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Hall-Spencer, Jason

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EFFECTS OF TOWED DEMERSAL FISHING GEAR ON BIOGENIC SEDIMENTS: A 5-YEAR-STUDY

Jason M. HALL-SPENCER

Institute of Biomedical and Life Sciences, Graham Kerr Building,
University of Glasgow, G 12 8QQ, Scotland.

Tel: + 44 141 3305985; Fax: + 44 1475 530601; ([e-mail: gbfa20@udcf.gla.ac.uk](mailto:gbfa20@udcf.gla.ac.uk))

RIASSUNTO

Effetti delle draghe da pesca su sedimenti biogenici: cinque anni di studio.

È stato condotto un esperimento di pesca di cappelletto (*Pecten maximus*), mediante draghe utilizzate per la pesca commerciale, su fondali composti da sedimenti depositati da alghe coralline non fissate al substrato. Questi sedimenti biogenici sono distribuiti in modo irregolare nelle acque costiere europee (fino a -30 m di profondità intorno al Regno Unito e fino a -120 m nel Mediterraneo) e rivestono una notevole importanza, a livello internazionale, per quanto riguarda la tutela dell'ambiente. Questo articolo descrive gli effetti, a breve e lungo termine, della pesca ai Pettinidi mediante draghe su sedimenti algali biogenici, sia in aree precedentemente soggette che non ad attività di pesca, presenti a Sud-Ovest della Scozia. Sono stati effettuati carotaggi di sedimenti ogni due anni, nel periodo 1994-'99, per valutare l'abbondanza di alghe coralline vive sia nelle aree sperimentali che in quelle di controllo. In una zona soggetta ad attività di pesca le alghe coralline vive coprono un'area <3%, e l'impiego sperimentale di draghe da traino non inficiava in maniera rilevante i valori di abbondanza. Le popolazioni di alghe coralline vive ad alta densità (copertura -20%) erano localizzate su un fondale che non era stato, in precedenza, soggetto ad attività di pesca. Pur essendo rimasta un'alta copertura di alghe coralline nell'area di controllo, si è avuta una riduzione del 70% della loro concentrazione nella zona dove è stata condotta la pescata sperimentale, senza che vi sia stato osservato alcun segno di recupero nei successivi 5 anni.

ABSTRACT

Experimental scallop fishing was carried out using towed commercial dredges on sediments deposited by unattached coralline algae in order to quantify their sensitivity to damage from current fishing practices. These biogenic sediments are patchily distributed in European coastal waters (to -30 m depth around the UK and to -120 m in the Mediterranean) and are of international conservation importance. This paper describes the short and long-term effects of scallop dredging on previously unfished and fished areas of biogenic algal sediment in SW Scotland. Sediment cores taken biannually from 1994-99 were used to assess live coralline abundance on marked test



and control plots. Living corallines had <3% cover at a fished site and experimental dredging had no discernible effect on their abundance. Dense populations of live coralline thalli (~20% cover) were located on a previously unfished ground. Although coralline cover remained high in control plots on the unfished site, experimental fishing led to ~ 70% reduction in live corallines on test plots with no signs of recovery over the subsequent 5 years.

Key words: scallop dredging, coralline algae, long-term impact, benthos, Scotland

INTRODUCTION

Widespread interest in the ecological effects of fishing has stimulated intense research in recent years (Lindeboom & de Groot, 1998; Jennings & Kaiser, 1998; Hall, 1999). A number of recent studies have established that towed demersal fishing gear can have dramatic immediate effects on the benthos (Watling & Norse, 1998), particularly where complex biogenic structures are found at the sea bed surface (Hall, 1999; Hall-Spencer & Moore, in press). Effects in the short term (weeks/months) are relatively well documented on continental shelf areas and are most pronounced for those types of gear that penetrate the substratum deeply, such as hydraulic dredges (Hall *et al.*, 1990; Pranovi & Giovanardi, 1994) and scallop dredges (Currie & Parry, 1996; Collie *et al.*, 1997; Hall-Spencer *et al.*, 1999; Bradshaw *et al.*, 2000).

The few studies that have monitored the after-effects of towed demersal fishing have usually been unable to detect long-term effects (years) on the benthos. The effects of gear impact on small areas within dynamic habitats, such as sandy sediments, are quickly diluted through the immigration of benthos from surrounding areas or through sediment redistribution (Hall *et al.*, 1990; Eleftheriou & Robertson, 1992). Even in larger-scale studies, the long-term effects of fishing have been difficult to differentiate from other sources of variability in the benthos such as population heterogeneity, large seasonal fluctuations and the effects of natural disturbances such as storms (Hall *et al.*, 1993; Currie & Parry, 1996; Kaiser *et al.*, 1998; Bradshaw *et al.*, 2000). In most cases, ecological studies have been undertaken in areas already modified by commercial fishing activity since there are now very few grounds that are a) suitable for fishing yet b) unmodified by towed fishing gear (Lindeboom, 1995; Tuck *et al.*, 1998). The benthos that remains in fished areas can represent modified communities that are generally resilient to gear impacts (Currie & Parry, 1996; Kaiser *et al.*, 1996; Bradshaw *et al.*, 2000).

The present study forms part of an investigation comparing natural variability with anthropogenic impacts on 'maerl' - a biogenic marine sediment deposited by calcareous rhodophytes (Grail & Glémarec, 1998; Hall-Spencer, 1998; Hall-Spencer & Atkinson, 1999). Maerl typically harbours a high biodiversity but is very slow-growing in European waters (Potin *et al.*, 1990; Canals & Ballesteros, 1997; Birkett *et al.*, 1998). NE Atlantic maerl beds are characterised by coarse sediment, clean water and



appreciable bottom currents. As such, they often provide good scallop fishing grounds. Of 242 maerl beds surveyed around the UK in the past 10 years, 29% were found to support the great scallop, *Pecten maximus* (Marine Nature Conservation Review database, 1999). Concerns have recently been expressed about the sensitivity of this biotope to towed demersal fishing gear (MacDonald *et al.*, 1996; Hall-Spencer, 1998; Hall-Spencer & Moore, 2000). Here, we report differences between fished and previously unfished areas of biogenic sediment in SW Scotland. A small-scale experiment is described which was designed to investigate the immediate and long-term effects of commercial scallop dredges on maerl-forming species.

METHODS

Site descriptions

Two grounds situated in the Clyde Sea area, SW Scotland were surveyed in detail (Fig. 1) and monitored from 1994-1999 by using a combination of Sprint® Remote Operated Vehicle (Perry Tritech Ltd), RoxAnn®, van Veen grab sampling (at least 6 grabs per site per quarter) and >230 h field observations using SCUBA. Site 1 was at Creag Gobhainn, Loch Fyne, where muddy maerl occurred parallel to the shore along a gently sloping strip over an area of 17.5 ha from -6 m to -14 m CD (Chart Datum). This site was unimpacted by towed demersal fishing gear due to rocky outcrops, its proximity to shore and the presence of a charted telecommunications cable (laid in 1968) in a designated 'trawling prohibited' area. Site 2 was situated on a shoal in Stravanan Bay -0.5 km off the SW coast of the Isle of Bute where coarse maerl covered an area of 6.75 ha from -6 m to -15 m CD. This site was used by scallop fishermen during 1994-99 (pers. obs.) and had been impacted by scallop dredges over the past four decades (G.A. Fisher, unpubl.). These sites were selected due to their proximity to facilities at the University Marine Biological Station, Millport: further descriptions of these sites are given by Hall-Spencer (1998) and Hall-Spencer & Atkinson (1999).

Fishing and sampling protocol

Surface and subsurface buoys were laid at -10 m CD in order to provide permanent markers for the 5 year study at 56°00.601'N 5°22.148'W (Site 1) and 55°45.323'N 5°04.265'W (Site 2). At each site, two buoys were laid 10 m apart to delimit the width of an area to be fished (test plot) while the opposite side of the permanent marker was untouched (control plot). In May 1994, divers took eight replicate sediment cores (PVC pipes 20 cm long, 10.3 cm diameter) on test plots and eight further cores on control plots (Fig. 2). Throughout the study, collected cores were kept upright, frozen within 5 h of collection and stored at -18 °C.

The day after coring, test plots were fished from RV Aora (15 m, 260 hp) using a set of three Newhaven scallop dredges (Fig. 3). The dredges were 77 cm wide, weighed 85 kg in air and had 10 cm long, 0.8 cm wide teeth mounted 8 cm apart (9 per dredge) on spring loaded tooth bars. A bag of linked 7 cm diameter steel rings extended behind

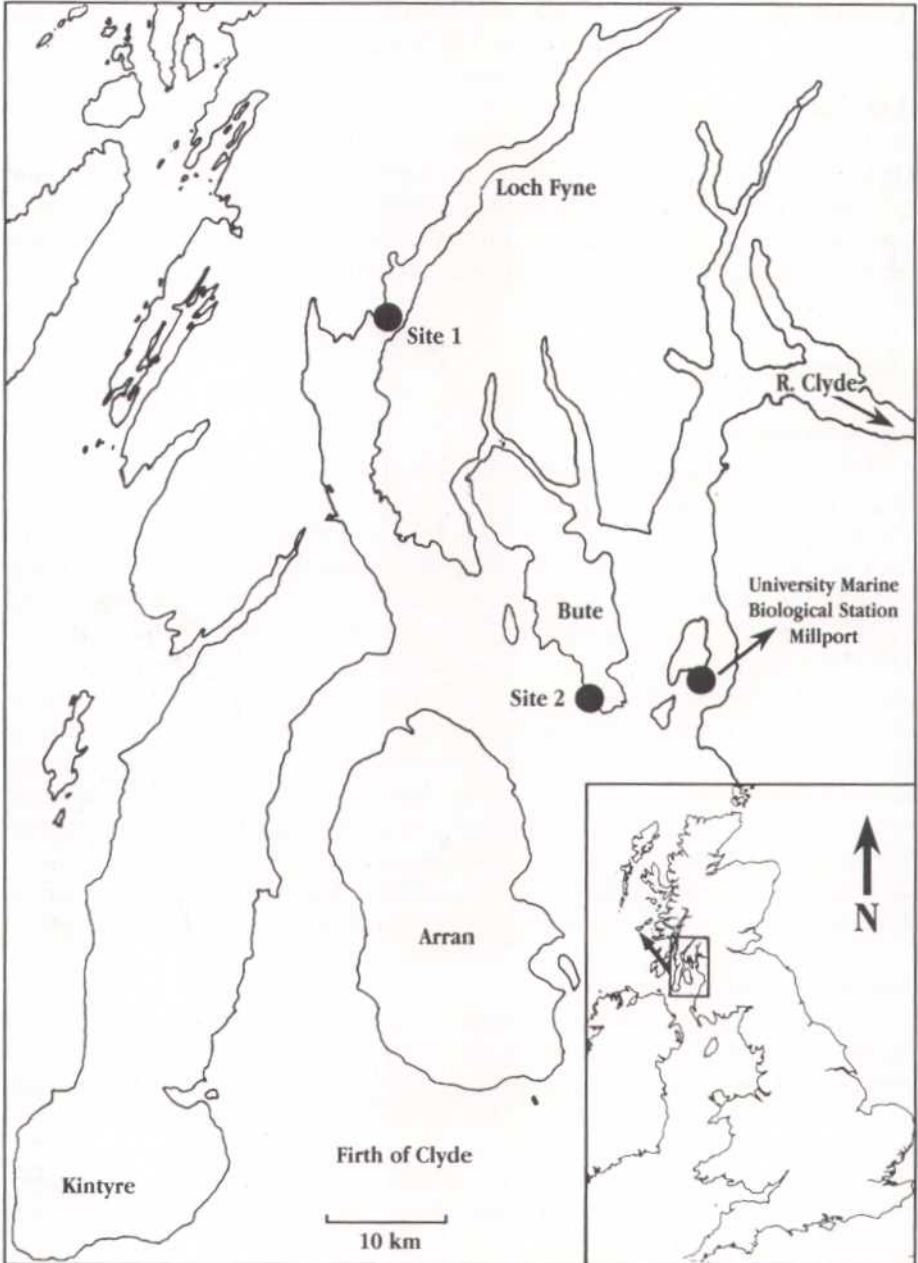


Fig. 1: Positions of two maerl sites in the Clyde Sea area, SW Scotland, sampled biannually from 1994-1999.

Fig. 1: Posizione di due siti con presenza di maerl nell'area del Clyde Sea, SW della Scozia, campionati due volte l'anno nel periodo 1994-1999.



Fig. 2: Diver equipped with SCUBA collecting a maerl gravel core at 10 m depth from a commercial scallop dredging site (Stravanan Bay, Bute), May 1995.

Fig. 2: Subacqueo durante un carotaggio per la raccolta di maerl a 10 m di profondità in un sito clove viene praticata la pesca commerciale di Pettinidi mediante draga (Stravanan Bay, Bute), Maggio 1995.

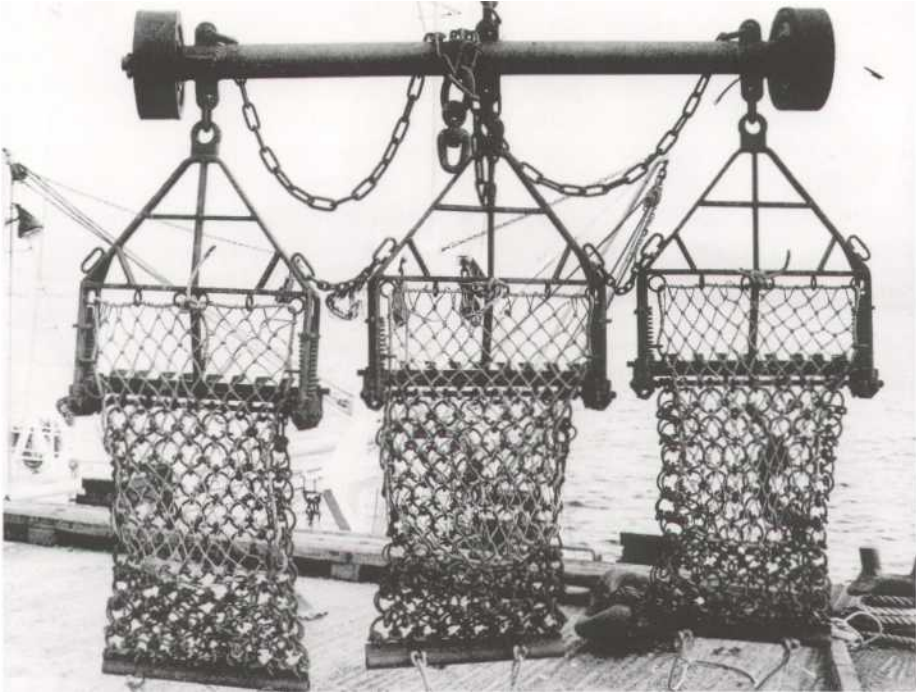


Fig. 3: A set of three Newhaven scallop dredges used in the present study, each dredge mouth measures 75 cm across.

Fig. 3: Un set di tre draghe per la pesca ai Pettinidi utilizzate per questa ricerca, la bocca di ciascuna draga misura 75 cm.



each tooth bar to retain the catch. The dredges were towed once over each test plot for -100 in giving fished areas of ~230 m².

Immediately after fishing, a pair of divers equipped with tape measures and writing slates recorded gross changes to surface topography on the test plots and took eight cores. Changes were also noted on the adjacent control plots where a further eight cores were taken. Labelled aluminium rods were pushed into the sediment to mark where cores were taken. This coring routine was repeated biannually for 5 years (until May 1999). The amount of living maerl on test and control plots was assessed as follows. Cores were first thawed and removed from the PVC pipes to note vertical stratification within the sediment. They were then wet sieved in fresh water through a 5 mm mesh. Maerl thalli that had been living at the time of collection (hereafter referred to as 'live thalli') were removed with forceps and laid in a monolayer on graph paper to obtain an estimate of the area covered by live thalli.

RESULTS

Differences in experimental sites

Sites I and 2 each had live maerl (Fig. 4) on level areas of sea bed within sheltered parts of the northern Clyde Sea area with no differences in depth (-10 m CD), salinity range (31-34 ssu), temperature range (6-14°C) or mean tidal amplitude (~3 m). Table 1I summarizes ecological differences between the sites which reflect that Site I was unfished with towed demersal gear whereas Site 2 had been scallop dredged for the past 40 years. An obvious difference between these two sites was that commercial dredge marks were common at the fished site but were absent in the unfished area. Population densities of the commercial scallop species *Aequipecten opercularis* (L.) and *Pecten maximus* (L.) were high at the unfished site (Fig. 4) whereas scallops were sparse and large individuals (> 7 years) were absent from the fished site. Other obvious differences between the two sites included lower molluscan diversity and lower live maerl cover at the fished site (Table 1).

Immediate and long-term effects of scallop dredging

Dives on test plots in the days after experimental dredging showed extensive physical disturbance along ~2.5 m wide tracks with three parallel furrows corresponding to the width of each dredge. Natural bottom features (e.g. sediment ripples, crab feeding pits and megafaunal burrows) were eliminated along the dredge tracks and boulders up to 1 m² had been dragged along the sediment surface. A shift in granulometric structure of the surface layer of sediment was evident by comparison with adjacent, unfished areas. Mud and sand had been brought to the surface of the tracks and maerl gravel had been sculpted into 3 cm high ridges at the edge of each dredge path. Live maerl was buried up to 8 cm below the sediment surface and biogenic carbonate structures (e.g. maerl thalli, echinoid test plates and bivalve shells) were crushed and compacted. The control plots nearby were relatively unaffected, although some silt had settled on the



Tab. 1. Differences between the sea bed and benthos of Site 1 (unfished) and Site 2 (fished); Clyde Sea area, Scotland

Tab. 1. Differenze rilevate tra il fondale e il bentos del Sito 1 (non soggetto a pesca) e Sito 2 (soggetto a pesca); zona del Clyde Sea, Scozia

Criteria	Site 1	Site 2
Presence of commercial scallop dredge tracks (seen on diver & ROV surveys)	None	Frequent
Molluscan diversity (data from Hall-Spencer, 1998)	107 spp.	82 spp.
Live maerl cover (in 30 0.1 m ² grab samples)	25.3%	1.8%
Mean population density of <i>Aequipecten opercularis</i> (\pm SD in 1 m ² quadrats)	0.27 \pm 0.06 m ⁻² (n=85)	None (n=224)
% <i>Pecten maximus</i> over 7 years old (n = number aged)	83% (n= 106)	None (n=57)
Mean population density of <i>P maximus</i> (\pm SD, in 1 m ² quadrats)	0.06 \pm 0.02 m ⁻² (n=85)	0.01 \pm 0.07m ⁻² (n=224)



upper surface of the maerl.

Sediment in cores taken prior to fishing was vertically stratified at both experimental sites. Site I had a 1-2 cm thick 'open lattice' layer of maerl gravel with a large amount of void space between particles. This 'open lattice' overlaid fine sand and mud mixed with shell and maerl fragments. Site 2 had less fine material and was characterized by a 3-8 cm thick layer of 'open lattice' maerl overlying coarse sand with pockets of fine sand and mud mixed with shell and maerl fragments. Cores taken on control plots had the same pattern of vertical stratification throughout the study whereas core samples taken from test plots immediately after fishing lacked vertical stratification. Thus test plot cores lacked an open lattice layer, had less interstitial space and had a greater proportion of fine particles at the sediment surface. Sampling over the subsequent 3 years revealed a gradual return to a clean, gravelly upper layer of maerl at both sites, presumably due to winnowing away of fine material by water movement. During spring tides, benthic currents exceeded 10 cm s^{-1} at both sites (Hall-Spencer & Atkinson, 1999).

Fig. 5 shows the mean area covered by live maerl in cores taken on test and control plots over a 5 year period. Live maerl was sparsely distributed in the previously fished area and experimental dredging had no discernible effect on their abundance. Table 1 and Fig. 5 show that live maerl was abundant in the previously unfished area (Site 1) and this remained the case on the control plot throughout this study. There was no difference in the amount of live maerl on the test plot immediately after experimental scallop dredging as buried thalli were still alive. Five months after dredging, however, there were 70-80% fewer live thalli in cores taken on test plots than in cores taken prior to fishing with no subsequent signs of recovery over the following 5 years (Fig. 6).

The sculpted ridges and troughs of the dredge tracks were persistent and remained visible within test plots for 2.5 years at Site I and 1.5 years at Site 2. Eventually, they were erased through bioturbation by large infauna (e.g. the thalassinidean *Upogebia deltaura* (Leach) and the holothurians *Neopentadactyla mixta* (Ostergren) and *Thyonidium drummondii* (Thompson)) and the feeding activities of whelks (*Buccinum undatum* L.), crabs (*Cancer pagurus* L., *Liocarcinus depurator* (L.), *Necora puber* (L.)), starfish (*Astropecten irregularis* (Pennant), *Mar hasterias glacialis* (L.), *Asterias rubens* L.) and fish (*Gallus morhua* L., *Pomatoschistus pietas* (Maim), *Pleuronectes platessa* L.). On shallow parts of maerl at Site 2 (-6 to -8 m CD), commercial scallop dredge tracks were erased by wave action during storms (Hall-Spencer & Atkinson, 1999).

DISCUSSION

Recent studies of the intensity and area swept by towed demersal gear indicate that most sedimentary habitats on the European continental shelf have been modified by fishing in the last 100 years (Kaiser *et al.*, 1996; Rijnsdorp *et al.*, 1998). In some places, such as the southern North Sea, fishing is thought now to be the main ecological



Fig. 4: Juvenile *Pecten maximus* (shell length 1.0 cm) on a control plot with living, red coloured maerl (*Phymatolithon calcareum*) at Creag Gobhainn, Loch Fyne, October 1999. Fig. 4: Esemplare di *Pecten maximus* allo stadio giovanile (lunghezza della conchiglia 1,0 cm) su un sito di controllo con presenza di maerl (*Phymatolithon calcareum*) vive, di colore rosso, a Creag Gobhainn, Loch FYne, Ottobre 1999.

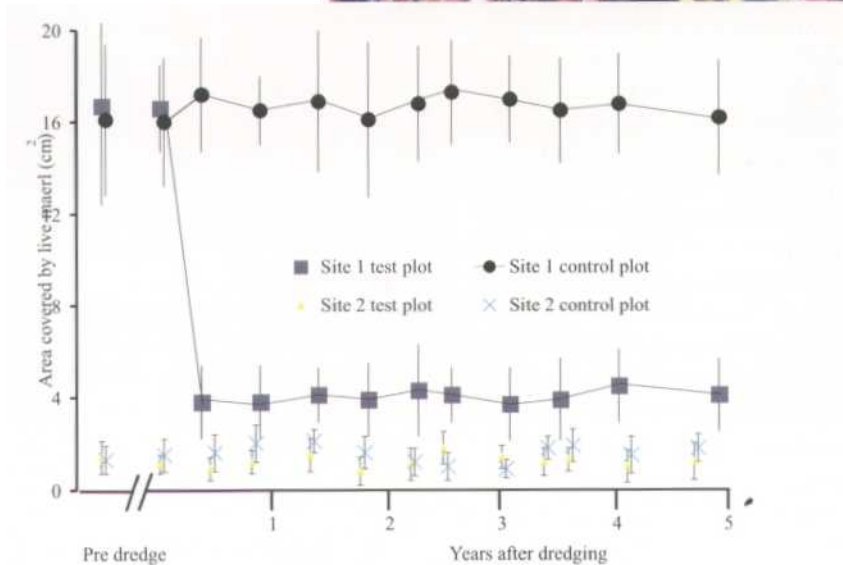


Fig. 5: Mean area of live maerl thalli retained on a 5 mm sieve from 80 cm² sediment cores taken biannually from 1994-1999 on test and control plots at Site 1 (previously unfished) and Site 2 (previously fished), Clyde Sea area, Scotland. Error bars are \pm one SD ($n = 8$).

Fig. 5: Valor medio dell'area ricoperta da talli di maerl vivi, raccolti con un setaccio da 5 mm, ricavata da carotaggi di 80 cm² eseguiti due volte l'anno nel periodo 1994-1999 in zone sperimentali e di controllo nel Sito 1 (non precedentemente sottoposto ad attività di pesca) e nel Sito 2 (precedentemente sottoposto ad attività di pesca), Clyde Sea, Scozia. L'errore equivale a \pm 1 SD ($n = 8$).



Fig. 6: Dead maerl photographed 1 year after experimental dredging on a test plot at Site 1, Creag Gobhainn in Loch Fyne, May 1995. Note loss of the red photosynthetic pigment phycobilin that characterises living maerl.

Fig. 6: Esempari morti di maerl fotografati 1 anno dopo le dragate sperimentali su un area di studio presente nel Sito 1, Creag Gobhainn nel Loch Fyne, Maggio 1995. Da notare la perdita del pigmento fotosintetico rosso ficobilina, caratteristica degli esemplari vivi.



structuring force on the benthos (Lindeboom & de Groot, 1998). A paucity of unimpacted control grounds has made the study of long-term effects of fishing difficult (Lindeboom, 1995), although useful insights have been obtained by sampling close to sea bed obstructions and in areas that are closed to fishing for military purposes (Hall *et al.*, 1993; Ball *et al.*, 2000; Hall-Spencer & Moore, 2000).

Had this study been restricted to previously fished areas of the Clyde Sea area (e.g. Site 2) it may have led to the conclusion that although scallop dredges have dramatic short-term effects (weeks), no long-term effects (years) could be detected against a background of natural variation. However, observations on a previously unfished site (Site 1) revealed highly significant long-term effects. This study adds to a increasing body of evidence that the major impact of towed demersal fishing occurs the first time an area is subjected to fishery pressure (Jennings & Kaiser, 1998). Impacts to maerl beds within the Clyde Sea area are likely to have occurred mainly in the past 30-40 years. Although small-scale scallop dredging began locally in the 1930s, with Scottish yields of $<3 \text{ t y}^{-1}$ (Elmhirst, 1945), it was not until the 1960s that landings increased sharply with the advent of more powerful boats, more efficient dredges and better processing facilities (Mason & Fraser, 1986). The environmental effects of scallop dredging is of continued concern since this is the most important UK mollusc fishery with 11300 t of scallops landed in Scotland alone in 1995 (Ministry of Agriculture, Fisheries and Food, 1996).

The immediate physical impacts of scallop dredging on biogenic algal deposits are similar to those observed on other types of sediment (Caddy, 1973; Eleftheriou & Robertson, 1992). The marked differences found between fished and unfished grounds confirm that maerl beds are especially fragile habitats. Legislation under European Council Directive 92/43/EEC (Conservation of Natural Habitats and of Wild Fauna and Flora, 1992) demands that exploitation of maerl habitats must be compatible with their maintainance at a "favourable conservation status" yet this may not be the case for many European maerl beds. Maerl is more sensitive to fishing than most sedimentary grounds, e.g. shifting sands especially, where evidence of towed gear impact is ephemeral (Hall, 1994; Curry & Parry, 1996; Kaiser *et al.*, 1998; Jennings & Kaiser, 1998). Parallels can be drawn with the effects of towed gear on other biogenic structures such as *Modiolus modiolus* (*L.*) reefs in the Irish Sea (Magorrian *et al.*, 1995) and *Posidonia oceanica* (*L.*) meadows in the Mediterranean (Ardizzone *et al.*, 2000) where impacts are long-lasting since they affect the survival of key habitat-structuring species with poor regenerative abilities.

The integrity of maerl habitats depends upon the survival of a surface layer of slow-growing algae which are unable to withstand prolonged burial due to lack of light. Scallop dredging can have effects (sediment redistribution, live maerl burial) that are clearly discernible for 5 years even on deliberately limited areas of impact (single tows of 3 dredges with $\sim 230 \text{ m}^2$ ground contact). Commercial scallop boats, which typically tow 16 dredges, are estimated to impact 6.6 km² of sea bed per 100 h fishing (Kaiser *et al.*, 1996). Clearly, repeated dredging of the area studied would considerably reduce the amount of living maerl given thallus burial, comminution and



smothering by silt.

That we found no recovery of dredged maerl over 5 years relates to the slow growth (Potin et al., 1990; Canals & Ballesteros, 1997) and poor recruitment of these coralline algae (Irvine & Chamberlain, 1994). These findings add impetus to moves to identify pristine maerl beds for conservation measures in Europe. In SE Spain, the Tabarca Island marine reserve has worked well in protecting local maerl beds from the effects of towed fishing gear (Ramos, 1985; pers. obs.) and similar projects are currently underway to establish management schemes for four candidate "marine Special Areas of Conservation" to protect maerl within the UK (Birkett *et al.*, 1998). In the light of this study, moves to protect biogenic reefs from towed gear impacts should be given a high priority in the future management of coastal resources.

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