

Chemical fluxes and mass balances in a marine fish cage farm. I. Carbon

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ABSTRACT: Carbon fluxes in a marine trout cage farm in the Gullmar Fjord, western Sweden, were measured to investigate how much of the carbon supplied to the farm was recovered in harvest, how much was lost to the environment, and the properties and fate of this environmental loss. The measured fluxes included fish food, juveniles, harvest, fish loss (death and escape), sedimentation from the cages, and benthic release measured with diver-operated flux chambers and a gas collection unit in situ. Carbon mass balances for the farm, based on the measured fluxes (flux method), were constructed for each of 2 consecutive growing seasons. Another mass balance (accumulation method) was based on the total carbon input with food and juveniles to the farm since it was started, the removal of carbon with harvested fish and fish loss, and the recovery of carbon in the sediment originating from the farm after 7 growing seasons. Some 21 to 22 % of the total carbon input to the farm was recovered in harvest, fish loss constituted 1 to 3 %, and 75 to 78 % (or 878 to 952 kg C per tonne of fish produced) was lost to the aquatic environment. On a seasonal basis and of the carbon input to the farm, solute release from the cages (probably CO₂ produced during fish respiration and excreted urea) removed 4 to 49 %, sedimentation of faeces and excess food removed 29 to 71 %, flux from the farm sediment of dissolved and gaseous carbon (total carbonate [C_t], methane and dissolved organic carbon) transferred 2 to 6 % back to the overlying water, and 23 to 69 % was accumulated in the sediment. (Ranges of values represent inter-seasonal variability.) The long-term (7 seasons) sediment accumulation of carbon amounted to 18 % of the total carbon input to the farm. Of the carbon deposited on the sediment surface 3 to 20 % was released back to the overlying water seasonally. C_t dominated the annual benthic fluxes. Loss to the environment of dissolved carbon (the sum of solute release from the cages and benthic flux) amounted to between 6 and 55 % of the carbon input to the farm (or 9 to 71 % of the total environmental loss) on a seasonal basis, and 58 % (or 76 % of the total environmental loss) on a long-term basis. This study constitutes the first step in an assessment of the eutrophication caused by the fish farm.

INTRODUCTION

Cultivation of fish in floating net cages in open fresh and coastal waters has been a rapidly growing industry in many parts of the world during the last 2 decades. Today, the cage farming technique is practised in numerous countries including Scotland, France, German Democratic Republic, Poland, all Scandinavian countries including the Faroe Islands, Canada, USA, Australia, New Zealand and Japan, and the development is expected to continue (Rosenthal 1986). Salmonid cultivation predominates in all these countries except Japan. Norway is the largest producer in the world of cultivated salmon, farmed in sea cages (Acke-

fors 1986a). Cage farming in coastal waters is the main form of fish aquaculture in Japan (Rosenthal 1986).

There is a growing concern about the impact of cage farming industry on the environment. The environmental impact includes dispersion of organic matter, nutrients and therapeutic agents such as antibiotics to surrounding waters and underlying sediments, effect on bottom fauna and plankton communities, and possible influence on wild fish stocks such as genetic interference and transmission of diseases. Knowledge on these topics is essential for decisions on the number and size of fish farms which can be established in a body of water. Such knowledge is also essential for the fish farmer as it relates to the health of the fish and hence to the profita-

bility of the farm. Studies of fluxes of particulate and dissolved substances in coastal cage farms, and of the exchange of matter between the farm and the environment, are also important in a wider biogeochemical context for an improved understanding of processes in marine organic-rich, hypertrophic systems.

General overviews of the environmental impact of fish cage farming have been presented (Phillips et al. 1985a, Ackefors 1986b, Weston 1986, Gowen & Bradbury 1987, Persson 1987, Waldichuk 1987, Håkanson et al. 1988). Stigebrandt (1986) modelled the environmental load from cage farms. The influence of physical site-specific characteristics, such as water exchange and bottom dynamic conditions, on the eutrophication effect in the recipient has been discussed by Håkanson (1986). Studies have shown enrichments of organic matter and nutrients in surrounding waters and sediments, oxygen depletion within cages, and qualitative and quantitative changes of the pelagic and benthic biota around and below farms (Beveridge & Muir 1982, Korzeniewski & Korzeniewska 1982, Penczak et al. 1982, Enell & Lóf 1983, Enell 1985, Ervik et al. 1985, Phillips et al. 1985b, Rosenthal 1985, Trojanowski et al. 1985, Persson 1986, Brown et al. 1987, Aure et al. 1988). Most of these studies were carried out in fresh or (in one case) brackish waters, and very few of them attempted to quantify the efflux of solutes from particles deposited on the bottom and the accumulation of material in the sediment. Continued successful growth of marine cage farming is strongly dependent on further direct measurements of the degree of environmental impact caused by this activity. Results from such measurements could also be used to tune existing models (Stigebrandt 1986, Aure & Stigebrandt 1989).

The main purpose of this paper is to estimate how much of the carbon supplied with fish food and juveniles to a marine trout cage farm in the Gullmar Fjord, western Sweden, is recovered in the harvested fish, how much is lost to the environment, how much of the environmental loss appears in particulate and in dissolved form, how much is returned in dissolved or gaseous form from the sediment to the overlying water, and how much is accumulated in the sediment. To this end we measured carbon fluxes in the farm and constructed mass balances based on the measured fluxes for each of 2 consecutive growing seasons. We also utilized another approach for construction of mass balances based on the total carbon input to the farm since it was started, the removal of carbon with harvested fish, and the recovery of carbon in the sediment below the cages after 7 growing seasons. These results form the basis of an assessment of eutrophication caused by the fish farm. In companion papers we will present fluxes and mass balances for nitrogen and phosphorus in the farm.

MATERIALS AND METHODS

Experimental site. The fish cage farm is situated in the Gullmar Fjord, western Sweden. Rainbow trout *Oncorhynchus mykiss* (Kendall 1988) have been cultivated in the farm since 1980, when the farm was established. The farm contains at most 15 net cages and then occupies an area of 25 × 40 m over a bottom depth of 18 to 21 m. Each cage is 7 × 7 m wide and 4 m deep. The cages are placed about 1.5 m apart and separated by rafts.

The farm is sheltered by a small island, which influences the current regime around and below the farm. Water circulation at the site is, however, sufficient to keep bottom water well-oxygenated throughout the year. The temperature of surface waters in this area normally ranges from 0 to 19 °C annually. The surface water has a salinity of typically 20 to 25 and the bottom water 30 to 35.

The sediment under the cages is situated below the halocline during the whole year. The top sediment layer is completely black, very loose and highly reducing. Underlying this layer is a light brown, more compacted sediment. Sulphate is almost depleted in the porewater of the top sediment layer, and sulfide concentrations are very high (10 to 13 mM). Nutrients in the porewater of this layer are also drastically enriched. During part of the year the sediment surface below the farm is almost entirely covered by a white *Beggiatoa* sp. bacterial mat. Gas bubbles are released from the sediment during late summer and until late fall.

The fish are equally distributed among the cages with normally 450 to 500 kg fish per cage at the beginning of the growing season, and 1700 to 2300 kg per cage at the end. The growing season is usually from April/May to December/January. Some years part of the fish production is kept in the farm during winter. Dry food (data in Table 1) is used and supplied to the fish by a combination of manual and automatic feeding.

Carbon in fish food and fish. Each of the various foods used in the farm was homogenized into fine particles and the water content determined from the weight loss at 105 °C after drying for at least 24 h. The dried food samples were analysed for carbon on a Carlo Erba CHN elemental analyser model 1106.

The carbon content of whole rainbow trout as a function of fresh weight was calculated according to Persson (1986), who gave the following relation

$$(\%C) = 0.55 \times 17.4 \times W^{0.099}$$

where W = fresh weight in g.

Sedimentation. The flux of particulate matter under the farm was measured using cylindrical sediment traps with a diameter of 80 mm. They had an aspect ratio of 6.2, which is within the range recommended in the literature to avoid selective trapping (Blomqvist &

Håkanson 1981). The traps were kept vertical by a gimbal and were oriented towards the current by a fin. Three traps, placed 6 m above the bottom (i.e. about 10 m below the bottom of the cages), were used on each occasion. No poisons or preservatives were used in the traps. Sampling periods were 1 to 2 d. Immediately after collection, the particles in each trap were suspended in 2 l of bottom water from the site. Three aliquots from each of the homogenized suspensions were filtered through precombusted Whatman GF/F glass microfibre filters. Particulate carbon determinations on the filters were carried out on a Carlo Erba CHN elemental analyser model 1106. The carbon concentration in blank filters without particulate material was negligible.

Benthic in situ fluxes. The flux chambers (Fig. 1) used in this study were similar to those previously described by Hall (1984), Rutgers van der Loeff et al. (1984) and Hall et al. (1989). They were constructed from 70 cm long sections of 50 cm diameter Plexiglas tube. When placed 10 cm into the sediment, each contained 118 l of water and covered 0.20 m² of sediment surface. The chambers were each equipped with a 7.5 cm long, 0.8 cm diameter, Teflon-coated magnetic stirring bar revolving at 75 rpm, Teflon valves for sampling, and a flexible membrane consisting of several layers of poly-

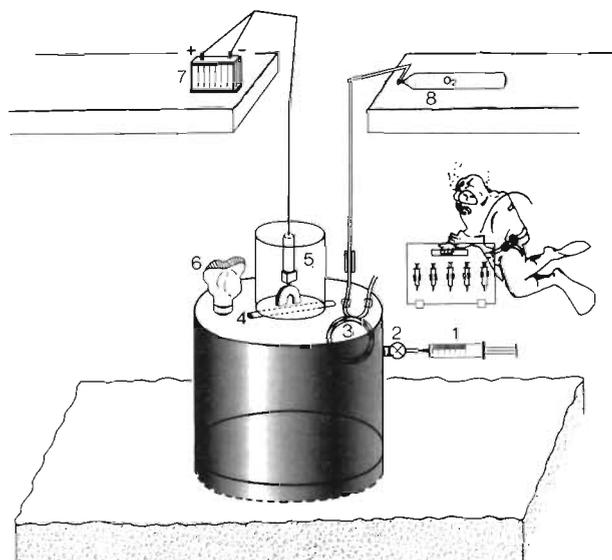


Fig. 1. The benthic flux chamber system used in this study. 1, Sample syringe; 2, Teflon valve; 3, Teflon tubing for oxygen diffusion (in the regulated chambers); 4, magnetic bar; 5, electric motor; 6, plastic bags serving as volume compensator; 7, 12 V battery; 8, oxygen cylinder. From Hall (1984)

ethylene bag that served to compensate the volume removed during sampling. Some of the chambers in each experiment were also equipped with a coil of FTP-Teflon tubing through which oxygen gas, supplied under pressure from the surface (up to 1.8 MPa), could

diffuse into the chamber. The chambers were thoroughly cleaned with detergent, then soaked in diluted hydrochloric acid and finally conditioned by suspending them in seawater at the experimental site for about 24 h. After this procedure they were carefully placed in position over the farm sediment within ca 2 m of each other and daily sampled by SCUBA divers.

Chamber experiments were carried out for 16 to 24 d in December 1984, May 1985, September 1985 and July 1986. Bottom water temperatures during the experiments ranged from 4 to 16 °C. From 2 to 4 chambers were used below the cages during each experiment. The height of the water column in the individual chambers was measured during each experiment, and the corresponding chamber volume used when calculating the benthic fluxes. In the *regulated chambers* dissolved oxygen was maintained close to the original level throughout the experiments by adjusting the pressure in the oxygen supply tubing. In the *unregulated chambers*, where no such adjustments were made, the respiration of benthic microorganisms was allowed to bring oxygen levels to zero.

Water samples from the chambers were collected in glass syringes (dissolved methane) and polypropylene syringes (pH, total alkalinity [A_t], calcium and dissolved organic carbon [DOC]). The pH (according to Almgren et al. 1975) was determined in the laboratory immediately after sampling. Methane samples were stored in gas-tight glass bottles with rubber stoppers, A_t and calcium samples in plastic bottles, and DOC samples were stored filtered and frozen in plastic bottles. Methane was determined on a gas chromatograph equipped with a flame ionization detector using a head-space technique. The column used was packed with Porapak Q (80/100 mesh) and N₂ was the carrier gas. A_t was determined by computerized titration using the method of Anderson & Wedborg (1983), calcium according to Anderson & Graneli (1982), and DOC using an Astro TOC analyser model 1850 with potassium hydrogen-phthalate as a standard. These determinations were carried out at the end of each experiment series or during the experiment. Total carbonate (C_t, also called sum-CO₂) was calculated from A_t and pH measurements. The C_t concentration that is determined by the titration was not used since CO₂ could have been lost during sample storage.

Gas fluxes. A frame was very gently placed over the sediment under the fish cages. In the frame, 20 cm above the sediment surface, a funnel was mounted up side-down, and above the funnel either a graduated cylinder or a glass bottle. These were also turned up side-down and filled with seawater. The gas content of the graduated cylinder was read off daily, and samples for gas analysis were taken frequently in the glass bottles after they had been closed with rubber stoppers at the bottom.

Installation of the 'gas frame', reading off the graduated cylinder and sampling were carried out by SCUBA divers. Determination of methane in the gas samples was performed with gas chromatography as described above, but here 100 μl of the sample was injected directly into the chromatograph. The gas flux at the bottom was converted to atmospheric pressure, temperature corrected to 0 °C, and the gas-carried methane flux calculated from the methane content of the gas and a mol volume of 22.4 l mol⁻¹ at this temperature.

Sediment. Sediment cores were collected by divers using 12 cm diameter Plexiglas core tubes. Five cores were taken in September 1984 along a diagonal transect, which went through the farm and continued 20 m outside it. Cores were also taken under the farm in May 1985 and July 1986. Immediately after collection, the cores were sectioned in vertical position under nitrogen in a glove box. Subsamples were taken for determinations of water content, which was determined from the weight loss at 105 °C after drying for at least 24 h. The density of wet sediment was calculated as a function of water and organic matter content (% of wet sediment) according to Håkanson (1981, and references therein). Concentrations of carbon in dried sediment samples were measured with (to remove calcium carbonates) and without acid treatment on a Carlo Erba CHN elemental analyser model 1106. The accuracy of the carbon determinations was checked against a standard reference sediment.

Estimation of the areal distribution of farm sediment. To estimate the amount of carbon accumulated on the bottom below and around the farm since it was started, the volume of sediment originating from the farm had to be estimated. As a start in accomplishing this, SCUBA divers used the black and loose appearance of the farm sediment and the existence of *Beggiatoa* sp. as indicators, when estimating the areal distribution of sediment originating from the farm. The divers swam above the bottom around the farm, sending up buoys to the surface from the edge of the farm sediment. The distance from each buoy to the farm rafts was measured and the total area calculated. The divers also measured the thickness of the farm sediment layer along the edges of the farm. The sedimentary distributions of carbon, water content and phosphorus were used as indicators of the thickness, but also of the areal distribution, in addition to diver observations, of the sediment layer originating from the farm (see below).

Assumptions made when constructing mass balances. The area used when calculating the total seasonal sedimentation from the farm corresponded to the actual number of cages utilized plus an additional 1 m around each cage. The sediment traps were positioned about 10 m below the bottom of the cages, and the sedimenting particles (faeces and excess food) were assumed to be

equally dispersed over this area at this depth. Five cages were utilized from the start of the 1985 season (beginning of May) until mid-July, when the growing fish were redistributed into 8 cages. This number of cages was utilized until the end of season (end of January). The corresponding numbers during the 1986 season were 4 (beginning of May to mid-June) and 7 (until mid-January). The sedimentation rate measured during one period was assumed to be valid through half of the time remaining until the next measurement. The sedimentation rate measured in November 1985 was assumed to be valid until the end of that season. Based on the annual variation of sedimentation rates during 1985, it was assumed that the high rate measured in September/October 1986 was not representative for the last 2.5 mo of that season. The sedimentation rate during these months was set to 10 % of the September/October 1986 value.

The sediment area used to calculate the total seasonal benthic flux was assumed to be equal to the area used to calculate the total seasonal sedimentation. The bottom area that had been influenced by sedimentation from the farm was 3.8 times larger than the farm itself (see below). However, we only considered a bottom area influenced by particle deposition during one season in the estimation of the total benthic flux during that season. As for the sedimentation measurements, the benthic fluxes measured during one period were assumed to be valid through half of the time remaining until the next measurement. Fluxes in December 1985 were assumed to be equal to the fluxes measured in December 1984 and to be valid until the end of the 1985 season. The bubble-transported methane flux was not measured in December 1984, and the DOC flux was not measured in December 1984 or in July 1986. The former was estimated from the September 1985 measurements assuming a constant ratio between the bubble and the dissolved methane fluxes. The latter was estimated from the May and September 1985 measurements assuming that the average ratio between the C_i and DOC fluxes obtained during these months was valid also during December 1984 and July 1986. Benthic fluxes during the 1986 season were measured in July and estimated for the other months assuming that the annual variation of each flux observed during 1985 was valid also during 1986. The methane bubble flux during late summer and fall of 1986 was estimated as for December 1984.

RESULTS

The measured carbon concentrations in the various fish foods used in the farm during the growing seasons of 1985, 1986 and 1980–86 are given in Table 1 together with food input data. The carbon content of all foods was within the range 50 to 52 % (dry w/w).

Table 1 Input data, measured carbon concentrations and water contents of the various fish foods used in the farm during 1985, 1986 and 1980 to 1986

Brand name	Input 1985	Input 1986 kg (wet wt) ^a	Input 1980–86	Carbon concentration % (dry w/w)	Water content % (wet w/w)
EWOS T51	34000	8900	178200	50.2–50.8 ^b	6.7–7.9 ^b
EWOS ET92		5700	5700	51.5	3.2 ^c
EWOS T45			4200	50.9 ^d	7.4 ^d
Danish Ørred (Ecolife 16)		3750	3750	49.6	8.4
Aller Mølle AT67			500	51.4	8.2
46/19			7000	51.8	10.0
Total input	34000	18350	199350		

^a Data given by the fish farmer
^b Samples from 2 batches analysed. One batch was analysed from each of the other foods
^c This value appears low
^d Samples of this food were not available. The mean of the carbon concentrations and water contents, respectively, of the other 5 foods analysed was used

The calculated carbon contents of whole rainbow trout are given in Table 2 together with juvenile input, harvest, fish production, fish size and fish mortality data. A fish mortality of 10% of the number of fish was estimated by the fish farmer (from comparisons of juvenile input and harvest, as well as carcass collection) for each year except 1986, when a mortality of 22% was estimated. The dead fish were removed from the farm and disposed on-shore. In addition to mortality loss, 1900 kg of fish escaped in August 1984. A carbon content of 18.2% (of fresh wt) in the escaped fish was used (the average carbon content of dead fish during 1980 to 1986; Table 2). The contributions of fish food, juveniles, harvest and fish loss (mainly mortality) to the turnover of carbon in the cages are given below. The average effective food conversion coefficients (the ratio between food input and fish production) in the farm were 2.18 to 2.33, 2.09 to 2.51 and 2.17 to 2.41 for 1985, 1986 and 1980 to 1986, respectively. The higher values are the gross coefficients including effects of fish loss and food waste. When the estimated weight of fish loss is considered part of the fish production, the lower values are obtained. The relatively large difference between gross and net values during 1986 reflects the higher mortality during that season.

The measured sedimentation rates of particulate carbon and benthic fluxes of C_t , DOC, and methane in dissolved and gaseous form are given in Table 3. The benthic fluxes given are initial fluxes, but in all chambers fluxes were constant during the first several days, or throughout the whole chamber incubation, in both regulated and unregulated chambers (examples are shown in Figs. 2 and 3). Oxygen was depleted in the unregulated chambers within 1 to 2 d during all experi-

Table 2. *Oncorhynchus mykiss*. Juvenile input, harvest, fish mortality, fish production and carbon concentrations of rainbow trout in the farm during 1985, 1986 and 1980–86

	Weight (kg) ^a (fresh wt)	Size (g) (fresh wt)	Carbon concentration ^b % (fresh w/w)
Juveniles			
1985	3700	160–380 ^a	15.8–17.2
1986	3200	160–450 ^a	15.8–17.5
1980–86	26340	100–1000 ^a	15.1–19.0
Harvest			
1985	18300	1160 ^c	19.2
1986	10500	1290 ^c	19.4
1980–86	109140	1320 ^c	19.5
Fish mortality			
1985	1020	580 ^d	18.0
1986	1480	650 ^d	18.2
1980–86	6960	660 ^d	18.2
Fish production (harvest minus juvenile input)			
1985	14600		
1986	7300		
1980–86	82800		

^a Data given by the fish farmer
^b Calculated for whole rainbow trout according to Persson (1986)
^c Average size calculated from the weight of harvest, and the number of fish remaining in the farm at the end of season after any fish loss (death and escape)
^d Estimated as half of the average size of harvested fish

ments, whereafter dissolved sulfide was rapidly released from the sediment. There was no significant or systematic difference between fluxes measured in the

regulated and unregulated chambers (Table 3), suggesting that dissolved carbon fluxes from this sediment were not strongly influenced by oxygen. Calcium fluxes were always (except in one chamber where it was taken up by the sediment) directed from the sediment to the overlying water and mostly low (normally constituting only 0.4 to 2.4 % of the C_t flux), suggesting that dissolution or precipitation of calcium carbonates did not significantly influence C_t fluxes from this sediment. The methane content of the gas samples collected at the bottom ranged from 97 to 99 % v/v.

Examples of the carbon distribution in the sediment directly under the cages are shown in Fig. 4. The contribution of calcium carbonates to the total carbon content of the sediment was 3 to 8 %, with the lower contribution generally in the surface layer. This is consistent with the small contribution of calcium carbonates to the measured benthic C_t fluxes. Concentrations of organic carbon were 21 to 30 % w/w in the top 3 cm of the sediment and gradually decreasing with depth. The sediment below the farm was enriched with organic carbon 10 to 15 times compared with the sediment 20 m outside the farm. There was some spatial variation of the

carbon distribution directly below the farm, but this variation was small compared with the drastic decrease in carbon concentration that was observed just at the edge of the farm (on one side of the farm). At 5 to 10 m outside this farm edge, carbon concentrations were much lower. The water content followed generally the distribution pattern observed for carbon. It was 95 to 98 % w/w in the top 2 cm of the farm sediment, gradually decreasing with depth and drastically decreasing with distance from the farm edge.

Phosphorus exhibited an almost constant distribution with depth in the top black and loose layer of the sediment (Hall & Holby unpubl.), below which it decreased rapidly and approached the concentrations observed outside the farm (at corresponding depths in the sediment). Phosphorus was for this reason considered the best indicator of the thickness of the sediment layer originating from the farm. A more detailed description of phosphorus in the sediment below and around the farm will be given elsewhere. Similar observations of sedimentary phosphorus distributions have been made in brackish water cage farms of the Baltic Sea (Persson 1986).

Table 3. Measured sedimentation rates, and benthic solute and gaseous fluxes of carbon under the farm during 1984, 1985 and 1986

Period	Sedimentation (mmol C m ⁻² d ⁻¹)	Benthic fluxes (mmol C m ⁻² d ⁻¹)			
		C_t	CH ₄ (diss.)	CH ₄ (in gas) ^a	DOC ^b
Dec 1984				0.778 ^c	4.128 ^c
Cham. A (unreg.)		49.63	0.009		
Cham. B (reg.)		31.65	0.003		
May 1985	2718 ^d			0	
Cham. A (unreg.)		134.5	1.404		
Cham. 1 (unreg.)		162.4	1.400		2.040
Cham. B (reg.)		123.8	1.804		2.424
Cham. 2 (reg.)		135.3	1.560		
Jun 1985	4649 ^d				
Sep 1985	4431 ^d			368.5–978.0	
Cham. 1 (unreg.)		480.4	3.967		112.4
Cham. B (reg.)		476.5	4.631		13.61
Cham. 2 (reg.)		553.4	6.266		
Oct 1985	3459 ^d				
Nov 1985	340.6 ^d				
May 1986	4088 ^e				
Jun 1986	6484 ^e				
Jul 1986	4948 ^e			0	12.93 ^c
Cham. A (unreg.)		90.41	0.139		
Cham. 2 (reg.)		164.4	0.136		
Aug/Sep 1986	5903 ^e				
Sep/Oct 1986	6138 ^e				
Nov–Jan 1986/87	613.8 ^c				

^a Measured with the 'gas frame' on the bottom. Values represent atmospheric pressure
^b Fluxes given as phthalate-carbon equivalents
^c Value estimated for this period (see text)
^d Value represents the mean of 9 subsamples (3 aliquots from each of 3 sediment traps)
^e Value represents the mean of 3 subsamples (1 aliquot from each of 3 sediment traps)

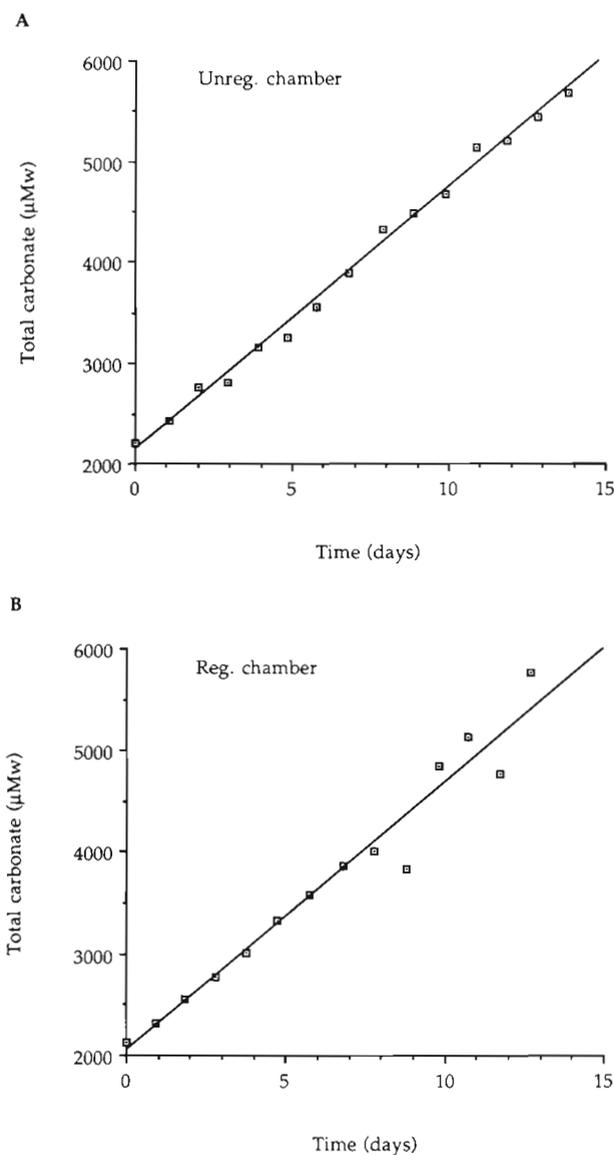


Fig. 2. Evolution of total carbonate ($\mu\text{mol per kg seawater}$) in (A) an unregulated and (B) a regulated chamber during the May 1985 flux chamber experiment

The areal distribution of fish farm sediment was calculated from the divers' measurements to be 3800 m^2 , which is 3.8 times the area of the farm itself (1000 m^2) and consistent with the horizontal distribution of phosphorus in the sediment. The farm sediment was more widely spread horizontally along one side of the farm (30 to 40 m from the edge of the outermost cages) than the other (10 to 15 m). At one corner of the farm, farm sediment was only observed within a distance of 5 m from the cages. The actual horizontal distribution pattern of farm sediment around the farm was most likely influenced by the number and exact position of cages used each year, the bottom topography and the current regime. The thickness of the sediment layer

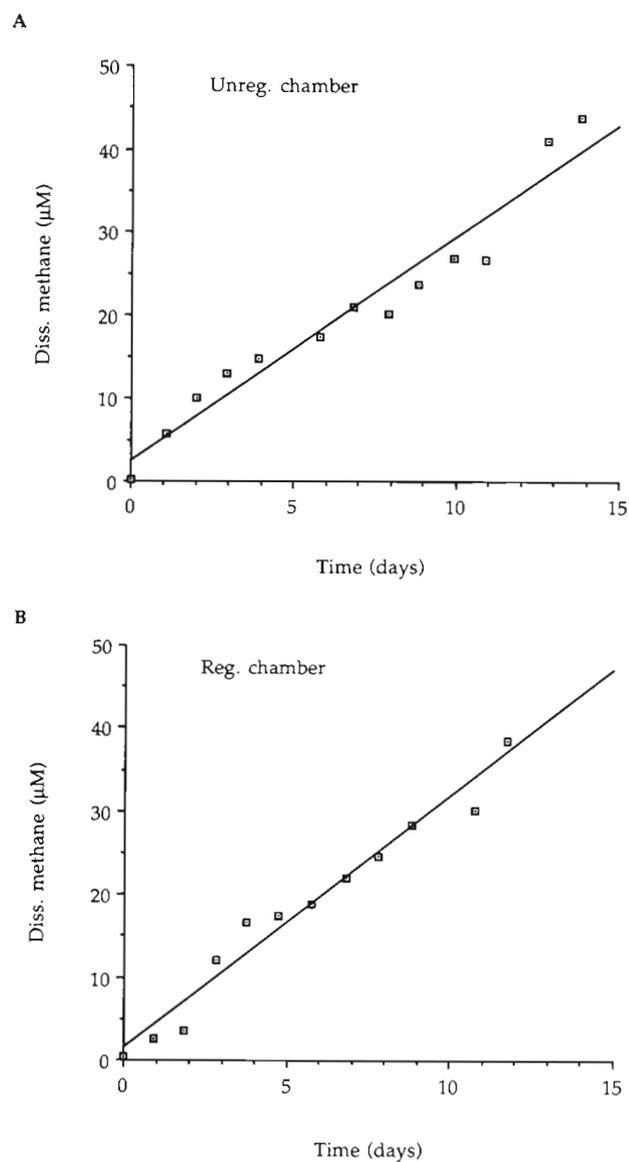


Fig. 3. Evolution of dissolved methane in (A) an unregulated and (B) a regulated chamber during the May 1985 flux chamber experiment

originating from the farm was 18 cm directly below the cages (as judged from vertical phosphorus distributions and extrapolated up to the end of the 1986 season) and became gradually thinner with distance from the farm edge (as measured by the divers and judged from phosphorus distributions). The total volume of wet farm sediment thus obtained was $458\,540 \text{ dm}^3$. A density of 1.078 g cm^{-3} of wet sediment was calculated. Using this value, an average water content in the sediment layer derived from the farm of 82.7 %, and an average dry weight based carbon content in the same sediment layer of 21.2 %, we obtained a net carbon accumulation in the sediment originating from the farm (since it was started) of $18\,130 \text{ kg C}$.

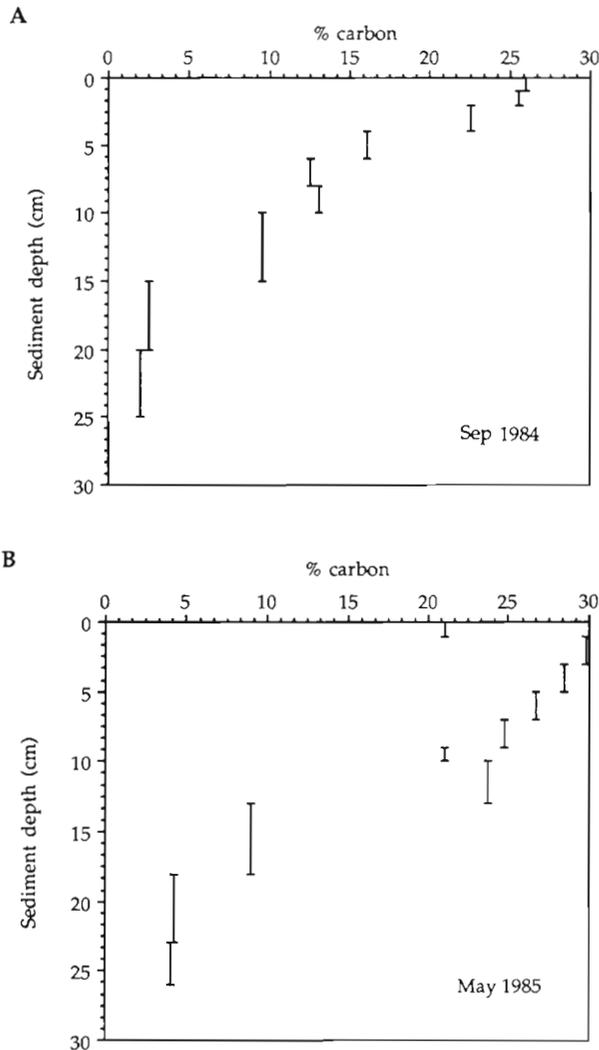


Fig. 4. Concentrations of organic carbon (% of dry weight) as a function of depth in the sediment below the fish farm in (A) September 1984 and (B) May 1985

DISCUSSION

Mass balances

Mass balances for fish farms need to be constructed in order to be able to estimate the recovery in harvested fish of the total input of an element to the farm, the total and the dissolved loss of the element to the environment, as well as the accumulation of the element in the sediment. Construction of mass balances is therefore crucial as a first step in an assessment of eutrophication effects caused by fish farming. The partitioning of the environmental loss into particulate and dissolved fractions is of great importance, as it relates to how the loss will be dispersed in the recipient waters, and, for nutrients, whether it will have a direct stimulatory effect on primary production.

We used 2 approaches when constructing carbon mass balances for the cage farm. The 'flux method' was based on calculations of the seasonal carbon fluxes carried by fish food, juveniles, harvest and fish loss, and on measurements of sedimentation and benthic fluxes of carbon. The 'accumulation method' was based on determinations of the total carbon input to the farm (since it was started) with food and juveniles, the carbon removal with harvest and fish loss, and the net accumulation of carbon in the sediment. Mass balances according to the flux method were constructed for the growing seasons 1985 and 1986 (Figs. 5 and 6). A long-term (7 yr) carbon balance was calculated according to the accumulation method for the consecutive growing seasons 1980 to 1986 (Fig. 7). The definitions of parameters used in the mass balances are given in Table 4.

Possible sources of error in the mass balances

Processes other than those taken into account in the mass balances may possibly add or remove carbon from the farm or the underlying sediment (examples shown in Figs. 5, 6 and 7). Such processes include the transport (with currents) of zooplankton through the cages, acting as external food for the fish. Taking the high supply rate of fish food to the cages and the large size of food pellets relative to zooplankton, we consider the addition of zooplankton a small contribution to the carbon budget of the farm. Solutes may be released to the water column from particles sedimenting out of the

Table 4. Definitions of parameters used in the carbon mass balances

Parameter	Definition
Total environmental loss	Input with fish food and juveniles minus removal with harvest and fish loss ^a
Dissolved environmental loss (flux method)	Benthic flux plus solute release from the cages
Solute release from the cages ^b	Total environmental loss minus sedimentation
Dissolved environmental loss (accumulation method)	Total environmental loss minus sediment accumulation
Sediment accumulation (flux method)	Sedimentation minus benthic flux

^a The fish loss was not considered part of the environmental (aquatic) loss, because it consisted mostly of dead fish and these were disposed on-shore

^b Probably mostly carbon dioxide produced during fish respiration and excreted urea (Gowen et al. 1985)

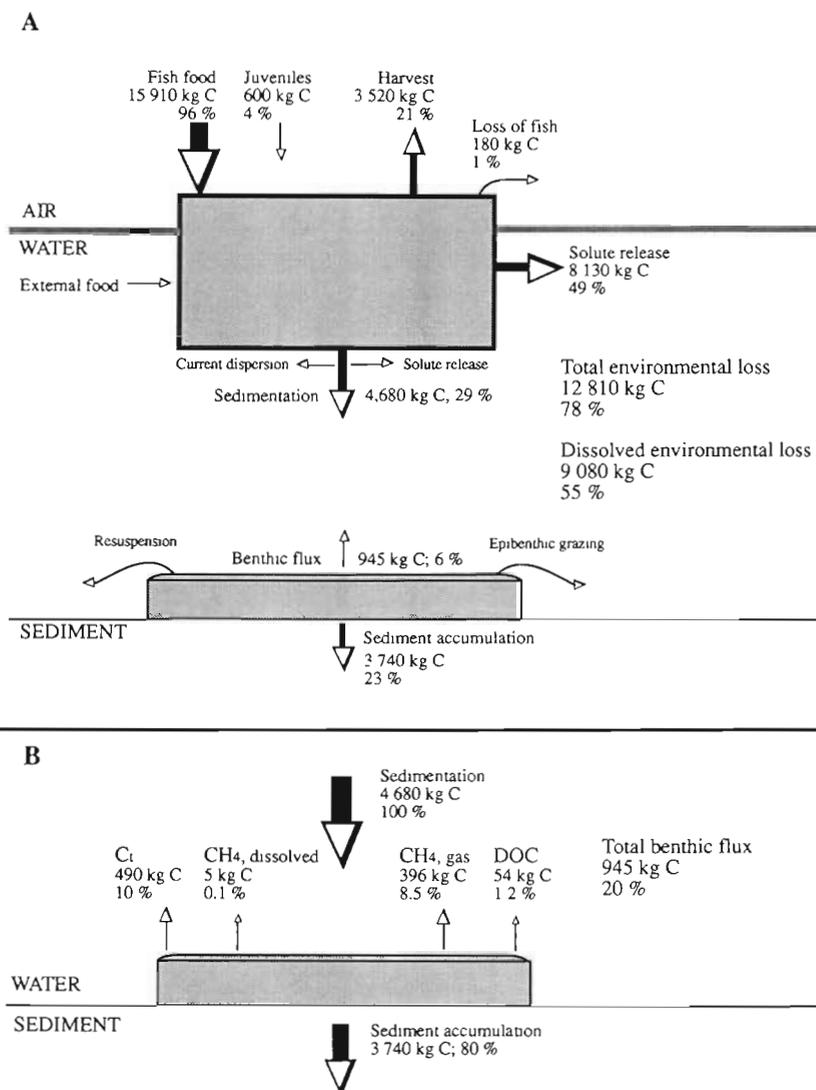


Fig. 5. Carbon mass balance in the cage farm constructed according to the flux method for the growing season of 1985. Fish loss is death and escape of fish. Unquantified arrows represent processes that constitute possible sources of error in the mass balance (see text). (A) Percentages are % of total carbon input to farm (the sum of fish food and juveniles). (B) Percentages are % of total carbon input to sediment

cages, before the particles reach the sediment. This release is then included in the mass balances, but in the solute release taking place directly from the cages, which is obtained as the difference between total environmental loss and sedimentation.

Sinking particles may also be dispersed by currents, or settled particles resuspended, away from the immediate vicinity of the farm, i.e. over a bottom area larger than the one used to calculate the benthic flux (on a seasonal basis) and larger than the area measured by divers to have been influenced by sedimentation from the farm (over the entire 7 yr period). This would result in our values of the benthic flux and of the long-term accumulation of carbon in the sediment being underestimates. The reported high sinking rates of particles from trout cage farms, and especially of food pellets (Gowen et al. 1985, and references therein), and the relatively shallow water column

below the bottom of the cages (ca 16 m), suggest that current dispersion of particles is not a major mechanism for the export of carbon from the farm. We have not estimated the importance of resuspension below the farm. This process may have contributed to the dispersion of deposited particles over the bottom area observed to have been influenced by the farm, and also beyond this area. It is important to note that the number and exact position of cages used in the farm varied between seasons. The bottom area influenced by the farm corresponds to the maximum number of cages ever used, and this area must be larger than the area that each season receives particles from the cages. Another possible pathway for the removal of carbon from the farm sediment is wild fish and mobile epibenthic animals eating excess food on the bottom. We occasionally observed fish below the cages during dives, but considering the high dissolved sulfide con-

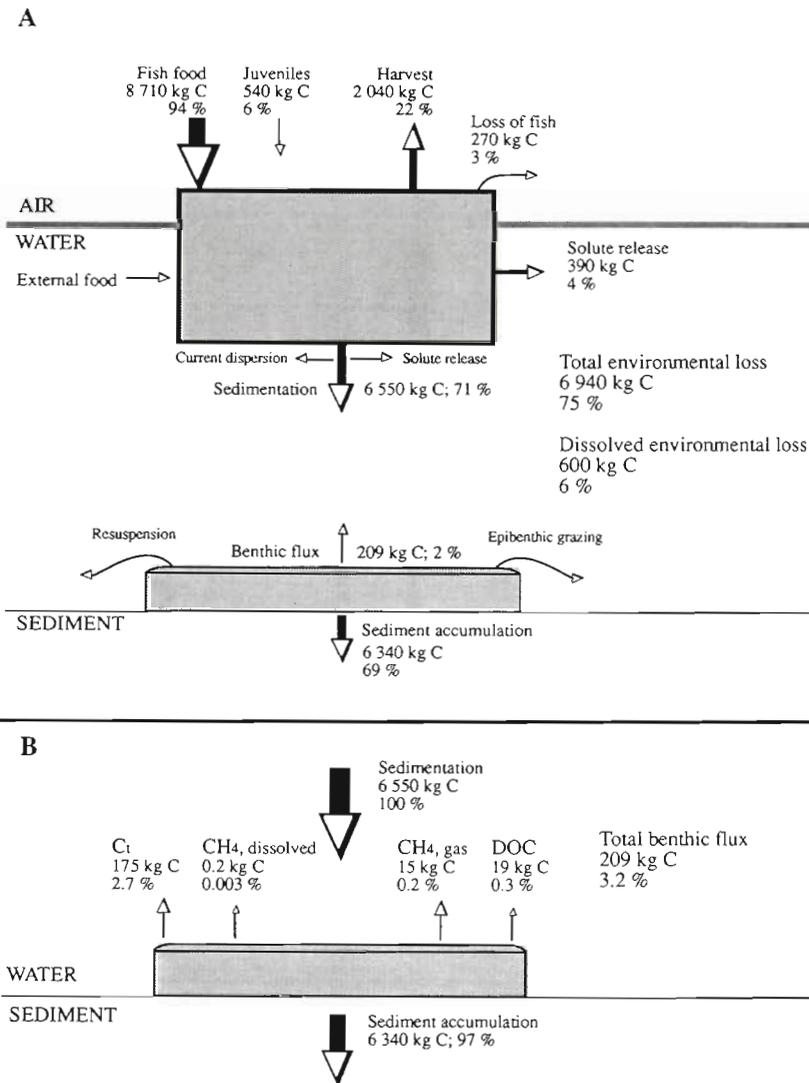


Fig. 6. Carbon mass balance in the cage farm constructed according to the flux method for the growing season of 1986. Fish loss is death and escape of fish. Unquantified arrows represent processes that constitute possible sources of error in the mass balance (see text). (A) Percentages are % of total carbon input to farm (the sum of fish food and juveniles). (B) Percentages are % of total carbon input to sediment

centrations in the sediment below the top few mm, grazing by fish and mobile epifauna on the bottom is most likely an insignificant process in the carbon budget of the farm. Observations in support of this conclusion were made by Avnimelech et al. (1981), who found that grazing by fish on fish pond sediments rich in food is limited, because of the anaerobic conditions in the sediment.

A possible additional carbon source to the farm sediment is the deposition of particulate 'external' carbon, produced outside the farm, on the bottom as well as in the sediment traps. The sedimentation rate of carbon 200 m outside the farm was $246.7 \text{ mmol C m}^{-2} \text{ d}^{-1}$ in May 1985, which is close to the annual mean sedimentation rate of carbon in the Gullmar Fjord (Lindahl 1988). Using this number as the background value for the farm, 'external' carbon constituted 4 to 8 % of the measured carbon sedimentation under the farm during

1985 and 1986. We do not consider this a large error in the mass balances or in the sedimentation measurements. A new method, based on high temperature catalytic oxidation, for the determination of DOC in seawater has recently been presented. Using this method, Sugimura & Suzuki (1988) reported DOC concentrations in ocean waters several times higher than those measured with previous techniques. In light of this new finding and the likely complex nature of the organic compounds released from the farm sediment, the DOC fluxes reported here may be underestimates.

Comparison of the 3 mass balances

The recovery in harvest of the total carbon input to the farm (Figs. 5, 6 and 7) agreed excellently in all 3 mass balances, being 21, 22 and 22 % during the

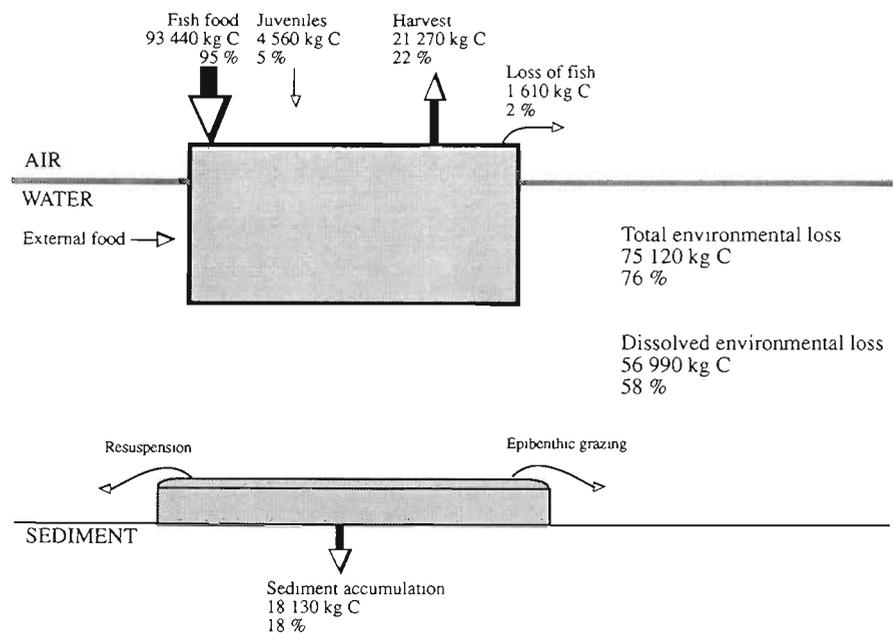


Fig. 7. Carbon mass balance in the cage farm constructed according to the accumulation method for the consecutive growing seasons of 1980 to 1986 (7 seasons). Fish loss is death and escape of fish. Unquantified arrows represent processes that constitute possible source of error in the mass balance (see text). Percentages are % of total carbon input to farm (the sum of fish food and juveniles)

1985, 1986 and 1980 to 1986 seasons, respectively. The contribution of fish loss (death and escape) to the overall carbon cycling in the farm was relatively small during all seasons (1 to 3%). Most of the carbon supplied to the farm was lost to the environment. The total environmental loss amounted to 878, 952 and 907 kg C per tonne of fish produced during the 1985, 1986 and 1980 to 1986 growing seasons, respectively. This corresponded to 78, 75 and 76% of the total carbon input to the farm during these 3 periods. The 3 mass balances agreed well with regard to the contribution of food (94 to 96%), juveniles (4 to 6%) and harvest to the turnover of carbon within the cages, as well as the fractions of the carbon input to the farm that was recycled in the sediment (benthic flux; 2 to 6%) and that was lost to the environment (see Figs. 5, 6 and 7). The agreement between the mass balances for 1985 and 1980 to 1986 with regard to sediment accumulation (18 to 23%) and dissolved environmental loss (55 to 58%) was also good. There was some discrepancy between the 1985 and 1986 seasons, however, in the individual contributions of sedimentation (29 to 71%), solute release from the cages (4 to 49%), sediment accumulation (23 to 69%) and dissolved environmental loss (6 to 55%) to the overall carbon cycling in the farm (Figs. 5 and 6). One possibility is that we measured an unusually high sedimentation during 1986. This would explain the low solute release from the cages, the high sediment accumulation, the small fraction of sedimented carbon that was released to overlying water (Figs. 5 B and 6 B) and the small dissolved environmental loss during that season compared to 1985.

Comparison with previous studies

There are no other complete carbon mass balances, including cycling and accumulation in the sediment, reported for fish cage farms with which we can compare. Penczak et al. (1982) calculated budgets for the cages of a lake water trout farm in Poland and estimated a total environmental loss of 750 kg C per tonne of fish produced. The recovery in harvest of the carbon supplied with fish food was 21.5%, which is close to our estimates of 21 to 22% (though we also included supply of juveniles to the farm). The total environmental loss reported in the Polish study was less than the values of 878 to 952 kg C per tonne of fish produced obtained here. Phillips et al. (1985b) constructed mass balances for the cages of a Scottish freshwater loch trout farm and found that, of the carbon input with food, 21.6% was retained in fish (harvest) and 78.4% was lost to the environment. These results are very similar to those we report. Phillips and coworkers calculated a total environmental loss of 560.5 kg C per tonne of fish production, with the major percentage of the carbon loss being in dissolved form. Gowen et al. (1985) predicted a carbon mass balance, based on theoretical calculations, for cages of coastal salmon farms. They found that, of the carbon supplied with food to the farm, 16% was retained in fish (harvest), 44% ended up as a particulate environmental loss (faeces and excess food) and 40% as a dissolved environmental loss (i.e. a total environmental loss of 84%). However, Gowen and coworkers assumed that 20% of the food input was wasted, i.e. not eaten by the fish. Based on their data, we calculated that urea-

carbon constituted 2 % of the dissolved environmental loss, the remainder being CO₂-carbon produced during fish respiration. We cannot fully check the consistency of our results with others until further studies of carbon fluxes and turnover in marine cage farms and underlying sediments have been reported.

The release of dissolved and gaseous carbon from the sediment to overlying water was dominated by C_i during all months except September, when gas-carried methane fluxes were as high or higher than C_i fluxes (Table 3). The C_i fluxes measured here are up to 50 times higher than C_i fluxes from 'normal' coastal sediments of the Gullmar Fjord at corresponding temperatures (Anderson et al. 1986). Blackburn et al. (1988) estimated C_i fluxes from sediments of marine fish ponds in Israel during spring. They obtained values of 107 to 187 mmol m⁻² d⁻¹ (including effects of fish bioturbation), which is within the range of C_i fluxes measured in this study. We found that, on a seasonal basis, C_i made the largest contribution to the total benthic carbon flux followed by gaseous methane or DOC. Dissolved methane made the smallest contribution to the seasonal benthic flux in both 1985 and 1986 (Figs. 5 B and 6 B). The methane content of the gas samples collected at the bottom in this study was 97 to 99 %. High methane concentrations in gas bubbles released from sediments (70 to 90 %) have also been reported from other marine fish cage farms in Norway (Samuelsen et al. 1988).

Environmental and biogeochemical implications

This study, having quantified the environmental carbon load from the farm and the composition of this load, contributes an assessment of the eutrophication effects caused by a fish farm. The exact environmental perturbations, which are also dependent on the physical, chemical and biological site-specific characteristics of the receiving system, cannot be understood from the present study. They have to be investigated separately and considered in relation to other possible sources of organic waste. It is clear, however, that the sediment derived from the farm is extremely organic-rich, highly reducing and very sulfidic. The seasonal range of measured benthic oxygen uptake rates was 90 to 180 mmol m⁻² d⁻¹, which is 12 to 15 times the rates measured on 'normal' coastal sediments of the Gullmar Fjord (Hall et al. 1989). The gas bubbles that are released from the sediment may contain low concentrations of hydrogen sulfide, which is very toxic to fish, and bring this compound to the cages, thereby forming a coupling between sedimentary processes and fish health. A fraction of the benthic ebullative flux is not dissolved in the water column, but constitutes a pathway for the trans-

fer of methane from the sediment to the atmosphere. The relatively low carbon sedimentation rates measured 200 m outside the farm, which were 13 to 22 times lower than below the cages, suggest no influence by the farm at this distance from the cages. The bottom area influenced by the farm was restricted to a distance of 5 to 40 m from the outermost cages. This is, however, a somewhat larger areal influence than what has been observed around other fish cage farms. Enell & Löf (1983), Weston (1986), Brown et al. (1987), Aure et al. (1988) and Kaspar et al. (1988) all observed bottom effects limited to the immediate vicinity of the cages.

Studies of the turnover and cycling of biogenic elements in coastal fish cage farms are important also for an improved understanding of biogeochemical processes in marine hypertrophic systems, such as the pathways of organic carbon decomposition in extremely organic-rich sediments. Such studies may also help predict the consequences of heavy eutrophication of coastal areas.

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