minimal information on currents. Chapter 2 describes a much more detailed model of benthic loading that can be used if complete time-series of current velocities are available, and Chapter 3 develops a theory of how benthic impacts evolve over time under the influence of benthic loading.

This chapter consists of three parts: first, a general model for calculating Benthic Carbon Loading from a single fish farm or cage; second, a discussion of the cumulative effects of several farms within close proximity to each other; and third, a brief discussion of the calculation of the oxygen demand of farmed fish and its relationship to benthic oxygen demand.

PART I. CALCULATION OF BENTHIC CARBON LOADING

The calculation of Benthic Carbon Loading actually involves two distinct models. The first is a Total Carbon Flux model which calculates the total amount of organic carbon released in particulate form from a cage or farm site. The second is a Benthic Distribution model which is used to compute the distribution of these particulates on the bottom under and near the site.

While there exist empirical relationships relating the Total Carbon Flux to other quantities, there are conceptual advantages to deriving it from the basic carbon budget. Strictly empirical relationships are seldom robust and maybe invalidated by something so simple as a change in the type of feed.

The major carbon transfers are shown in Figure 1:

$$\begin{tikzcd} & & \text{Respiration} & \\ \text{GROSS FEED} \arrow[r] & \text{Ingestion} \arrow[r] \arrow[d] & \text{Assimilation} \arrow[r] \arrow[u] \arrow[d] & \text{GROWTH} \\ & \text{Surplus Feed} & \text{Faeces} & \end{tikzcd}$$

Generally the dominant source of particulate organic carbon is surplus feed, but the faeces are also significant. The magnitudes of these fluxes are not well known, and estimates of surplus feed (the amount which is never ingested by the fish) range from below 5% to over 30% as discussed by Findlay and Watling (1994) in Chapter 4. Therefore, it is advisable to check this figure by constructing a carbon budget, which provides an estimate of the amount of feed ingested.

It is commonly assumed that of the ingested organic carbon, 20% is incorporated into fish growth, 20% is excreted as faeces, and the largest fraction, 60%, is respired (Gowen and Bradbury 1987; Silvert et al. 1990; Gowen et al. 1991; Bergheim et al. 1991). Since the growth of farmed fish is closely monitored, the budget can best be constructed by working from the growth measurement, as shown in Table 1. The parameter values used are taken from the literature for the most part, but most of them are not well known and should be treated with caution.

Table 1. Calculation of surplus feed.

---

0) Start with growth G, measured in g $fish^{-1}$ d^{-1}.

1) Convert to carbon equivalent. If we assume that whole fish are 8% carbon on a wet weight basis, then $GC = 0.08 \times G$, measured in g C $fish^{-1}$ d^{-1}.

2) If 20% of the ingested carbon (IC) goes into growth, then $IC = GC/0.20$.

3) The carbon content of feed is quite variable; but if we assume a figure of 45%, the amount of ingested feed (IF) is $IF = IC/0.45$, measured in g $fish^{-1}$ d^{-1}.

4) The surplus feed (SF) is simply the total feed (TF, generally a known quantity) minus the ingested feed, or:
$SF = TF - IF = TF - (0.08/(0.20 \times 0.45)) \times G = TF - 0.9 \times G$

Although this calculation seems straightforward, the parameter 0.9 which is obtained from the carbon content of fish and feed as well as the growth efficiency of the fish is very uncertain and probably cannot be determined with an accuracy of better than 10 or 20%. Because the surplus feed is determined by subtracting two terms of comparable magnitude, it is very sensitive to this parameter; and therefore the calculation should be verified by field observation whenever possible.

For example, suppose that we have a growth rate of 10 g d^{-1} for fish fed 15 g d^{-1} of feed, a Feed Conversion Ratio of 1.5. The calculation above gives $SF = 15 - 0.9 \times 10 = 6$ g d^{-1}, but a 10% increase or decrease in any of the parameters which go into the parameter value of 0.9 (say increasing it to 0.99) changes SF by about 15%. The lower the Food Conversion Ration, the more sensitive the calculation of surplus feed. The calculation of the amount of organic carbon in faecal pellets is very similar, since it is proportional to Ingested Carbon, which is calculated in Step 2 of Table 1.

Given the Total Carbon Flux calculated from Surplus Feed, as shown in Table 1, and faecal carbon, which is computed as 20% of Ingested Carbon, we can model the benthic distribution of this flux. Although there are different ways of doing this, all are variants of the basic Gowen model, as described in Chapter 2 (Gowen et al. 1994). The underlying calculation is that a particle that settles at speed S in water of depth Z takes time T = Z/S to reach the bottom, and during that time a current of speed V will displace the particle a distance D = VT = VZ/S. If $<V>$ is the mean current speed, then the mean displacement $<D> = <V>Z/S$, and the particles that originate within a cage will be distributed over an area that can be thought of as the shadow of the cage blurred by this amount. The details of how this area is calculated differ from model to model, although the exact value is not really very meaningful and thus it does not make very much difference which calculation is used. If we assume that the current direction, and hence the displacement, varies, then this blurred shadow will have a larger area than the cage, and then the Benthic Carbon Loading will be reduced by a factor which is the ratio of the shadow area to the area of the cage.

One way of calculating the area of the shadow which does not depend on the specific geometry of the cage is as follows. Consider a point which is a distance R from the centre of the cage. If a particle is released into the cage at this point and is displaced a distance D = VZ/S before reaching the bottom, its distance from a point under the centre of the cage will be:

$$R' = \sqrt{(R+D\cos\Theta)^2 + (D\sin\Theta)^2} = \sqrt{R^2 + 2RD\cos\Theta + D^2} \qquad (1)$$

as shown in Figure 2.

If we assume that the direction of the current is random, then the mean of $\cos\Theta$ is zero. Thus the mean value of $(R')^2 = R^2 + D^2$. If the area of the cage is A, then the edge of the cage is approximately $0.6\sqrt{A}$ from the centre (the exact relationship depends on the geometry; for a circular cage $A = \pi R^2$, so $R = 0.56\sqrt{A}$, for a square cage R ranges from $0.5\sqrt{A}$ to $0.7\sqrt{A}$ depending on whether we measure R in the middle of a side or at a corner). Thus the distance of a point on the edge of the shadow is approximately given by:

$$(R')^2 = (0.6\sqrt{A'})^2 = 0.36A' = R^2 + D^2 = (0.6\sqrt{A})^2 + D = 0.36A + D^2 \qquad (2)$$

or roughly $A' = A + 3D^2$.

It may be easier to visualize this by considering just a circular cage. In this case $A' = \pi(R')^2 = \pi(R^2+D^2) = A + \pi D^2$, which is effectively the same since in this crude calculation the difference between π and 3 is unimportant.

There are in fact much more important factors to take into account in using this kind of model. The first is that the assumption of a uniform settling speed is rarely justified. While feed pellets are uniform and fall at the same speed (although this depends on the size and constitution of the pellets), many of the pellets are partially consumed and through a combination of sloppy feeding and physical decomposition are broken into a range of sizes down to fine particles. Thus a calculation in which S is the settling speed of complete pellets will underestimate the area of the shadow.

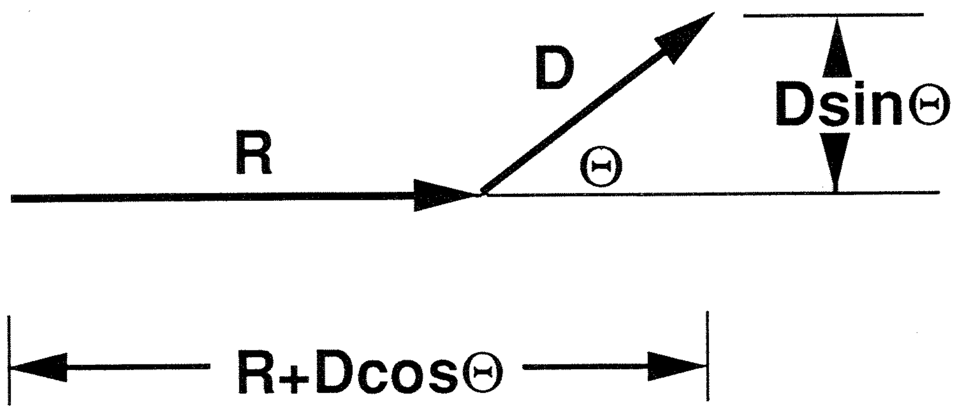

Figure 2. Geometry of deflection of a particle released at a distance R from the centre of the cage that is moved a distance D in an arbitrary direction by the current.

Another questionable assumption is that the currents are random, which is used in analyzing Figure 2 when we conclude that the mean value of $<\cos \Theta>$ is zero. If we consider a situation where the current is actually constant, then the entire shadow will simply be displaced a distance $D = VZ/S$ from under the cage and will have exactly the same area as the cage. We can resolve this difficulty by expressing the vector current as $V = V_m + (V-V_m)$ where V_m is the time-averaged mean current and we define $<V> = \sqrt{<(V-V_m)^2>}$ to calculate the random displacement of falling particles.

The third major problem with this calculation is that the organic carbon flux into the shadow area is not uniform. The actual flux is more likely to resemble that shown in Figure 3. Directly under the cage the benthic flux is almost the same as what comes out of the cage (i.e., the flux in g m^{-2} d^{-1} measured at the bottom is almost the same as what is measured at the cage itself), while towards the edge of the cage the benthic carbon loading falls off over a distance comparable to D. In a typical situation if the settling speed $S = 10$ cm s^{-1}, the depth $Z = 20$ m, and the mean current (or its variable component) is $\langle V \rangle = 5$ cm s^{-1}, the mean displacement $\langle D \rangle = \langle V \rangle Z/S = 10$ m, which is comparable to the size of a salmon pen.

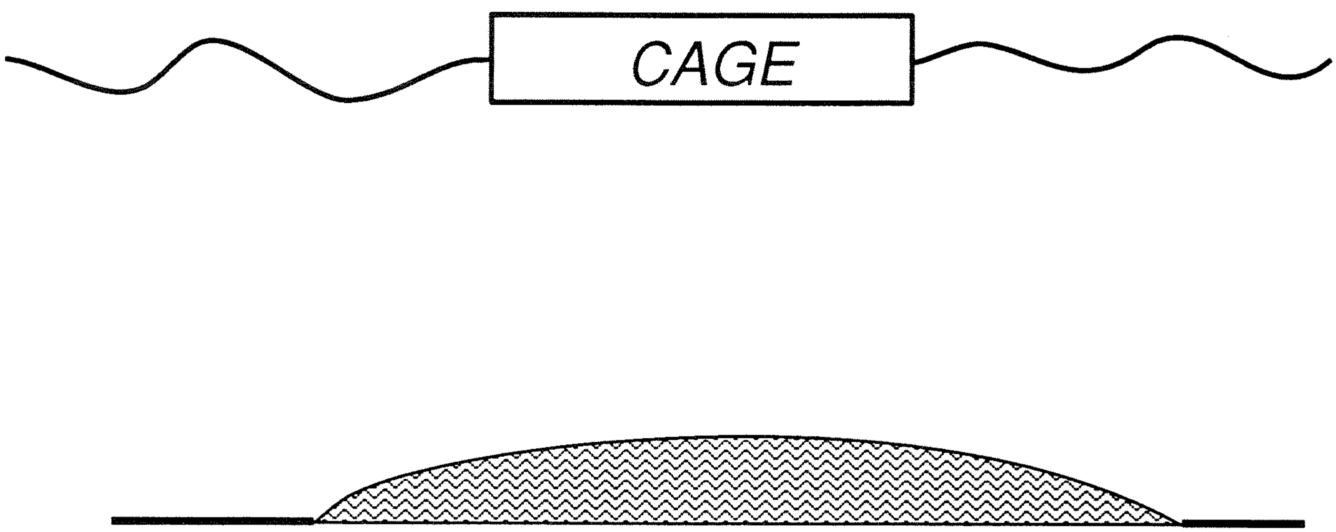

Figure 3. Representation of the variation in sediment accumulation under and near a cage site.

One way of visualizing the calculation of Benthic Carbon Loading is to imagine the fraction of time that particles are falling into a sediment trap. Directly under the centre of a large cage the trap will receive pellets whenever the fish are fed, although the pellets may originate in different parts of the cage. Under the edge of the cage the trap will receive pellets only half of the time, and so on. Thus the assumption of the uniform Benthic Carbon Loading is unrealistic and should be treated as a working assumption, since the loading of the entire depositional area is approximated by the average loading.

The calculation of Benthic Carbon Loading due to faecal pellets involves more complications than feed pellets. For one thing, it is not even clear that faeces should be treated as pellets at all. Field observations by divers indicate that faeces are generally extruded in the form of mucous strings which drift in the water column and sink very slowly. This could explain the difficulty in collecting intact faecal pellets under pens reported by Findlay and Watling (1994) in Chapter 4. It has also been reported that fish sometime mouth these faecal strings and eject them in a way that breaks them into smaller particles. It has also been suggested from diver observations that faeces may accumulate in the bottom of the pen and be released in a pulse during storm events (B. Hargrave, pers. comm.). Because of this, the faecal contribution to the Benthic Carbon Loading is very difficult to estimate, although in principle it can be calculated in the same way as the feed pellet contribution, but using a much smaller settling speed.

Because the calculation of the shadow area depends on the settling speed, the Benthic Carbon Loading must be calculated by adding together contributions from particles of different sizes. There is uncertainty of the proportion of waste food and faecal pellets in settled material, as discussed in Chapter 4. For the present calculation we assume that the dominant contribution comes from surplus feed, so only one term in the calculation is needed.

SAMPLE CALCULATION

A sample calculation of the Benthic Carbon Loading under a farm site will illustrate not only the calculations described above, but also the kinds of additional assumptions that are frequently needed to carry out the computations.

Suppose we need to estimate the Benthic Carbon Loading under a farm with a licensed production of 20 t y^{-1}, located in 20 m of water with a mean current of 5 cm s^{1}. We may or may not have access to data on the feed type and consumption, or the configuration of cages at the site.

The calculation of Total Carbon Flux is similar to that shown in Table 1. Since the licensed production is 20 t y^{-1}, we probably have to assume that this is also the actual production, which means that the amount of carbon converted into fish is GC = 0.08 $\times$ 20 t y^{-1} = 1,600 kg C y^{-1}. If this is 20% of the Ingested Carbon, the IC = GC/0.20 = 8,000 kg C y^{-1}. We may have detailed information on the feed used; but if we do not we might use the previous assumption that the feed is 45% carbon, so that Ingested Feed is IF = IC/0.45 = 18,000 kg y^{-1}.

If the Total Feed is known, then we can calculate the Surplus Feed by the formula SF = TF - IF. If we do not have this information, we may have to estimate the Total Feed. Using a Food Conversion Ration of 1.5 we get TF = FCR $\times$ Growth = 1.5 $\times$ 20,000 kg y^{-1} = 30,000 kg y^{-1}, so SF = TF - IF = 12,000 kg y^{-1}. 40% of the feed being wasted is a high value; but a well-run farm would probably keep statistics on how much feed was purchased, so that this crude way of estimating TF would not be necessary.

The area of the cages may be known; but if it is not, it can be estimated in the following way. New Brunswick has guidelines specifying a maximum production per unit volume of 18 kg m^{-3} y^{-1}; and since most of the pens are 5 to 6 m deep, this corresponds to an areal production of 100 kg m^{-2} y^{-1}. Thus a farm with a licensed production of 20,000 kg y^{-1} would require cages with a total area of at least 200 m^2.

The water depth is specified as 20 m; but it seems reasonable to assume that the feed is not strongly advected while falling through the cage, and therefore the appropriate depth to use is the mean depth under the cage, or roughly 15 m. Using a settling speed of 10 cm s^{-1} we thus get $<D> = <V>Z/S$ = (5 cm/s) × (15 m)/(10 m s^{-1}) = 7.5 m. The area of the shadow is thus at least $A' = A + 3D^2$ = (200 m^2) + 3 × (7.5 m)^2 = 370 m^2.

Finally we have a surplus feed estimate of 12,000 kg y^{-1}, which with a carbon content of 45% gives 5400 kg C y^{-1} distributed over an area of 370 m^2, or 15 kg C m^{-2} y^{-1}. This works out to be 40 g C m^{-2} d^{-1}, which is high but within the range of values reported from marine finfish aquaculture sites (see Table 1 of Chapter 5, Hargrave 1994). This is not surprising, given some of the assumptions made in the calculation. Of these the most drastic is the use of a high feed conversion ration, FCR = 1.5, leading to an estimate of 40% surplus feed. This demonstrates a serious problem in these calculations, in that the results are very sensitive to many of the parameter values commonly used. Even though it is possible to estimate Benthic Carbon Loading from very limited information, an elementary sensitivity analysis of the results can provide a convincing argument for direct measurement of some of the critical parameter values.

A further source of a possible overestimate of the Benthic Carbon Loading in this sample calculation is the assumption of a maximum stocking density in the cages of 100 kg m^{-2} y^{-1} annual production. This may be a necessary assumption if Benthic Carbon Loading is used to evaluate license applications in the absence of detailed site plans, but the potential bias needs to be recognized.

PART 2. CUMULATIVE EFFECTS OF SEVERAL FARMS

Coarse particulates such as surplus feed pellets fall mostly within 100 m or so of the cages, so their benthic impact is localized within the individual farm site. Soluble compounds and fine particles are transported by water movement, and the total load in the water column is the cumulative result of all the waste products released by farms in the area.

There are two ways of modelling these cumulative effects. The ideal approach is to use a hydrodynamic transport model to follow the wastes as they are released. A model of this type has been developed for Lime Kiln Bay in the L'Etang Inlet system under a contract supervised by R. Trites (to be published), and it is currently being used to track fish farm wastes and to compare their cumulative effects with those from other sources. Unfortunately, there are problems with this approach which limit its general utility as a method of assessing the environmental impacts of fish farms.

The most serious obstacle to the use of hydrodynamic models is cost, in both time and money. Each area requires development of a site-specific model, and the L'Etang model alone ran well over $100,000 and took several years to complete. As we gain experience in hydrodynamic modelling the costs and development times will certainly decrease, but it is likely that they will remain prohibitively high for general application for at least several years.

Even when hydrodynamic models are available, they are not perfect tools for evaluating environmental impacts. Such models have to use small space and time scales to be accurate, and it is difficult to use a model which tracks tidal currents with a time resolution of fractions of an hour to calculate average loadings over time scales of days or months.

An alternative approach to the estimation of cumulative effects is to use a simple flushing model to calculate the loading levels (Silvert 1992). In this model the water body is considered to be a well-mixed inlet with flushing time T. If the volume of the inlet is V, then the concentration C of dissolved or suspended material in the inlet is governed by the equation:

$$Dc/dt = I/V - C/T \qquad (3)$$

where the input term I includes all sources. More rigorously, C is the extent to which the concentration is elevated above natural levels; and the input term does not include mixing with outside water containing these natural levels.

In the steady state where $dC/dt = 0$, the equilibrium concentrations are given by $<C> = IT/V$. Although this is a far cruder result than what can be obtained by a detailed hydrodynamic calculation, the quantities T and V are relatively simple to estimate; and their estimated values for over 200 inlets in Atlantic Canada have recently been published (Gregory et al. 1992).

To illustrate this calculation, consider the estimation of the nitrogen loading of an inlet with a surface area of 4 km^2, a mean depth of 10 m, and fish production of 800 t y^{-1}, if the flushing time is 2 d. Because we are interested mainly in peak concentrations that could produce summertime bloom conditions, we should use peak summer nitrogen inputs rather than mean annual values. Simulation results (Silvert, to be published) confirmed by empirical calculation (S. Lall, pers. comm.) indicate that fish in the 3 to 4 kg range release about 0.1 mol-N d^{-1} of soluble nitrogen wastes. If the fish are harvested at about 4 kg, then the annual production of 800 t requires 200,000 fish, which would release about 20,000 mol-N d^{-1}. The resulting increase in nitrogen concentration is $C = IT/V = (20{,}000 \text{ mol-N d}^{-1})\times(2 \text{ d})/(4 \text{ km}^2\times 10 \text{ m}) = 1.0\ \mu\text{mol-N l}^{-1}$. This has to be added to the normal ambient level, so if typical summer values for nitrogen are 0.5 μmol N l^{-1}, the farms would triple this to 1.5 μmol-N l^{-1}, assuming that the additional nitrogen was not removed by increased primary production.

Although this sample calculation is very crude, it shows that there are many potential problems with the estimation of the cumulative effects of fish farms that cannot all be resolved by the use of more detailed hydrographic models. A major source of uncertainty is the calculation of the total nitrogenous inputs and knowledge of the nitrogen cycle in specific inlets, since this depends on several variables such as the type of feed and size distribution of the fish. Loss of nitrogen to the atmosphere has not been taken into account nor has the release of nitrogen from the enriched sediments under the cages. Given these uncertainties, the simple flushing calculation of water column loading is probably adequate for most purposes at this time.

This type of calculation can also be used to estimate holding capacity if one assumes that it is reasonable to regulate fish farms on the basis of estimated increases in nutrient concentrations. If we stipulate that farms should not be allowed to increase nitrogen levels more than 1 μmol N l^{-1} above natural levels, the above calculation could be used to establish that the holding capacity of the hypothetical inlet is 800 t, since that production of fish reaches the limit. More generally we can replace the steady-state solution of Equation 3 by:

$$H = (\Delta C/R)(V/T) \tag{4}$$

where H is the holding capacity, ΔC is the permitted increase in concentration (1.0 μmol-N l^{-1}), and R is the nitrogen release rate of the fish (R = 0.1 mol-N d^{-1}/4 kg = 0.025 mol-N d^{-1} kg^{-1}). A similar calculation has been carried out by Cranston (1994) in Chapter 6. A slightly more general approach is to include the freshwater input in the flushing calculation. In general the flushing time is given by V/Q where Q is the rate at which water is exchanged either through run-off or tidal exchange, so Equation 4 is equivalent to:

$$H = (\Delta C/R)Q \tag{5}$$

which basically says that the rate at which nutrients are added to the system, HR, is equal the rate at which they are removed, QΔC.

The holding capacity has been calculated for all of the 141 sites listed by Gregory et al. (1993) using both Equation 4 and Equation 5, and the results are shown in Table 2. The calculated holding capacities for 20 of these sites by Cranston (1994) in Chapter 6 are also shown and agree very well with the results of Equation 4. It is evident that the differences in the approximations and the inclusion of freshwater run-off introduces some differences in the two calculations. More importantly, it must be stressed that there are some very critical assumptions underlying these calculations, notably the assumption that all of the water that leaves the inlet is well-mixed and therefore carries out the ambient concentration of nutrients. Clearly these numbers must be treated with caution and sophistication; for example, it is absurd to think of growing 160 to 180 t of fish in the North West Arm of Halifax Harbour, a heavily polluted narrow inlet heavily used by recreational boaters! However, as a preliminary way of estimating the holding capacity of an inlet, the theory underlying Table 2 may be of value.

Benthic enrichment poses a different kind of problem, since fish farms release some fine particles which are light enough to travel large distances but which may settle out in times comparable to the flushing time of the inlet. We can modify the previous equation to give:

$$dC/dt = I/V - C/T - C/S \tag{6}$$

where S is the time it takes a particle to settle. In this case the steady state solution is:

$$<C> = ITS/(T+S)V(5) \tag{7}$$

and the rate at which material sediments out is:

$$V<C>/S = IT/(T+S) \tag{8}$$

or:

$$\text{Rate} = V<C>/SA = IT/(T+S)A \tag{9}$$

on an areal basis, where A is the area of the inlet ($V<C>$ is the total amount of suspended sediment in the inlet at any time).

The amount of organic carbon released directly from fish as fine suspended material is assumed to be small for calculations presented here. Although fine particulate matter may dominate material settled in sediment traps as discussed in Chapter 4, much of this material could come from disintegration of food and faecal pellets and not be directly released by fish. Since the amount of fine particulate matter release is unknown, it has been assumed to be small for the present calculation. Fish in the 3 to 4 kg range require about 1% of their body weight in feed d^{-1}, containing about 15% organic carbon, so it is unlikely that a fish would release more than 1 g C d^{-1} as fine particulates. Using the same parameter values as in the nitrogen loading calculation, 50,000 fish would release a maximum of 50 kg C d^{-1}, so if we assume that the settling time is the same as the flushing time (2 d), the mean carbon flux is:

$$\text{Rate} = IT/(T+S)A = (50 \text{ kg C d}^{-1})\times(1/2)/10 \text{ km}^2 = 0.0025 \text{ g C m}^{-2} \text{ d}^{-1} \tag{10}$$

which is negligible. Even if this amount were greatly increased by current patterns which concentrated the sediments in depositional areas, it seems unlikely that there could be enough accumulation of sediment anywhere but in the immediate vicinity of fish farms (due to the faster settling of larger particles) to have a significant impact on benthic productivity.

Table 2. Comparison of salmon holding capacity (t inlet[1]) in various embayment in eastern Canada (Nova Scotia and New Brunswick) calculated by Equations 4 and 5 in the text and by R. Cranston in Chapter 6 of this report.

Name of inlet	Equation 4	Equation 5	Cranston
Amet Sound	10738	12577	
Annapolis Basin	30460	42948	30600
Antigonish Harbour	784	1199	
Arichat Harbour	840	894	
Avon Bay	67858	152682	
Baie des Chaleurs	508241	522651	
Baie Verte	15754	18754	
Barrington Bay	5914	7101	
Barrington Passage	1763	2546	
Bay of Rocks	2325	2425	
Beaver Harbour	1326	1428	
Bedford Basin	1873	1953	1876
Blacks Harbour	195	306	192
Blind Bay	438	489	
Buctouche Bay	1616	2072	
Canso Harbour	241	271	240
Cardigan Bay	5053	5556	
Caribou Harbour	1324	1677	
Cascumpeque Bay	1865	2278	
Chedabucto Bay (Black Pt. to Cape Argos)	24000	24726	
Chedabucto Bay (Durell Is. to Guet Pt.)	66067	67746	
Cheticamp Harbour	210	240	
Chezzetcook Inlet	616	1002	
Chignecto Bay (Cap Enrage to Sand River)	260134	380063	
Chignecto Bay (Cape Chignecto to Martin Head)	605153	754478	
Clarke's Harbour	505	1044	
Cobequid Bay	70617	189549	
Cole Harbour	588	666	
Country Harbour	821	917	
Country Harbour / Isaacs Harbour	2319	2508	2327
Cumberland Basin	24941	57883	
Dover Bay	1339	1463	
Ecum Secum Inlet	564	641	
Fox Harbour	454	772	
Gabarus Bay	3383	3477	
Gegogan Harbour	713	802	
George Bay	78418	80385	
Glasgow Harbour	192	222	
Great Bras d'Or Inlet	2872	3016	
Green Bay	2612	2865	
Green Harbour	1018	1198	
Guysborough River	963	1086	
Halifax Inlet	10872	11379	

Table 2. Cont...

Name of inlet	Equation 4	Equation 5	Cranston
Hillsborough Bay	32786	38048	
Indian Harbour	1082	1158	1081
Inhabitants Bay	3049	3435	
Isaacs Harbour	219	258	
Jeddore Harbour	1885	2215	
John Bay	3272	3852	
Jordan Bay	3866	4306	
Jordan Bay / Green Harbour	7359	8127	
LaHave River	1910	2418	
Larry's River	1463	1610	
Lennox Passage East	2744	3066	
Lennox Passage West	2491	2746	
Letang Harbour	1918	2947	
Letang Harbour and vicinity	7735	9969	7755
Liscomb Harbour	1879	2190	
Liscomb Harbour / Gegogan Harbour	4359	4810	
Liverpool Bay	797	1081	798
Liverpool Bay and vicinity	4131	4536	
Lobster Bay	18839	25142	
Lockeport Harbour	2717	3092	
Lunenburg Harbour	2752	3107	
Mabou Harbour	341	432	
Mahone Bay	23790	24820	23870
Malpeque Bay	9783	10951	
Margaree Harbour	4	260	
Medway Harbour	1805	2545	
Merigomish Harbour	2108	2725	
Minas Basin (from Cape Chignecto)	1121799	1370088	
Minas Basin (from Cape Sharp)	580221	848376	
Mira Bay	5876	6287	
Miramichi Bay	36295	43018	
Molasses Cove	366	444	
Morien Bay	2106	2261	
Murray Harbour	1193	1558	
Mushaboom Harbour	1875	2110	
Musquodoboit Harbour	925	1667	
Necum Teuch Harbour	307	388	
Negro Harbour	854	2173	
Negro Harbour / Northeast Harbour	4754	5780	
New London Bay, P.E.I.	904	1067	
North West Arm (Halifax Harbour)	162	182	
Northeast Harbour	0	0	
Owl's Head	384	459	
Passamaquoddy Bay	37137	43022	36971
Passamaquoddy Bay & St. Croix River	73615	86185	
Pennant Bay	2669	2818	2667
Petpeswick Inlet	914	1132	
Pictou Harbour	2243	2679	

Table 2. Cont...

Name of inlet	Equation 4	Equation 5	Cranston
Popes Harbour	1043	1179	
Port Hebert	943	1226	
Port Joli	1947	2235	
Port La Tour	2544	3077	
Port Mouton	6238	6632	
Port Philip	252	704	
Portage Cove	33	37	
Porter's Lake	294	335	
Pubnico Harbour	2772	3729	2793
Pugwash Harbour	600	973	
Pugwash Harbour / Port Philip	1170	2039	
Quoddy Inlet	656	961	
Richibucto Harbour	1602	2091	
Rose Bay	1194	1342	
Rustico Bay	627	768	
Sable River	877	1288	
Saint John Harbour	10039	18662	10077
Sambro Harbour	717	790	
Shag Bay	376	427	
Shediac Bay	2350	2775	
Sheet Harbour	1882	2235	
Shelburne Bay	5329	6195	
Shelburne Harbour	2555	2966	2536
Shepody Bay	45380	107723	
Ship Harbour	681	812	
Shoal Bay	1566	1739	
Spry Bay	2259	2474	
St Croix River	15906	20999	15909
St Peters Bay, P.E.I.	630	724	
St. Anns Harbour	3357	3623	
St. Margaret's Bay	16939	17385	16950
St. Mary's Bay	185720	203817	
St. Mary's River	876	1206	
St. Peters Bay N.S.	830	925	
St. Peters Bay N.S. and vicinity	16648	17415	
Strait of Canso	3132	3252	3132
Summerside Harbour	1166	1765	
Sydney Harbour	3543	3776	3545
Sydney Harbour Northwest Arm	863	938	
Sydney Harbour South Arm	768	842	
Tangier Harbour	800	932	
Tatamagouche Bay	4714	5968	
Tor Bay	6700	7262	
Tracadie Bay	675	773	
Wallace Harbour	576	1080	
Wallace Harbour and vicinity	4838	5821	
Wedgeport & vicinity	32285	41910	32343
Whitehaven Harbour	1441	1558	
Wine Harbour	362	424	
Yarmouth Harbour	1266	1926	1268

PART 3. OXYGEN DEMAND BY FISH AND BENTHOS

Oxygen consumption by fish and oxidation of waste products are unlikely to lead to significant depletion of oxygen levels in well-mixed waters, but for completeness it is interesting to look at oxygen demand by fish and compare it with other oxygen fluxes.

Direct measurement of oxygen consumption indicates that a 4 kg salmon should consume about 10 g O_2 d^{-1}, but it is difficult to reconcile this with data on fish respiration. According to Bergheim et al. (1992) a fish respires about three times as much carbon as it converts into growth, so since a 4 kg fish uses about 3 g C d^{-1} for growth, it respires roughly 9 g C d^{-1} and would thus need 24 g O_2 d^{-1} for respiration alone. As a compromise, a figure of 20 g O_2 $fish^{-1}$ d^{-1} will be used, or 5 g O_2 kg^{-1} d^{-1}.

Given a maximum stocking density of 18 kg m^{-3} (assuming that the biomass and annual production figures are roughly the same), and usual cage depths of 5 to 6 m, we can assume an areal fish density of 100 kg m^{-2}. The corresponding oxygen demand is thus (5 g O_2 kg^{-1} d^{-1}) $\times$ (100 kg m^{-2}) = 500 g O_2 m^{-2} d^{-1} = 20 g O_2 m^{-2} h^{-1} in more usual terms. For a cage 10 m $\times$ 10 m containing 10 t of fish the total oxygen demand would be 2 kg O_2 h^{-1}.

In comparing these figures with Biological Oxygen Demand (BOD) for waste products and benthic oxygen demand, relative areas have to be taken into account. For example, heavily impacted bottoms may have demands of up to 400 mg O_2 m^{-2} h^{-1}, which is far less than the value of 20 g O_2 m^{-2} h^{-1} calculated above. However, if the impacted area extends 20 m beyond the cage, the total area is roughly 2500 m^2, and the total benthic oxygen demand would be 1 kg O_2 h^{-1}. This is significantly less than the oxygen consumed by fish respiration, but enhanced oxygen demand over a large area may be significant.

For calculations of the oxygen levels within cages it is essential to know the actual flow rates, which are affected by the structure of the nets, by fouling, and by the presence of fish. A detailed formalism for modelling some of these effects has recently been developed by Løland (1993), although there have been as yet insufficient field studies of the detailed hydrodynamics of aquaculture sites.

In ponds and other poorly mixed environments, oxygen depletion may pose serious problems. I hope that the present confusion about the oxygen consumption rates of fish will be resolved in the near future; but for the present, an oxygen demand of 100 to 200 g O_2 t^{-1} h^{-1} for the hourly oxygen demand per tonne of fish should be a reasonable approximation.

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CHAPTER 2

MODELLING THE SPATIAL DISTRIBUTION AND LOADING OF ORGANIC FISH FARM WASTE TO THE SEABED

by

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ABSTRACT

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Models describing the spatial distribution of particulate deposition from fish farms are reviewed and compared. These models do not allow for variable depth under the farm site and do not take into account possible variation in current with depth. A new model is presented which takes into account both realistic bathymetry and vertical gradients in current. A novel algorithm for simplifying the calculations is part of the model, which might otherwise be computationally impractical.

INTRODUCTION

A number of studies have demonstrated that intensive floating cage culture of salmonids generates large quantities of particulate waste in the form of uneaten food and faeces (Gowen and Bradbury 1987; Hall et al. 1990). The potential impact of this waste on the benthic ecosystem has been raised by environmental groups, government and non-governmental conservation, and wildlife agencies (Anon. 1988; NCC 1989). The potential for negative feedback, such that changes in the benthic environment may reduce the production potential of a particular site, has also been considered (Braaten et al. 1983). The changes in the benthos associated with particulate fish farm waste have been reviewed in detail by Gowen et al. (1992) and Mäkinen (1991) and include: alteration of the community structure of the benthic macrofauna (Brown et al. 1987; Ritz et al. 1989; Weston 1990); changes in aspects of sediment chemistry (Brown et al. 1987) including the accumulation of therapeutants in medicated feed (Samuelsen et al. 1988); and deoxygenation of sea-water overlying the enriched sediment (Tsutsum and Kikuchi 1983).

Understanding the effects of particulate fish farm waste on the benthos is the first stage in managing ecological change associated with fish farming. In order for regulatory authorities to set planning and discharge consents, however, a capability to predict the potential effects is essential. With respect to impacts on the benthic ecosystem, there are three levels of evaluation: assessment of topographic features on a regional scale of several kilometres (Håkanson et al. 1988); numerical modelling of the loading, settling, and dispersion rates of particulate waste using general information on water depth and current speeds (Hagino 1977; Silvert 1992); and numerical modelling of dispersion and loading based on site-specific water depth and current speeds (Gowen et al. 1989; Fox 1990; Weston and Gowen 1990). For many estuarine and fjordic coastlines, the use of topographic features is of limited value and will only permit generalisations regarding the accumulation or dispersion of waste to be made. The numerical modelling approach of Silvert (1994) based on loading rates, as described in Chapter 1, provides an opportunity to include a quantitative evaluation of benthic impact within a broader coastal zone management scheme (CZMS). Finally, such a CZMS may identify a requirement for a detailed EIA (Environmental Impact Assessment) for which the type of model developed by Gowen would be of use in providing a quantitative, site-specific prediction of dispersion and loading.

The purpose of this chapter is to review the utility of existing numerical models for predicting benthic loadings and suggest modifications to improve the ability of these models to predict the spatial distribution of particulate loadings that settle under fish cages.

OVERVIEW OF MODELS OF PARTICULATE DISTRIBUTIONS

The various models which have been developed to predict the dispersion and loading of particulate waste from fish farms all use the same conceptual approach.

These models are all based on a simple transport mechanism to calculate the horizontal displacement of particles as a function of water depth and current speed and direction (Fig. 1). Thus, the models discussed above only differ in detail as outlined below.

i) Hagino: This model utilises estimates of waste feed and faecal waste derived from experimentation, together with average current speed and direction. An important feature of this model is the use of a normal probability distribution of waste particle sizes based on the mean and standard deviation of measured particle sizes. The model appears to provide a good prediction of dispersion and loading when compared to a fish farm site (Hagino 1977).

ii) Gowen: This model uses approximate estimates of food wastage and faecal waste derived from dietary considerations (Gowen and Bradbury 1987) and separate, unique settling velocities for waste food and faecal particles. As it is presently formulated the model uses hourly mean values of current speed and direction recorded from a single depth over a spring-neap tidal cycle. Output from the model is in the form of a contour plot of organic carbon (g C m^{-2} d^{-1}). The model has been tested at a number of fish farm sites in Scotland (Gowen et al. 1988) and Puget Sound, Washington State (Weston and Gowen 1990).

iii) Fox: This model is based on a United States Environmental Protection Agency model to predict the dispersion of particulate waste from sewage treatment plant outfalls. Detailed current speed and direction data from a single depth are used; but in the model these data are analyzed to give a set of eight, 45° directional bins each with a mean speed. The model takes into account variation in bottom depth, a broad range of particle sizes, and post-depositional decomposition of sedimented carbon. The model is reported to give reasonable predictions of dispersion and loading which are similar to predictions obtained from the Gowen model. The model has been applied to a number of fish farm sites in Puget Sound, Washington State (Fox 1990).

iv) Silvert: This model incorporates a sub-model of fish growth to derive estimates of food consumption, wastage, and faecal output. Single settling velocities for waste food and faecal particles and mean current speed and direction were used in the original model (Silvert 1992), which has subsequently been refined and expanded as was described in Chapter 1 (Silvert 1994).

BASIC ASSUMPTIONS OF EXISTING BENTHIC MODELS

The models outlined above all have a number of implicit assumptions. It is important to recognize these, since otherwise there is a risk of drawing inappropriate conclusions.

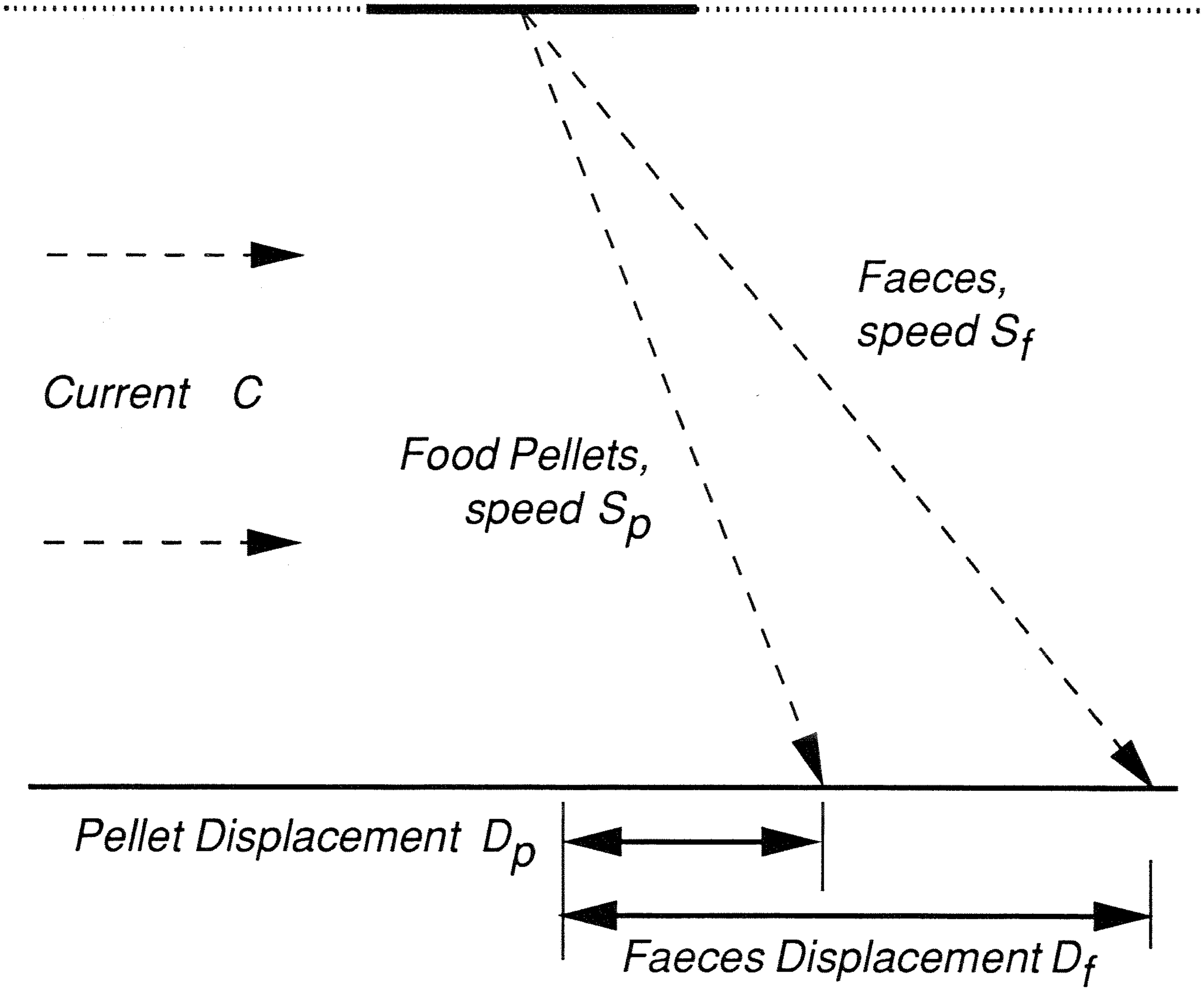

Figure 1. Representation of how earlier models of benthic deposition work. Particles settling at speed S take time Z/S to reach the bottom, where Z is the depth of the water. During that time a current of speed C transports them horizontally a distance CZ/S.

i) Post-depositional changes: Among the models discussed above, only the Fox model takes into account post-depositional changes in the form of decomposition of organic carbon. The model could therefore be used to predict changes in bottom conditions (Chapter 3, Sowles et al. 1994). None of the models incorporate resuspension and transport of material (but see Chapter 3), and this could invalidate many of the predictions. In one case Gowen et al. (1988) found uniform horizontal gradients in a number of variables (sediment redox potential, sediment carbon and dissolved oxygen in water overlying the sediment) beneath the farm. As a consequence Gowen et al. (1988) were unable to correlate predicted loading with the spatial distribution of these variables and concluded that high current speeds (up to 90 cm s^{-1}) were causing sediment resuspension and smoothing out any pattern in input to the sediment.

ii) Variation in bottom topography: Again, only the Fox model takes into account changes in the depth of the sea-bed in the vicinity of the cages. At many fish farm sites there is considerable variation in water depth, particularly when individual cage groups might be up to 100 m in length. At such sites the bottom slope may be as much as 1 in 5. Assuming that the sea-bed is of uniform depth is clearly an over simplification which could result in considerable error in the predicted pattern of dispersion and loading.

iii) Current speed and direction: The Gowen model, and to a lesser extent the Fox model, use detailed current data to predict dispersion and loading. In locations where flow is weak and may be influenced by wind (such as the upper regions of many coastal embayments) there is a considerable risk of erroneous predictions if models are run using limited time series current data. In general the reliability of any prediction increases with the amount of current data available.

The Gowen and Fox models use current data collected from a single depth, and none of the models take into account variation in current speed with depth. In river estuaries and fjordic embayments there can be a considerable reduction in flow with increasing depth, often with an abrupt reduction in flow where there is a strong, persistent pycnocline.

iv) Quantities of waste: Estimates of faecal waste can be derived from dietary considerations, which probably represents the simplest method since field measurements have a number of disadvantages. The use of a waste production sub-model (Silvert model) provides for changes in food consumption in relation to factors such as water temperature. There are no reliable methods for obtaining good estimates of food wastage and estimates are generally obtained from the food conversion ratio.

v) Settling velocity of waste particles: This has been approached in three different ways. The most simple approach (Gowen model) is to assume single settling velocities for waste food and faecal particles. A more sophisticated approach was adopted by Fox, who used a range of settling velocities. Finally, the Hagino model uses a normal distribution based on measurements of settling rates and the standard deviation of the measured rates to generate a range of settling velocities.

MODIFICATION OF THE GOWEN MODEL

Of the assumptions discussed above we consider that taking into account changes in current speed with depth, variable bathymetry, and a range of settling velocities would significantly improve the model predictions. Furthermore, these three modifications are amenable to computational solution. In this section we present one method of incorporating changes in current speed with depth and variable bathymetry into the Gowen model.

In the original Gowen model, the cage structure was represented by a two-dimensional grid which mapped waste material to a corresponding (larger) grid on the sea-bed, controlled by factors such as water depth, settling velocities, current speed, and direction. The modifications investigated keep the same basic structure.

Implementation of the variation in bottom topography has been achieved by assigning depth values to the bottom grid. Based on available knowledge of the area being studied, this may have varying degrees of detail. One possible approach is to represent the sea-bed by a tilted flat plane, which is of course simply a specific case of the more generalised form described above; but this offers only minimal computational advantages. The original model can obviously be recovered by setting the depths to a constant value.

The main difficulty in allowing for bottom topography and variable currents is the complexity of calculating where each particle strikes the bottom, since this involves finding the point of intersection of a curved trajectory and an uneven surface. We have solved this problem by working backwards from the bottom and solving for the location of the point of origin for particles which land at a particular spot on the bottom. Since the sea surface is a flat plane, this is much easier to calculate.

The disadvantage of this method is that the calculation has to be carried out for a larger area of the bottom than simply the area of the cage array. The detailed algorithm is as follows:

i) A matrix is constructed representing the bottom grid. The elements of the matrix represent organic carbon loading at that location.

ii) For each of the squares in the benthic grid the point-of-origin for particles falling at that location is calculated. For each step of the simulation one typically uses hourly values of current speed and direction.

iii) If the point-of-origin lies within the cage array, the corresponding element of the matrix is augmented.

iv) At the end of the simulation, the total value of each matrix element represents the total deposition at that grid location during a spring-neap cycle.

v) If more than one settling speed is used, the calculation represented by Steps ii and iii is repeated for each one.

Initial tests of this approach have shown that it can be implemented on a personal computer. Ways of further optimising the calculations, particularly using the speed and direction data, are being investigated.

The formulation for the displacement of a particle under constant velocity with depth (original Gowen model) can be given, for the back calculation method, by:

$$X_s = X_b - CZ/S \tag{1}$$

where X_s is the horizontal position of a particle at the surface, X_b the position of the point where the particle hits the bottom, C the current speed, S the settling speed, and Z the depth of the water over point X_b.

Variation in current flow with depth can be allowed for by interpreting Equation 1 in terms of infinitesimal displacements and integrating over the water column, giving:

$$X_s - X_b = -\int_0^Z C(z)dz/S \tag{2}$$

Again, as in the variable bathymetry extension, the original model is recovered if the velocity C(z) is constant (it is possible to allow for a variable settling speed with this approach too, but we have not felt it necessary at this stage).

It simplifies the calculation to use a standard functional form for the depth-dependent current speed C(z), although this is certainly not an essential assumption. C. Griffiths of Dunstaffnage Marine Laboratory (pers. comm.) has suggested that a power law equation of the form:

$$C(z) = C_s(z/Z)^m \tag{3}$$

gives a reasonable approximation of the standard tidal profile, where C_s is the current speed at the surface, z is interpreted as the depth above the bottom (where as before Z is the total depth), and m is an exponent with typical values in the range 0.15 to 0.2. This corresponds to a current profile which falls to zero at the bottom. Integrating this as shown in Equation 2 gives the result:

$$X_s = X_b - C_sZ/(m+1)S \tag{4}$$

which is very similar to the original Gowen model, except that the horizontal displacement for each particle is reduced by the factor $(1+m)^{-1}$, and of course the water depth Z is variable.

The functional form given in Equation 3 is simply one of many that may be used, and it is a reasonable approximation to the current flow in an unstratified water column when the depth does not change rapidly. In other situations the current profile has to be determined and integrated as shown in Equation 2. This is not always straightforward, since the necessary data are not always available; and it may be necessary to make a number of assumptions to obtain reasonable estimates of benthic deposition patterns.

Edwards and Sharples (1986) used river inflow and tidal kinetic energy mixing of freshwater to calculate an approximate depth for the surface mixed layer in Scottish sea-lochs. In most lochs the mixed layer depth was less than 10 m. In many fjordic estuaries there is a compensation flow of water at depth which balances the outflow of brackish surface water, which further complicates the situation.

At present there are no simple models which accurately predict changes in flow with depth, and the dependence of current on depth generally depends on a number of very site-specific factors (J. Loder and C. Hannah, Bedford Institute of Oceanography, pers. comm.). However, a simple modification of the Gowen model would be to assume a fixed pycnocline depth and treat flow in the two layers separately. In order to maintain a reasonable boundary condition of zero current on the bottom we can assume a constant current speed in the upper layer and a power law profile below the pycnocline similar to Equation 3. The result is a generalization of Equation 4, namely:

$$X_s = X_b - C_s(Z+mP)/(m+1)S \qquad (5)$$

where P is the depth of the pycnocline.

These alternative approaches do not predict dramatically different depositional patterns; using an exponent m of 0.2, the difference between the predicted displacement of a particle for Equations 1 and 5 would be less than 20%. Thus for weakly stratified flows with no strong current gradients, the original Gowen model, generalized for variable depth but with a constant current profile, gives a fairly good approximation. However, in cases where the current speed falls off significantly with depth, as might be the case if the bottom is very uneven and might have some very deep spots, Equation 1 could seriously overestimate the distance that particulates could be transported and therefore might significantly underestimate the benthic loading.

CONCLUSIONS

Simple models which predict the areal dispersion and loading of organic waste from floating cage fish farms are based on the principle of relating the dispersion of particles as a function of current flow and water depth. These models contain a number of assumptions which may not hold true, and hence there are varying degrees of error associated with each prediction. Despite these limitations all of the models discussed appear to give reasonable predictions of dispersion.

Of the assumptions discussed, changes in current speed with depth, variable topography, and a range of settling velocities are considered the most important and to be amenable to numerical solution. One method of incorporating changes in current speed with depth and variable bathymetry has been presented in this chapter. The next step will be to test these modifications against predictions made by the original model. A formal test to assess the performance of the modified model will require a detailed set of current measurements and supporting benthic data from a fish farm site.

To be of significant value as management tools it is necessary for these models to give a prediction of the effect of the loading rather than just the loading. Simple empirical relationships may exist between benthic variables such as sedimentary redox potential, the number of macrofaunal species, and organic loading which can be used to deduce the effect of a proposed development as discussed in Chapter 5 (Hargrave 1994). At the present time, however, there are no models which predict the effects of organic waste loading, that are of use as management tools.

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CHAPTER 3

THE EFFECT OF BENTHIC CARBON LOADING ON THE DEGRADATION OF BOTTOM CONDITIONS UNDER FARM SITES

by

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ABSTRACT

Sowles, J.W., L. Churchill, and W. Silvert. 1994. The effect of benthic carbon loading on the degradation of bottom conditions under farm sites, p. 31-46. *In* B.T. Hargrave [ed.]. Modelling Benthic Impacts of Organic Enrichment from Marine Aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

Twenty-three fish pen systems in Maine have been studied, and benthic conditions under these systems have been evaluated. A model has been developed to evaluate the amount of carbon accumulation and benthic deterioration under fish farms. An important feature of the model is that it deals with the dynamics of the interaction between the farm and the benthos and shows how bottom conditions change over time. There is a high degree of agreement between the predictions of the model and the field data. The model appears to provide a quantitatively useful tool to managers and regulators alike for estimating the potential benthic impacts of fish farming.

INTRODUCTION

The relatively recent arrival of salmonid pen culture to the east coast of the United States and Canada has presented new economic opportunities for many in the fishing industry. Traditional wild commercial stocks of pelagic and ground fish have shown a serious decline due to overharvest since implementation of the Magnusson Act in the United States and due to a combination of overfishing and adverse environmental conditions in Canadian waters. Fishing communities once dependent on these stocks welcome other means to support themselves. Aquaculture has established itself as one means of supplementing traditional fisheries while at the same time relieving pressure on depleted stocks.

With this new opportunity, however, come new challenges for the resource manager. Pen culture of salmonids has the potential to adversely affect surrounding waters through over-enrichment. Experiences on the west coast of North America (Weston 1986) and northern Europe (Gowan et al. 1988) suggest that pen culture operations can cause environmental problems such as anoxia of bottom waters (Rosenthal et al. 1988), displacement of benthic species (Lim and Gratto 1992; Rosenthal and Rangely 1989; Brown et al. 1987), noxious algae blooms, and release of toxic gases from sediments (Gowan and Bradbury 1987). This paper focuses on benthic impacts.

Environmental regulation of finfish pen culture in Maine and the Atlantic provinces of Canada, until recently, has been guided mostly by experiences from these other regions of the world having very different environmental conditions. Recognizing that the Bay of Fundy and the coast of Maine are dissimilar from these other regions, resource managers have relied on a combination of this "offshore" knowledge and professional judgement. However, the shifting regulatory environment resulting from uncertainty has been to some degree detrimental to both industry and resource management (Maine State Planning Office 1990).

While years of experience in water pollution management have afforded environmental regulators with an ability to predict and quantify impacts associated with various types and quantities of wastes, the simple algebraic equations normally used to predict the concentrations of specific constituents safely allowable in a receiving water for pipe discharges are inappropriate for pen culture permits. Most conventional waste assimilation or allocation models derive from simple dilution equations. Ambient concentrations of a "limiting" constituent are targeted to maintain existing or designated uses within the constraints of the receiving water's quality and flow. Aquatic organisms may be protected from acute or chronic toxicity effects, and human health may be protected from an unacceptable level of risk from exposure to pathogens by establishing an in-stream pollutant concentration target. Effluent concentration limits derived from demonstrated performance of practical technology are then diluted into a statistically derived "worst-case" in-stream low flow. The assimilation capacity developed is straightforward, especially in unidirectional systems such as large rivers with continuously monitored and statistically predictable flows. Once operational, the permitted concentrations and flows may be verified on a case-by-case basis by directly monitoring the discharge as it flows from the pipe.

Discharges from pen culture operations are quite different from discharges most environmental regulators are familiar with. In contrast to conventional land-based "pipe" discharges, waste loads of fish pen aquaculture operations are diffused over the entire operation area, move in three dimensions, and are not easily amenable to direct monitoring. Furthermore, as cause and effect relationships are obscured by unpredictable currents, depositional patterns, storms, and a rapidly evolving aquaculture technology, their impacts are not well understood.

Several predictive models have been developed in the past decade, but they are largely theoretical with little empirical verification. Interim siting guidelines were developed for Puget Sound aquaculture operations by relating potential benthic impact to horizontal current velocity, depth below pens, and annual production (Science Applications International Corp. 1986). In Maine, a predictive model (Maine Dept. of Environmental Protection 1988) adapted from the Puget Sound model was developed to rank potential associated benthic impacts of each operation. Silvert (1992) discussed the strengths and weaknesses of both approaches and proposed a non-linear model based on benthic flux as a more appropriate tool. The Maine model, while limited to ranking potential impact, served its intended purpose of enabling regulators to treating each operator equitably within the industry. Since the model was developed from a set of environmental conditions different from Maine, however, it was not considered capable of predicting actual impact. A new predictive tool reflective of Maine conditions was needed to provide an estimate of environmental risk.

In 1991, implementation of An Act Regarding Aquaculture (State of Maine 1991) provided an opportunity to test and refine Maine's model by requiring standardized environmental and operational monitoring of all net pen culture operations within Maine waters.

METHODS

SITE AND AQUACULTURE OPERATION VARIABLES

Twenty three pen systems were selected for study. Most pen systems were an array of contiguous net pens 6 m deep (Fig. 1); however, in some instances isolated 30 m diameter circles constituted a "system," and the total pen areas ranged from 690 to over 12,000 m^2. Information for each system was compiled for six operational and environmental variables: system age, water depth, mean current speed, areal feeding rate, and tidal range (Table 1). Because many sites were experimental during their first years of existence, age represented the number of years the site had been in "substantial" operation or in the case of small operations, steady state. System ages ranged from 1 to 10 yr with most systems about 4 yr old. Water depth was the average depth of water over which the pens were located measured at MLW. In some instances, bottom slopes were significant resulting in up to a 100% difference between shoal and deeper depths at a site. Depths ranged from as shallow as 6 m to as deep as 25 m. Current was estimated as the average speed near mid-water based on a combination of measurements (deployed current meters, window shade drogues, and professional judgement of divers). Current speeds ranged from 2.5 to 37.5 cm s^{-1}. Tidal range was determined from United States Coast and Geodetic Survey charts. Since most systems studies were in the Cobscook Bay area, tidal range was generally 6 m. Six sites located farther west had tidal ranges between 3 and 5 m. Feeding rate is defined as the weight of feed the operator recorded feeding over the 7-mo peak of the 1992 growing season, April though October, divided by the area of the pen system. Area is defined as the surface area or footprint of the pen array within the lease site. Since both moist and dry feed are used, feed was normalized to 100% dry weight using correction factors of 0.95 dry weight for dry feed and 0.65 dry weight for moist feed. Feeding rates varied considerably, from 15 to 90 kg m^{-2}; this reflects differences in stocking densities, year class, and husbandry.

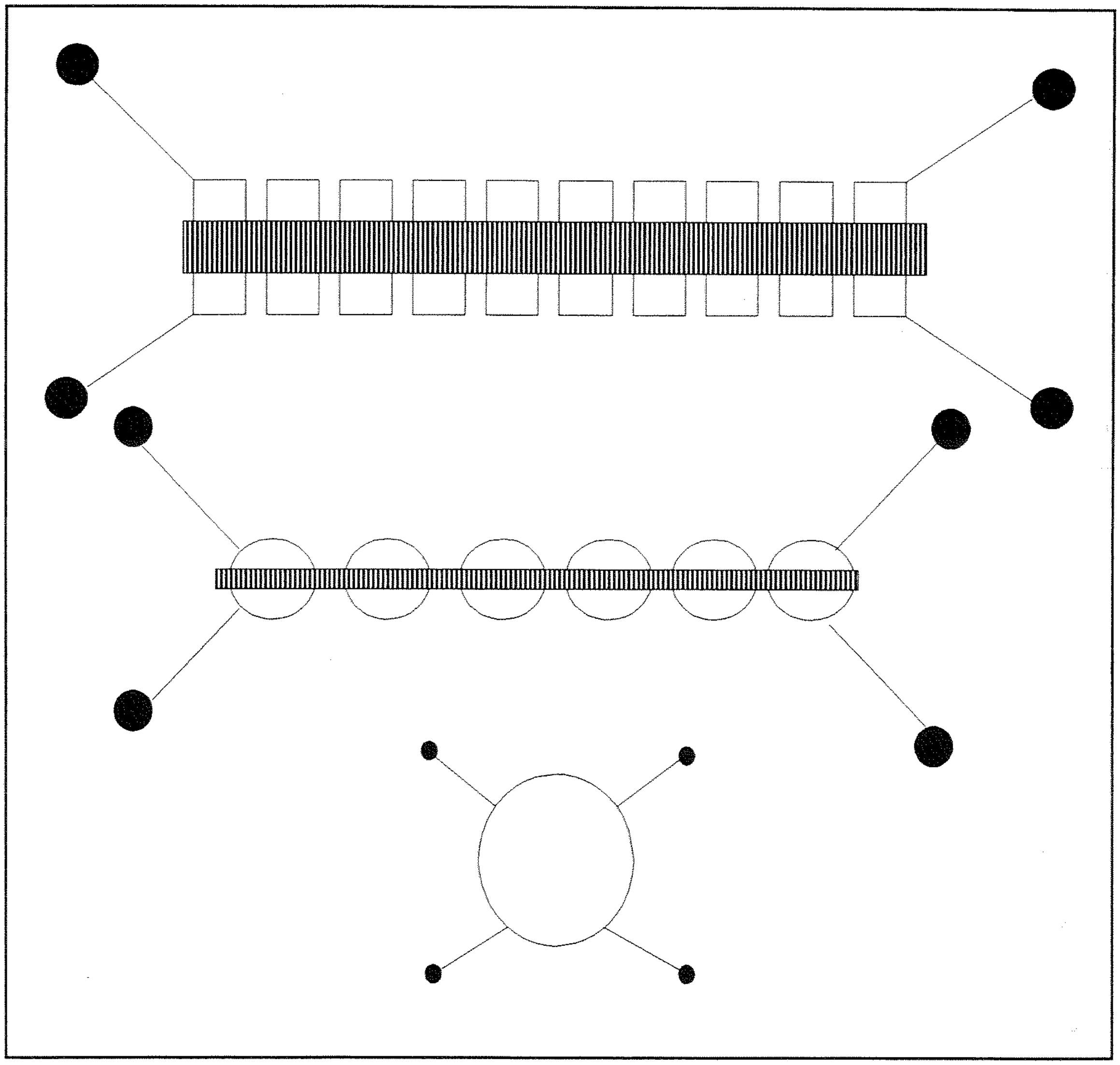

Figure 1. Standard pen configurations at sites in Maine.

Table 1. Operational and environmental variables used to develop benthic index (BI).

System	Year	Area (m^2)	Depth @ MLW (m)	Current ($cm\ s^{-1}$)	Tide (m)	Feed area^{-1} kg m^{-2} (7 mo)$^{-1}$	Mean score (BI)	Std. error of BI
1	1989	2304	8	8.0	7	38	3.75	0.50
2	1985	2880	9	18.0	7	48	3.25	0.50
3	1989	2880	15	18.0	7	57	2.88	0.25
4	1983	8640	15	12.5	7	29	2.83	0.29
5	1990	9000	15	17.5	7	40	1.83	0.29
6	1989	12130	12	3.0	4	87	2.83	0.76
7	1989	3375	6	37.5	7	60	1.75	0.50
8	1989	7200	11	35.0	7	58	0.75	0.50
9	1988	4602	12	35.0	7	38	0.50	0.58
10	1989	690	12	27.5	7	41	0.38	0.48
11	1990	920	9	17.5	7	15	0.63	0.48
12	1988	3140	8	5.0	4	69	2.13	0.25
13	1987	8190	6	4.5	7	54	2.75	0.50
14	1990	1222	12	30.0	7	27	0.00	0.00
15	1989	8550	11	27.5	7	55	0.63	0.48
16	1988	1728	8	12.5	7	55	1.63	0.48
17	1991	1000	14	30.0	7	34	0.17	0.29
18	1992	1215	12	5.0	4	36	1.67	0.29
19	1988	1620	6	2.5	5	79	2.50	0.50
20	1989	8316	15	2.5	3	40	2.17	0.29
21	1990	3905	25	4.0	3	74	1.00	0.00
22	1992	1347	9	10.0	7	23	0.67	0.58
23	1990	3150	11	15.0	7	30	1.67	0.58

BENTHIC SCORING

Benthic impact was assessed semi-quantitatively by visual observations by four professionals having direct knowledge and experience at each of the sites. Each site was scored on a scale of 0 to 4 where 0 equated to no perceptible difference from natural conditions and 4 to unacceptable benthic impacts. Scores of 1 to 3 reflected increasing levels of benthic enrichment based on type of impact and extent away from pen system. Unacceptable impacts were arbitrarily defined as any one of several conditions: azoic conditions or outgassing adjacent to or directly beneath the pens, *Beggiatoa* sp. mats, feed, and faeces build up extending more than 5 m away from pen footprint (the area of the bottom corresponding to the areal dimensions of the pen system), and hyper dominance of infauna extending more than 5 m away from the pen footprint. Raters scored each site based on a "composite" assessment of impacts they personally observed. The average of the individual scores was used as the benthic score for model development, and standard errors for the scores were also computed.

MODEL DEVELOPMENT

IDENTIFYING THE ROLE OF SITE AGE

Using the data collected in this survey, the Benthic Carbon Loading (BCL) for each site was calculated from the model described in Chapter 1 (Silvert 1994), and the calculated BCLs were compared with the Benthic Scores described in the previous section. The results of this comparison are shown in Figure 2; and although there is clear correlation, it is weak and the Benthic Carbon Loading (BCL) by itself is not a good predictor of the Benthic Score.

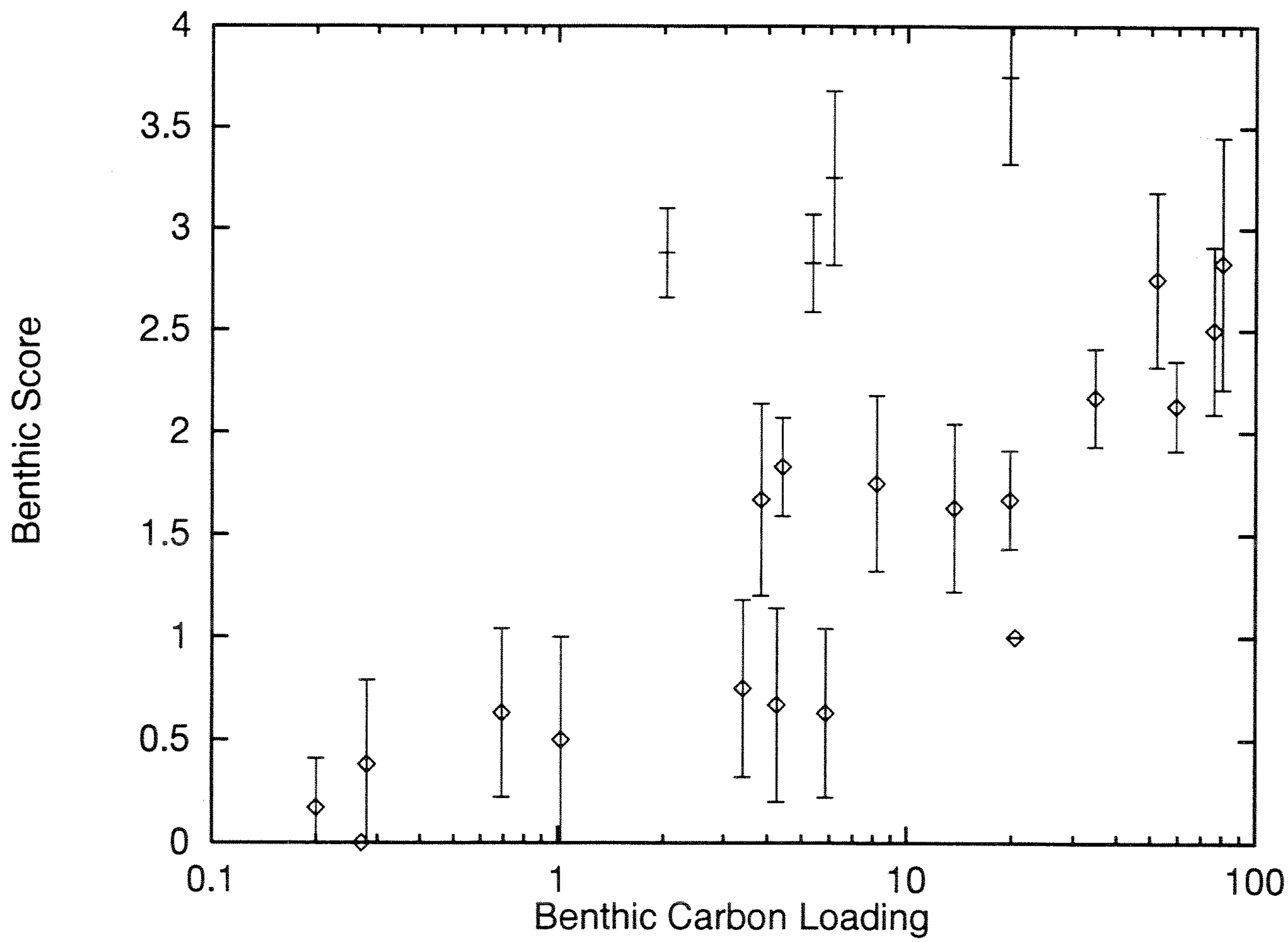

Figure 2. Benthic Scores plotted against Benthic Carbon Loading (BCL) (g C m^{-2} d^{-1}) as described by Silvert (1994) in Chapter 1 of this report. The sites represented by horizontal lines rather than circles are significantly older than the others or are in locations that have a long history of organic loading from other sources.

The inclusion of additional information about the sites greatly reduced the scatter about the regression line. Some of this information, such as that relating to husbandry practices, is confidential and cannot be released for publication. Other factors, such as the geometry of the site and proximity to headlands and other features which might affect the transport and settling of particulates, involve very site-specific considerations. Very few of the sites actually conform to any of the simple configurations shown in Figure 1; and although it is possible to modify the calculation of BCL to allow for different cage geometries, this type of detailed physical transport calculation is very labour-intensive and seemed inappropriate at the present stage of model development (see Gowen et al. 1994, Chapter 2 of this report, for details of this sort of calculation).

We noticed that points corresponding to older sites generally lay above the regression line, while points from more recently occupied sites were below it, which seems reasonable on biological grounds. We would expect that it would be the cumulative effect of several years of organic carbon loading that would have the greatest impact, and consequently that older sites would have higher scores for benthic degradation. Furthermore, among these older sites (represented by horizontal lines in Fig. 2) the Benthic Scores increased with BCL in the same way as with the newer sites, which suggests that there might be a limiting value to the score which depends on the loading. Based on these observations we proceeded to try to develop models which were both biologically reasonable and which agreed with the data.

MODEL FORMULATION

One type of common model which suggests itself in this situation is an uptake-clearance model of the form:

$$d(AC)/dt = (BCL) - k \times (AC) \qquad (1)$$

where AC is the Accumulated Carbon under the cage. The first term, (BCL), represents the input flux of organic carbon to the bottom, while the second term, $k \times (AC)$, represents loss by resuspension, microbial degradation, and consumption by fish and invertebrates (k can be interpreted as a combined resuspension + biological consumption rate). Although there are clearly many deficiencies in this model, particularly in the second term (these are discussed below), it seems to provide a reasonable starting point for the analysis.

By setting the derivative d(AC)/dt equal to zero we can calculate a limiting level of carbon accumulation, $AC_{lim} = (BCL)/k$, which is the level at which removal processes take away carbon at the same rate at which it is deposited (i.e., BCL). The Accumulated Carbon under a farm site at any time is:

$$AC = AC_{lim} \times [1 - exp(-kt)] \qquad (2)$$

where t is the age of the site and k is a constant. This function starts off at a value of zero when $t=0$ and increases asymptotically towards AC_{lim} as the age of the site increases.

Although it is reasonable to believe that AC (Accumulated Carbon) should be a better predictor of benthic impacts than BCL by itself, since it accounts for the length of time the bottom is impacted, a similar argument can be raised against it. We would not expect BCL to be a complete predictive variable, since it takes time for organic carbon to build up on the bottom and for adverse effects to occur. Similarly, the presence of a layer of organic matter on the bottom might not lead to instantaneous adverse consequences; it may take time for biological and chemical reactions to occur which lead to oxygen depletion, gas generation, and other deleterious effects. Consequently we have investigated the possibility that benthic impact is actually a second-order effect, driven by the existence of a layer of accumulated carbon in much the same way that organic carbon itself builds up as a result of loading.

If we represent the deterioration of the bottom by an uptake-clearance equation similar to Equation 1 we get:

$$d(BD)/dt = d \times (AC) - r \times (BD) \tag{3}$$

where BD is the degree of Benthic Deterioration and d is a degradation rate and r a recovery rate. This equation is based on the argument that the more accumulated carbon is present under the cages, the more rapidly the bottom deteriorates, and that this deterioration is balanced by some sort of recovery process.

The relationship between Accumulated Carbon (AC) and Benthic Deterioration (BD) over time is shown in Figure 3. It can be seen that AC begins to build up very rapidly as soon as the site begins operations, but that BD takes a while to begin to develop, since it is driven by AC and not directly by the Benthic Carbon Loading (BCL) itself; this is why we refer to benthic impacts as a second-order process.

In order to relate BD to the observed Benthic Scores described above, we have to deal with the constraint that the Benthic Scores are evaluated on a scale of 0 to 4, while the level of Benthic Degradation calculated from Equation 3 is unlimited. We have therefore arbitrarily mapped BD to a scale of 0 to 4 by using the transformation:

$$BI = 4 \times \sqrt{BD} \,/\, (1 + \sqrt{BD}) \tag{4}$$

to generate a Benthic Index (BI) lying between 0 and 4. Figure 4 shows the relationship between the observed Benthic Scores and the Benthic Index values predicted by this analysis. Comparison with the dashed line which corresponds to perfect agreement shows that this Benthic Index is a reasonable predictor of Benthic Scores as a measure of bottom conditions.

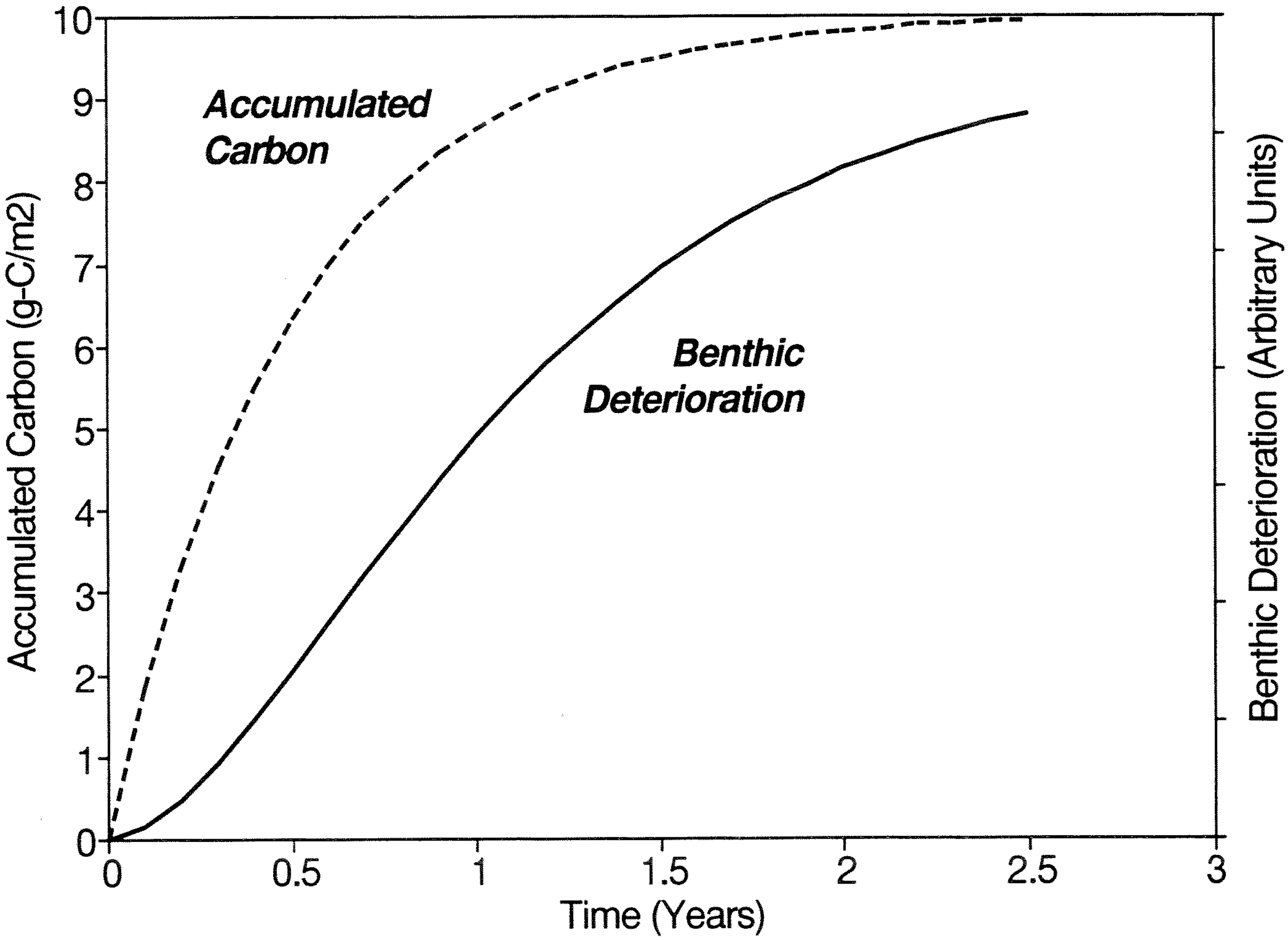

Figure 3. Comparison between dynamics of Accumulated Carbon (AC) and Benthic Deterioration (BD) based on the first and second order uptake-clearance models described in the text (Equations 1 and 3).

ANALYSIS AND INTERPRETATION OF THE MODEL

The results shown in Figure 4 indicate that a Benthic Index based on the second-order model of Benthic Deterioration is a reasonable predictor of benthic impacts, and that it might be worth refining the system of simple uptake-clearance models described in Equations 1 and 3. The greatest weakness in Equation 1, the first part of this model, is the set of assumptions that go into the clearance term, (AC)/k. We would not expect the rate of organic carbon removal to be strictly proportional to the amount present, since at low fluxes the natural grazers could probably remove any organic carbon that settles, while at high fluxes the amount sedimenting is likely to overwhelm natural removal mechanisms. The age of the carbon is also a factor, since the model does not distinguish between fresh food pellets and faecal matter on one hand, and aged buried

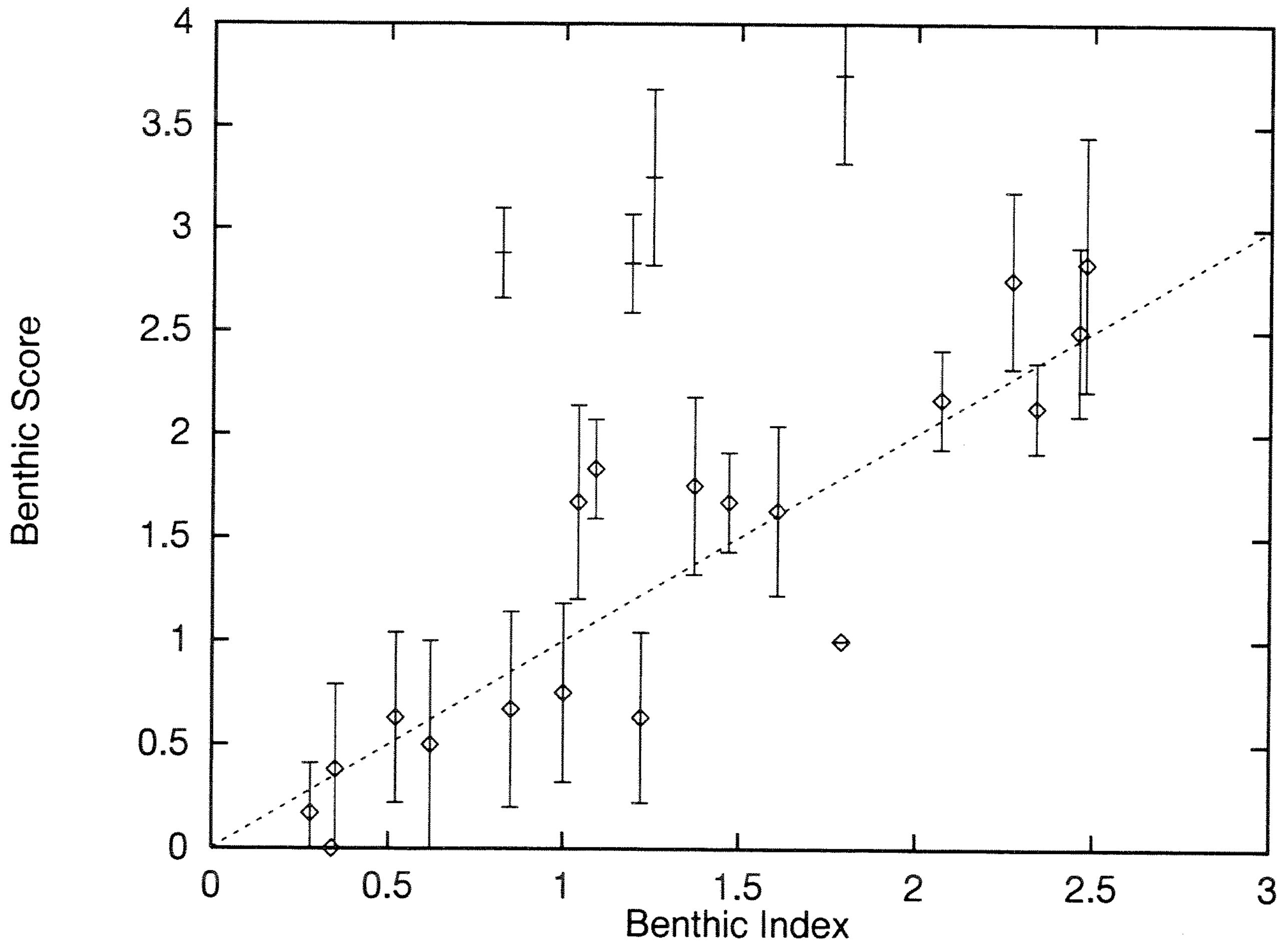

Figure 4. Benthic Scores plotted against predicted scores based on the uptake-clearance model. The four sites shown by horizontal lines above the other points represented by circles are considerably older or have different management histories than the others. The dashed line represents equality between predicted and observed scores.

organic carbon and *Beggiatoa* mats on the other. As for Equation 3 describing Benthic Deterioration, this is very speculative and we do not as yet have a direct measurement of any quantity which could be interpreted as a proxy for BD. However, the pattern of deterioration represented by the solid line in Figure 3 appears reasonable and appears to relate well to other, mostly anecdotal, information on bottom changes under fish cages. Certainly the basic shape of this curve, which suggests that it takes some time before serious benthic impacts are observed, seem reasonable to fish farmers and scientists with experience in this area.

Another weakness of the model, although one that might be more easily dealt with in the framework of this analysis, is that it assumes that the Benthic Carbon Loading is proportional to the amount of feed consumed. This ignores differences in feeding efficiency between farms, but it is known that significant differences exist. Information about these differences is often available to regulatory agencies, but it generally involves proprietary data and must be used with discretion.

DISCUSSION

The four "outliers" shown in Figures 2 and 4 might be explained by one or more of several factors unique to these sites. They are more heavily developed in terms of age and percent pen saturation of the lease area. The three most distant outliers are located within the same cove which has a history of organic loading from a fish processing plant. All four outliers are under the same ownership and management, suggesting that perhaps husbandry is a factor. And all four are unique in that they are sited within 25 m from other systems of similar size, and it is possible that the resultant density of pen structures reduces the actual currents in the vicinity of the cages. We tested this hypothesis by recalculating the predicted score using a current velocity of zero, which indeed moves the outliers to the right and reduces the disagreement between theory and experiment, as shown in Figure 5. However, the removal of current dispersion effects does not completely resolve the discrepancies, so it appears that other factors must be involved.

Although the limited data used in this study do not permit us to identify a definitive relationship between carbon loading and benthic impact, our model provides a good qualitative prediction that could be used as a tool for industry, regulators, and scientists alike. At present, it is the only means by which benthic degradation can be forecast for a specific pen culture operation. The forecast can be used in several ways. For new operations, environmental managers may use the model to design a baseline monitoring plan. A site/operation combination ranking high on the index may warrant a more intensive pre-start-up characterization than one ranking lower on the index. For existing operations ranking high, both spatial distribution and frequency of sampling might be increased at least until analysis of monitoring data supports a revision to the original monitoring plan. Given the scarce resources available to properly administer environmental and natural resource programs, the model enables managers to efficiently allocate their time and budget where it is likely to be the most effective. Rather than scrutinizing all operations at the same level of effort, those with the highest probability of causing an unacceptable impact would be monitored most closely.

From an industry perspective the model has perhaps even greater value. Knowing the regulatory environment in which permitting decisions are made can help avoid permitting delays by allowing an applicant to understand the decision-making process. The model may also benefit operators already in production in that the difference between the "expected" benthic score derived from the model and the actual benthic score observed through monitoring may reflect to some degree the efficiency of

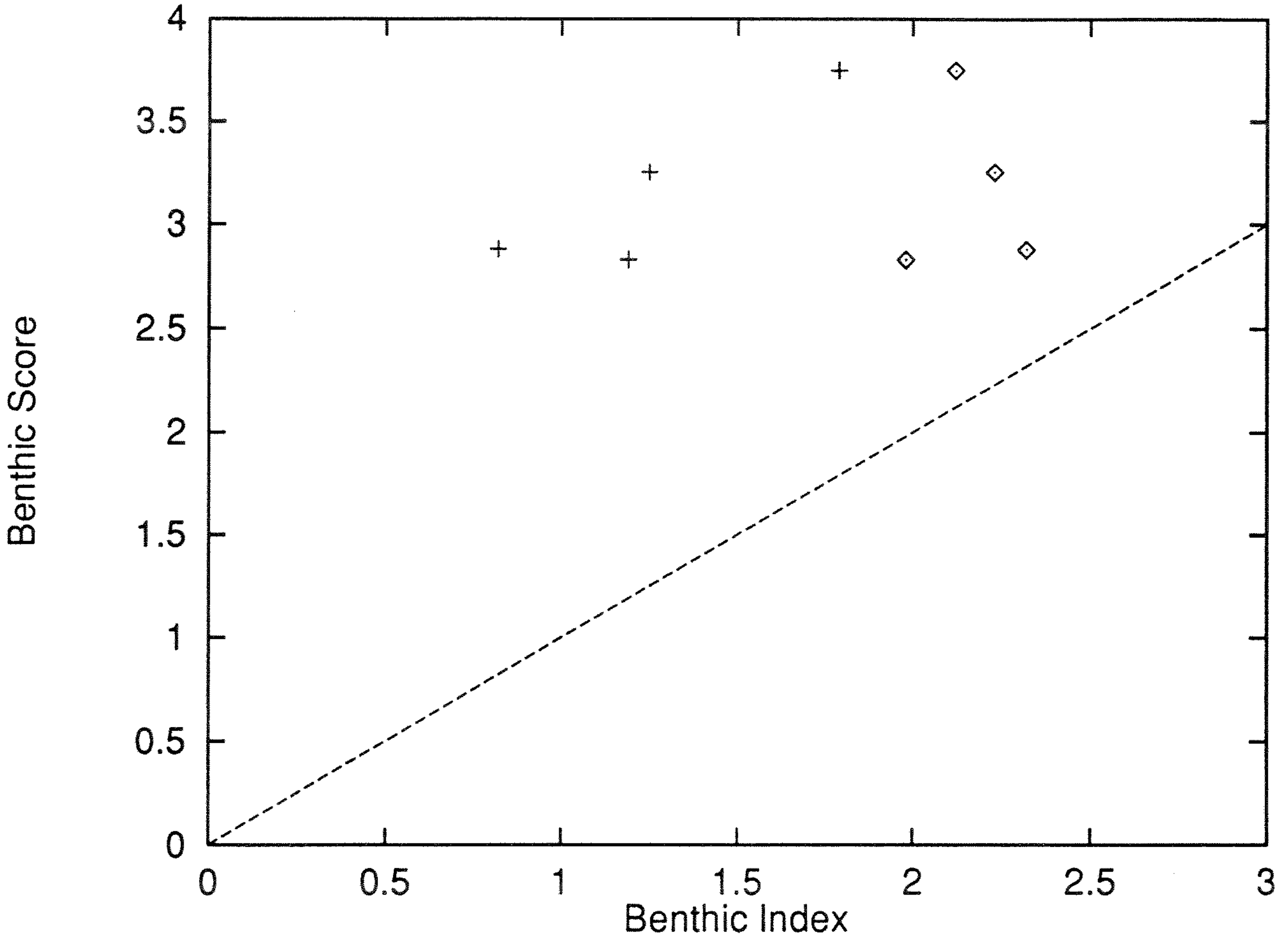

Figure 5. Effect of suppressing the dissipation of settling particles for the first four sites listed in Table 1. The crosses show the original positions of the points representing these sites, while the diamonds show the results obtained by setting the current speed equal to zero so that the depositional area is equal to the area covered by the pens. The Benthic Scores (Y-axis) are the same, but the predicted Benthic Index (X-axis) increases, which improves the agreement between Benthic Index and Benthic Score but does not fully resolve the discrepancies.

husbandry. Operations having higher observed impacts than predicted by the model might consider whether husbandry practices related to feed conversion is responsible. Clearly, converting more feed into fish flesh rather than enriching the benthic environment is desirable for all.

From a scientific perspective the model represents a step in the development of models describing the evolution of benthic impacts under continued exposure to benthic carbon loading. Already, in attempting to interpret and explain the outliers, several issues have arisen and provided quantitative information about the influence of husbandry on benthic impacts, the importance of pen configuration on organic deposition and clearance rates, and the effect of pre-existing and off-site near-field organic discharges.

This model is an important first step toward understanding the relationship between finfish pen culture operations, local environmental conditions, and benthic "quality." In the process of the model's development, inadequacies were revealed in sampling and data reporting protocols which offered valuable feedback on the monitoring program. The model appears to provide a reasonable prediction of benthic impacts associated with pen culture and is a substantial improvement over the formula used prior to its development. However, it is not a replacement for field verification and should be used with discretion. The model should be seen as a first step, one which we feel has been successful; and it shows promise for further refinement and offers a good basis for future research.

CONCLUSIONS AND RECOMMENDATIONS

Open-water net-pen aquaculture offers opportunities for fishermen attempting to harvest declining wild fish stocks. However, environmental managers and especially regulators have pointed out that environmental impacts must be minimized if productivity in the new industry is to be sustainable. Regulation has been largely arbitrary and inappropriately treated in the same manner as conventional industrial discharges. Consequently, the regulatory environment has been unpredictable. In 1991, the Maine legislature enacted an aquaculture law requiring development and implementation of a standardized monitoring program.

First-year monitoring results from 23 operations in downeast Maine have enabled us to develop an initial model which predicts the amount of benthic organic enrichment. Unlike site specific studies detailing local processes, this model is more robust and especially applicable to regulators who deal with coast-wide regulation and monitoring of an entire industry. The model may also be used by industry management to assess husbandry performance and scientists to formulate and test hypotheses.

Inherent in the development of any model lies the possibility of inappropriate application. We caution regulators and policy makers to use the model in conjunction with field verification monitoring before any decisions are made.

This initial work shows the potential for fruitful research into the development of a more refined benthic impact model through a more standardized data collection. We recommend that aquaculture monitoring programs focused on these environmental and operational conditions in collaboration with research programs be used to build the next generation models.

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CHAPTER 4

TOWARD A PROCESS LEVEL MODEL TO PREDICT THE EFFECTS OF SALMON NET-PEN AQUACULTURE ON THE BENTHOS

by

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ABSTRACT

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Particulate organic carbon flux to sediments was measured at several coastal Maine salmon production facilities by deploying sediment traps proximal to salmon net-pens. Several parameters critical to modelling the waste stream originating within the net-pens were also evaluated. Organic carbon and nitrogen content and settling rates of salmon feeds and faeces were measured. We combined this information with numbers of food pellets collected in traps deployed beneath the pens to construct a simple mass balance model for the net-pen system. Predicted rates of organic carbon flux were two to ten times higher than rates actually measured. The response of the benthic community to increased rates of organic carbon deposition was examined by determining the changes in rates of benthic oxygen consumption and carbon dioxide production. Both rates were directly correlated to organic carbon sedimentation implying that particulate wastes originating within the net-pens were rapidly degraded on or within the sediments if an adequate oxygen supply was maintained. A comparison of the maximum rate of oxygen delivery (calculated from data for current speed and a standard diffusion model) to the oxygen demand generated by the waste stream allowed accurate prediction of the development of *Beggiatoa*-type mats, a common endpoint associated with deteriorated benthic conditions under salmon net-pens.

INTRODUCTION

Any successful model which will predict, *a priori*, the impact (or lack thereof) of salmon production on the benthos must consider three separate, but related, problems. They are: 1) predicting the amount of the waste generated within the pen on the basis of projected salmon production figures, 2) the distribution of the waste stream within the environment, and 3) the effect of the increased organic loading on the benthos. While other chapters within this report address the specific information necessary to solve each of these modelling problems, our goal was to provide empirical measures of some of the critical modelling parameters, plus rate measures that will allow evaluation of the predictive models. The determination of the waste pool associated with a particular production facility requires knowledge of the amount of food introduced into the environment and its distribution between food consumed and food wasted. Ingested food is further partitioned between food assimilated and fish faeces. Once the size of the waste pool is determined the next task is to determine its distribution (as a rate function) within the environment. The final task is to use the rate of organic matter enrichment to predict changes in the benthos. This requires some measure of the assimilative capacity of the benthos. Unfortunately, solid data on many important parameters are still unavailable, forcing the use of estimates based on laboratory studies or first principles.

As stated above, one of the goals of this research project was to provide empirical measurements of parameters critical to modelling the benthic impacts of salmon production in the marine environment. They are: 1) the carbon and nitrogen content of the commonly used feeds, 2) the carbon and nitrogen content of salmon faeces, 3) the sinking rate of the commonly used moist and dry foods, 4) the sinking rate of salmon faeces, and 5) the percentage of waste feed. The first four were obtained by direct measurements, and the fifth was estimated using the number of food pellets caught in sediment traps placed below net-pens. Not specifically addressed was the amount of food fed per unit of salmon produced (and the variation in this ratio). High-quality information regarding this ratio can be obtained from production records submitted to the State of Maine Department of Environmental Protection as presented in Chapter 3. In addition, we constructed a simple mass-balance model and used this model to compare predicted and measured flux rates. Finally, we attempted to estimate the assimilation capacity of the benthic community.

MATERIALS AND METHODS

STUDY SITES

We have studied five salmon net-pen facilities within Maine coastal waters; the results from three of these studies were utilized in this report. The sites selected for the variable conditions such as current velocity and sediment depositional characteristics were: 1) a depositional site in Eastport, 2) a erosional site in Eastport, and 3) two individual net-pens located on a single lease site in Toothacher Cove, Swans Island

(Table 1). Exact fish biomass figures are not yet available; but we estimate that the Eastport-depositional site contained approximately 250,000 kg, the Eastport-erosional site 41,500 kg, and the two sea-pens at the Swans Island site 15,000 and 7,200 kg of fish biomass, respectively.

Table 1. Summary of ambient physical and chemical characteristics and fish biomass of the three study sites.

	Study sites		
Measure	Eastport-depositional	Eastport-erosional	Swans Island
Depth (MLW, m)	11.9	11.4	14.3
Sediment type	muddy sand	poorly sorted gravel	muddy sand
Current Velocities[1]:			
Maximum (cm s^{-1})	35.8	52.3	17.6
Average (cm s^{-1})	12.6	21.5	2.1
>25 cm s^{-1} (% time)	8.7	61.7	0.0
Mass accumulation rate (g m^{-2} d^{-1})	7.29±1.29	14.45±1.70	3.87±0.59
Salmon biomass (kg)	250,000	41,500	22,200

[1] Measured 1 m above the sea floor.

SAMPLING

We attempted to sample monthly throughout the year, but rarely were able to complete sampling during the winter months. All samples were taken using diver-deployed cores and sediment traps. Not all sediment trap deployments were successful. During some sampling periods, inclement weather, disturbed or lost trap holders due to aquaculture and fishing activities, equipment failures, diver disorientation, and technician error prevented measurement of sedimentation rates as discussed below.

PHYSICAL AND PRODUCTION DATA

Recording current meters were placed 1 m above bottom at all sites and refreshed monthly. Water depths, sediment granulometry, temperature data, the number and weight of fish per pen, the number, size, configuration, and orientation of the pens in use, and the size and type of food being fed were (when possible) obtained from the operators of the net-pen facilities.

ORGANIC MATTER SEDIMENTATION

Organic carbon sedimentation was measured using cylindrical sediment traps. Traps were designed to minimize sampling biases, incorporating the latest technological advances (Butman, 1986; Butman et al. 1986; Baker et al. 1988), and have been designed to minimize sampling bias. The amount of material collected in the traps was determined gravimetrically, and total organic carbon and nitrogen content were determined. When possible, the total number of intact waste food and faecal pellets were counted prior to processing of settled material collected in sediment traps. Fresh faecal pellets were obtained using the same sediment traps, except they were suspended ≈0.5 m from the bottom of the net-pens using a rope bridle, no preservatives were used, and deployments were from 2 to 6 h.

ORGANIC CARBON AND NITROGEN DETERMINATIONS

All measures of organic carbon and nitrogen in settled material were made using a Carlo Erba elemental analyzer. Sediment samples were first treated with fumes of hydrochloric acid to remove any inorganic carbon (usually present as calcium carbonate). Organic carbon and nitrogen were also measured in wet and dry feed used by many growers in the Eastport-Lubec area. Values are reported as mg g^{-1} dry weight of feed.

BENTHIC OXYGEN CONSUMPTION

Benthic oxygen consumption was estimated by measuring the loss of oxygen from water overlying undisturbed sediment in cores over time. Undisturbed sediment cores were collected at sites of sediment trap deployment and immediately transported to shore-side facilities. Overlying water was sampled for initial dissolved oxygen concentrations using micro-Winkler titration, and the cores were sealed with air-tight caps such that approximately 400 ml of ambient sea water overlay the sediment. A small, caged magnetic stir bar was glued to the underside of the sealing cap and the overlying water was stirred at the lowest possible rate to avoid sediment resuspension. Cores were incubated at ambient temperature for 2 to 4 h (depending on temperature) in the dark. At the end of the incubation period the cores were unsealed and samples were taken to determine final oxygen concentration.

MICROBIAL END-STAGE METABOLIC PROCESSES

Benthic ΣCO_2 flux was measured simultaneously with benthic oxygen consumption using the method of Edmond (1970) as modified (where appropriate) by Bradshaw et al. (1981) to measure ΣCO_2. Samples of overlying water (120 ml) were removed prior to and after incubation and titration alkalinity determined. ΣCO_2 was calculated using the modified Gran equation.

RESULTS

ORGANIC CARBON AND NITROGEN CONTENT OF THE COMMONLY USED FEEDS

Table 2 gives the organic carbon and nitrogen content, and the C/N ratio for several commonly used salmon feeds. Feeds averaged 463 ($\pm$ 61) mg of organic carbon g^{-1} of feed; values range from 317 to 529 mg C g^{-1} dry weight of feed. Moist feed produced by Connors Brothers and used by many growers in the Eastport-Lubec area was the most variable. Feeds averaged 76 ($\pm$ 9) mg of nitrogen g^{-1} of feed; values ranged from 58 to 88 mg N g^{-1} dry weight of feed. Connors Brothers moist feed was again the most variable. The carbon to nitrogen ratio for these feeds was 6.11 $\pm$ 0.45.

ORGANIC CARBON AND NITROGEN CONTENT OF SALMON FAECES

Efforts to determine the organic carbon and nitrogen content of salmon faeces, as well as settling rates as discussed below, were hampered by the apparent rarity of intact faecal pellets. We deployed sediment traps within net-pens for 2 to 6 h on six occasions in an effort to collect faeces. In all cases, a fine flocculent material was collected; but in only two cases (April 30, 1992, and October 6, 1992) were intact faecal pellets recovered. Organic carbon and nitrogen content in faeces were reduced from values in food pellets by 50 to 70% and C/N ratios were increased from values of about 6 to 15 (Table 3).

SINKING RATE OF SALMON FAECES

An average sinking rate of 2.0 cm s^{-1} for salmon faeces was determined from intact faecal pellets collected in April and October by allowing one pellet from each sample to settle over a distance of 10 cm.

Table 2. Organic carbon and nitrogen content (mg g^1 dry weight) of several salmon feeds.

Feed	Date collected (mo/d/yr)	Carbon	Nitrogen	C/N
Connors	Not recorded	528.6	82.3	6.42
Connors	12/19/91	351.5	55.4	6.34
Connors	01/29/92	488.6	76.2	6.73
Connors	02/24/92	317.0	57.6	5.49
Connors	10/08/92	474.2	80.0	5.87
Connors	07/17/93	466.6	78.4	5.95
Ecoline (3)	Not recorded	443.1	74.5	5.95
Fundy's Choice	05/04/91	439.6	74.3	5.92
Fundy's Choice	12/19/91	491.9	73.7	6.67
Moore Clark (8.5)	Not recorded	509.4	76.5	6.66
Moore Clark	12/19/91	468.8	87.8	5.34
Sure Gain	Not recorded	512.9	84.9	6.04
Sure Gain	05/04/91	480.7	84.8	5.67
Sure Gain	12/19/91	515.1	78.4	6.57

Table 3. Organic carbon and nitrogen content (mg g^1 dry weight) and C/N ratio of salmon faecal pellets.

Site	Date collected (mo/d/yr)	Carbon	Nitrogen	C/N ratio
Broad Cove	04/30/92	246.41	16.71	14.75
Toothacher Cove	10/06/92	188.33	12.05	15.72

SINKING RATE OF THE COMMONLY USED MOIST AND DRY FOODS

Table 4 gives the type, size (diameter and length), and sinking rates for several commonly used food pellets. Sinking rates ranged from 5.5 to 15.5 cm s^{-1}, with the commonly used Connors' moist food sinking at a rate of 10 cm s^{-1}. Neither type (dry or moist), pellet length, nor diameter appeared to be a good predictor of sinking rate.

Table 4. Size and sinking rate of several salmon food pellets.

Feed	Type	Size (mm)[1]	Sinking rate (cm s^{-1})[2]
Ecoline 3	Dry	3.9 ± 0.2 x 3.1 ± 0.1	5.5 ± 1.0
Moore Clark 8.5	Dry	11.8 ± 0.3 x 8.8 ± 0.2	7.4 ± 2.3
Sure Gain	Dry	14.7 ± 1.6 x 9.8 ± 0.1	15.5 ± 1.3
Moore Clark 3.5	Dry	5.8 ± 0.6 x 3.6 ± 0.1	7.0 ± 0.7
Fundy's Choice	Dry	9.6 ± 0.4 x 6.8 ± 0.1	8.6 ± 1.6
Connors (12/19/91)	Moist	30.0 ± 10.0 x 9.2 ± 0.3	9.8 ± 0.8
Connors (01/30/92)	Moist	30.0 ± 10.0 x 9.2 ± 0.3	10.0 ± 2.4

[1]Mean ± S.D., for five replicate pellets.
[2]Mean ± S.D., for ten replicate pellets.

THE PERCENTAGE OF WASTE FOOD REACHING THE SEDIMENT-WATER INTERFACE

The percentage of waste food reaching the sediment-water interface was calculated by counting numbers of food pellets present in the sediment traps deployed under or near a pen and by estimating the total area of sea bottom receiving pellets and the total number of pellets offered as food during the trap deployment period. This seemingly simple task proved most daunting. Appendix 1 lists by site, date, and station (the position of the trap relative to the pen) all of the successful sediment trap deployments. Numbers of observed food and faecal pellets are also presented in Appendix 1. Both were rarely observed in large numbers. Indeed, of 49 traps deployed within the expected settlement zone (see below) and assayed for the presence of waste food, only 19 traps contained food pellets. Similarly, of a possible 41 traps that might have contained intact faecal pellets, only 7 traps were observed to contain them. On average, traps deployed within the settlement zone contained 3.2 ± 7.1 waste food pellets and 1.3 ± 4.0 intact faecal pellets.

The area of sea bottom receiving pellets was calculated by:

$$A_{fp} = \pi\ ((D \times SR_{fp} / V) + (P_d /2))^2 \qquad (1)$$

Where A_{fp} is the area of bottom onto which pellets are assumed to settle; D is the mean depth of water below the net-pen (i.e., depth of water below the bottom of the net-pen at MLW + 1/2 the average tidal cycle); SR_{fp} is the settling rate of the food pellets; V is the average current velocity, and P_d is the diameter of the pen. It was assumed that distribution of pellets within this area followed a Poisson distribution. The number of pellets being fed within the deployment period was calculated from feeding logs provided by the aquaculturist. Estimates of the number of pellets per kilogram of feed were provided by the manufacturer and cross-checked by wet weight measurements conducted in our laboratory.

Only in two cases (Toothacher Cove Pen 2, 11/6-11/19/91; Toothacher Cove Pen 10 11/5-11/20/91) were we able to successfully gather all of the necessary data (see Appendix 1) to allow for this calculation. Waste food estimates for these two cases were 11.0 and 5.0%, respectively (Table 5). Clearly, these two estimates are less than the "typical" 20-30% waste food used in most models.

Given the large number of times that few, if any, food pellets were observed in traps deployed beneath pens these estimates are likely the upper limits of the percentage of food wasted. For example, during the period 7/14-8/4/92 no pellets were recovered in traps from Toothacher Pen 2. If the calculation of percentage waste food is made using an average of 1 pellet $trap^{-1}$ and the kilograms of food fed from 1991 the calculated waste food is 0.90%, indicating the percentage waste food for this time period is likely less than this value. Additional estimates can be made when further production figures are obtained from other aquaculturists in the study area.

COMPARISON OF PREDICTED AND MEASURED ORGANIC CARBON FLUX

We constructed a simple mass-balance model where organic carbon entering the pen was partitioned into two classes, assimilated or non-assimilated. Assimilated carbon was assumed to be removed from the system, and all non-assimilated carbon was assumed to settle to the sea bottom as either waste food or faeces. Finally, the rate of carbon flux to the sediments at the edge of the net-pens was calculated as the sum of the flux due to waste food and the flux due to faecal pellets. We illustrate this approach with two examples from Toothacher Cove. In both examples we use a single circular pen and short ($\approx$2 wk) sediment trap deployments in November of 1991. Table 6 summarizes all necessary production and environmental data.

The first example is from a Pen 2, for the period November 6-19, 1991. For this period the aquaculturists reported that 690 kg of Ecoline 3.5 was supplied as food. Food pellets contained $\approx$45% organic carbon (see Table 2) and approximately 8% water. The organic carbon fed during this time can then be calculated as:

$$C_{fd} = (F \times 0.08)(0.45) = 285{,}600 \text{ g} \qquad (2)$$

where F = the food fed in grams and C_{fd} = organic carbon (g) entering pen as food. The organic carbon in the waste food and faeces then becomes:

$$C_{wf} = C_{fd} \times \%WF \qquad (3)$$

and:

$$C_{fc} = (C_{fd} - C_{wf})(1-A) \quad (4)$$

or:

$$C_{wf} = 31{,}311 \text{ g and } C_{fc} = 45{,}783 \text{ g} \quad (5)$$

Where C_{fc} = organic carbon (g) exiting pen as faeces, C_{wf} = organic carbon leaving pen as waste food, %WF = the percentage of food wasted as calculated from our sediment trap data, and A = assimilation (we used a value of 0.82 - obtained from Moore Clark for their Select® formulation). We assumed that waste food was distributed homogeneously horizontally across the pen by hand feeding. It was also assumed that there would be lateral diffusive movement, no dispersion within the pen, and that currents would disperse the material equally in all directions. The rate of organic carbon deposition then becomes a function of the organic carbon leaving the system divided by the area of sea bottom receiving the waste divided by the number of days of fish production or:

$$FLUX_{wf} = C_{wf} / A_{fp} / d \quad (6)$$

and:

$$FLUX_{fc} = C_{fc} / A_{fc} / d \quad (7)$$

or:

$$FLUX_{wf} = 5.92 \text{ g C m}^{-2} \text{ d}^{-1} \text{ and } FLUX_{fc} = 3.37 \text{ g C m}^{-2} \text{ d}^{-1} \quad (8)$$

Where $FLUX_{wf}$ = the predicted rate of organic carbon deposition due to waste feed and $FLUX_{fc}$ = the predicted rate of organic carbon deposition due to salmon faeces. The total predicted flux at the pen edge is the sum of these two rates or 9.39 g C m^{-2} d^{-1}. During this period we measured carbon flux rates of 3.4, 3.8 and 4.5 g C m^{-2} d^{-1} in three traps deployed at the edge of this net-pen.

The second example is from Pen 10, for the period November 5-20, 1991. For this period the aquaculturists reported 1423 kg of a 50/50 mix of Ecoline 5 and Moore Clark Select 5 was fed. This food is ≈45% carbon (see Table 2), and we estimate the combined water content as 15%. The carbon fed during this time can then be calculated as:

$$C_{fd} = (F \times 0.15)(0.45) = 544{,}298 \text{ g} \quad (9)$$

the organic carbon in the waste food and faeces then becomes:

$$C_{wf} = 27{,}233 \text{ g and } C_{fc} = 93{,}072 \text{ g} \quad (10)$$

Calculated flux rates were:

$$FLUX_{wf} = 5.82 \text{ g C m}^{-2} \text{ d}^{-1} \text{ and } FLUX_{fc} = 8.50 \text{ g C m}^{-2} \text{ d}^{-1} \quad (11)$$

Table 5. Calculation of percent waste food.

Toothacher Cove	Pen 2	Pen 10
Food Fed:		
Type	Econoline 3	Econoline 5 & Moore Clark 5.5 (50/50)
Amount (kg)	690	1,423
Pellets g^{-1} (estimated)	13.2	6.0
Area of bottom receiving food (m^2) for:	407.11	311.94
P_d (m)	15.24	15.24
V (cm s^{-1})	2.00	2.00
SR (cm s^{-1})	5.50	6.50
D (m)	10.36	7.62
Average pellets in trap within this area	19.25	10.75
Area of traps (m^2)	0.00785	0.00785
Pellets hitting bottom	998,327	427,178
Total pellets fed	9,110,770	8,540,000
Percent waste	10.96	5.00

The total predicted flux at the pen edge is the sum of these to rates or 14.32 g C m^{-2} d^{-1}. During this period we measured organic carbon flux rates of 3.7, 1.4 and 1.0 g C m^{-2} d^{-1} in three traps deployed at the edge of this net-pen.

The range of sedimentation rates measured with sediment traps deployed at the edge of pens in Toothacher Cove in November 1991 (1.0 to 4.5 g C m^{-2} d^{-1}) encompasses values observed at the edge of salmon net-pens in Bliss Harbour, L'Etang Inlet (1.2 to 1.4 g C m^{-2} d^{-1}) (Chapter 5). However, the rates predicted by the model are two- to ten-fold higher than these values. The simplest interpretation of this inconsistency is that one or more of the assumptions on which the model calculations and measurements of sedimentation are based are false. These assumption include: 1) the sediment traps were properly designed for the environment; 2) the traps accurately collect the range of particle sizes emanating for the pen; 3) the carbon content of trapped material is stable under salt preservation; 4) that sieving through a 500 μm sieve does not remove any waste material; 5) the distribution of food, waste food, and faeces is uniform within the pen (in particular, that the vertical walls of the net-pens do not concentrate waste food particles); 6) particles start to disperse at the bottom of the net-pen (i.e. the particles do not exit the sides of the pens; 7) faeces exit as intact particles; 8) the bottom of the net-pen allows free exit for all particles (both temporally and spatially); 9) all non-assimilated food is distributed between either waste food or faecal pellets; and 10) current velocities measured 1 m from the bottom accurately represent the currents dispersing pellets.

Table 6. Production and environmental parameters used in modelling benthic carbon flux.

	Pen 2	Pen 10
Pen diameter (m)	15.24	15.24
Pen depth (m)	7.32	7.32
Water depth at MLW (m)	16.15	13.41
Tidal range (m)	3.05	3.05
Average tidal velocity (cm s^{-1})	2.1	2.1
Average number of fish	7,424	17,653
Average weight of fish (g)	544	500
Food Fed:		
Type	Econoline 3	Econoline 5 & Moore Clark 5.5 (50/50)
Amount (kg)	690	1,423
Period (d)	13	15
Pellets g^{-1}	13.2	6.01
Percent water	8.0	15.01
Percent carbon[2]	45.0	45.0
Assimilation efficiency	0.82	0.82[1]
Percentage waste food[3]	11.0	5.0
Area of bottom receiving food[4] (m^2)	407.11	311.94
Area of bottom receiving faeces[5] (m^2)	1,014.49	729.99

[1]Estimated.
[2]From Table 2.
[3]From Table 5.
[4]From Table 5.
[5]Calculated as for area of bottom receiving food, except a settling velocity of 2 cm s^{-1} was used.

It is not clear which, if any, of the above assumptions is false, although the sensitivity of the model to each parameter can be tested by altering values and noting the resultant changes in calculated flux rates. Using data for Pen 10 as an example, if particle dispersion is assumed to occur at mid-depth rather than at the bottom of the pen, total organic carbon flux rates decrease from 11.28 to 9.60 g C m^{-2} d^{-1}, respectively. Unfortunately, space limitation precludes a complete analysis and discussion of all the assumptions listed above. In addition, verification of the accuracy of the sediment traps will require experimental verification.

SEDIMENTARY ORGANIC CARBON DECAY RATE CONSTANTS

One measure of sediment assimilative capacity is the ability of the benthic community to decompose organic matter. If the organic matter is assumed to be uniform in nature, the process can be described by the equation

$$dG/dt = -kG \tag{12}$$

where G is the concentration of organic matter (expressed as organic carbon) and k is the organic matter decay rate constant (Berner 1980). Estimates of organic matter decay rate constants reported in the literature range over five orders of magnitude, from a low of 1.4 x 10^{-1} d^{-1} for labile algal material (Hendrichs and Doyle 1986) to a high of 1.6 x 10^{-6} d^{-1} for refractory organic material (Jahnke 1990).

Three methods for the estimation of organic carbon remineralization rates and hence organic carbon decay rate constants are commonly employed (Berner 1980; Martens and Klump 1984). They are: 1) establishment of a sedimentary carbon budget by direct measurement of the flux of organic matter to the sediment, the flux of ΣCO_2 across the sediment-water interface, and burial of residual organic carbon not degraded; 2) determination of organic carbon remineralization rates using kinetic models of the vertical distribution of sedimentary organic carbon; and 3) indirect determination of organic carbon remineralization rates from measured rates of sulfate reduction and O_2 consumption.

We were able to measure all of the above parameters with the exception of the rates of sulfate reduction, yet we were unable to calculate any organic matter decay rate constants. We had assumed, during the experimental design phase of this project, that we would be able to determine a sedimentation rate from our sediment trap data - this assumption proved false due to the importance of sediment resuspension within our study areas. While we are unable to provide an estimate of k, we have learned a great deal concerning the rates of degradation of the organic matter originating in net-pen systems and how the rate of flux of this material to the sediments affects benthic processes. This effort has culminated in a process-level model that attempts to explain some of the apparent site-specific differences observed between various salmon production sites. The remainder of this report details these findings.

Critical to an understanding of assimilative capacity is knowledge of the ratio of organic carbon flux to the sediment versus organic carbon degradation at and within the sediment. We explored this relationship by comparing fluxes of organic carbon to and from the sediments using sediment traps and increases in total CO_2 in sealed benthic incubation chambers, respectively. Results from cores covered by bacterial mats were excluded from this analysis as these communities are chemolithotrophic. There was a strong linear (b=1.0) relationship between organic carbon sedimentation measured over a 15- to 30-d period prior to the measurement of benthic respiration) and degradation as measured by CO_2 flux across the sediment-water interface (Fig. 1). This finding indicates that the waste stream (waste food and salmon faeces) from salmon net-pen facilities was readily and rapidly digested.

In aerobic metabolic systems, approximately 1 mole of O_2 is required to reduce 1 mole of organic carbon. In aerobic marine sediments this ratio is normally shifted to approximately 0.7:1 due to sulfate-reduction by anaerobic bacteria. By simultaneously measuring O_2 uptake and CO_2 production in benthic respiration chambers we have confirmed this general trend for marine sediments beneath and near salmon net-pens when benthic bacterial mats were not present (Fig. 2).

These observations lead to an important conclusion: recently deposited salmon production wastes (feed and faeces) will generate a sediment oxygen demand proportionate to the rate of carbon deposition. The molar ratio for carbon flux to oxygen demand in our study was 1:0.7. The validity of this conclusion was tested by substituting sediment O_2 uptake rates for CO_2 production rates and comparing these values to the average organic carbon sedimentation rates (Fig. 1). Results of the comparison (Fig. 3) show that there was a strong correlation between the amount of carbon arriving at the sediment-water interface and the sediment O_2 demand. This observation has profound implications for the siting and regulation of salmon net-pens.

At current speeds of ≈ 10 cm s^{-1} or less, water flow will characteristically be smooth-turbulent. This implies that the water column will be well mixed and a viscous sublayer will exist at the sediment-water interface. Under such conditions, and in the absence of biogenic transport, transport of O_2 to the sediments will be diffusion limited and dependent on O_2 concentration, temperature, and the thickness of the viscous sublayer. Jorgensen (1989), using O_2 microelectrodes, has measured the thickness of viscous sublayer at relevant current flow rates. Using these data we have calculated theoretical oxygen delivery rates to the sediment. Figure 4 shows the current speed dependent O_2 delivery rate for sea water at 20°C. Using the relationship outlined above (i.e., a molar ratio for carbon flux to oxygen demand of 1:0.7), it is possible to calculate the maximum carbon flux rates that will not deplete the sediments of molecular oxygen (Fig. 4). Clearly, as current speeds approach zero the theoretical maximum delivery rate of organic matter that will not deplete sediment molecular oxygen falls below 4 g C m^{-2} d^{-1}. It is important to note that O_2 uptake rates are closely linked to current speed and change rapidly (seconds to minutes). This suggests that instantaneous, rather than average, current velocities are critical for evaluating a potential aquaculture site. In addition, macrofauna are most likely to be the first

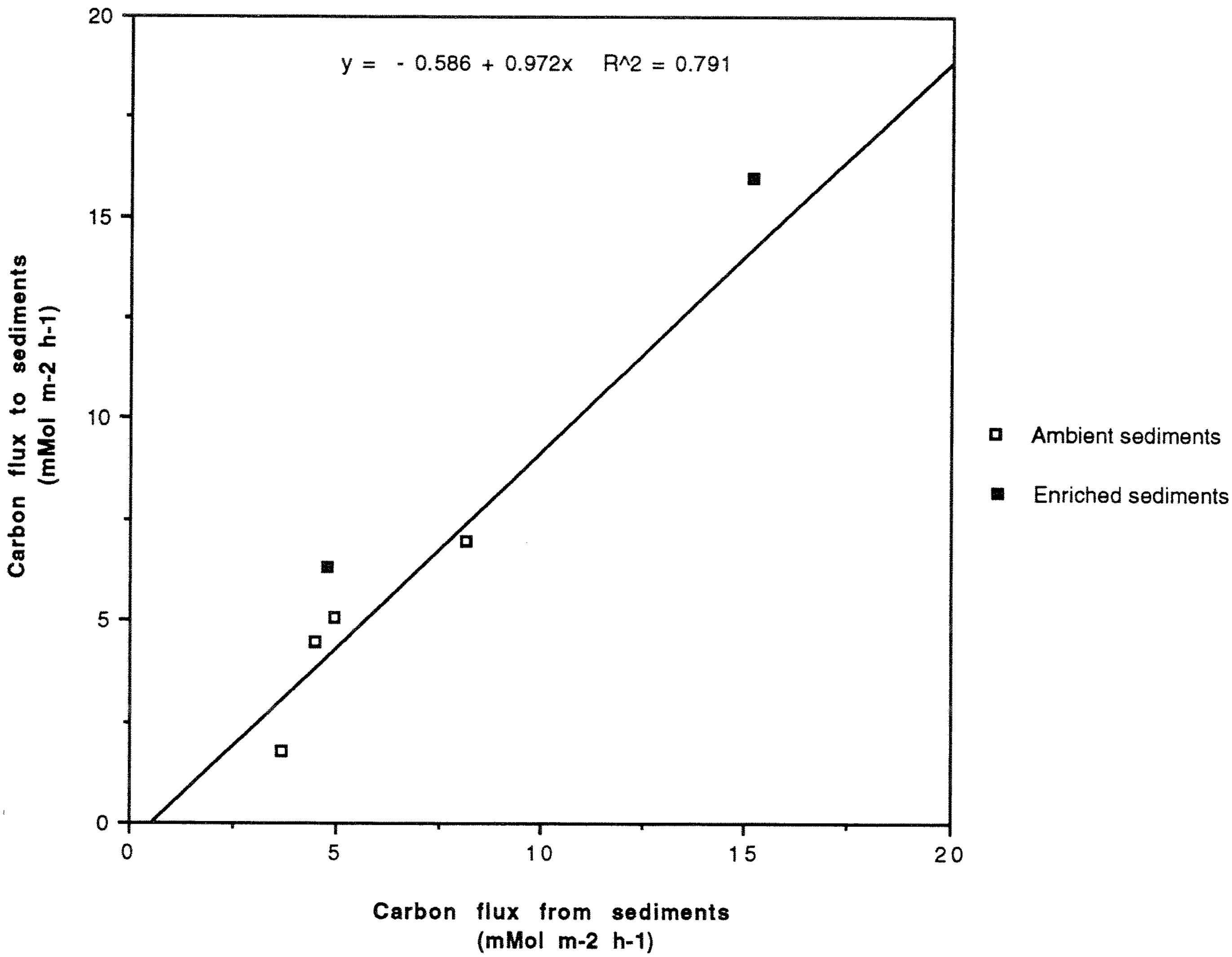

Figure 1. Comparison of organic carbon flux to and from sediments beneath and near salmon net-pens. Each point represents a comparison of the hourly flux of organic carbon to the sediments determined by sediment traps and the average flux of CO_2 from the sediment (three independent incubations) determined using undisturbed sediment cores and sealed incubation chambers.

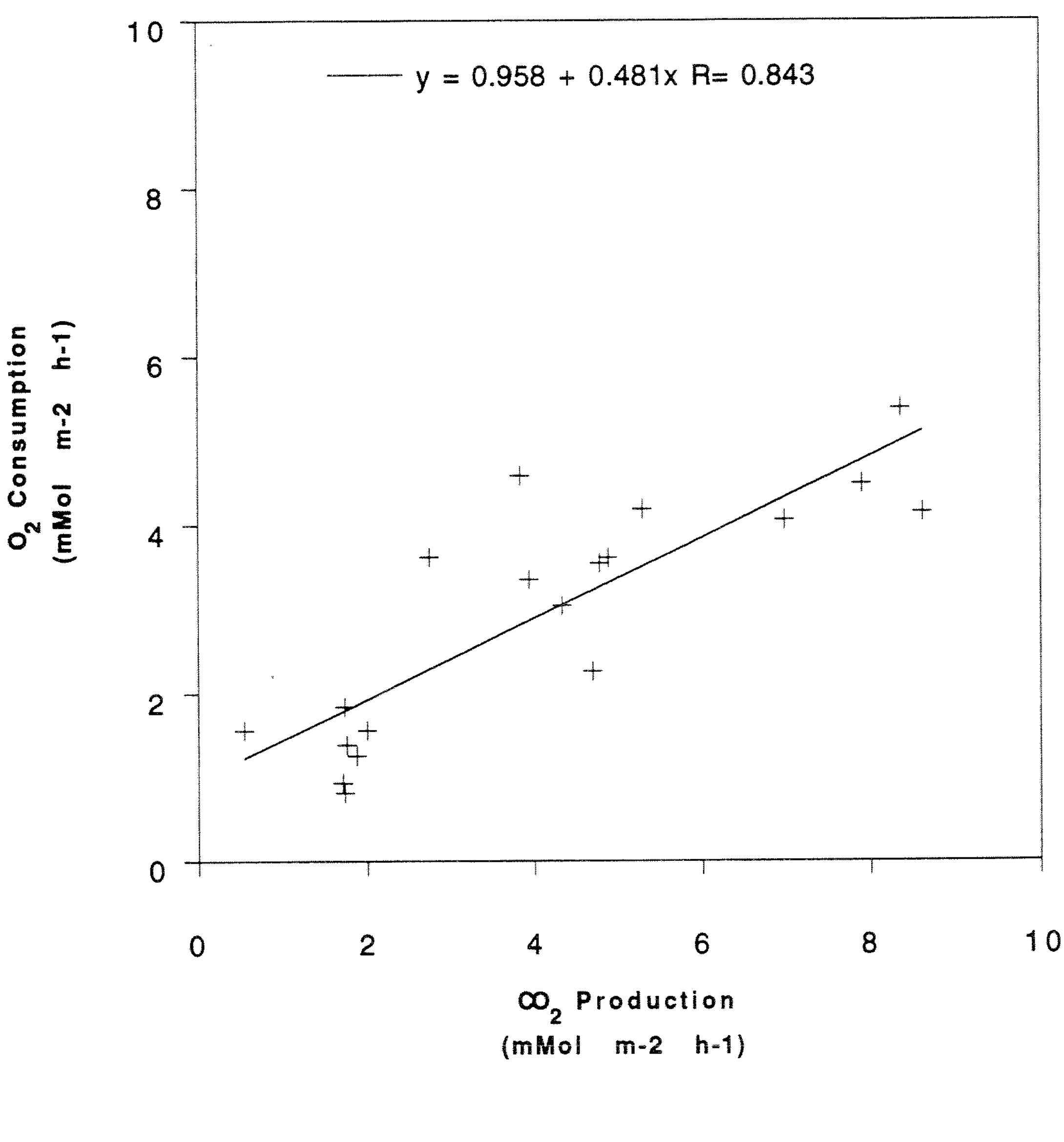

Figure 2. Comparison of O_2 consumption to CO_2 production in marine sediments. Each point represents the results of a single incubation of an undisturbed sediment core. Sediments were collected from three salmon production facilities (both proximal and distal stations) and a control site on the Damariscotta River.

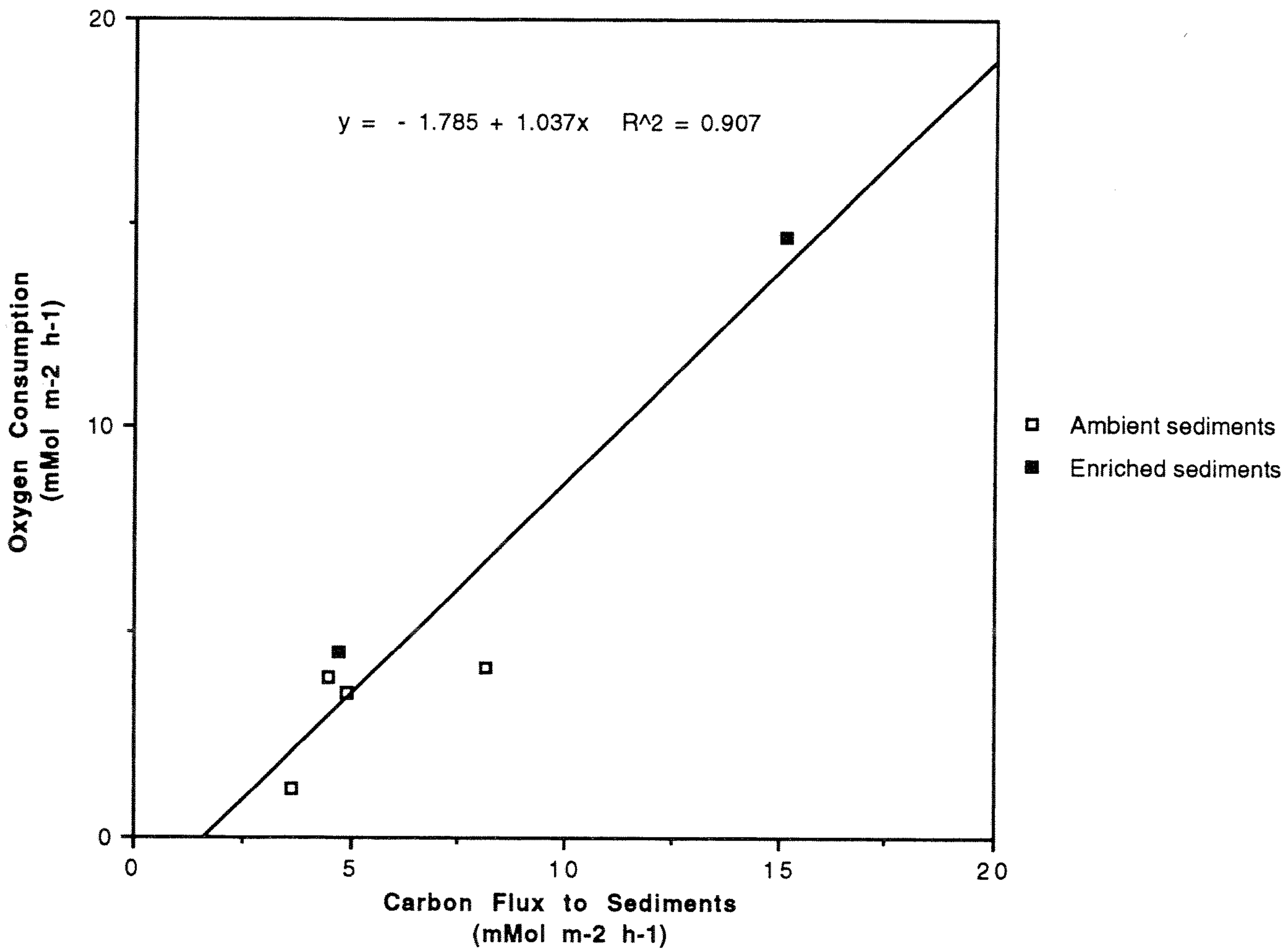

Figure 3. Comparison of carbon flux to and O_2 consumption by sediments beneath and near salmon net-pens. Each point represents a comparison of the hourly flux of organic carbon to the sediments determined by sediment traps and the average O_2 consumption of the sediment (three independent incubations) determined using undisturbed sediment cores and sealed incubation cambers.

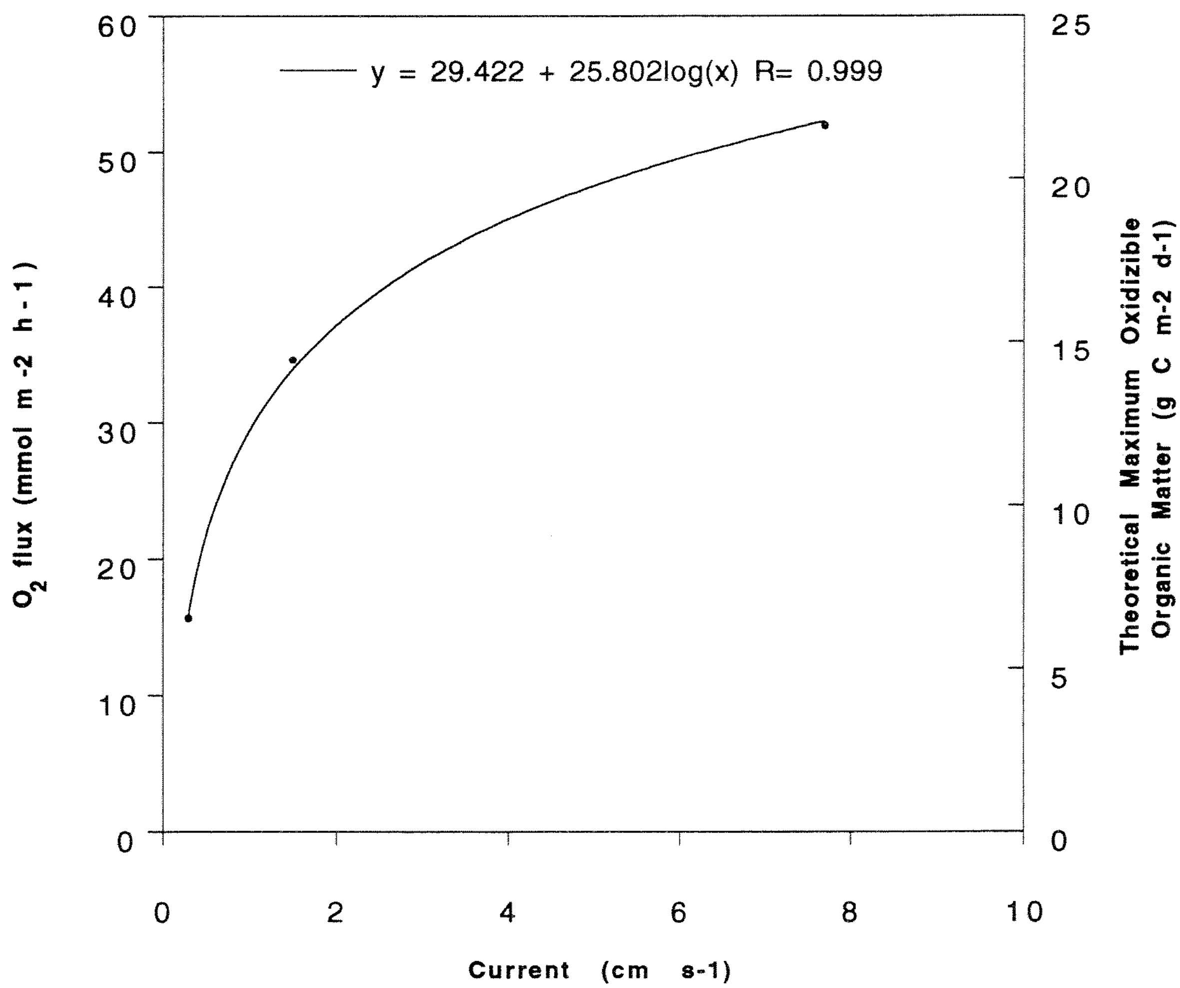

Figure 4. The rate of O_2 delivery to sediments as a function of current speed. Delivery rates were calculated using the oxygen diffusion rate and O_2 saturation concentration for sea water (at 20°C), and the thickness of the viscous sub-layer given in Jorgensen (1990). Axis Y_1 show rates in mMoles of O_2 m^{-2} h^{-1} and axis Y_2 shows the theoretical maximum rate of aerobic oxidation of organic matter (g C m^{-2} d^{-1}) calculated using the relationships detailed in Figures 1 to 3.

organisms to be negatively affected by anoxia. Species not able to withstand prolonged deprivation of dissolved oxygen will be killed within the first 2 h of the onset of anoxic conditions. Therefore, we have evaluated the effects of current speed on the basis of the minimum mean current speed for any 2 h period.

We have intensively studied four salmon production sites (Toothacher Cove - Pen 2, Toothacher Cove - Pen 10, Eastport - Depositional, and Eastport - Erosional) during 1991-1992. The highest average carbon fluxes were measured at the Eastport - Depositional site, yet sediments at this site remain aerobic and support abundant macrofauna. In contrast, moderate carbon fluxes were measured at the two Toothacher Cove sites and these sediments developed *Beggiatoa*-type bacterial mats. This apparent inconsistency in impact level can be explained by plotting these sites in terms of their organic carbon sedimentation rates verses their minimum average 2 h current velocities (Fig. 5).

The minimum 2 h average current velocity was 0 cm s^{-1} at the two Toothacher Cove sites. The estimated theoretical maximum rate of aerobic oxidation of organic matter at this current velocity is >4.0 g C m^{-2} d^{-1}, and the maximum measured rates of sedimentation at the two pens were 6.7 and 8.0 g C m^{-2} d^{-1}, respectively. Clearly, organic carbon supply outstripped the rate of aerobic decomposition of deposited organic matter. *Beggiatoa*-type mats formed beneath the pens at both sites and macrofaunal abundance was severely decreased.

In contrast to the Toothacher Cove sites, the minimum 2 h-average current velocity at the Eastport - Depositional site was 3 cm s^{-1}. The estimated theoretical maximum rate of aerobic oxidation of organic matter at this current velocity is $\approx$ 17.0 g C m^{-2} d^{-1}. While the maximum measured rate of sedimentation at this site was 1.6 times higher than at the Toothacher Cove sites (13.0 vs. 8.0 g C m^{-2} d^{-1}), sufficient O_2 was supplied to the sediments allowing for aerobic decomposition of the organic waste stream. This site supported an abundant and diverse macrofaunal community.

At the fourth site, Eastport - Erosional, the estimated theoretical maximum rate of aerobic oxidation of organic matter greatly exceeded the maximum measured rates of sedimentation and there were few, if any, detectable effects of the net-pens upon the benthic community.

In summary, we were not able to estimate an appropriate sedimentary carbon decay rate constant for waste food or salmon faeces due to the inadequacy of our mass accumulation rate as estimates of sedimentation rate; but this research effort has led to an understanding of the basic physical and biological principles behind apparent site-specific benthic responses that arise from salmon net-pen aquaculture.

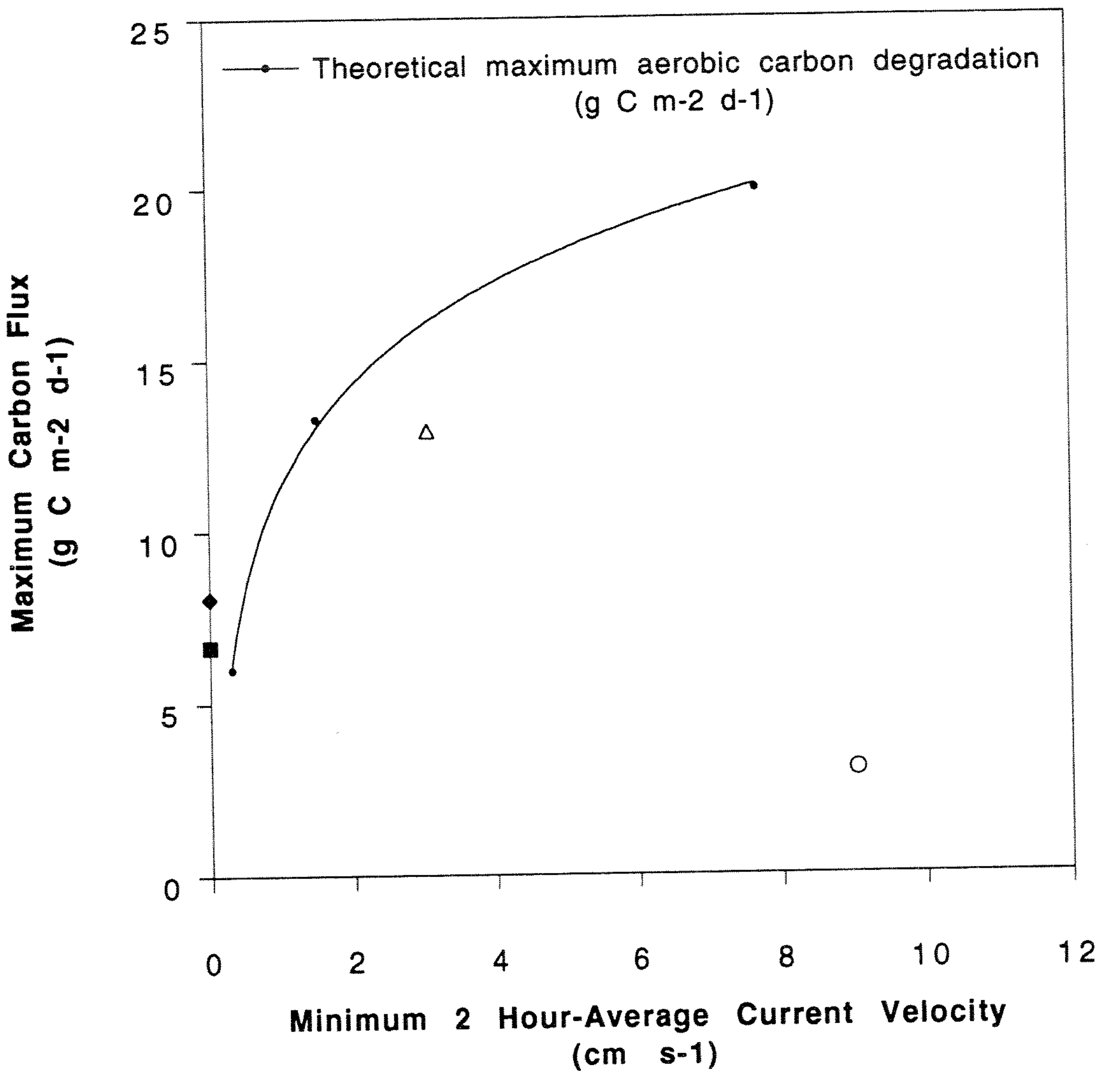

Figure 5. Comparison of the maximum measured carbon flux to the sediments beneath four salmon net-pens and the theoretical maximum rate of aerobic carbon degradation. For points above and left of the curve carbon flux to the sediment exceeds the theoretical maximum rate of aerobic carbon degradation and for points below and right of the curve carbon flux to the sediment is less than the theoretical maximum rate of aerobic carbon degradation.

IMPLICATIONS FOR SITING SALMON NET-PEN PRODUCTION FACILITIES

The implications of our work for siting decisions concerning salmon production cages are many. The organic carbon and nitrogen content of salmon feed and faeces reported within are empirical measures and require no further discussion. Of important note is the discrepancy of assimilation factors commonly used and those currently advertised (or at least available) from food manufacturers. Many models use an assimilation factor of 0.7 while food manufactures provide factors of 0.8 or greater. An important area of discrepancy between most models and our research is the percentage of waste food. For any model predicting the waste flux from the pen the most sensitive parameter (other than food fed) will be the percentage of waste food. The waste food percentages reported in the literature range from a high of 40% (Weston 1986) to a low of 1% (Rosenthal et al. 1988; Thorpe et al. 1990). Most models - or more accurately their authors - settle on a median value of 15 to 20%. Our research does not support the use of this value. While in one case we did estimate a percentage waste food as high as 11%, this was an extreme case and the high frequency of traps containing few or no food-pellets suggest that much lower values are the norm. We suggest that a 5% waste food figure (or lower) is more likely the case for modern farms in Maine (especially those that hand-feed). In addition, we suggest that all modelling attempts be run at three waste food percentages: a high of 15%, a low of 1%, and a middle value of 5%.

The parameters measured during this study that have direct bearing on distribution of the waste stream within the environment were the sinking rate of salmon food and faecal pellets. The relative infrequency with which intact faecal-pellets were observed within the sediment traps is also of potential relevance. Food pellets showed a range of sinking rates (5.5 to 15.5 cm s^{-1}) with mean and median values of 9.1 and 8.6 cm s^{-1}, respectively. No strong relationship between size or food type (dry or moist) and sinking rate was found. Fresh faecal pellets were found to sink much slower (2.0 cm s^{-1}) than the value commonly used for modelling the behaviour of salmon faecal pellets (4.0 cm s^{-1}). This lower sinking rate has the consequences within our model of increasing the area of sea bottom that will receive wastes and decreasing the predicted sedimentation of organic carbon unit^{-1} area.

Our extreme difficulty in collecting intact faecal pellets taken in conjunction with the few intact pellets collected by our sediment traps has led us to conclude that intact faecal pellets may be the exception rather that the rule. Indeed, faeces, for the most part, may exit a net-pen system as fine particulate matter rather than discrete pellets. This may be the result of several processes. Currently used types of food pellets fed to salmon in Maine are highly digestible and thus do not lead to the production of discreet faecal pellets. Alternatively, faecal pellets may be produced but they are broken by the swimming activity of the fish. These pellet fragments may be consumed by fish that mistake them for food pellets that are broken during capture or rejection. Finally, pellets may be produced but they are broken by contact with the nets, or pellets are produced but are retained on the bottom net where they age, degrade, and break apart. The consequence is to decrease the settling rate of the faecal matter which leads to an increased area of sea bottom that receives the wastes and hence a decrease in organic carbon loading $unit^{-1}$ area.

Our finding that the balance between O_2 and carbon flux to the sediment controls benthic respiration is paramount to understanding the effects of the increased organic loading on the benthos (Problem 3 mentioned in the Introduction section of this chapter). The implications for siting of net-pens are many. First, current flow becomes critical. Mean surface current velocities are not specific enough to make usable predictions. Current velocity ranges (as commonly reported) are even less useful. Near-bottom flow rates measured every 0.5 to 1 h are necessary to determine the average minimum flow during periods of low flow. This parameter is important because of the high dependency of benthos on oxygen and the importance of flow (or lack of flow) on the delivery oxygen to the benthos. Increases in the minimum flow rate from 0.3 to 1.0 to 3.0 cm s^{-1} increase the estimated theoretical maximum rate of aerobic oxidation of organic matter from 6.1 to 11.3 to 16.0 g C m^{-2} d^{-1}, respectively. Clearly, sites that exhibit stoppages of flow of 2 h or greater are at greater risk of developing anaerobic sediments than sites where current flow is not interrupted. The maintenance of even moderate flows will greatly reduce this risk. Small farms (feeding rates of 300 g of food m^{-2} of pen surface area d^{-1}), carefully operated (food waste $\geq 1\%$ of food fed), could theoretically be located in these low flow areas as predicted carbon loadings are ≥ 5.0 g C m^{-2} d^{-1}. Control of waste food is a critical factor in the relationships between flow, oxygen supply, and organic carbon decomposition. Although water depth and current speed play a role, a doubling in the percentage waste food from 5% to 10%, in general, leads to a two-fold increase in predicted organic carbon loading. If these increased rates of organic carbon sedimentation are realized under field conditions, then minimizing waste food becomes critical to reducing impacts on the benthos. It is important to note that current economic conditions and husbandry practices work to decrease waste food.

RECOMMENDATIONS

On the basis of the work presented here, we make the following recommendations with respect to future research on the environmental effects of salmon net-pens and on the current regulation of the industry:

1. Agencies concerned with the regulation of the salmon net-pen industry should seek or fund independent verification of our finding that the balance between O_2 and carbon flux to the sediment controls benthic respiration. While we are currently actively working to confirm this finding, it is important that independent confirmation be sought. If our finding is confirmed, our efforts will lead to the first process-level understanding of the response of the benthos to salmon net-pens.

2. Agencies concerned with the regulation of the salmon net-pen industry should seek or fund research to determine the state in which the majority of salmon faeces (as

discrete pellets or smaller fragmented particles) exit net-pen systems. If faecal pellets are as rare as we observed, then most of the faecal material leaving the pens will be in small (and presumably slower settling) particles. This will lead to larger areas of impact but should reduce the magnitude of the benthic flux, in turn, reducing the magnitude of the effects within the area.

3. Agencies concerned with the regulation of the salmon net-pen industry should continue to support the concurrent efforts to model the effluent stream of salmon net-pens and to empirically determine the effluent stream of salmon net-pens. While predictive models will provide the only cost-effective method of siting net-pens, they are virtually without value until their predictions are verified by empirical data (due the great number of unsubstantiated assumptions necessary within the models).

4. Agencies concerned with the regulation of the salmon net-pen industry should continue to support the "status quo" for current siting and monitoring regulations within Maine waters. The current memorandum of agreement for siting and regulating salmon net-pens contains sufficient safeguards for the nearshore environment. This recommendation is not based on the existence of highly accurate regulations guaranteed to protect the environment within the memorandum; it is based on the relatively benign nature of the waste stream originating with salmon net-pens. Our studies have shown that the waste is rapidly decomposed within the marine environment. Even under a "worst-case scenario" where *Beggiatoa*-type mats form on the sediment surface, the solution to the problem is to reduce the waste stream. Unlike many other waste streams that will persist within the environment (heavy metals, toxic xenobiotics, radioactive materials, etc.), the wastes from the production of salmon within the marine environment will rapidly decay if provided sufficient oxygen. This relatively low risk (i.e., short-term and localized damage) should minimize the need for a "zero-mistake" management approach. It is our opinion that the current regulations strike an acceptable compromise between the need for economic development of Down-East Maine and protection of the marine environment. This does not preclude future modification of existing regulations as our results are confirmed and predictive models are refined allowing the incorporation of process-level understanding of pelagic-benthic interactions within existing siting and monitoring regulations.

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Appendix 1. Recorded counts of food and faecal pellets in successful sediment trap deployments from 1991 throughout 1993.

Site and date		Station[1]	Number of pellets		Comments
			Food	Faeces	
Eastport - Depo 2	5/19-7/8/92	P1M sm	0	0	
Eastport - Depo 2	5/19-7/8/92	P1M lg	0	0	
Eastport - Depo 2	5/19-7/8/92	P10M lg	0	0	
Eastport - Depo 2	5/19-7/8/92	P20	0	0	
Eastport - Depo 2	7/8-7/29/92	P1Msm	0	0	
Eastport - Depo 2	7/8-7/29/92	P1Mlg	0	0	
Eastport - Depo 2	7/8-7/29/92	P10M	0	0	
Eastport - Depo 2	7/8-7/29/92	A10Msm	0	0	
Eastport - Depo 2	7/8-7/29/92	A20Msm	0	0	
Eastport - Depo 2	7/8-7/29/92	A20Mlg	0	0	
Eastport - Depo 2	7/29-8/25/92	P1Msm	0	0	
Eastport - Depo 2	7/29-8/25/92	P10Msm	0	0	
Eastport - Depo 2	7/29-8/25/92	P10Mlg	0	0	
Eastport - Depo 2	7/29-8/25/92	P20Msm	0	0	
Eastport - Depo 2	7/29-8/25/92	P20M	0	0	
Eastport - Depo 2	7/29-8/25/92	A1M	0	0	
Eastport - Depo 2	7/29-8/25/92	A10M	0	0	
Eastport - Depo 2	8/25-9/10/92	A10M	0	0	
Eastport - Depo 2	8/25-9/10/92	A20M	0	0	
Eastport - Depo 2	8/25-9/10/92	P10M	5	3	
Eastport - Depo 2	8/25-9/10/92	P20M	0	1	
Eastport - Depo 2	9/10-10/8/92	A1M	0	0	
Eastport - Depo 2	9/10-10/8/92	A10M	0	0	
Eastport - Depo 2	9/10-10/8/92	A20M	0	0	
Eastport - Depo 2	9/10-10/8/92	P10M	0	0	
Eastport - Depo 2	9/10-10/8/92	P20M	0	0	
Eastport - Depo 2	10/8-12/2/92	A1M	0	0	
Eastport - Depo 2	10/8-12/2/92	A10M	0	0	
Eastport - Depo 2	10/8-12/2/92	A20M	0	0	
Eastport - Depo 2	10/8-12/2/92	P10M	0	0	
Eastport - Depo 2	10/8-12/2/92	P20M#1&#2	0	0	
Eastport - Depo 1	6/20-7/18/91	PN1	ND	ND	
Eastport - Depo 1	6/20-7/18/91	PN25	ND	ND	
Eastport - Depo 1	6/20-7/18/91	PE1A	ND	ND	
Eastport - Depo 1	6/20-7/18/91	PE1B	ND	ND	
Eastport - Depo 1	6/20-7/18/91	AE100A	ND	ND	
Eastport - Depo 1	6/20-7/18/91	AE100B	ND	ND	

Appendix 1 (cont.)

Site and date		Station[1]	Number of pellets		Comments
			Food	Faeces	
Eastport - Depo 1	7/18-9/5/91	PN1	ND	ND	not done - excess sediment
Eastport - Depo 1	7/18-9/5/91	PN25	ND	ND	not done - excess sediment
Eastport - Depo 1	7/18-9/5/91	PE1A	ND	ND	not done - excess sediment
Eastport - Depo 1	7/18-9/5/91	PE1B	ND	ND	not done - excess sediment
Eastport - Depo 1	7/18-9/5/91	AE100A	ND	ND	not done - excess sediment
Eastport - Depo 1	7/18-9/5/91	AE100B	ND	ND	not done - excess sediment
Eastport - Depo 1	9/5-10/18/91	PN1	ND	ND	
Eastport - Depo 1	9/5-10/18/91	PN25	ND	ND	
Eastport - Depo 1	9/5-10/18/91	PE1A	ND	ND	
Eastport - Depo 1	9/5-10/18/91	PE1B	ND	ND	
Eastport - Depo 1	9/5-10/18/91	PE25A	ND	ND	
Eastport - Depo 1	9/5-10/18/91	PE25B	ND	ND	
Eastport - Depo 1	9/5-10/18/91	AE100A	ND	ND	
Eastport - Depo 1	9/5-10/18/91	AE100B	ND	ND	
Eastport - Depo 1	3/18-4/30/92	PE1A	0	0	
Eastport - Depo 1	4/30-7/9/92	P1EA #1	ND	ND	not done - excess sediment
Eastport - Depo 1	4/30-7/9/92	P1EA #2	ND	ND	not done - excess sediment
Eastport - Depo 1	4/30-7/9/92	E1MA	ND	ND	not done - excess sediment
Eastport - Depo 1	7/9-7/31/92	N25M	3	4	
Eastport - Depo 1	7/9-7/31/92	PN1M	0	1	
Eastport - Depo 1	7/9-7/31/92	E25B	2	0	
Eastport - Depo 1	7/9-7/31/92	Curr Meter A	0	1	new traps established
Eastport - Depo 1	7/9-7/31/92	E1A	4	11	
		E1B	6	12	
Eastport - Depo 1	7/30-8/26/92	PE1MA	0	0	
Eastport - Depo 1	7/30-8/26/92	PE25MA	0	0	
Eastport - Depo 1	7/30-8/26/92	PE25MB	0	0	
Eastport - Depo 1	7/30-8/26/92	Curr Meter A	0	0	
Eastport - Depo 1	8/26-9/23/92	PE1MB	0	0	
Eastport - Depo 1	8/26-9/23/92	PE25MA	0	0	
Eastport - Depo 1	8/26-9/23/92	Curr Meter	0	0	
Eastport - Depo 1	8/26-9/23/92	Curr Meter B	0	0	
Eastport - Depo 1	9/22-10/8/92	PE25MA	4	3	
Eastport - Depo 1	9/22-10/8/92	Curr Meter A	0	0	
Eastport - Depo 1	9/22-10/8/92	PE1MB	4	20	
Eastport - Eros 1	6/18-7/17/91	(N 15M)	ND	ND	
Eastport - Eros 1	6/18-7/17/91	(N 30 M)	ND	ND	
Eastport - Eros 1	6/18-7/17/91	(W 30 M)	ND	ND	
Eastport - Eros 1	6/18-7/17/91	(W15 M)	ND	ND	
Eastport - Eros 1	6/18-7/17/91	Pen East 1	ND	ND	

Appendix 1 (cont.)

Site and date		Station[1]	Number of pellets		Comments
			Food	Faeces	
Eastport - Eros 1	6/18-7/17/91	PE15	ND	ND	
Eastport - Eros 1	6/18-7/17/91	PE30	ND	ND	
Eastport - Eros 1	6/18-7/17/91	PSouth 1	ND	ND	
Eastport - Eros 1	6/18-7/17/91	PS30	ND	ND	
Toothacher Cove, Pen 10	6/5-7/3/91	A-PE 1	ND	ND	
Toothacher Cove, Pen 10	6/5-7/3/91	B-PE 10	ND	ND	
Toothacher Cove, Pen 10	6/5-7/3/91	C-PE 20	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	P WEST	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PW10	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PW20	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	P EAST	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PE10	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PE20	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	P NORTH 1	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PN10	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PN20	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	P SOUTH 1	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PS10	ND	ND	
Toothacher Cove, Pen 10	7/3-7/31/91	PS20	ND	ND	
Toothacher Cove, Pen 10	7/31-9/18/91	P WEST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	PW10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	PW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	P NORTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	PN10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	PN20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	P EAST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	P SOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	PS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	PS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	A WEST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	AW10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	AW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	A SOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	AS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	7/31-9/18/91	AS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	P EAST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	P WEST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	PW10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	PW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	P NORTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	PN10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	PN20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	P SOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	PS10	ND	ND	not done - excess sediment

Appendix 1 (cont.)

Site and date		Station[1]	Number of pellets		Comments
			Food	Faeces	
Toothacher Cove, Pen 10	9/18-11/5/91	PS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	A NORTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	AN10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	AN20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	A SOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	AS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	AS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	A WEST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	AW10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	9/18-11/5/91	AW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 10	11/5-11/20/91	ANORTH 1	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AN10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AN20	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	ASOUTH 1	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AS10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AS20	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AWEST 1	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AW10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	AW20	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	P SOUTH 1	10	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PS10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PS20	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PWEST 1	21	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PW10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PW20	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PNORTH 1	3	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PN10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PN20	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PEAST 1	9	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PE10	0	ND	not done - technician error
Toothacher Cove, Pen 10	11/5-11/20/91	PE20	0	ND	not done - technician error
Toothacher Cove, Pen 10	6/10-7/13/92	AS1M	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	AS10M	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	AS20M	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	AN1	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	AN10	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	AN20	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	PE10	0	0	
Toothacher Cove, Pen 10	6/10-7/13/92	PN20	0	0	
Toothacher Cove, Pen 2	7/2-9/17/91	A SOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/2-9/17/91	AS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/2-9/17/91	AS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/2-9/17/91	AWEST 1	ND	ND	not done - excess sediment

Appendix 1 (cont.)

Site and date		Station[1]	Number of pellets		Comments
			Food	Faeces	
Toothacher Cove, Pen 2	7/2-11/6/91	AW10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/2-11/6/91	AW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/11-7/30/91	A N 1M	0	0	
Toothacher Cove, Pen 2	7/30-9/17/91	AEAST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	AE10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	AE20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PNORTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PN10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PN20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PSOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PEAST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PE10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PE20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PW10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/30-9/17/91	PW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PSOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PNORTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PN10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PN20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PEAST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PE10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PE20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	PWEST 1M	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	ASOUTH 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	AS10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	AS20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	AEAST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	AE10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	AE20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/6/91	AWEST 1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/19/91	PW20	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	9/17-11/19/91	PW 10M	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	11/6-11/19/91	AEAST 1	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	AE10	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	AE20	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	ASOUTH 1	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	AS10	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	AS20	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	AWEST1	0	ND	not done - technician error

Appendix 1 (cont.)

Site and date		Station[1]	Number of pellets		Comments
			Food	Faeces	
Toothacher Cove, Pen 2	11/6-11/19/91	AW20	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PNORTH1	41	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PN10	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PN20	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PSOUTH1	9	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PS10	10	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PS20	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PEAST1	17	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PE10	0	ND	not done - technician error
Toothacher Cove, Pen 2	11/6-11/19/91	PE20	0	ND	not done - technician error
Toothacher Cove, Pen 2	6/11-7/14/92	A E1M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	A E10M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	A E20M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	AS1M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	AS10M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	AS20M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PE1M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PE10M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PE20M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PN1M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PN10M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PN20M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PS1M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PS10M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PS20M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PW1M	0	0	
Toothacher Cove, Pen 2	6/11-7/14/92	PW10M	0	0	
Toothacher Cove, Pen 2	7/14-10/16/92	AN1	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/14-10/16/92	AN10	ND	ND	not done - excess sediment
Toothacher Cove, Pen 2	7/14-10/16/92	AN20	ND	ND	not done - excess sediment
Lubec - Depo 1	7/8-7/23/93	UNDER PEN A	0	0	
Lubec - Depo 1	7/8-7/23/93	UNDER PEN B	3	0	
Lubec - Depo 1	7/8-7/23/93	PEN EDGE A	0	0	
Lubec - Depo 1	7/8-7/23/93	PEN EDGE B	4	0	
Lubec - Depo 1	7/8-7/23/93	5M A	1	0	
Lubec - Depo 1	7/8-7/23/93	5M B	1	0	
Lubec - Depo 1	7/8-7/23/93	A 100M A	0	0	
Lubec - Depo 1	7/8-7/23/93	A 100M B	0	0	

[1] Trap positions indicated by: 1) *P* for pen or *A* for ambient; 2) *N*, *E*, *S*, or *W* for north, east, south, or west; and 3) *##M* for distance from pen or imaginary pen (at ambient sites). An *A* or *B* indicates replicate traps at a single site.

CHAPTER 5

A BENTHIC ENRICHMENT INDEX

by

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ABSTRACT

Hargrave, B.T. 1994. A benthic enrichment index, p. 79-91. *In* B.T. Hargrave [ed.]. Modelling Benthic Impacts of Organic Enrichment from Marine Aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

Organic carbon sedimentation rates (SR) proximal to and distant from coastal sites used for finfish and molluscan aquaculture were summarized and compared to a new variable (benthic enrichment index, BEI) calculated as the product of organic carbon and redox potentials in surface sediments at various sites. The logarithm of SR was inversely correlated with BEI for values of $SR > 1$ g C m^{-2} d^{-1}. When organic carbon sedimentation exceeds this rate, carbon loss through benthic aerobic and anaerobic respiration is insufficient to prevent organic carbon accumulation and anoxic conditions prevail. This leads to negative redox potentials and negative values for BEI. The empirical relationship predicts when the waste stream from an aquaculture site is of sufficient magnitude to cause sediments to become permanently anoxic and dominated by white sulfur bacterial mats.

INTRODUCTION

Although models have been proposed to predict impacts of organic loading under various aquaculture and environmental conditions, as discussed in the Preface of this report, few general relationships have been derived to show how geochemical change in sediments under net-pens are altered by increasing organic matter supply. Such relationships, if generally applicable, could be used to describe existing conditions or to predict changes with the establishment of new or expanded aquaculture development.

Sediments at existing or planned aquaculture sites might be used to rank impacts of benthic organic enrichment. Sediment grain size, water content, organic matter concentration, oxidation-reduction potentials, and benthic macrofauna biomass have been used as input parameters in models and aquaculture impact monitoring programs (Gowen and Bradbury 1987; Gowen et al. 1989; Wildish et al. 1990; Silvert 1992). However, these measurements have not previously been directly compared to the deposition (sedimentation) of organic matter under and adjacent to aquaculture sites. Recently, a series of measurements of particulate organic matter sedimentation have been made under finfish and molluscan aquaculture sites (Table 1). The rates of organic carbon sedimentation, spanning two orders of magnitude, are in general one to two orders of magnitude greater than values (0.1 to 1 g C m^{-2} d^{-1}) observed in temperate marine coastal waters where particle fluxes are not elevated through sediment resuspension, river, or urban discharges (Hargrave 1985).

Gowen et al. (1991) reviewed many of the variables that prevent direct comparisons of measurements of sedimentation under fish pens derived from different studies. They concluded that the most important variable affecting organic carbon flux through salmonid cage farms is fish stocking density. In three different studies, the proportion of organic carbon retained by harvested fish was constant (21 to 23% of the total organic carbon fed) with 75 to 85% lost as soluble and insoluble inorganic and organic carbon. Losses of organic carbon through particulate matter sedimentation were highly variable between studies (7 to 66%), indicative of problems associated with using sediment traps to measure particulate fluxes from cages.

AN INDEX OF BENTHIC ENRICHMENT

Benthic impacts due to organic enrichment could be predicted by comparing rates of organic carbon sedimentation with variables expected to change with organic loading. For example, measurements such as macrofauna biomass or dissolved nutrient and gas fluxes across the sediment surface could be used along with data for sediment texture and variation in current speed to derive an index for estimating impacts of increased organic loading. However, these data are usually not available for most existing or potential aquaculture sites.

Table 1. Organic carbon sedimentation associated with different marine aquaculture sites for various molluscan and finfish species. Data as ranges or single observations from published and unpublished sources are ranked by magnitude of flux rates. "*" indicates that organic matter collected in sediment traps was assumed to contain 50% organic carbon.

Location	Species	Sedimentation $g\ C\ m^{-2}\ d^{-1}$	Reference
Finfish:			
Norwegian fjord	Salmon	4-181*	Ervik et. al. (1985)
Gullmar Fjord	Trout	33-78	Hall et al. (1990)
Scottish sea lochs	Salmon	23-35*	Gowen et al. (1991)
Japanese coast	Salmon	12-17*	Kadowaki et al. (1980)
Bay of Fundy	Salmon	0.3-15	Hargrave (unpublished data)
Norwegian coast	Salmon	1-15*	Hansen et al. (1991)
Maine coastal inlet	Salmon	1-6	Findlay et al. (1994)[1]
Nubeema, Tasmania	Salmon	2-6	Ye et al. (1991)
Shellfish:			
French coast	Oysters	8-99	Ottman and Sornin (1985) *In* Pillay (1992)
Hiroshima Bay, Japan	Oysters	14	Anakawa et al. (1973) *In* Pillay (1992)
Swedish coast	Mussels	2-3	Dahlback and Gunnarsson (1981)
Nova Scotia inlet	Mussels	2-4	Grant et al. (1994)
N. Baltic Sea	Mussels	0.1-1	Kautsky and Evans (1987)

[1] See also Chapter 4.

Oxidation reduction (redox) potentials and organic carbon are two variables that are often used to characterize sediments in studies of eutrophication associated with aquaculture operations (Wildish et al. 1990). Limitations to the use of sedimentary organic matter as an index of organic enrichment have been identified (Abdullah and Danielsen 1992; Rowan et al. 1992). However, in areas not receiving large amounts of terrigenous or inorganic material which will dilute deposited organic matter, a relationship between rates of supply of particulate organic carbon through sedimentation and the amount of organic carbon stored in sediments might be expected. Correlations between sediment accumulation and organic carbon burial rates are discussed further in

Chapter 6. Actual organic carbon burial rates in sediment reflect the relative time scales of sedimentation, remineralization, and the kinetics of aerobic and anaerobic mineralization processes (Henrichs and Reeburgh 1987; Burdige 1991).

Data for sediment organic carbon (SOC as mol C m^{-2} for the upper 1 cm layer) and redox potential (Eh as mv at 1 cm depth) were compared with measured rates of organic carbon sedimentation (SR as g C m^{-2} d^{-1}) for locations where all three variables were measured (Table 2). The data are representative of marine areas where molluscan and finfish aquaculture occurs (Gullmar Fjord, Bliss Harbour, Upper South Cove, Swans Island), experimental marine mesocosms (MERL), coastal embayments (St. Margaret's Bay, Bedford Basin, St. Georges Bay, and continental shelf areas near Nova Scotia (Georges Bank and Scotian Shelf). Correlation matrices (Table 3a) showed that a significant ($p<0.05$) negative correlation existed only between $\log_{10}$ SR and Eh ($n=23$, $r^2=0.65$). There was a positive relationship between $\log_{10}$ SR and SOC and an inverse relation between Eh and SOC.

A new variable (benthic enrichment index, BEI) was calculated as SOC x Eh. For the total data set ($n=23$), this index was inversely correlated with untransformed and logarithmically transformed values of SR ($r^2=0.64$). A plot of SR against BEI (Fig. 1) shows that data clustered into two groups for values of SR $>$and$<$ 1 g C m^{-2} d^{-1}. There was no relationship ($p<0.05$) between SR and BEI for values of SR <1 g C m^{-2} d^{-1}. Excluding this data ($n=10$) from the regression calculations increased the correlation coefficient between SR and BEI ($n=13$, $r^2=0.73$).

Analysis of residuals in the regression showed that data from the MERL mesocosms with dissolved nutrient enrichment were separated from other values with measures of SR lower than would be predicted on the basis of data from other areas. Removal of these data ($n=5$) from the regression calculations improved all correlation coefficients (Table 3b). Log_{10} SR was significantly correlated with SOC ($r^2=0.70$), but not with Eh ($r^2=0.33$) as in the full data set. A step-wise multiple linear regression showed that Eh and SOC accounted for 33.3% and 41.8% of the total variance, respectively. Although SOC and Eh were not significantly correlated, there was an inverse relation between the two variables ($r^2=0.45$) (Table 3b). The relationship between $\log_{10}$ SR and BEI for the reduced data set was described by the equation:

$$\mathrm{BEI} = 956.4 - 3508.6 \log_{10} \mathrm{SR} \qquad (n=9,\ r^2=0.81,\ p=0.001) \tag{1}$$

A similar calculation for the data from the MERL mesocosms enriched with dissolved nutrients yielded the equation:

$$\mathrm{BEI} = 890.1 - 7381.1 \log_{10} \mathrm{SR} \qquad (n=5,\ r^2=0.91,\ p=0.012) \tag{2}$$

Table 2. Summary of sediment organic carbon (SOC), oxidation reduction potentials (Eh) (in the upper 1 cm layer of surface sediment) and organic carbon sedimentation rates (SR) in different coastal locations. A benthic enrichment index (BEI) is calculated as the product of SOC (mol C m^{-2}) x Eh (mv). Values of SR are individual measurements or calculated as means from ranges in parentheses.

Location	SOC (mol C m^{-2})	Eh (mv)	BEI	SR ($g\ C\ m^{-2}\ d^{-1}$)		Reference
Gullmar Fjord (Sweden)	36.0	-160	-5760	56	(33-78)	Hall et al. (1990)
Bliss Harbour	17.5	-160	-2800	15	(pen)	Hargrave et al. (1993)
L'Etang Inlet	17.5	-100	-1750	7.8	(pen)	and B.T. Hargrave
N.B.	10.5	+130	+1365	1.2	(pen edge)	(unpubl. data)
(Canada)	12.1	+85	+1029	1.4	(pen edge)	
	10.1	+109	+1101	0.3	(control)	
	10.5	+124	+1302	0.3	(control)	
Upper South Cove, N.S. (Canada)	18.3	-100	-1830	11.6	(0.5-22.7) (mussel lines)	Grant et al. (1994) and B.T. Hargrave (unpubl. data)
Swans Island	15.0	-100	-1500	3.0	(pen edge)	Findlay et al. (1994) and
Maine (U.S.A.)	10.5	+100	+1050	0.7	(control)	Chapter 4 of this report
Georges Bank	2.7	+200	+534	2.0		B.T. Hargrave (unpubl. data)
MERL	50	-50	-2500	3.2	(x32)	Sampou and Oviatt (1991a)
Mesocosms	40	-50	-2000	2.3	(x16)	(carbon flux calculated
with nutrient	40	-5	-200	1.2	(x8)	as the sum of benthic
enrichment	40	+5	+200	1.1	(x4)	aerobic + anaerobic
	40	+50	+2000	0.9	(control)	respiration)
with sludge enrichment	5.0	-200	-1000	4.1	(8S)	Sampou and Oviatt (1991b)
St. Margaret's Bay, N.S. (Canada)	5.5	+150	+825	0.8		B.T. Hargrave (unpubl. data) and Webster et al. (1975)
Bedford Basin	12.6	+69	+868	0.2	(0.1-0.3)	Prouse and Hargrave (1987),
Halifax Inlet	6.6	+164	+1082	0.6	(0.4-0.8)	Hargrave (1980); Novitsky
N.S. (Canada)						(1990)
St. Georges Bay	3.5	+150	+525	0.2	(0.05-0.5)	Hargrave and Phillips (1986)
N.S. (Canada)	2.5	+200	+500	0.06	(0.05-0.1)	B.T. Hargrave (unpubl. data)
Scotian Shelf	2.4	+200	+480	0.14	(0.1-0.2)	B.T. Hargrave (unpubl. data)

Table 3. Matrix of Pearson correlation coefficients for linear regressions between variables listed in Table 2. Full data set includes all data. Values from sites where SR$<$1.0 g C m^{-2} d^{-1} and for mesocosms enriched with dissolved nutrients were removed to create the reduced data set. Values underlined indicate that correlations are significant ($p<0.05$).

A. Full data set (n=23)

Variable	SOC	log SOC	Eh	BEI
Eh	-0.438	-0.581	-	-
BEI	-0.406	-0.428	0.787	-
SR	0.284	0.316	-0.496	-0.779
log SR	0.426	0.536	-.805	-0.808

B. Reduced data set (n=9).

Variable	SOC	log SOC	Eh	BEI
Eh	-0.454	-0.545	-	-
BEI	-0.849	-0.718	0.725	-
SR	0.896	0.647	-0.451	-0.847
log SR	0.838	0.685	-.577	-0.902

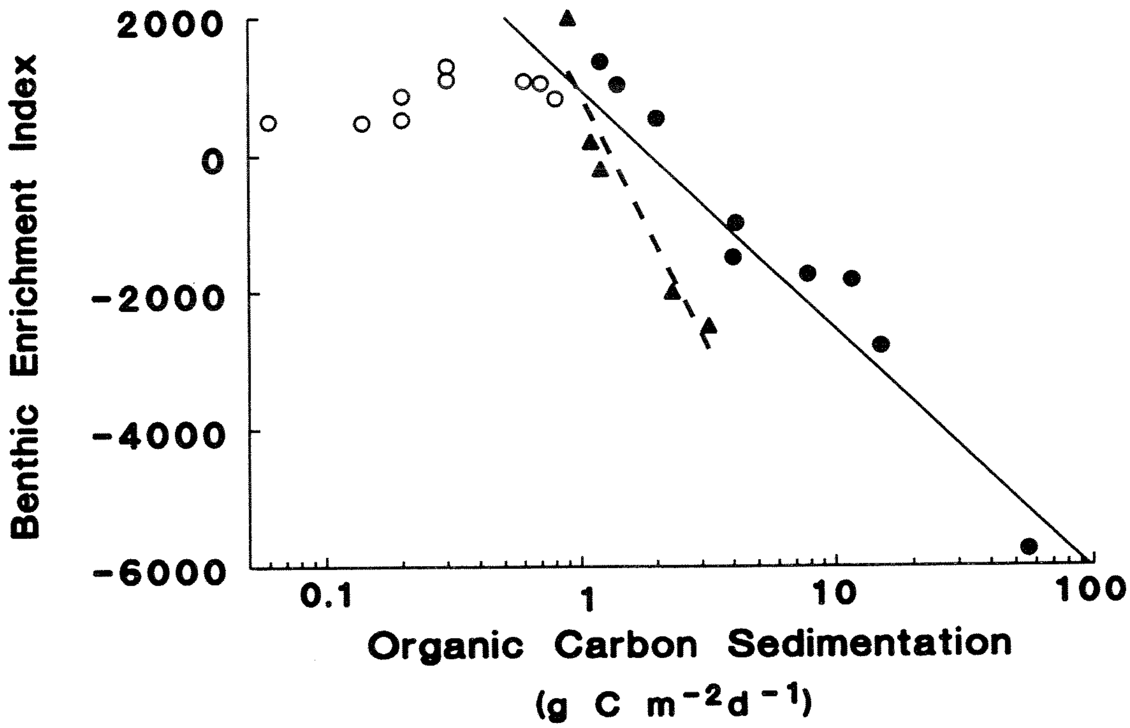

Figure 1. The relationship between organic carbon sedimentation rate (SR) and a benthic enrichment index (BEI) (organic carbon content (mol C m^{-2}) in the surface 1 cm sediment layer x redox potential (mv) measured at 1 cm depth). Data from Table 2. Data from sites with SR >1 g C m^{-2} d^{-1} plotted as solid circles (solid regression line BEI = 956.4-3508.6 $\log_{10}$ SR, n=9, r^2=0.814) and from mesocosms enriched with dissolved nutrients as solid triangles (dotted regression line BEI = 890.1-7381.1 $\log_{10}$ SR, n=5, r^2=0.911). Open circles indicate data from sites where SR <1 g C m^{-2} d^{-1}.

DISCUSSION

The production of anoxic conditions in sediments as a result of high rates of organic matter supply has been noted in previous studies as discussed in Chapter 4. Data presented in this chapter show that organic carbon sedimentation rates greater than approximately 1 g C m^{-2} d^{-1} can be predicted from the BEI index. The inverse relationship between organic carbon input and the product of Eh potentials and organic carbon accumulated in surface sediments is primarily due to the inability of organisms responsible for aerobic and anaerobic respiration to decompose organic matter deposited organic matter at high rates which leads to sulfide accumulation and increasingly negative redox potentials. Oviatt et al. (1987) observed that input of sewage sludge to marine sediments at rates >1 g C m^{-2} d^{-1} caused anoxic conditions. Additions of dissolved inorganic nutrients to mesocosms to create a eutrophication gradient stimulated organic carbon sedimentation and anaerobic metabolism was the predominant pathway for benthic carbon remineralization in anxoic sediments. Rates of sulfate reduction were correlated with seasonal changes in temperature, while Eh potentials decreased and organic carbon stored in sediments increased as a result of increased organic matter sedimentation.

Regions of rapid sedimentation are associated with accumulation of sediment organic matter on a global scale (Premuzic et al. 1982; Reimers et al. 1992). However, the preservation of organic carbon in sediments is not only related to the supply of particulate organic matter. The source of sedimented organic matter, the presence of redox sensitive metals such as iron and manganese, and bioturbation due to animal burrowing and feeding activities also control rates of organic carbon burial and degradation (Berner 1982; Emerson et al. 1985; Burdige 1991). Biotic processes such as the colonization of surface sediments by microbial mats can also alter thresholds of sediment erosion (Grant and Gust 1987).

The correlation between organic carbon sedimentation and BEI was maximized for data from locations where SR was >1 g C m^{-2} d^{-1} (Table 3b). Eh potentials in surface sediments at these sites are predominantly negative, yielding a negative BEI (Table 2). Although information for sediment grain size was not available for all sites listed in Table 2, organically rich deposits with low Eh potentials tend to be fine-grained and to occur in areas where current speeds are relatively low (<10 cm s^{-1}). Effects of sediment grain size, water current speed, and dissolved oxygen supply to the sediment water interface are thus embedded in the calculation of BEI by inclusion of the variables SOC and Eh.

The different intercepts and slopes for the regressions between $\log_{10}$ SR and BEI for sites with SR >1 g C m^{-2} d^{-1} (Equation 1, $b=-3508.6$) and from mesocosms enriched with dissolved nutrients (Equation 2, $b=-7381.1$) show that the source of organic matter added to sediments is important in controlling redox conditions and organic carbon burial rates. Data from the mesocosms, where the input of dissolved nutrients stimulated phytoplankton production, indicate that small increases in organic carbon sedimentation rate lead to relatively large changes in BEI. By substitution in Equation 2, increasing SR from 0.9 to 3.2 g C m^{-2} d^{-1} decreases BEI from +1228 to -2839. A similar substitution in Equation 1 for data from sites where sedimentation exceeded 1 g C m^{-2} d^{-1} results in values of BEI that decrease from +1117 to -816.

Organic carbon freshly produced by phytoplankton and macrophytes is rapidly mineralized by bacterial activity with turnover times of weeks to months (Hargrave and Phillips 1989; Sun et al. 1991). With aging, however, organic matter such as that buried in sediments becomes refractory to degradation resulting in longer decay times (Boudreau and Ruddick 1991). For this reason, residual organic carbon buried in sediments is in itself generally considered not to be a useful measure of nutritional quality for benthic deposit feeding invertebrates (Watling 1991). However, the present comparison of data from different aquaculture sites and enriched mesocosms shows that when organic carbon sedimentation is >approximately 1 g C m^{-2} d^{-1}, organic carbon accumulates in sediments at rates that exceed mineralization. Anoxic conditions that predominate under conditions of high rates of organic carbon input result in low Eh potentials associated with anaerobic microbial respiration.

The product of sedimentary organic carbon and Eh, which can be used to determine a BEI value, could be used to determine the level of benthic organic enrichment measured as SR at existing or potential aquaculture sites (Fig. 1), avoiding the need to actually measure sedimentation rates. The threshold value of 1 g C m^{-2} d^{-1} with an associated BEI value of about +1000 can further be used as a standard for site assessment where sites that maintained this value would be considered not to be negatively impacted by excess organic matter input. When related to current speed and sediment texture, these variables could be used to quantitatively position a benthic site along a scaled continuum between oxic and anoxic conditions that change as a result of variable rates of organic carbon supply below and above the threshold value of approximately 1 g C m^{-2} d^{-1}. Also, since SOC and Eh have been measured in surface sediments in many previous studies, where measurements of SR do not exist, Equation 1 or 2 could be used to predict SR for locations where values exceed 1 g C m^{-2} d^{-1}. Location of net-pen aquaculture development at these sites should probably be discouraged since increased organic carbon sedimentation would serve to reduce BEI values to below +1000. If aquaculture development occurred, the degree of impact could be determined by monitoring SOC and Eh and calculating values of BEI over time.

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CHAPTER 6

DISSOLVED AMMONIUM AND SULFATE GRADIENTS IN SURFICIAL SEDIMENT PORE WATER AS A MEASURE OF ORGANIC CARBON BURIAL RATE

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ABSTRACT

Cranston, R. 1994. Dissolved ammonium and sulfate gradients in surficial sediment pore water as a measure of organic carbon burial rate, p. 93-120. *In* B.T. Hargrave [ed.]. Modelling Benthic Impacts of Organic Enrichment from Marine Aquaculture. Can. Tech. Rep. Fish. Aquat. Sci. 1949: xi + 125 p.

The rate at which organic carbon accumulates in marine sediment columns depends on the supply and removal rates. Removal processes include organic matter mineralization which in anoxic sediments consumes sulfate and releases ammonium. The downcore concentration profiles for these ions are related to the remineralization rate, which is ultimately dependent on the supply of organic carbon.

Dissolved ammonium and sulfate profiles in sediment pore water have been measured in cores from more than 100 sites and compared with known organic carbon burial rates. A relationship between these variables has been estimated for carbon burial rates between 10^{-6} and 10^{1} g C m^{-2} d^{-1}. The low rates correspond to deep ocean regions with sediment accumulation rates on the order of mm ka^{-1} where sediment organic carbon concentrations are <0.1%. The high rates occur in impacted areas found under salmon aquaculture cages where sediment accumulation rates are on the order of cm a^{-1} and sediment organic carbon concentrations are ≥3%. By measuring dissolved ammonium and sulfate gradients, an immediate, inexpensive estimate of organic carbon burial rate can be interpolated from the gradient/carbon burial relationship.

Since the gradients are in equilibrium with present-day redox reactions, the proposed method provides quantitative estimates of present-day organic carbon burial rates, rather than an estimate of burial rate in the past. It minimizes errors due to bioturbation and physical mixing in the sediment. After a mixing event, the sediment record is permanently disturbed, while the pore water components immediately begin to be redistributed as they diffuse through the mixed sediment. This compensates for core sampling problems, such as mixing or loss of upper sediment core layers. The method can be applied to compare present-day organic carbon burial rates at proposed aquaculture sites and to monitor burial between and within regions. Flux estimates measured with sediment traps measure organic carbon "on the way to the bottom," much of which is remineralized, grazed, and physically removed from the site. Analyses of sulfate and ammonium gradients in sediment pore water allows an estimate of net organic carbon burial following all removal processes.

INTRODUCTION

Deep sea sediment columns often receive limited amounts of organic carbon, resulting in low carbon burial rates and organic carbon concentrations. For example, deep Pacific Ocean sediments receive 2 x 10^{-5} g C m^{-2} d^{-1}, and contain 0.2% organic carbon (Berner 1982). Redox conditions reflect this, as there is minimal organic matter remineralization and limited amounts of ammonium released to and sulfate consumed in the pore water. In coastal areas where increased amounts of organic matter reach the sediment, more organic mineralization occurs in the sediment column. For example, average continental margins receive 0.3 g C m^{-2} d^{-1} and the sediments contain on average 2% organic carbon (Berner 1982). This increases the rate that ammonium is released to and sulfate consumed in the pore water. As a result, concentration-depth relationships are enhanced as organic matter deposition increases.

Relationships between organic carbon concentrations and sediment accumulation rates have been provided by many workers, implying that more organic carbon is preserved as sediment accumulation rates increase (e.g. Muller and Suess 1979; Stein 1986; Henrichs and Reeburgh 1987; Ingall and Van Cappellen 1990; Betts and Holland 1991). Redox potentials, as discussed in Chapter 5, also tend to correlate with either sediment organic carbon concentrations or the sedimentation of freshly deposited organic carbon. In this chapter, the concentration of organic carbon and sediment accumulation rate are combined to provide an actual measure of carbon burial flux which is then compared to redox processes.

Problems associated with carbon burial estimates can result because of sample quality. Coring problems, or reworking of the sediment column, result in large error bars on the deposition rate estimates. As a result, there is very little reliable information on present-day carbon burial rates. This is especially true for shallow coastal waters where resuspension due to wind, wave, and tidal action is increased. If surficial sediments are used to indicate redox reactions, samples from the top of the sediment column are required. Often, surface sediment layers are lost or disturbed during collection. Piston coring can miss sediment sections, or suction in excess sediment, thus degrading the stratigraphic record. Most measures of sediment accumulation and carbon burial are averaged over long time frames. For assessing the benthic impacts of organic matter accumulation associated with finfish and molluscan aquaculture, it is more important to determine the present-day carbon burial rates, rather than rely on estimates averaged over decades or millennia.

The purpose of this chapter is to present a method which estimates present-day carbon burial rates (CBR) using ammonium and sulfate concentration-depth variations. Results are summarized from more than 100 sites, many of which have independent measurements of carbon burial rate, based on organic carbon measurements and sediment accumulation rate estimates from isotopic, biostratigraphic, and observational data.

METHODS

Ammonium and sulfate gradients were measured by collecting sediment cores and removing pore water at selected intervals downcore. Many of the pore water data presented are for samples collected and analyzed by the geochemistry laboratory at the Atlantic Geoscience Centre. About 40 ml of wet sediment was selected from each depth interval and placed in a 50 ml plastic centrifuge tube. The samples were centrifuged at 6000 g's in a 4°C cold room. The pore water was decanted off the sediment and filtered through 1 μm pore diameter filters. Ammonium was measured using 100 μl of pore water and a standard colorimetric method (Solorzano 1969). Sulfate was measured in 50 μl of pore water using a turbidometric method which requires barium chloride to precipitate sulfate.

The remainder of the results are taken from the literature, including the Ocean Drilling Program and a number of site-specific studies, where pore water analyses were done, where sediment accumulation rates were measured, and where organic carbon content of the sediment samples was known. A summary of the site data and references is available in Table 1. CBR is calculated as the product of the organic carbon concentration and sediment accumulation rate. When multiplied and converted, final units are g C m^{-2} d^{-1}, which is defined as the carbon burial rate (CBR).

RESULTS AND DISCUSSION

The controlling condition for ammonium production and sulfate consumption is that anoxic sediments are present. This occurs at some depth downcore in the sediment column, since oxygen and other oxidants are generally available to diffuse into the sediment from overlying water. If very little carbon is being buried, oxygen will penetrate many metres into the sediment column, and the anoxic reactions will occur below this depth. As a result, long sediment cores are required. If CBR is high, oxidants may penetrate millimetres to decimeters into the sediment column. In these cases, shorter sediment cores are required to collect pore waters from the anoxic zone. Data from over 200 sites are presented in Table 1.

Examples of different carbon burial rates are shown in Figures 1 to 4. Figure 1 includes concentration profiles for ammonium, sulfate, and organic carbon in a low depositional site in the South Indian Ocean (Ocean Drilling Project ODP Site 745). Note that the length of the core exceeds 200 m. Sediment accumulation rate is 30 m ma^{-1} and the carbon burial rate is 8 x 10^{-5} g C m^{-2} d^{-1}, calculated from the average organic carbon concentration and known sediment accumulation rate. Figure 2 depicts the concentration profiles for the same variables in Emerald Basin on the Scotian Shelf, in 240 m of water (Site 92003-19). The core length is 7.5 m, the sediment accumulation rate is 400 m ma^{-1}, and the carbon burial rate is 0.02 g C m^{-2} d^{-1}. Figure 3 shows the concentration profiles for a site in Halifax Harbour (Site hfxcreed33). The core length is 60 cm, the sediment accumulation rate is 3000 m ma^{-1}, and the carbon burial rate is 0.3 g C m^{-2} d^{-1}. Figure 4 shows the concentration profiles

Table 1. Summary of ammonium and sulfate gradients and organic carbon for all sites and sediment accumulation rates and calculated carbon burial rates for selected sites. Reference numbers refer to the following references: 1) Cranston (unpubl. data); 2) Cranston (1991); 3) Buckley et al. (1989a) (md45-GME); 4) Buckley et al. (1991); 5) Fitzgerald et al. (1989); 6) Fitzgerald et al. (1991); 7) Winters et al. (1987); 8) Buckley et al. (1989b) (md45-nares); 9) Ocean Drilling Program; 10) LeBlanc et al. (1991); 11) Reimers et al. (1992); 12) Rosenfeld (1983); Westrich (1983); 13) Jorgensen et al. (1990); 14) Sholkovitz (1973); 15) Elderfield et al. (1981); and 16) Val Klump and Martens (1989).

Site	Water Depth (m)	Latitude (degrees)	Longitude (degrees)	Organic Carbon (%)	Sediment Accumul. Rate (m/ma)	Sulfate Gradient (-mM/m)	Ammonium Gradient (mM/m)	Carbon Burial Rate (gC/m^2/d)	Reference
sachem	3	41.3	-73.7	2.00	5000	4E+01	1E+01	2E-01	12
nargbay	3	41.7	-71.4	3.00	2000	4E+01	1E+01	1E-01	15
let920609	6	45.1	-66.7	0.80			4E-01		1
let920607	6	45.1	-66.8	2.10		1E+02	6E+00		1
hfx8900924	8	44.6	-63.6	4.00	2000	5E+01	4E+00	2E-01	10
let920604	8	45.1	-66.8	1.40			2E+00		1
hfxcreed35	8	44.6	-63.6	6.00		1E+02	1E+01		4
northcar	8	34.6	-76.5	3.00	100000	3E+02	8E+01	6E+00	16
hfxcreed33	8	44.6	-63.6	5.00	3000	1E+02	8E+00	3E-01	4
foam	9	41.3	-73.7	1.00	1000	2E+01	4E+00	2E-02	12
hfx8903902	10	44.6	-63.6	2.00		6E+01	4E+00		4
hfxcreed32	10	44.6	-63.6	6.00	6000	6E+01	3E+00	7E-01	4
hfx9001014	10	44.6	-63.5	1.00		4E+01	3E+00		6
let920603	12	45.1	-66.8	1.30			2E+00		1
let920605	12	45.1	-66.8	1.20			9E-01		1
let920608	12	45.0	-66.8	1.40			7E-01		1
let9102400	15	45.0	-66.8	2.80			2E+01		1
let9102250	15	45.0	-66.8	7.10	20000	3E+02	6E+01	3E+00	1
let9102500	15	45.0	-66.8	1.80			3E+00		1
let91020	15	45.0	-66.8	1.80			2E+00		1
hfx8900916	15	44.6	-63.6	4.00	4000	3E+01	6E+00	3E-01	10
bh	15	41.3	-73.9	3.00	50000	3E+02		3E+00	12
let9102150	15	45.0	-66.8	6.20	10000	5E+02	1E+02	1E+00	1
hfx8900903	15	44.7	-63.7	5.00	2000	2E+01	3E+00	2E-01	10
let920601	15	45.0	-66.8	1.10			2E+00		1
nwc	16	41.2	-73.9	1.00	500	2E+00		1E-02	12
bbj90-8	17	56.0	10.5	3.00	500	1E+01	9E-01	3E-02	13
bliss910804	17	45.0	-66.8	1.50			2E+00		1
bliss910801	18	45.0	-66.8	1.80			5E+00		1
bliss910802	18	45.0	-66.8	1.60			2E+00		1
bbj90-12	20	58.2	10.1	2.00	3000	3E+01	2E+00	1E-01	13
bliss910803	24	45.0	-66.8	1.60			1E+00		1
bbj90-2	25	55.5	10.8	3.00	300	1E+01	1E+00	2E-02	13
bbj90-10	73	57.8	11.1	2.00	6000	3E+01	3E+00	2E-01	13
hfx8900901	70	44.7	-63.6	4.00	3000	4E+01	8E+00	2E-01	10
hfx9001016	70	44.7	-63.6	5.00	3000	2E+02	1E+01	3E-01	6
hfx9001018	70	44.7	-63.6	7.00	2000	1E+02	8E+00	3E-01	6
8801001	70	44.7	-63.6	3.00			3E+00		5
8801007	200	43.6	-63.6	1.00		8E-01	1E-01		5
9200316	200	43.7	-62.8	0.40		5E+00	3E-01		1
slope8801002	200	43.7	-62.8	0.50	200	2E+00	2E-01	2E-03	5
8801009	200	43.6	-63.6	1.00		8E-01	1E-01		5
9200303	236	43.9	-62.8	2.00			3E-01		1
slope8801005	240	43.9	-62.8	0.90	400	3E+00	2E-01	7E-03	5
9200317	240	43.9	-62.8	2.00	400	6E+00	5E-01	2E-02	1
9200304	243	43.9	-62.8	2.00		3E+00	3E-01		1
9200302	243	43.9	-62.8	2.00		4E+00	3E-01		1
9200318	266	43.9	-62.9	2.00			6E-01		1
pok9200319	270	43.9	-62.9	1.30		1E+01	8E-01		1
687	307	-12.9	-77.0	3.00	65	8E-01	1E-01	4E-03	9
725	311	18.5	57.7	0.70	120	2E-01		2E-03	9
739	412	-67.3	75.1	0.50	20	9E-02	5E-03	2E-04	9
684	426	-9.0	-79.9	4.00	30	9E-01	2E-01	2E-03	9
686	447	-13.5	-76.9	2.50	160	2E+00	6E-01	8E-03	9
815	465	-19.2	150.0	0.10	20	2E-01	2E-02	4E-05	9

Table 1. Cont…

Site	Water Depth (m)	Latitude (degrees)	Longitude (degrees)	Organic Carbon (%)	Sediment Accumul. Rate (m/ma)	Sulfate Gradient (-mM/m)	Ammonium Gradient (mM/m)	Carbon Burial Rate (gC/m^2/d)	Reference
741	551	-68.4	76.4	0.20	20	2E-01	5E-03	8E-05	9
737	564	-50.2	73.0	0.30	9	3E-02	2E-03	5E-05	9
819	565	-16.6	146.3	0.50	100	1E+00	7E-02	1E-03	9
9200322	570	42.9	-62.2	0.70			2E-01		1
santabar	580	34.3	-120.0	3.00	2000	2E+01	3E+00	1E-01	14
okhotsk9104118	590	42.5	131.3	1.00			5E-01		2
724	590	18.4	57.8	1.00	86	5E-01		2E-03	9
okhotsk910373	600	44.2	136.4	0.80			7E-01		2
okhotsk910267	601	53.2	144.4	1.00		1E+01	5E-01		2
okhotsk910257	628	53.4	144.4	1.00			5E-01		2
okhotsk910252	642	53.4	144.4	1.00		2E+01	7E-01		2
okhotsk910251	643	53.4	144.4	1.00		1E+01	6E-01		2
okhotsk910254	645	53.4	144.4	1.00		1E+01	5E-01		2
okhotsk910256	645	53.4	144.4	1.00		3E+00	3E-01		2
okhotsk910253	646	53.4	144.4	1.00		3E+01	2E+00		2
okhotsk910384	700	44.2	136.4	1.00		3E+00	2E-01		2
okhotsk910244	708	54.4	144.1	1.30		9E+01	3E-01		2
okhosk910240	708	54.4	144.1	1.30		6E+01	3E-01		2
okhotsk910241	708	54.4	144.1	1.30		3E+01	2E-01		2
okhotsk910243	725	54.4	144.1	2.00		1E+01	1E+00		2
818	749	-18.1	150.0	0.30	60	3E-02		3E-04	9
9200330	750	42.9	-62.2	0.80		3E+00	2E-01		1
9200332	750	42.9	-62.2	0.80			2E-01		1
okhotsk9104102	780	42.5	132.7	1.00			2E-01		2
okhotsk910133	798	50.5	155.3	0.60		2E+01	8E-01		2
okhotsk910119	800	50.5	155.3	0.80		7E+00	5E-01		2
okhotsk910110	800	50.5	155.3	0.90			6E-01		2
okhotsk910101	800	50.5	155.3	0.90			2E-01		2
okhotsk910117	800	50.5	155.3	1.00			6E-01		2
reimer-j	800	35.6	-121.6	3.00	100		5E-01	6E-03	11
okhotsk910118	804	50.5	155.3	0.60		1E+01	6E-01		2
740	807	-68.7	76.7	1.00	20	6E-02		4E-04	9
723	808	18.1	57.6	1.00	170	6E-01		3E-03	9
okhotsk910269	822	52.8	144.8	2.00		9E+00	8E-01		2
okhotsk910270	833	52.6	144.9	2.00		1E+01	5E-01		2
okhotsk9104101	870	42.5	132.7	0.90			6E-01		2
okhotsk9104110	900	42.5	131.8	0.80			3E-01		2
okhotsk910383	900	44.2	136.5	1.00		2E+00	1E-01		2
japan910383	900	44.2	136.5	1.00		3E+00	2E-01		2
798	903	37.1	135.0	2.00	100	3E+00	2E-01	4E-03	9
727	914	17.8	57.6	2.00	110	5E-01		4E-03	9
okhotsk910385	962	44.1	136.6	1.00		3E+00	2E-01		2
slope9102074	972	47.0	-43.5	0.40			3E-02		1
slope9102069	990	43.3	-49.1	1.00			7E-01		1
reimer-k	1000	35.6	-121.8	4.00	80		3E-01	6E-03	11
754	1060	-30.9	93.6	0.05	3	7E-03		3E-06	9
slope9102040	1060	44.7	-55.6	0.70		3E+01	2E+00		1
749	1070	-58.7	76.4	0.05	1	9E-03		1E-06	9
730	1070	17.7	57.7	2.00	50	2E-01		2E-03	9
slope9200323	1090	42.8	-62.2	1.00			3E-01		1
752	1090	-30.9	93.6	0.05	2	3E-03		2E-06	9
slope9102079	1140	47.5	-46.6	0.40	300	4E+00	2E-01	2E-03	1
okhotsk9104111	1150	42.5	131.8	0.70			5E-01		2
753	1180	-30.8	93.6	0.05	1	1E-03		5E-07	9
okhotsk910386	1190	44.1	136.8	2.00		3E+00	3E-01		2

Table 1. Cont…

Site	Water Depth (m)	Latitude (degrees)	Longitude (degrees)	Organic Carbon (%)	Sediment Accumul. Rate (m/ma)	Sulfate Gradient (-mM/m)	Ammonium Gradient (mM/m)	Carbon Burial Rate (gC/m^2/d)	Reference
9200329	1220	42.7	-62.2	0.80			3E-01		1
748	1290	-58.5	79.0	0.05	5	4E-02		5E-06	9
slope9102029	1330	44.7	-55.5	0.70		2E+01	1E+00		1
slope8801018	1340	42.8	-61.6	0.70		3E+00	2E-01		5
762	1360	-19.9	112.2	0.10	20	3E-02		4E-05	9
763	1370	-20.6	112.2	0.10	20	6E-02		4E-05	9
reimer-d	1400	36.2	-122.3	2.00	10		2E-01	4E-04	11
728	1430	17.7	57.8	2.00	45	4E-01		2E-03	9
9200324	1470	42.6	-62.2	0.90			2E-01		1
okhotsk910388	1500	44.1	136.9	1.00			2E-01		2
756	1520	-27.3	87.6	0.05	5	6E-02		5E-06	9
707	1550	-7.5	59.0	0.05	11	2E-03		1E-05	9
751	1630	-57.7	79.8	0.10	3	2E-02		6E-06	9
757	1650	-17.0	88.2	0.05	10	6E-02		1E-05	9
japan910393	1735	44.0	137.0	1.00		5E+00	1E-01		2
9200328	1740	42.5	-62.2	0.90			2E-01		1
okhotsk910393	1740	44.0	137.0	0.70		4E+00	1E-01		2
792	1790	30.4	140.4	0.40	80	2E-01	2E-02	6E-04	9
okhotsk910394	1800	44.0	137.0	0.50		5E+00	1E-01		2
703	1800	-47.1	7.9	0.05	6	5E-03		6E-06	9
okhotsk910389	1820	44.0	137.0	1.00			2E-01		2
okhotsk910395	1850	44.0	137.0	1.00			1E-01		2
slope9102044	1910	44.5	-55.6	0.70			7E-01		1
721	1940	16.7	59.9	1.00	43	3E-01		8E-04	9
760	1970	-16.9	115.5	0.10	20	5E-02		4E-05	9
9200325	1980	42.5	-62.2	0.90			3E-01		1
722	2030	16.6	59.8	0.80	47	3E-01		7E-04	9
714	2040	5.1	73.8	0.50	40	2E-01	8E-03	4E-04	9
799	2070	39.4	133.9	1.00	100	2E+00	7E-02	2E-03	9
689	2080	-64.5	3.1	0.05	9	1E-02		9E-06	9
698	2140	-51.5	-33.1	0.05	10	1E-02		1E-05	9
761	2190	-16.7	115.5	0.10	10	2E-02		2E-05	9
790	2220	30.9	139.8	0.20	80	2E-01	7E-03	3E-04	9
738	2250	-62.7	82.8	0.05	1	9E-03	5E-05	1E-06	9
715	2270	5.1	73.8	0.50	15	2E-02		1E-04	9
744	2310	-61.6	80.6	0.05	5	5E-03	6E-05	5E-06	9
705	2320	-13.2	61.4	0.10	8	1E-02		2E-05	9
731	2370	16.5	59.7	0.50	38	2E-01		4E-04	9
9200331	2400	42.4	-62.2	0.70			2E-01		1
9200326	2410	42.4	-62.2	0.70			2E-01		1
857	2420	48.4	-128.7	0.50	30	2E-01	3E-02	3E-04	9
9200327	2420	42.4	-62.2	0.70		6E+00	4E-01		1
719	2500	-1.0	80.0	0.50	81	2E-01		8E-04	9
717	2500	-1.0	80.0	1.00	260	6E-01		5E-03	9
718	2500	-1.0	80.0	1.00	120	1E-01		2E-03	9
806	2520	0.2	159.4	0.20	20		5E-03	8E-05	9
704	2530	-46.9	7.4	0.20	20	3E-02		8E-05	9
796	2570	42.9	139.4	1.00	70	1E+00	1E-01	1E-03	9
833 odp	2630	-14.9	167.9	0.20	322	4E-01	5E-02	1E-03	9
827	2800	-15.3	166.4	0.40	344	6E-01	3E-02	3E-03	9
807	2800	3.6	156.6	0.20	20	3E-02	2E-03	8E-05	9
794	2810	40.2	138.2	0.50	30	9E-02	2E-02	3E-04	9
797	2860	38.6	134.5	1.00	50	5E-01	4E-02	1E-03	9
713	2900	-4.2	73.4	0.05	7	1E-02		7E-06	9
690	2910	-65.2	1.2	0.08	12	3E-02	2E-03	2E-05	9

Table 1. Cont...

Site	Water Depth (m)	Latitude (degrees)	Longitude (degrees)	Organic Carbon (%)	Sediment Accumul. Rate (m/ma)	Sulfate Gradient (-mM/m)	Ammonium Gradient (mM/m)	Carbon Burial Rate (gC/m^2/d)	Reference
758	2920	5.4	90.4	0.05	15	8E-03		1E-05	9
782	2960	30.9	141.3	0.20	50	3E-02	2E-03	2E-04	9
709	3040	-3.9	60.5	0.10	11	3E-03	1E-04	2E-05	9
786	3060	31.9	141.2	0.40	9	4E-02	2E-03	7E-05	9
683	3072	-9.0	-80.4	3.00	25	7E-01	2E-01	1E-03	9
828 odp	3090	-15.3	166.3	0.40	60	7E-02	3E-02	5E-04	9
832 odp	3090	-14.8	167.6	0.20	356	1E+00	7E-02	1E-03	9
805	3190	1.2	160.5	0.10	20		3E-03	4E-05	9
reimer-g	3300	36.1	-122.7	3.00	200		6E-01	1E-02	11
795	3300	41.0	139.0	1.00	50	5E-01	4E-02	1E-03	9
846	3300	-3.1	-90.8	0.80	40	1E-01	3E-02	6E-04	9
847	3330	0.2	-95.3	0.50	30	3E-02	2E-02	3E-04	9
844	3410	7.9	-90.5	0.50	10	1E-02		1E-04	9
803	3410	2.4	160.5	0.10	10		1E-03	2E-05	9
slope9102013	3450	41.8	-62.3	0.50			1E-01		1
slope9102014	3530	41.8	-62.4	0.50			1E-01		1
700	3600	-51.5	-30.3	0.05	10	2E-02		1E-05	9
reimer-h	3600	35.9	-123.0	2.00	50		4E-01	2E-03	11
slope9102059	3620	41.8	-50.1	0.50			1E-01		1
845	3700	9.6	-94.6	1.00	10	9E-02	6E-03	2E-04	9
reimer-L	3700	35.5	-122.1	3.00	200		4E-01	1E-02	11
699	3710	-51.5	-30.7	0.10	13	2E-02		2E-05	9
851	3760	2.8	-110.6	0.10	20	4E-02	2E-03	4E-05	9
850	3790	1.3	-110.5	0.30	20	3E-02	5E-03	1E-04	9
682 odp	3790	-11.3	-79.1	3.00	26	6E-01	1E-01	1E-03	9
slope8801028	3820	41.5	-62.2	0.50		1E+00	7E-02		5
710	3820	-4.3	61.0	0.05	9	3E-02	4E-04	8E-06	9
688	3820	-11.5	-78.9	3.00	100	2E+00	3E-01	6E-03	9
849	3840	0.2	-110.5	0.20	30	8E-02	8E-03	1E-04	9
848	3850	-3.0	-110.5	0.10	20	1E-02		4E-05	9
804	3860	1.0	161.6	0.10	10		9E-04	2E-05	9
852	3860	5.3	-110.1	0.10	10	5E-03		2E-05	9
766	4000	-19.9	110.4	0.10	5	2E-02	5E-04	1E-05	9
720	4040	16.1	60.7	0.30	54	4E-01		3E-04	9
745	4080	-59.6	85.9	0.14	30	5E-02	4E-03	8E-05	9
708	4110	-5.5	59.9	0.20	15	3E-02		6E-05	9
slope9102012	4340	41.3	-61.8	0.40			1E-01		1
842	4430	19.3	-159.1	0.10	4	7E-02	8E-04	8E-06	9
711	4430	-2.7	61.2	0.05	5	1E-02		5E-06	9
701	4640	-52.0	-23.2	0.30	25	3E-02		1E-04	9
808	4680	32.4	134.9	0.60	900	5E+00	3E-01	1E-02	9
841	4810	-23.4	-175.3	0.10	11	8E-02	1E-03	2E-05	9
685	5070	-9.1	-80.6	2.50	100	1E+00	2E-01	5E-03	9
esope37	5370	31.5	-24.9	0.50		3E-01	3E-02		3
esope24	5380	31.4	-24.8	0.50		3E-01	2E-02		3
gme10	5400	31.3	-25.5	0.40	100	3E-01	2E-02	8E-04	3
765	5720	-16.0	117.6	0.80	30	1E-01	4E-03	5E-04	9
nares48	5800	22.9	-63.4	0.10	10		2E-03	2E-05	8
nares60	5800	23.5	-63.5	0.30	30		1E-02	2E-04	8
nares56	5800	24.0	-64.5	0.40	40		2E-02	3E-04	8
nares844615	5840	22.8	-63.4	0.40			5E-03		7
nares844618	5840	22.7	-63.4	0.40			6E-03		7
nares844609	5850	22.8	-64.5	0.40			6E-03		7
nares844622	5850	22.9	-63.5	0.40			7E-03		7

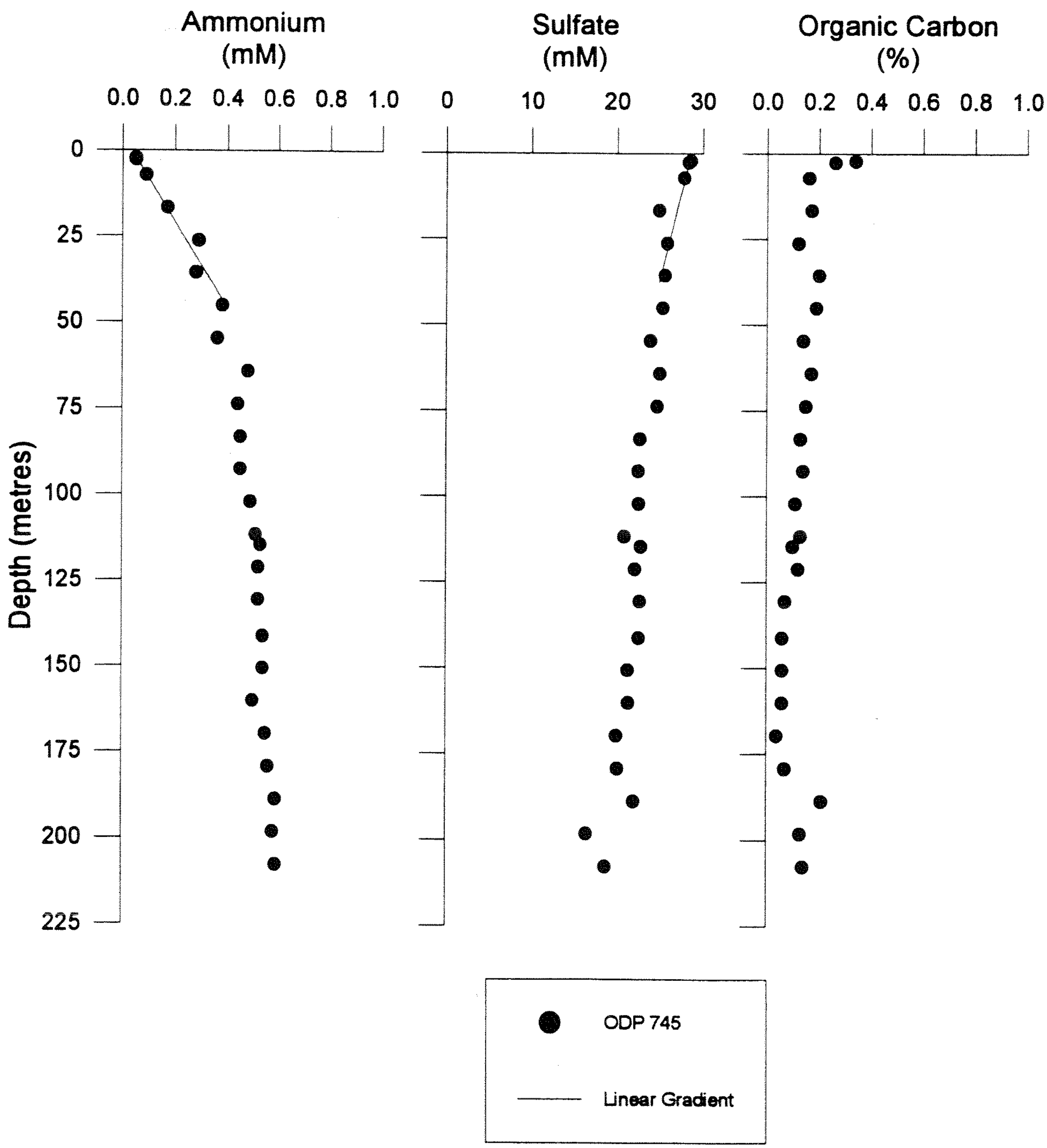

Figure 1. Concentrations of ammonium and sulfate in pore water and of organic carbon in sediment for ODP Site 745 in the South Indian Ocean, a site of low sediment accumulation rates.

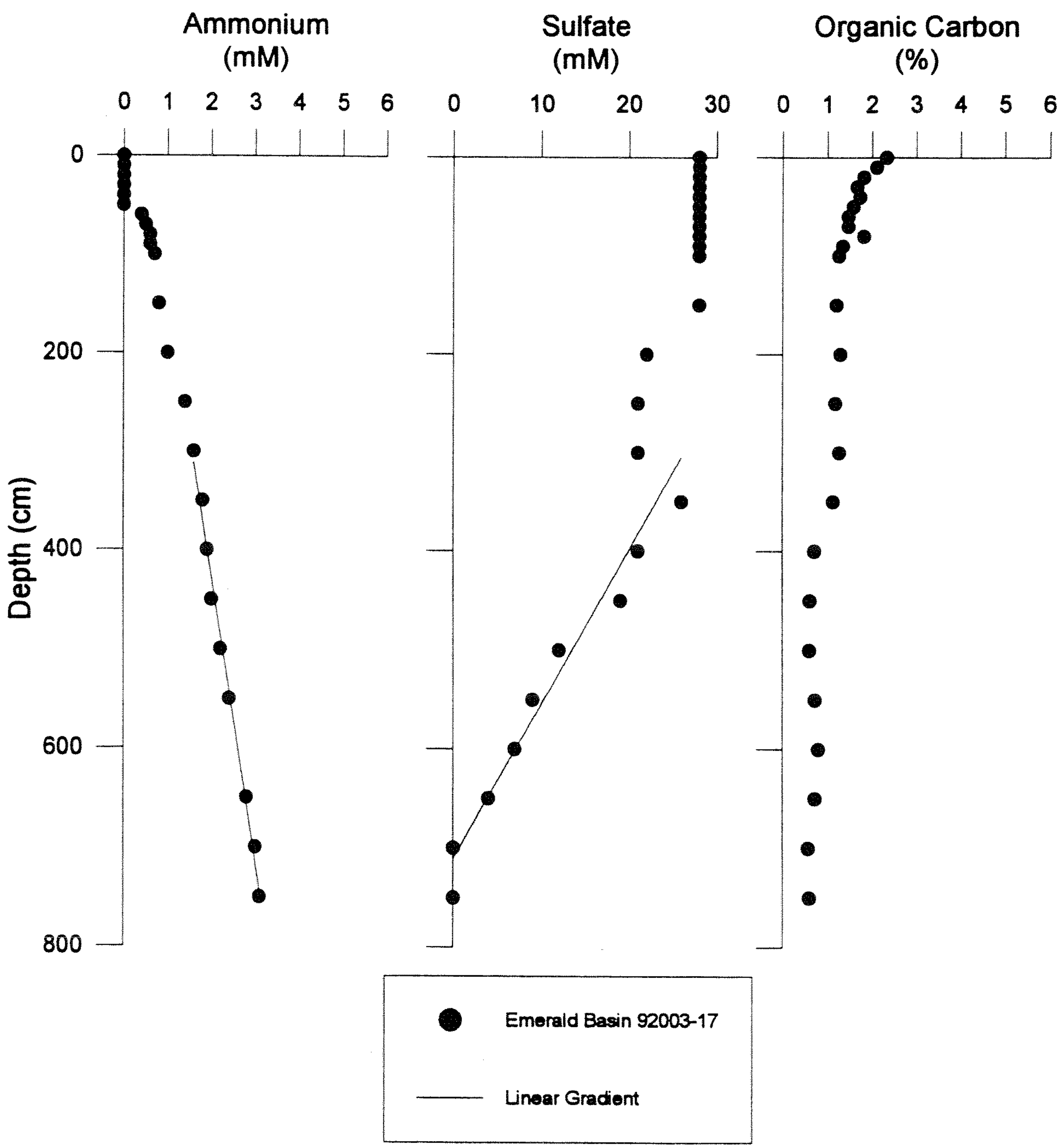

Figure 2. Concentrations of ammonium and sulfate in pore water and of organic carbon in sediment for Emerald Basin Site 92003-17.

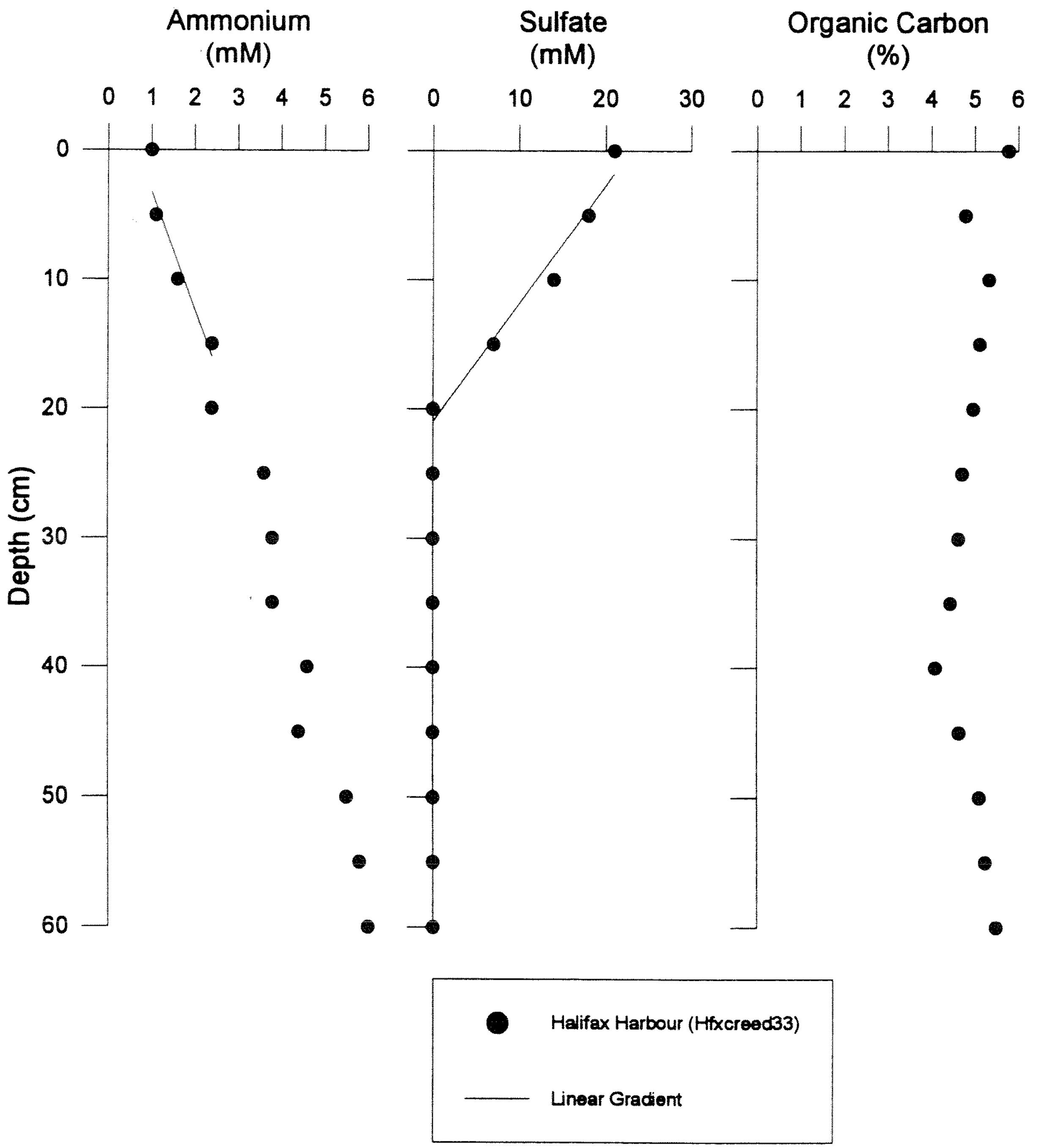

Figure 3. Concentrations of ammonium and sulfate in pore water and of organic carbon in sediment for Halifax Harbour Site Hfxcreed33.

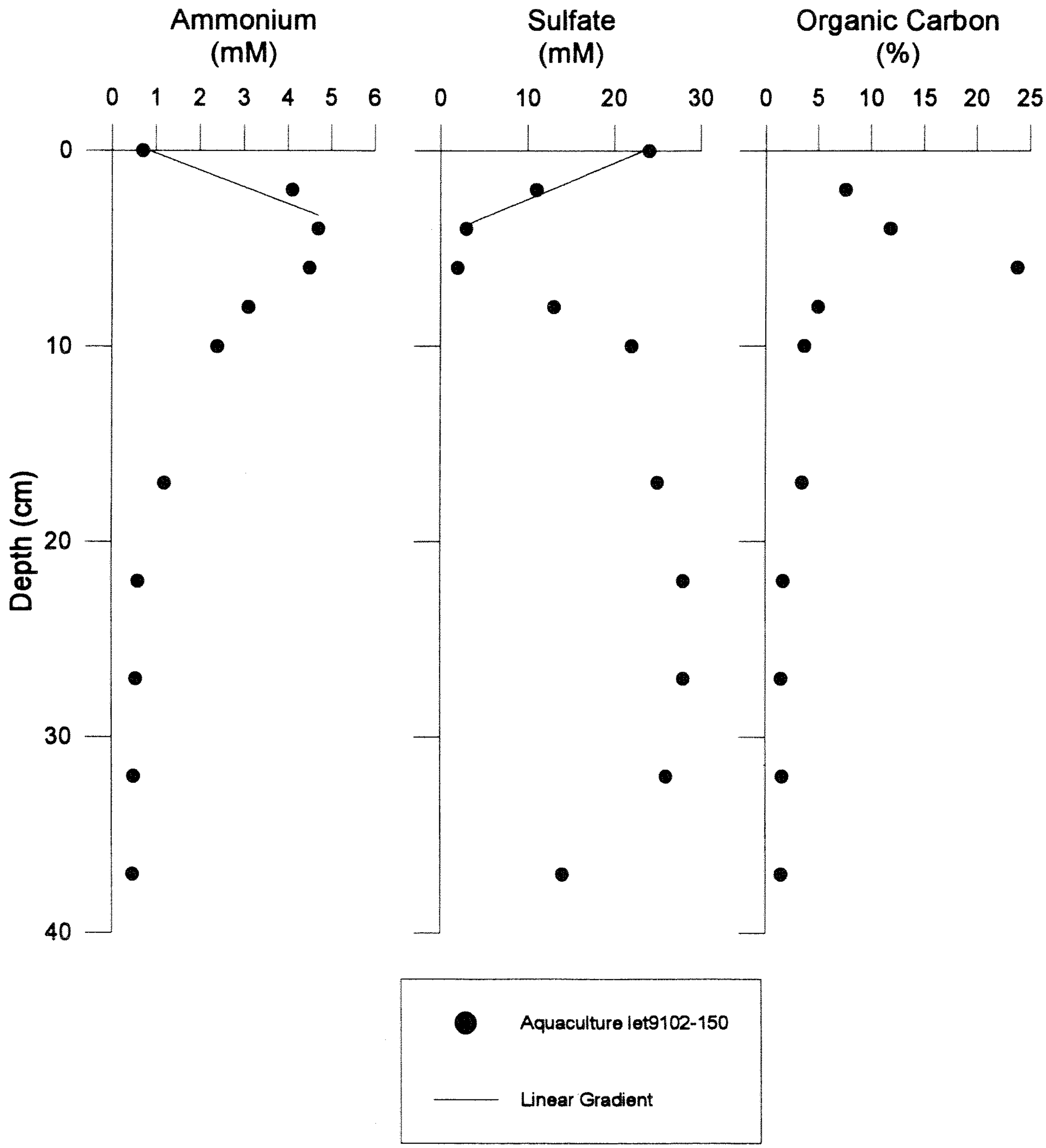

Figure 4. Concentrations of ammonium and sulfate in pore water and of organic carbon in sediment for L'Etang aquaculture Site let9102-150.

at an aquaculture site under a salmon cage in the L'Etang estuary in New Brunswick, Canada (Site let 9102-150). Core length is 40 cm, the sediment accumulation rate is 10000 m ma^{-1}, and the carbon burial rate is 1 g C m^{-2} d^{-1}.

The observation that pore water gradients are interrelated is shown in Figure 5, where ammonium and sulfate gradients for 110 sites (see Table 1) are plotted. Redfield ratios suggest that sulfate reduction and ammonium production should result in a sulfate/ammonium gradient ratio of 6.5, corrected for adsorption and diffusivity (Christensen et al. 1987). A best fit line to observations from 110 sites provides a S/N ratio of 15. At three sites where methane gas was venting from the seafloor (Cranston 1991; Ginsburg et al. 1993), sulfate gradients were enhanced relative to ammonium gradients. Sulfate is reduced during anaerobic methane oxidation, thus strengthening the sulfate gradient (Iversen and Jorgensen 1985).

With this discussion as a background, analyses of the results listed in Table 1 are now presented. The initial summary of the data can be shown using Pearson Correlation coefficients for the data set. Since water depth, concentration gradients, sediment accumulation rates, organic carbon concentrations, and carbon burial rates vary by orders of magnitude, $\log_{10}$ transformations of the data were carried out (Table 2). All correlations were significant at $p<0.001$.

The highest correlation coefficients occur between sulfate and ammonium gradient data ($r=0.95$, Fig. 5) and between these gradients and carbon burial rate ($r=0.95$ and 0.97, respectively, Fig. 6 and 7). Both gradients have lower correlation coefficients with sediment accumulation rate and sediment organic carbon content than with carbon burial rate. The relationship between sediment accumulation rate and organic carbon shows more scatter (Fig. 8), reflecting differences in organic carbon quality and preservation. Figure 9 shows the relationship between water depth and carbon burial rate. Best fit lines, correlation coefficients, and aquaculture cage site data are indicated on the scatter plots.

The sulfate and ammonium gradient relationships with carbon burial rate (CBR) can be summarized with the following equations, based on the least squares lines shown in Figures 6 and 7:

$$CBR = 0.0024 \times (\text{-Sulfate Gradient})^{1.1}$$

$$CBR = 0.023 \times (\text{Ammonium Gradient})^{1.0}$$

CBR units are g C m^{-2} d^{-1} and the gradient units are mM m^{-1}.

These relationships can be used to estimate present-day carbon burial rate when either or both sulfate and ammonium gradients are known. This limits the need to use more expensive methods to determine carbon burial rates, such as sediment rates or sediment accumulation rates (e.g. ^{210}Pb, ^{14}C). The estimated burial rates are reliable to $\leq$ one order of magnitude. If cores are collected within one study area, comparative

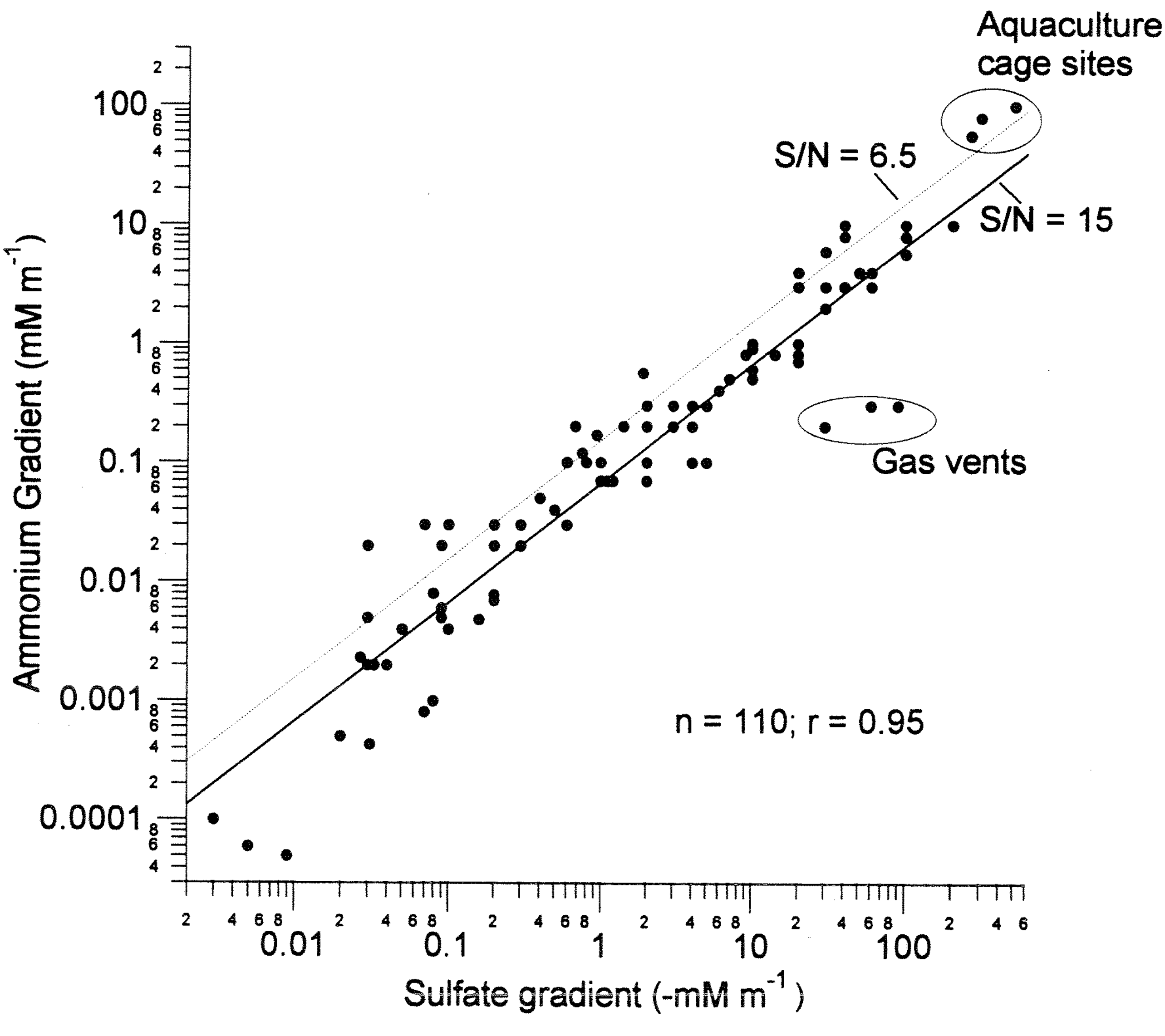

Figure 5. Relationship between ammonium and sulfate gradients. The line labelled "S/N=6.5" is the Redfield ratio for sulfur/nitrogen in organic matter. The best fit line (n=110, r=0.95), labelled "S/N=15," is the observed sulfate/ammonium ratio. The three points labelled "gas vents" appear to have higher sulfate gradients due to sulfate reduction by methane gas.

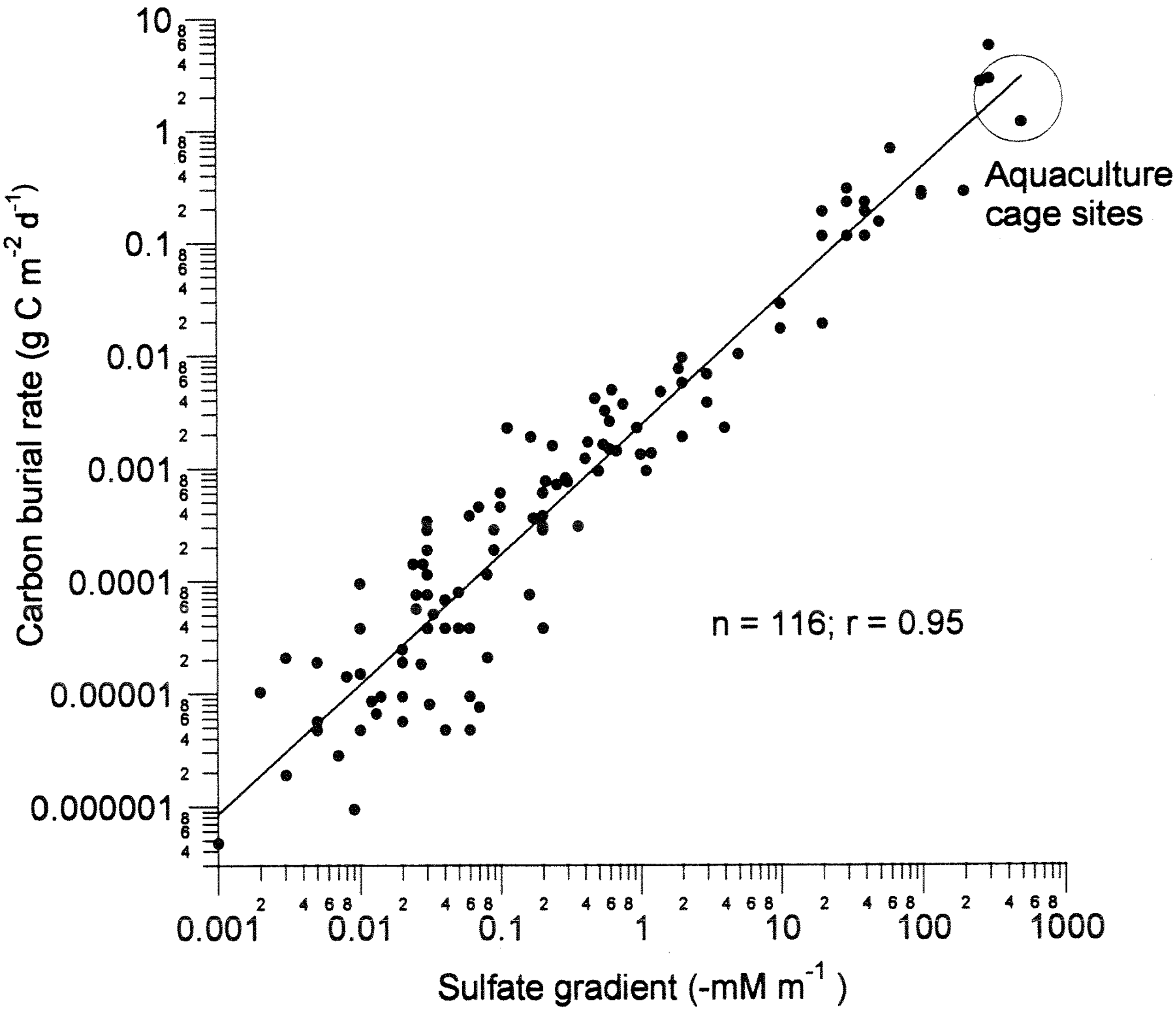

Figure 6. Relationship between sulfate gradients and carbon burial rates (calculated from known sediment accumulation rates and organic carbon concentrations).

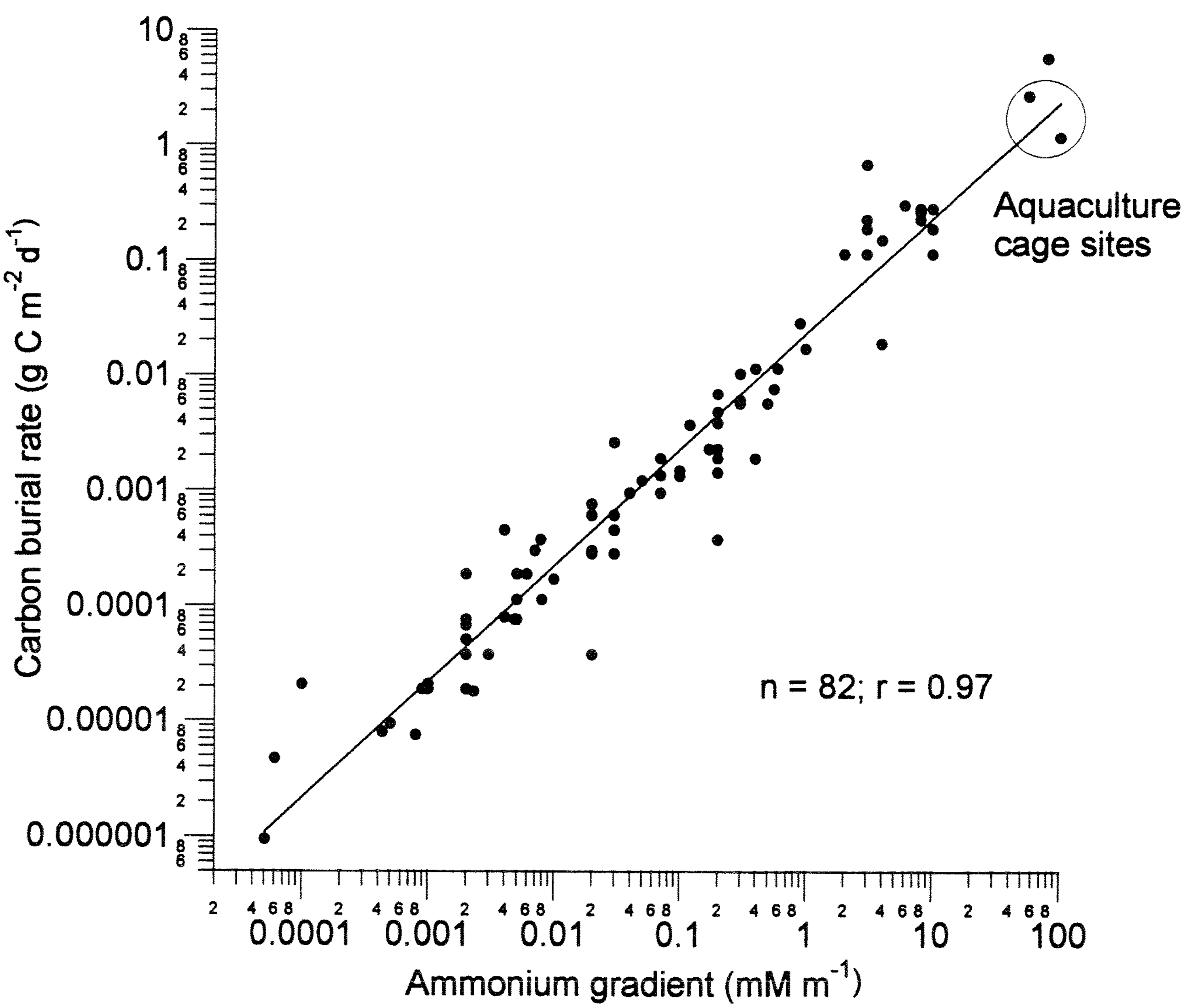

Figure 7. Relationship between ammonium gradients and carbon burial rates (calculated from known sediment accumulation rates and organic carbon concentrations).

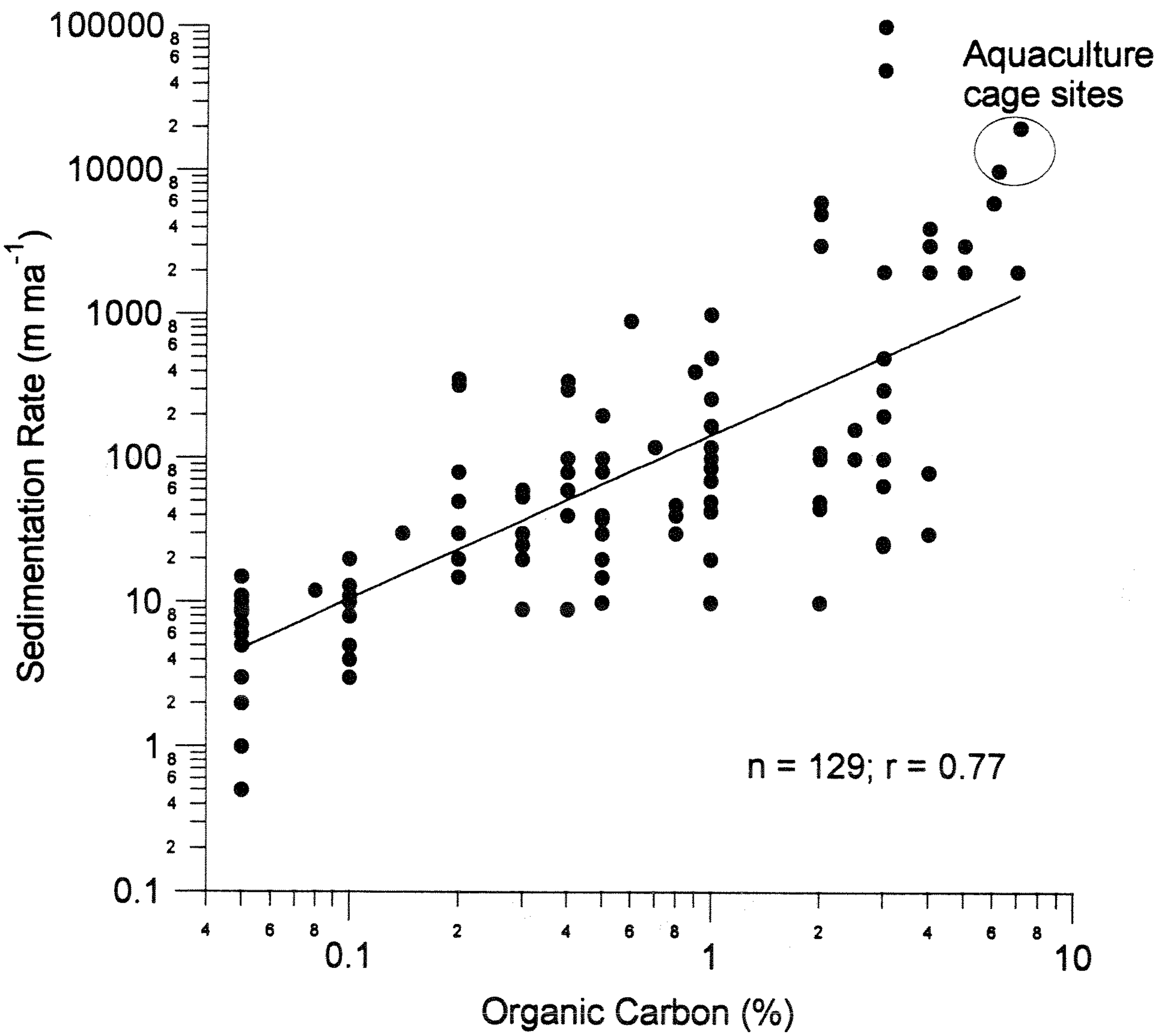

Figure 8. Relationship between organic carbon concentration and sediment accumulation rates.

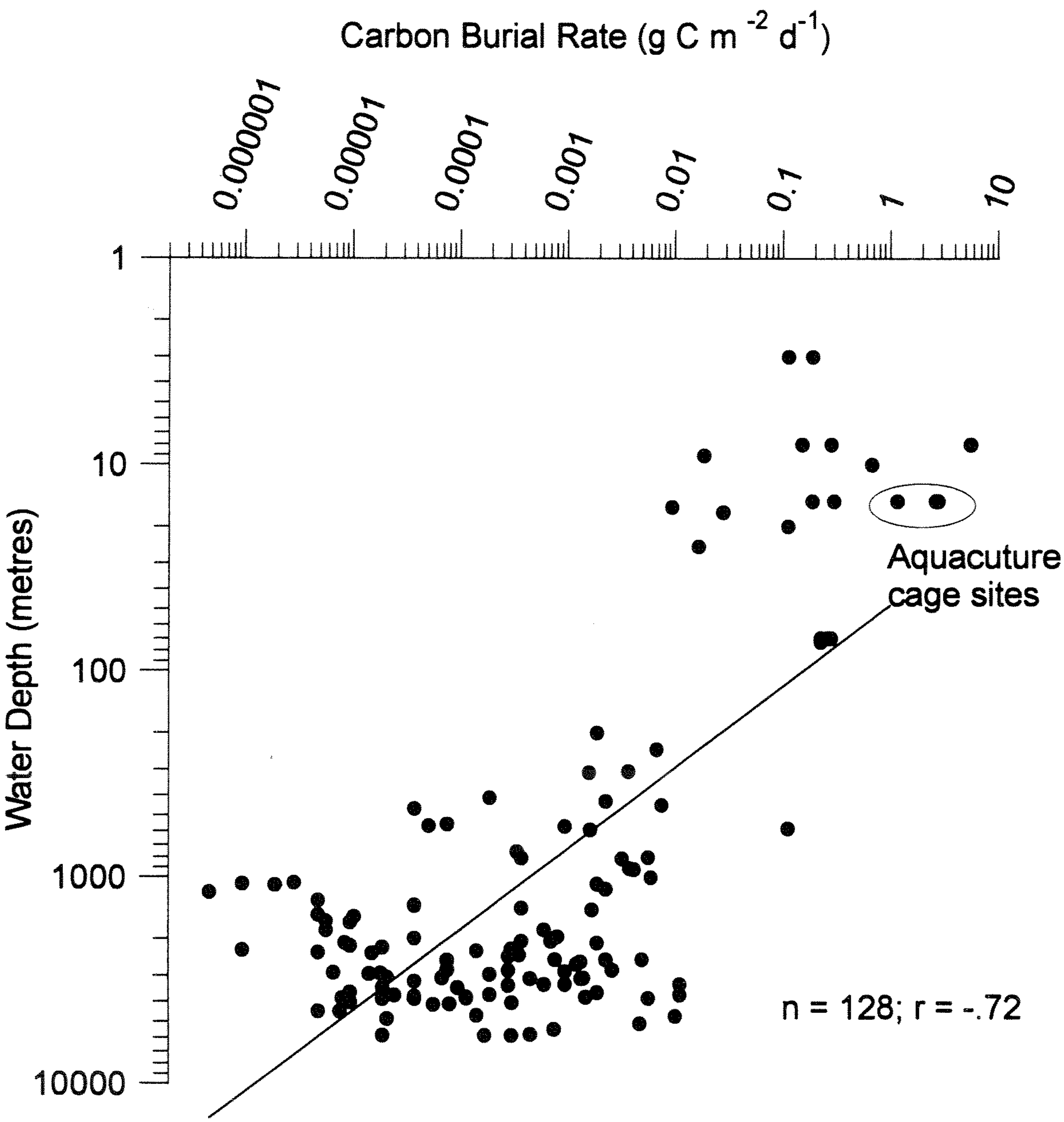

Figure 9. Relationship between carbon burial rate and water depth.

Table 2. Pearson correlation coefficients for data in Table 1. Numbers in parentheses indicate sample size. All coefficients are significant at $p<0.001$.

Variable	Sulfate gradient	Ammonium gradient	Carbon burial	Sediment accumulation rate	Organic carbon rate
Ammonium gradient	0.95 (110)	-			
Carbon burial rate	0.95 (118)	0.97 (82)	-		
Sediment accumulation rate	0.94 (116)	0.91 (82)	0.96 (129)	-	
Organic carbon	0.83 (157)	0.85 (173)	0.91 (129)	0.77 (129)	-
Water depth	-0.69 (157)	-0.72 (173)	-0.71 (129)	-0.74 (129)	-0.56 (220)

estimates of present-day carbon burial rates within the area are reliable to one significant figure. Errors arise when core quality is poor and when sample storage or analytical problems affect the sulfate and ammonium results.

When sediment temperatures vary, microbial activity will change, thus affecting the gradients. Efforts are now under way to determine if extremely low temperatures, which reduce microbial activity, alter the release of ammonium and consumption of sulfate, even though carbon burial is occurring. Seasonal variations in sediment metabolism in nearshore sediments can change by an order of magnitude; however, the lower activity occurs in the winter when the carbon supply rate is also decreased. Holmer and Christensen (1992) concluded that temperature variations can account for approximately 40% of the seasonal variation in benthic metabolism, while organic matter supply accounts for a majority of the observed variation in metabolic rates. For most of the sites included in Table 1, the sediment temperatures, especially in deeper water, are rather similar and tend not to change seasonally. In the L'Etang area, ammonium gradients that were measured at "similar" locations in February, June, and August agree to within a factor of two with each other, even though the bottom water temperature varied from 0 to 10°C.

This method provides a simple, inexpensive method to measure present-day carbon burial rate. Knowing carbon burial rates locally or regionally provides information on the input and sediment capture efficiency of an area. Within an area, variations in carbon burial rates can be used to understand depositional and flushing processes, as well as to identify specific sites that have very high or very low flushing and/or depositional rates. This method is particularly useful for comparing rates within a region, since systematic variations due to temperature and organic matter quality are minimized. Based on CBR and ammonium concentrations, estimates of potential salmon production limits for 20 eastern Canada inlets are presented below.

The primary sedimentation of organic carbon has often been measured using sediment traps placed at cage sites (see Chapters 4 and 5), although the amount of carbon finally buried is lower due to grazing by wild stocks, microbial metabolism, pellet disintegration, resuspension, and bioturbation. Sediment traps at Sea Farm Canada Ltd. placed under a pen at a salmon farm site in Bliss Harbour, L'Etang Inlet, New Brunswick, collected between 8 and 15 g C m^{-2} d^{-1} in June 1990 while those on the perimeter of the pens collected approximately 1 g C m^{-2} d^{-1} (Table 2, Chapter 5). The sulfate and ammonium gradient method was used to estimate a carbon burial flux for that site (let9102-150) of 1 g C m^{-2} d^{-1} in February, 1991, while the actual flux based on sediment accumulation rate and organic carbon content was 3 g C m^{-2} d^{-1} (Table 1). The proposed method is a direct measure of present-day, net carbon burial flux, and in most cases will be more accurate than calculating CBR by applying a correction or efficiency factor to calculate this flux from sediment trap data.

ESTIMATING CARRYING CAPACITIES FOR SALMON AQUACULTURE IN SELECTED INLETS

When carbon burial rates and ammonium budgets are established for an area, salmon loading capacities can be estimated. This approach is applied to 20 inlets in Eastern Canada. Volume, area, and flushing time for inlets are from Gregory et al. 1993.

LOADING LIMIT ESTIMATES FROM SEDIMENT CARBON BURIAL

In the L'Etang area, background carbon burial values of 0.01 to 0.05 g C m^{-2} d^{-1} do not have negative impacts on the ecosystem. Under cages, values of 1 to 5 g C m^{-2} d^{-1} are known to overwhelm the sedimentary environment, producing high concentrations of ammonium, H_2S and CH_4. Anthropogenic loading and natural sedimentary processes such as occur in Halifax Harbour can provide 0.2 to 0.7 g C m^{-2} d^{-1} in terms of carbon burial (Table 1). Such rates appear to be stressful to benthic communities, resulting in widespread anoxia in the sediment column. If 0.05 g C m^{-2} d^{-1} is a background level of carbon burial typical of unimpacted areas, where anoxic conditions do not extend to the sediment surface, a value of 0.05 g C m^{-2} d^{-1} could be used as the allowable carbon enhancement to an overall inlet. This enhancement is used in the following calculation to determine the level of salmon production which would create a two-fold increase in carbon burial.

During salmonid culture operations, for every 100 g C added as food, 20 g C is harvested in salmon and 20 g C is lost as settling faeces and unconsumed food pellets (Hall et al. 1990; Gowen and Bradley 1987). Based on observations in L'Etang Inlet (Table 2, Chapter 5, R. Cranston, unpubl. observ.), on the order of half of the settling carbon reaching the bottom remains in the sediment. Thus, for every 100 g C added to a fish pen, 20 g C is harvested as salmon and 10 g C accumulates in the sediments. Salmon contains 8% carbon by weight (Table 1, Chapter 1), thus 20 g C in salmon represents 250 g salmon, wet weight. For every 250 g salmon produced, 10 g C accumulates as organic carbon in the sediments.

A general equation can be applied to each inlet by multiplying the allowable enhancement flux (0.05 g C m^{-2} d^{-1}) with the inlet area and the salmon production/carbon burial ratio (25 g salmon g^{-1} C). A combined equation is:

$$(0.05\ \mathrm{g}\ C_{waste}\ \mathrm{m}^{-2}\mathrm{d}^{-1})(\text{area in km}^2)(10^6\ \mathrm{m}^2\ \mathrm{km}^{-2})(365\ \mathrm{d\ a}^{-1})(25\ \mathrm{g}_{salmon}\ \mathrm{g}^{-1}\ C_{waste}) \quad (1)$$

Simplifying by cancelling units, using appropriate unit conversion, and rounding to one significant figure gives:

$$\text{inlet area (in km}^2\text{) x } 450 = \text{tonnes salmon a}^{-1} \quad (2)$$

This equation is used to calculate the salmon production limit (based on sediment carbon loading limit) estimate in Table 3.

LOADING LIMIT ESTIMATES FROM WATER COLUMN DISSOLVED AMMONIUM CONCENTRATIONS

A similar loading estimate can be obtained based on water column considerations where ammonium is the ion to be limited. In L'Etang Inlet, background ammonium concentrations in the water column are 1 to 10 μM (Hargrave et al. 1993). Enhanced levels of ammonium can cause phytoplankton blooms which may be directly toxic to fish or through decomposition create anoxic conditions in the water column, thus posing a threat to salmon farms. An estimate for allowable ammonium concentration might be 2 μM (two times a background level of 1 μM). Each flush of an inlet removes 66% of the dissolved material added to the inlet during the flush time (Gregory et al. 1993). If the ammonium enhancement is 1 μM and each flush will remove 0.66 μM, in order to reach a steady state enhanced concentration, 0.66 μM multiplied by the volume of the inlet is the amount of ammonium that can be added during each flushing time.

From salmon production estimates, for every 1000 g of salmon harvested on the order of 80 g N is excreted as soluble ammonium to the water column (Hall et al. 1992; Gowen and Bradbury 1987). In addition to the ammonium released directly by excretion, sediments release about 10% of the annual accumulated carbon and nitrogen as a result of anoxic mineralization, adding an additional 4 g N as ammonium to the water column. Based on these ammonium release estimates per weight of salmon produced (84 g N kg^{-1} salmon, knowing the volume and flushing time of an inlet, and selecting an ammonium enhancement level (e.g. 1 μM), a water column carrying capacity or limit can be estimated. A combined equation is:

$$(\text{vol. } 10^6 \text{ m}^3)(\text{fl. time d}^{-1})(10^3 \text{ L m}^{-3})(0.66 \times 10^{-6} \text{ mol N L}^{-1})(14 \text{ g N mol}^{-1} \text{ N})$$
$$(365 \text{ d a}^{-1})(1\text{kg}_{\text{salmon}}/84 \text{ g N}) \quad (3)$$

A summary equation for these conditions and assumptions is (simplified by cancelling units, by using appropriate unit conversions, and by rounding to one significant figure):

[Volume of inlet (in 10^6 m^3)/flushing time (in days)] x 40 = tonnes of salmon a^{-1} (4)

This equation is used to calculate the salmon production limit (based on water column ammonium buildup) estimated in Table 3.

Region	Area	Volume	Ave.	Flush	Salmon	Salmon	Salmon
		x 10^6	Depth	Time	Production	Production	Production
	(km^2)	(m^3)	(m)	(days)	(sediment)	(water col.)	Ratio
					tonne/a	tonne/a	(sediment
							water col.)
Annapolis Basin	66.5	612	9.2	0.8	29925	30600	1.0
Passamaquoddy	86	1733	20.2	1.9	38700	36971	1.0
L'Etang Region	18.6	206	11.1	1.1	8370	7755	1.1
St. Croix River	40	406	10.2	1.0	18000	15909	1.1
Saint John Harbo	25.6	169	6.6	0.7	11520	10077	1.1
Black's Harbour	0.5	3	6.0	0.6	225	192	1.2
Yarmouth Harbou	3.9	21	5.4	0.7	1755	1268	1.4
Wedgeport Inlet	144	849	5.9	1.1	64800	32343	2.0
Pubnico Harbour	12.5	64	5.1	0.9	5625	2793	2.0
St. Margaret's Ba	138	5191	37.6	12.3	62100	16950	3.7
Shelburne Harbo	21	140	6.7	2.2	9450	2536	3.7
Liverpool Bay	6.7	54	8.1	2.7	3015	798	3.8
Bedford Basin	16.2	510	31.5	10.9	7290	1876	3.9
Canso Harbour	2.1	14	6.7	2.3	945	240	3.9
Mahone Bay	209	4227	20.2	7.1	94050	23870	3.9
Pennant Bay	23.8	400	16.8	6.0	10710	2667	4.0
Indian Harbour	9.8	116	11.8	4.3	4410	1081	4.1
Strait of Canso	30	672	22.4	8.6	13500	3132	4.3
Country Harbour	25.6	286	11.2	4.9	11520	2327	5.0
Sydney Harbour	52	517	9.9	5.8	23400	3545	6.6

Table 3. Area, volume, and flushing times from Gregory et al. (1993) for selected eastern Canadian coastal inlets used to calculate salmon aquaculture loading capacities. Average depth and salmon production limits, based on sediment carbon loading limits and on allowable enhanced dissolved ammonium concentrations in the water column are calculated as described in the text.

The ratios between estimated production for sediment carbon and ammonium load limits are included in Table 3. The two production estimates are plotted in Figure 10. Points falling near the 1:1 line show that the estimated salmon production from the ammonium water column limit are similar to the salmon production estimated from the sediment carbon loading limit. These inlets are influenced by Fundy tides, indicating that the high tides flush these inlets well (i.e flushing times are less than 2 d). When the flushing times are greater than 2 d, the salmon production ratio is greater than 3, indicating that ammonium buildup in the water column may be the limiting factor to load capacity.

The exercise illustrates a simple approach using approximate estimates of organic carbon burial rates, ammonium releases, and carbon/nitrogen budget estimates. The calculated maximum salmon production that would not exceed acceptable loading limits based on increments in dissolved ammonium and organic carbon burial may also be compared to actual salmon production in a specific area. For example, during 1992 total numbers of fish per site listed in permits granted by the New Brunswick Department of Fisheries and Aquaculture amounted to 2.2×10^6 for all regions of L'Etang Inlet (B. Chang, pers. comm.). If 50% of the fish are 2 y old and these are harvested each year at an average weight of 4.5 kg $fish^{-1}$, annual salmon production in L'Etang Inlet would be approximately 5000 t y^{-1}. This is approximately 60% of the salmon production limit calculated by the two methods for L'Etang Inlet (8000 t y^{-1}) (Table 3). There are many other considerations to be made, but Table 3 is a useful way to compare potential production in different inlets and to show that loading limits may be constrained by either carbon accumulation in the sediment, or by ammonium accumulation in the water column.

SUMMARY

Dissolved ammonium and sulfate profiles in sediment pore water have been measured in more than 100 sites and compared to known carbon burial rates. A relationship between these variables has been determined for carbon burial rates ranging from 10^{-6} to 10 g C m^{-2} d^{-1}. The low rates correspond to deep ocean regions with sediment accumulation rates on the order of mm ka^{-1} where sediment organic carbon concentrations are <0.1%. The high rates occur in impacted areas found under salmon aquaculture cages where sediment accumulation rates are on the order of cm a^{-1} and sediment organic carbon concentrations are >5%. By measuring the dissolved ammonium and sulfate gradients, a rapid, inexpensive estimate of carbon burial rate can be interpolated from the gradient/carbon burial relationship.

Since the gradients are in equilibrium with present-day redox reactions, the proposed method provides quantitative estimates of present-day carbon burial rates, rather than an estimate of carbon burial rate in the past. It minimizes errors due to bioturbation and physical mixing of the sediment. After a mixing event, the sediment record is permanently disturbed, while the pore water components immediately begin to

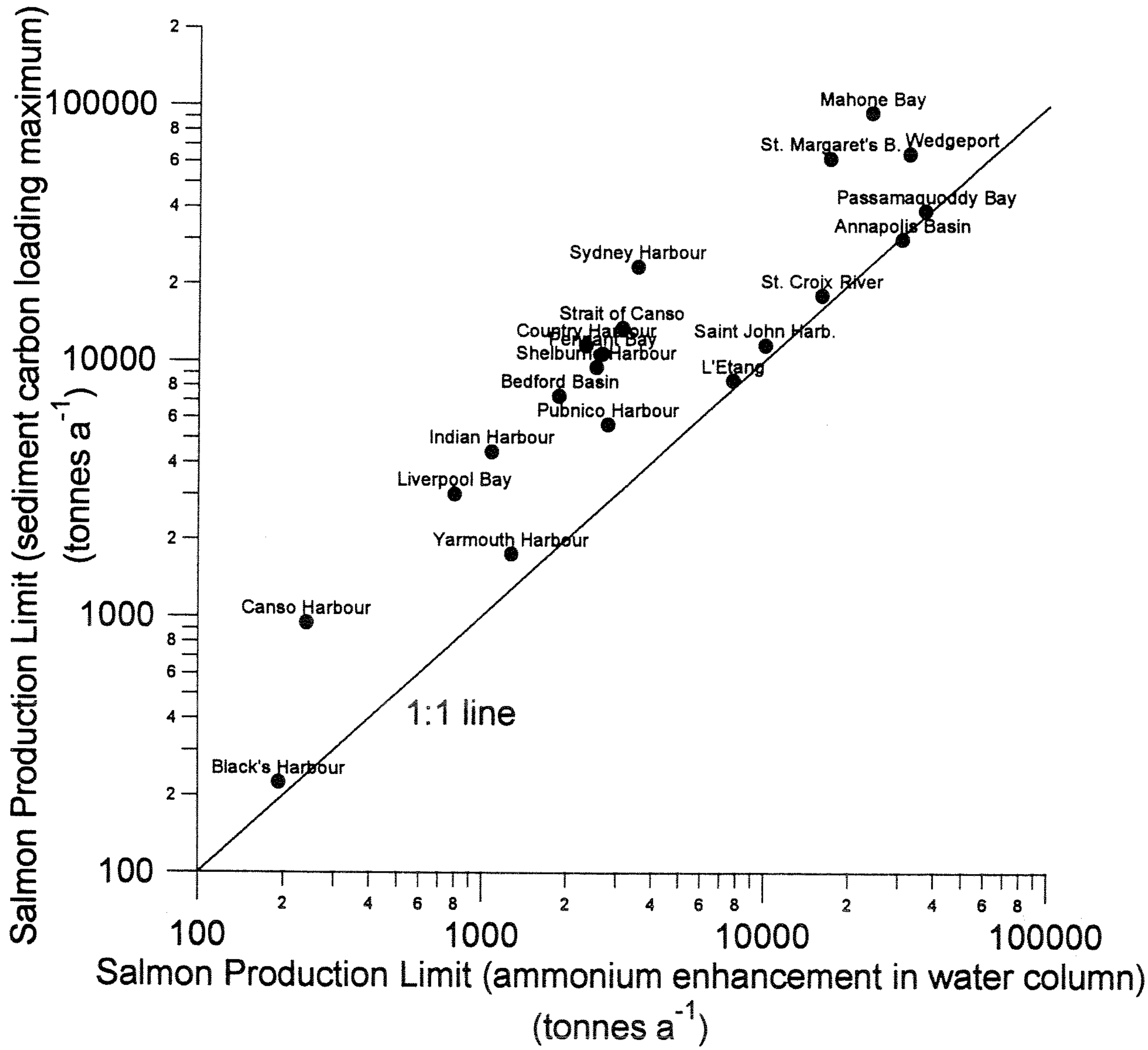

Figure 10. Salmon production limits, based on sediment carbon loading maxima, compared to salmon production limits, based on allowable enhanced dissolved ammonium concentrations in the water column, for selected Eastern Canadian inlets.

be redistributed as they diffuse through the mixed sediment. This compensates for core sampling problems, such as mixing/missing the top of sediment cores. The method can be applied to compare present carbon burial rates at proposed aquaculture sites and to monitor carbon loading between and within regions. Carbon depositional flux estimates from sediment traps measure the carbon "on the way to the bottom," much of which is remineralized, grazed and physically removed from the site. The method described here provides a direct measure of net carbon burial following all processes of removal.

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SUMMARY AND CONCLUSIONS

Chapters in this report have reviewed current models that describe and predict impacts of benthic organic matter enrichment due to fin-fish net-pen aquaculture. New models are also proposed that could be used to make standardized calculations for site assessment of existing or new sites.

Chapter 1 presents a generalized method for calculating benthic carbon loading (BCL) from two models - one which calculates the amount of particulate organic carbon released from a net-pen site and a second which describes the distribution of settled material on the bottom under and near the site. Variables needed for the calculations are numbers of fish, parameters for a carbon budget through each fish, water depth, current speed, and sedimentation rates of fish faeces and unconsumed food pellets. There is considerable uncertainty in calculating some variables (for example sedimentation rates of particles of various types and sizes). A major conclusion is that substantial carbon loading occurs directly under or proximal to cages because of the rapid deposition of ungrazed food pellets. Measurements of sedimentation under and immediately adjacent to net-pens in coastal areas of Maine and New Brunswick (presented in Chapters 4 and 5), however, show that unconsumed food pellets did not constitute a large fraction of material settled in sediment traps. Soluble and fine particulate matter are transported by water movement and therefore if impacts occur they could be at locations distant from the aquaculture site. A hydrodynamic model is described that can be used to study the transport of dissolved and fine particulate matter on an inlet-wide basis. Finally, comparisons of calculated oxygen demand by fish at usual stocking densities and sediments under net-pens show that oxygen consumed through fish respiration greatly exceeds the benthic demand. However, if the impacted area extends 20 m beyond the cages, the total oxygen demand by fish and sediments are of a similar magnitude. This explains why oxygen depletion may occur in shallow areas with reduced water circulation.

Chapter 2 reviews various models that have been developed to predict the dispersion and loading of particulate wastes from fish farms. All of these models are conceptually similar and based on a calculation of horizontal displacement of settling particles as a function of water depth, current speed, and direction. Of the numerous assumptions, it is concluded that changes in current speed with depth, variable bottom topography, and a range of particle settling velocities are the most amenable to numerical solution for improvement of existing models.

Chapter 3 presents data from 23 net-pen systems in Maine where benthic impacts assessed by a semi-quantitative benthic score (BS from 0 to 4) are compared to estimates of BCL (described in Chapter 1) calculated for the same sites. Values of BS are based on visual observations ranging from no perceptible difference from natural conditions (BS=0) to unacceptable impacts (BS=4) such as the presence of azoic sediments, gas release, formation of sulfur bacterial mats, horizontal extent of faeces buildup and dominance of infauna. Increasing scores from 1 to 3 reflect increments of benthic enrichment based on the type of impact and horizontal extent. There is a weak non-linear positive correlation between BCL and BS. A threshold in degree of negative benthic impacts occurs for benthic loading rates between 1 and 10 g C m^{-2} d^{-1} with low (<1) values of BS when BCL is <1 g C m^{-2} d^{-1} and higher (>1.5) values when organic carbon input exceeds 10 g C m^{-2} d^{-1}. This threshold is similar to the level of organic carbon sedimentation (1 g C m^{-2} d^{-1}) above which negative redox potentials and organic carbon accumulation have been observed in sediments at other aquaculture sites presented in Chapter 5. Calculations presented in Chapter 4 also show that the estimated rates of maximum aerobic oxidation of sediment organic carbon at current speeds between 0.3 and 1 cm s^{-1} range from 6 to 11 g C m^{-2} d^{-1}. Rates of particulate organic carbon supply in excess of these levels will result in accumulation of organic carbon. Anaerobic respiration of sedimented organic becomes dominant with the formation of anoxic conditions associated with deteriorated benthic conditions. The lack of linear response in BS values to increased levels of BCL between 1 and 10 g C m^{-2} d^{-1} could be explained by the transition from aerobic to anaerobic conditions.

A second model is developed in Chapter 3 to derive a benthic index (BI) based on the accumulation of sediment organic carbon that leads to deterioration of benthic conditions and the transition from aerobic to anaerobic conditions. It is assumed that no organic carbon accumulates in the sediment if rates of organic carbon deposition and removal are equal. An uptake-clearance model is used to describe benthic deterioration (BD) based on the balance between organic carbon accumulation and a recovery rate that decreases as organic carbon accumulates. Values of BD, determined directly by organic carbon accumulation and indirectly by benthic carbon loading and scaled from 0 to 4 to correspond to the scale of values of BI, are close to the empirical values of BS. High BS values relative to those predicted from BI occur at sites that have been occupied longer than other sites studied. The most discrepant sites are located in a cove that receives effluent from a fish processing plant and have a common management history. It is concluded that values of BS and BI can serve as robust, non-site-specific indices that be used by industry and regulatory agencies for aquaculture site monitoring and assessment of husbandry performance.

Chapter 4 summarizes observations of organic carbon and nitrogen content and sinking rates of food pellets and fish faeces, and the percentage of waste food settled in sediment traps deployed under and adjacent to five net-pen facilities in Maine. The empirical measures of sedimentation were combined with estimates of food delivery to pens to construct a mass balance model of assimilated and unassimilated organic carbon settled to sediments as food and faecal pellets. Sedimentation rates measured at the edges of net-pens (1 to 4.5 g C m^{-2} d^{-1}) are similar to those observed in similar studies at other salmon aquaculture sites summarized in Chapter 5, but fluxes predicted from the model were higher by three to four times. In addition, mass balance model calculations indicated that rates of organic deposition due to waste feed and faeces should be about equal, yet neither food nor faecal pellets were observed in large numbers in sediment trap samples. If faeces disintegrate into fine particles with low sinking rates, these could be transported horizontally to result in sedimentation over a much larger area than would occur if larger particles settled more rapidly immediately under and adjacent to net-pens. Model sensitivity calculations showed that the depth at which particle dispersion occurs within a net-pen affects sedimentation rates.

Despite the inconsistency between calculated and observed measures of organic carbon sedimentation, data presented in Chapter 4 indicate that rates of deposition in excess of 1 g C m^{-2} d^{-1} often occur under and adjacent to net-pen sites. Oxygen supply to sediments must be maintained at these high rates of organic carbon loading to prevent the formation of anoxic conditions. Measurements of benthic CO_2 production at three net-pen sites in Maine were linearly related to organic carbon deposition measured 2 to 4 wk previously. There was also a linear relation between benthic oxygen and CO_2 flux for these sites indicating that particulate waste food and faeces were rapidly decomposed at a rate proportional to the organic carbon supply. For the period of study there would have been no accumulation of organic matter in sediments under net-pens at these locations since the organic carbon supply was balanced by respiration. Calculations of diffusive oxygen flux and current speed are used to estimate the theoretical rate at oxygen must be supplied to prevent oxygen depletion at the sediment-water interface. Estimates show that as current speeds approach zero, the theoretical maximum delivery rate of organic carbon that will not deplete molecular oxygen in surface sediments is about 4 g C m^{-2} d^{-1}. This is approximately the level of benthic carbon loading shown to mark a threshold for the onset of deteriorated benthic conditions based on the benthic index derived in Chapter 3 (BI values >1) and it is within the range of sedimentation rates above which anoxic conditions lead to negative redox potentials and increases in organic carbon accumulation described in Chapter 5 .

An important observation in Chapter 4 is that sediment oxygen uptake rates increase with increased current speed and that changes occur rapidly (within seconds to minutes). Thus, instantaneous rather than average current velocities are critical for determining the level of oxygen supply to sediments and hence the potential for organic matter accumulation. As observed in other studies, macrofauna are usually the first benthic organisms to disappear following the onset of anoxic conditions since they cannot survive extended periods without oxygen supply. After the disappearance of macrofauna, benthic communities dominated by infauna are progressively replaced by those where white sulphur bacterial mats cover surface sediments indicative of permanent anoxic conditions at the sediment surface. The duration of minimum current velocities which allow anoxic conditions to develop at the sediment-water interface are therefore critical in determining the types of organisms that dominate a benthic community. It is concluded that aquaculture sites with relatively high current velocities (>3 cm s^{-1}) can support an abundant and diverse community of macrofauna and infauna even under conditions of high organic carbon loading since dissolved oxygen delivery to sediments can be sustained. At locations where current velocities are reduced such that low (< 1 cm s^{-1}) velocities occur for prolonged periods (1 to 2 h) during each tidal cycle, maximum rates of aerobic organic carbon oxidation that can be maintained are lower and there is the potential for accumulation and burial of undegraded organic matter.

Chapter 5 reviews previous measurements of particulate organic carbon sedimentation observed by deployment of sediment traps at finfish and molluscan aquaculture sites. Rates vary by over three orders of magnitude (0.1 to >100 g C m^{-2} d^{-1}). While there are many problems in attempting to compare results from different studies, benthic impacts arising from such a large range of organic carbon input values are also likely to be highly variable. Surface sediment oxidation reduction potentials (Eh) and organic carbon (SOC) from various aquaculture sites and other locations are correlated with rates of organic carbon sedimentation measured at the same sites. A benthic enrichment index (BEI), the product of Eh and SOC, is inversely correlated with the logarithm of organic carbon sediment at >1 g C m^{-2} d^{-1}, but there was no significant relationship at lower sedimentation rates.

As described in Chapters 3 and 4 and discussed above, sedimentation of organic carbon at rates >1 g C m^{-2} d^{-1} can lead to the formation of anoxic conditions if low current velocities prevent the supply of dissolved oxygen to maintain aerobic respiration. Values of sedimentation lower than this threshold allow aerobic conditions to be maintained. These conditions are reflected in positive Eh potentials and low rates of organic carbon accumulation with values of BEI>0. These sites would be considered to have a low degree of impact and correspond to BI values <1 (Chapter 3). Negative Eh potentials, indicative of anoxic and reducing conditions, occur as a result of high rates of sedimentation at sites where current velocity, dissolved oxygen supply, and aerobic respiration are all reduced. As organic carbon loading increases and anoxic conditions become more persistent, Eh potentials fall lower and organic carbon accumulation increases leading to larger negative values of BEI. This corresponds to the progression of BI values from 1 to 4 as the negative impacts of anoxic conditions increase as described in Chapter 3. It is concluded that values of both the BEI and BI can be used to characterize the present state of any benthic system with respect to changes expected as a result of increased organic matter loading. Both indices may be used to regulate or monitor benthic conditions at aquaculture sites.

Chapter 6 summarizes observations of dissolved ammonium and sulfate gradients in sediment pore water measured at over 100 sites where independent estimates of organic carbon burial (accumulation) exist. Increases in ammonium and decreases in sulfate dissolved in pore water with depth in the sediment are assumed to arise from redox reactions that are in equilibrium with present-day carbon burial rates. Combined data from all locations shows that ammonium and sulfate gradients are positively and linearly related to each other and to carbon burial rates. Measured ammonium and sulfate gradients and calculated carbon burial rates (approximately 1 g C m^{-2} d^{-1}) in three cores from an aquaculture site in L'Etang Inlet, New Brunswick, are about three times higher than values derived for cores from Halifax Harbour, an enriched Nova Scotia embayment, and 50 times higher than burial rates in a continental shelf basin off Nova Scotia. It is concluded that measurements of dissolved ammonium and sulfate gradients in sediment pore water provides a rapid, inexpensive method to directly estimate present-day net organic carbon burial rates. The summary of data from various coastal, continental shelf, and deep ocean sites demonstrates the broad range of values that occur and provides a scale for comparison of data that could be collected to characterize aquaculture sites on the basis of net organic carbon burial rates.

Finally, calculations are presented in Chapters 1 and 6 to show how estimated release rates of dissolved ammonium and particulate organic carbon by net-pen cultured salmon may be used to determine the maximum capacity of an inlet for aquaculture production. The calculations are referenced to assumed "baseline" concentrations of dissolved ammonium in the water column (1 μM) and carbon burial rates (0.05 g C m^{-2} d^{-1}) typical of values observed in the absence of salmon aquaculture. It is assumed that enhancement of baseline values of dissolved ammonium and organic carbon burial by these increments due to salmon production would be tolerable. Annual salmon production limits are then calculated to determine the biomass of fish that would yield this increment.

Salmon production limits calculated on the basis of tidal flushing and increments in water column ammonium and carbon burial rates for 20 inlets in Nova Scotia in Chapters 1 and 6 show almost perfect agreement. Holding capacities vary from 240 t to 30,000 t $inlet^{-1}$ with highest values in the largest embayments. The two calculations are in close agreement for well flushed (<2 d) inlets but for those with longer flushing times, ammonium buildup appears to be more important than increased carbon burial for determining maximum holding capacity. The calculations illustrate that numbers of fish cultured may be constrained by either accumulation of dissolved nutrients in the water column or increased organic carbon burial. The relative importance of these potential impacts differs depending on hydrographic features that determine flushing times for a particular inlet.