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Accumulation and mineralization of fish farming residuals in the benthic footprint area at two different farming locations in a Faroese fjord

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Abstract

In order to quantify the accumulation of farming derived organic carbon in the footprint area of two hydrographically different fish farming sites in a Faroese fjord, a simple distribution estimate based on current profile records was combined with measurements of organic carbon content in the sediment. One farming site (F1) had low near seabed current speeds while they at the other site (F2) frequently exceeded 10 cm s^{-1}. The calculated estimate successfully described the organic carbon distribution at F1, while frequent resuspension events at F2 implied not valid estimates. The total inorganic carbon (DIC) release from the sediment due to mineralization increased linearly with increasing sediment organic carbon content. There furthermore was a substantial release of dissolved organic carbon from the fish farming impacted sediments. As for the DIC efflux the sediment oxygen uptake increased with increasing organic carbon content of the sediment. The average DIC efflux was, however, 1.8 times higher than the sediment oxygen uptake, reflecting a continuous accumulation of reduced metabolites in the farming affected sediments. The ammonium efflux from the sediments was site specific, with considerably higher effluxes at low current site (F1) than at the site with stronger resuspension (F2).

Keywords: Benthic impact, Organic carbon, Footprint size, Mineralization, Accumulation

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Introduction

Fish farming in coastal areas is a growing industry world-wide, and the competitive use of coastal resources has highlighted the importance of satisfactory control measures to protect the natural environment (FAO 2009). The most evident environmental impact of fish farming is the accumulation of organic carbon in the seabed below the farm, and decreased oxygen availability for the benthos beneath fish cages (Holmer & Kristensen 1992, Christensen et al. 2003, Hyland et al. 2005), and in accordance with this benthic conditions receive a lot of attention in the environmental monitoring of aquaculture.

In order to assist regulators in site evaluation and decision making, and to support the farmers in their planning and husbandry, models are developed to predict the benthic load from aquaculture activities. The most cited models are the MOM (Ervik et al. 1997, Stigebrandt et al. 2004), DEPOMOD (Cromey et al. 2002a), and AWATS (Panchang et al. 1997, Dudley et al. 2000) systems. The MOM model estimates the holding capacity with regard to the benthic fauna and the water quality in the net pen and in the surroundings of the fish farm (Ervik et al. 1997, Stigebrandt et al. 2004). The DEPOMOD and AWATS systems are both particle tracking models. One main difference between these two models is the inclusion of the current. In DEPOMOD the main practice is to use a record of the current profile to determine the particle dispersion. In this way the vertical variations may be included, but it neglects spatial changes in hydrodynamics (Cromey et al. 2002a). The AWATS system includes a 2D-hydrodynamic model, but neglects possible vertical shears in the current (Panchang et al. 1997, Dudley et al. 2000). The spatial grid resolution in hydrodynamic models covering relatively large areas like fjords or fjord systems is at the best of the same order as the largest cages, which makes them less attractive when studying small scale variations in the close vicinity of fish farms. However, they have clear advantages in studying the distant fate of aquaculture waste.

Clearly modeled estimates are the way to go in evaluating the benthic impact of farming. However, in order to evaluate the amount of carbon accumulated at the seabed from food input to the farm, detailed knowledge on the farming practice and site has to be obtained, since the nature of the benthic impact is highly dependent on site characteristics such the hydrodynamic conditions, temperature and water depth (Kempf et al. 2002), as well as the farm size and husbandry methods (Menthe et al. 2006, Mayor et al. 2010). In addition, different benthic communities express variable sensitivity to elevated sedimentation and eutrophication (Heilskov et al. 2006, Macleod et al. 2007, Pusceddu et al. 2007). Some of these variables are readily quantified, such as hydrographics, but even parameters such as food loss is not easy to quantify (Chamberlain & Stucchi 2007), while the dependence on site specific benthic conditions are even harder to access.

In the Faroe Islands fish farming is licensed in the vast majority of fjords and sounds. The farmers are required by the authorities to undertake bottom samples during the production cycle. If given parameters are exceeding certain threshold values, the farmer is by the authorities ordered to reduce the feeding or eventually to move the entire plant. For reliable samples repeated over time it is of importance that they are not taken in areas with large gradients in the footprint.

In this study we combine measurements of sediment organic carbon content with a simple distribution estimate based on current profile records, and the accumulation of farming derived organic carbon in the footprint area is quantified at two farms in a Faroese fjord, further we evaluate to what extent this may be used as a guideline for environmental survey sample strategy for farming sites. The correlation between the amount of organic carbon and the oxygen consumption of the sediment, mineralization of organic carbon and nutrient release from the sediment is likewise investigated.

Materials and methods

Study site. The study sites were in Kaldbaksfjørður, in Faroe Islands, a 6.6 km long and 500 - 1700 m wide fjord, with a surface area of 5.41 km^2. At the entrance of the fjord there is a ~30 m deep sill, while the maximum depth in the fjord is 60 m (Fig. 1). The water circulation in Kalbaksfjørður is mainly wind or estuarine driven. The area outside the fjord is characterized by a semiamphidromic point for the semidiurnal tidal constituents, which otherwise are the dominant constituents for the region. This implies weak tidal currents in the fjord.

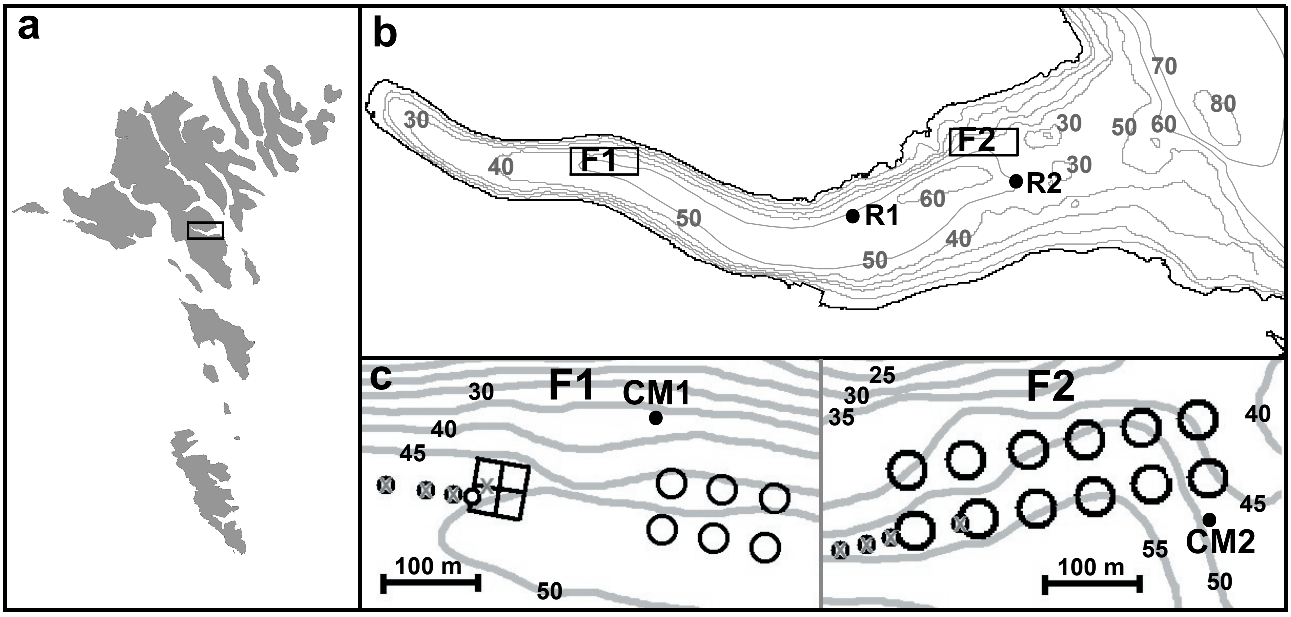

Figure 1. Map of the Faroe islands (a) and Kaldbaksfjørður (b). The farming sites F1 and F2 and the respective reference sites R1 and R2 are marked. (c) The farming sites F1 and F2 where the locations of the floating net cages are shown. The position of the current measurements are marked CM1 and CM2. Sediment sampling stations are indicated by black circles, while grey x indicate the sedimentation sampling stations.

Fish farming in the fjord represents the major source of anthropogenic organic carbon and nitrogen, since most of the catchment area is uncultivated and uninhabited (Mortensen 1990). In the fjord there are two areas of 0.21 and 0.25 km^2 licensed for fish farming.

In 2006 farming started in late April when 100 - 105 g large juvenile rainbow trout (*Oncorhynchus mykiss*) were introduced into cages at the inner location (F1; Fig. 1). The study at F1 started with the onset of farming and continued until September 2006. At F2 farming started in October 2006 when ~ 1 kg large trout were moved from F1 to F2, and ended in May 2007. At F1 four square cages of 20 x 20 m mounted in a platform (2025 m^2) were investigated. During the study period of 4.6 months the biomass increase at the platform was 151 tonnes while the food usage was 169 tonnes. At F2 the entire farming operation (12 circular net cages, 40 m diameter) was included in the study. The total biomass increase was 1596 tonnes while the total food usage was 1959 tonnes during the 7.5 months study period at F2.

Sediment time series samples were collected 5 times at the western edge of the platform and one circular cage at F1 and F2, respectively (Fig. 1c). In addition samples were collected at various distances to the farms at the end of the study periods. Reference samples for F1 and F2 were collected at the respective stations R1 and R2 (Fig. 1b).

Sedimentation traps (KC-Denmark; Lundsgaard et al. 1999) were moored to the platform (F1) at 20, 40 and 45 m depths, to a circular cage at F2, and at the reference station (R1) at 20, 40 and 50 m depths (Fig. 1). At the end of the study periods, sedimentation traps were moored (20 and 40 m dept) at the same locations as the sediment sampling locations (Fig. 1c).

Farming activity. Information on the fish biomass and food usage was derived from the farmer. The farming operation was controlled by a farm control monitoring system with automatic feeding (http://www.akvagroup.com). The water content in food pellets used at the farms was 6%, and the organic carbon (OC) content of the food was 51.5% and 51.9% of dry weight at F1 and F2, respectively (á Norði et al. submitted).

Currents. Depth profiles of the current were obtained for the two farming sites by deployments of a bottom mounted 300 kHz RDI Workhorse Acoustic Doppler Current Profiler (ADCP) in the vicinity of the farming plants (Fig. 1c). At F1 the instrument was deployed at 38 m depth on the 22’th of March, and at 48 m depth on 11’th of September at F2. The instrument was installed for 64 and 70 days at F1 and F2, respectively. At both deployments the instrument was measuring in 2 m vertical bins with the center of the first bin about 3 m above the bottom and with a time interval of 20 minutes.

Footprint estimation. The sinking velocities of fecal and feed pellets is found to be about 3.2 ±1.1 cm s^{-1} and 10 cm s^{-1}, respectively (Cromey et al. 2002b). Fecal pellets represented the vast majority of sinking particles below the fish farming sites (á Norði, et al. submitted) and thus only the sinking velocity of fecal material is included in the footprint estimation. The vertical mean of the horizontal current components of the two measurements is estimated from the deepest registration up to the bottom of the cages (Fig. 2). The vertical travel distance of the material is assumed to be from the bottom of the cages down to a flat bottom. With these assumptions the distribution plots of the currents directly indicate the shape of the average footprint from a single point source.

In order to mimic the footprint from the fish farms, they are included into a grid with a resolution of 4 m. In the estimations each grid cell required to represent the plant is considered as a point source. The footprint is obtained by summing up the distributions from all the cells needed to represent the actual fish farm.Both current records exceeded two months, and are assumed to be fairly representative for the current variations at the two sites. However, as with the most common practice with the DEPOMOD model, the spatial variations in the currents across the site are neglected with the used approach.

At the inner part of the fjord (F1) the fish farm consisted of 4 squared 20 x 20 m cages with approximately 4 m between each cage. In the calculations each cage is then represented by 5 x 5 grid cells. The bottom of these cages was at 12 m depth, and the bottom depth under the plant was 48 m. At the outer part of the fjord the fish farm (F2) consisted of 12 circular 15 m deep cages, organized in two rows with 6 cages in each and about 70 m between the centers of the cages (Fig. 1). The surface area of the circular cages was 1256m^2, and in the calculations squared cages are assumed, covering the same area. The bottom depth under this plant was 52 m.

In the distribution estimate the amount of released waste material is normalized to 100 units m^{-2} in average for the entire farm. However, during the farming period the total food input to the individual cages differed slightly between cages, and these relative changes are included in the released material input in the estimate.

Sedimentation. The sedimentation traps were replaced approximately every second week. The traps were preserved with formaldehyde added to dense seawater (salinity ~60). Preserved traps, might be compromised by migrating zooplankton caught in the traps (Gundersen & Wassmann 1990). However since the amounts in the traps in the present study were extremely high relative to natural conditions, these effects are considered to be relatively small.

Sub-samples of the trap content were filtered on pre-combusted (475°C) and pre-weighed Whatman GF/F filters. The filters were flushed with artificial saline water in order to remove dissolved organic carbon and nitrogen. The amounts of total particulate material (TPM) were determined as the weight gain of the filters after filtration and drying at 60°C. Particulate organic carbon (POC) and particulate nitrogen (PN) content was determined from sub-samples of the dried filters, after fuming with HCl.

In order to quantify the vertical fluxes of fish farming derived organic carbon at the farms, the fluxes at the reference station were subtracted.

Sediment characteristics. Sediment was retrieved with a HAPS bottom corer (KC-Denmark; Kanneworf & Nicolaisen 1973). Only cores with a clear water phase were used. For further analysis sub-cores were collected in Plexiglas tubes (i. d. 5.6 cm). Cores were kept dark and at bottom water temperature during transport to the laboratory, which was reached within 4 hours of sampling. For determination of porosity, total organic carbon (TOC) and total nitrogen (TN) content, two cores from each station were sectioned into 1 cm intervals down to 4 cm depth, and 2 cm intervals down to 12 cm depth. Sediment porosity was determined from density and water content measured as the weight loss after drying at 70°C for ~48 hours. TOC and TN were measured on a CE 440 Elemental analyzer after the sediment had been homogenized, acidified (4-5% H_2SO_4), and dried. Total organic carbon and nitrogen (g m^{-2}) in the top 12 cm of the sediment, was determined from the TOC and TN content of the dried sediment, and the sediment water content. In order to estimate the amount of fish farm derived organic carbon and nitrogen in the sediment below the fish farms F1 and F2, the background content at the respective reference stations R1 and R2 was subtracted.

Sediment water fluxes. Upon return to the laboratory three sediment cores from each station were submerged in an incubation tank, holding bottom water at *in situ* temperature ± 0.5°C. Cores were held in the tank for 12-24 hours prior to the onset of flux measurements. On most sampling dates, the *in situ* bottom water O_2 was 90% - 100% air saturated, and the water in the incubation tank was flushed with air, maintaining the ambient water at 100% air saturation. However, on two sampling dates, 22 August and 7 September, the *in situ* oxygen concentration of the bottom water was only ~50% of air saturation. On 22 August, the incubation water was flushed with a mixture of nitrogen and air, keeping the oxygen concentration at 60% air saturation, but on 7 September (the transect measurement at F1), the incubation water was flushed with air and these data have to be treated with caution. In order to ensure well mixed conditions in the core liners, small teflon-coated magnets were attached to the inner wall of the cores, receiving momentum from an externally rotating magnet (Rasmussen & Jørgensen 1992). The stirring resulted in a measured effective diffusive boundary layer (DBL) thickness of 344 µm ± 25 (SE).

Sediment water exchange rates of dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), nitrate and ammonium and the total oxygen uptake (TOU) was measured in closed sediment cores, with an internal water height of ~8 cm. During the incubation the oxygen concentration was monitored with a Clark type oxygen minielectrode (Revsbech 1989, Glud et al. 1995). Seawater samples for measurements of O_2, DIC, DOC, NO_3^- and NH_4^+ concentrations were taken at the start of the incubation and when the oxygen concentration inside the respective cores had decreased 15-20 %. Samples were stored and analyzed as in á Norði et al. (submitted) and is only described shortly here. The oxygen content was determined by Winkler titration (Grasshoff et al. 1999), DIC content was determined on an infrared gas analyzer (ADC-225-MK3), NH_4^+ was measured manually by the salicylate-hyperchlorite method (Bower & Holm-Hansen 1980). NO_3^- was measured on an autoanalyzer (Grasshoff et al. 1999).

Sediment-water fluxes were calculated linearly from the change in solute concentration during incubation, accounting for incubation time and enclosed water volume. A linear decline in the O_2 concentration was confirmed from the continuous recording of the minielectrode. Sediment uptake rates are defined as negative fluxes while release from sediment is defined as positive fluxes. The increases in fluxes due to farming activity were determined by withdrawing the fluxes at the reference stations. The presented fluxes at F1 are from á Norði et al. (submitted), which presented total fluxes and not increased fluxes due to farming activity.

Oxygen microprofiles were measured in the same cores 12 h after the flux measurements had been terminated by cap removal. Profiles were obtained with a Clark-type microelectrode equipped with a guard cathode and an internal reference (Revsbech 1989). The tip diameter was ~10 µm, stirring sensitivity <1% and the 90% response time was <1 s (Gundersen et al. 1998, Glud et al. 2000). Three profiles were measured in each core, adding up to a total of 9 microprofiles at each measuring event. The microelectrodes were positioned by a motor driven micromanipulator and profiles were measured at a depth resolution of 50 µm. The sensor current was measured by a picoammeter connected to an A/D converter, which transferred the signal to a PC (Revsbech & Jørgensen 1986). The microelectrode was calibrated by two-point calibration from the signal in the well mixed air-saturated water and the signal of the anoxic sediment.

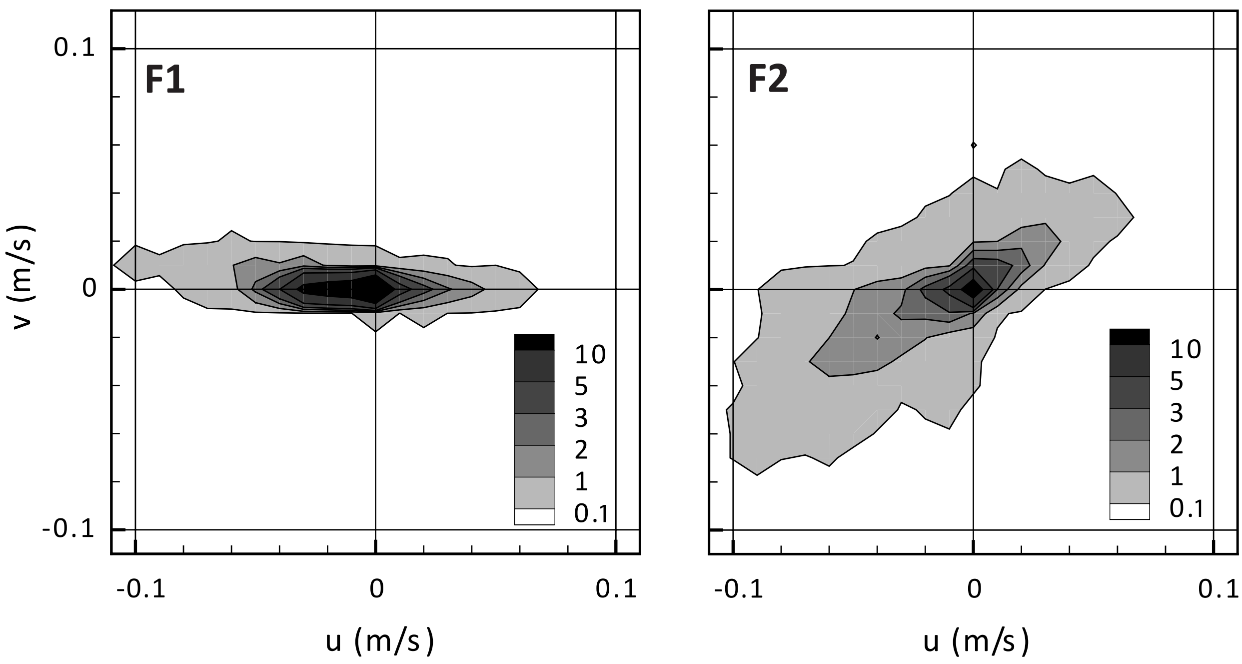

Figure 2. Frequency plots of the vertical mean velocity components estimated for the registrations at depths below the bottom of the cages at sites F1 (left) and F2 (right). The contours show the percentage of the registrations in 1 cm s^{-1} intervals for both velocity components.

The diffusive oxygen uptake (DOU) was calculated from the d microprofiles by DOU = $-D_0\ \delta C/\delta Z$, where D_0 is the temperature corrected molecular diffusion coefficient and C the oxygen concentration at depth Z within the DBL (Jørgensen & Revsbech 1985).

Results

Currents. The current speed at F1 was less than 10 cm s^{-1} more than 95% of the time at all depths with generally weaker currents towards the bottom. The dominant current direction was along the fjord, with a residual inward current of 1-2 cm s^{-1} at all depths (Fig. 2). Although the profiler was

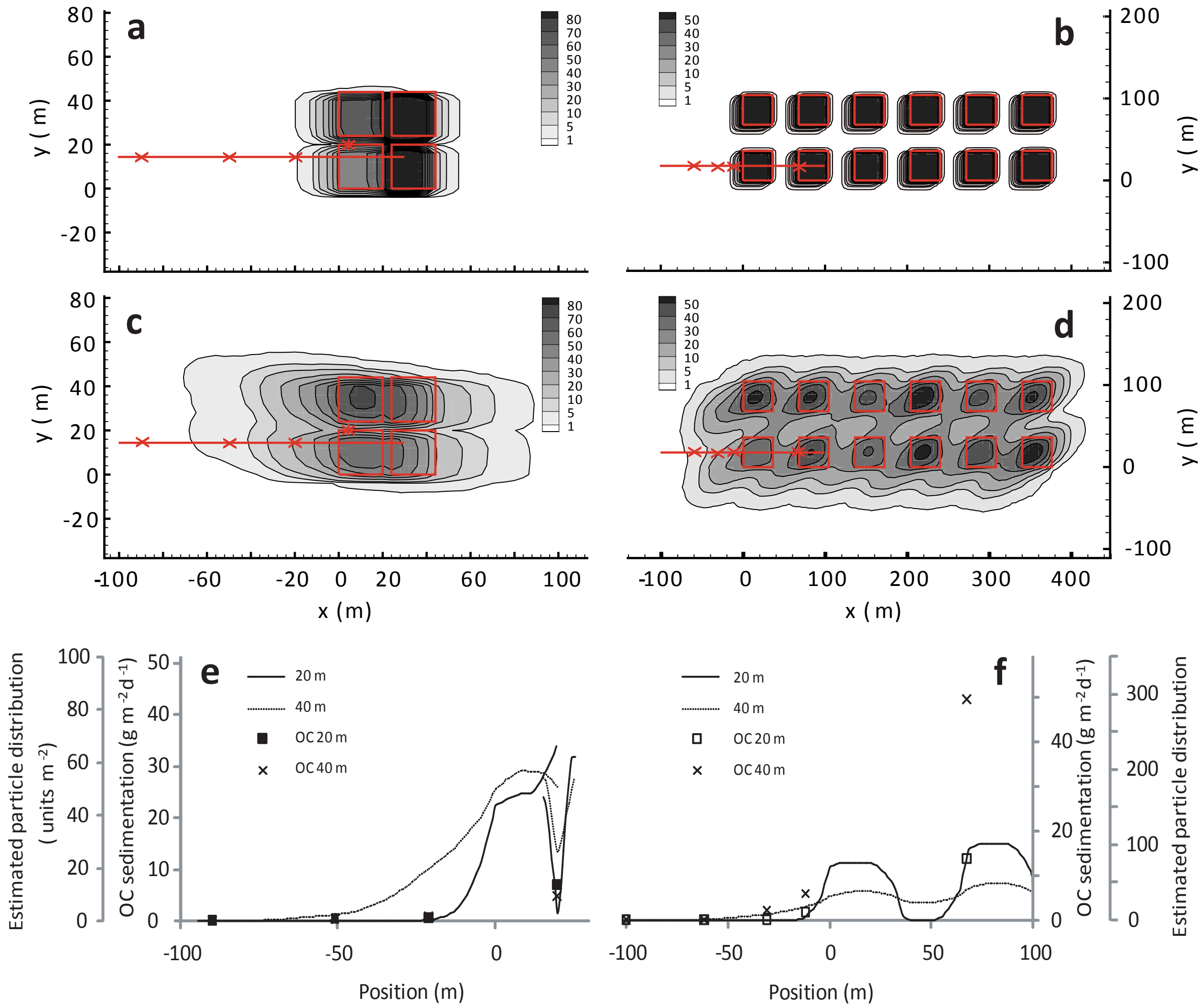

Figure 3. Estimated distribution of farming residuals at 20 (a,b) and 40 m (c,d) depth in the water column, and measured vertical fluxes of farming derived organic carbon (OC) at same depths (e,f). The farm (F1) is presented in the left panel while F2 is in the right panel. Red squares represent the cages. X indicate the positions of vertical flux measurements, while lines indicate the estimated values presented in e and f. The measurements are fitted to the estimate by the relative amounts of farming derived organic carbon at 20 m depth.

positioned slightly closer to land and at shallower water than the plant (Fig. 1), other measurements in the area (data not shown) indicate, that these measurements are representative for the area.

The energetic flow pattern at F2 was stronger and more complicated than at F1. Close to the surface and also close to the seabed the current speeds were in the range from 10-20 cm s^{-1} in 20% and 15% of the registrations, respectively, while the area at 20-24 m depth was relative slack with speeds less than 13 cm s^{-1}. However, the mean horizontal speeds rarely exceed 10 cm s^{-1} in the frequency plot (Fig. 2) due to the lower speed area at mid-depths, and due to the fact that the uppermost registrations were not included. Close to the surface the currents were mainly parallel to shore (W-E direction), while the currents close to the bottom were in the SW-NE direction. The residual currents were in the same directions, with a mean flow directly into the fjord in the uppermost layers, and towards SW in the deeper layers.

Sedimentation. The estimated distribution of farming derived particles at 20 m depth in the water column implies very steep gradients adjacent to the net cages where the sedimentation measurements were conducted (Fig. 3a and 3b), while the particles at 40 m depth were more dispersed (Fig. 3c and 3d). Within 3 m of the sedimentation measurements the modeled particle distribution at 20 m depth changed 1.6 - 2.8 folds at most measuring sites. However, the trap neighboring the cages in the platform at F1 was positioned where the estimate suggests a 9 fold change in particle distribution within 3 m (Fig. 3a and 3c). Due to variations in currents the sedimentation measurements are assumed to represent the average sedimentation surrounding the traps.

Despite the steep gradients the measured vertical fluxes of particulate organic carbon at various distances to the net cages corresponded quite well to the predicted particle distribution at 20 m depth at both farms ($p < 0.01$; Fig. 3e and 3f). The vertical POC fluxes at 40 m depth did, however, not correspond to the estimates. At F1 the fluxes were smaller than estimated, while the opposite was observed at F2 (Fig. 3e and f). Close to the seabed the measurements might be influenced by resuspension. Time series measurements adjacent to the net cages show that the vertical OC flux increases with depth at both farms (n= 10 at F1 and n=8 at F2). At F1 the increase was somewhat less than predicted by the calculations, while the vertical flux at F2 increases although it according to the estimate was expected to decrease. Assuming

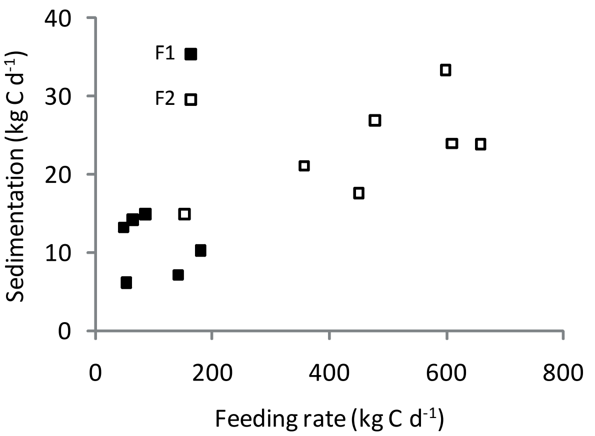

Figure 4. The relation between organic carbon supply with food to individual net cages and sedimentation of farming residuals at F1 and F2. The sedimentation is measured with traps at 20 m depth at one point and extrapolated to the total sedimentation surrounding the net cage based on the estimates in Fig 3a and 3b.

that this discrepancy was due to resuspension, 64 ± 7 (SE)% of the vertical OC flux at 40 m depth was due to resuspension, while 97 ± 20 % of the flux at 50 m depth was due to resuspension.

Taking into account the different amounts of particle release in the cages adjacent to the sediment traps, and the proportion of particles captured in the traps at 20 m depth neighboring the net cages at F1 and F2, sedimentation of fish farming residuals increased linearly with increasing organic carbon input with food ($p < 0.05$; Fig. 4), and the sedimentation at the two fish farms corresponded to 10.0 ± 2.1% of the organic carbon supplied to the cages with food.

Benthic footprint. According to the estimate the benthic footprint was quite patchy, with highest amounts of farming residuals directly below the cages (Fig. 5a and 5b). At F1 farming residuals (> 1 unit m^{-2}) are found at a maximum distance of 93 m from the cages at F1 and 116 m from the cages at F2 in the net current direction. The total benthic footprint at F1 was 10336 m^2, while it was 107280 m^2 at F2. Thus the footprint at F2 covered half of the licensed area for farming (0.21 km^2). As for the sedimentation the distribution of farming residuals resembled the estimated distribution well at F1 ($P < 0.025$). However, at F2 the correlation between the estimate and the observations was not significant ($p=0.074$; Fig. 5c and 5d).

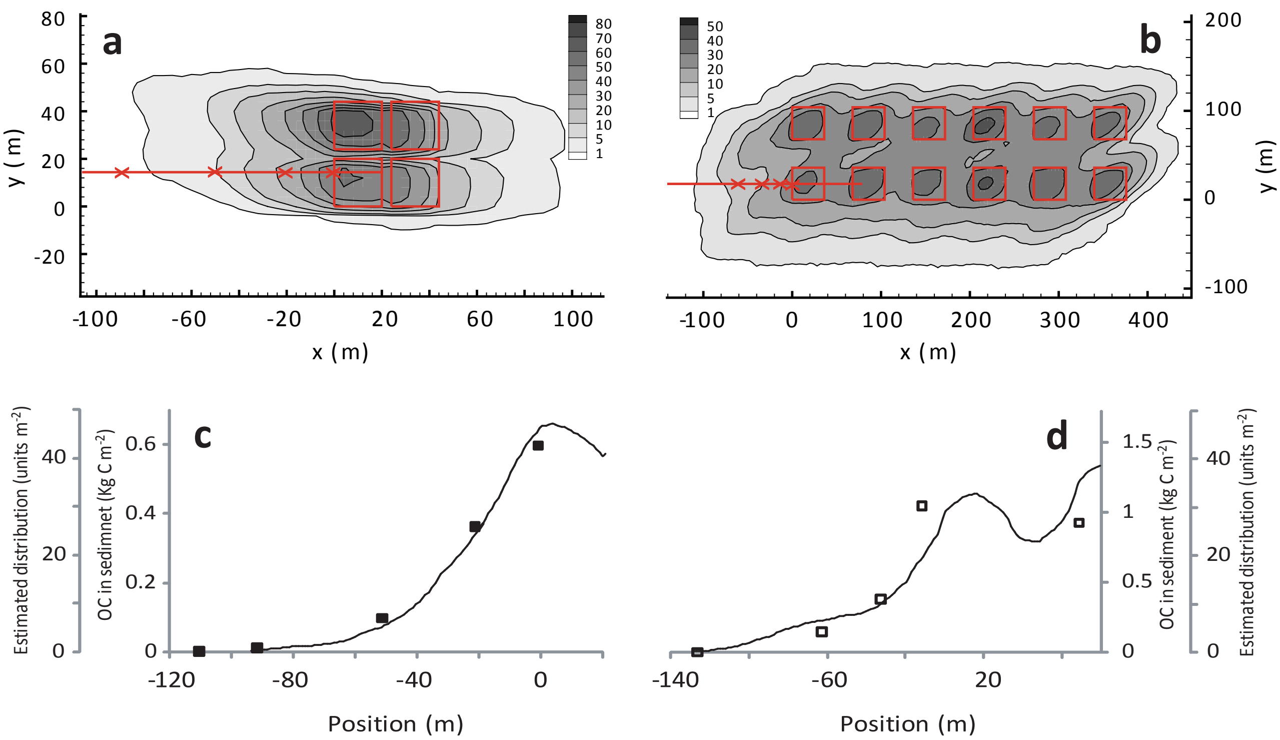

Figure 5. Estimated particle distribution in the sediment at F1(a) and F2(b). Red x and lines indicate the position of sediment measurements and the estimated values presented in c and d. Estimated (line) and measured (squares) distribution of farming derived organic carbon at the seabed surrounding the fish farms F1 (c) and F2 (d). The measurements are fitted to the estimate by the relative amounts of farming derived organic carbon (OC) in the sediment.

At the times of the two transects the measurements extrapolated over the entire estimated footprint area imply that the total amount of farming derived organic carbon in the benthic footprint areas at F1 was 2.2 tonnes, corresponding to 2.7% of the OC input with food. At F2 the total amount of organic carbon at the seabed can not be quantified from the OC measurements and the estimate, since the correlation was not significant.

Sediment description. Prior to farming the sediment at station F1 resembled the sediment at the reference station R1 (Table 1), while the sediment at F2 resembled R2. The two reference stations were, however, quite different. At station R1 the sediment consisted of equal proportions of mud and sand, while the sediment at station R2 contained less mud, and coarser sand (Table 1). The organic carbon content at station R1 was 1.78 ± 0.07 mmol C g^{-1} (dw) while it was considerably less at station R2 (1.14 ± 0.32 mmol C g^{-1} (dw)). Likewise did the sediment at R1 contain more nitrogen than at R2 (Table 1).

Farming activity immediately changed the benthic conditions, at both farming sites. First, distinct fecal and food pellets were observed along with patches of the sulphur bacteria *Beggiatoa* spp. (Jørgensen and Revsbech 1983). After 2 months with farming activity the sediment at both farming sites was covered by *Beggiatoa*. A black zone emerged, which early in the farming period was located at the top of the sediment, progressed down through the sediment as farming continued. After four months with farming at F1, (total food usage 38 kg C m^{-2}) (60 tonnes C) methane bubbles were observed from ~6 cm depth in the sediment and down to the maximum sampling depth

Table 1. Sediment characteristics (± SE) at the reference stations R1 and R2 and of the farming stations F1 and F2 when visible affected by farming. Values are mean from 0-4 cm depth. Numbers in brackets are the total number of samples.

	R1	F1	R2	F2
Grain size (% of total)				
<63 µm	45 ± 1.7 (2)	52 ± 2.0 (7)	34 ± 6.4 (6)	26 ± 1.7 (12)
63-250 µm	48 ± 3.9 (2)	36 ± 1.8 (7)	41 ± 8.4 (6)	48 ± 1.5 (12)
250 - 710 µm	6 ± 5.0 (2)	9 ± 0.8 (7)	15 ± 3.8 (6)	19 ± 1.9 (12)
710 - 2000 µm	1 ± 0.7 (2)	3 ± 0.6 (7)	8 ± 2.0 (6)	8 ± 0.5 (12)
> 2000 µm	0 (2)	0 (2)	1 ± 0.8 (6)	1 ± 0.3 (12)
Density (g cm^{-3})	1.29 ± 0.02 (27)	1.07 ± 0.03 (20)	1.53 ± 0.07 (4)	1.22 ± 0.05 (11)
Porosity	0.71 ± 0.01 (27)	0.75 ± 0.02 (20)	0.67 ± 0.05 (4)	0.74 ± 0.02 (11)
Organic carbon (mg g^{-1} (dw))	1.78 ± 0.07 (31)	5.55 ± 0.37 (24)	1.14 ± 0.32 (5)	6.43 ± 1.86 (11)
Nitrogen (mg g^{-1} (dw))	0.2 ± 0.01(31)	0.41 ± 0.02 (24)	0.14 ± 0.04 (5)	0.41 ± 0.10 (11)

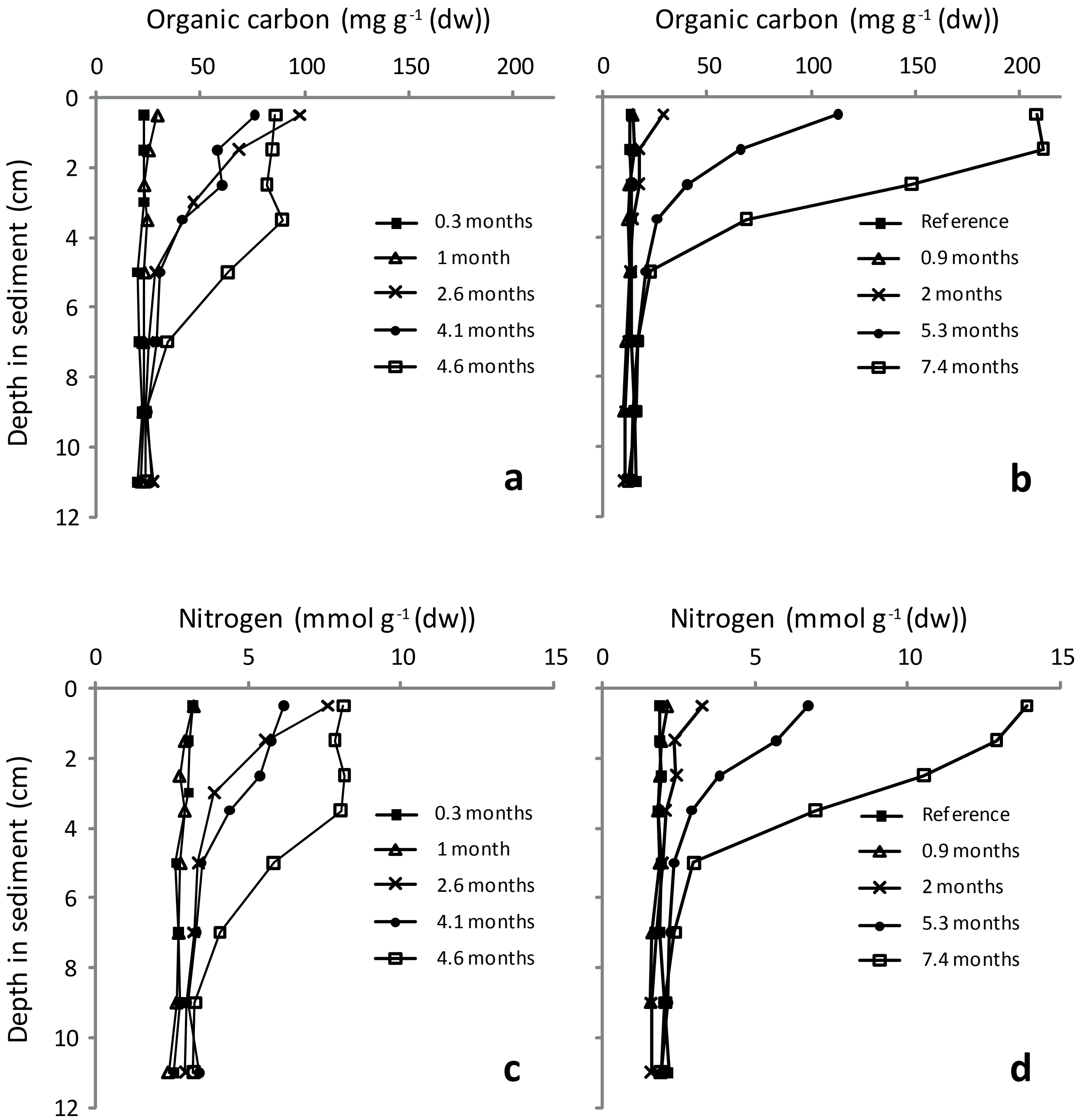

Figure 6. Organic carbon (a,b) and nitrogen content (c,d) in sediment below the fish farms F1 (a,c) and F2 (b,d) at various stages after farming started at the sites.

(>12 cm). At F2 methane bubbles were only observed at the last sampling, 7.4 months after farming had started at the site. At that time the total amount of food input since the onset of farming was 61 kg C m^{-2} (955 tonnes C). Farming activity considerably increased the organic carbon and nitrogen content in the sediment (Table 1), and continuous accumulation was observed at both sites during the sampling period (Fig. 6).

Mineralization rates in the footprint. From a total of 17 measurements of farming affected sediment at the two different farming sites (time series and transects), the relations between sediment water fluxes (elevated to reference fluxes) and farm derived organic carbon content in the sediment was investigated (Fig. 7).

The diffusive oxygen uptake (DOU) showed no difference between the two farming sites, nor was there a correlation between sediment OC content and DOU, but rather DOU asymptotically approached a saturation level of 140 mmol m^{-2} d^{-1} (Fig. 7a). The total oxygen uptake (TOU) on the other hand, increased linearly with increasing OC content ($p < 0.0001$). However, this linearly

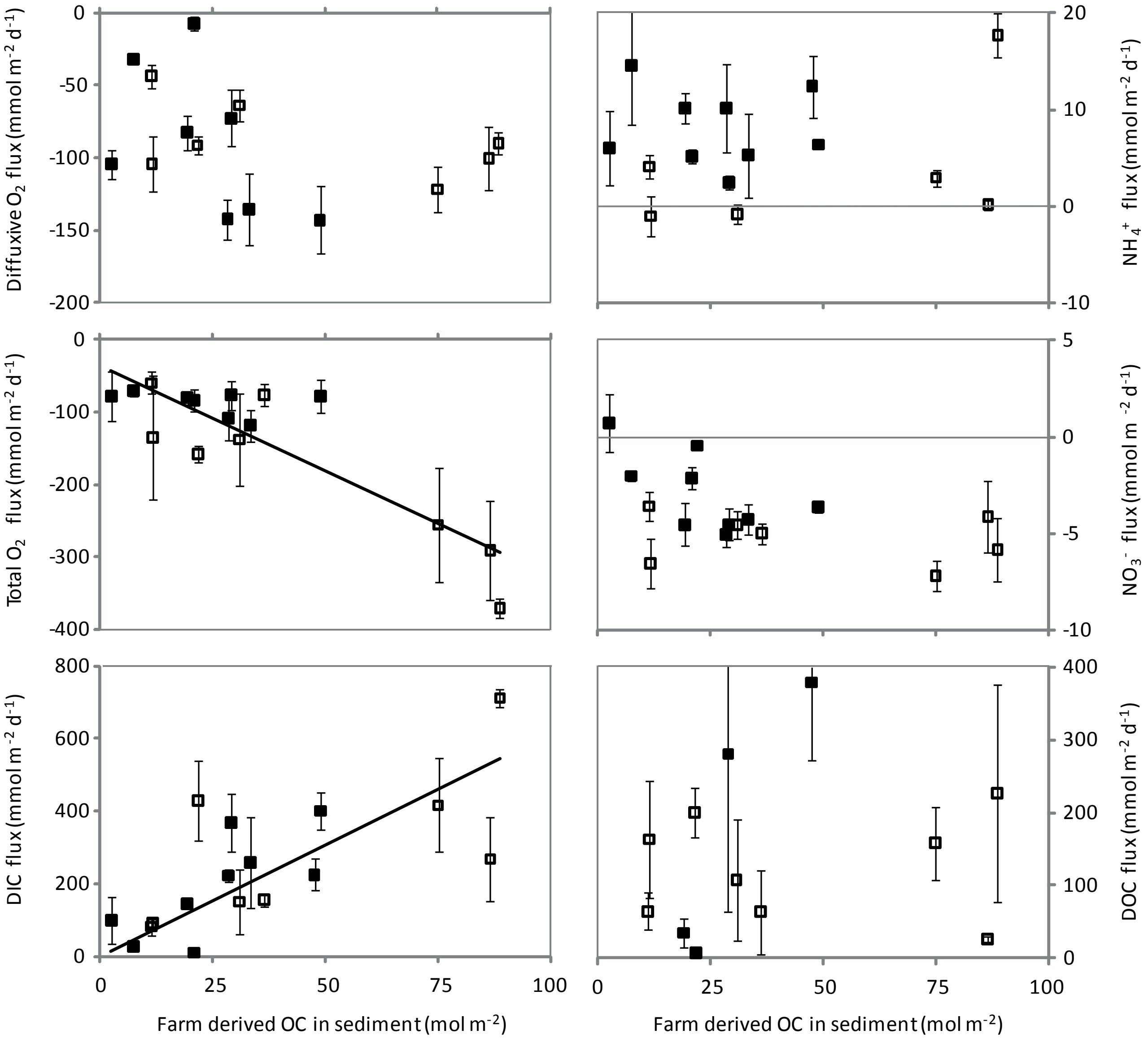

Figure 7. Fish farm elevated sediment- water fluxes in relation to the amount of farm derived organic carbon in the sediment at F1(solid squares) and F2(open squares). Error bars indicate the standard error, n= 9 in DOU and 3 in the other measurements. The presented oxygen fluxes were incubated with 100% air saturated water.

increase was dependent on the three measuring points with highest OC content in the sediment (Fig. 7b).

The DIC effluxes did not differ between the two farming sites (Fig. 7c), and showed a linear increase with increasing sediment OC content ($p < 0.001$). In average the DIC efflux was 1.8 ± 0.4 (SE) times higher than the total oxygen uptake at same level of OC content in the sediment. The NH_4^+ sediment water exchange rates seemed to differ between the two farms, with lower effluxes at F2 than F1 at the same organic loads (Fig. 7d), and at F2 the ammonium fluxes were directed into the sediment in some of the samples, while this was never the case at F1. However, the nitrate flux did not differ between the two sites, showing a sediment nitrate uptake at both farming locations (Fig. 7e). As for DOU, the nitrate uptake seems to saturate at ~7 mmol m^{-2} d^{-1}. The dissolved organic carbon (DOU) sediment-water fluxes were highly variable within replicates as well as between samples, and showed no relation to the OC content in the sediment (Fig. 7 f). However, in order to get an estimate of the total carbon loss from the sediment due to microbial degradation processes the average DOC efflux as percent of the DIC efflux was used (58 ± 14%).

During the study period of 4.6 and 7.4 months at the respective farms F1 and F2, the OC content in the sediment at the edge of the net cages was measured 5 times at each farm. From the changes in OC content over time (Fig. 6) and the relation between DIC efflux and OC content the total carbon mineralization of organic material (including DOC loss) in the footprint areas were estimated to be 2.1 tonnes C at F1 and 61.2 tonnes C at F2. Thus the mineralization and accumulation rates of organic carbon in the sediment were of similar magnitude.

Discussion

Sedimentation. The estimated distribution on sinking farming derived particles, demonstrate the difficulty in extrapolating trap data from point sources to larger areas, and that this should be done with great caution, preferentially deploying the traps directly below the cage (Fig. 3). In addition, resuspension might confound trap data close to the seabed (Pejrup et al. 1996).

The relative proportion of feed that sediment below fish farms is highly variable but typically range between 29 and 71% (Hall et al. 1990, Holmer et al. 2002, Kempf et al. 2002). Our estimates are somewhat lower reaching 10.0 ± 2.1% but the estimate predicts steep and poorly resolved gradients (Fig. 3), and should thus be treated with caution.

The vertical OC fluxes increased with water depth at both farms. However, the increase at F1 was smaller than predicted by the estimated particle distribution, especially at the measurement conducted 10 m away from the farm (Fig. 3). Never-the-less, the small increase in OC fluxes with depth demonstrate that the resuspension at F1 was small. At F2 the fluxes on the other hand increased substantially with depth, and in average 64 ± 7 % of the POC flux at 40 m depth was resuspended material, while almost the entire vertical POC flux at 50 m depth was resuspended

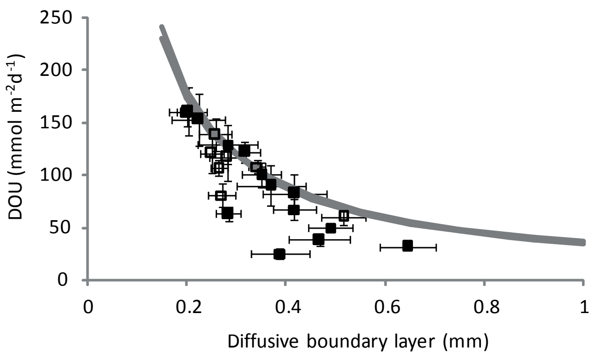

Figure 8. Relation between the sediment diffusive oxygen uptake (DOU) and the diffusive boundary layer (DBL) thickness at the fish farms F1 (Solid squares) and F2 (open squares). Horizontal and vertical error bars represent the SE of the DBL thickness and DOU respectively (n=9) Grey lines indicate the maximum DOU at the respective DBL thickness at 7-11°C when the bottom water is 100% air saturated with oxygen.

material. Although the depth integrated mean current speeds at the two farming sites were quite similar, 3.5 and 5.2 cm s^{-1} at F1 and F2, respectively, the near seabed currents differed significantly. At F2 the current speed 3 m above the seabed exceeded 10 cm s^{-1} for 15% of the time, while such near bed current speeds were hardly observed at F1. These events with stronger currents at F2 apparently have large influences on sediment resuspension. According to Cromey et al. (2002b) and Yokoyama et al. (2006), the threshold for sediment resuspension below fish farms is in the range of 10 cm s^{-1} (Cromey et al. 2002b, Yokoyama et al. 2006), which agrees well with our observations of regularly resuspension events at F2.

Sediment water fluxes in relation to organic carbon content. The two fish farms in Kaldbaksfjørður were highly different in many aspects, such as total farming activity, farm size, hydrodynamics, and benthic conditions prior to the onset of farming. However, the relation between fish farm derived organic carbon and sediment water exchange rates at the two sites were quite similar in the majority of the measured parameters (Fig. 7).

The finding that the diffusive oxygen uptake reached a maximum (Fig. 7a) is not surprising, since the sediment was covered by *Beggiatoa* and the sediment-water interface practically was anoxic at the majority of the samplings (data not shown). The diffusive oxygen uptake was close to the theoretical maximum in most measurements (Fig. 8), which also could be expected at the given sediment conditions (Jørgensen & Revsbech 1983, Jørgensen & Boudreau 2001). In coastal areas the average *in situ* DBL thickness is around 400µm (Glud 2008), which is close to the average DBL thickness of 344 ± 25 µm in our incubations.

TOU in sediments below fish farms ranges between 50 and 500 mmol m^{-2} d^{-1} (Hall et al. 1990, Hargrave et al. 1993, Findlay & Watling 1997, Christensen et al. 2000, Holmer et al. 2002, 2003, Kempf et al. 2002, Nickell et al. 2003, Alongi et al. 2009) which more than covers the TOU range of the farming affected sediment in Kaldbaksfjørður (Fig. 7b). Virtually all studies measure increased sediment organic carbon content and increased TOU in sediments below fish farms relative to unaffected sediments. However, the connection between TOU and OC content in the sediment below fish farms is not so unanimously. E. g. Nickell et al (2003) found increasing TOU with increasing OC, while the opposite relation between OC sedimentation rates and sediment TOU was found by

Holmer et al. (2003). In our study we found increasing TOU with increasing farming derived OC in the sediment, however this relation was highly dependent on the 3 samples from F2 with highest OC content.

The DIC efflux increased linearly with increasing organic carbon content in the sediment (Fig. 7c). This is a common observation, since mineralization rates in sediments generally are dependent upon the quality and quantity of organic matter (Fenchel et al. 2000). However, some studies have found metabolic thresholds for sediments enriched with organic carbon (Holmer et al. 2003, Valdemarsen et al. 2009). In our study the maximum increase in DIC efflux due to farming activity (711 ± 23 mmol m^{-2} d^{-1}) is at the higher end of reported DIC effluxes from farming affected sediments (Hall et al. 1990, Holmer & Kristensen 1992, Holmer et al. 2002, 2003). The 1.8 ± 0.4 (SE) times higher DIC efflux than total oxygen uptake in the sediment demonstrates that TOU is a poor indicator for total mineralization rates in fish farming sediments, due to the continuous build up of reduced metabolites in farming affected sediments (Glud 2008). However since the DOC efflux was substantial (although variable), our results also indicate that the DIC efflux alone does not necessary describe the total organic carbon loss from fish farm sediments due to diagenetic activity. Moreover methane bubbles were observed in the sediment. These were site specific, occurring at significant lesser amounts of fish farm derived organic carbon at F1 (~0.6 kg C m^{-2}) than at F2 (~1 kg C m^{-2}). Out-gassing of methane can be a significant contributor to the carbon efflux from sediments below fish farms (Hall et al. 1990).

The nitrate fluxes were directed into the sediment, and seemed to reach a maximum (Fig. 7e) probably due to limitation in nitrate transport from the water column which contained 3.4 to 11.5 µM nitrate, depending on season. Sediment nitrate uptake is general in sediments below fish farms (Hall et al. 1992, Christensen et al. 2000, Holmer et al. 2002). The ammonium effluxes were elevated relative to the reference conditions in the majority of the incubations (Fig. 7d) which is common in farming affected sediments (Hall et al. 1992, Holmer & Kristensen 1992, Christensen et al. 2000, Holmer et al. 2002, Nickell et al. 2003). However, the fluxes at F2 were much smaller than at F1, and the sediment even took up ammonium in some of the samples. Similar observations have been made in farming affected sediments with naturally low organic carbon and nitrogen content (Heilskov et al. 2006) as is the case at F2 in our study (Table 1), and the low ammonium fluxes were attributed to increased bacterial NH_4^+ incorporation. The site specific ammonium fluxes demonstrate the difficulty of evaluating the amount of nutrient release from the sediments, and their contribution to primary production, and our results demonstrate that even within the same fjord the nutrient release from the sediment can be significantly reduced by farming at sites with coarse sediments and low organic carbon and nitrogen content.

Estimate on the carbon accumulation in footprint area. At the inner farming location (F1) the relation between the spatial measurements of organic carbon and the estimate was significant (Fig. 5). This implies that this simple distribution estimate combined with a few sediment samples is a useful tool to evaluate the accumulation of fish farming derived organic carbon in the footprint area at F1. The estimate revealed that the total accumulation of organic carbon amounted to 2.7% of the organic carbon input with food.

The agreement between estimates and observations is, however, highly dependent on hydrodynamic conditions. At F2 the estimate did not correlate to the measurements of fish farm derived organic carbon (Fig. 5). This is not surprising, given the observed resuspension at this site since the estimate does not include relocation of material after reaching the seabed. The sediment samples at F2 however show considerable high amounts of fish farm waste in the footprint area, and thus the resuspension events were not strong enough or frequent enough to "wash out" the farming residuals. In fact the accumulation at the measured sites in F2 was higher than at F1 even when considering the different amounts of food usage. Near seabed current speeds at F2 exceeding 10 cm s^{-1} were consistently in a SW direction, and since the sediment samples were taken at the SW part of the farm, it is likely that some of the measured organic material originated from the sediment below the cages to the east and north of the sediment measurements. Quantifying the total amount of organic carbon in the footprint area at F2 by means of the estimate and measured sediment samples thus is subjected to substantial inaccuracies.

Thus for farming sites with low current speed the simple dispersion estimate seems to be a useful tool as guideline for environmental influences and limitations, when combined with a limited number of field measurements. The study furthermore illustrates quite strong gradients in sedimentation and the sediment footprint underneath the cages at such sites. This underlines the importance of location for representative field samplings. At farming sites where currents are strong enough to cause resuspension and relocation of the farming derived particles (> ~10 cm s^{-1}) (Cromey et al. 2002b, Yokoyama et al. 2006), the presented estimates alone are not sufficient for environmental assessment.

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