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Effects of scallop dredging on macrobenthic communities in west Iceland

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Effects of scallop dredging on benthic communities in Breidafjordur, west Iceland, were investigated by analysing bycatch data from scallop stock assessment surveys and effort data from the commercial scallop fishery. Bycatch constituted 28% of the total catch, with eight benthic macrofaunal taxa alone making up nearly 98% of the bycatch. *Modiolus modiolus* and *Cucumaria frondosa* dominated in terms of abundance and biomass in most of the study area regardless of intensity of fishing effort, although both have been identified as sensitive to fishing in other studies. The macrofaunal benthic community in Breidafjordur consisted mostly of hard-shelled molluscs, holothurians, crabs, and starfish. Emerging epifauna was absent in the samples taken since 1993. These results suggest that our study was carried out within an already altered community that would have suffered the greatest impact during the early years of the scallop fishery. However, the available data are not enough to endorse this assumption with complete certainty.

Keywords: benthic communities, bycatch, dredging, macrobenthos, scallop fisheries.

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Introduction

Scallop dredging can have a considerable impact on benthic fauna and habitats, including increasing the mortality of animals killed or injured following direct contact with the gear (Lindeboom and de Groot, 1998; Jenkins *et al.*, 2001) and that subsequently become more prone to predation (Kaiser and Spencer, 1994a; Veale *et al.*, 2000a).

Long-lived, slow-growing epifaunal species often have a fragile body structure and are especially sensitive to encounters with fishing gear, whereas taxa protected by exoskeletons or thick shells are more resilient (Gislason, 1994; Hall-Spencer and Moore, 2000; Kaiser *et al.*, 2000; Hall-Spencer *et al.*, 2002). The differential mortality among taxa and habitat destruction may cause important short- and long-term changes in faunal assemblages that are regularly disturbed by fishing gear (Collie *et al.*, 1996; Kaiser *et al.*, 1996, 1998; Hill *et al.*, 1999; Bradshaw *et al.*, 2000; Jenkins *et al.*, 2001; Kenchington *et al.*, 2001).

Scallop dredges can also increase the sediment load, relocate boulders, and destroy topographic features (Caddy, 1973; Eleftheriou and Robertson, 1992; Gislason, 1994). However, the magnitude of the impact on benthic communities depends to a large extent on the bottom type. Benthic communities adapted to sandy substrata are, in general, more resilient to disturbance than those living on gravelly, sandshell seabeds (Eleftheriou and Robertson, 1992; Hall, 1998).

Scallop dredging targeting *Chlamys islandica* commenced in Iceland in 1969 (Eiríksson, 1970). The largest scallop beds are located in Breidafjordur, west Iceland, and they have provided between 80% and 100% of the total annual catches since the beginning of the fishery (Anon., 2003). During the years 1983–1987, the years of biggest catches, up to 25 vessels took part in the Breidafjordur scallop fishery (Eiríksson, 1997). However, because of declining stocks, just eight vessels between 15 m and 33 m long took part in the fishery in 2001, and four in 2003. Fishing effort has always been unevenly distributed throughout the year. There is very little dredging from March to July, only 0.5–2% of the annual catch being landed then. Catch percentage for the remaining months ranged from 6% to 19% of the total, with maximum landings (always >11%) between September and December.

The bycatch data analysed here have been collected since 1993 during the annual scallop stock assessment surveys within Breidafjordur. The fishing effort data are from commercial fishery logbooks and span the years 1972–2001.

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This contrasts with many previous studies that have investigated the effects of scallop dredging using manipulative field experiments, in which the size of the study areas was much smaller and the time scales shorter (Eleftheriou and Robertson, 1992; Currie and Parry, 1994; Jenkins *et al.*, 2001).

The goal of this study was to investigate the effects of dredging on the spatial and temporal trends of non-target species caught as bycatch in the scallop fishery in Breidafjordur, west Iceland.

Material and methods

The scallop fishery is concentrated in the inner part of Breidafjordur (Figure 1). The scallop beds are generally between 20 m and 70 m deep. Their distribution is greatly influenced by bottom topography, characterized by many channels with orientation SW−NE or W−E within which the scallops accumulate (Eiríksson, 1986). For management purposes, the scallop fishing grounds are divided into 18 rectangles measuring 5 × 6 nautical miles (9.3 × 11.1 km).

Survey and commercial data

Bycatch data were collected in Breidafjordur between 1993 and 2001 during the annual scallop stock survey carried out in April. Between 108 and 130 samples were taken annually at fixed stations, their geographical position recorded with GPS. The dredge originally used in the surveys was a sledge dredge 1.5 m wide and weighing 470 kg, substituted since 1998 by a roller dredge of Icelandic design, 1.2 m wide and 835 kg in weight. Towing speed was 3−5 knots and tow duration 5−10 min. The average swept area and standard deviation (s.d.) per tow were 978 ± 150 m^2. Total catch from each tow was weighed and a random subsample of 25 kg was taken. Scallops and all bycatch species were subsequently counted and weighed to estimate their abundance and biomass.

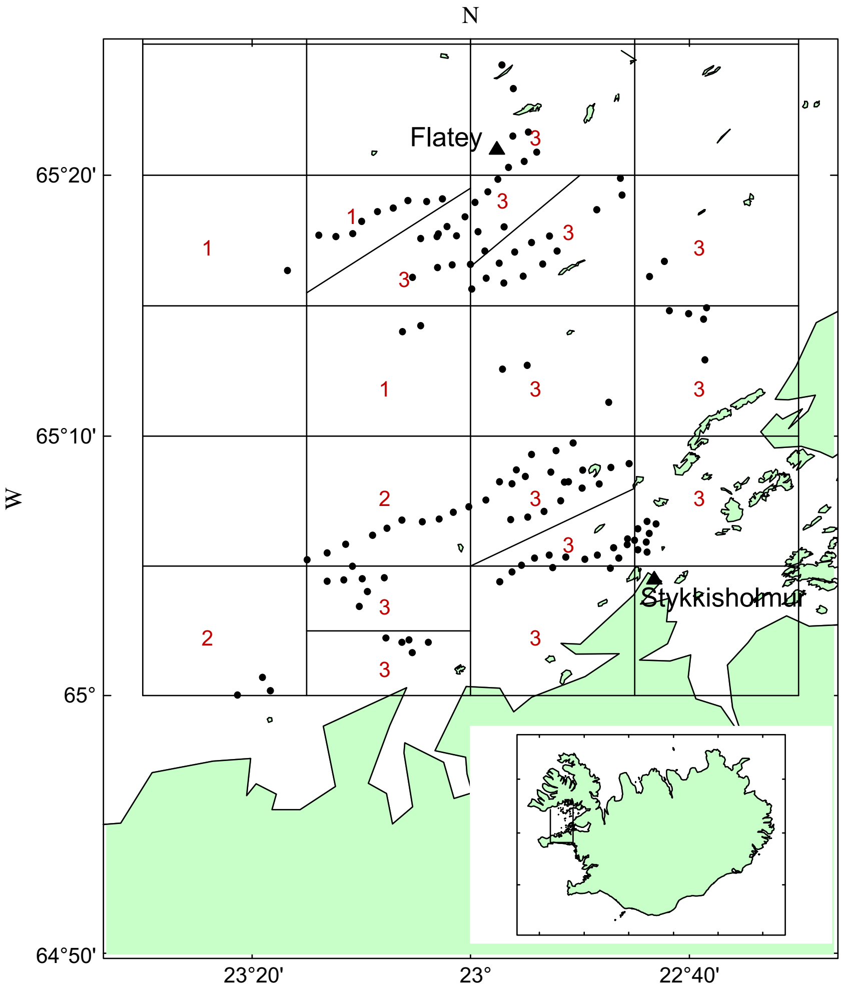

Figure 1. Map of the study area showing its location within Iceland (insert). The grid shows the division into rectangles used in surveys and commercial fishing, and the dots show the sampling stations. Note that four of the rectangles are split into two parts.

Table 1. Average biomass, abundance, and production ± s.d. for all bycatch species captured in annual scallop stock assessment surveys during the period 1993−2001. The last column indicates the percentage of sampling tows in which each taxon was found.

Taxa	Biomass (kg 1 000 m^{-2})	Abundance (number 1 000 m^{-2})	Production (kg 1 000 m^{-2})	% Tows
Anthozoa				
Unidentified species	1.61 ± 1.08	24.23 ± 20.94	0.51 ± 0.35	0.9
Mollusca				
Arctica islandica	5.22 ± 22.49	56.09 ± 80.84	1.28 ± 3.55	11.2
Astarte borealis	2.37 ± 17.24	155.09 ± 296.14	0.75 ± 2.31	17.5
Astarte crenata	1.39 ± 1.58	271.22 ± 318.67	0.89 ± 1.02	1.8
Astarte elliptica	1.27 ± 2.90			1.7
Clinocardium ciliatum	1.45 ± 2.10	48.48 ± 57.97	0.56 ± 0.78	9.8
Musculus discors	0.10 ± 0.03			0.2
Musculus niger	0.49			0.1
Modiolus modiolus	33.96 ± 68.59	961.05 ± 2 061.27	12.73 ± 25.78	93.6
Mytilus edulis	2.99 ± 9.37	91.31 ± 207.64	1.13 ± 3.31	12.8
Serripes groenlandicus	1.77	50.62	0.68	0.2
Buccinum undatum	3.68 ± 4.83	151.18 ± 176.94	1.53 ± 1.93	75.3
Neptunea despecta	2.08 ± 1.95	30.91 ± 35.35	0.65 ± 0.58	28.5
Colus islandicus	0.44 ± 0.55	28.12 ± 21.64	0.19 ± 0.18	0.3
Crustacea				
Carcinus maenas	3.46	95.62	1.31	0.1
Eupagurus bernhardus	2.21 ± 2.92	101.53 ± 113.51	0.92 ± 1.12	67.4
Hyas araneus	12.41 ± 13.29	130.84 ± 129.27	3.58 ± 3.72	66.5
Munida tenuimana	0.06	28.12	0.05	0.1
Pandalus montagui	0.14 ± 0.23	30.23 ± 8.83	0.08 ± 0.10	0.6
Sclerocrangon boreas	0.34 ± 0.25	24.84 ± 13.36	0.17 ± 0.12	0.6
Sclerocrangon ferox	0.38 ± 0.41	28.80 ± 2.55	0.18 ± 0.15	0.2
Echinodermata				
Asteroidea	6.49 ± 14.89	69.54 ± 67.17	1.69 ± 3.06	78.2
Cucumaria frondosa	37.40 ± 41.91	82.59 ± 84.89	7.06 ± 7.56	68.1
Echinus esculentus	16.00 ± 24.86	182.15 ± 282.39	4.59 ± 7.04	83.9
Strongylocentrotus droebachiensis	10.61 ± 16.58	254.57 ± 289.77	3.76 ± 5.29	80.4
Tunicata				
Unidentified species	2.84 ± 3.18			0.4
Cnidaria				
Unidentified species		399.56		0.1
Pisces				
Agonus cataphractus	0.43 ± 0.34	19.04 ± 12.20	0.18 ± 0.12	1.1
Ammodytes marinus	0.02 ± 0.00	22.50	0.02 ± 0.00	0.1
Ammodytes tobianus	0.07 ± 0.04	21.46 ± 9.17	0.05 ± 0.02	0.6
Anarhichas lupus	2.78 ± 5.60	5.42 ± 7.67	0.43 ± 0.85	0.2
Cottunculus microps	5.34 ± 2.38	70.19 ± 52.18	1.53 ± 0.17	0.2
Gadus morhua	2.44 ± 4.97	110.66 ± 442.62	0.50 ± 0.88	1.2
Hippoglossoides platessoides	0.80 ± 1.23	23.96 ± 28.78	0.30 ± 0.44	0.6
Leptagonus decagonus	0.07 ± 0.12	25.48 ± 11.19	0.04 ± 0.06	0.3
Limanda limanda	1.58 ± 1.20	29.7 ± 17.5	0.48 ± 0.29	1.5
Lumpenidae	0.27 ± 0.11	45.45 ± 18.45	0.17 ± 0.07	
Microstomus kitt	3.73 ± 4.73	19.69 ± 8.75	0.75 ± 0.79	0.2
Myoxocephalus scorpius	3.41 ± 5.21	34.56 ± 26.71	0.83 ± 1.14	1.9
Pholis gunnellus	72.7 ± 105.6	20.3 ± 7.1	47.2 ± 57.9	0.3
Pleuronectes platessa	5.27 ± 6.48	19.8 ± 15.3	1.26 ± 1.37	1.6
Pollachius virens	0.66 ± 0.00	30.4	0.29	0.1
Dypturus batis	7.62 ± 0.00	89.99	2.30 ± 0.00	
Ambliraja radiata	2.78 ± 4.38	9.22 ± 7.19	0.55 ± 0.70	1.0
Triglops murrayi	2.60 ± 5.53	74.24 ± 98.57	0.94 ± 1.91	0.4

Catch and fishing effort data from the commercial fishery spanning the period 1972−2001 were obtained from logbooks, mandatory in the Icelandic scallop fishery since 1972. Fishing effort is recorded as dredging time, the time elapsed between the first and last hauls of each fishing day.

Data analysis

The rectangles were used as a spatial unit for the analysis, because logbooks lack position data for individual tows. The area of the scallop beds within each rectangle is known. Therefore, assuming that effort is evenly distributed within the scallop beds, fishing pressure (as a percentage of scallop bed dredged) was estimated with the equation used in the ICES Ecosystem Effects of Fishing Working Group Report (Anon., 1991):

$$\%\ \text{Area of scallop bed dredged} = 100\left(dvt\right)A_i^{-1},$$

where d is the dredge width (km), v the towing speed of the vessel (km h^{-1}), t the time spent fishing (h), and A is the area of scallop bed i (km^2).

Fishing is mainly during autumn, whereas the stock assessment surveys are in April. For this reason, annual fishing effort was estimated for the period April−March, called the fishing year hereafter, to compare the data of any given survey with the fishing pressure of the previous 12 months.

For bycatch data, we considered convenient to use the variable production because it includes biomass and abundance data. Production was estimated according to Warwick and Clarke (1993) and Veale *et al.* (2000b):

$$P = \left(BA^{-1}\right)^{0.73} \times A,$$

where B is the biomass (kg) and A the abundance. BA^{-1} is the mean body size and 0.73 the average exponent of the regressions of annual production and body size for macrobenthic invertebrates (Brey, 1990). As the area dredged during the surveys is known, production per km^2 was used to permit comparisons with fishing pressure. The production data ($P\ \text{km}^{-2}$ hereafter) required log-transformation, which decreased the variance but also conferred more weight to the scarcer species. Thus, scarcer species were eliminated or aggregated to the taxonomic level of genus.

One-way ANOVA tests were performed to investigate whether $P\ \text{km}^{-2}$ of the most abundant bycatch taxa differed between the sledge and roller dredges. The remaining taxa were excluded because they were too scarce to permit statistical analysis. Multivariate analysis was performed on survey and commercial fishery data. Hierarchical cluster analysis (Clarke and Warwick, 2001) was used on log-transformed mean $P\ \text{km}^{-2}$ data of invertebrate bycatch taxa from each rectangle, to categorize the rectangles into groups (called areas hereafter) within which the faunal assemblage was homogeneous and environmental conditions were assumed similar. Differences between the faunal assemblages of the different areas were compared with similarity percentage (SIMPER) analysis and analysis of similarities (ANOSIM; Clarke and Warwick, 2001).

The diversity indices, number of species (S), Pielou's evenness (J'), and Shannon's diversity (H', $\log_e$), were estimated for each area using mean production data grouped by area and year. To determine the level of disturbance, the relationship between biomass and abundance of bycatch species within each area was analysed with partial and cumulative abundance/biomass comparison (ABC) plots (Warwick and Clarke, 1994). The relationship between mean production of bycatch taxa and total fishing pressure within each area was investigated for each of the most abundant taxa with regression tests. The data were aggregated per rectangle and year and were not transformed because the deviation from normality was very slight.

Three-way ANOVA was used to test for differences in log-transformed $P\ \text{km}^{-2}$ of the most abundant taxa among years, areas, and depths, as well as the existence of interactions between year and area, the only factors that were orthogonal to each other. Depth was considered as a factor because the same stations are sampled every year.

Results

In all, 42 bycatch taxa were recorded in the annual scallop stock assessment surveys from 1993 to 2001, including 19 demersal fish species, two pelagic invertebrate taxa, and the burrowing bivalve *Arctica islandica*. These taxa were excluded from the analysis because the dredge samples them inadequately. Only eight of the remaining benthic taxa were present in more than 60% of the tows (Table 1).

The estimated total weight of benthic bycatch in the surveys was 97.7 t or 32.8% of the weight of the scallop catch. Biomass of the 10 most abundant bycatch species (*Modiolus modiolus*, *Cucumaria frondosa*, *Echinus esculentus*, *Hyas araneus*, *Strongylocentrotus droebachiensis*, Asteroidea, *Buccinum undatum*, *Eupagurus bernhardus*, *Arctica*

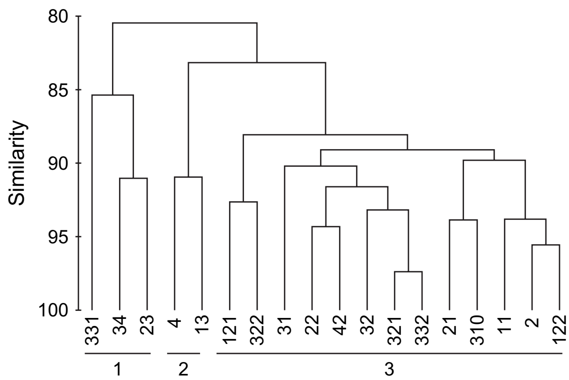

Figure 2. Results of the cluster analysis performed on average production data for the period 1993−2001 within each rectangle, indicating the three areas in which the study area was divided and the rectangles included within each of them.

Table 2. Results from SIMPER analysis. Mean and standard deviation of production (t km^{-2}) of the macrobenthic bycatch taxa in each of the three areas. Columns 1−2, 1−3, and 2−3 show the contribution of each taxon to the dissimilarity between areas in pairwise comparisons, expressed as a percentage.

Taxa	Area 1	Area 2	Area 3	1−2	1−3	2−3
Mollusca						
Astarte spp.		0.7 ± 0.4	3.8 ± 4.0	18.6	18.2	5.9
Buccinum undatum	3.5 ± 4.1	0.4 ± 0.2	9.4 ± 6.5	4.5	3.7	8.1
Clinocardium ciliatum	1.0 ± 90.4	0.4 ± 0.4	2.7 ± 2.8	4.2	6.4	6.8
Modiolus modiolus	119.8 ± 79.8	5.1 ± 0.6	85.2 ± 4.9	8.2		7.4
Mytilus edulis		1.3 ± 1.1	5.3 ± 5.9	20.1	20.8	4.1
Neptunea despecta	3.0 ± 0.7	2.3 ± 0.1	4.7 ± 1.7			
Colus islandicus		0.1 ± 0.2	0.3 ± 0.8	8.4		9.1
Crustacea						
Eupagurus bernhardus	1.8 ± 1.3	0.5 ± 0.6	6.5 ± 3.5	5.2	3.8	8.3
Hyas araneus	6.8 ± 8.5	1.3 ± 1.7	24.1 ± 8.8	5.6	4.8	10.1
Pandalus montagui			0.2 ± 0.3		5.5	6.3
Sclerocrangon spp.			0.3 ± 0.6		5.2	5.9
Echinodermata						
Asteroidea	3.7 ± 3.6	11.4 ± 1.4	10.9 ± 5.5	4.3		
Cucumaria frondosa	26.7 ± 28.3	10.9 ± 2.6	49.9 ± 22.1		3.8	4.1
Echinus esculentus	34.9 ± 49.5	6.9 ± 0.7	24.6 ± 9.3		3.5	
Strongylocentrotus droebachiensis	9.6 ± 13.3	0.4 ± 0.4	25.2 ± 18.4	6.6	6.1	11.1
Cnidaria						
Anthozoa	0.6 ± 1.1		0.9 ± 1.6	6.7	8.4	6.7

islandica, and *Neptunea despecta*) amounted to 96.7 t or 98.9% of the total biomass of benthic bycatch. *M. modiolus* and *C. frondosa* alone constituted 32.3% and 25.3%, respectively, or 31.6 t and 24.6 t of the benthic bycatch. Biomass of the remaining most abundant taxa ranged from 0.5% or 0.56 t for *N. despecta* to 12.8% or 12.4 t for *E. esculentus*.

Catchability of scallops was significantly greater with the roller dredge than with the sledge dredge when tested with one-way ANOVA ($F = 29.3$, $p < 0.001$), but only three of the bycatch taxa differed significantly in P km^{-2} between the two dredge types: *N. despecta* ($F = 4.35$, $p = 0.03$), *E. esculentus* ($F = 31.08$, $p < 0.001$), and *C. frondosa* ($F = 38.48$, $p < 0.001$). However, the annual variability of their abundance in the catch was so large that it was difficult to ascertain the difference was due to the dredge alone.

The survey area was split into three areas by cluster analysis (Figure 2). Areas 1 (4.1% of the total scallop bed area,

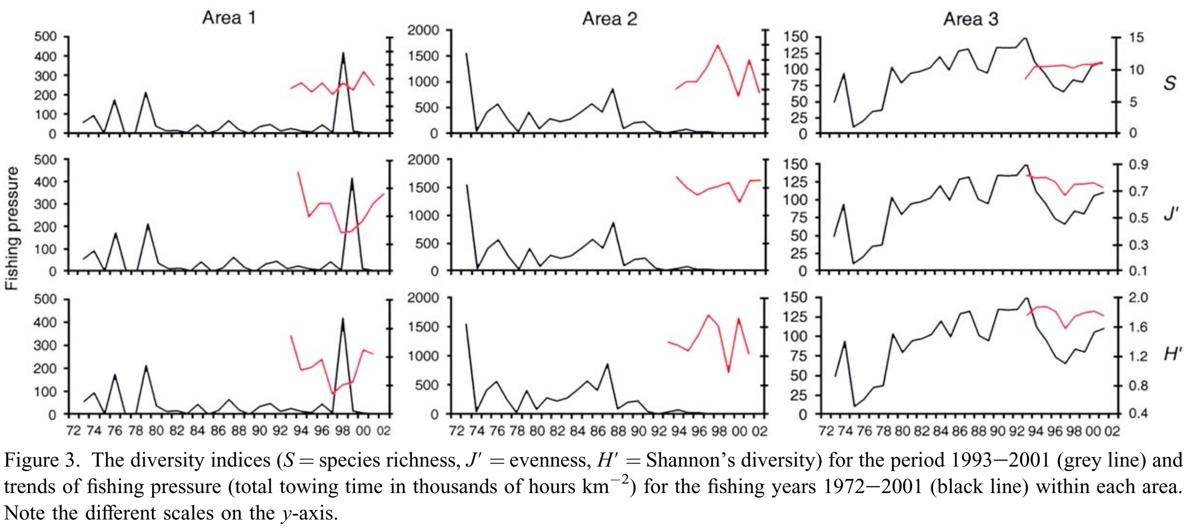

Figure 3. The diversity indices (S = species richness, J' = evenness, H' = Shannon's diversity) for the period 1993−2001 (grey line) and trends of fishing pressure (total towing time in thousands of hours km^{-2}) for the fishing years 1972−2001 (black line) within each area. Note the different scales on the y-axis.

Table 3. Mean and standard deviation (s.d.) of the diversity indices, species richness (S), evenness (J'), and Shannon's diversity (H') for each area.

	S		J'		H'	
Area	Mean	s.d.	Mean	s.d.	Mean	s.d.
1	6.79	2.41	0.54	0.19	1.03	0.48
2	7.62	2.63	0.70	0.10	1.39	0.34
3	10.43	1.35	0.76	0.10	1.77	0.26

mean depth 47.4 m) and 2 (5.7%, 36.1 m) are in the outer part of the study area, whereas area 3 (90.2%, 35.7 m) is in the inner part of the study area, towards the head of the fjord. Production of the most abundant bycatch taxa within each area is shown in Table 2.

An ANOSIM test revealed that the benthic assemblages differed significantly between the three areas (global $R = 0.8$, $\alpha = 0.001$). The pairwise tests concluded that areas 2 and 3, and 1 and 3 were highly significantly different, with R-values of 0.82 and 0.84 and p-values of 0.01 and 0.002, respectively. Although R-values in the range 0.5−0.74 indicate similarities in the faunal assemblages of two areas that are clearly different (Clarke and Gorley, 2001), the test comparing areas 1 and 2 was not significant despite R being 0.6. Statistical power was probably too low to detect these differences because of the limited number of samples.

Species richness showed a slightly increasing trend in areas 1 and 3, but it was very variable in area 2. Trends for evenness and Shannon's diversity showed an overall decrease during the period 1993−2001 and were similar within areas 1 and 3. They differed slightly in area 2, where evenness had a decreasing trend but diversity showed no trend (Figure 3). Mean values of the diversity indices are listed in Table 3.

Spatial distribution of fishing effort shifted between areas during the period 1973−2001. In area 1 fishing effort was

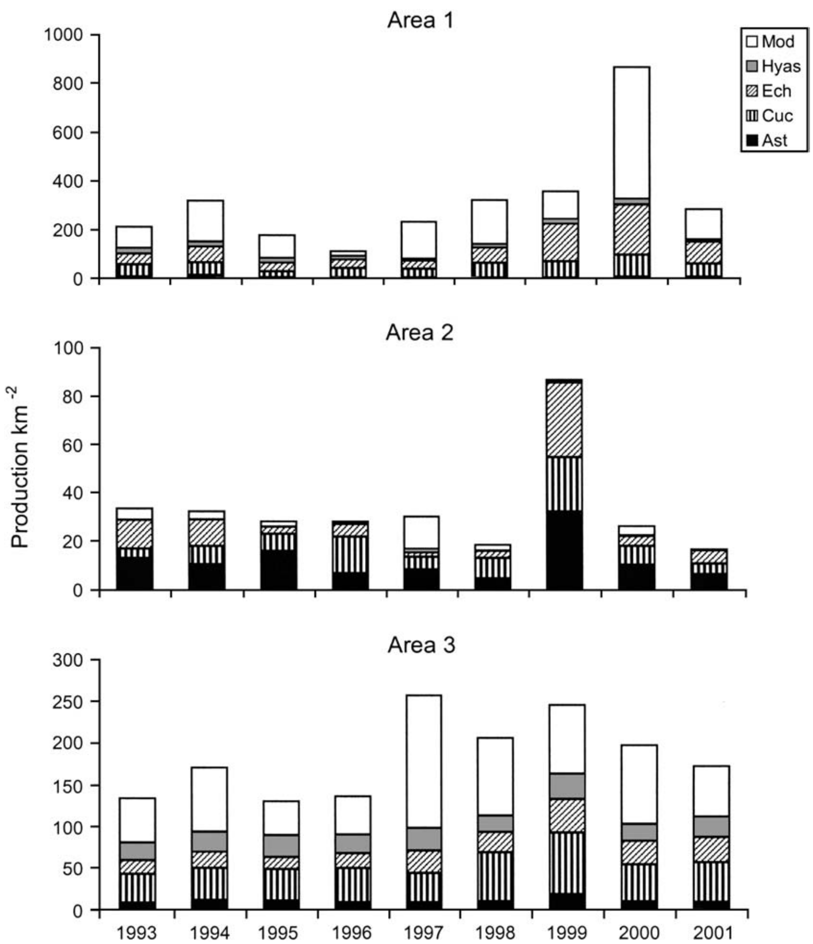

Figure 4. Annual variability in production ($P\ km^{-2}$) of the bycatch taxa that have a major influence on the diversity and evenness indices within each area. Note the different scales on the y-axis. Mod = *Modiolus modiolus*, Hyas = *Hyas araneus*, Ech = *Echinus esculentus*, Cuc = *Cucumaria frondosa*, Ast = Asteroidea.

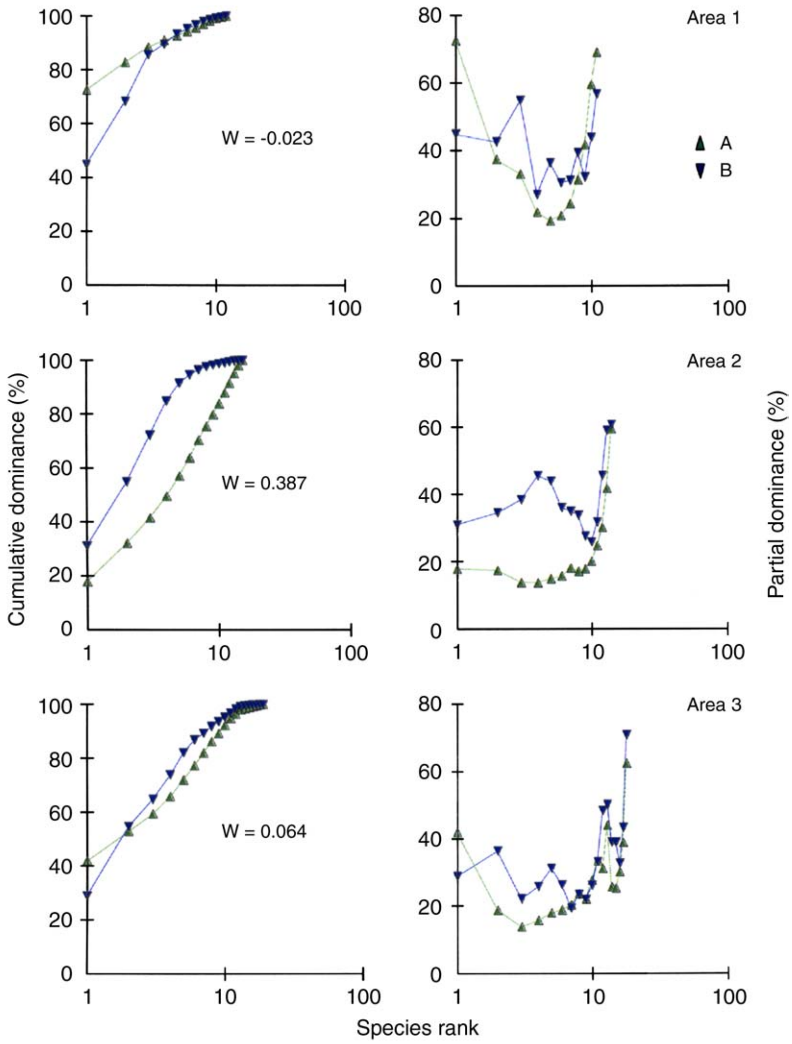

Figure 5. ABC plots of cumulative (left column) and partial dominance (right column) for each of the three areas. *A* is the average abundance in number km^{-1} and *B* the average biomass in g km^{-2}.

most pronounced from 1973 to 1979, but it still accounted for <27% of the total annual effort in Breidafjordur. Since 1980 it decreased there to <10%, except for the sudden increase in 1998 as a consequence of intensive fishing of a small scallop bed. Fishing effort in areas 2 and 3 varied greatly. In area 2 it decreased from 12−48% to <10% during the years 1987−1991, and it has remained very low since. However, fishing effort in area 3 increased steadily from 47−79% during the years 1980−1987 to 82−100% since 1988, except for a decline between 1993 and 1997 (Figure 3).

Evenness and diversity seemed influenced by variability in production of the dominant species in each area, often with lower values in years of enhanced production of *M. modiolus*, Asteroidea, *E. esculentus*, and *C. frondosa* (Figure 4). However, interpretation of the trends of diversity indices is not straightforward. Evenness was positively correlated with fishing pressure in area 3 ($F = 5.75$, $p = 0.012$), but dispersion was rather high (0.05). No other significant correlations were found.

The ABC plots based on abundance and biomass data of the bycatch taxa for the period 1993−2001 showed more pronounced signs of disturbance and had lower values for the W statistic in areas 1 and 3 than in area 2 (Figure 5).

Significant positive relationships between total fishing pressure and mean P km^{-2} were found in area 1 for *H. araneus* ($F = 7.98$, $p = 0.030$) and in area 3 for *B. undatum* ($F = 36.24$, $p < 0.001$), *E. bernhardus* ($F = 14.51$, $p < 0.001$), *H. araneus* ($F = 9.16$, $p = 0.003$), and *S. droebachiensis* ($F = 18.04$, $p < 0.001$; Figure 6).

P km^{-2} of all taxa differed significantly between areas, and for most taxa also between years and depths. The exceptions were Asteroidea and *H. araneus*, with no significant differences between years, and *N. despecta*, with no significant differences between depths. The interaction of

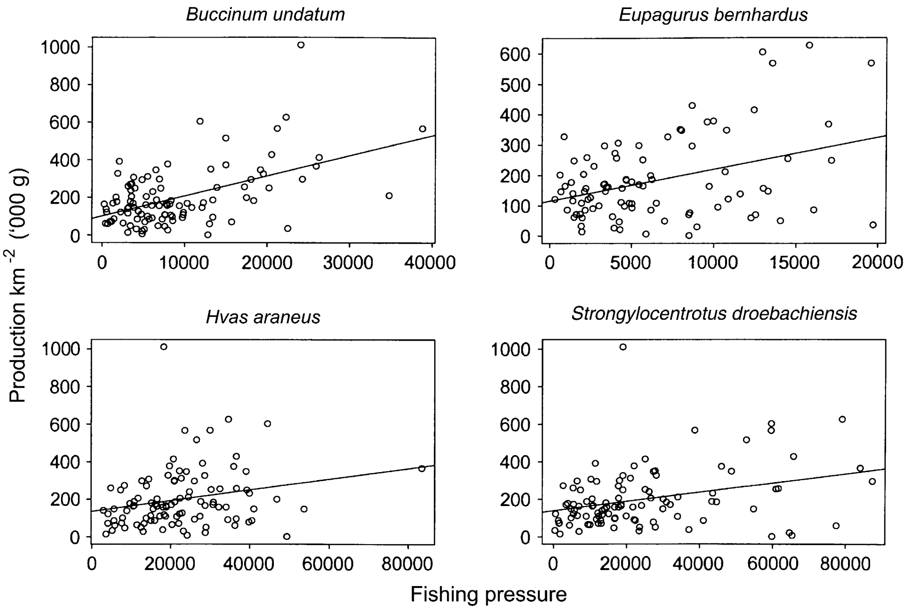

Figure 6. Regression of average production and fishing pressure for four species in area 3 for which the slope and intersection were significantly different from zero. The data were aggregated per fishing year and rectangle.

area–year was significant for *B. undatum*, *E. esculentus*, and *M. modiolus* (Table 4).

Discussion

The macrobenthic community in Breidafjordur was in several aspects similar to disturbed communities elsewhere (Riesen and Reise, 1982; Collie *et al.*, 1997, 2000; Freese *et al.*, 1999; Hill *et al.*, 1999; Kaiser *et al.*, 2000). Diversity and species richness were in general low, and among the dominant taxa were starfish, large bivalves, hard-shelled gastropods, and crabs.

All three diversity indices showed their highest values in area 3 despite the higher fishing pressure there, although it had been anticipated that species richness and diversity would be greater at undisturbed sites (Currie and Parry, 1994; Collie *et al.*, 1997; Veale *et al.*, 2000b), which is not the case here.

The increase in species richness in disturbed communities is usually concomitant with decreases in diversity and evenness caused by the punctual increase of one or more species (Warwick, 1996). We agree with this statement, because the trends of evenness and diversity generally decreased. *M. modiolus* and *C. frondosa* were dominant in areas 1 and 3, both areas of higher fishing pressure. Neither species seemed to be vulnerable to handling of the catch on fishing vessels. Unlike crabs and sea urchins, which very often had broken limbs and/or tests, holothurians and bivalves rarely suffered clear physical damage (EGG, pers. obs.). We lack data to investigate whether physiological stress increases significantly the mortality of these species, but their abundance suggests that they are rather resilient. However, they have been described as sensitive to fishing activities elsewhere (Kaiser *et al.*, 1996; Collie *et al.*, 1997, 2000; Hall-Spencer and Moore, 2000). The abundance of *E. esculentus* and *S. droebachiensis* in area 3, where fishing effort is greatest, was unexpected, but it may be explained by the proximity of extensive kelp forests.

The positive significant relationship between production of *B. undatum*, *E. bernhardus*, and *H. araneus* and fishing pressure was most likely a long-term scavenger response, as shown in other studies (Kaiser and Spencer, 1994a, b; Currie and Parry, 1996; Collie, 1998; Lindeboom and de Groot, 1998). The ANOVA test detected strong location effects on $P\,\mathrm{km}^{-2}$ for all taxa. These could be due to differences in bottom type and biotic factors among areas, but we lack data to discuss these results further. *M. modiolus*, *E. esculentus*, and *C. frondosa* had significant differences in $P\,\mathrm{km}^{-2}$ among years. *E. esculentus* and *C. frondosa* also gave significant results when testing the differences among dredges, so increased catchability may account partly for their annual variability. Nevertheless there were few significant interactions of area–year, suggesting that variability in $P\,\mathrm{km}^{-2}$ among areas and years follows similar trends for most taxa. Depth also influenced greatly the $P\,\mathrm{km}^{-2}$ of most taxa. Considering that areas 2 and 3 had similar mean depth, the question remains to what extent are the differences in macrobenthic fauna between areas 2 and 3 attributable to the low fishing pressure in area 2 since 1988, as the ABC plot suggests.

The data used in this study do not show evidence of any major impact of scallop dredging on the distribution and

Table 4. Results of the three-way ANOVA used to test for effects of area, year, depth, and area–year interaction on production of the most abundant bycatch taxa.

Taxon and parameter	d.f.	*MS*	*F*	*p*	Taxon and parameter	d.f.	*MS*	*F*	*p*
B. undatum					*N. despecta*				
Area	2	53.85	34.11	0.000	Area	2	14.34	21.27	0.000
Year	8	3.68	2.33	0.017	Year	8	1.58	2.34	0.019
Depth	82	1.52	0.96	0.569	Depth	69	0.48	0.71	0.945
Area:year	13	3.66	2.32	0.005	Area:year	16	1.08	1.61	0.068
E. bernhardus					*H. araneus*				
Area	2	50.26	30.16	0.000	Area	2	68.45	55.65	0.000
Year	7	3.76	2.25	0.028	Year	8	1.49	1.21	0.288
Depth	80	1.88	1.13	0.216	Depth	80	0.97	0.79	0.900
Area:year	13	2.14	1.28	0.217	Area:year	9	2.45	1.99	0.037
Asteroidea					*C. frondosa*				
Area	2	9.48	5.29	0.005	Area	2	34.32	31.59	0.000
Year	8	3.53	1.97	0.046	Year	8	7.84	7.21	0.000
Depth	79	2.52	1.4	0.014	Depth	81	1.33	1.22	0.100
Area:year	16	2.32	1.29	0.193	Area:year	16	0.88	0.81	0.665
E. esculentus					*S. droebachiensis*				
Area	2	166.84	96.98	0.000	Area	2	225.37	102.84	0.000
Year	8	12.05	7.00	0.000	Year	8	9.12	4.16	0.000
Depth	77	2.13	1.23	0.089	Depth	81	3.51	1.60	0.001
Area:year	16	5.75	3.34	0.000	Area:year	14	2.89	1.32	0.189
M. modiolus									
Area	2	257.09	144.69	0.000					
Year	8	25.75	14.49	0.000					
Depth	83	2.78	1.55	0.001					
Area:year	16	2.91	1.64	0.053					

abundance of bycatch taxa. However, scallop dredging started in 1972, whereas bycatch data are available only since 1993, and the available information from non-dredged areas is limited. This is a major drawback in this study, because we cannot compare the dredged areas with other locations that are rarely or not dredged at all. Previous studies (Kaiser *et al.*, 1996; Jennings and Kaiser, 1998; Hall-Spencer and Moore, 2000) concluded that the effects of fishing are more severe in the early phases of a fishery, and involve the loss of epibenthic biogenic structures and their associated fauna. It is therefore most likely that scallop dredging had already altered the benthic community by removing effectively the sensitive species before bycatch data collection started. Therefore, the current macrofaunal assemblage might be very resilient towards physical disturbance; hence, the apparently small effect of fishing effort on the benthic community in Breidafjordur.

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