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Organic enrichment at salmon farms in the Bay of Fundy, Canada: DEPOMOD predictions versus observed sediment sulfide concentrations

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ABSTRACT: A model for predicting benthic impacts of fish farms (DEPOMOD) was used to predict organic carbon deposition rates at 6 salmon farms in the southwestern New Brunswick (SWNB) area of the Bay of Fundy, Canada. Model predictions of the seafloor area with elevated deposition rates were compared to the areas of seafloor with elevated observed sulfide concentrations. DEPOMOD predictions with resuspension appeared to overestimate the rate of resuspension of waste particles where current speeds were moderate to high; therefore, model runs without resuspension were used for comparisons. There were no consistent relationships between current speeds and the predicted (without resuspension) area with elevated deposition rates and the areas with elevated sulfide concentrations. There was a positive relationship between the areas with elevated deposition rates and the areas with elevated sulfide concentrations at 3 sites, with a better fit when the DEPOMOD runs used average daily feeding rates during 1 mo periods including the date of sediment sampling (compared to average feeding rates during 3 mo feeding periods ending near the date of sampling). Because the predicted area with elevated deposition rates (without resuspension) was strongly correlated with the feeding rate, it is important that the appropriate feeding rate be used in model runs. At sediment sampling stations where predicted deposition rates were low, sulfide concentrations were usually low; however, at sampling stations where predicted deposition rates were elevated, sulfide concentrations showed high variability. Implications for the use of DEPOMOD for management of the salmon aquaculture industry in SWNB are discussed.

KEY WORDS: Model assessment · Organic enrichment · Salmon farming · Sulfide

INTRODUCTION

Salmon farming in Atlantic Canada began in 1978. The most intensive area of salmon farming within this region is in southwestern New Brunswick (SWNB) in the Bay of Fundy. In a small area, roughly 60 × 60 km, there are >90 salmon farm leases, of which about half are currently active. Salmon aquaculture production in SWNB in 2012 was 30 217 t (Statistics Canada 2013). This salmon farming area is somewhat unique, characterized by large tides, shallow water, cold winter water temperatures, and the high density of farms (Chang et al. 2014).

Current environmental monitoring practices for salmon farms are largely based on assessing benthic impacts, with the goal of protecting benthic environmental quality (Wilson et al. 2009). In SWNB, as well as in many other jurisdictions, sediment sulfide concentration is used as an indicator of benthic environmental quality in regulatory monitoring at fish farms (NBDELG 2012a,b). Relationships have been established between sulfide concentrations and benthic biodiversity based on data from various salmon farming areas (Brooks & Mahnken 2003, Hargrave et al. 2008, Hargrave 2010). Data collected at 4 salmon farms in SWNB have shown an association between

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increasing sediment sulfide concentration and decreasing benthic infaunal biodiversity, although there is considerable variability in the relationship (Wildish et al. 2001, 2004b, Chang et al. 2011a). The advantages of using geochemical monitoring (such as sulfides) are cost-effectiveness and the short time to obtain results (within one or a few days) compared to traditional benthic biodiversity analyses, which can take several weeks or months to complete (Wildish et al. 2001). The rapidity in obtaining results from geochemical monitoring means that when poor results are obtained, farms can quickly begin to implement remedial actions.

Regulators are also interested in predicting the potential impacts of new farm proposals. The most commonly used tool for this purpose is DEPOMOD, a model that was developed in Scotland to predict benthic impacts of salmon farms (Cromey et al. 2000, 2002). DEPOMOD predicts the spatial distribution of carbon deposition in the vicinity of fish farms, based on the feeding rates in each cage, current velocities measured at the site, and bathymetry. This model has also been used to study benthic impacts at salmon farms in British Columbia (BC), Canada (Chamberlain et al. 2005, Chamberlain & Stucchi 2007), and New Zealand (Keeley et al. 2013a). Chamberlain & Stucchi (2007) reported that DEPOMOD predictions showed similar patterns to field data collected near salmon farms in BC, as did Keeley et al. (2013a) in New Zealand. Recently, DEPOMOD has been used to predict benthic impacts of proposed salmon farms on the Atlantic coast of Canada (Page et al. 2009a, DFO 2011, 2013a, 2013b); however, there is a scarcity of data on the validity of the model predictions in this region.

The objectives of this study were to compare model predictions with spatially intensive measurements of organic enrichment at some marine salmon farms in SWNB. Ideally, DEPOMOD predictions of carbon deposition rates should be compared to actual measurements of the same variable. Carbon deposition rates are typically measured in the field using sediment traps. However, because of the difficulties in deploying sediment traps, most validation studies for deposition models have collected sediment trap data from only a few sampling points at each farm (e.g. Cromey et al. 2002, Stucchi et al. 2005, Brigolin et al. 2009, Jusup et al. 2009). Furthermore, Stucchi et al. (2005) noted that the accuracy of sediment trap measurements is largely unknown.

For these reasons, we decided to use sediment sulfide (S^{2-}) concentration data to estimate the actual organic enrichment under farms. Sulfide concentration is relatively easy to measure in sediments (Wildish et al. 1999, 2004a), facilitating the collection of spatially intensive data. It is also the main indicator used in the SWNB regulatory monitoring program (NBDELG 2012a,b). Exact comparisons between the model predictions and observed sulfide concentrations are not possible since 2 different variables are involved. However, there is a positive relationship between sulfide concentrations in surface sediments and organic matter sedimentation (Holmer et al. 2005, Hargrave et al. 2008, Hargrave 2010). Sediment classifications in SWNB based on sulfide concentrations, with estimated equivalent carbon deposition rates, are shown in Table 1.

In this study, we used data from 6 salmon farms in SWNB to compare the area of seafloor with elevated impacts based on DEPOMOD-predicted carbon deposition rates and the area of seafloor with elevated sulfide concentrations based on spatially intensive sediment sampling. The 6 farms represented a range of sizes, times within the production cycle, and current-velocity regimes in the SWNB area. Elevated sediment sulfide concentrations were defined as levels at which adverse environmental impacts on benthic sediments were likely occurring (Hypoxic B or worse), which is at sulfide concentrations >3000 µM

Table 1. New Brunswick Environmental Management Program (EMP) site ratings based on sediment sulfide concentration (NBDELG 2012a) and corresponding carbon deposition rates as predicted by DEPOMOD (Chamberlain & Stucchi 2007, Hargrave et al. 2008, Hargrave 2010)

EMP rating	Impact on marine sediments under farm (from EMP)	Biodiversity of macrobenthic infauna	Sediment sulfide concentration (µM)	Carbon deposition rate (DEPOMOD) (g C m^{-2} d^{-1})
Oxic A	Low	High	<750	<1.0
Oxic B	Low	Moderate	750–1500	1.0–2.0
Hypoxic A	May be causing effects	Reduced	1500–3000	2.0–5.0
Hypoxic B	Likely causing effects	Reduced	3000–4500	5.0–7.5
Hypoxic C	Causing adverse conditions	Reduced	4500–6000	7.5–10.0
Anoxic	Causing severe damage	Very low	>6000	>10.0

according to the regulatory monitoring program in SWNB (NBDELG 2012a). The carbon deposition rate approximately equivalent to this sulfide concentration is 5 g C m^{-2} d^{-1} (Hargrave et al. 2008, Hargrave 2010). Comparisons were also made between the observed sulfide concentration at each sediment sampling station and the predicted carbon deposition rate at the same stations. We also discuss some implications of the findings to management of the salmon farming industry. Some preliminary results and a technical report including additional details on the methodology for this study have been previously published (Page et al. 2007, 2009b, Chang et al. 2012).

MATERIALS AND METHODS

Study sites

The study sites were 6 salmon farms in SWNB (66° 40′ W, 44° 52′ N). This is a macrotidal area, with a maximum tidal range of 8.3 m; water circulation in this area is largely tidally driven (Trites & Garrett 1983). The mean water depths under the cage arrays at the study sites ranged from 14.6 to 22.2 m (relative to lowest normal tide), corresponding to total water depths of 18.4 to 26.0 m (at mean tide). All sites were located over fine sediments (mud or sand). Cage locations were estimated based on GPS coordinates of sediment sampling stations, aerial photos, and/or farm site plans. All 6 farms grew Atlantic salmon *Salmo salar* in net cages suspended from floating, circular, plastic collars, mostly 70 or 100 m in circumference, arranged in arrays of 2 to 4 rows with 10 to 25 m of water between adjacent cages, except at Site D where the 2 rows of cages were ~150 m apart. Sites H and I were integrated multi-trophic aquaculture (IMTA) sites, also growing mussels (at both sites) and seaweeds (at Site H).

Site A was in its 5th year of operation at the time of sediment sampling; all other farms had been operating >10 yr. At all 6 study sites, the cage array locations did not change during the 2 most recent year-classes, although there were some small changes in the numbers of cages at some sites, and at Site D, 1 cluster of smaller cages was replaced with a row of larger cages. All sites were fallowed between the 2 most recent year-classes; at Site C, the fallow period was unknown (but was ≤4 mo); at the other 5 sites, the fallow periods ranged from ~6 mo at Sites A and D to 28 mo at Site H. All 6 sites were classed as oxic in the annual regulatory monitoring (which requires sediment sulfide sampling under the edges of the cages with the highest biomasses) in the year prior to stocking of the year-class present at the time of sediment sampling or during the first year of that year-class.

In SWNB, smolts are usually stocked in marine cages in the spring (April to June) or fall (September to November) and are harvested after a grow-out period of ~2 yr. All farms in this study had stocked smolts 1 to 2 yr prior to the date of sediment sampling. The number of fish stocked per site and the number and biomass of fish at the time of sediment sampling are presented in Table 2. Feeding rates follow a strong seasonal cycle, peaking in the fall, with highest rates in the second year. At the time of sediment sampling, feeding rates were at or near the maximum for the year at Sites A, C, and H. At the other 3 sites, feeding rates had declined from earlier in the year: at Site D, the feeding rate in the month of sampling was about one-third less than the previous month, due to harvesting; at Site G, the feeding rate during the month of sediment sampling was very low, <10% of the rate 2 mo earlier, due to chemical treatments for sea lice; at Site I, where harvesting was completed 5 d prior to sampling, the average daily feeding rate during the month-long period ending at the end of harvesting was <10% of the peak rate earlier in the year (in late May to early June).

Table 2. The number of fish stocked per site and the number and biomass of fish at the time of sediment sampling. n = number of fish. Sites D and G were partially harvested at the time of sediment sampling, while Site I completed harvesting 5 d prior to sediment sampling

	Stocking		Fish at time of sediment sampling			Month of max. feeding
Site	Date	n	Date	n	Biomass (t)	in year of sampling
A	Fall 2004	484 000	22-Sep-05	461 600	480	Oct 2005
C	Spring 2005	448 000	12-Sep-06	408 500	1720	Sep 2006
D	Spring 2005	300 000	24-Jul-07	55 400	280	Jun 2007
G	Spring 2009	204 000	8-Sep-10	143 000	510	Jun 2010
H	Fall 2007	497 000	11-Sep-09	439 400	1770	Aug 2009
I	Spring 2009	555 000	27-Sep to 3-Oct-11	0	0	May 2011

Current-velocity measurement

Current meters were deployed for durations of 35 to 97 d at locations 50 to 350 m from the cage arrays at each site (see Fig. 3). Only deployments CM-A2 and CM-A3 (at Site A) were concurrent with the sediment sampling. The 2 deployments at Site H ended just 1 mo prior to sediment sampling. All other deployments were at different times of the year and/or in different years than the sediment sampling. At 5 sites (all except Site G), Teledyne RD Instruments Workhorse Sentinel Acoustic Doppler Current Profilers (ADCPs) were moored ~1 m above the seafloor at 2 or 3 locations at each farm; the ADCPs measured current speed and direction at 1 m depth intervals throughout the water column. At Site G, currents were measured using InterOcean S4 current meters, moored 6 m above the seafloor at locations on either side of the cage array in 2001 (the cage array location was different at the time of sediment sampling in 2010); the S4 meters measured current velocities only at the deployment depth. Further details on current measurements at Sites A–H can be found in Chang et al. (2012).

DEPOMOD predictions

DEPOMOD v. 2 (Cromey et al. 2000, 2002) was used to predict the organic carbon deposition rate on the seafloor in the vicinity of each farm. Model input values are shown in Table 3. The grid generation module defined the model prediction domain, which was determined by the size and number of grid cells. Grid cell dimensions were the same for the major and minor grids: 10 × 10 m at all farms. At all farms, the major grid was 99 × 99 cells, the minor grid was 98 × 98 cells, and the model domain size was 1 × 1 km. Bathymetry data (depths below lowest normal tide) were obtained from the Canadian Hydrographic Service (CHS, Dartmouth, Nova Scotia). Estimates of the water depths at the centre of each DEPOMOD grid cell were linearly interpolated from the CHS data. The centre coordinates of each cage were also entered in the grid generation module.

The particle tracking module was then used to predict where waste particles were deposited on the seafloor, assuming continuous release of food, which is the typical feeding scenario used (Cromey et al. 2000). The particle tracking module has 2 main parts: fish farm characteristics and the particle tracking model set-up. Fish farm characteristics include particle information (water content, digestibility, % wasted, carbon content, and settling velocity of feed particles; carbon content and settling velocity of feces) and cage set-up (cage diameter and depth;

Table 3. DEPOMOD input values

Parameter	Value
Grid generation module[a]	
Grid cell dimensions (major and minor grids)	10 × 10 m
Number of major grid cells	99 × 99
Number of minor grid cells	98 × 98
Particle tracking module	
Material type	Carbon
Release type	Continuous release of feed
Particle information[b]	
Feed water content	10%
Feed digestibility	90%
Feed wasted as % of feed fed (default value)	3%
Carbon as % of feed pellets (dry weight)	57%
Carbon as % of feces (dry weight)	33%
Settling velocity of feed pellets (1 particle group)	11.0 cm s^{-1}
Settling velocity of feces (1 particle group: mean ± SD)	3.2 ± 1.1 cm s^{-1}
Current velocity data[c]	
Current velocity layers	Near-surface, mid-depth, near-bottom (except 1 layer only at Site G)
Current velocity time step (default value)	3600 s (1 h)
Turbulence model (default values)	
Random walk model	Yes
Dispersion coefficient x	0.100 $m^2 s^{-1}$
Dispersion coefficient y	0.100 $m^2 s^{-1}$
Dispersion coefficient z	0.001 $m^2 s^{-1}$
Particle trajectory model (default values)	
Number of particles (for each particle type, per cage, at every time step)	10
Trajectory evaluation accuracy (model time step)	High (60 s)
Resuspension module	
Number of loops to run model for (Cromey et al. 2000)	2
Consolidation time of particles (default value)	4 d

[a]Values set by user; [b]from D. Stucchi & J. Chamberlain, Fisheries and Oceans Canada, Sidney, British Columbia, unpubl. report (2005); [c]Cromey et al. (2002)

feed input per cage). We used particle information values recommended when using DEPOMOD in BC (D. Stucchi & J. Chamberlain, Fisheries and Oceans Canada, Sidney, BC, unpubl. report, 2005; see also Chamberlain et al. 2005, Chamberlain & Stucchi 2007). Cage diameters were known, but the depths of the nets (from the water surface to the bottom of the net cage) were unknown, so a net cage depth of 10 m (which is typical for salmon cages in SWNB) was used; DEPOMOD released particles from random starting positions within each cage. Feeding rate data were provided by farm operators. The DEPOMOD user manual recommends using the feed input for the month when benthic sampling occurs (Cromey et al. 2000). We used the average daily feeding rate per cage during a 1 mo period which included the sediment sampling date (Table 4), except at Site I, which completed harvesting 5 d prior to sampling; at Site I, we used the average feeding rate during the 32 d period ending on the last week of harvesting. We also ran the model using average daily feeding rates during 3 mo periods ending near the sediment sampling dates (Table 4). Feeding rates varied widely among cages at Sites A, D, and I but were relatively similar among cages at the other sites.

The particle tracking module set-up includes current-velocity data input, a turbulence model, and a particle trajectory model. Hourly water current-velocity records were extracted for 3 depth layers (near-surface, mid-depth, and near-bottom), except at Site G, where data from only 1 depth (6 m above the seafloor) were available. The near-surface layer was a constant depth below the water surface (2.5 to 3.5 m, depending on the deployment), the near-bottom layer was a constant distance above the seafloor (3.1 to 4.7 m), and the mid-depth layer was a constant distance above the seafloor, approximately mid-way between the near-surface and near-bottom layers. For the turbulence model, the random walk model was selected, with default values for the dispersion coefficients. For the particle trajectory model, default values were used. DEPOMOD assumes a temporally constant water depth, based on the mean tidal height. The cage set-up allows a maximum of 30 cages per run; therefore, at Site H, which had 33 cages (in 3 rows), DEPOMOD was run twice (using the same model domain and grid), first with 2 rows of cages, and then only with the third row; the deposition estimates per grid cell from the 2 model runs were then summed.

The resuspension module was then run to calculate the carbon deposition rate at the centre of each grid cell. This module was run without resuspension activated and then with resuspension. The resuspension parameters are invariable: the critical shear stress for resuspension is 0.0179 N m^{-2} (~9.5 cm s^{-1} near bottom current speed), the critical shear stress for deposition is 0.004 N m^{-2} (~4.5 cm s^{-1} near bottom current speed), and the erodibility constant is 7×10^{-7} kg m^{-2}

Table 4. Feeding rates during 1 mo and 3 mo periods used in DEPOMOD runs at 6 salmon farms. 1 mo periods: feeding rate periods included the sediment sampling dates, except at Site I which completed harvesting 5 d prior to sampling. 3 mo periods: sediment sampling date was near the end of the 3 mo period

Site	Cages Circum. (m)	n	Sediment sampling date	Feeding rates per cage (kg d^{-1}) Time period	Min.	Max.	Mean	Total feeding rate (kg d^{-1})
1 mo period								
A	100	13	22-Sep-05	Sep-05	203	1 045	474	6 644
	50	1						
C	100	15	12-Sep-06	Sep-06	850	1 012	948	14 218
D	100	10	24-Jul-07	24-Jun to 24-Jul-07	0	939	493	4 927
G	70	15	8-Sep-10	8-Aug to 11-Sep-10	8	28	19	283
H	70	33	11 to 15-Sep-09	16-Aug to 12-Sep-09	373	506	439	14 485
I	100	17	27-Sep to 3-Oct-11	21-Aug to 24-Sep-11	0	301	54	915
3 mo period								
A	100	13	22-Sep-05	Jul to Sep-05	151	845	354	4 962
	50	1						
C	100	15	12-Sep-06	Jul to Sep-06	728	885	831	12 462
D	100	10	24-Jul-07	May to Jul-07	228	695	524	5 240
G	70	15	8-Sep-10	13-Jun to 11-Sep-10	55	159	132	1 971
H	70	33	11 to 15-Sep-09	14-Jun to 12-Sep-09	368	416	391	12 917
I	100	17	27-Sep to 3-Oct-11	Jul to Sep-11	0	369	115	1 959

s^{-1}. Resuspension only affects unconsolidated particles; the default consolidation time of 4 d was used. The module was run for 2 loops of each current meter record, as recommended in the DEPOMOD manual in order to achieve a steady state solution (Cromey et al. 2000). The output selected was carbon flux per year (g C m^{-2} yr^{-1}), which was divided by 365 to calculate daily flux (g C m^{-2} d^{-1}).

Contour plots of the predicted carbon deposition rates at the centre of each grid cell were produced using MapInfo Professional (v. 8.0) and Vertical Mapper (v. 3.1.1). The interpolation technique was Rectangular, as recommended by the Vertical Mapper user guide (MapInfo Corporation 2005) when data points are evenly distributed, as in DEPOMOD outputs. Default values for Cell size and Search radius were used. The contour intervals were defined by the carbon deposition rates corresponding to the sediment classification in Table 1. Calculations of the seafloor areas within the contour intervals assumed a flat bathymetry.

Mass balance calculations compared the DEPOMOD-predicted total rate of waste production by a farm (waste feed and feces) with the predicted rate of waste deposition on the seafloor within the model domain. The total rate of waste production was calculated as the total feeding rate (all cages combined) multiplied by the rate of waste production per unit of feed. The waste production rate per unit of feed was calculated by DEPOMOD based on the input feed particle information. Using the feed particle information values in Table 3, the model estimated the waste production rate (waste feed plus feces) to be 0.044 kg C kg^{-1} feed, comprised of 34% waste feed and 66% feces. The waste deposition rate within each grid cell was calculated as the deposition rate at the centre of each grid cell (in g C m^{-2} d^{-1}) multiplied by the area of each grid cell (100 m^2). The total rate of waste deposition within the model domain was calculated as the sum of the predicted waste deposition rates in all grid cells.

Sediment sampling and sulfide measurement

Sediment sulfide data were collected at all 6 sites during summer to fall (July to October) as part of other studies or monitoring. Details on sampling at Sites A, C, and D were reported by Chang et al. (2011b) and at Site I by Chang et al. (2013). Sediment sampling at Sites G and H was conducted for regulatory Tier 2 monitoring (unpubl. data provided by New Brunswick Department of Environment and Local Government [NBDELG], Fredericton, New Brunswick). At Sites A, C, D, and I, sediment samples were collected using surface deployed Hunter-Simpson grab samplers, except some samples at Site I were collected using an Ekman grab sampler. Samples were taken under the edges of some cages, between some cages, and up to 100 m or more outside the cage array (see Fig. 3); the number of sample stations ranged from 33 (Site C) to 168 (Site I). Grab samples were ~0.096 m^2 at Site A, 0.024 m^2 at Sites C and D, and 0.024 to 0.052 m^2 at Site I. One grab sample was taken per station. Sulfide concentrations (total S^{2-}, in µM) were measured in triplicate 5 ml subsamples taken from the top 2 cm of the sediment surface of each grab sample. At Sites G and H, sediment core samples were collected following Tier 2 monitoring protocols (NBDELG 2012b). Triplicate cores (~20 cm long × 5 cm diameter) were collected by divers at stations under, but not outside of, the cage arrays: 4 stations under the edges of each corner cage, under the outer edges of all other perimeter cages, and at stations approximately mid-way between all cages; there were 46 stations at Site G and 88 at Site H. The sulfide concentration was measured in one 5 ml subsample taken from the top 2 cm of sediment of each core sample. At all 6 sites, sediment sulfide concentrations (µM) were measured using Orion 9616BN silver/sulfide electrodes connected to Accumet meters, following the method described by Wildish et al. (1999, 2004a) and NBDELG (2012b). All samples were stored in ice and analyzed on the day of sampling or the next day, except at Site A, where half of the samples were analyzed on the day after sampling and the rest on the second day after sampling; a comparison of samples analyzed on the 2 days at this site (at least 1 subsample from each station was analyzed each day) indicated no systematic differences (Page et al. 2007). Contour plots of the sediment sulfide concentrations (means of triplicate subsamples at each sampling station) were produced using MapInfo software. Sampling station locations were based on GPS coordinates. The interpolation technique was Natural Neighbor (Simple), based on recommendations in the software user guide (MapInfo Corporation 2005). Default values were used for Cell size and Aggregation distance. The Surface Solution Type used was smoothed, without overshoot (the default). The contour intervals were defined by the sulfide concentrations corresponding to the sediment classification in Table 1. Calculations of the seafloor areas within the contour intervals assumed a flat bathymetry.

Comparison of DEPOMOD predictions and sediment sulfide data

For comparisons between DEPOMOD predictions of carbon deposition and observed sediment sulfide concentrations, we used the nomogram for benthic organic enrichment zonation in Hargrave et al. (2008; see Table 1). For each site, the area of seafloor with elevated impacts based on sulfide concentration data was compared to the area of seafloor with elevated impacts based on DEPOMOD-predicted carbon deposition rates. The threshold for elevated seafloor impacts was defined as Hypoxic B conditions, i.e. DEPOMOD-predicted deposition rate > 5 g C m^{-2} d^{-1} or sulfide concentration > 3000 µM. We also compared the observed sulfide concentration at each sampling station with the DEPOMOD-predicted carbon deposition rate in the corresponding model grid cell (averages of values from model runs using 2 to 3 current meter deployments per site); these data were also compared to the relationships between organic carbon deposition (C, in g m^{-2} d^{-1}) and sediment sulfide concentration (S, in µM S^{2-}) described by Eqs. (13) and (14) in Hargrave (2010):

$$\text{Eq. 13:}\quad S = -25 + 632 \times C$$

$$\text{Eq. 14:}\quad S = 62.2 + 487 \times C$$

The sampling stations where the DEPOMOD predicted deposition rate was low (<5 g C m^{-2} d^{-1}, based on the average values from model runs using 2 to 3 current meter deployments per site) were classed according to their sulfide concentration (as in Table 1). The sampling stations where the predicted deposition rate was elevated (>5 g C m^{-2} d^{-1}) were similarly classed.

RESULTS

Current velocities

Current meter data are summarized in Table 5 and Fig. 1. Current velocities were similar among the 3 depths in most deployments. At Sites A, C, and G, the velocities among the different current meter deployments at each site were similar, while at the other sites, there were greater differences among the deployments. The percentage of near-bottom current speeds greater than the DEPOMOD resuspension threshold (>9.5 cm s^{-1}) was ≥43 % in all deployments at Sites A, C, and I and at 1 deployment each at Sites D and H, but ≤22 % in all other deployments.

DEPOMOD predictions

When DEPOMOD was run without resuspension, all model particles remained within the 1 × 1 km model domain at all sites, using both feeding rate periods. When DEPOMOD was run with resuspension, there was a negative relationship between the percentage of near-bottom current speeds above the resuspension threshold (>9.5 cm s^{-1}) and the percentage of waste particles remaining within the

Table 5. Current speeds and percentages of near bottom records >9.5 cm s^{-1} for hourly records from current meter (CM) deployments at 6 study sites (see Fig. 3 for locations). Current speed values are medians, with maxima in parentheses. At Site G, current velocities were measured at only 1 depth, 6 m above the seafloor. nd: not determined

Site	Current meter	Period (no. of days)	Near-surface	Mid-depth	Near-bottom	% near-bottom >9.5 cm s^{-1}
A	CM-A1	Jan–Apr 05 (97)	7.1 (40.2)	7.6 (32.1)	8.9 (36.2)	46
	CM-A2	Sep–Oct 05 (41)	8.2 (34.3)	8.8 (33.8)	9.4 (35.1)	50
	CM-A3	Sep–Oct 05 (41)	9.3 (41.4)	9.3 (34.6)	9.5 (31.0)	50
C	CM-C1	Aug–Oct 03 (48)	13.1 (30.0)	11.6 (30.0)	11.5 (32.3)	66
	CM-C2	Jun–Aug 09 (80)	10.1 (28.6)	11.9 (28.3)	9.4 (29.9)	49
D	CM-D1	Jun–Aug 09 (61)	4.6 (34.5)	4.5 (21.5)	5.8 (16.2)	11
	CM-D2	Sep–Nov 09 (58)	7.8 (28.2)	8.3 (26.1)	9.4 (24.5)	49
G	CM-G1	Jul–Aug 01 (37)	nd	nd	2.6 (16.5)	<1
	CM-G2	Jul–Aug 01 (35)	nd	nd	2.6 (14.5)	<1
H	CM-H1	Jun–Aug 09 (60)	6.5 (31.4)	8.0 (24.7)	8.2 (31.2)	43
	CM-H2	Jun–Aug 09 (59)	7.1 (20.9)	5.4 (19.1)	5.9 (21.1)	22
I	CM-I1	Dec 11–Jan 12 (40)	13.0 (30.9)	12.8 (30.6)	13.0 (30.7)	72
	CM-I2	Dec 11–Jan 12 (40)	7.4 (25.2)	7.4 (26.7)	8.8 (29.6)	45
	CM-I3	Dec 11–Jan 12 (40)	14.8 (53.4)	13.6 (49.3)	12.5 (47.4)	75

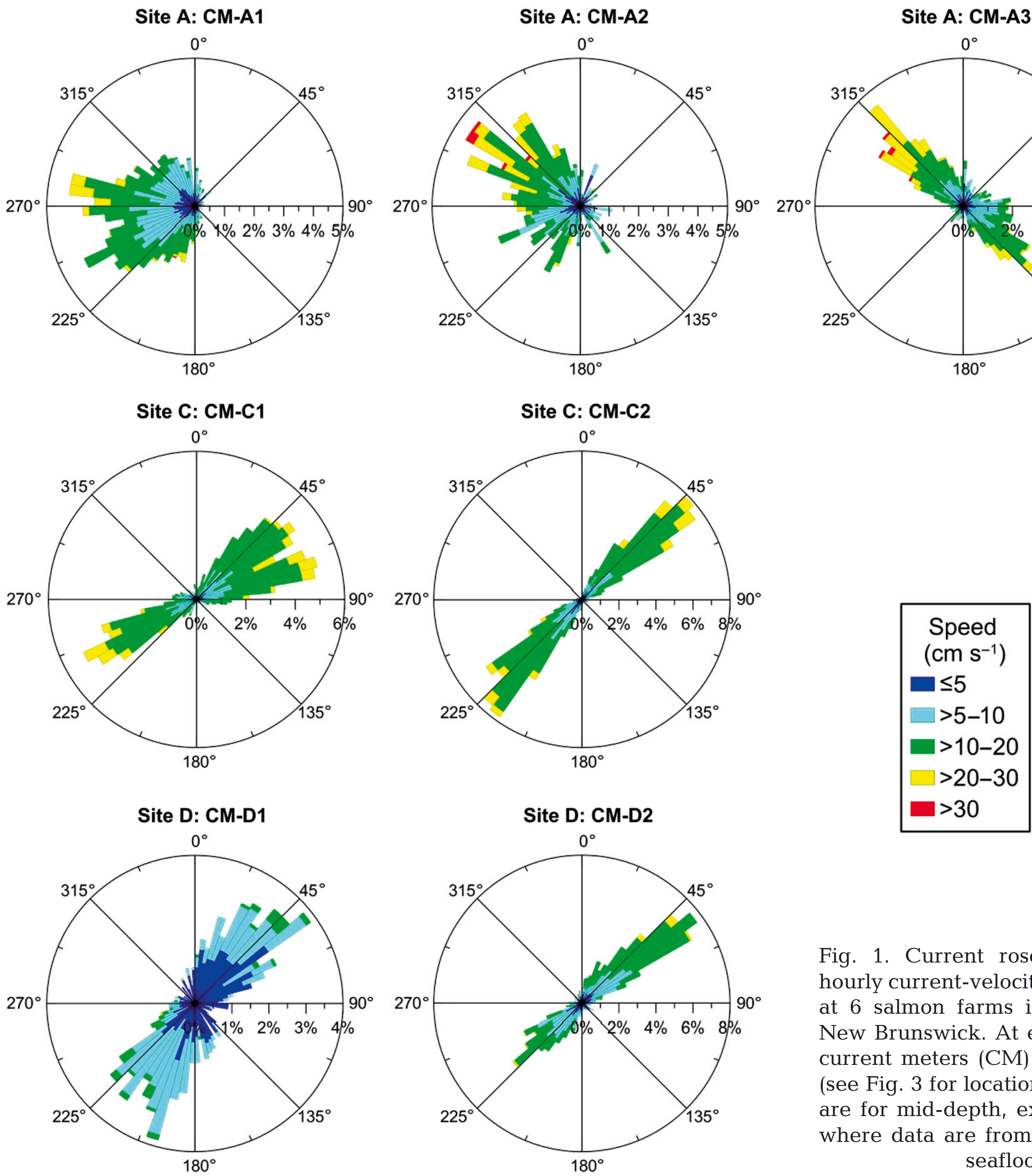


Fig. 1. Current rose diagrams for hourly current-velocity data collected at 6 salmon farms in southwestern New Brunswick. At each site, 2 to 3 current meters (CM) were deployed (see Fig. 3 for locations). Data shown are for mid-depth, except at Site G, where data are from 6 m above the seafloor

model domain. Fig. 2 shows the relationship when the model was run with resuspension using 1 mo feeding rates; the relationship was similar using 3 mo feeding rates (not shown). With both feeding rate periods, when about half or more of the near-bottom current speed records were above the resuspension threshold, <5% of waste particles remained within the model domain (with resuspension); when fewer current records were above this threshold, more particles remained in the model domain. Consequently, resuspension caused almost all particles to be transported outside the model domain at Sites A, C, and I, while at the other sites, the percent of particles remaining in the domain was higher.

Contour plots for DEPOMOD predictions without resuspension using average feeding rates during 1 mo periods at the time of sediment sampling are shown in Fig. 3. The seafloor areas with elevated predicted deposition rates were directly under the cage arrays or within 40 m of the cages. Using the settling velocities given in Table 3, feed particles would sink from the surface to the seafloor in 2.8 to 3.9 min, and feces would take 9.6 to 13.5 min (based on average sinking rates and average water depths at each site).

The predicted seafloor areas with elevated deposition rates (>5 g C m^{-2} d^{-1}) from DEPOMOD runs without resuspension and using different feeding rates (averages during 1 and 3 mo periods) are shown

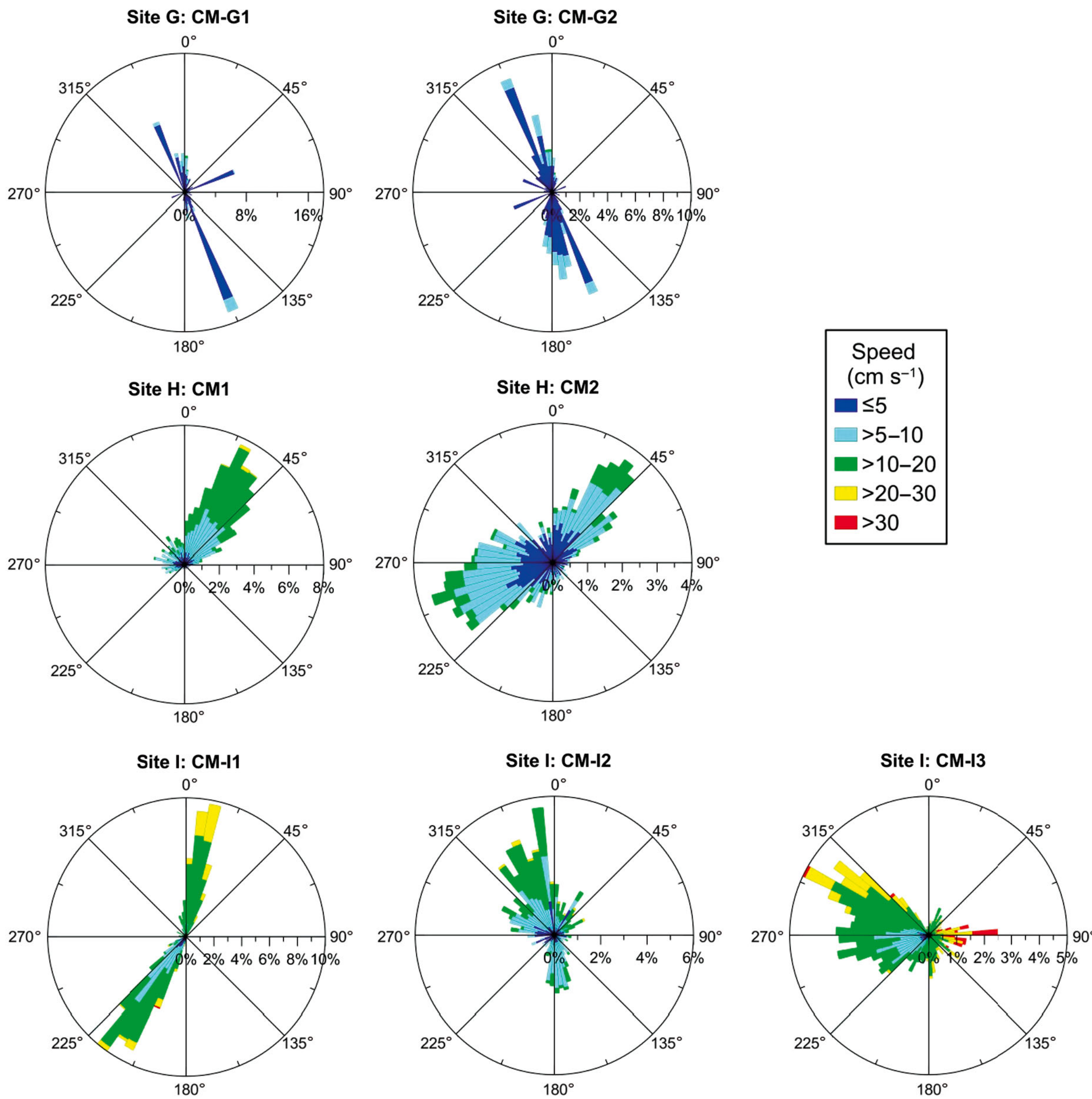


Fig. 1 (continued)

in Table 6. Using 3 mo feeding rates resulted in smaller areas with elevated deposition rates (compared to using 1 mo feeding rates) at Site A; little difference at Sites C and H; and larger areas at Sites D, I, and especially G.

Without resuspension, there were large areas with elevated predicted deposition rates at Sites A, C, D, and H using both feeding rate periods (Table 6). At Site G, there were no areas with predicted elevated deposition rates using the 1 mo feeding rate, but there were some areas with elevated deposition rates using the higher 3 mo feeding rate. At Site I, the areas with predicted elevated deposition rates were small using both feeding rates (but were larger using the 3 mo feeding rates). At Sites A, D, and I, there was considerable spatial heterogeneity in the seafloor deposition; the highest deposition rates were under cages receiving the most feed (Fig. 3). At Sites C and H, feeding rates were similar among cages, and the predicted impacts were relatively evenly spread under the cage array. At Site G using 1 mo feeding rate, the predicted deposition rates were low throughout, while using the 3 mo feeding rate, some elevated deposition rates were found under all cages, except at 1 cage that received much less feed (not shown).

Among sites, there was no clear relationship between the DEPOMOD-predicted seafloor area

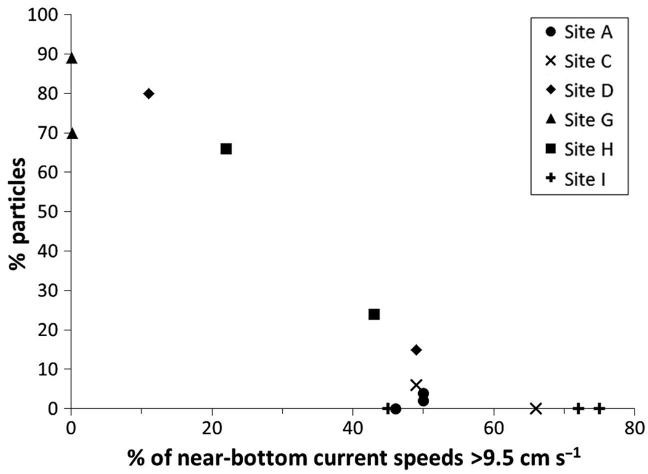


Fig. 2. Percentage of model particles remaining within the 1 × 1 km model domain when DEPOMOD was run with resuspension vs. the percentage of near-bottom current speeds exceeding the threshold for resuspension (>9.5 cm s^{-1}). When DEPOMOD was run without resuspension, all particles remained within the model domain at all sites. The results shown are for model runs using average daily feeding rates during 1 mo periods including the date of sediment sampling

with elevated deposition rates without resuspension (>5 gC m^{-2} d^{-1}) and the current speed using 1 mo feeding rates (Fig. 4a). There was a positive relationship between the DEPOMOD-predicted seafloor area with elevated deposition rates without resuspension and the feeding rate using 1 mo feeding rates (Fig. 4b). Both relationships were similar using 3 mo feeding rates (not shown); the main difference was that at Site G, there were no areas with elevated deposition rates using the 1 mo feeding rate, but using the 3 mo feeding rate, there was an area with elevated deposition rates equivalent to about half of the total cage area.

With resuspension, the areas with elevated predicted deposition rates (>5 gC m^{-2} d^{-1}) were very small (Table 6), except where current speeds were low, as at Site D (CM-D1) and Site H (CM-H2). At Site G, where current speeds were very low, there were no areas with elevated deposition rates with resuspension using the 1 mo feeding rates (because there were none without resuspension), but with 3 mo feeding rates, there were some elevated deposition rates directly under most cages (almost the same area as without resuspension). When ≥45% of the near-bottom current speed records were >9.5 cm s^{-1}, very little or no seafloor had elevated impacts (Fig. 4c); when <45% of near-bottom current speeds were above this threshold, more seafloor was impacted, except at Site G. There was no clear relationship between the DEPOMOD-predicted seafloor area with elevated deposition rates (with resuspension) and the feeding rate (Fig. 4d).

Sulfide concentrations

Sulfide concentrations showed spatial heterogeneity at all sites (Fig. 5). At Site A, some patches with elevated sulfide concentrations (>3000 µM S^{2-}) were found under or near some of the cages with higher feeding rates. At Site C, there was a small area with elevated sulfide concentrations at the shallowest corner of the site. At Site D, there were small areas with elevated sulfide concentrations under some of the cages with higher feeding rates. At Site I, there were some small areas with elevated sulfide concentrations under some cages in the shallower portion of the site. At Sites A, C, D, and I, the areas with elevated sulfide concentrations did not extend beyond 60 m from the cages. At Sites G and H, elevated sulfide concentrations were found under two-thirds or more of the cage array areas. However, at Sites G and H, sampling did not extend outside the cage arrays; therefore, the estimated areas with elevated impacts may be underestimated at these sites.

There were no clear relationships among sites between the seafloor area with elevated sediment sulfide concentrations and either the current speed or feeding rate (Fig. 6). The relationship between feeding rates per cage and the sulfide concentration in the vicinity of each cage was variable. At Sites A and D, where feeding rates varied among cages, the highest sulfide concentration areas were sometimes, but not always, under or near cages receiving higher rates of feed. At Site I, where feeding rates among cages were variable, but low, the highest sulfide concentrations were not near the cages receiving the most feed. At Sites G and H, where feeding rates were relatively even among cages, sulfide concentrations were relatively even under most or all cages, but at another site where feeding rates were high at all cages (Site C), elevated sulfide concentrations were found only under 1 corner of the cage array.

Comparison of DEPOMOD predictions and sediment sulfide data

There was a positive relationship between the DEPOMOD-predicted areas with elevated deposition rates and the areas with elevated sulfide concentrations at Sites A, H, and I in DEPOMOD runs without resuspension using 1 mo feeding rates; however,

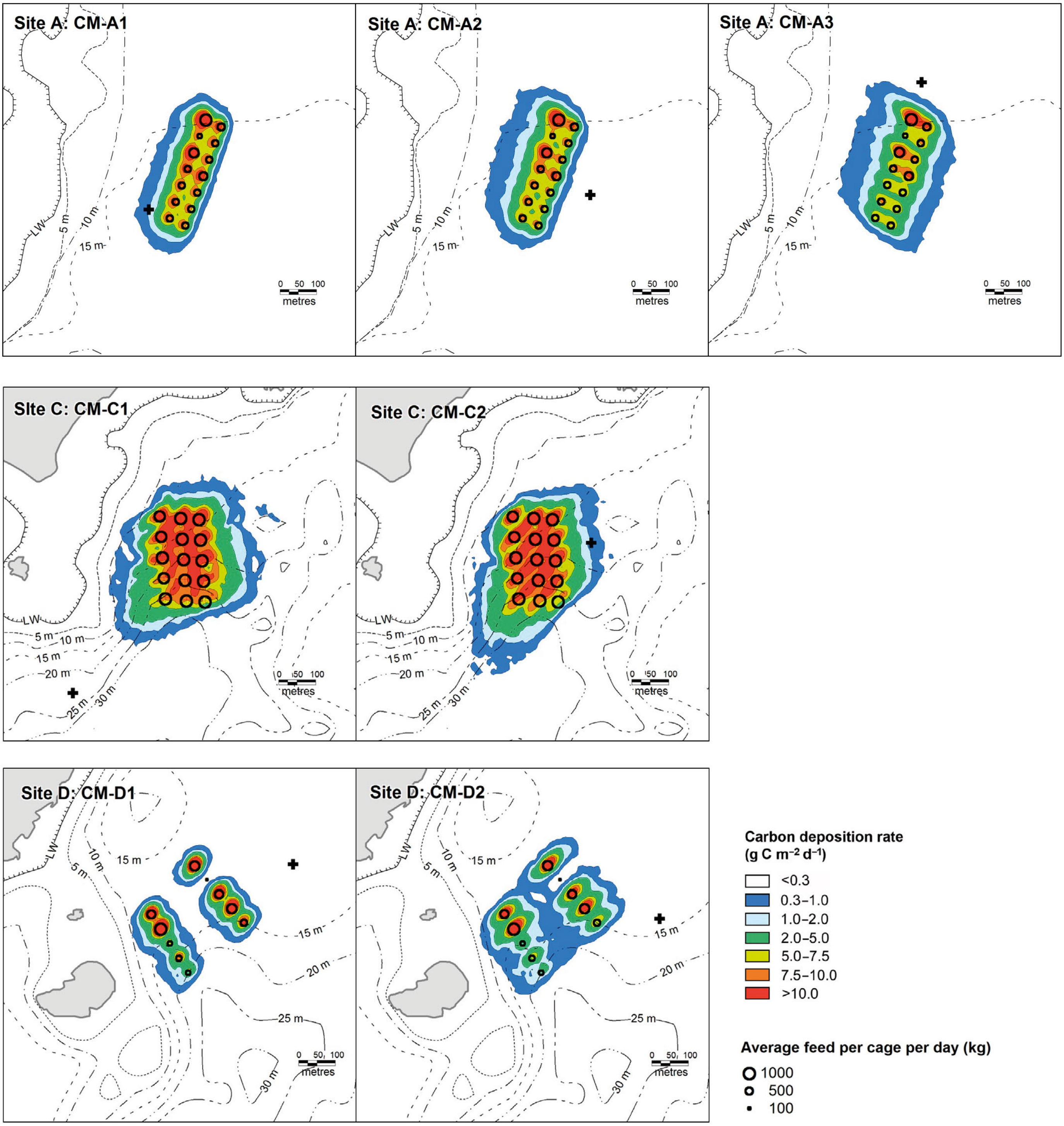


Fig. 3. Contour plots of DEPOMOD-predicted carbon deposition rates at 6 salmon farms in southwestern New Brunswick (SWNB), Canada, without resuspension. The plots show model runs using current velocity data from 2 to 3 current meter (CM) deployments at each site (+) and average daily feeding rates during 1 mo periods including the date of sediment sampling. At Site G, the current meter deployments were prior to the cage layout at the time of sampling (at the time of the deployments, the cages were between the 2 m locations). Circles indicate cage locations, with circle sizes proportional to the average daily feeding rate. Background carbon deposition rates in SWNB are reported to be <0.3 g C m^{-2} d^{-1} (Hargrave 1994). Each plot shows a 1 × 1 km area, representing the model domain. Depths are below normal lowest tide (LW)

(Fig. 3 continued on next page)

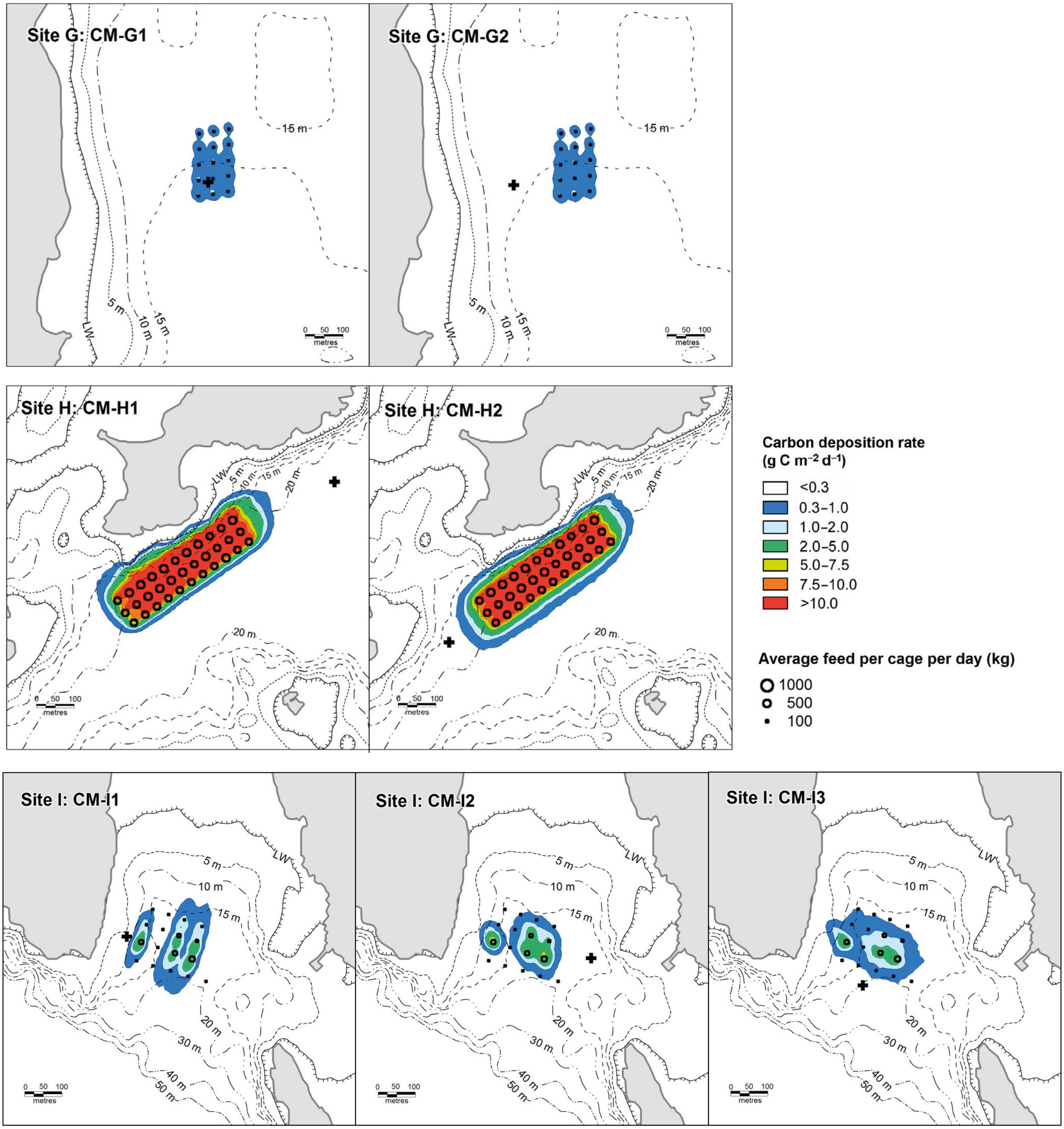


Fig. 3 (continued)

at Site G, there was no area with elevated deposition rates, but there was a large area with elevated sulfide concentrations, while at Sites C and D, the predicted areas with elevated deposition rates were much larger than the areas with elevated sulfide concentrations (Fig. 7a). The relationship was similar, but not as close, when using DEPOMOD predictions (without resuspension) with average feeding rates during 3 mo periods (Fig. 7c). With resuspension, the predicted areas of elevated impact were much smaller in most cases, and there was no clear relationship between the predicted and observed areas of impact using either feeding rate period (Fig. 7b,d).

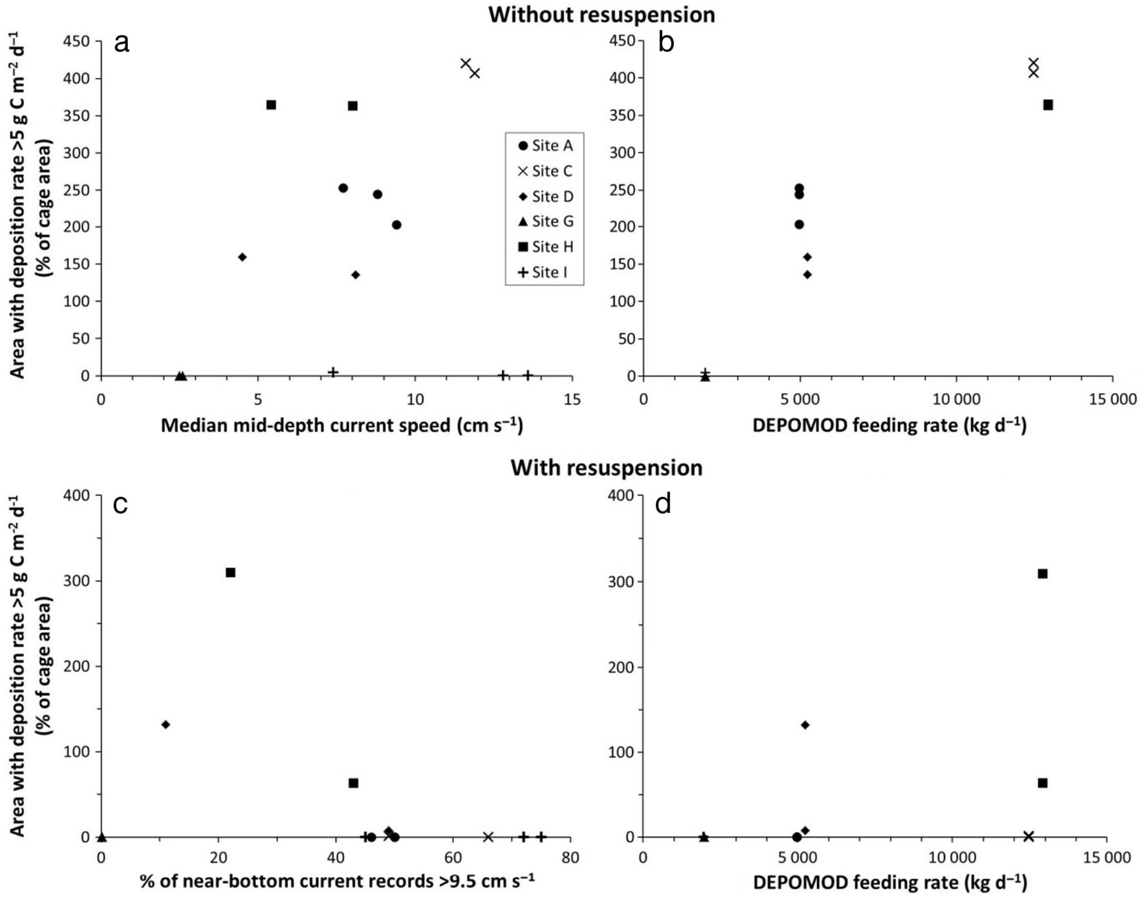


Fig. 4. Relationships between the DEPOMOD-predicted seafloor area with elevated impacts (deposition rate >5 g C m^{-2} d^{-1}) (a,b) without and (c,d) with resuspension and (a) the median mid-depth current speed, (b) the average daily feeding rate per site during 1 mo periods including the sediment sampling date, (c) the percentage of near-bottom current speeds > 9.5 cm s^{-1}, and (d) the average daily feeding rate per site during 1 mo periods including the sediment sampling date. Areas are normalized relative to the total surface area of cages at each site

Table 6. Areas with elevated carbon deposition rates (>5 g C m^{-2} d^{-1}) predicted using DEPOMOD at 6 study sites. Predictions were made without and with resuspension, using average daily feeding rates per cage during 1 and 3 mo periods ending near the date of sediment sampling

Site	Current meter	% near-bottom current speeds >9.5 cm s^{-1}	Area (m^2) with DEPOMOD deposition rate >5 gC m^{-2} d^{-1} — Without resuspension: 1 mo feeding rate	Without resuspension: 3 mo feeding rate	With resuspension: 1 mo feeding rate	With resuspension: 3 mo feeding rate
A	CM-A1	47	26700	16900	0	0
	CM-A2	50	25800	13400	0	0
	CM-A3	50	21400	9400	0	0
C	CM-C1	66	50200	46300	0	0
	CM-C2	49	48600	44900	200	0
D	CM-D1	11	12700	15000	10500	12200
	CM-D2	49	10800	11300	600	0
G	CM-G1	<1	0	2500	0	2100
	CM-G2	<1	0	2500	0	2400
H	CM-H1	43	46800	45300	8200	4300
	CM-H2	22	47000	45500	39600	37600
I	CM-I1	72	0	700	0	0
	CM-I2	45	600	1900	0	0
	CM-I3	75	0	300	0	0

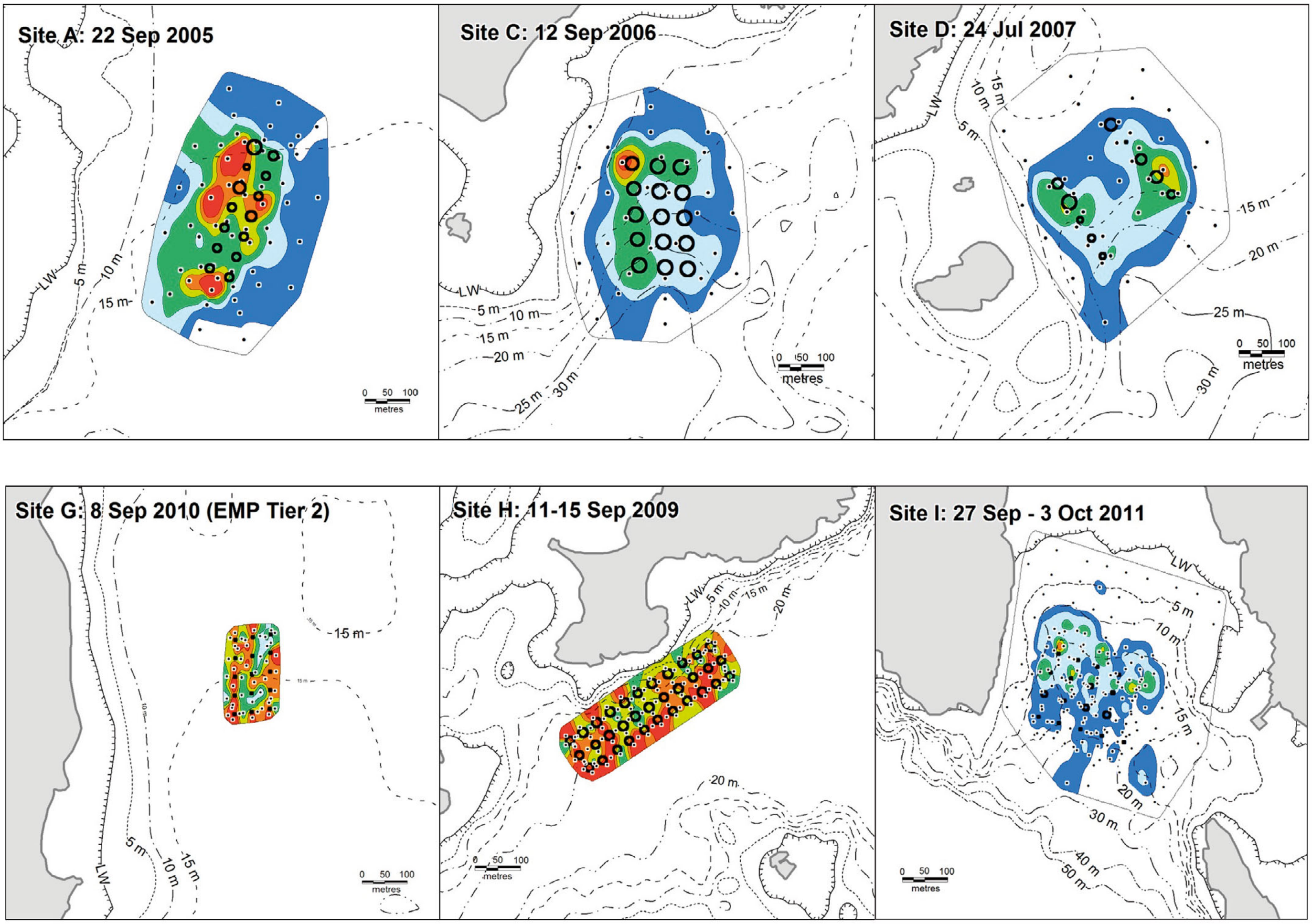
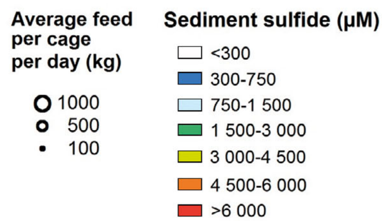


Fig. 5. Contour plots of sediment sulfide concentrations at 6 salmon farms in southwestern New Brunswick (SWNB). Black dots indicate sampling stations. Circles indicate cage locations, with circle sizes proportional to average daily feeding rates during 1 mo periods including the sediment sampling date. Background sediment sulfide concentrations in SWNB are generally <300 µM, based on data collected at reference sites (away from fish farms and other pollution sources) in SWNB (Hargrave et al. 2008) and unpublished data (provided by the New Brunswick Department of Agriculture, Aquaculture & Fisheries, St. George, New Brunswick) collected since 2000 at finfish farm sites prior to the start of operations. Each plot shows a 1 × 1 km area, representing the DEPOMOD model domain. Depths are below normal lowest tide (LW). At Sites G and H, there were no sampling stations outside the cage arrays



The relationships between the observed sediment sulfide concentrations at each sampling station and the predicted deposition rate (without resuspension, using 1 mo feeding rates) in the corresponding DEPOMOD grid cell at each site are shown in Fig. 8a. The fit of the 2 equations used by Hargrave (2010) to describe the relationship between these 2 parameters was relatively good at low predicted deposition rates (<5 g C m^{-2} d^{-1}), except for at Site G and some stations at Site A. With high predicted deposition rates (>5 g C m^{-2} d^{-1}), there was greater variability in sulfide concentrations, and the fit to the Hargrave equations was especially poor for Site C (using both feeding rate periods). When using 3 mo feeding rates for DEPOMOD, the relationship was very similar, except at Site G, the predicted deposition rates were higher but still mostly less than predicted by the Hargrave equations (Fig. 8b).

When we examined all sampling stations at the 6 sites where the DEPOMOD-predicted deposition rate without resuspension was low (<5 g C m^{-2} d^{-1}; classed as Oxic A to Hypoxic A) using 1 mo feeding rates, 85 % of these stations were also classed as Oxic A to Hypoxic A according to the sediment sulfide concentration (<3000 µM) (Fig. 9a), and when we examined all sampling stations where the DEPO-

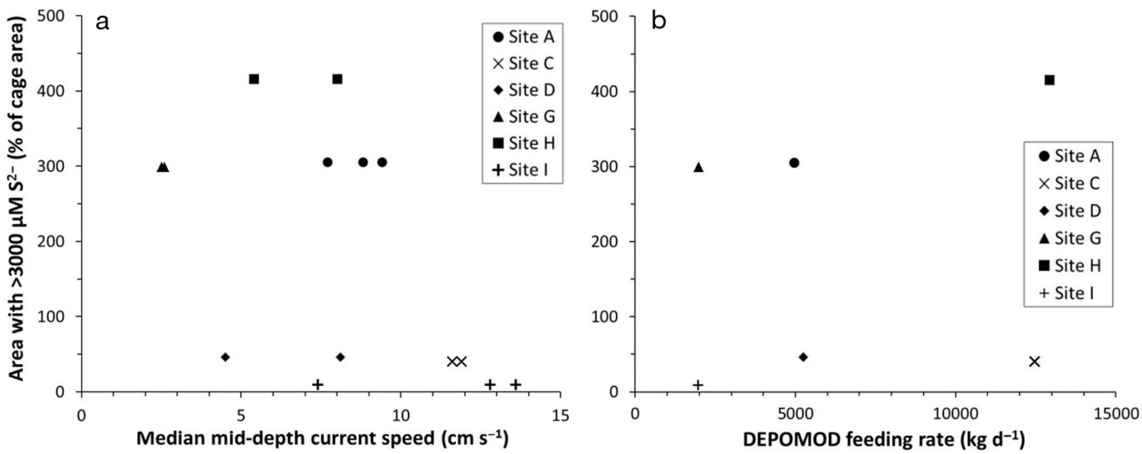


Fig. 6. Relationships between the seafloor area with elevated sulfide concentrations (>3000 µM S^{2-}) and (left) the median mid-depth current speed and (right) the average daily feeding rate per site during 1 mo periods including the sediment sampling date. Areas are normalized relative to the total surface area of cages at each site

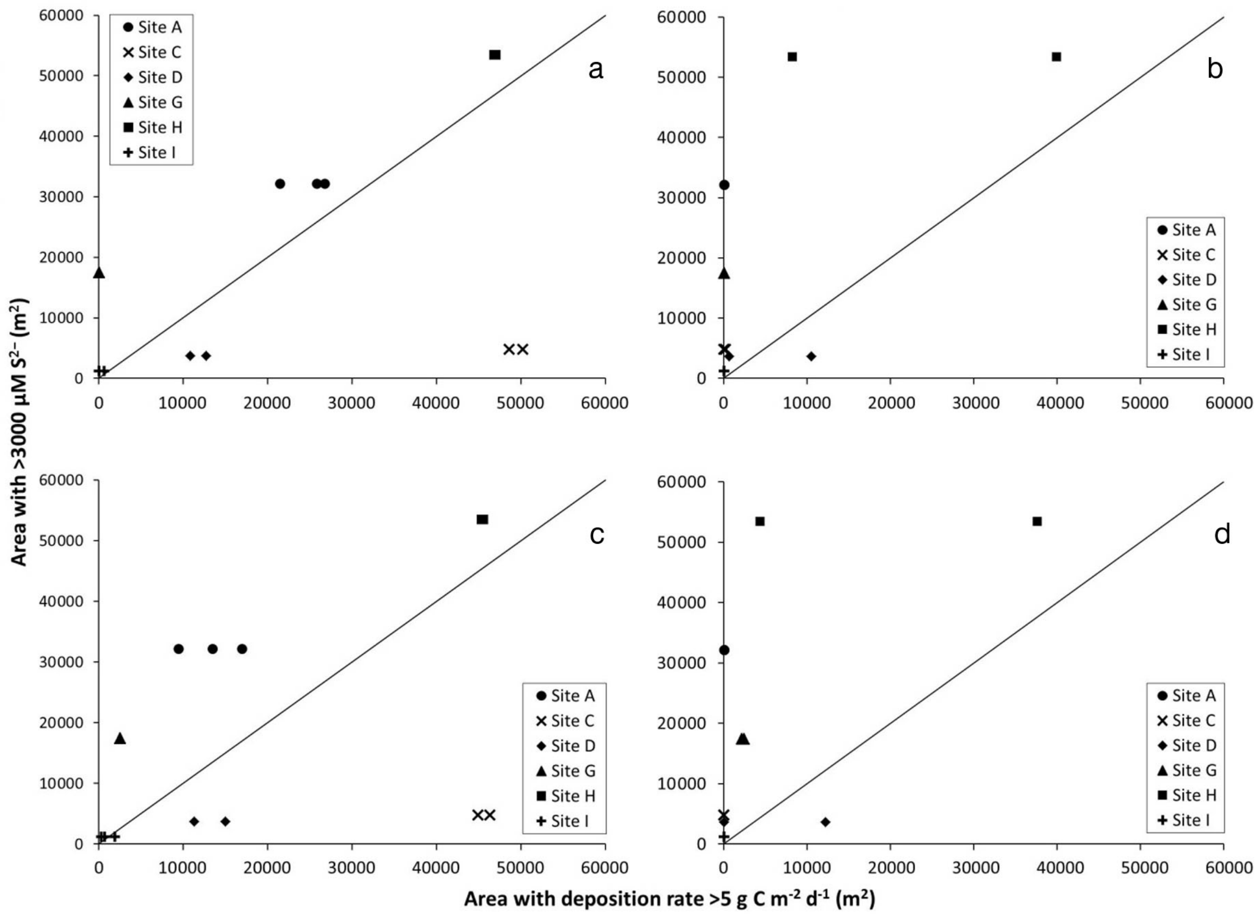


Fig. 7. Relationships between the spatial extent of seafloor area with elevated impacts predicted by DEPOMOD (deposition rate >5 gC m^{-2} d^{-1}) and the seafloor area with elevated sediment sulfide concentrations (>3000 µM). DEPOMOD runs using average daily feeding rates during (a,b) 1 mo periods and (c,d) 3 mo periods including the sediment sampling date, (a,c) without resuspension and (b,d) with resuspension. The line represents a 1:1 relationship

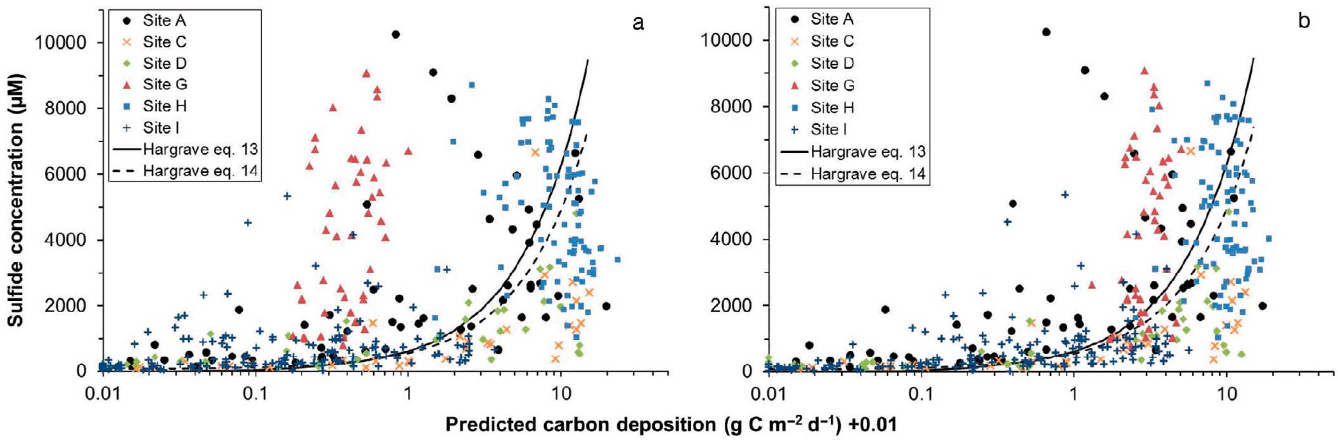


Fig. 8. Relationship between DEPOMOD-predicted carbon deposition rates (without resuspension) and observed sediment sulfide concentrations at 6 salmon farms in southwestern New Brunswick. DEPOMOD was run using average daily feeding rates during (a) 1 mo and (b) 3 mo periods ending near the date of sediment sampling. Deposition rates are for the DEPOMOD grid cell corresponding with each sediment sampling station and are averages of DEPOMOD runs using 2 to 3 current-velocity datasets per site. Sites G and H did not include any sampling stations outside the cage array. Lines represent equations for the relationship between these variables from Hargrave (2010)

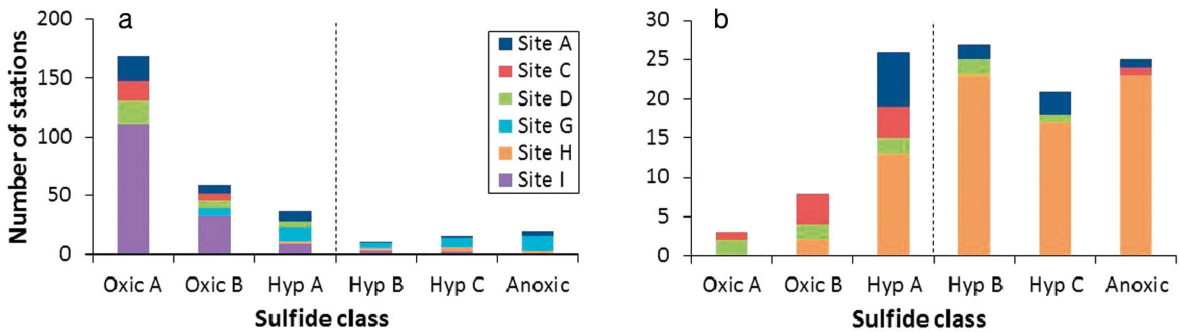


Fig. 9. Numbers of sediment sampling stations at 6 salmon farms classed according to their sulfide concentrations (see Table 1 for classification scheme). Predicted carbon deposition rates (without resuspension) used average daily feeding rates during 1 mo periods including the date of sediment sampling. (a) Stations where DEPOMOD-predicted deposition rates (without resuspension) were low (<5 g C m^{-2} d^{-1}; Oxic A to Hypoxic A according to deposition rates). (b) Stations where DEPOMOD-predicted deposition rates were elevated (>5 g C m^{-2} d^{-1}; Hypoxic B to Anoxic according to deposition rates). Deposition rate data are as in Fig. 8a

MOD predicted deposition rate without resuspension was elevated (>5 g C m^{-2} d^{-1}; classed as Hypoxic B to Anoxic), 66 % of these stations were also classed as Hypoxic B to Anoxic according to the sediment sulfide concentration (Fig. 9b). When using 3 mo feeding rates, the results were similar: 86 % of the stations classed as low impact according to DEPOMOD were also low according to sulfide concentrations, while 66 % of the stations classed as elevated according to DEPOMOD were also classed as elevated according to sulfide concentrations (graph not shown). Using both feeding rate periods, over half of the stations with low predicted deposition rates but high sulfide concentrations were from Site G.

DISCUSSION

Comparisons of DEPOMOD predictions and sulfide concentration data

The object of this project was to compare 2 methods used in SWNB (and some other jurisdictions) to evaluate the impacts of salmon farming on benthic sediments in the vicinity of farms: DEPOMOD predictions of carbon deposition rates, which are used to evaluate proposed new farms, and sulfide measurements in the seafloor sediments, which are used to monitor operating farms. A general relationship between these variables is expected: as the carbon deposition rate (due to farm wastes) increases and carbon accumulates on the seafloor, the sulfide concentration in the sediments is also expected to increase (Holmer et al. 2005, Hargrave et al. 2008, Hargrave 2010). However, a precise relationship between these variables would not be expected for several reasons, including the following: these are 2 different variables, with no direct chemical relationship between them; one variable is a concentration and the other is a rate; there is expected to be a temporal difference between the time when organic wastes are deposited on the seafloor and the time when changes in sulfide concentrations are observed, and this time lag may vary among sites and seasons due to factors such as temperature, currents, and waves; and there is error inherent in measuring both variables. While recognizing these problems, we chose to make comparisons using these variables because they are both being used by regulators and, in the case of sulfides, data were available for our study sites. The logistics of deploying sediment traps precluded the ability to collect carbon deposition data at the same spatial intensity that could be attained for sediment sulfide concentrations. Also, as mentioned in the introduction, there are questions regarding the accuracy of data collected by sediment traps (Stucchi et al. 2005, Giles 2008).

We used Hypoxic B (sulfide concentrations >3000 µM S^{2-}, approximately equivalent to carbon deposition rates >2.0 g C m^{-2} d^{-1}) as the threshold for elevated impacts. This level is defined by the regulators of the SWNB finfish aquaculture industry as the level at which farms are likely causing adverse environmental effects on sediments (NBDELG 2012a). While a number of studies indicate that biological impacts can start occurring at lower levels (Cranston 1994, Hargrave 1994, 2010, Hargrave et al. 2008, Keeley et al. 2013b), we used this level because in SWNB it is the threshold that triggers remedial actions such as additional monitoring and possibly reduction in biomass (NBDELG 2012a).

When DEPOMOD was run without resuspension, there was a positive relationship between the predicted area of impact (area with elevated carbon deposition rates) and the observed area of impact (area with elevated sulfide concentrations) at only half of the 6 study sites (A, H, and I), with the overall fit of the relationship improved by using average feeding rates during the month when sediment sampling occurred, compared to using average feeding rates during 3 mo periods. At Sites A and H, both the predicted (using DEPOMOD without resuspension) and observed (sulfide concentrations) areas of impact covered most or all of the cage array area, while at Site I, both the predicted and observed areas of impact were small. At Site I (where feeding rates were low, but varied among cages), the highest sulfide concentrations were under the shallowest cages, rather than under the cages receiving the most feed, suggesting a depth effect.

The 3 sites where the fit between predicted and observed areas of impact was poor (using DEPOMOD without resuspension) were Sites C, D, and G. At Site C, DEPOMOD predicted high deposition rates under all cages (feeding rates were high at all cages), while elevated sulfide concentrations were found under only 1 corner of the site; the high sulfide area was in the shallowest part of the cage array, suggesting that depth may have been a factor. At Site D, DEPOMOD predicted highest impacts under the cages receiving the most feed (feeding rates varied widely among the cages). The highest sulfide levels were found near some of the cages receiving the most feed, but low sulfide concentrations were found near other cages with relatively high feeding rates; overall, the predicted area of impact was much larger than the observed area of impact. At Site G, feeding rates were low at all cages, resulting in DEPOMOD predictions of no areas with elevated deposition rates (using 1 mo feeding rates) or small areas (using 3 mo feeding rates), while elevated sulfide concentrations were found under most of the cage array. The importance of feeding rates on predicted and observed sediment impacts are further discussed below.

When DEPOMOD was run with resuspension, most or all of the waste particles were transported outside the model domain, except where current speeds were low. As a result, there was no overall relationship between the model predicted (with resuspension) areas with elevated deposition rates and the areas with elevated sulfide concentrations.

A study at a BC salmon farm (Chamberlain & Stucchi 2007) found good correspondence between DEPOMOD predictions without resuspension and observed impacts. However, as in our study, when DEPOMOD was run with resuspension, most or all waste was transported outside the model domain, even though field data indicated that sediment and benthic impacts were occurring. The authors attributed this to the use of the fixed critical erosion threshold current speed of 9.5 cm s^{-1}, citing Sutherland et al. (2006) who found that current speeds of ~16 cm s^{-1} or higher were needed to transport feed pellets deposited on the seafloor. Keeley et al. (2013a) studied the effects of resuspension when using DEPOMOD at 6 salmon farms in New

Zealand. At non-dispersive (low current) sites, model predictions of the area of impact with resuspension were similar to results without resuspension (and consistent with field observations), while at dispersive (high current) sites, DEPOMOD with resuspension predicted that all wastes would be transported outside the model domain (even though impacts were observed). In the New Zealand study, dispersive sites were defined as those having >50% of near-bottom current speeds >9.5 cm s^{-1} (the DEPOMOD resuspension threshold speed).

The data from these studies suggest that a higher critical threshold speed for resuspension might provide better model predictions. However, Keeley et al. (2013a) were able to test higher resuspension threshold speeds (12 and 15 cm s^{-1}) but found that, at dispersive sites, all wastes were still transported outside the model domain. They also noted that if feed wastage rates are low, most of the waste particles will be feces, which resuspend at lower current speeds than feed pellets, suggesting that higher resuspension thresholds may not be appropriate in such cases. (At a feed wastage rate of 3%, using the waste particle information from our study, DEPOMOD estimates that 66% of wastes will be feces, and at 1% wastage, 85% will be feces.) As has been previously commented, using any single value for the resuspension threshold speed may result in inaccurate model predictions (Sutherland et al. 2006, Chamberlain & Stucchi 2007).

Chamberlain & Stucchi (2007) and Keeley et al. (2013a) suggested another possible explanation for the model results with resuspension at higher current sites: the DEPOMOD predictions with resuspension may have been accurate, i.e. there was initially a high rate of carbon deposition occurring (as indicated by the DEPOMOD predictions without resuspension), which caused the observed benthic impacts, but resuspension (due to the high currents) was continually transporting the waste away, resulting in little or no accumulation of organic waste near the farms. In New Zealand, this hypothesis was supported by field data indicating no evidence of organic matter accumulation, but reduced macrofauna, at well-flushed sites (Keeley et al. 2013a,b).

In our study, when ≥50% of the near-bottom current speeds were greater than the DEPOMOD resuspension threshold (9.5 cm s^{-1}), the model, with resuspension, predicted that most or all of the waste particles would be transported outside the model domain, resulting in very small or no areas with elevated deposition, as in the BC and New Zealand studies. However, some areas of seafloor with elevated sulfide concentrations were detected at all 6 sites, which would only be expected if some organic matter were accumulating (Holmer et al. 2005). This suggests that the resuspension module was overestimating the amount of waste transport outside the model domain in SWNB, at least where current speeds were moderate to high. Because of this overestimation, we used model runs without resuspension for our comparisons with sulfide data. However, because some resuspension is likely occurring, especially at higher current sites, omission of resuspension would be expected to negatively affect the accuracy of predictions (Giles 2008).

For DEPOMOD runs without resuspension, there was no clear effect of current speed on the predicted area of elevated impact at our study sites. There was also no clear effect of current speed on the observed area with elevated sulfide concentrations among the sites. It would be expected that higher current speeds would result in smaller areas with elevated impacts, due to greater dispersion of wastes. The lack of a clear relationship at our study sites may be related to the shallow depths and, hence, short settling times of waste particles. At our study sites, average settling times would be about 3 to 4 min for waste feed particles and 10 to 14 min for feces. As a result, the predicted horizontal displacement of waste particles was small, and most wastes were predicted to land directly under or within a few meters of cages (without resuspension). The sulfide data also indicated that elevated concentrations occurred close to cages (although data were insufficient to confirm this at Sites G and H). At Site A, where currents were predominantly to the west and northwest, the sulfide data indicated a shift in the high concentration areas in those directions (more than predicted by DEPOMOD). There was a greater effect of current speed on the predicted deposition rate when DEPOMOD was run with resuspension. However, it appears that the resuspension module overestimated the amount of transport away from farms with moderate to high current speeds (see above).

The feeding rate had a stronger effect on DEPOMOD predictions. In model runs without resuspension, there was a strong positive relationship among sites between the feeding rate and the predicted area with elevated deposition rates, and within sites, predicted deposition rates were highest under the cages receiving the most feed. However, when the model was run with resuspension, there was no clear effect of the feeding rate; as noted above, model runs with resuspension resulted in little or no area with elevated deposition rates in most cases.

There was also no clear relationship among sites between the feeding rate and the observed area with elevated sulfide concentrations. Within sites, the relationship between feeding rate per cage and the sulfide concentration in the vicinity of each cage was not consistent. At sites where feeding rates varied among cages (Sites A, D, and I), the highest sulfide concentrations were sometimes, but not always, near the cages receiving the most feed. At 3 sites where feeding rates were relatively even among cages, sulfide concentrations were relatively even under the cages at 2 sites (G and H), but not at the other site (C).

At farms where feeding rates have declined rapidly, observed sediment sulfide concentrations may reflect earlier, higher feeding rates. In 2 previous studies in SWNB, maximum sediment sulfide concentrations were found to occur after the time of maximum feeding rates (Wildish et al. 2005, Page et al. 2011). In the present study, we obtained sediment data at 2 sites (G and I) where feeding rates had declined significantly at the time of sediment sampling and were <10% of the peak feeding rates earlier in the year, resulting in DEPOMOD predicting very little or no areas with elevated deposition rates when using feeding rates near the time of sampling. However, at Site G, sediment sulfide concentrations were high: despite the low feeding rates during the month prior to sediment sampling, the sediments showed no signs of recovery. This site was also subject to Tier 2 monitoring in October 2011 (the year after the monitoring reported in the present study); despite having been fallow for 4 mo, 83% of the seafloor area under the cage array had elevated sulfide concentrations in late 2011 (NBDELG, Fredericton, New Brunswick, unpubl. data). The very low current speeds at Site G may have been a factor in the slow recovery of sediments. At Site I, where feeding rates were also low during the month prior to sediment sampling, the observed sediment sulfide concentrations were mostly low. At this site, feeding rates had been declining for a longer period, and this site also had higher current velocities, which may have aided in sediment recovery. Currents can influence benthic impacts through their role in determining the spatial distribution of settled wastes (including resuspension effects) and by supplying oxygen required for degradation of wastes (Findlay & Watling 1997).

The relationship between sediment sulfide concentrations at individual sampling stations and the DEPOMOD-predicted carbon deposition rates (without resuspension) in the corresponding grid cells showed considerable variability, especially at higher deposition rates, but generally fit the Hargrave (2010) equations, except at Sites C and G. These equations were derived mostly using BC data (see Chamberlain & Stucchi 2007), and there was considerable variability in the relationship between these parameters in those data, especially at higher predicted deposition rates (as in our study). In the BC study, DEPOMOD-predicted deposition rates (without resuspension) < 1 g C m^{-2} d^{-1} were associated with sulfide concentrations < 1300 µM; deposition rates of 1 to 5 g C m^{-2} d^{-1} were associated with sulfide concentrations of 388 to 2400 µM; and deposition rates >5 g C m^{-2} d^{-1} were associated with sulfide concentrations of 831 to 7870 µM (Chamberlain & Stucchi 2007). These values fall within the ranges found in in our SWNB study.

Sources of uncertainty

Some errors in the model predictions will occur as a result of the input values used. For feeding rates, we first used the rate during the month of sediment sampling, as recommended in the DEPOMOD user manual. This choice appears reasonable for farms where feeding rates have been relatively constant over the previous few months. However, at farms where feeding rates have changed significantly, such as Site G, determining the appropriate feeding rate may be difficult because of the time lag between carbon deposition and sulfide concentrations, which may vary between sites and may depend on several factors (such as the rate of change in feeding rates, the current velocity, and the time of year). As an alternative to the average feeding rate during the month of sediment sampling, we also ran the model using the average feeding rate during a 3 mo period ending near the time of sediment sampling. This produced slightly better predictions of the area of impact (compared to observed impacts) at Site G where feeding rates had declined greatly, but poorer predictions at Site A where feeding rates had been increasing. At Site G, the average feeding rate during the 3 mo period was still only about half of the peak daily average feeding rate earlier in the same year (in June and July). Running DEPOMOD at Site G using the peak feeding rate would produce a much larger estimate of the area with elevated deposition rates, similar in size to the area with elevated sulfide concentrations, as well as a closer match to the Hargrave equations for the relationship between predicted deposition rates and sulfide concentrations at the sampling stations (see Chang et al. 2012).

These results demonstrate that caution must be used when running DEPOMOD at sites where feeding rates have been changing; in such cases, the choice of feeding rate input can have a large effect on the predictions.

The values we used for feed particle information were recommendations for use with DEPOMOD at BC salmon farms (D. Stucchi & J. Chamberlain, Fisheries and Oceans Canada, Sidney, BC, unpubl. report, 2005). The feed wastage rate we used (3%) is also the DEPOMOD default value. Feed wastage rates of ≤5% have been reported for salmon farms in BC (Brooks & Mahnken 2003, Chamberlain & Stucchi 2007) and Scotland (Cromey et al. 2002). Recent estimates from SWNB (M. Szemerda, Cooke Aquaculture, pers. comm. 2012) and New Zealand (Cairney & Morrisey 2011) indicate that current feed wastage rates may be as low as ≤1%. Therefore, the DEPOMOD default value of 3% seems reasonable. In a preliminary study at 5 of our study sites (A to H), using DEPOMOD without resuspension and feeding rates during 1 mo periods, we found that increasing the wastage rate from 3% to 5% resulted in a 23% increase in the total waste produced and 6 to 19% increases in the areas with elevated deposition rates (except at Site G, where there were no areas with elevated deposition rates), while decreasing the wastage rate to 1% resulted in a 23% decrease in the total waste production and 11 to 25% decreases in the areas with elevated deposition rates (except at Site G) (Chang et al. 2012).

Petersen et al. (2005) and Corner et al. (2006) reported water content of 5 to 6% and carbon content of 51 to 52% in large salmon feed pellets, compared to the values of 10% and 57%, respectively, used in our study and the 9% and 49% DEPOMOD default values. However, use of these different values would have very little effect on the predicted waste output: using 5 to 6% for water content and 51 to 52% for carbon content, the total fecal waste produced per 100 kg (wet weight) of feed used would change from 2.9 kg to 3.0 kg (dry weight), while there would be no change in the amount of waste feed. We used a value of 90% for feed digestibility, which is consistent with recent estimates for farmed salmon feeds (Stucchi et al. 2005, S. C. Backman, Skretting Canada, pers. comm. 2012) but slightly higher than the DEPOMOD default value (85%). The value we used for carbon content of feces was 33%, based on findings of Chen et al. (1999); this is similar to the DEPOMOD default value (30%).

For waste settling rates, we used the values recommended for running DEPOMOD at BC salmon farms: 11.0 cm s^{-1} for feed pellets and 3.2 ± 1.1 cm s^{-1} (mean ± SD) for feces. The DEPOMOD default value for feed pellets is 9.5 cm s^{-1}. Salmon feed pellets currently used in SWNB are manufactured to have sinking rates of 10 to 11 cm s^{-1} (S. C. Backman pers. comm. 2011), which is consistent with the value we used. Published data on fecal settling rates for salmon are highly variable; a review by Reid et al. (2009) reported that measured settling rates for salmonid feces ranged from 0.7 to 9.2 cm s^{-1}. The value we used was the DEPOMOD default value (see Cromey et al. 2002). There are no data available on fecal settling rates for salmon farms in SWNB to confirm if this rate is accurate in our situations. Because of the difficulties in obtaining fecal settling data specific to each situation, most users of DEPOMOD would use default values or values from the scientific literature. However, it has been noted that models such as DEPOMOD are sensitive to fecal settling data, so using data obtained from other farm settings can introduce uncertainty (Cromey & Black 2005, Reid et al. 2009). This would be especially true if feed wastage rates are <5%, because at these wastage rates, DEPOMOD predicts that the majority of wastes will be feces.

Other potential sources of error in DEPOMOD predictions include: current velocity data, bathymetry data, cage movement, and lack of data on waves and storms. At proposed farms, current meters will be deployed in the absence of cage infrastructure, while at operating farms, current meters must be deployed a short distance from the cage array to avoid interference with farm operations. Actual current velocities within a cage array may be somewhat different, due to the effects of the cage infrastructure on currents (including the effects of mussel and kelp infrastructure at IMTA sites); it is likely that the cage infrastructure will decrease the current speed in the immediate vicinity of the farm (Sutherland et al. 2001). Each DEPOMOD run uses current velocity data from only one meter deployment. The model assumes that the currents are the same throughout the model domain; therefore, small-scale variations in currents within the model domain are not considered. Differences from actual currents will likely increase with the distance from the current meter location. Another factor is that most of our current meter deployments were not concurrent with the sediment monitoring, so seasonal and inter-annual variations in currents are potential sources of error (see Cromey & Black 2005).

Bathymetry data are generally not available at the spatial scale of DEPOMOD grid cells (10 m in our study), so interpolation from available data is usually required. At our study sites, the average distance between depth soundings ranged from 23 m at Site C to 98 m at Site G. This means that fine-scale depth variations may be missed, which could affect the accuracy of model predictions. Complex seafloor topography may prevent resuspension of wastes, even during period of high current speed (Hall-Spencer et al. 2006). However, this was probably not an important factor in our study, since all of the sites were located over relatively flat, fine sediments.

Another potential source of error is that DEPOMOD does not include horizontal and vertical movement of cages. Such movement could spread out the wastes from each cage over a larger area, thus decreasing the intensity of impacts. A preliminary study at a Scottish salmon farm, where the maximum horizontal cage movement was ~10 m over 16 h, found no significant effect of cage movement on predicted deposition rates (Cromey & Black 2005). In another Scottish study, using a GIS-based model, where maximum horizontal cage movement was 7 to 10 m, incorporating cage movement reduced the predicted feed and fecal settlement at the cage centre by 23% and 11%, respectively (compared to predictions with static cages), but there was little difference at distances >5 m from the cage edge (Corner et al. 2006). GPS data collected in SWNB (where tides are greater than in Scotland) indicate horizontal movement of cages of up to 10 to 40 m over a tidal cycle at 5 farms (Hanke 2010, S. Smedbol unpubl. data); therefore, horizontal cage movement may cause greater dispersion of wastes at some farms in SWNB.

The effects of waves and storms were not included in our model runs. Waves and storms can be more important in shallow waters, especially in exposed locations. However, data on waves and storms are more difficult to obtain and were not available for our study sites. When running DEPOMOD with resuspension, there is an option to enter wind speed and direction and fetch data, but this module has apparently not been validated for fish farms (Cromey et al. 2002), and we did not attempt to use it. Cromey et al. (2002) indicated that DEPOMOD has limited scope in shallow sites (<15 m water depth below cages) where the cages may be subject to a large degree of wind-wave resuspension. Our SWNB study sites were in relatively shallow waters (average depths 15 to 22 m), but all except Site A were in relatively sheltered locations. At a salmon farm in Maine, USA, in a similar environment to SWNB (~150 km southwest of our study area, with an average depth of 16 m below mean low tide), episodic storm-related resuspension was found to be an important factor affecting benthic impacts (Findlay et al. 1995). In other areas of Atlantic Canada, such as in Nova Scotia, many fish farms are located in shallow, exposed sites where wind-wave resuspension may dominate; at such sites, DEPOMOD predictions without resuspension must be used with caution.

There is also error inherent in the sulfide concentration data. The distance between sampling stations was typically ~25 m or more, and there were sometimes wide variations in sulfide concentrations among subsamples from the same grab sample (Chang et al. 2011b). In BC, Brooks & Mahnken (2003) reported a high degree of variability in sediment sulfide concentrations near salmon farms at a scale of ±5 m. That report also noted that subtle differences in protocols and/or techniques can result in significant differences in sulfide results. In our study, the methods used to collect and analyze sulfide concentrations in sediments at all 6 sites followed the standard operating procedures established by the government regulatory agency (NBDELG 2012b), which were based on research conducted in SWNB by Wildish et al. (1999, 2004a). At 4 of the sites (A, C, D, and I), sulfide data were obtained using mostly the same personnel (including the authors of this report), equipment, and methods. However at 2 sites (G and H), data were from regulatory Tier 2 monitoring, conducted by 2 consulting firms. Although the methods used for these 2 sites also followed the approved protocols, there were some significant differences from the methods used at the other 4 sites. At Sites G and H, sediment core samples were collected by divers (compared to surface-deployed grab samplers at the other 4 sites). Furthermore, monitoring at Sites G and H did not extend outside the perimeter of the cage arrays, so the areas with elevated sulfide concentrations may be underestimated at these 2 sites. Brown et al. (2011) have examined some factors that can affect the accuracy of sediment sulfide measurements and have suggested some ways of modifying analytical techniques to improve accuracy. Our laboratory is currently involved in research on some other factors that may affect the accuracy of sediment sulfide measurements at salmon farms.

Site history was unlikely to have had a major influence on sulfide concentrations at our study sites. All of these sites had fallowed prior to introduction of the year-class present at the time of sediment sampling, and all received oxic ratings in the annual Tier 1 monitoring immediately prior to, or during the first year of, the year-class present during sediment sampling.

CONCLUSIONS

DEPOMOD predictions (without resuspension) of the seafloor area with elevated deposition rates were similar to the area with elevated sulfide concentrations at half of our study sites, but the agreement was poor at other sites. DEPOMOD predictions without resuspension were strongly correlated with the feeding rate input, but if feeding rates have been variable, determining the appropriate rates to use in the model can be difficult; therefore, in these situations, the model must be used with caution. The correlation between DEPOMOD-predicted (without resuspension) deposition rates and observed sulfide concentrations at the sediment sampling stations showed general agreement with established relationships, except at Sites C and G, but variability was high, especially where predicted deposition rates were high. The absence of more precise relationships between predicted carbon deposition rates and sulfide concentrations is not surprising, since the comparisons involve 2 different variables.

Nevertheless, the results suggest that DEPOMOD can provide some useful information for resource managers in SWNB. The most common scenario for using DEPOMOD will likely be in predicting maximum impacts at proposed farms. In such cases, if the model runs using proposed maximum feeding rates predict low impacts, there is probably a low risk of significant impacts due to organic carbon deposition on the seafloor. On the other hand, when DEPOMOD predicts high impacts, there is some risk that poor sediment conditions may occur, but there is also a good likelihood that actual sediment conditions may not become degraded. For the resource manager, the DEPOMOD results would suggest a 'conservative' scenario; i.e. it is unlikely that actual conditions would be worse than the DEPOMOD predictions. However, from the fish farmer's viewpoint, it could mean the rejection of a site (or downsizing of the proposal) when there is a strong possibility that seafloor impacts might not occur. In such cases, other inputs must be considered in the decision-making process.

Our study suggests that in SWNB, factors other than feeding rates, current velocities (from single current meter deployments), and water depths are also important in determining where the most highly impacted sediments will occur and the overall degree of impact. Fine-scale heterogeneity in water circulation, waves (especially in shallow water or exposed sites) and large resuspension events such as storms may be important factors in determining the actual distribution of organic deposition under fish farms. In other salmon farming areas, non-tidal water currents and waves are likely to have a greater impact than in SWNB, where tidal currents predominate. At some operating farms, historical site use may influence sediment conditions, but this was probably not a major factor at our study sites.

In our study, we used DEPOMOD predictions without resuspension in our comparisons. This is because DEPOMOD with resuspension resulted in predictions that appeared to greatly underestimate the areas of elevated impacts in most cases, especially where current speeds were moderate to high. Since some amount of resuspension is likely occurring, especially at higher energy sites, an improved resuspension module may produce more accurate predictions.

The lack of consistent agreement between DEPOMOD predictions and sediment sulfide data is an indication of the risk in depending on any single indicator of benthic impacts. In their New Zealand study, Keeley et al. (2012) found that none of the individual benthic indicators they evaluated were able to consistently discriminate levels of enrichment under a range of environmental conditions. Keeley et al. (2013b) also found that the sulfide concentration was not always a reliable indicator of biological condition. The regulatory monitoring program in SWNB currently classifies operating farms based on sediment sulfide concentration. As was noted by the researchers involved in developing the SWNB sulfide monitoring program (Wildish et al. 2004a), another independent approach is required to support sulfide measurements.

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