Bottom trawling resuspends sediment and releases bioavailable contaminants in a polluted fjord

C. Bradshaw [a,*], I. Tjensvoll [a], M. Sköld [b], I.J. Allan [c], J. Molvaer [c], J. Magnusson [c], K. Naes [c], H.C. Nilsson [b]

[a] Department of Systems Ecology, Stockholm University, S-10691 Stockholm, Sweden
[b] Institute of Marine Science, Department of Aquatic Resources, Swedish University of Agricultural Sciences, Turistgatan 5, S-453 21 Lysekil, Sweden
[c] Norwegian Institute for Water Research (NIVA), Gaustadalleen 21, NO-0349 Oslo, Norway

ABSTRACT

Sediments are sinks for contaminants in the world's oceans. At the same time, commercial bottom trawling is estimated to affect around 15 million km^2 of the world's seafloor every year. However, few studies have investigated whether this disturbance remobilises sediment-associated contaminants and, if so, whether these are bioavailable to aquatic organisms. This field study in a trawled contaminated Norwegian fjord showed that a single 1.8 km long trawl pass created a 3—5 million m^3 sediment plume containing around 9 t contaminated sediment; ie. 200 g dw m^{-2} trawled, equivalent to c. 10% of the annual gross sedimentation rate. Substantial amounts of PCDD/Fs and non-ortho PCBs were released from the sediments, likely causing a semi-permanent contaminated sediment suspension in the bottom waters. PCDD/Fs from the sediments were also taken up by mussels which, during one month, accumulated them to levels above the EU maximum advised concentration for human consumption.

1. Introduction

Bottom sediments are considered sinks for many contaminants entering the marine environment. Many contaminants associate readily with sediment particles and to particulate organic matter, organic molecules, colloids and black carbon in sediments (Cornelissen et al., 2005; Olsen et al., 1982; Schwarzenbach et al., 2003). However, if sediment is disturbed, for example by waves, currents, bioturbation, boat wash, dredging or bottom trawling, these particle-associated contaminants can be resuspended into the overlying water (e.g., Hedman et al., 2009; Jonas and Millward, 2010; Nelson et al., 1987; Olsen et al., 1982).

Laboratory experiments have shown that changes in chemical equilibrium may also lead to desorption of contaminants from the particulate to the dissolved phase, depending on the properties of the sediment (Cantwell et al., 2008; Latimer et al., 1999) or the overlying water (Atkinson et al., 2007), resuspension time (Feng et al., 2008; Friedman et al., 2011), contaminant concentrations and contaminant chemical properties such as K_{OW} (Friedman et al., 2011).

Particle-associated and dissolved contaminants that are suspended or released from sediments may be available for uptake by organisms, either through particle uptake or through transport across membranes (Eggleton and Thomas, 2004). Bioavailability and uptake depends on the type, chemical form and partitioning of the contaminant, physico-chemical properties of the sediment and water, and habitat and feeding mode of the organisms (Luoma, 1983). Dredging operations have been shown to enhance uptake of PAHs and metals (Bocchetti et al., 2008) and PAHs and PCBs (Bellas et al., 2007) to caged mussels in the field, but the majority of work in this area is also based overwhelmingly on laboratory studies (Roberts, 2012). The general applicability and relevance of these lab investigations to field conditions is uncertain, since, although there is a large body of data concerning contaminant concentrations in field sediments and organisms from monitoring studies, these data are rarely collected in the context of field measurements of contaminated sediment resuspension, release of contaminants from such sediments, and bioavailability of these contaminants to organisms (Roberts, 2012). Bottom trawling as an agent of contaminated sediment resuspension is particularly under-investigated.

Bottom trawling for fish and invertebrates, mostly for human consumption, is a globally important economic activity. An area half the size of the world's continental shelf is trawled every year (Watling and Norse, 1998), mainly on soft substrates. Most bottom fishing gears disturb the seabed, often deliberately in order to cause benthic organisms to swim up into the net. Despite the well-documented impacts of bottom fishing on benthic communities

---

* Corresponding author.
  E-mail address: clare@ecology.su.se (C. Bradshaw).

(e.g., Bradshaw et al., 2002; Jennings and Kaiser, 1998; Thrush and Dayton, 2002), there is surprisingly little information on the importance of disturbance of the seabed by trawling on resuspension and/or remobilisation of sediment-associated contaminants and nutrients, or on the implications of this for the ecosystem. From the few studies that have been carried out, it is clear that bottom-contacting fishing gears disturb sediments (e.g., Durrieu de Madron et al., 2005; Floderus and Pihl, 1990; O'Neill and Summerbell, 2011; Palanques et al., 2001), disrupt geochemical processes on the seafloor (Falcão et al., 2003; Pilskaln et al., 1998; Trimmer et al., 2005), and increase nutrient efflux (Durrieu de Madron et al., 2005; Falcão et al., 2003; Krost, 1990; Percival et al., 2005; Warnken et al., 2003). Resuspension may also alter pollutants' chemical forms and thus their bioavailability and toxicity (Cotou et al., 2005) and/or enhance the transfer of organic pollutants in the benthic food chain, through the mobilisation of contaminated particles (Charles et al., 2005). Given that bottom trawls disturb the sediment surface to depths of at about 20 cm (Hiddink et al., 2006) and that trawling activity is so extensive, it is highly likely that large quantities of sediment, and potentially contaminants, are resuspended by trawling activities.

This study provides new data to help fill this knowledge gap by quantifying i) the suspension of sediment immediately after the passage of a bottom trawl in a contaminated Norwegian fjord and ii) the release to the bottom water, bioavailability and uptake of sediment-associated contaminants during one month in the same fjord, using semi-permeable membrane devices (SPMDs) and a model marine organism, the blue mussel Mytilus edulis.

2. Methods

2.1. Study site

Eidangerfjord is one of five branches of the Grenlandfjords system in southern Norway. It is a typical fjord with a U-shaped cross-section under water, maximum depth of c. 118 m, a sill at the mouth (at 50 m water depth) and a stratified water column. Residence time for the bottom water is 5–8 months, typically with a stagnation period between May and October (Molvær and Stigebrandt, 1991). In 1951, Norsk Hydro established a magnesium production plant in a neighbouring branch, Frierfjord. As part of the production process, by-products (dioxins and other chlorinated organic contaminants) were released into Frierfjord, leading to high concentrations of dioxins in the Grenlandfjords ecosystem. During the mid-1970s and late 1980s restrictions and improved effluent treatment reduced this contaminant discharge, but contaminant concentrations remained high in water, sediment and biota (Knutzen et al., 2003; Persson et al., 2002; Schlabach et al., 1998). The plant was closed in 2002, but the legacy of contamination remains, particularly in the sediments of Frierfjord, and to a lesser degree in Eidangerfjord. PAHs are also released into the Grenlandfjords, mainly from a ferro-manganese plant (Næs, 1999), though discharges have decreased by 90% since 2000. Due to the continued high contaminant load, Norwegian authorities recommend not to consume eel, herring, mackerel and crabs from Eidangerfjord. However, there is a small prawn fishery in the fjord (1–2 trawls per week). Around 2–4 boats are active, mainly when bad weather prevents them fishing further afield. They use small otter trawls; e.g., the one used in this experiment was a demersal shrimp trawl with two 170 kg, 1.6 × 0.8 m wooden otter boards each with an iron shoe, which were attached to the sweeps by 3–4 m (12 kg) of chain. The trawl door spread was 25 m. The 35 mm mesh net was equipped with a 60 m long groundrope, and the headrope was 48 m long. Fishing boats usually trawl Eidangerfjord in a several km long loop that runs parallel with the fjord sides (Fig. 1b).

2.2. Field experiment to quantify sediment suspension caused by bottom trawling

The experiment was carried out in Eidangerfjord between 2 and 4 June 2008. Trawling took place four times during the experiment (twice on 3 June, twice on 4 June) by the prawn trawler Tine Marlin using standard fishing gear (see Section 2.1) along a c. 1800 m long track. The tracks passed between two pairs of fixed measuring stations (N and S, Fig. 1b), c. 1200 m from each other and with c. 125 m between the paired buoys, where pairs of Aanderaa RCM9 current meters were deployed 2 m above the seabed (Fig. 1a). The instruments measured water current speed and direction and turbidity once a minute during deployment to provide spatio-temporal information on the spread of the sediment plume. To describe the vertical profile of the plume in more detail, turbidity measurements were also taken behind the trawler using three CTDs (one Falmouth CTD and 2 SEACAT Seabird) with

Seapoint turbidity meters deployed from R/V Trygve Braarud and mounted on a Rosette water sampler. Vertical turbidity profiles were taken with the CTDs measuring continuously from just above the seabed to c. 15 m above the seabed, at a distance of c. 500 m from the trawler (c. 350 m from the otter boards; Fig. 1d), directly in, and at 30 m and 60 m to each side of the trawl track (Fig. 1e). These five profiles took just over 10 min to complete. In addition, 27 Rosette water samples were taken close to (mainly within 5 m of) the seabed in parallel with these turbidity measurements and total suspended material (TSM) determined in order to provide data for calculation of TSM from turbidity. The particle size distributions were also measured in seven of these samples using a Coulter counter.

The sinking rate of suspended particles was calculated using Stokes' law:

$$V = \frac{d^2(\rho_s - \rho_w)g}{18\mu}$$

where: V = sinking rate of particles in the water (m s^{-1}); d = particle diameter (m); g = 9.81 m s^{-2}; ρ_s = particle density (kg m^{-3}); ρ_w = water density (kg m^{-3}); μ = dynamic viscosity of water (kg m^{-1} s^{-1}). For field conditions during the experiment (salinity of 34.5, 6 °C), ρ_w is 1027 kg m^{-3}; μ is 1.6·10^{-3} kg m^{-1} s^{-1} ρ_s is taken as 2600 kg m^{-3}.

Basic characteristics of the bottom sediment (% <63 μm by weight, water and total organic carbon content) were measured in the top 2 cm of two sediment cores taken with a Gemini gravity corer (0.005 m^2) at station 2 (Fig. 1b) in May 2009.

2.3. Field experiment to evaluate longer term (1 month) effects of bottom trawling

Semi-permeable membrane devices (SPMDs; from Exposmeter AB) were deployed in the field to measure the amount of dissolved organic contaminants in the water column. The SPMDs were of standard size and design; 92 cm-long and 2.5 cm-wide lay-flat low density polyethylene tubing filled with 1 ml triolein. SPMD site control samplers were used during deployment and retrieval of the samplers to evaluate potential contamination from the air and handling of the devices and to measure the initial concentrations of performance reference compounds (PRCs). To measure the total bioavailable amount (dissolved and particulate) of contaminants in the water column, caged blue mussels, Mytilus edulis, (c. 40 per cage, fresh from Scanfjord AB mussel farm, Lysekil, Sweden) were used. Since these were destined for human consumption, we assume that contaminant levels before deployment were low or below the detection limit.

The SPMDs and mussels were deployed on three ropes placed as close as possible to the trawl tracks in the deep basin of the fjord (water depth 90–100 m) (Fig. 1b and c), with the lower set of samplers c. 1.5 m above the seabed (referred to as BW (bottom water) stations in the following text). It was assumed that these were exposed to sediment resuspended during trawling (see Section 3.1). Another set of samplers (referred to as OW; open water) were placed c. 19 m above the seabed, and were thus not exposed to resuspended sediment, but were well below the halocline/thermocline and the level of the fjord sill. Both OW and BW were in the same water layer within the stratified fjord and therefore exposed to the same oxygen concentration, temperature, salinity, water pressure and water exchange processes. The samplers were left in place from 4 June to 7 July 2008. During this period, one fishing boat was in operation, and trawled on five occasions; 13th, 17th, 20th, 24th June and 2nd July. Water temperature at the sampling depth during this period was constant at c. 6 °C. After one month all samplers were retrieved, but one OW mussel sample was later lost due to handling error. Two sediment samples were taken with a Gemini gravity corer (0.005 m^2) at station 2 (Fig. 1b) in May 2009 and the top 2 cm used for contaminant analysis.

Ideally, similar samplers would have been placed in control areas for comparison. However, suitable control areas with similar physical characteristics, similar contaminant loads, and most importantly, lack of fishing impacts (including export of resuspended sediment from trawled areas) could not be found. We therefore decided to use a 'weight of evidence' approach, focussing on the differences in concentrations between OW and BW and interpreting them in the context of a) the spatio-temporal quantification of suspended sediment (as described in Section 2.2) and b) data from towed SPMDs (Allan et al., 2011) where short-term pre- and post-trawl concentrations are available.

2.4. Analysis of contaminants in samples

SPMDs were analysed for four perdeuterated performance reference compounds (PRCs; acenaphthene-d_{10}, pheneanthrene-d_{10}, fluoranthene-d_{10} and chrysene-d_{10}), polychlorinated dibenzo-p-dioxins and -furans (PCDD/Fs), non-ortho polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs). Sediments were analysed for PCDD/Fs, PCBs and PAHs and blue mussels for PCDD/Fs and PCBs. PCDD/F and PCB analyses were conducted at the Norwegian Institute for Air Research (NILU) while those for PAHs and PRCs from the SPMDs were done at the Norwegian Institute for Water Research (NIVA). SPMDs were extracted by dialysis with hexane (2 × 24 h extraction). Extracts were reduced and split into two fractions for analyses of a) PAHs and PRCs and b) PCDD/Fs and PCBs.
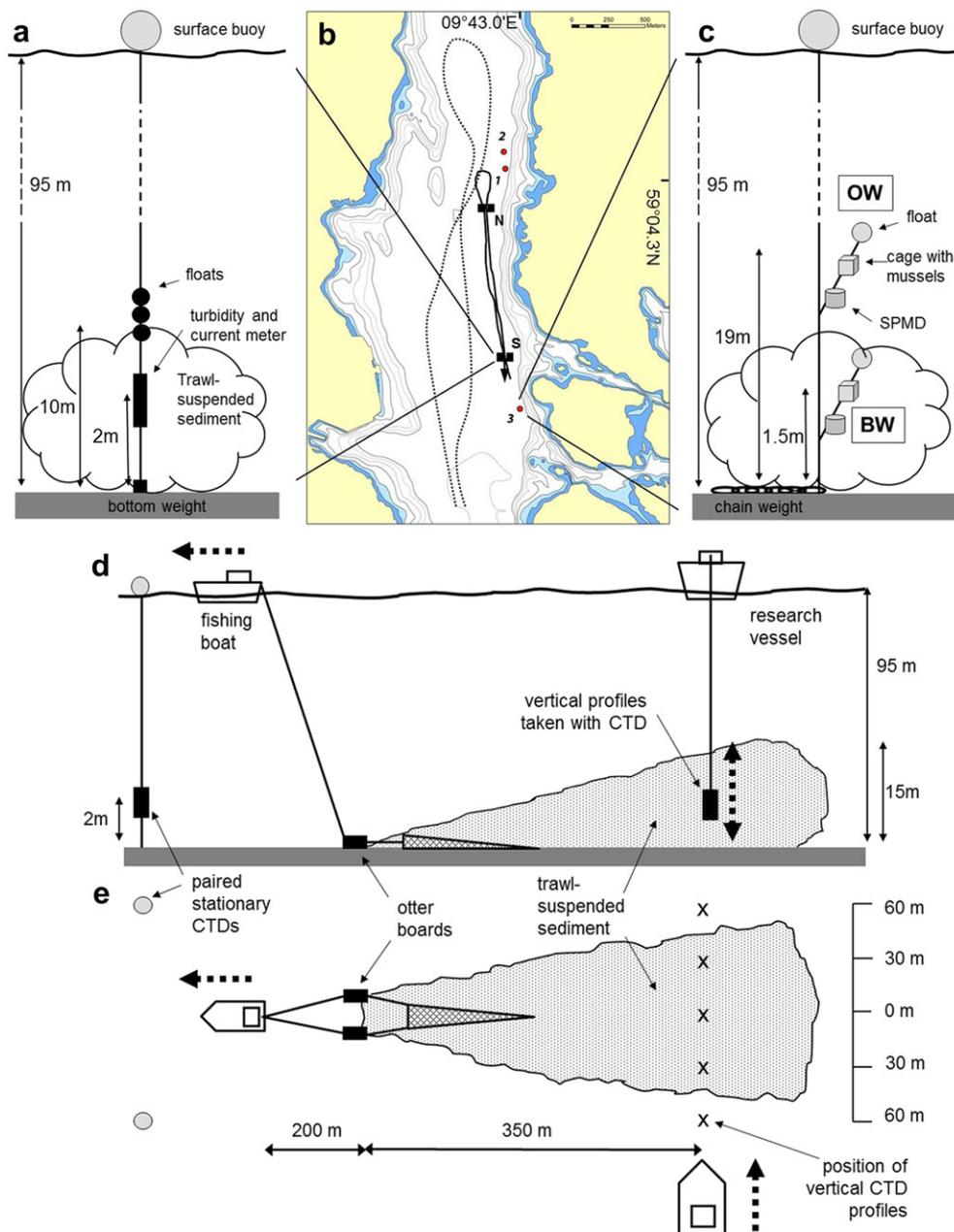
Fig. 1. Eidangerfjord (b) showing the positions of the two paired stations for measurement of suspended sediment (squares: North and South, details in a)), the path of the trawler during the experiment (solid line), and the three sampling stations where SPMD and mussels were deployed (circles; details in c)) and a typical bottom trawling track during June 2008 (dotted line). BW = near bottom samplers, OW = open water samplers. d) shows how spatial measurements of turbidity in the sediment plume were taken using towed CTDs on the research vessel; e) is the same, shown from above.

2.4.1. PAHs and PRCs

Sediment extractions were with accelerated solvent extraction using dichloromethane and cyclohexane (1:1 vol:vol) at 100 °C and 2000 psi. Gel permeation chromatography was used for clean-up of extracts from sediments and SPMDs which were then analysed by gas chromatography/mass spectrometry. For more details, see Harman et al. (2008) (SPMDs) and Ruus et al. (2005) (sediments).

2.4.2. PCDD/Fs and non-ortho PCBs

Sediment samples were dried prior to analysis, while blue mussels were frozen directly after retrieval and soft tissue samples of several individuals later pooled for analysis. Extraction of sediments and mussels was by Soxhlet extraction with acetone then toluene. Extracts were concentrated and the solvent exchanged to cyclohexane/dichloromethane (1/1). Sample clean-up was with two KOH-silica filled columns and eluted onto a final AX21 activated charcoal column. Elution of PCDD/Fs and non-ortho PCBs was done by toluene flushing of the final column in reversed flow direction. Further clean-up was with sulphuric acid-coated silica and

aluminium oxide. Analysis was performed using a gas chromatograph linked to a high resolution Micromass AutoSpec Ultima mass spectrometer. Full details can be found in Knutzen et al. (2003) and Ruus et al. (2006).

2.5. Data treatment

Concentrations of individual compounds were often below the limit of detection (LOD), especially for the SPMDs and thus also water concentrations (e.g. 81%, 18%, 42% and 39% of samples <LOD for PCDDs, PCDFs, PCBs and PAHs, respectively). We could not accurately estimate these concentrations from other data and simplified standard estimations such as setting <LOD data to zero, half-LOD or LOD would have led to inaccurate or misleading treatment averages. Thus, despite substantially reducing the dataset, we only used data based on measured values (>LOD).

Water concentrations of different organic compounds were calculated from the concentrations measured in the whole SPMD, together with the average PRC dissipation rate for acenaphtene-d_{10} and pheneanthrene-d_{10} according to the

polynomial model given by Huckins et al. (2006). Sampling rates were 2.9 (s.d. 0.5) and 4.6 (0.7) L d^{-1} for acenaphthene and phenanthrene, respectively (Allan et al., 2011). PRC dissipation was similar for SPMDs at OW and BW, enabling reliable comparison between the two depths.

Concentration of the contaminants in sediment porewater (C_{pw}) was calculated from sediment concentrations (C_{sed}) and mass fraction organic carbon (f_{OC}) of the sediment using the organic matter partitioning model (Chiou et al., 1983; Karickhoff, 1981): $C_{pw} = C_{sed}/(f_{OC} \times K_{OC})$. K_{OC}, the particulate organic carbon normalised solid–water distribution coefficient, was taken to be $0.33 \times K_{OW}$ (Seth et al., 1999). We chose not to use a model including black carbon, since a) there is uncertainty whether this improves predictions in this fjord system (Cornelissen et al., 2010; Persson et al., 2005) and b) we did not ourselves measure BC; using values from other studies, which vary from 0.4 to 7.3 mg BC/g dw sediment (Cornelissen et al., 2010; Persson et al., 2005), would have introduced further uncertainty.

Contaminant concentrations in mussels were calculated on a wet weight basis and a lipid basis. The latter used the assumption that shell-free wet weight comprises 2% lipid (National Food Administration, Sweden). The condition index of the blue mussels was calculated as the ratio of shell length: g dw soft tissues.

Statistical analysis to determine if contaminant concentrations in water or mussels differed between OW and BW was done using the programme PRIMER 6 with PERMANOVA (Anderson, 2001, 2005; McArdle and Anderson, 2001). Permutation of residuals was under a reduced model with 10 000 permutations. Only congeners for which data was available for all replicates were used. Canonical analysis of principal coordinates (CAP) was used to evaluate potential associations, and proportional significance of each contaminant was assessed using correlation analysis (two–tailed significance of the correlation coefficient r) of CAP scores.

A multiple regression was also performed (in Statistica 9) to ascertain which of the variables sediment concentrations (C_{sed}, a proxy for $C_{particles}$), porewater concentration (C_{pw}) or water concentration (C_{OW} or C_{BW}) best explained contaminant concentrations in mussel tissues at OW and BW. Only the congeners for which >LOD data was available for all sample types were used.

3. Results

3.1. Quantification of sediment suspension caused by trawling

The bottom sediment was a fine organic-rich mud; the <63 μm fraction comprised 88.5% (±3.5 s.e.) by dry weight, water content was 64.5% (±1.5) by weight and total organic content was 29.4 (±0.45) μg C mg dw^{-1}. Total suspended matter in the suspended plume was strongly correlated with turbidity: TSM (mg dw l^{-1}) = 1.1 × turbidity (NTU) −0.25 ($R^2 = 0.98$). Measurements of turbidity taken pre-trawling (2 June 2008) indicated that background levels of total suspended matter (TSM) were around 0.5–1.0 mg dw L^{-1} in the surface brackish water, 0.1–0.2 mg dw L^{-1} at mid-depths and slightly over 0.5 mg dw L^{-1} in bottom waters, with a sharp increase to c. 1.3 mg dw L^{-1} directly over the seabed (Fig. 2). Directly after the passage of the trawl, increased TSM was recorded at the fixed measuring stations (2 m above the seabed); up to 6 mg dw L^{-1} (turbidity 5.7 NTU) at the western station (Fig. 3). During the 4 trawling events (7 passes through the gates) over 2 days, TSM of up to 35 mg dw L^{-1} was measured in bottom waters to a depth of 15–18 m above the sediment (Fig. 2). This is equivalent to around 70 times greater than background concentrations. Depending on the direction of the currents when the trawl passed through the 'gate', this signal was more or less strong and persisted for different lengths of time. The shape of the suspended sediment plume clearly reflected the construction of the trawl, with two side plumes created by the otter boards (and subsequently spread outwards and upwards by turbulence and by weak tidal currents (average 2–3 cm s^{-1})) and a low-lying central area probably caused by the net and ground ropes (Fig. 4). Only 30–60 min after a single trawl pass the plume thus had a horizontal extent of 120–150 m and was 15–18 m high, ie. along the whole 1.8 km track it occupied a water volume of c. 3–5 million m^3. Using a conservative estimate of an increase in TSM of 1–3 mg dw sediment L^{-1} throughout the suspended plume, the total amount of sediment resuspended along a single 1.8 km trawl track was thus c. 3–15 t dw (average 9 t dw).
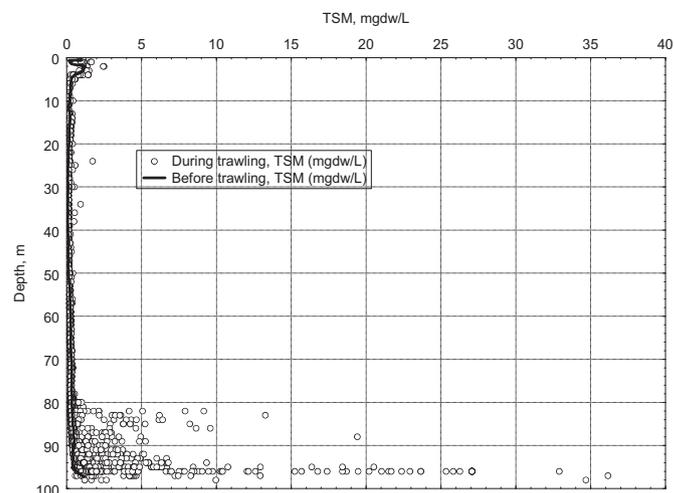
Fig. 2. Summary of total suspended matter (TSM) (mg dw L^{-1}) data from 4 June 2008, calculated from turbidity measurements taken with the mobile CTD after the trawler had passed. The cloud of suspended sediment is visible up to 15–18 m above the seabed (which is at c. 98 m). The solid black line shows TSM one day prior to trawling.

The particle size distribution varied little between the water samples. 95% of the particles had a diameter <10 μm and more than 50% <4 μm (Fig. 5). The sinking speed of the particles was estimated to range from <0.2 m to >108 m day^{-1} (Fig. 5). Given that most particles are in the smaller end of the size spectrum, and that sediment was suspended to 15–18 m above the seabed, this suggests that most particles will take many days to settle out (Fig. 5). This is consistent with the sediment plume still being clearly detectable on the second day of measurements.

3.2. Contaminant concentrations and congener profiles

Contaminants were detected in all the sample types (Fig. 6), though some congeners were below <LOD in some samples (Table 1). All dioxins except 1,2,3,4,6,7,8-HpCDD, several furans and nearly half the PAHs were below the LOD in the SPMDs (<3–6 pg SPMD^{-1} for PCDD/Fs, <5 ng SPMD^{-1} for PAHs). The high amounts of OCDF, 1234678-HpCDF, 1234789HpCDF and 123478/123479-
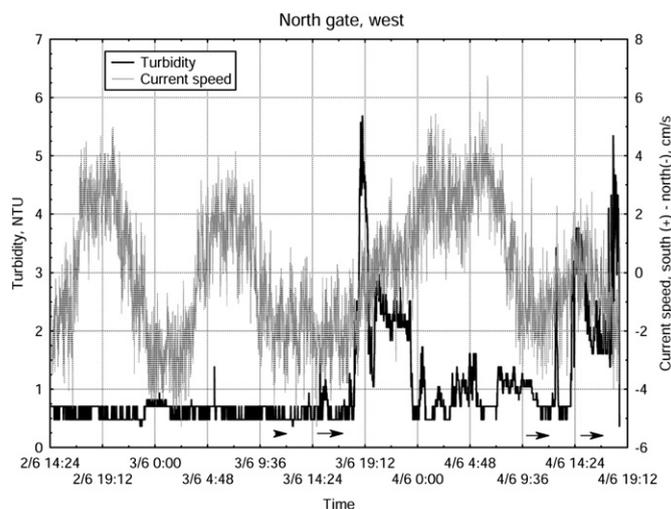
Fig. 3. Turbidity (NTU) and current speed (cm s^{-1}) in a north–south direction, measured at the northern station (western gate), 2–4 June 2008. Arrows indicate trawling events. Turbidity (NTU) = [TSM (mg dw L^{-1}) + 0.25]/1.1.
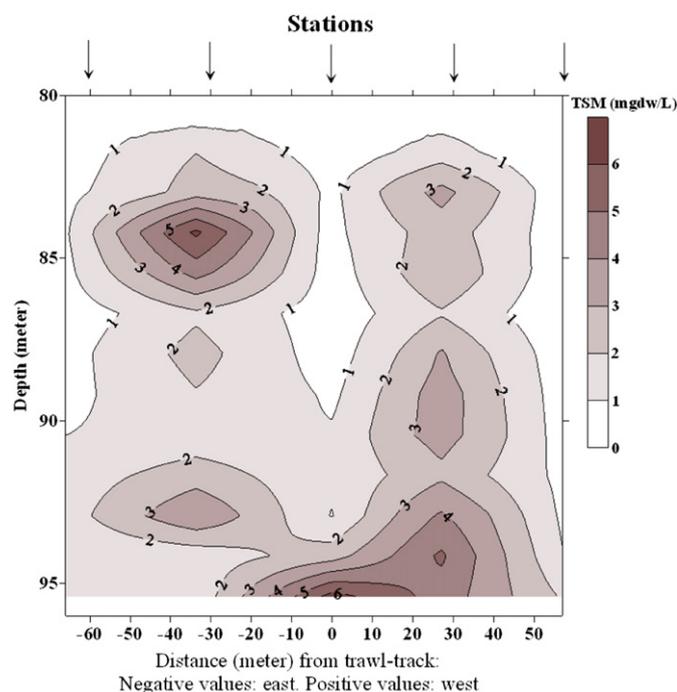
Fig. 4. Summary of five depth profiles (marked by vertical arrows) of total suspended matter made with towed CTDs across the trawl track after the single passage of the trawl, 4 June. A vertical cross-section across the trawl track (centred at 0) is shown, with a 15 m thick and >120 m wide sediment plume. The seabed is at the bottom of the figure.

HxCDF in the sediment reflected the original emission profile from the magnesium plant. This higher relative abundance of higher molecular weight/K_{ow} PCDD/Fs in sediment contrasts with the calculated concentrations of contaminants in the porewater (C_{pw}),
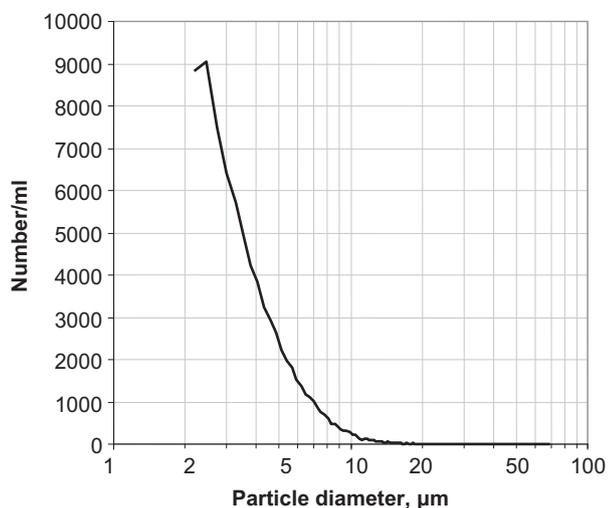
	particle diameter (μm)	2	5	10	15	20	50
sinking rate	m h^{-1}	0.01	0.05	0.18	0.41	0.72	4.5
	m day^{-1}	0.2	1.1	4.3	9.8	17.3	108
time to sink 15 m	days	87	13.8	3.5	1.5	0.9	0.1

Fig. 5. Particle size distribution in a water sample taken 600 m behind the prawn trawler, 4 June 2008 (water depth 92–98 m). Below are the sinking rates for selected particle sizes, calculated using Stokes' Law, and the calculated time taken for each size class to settle to the seabed from a 15 m high sediment plume.

where lower molecular weight/K_{ow} molecules were relatively more abundant for each group of chemicals (PCDD/Fs, non-ortho PCBs, PAHs) (Fig. 6), as would be expected from the partitioning coefficients of the various congeners. Note that C_{pw} for some congeners may have been overestimated due to using the OMP model that does not account for adsorption to black carbon (e.g., Lambert et al., 2011).

There was an overall significant difference between BW and OW water contaminant concentrations (PERMANOVA pseudo-$F_{1,5} = 6.6385$, $p = 0.0148$). Subsequent correlation analysis of CAP scores against variables for the first order CAP axes revealed that this difference was explained by higher concentrations of the high K_{ow} PCDD/Fs in the BW samples than the OW samples, while for lower K_{ow} PAHs and non-ortho PCBs, concentrations were similar or higher in OW samples ($r > 0.811$, df $= 4$, $p < 0.05$).

In general, concentrations in the water (C_w) were higher when sediment concentrations (C_{sed}) were higher (Fig. 7). However, there were congener-specific differences; PAHs had relatively higher C_w for a given C_{sed}. For PCDD/Fs, there was a clear difference between the BW and OW samples, with BW having relatively higher C_w than OW samples for a given sediment concentration (Fig. 7). The relationship may also reflect the non-linearity of sorption to black carbon in the sediments (e.g., Accardi-Dey and Gschwend, 2002).

BW mussels had higher concentrations of PCDD/Fs than the OW mussels (Fig. 6a, Table 1), with the opposite trend for non-ortho PCBs, however, this was not significant (PERMANOVA pseudo-$F_{1,4} = 0.89989$, $p = 0.4399$), possibly due to high between-replicate variability. Multiple regressions of concentrations in mussels (C_{muss}) against concentration in sediment, porewater and water (C_{sed}, C_{pw} and C_w, respectively) were significant for both OW (adjusted $R^2 = 0.872$, $p = 0.00018$) and for BW (adjusted $R^2 = 0.937$, $p = 0.00001$). Correlations with all variables were stronger in the bottom water than the overlying water (Table 2). Sediment concentrations were the best predictor of C_{muss} (for the 12 congeners for which >LOD data was available), both in the bottom water and the overlying water (Table 2). The mussels' condition index was similar for OW and BW (0.11 ± 0.03, 0.12 ± 0.03 respectively), indicating that their health was comparable between the two sampling depths.

4. Discussion

4.1. Suspension of sediment by trawling

A single pass of a bottom trawl in Eidangerfjord suspended a plume of sediment 15–18 m above the seabed and 120–150 m wide. Total suspended matter (TSM) in the plume reached around 70 times background levels, ie. TSM up to 35 mg dw L^{-1}. This is comparable to values measured in the few other published experimental trawling studies (Table 3). Due to their small size, most of the particles will take several days to settle out from a 15 m high plume, even if flocculation occurs. It is thus likely that there is a semi-permanent layer of elevated TSM in the fjord. Since trawling occurred five times during the one month of sampling in the second part of our field study, and water export from the fjord bottom water is slow (months, see below), the BW samplers were probably surrounded more or less continuously by suspended particles.

4.2. Relevance of trawl-induced sediment resuspension for the fjord as a whole

Trawl-induced sediment resuspension may comprise a large part of the total sediment resuspension in the Grenlandfjords. Saloranta et al. (2006) and Persson et al. (2006) have previously estimated gross annual sedimentation rates as 2700 and 1968 g dw m^{-2} y^{-1}, respectively. Their estimates implicitly include
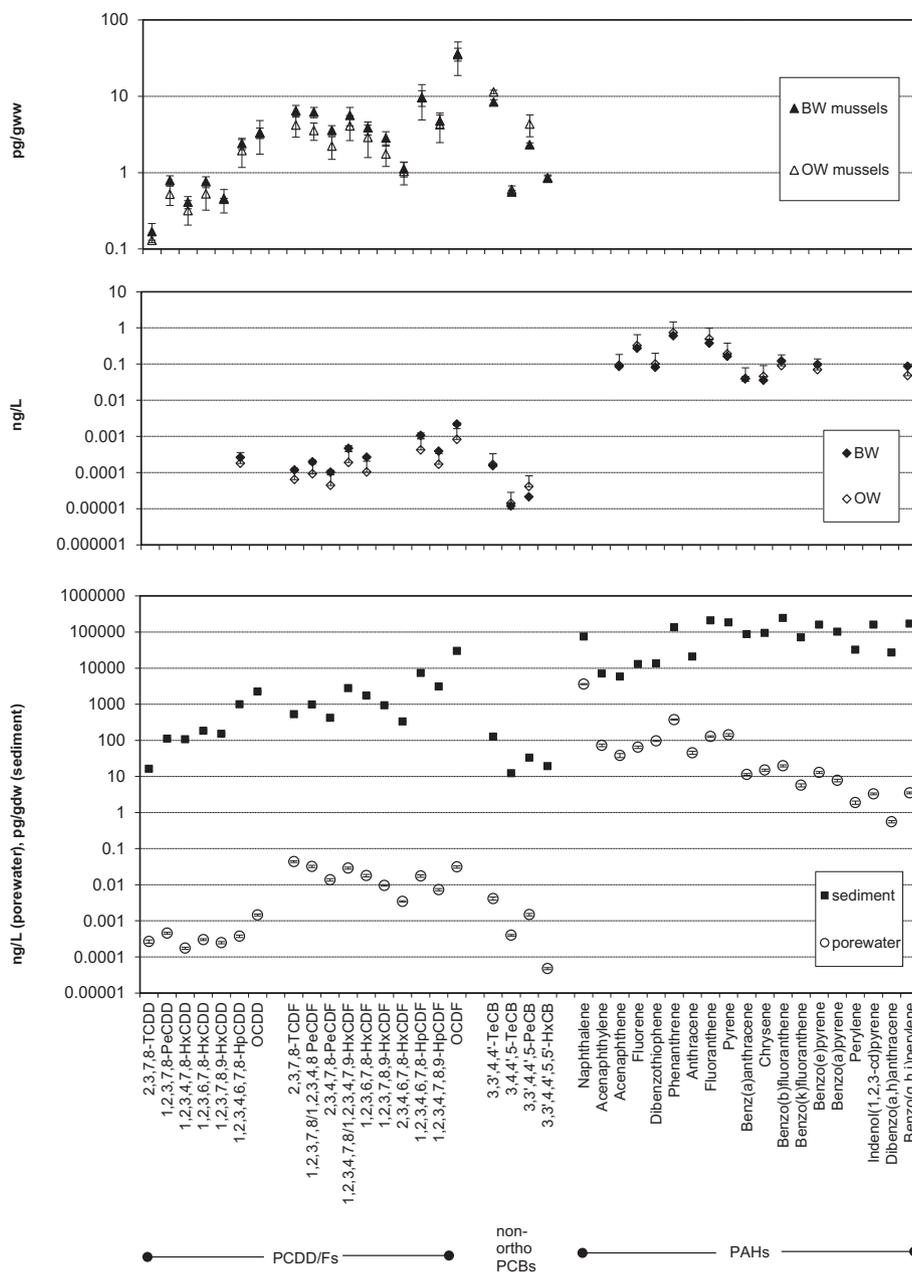
Fig. 6. Concentrations of the different congeners in a) mussels in bottom water (pg gww^{-1},▲) and in overlying water (pg gww^{-1}, △), b) bottom water (ng L^{-1}, ◆) and overlying water (ng L^{-1}, ◆), and c) sediment (pg g dw^{-1}, ■) and porewater (ng L^{-1}, ○). Only concentrations >LOD are shown. Data are means ± standard error; for details see Methods and Table 1. Note the different scales and that no PAH data is available for mussels.

resuspension by trawling, since they are based on gross deposition estimates near the seabed. We estimated that the average amount of sediment resuspended along a 1.8 km trawl track was 9 t dw. With a track width equal to the width between the otter boards (25 m) this gives a sediment resuspension of 200 g dw m^{-2}. Resuspension from a single trawl pass is thus equivalent to 7–10% of annual gross sedimentation in the Eidangerfjord.

Assuming that the resuspended particles have the same contaminant concentration as bottom sediment (mean 52106 pg g dw sediment^{-1} (ΣPCDD/Fs), s.d. 5276, n = 2), we can estimate the total amount of PCDD/Fs resuspended by trawling as 261 mg per 1 km seabed trawled or 78 g PCDD/F y^{-1}. These estimates take into account neither differences in particle quality (e.g. particle size, organic content, black carbon content) nor contaminant partitioning between bottom sediment and suspended

sediment. However, they give a valuable indication of the scale of contaminant resuspension by trawling.

Water currents and turnover rates are slow in the Grenlandfjord system, especially in the deep water where sediment is suspended by trawling. Over a period of 2 days in Eidangerfjord we measured average bidirectional (tidal) current speeds of around 2–3 cm s^{-1} (at c. 2 m above the seabed) and Persson et al. (2006) estimated an outflow rate from the deep water of 8 m^3 s^{-1}. Deep water residence time in the outer fjords is estimated at 5–8 months (Molvær and Stigebrandt, 1991). Thus, most of the resuspended particles and associated contaminants will not be flushed out, but rather remain in the fjord bottom waters. Given the small particle size and slow sinking speeds already described, it is likely that a semi-permanent contaminated sediment suspension will be created above the seabed by trawling activity. A lack of contaminant export is

Table 1
Summary of raw data (averages ± s.e., (n)). Values <LOD were removed before calculation of averages. Blank cells indicate no data.

Compound	K_{ow}	Sediment (pg g dw⁻¹)	Water (ng L⁻¹)		SPMD (pg SPMD⁻¹)		Mussel (pg gww⁻¹)	
			OW	BW	OW	BW	OW	BW
Dioxins								
2,3,7,8-TCDD	6.80	16.35 ± 1.75 (2)					0.13 (1)	0.17 ± 0.05 (2)
1,2,3,7,8-PeCDD	7.40	111 ± 6 (2)					0.52 ± 0.15 (2)	0.79 ± 0.12 (3)
1,2,3,4,7,8-HxCDD	7.80	107 ± 7 (2)					0.32 ± 0.11 (2)	0.41 ± 0.07 (3)
1,2,3,6,7,8-HxCDD	7.80	184.5 ± 11.5 (2)					0.53 ± 0.20 (2)	0.76 ± 0.12 (3)
1,2,3,7,8,9-HxCDD	7.80	152 ± 11 (2)					0.46 (1)	0.45 ± 0.15 (2)
1,2,3,4,6,7,8-HpCDD	8.44	995 ± 79 (2)	1.79E-04 (1)	2.61E-04 (1)	6.24 (1)	8.58 (1)	1.95 ± 0.77 (2)	2.41 ± 0.41 (3)
OCDD	8.20	2241 ± 129 (2)					3.27 ± 1.53 (2)	3.32 ± 0.52 (3)
Furans								
2,3,7,8-TCDF	6.10	530.5 ± 29.5 (2)	6.38E-05 ± 5.31E-06 (3)	1.17E-04 ± 1.32E-05 (3)	10.20 ± 0.78 (3)	17.26 ± 4.10 (3)	4.21 ± 1.28 (2)	6.46 ± 1.13 (3)
1,2,3,7,8/1,2,3,4,8 PeCDF	6.50	986.5 ± 69.5 (2)	9.14E-05 ± 9.40E-06 (3)	1.95E-04 ± 2.36E-05 (3)	12.21 ± 0.58 (3)	24.15 ± 6.15 (3)	3.55 ± 0.91 (2)	6.19 ± 0.95 (3)
2,3,4,7,8-PeCDF	6.50	420.5 ± 32.5 (2)	4.39E-05 ± 1.05E-05 (2)	1.02E-04 ± 1.29E-05 (2)	5.56 ± 1.80 (2)	12.47 ± 5.46 (2)	2.22 ± 0.73 (2)	3.59 ± 0.53 (3)
1,2,3,4,7,8/1,2,3,4,7,9-HxCDF	7.00	2788.5 ± 171.5 (2)	1.90E-04 ± 2.95E-05 (3)	4.66E-04 ± 7.75E-05 (3)	19.08 ± 2.87 (3)	42.91 ± 12.39 (3)	4.09 ± 1.47 (2)	5.62 ± 1.53 (3)
1,2,3,6,7,8-HxCDF	7.00	1733 ± 142 (2)	1.03E-04 ± 1.45E-05 (3)	2.62E-04 ± 1.18E-05 (3)	10.41 ± 1.54 (3)	24.37 ± 5.83 (3)	2.88 ± 1.31 (2)	3.87 ± 0.74 (3)
1,2,3,7,8,9-HxCDF	7.00	927.5 ± 41.5 (2)					1.76 ± 0.55 (2)	2.84 ± 0.60 (3)
2,3,4,6,7,8-HxCDF	7.00	334.5 ± 15.5 (2)					1.04 ± 0.34 (2)	1.12 ± 0.24 (3)
1,2,3,4,6,7,8-HpCDF	7.64	7384 ± 560 (2)	4.22E-04 ± 6.67E-05 (3)	1.07E-03 ± 1.76E-04 (3)	26.96 ± 3.96 (3)	62.73 ± 19.35 (3)	9.55 ± 4.65 (2)	9.63 ± 2.26 (3)
1,2,3,4,7,8,9-HpCDF	7.64	3089 ± 220 (2)	1.68E-04 (1)	3.88E-04 (1)	10.86 (1)	23.67 (1)	4.24 ± 1.77 (2)	4.75 ± 0.94 (3)
OCDF	8.00	30105.5 ± 2132.5 (2)	8.21E-04 ± 8.35E-05 (2)	2.19E-03 ± 1.87E-04 (2)	37.13 ± 7.19 (2)	96.66 ± 40.00 (2)	35.08 ± 16.32 (2)	35.80 ± 6.87 (3)
non-ortho-PCBs								
3,3′,4,4′-TeCB	6.50	127 ± 12 (2)	1.66E-04 ± 3.45E-05 (3)	1.53E-04 ± 7.43E-06 (3)	22.29 ± 4.55 (3)	18.86 ± 0.43 (3)	11.39 ± 0.59 (2)	8.40 ± 0.64 (3)
3,4,4′,5-TeCB	6.50	12.2 ± 0.6 (2)	1.41E-05 ± 2.24E-06 (2)	1.19E-05 ± 2.02E-06 (2)	1.77 ± 0.44 (2)	1.44 ± 0.03 (2)	0.60 ± 0.07 (2)	0.56 ± 0.03 (3)
3,3′,4,4′,5-PeCB	6.36	33.15 ± 2.55 (2)	4.06E-05 (1)	2.10E-05 (1)	5.87 (1)	2.88 (1)	4.33 ± 1.37 (2)	2.32 ± 0.14 (3)
3,3′,4,4′,5,5′-HxCB	7.62	19.25 ± 1.25 (2)					0.85 ± 0.05 (2)	0.85 ± 0.09 (3)
PAHs								
Naphthalene	3.33	7.45E+04 ± 2.50E+03 (2)						
Acenaphthylene	4.00	7.05E+03 ± 5.50E+02 (2)						
Acenaphthene	4.20	5.85E+03 ± 7.50E+02 (2)	9.21E-02 ± 6.41E-03 (3)	8.59E-02 ± 4.86E-03 (3)	7.00E+03 ± 6.66E+02 (3)	6.30E+03 ± 5.04E+02 (3)		
Fluorene	4.32	1.30E+04 ± 1.00E+03 (2)	3.23E-01 ± 2.02E-02 (3)	2.73E-01 ± 1.70E-02 (3)	2.93E+04 ± 2.85E+03 (3)	2.37E+04 ± 2.67E+03 (3)		
Dibenzothiophene	4.16	1.35E+04 ± 5.00E+02 (2)	9.98E-02 ± 9.85E-04 (3)	8.13E-02 ± 3.85E-03 (3)	7.10E+03 ± 3.21E+02 (3)	5.60E+03 ± 4.48E+02 (3)		
Phenanthrene	4.57	1.35E+05 ± 5.00E+03 (2)	7.31E-01 ± 7.33E-03 (3)	6.08E-01 ± 2.72E-02 (3)	8.87E+04 ± 6.12E+03 (3)	6.97E+04 ± 7.36E+03 (3)		
Anthracene	4.68	2.10E+04 ± 2.00E+03 (2)						
Fluoranthene	5.23	2.10E+05 ± 1.00E+04 (2)	4.90E-01 ± 5.24E-03 (3)	3.75E-01 ± 1.56E-02 (3)	8.60E+04 ± 7.57E+03 (3)	6.07E+04 ± 1.00E+04 (3)		
Pyrene	5.13	1.85E+05 ± 1.50E+04 (2)	1.89E-01 ± 1.66E-02 (3)	1.62E-01 ± 8.89E-03 (3)	3.23E+04 ± 2.60E+03 (3)	2.57E+04 ± 2.85E+03 (3)		
Benz(a)anthracene	5.90	8.75E+04 ± 5.50E+03 (2)	3.91E-02 ± 2.11E-04 (3)	3.93E-02 ± 6.12E-03 (3)	6.70E+03 ± 6.35E+02 (3)	6.17E+03 ± 8.29E+02 (3)		
Chrysene	5.81	9.40E+04 ± 6.00E+03 (2)	4.55E-02 ± 2.49E-03 (3)	3.55E-02 ± 2.47E-03 (3)	7.90E+03 ± 6.03E+02 (3)	5.67E+03 ± 5.84E+02 (3)		
Benzo(b)fluoranthene	6.11	2.45E+05 ± 1.50E+04 (2)	8.97E-02 ± 6.06E-03 (3)	1.20E-01 ± 9.68E-03 (3)	1.43E+04 ± 1.20E+03 (3)	1.77E+04 ± 2.65E+03 (3)		
Benzo(k)fluoranthene	6.11	7.10E+04 ± 7.00E+03 (2)						
Benzo(e)pyrene	6.11	1.60E+05 ± 1.00E+04 (2)	6.93E-02 ± 3.41E-03 (3)	9.76E-02 ± 7.73E-03 (3)	1.11E+04 ± 9.00E+02 (3)	1.43E+04 ± 2.26E+03 (3)		
Benzo(a)pyrene	6.13	1.02E+05 ± 8.50E+03 (2)						
Perylene	6.25	3.25E+04 ± 3.50E+03 (2)						
Indenol(1,2,3-cd)pyrene	6.70	1.60E+05 ± 1.00E+04 (2)						
Dibenzo(a,h)anthracene	6.70	2.70E+04 ± 2.00E+03 (2)						
Benzo(g,h,i)perylene	6.70	1.70E+05 ± 1.00E+04 (2)	4.77E-02 (1)	8.70E-02 (1)	5.80E+03 (1)	1.00E+04 (1)		
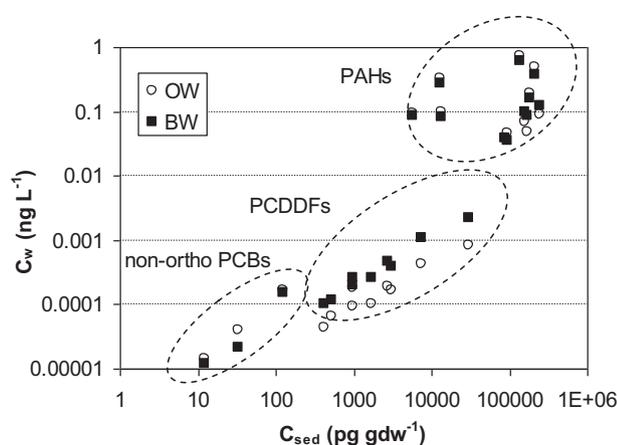
Fig. 7. Relationship between concentration of organic contaminants in the sediment and in the water 1.5 m (BW) and 19 m (OW) above the seabed. Only data where concentrations were >LOD are included. Data are averages of replicate samples (*n* varies between 1 and 3 depending on available data). Note the log scale on both axes.

supported by the fact that the concentrations in sediment measured in this study were of the same order of magnitude as samples taken nearby more than ten years previously (Station 16b in Næs, 1999) indicating minimal net removal and/or degradation of contaminants from the fjord sediments.

4.3. Elevation of contaminant concentrations in bottom waters

A clear decrease in concentration was seen for most PCDD/Fs from near bottom water (BW) to overlying water (OW), suggesting that the sediment was the main source of these dissolved contaminants. However, the gradient between the BW and OW was much less than expected due to diffusion and advection alone. For example, the relative difference between porewater and BW concentrations in our study was orders of magnitude lower than that measured for the same congeners by Persson et al. (2006) in the adjacent (untrawled) Frierfjord, indicating relatively higher bottom water concentrations in our study.

Suspension of contaminated sediment, for example by bottom trawling, will also release contaminated porewater and cause changes to the partitioning of sediment-bound contaminants, potentially increasing their release to the water phase. In a parallel study, concentrations of freely dissolved PAHs were increased by up to $1.6\times$ (Allan et al., 2011) and PCDD/Fs by an order of magnitude during the 5 h immediately after trawling, and on a scale of only metres from the trawl. Over the larger spatial and temporal scales (tens of metres, 1 month) of the second part of this study, the concentration gradient was less pronounced, as would be expected. However, clear differences were still seen; for example, OW had 1.5—2.9 times lower concentrations of PCDD/Fs and 1.3—1.8 times less benzo(b)fluoranthene, benzo(e)pyrene and benzo(ghi)perylene than BW where trawling had an influence. Although we do not have comparative data from an untrawled site, these data,

Table 2
Results of multiple regressions to determine best predictor variables of mussel tissue concentrations. Beta values show the relative contribution of each variable to the prediction. * indicates statistical significance ($p < 0.05$).

Mussels in overlying water (OW)				Mussels in bottom water (BW)			
Variable	*p* level	Correlation	Beta value	Variable	*p* level	Correlation	Beta value
C_{OW}	0.56	0.92	0.23	C_{BW}	0.57	0.93	−0.19
C_{sed}	0.09	0.94	0.74	C_{sed}	0.01*	0.97	1.12
C_{pw}	0.88	0.30	−0.02	C_{pw}	0.20	0.43	0.11

Table 3
Total suspended matter (TSM) concentrations caused by trawling compared to background values. All TSM values are assumed to be in mg dry weight, though this is not always explicitly stated in the source reference.

TSM caused by trawling (mg L^{-1})	Background TSM (mg L^{-1})	Reference
Maximum c. 6	2—3	Palanques et al. (2001)
15—70	1.5—2.5	Durrieu de Madron et al. (2005)
100—500	Not given	Schubel et al. (1979)
>250	50—100	Schoellhamer (1996)
225—350	10—25	Dellapenna et al. (2006)
1280	50	Dounas (2006)
660—7000	2.5—10	O'Neill and Summerbell (2011)
35	**c. 0.5**	**This study**

together with the clearly elevated turbidity in bottom waters caused by trawling, the persistence of this turbidity, the contaminated nature of the bottom sediment, and the repeated trawling during the experimental exposures all suggest that the contamination gradient was at least partly due to trawl-induced suspension of contaminated sediment.

Less hydrophobic, lower molecular weight congeners had similar concentrations in OW and BW. This was also the case in the short-term study that compared water concentrations before and directly after trawling (Allan et al., 2011). This could either indicate that these congeners are also entering the fjord from the atmosphere or surface run-off, or that these more mobile contaminants were already elevated in bottom waters previous to the field experiment, possibly due to continuous trawling activity (ie. there was a weaker concentration gradient between water and sediment).

4.4. Uptake of contaminants from the water column by exposed marine organisms

The gradient in concentrations of many dissolved contaminants in the water column was mirrored by a depth gradient in tissue concentrations of PCDD/Fs in field-exposed blue mussels; up to three times higher in BW mussels than OW mussels at the same sampling station. The contaminants in the surrounding environment were therefore clearly bioavailable and taken up by these organisms.

If mussels took up contaminants solely through passive diffusion, like SPMDs, one would expect C_{lipid} to be around 4% of that of C_{SPMD}, since mussels of the size used here have an approximate gill area of 17.5 cm^2 (Bayne, 1976); 4% of an SPMD (surface area 460 cm^2). However, C_{lipid} are an order of magnitude higher in mussels than for SPMDs and this can be attributed to the fact that mussels are active filter-feeders, filtering c. 2.5 L h^{-1} mussel^{-1} for mussels of the size used in this study (Bayne, 1976), or c. 1750 L water per mussel during the study exposure. Not only will this substantially increase a mussel's exposure to dissolved contaminants, compared to an SPMD, but at the deep stations (BW) there were also elevated concentrations of particles in the water, probably due to trawling. Since there was a more or less constant supply of contaminated particles and water to the deep water, it is unlikely that there was any depuration from SMPDs or mussels during this time, and these contaminants are slow to be biodegraded by mussels (Gilek et al., 1997; Hektoen et al., 1994). Multiple regression analysis also supported the suggestion that suspended sediment particles are the most likely source of contaminants by indicating that sediment concentrations were the best predictor of mussel concentrations (Table 2), as does the work of Axelman et al. (1999). Body burdens of organic contaminants in caged mussels have been shown to increase as a result of sediment resuspension by human activities, e.g. dredging of harbours (Bellas et al., 2007; Bocchetti et al., 2008).

Blue mussels do not naturally occur at these depths in the fjord, but were used here as tried and tested field monitoring organisms (e.g., Axelman et al., 1999; Bellas et al., 2007; Bocchetti et al., 2008; Booij et al., 2006). However, by using mussels as indicators for bioavailability we show that suspended and/or desorbed contaminants are potentially bioavailable to naturally-occurring fauna, particularly suspension- and filter-feeders and benthic species like prawns and demersal flatfish.

4.5. Potential effects on organisms and implications for humans

There is very limited data on the toxicity of PCDD/Fs to marine organisms with which to compare our results. However, field-exposed mussels that have bioaccumulated PAHs have been shown to have increased levels of various cellular biomarkers (Bellas et al., 2007; Bocchetti et al., 2008; Giannapas et al., 2012) and the same can be expected for PCDD/Fs. As a gross approximation of toxicity, we can estimate the Hazard Quotient (HQ) for the blue mussels in our study (Jones et al., 1999; Micheletti et al., 2007). The HQ is the body concentration in toxic equivalents (TEQ) divided by the Internal No Effect Concentration (INEC); values >1 indicate potential toxicity. INEC values for PCDD/Fs and mussels are not available, but using the range 0.057–0.699 ng TCDD g lipid^{-1} for fish (Steevens et al., 2005), we get HQs of 8–88 for OW mussels and 12–129 for BW mussels.

EC Regulation 1881/2006 (EC, 2006) advises a dietary maximum of 4 pg gww^{-1} (ΣPCDD/Fs) for a range of marine organisms including blue mussels. At 5.36 ± 1.61 pg gww^{-1} the BW mussels are over this limit after only one month of exposure. The recommended Tolerable Daily Intake (TDI) for dioxin-like compounds is 1–4 pg TEQ/kg body weight/day (van Leeuwen et al., 2000; WHO, 1998). For an average adult weighing 70 kg this is equivalent to 70–280 TEQ day^{-1}. With TEQs for ΣPCDD/Fs of 3.25 ± 1.24 and 5.36 ± 1.61 pg gww^{-1} for OW and BW mussels in this study, a person could not safely consume more than 22–86 gww OW mussels or 13–52 gww BW mussels per day. Our TEQ values are consistent with wild blue mussels collected a c. 2 km away (Croftholmen: WHO-TEQ$_{PCDD/F}$ = 2.64, Knutzen et al., 2003). That study also measured similar or higher WHO-TEQ$_{PCDD/F}$ for crabs and several fish species. Our data indicate how uptake of PCDD/Fs during only 1 month in these waters can result in potentially toxic levels of these contaminants in human food, and that the closer the organisms live to the sediment, the more contaminants they will accumulate.

Acknowledgements

References

Accardi-Dey, A., Gschwend, P.M., 2002. Assessing the combined roles of natural organic matter and black carbon as sorbents in sediments. Environmental Science and Technology 36, 21–29.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26, 32–46.

Anderson, M.J., 2005. PERMANOVA: a FORTRAN Computer Program for Permutational Multivariate Analysis of Variance. Department of Statistics, University of Auckland, New Zealand.

Allan, I.J., Nilsson, H.C., Tjensvoll, I., Bradshaw, C., Næs, N., 2011. Mobile passive samplers: concept for a novel mode of exposure. Environmental Pollution 159, 2393–2397.

Atkinson, C.A., Jolley, D.F., Simpson, S.L., 2007. Effect of overlying water pH, dissolved oxygen, salinity and sediment disturbances on metal release and sequestration from metal contaminated marine sediments. Chemosphere 69, 1428–1437.

Axelman, J., Næs, K., Näf, K., Broman, D., 1999. Accumulation of polycyclic aromatic hydrocarbons in semipermeable membrane devices and caged mussels (Mytilus edulis L.) in relation to water column phase distribution. Environmental Toxicology and Chemistry 18, 2454–2461.

Bayne, B.L., 1976. Marine Mussels: Their Ecology and Physiology. Cambridge University Press, Cambridge.

Bellas, J., Ekelund, R., Halldórsson, H.P., Berggren, M., Granmo, Å., 2007. Monitoring of organic compounds and trace metals during a dredging episode in the Göta Älv Estuary (SW Sweden) using caged mussels. Water Air & Soil Pollution 181, 265–279.

Bocchetti, R., Fattorini, D., Pisanelli, B., Macchia, S., Oliviero, L., Pilato, F., Pellegrini, D., Regoli, F., 2008. Contaminant accumulation and biomarker responses in caged mussels, Mytilus galloprovincialis, to evaluate bioavailability and toxicological effects of remobilized chemicals during dredging and disposal operations in harbour areas. Aquatic Toxicology 89, 257–266.

Booij, K., Smedes, F., van Weerlee, E.M., Honkoop, P.J.C., 2006. Environmental monitoring of hydrophobic organic contaminants: the case of mussels versus semipermeable membrane devices. Environmental Science & Technology 40, 3893–3900.

Bradshaw, C., Veale, L., Brand, A., 2002. The role of scallop-dredge disturbance in long-term changes in Irish Sea benthic communities: a re-analysis of an historical dataset. Journal of Sea Research 47, 161–184.

Cantwell, M.G., Burgess, R.M., King, J.W., 2008. Resuspension of contaminated field and formulated reference sediments. Part I: evaluation of metal release under controlled laboratory conditions. Chemosphere 73, 1824–1831.

Charles, F., Lopez-Legentil, S., Gremare, A., Amouroux, J.M., Desmalades, M., Vetion, G., Escoubeyrou, K., 2005. Does sediment resuspension by storms affect the fate of polychlorobiphenyls (PCBs) in the benthic food chain? Interactions between changes in POM characteristics, adsorption and absorption by the mussel Mytilus galloprovincialis. Continental Shelf Research 25, 2533–2553.

Chiou, C.T., Porter, P.E., Schmedding, D.W., 1983. Partition equilibria of non-ionic organic compounds between soil organic matter and water. Environmental Science & Technology 17, 227–231.

Cornelissen, G., Gustafsson, O., Bucheli, T.D., Jonker, M.T.O., Koelmans, A.A., Van Noort, P.C.M., 2005. Extensive sorption of organic compounds to black carbon, coal, and kerogen in sediments and soils: mechanisms and consequences for distribution, bioaccumulation, and biodegradation. Environmental Science & Technology 39, 6881–6895.

Cornelissen, G., Broman, D., Næs, K., 2010. Freely dissolved PCDD/F concentrations in the Frierfjord, Norway: comparing equilibrium passive sampling with "active" water sampling. Journal of Soils and Sediment 10, 162–171.

Cotou, E., Gremare, A., Charles, F., Hatzianestis, I., Sklivagou, E., 2005. Potential toxicity of resuspended particulate matter and sediments: environmental samples from the Bay of Banyuls-sur-Mer and Thermaikos Gulf. Continental Shelf Research 25, 2521–2532.

Dellapenna, T.M., Allison, M.A., Gill, G.A., Lehman, R.D., Warnken, K.W., 2006. The impact of shrimp trawling and associated sediment resuspension in mud dominated, shallow estuaries. Estuarine, Coastal and Shelf Science 69, 519–530.

Dounas, C.G., 2006. A new apparatus for the direct measurement of the effects of otter trawling on benthic nutrient releases. Journal of Experimental Marine Biology and Ecology 339, 251–259.

Durrieu de Madron, X., Ferré, B., Le Corre, G., Grenz, C., Conan, P., Pujo-Pay, M., Buscail, R., Bodiot, O., 2005. Trawling-induced resuspension and dispersal of muddy sediments and dissolved elements in the Gulf of Lion (NW Mediterranean). Continental Shelf Research 25, 2387–2409.

Eggleton, J., Thomas, K.V., 2004. A review of factors affecting the release and bioavailability of contaminants during sediment disturbance events. Environment International 30, 973–980.

European Commission, 2006. Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. Official Journal of the European Union L 364, 5–24.

Falcão, M., Gaspar, M.B., Caetano, M., Santos, M.N., Vale, C., 2003. Short-term environmental impact of clam dredging in coastal waters (south of Portugal): chemical disturbance and subsequent recovery of seabed. Marine Environmental Research 56, 649–664.

Feng, J., Shen, Z., Niu, J., Yang, Z., 2008. The role of sediment resuspension duration in release of PAH. Chinese Science Bulletin 53, 2777–2782.

Floderus, S., Pihl, L., 1990. Resuspension in the Kattegat: impact of variation in wind climate and fishery. Estuarine Coastal and Shelf Science 31, 487–498.

Friedman, C.L., Lohmann, R., Burgess, R.M., Perron, M.M., Cantwell, M.G., 2011. Resuspension of polychlorinated biphenyl-contaminated field sediment: release to the water column and determination of site-specific K($_{DOC}$). Environmental Toxicology & Chemistry 30, 377–384.

Giannapas, M., Karnis, L., Dailianis, S., 2012. Generation of free radicals in haemocytes of mussels after exposure to low molecular weight PAH components:

immune activation, oxidative and genotoxic effects. Comparative Biochemistry and Physiology Part C 155, 182–189.

Gilek, M., Björk, M., Broman, D., Kautsky, N., Kautsky, U., Näf, C., 1997. The role of the blue mussel, Mytilus edulis, in the cycling of hydrophobic organic contaminants in the Baltic Proper. Ambio 26, 202–209.

Harman, C., Tollefsen, K.-E., Bøyum, O., Thomas, K., Grung, M., 2008. Uptake rates of alkylphenols, PAHs and carbazoles in semipermeable membrane devices (SPMDs) and polar organic chemical integrative samplers (POCIS). Chemosphere 72, 1510–1516.

Hedman, J.E., Tocca, J.S., Gunnarsson, J.S., 2009. Remobilization of polychlorinated biphenyl from Baltic Sea sediment: comparing the roles of bioturbation and physical resuspension. Environmental Toxicology and Chemistry 28, 2241–2249.

Hektoen, H., Berge, J.A., Ingebrigtsen, K., Knutzen, J., Oehme, M., 1994. Elimination of polychlorinated dibenzofurans and dibenzo-p-dioxins from blue mussel (Mytilus edulis) and tissue distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD). Chemosphere 29, 1491–1499.

Hiddink, J., Jennings, S., Kaiser, M., Queirós, A., Duplisea, D., Piet, G., 2006. Cumulative impact of seabed trawl disturbance on benthic biomass, production and species richness in different habitats. Canadian Journal of Fisheries and Aquatic Science 63, 721–736.

Huckins, J., Booij, K., Petty, J., 2006. Theory and modeling. In: Huckins, J., Booij, K., Petty, J. (Eds.), Monitors of Organic Chemicals in the Environment. Semipermeable Membrane Devices. Springer, New York, pp. 45–81.

Jennings, S., Kaiser, M., 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology 34, 203–314.

Jonas, P.J.C., Millward, G.E., 2010. Metals and nutrients in the Severn Estuary and Bristol Channel: contemporary inputs and distributions. Marine Pollution Bulletin 61, 52–67.

Jones, D.S., Barnthouse, L.W., Suter, G.W., Efroymson, R.A., Field, J.M., Beauchamp, J.J., 1999. Ecological risk assessment in a large river-reservoir: 3. Benthic invertebrates. Environmental Toxicology & Chemistry 18, 599–609.

Krost, P., 1990. The impact of otter-trawl fishery on nutrient release from the sediment and macrofauna of Kieler Bucht (Western Baltic). Berichte aus dem Institut für Meereskunde 200, 167.

Karickhoff, S.W., 1981. Semi-empirical estimation of sorption of hydrophobic pollutants on natural sediments and soils. Chemosphere 10, 833–846.

Knutzen, J., Bjerkeng, B., Næs, K., Schlabach, M., 2003. Polychlorinated dibenzofurans/dibenzo-p-dioxins (PCDF/PCDDs) and other dioxin-like substances in marine organisms from the Grenland fjords, S. Norway, 1975–2001: present contamination levels, trends and species specific accumulation of PCDF/PCDD congeners. Chemosphere 52, 745–760.

Lambert, M.K., Friedman, C., Luey, P., Lohmann, R., 2011. Role of black carbon in the sorption of polychlorinated dibenzo-p-dioxins and dibenzofurans at the Diamond Alkali Superfund Site, Newark Bay, New Jersey. Environmental Science and Technology 45, 4331–4338.

Latimer, J.S., Davis, W.R., Keith, D.J., 1999. Mobilization of PAHs and PCBs from in-place contaminated marine sediments during simulated resuspension events. Estuarine, Coastal and Shelf Science 49, 577–595.

Luoma, S.N., 1983. Bioavailability of trace metals to aquatic organisms – a review. Science of the Total Environment 28, 1–22.

McArdle, B.H., Anderson, M.J., 2001. Fitting multivariate models to community data: a comment on distance based redundancy analysis. Ecology Letters 82, 290–297.

Micheletti, C., Critto, A., Marcomini, A., 2007. Assessment of ecological risk from bioaccumulation of PCDD/Fs and dioxin-like PCBs in a coastal lagoon. Environment International 33, 45–55.

Molvær, J., Stigebrandt, A., 1991. Investigation of Eutrophication in the Grenlandsfjords 1988–89. Report 3. Vannutskiftning i fjordene. Norwegian State Pollution Monitoring Programme, Report No 450/91 (in Norwegian).

Næs, K., 1999. Monitoring of Contaminants in Sediments from the Grenlandsfjords 1997. Norwegian State Pollution Monitoring Programme, Report No 765/99 (in Norwegian with English Summary).

Nelson, W., Phelps, D., Galloway, W., Rogerson, P., Pruell, R., 1987. Effects of Black Rock Harbor Dredged Material on the Scope for Growth of the Blue Mussel, Mytilus edulis, After Laboratory and Field Exposures. US Environmental Protection Agency Technical Report D-87-7.

Olsen, C.R., Cutshall, N.H., Larsen, I.L., 1982. Pollutant-particle associations and dynamics in coastal marine environments: a review. Marine Chemistry 11, 501–533.

O'Neill, F.G., Summerbell, K., 2011. The mobilisation of sediment by demersal otter trawls. Marine Pollution Bulletin 62, 1088–1097.

Palanques, A., Guillén, J., Puig, P., 2001. Impact of bottom trawling on water turbidity and muddy sediment of an unfished continental shelf. Limnology & Oceanography 46, 1100–1110.

Percival, P., Frid, C., Upstill-Goddard, R., 2005. The impact of trawling on benthic nutrient dynamics in the North Sea: implications of laboratory experiments. American Fisheries Society Symposium 41, 491–501.

Persson, N.J., Gustafsson, Ö., Bucheli, T.D., Ishaq, R., Naes, K., Broman, D., 2002. Soot-carbon influenced distribution of PCDD/F in the marine environment of the Grenlandsfjords, Norway. Environmental Science & Technology 36, 4968–4974.

Persson, N.J., Bucheli, T.D., Gustafsson, Ö., Broman, D., Næs, K., Ishaq, R., Zebühr, Y., 2005. Testing common sediment-porewater distribution models for their ability to predict dissolved concentrations of POPs in the Grenlandsfjords, Norway. Chemosphere 59, 1475–1485.

Persson, N.J., Cousins, I.T., Molvær, J., Broman, D., Næs, K., 2006. Modelling the long-term fate of polychlorinated dibenzo-p-dioxins and furans (PCDD/Fs) in the Grenland Fjords, Norway. Science of the Total Environment 369, 188–202.

Pilskaln, C.H., Churchill, J.H., Mayer, L.M., 1998. Resuspension of sediment by bottom trawling in the Gulf of Maine and potential geochemical consequences. Conservation Biology 12, 1223–1229.

Roberts, D.A., 2012. Causes and ecological effects of resuspended contaminated sediments (RCS) in marine environments. Environment International 40, 230–243.

Ruus, A., Schaanning, M., Oxnevad, S., Hylland, K., 2005. Experimental results on bioaccumulation of metals and organic contaminants from marine sediments. Aquatic Toxicology 72, 273–292.

Ruus, A., Berge, J.A., Hylland, K., Bjerkeng, B., Bakke, T., Naes, K., 2006. Polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in the Grenland Fjords (Norway) – disposition, levels, and effects. Journal of Toxicology and Environmental Health A 69, 185–200.

Saloranta, T., Armitage, J., Naes, K., Cousins, I., Barton, D., 2006. SF-Tool Multimedia Model Package: Model Code Description and Application Examples from the Grenland fjords. Norwegian Institute for Water Research, Report No 5216–2006.

Schlabach, M., Knutzen, J., Bjerkeng, B., Becher, G., 1998. Tracing of the PCDF/PCDD contamination from the Frierfjord along the Norwegian south coast. Organohalogen Compounds 36, 505–508.

Schoellhamer, D.H., 1996. Anthropogenic sediment resuspension mechanisms in a shallow microtidal estuary. Estuarine Coastal and Shelf Science 43, 533–548.

Schubel, J.R., Carter, H.H., Wise, W.M., 1979. Shrimping as a source of suspended sediment in Corpus Christi Bay (Texas). Estuaries 2, 201–203.

Schwarzenbach, R., Gschwend, P., Imboden, D., 2003. Environmental Organic Chemistry, second ed. John Wiley & Sons, New York.

Seth, R., Mackay, D., Muncke, J., 1999. Estimating the organic carbon partition coefficient and its variability for hydrophobic chemicals. Environmental Science & Technology 33, 2390–2394.

Steevens, J.A., Reiss, M., Pawlisz, A.V., 2005. A methodology for deriving tissue residue benchmarks for aquatic biota: a case study for fish exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin and equivalents. Integrated Environmental Assessment and Management 1, 142–151.

Thrush, S.F., Dayton, P.K., 2002. Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. Annual Review of Ecology and Systematics 33, 449–473.

Trimmer, M., Petersen, J., Sivyer, D.B., Mills, C., Young, E., Parker, E.R., 2005. Impact of long-term benthic trawl disturbance on sediment sorting and biogeochemistry in the southern North Sea. Marine Ecology Progress Series 298, 79–94.

van Leeuwen, F.X., Feeley, M., Schrenk, D., Larsen, J.C., Farland, W., Younes, M., 2000. Dioxins: WHO's tolerable daily intake (TDI) revisited. Chemosphere 40, 1095–1101.

Warnken, K.W., Gill, G.A., Dellapenna, T.M., Lehman, R.D., Harper, D.E., Allison, M.A., 2003. The effects of shrimp trawling on sediment oxygen consumption and the fluxes of trace metals and nutrients from estuarine sediments. Estuarine Coastal and Shelf Science 57, 25–42.

Watling, L., Norse, E.A., 1998. Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. Conservation Biology 12, 180–1197.

WHO, 1998. Assessment of the Health Risks of Dioxins: Re-evaluation of the Tolerable Daily Intake (TDI). Executive Summary of the WHO Consultation, May 25–29 1998, Geneva.