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Measuring and modelling the dispersal of salmon farm organic waste over sandy sediments

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ABSTRACT: Fish farm waste dispersal models are widely used but have only been directly validated to a limited extent. Two shallow (<20 m) Atlantic salmon farms (Bay of Meil and Quanterness) in Orkney, Scotland were studied. Bay of Meil has peak near-bed currents of 9.7 cm s^{-1} whereas Quanterness has flows up to 31.6 cm s^{-1}. Sediment tray traps which allow resuspension to occur were deployed at each site. The patterns of particulate organic carbon (POC) deposition into the traps were in broad agreement with the observed water current directions and results from infaunal benthic monitoring. Despite the markedly different flow regimes at the 2 sites, most of the deposition occurred within 210 m of the cage perimeters. POC footprints were then modelled using the particle tracking model NewDEPOMOD. For Bay of Meil, a footprint was obtained using the recommended parameter defaults, but the spatial extent was too constrained compared to the sediment tray results. For Quanterness, all simulated particles were lost from the model domain and the critical erosion shear stress had to be increased to unrealistic levels to obtain a footprint. The failure to find a common set of parameter values applicable to both sites, despite their similar depths and sandy seabed, suggests that there remain unresolved issues, likely in how NewDEPOMOD handles waste resuspension. The sediment trays provided a direct method for quantifying the organic carbon deposition, facilitating direct validation of the dispersal model and demonstrating that further research is needed on fish farm waste dispersal at coarser sediment sites.

KEY WORDS: Benthic · Modelling · Salmon · Waste dispersal

1. INTRODUCTION

In 2020, global production of Atlantic salmon *Salmo salar* reached 2.7 million tonnes (FAO 2022) with 87% of the total coming from Norwegian, Chilean, and UK (predominantly Scottish) farms. Most of the rearing takes place in open sea cages, and this produces large amounts of faeces which, along with uneaten fish pellets, sink through the water column to the seabed. Whilst intact waste pellets may be directly consumed by predators such as fish (Uglem et al. 2020) and invertebrate scavengers (Sardenne et al. 2020), faecal material tends to be the main contributor to benthic organic enrichment. The responses of marine benthic communities to organic enrichment are well understood from foundation studies conducted in the 1970s on the impacts of sewage and pulp mill waste. Those early insights, including the iconic conceptual model of Pearson & Rosenberg (1976), have proven to be broadly applicable to the benthic impacts of open cage fish farming.

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Many benthic communities can tolerate mild to moderate organic enrichment, and this may actually lead to increases in macrofaunal production and species diversity (Hargrave 2003, Macleod et al. 2007, Keeley et al. 2013b). However, with increasing loading, biological oxygen demand will eventually exceed the system's replenishment capacity, leading to low oxygen levels (hypoxia). Under mild to moderate hypoxia, anaerobic microbial respiration becomes increasingly dominant and releases free hydrogen sulphide as a by-product. At this stage, the surficial sediment often becomes dominated by opportunistic species which can tolerate hypoxia and elevated sulphides (Pereira et al. 2004, Tomassetti & Porrello 2005, Bannister et al. 2014, Keeley et al. 2019). Under these conditions the abundance of larger macrobenthic bioturbators tends to decline, which leads to further reductions in oxygenation of the sediment (Kristensen 2000, Heilskov & Holmer 2001, Cathalot et al. 2012). Under extreme organic enrichment, the sediment can become completely anoxic with hydrogen sulphide being released across the sediment–water interface, and with mats of sulphide reducing bacteria (*Beggiatoa* sp.) developing, giving the sediment surface a characteristic white appearance (Hamoutene 2014). The Pearson & Rosenberg (1976) model appears widely applicable to organic enrichment from fish farms (Wildish et al. 2004) but the precise community responses will depend on local conditions (Brown et al. 1987, Weston 1990, Hargrave et al. 1993, Sowles et al. 1994, Cromey et al. 1998, Wilding et al. 2012, Keeley et al. 2013b). Important site-specific factors include the species being reared (Cromey et al. 2009, Weise et al. 2009), the feeding routine (Cromey et al. 2002a), seasonality and water temperature (Brown et al. 1987, Hargrave et al. 1993), the seabed's physical (Kalantzi & Karakassis 2006) and biological characteristics (Keeley et al. 2013b), and the water current regime. The latter factor is especially important as it affects both the dispersal of the organic waste and the degree to which the surficial sediments are ventilated (Findlay & Watling 1994, Cromey et al. 2002a, Keeley et al. 2013b).

In Scotland, most salmon farms are located within sea lochs along the western coasts and around the Orkney and Shetland Islands. The biomass which can be reared at each site is set through the Controlled Activities Regulations (CAR) licence issued by the Scottish Environmental Protection Agency (SEPA). A key consideration for SEPA is the area of seabed impacted by organic waste. Following the granting of a licence, the farm must collect benthic grab samples close to the time of peak salmon biomass along and orthogonal to the expected axes of maximum impact (SEPA 2019b). In addition, benthic samples are collected at a nearby reference location. The benthic samples are taxonomically analysed to derive the Infaunal Quality Index (IQI) which was developed under the European Union's Water Framework Directive and replaced the 'Infaunal Trophic Index' (ITI) of Word (1979). The IQI is a composite of taxa diversity, the AZTI Marine Biotic Index, and Simpson's evenness (Phillips et al. 2014). Samples with IQI values above 0.64 are classed as being at a 'Good' or 'High' ecological state while those below 0.44 are classed as 'Poor' or 'Bad'. SEPA also defines a mixing zone as an area of equivalent size to that within a path drawn 100 m from the cage edges, but the shape of the mixing zone is generally an ellipse determined by the dominant tidal currents at the site. At the limit of the mixing zone, the seabed ecological status as measured by the IQI must be 'Good' or 'High'.

For established sites there will be a history of benthic sampling, but for a new site this track record of observed benthic impact will be lacking. In these situations, or if a company wishes to expand an existing operation, computer simulations are used to predict the maximum permissible salmon biomass which will not fail the benthic environmental standard. This requires relating the predicted organic carbon deposition rate and footprint with the likely environmental impact, i.e. having a known relationship between the predicted carbon flux and IQI. Based on analysis of historical benthic sampling at Scottish farms, SEPA concluded that the 0.64 IQI boundary generally coincides with a predicted waste deposition rate of 250 g m^{-2} integrated over a year, where the flux is based on the daily average during the final 90 d of the computer simulation run close to peak biomass. The 250 g m^{-2} yr^{-1} threshold corresponds to a waste flux of 0.7 g m^{-2} d^{-1} and, assuming a faecal carbon content of 30% (SEPA 2019a), to 0.21 g C m^{-2} d^{-1}. As mentioned above, the basic requirement for a proposed farm (or farm extension) is that the predicted impact area at peak salmon biomass with IQI less than 0.64 should not exceed the mixing zone area. There are further requirements in terms of the modelled mean deposited mass within this boundary, which should not exceed 2000 g m^{-2} yr^{-1}. A further modification was recently issued by SEPA in relation to wave-exposed sites based on the exposure index of Burrows et al. 2008. Sites with a wave exposure index greater than 2.8 are permitted to increase the mean deposited mass within the mixing zone up to 4000 g m^{-2} yr^{-1} with an additional 20% increase in the spatial extent (SEPA unpublished data). Therefore, for licencing purposes it is important to be able to model the likely organic waste deposition rate and footprint with sufficient skill to allow the setting of biomass limits which will not lead to non-compliant benthic impacts. It is also important to be able to predict whether organic waste may be deposited onto any nearby sensitive habitats.

In Scotland, the most widely used simulator of fish farm waste dispersal is DEPOMOD. The software was originally developed by Cromey et al. (2002a) but has since been updated several times to the current version, NewDEPOMOD (SRSL 2021). Table 1 shows the evolution of the main features in the DEPOMOD family of models. In NewDEPOMOD waste is tracked as discrete particles in a Lagrangian manner. The flow field can either be based on observations from one or more current meters or derived from a separate hydrodynamic model. Once settled, material can be resuspended when the near-bed current flow exceeds an erosion threshold. Resuspended material will then be transported further afield until the current drops below a critical settlement threshold at which point the particles resettle. Layer-based bed hardening and compaction are also incorporated so that organic material can become buried over time (SRSL 2021). Most of the validation work with DEPOMOD has taken place in relatively sheltered sea lochs along the Scottish west coast where the sediments are usually mud to muddy sand. At such sites, comparison of model predictions with results from sediment traps placed close to the cages (Cromey et al. 2002a), and with tracing fluorescent particles added to the waste (Cromey et al. 2002b), showed good agreement. The tracer study suggested that newly deposited organic material below salmon cages is easily eroded and a value of 0.018 N m^{-2} was chosen for the default critical shear stress (equivalent to a near-bed current speed of about 9.5 cm s^{-1}) and a value of 7×10^{-7} kg m^{-2} s^{-1} for erodibility. However, since those studies were completed, more farms have been developed or are being proposed in energetic locations (SEPA 2019b). Several studies have noted that it has been difficult to model farm waste footprints at such sites using DEPOMOD due to apparent

Table 1. The evolution of the DEPOMOD software family

Software	Main features	Operating system	Comments	Reference
BenOSS 2	Initial waste dispersal and settlement Waste resuspension; Benthic community impacts	DOS	Model for organic totals from sewage discharges into the marine environment	Cromey et al. (1988)
DEPOMOD	Initial waste dispersal and settlement Waste resuspension; Benthic community impacts	DOS	Evolution of BenOSS2 applied to salmon farms	Cromey et al. (2002a)
CODMOD	As DEPOMOD	DOS	Re-parameterisation of DEPOMOD for cod farm	Cromey et al. (2009)
MERAMOD	As DEPOMOD	DOS	Re-paramaterised for gilthead sea-bream *Sparus aurata*, sea-bass *Dicentrarchus labrax*	Cromey et al. (2012)
Auto-DEPOMOD	More user-friendly version of DEPOMOD. Used from 2005 by SEPA for modelling discharges from Scottish salmon farms; Only used single current flow; Only flat seabed; Limited in spatial extent (1 km^2)	Windows '98 to NT	Almost all dialog input centralised in one .ini file; Automatic iteration towards solutions; Checked against SEPA method with DEPOMOD; Used commercial package Surfer® for plotting results	
MACAROMOD	Re-paramaterised for gilthead sea-bream *Sparus aurata*, sea-bass *Dicentrarchus labrax* and meagre *Argyrosomus regius*	Windows '98 to NT	Essentially MERAMOD reparameterised for Macaronesian fish farms, but also allowing a larger spatial grid	Riera et al. (2017)
New-DEPOMOD	Rewritten AutoDEPOMOD; Allows larger model domain for simulating far-field deposition; Ability to use variable 3D current model output; Ability to include variable seabed topography	Java for Windows 2000 or later or Unix	NewDEPOMOD includes more functionality but also allows plugins and easier future upgrading; Removed commercial package Surfer® for plotting results	Black et al. (2016); SRSL (2021)

over-dispersion of the simulated particles (Chamberlain et al. 2005, Chamberlain & Stucchi 2007, Chang et al. 2012, Keeley et al. 2013a, Chang et al. 2014). These problems have been addressed by either turning off the resuspension module entirely or by increasing the erosional critical shear stress to a level where resuspension is effectively turned off. Given what is known about the erodibility of waste from the heavily impacted areas beneath salmon cages (Cromey et al. 2002b), turning off resuspension is likely to result in unrealistically constrained footprints (Broch et al. 2017). The bias associated with ignoring resuspension may be acceptable to regulators because it will be conservative in the sense of overestimating waste deposition close to the cages, where benthic impacts will be most pronounced (Keeley et al. 2013a). However, this approach may mean new sites, or expansions of existing sites, are erroneously evaluated as not worth proceeding with because the biomass which would pass the environmental limits will be uneconomic. Ignoring resuspension may also mean that the area or shape of the impact footprint is incorrectly predicted with the risk that the site may fail subsequent benthic monitoring. Finally, potential accumulation of waste further from the cages will not be apparent in the model outputs which could be a significant risk for any nearby sensitive habitats, such as rocky reefs and maerl beds (Airoldi 2003, Hall-Spencer et al. 2006, Sanz-Lázaro et al. 2011).

Because of the increasing numbers of fish farms located over sandy and coarser sediments, there is a need to improve our understanding and ability to model the waste deposition and resuspension processes at such sites. The present study aimed to directly measure organic waste deposition at 2 Scottish farms located over sandy sediments but with differing current regimes and to explore whether the observations could be satisfactorily modelled using NewDEPOMOD.

2. MATERIALS AND METHODS

2.1. Study sites

Two farms in the Orkney Islands (northern Scotland), at Bay of Meil and Quanterness, were studied (Fig. 1). The Bay of Meil farm (SEPA licence CAR/L/1003888/V4) lies to the east of Kirkwall and consists of 10 circular cages with a maximum licenced biomass of 884 t. The shore is rocky but shelves gradually to a seabed at 10–17 m charted depth. The cages are in 10 to 14.5 m of water when tidal elevation is considered. For site licencing, current meter data were collected between 6 and 21 November 2009 using a Teledyne RD Instruments Acoustic Doppler Current Profiler (ADCP) deployed around 90 m southwest of the centre of the cages (58° 59.717′ N, 002° 54.022′ W). Logistical constraints in 2021 due to the SARS-Cov-19 pandemic meant that a current meter could not be deployed coincident with the present study.

The farm at Quanterness (SEPA licence CAR/L/1001931/V1) comprises eight 90 m circumference circular cages with a maximum licenced biomass of 600 t. The adjacent shore is rocky but shelves to coarse sand at around 10 m charted depth with the cages being in 10 to 14.5 m of water when tidal elevation is considered. For site licencing, current meter data were collected between 16 June and 1 July 2009 using an RDI 600 kHz Workhorse ADCP deployed slightly west of the cages (59° 00.459′ N, 02° 59.148′ W). Additional current data were collected between 29 May and 17 June 2019, using an Aanderaa SeaGuard II ADCP at 59° 00.508′ N, 002° 59.134′ W and covering the period when the present study was undertaken.

2.2. Benthic sampling transects

Weighted lines were laid by divers from the edge of the cages out to temporary moorings along the axes of expected maximum and minimum impact. These transect lines were pre-marked at set distances of 0, 10, 20, 30, 40, 50, 75, and 100 m at Bay of Meil, and at intervals of 20 m from the cage edge at Quanterness to enable accurate placement of the sediment boxes and collection of the benthic cores used for sediment particle size and particulate organic carbon (POC) analyses.

2.3. Sediment sampling by diving

Divers collected duplicate sediment cores adjacent to each sediment tray location during the first sediment tray deployment at each site. Cores were collected using 50 ml plastic syringes (Terumo, Leuven) with the tubes cut flush at the nozzle ends. Each syringe was pushed into the sediment keeping the angle as vertical as possible while steadily withdrawing the plunger. Once filled, the syringe was withdrawn from the sediment and sealed with a second plunger. Cores were returned to shore, frozen at −20°C and transported in insulated containers to the SAMS laboratory, Oban for analysis.

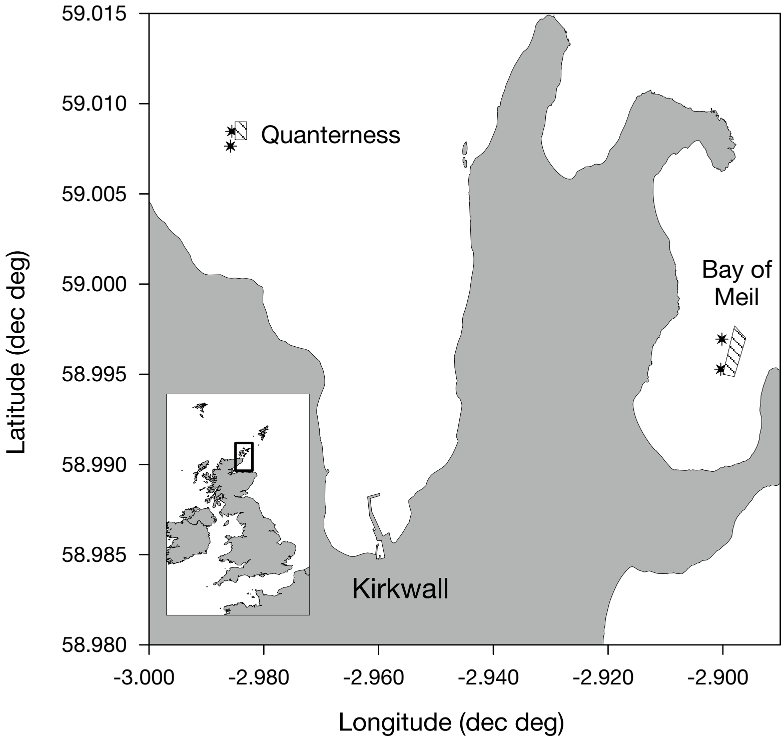

Fig. 1. Location of the study sites in the Orkney Islands, Scotland. Hatched areas: perimeters of the cages; asterisks: locations of the current meter deployments

2.3.1. Analysis of syringe cores for particle size analysis (PSA) and POC

The syringe core contents were freeze-dried (Harvest Right, scientific model) followed by sieving through a 1.18 mm screen. For PSA, sub-samples of 1 g of sediment from each syringe were placed in 50 ml centrifuge tubes, 5 ml of Calgon dispersant added, and the volume made up to 25 ml with water. Samples were then vortex mixed for 1 min and suspended sediment was introduced into an LS230 Beckman Coulter laser diffraction particle size analyser. Particle size control used standard 500 µm glass bead matrix (Coulter Control GB500/1) run at the beginning and end of each batch of sample analyses. Data were analysed using Gradistat (version 6) software (Blott & Pye 2001).

For POC analysis (Verardo et al. 1990), the remaining sieved sediment from each syringe core was ground and homogenised at 350 rpm for 3 min in a ball mill (model PM400, Retsch). A sub-sample (15–35 mg) was weighed from each core into a 2 ml glass ampoule and 1 ml of sulphurous acid added. Vials were left to degas for 8 h and then placed in a vacuum desiccator for at least 4 h. Following freeze-drying for a further 24 h, the samples were transferred into tin capsules and combusted in a Model 4010 EAS Elemental Combustion System Total Carbon and Nitrogen analyser (Costech Analytical Technologies). Quality control included calibration using acetanilide standards and running of blanks.

2.4. IQI sampling

As part of the farms' statutory environmental monitoring program, duplicate benthic samples were collected using a 0.045 m^2 Van Veen grab by farm staff. Bay of Meil was sampled on 31 Aug and 1 Sep 2021 and Quanterness was sampled on 3 and 5 July 2019. On recovery of the grab, the volume of sediment, any obvious smell of hydrogen sulphide, and the surficial appearance (colour, texture, presence of bacterial mats, feed pellets or visible faeces) were recorded. Grab contents were then passed through a 1 mm sieve and macrofauna preserved using 4% formalin. Subsequent taxonomic analyses were undertaken by environmental consultancies (Fish Vet Group unpublished data, Pharmaq Analytiq unpublished data). In the laboratory, the samples were rinsed, transferred to trays, and macrofauna removed and stored in vials containing 70% Industrial Methylated Spirit. Macrofauna were identified to the lowest taxonomic level possible using standard keys. The calculation of IQI from the taxonomic data was performed by using the standard fish farm benthic reporting framework used by SEPA.

2.5. Sediment tray traps

Because of the likely importance of waste resuspension at these sites, open trays which allowed for resuspension of surficial material (Grant 1985) were used rather than retentive sediment traps (White 1990). Plastic bakery boxes (Sistema® KLIP IT™, capacity 3.5 l, 85 × 238 × 264 mm) were filled with clean, medium-fine kiln dried marine sand (Specialist Aggregates). Because of the volumes required, it was not feasible to use sediment taken from the study sites, as in the original method (Grant 1985). At Bay of Meil, the trays were deployed on 7 July (spring tide) and 22 July 2021 (spring tide). Trays were deployed at Quanterness on 31 May (spring tide) and 10 June 2019 (neap tide). Seawater was added to each tray and the plastic lids clipped into place on board the support vessel. The boxes were lowered to the seabed and then buried by a diver along the transect lines so that the tray lips were as flush with the seabed surface as possible. The lids were then carefully removed, and the trays left in place for 7 d. At the end of this time, divers clipped the lids back *in situ* before recovering the boxes. Once ashore, the boxes were frozen at −20°C and then transported in insulated containers to the SAMS laboratory, Oban for further processing.

2.5.1. Measurement of POC in the sediment trays

Frozen sediment trays were freeze-dried over 2 drying cycles of 5–7 d, followed by a further 2–3 d. Once dried, any visible macrofauna or macroalgae on the sediment surface were removed and discarded. A salt correction was not made as the error contribution to the total sediment weight was estimated at less than 1% based on the volume of seawater in the boxes. The sediment was then gently mixed in a clean bucket to remove any lumps and the total weight recorded. To avoid over-loading the carbon detection instrument, triplicate sub-samples of between 20 and 60 g were weighed out from the dried and sieved contents of each sediment tray. Smaller sub-samples were taken from trays close to the cages as these were expected to have higher POC content. Each sub-sample was placed into a 2.5 l plastic mixing bottle, 500 ml of deionised water was added, and the contents mixed using a kitchen hand blender (Bosch) for 20 s. The sand was allowed to settle, following which the supernatant was gently poured into a vacuum filtration funnel fitted with a pre-ashed 47 mm diameter GF/F filter. The mixing and filtration steps were then repeated with another 500 ml of deionised water. Both mixing and filtration steps were then repeated once more, but this time pouring the supernatants into a second clean GF/F filter cup. Vacuum suction was then applied for up to several hours after which the filter papers were removed and placed overnight on a perforated tray in a desiccator over a small amount of 37% HCl to remove any inorganic carbon. The following day, the acidified filters were placed in a warm oven and dried overnight at 50°C. Blank extractions were also performed using the clean sediment used to fill the sediment trays. Dried filter papers were then individually wrapped in foil and shipped to the University of Essex where the quantity of organic carbon on each filter was determined using a PrimacsMCS analyser (Skalar). Calibrations were made at low, medium, and high detection ranges using desiccated acetanilide.

The amounts of POC deposited into each sediment tray were estimated taking into account the retention efficiencies of the GF/F filters. The estimated retention efficiency of POC on a filter was calculated as

$$E = 1 - (\mathrm{POC}_2/\mathrm{POC}_1) \tag{1}$$

where E is the capture efficiency, and POC_1 and POC_2 are the quantities of POC (mg) measured on the first and second wash filters, respectively. Any samples where the capture efficiency was estimated to be less than 0.5 were repeated. The estimated POC (mg) in each sediment sub-sample was then calculated as

$$\mathrm{POC}_{\mathrm{sub}} = \mathrm{POC}_1 + \mathrm{POC}_2 + [\mathrm{POC}_2 \times (1 - E)/E] \tag{2}$$

The POC in each sediment sub-sample ($\mathrm{POC}_{\mathrm{sub}}$) was corrected for the background organic carbon present in the clean sand used to fill the boxes as

$$\mathrm{POC}_{\mathrm{corr}} = \mathrm{POC}_{\mathrm{sub}} - (\mathrm{POC}_{\mathrm{blank}} \times S_{\mathrm{sub}}) \tag{3}$$

where $\mathrm{POC}_{\mathrm{blank}}$ is the mean POC (mg g^{-1}) in the clean sand from the blank measurements and S_{sub} is the weight of the sub-sample (g) taken from the tray. The total POC (mg) present in each sediment tray was then estimated as

$$\mathrm{POC}_{\mathrm{tray}} = \mathrm{POC}_{\mathrm{corr}} \times \mathrm{S}_{\mathrm{tray}} / S_{\mathrm{sub}} \tag{4}$$

where $\mathrm{S}_{\mathrm{tray}}$ is the total weight (g) of dried sediment in the tray and $\mathrm{S}_{\mathrm{sub}}$ is the weight of the sub-sample (g) taken from the tray. The surface area of each sediment tray was 0.0504 m^2, so the estimate of the POC deposited over the 7 d deployment period (mg m^{-2}) is given by

$$\mathrm{POC}_{\mathrm{dep}} = \mathrm{POC}_{\mathrm{tray}} / (0.0504) \tag{5}$$

The final estimates of the organic carbon deposited into each sediment tray were taken as the mean of the triplicate analyses from each tray. Given that the trays were initially filled with clean sand, it was assumed that breakdown of the organic material in the traps would be negligible over the 7 d deployments.

2.6. Modelling of organic particulate waste dispersal and settlement using NewDEPOMOD

A full description of NewDEPOMOD (v1.4.0-rc02-WORLD edition) can be found in SRSL (2021). For Bay of Meil, the model had to be driven using current data collected in 2009, with the period extracted corresponding to the tidal state during the sediment tray deployments. For Quanterness, water current data was taken from the 2019 data collected during the time the sediment trays were deployed. Modelled bathymetry was based on the original site licence data augmented with any recent Admiralty surveys (https://datahub.admiralty.co.uk). Salmon feed inputs and biomass data were supplied by Cooke Aquaculture based on the feed quantities used and harvest records at each site.

Firstly, models were run using recommended SEPA defaults (SEPA 2019a), except for some specific adjustments required to accommodate the smaller spatial mesh due to the sediment tray spacing at each site (Table 2). Secondly, the effects of adjusting 5 parameters often used for tuning NewDEPOMOD models were tested across a range of values from low to high (Table 3). Each of the parameters were adjusted in turn whilst keeping the remaining parameters set to the mid-value. Modelled organic carbon depositions were compared with the sediment tray results using scatterplots and calculation of the root mean squared error (RMSE). A multiple linear regression was performed to rank the sensitivity of the results to tuning. The RMSE value for the model fit was used as the dependent variable, with the 5 parameters that were altered as the independent variables. The p-value was used to assess which parameters held the most significance. If a parameter showed low significance ($p \geq 0.05$), then the SEPA default was used in further model runs. For parameters that showed high significance ($p \leq 0.05$), further runs were carried out using values near those that showed the best RMSE values in initial runs. For Quanterness, models were run both including and removing the tidal residual flow as recommended by SEPA for modelling highly dispersive sites (SEPA 2019a).

Table 2. Parameter adjustments from SEPA recommended default settings for the NewDEPOMOD baseline model runs. Note that the last 3 parameters are unitless

Parameter	Units	SEPA default	Value used	Reason
Bathymetry.bufferZoneWidth	m	100	250	Allows particles to move further in a timestep — model crashes with low value when resuspension is very active
Bathymetry.minimumSurfaceDX/DY	m	25	10	Sediment trays were closer to each other than 25 m — a smaller resolution was required to allow sediment box locations to be in separate cells
Bathymetry.surfaceDX/DY	m	25	10	
Transports.BedModel.surfaceDX/DY	m	25	10	
Transports.BedModel.contractionT50		Infinity	900	To allow tuning of the bed model
Transports.BedModel.expansionT50		1	14400	
Transports.BedModel.releaseParticles. particlesPerArea		0.0016	0.01	Maintains the setting of 1 resuspension particle per bathymetry cell

Table 3. Parameter values used in model tuning. A range of values were tested between Low and High levels. The variable names used in NewDEPOMOD are Resuspension height: Transports.BedModel.releaseHeight.height; Hydraulic roughnes: Transports.bottomRoughnessLength.smooth; Critical shear stress: Transports.BedModel.tauECritMin; Layer mass: Transports. BedModel.dLayerMass; Horizontal bed dispersion: Transports.suspension.walker.dispersionCoefficient X and Y. Values for hydraulic roughness were increased in a logarithmic manner so only 6 values plus the default were tested

Parameter	Default	Low						High
Resuspension height	0.12	0	0.0144	0.0555	0.12	0.44	1	2
Hydraulic roughness	0.001273	0.00001		0.0001	0.001	0.01	0.1	1
Critical shear stress	0.02	0.00002	0.0002	0.002	0.02	0.2	2	20
Layer mass	3375	5	15	45	130	385	1140	3375
Horizontal bed dispersion	0.1	0.1	0.25	0.5	0.75	1	1.5	2

3. RESULTS

3.1. Water currents

Bay of Meil is less energetic than Quanterness with mean flows of 3.2, 3.3, and 3.4 cm s^{-1} at 5.6, 4.1, and 2.1 m above the seabed, respectively, based on the 2009 data. Near seabed peak flows were up to 9.7 cm s^{-1} towards the south (Fig. 2a). At Quanterness, mean flows from the 2009 ADCP deployment were 15.4, 13.1, and 11.3 cm s^{-1} at 6.1, 4.6 (net depth), and 2.1 m (taken as near-bed for licencing purposes) above the seabed, respectively. Near-bed peak flows were up to 31.6 cm s^{-1} in a predominantly southeast direction. Mean flows from the ADCP deployment in 2019 were higher at 45.7, 16.0, and 13.9 cm s^{-1} (at the same depths as in 2009) with near-bed peak flows up to 32.4 cm s^{-1}, again orientated predominantly to the southeast (Fig. 2b).

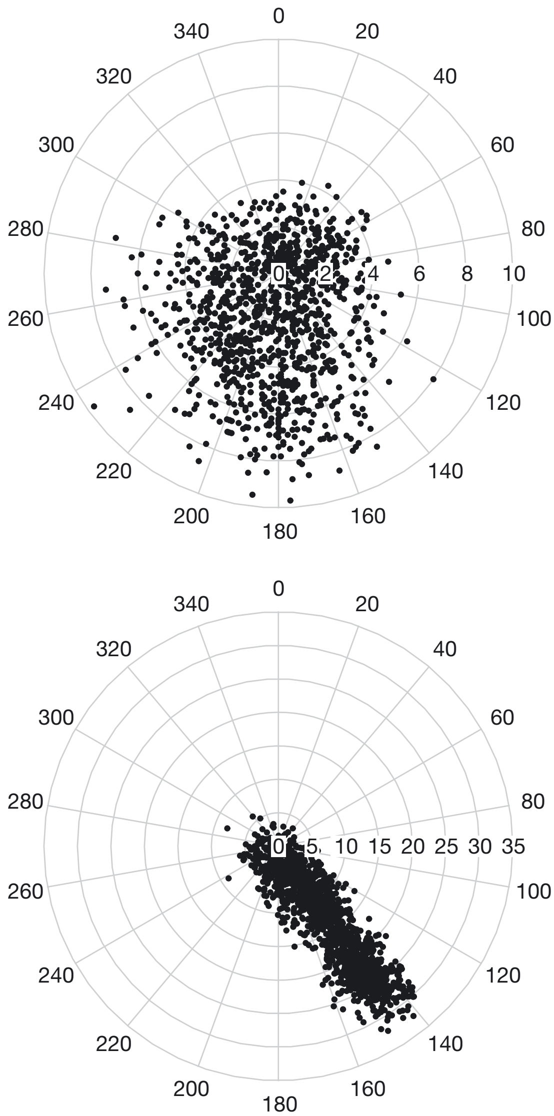

Fig. 2. Near-bed current speeds and directions at (a) Bay of Meil 2009 data and (b) Quanterness 2019 data. Radii indicate speed (cm s^{-1}); note the difference in radii scaling between the 2 sites

3.2. Particle size and POC in sediments

The sediment at Bay of Meil is dominated by fine and very fine sands (Table 4). Coarser sands were found along the east transect which ran into bare rock beyond 30 m from the perimeters of the cages whilst finer sands dominated to the west and north. The maximum percentage of clay particles (2.6%) occurred 20 m from the cages in the south-easterly direction. Sediment POC levels were generally low (<0.5%) but were slightly elevated (1.3%) at the cage edge along the southerly transect.

Sediments at Quanterness are dominated by fine and very fine sands (Table 4) although the percentage of coarse sand and silt was a little higher along the southwest transect. The maximum percentage of clay particles was 3.3% and found 40 m from the cages in the southwest direction. There was no evidence of high levels of organic enrichment in the sediments, with a maximum POC content of 1.3% at 40 m from the cages along the southwest transect. There was also slight enrichment to a similar level near the cages along the east and southeast directions.

It must be cautioned that the POCs from the syringe cores are averaged down to a depth of about 5 cm and surficial POC levels are likely to have been higher. The original intention had been to slice the cores, preserving the vertical structure. However, because of the coarse sediment, material in the cores tended to become partially mixed during collection by the divers and recovery of the cores to the fieldwork vessel.

3.3. Infaunal quality index (IQI)

Cage edge impacts at Bay of Meil were apparent along all 4 transects, with elevated densities of enrichment polychaetes and low IQI scores (Table 5). Along the westerly and northerly transects, degraded conditions were confined close to the cage edges but grabs with 'Moderate' status persisted out to 150 m in the southerly direction. The easterly transect could not be sampled beyond 25–55 m as the ground be-

Table 4. Average percentages by particle size class from N syringe cores collected along sampling transects at Quanterness and Bay of Meil. Samples for PSA were taken from duplicate syringe cores collected adjacent to the sediment tray locations shown in Figs. 3 & 4. V. = very

Site	Transect direction	N	V. coarse sand	Coarse sand	Med sand	Fine sand	V. fine sand	V. coarse silt	Coarse silt	Med silt	Fine silt	V. fine silt	Clay
Bay of Meil	East	4	17.5	14.8	8.8	17.5	23.7	12.3	4.3	3.3	2.3	0.9	1.6
	South	8	9.2	12.6	14.7	21.7	25.0	13.0	3.8	2.4	1.6	0.8	1.4
	West	8	0.5	0.4	3.3	39.0	43.9	9.9	1.3	0.9	0.8	0.1	0.3
	North	8	0.3	0.3	3.2	39.7	43.8	9.3	1.3	0.9	0.8	0.1	0.4
Quanterness	East	9	1.0	4.7	14.0	48.3	16.2	4.3	3.7	2.8	2.1	0.8	1.4
	Southeast	9	1.2	3.8	10.6	48.8	21.4	5.1	3.6	2.3	1.6	0.7	1.2
	Southwest	4	1.3	9.0	16.0	31.4	15.7	7.5	6.6	5.2	3.9	1.6	2.4
	Northeast	4	2.2	7.9	16.1	46.1	15.0	3.3	2.9	2.4	1.9	1.0	1.5

Table 5. Results of enhanced benthic monitoring at Bay of Meil and Quanterness. Dist = distance from cage edge; EP dens = density of enrichment polychaete species; IQI = Infaunal Quality Index (ver 4); Eco = ecological status from the IQI value (Mod: moderate; Rock: sampling was not possible due to rocky substrate; see Sections 1 & 2.4 for further description of Eco designations). The locations of the grab sampling transects at the 2 study sites are indicated by the coloured dots in Figs. 3 & 4

Site	Transect direction	Dist (m)	EP dens (m^{-2})	IQI	Eco
Bay of Meil	East	0	17.021	0.41	Poor
		25	22	0.51	Mod
		55			Rock
		76			Rock
	South	0	878	0.29	Poor
		28	378	0.59	Mod
		39	0	0.61	Mod
		50	11	0.64	Mod
		76	122	0.60	Mod
		102	33	0.65	Good
		151	100	0.60	Mod
		211	44	0.72	Good
	West	0	26.675	0.22	Bad
		26	389	0.62	Mod
		50	300	0.75	Good
		101	133	0.75	High
		152	56	0.84	High
		203	133	0.81	High
		274	78	0.75	Good
	North	0	27.753	0.23	Bad
		23	44	0.73	Good
		51	67	0.76	High
		80	0	0.73	Good
		102	0	0.78	High
		151	56	0.81	High
		208	44	0.72	Good
Quanterness	East	0	6.610	0.28	Poor
		44	10.943	0.27	Poor
		98	1.767	0.45	Mod
		118	367	0.70	Good
		167	333	0.74	Good
		200	11	0.73	Good
		253	11	0.79	High
	Southeast	0	45.229	0.24	Bad
		60	2.111	0.26	Poor
		111	11	0.64	Mod
		165	44	0.71	Good
		217	67	0.72	Good
		275	89	0.73	Good
		319	44	0.66	Good
	Southwest	0	422	0.63	Mod
		36	67	0.72	Good
		58	267	0.79	High
		90	6.777	0.32	Poor
		102	744	0.72	Good
		122	0	0.78	High
		173	189	0.80	High
	Northeast	0	111	0.68	Good
		33	11	0.71	Good
		48	11	0.88	High
		66	2.166	0.54	Mod
		92	67	0.66	Good
		123	89	0.67	Good
		155	33	0.74	Good

comes rocky. The infaunal data from this site suggests that most impact occurs close to the cages but extends beyond 100 m in a southerly direction.

At Quanterness, benthic impacts were most apparent at the east and southeast cage edges with high densities of enrichment polychaetes and low IQI scores (Table 5). Benthic impacts also extended further out along these transects where 'Good' conditions were not reported until 118 and 165 m from the east and southeast cage edges, respectively. The ecological status of most of the other grab samples was 'Good' or 'High', except for single anomalous samples along the west and north transects. Overall, the IQI results suggest that most of the impact occurs relatively close to the cages but extends further out in the easterly and south-easterly directions, as might be expected from the measured near-bed water currents (Fig. 2b).

3.4. Sediment tray results

The original intention had been to deploy sediment trays during spring and neap tides at both sites but because of logistical constraints on diver availability, both deployments at Bay of Meil took place during spring tides. The implications of this for interpretation of the Bay of Meil sediment tray results are likely to be relatively minor because this site has much lower near-bed flows than Quanterness. For example, under neap conditions, near-bed flows at Quanterness reach around 25 cm s^{-1} whilst at Bay of Meil, flow is only up to 10 cm s^{-1}, even during spring tides.

At Bay of Meil, trays beyond 40 m along the easterly transect could not be placed flush with the sediment surface as the ground becomes exposed rock. Not all the sediment trays were successfully recovered. At Bay of Meil, one tray was lost from the first deployment, while at Quanterness, one tray was lost from the neap tide deployment and 4 from the spring tide deployment. In addition, sediment trays from both deployments at 160 m along the south-easterly transect were largely empty on recovery so their contents could not be analysed. At Bay of Meil, the maximum estimated organic carbon deposition was 96 g C m^{-2} over 7 d (Fig. 3b). Despite both deployments for Bay of Meil being under spring tide conditions, there was a noticeable difference in deposition patterns, especially along the northern and southern transects. Deposition was highest to the south during the second deployment leading to noticeable elevated deposition at the cage edge and out as far as 75 m.

The maximum estimated organic carbon deposition at Quanterness was 55 g C m^{-2} over 7 d (Fig. 3a). This was lower than at Bay of Meil which may reflect the lower salmon biomass at this site but also slightly further dispersion of material under stronger flow conditions. During the spring tide deployment, the highest deposition occurred within 20 m of the cage

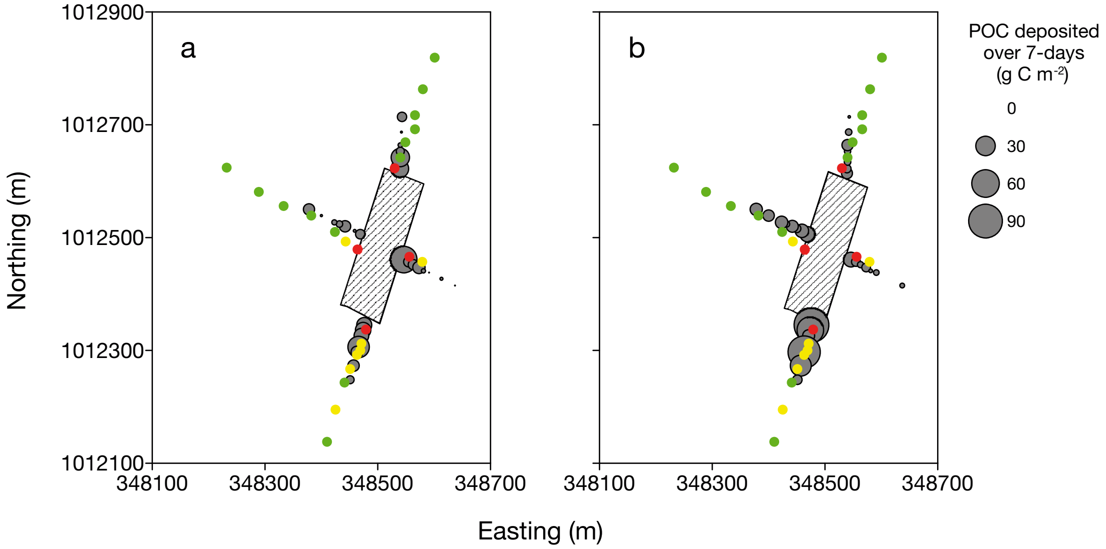

Fig. 3. Patterns of particulate organic carbon (POC) deposition at Bay of Meil estimated from sediment trays compared with infaunal quality index (IQI) values derived from benthic grabs. Grey circles: POC deposition is from sediment trays deployed during (a) spring tides 7–14 July and (b) spring tides 22–29 July; the grey circle areas are proportional to the amount of POC deposited into the sediment trays over the 7 d deployments. IQI results derived from the benthic grabs — green circles: 'High' or 'Good', yellow circles: 'Moderate', and red circles: 'Poor' or 'Bad' ecological status. Hatching: approximate perimeter of the cages

edge along the east and southeast transects but there was also evidence of slightly elevated deposition at around 40 m along the southwest and northeast transects. During the neap tide deployment deposition of waste material was lower, especially in a south-easterly direction and the maximum rate of deposition (38.6 g C m^{-2}) was at the start of the easterly transect.

3.5. Comparison of POC deposition with benthic faunal observations

Although the transects for the sediment trays and benthic grabs were not exactly aligned (Figs. 3 & 4), positions were sufficiently close to suggest comparisons between the results might not be unreasonable. As the total number of benthic grabs collected was less than the number of sediment trays, IQI values were compared with POC deposition estimated from the nearest tray. Although some locations with low estimated deposition were associated with grabs yielding poor ecological state, the overall spatial patterns from the sediment trays and benthic grabs seem to be in broad agreement. At Bay of Meil, the heaviest POC deposition occurred along the southerly transect which was also the direction in which the less than 'Good' IQI state extended the furthest (Fig. 3). At Quanterness, elevated POC deposition and poorer IQI states occurred along the easterly and south-easterly transects (Fig. 4). Considering both sites, POC deposition of more than 20–30 g C m^{-2} over 14 d (equivalent to 1.4–2.1 g C m^{-2} d^{-1}) tended to be associated with poor ecological state (IQI < 0.64), although some locations with lower estimated waste deposition also had less than 'Good' states (Fig. 5).

The sediment tray results also allow an estimation of the distance within which 90% of the organic waste was deposited over 14 d assuming deposition declines exponentially (Fig. 6). The decline curves tended to underfit the estimated POC deposition rates at the cage edges so that the 90% deposition distances can only be considered indicative. Bearing this in mind, dispersal of 90% of the mass deposited was estimated to extend only a further 60 m out at Quanterness, despite the markedly faster near-bed current regime at this site when compared with Bay of Meil (Fig. 2).

3.6. Modelling using NewDEPOMOD

For Bay of Meil, the best fitting model was produced using the SEPA default parameters except for resuspension height which had to be increased to produce any spread of resuspended material (Table 6). However, when compared with the sediment trap results, particle resuspension and re-dispersal appeared to be too constrained with the southerly extension seen in both the sediment tray results and IQI states not being reproduced (Fig. 7).

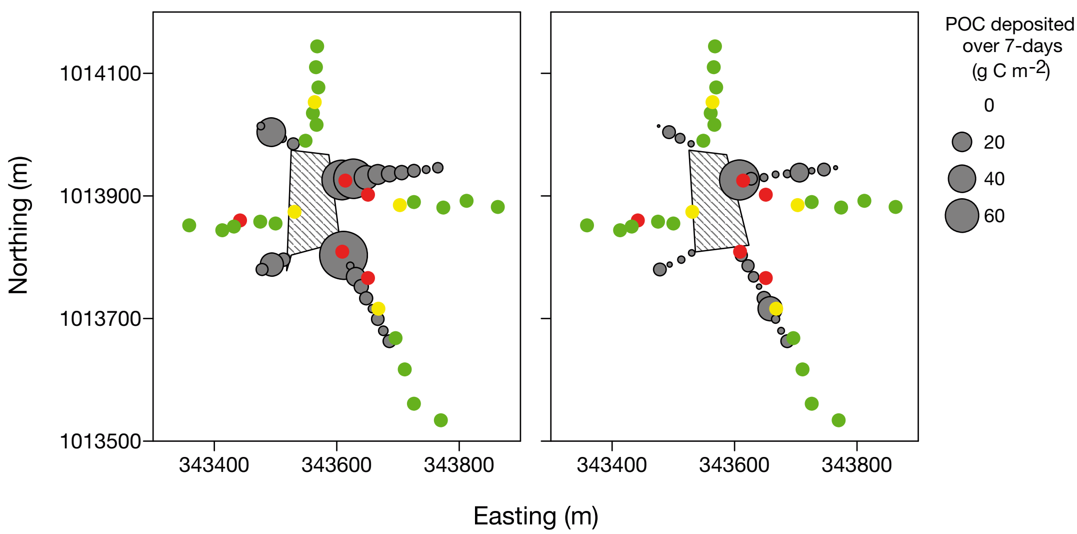

Fig. 4. Patterns of POC deposition at Quanterness estimated from sediment trays compared with IQI derived from benthic grabs. Grey circles: POC deposition from sediment trays deployed during (a) spring tide 31 May–7 June and (b) neap tide 10 June–17 June; the grey circle areas are proportional to the amount of POC deposited into the sediment trays over the 7 d deployments. IQI results derived from the benthic grabs — green circles: 'High' or 'Good', yellow circles: 'Moderate', and red circles: 'Poor' or 'Bad' ecological status. Hatching: approximate perimeter of the cages

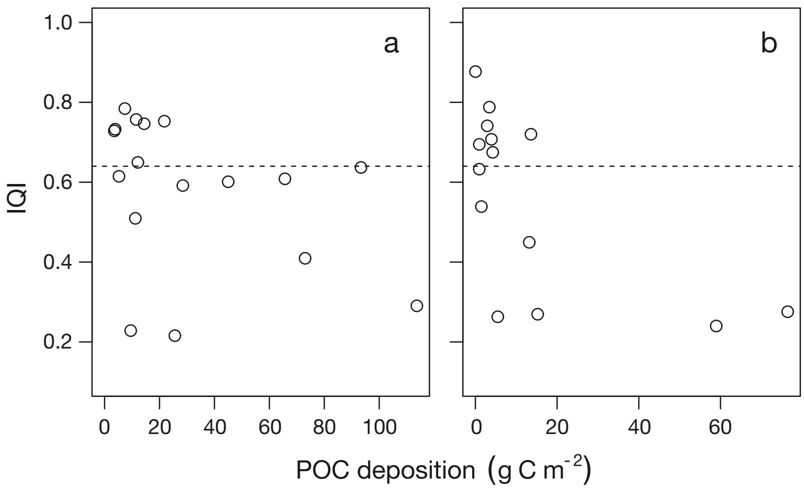

Fig. 5. IQI from benthic grabs against estimated POC deposition as the sum of the two 7 d sediment box deployments from the closest sediment trays at (a) Bay of Meil and (b) Quanterness. Dashed horizontal lines: the IQI boundary between 'Good' and 'Moderate' ecological states

4. DISCUSSION

4.1. Modelling waste dispersal at Bay of Meil and Quanterness

Even though Bay of Meil has relatively weak near-bed currents compared with Quanterness, the seabed at both sites is sandy with POC content of <1.2%. Thus, neither location is strongly depositional, and transport needs to be accounted for when modelling fish farm waste dispersal (Broch et al. 2017). Although a modelled footprint could be obtained for Bay of Meil using NewDEPOMOD default parameter values, the footprint appeared to be too constrained when compared with the sediment tray and IQI results. In contrast, modelling of Quanterness using the default parameters failed to produce any benthic footprint with all simulated particles being lost from the model domain. This finding agrees with several other modelling studies conducted at higher energy sites (Chamberlain et al. 2005, Chamberlain & Stucchi 2007, Chang 2012, Keeley et al. 2013a, Chang et al. 2014). Over-dispersion has been reported when modelling sites with peak near-bed current speeds above 20 cm s^{-1} (Chang 2012, Chang et al. 2014) or 30 cm s^{-1} (Keeley et al. 2013a), while at the Canadian site studied by Chamberlain et al. (2005), the maximum near-bed current speed was 36.2 cm^{-1}, which is

For Quanterness, application of the SEPA defaults led to all waste particles being moved out of the model domain. Removal of the tidal residual current, as recommended by SEPA for high-energy sites, also failed to produce a depositional footprint. To produce a depositional footprint the critical shear stress (tau$_e$crit) had to be increased substantially (Table 6). Although the modelled footprint did extend a little further out to the east and southeast compared with the westerly and northerly directions, the sediment tray and IQI results suggested that dispersal was still being underestimated (Fig. 8).

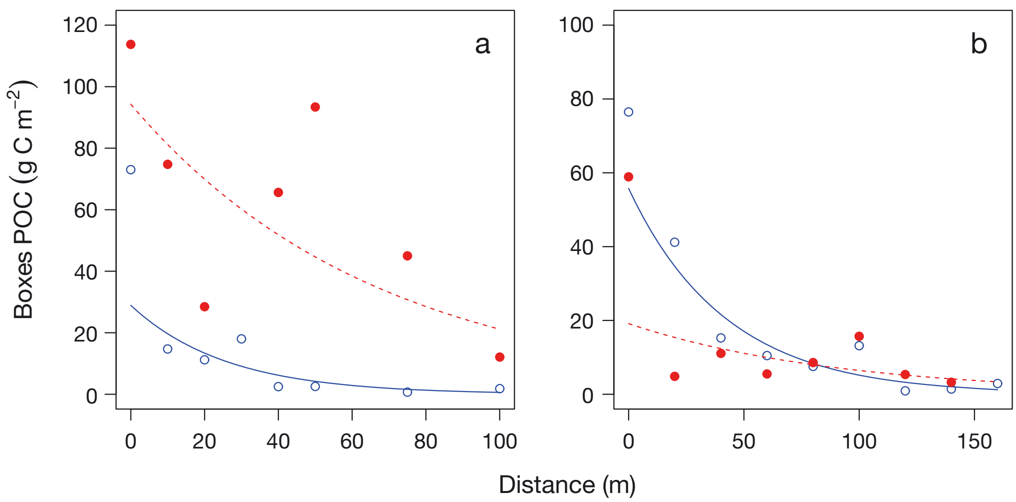

Fig. 6. POC deposition over the 14 d sediment tray deployments fitted with exponential declines with distance for the maximally impacted transects at (a) Bay of Meil and (b) Quanterness. Blue open circles and solid lines: the easterly transects; red filled circles and dashed lines: (a) southerly transect at Bay of Meil, and (b) south-easterly transect at Quanterness

Table 6. Parameter values for the best-fitting fixed erosional critical shear stress NewDEPOMOD models for Quanterness and Bay of Meil using the full current meter data (without the tidal residual current removed). Values corresponding to NewDEPOMOD defaults are indicated in italics. The RMSEs are calculated over both the sediment tray deployments

Parameter	Bay of Meil	Quanterness
RMSE	20.5137	9.1085
Resuspension height	2	0.44
Hydraulic rough	*0.001*	*0.001*
Critical shear stress	*0.02*	20
Layer mass	*3375*	*3375*
Dispersion	*0.1*	*0.1*

similar to that at Quanterness. Thus, despite both Orkney sites being shallow (<20 m deep) and with sandy sediments, we were unable to identify a common set of parameter values which would produce modelled footprints comparable with the sediment tray observations. DEPOMOD v2 was previously validated by Cromey et al. (2002a) at 2 Scottish farms with near-bed peak current speeds of 30.8 and 24.2 cm s^{-1} and which were described as dispersive and depositional, respectively. In that validation, model predictions were compared with the contents of sediment traps, although these were only deployed for 24 h, and through model comparison with ITI results from benthic grabs. It is not clear why Cromey et al. (2002a) were able to obtain a modelled benthic footprint at the dispersive site, although it may be that the periods of strong flow in their data occurred for short enough times to avoid removal of particles from the model domain. At the cage edges in the present study, modelled and observed deposition rates were broadly similar, being up to 100 g C m^{-2} over 14 d. This suggests that the conversion of feed inputs and simulation of the initial sinking of waste to the seabed are probably reasonably accurate, suggesting that there is something incorrect in how NewDEPOMOD treats the resuspension and further dispersal of settled organic waste from coarser sediments. The effects of accounting for sediment specific differences in erosional critical shear stress have also been demonstrated by Carvajalino-Fernández et al. (2020b). In that study, Regional Ocean Modeling System (ROMS) hydrodynamic models were coupled with sediment-dependent resuspension thresholds to simulate dispersal of salmon farm waste within the Altafjoden and Frøya Archipelago, Norway. Model runs were also performed with no resuspension or with a fixed erosional critical shear stress of 0.018 N m^{-2}. The shapes of the resulting particulate organic matter (POM) footprints were clearly affected by the choice of resuspension scheme, although comparisons with sediment trap data suggested that the spatially varying threshold model was still underpredicting POM deposition out to 600 m from the cages.

To date there have been a limited number of studies where laboratory or field flumes have been used to directly measure fish farm waste resuspension. Law et al. (2016) used a Gust Microcosm Erosion Chamber where faecal particles began to resuspend at a stress of 0.01 N m^{-2} but a large increase in resuspension occurred at 0.08 N m^{-2}, regardless of substrate type. However, much less faecal material was eroded when the underlying substrate was coarser. On cobble, less than 25% of the material was eroded whereas nearly complete resuspension of the faecal particles occurred when the substrate was mud. Averaged over stresses up to 0.6 N m^{-2}, the erodibility parameter estimates for salmon waste on mud were estimated at 1.3×10^{-6} kg m^{-2} s^{-1}, sand at 3.5×10^{-7} kg m^{-2} s^{-1}, sand gravel at 6.0×10^{-7} kg m^{-2} s^{-1}, and sand cobble at 5.8×10^{-7} kg m^{-2} s^{-1}. Using a horizontal flume, the resuspension of intact salmon faecal pel-

Table 7. Estimated distances from cage edge encompassing 90% of the POC deposition along the transects of maximum dispersal as estimated from exponential decline curves fitted to the sediment tray trap data. POC deposition = Intercept*exp(Slope*Distance). BOM = Bay of Meil, QUA = Quanterness

Site	Direction	Intercept	Slope	Total deposition along 1 m wide transect extended to 2000 m (g)	90% of total deposition (g)	Distance corresponding to 90% of total deposition (m)
BOM	East	28.910	−0.0385	750	675	60
	South	94.274	−0.0149	6300	5670	150
QUA	East	55.758	−0.0237	2350	2115	120
	Southeast	19.129	−0.0109	1760	1584	210

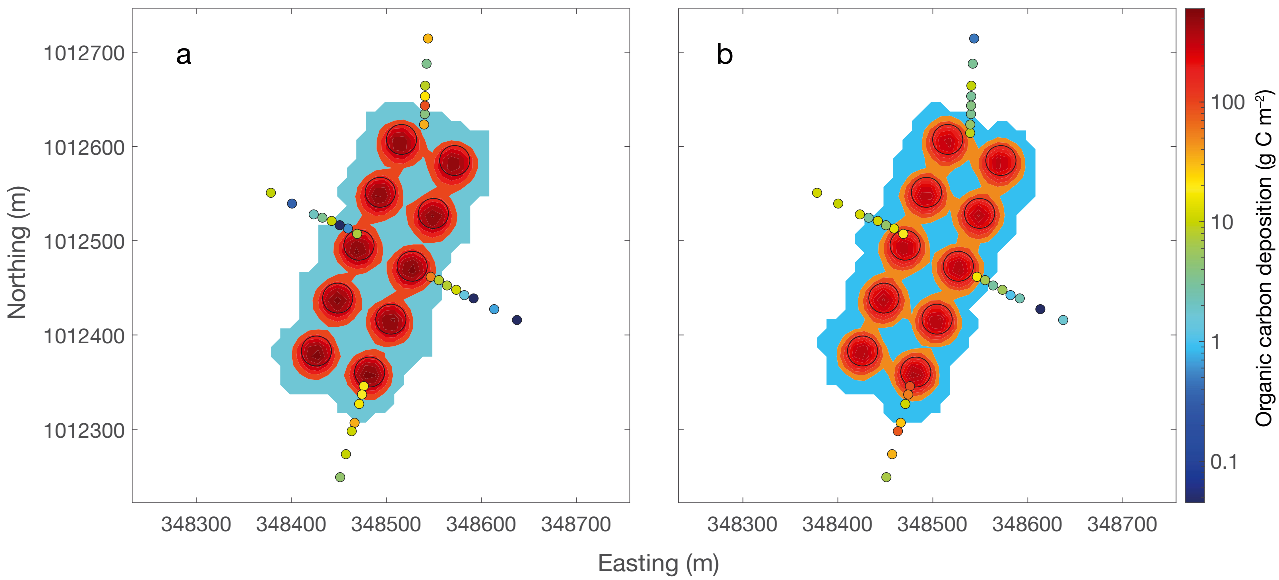

Fig. 7. Modelled deposition footprint from the best fitting NewDEPOMOD model for Bay of Meil. Large open circles: salmon cages; filled contours: predicted organic carbon deposition over 7 d; small solid circles: observed organic carbon deposition from the 7 d sediment tray deployments during (a) spring tide deployment and (b) second spring tide deployment

lets was studied by Carvajalino-Fernández et al. (2020a). Pellets began to saltate at around 5.4 cm s^{-1} and resuspend with current speeds of 5–20 cm s^{-1}. Substrate type did not appear to affect saltation but did influence resuspension. Mean values for the critical erosional shear stress were 0.06 N m^{-2} on slate and 0.07 N m^{-2} on mud but 0.12 N m^{-2} on sand and 0.32 N m^{-2} on fractured rock. However, Carvajalino-Fernández et al. (2020a) cautioned that these results are unlikely to apply to broken down and flocculent material, for which lower critical shear stresses were expected. Although uneaten feed pellets usually comprise a small fraction of the overall waste, understanding their transport has also been of interest. In flume experiments, feed pellets became wedged in cobble crevices but on mud, pellets began to saltate

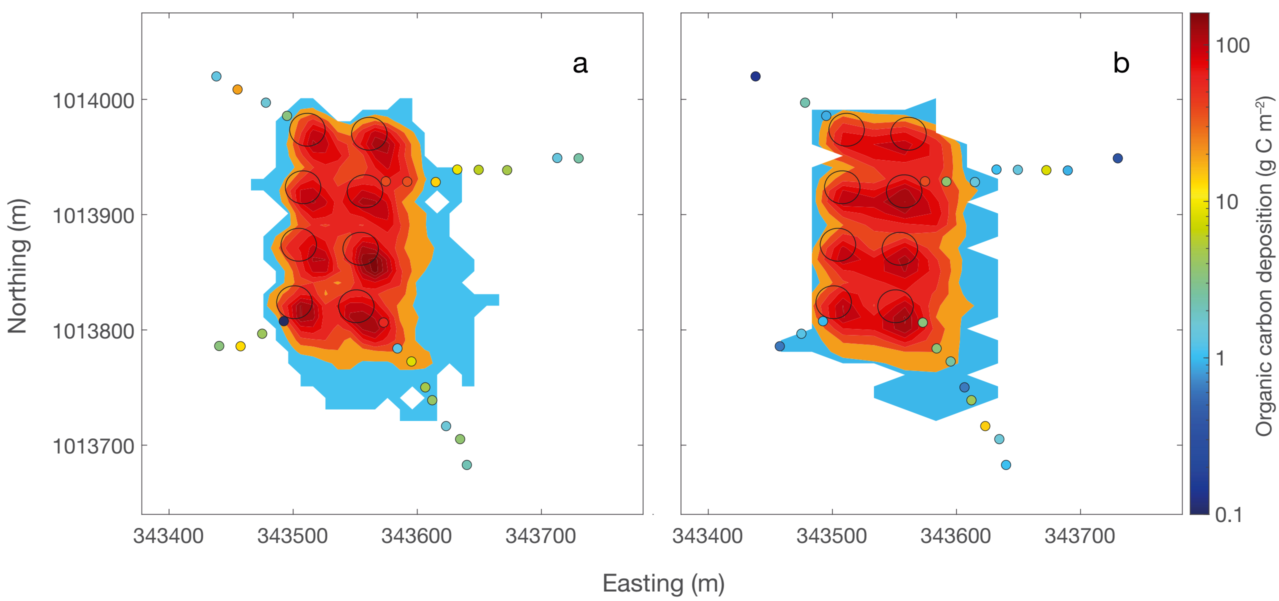

Fig. 8. Modelled deposition footprint from the best fitting NewDEPOMOD model for Quanterness. Large open circles: salmon cages; filled contours: predicted organic carbon deposition over 7 d; small solid circles: observed organic carbon deposition from the 7 d sediment tray deployments during (a) spring tide deployment and (b) neap tide deployment

at 0.08 N m^{-2} while on sand cobble, this did not occur until the shear stress reached 0.16 N m^{-2} (Law et al. 2016). Once pellets were moving, there was little difference in the overall horizontal distance travelled (2–3 cm s^{-1} at 0.16–0.24 N m^{-2}). At stress levels above 0.16 N m^{-2}, exposed pellets began to break up but buried pellets required shear stresses above 0.48 N m^{-2} to become re-exposed. A general conclusion from these laboratory experiments is that coarser substrates provide spatial refuges for waste particles whereas consolidated mud or heavily organically enriched sediment provides less protection from erosion. Furthermore, sediments frequently contain biofilms which render the substrate 'sticky' affecting resuspension. This may be especially true of shallow sediments with sufficient illumination for films of benthic diatoms to develop (Grant et al. 1986).

Whilst laboratory studies have provided insights into how fish farm waste behaves under different water flow stresses, it can be difficult to fully simulate field conditions (Adams et al. 2020). Using an annular field flume, critical water speeds for sediment resuspension were estimated at between 33 and 55 cm s^{-1} at 100 cm above the bed (Dudley et al. 2000). However, Cromey et al. (2002b) noted that applying these critical speeds to Scottish fish farms would result in virtually zero resuspension, which was considered unrealistic. A series of experiments are described by Adams et al. (2020) at 9 Scottish farms where a small annular flume was deployed close to the cage edges and a larger flume at distances of 100–500 m. Sediments ranged from coarse sand to silt, and all showed signs of organic enrichment. The flume results suggested an average erosional critical shear stress of 0.02 N m^{-2} (range 0.01–0.04 N m^{-2}) for heavily organically enriched sediments such as those found close to the cage edges. This value is very close to the default critical erosional shear stress of 0.018 N m^{-2} used in DEPOMOD. However, away from the cage edges higher critical shear stresses were required to initiate resuspension (mean 0.19 N m^{-2} but up to 0.74 N m^{-2}). A revised estimate for the erodibility constant was also calculated as 0.031 kg m^{-2} s^{-1} and this was applied in NewDEPOMOD. However, it should be noted that the form of the relationship between erodibility and excess shear stress was also changed in NewDEPOMOD, so that the erodibility constants are not directly comparable (Fox et al. 2022). However, none of the studies cited above have suggested that the erosional critical shear stress should be as high as 20 N m^{-2}, a value which practically shuts off resuspension, but which was required when modelling Quanterness to obtain a benthic footprint.

An important observation is that despite the very different current regimes at Bay of Meil and Quanterness, the distance within which 90% of the waste was deposited was only about 60 m greater at the more energetic site (Table 7). NewDEPOMOD models resuspend material as discrete particles with similar properties to the initial waste, but the real-life process may be more complex. Rapid consolidation of organic material, perhaps by the formation of biofilms (Droppo et al. 2007), may mean that settled waste becomes harder to erode than previously thought and that the characteristics of the waste change quite quickly following settlement.

The decomposition of organic particles within surface sediments is also only partially understood. The concept of assimilation capacity (AC) is intended to express the ability of biological processes to mineralize organic matter without depleting limiting substrates, especially oxygen (Bravo & Grant 2018). For example, when sufficient organic accumulation occurs, anaerobic decomposition via sulfate reduction allows the concentration of free sulphide to increase which is a result of the AC of the near field benthos being overwhelmed. In highly dispersive sites, AC should never be exceeded because the accumulation of POM would be small enough to be aerobically degraded (Bravo & Grant 2018). However, that assumes that the organic waste is rapidly dispersed over a large area and does not accumulate close to the farm. If processes such as organic material becoming trapped between sediment grains, or the development of biofilms increasing cohesion, are significant, then the residence time of POM in the sediment can become sufficient for enrichment to occur, resulting in increased benthic oxygen demand and measurable changes in sediment chemistry and biology. Accumulation of POM in the sediment trays and measurable changes in benthic ecological quality were observed at both Bay of Meil and Quanterness, despite the latter site having strong water flows. The present results show that high flows should not automatically be assumed to result in wide dispersal and dilution of organic waste from fish farms, at least for shallow sites.

4.2. Study caveats

Due to logistical constraints caused by the Covid-19 pandemic, we were unable to deploy an ADCP concurrent with the Bay of Meil sediment tray deployments, so the current meter data collected for licencing purposes in 2009 had to be used to drive the NewDEPOMOD model. A short set of ADCP data collected at Bay of Meil in April 2018 also showed higher and more variable direction for water speeds at all 3 reported depths. This was likely due to wind forcing, and demonstrates how local meteorological conditions can impact water flows at such shallow sites. Such effects may explain the variable directions of waste deposition seen in the first and second sediment tray deployments, which were not reproduced by the model when driven using the historical current data.

Forcing dispersal models using data collected from single current meters can also be criticised as it may not capture local spatial variability (Broch et al. 2017). NewDEPOMOD does have the ability to accept multiple current meter data or spatially varying flow fields from a hydrodynamic model such as FVCOM. However, given the computational complexity in running high-resolution 3-dimensional oceanographic models, they are not widely used in the aquaculture industry, and remain largely a research tool. Although the high-resolution hydrodynamic model WeStCOMS, which covers most of the Scottish west coast, is available, it has not yet been extended to include the Orkney Islands (Davidson et al. 2021).

The present study used an approach described in Grant (1985) where clean sediment was deployed in metal trays at a Nova Scotian beach to measure organic carbon deposition during flood and resuspension during ebb tides. Whilst sediment traps have been previously deployed around fish farms (Cromey et al. 2002a, Stucchi et al. 2005, Riera et al. 2017), classical cone or parallel-sided traps do not allow for resuspension of the settled material and so measure gross rather than net deposition. In contrast, our approach was designed to allow material to be resuspended and redistributed in a more natural manner. Although the carbon deposition results seemed credible, there are several uncertainties with the technique. Trays were deployed for 7 d which may be insufficient to capture variability in deposition due to periods of strong wind or freshwater run-off, although our deployments were over longer periods than those in many previous studies where collection was over only 24 h (Cromey et al. 2002a, Stucchi et al. 2005, Riera et al. 2017). It should be noted that if deployments of tray traps are too long, then the sediment will be replaced by ambient material, and the contents of the tray will reflect other waste reduction processes including biological degradation. Deployments of 7 d were assumed to avoid substantial losses due to biological degradation whilst allowing deposition of sufficient organic carbon to permit quantification. The main practical limitation to the technique was the dive time required for positioning and retrieving the trays. Diving at Scottish fish farms generally uses surface supplied compressed air, so bottom times and depths are limited and at deeper locations, a different approach would be required to deploying and recovering sediment trays. It would also have been preferable to place replicate sediment trays at each location to better capture small-scale spatial variability and further traps to act as references measuring background carbon deposition. There are few measurements of background carbon flux for Scottish waters but Overnell & Young (1995) estimated a background deposition originating from phytoplankton of 82 mg C m^{-2} d^{-1} in Loch Linnhe whilst Brigolin et al. (2009) reported a background carbon flux of 38 mg C m^{-2} d^{-1} in Loch Creran. The lowest level of deposition into the sediment trays was recorded at Quanterness and was 140 mg m^{-2} over 7 d, or 20 mg C m^{-2} d^{-1} which is a similar order of magnitude to the estimates of background deposition mentioned above. This suggests that the sediment tray method should have been sensitive enough to detect organic carbon deposition rates down to typical background levels. Unfortunately, setting up additional reference trays would have increased the dive time beyond what was available. Further comparisons between classical retentive sediment traps and the open sediment trays could be informative as this could provide a direct measure of how important resuspension is at more energetic sites. Nonetheless, in terms of spatial coverage, our data represent some of the most detailed direct measurements of organic waste deposition around fish farms ever achieved.

The relationships between organic carbon deposition and benthic community responses can also be quite variable, especially when considered at fine spatial scales. Results from different studies are thus hard to compare because of the use of different measures of impact. For example, Findlay & Watling (1997) suggested that many marine sediments should be able to assimilate quite high amounts of organic material (5–20 g C m^{-2} d^{-1}). However, other studies have noted negative biological impacts at lower deposition rates. Hargrave et al. (2008) suggested that bioturbators may begin to decline at deposition rates above 5 g C m^{-2} d^{-1} whilst anoxia may develop at deposition rates as low as 1 g C m^{-2} d^{-1} (Hargrave 1994). At a site in British Columbia, transition between oxic and anoxic states was noted at ~1 and 5 g C m^{-2} d^{-1} (Chamberlain & Stucchi 2007). Current speeds are clearly also important as a flux of 5 g C m^{-2} d^{-1} may result in noticeable impacts at nondispersive sites while at dispersive sites 11.2 g C m^{-2} d^{-1} was required to elicit similar effects (Keeley et al. 2013a). Based on sediment sulphide standards used in Canada, the safe assimilative capacity for poorly flushed environments was suggested to lie between 0.6–22.1 g C m^{-2} d^{-1}, but might be almost indeterminate for well-flushed locations (Bravo & Grant 2018). Based on the sediment tray results, a transition from 'Good' to 'Moderate' IQI seemed to be associated with carbon deposition rates of between 20 and 30 g m^{-2} over 14 d for the Orkney sites studied. This equates to an average daily flux of 1.4–2 g m^{-2} d^{-1}, which is at the lower end of the ranges cited above, but higher than the SEPA threshold of 0.33 g C m^{-2} d^{-1}, where a transition to an IQI of below 0.64 is expected. Some low ecological state grabs in the Orkney data were also associated with low or moderate carbon deposition as estimated from the nearest sediment tray, but as grabs were not collected exactly where sediment trays were placed, such differences could have arisen from small-scale spatial variability in waste settlement or retention, patchiness in benthic community response or inaccuracies in the sediment tray results.

5. CONCLUSIONS

Open sediment tray traps were deployed at 2 salmon farms at shallow sandy sites and provided a novel approach to directly measuring organic waste deposition whilst allowing for resuspension. The general spatial patterns in both sediment tray and benthic impact (IQI) results were consistent with the expected directions of maximum impact based on observed water currents at the sites. However, despite the different flow regimes at the 2 sites, dispersal of organic waste appeared to occur over similar distances. Based on the sediment tray results, around 90% of the POC was estimated to settle within 150 m of the cage perimeter at the slower flow site (Bay of Meil) and within 210 m of the cage perimeter at the faster flow site (Quanterness).

Attempts to find a common set of parameter values which would allow the NewDEPOMOD model to produce POC footprints with reasonable agreement to the observations failed. The only approach which produced credible footprints was ad hoc tuning, which resulted in substantially different values between the sites for important parameters such as the critical erosion shear stress (tau $_e$crit). This suggests that following settlement to the seabed there is a problem in how NewDEPOMOD models particle resuspension and transport, an issue which seems to be particularly relevant for sites located over coarser sediments. The present study adds to the body of literature where particle-based models have been applied to simulate the benthic footprint of fish farm waste, but where predictions have failed to match either direct and/or indirect observations (Chamberlain et al. 2005, Chamberlain & Stucchi 2007, Chang 2012, Keeley et al. 2013a, Chang et al. 2014; Carvajalino-Fernández et al. 2020b).

At the present time, modellers are spending considerable amounts of time fitting NewDEPOMOD models to individual farms by ad hoc tuning, and established timeseries of benthic infaunal monitoring are required for impact validation. Such an approach is neither desirable from a scientific view nor economically optimal. This suggests that further research is required into both the real-life processes and modelling of organic waste dispersal from fish farms.

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