

Impacts of biodeposits from suspended mussel (*Mytilus edulis* L.) culture on the surrounding surficial sediments

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The effects of increased sedimentation on the macrobenthic community, physical structure, and biogeochemistry of the surficial sediment around two farms in south-west Ireland were examined in conjunction with current characteristics. Both farms had been in production for over eight years, were of reasonably large size (>100 MT) and located in low-energy environments. At one site, the benthic community was subjected to bulk sedimentation and organic enrichment and reduced macrobenthic infaunal diversity and elevated levels of organic carbon were recorded close to the farm. In general, effects were restricted to a radius of 40 m around the farm. Conversely, at the second site, there were no observed effects of mussel biodeposits on the benthos and a diverse macrobenthic community persisted. We propose that variations in the dispersion of biodeposits caused by local current patterns had a significant influence on the impact observed, and that this could also account for differences reported in other studies.

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Introduction

The effects of biodeposits from suspended mussel culture on the local benthic environment have been considered in a number of studies (Tenore *et al.*, 1982; Kaspar *et al.*, 1985; Baudinet *et al.*, 1990; Hargrave, 1994). The reported effects on the physicochemical and biological structure of the surrounding surficial sediments were generally similar, although the extent and degree of these differed considerably among locations.

Heavy sedimentation of faeces and pseudofaeces beneath mussel farms effectively lead to organic enrichment and thus alter macrofaunal communities (Mattsson and Lindén, 1983; Kaspar *et al.*, 1985; Tenore *et al.*, 1982, 1985). Sedimentation rates up to three times higher than at reference sites have been reported (Dahlbäck and Gunnarson, 1981). Grenz (1989) suggested that average biodeposits in suspended culture could reach quantities up to 345 kg m⁻² year⁻¹. The

increased sedimentation rate may select for species that are more tolerant of low oxygen levels or of the instability of finer textured, high organic sediments (Tenore *et al.*, 1982). However, Jaramillo *et al.* (1992) and Grant *et al.* (1995) did not record such changes under mussel farms and Baudinet *et al.* (1990) concluded that mussel culture had little impact at their site. Grenz *et al.* (1990) reported that deposition originating from phytoplankton blooms confounded the effects of mussel biodeposition on sediment organic matter.

Macrofaunal abundance is commonly reduced at mussel farms (Tenore *et al.*, 1982; Mattsson and Lindén, 1983; Jaramillo *et al.*, 1992; Grant *et al.*, 1995) with concomitant decreases in infaunal diversity (Mattsson and Lindén, 1983). However, Grant *et al.* (1995) did not find the classic eutrophication responses such as hypoxic conditions and sulphidic sediments, and the macrofaunal biomass beneath the mussel long-lines at times surpassed the biomass at the reference site.

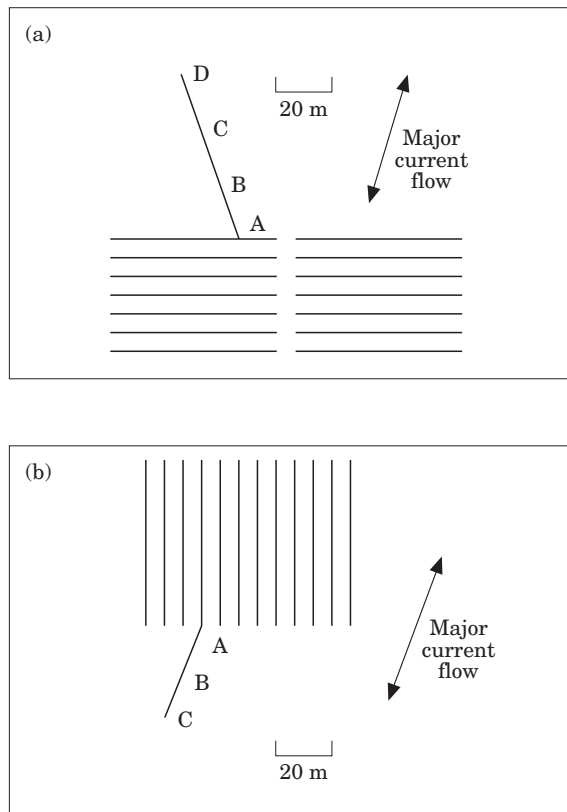


Figure 1. Schematic diagrams of the layout of (a) farm site 1 and (b) site 2 with direction of major current flow and location of sampling stations.

We provide a description of the benthic environmental conditions at two suspended mussel culture sites in southwest Ireland. The project was designed to examine local effects of individual farms, or of selected groups of mussel lines, rather than the bay-wide effects reported in several other studies.

Materials and methods

Site selection and field sampling

The criteria used for site selection were that the farms were in areas of low hydrological energy and shallow depth, thus reducing the dispersion of biodeposits according to the basic dispersion model proposed by Gowen *et al.* (1988). The two sites (Figure 1) were in adjoining bays (approximately 35 km apart) forming deep marine inlets in flat-floored valleys of glacial origin on the southwest coast of Ireland. The environments were similar and the farms had a relatively large annual production for the area (Table 1).

An array of two electromagnetic current meters (S4, InterOcean Systems) was deployed close to each farm site. The meters were moored from a single line and

placed at approximately 2 m and 8 m above the seabed. Current speed and direction were recorded over a 14-d period (equivalent to one tidal cycle), after which the meters were recovered. During August 1998, benthic core samples (90-mm long, internal diameter 81 mm) were obtained by divers. Transects were laid out from the farms along a gradient of deposition, taking into account the current meter measurements and *in situ* observations. Sample stations were established along these transects at either 10-m or 20-m intervals (Figure 1). Core samples were analysed for benthic infauna (five replicates, 0–9 cm depth), elemental carbon and nitrogen and granulometry (three replicates, 0–6 cm depth) and sediment redox (0–6 cm). The samples for faunal analysis were fixed in buffered 10% formalin saline containing 0.1% Rose Bengal stain. Each sample was subsequently sieved through a 500- μ m mesh and the retained fauna were sorted, identified and counted.

Immediately on retrieval of the cores, the redox potential of the sediment was measured at 0.5- to 1-cm intervals using a redox probe (30 \times 1 cm; Russell pH), standardized in Zobell's solution (Zobell, 1946), and a waterproof pH-redox meter (Russell RL100). Sediment samples were taken for carbon and nitrogen measurement and particle size analysis by slicing into 1-cm sections, followed by freezing. Results are reported from the surficial layer (0–1 cm depth) only, as this was considered to be the most significant with regard to the flux of biodeposits to the seabed.

Analysis

The macrofauna were identified to species level, as far as possible, with the use of standard taxonomic keys. Nomenclature followed Howson and Picton (1997).

Granulometric composition was determined using a Malvern Mastersizer/E (Malvern Instruments) with a 300-mm lens. From the computed results, the percentage <63 μ m was determined as an index of the silt/clay composition.

Aliquots of freeze-dried samples for elemental carbon and nitrogen analysis were initially ground to a fine powder using an automated mill and then heated with acid to remove the carbonate fraction. The samples were analysed using a Perkin Elmer 2400 Elemental analyser.

At each site and for each sample, non-metric multi-dimensional scaling (MDS) was carried out on root-transformed quantitative infaunal data using the Bray–Curtis similarity measure (PRIMER software; Clarke and Warwick, 1994). Where necessary, the physico-chemical data were transformed to achieve normality. One-way ANOVA with Tukey's pairwise comparison test ($p < 0.05$) was carried out on the physico-chemical results of the surficial sediment layer (0–1 cm depth).

Table 1. Farm characteristics and current measurements at Sites 1 and 2.

	Site 1		Site 2	
Farm type	Longline		Longline	
Production (t yr ⁻¹)	100		150	
Depth (m)	12–15		13–15	
Age of farm (yr)	~8		~14	
Current meter	Upper	Lower	Upper	Lower
Height above seabed (m)	8	2	8	2
Mean current velocity (cm s ⁻¹)	3.11	3.14	2.28	2.85
Residual current velocity (cm s ⁻¹)	1.25	0.75	0.76	0.40
Residual direction (deg)	52	277	226	102

Table 2. Top five ranked species (abundance), numbers and percentage of total across stations at the two farm sites.

Site 1		#	%	Site 2		#	%
A	<i>Aphelochaeta marioni</i>	81	32.2	A	<i>Ophryotrocha hartmanni</i>	295	62.9
	<i>Cirratulus cirratus</i>	28	11.1		<i>Aphelochaeta vivipara</i>	40	8.5
	<i>Podarkeopsis capensis</i>	27	10.7		Nematode spp.	27	5.7
	Cossuridae spp.	23	9.1		<i>Tharyx killariensis</i>	20	4.2
	<i>Prionospio fallax</i>	16	6.3		<i>Chaetozone setosa</i>	15	3.2
	Total		69.7		Total		84.6
B	Tanaidae spp.	27	21.2	B	Gammaridae spp.	113	41.3
	<i>Podarkeopsis capensis</i>	16	12.6		<i>Ophryotrocha hartmanni</i>	30	10.9
	<i>Scoloplos armiger</i>	12	9.4		<i>Nephtys hombergii</i>	19	6.9
	<i>Aphelochaeta marioni</i>	12	9.4		<i>Aphelochaeta vivipara</i>	15	5.4
	Gammaridae spp.	11	8.6		<i>Abra alba</i>	9	3.3
	Total		61.4		Total		68.1
C	Tanaidae spp.	19	15.0	C	Gammaridae spp.	89	42.5
	<i>Podarkeopsis capensis</i>	18	14.2		<i>Melinna palmata</i>	16	7.6
	Cossuridae spp.	14	11.1		<i>Abra alba</i>	16	7.6
	<i>Protodorvillea kefersteini</i>	12	9.5		Tanaidae spp.	12	5.7
	<i>Aphelochaeta marioni</i>	11	8.7		<i>Protodorvillea kefersteini</i>	7	3.3
	Total		58.7		Total		66.9
D	Cossuridae spp.	23	18.4				
	Tanaidae spp.	21	16.8				
	<i>Ophryotrocha hartmanni</i>	9	7.2				
	<i>Protodorvillea kefersteini</i>	8	6.4				
	<i>Scoloplos armiger</i>	8	6.4				
	Total		55.2				

Results

The residual current velocities at Site 2 were very low when compared to Site 1. At both sites, the upper-layer and lower-layer current meters recorded different residual current velocities and directions (Figure 1, Table 1). Generally, Site 2 was characterized by thick layers of mussel faeces under and around the farm, decreasing in thickness with distance from the farm.

A total of 35 and 33 macrofaunal species were identified at Sites 1 and 2, respectively. The top five species for each station are listed in Table 2. Species composition at Site 1 was dominated by the poly-

chaetes *Aphelochaeta marioni* (Saint-Joseph, 1894) and *Cirratulus cirratus* (O. F. Müller, 1776). These were especially dominant at stations near the farm and decreased in abundance at stations further away. The dominant species at Site 2 were (Station A) the polychaetes *Ophryotrocha marioni* (Huth, 1933), *Aphelochaeta vivipara* (Christie, 1984), and *Tharyx killariensis* (Southern, 1914) and (Stations B and C) gammarid amphipods living in the surficial layer.

Shannon–Wiener diversity indices (Table 3) revealed little difference among stations at Site 1, implying little direct impact from the farm. At Site 2, diversity was low close to the farm and markedly higher at Stations B and C.

Table 3. Comparison of Shannon-Wiener diversity index (I) and physico-chemical characteristics of the surficial sediments (mean \pm s.d., $n=3$; except Site 1, Station D where $n=4$; with associated p and F values from ANOVA with Tukey's pairwise comparison test) at each station at the two farm sites (C: total organic carbon content; N: total nitrogen; S: silt/clay content as fraction $<63 \mu\text{m}$).

Site	Station	I	C (%)	N (%)	S (%)
1	A	2.3	2.73 ± 0.15	0.37 ± 0.03	84.53 ± 6.44
	B	2.5	2.93 ± 0.18	0.42 ± 0.04	89.70 ± 1.11
	C	2.7	3.12 ± 0.16	0.43 ± 0.03	90.80 ± 0.21
	D	2.7	2.90 ± 0.22	0.45 ± 0.03	87.37 ± 3.35
	F-value	—	2.84	2.79	0.51
	p-value	—	ns	ns	ns
2	A	1.5	3.45 ± 0.56	0.58 ± 0.08	88.48 ± 2.73
	B	2.2	2.56 ± 0.24	0.41 ± 0.04	72.40 ± 9.78
	C	2.3	2.03 ± 0.18	0.27 ± 0.01	55.00 ± 2.75
	F-value	—	11.38	27.80	22.80
	p-value	—	0.009	0.001	0.002
			$A \neq B \neq C$	$A \neq B \neq C$	$A \neq B \neq C$

The MDS plots (Figure 2) illustrated the overall similarity among samples and stations. The plot for Site 1 suggests an alteration in species composition (indicated by arrow) with distance from the farm. However, the high stress value (0.21) demands caution in interpretation (Clarke and Warwick, 1994). Therefore, it is unclear whether the observed changes in species composition along the transect are significant. The plot for Site 2 indicates a definite trend in faunal composition and alteration of the community structure around the farm. At Station A, the community was impoverished and dominated by opportunistic, deposit-feeding polychaetes. This fauna was replaced by a more diverse community at Station B, leading to a community typical of background conditions at Station C.

Eh values (Figure 3) were positive at all stations at Site 1, with maximum values recorded closest to the farm (Station A). At Site 2, the values were lowest and negative at Station A and increased with distance from the farm, becoming positive throughout the core depth at Station C.

The total organic carbon contents of the surficial sediment layer were closely related to the distribution of fine particles ($\% <63 \mu\text{m}$); and similar trends were observed for total nitrogen (Table 3). There were no significant differences among stations at Site 1 for any of these variables. In contrast, all variables were significantly different ($p < 0.05$) among the stations at Site 2. Total organic carbon, total nitrogen, and silt content were highest at Station A and lowest at Station C.

Discussion

Alteration of macrofaunal community structure by organic enrichment has been a keystone of benthic

environmental studies (Findlay *et al.*, 1995) since the seminal review of Pearson and Rosenberg (1978). In our study, biodeposits from mussel farms were similarly shown to affect the benthic infaunal community structure of the surficial sediment only at one of the two sites investigated (Site 2), where the benthos was clearly subjected to elevated levels of sedimentation and organic enrichment. The effects were localised and the community was similar to those typical of unaffected conditions beyond a 40-m radius of the farm site. Conversely, at Site 1, no alterations of the community structure were observed and an infaunal community typical of background conditions appeared to persist under the farm.

The results for Site 2 fit broadly into those obtained by previous studies of benthic effects of suspended mussel culture, which generally demonstrate a reduction in macrofaunal diversity beneath mussel farms (Tenore *et al.*, 1982; Mattsson and Lindén, 1983; Kaspar *et al.*, 1985; Jaramillo *et al.*, 1992). Dahlbäck and Gunnarson (1981) and Tenore *et al.* (1982) showed high rates of faeces and pseudofaeces sedimentation beneath mussel lines, which were confirmed at Site 2. Increased sedimentation through biodeposition increased surficial sediment organic matter (Kaspar *et al.*, 1985), leading to an impoverished benthic community around the culture site in a sediment consisting of organically enriched fine grey flocculent material.

Negligible effects similar to those found at Site 1 were reported by Baudinet *et al.* (1990) and Grant *et al.* (1995). Both studies concluded that biodeposits from the respective mussel farms had little impact on the benthos. The alterations observed by Grant *et al.* (1995) were attributed to fall-off from the mussel lines, but changes caused by organic enrichment were considered minimal.

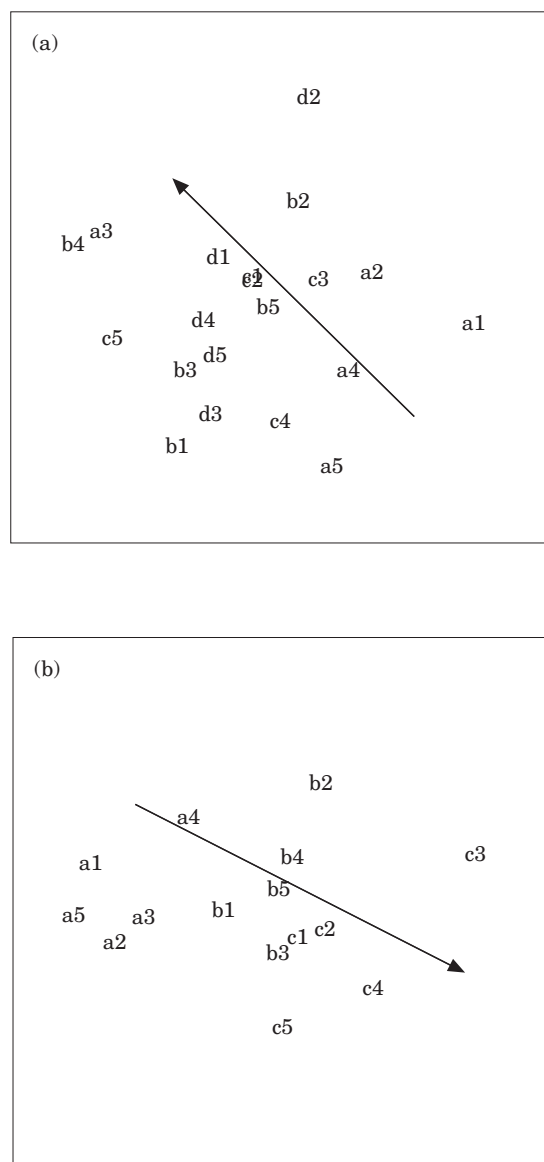


Figure 2. Multidimensional scaling plots of the overall faunal similarity of the sampling stations at (a) Site 1 (stress=0.21) and (b) Site 2 (stress=0.11) (arrows indicate trend in faunal composition of the different sampling stations).

Apparently, the degree and extent of effects from mussel cultures differ considerably between locations. A number of factors have been suggested for explaining these differences, including age of farm, stocking densities on ropes, physical structure of farm (density and orientation of ropes, distance from the bottom), hydrodynamics, sediment adsorption, etc. We consider that one important factor determining the final fate of faecal matter, and any subsequent impact, is the dispersion of biodeposits from the farm site. Information on current speed and direction, water depth and accurate estimates

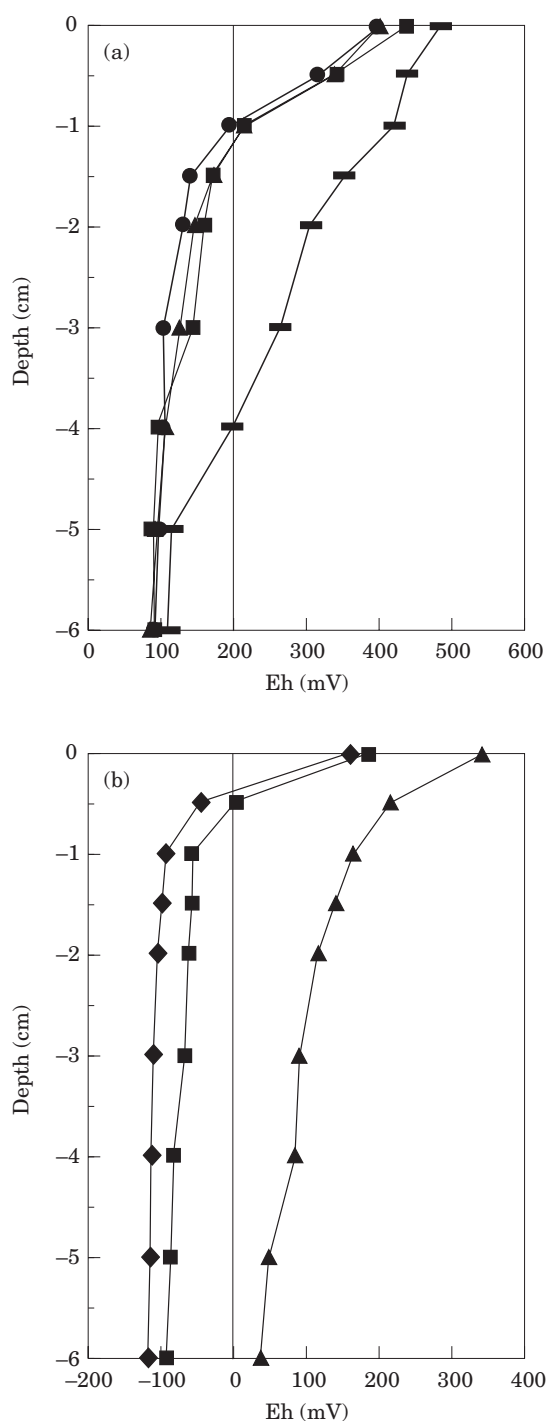


Figure 3. Plots of the redox potential of the sediment with depth at (a) Site 1 (—■—, A; —▨—, B; —▲—, C; —●—, D) and (b) Site 2 (—◆—, A; —■—, B; —▲—, C).

of settling velocities of biodeposits is required to examine this aspect, in accordance with the model developed by Cromey *et al.* (2000) to predict benthic

impacts at mariculture sites. Additional factors that we considered important were the production tonnage of the farm and the food availability to the mussels (i.e. suspended particulate matter) at the time of sampling.

The majority of our observations of low settling velocities, $<0.5 \text{ cm s}^{-1}$ for faeces and $<0.8 \text{ cm s}^{-1}$ for pseudofaeces, suggest that the area over which biodeposits reach the sediment is strongly influenced by water movement and depth. Resuspension of material could also affect the area over which the biodeposits finally settle. However, because of the low current velocities around the farms investigated, resuspension was probably not an important factor. Still, due to the low settling velocities of the particles involved, slight variations in current velocity and direction and water depth around the farms could have a great effect on the dispersion of the biodeposits and thus on rate of organic enrichment within the dispersion area.

Biodeposits from mussel farms thus may or may not have significant effects on benthos and the differences observed in this and other studies might be explained by the various dispersion factors. The higher average current velocities at Site 1 compared to Site 2 are consistent with this explanation. Thus, the potential for mussel farms to affect the benthos in the surrounding area is always site specific and hydrographic information is essential to select least destructive culture sites and to calculate appropriate biomass limits per unit.

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