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1. INTRODUCTION

This Volume 2 Ecodredge Investigations, describes the experimental work carried out over the course of 3 years 1999-2001. It is divided into 3 main [Sections 3.1](#) environmental impact which includes physical, chemical, biological both for individuals and community impacts, [3.2](#) covers selectivity both species and size, and finally [3.3](#) covers experiments with design of dredge components and whole dredge design.

Finally there is a discussion [Section 4](#) which includes cross references to all the sections in the Review and the Investigations volume and also updates the literature to the end of 2002. [Section 4](#) includes discussion of the measurement of environmental effects and selectivity of dredge fisheries. The main findings of the project in terms of the environmental effects of dredging in relation to selectivity, dredge design and fishing effort are discussed and considered in relation to management measures.

2. OBJECTIVES

The objectives of these investigations were as follows (Numbers refer to objectives of the Ecodredge project);

- 2) To develop techniques appropriate to each fishery to study the selectivity and environmental effects of dredging.
- 3) To examine and quantify the role of dredge components in the selectivity of *Aequipecten opercularis* and *Pecten maximus* and to consider possible technical measures to improve selectivity.
- 4) To study the selectivity of dredges used on Portuguese clams.
- 5) To examine the mode of action of dredging on affected species and the seabed, and to develop means to reduce physical impacts.
- 6) To study the incidental mortality, biological stress, ecological, physical and chemical environmental effects of dredging on a seasonal basis.
- 7) To study the role of selectivity components and dredge design on the environmental effects of dredging.

3. INVESTIGATIONS

3.1 Environmental effects

Environmental effects of dredging are reviewed in [Review Section 5](#). In this Section the experimental work on the environmental effects of dredging are described. These effects are divided into; physical, chemical and biological effects. The biological effects are further categorised into behavioural, physiological and damage effects on individuals, finally there is a section on community effects.

3.1.1. Physical Effects

Introduction

The four papers in this section use instrumentation, modelling, simulations, video observations and estimates of damage indices to describe the physical effects of dredging on the seabed habitats and species encountered. The aim was to understand the mechanical effects of the harvesting process with a view to minimising these effects on the habitats and species encountered.

3.1.1.1 SCALLOP (*PECTEN MAXIMUS*) DREDGE INTERACTIONS WITH THE SEABED AND SCALLOPS

Introduction

Spring toothed (Newhaven) dredges are widely used in UK in some French scallop fisheries. They were developed in the 1970s and early trials showed (Chapman et al. 1977) that, in contrast to the fixed tooth dredges, they caught fewer stones and other matter for an equivalent quantity of scallops. Spring toothed dredges did not build up stones in front of the dredge whilst in motion and there were no mounds of material at the side of the dredge tracks post dredging. They are adapted for use on areas where large stones and rocky outcrops occur; the springs enable the dredges to work these areas and catch the scallops living in the areas of sand and gravel sediments between the harder substrates.

Fishermen use various empirical methods for tuning the dredges, through adjustment of springs and the warp-depth ratio (Melhuish 1995) but there remains very little information on the action of the dredges on the seabed and scallops. Cover and Sterling (1994) have studied dredge dynamics in Australian toothed mud dredges ([Review Section 3.3.1](#)). The results indicated high point loading and intermittent action on the seabed meant that very high intermittent contact pressures could occur between the dredge and the seabed. For spring toothed dredges most of the high static contact loading came at the teeth, although no studies of their dynamics were performed. Thus this study centres on the action of the teeth on seabed sediments and factors affecting dredge dynamics.

In this Section we describe the development of experimental instrumentation for sprung toothed scallop dredges and its use in describing the interactions of dredges with the seabed. The approach used devised and constructed an instrumentation package designed to record data on several key parameters of dredge dynamics and to relate these results to available models of dredge teeth being drawn through a seabed soil. Also described is the design and output of the instrumented scallop used in [Section 3.1.1.2](#) and a description of video observations of dredge dynamics and scallop capture taken by Port Erin Marine Laboratory using their vessel RV Roagan.

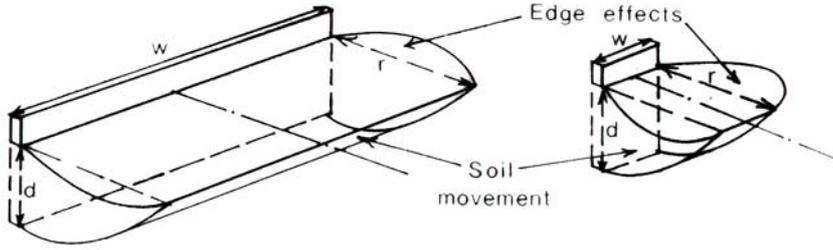
Theoretical considerations

The force required to pull the teeth of a scallop dredge through the seabed is a function of several factors. The primary force is that required to remove the soil in front of the tooth. However consideration should be given to the friction forces between the sides of individual teeth against the sediment.

Many analyses of this type are based on a back analysis of data, from which empirical correlations have been developed. In this case, this is not possible as no data exists and only limited information regarding soil type is available. However theoretical models are available to assess the force required to drag a tine through a soil. These have largely been developed as part of agricultural engineering to assist the design of earth tillage equipment and are briefly discussed below.

The soil failure in front of a moving body is a passive failure as shown in Figure 1a below. For a wide body at shallow depth, the end effects are small and may be

ignored. However as the depth to width ratio decreases (Figure 1b), the end effects become significant and must be taken into account.



a : Large width/depth ration

b : Small width/ depth ratio

Figure 1 : Effect of width to depth ratio in soil failure.

Mathematical models for this analysis have been proposed (Spoor and Godwin 1978 and McKyes and Ali 1977). Both are similar in approach and give similar results. For the purposes of this study, use has been made of McKyes and Ali. The basic equation 1 for this analysis is similar to Terzaghi's bearing capacity equation (Smith 1982) and used a variety of N factors to describe soil properties.

$$P := (\gamma \cdot d^2 \cdot N_\gamma + c_u \cdot d \cdot N_c + q \cdot d \cdot N_q + c_a \cdot d \cdot N_{ca}) \cdot w \quad 1$$

where :

$$N_\gamma := \frac{1}{2} \cdot \frac{\cotd(\alpha) + \cotd(\beta)}{(\cosd(\alpha + \delta) + \sind(\alpha + \delta) \cdot \cotd(\beta + \phi))}$$

$$N_c := \frac{(1 + \cotd(\beta) \cdot \cotd(\beta + \phi))}{\cosd(\alpha + \delta) + \sind(\alpha + \delta) \cdot \cotd(\beta + \phi)}$$

$$N_q := 2 \cdot N_\gamma$$

$$N_{ca} := \frac{(1 - \cotd(\beta) \cdot \cotd(\beta + \phi))}{\cosd(\alpha + \delta) + \sind(\alpha + \delta) \cdot \cotd(\beta + \phi)}$$

- α = rake angle of share (tooth or tine) to horizontal from the base of the tine
- β = soil failure angle to horizontal from the base of the tine.
- ϕ = angle of internal friction of soil; this is the angle of the slope of a soil which the soil adopts when it is at equilibrium. This is a function of the angularity of the particles, the variety of particle size grades ('grading') and the soil's degree of compaction.
- δ = soil steel interface friction
- c_u = undrained shear strength
- c_a = soil steel interface adhesion
- d = depth of tooth
- w = width of tooth

As the depth to width ratio further decreases, the upward failure noted above ceases to occur and the soil is pushed sideways below a certain depth, rather than upwards by

the passage of the share as shown in Figure 2. The depth at which this transition occurs is referred to as the critical depth.

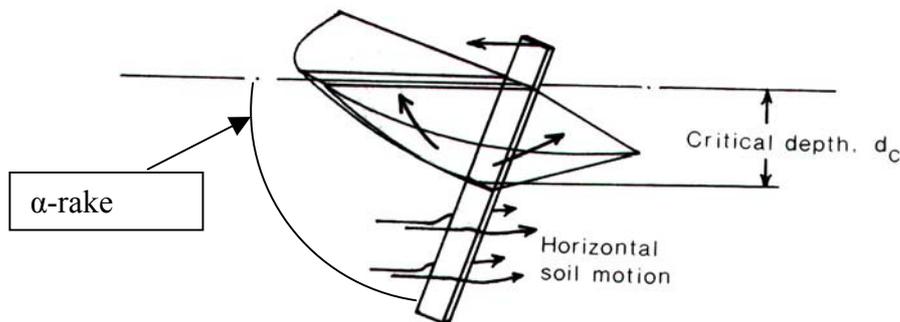


Figure 2 : Critical depth with shallow (upward) soil failure above, and deep seated (sideways) soil failure below.

The development of a critical depth is a function of both the share geometry (depth to width ratio and rake angle) and the soil characteristics. It is best found by assuming a critical depth and analysing the resistance developed above and below this depth. An iterative procedure is followed, varying the critical depth to identify the lowest total resistance for any given total depth.

In this study, it was found that the soils considered generally failed in a shallow manner and that the critical depth was below the depth of tooth penetration experienced. Therefore only shallow seated failures were considered.

Frictional forces are developed both in pulling the tooth over the seabed soils, and from the soil on the side of the tooth. Scallops tend to be concentrated in sandy or gravelly soils, in which the soil-steel interface friction angle (δ) is well documented and is typically in the range 20° to 25° . The frictional force experienced is a function of the vertical load acting on the tooth and the soil-steel interface friction angle. As both variables are fairly well defined this element of the total force experienced may be estimated with a reasonable degree of accuracy.

The frictional force on the side of a tooth is dependent on the horizontal stress between the soil and the tooth and the interface friction in the case of a granular soil. However, for the relatively short longitudinal dimension of the scallop dredge teeth and depth of penetration associated with the dredges, this force is considered to be minimal and not to have a significant effect. It may therefore be ignored in this analysis.

Using the above mathematical relationships it is possible to use the soil models to investigate the forces on a dredge tooth as it is dragged through the seabed sediments. Variables that have been considered in this analysis are:

Angle of internal friction of seabed sand (ϕ)	30° and 35°
Tooth angle to horizontal (α)	90° and 70°
Depth range	20 to 100mm

The results of this analysis are presented as Figures 3 to 6.

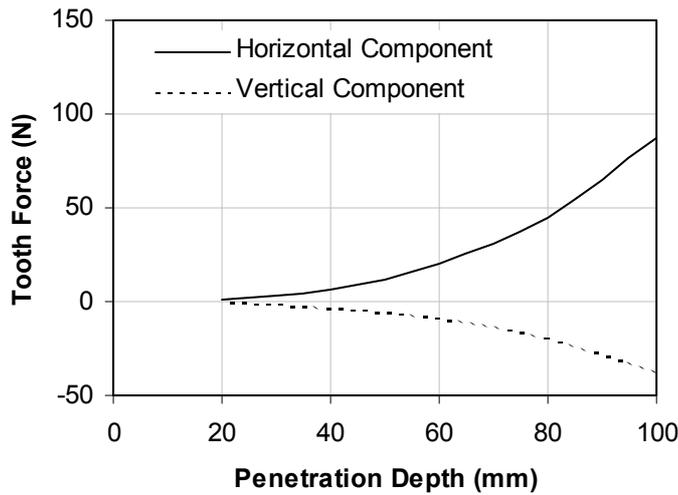


Figure 3: Theoretical tooth forces for a rake angle of 90° and $\phi = 35^\circ$

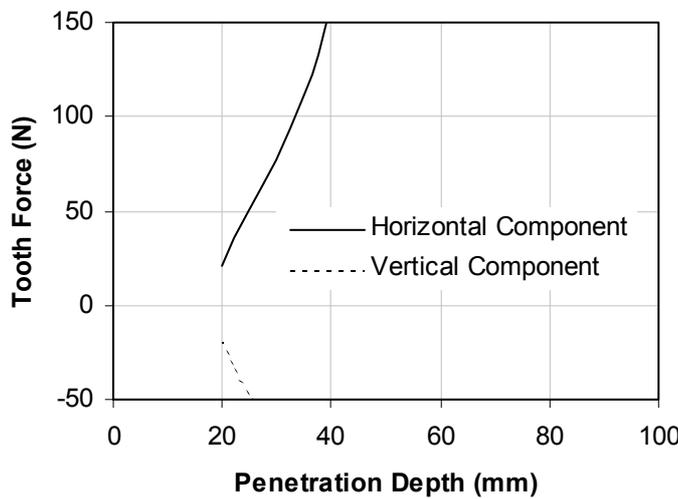


Figure 4: Theoretical tooth forces for a rake angle of 70° and $\phi = 35^\circ$

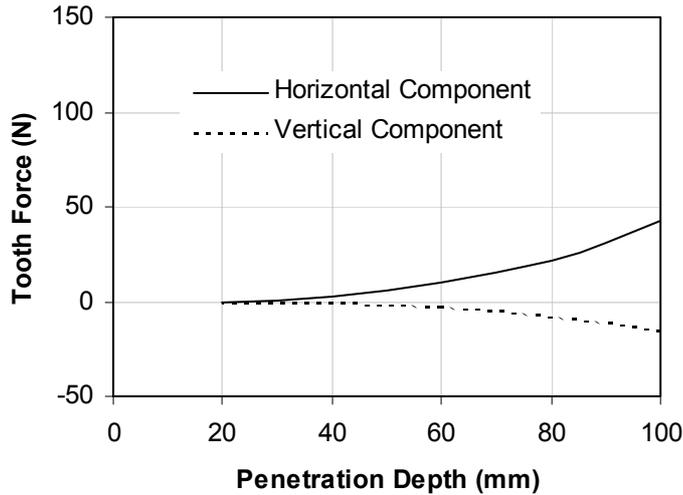


Figure 5: Theoretical tooth forces for a rake angle of 90° and $\phi = 30^\circ$

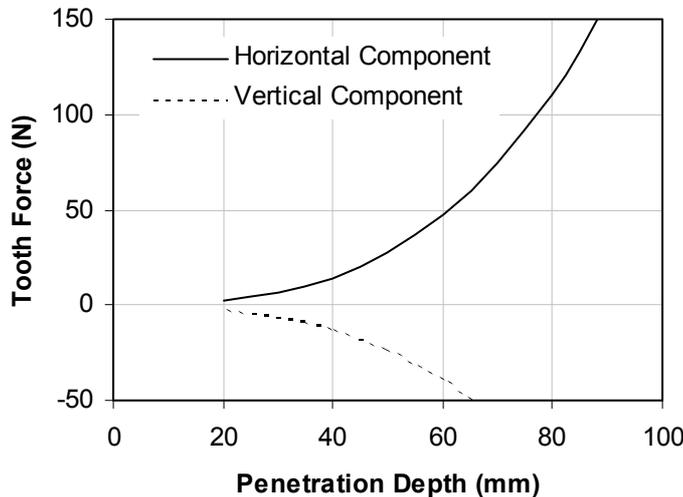


Figure 6: Theoretical tooth forces for a rake angle of 70° and $\phi = 30^\circ$

The results of these analyses show that the horizontal load increases by a factor of two between the angles of internal friction of the sediment (soil) of 30° to 35° , and there is an increase by a factor of more than three with a change of rake angle from 70° to 90° . The explanation for this effect is that a granular soil (such as a sand or gravel) derives its strength from the frictional action of one sand grain moving over another. As the frictional force at any given depth, is directly related to the self weight of overlying soil (note that water depth does not affect the frictional resistance due to the concept of effective stresses) then a change in the angle of internal friction does not have a great effect as a change in the rake angle at the shallow depths considered.

However as the rake angle reduces, the tooth tends to push the soil downwards as it penetrates. This clearly increases the stress between the soil particles and hence increases the frictional resistance seen by the tooth. Thus both horizontal and vertical components increase dramatically as the tooth angle changes.

Interaction between multiple teeth

The failure zone around a single tooth on the dredge bar extends to either side of the tooth. The lateral extent of this failure zone is a function of both the depth of penetration and the width of the tooth. This generally produces a cone of soil disturbance. Based on classical soil mechanics, the lateral extent of disturbance can be estimated as shown in Figure 7. For typical angles of internal friction of seabed sands, the angle of soil disturbance to each side of a tooth will be 1 Horizontal: 2 Vertical. Note is made that the shape of the zone of disturbed soil changes at large depth to width ratios, however as noted above, it has not been considered necessary to discuss this in details as all soil failures are considered to be shallow seated.

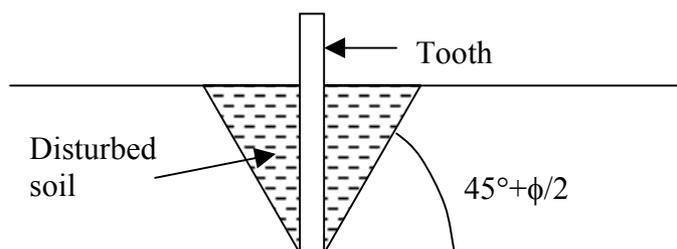


Figure 7: Extent of disturbed soil around a tooth penetrating the surface.

For widely spaced teeth, there is no interaction and the volume of soil disturbed is essentially equivalent to two isolated teeth. As the spacing closes to less than the approximate depth of penetration, some degree of interaction will occur. Given the physical dimensions of a scallop dredge tooth bar, with teeth of 100mm length (before wear) and a spacing of 89mm, the possibility of interaction is relatively low.

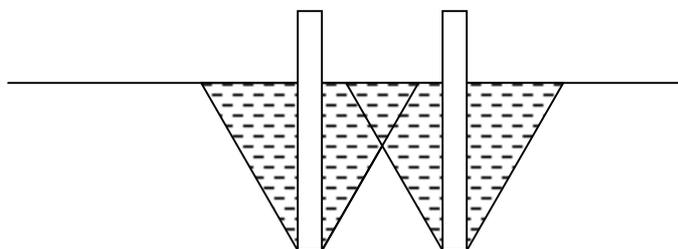


Figure 8 : Interaction between two adjacent teeth.

As the spacing further closes, the two teeth will interact so closely that they can be considered as a single tooth, equivalent to the overall width of the blades. For the relatively widely spaced scallop dredge teeth this is clearly not the case and need not be considered.

Methodologies

Gear

The dredges used in these studies were the conventional Metallco dredges using French teeth (Figure 9) with approximately 12mm cut off the end of the teeth. It was found best to place the instrumented dredge as the central dredge on the bar with two dredges on the inboard and outboard positions with sufficient space between the dredges to avoid collisions.

Design and development of novel instrumentation-Instrumented scallop

The instrumented scallop was devised to describe the accelerations and rotations experienced by a scallop (*Pecten maximus*) during the process of being dredged. The scallop analogue, a high impact resin cast contains three orthogonal accelerometers mounted in the x, y and z axes of the scallop, (see Figures 10 and 11) electronic circuitry to perform a data-logging function which can be controlled from a PC. The resin-encapsulated system was modelled on the size and shape of a 110mm scallop of similar relative density and centre of gravity. The adaptation of the current accelerometer technology was particularly useful in identifying the following parameters associated with the instrumented scallop:

- i The orbit of rotation of a scallop in all three planes x, y and z, in order to assess the 'rotating effects' in the dredge chain mail bag;
- ii Acceleration inside the bag due to high frequency impact from stones (collision) and seabed disturbance during traverse across the seabed.

The accelerometer data was processed in the following ways in order to calculate the acceleration and rotation.

Rotation

1. A 10 reading (equivalent to 1 second at 10Hz) running mean was calculated for acceleration in the x, y and z planes.
2. From these results the mean angles of incidence and declination were calculated thus:

$$\text{Mean Heel angle} = \text{Arctangent} (X\text{acceleration}/Z \text{ acceleration}) \times 180/\pi$$

$$\text{Mean Incidence angle} = \text{Arctangent} (Y\text{acceleration}/Z \text{ acceleration}) \times 180/\pi$$

These parameters can be calculated since since the Z axis acceleration is equal to gravity at rest (Figure 11). (The designers assumed that the natural orientation of the scallop was flat side down which is of course incorrect but irrelevant in this case). The arctangent converts ratios between the x and z and y and z accelerations to degrees away from rest in these two directions.

Resultant Modulus of Acceleration

This was calculated from the running mean accelerometer data described above based on pythagorus' theorem (note squaring the acceleration also removes the sign from the acceleration):

$$\text{Resultant Modulus} = (\sqrt{X^2 + Y^2 + Z^2}) - 1$$

At rest this parameter would have a value of 1 due to gravity. Hence this formula calculates the acceleration in a given direction. Subtracting 1 allows the description of acceleration net of gravity.

Assessment of motion and acceleration of the dredge frame.

The instrumented scallop was fixed to the dredge frame on the central spine of the frame where it is welded to the crosspiece (Figure 12). The data acquired during towing on a variety of seabed types, eg soft sand, hard stony, was used to assess the amount of acceleration and rotation of the dredge on the seabed. This influenced the design of the logging system on the final instrumentation package and provided comparative data on the amount of acceleration in these locations. Comparisons were also made between instrumented scallops placed loose in the dredge bag and those clamped to the framework simultaneously.

Parameters measured on the scallop dredge

The design and calibration of the scallop dredge instrumentation is shown in Figures 13-16. The following parameters were recorded into the self-contained data-logger, recessed for protection into the dredge tow frame:

- i *Total dredge tension* – transducer:
Shear pin load shackle rated @ 5000N (overload rating of 50%)
- ii *Tooth angle* – (see * iv)
Measured between the dredge frame (as a reference) and the tooth bar.
Transducers: two compressive loadcells, each mounted in series with the coil springs controlling the deflection of the tooth bar, and calibrated for signal output to angular deflection of the tooth bar. This was later modified to two shear pin loadcells located in the as the bolt retaining the spring in place.
- iii *Tooth depth* – into the seabed.
Transducers – two digital rotational decoders attached to a trailing arm, which mechanically follows the seabed contour adjacent to and in line with the outboard sides of the tooth bar. The trailing arm mechanically follows the contour of the seabed, and any changes are translated via this mechanism into an angular displacement of the rotary encoder. Initial calibration is carried out on a flat rigid surface by elevating the foot attached to the base of the arm in 20mm vertical steps. The corresponding angular displacements are recorded in the data logger's memory and used in a calibration over the full range of tooth depth. It should be noted that the pivot point of the encoder is sufficiently far away from the towing point of the dredge (that is the load shackle) such that any changes in elevation of the dredge bar produce very small changes on the movement of the trailing arm.
- iv The angles of incidence and heel of the dredge frame plane. Using 3 accelerometers mounted into the data logger housing and referenced to this plane (see above for calculation of angles).

In order to determine the rake angle (α) of the dredge teeth to an assumed horizontal seabed the angle of incidence of the frame (in the housing direction) needed to be computed together with the tooth angle to the framework. The shear pin load cell

shown detects the load introduced in the spring via the axial bolt, which passes through the coils. As the tooth bar swings backwards an output from the shear pin strain gauge bridge proportional to spring tension is recorded by the data logger. This is calibrated against angular deflection by deflecting the tooth bar backwards at known intervals ie in 10 degree stages up to 40 degrees. The subsequent data is stored in the Eprom. In order to deduce the angle at which the teeth enter the seabed the dredge frame angle to an assumed horizontal seabed must be recorded. This angle is computed from the output of a triaxial accelerometer recessed into the frame in the same housing as the data logger as described above.

Data logger

Recessed into the dredge framework, to improve its protection from impacts, the data-logging system comprises of:

- a pressurised stainless steel housing tested to 20 bar (200metres) containing the necessary circuitry to sample the above parameters at two selectable sampling speeds of 10 and 20 Hz
- 3 orthogonal accelerometers, referenced to the dredge frame;
- 5 Mb of data storage (RAM);
- a μ processor with embedded software
- a real time clock and RS232 serial port for downloading data to a laptop PC with the menu facility for start/stop logging and its respective duration to maximum of 40 minutes.

Protection

The instrumentation was protected from impacts by metal bars designed to be clamped to the dredge forward of the digital rotational decoders See Figure 16. This would of course add marginally to the weight and drag of the gear. The priority was to protect the instruments.

Field Testing

The system was field tested in August 2000. Observations were made using an underwater video camera of the trailing arm in action (Figure 16). It was observed to follow the contours satisfactorily and to avoid being affected by stones and sediment being ploughed up in front of the dredge. After this cruise the aluminium shoe, which was at the base of the trailing arm, was observed to snag on seaweed and other matter. This was modified to avoid this problem. Also the compressive load cells designed to monitor spring tension were replaced with shear pins (see above) after this trial. This was because it was found difficult to seat the load cells satisfactorily between the spring and the tooth arm. Some data were obtained but there was insufficient consistency to obtain a satisfactory result. This was because of the mechanical problems with the load cells and it was found that weather conditions (wind force 4) meant that it was very difficult to dredge consistently downwind.

Field work

A cruise was undertaken on MFV De Bounty on fishing grounds off the South and west of the Isle of Man in calm weather. A total of 23 hauls of 10minutes duration

were made on 3 different scalloping grounds; Bradda inshore, Bradda offshore and the Chickens. The spring tension was set at 8 kgfm using the adapted torque wrench described in [Section 3.2.2.3](#) and the speed over the ground of 2.4 knots (1.2m/sec) were kept constant. Care was taken to ensure a consistent Warp/depth ratio and the areas dredged were close to being horizontal sea bed. For some hauls the dredge was pre packed with stones to emulate a full dredge.

Tows were made in alternating directions; as far as possible emulating commercial practice in a restricted area on each ground. Sediment samples were using a grab at three stations on Bradda inshore and offshore and also two at the Chickens. The environment on the Chickens ground proved to harsh for the sensors; both the tension and acceleration sensors ceased to function although some results were obtained for the tooth angle and tooth depth it was decided to terminate the trial.

Grab samples

Grab samples of sediment were collected at 3 locations on Bradda inshore and offshore and at 2 locations on the Chickens. These locations represented the beginning middle and end of the tows. The rocky nature of the seabed at the Chickens made collecting sediment samples from this location difficult. These locations were in distributed evenly along the area dredged and are shown in Table 1.

Table 1 Grab sample locations

Grab Sample	Position	
	Lat.	Long.
Offshore Bradda 1	54N005.288	004W48.419
Offshore Bradda 2	54N004.804	004W48.353
Offshore Bradda 3	54N004.269	004W48.585
Inshore Bradda 1	54N005.22	004W53.3
Inshore Bradda 2	54N005.429	004W53.715
Inshore Bradda 3	54N005.45	004W54.486
Chickens 1	53N056.915	004W47.28
Chickens (No sample)	53N56.809	004W47.521
Chickens 2	53N56.812	004W48.261

Video observations

RV Roagan (Research vessel for PEML¹) has collected 3 hours of video (taken over a number of years) taken using a camera mounted on the dredge bar. In this video clear observations of the action of the dredge teeth on the seabed and observations of scallops being captured could be made. [Clips are shown on the accompanying file.](#)

¹ Port Erin Marine Laboratory Univ. of Liverpool

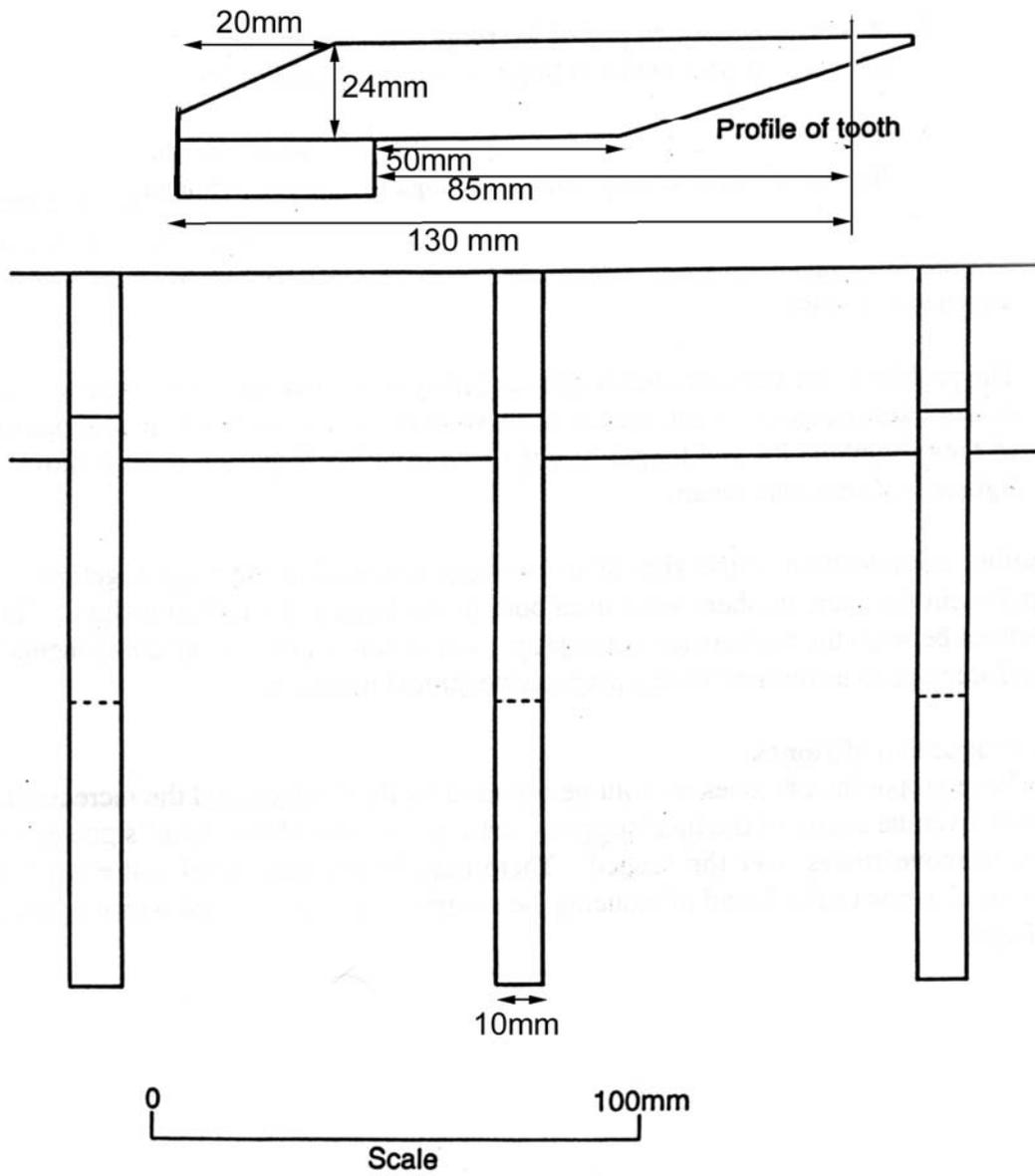


Figure 9: Profile of French teeth and arrangement on bar

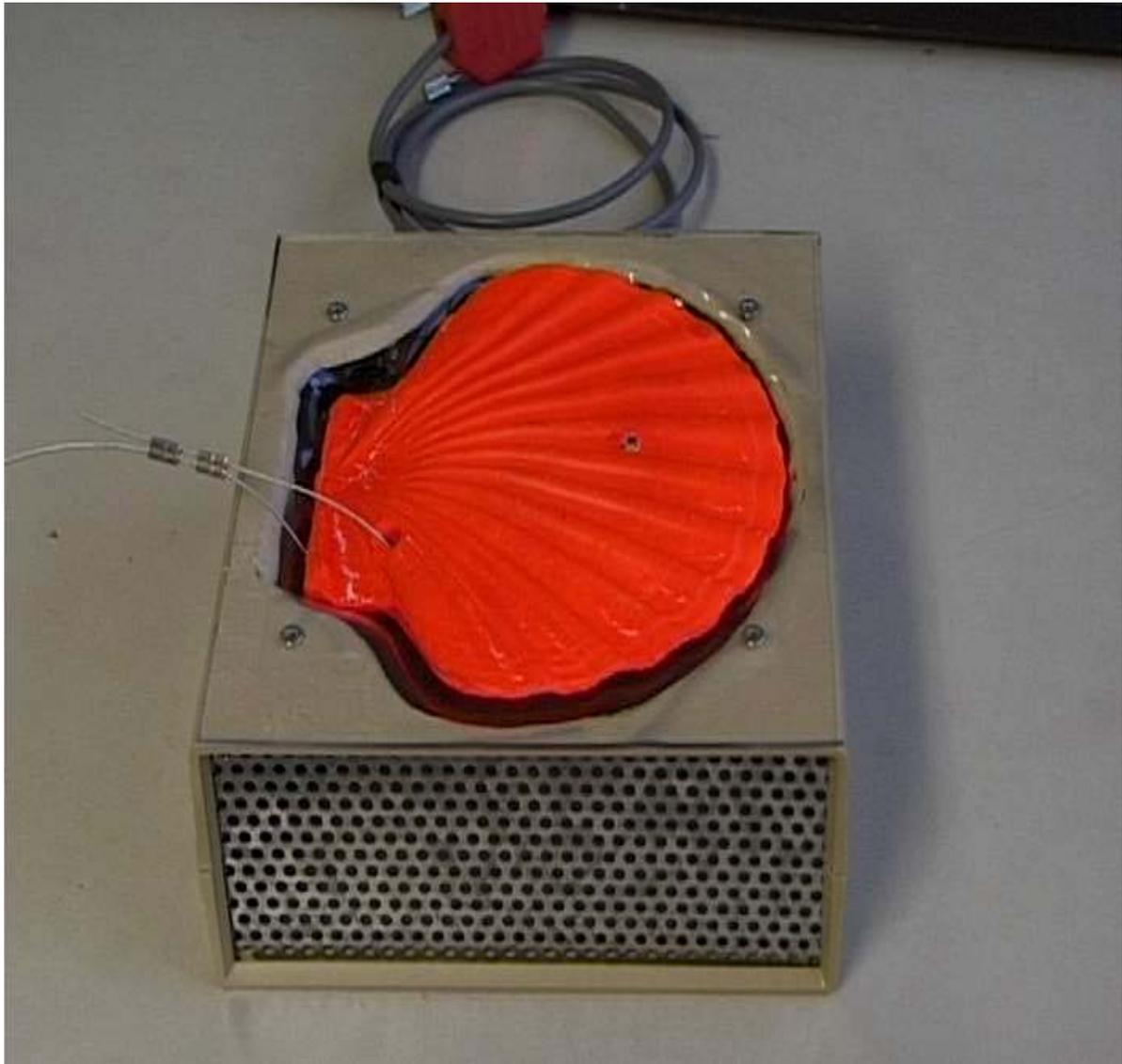


Figure 10 Instrumented scallop in cradle for downloading data and charging batteries

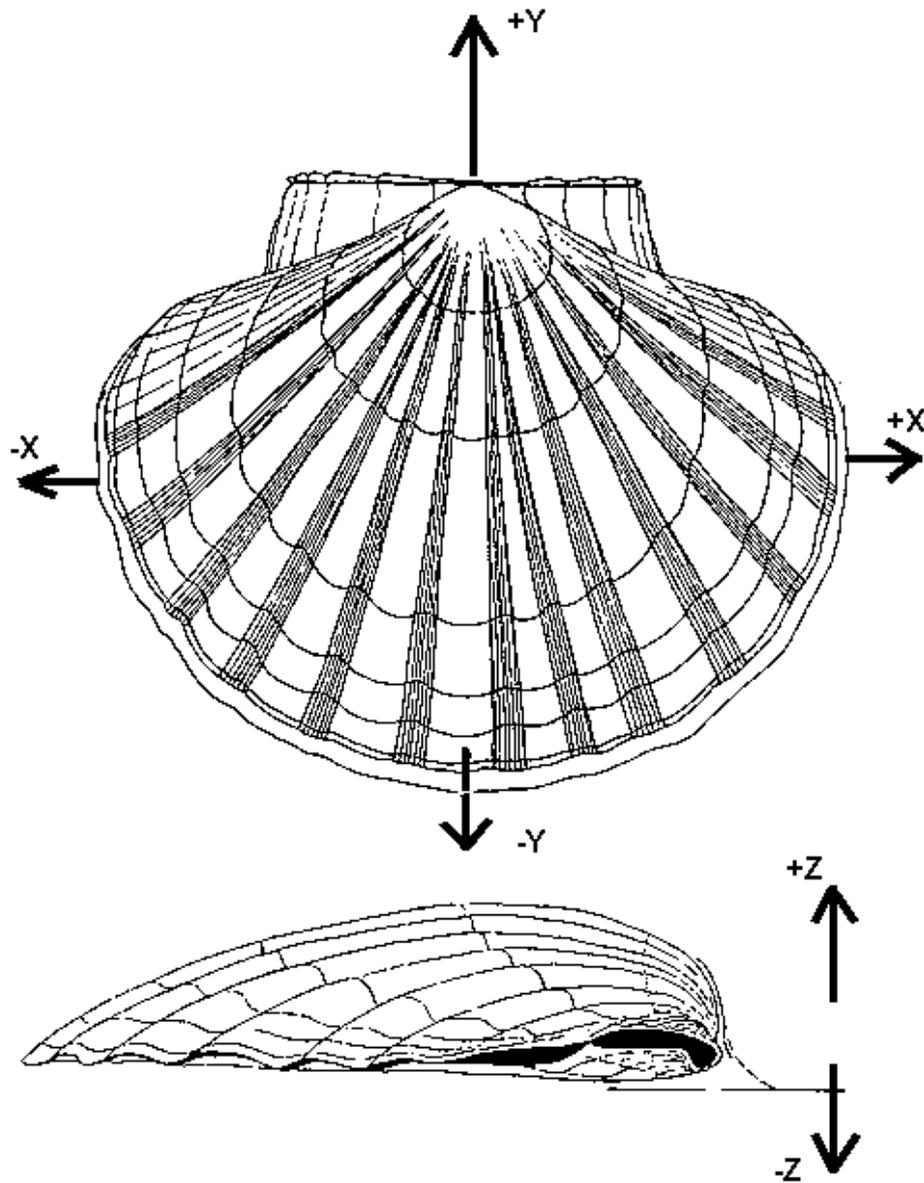


Figure 11 x, y, and z accelerations on instrumented scallop

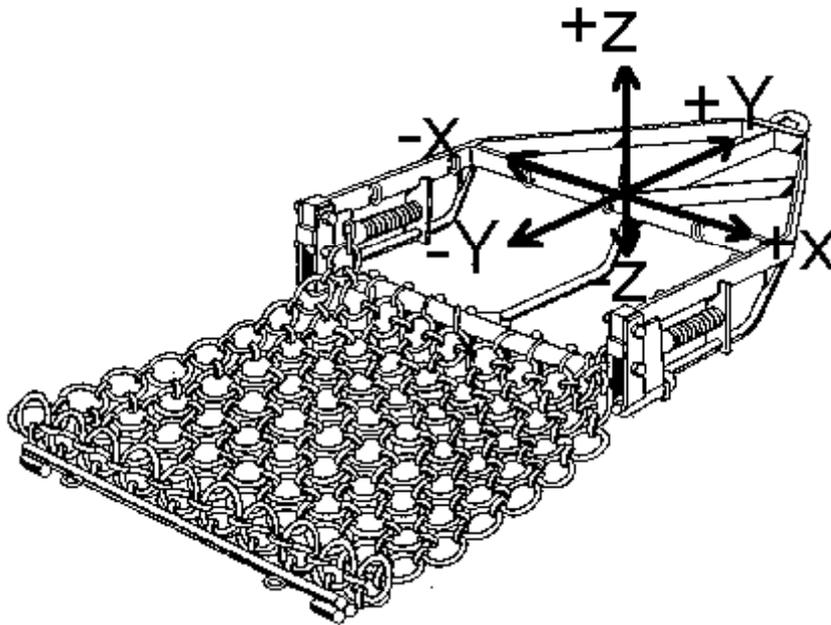


Figure 12: Deployment position on dredge of instrumented scallop on dredge frame

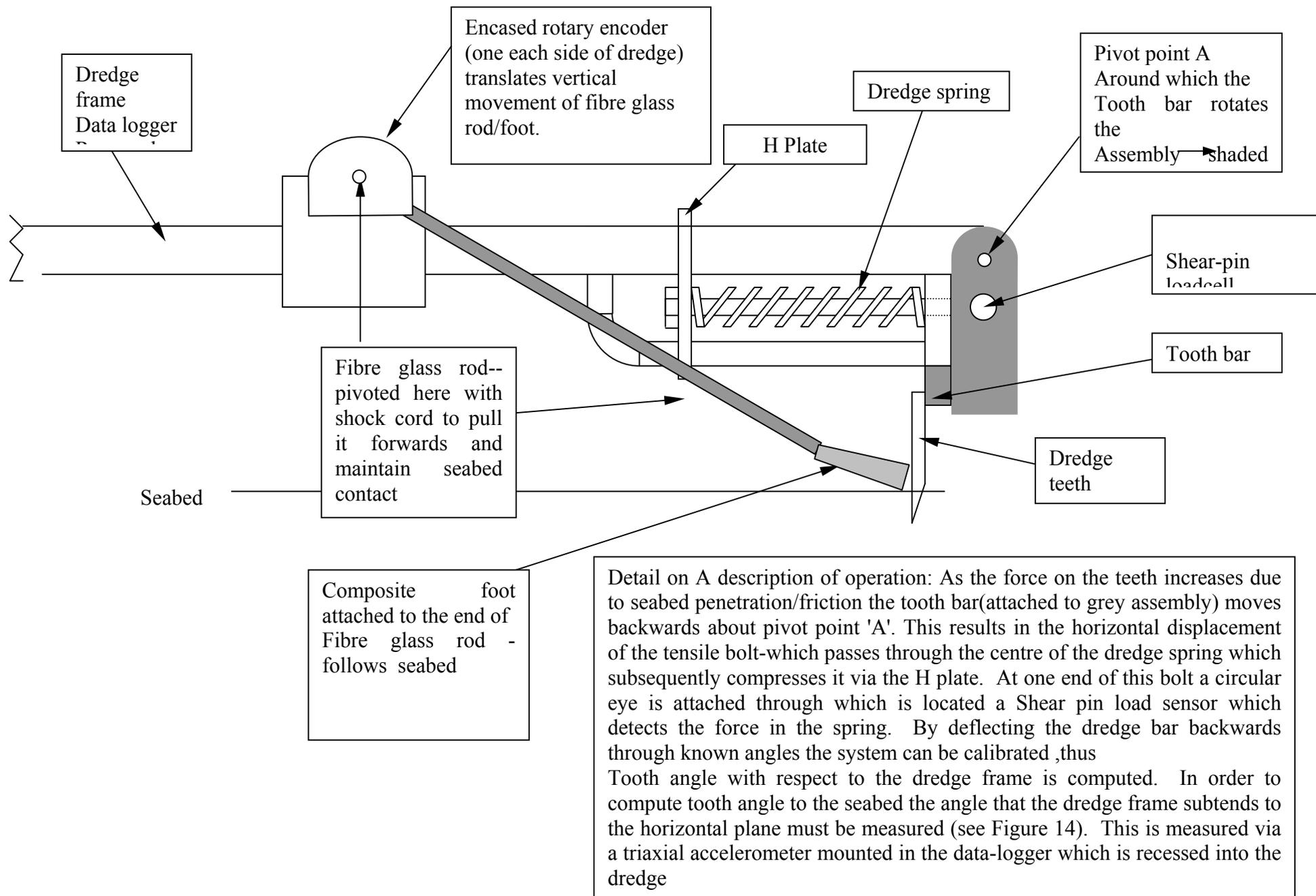


Figure 13 Side view of instrumented dredge

Tooth Angle computation (α) for dredge with 'nose up' elevation.

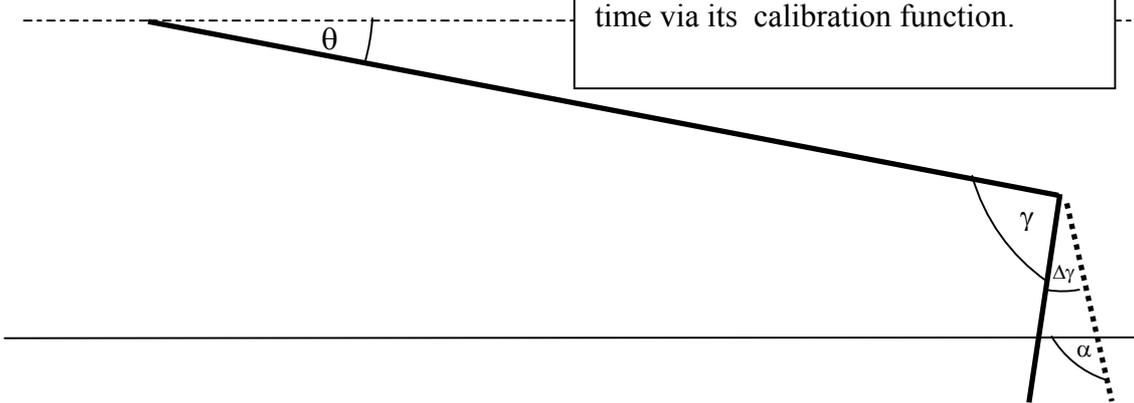
θ – angle of dredge frame to the horizontal plane

N.B. $\Delta\gamma$ is the angular change from a quiescent rest position to that when a frictional component force(p) is applied at the teeth due to penetration into the Seabed. It is proportional to the spring tension at any instance in time via its calibration function.

γ – angle of tooth bar to the dredge frame(at rest) before frictional component due to seabed penetration takes place(set angle)

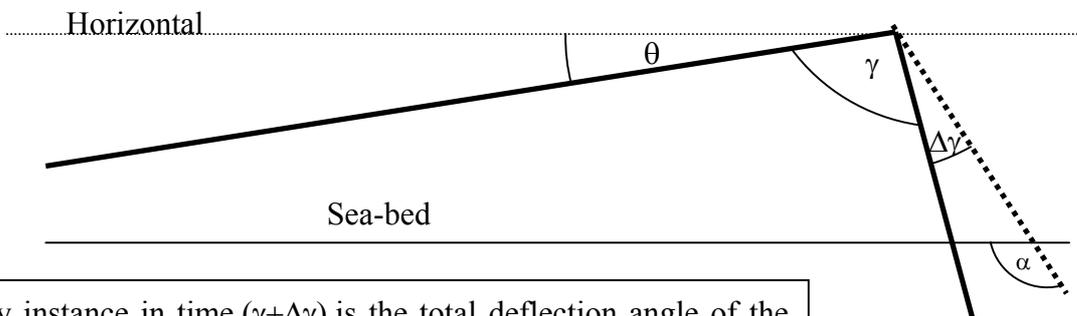
$\Delta\gamma$ – change in the tooth bar angle with respect to the dredge frame due to the forces applied to the teeth when immersed into the seabed substrate

α – angle of attack of tooth to (an assumed horizontal seabed)



At any instance in time. $(\gamma + \Delta\gamma)$ is the total deflection angle of the tooth bar with respect to the dredge frame.
 ...therefore $\alpha = ((\gamma + \Delta\gamma) - \theta)$

Tooth Angle computation (α) for dredge with 'nose down' elevation.



At any instance in time. $(\gamma + \Delta\gamma)$ is the total deflection angle of the tooth bar with respect to the dredge frame.
 ...therefore $\alpha = ((\gamma + \Delta\gamma) + \theta)$

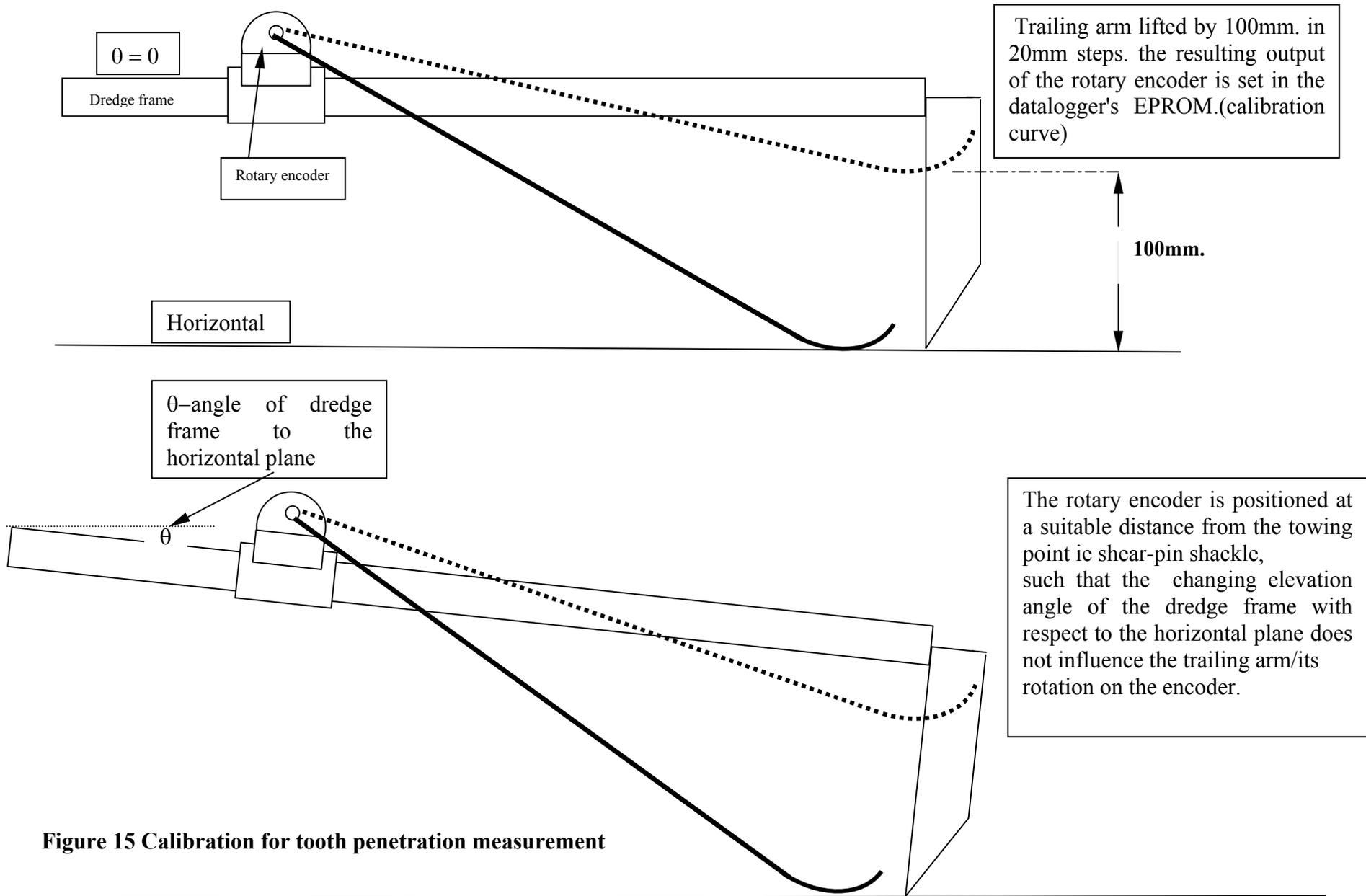


Figure 15 Calibration for tooth penetration measurement



Figure 16 Underwater photograph of instrumented dredge taken from video to check action of trailing arm. Note the protective bar across in front of the instrumentation. The skid at the base of the trailing arm in this picture was not the final design used; modifications were made to avoid the collection of weed on the skid.

Results

Instrumentation

Figure 17 and 18 show example plots of rotation and acceleration from the instrumented scallop. The results for the instrumented scallop show some variability. There were important differences between the scallop placed in the bag and those clamped on to the frame with very much more acceleration occurring on the dredge frame. The flexible link between the dredge frame and bag appears to act to reduce the acceleration on scallops in the dredge bag.

Figures 19-21 show results from the instrumentation. There were rapid variations in all the parameters. Thus in order to summarise the information by 10 minute tow the results were averaged in 10 units of 1 minutes and then these averaged to produce the outputs shown in Table 2. This shows by tow information and the mean results by tow and then by ground. For the dredge results the mean tooth rake angle shows consistency through all tows with an angle to the seabed (α) of 125° (as indicated in Table 2); this contrasts with the an angle to the seabed (α) of 90° and 70° described in the model (equation 1 and Figures 1-6). The mean tooth penetration depth of around 25-35mm is what would be expected from the observations of wear on teeth. The reduction of tooth penetration depth for those hauls where the dredges were packed with cobbles is consistent with the centre of gravity of the dredges being moved back with more weight being on the bag than the teeth. For the Chickens ground where the catch of cobbles was similar to the quantity packed into the dredge much higher apparent tooth penetration depths were observed. However this effect could be caused by many cobbles being swept around the sides of the teeth and thus pushing the trailing arms upwards.

Particle size distributions.

Particle size distributions are shown in Figure 22. These results show a more sandy component in two of the stations on the inshore Bradda location with the rest of the stations found to be gravely sand. This is at variance with the BGS² survey data for the area, which shows higher gravel content and more uniformity between the three grounds. It is not clear why these samples should be so different from the BGS results. However sample volume may have influenced the result, reducing the reported percentage gravel by weight. This may also apply to the Bradda samples. Some difficulty was experienced actually making the grab function at the chickens stations due to the large amounts of cobbles on the seabed.

Video observations

The video observations show three main aspects of dredge scallop interaction.

1. Interaction between scallops and dredges. The scallops made initial contact with the tip of the teeth as they raked through the seabed. The vertical force on the teeth rotates them out of the seabed (see Figure 23) rotation being in both planes until the animals were then seen on the face of the teeth. the flat side of the scallop faced the teeth, and the hinge downwards, being held there by hydrodynamic forces. They could remain here for a period of time; up to 30

² British Geological Survey

seconds has been observed although 1-3 seconds was more common. This was followed by the scallops being swept into the dredge, or round it. There is usually a clearly defined event that occurred; collision with a stone or other matter or movement of the teeth, which was a pre cursor to the scallop being dislodged from the teeth.

2. The dredges were observed to bounce. This effect, in which the dredges were seen to lift clear of the seabed with a ‘hopping’ motion, occurred intermittently during the tows. The effect appeared to be related to the springs resonating, the springs could be seen to be working backwards and forwards as the teeth lifted in and out of the sediment. The frequency of the motion, as judged by frame by frame analysis of the video 2-6Hz. In some cases there was a clearly observed precursor such as a change in speed or sediment composition there was no clear initiating event.

As well as scallops the dredges were seen to encounter rocks on the seabed. In some of these encounters the dredges were observed to bounce over them on the seabed; others were treated in a similar way as the scallops; a period of time being pushed along in front of the dredge, followed by passage into, under or round the dredge. The video does not contain observations of encounters with solid reefs although other sources (Lart et al. 1993) report the dredge impacting on solid substrate. Here the spring-loaded tooth bar can clearly be seen to respond to the hard substrate allowing the dredge to lift over obstacles.

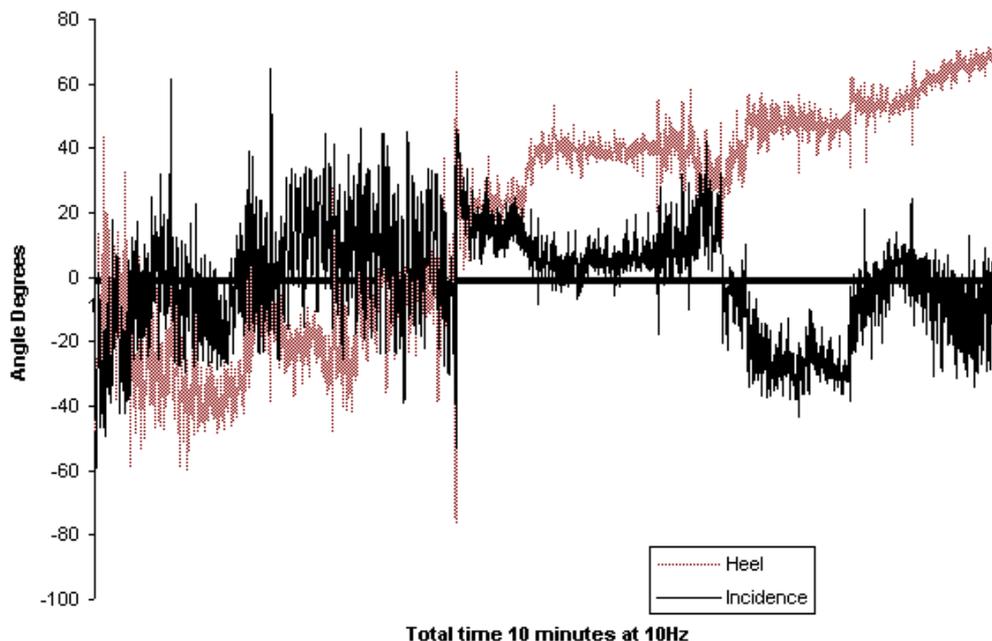


Figure 17 Rotation of the instrumented scallop during the course of 10minutes in the dredge bag.

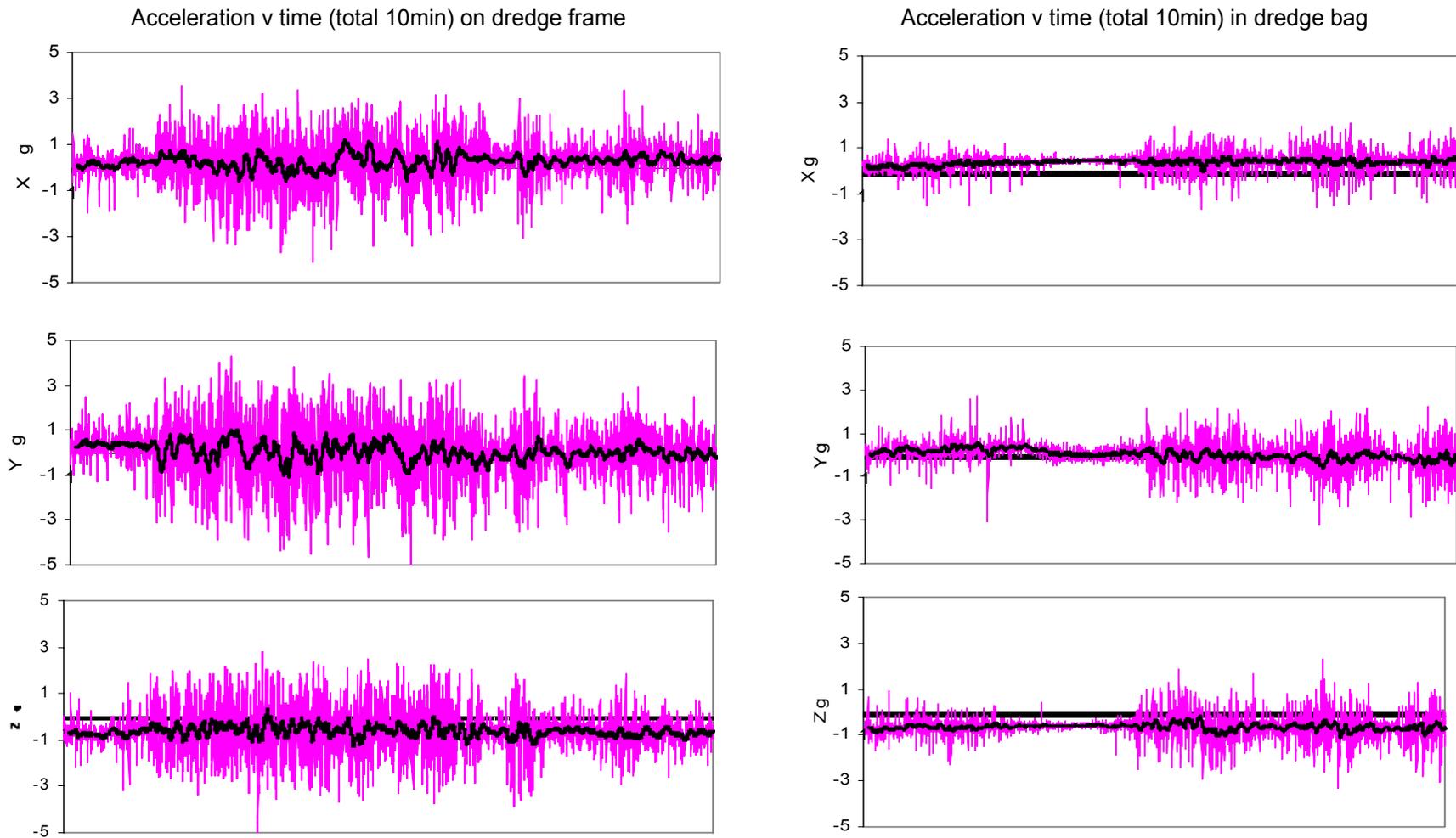


Figure 18 Synchronous results; 10 minutes at 10Hz from the instrumented scallop for acceleration on the dredge frame and loose in the dredge bag

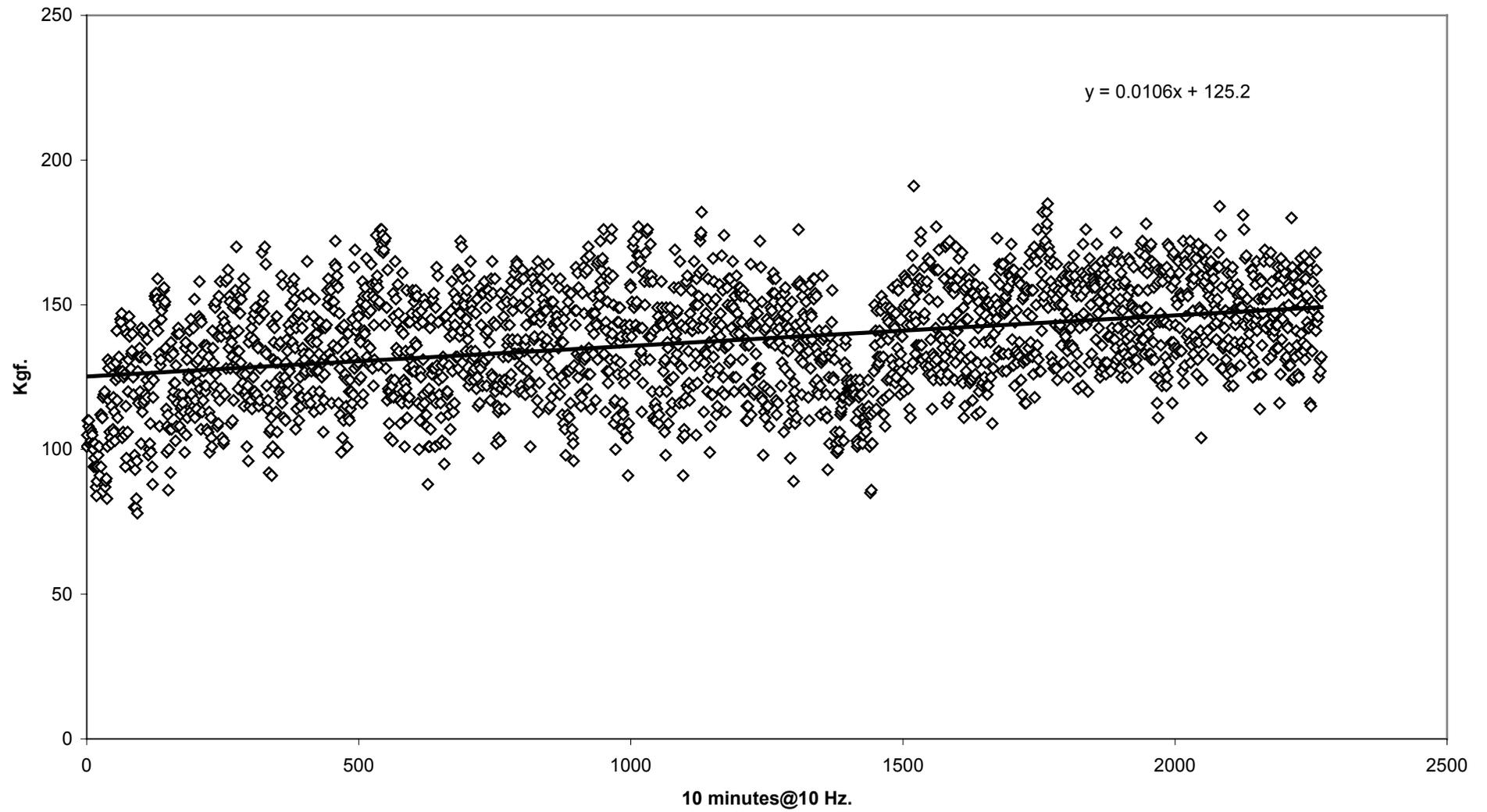


Figure 19 Dredge Tension vis time

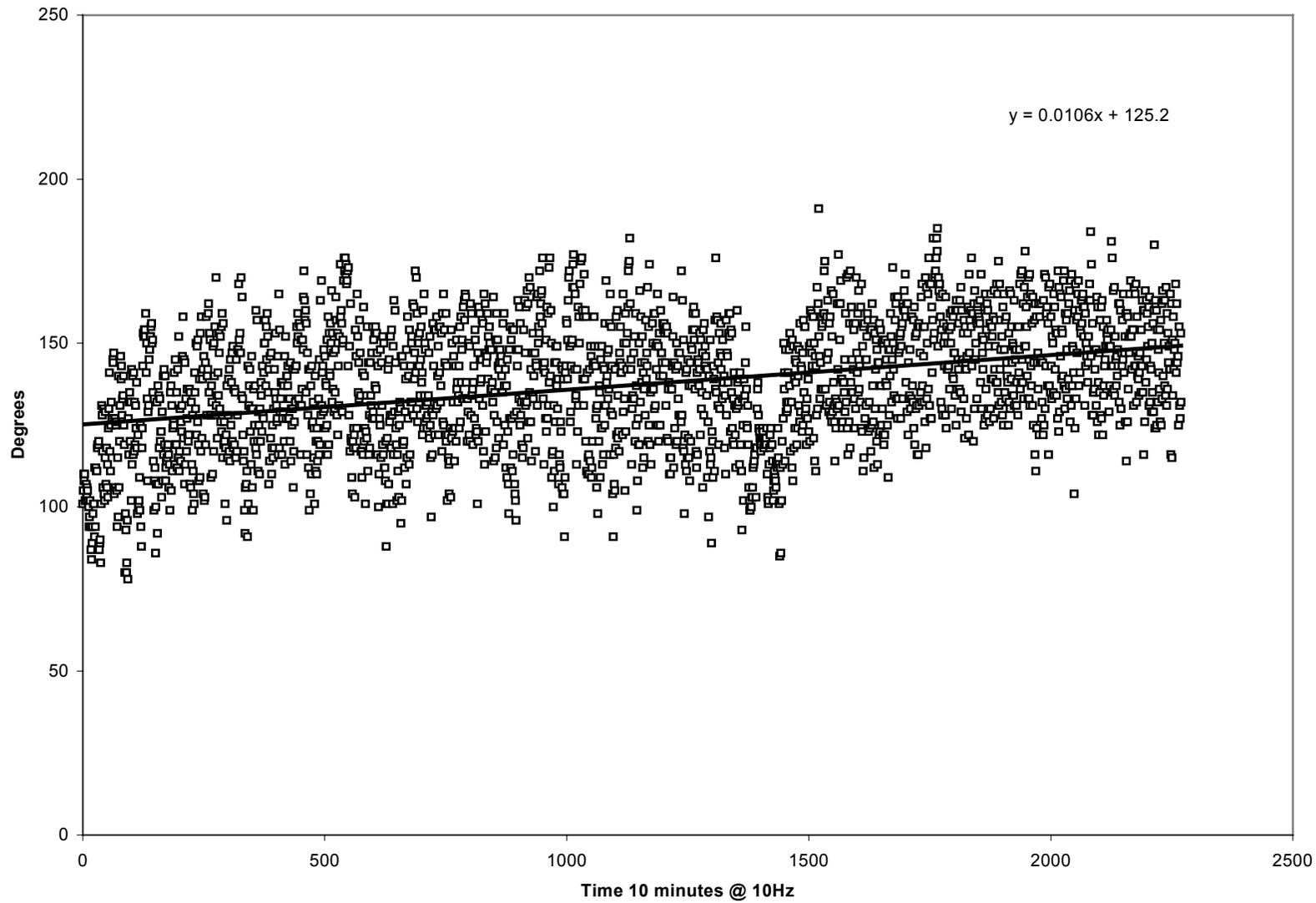
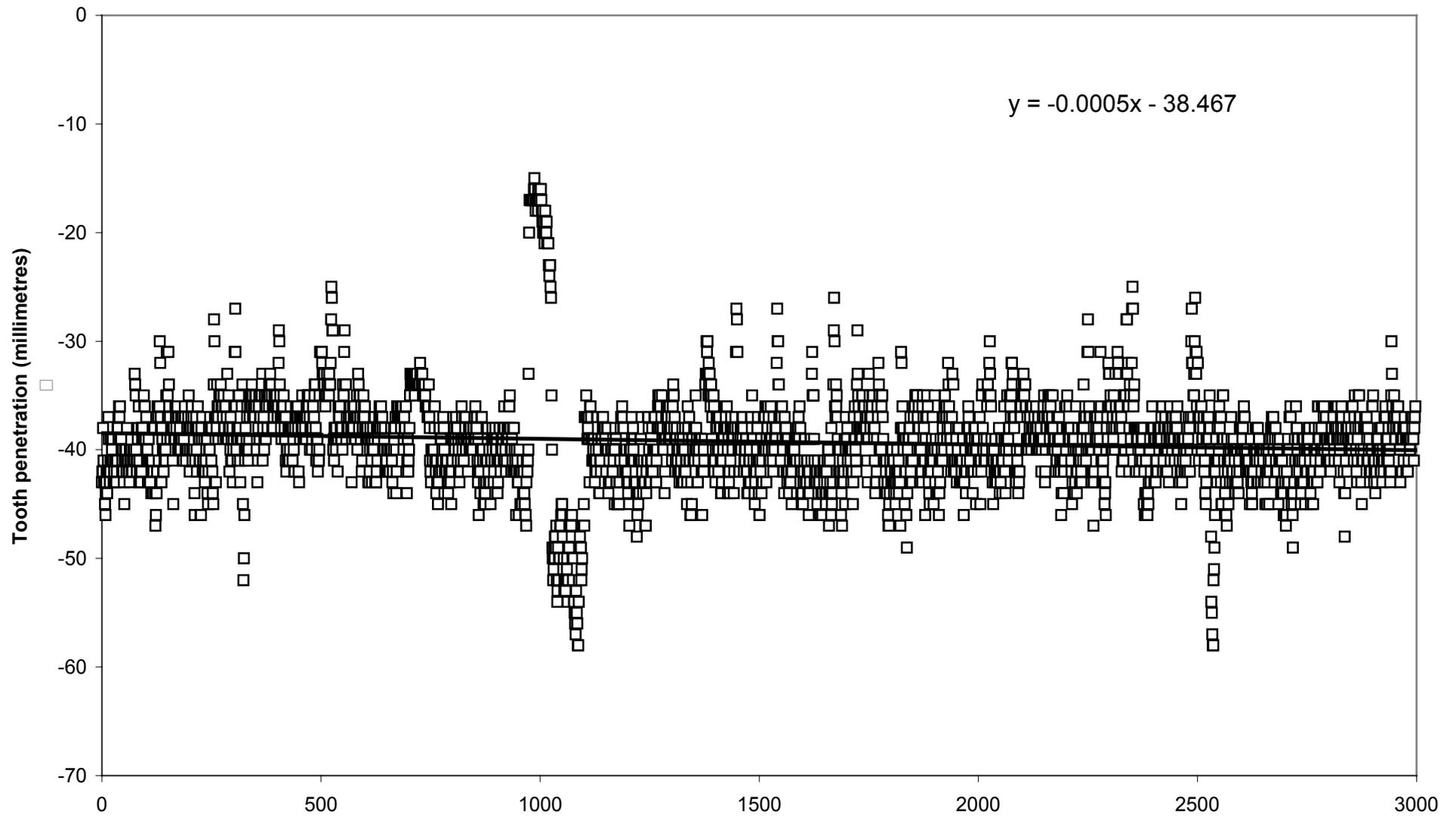


Figure 20 Tooth Angle (α) to the seabed vis time

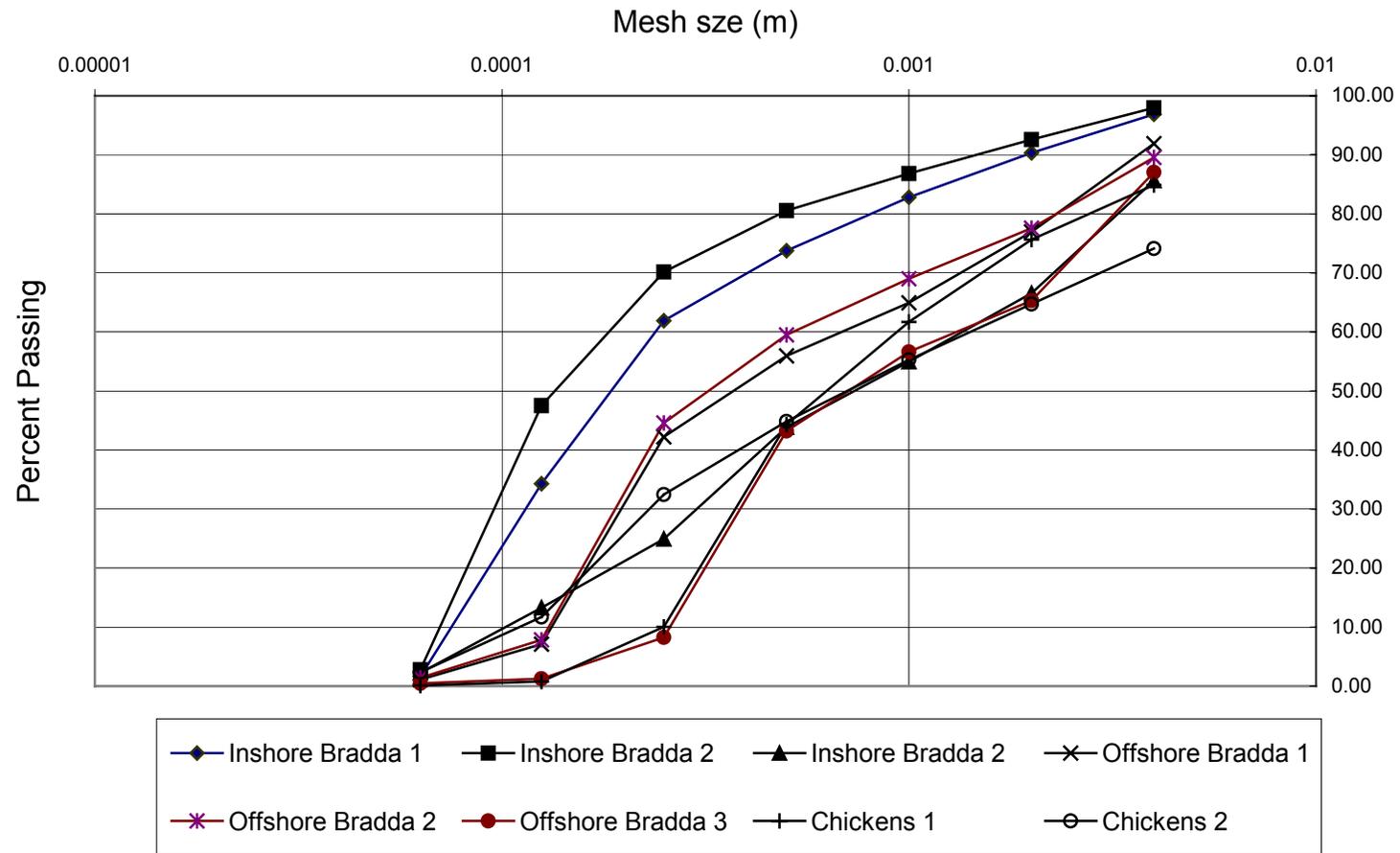


10 minutes @ 10 Hz.

Figure 21 Tooth penetration into the substrate vis time

Table 2 Mean results for measured parameters on 10 minute tows on 3 scallop grounds, mean results by ground also shown

Location	Depth (m)	Warp out (m)	Warp: Depth Ratio	Tension Kgf	Acceleration	Tooth rake angle α Deg	Tooth Depth Of penetration (mm)	Total Catch Kg	Scallops Kg
Bradda Inshore	36	100	2.8	158	0.98	123	7	14	3
	36	100	2.8	148	1.14	124	7	9	1
	35	100	2.9	178	1.13	125	13	19	2
	35	100	2.8	169	1.11	122	3	33	4
	36	100	2.8	117	1.27	130	53	14	1.5
	36	100	2.8	123	1.26	126	40	17	3.5
	36	100	2.8	112	1.18	121	25	12	2
	37	100	2.7	144	1.15	122	39	5	1
	36	100	2.8	155	1.21	123	21	5	1
	36	100	2.8	175	1.17	127	16	28	2
	36	100	2.8	166	1.28	124	31	22	2
	36	100	2.8	160	1.19	131	26	10	2
	35	100	2.8	*	1.02	*	*	12	3
	35	100	2.8	177	1.19	*	37	29	4
Mean Bradda Inshore			2.8	152	1.16	125	25	16	2.3
Dredges packed with 80Kg rocks: Bradda Inshore	34	100	2.9	155	*		7	*	*
	35	100	2.8	165	1.22		8	98	*
Bradda Offshore	44	126	2.9	115	1.14	126	43	15	5
	45	126	2.8	116	1.15	126	51	13	3
	45	126	2.8	112	1.22	125	23	15	3
	47	126	2.7	110	1.22	127	23	9	3
Mean Bradda Offshore			2.8	113.25	1.18	126	35	13	3.5
Chickens	62	170	2.7	*	*	133	68	63	1
	61	170	2.8	123	*	126	41	85	0.5
	62	170	2.7	*	*	133	68	85	3
Mean Chickens			2.8	123		131	59	78	1.5



Note: Sampling of sediments on the Chickens ground was difficult because of the presence of cobbles in the grab, only two grab samples were successfully made.

Figure 22 Particle size distributions from samples taken from scallop grounds in the locations of the instrumented trials

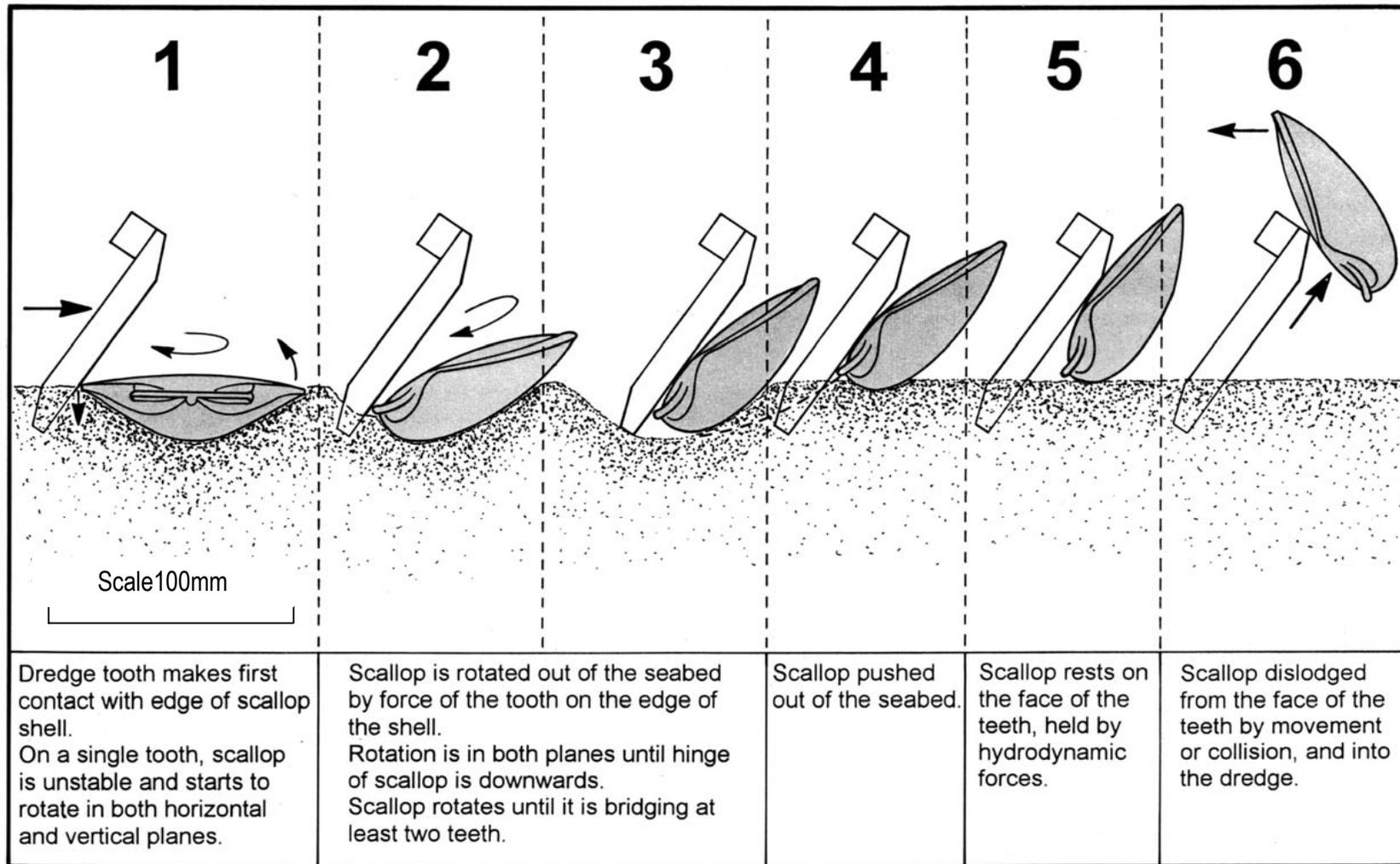


Figure 23 Hypothesised method of scallop capture by spring toothed dredge based upon video observations. Tooth angle set at 55° the mean angle and depth of penetration measured on the Bradda inshore ground.

Discussion

Model

The results of the tooth angle measurements indicate that the tooth adopts an angle of 90° or greater in the sediments. The model discussed above is designed to study the interaction of teeth (tines) at 90° or less to the sediments. Thus they cannot be used to directly describe the behaviour of the teeth in the sediments. However the model does indicate certain characteristics of tooth-sediment which may affect dredge behaviour.

The model indicates that were the teeth to be at 70° for example, the rate of change of both horizontal and vertical forces with increasing tooth penetration would be very much higher than at 90°. In the variable environment encountered by the dredges this could be expected to introduce instability in the dredges' movement. This might set the dredges into the periods of resonate bouncing described above. Thus the angle of the teeth at 90° or greater the teeth as described in this example may contribute to the stability of the dredges. The sedimentary environment is also variable which would result in changes in the sediment properties, for example the Φ value, which as indicated in the model, would result in variations in the horizontal and vertical forces on the tooth which would also induce instability.

Capture mechanism

The hypotheses for scallop a capture mechanism described above depends on rotating the scallops out of the seabed by the application of downward pressure on the edge of the scallop. There is no apparent requirement for the part of the teeth which penetrates the sediments to be at less than 90° to the sediments. As discussed above this would be likely to induce instability in the motion of the dredge. There could be advantages in raking the part of the tooth that is above the surface of the sediment at less than 90° since this would improve the passage of the scallop into the dredge, because the scallops were sometimes observed to remain on the front face of the teeth. 'French' teeth as used in this study are shaped with a bevelled edge at the top of the teeth (see Figure 9) and this probably helps this effect, but 'peg' teeth do not. It might be possible to extend this bevel further down the tooth. However, care would have to be taken to avoid a sudden change in the shape of the tooth, which would be expected to create instability as it penetrates the substrate.

This dredge is designed to operate over a variety of seabed types. The springs are a mechanism by which the dredges can be adapted to different seabed types; fishermen explore the levels of spring tension required by setting the springs on each dredge to different tensions and observing to see which dredge catches the most scallops. Many areas of scallop grounds exhibit considerable spatial variability in terms of substrate types; thus the springs constitute a mechanism by which the dredge can be adjusted to suit the conditions this enables a wider range for the fishery.

Many damaged scallops in the catches with shell damage on the flat valve of the shell, which is clearly the result of dredge teeth impacting from above. The results shown in [Section 3.1.1.2](#) suggest that damage levels in *Pecten maximus* are consistently different between grounds thus suggesting that the levels of equilibrium also vary between grounds. It may be that the most spatially variable grounds cause the most

variation in equilibrium and hence resonate bouncing which also caused the highest damage rates.

This suggests that one approach to reducing the impact of these dredges would be to improve the dredges' ability to adapt to changes in substrate without setting off the resonate bouncing which was observed. Mechanical approaches to this could be the use of some form of shock absorber on the springs, which would damp out the bouncing motion of the dredges. Another approach might be to spring each tooth individually, thus the effect of teeth encountering individual rocks would only affect those teeth and not the whole dredge bar.

Further work

Further work could be carried out to investigate the relationships between the behaviour of the dredge and the substrates that it passes over. One approach would be to design a robust instrument like the instrumented scallop with three accelerometers at right angles to each other but with sufficient memory and battery endurance to last for a whole haul. This instrument would be attached to the dredge frame and its outputs from the instrument and ground discrimination equipment compared and also with damage rates of the scallops. Many hauls on a commercial trip could be undertaken rather than just the few hauls with full instrumentation undertaken in this study. The full instrumentation package would probably be better deployed under controlled conditions where dredge behaviour on specific substrates, perhaps on beach trials.

Ultimately it might also be possible to design instrumentation for the Skipper to be able to monitor the amount of resonance occurring in the dredges by the use of accelerometers on the dredge frames. This could perhaps be related this to the output from the seabed texture discrimination systems (such as Rox AnnTM). This could enable adjustments of spring tension or other parameters such as speed to avoid this effect and hence perhaps reduce damage levels.

3.1.1.2 USE OF THE INSTRUMENTED SCALLOP TO ASSESS SPATIAL VARIABILITY IN DREDGE CAPTURE DISTURBANCE IN *PECTEN MAXIMUS*

Introduction

Fishing-induced damage can cause high levels of mortality in scallop populations (e.g. Medcof and Bourne 1964a). Such damage occurs in both captured (Medcof and Bourne 1964a) and non-captured, (Jenkins et al. 2001³, Caddy 1973, Butcher et al. 1981, Shepard and Auster (1991) scallops through the initial encounter with the toothbar, impaction or crushing by the steel mesh belly of the dredge or by other components of the catch whilst in the dredge, or by the sorting process on the deck of the fishing vessel. Of the scallops that are captured a significant proportion will be discarded owing to regulatory or market imposed minimum sizes. In addition, owing to the low efficiency of scallop dredges a large proportion of individuals encountering dredge gear will not be captured. Such individuals can show levels of damage equivalent to that in the catch (Jenkins et al. 2001). The level of damage in scallops will dictate their chance of survival, but even lightly damaged individuals are likely to show high levels of mortality (see [Section 3.1.4.2](#)). The proportion of scallops encountering fishing gear which sustain damage has been shown to vary. For example Medcof and Bourne (1964a) found between 11 and 42% of captured *Placopecten magellanicus* showed signs of external damage, while Shepard and Auster (1991) found that between 7 and 25% of this species which encountered gear but were not captured were damaged. Such variability may be caused by differences in ground type (Shepard and Auster 1991). The great scallop *Pecten maximus* typically shows low levels of visible external damage in comparison with many other non-target species following dredging (Jenkins et al. 2001³). However, given the intensive nature of fishing in areas of high scallop density and the low efficiency of gear such damage and subsequent mortality may have a significant impact on the exploitable stock.

We aimed to investigate spatial variability in the level of damage to *Pecten maximus* in dredge fisheries in the north Irish Sea and determine potential causes of such variability. ‘Instrumented scallops’ [Section 3.1.1.1](#) were used to assess variability among fishing grounds in the degree of mechanical stress experienced by captured scallops in the dredge bag to test the hypothesis that an increase in mechanical stress caused by high volumes of rocks leads to high levels of damage.

Methods

Variability in Pecten maximus damage levels

Bi-annual scallop stock surveys have been undertaken by Port Erin Marine Laboratory around the Isle of Man since 1990 at up to 13 scallop fishing grounds. Surveys are undertaken in October (just prior to the start of the fishing season for *Pecten maximus*) and in June (at the end of the season). During stock surveys between 3 and 4 tows of 2 nautical miles are carried out at each fishing ground using

³ See also [Section 3.1.4.1](#)

a gang of 4 scallop dredges. Following each tow the age, size structure and levels of damage to captured scallops are evaluated along with an assessment of bycatch.

The database of scallop stock surveys was used to investigate spatial and temporal variability in damage to individuals captured in scallop dredge gear. Seven grounds were followed over 2 years to determine if differences among fishing grounds in damage level were temporally consistent.

Investigation of potential causes of damage

During the scallop survey carried out in June/July 2001 damage levels in captured scallops and all major bycatch species were assessed on a 4 point damage scale (see [Section 3.1.4.1](#) for details). The number of individuals found within each damage category was used to calculate a mean damage index (MDI) for each species for each tow using the following formula:

$$\frac{\sum_{i=1}^{i=4} ni}{N}$$

where n_i = number of organisms of damage score i , N = total number of organisms

Visual estimates were made of dredge fullness prior to emptying and the volume of rocks in each dredge estimated using buckets of known volume.

Instrumented scallops were used throughout the survey to assess variability in mechanical stress experienced by scallops on different grounds. The design and output of the instrumented scallop are described fully in [Section 3.1.1.1](#). One or more instrumented scallops were placed in a dredge bag prior to shooting the gear and were retrieved from the bag at the end of the tow before emptying. The instrumented scallop was programmed in advance to begin recording a 10 minute period at approximately the mid point of each tow. At the end of each tow the data in each instrumented scallop were downloaded to a PC on board the research vessel and the internal battery charged if necessary. Eight instrumented scallops were available throughout the survey but the number of instrumented scallops available for any one tow depended on their state of charge. A major limiting factor in the number of instrumented scallops used was equipment loss (through damage to dredge gear) and failure; over the course of the survey 6 out of 8 instrumented scallops were either lost or malfunctioned. In total twenty-five 10 minute periods of data were successfully obtained from 5 grounds (Table 1).

Table 1 Details of instrumented scallop deployment. The dredge number refers to the position of the dredge on the towing beam (1= Inboard, 4 = Outboard). Between 1 and 3 instrumented scallops were deployed per tow. On the 4th tow at Chickens all 3 instrumented scallops were placed in the same dredge bag (Dredge 2). (See Review Section 4.3.1 for locations)

Date	Ground	Tow	Dredge
June 4 th	Laxey	1	4
		1	2,3,4
		2	2,3,4
June 5 th	Chickens	1	2,3
		3	1,2,3
		3	2,2,2
July 10 th	Bradda Inshore	1	2,3,4
		2	2,4
		3	2,4
July 13 th	Bradda Offshore	1	3,4
		3	2
July 16 th	Peel	2	3
		4	3

Results

Variability in Pecten maximus damage levels

Levels of external physical damage showed a high degree of variability among different fishing grounds. Over 4 scallop stock surveys covering 2 years the level of external damage observed in *Pecten maximus* varied from a minimum of 2% to a maximum of 18% of the total catch for a particular ground and date (Figure 1). There was a relatively high degree of temporal consistency in the ranking of different fishing grounds in the level of damage sustained by *Pecten maximus*. At some fishing grounds damage was consistently low (e.g. Bradda Offshore, Peel) and at others damage was consistently high (e.g. Targets, Laxey) (Figure 1).

Over two years (4 scallop surveys) the percentage of captured scallops which were under the minimum legal landing size varied among grounds from 7% to 73%. Over all grounds the percentage of undersized scallops in the catch was 46%. Both undersized (<110mm shell length) and marketable sized (>110mm shell length) scallops showed external signs of damage. Although a greater proportion of marketable sized scallops were damaged, over 4 surveys there was no consistent ranking of the 2 size classes (Figure 2).

Investigation of potential causes of damage

The output of the instrumented scallop consisted of 6000 recordings of acceleration in 3 planes (x,y,z) over a 10 minute period (one recording every 0.1 second). An example of a typical output is shown in Figure 3.

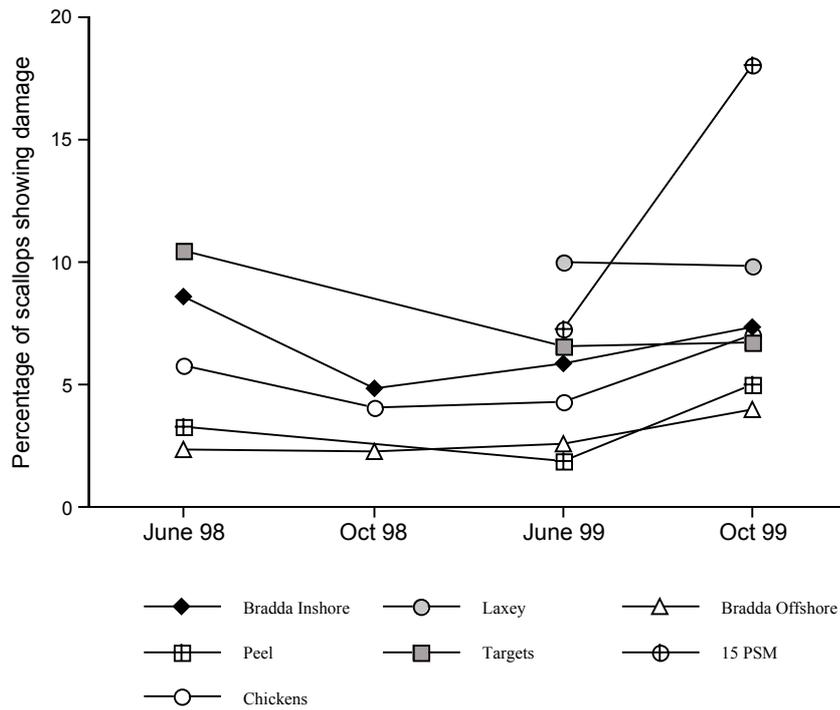


Figure 1 Percentage of *Pecten maximus* captured during bi-annual scallop stock surveys over 2 years showing signs of external physical damage at 7 different fishing grounds

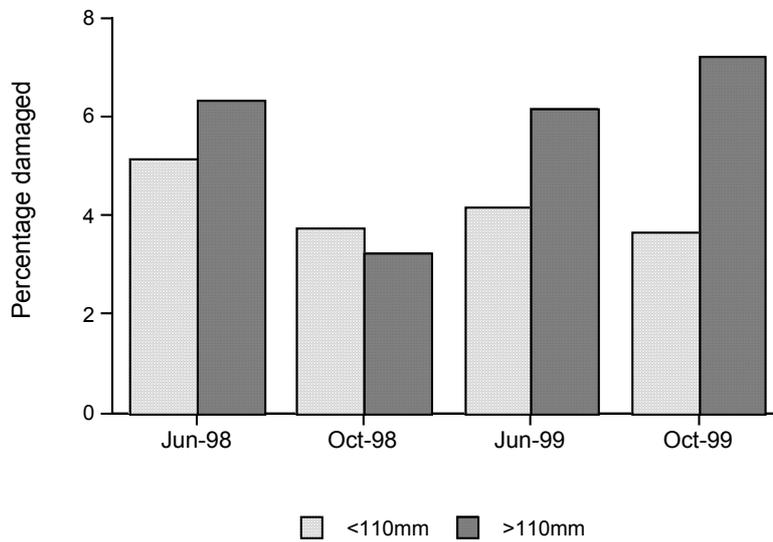


Figure 2 Percentage of undersized (<110mm shell length) and marketable size (>110mm shell length) *Pecten maximus* captured over 7 fishing grounds around the Isle of Man during bi-annual surveys which show external signs of damage.

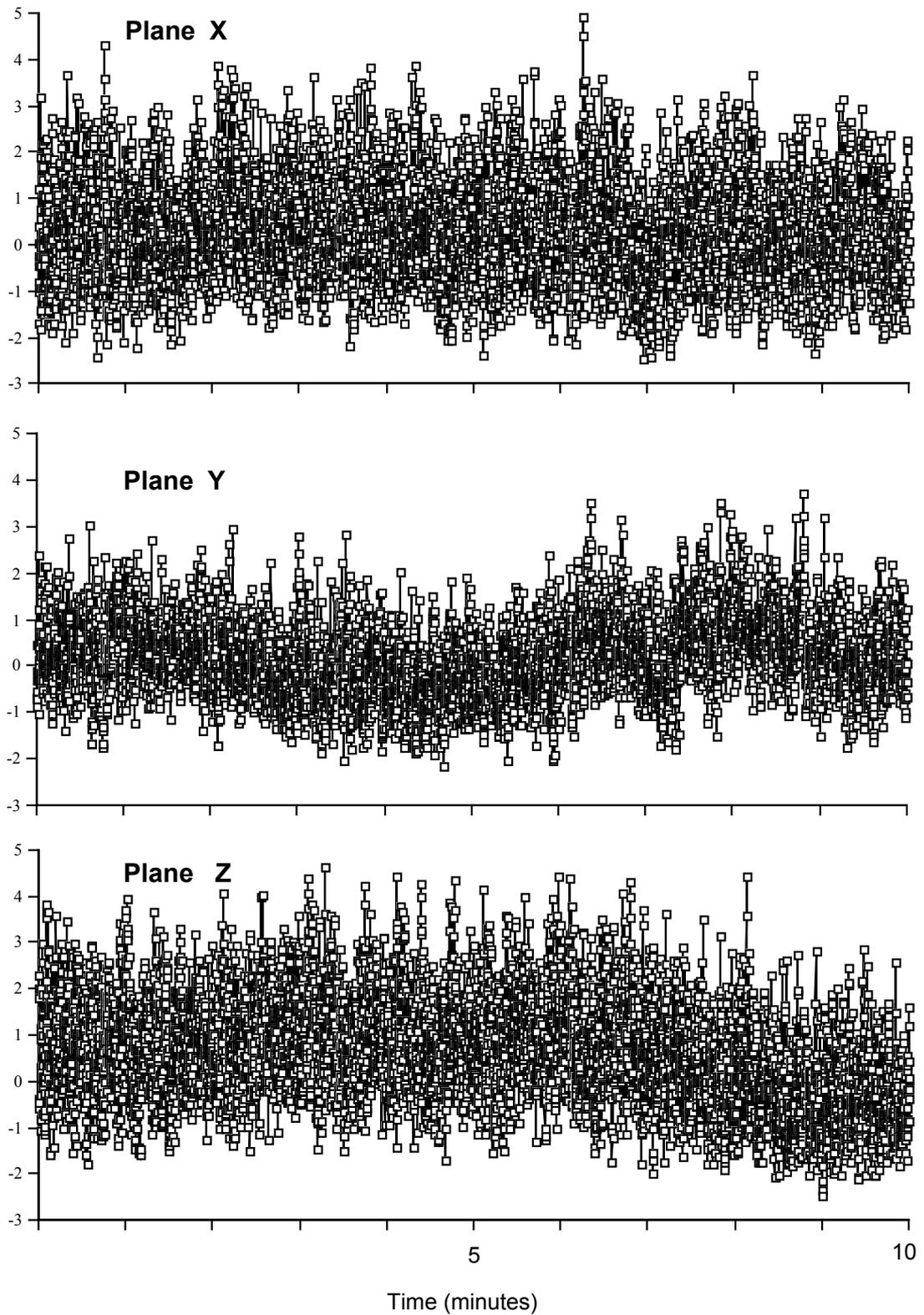


Figure 3 An example of a typical output from instrumented scallop. Each point represents the measure of acceleration in any one of 3 planes every 0.1 seconds.

This output was converted into a single variable (the Resultant modulus - see Section [3.1.1.1](#) for explanation) which gave an estimate of the mean acceleration experienced by the instrumented scallop over the 10 minute recording period. Our aim in assessing variability among grounds in the mechanical stress experienced by scallops was to deploy at least 3 instrumented scallops per tow over at least 3 tows per ground. However, owing to logistical problems and equipment failure it was not possible to maintain this level of replication throughout (see Table 2 for replication levels). The resultant modulus showed a high degree of variability both among individual instrumented scallops within a tow, among tows at a single ground and among grounds (Table 2, Figure 4). One way ANOVA of the resultant modulus for the three grounds in which data for 3 tows are available showed no significant difference among grounds. However, the power of this test is low due insufficient samples and these preliminary data do suggest large differences in the degree of mechanical stress experienced by scallops among grounds.

Table 2 The mean resultant modulus for all instrumented scallops deployed.

Ground	Tow	Replicate 1	Replicate 2	Replicate 3
Laxey	1	0.36		
	2	0.17	0.21	
	3	0.10	0.32	0.13
Chickens	1	0.23	0.12	
	2	0.09	0.18	-0.06
	3	-0.08	-0.05	
Bradda Inshore	1	0.28	0.39	0.06
	2	0.06	0.19	
	3	0.22	0.36	
Bradda Offshore	1	0.37	-0.03	
	2	-0.00		
	3			
Peel	1	0.53		
	2	0.58		
	3			

In order to investigate the potential causes of damage in *Pecten maximus* and the variability observed among fishing grounds the mean damage index of scallops was plotted against three variables: the volume of rocks caught in dredges, the dredge fullness, and the index of mechanical stress obtained from the instrumented scallop (the mean resultant modulus). In graphing these variables a single tow was used as a replicate and the damage sustained by all scallops captured in the 4 dredges was used to determine a mean damage index for the tow. Similarly where more than one instrumented scallop was used per tow, the mean resultant modulus for the tow was calculated. There was no apparent relationship between the mean damage index in *Pecten maximus* and any of the three variables measured (Figure 5). Investigation of how the index of mechanical stress obtained from instrumented scallop varied with the catch characteristics (volume of rocks caught, dredge fullness) revealed no clear pattern.

Three commonly captured bycatch species, the edible crab *Cancer pagurus*, the common starfish *Asterias rubens* and the seven armed starfish *Luidia ciliaris* were used to investigate further the relationship between damage sustained and the recorded physical variables (Figure 6). Only one significant regression was found; there was a significant positive relationship between the damage sustained by *Asterias rubens* and the volume of rocks caught in dredges.

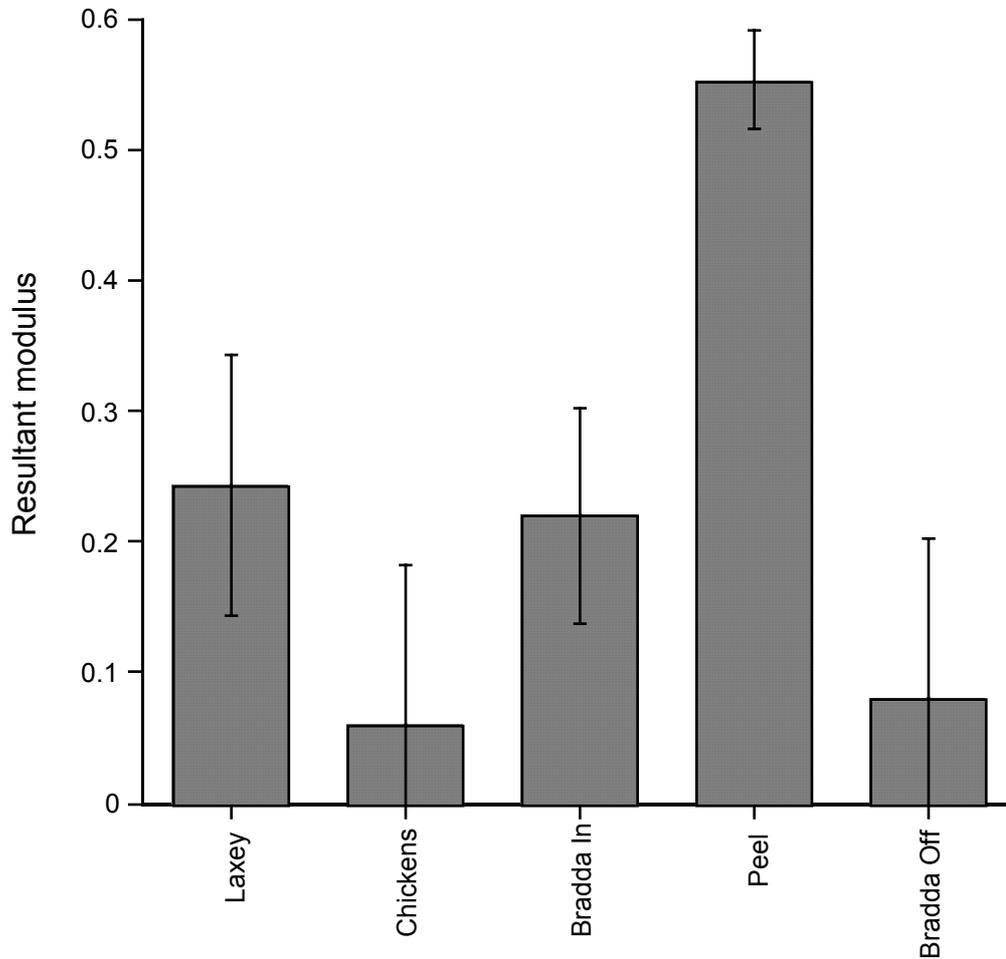


Figure 4 The mean resultant modulus measured by instrumented scallop for 5 scallop fishing grounds around the Isle of Man. The results are the average of 3 tows in the case of Laxey, Chickens, Bradda Inshore and 2 tows in the case of Peel and Bradda Offshore. Error bars \pm 1SE

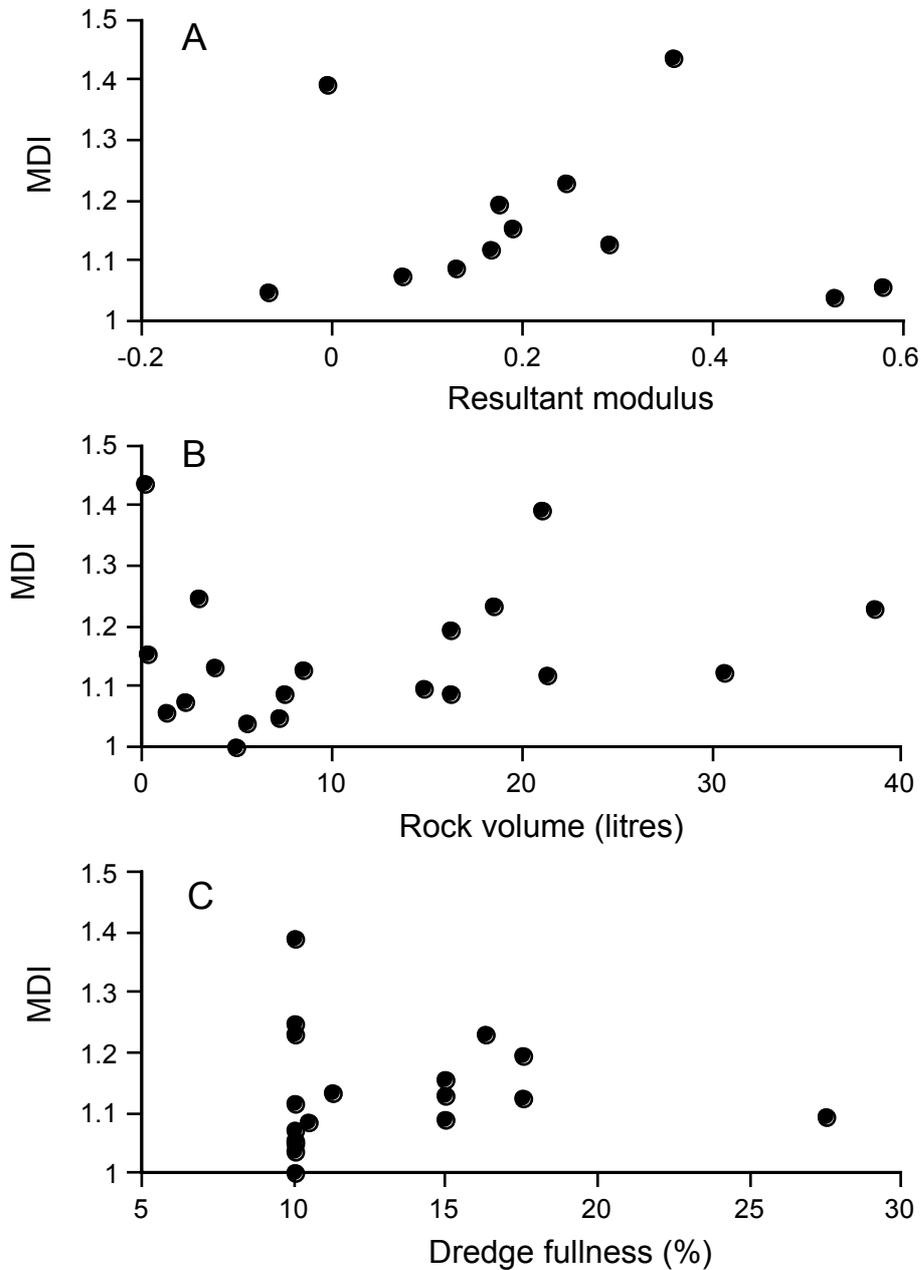


Figure 5 Relationship between the mean damage index in *Pecten maximus* and A) the resultant modulus calculated from the instrumented scallop output, B) the volume of rocks caught and C) the fullness of dredges. Each point represents an estimate of the variable for a single tow on a particular ground.

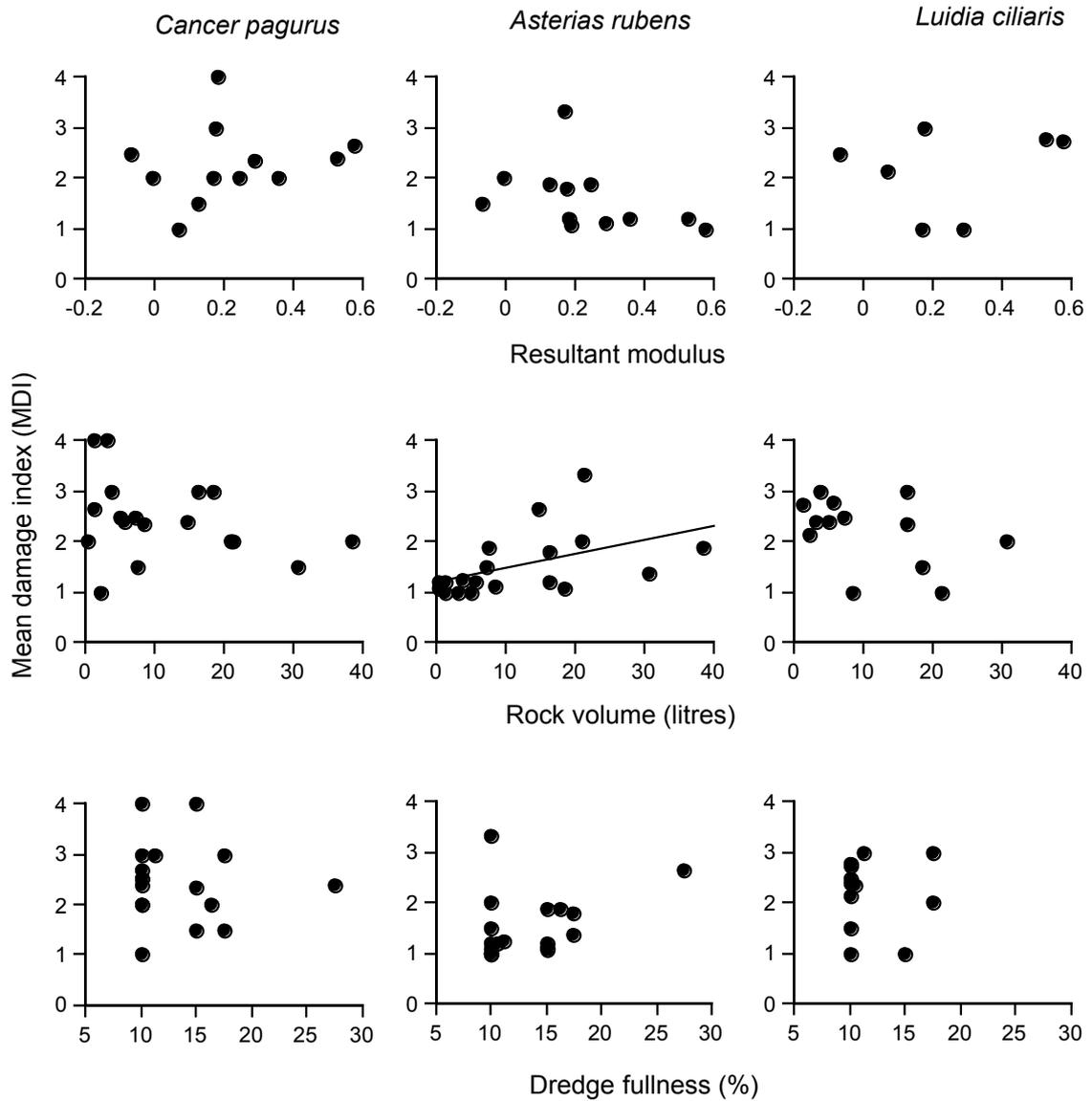


Figure 6 Relationship between the mean damage index in three bycatch species *Cancer pagurus*, *Asterias rubens* and *Luidia ciliaris* and the resultant modulus calculated from the instrumented scallop output, the volume of rocks caught and the fullness of dredges. Each point represents an estimate of the variable for a single tow on a particular ground. Regression line is fitted only where a significant regression was found.

Discussion

There was clear temporally consistent variability in the level of damage sustained by *Pecten maximus* among different fishing grounds. Thus, given the high probability of mortality in damaged scallops ([Section 3.1.5.1](#)) it is likely there is a high degree of variability in fishing induced mortality among grounds. The cause of such variability among grounds is unclear.

The output from instrumented scallop will be affected by a number of different variables, including seabed roughness, volume of rocks captured by dredges and the total volume of catch. We found no relationship between the output from instrumented scallop and measures of rock and catch volume indicating that any relationship will be a complex one involving other interacting factors such as seabed roughness.

There was no clear relationship between the output from instrumented scallop and the damage indices for *Pecten maximus* and bycatch species. There were large differences among individual tows and among grounds in the output of instrumented scallop indicating that scallops will be subject to different degrees of mechanical stress during capture. The lack of a relationship indicates that such variability of mechanical stress whilst in the dredge bag may have little bearing on the degree of damage sustained. One possibility for the lack of a relationship in *Pecten maximus* is that the majority of damage in this hard shelled species does not occur in the dredge bag, but at the teeth. Damaged scallops frequently show damage which is clearly caused by puncturing by dredge teeth.

The variability among grounds in damage level in *Pecten maximus* was not reflected in variability among grounds in instrumented scallop output. For example the ground with the greatest degree of mechanical stress (Peel) had one of the lowest levels of damage. At Laxey, where damage levels are high, the output from instrumented scallop indicated intermediate levels of mechanical stress. However it is clear that more replication is required to accurately characterise grounds in the level of stress experienced by scallops.

Our results do not offer a clear explanation of the causes of damage in *Pecten maximus*, but do point to the fact that damage at the point of contact with the teeth may be important. If this is the case then radical gear designs which avoid the use of teeth for scallop capture would result in a reduction in scallop damage.

3.1.1.3 THE USE OF INSTRUMENTED SCALLOP TO ASSESS DISTURBANCE IN SIMULATED AND FIELD DREDGE CAPTURE.

Introduction

The aim of this experiment was to use the instrumented scallop to compare acceleration and pitch and roll in field dredge capture in order to construct a dredge simulator.

Methodologies

The simulator was a meat turner manufactured by Hans Lenze, Bosingfeld/Lippe. It had a stainless steel barrel with a diameter of 60cm and a depth of 80cm and was powered by two motors each 3.8/220 Volts with 1.15/2 Amps. During dredge simulation experiments it was rotated at an angle of 45° and had a variable speed controller. The simulator was filled with seawater and small rocks, stones and sand in order to mimic conditions in a scallop dredge bag. The scallops were placed in a mesh bag and suspended from a rope across the top of the barrel.

The instrumented scallop is described in [Section 3.1.1.1](#). Three and four replicate tows (containing the instrumented scallop) respectively were carried out in the Isle of Man (spring toothed dredge) and the Rade de Brest (modified water jet dredge). Descriptions of these dredges can be found in [Review Section 3.2.1](#) and [Section 3.3.3.1](#) respectively. The instrumented scallop was placed in the simulator and 4 replicate dredge events were carried out for two different speeds.

Acceleration and angle of pitch and roll were measured every tenth of a second for ten minutes and the mean movement was calculated for each dredge event.

Results and Discussion

The graphs showed that even though there was a variation between the different dredge treatments there was also a variation between the replicate tows of each dredge treatment. The mean acceleration and pitch and roll results of all the replicate tows for the simulation experiments fell in between the minimum and maximum mean results for the field trials. Therefore the dredge simulator was used for further experiments to investigate the effect of dredging on undersized scallops.

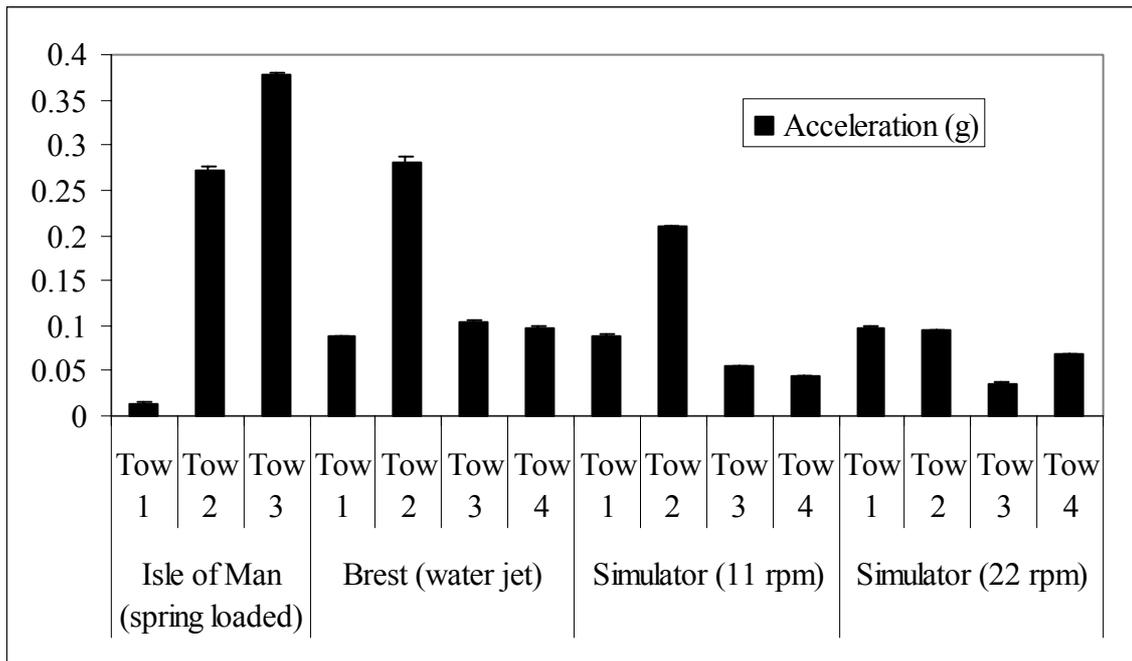


Figure 1 – Mean acceleration of the instrumented scallop placed in two different dredges (spring loaded and modified water jet) and a dredge simulator at two different speeds (11 and 22 rpm).

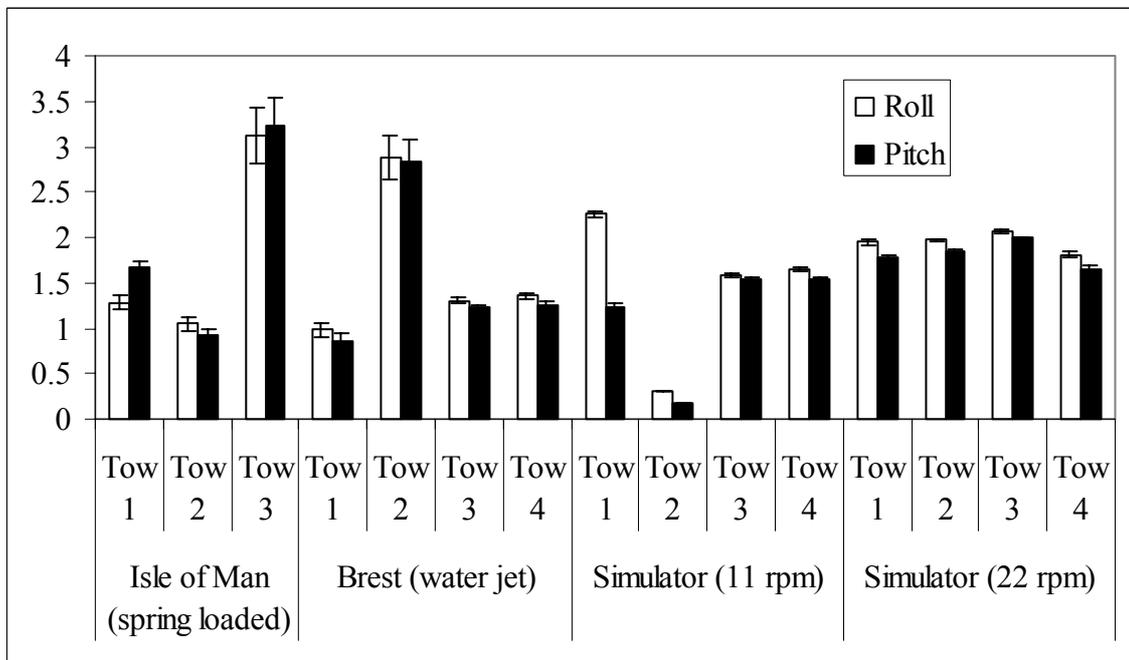


Figure 2 – Mean angle difference of pitch and roll of the instrumented scallop placed in two different dredges (spring loaded and modified water jet) and a dredge simulator at two different speeds (11 and 22 rpm).

3.1.1.4 INSTRUMENTATION AND PERFORMANCE ASSESSMENT OF THE TURBO SOFFIANTI HYDRAULIC DREDGE SYSTEM

Introduction

The ECODREDGE research contract has been established with European Union support to evaluate and improve shellfish dredge design and fishing effort in relation to technical conservation measures and environmental impact. Included in the research programme is the instrumentation and monitoring of a turbo soffianti dredge used in Italian waters.

The objective of this part of the research contract was to install instrumentation on the turbo soffianti dredge and investigate how the dredge interacted with the seabed. This part of the ECODREDGE contract was carried out with the assistance of CNR in Venice and the University of Padova.

A turbo soffianti dredge was instrumented to determine the water pressure available at the nozzles and the depth to which the sand at the seabed was fluidised. Trials were performed at two sites near Venice and results of these tests are presented and discussed. The main variables of water jet pressure and speed over the seabed were varied to investigate the performance of the dredge and the depth to which sand was fluidised.

Research work has been done investigating the effect of water jets on sand as part of studies relating to hydraulics and sedimentology. A literature review has been performed and is summarised in this report. Additional laboratory testing to complement this work was performed and results of this testing are presented and analysed.

The laboratory research work has been extended to the results of the field trials and comments are made on the validity of the models developed.

The lifting of large volumes of sediments into suspension is undesirable and fishing activities are limited to regulate the resulting disturbance. Based on the results of the trials, comments are made regarding the volume of material which may be lifted into suspension by hydraulic dredges. Suggestions are also made regarding how a dredge may be modified to reduce the depth of disturbance into the seabed. In selecting suitable methods, emphasis has been placed on recommending modifications which are easily identifiable by simple observations and measurements.

Instrumentation

Two turbo soffianti dredges were used for the trials. Both were of similar design as described by Brambati and Fontolan (1990). The instrumentation fitted to the dredges comprised:

- A pressure transducer mounted on the dredge.
- A displacement sensor to determine the depth of seabed disturbed.

The pressure transducer was mounted on the dredge to enable the water pressure to be measured as close to the jet nozzles as possible and hence remove the need to allow for any head losses and hydrostatic head of water.

The displacement sensor was mounted on the dredge to determine the depth to which the seabed was disturbed. The measurement was made using a spring loaded trailing arm connected to a transducer. For the first set of trials at Jesolo, the trailing arm was mounted on the front of the dredge to run on the seabed immediately in front of the cutter bar (lama). However, during commercial fishing operations between fitting the instrumentation and performing the trials, the bracket broke. As a result it was necessary to reconfigure the position of the trailing arm. The revised location was to the side of the dredge and behind the cutter bar. Reference for the displacement sensor was the underside of the skids supporting the dredge.

For the second set of trials at San Pietro, the equipment was modified to locate the fluidisation depth measuring lever on the rear of the dredge. However the dredge design was slightly different and it was felt that the lever would become damaged during deployment. The lever was therefore placed on the side of the dredge, but still behind the jet nozzles due to the tapered shape of the dredge.

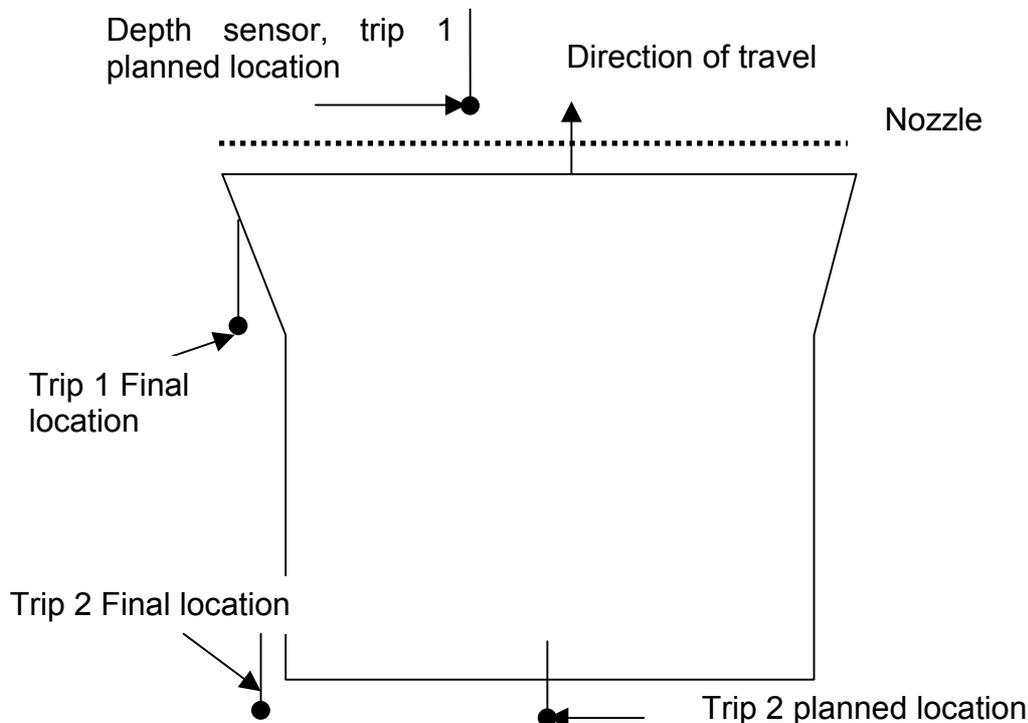


Figure 1 Plan view of dredge showing location of depth of fluidisation sensor on the two trips trip 1 Jeselo and trip 2 San Peitro

Field Operations

Field trials were conducted at two locations. The first series of trials was performed in 1999 at Lido di Jesolo, the second series was performed in 2001 near San Pietro. Both sets of field work are described below.

Lido di Jesolo

Field Work

The first series of trials with the instrumented dredge were performed using the fishing vessel *Ketty* on 10th June 1999. The crew were Marino Padovan, Cristian Chiaranda, Peter Allan and Michele Pellizzato. Adverse weather conditions had prevented the trials commencing for two days. Thus the time available for the trials was substantially reduced. The trials were performed at two sites close to Faro Piave (Lido di Jesolo).

Fishing operations on this vessel are performed by dropping an anchor, and sailing a short distance away. Dredging was conducted by hauling in on the anchor line. It was possible to independently vary jet pressure and speed of the dredge over the seabed.

During the trials, the speed was monitored by timing the rate at which a predetermined length of cable was reeled in. For the trials, a 100m dredge was performed, and the speed was maintained constant throughout any one dredge. Tests were also performed at various different pressures.

Variables which were not altered included the nozzle diameter and the stand off distance of the nozzle above the seabed. Both are set by the configuration of the dredge and cannot be altered without major modifications to the dredge.

The nozzle diameter on the dredge was 6.5mm and the stand off distance of the nozzle above the seabed was approximately 40mm.

The trials are summarised below:

Table 1 Summary of Dredge Trials

Trial No.	Gauge Pressure (bar)	Time (sec)	Length (m)	Speed (m/s)
1	3	119	100	0.84
2	2.5	255	100	0.39
3	3	88	100	1.14
4	1.5	378	100	0.26

Site 1 : 5m water depth, 0.34 NM from shore

Trial No.	Gauge Pressure (bar)	Time (sec)	Length (m)	Speed (m/s)
5	3	155	100	0.65
6	2.5	243	100	0.41
7	3	85	100	1.18
8	1.5	418	100	0.24

Site 2 : 7m water depth, 0.56 NM from shore

For trials 1 and 5, the crew were asked to dredge at a speed and pressure which was typical of their normal fishing method. Speed of the dredge over the seabed and pressure of the water jets was varied around these normal values. Due to time constraints as full a variation as may have been desirable was not possible.

Results from the trials are shown as Figures 14-21 (p74-77). These show the output from the pressure transducer, after correcting for the hydrostatic head of water (water depth) and the trailing arm following the surface of the undisturbed seabed. Depth of fluidisation in trial 7 was greater than 80mm for periods of time during the dredge. The maximum extension of the depth measurement arm was 80mm. It would therefore appear that the depth of fluidised soil was greater than 80mm. Possible explanations include a particularly loose, fine soil, susceptible to fluidisation, or bed forms such as sand ripples.

For analysis of the trial results an average of the three variables (speed, pressure and depth of fluidisation) was made. This was based on a visual interpretation of Figures 14 to 21. The values derived are presented as Table 2.

Table 2 Summary Results of Dredge Trials

Trial No.	Water Depth	Pressure		Speed	Depth of Fluidisation	Water Volume	Nozzle Velocity
		Total	Differential				
	(m)	(bar)	(bar)	(m/s)	(mm)	(m ³ /s)	(m/s)
1	5	3.84	3.34	0.84	34	6.5E-04	19
2	5	2.31	1.81	0.39	38	5.7E-04	17
3	5	2.43	1.93	1.14	29	5.9E-04	18
4	5	1.13	0.63	0.26	36	3.4E-04	10
5	7	2.53	1.83	0.65	32	5.7E-04	17
6	7	2.20	1.50	0.41	37	5.2E-04	16
7	7	2.51	1.81	1.18	30*	5.7E-04	17
8	7	1.50	0.80	0.24	37	3.8E-04	11

* Arm reached maximum extension – typical value quoted.

Sediment Analysis

A sample of sediment was collected from the dredge following the trials. The exact location from which this sample came is not known. Three further samples of sediment were collected by CNR as part of their work for the same research contract. Sample locations were in a line from Jesolo across the site of the trials in water depths of 4m, 6m and 8m. All four samples were tested to determine their particle size distribution. Results of this analysis are presented as Figure 22 (p78).

The close similarity of the sand is demonstrated by the particle size analysis. A slight increase in fines content is noted as the water depth deepens from 6m to 8m. The sample recovered from the dredge falls within the envelope of the samples tested and indicates that the samples are representative of the seabed soils present at the time of the dredge trials. Typical d_{50} size is 0.07mm.

San Pietro

The design of clam dredges and method of fishing used in this area is slightly different to that at Jesolo. For these trials the fishing vessel Nedduno was used, and crewed by fishermen Mose Vianello and Giovanni Vianello, with testing operations done under the supervision of Michele Pellizzato and Julia Fernley.

While the dredging operation is essentially similar to that performed at Jesolo, in this case the fishing vessel pulls the dredge backwards under its own power. As the water pump supplying jet water to the nozzles, and the propeller are mechanically linked, it was not possible to independently vary the speed of the dredge over the seabed and the nozzle water pressure. Based on experience, the fishermen were able to give an indication of the speed of the dredge and water jet pressure.

Testing took place on 12th September 2001 and a total of five trials were performed. A summary of tests performed is given in Table 3. As only the engine speed could be varied, a series of tests was performed at different engine speeds. Speed of the dredge over the seabed, and the water jet pressure quoted in Table 3 are based on the fishermen's experience. During the fifth dredge, the equipment became damaged.

Table 3 Details of trials performed at San Pietro

Test	Engine speed (rpm)	Dredge Speed (m/s)	Water Jet Pressure (bar)	Notes
1	1200	1.0	2	Normal fishing operations
2	1400	1.2	5	Chain caught under skid, vessel not thought to move as fast as normal for engine speed.
3	1100	0.8	<1.5	Data lost due to problem with logger
4	1400	1.2	5	Depth measuring lever damaged slightly during test.
5	1100	0.8	<1.5	Lever bent during test and transducer pod became detached.

Results from this set of tests have been calibrated and the variables of pressure and depth have been determined from the test results. This analysis is shown as Table 4. Results are presented graphically as Figures 23 to 26 (p78-80).

Table 4 Calibrated results of San Pietro Trials

Test	Dredge Speed (estimated) (m/s)	Water Jet Pressure (bar)	Nozzle Velocity (m/s)	Depth of Scour (mm)	Notes
1	1.0	1.29	15.3	86	Normal fishing operations
2	1.2	4.94	30.0	91	Chain caught under skid, vessel not thought to move as fast as normal for engine speed.
4	1.2	4.26	27.8	93	Depth measuring lever damaged slightly during test.
5	0.8	0.86	12.5	85	Lever bent during test and transducer pod became detached.

Inspection of the results indicates some scatter in the data. It is not clear why this is the case, however a probable explanation is use of a piston pump which does not deliver a steady flow.

Sediment Analysis

Results of four particle size distribution analyses have been provided by University of Padova. Results of these analyses are presented as Figure 27 (p82). The results show a slightly wider grading than that of the Jesolo samples with a d_{50} size in the range 0.06mm to 0.12mm. The samples fall towards either limit of the grading envelope. This may be indicative of a change in nature of the sand across this fishing ground.

Theoretical analysis of Water jets acting on sand

Introduction

An assessment of the ability of a water jet to erode sand may be approached from a simple sediment mechanics perspective of water velocity required to erode sand, or the combined hydraulics and sediment mechanics work performed on the scouring action of a jet directed across the surface of a sand bed.

Erosion Potential of Sand

Sediment mechanics is concerned with the mobility and deposition of soils under the action of surface currents of both air and water. As such it can provide a useful basis for estimation of the current velocity required to remove sand particles from a bed.

Sediment mechanics theories indicate that the turbulence of a current passing over a seabed creates an uplift force which overcomes the self weight of the particle. For fine grained soils (clays and silts), relatively fast currents are required to generate sufficient surface drag to lift particles (due to the smooth nature of the surface and to overcome adhesion), despite their low weight. As soil becomes coarser (fine sand), greater turbulence is generated and individual soil particles may be lifted out of the seabed by lower velocity currents. However coarse soils such as gravels have much greater self weight and the required current velocity increases again. It may also be noted that relative density or angle of friction is not considered to have a significant effect. This relationship was first recognised by Hjulstrom in 1935 and has since been modified and refined by various workers (Miller et al. 1977; Mantz 1983).

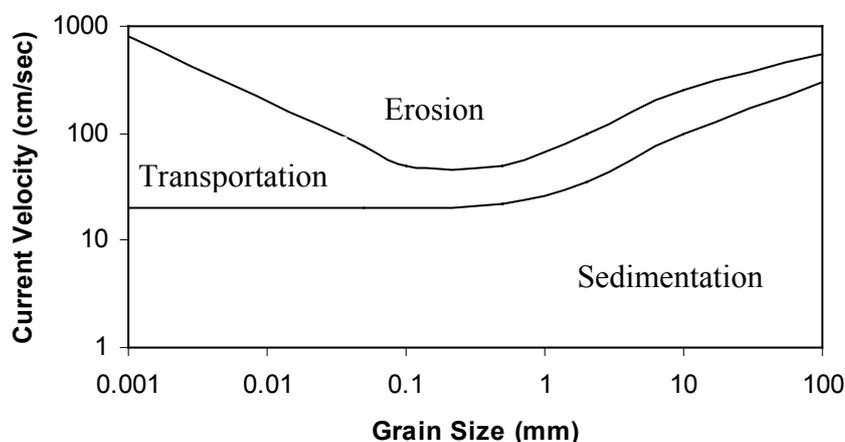


Figure 2 Grain Size/Current Velocity for Transportation and Erosion

A feature of the Hjulstrom curve (Figure 2), of interest here, is the relative ease with which fine sands, in the range 0.06mm to 1mm grain size are moved. Note is also made of the low current velocities required to transport clay (<0.002) and fine silt particles combined with the relatively high current velocities for erosion of the same soil. It should be noted that the curve is strictly only applicable to uniformly graded materials. However nearshore sands, such as those on shell fishing grounds may be considered to be uniform and therefore the curve is generally applicable. The particle size distribution analyses for Jesolo, and San Pietro, described in the previous section may both be considered to be uniform.

Scour of Water Jets

The mechanics of a water jet scouring a sand bed have been investigated by a number of workers. The primary interest for research in this area is associated with scour from pipelines and outfalls. As such, all the work traced considers a static pipe outlet, rather than the case of an outlet moving over the surface of a sand bed. However the work is considered to give some useful indications of the interaction of a water jet with a sand bed. Several papers on this subject are listed in the bibliography. The areas discussed below consider both a vertical jet and an oblique jet. A number of papers also discuss a horizontal jet acting parallel to the surface of a sand. As this is not applicable to the case of a hydraulic dredge, these are not discussed further.

Critical Scour Froude Number

For scour of a sand to occur under the action of a water jet, it is necessary for a certain velocity to be achieved. This is a function of both the nozzle diameter and the mean particle diameter, related by the Froude number. Rajaratnam and Beltaos (1977) investigated this value and concluded that:

$$F_r = \frac{U}{\sqrt{g(\rho_s - \rho_f)d_p / \rho_f}} = 0.44\beta \left(\frac{d_p}{d_j} \right)^{-0.39}$$

Where	F_r = Froude number
	U = jet velocity at impact point
	ρ_s = density of solid particles
	ρ_f = jet fluid density
	g = acceleration due to gravity
	d_p = mean particle diameter
	d_j = nozzle diameter
	β = a constant, 6.3 for a stationary fluid

From the above equation it is possible to calculate the minimum diameter/minimum flow velocity required to scour sand. This is shown as Figure 3. It can be seen that in practice this lower bound value is unlikely to be approached in sands, however it may be approached or exceeded in medium and coarse gravels.

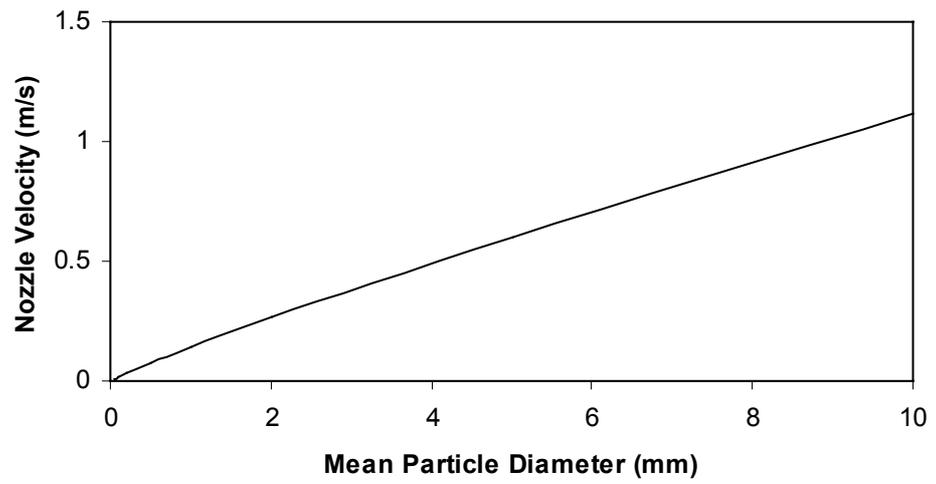


Figure 3 Critical velocity for a 10mm diameter nozzle

Vertical Water Jets

The profile of a sand bed resulting from the action of a vertical water jet has been discussed in some detail by Niven and Khalili (1998). They performed a series of experiments investigating the scour profile as a nozzle was advanced into a sand bed, however the work was performed at a pressures lower than would normally be associated with a jet trenching tool. Five different sands were used ranging from a coarse sand to a silty clayey sand. A typical profile obtained is shown as Figure 4. The profile and corresponding nozzle tip depth are denoted by test number on the chart.

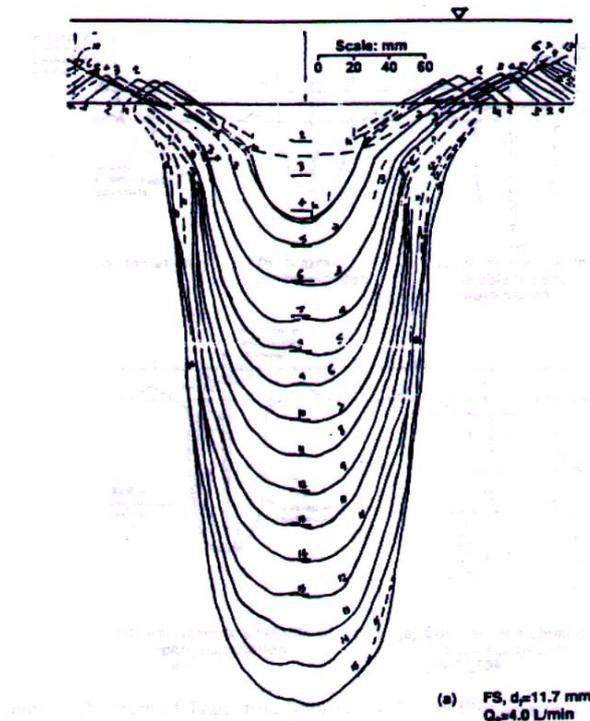


Figure 4 Scour profiles at different nozzle depths

Interesting features to note are the well defined and steep sided nature of the cavity formed. This cavity was formed in a period ranging from 1 to 8 minutes. Unfortunately no details of the rate of formation are given in different soils. In all cases a ridge of sand was formed around the cavity as sand was washed out. The cavity represented a steady state condition between sand rolling back in and being ejected out by the flowing water.

The results were normalised and it was found possible to achieve a good fit of data without reference to the median sand particle diameter. An example is shown in Figure 5.

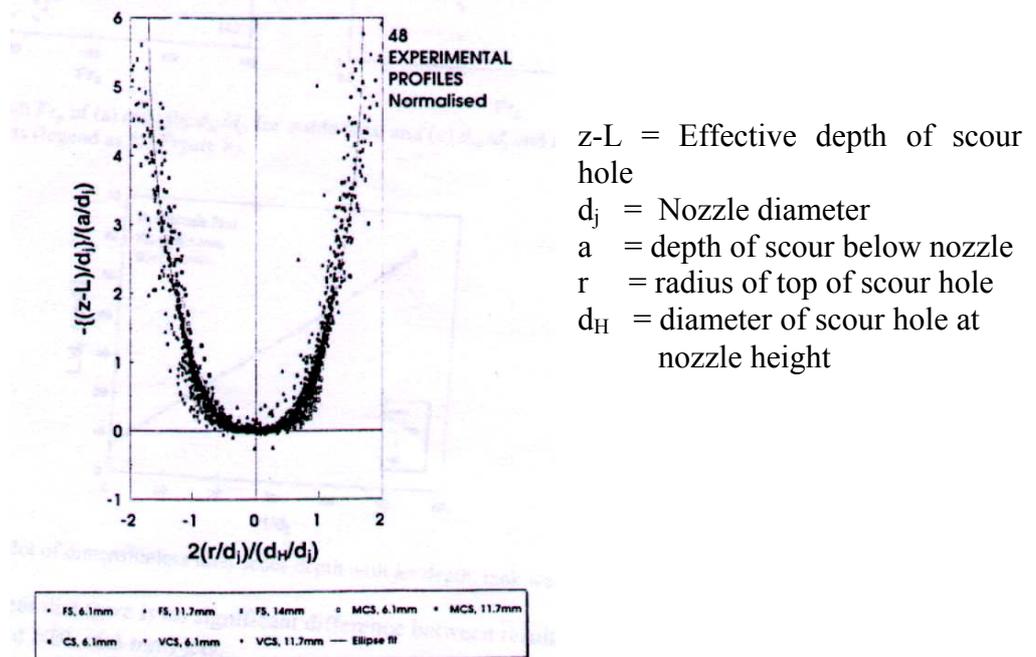


Figure 5 Example of Normalised Scour Profile

These results suggest that the depth of scour is more highly dependant on the hydraulic aspects of the jet nozzle than the nature of the sand. However no measurement of time for stabilisation was made and this therefore cannot be assessed.

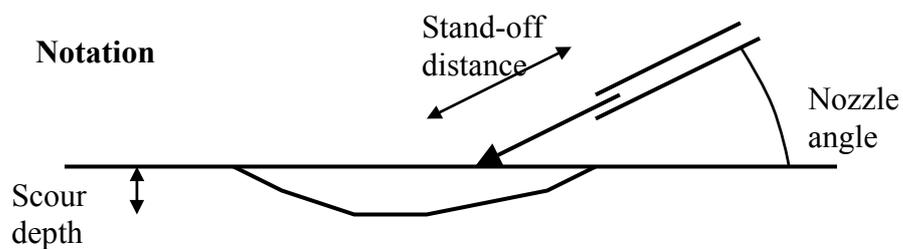
Tests with a smaller diameter nozzle, and a correspondingly lower water flow, indicated that as depths increased, the cavity would become unstable with a fluidised zone oscillating from side to side around the nozzle pipe. Ultimately this stabilised to a fluidised bulb of sand around the nozzle. The development of such a situation is of interest, but will not be applicable to a hydraulic dredge.

Effect of Jet Angle

The angle at which a water jet impacts on a sand bed will clearly affect the nature of the scour hole produced. A valuable contribution was made to this subject by Rajaratnam et al. (1995). This work aligned a jet nozzle at a series of different angles and elevations to a sand bed and with varying flow rates. A summary of the results obtained is given as Table 2. This is useful as it allows the data to be manipulated in ways other than those reported by Rajaratnam et al. (1995) and for this reason details are provided.

Table 1 Summary Test Results after Rajaratnam et al. (1995)

Nozzle Angle (°)	Median Sand Diameter (mm)	Stand-off distance (mm)	Nozzle Velocity m/s	E ($=f[F_0, H, d_j]$)	Scour hole depth (mm)
45	2.38	42	2.53	1	48
45	2.38	41	2.46	0.99	44
45	2.38	41	1.58	0.63	26
45	2.38	61	2.77	0.91	55
45	2.38	58	2.04	0.69	42
45	2.38	58	1.39	0.47	31
45	2.38	57	1.84	0.63	34
45	1.2	41	1.86	1.05	42
45	1.2	41	2.31	1.3	60
45	1.2	41	2.52	1.42	69
45	1.2	59	1.62	0.76	60
45	1.2	58	2.53	1.2	69
45	1.2	58	1.62	0.77	56
45	1.2	41	1.42	0.8	44
60	1.2	19.6	2.09	1.71	38
60	1.2	18.4	2.08	1.75	41
60	1.2	17.3	2.26	1.96	43
60	1.2	17.3	1.76	1.53	31
60	1.2	32.3	1.89	1.2	38
60	1.2	33.5	2.45	1.53	56
60	1.2	34.6	2.05	1.26	41
60	1.2	34.6	1.12	0.69	23
30	1.2	74	1.98	0.83	83
30	1.2	74	1.43	0.6	54
30	1.2	74	1.23	0.52	44
30	1.2	98	2.16	0.79	93
30	1.2	98	1.27	0.46	45
30	1.2	98	1.63	0.6	67
30	1.2	98	1.43	0.52	56
10	1.2	248	1.27	0.29	28
10	1.2	248	1.53	0.35	35
10	1.2	248	2.24	0.51	92
10	1.2	248	1.74	0.4	48



It should be noted that Rajaratnam et al. (1995) only recorded data after the tests had been running for 24 hours or more. No indication is given of the time taken for the sand erosion to reach equilibrium conditions. The variability of the soil was taken into account by a calculated constant E , which is a function of the Froude number, the stand-off distance and the nozzle diameter.

The data has been manipulated to investigate the depth of the scour hole at different nozzle angles. The results of this analysis are presented as Figure 6. The most interesting factor of this analysis is that at a shallow nozzle angle (10°), there is a significant variation in scour depth for a relatively small change in the E value. However as the angle increases to 45° , at which angle two sand sizes were tested, the scour depth becomes much more constant and shows little variation between the two materials. Progressing further to 60° , the change in scour depth is negligible.

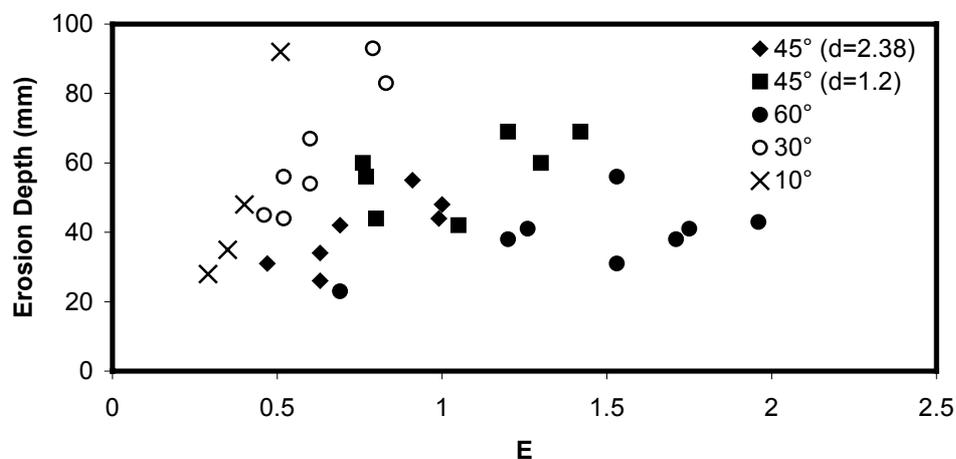


Figure 6 Erosion Depth V E at different nozzle angle (to the horizontal)

A study of Figure 6 suggests that higher nozzle angles produce scour depths which are less influenced by stand-off distance and grain size. This may suggest that improved out of straightness during trenching could be achieved by use of high angles.

The action of a water jet on a sand bed has been discussed by a number of workers (eg Rajaratnam et al. 1995, Niven and Khalili 1998). Most workers in this field have not found it possible to develop rigorous mathematical relationships. This is due to the large number of variables, including particle grain size, jet diffusion in the vicinity of the scour hole and the movement and effect of scoured materials. For this reason most workers have developed empirical models.

A brief discussion of the models available is provided below. The relationships provide a basis for the assessment the performance of the hydraulic dredge and identifying the major factors which affect the performance.

Niven and Khalili (1998) suggested the following general form for scour depth equations:-

$$\frac{d_s}{h} = f[F_r, R_e]$$

Where d_s = depth of scour hole
 h = nozzle stand off height
 F_r = Froude number
 R_e = Reynolds number

Both the Froude number and the Reynolds number are relatively complex terms based on a number of variables. Rajaratnam et al. (1995) suggested a simpler form for the above equation:-

$$d_s = K \cdot \frac{q^s \cdot H^y}{d^z}$$

Where d_s = depth of scour hole
 H = nozzle pressure
 d = mean particle diameter
 q = jet water volume
 K, s, y, z = constants

Rajaratnam et al. (1995) state that analysis of experimental data indicates s and y to be approximately 0.6 and 0.25 respectively. K is noted to be typically in the range 0.2 to 3.0, while z may vary between 0 and 0.5.

Laboratory testing

The literature review gave a useful indication into the depth of scour of water jets and the effect of nozzle angle. However it was not possible to find any appreciable indication of the effect of time of exposure or effect of speed of a nozzle traversing over the surface of a sand. As this is critical to an assessment of the depths to which a hydraulic dredge may penetrate, further investigation was performed in a programme of laboratory tests.

The objective of this exercise was to identify the effect of time of exposure, and enable this to be correlated to the speed of a dredge over the seabed. Testing was performed in a specially designed tank at the University of Newcastle upon Tyne. A series of different sands and pressures were used for the testing.

Test configuration was to direct a vertical jet at the sand bed. This is shown in Figure 7. Water pressure was from a mains supply giving up to 4bar at the nozzle. Two valves were used, a needle valve to provide a fine adjustment to the flow, and a ball valve for quickly turning the flow on and off. The procedure used was to set the needle valve to give the correct pressure at the nozzle without the sand bed present. The sand bed was then placed under the nozzle and the time of exposure controlled using the ball valve.



Figure 7 Experimental set up for laboratory jetting trials

Depth of fluidisation of the sand was monitored by placing small metal markers on the surface of the sand. These sank down in the sand during exposure to the water jet and remained in position when the water jet was turned off and the sand settled out of suspension. It was possible to find these with the assistance of a magnet. Three markers were used to ensure consistency of results. The close correlation between markers confirmed that this was a representative method of measurement. In addition to performing tests with a controlled time period, a small number of tests were also performed with the jet nozzle traversing over the surface of the sand.



Figure 8 Model sand bed with depth of fluidisation markers exposed

Tests were performed in sand of three gradings. Particle size distribution analyses are presented as Figure 9. Results of the tests performed are presented in Table 2 (a toc). For all tests a nozzle diameter of 5mm was used. This approximates to the size used on the turbo soffianti dredges.

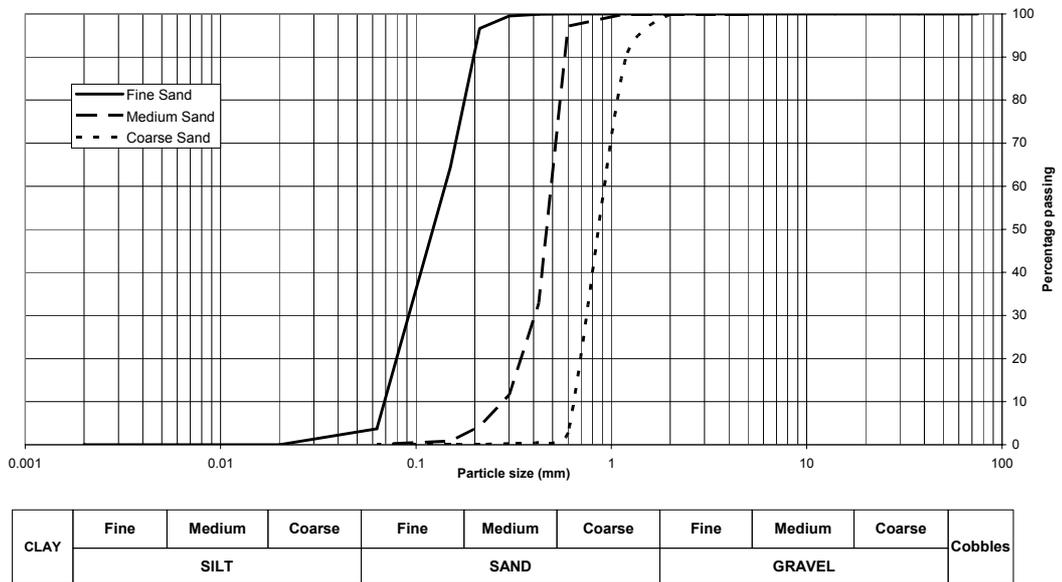


Figure 9 Particle size distribution analyses for sand

Table 2a Medium grained sand, static nozzle

Run	Stand off distance (mm)	Flush time (sec)	Pressure (kPa)	Static Depth (mm)	Depth Marker A (mm)	Depth Marker B (mm)	Depth Marker C (mm)	Depth Average (mm)
1	85	2	27	10	74	79	84	79
2	85	5	27	9	110	110	110	110
3	85	2	53	12	101	101	108	103
4	85	5	53	10.5	110	122	122	118
5	130	2	27	15	64	64	71	66
6	130	5	27	13	88	91	91	90
7	130	2	53	15	75	74	71	73
8	130	5	53	14	113	121	125	120
9	175	2	27	21	51	50	51	51
10	175	5	27	21	82	82	79	81
11	175	2	53	23	63	64	64	64
12	175	5	53	22	99	101	101	100

Table 2b Fine grained sand, static nozzle

Run	Stand off distance (mm)	Flush time (sec)	Pressure (kPa)	Static Depth (mm)	Depth Marker A (mm)	Depth Marker B (mm)	Depth Marker C (mm)	Depth Average (mm)
1	85	2	27	14	52	55	55	54
2	85	5	27	18	86	87	87	87
3	130	2	27	25	35	39	35	36
4	130	5	27	26	57	59	61	59
5	175	2	27	23	23	23	23	23
6	175	5	27	40	40	40	40	40

Table 2c : Coarse grained sand, static nozzle

Run	Stand off distance (mm)	Flush time (sec)	Pressure (kPa)	Static Depth (mm)	Depth Marker A (mm)	Depth Marker B (mm)	Depth Marker C (mm)	Depth Average (mm)
1	85	2	27	11	82	76	76	78
2	85	5	27	12	95	98	100	98
3	85	2	53	10	116	120	120	119
4	85	5	53	11	120	120	120	120
5	130	2	27	13	73	68	73	71
6	130	5	27	15	88	89	88	88
7	130	2	53	13	104	106	105	105
8	130	5	53	13	120	120	120	120
9	175	2	27	15	56	58	59	58
10	175	5	27	17	72	74	74	73
11	175	2	53	21	94	94	93	94
12	175	5	53	40	119	119	119	119

Results from these tests have been analysed using Rajaratnam et al. (1995). For initial analysis, the relationship was simplified to:

$$d_s \propto q^s \times H^y / d^z$$

The depth of scour was calculated as the sum of the stand off distance and the depth of fluidisation. The constants s was taken as 0.6 with y and z taken as 0.25.

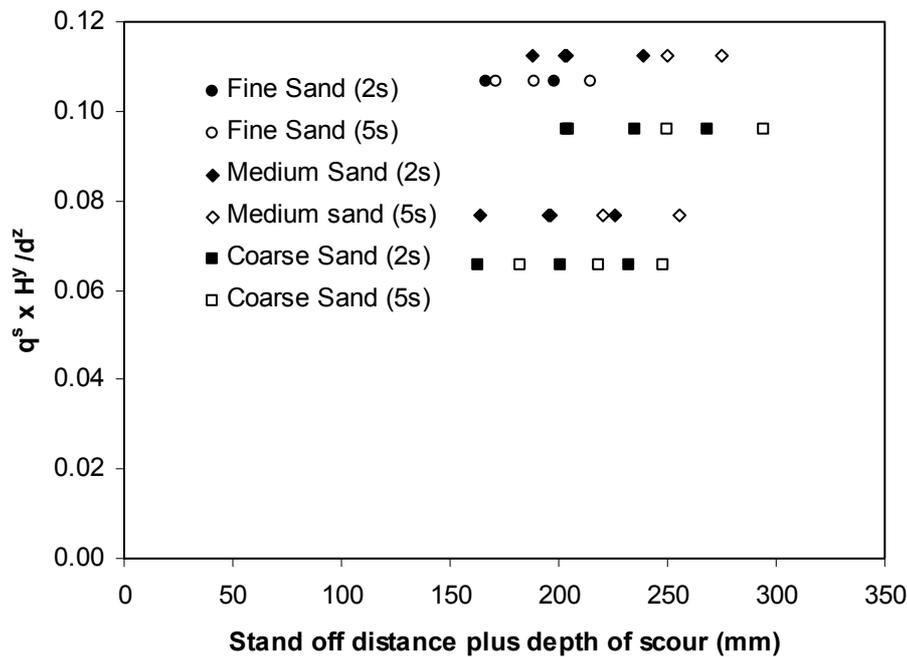


Figure 10 Jetting test results normalised using Rajaratnam et al. (1995)

Inspection of the results presented in Figure 10 shows a trend between total depth of scour (stand off distance plus depth of scour) and the pressure / volume / particle size relationship proposed by Rajaratnam et al. (1995). Features which are interesting to note are that the coarse sand is scoured to a greater depth than fine sands, however the difference in depth is relatively small given the change in grading of the sand. Also the difference in depth of scour between 2 seconds exposure to the water jet, and 5 seconds exposure is relatively small component of the total depth, typically a further 20 or 30mm of a total depth of 200mm.

The data suggest that the most significant factor in the depth of scour is the time of exposure, with a very short time period being required to excavate a scour hole. Unfortunately it was not possible to repeatedly control the exposure of the sand to the jet to less than 2 seconds, however it appear that 90% of the scour is generated within this time period. Pressure has an effect, but within the range investigated, this effect is relatively small.

To further investigate the effect of a moving nozzle, a series of tests were performed with the nozzle moving over the surface of the sand. The manner in which the tests were performed was similar to the above tests with the nozzle positioned statically. Depth markers were again used, being placed along the line of traverse of the nozzle. Tests were performed on medium grained and coarse grained sand.

Results are presented as Tables 3a and 3b overleaf.

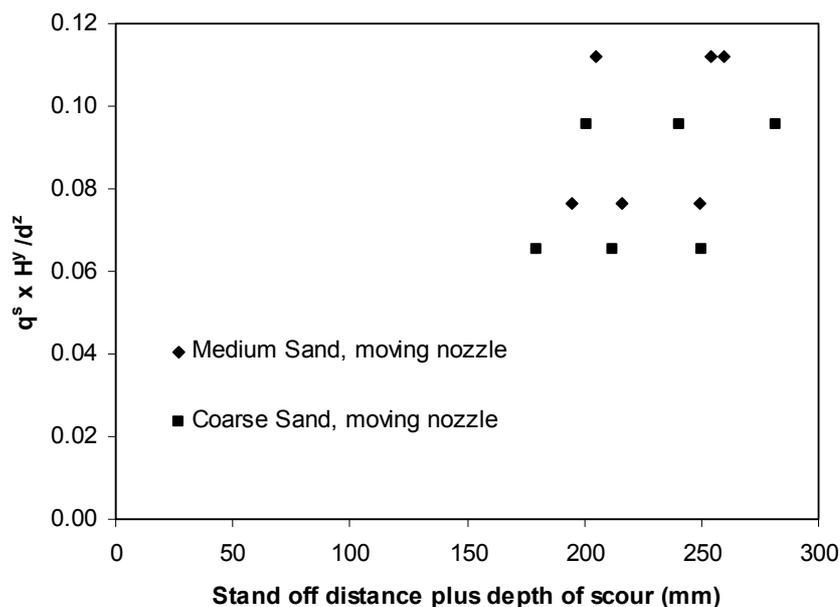
Table 3a Test results for moving nozzle in medium grained sand

Run	Stand off distance (mm)	Pressure (kPa)	Nozzle speed (m/s)	Depth of scour (mm)
1	85	27	0.014	110
2	85	53	0.014	120
3	130	27	0.014	86
4	130	53	0.014	124
5	175	27	0.014	74
6	175	53	0.014	85

Table 3b : Test results for moving nozzle in coarse grained sand

Run	Stand off distance (mm)	Pressure (kPa)	Nozzle speed (m/s)	Depth of scour (mm)
1	85	27	0.014	95
2	85	53	0.014	116
3	130	27	0.014	82
4	130	53	0.014	111
5	175	27	0.014	75
6	175	53	0.014	107

The results of these tests have been plotted, normalised by Rajaratnam et al. (1995) relationship and are presented as Figure 10. Inspection of these results, and comparison with Figure 10, indicates that the depth of scour is greater than for a static nozzle, despite shorter exposure time at any given location. A possible explanation for this is that the nozzle is clearing a path and a more efficient flow pattern develops as sand is cleared from the track of the nozzle. This is in contrast to the turbulence, and loss of energy associated with the small diameter scour hole created by a static nozzle, as return water washes past the jet water.

**Figure 11 Normalised depth of scour for a nozzle moving at 0.014m/second.**

Discussion

Field Trials

The results of the testing indicate that the hydraulic dredge is fluidising the seabed to a depth typically in the range 30 to 100mm. A noticeable feature of the two sets of trials at Jesolo and San Pietro was the variability in the depth of fluidisation. This may be due to a variety of factors including changes in the nature of the seabed sediments, changes in the design and efficiency of the dredge, and the location of the depth-measuring sensor. This last factor may be significant with the Jesolo trials being conducted with the arm at the limit if the dredge nozzles (see Figure 1 above). The location during the Jesolo trials, which was found necessary to avoid the loss of the trailing arm of the sensor, may have resulted in the arm not being in the area of full fluidisation.

Analysis of the data from the two sets of trials has been performed using Rajaratnam et al. (1995) relationship to normalise the results in terms of nozzle flow rate and pressure, and sediment size. The results of the analysis are presented as Figure 12 below.

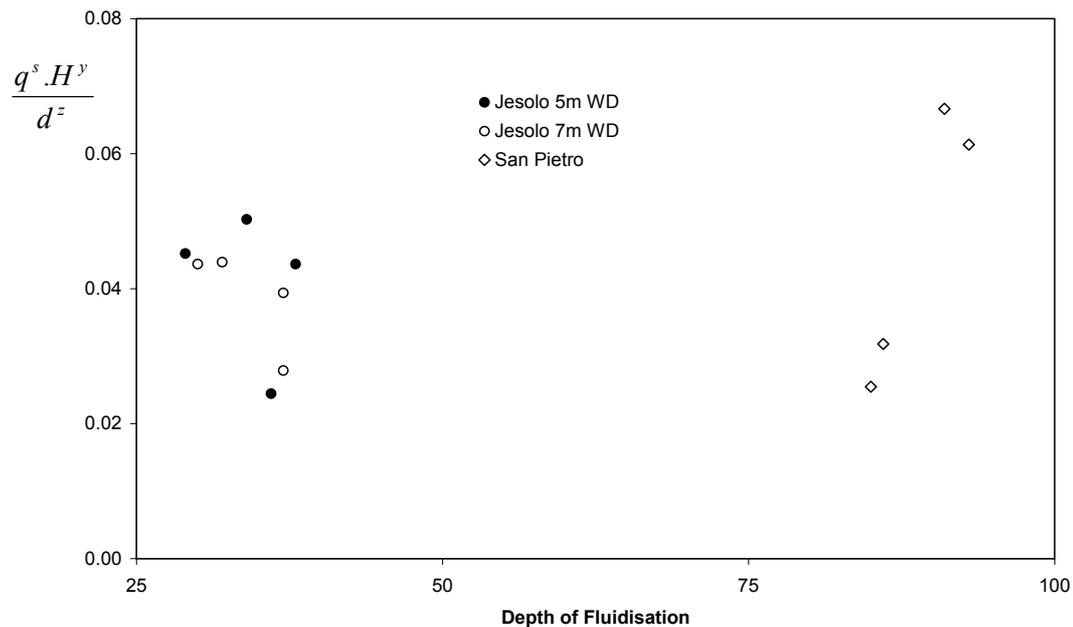


Figure 12 Depth of fluidisation normalised by flow rate and nozzle pressure

The results presented in Figure 12 show some scatter. However no allowance has been made for speed of the dredge over the seabed in these analyses. This has been done in Figure 13, by dividing the nozzle and sediment normalising factor by speed. The results of this analysis are shown as Figure 13.

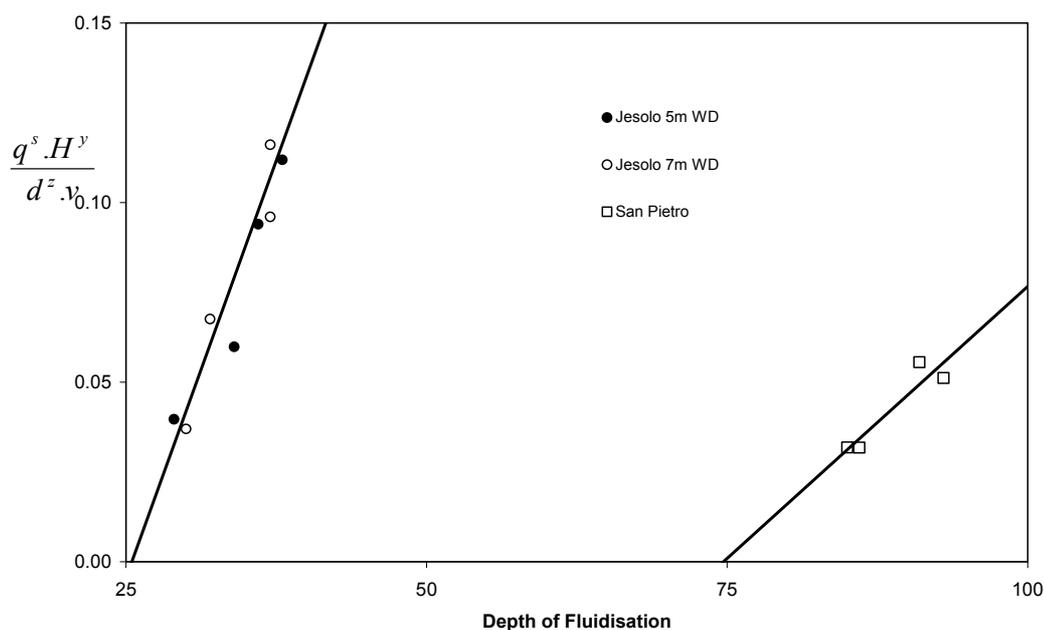


Figure 13 Depth of fluidisation vs normalised nozzle and sediment parameters

Inspection of Figure 13 indicates that while a good correlation is possible between data from one site, a wide variability is present in the two data sets. Other than the explanation given above for possible detail variables in the dredge and location of the measuring arm, it is not clear why such a large difference should be present. The most probable reason is due to the location of the trailing arm depth of fluidisation sensor. It probable that greatest reliance should be placed on the San Pietro results since these were taken with the depth sensor in the middle of the area of fluidisation.

For the yield to be maximised the clams must be washed into the dredge. The depth of fluidisation being achieved by the dredge is greater than the depth of the blade. This implies that the clams must be lifted into suspension by jet water. To lift the clams into the cage, the volume of water flowing must be of a greater velocity than the free fall velocity of the clam itself. The typical water volume issuing from the dredge nozzles is $5.5 \times 10^{-4} \text{ m}^3/\text{s}$. Assuming each nozzle has an approximately square area of influence based on the nozzle spacing, then the upward water velocity may be calculated to be about 0.6m/s. The settlement velocity of a clam may be approximated using Gibb's equation for the settlement velocity of submerged particles (Gibbs 1971). For a particle of approximately 10mm diameter, with a submerged density of 5 kN/m^3 , the settlement velocity can be calculated to be 0.6m/s. Velocities greater than the settlement velocity are required if the clam is to be lifted into suspension in the water column. This calculation suggests that the upwards velocity of the water the minimum required to lift the clams into suspension.

Failure to reliably lift the clams into suspension implies that the dredge scrapes them off the seabed. The fine grained nature of the sand suggests that the sand particles will be lifted into suspension more readily than the clams and hence the clams will collect at the top of the undisturbed sand. It is possible that some clams get washed to the bottom of the depth of fluidsation, and hence do not get picked up by the dredge.

The dredge lifts sediment into suspension to a depth of between 30mm and 90mm below the seabed. For a 2.5m dredge this equates to lifting between 9m³ and 27m³ of sediment into suspension for each 100m of dredging performed. Assuming a typical work rate of 20 tows of 500m length, then between 900m³ and 2700m³ (1600 to 4800 tonnes) of sediment are disturbed by one fishing boat in a single day.

Laboratory Testing

A programme of laboratory trials has been conducted. The results indicate that a relatively deep scour hole was possible at pressures significantly less than those used by commercial dredges. This hole formed very rapidly and it was not possible to determine a meaningful relationship for the effect of time of exposure to the rate of scour.

The results did indicate that it would be possible for a dredge, operating either slowly, or while stationary at the beginning or end of a dredge, to excavate a relatively large scour hole. Based on a correlation of pressure and nozzle size, such scour holes could be in excess of 300mm deep.

Summary and Conclusions

The results of the testing indicate the dredge to be disturbing a significant volume of sand in the seabed. In general the dredge is achieving its design objective of fluidising the sand to a depth equivalent to, or greater than the preset depth of the scraper. Particular factors which should be considered are optimising the depth of disturbance and reducing the effect that operator controlled variations in the system can make to the depth of fluidisation. The design should also maximise the catch and reduce any potential sources of injury to the clams. Specific findings of this work are as follows:

1. A good correlation between the dredge data at a single site has been obtained, based on theoretical relationships. However a high degree of difference is present between the sites at which trials were performed. It is probable that this is due to a combination of factors including variations in the dredge, sediment grading and location of measurement. Further studies should consider placing multiple sensors for the assessment of the depth of scouring.
2. Given the natural variability in the seabed topography, it is thought unlikely that it will be possible to reduce the depth of the scraper below the current pre-set depth of 30mm. Indeed this may be considered quite a low depth should any bedforms (ripples) be present.
3. The volume of water being ejected at normal operating pressures is estimated to be marginal in its capacity to lift the clams into the water column for collection by the dredge. Failure to adequately lift the clams suggests that they could be damaged by the scraper blade on the dredge or pass under the dredge without being caught. In practice, some clams being left on the seabed may be advantageous as it reduces the efficiency of the dredge perhaps leaving clams for breeding.
4. The work done to date forms a useful basis for optimisation of some of the performance variables of the turbo soffianti hydraulic dredge. Further work with marine biologists to identify the natural depth at which the clams live would help optimisation of these variables.
5. The trials performed to date have concentrated on the physical effects of the dredging on the seabed. It would be useful to compare the results with an assessment of clam mortality and damage to assess speed and pressure, which provide optimum operating conditions.
6. Increasing the depth of fluidisation is unlikely to be desirable both due to the increased level of disturbance to the seabed and the possibility that the dredge will pass over clams sinking to the base of the fluidised seabed. The trials and the laboratory tests have both indicated that relatively deep scour is possible.
7. Research work by others has indicated that use of an angled nozzle on the dredge would reduce the influence of pressure and mean grain size of the sand on the depth of fluidisation thus reducing the variation in the optimum operating parameters at different sites. The washing action of an inclined nozzle may also result in the clams being transported more efficiently into the cage of the dredge.

Based on results of laboratory tests (Figure 6), it would appear that a nozzle angle of 45° , could be close to an optimum angle.

FIGURES

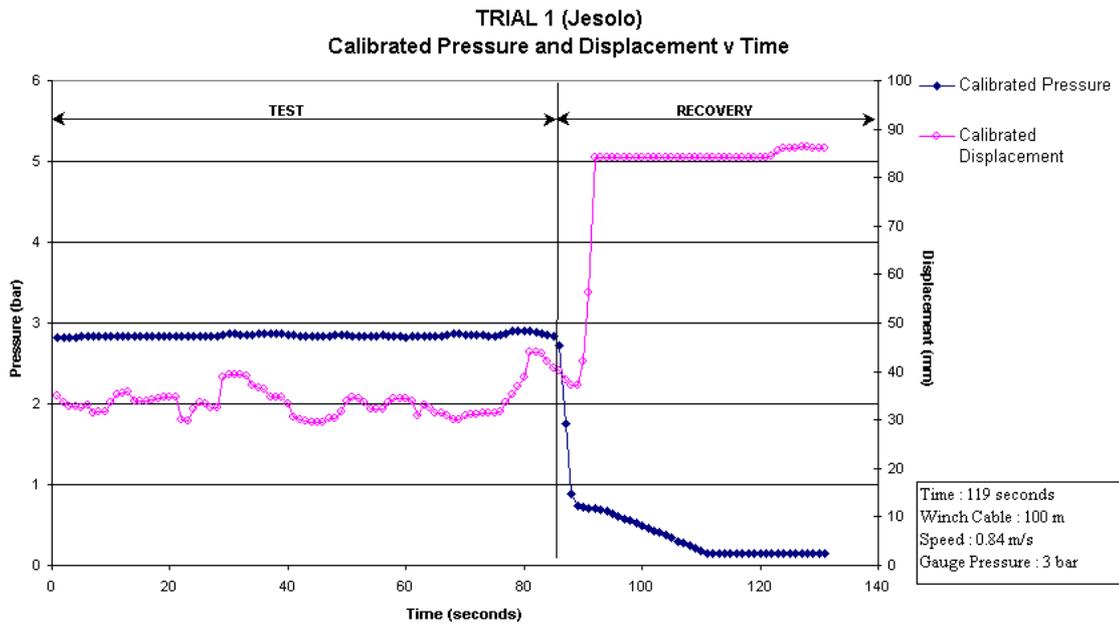


Figure 14

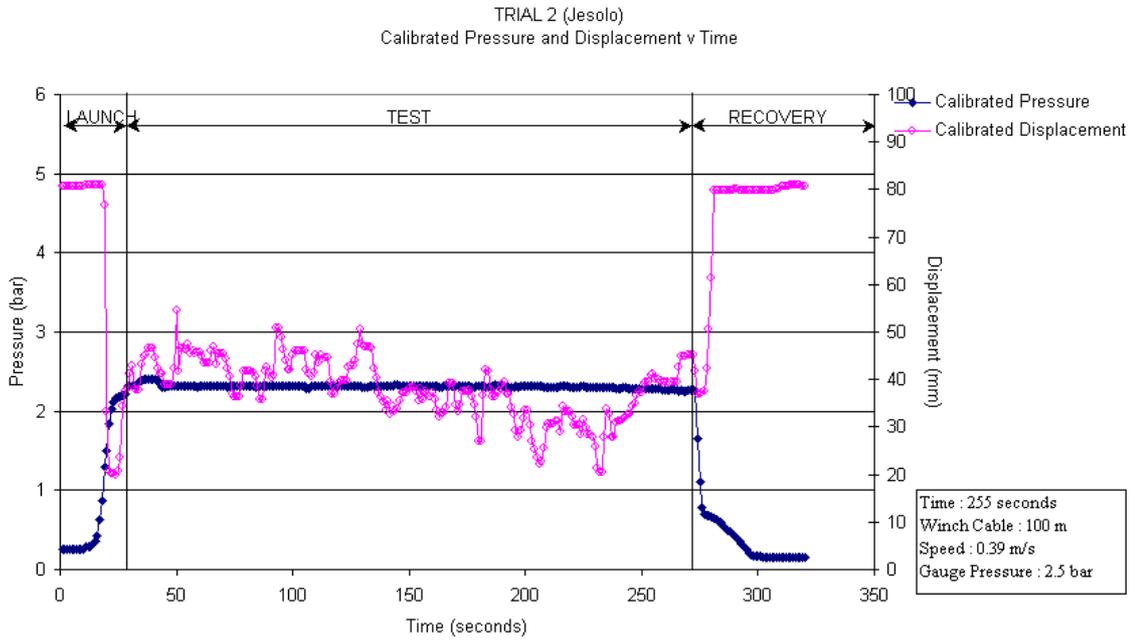


Figure 15

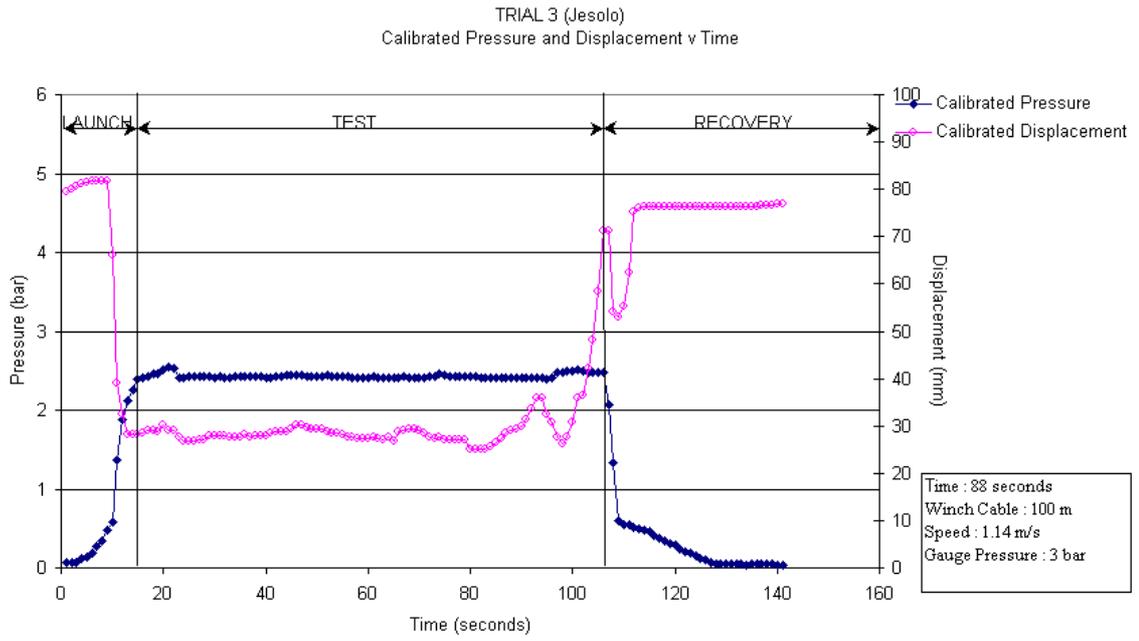


Figure 16

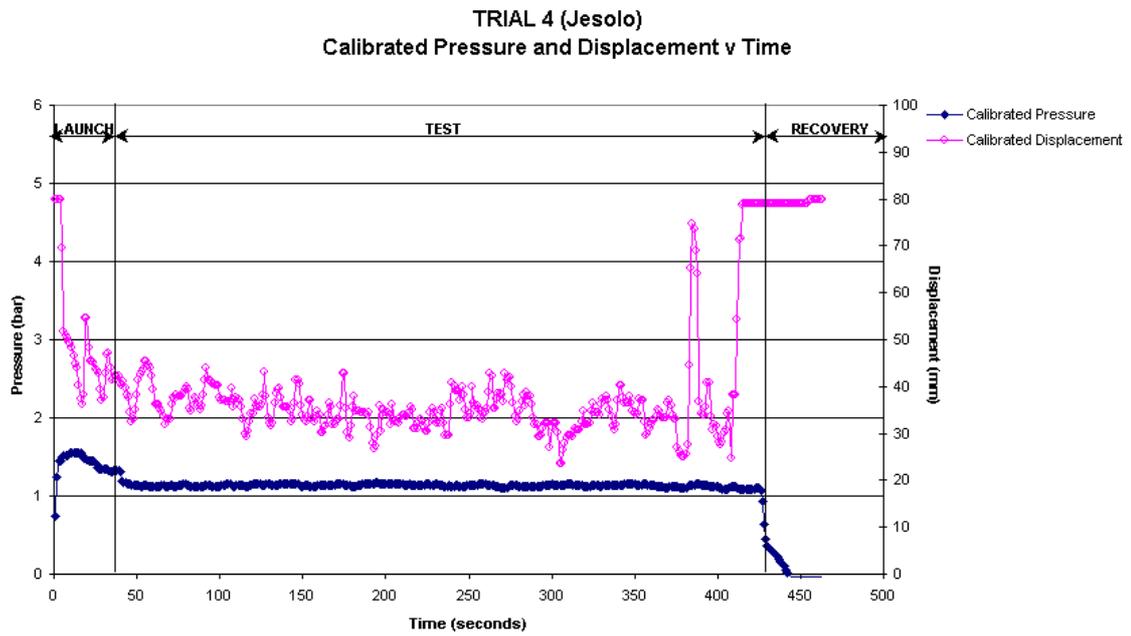


Figure 17

TRIAL 5 (Jesolo)
 Calibrated Pressure and Displacement v Time

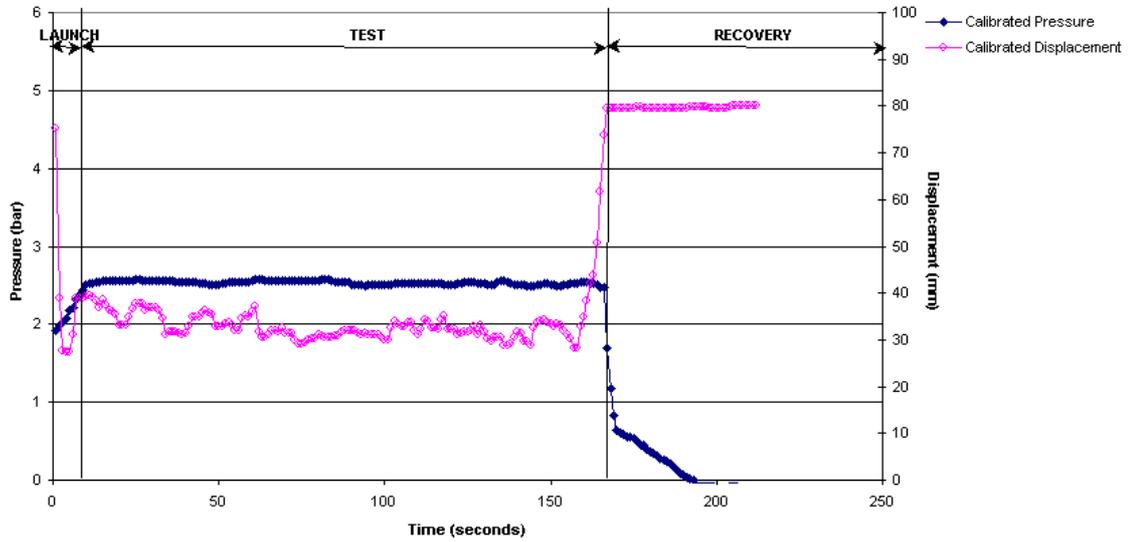


Figure 18

TRIAL 6 (Jesolo)
 Calibrated Pressure and Displacement v Time

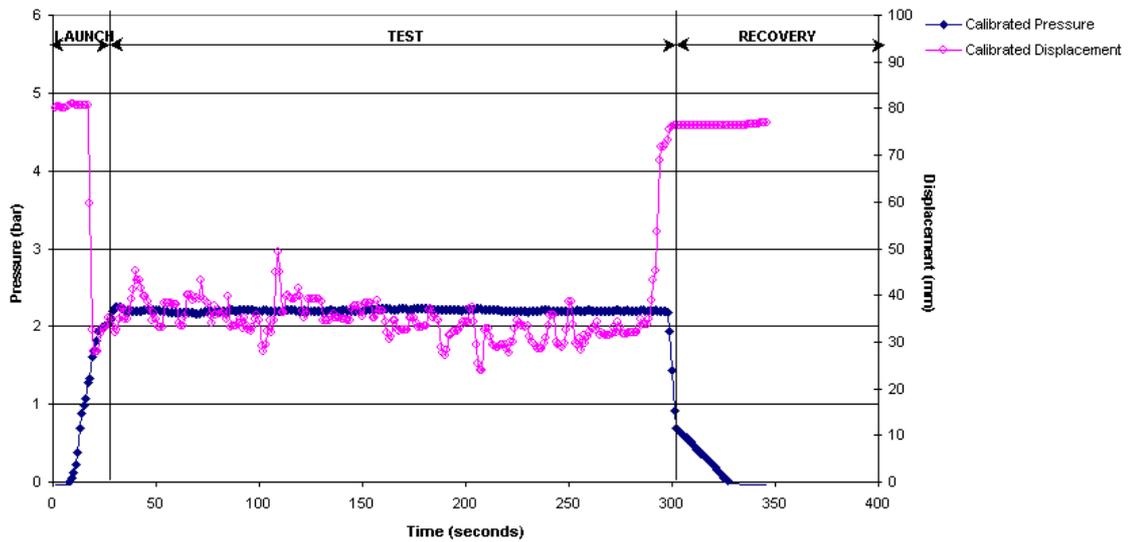


Figure 19

TRIAL 7 (Jesolo)
 Calibrated Pressure and Displacement v Time

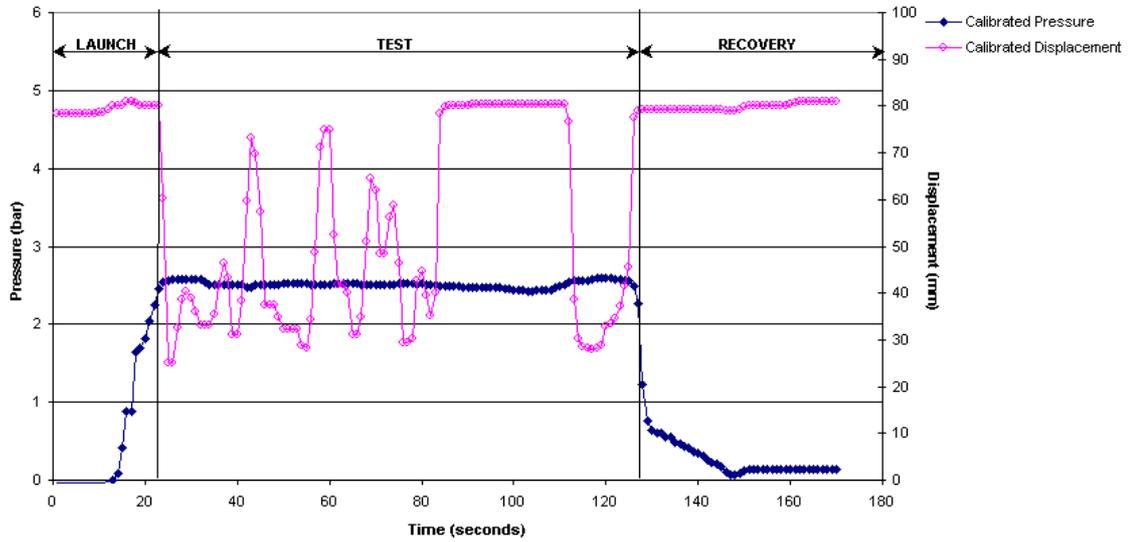


Figure 20

TRIAL 8 (Jesolo)
 Calibrated Pressure and Displacement v Time

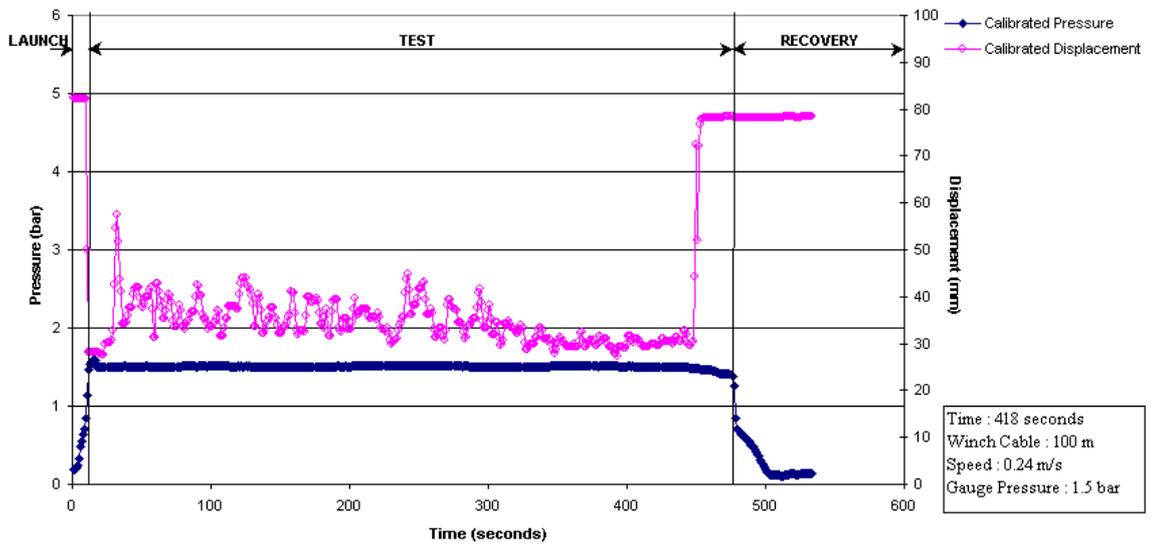


Figure 21

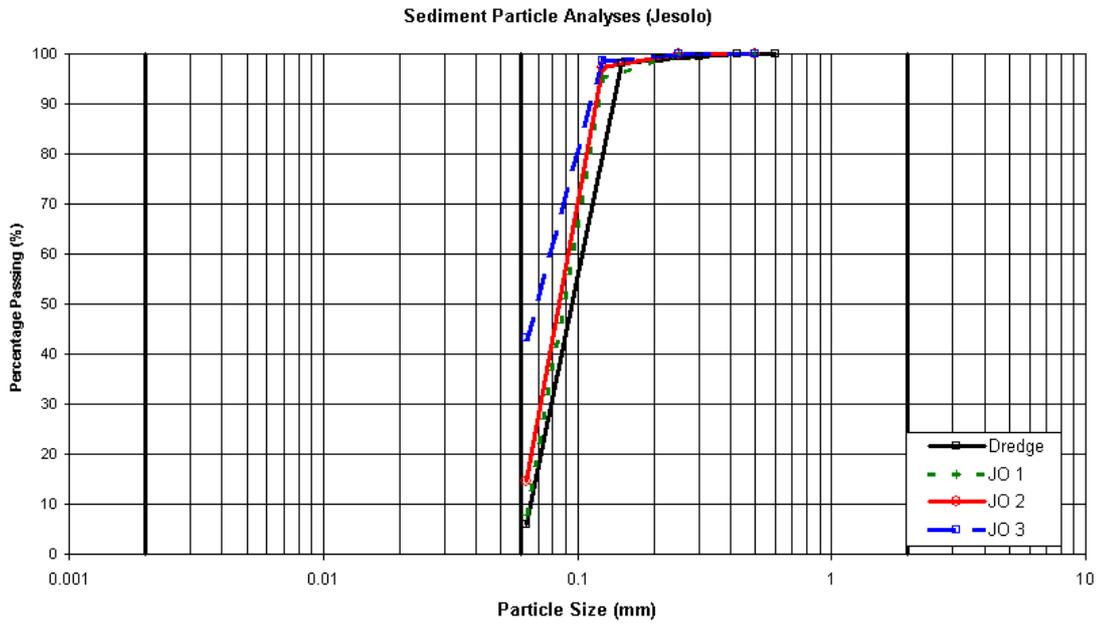


Figure 22

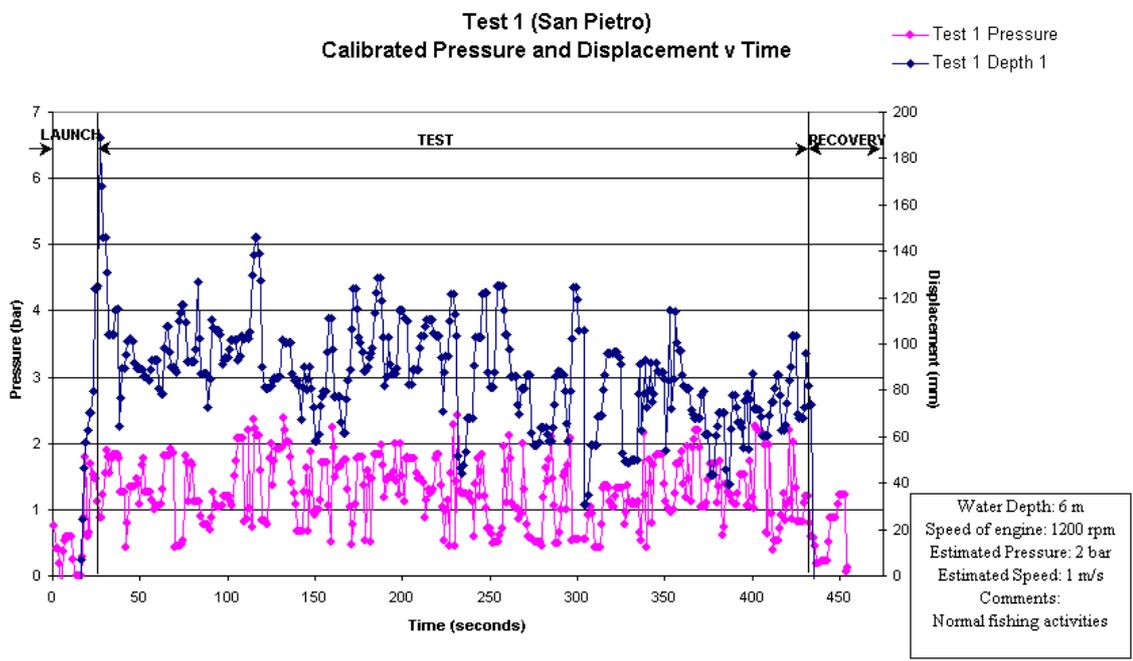


Figure 23

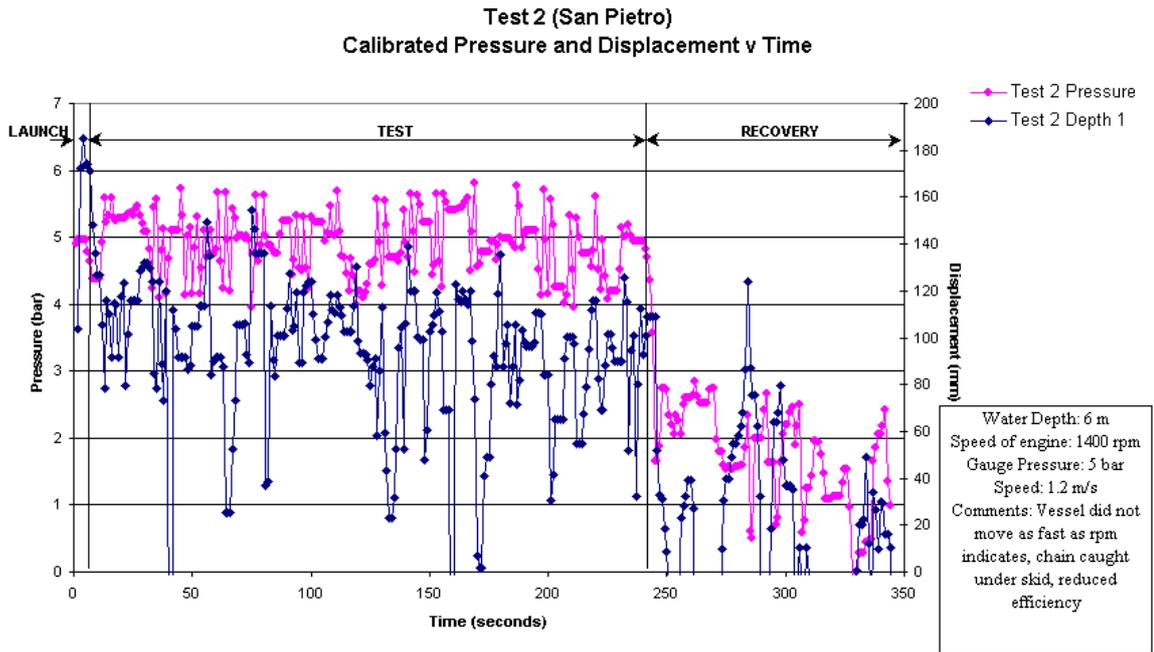


Figure 24

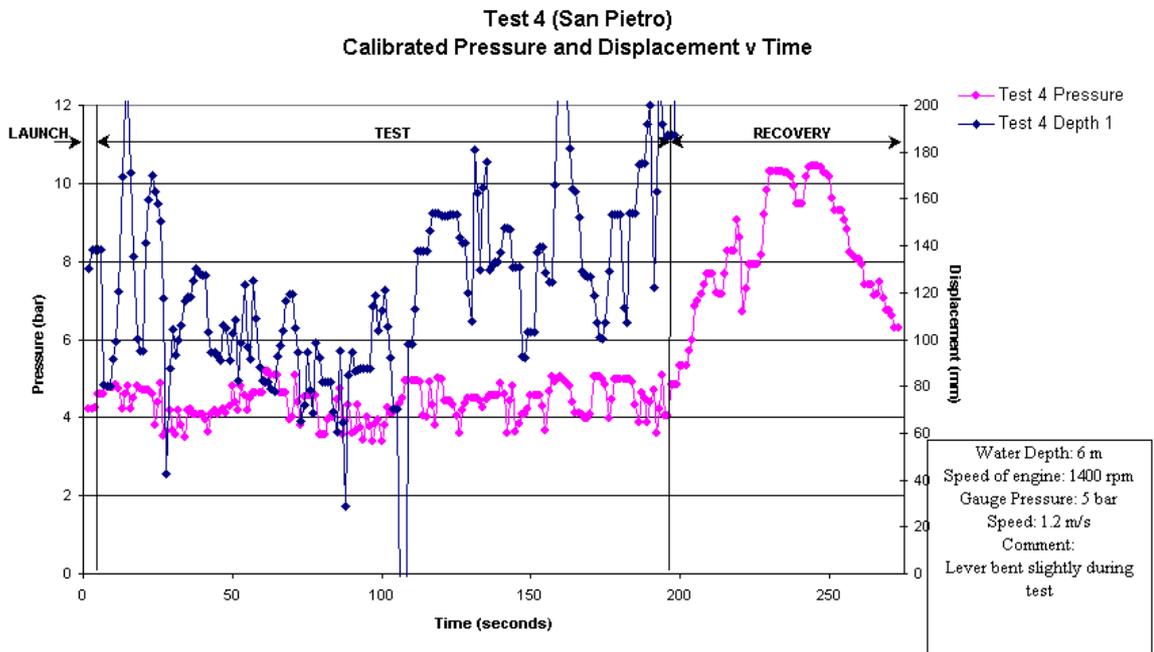


Figure 25

3.1.1.5 SUMMARY STATEMENT

The novel instrumentation developed and used in this study was successful in measuring some key parameters of the environmental effects of dredging. For scallop dredging these included acceleration and rotation on the scallops and tooth penetration into, and rake angle of the teeth to, the seabed. Although there was substantial variation in these parameters, there was some consistency. In particular the angle of the teeth was angled forwards in the sediments at approximately 125° which was opposite to the theoretical models developed for ploughing where the tines are angled at $\leq 90^\circ$ to the seabed. This precluded the use of these models to study the mechanical effects on the sediments directly, but they did indicate that if the sprung teeth were at $< 90^\circ$ to the seabed instability would be more likely. Video observations enabled the elucidation of a possible mechanism by which the scallops could be captured by the teeth, with the scallops being rotated out of the sediments by the downward force on the scallop by the tooth. They were then held on the forward face of the teeth until being dislodged into the bag or lost from the dredge.

The observations of types of damage and the lack of correlation between the outputs of the instrumented scallop (in terms of acceleration and rotation) in the dredge bag and damage levels of captured organisms suggests that most damage is done during the initial encounter with the scallop dredge teeth. Damage levels appear to be consistent from year to year on particular grounds; thus they may be substrate related. Video observations suggest that the dredge can undergo periods of instability resulting in bouncing behaviour perhaps mediated by changes in substrate or speed. This suggests that the spatial heterogeneity of the seabed may be related to the damage levels. Areas where dredges are encountering different or unsuitable substrates could more likely to promote dredge instability and increase damage rates. Possible mitigating measures include improved damping of the tooth springs to counter resonating effects or relating seabed texture to dredge behaviour and hence enabling Skippers to avoid instability by changes in spring tension or speed.

The instrumented scallop was successfully used to show that the simulator produced similar levels of acceleration and rotation to that found in the dredge bag. This enabled the simulator to be used to investigate physiological effects of dredging.

Instrumentation and modelling of the effects of turbo soffianti dredging targeting *Chamellea gallina* in the Adriatic produced consistent relationships between dredging speed, nozzle pressure, and fluidisation depths which were consistent with empirical models developed for seabed trenching using water jets. This approach has the potential for the design of minimal impact dredging systems designed only to impact the sediment sufficiently to harvest the bivalves.

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3.1.2. Chemical effects

Introduction

The use of mobile fishing gears by commercial fishing fleets unquestionably produces a variety of direct and indirect changes on the benthic ecosystem (Newell 1998; Riemann and Hoffmann 1991). These impacts will depend on many factors such as marine sediment characteristics, type of dredging, fishing operations, depth, tidal strength, currents and period of the year (de Groot 1984; Prena 1996). Changes in the biology, sedimentology, biochemistry and benthic environment are a result of this activity (Mayer et al. 1991). The microphytobenthos is overturned, being exported from the fishing ground (De Jonge 1987); the nutrients are rapidly released to the water column changing the chemical and biological stability of sediment and biogeochemical processes along vertical profiles of sediment are disrupted (Fanning 1982). Redox-dependent sediment phosphate release has been observed in several continental margin environments (Aller 1980; Ingall 1994). Retention and release of phosphorus from sediments have been a privileged subject of research due to the importance of this element in the production and distribution of plankton in lakes, estuaries and coastal systems (Nixon 1981; Froelich 1982; Benitez-Nelson 2000).

While fishing, the bivalve dredges digs a furrow along its towpath provoking a sediment movement that occurs mainly through slumping of the sides of the furrow. Subsequently, the furrow is in-filled by fine particles transported by tidal currents acting as a sediment-trap (Newell 1998). Simultaneously with the seabed disturbance, sediment clouds of large dimensions are produced into the water column and generally settles out within minutes to hours, as shown by underwater video images (Mayer et al. 1991; Gaspar 1996b). The horizontal dispersal of this material will be accentuated if currents are present.

To understand the dynamic of marine sediment over continental shelves, many studies have been focused upon sediment re-suspension by the action of winds and turbulent tidal currents (De Jonge 1995) and by the influence of marine animals (Smith 1987). The mechanism of re-suspension of sediment by the action of fishing gear has been largely overlooked (Churchill 1989). Thus there are a lack of studies concerning geochemical changes resulting from this activity.

The aim of the present study was to evaluate the biogeochemical changes caused by clam dredging and the subsequent recovery of the seabed at different depths. Nutrient changes in pore water and near bottom water and concomitantly biological and chemical changes in the upper sediment layers were observed before, during and after dredging.

Objectives

The seabed disturbance by fishing activity imply changes in physical, chemical and biological stability of sediment. With this study we assessed the impact of dredging activity on the seabed at different depths. Two dredge types were studied; Portuguese toothed dredges off the south coast of Portugal and Italian Turbo Soffianti (hydraulic dredges) in the Northern Adriatic.

The specific objectives of studies carried out were the following:

1. To characterize the sediment composition (grain size and chemical) of the habitats studied.
2. To characterize the seasonal pattern of hydrology and chemistry of the water column during turbo soffianti dredging in the Northern Adriatic.
3. To evaluate the changes of physical parameters in the water column during dredging.
4. To estimate the grain size and chemical changes before and after dredging.
5. To evaluate the changes of chlorophyll *a*, phaeopigments, organic carbon and nitrogen in the superficial layer of sediment (ridge and furrow).
6. To evaluate the changes of nutrients in pore water of the upper sediment layers in relation to the 'ridge and furrow' formation observed after dredging.
7. To evaluate oxidation process in the upper sediment layers through the phosphate retention into iron oxides during dredging activity.
8. To evaluate the benthic fluxes of nutrients before and after sediment turnover.

In this Section we report the results of three sets of experiments, two carried out off the S. Coast of Portugal ([Sections 3.1.2.1 & 2](#)) examining the effect of Portuguese toothed dredges and one ([Section 3.1.2.3](#)) examining the effects of Turbo soffianti dredges used in the Northern Adriatic.

The studies differed in their emphasis as follows:

[Section 3.1.2.1](#) covers Objectives 1,3,5,6,7 and 8 but majors on Objective 7

[Section 3.1.2.2](#) covers Objectives 1,3,5,6 and 8 but majors on Objective 8.

[Section 3.1.2.3](#) covers Objectives 1,2,3,4,5,6 and 8.

3.1.2.1 THE IMPACT OF DREDGING OPERATIONS IN COASTAL WATERS VILAMOURA- ARMONA; SOUTH OF PORTUGAL: PHYSICAL AND CHEMICAL DISTURBANCE AND SUBSEQUENT RECOVERY ON THE SEABED.

Methodologies

Study area and bivalve fishery

The Algarve coast, located at the south of Portugal extends by 100 Km in length. According to Moita 1986) sediments in shallow areas, from coastal line to 25 meters depth, range from coarse sand (500 μm) to fine sand (125 μm). Tides are semi-diurnal and the mean amplitude between tides is 2 m, with a maximum of 3.5 and a minimum of 0.3 (Melo 1989). The drift currents tend to run along the shore, to 30 m or more. The current speed may reach under extreme conditions 0.5 m s^{-1} (Moita 1986).

The bivalve fishery is a very important activity along the south coast of Portugal. The dredge fleet directs its fishing effort towards four species: *Spisula solida*, *Donax trunculus*, *Ensis siliqua* and *Pharus legumen*. On this part of the Portuguese coast, these species are distributed along a narrow strip of shallow waters (1 to 15 meters depth) and on sandy bottoms. Taking into consideration the distribution of both commercial bivalve species and fishing effort two stations (VL and AR) were selected (Fig.1). Station VL (37°03'55''N, 08°09'00''W; 10 meters depth) was located off Praia da Falésia (*Spisula solida* fishing ground) and st. AR (37°01'50''N, 07°45'00''W; 3 meters depth) was located in the adjacent coast of Armona island (*Donax trunculus* fishing ground).

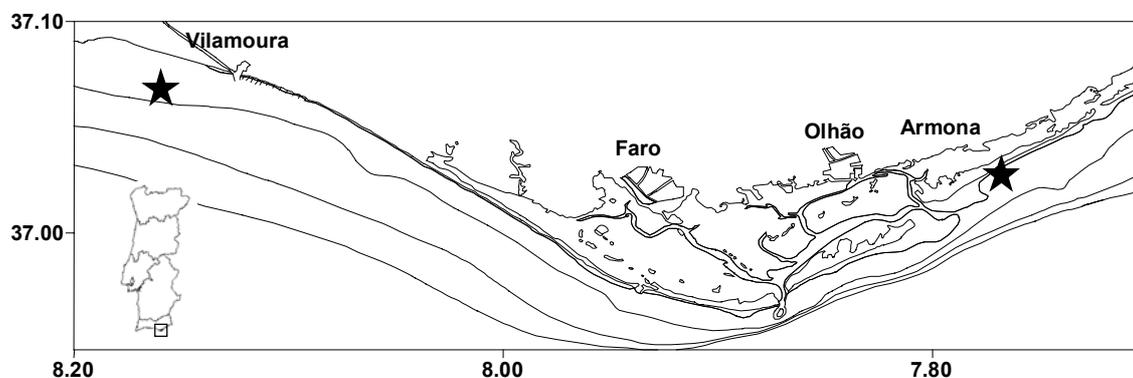


Figure 1 Map of Coastal Zone (South of Portugal) with the locations of the study sites ★(VL and AR).

Field methodology

In July 1999, dredging was carried out (teeth length-20 cm), at 10 meters depth at st. VL. Sediment cores and near bottom water were collected before dredging (control samples), during dredging (0) and 5, 15, 30 and 60 minutes after dredging at the same place. Sediment cores were collected by divers with PVC tubes (5-cm length and 5-cm diameter), filled up with sediment and water avoiding the presence of air inside the core. Posteriorly, these corers were kept in a refrigerated chamber. Each sampling comprised 3 cores collected gently from the furrow and from the ridge

designed along the dredging path. Near bottom water was collected concomitantly with polyethylene vials. In September 1999, another fishing operation was carried out with the same dredge, at 3 meters depth in st. AR. Sediment cores and near bottom water were collected before dredging, during dredging and 5, 20, 60, 120, 240 and 420 minutes after dredging at the same place. A broad time scale was used, aiming to augment the probability of the seabed recovering during the observation. The same sampling methodology used at st. VL was followed at st. AR.

Laboratory methodology

The sediment cores were rapidly analysed. In order to minimise changes resulting from exposure of sediment surfaces to the atmosphere the sediment cores were rapidly sliced into thin layers. Each slice was then centrifuged to extract pore water. Each integrated sample of pore water was composed of water extracted from three cores (0 – 2 cm layer). Samples of pore water and near bottom water were filtered through 0.45 μm Nuclepore membranes and kept at -20°C until the analysis of ammonium (NH_4^+), nitrates (NO_3^-), total organic nitrogen (TON), phosphates (HPO_4^{2-}) and silicates ($\text{Si}(\text{OH})_4$) in an “SKALAR” autoanalyser according to the methods Technicon Industrial Systems (Grasshoff 1976). The detection limit used in the present method was 0.2 μM for ammonium and silicates and 0.05 μM for nitrates and phosphates.

The first two layers of sediment (0 – 2 cm and 2 – 4 cm) were dried to 80°C until a constant weight and ground to a fine powder. The homogeneous mixing from three cores composed each sediment sample. In the powder samples, the total organic carbon, amorphous iron oxides (FeOOH) and the phosphate bound to iron oxides (P-FeOOH) were determined. Total carbon and total inorganic carbon (loss on ignition) in the solid phase was determined using a CNH analyser “NC 2500 CE instruments” with acetanilide as reference material (Byers 1978), the amorphous iron oxides (FeOOH) and the phosphate associated to iron oxides (P-FeOOH) were removed from the sediments by ascorbate extraction following the procedure described in Anschutz et al. 1998). Iron oxides were analysed by flame A.A. and phosphate by UV spectrophotometry. Phosphate was determined colorimetrically as the molybdenum complex (Grasshoff 1976), using a spectrophotometer Hitachi 150-12. The organic matter content was determined in each layer by “loss on ignition” and porosity by weight difference. The chlorophyll *a* and phaeopigments were extracted with acetone (90%) from the upper sediment layer and determined by spectrophotometry according to Lorenzen 1967).

Laboratory experiment

To understand the field results concerning phosphorus sorption a microcosm experiment was undertaken. Sandy sediment collected by divers was placed inside three transparent plexiglass chambers ($15 \times 15 \times 15 \text{ cm}^3$) and covered with seawater. Each chamber was filled with 1000 g of sediment and 1.5 L of seawater. During 24 h the sediment was maintained stable and a constant aeration was applied to maintain saturation levels of oxygen. After this period the sediment was strongly stirred. Samples of near bottom water, pore water and superficial layer of sediment were collected within the following time scale: (before stirring), during stirring and 10, 30, 60, 180, 330, 1340, 1760 and 1780 minutes after stirring. Phosphates (HPO_4^{2-}) were determined in near bottom water and pore water. Phosphorus sorbed into iron oxides

(P-FeOOH) and iron oxides (FeOOH) were determined in the superficial layer of sediment (0 – 2 cm).

Data analysis

To determine the significance between data obtained in pore water and sediment from the two studied sites, two matched samples (Scherrer 1984) was applied the non-parametric Wilcoxon's test. To obtain a multivariate concept of all information and its variability within the time Principal Component Analysis (PCA) was applied (Legendre and Legendre 1984).

Results

Sediment: biological and chemical changes on seabed

Sediment characteristics

Table 1 shows values of grain-size of particles, carbonate content (CaCO_3), porosity (ϕ), loss on ignition (LOI) and total organic carbon (TOC) of sediments collected at the two studied stations.

Table 2 Values of particles grain-size, carbonate content (CaCO_3), porosity (ϕ), loss on ignition (LOI) and total organic carbon (TOC), of sediments collected at different depths in st. VL and at st. AR.

Station VL											
layers (cm)	grain size (%)							CaCO ₃ (%)	porosity	LOI (%)	COT (%)
	sand				silt+clay						
	> 2mm	1mm	500 μm	250 μm	125 μm	63 μm	<63 μm				
0 - 2	0	6	27	57	5	1	4	11	0.301	0.874	0.21
2 - 4	1	9	29	52	5	0	4	13	0.268	0.969	0.52
4 - 6	-	-	-	-	-	-	-	-	0.302	0.781	0.43
10 - 12	2	8	31	53	4	1	1	9	-	-	-
19 - 21	4	9	30	49	5	1	2	10	-	-	-
Station AR											
layers (cm)	grain size (%)							CaCO ₃ (%)	porosity	LOI (%)	COT (%)
	sand				silt+clay						
	> 2mm	1mm	500 μm	250 μm	125 μm	63 μm	<63 μm				
0 - 2	0	1	13	60	25	0	1	6	0.329	0.662	0.34
2 - 4	0	1	15	60	24	0	0	5	0.290	0.600	0.09
4 - 6	-	-	-	-	-	-	-	-	0.316	0.592	0.06
10 - 12	0	3	23	47	27	0	0	6	-	-	-
19 - 21	0	1	18	52	28	0	1	6	-	-	-

The sediment characteristics from VL station were different from those found in the AR station. Sediment in st. VL contain more fine particles (silt+clay) than the sediments from st. AR. The percentage of CaCO_3 and LOI was also higher in st. VL. The finer particles found in the seabed from the deeper location may indicate that these surfaces are less disturbed by natural processes than shallow location which was more exposed to winds and tidal currents. Sediment profiles at st. VL, show clearly an increase of finer particle (<63 μm), from deeper layers (10 – 20 cm) towards the surface (0 – 4 cm). The organic matter content and the total organic carbon were nearly 2 times higher in the station with more abundant fine particles (VL).

Chlorophyll *a*

Chlorophyll *a* obtained in the ridge increased slightly after dredging at st. VL (Fig. 2). This increase from $1.2 \pm 0.22 \mu\text{g g}^{-1}$ (before dredging) to $1.7 \pm 0.30 \mu\text{g g}^{-1}$ (after dredging), may be related to the accumulation of particles settled after the cloud of re-suspended sediment. The concentrations obtained concomitantly in the deeper layer that became exposed during the tow (furrow) were significantly lower (Wilcoxon's test $p \leq 0.01$) than in the ridge. At st. AR the chlorophyll *a* in the furrow was also significantly lower than that obtained in the ridge (Wilcoxon's test $p \leq 0.01$) where concentrations were maintained approximately constant ($1.5 \mu\text{g g}^{-1}$) over 60 minutes (Fig. 2). Seven hours after dredging chlorophyll *a* in the furrow remained lower than the control value ($0.7 \pm 0.04 \mu\text{g g}^{-1}$).

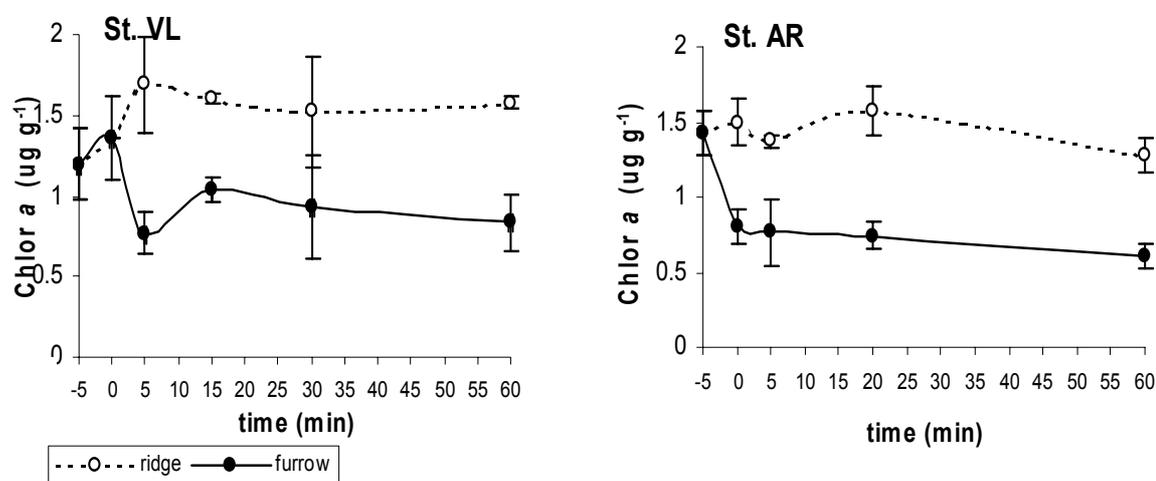


Figure 2 Variations of chlorophyll *a* ($\mu\text{g g}^{-1}$) obtained in the ridge and furrow before dredging (-5); during dredging (0) and after dredging at st. VL and st. AR.

Ascorbate-extractable iron (FeOOH) vs ascorbate-extractable phosphorus (P-FeOOH)

During and after dredging the concentration of phosphorus removed from the sediments by ascorbate extraction ($4\text{-}5 \mu\text{mol g}^{-1}$) exceeds the concentration of iron oxides formed in the studied sediments ($1.5\text{-}2 \mu\text{mol g}^{-1}$). The molar ratio (P:Fe) was calculated over the period of 60 minutes and its variability in the ridge at st. VL is illustrated in Figure 3. An instantaneous and sharp adsorption of P into iron oxides occurred when oxygen penetrates in the sediment. The same process was observed at st. AR however, the higher values of P:Fe remained for a period (Fig. 3).

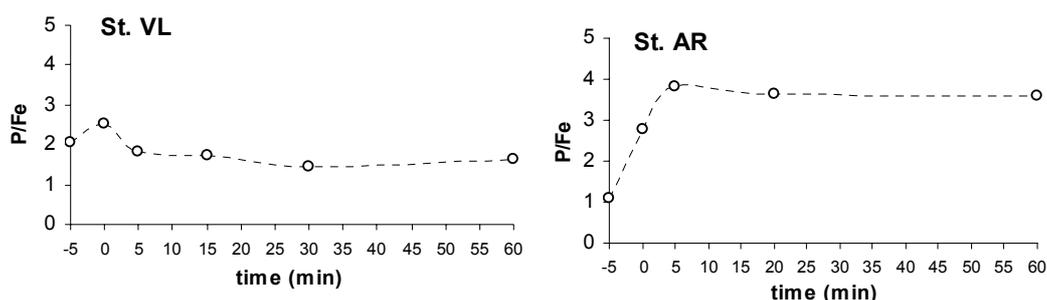


Figure 3 Variation of the ratio P:Fe obtained in the ridge before dredging (-5); during dredging (0) and after dredging at st. VL and at st. AR.

To confirm these findings in the seabed sediment turnover was simulated in the laboratory under the following physical conditions: water temperature 22°C - 25°C and oxygen saturation 80% - 90%. On the basis of the mean concentrations of P-FeOOH and FeOOH obtained in the superficial sediment in the three chambers it the molar ratio P:Fe was calculated. This ratio increased sharply during and after the sediment stirring (Fig. 4) and the recovery tended to start 24 h after perturbation (P:Fe=2.5). The plot of P sorbed into iron oxides vs. HPO_4^{2-} dissolved in pore water showed an inverse linear relationship ($P_{\text{sorbed}} = -0.22 \text{HPO}_4^{2-} + 8.8$; $r^2=0.5$; $n=10$; $p \leq 0.05$). This means that the most part of the phosphorus sorbed into iron oxides came from pore water since a slight increase in phosphorus overlying water was recorded. The HPO_4^{2-} dissolved in near bottom water increased from $5.8 \mu\text{M}$ before stirring to $7.7 \mu\text{M}$ during stirring and 60 minutes later returned to the initial value.

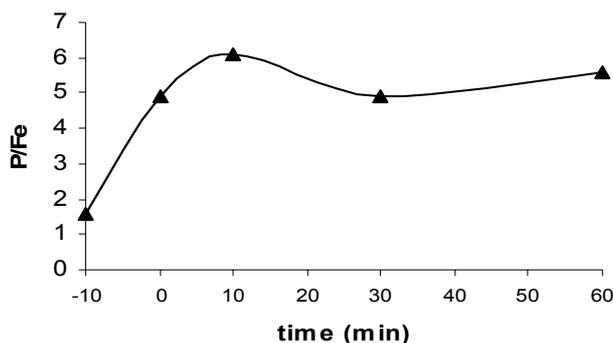


Figure 4 Variation of the ratio P:Fe obtained in the microcosm experiment before stirring (-10); during stirring (0) and after stirring.

Pore water and in near bottom water: chemical changes

In Table 2 are shown the concentrations of NH_4^+ , NO_3^- , TON, HPO_4^{2-} and Si(OH)_4 , obtained in near bottom water before, during and after dredging operation at both stations.

Table 2 Concentrations of NH_4^+ , NO_3^- , TON, HPO_4^{2-} and Si(OH)_4 , obtained in near bottom water at control situation, during dredging (0) and post-dredging at st. VL and st. AR.

station VL					
Time scale (min)	NH_4 (μM)	NO_3 (μM)	TON (μM)	PO_4 (μM)	Si(OH)_4 (μM)
Control	10.18	0.33	1.4	0.09	0.64
0	7.59	0.10	6.9	0.31	1.20
5	13.02	0.10	2.1	0.37	0.15
15	8.82	0.07	4.6	0.20	0.20
60	11.21	0.20	5.6	0.15	0.52
station AR					
Time scale (min)	NH_4 (μM)	NO_3 (μM)	TON (μM)	PO_4 (μM)	Si(OH)_4 (μM)
Control	7.52	0.02	6.4	0.31	0.41
0	7.36	0.02	2.6	0.41	1.40
5	7.52	0.08	5.8	0.61	0.43
20	5.51	0.06	10.0	0.66	0.46
60	8.48	0.07	7.9	0.61	0.57
120	5.43	0.05	6.6	0.41	0.39
240	6.87	0.09	3.3	0.61	0.41
420	6.23	0.06	5.3	0.46	0.99

Nitrogen compounds

In the st. VL ammonium in pore water decreased considerably during the first five minutes after dredging (311 to 124 μM) and concentrations obtained in the ridge and

furrow were very close (Fig. 5). However, the expected increase of ammonium in near bottom water was not clearly observed. Concentrations oscillated between 8 μM and 13 μM over the studied period (Table 2). The intense dilution and tidal currents as expected to occur in the water column may explain these results. Fifteen minutes after dredging the ammonium in ridge pore water reaches a concentration similar to that obtained before dredging, suggesting a rapid recovery of ammonium in pore water. At st. AR, the ammonium in pore water was two times lower than that obtained at st. VL and the values observed both in the ridge and furrow, were maintained similar through out the studied period (Fig 5).

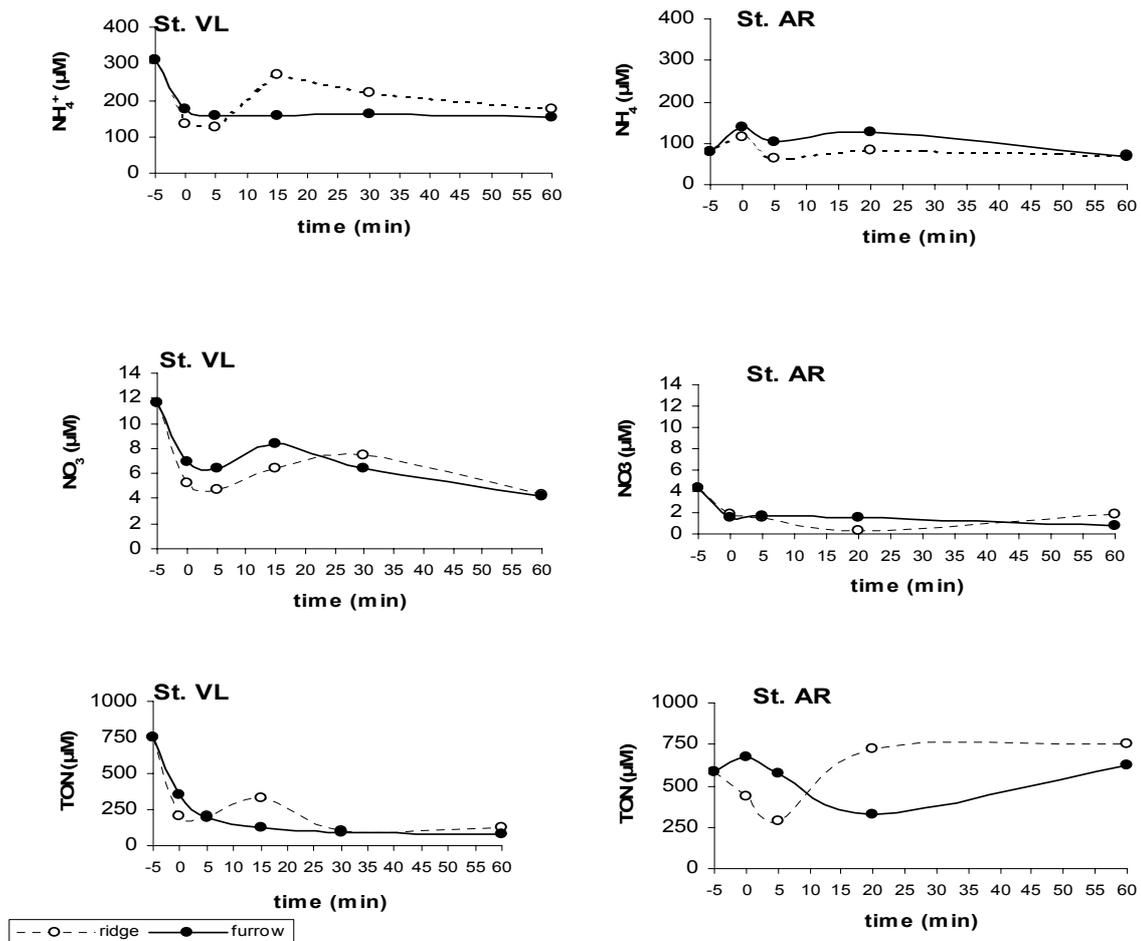


Figure 5 Variation of ammonium, nitrates and total organic nitrogen (μM) obtained in the ridge and furrow before dredging (-5); during dredging (0) and after dredging at st. VL and at st. AR.

Nitrates in pore water decreased during and after dredging in both stations (Fig. 5) and, the concentrations obtained in ridge and furrow were not significantly different (Wilcoxon test). At st. AR, two hours after dredging, nitrates reached the initial value. Nitrates were probably exported to water column during the dredging operation however, the corresponding increase in near bottom water was not detected. The concentrations in near bottom water oscillated within a narrow range (Table 2).

Total organic nitrogen decreased sharply from 750 μM to 200 μM (st. VL) during and after dredging (Fig. 5) and the concentrations in ridge and furrow pore water were not significantly different. The increase of organic nitrogen in near bottom water during dredging from 1.4 μM to 6.9 μM (Table 2) may result from transport sediment-water during sediment turnover. The decrease of organic nitrogen in pore water (from 585 μM to 283 μM) was also observed in st. AR within the first 15 minutes after dredging. Twenty minutes after dredging the recovery may be reached and after seven hours the concentrations were maintained constant (575 μM).

Phosphates

In the ridge of st. VL, phosphates dissolved in pore water decreased during dredging from 4.5 μM to 1.3 μM (Fig. 6). A similar trend was also observed in the furrow. The decrease in pore water corresponded to an increase in near bottom water from 0.09 μM to 0.37 μM (Table .2). Part of phosphate dissolved in pore water is probably exported to the water column. In the ridge of st. AR phosphates decreased from 20 μM to 16 μM during the first 5 minutes after dredging however, the transport of this compound to water column was not observed (Table 2).

Silicates

In the ridge of st. VL, the silicates decreased sharply during and after the first 5 minutes after dredging (from 5.0 μM to 0.5 μM) while, in the furrow, concentrations were maintained approximately constants (Fig. 6) and significantly higher than those obtained in the ridge ($p \leq 0.05$). Sixty minutes after dredging levels of silicates had recovered in the pore water.. At st. AR a similar pattern was observed in the silicate evolution, that is, a sharp decrease in the ridge after dredging while, in the furrow values remained nearly constant within the first hour. Seven hours after dredging the silicates didn't change. During dredging operation, in both stations, a clear increase of silicates in near bottom water was observed ($\approx 60\%$).

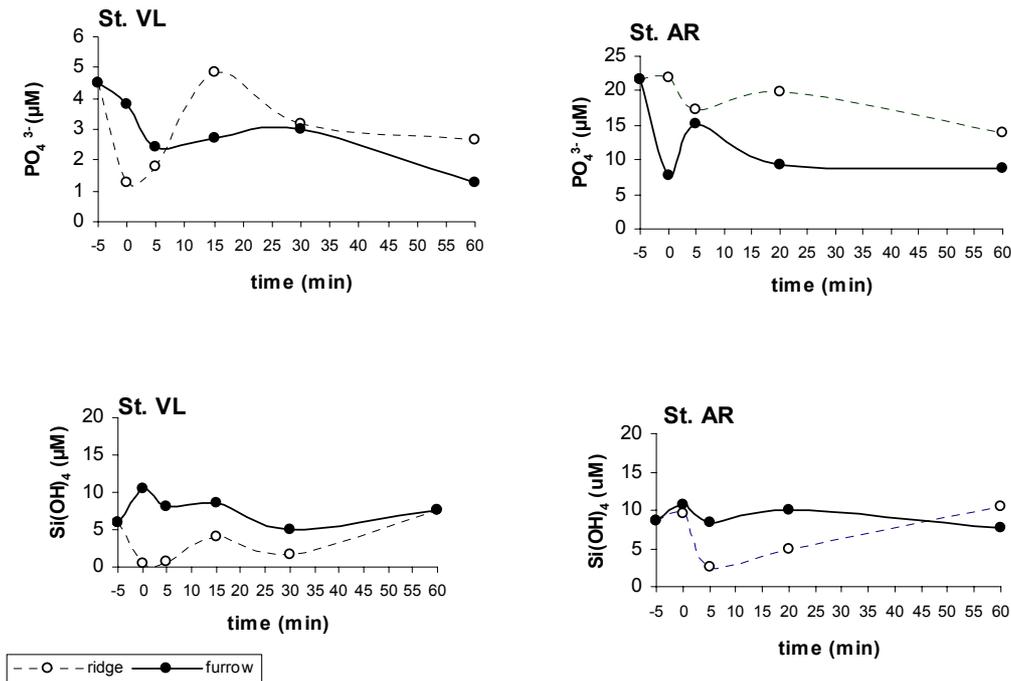


Figure 6 Variation of phosphate and silicate concentrations (μM) obtained in the ridge and furrow before dredging (-5); during dredging (0) and after dredging at st. VL and at st. AR.

Benthic fluxes of nutrients pre-dredging and post-dredging

On the basis of the concentrations of nutrients obtained in pore water and in near bottom water, the nutrient fluxes (F) across the sediment water interface by the Fick's first law of diffusion were calculated (Berner 1980):

$$F = -\phi D_s (C_o - C_p) / (\Delta x)$$

where: ϕ =porosity

D_s =molecular diffusion coefficient

C_o = $[\text{NH}_4^+]$ in near bottom water

C_p = $[\text{NH}_4^+]$ in pore water

Δx = distance across the interface where concentrations were measured (0 – 2 cm)

$D_s = 14 \times 10^{-6}$ to $22 \times 10^{-6} \text{ cm}^2 \cdot \text{s}^{-1}$ for NH_4^+ and NO_3^- ; $D_s = 5 \times 10^{-6}$ to $8 \times 10^{-6} \text{ cm}^2 \cdot \text{s}^{-1}$ for PO_4^{3-} and $D_s = 3 \times 10^{-6} \text{ cm}^2 \cdot \text{s}^{-1}$ for $\text{Si}(\text{OH})_4$ considering temperature range of 10 – 26 °C (Aller, 1980).

The estimated fluxes of ammonium, nitrates, phosphates and silicates in st. VL and st. AR, during the sampling period are presented in Table 3. The decrease of ammonium, nitrates, phosphates and silicates in pore water observed at both stations during and after dredging implies the decrease of benthic fluxes of nutrients during and after sediment turnover.

Some differences between the ridge and furrow were observed. Generally, at st. VL, the benthic fluxes of ammonium, nitrates, phosphates and silicates decreased sharply during sediment turnover. The recovery varies with the nutrient, occurring within minutes to few hours after dredging. At st. AR the fluxes of ammonium and nitrates were notably lower than those obtained at st. VL. The variation of ammonium fluxes over the studied period was not well defined, while, nitrate fluxes decreased clearly (near 50%) during sediment turnover and, seven hours after dredging remain lower than the control value (Tab. 3). In the ridge the silicate fluxes decreased during the first sixty minutes, after which the flux recovery may be reached ($0.3 \text{ nmol cm}^{-2} \text{ d}^{-1}$).

Table 3 Benthic fluxes of ammonium, nitrates, phosphates and silicates ($\text{nmol cm}^{-2} \text{ d}^{-1}$), at control situation, during dredging (0) and post-dredging at st. VL and and st. AR.

Station VL								
	NH_4 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)		NO_3 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)		PO_4 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)		Si(OH)_4 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)	
Time scale (min)	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow
control	70.20	70.20	2.65	2.65	0.25	0.25	0.21	0.21
0	30.01	38.89	1.19	1.61	0.09	0.32	0.01	0.40
5	25.89	34.13	1.09	1.47	0.13	0.19	0.02	0.31
15	61.11	35.12	1.47	1.94	0.42	0.22	0.16	0.33
30	49.10	34.99	1.71	1.47	0.27	0.26	0.06	0.19
60	38.36	32.66	1.90	1.86	0.22	0.10	0.29	0.29

Station AR								
	NH_4 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)		NO_3 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)		PO_4 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)		Si(OH)_4 ($\text{nmol cm}^{-2} \text{ d}^{-1}$)	
Time scale (min)	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow
control	17.03	17.03	0.99	0.99	1.92	1.92	0.32	0.32
0	25.11	31.03	0.43	0.36	1.95	1.56	0.32	0.36
5	12.88	22.54	0.33	0.38	1.50	1.32	0.08	0.31
20	18.31	28.05	0.06	0.35	1.74	0.76	0.17	0.37
60	14.63	13.36	0.41	0.16	0.86	0.55	0.39	0.28
120	12.75	11.91	0.17	0.17	1.25	0.92	0.32	0.31
240	25.29	15.42	0.47	0.59	1.50	0.62	0.44	0.54
420	11.62	12.58	0.15	0.65	1.06	0.69	0.32	0.22

Discussion

During dredging a furrow is formed exposing the underlying sand with a spoil ridge on either side of the depression. In the present study the chemical and biological characteristics observed in the furrow often contrasts with the ones obtained in ridge. The sediment dragged by the teeth of the dredge along the path is overturned accumulating particles in the ridge, mostly microphytobenthos. This may explain the increase of chlorophyll *a* and phaeopigments in the ridge after dredging operation while the lower values of chlorophyll *a* in the furrow are due the lower production of microphytobenthos in deeper sediment layers (Boon 1998).

The sediment disturbance caused by dredging operations implies an oxidation in the upper sediment layers. When an oxidised surface layer is present, substantial amounts

of phosphates can be retained in the sediment through adsorption to iron oxides (Krom 1981; Sundby 1992; Slomp 1998; Anschutz et al. 1998). This adsorption processes generally results in a buffering of pore water HPO_4^{2-} concentrations to low values in the oxidized sediment zone (Froelich 1988; Sundby 1992) thus, the diffusive transport of HPO_4^{2-} to the overlying water is limited. The effective retention of phosphate in the first layers of sediment through adsorption into iron oxides, observed during dredging operations, means that these sediments have a high and instantaneous capacity to adsorb phosphorus when clouds of particles are formed hindering the release of phosphate to the water column. In the microcosm experiment only a small fraction of phosphate was released to water column during sediment perturbation (HPO_4^{2-} in near bottom water increased $2 \times 10^{-3} \mu\text{mol cm}^{-3}$) contrasting with the simultaneous adsorption of phosphate into the upper sediment ($4 \mu\text{mol cm}^{-3}$ dry sed). The increase of phosphorus by volume of sediment was 2×10^3 times higher than its increase by volume of water indicating that the amount of phosphate released to water column is insignificant when compared with the amount of phosphate retained in the sediment.

The sediment is a sink of nutrients (Henriksen 1983; Sundby 1992; Forja 1994) thus, during turnover when the pore water of superficial sediment is renewed, the nutrients are released to water column (Fanning 1982). According to the results found in the present study ammonium, nitrates, organic nitrogen, phosphates and silicates decreased in pore water after sediment turnover probably being exported to the water column. The consequent decrease in benthic fluxes of nutrients during and after turnover indicates that the sediment function as a source of nutrients during turnover. However, while the re-establishment of seabed nutrients is not reached, the sediment has a lower contribution to the nutritive enrichment of water column. The increase in organic nitrogen, phosphates and silicates in near bottom water during dredging, in spite of the intense dilutions in water column, reinforce the evidence that nutrients are exported from the sea bed. Other studies on the ecological consequences of dredging corroborate these findings. Riemann and Hoffmann (1991) demonstrate that silicates had a surprisingly high difference in concentration in the water column of a dredged area compared with a control area and consider that the water column receives organic and inorganic nutrients which may cause significant changes in the nutrient level. Thus, it is expected that phytoplankton primary production may increase during dredging activity if it is controlled by nutrients. However, the retention of phosphate in the upper sediment layers through sorption into iron oxides during and after sediment turnover hinders, for the most part, its release to the water column. If large areas of seabed are frequently disturbed by mobile fishing gears implying nitrogen release to the water column and phosphate retention in the upper sediment layers some changes in N/P stoichiometry may be expected from this fishing activity.

Multivariate analytical methods (PCA) allows an ordination that arrange particulate and dissolved compound along axes according to their both similarity and correlation with the dredging time scale. The scatterplot corresponding to the deeper station-VL (Fig. 7) indicates that the group of nutrients dissolved in pore water (TON , NH_4^+ , NO_3^- , HPO_4^{2-} and $\text{Si}(\text{OH})_4$) reaches the maximum correlation in the control situation (before dredging) meaning that concentrations of these nutrients decrease during and after dredging. The other group of compounds (chlorophyll *a*, phaeop, iron oxides and phosphorus sorbed to iron oxides), behave in an opposite manner to the nutrients dissolved in the pore water, reaching the maximum correlation with the instantaneous

turnover (during dredging). Chlorophyll *a* and phaeopigments coming from the dredge path are accumulated in the overturned sediment and the iron oxides and phosphorus sorbed on to iron oxides are instantaneously formed on the removed layers. The scatterplot corresponding to the shallow st. AR (Fig. 7) shows a higher dispersion of chemical compound on the plan of analysis. This dispersion means that dissolved and particulate compounds do not have a well-defined correlation with the dredging time scale as it was found in st. VL. However, the chlorophyll *a* and phaeopigments are correlated with the instantaneous turnover (during dredging) and nitrates and organic nitrogen are well correlated with the control situation. This analysis shows the differences in dredging impact in studied stations with different depths and sediment characteristics. A possible explanation for the unclear pattern found in the st. AR may be the instability created at the seabed by its exposure to winds, waves and tidal currents that tend rapidly to restore the sediment characteristics in shallow waters as demonstrated by (Krost 1990).

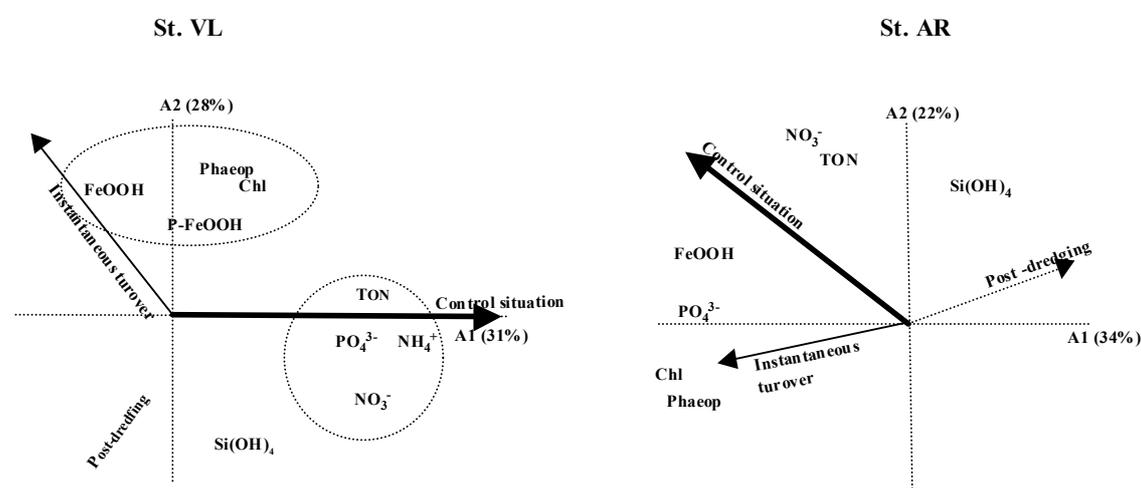


Figure 7 Scattergram (PCA) of chemical parameters (NH_4^+ , NO_3^- , TON, HPO_4^{2-} , Si(OH)_4 , Chlor, Phaeop, FeOOH and P-FeOOH) obtained within the studied time scale (control situation, instantaneous turnover and post-dredging) at st. VL and at st. AR.

3.1.2.2 EFFECTS OF DREDGING OPERATIONS ON CHEMICAL CHANGES IN NEAR BOTTOM WATER AND SEABED (LAGOS- SOUTH COAST OF PORTUGAL).

Methodologies

Study area and mobile gear fisheries

The Algarve coast, located at the south of Portugal, extends by 100 Km in length. According to Moita (1986), sediments in shallow areas, from coastal line to 25 meters depth, range from gross sand (500 μm) to thin sand (125 μm). Tides are semi-diurnal and the mean amplitude between tides is 2 m, with a maximum of 3.9m and a minimum of 0.3m (Melo 1989). The drift currents tend to run along the shore, to 30 m or more. The current speed may reach in extreme conditions 0.5 m s^{-1} (Moita 1986).

The bivalve fishery is a very important activity along the south coast of Portugal. The dredge fleet directs its fishing effort towards four species: *Spisula solida*, *Donax trunculus*, *Ensis siliqua* and *Pharus legumen*. On this part of the Portuguese coast, these species are distributed along a narrow strip of shallow waters (1 to 25 meters depth) and sandy bottoms. Taking into consideration the distribution and the fishing effort, two stations were selected for the present study represented in Fig. 1. The station LG_A located in the adjacent coast (main *Ensis siliqua* fishing ground) and the station LG_B located in the adjacent coast off Lagos under the influence of Alvor lagoon output (main *Donax trunculus* fishing ground), at depths of 12 and 7 meters respectively.

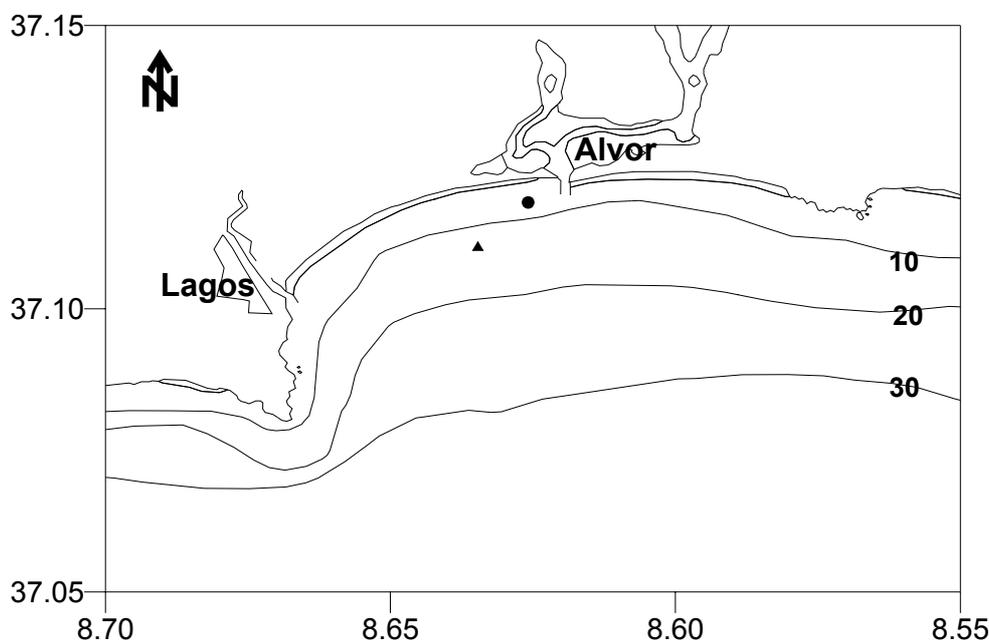


Figure 1 Map of Coastal Zone (South of Portugal) with the locations of the study sites (● - LG_A and ▲ - LG_B).

Field methodology

In July 2000, a mobile fishing gear operation was carried out, with a commercial dredge equipped with a tooth bar with 25 cm teeth, at 12 meters depth in st. LG_A. Sediment cores and near bottom water were simultaneously collected before, during and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at the same site. Sediment cores were collected by divers with PVC tubes (5 cm length and 5 cm diameter), filled up with sediment and water avoiding air presence inside the core and kept in a refrigerated chamber. Each sampling comprised 3 cores carefully collected in the ridge and furrow designed along the dredge towpath. Near bottom water was collected concomitantly with polyethylene vials. In September 2000, it was carried out other fishing operation with the same dredge, at 7 meters depth in st. LG_B. Sediment cores and near bottom water were collected before, during and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging. A broad time scale was used, since may augment the probability of reaching seabed recovery. The same sampling methodology used in st. LG_A was followed in st. LG_B.

A multi-parameter deep-sea probe (YELLOW SPRING-mod. YSI 6820), was used *in situ* to determine dissolved oxygen, pH, temperature, turbidity and salinity. These physical parameters were measured before dredging (control situation), during dredging (instantaneous turnover) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging.

Laboratory methodologies

The sediment cores were rapidly sliced into thin layers in order to minimise changes resulting from exposure of sediment surfaces to the atmosphere. Each sliced surface was not exposed for more than 2 minutes and was centrifuged to extract pore water. Each integrated sample of pore water was composed of water extracted from three cores (0 – 2 cm). The upper two layers of sediment (0 – 2 cm and 2 – 4 cm) were dried at 80°C until a constant weight and ground to a fine powder. The homogeneous mixing of three cores composed each sediment sample. Samples of pore water and near bottom water were filtered through 0.45 µm Nuclepore membranes and kept at –20°C until the analysis of ammonium, nitrites, nitrates, phosphates and silicates using an autoanalyser according to the methods Technicon Industrial systems (Grasshoff 1976). The detection limit was 0.2 µ M.

The powder samples were used to determine the total carbon and nitrogen, inorganic carbon and nitrogen (loss on ignition) in the solid phase using a CNH analyser “NC 2500 CE instruments” with acetanilide as reference material (Byers 1978). The organic matter content was determined by “loss on ignition” and porosity;

($\phi = \frac{V_{porewater}}{v_{we\ sed} + V_{porewater}}$) was determined in each layer.

The chlorophyll *a* and phaeopigments were determined in the first layer of sediment (0 – 2 cm) according to Lorenzen (1967), using a spectrophotometer Hitachi 150-12.

To determine the significance between data obtained in pore water and sediment, a non-parametric Wilcoxon-Man-Whitney test (Scherrer 1984) was applied.

Results

Water column: physical characteristics

The physical parameters: dissolved oxygen (DO); pH; temperature (Temp); turbidity (Turb) and salinity (Sal), measured at surface and bottom, before dredging (control), during dredging (instantaneous turnover) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging are represented in Table 1 (st. LG_A) and in Table 2 (LG_B).

Table 1 Values of depth and dissolved oxygen (DO), pH, temperature (Temp), turbidity (Turb) and salinity (Sal) measured at surface and bottom in st. LG_A before dredging, during dredging and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging.

		st. LG _A					
		Depth (m)	Temp (°C)	OD (mgL ⁻¹)	pH	Turb (NTU)	Sal (ppt)
before dredging		0.6	15.8	7.18	8.04	2.4	36.9
		12.3	15.4	7.37	8.02	4.0	36.9
during dredging		0.5	15.6	7.03	8.05	3.1	36.7
		12.5	15.1	6.01	8.04	13.9	36.1
post-dredging	15 min	0.5	15.7	6.09	8.04	1.7	36.8
		12.5	15.1	4.92	8.02	2.7	36.8
	30 min	0.5	15.6	7.05	8.05	1.9	36.8
		12.5	15.1	5.98	8.05	7.4	36.8
	60 min	0.5	15.5	7.07	8.03	1.5	36.8
		12.7	15.1	6.22	8.01	7.5	36.8
	2 h	0.5	15.5	6.99	8.03	2.3	36.8
		12.7	15.0	5.85	8.02	1.8	36.8
	4 h	0.5	15.4	7.26	8.04	2.2	36.9
		12.4	15.2	5.79	8.01	1.5	36.8
	6 h	0.5	15.4	7.11	8.03	2.5	36.9
		10.0	15.7	5.35	8.00	1.8	36.8
	12 h	0.5	15.5	7.36	8.04	1.4	36.9
		11.9	15.4	5.73	8.07	1.6	36.8
	24 h	0.6	15.2	6.94	8.03	1.3	36.8
		11.7	15.0	6.09	7.83	1.3	36.8

Values of temperature, dissolved oxygen and pH decreased in the near bottom water while, the inverse was recorded for the turbidity in both stations (LG_A and LG_B). This variation down the water column was as expected in coastal waters not subject to remarkable turbulence.

The effect of the dredging operation was reflected on the values of dissolved oxygen, pH and turbidity. Dissolved oxygen in near bottom water (st. LG_A) decreased during sediment turnover and remained constant after dredging. At st. LG_B, oxygen variation was not observed. The pH increased slightly during sediment turnover in both stations and reached recovery a few minutes after dredging. This suggests that the release of CO₂ through the upper sediment is not important. The turbidity in near bottom water increased 4 times during sediment turnover and rapidly (first minutes post-dredging) reached the initial value. Values of salinity were maintained

approximately constant from surface to bottom of the water column and within the areas affected by dredging varying between 36.7 and 36.9%.

Table 2 Values of depth and dissolved oxygen (DO), pH, temperature (Temp), turbidity (Turb) and salinity (Sal) measured at surface and bottom at st. LG_B before dredging, during dredging and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging.

		st. LG _B					
		Depth(m)	Temp (°C)	OD (mgL ⁻¹)	pH	Turb (NTU)	Sal (ppt)
	before dredging	0.4	15.3	6.68	8.05	2.8	36.9
		11.7	15.0	5.36	8.02	3.2	36.8
	during dredging	0.5	15.3	6.01	8.06	2.0	36.9
		11.1	15.0	5.76	8.04	12.0	36.8
post-dredging	15 min	0.5	15.3	5.75	8.05	2.3	36.9
		12.5	15.0	5.59	8.03	5.4	36.8
	30 min	0.5	15.3	6.48	8.05	1.6	36.9
		12.4	15.0	5.88	8.03	4.7	36.7
	60 min	0.4	15.3	6.50	8.04	1.9	36.9
		12.6	14.9	5.27	8.02	2.4	36.8
	2 h	0.4	15.3	6.47	8.05	1.5	36.9
		13.0	14.9	5.03	8.04	2.7	36.8
	4 h	0.4	15.6	6.63	7.98	2.0	36.8
		12.0	15.0	5.07	7.99	2.7	36.8
	6 h	0.5	16.1	6.87	7.89	2.3	36.9
		11.6	15.1	5.65	7.91	2.4	36.8
	12 h	0.2	16.1	7.32	8.02	2.1	36.8
		11.7	15.1	5.30	8.01	3.6	35.8
24 h	0.5	15.6	7.29	7.97	1.4	36.9	
	11.7	15.2	6.83	7.61	2.0	36.8	

Sediment characteristics

Table 3 shows the values of porosity (ϕ), loss on ignition (LOI) and total organic carbon (TOC), of the upper two layers of sediment (0 – 2 cm) and (2 – 4 cm) collected at st. LG_A and at st. LG_B. Sediment at st. LG_B had porosity and total organic carbon slightly higher than that found at st. LG_A. This may result from the influence of outflow of lagoon water (Alvor) in the area of the station LG_B (Fig.1).

Table 3 Values of porosity (ϕ), loss on ignition (LOI) and total organic carbon (TOC), of sediments collected at st. LG_A and st. LG_B.

Sed. layers	Porosity		Total organic matter (%)		Total organic carbon (%)	
	LG _A	LG _B	LG _A	LG _B	LG _A	LG _B
(0 - 2 cm)	0.33± 0.028	0.36± 0.011	1.36± 0.124	1.27± 0.129	0.28± 0.154	0.580± 0.299
(2 - 4 cm)	0.31± 0.021	0.39± 0.022	1.39± 0.112	1.20± 0.111	0.15± 0.087	0.403± 0.122

Chlorophyll a and phaeopigments in the seabed

During sediment turnover it was observed that there was a decrease of chlorophyll *a* in the ridge and furrow at both stations (Fig. 2). This suggests that part of the microphytobenthos may be dispersed into the surrounding area when the cloud of particles is formed. However, immediately after dredging chlorophyll *a* obtained from the ridge samples was significantly higher (Wilcoxon's test $p \leq 0.05$) than that found in the furrow for both stations. Concentrations in deeper layers that became exposed when the dredge teeth penetrated into sediment were lower than those obtained in the top layer, which follows the normal pattern of microphytobenthos production in profiles of marine sediments (Boon 1998). The recovery of microphytobenthos on the seabed seems to be reached nearly 24 hours after dredging when chlorophyll levels became very similar in the ridge and furrow.

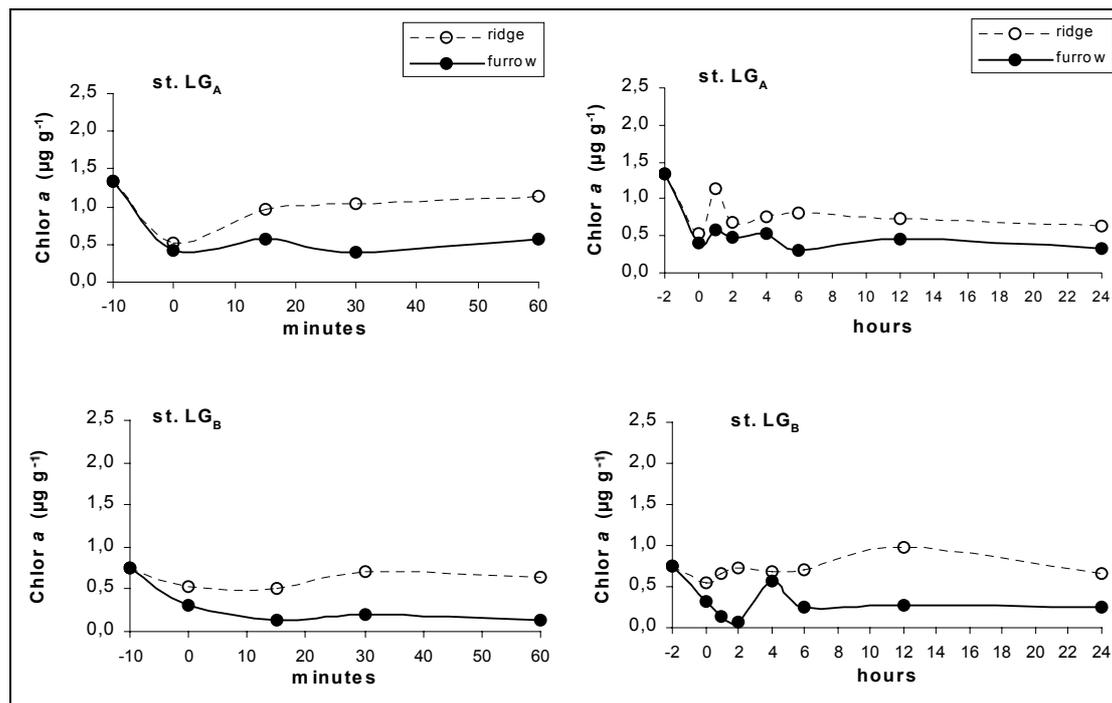


Figure 2 Variation of chlorophyll *a* ($\mu\text{g g}^{-1}$) in the ridge and furrow before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

The variation of phaeopigments levels over the studied time scale was similar to that obtained for chlorophyll in *a* (Fig. 3). At st. LG_A phaeopigments in the ridge were significantly higher than in the furrow (Wilcoxon's test $p \leq 0.05$) while, at st. LG_B the concentrations in the ridge and furrow were similar.

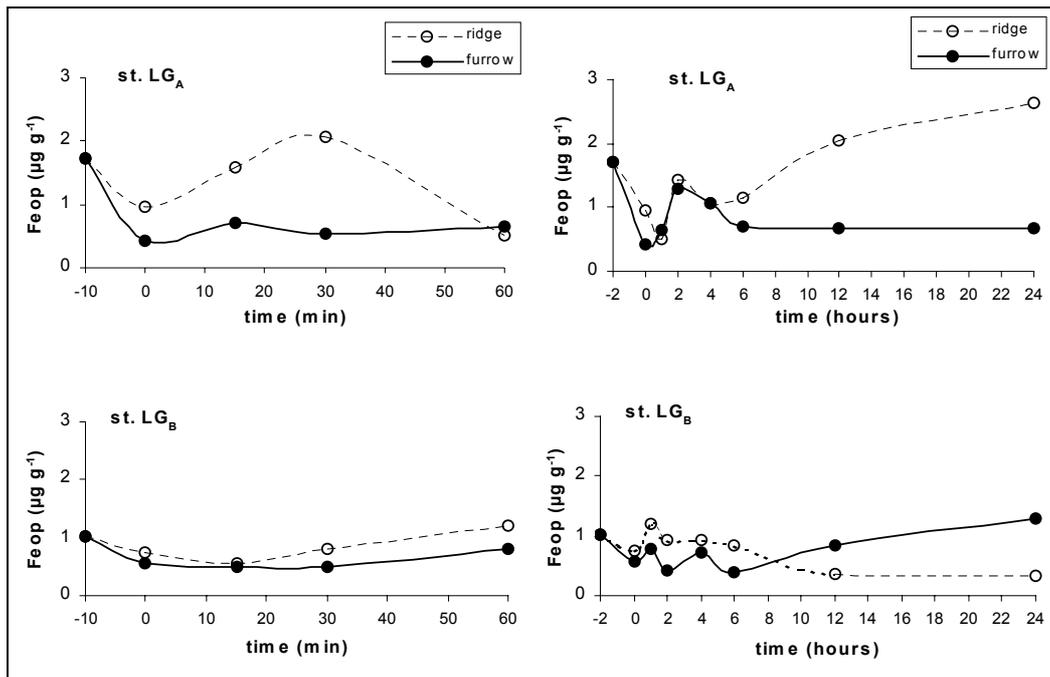


Figure 3 Variation of phaeopigments ($\mu\text{g g}^{-1}$) in the ridge and furrow before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

Pore water and near bottom water: chemical change***Ammonium***

The ammonium variation in the ridge pore water (st. LG_A) was not very clear (Fig. 4). However, concentrations were generally higher than the others found in furrow, over the period of 60 minutes. The recovery was generally reached 24 hours after dredging. In st. LG_B, during dredging the ammonium decreased sharply in the furrow (from 114.5 to 28.5 μM) and decreased slightly in the ridge (from 114.5 to 83.8 μM). Concentrations of ammonium were maintained higher in the ridge and may reach the recovery in pore water, 24 hours after dredging.

Figure 5 illustrates the ammonium variation in near bottom water over the time scale of minutes and hours. At st. LG_A, ammonium varied within a narrow range during the 24 hours of observation (from 0.57 to 1.19 μM). Oppositely, at st. LG_B, ammonium increased sharply (from 0.55 to 2.34 μM) during the dredging operation. The decrease of ammonium in pore water, in this station, during sediment turnover and its concomitant increase in near bottom water may indicate the transport of sediment-water due to the renewal of pore water during fishing activity.

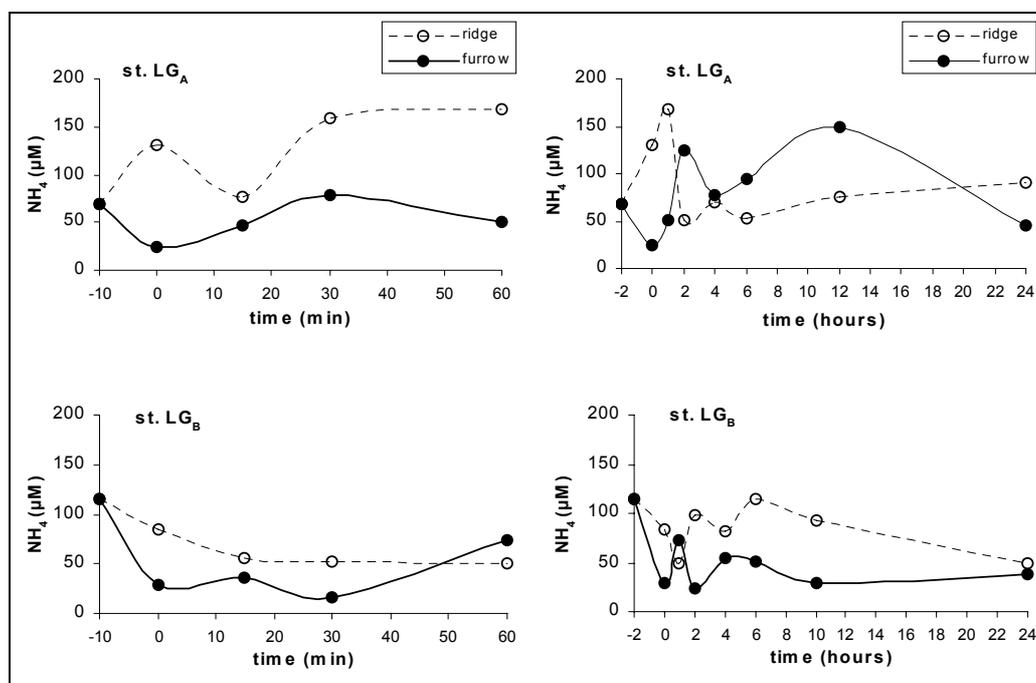


Figure 4 Variation of ammonium (μM) in the ridge and furrow pore water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

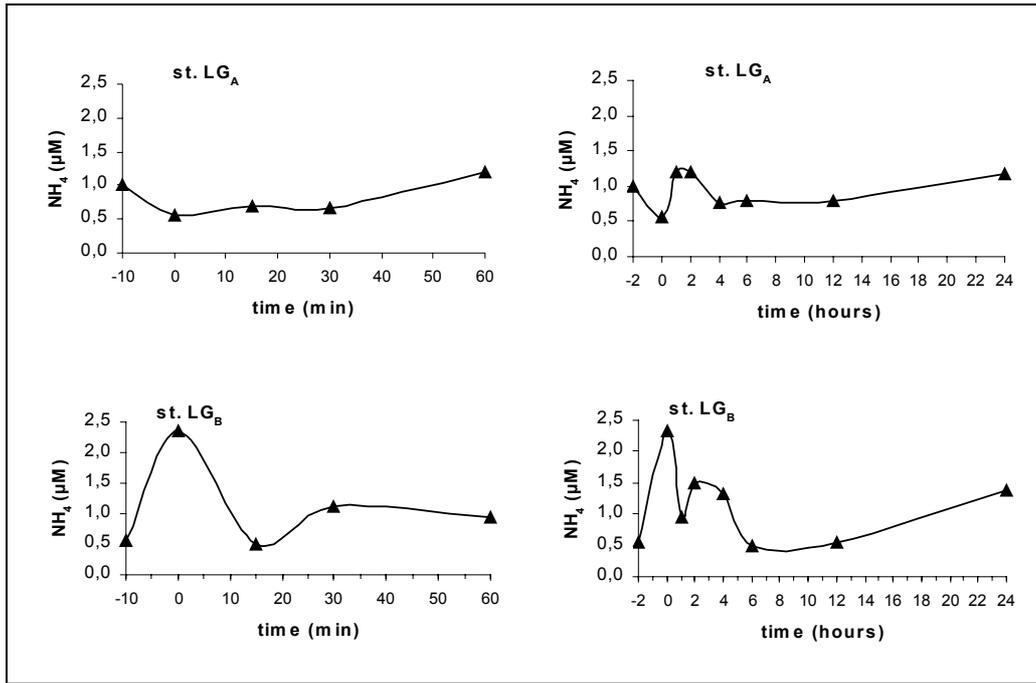


Figure 5 Variation of ammonium (μM) in near bottom water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

Nitrates

The decrease of nitrates during dredging occurred in the ridge and furrow but it was clearer at st. LG_A (Fig. 6). In this station nitrates remained higher in the ridge 24 hours after dredging while at st. LG_B the recovery of nitrates in pore water was reached 10 hours after dredging.

The expected increase of nitrate in near bottom water during dredging (Fig. 7) was not observed probably due the intense dilution in seawater. At st. LG_B nitrates concentration were $<0.2 \mu\text{M}$ and remained constant during the period of observation (24 hours).

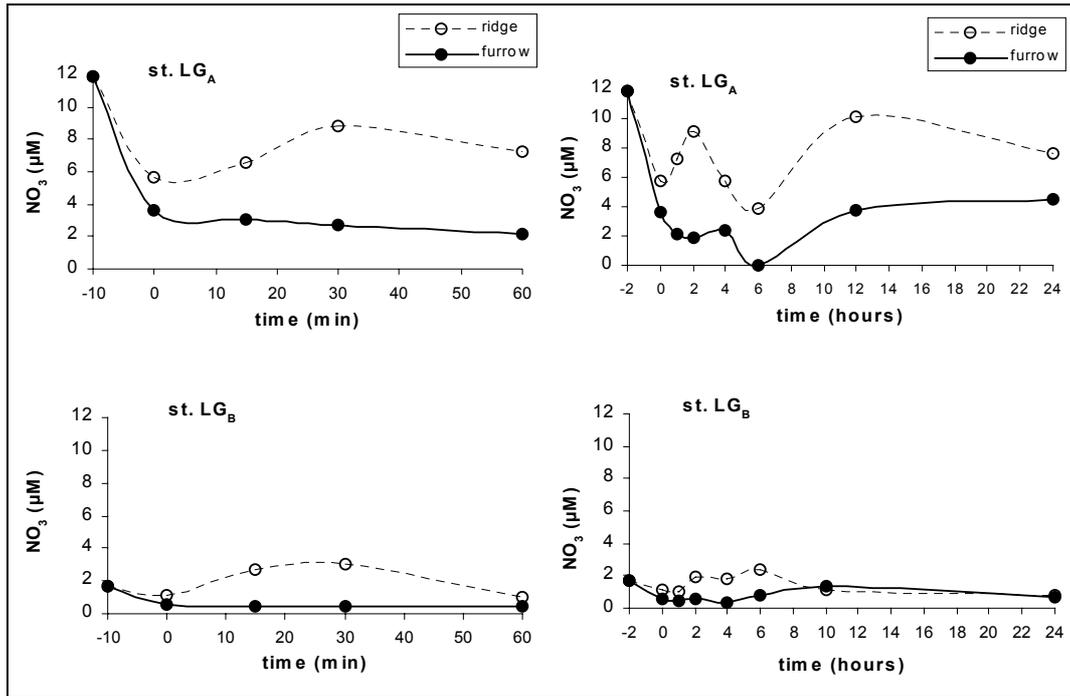


Figure 6 Variation of nitrate (μM) in the ridge and furrow pore water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

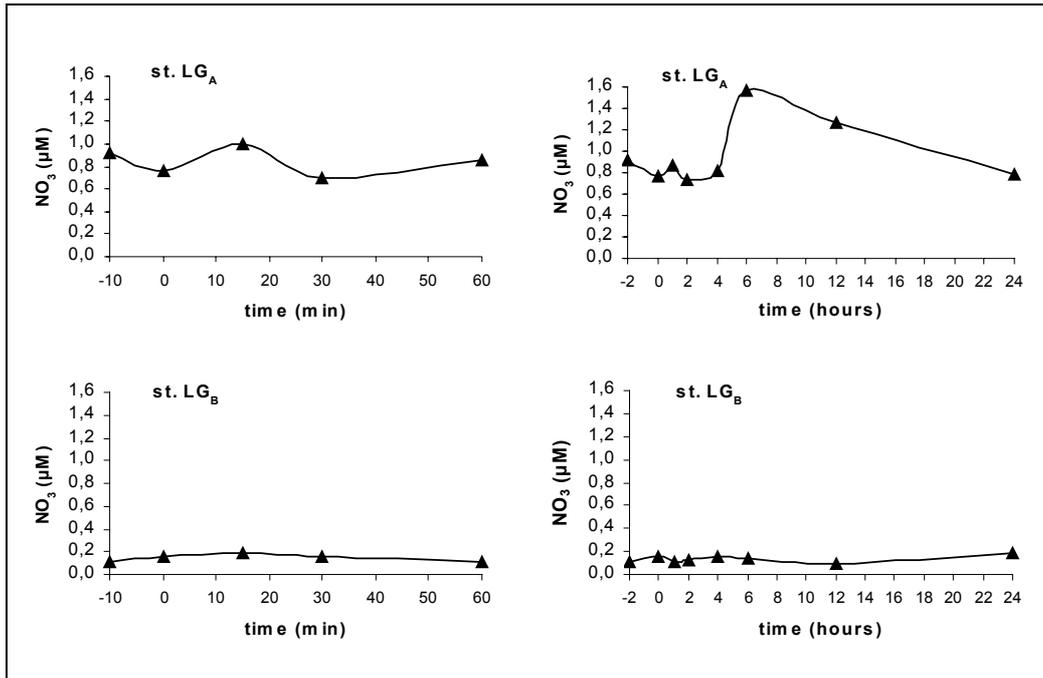


Figure 7 Variation of nitrate (μM) in near bottom water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B .

Total organic nitrogen (TON)

During dredging, at st. LG_A the total organic nitrogen decreased in the ridge and furrow from 477 to 344 μM and from 477 to 193 μM respectively (Fig. 8). The sharp decrease observed in the furrow suggests that organic nitrogen decrease with depth in the sediment. The decrease observed in the ridge may result from pore water renewal. Similar results were obtained in st. LG_B . The organic nitrogen decreased in the ridge and furrow from 893 to 772 μM and from 893 μM to 165 μM respectively. Some oscillations were recorded, mainly in the furrow however, in the ridge the recovery was reached after 60 minutes.

Figure 9 illustrates the variation of total organic nitrogen in near bottom water over the time scale of minutes and hours. At st. LG_A , organic nitrogen increased sharply from 0.55 μM to 2.34 μM during the dredging operation and immediately after. The decrease of TON in pore water during sediment turnover and the concomitant increase in near bottom water indicate its transport from sediment to water column due to fishing activity.

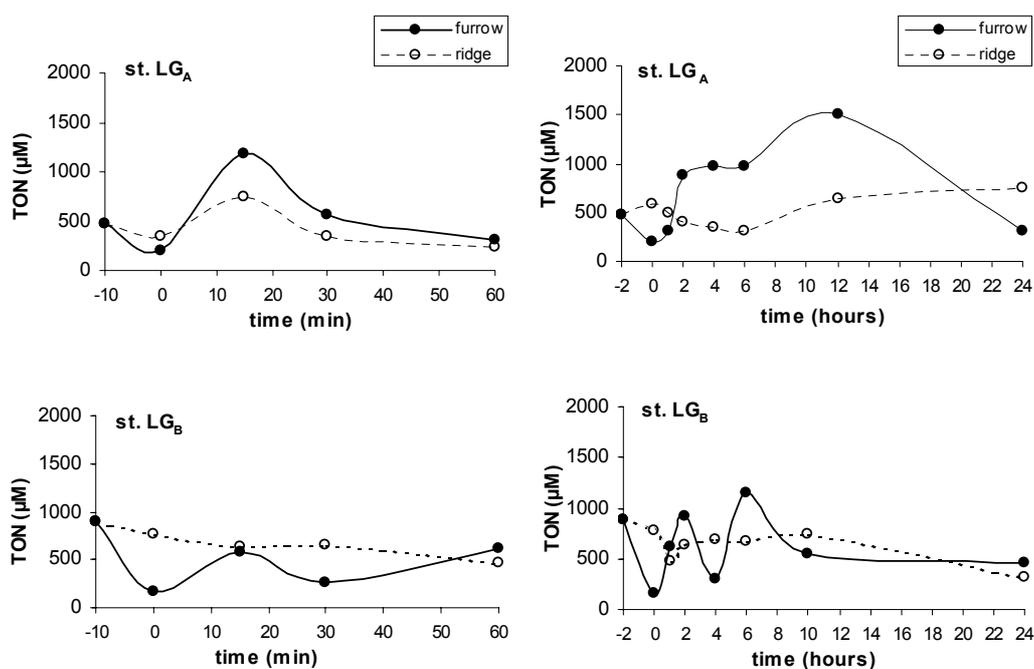


Figure 8 Variation of total organic nitrogen (μM) in the ridge and furrow pore water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and

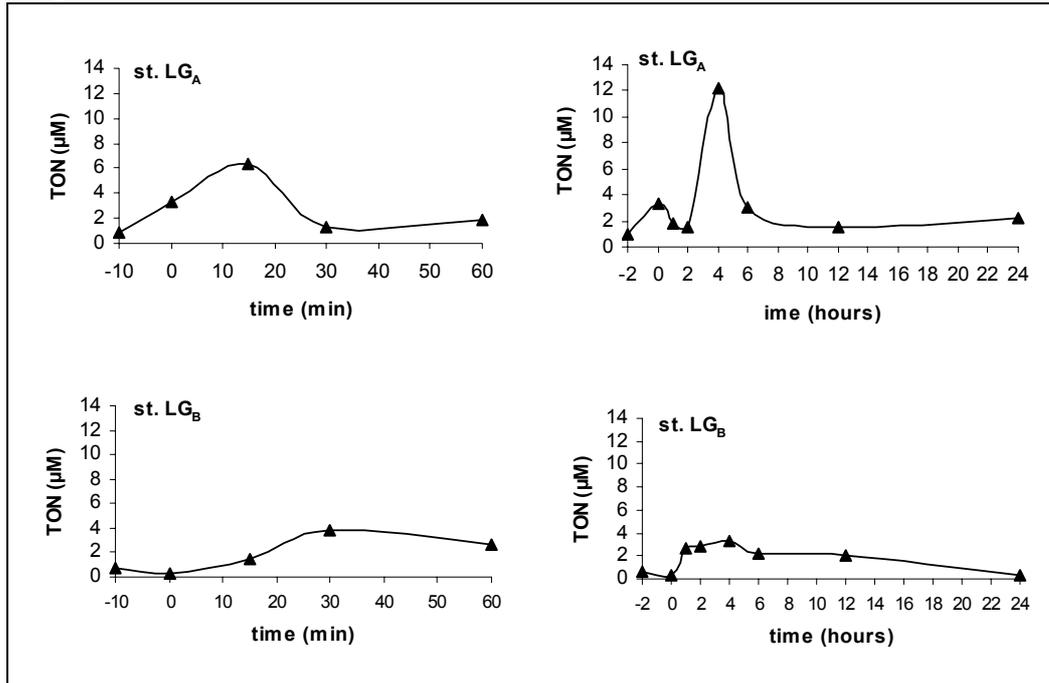
LG_B.

Figure 9 Variation of total organic nitrogen (μM) in near bottom water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

Phosphates

At st. LG_A phosphates decreased during sediment turnover and the recovery in pore water may be reached 24 hours after dredging. At st. LG_B phosphates in the ridge remained higher than in the furrow during the studied period (Fig. 10) and tend towards the recovery 24 hours after dredging.

In both stations (LG_A and LG_B) the expected increase of phosphate in near bottom water was not observed (Fig. 11). The intense dilution in seawater may superimpose on the sediment-water transport.

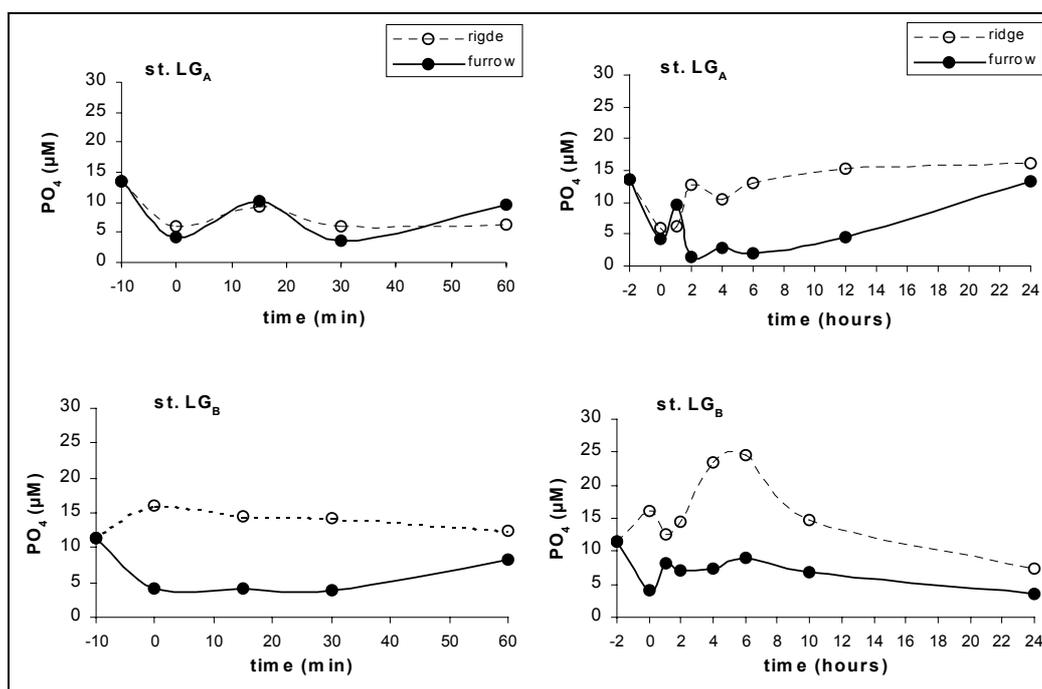


Figure 10 Variation of phosphates (μM) in the ridge and furrow pore water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

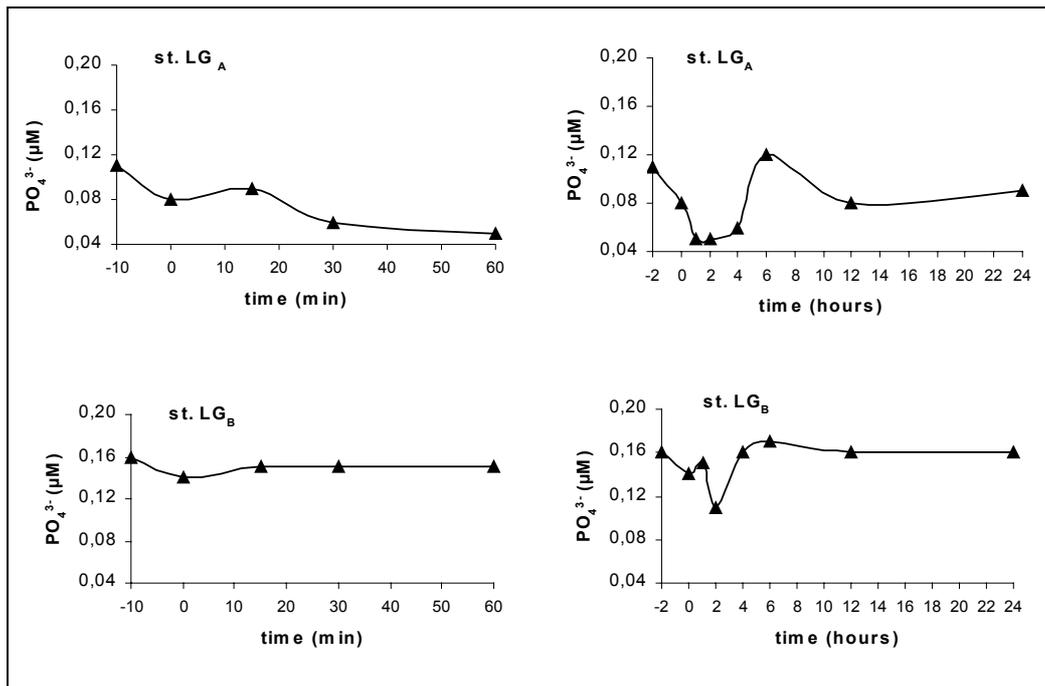
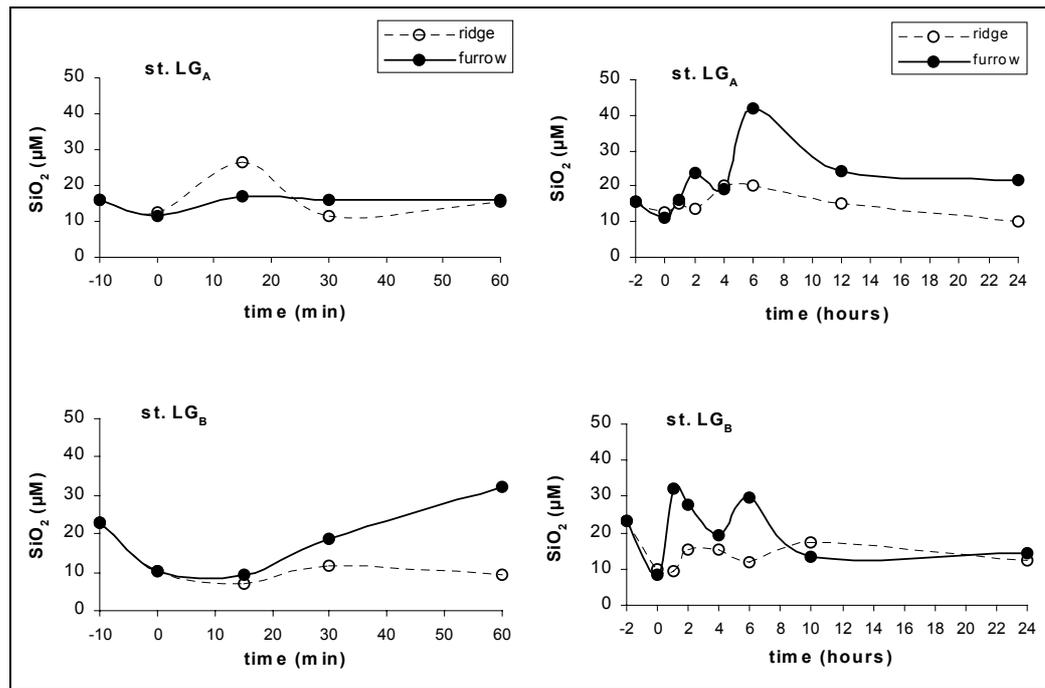


Figure 11 Variation of phosphates (μM) in near bottom water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B.

Silicates

At st. LG_A silicates decreased slightly during turnover (from 15 to 11 μM). In the first sixty minutes after dredging, concentration varied within a narrow range (Fig. 12). At st. LG_B silicates decreased sharply during turnover (from 23 to 10 μM). In this station silicates in pore water clearly reached recovery 10 hours after dredging (Fig 12). In both stations silicates tend to remain higher in the furrow, suggesting that silicates increase with the depth.

At st. LG_A silicates in near bottom water appeared constant over the studied period (24 hours) while, at st. LG_B silicates increased considerably during sediment turnover (Fig 13). In this station we could observe a decrease of silicates in pore water (10 μM) and a concomitant increase in near bottom water (10 μM) during dredging. This points to the transport of silicate in the sediment in the water during turnover caused by the fishing activity.



Figure

re 12 Variation of silicates (μM) in the ridge and furrow pore water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B .

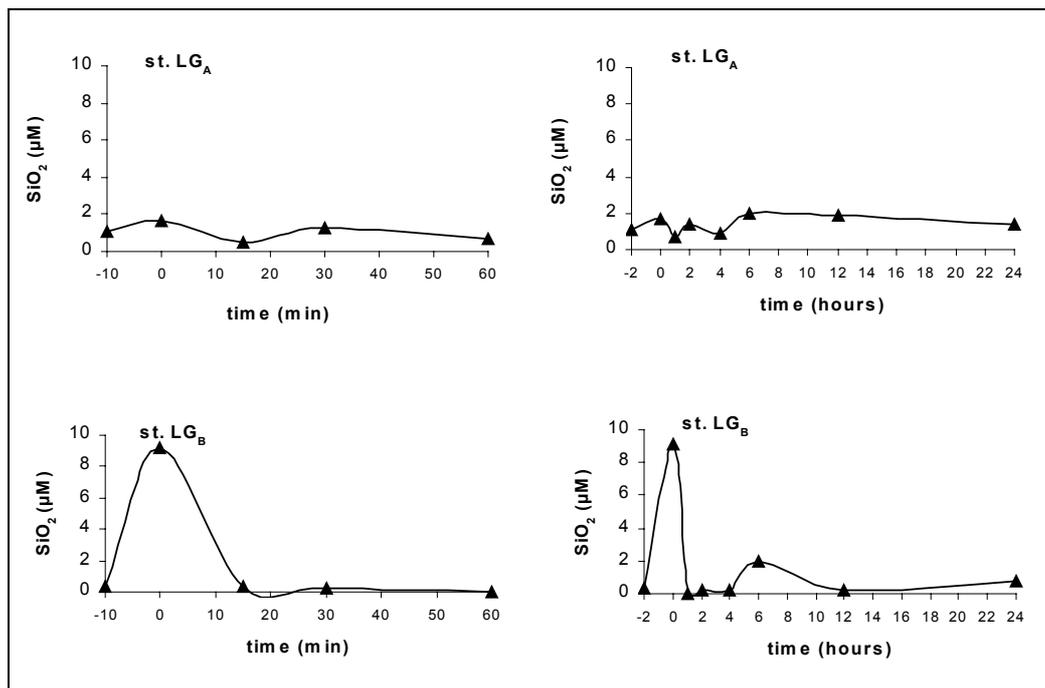


Figure 13 Variation of silicates (μM) in near bottom water, before dredging (-10, -2); during dredging (0) and 15 min, 30 min, 60 min, 2 h, 4 h, 6 h, 12 h and 24 h after dredging at stations LG_A and LG_B .

Benthic fluxes of nutrients before and after dredging

The decrease of ammonium, nitrates, phosphates and silicates in pore water was generally observed at st. LG_A and st. LG_B during and after dredging. This implies

changes of sediment-water fluxes pre and post-dredging. On the basis of the concentrations of those nutrients obtained in pore water and in near bottom water, the nutrient fluxes (F) may be calculated across the sediment water interface by Fick's first law of dilution (Berner 1980):

$$F = -\phi D_s (C_o - C_p) / (\Delta x)$$

where: ϕ =porosity (0,3); $D_s = 14 \times 10^{-6}$ to $22 \times 10^{-6} \text{ cm}^2 \cdot \text{s}^{-1}$ for NH_4^+ and NO_3^- ; $D_s = 5 \times 10^{-6}$ to $8 \times 10^{-6} \text{ cm}^2 \cdot \text{s}^{-1}$ for PO_4^{3-} and $D_s = 3 \times 10^{-6} \text{ cm}^2 \cdot \text{s}^{-1}$ for SiO_2 considering temperature range of 10 – 26 °C (Aller, 1980); $C_o = [\text{NH}_4^+]$ in near bottom water; $C_p = [\text{NH}_4^+]$ in pore water and $\Delta x =$ distance across the interface where concentrations were measured (0 – 2 cm).

The estimated fluxes of ammonium and nitrate in st. LG_A and st. LG_B , during the time scale of minutes and hours are illustrated in Figure 14. The estimated fluxes of phosphates and silicates in st. LG_A and st. LG_B , during the time scale of minutes and hours are illustrated in Figure .15.

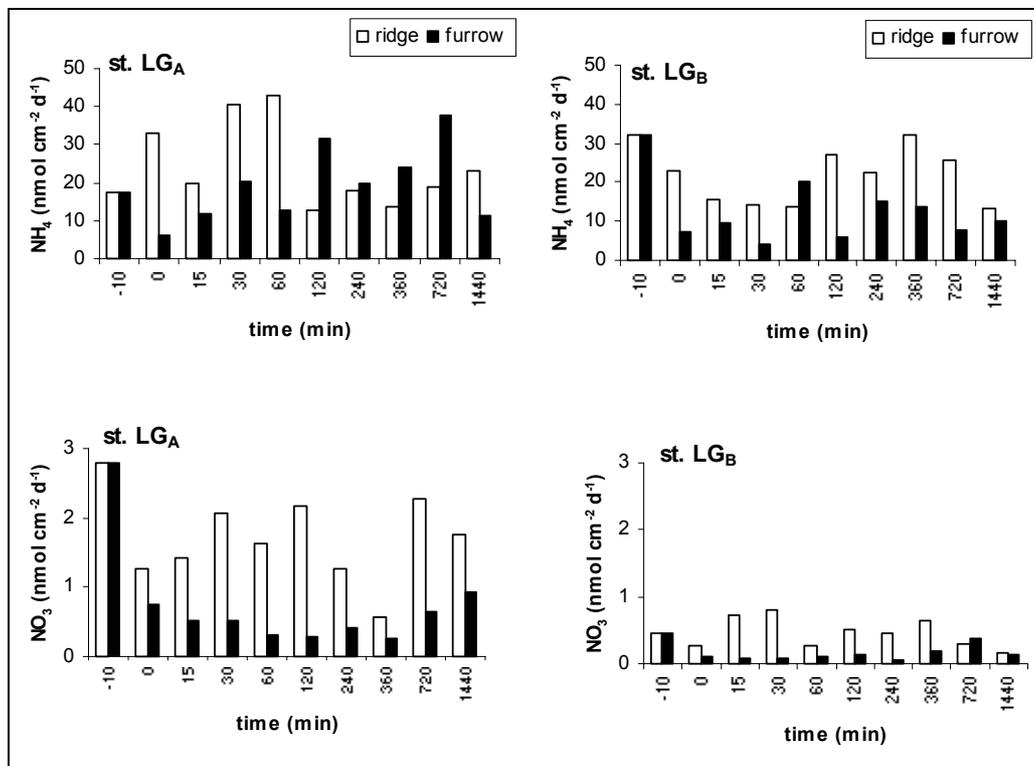


Figure 14 Benthic fluxes of ammonium and nitrates ($\text{nmol cm}^{-2} \text{ d}^{-1}$), before dredging (-10); during dredging (0) and 15, 30, 60, 120, 240, 360, 720 and 1440 minutes after dredging at stations LG_A and LG_B .

Generally the benthic fluxes of nutrients decreased during and a few minutes after dredging. This reinforces the theory that nutrients are transported from the sediment to water column when sediment is disturbed. The Figures show that the fluxes of ammonium, nitrates and phosphates remain clearly lower in the furrow dug in seabed. This means that levels of these nutrients are lower in deeper sediment layers. Oppositely, levels of silicates tend to be higher in deeper layers. The differences of fluxes between ridge and furrow were maintained during the studied period of 24 hours suggesting that, for the existing physical conditions (temperature, wind,

currents) and characteristics of mobile fishing gears, the recovery of seabed may be longer than a day.

Generally, the ridge suffers a more exhaustive renewing of pore water than the sediment of the furrow and molecular fluxes decrease during and a few minutes after dredging. In the furrow the fluxes were lower than in the ridge.

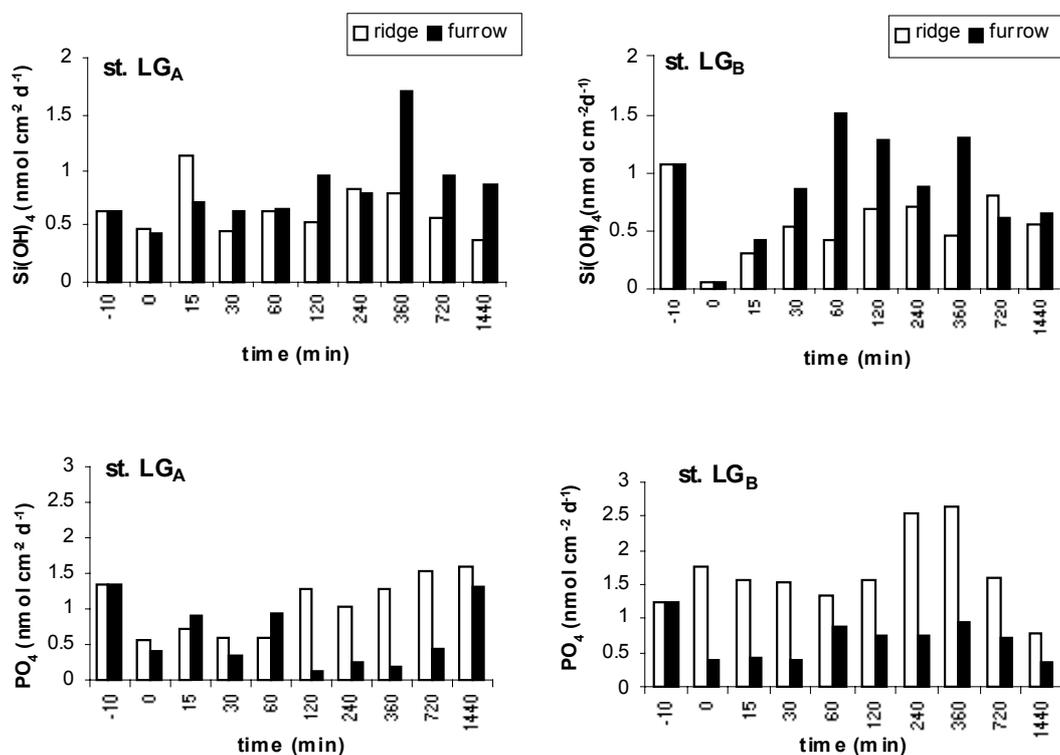


Figure 15 Benthic fluxes of silicates and phosphates ($\text{nmol cm}^{-2} \text{d}^{-1}$), before dredging (-10); during dredging (0) and 15, 30, 60, 120, 240, 360, 720 and 1440 minutes after dredging at stations LG_A and LG_B .

Discussion

The slight variation of dissolved oxygen, pH and turbidity during, and a few minutes after sediment turnover, indicate that sediment disturbance causes a rapid and insignificant disturbance in physical parameters. The slight increase of pH in near bottom water reflects the fact that the sediment is organically poor as observed in the studied areas (Table 2). The rapid recovery of turbidity indicates that the cloud of particles formed during turnover is rapidly deposited and/or transported by currents.

When the mobile gear digs the furrow, deeper layers of sediment are exposed and its chemical and biological characteristics often contrast with those of the ridge formed by the overturn of the upper layers of sediment. The sediment along the path dragged by the teeth of the mobile gear is overturned and the microphytobenthos coming from all towpaths are dispersed in the surrounding area. After the dredging operation, the lower concentrations of chlorophyll *a*, obtained in the furrow are as a result of the lower production of microphytobenthos in deeper sediment layers that become exposed (Boon 1998).

The sediment is a sink of nutrients; (Henriksen 1983; Sundby 1992 and Forja 1994) thus, during turnover, when the pore water of superficial sediment is renewed, the nutrients are release to the water column (Fanning 1982). According to the results found in the present study, ammonium, nitrates, organic nitrogen, phosphates and silicates generally decrease in pore water after sediment turnover, being probably exported to the water column. The decrease of the benthic fluxes of nutrients during and the few minutes after turnover indicates that sediment function as a source of nutrients during disturbance. However, when the re-establishment of the seabed is not reached, the sediment has a lower contribution to the nutritive enrichment of the water column. The increase of ammonium, organic nitrogen and silicates observed in near bottom water during dredging corresponds to a decrease in these constituents in pore water, reinforcing the theory that nutrients are exported from seabed to the water column. Other studies on ecological consequences of dredging corroborate these findings. Riemann and Hoffmann (1991) demonstrate that silicates had a surprisingly higher concentration in the water column of a dredged area than in a control area. They also refer that water column receives organic and inorganic nutrients that may cause significant changes in the nutrient level. If mobile gears causing nutrient release frequently disturb large areas of seabed it is expected that changes in the primary production of phytoplankton may occur.

3.1.2.3 ECOLOGICAL DISTURBANCE OF DREDGING IN A COASTAL AREA OF THE NORTHERN ADRIATIC SEA: PHYSICAL AND CHEMICAL EFFECTS OF ITALIAN FISHERIES ON THE SEABED AND THE UPPER WATER COLUMN.

Introduction

In this section, results concerning the seasonal study performed on *Chamelea gallina* clam beds are shown. As requested in the Task 7 of the project, the research activity characterized the seasonal pattern of the various ecosystem components affected by hydraulic dredging.

With this aim, attention of the Italian working group has been paid on the following topics:

1. To characterize the seasonal pattern of hydrology and chemistry of the water column during dredging.
2. To characterize the sediment composition (grain size and chemical) required for establishing the *C. gallina* habitat.
3. To estimate the physical and chemical changes of the properties of the near bottom water and sediments before and after dredging.
4. To estimate the grain size and chemical changes before and after dredging.
5. To calculate the fluxes of dissolved nutrients from sediments due to fishing activity.

Methodologies

In the framework of the Ecodredge EU program, six cruises were carried out, according to the following Table:

Table 1. Cruises carried out.

<i>Cruise</i>	<i>Seasons</i>	<i>Date</i>	<i>Topics</i>	<i>Matrices</i>	<i>Stations</i>
FAIR 01	Summer	1999, June	07-16 Preliminary experiment	Water column, sediments	L01, L02, L03, L04, J01, J02, J03, J04 L02 (experiment)
FAIR 02	Winter	2000, February	07-09 Seasonal features	Water column	L01, L02, L03, L04, J01, J02, J03, J04
FAIR 03	Spring	2000, May	04-18 Seasonal features	Water column	L01, L02, L03, L04, J01, J02, J03, J04
FAIR 04	Summer	2000, July	04-05 Seasonal features	Water column	L01, L02, L03, L04, J01, J02, J03, J04
FAIR 04B	Summer	2000, August	31 Experiment	Water column, pore water and sediments	L02
FAIR 05	Autumn	2000, October	17-24 Seasonal features	Water column	L01, L02, L03, L04, J01, J02, J03, J04

dissolved nutrients from pore water into the near bottom water; iii) the changes in chlorophyll concentration from sediments, before and after dredging. To survey the water column, CTD casts and samples were also performed, before and some hours after dredging, in which dissolved nutrients, POC and TN, chlorophyll and particles spectra were determined.

As regards sediments, during the same cruises for the water column, samples taken were to characterize the grain size distributions, the porosity, the Eh potential and the pH, the organic carbon and nitrogen in sediments (both before and after dredging).

Methods on the water column.

CTD casts data were filtered and averaged to dbar. Dissolved oxygen determinations were made on-board by a DOSIMAT burette following the Winkler method. Dissolved nutrients (N-NH₃, N-NO₂, N-NO₃, Si-SiO₄, P-PO₄) were filtered on Whatman GF/C glass fiber filters (porosity=1 µm) and analyzed in a SYSTEAL-ALLIANCE autoanalyzer, following the methods described in Hansen and Koroleff (1999). Chlorophyll a samples, filtered on the same way as nutrients, were homogenised in ULTRA-TURRAX (20000 rpm) for 30 seconds and extracted in acetone 90% at 8°C for 12h. Fluorescence readings were measured in a PERKIN-ELMER luminescence spectrometer on clarified extracts, before and after addition of 1 drop of HCl 1 N. Chlorophyll α and phaeopigments concentrations were obtained by the equations described in Holm-Hansen et al. (1965). POC and TN analyses were carried out on Whatman GF/F filters (porosity=0.7 µm), after desiccation at 60°C and addition of HCl 1N to remove the inorganic fraction, using a CHN PERKIN ELMER elemental Analyzer, following Hedges and Stern (1984). Particle size spectra were determined with a multichannel COULTER COUNTER particle analyzer: the observed particle size ranged from 2 to 100 µm (Sheldon et al. 1972).

Methods on pore water and sediment.

Pore water and sediments were sampled by scuba-divers at different time scale (45 minutes prior to dredging and 15, 45, 150 minutes post dredging), using open 50 ml syringes (28x80 mm) for nutrients and 5 ml syringes (12x40 mm) for chlorophyll, gently forced into the sediment and plugged to avoid the later contact with air. Three replicates were made for each site, both on the furrow and in the track, immediately frozen at -30°C on board. In the laboratory, the sediment cores were sliced, the top layer (0-2 cm) centrifuged to extract pore water and filtered with Whatman GF/C glass fibre filters (1 µm porosity). Dissolved nutrients were determined following the same procedure as in the water column.

Chlorophyll and phaeopigments were determined in the top layer of the sediment cores (0-1 cm), according to the same method as in the water column (by fluorometry; Holm-Hansen et al. 1965).

Methods for sediments analysis

The sediments samples were carried out by a Van Veen grab from the ship; pH (referring to a solution of 7.1 pH) and Eh (referring to a Zobell 1946, solution) were immediately measured. Subsamples were taken, to determine organic carbon and nitrogen, porosity of sediments and grain size distribution. Similar analysis were

done on sediments collected in stations L02 and J02 just before and after the dredging action. In these stations, some cores were collected and analyzed in laboratory about grain size, organic carbon and nitrogen concentrations. As regards porosity, wet weight was related to dry weight, obtained by desiccating sediments at 80°C; for grain size distribution, other steps were added, such as elimination of organic matter with 10% H₂O₂ washings, 80°C sediment drying and then mechanical sieving using a 500 µm mesh, 250 µm, 125 µm and 63 µm, finally weighting each fraction. The distinction of different grain size classes was performed following the Wentworth (1922) scale. Organic carbon and nitrogen were analyzed in a CHN PERKIN ELMER elemental Analyzer, after drying at 60°C and sediment homogenization in a mortar.

Results

Water column

Winter conditions (February, FAIR_02): temperature showed the low values typical of the winter season (range: 5-8°C; Fig. 2). In this cruise, very high salinity waters, due to reduced rainfalls in the previous months, were detected very close to the coast (up to 38 PSU 3 miles off-shore; Fig. 3). The low riverine discharge is evident in transparency (a surface minimum of about 50% in transmittance; Fig. 4), relative high when compared to other situations of high suspended matter transport. The chlorophyll concentrations reveal the early growth of phytoplankton, more pronounced in the distal stations of the Jesolo transect (up to 3 µg/dm³; Fig. 5).

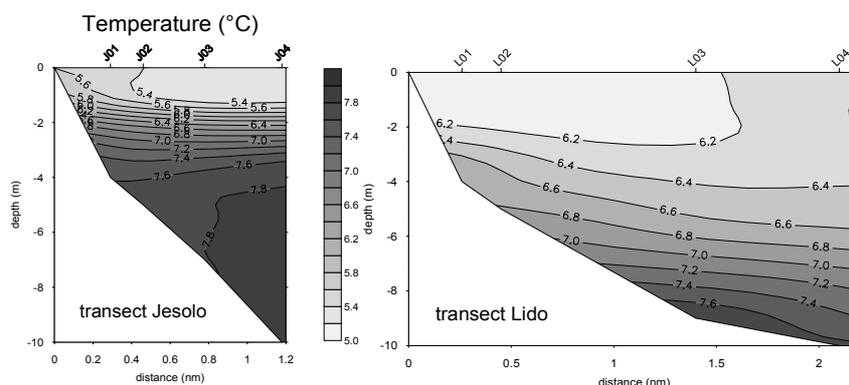


Figure 2 February 2000: sections of temperature (°C) by CTD.

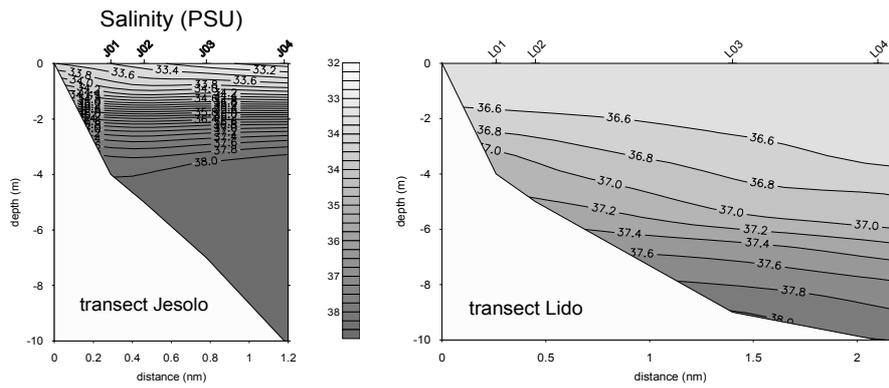


Figure 3 February 2000: sections of salinity (PSU) by CTD.

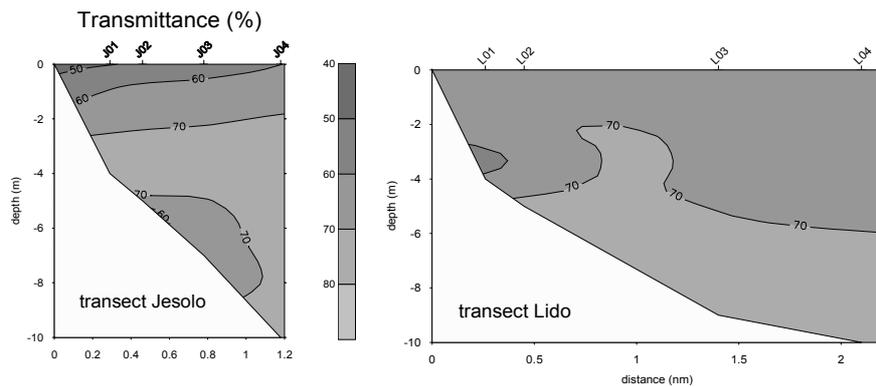


Figure 4 February 2000: sections of transmittance (%) by CTD.

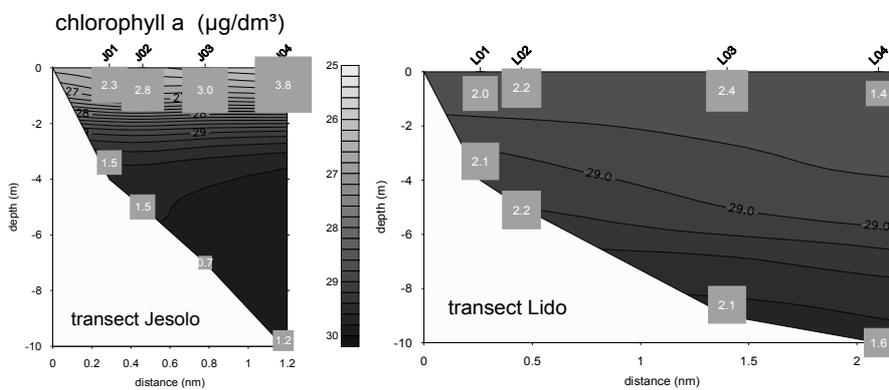


Figure 5 February 2000: chlorophyll a ($\mu\text{g}/\text{dm}^3$) by discrete samples, represented by squares proportional to its concentration, superimposed on the density field (kg/m^3).

Spring conditions (May, FAIR_03): due to the atmospheric warming, water temperature increased, leading to vertical stratification (range 13-21°C; Fig. 6), shown in the salinity distribution too (range 29-36 PSU; Fig. 7). At the furthest stations of the Lido transect, a small gyre was evidenced by the presence of particulate material, as detected by transmittance (Fig. 8) and chlorophyll values (Fig. 9).

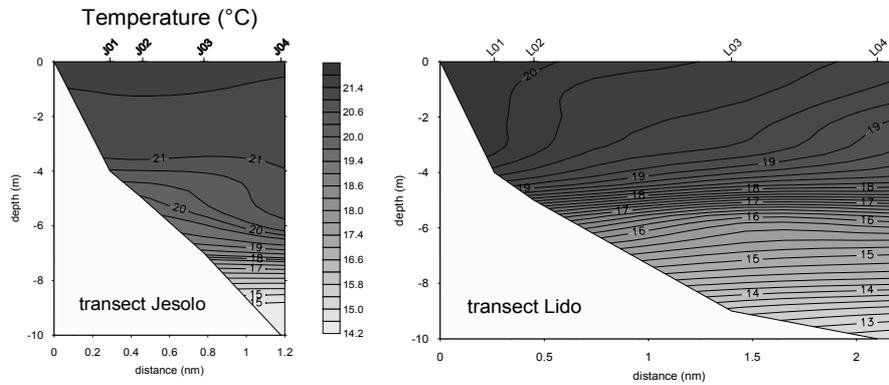


Figure 6 May 2000: sections of temperature (°C) by CTD.

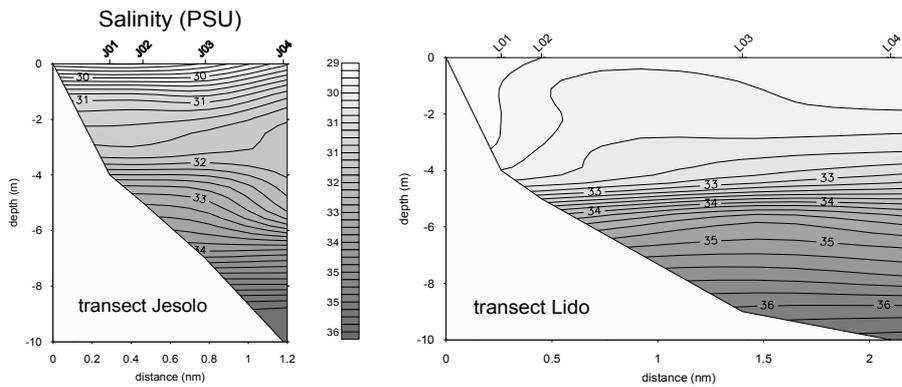


Figure 7 May 2000: sections of salinity (PSU) by CTD.

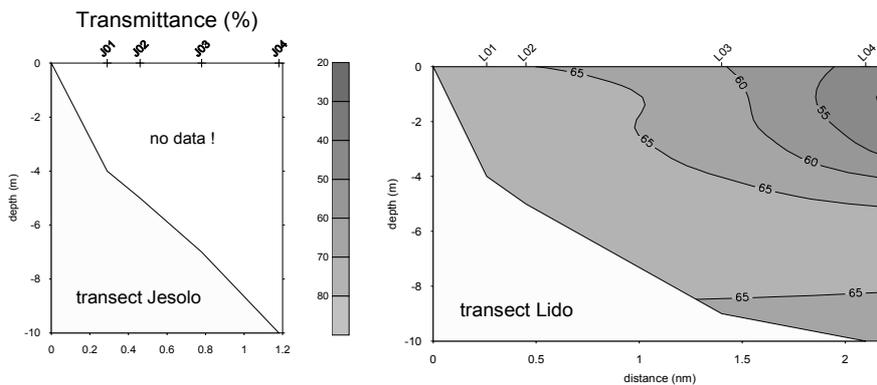


Figure 8 May 2000: sections of transmittance (%) by CTD.

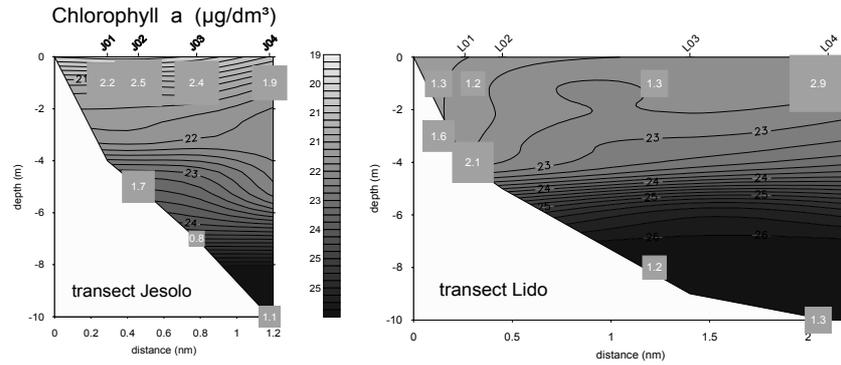


Figure. 9 May 2000: chlorophyll a ($\mu\text{g}/\text{dm}^3$) by discrete samples, represented by squares proportional to its concentration, superimposed on the density field (kg/m^3).

Summer conditions (July, FAIR_04): temperature showed the high values typical of the summer period (a surface maximum of about 26°C), that occurred through almost the whole water column (T excursion between surface and bottom= $1\text{-}2^\circ\text{C}$). Salinity vertical distribution was quite different in the two transects (Fig. 10): at Jesolo the dilution effect of the Piave river runoff was evident, increasing the phytoplankton biomass up to values of $10 \mu\text{g}/\text{dm}^3$, while at Lido the water column was quite homogeneous, with chlorophyll concentrations around $3\text{-}4 \mu\text{g}/\text{dm}^3$ (Fig. 11).

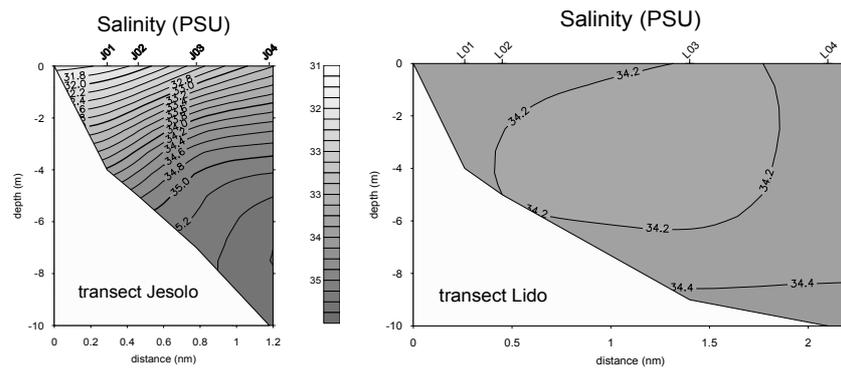


Figure 10 July 2000: sections of salinity (PSU) by CTD.

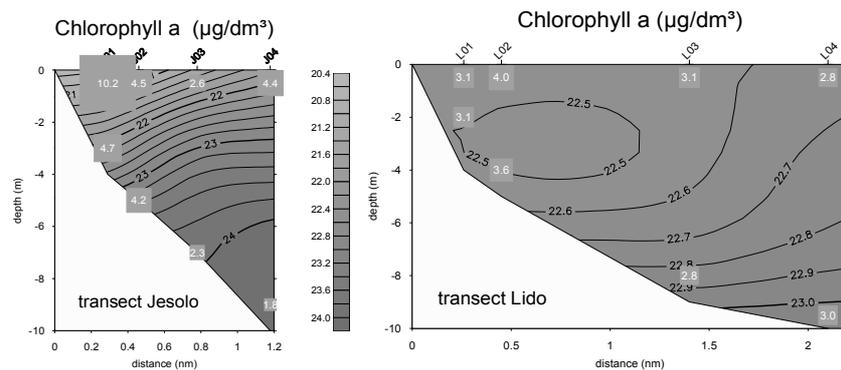


Figure 10 July 2000: chlorophyll a ($\mu\text{g}/\text{dm}^3$) by discrete samples, represented by squares proportional to its concentration, superimposed on the density field (kg/m^3).

Autumn conditions (October, FAIR_05): the atmospheric cooling reflected in the progressive loss of heat from the surface layer of the water column, where lower temperatures were detected (about 17°C at surface, up to 19°C at bottom, Fig. 12).

Because of the presence of a water lens at the distal stations, due to the plume of the Piave river, colder (about 16°C) and fresher (24 PSU) water bodies were observed in front of Jesolo (Fig. 13). This outflow was not evidenced by the content in particulate matter, transparency showing minimum at bottom due to resuspension, the same picture as in front of Lido (Fig. 14). We do not exclude they could be simultaneous dredging by other fishing-vessels observed in the neighborhoods. Phytoplankton biomass showed even sustained values as usually does in the autumn season: about 5-6 $\mu\text{g}/\text{dm}^3$ at Jesolo, from 6 at surface to less than 2 $\mu\text{g}/\text{dm}^3$ at bottom al Lido (Fig. 15).

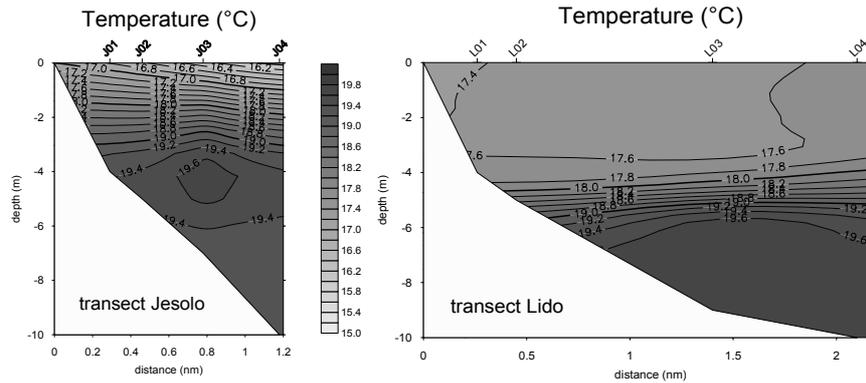


Figure 12. October 2000: sections of temperature (°C) by CTD.

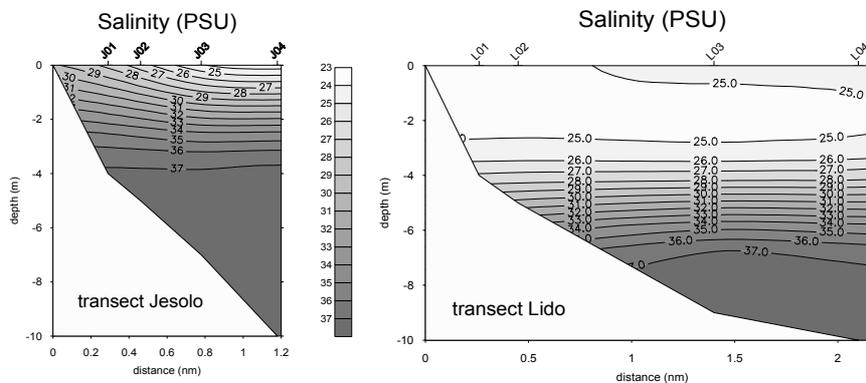


Figure 13. October 2000: sections of salinity (PSU) by CTD.

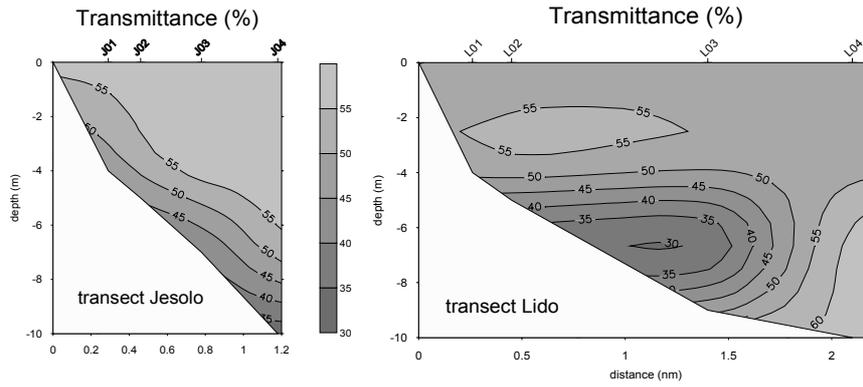


Figure 14. October 2000: sections of transmittance (%) by CTD.

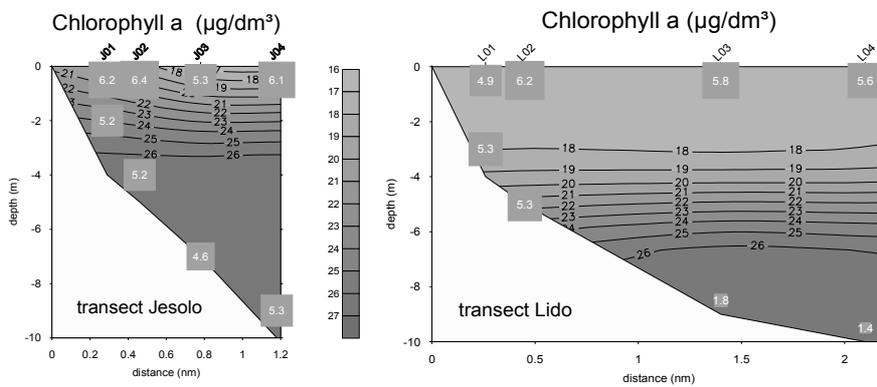


Figure 15. October 2000: chlorophyll a ($\mu\text{g}/\text{dm}^3$) by discrete samples, represented by squares proportional to its concentration, superimposed on the density field (kg/m^3).

Modifications in the hydrochemistry of the water column after dredging

The June 1999 preliminary experiment showed as modifications induced by dredging were mainly revealed by:

1. the increase of turbidity: after the initial perturbation, the system return to the original conditions in 1h (Lido) - 1.5 h (Jesolo), according to the sediment quality and tidal current (Fig. 16);
2. the ratio between chlorophyll degradation products/alive chlorophyll, that increased after dredging, due to the resuspension of dead phytoplankton as well as phytobenthos cells (Fig. 17).

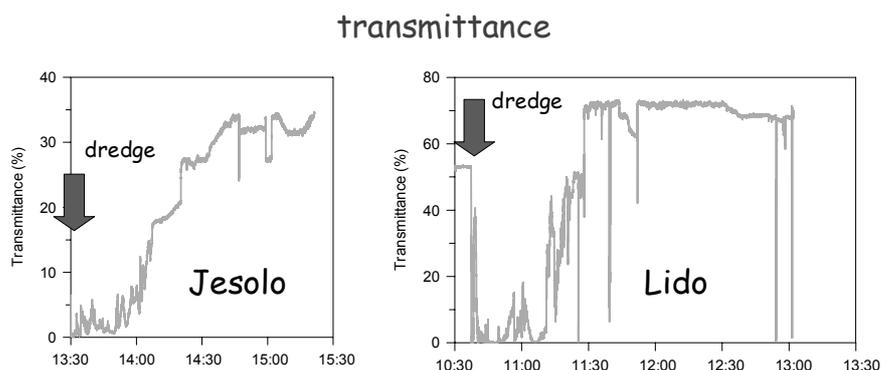


Figure 16. June 1999: transmittance variations caused by dredging.

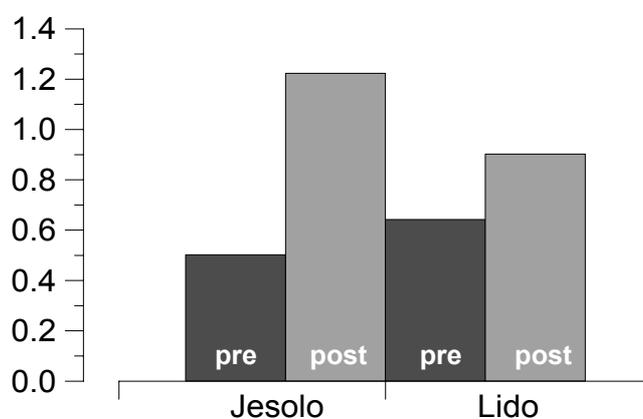


Figure 17. June 2000: ratios chlorophyll degradation products/alive chlorophyll.

The August 2000 experiment had the following results:

3. transparency decreased on the whole water column due to fishing activity, after which it increased again; 2 hours afterwards, values higher than the starting ones were measured, due to more transparent water being advected laterally at high tide (Fig. 18);
4. particle size distribution and frequency, as detected by Coulter Counter, showed an increase of smaller fractions (less than 10 μm) till 60 minutes after dredging, then a decrease. A new increase at 150 minutes appeared (second oval at right), not explainable by transmittance data, but probably due to current (Fig. 19);
5. all of the dissolved nutrients do not reveal any significant changes in the water column due to dredging;
6. particulate organic carbon (POC) and chlorophyll/pheopigment ratios seem to be the best indicators of dredging activity, the former increasing at bottom from 203 to 282 $\mu\text{g}/\text{dm}^3$ (Fig. 20), the latter decreasing from 1 to 0.7 (Fig. 21).

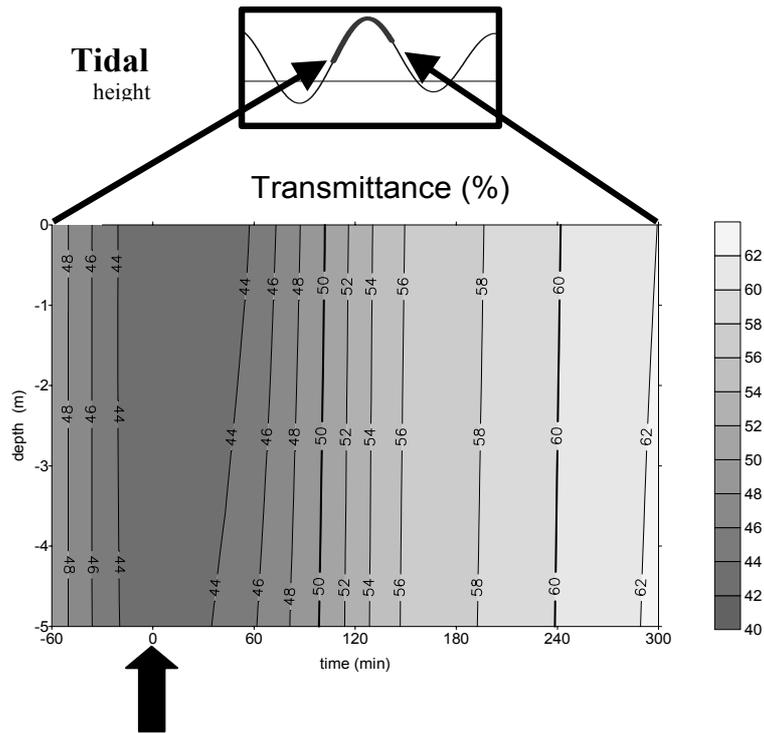


Figure 18. Modifications induced by dredging on the water column: transparency (%).

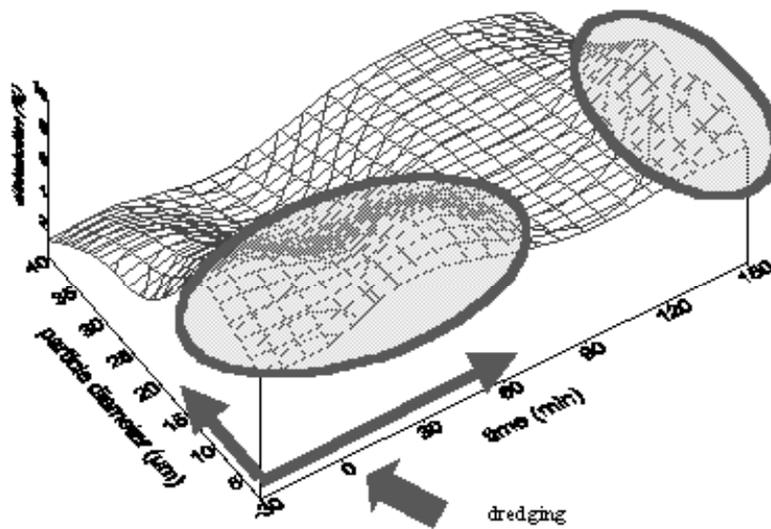


Figure 19. Modifications induced by dredging on the water column: particle size distribution by Coulter Counter.

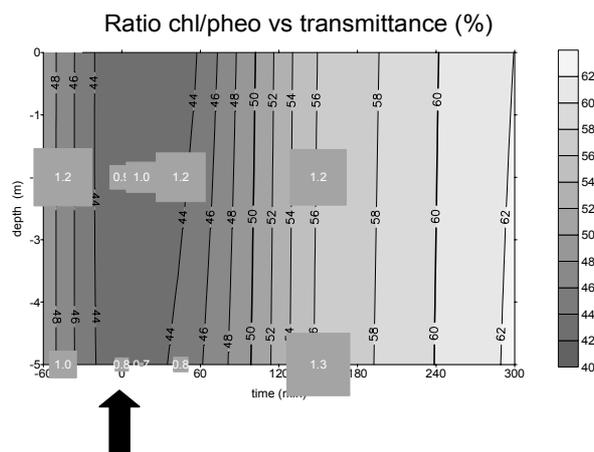


Figure 20. Modifications induced by dredging on the water column: POC ($\mu\text{g}/\text{dm}^3$) vs transmittance (%).

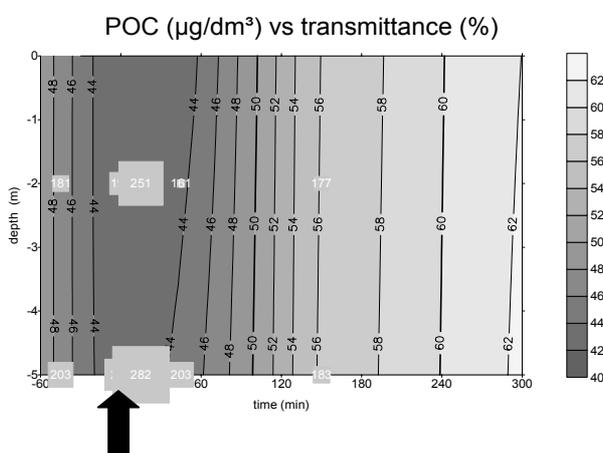


Figure 21. Modifications induced by dredging on the water column: chlorophyll a/phaeopigments ratio vs transmittance (%).

Modifications in pore water and sediments

Dissolved nutrients variations observed in pore water, as well as chlorophyll differences in sediments, have been studied in cores sampled by scuba divers, both in the furrow and track. The results are the following:

- a) ammonia showed a different behaviour: at the edge, dredging released this nutrient into the water column, while in the furrow its concentrations increased in pore waters; short-time trends are different too, with a slight increase at the edge and a decrease in the furrow (Fig. 22);
- b) the oxidized forms on nitrogen (nitrite plus nitrate, NO_x) seemed to behave similarly: a sharp decrease in pore waters has been measured, followed by a slight increase. 150minutes do not seem to be enough to complete the recovery time of these forms of nutrients (Fig. 23);
- c) orthosilicates showed an opposite trends, fishing causing an increase in pore waters concentrations. For silicates, 150 minutes was a sufficient time to recover the initial concentrations (Fig. 24);

d) chlorophyll concentrations showed a general decrease after dredging, but a different trend was observed in sediments: at the edge, the starting conditions were reached after 60minutes, while in the track the process of recovery took more time (150minutes: Fig. 25). The trend on chlorophyll degradation products (phaeopigments), in the furrow and track is similar to active chlorophyll. The decrease in concentration after 20minutes continued more markedly until 60minutes after dredging, increasing then until they reached the starting values, except in the furrow, where it ended higher than the initial levels (Fig. 26).

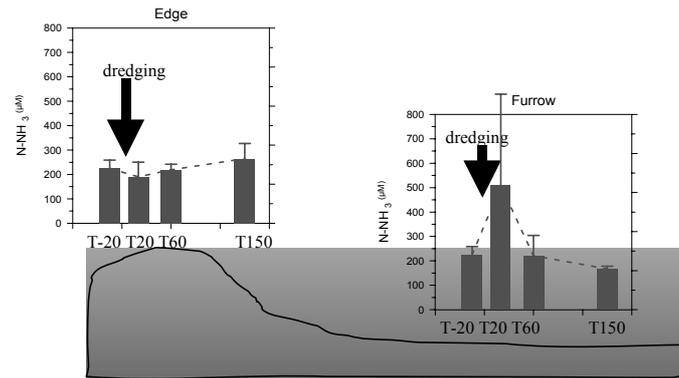


Figure 22. Modifications induced by dredging in pore water: ammonia (μM).

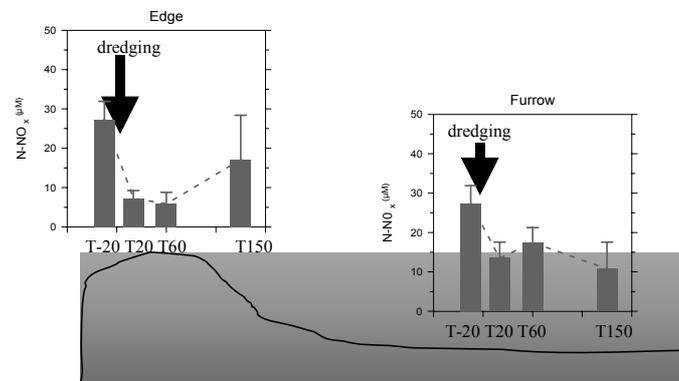


Figure 23. Modifications induced by dredging in pore water: nitrite plus nitrate (N-NO_x; μM).

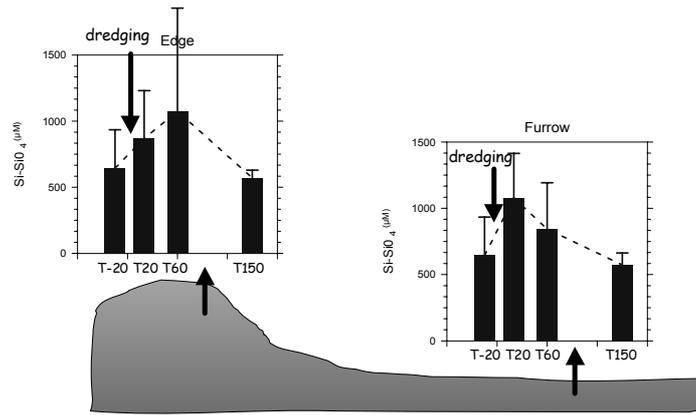


Figure 24. Modifications induced by dredging in pore water: orthosilicates (μM).

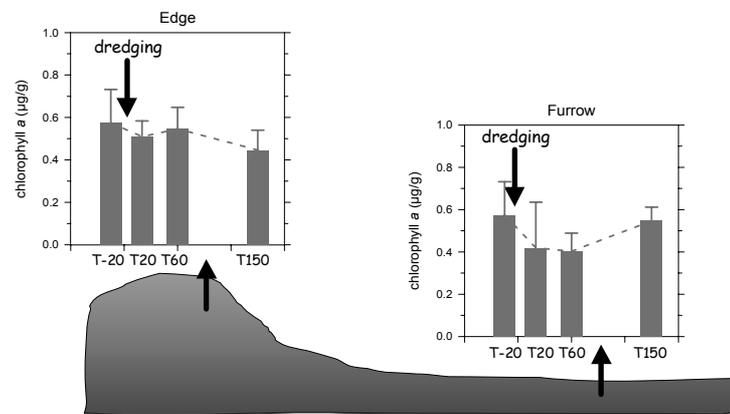


Figure 25. Modifications induced by dredging in sediment: chlorophyll a (μg/dm³).

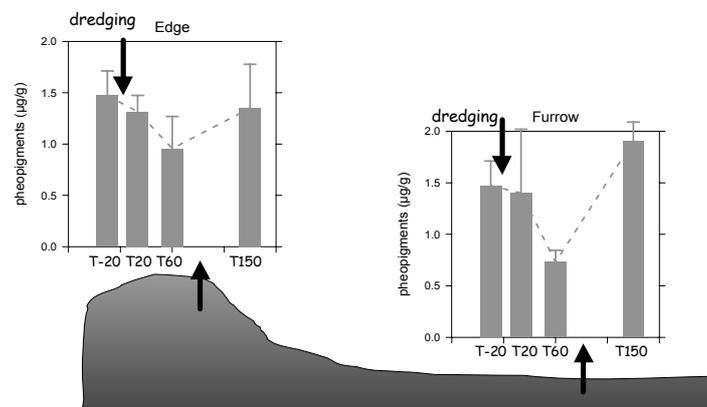


Figure 26. Modifications induced by dredging in sediment: phaeopigments (μg/dm³).

Therefore, some patterns are difficult to explain from the point of view of a simple observation about different concentrations, as measured before and after dredging. Thus we performed calculations of fluxes from pore into near bottom water for the main forms of dissolved nutrients. Results showed that ammonia (N-NH₃), oxidised nitrogen (NO_x) and orthosilicate (Si-SiO₄) exhibited a different behaviour mainly due to their chemical state:

- Ammonia has a consistent behaviour in the furrow, increasing its flux immediately after fishery (from 20 to 60x10³ nm/cm²/day, Fig. 27), as one could expect from a reduced form, common on the seabed, and resuspended by dredging activity;
- N-NO_x has a consistent behaviour in the furrow too, decreasing its flux immediately after dredging (from 2500 to 500 nm/cm²/day, Fig. 28), a quite common exchange for an oxidised chemical compound;
- Orthosilicate has a consistent behaviour both on furrow and on the ridge, increasing its flux immediately after dredging (from 20 to 40x10³ nm/cm²/day, Fig. 29);

All the measured fluxes appear to be of the same order of magnitude as those calculated from other authors from the Portuguese (see [Sections 3.1.2.1-2](#)). Except for N-NO_x, all the calculated fluxes return to their original values after 1h 30minutes.

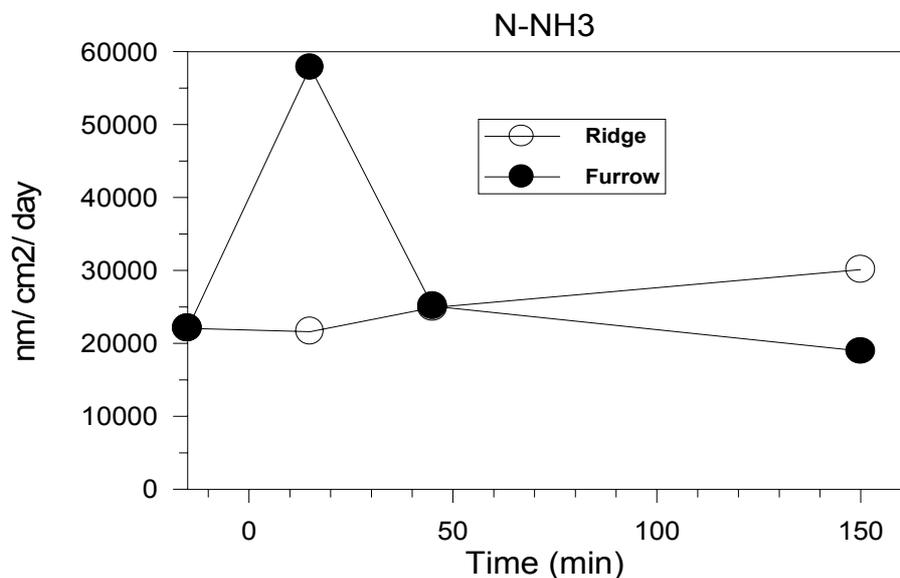


Figure 27. Fluxes of ammonia (N-NH₃) calculated from pore waters into near bottom waters.

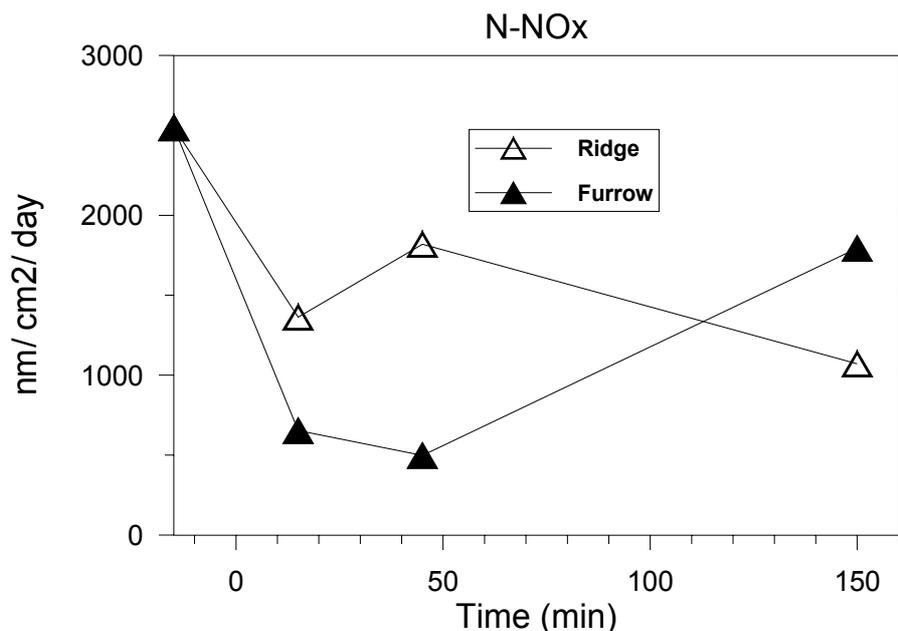


Figure 28. Fluxes of oxidized nitrogen (N-NO₂+N-NO₃) calculated from pore waters into near bottom waters.

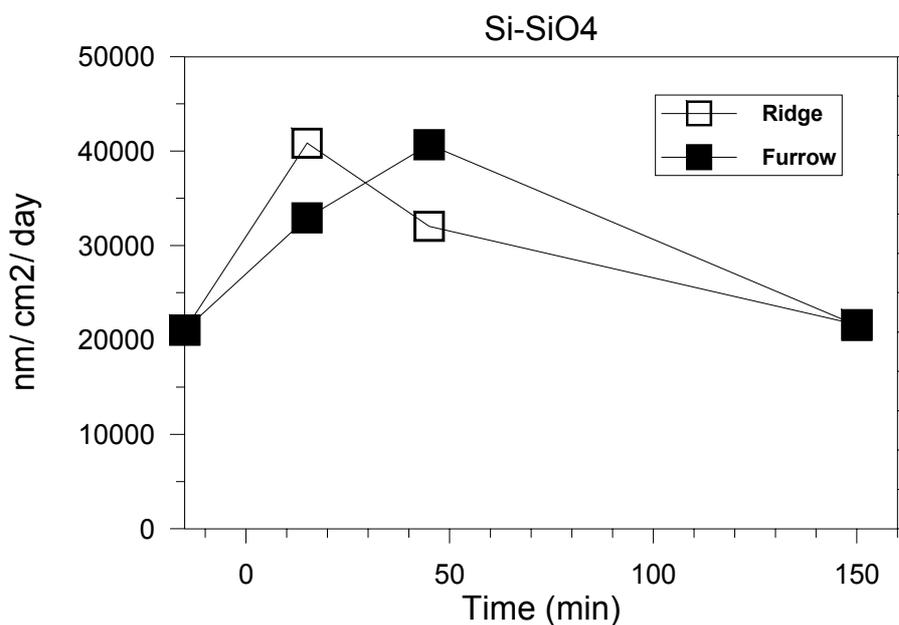


Figure 29. Fluxes of orthosilicate (Si-SiO₄) calculated from pore waters into near bottom waters

Modification in sediments

Table 2 shows the sediment characteristics along the two transects. The optimal habitat for *C. gallina* is sandy sediment and positive potential referred to the calomel potential (Barillari et al. 1978), so at station L03 and J02 it is possible to marked approximately the outer limits of this habitat. The porosity increased too, going offshore because the quantity of fine sediments increased. Organic carbon and

nitrogen have the same trend; they increase with depth but the quantities in Jesolo are higher than Lido (this may be because of the river input of Piave that occurred much more at Jesolo than Lido, and so organic matter are transported by this river and sunk along the Jesolo shore).

Table 2 Sediment characteristic of seabed along the two transects

Station	Depth (m)	Temperature (°C)	pH	Eh (mV)	Porosity (%)	Sand (%)	Silt&clay (%)	C org. (%)	Nitrogen (%)
L1	3,7	23,6	6,25	164,4	42	97,50	2,50	0,07	0,02
L2	5	23,5	6,5	180,1	44	98,01	1,99	0,11	0,02
L3	8	23,6	6,35	-24	50	64,51	35,49	0,34	0,05
L4	11	20,2	6,36	-137	55	43,00	57,00	0,45	0,06
J1	4	24,5	6,1	93	47	92,32	7,68	0,09	0,02
J2	6	24,5	6,43	-19	49	85,30	14,70	0,11	0,03
J3	8	24,5	6,48	-131	55	56,89	43,11	0,50	0,06
J4	11	22,8	6,61	-120	67	10,85	89,15	0,87	0,10

In Figures 30 and 31 it is possible to compare the Eh potential and sand percentile, with the aim being to find a similar trend going off-shore and to see environmental changes at stations L03 and J02. Of course, the fishing area is smaller at Jesolo compared with Lido, because morphological and chemical differences between these areas are evident: at Jesolo the 11 m depth is reached just 1 mile far from the coast, but at Lido it is further out at up to 2 miles, with the sandy area more extended than at Jesolo. The same picture is evident in the Eh distribution too.

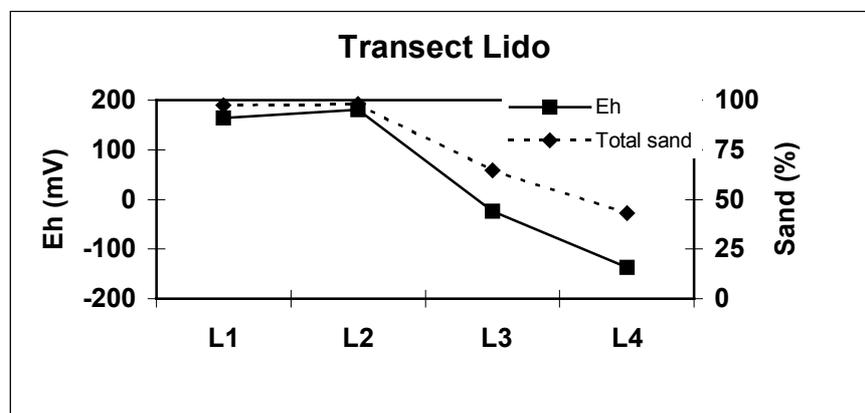


Figure 30: comparison between Eh potential and sandy composition of sediments along Lido transect

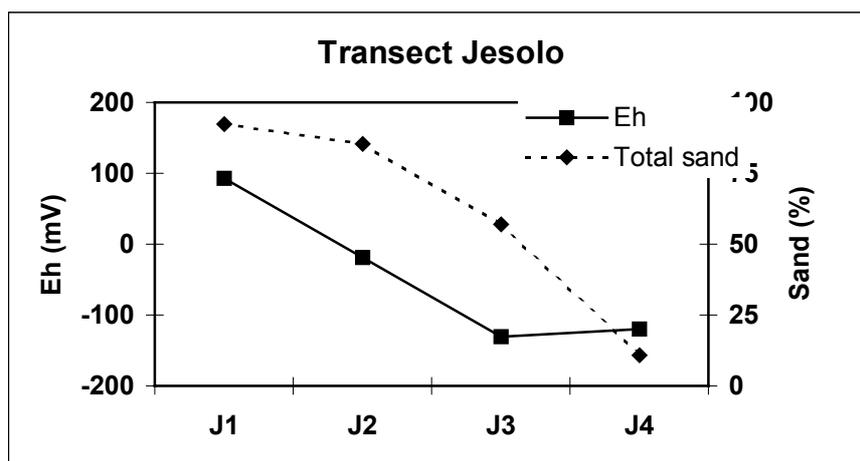


Figure 31: Comparison between Eh potential and sandy composition of sediments along Jesolo transect

A comparison of before and after dredging data does not demonstrate an environmental change (Tab. 3), except perhaps for porosity, increasing after dredging. As regards nitrogen, the concentration detected was very low, reaching the lower limit of instrumental sensitivity.

Table 3. Comparison of data before and after dredging, in stations L02 and J02

	Sand (%)	Silt&clay (%)	Porosity (%)	C _{org.} (%)	N (%)
LIDO 2 before	97,16	2,84	38	0,05	0,02
LIDO 2 after	95,67	4,33	44	0,08	0,02
JESOLO 2 before	90,81	9,19	41	0,08	0,02
JESOLO 2 after	90,23	9,77	49	0,07	0,02

Core sample analyses (the “after” sample was collected inside the furrow), instead, may demonstrate the rate at which sediments are dispersed, in which only the top layer (0-2 cm) is slowly disturbed at Lido (see Fig. 32), while at Jesolo (Fig. 33) all the top layer (0-8 cm) shows a dredge disturbance, with all the upper sediment being removed; after dredging, at Jesolo the first centimeters of seabed had sediment with a percentage of sand higher than before dredge action.

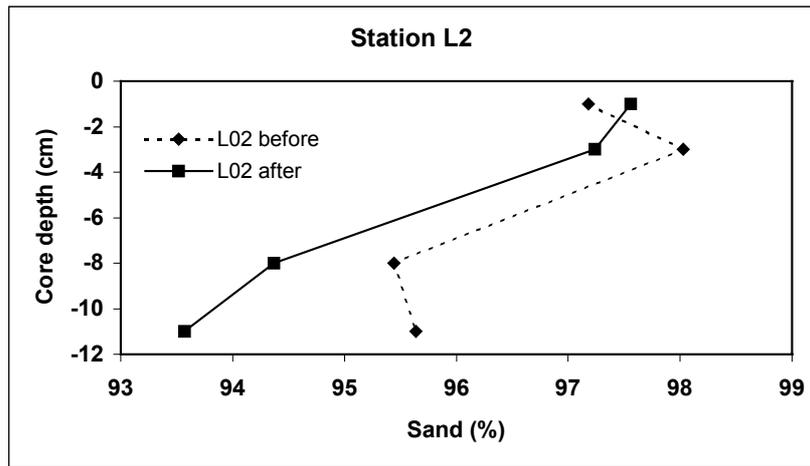


Figure 32: total sands in a core sample collected before and after dredging inside the furrow on station Lido 02

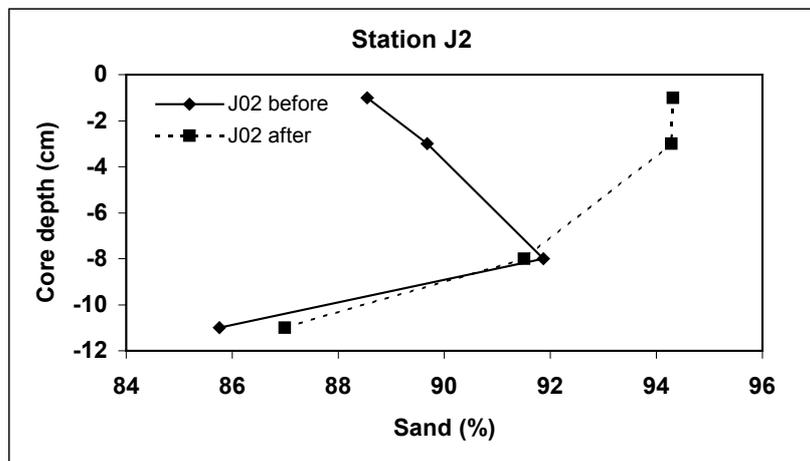


Figure 33: total sands in a core sample collected before and after dredging inside the furrow on station Jesolo 02

Discussion and Conclusions

The results we obtained suggest the following considerations:

7. Observations of the hydrological features of the fishing area, the marked physical variability of the two transects is once more evidenced; the temperature budget measured in the sampling stations, coupled with the river runoff, mainly drives the oceanographic regime of the coastal area. In this context, particular events, such as the exceptional level of salinity up to 38 PSU, detected at the sea bed very close to the coast in winter, could play a further role in adding “natural” stress on benthos communities.

8. The habitat of *C. gallina* can be correctly established by analyzing the sediments grain size and chemical composition. When sediment Eh becomes negative and sandy is replaced by mud, clams tend to disappear.

9. The first attempt to evaluate the modifications induced by dredging between sediment and the overlaying water column produced some good information, but some discrepancies too, mainly due to the different behaviour of the observed parameters.

10. The apparently unchanged environment before and after dredging may be due to different causes, mainly induced by difficulties in sampling:

- it is difficult to sample exactly inside the furrow, sometimes not very easy to find it in a highly disturbed (by fishing) seabed;
- sometimes there are many doubts about the exact position of the furrow, because divers, after fishing, encountered a lot of problems mainly due to the high turbidity.

11. For dissolved nutrients, calculations performed on pore and overlaying water concentrations, to assess the quantity of exchanges between these two matrices, add more useful information on fluxes of chemical parameters due to fishing activity.

12. The morphology and chemistry of the studied seabed does not change very much, because of the high and continuous fishing pressure in almost all the areas.

13. In the location investigated, the coastal Northern Adriatic, we were not able to have a reserved closed area free from fishery disturbance: so it was impossible for us to have a comparison between fished and non-fished areas.

14. Thus finally, the differences we detected between before and after-dredging modifications have to be considered tentative, mainly because of the impossibility to find a relatively undisturbed sediment.

Conclusions

South coast of Portugal

1. Studies were carried out on fishing grounds on which fishermen exploit *Spisula solida*, *Donax trunculus*, and *Ensis siliqua*. The habitats were typical of these species on this coast.
2. Physical parameters reflect a slight effect of dredging activity in near bottom water. Turbidity increase sharply during turnover and within few minutes reaches the control value. The rapid sedimentation of re-suspended particles and/or its transport by currents may explain the results obtained.
3. The microphytobenthos is overturned by mobile fishing gear being accumulated in the ridge formed. Benthic particles slump down the sides of furrow and the recovery of the appearance of the seabed is reached nearly 24 hours after dredging.
4. The nutrients (ammonium, nitrates, organic nitrogen, phosphates and silicates) decrease in the pore water after sediment turnover being exported to the water column.
5. The benthic fluxes of nutrients decrease during and after turnover indicating that sediment function as a source of nutrients during turnover however, while the re-establishment of seabed is not reached the sediment has a lower contribute as source of nutrients. The results point to a seabed recovery in around a day.
6. The physical effect caused by dredging operations implies an oxidation in the upper sediment layers. The phosphate is retained in the first layers of sediment through adsorption on to iron oxides during dredging operations. Sediments have a high and instantaneous capacity to adsorb phosphorus when clouds of particles are formed hindering the phosphate release to water column.
7. When the seabed is disturbed by mobile fishing gears nitrogen is release to water column and phosphate is retained in the upper sediment layers. Thus, some changes in N/P stoichiometry may be expected from this fishing activity if large areas were frequently disturbed.

Northern Adriatic Sea

1. The habitat of *C. gallina* can be established by analyzing the sediment grain size and chemical composition. When sediment Eh becomes negative and sand is replaced by mud, clams tend to disappear.

Modifications in the hydrochemistry of the water column after dredging:

2. The increase of turbidity: after the initial perturbation, the system return to the original conditions in 1h (Lido) - 1.5 h (Jesolo), according to the sediment quality and tidal current.
3. The ratio between chlorophyll degradation products/alive chlorophyll, that increases after dredging, due to the resuspension of dead phytoplankton as well as phytobenthos cells.
4. Transparency decreased on the whole water column due to fishing activity, after which it increased again; 2 hours afterwards, values higher than the starting ones were measured, due to more transparent water advected laterally by high tide.
5. Particle size distribution and frequency, as detected by Coulter Counter, showed an increase of smaller fractions (less than 10 μm) until 60minutes after dredging, then a decrease. A new increase at 150minutes appeared (second oval at right), not explainable by transmittance data, but probably due to current.
6. All the dissolved nutrients do not reveal any significant changes in the water column due to dredging while, particulate organic carbon (POC) and chlorophyll/phaeopigment ratios seem to be the best indicators of dredging activity.

Modifications in pore water and sediments:

7. Ammonia showed a different behavior: in the edge, dredging released this nutrient into the water column, while in the furrow its concentrations increased in pore waters; short-time trends are different too, with a slight increase at the edge and a decrease in the furrow.
8. The oxidized forms of nitrogen (nitrite plus nitrate, NO_x) seemed to behave similarly to ammonia: a sharp decrease in pore waters has been measured, followed by a slight increase. 150minutes do not seem to be enough to complete the recovery time of these forms of nutrients.
9. For silicates, 150minutes was a sufficient time to recover the initial concentrations.
10. Chlorophyll concentrations showed a general decrease after dredging, but a different trend was observed in sediments: at the edge, the starting conditions were reached after 60minutes, while in the track the process of recovery took more time (150minutes). As regards chlorophyll degradation products, their trend in furrow and track is in accordance: the decrease in concentration after 20minutes

continued more markedly 60minutes after dredging, increasing then until reaching the starting values, except in the furrow, where it was higher than the initial levels at 150 minutes.

3.1.2.4 SUMMARY STATEMENT

The observed decrease of benthic fluxes of nutrients during and after turnover due to dredging indicates that sediments function as a source of nutrients during this turnover. Thus, it is expected that phytoplankton primary production may increase during dredging activity if it is controlled by nutrients. However, the retention of phosphate in the upper sediment layers through sorption into iron oxides observed during and after sediment turnover hinders for the most part of its release to water column. If large areas of seabed are frequently disturbed by mobile fishing gears implying nitrogen release to water column and phosphate retention in the upper sediment layers some changes in the N/P stoichiometry may occur from this fishing activity and the expected increase of primary productivity may be negligible.

The physical and chemical effects caused by dredging on the seabed described in this study not point to large effects along the narrow strip of shallow waters (3, 7, 10 and 12 metres depth) and sandy bottoms where the studies were carried out. The instability created at the seabed by its exposure to winds, waves and tidal currents may contribute to the rapid restoration of the sediment characteristics in shallow waters. Thus, as bivalve fisheries on the south coast of Portugal and Adriatic sea are only undertaken in shallow waters with sandy bottoms submitted to natural disturbance, we may expect that dredging does not cause an important impact on chemical parameters.

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3.1.3. Biological – Individual Physiological and Behavioural Studies

Introduction

The eight contributions in this Section of the report are concerned with impacts of shellfish dredging on the target species (scallops and clams) and the implications for survival of these discarded animals. Together these eight contributions highlight the impact of three different types of dredging on three different target species in three different geographical locations around Europe.

The first four papers deal with scallops in the Irish Sea and west coast of Ireland and the first paper ([Section 3.1.3.1](#)) assessed the effects of simulated dredging on the timing and magnitude of the swimming escape response in scallops and tested the general hypothesis that dredging has a deleterious effect on the ability of *P. maximus* to escape from predators. The next three papers ([Sections 3.1.3.2 to 3.1.3.4](#)) used Adenylic Energetic Charge (AEC) and behaviour as acute indicators to measure stress in undersized scallops. They used these techniques to measure the effects of dredging in scallops for different; grounds, seasons, sized animals, tow lengths, tow speeds, emersion periods following dredging and repeated dredging (to investigate a cumulative stress response).

The next two papers ([3.1.3.5 and 3.1.3.6](#)) deal with the acute and chronic responses of the striped venus clam *Chamelea gallina* to hydraulic dredging. Samples were collected in two areas along the North Adriatic coast using different fishing approaches and at different times of the year. The acute indices used can be divided in to immunological (hematocrit value, phagocytic index and enzyme activity indices) biochemical (AEC) and behavioural (reburrowing ability). The chronic indices used included; survival in air, filtration rate, respiration rate, absorption efficiency, scope for growth, condition index, catalase, protein content, latency lysosomal enzyme, RNA/DNA ratio and nucleic acid contents.

The final two papers ([3.1.3.7 and 3.1.3.8](#)) deal with the impact of dredging in the clam *Spisula solida* in Southern Portugal. The response of this clam under stress imposed by dredging was analysed in terms of reburrowing and different indices of biochemical condition (acute: AEC ATP/mg dry weight; chronic: RNA/DNA ratio, neutral/polar lipids ratio). The impact of dredging in different seasons and sites was investigated and the changes in the reburial time of the clams caused by fishing disturbance and its consequence to indirect mortality was discussed.

3.1.3.1 THE EFFECT OF DREDGE CAPTURE ON THE ESCAPE RESPONSE OF THE GREAT SCALLOP, *PECTEN MAXIMUS* (L.): IMPLICATIONS FOR THE SURVIVAL OF UNDERSIZED DISCARDS

Introduction

In most fisheries, a proportion of the captured target species will be discarded because of their small size. Many studies have shown that damage, resulting from capture or encounter with fishing gear, can cause high levels of mortality (see Chopin and Arimoto 1995; Kaiser and Spencer 1996a). However, the effects of stress or slight damage caused by encounter with gear on survival rates are not at all clear. In common with many types of fishing gear, scallop dredges typically catch large numbers of undersized individuals of the target species. For example in the Irish Sea, 'Newhaven' spring toothed scallop dredges catch an average of 35% of scallops under the minimum legal landing size of 110mm shell length. The fate of these undersized discards, the majority of which are undamaged, when returned to the sea bed is of obvious importance to the long term sustainability of exploited populations.

Animals which are disturbed or damaged by demersal fishing gear on the seabed, and those which are captured and subsequently discarded, attract mobile predators and scavengers (Kaiser and Spencer 1994; Kaiser et al. 1996; Kaiser and Spencer 1996a). It is likely, therefore, that undamaged scallops, discarded along with large quantities of damaged invertebrate by-catch (Veale et al. 2001), will be subject to high levels of predator activity. Scallops show a characteristic escape response to predators (Brand 1991), reacting by either closing the shell, jumping or swimming. Shepard and Auster (1991) speculated that the swimming ability of scallops may be impaired by the passage of a dredge. A reduction in the scallop's ability to escape predators has important implications for survival of undersized discards.

We assessed the effect of simulated dredging on the timing and magnitude of the swimming escape response in the great scallop, *Pecten maximus* to test the general hypothesis that dredging has a deleterious effect on the ability of *P. maximus* to escape from predators.

Methodologies

Collection and maintenance in the laboratory

Scallops beneath the minimum legal landing size of 110mm shell length were collected using spring toothed scallop dredges from fishing grounds off the Isle of Man, British Isles. They were collected from two locations, the 'Chickens' scallop ground (53° 59' N 4° 53' W) and the Laxey Bay scallop ground (53° 14' N 4° 21' W). After sorting on deck, scallops were placed in seawater and transported to Port Erin Marine Laboratory, where they were maintained indoors in tanks (1m x 1m x 0.4m) with fresh running seawater. The animals were not given supplementary food but were used in experiments within 3 weeks of collection. All scallops were given at least 10 days to recover from the dredging process before use in experiments.

Assessment of swimming ability

A preliminary experiment was carried out to determine the relationship between the number of valve adductions and the distance travelled by *Pecten maximus* (size range

90-110mm) during swimming. A single scallop was placed in the centre of a large circular tank (depth 0.9m, diameter 2.8m) and stimulated to swim by touching the tube feet of a single arm of the starfish *Asterias rubens* onto the scallop's mantle edge. The distance moved in a single bout of swimming from the tank centre was measured and the number of valve adductions counted. The scallop was placed back in the centre of the tank and stimulated to swim again. Scallops were used in this way until swimming responses ceased. At least 6 observations of each adduction number, between 1 and 14, were made.

General experimental procedure

In order to assess the effect of dredge disturbance on the swimming escape response of scallops, we took the same approach as previous studies of scallop swimming (Thomas and Gruffydd 1971; Stephens and Boyle 1978). The lower (right) valve of each scallop was attached to the substratum to prevent movement of the animal whilst performing valve adductions. This was achieved by gluing a nylon nut to the lower valve of each scallop using Araldite RapidTM adhesive. During the gluing procedure scallops were exposed to the air for a maximum of 10 minutes. The head of a nylon bolt was glued to a brick so that the scallop could be quickly and easily screwed to the bolt and immobilised. Scallops were given at least 3 days to recover after the gluing procedure.

A system of 6 circular experimental tanks (diameter 0.38m x depth 0.28m), each with a brick and bolt for immobilising a single scallop, were set up with running seawater. The experimental procedure consisted of removing a set of scallops from the storage tanks, subjecting them to the appropriate treatment (see below) and then quickly screwing them to the single brick in each experimental tank. This could be achieved underwater to avoid additional stress. Once immobilised in the experimental tanks, all scallops, whatever the experimental treatment, were given 15 minutes to recover from the handling process. This time was chosen as it allowed even the most stressed scallops time to open their valves. After 15 minutes scallops were stimulated to 'escape' using the predator *Asterias rubens*. A number of individuals of this species, of similar size (15-18 cm diameter), were maintained in seawater tanks, and a different individual used for each set of scallops. To stimulate an escape response, a starfish was introduced to a scallop, so that the tube feet of a single arm touched the mantle edge. Care was taken to avoid creating shadows and displacing water, which could cause the scallop to close its valves before the starfish could be applied. The starfish arm was left in place until a response was initiated. The time from the first stimulation until an escape response was performed was recorded using a stopwatch, and the number of valve adductions counted. After valve adductions had ceased, the scallop was again stimulated in the same way, and the number of adductions counted. This was continued until the valves closed firmly or until there was no response after 60 seconds.

In this way a single response time (the time from the first stimulation until an escape response was performed) was recorded, plus a series of adductions. Adductions were either in the form of single valve claps, or rapidly repeated valve claps. Preliminary observations on swimming, described above, showed that a high total number of adductions could be made, either in the form of a few bouts of sustained valve claps (swimming), or by numerous single adductions (jumping). It is known that the jumping response occurs in bouts of between 1 and 3 valve claps (Brand 1991). In

order to differentiate between the swimming response and the jumping response, the percentage of the total number of valve claps which occurred in bouts of four or more was calculated. This separation between 3 and 4 valve adductions was justified by the results of the preliminary swimming experiment which showed a doubling in the distance travelled, from 20 cm to 40cm, in scallops which exhibited 4 valve claps compared with 3 (Figure 1).

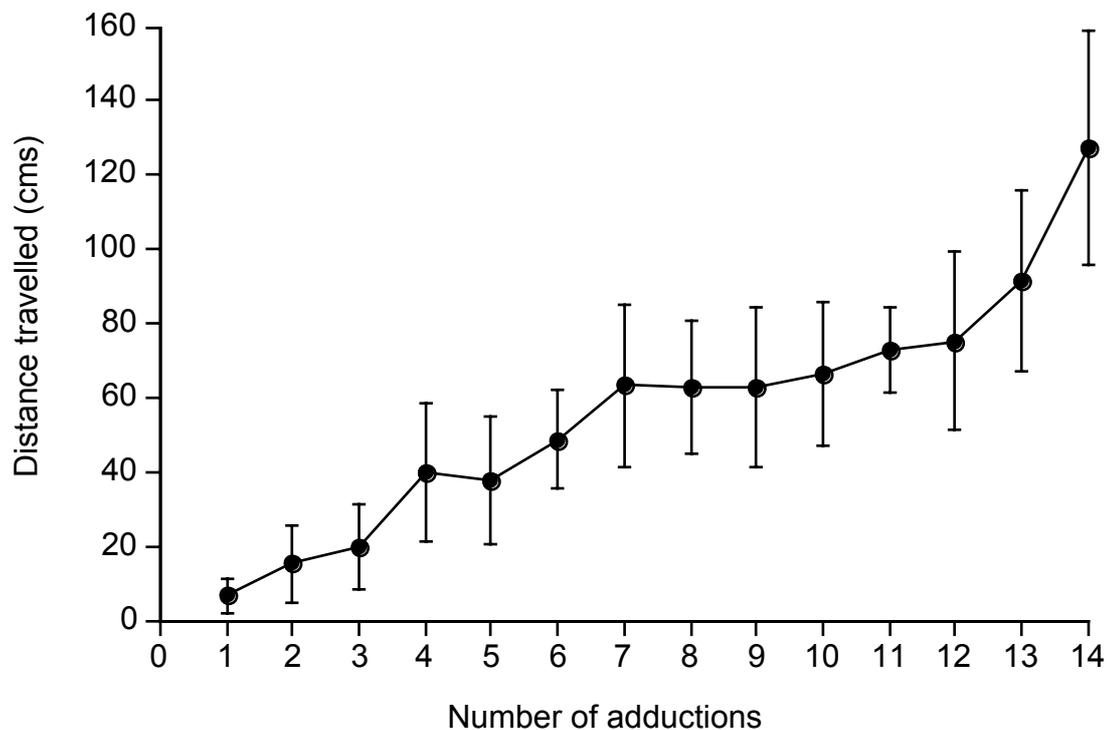


Figure 1. The relationship between the number of consecutive valve adductions performed by *Pecten maximus* during an ‘escape response’ and the distance travelled over the ground. Error bars = \pm 1SD

Simulation of dredging

In order to simulate the effects of dredging on scallops caught in a typical dredge used in the Irish Sea, a nylon mesh bag (30 x 50cm) was prepared containing an assortment of stones varying in diameter from 6-12cm and with a total weight of 7 kg. Between four and seven scallops (depending on the experiment) were placed in the bag, which was then placed in a 60l tank of seawater containing a thin layer of sand. The bag was attached via two overhead pulleys to a 22 cm arm attached to the axle of an electric motor, which turned at the rate of 38 times per minute. Action of the motor caused the arm to turn, which caused the bag of scallops and stones to lift approximately 10cm and to fall under its own weight onto the base of the tank. Throughout the process the scallops were continuously submerged. For all experiments, scallops were ‘dredged’ for 40 minutes. This time period represents the lower end of the range of tow durations for scallop fishing boats in the north Irish Sea.

Experiment 1: Effect of dredging in different seasons and on different sized scallops.

Scallops were collected from the 'Chickens' scallop ground on March 15th, 2000, and September 12th, 2000. These dates occur during periods of minimum and maximum water temperatures in the north Irish Sea. The mean temperature of the water in laboratory tanks at the time of experimentation was 8°C (range: 7.4-10°C) in the spring and 15°C (range 15-16°C) in the autumn. Two size classes of undersized scallops were collected, 80-95mm and 105-110mm shell length. For each size class of scallop there were two treatments, non-dredged and dredged, with each size class-treatment combination replicated five times. Each replicate consisted of 4 scallops which were treated as sub-samples and averaged to give a single value.

Experiment 2: Determination of the time taken for scallops to recover following dredging

Undersized scallops (80-110 mm) were collected from the 'Chickens' scallop ground on April 28th, 2000. The experiment consisted of seven experimental treatments and one control. Scallops underwent simulated dredging for 40 minutes and were then left to recover in holding tanks with running seawater for varying lengths of time: zero, one, two, four, six, eight, and twenty-four hours. In addition, there was a single control treatment in which no dredging took place. For each of the eight treatments, four replicates were used, with each replicate consisting of seven scallops.

Experiment 3: Determination of the interactive effects of dredging, exposure to air and recovery time

Undersized scallops (80-110 mm shell length) were collected from Laxey Bay on November 2nd, 2000. A three-way factorial experiment was designed to investigate the interactive effects of dredging, exposure to air (hereafter termed desiccation) and recovery time. All three factors had two levels, presence and absence, resulting in eight orthogonal treatments which were replicated four times. Five scallops were used for each replicate. Simulated dredging was undertaken for 40 minutes, whilst scallops were exposed to the air for 20 minutes at a temperature of 10-13 °C. The recovery period was either zero or one hour.

Data analysis

Data were analysed using ANOVA. Prior to using ANOVA, Cochran's test was used to test for heterogeneity of variance. Multiple comparisons of levels within significant factors were made using Student Newman Keuls (SNK) tests.

Results

General observations of swimming and adduction patterns

There was a large degree of variability in the swimming response of scallops within a single treatment. For example, in control individuals the total number of adductions performed ranged from 1-38, whilst the response time ranged from 1-58 s. The total number of valve adductions performed in one bout ranged from 1 to a maximum of 25. In dredged scallops, although the swimming response was typically very low, some scallops performed very well, with a maximum of 18 adductions in total. Dredged scallops frequently took long periods to respond at each stimulation, and the adductions were often relatively weak, with a shallow gape and slow valve movements. Adductions judged to be too weak to move the scallop were ignored.

Experiment 1

Simulated dredging caused a significant increase in the time taken for scallops to respond to the predator stimulus for both large and small scallops, in both spring and autumn (Figure 2, Table 1). The average response time for all scallops tested was 8 seconds in non-dredged scallops, compared with 22 seconds in dredged scallops. There was a significant effect of the factor size, with large scallops (105-110mm) being slower to respond than small scallops (80-95mm) (Figure 2, Table 1).

The effect of simulated dredging on the number of adductions shown after the first stimulation by *Asterias rubens* ('first adduction') was dependent on both scallop size and season, as shown by the significant three way interaction (Table 1). Adduction number was significantly reduced for all treatments, except in large scallops in the spring. In all dredged scallops only 3 individuals out of 80 showed more than one adduction in their first response. There was a clear difference in the number of adductions between seasons in control scallops, with a significantly higher number in the autumn (Figure 2).

The total number of adductions was significantly lower in scallops that had undergone simulated dredging (Figure 2, Table 1). This was true irrespective of size or season. The mean number of adductions was 17 in controls, compared with only 4 in dredged animals. There was no difference in adduction number between seasons or sizes. The percentage of the total number of adductions which occurred as a swim response (in bouts greater than three valve claps) was significantly lower in dredged scallops (Figure 2, Table 1). In addition, there was a greater proportion of swimming bouts in the autumn compared with spring.

Table 1 ANOVA of four response variables in experiment 1 carried out in spring and autumn 2000.

	Source	df	MS	F	P
Response time	Dredge	1	31.71	38.2	<0.001
	Season	1	1.06	1.27	>0.25
	Size	1	6.59	7.94	<0.01
	Dredge x Season	1	1.58	1.91	>0.15
	Dredge x Size	1	0.42	0.51	>0.4
	Season x Size	1	1.65	1.99	>0.15
	Dredge x Season x Size	1	0.21	0.25	>0.6
	Residual	32	0.83		
First adduction	Dredge	1	17.78	645.79	<0.001
	Season	1	6.27	227.80	<0.001
	Size	1	0.57	20.64	<0.001
	Dredge x Season	1	7.34	266.83	<0.001
	Dredge x Size	1	0.29	10.90	<0.01
	Season x Size	1	0.84	30.39	<0.001
	Dredge x Season x Size	1	0.36	13.21	<0.001
	Residual	32	0.03		
Total adductions	Dredge	1	1565.63	114.54	<0.001
	Season	1	24.94	1.82	>0.15
	Size	1	6.88	0.50	>0.4
	Dredge x Season	1	45.33	3.32	>0.05
	Dredge x Size	1	2.80	0.21	>0.6
	Season x Size	1	15.52	1.14	>0.25
	Dredge x Season x Size	1	24.15	1.77	>0.15
	Residual	32	13.67		
% swim	Dredge	1	49799.06	174.77	<0.001
	Season	1	2872.25	10.08	<0.01
	Size	1	112.02	0.39	>0.5
	Dredge x Season	1	1164.77	4.09	>0.05
	Dredge x Size	1	30.38	0.11	>0.7
	Season x Size	1	368.41	1.29	>0.25
	Dredge x Season x Size	1	19.10	0.07	>0.7
	Residual	32	0.83		

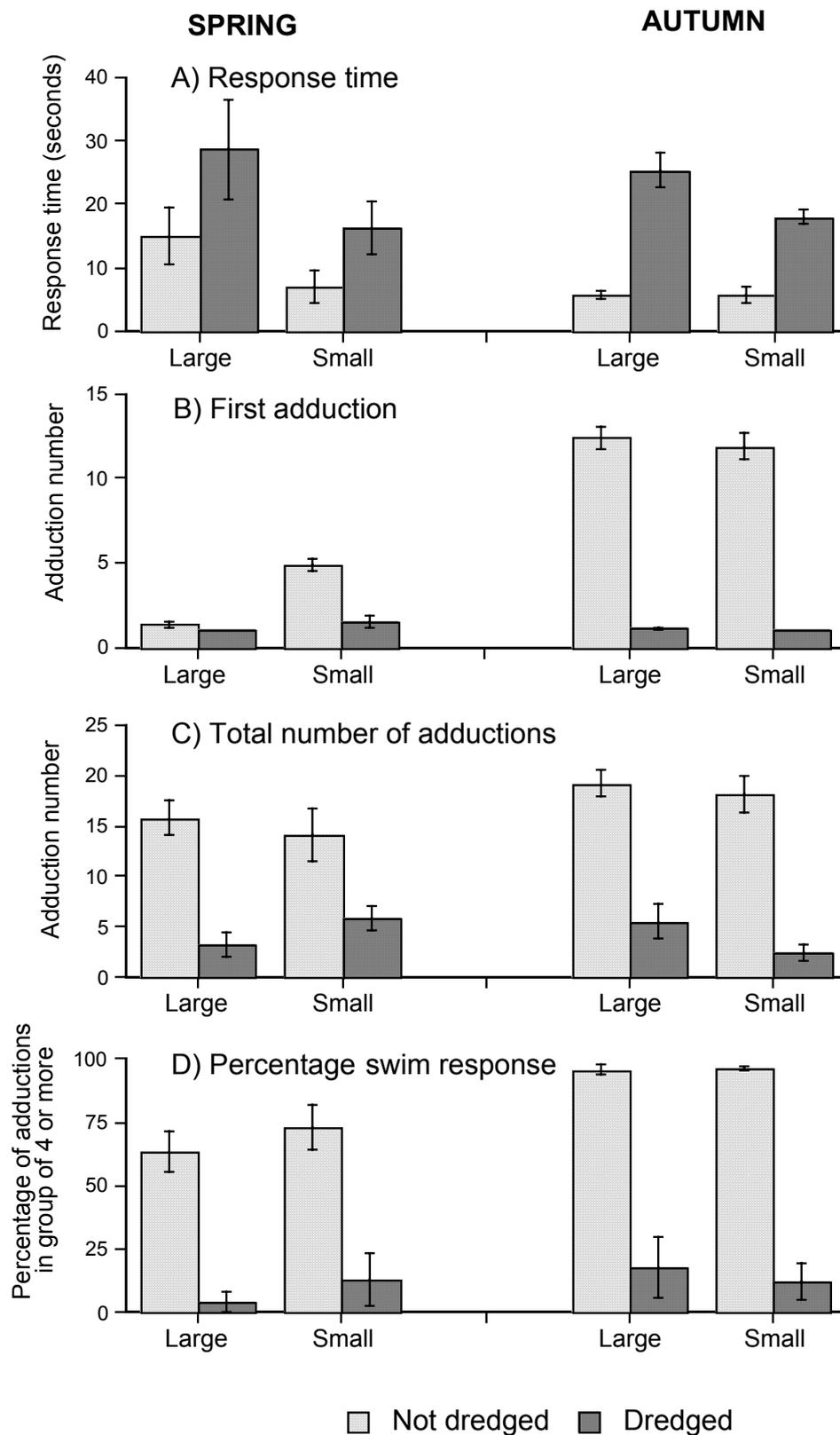


Figure 2. The effect of dredge simulation on the responses of ‘large’ (105-110mm shell length) and ‘small’ (80-95mm shell length) *Pecten maximus* following stimulation by the predatory starfish *Asterias rubens* in spring and autumn 2000. Error bars = \pm 1SE.

Experiment 2

For all four variables measured there was a significant effect of dredging; response time increased from a mean of 9 seconds to 40 seconds, the mean number of adductions in the first response decreased from 10.3 to 1, the total number of adductions declined from 21.6 to 4.6, while the percentage of adductions which occurred as a swim response declined from 92% to 13% (Figure 3). The rate of recovery after simulated dredging varied depending on the response variable. Both the response time and the percentage of adductions performed as a swim response showed no significant difference to control levels after only one hour (Figure 3, Table 2). However, both the number of adductions in the first response and the total number of adductions were still lower than control levels after 24 hours of recovery. The total number of adductions showed a clear pattern, with a partial recovery after only one hour, but no further recovery over the following twenty-three hours (Figure 3, Table 2). The number of adductions in the first response showed no recovery eight hours after dredging. After 24 hours the number of adductions was significantly greater than at zero hours but was still lower than the control scallops.

Experiment 3

The escape response of scallops was tested immediately (zero hours) and one hour after experimental treatments. Scallops showed a similar response to the effect of simulated dredging and exposure to air at both times, indicated by the non-significant dredge x time and desiccation x time interactions (Figure 4, Table 3). Thus, after one hour, scallops were still negatively affected by the treatments. Simulated dredging had a significant negative effect on all 4 response variables (Figure 4, Table 3). Exposure to air had a negative effect on all response variables except the number of adductions in the first response. We had hypothesised that there would be no effect of exposure to air in dredged scallops. This would be expressed as a significant dredge x desiccation interaction. However, this interaction was non-significant; there was a negative effect of desiccation for both dredged and non-dredged scallops. For all response variables, at both zero hours and one hour, those scallops subject to both dredging and exposure to air were the most negatively affected (Figure 4).

Table 2 ANOVA of four response variables in experiment 2. The factor treatment refers to comparison between the control and 7 dredged treatments with differing periods of recovery.

	Source	df	MS	F	P
Response time					
Transform: Ln (x+1)	Treatment	7	0.66	3.89	<0.01
C = 0.2708	Residual	24	0.17		
SNK test:	Zero > All other treatments				
First adduction					
Transform: None	Treatment	7	37.29	17.05	<0.001
C = 0.2950	Residual	24	2.19		
SNK test:	Control > All other treatments Zero and One < Twenty-four				
Total adductions					
Transform: None	Treatment	7	93.16	7.48	<0.001
C = 0.3458	Residual	24	12.45		
SNK test:	Control > All other treatments Zero < One, Six, Eight, twenty-four				
% SWIM					
Transform: None	Treatment	7	2355.65	9.33	<0.001
C = 0.2436	Residual	24	252.36		
SNK test:	Zero < All other treatments				

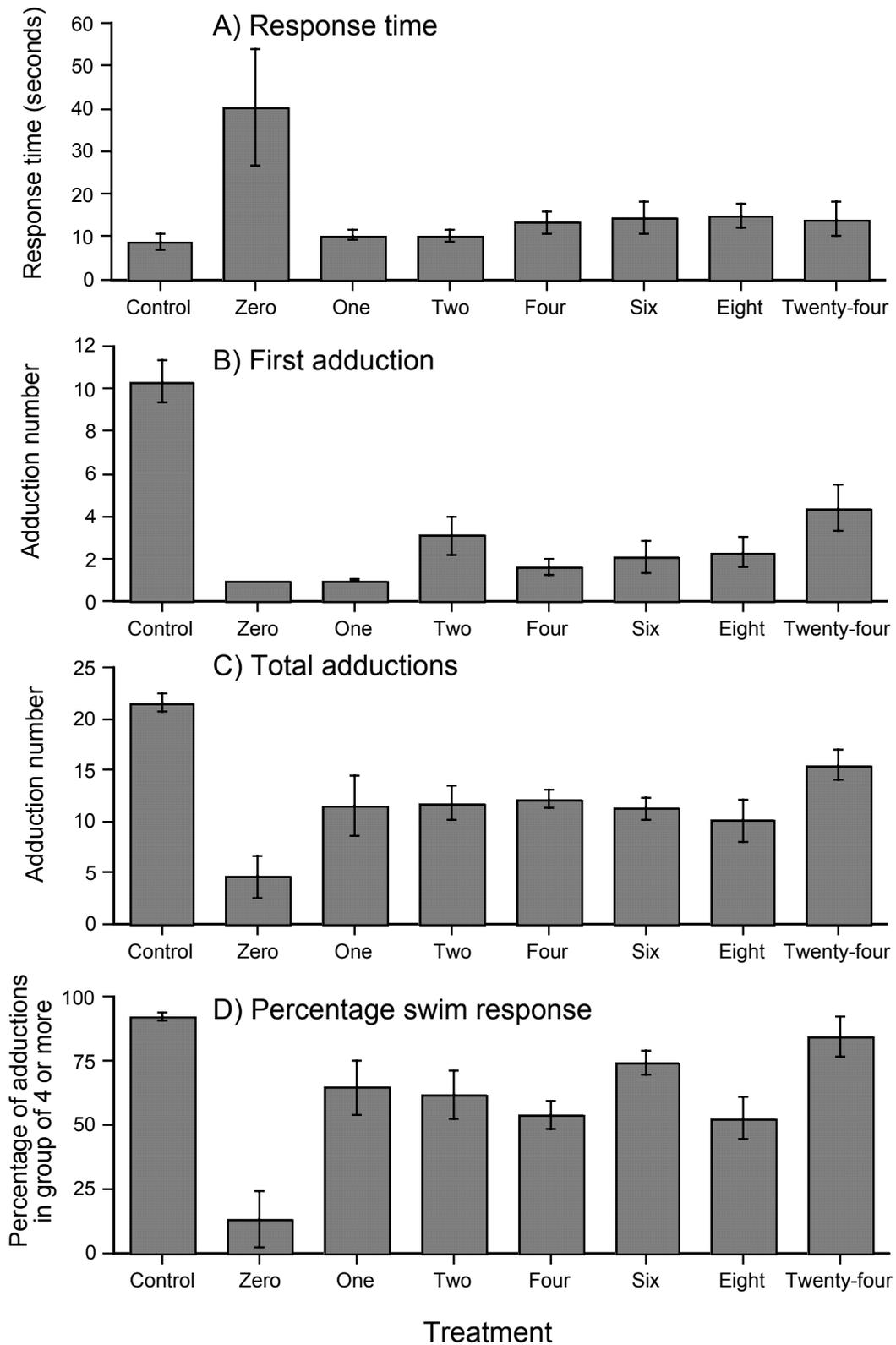


Figure 3. The time taken to recover the ‘escape response’ in *Pecten maximus* following simulated dredging. All treatments except the control were dredged and given differing periods of time to recover (zero to 24 hours) before stimulation with the predatory starfish *Asterias rubens*. Error bars = ± 1 SE.

Table 3 ANOVA of four response variables in experiment 3.

	Source	df	MS	F	P
Response time	Dredge	1	322.47	8.44	<0.01
	Desiccation	1	182.49	4.78	<0.05
	Time	1	50.54	1.32	>0.25
	Dredge x Desiccation	1	49.96	1.31	>0.25
	Dredge x Time	1	81.87	2.14	>0.15
	Time x Desiccation	1	4.49	0.12	>0.7
	Dredge x Desiccation x Time	1	7.59	0.20	>0.6
	Residual	24	38.21		
First adduction	Dredge	1	12.14	20.67	<0.001
	Desiccation	1	3.13	5.33	<0.05
	Time	1	1.14	1.94	>0.15
	Dredge x Desiccation	1	0.05	0.08	>0.75
	Dredge x Time	1	0.14	0.02	>0.85
	Time x Desiccation	1	0.26	0.44	>0.5
	Dredge x Desiccation x Time	1	0.19	0.33	>0.55
	Residual	24	0.59		
Total adductions	Dredge	1	525.82	26.75	<0.001
	Desiccation	1	411.72	20.95	<0.001
	Time	1	40.46	2.06	>0.15
	Dredge x Desiccation	1	3.39	0.17	>0.6
	Dredge x Time	1	45.56	2.32	>0.1
	Time x Desiccation	1	22.92	1.17	>0.25
	Dredge x Desiccation x Time	1	51.21	2.61	>0.1
	Residual	32	19.66		
% SWIM RESPONSE					
	Dredge	1	26694	96.88	<0.001
	Desiccation	1	1285	4.67	<0.05
	Time	1	419	1.52	>0.2
	Dredge x Desiccation	1	138	0.50	>0.45
	Dredge x Time	1	947	3.44	>0.05
	Time x Desiccation	1	51	0.19	>0.65
	Dredge x Desiccation x Time	1	149	0.54	>0.45
	Residual	32	275		

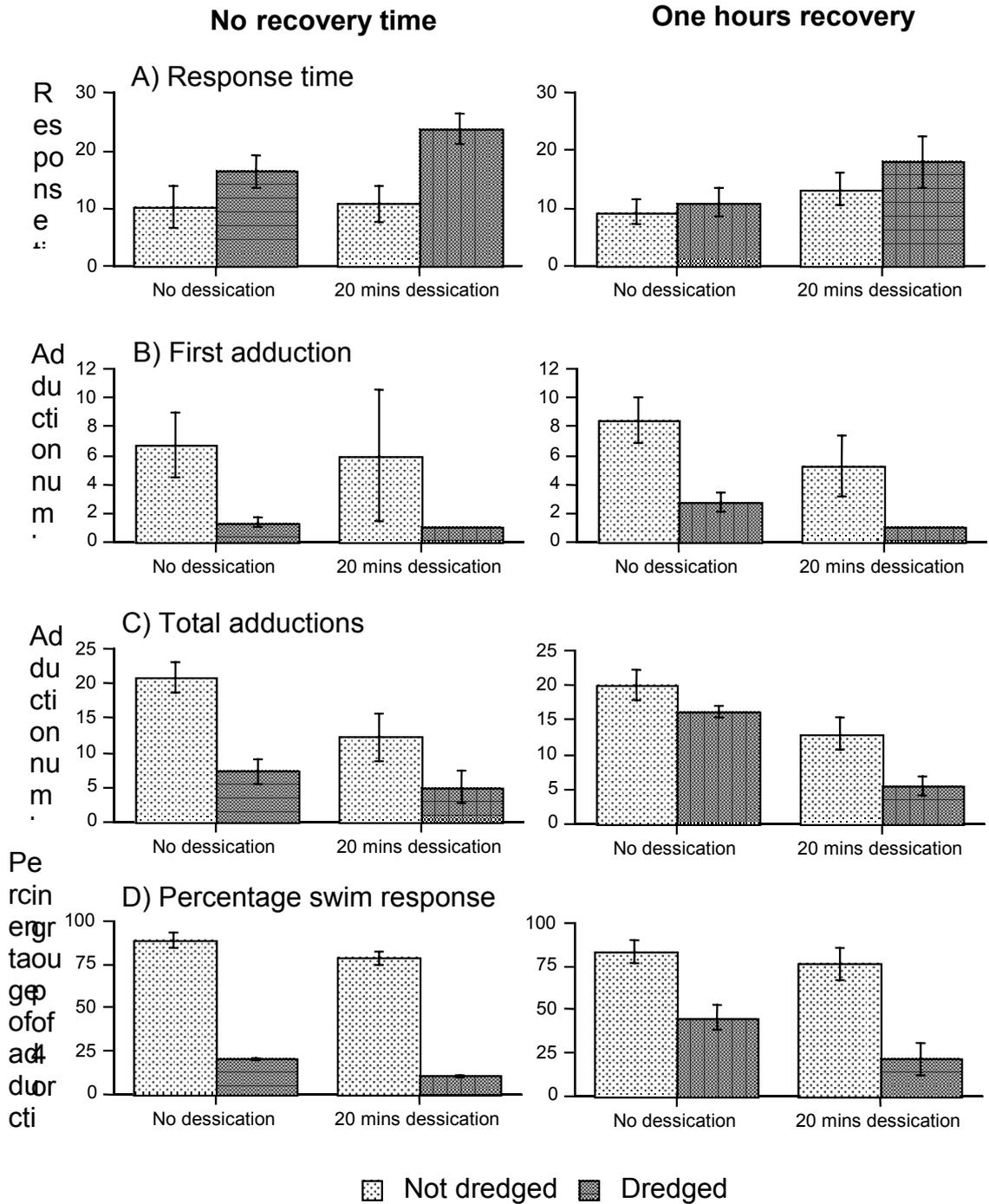


Figure 4. The effect of dredge simulation and exposure to air on the responses of *Pecten maximus* following stimulation by the predatory starfish *Asterias rubens*. *P. maximus* was given both zero and one hours recovery time before stimulation. Error bars = \pm 1SE

Discussion

Simulated dredging (including exposure to air) clearly reduced the ability of *Pecten maximus* to escape from an introduced predator. In *Pecten maximus*, both the increase in response time and the reduction in swimming ability after dredging suggest an increase in vulnerability to predation, particularly from slow moving predators such as starfish and gastropod molluscs. There are numerous factors associated with dredge capture which may cause stress to scallops and a subsequent reduction in swimming ability. The physical impact of dredge teeth at the point of capture, physical impact with rocks during the period within the dredge bag and exposure to high suspended sediment levels are all potential causative factors. Alternatively, scallops may simply be exhausted by attempting to swim whilst in the dredge itself.

The experiment to determine recovery time in *Pecten maximus* clearly showed that a scallop's ability to escape a predator is at its lowest immediately after dredging. A limited recovery was shown after one hour in experiment 2 but at this stage, and for the following 23 hours, scallops were still negatively affected by dredge disturbance. The implications of recovery time to a scallop's chance of survival when returned to the seabed depend on the time taken for predators to aggregate to discarded material. Studies of predator aggregation to dredge and trawl tracks and to discarded material demonstrate that a variety of taxa can aggregate quickly, from a few minutes to hours (Ansell et al. 1988; Kaiser and Spencer 1996a). Thus, it is likely that discarded scallops will be subject to elevated levels of predatory starfish activity within the period during which we have shown a reduction in the ability to escape. In addition, *A. rubens* is relatively resilient to the effects of demersal trawls (Kaiser and Spencer 1996b) and dredges (Veale et al. 2001). Thus, high numbers of starfish, discarded with undersized scallops, may be able to prey on *P. maximus* immediately on return to the seabed.

In conclusion, we have demonstrated a reduction in swimming ability in captured undersized scallops. These data, together with numerous studies of predator aggregation to discarded material, indicate there is the potential for high levels of mortality in undersized discards of *Pecten maximus* and in impacted but uncaptured individuals. There is clearly a need to carry out realistic field-based tests of survival of stressed and damaged organisms impacted by demersal fishing.

3.1.3.2 USE OF AEC AND BEHAVIOUR AS ACUTE INDICATORS OF STRESS IN THE GREAT SCALLOP *PECTEN MAXIMUS*: IMPLICATIONS FOR THE SURVIVAL OF UNDERSIZED SCALLOPS.

Introduction

The main types of obvious sublethal damage caused to scallops during the dredging process include chipped valve margins, separation of the hinges, desiccation and sediment in the mantle cavity. It is common practice for fishermen to return damaged and undersized scallops (the minimum landing size for Ireland is 110mm shell length) to the beds in the belief that they will survive and recruit to the fishery. However the performance of these traumatized juveniles has not, to our knowledge, been previously investigated. It would be reasonable to hypothesize that the longer a scallop had been exposed to disturbances such as “rocking and rolling” in the dredge bag or to desiccation once landed on deck, the more its ability to recover and survive once returned to the sea bed, would be compromised. Shepard and Auster (1991), estimated that the percentage of damaged scallops left behind in the dredge track can be as high as 7%. Undersized scallops returned to the sea have been found to be more susceptible to predation. Caddy (1973) observed that predatory fish and crabs were attracted to dredge tracks within one hour of dredging at densities of up to 30 times greater than in areas outside the tracks. Gruffydd (1972) suggested that this type of mortality is much more difficult to estimate. Therefore, in this study, biochemical and behavioural indices were used to assess the stress effect of dredging on undersized scallops both in the field and in the laboratory.

The role of the adductor muscle

The large adductor muscle is divided into two parts, the striated and the smooth. The striated part acts as the main site for storage of energy reserves and is used when swimming, jumping and re-cessing. The smooth muscle is used to keep the shell closed for long periods of time, for example while resisting predator attack and during exposure to air. Glycogen reserves are accumulated seasonally during optimal periods and are then depleted to contribute to metabolic requirements for maintenance or gonadal development. The timing of events throughout the cycle varies but, in general, glycogen content increases during times of optimal temperature and food availability in the spring and summer to reach a peak in the autumn and decreases during the winter (Ansell 1978; Ansell et al. 1988).

Adenylic Energetic Charge (AEC)

Maguire et al. (1999c) assessed the usefulness of various techniques for stress assessment in scallops and found and behaviour to be the most effective in the measurement of acute stress. Other techniques commonly used are condition index (Rogan et al. 1991), carbohydrate content (Kaufmann et al. 1994), total oxyradical scavenging capacity (Regoli et al. 1998), RNA:DNA (Lodeiros et al. 1996), lysosomal membrane fragility (Pelletier et al. 1991) and oxygen consumption and ammonia excretion (Cheung and Cheung 1995), however these tests are only useful for the assessment of chronic stress. AEC was first proposed as a stress index by Atkinson (1968), who suggested that modulations in the levels of adenylyl-phosphate reflected variations of enzyme activity at key points in metabolic pathways that yield

energy in the form of high energy adenine-phosphate bonds. These variations are a result of external stress. In other words, the more stressed an animal becomes, the more energy it uses to counteract the stress thus lowering its AEC level.

AEC is defined by the ratio: $AEC = (ATP + 0.5ADP) \div (ATP + ADP + AMP)$

Where: ATP = adenosine tri-phosphate, ADP = adenosine di-phosphate and AMP = adenosine mono-phosphate. The tri-phosphate bond of the ATP molecule has maximum energy, the di-phosphate bond of ADP is half as rich and the mono-phosphate bond (AMP) lacks energy. The AEC ratio ranges from 0 to 1 i.e. 0 (all nucleotides are AMP) and 1 (all nucleotides are ATP). Therefore the level of these bonds can be used as a measure of the energy directly available to the cells at that particular time. High AEC levels (>0.8) have been found in organisms living in optimal conditions where the animals were growing and reproducing. Levels between 0.5 and 0.7 have been found in organisms whose environment was limiting in some way, such animals had reduced growth rates and did not reproduce, but recovered when returned to optimal conditions. Organisms whose AEC levels were less than 0.55 had a negative scope for growth and did not recover (Ivanovici 1980). Many studies have been carried out using AEC as a stress index and in nutritional studies on different bivalves e.g. the mussels *Mytilus edulis* (Wijsman 1976a) and *M. galloprovincialis* (Isani et al. 1997), the oysters *Crassostrea gigas* (Moal et al. 1989b), the oyster *Crassostrea angulata* (Madureira et al. 1993.) and the scallop *P. maximus* (Duncan 1993, Fleury 1997, Maguire et al. 1999a; Maguire et al. 1999b). Maguire et al. 1999c) conducted an experiment to assess the effect of stress on the AEC level of both the smooth and striated muscle of the king scallop. A greater significant AEC decline was shown between treatments in the striated muscle. Thus the AEC level in the striated muscle was used in these investigations.

Behavioural tests

Scallops can swim relatively long distances in an oriented way and can recess into the sediment. These activities require a large energetic cost, and scallops already stressed from dredging would be less able to escape from predators by recessing or swimming when returned to the sea (Thompson et al. 1980). The energy demand for swimming in scallops has been recorded as $0.6\mu\text{moles ATPg}^{-1}$ by Gäde et al. (1978), who showed that swimming exhaustion occurred after fifteen seconds at three claps per second, and who suggested that cessation of swimming was due to exhaustion rather than habituation to the stimulus. Many studies have examined the relationship between AEC levels and certain behavioural traits in bivalves e.g. *P. maximus* (Fleury et al. 1997; Maguire et al. 1999b) and the Baltic clam *Macoma balthica* (Sokolowski et al. 1999). Subsequently behavioural changes in bivalves have been used as a stress detectors e.g. the recessing and righting ability of the scallops *P. maximus* (Maguire et al. 1999c; Minchin et al. 2000) and *Mizuhopecten yessoensis* (Tyurin 1991.), burrowing in the marine clam *Venerupis decussates* (Stephenson and Taylor 1975) and shell closure in the mussel *Mytilus edulis* (Kramer et al. 1989).

Dredge simulator

A dredge simulator was used in these experiments to eliminate certain variables e.g. ground type and weather conditions that field studies may create. During the exploratory phase of these experiments an instrumented scallop was developed that

measured acceleration and angle of pitch and roll of scallops in the dredge bag. Measurements were taken every 0.1 seconds for 10 minutes. The measurements taken from a “real” dredge were compared to those taken from the simulator (Section [3.1.1.3](#)). The simulator was fine tuned until there was no difference in the acceleration and angle of pitch and roll between the “real” and simulated dredge. A speed of 22 rpm in the simulator equated to 2.5 knots in a “real” dredge. In this paper we will show that the AEC levels in scallops sampled from a dredge were similar to those sampled from the dredge simulator for the same time period.

In this paper we looked at the effect of dredging on both AEC levels and the recessing and righting ability of undersized scallops. Both laboratory (using a dredge simulator) and field experiments were carried out to examine the stress effect of different dredge tow lengths, scallop sizes and emersion periods.

Materials and Methods

The data presented in this paper comes from two field trials and four laboratory experiments.

Field Experiments

The field trials were carried out on commercial scallop grounds off Valentia Island, Co. Kerry, SW Ireland, (51.56°N, 10.18°W) and the Chickens ground, Isle of Man in the North Irish Sea (54.05°N, 04.38°W). The scallops were caught at both sites using a spring-loaded dredge (Width – 75cm) with a toothed cross bar (tooth spacing – 66mm; tooth length – 100mm), a collecting bag made from case hardened 65mm x 8mm steel rings (Diameter – 70mm) and a mesh bag (Mesh Size – 100mm).

Valentia

The two dredges were towed at a speed of one knot for fifteen minutes at the Valentia site in December 1999. At the end of each tow 15 scallops (shell length 99.1 ± 10.09 mm) from each dredge were sampled for AEC analysis. Three tows in total were carried out. One hundred and twenty scallops were removed from the dredge and placed in polystyrene boxes which contained seaweed and transported by van for 3 hours. After transport a further 30 scallops were sampled for AEC analysis and the remainder were placed in 1m depth round tanks containing flowing seawater where recovery was monitored. 30 scallops were sampled after 24 and 48 hours and after one week.

Chickens Ground

At the Isle of Man site eight dredges were towed at 2.5 knots for thirty minutes in August 1999. At the end of each tow 30 scallops (shell length 102.3 ± 15.59 mm) were taken from three random dredges and sampled for AEC analysis. Six tows in total were carried out. 17 scallops were removed from the dredge, placed in tanks containing seawater and allowed to recover for one week, before being sampled. These animals were used as a control.

Laboratory Experiments

All scallops used in the laboratory experiments were undersize (<110mm shell length) and were collected from Valentia Island, Co. Kerry. The experiments took

place between January and April 2000. The animals were acclimated in seawater tanks for one week at a temperature of 10°C and a salinity of 35‰. For each experiment five replicates of four scallops were used per treatment. Each replicate was a separate dredge event. After dredge simulation ten scallops per treatment (2 from each dredge event) were sampled for AEC analysis and behaviour was monitored for the remaining 10 animals. Recovery was monitored biochemically, after 72 hours, 10 adductor muscle samples from these scallops were taken for AEC analysis.

Description of dredge simulator

The simulator was a meat turner manufactured by Hans Lenze, Bosingfeld/Lippe. It had a stainless steel barrel with a diameter of 60cm and a depth of 80cm and was powered by two motors each 3.8/220 Volts with 1.15/2 Amps. During dredge simulation experiments it was rotated at an angle of 45° at a constant speed of 22rpm. Preliminary results had shown that the AEC levels in scallops sampled from a dredge were similar to those sampled from the dredge simulator using those settings for angle and speed for the same time period. The simulator was filled with seawater and small rocks, stones and sand in order to mimic conditions in a scallop dredge bag. The scallops were placed in a mesh bag and suspended from a rope across the top of the barrel.

Adenylic Energetic Charge (AEC)

The scallops were rapidly dissected and a small portion (0.2g) of the striated adductor muscle was removed and placed in liquid nitrogen. It was stored and analysed within one month using the technique of Moal et al. (1989a).

Behavioral Studies

The scallops were labelled with a permanent marker and placed upside-down (flat side down) in a tank with re-circulating seawater. The bottom of each tank was covered with 10cm of sediment (collected from a scallop bed) with a predetermined granulometry of “sandy silty gravel” as defined in the Wentworth scale. The righting and recessing time was monitored every 12 hours for 72 hours (unless otherwise stated). The scallops were recorded as recessed = 3 (completely covered by substrate), semi-recessed = 2 (half covered by sediment), righted = 1 (flipped over but not recessed) or not righted = 0. The sum of the individual scores was calculated every 12 hours.

Experiment 1. Stress in relation to the size of shellfish and tow length.

The effects of tow length and scallop size on stress levels were examined using a two way factorial experiment. Three different size classes of scallops were used with shell lengths of; small (59.1 ± 4.8 mm shell length), medium (70.9 ± 2.2 mm shell length) and large (81.7 ± 3.6 mm shell length). Scallops from each size class were placed in the dredge simulator for 0 (no dredge simulation) 15 or 30 minutes respectively, after which, samples were taken for AEC analysis and behaviour was monitored.

Experiment 2 Comparison of the effect of scallop size with the stress period extended to a 45 minute simulated dredging period.

In this experiment there were four treatments. Two different size classes of scallops were used small (72.9 ± 3.4 mm shell length) and large (107.6 ± 8.7 mm shell length). Scallops from each size class were placed in the simulator for a 0 or 45 minute period. Samples were taken for AEC analysis only.

Experiment 3 Comparison of the effect of emersion and physical stress on scallops.

In this experiment we looked at the combined effect of dredging and emersion on scallops using a two way factorial experiment. The scallops (102.7 ± 9.3 mm shell length) were placed in the simulator for 0 (no dredge simulation), 15 or 30 minutes respectively. After the physical stress the scallops were subjected to a 0 or 6 hour emersion stress. Following the treatment, samples for AEC were taken and behaviour was monitored.

Also a similar experiment was carried out whereby scallops were either placed in the simulator for different lengths of time (0, 0.5, 1, 2, 4, 8, 16 or 32 minutes respectively) or emersed from water for different lengths of time (0, 2.5, 5, 10 or 20 minutes respectively). Following the treatment, samples for AEC were taken.

Experiment 4 Comparison of stress levels in clamped scallops and simulated dredged scallops.

Preliminary studies had shown that scallops closed their valves tightly while in a dredge. This experiment was carried out to assess whether AEC levels decreased solely because of shell closure or if the physical movement of the dredge had an added stress effect. The scallops (101.7 ± 8.8 mm shell length) were held clamped underwater using a standard G clamp or placed in the simulator. In this experiment there were five treatments; a control – No simulated dredging or clamping (A), 15 minutes in the simulator (B) or clamped (C), 30 minutes in the simulator (D) or clamped (E). Samples were taken for AEC analysis only.

Statistical Analyses

After testing for normality non parametric data were normalized by log transformation or arcsine square root transformation. Depending on the number of factors one-way or two way analyses of variance (ANOVAs) and general linear models were used to test significant differences between treatments and a posteriori Tukey test was used to contrast treatments. The level of significance was set at 0.05.

Results

Field Experiments

Figure 1 shows the effect of dredging on the AEC level of scallops from Valentia and the Isle of Man respectively. The AEC level in the striated muscle after dredging, was similar for both sites ranging from 0.46 to 0.54 even though the length of tow was 15 minutes in Valentia and 30 minutes in the Isle of Man. The AEC level decreased further (0.37 ± 0.17) after the additional effect of transport but this was not significant. However, the AEC level increased significantly ($F_{41,5} = 4.54$, $p < 0.005$)

when returned to seawater after 24 hours (Fig. 1). In the Isle of Man the AEC level (Fig. 1) decreased significantly from the control for all tows ($F_{172,6} = 8.11, p < 0.001$).

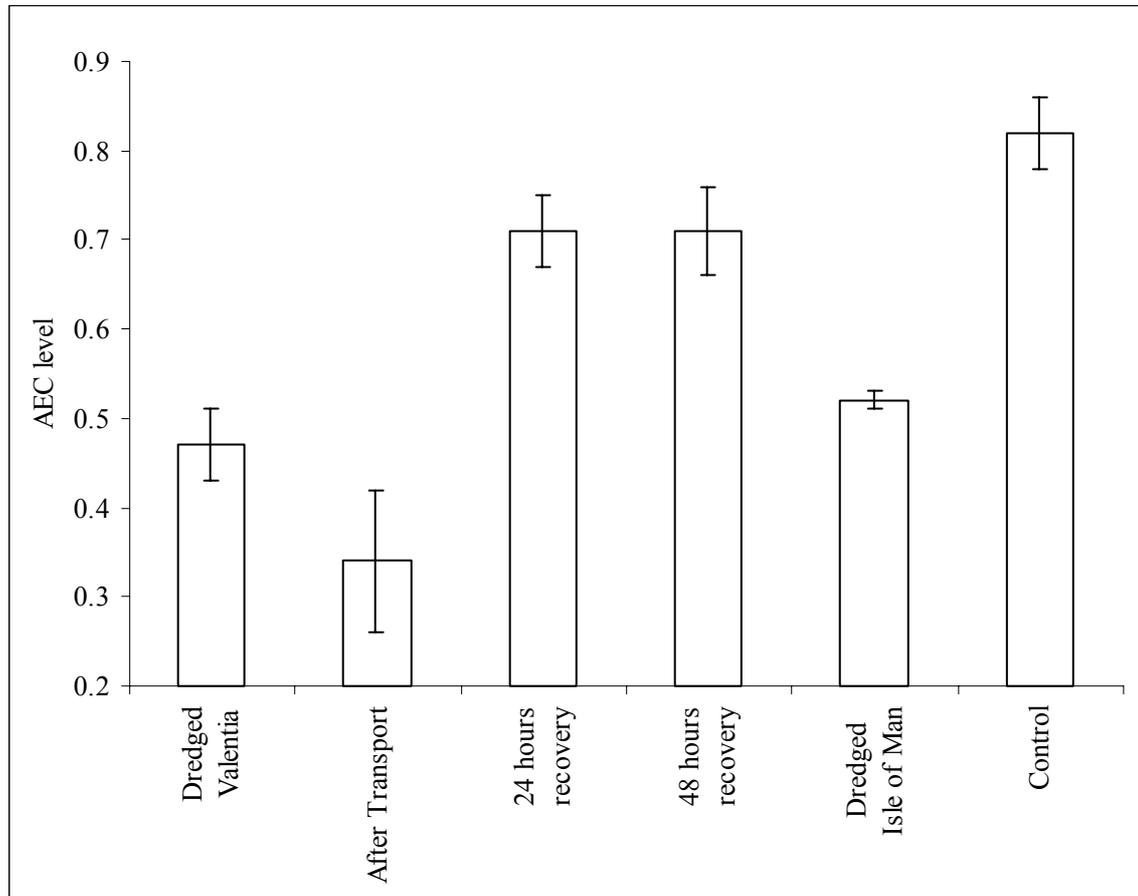


Figure 1. The effect of dredging and recovery on the AEC ratio (mean \pm s.e.) of scallops from Valentia and the Isle of Man.

Experiment 1. Stress in relation to the size of shellfish and tow length.

Figure 2 shows the effect of simulated dredging on the AEC level of different sized scallops. The smallest scallops had a significantly higher AEC level ($F_{88,2} = 4.58, p < 0.05$) than the larger animals. The AEC levels in scallops significantly decreased ($F_{88,2} = 50.25, p < 0.001$) from the control (no dredge simulation) to animals treated to 15 and 30 minutes in the simulator. The AEC results in this experiment were similar to those found in the field trials, as there was no further decrease in AEC levels from 15 to 30 minutes of dredge simulation. Scallops from these treatments (D to I) recovered significantly to within 5% of their control levels ($p < 0.01$) when returned to seawater for three days.

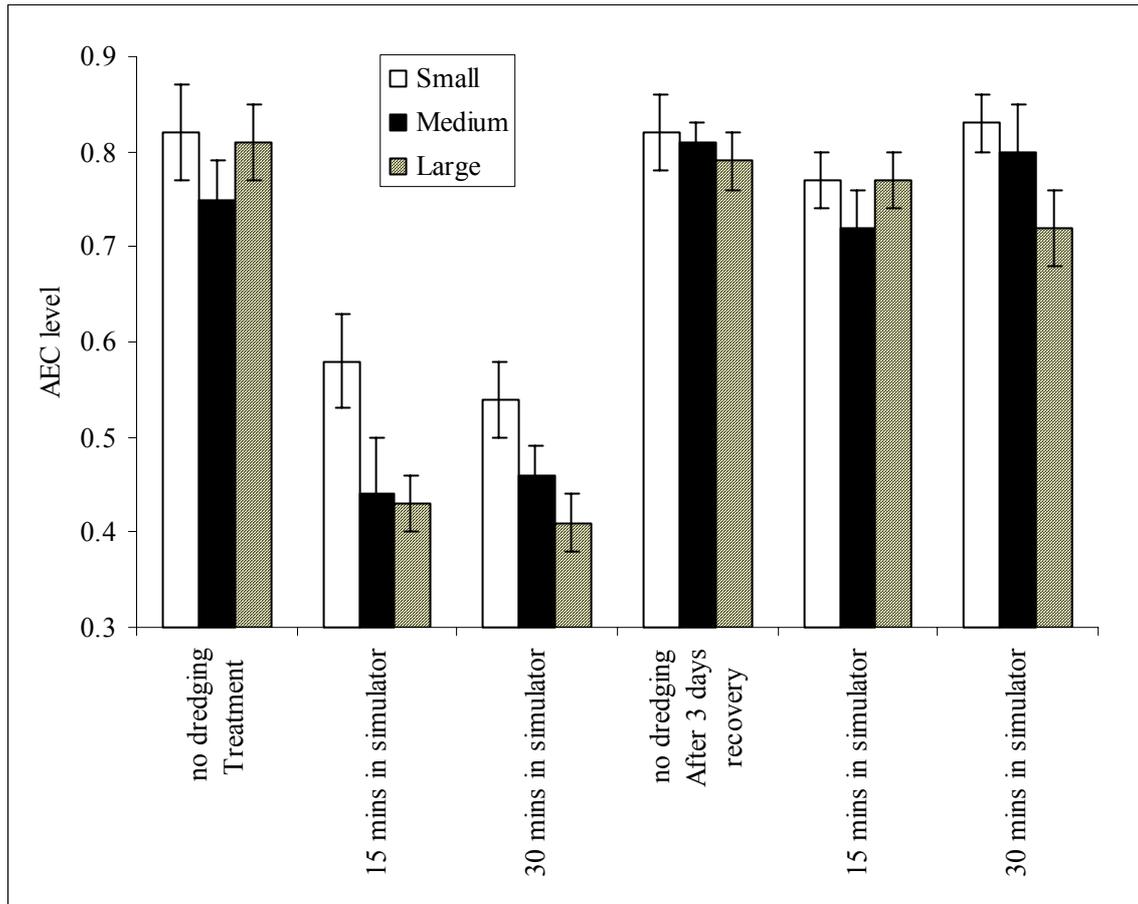


Figure 2. The effect of simulated dredging for 0, 15 and 30 minutes on the AEC ratio (mean \pm s.e.) of different sized (60 mm, 70 mm and 80 mm) scallops.

Figure 3 shows the effect of simulated dredging on the recessing and righting ability of different sized scallops. A similar pattern to the AEC results was found. In general the smaller scallops were more active than the larger animals and had a higher behavioural score. However this result was not significant. A significant difference was found ($F_{81,2} = 4.95$, $p < 0.01$) between the treatments with time as a variable, with the control animals (no dredge simulation) being significantly more active than scallops from the other treatments (15 and 30 minutes in the simulator).

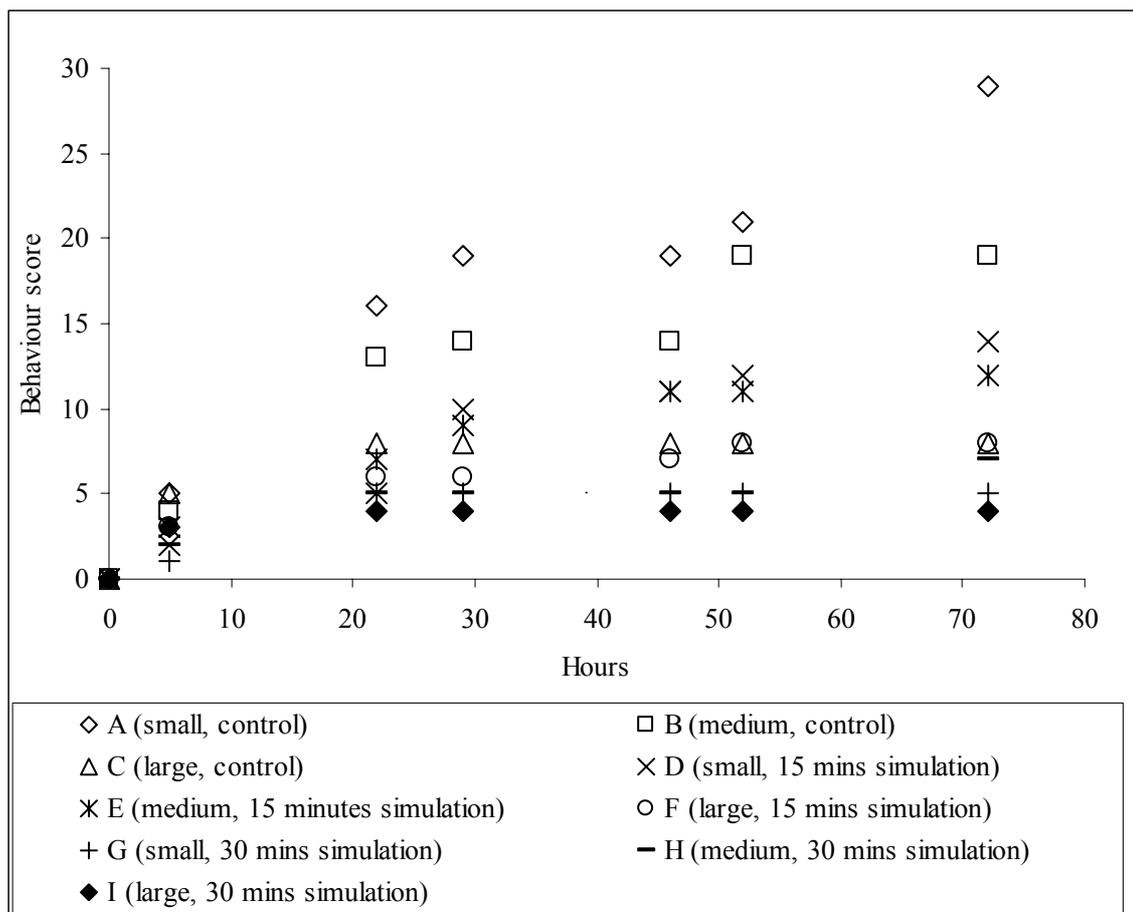


Figure 3. The effect of simulated dredging (no dredging, 15 and 30 minutes) on the righting and recessing ability of different sized (60 mm, 70 mm and 80 mm) scallops.

Experiment 2 Comparison of the effect of scallop size with the stress period extended to a 45 minute simulated dredging period

Similarly, after 45 minutes in the simulator (Fig. 4) the smaller animals had a higher AEC level (0.51 ± 0.15) than the larger animals (0.38 ± 0.17) however this result was not significant. The AEC levels in scallops significantly decreased ($F_{45,1} = 53.11$, $p < 0.001$) from the control (no dredge simulation) to animals treated to 45 minutes in the simulator. The results were similar to those found in the field trials and earlier laboratory trials in that there was no difference in AEC levels between animals collected in a dredge and those placed in the simulator for 15, 30 or 45 minutes.

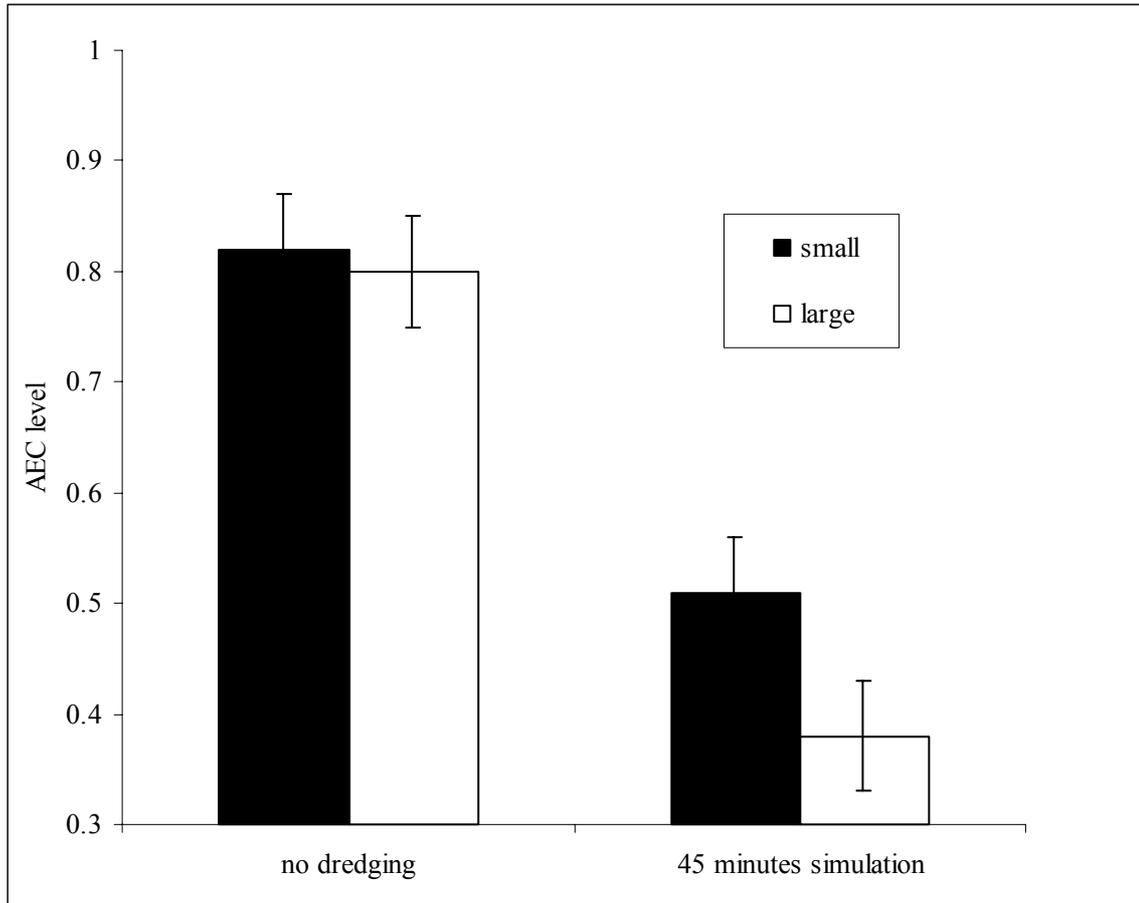


Figure 4. The effect of 45 minutes of simulated dredging on the AEC ratio (mean \pm s.e.) of small (73mm) and large (108mm) scallops.

Experiment 3 Comparison of the effect of emersion and physical stress on scallops

Figure 5 shows the combined effect of dredging followed by emersion the AEC ratio of scallops. The AEC decreased significantly from the control for all treatments. A two way ANOVA was carried out and both emersion and dredge simulation had a significant effect on the AEC levels ($F_{59,1} = 10.78$, $p < 0.001$ and $F_{59,2} = 12.91$, $p < 0.001$ respectively). Again, there was no difference between the AEC level in scallops from treatment B (15 minutes in the simulator; 0.47 ± 0.16) and treatment C (30 minutes in the simulator; 0.47 ± 0.19). However scallop AEC levels decreased in treatment E (15 minutes in the simulator followed by a 6 hour emersion to 0.41 ± 0.21) and further decreased in treatment F (30 minutes in the simulator followed by a 6 hour emersion to 0.34 ± 0.22) but not significantly. Scallops from all treatments (B to F) significantly recovered once returned to seawater after three days ($p < 0.05$).

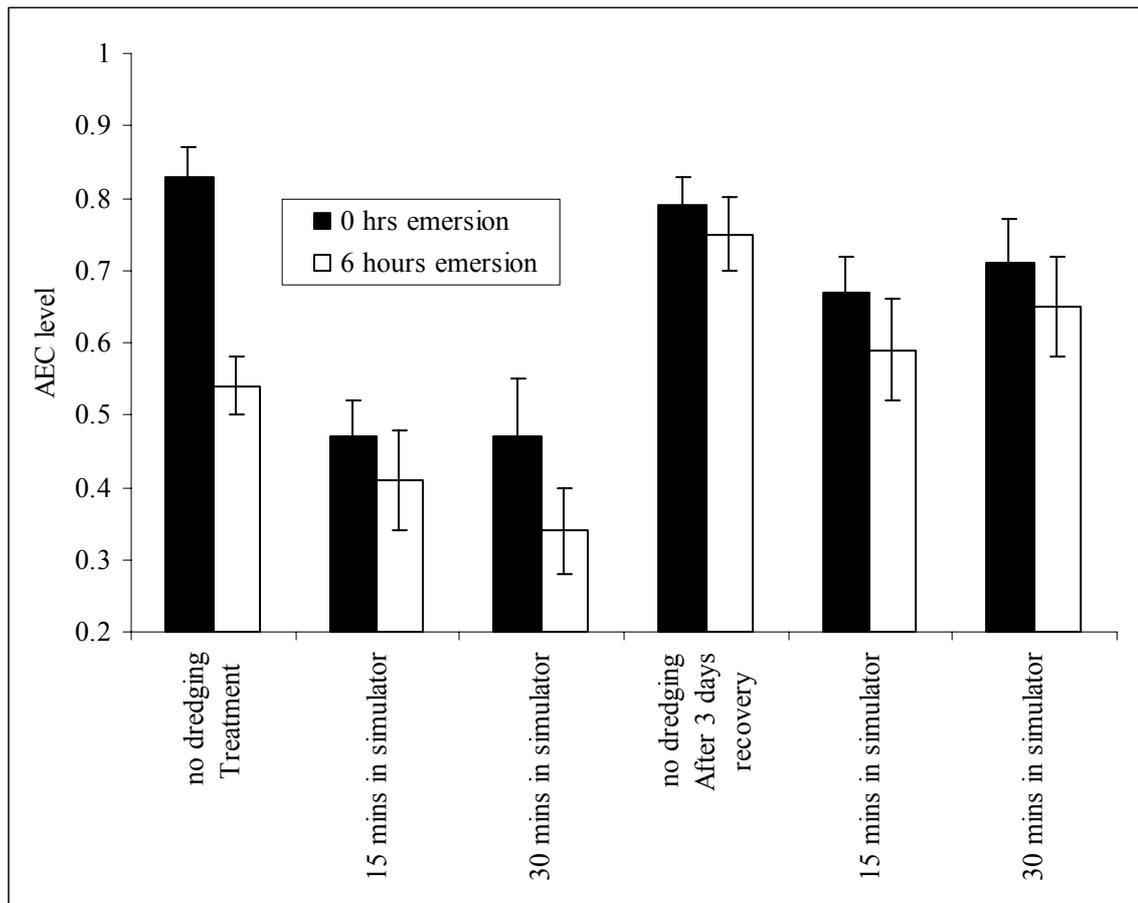


Figure 5. The combined effect of simulated dredging followed by emersion on the AEC ratio (mean \pm s.e.) of scallops.

Figure 6 shows the combined effect of dredging followed by emersion on the recessing and righting ability of scallops. The control scallops were most active in the first few hours and at the end of the 3 days period had a score of 20, which was exactly the same as treatment B (15 minute simulation). However the scallops from group B were more active after 6 hours in seawater so presumably needed this time to recover from the dredge simulator before righting and recessing. Group C (30 minutes simulation), D (6 hours emersion) and E (15 minutes in the simulator, 6 hours emersion) displayed a similar gradual increase in activity and at the end of the experiment had a score of 14, 15 and 13 respectively. Only 8 individuals righted themselves from group F (30 minutes in the simulator followed by a 6 hour emersion). A two way ANOVA was carried out and both emersion and dredge simulation had a significant effect on the righting and recessing ability of scallops ($F_{51,1} = 11.71, p < 0.001$ and $F_{51,2} = 5.5, p < 0.01$ respectively).

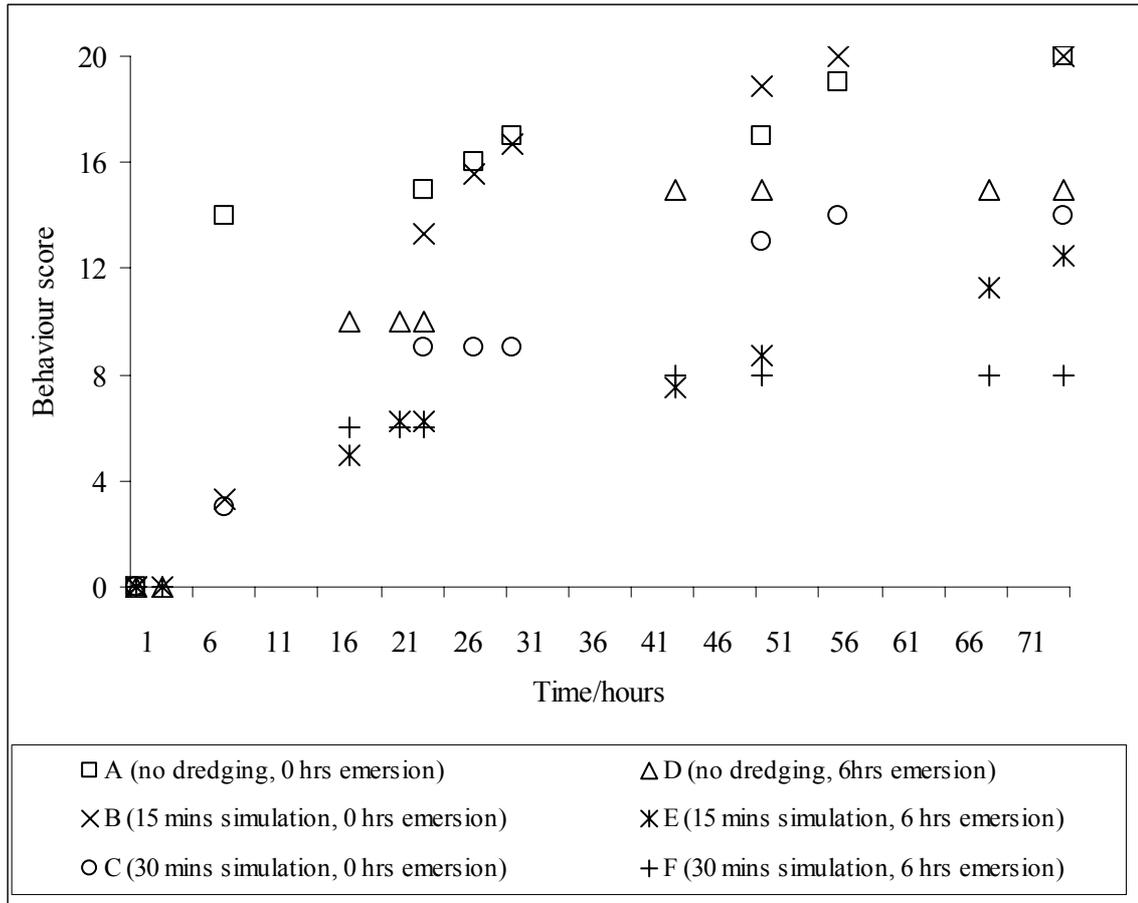


Figure 6. The combined effect of simulated dredging followed by emersion on the righting and recessing ability of scallops.

Figure 7 shows the effect of dredge simulation or emersion for different lengths of time on the AEC level of scallops. Although statistical analysis could not be carried out to compare the stressors (stress times were different between dredge simulation and emersion treatments), figure 7 indicated that over time scallops subjected to dredge simulation had a lower AEC level than the animals treated to an emersion stress. A one way ANOVA was carried out for each stressor group and the AEC levels significantly decreased when scallops were subjected to both dredge simulation ($F_{148,7} = 1763.41, p < 0.001$) and emersion ($F_{67,5} = 117.04, p < 0.001$). A Tukey test was carried out to contrast treatments and the results are shown in Figures 8 and 9.

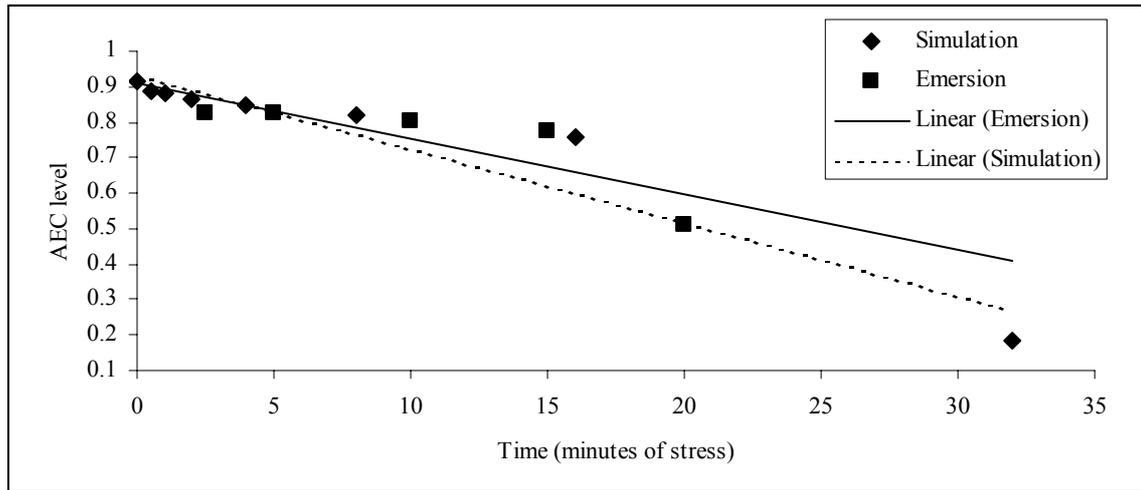


Figure 7. The effect of simulated dredging and emersion for different lengths of time on the AEC ratio of scallops.

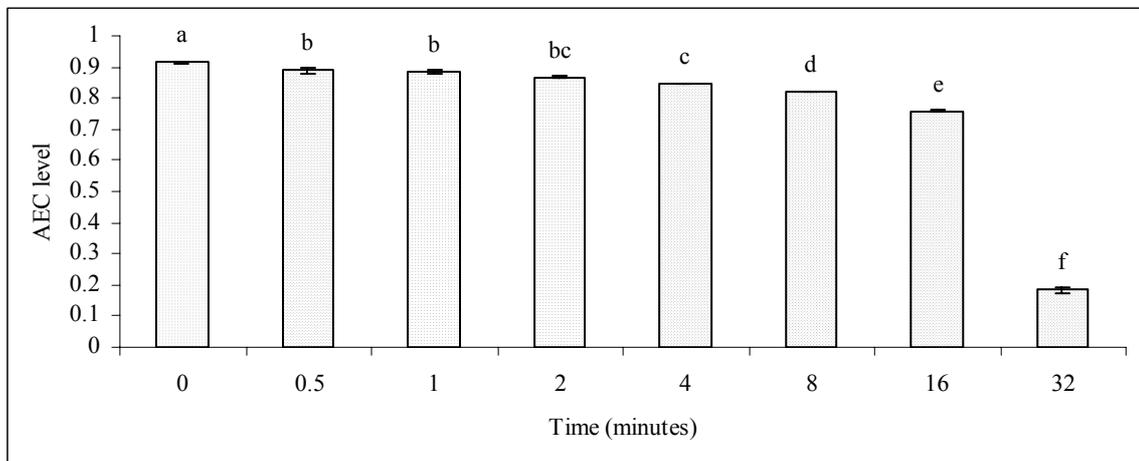


Figure 8. The effect of simulated dredging on the AEC ratio (mean \pm s.e.) of scallops. Bars sharing a common letter are not significantly different at $p < 0.05$.

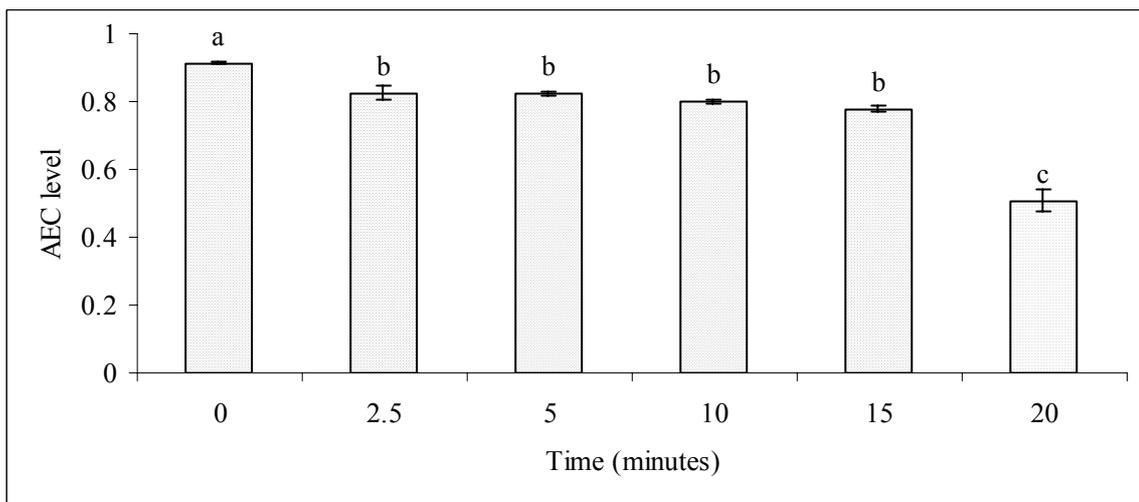


Figure 9. The effect of emersion on the AEC ratio (mean \pm s.e.) of scallops. Bars sharing a common letter are not significantly different at $p < 0.05$.

Experiment 4 Comparison of stress levels in clamped scallops and simulated dredged scallops

Figure 10 compares the effects of dredging and clamping on the AEC level of scallops. The AEC level in the simulated dredged scallop treatments was lower than the AEC level of the clamped scallop treatments but this result was not statistically significant. There was a significant difference between the simulated dredged scallops and the control scallops ($F_{45,4} = 6.37$, $p < 0.001$), but not between the control and clamped scallops.

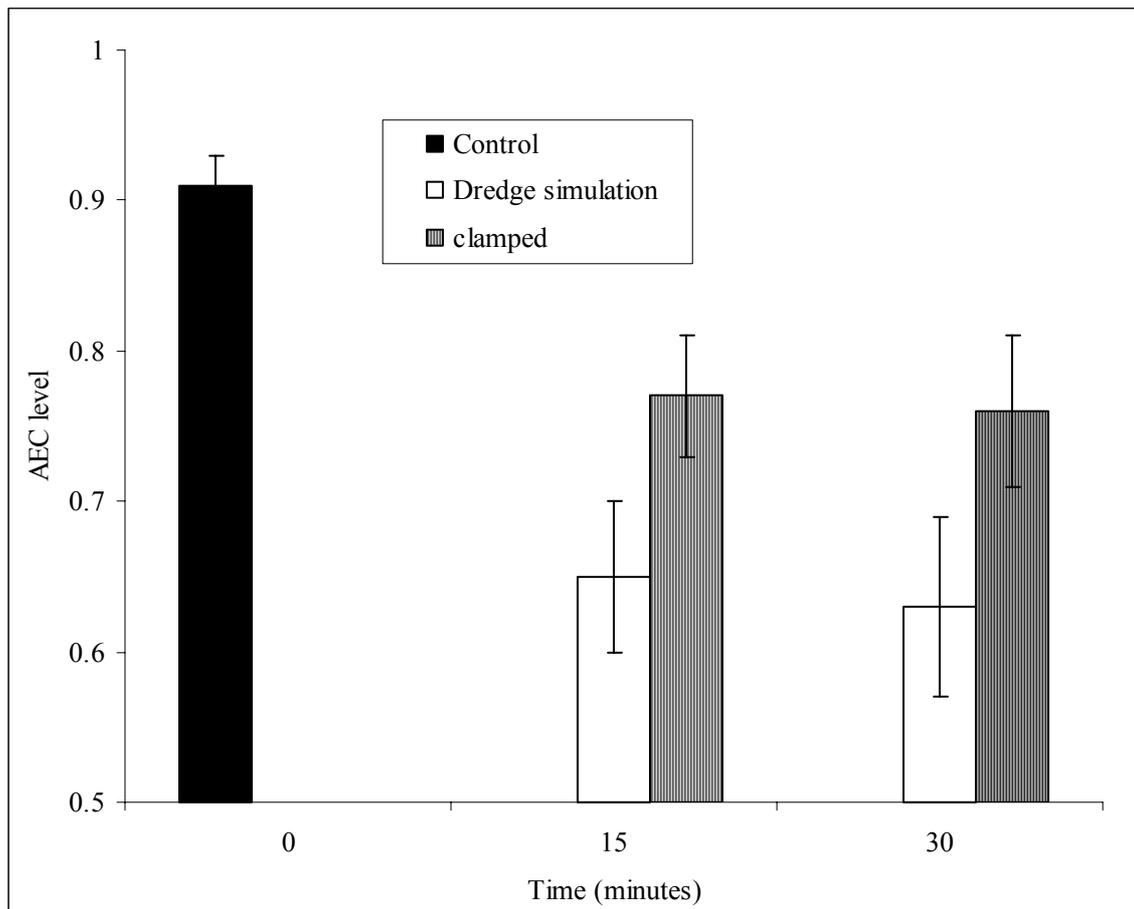


Figure 10. Comparing the effects of simulated dredging and valve clamping on the AEC ratio (mean \pm s.e.) of scallops.

Discussion

Similar AEC results were found between the laboratory and field experiments. AEC levels decreased gradually in the striated muscle to approximately 0.5 after 15 minutes of dredging, but did not decrease further after 30 and 45 minutes dredging and simulated dredging. Ivanovici (1980) and Duncan (1993) suggested that AEC levels in molluscs below a value of 0.55 resulted in a negative scope for growth, which eventually would lead to mortality. They suggested recovery was impossible even after a return to optimal conditions. However, in this study stressed scallops with AEC levels of <0.5 recovered when returned to optimal conditions after 3 days. Similarly, Wijsman (1976b) studied AEC and mortality in *M. edulis* from different habitats in relation to anaerobiosis. The energy charge decreased during exposure to

air over a seven day period from 0.85 to 0.72. Animals that died during the experiment had an AEC level of 0.72 however levels of 0.24 were found in some animals. Recovery was rapid and ATP levels were restored after a period of two to three days. Maguire et al. (1999a) found that after 12 hours dry transport of scallop spat AEC levels had decreased to 0.42, but after only two hours of recovery in seawater levels had increased to 0.68, and after 24 hours recovery levels had increased to 0.82.

Our results showed that dredged small scallops (<65mm shell length) had a higher AEC level than dredged larger animals (>70mm shell length). Similar results were also found by Maguire et al. (1999c) who dry transported two different sized groups of scallops. After 12 hours of transport the smaller animals (50mm) had a higher AEC level (0.56) than the larger animals (65mm; AEC level 0.42). In this study righting and recessing response was also greater in smaller animals. Hagen (1994.) found a negative linear relationship between the righting coefficient and body size in the green sea urchin *Strongylocentrotus droebachiensis*. Furthermore Fleury et al. (1997) completed a study of the recessing behaviour of scallops according to three different sized groups and found that 30mm scallops recessed most effectively.

In this study no further decrease in AEC levels was found in scallops subjected to 15, 30 or 45 minutes in the dredge bag or in the dredge simulator. However, when dredging was followed by emersion, AEC levels dropped considerably. Many studies have noted a decrease in the AEC levels of marine animals during prolonged emersion periods. Ebberink et al. (1979.) measured the energy demand of the posterior adductor muscle of *M. edulis* during exposure to air and found that energy expenditure was reduced by a factor of 5.3 after twelve hours. Moal et al. (1989b) found a decrease in AEC levels in the oyster *Crassostrea gigas* after a three hour emersion period. In *P. maximus* AEC levels decreased after an emersion period of two hours (Maguire et al. 1999c) and decreased from 0.88 to 0.4 after a 48 hour emersion period (Duncan 1993). In this study a significant decrease in the AEC level was found after only 2.5 minutes.

In experiment three, simulated dredging followed by emersion had a deleterious effect on righting and recessing speed in scallops. Similarly, Maguire et al. (1999c) found that the mean recessing speed of scallops under optimal conditions was 18.3 hours, which increased to 40.3 hours after scallops were treated to a twelve hour emersion period. Minchin et al. (2000) have suggested that scallops that remained unrecessed for long periods were more easily preyed upon. Predation by the edible crab *Cancer pagurus* and the starfish *Asterias rubens* is the major cause of mortality in seeded king scallops, shown by studies in Scotland, Ireland and Norway (Fleury 1997). Therefore undersized stressed or damaged scallops could have a much reduced chance of survival when returned to the seabed.

Throughout this study it had been observed that the scallops closed their valves tightly while in the dredge or during dredge simulation in the laboratory. Many studies have shown that bivalves under stress will close their valves (Kramer et al. 1989; Tyurin 1991.). During valve closure the bivalves exhibit anaerobic metabolism. In energy terms, this metabolism is seven times less efficient than aerobic metabolism and a dramatic drop in AEC levels has been shown by Zwaan (1977). In this study AEC levels dropped from 0.91 to 0.76 in scallops that were clamped for 30 minutes but

AEC levels dropped to 0.63 in animals that were subjected to dredge simulation for the same length of time. This would suggest that the physical movement of the scallops within the simulator combined with valve closure has an added stress effect.

This study has shown that undersized dredged scallops are stressed and may be less able to avoid predation when discarded and mortality levels may even be higher than previous estimates. At the moment many scallops stocks are overfished to the point of extinction. The total European capture fishery has declined from 48,571 tonnes in 1973 to 24,475 tonnes in 1996 (FAO statistics). Therefore, it is vitally important that this additional mortality be taken into account during stock assessment. To measure the level of all unaccounted mortality has not been previously quantified for scallops but it may be possible using the catch mortality model developed by Chopin et al. (1996). Overall, the number of discards needs to be reduced. This may be achieved by increasing the selectivity of the fishing gear, perhaps by either increasing the size of the belly rings or increasing the space between the teeth to allow larger scallops to be caught.

Conclusions

1. AEC levels decreased gradually in the striated muscle to approximately 0.5 after 15 minutes of dredging, but did not decrease further after 30 and 45 minutes dredging and simulated dredging.
2. AEC levels of simulated dredged scallops returned to normal after 3 days in optimal conditions.
3. Simulated dredged smaller scallops (<65mm shell length) had a higher AEC level and were more active than dredged larger animals (>70mm shell length).
4. Simulated dredging followed by emersion increased the depression of AEC levels and reduced the righting and recessing speed of scallops.
5. Physical movement of the scallops within the dredge simulator combined with anaerobic respiration during valve closure has an added stress effect during dredging.

3.1.3.3 TEMPORAL AND SPATIAL VARIABILITY IN DREDGING INDUCED STRESS IN THE GREAT SCALLOP *PECTEN MAXIMUS*.

Introduction

Most fishing techniques aim to capture a single or limited number of target species over a minimum size. This size may be dictated by markets or by government legislation. In scallop dredging, as in many other fisheries, a proportion of the captured target species are discarded owing to their small size. These undersized discards may be damaged during capture, for example suffering chipped valve margins or separation of the hinges, or may suffer stress from fatigue and desiccation. The level of mortality of these discarded scallops is unknown but may be considerable. Several studies have speculated on the levels of dredging induced mortality in relation to scallop size, sorting time and conditions on deck (Medcof and Bourne 1964b), substrate type (Naidu 1988), catch weight (Chapman et al. 1977) and type and performance of gear (Gruffydd 1972).

Damaged scallops which are left on the seabed, or are discarded, are likely to show high mortality owing to the aggregation of predators to dredge tracks (e.g. Caddy 1973; Kaiser and Spencer 1994; Medcof and Bourne 1964b) and to damaged bycatch (Veale et al. 2000a). However, the majority of undersized discarded scallops show no signs of external physical damage. In the Irish Sea Jenkins et al. (2001) and Section [3.1.4.1](#) showed that less than 10% of scallops (*Pecten maximus*) encountering dredges showed any signs of damage while Shepard and Auster (1991) estimated that damage to the scallop *Placopecten magellanicus*, which encounter dredges but are not captured, can vary between 7% and 25% depending on substrate type. Although discarded scallops may have low levels of damage, they are likely to be highly stressed from the process of capture. Little is known regarding the effects of fishing induced stress on subsequent survival of scallops.

Dredging induced stress can be assessed in scallops using a variety of biochemical indicators. These have been discussed in the introduction of [Section 3.1.3.2](#). In this paper they used AEC as a stress index to investigate the effect of dredge capture on undersized scallops. They found no difference in AEC levels between different lengths of tow (15, 30 or 45 minutes although shorter periods of simulated tow produced lower stress effects) but found that emersion following dredging had an added stress effect. In this paper we aimed to investigate different aspects of dredging on scallops i.e. effects of different seasons, ground types and to assess the stress caused to scallops that come in contact with dredges but remain uncaught on the sea bed.

We aimed to examine dredging induced stress in the great scallop *Pecten maximus* on a seasonal basis to determine if the seasonal reproductive cycle in this species affected the extent to which it is negatively affected by dredge capture. Seasonal variability in AEC levels has been examined in a number of invertebrate species including the crayfish, *Procambarus acutus acutus* (Dickson and Geisy 1982), the oysters *Ostrea edulis* and *Crassostrea gigas* (Moal et al. 1989b; Moal et al. 1991), the mussel *Mytilus edulis* (Zarogian et al. 1982.) and the clam *Cardium sp.* (Picado et al. 1988). Such variability has generally been associated with the reproductive cycle. Seasonal

changes in biochemical composition of the scallop *P. maximus* have been well documented (Ansell 1978; Maguire and Burnell 2001). Carbohydrate reserves are built up during periods of greater food availability during the summer and used up in the winter. In *P. maximus* the main period for gonadal proliferation takes place between November and February by the mobilisation of glycogen and protein reserves from the adductor muscle and lipid from the digestive gland. *P. maximus* may spawn over a long period from spring to autumn (Ansell 1978). Gametogenesis represents a period of high demand for energy in scallops and when external food supplies are limited, gamete production occurs at the expense of biochemical components in somatic tissues (Calow 1985). In this study on each sampling day we measured the glycogen content of both the striated adductor muscle and the gonad in order to assess the reproductive state of the scallops. Brokordt et al. (2000) showed that the reproductive state of the scallop *Chlamys islandica* affected its ability to escape, presumably as a result of changes in the energetic state of the individual. Therefore, it would be reasonable to hypothesize that seasonal variability in the energetic state of scallops would affect the degree to which they are negatively affected by dredging.

In addition to the major aim of this study, the determination of temporal variability in dredging induced stress, two further objectives were pursued. Firstly, field trials were conducted at three different fishing grounds to assess the degree to which dredging induced stress varies spatially. Fishing grounds were specifically chosen with contrasting bottom characteristics. Secondly, diving was undertaken to collect scallops, which encountered dredges but were not captured. AEC was measured to determine the extent to which these scallops were negatively affected by this experience.

Materials and Methods

Seasonal field trial

The field trial was carried out on the commercial scallop ground known as the Chickens, off the south west of the Isle of Man in the North Irish Sea (53° 58.75' N 04° 52.71' W). The scallops were caught using a Newhaven type spring-toothed scallop dredge (width – 75cm) with a toothed cross bar (tooth spacing – 66mm; tooth length – 100mm), a collecting bag made from case hardened steel rings (diameter – 78mm) and a net mesh cover (mesh size – 70mm). Eight dredges were towed at a speed of 5 kph for forty five minutes on 4 dates, March 15th, June 5th, October 6th (2000) and February 14th 2001. On each date two tows were made. At the end of each tow at least 15 scallops below the minimum legal landing size (<110mm shell length) were taken from two randomly selected dredges and samples of muscle and gonad taken for AEC and glycogen analysis. Each scallop was opened and two small portions of the striated adductor muscle and one portion of gonad weighing approximately 0.2 g were removed using a scalpel. Samples were wrapped separately in tin foil and immediately placed in liquid nitrogen.

On each date 10 undersized scallops were taken from the dredges and transferred alive to Port Erin Marine Laboratory and placed in tanks containing running seawater. Samples of muscle were taken one week later as described above. These samples acted as controls for AEC analysis. A preliminary experiment showed that the AEC levels of post-dredged scallops returned to normal after 24 hours (at most) of recovery in seawater (Maguire et al. 2002b see also [Section 3.1.3.4](#)). Scallops placed in tanks

for a full week had made a complete recovery and thus this protocol was adopted for the control animals in this study.

Comparison of scallops from different grounds

This field trial was carried out in June 2000 on three commercial scallop fishing grounds off the Isle of Man; Chickens (53° 58.75' N 04° 52.71'W), Laxey (54° 13.50' N 04° 21.38'W) and Bradda Inshore (54° 06.5' N 04° 47.76'W). The substratum on all grounds was generally coarse sand or gravel, often overlain with pebbles, cobbles and dead shell. The abundance of pebbles and cobbles, as indicated by the number caught during dredging, varied greatly among the three grounds sampled. At Laxey there were relatively few and the substratum was predominantly sandy with areas of dead maerl. The Chickens and Bradda Inshore grounds were both made up of variable gravelly sediments with large numbers of pebbles and cobbles at Bradda Inshore and intermediate amounts at Chickens.

Scallops were caught using the same method as above. Samples of muscle for AEC analysis were taken from 15 undersized scallops from random dredges from two tows.

Stress in non-captured scallops

In June 2000 dredging was undertaken on the Bradda Inshore scallop ground as part of a study into the impacts of dredging on benthic megafauna (see Jenkins et al. 2001 for details and [Section 3.1.4.1](#)). Divers collected scallops which had encountered dredges but not been captured by following the tracks on the seabed formed from the passage of the dredge. 42 scallops from within the dredge tracks (impacted scallops) and 14 scallops from outside the tracks (control scallops) were collected between 15 and 45 minutes after the passage of the dredges. Dredging also took place and two tows were carried out. For each tow, 15 undersized animals were removed from the dredge bags. On surfacing all scallops were immediately opened and samples of striated adductor muscle taken for AEC analysis.

Biochemical analyses

All samples were stored in liquid nitrogen until analysis. Adenylic Energetic Charge (AEC) of the striated adductor muscle was determined using the technique of Moal et al. (1989a). Maguire et al. (1999b) conducted an experiment to assess the effect of stress on the AEC level of both the smooth and striated muscle of the king scallop. A greater significant AEC decline was shown between treatments in the striated muscle. For glycogen analysis samples were withdrawn from the liquid nitrogen and freeze dried using a HETOSICC CD 53-1 freeze dryer. The glycogen content was analysed using a miniaturisation of the Dubois et al. (1956.) method.

Statistical Analyses

After testing for normality, non parametric data were normalized by log transformation or arcsine square root transformation. One-way analyses of variance (ANOVAs) were used to test significant differences between treatments and *a posteriori* Tukey test was used to contrast treatments. The level of significance was set at 0.05.

Results

Seasonal field trial

Figure 1 shows the effect of dredging on the AEC level of scallops from the Chickens scallop fishing ground off the coast of the Isle of Man. The AEC level decreased significantly in the dredged scallops from the control samples at each sampling period ($p < 0.001$). However, the AEC levels of the control samples varied throughout the year but not significantly. The highest AEC levels in the control animals were found in October (0.90 ± 0.01) and the lowest levels were found in February (0.77 ± 0.05). Because of this seasonal AEC variation in the control animals the % AEC decrease from the control to dredged scallops was calculated for each season. The AEC reduction was highest in February ($44.8\% \pm 3.3$) and June ($43.4\% \pm 2.9$) and significantly less ($F_{136,3} = 3.476$, $p < 0.05$) in March ($33.6\% \pm 2.8$). The lowest absolute value of AEC in the striated muscle of scallops following dredging was also in February (0.43 ± 0.03) and the highest was in October (0.53 ± 0.01). The AEC level reached in scallops sampled in February was significantly different ($F_{136,3} = 4.942$, $p < 0.05$) from all the other dates sampled except for June. No difference in the AEC levels of dredged scallops was found between any of the other sampling dates.

The % glycogen content in the striated muscle of the animals showed a similar pattern to the AEC results found in the control animals (Figure 2). The lowest glycogen levels were found in the scallops sampled in February ($4.38\% \pm 0.27$), with levels over three times higher in October ($F_{165,3} = 83.32$, $p < 0.001$). The % glycogen content in the gonad also varied significantly throughout the year ($F_{156,3} = 331.42$, $p < 0.001$). The glycogen levels in the gonad ranged from a minimum in October ($0.30\% \pm 0.05$) to a maximum in June ($10.33\% \pm 0.43$).

Sea water temperatures decreased from a maximum on August 29th 2000 (15.7°C) to a minimum value on the 17th of January 2001 (7.3°C). On the specific sampling days the temperatures are shown on Figure 1.

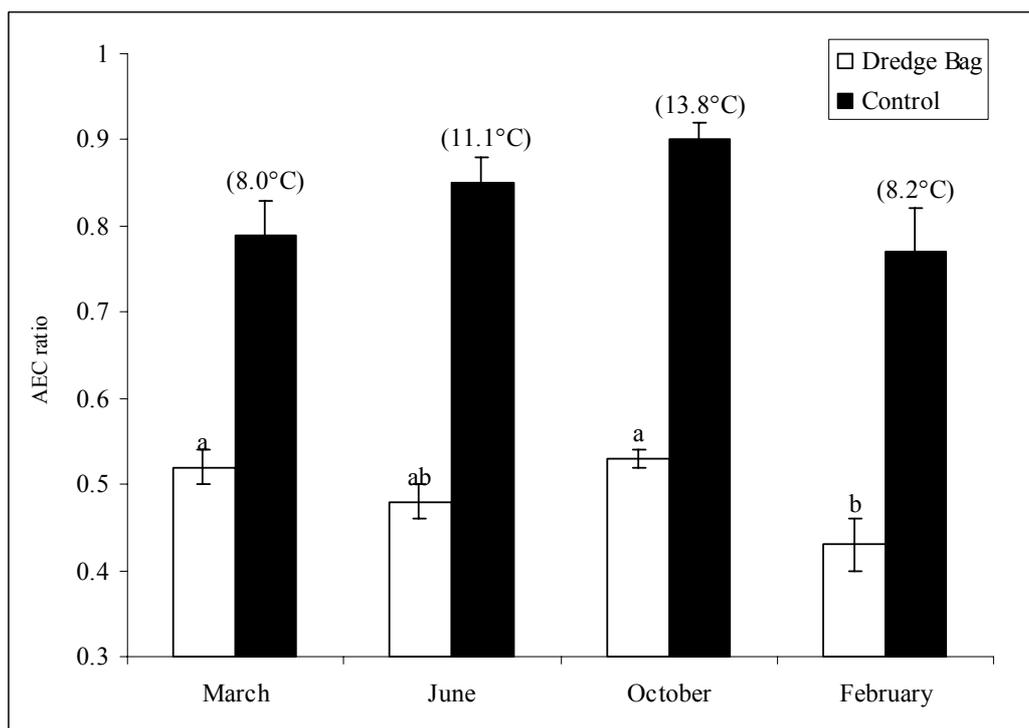


Figure 1. The seasonal effects of dredging on the AEC level (mean ± s.e) of scallops from Chickens (March 2000 to February 2001). Sea water temperatures shown in parenthesis. Any two means sharing a common letter between each treatment are not significantly different at $p < 0.05$ (Tukey test).

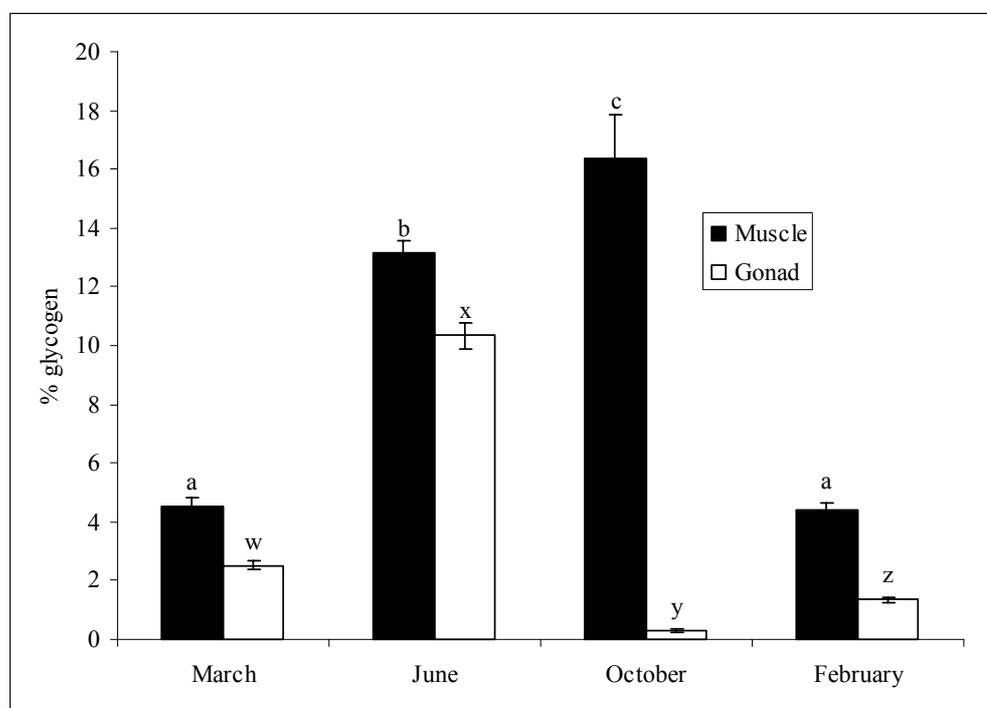


Figure 2. The seasonal % glycogen content (mean ± s.e) of scallops from Chickens (March 2000 to Feb. 2001). Two means sharing a common letter between each treatment are not significantly different at $p < 0.05$ (Tukey test).

Comparison between scallops from different grounds.

There was no difference in the AEC levels of scallops dredged at different sites in June (Figure 3). The AEC level varied in scallops from 0.46 to 0.48 on all three grounds.

Stress in non captured scallops.

Scallops that came in contact with the dredge but remained uncaught had a significantly lower AEC level than the control scallops ($t_{69} = 7.401$, $p < 0.001$). However, the reduction in AEC was not as great as in captured scallops. Figure 4 showed that the AEC level in captured scallops was significantly lower ($t_{55} = 6.869$, $p < 0.001$) than in the non- captured scallops.

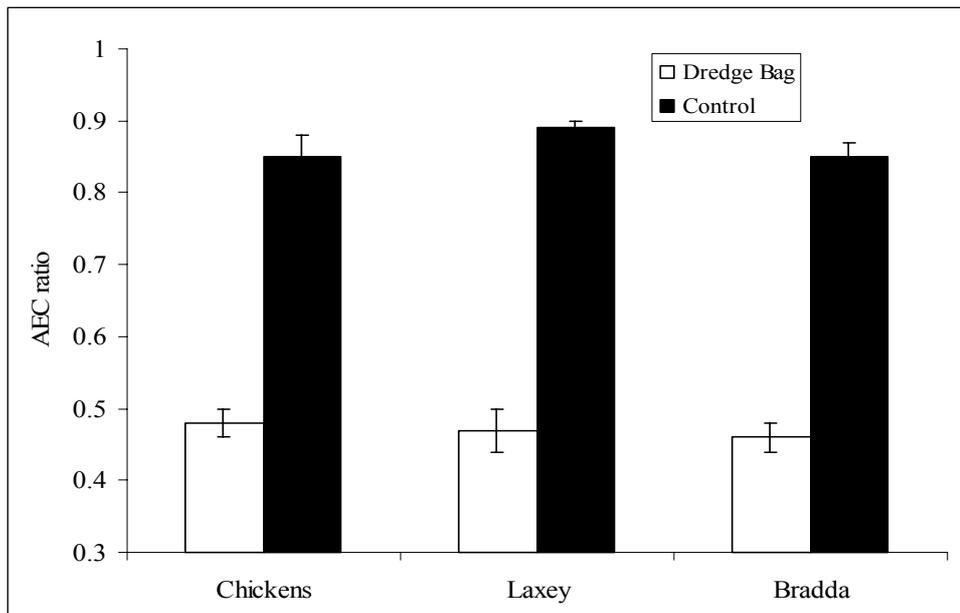


Figure 3. The effects of dredging on the AEC ratio (mean \pm s.e) of scallops from three sites off the Isle of Man (June 2001).

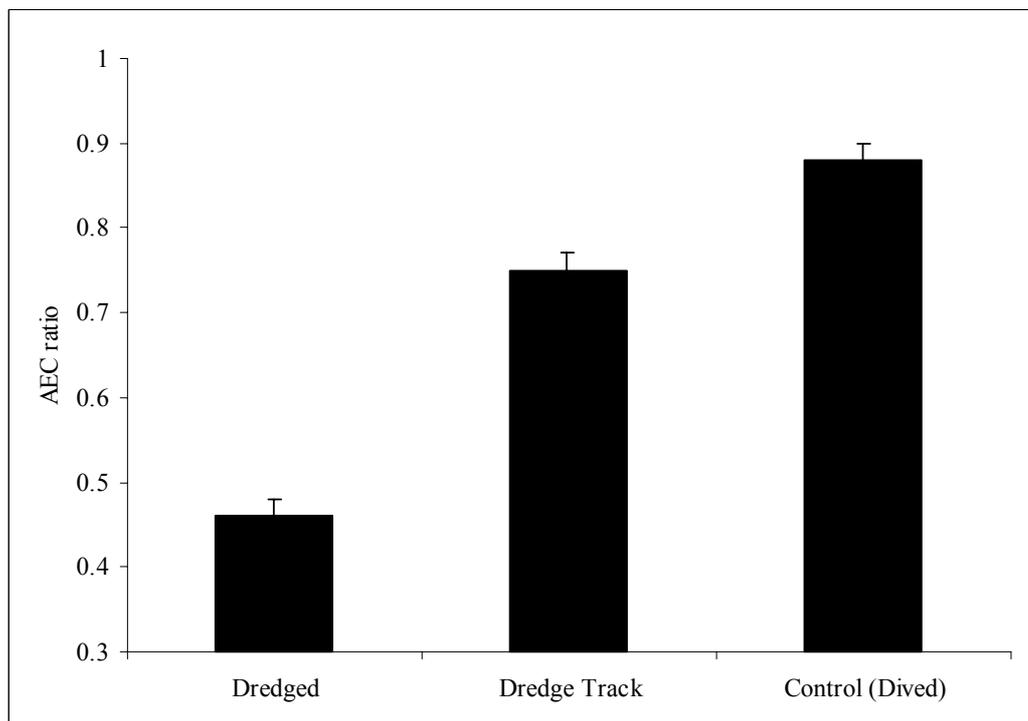


Figure 5. Comparison of the AEC levels (mean \pm s.e) of dredged scallops and scallops left behind on the dredge track from Bradda in June.

Discussion

The results from this study suggest that the AEC response to dredging of undersized scallops is different according to the season of the year. Lowest AEC levels in dredged and control scallops were found in February and highest levels in October. This coincided with the troughs and peaks of the glycogen content in the adductor muscle of the test animals. Reserves are built up during periods of greater food availability in the summer. However, energy requirements for maintenance are high therefore rapid utilisation of reserves takes place during the winter. Many demands are placed on the metabolism of scallops during the winter from the stresses imposed from internal physiological drives such as gonadal proliferation. However, during unfavourable winter conditions these energy demands may be insufficient to support gonadal development (Ansell 1978). The greatest % AEC decrease from control to dredged scallops was also in February (44.8%). Therefore, the additional stress effect of dredging at the end of winter had a greater negative effect on the AEC level of scallops whose energy reserves were already low. Similarly, Brokordt et al. (2000) showed that the mobilization of reserves had a detrimental impact on the escape response of the Iceland scallop, *Chlamys islandica*.

Moal et al. (1989b), also found that the AEC response to the stressor aerial exposure was different according to the time of year in the oyster *Crassostrea gigas*. The animals were subjected to a three hour emersion period in January, May and July. Lowest AEC values were found in July and these levels were inversely correlated with temperature. Similarly, in our study a large % AEC decrease from control to dredged animals was recorded in June (43.4%) and October (41.1%) when temperatures were highest. The June sample coincided with the spawning season of the scallop population in the Isle of Man. Moal et al. (1991), also found low AEC levels during the summer in the farmed oysters *C. gigas* and correlated this decrease with the reproductive state of the oysters.

Many authors have suggested that AEC levels in molluscs below a value of 0.5 result in a negative scope for growth, which would eventually lead to mortality even if the animals were transferred to optimal conditions (Ivanovici 1980; Duncan 1993). However, in this study, scallops dredged in February and June had AEC levels <0.5. However, no mortality was recorded in the control scallops, which were kept for several days post dredging and would be expected to have similar AEC levels as the experimental scallops immediately after dredging. Similarly in a previous study by Maguire et al. (1999a), juvenile scallops were transported in polystyrene boxes for 12 hours and the AEC level of the spat decreased from 0.88 to 0.42. Recovery of these animals was initially rapid and AEC levels had increased to 0.68 after only two hours in optimal conditions and had returned to normal after 24 hours. In addition dredge captured scallops which are returned to the laboratory and maintained in running seawater showed consistently low levels of mortality (Jenkins, unpublished data). Although the stress and subsequent reduction in AEC levels in dredge captured scallops may not lead directly to mortality, such scallops may be more susceptible to predation, (Thompson et al. 1980) through a reduction in their escape response or inability to recess (Minchin et al. 2000). Maguire et al. (2002a) and [Section 3.1.3.2](#) found that the AEC levels of dredged scallops (30 minute tow length) had returned to normal after 3 days of recovery but had not recessed into the sediment in that time period. Jenkins and Brand (2001) and [Section 3.1.3.1](#) have shown that the escape response in *P. maximus* was significantly reduced following simulation of dredging

for up to 24 hours. Such observations suggest that predator induced mortality may be significant for stressed discards.

In general, the % glycogen content of these Isle of Man scallops was quite low in the striated adductor muscle ranging from 4.4% to 16.4%. In a study of a population of scallops in Bantry Bay, Ireland the glycogen content ranged from 16.5% to 45.9% (Maguire and Burnell 2001). Ansell (1978) studied the storage of reserves in the adductor muscle of some scallop populations in the U.K. and found that the % glycogen content generally varied from 2.2% to 24.0%, but in one population the glycogen content was as high as 39%. He also suggested that glycogen content could vary between sites and between different years.

There was no difference in the AEC values between the three different sites studied. The effect of the dredging process was the dominant stressor on the animals so much so that the subtle differences between the sites had little added impact on the AEC decrease. Similarly, Verschraegen et al. (1985.) found no difference in the AEC levels of the polychaete species *Nereis diversicolor* and *Nephtys* sp. between sites in the Western Scheldt estuary, Belgium. Also, Zaroogian et al. (1982.) found no difference in AEC levels of the mussel *Mytilus edulis* in two sites.

Scallops that came in contact with the dredge but remained uncaptured and were found on the dredge track were stressed (AEC level = 0.75). Although the AEC level was not as low as in captured scallops it is important to demonstrate that the impact of dredging is not confined solely to those scallops landed on deck. Owing to the relatively low efficiency of scallop dredges (Dare et al. 1993) far more scallops will encounter dredges and remain on the seabed than are captured. These non captured animals which include all sizes, may suffer enhanced mortality due to dredge induced stress in the same way as undersized discards. Also, it was interesting to note that the collection and transportation of scallops from the sea-bottom to the surface did not induce stress in the control animals.

Conclusions

1. AEC levels in dredged and control animals, and % glycogen in the striated muscle of dredged scallops, were lowest in February and highest in October.
2. The largest % AEC decrease from control to dredged scallops was also in February and the smallest decrease was in March.
3. There was no difference in the AEC level of dredged scallops from three sites with different sea bed types around the Isle of Man.
4. Scallops that came in contact with the dredge but remained un-captured on the dredge track had a reduced AEC level but this level was significantly higher than the AEC level of captured animals.

3.1.3.4 THE EFFECTS OF REPEATED DREDGING AND SPEED OF TOW ON UNDERSIZED SCALLOPS.

Introduction

Upon contact with the dredges, scallops are either caught or left on the seabed. A proportion of the caught scallops will be kept as catch while the remainder will be discarded, being undersized (<110mm shell length) or damaged (McLoughlin et al. 1991). The indirect mortality of discarded scallops is unknown but thought to be considerable and has been found to be directly related to the catch weight (Chapman et al. 1977), substrate type (Naidu 1988), type and performance of gear (Gruffydd 1972), scallop size, sorting time and conditions on deck (Medcof and Bourne 1964b). Murawski and Serchuk 1989) estimated that the total mortality of captured scallops including discards was 10% per tow. Of the scallops that are not caught, a significant proportion that have come in contact with the dredge may also be damaged and subsequently die. Shepard and Auster (1991), estimated that the proportion of damaged scallops left behind in the dredge track can be as high as 7% on sandy bottoms and 25% on rock bottoms. Discarded scallops have also been shown to be more susceptible to predation. Caddy (1973) observed that predatory fish and crabs were attracted to dredge tracks within one hour of dredging at densities of up to 30 times greater than in areas outside the tracks.

In Ireland, many local stocks are often overfished to the point of extinction as many boats will fish for small daily returns all year round. Therefore, many undersized scallops may be caught and returned to the sea bed over and over again by the local scallop fishing fleet. It has been shown that physiological responses of fish to stressful influences may be cumulative in certain finfish species (Donaldson 1981; Wedemeyer et al. 1984; Barton et al. 1986). However, to our knowledge, this has not been tested experimentally in shellfish. In this paper we looked at the effect of simulated dredging on both the AEC levels and the repositioning and righting ability of undersized scallops (these techniques have been discussed in the introduction of [Section 3.1.3.2](#)). The simulator was used to examine the stress effect of different dredge tow lengths, tow speeds and to determine if the scallop can respond to further successive dredging stress and to monitor the recovery after the repeated stresses.

Materials and Methods

Laboratory experiments

All scallops used in the laboratory experiments were under market size ($71.45 \pm 3.0\text{mm}$) and were collected from Clew Bay, Co. Mayo off the west coast of Ireland by divers. The experiments took place between March and September 2000. The animals were acclimated in seawater tanks for one week at a temperature of 11°C and a salinity of 35‰. For each treatment five separate dredge events took place containing four scallops each. After dredge simulation ten scallops per treatment (2 from each dredge event) were sampled for AEC analysis ($n=10$) and behaviour was monitored for the remaining 10 animals unless otherwise stated.

Dredge simulator

The simulator was originally a meat turner, which rotates at an angle of 45° at variable speed. It consists of a stainless steel barrel (diameter 60cm; depth 80cm), powered by two motors each 3.8/220 Volts with 1.15/2 Amps. Small rocks, stones and sand were placed in the simulator, which was constantly filled with seawater through a hose at a flow rate of 9 sec/l. The scallops were placed in a mesh bag and suspended from a rope across the top of the barrel.

Measurement of stress

Following each experimental treatment, scallops were either sampled for AEC or their behaviour monitored. Those sampled for AEC were immediately dissected and a small portion (0.2g) of the striated adductor muscle was removed and placed in liquid nitrogen. It was stored until analysis using the technique of Moal et al. (1989a).

Assessment of behaviour was made by placing scallops upside-down (flat side down) in a tank with recirculating seawater. The bottom of each tank was covered with 10cm of sediment (collected from a scallop bed) with a predetermined granulometry of “sandy silty gravel” using the Wentworth scale. Scallops were observed every 12 hours over a 72 hour period and their position relative to the sediment recorded. Scallops were either not righted, righted (flipped over but not recessed), semi-recessed (half covered by sediment) or recessed (completely covered by substrate) and were subsequently given a score of zero, one, two and three respectively. The sum of the individual scores was calculated every 12 hours.

Experiment 1. Stress in relation to the speed and duration of tows

The scallops were placed in the simulator for 15 or 30 minutes respectively at a rotation of 11 or 22rpm or were not dredged. After each treatment 10 scallops were sampled for AEC analysis and behaviour was monitored for 10 animals. Recovery was also monitored biochemically, after 2, 6 and 24 hours, 10 adductor muscle samples from these scallops were taken for AEC analysis.

Experiment 2 Cumulative effects of scallop stress at 11rpm

In this experiment there were nine treatments. The scallops were placed in the simulator for 15 or 30 minutes at a speed of 11 rpm or were not dredged. After each treatment 10 scallops in total were sampled for AEC analysis and behaviour was monitored for 24 hours for 10 animals. This treatment and sampling regime was repeated after 24 and 48 hours. The behaviour of the animals was monitored for a further 72 hours. Recovery was also monitored biochemically, after 2 and 24 hours, by taking 10 adductor muscle samples from these scallops were taken for AEC analysis.

Experiment 3 Cumulative effects of scallop stress at 22rpm

This experiment was similar to the previous experiment although the sampling regime was slightly altered. There were nine treatments and scallops were placed in the simulator for 15 or 30 minutes at a speed of 22 rpm or were not dredged. After each treatment 10 scallops were sampled for AEC analysis and behaviour was monitored for 24 hours for the remaining 30 animals. This treatment and sampling regime was repeated after 24 and 48 hours. The behaviour of the animals was monitored for 72

hours for the remaining 10 scallops. Recovery was also monitored biochemically, after 72 hours, 10 adductor muscle samples from these scallops were taken for AEC analysis.

Statistical analyses

Non parametric data were normalised by log transformation or arcsine square root transformation. Depending on the number of factors t-tests, one-way or two way analyses of variance (ANOVAs) and general linear models were used to test significant differences between treatments and a *posteriori* Tukey test was used to contrast treatments.

Results

Experiment 1. Stress in relation to the speed and duration of tows

The speed of the simulator had an important impact on the AEC level of the treatment scallops (Figure 1). The AEC decreased significantly from the control for all treatments ($F_{43,4} = 13.04$ $p < 0.001$). A Tukey test contrasted the treatments and there was no effect of length of simulation on AEC levels. However a significant difference was shown in the AEC levels between the scallops from the low to high speed treatments, which decreased from 0.70 ± 0.03 to 0.54 ± 0.03 respectively. Recovery was rapid and scallops from treatment B (15 minutes in simulator, low speed) had AEC levels of >0.8 after 2 hours. In the remaining scallop treatments (C-E) the AEC returned to levels of >0.8 after 6 hours.

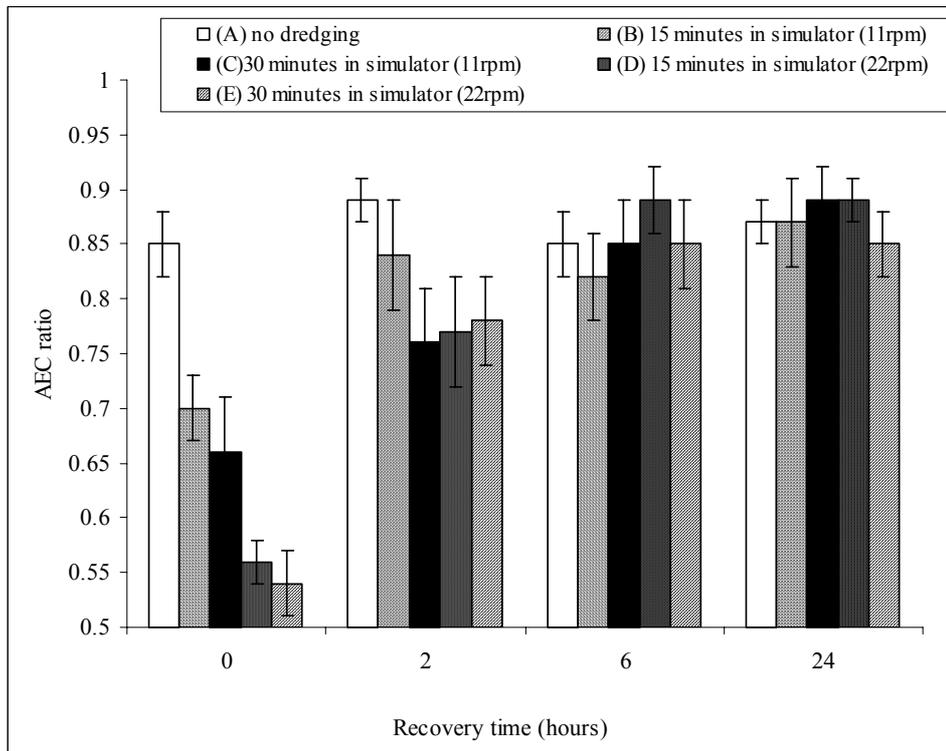


Figure 1. The effect of simulated dredging (no dredging, 15 and 30 minutes) at two different speeds (11 rpm and 22 rpm) on the AEC ratio (mean ± s.e.) of undersized scallops.

The behaviour results are shown in Figure 2. The control scallops were most active in the first few hours of the experiment and after 24 hours had a score of 22 and at the end of the 3 days period had a score of 24. At 24 hours, treatment B (15 minute simulation 11rpm) and treatment C (30 minute simulation 11rpm) had a score of 15 and 22 respectively which increased to 24 and 23 respectively after three days. Group D (15 minute simulation 22rpm) and E (30 minute simulation 22rpm) displayed a similar gradual increase in activity and at 24 hours had a score of 8 and 9 respectively and by the end of the experiment both groups had score of 14. An ANOVA was carried out after 24 hours recovery as this was the time when the non dredged scallops had recovered and speed had a significant effect on the righting and recessing ability of scallops ($F_{45,4} = 5.37, p < 0.01$) with groups D and E significantly different from group A. However there was no difference between groups A, B and C.

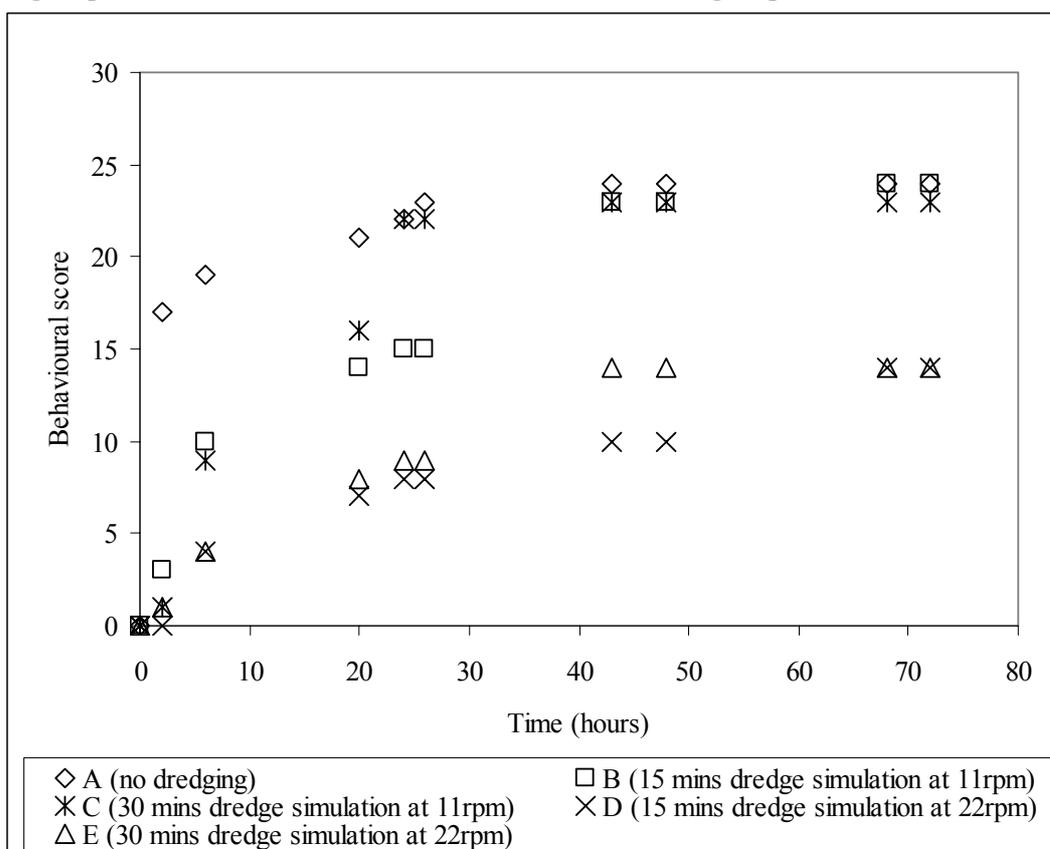


Figure 2. The effect of simulated dredging (no dredging, 15 and 30 minutes) at two different speeds (11 rpm and 22 rpm) on the behavioural score of undersized scallops (score = sum of not righted (0), righted (1), semi-recessed (2) and recessed (3) animals, n=10).

Experiment 2 Cumulative effects of scallop stress at 11rpm

The cumulative effects of dredging on the AEC levels of the test scallops are shown in Figure 3. There was a significant decrease in AEC levels from a mean of 0.85 ± 0.03 in control scallops to 0.70 ± 0.03 after 15 minutes dredge simulation (Treatment B). AEC levels then significantly decreased further when the scallops were subjected to further dredging after 24 hours (0.58 ± 0.03). No further decline was evident on repeated dredging at 48 hours. The results were similar for treatment C (30 minute dredge simulation, $T_0 = 0.66 \pm 0.05$; $T_{24} = 0.59 \pm 0.03$; $T_{48} = 0.57 \pm 0.03$). A two

way ANOVA was carried out and both dredge simulation and repeated dredging had a significant effect on the AEC levels $F_{2,238,2} = 46.22$ $p < 0.001$. Recovery was rapid and AEC levels returned to normal in the scallops from treatment B after only 2 hours. In treatment C, after 2 hours recovery in seawater scallop AEC levels were between 0.7 and 0.8 but returned to >0.8 after a 24 hour recovery period.

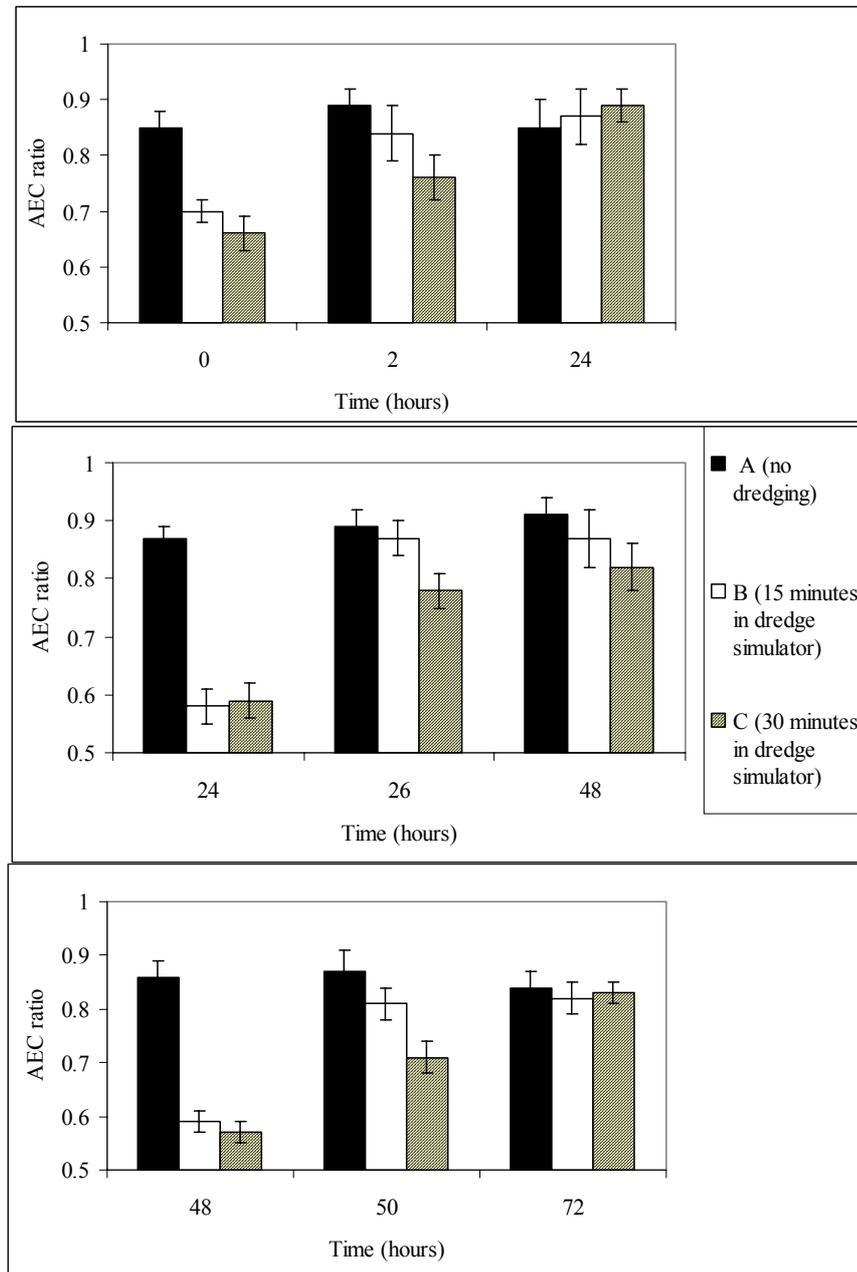


Figure 3. The effects of daily simulated dredging (11rpm) and recovery on the AEC ratio (mean \pm s.e.) of scallops.

The behaviour results followed a similar pattern to the results from experiment 1 (Figure 4). In the control group most of the animals righted and recessed during the first 24 hours after treatment. Activity was similar for treatments B (15 minute simulation) and C (30 minute simulation) after repeated dredging at Time 24 and 48

hours. A two way ANOVA was carried out and only dredge simulation had a significant effect on the behavioural score ($F_{81,2} = 7.56$ $p < 0.001$). However, the

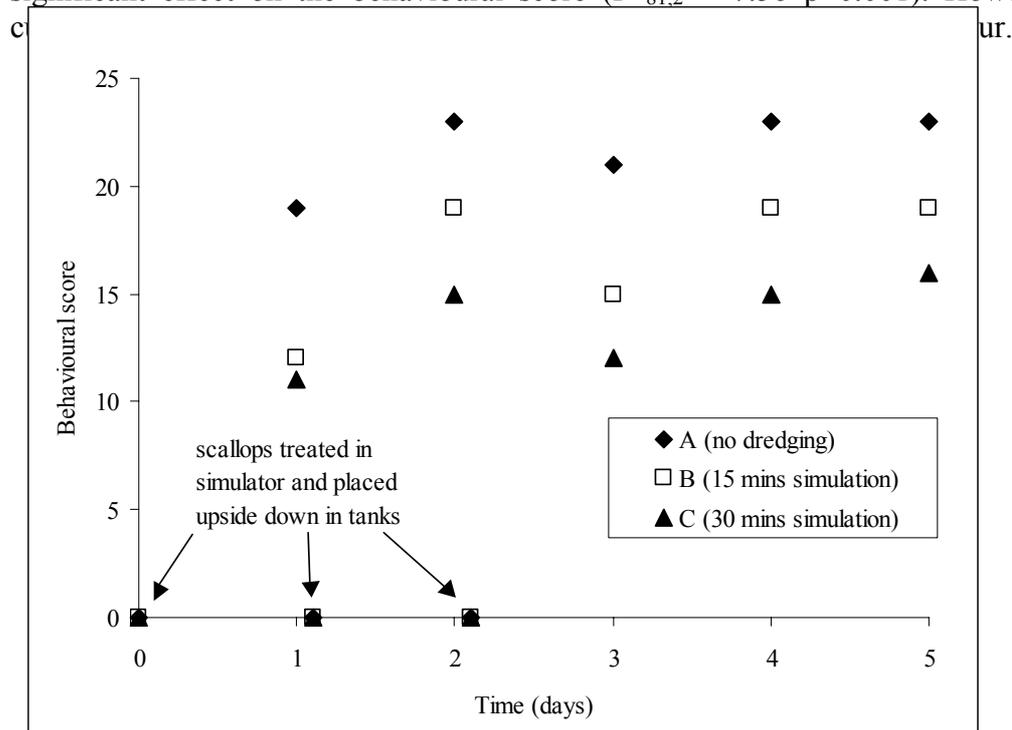


Figure 4. Cumulative effects of daily simulated dredging (11rpm) on the behavioural score of undersized scallops (score = sum of not righted (0), righted (1), semi-recessed (2) and recessed (3) animals, n=10).

Experiment 3 Cumulative effects of scallop stress at 22rpm

The AEC decreased significantly from the control for all treatments (Figure 5). A two way ANOVA was carried out and the length of tow had a significant effect on the AEC levels ($F_{89,2} = 46.22$ $p < 0.001$). There was a slight decrease in the AEC levels for both groups B (15 minute simulation) and C (30 minute simulation) following repeated dredge simulation 3 days in a row but it was not statistically significant. All the experimental scallops significantly recovered after three days.

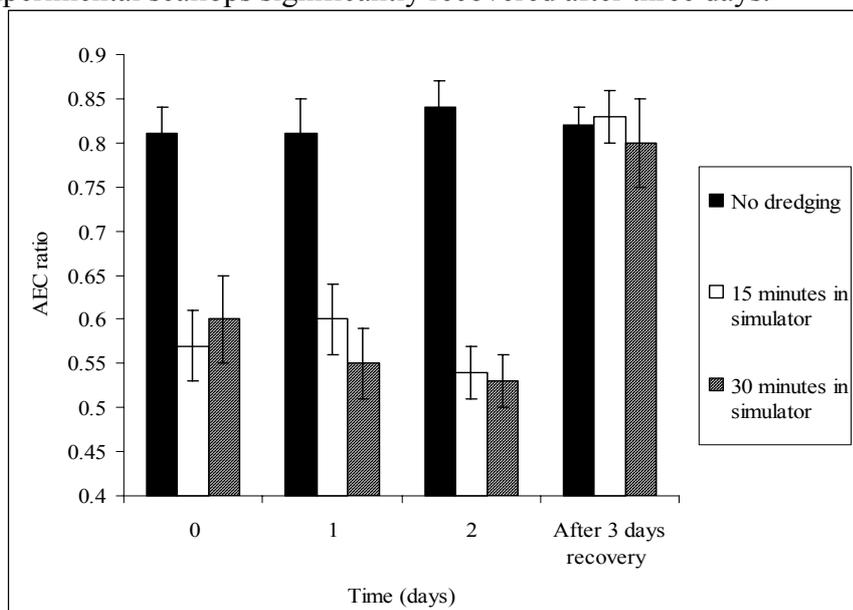


Figure 5. The effects of daily simulated dredging (22rpm) on the AEC ratio (mean \pm s.e.) of scallops.

The behaviour results are shown on Figure 6. The results displayed a similar pattern to the previous experiment. Scallop activity was similar for treatments B and C after 24 and 48 hours but decreased after three days of successive dredge simulation. In treatment C (30 minutes daily simulation) the activity of the scallops did not increase further over the following days of recovery.

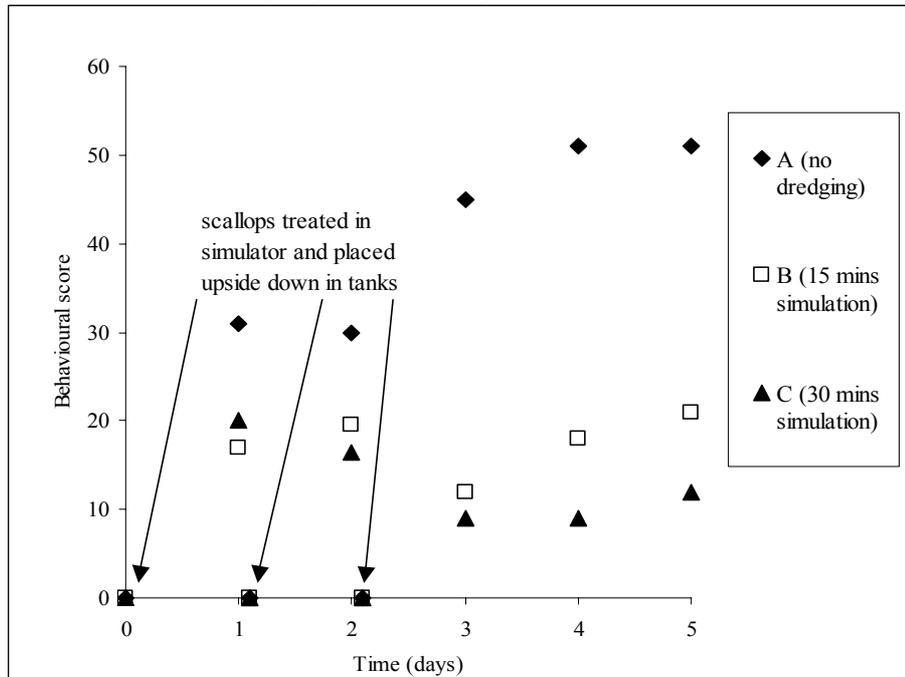


Figure 6. Cumulative effects of daily simulated dredging (22rpm) on the behavioural score of undersized scallops (score = sum of not righted (0), righted (1), semi-recessed (2) and recessed (3) animals, n=30).

Discussion

In this study the AEC levels significantly decreased in the striated muscle after 15 minutes dredge simulation at 11rpm and 22rpm respectively. However, after 30 minutes dredge simulation an AEC decrease was shown but was not significantly lower at both speeds. A similar result was found in an earlier paper by Maguire et al. (2002a) and [Section 3.1.3.2](#), AEC levels decreased in the striated muscle to approximately 0.5 after 15, 30 or 45 minutes dredging and simulated dredging. In this paper the length of tow between 15 and 45 minutes did not have an effect on AEC levels but a difference in AEC levels was found in dredged scallops of different sizes and scallops that were dredged followed by an extended emersion period. Shorter periods of simulated dredging produced a lower reduction in AEC ([Section 3.1.3.2](#)). To our knowledge this previous work is the only other study that used AEC as a stress index in relation to dredging. However, AEC has been used by several authors in relation to other stressors e.g. emersion and anoxia. Many of these studies have shown a gradual decrease in the AEC level over time due to an acute stress (Wijsman 1976b; Skjoldal and Bakke 1978; Ebberink et al. 1979.; Verschraegen et al. 1985.; Moal et al. 1989b; Veldhuizen-Tsoerkan et al. 1991, Madureira et al. 1993.; Lim et al. 1996; Isani et al. 1997; Page et al. 1998).

Duncan (1993) suggested that AEC levels below 0.55 in scallops resulted in eventual mortality even if the animals were returned to optimal conditions. However in this study acutely stressed scallops (from the higher speed treatments, recorded AEC levels <0.55) recovered quickly and had AEC levels of >0.8 after only 6 hours. In a previous study by Maguire et al. (2002a) and [Section 3.1.3.2](#), AEC levels of dredged scallops returned to normal after 3 days in optimal conditions. Similarly, Livingstone et al. (1981) measured stress and recovery in the scallop *Placopecten magellanicus*, after 15 minutes recovery the AEC level in the test animals increased from 0.76 to 0.90 in only 15 minutes. Maguire et al. (1999a) studied the effect of dry transport on juvenile scallops (*P. maximus*) and after 12 hours of transport, AEC levels had decreased to 0.42 in the test scallops. Recovery was rapid and the AEC level in the scallops increased to 0.68 after two hours recovery and 0.82 after 24 hours.

Similarly, in the behavioural test, differences in activity were related to the speed of tow but not the length of tow. Similarly, Maguire et al. (2002a) and [Section 3.1.3.2](#) found that dredged scallops were less active than control scallops but no difference was found between dredged treatments -15 or 30 minute tows- although shorter periods of stress induced less effect on the AEC level. Sokolowski et al. (1999), related reduced burrowing in the Baltic clam *Macoma balthica* to exposure to copper. They observed that increased copper concentration eventually affected metabolic activity and a higher use of the clams energy reserve. Similarly, in this study a decrease in behavioural activity and AEC level was demonstrated by the test scallops. It is vital that discarded scallops are able to swim and recess as quickly as possible to avoid predation (Minchin et al. 2000). Caddy (1973) observed that predatory fish and crabs were attracted to dredge tracks within one hour of dredging at densities of up to 30 times greater than in areas outside the tracks. Therefore undersized stressed or damaged scallops probably have a reduced chance of survival when returned to the seabed particularly if dredged at a high speed.

Each application of an additional dredge disturbance to the test scallops in Experiments 2 and 3 did not have a significant cumulative effect on the AEC level or behavioural activity of the animals. The period between disturbances was long enough for the scallops to recover significantly from the previous stress. After three days of simulated repetitive dredging at 11rpm the AEC had decreased to 0.59 ± 0.04 (15 minutes simulation) and 0.57 ± 0.03 (30 minutes simulation) respectively. However, after only two hours the AEC had increased to 0.81 ± 0.03 and 0.71 ± 0.03 respectively for both treatments. Similarly, the re-cessing and righting ability of the scallops was reduced after each disturbance but was not compounded following daily simulated dredging. Cumulative physiological stress responses in juvenile chinook salmon (*Oncorhynchus tshawytscha*) have been studied by Barton et al. (1986), who demonstrated a stepwise pattern of increased plasma cortisol and glucose concentration in fish subjected to a 30 second handling stress applied every three hours for six hours. The concentrations of plasma cortisol and glucose were still significantly altered 12 hours after the third disturbance but had returned to normal within 6 hours after a single handling. However, in the salmon study the fish had a shorter recovery period between disturbances (3 hours) than in this study where the recovery period was 24 hours. Therefore the stress effect of a single dredging disturbance or repeated dredging has the same effect on the energy charge and behaviour of scallops if the animals have 24 hours between disturbances. However, it

is not known whether a cumulative stress effect might occur in scallops if the recovery period was shorter.

This study has shown that undersized dredged scallops are stressed and may be less able to avoid predation when discarded and mortality levels may even be higher than previous estimates. The tow speed of dredging has an important impact on this mortality level. Scallops dredged at lower speeds recovered three times faster than animals dredged at higher speeds. Maguire et al. (2002a) and [Section 3.1.3.2](#) suggested that this additional mortality be taken into account during stock assessment using the catch mortality model developed by Chopin et al. (1996). At the moment scallops stocks are declining rapidly and the amount of post-dredge mortality needs to be reduced. One way to achieve this could be to reduce the towing speed of boats.

3.1.3.5 ACUTE BIOLOGICAL RESPONSES AS INDICES OF SHORT-TERM STRESS DUE TO HYDRAULIC DREDGING IN THE STRIPED VENUS CLAM *CHAMELEA GALLINA*

Introduction

By the end of the 1960s, fishing for the striped venus clam *Chamelea gallina* along the western coast of the Adriatic was significantly changed by the introduction and rapid spread of a powerful new tool, the hydraulic dredge. The most evident short/middle-term effect was an immediate increase in landings, which peaked at 100,000 metric tons per year in the early 1980s (Froglia 1989).

Only more recently, after dramatically decreased harvests had become established over time, overfishing of the resource became apparent and was finally recognised as at least one of the factors responsible for the observed decline in natural clam populations. Simultaneous failure in recruitment and widespread outbreaks of massive mortality in several clam beds along the western coast of the Adriatic have also been indicated as concurrent factors (Froglia 1989. ; Froglia 1998).

As a consequence, mainly within the framework of national research projects, much local effort has been applied to the urgent need to adopt regulations for eco-sustainable management of this important resource, monitoring abundances of the clam population, evaluating its dynamics, and investigating the causes of repeated outbreaks of mass mortalities (Froglia 1998).

Many studies have demonstrated the negative long-term effects of dredging on the sea bottom, related to both abiotic and biotic compartments (Caddy 1973; de Groot 1984; Pranovi and Giovanardi 1994; Currie and Parry 1996; Kaiser et al. 1996; Giovanardi et al. 1998; Pranovi 1998; Hall-Spencer et al. 1999b; Veale et al. 2000b). However, the use of biomarkers to describe the state of well-being of natural bivalve populations and to evaluate the effects of hydraulic dredging as a whole on the biological response of the target species and/or the by-catch, has been a less popular approach (Viarengo et al. 1991).

More recently, this project offered the opportunity to focus on the repeated disturbance caused by commercial hydraulic dredges on the target species, the hypothesis being that dredging directly damages clam populations. If commercial hydraulic dredging has significantly negative effects, substantially weakening the exploited clam population, it may be considered at least a partial cause of increased mortality and a factor contributing towards abnormal susceptibility to normal fluctuations in environmental parameters.

In the present work, an estimate of dredging impact was performed evaluating the effects of the acute mechanical stress, which clams experience during fishing operations, by applying biological indices at both cellular and organism level. In particular, at cellular level, bio- and cyto-chemical parameters were determined, whilst the behavioural assay based on the reburrowing ability and the shell damage were evaluated at organism level.

Considering the various sources of stress during commercial fishing operations, we did seasonal measurements on clam samples, collected using three different fishing methodologies: low water pressure, high water pressure with sorter, and high water pressure excluding the mechanical sieve. This procedure allowed to distinguish between the effect of hydraulic dredging and that one of mechanical sieving on both commercial and under-size clams. At the same time, different experiments, simulating fishing mechanical stress, were carried out in the laboratory.

Methodologies

Laboratory studies

Morphofunctional and cytoenzymatic characterization of *Chamelea gallina* hemocytes
Clams (shell length 20 to 30 mm) were collected with a hydraulic dredge at Venice Lido, in the west coast of the Northern Adriatic Sea in front of the Venice lagoon. They were kept in the laboratory for 3 days before experiments, in tanks filled with aerated seawater renewed every other day (salinity 35‰ ± 1; temperature 16°C ± 1) and fed with microalgae (*Isochrysis galbana* and *Phaeodactylum tricornutum*).

Hematocrit value and hemocyte culture

Hemolymph was collected from the anterior adductor muscle with a 1-ml syringe containing L-cysteine 10 mM (pH 7.5). The hematocrit value, i.e., the concentration of circulating cells, was obtained by counting cell concentrations in a Buerker hemocytometre. To prepare cell cultures, pooled hemolymph was centrifuged at 780 x g for 10 min and pelleted cells were resuspended in FSW, to a final concentration of 10⁶ cells/ml. Sixty µl of hemocyte suspension were placed in culture chambers made by glueing Teflon rings (15 mm internal diameter, 1 mm thick) on glass slides. Washed coverslips were laid over the Teflon rings smeared with vaseline to touch the drop of cell suspension. Culture slides were kept upside down for 30 min at room temperature to allow cells to adhere to coverslips. Hemocytes were then fixed for 30 min at 4°C in a solution of 1% glutaraldehyde and 1% saccharose in FSW, washed in phosphate-buffered saline (PBS, pH7.2) and stained for 10 min with a 5% Giemsa aqueous solution. Coverslips were then washed, mounted on glass slides with an aqueous medium (Acquovitrex, Carlo Erba), and cells were observed under a Leica DMLB lighy microscope connected to an image analysis equipment (Casti Imaging).

Hemocyte morphology

Hemocyte monolayers were fixed for 30 min at 4°C in a solution of 1% glutaraldehyde in FSW containing 1% saccharose. They were then washed in PBS and stained for 4 min in 10% Giemsa solution. Finally, hemocytes were mounted in Acquovitrex. Observation was carried out with a light microscope. Hemocytes were incubated in Neutral Red (Merck) solution (8 mg/l in FSW) and observed immediately to identify lysosomal compartments.

Adhesion assay

The number of hemocytes in 30 fields were evaluated and compared with that of monolayers on poly-L-cysteine treated coverslips to assess the adhesion index, being the percentage of adhering cells on untreated coverslips. The hemocytes were stained with Giemsa, mounted and observed with image analysis equipment.

Phagocytosis assay (Ballarin et al. 1994).

Hemocyte monolayers were incubated for 60 min at room temperature with 60 µl of either a yeast (*Saccharomyces cerevisiae*) suspension in FSW (yeast/hemocyte ratio = 10:1) or 1% carmine in FSW and incubated. Carmine was also pre-incubated for 30 min in CFH to examine opsonic activity. After incubation, uningested particles were washed out by dipping in FSW. Hemocytes were finally fixed, stained with Giemsa solution, mounted and observed under light microscope. Attached foreign particles were enumerated. The percentage of hemocytes with ingested yeast cells was expressed as the phagocytic index.

Cytoenzymology

Hemolymph was collected in Alsever's solution pH 7.5, and 300 µl of hemocyte suspension (10^5 cells/ml) were spun at 18 x g for 3 min (Cytospin 3, Shandon), air dried for 20 min and fixed in glutaraldehyde. Specific cytochemical reactions were employed to demonstrate the presence of several hydrolytic and oxidative enzymes on hemocytes. Substrates were omitted in control slides. For β -glucuronidase and acid phosphatase assays, cells were also pre-incubated for 60 min with 3 µm latex beads.

Acid Esterase (Lojda 1977). Hemocytes were washed with phosphate-citric acid (0.1M, pH 5.5) and incubated for 16h at 4°C in the following reaction mixture: 3 mg naphthol acetate (Sigma) dissolved in 500 µl acetone, 250 µl solution A (0.4 g new fuchsin (Sigma), 2 ml HCl 36%, 8 ml distilled water), 250 µl solution B (4% NaNO₂ in distilled water), 19 ml phosphate-citric acid buffer. Hemocytes were then washed with distilled water and mounted. Enzyme-reactive sites were stained pink-brown.

Acid Phosphatase (Lojda, 1979). Hemocytes were washed in sodium acetate (0.1M, pH 5.2) for 10 min and incubated for 3h at 37°C in a reaction mixture made by dissolving 10 mg naphthol AS-B phosphate (Sigma) in 400 µl dimethylformamide (DMF), 400 µl solution A, 400 µl solution B and 20 ml of sodium acetate buffer (0.1 M, pH 5.2). After incubation, hemocytes were washed in the buffer solution and mounted. Enzyme-reactive sites were stained red.

Alkaline Phosphatase (Burstone, 1962). Hemocytes were washed with Tris-HCl (0.1M, pH 9) for 10 min and incubated for 2h at 37°C in a reaction mixture similar to that used for acid phosphatase detection, though containing 20 ml of Tris-HCl buffer instead of sodium acetate. Hemocytes were then washed for 10 min in Tris-HCl and mounted. Enzyme-reactive sites were stained red.

Arylsulphatase (Goldfisher 1965). Hemocytes were washed for 10 min in sodium acetate, and incubated for 2h at 37°C in 0,16 g of p-nitrocatecholsulphate (Sigma) in 4 ml distilled water, 12 ml sodium acetate buffer, 4 ml 8% PbNO₃. After incubation, hemocytes were washed with distilled water and then with ammonium sulphide solution for 2 min. Finally, they were washed with distilled water and mounted. Enzyme-reactive sites were stained brownish-black.

Chloroacetyl Esterase (Moloney et al. 1960). Hemocytes were washed in PBS for 10 min and incubated for 1h at room temperature in a reaction mixture made by solving 6 mg naphthol chloroacetate (Sigma) in 1 ml DMF, finally added to 19 ml PBS containing 20 mg Fast Blue B (Fluka). They were then washed in PBS and mounted. Enzyme-reactive sites were stained blue.

STOP

β -Glucuronidase (Hayashi et al. 1964). Hemocytes were washed in sodium acetate for 10 min and incubated for 3h at 37°C in the following reaction mixture: 4 mg naphthol AS-BI β -glucuronide (Sigma) dissolved in 250 μ l DMF, 400 μ l solution A, 400 μ l solution B and 20 ml of sodium acetate buffer. After incubation hemocytes were washed in sodium acetate buffer and mounted. Enzyme-reactive sites were stained red.

Nonspecific Esterase (Gomori 1948). Hemocytes were washed with PBS for 10 min and incubated for 3h at 4°C in the following reaction mixture: 10 mg naphthol AS-D acetate (Sigma) dissolved in 1 ml acetone, added to 49 ml PBS containing 80 mg Fast Blue B (Sigma). Hemocytes were then washed in PBS and mounted. Enzyme-reactive sites were stained blue.

5'-Nucleotidase (Wachstein and Meisel 1957). Hemocytes were washed for 10 min in Tris-maleate (0.2M, pH 7.2) and incubated for 2h at 37°C in 20 mg adenosine-5'-monophosphate (AMP, Sigma) solved in 22 ml of distilled water, 20 ml Tris-maleate buffer and 3 ml of 2% aqueous solution of PbNO₃, containing 5 ml MgSO₄. Hemocytes were then washed with distilled water and immersed in 1% ammonium sulphide solution for 2 min. Finally, they were washed with distilled water and mounted. Enzyme-reactive sites were stained black.

Peroxidase (Graham and Karnovsky 1966). Hemocytes were washed in PBS for 10 min and incubated for 2h at 37°C in 0.5 mg/ml of 3-3'diaminobenzidine tetrahydrochloride (DAB, Sigma) in distilled water containing 0.02% H₂O₂. Then hemocytes were washed in distilled water and mounted. Enzyme-reactive sites were stained brown.

Phenoloxidase (Hose et al. 1987). Hemocytes were washed with PBS for 10 min and incubated for 30 min at 37°C in a saturated solution of L-dihydroxyphenyl-L-alanine (L-DOPA). They were then washed with PBS and mounted. Enzyme-reactive sites were stained black.

All experiments were repeated four times, using pools of hemolymph from 10 clams each. The percentage of positive hemocytes and the phagocytic index (percentage of phagocytic hemocytes) were evaluated on 1500 cells per pool.

Anoxic stress

Since caught clams remain exposed to air during fishing activity before re-immersed in seawater, it was important in our opinion to evaluate the effect of anoxic stress on immunological parameters.

Exposure to air

According to literature data regarding survival in air of *C. gallina* (LT₅₀ at 18°C is 4 days; (Brooks et al. 1991), we exposed clams to air, arranging them in closed boxes with 100% humidity at a temperature of 16°C \pm 1. Hemocytes were then collected from 10 specimens and pooled. Hemocyte samples were obtained from animals subjected to 0, 1, 2, 3 and 4 days of anoxia respectively. Clams were considered dead when an external stimulus did not cause any shell closing; dead clams were removed immediately. We had chosen three types of indicators of stress based on the

hemocytes functionality: the hematocrit value, β -glucuronidase and acid phosphatase activities, the phagocytic index and the adhesion index. Due to the low number and poor condition of collected hemocytes, only the hematocrit value was reported for the fourth day.

Recovery after air exposure

To evaluate the ability of clams to recover after anoxic stress, clams were exposed to air for 1, 2, and 3 days before re-immersing them in tanks with water and food, and followed for 1, 2, 3 and 4 days to study recovery. Hemolymph was collected immediately (T0) after exposure, and in the subsequent four days after re-introduction in aquaria (T1, T2, T3, and T4, respectively). Hematocrit, acid phosphatase and β -glucuronidase activities were assessed.

We analysed 3 pools of 10 clams for each experimental point. Acid phosphatase and β -glucuronidase activity and the phagocytic indexes were evaluated on 1500 cells per pool. Adhesion indexes were compared by χ^2 test and G-test (Sokal and Rohlf 1981).

Mechanical stress experiment on commercial size clams***Immunological parameters***

Clams from commercial dredging (about 25 mm) were acclimatised in the laboratory for 4 days in tanks containing sea water at a temperature of $16^{\circ}\text{C} \pm 1$ and fed with *Isocrysis galbana* before performing experiments.

Mechanical stress conditions were reproduced using a vortex mixer: clams were arranged in a box, shaken for 6 min at the maximum speed (40 hertz), kept 30 min out of water and finally processed. Timing for stress exposure was chosen taking into account that, immediately after stress, clams need to be processed alive and for a long time.

Samples of both control and stressed clams have been also maintained for 24h at a temperature of $+4^{\circ}\text{C}$. This approach is needed because of some constraints in field studies, when clams arrive at the laboratory in the afternoon: due to the long experimental time required for measurements, an immediate processing of collected samples was impracticable, and clams were refrigerated until the following morning. To determine recovery times of stressed organisms, clams were replaced in aquarium after mechanical shaking for 12h and 24h.

These experiments allowed us to verify the effect of mechanical stress and low temperature on hematocrit value, phagocytic index, adhesion index, acid phosphatase and β -glucuronidase activities.

We analysed 3 pools of 10 clams for each experimental point. Acid phosphatase and β -glucuronidase activities and the phagocytic indexes were evaluated on 1500 cells per pool. Adhesion indexes were compared by χ^2 test and G-test (Sokal and Rohlf 1981).

Biochemical and behavioural parameter

As for Adenylic Energetic Charge (AEC) and re-burrowing, mechanical stress due to dredging was simulated in the laboratory shaking manually two sub-samples of clams of 200 individuals, (from a sample commercially dredged at Jesolo, same station as in the field experiment). The shaking, 3 minutes long, was repeated every 30 min up to 6h. The treatment was performed inside a box as it was rotated (with a frequency of about 60 rpm). During each recovery period (of about 30 min) the animals were kept in aerated aquaria.

One of the two sub-samples was processed for the different analyses at the end of the treatment, and the other one was kept in aquarium for the following 24h before being processed. As controls, both clams collected manually and clams from the same dredged sample used for the experiment, but not mechanically stressed, were used.

Ten clams were analysed for AEC, whereas 20 clams were observed for the re-burrowing rate.

Mechanical stress experiment on under-sized clams

A laboratory experiment was carried out in order to evaluate the effects of dredging and mechanical sorting on under-sized clams which are rejected into the sea bed after passing the sieve during the normal fishing activity.

Under-sized clams (13-19mm), commercially dredged and acclimatised for 4 days in laboratory, were shaken for 10 min using a vortex at maximum speed (40 hertz), the same operation being repeated once a day in the following three days. AEC, re-burrowing and damage index were evaluated.

Ten clams were analysed for AEC, whereas 20 clams were observed for the re-burrowing rate.

Field study

Four seasonal samplings on natural *C. gallina* beds were carried out in February, May, July and October 2000, with the aim to evaluate the effects of different fishing systems on a seasonal basis.

Clam samples were collected in two fishing areas along the west coast of the North Adriatic Sea: Jesolo and Lido at about 5 m depth, using four fishing methodologies:

- dredging at high water pressure (~ 2.5 atm) and using a mechanical sieve for sorting (as in commercial fishing), to verify the seasonal responses of clams to the fishing system (HPS samples);
- dredging at high water pressure without sorting, to detect the responses effectively due to the hydraulic dredging system (HP samples);
- dredging at low water pressure (~ 1 atm) without sorting, to get a less manipulated sample (LP samples);
- manually by divers, to obtain a control sample (M samples, not collected in February at both sites and in July at Jesolo).

With the aim to standardise the biological responses, commercial size samples were used in all analyses.

During this study we had to face with a number of problems regarding the collection of control samples by divers. In February and October we were not able to collect any due to very low water temperature; and, more generally, we had to use samples made up by heterogeneous size clams, as it was difficult to reach the minimum number of commercial size clams, when we needed a control sample.

The following indices were studied:

- haematocrit,
- phagocytic index,
- acid phosphatase activity index,
- β -glucuronidase activity index,
- AEC,
- re-burrowing,

During the samplings of July and October 2000, the possible stress due to dredging and mechanical sorting were evaluated also on under-sized clams. Animals (13-19 mm length) were collected from *by catch* and the following measurements were performed: AEC and re-burrowing time.

We analysed 3 pools of 10 clams for each experimental point in the immunological measurements. Acid phosphatase and β -glucuronidase activity and the phagocytic indexes were evaluated on 1500 cells per pool.

Ten clams were analysed for AEC, whereas 20 clams were observed for the re-burrowing rate.

Stress indices

Haematocrit value and haemocyte culture

Haemolymph was collected from the anterior adductor muscle with a 1-ml syringe containing L-cysteine. The haematocrit value, i.e., the concentration of circulating cells, was obtained by counting cell concentrations in a Buerker haemocytometre. To prepare cell cultures, pooled haemolymph was centrifuged at $780 \times g$ for 10 min and pelleted cells were resuspended in FSW, to a final concentration of 10^6 cells/ml. Sixty μ l of haemocyte suspension were placed in culture chambers for 30 min at room temperature to allow cells to adhere to coverslips. Haemocytes were then fixed for 30 min at 4°C in a solution of 1% glutaraldehyde and 1% saccharose in FSW, washed in PBS and stained for 10 min with a 5% Giemsa aqueous solution. Coverslips were then washed, mounted on glass slides with Acquovitrex and cells were observed under a light microscope connected to an image analysis equipment (Casti Imaging). Statistical analysis of hematocrit values was performed using Student's t-test (Sokal and Rohlf 1981).

Phagocytosis assay

Haemocyte monolayers were incubated for 60 min at room temperature with a yeast (*Saccharomyces cerevisiae*) suspension in FSW (yeast/haemocyte ratio = 10:1). After incubation, uningested yeast cells were washed out by repeatedly dipping in FSW. Haemocytes were then fixed, stained, and observed under the light microscope. The percentage of haemocytes with ingested yeast cells was expressed as the phagocytic index (Ballarin et al. 1994). Phagocytic indexes were statistically compared by the G-test.

Enzyme activity assay

Haemolymph was collected in Alsever's solution, and 300 μ l of haemocyte suspension (10^5 cells/ml) were cytopun at $18 \times g$ for 3 min, air-dried for 20 min and fixed as described above. Haemocytes were washed in sodium acetate buffer for 10 min and incubated for 3 h at 37°C in the specific reaction mixture for acid phosphatase (Lojda et al. 1979) and β -glucuronidase (Hayashi et al. 1964) activities. After incubation, haemocytes were washed in sodium acetate buffer, coverslips mounted, and observed with the image analysis equipment. Enzyme-reactive sites were stained red. Enzyme activity indexes were defined as the fraction of cells showing positivity for the enzymes as determined by using an image analysis equipment. Enzyme activity indexes were statistically compared by the G-test.

Adenylate Energy Charge (AEC)

The effect of a short term stress on the biochemistry of the animal can be measured by its level of AEC, which is a measure of its energetic status. (see [Section 3.1.3.2-4](#)) The striated muscle of 10 specimens was removed and frozen in liquid nitrogen until the extraction procedure. Ice cold TCA (1ml, 0.5 M) was added to each sample, mixed in a Potter homogenizer, and centrifuged for 10 min at 12000 rpm. The supernatant was neutralised with fresh amine freon solution 0.5 M and stored at -18°C until High Performance Liquid Chromatography (HPLC) analysis (Moal et al. 1989a).

The nucleotides were separated in a C18 column and detected using an ultra-violet light at 254 nm. A NaH₂PO₄ buffer pH 6 containing 0.005M tetrabutylammonium and 5% methanol was used to elute the nucleotides. The separation took 20 minutes at flow rate of 1 ml/min.

AEC was calculated using the following equation: $AEC = (ATP + 0.5ADP) / (ATP + ADP + AMP)$.

AEC values were statistically compared using ANOVA.

Reburrowing rate

This technique evaluates the re-burrowing time of the bivalves once relocated on aquarium after having been captured. This is carried out in order to have an indication of the indirect mortality resulting from an increased exposure to predators (Phelps 1989).

Twenty individuals were placed in aquaria (42x24x26 cm) with a fine sediment obtained by sieving natural beach sediment. The aquaria were kept in a thermostatic chamber simulating original water temperatures. A web cam was employed for the continuous observation of the re-burrowing behaviour. The camera was set for recording a frame every 30 min over a time period 10h long. Subsequently the RT50 (the time requested for the reburrowing of the 50% of the sample) was calculated according to the method of Kaplan and Meier (1958) and the reburrowing curves were compared using the Gehan and Wilcoxon test (Gehan 1965), both included in the software STATISTICA. Moreover the % of reburrowed clams over a fixed time period was also determined.

Multivariate analysis

The Principal Component Analysis (PCA) was performed on all the above, considered biological parameters with the aim of evaluating in a scatterplot possible site and pressure effects on clam responses (Lagonegro and Feoli 1986).

Results

Laboratory studies

*Morphofunctional and cytoenzymatic characterization of *Chamelea gallina* hemocytes*

Two main types of circulating hemocytes were identified: hyalinocytes and granulocytes. Hyalinocytes (79.2%) were generally devoid of granulations and had a large, dark blue basophilic nucleus surrounded by light blue cytoplasm. Granulocytes (16.5%) were smaller than hyalinocytes, with smaller, dark blue nuclei and blue-stained basophilic granules in their cytoplasm. Hemocytes with a strongly basophilic nucleus and a thin, light blue peripheral cytoplasm were also found (4.3%): they probably represent blast cells or undifferentiated hemocytes. Both granulocytes and hyalinocytes can assume a spreading (irregular shape with pseudopodia of varying lengths) or a round morphology (Table 1). Among granulocytes, 81.1% were spreading cells, whereas 18.9% were round cells; and 53.2% of hyalinocytes were spreading cells and 46.8% were round cells. Table 2 reports the mean values of cell (C) and nucleus (N) sizes (longest axis excluding pseudopodia and longest nuclear diameter, respectively), measured by image analysis, and the N/C ratio. Hyalinocytes (C=11.51 μ m, N=4.74 μ m) were bigger than granulocytes (C=9.42 μ m; N=4.48 μ m), but had a smaller N/C ratio. Blasts were the smallest cell type and had the highest N/C ratio. Spreading cell dimensions were more variable.

A low number (<1%) of binucleated circulating cells was also observed.

A haematocrit value (concentration of circulating cells) of 1.2-2.4 10^6 cells/ml, evaluated on pools of 10 clams, shows seasonal variations, being significantly higher in spring and summer (Figure 1).

A low percentage (6.3% \pm 1.3) of hemocytes were able to phagocytise yeast cells, whereas most of them (92.7% \pm 5.4) adhered to glass slides. The adhesion assay proved that all cell types adhere to, and spread on glass coverslips. In phagocytic assay in the presence of carmine particles: 2.8% (\pm 1.01) of granulocytes and 9.8% (\pm 0.2) of hyalinocytes showed phagocytic ability. After pre-incubation with CFH, these percentages increased (phagocytic granulocytes: 10% \pm 1.7, phagocytic hyalinocytes 12.7% \pm 1.5). Most hemocytes contain lysosomes inside their cytoplasm, as revealed by neutral red accumulation. Both granulocytes and hyalinocytes were positive for hydrolytic enzymes (Table 3); 79% of hemocytes were positive for chloroacetyl esterase, 67% were positive for arylsulfatase and 65% were positive for non-specific esterase. Acid phosphatase activity occurred in 38% of cells; the lowest frequency (6%) was observed for β -glucuronidase activity. None of the oxidative enzymes assayed were detected in clam hemocytes. Phagocytic activity changed enzyme levels (Table 4).

The percentage of cells positive for β -glucuronidase significantly increased ($p < 0.001$) after the activation of hemocytes by pre-incubation with foreign particles (latex beads, 3 μ m diameter).

Table 1 Morphological characterization of *C. gallina* hemocytes obtained on Giemsa stained slides: mean percentage of various cell types (+ s.d., n=4).

Hemocytes morphology				
% granulocytes		% hyalinocytes		% blast cells
16.5 ± 1.3		79.2 ± 1.3		4.3 ± 1.1
% spreading	% round	% spreading	% round	
81.1 ± 2.5	18.9 ± 2.5	53.2 ± 2.6	46.8 ± 2.6	

Table 2 Mean values (+ s.d.) of nuclear and cell diameter and their ratio (N/C) in hemocytes of *C. gallina*. (+ s.d., n=sample size).

	Granulocytes (n=100)	Hyalinocytes (n=100)	Blast cells (n=50)	Spreading cells (n=100)	Round cells (n=100)
Cell					
diameter (µm)	9.42 ± 4.05	11.51 ± 3.92	5.97 ± 0.72	10.73 ± 4.27	7.71 ± 1.80
Nuclear					
diameter (µm)	4.48 ± 0.73	4.74 ± 0.90	4.26 ± 0.55	4.60 ± 0.84	3.86 ± 0.64
N/C	0.48 ± 0.15	0.41 ± 0.10	0.71 ± 0.11	0.43 ± 0.12	0.50 ± 0.11

Table 3 Enzymatic activity of *C. gallina* hemocytes: mean percentage of positive cells (\pm s.d., n=4).

	Enzimatic Activity	%granulocytes (mean \pm d.s.)	%hyalinocytes (mean \pm d.s.)
Hydrolase	Arylsulphatase	25.3 \pm 3.1	41.2 \pm 2.2
	Chloroacetyl Esterase	27.9 \pm 1.2	51.3 \pm 0.7
	Acid Esterase	-	-
	Nonspecific Esterases	36.4 \pm 2.5	28.2 \pm 1.8
	5'-Nucleotidase	-	-
	Acid Phosphatase	12.3 \pm 1.0	25.6 \pm 0.6
	Alkaline Phosphatase	-	-
	β -Glucuronidase	2.8 \pm 1.2	3.6 \pm 0.5
Oxidase	Phenoloxidase	-	-
	Peroxidase	-	-

Table 4 Enzymatic activity of *C. gallina* hemocytes: mean percentage of positive granulocytes and hyalinocytes with and without pre-incubation with latex (\pm s.d., n=4). Statistical comparison was performed by t-test (*) $p < 0.001$, n.s. $p > 0.05$).**

Enzimatic Activity	Normal condition		Pre-incubated (30 min) with latex (3μm diameter)	
	%granulocytes (mean \pm d.s.)	%hyalinocytes (mean \pm d.s.)	%granulocytes (mean \pm d.s.)	%hyalinocytes (mean \pm d.s.)
β-Glucuronidase	2.8 \pm 1.2	3.6 \pm 0.5	9.5 \pm 1.8 ***	5.8 \pm 0.6 ***
Acid Phosphatase	12.3 \pm 1.0	25.6 \pm 0.6	13.1 \pm 2.1 n.s.	23.9 \pm 3.2 n.s.

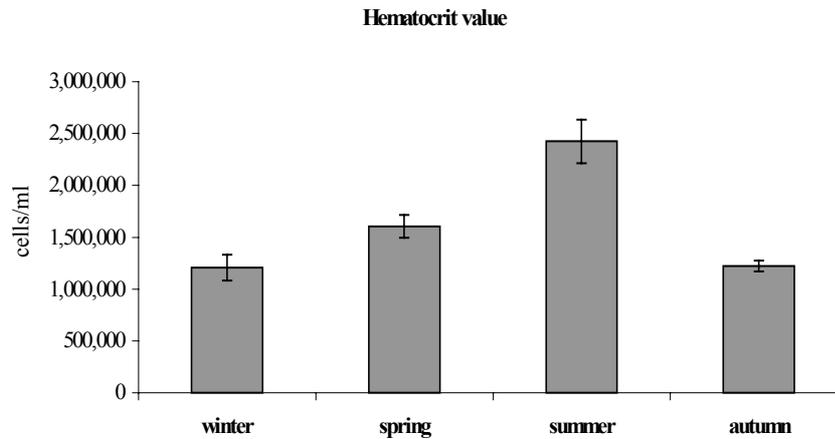


Figure 1 Seasonal variations of hematocrit values in *C. gallina* hemolymph (mean \pm s.d., n=3).

Anoxic stress

Effects of air exposure.

The hematocrit value decreased significantly ($p < 0.01$) after one day of air exposure and continued to decrease progressively until the fourth day (Figure 2a). In parallel, the fraction of acid phosphatase-positive cells decreased significantly ($p < 0.001$) from the first day out of water (Figure 2b), whereas β -glucuronidase activity index (Figure 2c) followed an opposite course, significantly ($p < 0.001$) increasing after the first day of treatment.

The phagocytic index significantly ($p < 0.001$) decreased after the first day of air exposure (Figure 3a). A similar behaviour was observed for the adhesion index, with a significant ($p < 0.05$) decrease in adherent cells from the first day of treatment (Figure 3b). Conversely, a significant ($p < 0.001$) increase in apoptotic cells was observed after the first day out of water, whereas the number of spreading and round cells decreased progressively, that of spreading cells more rapidly than round cells (Figure 3c).

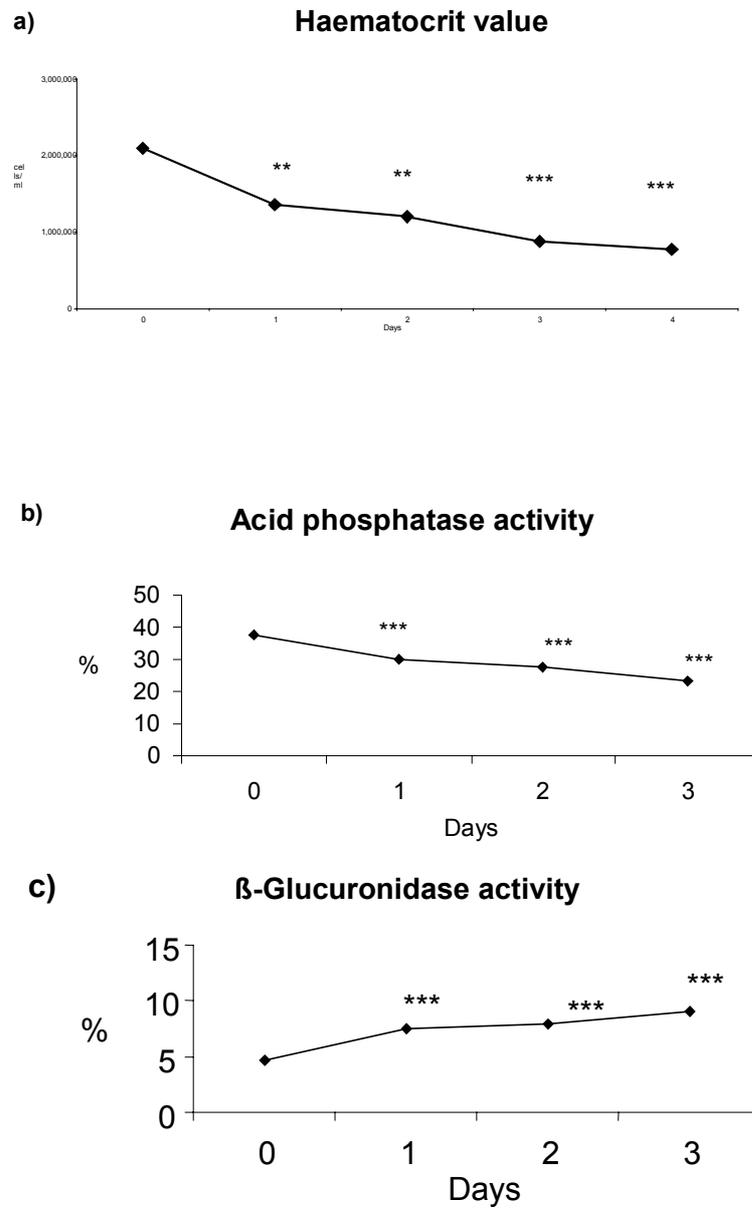


Figure 2 Laboratory study. Results of anoxic stress:

- a) hematocrit value (test-t);
 - b) acid phosphatase activity (G-test);
 - c) β -glucuronidase activity (G-test).
- Statistical significance: *p<0.05 **p<0.01 ***p<0.001

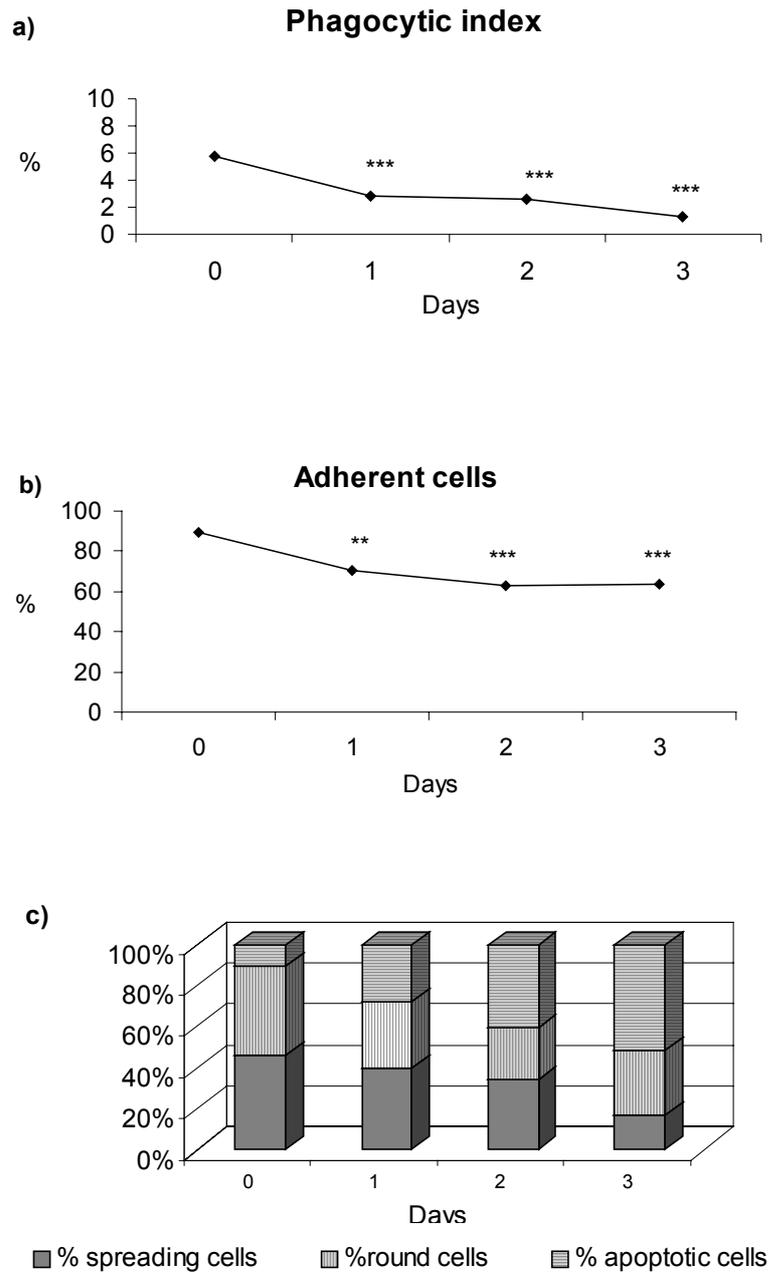


Figure 3 **Laboratory study. Results of anoxic stress.**
a) **Phagocytic index (G-test).**
b) **Adherent cells (G-test).**
c) **Different types of adherent cells.**
Statistical significance: * $p < 0.05$ ** $p < 0.01$ *** $p < 0.001$

Recovery after air exposure.

Clams exposed to air for one day and returned into aquaria showed a progressive increase in hematocrit value becoming significant ($p < 0.01$) with respect to T0 after two days (Figure 4a). At T3, experimental values returned to those of untreated animals. In animals exposed to air for 2-days before re-immersion, a significant ($p < 0.01$) increase in the hematocrit was observed at T1: it remained approximately constant until T3 and then decreased to initial values at T4. In clams left out of water for 3 days, a significant ($p < 0.01$) increase in the hematocrit occurred at T1: the value was similar at T2, whereas at T3 it dropped and was comparable to that obtained at T0; it further decreased and was significantly different from T0 value at T4. Therefore, hematocrit values reached control values after the whole period of recovery only in clams exposed to air for a single day.

The decrease in acid phosphatase activity index observed after air exposure was easily recovered (already at T1) in clams left out of water for only one day. In clams exposed to air for 2 days, the index returned to control values at T1, but decreased again at T2, T3, and T4, reaching values significantly ($p < 0.001$) different from T0. Clams left out of water for 3 days before re-immersion, showed a decrease of the percentage of acid phosphatase-positive cells at T1 ($p < 0.001$) different from T0; no further significant changes were observed at T2, T3, and T4 (Figure 4b).

β -Glucuronidase activity index which, as reported above, significantly increased at T0, returned to control values at T1-T4 in clams kept out of water for 1 day. A similar trend was observed for clams exposed to air for 2 days, whereas in clams exposed for 3 days, the increase in the fraction of positive hemocytes remained compatible to T0 at T1 and T2 and significantly ($p < 0.01$ and $p < 0.05$, in comparison with T0, respectively) decreased, at T3 and T4; these values were always significantly higher than controls (Figure 4c).

The above three indices return to values comparable with controls only in clams exposed to air for 1 day. After 2 days of air exposure, an attempt of recovery was observed (from T1 to T3), but at T4 the values were not significantly different from the exposed ones. Clams exposed to air for 3 days did not recover.

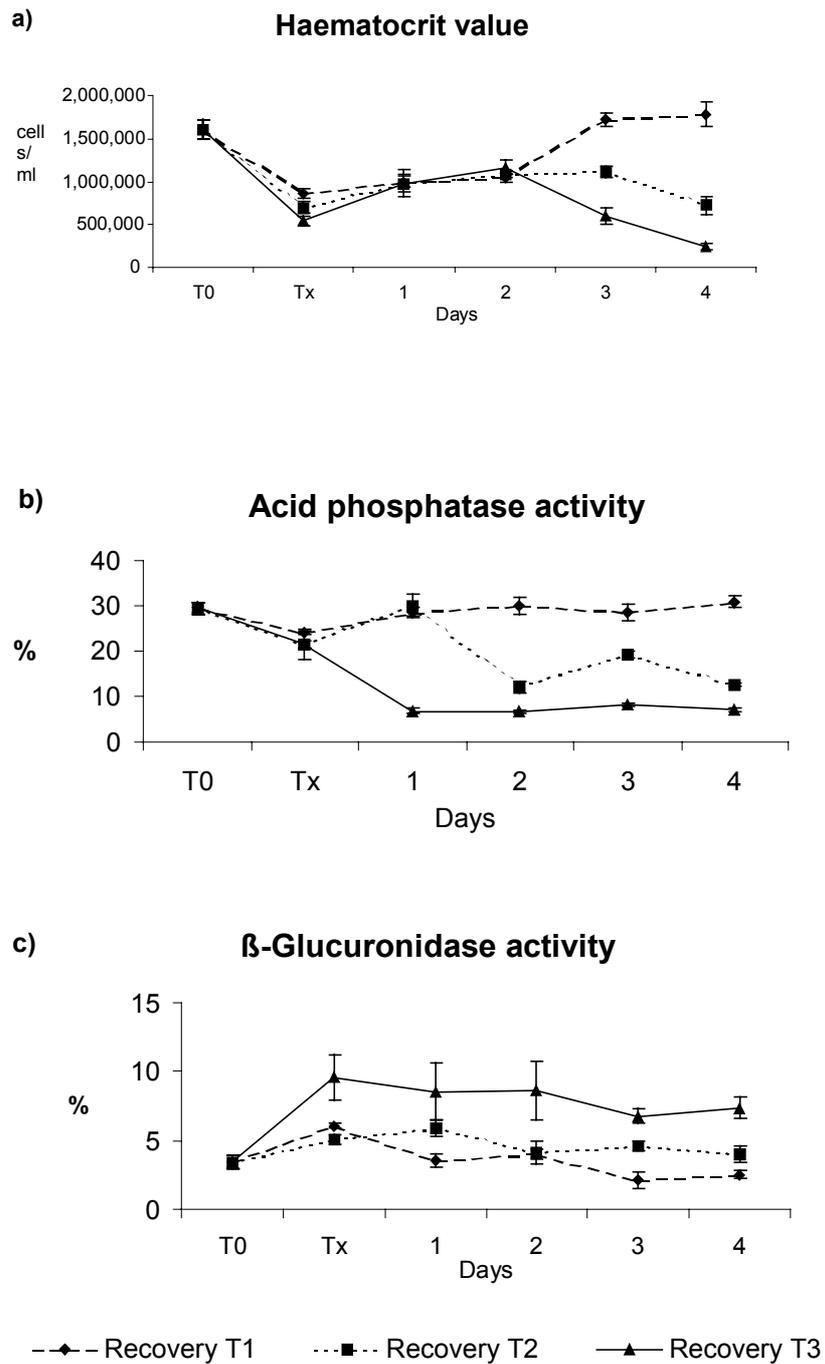


Figure 4 Laboratory study. Results of anoxic stress after a recovery period of 1-2-3 days in seawater.

- a) Haematocrit value (test-t).
- b) Acid phosphatase activity (G-test).
- c) β -glucuronidase activity (G-test).

Mechanical stress experiment on commercial size clams

Immunological parameters

A similar pattern was observed in the haematocrit value, phagocytic index and acid phosphatase activity, showing a minimum in treated and 24h refrigerated clams and an increase up to control values after the recovery periods (Figures 5a,b 6b). In contrast β -glucuronidase activity increased significantly ($p < 0.01$, G-test) in stressed clams and decreased after the recovery period, reaching control values only after 24h of recovery (Figure 6a).

A significant decrease of adherent cells ($p < 0.001$, G-test) was observed after the mechanical stress, followed by a significant recovery after 12h. An increase of apoptotic and round cells and a decrease of spreading cells were noticed in the treated clams. After a recovery of 24h, the frequencies of apoptotic and round cells were lower than in the control, whereas spreading cells increased until values three times higher than in the control (Figure 7).

All these indices showed significant variations in stressed clams compared with controls. Refrigeration, used to store clams overnight before processing seasonal sampling, increased the difference with respect to controls. A full recovery is however shown, in all the assayed indices, after 12h in seawater. The adhesion index gives no additional information compared with the others immunological parameters.

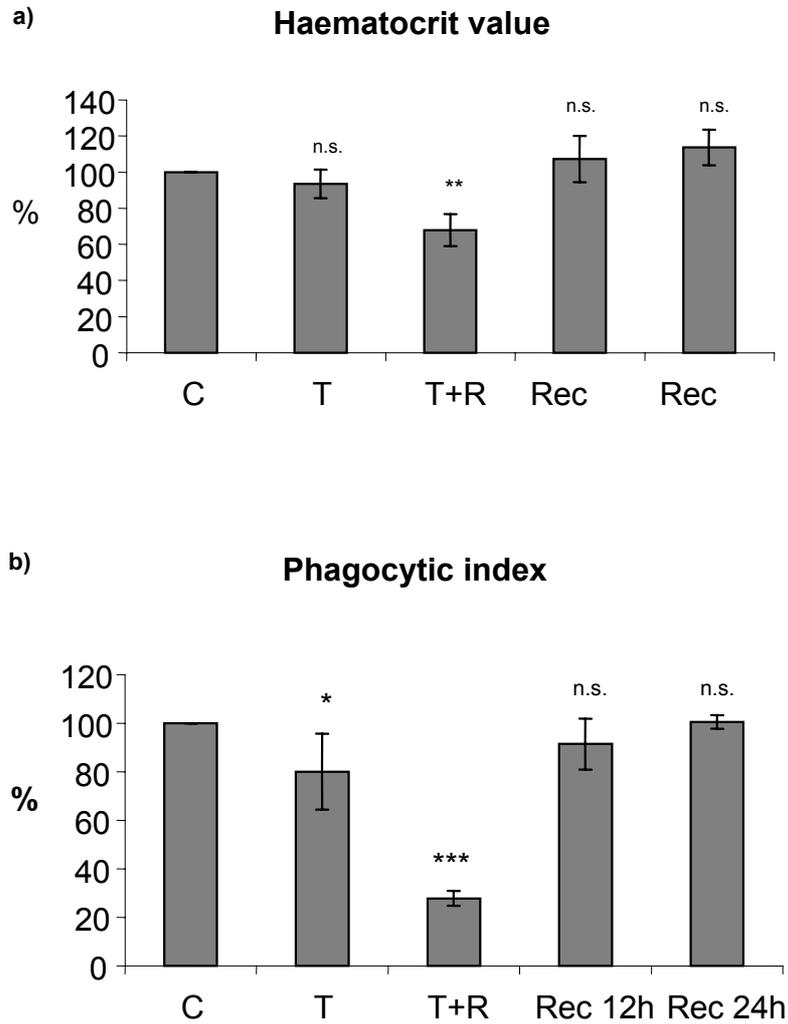


Figure 5 Laboratory study. Results of mechanical stress.

a) Haematocrit values (test-t).

b) Phagocytic index (G-test).

C = control; T = treated; T+R = treated and refrigerated.

Rec 12h = recovery after 12 hours; Rec 24h = recovery after 24 hours.

Statistical significance: *p<0.05 **p<0.01 *p<0.001**

Each value is compared with 100% control.

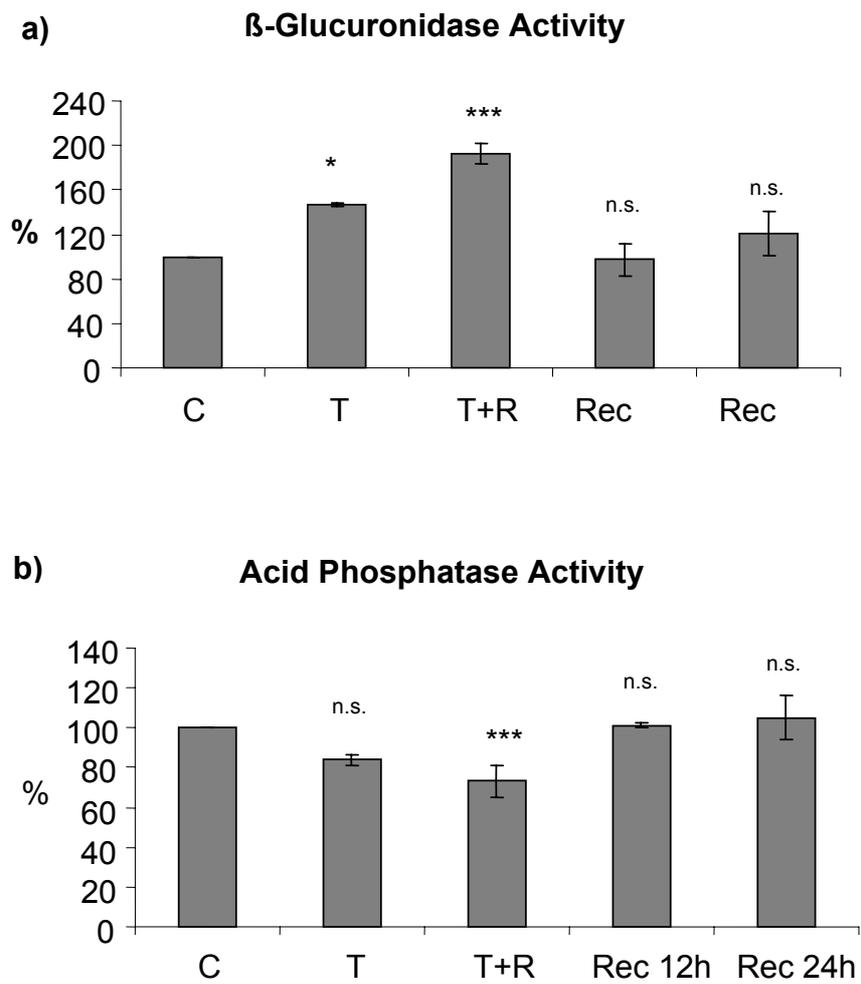


Figure 6 Laboratory study. Results of mechanical stress.

a) β -glucuronidase activity (G-test).

b) Acid phosphatase activity (G-test).

C = control; T = treated; T+R = treated and refrigerated.

Rec 12h = recovery after 12 hours; Rec 24h = recovery after 24 hours.

Statistical significance: * $p < 0.05$ ** $p < 0.01$ *** $p < 0.001$

Each value is compared with 100% control.

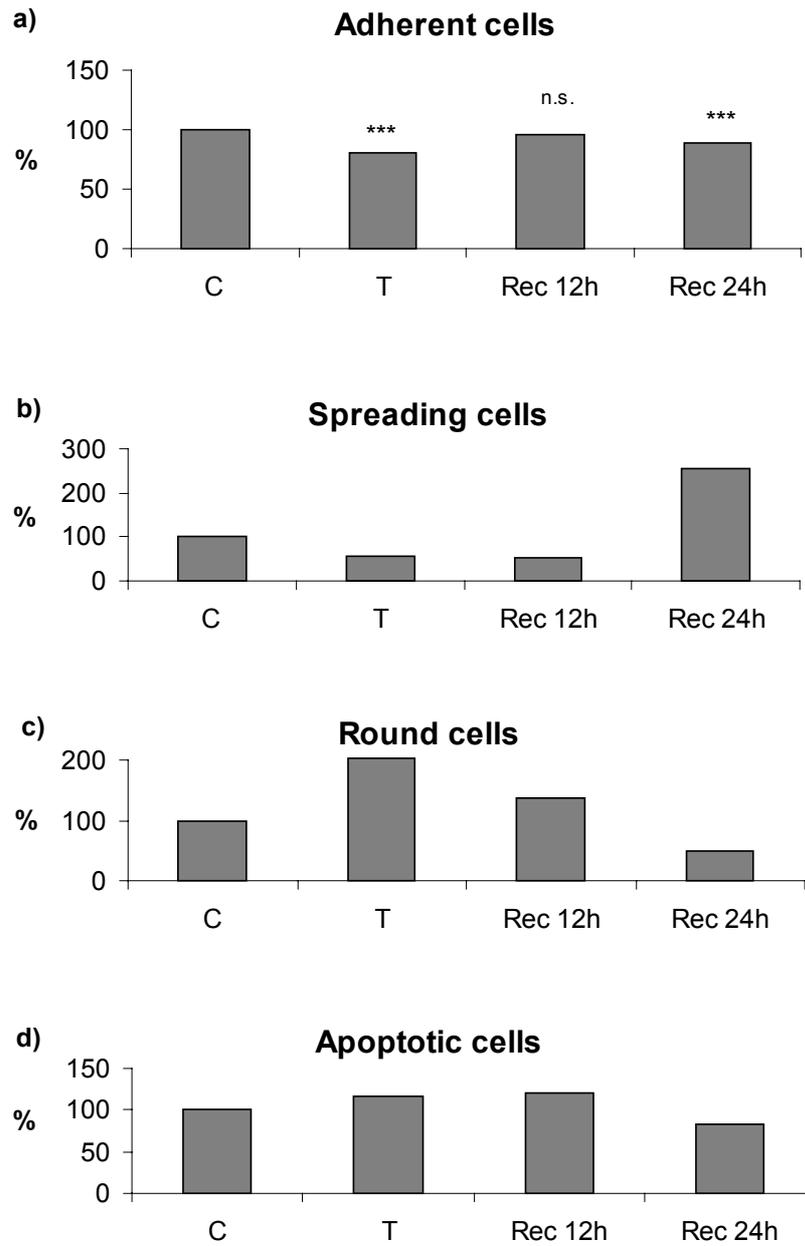


Figure 7 Laboratory study. Results of mechanical stress.
a) Adherent cells (G-test).
b-c-d) Frequencies of different cell types.
C = control; T = treated; T+R = treated and refrigerated.
Rec 12h = recovery after 12 hours; Rec 24h = recovery after 24 hours.
Statistical significance: *p<0.05 **p<0.01 *p<0.001**
Each value is compared with 100% control.

Mechanical stress experiment on commercial size clams

Biochemical and behavioural parameters

The AEC, results shown in Figure 8 indicated that this parameter could be considered a good indicator of stress condition; in fact the treated sample showed a significant lower value in comparison with all the controls, and it was lower even than that reached by the 24h recovered sample.

The reburrowing behaviour, the percentage of the reburrowed clams in 10 hours, normalized to their controls, are presented in Figure 9. The control sample showed the maximum percentage of reburrowing and for the treated ones the percentage is lower than both control and recovery samples although not significant. Considering the reburrowing curves, presented in figure 10, it became more evident that the value of RT50 is the shortest for the manually collected sample (less than 2 hrs), it is longer (5 hrs) for the commercially dredged sample, and likely to be more than 10 hrs for the treated clams.

Mechanical stress experiment on under-sized clams

Biochemical and behavioural parameters

The AEC showed very low values both in control and treated samples possibly due to the presence of endogenous stress which could have masked the effects of the mechanical shaking (Figure 11). In fact the values did not significantly differ in treated and control samples at both times 0 and 48 hours. Only at time 24 hours the control sample showed significantly higher value in comparison with the treated one. The percentage of reburrowed clams showed no difference between control and treated samples at any time, even if the trend was for decreasing values in the treated clams, with the most evident difference at day 3 (Figure 12).

Also for the reburrowing curves and relative RT50 values no significant differences were present between control and treated samples at any time (Figure 13).

ADENYLATE ENERGY CHARGE

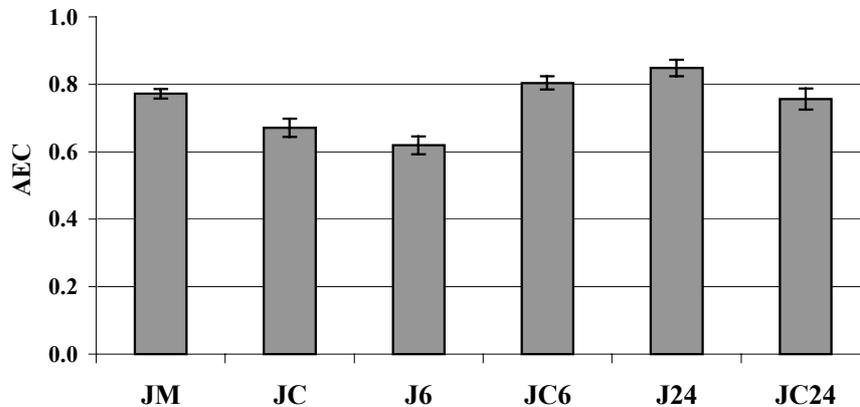


Figure 8 Laboratory study. Adenylate Energy Charge (AEC) in *C. gallina* subjected to a shaking experiment (mean ± Err.St., n = 20). Statistical comparison (ANOVA): * = p < 0.05; ** = p < 0.01; *** = p < 0.001; n.s. = not significant.

Legend: JM = control sample collected by divers; JC = control sample collected by commercial dredge and processed immediately; J6 = 6 hours treated sample collected by commercial dredge and ; JC6 = control sample collected by commercial dredge and processed after 6 hours; J24 = 6 hours treated sample collected by commercial dredge and processed after 24 hours recovery; JC24 = control sample collected by commercial dredge and processed after 24 hours.

% reburrowed clams in 10 hours

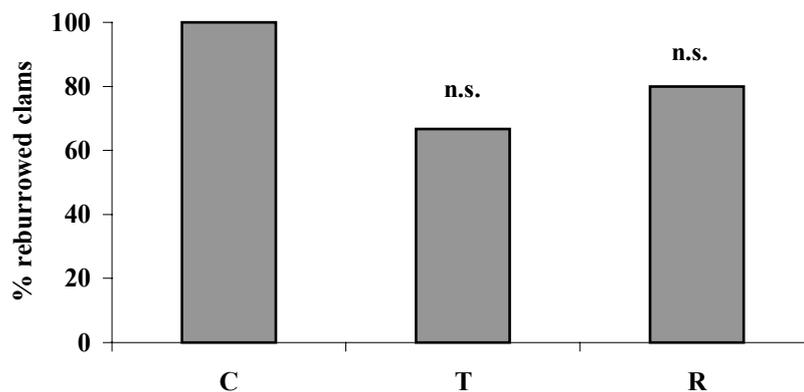


Figure 9 Laboratory study. % of reburrowed clams in 10 hours in *C. gallina* subjected to a shaking experiment. Statistical comparison (G test): n.s. = not significant.

Legend: C = control; T = 6 hours treated samples; R = 6 hours treated samples and processed after 24 hours recovery.

Shaking experiment - Reburrowing curves

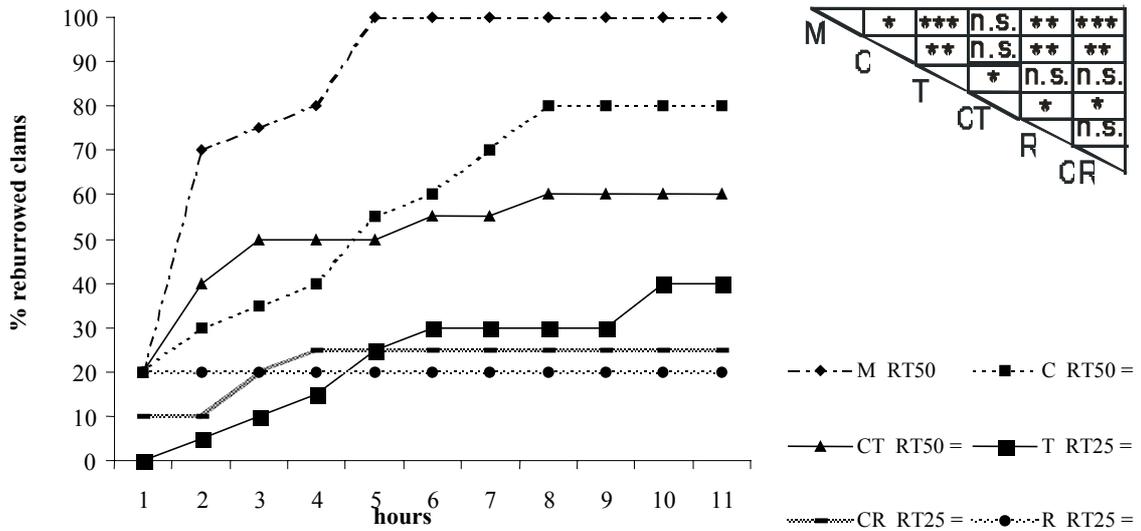


Figure 10 Laboratory study. Reburrowing curves in *C. gallina* subjected to a shaking experiment. Statistical comparison (Gehan & Wilcoxon test): * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$; n.s. = not significant.
 Legend: M = control samples collected by divers; C = control samples collected by commercial dredge; CT = control samples collected by commercial dredge processed after 6 hours; T = 6 hours treated samples collected by commercial dredge; RC = control sample R = 6 hours treated samples and processed after 24 hours recovery.

Shaking Exp. on undersized clams
Adenylate Energy Charge

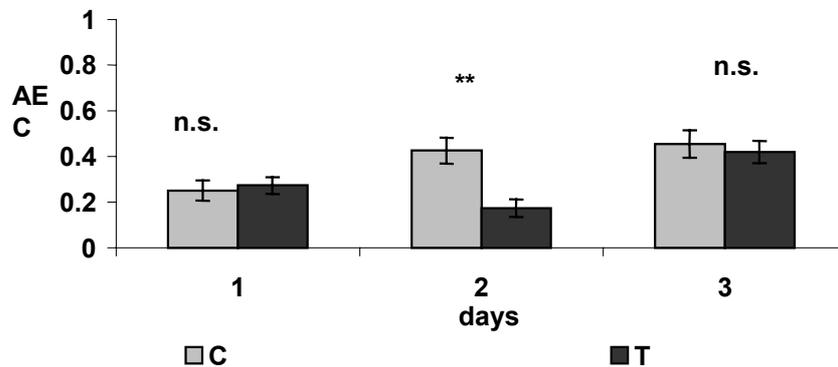


Figure 11 Laboratory study. Adenylate energy charge in *C. gallina* subjected to a shaking experiment (mean \pm st. err., $n = 10$). Statistical comparison (ANOVA): ** = $p < 0.01$; n.s. = not significant.
 Legend: C = control; T = treated.

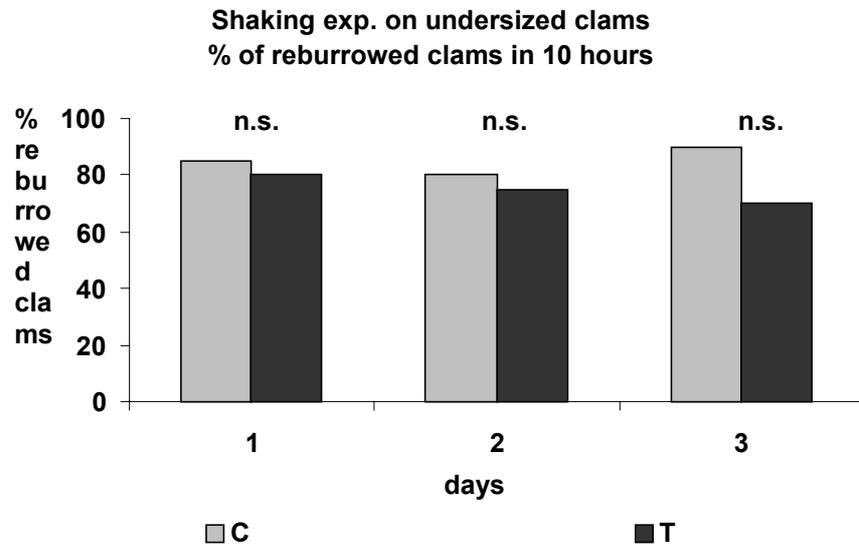


Figure 12 Laboratory study. % of reburrowed clams in 10 hours in *C. gallina* subjected to a shaking experiment. Statistical comparison (G test): n.s. = not significant.
Legend: C = control; T = treated.

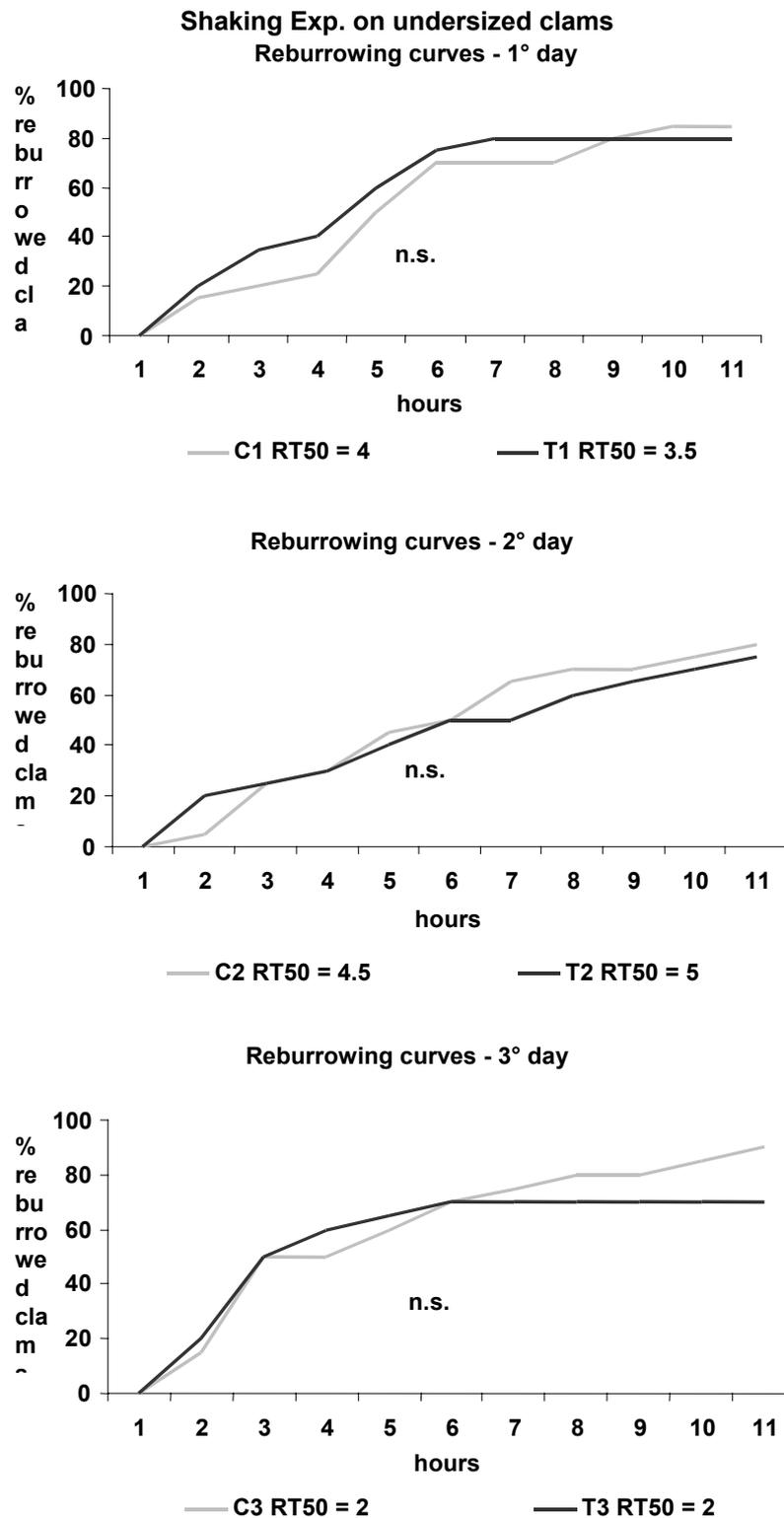


Figure 13 Laboratory study. Reburrowing curves in *C. gallina* subjected to a shaking experiment. Statistical comparison (Gehan & Wilcoxon test): n.s. = not significant. Legend: C = control; T = treated.

Field study

Immunological parameters

As for the hematocrit value, the LP samples showed significantly higher values in comparison with HP and HPS samples, the latter giving the lowest values observed, in all the seasons and for both sites (Figure 14). The results exhibited a seasonal trend with minimum values in spring or summer for the three fishing methodologies (Figure 15). Comparing the two sampling sites, LP and HPS values were significantly different in winter, spring and summer, while HP samples were significantly different only in winter and spring.

The phagocytic index showed significant differences among the fishing methodologies. The LP values were always significantly higher than the others, while no difference was found between HP and HPS values. A particular situation was observed in winter when HP samples results were lower than the others for both sites (Figure 16). With regards to the seasonal pattern, a maximum is reported in summer for the LP and HP samples, while the HPS values resulted more or less constant during the four seasons (Figure 17).

Generally in all the seasons and for both sites, the acid phosphatase activity exhibited a pattern similar to that described above for the hematocrit value, the lowest values being observed in HPS samples (Figure 18). This index showed a seasonal pattern with a minimum in spring time. The difference between the two sites was not highlighted by this parameter (Figure 19).

β -glucuronidase activity index showed an opposite trend in comparison with the other immunological parameters, revealing the higher values in HPS samples. This parameter generally did not show any difference among the various fishing methodologies both at Lido and Jesolo (Figure 20). A slight seasonal trend was observed at Jesolo for HP and HPS samples recording the higher activity in summer, while LP values remained constant during the year (Figure 21).

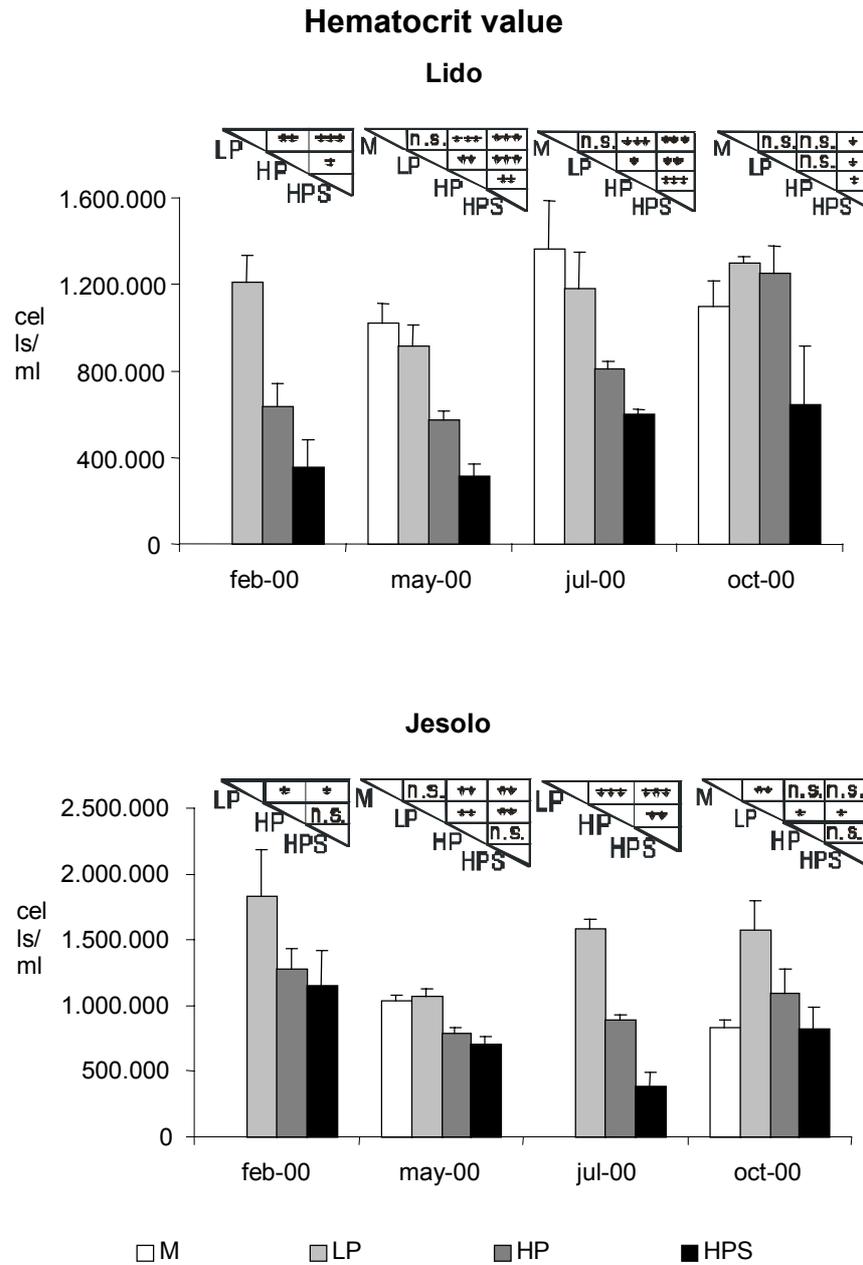


Figure 14 Field study. Haematocrit value in *C. gallina* collected at Lido and Jesolo using different fishing system (mean + s.d., n=3). Statistical comparison (Student t test): * p<0,05; ** p<0,01; *p<0,001; n.s. not significant. Legend: M=manual; LP=low pressure; HP=high pressure; HPS=high pressure and mechanical sorter.**

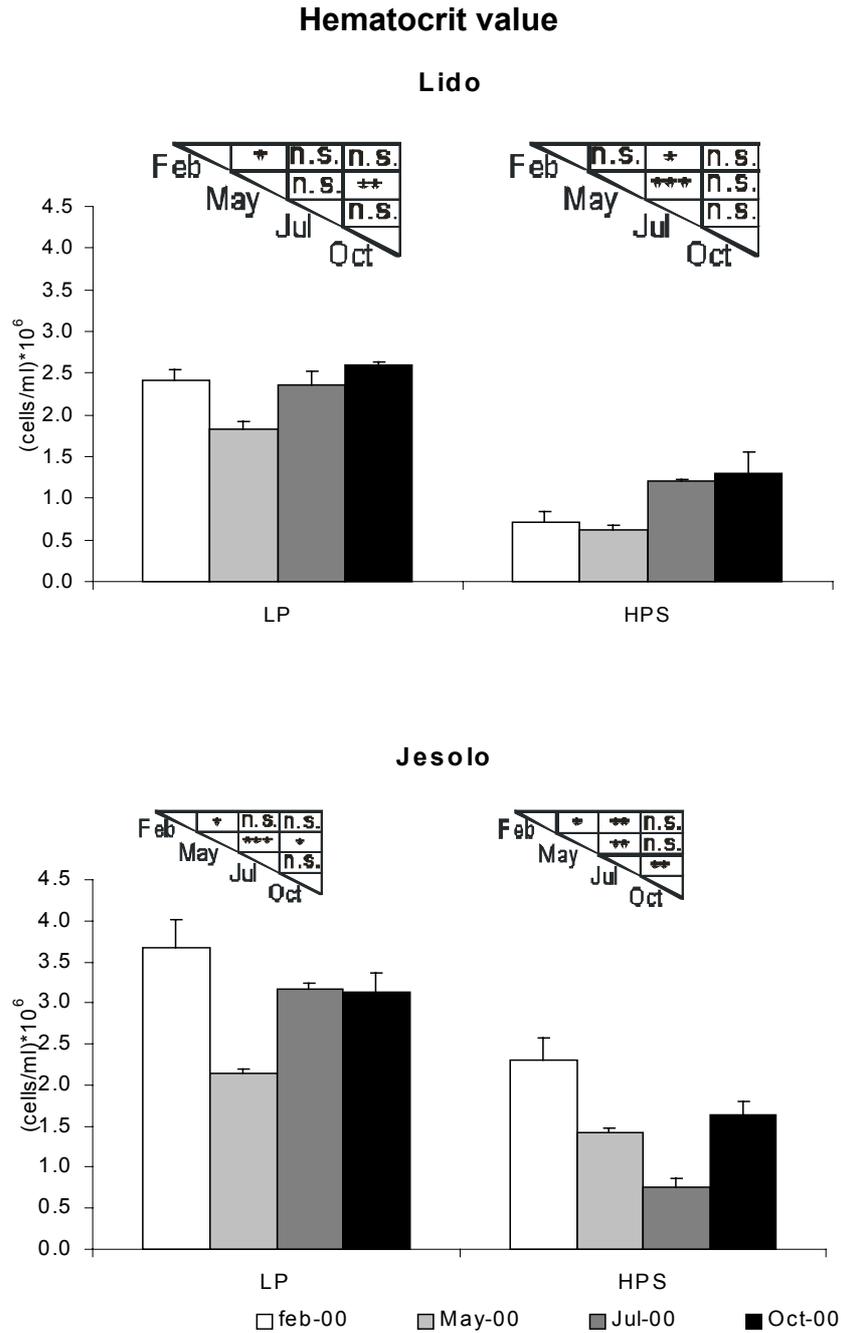


Figure 15 Field study. Haematocrit value seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido and Jesolo, (mean + s.d., n=3). Statistical comparison (Student t test): * p<0,05; ** p<0,01; *** p<0,001; n.s. not significant).

Phagocytosis

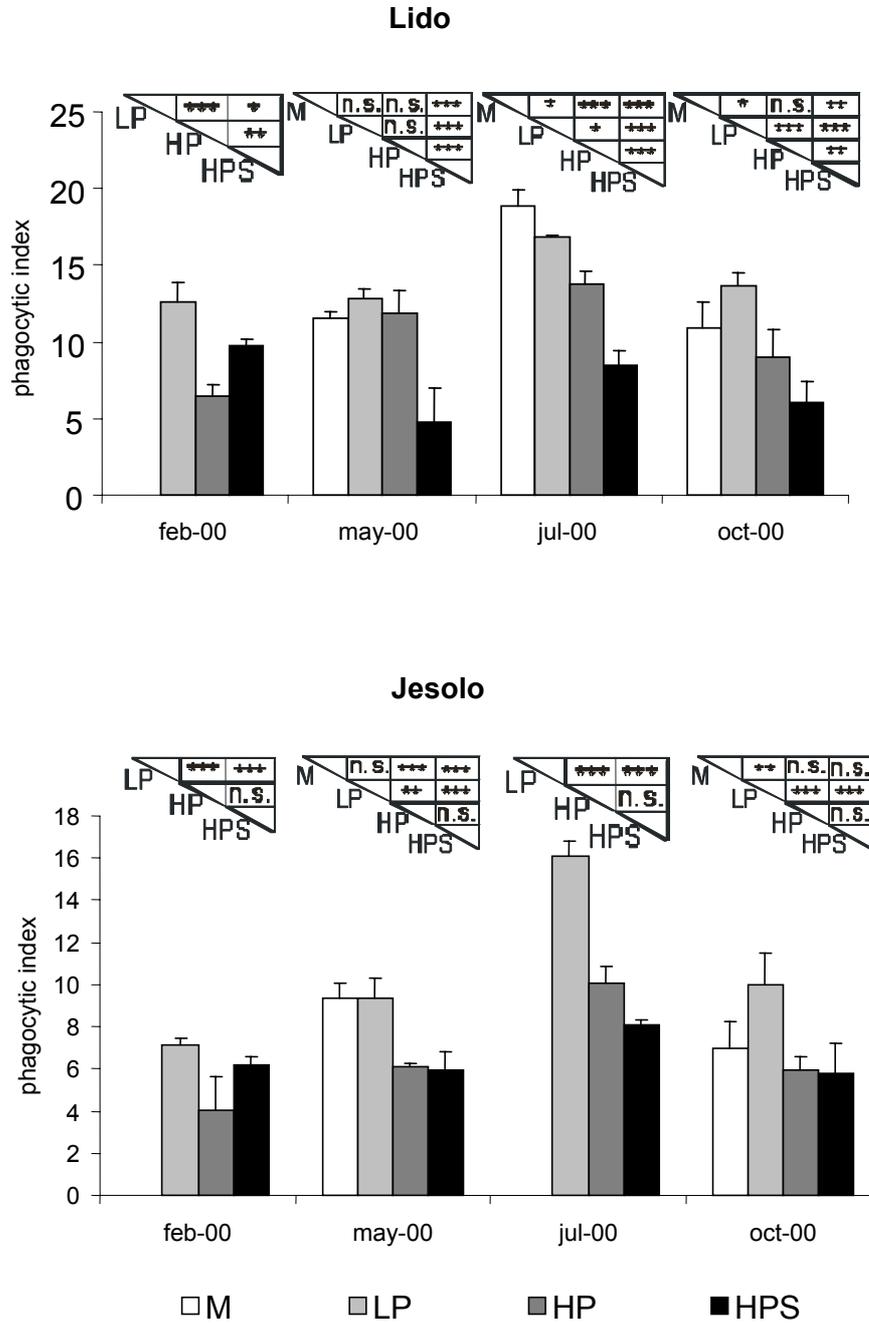


Figure 16 Field study. Phagocytosis in *C. gallina* collected at Lido and Jesolo using different fishing system (mean + s.d., n=3). Statistical comparison (G-test): * p<0,05; ** p<0,01; ***p<0,001; n.s. not significant. Legend: M=manual; LP=low pressure; HP=high pressure; HPS=high pressure and mechanical sorter.

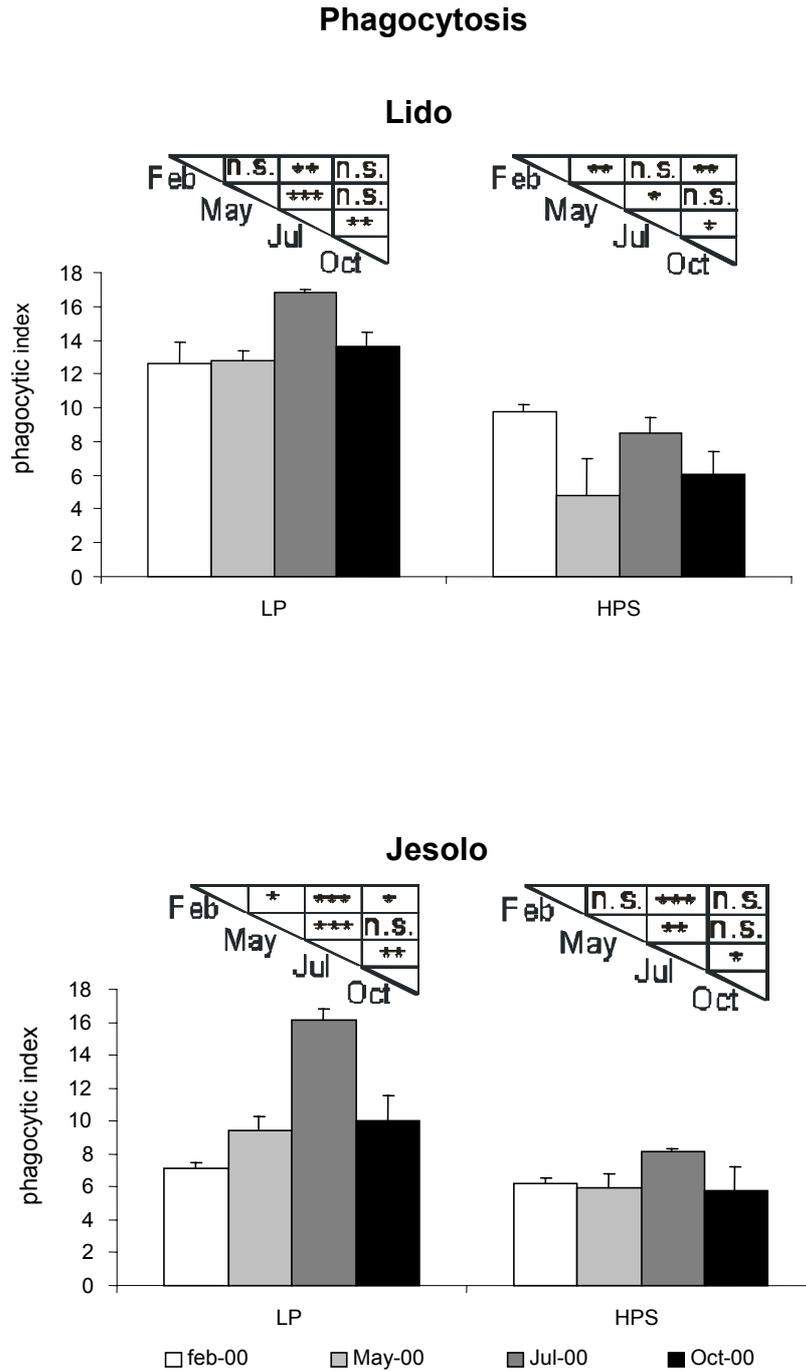


Figure 17 Field study. Phagocytic index seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido and Jesolo, (mean + s.d., n=3). Statistical comparison (G-t test): * p<0,05; ** p<0,01; *** p<0,001; n.s. not significant).

Acid phosphatase activity

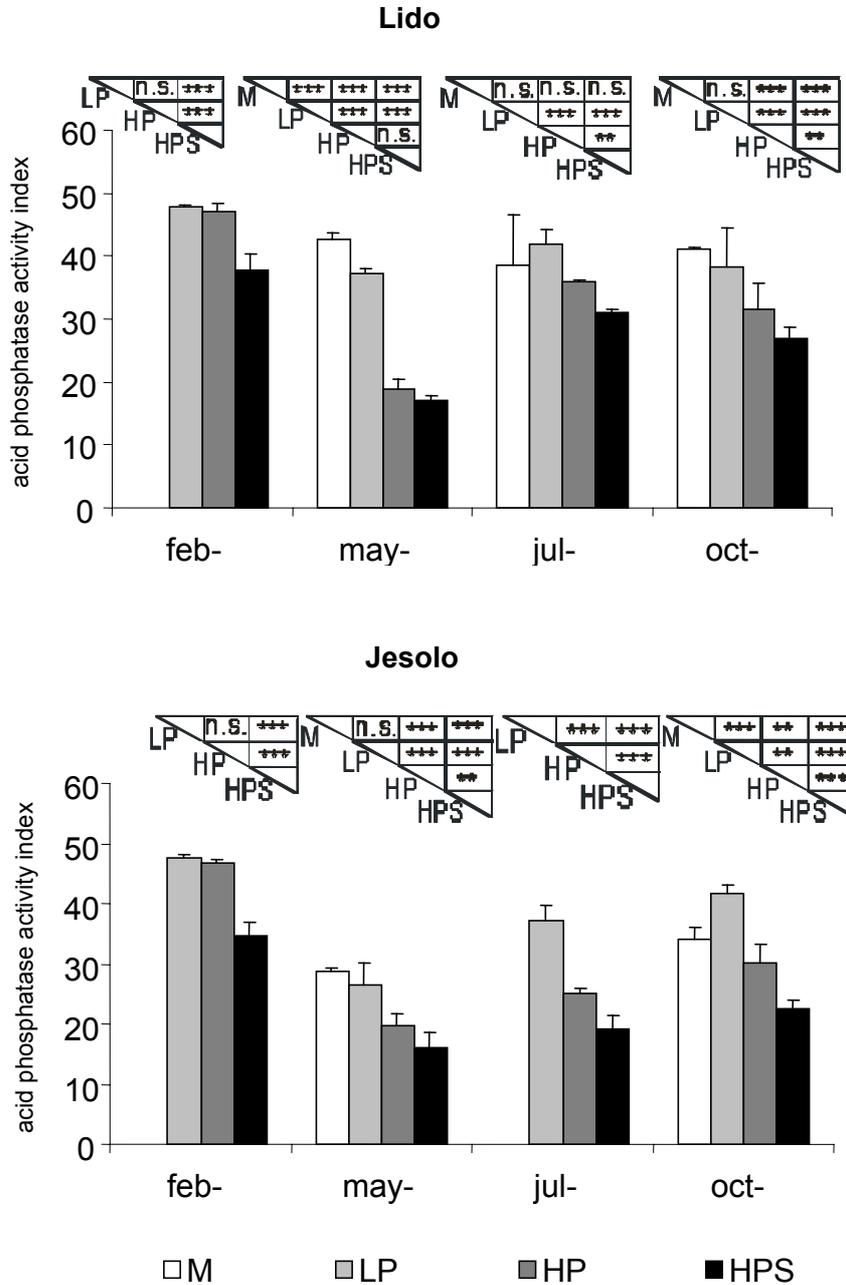


Figure 18 Field study. Acid phosphatase activity in *C. gallina* collected at Lido and Jesolo using different fishing system (mean + s.d., n=3). Statistical comparison (G-test): * p<0,05; ** p<0,01; ***p<0,001; n.s. not significant. Legend: M=manual; LP=low pressure; HP=high pressure; HPS=high pressure and mechanical sorter.

Acid phosphatase activity

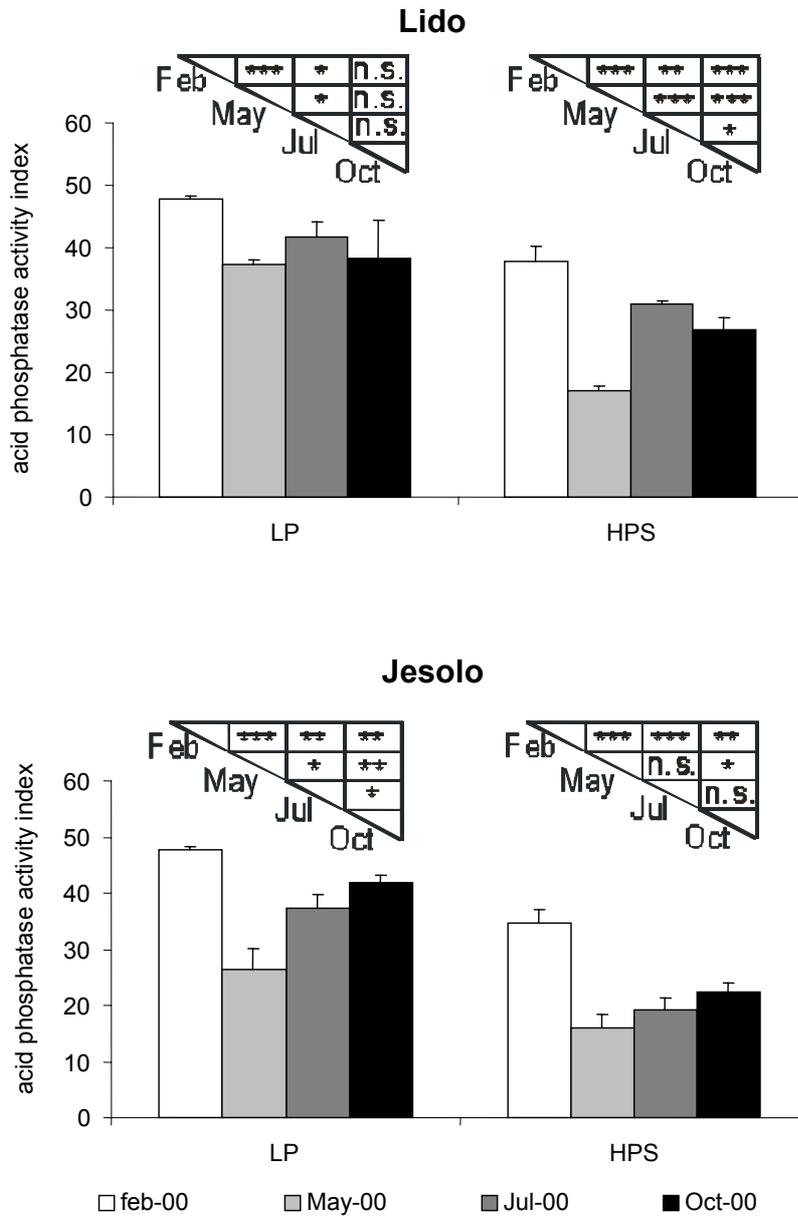


Figure 19 Field study. Acid phosphatase activity seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido and Jesolo, (mean + s.d. n=3). Statistical comparison (G-test): * p<0,05; ** p<0,01; *** p<0,001; n.s. not significant).

β-Glucuronidase activity

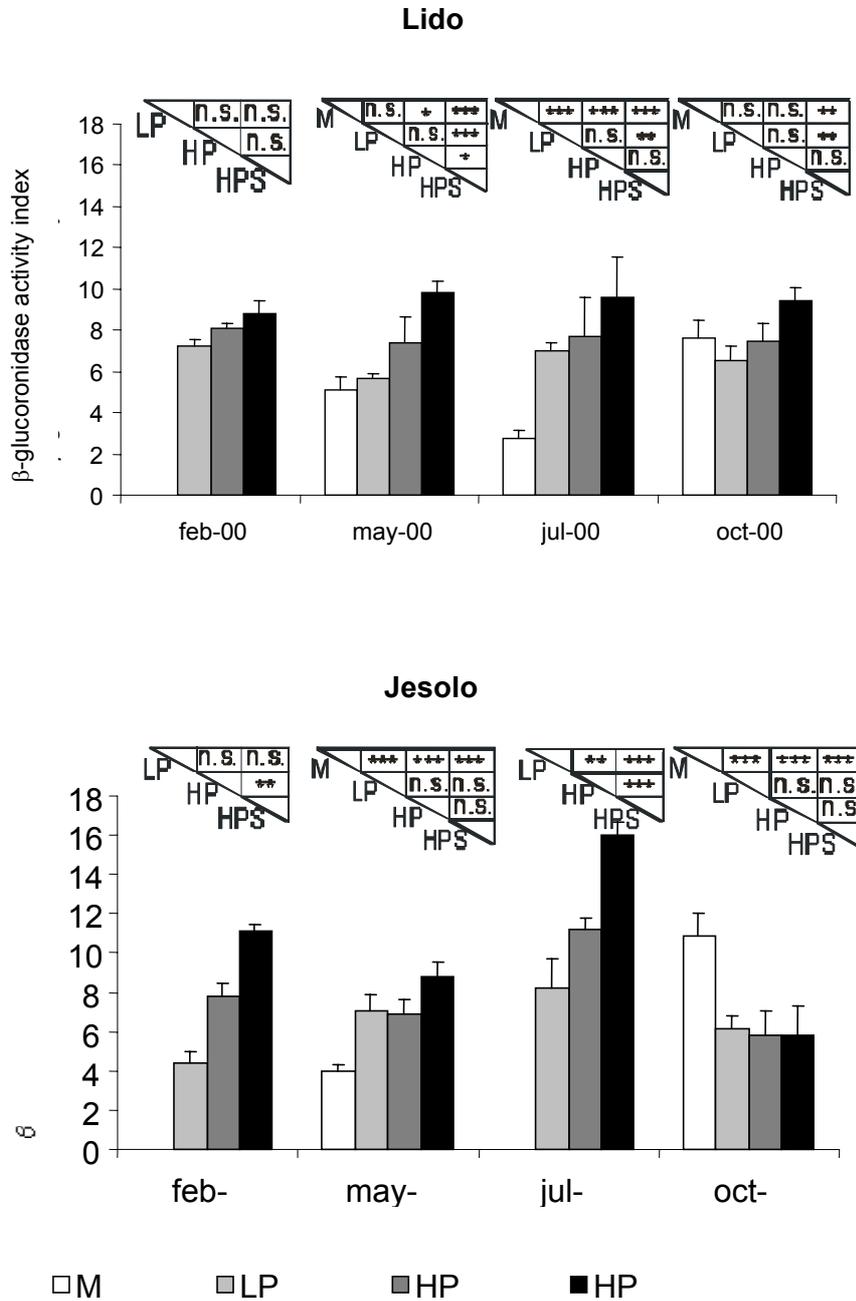


Figure 20 Field study. β-glucuronidase activity in *C. gallina* collected at Lido and Jesolo using different fishing system (mean + s.d., n=3). Statistical comparison (G-test): * p<0,05; ** p<0,01; ***p<0,001; n.s. not significant. Legend: M=manual; LP=low pressure; HP=high pressure; HPS=high pressure and mechanical sorter.

β -Glucuronidase activity

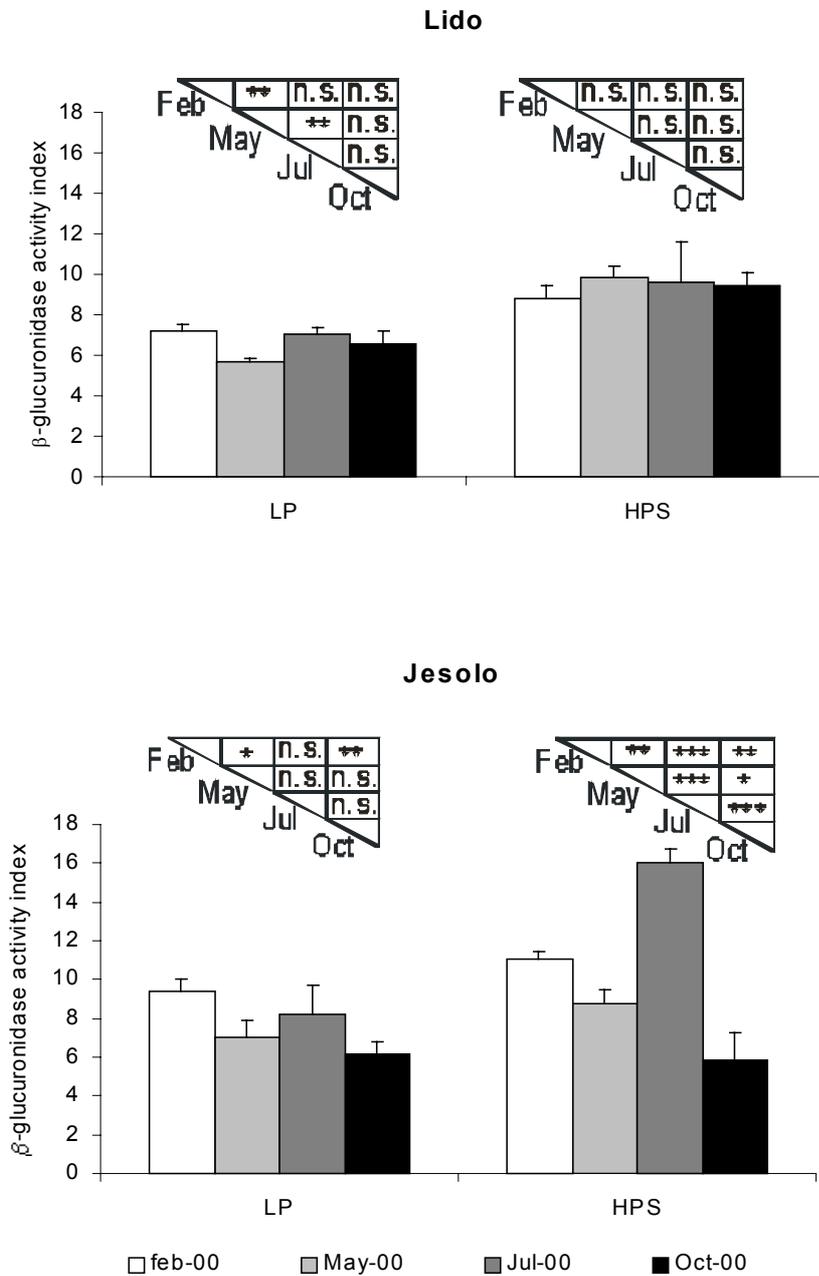


Figure 21 Field study. β -glucuronidase activity seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido and Jesolo, (mean + s.d. n=3). Statistical comparison (G-test): * p<0,05; ** p<0,01; *** p<0,001; n.s. not significant).

Field study

Biochemical and behavioural parameters

With regards to the pressure effects on the AEC response, similar results were obtained for both Lido and Jesolo samples (Figure 22). In February the LP samples showed significant higher values in comparison with both HP and HPS samples ($p < 0.01$; $p < 0.001$ at Lido and $p < 0.05$ at Jesolo), while in May and October the values were similar in all the samples (except for LP samples in May at Jesolo which resulted to be significantly lower ($p < 0.05$) with respect to HP sample). In July a significant lower value in HPS sample in comparison with HP one at Lido ($p < 0.05$) and a significant higher value in M sample in comparison with all the others ($p < 0.01$, $p < 0.001$) at Jesolo were detected. Looking at the specific results for LP and HPS samples (Figure 23), a seasonal trend was evident at both sites with summer values significantly lower with respect to the other seasons.

As regards the comparison between under-sized and commercial clams, the AEC did not reveal any statistical difference at Lido in July and Jesolo in October, whilst a significant higher value in commercial samples with respect to undersized one ($p < 0.01$) was evidenced at Lido in October (Figure 24).

Although the reburrowing rate, as percentage of reburrowed clams over a time period 10 hours long, in general did not show any differences among samples collected with different fishing systems both at Lido and Jesolo, in February more marked differences among the various fishing systems than in the other months were indicated. The only significant difference was shown between M and HP samples ($p < 0.05$) at both sites in October (Figure 25).

The values of the reburrowing rate, expressed as reburrowing curves and relative RT50, generally showed trends similar to those indicated by the percentage of reburrowing. Only in February at Lido LP sample showed higher reburrowing rate than HP which presented higher rate than HPS, although the differences were not significant. In May and July no significant differences were found among samples, while in October a significant lower reburrowing rate in M sample with respect to the other samples ($p < 0.05$) was present (Figure 26). At Jesolo in February the LP sample showed a significant higher reburrowing rate in comparison with HPS ($p < 0.05$). In May all the curves were quite similar. In July HPS sample showed an high RT50 value and the related curve resulted significantly different from LP and HP ones ($p < 0.05$). In October, a significant lower reburrowing rate in M sample with respect to the other samples ($p < 0.05$, $p < 0.001$) was present (Figure 27).

Regarding to the seasonal effects, looking at the specific results for LP and HPS samples higher percentage of reburrowed clams in July and/or October in both sites were indicated (Figure 28). The related reburrowing curves showed the same trend as well: higher reburrowing rate in July at Lido (Figure 29) and in July and October (respectively in LP and HPS samples) at Jesolo were presented (Figure 30).

As regards the comparison between under-sized and commercial clams, the reburrowing rate, expressed as both percentage of reburrowed clams and RT50, didn't

show any significant difference at the two sites both in July and in October (Figures 31, 32).

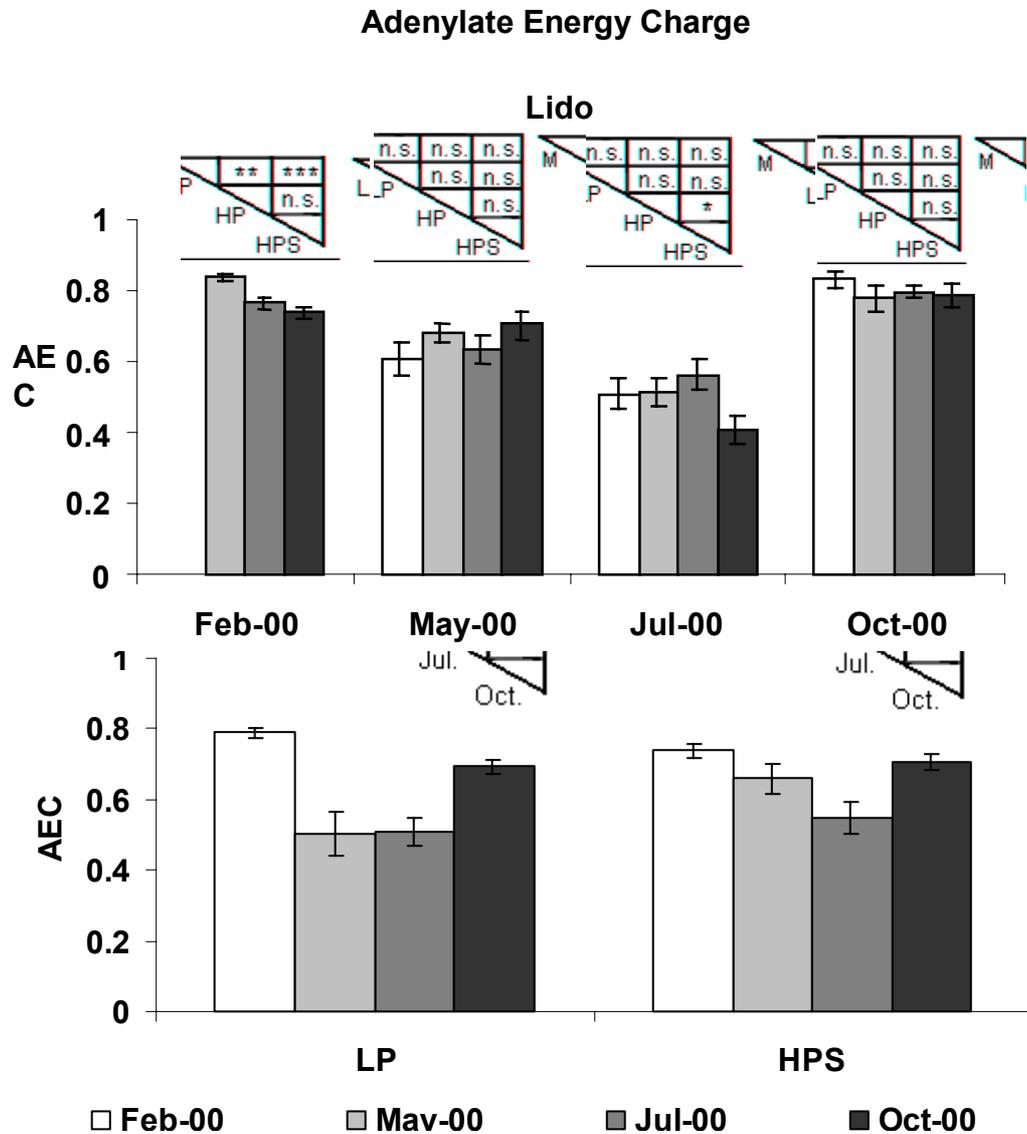


Figure 22 Field study. Adenylate energy charge (AEC) in *C. gallina* collected at Lido and Jesolo using different fishing systems (mean \pm st. err., $n = 10$). Statistical comparison (ANOVA): * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$; n.s. = not significant. Legend: M = manual; LP = low pressure; HP = high pressure; HPS = igh pressure and mechanical sorter.

Adenylate Energy Charge

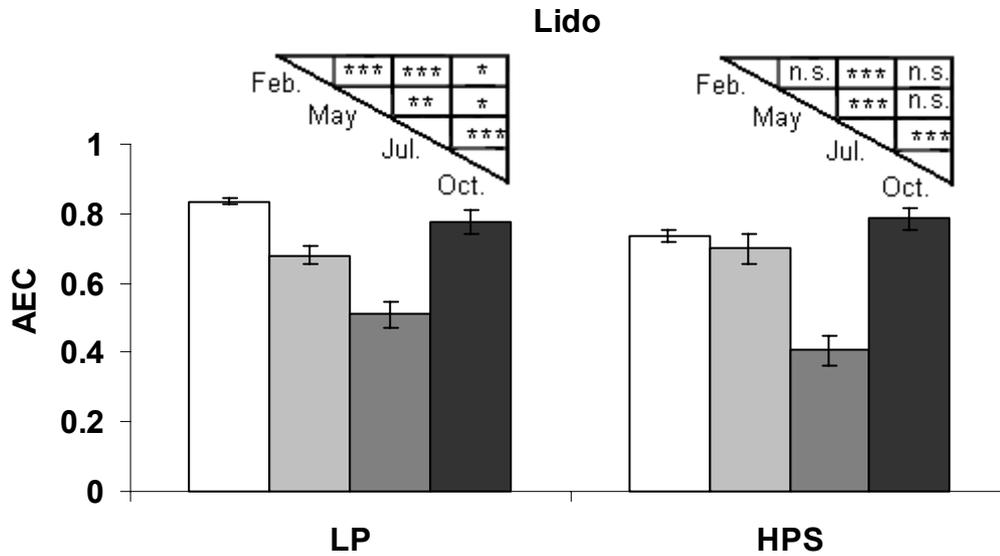


Figure 23 Field study. Adenylate energy charge seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido and Jesolo, (mean \pm st. err., n = 10). Statistical comparison (ANOVA): * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$; n.s. = not significant.

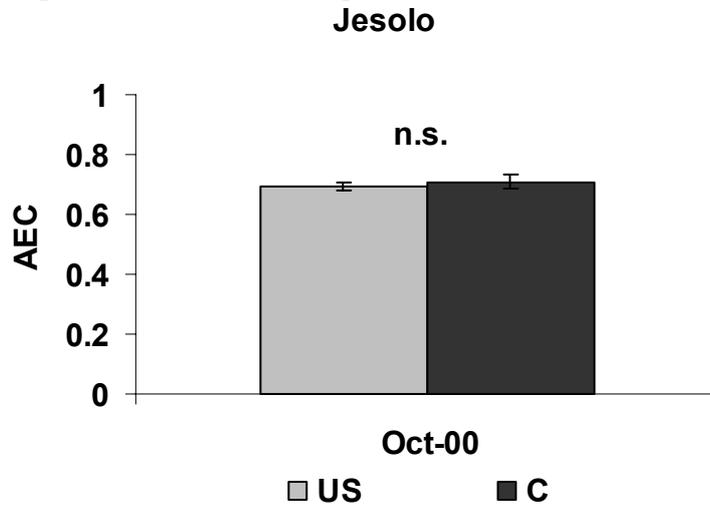


Figure 24 Field study. Adenylate energy charge comparison between undersized (US) and commercial (C) clams *C. gallina* collected at Lido and Jesolo, (mean \pm st. err., n = 10). Statistical comparison (ANOVA): ** = $p < 0.01$; n.s. = not significant.

% of reburrowed clams in 10 hours

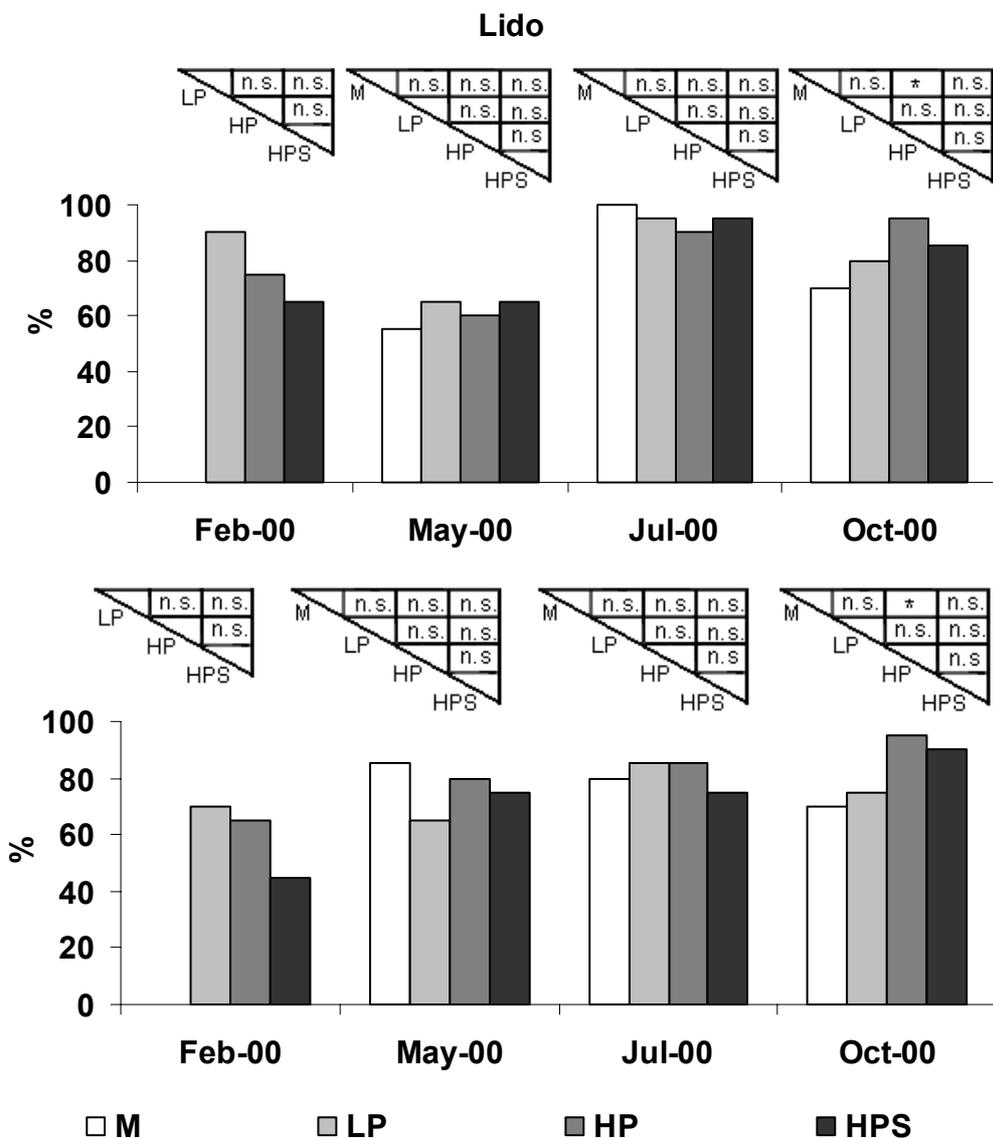


Figure 25 Field study. % of reburrowed clams over time period 10 hours long in *C. gallina* collected at Lido and Jesolo using different fishing systems (n = 20). Statistical comparison (G test): * = p < 0.05; n.s. = not significant.

M = manual; LP = low pressure; HP = high pressure; HPS = high pressure and mechanical sorter.

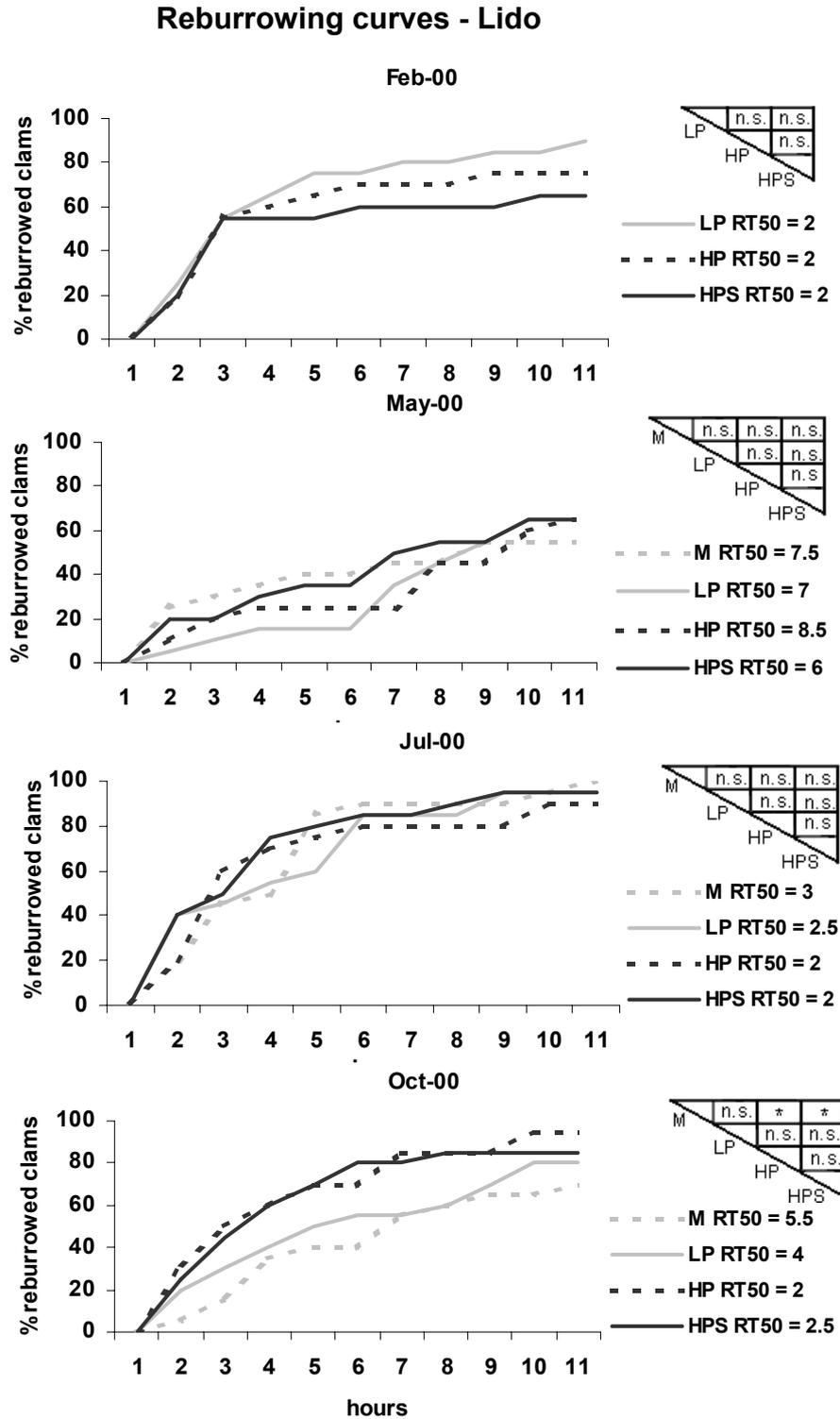


Figure 26 Field study. Reburrowing curves in *C. gallina* collected at Lido using different fishing systems (n = 20). Statistical comparison (Gehan & Wilcoxon test): * = p < 0.05; n.s. = not significant. M = manual; LP = low pressure; HP = high pressure; HPS = high pressure and mechanical sorter.

Reburrowing curves - Jesolo

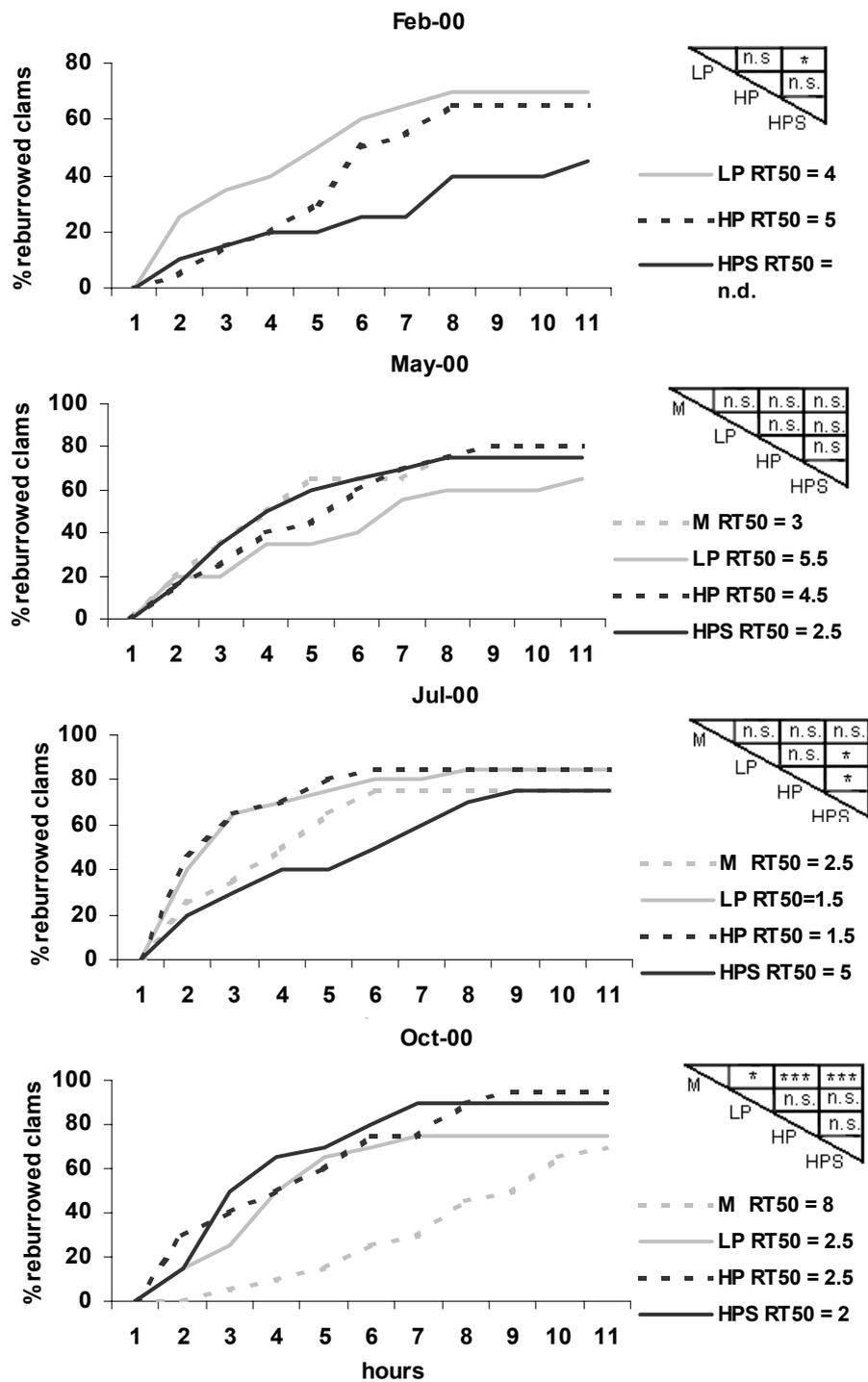


Figure 27 Field study. Reburrowing curves in *C. gallina* collected at Jesolo using different fishing systems (n = 20). Statistical comparison (Gehan & Wilcoxon test): * = p < 0.05; *** = p < 0.001; n.s. = not significant.
M = manual; LP = low pressure; HP = high pressure; HPS = high pressure and mechanical sorter.

% reburrowed clams in 10 hours

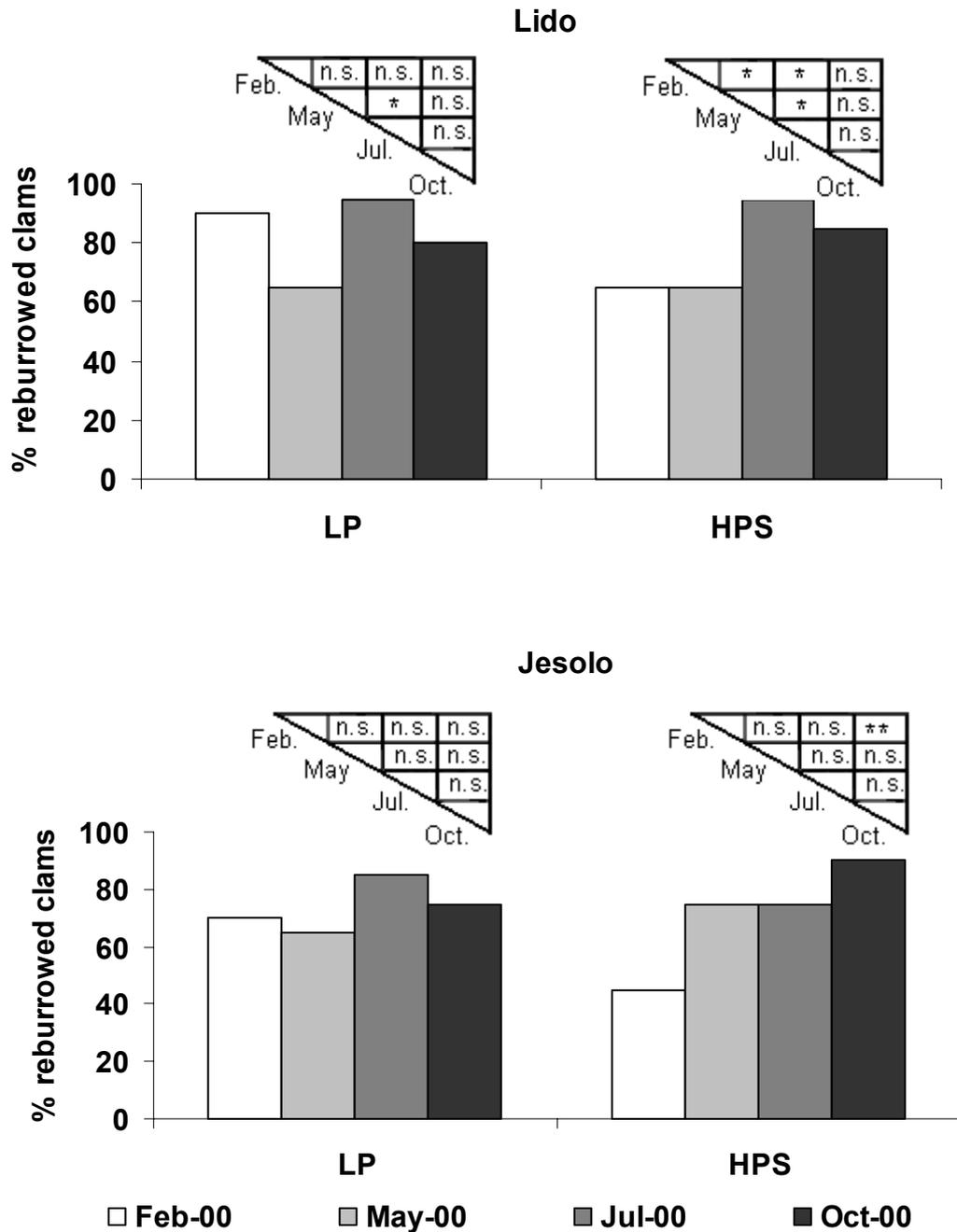


Figure 28 Field study. % of reburrowed clams seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido and Jesolo, (n = 20). Statistical comparison (G test): * = p < 0.05; ** = p < 0.01; n.s. = not significant.

Reburrowing curves - Lido

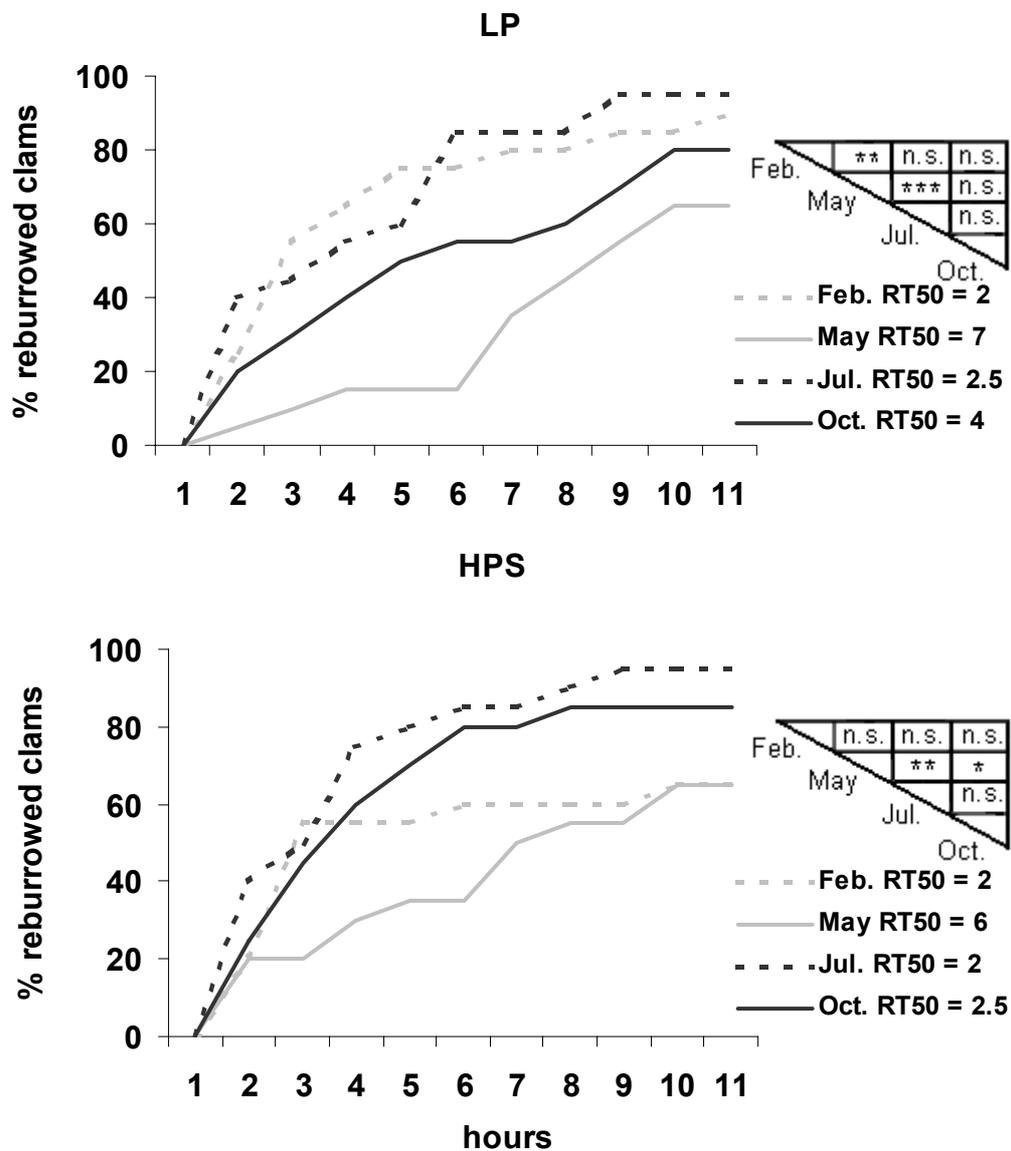


Figure 29 Field study. Reburrowing curves seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Lido, (n = 20). Statistical comparison (Gehan & Wilcoxon test): * = p < 0.05; ** = p < 0.01; *** = p < 0.001; n.s. = not significant.

Reburrowing curves - Jesolo

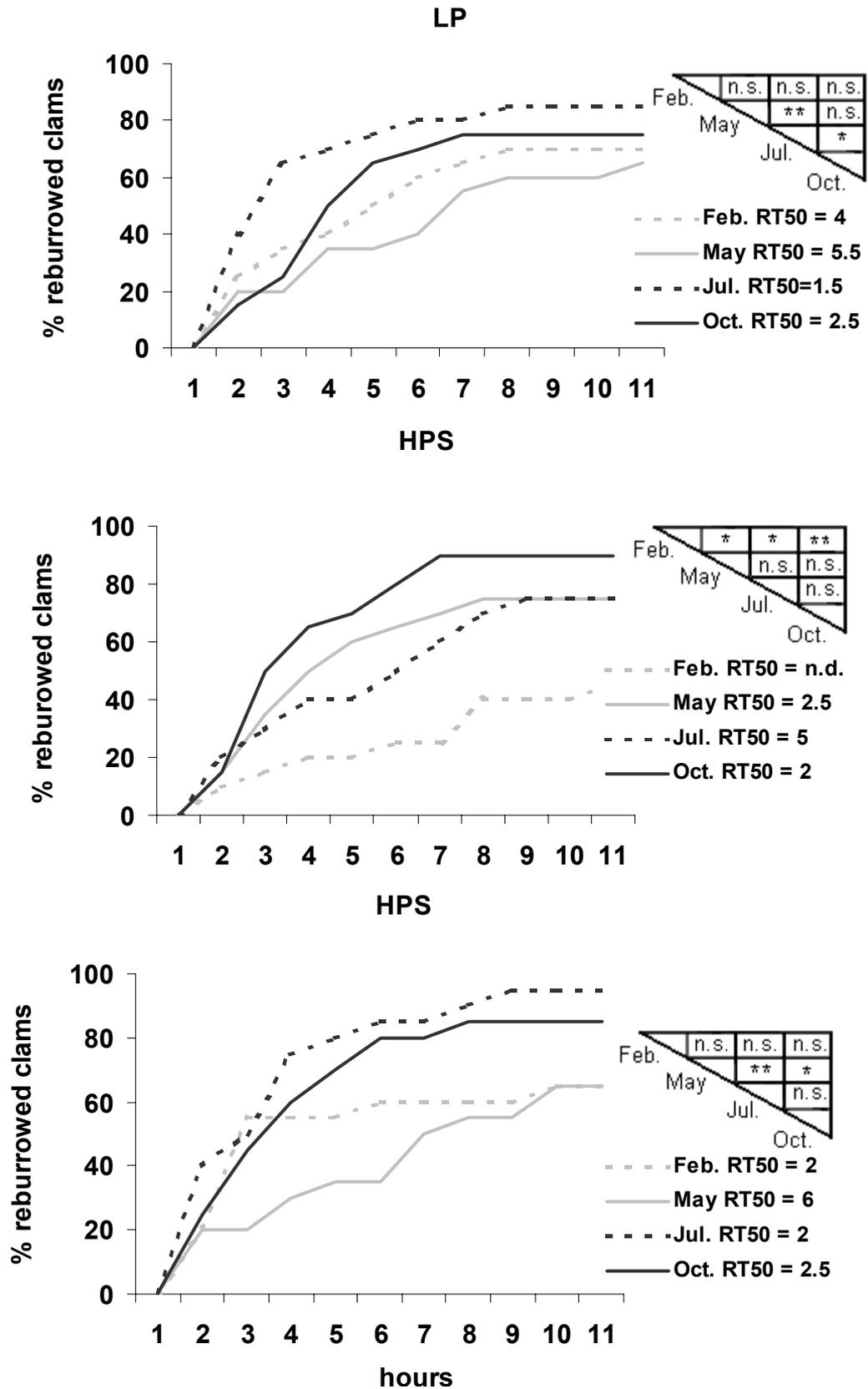


Figure 30 Field study. Reburrowing curves seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure and mechanical sorter (HPS) at Jesolo, (n = 20). Statistical comparison (Gehan & Wilcoxon test): * = p < 0.05; *** = p < 0.001; n.s. = not significant.

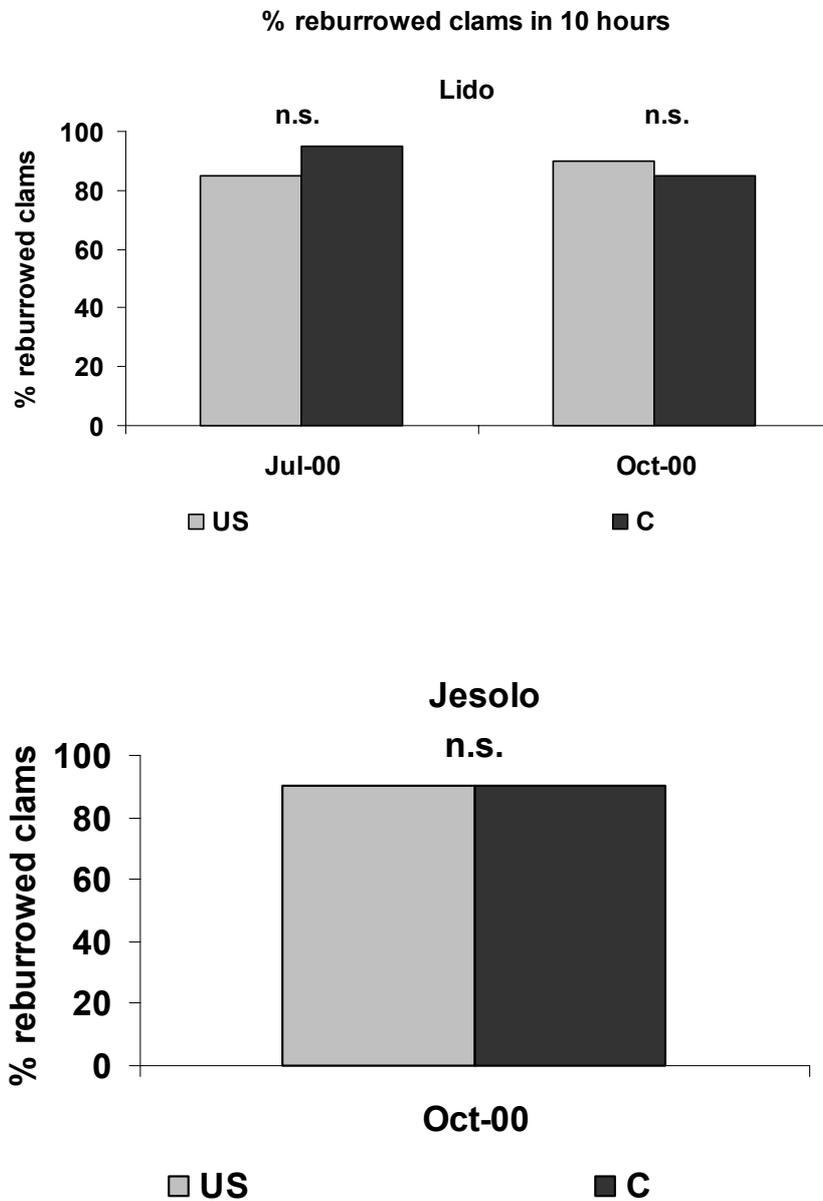


Figure 31 Field study. % of reburrowed clams comparison between undersized (US) and commercial (C) clams *C. gallina* collected at Lido and Jesolo, (n = 20). Statistical comparison (G test): n.s. = not significant.

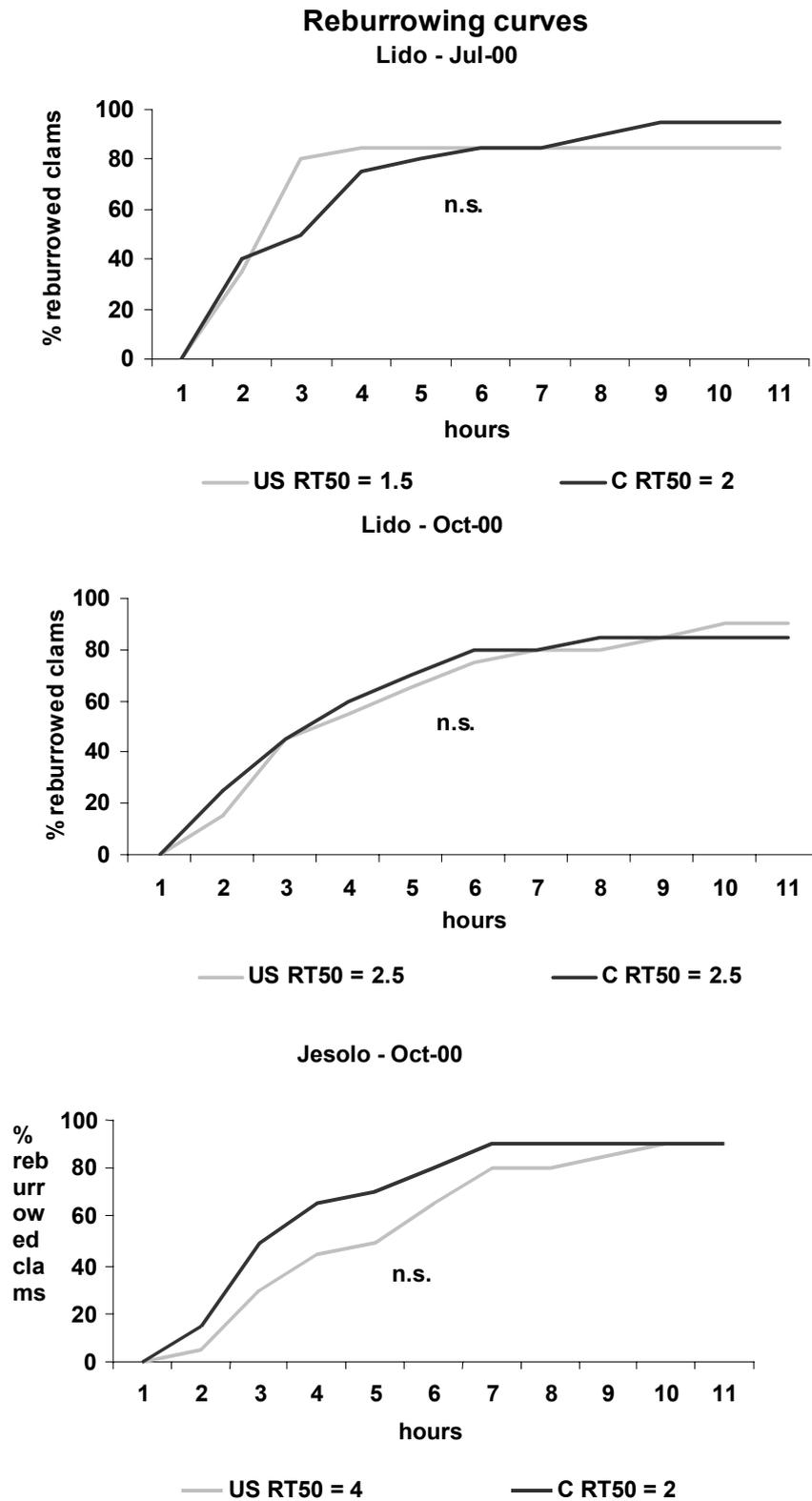


Figure 32 Field study. Reburrowing curves comparison between undersized (US) and commercial (C) clams *C. gallina* collected al Lido and Jesolo, (n = 20). Statistical comparison (Gehan & Wilcoxon test): n.s. = not significant.

Multivariate analysis

The scatterplot in Figure 33 showed a marked separation along the axis of the Factor 2 between the two studied sites (Lido and Jesolo) as well as among the dredging systems applied, according to the Factor 1. For both factors, high values of percentage of the total variance explained were observed: 49% for Factor 1 and 25% for Factor 2 (total explained variance 74%).

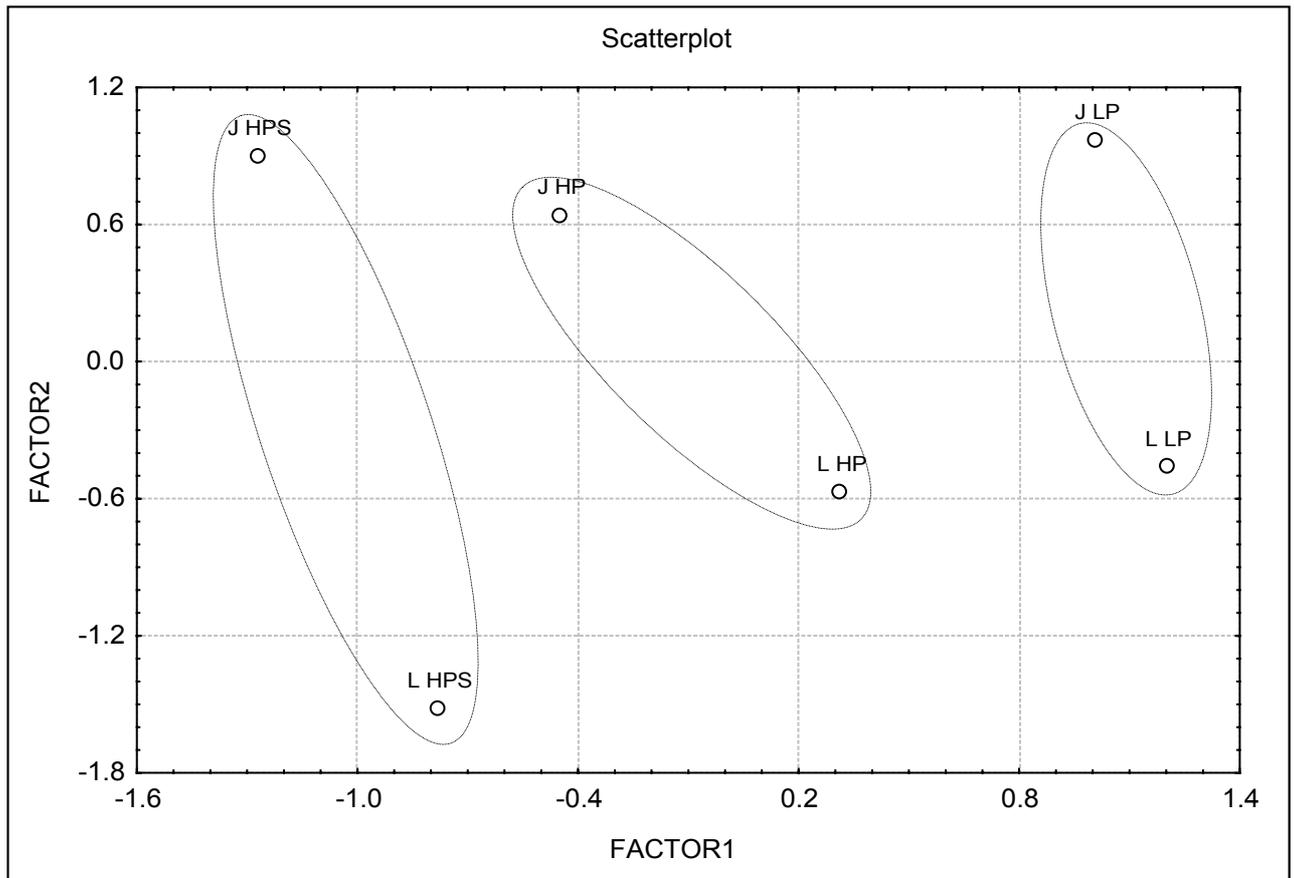


Figure 33 Samples arrangement obtained by PCA analysis considering the whole set of data obtained for the acute stress indices.

Discussion

The biomarker approach used in this study for evaluating mechanical stress due to fishing activities on the target species has been proposed as a relatively new tool. It has been extensively applied since the '80s in monitoring programs of the effects of anthropogenic stress, namely xenobiotics introduced into marine environment (Goldberg et al. 1978 ; Bayne et al. 1985 ; Bayne et al. 1988). The present work was focused on the evaluation of the responsiveness of a number of biomarkers to the specific stress caused by commercial hydraulic dredging. Biomarkers could be applied at different levels of biological organization, from cells to population/community, and data obtained at cellular/organism level are the most suitable to point out early signs of acute stress condition.

In this study, biochemical (Adenylic Energy Charge, AEC) and immunological (hematocrit value, phagocytic index, enzyme activity indices) parameters have been determined at cellular level, whereas behavioural assay based on the reburrowing ability and shell damage index at organism.

Indicators

A morphofunctional and cytoenzymatic characterization of *Chamelea gallina* hemocytes was carried out as a prerequisite for further study in the application of immunological parameters as stress indices. In accordance with Cheng (1981) and numerous other authors (see the review of Hine 1999), the hemocytes of *C. gallina* can be grouped into two main categories, namely granulocytes (cells containing many granules) and hyalinocytes (with few or no cytoplasmic granules). Earlier studies on Venerid hemocytes indicated the presence of only one cell type, the granulocyte (Cuenot 1891, Zack 1955, both cited in (Cheng 1981). Nevertheless the existence of two hemocyte types was proposed by Huffman and Tripp (1982) in *Mya arenaria* and by Auffret 1985) in *Ruditapes philippinarum*. In our species hyalinocytes were more abundant than granulocytes. The cytoplasm of some hemocytes exhibited yellow-orange inclusions larger than granules, most likely lipofuscin-containing phagosomes (Carballal et al. 1997b).

Although most of the hemocytes contained hydrolytic enzymes, only 6% ingested yeast *in vitro*. This low activity was also reported from mussels *Perna perna* (Barracco et al. 1999) and *Mytilus galloprovincialis* (Cajaraville et al. 1995) and might be attributed to the lack of humoral opsonins in experimental conditions, normally present in the hemolymph (Hardy et al. 1977). A higher percentage of phagocytic cells occurs after pre-incubation of latex beads with CFH; this suggests the presence of opsonin(s) in the hemolymph, in agreement with what reported by Renwranz and Stahmer (1983) for *Mytilus edulis*.

The presence of binucleated circulating cells (both granulocytes and hyalinocytes) suggests that differentiated hemocytes retain the ability to divide and supports the hypothesis that granulocytes and hyalinocytes represent two different cell lines, in agreement with the results reported by Moore and Lowe (1977) in *M. edulis* and by Carballal et al. (1997b) in *M. galloprovincialis*, although hyalinocytes are considered by some authors (Ottaviani et al. 1998) as a young differentiation stage of hemocytes, leading to granulocytes. Moreover, other studies have confirmed various stages of

granulocyte differentiation with parallel growth of their specific granules (Cajaraville and Pal 1995; Carballal et al. 1997b).

In winter, the haematocrit value was $1.2 \cdot 10^6$; it significantly increased in spring ($1.6 \cdot 10^6$, $p < 0.05$) and summer ($2.4 \cdot 10^6$, $p < 0.001$), whereas it decreased to winter levels in autumn. This variation may reflect dependence on the biological characteristics of the individuals, their physiological state and the environmental conditions (Auffret and Oubella 1995). The number of hemocytes were influenced by seasonal factors, as reported by Santarè et al. (1994) in *M. galloprovincialis*. Feng (1965) reported that the number of circulating hemocytes in *Crassostrea virginica* is influenced by the amount of food and the temperature. In this study, the highest value was found in summer, when the highest temperature and abundant food availability occurred in the Adriatic Sea.

Also evaluated were the presence of various enzyme activities in *C. gallina* hemocytes by using cytoenzymatic techniques. Both hyalinocytes and granulocytes show positive evidence for certain hydrolytic enzymes inside small granules, which probably correspond to lysosomes (Hayashi 1971). As in other bivalves, not all granules in hemocytes are reactive for lysosomal enzymes (Huffman and Tripp 1982; Auffret 1989, Pipe 1990).

β -glucuronidase activity was infrequent in *C. gallina* hemocytes, in agreement with results reported for other bivalves (Bayne et al. 1979, Pipe 1990, Carballal et al. 1997a, Lopez et al. 1997, Ottaviani et al. 1998). β -glucuronidase is considered a non-specific humoral defence molecule, owing to the fact that it can hydrolyse acid mucopolysaccharides, which are constituents of bacterial cell walls (Cheng 1976). Acid phosphatase is a typical marker for lysosomes (De Duve 1963), even though extralysosomal localisations of these enzymes have been described in several kinds of cells (Borgers and Verheyen 1985). Particularly in bivalves, a high activity of acid phosphatase was found in *R. decussatus* (Lopez et al. 1997), *M. galloprovincialis* (Ottaviani et al. 1998), *Mercenaria mercenaria* (Yoshino and Cheng 1976), *M. arenaria* (Huffman and Tripp 1982), *C. virginica* (Downs 1988), *Ostrea edulis* and *Crassostrea gigas* (Auffret 1989), *P. perna* (Barracco et al. 1999), *M. galloprovincialis* (Bayne et al. 1979; Cajaraville et al. 1995; Cajaraville and Pal 1995, Carballal et al. 1997a), *M. edulis* (Moore and Lowe 1977). Its presence was not reported in *T. philippinarum* (Cima et al. 2000). β -glucuronidase activity, but not acid phosphatase activity, increases after cell activation upon the recognition of foreign particles; this indicates a certain degree of regulation of the enzyme activity, in agreement with data obtained under anoxic conditions (Pampanin et al. 2001).

Clam exposed to air initially slightly open their shells at some point having limited oxygen exchange (hypoxia). Gaping decreases and clams close their shells and become anoxic until death. To study hemocyte alterations during air exposure in *C. gallina*, the course of some indices were followed that are known to have negative repercussions in immune defence mechanisms (Hawkins et al. 1993). The hematocrit gives an indication of cell proliferation and the presence of inflammatory foci able to attract hemocytes (Pipe et al. 1995). Decreases observed after air exposure may be the consequence of cellular lysis, reduced proliferation of hemocytes, or increased movement of cells from circulation into tissues (Marigomez et al. 1990; Suresh and Mohandas 1990; Pipe 1990).

The adhesion index estimates the ability of cells to adhere to a substratum and is related to the integrity of the cell plasma membrane and cytoskeletal organisation (Alvarez and Friedl 1992). Adhesion requires the transduction of signals originating from the interaction of adhesion molecules with ligands, and leads to changes in energy dependent cell morphology (Hynes 1999). Most of the haemocytes can adhere to uncoated coverslips, probably through the involvement of integrin-like molecules and ligands containing the RGD sequence (RGD: Arg-Gly-Asp). RGD-dependent adhesion was observed in the gastropod *Biomphalaria glabrata* (Davids and Yoshino 1998), the synthesis and secretion of the RGD-containing fibronectin was reported in *M. galloprovincialis* haemocytes (Panara et al. 1996). Reduced ability to adhere in anoxic conditions reflects the low energy charge of cells.

The increase in the number of circulating apoptotic cells following an anoxic period fits previously reported induction of apoptosis by various stressors in cells of marine organisms (Grosvik and Goksoyr 1996; Cima and Ballarin 1999).

A decrease in phagocytic activity during stress is in agreement with many studies focusing on the effects of pollutants (Beckmann et al. 1992; Coles et al. 1995; Ordas et al. 1999). Phagocytosis requires phagocytes to adhere to extra-cellular matrix and foreign particles. This is achieved through the formation of podosomes and pseudopodia and is mediated by cytoskeletal changes (Wei et al. 1997) which, in turn, involves consumption of ATP which is not available during hypoxic/anoxic stress. This is also confirmed by the progressive reduction of spreading cells with increasing periods of air exposure. The decrease in the phagocytic index in stress conditions can also be influenced by a reduction in circulating phagocytic cells consequent upon the lower hematocrit value.

Acid phosphatase and β -glucuronidase are detectable in *C. gallina* hemocytes: both enzymes are located in lysosomes and the different amount of positive cells probably reflects the different nature of the two enzymes: the former constitutive and the latter inducible. In support of this hypothesis, in the related species *T. philippinarum*, β -glucuronidase activity is detectable in a small fraction of hemocytes only after stimulation by preincubation with yeast cells (Cima et al. 2000). The observed decrease in acid phosphatase activity after air exposure is probably the result of several events, e.g., metabolic arrest due to anaerobiosis and alteration of the lysosomal integrity caused by hypoxic/anoxic stress (Lowe et al. 1995b; Lowe et al. 1995a). Conversely, increase in the activity of β -glucuronidase after air exposure is probably due to induction of this enzyme as a response towards proliferation of anaerobic bacteria (De Zwaan and Eertman 1996).

As regards the ability to recover after air exposure, only three indices were followed, hematocrit and the frequency of cells showing positivity for the two enzyme activities. It is interesting to note that in clams exposed to air for one day, the hematocrit did not exceed control values after four days of recovery. In clams exposed for more than one day, the concentration of hemocytes was no longer restored. A similar pattern was observed for the two enzymes, whose activity index reached control values during the recovery period only in clams exposed to air for 1 day. Indeed, they remained considerably lower than controls in cells from clams exposed to air for longer periods. This is probably related to the remarkable and long-lasting

reduction in the number of circulating hemocytes. Nevertheless, *C. gallina* hemocytes do not show a good ability to recover, probably due to incapacity to adapt their responses.

It is known that sublethal stress events, such as hypoxia, reduce the functionality of hemocytes and can compromise the ability of animal to survive in natural ecosystem. The development of indices assessing biological impairment can be useful to managers of ecosystems to prevent depletion of natural resources. The present study can be considered a contribution to the comprehension of natural mortality repeatedly occurring along the western Adriatic coasts, moreover it clearly shows that changes in hemocyte functionality can be used as ecophysiological indices.

Laboratory response to stress

The shaking experiments were carried out with the aim to verify the sensitivity of several biological responses to point out a stress condition in clam populations subjected to dredge fishing.

As immune defences completely rely on haemocyte activity in invertebrates and stress is usually reported to impair immune system, we tried to define some parameters useful to put in evidence, at the cellular level, the effects of stress on cell physiology. Haemocytes were used as reference cells. The value of haematocrit decreased in stressed animals, and this is in agreement with previous observations that had related this parameter both to cell proliferative activity and to the presence of inflammatory foci able to attract haemocytes.

Phagocytic index is dependent on the physiological status of haemocytes, which must be able to recognise and bind foreign particles, and finally ingest them. It resulted to be sensitive enough to put in evidence mechanical stress on haemocytes. Adhesion index, although a sensitive parameter, resulted useless as it does not provide additional information with respect to the previous one.

Only two hydrolytic enzymes (abundant in phagocytes), among those tested, were used as stress indicators, namely acid phosphatase and β -glucuronidase. The latter seems to be an inducible enzyme, whose activity increases in haemocytes from stressed animals.

All these indices showed significant variations in stressed clams compared with controls. Refrigeration, needed to storage clams overnight before processing them, increased the difference with respect to controls. However, a full recovery was clear for all the indices, just after 12 hours in seawater.

Field Results

Generally field results confirmed the laboratory data. The hematocrit value was able to distinguish samples collected with different dredging systems revealing the effects of water pressure and mechanical sorting. This behaviour can be due to increased movement of cells from circulation into tissues under stress condition, reduced proliferation of hemocytes or can be the consequence of cellular lysis. This parameter showed a seasonal trend, with spring values significantly lower and winter-autumn values higher for each dredging system. During the winter period, the increase of

hemocytes can be the result of the higher energy invested in tissue growth, including hematopoietic activity, in the no-gametogenesis period. After spawning period (hot months), the hemocyte number decrease, since there was a migration of cells from hemolymph to gonads to help the elimination of remaining cellular fragments.

The phagocytic index reflected the effects of different fishing systems, even though the effect of mechanical sorting was not highlighted. The higher values reported in winter for HPS samples in comparison with HP ones may be due to the stimulation of hemocytes. A seasonal trend was observed with generally higher phagocytic index in summer and a lower activity in winter. Phagocytic activity showed a significant positive correlation with temperature, as demonstrated for other invertebrates.

Acid phosphatase and β -glucuronidase are a typical lysosomal enzymes in metazoa, and their levels of activity appear to be altered by stress conditions. The different amount of cells positive for acid phosphatase probably reflected the constitutive nature of the enzyme. Also the acid phosphatase activity indicated the effects of water pressure and mechanical sorting; the variation of this activity can be due to an alteration of the lysosomal integrity caused by stress conditions. This index showed a seasonal trend, with spring values significantly lower and winter-autumn values higher for each dredging system.

Unlike acid phosphatase, β -glucuronidase is probably an inducible enzyme, which can show a different behaviour under stress conditions. This index was not able to reveal any statistical difference among samples collected with different dredging systems in all sampling periods. Contrary to what we observed in the laboratory experiments, the stimulus produced during commercial fishing activity was not enough to promote a variation of the enzyme activity. This parameter did not reveal a seasonal trend too, even though a slight increase was detected in summer, the variation of environmental parameters probably did not represent a stimulus suitable for the enzyme induction.

The adenylate energy charge (AEC) measures the metabolic energy available to an organism. It is evaluated by the ratio: $(ATP + 0.5 ADP) / (ATP + ADP + AMP)$, considering the various nucleotide forms of adenosine with phosphoric acid present at a given time in the living cell (Atkinson 1968; Atkinson 1972 ; Atkinson 1977). Values, between 0 and 1, are used as an index of the energy status of living organisms. The AEC has been applied to describe the metabolic characteristics of various marine invertebrates (Ivanovici 1980; Gäde and Meinardus 1981; Moal et al. 1989a; Maguire et al. 1999a). High AEC values (0.8–0.9) are typical of organisms living in a non-limiting environment, where they can grow and reproduce at optimal rates. Values between 0.5 and 0.7 characterise organisms living in a limiting environment (i.e., low nutrients, oxygen depletion, etc.) and sometimes exhibiting impairment in growth rate and/or failure in reproductive effort. The AEC ratio, as a very quick response to environmental modifications (hours for molluscs), has been used extensively to evaluate the effects on specific organisms of various kinds of environmental and human disturbance, such as anoxia, variations of temperature and salinity, exploitation of coastal and shelf sea (Wiebe and Bancroft 1975 ; Wijsman 1976a; Ivanovici 1980; Gäde and Meinardus 1981). Referring specifically to the possible disturbing action of fishing, Skjoldal (1981) determined AEC levels in tropical zooplankton from the Great Barrier Reef and found that *Eucalanus subcrassus* had significantly lower AEC values when collected during 30 minutes

than during 3 minutes' towing, indicating capture stress. Maguire (see [Section 3.1.3.2](#) for details of the results in this Project.) also found decreasing AEC levels in *Pecten maximus* captured by dredge in comparison with hand-harvested samples, and after being treated in an experiment device simulating the fishing gear operating system.

The laboratory results, simulation of repeated fishing of undersized clams did not provide any clear indication. The very low AEC values in both controls and treated specimens may have indicated endogenous stress due to reproductive effort (Froggia 1975; Marano et al. 1980; Nojima and Russo 1989), ultimately masking the effects of mechanical stress. On the other hand, the results obtained from laboratory experiment on commercial size clams gave indications about the sensitivity of this parameter, suggesting its use as an index of pressure stress and confirming it as a rapid and precise method of evaluating the effects of environmental perturbation on organisms.

The mean AEC levels recorded in our field study at both sites was about 0.8 in the cold season samplings (February and October) and ranged from 0.4 to 0.7 in the warm period samplings (May and July). Various authors detected, in different natural population of bivalves, values in the range 0.7-0.9. Moal et al. (1989b) found 0.73 in *Crassostrea gigas* harvested off Brest (Brittany, France) and similarly, Gäde and Meinardus (1981) measured AEC levels in the range 0.88–0.9 for *Cardium edule*; Maguire et al. (1999a) found 0.9 for *Pecten maximus* and the snail *Pyrazus ebeninus*, from Jervis Bay (New South Wales) showed AEC levels of 0.8-0.9 (Ivanovici 1980). In this sense, the lowest values (0.5 - 0.4), always detected in July, could be indicative of seasonal effect more than the effect of different mechanical fishing systems. The AEC index indicated a pressure effect on samples collected with different dredging systems, specially in winter time when it seemed more successful in evaluating the effects of the mechanical stress. Only in February this parameter showed significant differences between LP an HPS samples in both sites, while the use of the sorter never influenced the biological response of the clams. The lack of significant differences among samples in the other months could be due to the presence of high endogenous and environmental stress (reproductive period, temperature, salinity, dissolved oxygen, food availability, etc.) that could have hidden the mechanical effects of dredging. The seasonal effect detected is likely due to the reproductive effort which could had depleted the energetic resources of the organisms. In general no significant differences were found between undersized and commercial clams.

The ecological relevance of burrowing behaviour in clams becomes apparent when we consider their need to avoid predation (Doerding 1982; Pearson et al. 1981). On this basis, a rapid bioassay has been developed and successfully used in toxicity sediment tests (Phelps et al. 1985; Phelps 1989). It gives an indication of the indirect mortality resulting from increased exposure to predators (Phelps 1989) and can provide a short-term response (minutes to hours) to the introduction of chemical or environmental perturbation.

The laboratory simulation of repeated fishing actions on commercial size clams showed an effect of the mechanical stress imposed, revealed by the decrease of the reburrowing rate as both percentage of reburrowed clams and RT50. Similarly, looking at the results of the laboratory simulation on undersized clams, the shaking treatment seems to have affected reburrowing behaviour, as treated samples showed a

reduction in the percentage of reburrowing clams with respect to controls, especially after three days' treatment, although differences were not statistically significant.

The reburrowing rates found in our field study on *C. gallina* (RT50 = 1.5 – 5.5 h) were higher than those calculated for *Mya arenaria*, which is considered a suitable organism for rapid burrowing bioassays in estuarine sediments, having an average burrowing speed in controls of ET50 (elapsed time for 50% population) of 0.45 h (Phelps 1989). Also *Prototarca staminea*, used in a toxicity test with Cu-enriched sediment, showed higher burrowing rates in controls (ET50 = 0.2–1 h) (Phelps et al. 1985) - like *Ensis siliqua* and *E. ensis* from North Wales, with a maximum ET 50 of 0.25 h (Henderson and Richardson 1994). These considerations are helpful in highlighting how long *C. gallina* can be left on the seabed after any kind of sediment perturbation, and show that it is a very responsive organism for evaluating mechanical stress.

In general, samples from Lido seemed to be less stressed than those from Jesolo. The different grain size distribution in sediments (sandier at Lido, muddier at Jesolo) may have influenced the speed and proper operation of the hydraulic dredge, reflecting a different impact on the seabed (more severe on muddy bottoms). The high percentage of mud in sediments at Jesolo may have contributed to poor environmental conditions for *C. gallina*, which preferably lives in sandy seabeds near the coastline where medium, fine and very fine sand is normally present at percentages exceeding 90%, with Eh greater than 300 mV (Vatova 1949 ; Vatova 1966 ; Barillari et al. 1978; Backeljia et al. 1994). These behavioural parameters indicated a pressure effect especially during winter time when more marked differences in the reburrowing rates among samples were reported although they were not always statistically significant. Also in this case the presence of different internal or environmental stress could have hidden the effect of the dredging action. A seasonal trend was present with generally higher reburrowing rates during the summer and autumn periods possibly due to the related increase of the metabolic rates. With regards to the size effect, no significant differences were found between undersized and commercial clams.

Comparing the results obtained in each season at the highest level of mechanical stress (HPS), the greatest impact revealed by the biological responses was observed in the warm season (May and July) in both sites. This could be related to the increased susceptibility of clams to exogenous stress when internal stress conditions are present (i.e. during reproductive period) (Valli et al. 1985).

Even though the Principal Components Analysis exhibited a clear separation between the two clam populations, this biomarker approach did not revealed significant geographical differences in fishing impact, probably due to the heavy exploitation in both areas.

Although our overall results indicate that these acute stress indices can be used to measure the impact of hydraulic dredging on *C. gallina*, further research should aim at studying the influence of endogenous and environmental variables (reproductive period, temperature, salinity, dissolved oxygen, food availability, etc.) on the biological responses of this clam to mechanical stress.

3.1.3.6 CHRONIC BIOLOGICAL RESPONSES AS INDICES OF SHORT-TERM STRESS DUE TO HYDRAULIC DREDGING IN THE STRIPED VENUS CLAM *CHAMELEA GALLINA*

Introduction

The striped venus clam *Chamelea gallina* is an infaunal venerid bivalve widespread on well-sorted fine sand of shallow waters (0-10 m in depth), along the western coast of the North Adriatic. Clam harvesting traditionally represents one of the most relevant resources of the local fisheries. In the 1970s, the exploitation of clam beds steadily and considerably increased, due to improvements in fishing technology and the introduction of hydraulic dredging (Frogia 1989).

One of the aims of Italian team in the ECODREDGE Project was to evaluate possible stress conditions caused by hydraulic dredge fishing on natural populations of clam *Chamelea gallina*. To this end, a battery of biomarkers, both general and specific, was chosen, being these biological indices considered as indicative of natural and/or anthropogenic stress (Bayne et al. 1985 ; Mayer et al. 1992; McCarthy and Shugart 1990 ; Moore 1990). The over-fishing which *C. gallina* populations of the North Adriatic Sea have undergone since the introduction of the hydraulic dredges may have contributed to induce a general stress syndrome measurable as variations of the chronic biomarkers.

However, the heavy exploitation of clam fishing areas along the North Adriatic coast made the plan of defining a sanctuary area, of a suitable size, impracticable. As a consequence, no comparison could be done between clam populations suffering from chronic stress due to fishing and a less stressed natural population (to be considered as control). Therefore we gave up the first plan and attempted an indirect estimate of dredging impact by evaluating the effects of the acute mechanical stress which clams experience during fishing operations using chronic biomarkers.

Clam samples were collected in two sites showing different environmental characteristics, mostly concerning sediment grain size and riverine influx, in order to detect possible variations in fishable clam well-being due to environmental conditions.

During the first year of activity (1999), a series of laboratory experiments was carried out mainly with the aim to test and verify if chronic indices could be able to point out acute mechanical stress in *C. gallina*. Also, experimental times and protocol were set up.

The same chronic indices applied in laboratory experiments were utilised to evaluate effects of acute mechanical stress on *C. gallina* seasonally collected in two sites (Lido and Jesolo) along the North Adriatic coast from February to October 2000.

Methodologies

Laboratory studies

Mechanical stress experiments on commercial size clams

Physiological rates and survival in air test

Clams from commercial dredging were acclimatised in the laboratory for four days in tanks with sea water at $17\pm 1^\circ\text{C}$ and 35‰ and fed with *Isocrysis galbana*.

Mechanical stress conditions due to fishing were simulated using a vortex mixer: clams were arranged in a box, shaken for 6 min at the maximum speed (40 hertz) and then kept 30 min out of water before evaluating filtration and respiration rates and survival in air. Timing for stress exposure was chosen taking into account that, immediately after stress, clams need to be processed alive and for a long time.

Samples of both control and stressed clams have been also maintained for 24 hrs at a temperature of 4°C . This approach is needed because of some constraints in field studies, when clams arrive at the laboratory in the afternoon: due to the long experimental time (18 hrs) required for physiological measurements, an immediate processing of collected samples is impracticable, and clams must be kept refrigerated until the following morning. These experiments allowed us to verify the effect of low temperature on filtration and respiration rates of clams.

Food absorption efficiency was estimated in the laboratory to calculate Scope for Growth: clams were fed in continuous with *Isocrysis galbana* at concentration of 15000 cell/ml for 24h; the organic matter content in seston was evaluated filtering sea water from tanks.

Moreover, clams were replaced in aquarium after the mechanical shaking for 12hrs and 24hrs to determine recovery times of stressed organisms, and then physiological measurements and survival in air test were evaluated.

Biochemical and histochemical biomarkers

As for biochemical and histochemical biomarkers, mechanical stress due to dredging was simulated in the lab shaking manually two sub-samples of 200 clams each. A 3-minute shaking was repeated every 30 minutes up to 6 hours. The treatment was performed inside a box rotated at estimated frequency of about 60 rpm. During the 30 min intervals, the animals were kept in aerated aquaria.

One of the two sub-samples was processed for the different analyses at the end of the treatment, the other one was kept in aquarium for the following 24 hrs before being processed. As controls, both clams collected manually and clams from the same dredged sample used for the experiment, but not mechanically stressed, were used.

Mechanical stress experiments on under-sized clams

Survival in air test was performed in a laboratory experiment carried out in order to evaluate the effects of dredging and mechanical sorting on under-sized clams which are rejected into the sea bed after sieved during the normal fishing activity.

Under-sized clams (13-19mm), commercially dredged and acclimatised for 4 days in the laboratory, were shaken for 10 minutes using a vortex at maximum speed (40

hertz), the same operation being repeated once a day in the next three days (3 control and 3 treated samples).

Effects of temperature and body size on physiological rates

The influence of body size and temperature on physiological rates (filtration and respiration) was studied. The effect of body size can be removed from physiological measurements using the simple allometric equation:

$$Y=aX^b$$

where:

- Y is the physiological parameter (clearance rate, CR, or respiration rate, RR)
- X is the body size expressed as dry weight (DW) of experimental organisms
- a is the intercept of the regression line and represents the value of physiological measurements/unit of dry weight (1 g)
- b is the slope of the regression line.

When there is no significant difference in slopes at different temperatures, it is possible to estimate a common regression coefficient b, which will be used to recalculate the intercept.

The experiments were carried out on different size clams (from about 0.05g to 0.5 g dry weight) at three temperatures: 8°C, 16°C and 24°C related to winter, spring-autumn and summer conditions, respectively. Before any measurement, clams were kept in the laboratory for five-six days at the experimental temperature and fed with *Isocrysis galbana*.

Statistical comparison of a and b values were performed using Covariance Analysis-Parallelism test. Significance of correlation coefficient (r) was also determined.

Field study

With the aim to evaluate the effects of different fishing systems on a seasonal basis, four samplings on natural *C. gallina* grounds were carried out in February, May, July and October 2000.

Clam samples were collected in two fishing areas along the west coast of the North Adriatic Sea: Jesolo and Lido at about 5 m depth, using four fishing methodologies:

- dredging at high water pressure (~ 2.5 atm) and using a mechanical sieve for sorting (as in commercial fishing, HPS samples);
- dredging at high water pressure without sorting (HP samples);
- dredging at low water pressure (~ 1 atm) without sorting (LP samples);
- manual SCUBA collecting (M samples, not collected in February at both sites and in July at Jesolo).

To standardise the biological responses, when possible, commercial size clams were used for analysing all the above reported stress indices. In addition, in the samplings of July and October 2000, possible stress due to dredging and mechanical sorting was evaluated on under-sized clams. Organisms (13 - 19 mm length) were collected from *by catch* and survival in air test was performed.

Stress indices

Physiological parameters

Physiological parameters were measured on 16 individual clams per sample according to Widdows (1985).

After seasonal sampling, clams were transferred to the laboratory in refrigerated boxes and then maintained at low temperature until the following morning: this approach was needed because of the long time required for measurements.

Clams were acclimatised for 30 min in filtered sea water before measurements. All physiological parameters were evaluated at the same temperature measured in the sea water during sampling.

Filtration Rate

Filtration rate, expressed as volume of water cleared of suspended particles per unit time, was determined using a static approach: each animal was put in a glass tank containing 1 l filtered seawater and 15000 cell/ml of *Isocrysis galbana*. Measures were carried out every 30 min for 2 hours using a Coulter Counter (Model Z2).

Respiration Rate

Rates of oxygen consumption were monitored for 60 min using closed plexiglas respirometers (about 600 ml capacity). Air-saturated filtered seawater was added to each respiration chamber provided with a calibrated oxygen electrode (Strathkelvin 1302) which was connected to a multi-channel oxygen system (Strathkelvin 928 Interface).

Absorption efficiency

Food absorption efficiency was estimated according to the method of Conover (1966); the proportion of organic matter in the seston was compared with that in the faeces.

The organic matter content (OM) in the seston was evaluated by filtration of sea water from each sampling site on Whatman GFC fibre filters. OM in the faeces was determined on four sub-samples of 50 clams in tanks containing filtered sea water; after 4 hrs faeces were collected on Whatman GFC fibre filters. Filters with seston and faeces were dried in oven at 60°C, weighted after 48 hrs and then ashed at 450°C and re-weighted.

SFG

The physiological measurements were weight-corrected to a 'standard' body mass of 0.2 g dry weight (coefficient $b=0.884$ for filtration and $b=0.799$ for respiration rate) and then converted to energy equivalents ($J h^{-1}$) for the calculation of an energy budget and thus scope for growth.

Scope for growth represents the difference between the energy absorbed from food minus the energy expenditure associated with respiration and provides an instantaneous measure of the potential growth of an animal (Smaal and Widdows 1994).

Statistical comparisons were performed using Anova test.

Survival in air

For the survival in air test 30 animals per sample were used: clams were subjected to anoxia by air exposure at 18°C in humidified chambers (Eertman et al. 1993). Survival was assessed daily until 100% mortality rate was reached.

Analysis of survival was performed according to the method of (Kaplan and Meier 1958); the significance of differences between groups was tested using the Gehan and Wilcoxon test (Gehan 1965).

Biochemical analyses

The activity of the anti-oxidant enzyme catalase was determined on three replicated pools (3 g each) of clam digestive glands (Livingstone et al. 1992). The digestive glands of clams were dissected out, damp-dried, immediately frozen in liquid-nitrogen and stored at -80°C prior to analyses. The glands were homogenised with buffer (w : v = 1 : 4) (0.606 g tris HCl 10mM, 875.58 g Sucrose, 5.59 g KCl in 500 ml milliQ, pH 7.6) by means of a potter at 400 rpm (Livingstone 1988).

The homogenated glands were centrifugated at 2000 rpm for 10 minutes at 4°C, and the supernatant was centrifugated at 9000 rpm for 40 minutes at 4°C. The cytosolic fraction was obtained from the supernatant, while the pellet fraction was obtained from the pellet after resuspension with the buffer. The samples were stored at -80°C before the analyses. Cytosol catalase activity was determined by the decrease in absorbance at 240 nm, due to the H₂O₂ consumption (Aebi 1974). Protein concentration was determined according to Lowry et al. (1951).

Histochemical analyses

Latency of lysosomal enzymes

The latency of the enzyme N-acetyl-β-hexosaminidase was evaluated assessing the fragility of the lysosomal membranes according to the methods of Moore (1976). A small piece of freshly excised tissue of digestive gland from each of 10 individuals was supercooled at -70°C in hexan bath and stored at -30°C until required for sectioning (Bitensky et al. 1973). Sections (10 μm) were cut in a cryostat at -25°C. Latent lysosomal activity of the lysosomal enzymes N-acetyl-β-hexosaminidase was demonstrated in the digestive cells using naphthol AS-BI substrates and post-coupling with diazonium salts were pretreated in a staining jar for 0 – 35 min at intervals of 5 minutes in 0.1M citrate buffer pH 4.5 containing 2.5% NaCl (w:v) at 37°C in order to labilise the lysosomal membranes. Following this pretreatment sequence the set of slides was transferred to the substrate incubation medium containing naphthol AS-BI-β-D-glucosaminide (Sigma) and low viscosity polypeptide (Polypep, Sigma) to act as a section stabiliser. The slides were subsequently transferred to 0.1M phosphate buffer pH 7.4 containing a diazonium coupler, fast violet blue, at room temperature for 10 minutes. The slides were then rinsed rapidly in a running tap water, fixed for 10 minutes in calcium formol containing 2.5% NaCl (w:v) at 4°C, rinsed in distilled water and mounted in aqueous mounting medium. The labilisation period is the time

of pretreatment required to fully labilise the lysosomal membranes resulting in maximal staining intensity for the enzymes (Moore 1976).

February samples were not analysed due to problems in their processing.

RNA/DNA ratio

Nucleic acid content was measured according to the method of Caldarone and Buckley (1991). Individual clams were analysed for the nucleic acid content and the RNA/DNA ratio. Small pieces of gill were dissected, placed in eppendorfs and immediately frozen in liquid nitrogen and stored at -80°C .

Nucleic acids were extracted from the tissue by adding 150 μl of 1% sarcosine (sodium N-laurosarcosine) in Tris-EDTA buffer (pH=8.0), and subsequently 1.35 ml Tris-EDTA buffer, then centrifugating at 2500 xg for 5 min at $+4^{\circ}\text{C}$. Aliquots of the supernatant were used for the analysis of fluorescence-photometric measures using ethidium bromide, a specific nucleic acid fluorochrome dye. The fluorescence due to total RNA was calculated as the difference between total fluorescence (RNA+DNA) and the fluorescence after ribonuclease A treatment, which was assumed to be due to DNA.

The fluorescence was determined by exciting at 365 nm and reading at 610 nm with a spectrofluorometer (Perkin-Elmer). The concentration of nucleic acids were determined by running standard curves of DNA and RNA standard solutions in the appropriate range of values.

Condition index

Condition index was determined according to the method of Walne and Mann (1975)

$$\frac{\text{Dry weight of meat (g) x 1000}}{\text{Dry weight of shell (g)}}$$

Measurements were made seasonally on 50 clams from each sampling site; meat and shell of individual clams were put separately in oven at 60°C and weighted after 48 hr.

Statistical comparisons were performed using Anova test.

Multivariate analysis

The Principal Component Analysis (PCA) was performed on all the above considered biological parameters with the aim of evaluating in a scatterplot possible site and pressure effects on clam responses (Lagonegro and Feoli 1986).

Results

Laboratory studies

Mechanical stress experiment on commercial size clams

Physiological rates and survival in air test

Clams subjected to mechanical stress showed lower physiological rates than control clams: both filtration and respiration rates resulted significantly different ($p < 0.05$) when control and stressed organisms were compared. The same decreasing trend was observed also in SFG, although the difference was not significant. SFG was calculated according to a food absorption efficiency value of 40.7% determined at the experimental organic matter content of 0.356 mg/l (corresponding to 15000 cells/ml). Overnight refrigeration caused a slowing down of physiological rates, particularly strong for clearance rate; the trend observed in non-refrigerated samples was maintained even if a significant difference was observed only for respiration rate ($p < 0.05$) (Figure 1).

Survival in air showed a significantly lower LT_{50} value in stressed clams compared to control (6 days vs 10 days; $p < 0.001$) (Figure 2).

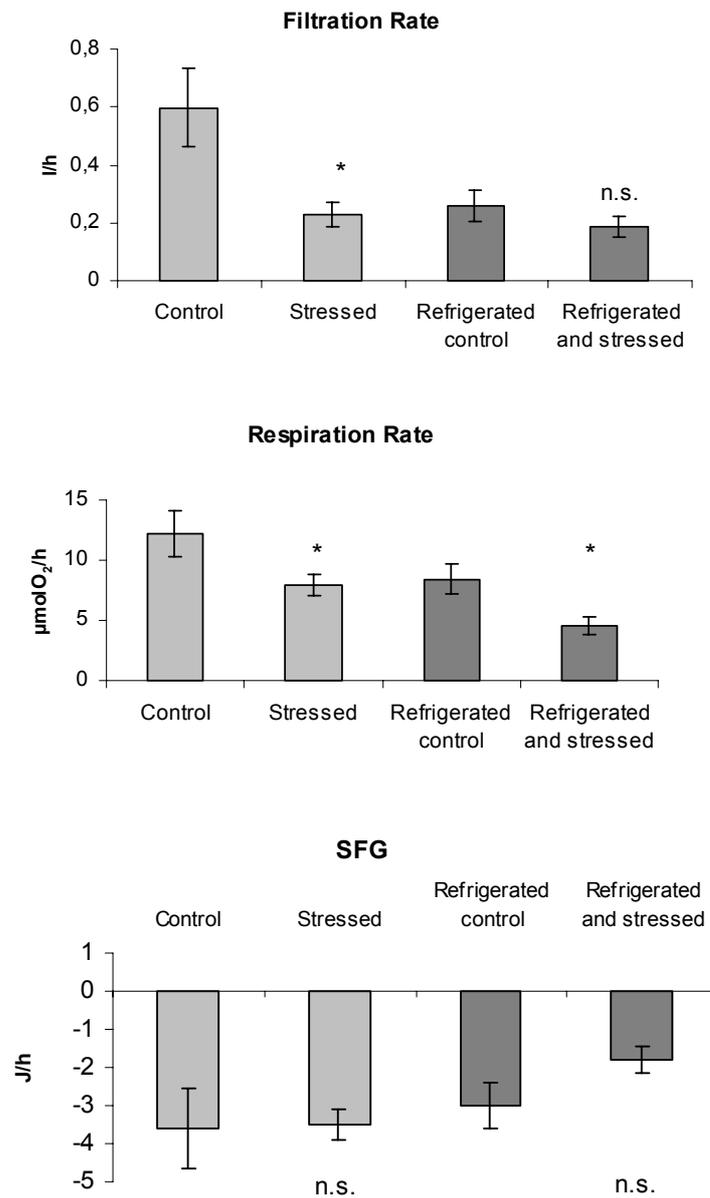


Figure 1 Physiological measurements in *C. gallina* after mechanical stress and after mechanical stress plus refrigeration. Mean \pm e.s. Statistical comparison Anova: * $p < 0.05$.

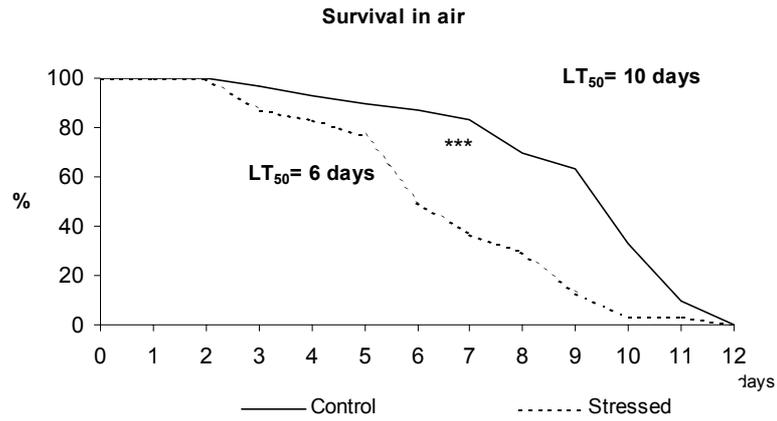


Figure 2 Survival in air of *C. gallina* after mechanical stress. N=30. Statistical comparison Wilcoxon & Gehan: * p<0.001.**

Recovery experiment

Soon after 12 hrs, physiological rates of stressed clams showed no significant differences when comparing with control ones. Nevertheless the lowest values of filtration rate and SFG and highest value of respiration rate (if compared with the 24 hrs values), demonstrated that recovery was taking place (Figure 3).

Similar results were obtained in survival in air test: the LT_{50} values for the 24 hrs recovered clams rose up to 6 days, the same value as in control, while 12 hrs recovered clams exhibited a significantly lower LT_{50} of 5 days ($p < 0.001$; Figure 4).

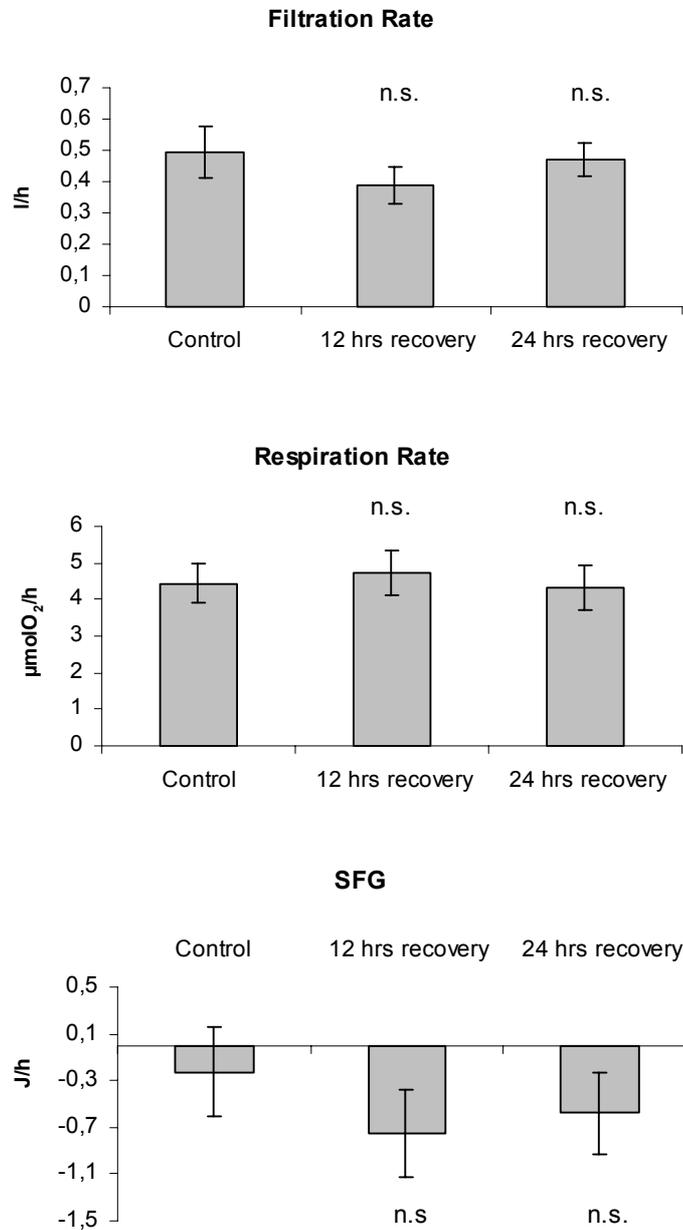


Figure 3 Physiological measurements in *C. gallina* after 12 hr and 24 hr post-stress recovery periods. Statistical comparison Anova.

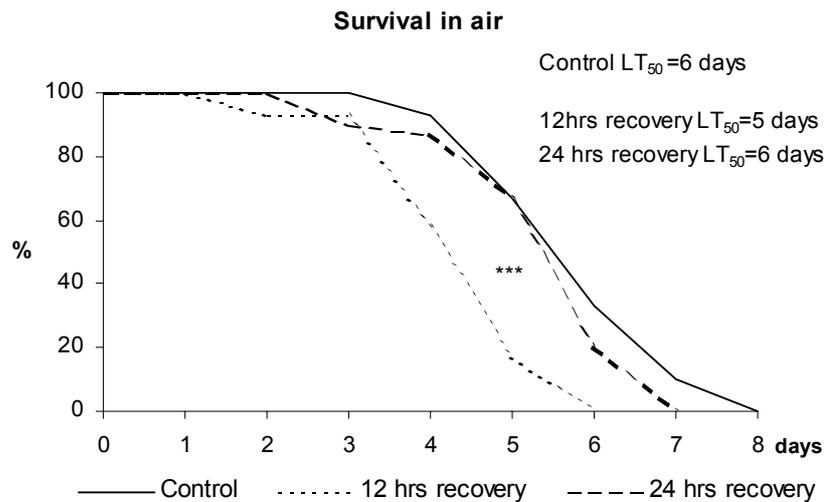


Figure 4 Survival in air of *C. gallina* after 12 hr and 24 hr post-stress recovery periods. N=30. Statistical comparison Wilcoxon & Gehan: *** $p<0.001$.

Biochemical and histochemical biomarkers

The activity of SOD (Figure 5) in the stressed sample showed values slightly higher than in its control and was significantly higher than in the 24 hrs recovered sample, while there was no difference between manual and dredged samples. In general, the 6 hrs treated sample exhibited a little induction of enzyme activity with respect to the no-treated control and a certain capacity of recovery after 24 hours.

The catalase activity was measured both in cytosol and pellet fractions and the values summed up as total activity. The data showed a significant induction in the treated sample with respect to the control and even to the recovered sample, which is not different from its control (Figure 6). The protein level did not give clear indication of a stress effect due to mechanical treatment. Only the cytosolic proteins showed a slight depletion in the stressed sample and no difference was observed between the recovered sample and its control (Figure 7). Measures of pellet proteins were inconclusive: stressed samples showed values significantly higher than the controls, while there was no statistical difference between recovered samples and their controls (Figure 8). As for the content of G-25 proteins it was quite similar in all samples except for the 24 hr control that showed a significant depletion (Figure 9).

The latency of N-acetyl- β -hexosaminidase in the laboratory experiment resulted a sensitive biomarker of mechanical stress. In fact experimental samples showed values significantly lower than the manually collected and dredged ones, although there was no difference between treated samples and their control and between recovered samples and their control (Figure 10).

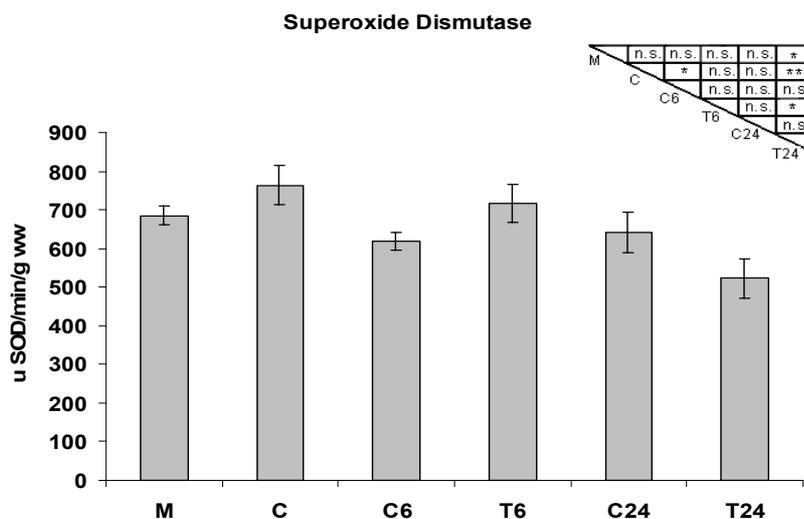


Figure 5 Superoxide dismutase activity in *C. gallina* subjected to a shaking experiment (mean \pm s. e., $n = 3$). Statistical comparison Anova: * = $p < 0.05$; ** = $p < 0.01$; n.s. = not significant.

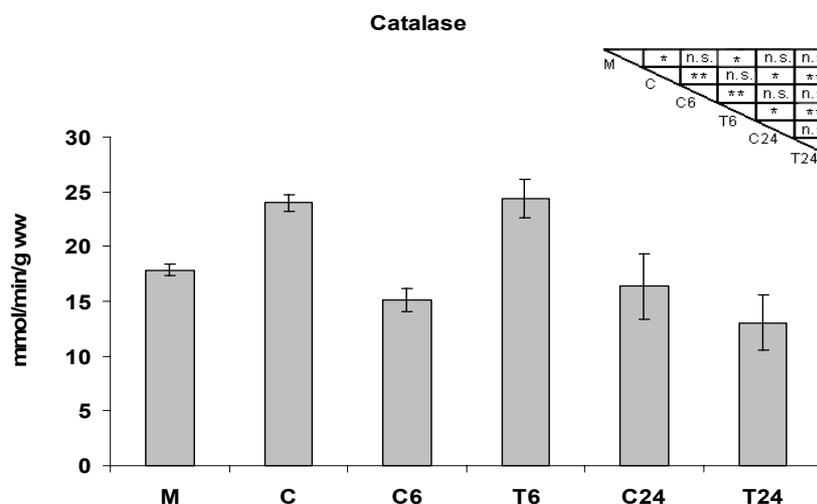


Figure 6 Catalase activity in *C. gallina* subjected to shaking experiment (mean \pm s. e., $n = 3$). Statistical comparison Anova: * = $p < 0.05$; ** = $p < 0.01$; n.s. = not significant.

Legend: M = control sample manually collected; C = control sample collected by commercial dredge and processed immediately; C6 = control sample collected by commercial dredge and processed after 6 hours; T6 = 6 hour-treated sample collected by commercial dredge and processed immediately; C24 = control sample collected by commercial dredge and processed after 24 hours; T24 = 6 hour-treated sample collected by commercial dredge and processed after 24 hours recovery.

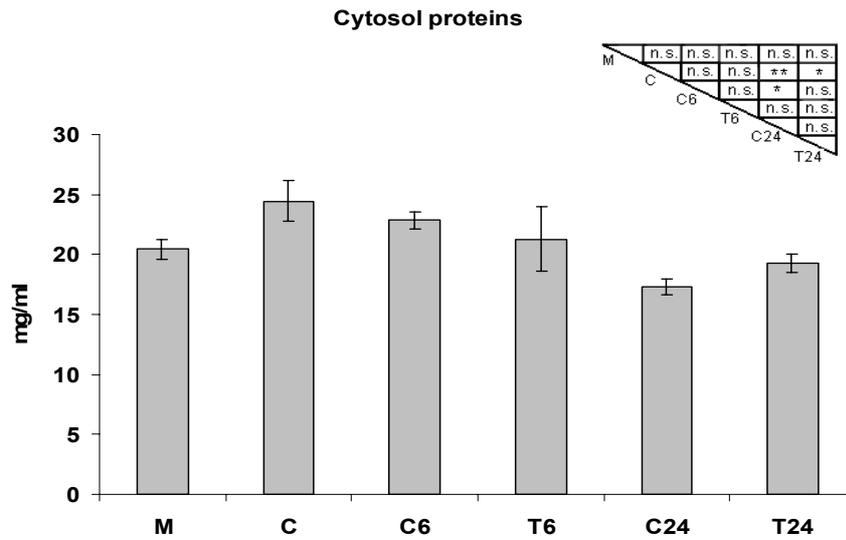


Figure 7 Cytosolic protein content in *C. gallina* subjected to shaking experiment (mean \pm s. e., n = 3). Statistical comparison Anova: * = p < 0.05; ** = p < 0.01; n.s. = not significant.

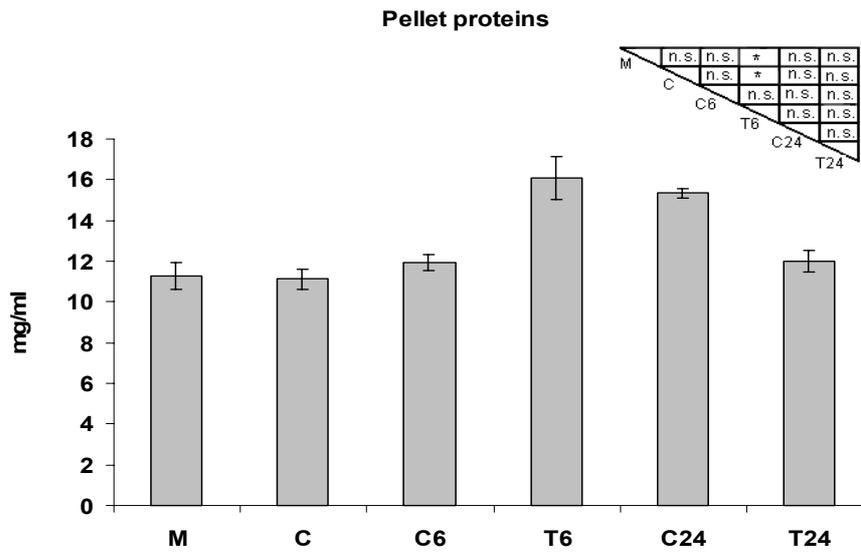


Figure 8 Pellet protein content in *C. gallina* subjected to shaking experiment (mean \pm s. e., n = 3). Statistical comparison Anova: * = p < 0.05; n.s. = not significant.

Legend: M = control sample manually collected; C = control sample collected by commercial dredge and processed immediately; C6 = control sample collected by commercial dredge and processed after 6 hours; T6 = 6 hour-treated sample collected by commercial dredge and processed immediately; C24 = control sample collected by commercial dredge and processed after 24 hours; T24 = 6 hour-treated sample collected by commercial dredge and processed after 24 hours recovery.

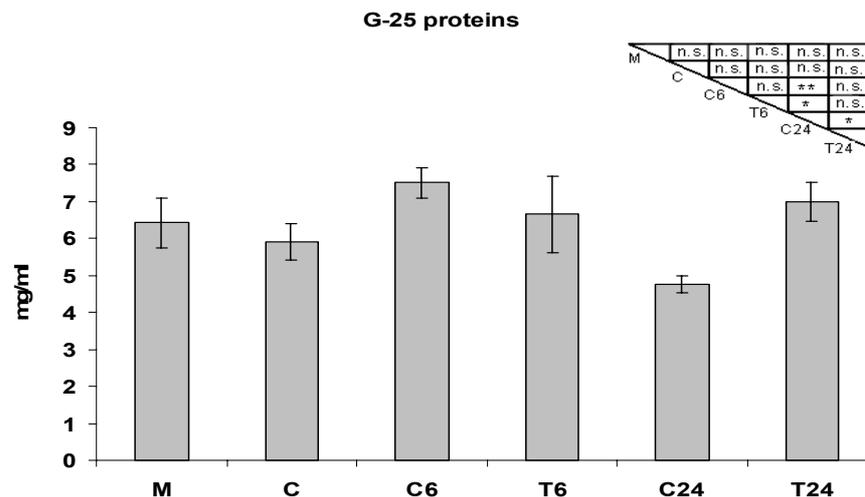


Figure 9 G-25 protein content in *C. gallina* subjected to shaking experiment (mean \pm s. e., n = 3). Statistical comparison Anova: * = p < 0.05; ** = p < 0.01; n.s. = not significant. Legend: M = control sample manually collected; C = control sample collected by commercial dredge and processed immediately; C6 = control sample collected by commercial dredge and processed after 6 hours; T6 = 6 hour-treated sample collected by commercial dredge and processed immediately; C24 = control sample collected by commercial dredge and processed after 24 hours; T24 = 6 hour-treated sample collected by commercial dredge and processed after 24 hours recovery.

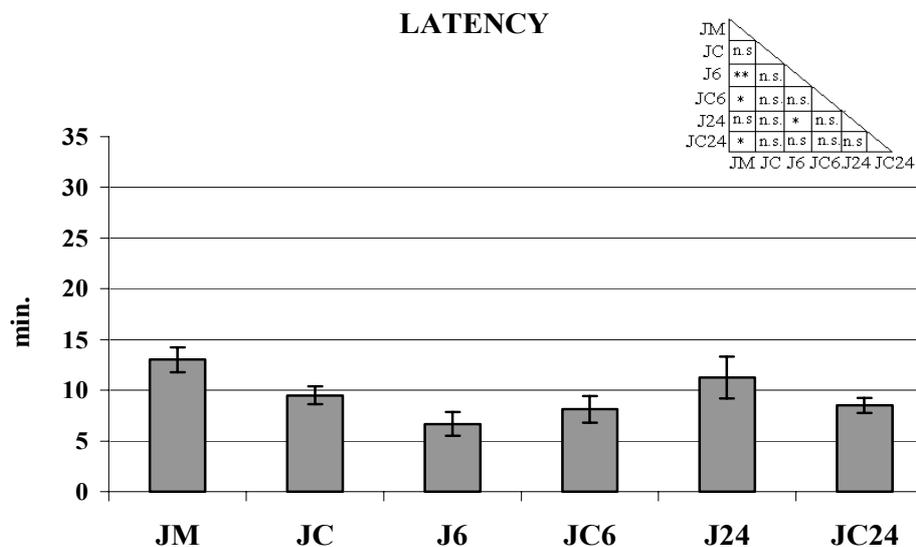


Figure 10 Latency of N.A.H. enzyme in *C. gallina* subjected to a shaking experiment (mean \pm s. e., n = 10). Statistical comparison Anova: * = p < 0.05; ** = p < 0.01; n.s. = not significant.

Legend: JM = control sample collected by divers; JC = control sample collected by commercial dredge and processed immediately; J6 = 6 hour-treated sample collected by commercial dredge and ; JC6 = control sample collected by commercial dredge and processed after 6 hours; J24 = 6 hour-treated sample collected by commercial dredge and processed after 24 hours recovery; JC24 = control sample collected by commercial dredge and processed after 24 hours.

Effects of temperature and body size on physiological rates

The results of the evaluation of body size effect on filtration and respiration rates for three different temperatures (8°C, 16°C and 24°C) are reported in Figures 11, 12.

The values of b and a coefficients of the regression lines calculated at 8, 16 and 24°C were compared by covariance analysis - parallelism test: no significant differences were observed in slopes (b values) for both filtration and respiration rates. The intercepts a of the regression lines for filtration showed significant differences when comparing the values at 24°C with both those at 8°C (p<0.01) and 16°C (p<0.001); whereas, no differences were observed between a values at 8 and 16 °C. No differences were detected when comparing a values of regression lines for respiration. The values of a, b and r (correlation coefficient) for each experimental condition are reported in Tables 1, 2.

The mean values of b coefficient were calculated for both filtration and respiration rates: b=0.884 and b=0.799, respectively.

Table 1 Values of a, b (± s.e.) and r of regression lines for filtration rates at 8, 16 and 24°C.

	N° animals	Intercept (l/h/g)	a Regression coefficient b	Correlation coefficient
8°C	27	0.994±0.140	0.894±0.173	0.719***
16°C	40	0.846±0.158	0.893±0.214	0.560***
24°C	31	1.552±0.147	0.866±0.199	0.629***

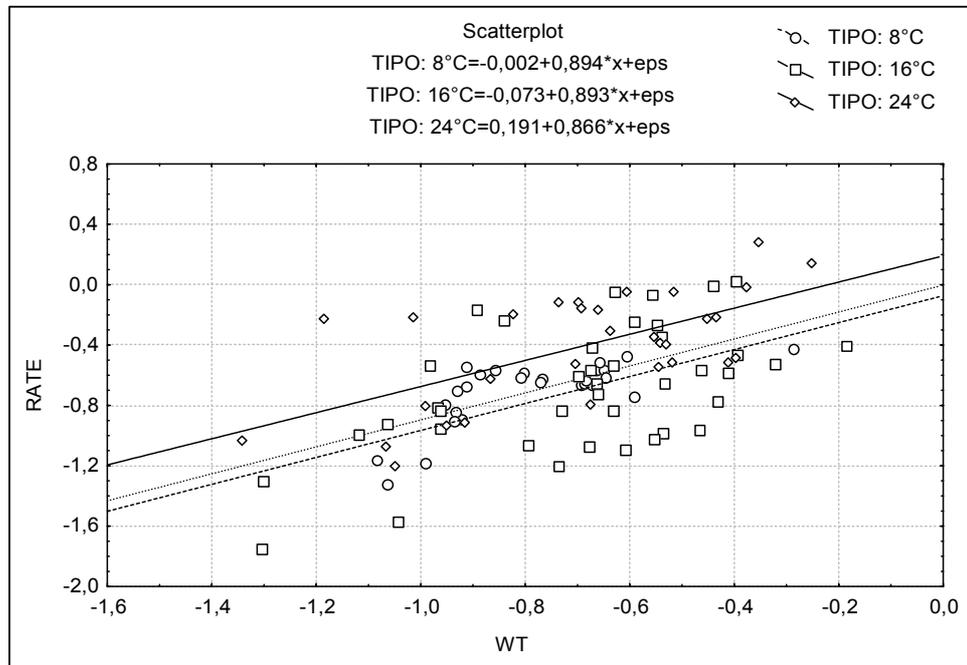
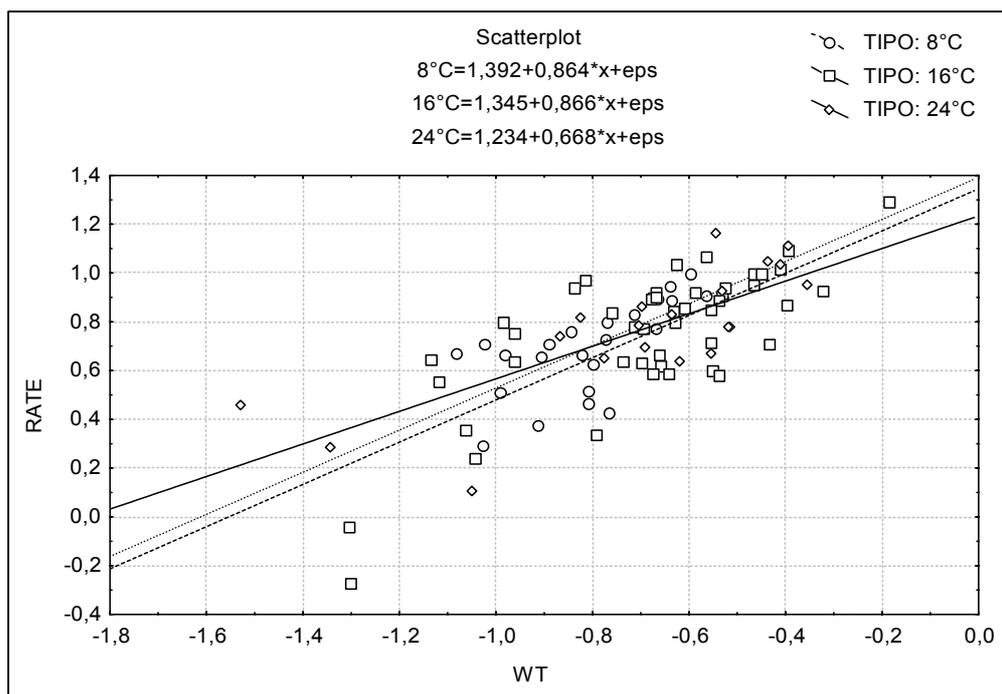


Figure 11 Linear regression relating dry weight of *C. gallina* to filtration rate at different temperatures.

Table 2 Values of a, b (\pm s.e.) and r of regression lines for respiration rates at 8, 16 and 24°C.

	N° animals	Intercept (l/h/g)	a Regression coefficient b	Correlation coefficient
8°C	24	24.683 \pm 0.166	0.864 \pm 0.203	0.671***
16°C	46	22.156 \pm 0.083	0.866 \pm 0.114	0.753***
24°C	20	1.552 \pm 0.096	0.668 \pm 0.126	0.779***


Figure 12 Linear regression relating dry weight of *C. gallina* to respiration rate at different temperatures.

Mechanical stress experiment on under-sized clams

Survival in air

Mechanical stress did not affect survival times of under-sized clams treated the first and second day of experiment: no significant differences were observed comparing LT_{50} values and survival curves of control and treated animals. Stressed clams showed LT_{50} values significantly lower than controls ($p < 0.001$) only when mechanical shaking was repeated once a day for three days (Figure 13).

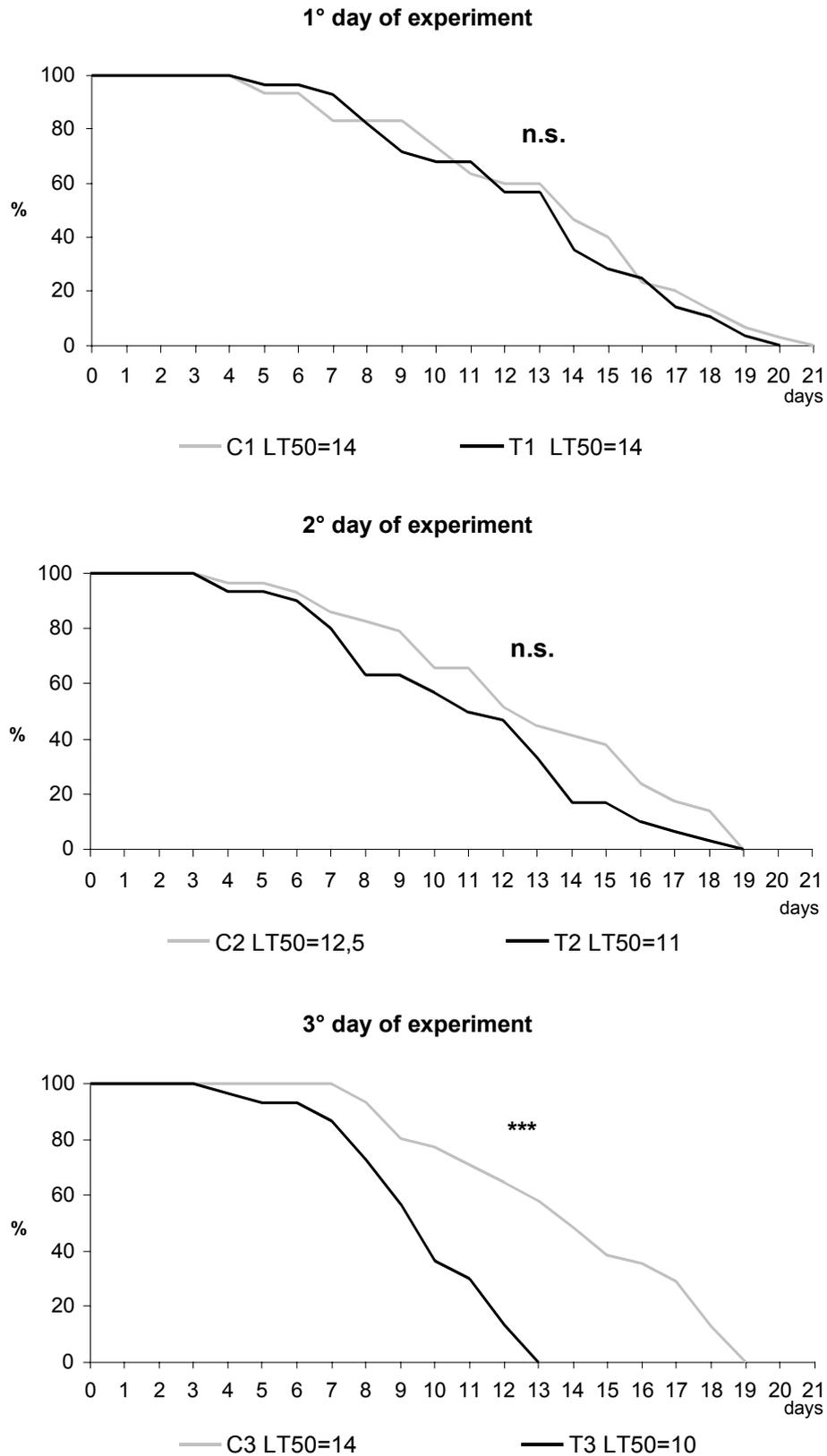


Figure 13 Survival in air of under-sized clams *C. gallina* subjected to mechanical stress. LT_{50} expressed in days, Wilcoxon & Gehan: $***p < 0.001$. Legend: C= control; T= treated.

Field studies

Catalase activity

The catalase activity generally showed no statistical differences among the different fishing systems at both sites, except for the presence of significant higher value in July for M samples at Lido and Jesolo, and in October at Lido (Figure 14).

Looking at the specific results for LP and HPS samples, significant lower values in February in comparison with the other months at both stations indicated a possible seasonal effect (Figure 15).

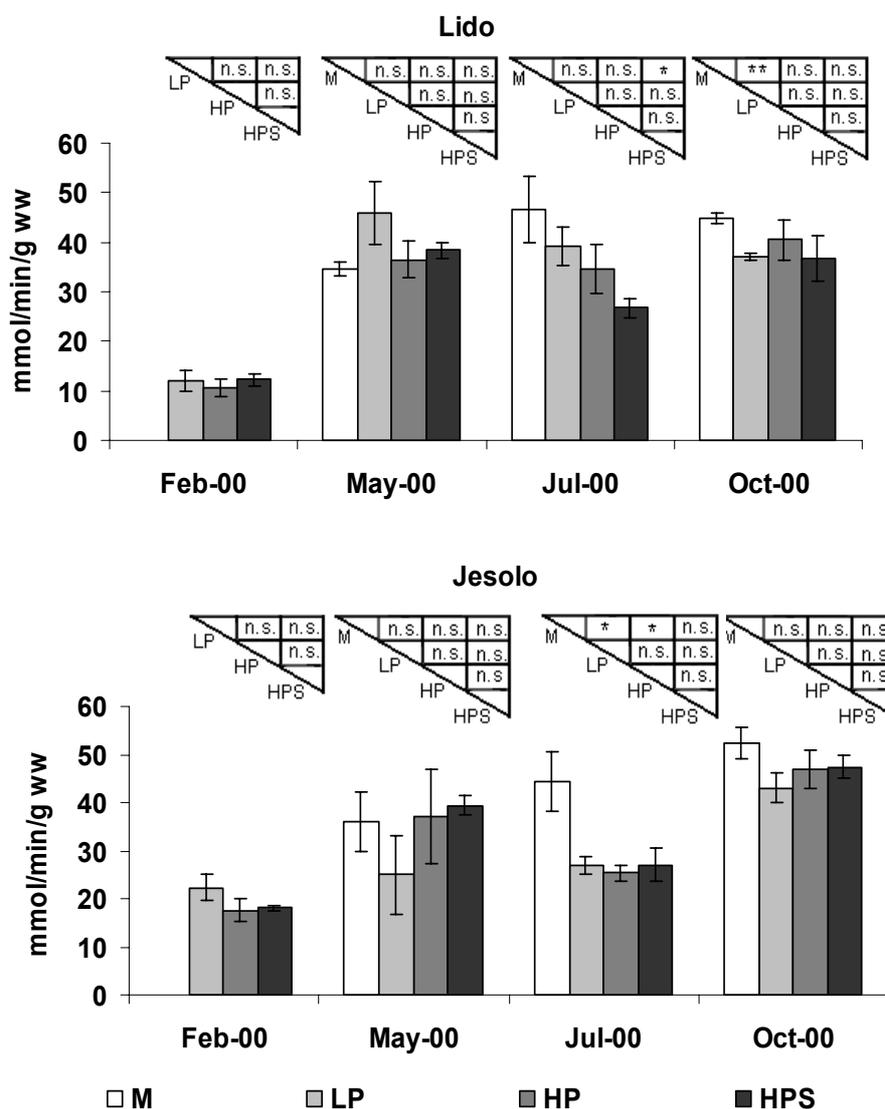


Figure 14 Catalase activity in *C. gallina* collected at Lido and Jesolo using different fishing systems (mean \pm s. e., n = 3). Statistical comparison Anova: * = p < 0.05; ** = p < 0.01; n.s. = not significant. Legend: M = manual; LP = low pressure; HP = high pressure; HPS = high pressure and mechanical sorter.

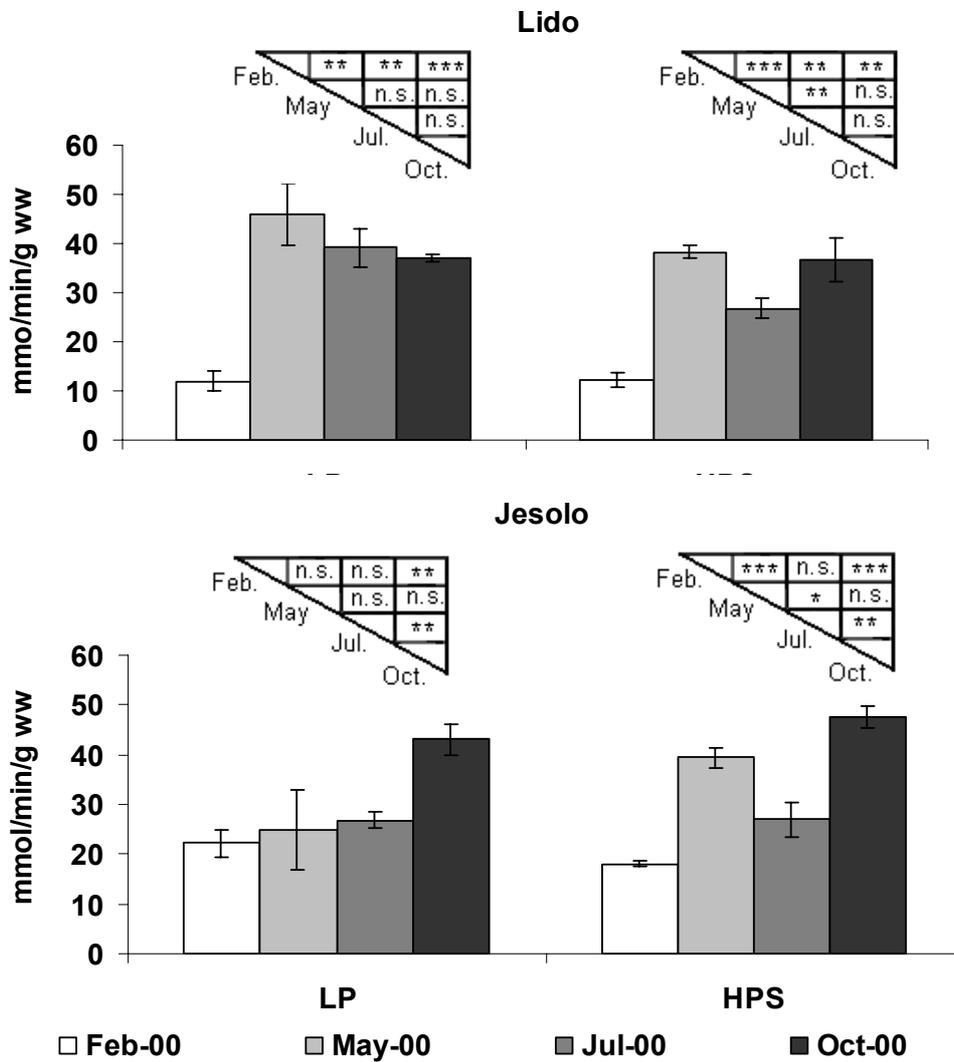


Figure 15 Catalase activity seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure plus mechanical sorter (HPS) at Lido and Jesolo, (mean \pm s. e., n = 3). Statistical comparison Anova: * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$; n.s. = not significant.

Protein content

No significant differences among the various fishing systems at both Lido and Jesolo were detected, except for lower values in HPS sample in comparison with M sample ($p < 0.05$) at Lido (October) and in HPS sample with respect to all the others ($p < 0.05$) at Jesolo (July). A significant higher value ($p < 0.05$) was also detected in HPS sample when compared with HP at Jesolo in February (Figure 16). As for the seasonal comparison, both the results reported for LP and HPS samples at Lido and Jesolo showed the same trend, with values significantly higher in May (Figure 17).

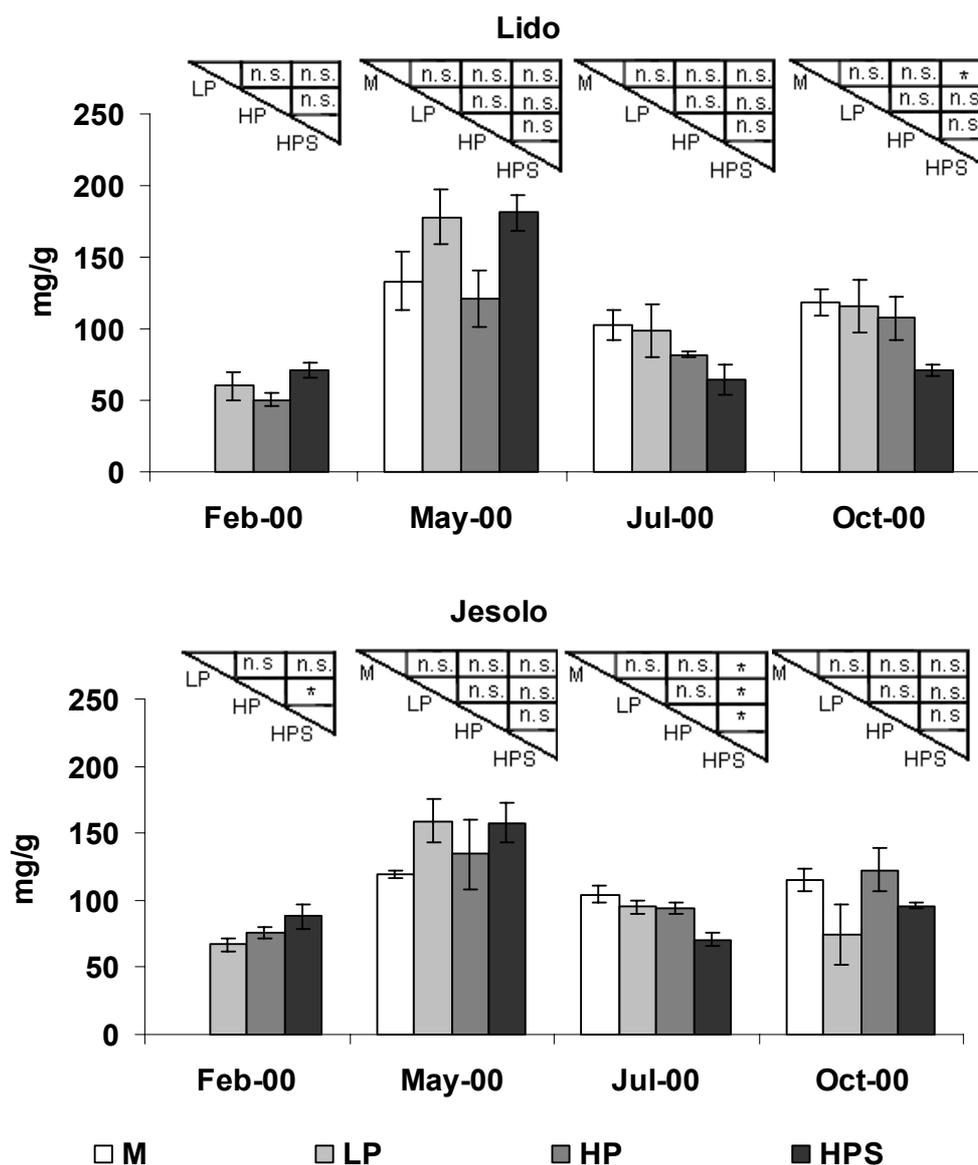


Figure 16 Protein content in *C. gallina* collected at Lido and Jesolo using different fishing systems (mean \pm s. e., $n = 3$). Statistical comparison Anova: * = $p < 0.05$; n.s. = not significant. Legend: M = manual; LP = low pressure; HP = high pressure; HPS = high pressure and mechanical sorter.

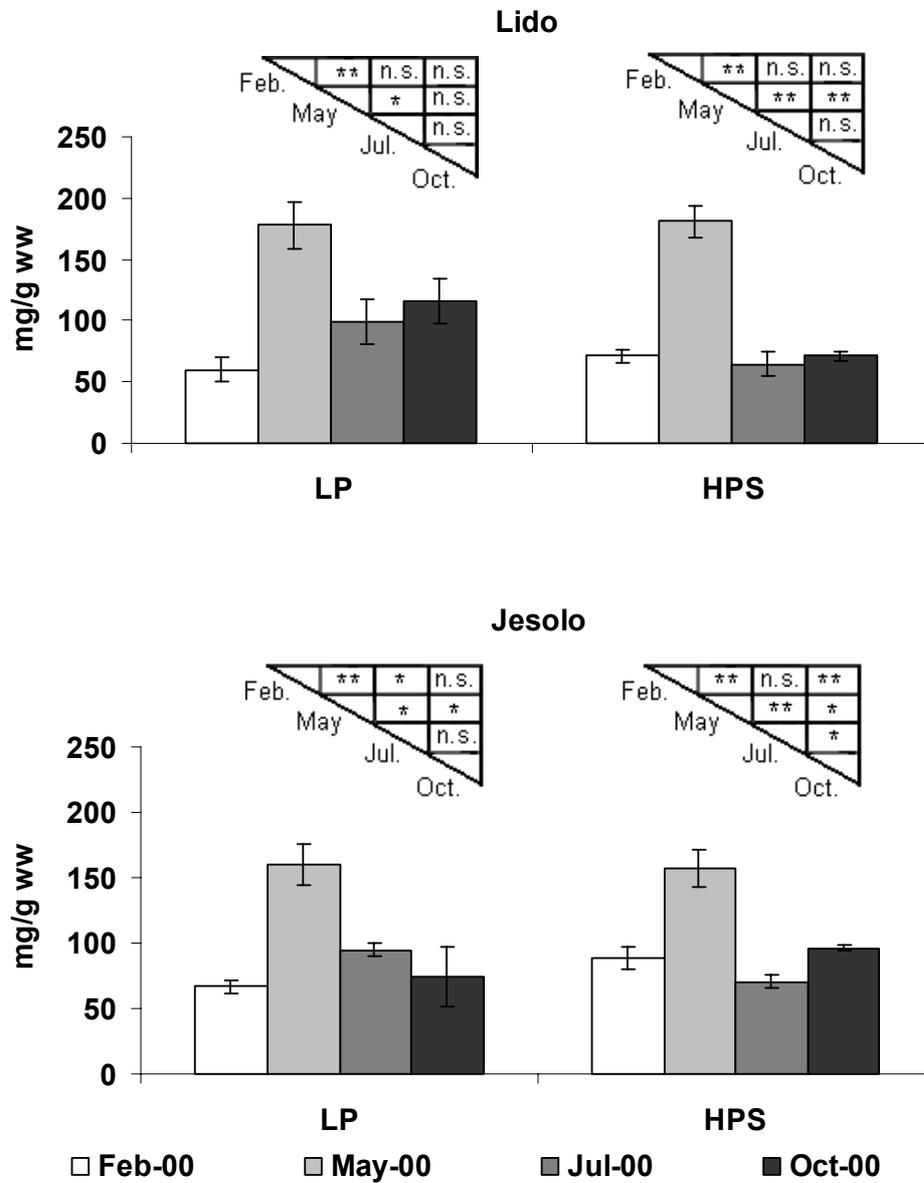


Figure 17 Proteins content seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure plus mechanical sorter (HPS) at Lido and Jesolo, (mean \pm s. e., n = 3). Statistical comparison Anova: * = p < 0.05; ** = p < 0.01; n.s. = not significant.

Latency

The latency of the enzyme N-acetyl-β-hexosaminidase did not reveal any statistical difference among the fishing systems at both Lido and Jesolo, except between M and HP samples at Lido in July ($p < 0.05$) (Figure 18). As for the seasonal comparison, the results for LP and HPS samples did not show any seasonal trend at Lido, whilst significantly higher values in May than in October ($p < 0.01$) were detected at Jesolo (Figure 19).

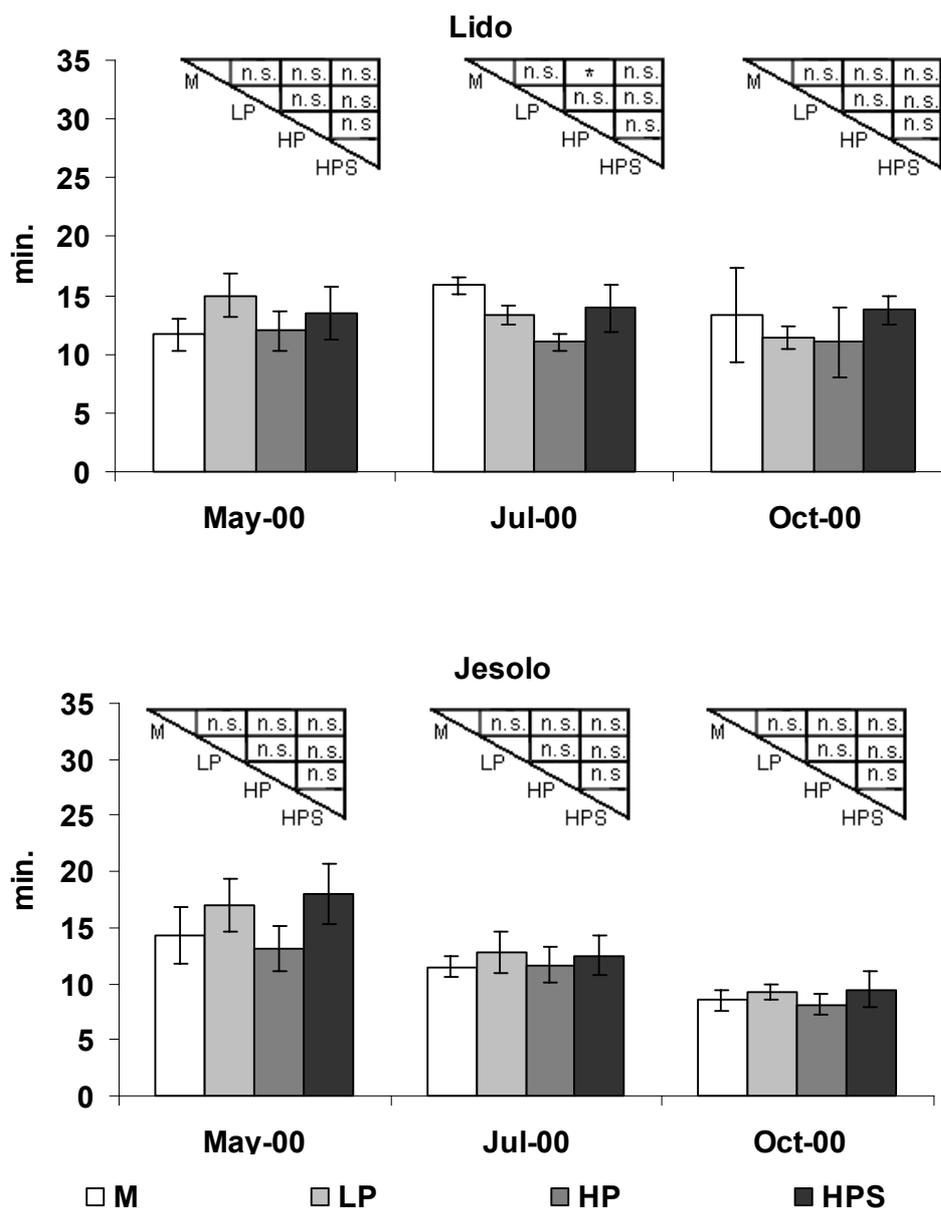


Figure 18 Latency of N.A.H. in *C. gallina* collected at Lido and Jesolo using different fishing systems (mean ± s. e., n = 10). Statistical comparison Anova: * = $p < 0.05$; n.s. = not significant. Legend: M = manual; LP = low pressure; HP = high pressure; HPS = high pressure and mechanical sorter.

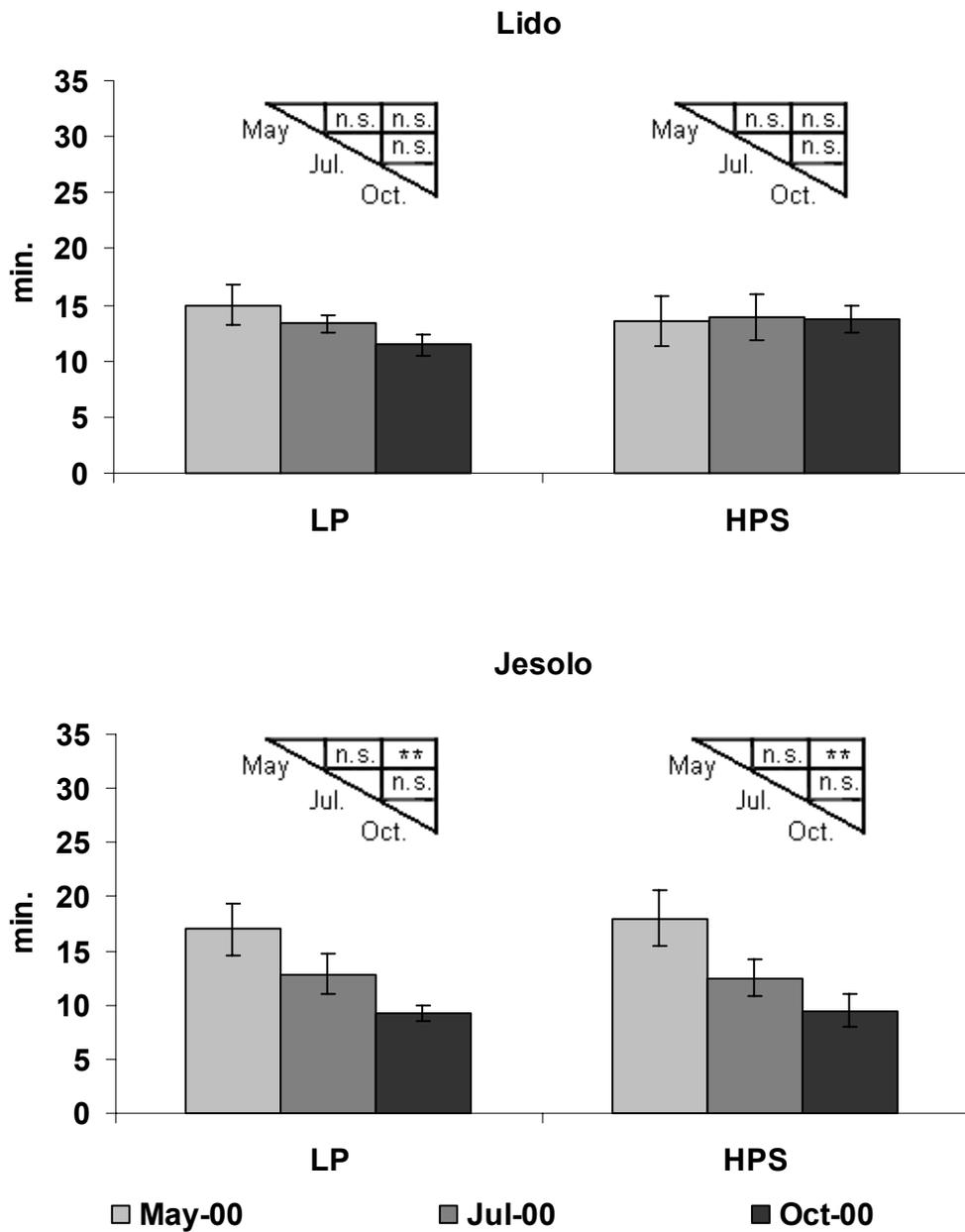


Figure 19 Latency of N.A.H. seasonal comparison in *C. gallina* collected at low pressure (LP) and high pressure plus mechanical sorter (HPS) at Lido and Jesolo, (mean \pm s. e., n = 10). Statistical comparison Anova: ** = p < 0.01; n.s. = not significant.

Filtration rate

In clams collected manually at Lido, filtration rates were higher than those of dredged clams in May and July, even if a significant difference was observed only in July in comparison with HPS sample ($p < 0.01$). The filtering activity of clams collected using hydraulic dredge showed a decreasing trend when water pressure increased and mechanical sorter was used: HPS samples were significantly different in comparison with LP samples in February and October ($p < 0.01$) and in comparison with HP sample only in October ($p < 0.05$). At Jesolo, filtration value of manually collected clams was higher with respect to HPS sample in May and October. Dredged clams showed similar values in February, whereas the expected trend was observed in May and October (Figure 20). Comparing the filtration rates on seasonal basis, at Lido the highest values were observed in October: in LP samples October value was significantly different in comparison with the other seasons, whereas in HPS samples February value was significantly lower than the others. At Jesolo, clam filtering activity was highest in July: the mean values in this month were statistically different with respect to February LP sample ($p < 0.01$) and the other HPS samples ($p < 0.001$), respectively (Figure 21).

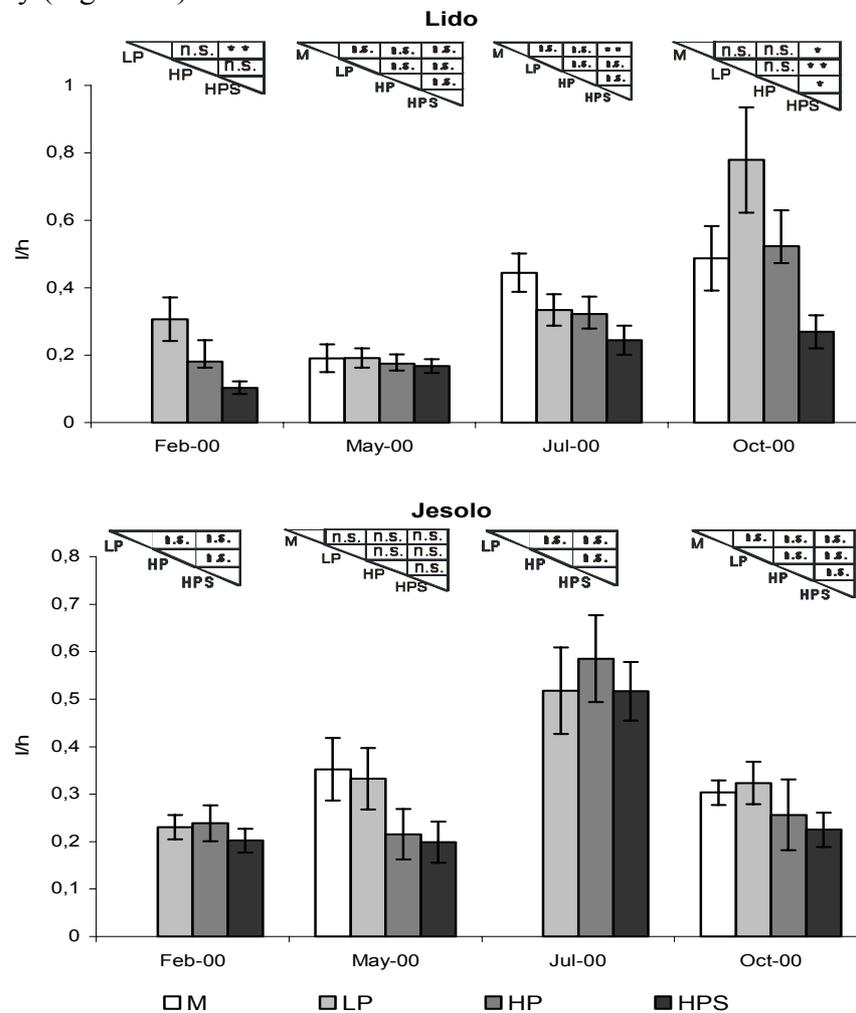


Figure 20 Filtration rate in *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.e.; Anova: * $p < 0.05$; ** $p < 0.01$.

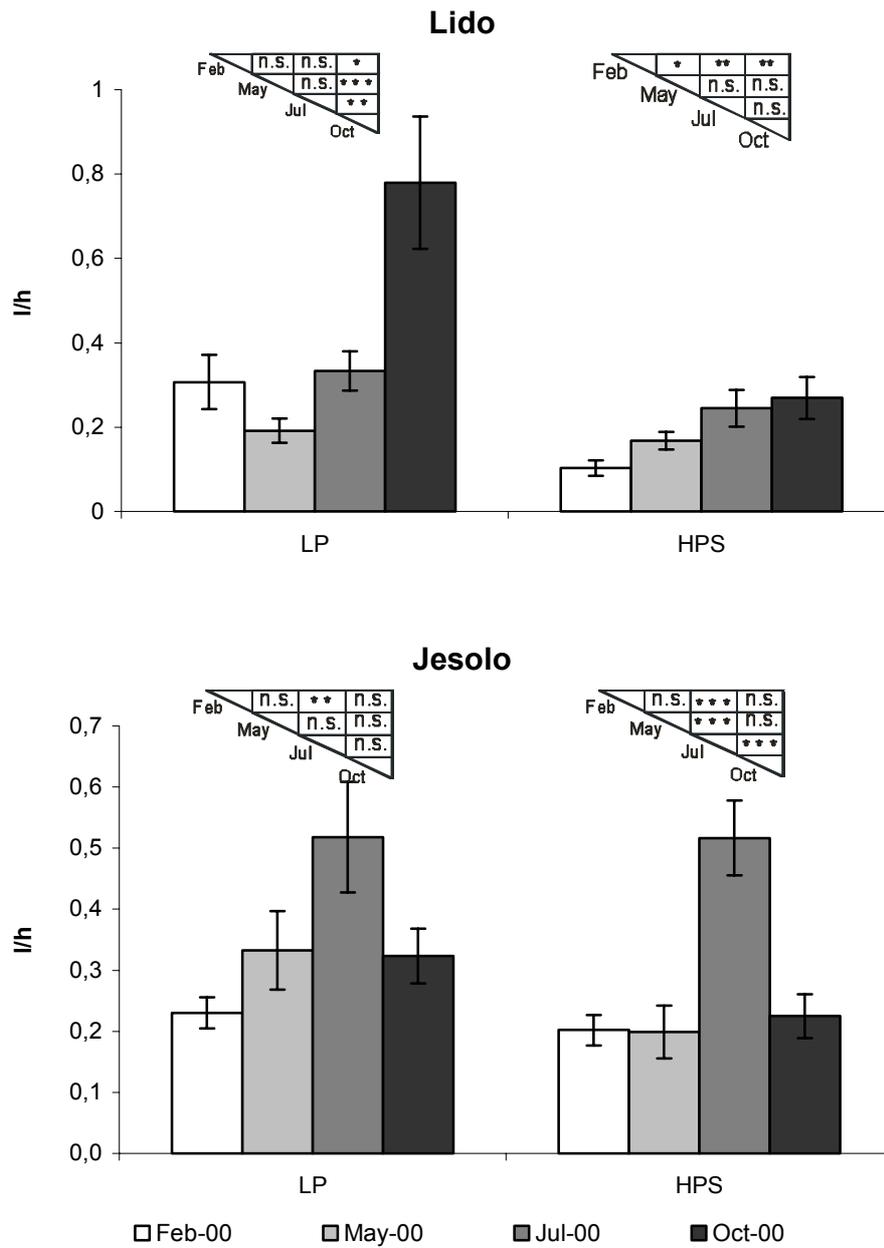


Figure 21 Seasonal comparison of filtration rates of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido and Jesolo. Mean \pm s.e.; Anova: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

Respiration rate

Respiration rates showed similar values: no significant differences were found in clams collected at Lido using the various fishing methodologies. At Jesolo, LP sample

was significantly higher than HPS in February ($p<0.01$). Clams collected manually in October showed a rate significantly lower than dredged clams ($p<0.01$ and $p<0.001$) (Figure 22).

Generally, seasonal comparison showed higher respiration values in February, always significantly different in both LP and HPS samples at Lido, and only in comparison with October LP sample at Jesolo (Figure 23).

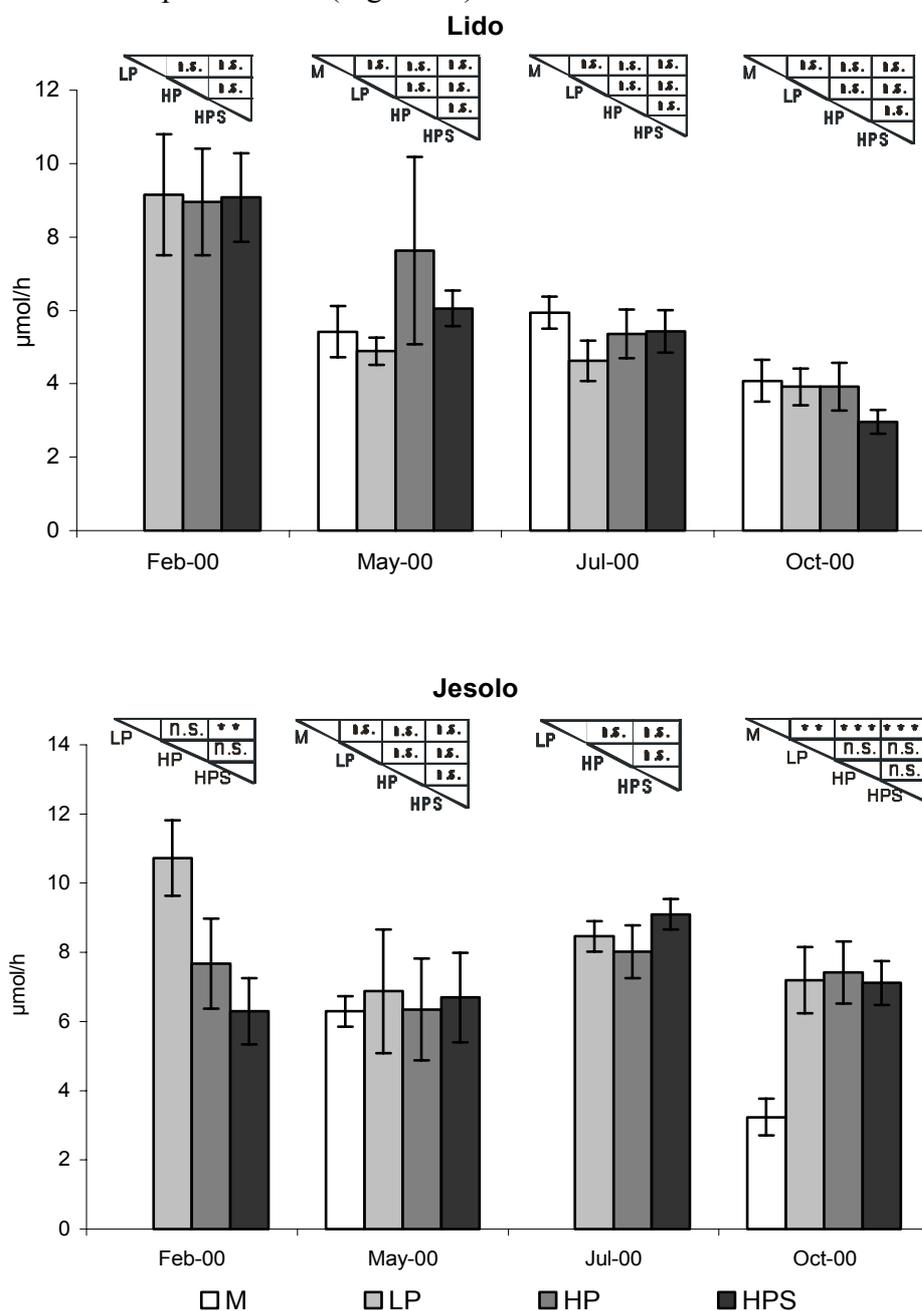


Figure 22 Respiration rate in *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.e.; Anova: ** $p<0.01$; *** $p<0.001$.

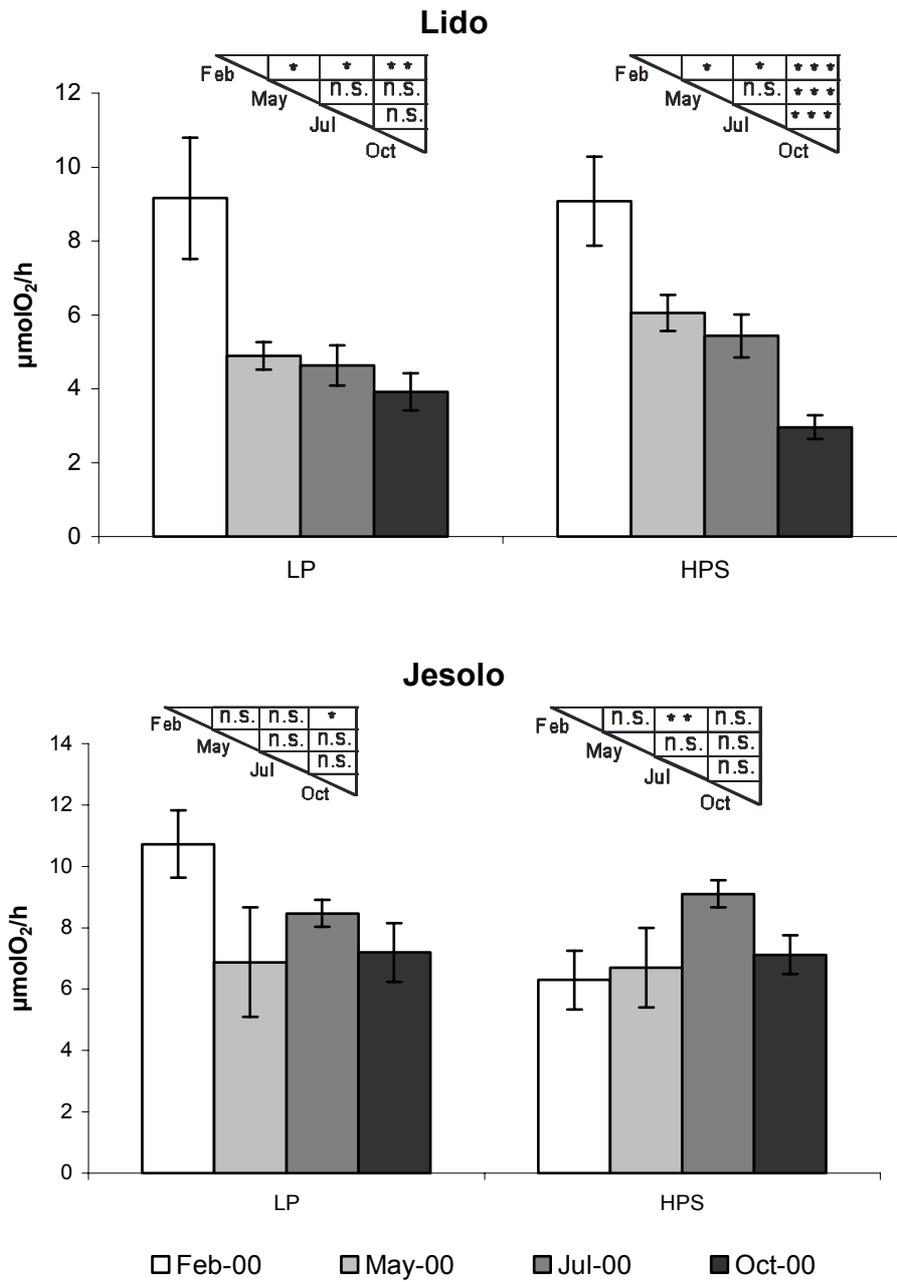


Figure 23 Seasonal comparison of respiration rates of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido and Jesolo. Mean \pm s.e.; Anova: * $p < 0.05$; *** $p < 0.001$.

Absorption efficiency

The values of organic matter content (OM) in seston and those of food absorption efficiency (EA) of clams were reported in Table 3. The lowest OM and highest AE values were found in February in both sites; whereas the lowest AE values were observed in clams collected in October.

Table 3 Organic matter content (OM) in seston and food absorption efficiency of clams (AE) at Lido and Jesolo.

	<i>Lido</i>		<i>Jesolo</i>	
	OM mg/l	AE	OM mg/l	AE
Feb-00	0.38	90.5%	0.33	86.6%
May-00	0.89	71.4%	1.13	82%
Jul-00	1.36	80.5%	0.77	82.8%
Oct-00	0.73	61.6%	0.83	54.3%

Scope for growth

This index generally showed a trend reflecting different stress levels of the fishing systems applied, both at Lido and Jesolo sites: manually-collected clams showed values significantly higher than HPS sample in July ($p < 0.05$) and at Jesolo in October ($p < 0.001$), and between LP and HPS at Jesolo in February ($p < 0.01$) (Figure 24).

Scope for growth exhibited a strong seasonal variation in both sites: lower values were observed in February, significantly different in comparison with the other months, except in HPS samples in October at Lido and in May and October at Jesolo. Moreover, higher values were generally observed in July: in both LP and HPS samples, measurements were significantly different in comparison with May at Lido ($p < 0.001$ and $p < 0.01$, respectively) and with October at Jesolo ($p < 0.01$ and $p < 0.001$, respectively) (Figure 25).

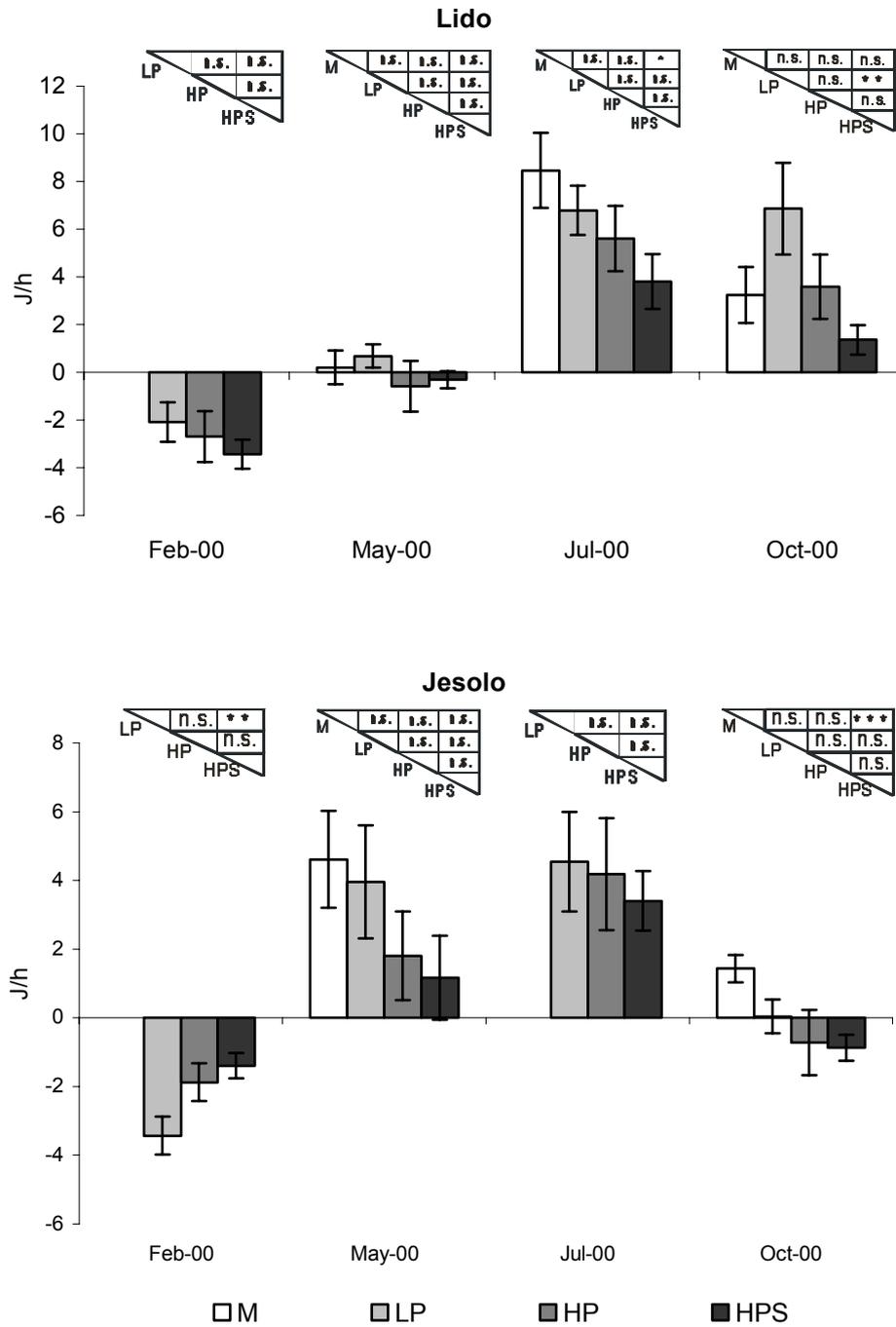


Figure 24 Scope for growth of *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.e.; Anova: * $p < 0.05$.

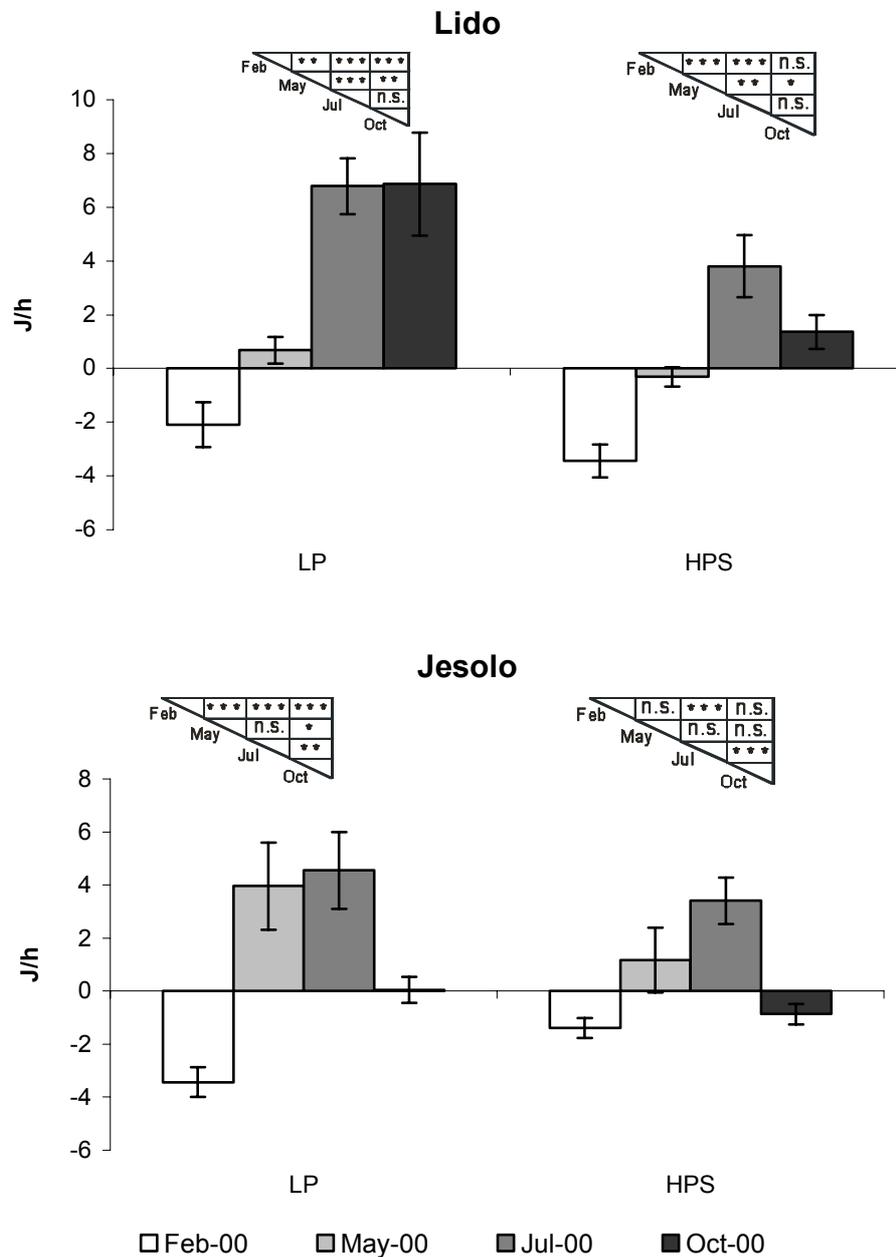
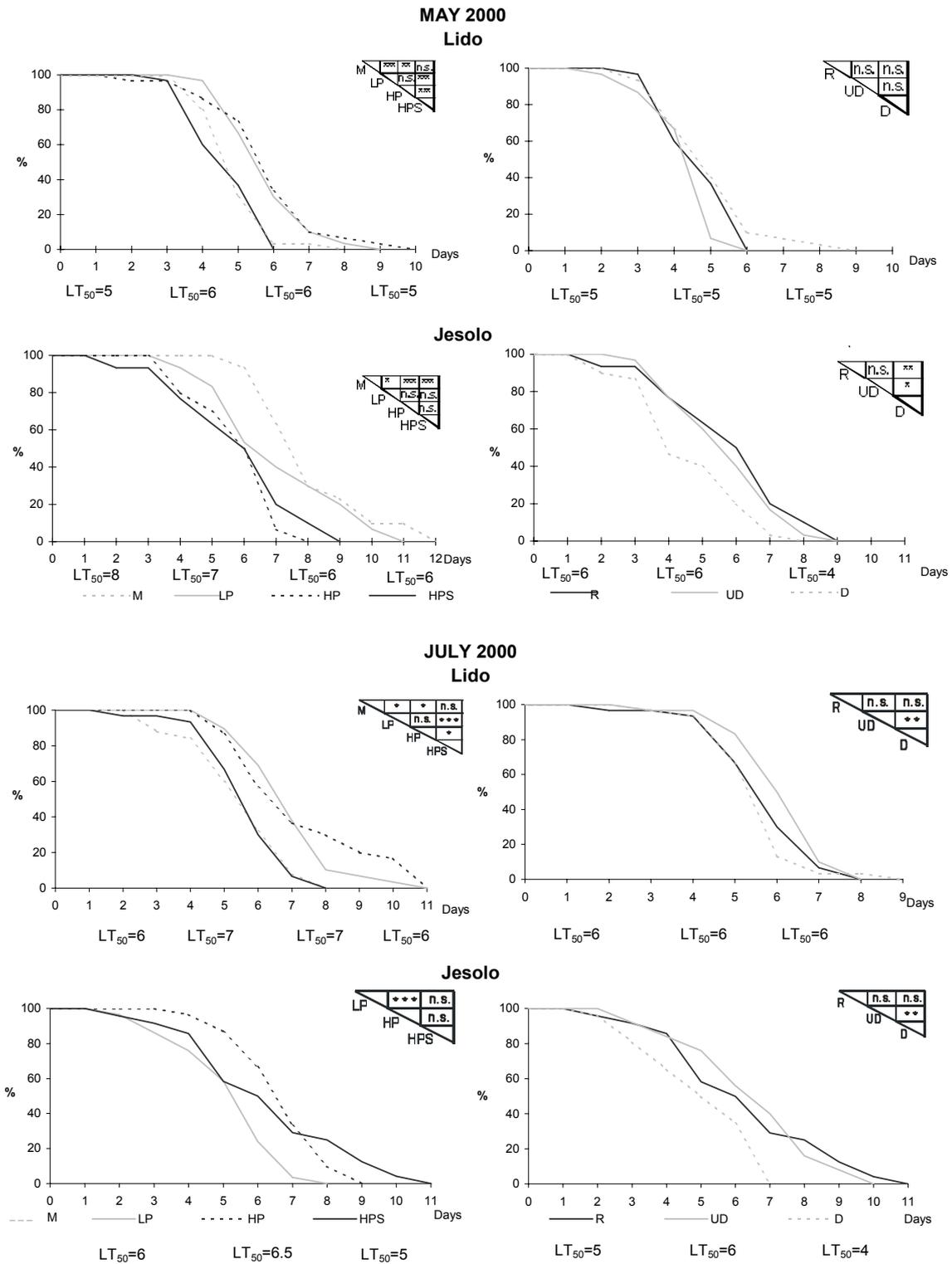


Figure 25 Seasonal comparison of scope for growth values of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido and Jesolo. Mean \pm s.e.; Anova: * $p < 0.05$; *** $p < 0.001$.



Figures 27 and 28 – Survival in air of *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). LT50 in days; N=30; Wilcoxon & Gehan: *p<0.05; **p<0.01; *p<0.001.**

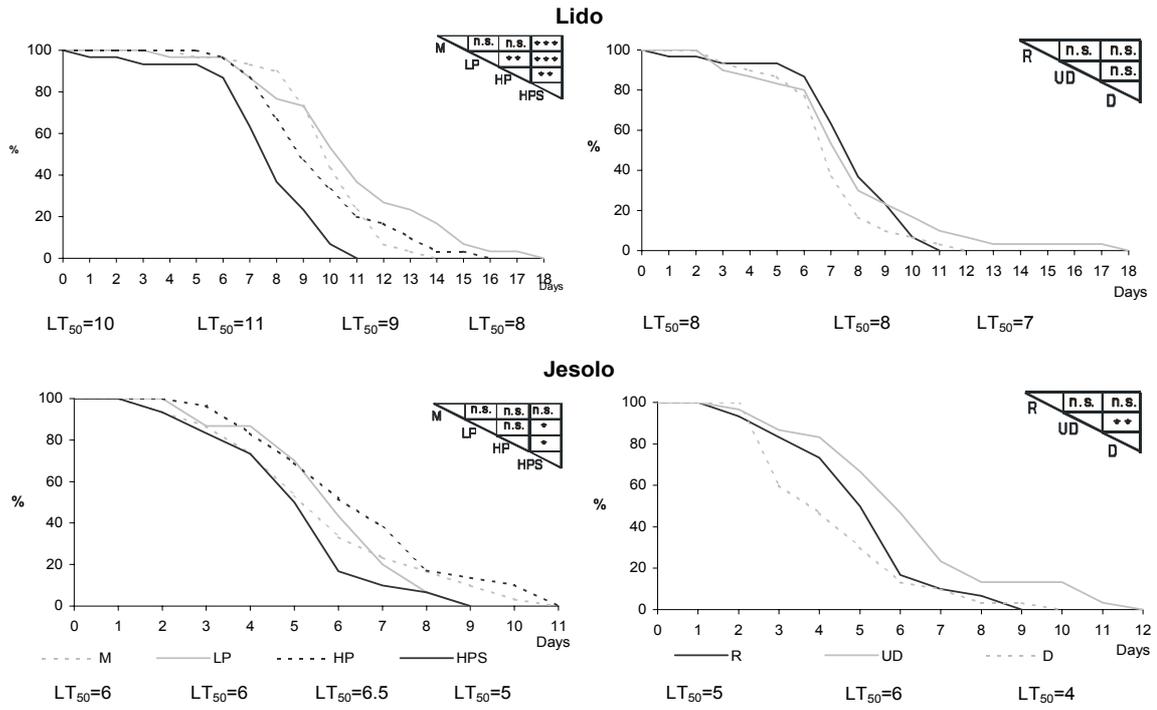


Figure 29 – Survival in air of *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). LT₅₀ in days; N=30; Wilcoxon & Gehan: *p<0.05; **p<0.01; *p<0.001.**

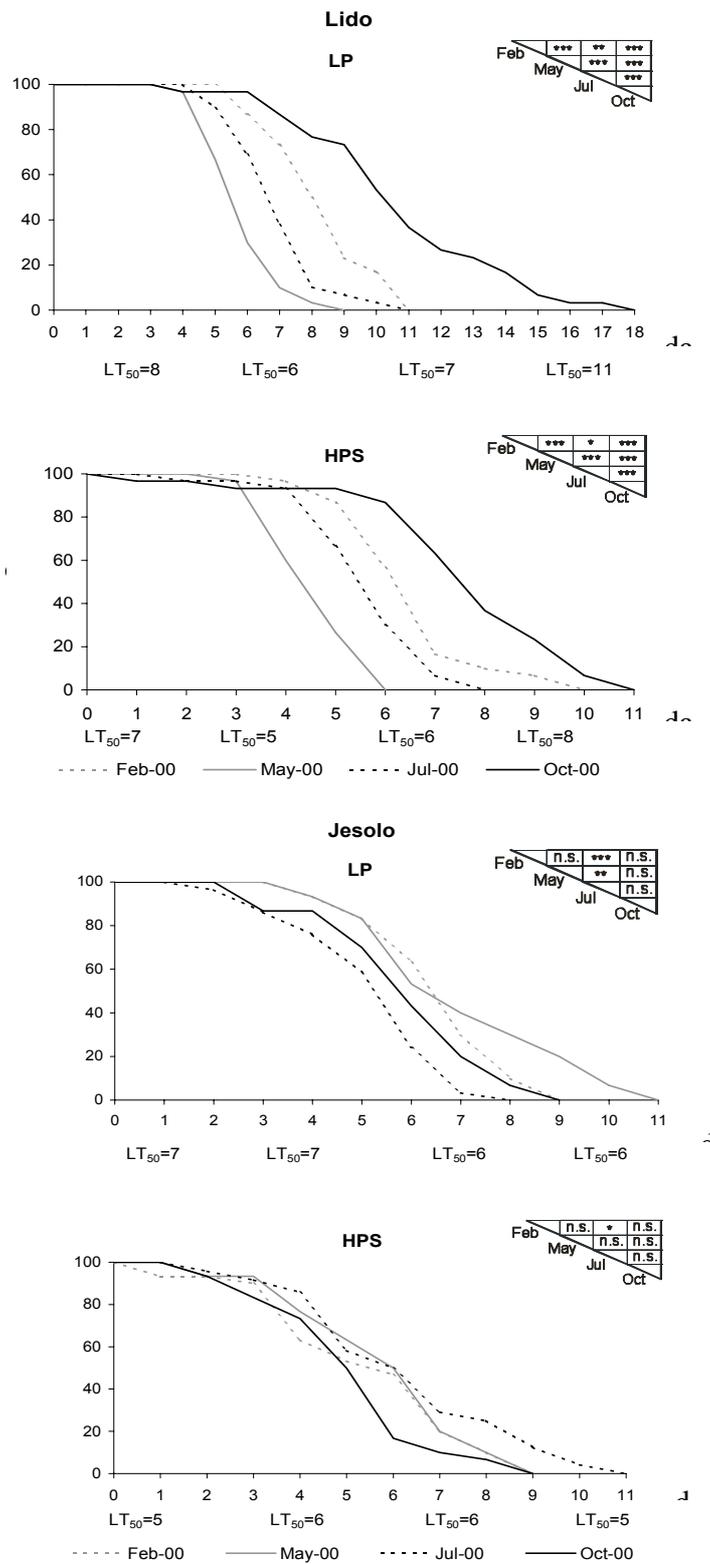


Figure 30 and 31 Seasonal comparison of survival in air of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido. LT50 in days; N=30; Wilcoxon & Gehan: *p<0.05; **p<0.01; ***p<0.001.

Condition index

In both sampling sites values observed in May significantly higher resulted with respect to the other months ($p < 0.001$). Condition index in February was always significantly higher than those measured in July and October ($p < 0.001$) (Figure 32).

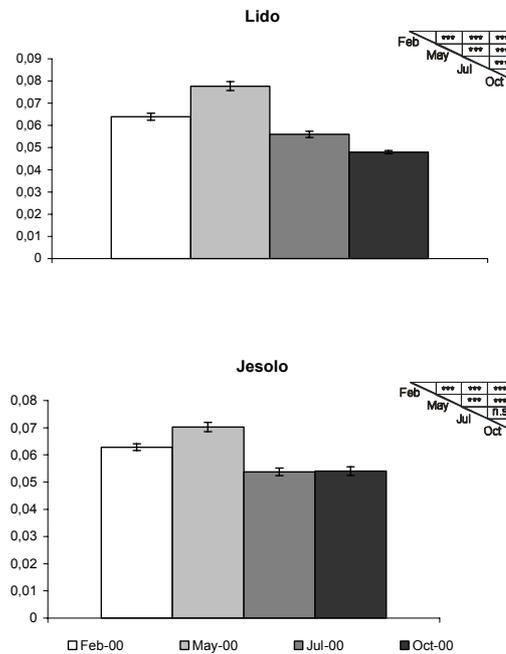


Figure 32 Seasonal comparison of condition index values of *C. gallina*. Mean \pm s.e.; Anova: *** $p < 0.001$.

RNA/DNA ratio

When comparing the various fishing methodologies, the RNA/DNA ratio of LP (Low Pressure) samples from Lido was often lower than the ones of the other samples at Lido, whereas this parameter did not reveal any particular trend in the samples collected at Jesolo (Figure 33). With regards to the seasonal variations, a maximum was reported in spring for the LP and HPS samples in both sites (Figure 34). As for the evaluation of the acid nucleic contents, the DNA was more or less constant in the samples collected with the three methodologies (Figure 35), but, when seasonally considered, it showed the highest values in summer and the lowest in winter for both sites (Figure 36). The RNA content showed variation for the three methodologies in every season, no effect of different fishing systems was detected (Figure 37). This parameter showed a clear seasonal pattern with higher values in spring and lower in autumn at Lido, a similar trend was not so evident in the samples collected at Jesolo (Figure 38).

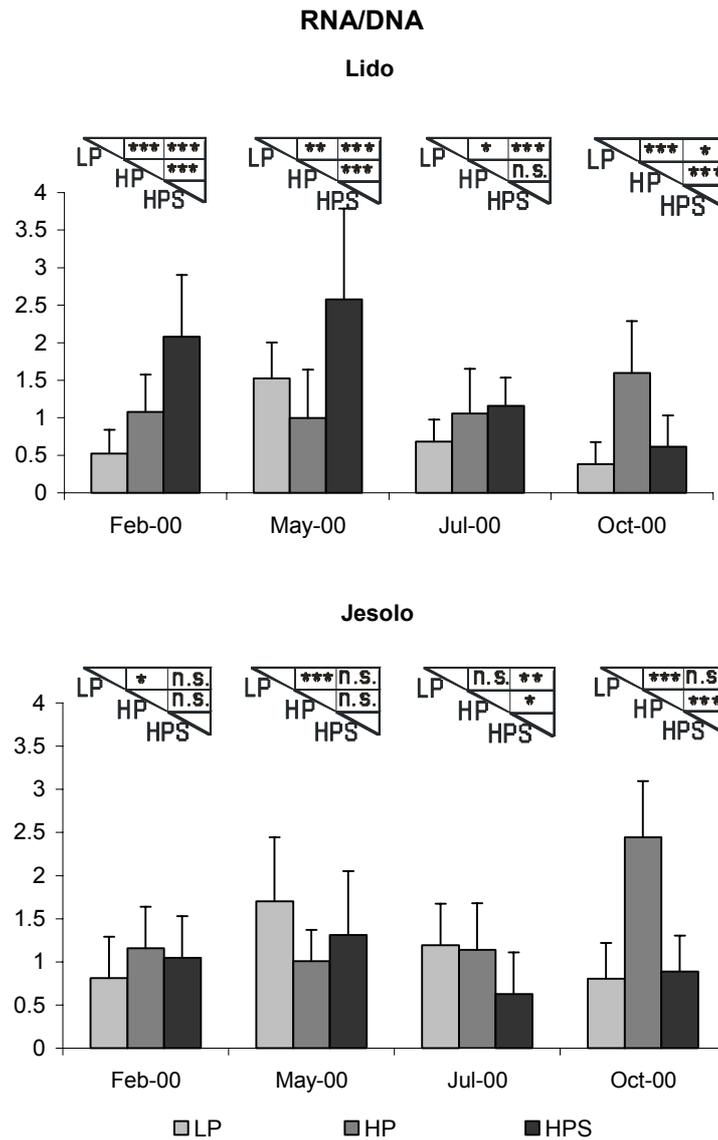


Figure 33 RNA/DNA ratio in gill of *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.d.; n=20; Statistical comparison Student t test: *p<0.05; **p<0.01; ***p<0.001.

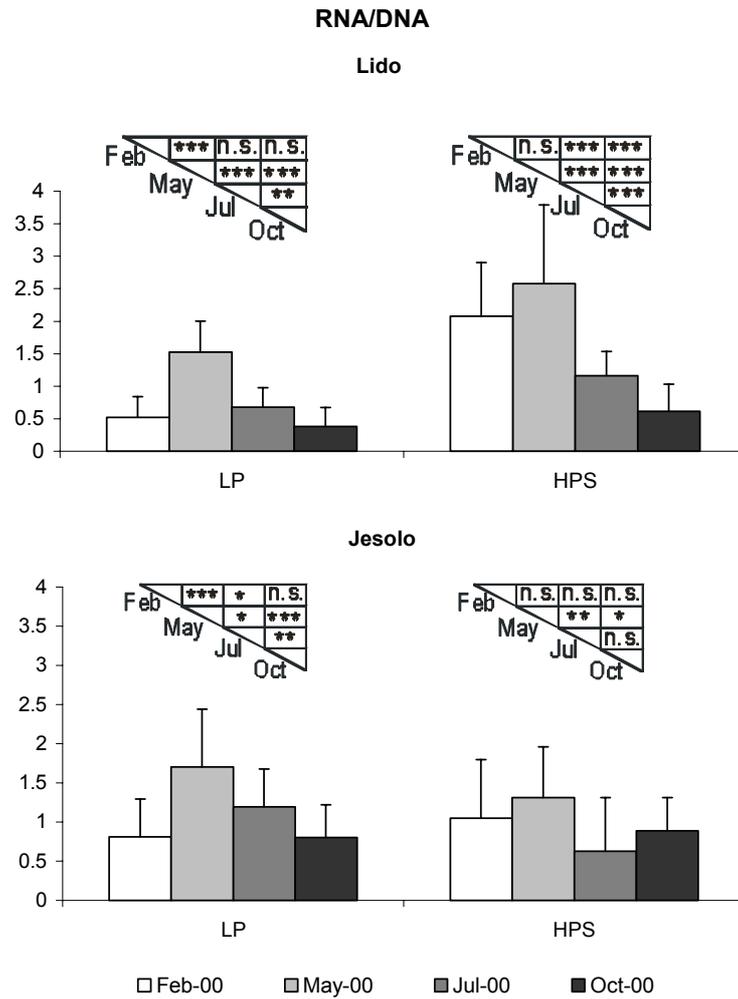


Figure 34 – Seasonal comparison of RNA/DNA ratio of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido and Jesolo. Mean \pm s.e.; Statistical comparison Student t test: * $p < 0.05$; ** $p < 0.01$; * $p < 0.001$.**

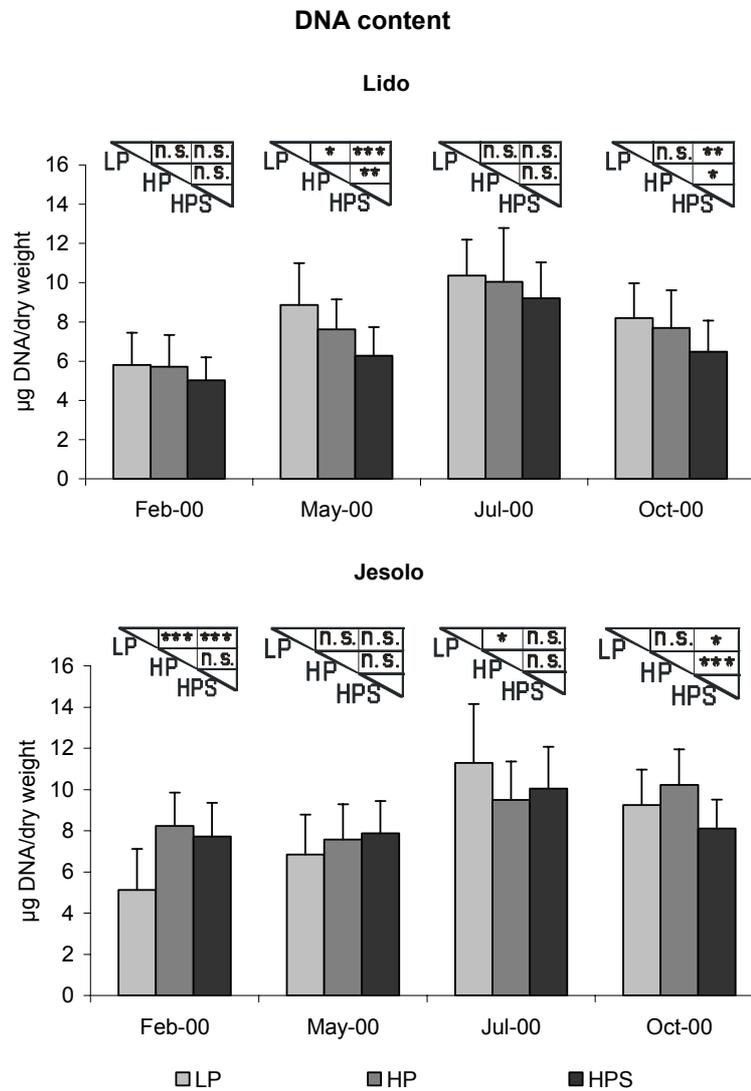


Figure 35 DNA content in gill of *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.d.; n=20; Statistical comparison Student t test: *p<0.05; **p<0.01; ***p<0.001.

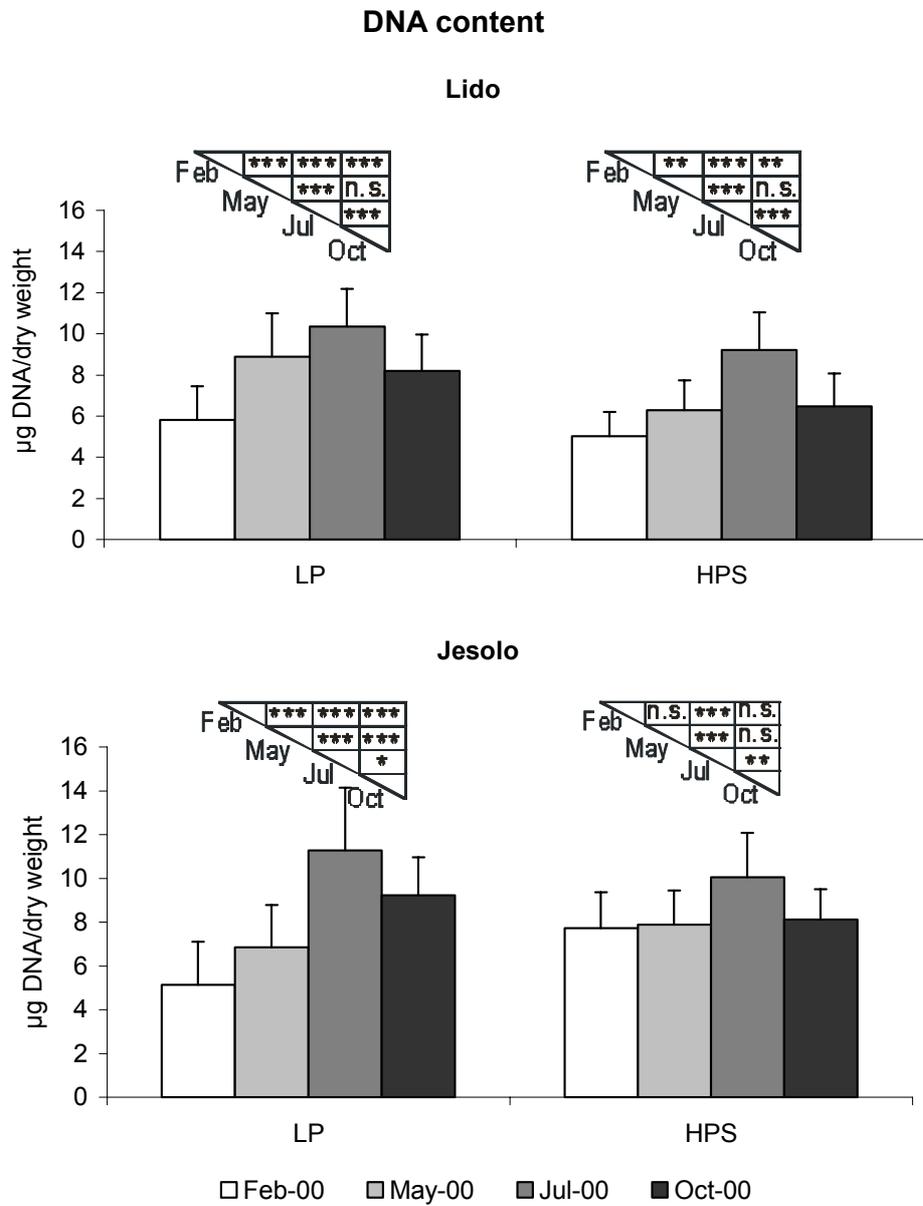


Figure 36 Seasonal comparison of DNA content of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido and Jesolo. Mean \pm s.e.; Statistical comparison Student t test: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

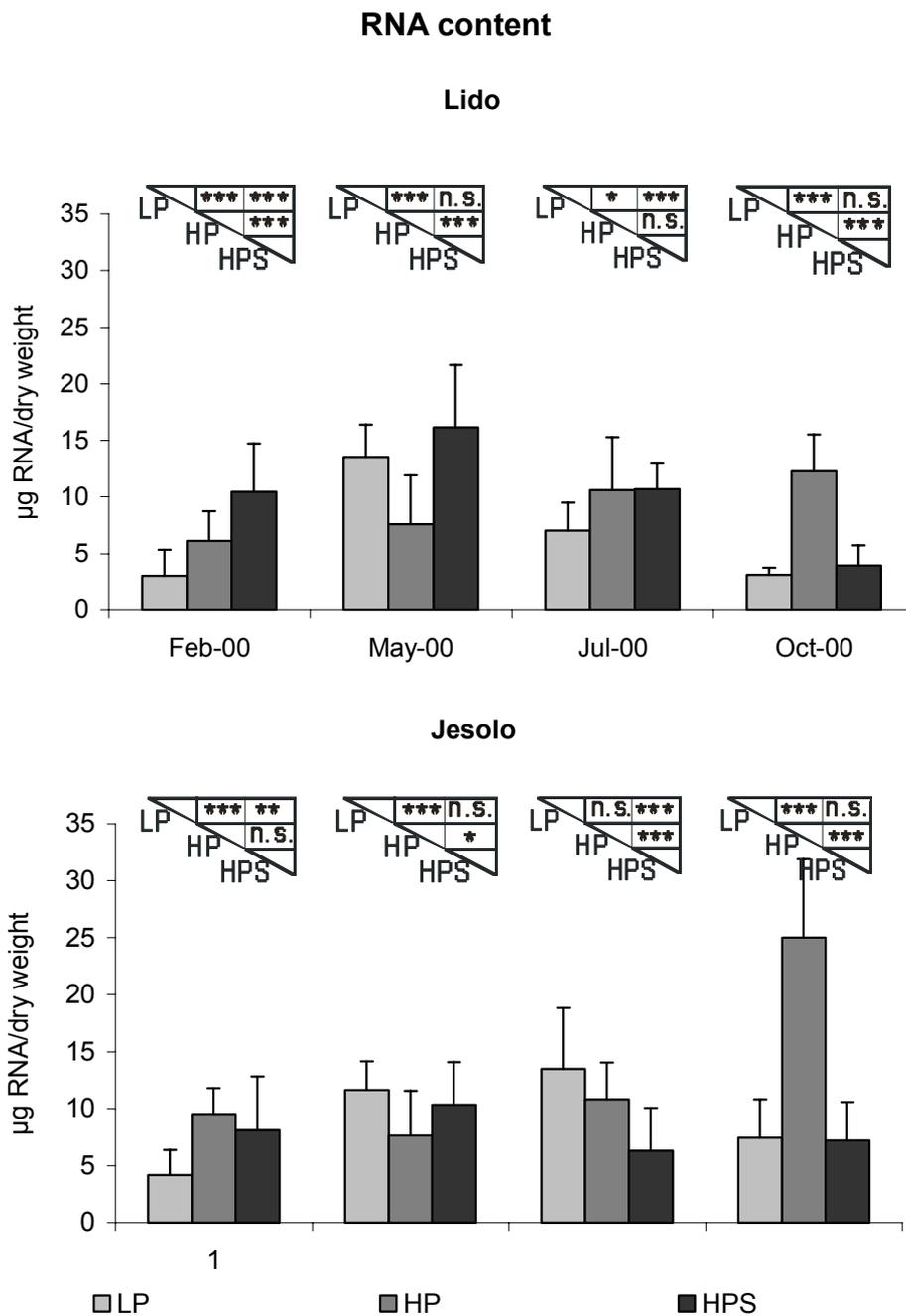


Figure 37 RNA content in gill of *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.d.; n=20; Statistical comparison Student t test: *p<0.05; **p<0.01; ***p<0.001.

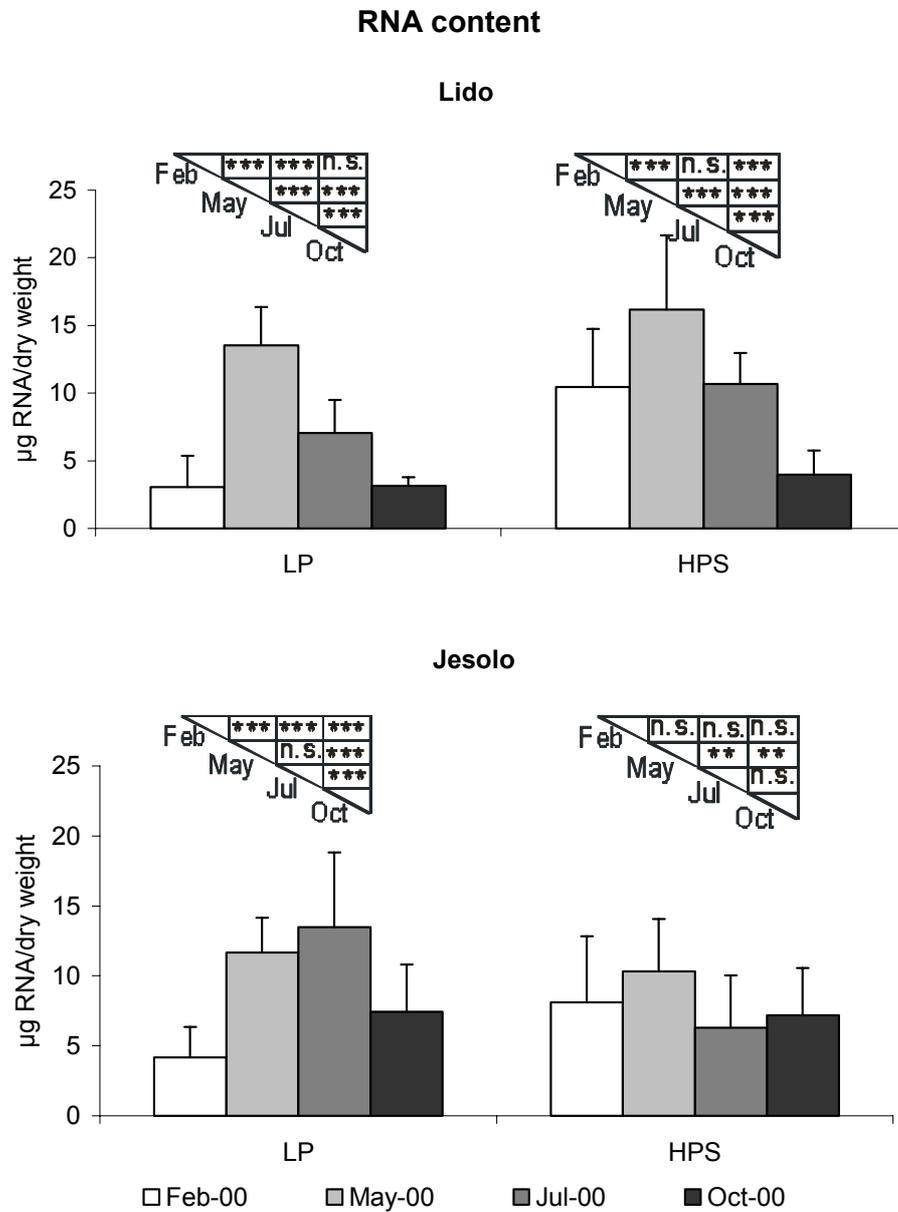


Figure 38 Seasonal comparison of RNA content of *C. gallina* dredged using low water pressure (LP) and high water pressure and mechanical sorter (HPS) at Lido and Jesolo. Mean \pm s.e.; Statistical comparison Student t test: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

Multivariate analysis

The scatterplot in Figure 39 showed a marked separation between the two studied sites (Lido and Jesolo) as well as among the dredging systems applied (mostly at Lido), according to a gradient from less to more impacting system. For both factors (e.g., site and dredging system), high values of percentage of the total variance explained were observed.

Eigenvalues

Estraction: Principal Component

	<i>Eigenval.</i>	<i>%Varian. Total</i>	<i>Cumul Eigenval.</i>	<i>Cumul %</i>
1	20.41304	58,32296	20,41304	58,32296
2	5.67800	16,22285	26,09104	74,54582

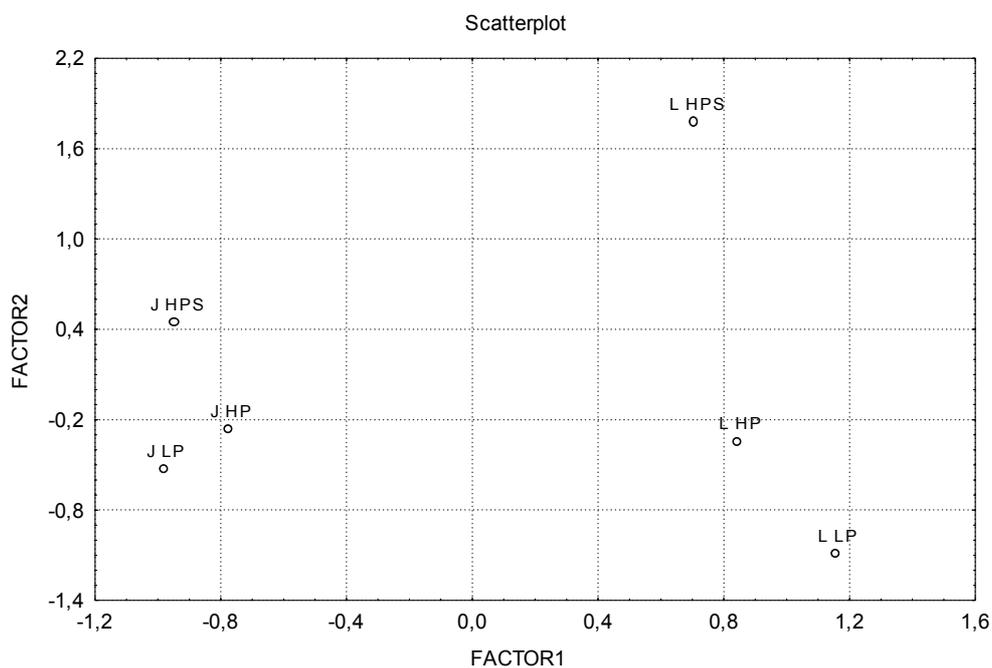


Figure 39 Samples arrangement obtained by PCA analysis applied on chronic biomarkers data.

Comparison between under-sized and commercial clams

Survival in air

Under-sized clams showed longer survival times than commercial size ones, statistically significant at Lido in July ($p < 0.05$) and at both sites in October ($p < 0.001$ at Lido and 0.05 at Jesolo) (Figures 40, 41).

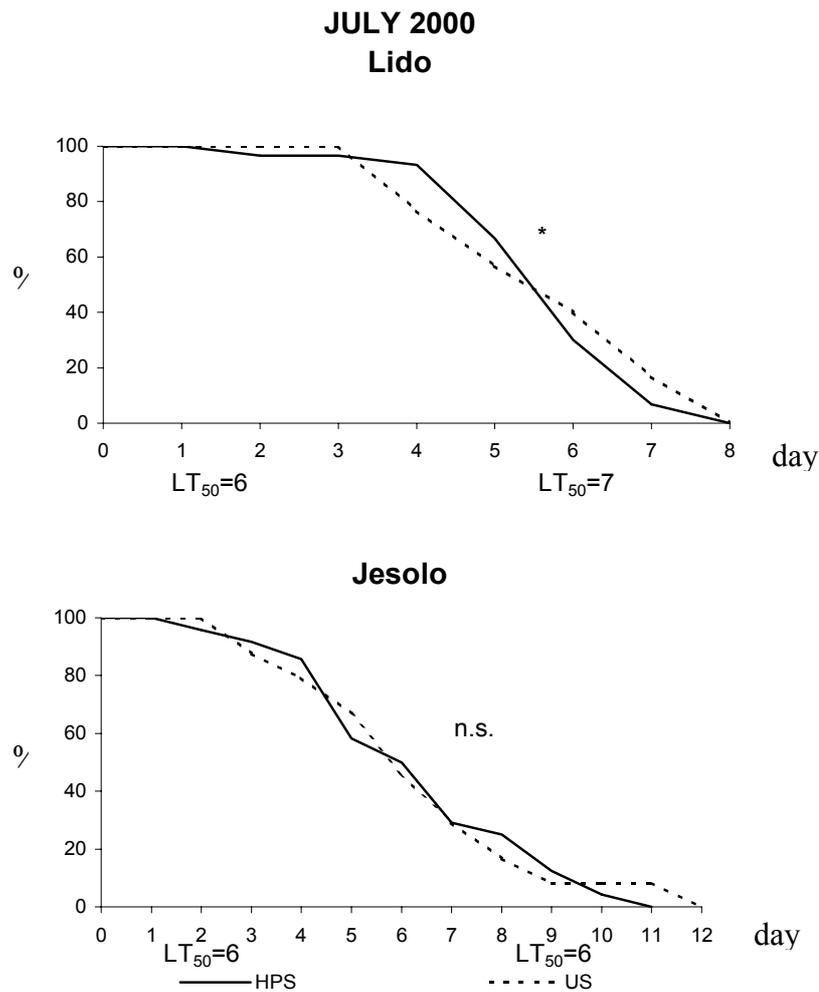


Figure 40 Survival in air on commercial (HPS) and under-sized (US) *C. gallina* collected in July using high water pressure and mechanical sorter. N=30; LT50 expressed in days; Wilcoxon & Gehan: * $p < 0.05$.

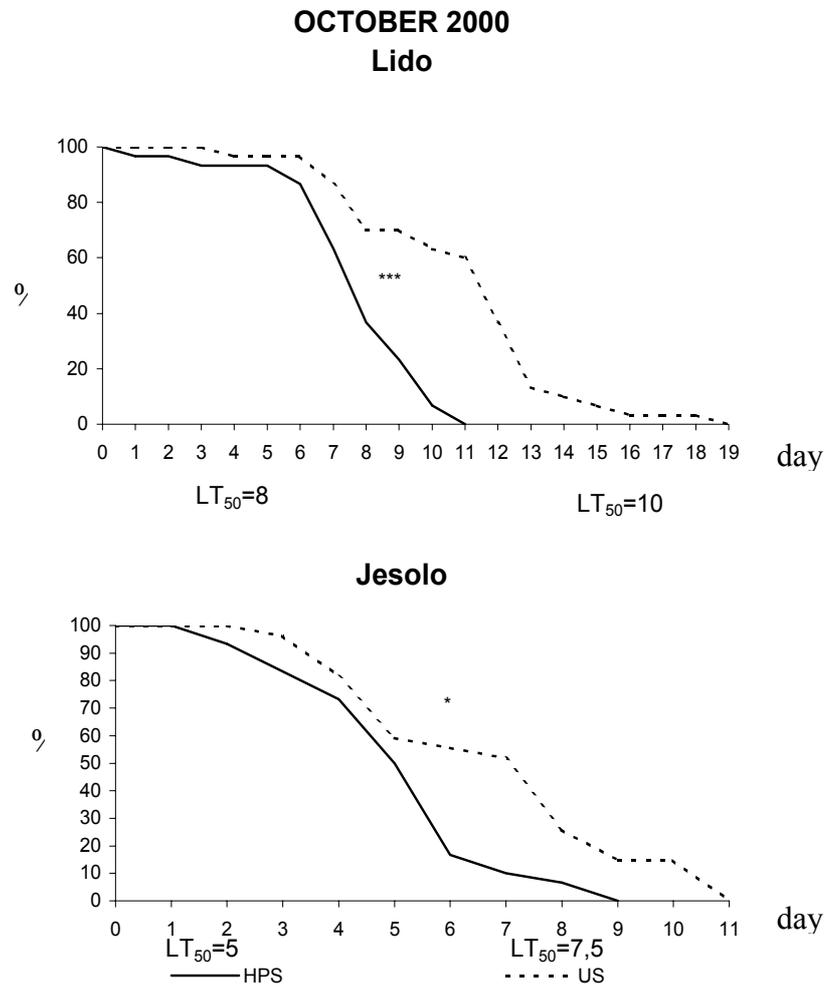


Figure 41 Survival in air on commercial (HPS) and under-sized (US) *C. gallina* collected in October using high water pressure and mechanical sorter. N=30; LT50 expressed in days; Wilcoxon & Gehan: *p<0.05; ***p<0.001.

Discussion

Laboratory study

Since the rates of most physiological processes are dependent on body size, the effects of this variable were removed from laboratory and field measurements by application of the allometric model relating physiological rates to body mass at different temperatures (Widdows 1978a; Widdows 1978b). The slopes of the regression lines for both filtration and respiration rates measured at 8, 16 and 24°C was found to be statistically different so it was possible to estimate a common regression coefficient which was used to standardised physiological rates. Moreover, filtration rate resulted more temperature-dependent than respiration: the filtering activity of *C. gallina* statistically increased at 24°C with respect to 8 and 16°C, whereas respiration rates showed no differences at the tested temperatures.

Laboratory experiments simulating mechanical stress pointed out a good responsiveness of filtration and respiration rates, both parameters decreasing when shaking was applied. Overnight refrigeration caused a flattening of values reducing the difference between control and treated clams even if the same pattern observed in non-refrigerated samples was shown. However, SFG revealed to be the less responsive measurement. Survival in air was the most sensitive biomarker in underlining stress conditions due to shaking. Among the chronic indices, it was the only one used to evaluate stress on under-sized clams, which showed longer survival times than commercial size clams (control LT₅₀ values of 14 and 10 days, respectively).

Moreover, survival times were significantly reduced in commercial clams after the first shaking treatment, while in under-sized clams a significant decrease was observed only when shaking was repeated once a day for three days. In this case, a significant ($p < 0.05$) increase of shell damage level was also detected.

Clams showed a good recovery potential, 24 hr values being similar to their controls for both physiological parameters and survival in air. The survival in air values at 12 hrs after shaking highlighted that clams needed a longer recovery period.

Among the different biochemical biomarkers applied in the laboratory experiments, only superoxide dismutase and catalase activity resulted sensitive enough to highlight acute mechanical stress on *C. gallina* and for this reason they had been evaluated also in the following field study.

As for histochemistry, the index applied in laboratory, namely the latency, showed to be sensitive to the mechanical stress imposed to the clams, although values reported for the controls both manually collected and fished were relatively low if compared with levels detected in similar bivalve species (Moore 1976)

Field study

A battery of chronic biomarkers, previously tested in laboratory experiments, were chosen with the purpose to make an indirect estimation of dredging impact by evaluating the effects of acute mechanical stress which clams experienced during fishing operations. Samples were taken at different stages in the fishing process or treated with different pressures or collected by divers controls with the aim to

distinguish stress levels associated with the main stages in the process of catching clams, namely pressure in water jets and mechanised sorting.

Some chronic indices were able to detect the effects of water pressure and sorting on clams. Only physiological rates and survival in air test seemed to be clearly influenced by the fishing methodologies: filtration and scope for growth generally showed decreasing trends as mechanical stress increased in both sampling sites, even if measurements rarely resulted in significant differences. Respiration rate was less responsive, being not distinctly affected by different levels of stress. The decrease of both filtration and scope for growth in dredged clams was able to provide a short-term response to the acute stress experienced by *C. gallina*, although these parameters are generally recognised as indicators of long-term stress (Smaal and Widdows 1994).

Survival in air of dredged animals decreased with increased stress, LT_{50} values of LP samples being higher than those of HP and HPS. Significant differences were generally observed when comparing survival curves of LP and HPS samples. However, this index rarely made possible a discrimination between LP and HP samples, based only on water pressure effect, whereas the effect of the sorter was clearly evident at Lido, as revealed by comparing HP and HPS samples. In clams fished using commercial system, a positive correlation between size and stress effects was observed: under-sized clams had longer survival times than commercial ones both at Lido and Jesolo, particularly in October. This result confirmed those obtained in laboratory experiment.

Among physiological parameters, scope for growth exhibited the highest seasonal variation: the observed pattern was similar at both Lido and Jesolo in LP as well as in HPS samples, showing the highest values in July and the lowest in February, mostly influenced by food availability and water temperature. In particular, negative SFG values in winter highlighted a general worsening of clam conditions in this period of the year, when animals are utilising their body reserves for metabolism maintenance (Bayne et al. 1985). Further information can be obtained by evaluating seasonal variations of the condition index; the highest value detected in May reflected the increased food availability that allows organisms to accumulate metabolic reserves after winter season (Beninger and Lucas 1984). The reduction of the index observed in July at both sampling sites was probably due to the employment of reserves for reproduction and spawning. Survival in air showed an opposite seasonal trend, LT_{50} values being higher in cold months and lower when water temperature increased. As suggested by Eertman et al. (1993) for mussels, higher temperature and reproductive conditions (Valli et al. 1985) lowered clams survival in the summer period, as well as the responsiveness of the survival in air test, since natural stressors, both endogenous and exogenous, came to play a predominant role (Marin et al. 2001).

Catalase, protein content and latency were not able to reveal either statistical differences or expected trends when comparing samples from the different stages of treatments dredged at any season. However all these indices showed seasonal variations. Generally, catalase activity exhibited significantly lower values in winter, reflecting low level of oxidative stress in this period of the year, whereas significant higher values of protein content were observed in spring before the reproductive period, then dramatically decreasing in summer. Similarly, latency showed higher values in May and lower value in October. These marked seasonal variations are

consistent with other studies on anti-oxidante enzymes (Viarengo et al. 1991 ; Livingstone and Nasci 2000) and on latency (Lowe and Da Ros 2000).

Like the above mentioned biochemical and histochemical biomarkers, RNA/DNA ratio and acid nucleic contents did not reveal a clear relationship with stress levels due to fishing system. According to the results of the other growth indices, namely scope for growth and condition index, the seasonal pattern, showing higher values in spring/summer, seemed to be mostly related with food availability and higher temperature values which accelerate metabolism and cause a faster cell turn over (Brett et al. 1969; Bullock et al. 1981).

Analysing, season by season, the biological responses of clams at the lowest (M or LP samples) and at the highest (HPS) level of mechanical stress experienced, greater differences were observed in July and October (see, for example, filtration rate at Lido, and SFG at both sites), thus underlining an increased sensitivity of clams to the impact of fishing in this period of the year. Our results indicate a decreased capability of clams to cope with mechanical stress during or immediately after spawning season, when endogenous stress is higher (Valli et al. 1985).

Despite the Principal Components Analysis exhibiting a clear separation between the two clam populations, the evaluation of the fishing impact through the biomarker approach in the different environmental conditions considered in this study highlighted a similar poor level of clam well-being and consequently similar stress responses in the two populations, both heavily exploited.

Finally, among the evaluated chronic indices, the highest responsiveness to acute mechanical stress due to fishing impact was showed by measurements detected at organism level and referring to a general short-term reaction. Cellular biomarkers were confirmed to be particularly suitable to underline long-term variations in well-being of clam populations.

Manually collected clams, we intended to use as controls, generally did not have the expected behaviour when biomarkers at both cellular and organism level were analysed. These results may be a consequence of the heterogeneous size of clams, as well as of their possible low well-being status when collected. Indeed, it can be hypothesised that divers more easily picked out clams on bottom surface, having these bivalves probably already suffered for stress conditions previously experienced.

Chronic indices suitable to highlight short-term stress revealed that the impact of hydraulic dredging has deleterious effects on commercial size clams and, in addition, must be considered particularly harmful even to under-sized clams which are first dredged and sieved, and then rejected. On the basis of laboratory results, we hypothesise that, once again in their habitat, which is severely modified by dredge towing, clams suffer altered physiological and behavioural rhythms, and, above all, loss of shell integrity, which makes them more vulnerable to potential predators. Since the same area in the clam ground may be dredged more than ten times per month (Pellizzato, pers. comm.), the overall stress undergone by growing clams is considerable.

3.1.3.7 INDICATORS OF THE EFFECTS OF DREDGING-INDUCED STRESS ON THE CLAM (*SPISULA SOLIDA*)

Introduction

The Portuguese bivalve fisheries industry, which today principally targets *Spisula solida*, has been commercially important since 1969, but the behavioural and physiological effects of shellfish dredging have received little attention worldwide (Maguire et al. 1999a; Maguire et al. 1999b; Maguire et al. 1999c;). Bivalves are caught with a dredge that can penetrate the sediment to a depth of 50 cm, depending on target species and sediment type. Despite the high efficiency of the Portuguese clam dredge (Gaspar 1996a), clams not captured may also die as a consequence of fishing. In Portugal, bivalves are subject to successive habitat disturbance by dredging. Undersized bivalves (those that can pass through the mesh of the dredge), which in the case of *S. solida* are individuals of less than 25 mm in length, are especially affected. The passage of fishing gear across the seabed leads to both direct and indirect mortality through subsequent predation (Kaiser and Spencer 1995). The extent of this additional mortality must be quantified if the total mortality associated with fishing is to be calculated (Kaiser and Spencer 1995). Murawski and Serchuk (1989) observed variable survival rates among “undersized” clams returned to the seabed, which provide potential food in the dredge track for scavengers and predators (Chícharo et al. 2002.).

The survival of non-damaged individuals may be related to the time they need to rebury. This time may be influenced by stress resulting from the passage of the dredge. The anthropogenic effects of dredge fishing are known to affect various physiological/biochemical processes associated with organism metabolism (Maltby 1999). Knowledge of organism-level responses to dredge-induced stress is essential for understanding its adverse effects and the strategies adopted by organisms to tolerate such stress.

Indices of acute and chronic physiological conditions can be used to analyse both mechanical stress and stress induced by aerial exposure of bivalves before their return to the sea bed. The reburying time of a clam can be used as a behavioural index of acute stress in natural environments. Burrowing behaviour is adaptive and allows clams to escape predation (Phelps et al. 1983). Therefore, if stress conditions affect burrowing behaviour, this could be reflected in increased mortality.

Maguire et al. (1999b) examined the usefulness of various techniques for stress assessment in scallops and found Adenylic Energetic Charge (AEC) and behaviour to be the most effective for the measurement of acute stress. AEC was first proposed as a stress index by Atkinson (1968), who suggested that modulations in the levels of adenylyl-phosphate reflected variations of enzyme activity at essential points in the metabolic pathways that yield high energy adenine-phosphate bonds. Such variations result from external stress.

The more stressed an animal becomes, the more energy it uses to counteract the stress, thus lowering its AEC level. AEC is defined by the ratio: $AEC = (ATP + 0.5ADP) \div (ATP + ADP + AMP)$, where: ATP = adenosine tri-phosphate, ADP = adenosine di-phosphate and AMP = adenosine mono-phosphate. The tri-phosphate bond of the

ATP molecule stores the most energy, the di-phosphate bond of ADP is half as energy-rich and the mono-phosphate bond (AMP) lacks useable energy. The AEC is a linear measure of the ratio of ATP concentration to total adenylate concentration, ranging in value from 0 to 1 in the fully charged state. An AEC ratio of 0 indicates that all nucleotides are AMP while a ratio of 1 signifies that all nucleotides are ATP. Therefore, the level of these bonds can be used as a measure of the energy directly available to the cells at that time. Another adenylic-derived index, ATP/mg dry weight, is also a very useful indicator of acute stress and has been used in other studies (Skjoldal 1981 ; Isani et al. 1995).

Nevertheless, not only acute stress should be analysed when studying the environmental effect of dredging. In ecological studies, analysis of seasonal changes is also of importance and this will only be achieved by studying the chronic indicators of stress. A variety of abiotic and biotic variables, such as spawning and recruitment periods, predation, temperature, and food availability, may account for temporal oscillations in benthic animal populations.

These seasonal changes, reflected in bivalve physiology, can be measured through chronic indices of their biochemical condition. Such indices can reflect overall environmental conditions to which the bivalves are subject. The RNA/DNA ratio is an eco-physiological index of activity (growth, reproduction, secretion, etc.) under a given environmental condition (Lucas and Beninger 1985). Determination of the RNA/DNA ratio has been used on a wide range of marine organisms, principally fish (Bulow 1970; Buckley 1984; Robinson and Ware 1988), crustaceans (Anger and Hirche 1990) and bivalves (Grémare and Vétion 1994; Chícharo and Chícharo 1995; Chícharo et al. 2002.). This index is based on the assumption that the amount of deoxyribonucleic acid (DNA), the primary carrier of genetic information, is stable under changing environmental situations within the somatic cells of a species (Clemmesen 1994), whereas the amount of ribonucleic acid (RNA) is known to vary with age, life-stage, organism size, disease-state and with changing environmental conditions (Bulow 1970). RNA is an obligate precursor in protein biosynthesis. Thus, the RNA/DNA ratio provides a self-calibrating index of protein synthesis. Bivalves in good condition tend to have higher RNA/DNA ratios than those in poorer condition (e.g. Chícharo and Chícharo 1995).

Another indicator of physiological condition is the energy storage index, the Neutral/Polar (N/P) lipids ratio proposed by Hentschel (1998). According to this author, quantifying neutral lipids (triglycerides) indicates the degree to which energy gain exceeds energy demand, while polar lipids (cholesterol and phospholipids), having a structural function in cell membranes, indicate body size and are less variable.

Therefore, these physiological indices can be useful in accurately assessing dredge fishery impacts in that they may allow the detection of variations in the condition of individuals more rapidly than the standard method of analysing community change using diversity indices (Kaiser and Spencer 1995). The indices also allow for estimation of survival rates, indirect mortality and comparison of different areas.

The aims of this work were to determine: 1) minimum index levels of biochemical stress in bivalves; 2) changes in different biochemical condition indices (acute:

adenylic-derived indices; chronic: RNA/DNA and N/P lipids ratios) of undersized (<25 mm) *Spisula solida* in response to cumulative stress imposed in a laboratory simulation of dredging activity; 3) changes in reburying time and adenylic-derived indices of *S. solida* in response to dredging stress in situ; and 4) seasonal changes in the condition of *S. solida* before and after dredging, according to RNA/DNA and N/P lipids ratios.

Methods

Study area

The Algarve coast of Portugal extends from Cabo São Vicente in the west (8° 59'W), to the border with Spain in the east (7° 24'W). Currents on the continental shelf of the Algarve are usually weak, with wind-caused drift currents that run along the shore from east to west, to a depth of 30 m or more, predominating over tidal currents. Normal current speeds are under 0.25 m/s, but may reach 0.5 m/s or more during extreme conditions (Fiúza, 1983). Water temperatures range between 14°C and 22°C. The study site was located off Vilamoura on a sandy fishing ground at a depth of 8 m (Figure 1).

Fishing gear used

A bivalve dredge of the type commonly used on the south coast of the Algarve was employed in this study. It is a large, heavy, iron structure, with a 25 mm mesh net bag and a metal semicircular parallel bar grid and toothed lower bar at the mouth. The mouth is 43 cm wide, the bars are 2.3 cm apart, and the teeth are 34 cm long, acting as a rake when the dredge is dragged through the sediment (Figure 2).

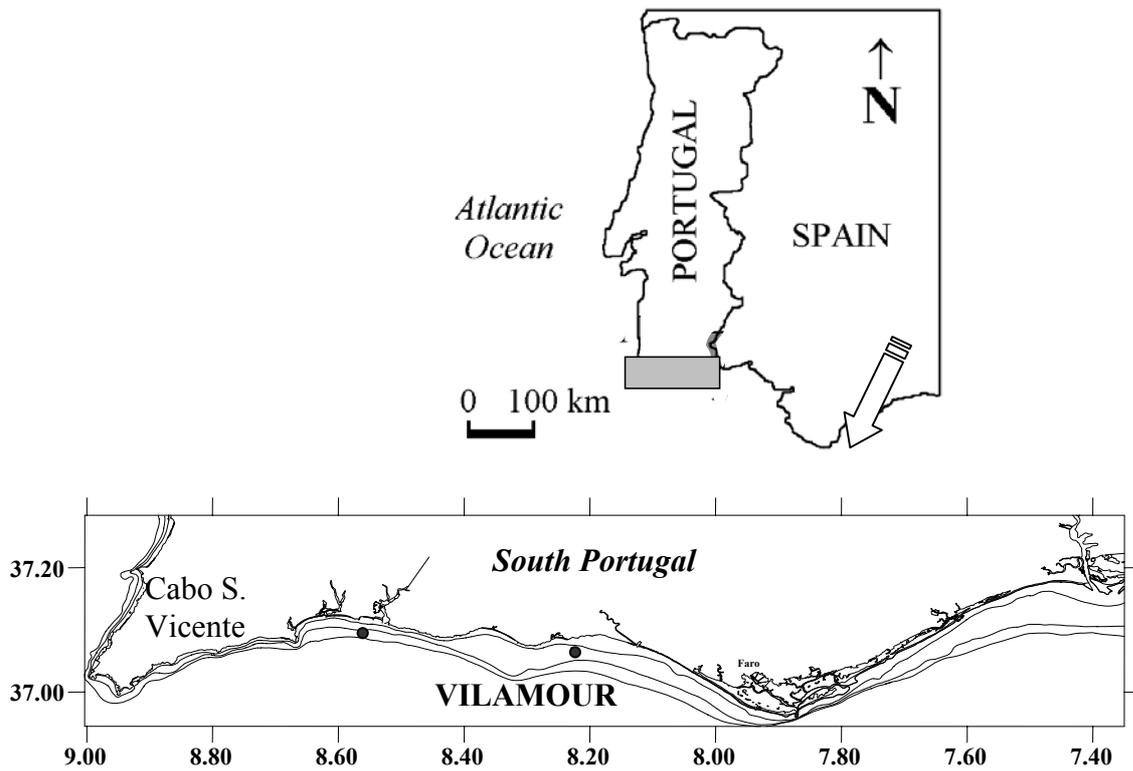


Figure 1 South Portugal (Algarve) coast map with indication of the sampling locations: fished area – Vilamoura;

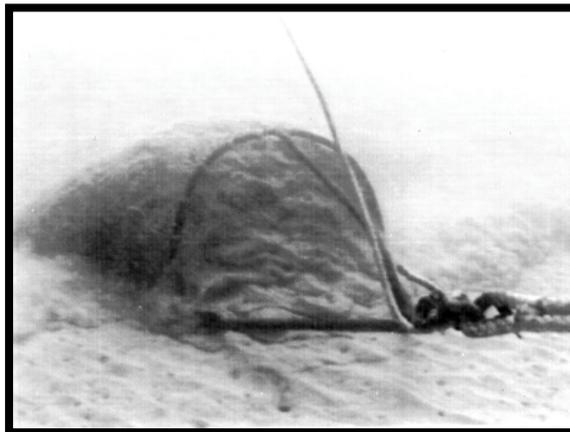


Figure 2 Photograph of the dredge showing the gear during towing.

Laboratory experiments

Minimum level of biochemical indices

To determine the minimum level on the biochemical indices (the critical point where survival of the bivalve is threatened directly or indirectly, a point not described previously), the clams were submitted to high stress levels by starvation over two days. The level of biochemical indices achieved after this period were then considered the minimum level required for survival in the subsequent experiment. Fifteen individuals of *Spisula solida* (<25 mm in length), kept in oxygenated, filtered seawater, were used in this experiment.

Cumulative stress experiment

A simulation of dredging stress on undersized bivalves, those able to pass through the dredge mesh, was developed under laboratory conditions. One hundred and twenty *Spisula solida* less than 25 mm in length were used. The bivalves were maintained for one day in oxygenated, filtered seawater and fed with the microalgae *Isochrysis* sp (4.27×10^6 cel/ml) before experimentation. To simulate dredging stress, the bivalves were subjected to 3 minutes of shaking every 30 minutes for a six hour period. After each shaking, 10 bivalves were removed, measured, weighed, and the foot divided into thirds. Each third was put in an eppendorf tube and frozen in liquid nitrogen. Later each third was analysed for the N/P lipids and RNA/DNA ratios (as a measure of chronic stress), and ATP level (measuring acute stress).

Field procedures

Aerial exposure of bivalve after dredging

Acute stress was analysed using experiments conducted between 28 and 30 September 1999 at Vilamoura fishing ground, on bivalves captured by the dredge and exposed to the air over differing periods. The stressor examined here is simultaneous anoxia, desiccation and temperature-induced stress. Bivalves were subjected to increasing levels of air exposure of 0, 30, 60, 120 and 360 minutes on the deck of the research vessel “Donax”, from IPIMAR (mean air temperature- 23 °C, direct sunlight (no clouds) and wind velocity 2-3 m.s⁻¹). The three indicators used to assess the effects of this acute stress on *Spisula solida* were AEC, $\mu\text{mol ATP/mg}$ dry weight and

reburying time. Fifteen individuals were assessed for each indicator and exposure period. Individuals to be analysed using the adenylic indices were immediately immersed in liquid nitrogen after the various exposure periods. Analyses of reburying times after different periods of aerial exposure were accomplished on board the research vessel. The bivalves were placed in a box containing seawater and local sea bottom sediment. The number of reburied individuals was recorded every minute over a 20 minute period. The reburying time of 15 unstressed *Spisula solida* (never exposed to the air) was recorded in situ using SCUBA divers.

Seasonal study

For the seasonal study, bivalves were carefully collected in situ using SCUBA divers, from an area which had not been dredged and from three separate dredge tracks generated from normal fishing procedures. Surveys were conducted at Vilamoura in spring (April) 1999 and summer (July) 1999. For each treatment, we collected 60 bivalves (15 per treatment-before and after fishing, in spring and summer). After collection, all individuals were immediately frozen and stored in liquid nitrogen, for later processing. In the laboratory, the foot of each bivalve was sectioned, and the dry weight determined after lyophilization. Samples were further processed for determination of RNA/DNA and N/P lipids ratios.

Biochemical procedures

Nucleic acids

Nucleic acids were extracted and purified from bivalve tissue homogenates and fluorescence-photometric measurements were made using ethidium bromide (EB), a specific nucleic acids fluorochrome dye (Chícharo et al. 2001). The fluorescence was determined by exciting at 365 nm and reading at 590 nm with a spectrofluorometer (Hitachi model 650-10). RNA fluorescence was calculated as the RNA+DNA fluorescence minus DNA fluorescence after RNase Treatment.

Lipids

Lipid extraction involved fluorescence-photometric measurements using Nile red (RD), a specific lipids fluorochrome dye (Hentschel 1998). Both neutral and polar lipids can be quantified simultaneously via spectrofluorometry of the same stained sample: neutral lipids – excitation 488 nm, emission 560 nm; polar lipids – excitation 549, emission 628 nm.

Adenylic-derived indices: AEC and µg ATP/mg dry weight

AEC and µg ATP/mg dry weight determinations were made through luminometry, using a Luminometer – Turner TD 20/20 according to the technique of Skjoldal and Bakke (1978). ATP was determined through measurement of light emission after adding luciferase. The light reaching the luminometer's photomultiplier tube is proportional to the amount of ATP present in the sample. ADP and AMP levels were measured indirectly after enzymatic conversion to ATP, using pyruvate kinase and myokinase. ATP levels were expressed as µmol ATP/mg dry weight. However, since AEC is conventionally measured using High Performance Liquid Chromatography (HPLC), we also compared this technique with luminometry. This experimental technique was partly developed in the Aquaculture Development Centre of the National University of Ireland (Cork) using oysters (*Crassostrea* sp). Striated

adductor muscle tissue was analysed for AEC using the technique of Moal et al. (1989a). Similar portions of *Crassostrea* tissue were also frozen in liquid nitrogen and transported to the University of the Algarve for luminometrical analysis. The ANOVA of the results revealed no significant differences between the two adenylic energetic charge methodologies ($p=0.57$) (Figure 3). A positive correlation was also revealed using linear regression ($R = 0.6$; $R^2 = 0.4$; $n = 15$) (Moal et al. 1989a) (Figure 4). However, as HPLC requires the use of expensive equipment and is more time-consuming, we chose to use the luminometry technique.

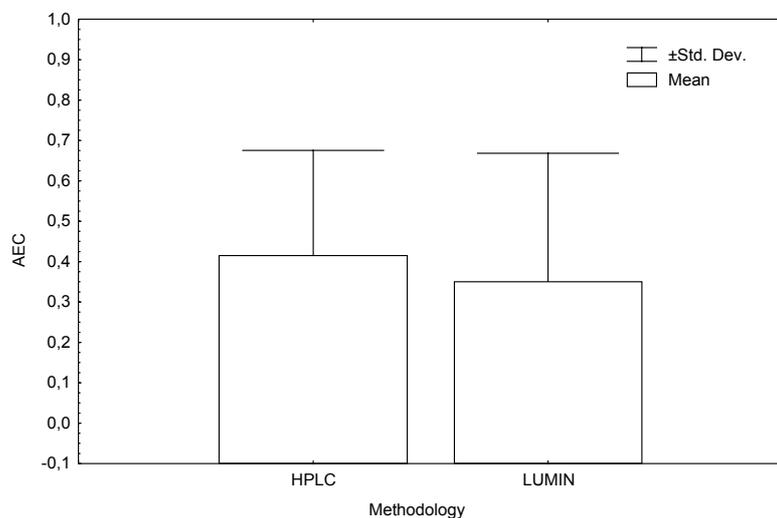


Figure 3 Comparison of two methodologies for adenylic energetic charge (AEC) determination on striated muscle of *Crassostrea* sp. ANOVA analysis between HPLC and luminometry techniques revealed no significant differences ($p=0.57$).

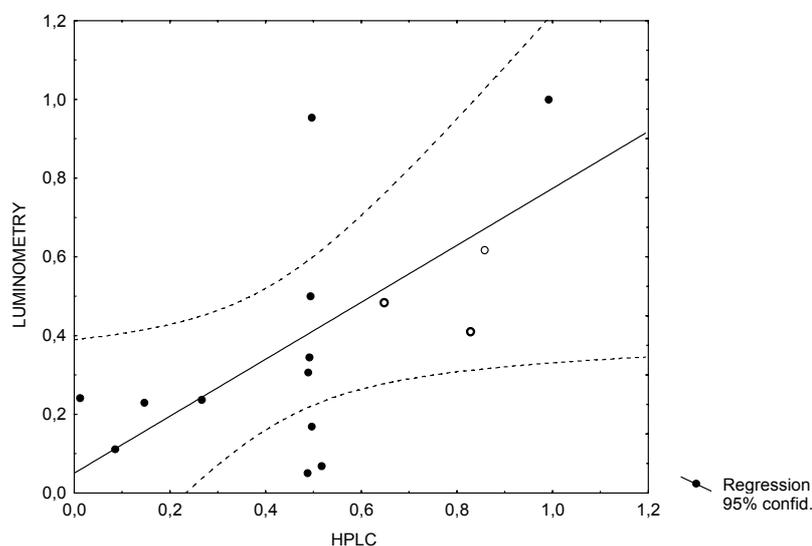


Figure 4 Linear regression between HPLC and luminometry techniques ($R = 0.6$; $R^2 = 0.4$; $n = 15$).

Data analysis

The homogeneity of variances and normality of data were tested using the Levene’s and Chi-Square tests, respectively. When the ANOVA assumptions were followed, we applied a one-way ANOVA to analyse whether the values were significantly different. When ANOVA assumptions were not followed, the non-parametric Kruskal-Wallis test was used. Where significant differences were found using the ANOVA, a Tukey test (HSD) was employed. In the experiments into reburial time, the cumulative frequency percentage of the individuals that reburied within 20 minutes was determined. A non-parametric Kruskal-Wallis test was applied to compare reburied individuals after 20 minutes, within differing aerial exposure periods, using the presence/absence transformation among the 15 individuals subjected to each aerial exposure period. All statistical analyses were made with the software package STATISTICA V.5.

Results

Minimum level of biochemical indices

The minimum level of neutral/polar lipids achieved after two days of starvation was 0.8 (Figure 5). The proposed critical level for $\mu\text{mol ATP /mg dry weight}$ is 0.2.

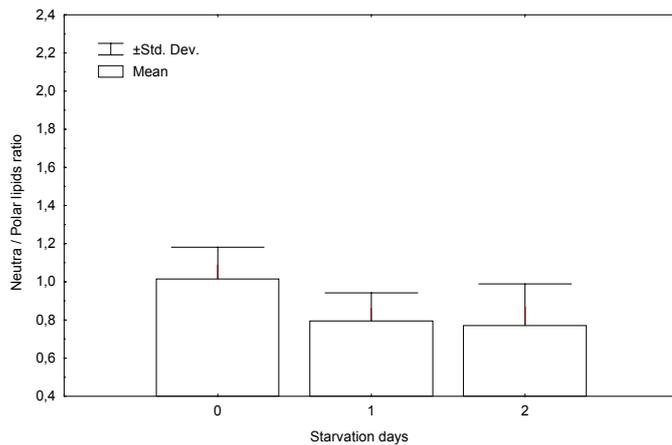


Figure 5 Variation of Neutral/Polar lipid ratios during starvation experience for *Spisula solida*.

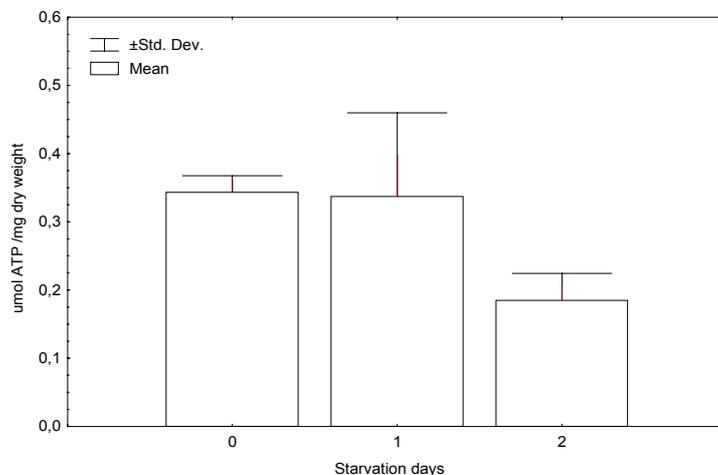


Figure 6 Variation of $\mu\text{mol ATP /mg dry weight}$ during starvation experience for *Spisula solida*.

Other minimum levels of physiological indices used to assess stress were based on the literature as described in Table 1.

Table 1 Minimum level of different biochemical indices to guarantee survival

Physiological index	Critical level	Species	Reference
RNA/DNA ratio	1	<i>Ruditapes decussatus</i>	Chícharo and Chícharo (1995)
Neutral /Polar lipids ratio	0.8	<i>Pseudopolydora kempii japonica</i>	Hentschel (1998)
μmol ATP/mg dry weight ratio	0.2	<i>Spisula solida</i>	Present study
AEC	0.3–0.5	<i>Spisula solida</i>	Present study
		<i>Pecten maximus</i>	Maguire et al. (1999b)

Cumulative stress experiment

With regard to the RNA/DNA ratio of *Spisula solida* subjected to cumulative stress, the Kruskal-Wallis test revealed significant differences (p=0.03) over time, with a decrease in the condition of the bivalves during the course of the experiment. However, minimum ratios, critical for survival, were not achieved during this experiment (Figure 7).

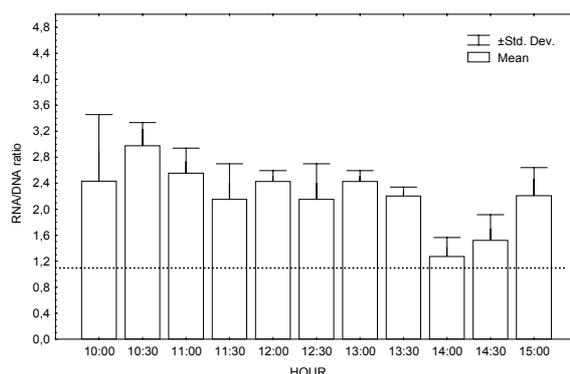


Figure 7 Comparison of the mean and standard deviation of RNA/DNA ratios between periods of cumulative stress (hours) on *Spisula solida*. Kruskal-Wallis: p=0.03. Note: the dotted line represents the critical level.

The N/P lipids ratios for *Spisula solida*, an indicator of the effects of chronic stress, exhibited different results from the above test for acute stress (Figure 8). In general, the N/P lipids ratios decreased with increasing cumulative stress. A one-way ANOVA test revealed these differences to be significant (F(11.48)=9.62; p<0.001). The corresponding Tukey (HSD) test revealed that the test at 1400 hours significantly differed from the test at other hours, as did the 13:00 hour test from the 13:30 test, 15:00 from 12:30 and 13:30, and 15:30 from 12:30 and 13:30 (Figure 8). However, the minimum ratio for survival was not achieved during this experiment.

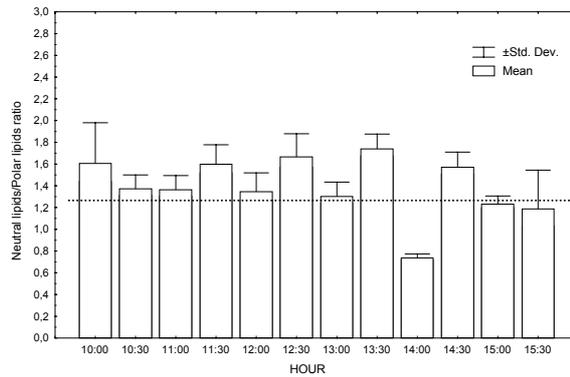


Figure 8 Comparison of the mean and standard deviation of Neutral/Polar lipid ratios between *Spisula solida* exposed to differing levels of cumulative stress (hours). $F(11.48)=9.62$, $p<0.001$. Note: the dotted line represents the critical ratio for survival.

We observed higher values for the AEC index for *Spisula solida* at 1400 hours (Figure 9). The one-way ANOVA test revealed significant differences ($F(10.40)=3.13$; $p=0.005$), while the Tukey (HSD) for an unequal number of samples revealed that the observed differences were only between the 1400 hours data and the 1130 to 1300 hours data. Lower values than the minimum were achieved especially between 1200 and 1300 hours.

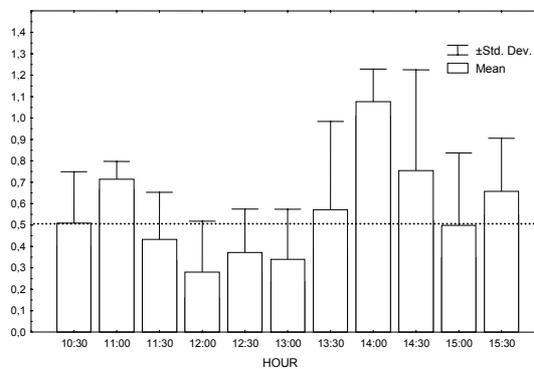


Figure 9 Comparison of the mean and standard deviation of AEC on *Spisula solida* groups exposed to cumulative stress (hours). $F(10.40)=3.13$; $p=0.005$. Note: the dotted line represents the critical level required for survival.

Values of the $\mu\text{mol ATP/mg dry weight}$ ratio, a physiological test of the effects of acute stress, were highest at 1230 hours (Figure 10). Only at the end of the experiment did values drop below the minimum, critical level. Test by one-way ANOVA revealed no significant differences ($F(6.19)=1.14$; $p=0.376$).

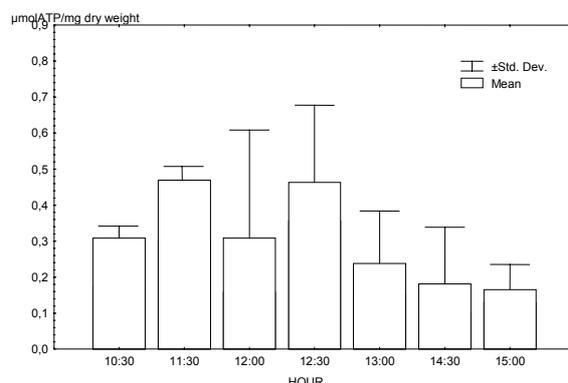


Figure 10 Comparison of the mean and standard deviation of the $\mu\text{mol ATP/mg}$ dry weight ratio between *Spisula solida* exposed to varying degrees of cumulative stress (hours). $F(6.19)=1.14$; $p=0.376$. Note: the dotted line represents the critical level for survival.

To ensure that the values of the adenylic-derived indices lower than the critical level were not skewed by the size of the small bivalves, a regression analysis between the indices and the total length of the bivalves was performed (Figures 11 and 12). However, an independent relationship was only achieved with the ATP/mg dry weight index.

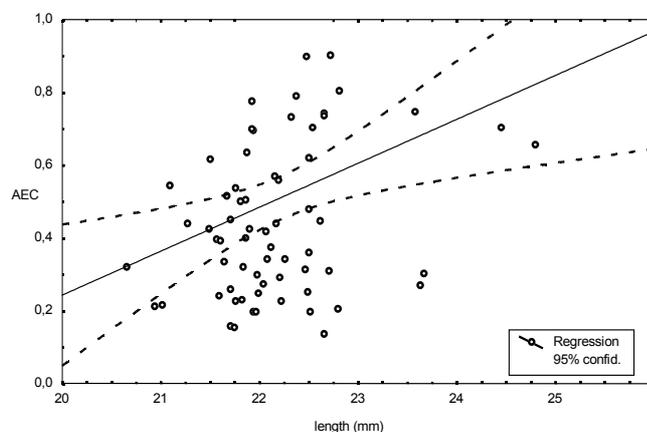


Figure 11 Regression between the length (mm) and AEC. $F(1.82)=0.005$; $p=0.003$; $R^2=0.321$.

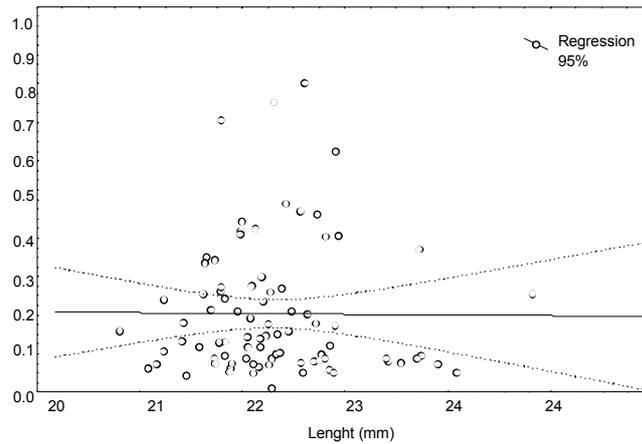


Figure 12 Correlation between the length (mm) and $\mu\text{mol ATP/mg}$ dry weight ratio for *Spisula solida* bivalves. $F(1.82)=0.005$; $p = 0.944$; $R^2 = 0.0006$.

Aerial exposure of bivalves captured by dredging

Adenylic-derived indices

Aerial exposure experiments on *Spisula solida* captured by the dredge showed differing AEC and $\mu\text{mol ATP/mg}$ dry weight ratios among bivalves subject to varying exposure times. An ANOVA revealed non-significant differences for the AEC, but significant differences in ATP/mg dry weight ratio values between groups. The Tukey (HSD) test for an unequal number of samples revealed significant differences only in the ATP/mg dry weight ratio between groups subjected to 0 and 60 minutes of aerial exposure compared with that subjected to 360 minutes of exposure (Figure 13).

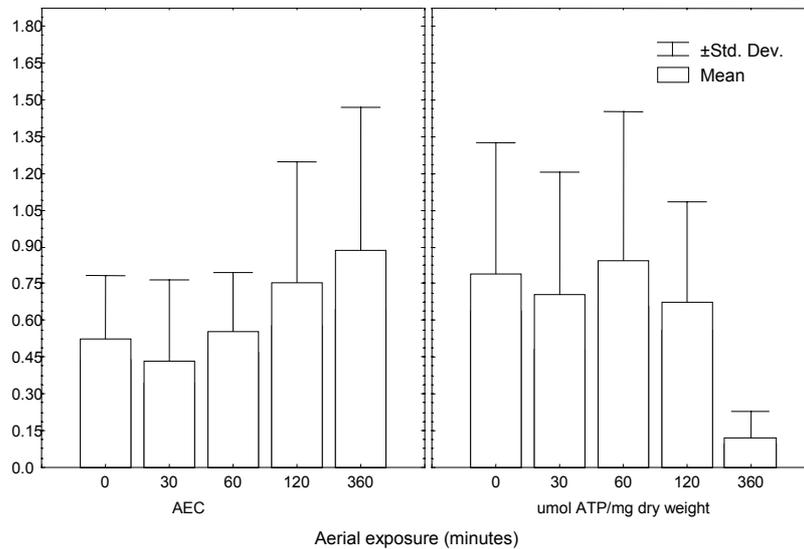


Figure 13 Comparison of the mean and standard deviation of 1) AEC level, and 2) $\mu\text{mol ATP/mg}$ dry weight ratio in captured *Spisula solida*, subjected to differing periods of aerial exposure. ANOVA results: 1) $F(4.89)=0.00096$; $p=0.97$; 2) $F(4.89)=0.813$; $p=0.520$.

Reburying time

Different periods of aerial exposure also affected the reburying time of captured bivalves after return to the sediment floor (Figure 14). Generally, an increase in the aerial exposure period led to an increased delay in the response to rebury, and a subsequent lower cumulative frequency percentage of reburied individuals after the 20 minute measurement period. However, individuals subjected to a 30 minute aerial exposure period started to rebury earlier than both controls.

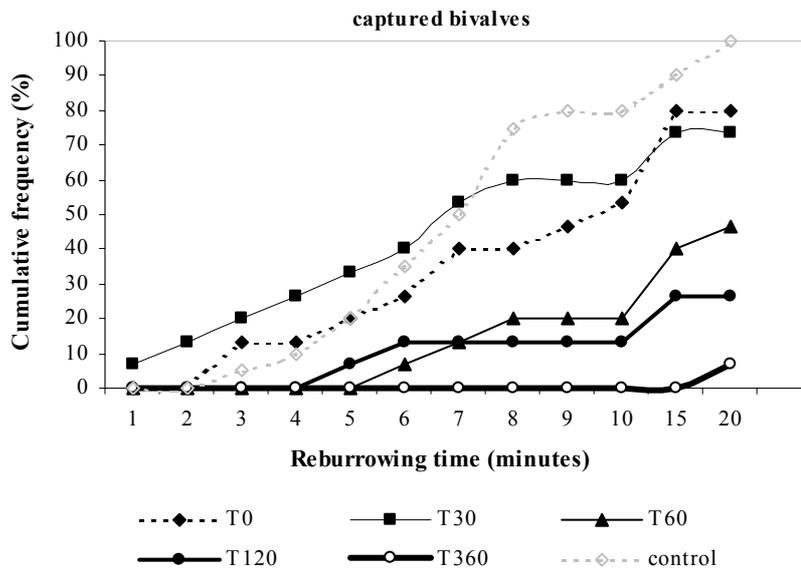


Figure 14 Reburying time of captured *Spisula solidus* under different aerial exposure periods on deck: 0, 30, 60, 120 and 360 minutes. The control line records the reburying time of unstressed individuals determined in situ. Each exposure group contained 15 individuals.

Table 2 shows the cumulative frequency percentage and number of reburied individuals after 20 minutes. An increase in the aerial exposure period was found to lead to a lower number of reburied individuals. A linear increase on the mean (\pm standard deviation) of the reburying time with an increase on the aerial exposure period was not found, despite an increase among the group exposed for 360 minutes. The group exposed for 30 minutes exhibited the lowest reburying time (5 ± 3 minutes).

Table 2 Captured and reburied *Spisula solida* with mean \pm standard deviation reburying time over 20 minutes on deck (aerial exposure periods: 0, 30, 60, 120 and 360 minutes) and in situ (control). N^{er}, number; min., minutes; cum. freq. (%), cumulative frequency percentage; reb., reburied; std. dev., standard deviation; *, value from one individual only.

Treatment	Experimental site	Aerial exposure period (min.)	Initial n ^{er} of ind.	N ^{er} of reburied ind. after 20 min.	Cum. freq. (%) reb. ind. 20 min.	Mean \pm std. dev. reburrowing time (min.)
control	<i>in situ</i>	0	15	15	100	8 \pm 4
T0	deck	0	15	12	80	8 \pm 4
T30	deck	30	15	11	73	5 \pm 3
T60	deck	60	15	7	47	11 \pm 4
T120	deck	120	15	4	27	10 \pm 5
T360	deck	360	15	1	7	19*

The Kruskal-Wallis test, when applied in the comparison of captured and reburied individuals subjected to different periods of aerial exposure, using the presence/absence transformation within the 15 individuals of each treatment group, demonstrated significant differences between groups ($H(5, N=90)=45.98, p=0.000$).

Comparison between adenylic-derived indices and reburying time of stressed bivalves

The mean and standard deviation of $\mu\text{mol ATP/mg}$ dry weight and reburying time for the different aerial exposure periods revealed a similarity in the results of these two variables. Both variables decreased as the aerial exposure periods increased (Figure 15). However, the AEC of the bivalves showed no such inverse relationship. The linear regression showed high R^2 values (0.95) for both indices, being positive for AEC and negative for $\mu\text{mol ATP/mg}$ dry weight.

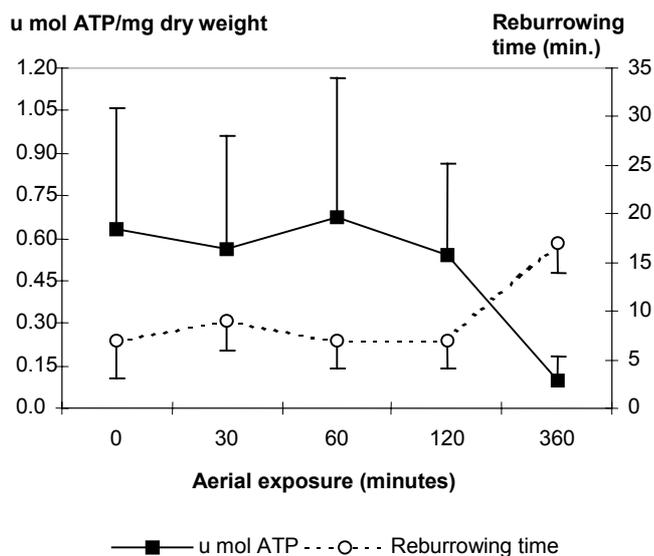


Figure 15 Mean and standard deviation of $\mu\text{mol ATP/mg}$ (dry weight) against reburrying time (of bivalves that reburied in the 20 minute period) among groups of *Spisula solida* subjected to varying periods of aerial exposure.

Seasonal changes

Seasonal changes in the RNA/DNA ratio were significant and comparatively more obvious than the changes arising through the direct impact of the fishery itself. Moreover, the condition, as exhibited in the RNA/DNA ratio, of those bivalves collected in April was generally lower than among individuals collected in July (Figure 16).

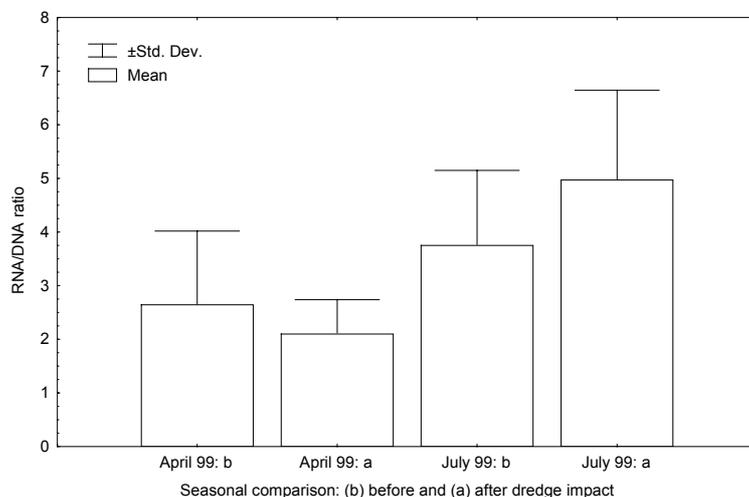


Figure 16 Seasonal comparison of bivalve condition (the RNA/DNA ratio) of *Spisula solida* collected in the field, with corresponding averages and standard deviations. (Season Main Effect: $F(3.36)=9.17$; $p<0.000121$). (The Tukey HSD revealed significant differences between April and July).

Seasonal changes in the N/P lipids ratios were significant between spring (April) and summer (July) (Figure 17).

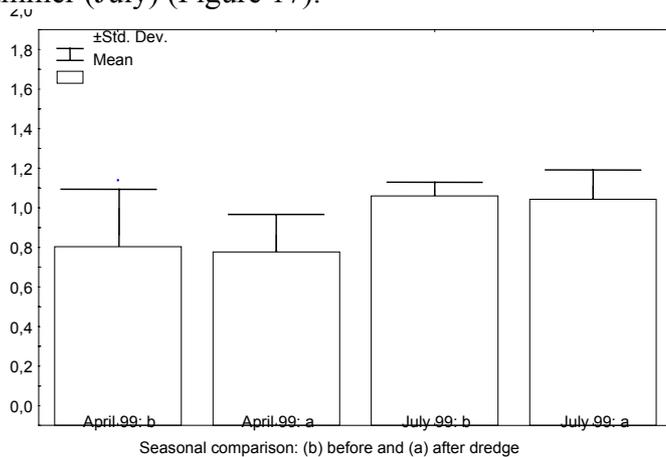


Figure 17 Seasonal comparisons of Neutral/Polar lipid ratios in *Spisula solida* collected at a fishing ground. (Month Main Effect: $F(3.32)=5.87$; $p<0.0067$). (Tukey HSD revealed significant differences between April and July).

Discussion

Minimum level of biochemical indices

To assess the impact of dredging on field-caught bivalves it is necessary to establish the level of the biochemical indicator being used below which the bivalve will be classified as stressed. Then, through an accurate application of the biochemical condition indices, it is important to determine the maximum stress level of the organism, which implies its minimal survival condition, that is, the critical level.

A few studies have determined the critical levels of several bivalves. Chícharo and Chícharo (1995) established that, among *Ruditapes decussatus* in Ria Formosa (Portugal), survival is not guaranteed when the RNA/DNA ratio is lower than 1. Maguire et al. (1999a) have shown that a stressed bivalve, with an AEC level of 0.3–0.5, would have a negative scope for growth and not recover, this then being the critical, minimum level for survival. The results of the determination of the minimum ratio of Neutral/Polar lipids described here were in accordance with previous studies with polychaetes (Hentschel 1998), suggesting a critical level of 0.8, corresponding to bivalves in low condition. When these level is achieved almost all lipid reserve were used.

The establishment of such critical levels plays an important role in the determination of bivalve condition, with widespread applications in assessing the impact of fisheries on resident populations. In this field there are mainly two type of stress imposed to the undersized bivalves: 1- cumulative mechanical stress due to successive disturbance of dredging in the same area and 2- aerial exposure of the bivalves retained by the net because of clogging and after some time on the deck rejected to the sea. The later stressor examined here is simultaneous anoxia, desiccation and temperature-induced stress.

Experiment on cumulative stress due to dredging

Results from the examination of N/P lipids and RNA/DNA ratios among laboratory-stressed bivalves suggest that these tests provide an inadequate index of cumulative stress in *Spisula solida*, especially in view of the brief time of the experiment. In fact, both indices decreased after the bivalves were shaken in this simulation of the stress that undersized bivalves experience from repeated dredging in the same area. However, differences were not significant nor were critical ratios reached. We therefore reject the hypothesis of a sudden increase in RNA concentration as a response to increased stress. This rejection is also based on the findings of Clemmesen (1994), that RNA/DNA ratios do not reflect increase in stress over a matter of hours. Rather it was found that a sudden increase in stress leads first to decreased activity of the ribosome followed later by an increase in their numbers, with fluorometric methods measuring only the ribosome content. A similar explanation seems to apply to the lipid index, in that stress applied for a few hours does not lead to a degradation of lipid reserves.

When biochemical indicators of acute stress (AEC and ATP concentration) were employed, obvious changes were observed among undersized *S. solida*. AEC values among bivalves subjected to simulated dredge stress dropped within the first hour before increasing. Differences in these AEC values, however, were not significant. ATP concentrations among similarly treated bivalves initially increased and then decreased significantly by the end of the experiment. Barthel (1984) associated initial breakdown of AEC and consequently ATP concentration, in other bivalves, to a shift to high energy demand along with low energy production. The subsequent increased ATP concentration may have been the result of increased energy production or reduced consumption.

Comparison of the results from the biochemical indices of dredging stress indicate the test for ATP concentration was the most accurate, showing a significant decrease with increased stress. Results from this test were also independent of bivalve length.

Aerial exposure of bivalve captured by dredging

In contrast to the results of Maguire et al. (1999c), the test for AEC in the in situ experiment and the laboratory simulation was shown not to have been an accurate measure of stress in *Spisula solida*. The difference may be specific to *S. solida* as Maguire et al. (1999c) worked with the bivalve *Pecten maximus*. In the present study, ATP concentrations in captured *S. solida* decreased significantly with critical minimum levels achieved after six hours of anoxia. The ATP decline was probably the result of increased energy consumption. Higher initial ATP concentrations recorded in the present study suggest that bivalves, in the various treatment groups, suffered greater stress than those monitored in the seabed. Other studies (e.g. Wijsman 1976a; Isani et al. 1995) examined the effects of up to 24 hours anoxic exposure and recorded decreases in ATP levels. However, the Isani et al. (1995) study carried out long-term anoxia experiments on *Mytilus edulis*, and observed a pattern of metabolic adaptations that was clearly biphasic, with the first (transition) stage characterized by a decline in ATP concentration which then increased in the second (stationary) stage.

In the present study, a similar relationship was observed between ATP concentration and time in captured bivalves. Therefore, the recorded increase in reburying time with longer periods of anoxia may have been influenced by decreased ATP concentrations. This indirect but significant relationship between reburying time and ATP concentration is very useful as it allows a rough determination of ATP concentration through examination of reburying time. Nevertheless, the reburying time appears to be a valuable indicator of stress levels in bivalves and a useful, indirect estimate of mortality. This test has the additional advantage of being easy to use in the field.

Seasonal changes

Normal seasonal differences were invariably higher than the stress-induced changes recorded. These results are to be expected as the biochemical indices test chronic stress only. RNA/DNA and N/P lipids ratios indicated lower bivalve condition in April. This decline is the result of spawning activity. Spawning in *Spisula solida* along the South Coast of Portugal occurs from February to May, with the greatest spawning activity in April (Gaspar 1996a). Cockles and clams lose condition during the spawning season (Boyden 1971). This loss of condition would be especially evident through analysis of muscle, as in this study, since during spawning proteins and lipids from muscle are redirected towards gonad development (Paon and Kenchington 1995). Increased values of RNA/DNA and N/P lipids ratios recorded in July, three months after spawning, may indicate that the bivalves had recovered from the intense physiological activity of gamete production. Further studies should examine the effects of dredging stress on *Spisula solida* during the spawning period, as acute stress effects could be higher than those detected in the present study.

Conclusion

We conclude from this study that both the cumulative, mechanical stress due to the passage of the dredge and temporary aerial exposure on the deck of the fisheries vessel affect the behavioural and physiological responses of undersized *Spisula solida* in ways that decrease its ability to escape predation and reduce its survival chances.

Acknowledgements

Thanks are also due to the EcoResources Group for help with the processing of laboratory samples and to the staff of the research vessel “Donax”, from IPIMAR. We also want to acknowledge Dr Julie Maguire who assayed samples at the University College of Cork.

3.1.3.8 CHANGES IN THE REBURIAL TIME OF *SPISULA SOLIDA* CAUSED BY FISHING DISTURBANCE AND ITS CONSEQUENCES TO INDIRECT MORTALITY

Introduction

The environmental effects of shellfish dredging have received special attention throughout the world (Caddy 1968; Caddy 1973; Meyer et al. 1981, Hall et al. 1990, McLoughlin et al. 1991, Dare 1974, Kaiser et al. 1998, Gilkinson et al. 1999a).

In Portugal, fishing for bivalves has been an important commercial enterprise since 1969. Today, the industry targets *Donax trunculus*, *Venus striatula*, *Pharus legumen*, *Ensis siliqua*, and *Spisula solida*. Bivalves are caught with a dredge that can penetrate into the sediment up to 50 cm, depending on target species and sediment type. Even if

clams not retained by the dredge bag may also die as a consequence of the fishing. The passage of the fishing gear across the seabed leads to both direct and indirect mortality through subsequent predation (Kaiser and Spencer 1995). The extent of this additional mortality must be quantified if we are to calculate total mortality associated with the fishing (Kaiser and Spencer 1994). Murawski and Serchuk (1989) observed variable survival rates among those clams returned to the seabed as “undersized”, and providing potential food in the dredge track for scavengers and predators (Ramsay and Kaiser 1998).

The survival of non-damaged individuals may be related to the time they need to rebury. Exposed individuals are preferential targets for opportunistic invertebrate (Wassenberg and Hill 1987) and fish predators (Kaiser and Spencer 1994). It is reasonable to predict that predation upon uncaught clams in the dredge track increases with exposure time. However, this time is probably influenced by stress resulting from the impact of the passage of the dredge. Thus, the indirect mortality of the uncaught *Spisula solida* attributable to predation will depend on the relation between time for reburial and the time needed for predators to reach the impacted area. If the reburial time is longer than the time taken for the initiation of predation, then an expected important indirect mortality can be attributed to predation activity on the dredge track. If exposed clams rebury themselves before significant numbers of predators can reach the area, then indirect mortality due to predation will be negligible.

The aims of this work were to analyse the: 1) impact of dredging on the exposed *Spisula solida* clams; 2) impact of dredging on the abundance of potential predators; and 3) time needed for predators to reach the dredged area relative to the exposed abundance of *Spisula solida* individuals to rebury themselves, and its implications to indirect mortality.

Methods

Study site

The Algarve coast of Portugal extends from Cabo São Vicente in the west (8° 59'W), to the border with Spain in the east (7° 24'W). Currents on the continental shelf of Algarve are usually weak, with wind-caused drift currents running along the shore,

from east to west, to a depth of 30 m or more, predominating over tidal currents. Normally, currents speeds are under 0.25 m/s, but during extreme conditions may reach 0.5 m/s or more (Fiúza 1983). Water temperatures were 14° C to 22°C. The study took place off Vilamoura on a sandy fishing ground 8 meters depth (Figure 1).

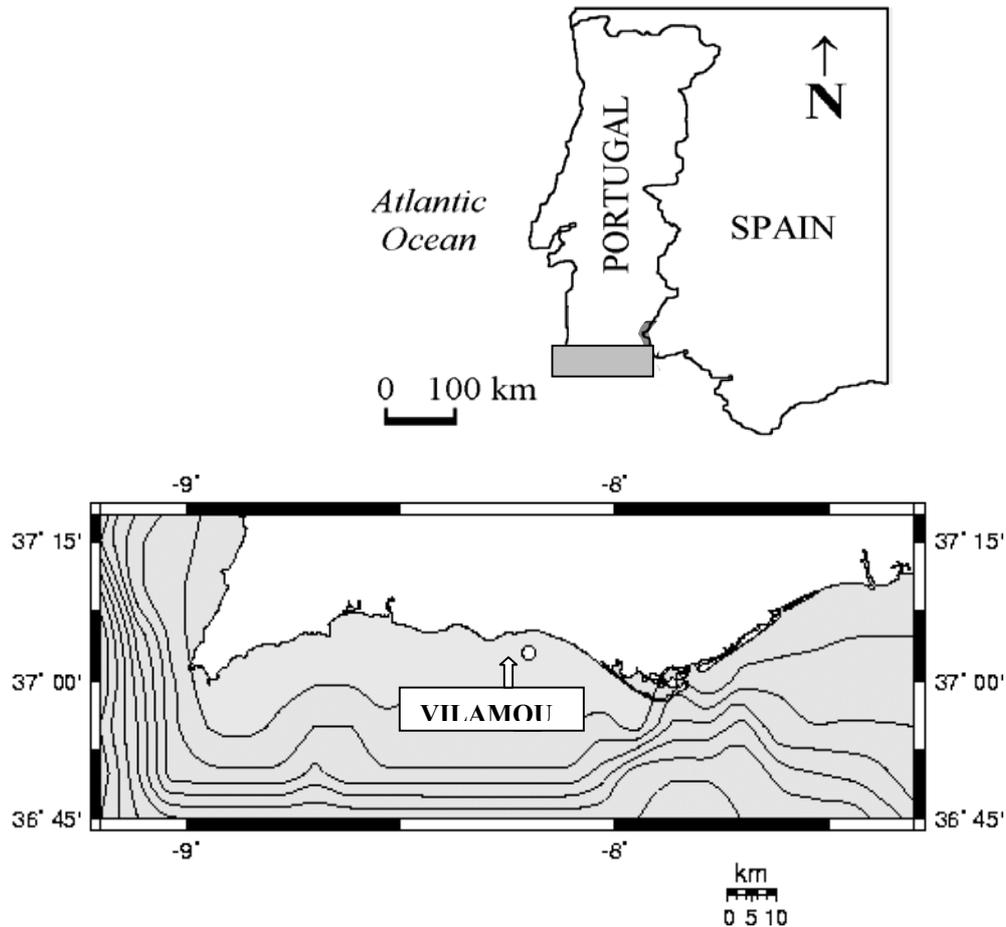


Figure 1 – Map of the South Portugal (Algarve) coast, showing the sampling location: Vilamoura.

Fishing gear used

Bivalve dredges used on the south coast of Algarve are large, heavy, iron structures, with a 25-mm-mesh net bag, and a metal semicircle parallel bar grid and toothed lower bar at the mouth. The mouth is 43 cm wide, bars are 2.3 cm apart, and the teeth are 34 cm long, acting as a rake when the dredge is dragged through the sediment (Figure 2).

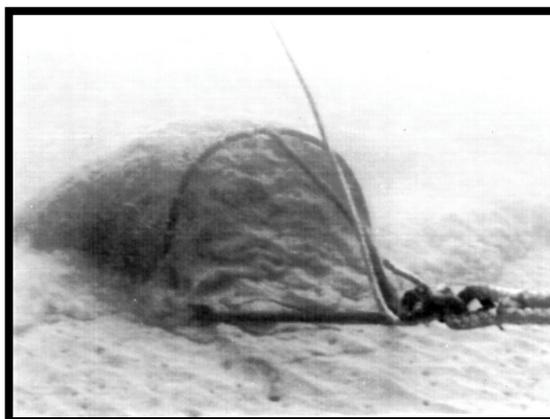


Figure 2- Dredge used in the sampling.

Sampling strategy and laboratory procedures

The in situ experiments were performed in July 2000. Three areas studied before dredging were control areas. Three independent sites were impacted by dredge, and within each site three independent samples were taken. The sampling strategy focused on two main objectives: first, determination of the time taken for *Spisula solida* individuals to rebury themselves; second, the time taken for predators to arrive.

To measure the time needed for *Spisula solida* to rebury themselves before there was any impact from fishing, divers carefully hand-collected three groups of 10 bivalves from the sediment. This procedure was done very gently to avoid stress on the clams. Immediately after collection, the clams were replaced on the seabed. Divers directly assessed the numbers of bivalves both reburied and exposed every three-minute for 30 minutes (this time was found to permit the reburial of almost all bivalves).

To determine the impact of fishing activity on exposed *Spisula solida* clams, normal dredge fishing was performed and three fifty-metre dredge-tracks were created (approximately 1 min of towing). Divers already near the sea bottom immediately collected 10 exposed clams from each dredge track. These were all placed in an area of 1 m² for ease of observation by the divers. The number of bivalves reburied and exposed was assessed every three-minute for 30 minutes. The number and level of damaged bivalves was recorded *in situ* as follows: level 1 (absence or low damage)- bivalve intact; level 2 (medium damage)- small breakage of the shell; and level 3 (severe damage)- more than two-thirds of the shell broken.

In a non-fished control area and to minimize the risk of predator avoidance divers buried three-square quadrats measuring 50 × 50 cm into the sediment, so that the structure was invisible, but leaving, each vertices identifiable. The numbers of predators that entered the quadrats marks during three-minute intervals were assessed for 30 minutes. Individuals re-entering the quadrat areas were counted once.

The fishing procedure was repeated three times. Immediately after passage of the gear, divers repeat the same procedure described above for the non-fished control area. Highly mobile species such as fish or cuttlefish were identified *in situ*. Benthic potential predator macrofauna species were collected and brought to the laboratory for

identification. Abundance of Predators considered just individuals with length at least twice the preys (*Spisula solida*- 20-25 mm).

For characterization of the area sediment analysis were carried out. Sediment samples collected and in laboratory organic mater in the samples was eliminated using H₂O₂ (130 volumes). Sediment was then dried on a stove WTB binder at 60° for 24 hours, and passed through ½φ interval sieves comprised between -2φ to 4φ (equivalent to 4mm to 0,063mm), using stack sieves Retsch AS 200 basic, during 10 minutes.

Data analysis

The effects of dredge and time on predators’ abundance and on the number of exposed bivalves were examined by a two way ANOVA performed with the Statistic V.5 software. Due to the absence of homogeneity of variances and of normality of the data (p<0.05), these were log transformed. As more than two comparisons were made, this analysis was followed by a means comparisons Tukey test to compare means between times and the results displayed on graphics.

Results

The study took place off Vilamoura, on a sandy fishing ground dominated by sediment grain size of 0.5 and 0.355-mm (Figure 3).

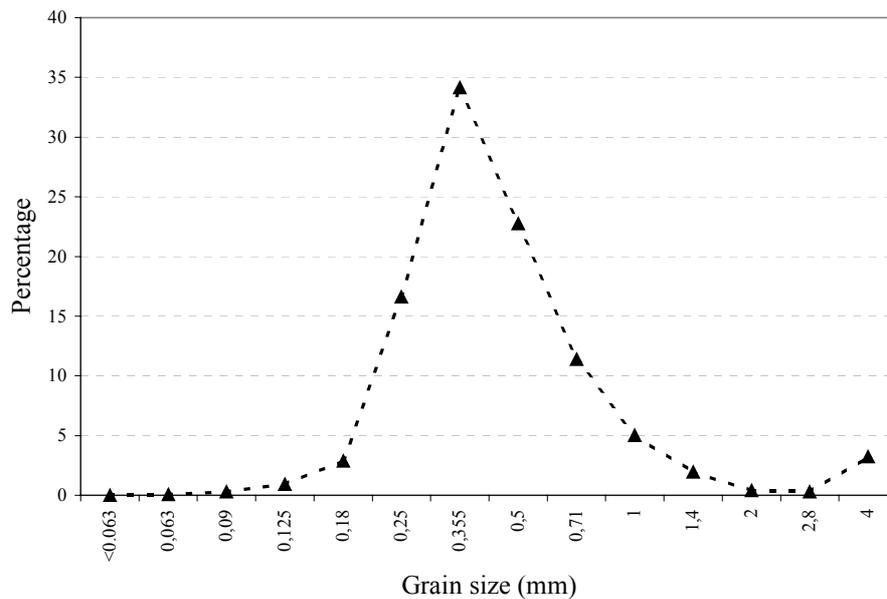


Figure 3- Distribution of sediment grain size in Vilamoura area (see figure 1 for details)

The number of exposed *Spisula solida* clams increased significantly ($p < 0.05$) after dredge impact (Figure 4, Table 1). According to Tukey test especially after 3 min of dredge impact. (Figure 5). All *S. solida* individuals from control areas were reburied 12 minutes after being exposed, while in the dredge impacted areas more than 30 minutes was needed for all bivalves to rebury themselves. Not only more clams reburied during the period of observations, but also they rebury faster in the control areas. In the control areas, 50% of the clams were completely reburied after three minutes, while in the dredge impacted areas this was increased to nine minutes, at this time all the clams in the control study were buried. In the impacted area, 80% of the clams were reburied after 15 minutes, 90% after 24 minutes, and some never reburied. Thus, 50% of the clams that suffered the impact of dredging took six minutes longer to rebury themselves than did the clams that did not suffered any impact (Figure 6).

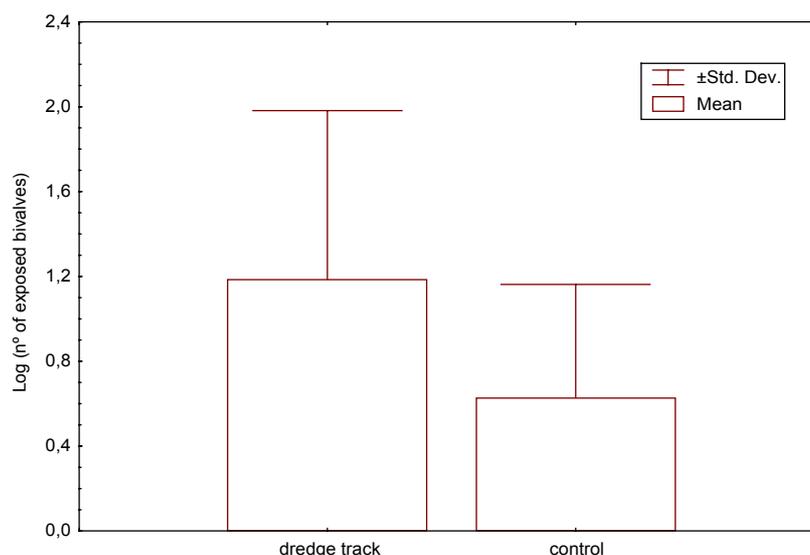


Figure 4 – Comparison between the number of exposed *S. solida* individuals, before and after the dredge impact (Std. Dev. is standard deviation for the mean)

Table 1 - Summary of a two way ANOVA, fixed effect for log number of exposed bivalves, with nested design:1-before and after dredge, 2-time (df - degrees of freedom, MS- Mean Square Error).

Effect	df	MS	df	MS	F	p-level
	Effect	Effect	Error	Error		
1	1	4.177	52	0.462	9.029	0.004*
2	17	1.037	52	0.462	2.245	0.001*

Comparisons between the abundance of potential predators before (control) and after the fishing disturbance, measured 30 minutes after the beginning of the experiments, showed that significant differences occurred ($p < 0.05$) (Table 2, Figure 7). In the control study, predator abundance was almost constant (5–6 individuals per square meter). Predators abundance on the dredge track started to increase three minutes after dredging (ranging from 6 to 21 individuals per square meter). During the following 27 minutes, average predator densities remained quite similar, with a minimum of 18 individuals per square metre (Figure 8).

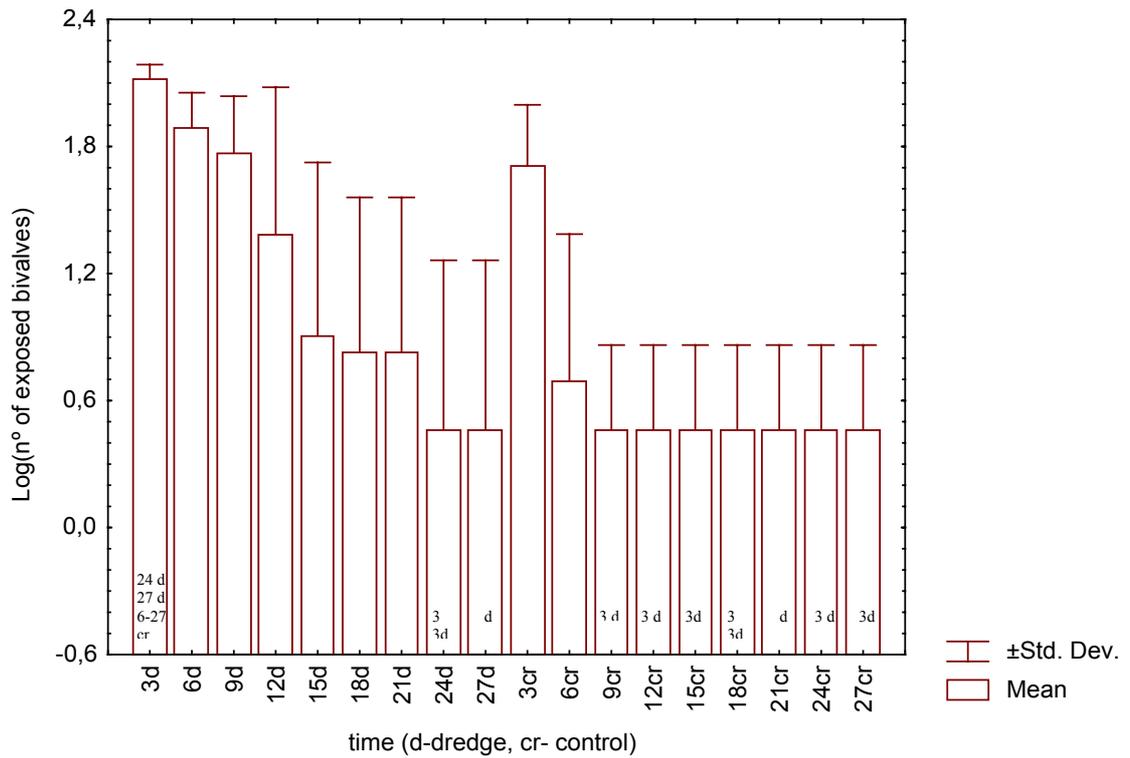


Figure 5 - Change in average of exposed *Spisula solida* over the 30-min. observation. Error bars are standard deviation. Number at each bar showed the significant differences (s.d.) between times, results of Tukey test ($p < 0.05$), which was done after the ANOVA, with significant F (Table 1).

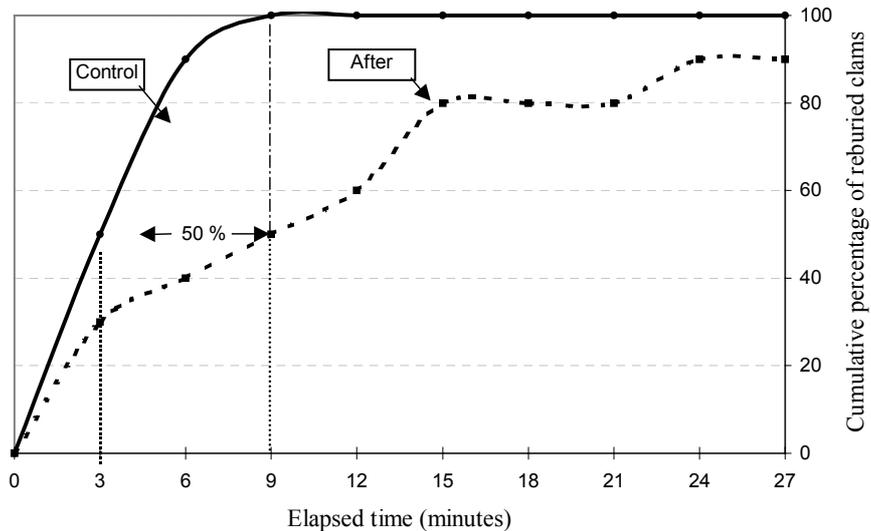


Figure 6 – Cumulative percentages of reburied *S. solida* clams, before (control) and after the dredge impact, during the 30 - minute experiment.

The echinoderm *Ophiura texturata* was the most abundant species moving to dredge tracks during the 30-minute experiment (72% to 100% of the total potential predators observed). In fact, it was also the first species to reach the impacted area, increasing its density in the first minute after the passage of the gear. The presence of fish *Pomatochistus* spp. was observed six minutes after the passage of the gear, while the anomuran *Diogenes pugilator* and the gastropod *Nassarius reticulatus* were observed nine minutes after (Figure 9). The flatfish *Solea solea*, cuttlefish *Sepia officinalis*, mullet *Chelon labrosus* and an unidentified blenniid fish were also observed on dredge track areas.

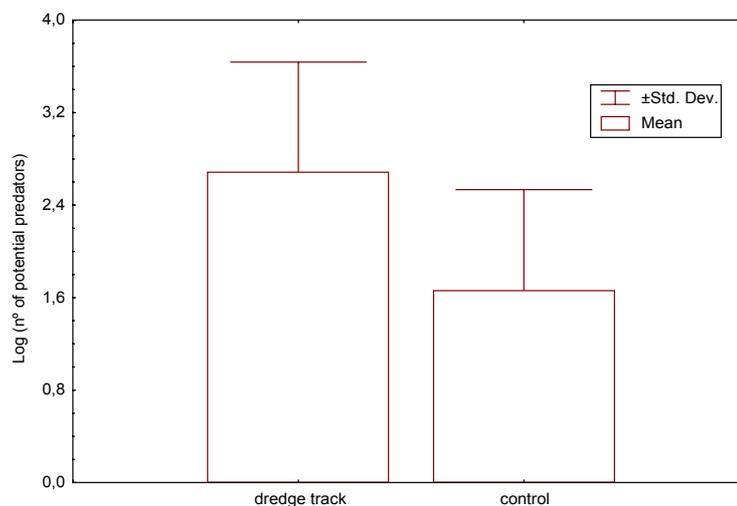


Figure 7 - Comparison between the average of potential predator abundance (log transformed data) that reach the study areas, before and after dredge impact.

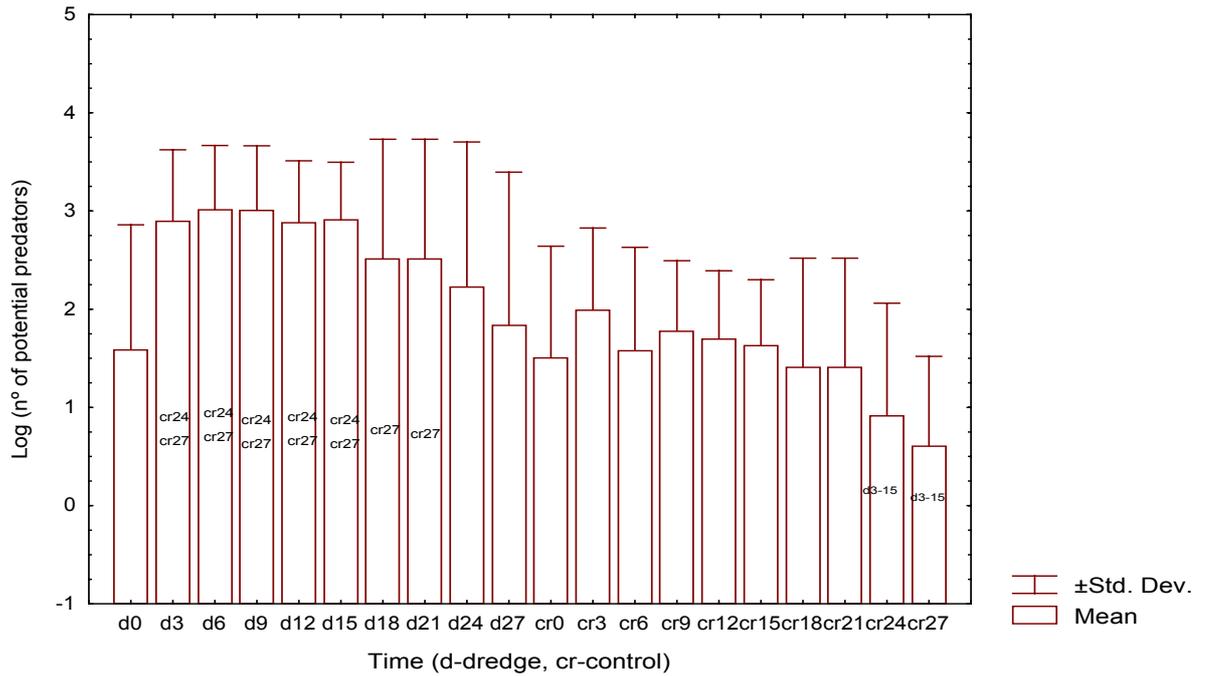


Figure 8 - Change in the average of potential predators abundance (log transformed data) over the 30-min. observation, before and after dredge impact. Error bars are standard deviation. Number at each bar showed the significant differences (s.d.) between times, results of Tukey test ($p < 0.05$), which was done after the ANOVA, with significant F (Table 2).

Our *in situ* observations revealed that the most damaged and/or exposed species due to dredging were the urchin *Echinocardium cordatum* and the echinoderm *Amphiura mediterranea*. Infaunal polychaete species, bivalves *Macra* sp. and *Callista chione*, seastar *Astropecten* sp., and gastropod *Cymbium olla* were affected. Damage to the fishing target species *Spisula solida* was, however, negligible. Less than 6% of the bivalves not caught on the dredge track were damaged, and less than 2% suffered severe damage. The results shows that most of the bivalves were classified in level 1 (absence or low damage)- 94.23 % (Figure 10).

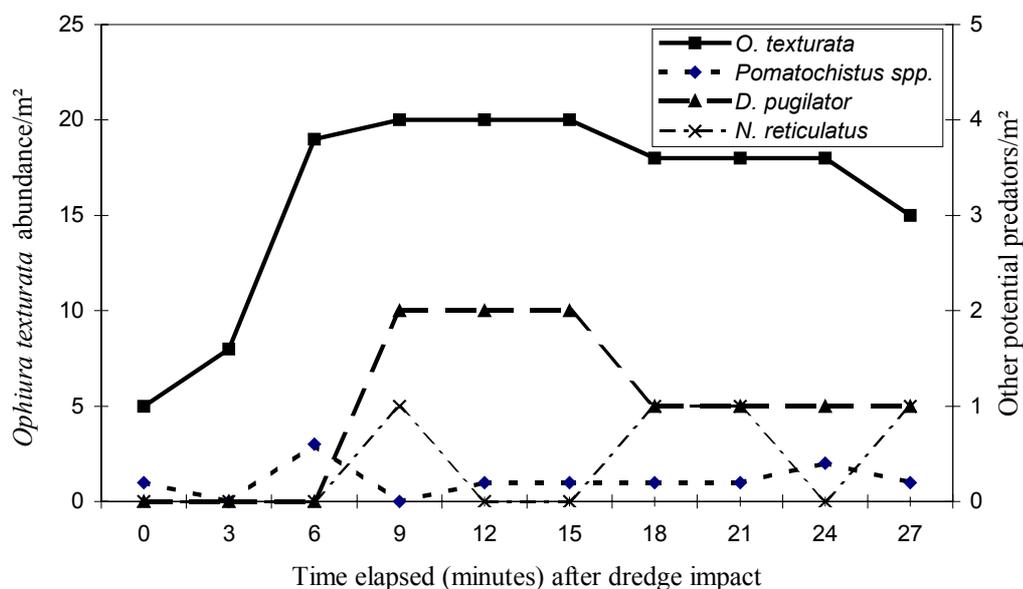


Figure 9 – Variations in the abundance of the potential predator species that reached the area after the dredge impact during the 30-minute experiment.

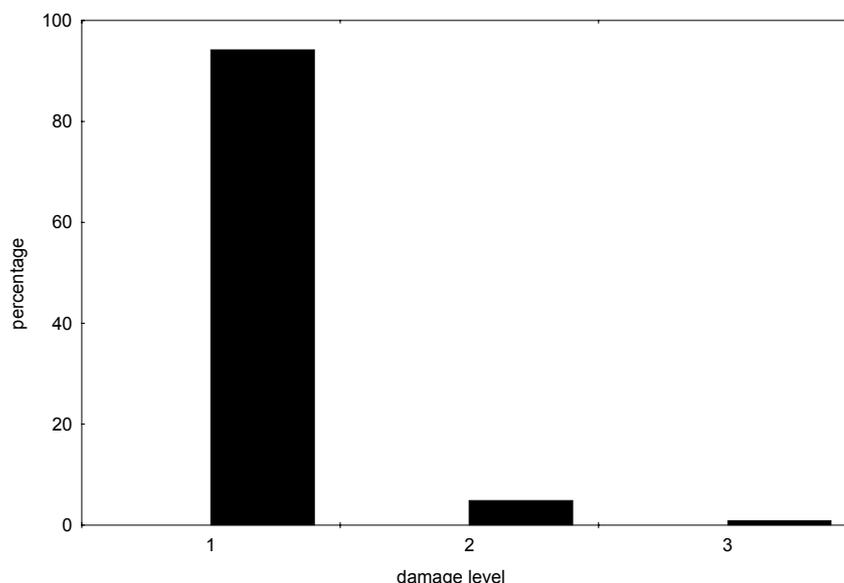


Figure 10 – Percentage of damaged *Spisula solida*. The number and level of damaged bivalves was recorded *in situ* as follows: level 1 (absence or low damage)- bivalve intact; level 2 (medium damage)- small breakage of the shell; and level 3 (severe damage)- more than two-thirds of the shell broken.

When comparing the time needed for the impacted clams to rebury themselves with the time needed for potential predators to reach the impacted area, at least 50% of the clams rebury themselves before the arrival of predators (Figure 11). Moreover, potential predator abundance became more important (> 50% of the total potential predators counted during the experiment) after 12 to 15 minutes, when 40 to 20% of clams were still exposed.

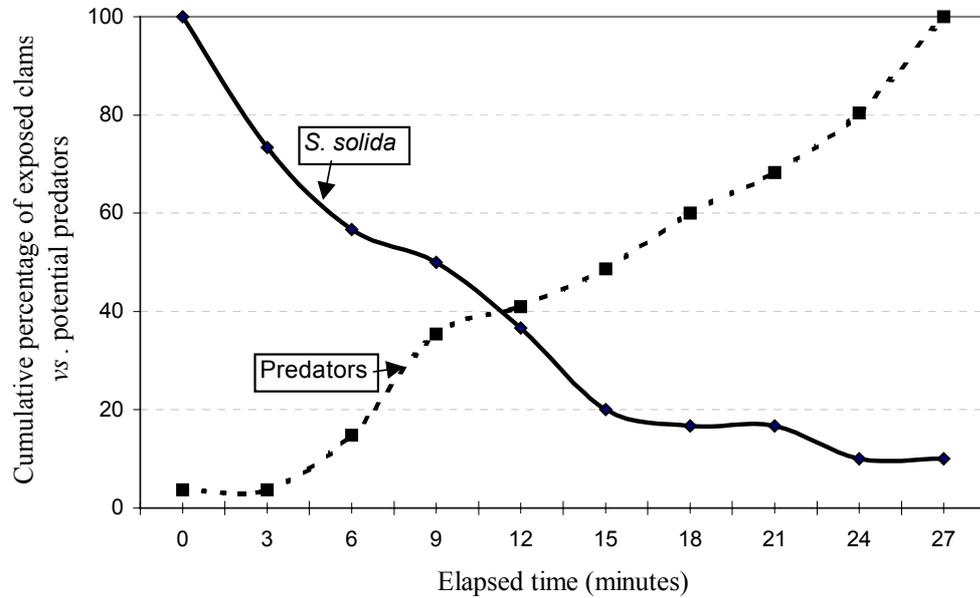


Figure 11 - Relation between the cumulative percentages of reburied *S. solida* and the abundance of potential predator species reaching the area after the dredge impact.

Table 2 - Summary of a two way ANOVA, fixed effect for log number of predators, with nested design:1-before and after dredge, 2-time (df - degrees of freedom, MS- Mean Square Error).

Effect	df Effect	MS Effect	df Error	MS Error	F	p-level
1	1	53.153	160	1.023	51.961	0.001*
2	19	4.611	160	1.023	4.503	0.001*

Discussion

Studies have indicated that a variety of fishing gear, such as beam trawls (Bergman and Hup 1992; Kaiser and Spencer 1994), otter trawls (van Dolah et al. 1987, van-Dolah et al. 1991; Rumohr and Krost 1991), and dredges (Veer et al. 1985), can cause mortality of some epi- and infaunal benthic organisms. Although the direct effects of this activity on benthic communities appear obvious, and may cause large-scale alterations to predator populations (Kaiser et al. 1998), the magnitude of the effects has been very difficult to evaluate and has often been considered equivocal (Thrush et al. 1995).

In our study, the impact of the passage of the dredge increased the time needed for *Spisula solida* individuals to rebury themselves. Thus, the mechanical impact of the dredge caused a reduction of bivalve reburial capacity. Interestingly, Breum (1970) reported that *S. subtruncata*, a species with a depth distribution similar to *S. solida*, exhibited increased burrowing activity when disturbed by wave action.

Disturbance to the sea bottom caused by dredging also affected the abundance of potential predators in the fishing area. The number of predators increased after passage of fishing gears, as was observed by Meyer et al. 1981), Kaiser and Spencer (1994), and Kaiser et al. (1998) observed specific differences in aggregating behaviour between species of the same genera, *Pagurus*. In addition, Evans and Tallmark (1996) reported rapid movement of whelks (*Buccinum undatum*) towards animals damaged or killed because of the impact of fishing. Ramsay et al. (1997c) observed a significantly higher proportion of the predator seastar *Asterias rubens* after fishing. In fact, fisheries can impact population structures in the short-term, driving the evolution of life-history changes in the long-term (Pechenik 1999).

In our study, the opportunist invertebrate *Ophiura texturata* was the most abundant and fastest species to reach the dredge track (less than three minutes). According to Feder (1981), *O. texturata* is a potential predator of meiofauna and small benthic macrofauna. Also *Pomatochistus* spp, that reached the impacted area six minutes after impact may effectively predate the exposed bivalves, according to Zander (1982) and Fitzhugh and Fleeger (1985). However, other species that appeared occasionally at the dredge track areas may have more importance on *Spisula solida* as predators, i.e. the flatfish *Solea solea*, cuttlefish *Sepia officinalis* (as observed by Najai and Ktari. 1979), and mullet *Chelon labrosus* (as reported by Gisbert et al. 1995). These species may also feed on fragile non-target species, such as *Echinocardium cordatum*, *Amphiura mediterranea* or polychaetes, all damaged heavily by passage of dredges'. Because half of the clams exposed on the dredge track needed nine minutes to rebury and 80% needed 15 minutes, when the last predator species arrived at the dredge track there were still 20 to 40% of the clams exposed; this predatory impact was smaller than expected, due to the high selectivity of the dredge (Gaspar 1996a). In fact, less than 5% of the bivalves evidenced medium or severe damage.

The impact of predators will probably be greater during the early post-larval period, when bivalves are much smaller, more fragile. In fact *Diogenes pugilator* and *Nassarius reticulatus*, species that occurred in the impacted area nine minutes after impact are described as scavengers or predators upon small meiofauna (Ramsay et al. 1996; Volvenko 1994). However, observations by Alexander (1993) on *Anadara ovalis* suggest that the time to reburrow increases with shell length. Therefore, if

similar behaviour occurs in *Spisula solida*, it can be expected that the main potential targets of the predator species found in the dredge tracks may escape, leaving available for predation the larger clams that, unless damaged, do not appear available as prey. Thus, it seems unlikely that a significant number of exposed *S. solida* would be effectively preyed upon.

Our results suggest that predatory action upon unharvested clams on the dredge track is not a major factor in the mortality of *Spisula solida* and that potential predators reaching the dredge track areas do greater damage to the fragile non-target species. However, since a relation could be established between the reburial time of impacted target species and the arrival of predators at the harvesting area, after fishing impact, this albeit minor aspect should be considered in indirect mortality estimations caused by fishing activities. Moreover it will be important to mention that the study tried to infer indirectly whether predation was increased by dredging, but mortality rates of clams were not actually measured and dredging in nature occurs repeatedly, which may have a greater effect than the one-pass tested in this study

3.1.3.9 SUMMARY STATEMENT

Acute biological responses as indicators of short-term stress due to spring 1 dredging in the scallop *Pecten maximus* (L.) in the Irish Sea and off the coast of s.w. Ireland

There is little information on the effect of dredging on the physiology and behaviour of discarded scallops. The first paper in this study examined the effect of simulated dredging on the scallops' behavioural response to stimulation by a predator; the results showed that swimming behaviour is affected for at least 24 hours after simulated dredging.

This study also used changes in adenylic energetic charge (AEC) and righting and reccessing behaviour as indicators of stress caused by the act of dredging on scallops. The effect of scallop size, tow speed, tow length, dredging followed by emersion, together with the effect of multiple dredge events were tested in the field using a commercial dredge and a laboratory based dredge simulator.

AEC levels decreased gradually in the striated muscle from 0.9 to approximately 0.5 after 15 minutes of simulated dredging, however no further difference was found between longer lengths of tow or length of time in the dredge simulator (15, 30 or 45 minutes). The AEC levels of dredged scallops returned to normal after 3 days in optimal conditions. Dredged smaller scallops (<65mm shell length) had a higher AEC level (0.54 – 0.58) and were more active than dredged larger animals (>70mm shell length, AEC level; 0.41- 0.46). Dredging followed by emersion had a deleterious effect on AEC levels and on the righting and reccessing speed of scallops. Physical movement of the scallops within the dredge bag combined with anaerobic respiration during valve closure has an added stress effect during dredging.

Tow speed had an important impact on both AEC and behaviour. The AEC level decreased from 0.85 to 0.70 and 0.56 after 15 minutes dredge simulation at low and high speeds respectively. In addition, the behavioural score declined in dredged animals only at the high tow speed. Recovery was also monitored and was rapid and scallops from the low speed treatments had AEC levels of >0.8 after 2 hours, whereas, the AEC returned to this level after 6 hours in the scallops from the higher speed treatments.). For scallops exposed to low speed simulation on a daily basis for three days, the reduction in the AEC level was less on the first day of simulated dredging than on subsequent days. However, recovery time was similar (2 hours) for each of the three days tested. Each application of an additional dredge disturbance at the higher speed did not have a significant cumulative effect on the AEC level or behavioural activity of the scallops. The period between disturbances was long enough for the scallops to significantly recover from the previous stress. However, it is not known whether a cumulative stress effect might have occurred in the test scallops if the recovery period was shorter.

Temporal and spatial variability in the stress caused to the great scallop *Pecten maximus* by dredge capture was also determined using the adenylic energetic charge (AEC) of the striated muscle as an index. Samples were taken on a seasonal basis from dredge captured scallops from a single fishing ground in the north Irish Sea. All samples were taken from individuals under the minimum legal landing size. On one

date samples were taken from scallops from three separate grounds with contrasting bottom characteristics. At all dates dredge captured scallops had a significantly reduced level of AEC compared to control scallops. This reduction was greatest in February when glycogen levels in the muscle tissue were shown to be at a minimum. There was little or no variability among fishing grounds in the effect of dredge capture on AEC levels. An additional aim of this work was to determine if scallops that encounter dredges but are not captured suffer a reduction in AEC levels. Scallops collected from dredge tracks immediately following the passage of dredges showed a significant reduction in AEC compared to control, scallops collected from outside the tracks. However, the reduction in AEC was not as great as in captured scallops.

Overall, AEC levels were not reduced enough to cause mortality, but the righting, rearing and swimming speed of scallops was greatly reduced after dredging. The experiment to determine recovery time in *Pecten maximus* clearly showed that a scallop's ability to escape a predator is at its lowest immediately after dredging. A limited recovery was shown after one hour but at this stage, and for the following 23 hours, scallops were still negatively affected by dredge disturbance. These data, together with numerous studies of predator aggregation to discarded material, indicate there is the potential for high levels of mortality in undersized discards of *Pecten maximus* and in impacted but un-captured individuals. There is clearly a need to carry out realistic field-based tests of survival of stressed and damaged organisms impacted by demersal fishing.

Acute biological responses as indicators of short-term stress due to hydraulic dredging in the striped venus clam *Chamelea gallina* in the adriatic

This project offered the opportunity to focus on the repeated disturbance caused by commercial hydraulic dredges on the target species. If commercial hydraulic dredging has significantly negative effects, substantially weakening the exploited clam population, it may be considered a partial cause of increased mortality repeatedly observed in the North Adriatic beds of *Chamelea gallina* in the last ten years or at least a factor contributing towards abnormal susceptibility to normal fluctuations in environmental parameters. Four seasonal samplings on natural *C. gallina* grounds were carried out in February, May, July and October 2000. Considering the various sources of stress during commercial fishing operations, clam samples were collected in two fishing areas along the west coast of the North Adriatic Sea (Jesolo and Lido) using four fishing methodologies: dredging at high water pressure and using a mechanical sieve for sorting (as in commercial fishing), dredging at high water pressure without sorting, dredging at low water pressure without sorting and manually by divers (control samples). At the same time, different experiments, simulating fishing mechanical stress, were carried out in the laboratory.

The biomarker approach used has been proposed as a relative new tool. An estimate of dredging impact was performed evaluating the effects of the acute mechanical stress by applying biological indices at both cellular and organism level. In particular, immunological (hematocrit value, phagocytic index, enzyme activity indices) and biochemical (Adenylic Energy Charge, AEC) parameters have been determined at cellular level, whereas behavioural assay based on the reburrowing ability at organism.

Generally, field results were in agreement with the laboratory data. Hematocrit value, phagocytic and acid phosphatase activity indices were able to distinguish samples collected with different dredging systems revealing both the effects of water pressure and mechanical sorting. On the contrary, statistical differences in β -glucuronidase activity among samples could be detected, suggesting its relative independence from mechanical stimulus. AEC and reburrowing revealed only the pressure effects, especially during winter time: the presence of different internal (reproductive effort) or environmental stress could have hidden the effect of the dredging action in any other season. All the applied indices showed a seasonal trend; phagocytic activity and reburrowing showed a significant positive correlation with temperature, as demonstrated for other invertebrates. Hematocrit value and acid phosphatase activity index showed the highest values during the winter period and the lowest in summer, revealing a correlation with the reproductive cycle of this specie. AEC revealed a negative correlation with temperature and it's also seemed to be affected by reproductive cycle. Only β -glucuronidase activity did not show a seasonal trend, even though a slight increase was detected in summer.

The Principal Components Analysis exhibited a clear separation between the two clam populations, even though it did not revealed significant geographical differences in fishing impact, probably due to the heavy exploitation which takes place in both areas.

Overall results indicate that the acute stress indices tested in the framework of this project could be used to measure the impact of hydraulic dredging on *C. gallina*, even though further research is needed to study the influence of endogenous and environmental variables (reproductive period, temperature, salinity, dissolved oxygen, food availability, etc.) on the biological responses of this clam to mechanical stress.

Chronic biological responses as indicators of short-term stress due to hydraulic dredging in the striped venus clam *Chamelea gallina* in the Adriatic

A battery of chronic biomarkers, previously tested in laboratory experiments, was chosen with the purpose to make an indirect estimation of dredging impact by evaluating the effects of acute mechanical stress which clams experienced during fishing operations.

Samples were collected in two exploited areas along the North Adriatic coast (Lido and Jesolo) using different fishing approaches with the aim to distinguish stress levels associated with the main components acting for catching clams: pressure in water jets and mechanised sorting. Only physiological rates and survival in air test seemed to be clearly influenced by the fishing methodologies: filtration and scope for growth generally showed decreasing trends as mechanical stress increased in both sampling sites; respiration rate was less responsive, being not distinctly affected by different levels of stress.

Survival time of dredged clams decreased with the increasing of stress, the LT_{50} values of samples collected using low water pressure being higher than those fished with high water pressure. A positive correlation between size and stress was observed: under-sized clams had longer survival times than commercial ones both at Lido and Jesolo, this results confirm those obtained in laboratory experiments.

Catalase, protein content and latency were not able to reveal either statistical differences or expected trends when comparing samples differently dredged at any season, however all these indices showed marked seasonal variation. In parallel with the above biochemical and histochemical biomarkers, RNA/DNA ratio and acid nucleic contents resulted to be unable to highlight a clear relationship with stress levels due to fishing activity.

Analysing the biological responses of clams at the lowest and at the highest level of mechanical stress experienced, greater differences were observed in July and October, thus underlining an increased sensitivity of clams to the impact of dredging in these periods of the year. These results indicated a decreased capability of clams to cope with the mechanical stress during or immediately after spawning season, when endogenous stress was higher.

Chronic indices suitable to highlight short-term effects revealed that the impact of hydraulic dredging has deleterious effects on commercial size clams. Also it must be considered particularly harmful to under-sized clams which are first dredged and sieved, and then rejected, all the more so as, in the studied zone, the same area in the clam ground may be dredged more than ten times per month.

Indicators of the effects of dredging-induced stress on the clam (*Spisula solida*) in Portuguese waters

Little attention has been given to the behavioural and physiological effects of dredging on clams. The response of the clam *Spisula solida* under stress imposed by dredging activity is analysed in terms of reburying time and different indices of biochemical conditions (acute: AEC, ATP/mg dry weight; chronic: RNA/DNA ratio, Neutral/Polar lipids ratio). Cumulative stress on undersized (<25 mm) *S. solida* due to repeated habitat disturbance by dredging was simulated in the laboratory and measured with *in situ* studies in April and July 1999, and July 2000 at Vilamoura, off the southern coast of Portugal. Laboratory simulation on undersized bivalves indicated a decrease of RNA/DNA and Neutral/Polar lipids ratios, but critical levels, beyond which survival is not guaranteed, were not achieved. However, for the acute biochemical indices critical levels were achieved, but only ATP/mg dry weight had a significant decrease, which was independent of bivalve length compared with AEC. An *in situ* study showed significant increases in reburying time and adenylic-derived indices of captured bivalves compared with unstressed bivalves. ATP/mg dry weight levels in ship-discarded bivalves decreased with increased exposure to air. However, normal seasonal differences were invariably higher than the dredging impact itself, with RNA/DNA and Neutral/Polar lipid ratios lower in April because of loss of condition from spawning. We concluded that the stress caused by dredging affects the behavioural and physiological responses of *Spisula solida*, which in turn reduce survival rates by decreasing their ability to escape from predators.

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3.1.4. Biological - Individual Damage Studies

Introduction

The four parts of this Section investigate the level of physical damage to target and non target organisms on the sea bed caused by dredging in three different fisheries using three contrasting types of dredge. The first two sections examine dredge induced damage in the scallop fishery of the Irish Sea using spring toothed dredges. The third Section is concerned with the Portuguese clam dredge fishery and the final Section with damage to the target species of hydraulic dredging in the northern Adriatic.

As well as description of the types of organism affected and the level of damage suffered, the different sections each address a different question. The first section ([3.1.4.1](#)) considers the difference in damage level to benthic megafauna in captured and non-captured organisms in the scallop fishery of the Irish Sea. This is followed by [Section 3.1.4.2](#) which experimentally assesses how the level of damage in undersized scallops affects their subsequent survival after discard. The third and fourth sections both investigate how changes in dredge design and operation can affect the level of damage. [Section 3.1.4.3](#) examines the effect of changes in the mesh size and tooth spacing in simple clam dredges in Portugal while the final ~~Section~~ [\(3.1.4.4\)](#) focuses on how changes in operation of the hydraulic dredge in the northern Adriatic can influence damage to undersized discards of the target clam species.

3.1.4.1 THE IMPACT OF SCALLOP DREDGING ON BENTHIC MEGAFUNA: A COMPARISON OF DAMAGE LEVELS IN CAPTURED AND NON-CAPTURED ORGANISMS

Introduction

The short term changes in the abundance of non-target organisms after the passage of scallop dredges (Currie and Parry 1996; Kaiser et al. 1996) may be caused by removal of biota in the bycatch, or as a result of high levels of mortality in organisms which encounter the dredge but are not captured. Such ‘unobserved mortalities’, resulting from gear encounters, are difficult to assess and have been largely ignored (Chopin and Arimoto 1995). We tested the hypothesis that dredge-induced damage is greater in bycatch organisms than in those left on the seabed by determining the level of damage in both groups of organisms *in situ*. Observations on dredge tracks were made by divers immediately after the passage of the fishing gear, and analysis of the catch was made on hauling. Simultaneous assessment of both bycatch, and organisms left on the seabed, were used to estimate capture efficiency for both target and non-target organisms. Data on capture efficiency and damage were used to predict the impact of scallop dredging on megafaunal communities.

Methodologies.

Dredging was undertaken by the RV Roagan on the ‘Bradda Inshore’ scallop ground off the west coast of the Isle of Man. One gang of four ‘Newhaven’ spring toothed dredges (see [Review Section 3.2](#) for description) was used on each side of the boat. These dredges used 78mm id (10mm wire) rings and 9 French teeth/bar (see [Section 3.2.2.4](#)). Ten tows were carried out over the period July 12th to July 21st 2000 in water ranging in depth from 29 to 35m. All dredging took place within an area of 2.74 km² which allowed clear separation of the dredge tracks. In accordance with common commercial practice, each tow lasted on average 45 minutes at a mean speed of 2.77 knots (5.13 kph). At the end of each tow the catch was sorted on deck and the abundance and damage score (see below) of all megafauna were recorded.

On each of the ten tows the dive support vessel RV Sula dropped two buoyed shot lines at timed intervals of 5 minutes immediately behind RV Roagan as dredging was being carried out. In this way, two positions, separated by approximately 400m, were marked between the tracks made by the two gangs of dredges. At each shot line up to two pairs of divers were deployed, each pair locating and surveying a separate dredge track with a width of 3.5m. During the course of the survey, dredge tracks were marked in this way during both the first and second half of the tows.

A short interval after the shot lines had been positioned, divers located the dredge track, and surveyed the track for as far as possible given limitations of air and decompression requirements. Initially, divers entered the water 5 minutes after shotline deployment. However, the reduction in visibility caused by sediment disturbance during dredging made surveying difficult. Thus, the majority of dives took place 15-20 minutes after the passage of the dredge when visibility had improved. The abundance and damage score of megafauna within the track, including the target organism *Pecten maximus*, were assessed. The sediment type and approximate depth of track furrows were also recorded. At the end of the survey a marker buoy was released. The distance between this marker and the shot line buoy

was determined by the dive support vessels on the surface using a Global Positioning System, and gave an estimate of the distance surveyed by the divers. For each of the 10 tows, between 2 and 4 pairs of divers were deployed making a total of 28 survey dives. The mean distance surveyed was 120m (range: 25m- 283m) and a total of 2.99 km of track made by one gang of dredges (10,500m²) was covered by the divers.

Damage levels of megafauna were assessed, both by divers and by scientists on board RV Roagan. This was done in an identical manner using a simple four point scale, with the criteria used appropriate for each taxonomic group (Veale et al. 2001); Table 1). The mean damage index (MDI) for each species was calculated using the formula :

$$\frac{\sum_{i=1}^{i=4} ni}{N}$$

where n_i = number of organisms of damage score i, N = total number of organisms

A t test was used to determine differences in the mean damage index between the bycatch and organisms left on the seabed. Data were first tested using an F test to verify homogeneity of variances.

Table 1 Damage scores for megafauna (after Veale et al. 2001)

	1	2	3	4
Crabs	In good condition	Legs missing/ small carapace cracks	Major carapace cracks	Crushed/ dead
Starfish	In good condition	Arms missing	Worn & arms missing/ minor disc damage	Major disc damage/ dead
Urchins	In good condition	< 50% spine loss	> 50% spine loss/ minor cracks	Crushed/ dead
Whelks	In good condition	Edge of shell chipped	Shell cracked or punctured	Crushed/ dead
Bivalves	In good condition	Edge of shell chipped	Hinge broken/ large crack	Crushed/ dead

Results

Observations of dredge tracks showed that, within the experimental area covered (2.74km²), sediment type varied from pure sand to a range of gravely sediments containing mud, sand, shell material, and stones, in varying proportions. Sediments varied over small spatial scales, with distinct changes apparent over 10's of metres. The dredge tracks were not always distinct, even though dives took place only 15-20 minutes after dredging. The depth of furrows made by the dredge teeth varied from zero up to 5-8cm, with the majority only 1-2cm deep. Tracks were more obvious, with deeper furrows, in coarse, loose sediment. A variety of dead and damaged species was observed within the dredge tracks (see below). In addition, three species were observed preying on dead or damaged individuals, the flatfish *Limanda limanda*, the dogfish *Scyliorhinus canicula* and the crab *Cancer pagurus*. These are all mobile species that could aggregate quickly on damaged organisms.

Damage levels (calculated as mean damage index) differed greatly among species (Table 2). Statistical comparison between damage levels in the bycatch and in organisms left on the seabed were carried out for each species using t tests. Data were pooled over the two tows on each of the five days of the survey to give five replicate damage estimates. Three species showed significant differences between the bycatch and the seabed: *Asterias rubens* and *Neptunea antiqua* were more severely damaged in the bycatch (df=8, t = 3.64, P<0.01; df=8, t = 2.47, P<0.05), whilst *Cancer pagurus* was more severely damaged in the dredge track (df=8, t = 2.63, P<0.05). No other significant differences could be detected.

Table 2 The mean damage index (MDI) of the common megafaunal species found in the bycatch, and left on the seabed in dredge tracks, following dredging using spring-toothed scallop dredges (ns: not significant, *: P<0.05). NB Species belong to the following taxonomic groups: ¹ Class Asteroidea, Phylum Echinodermata; ² Class Gastropoda, Phylum Mollusca; ³ Class Malacostraca, Phylum Crustacea; ⁴ Class Echinoidea, Phylum Echinodermata.

Species	Bycatch	Dredge track		Bycatch versus Dredge track
<i>Porania pulvillus</i> ¹		1.0	1.0	ns
<i>Crossaster papposus</i> ¹		1.1	1.1	ns
<i>Asterias rubens</i> ¹		1.4	1.1	*
<i>Astropecten irregularis</i> ¹		1.4	1.4	ns
<i>Neptunea antiqua</i> ²		1.7	1.2	*
<i>Liocarcinus</i> spp. ³		1.9	2.3	ns
<i>Cancer pagurus</i> ³		2.0	2.9	*
<i>Echinus esculentus</i> ⁴		2.2	2.1	ns
<i>Luidia ciliaris</i> ¹		2.6	2.1	ns

The proportions of the four damage scores for each species are shown in Figure 1. Comparison between the bycatch and those left on the seabed shows that for many species (*Crossaster papposus*, *Astropecten irregularis*, *Porania pulvillus*, *Echinus*

esculentus, *Pecten maximus*) proportions with each damage score were very similar. For other species there were clear differences. For example, in both crab species, *Cancer pagurus* and *Liocarcinus* spp., the proportion of crushed or dead (score 4) animals left on the seabed was nearly twice as high as in the bycatch.

The efficiency of capture of megafaunal epibenthos by scallop dredges varied greatly from just over 2% for *Astropecten irregularis* and *Neptunea antiqua*, to 25% for the edible crab *Cancer pagurus* (Figure 2). Capture of the target organism *Pecten maximus*, showed an efficiency of just under 19%, similar to previous estimates for spring toothed dredges on sandy gravel sediments (Chapman et al. 1977; Dare et al. 1993). Of the scallops left in the dredge track only 15% were still recessed in the sediment, indicating that the majority of uncaptured scallops had been lifted from the sediment by the passage of the dredge.

Combining data from the seabed and bycatch allowed prediction of the level of damage suffered by all organisms that encountered the dredge, whether caught or not (Table 3). This further illustrates that some species are little affected by dredging (e.g. *Porania pulvillus*, *Asterias rubens*), while in others over 40% are badly damaged (score 3 and 4).

Table 3 Predicted level of damage to megafauna in the path of spring toothed scallop dredges (bycatch and non-captured animals combined) as a percentage of individuals with each damage score. Damage scores as in Table 1.

Species	Damage score			
	1	23 4
<i>Asterias rubens</i>	93.2	6.2	0.4	0.2
<i>Crossaster papposus</i>	92.1	7.6	0.0	0.3
<i>Astropecten irregularis</i>	67.5	26.8	4.6	1.1
<i>Porania pulvillus</i>	99.8	0.2	0.0	0.0
<i>Luidia ciliaris</i>	53.3	38.6	15.3	20.4
<i>Echinus esculentus</i>	53.3	1.1	29.3	16.4
<i>Neptunea antiqua</i>	76.7	23.2	0.1	0.0
<i>Cancer pagurus</i>	36.8	13.9	6.1	43.2
<i>Liocarcinus</i> spp.	50.0	6.1	7.7	36.3
<i>Pecten maximus</i>	93.0	1.8	2.9	2.3

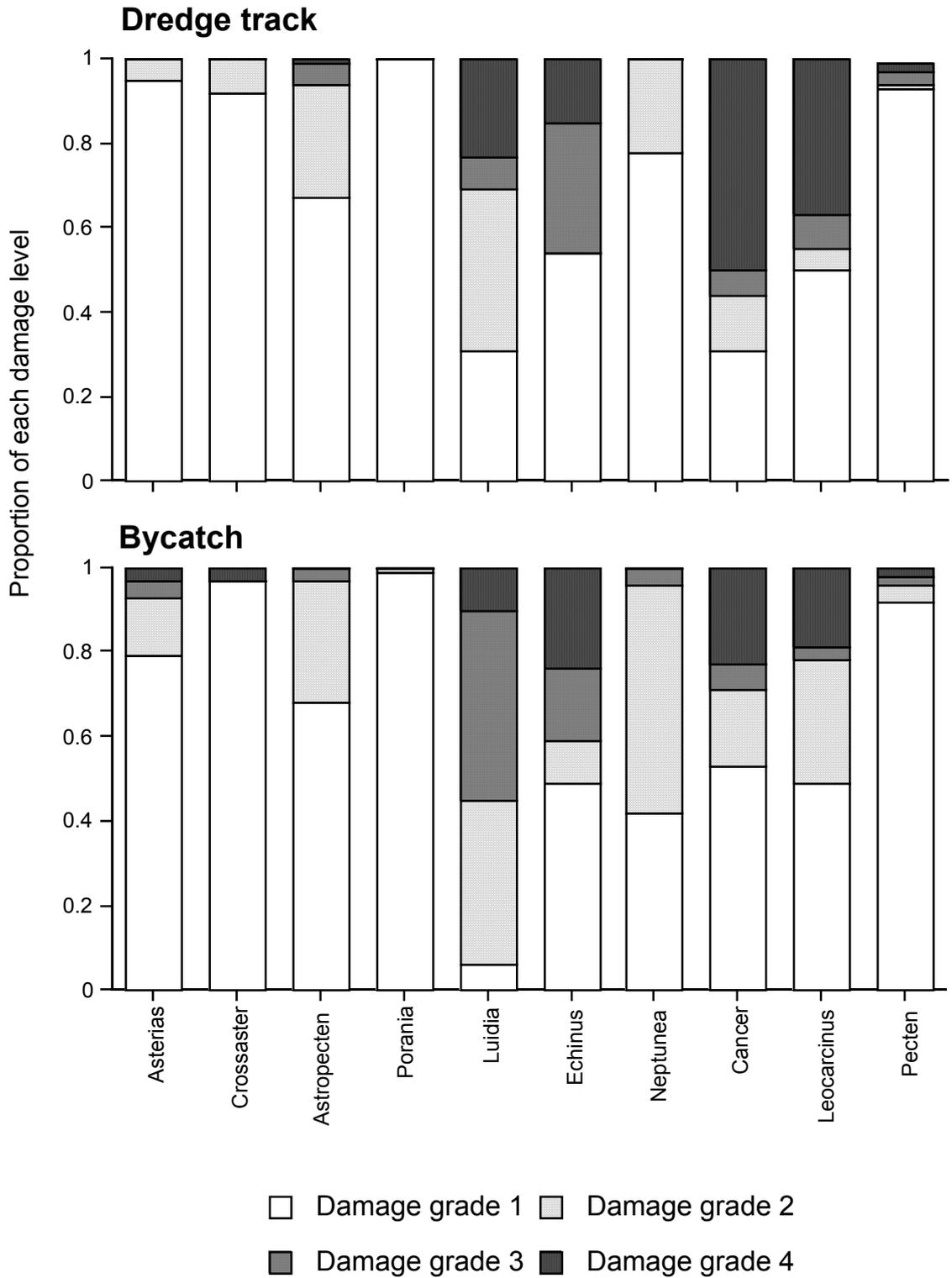


Figure 1 Proportion of individuals of 10 megafaunal species found for each of four damage scores in the bycatch and left in the dredge track.

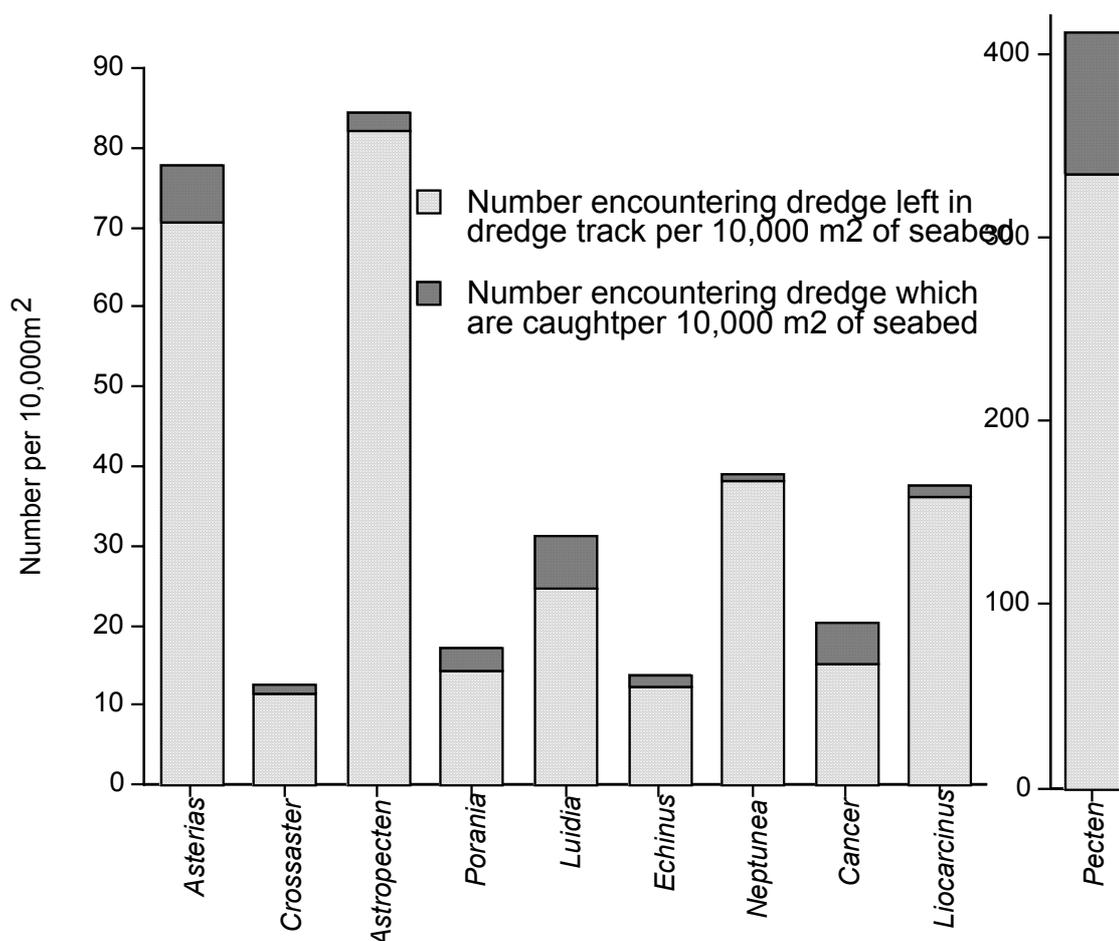


Figure 2 Density (calculated as number per 10,000m² of seabed) and efficiency of capture by spring toothed scallop dredges of 10 megafaunal species.

Discussion

The mean level of damage to a number of common megafaunal species showed surprisingly similar levels between captured organisms landed on deck (bycatch), and those encountering the dredge but left on the seabed. Where differences were apparent there was no consistency in the ranking of the two classes of fauna. This is the first study of demersal fishing gear to demonstrate equivalent levels of damage to benthic megafauna whether captured or not. We had hypothesised that damage would be greater in the bycatch, on the assumption that captured organisms would be subject to longer periods of mechanical disturbance whilst retained in the belly of the dredge. However, damage may also be caused by crushing as animals pass around or under the heavy gear or by the initial encounter with the tooth bar. Our results show that for one species at least (*Cancer pagurus*), fishing induced damage may be higher in individuals not captured, suggesting that crushing by gear passing over an individual may be more harmful than being captured and retained in the dredge belly.

Spring toothed scallop dredges are relatively inefficient at capturing the target organism *Pecten maximus*, with estimates ranging from 6-41% of scallops over 90mm in size depending on ground type (Dare et al. 1993). Our estimate of capture

efficiency for non-target organisms showed large variability among species but efficiency was generally low (<25%). Thus, the majority of megafauna which encounter scallop dredges remain on the seafloor. Given that levels of damage to organisms left on the seabed were similar and in some cases higher than in the organisms captured, it is clear that the majority of damage and subsequent mortality of benthic megafauna takes place in the dredge tracks, and not in the discarded bycatch.

3.1.4.2 DETERMINATION OF THE EFFECT OF DREDGE CAPTURE ON THE SURVIVAL OF UNDERSIZED SCALLOP DISCARDS

Introduction

Work by Partner 9 on stress in captured scallops and by Partner 3 (see Sections [3.1.3.1-4](#) and Jenkins and Brand in press; Maguire et al. 2002a; Maguire et al. 2002b; Maguire et al. in press) on the escape response in captured scallops suggested there was potential for high levels of mortality in undersized discarded scallops. The aim of this work was to determine relative levels of mortality in tethered scallops that had undergone dredge simulation and/or physical damage.

Methodologies

Undersized (<110mm shell length) individuals of the great scallop *Pecten maximus* were collected using spring toothed scallop dredges from the Laxey fishing ground in June 2001. These scallops were kept in tanks of running seawater at Port Erin Marine Laboratory for 2 weeks to recover from the dredging process.

In order to determine the effect of dredge capture on the survival of undersized scallop discards a field experiment was undertaken on the Bradda Inshore scallop ground during July 2001. The survival of scallops which had undergone varying degrees of stress and damage was assessed using a tethering experiment. The experimental design consisted of three treatments, a control where scallops were not stressed, a dredged treatment in which scallops were subjected to 40 minutes of simulated dredging in the laboratory and a damaged treatment in which scallops underwent simulated dredging followed by simulation of slight damage to the ventral margin of the valves. Both dredged treatments were also exposed to air for between 10 and 20 minutes. 5 replicates of each treatment, each consisting of 14 scallops were used.

The experimental set up consisted of a series of weighted bottom lines, 5 for each treatment along which scallops were tethered. The 34m long bottom lines were made of 5mm negatively buoyant rope. At 2m intervals along the bottom line a 1m long length of nylon fishing line (15 lb braking strain) was attached with a fishing swivel. At the end of each mono-filament line a small spring clip was tied to enable attachment of a single scallop. At one end of the bottom line a 10kg weight was attached and at the other a 30cm diameter steel ring. 15 such lines were prepared and wound around wooden frames to allow easy deployment.

On the Bradda Inshore fishing ground 15 concrete blocks, each with a rope and buoy attached to mark their position, were deployed at approximately 150m intervals in a depth of water of between 33-35m above LAT.

300 scallops ranging in size from 76mm to 110mm (shell length) were prepared for tethering by drilling a 1.5mm diameter hole in the shell ear and inserting a ring of Nichrome wire. This was done 3 days prior to the experiment to enable the scallops to recover from handling stress.

The experiment was deployed over 2 days, 10th-11th July 2001. Dredging was simulated by agitating scallops in seawater with stones and sand (see [Section 3.1.3.1](#) for details). Scallops were damaged by removing a small part of the shell using a pair

of pliers. Care was taken to avoid damaging internal tissues. Three bottom lines, (one for each treatment) each with 14 tethered scallops were deployed at a time. 28 scallops underwent dredge simulation and at the end of the 40 minute period, these and a further 14 control scallops were transferred quickly to a boat. Control scallops were transferred in seawater while the dredged treatments were left exposed to the air. Each line was deployed by slipping the steel ring at the end of the line over the buoy marking the position of a concrete weight such that the ring slipped down the marker rope to the seabed. As the line was paid out, a single scallop was attached to each mono-filament line and the boat was allowed to drift with the wind or tide away from the marker buoy. At the end of the line, the 10 kg weight was thrown overboard. To straighten the ground line the marker buoy was taken aboard the boat and the concrete weight slowly dragged in the opposite direction to the line deployment. The time from the end of dredge simulation or movement of control scallops until deployment into the sea varied from 10 to 30 minutes.

The planned duration of the experiment was 10 days. However owing to bad weather the tethered scallops were not retrieved until 20 days after deployment. On retrieval all scallops were examined and scored as either alive, dead with shell broken or dead with shell intact. For this analysis the damage inflicted as part of the experimental treatment was ignored.

During the same period as the tethered scallops were deployed, scallops protected from predators were placed on the seabed and their survival monitored. Four ‘crab pots’ which are used to trap the edible crab *Cancer pagurus* and the lobster *Homarus gammarus* were used as cages to hold scallops on the seabed. Two treatments were used, dredged scallops which had undergone simulated dredging for 40 minutes, and control scallops. 12 scallops from each treatment were each placed in each of two crab pots. These were placed on the seabed in the same general area as the experiment described above with each of the 4 pots separated by at least 30m.

Results

There was no significant difference in the level of mortality among the three treatments (Table 1, Figure 1). Over the three week period of the experiment there was a high level of mortality in tethered scallops in all treatments, with an overall mean of 48% of tethered scallops dying (Figure 1). Of the scallops that died, the majority were unbroken. Less than 8% of those that died, showed signs of shell breakage, while 15 % had a broken hinge. There was no difference in the size distribution of those scallops that survived and those that died. In comparison to the high mortality of tethered scallops exposed to predators, almost all of the scallops protected from predators survived.

Table 1 ANOVA of percentage mortality in dredged and non-dredged scallops

Source	df	MS	F	P
Treatment	2	1049.5	2.77	>0.1
Residual	9	378.8		

Summary and conclusions

This work failed to show any effect of simulated dredging on survival of scallops returned to the sea. There may be a number of reasons for this. As discussed by Jenkins and Brand (in press) and in [Section 3.1.3.1](#) an increase in stress and a reduction in the ability to escape from predators following dredge capture and subsequent discard shows there is potential for high levels of mortality. However this potential will only be realised where predator densities and distribution are such that aggregation occurs at discarded material. The predominant predator of *Pecten maximus*, the starfish *Asterias rubens* was relatively rare in the experimental area. Another predator of *Pecten maximus*, the crab *Cancer pagurus* was common but this species is known to predate scallops of a small size and may have been unable to successfully attack the size of scallops used in this experiment.

This preliminary work does not rule out the potential for high mortality in discarded scallops but does highlight the need to take into account predator distributions.

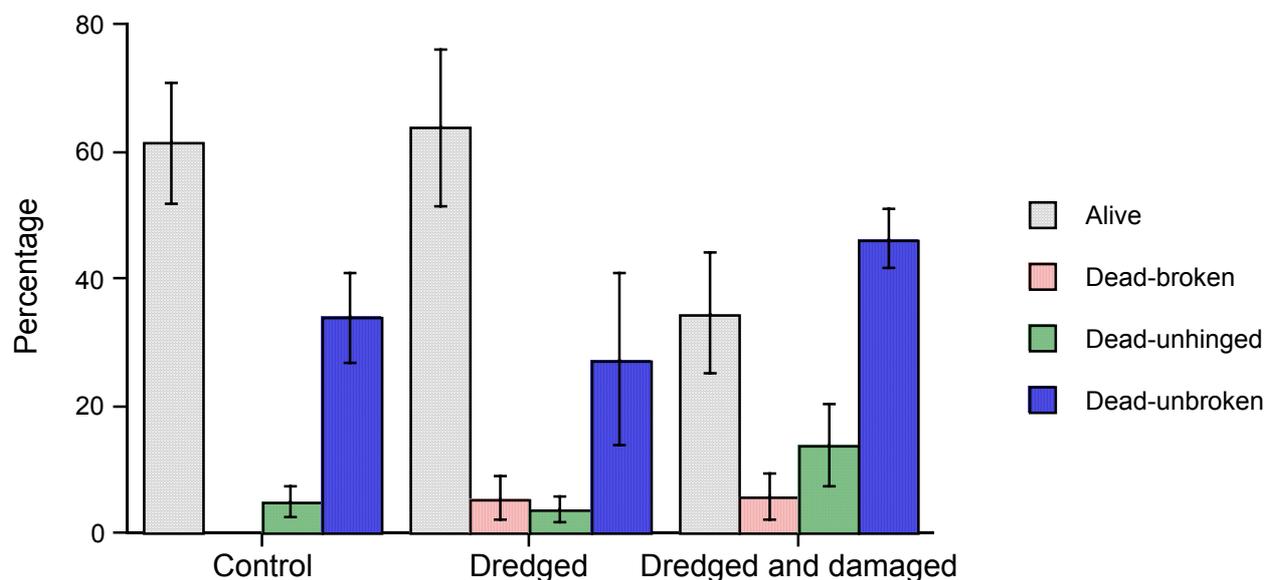


Figure 1 Level of survival and type of damage in tethered scallops that had undergone simulated dredging and physical damage.

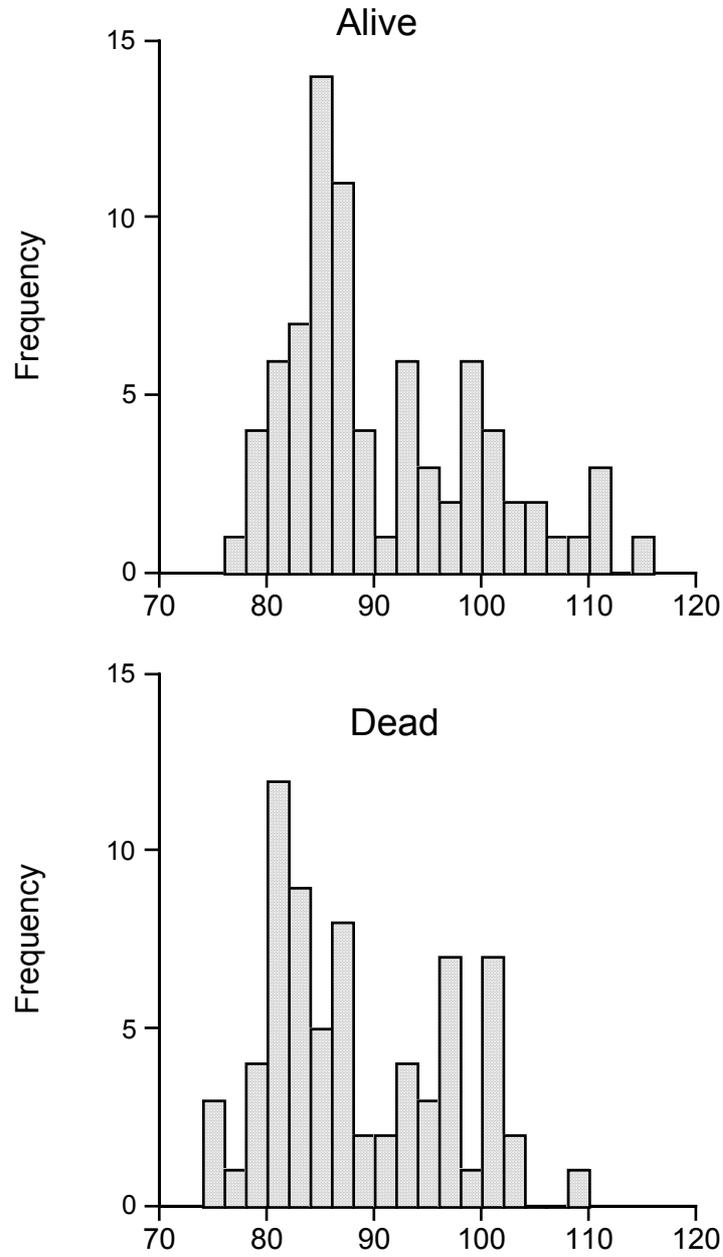


Figure 2 Size distribution of those tethered scallops that died and those that survived

3.1.4.3 INFLUENCE OF MESH SIZE AND TOOTH SPACING ON THE NUMBER OF DAMAGED ORGANISMS IN THE PORTUGUESE CLAM DREDGE FISHERY

Introduction

Demersal mobile fishing gears such as dredges and beam trawls cause a wide range of impacts on the marine environment. Towed along the sea floor, these gears exert physical impacts on the seafloor by resuspending and reworking bottom sediments. Dredges and trawls overturn, move and bury boulders, eliminate microtopography and may leave long-lasting grooves (e.g. Bridge 1972; Caddy 1973; Mayer et al. 1991, Churchill 1989). Lambert and Goudreau (1996) and Tuck et al. (2000) recorded sediment fluidisation in fished tracks. The sediment structure may also change because physical disturbance resuspends sediments leading to changes in the granulometry (Aschan 1991; Leth and Kuijpers 1996). Release of nutrients from the sediment (Krost 1990) and increase of oxygen consumption (Riemann and Hoffmann 1991) can also occur, which may slightly increase phytoplankton productivity. These physical changes may also have an effect on the benthos, either directly or indirectly. Dredging and trawling also inflicts damage both on epifaunal and infaunal species, affecting not only target and by-catch species, but also other benthic individuals, that are not caught but are exposed, damaged or killed during fishing operations. The effects of this kind of fishery to the benthic community could either be short-term or long-term (e.g. Peterson et al. 1987; Bergman and Hup 1992; Eleftheriou and Robertson 1992; Currie and Parry 1996; Kaiser et al. 1998). Therefore, potential ecosystem impacts include changes in community structure and food chains.

The magnitude of the impacts resulting from the fishing activity depends on a variety of factors, such as the type of gear employed, gear penetration depth into the sediment, water depth, nature of the substrate, structure of benthic communities, frequency with which the area is fished, towing speed, local environmental conditions (strength of the tides and currents), and time of the year (eg. de Groot 1984; Redant 1987; Churchill 1989; Krost 1990; Jenner et al. 1991; Mayer et al. 1991).

The effective management of any living resource requires the maintenance of a dynamic balance between the benefits of exploitation and minimising the impacts of exploitation (Brown et al. 1998). Along the Portuguese coast dredges are used to target infaunal bivalves species namely *Spisula solida*, *Donax trunculus*, *Chamelea gallina*, *Ensis siliqua* and *Callista chione*. The Portuguese bivalve dredges are fitted with teeth up to 50 cm long in order to dig target species out of the sediment. If the impact over the sediment is difficult to be minimised, the impact over the benthic community may be reduced by developing new fishing gears or by improving the older ones. According to Sangster (1994) animals may be damaged by different parts of the gear, or may find certain parts of the gear more stressful than others. In the case of the Portuguese dredges, the gear features that are more likely to induce damage on the macrofauna are tooth spacing and mesh size. Thus, the objective of the present study was to assess the effect of these two characteristics on the proportion of damaged individuals present in the catches, in order to determine if changes in tooth spacing and mesh size contribute to the reduction of the number of macrofaunal organisms damaged or killed by dredges.

Methodologies

Study area

The experimental field work was carried out during July 1999, off the Northwestern coast of Portugal in the Aguda region (41°01'00''N and 08° 39'00'' W), one of the most important *Spisula solida* fishing grounds in this part of the Portuguese coast (Fig.1).

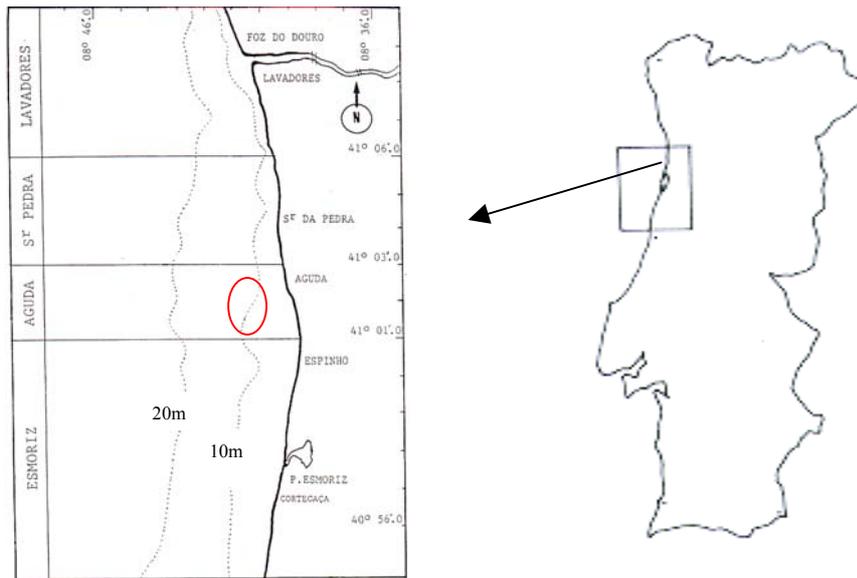


Fig. 1. Map showing Portugal and the sampling area in Aguda (ellipse)

The experiments were conducted at depths ranging from 8 to 10m, where generally the commercial clam fishery takes place. Sediment in the study area consists of well-sorted fine-medium sands and broken shells. Tidal currents in the area flow parallel to the shore usually in the South-North direction..

Fishing gear (dredge design)

The experimental fishing gear used in the present study was similar to the dredges used by the Portuguese northwestern commercial dredge fleet. Figure 2 shows a diagrammatic representation of the gear used during the experiments. The dredge weighs approximately 80 Kg and comprises a rectangular shape iron structure, with a toothed lower bar and a collecting net bag (approximately 4.5 m long). The gear's mouth is 150cm wide and has teeth of 7cm long. Welded to the dredge mouth structure are four metal shafts where the towing cable is tied.

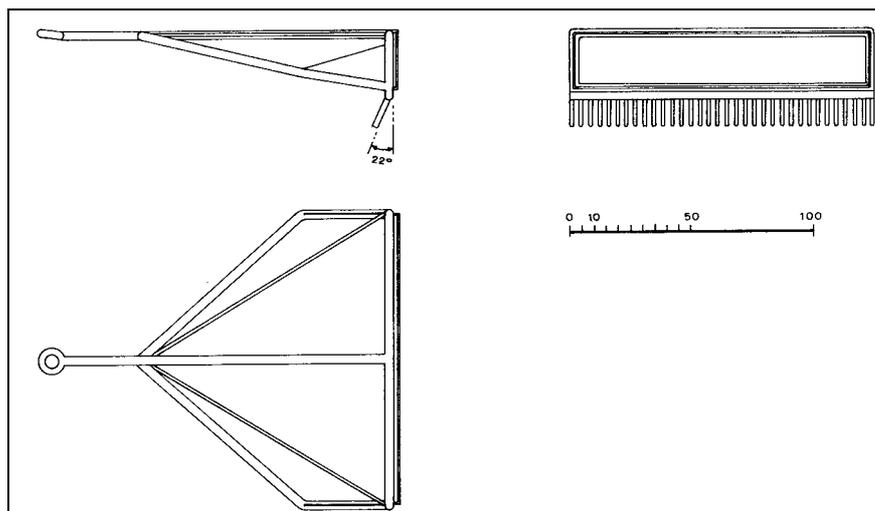


Figure 2 Gear used in the experiments (see also [Review Section 3.2.1](#))

Experimental design

All fishing work was undertaken by the commercial dredging vessel “*Narciso Sérgio*”. In order to evaluate the direct effect of mesh size and tooth spacing on the macrobenthic community, four mesh sizes (35, 40 and 50mm) and three-tooth spacings (2, 4 and 6 cm) were experimented. During the fishing operations, two dredges with different tooth spacing and mesh sizes were towed simultaneously side-by-side. For each, mesh size/ tooth spacing combination, three tows were performed. Every tow was conducted for 15 min each at 1.7-2.3 knots, which is the speed currently used by the commercial dredge fleet.

Before fishing operations, a cover bag with a 20mm diamond shape mesh was attached to the gear mouth. This method, generally used in trawling selectivity experiments, allowed us to assess the proportion of damaged organisms that passed through the main net. To ensure the normal flow of water through the net, the cover bag was 1.5 times longer and wider than the primary bags (Gaspar and Monteiro 1999).

On hauling the net, catches were sorted by *taxa* and a damage score was attributed on board to each specimen caught. The nomenclature adopted was that of Sanches (1989). The extent and type of damage was recorded following Gaspar et al. (2001) arbitrary scale (Table 1). To evaluate the effect of dredging on the number of damaged individuals, scores 2, 3 and 4 were used. To quantify mortality it was assumed that animals assessed as damage score 3 and 4 would die, while individuals scored as 1 and 2 would survive.

Table 1 – Criteria used in the attribution of a damage score to each *taxon*.

Score	1	2	3	4	
Crustacea	Bivalvia	In good condition	Edge of shell chipped	Hinge broken	Crushed / dead
	Gastropoda	In good condition	Edge of shell chipped	Shell cracked or punctured	Crushed / dead
	Anomura	In good condition	Out of shell intact	Out of shell and damaged	Crushed / dead
	Brachyura	In good condition	Legs missing / small carapace cracks	Major cracks	Crushed / dead
	Natantia	In good condition			Dead
	Osteichthyes	In good condition	Small amount of scales	Large amount of scales missing/ small cuts or wound	Dead

Data analysis

To evaluate the effect of tooth spacing on the number of damaged (scores 2, 3 and 4) and dead animals (scores 3 and 4), the data obtained for the same tooth spacing were used independently from mesh size. Similarly, the effect of mesh size was studied by using hauls with the same mesh size independently of the tooth spacing. Significant differences between the effects of tooth spacing, mesh size and their interactive effects, on the proportion of damaged and dead animals, were tested using Two-way ANOVA (*F*-test). Prior to ANOVA, data were analysed to test normality (Anderson Darling test) and homogeneity of variance (Bartlett’s method) among treatments. Whenever these assumptions were not met, the non-parametric test of Kruskal-Wallis (K-W) ANOVA on RANKS was used. In situations where the null hypothesis was rejected, the multiple comparison test of Tukey was performed. Statistical analysis were conducted using the MINITAB software, with a significance level of $\alpha=0,05\%$

Results

During the experiments a total of 30,715 individuals were caught and 24 species were identified (Table 2): seven species of bivalves, seven species of fishes, six species of crustaceans, two species of cephalopods and one species of gastropods. The most abundant species in the catches were the hermit crab *Pagurus spp.* (44.3%) and the white clam *Spisula solida* (41.2%). Swimming and sandy crabs composed 4.6% of total catches and were mainly represented by *Polybius henslowi*, *Macropipus marmoreus* and *Leucarcinus vernalis*. All these species are described by Sobral (1989) as characteristic of this coastal area seabed.

Table 2 – List of species present in the catches

Class Crustacea	
Natantia	
	<i>Crangon crangon</i> (Linnaeus, 1758)
Brachyura	
	<i>Atelecyclus undecimdentatus</i> (Herbst, 1783)
	<i>Leucarcinus vernalis</i> (Risso, 1816)
	<i>Macropipus marmoreus</i> (Linnaeus, 1758)
	<i>Polybius henslowi</i> (Linnaeus, 1758)
Anomura	
	<i>Pagurus spp.</i>
Class Bivalvia	
	<i>Donax vittatus</i> (da Costa, 1778)
	<i>Donax semistriatus</i> (Poli, 1844)
	<i>Donax venustus</i> (Poli, 1795)
	<i>Glycymeris glycymeris</i> (Linnaeus, 1758)
	<i>Macra coralina</i> (Linnaeus, 1758)
	<i>Spisula solida</i> (Linnaeus, 1758)
	<i>Venus fasciata</i> (da costa, 1778)
Class Gastropoda	
	<i>Nassarius sp.</i>
Class Cephalopoda	
	<i>Sepia officinalis</i> (Linnaeus, 1758)
	<i>Sepiolo sp.</i>
Class: Osteichhyes	
	<i>Arnoglossus laterna</i> (Walbaum, 1792)
	<i>Dicologlossa cuneata</i> (Mareu, 1881)
	<i>Solea vulgaris</i> (Linnaeus, 1758)
	<i>Solea lascaris</i> (Linnaeus, 1758)
	<i>Trisopterus luscus</i> (Linnaeus, 1758)
	<i>Trachinus vipera</i> (Cuvier, 1829)
	<i>Trigla lucerna</i> (Linnaeu, 1758)

The mean number caught and the proportion of damaged and dead individuals observed for each group or species and for each combination of tooth spacing/mesh size is shown in Table 3. The effect of mesh size and tooth spacing on the number of damaged (scores 2, 3 and 4) and dead (scores 3 and 4) individuals was investigated for the target species *Spisula solida*, for crabs (Brachyura) and for the overall community. As far as *Spisula solida* is concerned, the statistical analysis carried out showed that mesh size had no effect either on the percentage of damaged (ANOVA, $F=1.50$; $P=0.250$) or dead individuals (K-W, $H = 0.81$; $df = 2$; $P = 0.667$). However, significant differences were observed when the effect of tooth spacing was tested both in terms of the percentage of damaged individuals (ANOVA, $F=5.60$; $P=0.013$) and percentage of dead individuals (K-W, $H=12.09$; $df = 2$; $P = 0.002$). Application of the Tukey test revealed the existence of significant differences in the proportion of damaged individuals between tooth spacing of 40mm and 60mm (Tukey, $P=0.0161$), and between tooth spacing of 20 and 40mm (Tukey, $P=0.0197$) in the case of the proportion of dead individuals. No significant interactions between the effects of the two factors in the percentages of damaged individuals (T-W ANOVA, $F=2.18$; $P=0.112$) and dead individuals (K-W, $H=14.0$; $DF=8$; $P=0.082$) were observed. However, if tooth spacing has an effect on the proportion of damaged individuals, it

was expected that tooth spacing of 20mm should damage more individuals than tooth spacing of 60mm or vice-versa, which was not observed. Therefore, the results obtained are probably related to sampling rather than the effect of tooth spacing on the catch.

In the case of crabs, the results of the Two-way ANOVA analysis revealed that the percentage of damaged specimens is not affected either by tooth spacing (T-W ANOVA, $F = 0.83$; $P = 0.452$), mesh size (T-W ANOVA, $F = 0.72$; $P = 0.500$) or interactions between these factors (T-W ANOVA, $F = 2.27$; $P = 0.102$). No significant differences were also found for the effect of mesh size (K-W, $H = 0.61$; $df = 2$; $P = 0.739$), tooth spacing (K-W, $H = 0.91$; $df = 2$; $P = 0.633$) and interactions between tooth spacing and mesh size in the percentage of dead crabs (K-W, $H = 10.49$; $df = 8$; $P = 0.232$).

For the overall catches non-significant differences were found in the effect of tooth spacing, mesh size and interactions between these two gear specifications in the percentage of damaged (Table 4) and dead individuals (K-W, mesh size - $H = 1.59$; $DF = 2$; $P = 0.451$; K-W, tooth spacing - $H = 3.71$; $df = 2$; $P = 0.156$; K-W, interaction - $H = 7.89$; $df = 8$; $P = 0.444$).

Table 3. Comparison of the mean number of individuals damaged (scores 2 to 4) and dead (scores 3 and 4) for each *taxon* and for each combination of tooth spacing/ mesh size experimented.

Species/Group	Mesh size 20m												Mesh size 40m												Mesh size 60m																				
	20m				40m				60m				20m				40m				60m				20m				40m				60m												
	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo	Total	Da	ma	Mo									
Crustacea																																													
Branchyura	8	1	8	0	4	22	3	14	2	9	23	6	24	5	20	27	3	10	1	5	9	2	19	1	8	58	17	29	3	5	135	34	25	15	11	93	6	7	1	1	100	7	7	1	1
Natantia	11	1	6	1	6	3	0	11	0	11	39	1	3	1	3	3	0	0	0	0	5	0	0	0	0	90	3	3	3	3	79	1	2	1	2	22	2	7	2	7	27	2	6	2	6
Anomura	362	3	1	0	0	166	3	2	0	0	270	7	2	0	0	638	2	0	0	0	284	5	2	0	0	569	0	0	0	0	942	1	0	0	0	849	0	0	0	0	456	0	0	0	0
Total	382	4	1	1	0	191	6	3	2	1	332	13	4	6	2	668	5	1	1	0	298	7	2	1	0	717	19	3	6	1	1156	37	3	17	1	964	8	1	3	0	583	8	1	3	0
Bivalvia																																													
<i>S. solida</i>	548	27	5	6	1	556	23	4	2	0	320	39	12	2	1	690	53	8	13	2	347	16	5	2	1	426	32	7	3	1	443	47	11	9	2	555	39	7	4	1	331	28	8	4	1
<i>D. vittatus</i>	5	0	6	0	6	2	0	0	0	0	4	0	0	0	0	8	0	0	0	0	1	0	0	0	0	20	0	0	0	0	53	0	1	0	1	22	0	0	0	0	7	0	0	0	0
<i>V. fasciata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	100	0	100	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0					
<i>D. semistriatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0					
<i>D. venustus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0					
<i>G. glycymeris</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	50	0	50	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0					
<i>M. corallina</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0					
Total	554	27	5	6	1	559	23	4	2	0	324	39	12	2	1	699	54	8	13	2	348	16	5	2	1	446	42	9	3	1	497	47	9	9	2	578	39	7	4	1	338	28	8	4	1
Gastropoda																																													
Nassarius sp.	41	0	0	0	0	41	0	0	0	0	17	0	0	0	0	99	0	0	0	0	28	0	0	0	0	45	0	0	0	0	64	0	0	0	0	118	0	0	0	0	70	0	0	0	0
Cephalopoda																																													
	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	40	1	40	1	0	0	0	0	1	0	0	0	0	1	1	100	1	100
Osteichthyes																																													
<i>T. luscus</i>	1	0	25	0	25	5	3	67	3	67	0	0	0	0	0	3	0	13	0	13	4	1	33	1	33	3	3	100	1	40	0	0	0	0	0	1	0	50	0	50	0	0	0	0	0
<i>T. vipera</i>	4	0	8	0	8	0	0	0	0	0	3	1	22	1	22	4	1	25	1	25	4	1	36	1	36	3	0	13	0	0	0	0	0	0	0	6	1	22	0	0	6	1	12	0	0
<i>T. lucerna</i>	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	50	1	50	0	0	0	0	0	1	0	0	0	0	2	0	0	0	0	1	1	100	0	0
Soles	1	0	25	0	25	1	0	33	0	33	3	1	20	0	0	1	1	67	0	33	1	0	33	0	33	4	0	8	0	8	1	0	0	0	0	3	0	10	0	0	1	0	33	0	0
Total	8	1	13	1	13	7	4	50	4	50	7	1	20	1	10	8	2	25	2	21	11	3	28	4	38	10	4	42	2	16	2	0	0	0	0	12	2	17	0	3	7	2	32	0	0
TOTAL	985	32	3	8	1	799	33	4	8	1	680	54	8	9	1	1474	61	4	16	1	685	26	4	7	1	1221	66	5	11	1	1719	84	5	26	2	1672	49	3	7	0	1000	40	4	8	1

Table 4. Split-plot ANOVA for the effects of mesh size and tooth spacing on the percentage of damaged individuals for overall macrobenthic community.

Source of Variation	DF	SS	MS	F	P
Mesh	2	0.000730	0.000365	1.000	
0.387					
Tooth spacing	2	0.00175	0.000874	2.395	
0.120					
Mesh x tooth spacing	4	0.00348	0.000870	2.383	
0.090					
Residual	18	0.00657	0.000365		
Total	26	0.0125	0.000482		

After establishing that tooth spacing and mesh sizes did not have any effect on the proportion of damaged and dead individuals, data were pooled and the mean number of damaged and dead animals was obtained for a standard 15 minute haul (Table 5).

The analysis of Table 5 shows that the mean percentage of both damaged (4.24%) and dead individuals (0.97%) of the overall catch is very low. However the animals vulnerability to dredging differs according to *taxa*. The most sensitive *taxa* to this kind of fishery were cephalopoda (42.3%), osteichthyes (18.18%) and brachyura (6.2%). The high mean mortality observed for cephalopods can be explained by the fragility of their structure and small size of *Sepiola spp.*, which are probably killed by the weight of catches on hauling or even during sorting. Among fishes, the pouting *Trisopterus luscus*, was the most affected species. Over 9.8% of all flatfish (*Dicologlossa cuneata*, *Solea vulgaris*, *S. lascaris* and *Arnoglossus laterna*) that were retained in the net bag had a large amount of scales missing and severe wounds. Approximately 5% of the crabs caught had missing legs, while 7% showed major carapace wounds. Within bivalvia, the most affected species were *Venus fasciata* (33.3%) and *Glycymeris glycymeris* (20.0%). Nevertheless, it is important to emphasise that the mean number of individuals caught for these species was very low. All the other bivalve species present in the catches *Donax vittatus*, *Donax semistriatus*, *Donax venustus* and *Mactra corallina* were highly resilient to the effects of dredging. The shell structure of the gastropods and hermit crabs provides an efficient shelter against the fishing process, hence 100% were in perfect condition or only slightly damaged. During the sorting operation it was observed that shrimps were either in perfect condition (score 1) or dead (score 4), which may be related to the time they have been retained in the net during hauling.

Table.5. Comparison of the mean number of individuals damaged (scores 2 to 4) and dead (scores 3 and 4) for each taxon.

Species/Group	Score	1	2	3	4	Total	Damage		Mortality	
							n°	(%)	n°	(%)
Crustacea										
Brachyura		44.22	5.3	0.22	3.07	52.8	8.59	16.27	3.29	6.23
Natantia		30	0	0	1.04	31.0	1.04	3.35	1.04	3.35
Anomura		501.85	2.26	0	0	504.1	2.26	0.45	0	0.00
Total		576.1	7.6	0.2	4.1	588.0	11.9	2.0	4.3	0.7
Bivalvia										
<i>Spisula solida</i>		434.7	28.85	2.26	2.67	468.5	33.78	7.21	4.93	1.05
<i>Donax vittatus</i>		13.59	0	0.04	0.04	13.7	0.08	0.59	0.08	0.59
<i>Venus fasciata</i>		0.07	0	0	0.04	0.1	0.04	36.36	0.04	36.36
<i>Donax semistratus</i>		0.04	0	0	0	0.0	0	0.00	0	0.00
<i>Donax venustus</i>		0.04	0	0	0	0.0	0	0.00	0	0.00
<i>Glycymeris glycymeris</i>		0.15	0	0	0.04	0.2	0.04	21.05	0.04	21.05
<i>Mactra corallina</i>		0.15	0	0	0	0.2	0	0.00	0	0.00
Total		448.74	28.85	2.3	2.78	482.7	33.93	7.03	5.08	1.05
Gastropoda										
Nassarius sp.		58.11	0	0	0	58.1	0	0.00	0	0.00
Cephalopoda										
		0.3	0	0	0.22	0.5	0.22	42.31	0.22	42.31
Osteichthyes										
<i>Trisopterus luscus</i>		0.89	0.22	0	0.78	1.9	1	52.91	0.78	41.27
<i>Trachinus vipera</i>		2.7	0.26	0	0.37	3.3	0.63	18.92	0.37	11.11
<i>Trigla lucerna</i>		0.63	0.07	0	0.11	0.8	0.18	22.22	0.11	13.58
Soles		1.52	0.19	0	0.19	1.9	0.38	20.00	0.19	10.00
Total		5.74	0.74	0	1.44	7.9	2.18	27.53	1.44	18.18
Total		1088.96	37.15	2.52	8.57	1137.2	48.24	4.24	11.09	0.98

Discussion

Experiments to assess the effect of mesh size and tooth spacing on the catch of *Spisula solida* were undertaken with the aim of determining an optimal combination of these two characteristics, that minimises the dredging impact on the most vulnerable macrobenthic species, incidentally caught by the dredge. Thus, in the present study, it was expected to reduce the number of injured animals by increasing or decreasing tooth spacing or mesh size. However, our data showed that tooth spacing, mesh size or interactions between these two factors does not have an effect on the number of damaged macrofaunal individual's caught. This may be related to the way this gear is operated, as observed by divers during the experimental phase of this work. The tooth bar of the northwestern dredge penetrates 10-15cm into the sediment, acting as a rake that pushes sand to the front of the frame mouth creating a "sand wave". As a result of this bulldozing effect the animals burrowed in the sediment enter the dredge without passing through the space between the teeth. In the case of the mesh, while the dredge is being towed, the mesh of the net bag closes as it is stretched due to the weight of the material in the bag, preventing escape of the individuals from the bag. Therefore, independently of mesh size, when the dredge is towed through the sediment the retained individuals are susceptible to be injured due to the abrasive action between animals and/or between animals and debris (empty shells) that roll inside the bag. Thus, the probability of the retained animals becoming injured augments with increasing tow duration (Gaspar et al. 1998), especially in the case of fishes (Main and Sangster 1990; Van Beek et al. 1990).

The nature of the bottom is another factor that may contribute to increase the mortality induced by mobile fishing gears on benthic species. In fact, several authors (Hall 1994; Currie and Parry 1996; Jennings and Kaiser 1998; Kaiser et al. 1998) stated that the impact of towed gears over the macrobenthic community is lower in mobile sandy sediments, than in rocky, muddy or dirty bottoms. On this kind of grounds the codend or net bag fills with mud or stones, which could enhance the probability of damaging individuals during fishing or sorting operations (Drinkwater 1974; Lart et al. 1997). Houghton et al. (1971), observed that in hauls performed on sandy grounds the extent of damage inflicted to invertebrate species varied with the quantity of empty shells caught with them. In the Portuguese dredge fishery, the nature of the bottom does not constitute a problem, since the exploited species only form extensive and dense beds on very clean sandy bottoms (Gaspar 1996a).

The severity of injuries inflicted by dredging on different macrobenthic species seems to be related to their fragility and physical. For instance, whelks (*Nassarius sp.*) and hermit crabs (*Pagurus spp.*) were highly resistant to the effects of the entrapment in the net bag. These species are protected by a strong shell that provides an efficient protection against fishing operations. Low mortalities were also observed for the crabs *Atelecyclus undecimdentatus*, *Leucarcinus vernalis*, *Macropipus marmoreus* and *Polybius henslowi*, which was similar to the findings of Kaiser and Spencer (1995) for other crab species. A high proportion of the clam *Spisula solida* and donax clam *Donax vittatus*, the most abundant bivalve species in the area where this study was conducted, were undamaged, indicating that these species are well protected by their thick shells. In the present study, Cephalopoda and Osteichthyes were the most affected taxa. The high mortalities found for *Sepiolo spp.* seem to be related to their small size and soft structure. These animals were probably killed by the weight of catches on hauling the net or during sorting operations. De Groot and Apeldoorn (1971) had already observed the cephalopods sensitivity during the fishing process. Among fishes, the most vulnerable species was *Trisopterus luscus*. It is however important to remark that the occurrence in number of these two groups in the catches was very low. Our results are in accordance with Hall-Spencer et al. (1999a) and Giovanardi et al. (1998) for bivalves and gastropods. However, in the case of badly damaged and dead individuals for crabs and cephalopods, we found out a significantly smaller impact than those authors. For commercial beam trawls, de Groot and Lindeboom (1994) reported mortalities up to 50% for most crabs and molluscs. Despite the apparent robustness of bivalves, Lindeboom and de Groot (1998) found, for the North Sea, that some species (e.g. *Gari fervensis*; *Dosinia lupinus*; *Tellina fabula*) were susceptible to suffer mortalities up to 40%.

Nevertheless, the majority of animals present in the catches were very resistant to the fishing process, certainly explaining the low number of severely damaged and dead individuals (1%) found in this study for the overall macrobenthic community. In the present study, it was estimated that 4.24% of the discarded species were unlikely to survive due to the degree of damage suffered. In the northwestern fishery, the by-catch is discarded immediately after sorting, which is an important stage for the organisms' survival, since exposure to air inevitably causes stress and mortality if sorting times are long and conditions on deck unfavourable (Medcof and Bourne 1964b; Gaspar and Monteiro 1999). According to McLoughlin et al. (1991) damage is directly correlated with dredge efficiency. In the present work we did not gather data on the northwestern dredge efficiency, however we believe that the efficiency of this

gear is relatively low. In fact, *in situ* and video observations of clam dredging showed that shortly after the start of the tow, a sand buffer is formed in front of the gear mouth. This sand buffer pushes sediment sideways and above the dredge, limiting the amount of material that enters the bag. Therefore, it is likely that part of the animals that have been in contact with the dredge but that were not caught, may become severely damaged dying immediately or become susceptible to predation dying subsequently.

Taking into consideration the species vulnerability, the fishing strategy used by the local dredge fleet and the results of the bivalve surveys carried out periodically by IPIMAR since 1986, we can speculate about the long-term effects of this kind of fishery over the macrobenthic community. Due to the rough sea conditions observed all year round, the northwestern dredge fleet only operates during 5-6 months per year. This means that the fishing effort is distributed both spatially and seasonally, preventing the submission of benthic fauna to a constant effect of dredging. Moreover, the fishing strategy adopted by this fleet tends to concentrate fishing effort during a short period of time on a certain white clam bed, until the catch rates drop below economical acceptable levels, after which the clam bed remains unfished for periods that can reach 1-2 years. This fact leads to a highly patchy distribution of fishing effort and so we cannot talk about continuous and cumulative fishing effects for a specific white clam bed and associated community. The immediate effect of the fishing process is the reduction of the target species abundance. However, the fishing process also inevitably damages other macrobenthic species present in the area. Therefore, a decrease in abundance of the most vulnerable non-target species is also expected. It is interesting to emphasise that the same species found in the present work have been recorded in the bivalve surveys carried out since 1986, which indicates that during this period the macrobenthic species present in the area where this study was undertaken remained unchanged. Although some changes in abundance occurred, we do not know if this changes were due to the fishing process, due to natural causes or a result of the combination of both factors.

Nevertheless, we believe that the impact of this type of fishery upon the macrobenthic community could be minimised if a more efficient, and, at the same time, more selective dredge can be developed in order to reduce both the number of non-target individuals in the catch and to allow the rapid escape of the individuals during the tow.

3.1.4.4 EVALUATION OF SHELL DAMAGE IN THE CLAM *CHAMELEA GALLINA* CAPTURED BY HYDRAULIC DREDGING IN THE NORTHERN ADRIATIC SEA

Introduction

The impact of various types of fishing gears used to harvest molluscs has been assessed in several geographic regions (Hall et al. 1990; Collie et al. 1997; Currie and Parry 1996; Kaiser et al. 1996; Kaiser et al. 1998), including the North Adriatic Sea and the nearby Lagoon of Venice (Hall-Spencer et al. 1999b; Pranovi and Giovanardi 1994). Most of these studies focused on evaluating the effects of fishing on non-target species and ecosystem structure. Research studies on commercial target species have been mainly concerned with gear efficiency or stock assessment (Dare et al. 1994; Dare and Palmer 1994; Fifas and Berthou 1999). Recently, the need for an estimation of the effects of fishing activities on target species was highlighted in order to determine population sustainability and to establish adequate criteria for the management of the resource. Particular attention has been paid to the evaluation of the effects of dredging and sorting on bivalve molluscs subjected to commercial fishing. The impact of fishing gears must be considered especially harmful to undersized animals which are first dredged and sieved and then rejected into the seabed: discarded animals may be disturbed or suffer physical damages, with a consequent reduction in survival capability and an increase of the risk of predation. Simulated fishing disturbance caused a delay in the reburrowing response of the cockle *Cerastoderma edule* (Coffen-Smout and Rees 1999) and a reduction in the ability to perform an escape response in the whelk *Buccinum undatum* (Ramsay and Kaiser 1998). Moreover, Jenkins and Brand (2001 see also [Section 3.1.3.1](#)) demonstrated a decrease in the ability of swimming and escape from predators in captured undersized great scallop, *Pecten maximus*.

Dare (1974) found that up to 13% of mussels (*Mytilus edulis*) which passed through a rotary sorting machine underwent shell damage and many of them apparently suffered some internal damage which impaired their long-term survival out of water.

Gaspar et al. (1994) detected the presence of a series of shell margin breaks in the razor clam *Ensis siliqua* collected from a heavily dredged area. Authors suggested that these disturbances to shell growth were the results of repeated dredge damage.

Since the 70's, the exploitation of *Chamelea gallina* beds of the North Adriatic Sea considerably increased due to the improvements in fishing technology and the introduction of hydraulic dredging and mechanised sorting. The intense fishing effort on the resource and the specific features of the technique used for harvesting clams resulted in a general weakness of the *C. gallina* population and then in a dramatic reduction of fishable biomass (Del Piero and Fornaroli 1998; Del Piero et al. 1998). In the present work, an approach to evaluate mechanical stress due to dredging and sorting was attempted by detecting and quantifying shell damage in captured, discarded and undersized *C. gallina*.

Methodologies

With the aim to evaluate damage levels on clam shells caused by different fishing systems, sampling on natural *C. gallina* grounds was carried out in February, May, July and October 2000.

Clam samples were collected in two fishing areas along the west coast of the North Adriatic Sea, Jesolo and Lido, at about 5 m depth, using four fishing methodologies:

- dredging at high water pressure (~ 2.5 atm) and using a mechanical sieve for sorting (as in commercial fishing, HPS samples);
- dredging at high water pressure without sorting (HP samples);
- dredging at low water pressure (~ 1 atm) without sorting (LP samples);
- manual SCUBA collecting (M samples).

Damage levels were also detected on by-catch clam samples (BC) in May, July and October and on under-sized clams (US, 13-17 mm in length) in July and October. Moreover, in all samples shell lengths were measured. Damage caused by fishing activity was detected on clam shells (Figure 1), and six damage categories were scored (from 1 to 4) according to the following table:

<i>General conditions</i>	<i>Categories</i>	<i>Scores</i>
	Healthy	1
<u>Good</u>	Shell with previous repaired damage	1
	One scratched valve	1.5
<u>Damaged</u>	Both scratched valves	2
	Edge of shell chipped	2.5
<u>Crushed</u>	Crushed umbo	3
	Crushed shell	4

As organisms showing crushed umbo or crushed shell can be considered unable to recover, the percentage of crushed clams was used to quantify mortality caused by dredging.

Statistical comparisons were performed using Anova test.

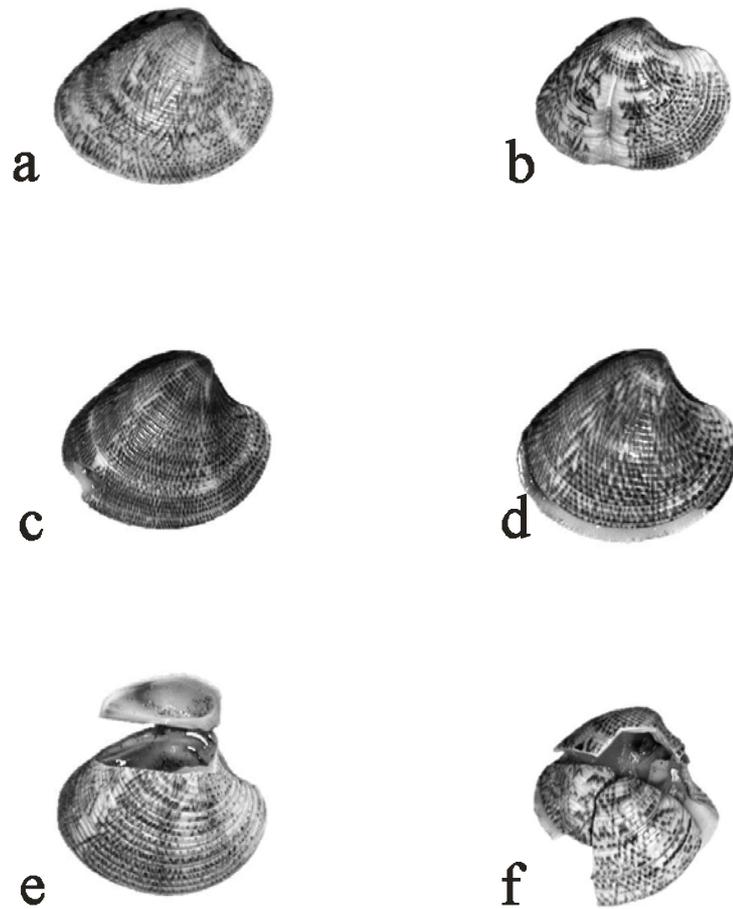


Figure 1- Various damage types detected in *C. gallina*:

- a) healthy clam;**
- b) shell with previous repaired damage;**
- c) shell having both scratched valves;**
- d) edge of shell chipped;**
- e) crushed umbo;**
- f) crushed shell.**

Results

At Lido, no significant differences were generally observed between LP and HP clams, whereas in HPS samples damage was significantly higher, when compared to HP value in February ($p < 0.001$), May ($p < 0.001$) and July ($p < 0.01$).

At Jesolo a different situation was observed: generally LP samples were significantly less damaged with respect to HP samples, which were significantly different from HPS samples only in May ($p < 0.01$). Manually collected clams showed the lowest damage levels and frequently the values were close to 1 (healthy clams) (Figure 2).

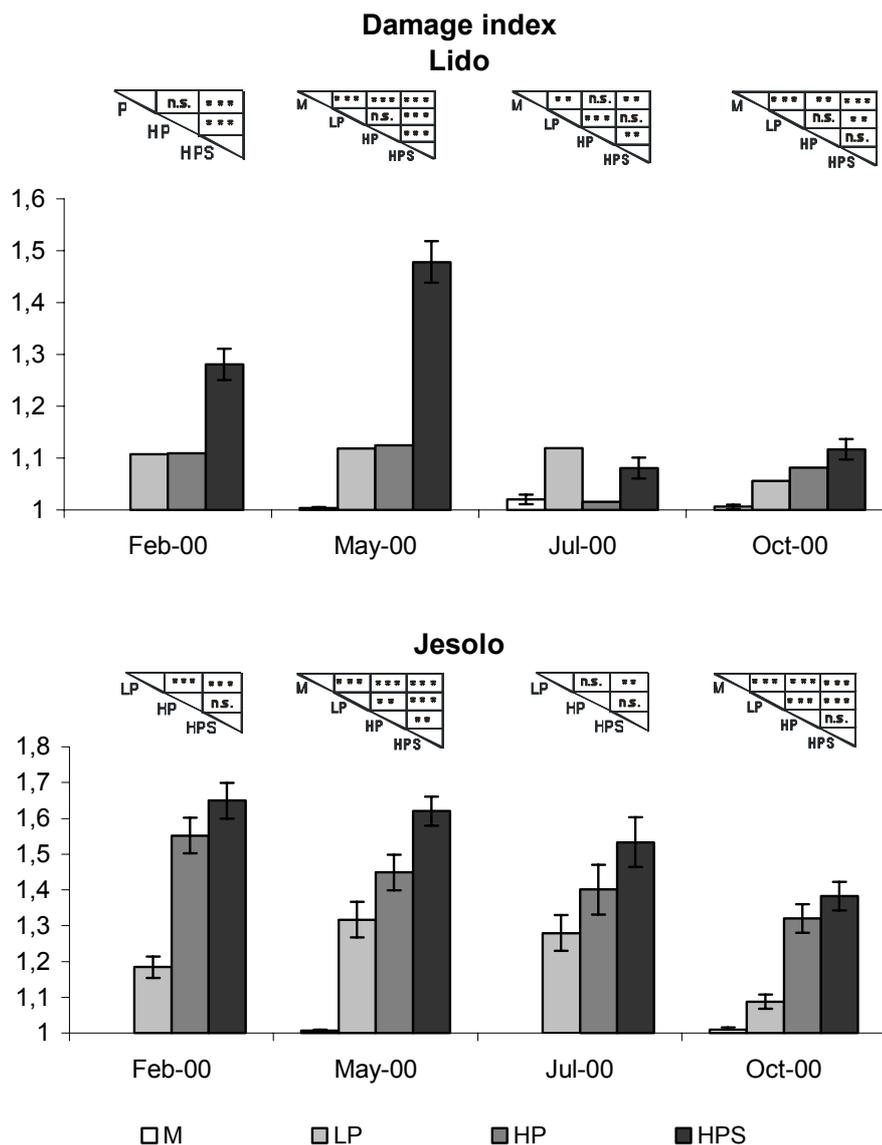


Figure 2 – Damage levels in *C. gallina* collected at Lido and Jesolo using different fishing systems (M=manual; LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.e.; Anova: : * $p < 0.05$; ** $p < 0.01$; * $p < 0.001$.**

The seasonal comparison exhibited no differences among LP samples at Lido, whereas HPS samples showed more variability and the highest damage value was observed in May. At Jesolo the lowest value of damage index in HPS clams was observed in October, being significantly different in comparison with February and May HPS values ($p < 0.001$).

The percentage of damaged clams collected in the two sampling sites during the year and those of damaged clams collected from by catch are shown in Table 1 and Table 2, respectively. Higher amounts of damaged animals were found in May, both at Lido and Jesolo, reaching the maximum value of 35% at Lido, even if in the other months higher percentages were observed at Jesolo. As concerns samples from the by-catch, Lido showed higher percentages of damaged clams: 26.8% in May and 22.6% in February.

<i>Lido</i>	<i>Feb-00</i>	<i>May-00</i>	<i>Jul-00</i>	<i>Oct-00</i>
LP	7.8%	8.8%	9.4%	7.2%
HP	11.1%	11%	3.6%	4.1%
HPS	20.5%	35%	9.6%	8.9%
<i>Jesolo</i>				
LP	12.9%	18.4%	12.5%	13.9%
HP	28.3%	24%	15.3%	8.5%
HPS	31.9	32.6%	29.3%	19.4%

Table 1 - Percentages of damaged captured clams in the two studied sites.

	<i>Lido</i>	<i>Jesolo</i>
May-00	26.8%	11.6%
Jul-00	13.2%	9.4%
Oct-00	22.6%	12.4%

Table 2 - Percentages of damaged discarded clams collected from by-catch.

The lowest mortality was found in HP samples at Lido and in LP samples at Jesolo: significant differences were generally observed when comparing these samples with HPS ones. Highest mortality was generally detected in HPS clams in both sites. Comparing Lido and Jesolo, mortality was significantly higher at Jesolo ($p < 0.001$), reaching 19.6% in February, whereas at Lido the maximum value, 7.8%, was found in May (Figure 3).

At Lido, the lowest value of shell length was detected in LP samples in February and October and in HP samples in May and July, whereas at Jesolo LP clams were always significantly smaller (Figure 4).

Under-sized clams generally showed lowest damage levels with respect to by catch and HPS samples. Significant differences were reported comparing all samples, except those in July (at Lido, only) (Figure 5).

Mortality

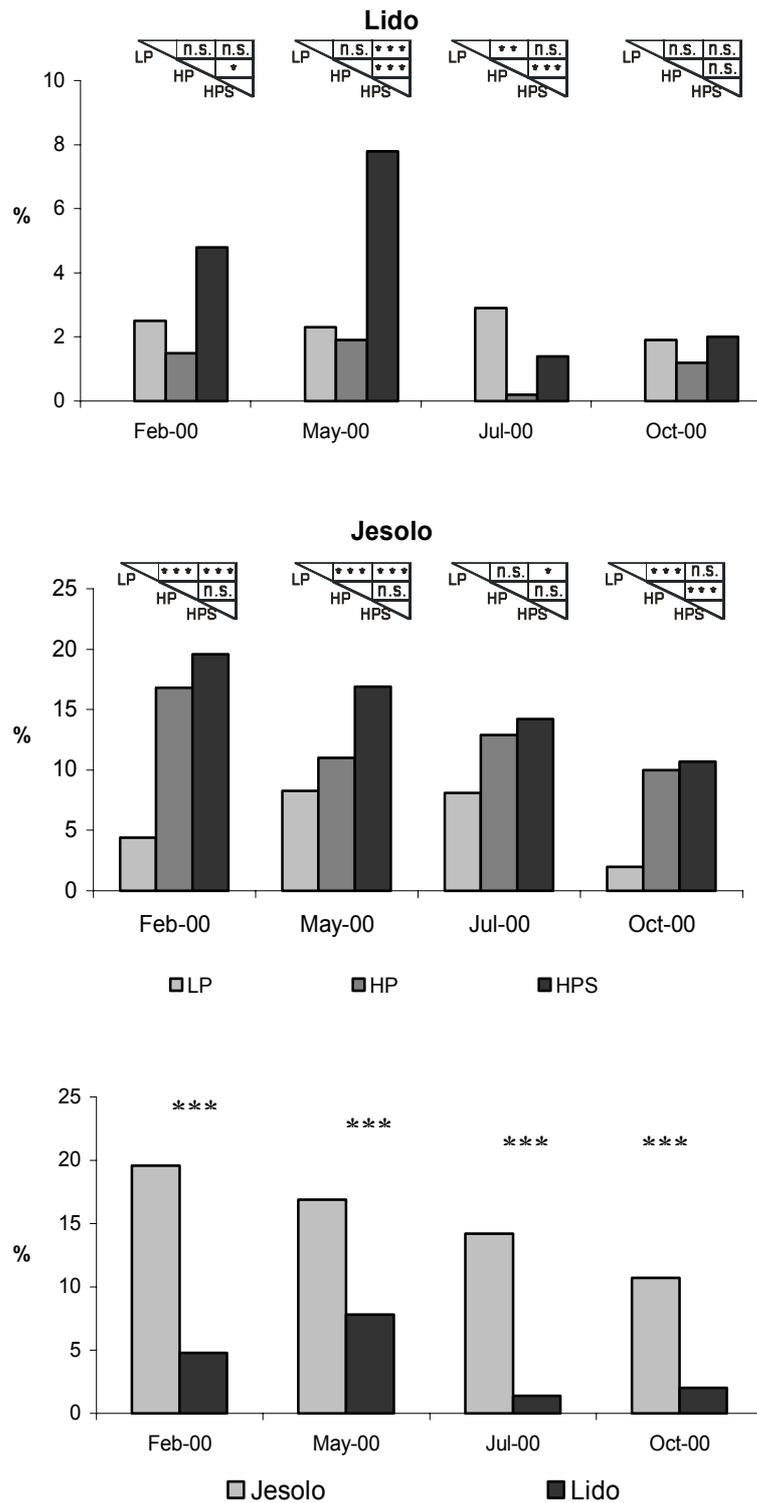


Figure 3 - Mortality values in *C. gallina* collected at Lido and Jesolo using different fishing systems (LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter), and comparing the HPS samples of the two sites.

Shell length

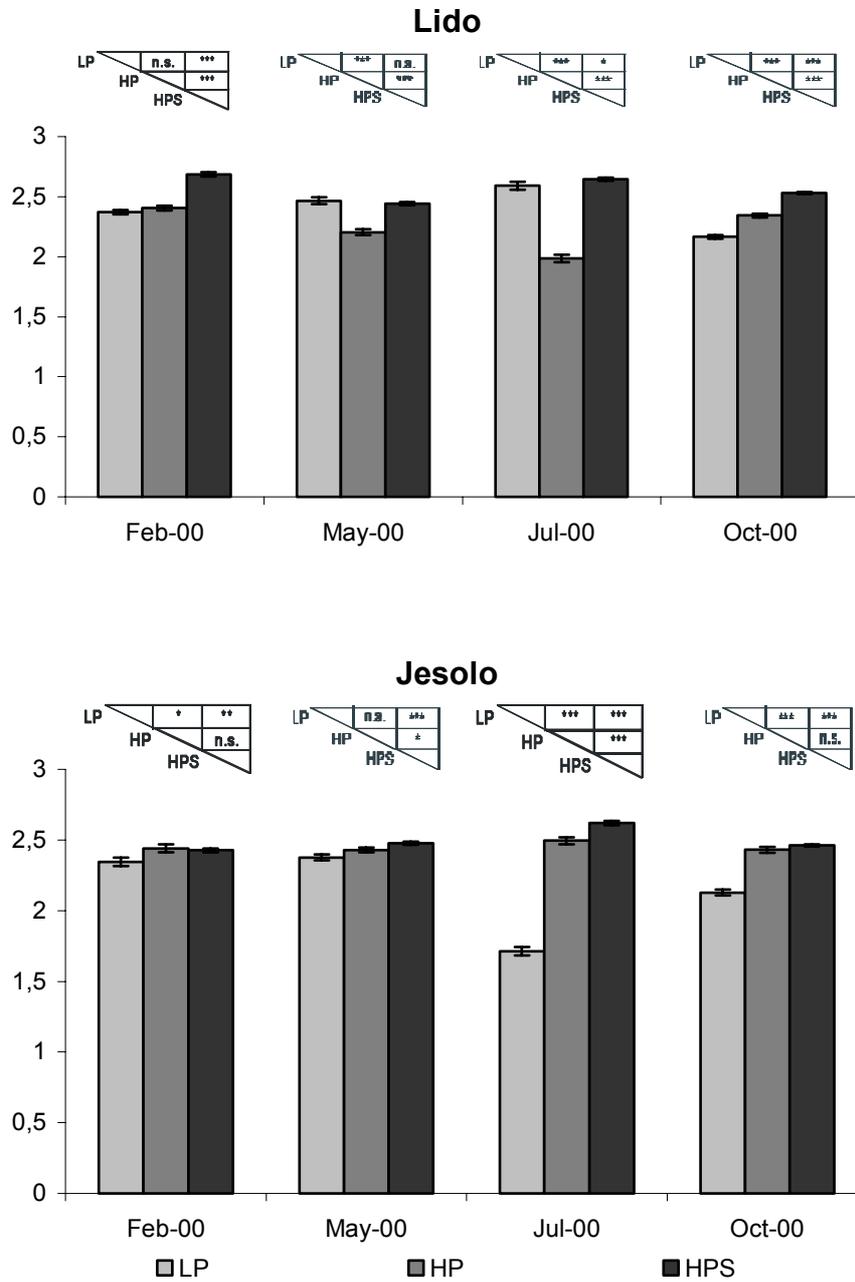


Figure 4 - Shell length in *C. gallina* collected at Lido and Jesolo using different fishing systems (LP= low pressure; HP= high pressure; HPS= high pressure and mechanical sorter). Mean \pm s.e.; Anova: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

Damage levels on undersized clams

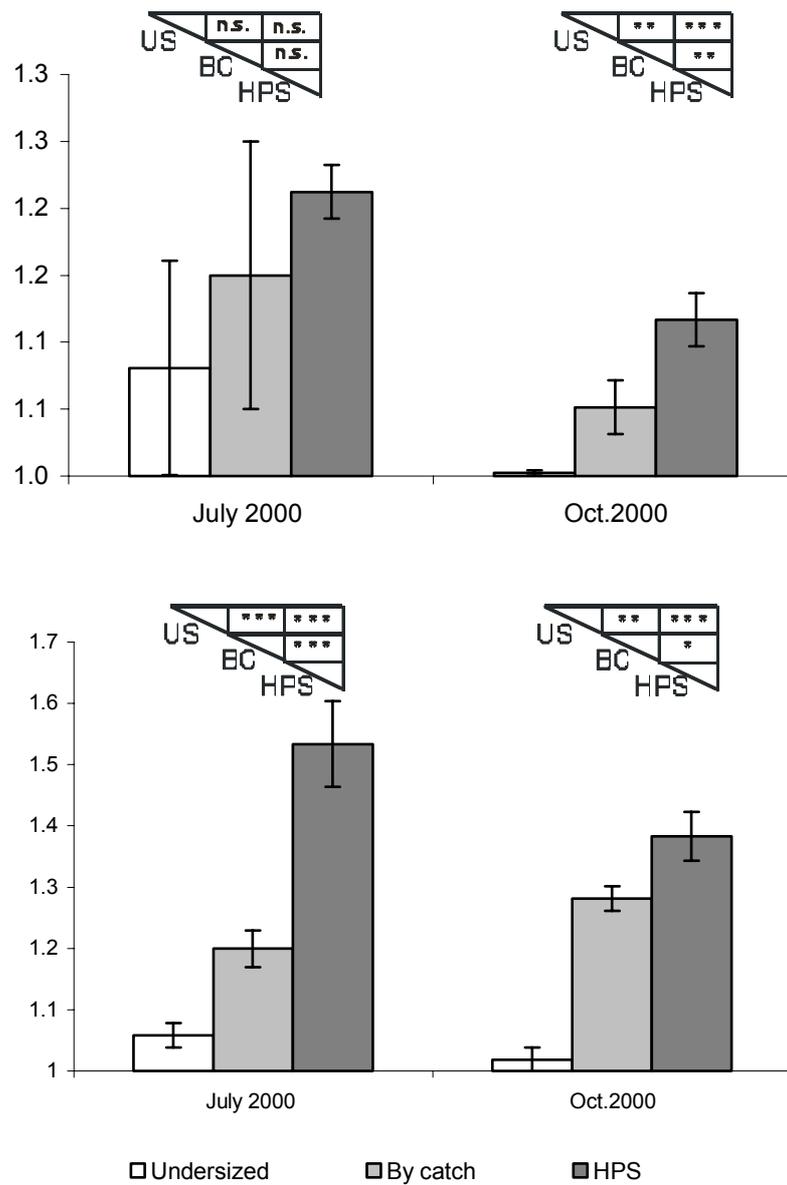


Figure 5 – Damage levels of commercial size (HPS), under-sized (US) and by-catch (BC) clams *C. gallina* collected using high water pressure and mechanical sorter at Lido and Jesolo. Mean \pm s.e.; Anova: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

Discussion

The overfishing which *Chamelea gallina* populations of the North Adriatic Sea have undergone since the introduction of the hydraulic dredges, has caused a dramatic reduction in fishable clam biomass (Del Piero and Fornaroli 1998; Del Piero et al. 1998), mainly due to the efficiency of the gear used to harvest clams and to the great number of vessels employed. Therefore, evaluation of the capability of clam populations to sustain fishing effort needs to be better estimated.

In this study, the impact of experimental dredging was assessed on clam populations in two different sampling sites along the North Adriatic coast by quantifying shell damage caused by fishing operations. In order to distinguish the effects of high water pressure and mechanised sieving, different levels of stress were applied, the highest being that used by the commercial fishing vessels and the lowest the manual sampling of clams by scuba divers.

The use of high water pressure and mechanised sorting increased the physical impact on clams, the highest levels being always detected in commercially dredged clams. These results confirmed the observations of Dare (1974) on dredged and sorted mussels.

Some differences were pointed out comparing the data from the two sampling sites, even if mechanical stress applied was similar: at Lido, damage to clam shells seemed to be mostly due to the action of mechanical sorter rather than to water pressure, as shown by significant differences between HP and HPS samples; at Jesolo the effect of high water pressure was more distinguishable, HP samples being particularly damaged with respect to LP ones.

At Lido, the lowest mortality and mean shell length values were often observed in HP samples, whereas at Jesolo an increase of both parameters was detected when physical impact increased. The relevant amount of empty broken shells in Lido sediments determined a tamping of the catch inside the gear which was particularly strong when high water pressure was used. This condition reduced clam shaking, and consequently the possible shell damage, as well as size selectivity. According to this hypothesis, mortality was always lower at Lido when comparing the two sampling sites.

Damage levels proportionally increased with the increasing of clams size: under-sized samples (13-17 mm) were less damaged than by-catch (18-24 mm); moreover, HPS samples (>25 mm) showed the highest damage levels.

Relevant percentages of damaged individuals were found in both captured and discarded clams, highlighting a strong and harmful impact of this type of fishing gear. More than 30% of commercially fished clams showed shell damage in the sample collected at Lido in May and in those from Jesolo in February, May and July. Clams collected at Jesolo exhibited not only higher mean damage level, but also higher number of damaged individuals with respect to Lido samples.

Finally, useful information can be given from by-catch samples, considering that all under-sized clams captured by fishing gear, sieved and then rejected into the seabed, contribute to the restocking of the natural populations. In both sampling sites, a

considerable fraction of discarded clams suffered shell damage reaching 27% at Lido and 12,5% at Jesolo: these damaged under-sized clams will probably have reduced survival capability. Actually it has been demonstrated that captured and discarded animals, both target or non-target, can be generally subjected to high levels of predation and mortality (Kaiser and Spencer 1995; McLoughlin et al. 1991). Although underwater observations in the studied areas did not reveal high predation pressure, the majority of predators being represented by crabs, the low percentage of repaired shells detected (from 1.2% to 6.7%) indicate that only a small fraction of damaged discarded clams are effectively able to recover.

3.1.4.5 SUMMARY STATEMENT

The physical damage caused by dredge fisheries to both target and non-target organisms was demonstrated in clam fisheries in Portugal and Italy and in scallop fisheries in the UK. Damage indices were used in a variety of different studies to understand the effects of dredging on benthic epifauna.

It is clear that not all dredge fisheries have the same damaging effect on benthic organisms. For example hydraulic dredging (with associated mechanical sorting) in the Adriatic Sea clearly causes greater damage to the target clam species than the simple clam dredges used in Portugal. Up to 30% of discarded *Chamelea gallina* captured using hydraulic dredges were damaged. In the Portuguese clam fishery only 7% of captured bivalves were damaged. In general in this fishery overall damage levels were relatively low with only 4% of individuals of all species showing signs of external damage. However as in previous studies different species showed vastly differing sensitivity. For example high levels of damage occurred in Cephalopoda and Osteichthyes with 42% and 18 % respectively of individuals showing damage.

Work on the scallop dredge fishery of the Irish Sea showed a range of damage levels in large epibenthos with some species appearing to exhibit virtually no signs of physical damage. Importantly the edible crab *Cancer pagurus* was one of the most sensitive species indicating the potential negative interaction of dredge fisheries with pot fishing for crabs. This study also revealed that the use of bycatch as the sole means of estimating damage to megabenthos is inappropriate. A combination of inefficient gear and high damage levels of individuals that encounter gear, but are not caught, means that the majority of damaged organisms remain on the sea bed.

In scallop dredge fisheries the target species *Pecten maximus* generally shows relatively low levels of damage following dredging. This has led to investigations of the stress experienced by scallops and potential modification of their behaviour (see previous section) as an additional way (beyond direct mortality caused by physical damage) in which discarded scallops may show high mortality. The work described in this section on mortality levels in stressed and damaged scallops showed no effect of either simulated dredging or simulated damage on mortality levels. However further work is required in this area.

Modification of gear in Portugal and modification of dredging method in Italy were tested to assess whether damage to benthic organisms could be reduced. The Italian study was made from the perspective of damage to undersized discards while the study in Portugal assessed damage to bycatch in general. In hydraulic dredging in Italy where damage rates were relatively high, it was possible to obtain a reduction in damage through a reduction in pressure and replacing mechanical sorting with sorting by hand. In contrast, in the Portuguese clam fishery there was no difference in the number of damaged individuals captured using gear with varying mesh size and tooth spacing. Knowledge of gear behaviour gained throughout this project may be used to explain such differences between different dredge fisheries. It may also be used in the future to determine the types of gear modification likely to result in a reduction in environmental damage.

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3.1.5. Biological Community

Introduction

The six contributions in this Section of the report are concerned with impacts of shellfish dredging at the biological community level. The first ([Section 3.1.5.1](#)) describes experiments to study the aggregation of predators and scavengers to undersized discards in the Irish Sea scallop dredge fishery, paying particular attention to the importance of damage level. The next four sections investigate different aspects of clam dredging impacts in southern Portugal. The first of these ([Section 3.1.5.2](#)) analyses spatial differences in the macrofauna along and across sections of dredge track, including comparisons between fished and recently non-fished areas. This is followed by a diving-based study ([Section 3.1.5.3](#)) of the new Portuguese clam dredge to assess the effect of dredging on the bottom substrate, to estimate the degree of damage caused to the macrobenthos and to study scavenger aggregation within the dredge tracks. The third Portuguese study ([Section 3.1.5.4](#)) assesses the effects of an intense dredging episode on meiofauna and macrofauna community structure and investigates the time required for recovery. [Section 3.1.5.5](#) compares the benthic community in a continuously fished area with that of an area that had not been fished for 4 years; it discusses differences in macrofauna and meiofauna diversity, abundance and biomass, as well as the relative importance of different trophic groups. The final contribution ([Section 3.1.5.6](#)) is a seasonal investigation of the effects of hydraulic clam dredging on the composition and structure of macrobenthic communities in the northern Adriatic, including assessments of damage sustained by organisms on the seabed and in the by-catch. Together these six contributions cover a range of community level impacts for three European bivalve fisheries, in different geographical regions, using different types of dredges.

3.1.5.1 PREDATOR AND SCAVENGER AGGREGATION TO DISCARDED BY-CATCH FROM A SCALLOP DREDGE FISHERY: IMPORTANCE OF DAMAGE LEVEL

Introduction

Dead and damaged organisms left in the track of demersal fishing gear or discarded as bycatch attract a wide range of mobile scavengers and predators (Medcof and Bourne 1964b; Caddy 1973; Kaiser and Spencer 1994). The movement of scavengers towards dead and damaged biota is probably caused by chemical stimulants released from damaged tissue (Lapointe and Sainte-Marie 1992; Zimmer-Faust 1993). Undamaged but stressed animals, especially those discarded and found on the seabed in high densities, may also attract predators. Numerous studies have reported aggregations of predators on high density patches of scallops produced in aquaculture seeding operations (Caddy 1988; Minchin 1991; Barbeau and Caswell 1999). There has been little study into the effects of varying stress or damage levels on predator aggregation.

We aimed to determine the rate and magnitude of predator/ scavenger aggregation to simulated discarded by-catch with varying degrees of stress and damage. Most experimental studies of predator aggregation have used either dead (Ramsay et al. 1997a; Ramsay et al. 1997b) or very badly damaged (Veale et al. 2000a) bait to produce the maximum attractive effect. In many fisheries, for example pelagic trawls, the majority of discarded material is dead. However, in demersal fisheries where a large part of the catch is made up of benthic invertebrates, mortality levels are lower and discards show varying levels of damage which may not lead directly to death (Kaiser and Spencer 1995). A mono-specific bait of *Pecten maximus*, the target species of the dredge fishery, was used with three levels of damage, from none to high. All three treatments underwent simulated dredging prior to deployment. *Pecten maximus* was used as bait for two reasons: firstly, undersized *Pecten maximus* represent a large proportion of the total discarded material in scallop dredge fisheries. Secondly, the fate of undersized discards is of obvious concern to the fishery itself. We hypothesised that all three types of bait would attract scavengers and predators but that the rate and magnitude of aggregation would increase with the degree of damage.

Methodologies

The aggregation of predators and scavengers to bait on a scallop fishing ground in the north Irish Sea was assessed using a fixed underwater video camera between August 1st and September 26th 2000. A Rovtech Systems low-light colour camera with two 250W red filtered lights was used and the signal transmitted to shore via a 1000m cable. The image could be viewed real time on a colour monitor and was recorded on a Panasonic SVHS time-lapse video recorder. The camera was positioned on a Dexian frame at an angle of 45°, 1m above the seabed such that an area 1m wide could be viewed. The depth of field varied depending on underwater visibility but was generally at least 3m. The camera was located approximately 600m offshore, due west of Port Erin Marine Laboratory in a depth of water 20m below LAT. The area of study was on a typical scallop fishing ground, but within an area closed to fishing since 1989. On three occasions throughout the period of study, the camera was

moved approximately 50 metres to limit the permanent association of organisms with the camera frame.

Over the period of study divers baited the camera on nine separate occasions. The bait consisted of 12 scallops (*Pecten maximus*) loosely tethered using monofilament fishing line passed through small holes drilled in the shell ear. The fishing line was attached to a peg secured in the substratum in front of the camera. The scallops used as bait were subjected to three different treatments. Immediately prior to deployment in front of the camera, all scallops were subjected to simulated dredging in which they were agitated in seawater with sand and rocks for 40 minutes (see Jenkins et al for details). The scallops were then either left undamaged, or subjected to low or high levels of damage. For low levels of damage a screwdriver was used to puncture the shell and remove fragments from the outer shell margin up to the mantle edge. In this way access was provided to internal body parts for predators but no soft tissue was damaged. For high levels of damage a hammer was used to break the shell in a number of places and cause damage to soft tissue. Each of the three treatments was replicated three times. For each replicate, the events following baiting were recorded for between four and seven days. In order to maintain independence between treatments, each replicate baiting event was either separated by a period of at least three days or the position of the camera was moved. In addition to the experimental treatments, three periods of at least four days were recorded with no bait.

Data analysis

Videotapes of the recorded periods were viewed on the time-lapse video recorder. Using the on-screen data and time display, instantaneous counts of all organisms were made every 10 minutes of real time. Because of the difficulty of determining whether certain organisms were feeding on the bait, counts were made of all organisms within the field of view. However, notes were made when organisms were clearly feeding. For graphical display, the average number of animals seen at six instantaneous counts was calculated to give a mean hourly abundance.

Results

Technical problems resulted in insufficient recording for two baiting periods (one low damage level and one high damage level). Thus data for only two replicates of each of the damage treatments are presented.

General observations

Lightly damaged scallops were not available as a food supply to many large organisms including most fish, although dogfish were seen attempting to feed. This food was only made available by the action of large crustaceans, mainly *Cancer pagurus*, but also *Liocarcinus* species, using cracks in the shell to break into the animal. On many occasions, plaice (*Pleuronectes platessa*) were seen waiting, and feeding at the chelae of crabs as they cut into soft tissue. Also the brittlestar, *Ophiocomina nigra*, although frequently dispersed during periods of intense activity by crabs and large fish, was attracted toward damaged scallops when *Cancer pagurus* was feeding. Highly damaged scallops provided a readily accessible, highly concentrated food source. This resulted in periods of intense feeding activity by large fish and crustaceans, which disturbed the sediment and resulted in the dispersion of smaller organisms, for example dragonets and brittlestars from the immediate area.

At times up to 12 plaice were feeding at once; at others a mixed assemblage of large gadoids, flatfish and *Cancer pagurus* occurred. When more than one edible crab was feeding there was frequent competition between individuals for food. On a number of occasions crabs attempted to remove scallops from the immediate area. Although no undamaged scallops were eaten, *Cancer pagurus* was seen, on a few occasions, attempting to open shells by exploring the ventral shell margins with its chelae. However, such attempts rarely lasted more than a few minutes.

Table 1. Percentage of instantaneous counts in which each of the taxa were recorded.

		Baited periods		Non baited periods	
		%	Rank	%	Rank
Echinodermata	<i>Ophiocomina nigra</i>	99.3	1	99.2	1
	<i>Astropecten irregularis</i>	0.3	13		
	<i>Asterias rubens</i>	0.1	14		
	<i>Marthasterias glacialis</i>	1.2	9		
	<i>Luidia ciliaris</i>	0.6	12		
Crustacea	Spider crabs	11.2	3	2.5	3
	<i>Cancer pagurus</i>	7.7	4	0.6	7
	<i>Liocarcinus</i> spp	3.4	7	1.1	4
	<i>Pagurus</i> spp.	0.7	11	0.8	6
	Lobster	0.1	18		
Pisces	<i>Callionymus lyra</i>	30.6	2	20.3	2
	<i>Scyliorhinus canicula</i>	2.7	8		
	Gadoids	5.2	6	1.1	5
	<i>Aspitrigla cuculus</i>	0.7	10		
	<i>Synagnathus</i> spp.	0.1	17		
	Flatfish	7.6	5	0.3	9
Mollusca	<i>Eledone cirrhosa</i>	0.1	15	0.6	8
	<i>Loligo</i> spp	0.1	16		
Mammalia	<i>Halichoerus grypsus</i> (Grey seal)	0.1	19		

Predator and scavenger activity

Nineteen taxa were identified over the period of study. However, the majority of these were only occasional visitors to the baited area (Table 1). For example, only 6 taxa appeared in more than 5% of counts at bait. The brittlestar *Ophiocomina nigra* and the demersal fish *Callionymus lyra* (dragonet) were locally very abundant, appearing in 99.3% and 30.6% respectively of instantaneous counts over all baited periods. Other common taxa present at the bait were the edible crab *Cancer pagurus*, unidentified spider crabs, large gadoids (probably cod and haddock) and flatfish, predominantly the plaice *Pleuronectes platessa*. Predatory starfish, *Asterias rubens*, *Astropecten irregularis*, *Luidia ciliaris* and *Marthasterias glacialis* were all observed rarely at the bait. During unbaited periods the brittlestar and dragonet, which were ubiquitous in the study area, were seen frequently. However, all other species were either not present or seen only rarely (Table 1).

The times of first appearance of the common taxa at the bait are shown in Figure 1. Locally very common species, such as *Ophiocomina nigra* and *Callionymus lyra* were either already present at the baited site or appeared very shortly after baiting for all damage grades. Large variability between replicate baiting events and the low number of replicates make generalisations regarding other taxa difficult. However, there was a trend toward relatively rapid aggregation to damaged bait, compared to slower aggregation to undamaged bait. Flatfish appeared consistently early at damaged bait, under 3 hours for all replicates, whereas *Cancer pagurus*, although showing a similar trend, exhibited more variability, appearing between 0.5 and 12 hours at the damaged treatments. A notable exception to the trend described above was spider crabs, which showed the slowest rate of aggregation to the highly damaged treatment.

The overall level of scavenger and predator activity for each treatment was calculated as the mean hourly abundance (i.e. the mean number of individuals of each taxa counted in each of the 6 instantaneous counts in one hour) (Figures 2 and 3). The timing of peak activity relative to the baiting event varied among treatments and replicates within treatments. Therefore the mean abundance of organisms was calculated over the full period (96 hours) of observation. The brittlestar *Ophiocomina nigra* was the most abundant of the megafauna recorded by the camera; there was a mean of approximately 10 individuals within the field of view for periods baited with undamaged scallops and with no bait (Figure 2). Baiting with damaged scallops caused a 2-3 fold increase in density. *Cancer pagurus* was the dominant predator of the tethered scallops but appeared in significant numbers only when damaged scallops were used (Figure 2). The mean abundance was highest for lightly damaged scallops, with over 3 times as many in this treatment than at badly damaged bait. The swimming crab *Liocarcinus* appeared with increasing frequency with an increase in the availability of food, from no bait to highly damaged bait, while spider crabs occurred at similar levels for all treatments except for lightly damaged bait where they occurred at a level over 20 times greater than background.

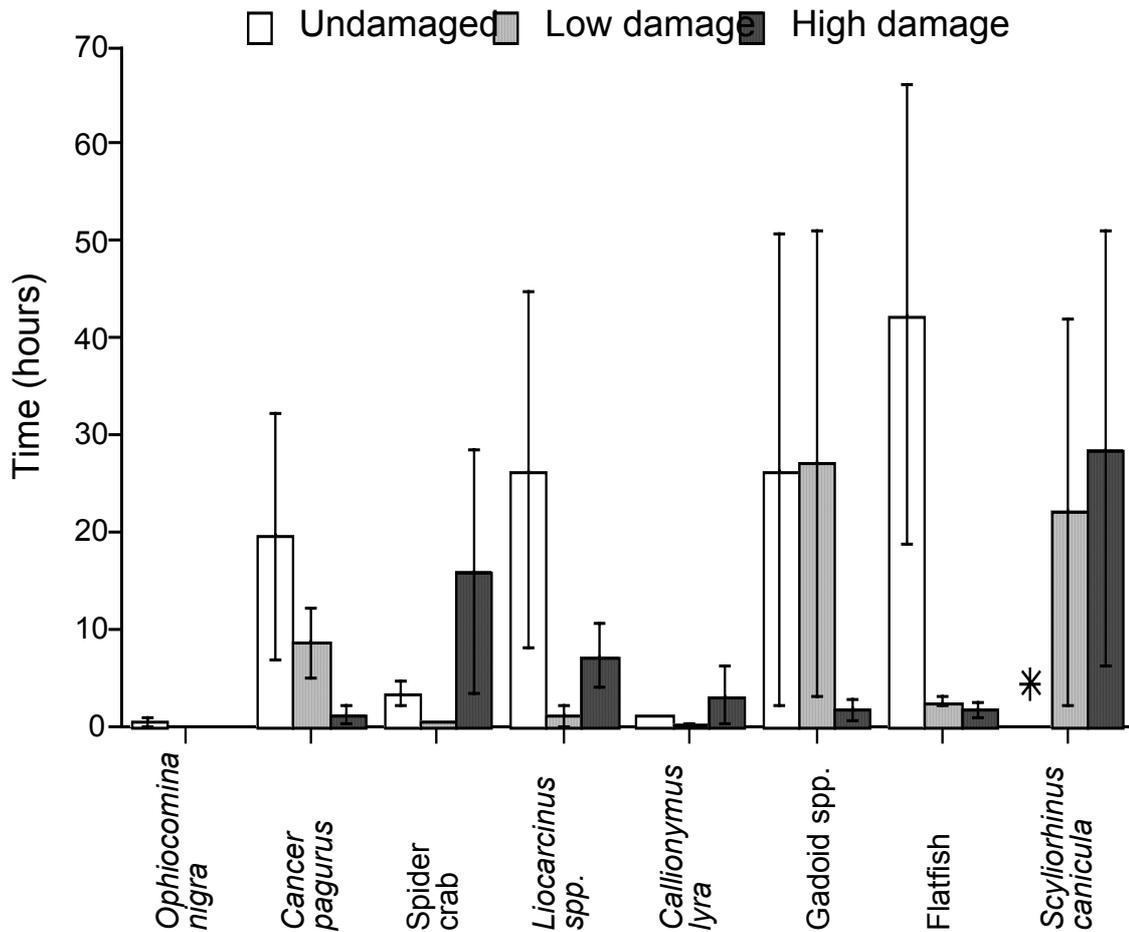


Figure 1. Mean number of hours after baiting for appearance of dominant scavengers in each of three treatments

Among the fish visiting the bait, flatfish were the most important consumers of scallops. However, dragonets (*Callionymus lyra*) were the most commonly seen fish (Figure 3). They were frequently seen during unbaited periods and showed no increase in density following baiting with undamaged scallops. Baiting with lightly damaged scallops caused an increase in abundance, whilst highly damaged scallops resulted in a decline. Dogfish were only recorded at damaged bait, while gadoids occurred at both undamaged and highly damaged scallops but were rarely seen with no bait or lightly damaged scallops. Flatfish, predominantly plaice, increased in abundance with increasing availability of food. However, the difference between the different damage treatments was large, with 8 times greater abundance where damage was high.

By pooling the two damaged treatments a one-way ANOVA was performed to determine differences between control periods (no scallops), dredged scallops and damaged scallops (Table 2). For three taxa, *Ophiocomina nigra*, *Cancer pagurus* and spider crab species, there was significantly greater abundance at damaged bait compared to undamaged bait. For all three comparisons there was no difference

between no bait and undamaged bait. No significant difference could be shown for the crab *Liocarcinus* and all fish species.

Occurrence of the 2 most important scavengers, *Cancer pagurus* and flatfish species, at bait is shown in more detail in Figures 4 and 5. The pattern of occurrence at bait was not simply a case of initial aggregation followed by dispersal. In *Cancer pagurus* there was some evidence of preference for nocturnal foraging, with aggregation during the night followed by dispersal during the day. However, this was not a consistent rule. The activity of *C. pagurus* in the lightly damaged treatment demonstrates the large variability typically found among replicates. In this case *C. pagurus* was abundant in the first 48 hours after baiting in one replicate but only found in any number in the second 48 hours in the other. Flatfish occurred sporadically in both the undamaged and lightly damaged treatments but showed clearer signs of strong aggregation to bait in the highly damaged treatment.

Scallop survival

The fate of the tethered scallops used as bait was determined from the video results and by divers at the end of each experimental period. All dredged, but undamaged scallops survived. The badly damaged scallops were all eaten within just over 24 hours for each of the 2 replicate periods. The scallops that were lightly damaged survived longer, but survival rates differed among replicates. In the first replicate, all scallops survived for 48 hours, and were eaten over the second half of the recorded period. In the second replicate, half of all scallops were eaten after only 12 hours, but some survived for up to 48 hours, whilst in the third baiting event (for which video records were incomplete) nearly half of all tethered scallops survived the full 96 hour period.

Discussion

There was a distinct difference in the community composition of scavengers compared to the study of Veale et al. (2000a) at the same site 4 years previously. Veale et al. (2000) found large aggregations of the predatory starfish *Asterias rubens* and *Astropecten irregularis*, and the crabs *Liocarcinus* species and *Pagurus* species. In general the scavenger community observed in our study differed markedly, with a greater dominance of fish and an increased importance of the edible crab *Cancer pagurus*. The differences in scavenger community found at bait in the two studies are likely to be a result of differences in the background density and distribution of scavengers.

Table 2. ANOVA of abundance estimates of different taxa found at bait of *Pecten maximus* with different damage levels. The two damage treatments were pooled and randomly reduced to three replicates. Data for *Cancer pagurus* and spider crabs are square root transformed to meet assumption of homogeneity of variance. Data for dogfish heterogenous.

Taxa	Source	df	MS	F	P
<i>Ophiocomina nigra</i>	Damage	2	341.69	9.06	<0.02
	Residual	6	37.71		
	SNK comparison	Damaged > Undamaged =Control			
<i>Cancer pagurus</i>	Damage	2	0.11	6.34	<0.04
	Residual	6	0.02		
	SNK comparison	Damaged > Undamaged =Control			
Spider crabs	Damage	2	0.18	6.28	<0.04
	Residual	6	0.03		
	SNK comparison	Damaged > Undamaged =Control			
<i>Liocarcinus</i> spp.	Damage	2	1.14×10^{-3}	3.56	>0.05
	Residual	6	3.19×10^{-4}		
<i>Callionymus lyra</i>	Damage	2	0.14	0.68	>0.5
	Residual	6	0.21		
Gadoid spp.	Damage	2	7.65×10^{-3}	1.02	>0.4
	Residual	6	7.5×10^{-3}		
Flatfish spp.	Damage	2	6.24×10^{-3}	2.69	>0.1
	Residual	6	2.32×10^{-3}		
<i>Scyliorhinus canicula</i>	Damage	2	2.32×10^{-3}	2.90	>0.1
	Residual	6	7.99×10^{-4}		

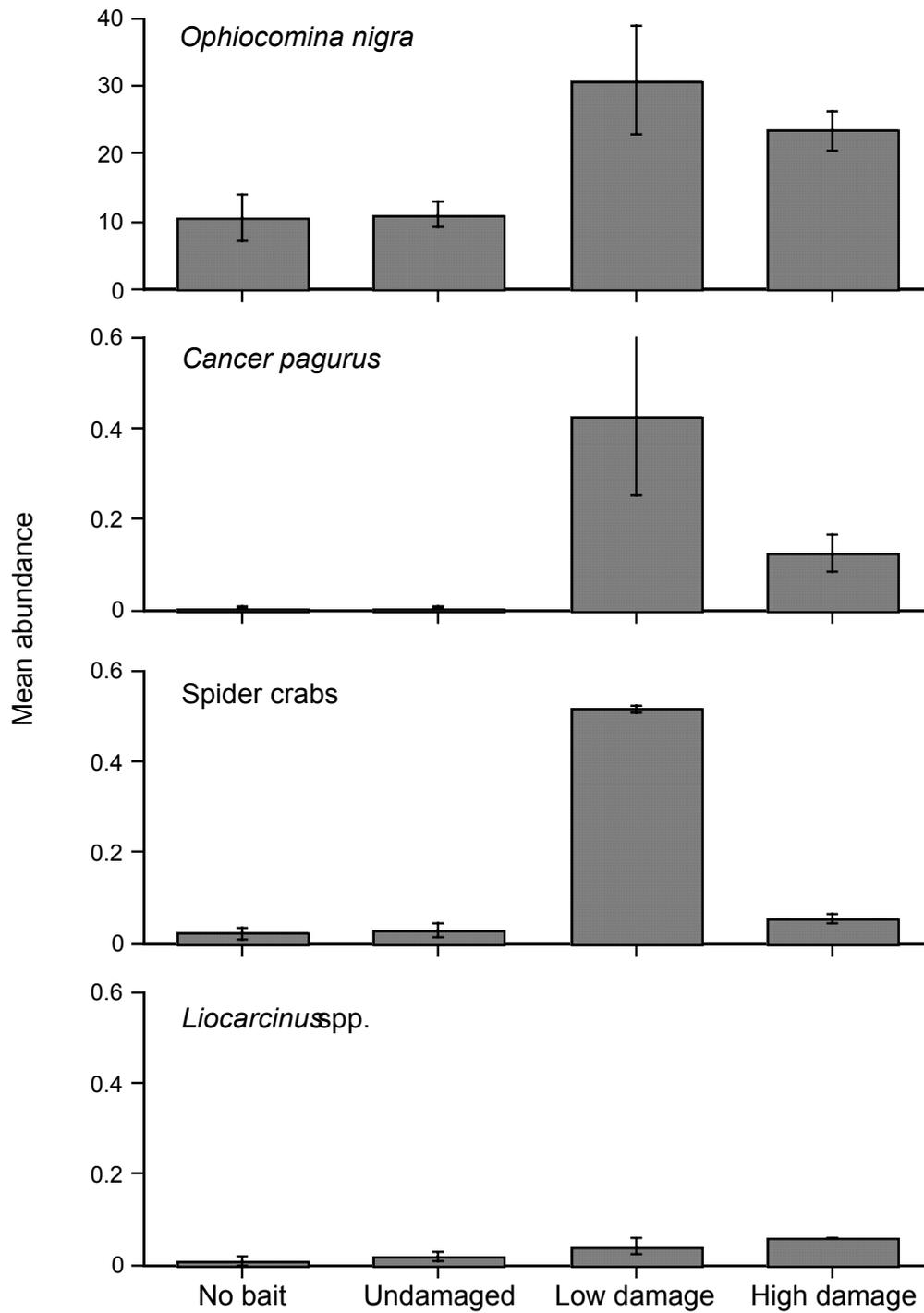


Figure 2. Mean hourly abundance of the dominant invertebrate scavengers at unbaited and baited treatments over the 96 hour sampling period

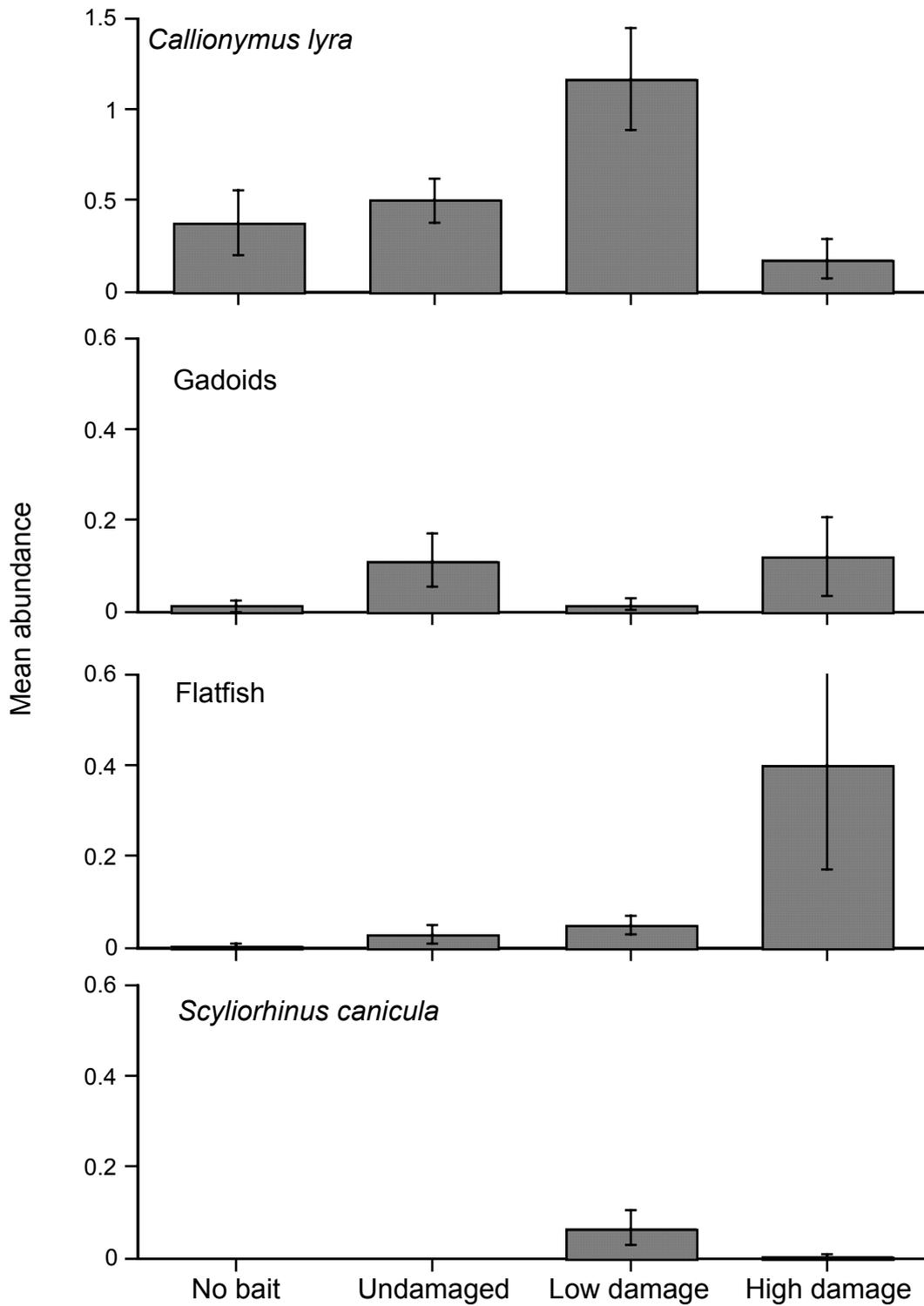


Figure 3. Mean hourly abundance of the dominant fish scavengers at unbaited and baited treatments over the 96 hour sampling period

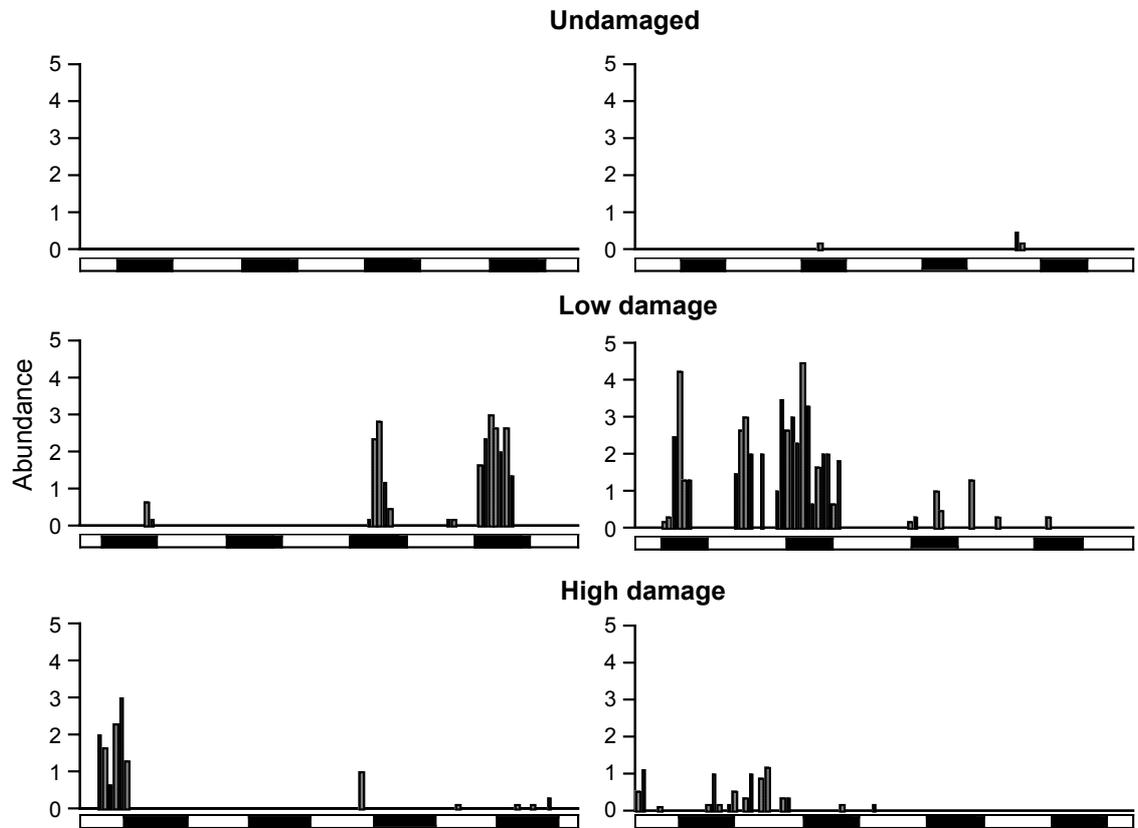


Figure 4. Mean hourly abundance of *Cancer pagurus* for two replicate periods in each of the three damage treatments.

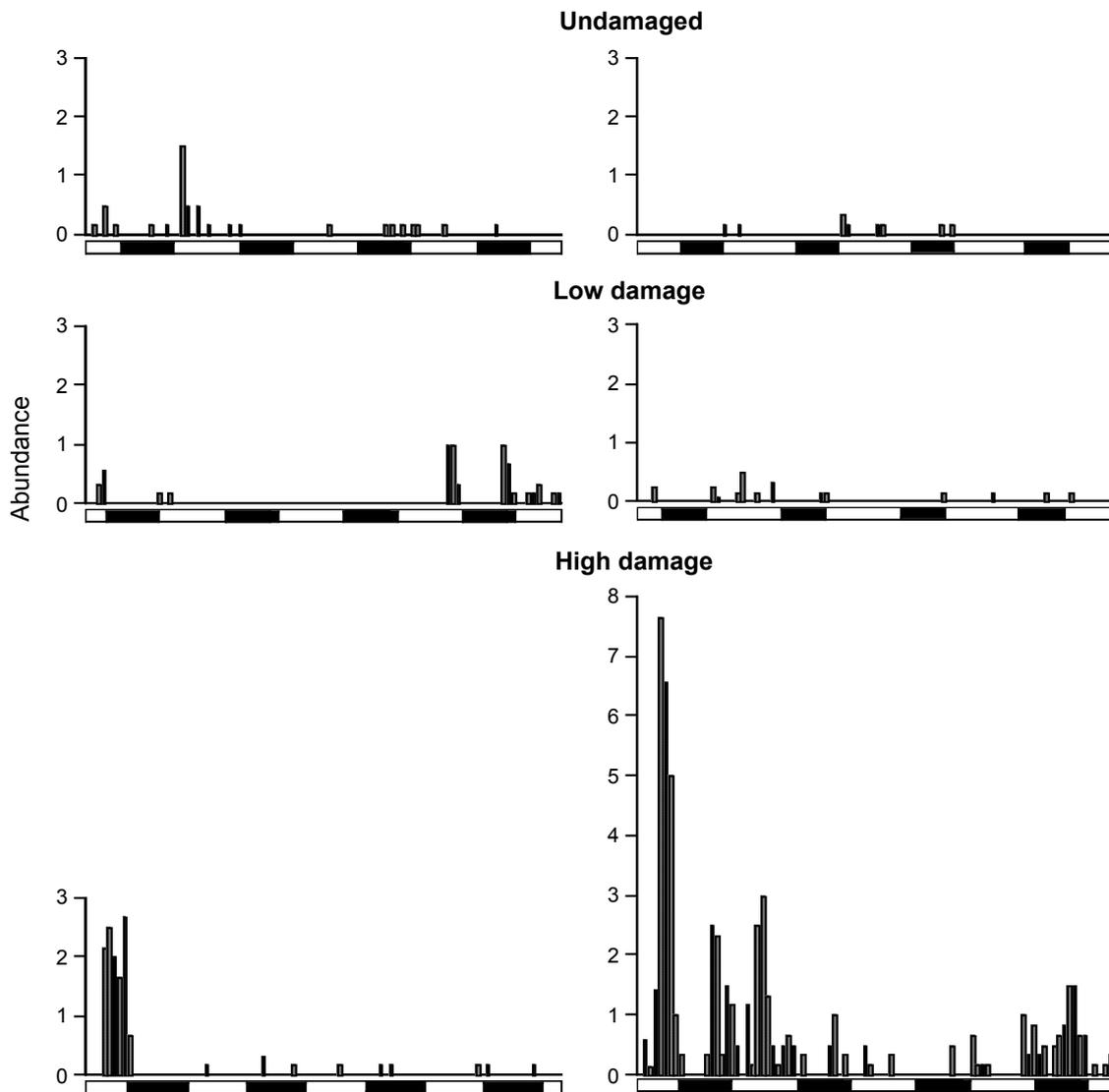


Figure 5. Mean hourly abundance of flatfish species for two replicate periods in each of the three damage treatments.

The community composition of scavengers will affect the way in which carcasses or prey are utilised. We found that the crab *Cancer pagurus* greatly facilitated feeding on *Pecten maximus* by other scavengers, particularly fish. It is likely that without this species much of the scallop biomass that was consumed by a range of scavenger species would have been unavailable. *Cancer pagurus* was not able to prey on undamaged *Pecten maximus*, presumably owing to their large size Lake et al. (1987). *Asterias rubens* may be capable of preying on larger scallops and hence its virtual absence may have affected the chance of undamaged scallops being preyed on and made available to the wider scavenger community.

Comparison of the rate and magnitude of scavenger aggregation to bait with different levels of damage was hindered by the loss of two replicates through technical problems. A large degree of variability was found in the timing, pattern and magnitude of aggregation among replicates of the same treatment. In addition, different taxa showed different patterns. In a preliminary assessment of the effect of damage level on scavenger aggregation, Veale et al. (2000) found no difference in the level of aggregation to bait of damaged and undamaged queen scallops (*Aequipecten opercularis*). Undamaged bait attracted *Asterias rubens* to a density 7 times that of background levels. However, these data were based on only one replicate. We found the density of scavengers at dredged but undamaged bait was equivalent to the density during unbaited periods. Thus, despite the unnaturally high density of scallops and the potential for the release of stress-induced products Minchin (1991), we found no evidence of scavenger aggregation. This was in marked contrast to the damaged scallops, where, despite large spatial and temporal variability, there was a significant increase in invertebrate scavenger density over the study period of 96 hours.

In summary, we showed that the level of damage to discarded biota had a clear effect on scavenger aggregation. Our results have implications for the design of demersal fishing gear. If damage to by-catch and non-catch can be reduced then there is the potential to reduce the level of fishing-induced mortality. Our work also showed a large degree of temporal variability in the rate and magnitude of scavenger aggregation over time-scales of weeks. In addition, through comparison with an earlier study we demonstrated large temporal variability over scales of years in the community of scavengers found at discard sites.

3.1.5.2 SPATIAL DIFFERENCES OF MACROFAUNA WITHIN CLAM DREDGE TRACKS AND THEIR IMPLICATIONS FOR SHORT-TERM FISHING EFFECT STUDIES

Introduction

The short-term environmental effects of dredging on the sea-bottom have received increased attention in recent years, and several studies have characterised changes in the ecosystem due to dredge fishing (e.g. Hall et al. 1990; Michael et al. 1990; Eleftheriou and Robertson 1992; Kaiser et al. 1996; Lambert and Goudreau 1996). However, only a few authors, such as Lambert and Goudreau 1996) and Meyer et al. 1981), have considered the importance of analysing different sections of the dredge track in fishing impact studies.

The most common approach in fishing effects studies has been to perform random sampling along dredge tracks (e.g. Medcof and Caddy 1971; Caddy 1973; Michael et al. 1990). However, random sampling within a dredge track can lead to biased data collection because, if the impact is not even throughout a track, results may be biased depending upon where samples are taken. Consequently there is an increased heterogeneity of variance in results, as noted by Hall et al. (1990), Eleftheriou and Robertson (1992) and the use of non-parametric, and frequently less powerful, statistical tests is necessary.

Information on the effects of clam dredge fisheries on benthos is scarce (Gaspar et al. 1998). In fact, few impacts of fishing are well documented and biological impacts are particularly difficult to investigate because of the complexity of benthic communities and our limited knowledge of their natural variability. Thus, there is the need to more adequately assess biological impacts (Currie and Parry 1996). With the increase in Portuguese exploitation of clams (*Spisula solida*, *Donax trunculus*, *Venus striatula*, *Pharus legumen* and *Ensis siliqua*), there is also an increased need to understand the environmental effects of these fisheries for appropriate fisheries management.

The aims of this study were: to analyse differences in macrofauna within sections of a bivalve dredge track in the southern coast of Portugal; to compare these differences between fished and a non-fished areas, and to suggest an appropriate sampling strategy to assess short-term dredging impact.

Methods

Study site

The Algarve coast (south Portugal) extends from Cabo São Vicente in the west (8° 59'W), to the border with Spain in the east (7° 24'W). Two study areas were selected off the western Algarve coast. The area off Vilamoura is a *Spisula solida* ground and is heavily fished, whereas the presently non-fished (control) area off Lagos was an *Ensis siliqua* fishing ground until it was exhausted in 1996. Both areas have similar physical characteristics in terms of coastline, bottom substratum (mostly sand) and direction of prevailing winds and currents (Dias 1987).

Gear description

Bivalve dredges used on the south coast of Algarve are large, heavy, iron structures, with a 25-mm-mesh net bag and a toothed lower bar to the mouth. The dredge mouth can be up to 150 cm wide and have teeth up to 50 cm long that act as a rake when the dredge is dragged through the sediment (Gaspar et al. 1994). After a dredging tow, a clearly defined track can be found on the seabed. A slope about 20 cm high forms the edges of this track.

In situ and video observations of clam dredging show the presence of a sand buffer (Figure 1), which forms approximately 10 m after the start of a tow, in front of the dredge mouth. This sand buffer pushes the sediment sideways, and limits the amount of material (both sediment and organisms) that enters the dredge mouth.

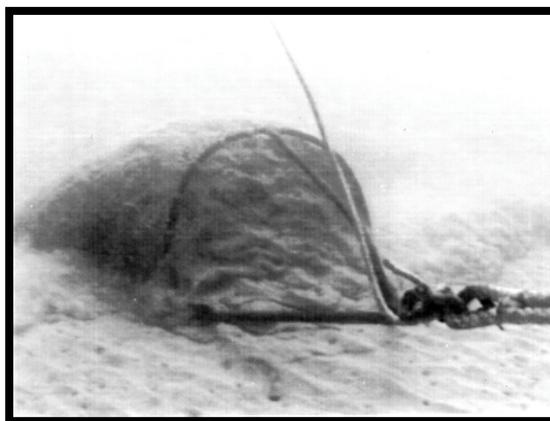


Figure 1. Photograph of the dredge showing the sand buffer formed in front of the gear during towing.

Sampling strategy and laboratory procedure

During September 1999, divers sampled both sampling areas before and after dredge fishing activity, within a depth range of 7–10 m. At both areas, three 50 m dredge tracks were created (approximately 1 min towing), following usual fishing procedures. These tracks were divided into three longitudinal sections (start, before the formation of the sand buffer; middle, approximately 3 m after the formation of the sand buffer; and end, the last 3 m of the track), and two transverse sections (inside the track and track-edge). Sediment from six quadrats (0.0625 m² to 0.15 m depth) from each section of every track was collected by SCUBA divers, before and after the fishing disturbance. Samples were sieved *in situ* through 1mm-mesh bags. The macrofauna collected were preserved on board in 70% ethanol. In the laboratory, organisms were identified to species level, and counted.

Data analysis

Macrofaunal abundance was converted to abundance per square metre (A) and total number of taxa (T), Shannon-Wiener diversity (H) and evenness (E) indices (Legendre and Legendre 1984) were calculated. The effect of transverse and longitudinal dredge track sections on H, E, A and T, in Lagos and Vilamoura was

examined by conventional two-way ANOVA (F-test). As more than two comparisons were made in each, all analyses were followed by a means comparisons Tukey test to compare means between dredge track sections and the results displayed on the figures. Statistical analyses were conducted using the STATISTICA 5.0 program.

Results

In both the fished and non-fished study areas, differences in macrofauna composition were observed following dredging. The most significant reduction in diversity, abundance and the number of taxa present occurred at the Lagos study area (Table 1).

Table 1: Mean and standard deviation of the Shannon-Wiener Diversity Index, evenness, abundance and number of taxa, before and after dredge disturbance at the Lagos and Vilamoura sampling sites.

	LAGOS		VILAMOURA	
	Before	After	Before	After
Shannon-Wiener Diversity Index (H)	1.21 ± 0.14	0.76 ± 0.20	0.88 ± 0.15	0.64 ± 0.22
Evenness (E)	0.87 ± 0.05	0.90 ± 0.12	0.82 ± 0.08	0.86 ± 0.23
Abundance (A)	694 ± 323	195 ± 135	160 ± 54	154 ± 78
Number of taxa (N)	15 ± 5	8 ± 3	5 ± 2	6 ± 2

Two-way ANOVA analysis showed significant differences in diversity, evenness, abundance and the number of taxa present between track sections at the Lagos study area. Off Vilamoura, only macrofauna evenness was significantly different across longitudinal sections of the track. The results indicate interactions in macrofauna diversity between longitudinal and transverse sections of the dredge track at both the areas studied, though only off Vilamoura (fished area) were these interactions also observed in macrofauna abundance and the number of taxa present (Table 2).

Application of the Tukey HSD Test (Figure 2) showed that significant differences in macrofauna abundance and number of species occurred between the dredge track and its lateral edges at the start of the track, at both Vilamoura and Lagos. At Lagos, this section of the track also showed significant differences in the Shannon-Weaver diversity index between the track and edges. Macrofauna community structure along the middle length of the dredge track did not differ significantly between the track and the edges at Vilamoura, but significant differences in the diversity index of these sections were found at Lagos. For both areas, no significant differences in any of the parameters analysed were observed between the track and the edges at the end of the dredge track.

Table 2 - Summary of *p* values of two-way ANOVA, fixed effect for Shannon-Wiener Diversity Index (H) and evenness (E), abundance (A) and number of taxa (T), at both the Lagos and Vilamoura sampling sites. 1- Transverse sections of the dredge track: Edge and Track, 2-Longitudinal sections of the dredge track: Start, Middle and End (*-*p*<0.05).

	LAGOS				VILAMOURA			
	H	E	A	T	H	E	A	T
1-Transverse sections	0.0088*	0.2667	0.0000*	0.0001*	0.4879	0.3161	0.2482	0.1490
2-Longitudinal sections	0.1644	0.0001*	0.0024*	0.0458	0.0449	0.0001*	0.7954	0.2744
Interactions	0.0023*	0.2482	0.1421	0.0087*	0.0013*	0.1753	0.0020*	0.0008*

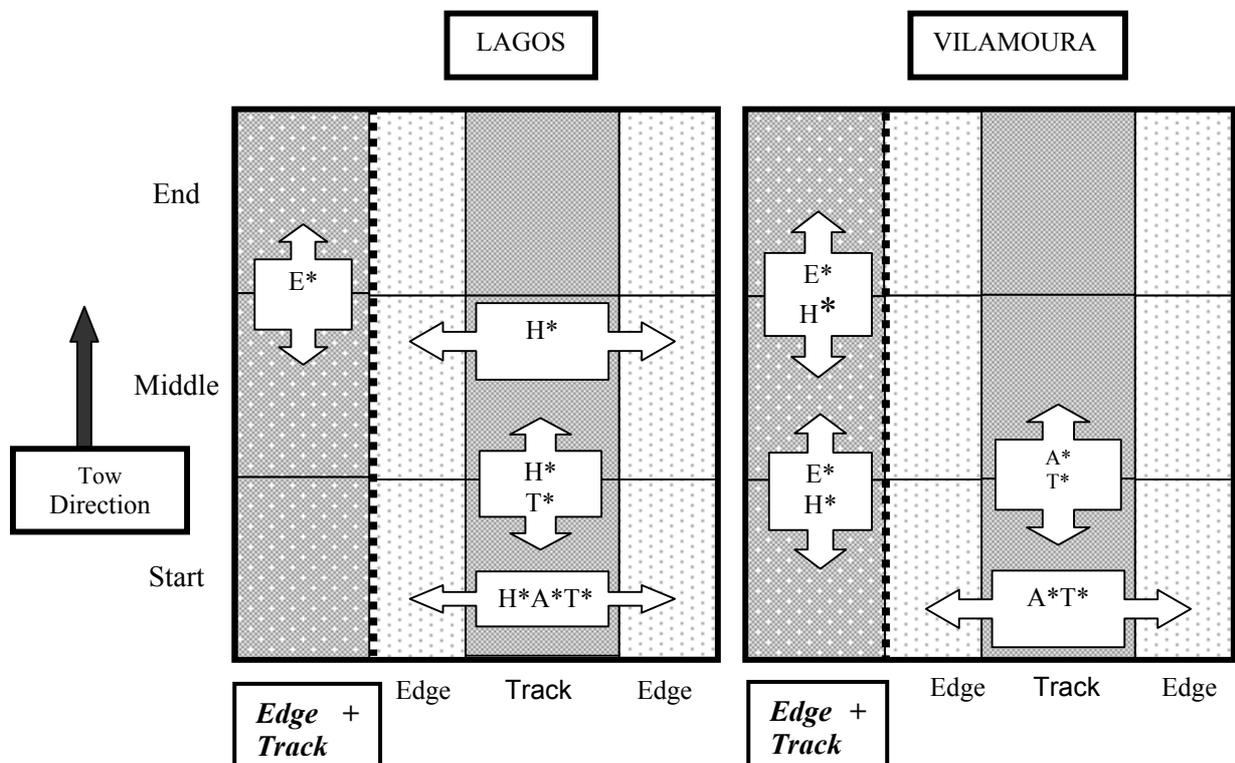


Figure 2. Arrows show significant differences among sections of the area affected by the dredge (track and edges), as determined by Tukey HSD test, for the population parameters analysed: H=Shannon-Wiener Diversity Index; E=Evenness; A=Abundance; and T=Number of taxa.

Significant differences between longitudinal sections of the dredge track were found at the Vilamoura study area. In this fished area, significant differences in diversity and evenness were found between the start and the middle, and between the middle and the end of the dredge track. At Lagos, only evenness was significantly different, and then only between the middle and end track sections. Differences between longitudinal sections of the lateral edges of the dredge track were also observed. In both study areas, these differences occurred only between the start and middle sections of the edges. Similarly, in both fished and non-fished areas, the number of taxa present varied significantly between these two sections. Furthermore, differences in the diversity index were found at Lagos, and macrofauna abundance differed significantly between these two sections of track edge at Vilamoura.

Discussion

As expected, macrofauna composition changed following dredging in both the fished and non-fished areas (e.g., Bergman and Hup 1992; Kaiser et al. 1998). Therefore, changes within track sections can be directly related to the fishing impact. This was more evident at the Lagos site, where abundance, number of taxa, diversity, and evenness of the macrofauna were all higher before fishing. In this non-fished area, more differences also occurred between track sections, and fishing was most disruptive in this more stable ecosystem.

In contrast, at the Vilamoura site, macrofauna composition must be adapted to the long-standing fishing impact to which it has been exposed and dredging caused fewer changes. This would explain why, in the fished area, only the evenness of the macrofauna community was significantly different between dredge track sections, whereas in the non-fished area, significant differences between track sections were observed for abundance, number of taxa, diversity, and evenness. Therefore, greater caution is needed when sampling the macrofauna of dredge tracks in non-fished areas, which are commonly used as control areas during studies (Tuck et al. 1998).

If fishing activity may explain overall differences in macrofauna composition between tracks at the fished and non-fished areas, differences among track sections may also result from the gear behaviour, as also noted by Eleftheriou and Robertson (1992) and Meyer et al. (1981). Our observations showed that approximately 10 m after the start of dredging a sand buffer forms in front of the gear. This results from the clogging of the mesh. After formation of this buffer, sediment and organisms are “mixed”, and then pushed downwards and sideways away from the gear. This explains the significant differences between the start and middle sections of the track, and between the track and its edges. However, after formation of the sand buffer and the subsequent “mixing”, the sediment and organisms pushed along the track or to the edges become more homogeneous, and fewer differences can be observed in composition of the macrofauna between track sections.

Thus, the decision where to sample within a dredge track will depend on the objective of a study. Specific analysis of the effects of the gear on the ecosystem (e.g. the fishing impact on a particular taxa or sediment type) can be most clearly determined if sampling is conducted at the start of the dredge track, before the sand buffer forms. However, more general analysis of the impact of fishing should also consider changes caused by the formation of a sand buffer, which affects the larger part of a dredge track.

Therefore, we suggest that for fishing impact studies, the behaviour of the gear should be assessed before sampling, and sampling design must consider eventual differences between longitudinal and transverse sections of the area affected by the gear.

3.1.5.3 SCAVENGING ON DAMAGED INDIVIDUALS LEFT ON DREDGE TRACKS

Introduction

Clam dredges are extensively used along the south coast of Portugal to catch the white clam *Spisula solida*, the striped venus *Chamela gallina* and the donax clam *Donax trunculus*. This kind of fishing gear disturbs the sediment and may have an impact on the structure and processes of the seabed. For example, changes in grain-size may occur due to the transport of fine sediments during sediment suspension. Clam dredges also induce mortality in non-target benthic species and undersized individuals of the target species that are discarded or left on the dredge path. Of the individuals that are not caught, but have come within the influence of the dredge, some may be so injured or stressed that they die almost immediately, while other may become susceptible to predation (McLoughlin et al. 1991). Animals dislodged or damaged by the passage of the dredge and left on the dredge track are known to attract mobile epifauna (eg. Medcof and Bourne 1964b; Caddy 1973; Kaiser and Spencer 1994).

Marine predators and scavengers have developed chemosensory abilities, which enable them to locate any source of food using water-borne signals (Miller 1978). The size and shape of an area around a food source within which scavengers detect and actively move towards the food is influenced by several factors, including depth, substratum, currents, temperature and reproductive condition (Himmelman 1988). Among these, currents are the major factor determining the ability of benthic scavengers to detect and locate the food source, since detection depends on water-borne substances emitted by the food (Okubo 1980; Zimmer-Faust 1983). McQuinn (1988) showed that the size and shape of the area from which scavengers are attracted to bait is influenced by the current speed and direction, and that the effective area can vary by more than an order of magnitude depending on current conditions. Kleerekopper et al. (1975) reported that under conditions of rapid, compared with very slow current flow, chemical and rheotactic gradients are steeper so animals may be more able to localise bait. Many observations of scavengers arriving at bait from downstream directions indicate that food is primarily detected by olfaction (Wilson and Smith 1984; Smith 1985; Atema 1988). Scavenger attraction might also result from the detection of sound or pressure waves generated by the dredge during the tow. The activity of scavengers within a dredge path is determined by the number of damaged individuals left on the track.

Recently, a new dredge was introduced into the Portuguese bivalve fishery, which is more efficient and selective than the traditional dredge used by the commercial fleet (Gaspar et al. In press). However, the general effect of dredging with this new dredge, both on the substrate and macrobenthic fauna, is still unknown. The present study reports the results of a diver-conducted investigation to assess the effect of dredging on bottom substrate, to estimate damage inflicted by the new dredge on the benthic macrofauna left on the dredge path and to evaluate scavenger aggregation within the track.

Methodologies

Experimental design

The study was undertaken off Lagos (south Portugal) on board the R.V. Donax during July 2000. The tows were conducted during ebb tide and performed parallel to the coastline in an east-west direction.

Track configuration and persistence

In order to evaluate the influence of depth and type of sediment on dredge track configuration and dredge track persistence, tows were performed at depths of 5 and 12 m, in sand and sandy-mud bottoms. For each combination depth/type of sediment 3 hauls were made with a dredge identical to those used by the commercial dredge fleet (Figure 1). All fishing hauls were conducted at a towing speed of 1-1.5 knots for a duration of 1min. The beginning and the end of the dredge track were marked by a buoy and the distance between buoys was measured. After each haul, divers swam along the dredge path to characterise the dredge tracks. For each tow, divers measured the track width (at the bottom and top of the track) and the track depth. These measurements were carried out in the middle of the dredge track. In order to evaluate how long the dredge track persisted the track depth was measured immediately after the tow and again at 1, 2, 4, 6, 12 and 24 hours. The experiments were made on 4 consecutive days.

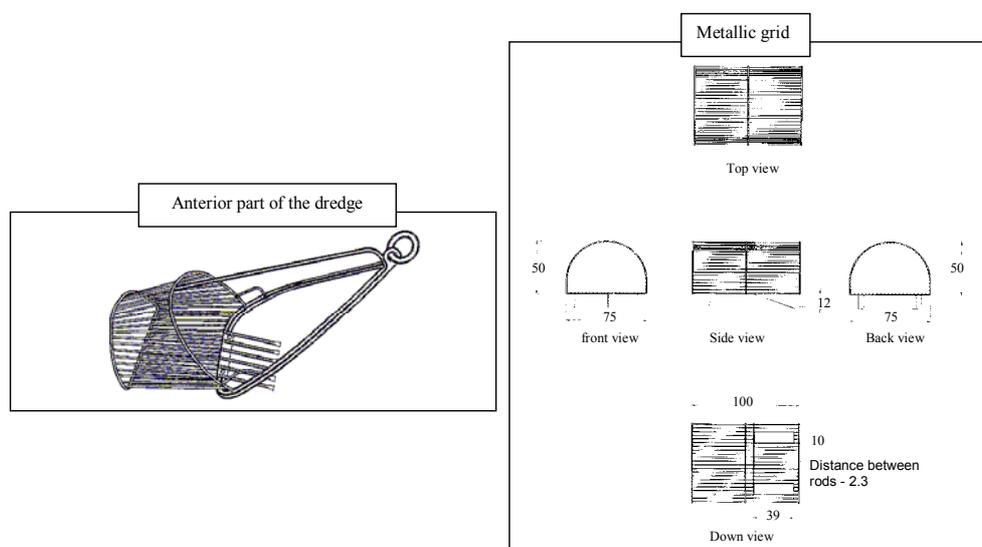


Figure 1. Schematic representation of the clam dredge used in the present study.

Mortality of macrofauna left on the track.

In order to estimate the number of damaged individuals left in dredge tracks, sediment samples were collected by divers. This study was only conducted on the hauls performed on sandy bottoms at 5 m depth. At each dredge path, 6 quadrats (area=0.0625m²) were collected in the beginning, middle and at the end of the track. Samples were sieved *in situ* through a 5-mm mesh bag. On board, the organisms collected were preserved in 70% ethanol. In the laboratory, the organisms were identified and counted, and a damage score was attributed to each specimen. The damage scale applied in this study had three scores, where score 1 corresponded to

individuals apparently undamaged, whilst score 3 corresponded to dead organisms. Prior to these experiments, the background densities of the most important scavenging species were estimated by means of six 100 m² diver transect surveys. For each survey, divers followed a 50 m long bottom line marked every 5 m and the number of organisms was counted within each mark on either sides of the line. The width of each transect was 2 m.

Aggregation.

Aggregation of predators and/or scavenging species within the dredge tracks was also assessed. For this purpose, divers made video recordings of the tracks. These recordings were made with a digital video camera, immediately after the haul and after 10, 20, 30 and 60 min. The divers swam along the dredge path recording damaged epifauna and activity by mobile epifauna. The filming procedures were carried out at a slow speed (approx. 7 m.min⁻¹), with fixed focus and with the camera held approximately 1 m from the bottom.

Data analysis

To estimate the aggregation of predators and/or scavengers in the dredge track, the video tape recordings were viewed and instantaneous counts of all individuals within the track were made every 1 min of real time. To test the accuracy of data gathered from underwater video images, independent reviews of the videotapes were made. Before further data processing, the number of scavengers was converted to abundance per square metre.

Statistical procedures followed Sokal and Rohlf (1981) and Zar (1996). Analyses of variance ANOVA or Kruskal-Wallis ANOVA were used to test differences, both in the number of scavengers within the dredge path through the duration of the experiments, and in the proportion of damaged individuals (scores 2 and 3) in the beginning, middle and end of the dredge path. Multiple comparisons were performed using the Student-Newman-Keuls-test. Prior to the application of the above tests, data were converted to square-roots, or to arcsine square root values for percentages. All statistical analyses were performed using SigmaStat statistical software.

Results

Track configuration and persistence.

During the tow, on sandy bottoms, the sediment cloud behind the dredge seldom exceeded 1 metre in height and settled quickly and evenly within 1 minute. On sandy-mud sediments, the cloud created during tows was 2-3 m in height and settled within 3-5 minutes. The dredge tracks were very similar in both sediments, with smooth and steeply cut walls and a flat floor. However, on sand bottoms slumping along the walls of the path began immediately after the tow. The dredge paths were 50 m long and 60 cm wide. The average track depth was slightly deeper on sandy-mud bottoms than on sand bottoms, being of 17 cm and 15 cm penetration depth respectively. The track shoulders formed during the tow varied along the dredge track. At the beginning of the dredge path the track shoulder was 5 cm wide and 5 cm high, whilst on the middle and end of the track, the shoulder was 15-20 cm wide and 3-5 cm high.

Figures 2 and 3 show the persistence of a dredge track, in sand and sandy-mud bottoms, over a period of 24 hours, at 5 and 12 metres depth respectively. On sand bottoms, at both depths, the track walls began slumping immediately after the tow. In the 2-hours old track the slumping of the walls created a more rounded depression and the track shoulder decreased in height. The dredge path average depth also decreased slightly. After 6 hours and 12 hours the mean depth of dredge track decreased to 11 cm and 7 cm respectively and the dredge shoulder almost disappeared. At the end of the observations (24-hour old track) the dredge track was difficult to recognise, since it blended in with general bottom features (sand ripples) and the dredge shoulders completely disappeared.

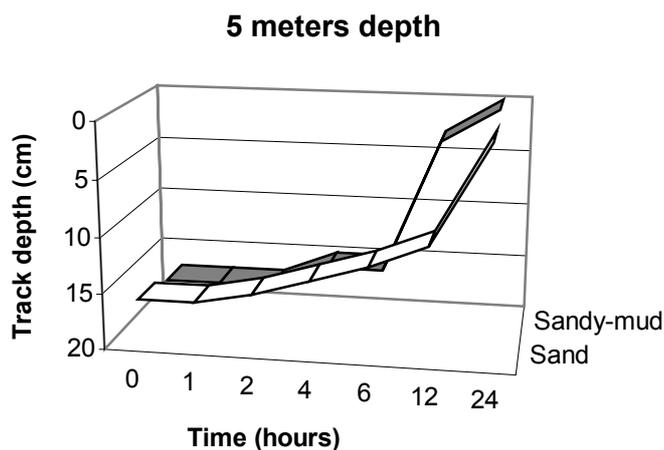


Figure 2. Persistence of dredge tracks on sand and sandy-mud bottoms at 5 meters depth.

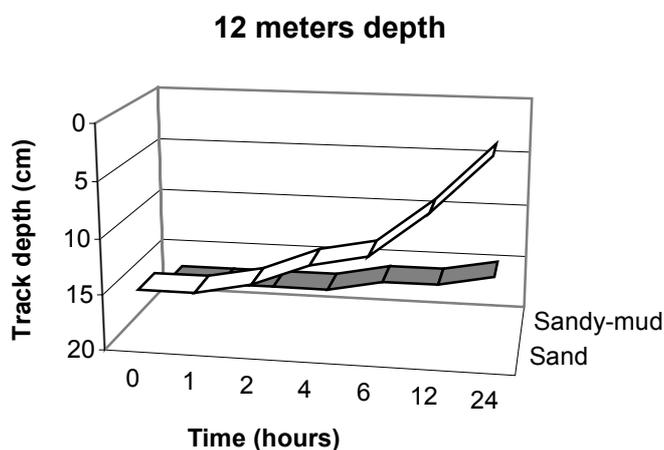


Figure 3. Persistence of dredge track on sand and sandy-mud bottoms at 12 meters depth.

On sandy-mud bottoms at 12 m depth the dredge track remained almost unchanged after 24 hours and the average track depth decreased by only two centimetres. The dredge track seemed to be filling in by sediment transport rather than by slumping of sediment from the walls and the dredge shoulders were also perfectly visible. At 5 m depth, the dredge track remained almost unchanged 6 hours after the tow but within 12 hours the dredge shoulder and path were hardly recognised and its mean depth decreased to 3 cm. After 24 hours the dredge track was indistinguishable. The reason for the rapid disappearance of the dredge track was the sea conditions, for at the end of this day there were intense south-easterly winds that caused the formation of waves (2 -3 m high). This waves action induced disturbance in bottom sediment, leading to the rapid disappearance of the track left by the dredge. For the first three days of this experiment, sea conditions were calm with weak tidal currents.

Mortality of macrofauna left on the track.

During the experiments, a total of 10 macrofaunal species were identified within the dredge path. The most abundant group was Bivalvia, comprising approximately 95% of the individuals left on the track. Within this Class the most abundant species were *Chamelea gallina*, *Donax trunculus*, *Spisula solida* and *Tellina tenuis*. The estimated proportion of damaged individuals that remained on the dredge track was very similar for the three tows performed, varying between 12.9% and 13.8%, which corresponded to 31 and 38 individuals respectively. No significant differences were observed in the proportion of damaged individuals in the beginning, middle and end of the dredge tracks (K-W, $H=0.746$, d.f.=2, $P_{\text{est}}=0.689$, $P_{\text{exact}}=0.746$). Hence, for each tow, data from all quadrats were pooled and the mean number of individuals per damage score determined for each species in order to evaluate the relative vulnerability to dredging of the species that comprised the by-catch (Table 1).

As expected, the most sensitive species to this kind of fishery were the thin-shelled bivalves such as *Tellina tenuis*, *Pharus legumen*, *Macra corallina stultorum* and *Lutraria anguistor*, as well as the sea urchin *Echinocardium cordatum*. Little or no damage occurred to the solid shelled bivalves such as *Chamelea gallina*, *Spisula solida* and *Donax trunculus*.

Table 1. Species collected inside the dredge track and mean number of individuals per damage score

	Score	1	2	3
Group				
Bivalvia				
<i>Chamelea gallina</i>		82		
<i>Donax trunculus</i>		31	4	1
<i>Ensis siliqua</i>		1		
<i>Lutraria anguistor</i>				1
<i>Mactra corallina stultorum</i>				1
<i>Pharus legumen</i>		1		2
<i>Spisula solida</i>		57	2	1
<i>Tellina tenuis</i>		45	3	12
Echinoidea				
<i>Echinocardium cordatum</i>		1		8
Brachyura				
<i>Liocarcinus depurator</i>		3		
Total		221	9	26

At the end of each tow, divers made immediate observations on the area of seabed fished over by the gear. On all occasions, they found that only heavily damaged individuals remained on the seabed, while all the other organisms were already burrowed. The only exceptions were two individuals of *Pharus legumen* and one individual of *Ensis siliqua*, which apparently were undamaged, suggesting that these organisms may have suffered internal damage.

The size-frequency distributions of the bivalve species of commercial interest (*Spisula solida*, *Chamelea gallina* and *Donax trunculus*) collected inside the dredge track are shown in Figure 4. For *Spisula solida* and *Chamelea gallina* the majority of the individuals that passed through the grid bars were undersized (<25mm shell length), reflecting the dredge's selectivity. In the case of *Donax trunculus*, individuals of a size both below and above the minimum landing length for the species (25mm) were found within the dredge track.

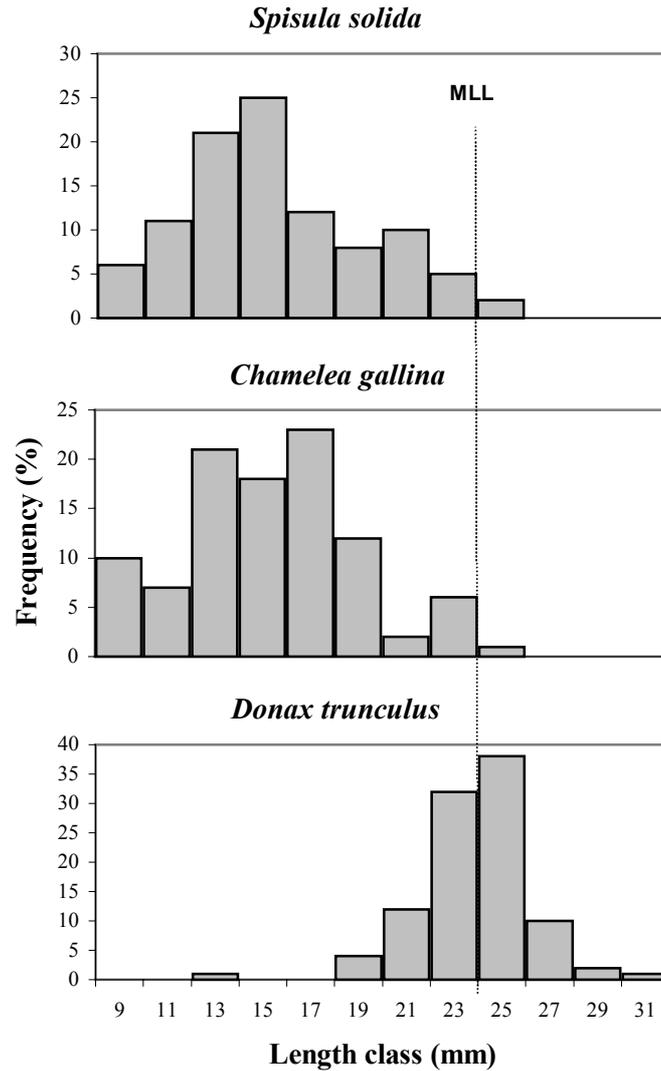


Figure 4. Size-frequency distributions of the commercial bivalve species collected by divers within the dredge tracks. Dotted line indicates the Minimum Landing Length (MLL) for the species.

Aggregation.

Prior to fishing operations, divers estimated the background densities of the most common scavenger species in the study area. Only two species were identified, the brittlestar *Ophiura albida* and the hermit crab *Pagurus spp.* The background densities recorded in these diving surveys were 0.62 and 0.3 individuals per m², for the brittlestar and for the hermit crab, respectively.

Divers observed the pronounced and rapid aggregation of scavengers in the dredge tracks immediately after the tow. Figure 5 shows the changes in mean abundance of these two species, throughout the duration of the experiments. In the case of the brittlestar, an initial increase in abundance was observed, being approximately 8 times greater than the background densities. However, after 10 minutes the mean abundance had decreased to 2.85 individuals per m², indicating that this species starts to disperse

very quickly. The K-W ANOVA on Ranks analysis showed significant differences in the mean abundance of brittlestars before and after the tow (K-W $H=16.128$; $df= 5$; $P=0.006$). The pair-wise multiple comparisons (Table 2) showed no significant differences in brittlestar density after 30 minutes in relation to the background density. The underwater video records showed that the direction of arrival of the brittlestars was opposite to that of the prevailing water current.

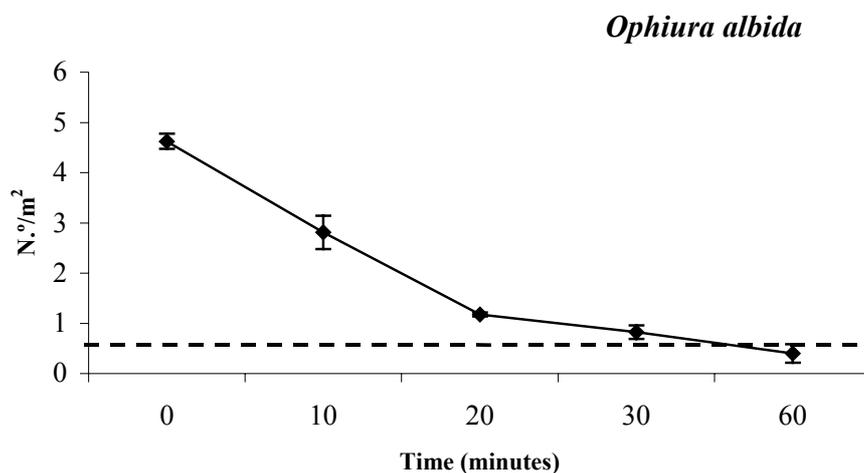


Figure 5. Mean density of *Ophiura albida* within the dredge track throughout the study period. Dashed line indicates the mean background density.

Table 2. Results of the pair-wise multiple comparisons (Student-Newman-Keuls-test) performed to test differences in the density of brittlestar throughout the study period.

Time (min.)	0	10	20	30	60	Background
0	-	**	**	*	*	*
10		-	**	**	**	**
20			-	**	**	**
30				-	ns	ns
60					-	ns

Notes: * = $p < 0.01$; ** = $p < 0.05$; ns = not significant

Changes of mean abundance estimated for the hermit crab in the dredge track are shown in Figure 6. Although a slight increase in the density of this species was observed throughout the duration of the observations, the ANOVA analysis revealed no significant differences ($P=0.710$) in the abundance of hermit crabs before and after the tow.

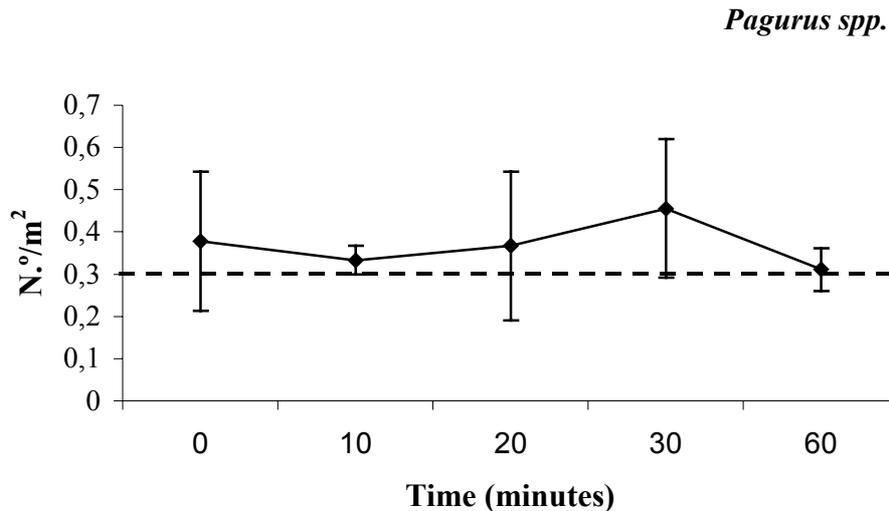


Figure 6. Mean density of *Pagurus spp.* within the dredge track throughout the study period. Dashed line indicates the mean background density.

Discussion

Physical effects of dredging include plowing and scraping the seabed, resuspension of sediments, and lowering habitat complexity. According to Jones (1992), plowing and scraping depend on towing speed, sediment type, depth, tides and currents, and gear configuration. In the present study it was found that the dredge track was deeper on sandy-mud sediments than in sandy sediments. It was also observed that the dredge track could be recognisable for a longer period in sediments comprising a large fraction of fines, varying from several days to a few hours in tows performed in sandy-mud and sandy sediments, respectively. Previous studies using hydraulic dredges have recorded sediment fluidisation in fished track areas (Lambert and Goudreau 1996). Kawling and Bakus (1979) suggested that disturbance of bottom sediments in the Bering Sea by a hydraulic dredge left traces that persisted from several days to several years. The disappearance of the tracks left on the bottom in both sediment types depends on both the tide or currents and wave action (this study; Messieh et al. (1991). Where the current movements are weak the dredge tracks may be recognisable for a long time and under these conditions even a relatively minor fishery may therefore have a significant cumulative effect on bottom microtopography (Caddy 1973).

The dredge passage also causes a large amount of sediment suspension in the water column. In our study, it was observed that sediment rapidly resettled independently of bottom type. However, no data were obtained on the effect of dredging on the grain size composition of the bottom sediments. Kawling and Bakus (1979), in an experimental study conducted in the Bering Sea, reported that hydraulic dredges had a minimal effect on the grain size, despite the resuspension of fine sediments that comprised only a small fraction of bottom sediments, consisting mainly of medium to coarse particles. Similar results were also reported by Eleftheriou and Robertson (1992). Nevertheless, Brambati and Fontolan (1990) stated that hydraulic dredging causes the transport of sediment, provoking geotechnical features on the seabed, influencing the stress resistance and consequently the critical erosion velocity.

Aschan (1991) reported that the scallop fishery conducted off Norway changed the bottom substrate from shell-sand to clay with large stones within a 3-years period.

On the south coast of Portugal the bivalve fishery only occurs in very shallow waters (between 1 and 15m depth) on sandy bottoms that primarily consist of medium to coarse sediments (Moita 1986). These areas are influenced by tides and relatively strong currents as well as wave action, therefore no changes in the granulometry are expected.

Damage and mortality of benthic organisms that were not caught but have come within the influence of the gear can occur due to the passage of the dredge over the seabed. In the present study, it was found that the most vulnerable species to this kind of fishery were the bivalves *Tellina tenuis*, *Pharus legumen*, *Macra corallina stultorum* and *Lutraria anguistor*, as well as the sea urchin *Echinocardium cordatum*. Nevertheless, our results showed that mortality induced by both dredges on the macrobenthic community left on the dredge path was very low. This may be explained by the high efficiency of capture and selectivity of the dredge used in this study (Gaspar et al. In press also [Section 3.3.3.2](#)), and/or the operational characteristics of the gears. McLoughlin et al. (1991) reported that the catching efficiency of the Australian scallop dredge was very low. According to these authors, 12-22% of the initial stock in Banks Strait was landed as catch while the rest of the stock was lost through direct and indirect mortality resulting from dredging. Michael et al. (1990) found that nearly 38% of the clams caught had severely or partly damaged feet, despite the high efficiency hydraulic dredges. For the New England hydraulic dredge, Lambert and Goudreau (1996) estimated a dredge efficiency of 90% for large surf clams (*Mactromeris polynyma*). These authors observed that two-thirds of the clams that remained on the bottom were damaged, which represents less than 10% of capturable clams. A small percentage of other species was also damaged.

Animals exposed or damaged by mobile fishing gears and left on dredge tracks is going to increase opportunistic feeding by invertebrates (e.g. Wassenberg and Hill 1987 Tuck et al. 2000) and predatory fishes (e.g. Kaiser and Spencer 1994). Aggregations of scavenging species after the passage of towed demersal fishing gears have been recorded in other studies on the effects of scallop dredges (e.g. Caddy 1973; Chapman et al. 1977), hydraulic dredges (e.g. Meyer et al. 1981; Lambert and Goudreau 1996) and beam trawls (e.g. Kaiser & Spencer, 1994; Ramsay et al. 1996). Caddy 1973) observed that predatory fish and crabs were attracted to tracks within 1 hour. Chapman et al. (1977) reported that damaged individuals left on the dredge path appeared to attract several fish and invertebrates including juvenile cod, adult plaice and dogfish, whelks and hermit crabs.

Predation on undamaged individuals may also occur. Meyer et al. (1981) stated that starfish were feeding on undamaged clams left on dredge tracks. Kaiser & Spencer (1994) found that the diet composition of whiting and gurnards altered after the passage of the beam trawl and suggested that these species were feeding on animals exposed or damaged by the gear. In our study, it was observed that animals damaged by dredging rapidly attract scavengers. Post-fishing diver observations showed that brittlestars were attracted into the track region immediately after the tow to scavenge on material disturbed by the dredge. Among scavengers *Ophiura albida* was the commonest species, with *Pagurus spp.* only occurring occasionally. Following

fishing, brittlestars were very abundant in the track area at densities 8 times greater than the background density. However, after 10 minutes the mean abundance of brittlestars decreased sharply, indicating that this species starts to disperse very quickly. These results are explained by the low number of damaged individuals left on the dredge tracks.

It is important to emphasise that almost all undamaged individuals within the track were already burrowed when the divers surveyed the tracks. This suggests that the stress induced by dredging is not high enough to greatly decrease the burrowing response speed of undamaged individuals, and therefore they burrow immediately after escaping through the bars of the retention grid.

In conclusion, the Portuguese clam dredge currently used by the commercial fleet is highly selective and induces low mortality both on individuals caught by the gear (Gaspar et al., in press) and on organisms left on dredge tracks (present study). However, long-term effects on the community structure may occur depending on fishing effort, since the abundance of the most vulnerable species such as thin-shelled bivalves species and sea urchins may decrease greatly or even disappear if a particular area is persistently fished over a long period.

3.1.5.4 CHANGES AND RECOVERY OF THE BENTHIC COMMUNITY OF A CLAM-DREDGED AREA OFF THE ALGARVE COAST (SOUTH PORTUGAL)

Introduction

Commercial bottom fishing is known to affect benthic communities by removing or injuring surface-living organisms in the path of the gear, or by digging into the sediment. This affects both the infauna and the epibenthos (Bergman and Hup 1992; Tuck et al. 1998). Thus, the environmental effects of shellfish dredging have been the subject of special attention throughout the world over recent decades (Caddy 1968, Caddy 1973; Meyer et al. 1981; Kaiser and Spencer 1996b; Jennings et al. 2001).

Several studies have shown that changes in biodiversity, taxonomic composition, feeding habits and community structure caused by fishing impact can be detected on a short-term scale (de Groot 1984; Currie and Parry 1996; Kaiser and Spencer 1996b; Pranovi and Giovanardi 1994). However, few studies have investigated the time required for the recovery of a disturbed community structure (Thrush et al. 1995; Kaiser et al. 1998).

In Portugal, bivalve fishing has played an important economic and social role since 1969. Bivalves are caught with a towed dredge that penetrates the sediment for up to 50 cm depth (Gaspar 1996a). Previous studies showed a negative impact in the benthic ecosystem caused by this fishing (Gaspar et al. 1994). However, recovery after the clam-dredging impact was not analysed. Thus, the aim of this study was to assess the damage and recovery of meiofauna and macrofauna benthic communities, after an intensive dredge fishing episode.

Methods

In September 2000, three replicate areas (each 625 m²) off Lagos (South Portugal: Figure 1) were intensively dredged for several hours, using traditional dredges and procedures. *In situ* corer and quadrat samples were randomly collected before and after the fishing disturbance in the sampled areas. Divers then collected samples after 24, 48 and 72 hours, 1 and 2 weeks, and 1 and 2 months from the three replicate impacted areas and from three other replicate non-fished control areas.

Sampling for macrofauna was accomplished by collection of 15 cm deep sediment samples in 0.0625 m² quadrats, using 10 replicates per area. Each of these replicates was sieved using 1 mm mesh, and the retained portion sorted by species.

Sampling for meiofauna was accomplished by collection of 0.001 m² cores, 15 cm into the sediment, using 10 replicates per area. The resulting samples were sieved between 150 µm and 1 mm mesh, and sorted by phylum or order. All samples were preserved in 90% ethanol.

For data analysis, we used four univariate measures of community structure: (1) the number of individuals per square metre; (2) the number of taxa; (3) the Shannon-Wiener diversity index (H') (Daget 1979); and (4) the evenness of distribution. To detect statistically significant differences (using $p < 0.05$) between samples taken before and immediately after dredging, the four univariate measures for

meiofauna and macrofauna were tested using one-way ANOVA (F test). The taxa contributing to dissimilarities between before and immediately after dredging were investigated using the similarity percentages procedure (SIMPER). Statistical analyses were accomplished with the use of STATISTICA V.5 (StatSoft) and PRIMER 5 software packages.

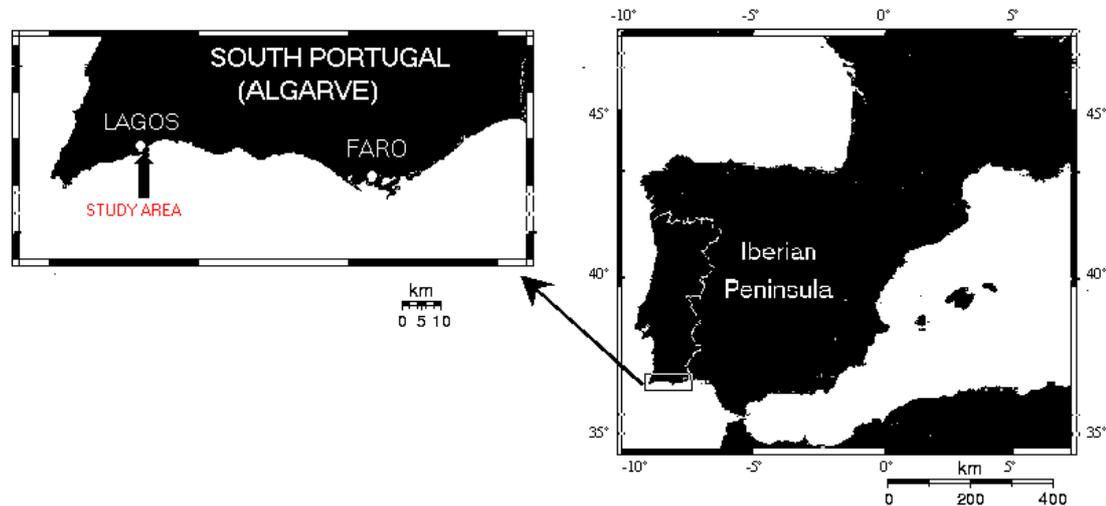


Figure 1 – Geographic localisation of the study area.

Results

The results showed a significant decrease in macrofauna abundance, numbers of taxa and diversity, and in meiofauna abundance and number of taxa, immediately after dredging (Figure 2 and Table 1). There was evidence of a greater impact to macrofauna than to meiofauna. This was shown by (1) non-significant changes in the meiofauna diversity index, (2) lower p -levels obtained for changes in the abundance of macrofauna (Table 1), and (3) higher average dissimilarities in macrofauna (71.17) than in meiofauna (25.77: Table 2).

The meiofauna taxa that contributed most to the dissimilarity between pre- and post-dredging samples were nematodes, Syllidae (polychaetes), copepods, oligochaetes and tanaids, respectively. The macrofauna species were *Siphonoecetes striatus*, *Diogenes pugilator*, *Branchiostoma lanceolatum*, *Urothoe poseidonis* and *Divaricela divaricata*, respectively (Table 2).

The meiofauna community structure had recovered by eight days after dredging. Concerning the macrofauna, two months later, seasonal changes had occurred and the effects of the fishing disturbance were no longer evident (Figure 2).

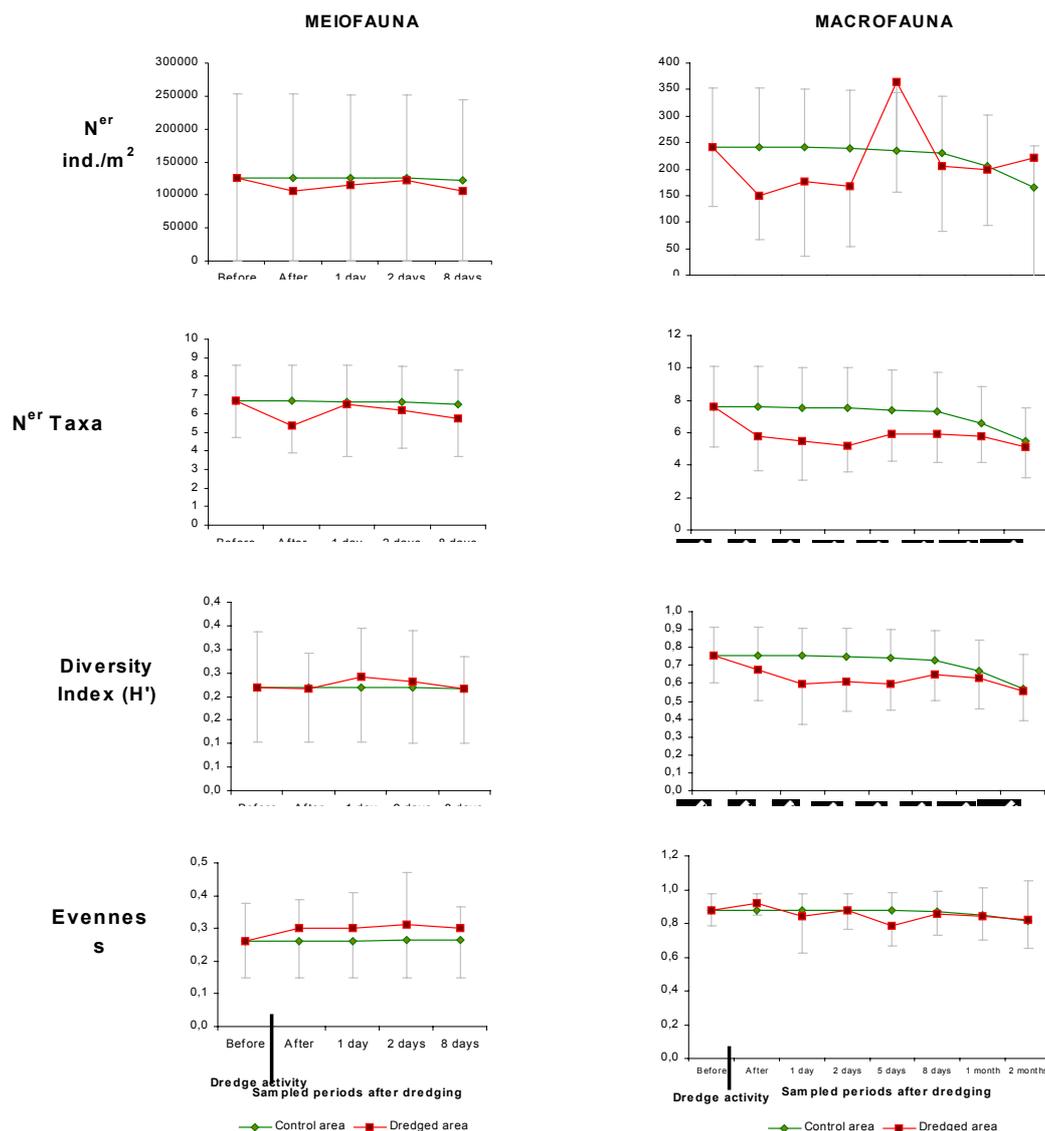


Figure 2. Changes and recovery of meiofauna and macrofauna due to dredging.

Table 1. *p*-level of eight one-way ANOVA (fixed effects) between pre-and post-dredging data for (1) number of individuals per square metre, (2) number of taxa, (3) Shannon-Wiener diversity index (H'), and (4) evenness of distribution; for meiofauna and macrofauna. * Significantly different ($p < 0.05$).

	N^{er} ind./m ²	N^{er} Taxa	Diversity Index (H')	Evenness
MEIOFAUNA	0.0281*	0.0049*	0,8882	0,1554
MACROFAUNA	0.0006*	0.0033*	0.0492*	0,0754

Table 2. Species contribution to the dissimilarities between before and immediately after dredging.

MEIOFAUN Average dissimilarity = 25.77			MACROFAU Average dissimilarity = 71.17		
TAXA	Contribution %	Cumulative contrib. %	TAXA	Contrib. %	Cum. %
Nematoda 1	73,35	73,35	<i>Siphonocetes striatus</i>	15,12	15,12
Syllidae	5,42	78,77	<i>Diogenes pugilator</i>	14,76	29,89
Copepoda 1	3,61	82,38	<i>Branchiostoma lanceolatum</i>	10,69	40,58
Copepoda 2	3,37	85,75	<i>Urothoe poseidonis</i>	10,33	50,90
Oligochaeta	3,13	88,88	<i>Divaricela divaricata</i>	5,53	56,43
Tanaidacea	2,12	91,00	<i>Ampelisca brevicornis</i>	4,08	60,51
Nematoda 2	1,86	92,86	<i>Echinocardium cordatum</i>	3,82	64,33
Turbelaria	1,57	94,43	Nephtyidae	3,28	67,61
Ostracoda	1,04	95,47	n. id. sipunculo	2,92	70,53
Nemertinea	1,03	96,50	<i>Chamelea gallina</i>	2,28	72,81

Discussion

The results of this study show that Portuguese clam dredging activity causes short-term changes in the benthic community structure in mobile sediments. Similar results in other world regions were observed by Bergman and Hup (1992) and Pranovi et al. (2000), using similar equipment.

While the meiofauna abundance and numbers of taxa revealed significant changes immediately after dredging, we consider that these changes reflect sediment redistribution rather than actual changes to the benthic communities, as organisms may be too small to suffer any damage from dredge disturbance. Thus, we believe that the impact of fishing on meiofauna is trivial, and this is corroborated by the fast recovery shown in our results.

The period during which our experiment took place, from September to November 2000, was relatively calm with only small weather disturbances (no storms), and was after the reproductive season of most of the invertebrate species sampled. However, according to Kaiser et al. (1998), the effects of fishing disturbance may last longer depending on whether or not they coincide with the peak settlement periods of benthic invertebrate larvae. Conversely, the effects may be negligible during periods of frequent natural disturbance, such as during winter storms. In agreement with; Kaiser et al. (1998), the implications of these secondary effects of dredging on seabed habitats suggest that these effects should be integrated into fisheries management strategies.

3.1.5.5 ECOLOGICAL CHARACTERISATION OF DREDGED AND NON-DREDGED BIVALVE FISHING AREAS OFF SOUTH PORTUGAL

Introduction

Commercial fishing is one of the greatest human impacts on the marine benthic ecosystem, as benthic habitats are disturbed when fishing gear is dragged across the seabed. The environmental effects of shellfish dredging have been receiving special attention throughout the world during recent decades (e.g. Caddy 1973; Conner and Simpson 1979; Eleftheriou and Robertson 1992; Hall et al. 1993; McLoughlin et al. 1991; Dare et al. 1993; Jennings and Kaiser 1998; Gilkinson et al. 1999b; Hall-Spencer and Moore 2000).

In Portugal, a dredge fishery has exploited clam populations since 1969, and currently has great socio-economic importance. Target species are *Spisula solida*, *Donax trunculus*, *Venus striatula*, *Pharus legumen* and *Ensis siliqua*. In the Algarve (South Portugal), 57 bivalve fishing boats with a maximum 100.8 HP are licenced. The dredges used are heavy iron structures with a net bag and a toothed lower bar at the mouth. The teeth are 10–50 cm long, depending upon the target species, and act as a rake when the dredge is dragged over the sea bottom (Gaspar 1996a).

Despite most studies having shown that it is possible to detect short-term changes in community structure in response to fishing disturbance (e.g. de Groot 1984; Currie and Parry 1996; Kaiser and Spencer 1996b), long-term changes (see Frid et al. 1999) are more difficult to evaluate and are consequently scarce in the published literature. When an area has been continuously fished for several decades, it can be difficult to distinguish between changes in the community caused by fisheries disturbance and those caused by natural phenomena (Currie & Parry, 1996). Kaiser et al. (1998) have referred to the importance of evaluating the ecological relevance of fishing disturbance versus natural perturbations, which will vary between different habitats. To analyse these long-term effects, comparisons are usually conducted between areas that are fished and non-fished, but interpretation can be difficult as non-fished areas usually differ physically from commercial fishing grounds. A variant on this approach is to examine fishing effects in an area that has been closed to fishing (Tuck et al. 1998; Bradshaw et al. 2000).

Traditionally, bivalve fishing grounds have been overexploited, and it is necessary to take measures to mitigate reductions in target species abundance. In the Algarve region these measures include TACs, minimum mesh size (25 mm) and closure periods (from May 1 until June 15 annually). This closure aims to protect spawning individuals. In heavily exploited areas, longer periods of closure could be needed. However, the complete closure of an entire fishing area can have high socio-economic impact. As an alternative, temporary closure of fishing sub-areas to allow their recovery can be a useful tool for fishery management.

In this study, we compare an area that had not been fished for four years (Lagos) with a continuously fished area (Vilamoura), and discuss differences in benthic macrofaunal and meiofaunal diversity, abundance, and biomass, as well as in the relative importance of feeding groups.

Methodologies

Study site

The Algarve coast (south Portugal) extends from Cabo São Vicente (8°59'W) in the west to the border with Spain in the east (7°24'W) (Figure 1). The continental shelf off Algarve is narrow. Both study sites—the continuously fished area, Vilamoura (37°05'N, 8°2'W), and the recently non-fished area, Lagos (37°08'N, 8°6'W)—are located in the western part of the Algarve coast. Drift currents ($\approx 0.25\text{--}0.50\text{ m s}^{-1}$, (Fiúza 1983) run westwards parallel to the shore at 30 m depth and predominate over tidal currents. The two locations have similar hydrological (water temperature, salinity, pH, etc), depth (7 to 9 m) and sea bottom characteristics (sandy sediment in both areas) (Moita 1986; Dias 1987; Teixeira 2000; Vieira et al. 2000). Due to the low depth of the sampling sites, changes in sediment characteristics (granulometry, organic content, etc) may occur as a consequence of weather conditions.

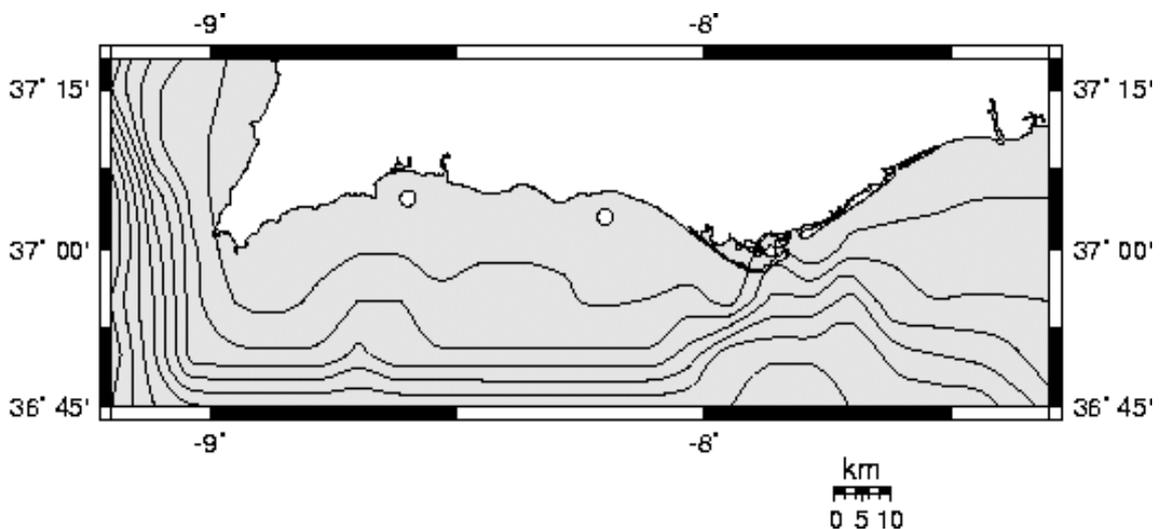


Figure 1. The South Portugal (Algarve) coast, showing location of the studied areas: Vilamoura (continuously fished area) and Lagos site (recently non-fished area).

Experimental design and sampling

Samples were collected at 7–9 m water depth during September 1999. At both study sites, 3 areas (approximately 50 m x 50 m) were analysed. At each of these replicated areas 18 quadrats ($0.0625\text{ m}^2 \times 0.15\text{ m}$) for macrofauna, and 12 cores ($0.001\text{ m}^2 \times 0.15\text{ m}$) for meiofauna were randomly placed and sampled by SCUBA divers. The sediment in the area of the quadrats was carefully dug with a spade into 1mm-mesh bags. Corers samples were sealed *in situ*, after collection. The number of replicates used was determined after considering a previous study in the same areas performed by Chícharo et al. (2000), and following Elliot (1977): $N = s^2 \times (m \times D)^{-2}$, where: N is the number of replicates, s^2 is the sample variance, m is the mean density per sample unit, and D is the sample precision (accuracy = S.E./m). Because spatial aggregation of benthic organisms changes over time and space, variance is highly sensitive to mean density (Downing 1989). To reduce this variability, sampling variance was calculated from the empirical algorithm for marine benthos by Veniza (1988):

$s^2 = 1.641 \times m^{1.219}$, where m is the mean density per sample. Macrofauna samples were sieved immediately after collection, on the surface, through 1mm-mesh bags.

Laboratory procedures

Meiofauna samples were sieved through a 150 μ m mesh. Both macrofauna and meiofauna samples were preserved in 70% ethanol. Organisms were sorted to the lowest possible taxonomic level and counted. Ash free dry weight was calculated for bivalve biomass determinations. These animals were dried at 60°C until constant weight was reached (at least 48 h) and then burned at 450°C for three hours in a muffle-furnace. Linear measurements of the less abundant species of bivalves, gastropods and echinoderms were recorded and used for macrofauna biomass calculations, following Gamito (1994) and, Sprung (1994a), Sprung (1994b). Meiofaunal biomass was calculated following Sprung (1994b), Banse (1982) and Knox (1986).

Data analysis

Species diversity was calculated using the Shannon-Wiener Index, H' (Daget 1979). Significant differences in meiofaunal and macrofaunal diversity and trophic groups between areas were analysed using one-way ANOVA (F test) ($P < 0.05$), performed with the Statistic V.5 software package. The specific contribution of macrofauna species to the observed differences in Bray-Curtis dissimilarity between sites was analysed using the SIMPER routine ("Similarity percentages") of the PRIMER (Plymouth Routines in Multivariate Ecological Research) software package (Clarke & Warwick, 1994). The selected value for the percentage cut-off was 100.0. Analyses were carried out using square-root transformations to normalize the data.

Categorization of macrofauna (Table 1) and meiofauna (Table 2) into trophic groups was based on Feder (1981), Commito and Shrader (1985), Rield (1986), Bemvenuti (1994), Gamito (1994), McKillup and McKillup (1994), Sprung (1993, 1994b); Fish and Fish (1996) and Skoeld and Rosenberg (1996). We considered carnivores and omnivores to be potential scavengers, as proposed by Lindeboom and de Groot (1998). Many species can exhibit different modes of feeding in particular situations, depending, for instance, on their nutritional condition. Taxa were assigned to one trophic group according to the most usual (more frequent) feeding habit they exhibit (based on literature references). Thus, macrofauna taxa that may act as both suspension-feeders and deposit-feeders were considered to be preferentially suspension-feeders and not deposit-feeders. Also, as it is difficult to differentiate meiofaunal scavengers from deposit-feeders, we only consider a deposit-feeding group and a carnivorous group. Herbivores include grazers, some of which can be detritivores.

Results

Benthic communities

The number of macrofaunal taxa found at the two study sites was 44 and 59 at Vilamoura and Lagos, respectively. Owing to difficulties in specific identification of meiofauna taxa, only nine different taxa were identified in the fished area and 15 in the recently non-fished area.

Table 1 – Trophic categorisation of macrofauna taxa at the continuously fished (Vilamoura) and the recently non-fished (Lagos) study areas: DF – Deposit-feeder; SF – Suspension-feeder; C – Carnivore; HV – Herbivore; SC – Scavenger. For calculation purposes taxa were assigned to the most common feeding habit (the first listed, when several are indicated).

TAXA	Feeding habits	Occurrence	
		Lagos	Vilamoura
MACROFAUNA			
<i>Nemertina</i>			
Nemertinea n.id.	Sc,C	x	
Sipuncula			
Sipuncula n.id.	D	x	
Annelida			
Polychaeta			
Aphroditidae	S,D	x	
Capitellidae	D	x	
Cirratulidae	D	x	
Eunicidae	D	x	x
Phyllodocidae	C	x	
Glyceridae	S	x	
Hesionidae	C	x	
Magelonidae	C	x	
Maldanidae	S,D	x	
Nephtydidae	Sc,C	x	x
Nereidae	O	x	x
Opheliidae	D	x	
Orbiniidae	D	x	x
Oweniidae	S	x	x
Sabellaridae	S	x	
Sabellidae	S	x	
Spionidae	S,D	x	
Terebellidae	D	x	
Mollusca			
Polyplacophora			
Poliplacophora n.id.	H		x
Bivalvia			
<i>Abra alba</i>			
<i>Abra ovata</i>	D	x	
<i>Acanthocardia aculeata</i>			
<i>Acanthocardia spinosa</i>			
<i>Acanthocardia tuberculata</i>	S,D	x	x
<i>Acanthocardia</i> sp.			
<i>Anomia ephippium</i>	S,D		x
<i>Chamelea gallina</i>	S	x	x
<i>Callista chione</i>	S,D	x	
<i>Clausinella brongniartii</i>	S,D		x
<i>Corbula gibba</i>	S	x	

TAXA	Feeding habits	Occurrence	
		Lagos	Vilamoura
MACROFAUNA			
<i>Donax semistriatus</i>	S	x	
<i>Donax trunculus</i>	S	x	
<i>Donax variegatus</i>	S	x	
<i>Donax vittatus</i>	S		x
<i>Dosinea exoleta</i>	S,D		x
<i>Ensis ensis</i>	S	x	
<i>Ensis siliqua</i>	S	x	
<i>Pharus legumen</i>	D	x	
<i>Loripes lucinalis</i>	S,D		x
<i>Lutraria anguistor</i>	S	x	
<i>Lyonsia norwegica</i>	S		x
<i>Mactra corallina</i>	S	x	x
<i>Mactra glauca</i>	S		x
<i>Mesalia brevia</i>	S,D	x	
<i>Modiolus modiolus</i>	S		x
<i>Musculus costulatus</i>	S		x
<i>Ostrea edulis</i>	S,D		x
<i>Pandora sp.</i>	S,D	x	
<i>Pecten sp.</i>	S,D	x	
<i>Petricola lithophaga</i>	S,D		x
<i>Pharus legumen</i>	D	x	
<i>Scrobicularia plana</i>	S,D	x	
<i>Spisula elliptica</i>	S		x
<i>Spisula solida</i>	S		x
<i>Tellina tenuis</i>	S,D		x
<i>Tellina incarnata</i>	S,D		x
<i>Thracia pubescens</i>	S,D		x
Gastropoda			
<i>Bulla striata</i>	H,D	x	
<i>Calyptrea chinensis</i>	Sc,C,D		x
<i>Cymbium olla</i>	D		x
<i>Gibberula miliaria</i>	H,D	x	
<i>Natica prietoi</i>	Sc,C	x	x
<i>Turritella communis</i>	S	x	
Crustacea			
Cirripedia			
<i>Balanus perforatus</i>	S		x
Ostracoda			
Ostracoda n.id.	H,D	x	
Cumacea			
<i>Pseudocuma longicornis</i>	D	x	
Tanaidacea			
<i>Apseudes latreillei</i>	D	x	
Isopoda			
<i>Eurydice pulchra</i>	C		x
Isopoda n.id.	H,D	x	x

TAXA	Feeding habits	Occurrence	
MACROFAUNA		Lagos	Vilamoura
Amphipoda			
Ampeliscidae	H,D		x
Podoceridae	H,D	x	x
Haustoriidae	H,D	x	x

Table 2. Trophic categorisation of meiofauna taxa at the continuously fished (Vilamoura) and the recently non-fished (Lagos) areas: DF – Deposit-feeder; SF – Suspension-feeder; C – Carnivore; HV – Herbivore; SC – Scavenger. For calculation purposes taxa were assigned to the most common feeding habit (the first listed, when several are indicated).

TAXA	Feeding habits	Occurrence	
MACROFAUNA		Lagos	Vilamoura
Decapoda			
Natantia			
<i>Processa parva</i>	Sc,C	x	x
<i>Crangon crangon</i>	Sc,C	x	x
Raptantia			
<i>Crangon crangon</i>	Sc,C	x	x
<i>Diogenes pugilator</i>	Sc,C		x
<i>Liocarcinus pusillus</i>	Sc,C		x
<i>Parthenope massena</i>	Sc,C		x
<i>Pirimela denticulata</i>	Sc,C		x
<i>Thia scutellata</i>	Sc,C		x
Echinodermata			
Asteroidea			
<i>Astropecten irregularis</i>	C		x
Ophiuroidea			
<i>Amphiura mediterranea</i>	S	x	x
<i>Ophiopsila annulosa</i>	Sc,C		x
<i>Ophiura texturata</i>	Sc	x	x
Echinoidea			
<i>Echinocardium cordatum</i>	D	x	x
<i>Psamechinus miliaris</i>	Sc		x
Hemichordata			
<i>Branchiostoma lanceolatum</i>	S	x	X

The mean abundance of macrofauna was higher in the recently non-fished area, 12.198 ± 4.439 individuals.m⁻², than in the fished area, 4.006 ± 1.328 individuals.m⁻². Similarly, mean meiofauna abundance was higher in Lagos, 49216.67 ± 43975.9 individuals.m⁻², than in Vilamoura, 41638 ± 34594.15 individuals.m⁻². However, mean biomass of macrofauna was higher in the fished area, 0.648 ± 0.558 g m⁻², than in the non-fished area, 0.612 ± 0.343 g m⁻². In contrast, mean biomass of meiofauna was higher in Lagos, 4.906 ± 4.663 g m⁻², than in Vilamoura, 0.128 ± 0.008 g m⁻².

Meiofaunal diversity was higher in the fished area ($H' = 0.516 \pm 0.093$) than in the non-fished area ($H' = 0.352 \pm 0.091$), whereas macrofaunal diversity was higher in the non-fished area ($H' = 1.014 \pm 0.136$) in comparison with the fished area ($H' = 0.583 \pm 0.137$) (Figure 2). One-way ANOVA results revealed significant differences in meiofaunal ($F(1,22) = 18.97$; $P < 0.0003$) and macrofaunal ($F(1,34) = 90.028$; $P < 0.0001$) diversity between the two areas. However, no significant differences ($P < 0.05$) were observed between abundance and biomass, for macrofauna and meiofauna, in both areas.

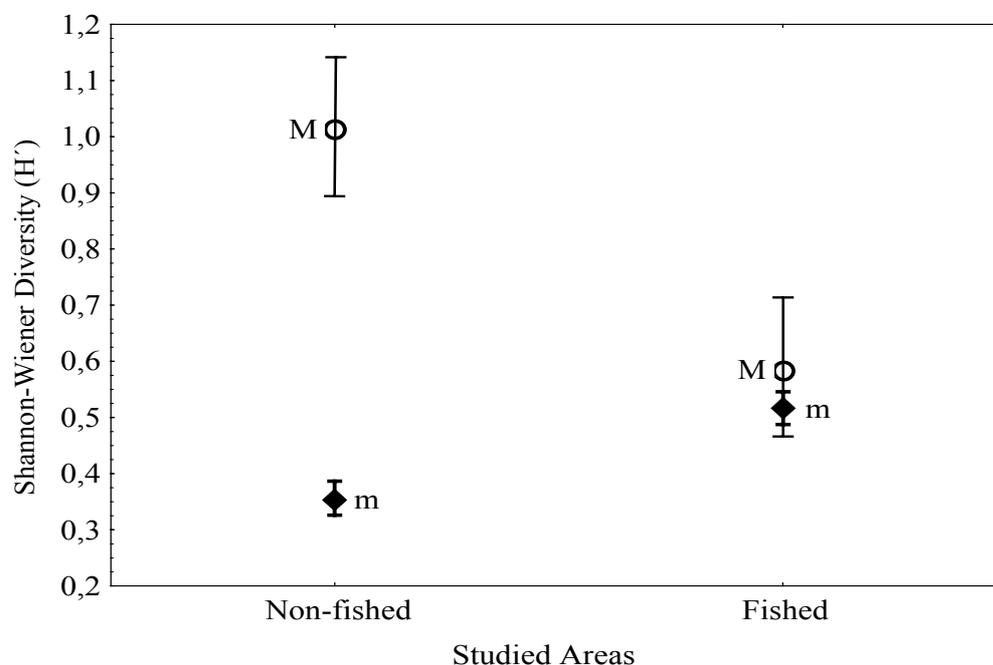


Figure 2. Macrofauna (M) and meiofauna (m) Shannon-Wiener (H') diversity calculated for the continuously fished (Vilamoura) and the recently non-fished (Lagos) site. Error bars are standard errors. Both macrofauna and meiofauna diversity between sites were significantly different ($P < 0.05$).

The Bray-Curtis dissimilarity value for the comparison between the fished and recently non-fished areas was 87.82%, as resulted from the SIMPER analysis. The taxa that most contributed to this value were Ampeliscidea, *Amphiura mediterranea*, Haustoriidae, Nemertinea and *Diogenes pugilator* (more abundant in the non-fished area) and *Spisula solida* (more abundant in the fished area). The recently non-fished area was dominated by Ampeliscidea and *Amphiura mediterranea*, but Haustoriidae, Nemertinea and *Diogenes pugilator* were also abundant. *Spisula solida* and Nephtydidae were the most abundant taxa in the continuously fished area (Table 3).

Table 3. Average abundance of macrofauna at the continuously fished (Vilamoura) and the recently non-fished (Lagos) study sites determined by SIMPER Bray-Curtis dissimilarity calculations.

TAXA	FISHED	UNFISHED	Cumulative percentage
	(Vilamoura)	(Lagos)	
	Average Abundance		
Ampeliscidea	0.00	11.79	10.63
<i>Amphiura mediterranea</i>	0.67	8.93	18.99
<i>Spisula solida</i>	5.74	0.00	25.07
Haustoriidae	3.52	5.82	29.99
Nemertinea	0.00	5.35	34.88
<i>Diogenes pugilator</i>	0.00	5.01	39.69
Capitellidae	0.00	4.95	44.26
Podoceridae	1.61	4.72	48.30
Eunicidae	0.00	3.52	51.81
Nephtydididae	3.64	3.20	55.24
Sipunculo 2	0.00	3.38	58.22
Nereidae	3.22	1.52	61.19
Lysianassidae	0.67	3.25	64.14
<i>Apseudes latreillei</i>	0.00	3.36	67.00
Cirratulidae	2.65	1.07	69.87
<i>Venerupis rhomboides</i>	0.00	3.22	72.43
<i>Processa parva</i>	0.67	1.30	74.09
Amphictenidae	0.00	1.43	75.47
<i>Ophiura texturata</i>	0.94	0.54	76.79
<i>Echinocardium cordatum</i>	0.00	0.89	78.05
<i>Eurydice pulchra</i>	0.00	1.59	79.27
<i>Divaricela divaricata</i>	0.00	1.27	80.37
<i>Gibberula miliaria</i>	0.00	1.44	81.47
Maldanidae	0.00	1.07	82.52
Sipunculo 1	0.00	1.21	83.54
<i>Branchiostoma lanceolatum</i>	0.67	0.44	84.55
<i>Chamelea gallina</i>	0.00	0.89	85.46
<i>Acanthocardia tuberculata</i>	0.00	0.67	86.35

Feeding habits

In the continuously fished area, suspension-feeders were the most abundant trophic group (63 individuals.m⁻²), followed by scavengers (51.04 individuals.m⁻²) and herbivores (37.31 individuals.m⁻²), and deposit-feeders were the least abundant trophic group (24.64 individuals.m⁻²) (Figure 3). Similarly, deposit-feeders were the least abundant trophic group in the non-fished area (97.92 individuals/m²), whereas herbivores were the most abundant (308.88 individuals.m⁻²), followed by suspension-feeders (177.84 individuals.m⁻²) and scavengers (135.36 individuals.m⁻²). There were significant differences between the abundance of herbivores at the two study sites ($F=(1,20)= 5.623$; $p=0.028$).

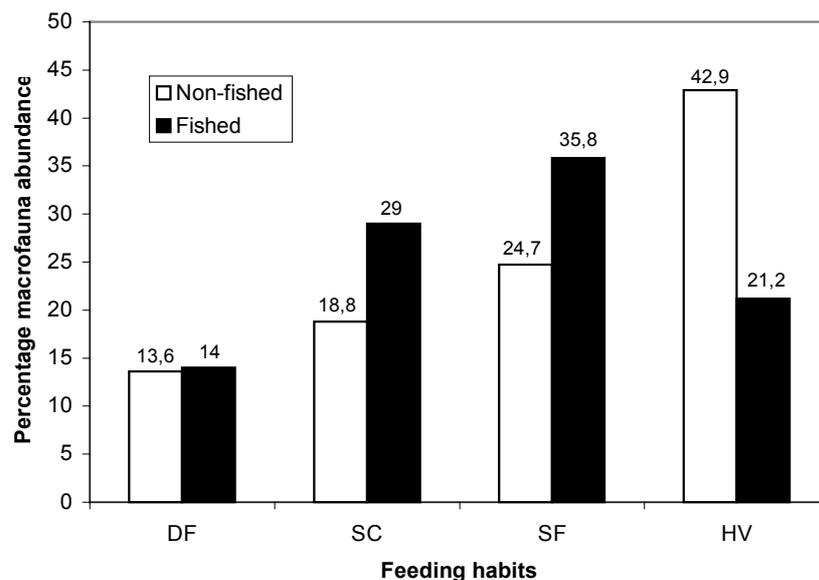


Figure 3. Percentage macrofaunal abundance at the continuously fished (Vilamoura) and the recently non-fished (Lagos) site by major trophic group: DF – deposit-feeder; SC – scavenger; SF – suspension-feeder; and HV – herbivore. Error bars are standard errors. For calculation purposes taxa were assigned to the most common feeding habit (the first listed, when several are indicated, in Table 1). (* significant differences $P<0.05$).

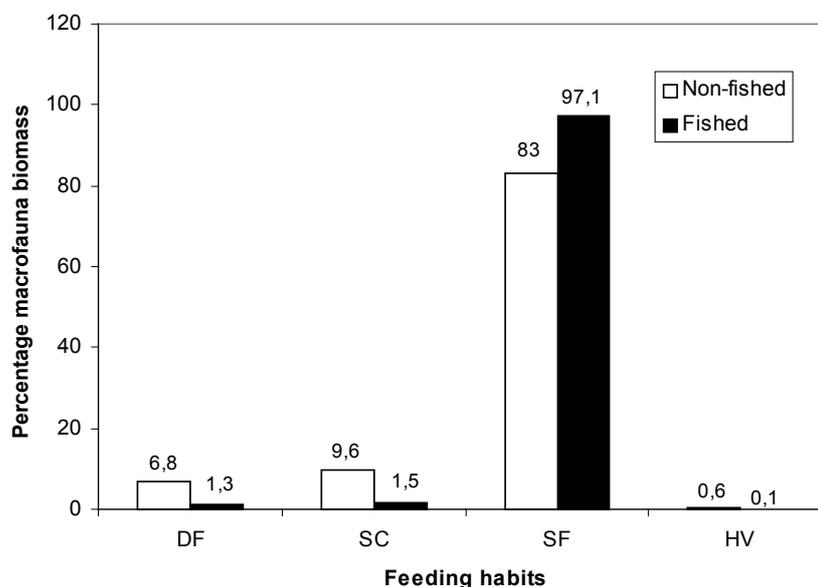


Figure 4. Percentage macrofaunal biomass at the continuously fished (Vilamoura) and the recently non-fished (Lagos) site by major trophic group: DF – deposit-feeder; SC – scavenger; SF – suspension-feeder; and HV – herbivore. Error bars are standard errors. For calculation purposes taxa were assigned to the most common feeding habit (the first listed, when several are indicated, in Table 1).

When considering total macrofaunal biomass, the suspension-feeders trophic group was the most important, both at the fished area (27.680 g m⁻²) and the recently non-fished area (29.968 g m⁻²) (Figure 4). In the fished area, Vilamoura, the biomass of the other feeding groups were 0.428 g m⁻² for deposit-feeders, 0.384 g m⁻² for scavengers, and 0.028 g m⁻² for herbivores. At Lagos, scavengers represented 3.493 g m⁻², and deposit-feeders and herbivores contributed with 2.466 g m⁻² and 0.201 g m⁻² of the total biomass, respectively. The most abundant meiofaunal trophic group was the deposit-feeders, followed in order by the suspension-feeders, carnivores and herbivores. Deposit-feeders total abundance at Lagos and Vilamoura was 696,169.75 individuals.m⁻² and 315,914 individuals.m⁻², respectively (percentages for each group in Figure 5).

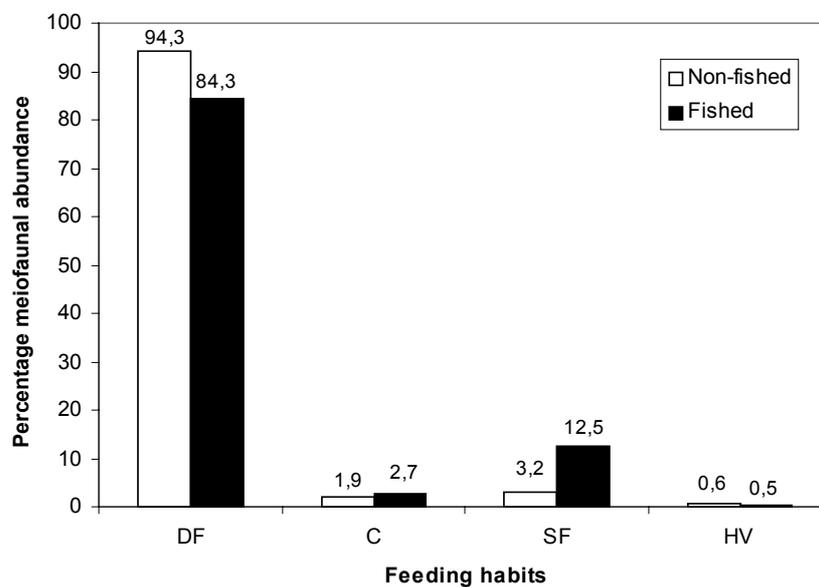


Figure 5. Percentage meiofaunal abundance at the continuously fished (Vilamoura) and the recently non-fished (Lagos) site by major trophic group: DF – deposit-feeder; C – carnivores; SF – suspension-feeder; and HV – herbivore. Error bars are standard errors. For calculation purposes taxa were assigned to the most common feeding habit (the first listed, when several are indicated, in Table 1).

However, when considering the biomass of meiofaunal trophic groups, clear differences between Lagos and Vilamoura were observed (percentages for each group in Figure 6). Whereas in the recently non-fished area (Lagos) the total biomass of deposit-feeders was 50.122 g m⁻², at Vilamoura the most important trophic group was the carnivores (1.110 g m⁻²), followed by the deposit-feeders (0.029 g m⁻²), herbivores (0.014 g m⁻²) and suspension-feeders (0.006 g m⁻²). At Lagos, carnivores made up 12.88 g m⁻² of the total biomass, suspension-feeders 10.009 g m⁻² and herbivores 0.589 g m⁻². There were significant differences only between the biomass of deposit-feeders ($F= (1,27)=4.455$; $P=0.044$) and carnivores ($F= (1,22)=11.412$; $P=0.003$) at both study sites.

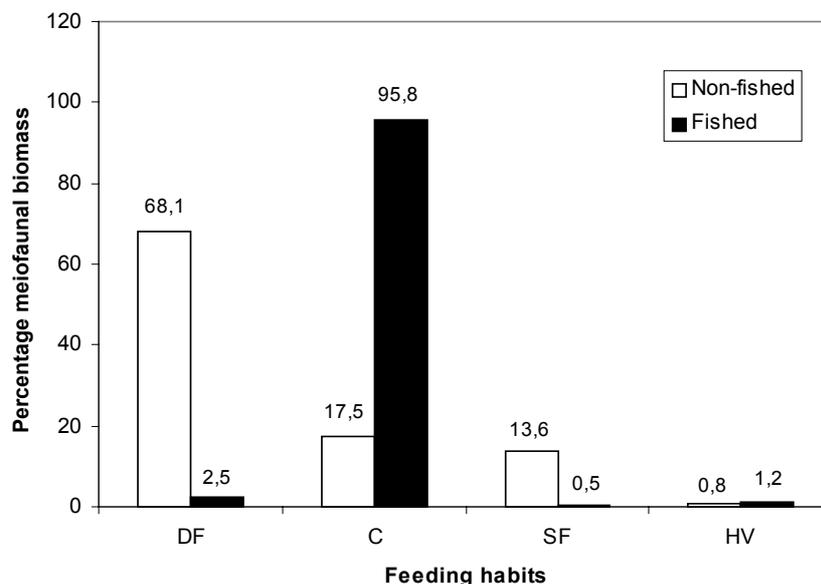


Figure 6. Percentage meiofaunal biomass at the continuously fished (Vilamoura) and the recently non-fished (Lagos) site by major trophic group: DF – deposit-feeder; C – carnivores; SF – suspension-feeder; and HV – herbivore. Error bars are standard errors. For calculation purposes taxa were assigned to the most common feeding habit (the first listed, when several are indicated, in Table 1).

Discussion

Differences of abundance, biomass, diversity and dominant trophic groups within macro- and meiobenthic communities were observed between the fished and recently non-fished locations. Macrofauna abundance and diversity were higher in the non-fished area (Lagos). The difference in these community parameters may be attributed to the continuous passage of the dredge teeth on the bottom in the fished area, damaging and killing mostly macrofauna species that are unable to pass between the teeth. In fact, fragile species like polychaetes, namely Nephtyidae, Nereidae and Cirratulidae, were still very abundant in the fished area. Larger animals, such as the bivalve *Acanthocardium tuberculata* (more than 60 mm) or fragile species like the Nemertinea, Sipunculidae or the sea urchin *Echinocardium cordatum*, live only at the non-fished area. Thus, the continuous fishing impact in the Vilamoura area probably resulted in a dominance of *r*-selection opportunistic short-lived species such as polychaetes, and a decrease of long-living sessile species such as sea urchins. Similar observations were described by Commito (1982), Bemvenuti (1994), Rumohr et al. (1998), Mortimer et al. (1999), Christensen et al. (2000) and Jennings et al. (2001).

Biomass of macrofauna species was higher in the fished area, as a result of the abundance of *S. solida* at the area. Meiofauna in the fished and non-fished area was dominated by Nematoda, Copepoda and Polychaeta. Higher values of abundance and biomass were observed at the recently non-fished area (Lagos). Despite this,

Shannon-Weiner diversity was lower in this area, owing to the dominance of Nematoda, which represented 90% of total meiofauna.

Demersal fishing activities provide food for scavengers in the form of damaged animals, which are left in the dredge track (Ramsay et al. 1998). The continuous fishing impact reduced the abundance of herbivores and increased the abundance of suspension-feeders and scavengers, as evidenced by Bray-Curtis dissimilarity results. The larger number of herbivores in the recently non-fished area, mainly amphipods, may be related to the presence of macroalgae that are unable to grow on the impacted area. Abundance of suspension-feeding and scavenger taxa, in the fished site, could be related to the input of dead organic material and to the increase of turbidity caused by the dredge.

Meiofauna are believed to be highly selective feeders with distinct and often highly specialised food niches (Kennedy 1994). Analysis of meiofauna as a potential indicator of anthropogenic perturbation in aquatic ecosystems has often been limited to monitoring surveys for pollution (Pranovi et al. 2000). However, analysis of the meiofauna may also reveal the existence of dredge disturbance, as seems to occur in this study. In fact, the level of dredge disturbance was the factor that dramatically changed, when the two areas are compared. Both temporal (seasonal) and spatial natural fluctuations of abundance and biomass in the two areas, even despite differences that may exist between them, are not expected to justify the magnitude of the changes observed. It may therefore be assumed that the observed differences in this study, in terms of biodiversity, relative abundance and biomass of feeding groups, and in the relation between opportunistic short-living and long-living sessile species, can be attributed to the different levels of fishing impact. Since there are no previous data available on the characteristics of the two areas before fishing activity began forty years ago, we are not able to positively state that the differences between the two sites are due to fishing or that the non-fished area has reached its climax. However, observed changes indicate a trend and we would recommend monitoring the Lagos area for a period of two more years before allowing fishing activities. Therefore, based on our results, we anticipate a period of 6 years for the recovery of the previously fished area. Closing areas for such a long period of time could be a management solution but is difficult to implement because of social and economic pressure by fishermen. An integrated approach, considering the contribution of restocking methods, could be useful in allowing a faster recovery of fishing impacted communities.

3.1.5.6 EFFECTS OF HYDRAULIC DREDGING ON THE BENTHIC COMMUNITY IN THE NORTH ADRIATIC SEA

Introduction

Fishing activities are known to produce changes in the structure of marine habitats, strongly influencing composition, diversity, biomass and productivity of the associated biota. The effects of trawls and dredges on infauna and epifauna, and related trophic interactions, have been recently reviewed by Jennings and Kaiser (1998). Moreover, the impact of bottom fishing-gear on the benthic community has been demonstrated in many studies carried out in various marine environments (Bergman and Hup 1992; Hall 1994; Kaiser and Spencer 1996b). Areas disturbed by fishing activity showed reduced numbers of organisms, biomass, species richness and diversity (Collie et al. 1997). In a study area around the Isle of Man (UK), Kaiser et al. (2000) found that "chronic" fishing activities caused a shift from communities dominated by sessile, emergent, high biomass species to communities dominated by infaunal, smaller-bodied organisms. Removal of emergent fauna led to increased homogeneity of seabed habitats in the areas of high fishing effort.

Short-term effects of fishing can be detected in animals encountering gears, captured and subsequently discarded. They may suffer extensive physical damage, with a consequent lower survival capability and an increased probability of attracting predators. High levels of mortality in certain benthic species may cause changes in community structure (Kaiser and Spencer 1995; Jennings and Kaiser 1998).

Recently, some studies were also carried out in the Adriatic Sea, one of the most exploited areas in the Mediterranean. In particular, the effects of Rapido trawling for scallops, *Pecten jacobaeus*, was assessed, highlighting the selective removal of a large proportion of the benthos; of the 78 taxa caught, lethal mechanical damage varied from <10% in resilient animals, such as hermit crabs, to >50% in soft-bodied organism, such as tunicates (Hall-Spencer et al. 1999b). In the lagoon of Venice, hydraulic dredging for clams (*Tapes* spp.) caused significant differences in total abundance and biomass, some of them persisting for a few months, as well as long-term effects on biocenoses, related to changes in sediment particle size and to the mechanical action of the dredge on marine Phanerogames (i.e. *Zostera* spp.) (Pranovi and Giovanardi 1994).

The North-Western Adriatic area investigated in this study is in the infra-littoral belt (Perès and Picard 1964) more precisely located in the "pre-lagoonal" zone, as described by Vatova (1940). It extends slightly more than one-and-a-half miles from the Venetian coast, and varies in depth between 3 and 12 m. The most representative biocenoses is *Chamelea gallina* + *Owenia fusiformis* (Vatova, 1940), growing on sediments made up mainly of fine, well-calibrated (SFBC) sands.

As part of the ECODREDGE Project, the composition and structure of the macrobenthic community was studied on a seasonal basis in order to establish their present status and then to determine changes due to the impact of fishing activities. The two transects considered in this study were placed in front of Venice-Lido and in front of Jesolo, running from the coastline out to sea for a maximum distance of 2 km, in an area of the seabed exploited by a hydraulic dredge fishery for the clam *C.*

gallina (3-10 m). Four sampling stations were established along each transect, at progressive depths of 3, 5, 8 and 10 m (Figure 1).

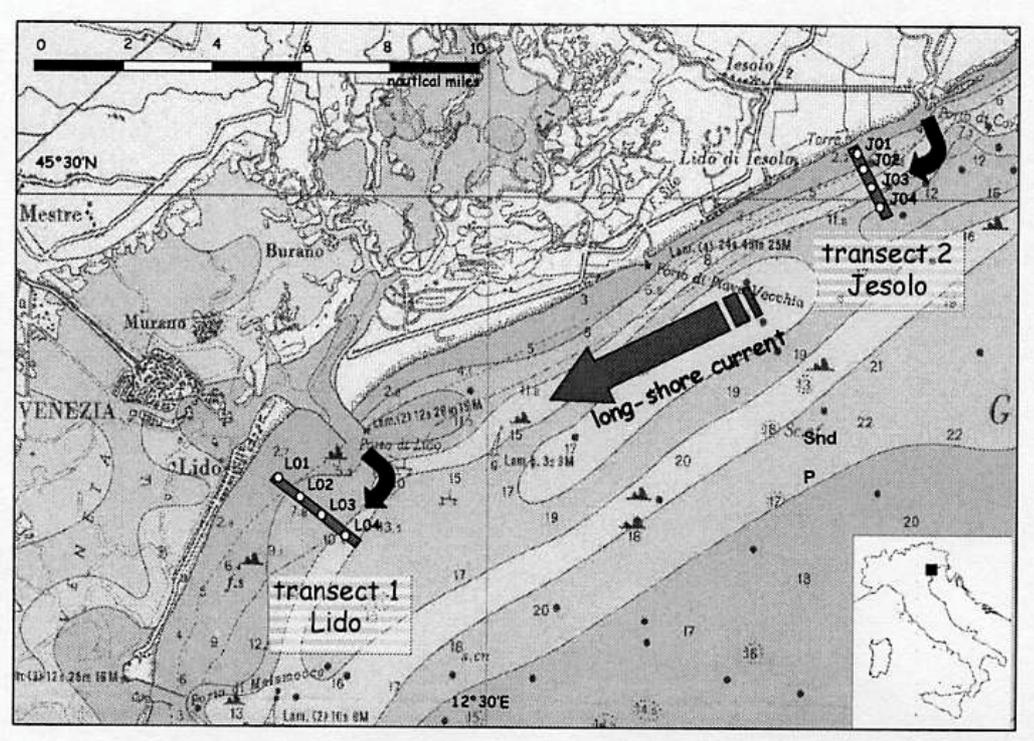


Figure 1. North-western Adriatic coast: location of the two transects.

Methodologies

Sampling

Sampling was carried out in June 1999, February, May and July 2000. An additional sampling was also made in October 2000 but only for station 2 of each transect.

For each site, five random replicate samples were collected in an area of about 10 m² using a van Veen grab of 347 cm² sampling area: the total area sampled was therefore 0.1 m². The choice of sampling methodology was determined by the features of the sediment, which is mostly sandy-silt: in this context the van Veen grab is considered to be the most suitable sampling equipment in comparison with the Ekman grab or sediment corers (Kutty and Desai 1968).

Macrozoobenthos was also collected during commercial clam dredging, both at low (1 atm) and high (3 atm) water pressure, applying a double net cover inside the mouth of the hydraulic dredge. The dredge tow was 100 m long, and the dredged area was of 26 m² at Jesolo and of 38 m² at Lido. An additional sample was collected from the by-catch.

In August 2000, further samples were taken at station 2 at Lido in order to assess the short-term effects of dredging by evaluating the qualitative and quantitative impoverishment of the benthic community. These samples were collected by divers using box-corers of 76.94 cm² sampling area: three replicates before dredging and other six after dredging, three in the dredge track and three on the edge of the furrow.

Immediately after collection, each sample was put into a plastic bag and fixed with formaldehyde 4%, buffered with $\text{Na}_2\text{B}_4\text{O}_7 \cdot 10 \text{H}_2\text{O}$.

All samples were sieved in the laboratory, using different mesh screens (from 3150 to 315 μm , Endecotts Test Sieve – London) to separate the organisms according to their size.

Sediment analyses

Sediment samples were analysed to determine grain-size composition (Buchanan 1984) and organic matter content (Gaudette et al. 1974).

Determination of abundance and biomass

Abundance (individuals.m⁻²) was calculated for each taxon on the basis of the mean of three replicates both in grab and cover samples. For biomass determination, all individuals in each taxon were oven dried (105°C for 24 hr) on pre-washed, dried and weighted fibreglass filters. Before measuring dry weight, the samples were allowed to cool in a desiccator for 12 hr.

Evaluation of damage on non-target species

The physical damage to benthic invertebrates collected using hydraulic dredging, at low (LP) and high (HP) water pressure, was assessed and compared with damage levels in grab-collected organisms. Observations concerned body integrity and, for molluscs, shell integrity; damage was expressed as percentage of organisms belonging to one of the following three categories (undamaged, not severely damaged/identifiable, severely damaged/unidentifiable) for polychaetes and crustaceans, and two categories for molluscs (undamaged and broken shell).

Additional samples were collected from by-catch to evaluate damage due to dredging and mechanised sorting on non-target species.

Data analysis

Multivariate analyses were performed by calculating Bray/Curtis similarity indices (Bray and Curtis 1957) using PRIMER (Plymouth Routines in Multivariate Ecological Research) software. Damage frequencies were compared using the G-test (Sokal and Rohlf 1981).

Results

Sediment analysis

At Lido, the sediment grain size at stations 1 and 2 was similar, with a prevailing component of medium-sized sand, whereas stations 3 and 4 showed increasing percentages of fine sand and silt. At Jesolo, station 4 differed from the other stations, being mostly silty; moreover, at a similar depth, the sediments of the seabed were generally finer at Jesolo than at Lido (Figure 2).

Sediment organic matter was more abundant at Jesolo than at Lido (Figure 3). In both cases, however, there was an increase in gradient from the coast towards the sea. The

high variation intervals observed in Jesolo station 3 and Lido station 4, indicated considerable seasonal variability.

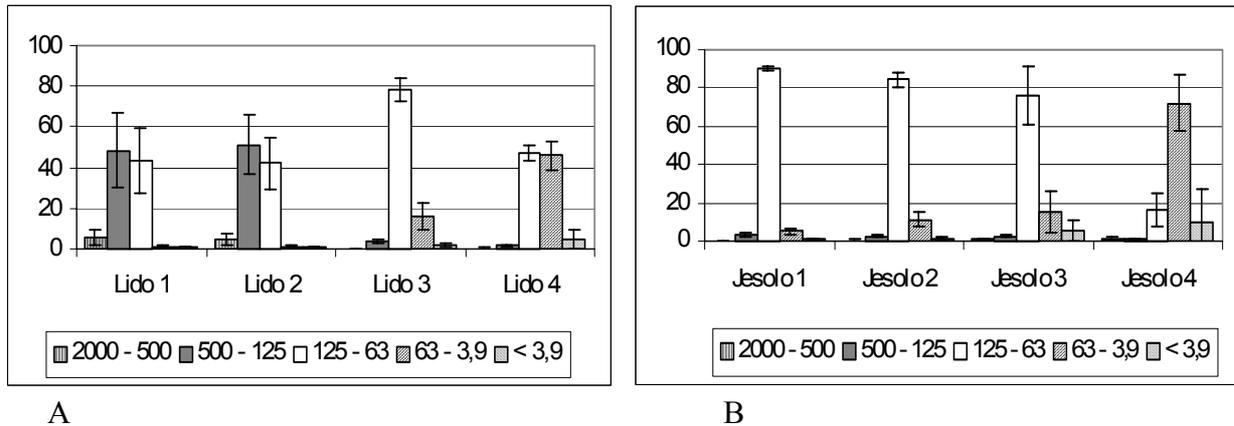


Figure 2. Mean sediment grain-size composition: transect Lido (A), transect Jesolo (B).

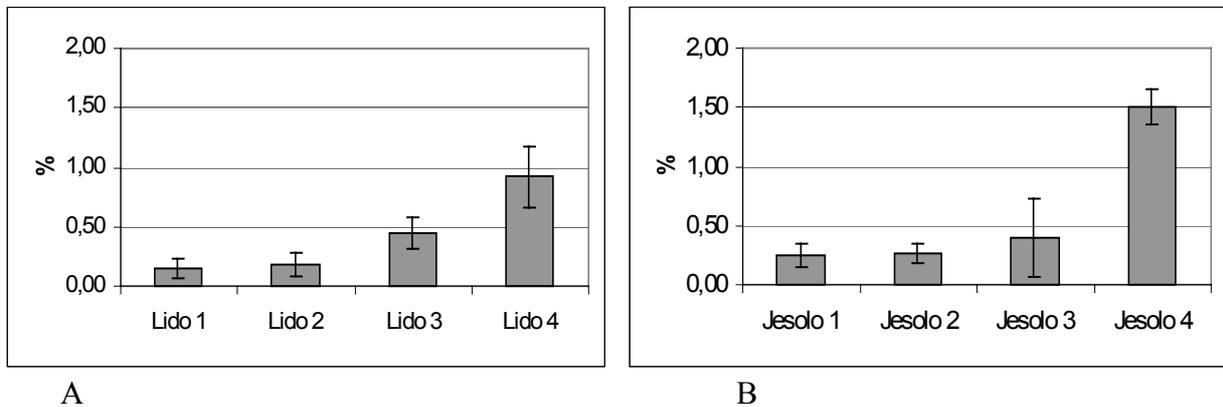


Figure 3. Mean sediment organic matter content in the stations of Lido transect (A) and Jesolo transect (B)

Determination of abundance and biomass

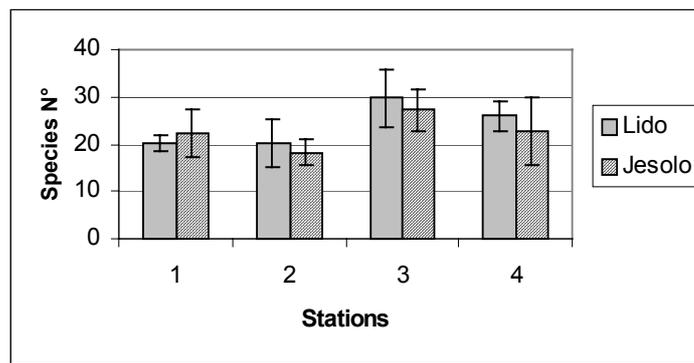
The benthic community in the whole area investigated was mainly composed of polychaetes, crustaceans and molluscs. Individuals belonging to other groups, like echinoderms, were occasionally observed in by-catch samples, but they are not reported in Table 1 where all the taxa found are listed. In both transects, station 3 (8 m depth, with a sandy-muddy sediment) was the richest in species and individuals (Figure 4).

Within the whole community, polychaetes were the most representative group, for both the number of species (Figure 5) and population density (Figure 6). The same pattern was not observed for biomass, especially in June-July, when molluscs prevailed, being individually larger (Figure 7). Analysing the seasonal trends of the main groups examined, differences were found between transects: polychaetes reached their maximum density at Jesolo in February and at Lido in June, while crustaceans were more abundant in July at Jesolo and in May at Lido.

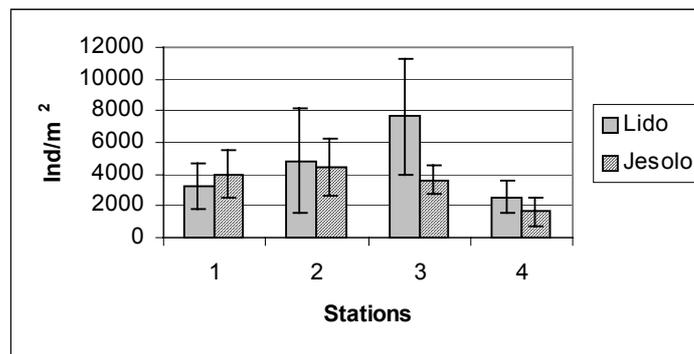
The similarity dendrogram performed with the biological data clearly highlighted two groups of stations, the first containing stations 1 and 2 of both transects, the second stations 3 and 4 (Figure 8A). However, if abiotic parameters are included in the multivariate analysis, station 3 of Lido is separated from station 3 of Jesolo and more similar to the stations of the first group (Figure 8B). The values of Shannon's Index, calculated for the entire community, ranged between 0.74 and 1.51 for Lido and between 0.76 and 1.59 for Jesolo, and did not reveal any particular space-time differences (Figure 9).

POLYCHAETA	MOLLUSCA	CRUSTACEA
Amphiglena mediterranea	Abra alba	Ampelisca diadema
Aonides sp.	Acanthocardia echinata	Amphilocheus brunneus
Aponuphis bilineata	Actaeon tornatilis	Amphithoe ramondi
ARABELLIDAE	Bittium reticulatum	Aora gracilis
Axiothella constricta	Buccinidae n.d.	Apseudes latreillii
Capitella capitata	Callista chione	Atylus guttatus
Chaetozone setosa	Cerastoderma glaucum	Bathyporeia guilliamsoniana
Dasybranchus caducus	Chamelea gallina	Calanoida n.d.
Drilonereis filum	Corbula gibba	Circolana sp.
Eteone sp.	Cyclope neritea	Corophium insidiosum
Euclymene oerstedii	Dentalium vulgare	Corystes cassivelaunus
Euclymene sp.	Ensis minor	Cragon crangon
Eunice vittata	Epithonium clathrus	Cypridinidae
Glycera sp.	Glycimeris sp.	Diastylis rugosa
Goniada sp.	Gourmya sp.	Harpacticoida n.d.
Heteromastus filiformis	Littorina sp.	Harpinia crenulata
Hydroides sp.	Mactra corallina	Hippomedon ambiguus
Lumbrineris sp.	Murex brandaris	individui n.d.
Magelona papillicornis	Mytilus galloprovincialis	Iphinoe serrata
Maldane sp.	Nassarius reticulatus	Jassa marmorata
Mediomastus capensis	Natica cruentata	Leptocheirus mariae
Micronephthys stammeri	Natica millepunctata	Leucothoe serraticarpa
Microspio mecznikovianus	Nucula nucleus	Lysianella della vallei
Neanthes sp.	Patella sp.	Megaluropus massiliensis
Nephtys hombergi	Sphaeronassa sp.	Microdeutopus versiculatus
Notomastus sp.	Spisula subtruncata	Microprotopus maculatus
Onuphis eremita	Tapes decussata	Nebalia sp.
Orbinia sp.	Tapes literatus	Paguridea
Owenia fusiformis	Tapes philippinarum	Perioculodes aequimanus
Perinereis cultrifera	Tapes sp.	Photis longicaudata
Phyllodoce sp.	Tellina pulchella	Processa edulis
Phylo foetida	Tellina sp.	Pseudocuma longicorne
Pirakia punctifera	Trunculariopsis trunculus	Pseudolirius kroyerii
Polydora ciliata	Venerupis sp.	Synchelidium haplocheles
Polydora sp.	Venus candida	Westwoodilla rectirostris
Prionospio caspersi	Venus sp.	Tot. 35
Prionospio cirrifera	Tot. 36	
Prionospio malmgreni		
Prionospio sp.		
Pseudobrania sp.		
SABELLIDAE		
Scoloplos armiger		
Spio sp.		
Spiophanes bombyx		
Sthenelais boa		
Tot.45		

Table 1. Total taxa found in the two areas studied

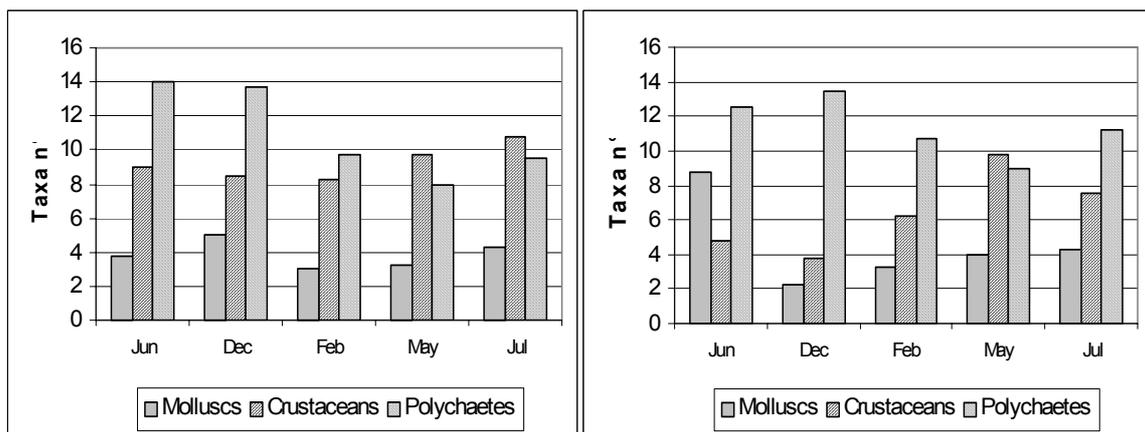


A



B

Figure 4. Variation in the number of taxa (A) and mean density of benthic invertebrates (B) along the transect of Lido and Jesolo (mean \pm s.d.).



A

B

Figure 5. Seasonal variation of taxa for each faunal group at Lido (A) and Jesolo (B).

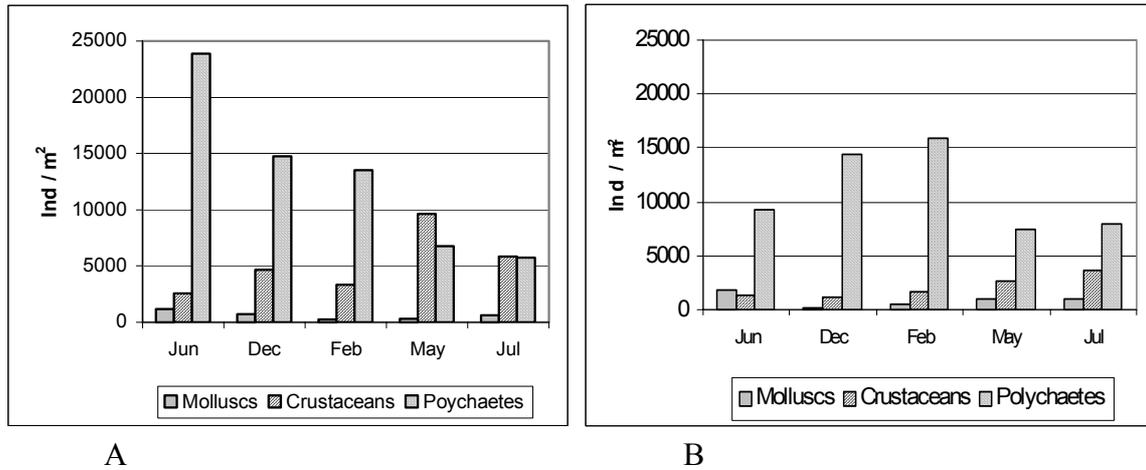


Figure 6. Seasonal variation of population density for each faunal group at Lido (A) and Jesolo (B).

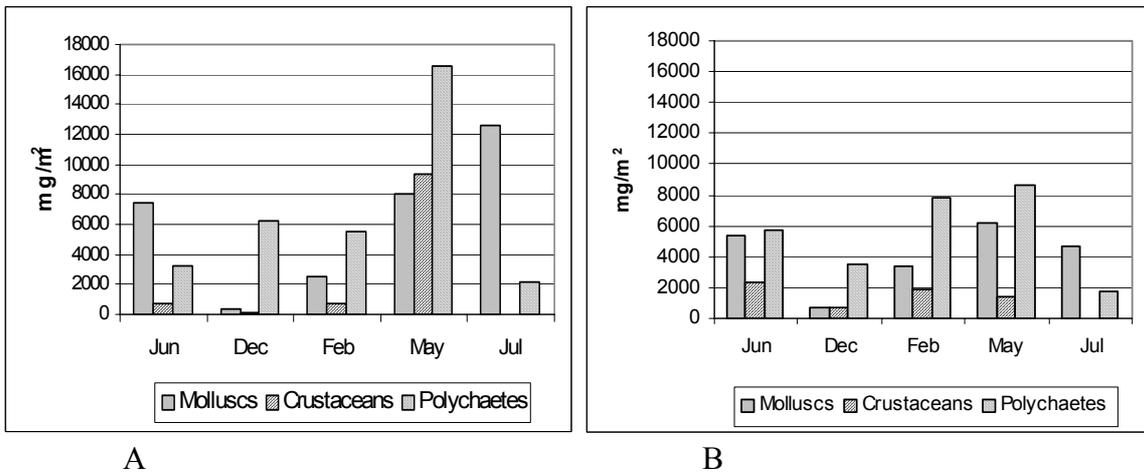


Figure 7. Seasonal variation of the biomass (mg/m dry weight) of macro-invertebrates in the whole transect of Lido (A) and Jesolo (B).

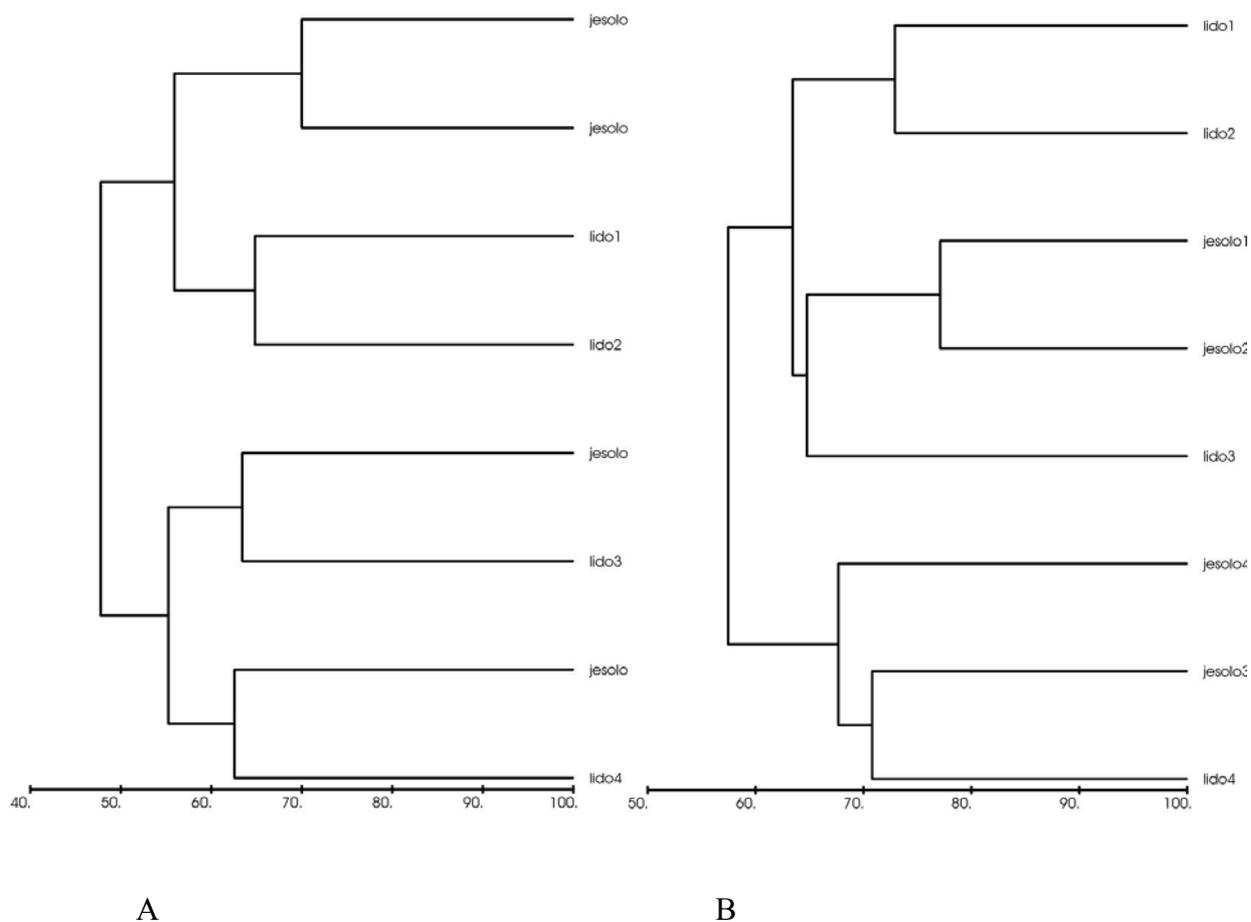


Figure 8. Dendrograms based on Bray-Curtis similarity indices for biological data only (A) and total data (chemical-physical parameters included) (B).

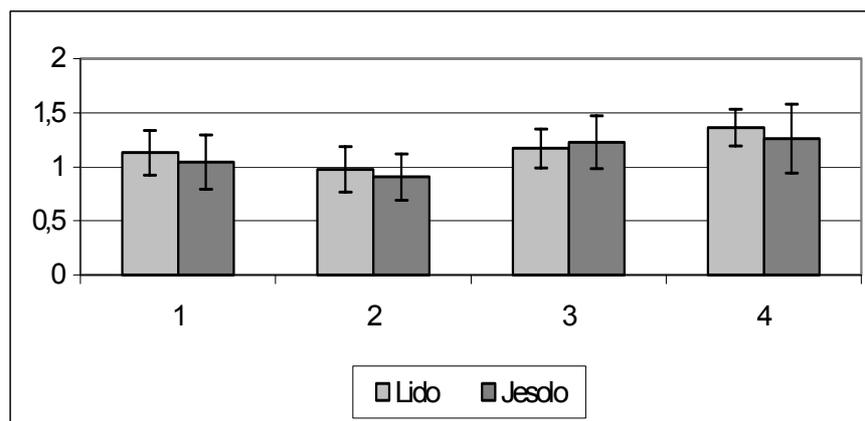


Figure 9. Mean values of Shannon Index in each station of the two transects (mean \pm s.d.).

Evaluation of damage to non-target species

Molluscs were less affected by dredging than the other taxa at both Lido and Jesolo, the percentage of damaged organisms reaching maximum values of 23% and 17%, respectively (Figures 10, 11). No significant differences were observed between van Veen grab (G) and cover collected (LP and HP) samples. A significant increase in damaged molluscs was detected in samples collected from the by-catch (bc), particularly at Jesolo ($p < 0.001$).

Crustaceans were more sensitive than molluscs to fishing impacts, showing the highest percentages of damaged animals at Lido in the February HP sample (66%) and at Jesolo in the February bc sample (22%). At Lido the number of damaged crustaceans rose with increasing water pressure, the lowest value being detected in grab samples, the highest in HP samples; however by-catch samples were less damaged than cover samples (Figure 12). At Jesolo, the percentages of damaged crustaceans were lower than at Lido and the highest values were often observed in the by-catch, being significantly different from grab samples in February and October ($p < 0.001$ and 0.01 , respectively), and from LP and HP samples in May and October ($p < 0.001$) (Figure 13).

Polychaetes were the most impacted group: at Jesolo in July 100% of animals collected using high water pressure showed physical damage. Higher percentages of damaged organisms were generally observed in HP samples at both sites (Figures 14, 15). Damage level in by-catch samples was undetectable due to the lack of this faunal component, which is mostly represented by small organisms not retained by the fishing gear. The mean values of the percentage of dredged molluscs, crustaceans and polychaetes showing physical damage are reported in Table 2.

The greater vulnerability of polychaetes was also confirmed by the analysis of the corer samples collected before and after the dredging, inside the furrow and along the

edges. Differences between before- and after-dredging samples mainly concerned the density of specimens in the various groups: the numbers of polychaetes fell by 50%, crustaceans by 24% and molluscs by 14% (Figure 16A). The number of species remained more or less unchanged for polychaetes, fell by about 23% for crustaceans and by 50% for molluscs (Figure 16B). As concerns damage evaluation, among the organisms that remained in the furrow after dredging, 51% of polychaetes and 17% of crustaceans were damaged: the increase of damage observed with respect to pre-dredging samples was significant for both polychaetes and crustaceans ($p < 0.001$ and 0.05 , respectively). Moreover, damage was significantly higher in the furrow than on the edge of the track ($p < 0.001$ for polychaetes and $p < 0.01$ for crustaceans). Molluscs were not damaged, but it must be noted that only a few species and a few specimens were found (Table 4 and Figure 17).

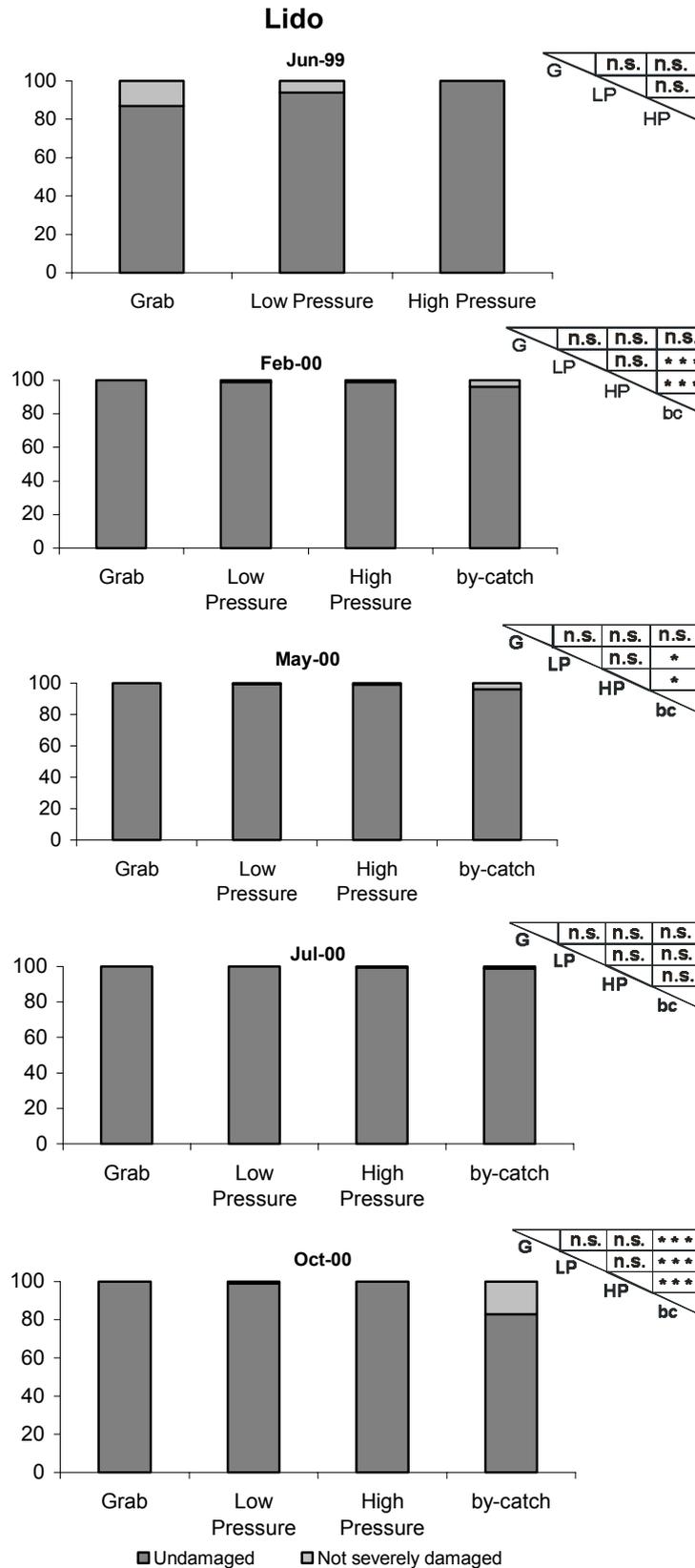


Figure 10. Mean percentages of damaged and undamaged molluscs at Lido (G= grab samples; LP= samples dredged using low water pressure; HP= samples dredged using high water pressure; bc= samples collected from the by-catch). G-test: *p<0.05; *p<0.001.**

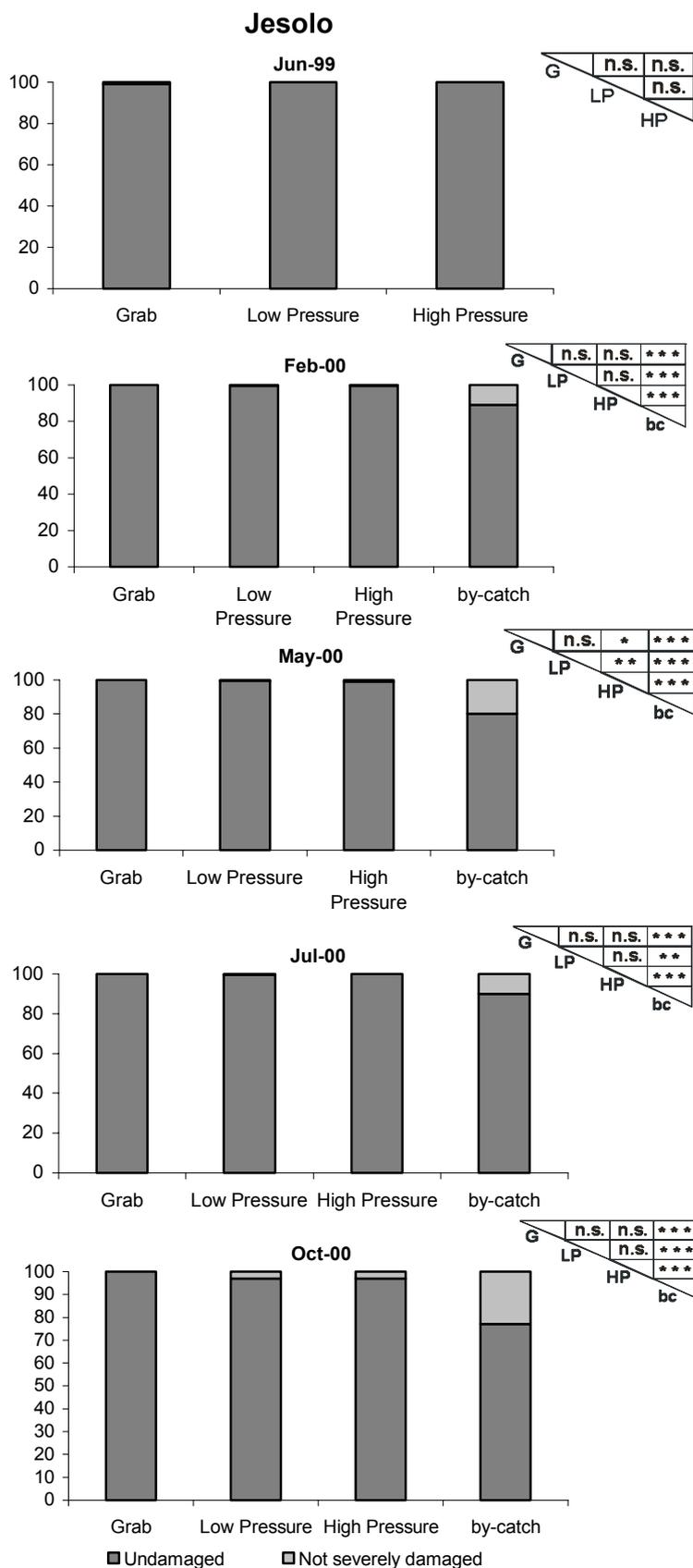


Figure 11. Mean percentages of damaged and undamaged molluscs at Jesolo (G= grab samples; LP= samples dredged using low water pressure; HP= samples dredged using high water pressure; bc= samples collected from the by-catch). G-test: *p<0.05; **p<0.01; ***p<0.001.

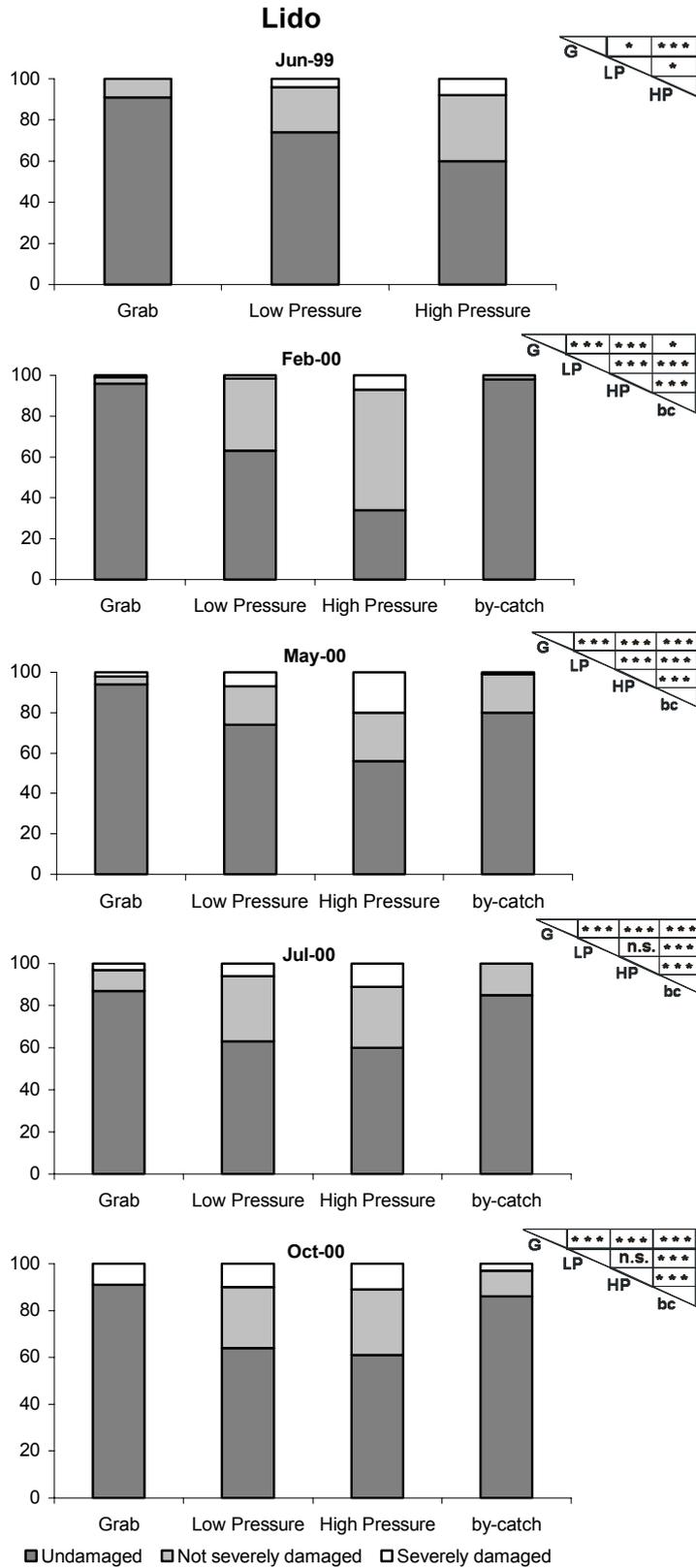


Figure 12. Mean percentages of damaged and undamaged crustaceans at Lido (G= grab samples; LP= samples dredged using low water pressure; HP= samples dredged using high water pressure; bc= samples collected from the by-catch). G-test: *p<0.05; *p<0.001.**

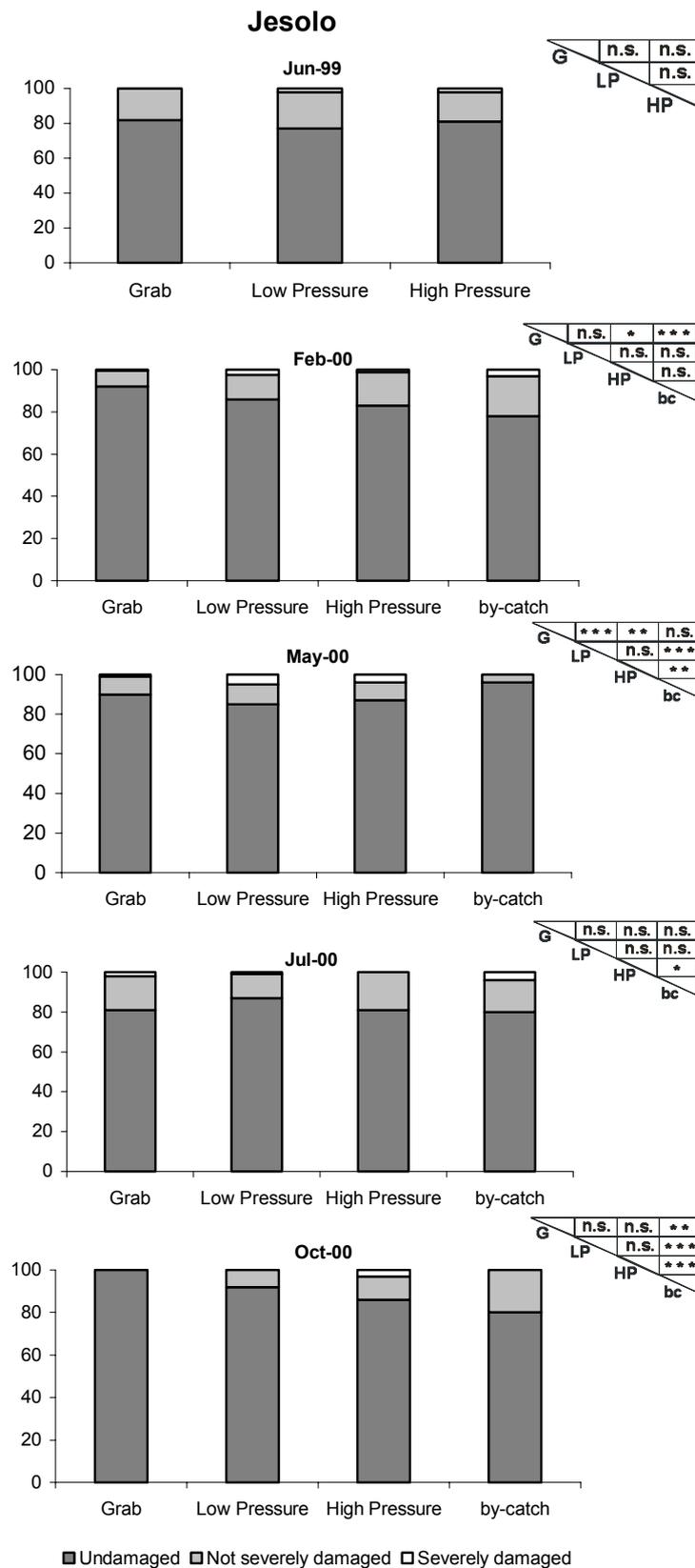


Figure 13. Mean percentages of damaged and undamaged crustaceans at Jesolo (G= grab samples; LP= samples dredged using low water pressure; HP= samples dredged using high water pressure; bc= samples collected from the by-catch). G-test: *p<0.05; **p<0.01; *p<0.001.**

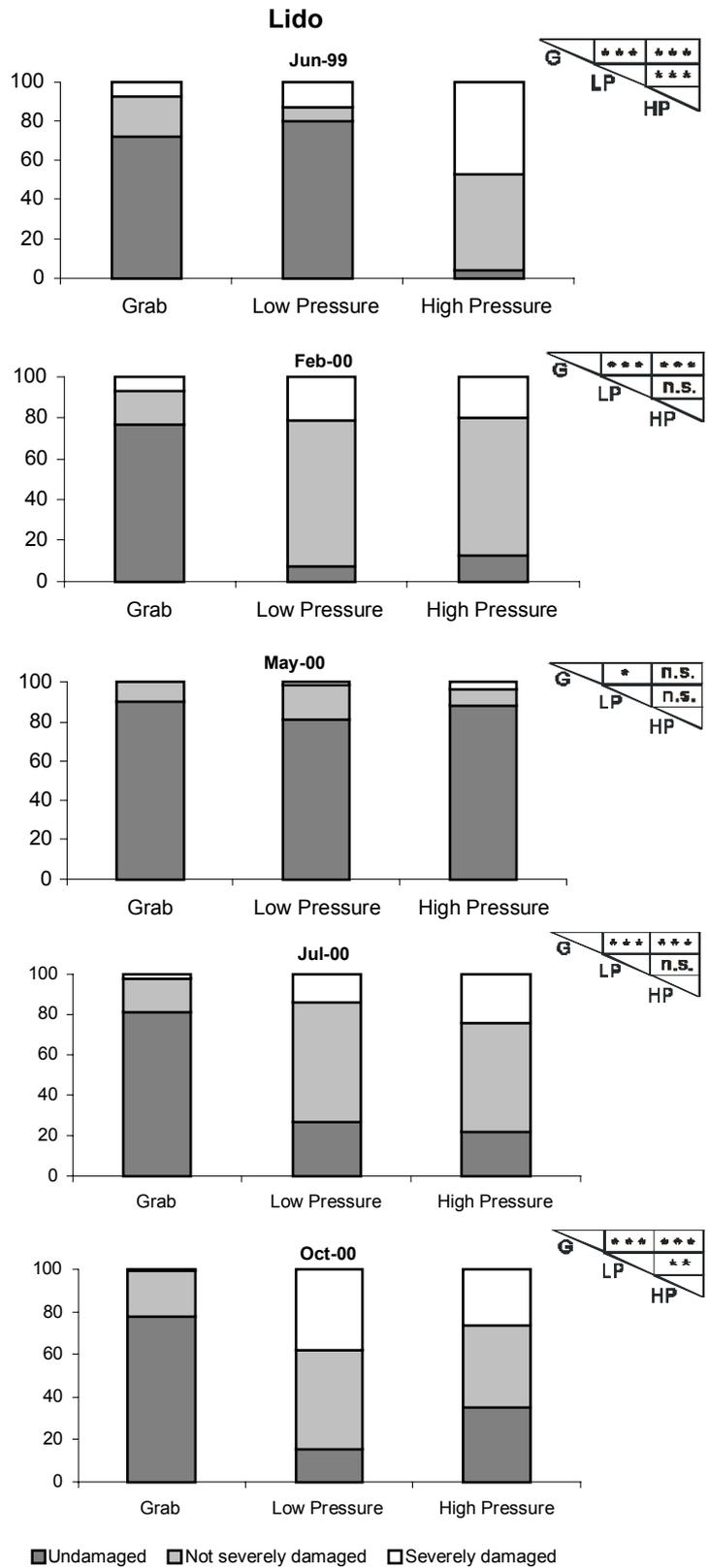


Figure 14. Mean percentages of damaged and undamaged polychaetes at Lido (G= grab samples; LP= samples dredged using low water pressure; HP= samples dredged using high water pressure; bc= samples collected from the by-catch). G-test: *p<0.05; **p<0.01; *p<0.001.**

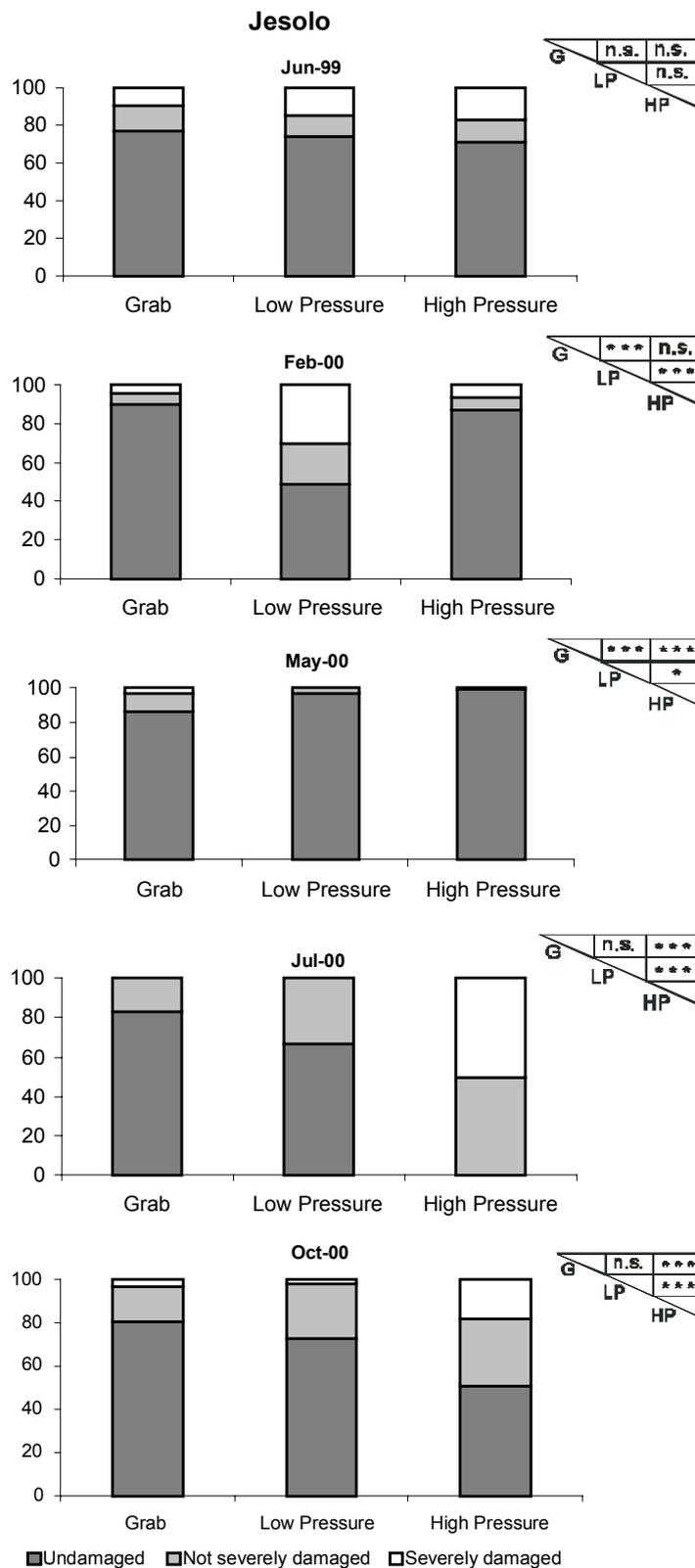
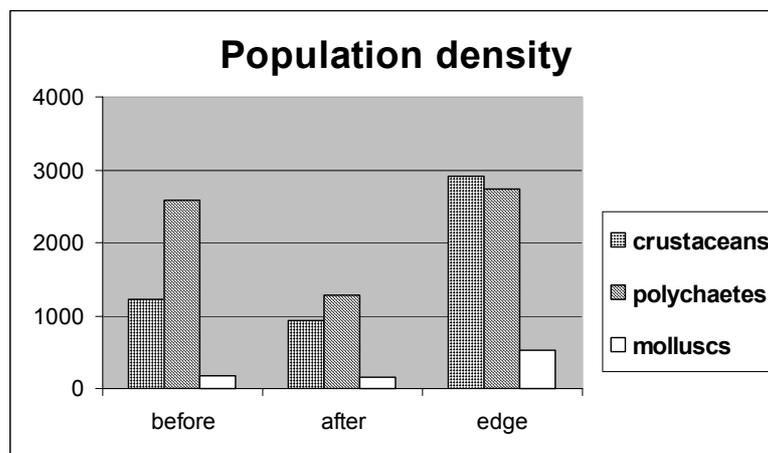


Figure 15. Mean percentages of damaged and undamaged at Jesolo (G= grab samples; LP= samples dredged using low water pressure; HP= samples dredged using high water pressure; bc= samples collected from the by-catch). G-test: *p<0.05; ***p<0.001.

Lido	Molluscs	Crustaceans	Polychaetes
Low Pressure	1%	33%	57%
High Pressure	2%	45%	68%
Jesolo	Molluscs	Crustaceans	Polychaetes
Low Pressure	1%	15%	28%
High Pressure	1%	15%	38%

Table 2. Percentages of damaged organisms of the three faunal groups in the two dredged areas.



A



B

Figure 16. Variation of richness (A) and population density (B) in samples collected in the furrow and edge after dredging.

	Molluscs	Crustaceans	Polychaetes
Track before- vs after-dredging	n.s.	*	***
Track before- vs edgeafter-dredging	n.s.	n.s.	***
Track after- vs edge after-dredging	n.s.	**	***

Table 3. Statistical comparison between before and after dredging samples. G-test: *p<0.05; **p<0.01; ***p<0.001.

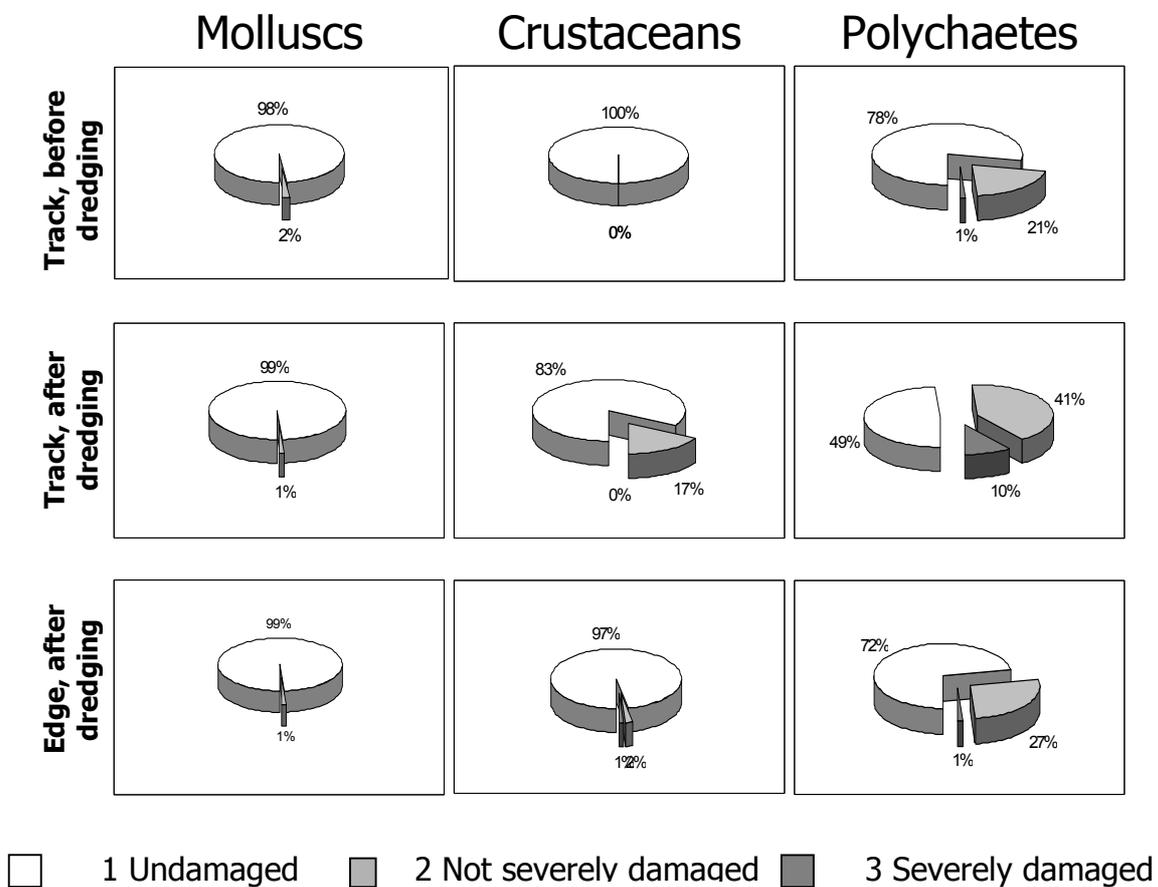


Figure 17. Percentages of damaged and non-damaged individuals collected using box-corers before and after hydraulic dredging.

Discussion

In the present study short-term effects of dredging on the benthic community were assessed by detecting physical damage to organisms that encountered the fishing gear on the seabed (cover samples) and in those captured, sorted and then discarded (by-catch samples). To obtain an indication of the basic damage level in the populations studied, samples collected with the van Veen grab were used.

Polychaetes were found only in cover collected samples, indicating that the high water pressure used during fishing did not allow these small organisms to be retained in the cage. However, they were the most affected group, reaching about 100% of damaged animals in some samples; these annelids, when they are not protected by thick tubes, are more vulnerable, confirming previous data reported for other soft-body organisms, such as tunicates, subjected to Rapido trawling (Hall-Spencer et al. 1999b). It was also observed that damage varied depending on size and body form: animals with body appendages, such as crabs, and those having a larger size were more subjected to physical impact, this condition being particularly highlighted in crustaceans and molluscs from by-catch samples.

Moreover, the comparison between Lido and Jesolo showed a significant variability in damage percentages, probably reflecting differences in both physical features of the grounds and composition of the faunal groups. Crustaceans and polychaetes were more damaged at Lido, both in low and high water pressure samples, where sediments were coarser and the amount of empty broken shells greater than at Jesolo, thus adding an abrasive effect on the animals to the mechanical impact of the dredging operation. Furthermore, the amphipod *Corophium insidiosum* was the most abundant crustacean species at Lido, whereas at Jesolo its presence was undetected.

Our results do not reveal a particular season in which the physical impact of dredging is greater. The seasonal trend in the percentage of damaged organisms differed in the various faunal groups, and inside each group, it may differ when comparing the two sampling sites. This situation could also be explained by the differences observed at Lido and Jesolo in both environmental conditions and the specific composition of faunal groups.

Evaluating chronic fishing disturbance on benthic communities is hardly feasible along the North Adriatic coast due to the heavy exploitation of the clam beds. This made our initial plan of defining a suitably wide sanctuary area impracticable, and prevented us making a comparison between fished and non-fished grounds. In this context, long-term modifications of the whole benthic community may be tentatively evaluated only if previous references are available. In the study area, decades of fishing activity have certainly changed the benthic populations, although the typical zoocenoses reported in the 1930s and 1940s by Vatova appeared to be more or less unchanged (Scardi et al. 1997). However, a more detailed examination of the quantitative presence of single species in the community may reveal great differences.

In the last few years, many studies have been carried out on Adriatic benthic communities (e.g., Orel et al. 1987, Orel et al. 1989; Albertelli et al. 1998), but the

locations and the depths explored often differed, and any comparison between different space-time situations is obviously not conclusive. As concerns the present research, the only data we can use for a comparison are those of Vatova, regarding the “prelagoonal” belt, at bottom depths ranging from 3 to 12 m (Vatova 1940; Vatova 1948; Vatova 1949 ; Vatova 1966). However, Vatova’s data have gaps concerning the collection and, to an even greater extent, the sorting methodologies. A careful examination of Vatova’s faunal lists shows that the entire “small” component was missing, including spionids, among the polychaetes, and copepods and most of the amphipods among the crustaceans, which in our samples are the most abundant components (70% of polychaetes, and 70-80% of crustaceans, respectively). This condition makes any ultimate consideration about long-term effects of dredging somewhat difficult and dubious, although normalising our data with respect to Vatova’s dataset, it clearly appears that both biomass and density have been reduced, mainly for polychaetes, which are subjected to the worst short-term effects.

The encounter with hydraulic dredges results in a great number of non-captured or captured and then discarded organisms that consequently can suffer severe physical damage. These animals will probably have a reduction in both survival capability and ability to avoid mobile predators and scavengers, as reported in previous studies (Hall-Spencer et al. 1999b; Kaiser and Spencer 1994). Moreover, the short-term effects of fishing activities include extensive and continuous changes in the structural complexity of the habitats, and the removal of a large proportion of benthic organisms, modifying the species ratio and causing long-term modifications to heavily exploited areas.

3.1.5.7 Summary Statement

The contributions presented in Section 3.1.5 cover a range of topics concerning the impact of three important European bivalve dredge fisheries on benthic communities. For each study, some of the conclusions are relevant to the particular fishery, benthic community or type of gear, while others are of more general applicability.

The study of aggregation to discarded undersized scallops in the Irish Sea scallop fishery ([Section 3.1.5.1](#)) highlighted the importance of damage level. The main predators and scavengers, *Ophiocomina nigra*, *Calliostoma lyra*, *Cancer pagurus*, *Liocarcinus* spp., spider crabs, dogfish, gadoids and flatfish (mainly plaice) all aggregated more rapidly to damaged, compared with undamaged, scallops. All dredged but undamaged scallops survived for the duration of the observations (4-7 days), while badly damaged scallops were all eaten within 24h; lightly damaged scallops survived longer but with a large variation between replicates. The crab, *Cancer pagurus*, greatly facilitates feeding by other scavengers, particularly fish, by breaking open the shell of lightly damaged scallops. This work has shown that if damage to uncaught and discarded animals can be reduced there is the potential to reduce fishing-induced incidental mortality.

Detailed investigations of the macrofauna remaining in the track after the passage of the Portuguese clam dredge ([Section 3.1.5.2](#)) recorded some statistically significant differences in species diversity, evenness, abundance and number of taxa both along and across different sections of the dredge track. These differences result from the behaviour of the gear, particularly the clogging of the mesh that occurs after about 10 m, forming a sand buffer across the dredge mouth. The spatial differences in the macrofauna were more evident at the unfished site, suggesting that the composition of the macrofauna at the fished site has become adapted to long-standing fishing impacts. These results highlight the importance of assessing gear behaviour before designing a sampling programme to study fishing impacts.

The diver-conducted study of the new Portuguese dredge ([Section 3.1.5.3](#)) found that the dredge tracks were deeper in muddy sand, compared with sand sediments, and persisted for a longer period, but this depended on the strength of water currents, tides and wave action. Few animals were left in the dredge track and mortality was low. Most undamaged animals burrowed very quickly, indicating that the stress of dredging does not greatly reduce burrowing ability. As a result, scavenger aggregation was small (mainly brittlestars) and dispersed rapidly (within 30 min). The study concludes that the new Portuguese clam dredge is very efficient and highly selective, and induces low mortality of individuals both caught by the gear and left on the track. However, there may be long-term community effects due to the damage to, and subsequent mortality, of thin-shelled bivalves and urchins over a long period.

Detailed studies of the benthos ([Section 3.1.5.5](#)) showed substantial differences in abundance, biomass, species diversity and dominant trophic group in macrofauna and meiofauna communities between a continuously fished area off southern Portugal and an area that has not been fished for 4 years. Macrofauna abundance and diversity were higher in the unfished area, but biomass was higher in the fished area due to the abundance of the large bivalve *Spisula solida*. Larger (e.g. the bivalve *Acanthocardium tuberculata*) and fragile (e.g. Nematode, Sipunculidae and the sea

urchin *Echinocardium cordatum*) species were present only in the recently unfished area. Continuous fishing appears to have resulted in the dominance of r-selection, opportunistic, short-lived species such as polychaetes, and a decrease in long-lived, sessile and fragile species, such as the sea urchins. In the meiofauna community there was a greater abundance and biomass in the unfished area but diversity was lower due to the dominance of Nematoda. Continuous fishing appears to have reduced the abundance of herbivores and increased suspension feeders and scavengers; the latter may be due to increased availability of organic material from animals killed by fishing gear, and increased turbidity. As there are no benthic community data available for these two areas before fishing began 40 years ago it is not possible to determine if the unfished area has reached its climax but it is suggested that a 6 year recovery period may be required.

An intensive episode of clam dredging in the recently unfished area (Lagos, S. Portugal) caused short-term changes in benthic community structure ([Section 3.1.5.4](#)). Meiofauna abundance and numbers of taxa decreased significantly immediately after dredging but recovered rapidly (within 8 days); these changes reflect sediment redistribution rather than actual changes to the meiofauna community. Dredging had a greater impact on the macrofauna: abundance, numbers of taxa and diversity all decreased significantly immediately after dredging and it was 2 months before seasonal changes had obscured the effects of fishing. It is concluded that seasonal factors such as invertebrate reproductive seasons and periods of winter storms should be taken into account in fisheries management strategies.

Italian studies of the effects of hydraulic dredging on macrobenthic communities at two locations in the north Adriatic ([Section 3.1.5.6](#)) show that this gear causes marked short-term changes in density and damages significant numbers of animals, both on the seabed and in the by-catch. Polychaetes were the most affected group, with population densities falling by 50% after dredging and up to 100% of individuals damaged by dredges using high water pressure. Crustacean densities fell by 24% and the numbers damaged increased with increased water pressure up to 66% in some samples. Molluscs were less affected by dredging with densities falling by 14% and damage levels of 17% and 23% recorded at the two locations. Of the animals left in the dredge track, 51% of polychaetes and 17% of crustaceans were damaged, and damage was significantly higher in the furrow than at the edge of the track. Crustaceans and polychaetes were generally more damaged at the location where the sediments were coarser and contained more broken shell. Some seasonal trends in damage level were noted for the different groups but there was no particular season in which the physical impact of dredging was greater. Overall it was noted that physical damage to benthic animals varied with size and body form: animals with appendages, such as crabs, or of a large body size were more affected, particularly in the by-catch samples. It was not possible to evaluate long-term changes due to fishing in this area because of the lack of suitable pre-fishing data or unfished areas.

These six studies show substantial community level effects of dredging for three important European bivalve fisheries. For the Irish Sea scallop fishery, the Italian clam fishery in the northern Adriatic and, to a lesser extent, the southern Portugal clam fishery, the gear and fishing techniques damage appreciable numbers of benthic animals both on the seabed and in the by-catch and there is scope for the development of less damaging fishing methods. In all three fisheries the extent and intensity of

fishing in all suitable areas is such that there is at least the potential to cause long-term, and possibly irreversible, changes in benthic community structure and this must be taken into account in fisheries management.

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3.2 Selectivity

Introduction

An understanding of gear selectivity both in terms of species and size are basic requirements for considering options for improving conservation. Such studies should include an understanding of the likely effect of selectivity measures on commercial catches. In this study of selectivity of dredges we examine the selective effects of dredge components on species and size selectivity. We also test selective dredge components in commercial practice to examine changes in selectivity as the components wear. Finally we consider how components wear during the life of the dredge and the use of three-dimensional modelling techniques in dredge selectivity design.

3.2.1. Species Selectivity

This Section presents a study of the seasonal behaviour of queen scallop (*Aequipecten opercularis*). There is also an investigation of species selectivity (king *Pecten maximus* and queen scallop *Aequipecten opercularis*) in relation to the presence or absence of teeth and the catch of queen scallop in relation to various modifications designed to catch swimming queen scallop.

Species selectivity in Portuguese clam dredges also discussed in [Section 3.2.2.2](#) although the majority of the results of this Section pertain to clam size selectivity.

3.2.1.1 SELECTIVITY OF QUEEN SCALLOP, *Aequipecten opercularis*, DREDGES IN THE IRISH SEA

Introduction

In the north Irish Sea the great scallop *Pecten maximus* is fished during the period November 1st to May 31st. During the summer closed season the majority of fishing boats target the queen scallop *Aequipecten opercularis*. Although in many cases commercial densities of these two species are spatially separated there are numerous areas where they co-occur. Owing to the different behaviour of these two scallops different gear types can be used to select the appropriate species. Features of scallop behaviour that could be exploited by gear designed to improve selectivity are associated mainly with habitat preference and mobility, especially in response to fishing gear.

Both *Pecten maximus* and *Aequipecten opercularis*, are bottom dwellers but assume different positions within the substrate ([Review Section 6](#)). *P. maximus* exhibits burying behaviour and is usually found recessed into the seabed, with the upper (left) valve level with the substrate. In contrast *Aequipecten opercularis* lies on the seabed surface. The queen scallop is generally considered a more mobile species than the great scallop. Underwater observations by divers have shown that queen scallops react to approaching fishing gear by swimming upwards in the water column (Chapman et al. 1979; Chapman 1981). The great scallop rarely shows this behaviour and owing to the recessed position of this species can only be targeted effectively using toothed gear that digs into the substratum.

The swimming activity of the queen scallop is likely to vary seasonally. Fishermen in the Irish Sea typically change from toothed dredge gear to trawls at the end of July to target this species. At this time of the year the queen scallop is considered to be more actively swimming and therefore can be targeted using bottom trawls. We aimed to assess seasonal variability in the response of queens to fishing gear in order to more accurately understand the efficiency of different gears throughout the year. This was achieved by adapting the conventional queen dredge gear to capture those individuals which avoided dredge capture by swimming upwards and passing over the dredge mouth. This gear was used approximately every 6 weeks over a 14 month period to determine the proportion of queens avoiding dredge capture. In addition individuals were transported alive to Port Erin Marine Laboratory where their swimming ability was tested using a standard methodology.

A number of different gear types have been used to target the queen scallop. However, currently three main types are utilised, the toothed dredge (essentially a modified great scallop dredge with smaller belly rings and shorter teeth), the skid dredge (a modification of the toothed dredge, with each dredge frame mounted on ski-like skids, the tooth bar removed and replaced with a tickler chain) and bottom trawls. Toothed dredges have increased potential for damaging benthic communities. In addition this type of gear can catch a bycatch of the great scallop. During the summer closed season for great scallops, this bycatch must be returned to the sea bed. Although little is known regarding the fate of discarded scallops there is the potential for mortality (Jenkins and Brand 2001 and [Section 3.1.3.1](#)). We aimed to investigate the species selectivity and damage to major bycatch species caused by toothed and

non-toothed queen gear. Also the minimum design requirements for the ‘cover’ or ‘top net’ were investigated with a view to considering a commercial version.

Methodologies

Seasonal variability in queen scallop behaviour

Two approaches were taken to determine the seasonal variability in queen scallop swimming behaviour. In the field, we determined the degree to which queens avoided dredge gear by swimming upwards in the water column. In the laboratory we assessed swimming ability directly.

A net (hereafter called the ‘top net’) was developed which could be deployed above a gang of four queen scallop dredges (Figure 1) to catch individuals which avoided dredge capture by swimming above the dredge mouth. This gear was initially used between August 8th and September 4th 2000 at 3 queen scallop fishing grounds around the Isle of Man in the north Irish Sea. At each fishing ground 3 tows of 2 nautical miles were made at a speed of between 2.5 to 3 knots (4.6-5.6 km/h). On hauling the gear the number of queen scallops and great scallops captured in each dredge and in the net were counted. A subsample of queen scallops were taken from the net and from one dredge and the shell height measured to the nearest millimetre. Following this successful trial, the same procedure was repeated at the Laxey fishing ground at between 4 and 14 week intervals over the following 14 months.

From March 2001, on each field sampling date, at least 50 queen scallops were taken alive to Port Erin Marine Lab where they were placed in running seawater tanks and allowed 7 days to recover from the dredging process. The procedure described in Jenkins and Brand (2001) and in [Section 3.1.3.1](#), was used to assess the swimming ability (response time and total number of adductions before exhaustion) of 27 queen scallops of shell height 50-75mm.

At each field sampling date from March 2001 the reproductive state of queen scallops caught in the dredges and caught in the net was assessed in 2 ways. Firstly 50 individuals from the dredges and 50 from the net were opened and the gonad staged (see Wanninayake 1994 for details). Secondly 30 queen scallops from each of the capture methods were frozen and subsequently gonad and tissue dry weight determined.

Assessment of queen gear species selectivity: Skid gear versus toothed gear

Two types of gear commonly used to target queen scallops in the Irish Sea were used in a trial to determine the level of species selectivity (i.e. how well the gear selected the queen scallop and avoided capturing the great scallop). The two gear types used were the standard queen dredge, hereafter referred to as toothed gear (see task 1 review for design) and the ‘skid dredge’. The skid dredge consists of a standard queen dredge with the tooth bar removed and replaced by a tickler chain (Figure 2). Each dredge is mounted on a pair of ski-like skids (approximately 8 cm wide by 50 cm long). Six of each of the two types of dredge were mounted on two 4.8 m long wheeled towing beams which were towed from either side of the boat.

A series of trials were carried out between the 1st and 6th of July and the 23rd and 24th of August at different queen scallop fishing grounds around the Isle of Man in the

north Irish sea on board the fishing vessel De Bounty (Table 1). The two types of gear were compared by towing for 30 minutes at a speed of 2.5-3 knots (4.6-5.6 km/h) using a warp to depth ratio of 2.5-3:1. Standard commercial fishing practices were followed throughout.

Table 1 Timing and location of trials to investigate species selectivity of different queen scallop dredge designs

Date	Location	Depth (m)	Tows
1.7.01	Port St Mary Offshore 51	80-100	3
1.7.01	Port St Mary Offshore 54	66-68	3
2.7.01	Port St Mary Offshore 51	80-100	3
2.7.01	Port St Mary Offshore 54	66-68	3
6.7.01	Laxey	26	5
23.8.01	Douglas South	40-66	5
24.8.01	Port St Mary Offshore 54	66-84	4

On hauling, the fullness of all dredges was assessed before the catch from each dredge was emptied into separate fishing boxes. Three dredges were selected at random from each gear type and sampled for queen scallops, great scallops, rock volume and invertebrate bycatch. Queen scallops were counted and a sub-sample selected at random from each gear type in which the shell height was measured. Great scallops were counted and the shell length and damage level of all individuals determined. Major species and species groups of the invertebrate bycatch were identified, counted and the level of damage assessed on a four point scale (see Jenkins et al. 2001 and [Section 3.1.4.1](#) for details). The volume of rocks caught by each of the sampled dredges was estimated.

An additional trial was carried out in which skid gear was modified in an attempt to increase catch rates of queen scallops by preventing loss over the top of gear. Two types of ‘cover net’ were designed which stretched from the dredge frames, forward to the towing chains. Two types of cover net were tried: Mark 1 without the glassfibre boom and the Mark 2 shown in Figure 3. These nets were designed to deflect queen scallops which swam upwards in the water column downwards into the dredge mouths. The mark 1 was used on both trials comparing catches against dredges with no top net then for 3 hauls with dredges with a top net in an effort to evaluate whether queen scallops were actually swimming. The mark 2 was solely compared with the top net.

Data analysis

Tests of gear were made in paired comparisons, with each tow consisting of two types of gear, one towed on either side of the boat. Data such as this were analysed using a paired t test. Laboratory experiments on swimming behaviour were analysed using one way Analysis of Variance.

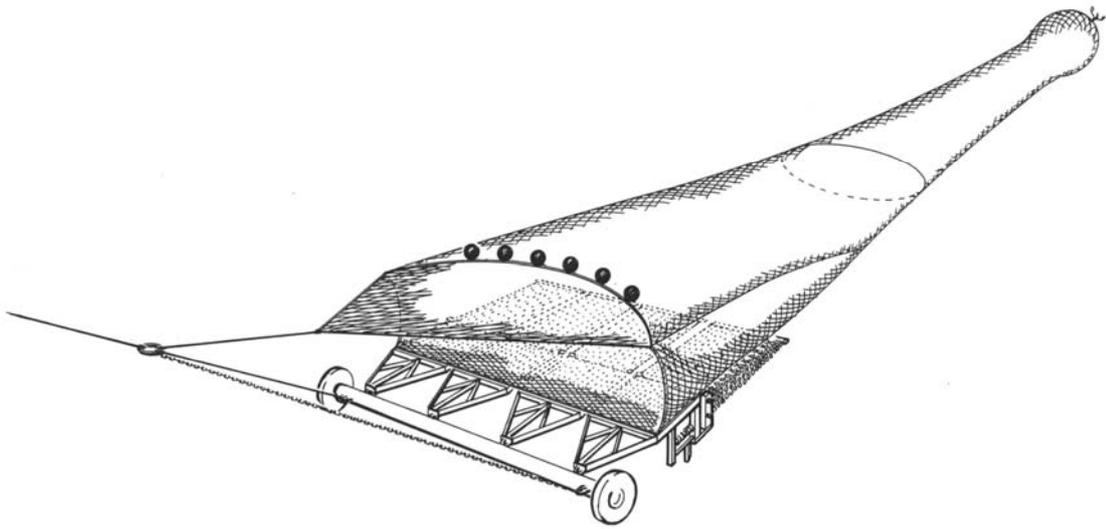


Figure 1 Line drawing of the 'top net' in position over a gang of four queen dredges

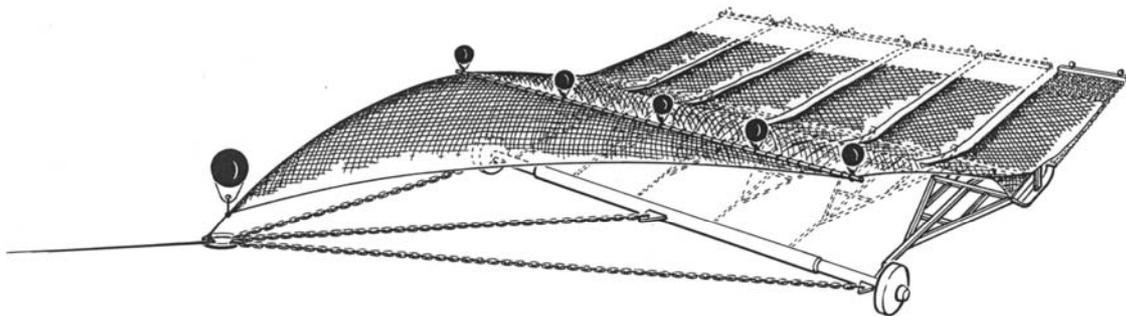


Figure 3 Line drawing of 'cover net' in position over a gang of four dredges

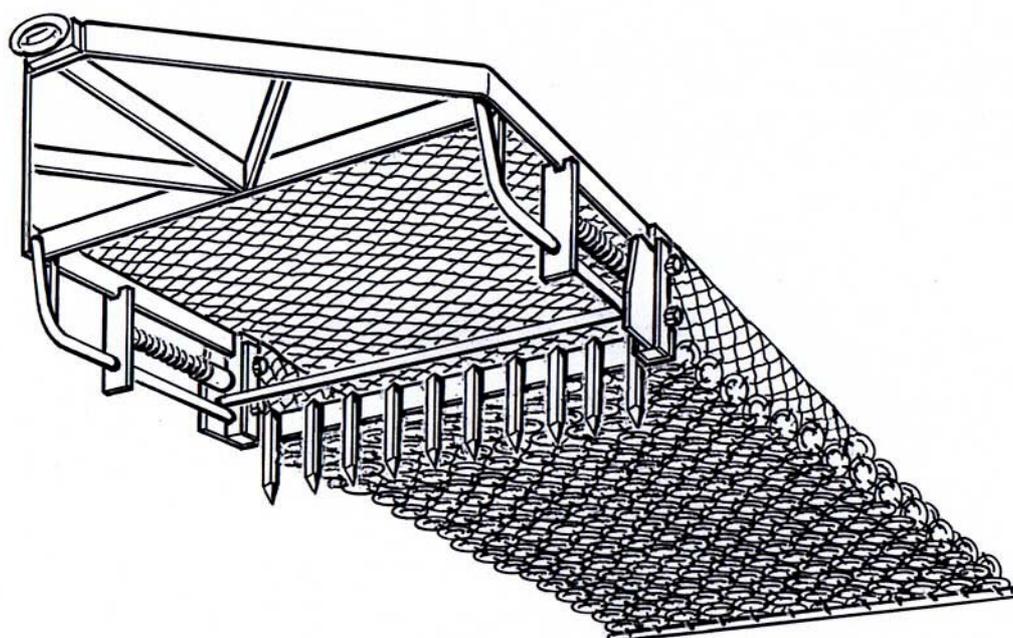
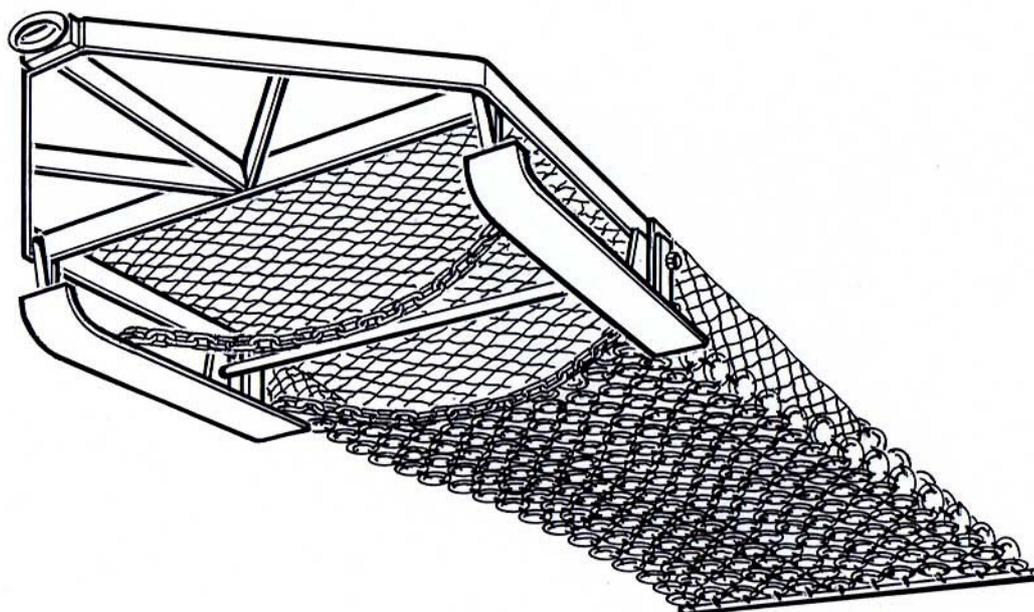


Figure 2 Skid (above) and toothed (below) queen scallop dredges.

Results

Seasonal variability in queen scallop behaviour

The number of queen scallops caught in the net was calculated as a percentage of the total catch in the 4 dredges. There was very high seasonal variability in the proportion of individuals captured in the net and therefore avoiding dredge capture. During the winter and spring very low numbers of queen scallops were found in the net (<3% of dredge catch). A high rate of net capture occurred during the summer and autumn with the highest level occurring in September 2000 when over twice as many individuals were found in the net compared to the dredges. In the year 2000, sampling was started too late to determine when queens began actively swimming. In 2001 net capture (and therefore presumably active swimming in response to the approach of gear) clearly increased from May to June (Figure 4a). Sampling is planned in October November and December to determine when swimming ceases. Dredging took place at all dates in a direction parallel to the tidal flow and at each date tows were undertaken both with the tide and against the tide. Examination of tidal data showed no relationship between the level of net capture and the direction of tow relative to tidal flow. There was no apparent relationship between the level of gonad development and net capture. Examination of the size structure of the queen scallops captured over the study period shows a shift from a distinctly bi-modal population structure from the summer 2000 through to March 2001 to a predominantly unimodal structure in the summer and autumn of 2001 (Figure 5). These data suggest low recruitment of queen scallops during 2000 resulting in very low numbers of juveniles in 2001. This difference may have caused the large difference in the maximum proportion of queen scallops found in the top net between 2000 (maximum value = 221%) and 2001 (maximum value = 61%). Comparisons between years and comparison of the size structure of queens found in dredges compared to the net suggest that juvenile queens are more likely to swim and therefore to avoid capture. However, the difference in size selectivity between the net (70mm mesh panels) and the dredge (60mm diameter rigid steel rings) prevent any firm conclusions.

In order to determine the proportion of marketable sized queen scallops which avoid dredge capture the data were re-calculated to include only those individuals over 55mm in shell height (Figure 4b). These data mirror the temporal trend described above and show that the number of queen scallops of marketable size which avoid dredge capture by swimming over the top of conventional queen dredges can be as high as 42% of the number captured in dredges. Queen scallops over 55mm in shell height can reasonably be expected to be retained by both the dredge and net and therefore a comparison of size structure of scallops over 55mm shell height captured by the 2 processes can be made. Fig 6 shows a similar size structure between dredge and net although the net clearly caught very few of the largest size classes (> 68mm shell height).

Comparison of field sampling with the swimming experiments carried out in the laboratory shows that with the increase in the level of net capture there was a corresponding change in the response time but not in the total number of adductions (Figure 4c,d). One way ANOVA shows that there was a significant decrease in the time taken to respond to the predator stimulus from May to the next laboratory sampling date in August. Response time further declined in September. There was

no significant difference among sampling dates in the total number of adductions (Table 2).

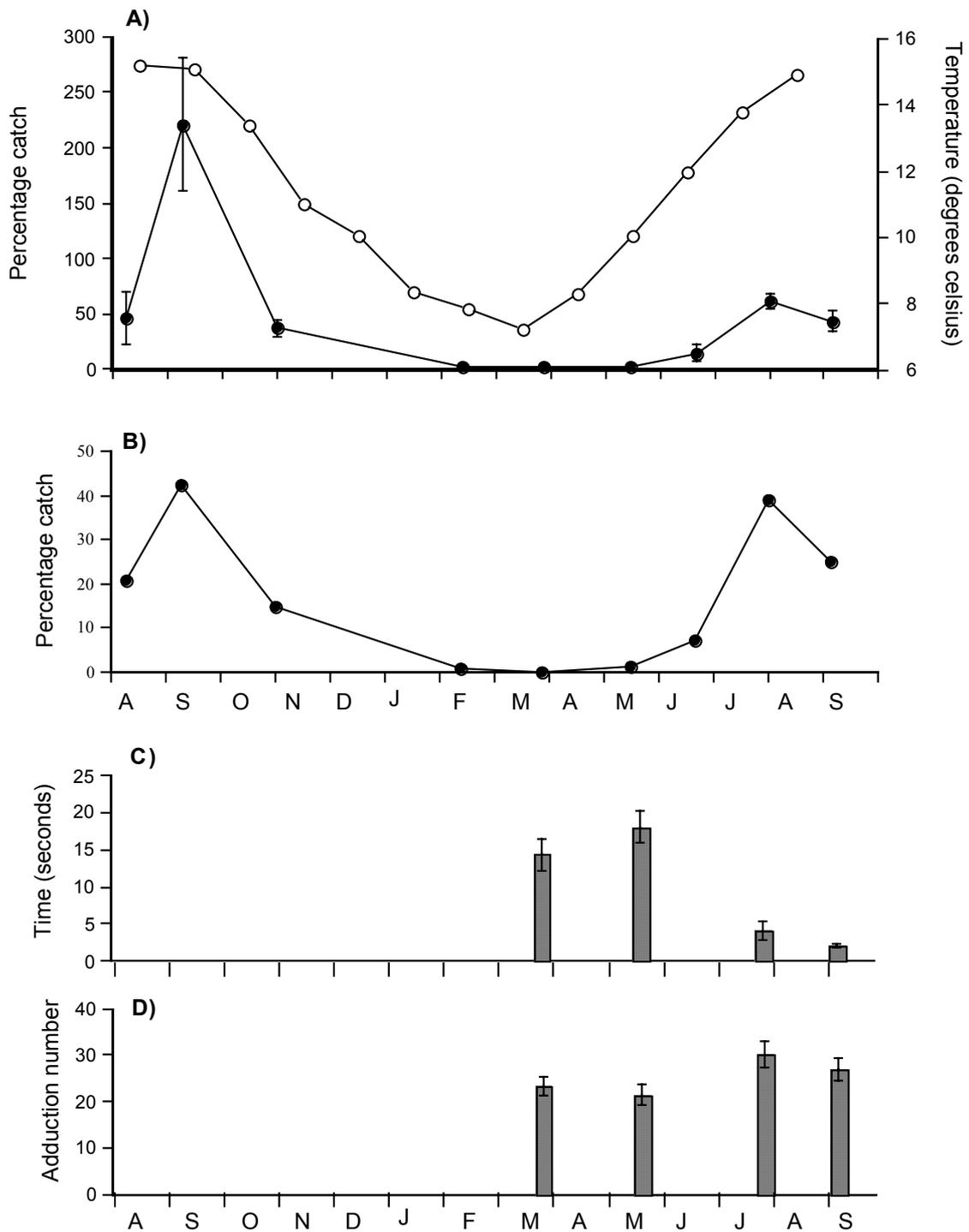


Figure 4 Assessment of seasonal swimming behaviour in the queen scallop *Aequipecten opercularis* (A) Number of queen scallops captured in the top net as a percentage of those captured in the dredges. (B) Number of queen scallops over 55mm shell height captured in the top net as a percentage of those captured in the dredges. (C) Response time in queen scallops induced to swim in the laboratory (B) Total number of adductions before exhaustion in queen scallops induced to swim in the laboratory.

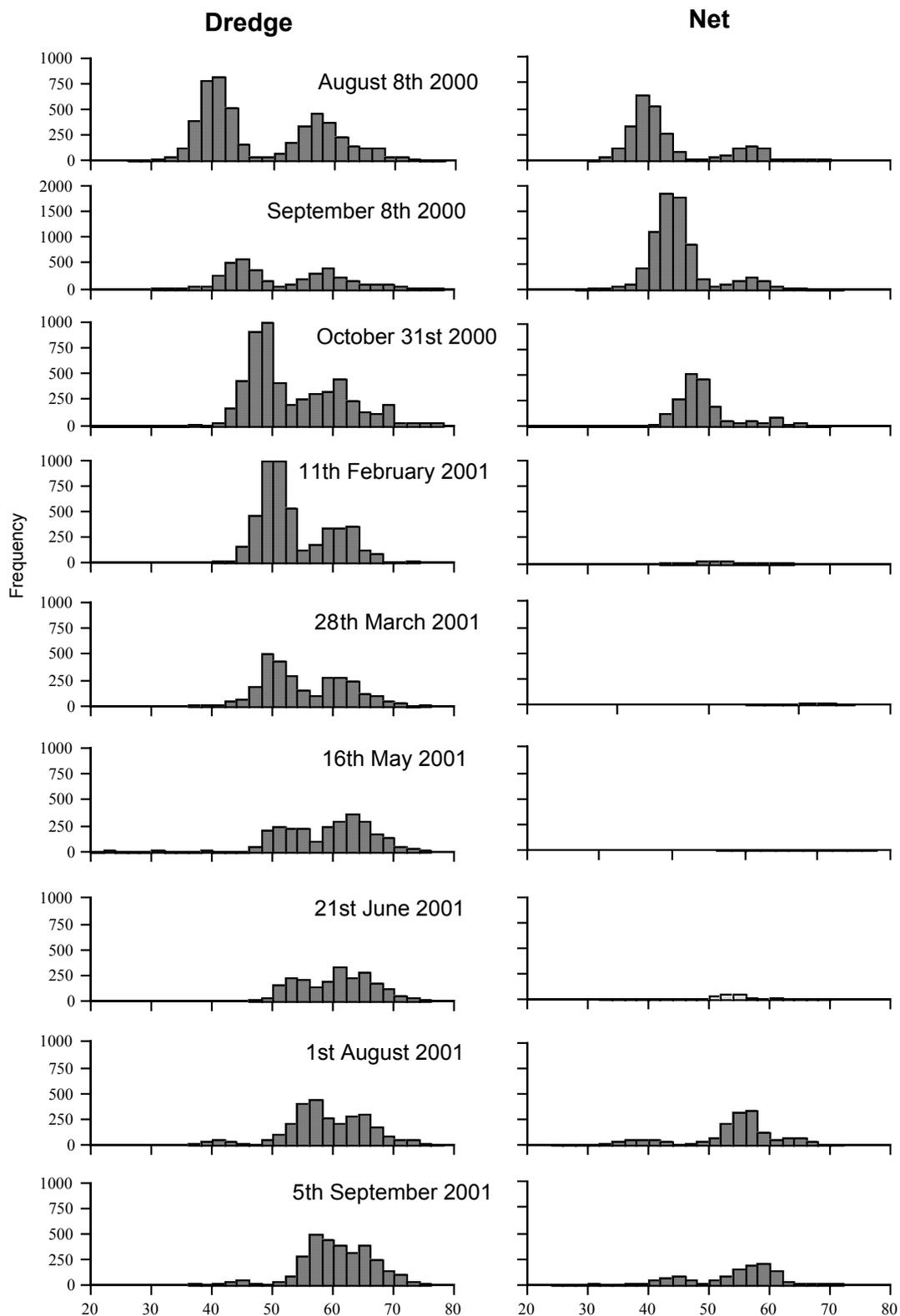


Figure 5 Size distribution and number of queen scallops captured in queen dredges and the top net over a 14 month period NB Note different y axis scale for September 2000

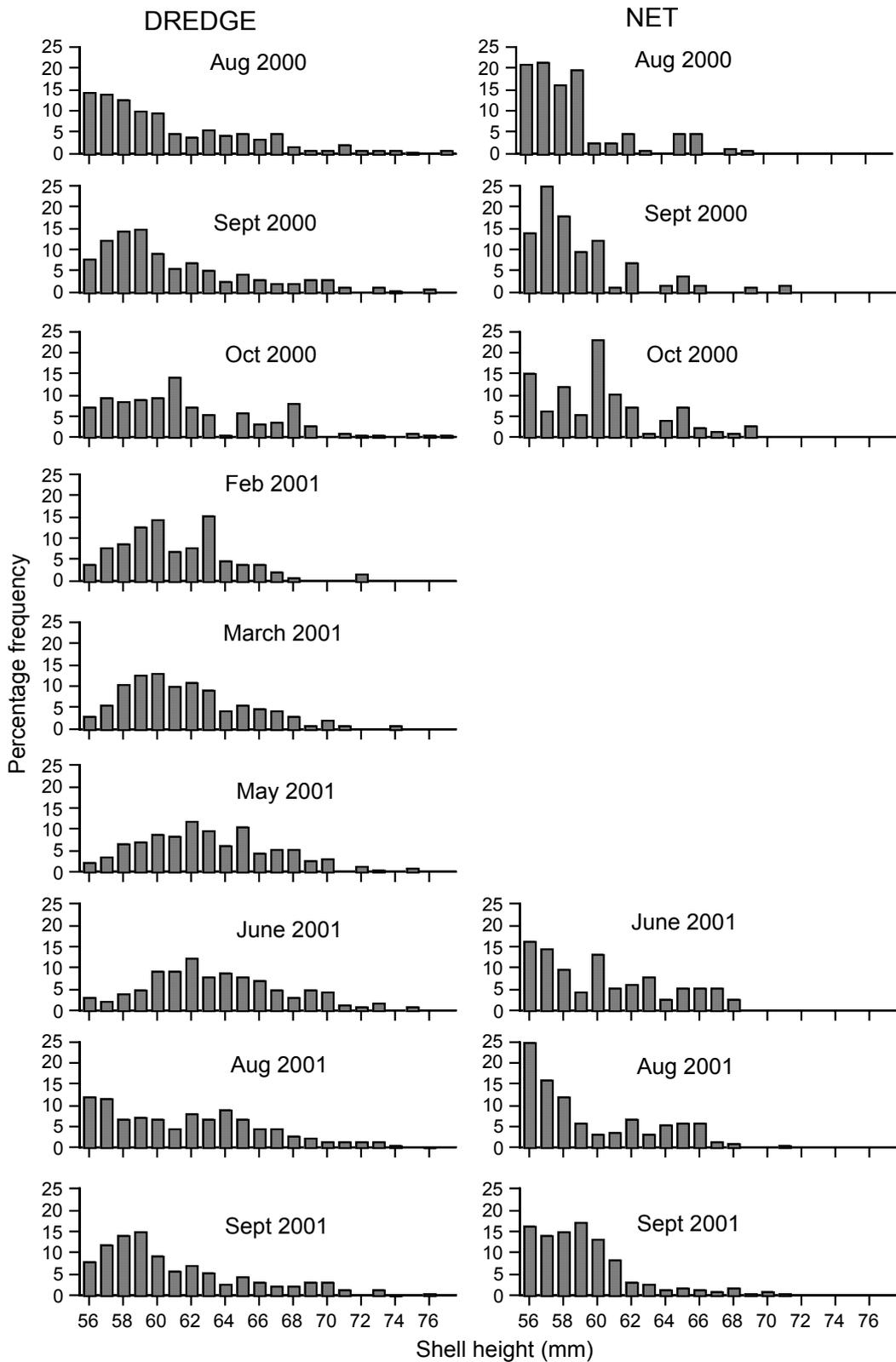


Figure 6 Size frequency distribution of marketable sized queen scallops (>55mm shell height) captured in queen dredges and the top net over a 14 month period

Table 2 One way ANOVA of response variables measured in laboratory swimming experiments at four separate dates.**A) Response time** (Ln transformed)

Source	df	MS	F	P
Date	3	28.73	66.05	<.0001
Residual	104	0.43		

SNK test March = May > August > September

B) Total adductions (Untransformed)

Source	df	MS	F	P
Date	3	368.22	2.39	>0.05
Residual	104	154.15		

Assessment of queen gear species selectivity: Skid gear versus toothed gear

Within individual fishing grounds there was large variability among tows in the capture rate of queen scallops for both gear types. Neither toothed or skid gear caught consistently higher numbers of queen scallops over the grounds fished (Figure 7 and 8). Paired t tests revealed a significant difference in the number of queen scallops caught per tow at only one ground, Laxey, where skid gear caught significantly higher numbers (24 % greater than toothed gear) (Table 3). The size distribution of queen scallops captured showed no difference between the two gear types (Figure 9).

Table 3 Paired t test results (two tailed) of queen scallop capture using two different gear types over different fishing grounds

Trial	Ground	t	df	P
July	PSM 51	1.98	5	>0.1
	PSM 54	1.08	5	>0.3
	Laxey	4.45	4	<0.05
August	Douglas South	1.99	4	>0.1
	PSM 54	1.04	3	>0.3

In both July and August trials, which covered a total of four different fishing grounds, the number of great scallops (*Pecten maximus*) captured by skid gear was extremely

low. Averaged over all five days of fishing, there were almost 20 times more scallops caught by toothed gear (Figure 10). Damage levels to scallops were consistently low for all fishing grounds in both gear types, with 2.3% and 3.9% of scallops showing external physical damage in skid and dredge gear respectively. The size distribution of scallops caught by the two gear types appeared to show no differences, although comparisons are difficult owing to the low numbers of scallops caught in skid gear.

The level of fullness of each dredge on completion of a tow (measured as a percentage of the greatest possible volume) showed little difference between the two gear types at both fishing grounds sampled in August (Figure 12, Table 4). Data are not available for % fullness in July. However there were clear differences in the volume of stones collected with a significantly greater volume collected at fishing grounds PSM 51 and PSM 54 in July (Figure 11). At Laxey during the July trial and at both fishing grounds in the August trial, no significant differences were found.

Table 4 Paired t test results (two tailed) of dredge % fullness and volume of rocks caught using two different gear types over different fishing grounds

Trial	Ground	t	df	P
% FULLNESS				
August	Douglas South	0.19	3	>0.8
	PSM 54	1.73	3	>0.15
Rock volume				
July	PSM 51	10.32	4	<0.01
	PSM 54	5.63	4	<0.01
	Laxey	2.59	3	>0.05
August	Douglas South	2.38	4	>0.05
	PSM 54	0.98	3	>0.3

The major bycatch species captured by both gear types consisted of the crabs, *Cancer pagurus* and *Pagurus* spp., the starfish *Porania pulvillus*, *Asterias rubens*, *Crossaster papposus*, *Luidia ciliaris* and *Henricia oculata*, the whelk *Buccinum undatum* and the urchin *Echinus esculentus*. Data are presented on the abundance and mean damage index of individuals of all the major bycatch species captured in both gear types for three fishing grounds in the July trial and two fishing grounds in the August trial (Figures 13-17). No clear trends are apparent. Comparisons among different fishing grounds at different dates showed that neither gear type appeared to catch consistently more individuals of any bycatch species. The damage caused to bycatch species was similar in both gear types with no consistent differences.

For all trials in which the mark 1 'cover net' was used in both July and August, there was no significant difference found between the number of captured queen scallops in standard skid gear and skid gear with attached net (Figures 18 and 19). The size frequency distribution of captured queens did not differ between the 2 gear types (Figures 20 and 21).

The number of queen scallops captured using skid gear was increased (relative to skid gear with the mark 2 'cover net') by the use of the 'top net' designed for use in the seasonal sampling described above (Figure 22). A paired t test showed a significantly greater catch of queen scallops over a 2 day period (29th-30th August) when the number of individuals in the 'top net;' and skid gear were combined ($t = 2.7$, $df = 8$, $P < 0.05$). Thus the 'top net' configuration was better than either cover net for catching swimming queenies. It also has the merit of the operator being able to judge when it is working because the cod end gets filled up with queen scallops. It was operationally better since the mark 2 cover net was prone to snagging on the dredge.

Summary and conclusions

The investigation into seasonality of queen scallop swimming showed clearly that there is a period of the year when very few queen scallops escape conventional dredge gear by swimming over the dredge mouths. From February to May very few queen scallops were found in the top net. The beginning of the period when queen scallops are likely to be caught by gear such as trawls, which do not scrape or dredge the substratum, is June; this period extends at least into mid or late autumn. Data showed that a large proportion of marketable sized queen scallops encountered by conventional dredge gear are lost, not through inefficiency at the tooth bar or belly collecting bag, but by the swimming response of the queens allowing escape over the top of the gear. Experimental work in the laboratory showed that the seasonality of the field results was a result not of changes in the ability of the scallops to perform valve adductions, but changes in the time taken to respond to a stimulus. This behavioural observation may be exploited by the use of devices to stimulate scallops to swim in advance of oncoming gear, for example by the use of tickler chains.

The assessment of species selectivity showed that, at least in mid summer, it is possible to capture queen scallops with a minimal bycatch of the great scallop *Pecten maximus*. The skid dredges caught equivalent levels of the target species the queen scallop (on one occasion significantly more were captured than toothed gear) and captured very few *Pecten maximus*. In addition, skid dredges caught significantly fewer rocks and on some grounds produced a much cleaner catch which was quicker and easier to sort. There appears to be no advantage in using toothed dredge gear at this time of year. Such trials require extending throughout the year to examine any seasonal variability in selectivity and catch rates. If consistent results are found this work presents a strong argument to implement technical measures to prevent targeting of queen scallops using toothed dredge gear.

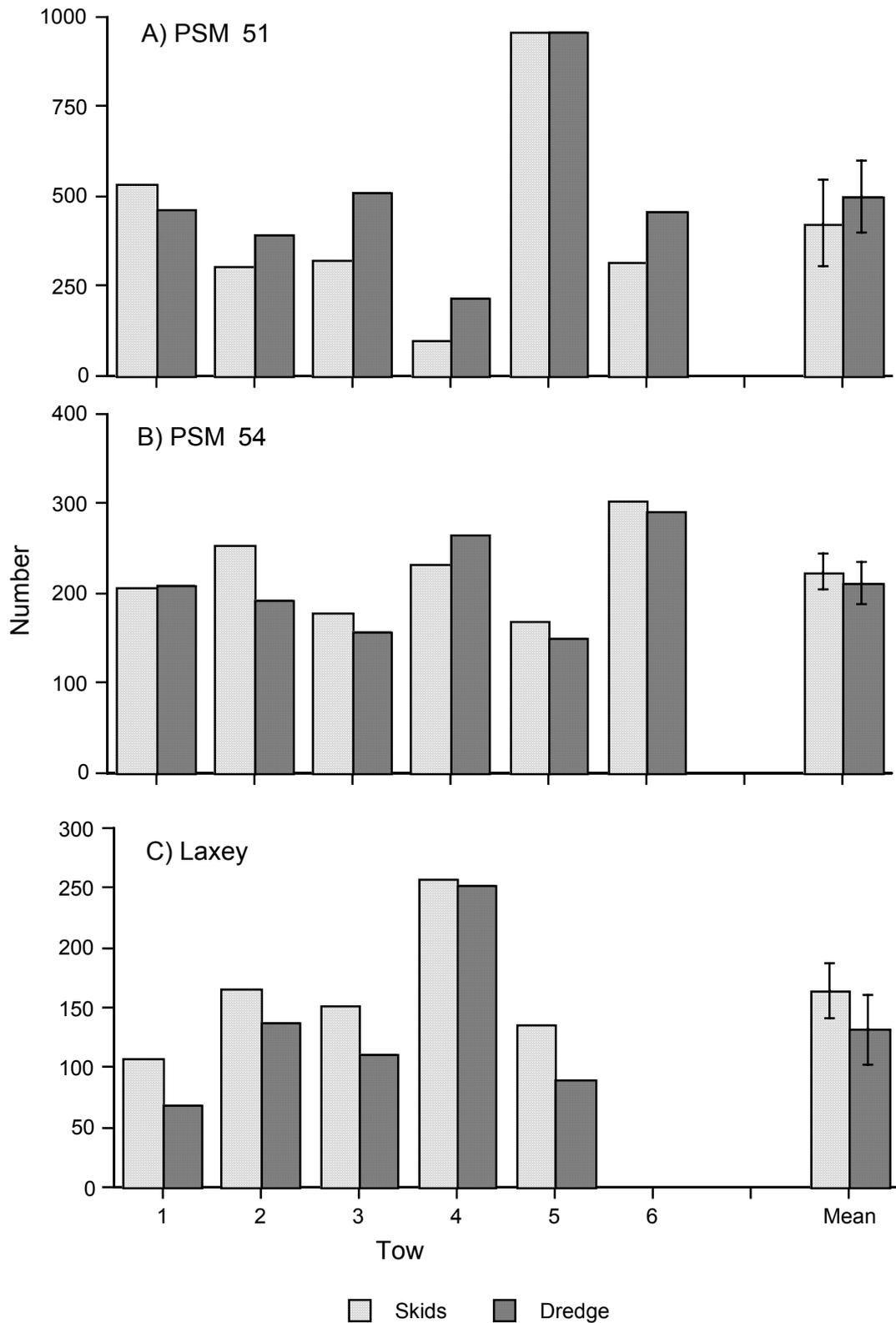


Figure 7 Mean number of queen scallops caught per dredge in two gear types, skids and toothed dredge at three fishing grounds, A) PSM 51, B) PSM 54 and C) Laxey during the July trial. Data are presented as mean values for each individual tow and the overall mean of all tows. SE \pm 1SE

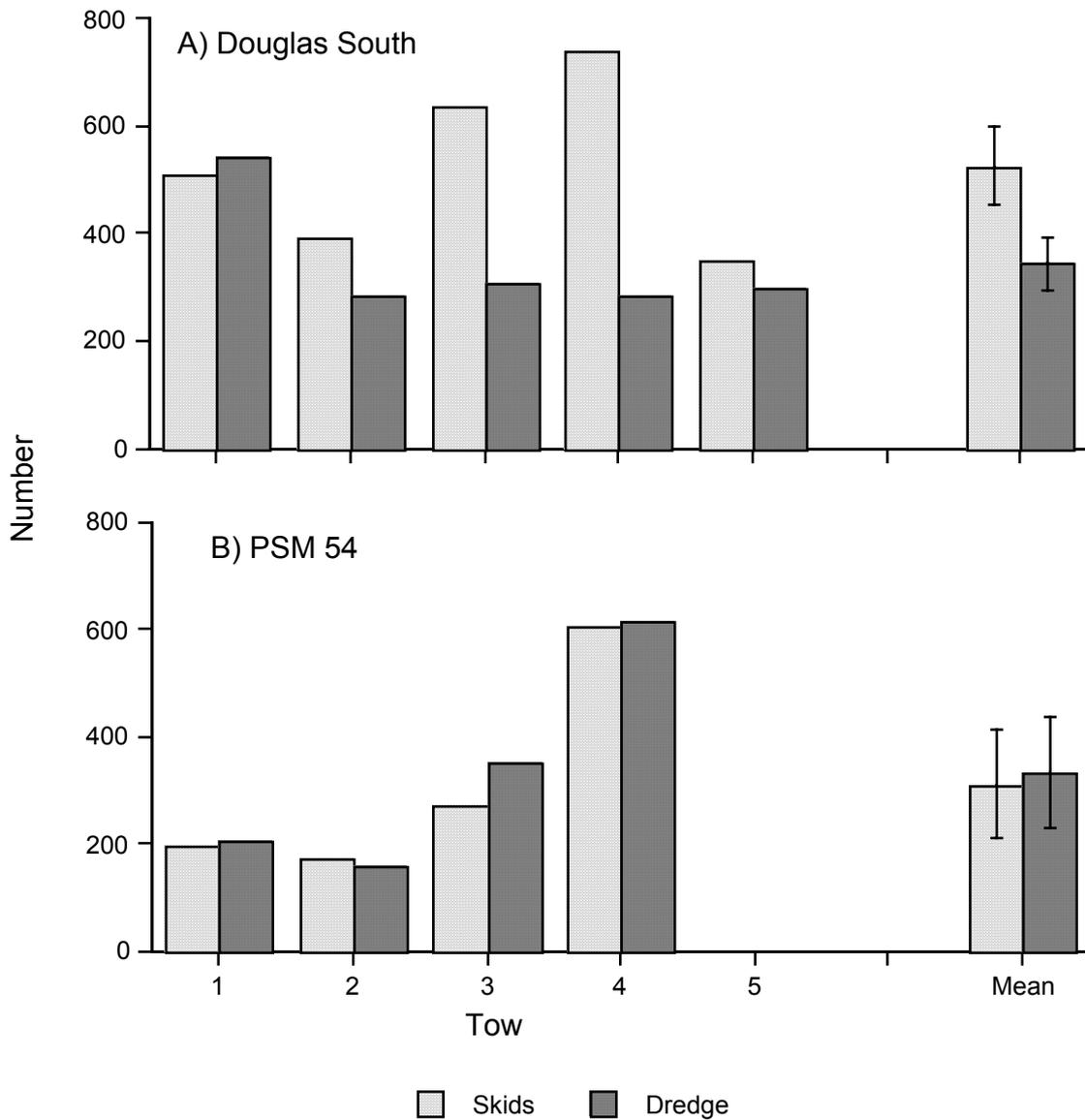


Figure 8 Mean number of queen scallops caught per dredge in two gear types, skids and toothed dredge at two fishing grounds, A) Douglas South and B) PSM 54 during the August trial. Data are presented as mean values for each individual tow and the overall mean of all tows. Error bars $\pm 1SE$.

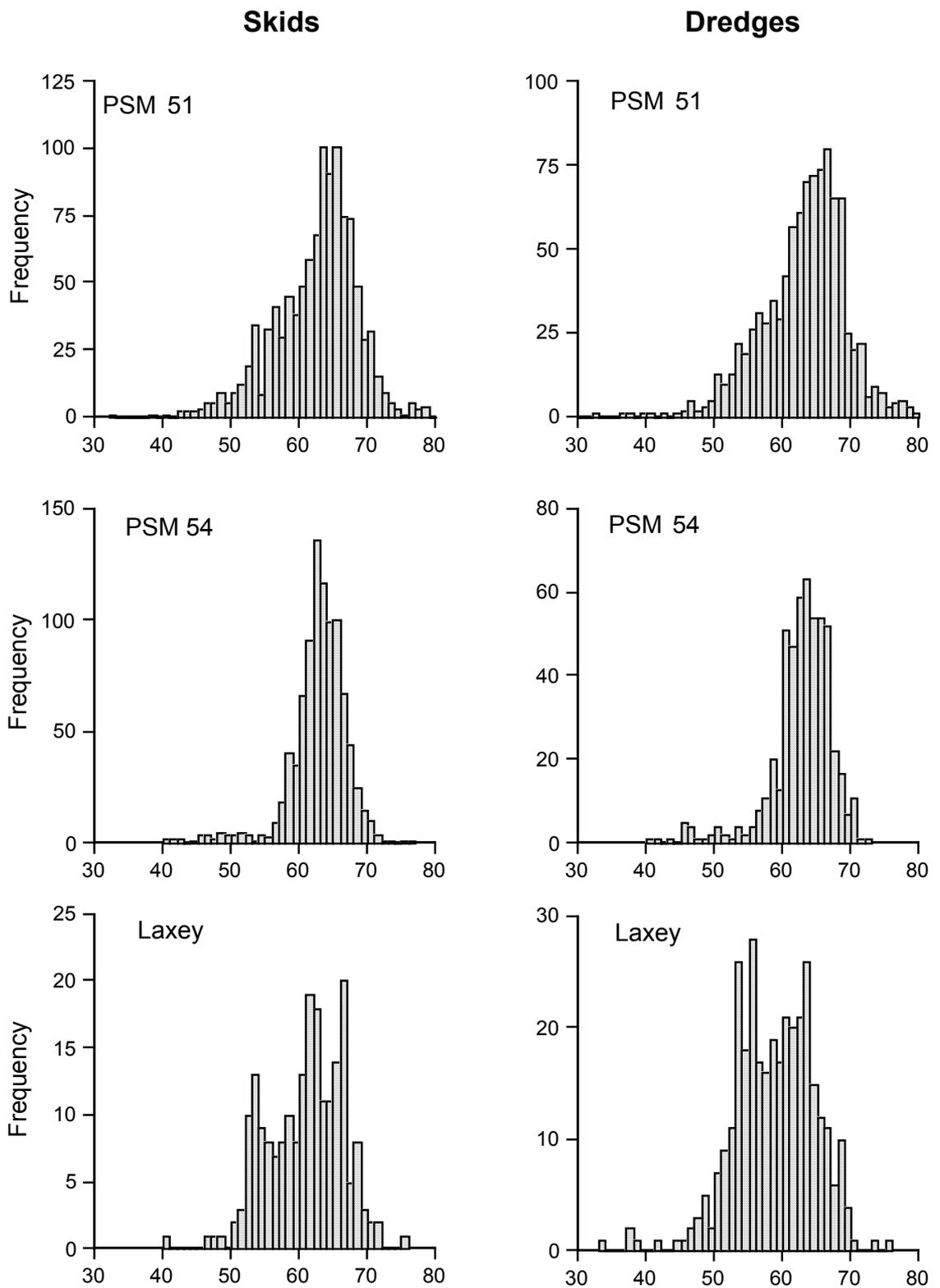


Figure 9 Size distributions of queen scallops captured using two different gear types (skids and dredges) at three fishing grounds during the July trial (Data from all tows combined)

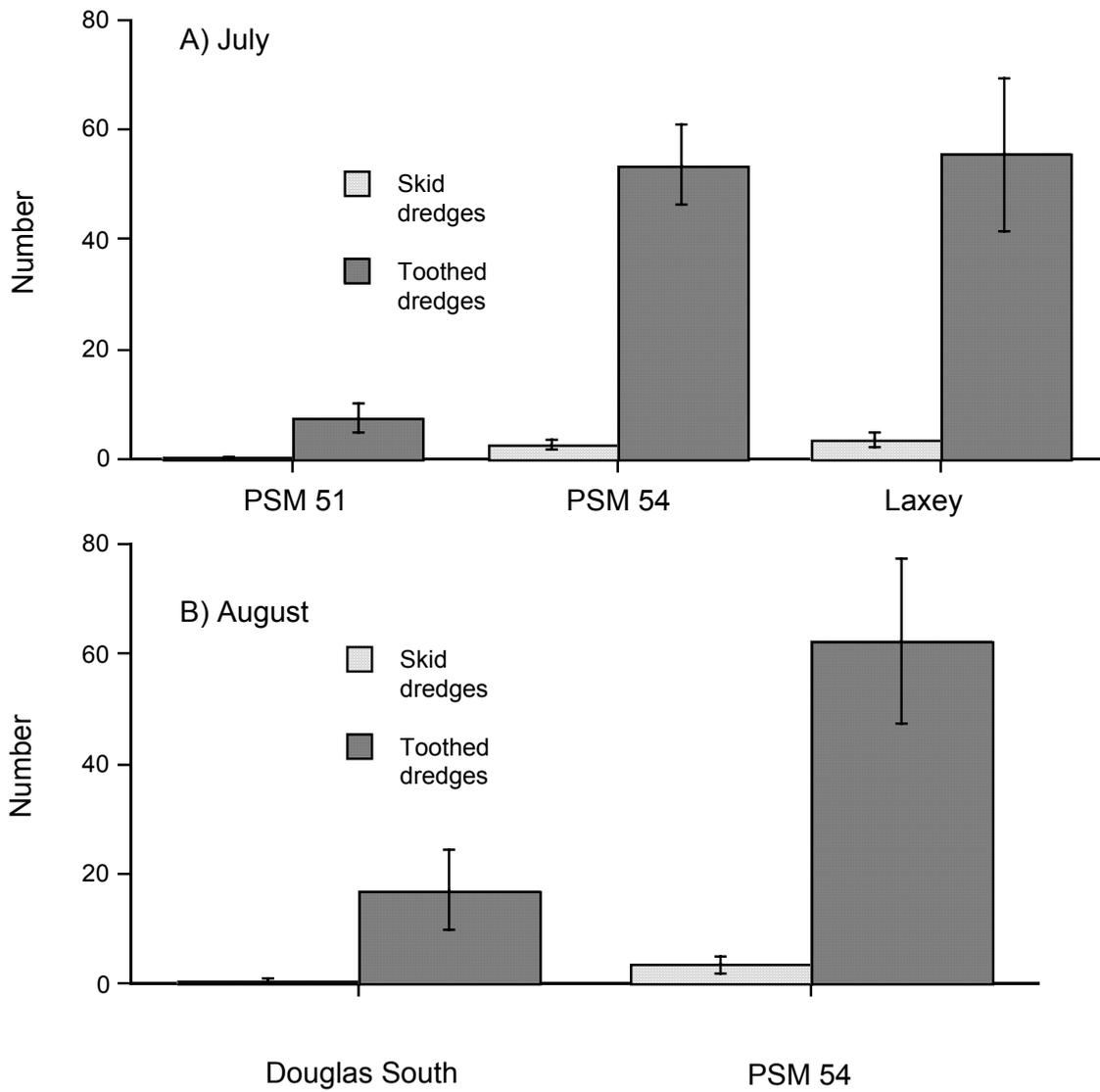


Figure 10 Mean number of great scallops (*Pecten maximus*) caught per dredge in fishing trials comparing skid and dredge gear A) July trial B) August trial. Error bars $\pm 1SE$

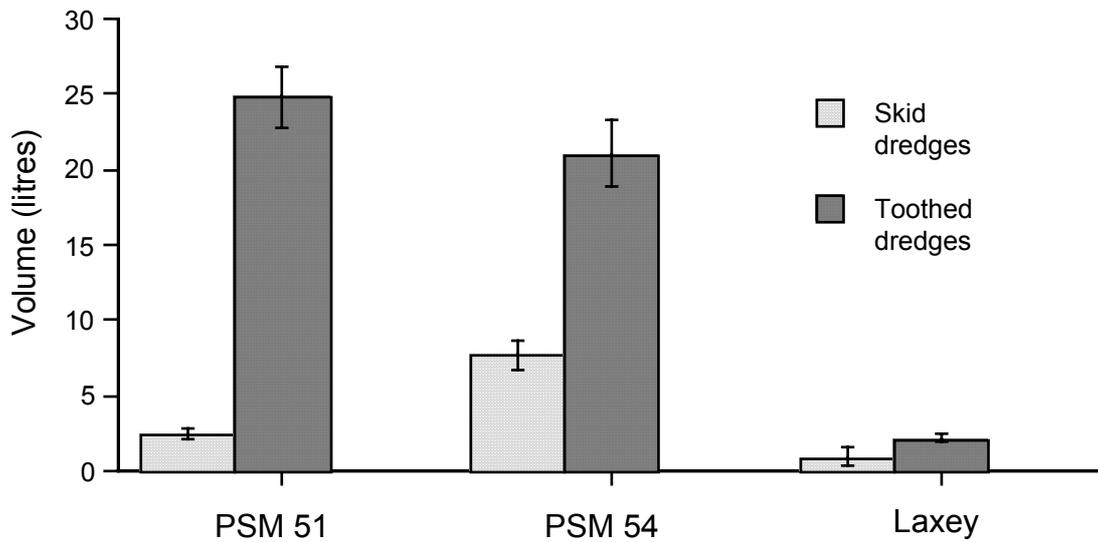


Figure 11 Mean volume of rocks caught per dredge in skid and toothed gear at 3 fishing grounds during the July trial

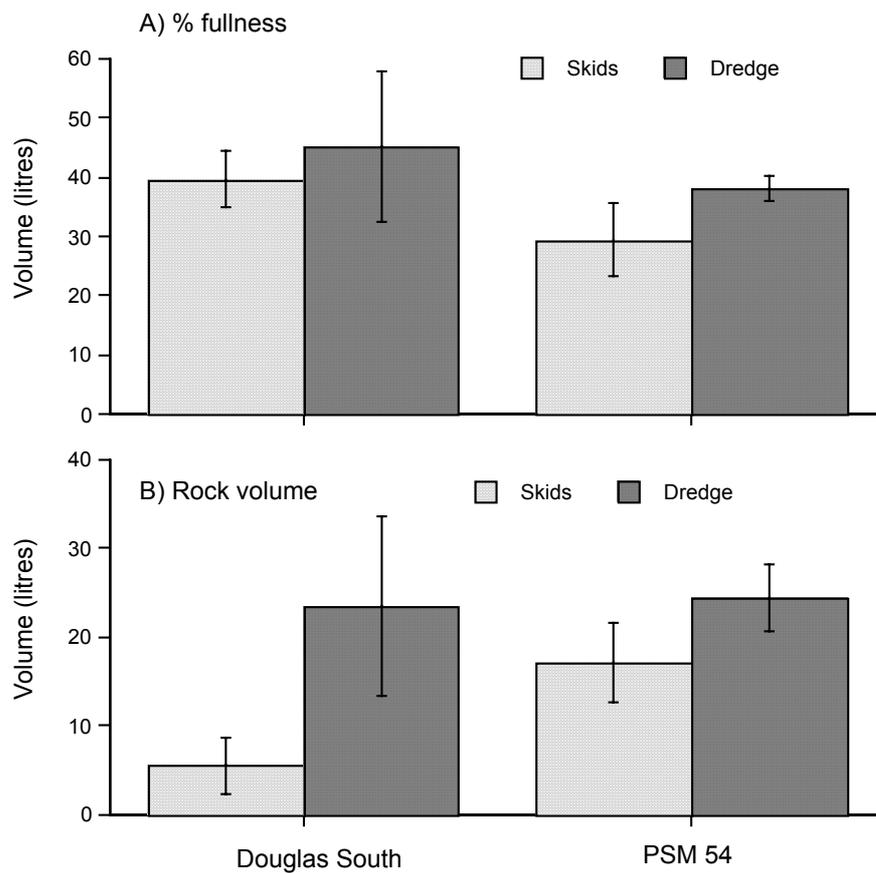


Figure 12 Percentage level of fullness of dredges (A) and volume of rocks (B) in skid and toothed dredge gear at 2 fishing grounds during the August trial. Error bars \pm 1SE

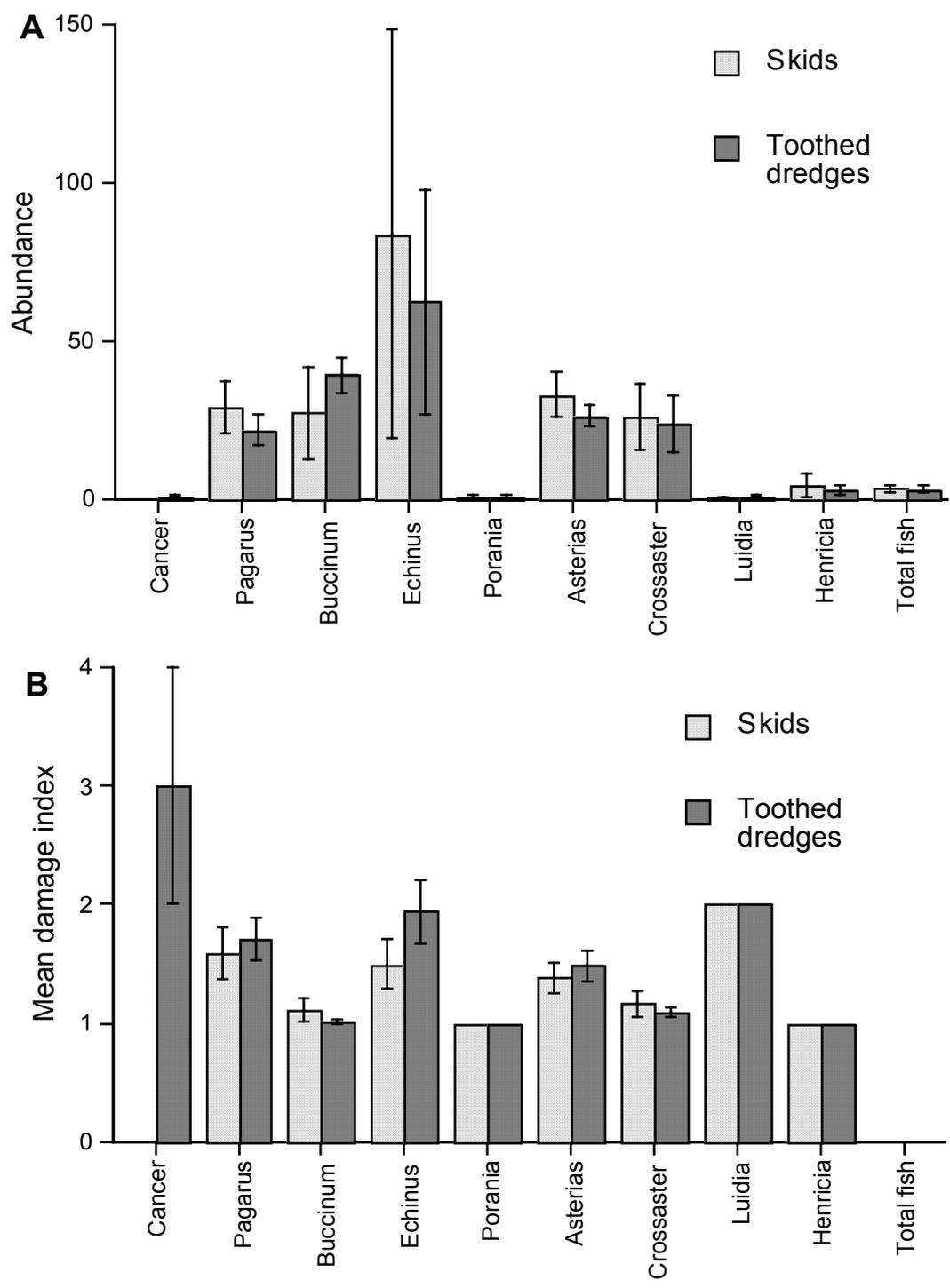


Figure 13 Mean number of individuals captured per tow (A) and mean damage level (B) of major bycatch species at the PSM 51 ground during the July trial in the two gear types, skid and toothed dredges. Error bars $\pm 1SE$

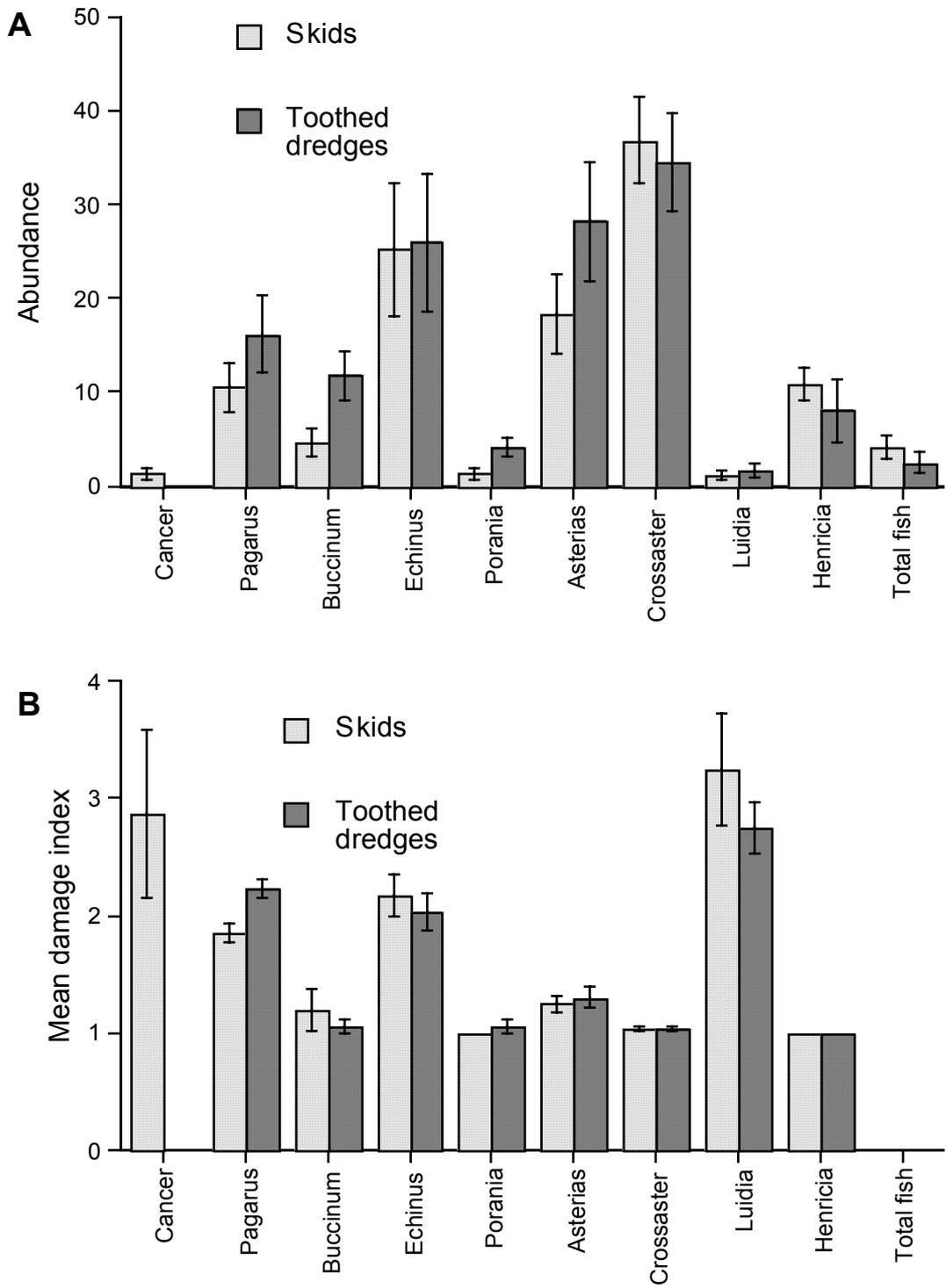


Figure 14 Mean number of individuals captured per tow (A) and mean damage level (B) of major bycatch species at the PSM 54 ground during the July trial in the two gear types, skid and toothed dredges. Error bars $\pm 1SE$

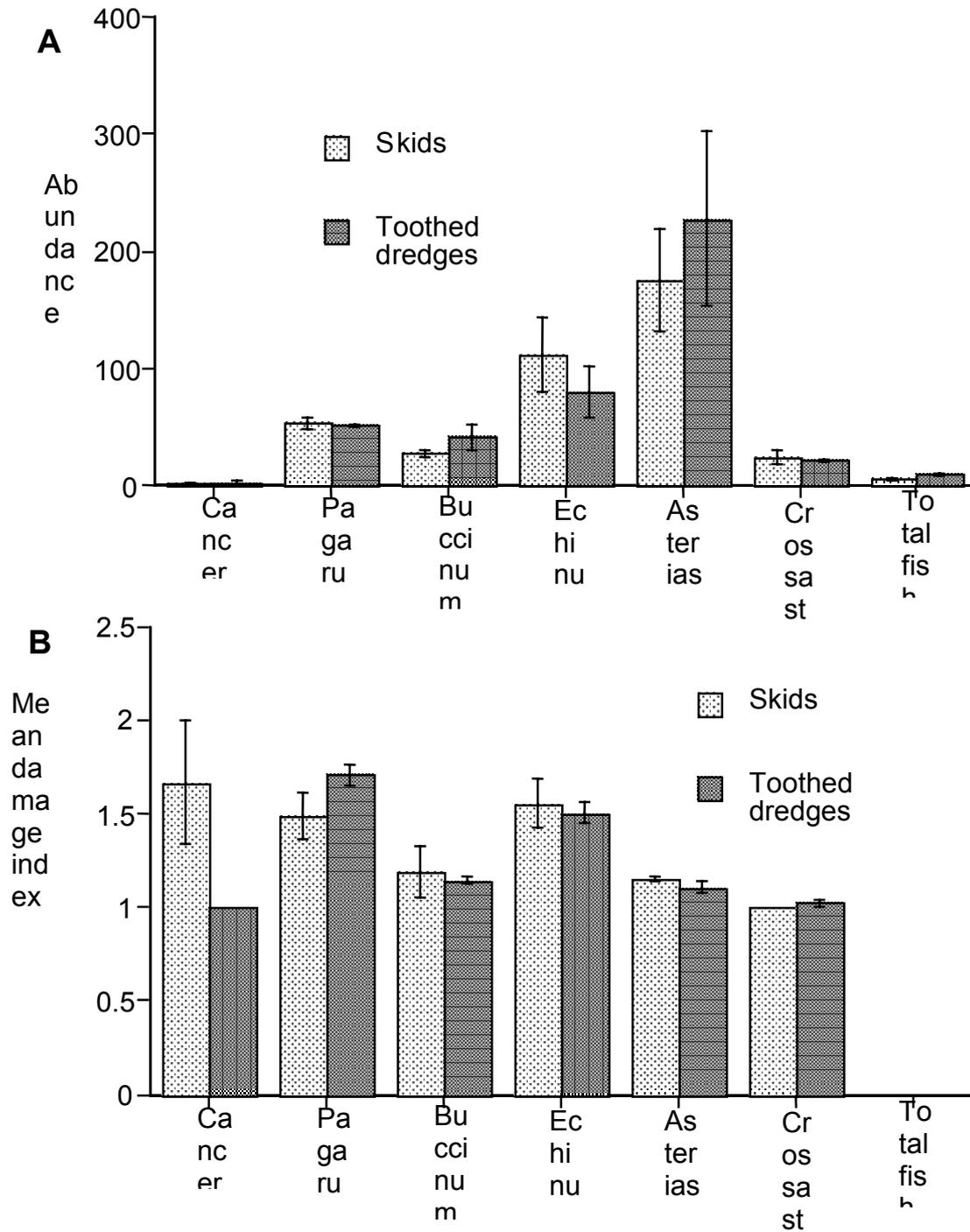


Figure 15 Mean number of individuals captured per tow (A) and mean damage level (B) of major bycatch species at the Laxey ground during the July trial in the two gear types, skid and toothed dredges. Error bars $\pm 1SE$

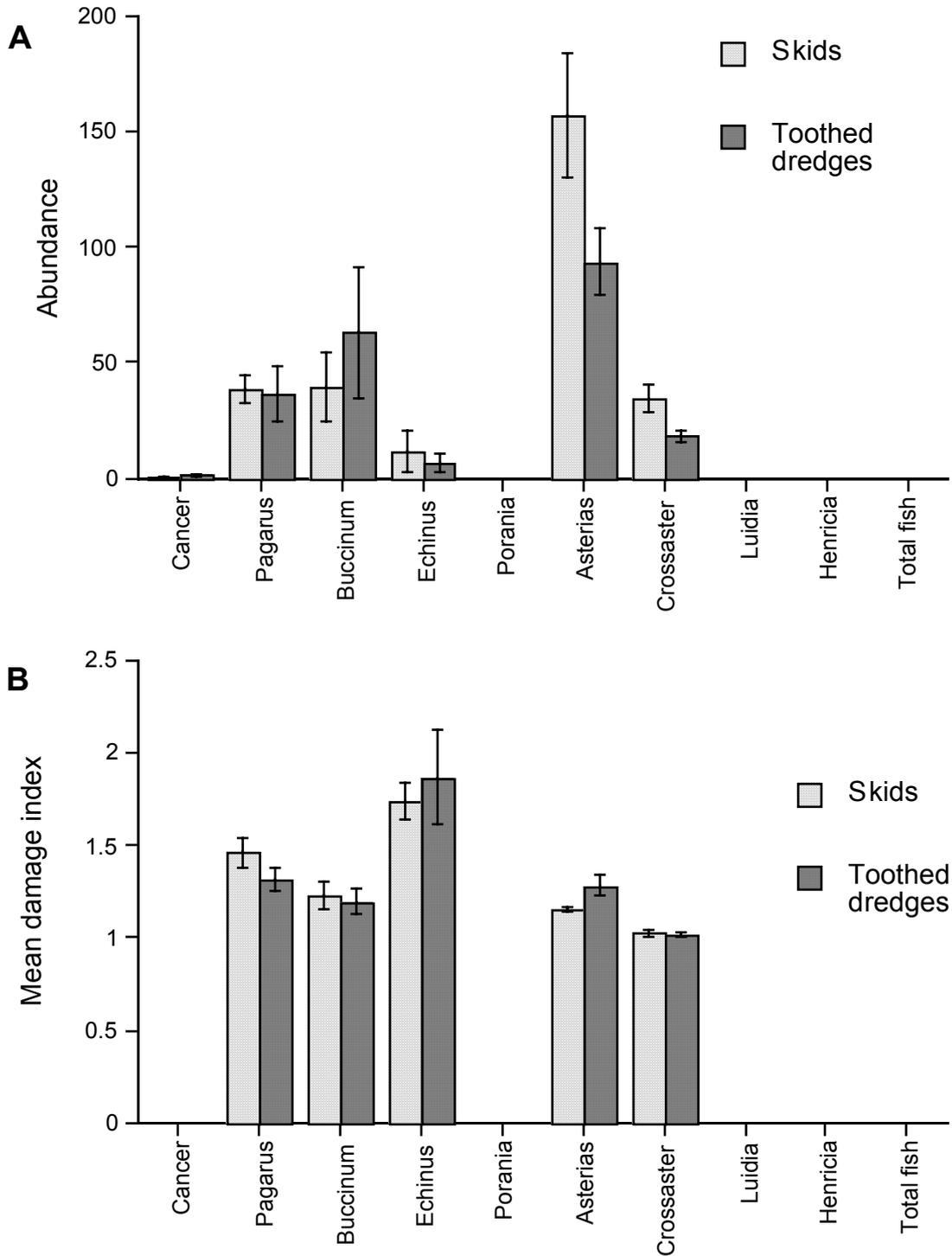


Figure 16 Mean number of individuals captured per tow (A) and mean damage level (B) of major bycatch species at the Douglas South ground during the August trial in the two gear types, skid and toothed dredges. Error bars $\pm 1SE$

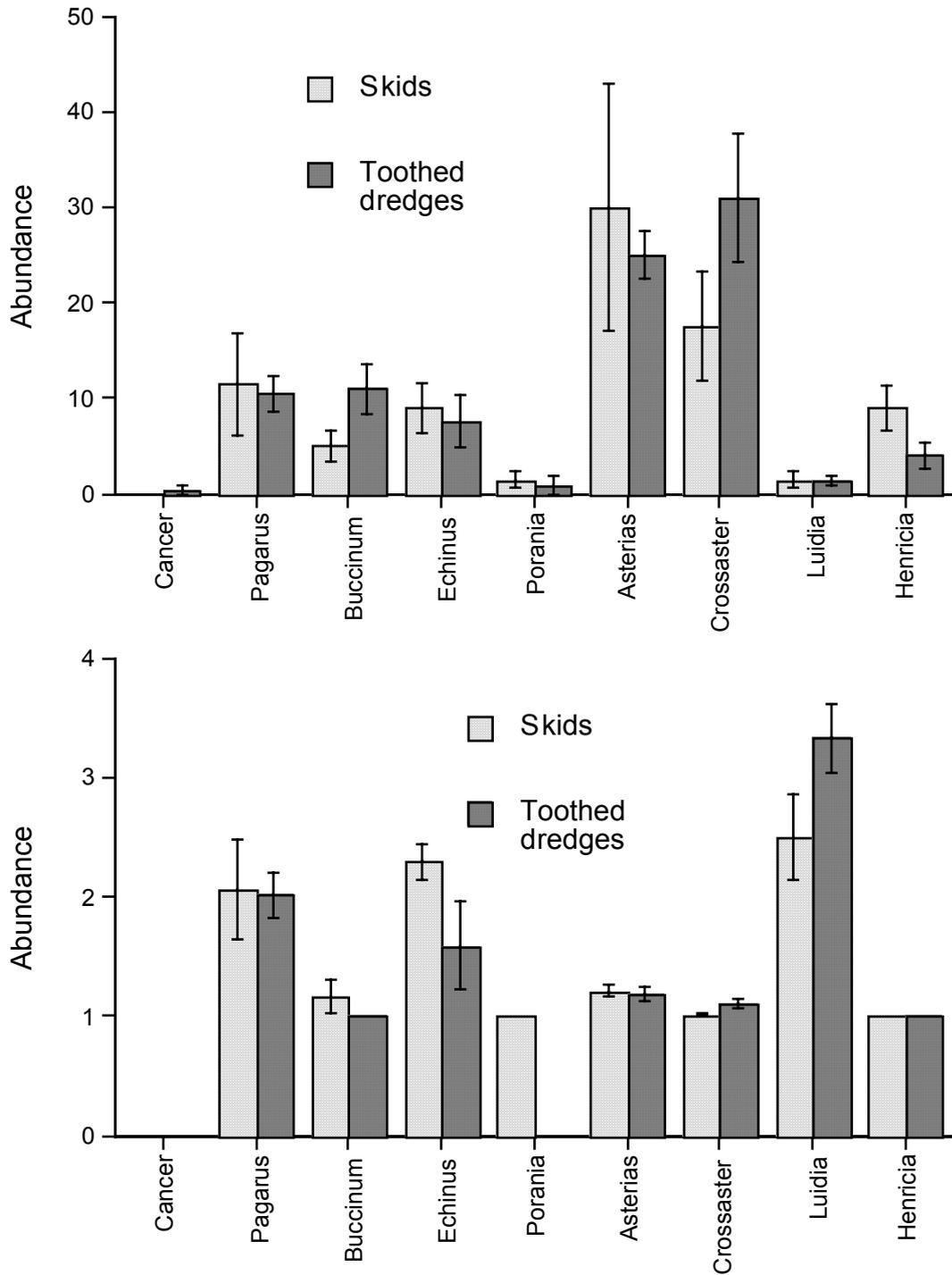


Figure 17 Mean number of individuals captured per tow (A) and mean damage level (B) of major bycatch species at the PSM 54 ground during the August trial in the two gear types, skid and toothed dredges. Error bars $\pm 1SE$

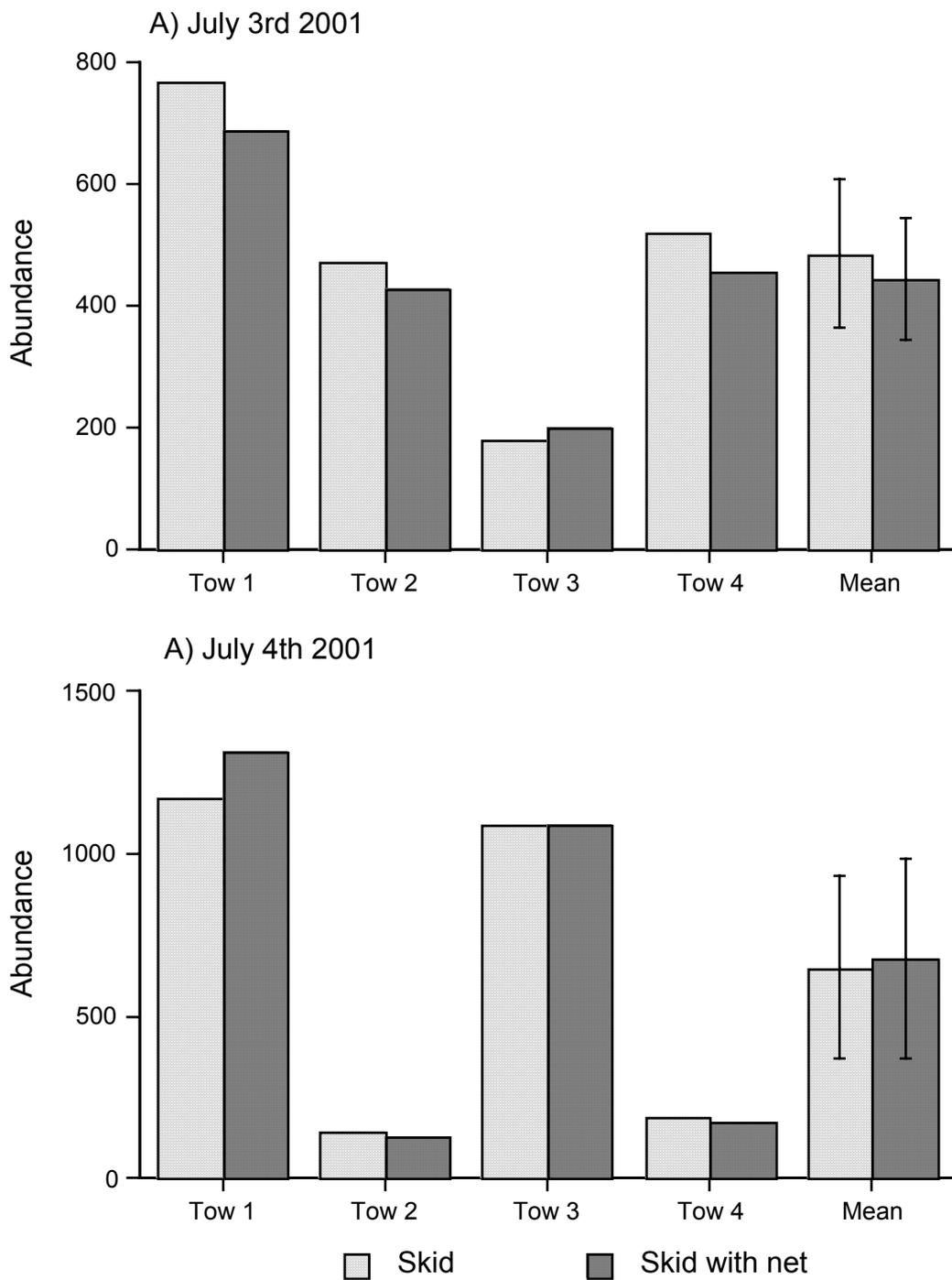


Figure 18 The number of queen scallops captured using skid gear with and without a the mark 1 cover net on two separate dates during the July trial. Error bars \pm 1SE

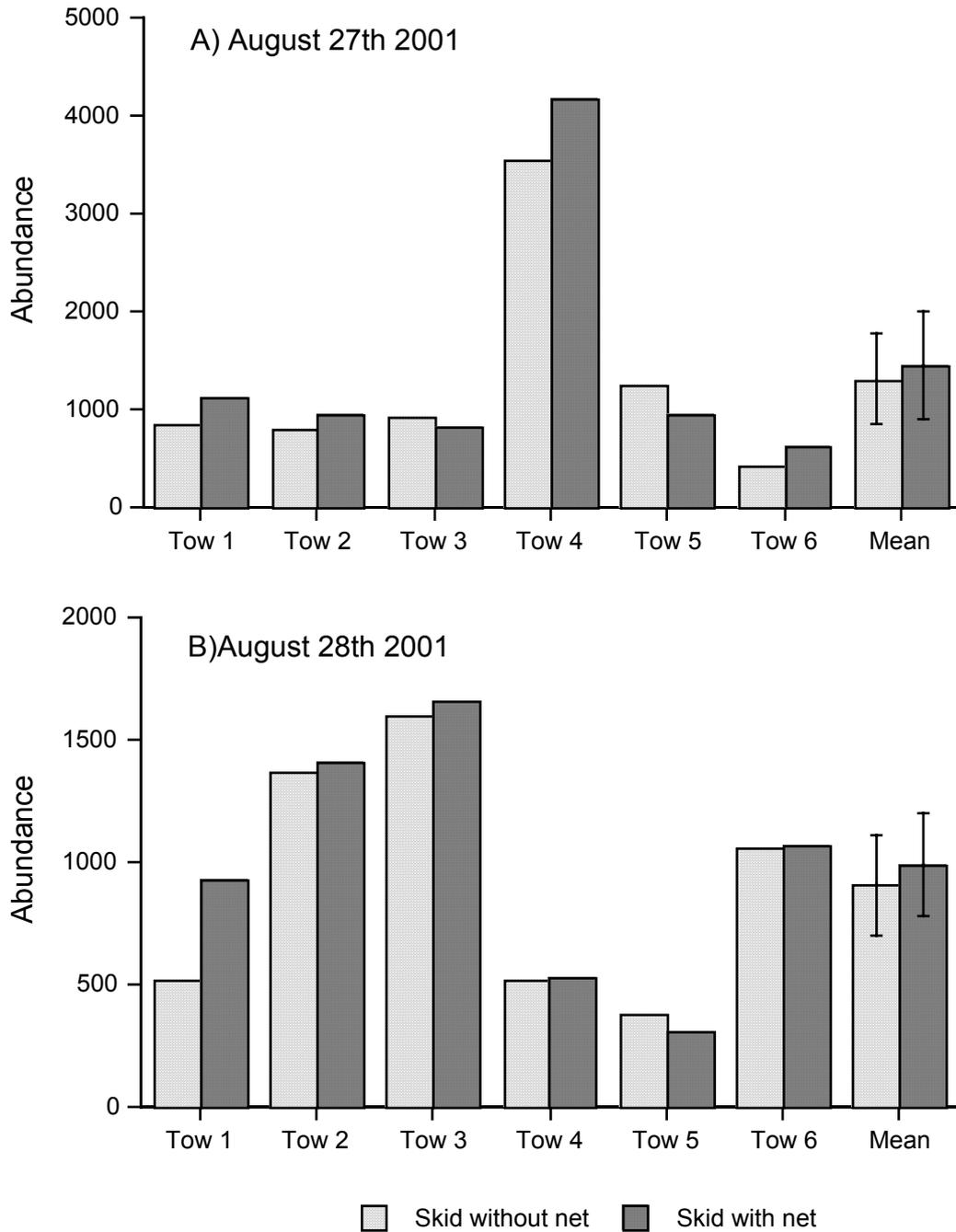


Figure 19 The number of queen scallops captured using skid gear with and without a mark 1 cover net on two separate dates during the August trial. Error bars \pm 1SE

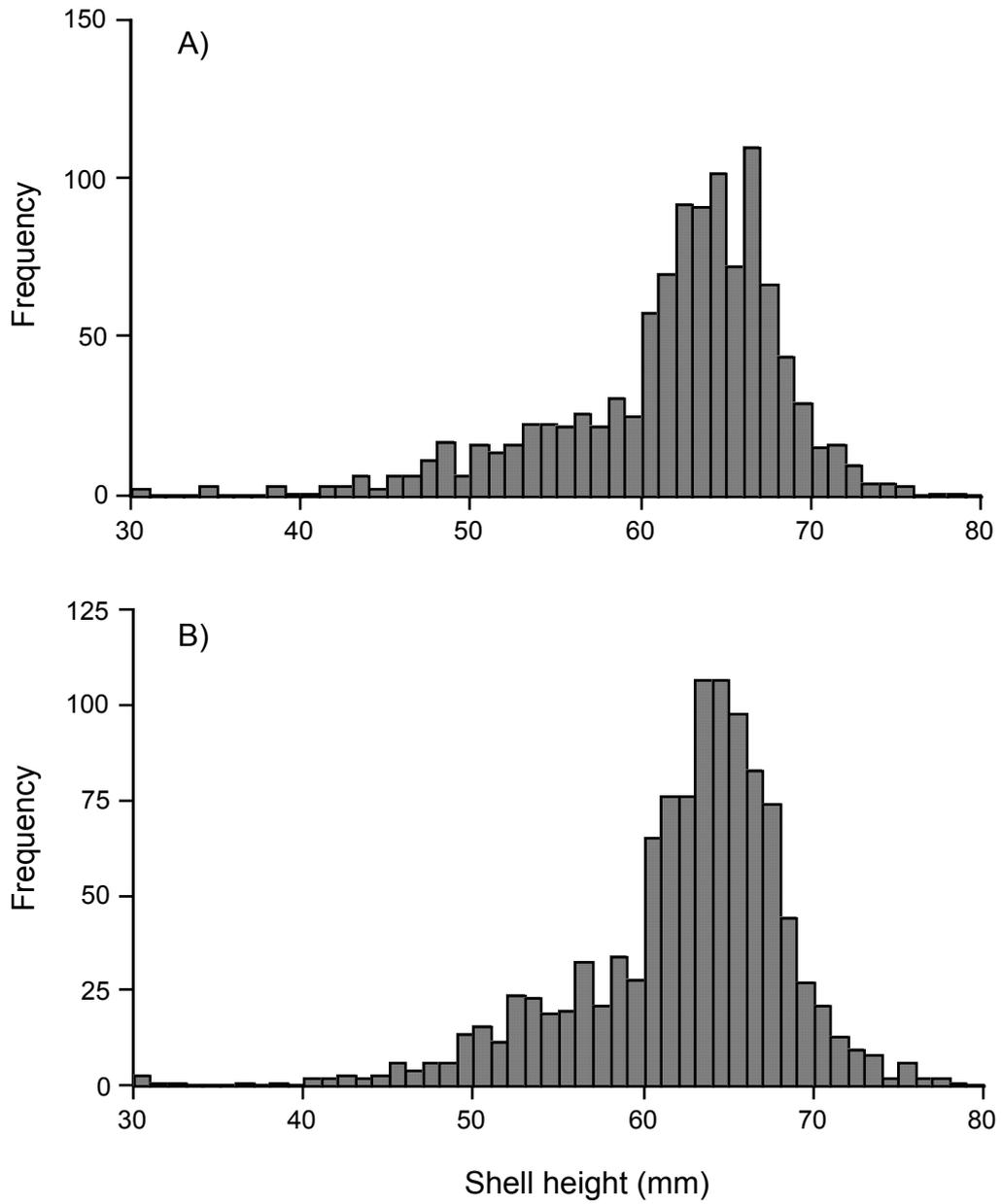


Figure 20 Size frequency distribution of queen scallops caught with skid gear on 3rd July 2001 at PSM 51(A) with no cover net and (B) with a mark 1 cover net.

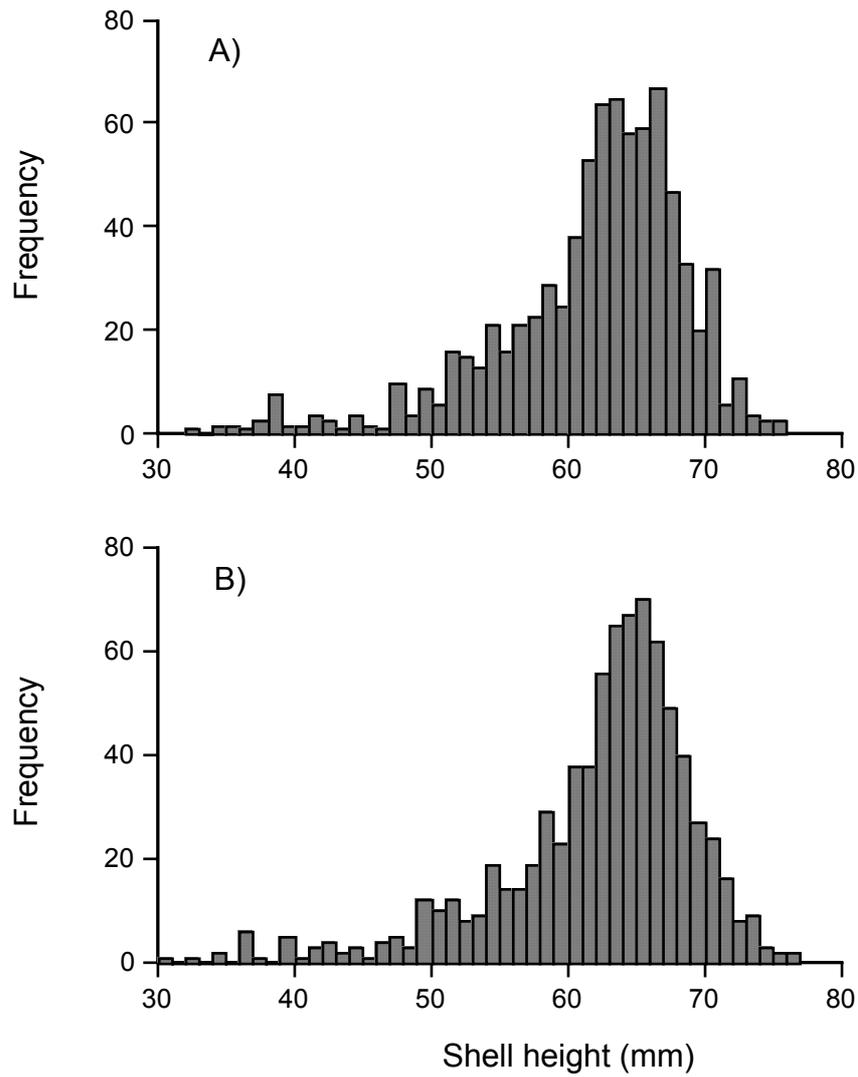


Figure 21 Size frequency distribution of queen scallops caught with skid gear on 27th August 2001 (A) with no cover net and (B) with a mark 1 cover net.

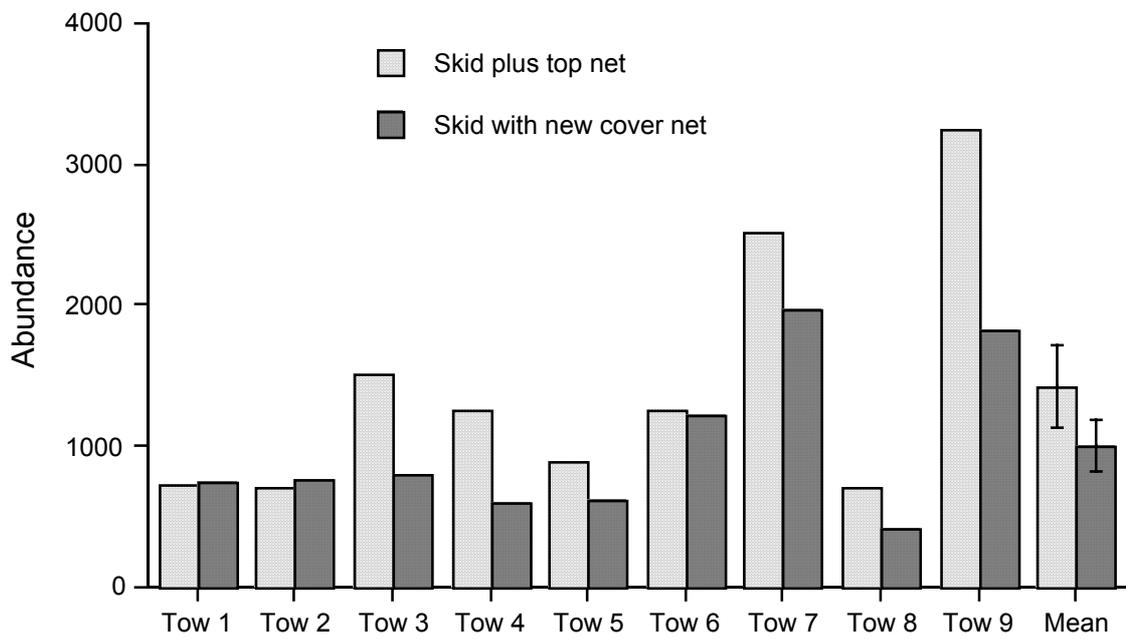


Figure 22 Comparison of total catch of queen scallops in skid gear with top net (as used in seasonal work) with skid gear and mark 2 cover net. Tows were carried out over 2 days, 1-3 on 29.08.01 and 4-9 on 30.08.01

3.2.2. Size Selectivity

3.2.2.1 THE EFFECT OF TOOTH SPACING AND MESH SIZE ON SELECTIVITY OF THE *CALLISTA CHIONE* AND *DONAX TRUNCULUS* DREDGE

Introduction

Among the Portuguese, bivalve fishing is an ancient and traditional activity. Until the 1960's the harvesting of bivalves was restricted to estuaries and lagoons. The exploitation of bivalve ocean beds along the Portuguese coast is recent and was initiated in 1969 by the Spanish fleet under the auspices of the "Portuguese-Spanish Fishery Agreement". The Spanish fleet directed its fishing effort mainly towards two species *Ensis siliqua*, *Spisula solida* and *Chamelea gallina*. The high economic value of these species in the external market soon aroused great interest in the Portuguese fleet. Thus, the fishing effort as well as the gear efficiency increased and as a consequence bivalve beds soon showed signs a depletion. During the past 20 years the fishing effort was also directed to other species, namely *Callista chione* and *Donax trunculus*. These species constitute the target of a specific fishing activity carried out by an artisanal fleet. The boats involved in the fishery are 4-15 m long, with engines of 17-150 Hp and a crew of 1 to 5 fishermen. Dredges are the gear used in this fishery. The basic structure of the Portuguese clam and razor dredge is a small, heavy semicircular iron structure, with a net bag and a toothed lower bar at the mouth. The lower bar has 12 to 14 teeth with a maximum length of 55 cm spaced 1.5 to 2.5 cm apart, acting as the lower leading edge of the dredge. Welded to this iron structure are three metal shafts forming a kind of hen's foot where the towing cable is attached.

One of the most important problems that managers are facing is the capture of undersized individuals. In fact, there are high discard rates in this fishery because adults and juveniles are found in the same areas. The mortality due to fishing is affected by the survival rates of discarded individuals. Recently Gaspar and Monteiro (1999) showed that undamaged juveniles of *Spisula solida* are extremely sensitive to the time of exposure on the deck. These authors estimated a fishing mortality rate of 0.003 per minute out of water. Because catches are usually sorted at the end of the trip mortality is high. This is a problem of particular importance during summer, when high temperatures increase mortality and densities of undersized individuals are higher. A good management option would be to avoid a large by-catch of undersized clams and razor clams in the first place. According to Lart et al. (1997), chain mail mesh size (belly ring size) and tooth spacing (see also [Sections 3.2.2.3-5](#)) are the most important dredge features that affect selectivity in scallop dredges. The effect of mesh size and tooth spacing in Portuguese clam dredges was studied with the objective of quantifying the catch of undersized individuals retained and estimating a minimum mesh size.

Methodologies

Experimental design

Selectivity experiments were carried on the southwest coast (off Comporta) and on the south coast (off Fuzeta) of Portugal (Figure 1), for *Callista chione* and *Donax*

trunculus respectively. The experiments were conducted on board on board of the IPIMAR's research vessel "DONAX". For all experiments samples were collected on sandy bottoms between 3 and 10m depth. The dredges used on this study were similar to those used by the commercial dredge fleet. For experimental purposes the dredges were fitted with interchangeable tooth bars. The specifications of the gears used in the three experiments are summarised in Table 1.

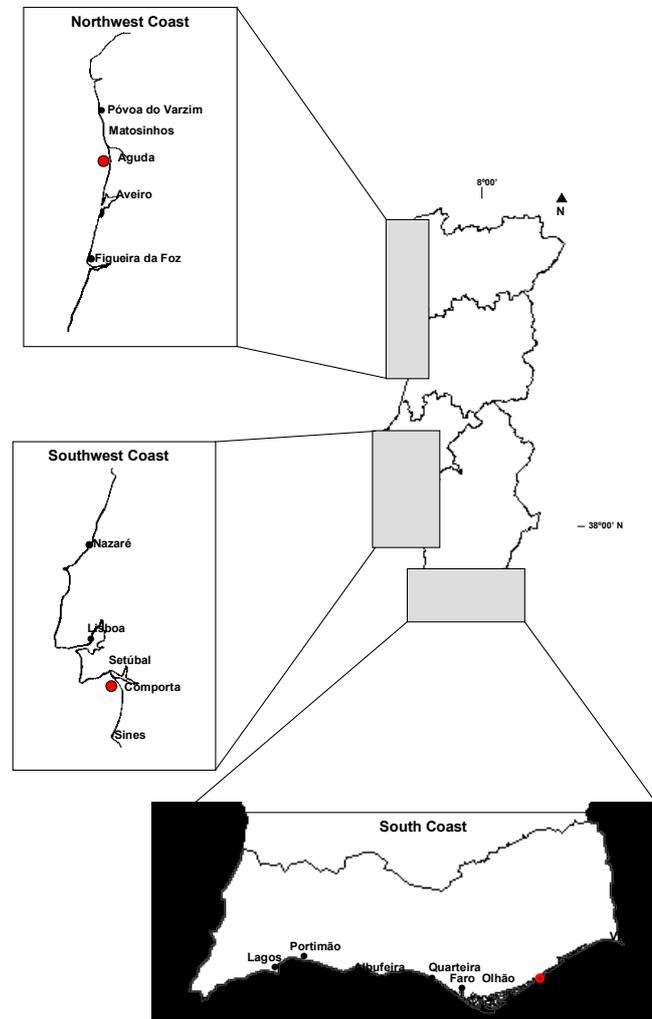


Figure 1 – Portugal coast map with indication of the sampling locations.

Table 1. Gear specifications of the dredges used in the selectivity experiments.

	Dredge for <i>Callista chione</i>	Dredge for <i>Donax trunculus</i>
Shape of the iron structure	Semicircular	Semicircular
Length of the mouth	60 cm	40 cm
Distance between rods	2.4 cm	0.8 cm
Tooth length	34 cm	13.5cm
Tooth angle	20°	20°
Length of the net bag	200 cm	200 cm

For *Callista chione*, six different mesh sizes 50, 60, 70, 80, 90 and 100 mm and four tooth spacing 20, 40, 60 and 80 mm were investigated. In the case of *Donax trunculus*, taking into consideration the results obtained for *Callista chione*, only the influence of mesh size was investigated. The mesh sizes were chosen to cover a range of dimensions and include the one currently at use (50 and 25 mm respectively for *Callista chione* and *Donax trunculus*). All meshes were measured across the diagonal of fully stretched nets. For each gear combination (teeth and mesh size) 5 hauls were made.

The dredge selectivity experiments were conducted by attaching a cover bag with a 20 mm mesh to the gear. This bag was 1.6 times longer and wider than the primary net bag and did not impede the natural flow of water through the net. Two dredges with equal teeth and different bag meshes were towed side by side. All fishing hauls were conducted at a towing speed of 1 to 1.5 knots with duration of 15 minutes. After each haul the catches in the bag and in the cover were sorted separately. For all species the standard length was measured to the millimetre and subsequently grouped into 2 mm classes, in the case of *Donax trunculus* and into 5 mm classes in the case of *Callista chione*.

Data analysis

It was assumed that the selection curves followed a logistic model, which is expressed by the following equation (Paloheimo and Cadima 1964):

$$P(L) = \frac{1}{1 + \exp^{-(a + b L)}}$$

Where: L is the centre of the length class; P (L) is the retention proportion for length class L; and \underline{a} and \underline{b} are parameters.

The logistic curves were fitted using non-linear regression with the routine NLIN, part of the statistical package SAS (SAS Institute Inc., 1989). This interactive curve fitting procedure employs a non-linear least-square regression method (Gauss-Newton) and estimates \underline{a} and \underline{b} from the above equation. Useful parameters of a selection curve are $L_{50\%}$ (theoretical length at which 50% of the individuals are retained) and the selection

range, SL (difference between $L_{75\%}$ and $L_{25\%}$). These two parameters are related to the parameters of the logistic equation by the expressions:

$$L_{50\%} = -\frac{a}{b}$$

and

$$S.R. = 2 \times \frac{\ln 3}{b}$$

Results

Callista chione

The effect of tooth spacing on the selectivity of the dredges was analysed first. In these experiments mesh size was ignored and catches in the bag and cover were added together for hauls with the same tooth spacing. The length distributions of the catch are represented in Figure 2.

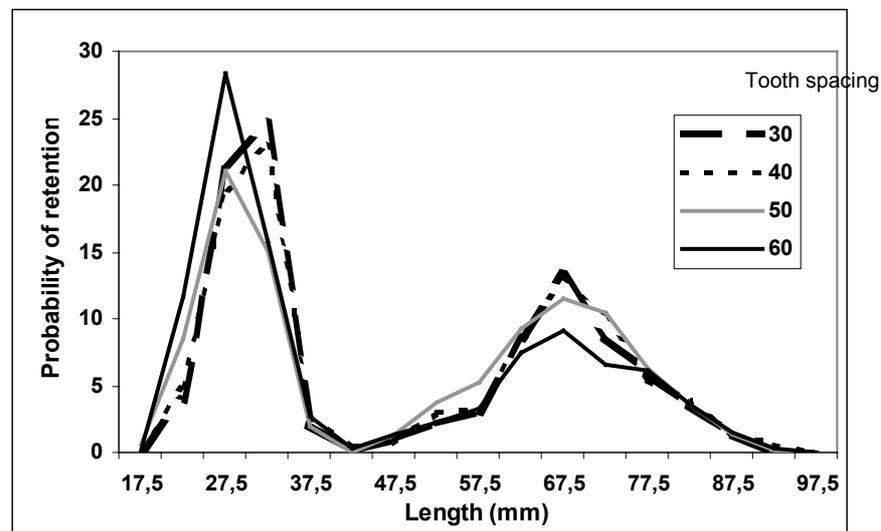


Figure 2 Length frequency distributions by tooth spacing obtained for all experiments.

If tooth spacing had a selective effect in the size range caught, the length frequencies obtained for each tooth spacing would be different. One would expect that narrower tooth spacing would correspond to length distributions with a higher proportion of smaller individuals. This hypothesis was not confirmed and the opposite was found the wider tooth spacing corresponded to higher proportion of small individuals. These results led to the conclusion that tooth spacing, within the ranges studied here, does not affect the selectivity of the dredge.

After establishing that tooth spacing was of no importance for the selectivity of this dredge, the effect of mesh size was studied ignoring this factor, that is, data from the hauls with the same mesh size were pooled, ignoring the tooth spacing used. Figure 3 shows the estimated selectivity curves for *Callista chione*. Table 2 summarises the parameters obtained for different mesh sizes.

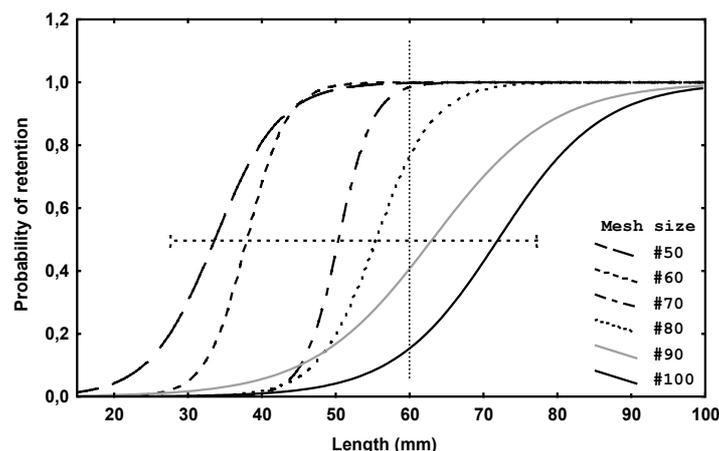


Figure 3 - Selectivity curves estimated for *Callista chione*. The vertical line represents the minimum legal landing length (MLL) of this species. The horizontal line represents $L_{50\%}$.

Table 2 – Selectivity parameters for *Callista chione* where: a and b are the parameters of the logistic curve, r is the coefficient of determination, $L_{50\%}$, $L_{75\%}$ and $L_{25\%}$ are the lengths at which 50%, 75% and 25% of the individuals are retained, SR is the selection range ($L_{75\%} - L_{25\%}$) and SF is the selection factor ($L_{50\%} / \text{mesh_size}$).

Selectivity parameters								
Mesh	a	b	r	$L_{25\%}$	$L_{50\%}$	$L_{75\%}$	S.R	S.F.
50mm	-7.6669	0.2279	0.9977	28.82	33.64	38.46	9.641	0.673
60mm	-14.031	0.3696	0.9484	35.00	37.97	40.94	5.946	0.633
70mm	-21.638	0.4295	0.9847	47.82	50.38	52.94	5.116	0.720
80mm	-14.340	0.2586	0.9919	51.20	55.45	59.70	8.497	0.693
90mm	-7.7265	0.1125	0.9737	54.10	63.07	72.04	17.936	0.701
100mm	-10.252	0.1424	0.9799	64.28	71.99	79.71	15.429	0.720

For mesh sizes 50, 60, 70, 80, 90 and 100 mm the 50% retention lengths are, respectively, 33.64, 37.97, 50.38, 55.45, 63.07 and 71.99 mm. Corresponding selectivity intervals ($L_{25\%} - L_{75\%}$) intervals were 9.64, 5.95, 5.12, 8.5, 17.94 and 15.43 mm. For Portuguese waters The MLL for this species is 50mm, which corresponds to the $L_{50\%}$ estimated for 70mm mesh size. For the other meshes investigated the $L_{50\%}$ obtained was below (for 50 and 60 mm meshes) or above (for 80, 90 and 100mm meshes) the MLL. The selection range estimated for these five meshes does not include the MLL for this species.

Donax trunculus

Taking into consideration the results presented above, the selectivity experiment carried out for *Donax trunculus* only contemplated the influence of mesh size. A total of 3529 *Donax* individuals were caught in 20 hauls. The selectivity curves estimated for this species are shown in Figure 4. Table 3 gives the estimated selectivity curves parameters for each mesh.

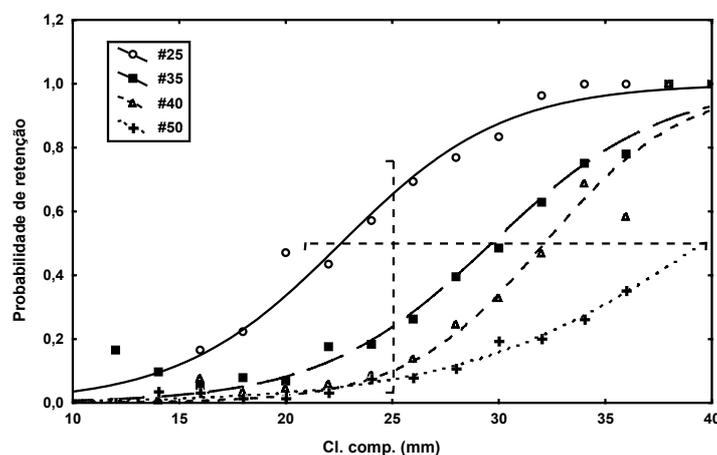


Figure 4 - Selectivity curves estimated for *Donax trunculus*. The vertical line represents the minimum legal landing length (MLL) of this species. The horizontal line represents $L_{50\%}$.

Table 3 – Selectivity parameters for *Donax trunculus* where: a and b are the parameters of the logistic curve, r is the coefficient of determination, $L_{50\%}$, $L_{75\%}$ and $L_{25\%}$ are the lengths at which 50%, 75% and 25% of the individuals are retained, SR is the selection range ($L_{75\%} - L_{25\%}$) and SF is the selection factor ($L_{50\%} / \text{mesh_size}$).

Selectivity parameters								
Mesh	a	b	r	$L_{25\%}$	$L_{50\%}$	$L_{75\%}$	S.R	S.F.
25mm	-6.1998	0.2628	0.9874	19.41	23.59	27.77	8.3608	0.9436
35mm	-7.6633	0.2500	0.9829	26.26	30.65	35.05	8.7896	0.8757
40mm	-10.185	0.3073	0.9797	29.57	33.14	36.72	7.1500	0.8285
50mm	-7.0398	0.1734	0.9894	34.26	40.59	46.93	12.671	0.8118

The 50% retention lengths estimated for mesh sizes 25, 35, 40 and 50 mm were 23.59, 30.65, 33.14 and 40.59 mm, respectively. Only the value estimated for the 25mm mesh size is below the 25 mm MLL for this species. The selection range for this mesh is very wide, which leads to the capture of a high proportion of undersized individuals. For the other three mesh sizes investigated the selection range does not include the MLL. Assuming the selection factor is more or less constant for a given gear and

species, and is approximately equal to the ratio $L_{50\%}/\text{mesh_size}$, the indicated mesh, chosen to obtain a $L_{50\%}$ close to 25 mm, would be 28 mm.

Discussion

Dredge selectivity may be affected by several factors, namely the the nature of the bottom, the duration of the tow and the design of the dredge (Drinkwater 1974). According to this author, the nature of the bottom is one of the most important factors that influence the selectivity of the scallop dredges. This author reported that the dredge is likely to bounce in some places and fill with mud and stones in others. Therefore the selectivity will depend partially on the nature of the bottom deposits scraped up by the dredge. In the Portuguese dredge fishery, the nature of the bottom does not constitute a problem, since the exploited species only forms extensive and dense beds on very clean sandy bottoms. The second factor affecting selectivity is the duration of the tow, which was not studied in the present work. However, the experiments were designed so that this factor would be constant. Of the factors inherent to the dredge itself, tooth spacing and the mesh size were considered to be potentially the most important. In the present study, the effect of tooth on the catch of *Callista chione* spacing was analysed first. Our results showed, with the dredge design used, the spacing of the teeth was of no importance, therefore the tooth bar must act as a hoe. On the contrary, and as expected, the effect of mesh size on the catches is clear for *Callista chione* and *Donax trunculus*, with the retained proportion of undersized clams decreasing with the increasing mesh size. Management of these species usually combines a minimum legal landing length (MLL) with a minimum mesh size (MMS). The objective is to aid enforcement by discouraging fishermen from using very small meshes that lead to the rejection of a large proportion of the catch. In the present study, a 50% selection length of approximately 25 mm was the target for the Donax dredge because this value is the current minimum landing size for *Donax trunculus*. Similarly, for the *Callista* dredge, our target was a 50% selection length of 50 mm, the minimum legal length for *Callista chione*. The present experiments suggests that the optimum mesh for *Callista chione* would be 70mm and for *Donax trunculus* would be 28mm.

3.2.2.2 SELECTIVITY OF THE *SPISULA SOLIDA* DREDGE IN RELATION TO TOOTH SPACING AND MESH SIZE

Introduction

Despite the occurrence of several bivalve species on the northwestern coast of Portugal with economic interest, including *Spisula solida*, *Ensis siliqua*, *Donax vittatus* and *Glycymeris glycymeris*, the commercial dredge fleet only directs its fishing effort towards the white clam *Spisula solida*. In 1963, when the fishery started, the dredge fleet comprised of 63 fishing boats. In the late 1980's and early 1990's this increased markedly, involving 90 boats and approximately 500 professionals by 1994. At that time the white clam beds showed signs of overfishing and although some regulating measures were adopted, including daily quotas, it was not possible to stop the decline, leading to the collapse of the stock and closure of the fishery in 1995.

After several years of very poor stocks, a bivalve survey carried out by IPIMAR in 1999 showed a moderate recovery of the *Spisula solida* beds. The fishery was cautiously reopened during that same year, restricting the number of boats to 11 and to a daily quota of 240 kg/boat. Results obtained from a survey in 2001 revealed that the abundance of *Spisula solida* was still increasing. Thus, in order to change current legislation accordingly, consultation is in progress to define a suite of new technical measures including the number, size and selective properties of the dredges as well as the number of licences and permitted type.

Good management requires the adaptation of fishing gear specifications to the biology/ecology of target species. Thus, gear restrictions, such as minimum mesh size, must be enforced in order to prevent the harvest of undersized individuals, allowing them to both grow to a more valuable market size and to reach a size at which they can reproduce at least once before capture. Therefore, the minimum mesh size stipulated for a certain fishery usually takes into consideration the minimum legal size of the target species. Knowledge of both the selection range and 50% retention length is necessary to determine which mesh size should be adopted in a fishery and to assess the effect of any change in mesh regulation (Pope et al. 1975).

Studies concerning selectivity of bivalve dredges are scarce when compared with the vast literature available on the selectivity of trawls. Furthermore, most studies deal with the selectivity of the scallop dredge (e.g. Bourne 1966; Caddy 1971, Caddy 1972; Chapman et al. 1977; Serchuk and Smolowitz 1980); Worms and Lanteigne 1986; Lart et al. 1997).

Within dredge fisheries, the gear specifications that have received most attention in relation to selectivity studies have been mesh size, ring size and tooth spacing. The selective properties of tooth spacing, however, have been unclear. This paper reports on the effects of mesh size and tooth spacing on the catch of the white clam dredge used off the northwestern coast of Portugal. The selectivity experiments were developed in order to advise which mesh size and tooth spacing should be implemented in this *Spisula solida* fishery.

Methodologies

Experimental design

The fishing trials were conducted during July 1999, near the Aguda region, on the northwestern coast of Portugal, one of the country's most important white clam fishing grounds. All fishing operations were carried out on board the commercial fishing vessel "Narciso Sérgio". This boat has an overall length of 15m, 135HP and is of 16.54 GRT. The dredges used in the selectivity experiments were identical to those used by the commercial dredge fleet operating in the area.

The Portuguese northwestern clam dredge comprises of a heavy rectangular metal bar mouth (2m long and 0.35m high), a tooth bar (tooth length = 6.5cm) and a netting bag (4.5m long). Welded to the mouth structure is a triangular frame and onto which the towing cable is attached. The gear weighs approximately 80kg. Detailed gear specifications of the experimental dredge are presented in Figure 1.

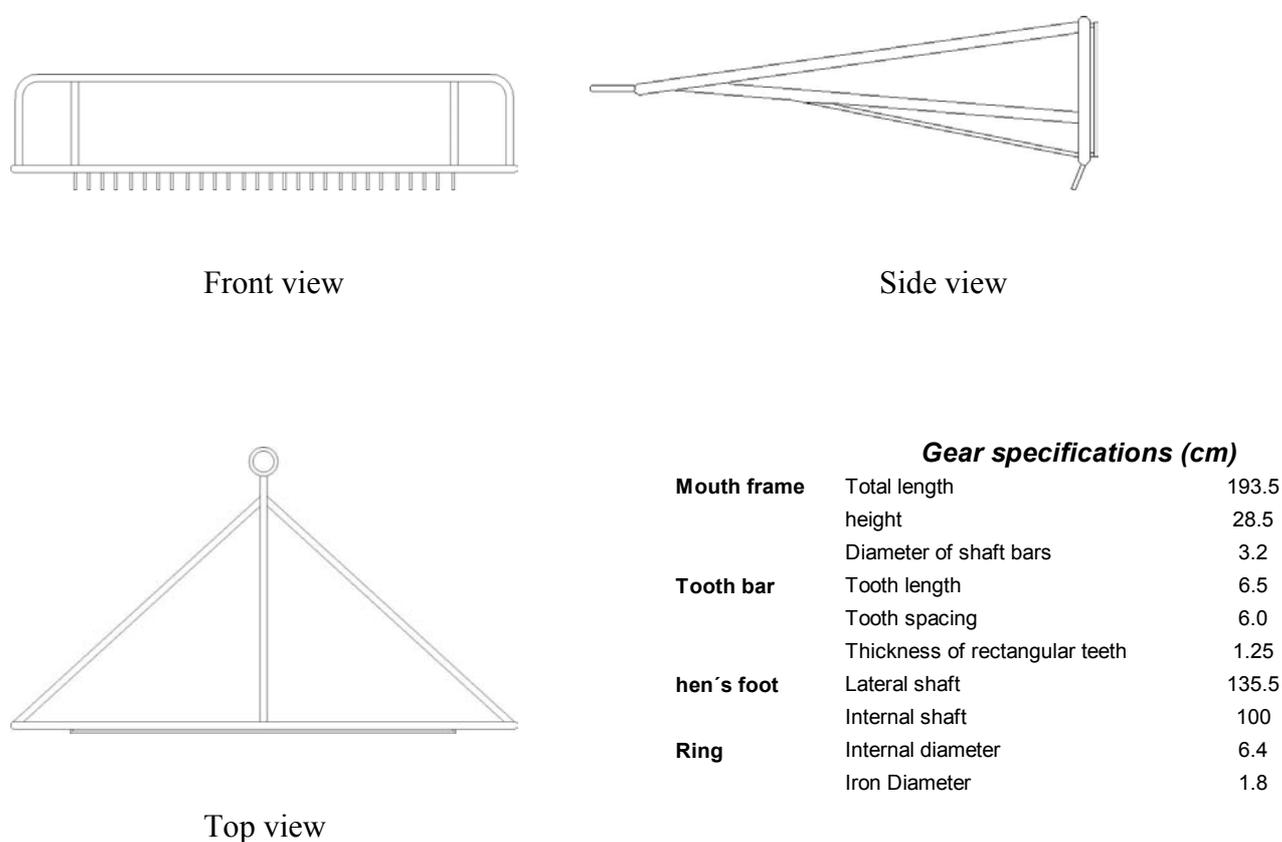


Figure 1. Diagram of the clam dredge used off the northwestern coast of Portugal and technical specifications.

Three mesh sizes (35, 40, 50mm) and three sizes of tooth spacing (20, 40 and 60mm) were investigated. All net bags used were constructed from green multiple-twisted polyethylene twine and designed in order to have the same width and length. All mesh was measured across the diagonal of fully stretched nets. In order to assess the effect

of tooth length on the catches of *Spisula solida*, a dredge without a tooth bar was used.

During these experiments the primary collection bag was covered with a second bag with a 20mm-diamond mesh size. This covering mesh was constructed from nylon twine and was approximately 1.5m longer and wider than the primary bag, allowing a natural flow of water through the net bag.

During the fieldwork two dredges were towed side by side. For each tooth spacing/mesh size combination three tows were performed. All tows were conducted at depths of between 8 and 10m, at a constant speed of 2 knots and with a tow duration of 15 minutes.

Samples from the bag and cover bag were sorted separately. For the target species, the standard shell length was measured to the nearest lower millimetre, and subsequently individuals were grouped into 2mm length classes.

Data analysis

Logistic selectivity curves of individual hauls were fitted using the SELECT method developed by Millar (1991). Mean selectivity curves for each tooth spacing and mesh size were also estimated using the Fryer model of between-haul variation (Fryer 1991). The Fryer model is a variant of Laird and Ware (1982) effects model and differs from the last by the fact that the covariance level is known.

The Fryer model is applied under the assumption that selectivity curves from individual hauls vary around a mean selectivity curve according to a multivariate normal distribution. It was considered that this assumption was satisfied for all hauls performed during the study. The explanatory variables, i.e. tooth spacing and mesh size, were considered as fixed effects, allowing assessment of the influence of these gear specifications in the 50% length retention ($L_{50\%}$) and selection range (SR).

For all analyses, the residual maximum likelihood (REML) procedure was used to fit the selection curves. The total deviance across all data was used as a measurement to explain the goodness of fit. The analyses were performed using the software package iCC 2000 SELECTIVITY available through the internet : WWW.CONSTAT.DK/ICC.

In order to assess the influence of tooth length in the catches six tows were performed using two dredges, with teeth and without teeth, but with the same mesh size (20mm). Size frequency distributions obtained from these two dredges were constructed from raw data and compared using the Kolmogorov-Smirnov test (Zar 1996).

Results

The selectivity parameters ($L_{50\%}$ and SR) estimated and the goodness of fit for each haul obtained using the SELECT model are shown in Table 1. The analysis of p-values showed that the logistic function could be used for the majority of the hauls to describe the *Spisula solida* selection curves. Figure 2 shows the estimated selectivity curves for each mesh size/ tooth spacing combination.

Table 1. Selectivity parameters ($L_{50\%}$ and SR) estimated and the goodness of fit for each haul using SELECT.

Haul	Tooth spacing (mm)	Mesh size (mm)	$L_{50\%}$ (var $L_{50\%}$)	SR (var SR)	G. of fit (p-value)
1	20	35	26.13 (0.03)	2.80 (0.06)	0.0927
2	20	35	26.24 (0.12)	4.07 (0.25)	0.0085
3	20	35	24.93 (0.14)	3.92 (0.25)	0.0295
4	40	35	26.62 (0.02)	2.49 (0.04)	0.4177
5	40	35	27.15 (0.13)	3.58 (0.31)	0.0008
6	40	35	26.49 (0.03)	2.11 (0.06)	0.9902
7	60	35	24.75 (0.10)	1.95 (0.12)	0.9980
8	60	35	24.49 (0.06)	3.41 (0.11)	0.2703
9	60	35	24.28 (0.08)	3.16 (0.12)	0.7290
10	20	40	27.37 (0.05)	4.30 (0.10)	0.4775
11	20	40	26.53 (0.36)	4.58 (0.10)	0.3672
12	20	40	25.47 (0.21)	6.53 (0.87)	0.0233
13	40	40	28.13 (0.09)	4.88 (0.23)	0.4092
14	40	40	25.45 (0.17)	3.60 (0.27)	0.5098
15	40	40	31.14 (1.25)	8.32 (3.41)	0.0000
16	60	40	26.22 (0.15)	4.02 (0.29)	0.0158
17	60	40	24.09 (0.09)	3.41 (0.14)	0.9969
18	60	40	24.04 (0.12)	5.85 (0.27)	0.1865
19	20	50	32.25 (0.10)	5.05 (0.46)	0.0974
20	20	50	32.79 (0.17)	5.25 (0.27)	0.1257
21	20	50	32.66 (0.62)	7.10 (1.03)	0.0052
22	40	50	33.08 (0.26)	5.89 (0.34)	0.4870
23	40	50	28.83 (0.10)	6.95 (0.29)	0.0617
24	40	50	32.90 (0.31)	6.07 (0.55)	0.1983
25	60	50	31.87 (0.30)	6.61 (0.55)	0.0106
26	60	50	33.23 (0.54)	6.25 (0.87)	0.5033
27	60	50	31.29 (0.33)	6.44 (0.72)	0.0484

The effect of tooth spacing and mesh size on the catch was then investigated using the Fryer model. The results obtained, after all potential effects had been considered, are given in Table 2 along with their standard deviations and p-values. The p-values observed showed that tooth spacing does not have a significant effect on the selection range (SR). This parameter was therefore removed and a new analysis was performed. From Table 3 it can be observed that $L_{50\%}$ is not significantly affected by tooth spacing. Mesh size, however, was shown to have a highly significant ($P < 0.00001$) influence, both in terms of $L_{50\%}$ and SR. These results lead to the conclusion that tooth spacing, within the ranges studied here, does not affect the selectivity of the dredge.

After proving that tooth spacing does not affect the selectivity of the Portuguese northwestern dredge, new Fryer mean selectivity curves were then estimated for the three mesh sizes investigated ignoring tooth spacing. The estimated selectivity parameters are given in Table 4.

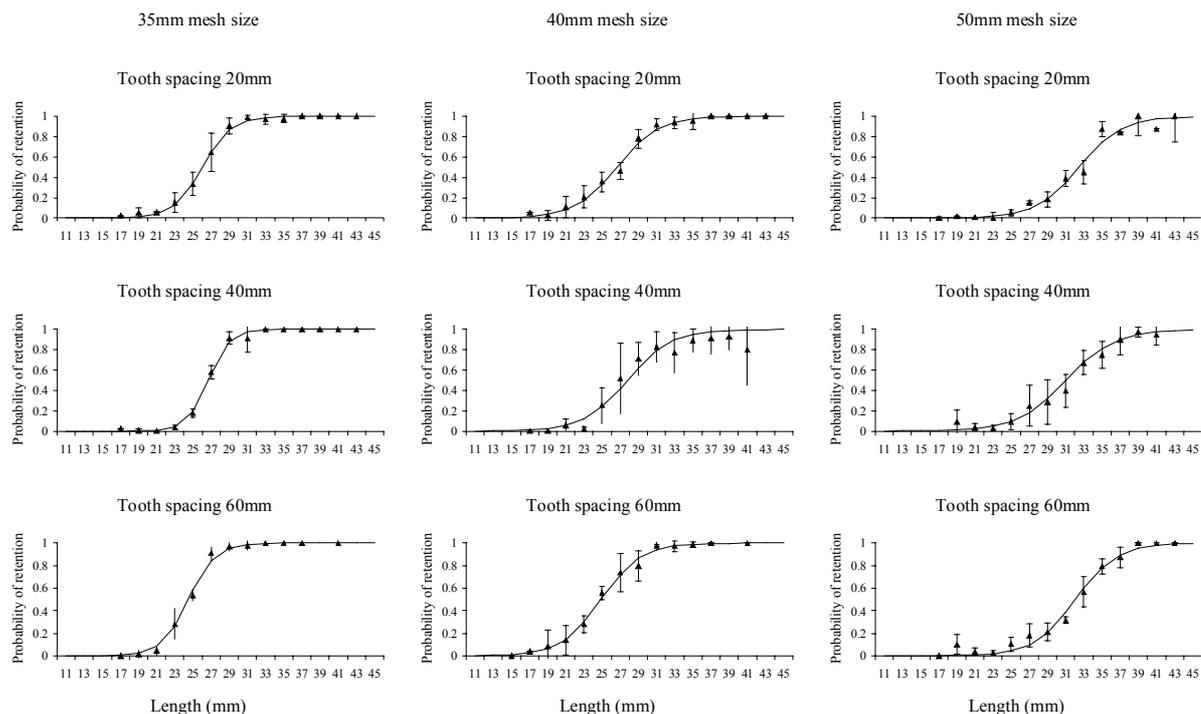


Figure 2. Selectivity curves estimated for *Spisula solida* for each mesh size and tooth spacing combination.

Table 2. Fryer model. Summary of the results obtained when all potential effects were considered.

Parameter	Estimate	Std.Dev	t-values	DF	p-value
Intcpt(L _{50%})	16.2900	1.7940	9.08	48	0.0000
Intcpt(SR)	-1.5180	0.8154	-1.86	48	0.0687
T. spacing (L _{50%})	-0.8046	0.4291	-1.88	48	0.0669
T. spacing (SR)	0.0108	0.1996	0.05	48	0.9570
M. size (L _{50%})	3.1270	0.4280	7.31	48	0.0000
M. size (SR)	1.5010	0.1989	7.55	48	0.0000

Table 3. Fryer model. Summary of the results obtained after removing the effect of tooth spacing (SR).

Parameter	Estimate	Std.Dev	t-Values	DF	p-Value
Intcpt(L _{50%})	16,2900	1,7940	9,08	48	0,0000
Intcpt(SR)	-1,5100	0,7638	-1,98	48	0,0537
T. spacing (L _{50%})	-0,8035	0,4287	-1,87	48	0,0669
M. size (L _{50%})	3,1260	0,4278	7,31	48	0,0000
M. size (SR)	1,5010	0,1943	7,72	48	0,0000

Table 4. Parameter estimates for each mesh size investigated. Parameter estimates are quoted along with associated standard deviations and t-values. Also shown are the parameter variance and the between-haul variation.

	Mesh size (mm)					
	35		40		50	
	L _{50%}	SR	L _{50%}	SR	L _{50%}	SR
Estimate	25.68	2,958	26.35	4,629	31.94	6,009
Std. Desv.	0.3517	0.2427	0.6313	0.3507	0.4502	0.2624
t-Value	73.02	12.19	41.73	13.20	70.96	22.90
DF	16	16	16	16	16	16
Significance level	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001
Parameter Variance						
L _{50%}	0.1237	-0,0120	0.3985	0,0645	0.2027	-0,0452
SR	-0,0120	0.0589	0,0645	0,1230	-0,0452	0,0689
Between haul variation						
L _{50%}	1,0350	-0.1003	3,3870	0.5672	1,6260	-0.5049
SR	-0.1003	0.4002	0.5672	0.7764	-0.5049	0.2888

As expected, both L_{50%} and SR increase with an increase with mesh size. The 50% retention lengths obtained for mesh sizes of 35, 40 and 50 were 25.68, 26.35 and 31.94mm respectively. For all three mesh sizes, the estimated L_{50%} were above the Minimum Landing Size (MLS) stipulated for *Spisula solida* (25mm). The estimated selection ranges have a minimum of 2.96mm for the 35mm-mesh size and a maximum of 6.01mm for the 50mm-mesh size. Only the SR estimated for mesh sizes of 35mm and 40mm comprehends the MLS for the species (L_{25%} = 24.30mm and 23.98 for mesh size of 35mm and 40mm, respectively). The large between-haul variation was recorded for the 40mm-mesh size (Table. 4) both for L_{50%} and SR, which ranged from 24.04 to 31.14mm and from 3.41 to 8.32mm, respectively (see table 1).

Table 5. Results of the pair-wise comparison performed between L_{50%} and SR per mesh size.

Mesh size (mm)	Parameter	40		50	
		L _{50%}	SR	L _{50%}	SR
35	L _{50%}	0.3939		0.0000	
	SR		0.0003		0.0000
40	L _{50%}			0.0000	
	SR				0.0016

In order to determine the existence of between-mesh size differences in selectivity in terms of L_{50%} and SR, a pair-wise comparison was performed. The model was then refitted in order that mesh size, after being fixed, acts as the explanatory variable. No significant differences were found for mesh sizes of 35 and 40mm mean selection

curves as far as the $L_{50\%}$ is concerned (Table 5). Contrary to this, when the SR estimated for these two mesh sizes were compared significant statistical differences were obtained. Between 35-50mm and 40-50mm meshes, significant differences were observed for both $L_{50\%}$ and SR.

Histograms of size frequency distribution for dredges with and without teeth are shown in Figure 3. Results from the Kolmogorov-Smirnov test ($D'=25.4 > D_0$ ($\alpha=0.05$), $H_0: F_A(x) = F_B(x)$, is rejected) indicate that the size distributions were significantly different.

From Figure 3 it can be seen that catches from the dredge without teeth are made up of a significant fraction (51.66%) of undersized individuals (shell length inferior to 25mm), while with dredge equipped with teeth the catches consisted of a higher proportion (73.76%) of individuals with a size greater than to the legal minimum. These results clearly show that tooth length has a significant influence on the catch length composition as far as this *Spisula solida* fishery is concerned.

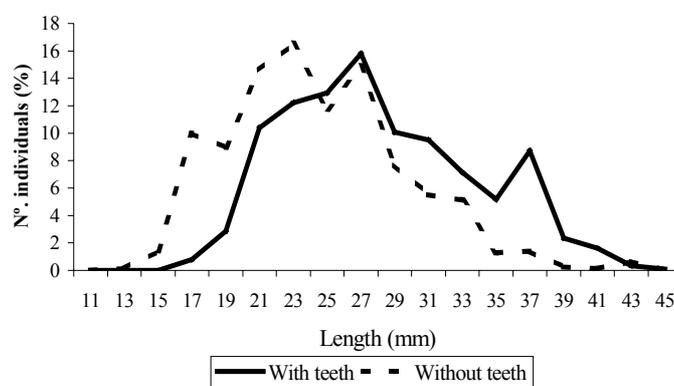


Figure 3. *Spisula solida* length-frequency distributions for dredges with and without teeth.

Discussion

Dredge selectivity may be affected by several factors, including the type of seabed, tow duration and velocity, the hanging coefficient of the net bag, the twine material and its diameter, tooth spacing and mesh size. Only tooth spacing and mesh size were investigated during this study and great care was taken to ensure that all other factors would be constant during the experiments.

Gaspar et al. (1999) noted that tooth spacing was of no importance in the selectivity of the dredges used on the south coast of Portugal. The selectivity experiments carried out by these authors, however, covered a very narrow range of tooth spacings (from 15 to 25mm) that never exceeded the MLS for the species (25mm), possibly influencing the results obtained. To overcome this uncertainty and to find out if tooth spacing has in fact an effect on selectivity, a wider range of tooth spacing was used in the present study, with the largest tooth spacing being larger than the L_{∞} estimated for this species (45.08mm; Gaspar 1996a).

The results obtained in the present work corroborate the findings of Gaspar et al. (1999) and clearly showed that for the *Spisula solida* fishery the spacing of the teeth

does not have any effect on selectivity. Therefore, the tooth bar must act as a hoe, creating a bulldozing effect. The different values of $L_{50\%}$ and SR estimated for hauls undertaken with different tooth spacing and for a given mesh size, are mainly concerned with between-haul variation rather than with the effect of tooth spacing. The cause of between-haul variation is unclear, but could arise from changes in uncontrolled factors, namely spatial distribution and the structure of stocks (Millar and Fryer 1999).

Our results show the influence of tooth length on the size composition of catches. The comparison of length frequency distributions of *Spisula solida* between dredges with and dredges without teeth show that catches from the latter dredge contained a greater fraction of juveniles. The catches from the dredge equipped with teeth were composed of a higher proportion of individuals of a size greater than the minimum legal size.

As was expected, this result indicated that larger clams burrow deeper in the sediment than smaller ones. Tooth length is therefore directly related to the capture efficiency of dredges. The use of teeth allows increasing the gear efficiency for the capture of larger clams, which become, in this way, more vulnerable to dredging. For the purpose of stock management, therefore, the tooth length stipulated for a certain bivalve fishery must take into consideration some ecological characteristics of the species, in particular the maximum burrowing depth.

The effect of tooth spacing on selectivity was found to be highly significant in the Japanese surf clam and sunary surf clam hydraulic dredge fisheries (Nashimoto et al. 1983; Nashimoto 1984, Nashimoto 1985). Other studies concerning the selectivity of scallop dredges in relation to tooth spacing, mesh size and ring size have demonstrated that the mesh or ring effects are the most important (Baird and Gibson 1956; Drinkwater 1974; Lart et al. 1997).

Results, presented by Gaspar *et al.* (1999) and those presented in this study suggest that in bivalve fisheries the effect of tooth spacing on selectivity depends on the way fishing gear are operated. Therefore, it can be concluded that in non-hydraulic dredges equipped with a tooth bar and used to catch infaunal bivalve species, tooth spacing never has an effect on selectivity. Nevertheless, we believe that tooth spacing can not be dissociated from tooth length. In fact, there would exist a particular spacing between teeth after which the length of teeth would have no effect on the catch. Consequently, the dredge acts as if a tooth bar is not presented.

The selectivity effect of mesh size on selectivity was also assessed in the present study. In commercial clam fisheries the importance of mesh size regulation is extremely important in order to prevent the capture of juveniles. Restriction on mesh size must take into account the minimum landing size (MLS) stipulated for the species. In Portugal, the current MLS for *Spisula solida* is 25mm, thus a 50% selection length of approximately 25mm is the target for this fishery. For all mesh sizes experimented with, the $L_{50\%}$ estimated were above the MLS. It is worth noting, that the selection ranges estimated for the three mesh sizes were very narrow (2.96, 4.63 and 6.01mm for mesh sizes of 35, 40 and 50mm, respectively). Therefore, the $L_{25\%}$ estimates for mesh sizes of 35 and 40mm also approaches the MLS stipulated for this species.

These results suggest that a high percentage of juveniles pass through the mesh bag during the tow and while the net bag is being washed, which contrasts with those reported by Gaspar et al. (1999). During the selectivity experiments carried out on the south coast of Portugal, the authors estimated wider SR for *Spisula solida* and for the same mesh sizes investigated in the current study. During the experimental phase of this study, scuba divers observed that during the haul the mesh near the mouth of the dredge remain open favouring the escape of undersized clams through the net bag. Thus, the difference in the SR estimated for dredges used on the south and northern coasts of Portugal may be related to the dredge mouth length, hanging coefficient and/or length of the net bag. The influence of these factors on selectivity was also reported for trawls by Reeves et al. (1992).

From a management point of view, and taking into consideration the results obtained in the present study, it seems reasonable to implement a minimum mesh size of 40mm in the Portuguese northwestern *Spisula solida* fishery, which corresponds to an increase of 15mm in the current legal mesh size (25mm).

3.2.2.3 SELECTIVITY MECHANISMS IN SPRING-TOOTHED SCALLOP (*PECTEN MAXIMUS*) DREDGES

Introduction

This study follows on from Lart et al. (1997). The aim of both studies is to investigate selectivity mechanisms in sprung toothed (or Newhaven) scallop dredges with a view to improving selectivity of these dredges through technical measures.

The study focuses on factors that could potentially be regulated by technical measures. These are:

- Tooth spacing;
- Mesh size;
- Ring size of the chain mail that makes up the back and belly of the dredge.

The previous study, carried out in Scotland using FV Kelly (Lart et al. 1997), investigated the role of tooth spacing, mesh size and the chain mail ring size on the belly of the dredges. It was found that tooth spacing and belly ring size contributed to selectivity with some loss of marketable scallops during selection due to tooth spacing. Mesh size in the back of the dredges played no part in selection, however ring size in the dredge back was not tested. Also due to constraints inherent in the experimental design, tooth spacing was not compared directly but by paired hauls on subsequent days fishing.

It also recognised that factors such as ground type and the wear on the gear can affect selectivity, as well as variations in gear design. It was thus decided to carry out further cruises in the Irish Sea (FV *August Rose*) and Eastern English Channel (FV *Seafalke*) to investigate these aspects of selectivity mechanisms in scallop dredges and to elucidate whether there were similar selectivity mechanisms in these dredge fisheries.

Objectives

The results of three cruises are described in this Section. The first one (FV Kelly) which used similar methods to those described in detail here is fully described in Lart et al (1997). Some further analyses of the results of this cruise are presented here, for comparison with the results of the two later cruises. The objectives of the three cruises were as follows:

Kelly (West of Scotland)

To investigate the effect of tooth spacing, mesh size and belly ring size

August Rose (Irish Sea):

- 1 To investigate the effect of tooth spacing on selectivity.
- 2 To investigate the effect of wear on the bellies on selectivity.

Seafalke (Eastern English Channel):

- 1 To investigate the effects of:
 - Tooth spacing,
 - ring size of the dredge backs,
 - ring size of the dredge bellies on the selectivity of the dredges.
- 2 To further investigate the effect of wear on the bellies on dredge selectivity.

Materials and Apparatus***Scallop dredges***

Figure 1 shows a perspective drawing of a Newhaven scallop dredge. These dredges are deployed in gangs of up to 12 or more per bar (Figure 6) on each side of the vessel. The 4 main components are:

- 1 The dredge frame on which the springs and sprung tooth bar are mounted.
- 2 The belly of the dredge which is constructed of chain mail and is dragged along the sea floor.
- 3 The chain mail back of the dredge, which consists of chain mail forming a bag with the belly of the dredge.
- 4 The mesh which covers the forward part of the dredge. In the August Rose's dredges this was present, but in the Seafalke's dredges this part was absent with the chain mail back extending forward to the dredge frame.

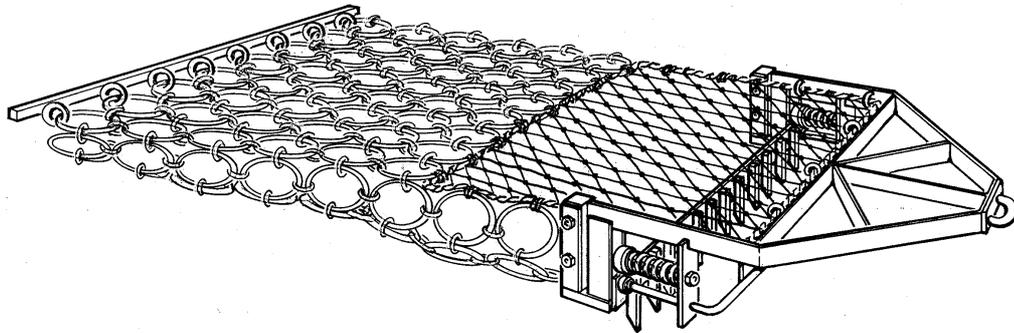


Figure 1 **Newhaven scallop dredge**

Table 1 Mean Longest Internal Dimension (MLID) and Standard Deviation (SD) of twenty belly rings selected at random from each of twelve dredge bellies from MFV August Rose.

Partly worn bellies (6)		Fully worn bellies(6)	
MLID	SD	MLID	SD
77.5	2.56	89.8	4.89
78.3	1.66	90.1	5.61
79.1	2.84	91.2	8.28
77.0	0.75	89.3	7.18
80.0	3.45	89.5	7.78
80.3	3.45	90.9	6.39

Table 2 Gear specification *FV Seafalke*

Component	Specification	Measured dimension: *
Dredge bars	8.1m with 8 x 76cm dredges attached (normal arrangement is 10 x 76cm dredges (see Fig 6).	
Dredge frames	Standard BEJAY 76cm	
Tooth bar	705mm hardened bar, teeth set behind bar rather than in front (Figure 2).	
Teeth	Hardened 'Peg teeth'. Teeth were 130mm x 20mm diameter.	Distance between teeth 10 Teeth/bar= 60mm 9 Teeth/bar=67mm 8 Teeth/bar=77mm
Bellies	Large: 8 rings long × 8 rings across plus 1 ring turnups Small: 9 rings long × 9 rings across plus 1 ring turnups	
Belly rings	Large: nominal internal diameter = 75mm, bar thickness=10mm Small: nominal internal diameter = 64.4mm, bar thickness = 10mm.	
Belly washers	Both large and small: external diameter=45mm, internal diameter=22mm, thickness=9mm	
Backs	Large backs 14 × 8 rings Small backs 15 × 9 rings	
Back rings	Large: = 72mm bar thickness = 8mm. (As vessels normal gear) Small: nominal internal diameter = 63mm, bar thickness = 8mm. Internal diameter of the forward row of backs adjusted to ensure the tensions remained equal; Small backs on small bellies=29mm Large backs on small bellies = 56 mm Small backs on large bellies = 40 mm	
Back washers	Both large and small: external diameter=38mm, internal diameter=18mm, thickness=7mm	

Tooth Bar Arrangement on the MFV Seafalke Dredges

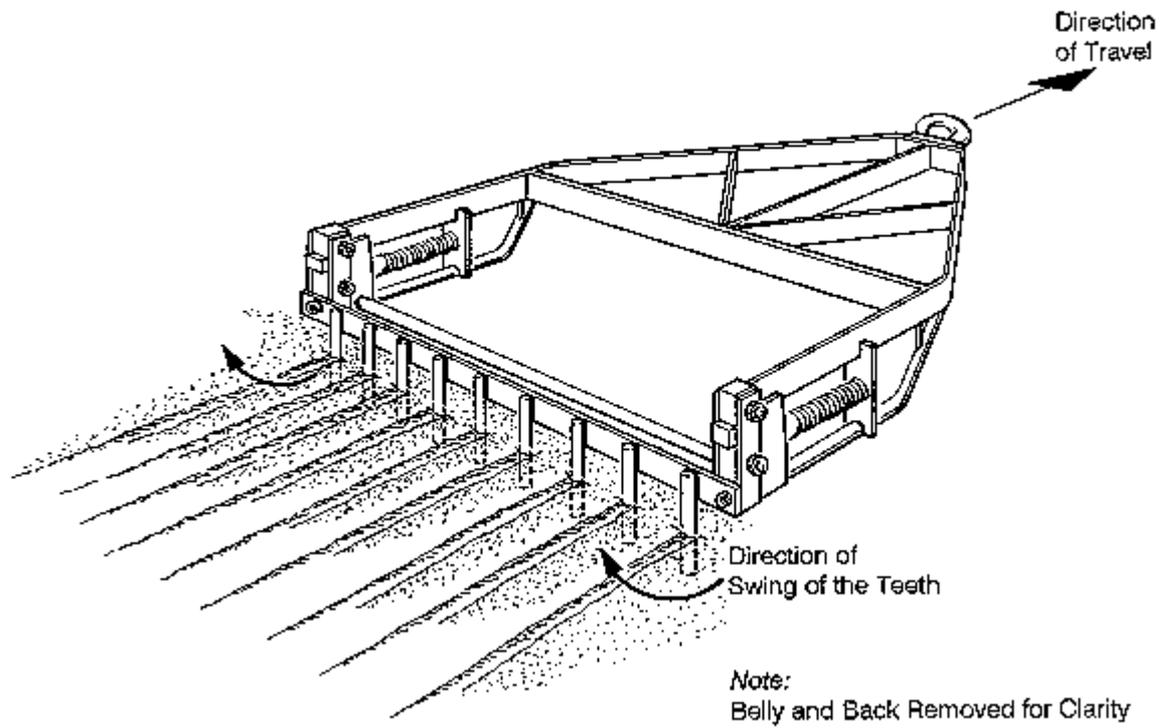


Figure 2 Tooth bar arrangement on *MV Seafalke's* dredges

Gear parameters

August Rose

For the August Rose trial, the experimental gear used was the vessel's own. The 12 experimental dredge bellies consisted of six bellies, which had been used, but they were considered to be at the start of their working life and six bellies which were towards the end of their working life. The mean longest internal dimension of the rings of these two sets of bellies are shown in Table 1 and the percentage difference between longest internal dimension and the original circular rings was 6.4% and 18.5% respectively. The control gear and tooth configurations were as described in Lart et al. (1997); they were standard Oban Scallop gear frames with French teeth set at 9/bar and 10/bar.

Seafalke

Bellies and backs

The gear specification of the Seafalke trial is described in Table 2. No mesh was used in this gear, the chain mail rings extending forward to the front of the dredge frame. The large rings were based on the vessels normal gear with 74mm internal diameter rings. The size of the small rings in both the backs and bellies was such that the bellies and backs were the same overall dimensions but with an extra row of rings in them for the small ringed backs and bellies. Where small backs and bellies were combined with large backs and bellies, care was taken to ensure that the lengths of the backs was congruent with there being the same tension in both combinations. Hence at the forward end of the backs a row of small rings was used, with its diameter dependent upon whether the back was being used with a large belly or a small belly.

Tooth bar Configuration

The tooth bar configuration used on the Seafalke trial is shown in Figure 2. This set up was different from that used on the Kelly (Lart et al. 1997) and the August Rose which used 'French teeth' in that 'peg' teeth were used and the tooth bar was bolted on the opposite way round with the teeth on the after (rear) side of the bar.

Data acquisition systems

Vessel and surface data

A number of parameters were recorded synchronously on the Delta-T data-logger. The acquisition scheme is shown in Figure 3. At the end of each tow the following data were downloaded onto a PC spreadsheet for later analysis:

- 1 ***Port and starboard towing loads:*** using strain-gauged loadcells connected in-line with the main towing wires (warps). These load cells were calibrated before and after the sea trip and found to be consistent. In the Seafalke trials Scanmar load shackles were used attached to the apex of the dredge bridle in place of the shackles, transmitting the tension sonically to the towed transducer.
- 2 ***Vessel speed through the water:*** using an impeller type log deployed via a telescopic towing boom off the starboard side of the vessel. Care was taken to

ensure that the impeller was not influenced by the vessel wake or other ship's noise.

- 3 ***Vessel speed over the ground:*** this parameter was logged autonomously by the GPS. Positions were logged every 30 seconds and at the end of each tow the total data was downloaded onto the PC. It was then analysed in a specifically designed spreadsheet which computed speed over the ground and total distance traversed over each towing period.
- 4 ***Haul parameters:*** the following parameters were recorded manually for each haul and the data are shown in Tables 6-8:
 - Location.
 - Time shot/hailed.
 - Wind and sea state.
 - Warp length and mean depth recorded at the during the tow.

Tooth bar spring tension

The tooth bar spring tension was estimated using an adapted torque wrench as Figure 3. Tensions were measured with the dredges hung freely below the dredge bar, above the rail of the vessel whilst in port. Torque was applied until the tooth bar just began to move relative to the frame. There is a small component of torque due to movement of the dredge itself. Thus, these measurements are not directly comparable with those taken with the dredge in a fixed position. However it was possible to reproduce the results and the technique corresponded to the Skipper's method of adjustment.

Scallop length

Scallop lengths were measured to the nearest 5mm below using the apparatus described in Figure 4. The results were recorded initially on the white plastic plate attached to the slider arm of this apparatus and then onto paper records for entry into spreadsheets.

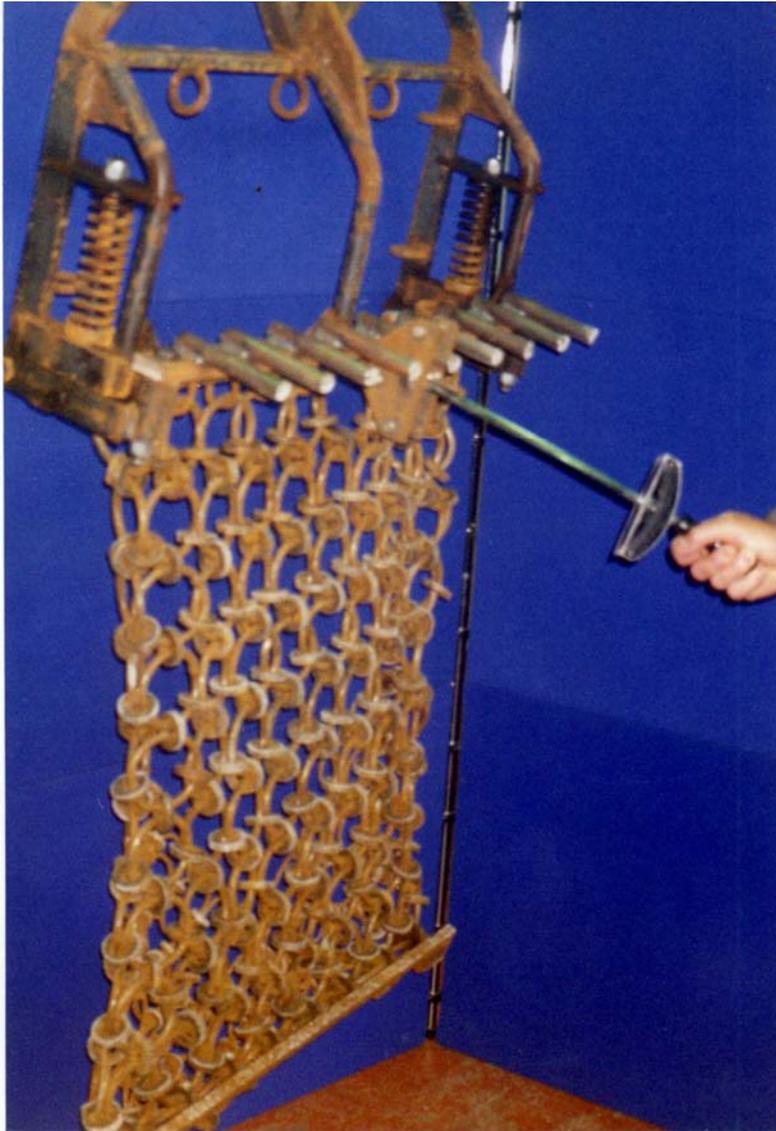


Figure 3 Converted torque wrench used for adjusting spring tensions. This device was fitted to the tooth and torque applied until the tooth bar just began to move relative to the frame. A reading was obtained on the wrench of the torque.



Figure 4 Apparatus designed for measuring scallops. The scallop was placed in the right angle of the apparatus and the slider pushed up against the scallop; the scallop length was read off the ruler.

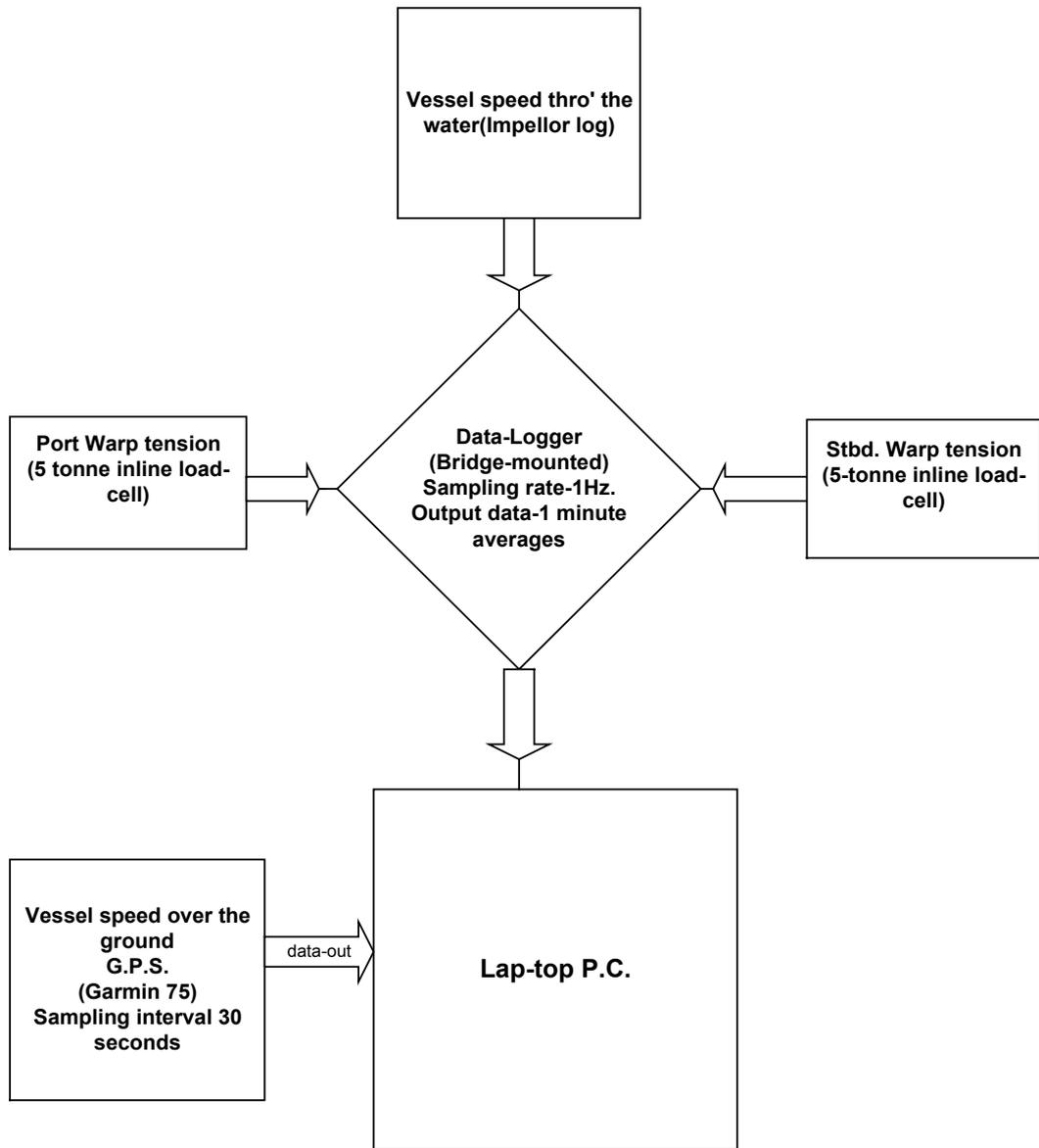


Figure 5 Vessel/surface data acquisition block diagram

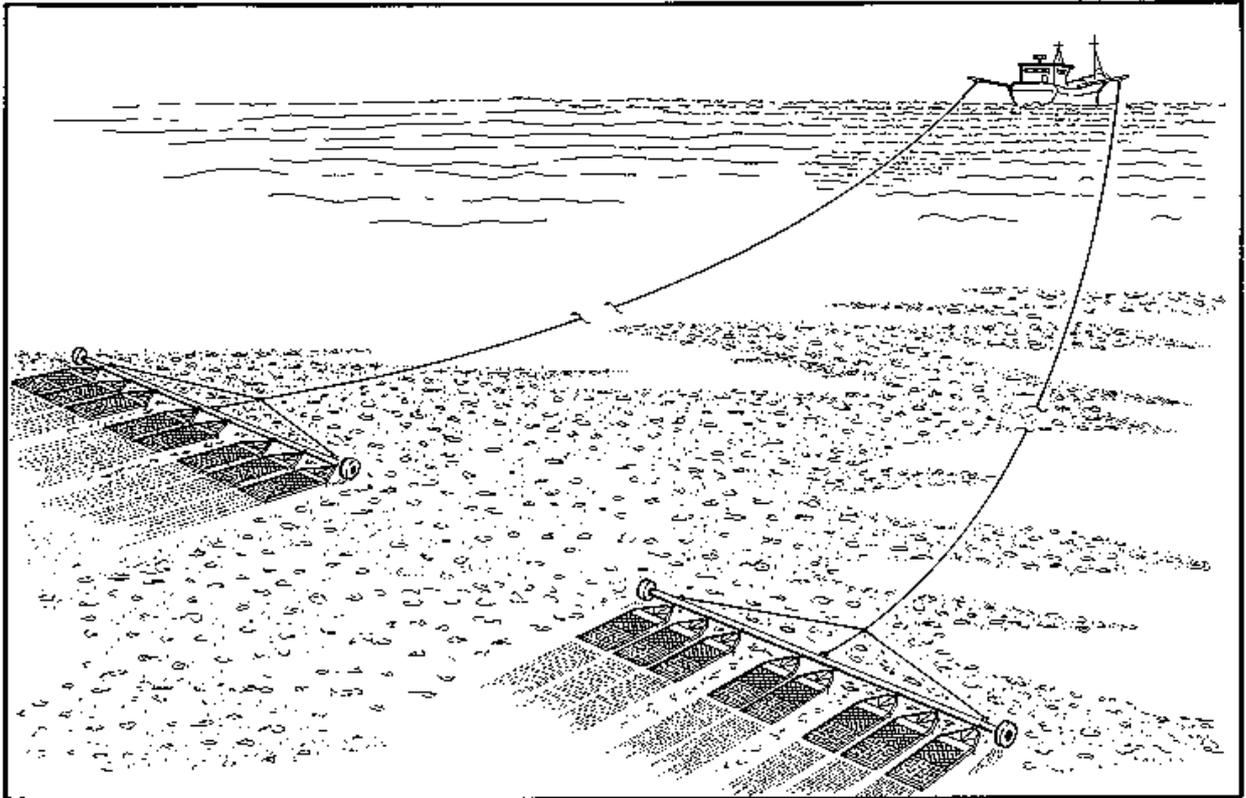


Figure 6 Towing arrangement on FVs August Rose and Seafalke

Method

Experimental design

The experimental design for these trials was based up Lart *et al* (1997). The intention was to design experiments to assess the contribution of the various factors; large and small rings in the backs and bellies, or tooth spacing, or old or new bellies. This had to be done within a reasonable time scale and in such a way as to eliminate unwanted variables such as those due to the position of the dredges on the bar or changes in the population of scallops. Achieving this would make it possible to attribute variation to catch variations with confidence to the variants being studied. The major sources of variation were considered to be:

- The way in which the scallop population is naturally distributed on the seabed and therefore available to the gear.
- The conditions under which each dredge configuration is deployed; its position on the bar, whether the bar is port or starboard.

Both of these were accommodated into the experimental design and in the way in which the data were analysed.

- Control dredges were used which exerted constant fishing effort, giving a reference against which the experimental dredges could be compared.
- Efforts were made to carry out the same number of hauls with each dredge configuration in each set of conditions. For analytical purposes, these conditions are termed block structures.
- The use of the statistical package, Genstat™ (Version 5.3.2). This enables incorporation of information from the controls as 'co-variates' and minimises variation between block structures. This improves the detection of significance in the factors investigated.

Gear configurations

Gear configurations are discussed at length in Lart *et al* (1997) (p14). For these two trials the design was simplified because neither of the two vessels operated the Lead/Lag (Short wire/Long wire) system of deployment which is described in the above report and [Section 3.2.2.4](#).

Dredge deployment

The dredges were deployed in two groups of 3 on each end of the bars (experimental dredges) and one group of 2 (control dredges) in the middle of each bar. Positions 4 and 7 were left vacant to avoid interference between the control and experimental dredges. See Figure 6.

Experimental dredges

August Rose

The experiment on the August Rose cruise was intended to investigate the effect of tooth spacing and worn bellies on selectivity. The dredges were deployed as shown in Table 3. In this, the degree of wear was varied by end of the bar and the tooth spacing

was varied by side. The teeth were switched side for side as described in Table 3 so that each day an equal number of hauls were fished using the two tooth bar spacing for each dredge. Thus, there was an equal quantity of effort from each experimental dredge with each tooth spacing.

The deployment of both the most worn experimental dredges on the inboard and the less worn ones on the outboard end was, with hindsight, a mistake; they should have been equally on the inboard and outboard ends. However, there were observations prior to this cruise that indicated no bias due to position on the bar.

Seafalke

This trial was in two parts, days one to 4 was designed to investigate the relative effects of tooth spacing back ring size and belly ring size on selectivity using all new gear. Days 5 and 6 investigated the effects of two levels of wear on the catches in the bellies and also two levels of tooth spacing (8 and 9 to the bar). The dredge deployment is shown in Tables 4 and 5.

In the first part of the trial (days one to 4) the bellies were varied by side and the backs varied by end of bar (Table 4). The teeth were changed at mid-day each day, since conditions were more likely to vary by day than by half day. Thus, comparisons could be better made between half days of fishing with each tooth spacing. Also, it meant that data was obtained from each tooth spacing from an equal number of morning and afternoon hauls, designated as Periods 1 and 2 in Table 4. Half way through the trial the gear was 'mirror imaged' so that the dredges on the inboard end were switched to outboard on the other side and *vice versa*.

For the final part of the experiment, days 5 and 6, very worn dredge bellies were compared with new (small and large) and half-worn bellies. These bellies' wear patterns were investigated in some detail in [Section 3.2.2.5](#). Also tooth spacing was compared, in this case 8 teeth were compared with 9 teeth to the bar, on opposite bars again switched by half day.

Control dredges

The control dredges were intended to monitor the likely catches of the least and most selective dredges. They were deployed in the middle two positions of each bar. These dredges consisted of the anticipated most and least selective combinations:

- large tooth spacing, large mesh (no mesh on the Seafalke trial), large back and large belly ring size.
- small tooth spacing, small mesh (no mesh on the Seafalke trial), small back and small belly ring size.

Each pair of control dredges was fished side by side shown in Tables 3, 4 and 5. Although this means that the two dredges in the pair may influence each other, the intention was, in most analysis, to combine the results of these control dredges. Apart from adjustments of the springs, and changing tooth bars to ensure that the amount of wear was the same as on the experimental dredges and necessary repairs, no alterations were made to these dredges.

Table 3 Experimental Design: August Rose

Port Side

Day	Hauls	Experimental			Control						Experimental		
		Outboard			Outboard			Inboard			Inboard		
		3 dredges			1 dredge			1 dredge			3 dredges		
		Teeth	Mesh	Belly	Teeth	Mesh	Belly	Teeth	Mesh	Belly	Teeth	Mesh	Belly
1	1-2	9	L	Part worn	9	L	L	10	S	S	9	L	Fully worn
1	3-5	10	L	Part worn	9	L	L	10	S	S	10	L	Fully worn
1	6	9	L	Part worn	9	L	L	10	S	S	9	L	Fully worn
2	7-8	10	L	Part worn	9	L	L	10	S	S	10	L	Fully worn
2	9-11	9	L	Part worn	9	L	L	10	S	S	9	L	Fully worn
2	12	10	L	Part worn	9	L	L	10	S	S	10	L	Fully worn

Starboard Side

Day	Hauls	Experimental			Stbd Control						Experimental		
		Inboard			Inboard			Outboard			Outboard		
		3 dredges			1 dredge			1 dredge			3 dredges		
		Teeth	Mesh	Belly	Teeth	Mesh	Belly	Teeth	Mesh	Belly	Teeth	Mesh	Belly
1	1-2	10	L	Fully worn	9	L	L	10	S	S	10	L	Part worn
1	3-5	9	L	Fully worn	9	L	L	10	S	S	9	L	Part worn
1	6	10	L	Fully worn	9	L	L	10	S	S	10	L	Part worn
2	7-8	9	L	Fully worn	9	L	L	10	S	S	9	L	Part worn
2	9-11	10	L	Fully worn	9	L	L	10	S	S	10	L	Part worn
2	12	9	L	Fully worn	9	L	L	10	S	S	9	L	Part worn

Key for tables 4-1 tp 4.3 (see Table 3.1 for sizes)

Rings

L – Large

S – Small

Tooth Spacing

10 = 10 teeth/bar

9 = 9 teeth/bar

8 = 8 teeth/bar

Table 4 Experimental Design: Seafalke Days 1-4

Period	Day	AM	Hauls	Port Side												Starboard Side														
				Experimental						Control						Experimental			Control			Experimental								
				or			OB			OB			IB			IB			IB			IB			OB			OB		
				PM			3 dredges			1 dredge			1 dredge			3 dredges			3 dredges			1 dredge			1 dredge			3 dredges		
			Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly				
1	1	AM	1-4	10	L	L	10	S	S	9	L	L	10	S	L	10	L	S	10	S	S	9	L	L	10	S	S			
2		PM	5-8	9	L	L	10	S	S	9	L	L	9	S	L	9	L	S	10	S	S	9	L	L	9	S	S			
2	2	AM	9-12	9	L	L	10	S	S	9	L	L	9	S	L	9	L	S	10	S	S	9	L	L	9	S	S			
1		PM	13-16	10	L	L	10	S	S	9	L	L	10	S	L	10	L	S	10	S	S	9	L	L	10	S	S			
1	3	AM	17-19	10	L	S	10	S	S	9	L	L	10	S	S	10	L	L	10	S	S	9	L	L	10	S	L			
2		PM	20-22	9	L	S	10	S	S	9	L	L	9	S	S	9	L	L	10	S	S	9	L	L	9	S	L			
2	4	AM	23-25	9	L	S	10	S	S	9	L	L	9	S	S	9	L	L	10	S	S	9	L	L	9	S	L			
1		PM	26-28	10	L	S	10	S	S	9	L	L	10	S	S	10	L	L	10	S	S	9	L	L	10	S	L			

Table 5 Experimental Design ; Seafalke days 5 and 6

Period	Day	AM	Hauls	Port Side												Starboard Side														
				Experimental						Control						Experimental			Control			Experimental								
				or			OB			OB			IB			IB			IB			IB			OB			OB		
				PM			3 dredges			1 dredge			1 dredge			3 dredges			3 dredges			1 dredge			1 dredge			3 dredges		
			Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly	Teeth	Back	Belly				
	5	AM	29-31	8	L	S	10	S	S	9	L	L	8	Very Worn	9	New	10	S	S	9	L	L	9	Very Worn						
3		PM	32-34	9	L	S	10	S	S	9	L	L	9	Very Worn	8	New	10	S	S	9	L	L	8	Very Worn						
3	6	AM	35-38	8	L	S	10	S	S	9	L	L	8	Half Worn	9	New	10	S	S	9	L	L	9	Half Worn						
		PM	39-42	9	L	S	10	S	S	9	L	L	9	Half Worn	8	New	10	S	S	9	L	L	8	Half Worn						

Note; No mesh present on these dredges

Day and haul routine

The spatial pattern of fishing was as near identical as possible, without actually fishing down the same track, each day for the August Rose, and each half day for the Seafalke. The daily routine consisted of 6 or 8 50 minute hauls, starting one hour later each day where feasible to keep tidal conditions as constant as possible.

At the end of each day (August Rose), half day (Seafalke) the spring tensions were checked and re-adjusted to the level favoured by the fishermen using the adapted torque wrench described in Figure 3)

The levels to which the spring tensions were set were determined in consultation with the fishermen:

- For the August Rose they were set at 7.0kgfm
- For the Seafalke they were set at 2.5 kgfm.

These contrasting levels were set for the grounds on which the fishing was carried out based on the experience of fishermen.

Locations fished

The locations fished are shown in Figures 7 and 8. These were considered to be typical of the two locations.

Catch monitoring

Each dredge was then tipped and the scallop catches placed in a basket. All the scallops (*Pecten maximus*) from each dredge were measured using the apparatus described in Figure 4; there was no requirement to sub sample.

Haul parameters

Tables 6, 7 and 8 show the haul parameters for all the hauls recorded in these experiments.

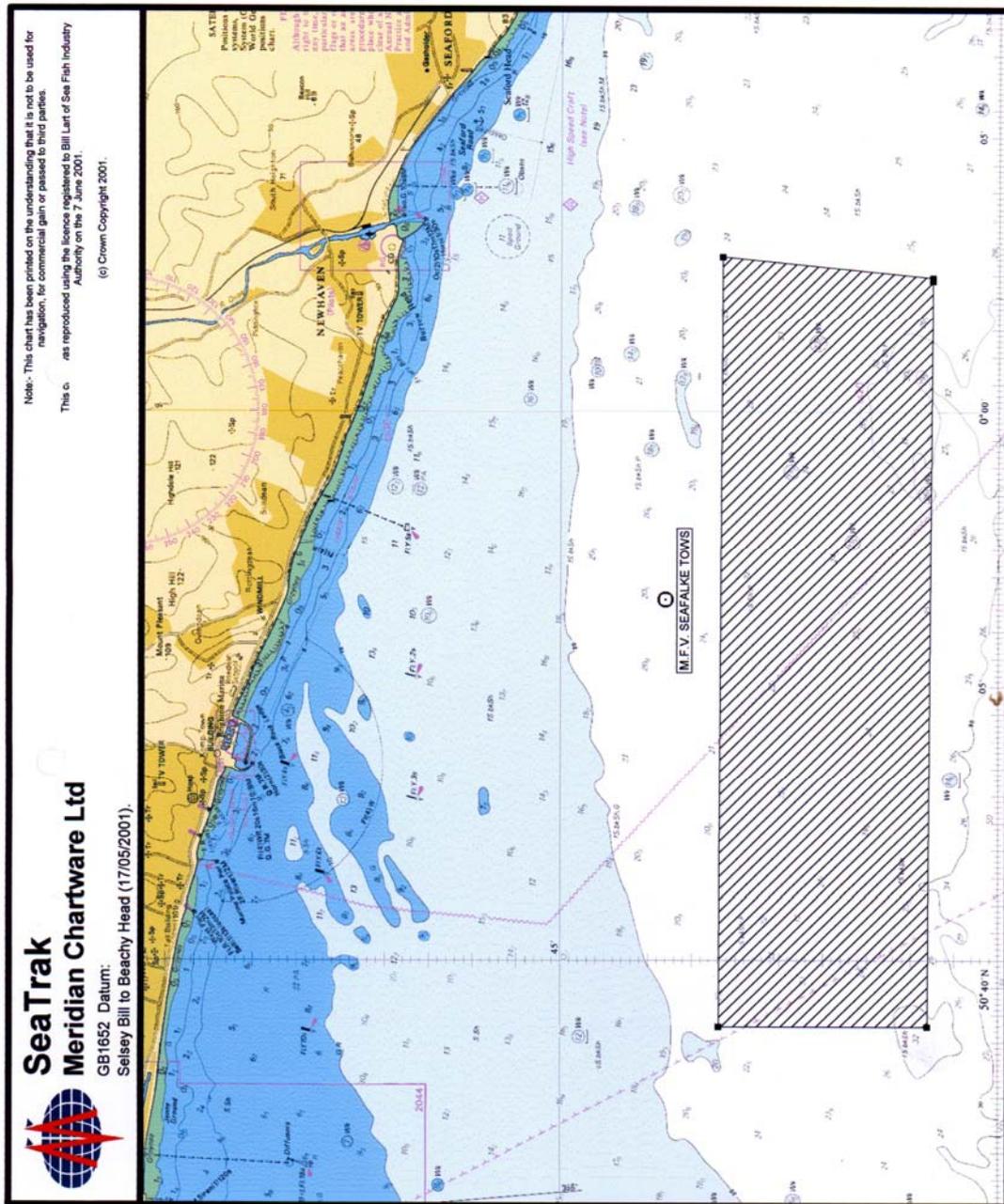


Figure 8 Fishing locations: Seafalke Mar 2000

Table 6 Log of Haul Parameters: August Rose Irish Sea 5/12-6/12/98

Day No.	Haul No.	Time		Wind state	Sea state	Warp-out Starboard (metres)	Depth (metres)	Mean speed over ground (knots)	Distance covered* n-m	Mean speed thro' water (knots)	Mean total tension (Tonnes)
		Shot	Hauled								
1	1	05:10	06:00	4-5	5	138	37		1.79		
1	2	06:40	07:30	5	5	138	37		1.86		
1	3	09:10	10:00	5-6	6	138	38		2.26		
1	4	10:15	11:10	5-6	6	138	40		1.44		
1	5	11:23	12:15	5	5	138	34		1.99		
1	6	12:40	13:30	4-5	4-5	138	45		2.1		
2	7	08:55	09:40	0-1	0-1	138	35	2.7	2.33	4.2	2.9
2	8	09:50	10:40	0-1	0-1	138	38		1.93		
2	9	11:25	12:20	0-1	0-1	138	44		2.52		
2	10	12:25	13:25	0-1	0-1	138	40		2.21		
2	11	13:45	14:30	0-1	0-1	138	35	4.4	1.96	4.4	2.6
2	12	15:00	15:45	0-1	0-1	138	40		2.17		

Total distance covered over the ground: Day 1 = 11.44 n-m Day 2 = 13.12 n-m

Table 7 Log of Haul Parameters: Seafalke, Eastern Channel

Day No.	Haul No.	Time		Wind state	Sea state	Warp-out (metres)	Depth (metres)	Mean speed over ground (knots)	Mean speed thro' water (knots)	Mean total tension (Tonnes)
		Shot	Hauled							
1	1	07:45	08:35	SW 2-3	1-2	82	27	3.0	3.2	3.2
1	2	09:07	09:57	SW 2-3	1-2	93	31	4.0	2.6	3.6
1	3	10:38	11:28	SW 2-3	1-2	106	34	3.4	4.7	3.5
1	4	11:45	12:35	SW 3	2-3	113	36	2.8	4.7	3.5
1	5	14:35	15:25	SW 3	1-2	100	34	3.5	3.5	3.5
1	6	15:55	16:45	SW 3	1-2	104	35	2.8	3.7	3.2
1	7	17:15	18:05	SW 2-3	1	93	32	3.4	4.0	3.3
1	8	18:22	19:12	SW 2	0	106	33	3.7	2.6	3.2
2	9	08:16	09:06	SW 3	2	106	27	3.1	3.1	3.3
2	10	09:35	10:25	SW 3	2	91	31	4.0	2.8	3.2
2	11	10:42	11:32	SW 3-4	2	99	33	3.4	2.9	3.3
2	12	12:15	13:05	SW 4	3	101	35	2.8	4.5	3.8
2	13	15:10	16:00	SW 3-4	3	92	32	3.1	3.6	4.6
2	14	16:25	17:15	SW 3	3	101	34	3.5	3.5	3.1
2	15	17:40	18:30	SW 3	2	104	35	3.9	3.5	3.1
2	16	18:48	19:38	SW 3	2	99	32	3.0	3.4	3.2
3	17	07:04	07:54	WSW 3	1	88	29	3.0	3.4	3.3
3	18	08:45	09:35	WSW 3	1	88	29	3.1	3.4	3.4
3	19	10:53	11:43	WSW 3	1	88	29	3.4	2.9	3.6
3	20	13:02	15:45	W 2	0	88	30	3.2	3.5	3.4
3	21	14:38	15:28	W 2	0	91	32	3.9	3.0	3.5
3	22	15:50	16:40	W 2	0	93	31	3.7	3.2	4.2
4	23	07:40	08:30	N 3	0-1	91	29	3.4	3.1	
4	24	09:16	10:07	N 4	2	82	27	3.0	4.0	
4	25	10:26	11:16	N 5	3	91	29	3.3	3.1	
4	26	12:58	13:48	N 4	3	91	29	3.4	3.2	
4	27	14:21	15:11	N 3	2	91	29	3.0	2.1	
4	28	15:40	16:30	N 3	2	91	29	3.2	3.1	

Table 8 Log of Haul Parameters: Seafalke Eastern Channel (Continued)

Day No.	Haul No.	Time		Wind state	Sea state	Warp-out (metres)	Depth (metres)	Mean speed over ground (knots)	Mean speed thro' water (knots)	General comments (ground type etc.)
		Shot	Hauled							
5	29	06:58	07:48	N 5	3	87.8	30	3.0	3.4	
5	30	08:06	08:56	N 5	3	87.8	30	3.25	3.4	
5	31	09:13	10:03	N 4	2	87.8	26	3.5	3.7	
5	32	10:45	11:35	N 4	2	80	25	3.6	3.6	
5	33	11:50	12:40	N 4-5	3-4	82.3	27	2.9	3.3	
5	34	12:55	13:47	N 3	2	83	27	3.6		
6	35	06:25	07:15	N 3	0	83	27	3.2		
6	36	07:35	08:25	N 4	3	83	29	3.6		
6	37	08:40	09:30	N 4	2	83	27	3.5		
6	38	09:45	10:35	N 4	2	87	29	3.1		
6	39	11:15	12:05	N 3	2	80	26	3.5		
6	40	12:20	13:10	N 3	2	82	26	3.5		
6	41	13:25	14:15	N 3	2	82	26	3.7		
6	42	14:34	15:24	N 3	2	82	26	3.0		

Results

Environment and physical parameters

The weather conditions and physical parameters (where measured) are described in Tables 6,7 and 8. With the exception of the first day of the August Rose trial, when sea state was either moderate (Beaufort 4) to fresh (Beaufort 5) sea conditions were either calm (Beaufort 0-1) or light (Beaufort 2-3) throughout these trials. Speeds through the water and over the ground (where measured) and total warp tension were comparable. The main differences between the two trials are therefore:

- i The by catch from, the area dredged off the Isle of Man by August Rose contained very few stones, substantial quantities of benthos were captured and the seabed was relatively flat. In contrast, the by catches from the Seafalke trials in the Eastern Channel consisted of chalky rocks and very little benthos.
- ii The design of the gear, mesh being present on the August Rose's dredges, which used French teeth [Section 3.1.1.1](#) bars bolted on in the conventional fashion. In contrast, in the Seafalke trial, the mesh was absent with the 'peg' teeth set on the tooth bolted (Figure 2) on the opposite way round from the August Rose. Also, there was around 1/3 the tension in the tooth bar springs on the Seafalke trial. These measures (reversing the tooth bar use of 'peg' teeth) are intended to reduce the catch of stones in the gear.

The physical conditions on the FV Kelly trials in Western Scotland (Lart et al. 1997) were different. In these trials the skipper set the tooth bars up at 7.5 kgfm (similar to the level set on the August Rose, but very much more than the level set on the Seafalke) and the speed over the ground was less. Warp tensions were less at an average of 2.5 tonnes; this would be expected due to the reduced number of dredges.

Variation in selectivity and catch per effort

This is divided into variation due to environmental factors, experimental design and the factors of interest; tooth spacing, ring size in the backs and bellies and wear on the gear. The design of the experiment was intended to reduce to a minimum the effect of environmental factors and to avoid unwanted variation due to a balanced experimental design.

These data were analysed to investigate;

1. Variation in the results due to environmental factors or variations in the scallop populations.
2. Variation due to the position of the experimental dredges (inboard and outboard) on the bar and between sides (port and starboard).
3. Variation due to the factors being investigated for their contribution to scallop dredge selectivity; tooth spacing, ring size in the chain backs and bellies, wear on the bellies.

To do this, the following analysis was carried out:

- 1) Analysis of variance (ANOVA) using the treatment structures and interaction terms described in Table 9 and 10 in the following variates:
 - a) The mean length of scallops, this is intended to detect differences in selectivity.
 - b) The mean total number of scallops per haul. If there is significant selectivity then this would be expected to decrease. However if there is no significant selectivity but a significant decrease in the number of scallops per dredge per haul then this suggests a reduction in overall efficiency.
 - c) In order to pinpoint the selectivity range of the factors identified as being the catch per haul of the length groups was compared by ANOVA. This was done cumulatively, starting with the longest length range of 150mm, then 150 + 145 ... down to 60mm scallops. Where there was significantly less catch in the higher level of the factor investigated was considered to approximate the size selection range.
- 2) Aggregate Length-frequency distributions in a selection of treatments (for results from the West of Scotland cruise on the FV Kelly see Lart et al. (1997)). These are shown in Figure 9a calm and rough weather; Figure 10 tooth spacing; Figure 11 backs and bellies and 12 worn bellies.

The percentage discard rate:

$$\% \text{Discard} = \frac{\text{Scallops below MLLS}^4 \text{ (Number)}}{\text{Total Catch (Number)}} \times 100\%$$

is also shown in these Figures. However since the MLLS varies between this study and Lart et al. (1997) the significance of differences between discard rates were not tested.

- 3) Ratios of large/small against length. These graphs describe the relative selectivity of the gear in relation to the length of the scallop. Figure 13 shows the results for tooth spacing, Figure 14 for bellies. The significance levels for 1c above are shown on these Figures.

Variation inherent in the design

The structure of the experimental design of the Seafalke trial enables variation due to side and inboard and outboard position on the dredges to be investigated. No significant difference could be detected in mean length of scallop or for catch per effort for either of these factors (Table 9). For the August Rose trial, due to an error, in the design the factor inboard-outboard could not be examined. However, a balanced design had been undertaken during the previous 4 days (unpublished data)

⁴ MLLS= Minimum Legal Landing Size

on this vessel and no significant difference between the inboard and outboard locations was detected.

Environmental and experimental factors

The scallop populations available for capture and the weather and ground conditions are the most important environmental factors. The control dredges give an indication of changing populations and the results from the control dredges also show the variation in catch rates due to weather conditions if similar populations are assumed.

Irish Sea

During the two days of the August Rose trial there was a significant reduction in catch per haul in the controls in the rough sea conditions, there was also a small increase in the mean length of the scallops captured in the rough conditions, although this was just outside normally accepted significance level at $p=0.056$ (Table 9). Figure 9a compares the Length Frequency distributions of the controls for these two days.

This indicates a major reduction in catch per effort and possibly a slight improvement in selectivity in rough sea conditions. This effect could be due to the loads transmitted down from the vessel affecting the action of the dredge on the seabed or due to the vessel not covering the same distance over the ground. The former is more likely in this case since the distance covered over the ground was greater for day 1 the rough day than day 2 the calm day (Table 6).

East Channel

The control catches were compared between periods when the 10/bar tooth (Period 1) and 9/bar (Period 2) tooth bars were used. It is important to compare controls, since these two levels of the tooth spacing factor were be tested between these two periods. Thus if there was a difference in the populations of scallops available to the gear, as indicated by the catches in the control results periods, this would make this comparison difficult.

Table 9 shows that there is a difference and the mean length of the scallops in the control gear from the two periods but no significant difference in catch per dredge per haul. The length-frequency distributions of the controls were also dissimilar Figure 9b. These results suggest that it could be difficult to compare tooth spacing on this basis. It may be possible to use the control results as a covariate.

Experimental treatments

Tooth spacing

Two types of teeth were tested in these trials ‘French’ or pointed teeth (see Lart et al. 1997 and [Section 3.1.1.1](#) for details) as used on the Kelly (West of Scotland) and (Irish Sea) and ‘peg’ or rod shaped teeth as used on the Seafalke (East Channel). The Kelly and August Rose trials compared 10 and 9 teeth/bar on successive days and on opposite sides of the vessel respectively. On the Seafalke trial 10 and 9 teeth to the bar were compared on successive days and 8 and 9 teeth to the bar were compared on opposite sides of the vessel.

West of Scotland

Selectivity is evident, as revealed by the significant increase in the mean length of scallops and reduction in discards (Lart et al. 1997) in the 9 teeth /bar when compared with 10/bar. However as in the East Channel results these are from hauls paired in successive periods (see below). There is also significant loss of scallops up to 125mm.

Irish Sea

In these trials the two levels of tooth spacing were fished in parallel on opposite sides of the vessel. There is evidence of increased selectivity in the 9teeth/bar results (Figure 10a) and that the upper end of the selection range is around 95-100mm because there is no significant difference between the catch/haul of the 9 and 10/bar tooth spacing above that size (Figure 12b).

East Channel

Two comparisons were made;

- a. Between 9 and 10 teeth/bar made during the first 4 days of the trial periods 1 & 2.
- b. Between 8 and 9 teeth/bar made in the final two days of the trial.

Since the comparison between tooth spacing of 9 and 10 to the bar was carried out during successive periods, the intention was to use the controls, which were fished consistently through the experiment to enable a link between the two periods. The results from the controls indicate that different length distributions of scallops were caught in the controls during the two periods (see above). Analysis of variance (Table 10) comparing the mean lengths of scallops captured in 9 and 10 teeth/bar, carried out with and without covariates from the control does not clarify the results. However, there is no suggestion of selectivity in the length frequency (Figure 10a) results from the tooth spacing of 9 and 10/bar.

For the 8 and 9 teeth/bar there is direct comparison between the two sides of the vessel thus the use of the control as a covariate is not required. There is no suggestion of increased selectivity in the comparative length-frequency distributions (Figure 10c) between 8 and 9/bar and the analysis of variance (Table 10) indicates no significant difference in the mean length, or catch per haul.

Backs and bellies

Backs

In the West of Scotland trials (Lart *et al.*, 1997) the ring sizes of the backs were matched closely to bellies thus the examination of this factor is not feasible. In the Seafalke trials the experiment was designed so that the effect of backs could be examined in combination with belly ring size. Figure 11c and Table 10 shows no evidence of increased selectivity in the larger backs and there was a significant reduction in catch per haul.

Bellies

Although there is no significant difference in the mean length of the catches in the two sizes of bellies in the Seafalke (East Channel) trial (Table 10), examination of the ratio large/small (Figure 14) shows a reduction in the proportion of scallops below 85mm length in the larger ring (75mm id) ring size. Analysis of variance showed that there was a significant (at $p < 0.05$) reduction in the catch per haul in the larger (75mm) ring suggesting that this (85mm) is the upper end of the selection range. This compares with the results from the Kelly (West of Scotland) trial in which there is a significant reduction in the catch of scallops below 90mm length in the larger (74mm id) ring size (Figure 14). Given the differences in vessel, manufacturer, gear, ground type and location this is good correspondence between these results.

Effect of wear

In both the Irish Sea and East Channel results there is evidence of increased selectivity with increasing wear (Figure 12). There is evidence for loss of scallops larger than the MLLS in the East Channel results. However, these losses are in the range 110-115mm which is at the smaller end of the marketable size range. A proportion of these are likely to be discarded in the normal course of events (unpublished discard studies) in order to ensure compliance with the 110mm MLLS; thus they might not be perceived to be losing significant quantities of marketable scallops.

Table 9 Environmental and experimental treatments

Treatment	Mean Scallop Length		Significance	Interactions and notes	Mean Total Catch per haul		Significance	Interactions and notes
	Rough	Calm			Rough	Calm		
August Rose, Irish Sea	Calm vis rough	103	102	p=0.056	35	51	p>0.001	
Seafalke, South coast	Period	Period 1	Period 2		Period 1	Period 2		
		112	113	p=0.010	26	30	Not significant	
	Side	Port	Starboard		Port	Starboard		
		112	112	Not significant	95	95	Not significant	
	Inboard vis outboard	Inboard	Outboard		Inboard	Outboard		
		112	112	Not significant	I=95.3	O=95.3	Not significant	

Table 10 Experimental treatments

Vessel and Location	Treatment	Mean Scallop Length		Significance	Interactions and notes	Mean Total Catch per haul		Significance	Interactions and notes
		Large	Small			Large	Small		
Kelly, West of scotland	Tooth spacing 10 vis 9/bar	105	104	p=0.006	No significant	61	77	p<0.001	No significant
	Mesh size	104	104	Not significant	Interactions	67	71	Not significant	Interactions
	*Belly & back ring size	106	103	p=0.001	Tooth vis belly	64	74	p=0.008	Tooth vis belly
August Rose, Irish Sea	Tooth spacing 10 vis 9/bar	107	105	p<0.001		35	41	p<0.006	
	Old vis New bellies	105	107	p<0.001		New= 38.4	O= 37.1	Not significant	
Seafalke, South coast	Tooth spacing								
	10 vis 9/bar	111	112	p=0.03	Covariate=Mean length of control	90	101	p=0.041	Covariate=Numbers in control
	10 vis 9/bar	112	112	Not significant	No Covariate	98	92	Not significant	No Covariate
	8 vis 9/bar	105	105	Not significant		48	46	Not significant	
	Back ring size	112	112	Not significant		87	104	P=0.034	No significant interactions
	Belly ring size	112	112	Not significant		90	101	Not significant	Backs vis bellies
	New vis	104		Not significant		59			
	Half worn bellies	105				41		p>0.001	
Full worn bellies	108		p<0.001		27				

*Belly and back both the same in this experiment

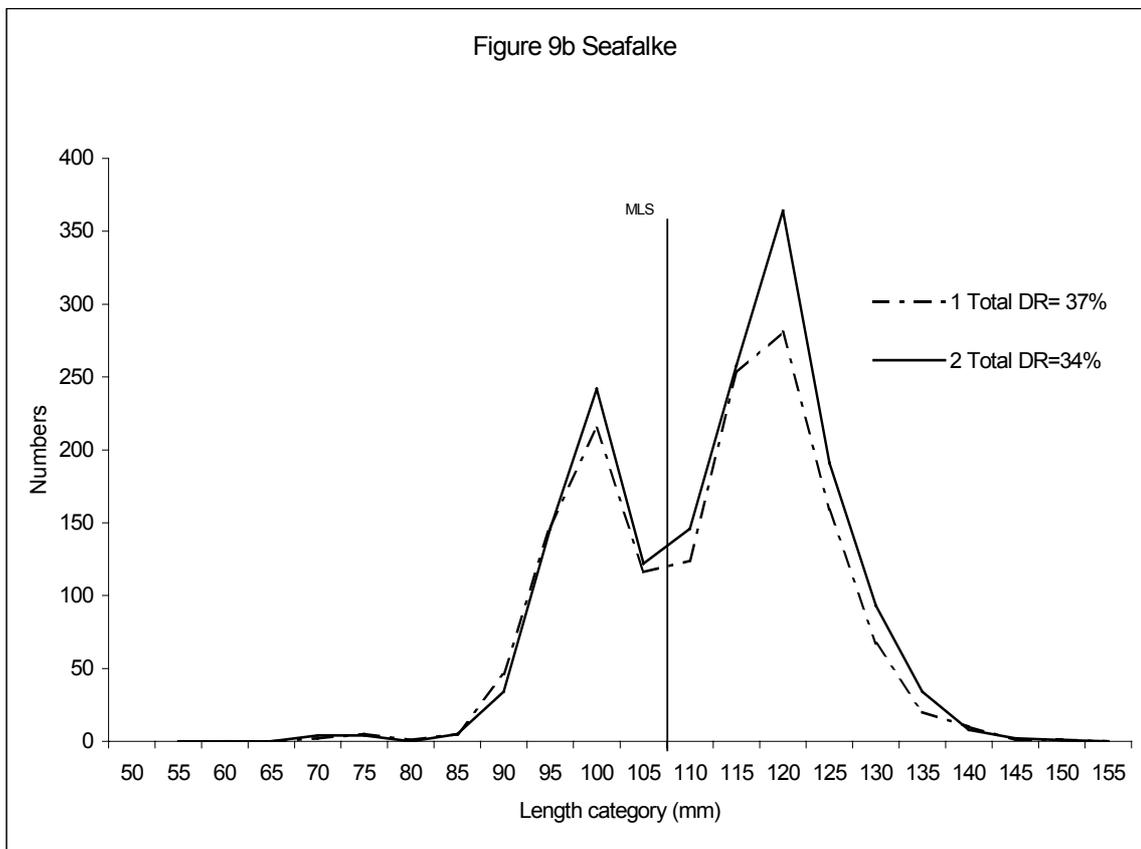
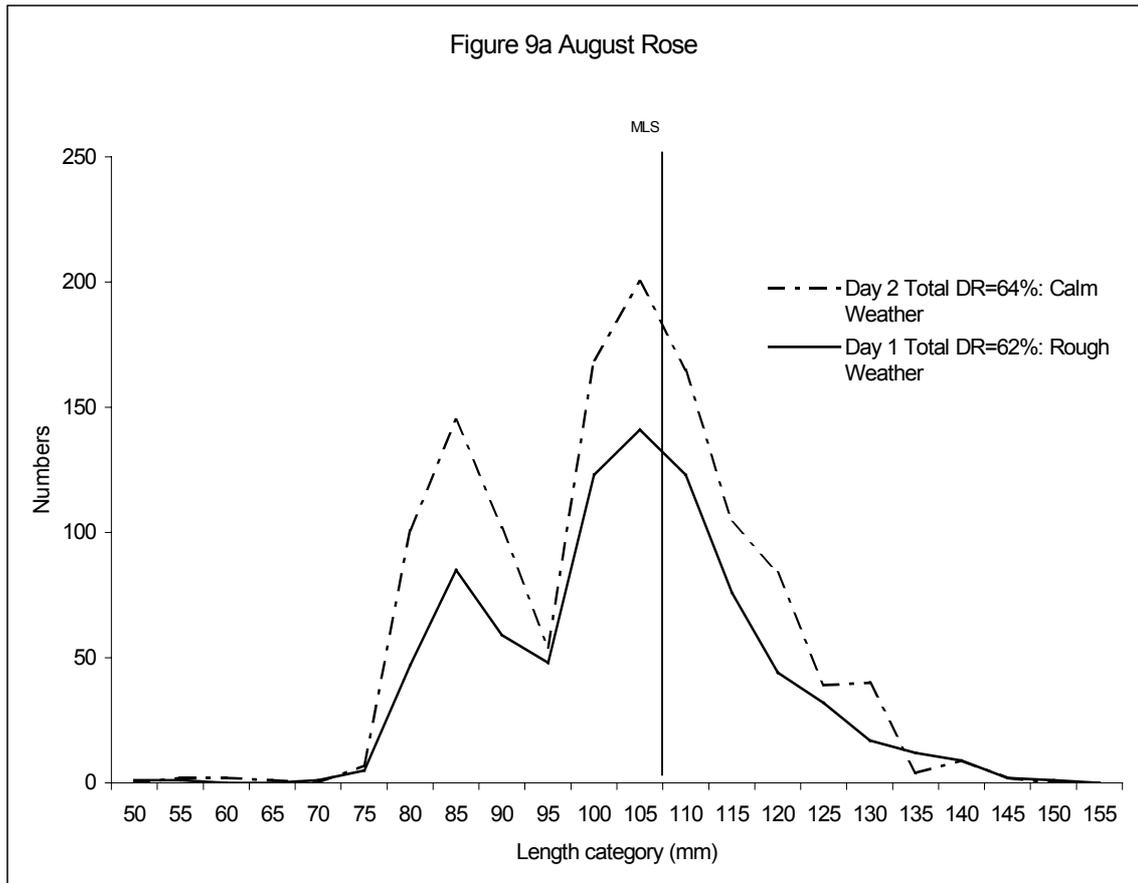


Figure 9 Aggregate Length-Frequency distributions of controls

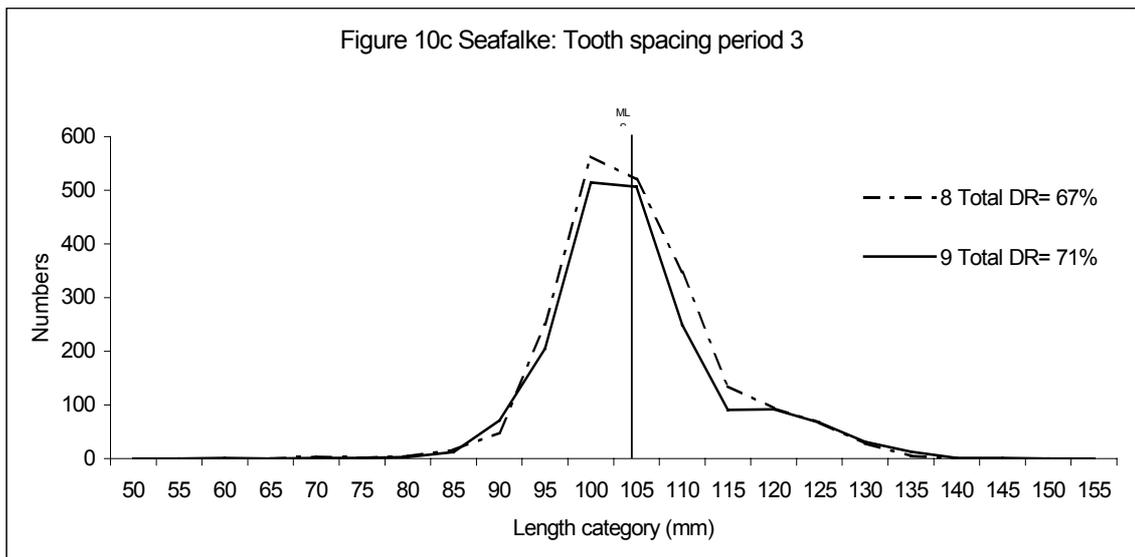
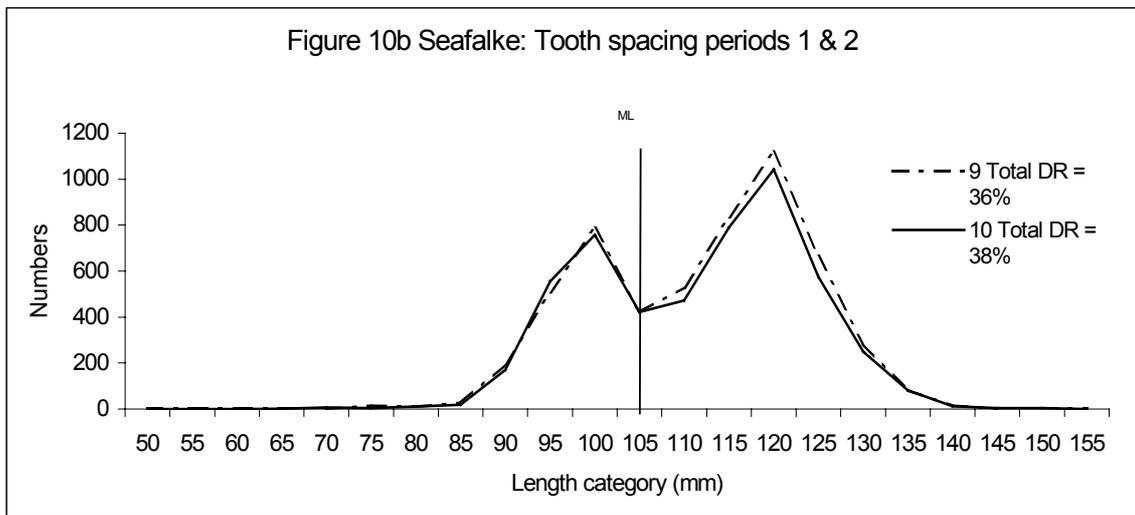
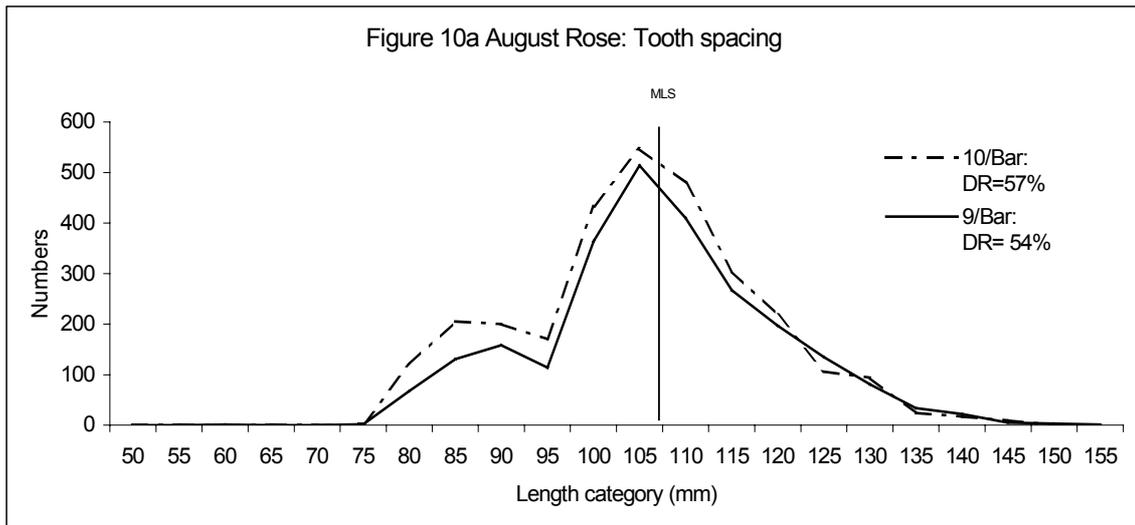


Figure 10 Aggregate Length-Frequency distributions for tooth spacing

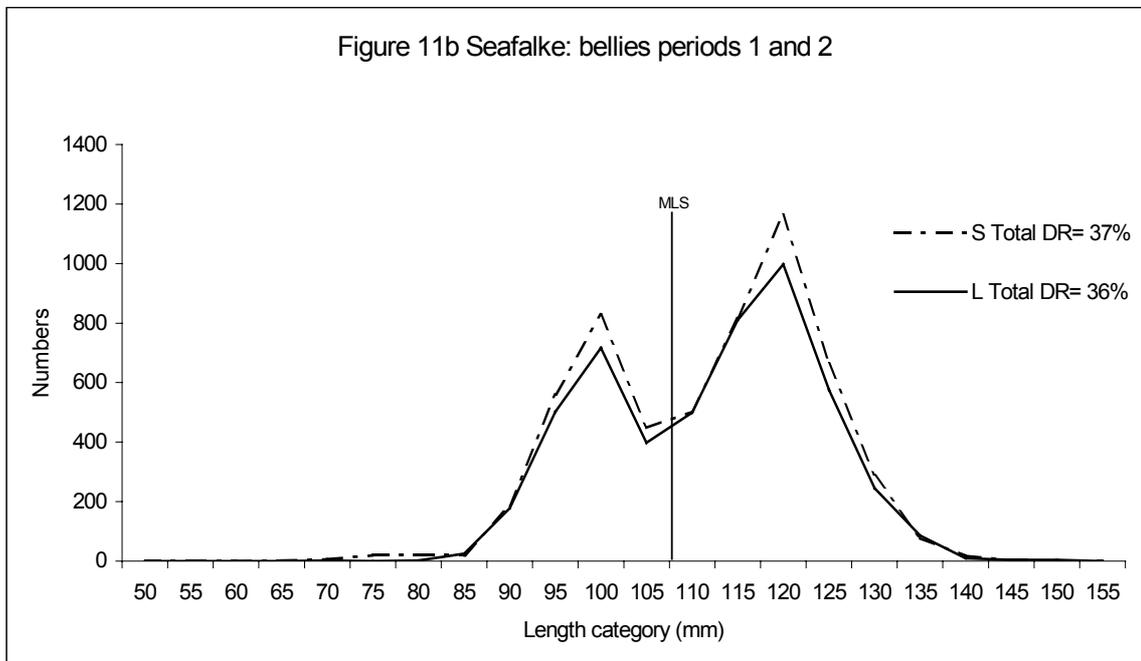
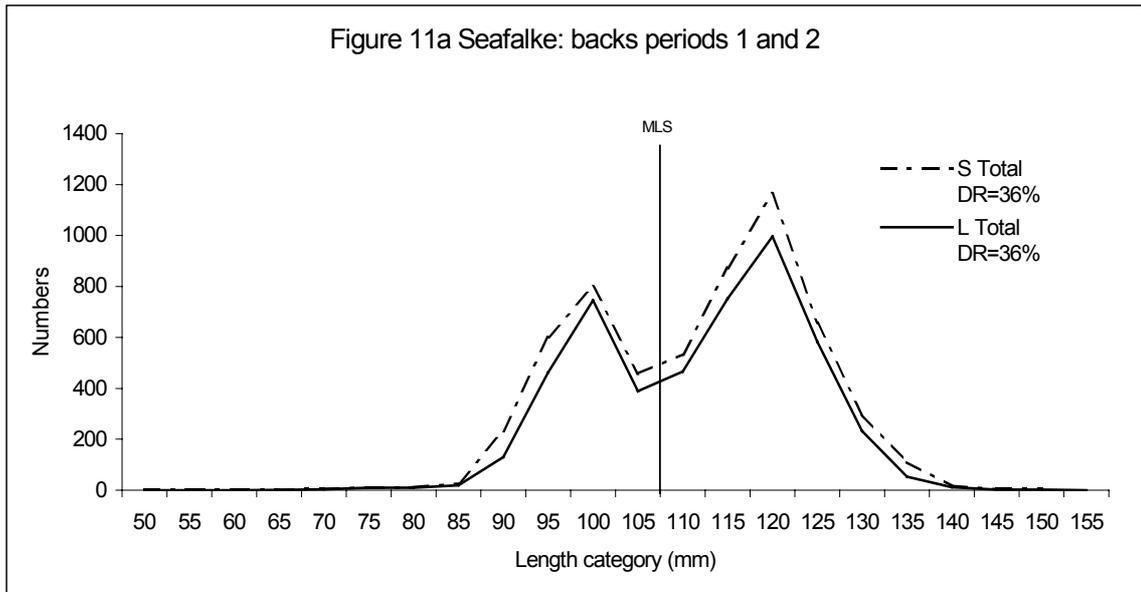


Figure 11 Aggregate Length-Frequency distributions for backs and bellies

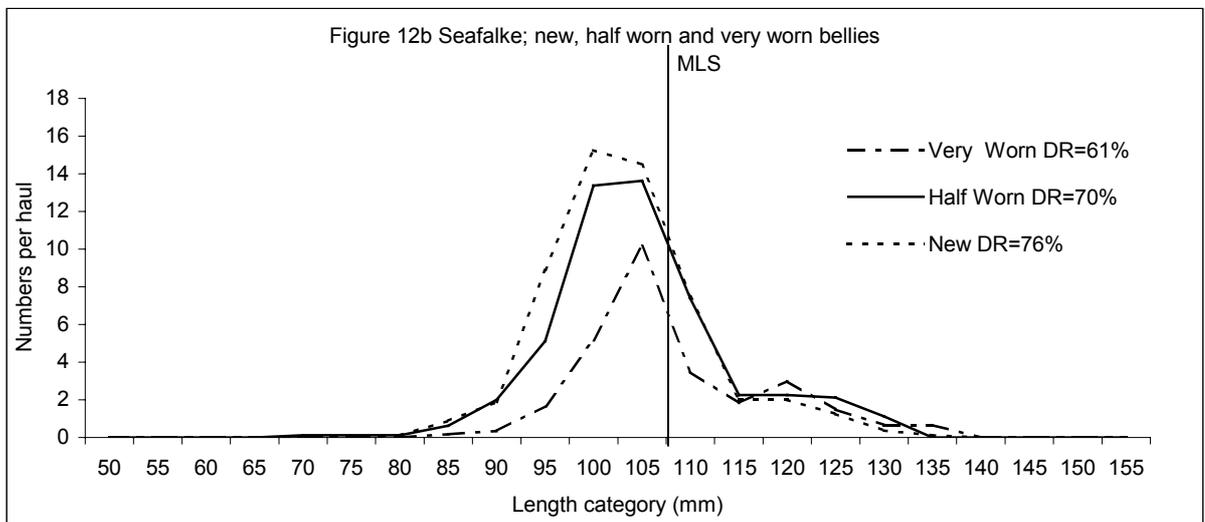
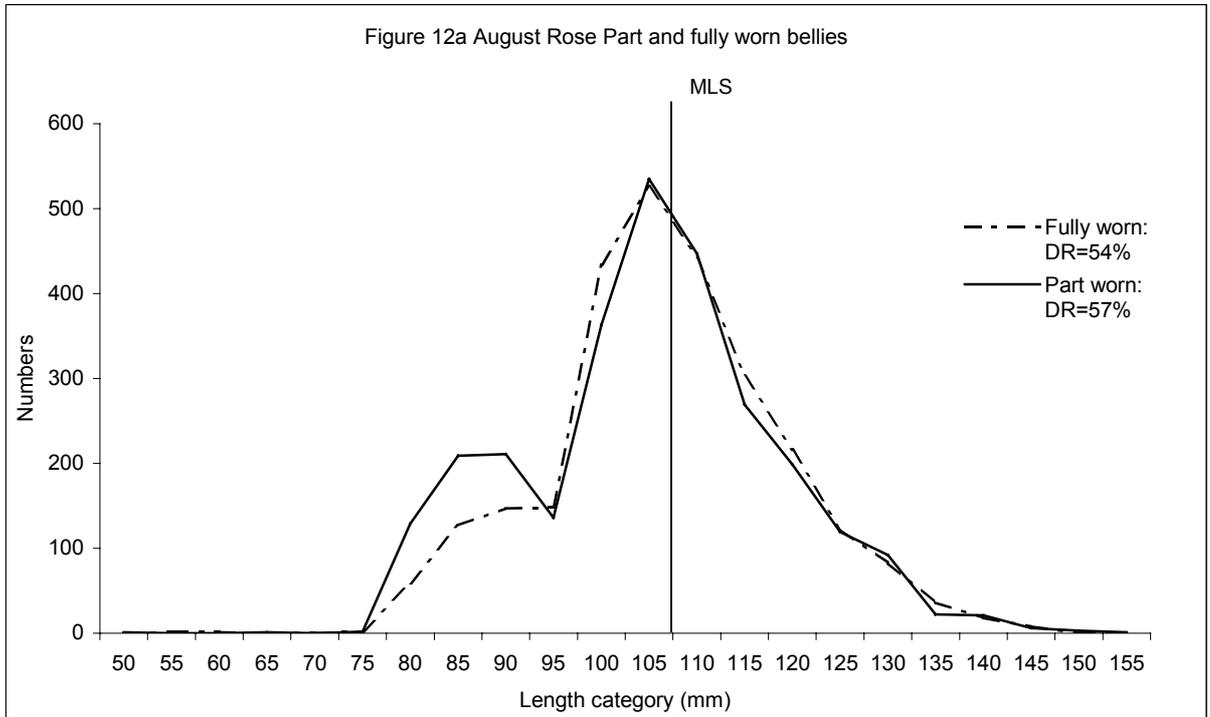


Figure 12 Aggregate Length-Frequency distributions for worn bellies

Figure 13a Ratio Large/Small Kelly (West of Scotland); Tooth spacing 9 vis 10/bar

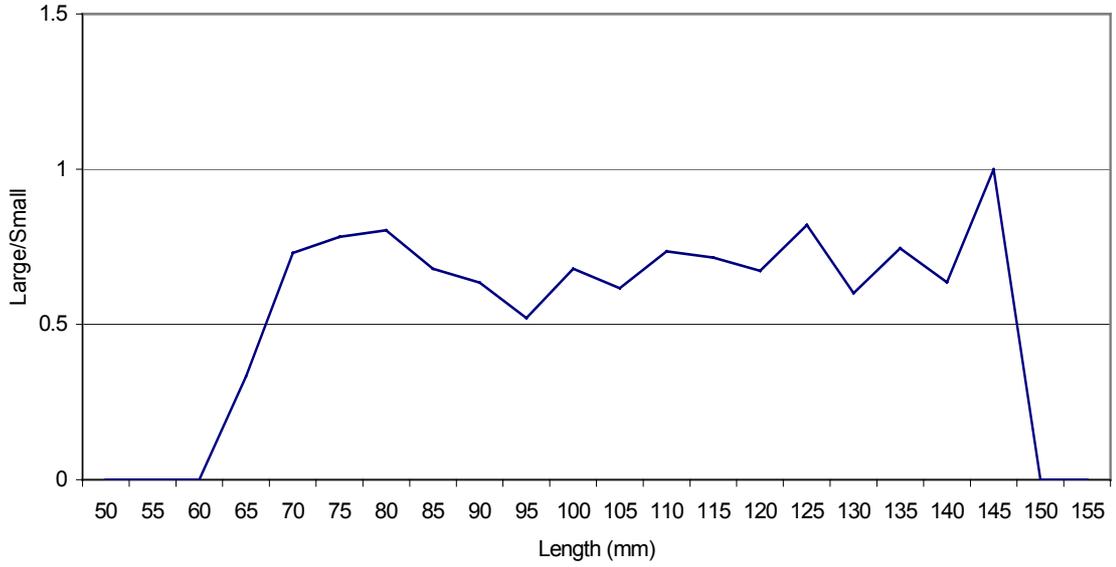


Figure 13b Ratio Large/Small Irish sea (August Rose); Tooth spacing 9 vis 10/bar

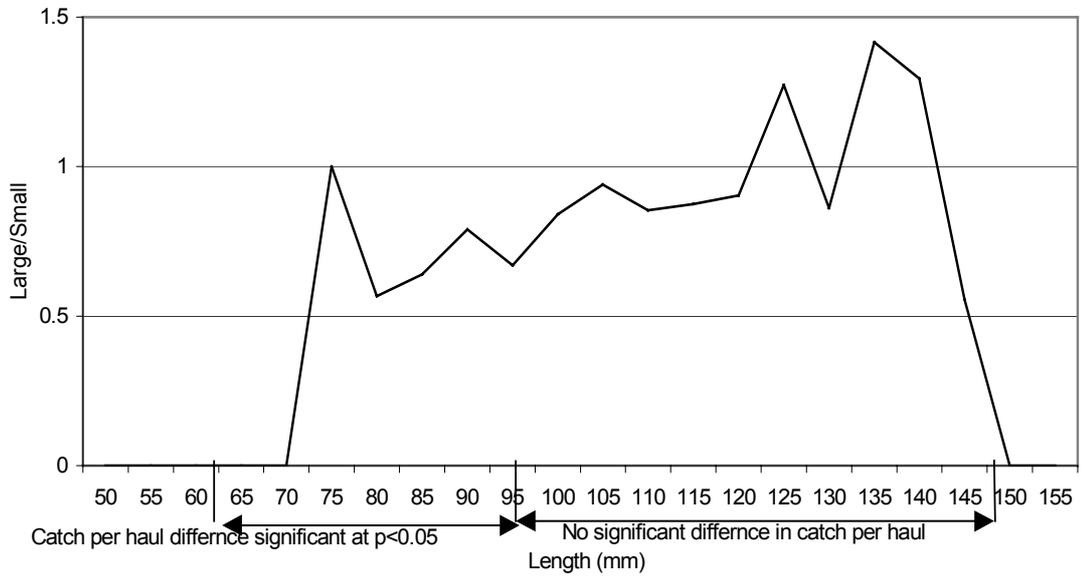


Figure 13 Ratio of catch in Large/Small tooth spacing vis length

Figure 14a Seafalke 75mm vis 64mm

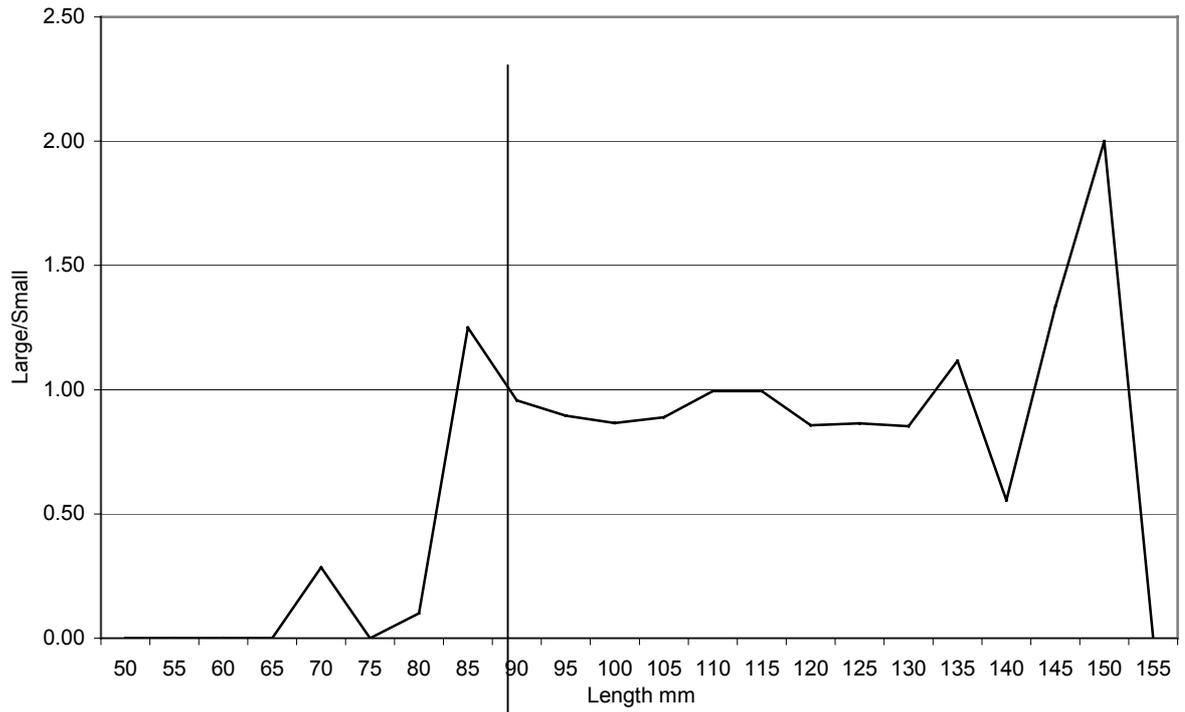


Figure 14b Kelly (W. of Scotland) 73mm vis 63mm

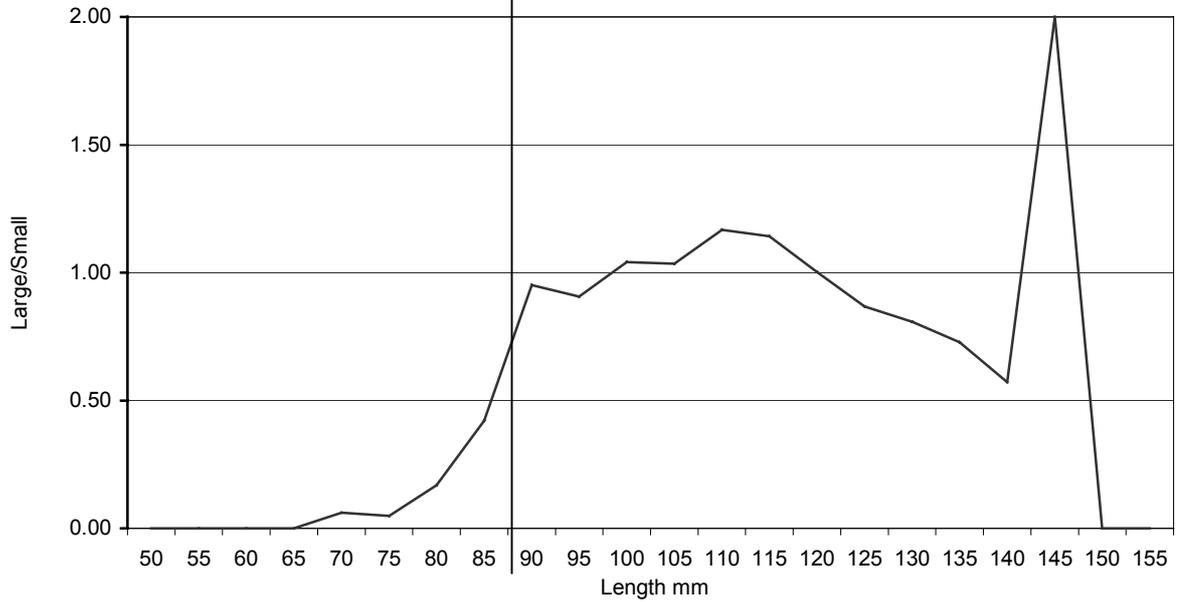


Figure 14 Ratio of catch in Large/Small bellies against length

Discussion

Selectivity mechanism

Taken together with the results from Lart et al. (1997) these results show that the most consistent factor affecting selectivity was the ring size in the bellies of the dredges. In the results from both areas the selectivity ranges were similar with the upper end of the selectivity range corresponding to 85-90mm for 74-75mm rings. Since the MLLS is 100mm to 110mm dependent upon the area fished, there appears to be scope for a larger ring size which could reduce discards further.

Although there is some evidence for selectivity due to changes in the number of teeth to the bar and hence tooth spacing, the increases in selectivity due to this change were not consistent. When selectivity at the teeth was apparent in the Irish Sea trials the upper end of the selection range for 9teeth/bar was 95-100mm suggesting that this could be the correct spacing for an MLS of 100mm. However, the variable way in which the teeth were set on the bar and the tension at which the springs were set suggests that this would be a less reliable parameter for improving selectivity than changes in ring size.

Selection at the backs was not demonstrated in the Seafalke trials and not tested in the Kelly trials where backs were matched to bellies, and comparison made between mesh sizes. It appears that selectivity does not occur at the back or mesh and there is evidence for reduced overall efficiency with an increase in the ring size of the back of the dredge.

Wear on the gear

Increased selectivity does appear occur with increasing wear and there is some evidence for a loss of overall efficiency in these results. The rate of wear would be expected to depend on the sea bed conditions and vary from boat to boat.

Conclusions

- The most reliable selectivity mechanism was the belly ring size. Similar results were obtained from belly ring selectivity from both the East Channel and Scottish trials.
- Tooth spacing was less consistent in selectivity; seabed type, tooth shape and the way in the teeth were set up and the springs tensioned may have an effect on selectivity. When selectivity at the teeth does operate the spacing corresponding to 67mm between teeth (9 teeth/bar) appears to correspond to an MLLS of 100mm. Further trials are required to confirm these results and to investigate the use of 77mm spacing (8 teeth/bar) for selecting to 110 mm MLLS.
- Wear on the bellies can result in increased selectivity but there is evidence of loss of marketable scallops.
- There is scope for increased belly ring size of the bellies improving selectivity of the dredges but observations of the rate of wear would be required.

3.2.2.4 SIZE SELECTIVITY OF SPRING TOOTH *PECTEN MAXIMUS* DREDGES

Introduction

[Section 3.2.2.3](#) and Lart et al. (1997) examined selectivity mechanisms in sprung tooth *Pecten maximus* dredges. It was concluded that belly ring size was the most consistent determining factor in the selectivity of the dredges and that under some circumstances tooth spacing can affect selectivity. It was also found that wear on the bellies had an effect on selectivity. This Section reports on trials aimed at quantifying belly and tooth spacing and selectivity and examining the effect of wear on the bellies and tooth length on selectivity.

Three sets of sea trials are reported on in this Section. They were carried out on FV Star of Annan fishing in the Tiree passage off the West of Scotland, which investigated belly ring size, FV De Bounty, which investigated tooth spacing and tooth length, fishing in the Irish sea off the Isle of Man. Finally, FV Alena fishing off Newhaven on the South coast of England which also investigated larger ring sizes.

Materials and apparatus

The gear specifications of the three vessels are described in Tables 1 to 3 and Figure 1. The dimensions of the belly rings and tooth spacings were chosen for the following reasons:

Star of Annan.

The vessel's normal gear used 75mm 10mm thick rings which is the size normally used on many vessels in the fleet and was the size used for previous experiments. The use of this size of belly resulted in discards of scallops (see [Section 3.2.2.3](#)) and it was decided to explore whether larger bellies would result in improved selectivity and would be durable enough for use in commercial practice. The size of ring was constrained by the need to have whole numbers of rings across the belly and keep belly dimensions the same as the original 75mm rings. Thus the three ring sizes were calculated on that basis; 75mm id with 10mm wire thickness being the standard commercial ring size with 10 rings wide and 85mm id with 10mm wire thickness and 80mm id with 12mm wire thickness at 9 rings wide. The washers on the 85mm id rings were the same as the 75mm rings but it is normal practice to use larger, thicker washers on the thicker rings so this practice was followed. All the backs were 75mm id 8mm wire thickness rings which is common through the industry.

These three ring sizes were compared against 60mm id, 10mm thickness rings as being considered to be capable of catching the population. Ring sizes as small as 50mm have been used to sample populations of scallops Fifas and Vigneau (in prep) but it was considered that this was not a viable option with the amount of stones and other debris encountered in fisheries using sprung toothed gear. To investigate whether small scallops were passing through the back rings, a small mesh cover was placed over the back of one of the small ringed dredges. During the whole 36 hauls of the trip a total of 3 scallops, all smaller than 75mm shell length, were found in the cover. Whilst this does not account for scallops lost through the bellies it does suggest that only the very smallest scallops passed through the small rings used.

Approximately 12mm was removed from the tips of all the teeth prior to the experiment to bring them down to the starting lengths described in Table 1. This is because when manufactured the tips of the teeth are very sharp so the rate of wear is high in the first few days of use.

The dredges were deployed in 3 sets of two in pairs on both bars of the vessel as shown in Figure 1. The experimental ring sizes were all on one side of the vessel and the population dredges were on the other side. On the first voyage; a charter voyage, the dredges of each group were separated by a space, for the subsequent wear trials the spaces were filled with normal commercial size dredges. See below for full details of the experimental design.

The spring tension was set, on the advice of the Skipper, to 4.0 kgf-m and kept consistent throughout the trials by checking every 3 hauls using the apparatus shown in [Figure 3 Section 3.2.2.3](#).

De Bounty

The gear specification used on this set of trials is as shown in Table 2. Up until this trial there was not much information on selectivity at the teeth. Thus normal commercial dredges of circa 78mm id ring size were used to examine the effects of different tooth spacing in commercial use on the catches of these dredges. To explore the use of very small tooth spacing on the capture of small scallops with a view to examining the feasibility of catching the whole population of scallops two sprung toothed queenie dredges using small rings were placed on the centre position on each bar. The larger ringed scallop dredges occupied the outer two positions on each bar. There was a gap between the queenie dredges and the scallop dredges, to avoid interference between the dredges.

The spring tension was set, on the advice of the skipper to 7kgf-m and checked every three hauls with the modified torque wrench. For the shortened teeth the spring tension was set to 6.2kgf-m because it was calculated that this would result in the same moment at the teeth and it fitted commercial practice which is to decrease spring tension as the teeth wear.

Alena

The gear specification used is shown in Table 3. On this boat the normal complement of dredges was 6 per side of 76cm dredges. Instead one dredge was placed in the middle of the bar and two pairs of dredges were placed in the inboard and outboard positions of each bar with a gap between these and the middle dredge. As for the Star of Annan trial, the experimental dredges were compared with a standard 60mm-ringed dredge.

The choice of ring sizes was made as follows. The vessel herself was using 85mm id 12mm thick rings, so these were placed in the middle position on the bar. The other two sizes were calculated by taking the selection factor calculated for thick and thin rings from the Star of Annan trials and estimating an expected L_{50} . For the 88mm id (12mm wire diameter) rings this produced an expected L_{50} of 94mm and for the 92mm rings an expected L_{50} of 105mm. This was intended to be approximately the same relationship to the MLLS of 110mm in force in the scallop fishery in ICES

division VIId where Alena was operating as 80mm and 85mm diameter rings were to the 100mm MLLS in Scottish waters.

The spring tension was set, on the advice of the Skipper, to 7 kgf-m and kept consistent throughout the trials by checking every 3 hauls using the apparatus shown in [Figure 3 Section 3.2.2.3](#).

The setting up of these dredges was somewhat of a compromise. Initial estimates showed that there would not be room for a whole number of rings on the bar. Although discussion with dredge manufacturers suggested 9 rings across would be a suitable dimension. When actually placed on the bar there was not properly room for all the rings so for both the ring sizes there was some overlap of the rings at the forward end of the dredge. The bellies were well stretched out at the rear end of the dredge and the overlapping only affected the first 2-3 rows of rings. Nevertheless this may have affected efficiency. There was a similar effect on the fisherman's dredge, the belly of which was also protected against abrasion by a network of anti chafe devices.

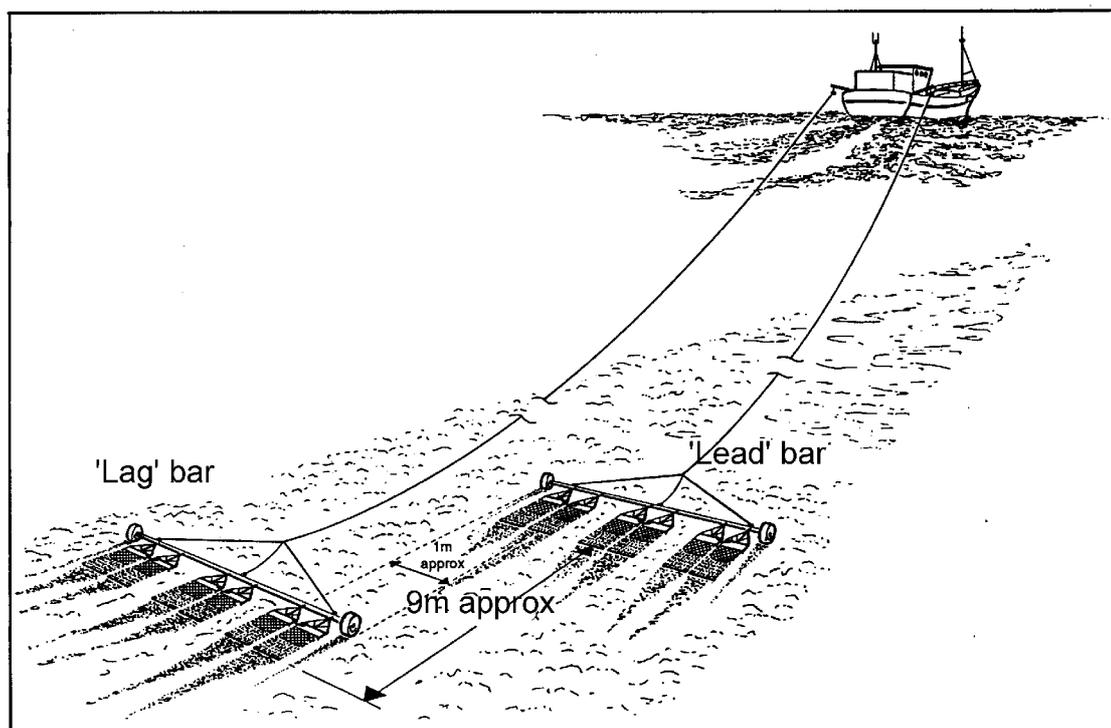


Figure 1 Diagram of the towing arrangement on the FV Star of Annan; underwater distances are estimated from dimensions measured at the surface. Overlap of the two bars was approximately 1m. Both the two other -FVs De Bounty and Alena- vessels towed their dredges from towing booms in the forward part of the vessel with no overlap.

Table 1 Gear specification *FV Star of Anan*

Component	Specification	Measured dimension(mm) \pm SD
Dredge bars	7.16m with 6 x 76cm dredges attached (normal arrangement is 8 x 76cm dredges.	
Dredge frames	Standard Oban Scallop Gear 76cm	
Tooth bar	705mm hardened bar	
Teeth	Hardened 'French teeth '. Teeth were 90mm x 10mm at start of experiment (tips removed).	Distance between teeth 9 Teeth/bar=77mm
Mesh	82mm blue steel single strand 20 mesh across x 13 down	81.7 \pm 2.6
Belly rings	60mm: 60mm ID x 10mm wire 75mm: 75mm ID x 10mm wire 85mm: 85mm ID x 10mm wire 80mm: 80mm ID x 12mm wire	60 75 \pm 1 85 \pm 2.6 80 \pm 0.69
Bellies	60mm: 14 long x 12 rings wide including turn-ups 74mm: 11 long x 10 rings wide including turn-ups 85mm: 10 long x 9 rings wide including turn-ups 80mm: 10 long x 9 rings wide including turn-ups	
Belly washers	60mm: 10mm thick; 24mm ID 45mm OD. 74mm: 10mm thick; 24mm ID 45mm OD. 85mm: 10mm thick; 24mm ID 45mm OD. 80mm: 12mm thick; 27mm ID 48mm OD.	
Back rings	60mm: 65 mm ID x 8mm wire All others: 75mm ID x 8mm wire	
Backs	60mm: 6 long x 9 wide All others: 5 long x 8 wide	
Back washers	external diameter=7mm, thick 38mm OD, 18mmID	

Table 2 Gear specification *FV De Bounty*

Component	Specification	Measured dimension: (mm) + SD
Dredge bars	5.7m with 5 x 76cm dredges attached in 3 groups on each: 1 queenie dredge in the middle with 2 pairs of king scallop dredges on either side with a gap between the queenie gear and the king gear (normal arrangement is 8x60cm dredges).	
Dredge frames	Standard METALCO Scallop Gear 76cm	
Tooth bar	705mm hardened bar	
Teeth	Hardened 'French teeth'. Teeth were 90mm x 10mm at start of experiment (tips removed) and 85mm long at the end. For the shortened teeth experiment the teeth were cut down to circa 60mm with a squared off end.	Distance between teeth 17 teeth/bar = 32mm 10 teeth/bar = 65mm 9 teeth/bar = 75mm 8 teeth/bar = 87mm
Mesh	70mm mesh	
Belly rings	Queenie rings: 60mm 8mm wire King scallop: 78mm 10mm wire	
Bellies	60mm: 14 long × 12 rings wide including turn-ups 78mm: 11 long × 10 rings wide including turn-ups	
Belly washers	60mm: 8mm thick; 17mm ID 40mm OD. 78mm: 10mm thick; 24mm ID 45mm OD.	
Back rings	60mm: 65 mm ID x 8mm wire All others: 75mm ID x 8mm wire	
Backs	60mm: 6 long x 9 wide All others: 5 long x 8 wide	
Back washers	external diameter=7mm, thick 38mm OD, 18mmID	

Table 3 Gear specification *FV Alena*

Component	Specification	Measured dimension: mm+ SD
Dredge bars	4.0m with 5 x 76cm dredges attached (normal arrangement is 6 x 76cm dredges.	
Dredge frames	Standard BEEJAY 76cm	
Tooth bar	705mm hardened bar	
Teeth	Hardened 'Peg '. Teeth were 80mm x 20mm at start of experiment (tips removed).	Distance between teeth 9 Teeth/bar=65mm
Mesh	82mm blue steel single strand 20 mesh across × 13 down	81.7 ± 2.6
Belly rings	60mm: 60mm ID x 10mm wire 85mm: 85mm ID x 12mm wire (worn ship's gear) 88mm: 88m ID x 12mm wire 92mm: 92mm ID x 10mm wire	60 85 88± 0.8 92±1
Bellies	60mm: 14 long × 12 rings wide including turn-ups 85mm: 10 long × 9 rings wide including turn-ups 88mm: 10 long × 9 rings wide including turn-ups 92mm: 10 long × 9 rings wide including turn-ups	
Belly washers	60mm: 10mm thick; 24mm ID 45mm OD. 85mm: 10mm thick; 24mm ID 45mm OD. 88mm: 12mm thick; 27mm ID 48mm OD. 92mm: 10mm thick; 24mm ID 45mm OD.	
Back rings	60mm: 65 mm ID x 8mm wire All others: 75mm ID x 8mm wire	
Backs	60mm: 6 long x 9 wide All others: 5 long x 8 wide	
Back washers	external diameter=7mm, thick 38mm OD, 18mmID	

Data acquisition

The location, times of shooting and hauling, wind and sea state, warp length and mean depth recorded during the tow were recorded. When the catch was tipped on to the deck the scallops from each dredge were separated (using a white bag normally used for storing catch layered between the catches; this was especially employed on the commercial voyages). The scallops from each dredge were damaged graded and the length measured to 5mm below using the apparatus shown in [Figure 4 Section 3.2.2.3](#). All scallops were measured; there were no raising factors required.

Experimental design

Star of Anann

Prior to the trial a reconnaissance trip was carried out to select suitable areas for study with a sufficient length distribution of scallops. A suitable area was found in the Tiree passage, which is an important scalloping ground in this area (Figure 2). The dredge configuration on this vessel was as shown in Figure 1. The dredges were set in 3 pairs of two dredges on each bar towed from booms on the quarters of the vessel. The experimental dredges were set on one side and the 60mm standard dredges intended to catch the available population on the other side of the vessel. The daily routine consisted of 6, 50 minute hauls in the same order each day. Tows were conducted down parallel tracks but not exactly the same track. The lead and lag dredges were alternated between port and starboard to ensure that the experimental and population dredges had equal exposure to the two levels of this factor, and the pairs of experimental dredges moved on along the bar each day. Thus over a period of three days the experimental dredges were fished in each of the positions on the bar. At the end of three days the experimental and population dredges were moved across to the other side of the vessel and the trial repeated so that each of the dredges had been fished in each of the positions over the course of the six-day trial.

After the commercial trial was completed it was agreed that the experimental dredges should be left on the vessel to conduct trials of the durability and comparative selectivity under commercial conditions. Since this was a commercial operation the three dredges which had been removed from the bars were replaced and the vessel fished in her normal commercial activities. The catches were monitored over 2 periods of three days during the following 3 months of fishing. This was carried out in a similar way to the trials described above; catches from the three dredge types were separated and all scallops measured before being returned to the fishermen. There was no corresponding population dredge; the dredges were simply compared between themselves. The dredges were moved along the bar in the same manner described above but because only three days were observed the dredges were only set on one bar; the lead bar with the short wire length. The longest internal measurement of the rings was made at each stage of the trial with the central column of rings and three rows of rings measured at the top, middle and end of the dredge.

The initial trails were carried out in the last week in March 2001 and the wear trials were carried out in the following 3 months until mid June 2001. Observations were made in May and June (see Results for effort).

De Bounty

Fishing was carried out on grounds around the Isle of Man, locations are shown in Figure 3. The trials were carried out during August 2001, this period is closed for king scallop fishing so a derogation was obtained and all scallops were returned to the sea at the end of the tow.

The following levels of tooth spacing were compared on the standard scallop dredge gear:

8 vis 9 teeth/bar,
8 vis 10 teeth/bar
short 8teeth/bar vis long 8 teeth/bar

On the queenie gear set in the middle position of each bar:

17 teeth/bar vis 10 teeth/bar on the queenie dredge.

Each comparison consisted of 12 hauls usually over a period of 2 days, half way through the teeth were switched from one side of the vessel to the other and similar hauls made on the second half of the trial where weather and other factors permitted.

Care was taken to ensure that teeth were compared with the same degree of wear on them. Approximately 12mm was removed from the tips of all the teeth prior to the experiment to bring them down to the starting lengths described in Table 2. This is because when manufactured the tips of the teeth are very sharp so the rate of wear is very high in the first few days use.

In addition to the above trials a commercial comparison was made between a set of 2 85mm 10mm wire bellies with the vessel's normal gear; 78mm 10mm wire. This was carried out in a similar manor to the commercial wear trial on the Star of Annan commencing November 2001 with a later observation made in April 2002. Both these sets of bellies had undergone the same novel hardening process and the trial was carried out to see how they performed. They have remained on the vessel and will be tested further in the forth coming season (2002/3).

Alena

Fishing was carried out during November 2001 on grounds off Newhaven. It had been intended to carry out 4 days of fishing with the experimental dredges occupying each of the 4 positions on the two dredge bars; port inboard and outboard, starboard inboard and outboard with the vessel's normal dredge on the central position. However weather constraints meant that 3 days fishing were achieved two of 8 hauls and 1 of 3 hauls. It was also not possible to fish exactly the same hauls each day as in the Star of Annan trial.

In this trial the gear was mirror imaged each day, so that the dredge which had been inboard port one day was outboard starboard the next.

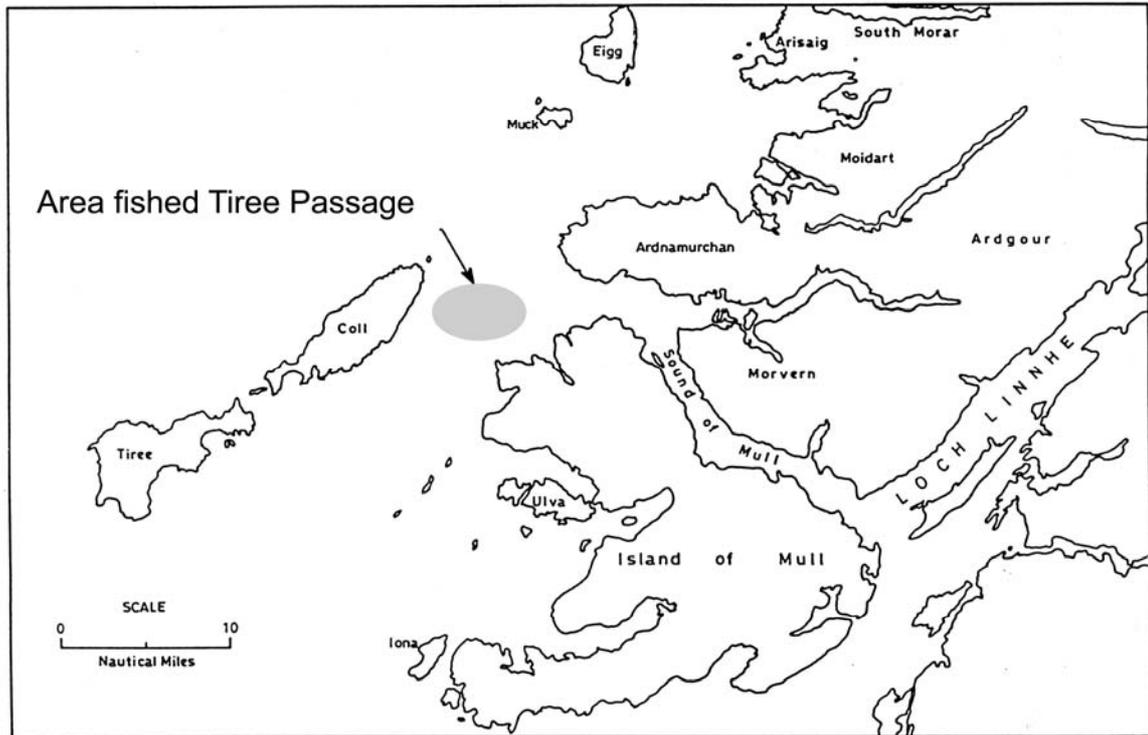


Figure 2 Location map Tiree passage: Area fished by FV Star of Annan March 2001

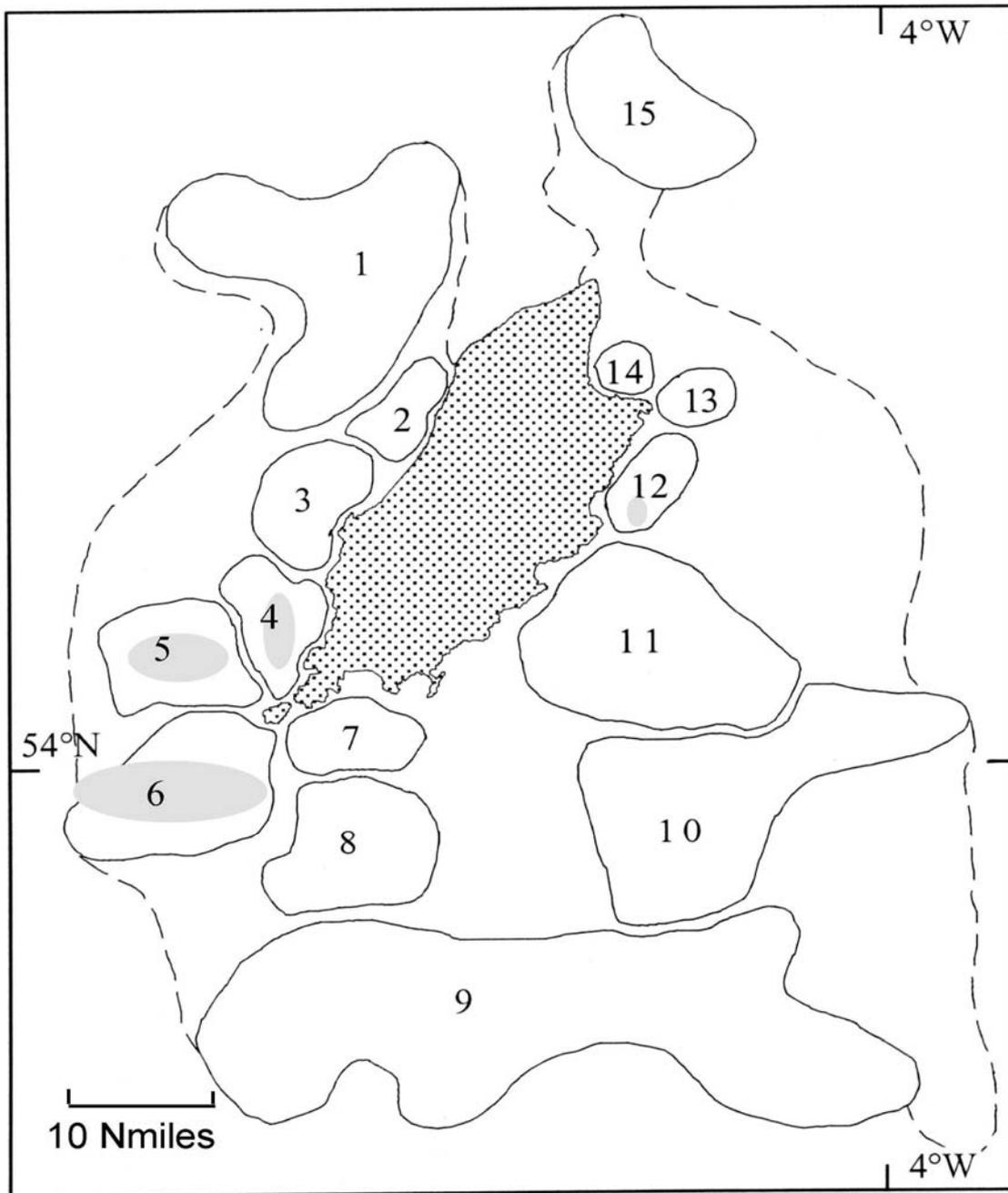


Figure 3 Location map: Grounds fished around the Isle of Man; FV De Bounty August 2001. See [Review Section 4.3.1](#) for ground names.

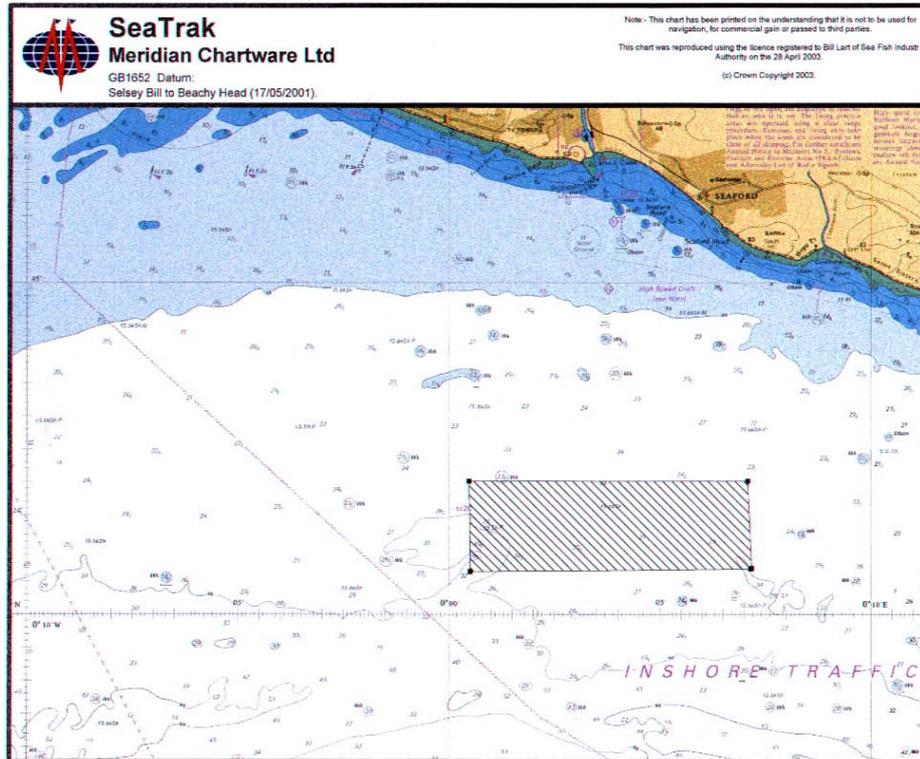


Figure 4 Location map: grounds fished off Newhaven FV Alena November 2001. Rectangle is bounded by approx. $50^{\circ}40.2' N$, $50^{\circ}42.2' N$ and $0^{\circ}0.9'$, $0^{\circ}0.5'$

Data analysis

Variation inherent in the design

The trials were designed in order that comparison could be directly made between the various belly ring sizes and tooth spacing as well as with the population dredges. This was achieved by fishing the dredges equally in different positions on the same bar and/or on opposite bars port and starboard. On FV Star of Annan there was an additional complication in that the dredges were either lead or lag. Thus there is a requirement to design the experiment to reduce bias from these source to a minimum. This was achieved by ensuring that the experimental dredges were exposed to equal quantities of effort in each of the positions on the bar and on each bar; port and starboard and to an equal number of hauls as lead or lag dredges (where required). Where feasible the tows were made in the same locations each day although not down exactly the same track. This design enables the examination of these results to establish whether there were any of these factors could have had important effects on these results. This was carried out in three ways; by analysis of variance (ANOVA) comparing the mean lengths of the scallop caught in the dredges and the mean catch per efforts of the dredges by haul. Comparison of the aggregate length frequency distributions of scallops in the various factors was made using the Kolmogorov-Smirnoff⁵ two sample test (Siegel 1988). This non-parametric test detects differences by assessing the cumulative distributions of the two samples.

Experimental Treatments

Experimental treatments; belly ring size, tooth spacing and length and wear on the bellies were compared by ANOVA as above, by comparison between length-frequency distributions and by comparing the relative efficiencies:

$$\text{Relative efficiency for a given size}(e_i) = \frac{\text{Number of Scallops in first dredge}}{\text{Number of Scallops in second dredge}}$$

Where the first and second dredges are more and less selective dredges respectively. The method used described by Fifas and Vigneau (in prep). The relative efficiency of each length group (e_i) were fitted to the model taking the form:

$$e_i = \frac{e_{\max}}{1 + \exp[-\alpha \cdot (L_i - L_{50})]} + \xi_i$$

Where:

e_{\max} = maximum asymptotic efficiency; this is considered independent of scallop size; it depends on the physical and mechanical characteristics of the dredge

α = Parameter linked to the deviation of selection of the experimental dredge, defined by the difference $L_{25} - L_{50}$ of maximum asymptotic efficiency $\alpha = 2 \cdot \ln(3) / (L_{25} - L_{75})$

L_{50} = size corresponding to 50% of the maximum asymptotic efficiency

ξ_i = unexplained residual error.

The curve was fitted by minimising the sum of weighted (by the number of scallops in both the dredges compared) residual squares using the solver procedure on excel.

⁵ Referred to as the K & S test in the Figures.

This procedure attributes most weight to those size classes which contained the most scallops and thus probably reduces uncertainty in parameters α and L_{50} (see Fifas et al. for details where the method for estimating the coefficients of variation and standard deviations are described).

Results

Variation inherent in the design

This was investigated by comparing mean lengths of the scallops, catch per dredge pair per (results from replicate dredges in each pairs were added together) haul and the length-frequency distributions of the scallops (by ANOVA and Kolmogorov-Smirnoff tests respectively see above) shown in Tables 4 and 5 and Figures 5 and 6.

Although many of the factors inherent in the design; inboard, middle, and outboard dredges, and lead and lag dredges showed significant differences between the different levels of these factors in terms of mean length and in the length distributions, these differences are very small. There are significant differences in the catch per dredge pair per haul per haul by day; this is inevitable because conditions are likely to vary by day and there inevitably variation in the distributions of scallops. In the Alena trial case there was a partial shift in the location of fishing because of poor catches at the first site.

For the Star of Annan trials the main differences appear to be on the right hand side of the curve away from the selectivity range. The only exception to this is the lead/lag factor where there is some evidence of smaller scallops captured in the lead bar. However since there is no evidence of interaction between the position on the bar and the lead/lag factor this suggests that this is not an effect of overlap between the two bars (Figure 1). All dredges were exposed to equal effort as lead and lag dredges during the charter trips so this factor should be cancelled out.

Differences observed between the experimental factors, ring size or tooth spacing was much greater than the factors discussed above so it was not considered necessary to allow for differences due to these factors in subsequent analysis.

Table 4 Environmental and Experimental factors : Star of Annan

Mean Scallop Length (mm)	Significance	Interactions and notes	Scallop numbers per haul	Significance
110			52	
106	p=0.005		73	p<0.001
106			66	
107			71	
107			64	
106			66	
106.44			63	
107.26	p=0.013	No significant interactions	64	N S
106.74		between position and wire	68	
106.43	p<0.001	length	67	N S
107.23			63	
107.09	p=0.018		66	N S
106.54			65	
105.62	p<0.001		68	N S
107.22			64	
106.87			63	
110.48			59	

Table 5 Environmental and Experimental factors: Alena

Treatment	Level	Mean Scallop Length (mm)	Significance	Mean Total Catch per haul Scallop numbers	Significance	
Day	1	106.26	p<0.001	56	p=0.002	
	2	104.03		96		
Position on bar	Inboard	104.27	p=0.017	76	Not Significant	
	Outboard	105.24		75		
	Port	106.02	p<0.001	66	Not Significant	
	Starboard	103.92		88		
Belly	60	103.13	p<0.001	53	Significant at	
	88	107.03		59		p<0.001
	92	112.28		32		

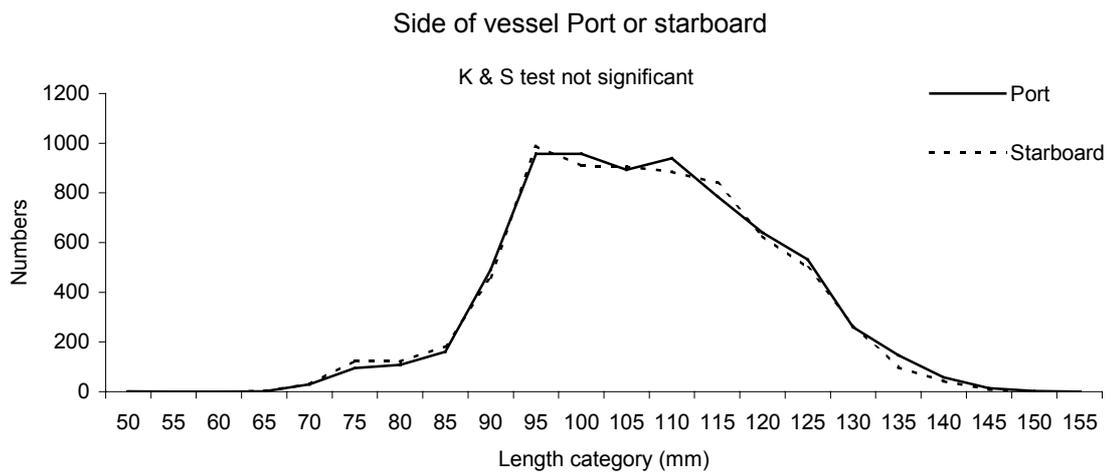
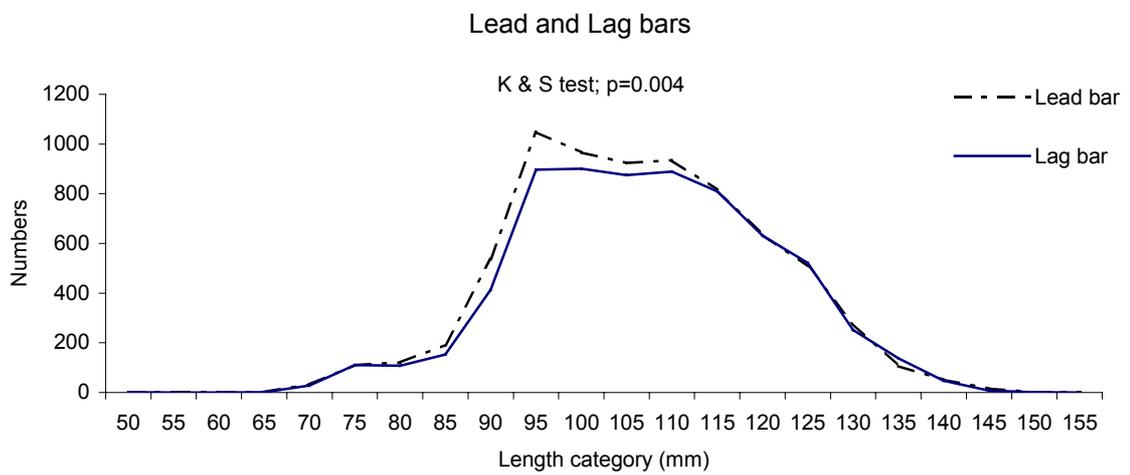
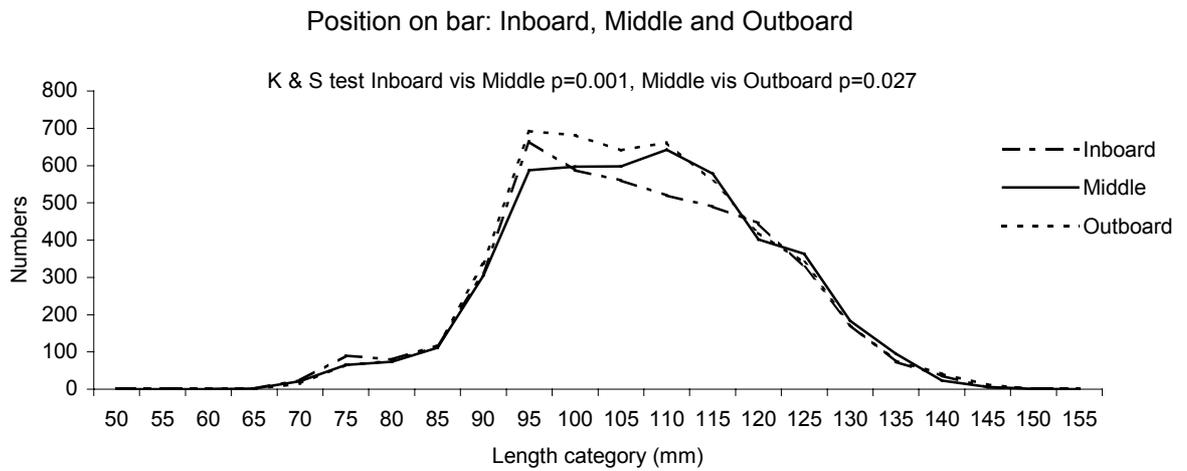


Figure 5 Experimental factors: Aggregate Length frequency distributions, Star of Annan

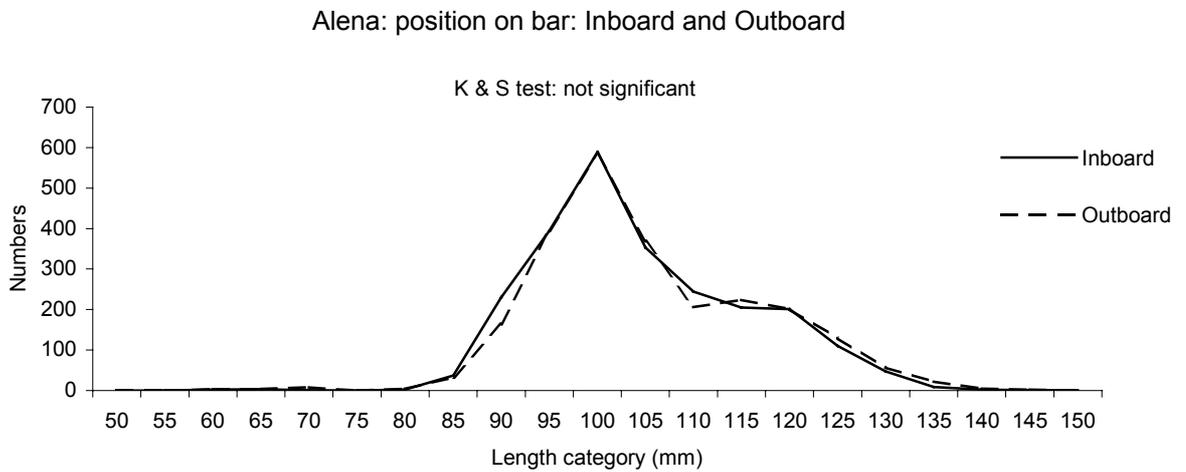
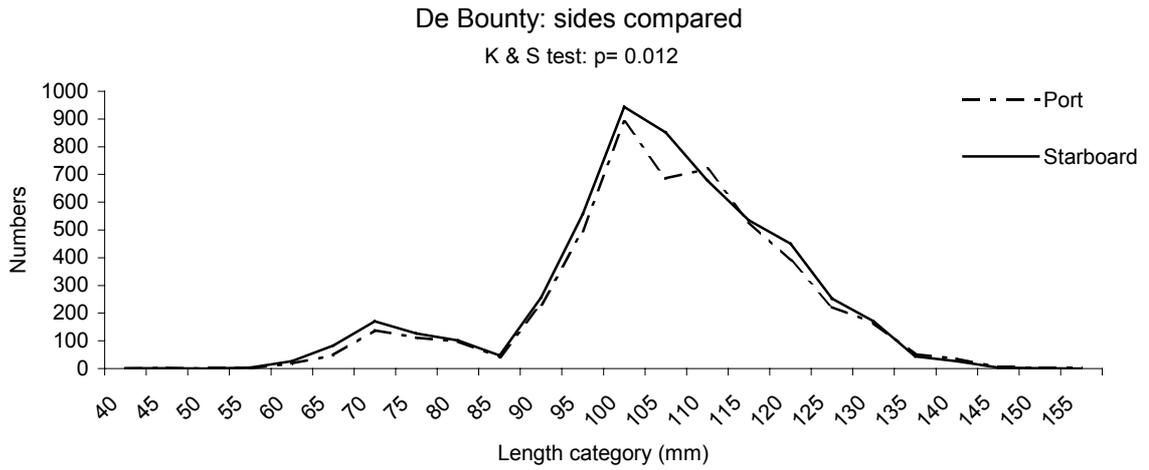


Figure 6 Experimental factors: Aggregate Length frequency distributions, De Bounty

Dredge selectivity

The selectivity measurements are described below in the order which the scallops encountered the dredges; tooth followed by ring selectivity.

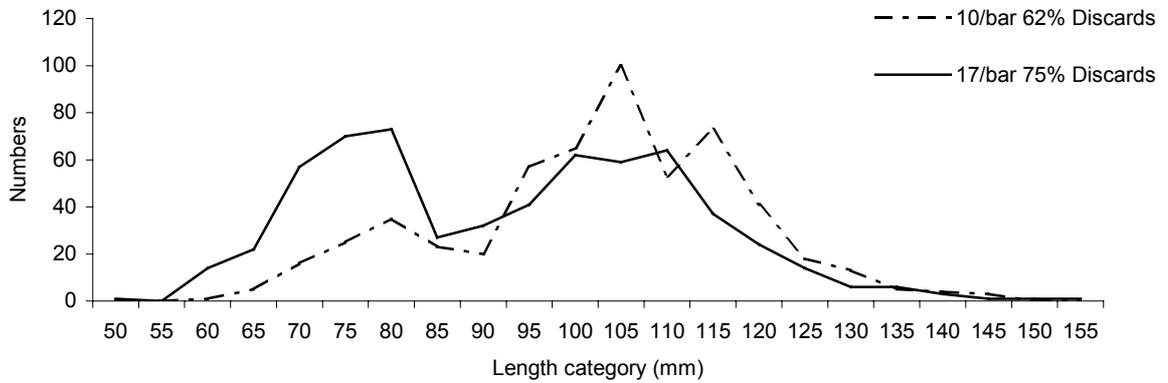
Tooth selectivity

Previous studies ([Section 3.2.2.3](#) and Lart et al. 1997) had indicated that tooth selection could occur. However it is not as consistent as ring selectivity and this study was intended to examine the effect further. Making measurements of selectivity constants was a lower priority, particularly because there was uncertainty as to what to regard as a 'population' catching gear in the context of tooth spacing. Very close together teeth and small rings might have resulted in poor performance by the dredges, hence these configurations were restricted to the middle dredges on the bar.

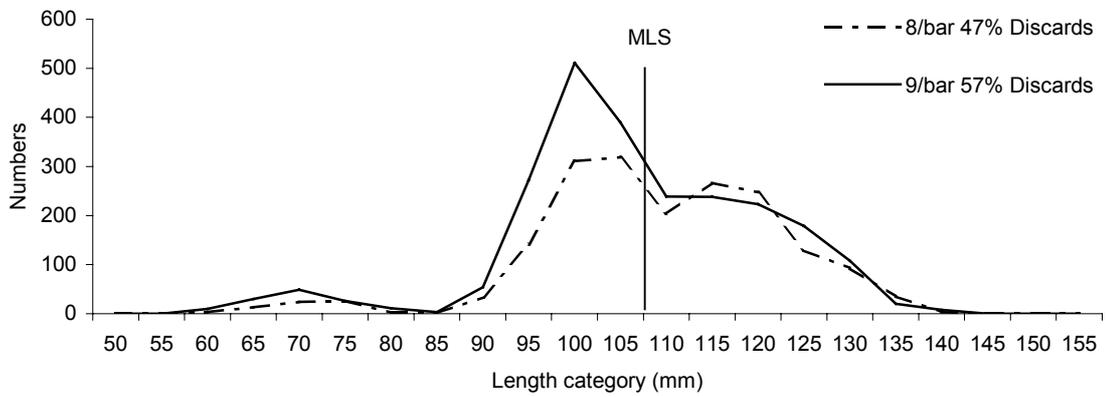
The comparative length frequency distributions of 10 vis 9/bar (August Rose data; see [Section 3.2.2.3](#)) 9 vis 8/bar, 10vis 8/bar and 17vis 10/bar are shown in Figure 7, and the comparison short vis long (8/bar) is shown in Figure 8. All these graphs show some signs of selectivity either by increasing the length or distance between the teeth. This is indicated by the length frequency distributions of catches converging in the larger size categories, only the results from 10 vis 8/bar and 17vis 10/bar were sufficiently consistent to fit selectivity curves to. These are shown in Figures 9 and 10 and the estimates of selectivity parameters shown in Table 6.

These results indicate that selectivity does occur at the teeth. The results from 8/bar indicate that this tooth spacing is suitable where the MLS is 110mm. This was the tooth spacing used by the vessel chartered (De Bounty) and also other vessels in the Irish Sea. The comparison between the 17/bar and 10/bar indicates that there are more small scallops available to the gear than would appear if the 10/bar dredges were used but that the relative efficiency on the larger scallops appears to be reduced.

17/bar vis 10/bar with queenie rings on the bellies



9 teeth/bar vis 8 teeth/bar



10 teeth/bar vis 8 teeth/bar

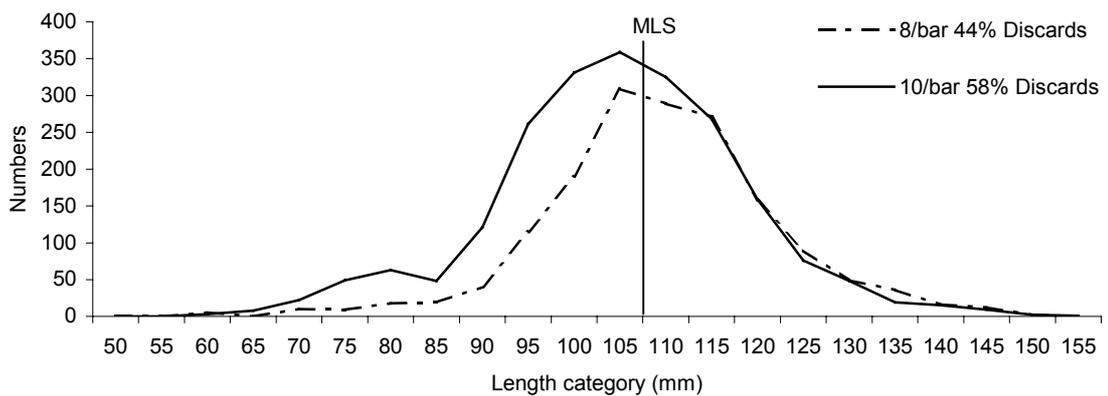


Figure 7 Tooth spacing comparisons: Aggregate length frequency distributions

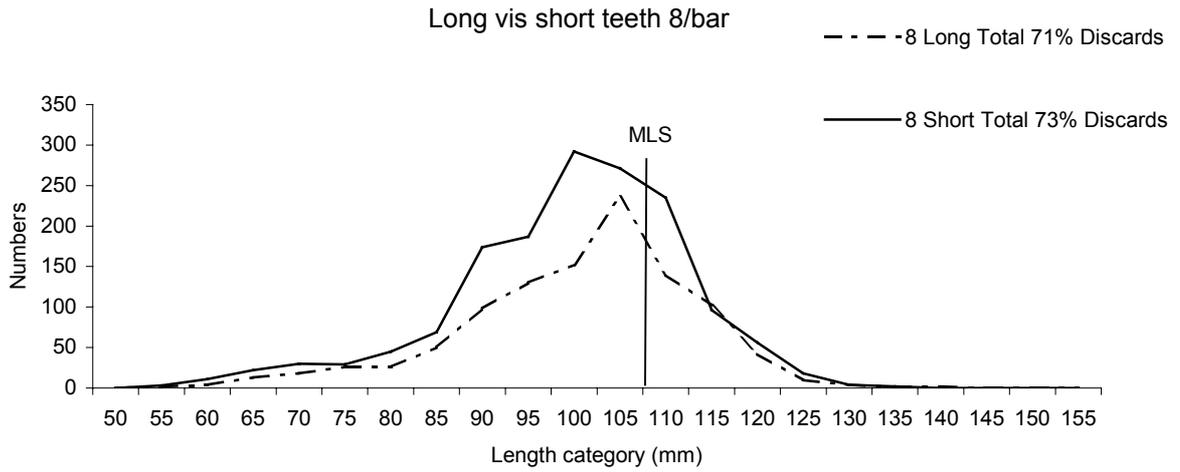


Figure 8 Tooth length comparisons: Agreggate length frequency distributions

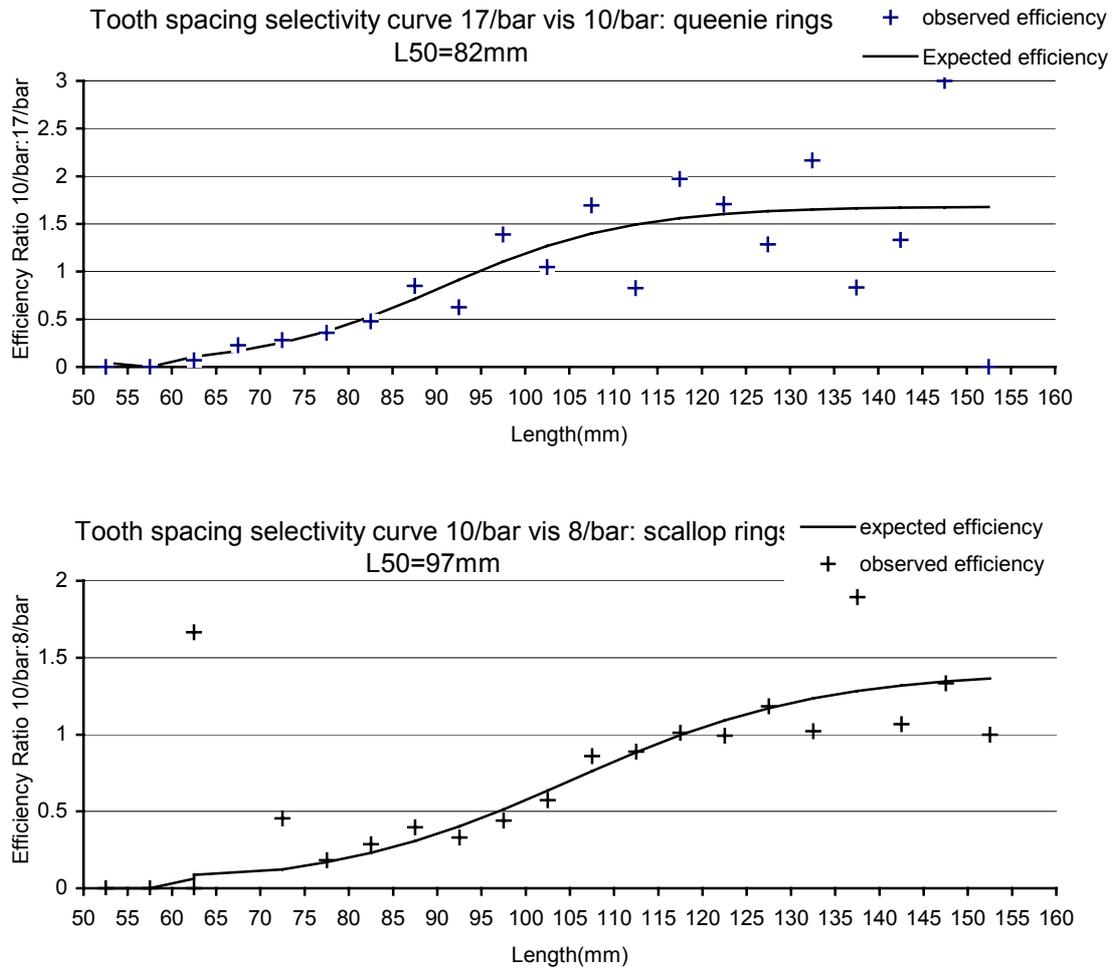


Figure 9 Tooth spacing Selectivity curves

Table 6 Selectivity parameters; Tooth spacing

Tooth spacing	L ₅₀ (mm)			Maximum asymptotic efficiency			α		
	Estimate	SD	CV %	Estimate	SD	CV %	Estimate	SD	CV %
10 vis 8	97	12.2	12	1.41	0.36	26	0.07	0.05	65
17 vis 10	82	8.4	9	1.68	0.27	16	0.10	0.1	69

10 vis 8 used 78mm diameter rings, 17 vis 10 used 60mm diameter rings

Ring size; new bellies

The length frequency distributions of the 4 belly ring sizes tested on the Star of Annan voyage (West of Scotland) are shown in Figure 10 and those from the Alena voyage (South coast) in Figures 11. In each case the population dredges were made comparable with the catches of the experimental dredges. For the Star of Annan voyage results the total catch for all the length groups in the population dredges was divided by 3 and for the Alena results the total results from the inboard and outboard dredges were divided by 2. The results for the middle dredges for the Alena voyage were compared directly between the population dredges and the vessel's own dredges. The curves fitted by the method described above are shown in Figures 12 and 13. In all the estimates of L_{50} and the selectivity curves allowance has been made for the fact that the scallops were measured into 5mm categories by adding 2.5mm to the estimates and plotting the X axis accordingly. The estimated e_{max} , L_{50} and α standard deviations, and coefficients of variation of these estimates are shown in Table 7.

The results for the Star of Annan voyage indicate good selection with smooth, steep curves for both the 75 and 85mm diameter rings (10mm wire thickness) with the curves fitting well to the data. The 80mm (12mm wire thickness) diameter rings' results are very similar to those obtained for the 75mm rings. There is no evidence of loss of larger marketable scallops (>100mm length in this fishery) with the use of any of the new experimental gears.

The MLLS⁶ for scallops in ICES division VIIId where the Alena voyage was carried out is 110mm as compared with 100mm in the Scottish waters. Thus this was considered a good location to carry out the study of larger ring sizes. As discussed in the description of the gear the expected L_{50} s for ring sizes 88mm id (12mm wire) and 92mm (10mm wire) were 94mm and 105mm respectively. The estimated L_{50} s for these results were 98mm and 108mm. This suggests that there is a slight increase in the selection factor in the larger ring sizes. The asymptotic efficiency (e_{max}) of both the experimental dredges was not as high as the ships dredge which would therefore lead to a reduction in the catch of marketable scallops. The poor efficiency of these bellies may be as a result of the crowding of the rings on to the dredge bar discussed in the gear descriptions. The fishermen's 85mm gear was worn and covered in a protective matrix of chains covered in small rubber discs to prevent chafe. Thus these results should be treated with some caution. This dredge's ring size was 85mm id (12mm wire), which had an estimated L_{50} 97mm which was very close to the L_{50} of 96mm estimated for the 85mm id (10mm wire) used on the Star of Annan voyage.

⁶ Minimum Legal Landing Size; abbreviated to MLS in the Figures.

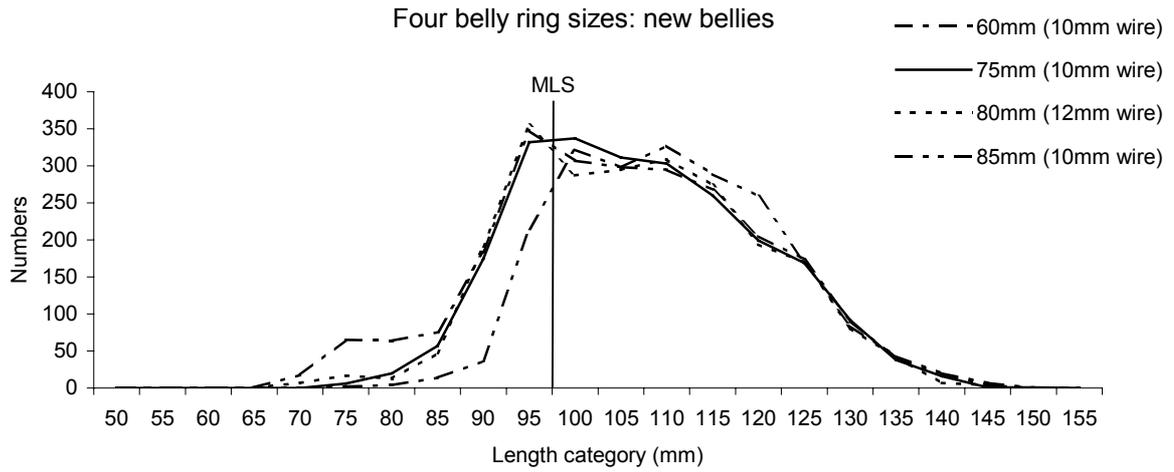


Figure 10 New bellies: Aggregate length frequency distributions, Star of Annan

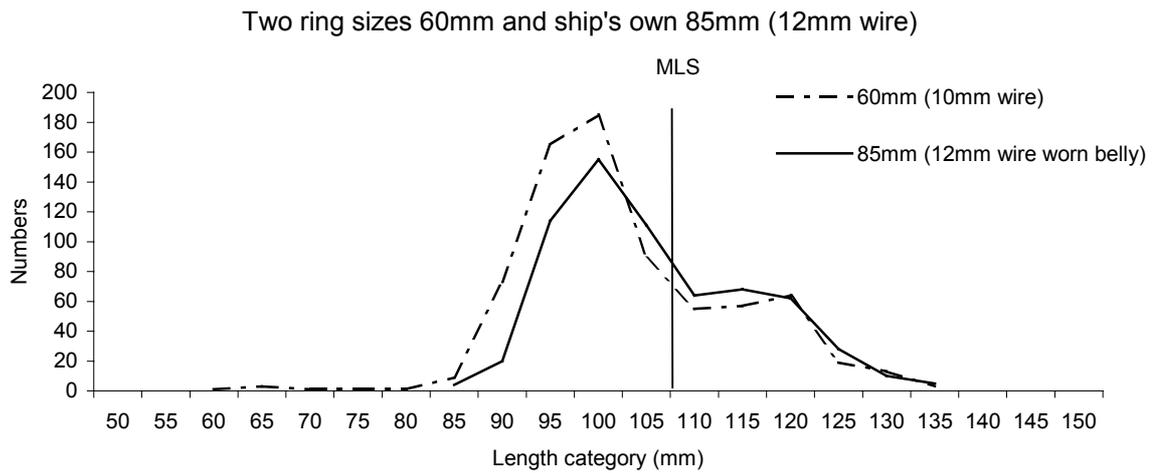
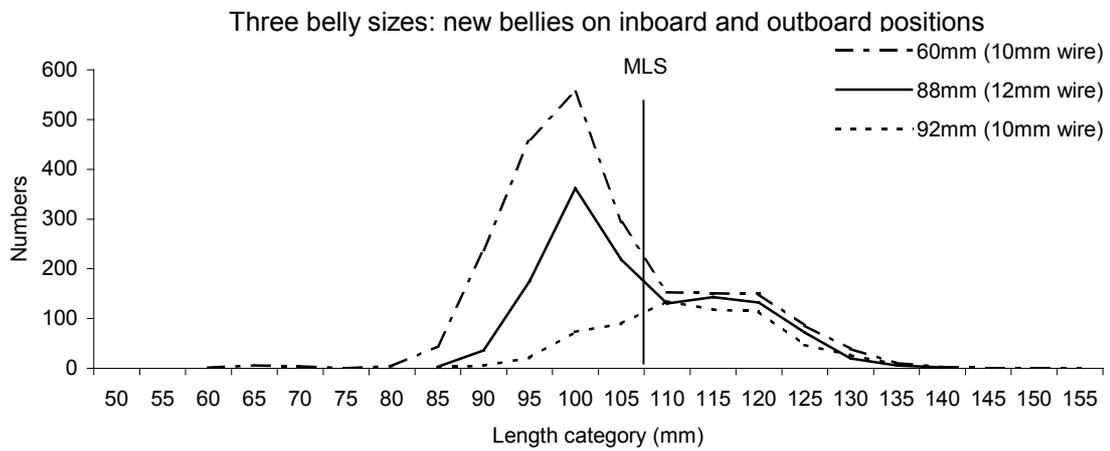


Figure 11 New bellies (except ship's own) aggregate length frequency distributions, Alena

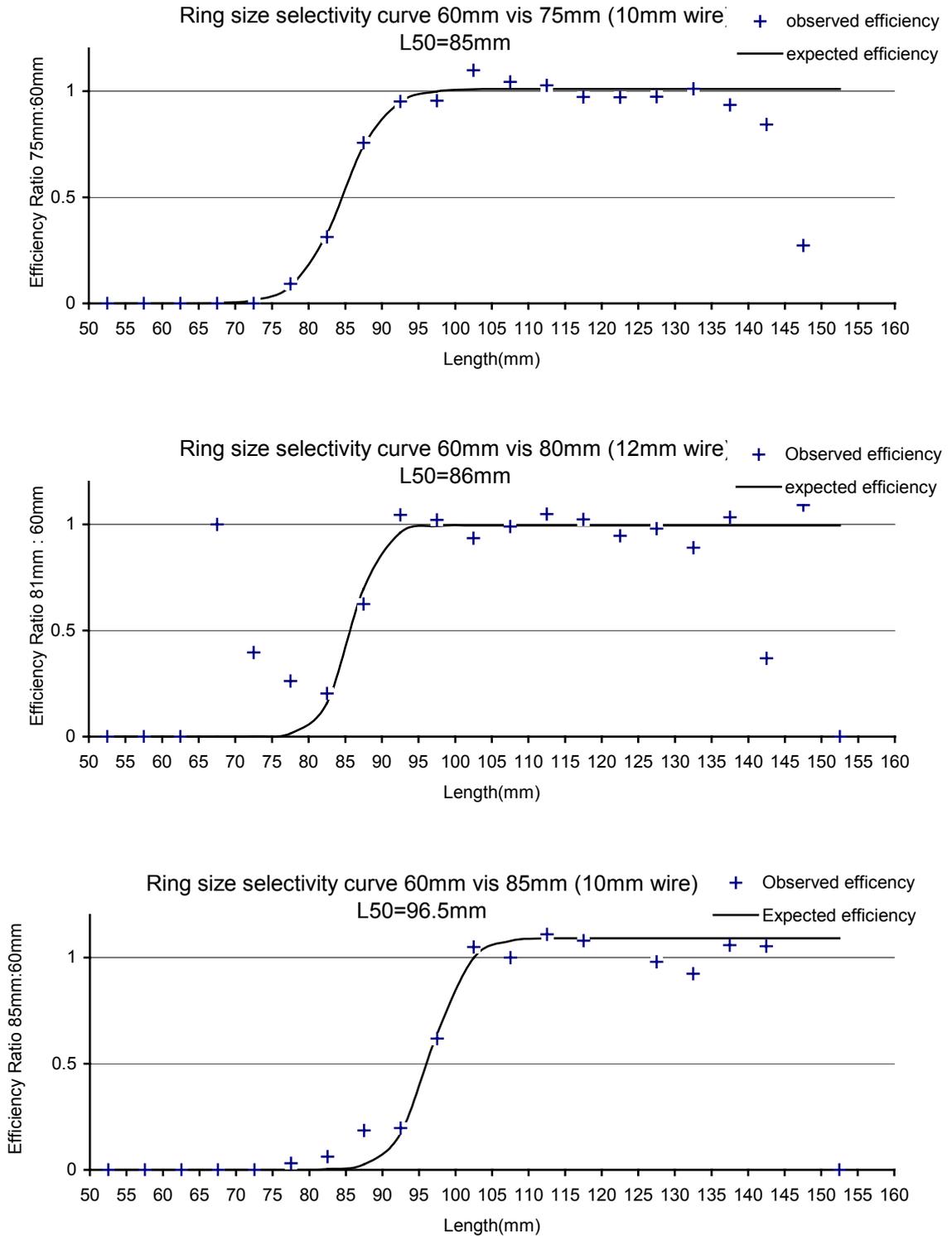


Figure 12 New dredge bellies: Selectivity curves, Star of Anann

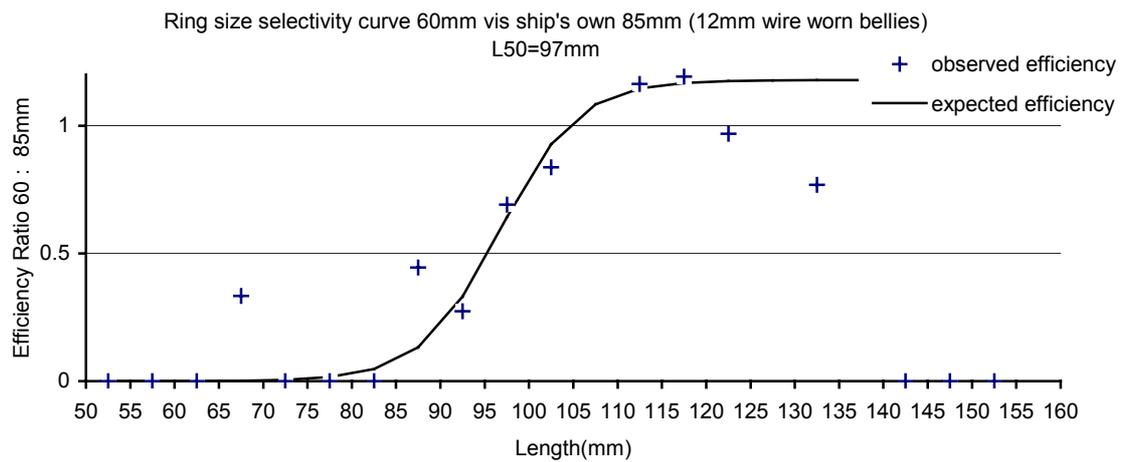
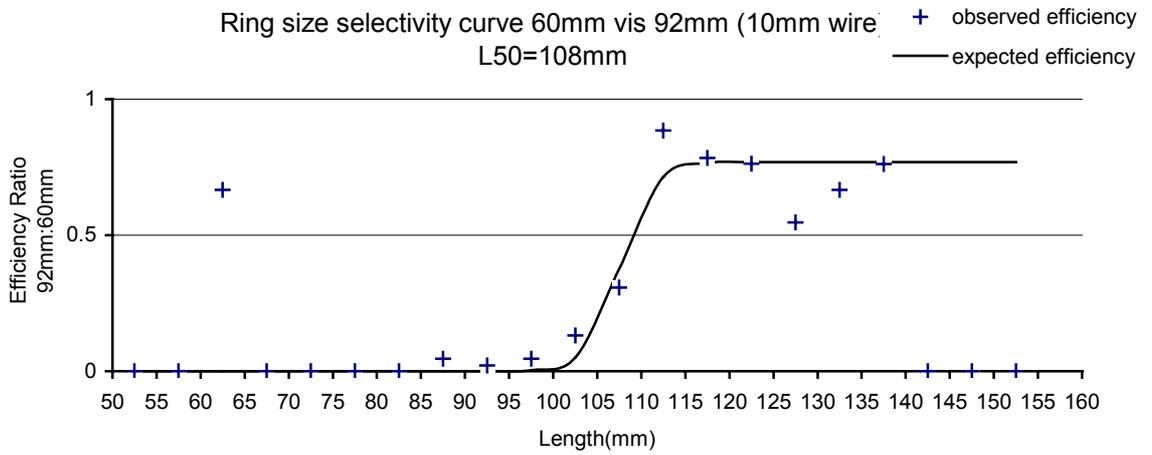
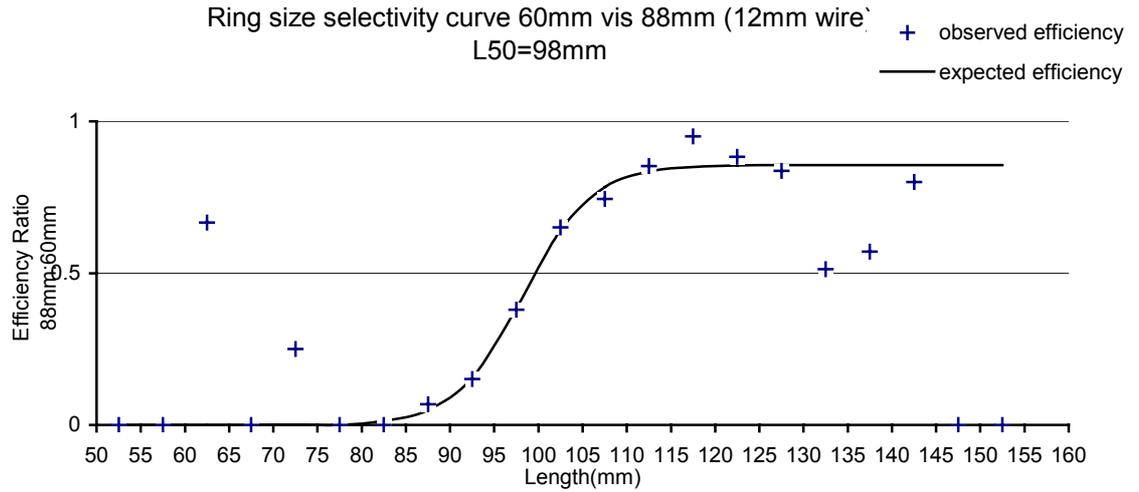


Figure 13 New dredge bellies: Selectivity curves, Alena

Table 7 Selectivity parameters: New bellies relative to 60mm id bellies

Ring dimensions		Estimates of L ₅₀ , L ₂₅ & L ₇₅ (mm)					Selection factor	Maximum asymptotic efficiency			α		
Ring id	Wire od	L ₅₀	SD	CV %	L ₂₅	L ₇₅		Estimate	SD	CV %	Estimate	SD	CV %
75	10	84.6	5.1	6	78	91	0.89	1.0	0.2	15	0.35	0.5	152
85	10	96.6	2.9	3	91	102	0.88	1.1	0.1	10	0.41	0.4	104
92	10	107.6	3.8	4	103	112	0.90	0.8	0.2	22	0.52	1.1	219
80	12	85.8	3.3	4	81	90	0.93	1.0	0.1	12	0.50	0.7	134
85	12	96.7	4.2	4	87	106	0.88	1.2	0.2	14	0.22	0.2	81
88	12	98.3	5.2	5	90	107	0.86	0.9	0.1	17	0.26	0.3	117

Ring size; changes with wear

This reports two experiments; one on the Star of Annan where all the experimental dredges were left for the following 3 months and one on the De Bounty 85mm id rings were compared over the course of a season with the vessel's usual 78mm id rings. Those used on the Star of Annan were case hardened by the normal process whilst those on the De Bounty were through hardened.

The results are presented in terms of the changes in catches over the period of use (Figure 14 and 15) and the changes in the longest dimensions of the rings Figure 16 and 17. Instead of referencing the results to the population dredges used in the first experiment, curves were fitted for the Star of Annan results relative to the 75mm id ring, which was the normal ring size used in the fishery; see Table 8.

In the Star of Annan and Seafalke ([Section 3.2.2.3](#)) results there is clear evidence of changes in the selection characteristics of the ring sizes as they age. In the Star of Annan results, for the first observation, after 40 days fishing the length-frequency distribution of the catch in the 85mm id (10mm wire) ring had lost some smaller marketable scallops (between 100 and 110mm) whilst the other two ring sizes showed similar length frequency distributions as they did in the first voyage. By the final observation at 60days the 85mm id rings were no longer considered usable but the catches of the 75mm (10mm wire) and 80mm rings had changed relative to each other with the 80mm rings becoming more selective. The Seafalke results ([Section 3.2.2.3](#)) show a similar pattern of change. The measurements of the ring sizes over the period of the experiment shows, as would be expected the 80mm rings to have the most stable dimensions over time. It seems possible that the change in selectivity is due to the wear on the washers between the rings resulting in larger apertures between the rings, with less distortion due to changes in the ring shape.

In the comparison on De Bounty operating in the Irish Sea there is less change in the relative catches of the two ring sizes and less relative distortion of the rings. Because the length-frequency distributions are so close together it is not possible to fit a curve to compare the two ring sizes. However there is some loss of scallops between 100 and 110mm from the 85mm rings on the second observation, which is a similar pattern to above. Catches of scallops larger than the MLS were similar for both ring sizes with slightly larger catch in the 85mm rings on both occasions.

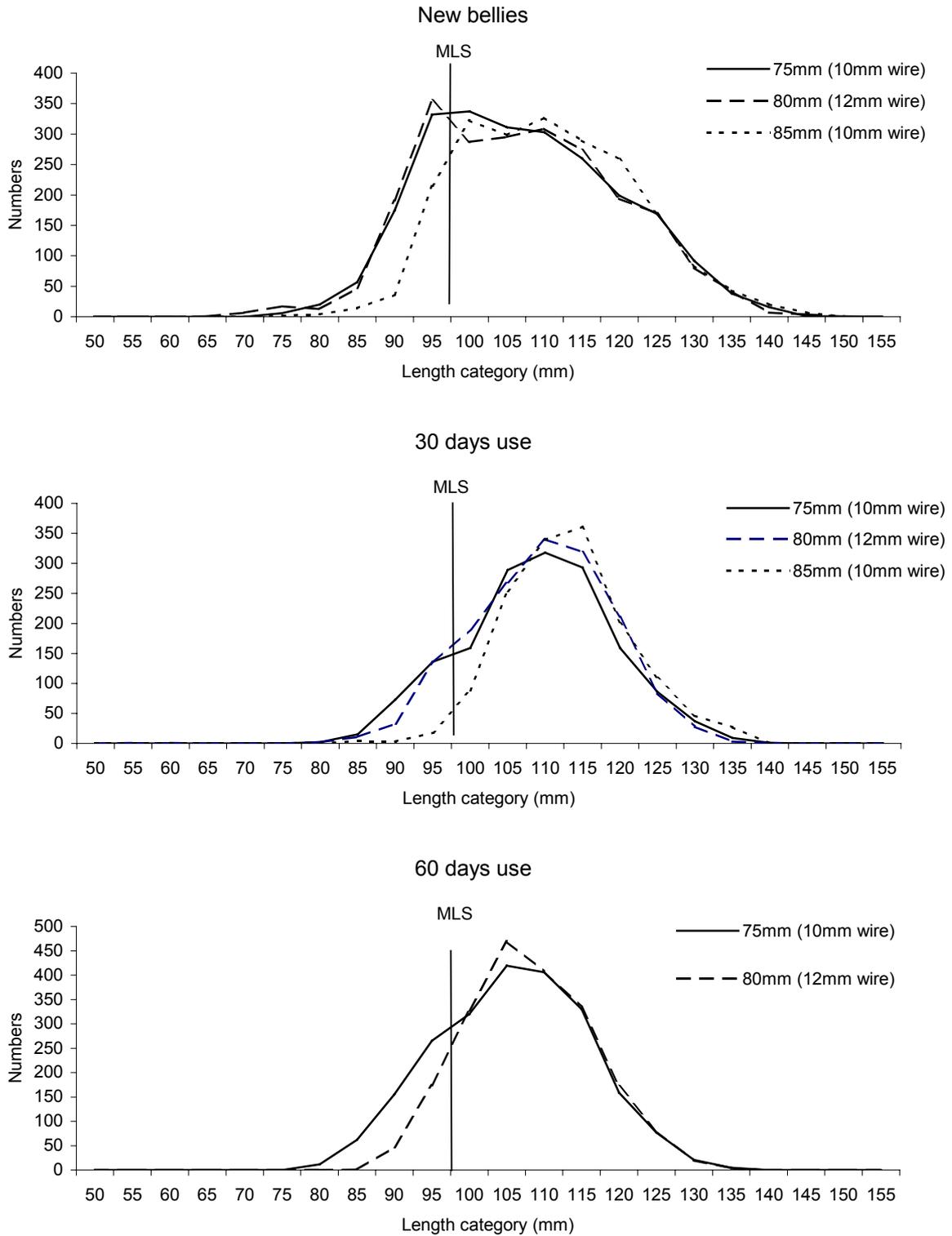


Figure 14 Changes with wear on bellies: Aggregate length frequencies Star of Annan using normal case hardened rings

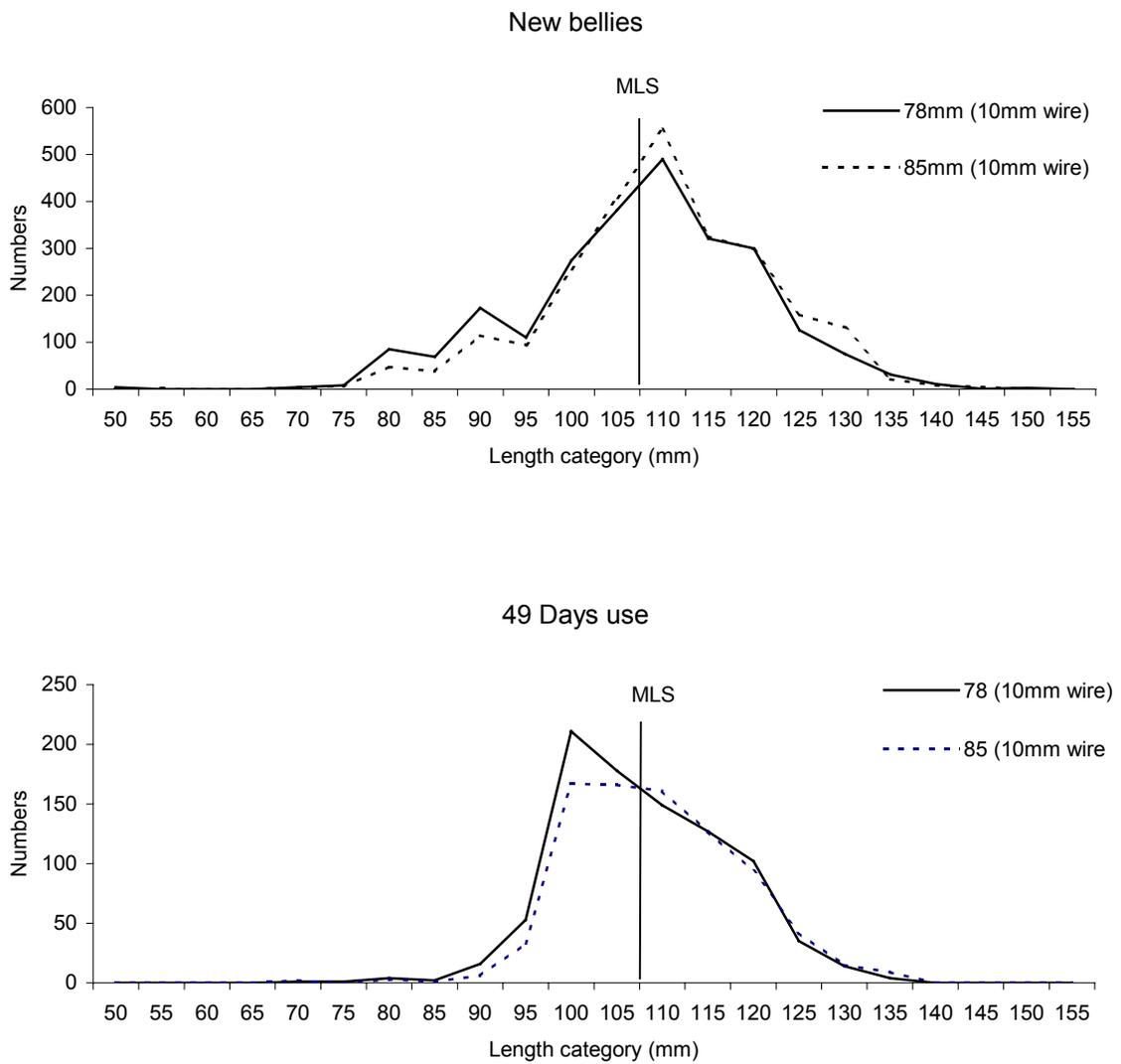


Figure 15 Changes with wear on bellies: Aggregate length frequencies: De Bounty using through hardened rings

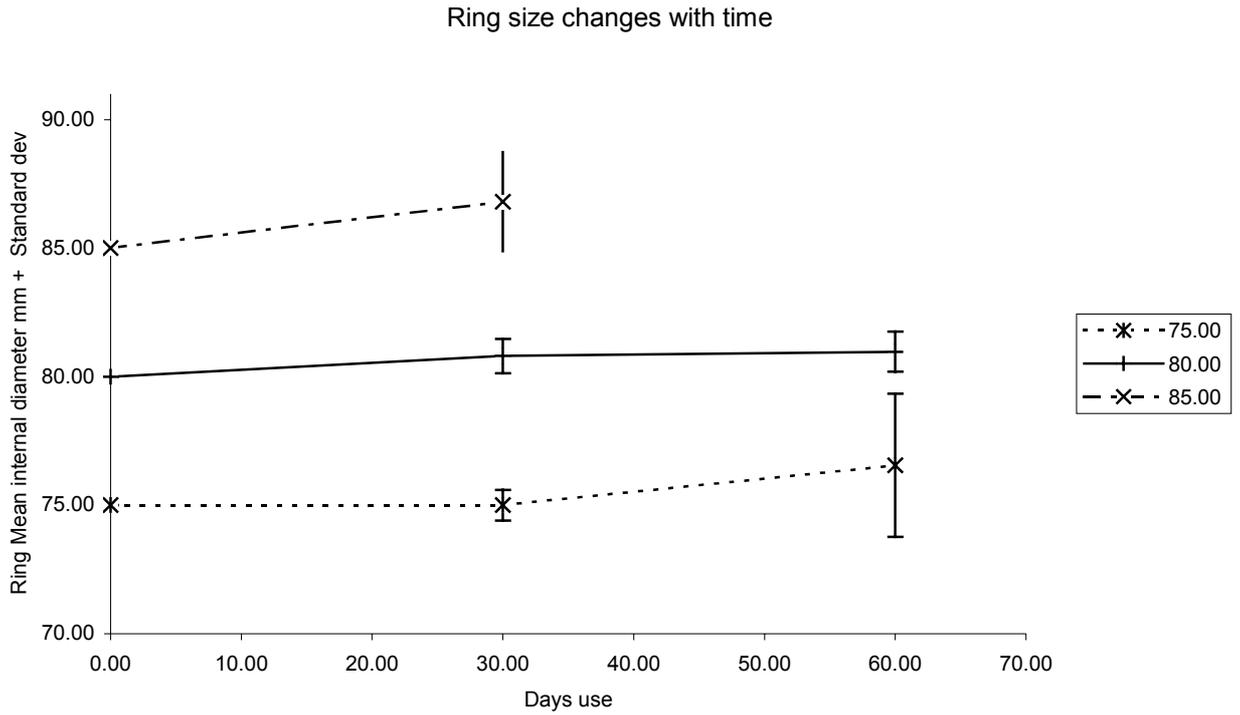


Figure 16 Changes with wear on bellies: Longest internal measurement of belly rings Star of Annan

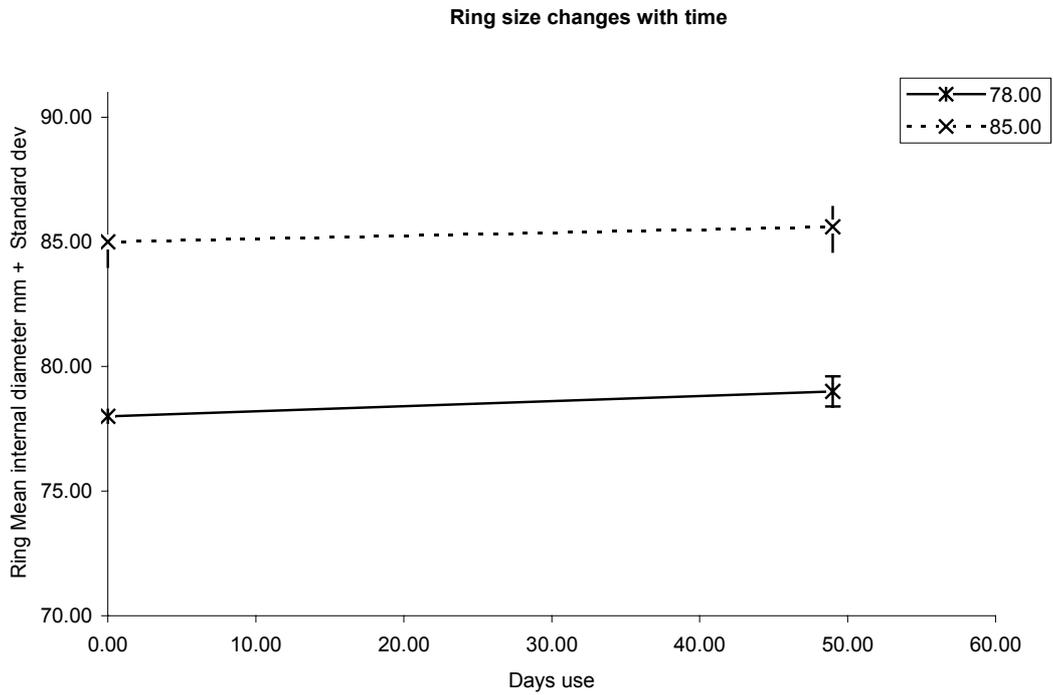


Figure 17 Changes with wear on bellies: Longest internal measurement of belly rings, De Bounty

Table 8 Selectivity parameters; Changes relative to 75mm id bellies with time

Observation	Ring dimensions		L ₅₀			Maximum assymtotic efficiency			α		
	Ring id	Wire od	Estimate	SD	CV %	Estimate	SD	CV %	Estimate	SD	CV %
1	80	12	37.5	2.5E+13	6.8E+13	1.00	0.28	28	0.67	4.521E+11	6.768E+13
1	85	10	94.0	18.1	19	1.13	0.57	50	0.23	0.8	359
2	80	12	88.9	3.7	4.2	1.09	0.13	12.3	0.31	0.31	100
2	85	10	102.9	6.9	6.7	1.40	0.31	22	0.17	0.17	100
3	80	12	93.3	0.8	0.8	1.06	0.03	3.2	0.39	0.10	26

Catches of larger scallops

Rudders et al. (2001) examined selectivity of dredges targeting *Placopecten magellanicus* and found that there were not only less small scallops caught in larger ring sizes but there was increased efficiency on the larger scallops. The results from both the Star of Annan and De Bounty (but not Alena) trials tend to support this effect in *Pecten* dredges with length distribution in the larger ring sizes being shifted slightly towards larger scallops. To examine this effect, the catch in the 85mm rings expressed as a percentage of the total catch was calculated by haul for all the hauls (both new and worn rings: a total of 73 hauls) on the Star of Annan and De Bounty was calculated thus.

$$\% \text{ Catch in 85mm rings} = \frac{\text{Catch in 85mm ringed bellies}}{\text{Catch in 85mm rings} + \text{Catch in normal rings}} \times 100\%$$

For the Star of Annan hauls the normal rings used were 75mm; for De Bounty they were 78mm.

If the dredges with the different ring sizes exhibited equal fishing power then 50% of scallops would be caught in both dredges; if one dredge with one ring size was more efficient than the other for a given length then the results would be skewed correspondingly. Figure 18 shows this comparison for the 85mm rings compared with normal rings, for all the Star of Annan and De Bounty catches. The relative differences in efficiency due to selectivity can clearly be seen on this graph, and there is a trend towards increased efficiency for scallops of larger than 120mm in the larger size groupings. However, analysis of variance (ANOVA) comparing the catch per haul of scallops of 120mm or larger did not find a significant difference between the two ring sizes.

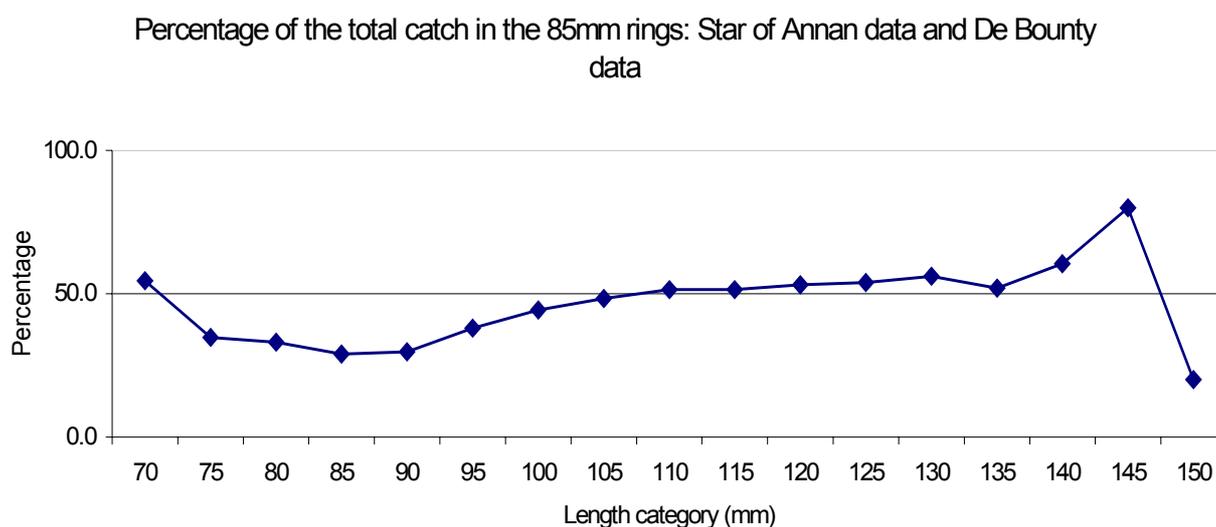


Figure 18 Percentage of catch in the larger (85mm rings). by length in 73 hauls .

Discussion

Experimental design

The experimental design used in these trials, where factors affecting selectivity are compared directly between dredges of different configurations on the same bar has advantages and risks. It enables comparison directly between experimental configurations as well as with the standard or population dredges and it enables an efficient approach to the factors that can be examined since two factors and interactions can be examined at the same time ([Section 3.2.2.3](#)). The main risks are of interference between the dredges. Two possible types of interference are a) in terms of catch, which should be eliminated by putting a space between dredges and b) mechanically with unequal drag forces drag along the bar from the different variants. Efforts were made, during the Star of Annan trial, to compare *in situ* tensions at the dredge shackle for dredges with 2 different ring sizes using the load cells and data loggers described in [Section 3.1.1.1](#), but conditions proved to be harsh for the sensors. Precautions were taken against this effect were to make differences between variants along the bar small and to only vary tooth spacing on the middle two dredges where mechanical leverage along the bar is non-existent. With experience, Skippers (and eventually, scientists) can detect faults in the operation of the gear through feeling vibrations on the wire and through the vessel. There were none of these problems observed and the Skippers involved were able to use the gear commercially on a sustained basis for the wear trials. This suggests that there was no mechanical interference between the dredges. The precautions taken against inherent differences between locations on the bars were to move the dredges along and between the bars during the course of the experiment, in a balanced design. Although differences were found between these factors, they were much smaller than those found between ring size or tooth spacing and were probably related mostly to spatial variations in the distribution of scallops.

Selectivity Characteristics

The results reported in this study compare the relative efficiency of the various ring sizes or tooth spacing with each other or to a standard. Other authors (see [Review Section 6](#) and Fifas and Vigneau (in prep)) have investigated absolute efficiency of dredges referring the catches in the dredges to the population defined by diver survey or pre-seeded tagged scallops. These studies suggest that the absolute efficiency of the gear increases to a maximum and then declines with size. From a commercial perspective the relative efficiencies are relevant, for population surveys the absolute efficiencies may be more relevant. In this context the results for the very small tooth spacing (17teeth/bar and 32mm between teeth) are of interest, because they show that there are more smaller scallops (of between 55 and 85 mm length) available for capture by the gear if 17teeth/bar are used instead of 10teeth/bar. There also appears to be a fall off in the efficiency of capture of larger scallops in queenie dredges using 17teeth/bar. Thus surveys aimed at catching a cross section of the scallop population should perhaps use a range of tooth spacing.

These results are compared with those obtained for French depressor plate dredges in Table 9. These results are based on Table 5-1 in the review but allowance is made for the measured dimension in this Table being shell height and the L_{50} is interpolated from the nearest ring size using the selection factors quoted in Table 5-1. There is good agreement between the two sets of results.

Table 9 Comparison between L₅₀s obtained in this study (new bellies with 10mm wire) and L₅₀s obtained for French depressor plate gear [See Review Section 6](#)

Ring size (10mm wire: new bellies)	Estimated L ₅₀ (Scallop Length in mm)	
	Sprung toothed dredges	Depressor plate dredge
75	85	83
85	97	97
95	108	108

Changes in selectivity with belly wear

In all four studies of changes in selectivity with belly wear (August Rose and Seafalke [Section 3.2.2.3](#) Star of Annan and De Bounty this Section) the effects were very similar. As the dredge bellies wear the ascending limb of the length frequency curve moves to the right showing an improvement in selectivity. Relative to the 75mm rings the 85mm rings with normal hardening did not appear robust enough for commercial practice but the through hardened ones were more robust and are still in use at time of writing (Nov 2002). The 80mm id 12mm wire rings were relatively unstretched compared with the 75mm id rings and also appear to be more selective the ascending limb of the length frequency curve moving too the right of the 75mm results on the final observation.

Table 10 below shows the relative tensile strength of the bellies relative to 75mm bellies based on stress calculations and allowing for the different numbers of rings across the different bellies. For new 75mm rings (10mm wire) the force required to bend them is a load of around 200kgf per ring. This compares with the load on the dredge at the point of attachment to the bar of 100-200kgf ([Section 3.1.1.1](#)). If all the tension from one or two dredges came on one ring (as when an obstruction was encountered) then new rings might be bent but for normal circumstances they are well in excess of requirements. Thus the elongation of the rings is most likely a consequence of the rings losing material around the points where the washers abrade them. This will not only result in reduced thickness at this point and hence a loss of strength but also a concentration of the stress at these points in the rings.

Table 10 Relative strength of bellies (10mm wire 9 rings across) to a 75mm ring (10mm wire belly 10 rings across) based on Warnock and Benham (1965)

Ring internal diameter mm	Wire thickness	Relative Strength %
80	12	141
85	10	81
88	12	132
92	10	76

Thus the key to avoiding the rings becoming elongated is avoiding the grooving effect. Hence the success of the hardened rings. However, the ratios of tensile strength will remain the same between the different ring sizes for the same material since this is a function of geometry and not material. If larger hardened rings were contemplated to improve selectivity the condition of failure would be an important factor. If both large and small rings failed because of a lack of tensile strength then the smaller ring size would be expected to exhibit improved durability. However, if the source of failure was due to abrasion of the washers, then improving the hardness of both rings and washers and using a larger ring size might improve the selectivity of the gear on a sustained basis.

The wear results from the 80mm 12mm wire rings compared with the 75mm rings are of interest in this context. Here there almost no change in the internal diameter of the 80mm rings over the period of use but towards the end of the experiment they become more selective in relation to the 75mm rings, which had been significantly elongated. Thicker, heavier rings would not necessarily be the best solution since it would increase the overall weight of the gear and would not be acceptable to all fishers. However, these results suggest that if ring integrity can be maintained there may be advantages in changing the shape of components, for example thicker washers shaped to avoid turning and hence possibly reduce abrasion (See [Section 3.2.2.5](#)).

3.2.2.5 SPRING-TOOTHED SCALLOP (*PECTEN MAXIMUS*) DREDGES: WEAR PATTERNS ON BELLY RINGS

Introduction

The king scallop, *Pecten maximus*, is targeted using a ‘Newhaven’ spring scallop dredge, Figure 1, whose ‘belly’ (lower panel of the collecting bag) comprises a matrix of steel rings and washers, Figure 2. The selectivity of the dredge has been investigated in [Section 3.2.2.3](#), [3.2.2.4](#) and Lart et al. (1997) where studies conducted on new dredges showed that selectivity was related to ring size and tooth spacing. Inspection of worn dredges showed changes in the ring matrix over time, and trials showed that the selectivity characteristics of worn dredges were different ([Section 3.2.2.3-4](#)); the main effect being differences in catch composition in terms of animal size and numbers.

Since *consideration* of technical measures designed to improve belly selectivity requires an understanding of wear patterns the aim of the research was to investigate wear patterns of the ring of the belly to gain an insight into the mechanisms that influence such changes in selectivity.

Objectives

The objectives were to:

- i) describe typical wear patterns of belly components (rings and washers);
- ii) investigate changes in wear over time;
- iii) investigate the location of greatest wear within a belly

Method

The research was conducted in two phases; the first phase involved identifying typical wear characteristics and the second phase involved measuring specific dimensions of belly components (ring and washers).

Wear measurement

Data on typical wear characteristics was obtained by discussion with fishermen and direct inspection of ring bellies deployed for different periods at sea. Data on unworn bellies was obtained by measuring new bellies obtained from the factory.

More detailed information was obtained by mapping a fully worn belly used in Scotland (MFV Kelly) and both partly worn and fully worn bellies used on the south coast of England (MFV Seafalke) as used in the trials described in [Section 3.2.2.3](#).

Measurement of belly components

Each component (ring/washer) of the belly was allocated a position according to a grid layout, Figure 3. Columns of rings are denoted numerically 1 to 10; column 1 and 10 referring to the sides, or ‘turn-ups’, of the belly and columns 2 to 9 denoting the flat ‘belly’ section. Washers were categorised according to their orientation within the belly matrix; longitudinal washers, which connect columnar rings, and horizontal washers, which connect rings within rows, Figure 4. The Longest Internal Dimension

(LID) of each component was recorded using an electronic Vernier caliper, as was the presence of grooving.

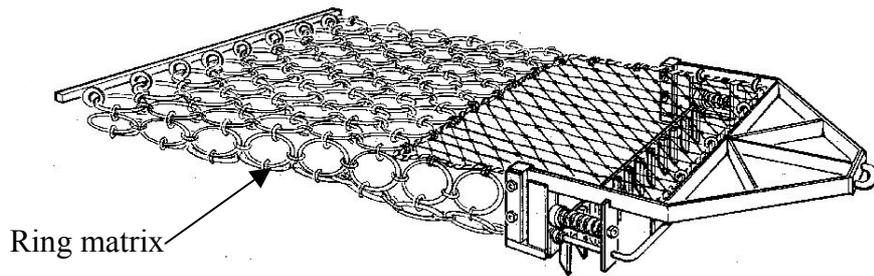


Figure 1 Newhaven sprung scallop dredge

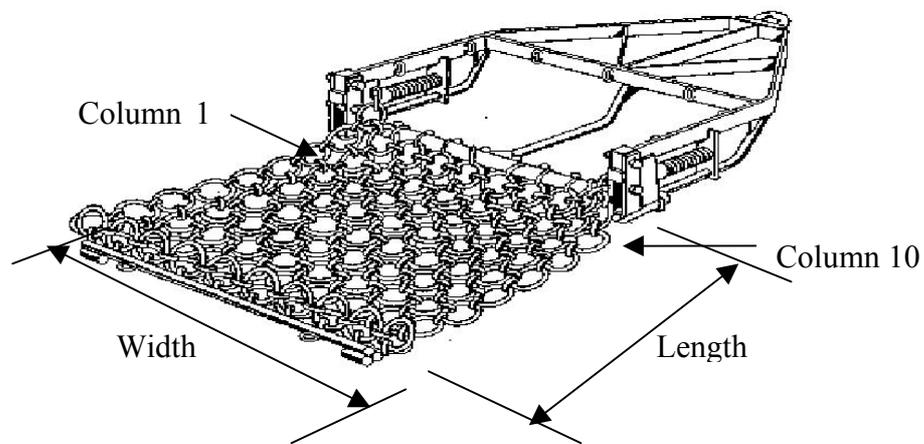


Figure 2 Belly ring matrix of the lower section of the collecting bag fabricated from steel rings and washers. A typical belly is 10 rings wide by 9-12 rings long. Rings denoted by columns 1 and 10 comprise the sides of the belly ('turn-ups'), and the section that comprises the rings from column 2 to 9 is known as the belly.

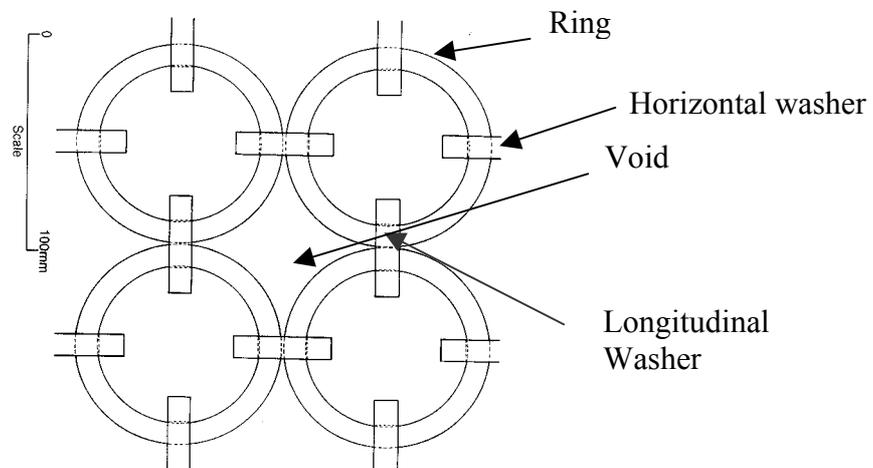


Figure 3 Section of belly ring matrix showing arrangement of rings and washers (longitudinal and horizontal), and void (space between rings).

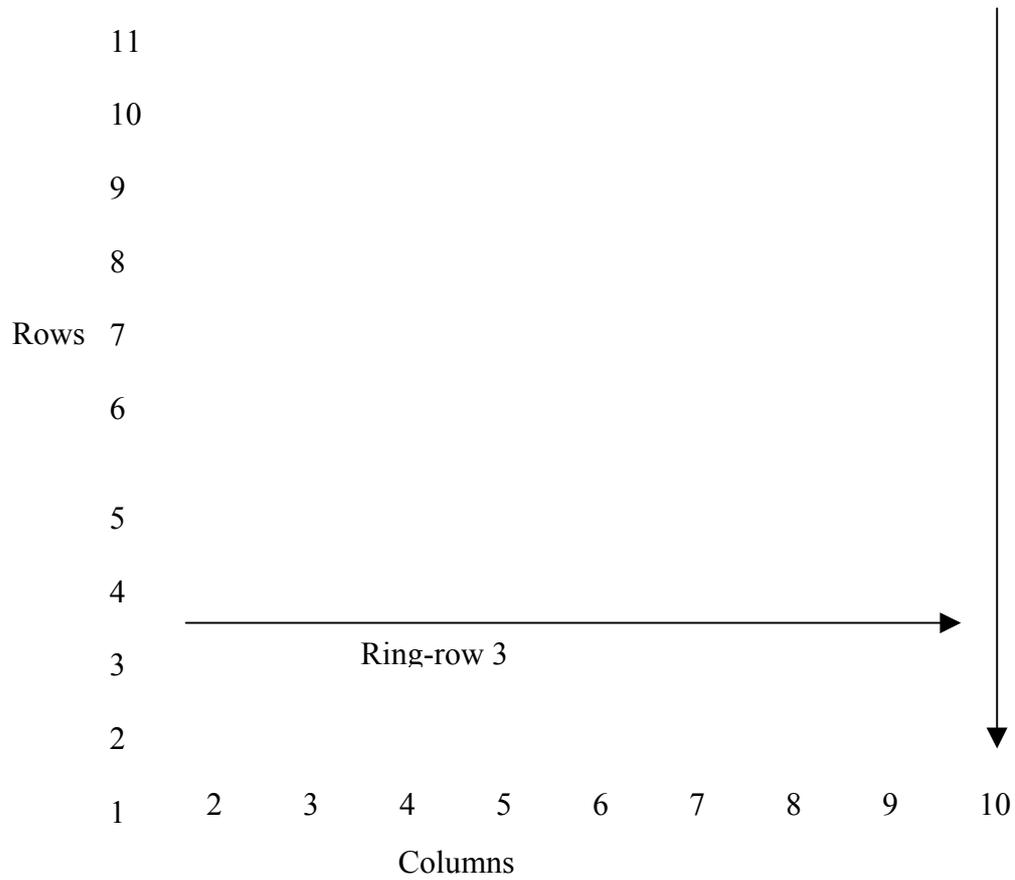


Figure 4 Grid layout for allocating locations of compartments

Component location is described by grid reference that refers to the columns and rows of rings of the matrix. Thus 'Ring-column 6' and 'Ring- row 4' refers to the rings that comprise column 6 and row 4 respectively. The location of a washer is described by the location of the ring to which it is attached posteriorly or dextrally (ie to the lower or right hand side of a ring). The orientation of a washer is described as 'longitudinal' or 'horizontal'; the former connecting rings in a column, the latter connecting rings in a row. Row 1 denotes the row of rings adjacent the tooth bar located at the front of the belly.

Results

Wear characteristics; belly rings

Typical wear can be characterised by three types: non-specific erosion, point erosion and elongation.

Non-specific erosion describes a reduction in diameter along the length of the steel rod from which the ring has been fabricated, Figure 5, causing the internal diameter to increase and the external diameter to decrease. Surfaces are smooth and exhibit no signs of pitting or indentation

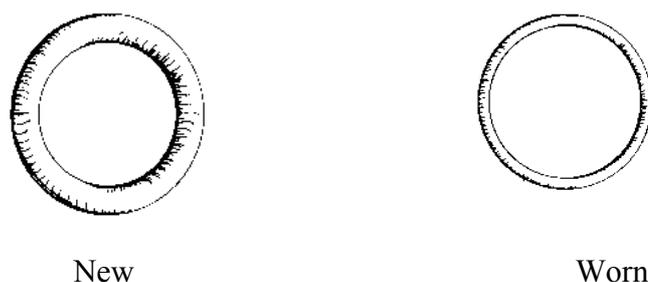


Figure 5 General erosion of a worn ring, showing reduction of rod diameter, as compared with a new ring.

Point erosion describes loss of material at specific locations on the inner circumference with little reduction in the internal diameter but considerable increase in the LID, Figure 6.



Figure 6 Point erosion on inner circumference of a worn ring, as compared with a new ring.

Elongation describes ring deformation where the symmetry has changed from annular to oval, Figure 7.



Figure 7 Elongation differences between new and worn rings showing change of symmetry

All three conditions were exhibited by rings within the matrix, whereas washers exhibited only erosion and grooving. The absence elongation was due, probably, to the washer's more robust construction (small ratio of internal diameter to rod diameter) compared with that of the ring (larger ratio of internal diameter to rod diameter).

Wear characteristics: washers

As with the rings two characteristic wear patterns were observed. Figure 8 shows a new washer compared with one with non specific wear and Figure 9 shows notching effects.

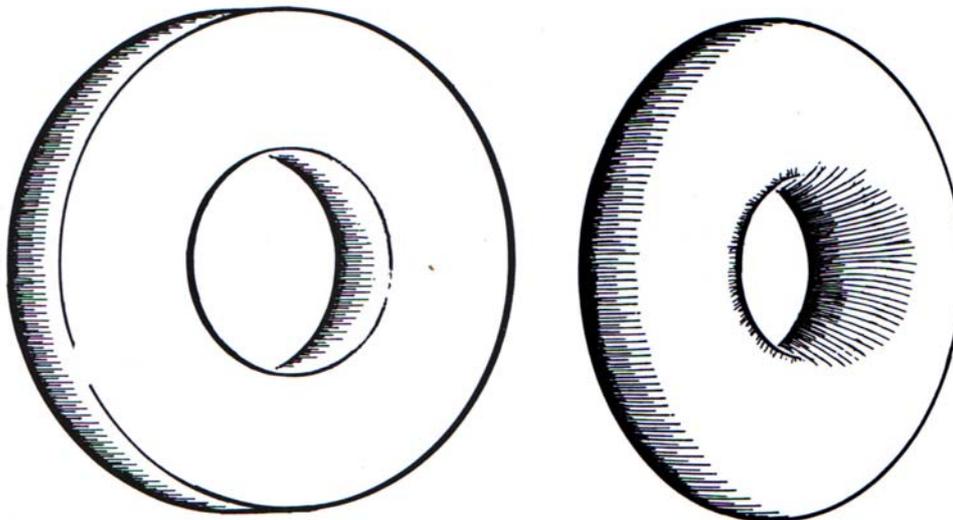


Figure 8 New washer compared with washer with general wear. Size when new 45mm outside diameter 22mm internal diameter thickness 9mm

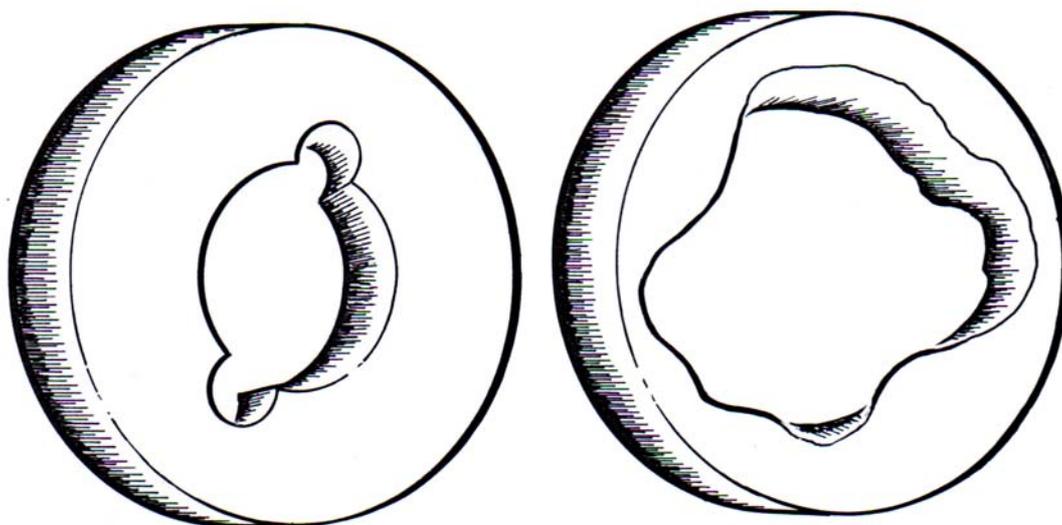


Figure 9 Notched and severely notched washers

Belly wear

The results are presented graphically and describe wear characteristics, in terms of the Longest Internal Dimension (LID), of belly components used in different locations. Patterns of wear are described using contour plots that describe the location, and magnitude of wear in terms of level categories; however, contour plots cannot be compared directly due to variation in the categories' ranges. The magnitude of wear is described precisely by line graphs that show the mean LIDs for components (ring and washer) comprising each column and row of a matrix.

Scotland:

Rings

The results show that wear occurred unevenly throughout the belly (fully worn) and was most pronounced towards the front and rear middle section, Figure 10. Least wear was exhibited by rings in the sides ('turn-ups), front and rear of the belly. Variation in the amount of wear across the width of the belly (79-91mm) was greater than that along the length (87-91mm), Figures 11 and 12, which show the mean LIDs for rings in each column and row. Point erosion was exhibited by all rings, except for those at the corners of the belly, Figure 13, and occurred mainly at the upper and lower loci of each ring's inner circumference, Figure 6.

Washers

Wear patterns for both washer categories (longitudinal and horizontal) were uneven but similar since greatest wear occurred down the middle and sides (columns 1 and 9) of the belly, Figure 14. However, the longitudinal washers exhibited greater wear (26.5-30mm), compared with that of horizontal washers (23-24.5mm), whereas variation in wear across the width and along the length of the belly was similar for each category (approximately 3 mm), Figure 15. Point erosion was exhibited only by the longitudinal washers, and occurred at the front and along the sides of the belly, Figure 16.

South coast:

Rings

Wear patterns for both the partly worn and fully worn bellies were similar, though the magnitude of wear was greater in the latter. Wear occurred unevenly but was most pronounced towards the central rear section of the belly, whereas least wear occurred at the front and along the sides ('turn-ups) of the belly. Figure 17. Variation in wear across the width and along the length of the belly was similar for each category (approximately mm), Figures 18 and 19. In contrast, point erosion of the two bellies exhibited more differences than similarities, occurring in the central belly section of the partly worn belly, Figure 20, but throughout the belly section of the worn matrix, Figure 21. Point erosion was absent from side/turn-ups of both bellies.

Washers

Wear patterns for both the horizontal and longitudinal fully worn washers were similar, with greatest wear occurring towards the central rear section of the, though much greater in extent in the latter, Figures 22, 23, 24 and 25.

Point erosion occurred in only the longitudinal washers, and was restricted to central front section of the belly, Figure 26. In contrast, both categories of partly worn washers exhibited no point erosion and negligible wear, their dimensions differing little from those of unworn components.

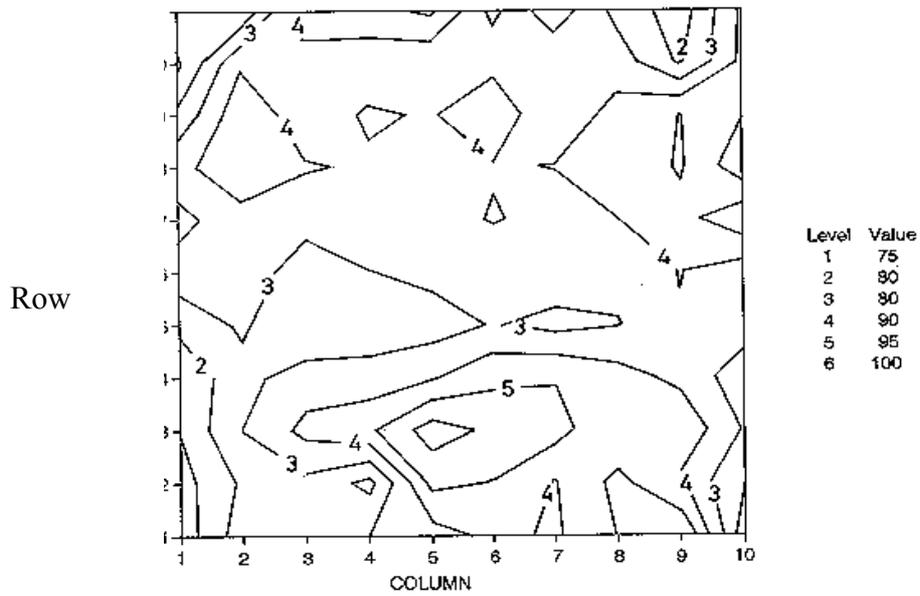


Figure 10 Contour plot showing location of greatest LID of fully worn rings of belly used in Scotland. Row 1 is the first row of rings at the front of dredge.

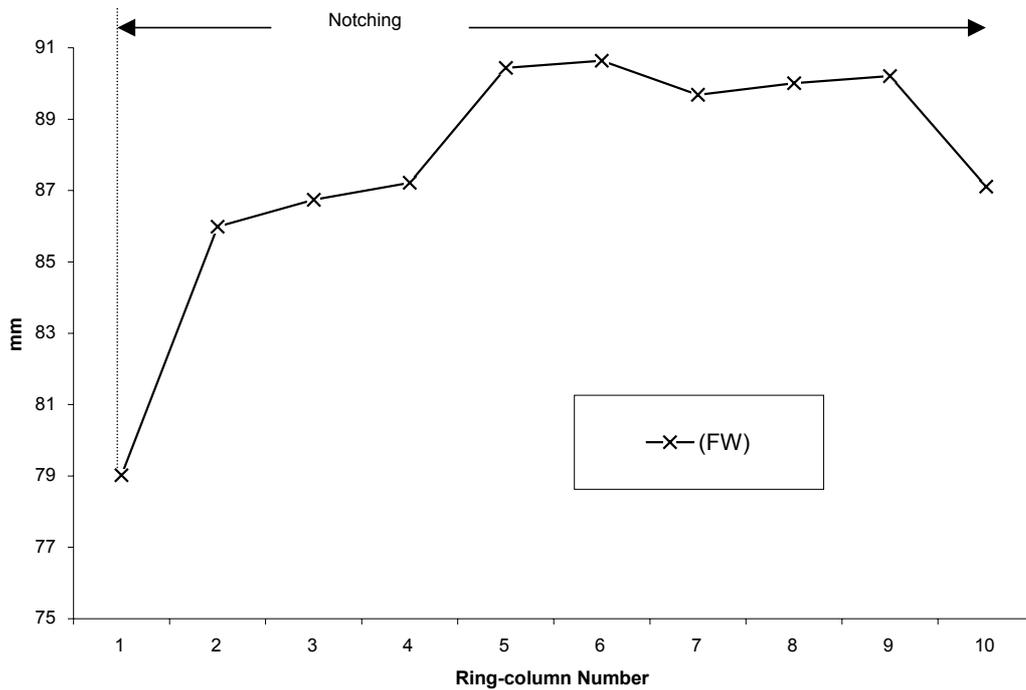


Figure 11 Longest internal dimension of fully worn rings, by column, of belly used in Scotland.

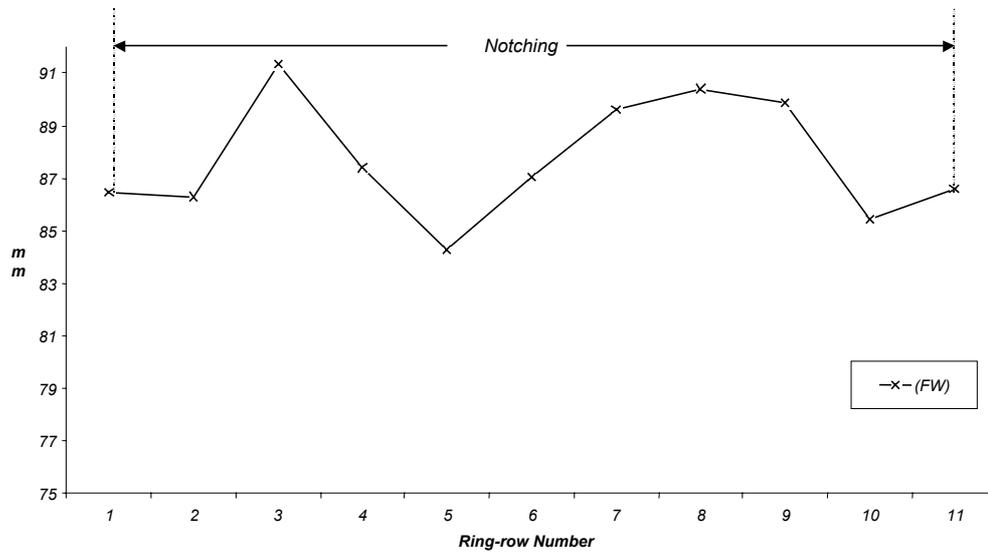


Figure 12 Longest internal dimensions of fully worn rings by row of belly used in Scotland.

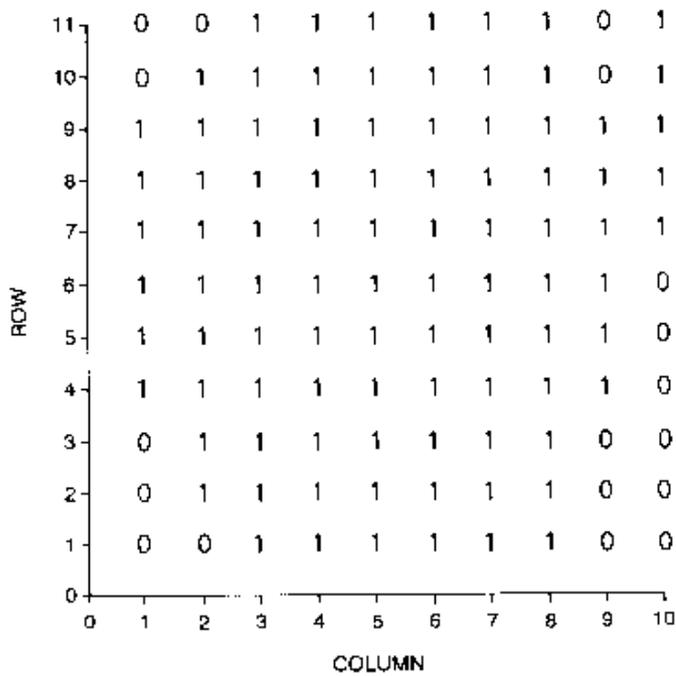


Figure 13 Groove contour plot showing grooving (denoted by 1) in fully worn rings of belly used in Scotland.

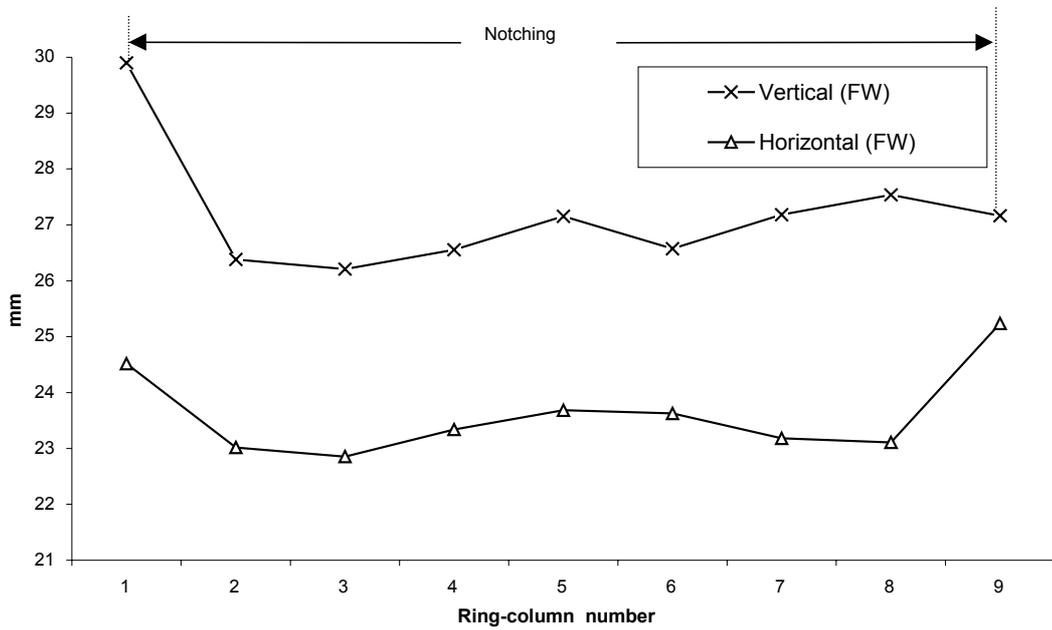


Figure 14 Longest internal dimension of fully worn washers by column of belly used in Scotland.

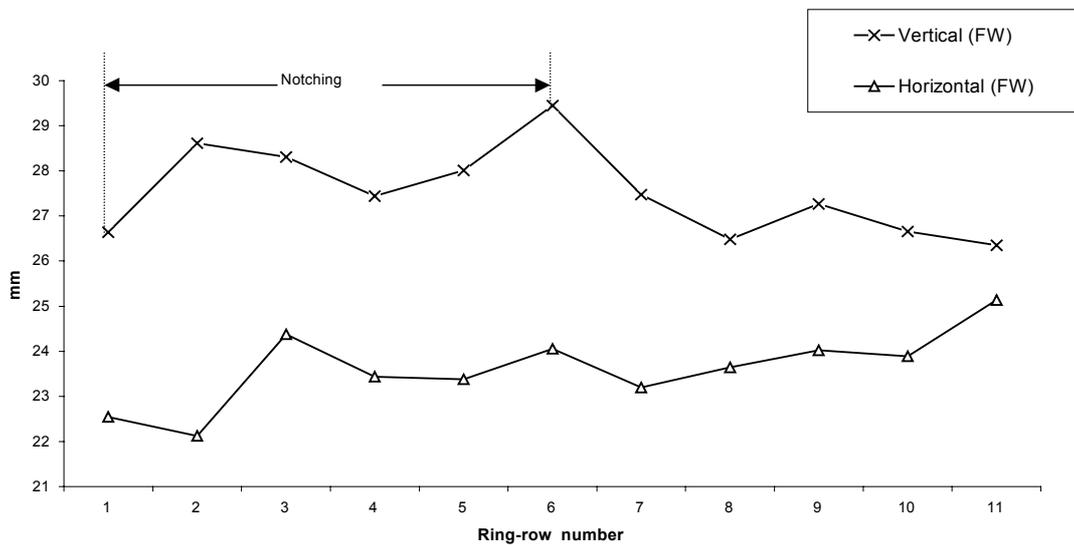


Figure 15 Longest internal dimension of fully worn washers by row of belly used in Scotland.

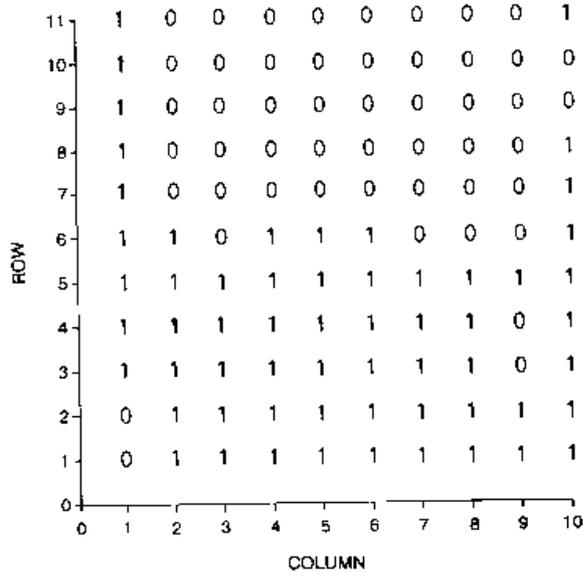


Figure 16 Location of grooving (denoted by 1) in fully worn longitudinal washers of belly used in Scotland

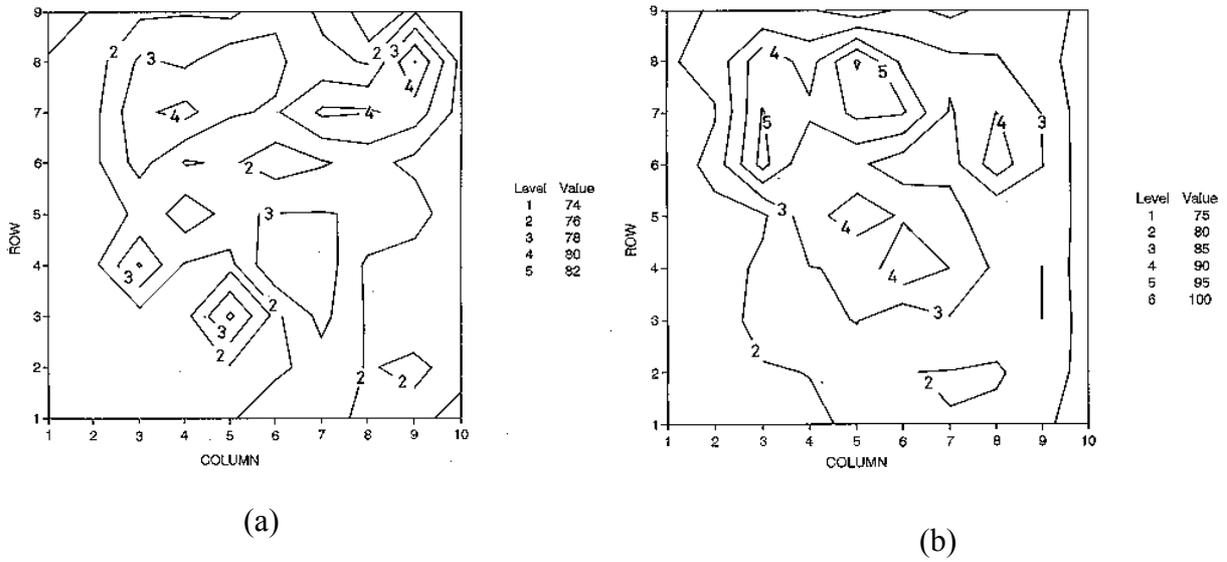


Figure 17 Contour plots showing location of greatest wear in partly worn (a) and fully worn (b) rings of belly used in the South Coast.

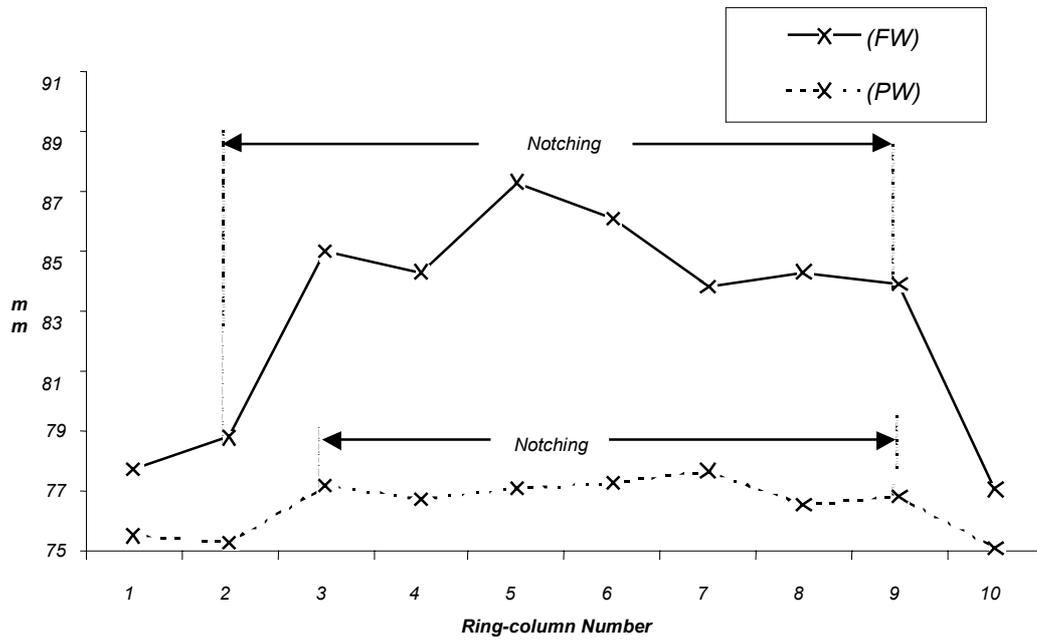


Figure 18 Longest internal dimension of fully worn and partly worn rings, by column, of belly used in the South coast.

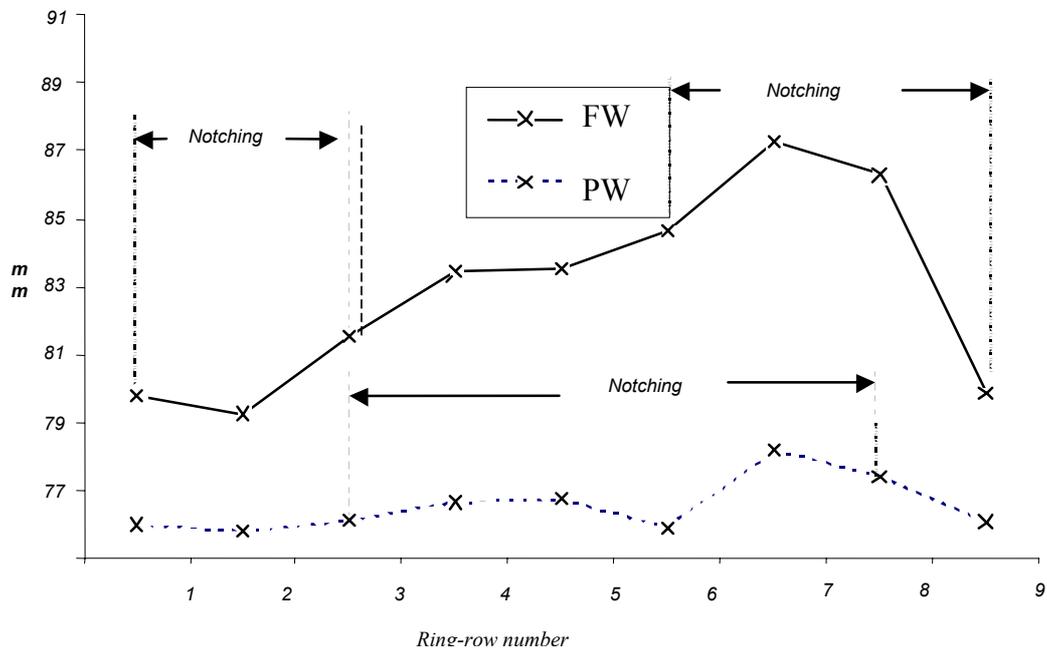


Figure 19 Longest internal dimension of fully worn and partly worn rings, by row, of belly used in the South coast

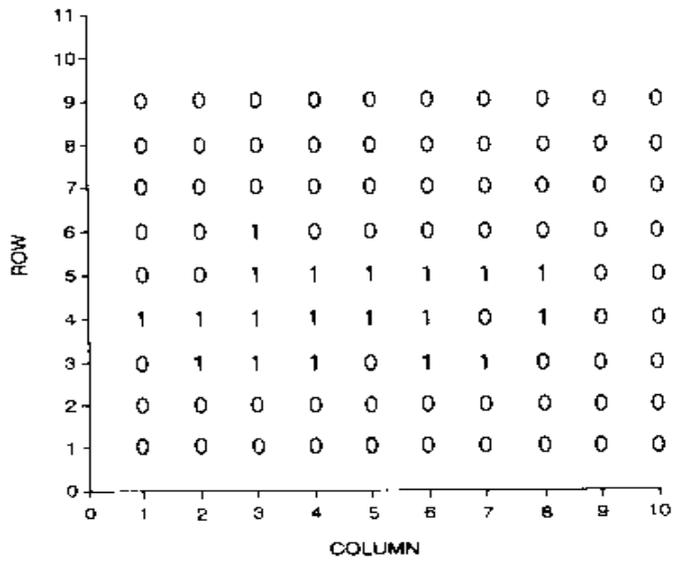


Figure 20 Contour plot showing location of greatest wear in partly worn ring bellies used in the South Coast

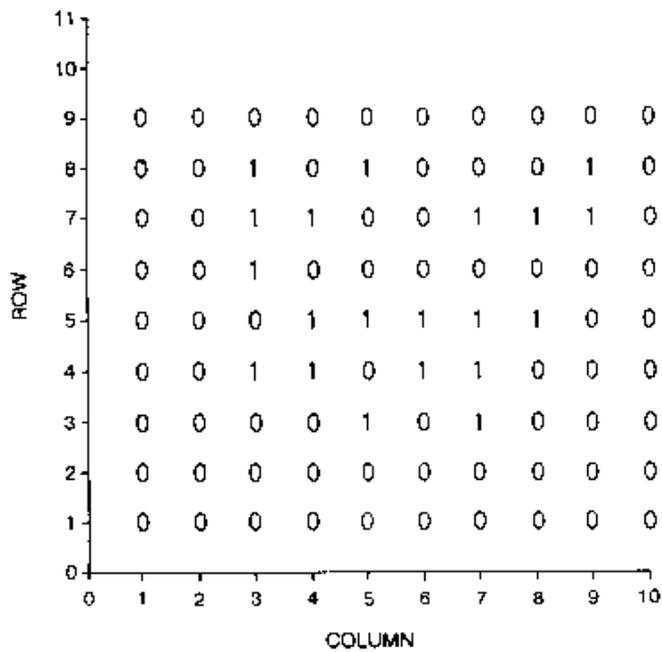


Figure 21 Contour plot showing location of greatest wear in fully worn ring bellies used in the South Coast

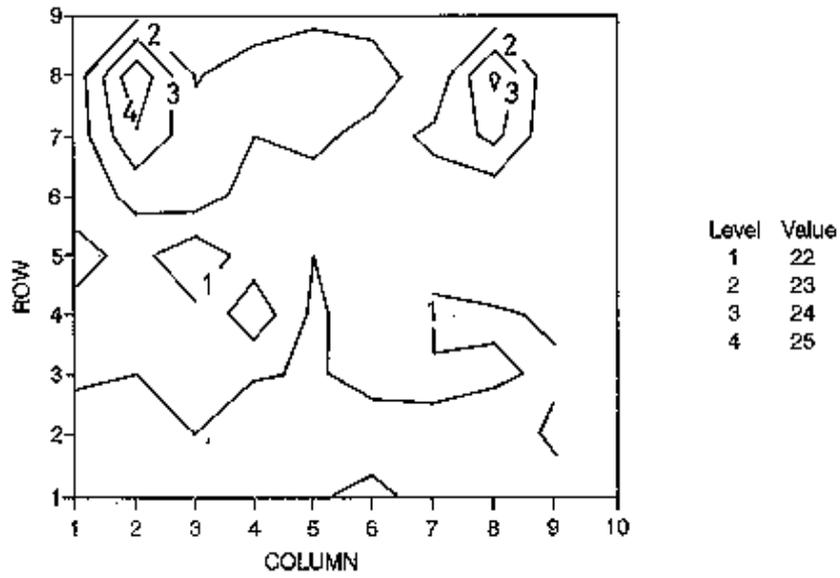


Figure 22 Contour plot showing location of greatest wear in partly worn longitudinal washers of belly used in the South Coast

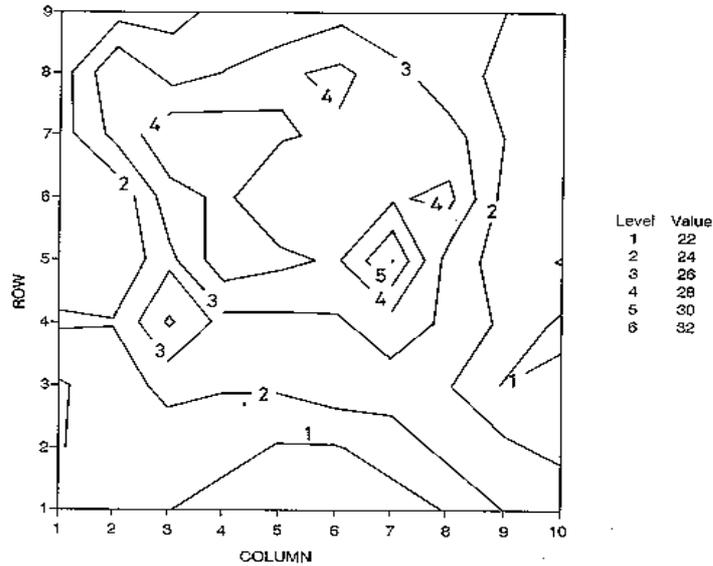


Figure 23 Contour plot showing location of greatest wear in fully worn longitudinal washers of belly used in the South Coast.

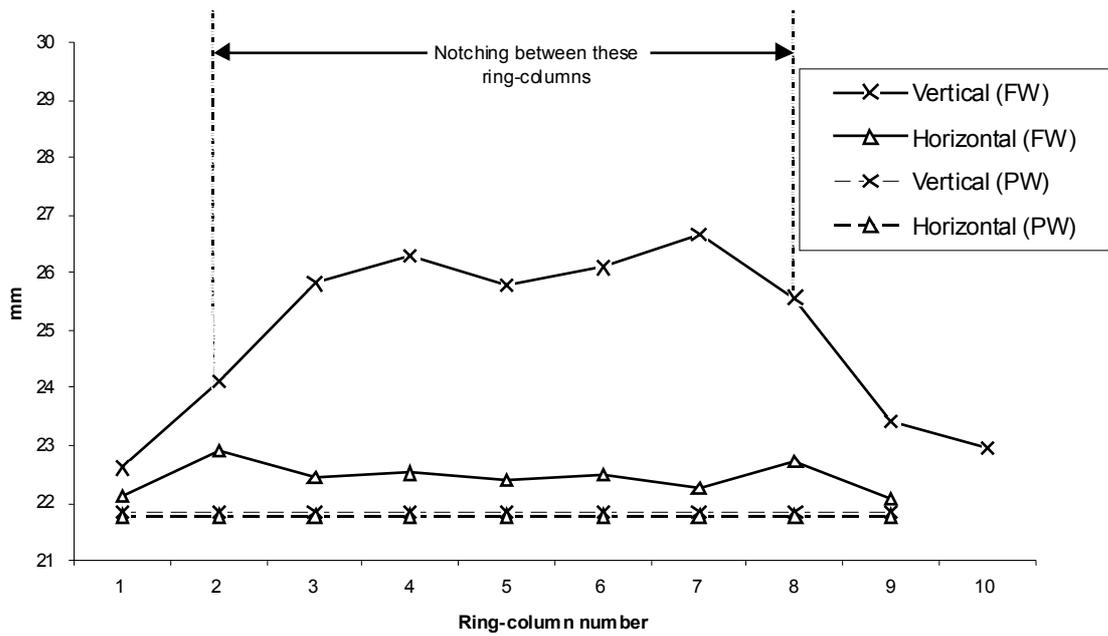


Figure 24 Longest internal dimension of fully and partly worn washers, by column, of belly used in the South Coast.

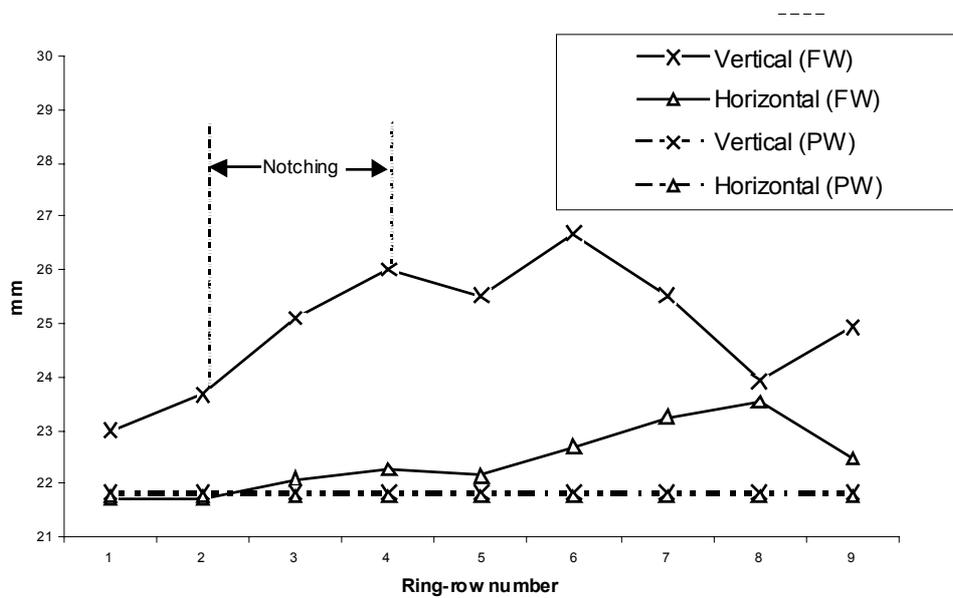
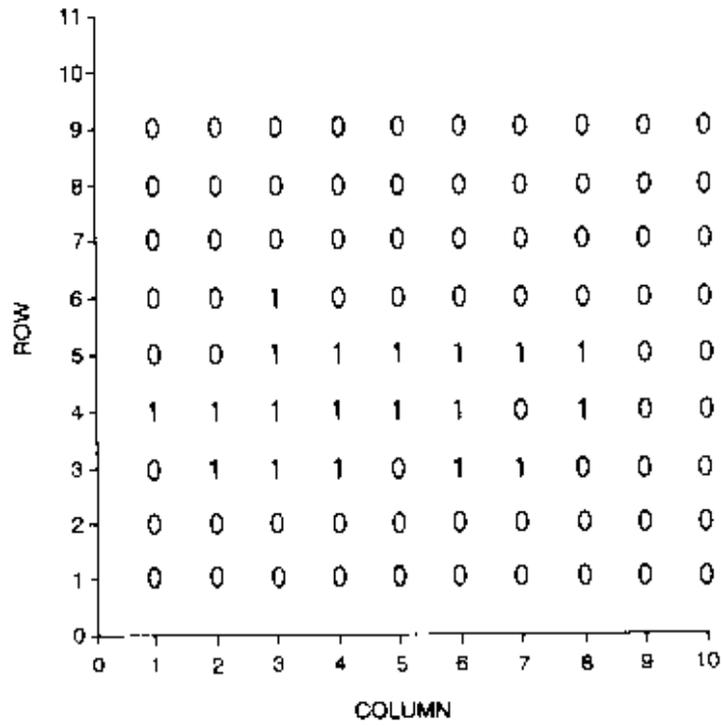


Figure 25 Longest internal dimension of fully and partly worn washers, by row, of belly used in the South Coast.

Figure 26 Location of grooving (denoted by 1) in fully worn longitudinal washers



used in the South Coast.

Proportionate change in dimensions

Table 1 summarises the overall mean percentage differences in the dimensions studied. These results show that the longitudinal washers are the component whose dimensions are most affected by the wear.

Table 2 Percentage increases in mean initial dimensions of dredge bellies subjected to different periods of use in different locations.

Mean Longest Internal Dimension (MLID)						
Location	Rings		Washers			
			Longitudinal		Horizontal	
	Partly worn	Fully worn	Partly worn	Fully worn	Partly worn	Fully worn
Scotland	----	15.4**	----	25.0**	----	7.3**
South Coast	0.6*	9.1**	0.0	13.2**	0.0	2.0**

* low variance ** high variance

Discussion

Physical causes

Change within a belly matrix is associated with effects arising from two main physical sources; friction and tension. Friction arises from contact with the seabed, with contents within the collecting bag and between matrix components. Contact with the substrate could induce annular rotation that can cause material loss from all surfaces and results in smooth worn inner surfaces. In contrast, abrasion between components without annular rotation causes rapid loss of material from the same locii on the inner circumference (point erosion), and the formation of grooves. Contact with contents within the collecting bag will cause mainly erosion from the outer circumference resulting in a reduced rod diameter. Tension within a ring matrix is caused by forward motion of the dredge and drag from seabed resistance. Contact with seabed objects can cause acute tensile loads that result in ring deformation (elongation) of variable magnitude.

The rate of component wear is likely to increase with time due physical and material changes in belly components. The wear rate is likely to be low, initially, due to the case hardened exterior of the components, but is likely to increase after erosion of the case hardening since the abrading surfaces would comprise soft mild steel. Loss of material would increase, also, the area of abrading surfaces causing further material loss, and increased tolerances allowing greater flexibility and increased scope for erosion. Rate of wear is likely to be affected, also, by component rotation since greater friction between abrading surfaces would increase the rate of material loss and groove formation.

Variance of fully worn components is greater than that of less worn equivalents indicating different wear rates for individual components.

Spatial change

The wear of all three components is most pronounced in the central rear section of the belly, with least wear occurring along the front and turn-ups (sides). This pattern is consistent with catch entering the dredge, collecting at the rear of the ring matrix thereby increasing frictional forces between the underlying components and substrate.

Within

the rear section point erosion occurs only in the rings, and is caused, probably, by the undulating motion of the matrix and lack of annular rotation. In contrast, the inner and outer surfaces of the longitudinal washers, which lie in the same longitudinal axis as the rings, are smooth and highly worn indicating that the washers rotate during passage over the substrate.

Areas of least wear occur at the front and along the sides of the matrix. During towing the teeth do not penetrate fully into the seabed which would result in the front of the belly being raised so reducing contact with the seabed and flexing between components. This process would explain the unworn outer surfaces and the longitudinally grooved inner surfaces of both rings and longitudinal washers within this section of the matrix.

Lateral forces across the matrix, however, are low, as indicated by the lightly, evenly worn inner surfaces of the laterally orientated horizontal washers. Forces applied during towing act longitudinally or longitudinally; any lateral component (of force) would arise mainly from the action of the dredge contacting the seabed or obstructions.

The results suggest that the location of wear remains similar throughout the life of a matrix since similar trends were observed in both part worn and fully worn bellies used under the same conditions.

Conclusions

It has been possible to describe typical erosion patterns in matrix components over time. Typical wear can be characterised by three types: 'non-specific erosion', which describes a reduction in diameter along the length of the steel rod; 'point erosion', which describes loss of material at specific locations on the inner circumference, and 'elongation', which describes ring deformation where the symmetry has changed from annular to oval. The wear of all three components is most pronounced in the central rear section of the belly, with least wear occurring along the front and turn-ups (sides).

3.2.2.6 GEOMETRICAL MODELLING OF SCALLOP (*PECTEN MAXIMUS*) SELECTIVITY

The objectives of the geometrical investigations were:

- 1 To establish relationships between the three principal dimensions of the scallop.
- 2 To create a three dimensional computer model of the scallop and appropriate apertures on the dredges.

To simulate the passage of the scallops through the apertures and compare these results with the selectivity trials.

Method

Scallop shape definition

In order to describe the relationship between scallop length, width and height, a sub-sample of 184 scallops from each of the locations West of Scotland, Irish Sea and English Channel was measured in these three dimensions length, width and height to the nearest mm below using callipers. The height: length and width: length ratios ([See Figure 1-1 Review p5 for definitions](#)) were plotted against length and regression lines calculated (Figure 4).

Modelling the scallop and gear

A computer model of the scallop shell using Idears Artisan™ software was created using the dimensions obtained above and also a scanned image. The model was constructed such that it could be made fatter or thinner in order to model variation in the shape of the shell both with size and the normal variation in shape. These changes in shape were modelled using the equations derived from the graphs shown in Figures 4. The height: length variation was modelled at plus or minus 7.5% of the mean and the width: length ratio was modelled at plus or minus 17.5% of the mean. These were derived from Figure 4 which shows the appropriate +/- lines.

Models of belly rings were also created within the software using the specifications derived from the field studies data.

Selectivity modelling

Two types of selectivity were modelled:

- 1 Through the rings selectivity
- 2 Between rings or 'ring lifting' selectivity.

In order to model through the rings' selectivity, Figure 1, the ring was rotated round the scallop's smallest dimension. The model was set up so that the ring cut the scallop if it came into contact with it. The model was iterated until the shell was able to clear the ring. This modelled the largest scallop that could pass through the ring of a given diameter. The washers were modelled in the same way as the rings. The scallops were not reduced in size to fit between the washers but washer grazing effects (Figure 2) were noted when they occurred. The model was run for scallops of the mean dimension for all the results and then the extreme variations derived from the length-

width-height relationships. These were height: length ratio minus 7.5% and width: length ratio minus 17.5% and height: length ratio plus 7.5% and width: length ratio plus 17.5% as the smallest and largest scallop shapes respectively.

In order to model between the rings' selectivity (Figures 3) the ring positions were fixed with a lift of 35° and an internal diameter of 73mm; the other dimensions were as specified in Lart et al (1997). The scallop (set to mean dimensions) starting from a set distance from the rings, was continuously moved towards the rings with a bias in favour of passing through. Random movements were made using a total of up to 10,000 translations or rotations. Each time a movement was made, the software tested to see whether the scallop and rings were overlapping. If they were, the movement would be reversed.

For the ring size and ring lift described above the different sized scallops were run on this model. The model was run with increasing sizes of scallop until no further movement was possible, or the centre of the scallop had passed through a pre-defined plane when the scallop was considered to have passed through between the rings. The number of movements (translations or rotations) before the scallop cannot move or has passed through the aperture between the rings was recorded.

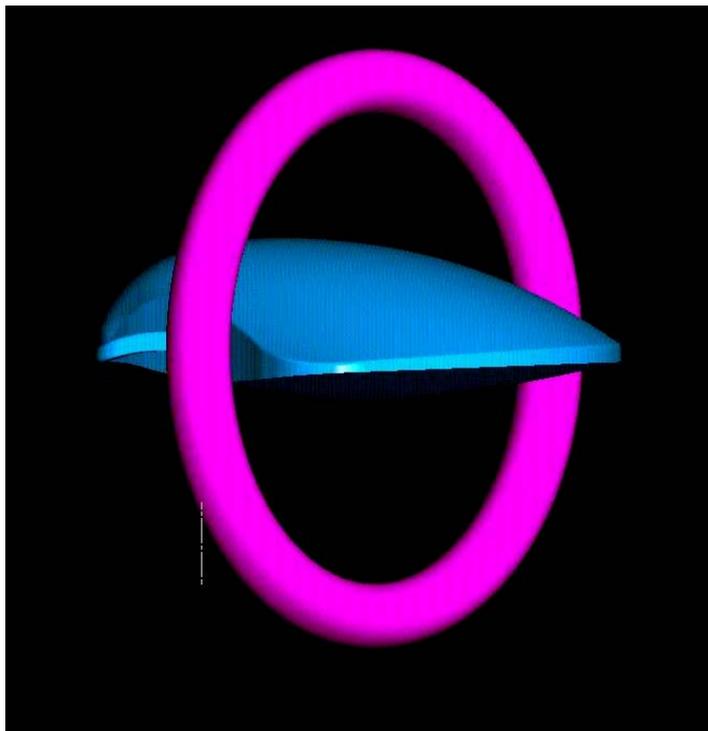


Figure 1 Mechanism of through ring selectivity

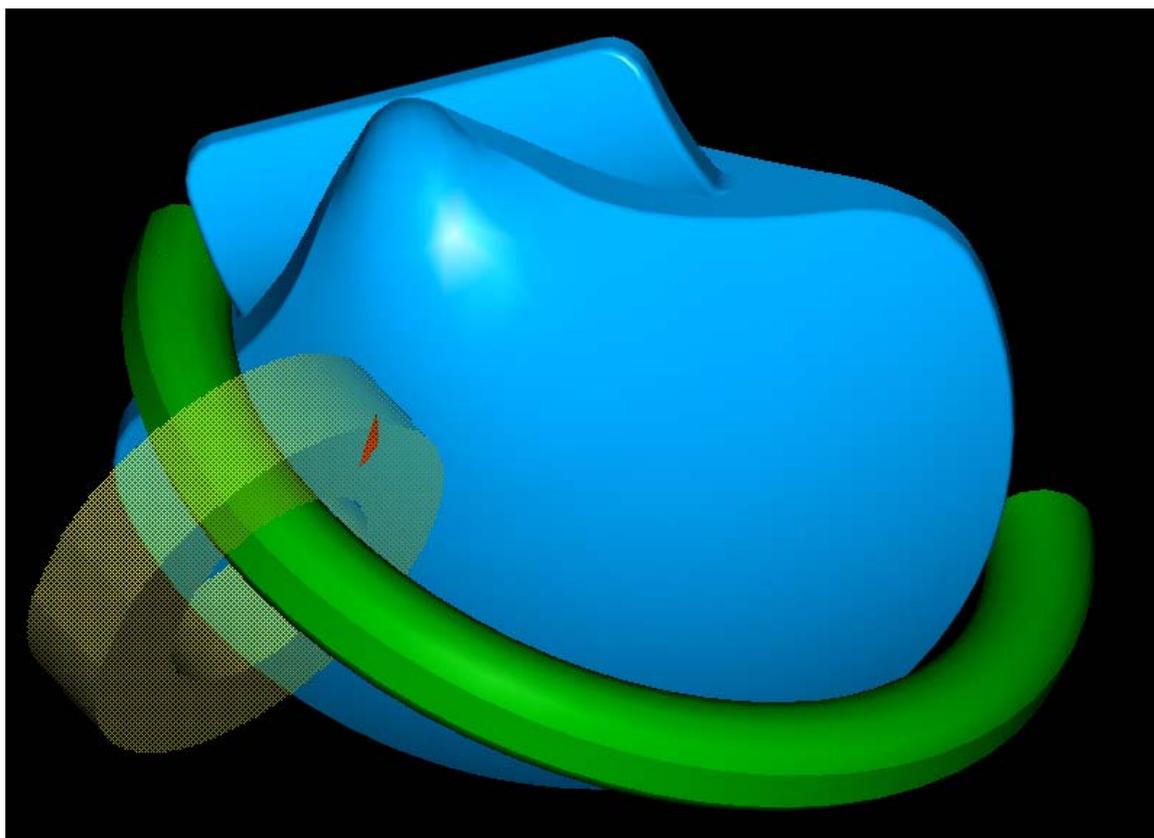


Figure 2 Washer grazing effects

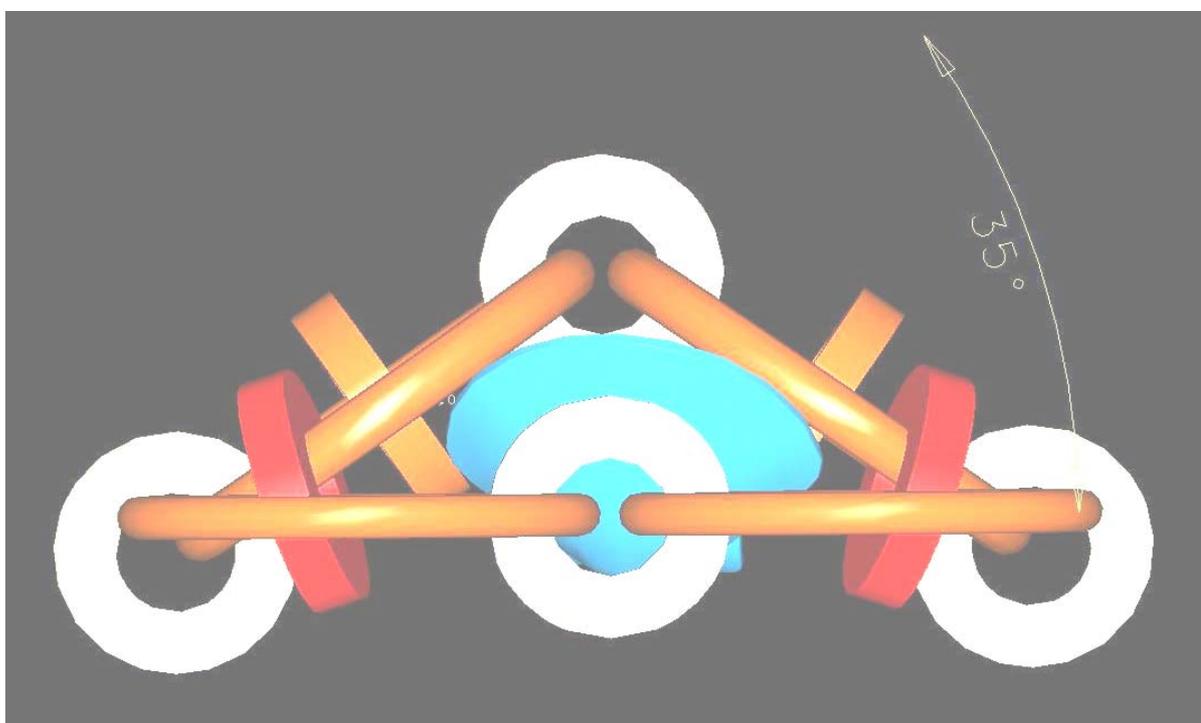


Figure 3 Mechanism of between the rings selectivity

Results

Scallop shape

In order that the model of a scallop shell could be constructed the scallop length was plotted against the length/height and length/width ratio, for the three populations studied. These results are shown in Figures 4. Although there were significant differences between populations the actual variations were quite small compared with the overall variation and ring selectivity does not vary very much between locations ([Section 3.2.2.3](#)). Thus for modelling purposes it was decided to take the limits of variation of both length: height and length: width ratio as described by the parallel lines shown in the Figures 1 and 2. That is $\pm 7.5\%$ for height: length and $\pm 17.5\%$ for height:width.

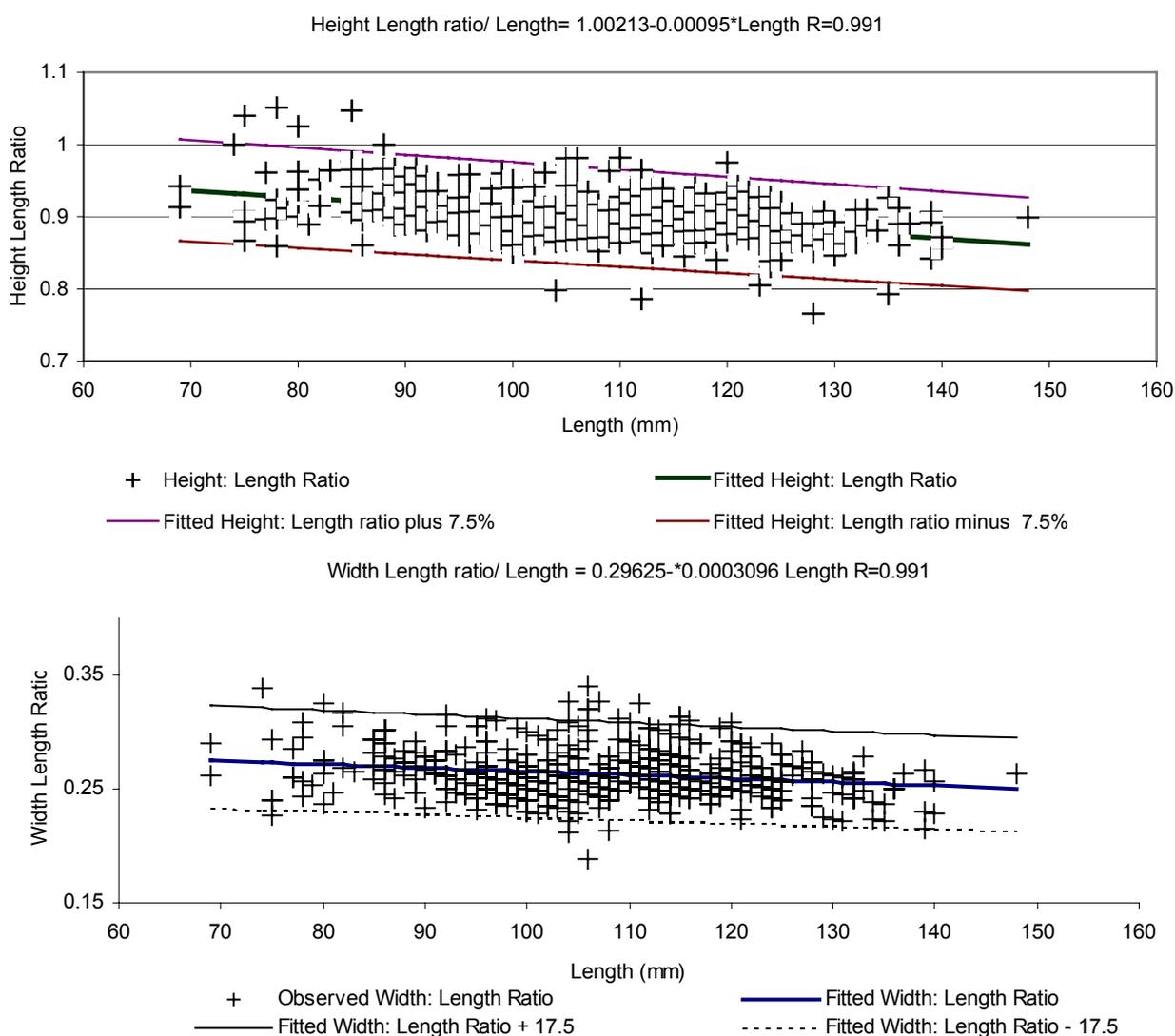


Figure 4 Relationships between scallop shell height and length and width and length

Through ring selectivity

The modelled results for through ring selectivity are shown in Figure 5-10 and compared with the results from the field trials. The observed L_{50} s are compared with the modelled values in Table 1. In addition to these results the selectivity of the 60mm (10mm) wire diameter population bellies was modelled. These results indicated an L_{50} of 70mm with the upper and lower boundaries being 68 and 72mm; both the L_{50} and the lower boundaries indicated washer grazing would take place, in the case of the lower boundary there was severe washer grazing.

For the 75, 80, 85(both 10 and 12mm bar) rings the comparison is more valid than for the 88 and 92mm rings because there was a loss of efficiency in these larger ring sizes. In all cases the modelled L_{50} was greater than the observed L_{50} with a tolerance of between 1 and 8mm. Washer grazing effects were confined to the largest scallops for their length.

Table 1 Observed L_{50} s compared with modelled L_{50} s.

Ring specification		Length at 50% retention (L_{50})		
Internal diameter (mm)	Wire diameter (mm)	Observed (mm)	Modelled (mm)	Difference (mm)
75	10	85	89	4
85	10	96.5	101	4.5
92	10	108	109	1
80	12	86	94	8
85	12	97	100	3
88	12	98	104	6

The two ends of the modelled curve are the most difficult to model. The method used to describe scallop shape is probably most valid for those scallop shapes nearest the mean. The extremes were modelled but it should be commented that the extreme combinations chosen may be rare in the population. Also the shape of the 'ears' of the scallop which could be crucial to the passage of the scallop through the ring can be quite variable. Thus the results for the upper and lower ends of the modelled curves are less reliable compared with those for the mean. It should also be taken into consideration that the scallops in the selectivity trials were measured to the nearest 5mm below but the scallops measured for modelling and the models themselves were measured to the millimetre.

The modelled and observed curve, in most cases (the exception being the 80mm 12mm wire bellies), converged towards the top of the curve. This implies that the shorter thinner scallops have a higher probability of passing through the rings. This could be accounted for by washer grazing effects, the wider scallops being more likely to encounter the washers. Alternatively it could indicate that some between the rings selectivity was occurring.

Between the rings selectivity

Modelling selectivity between the rings was more difficult due to the dynamic nature of the interaction between the rings and scallops. Figure 12 shows the number of moves required by scallops of a range of lengths took before they passed through between the rings. No scallop of greater than 72mm passed through.

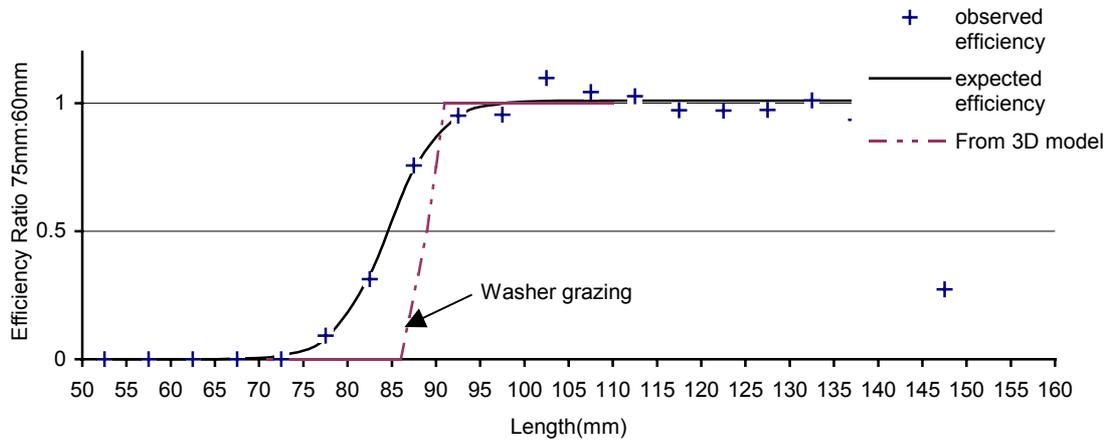


Figure 5 Observed vis modelled selectivity curves 75mm rings (10mm wire)

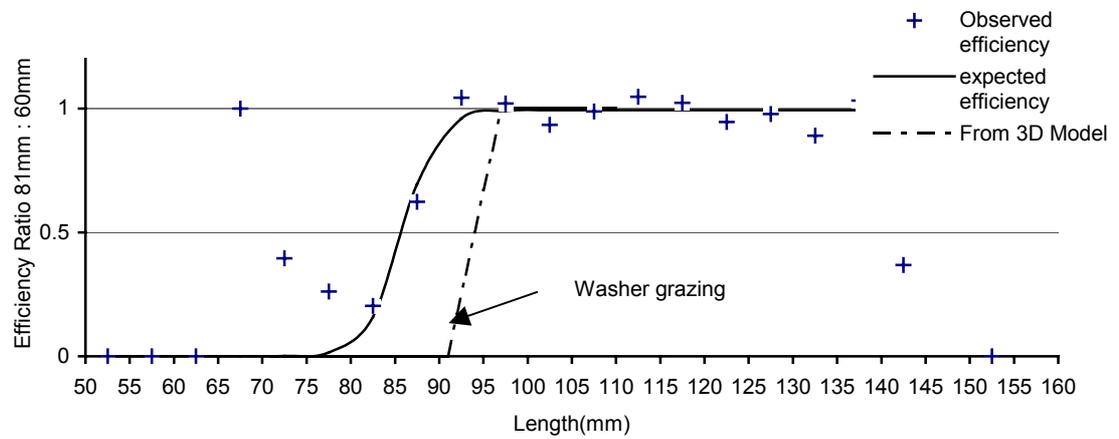


Figure 6 Observed vis modelled selectivity curves 80mm rings (12mm wire)

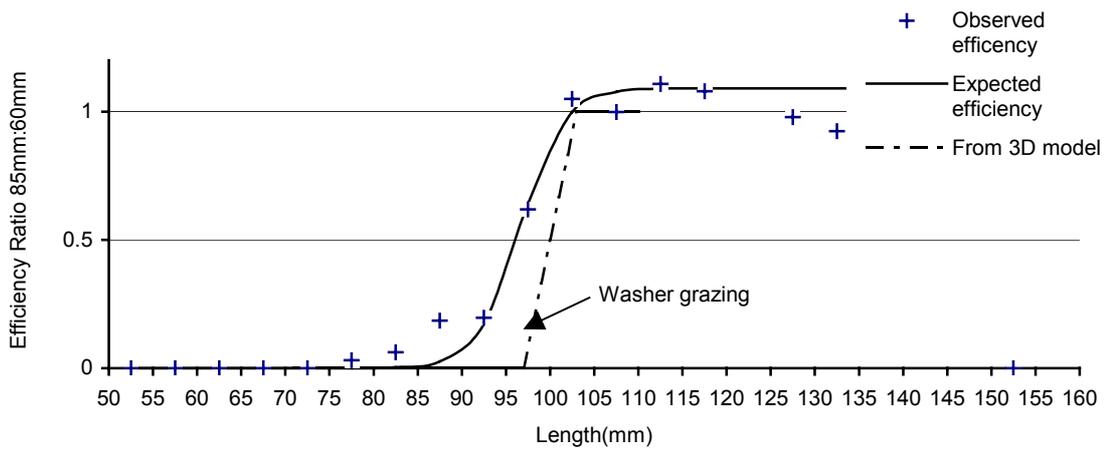


Figure 7 Observed vis modelled selectivity curves 85mm rings (10mm wire)

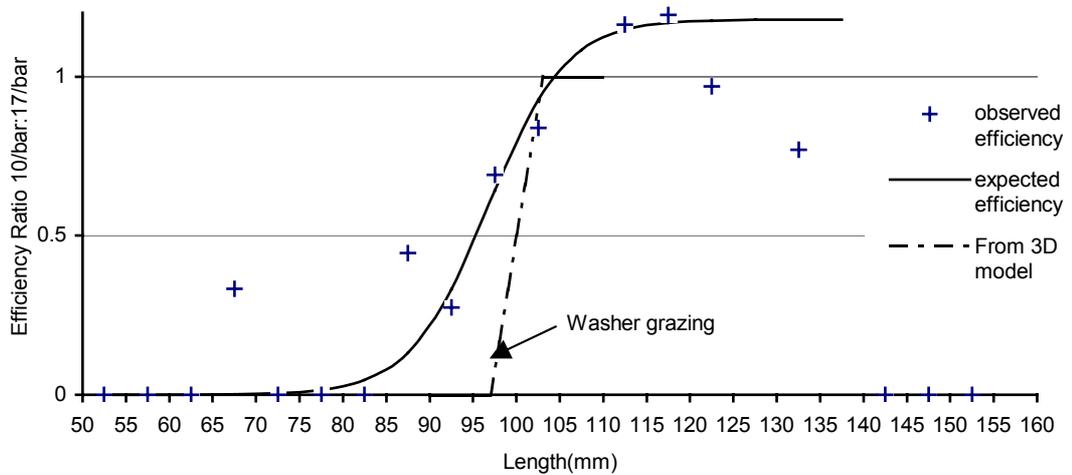


Figure 8 Observed vis modelled selectivity curves 85mm rings (12mm wire); worn bellies.

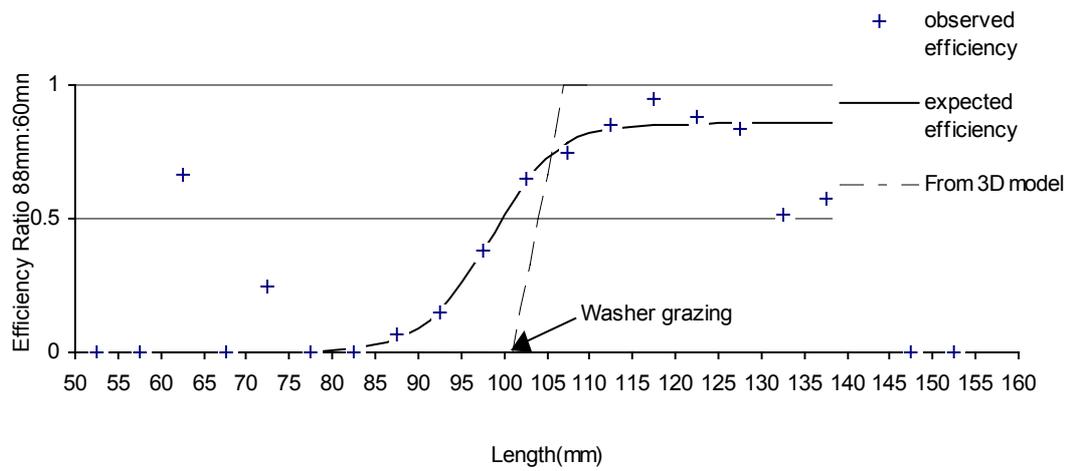


Figure 9 Observed vis modelled selectivity curves 88mm rings (12mm wire).

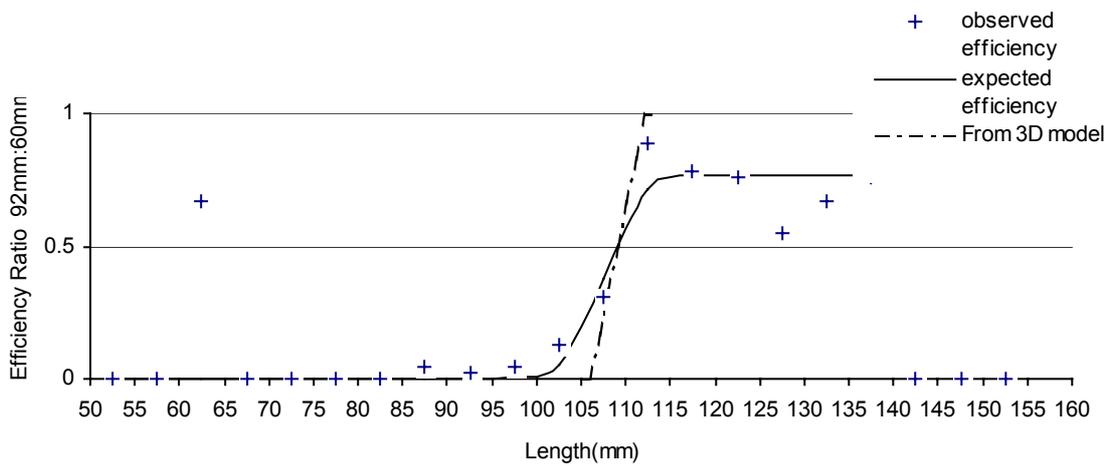


Figure 10 Observed vis modelled selectivity curves 92mm rings (10mm wire)

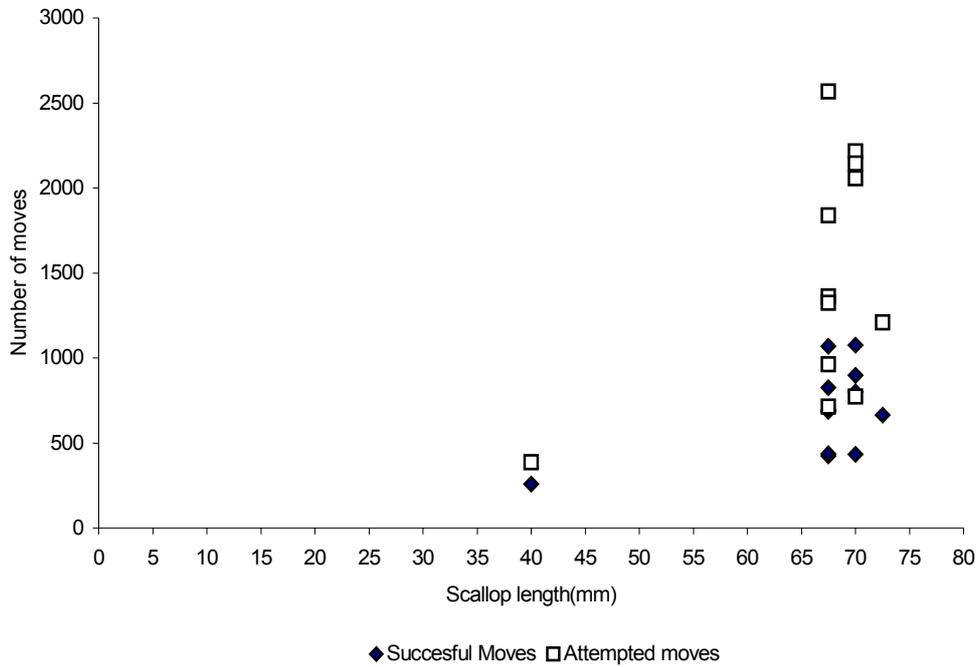


Figure 11 Between the rings selectivity: Attempted and successful moves against scallop size using scallops of the mean shape and 73mm rings (10mm wire).

Discussion

Selectivity apertures

These results support the hypotheses that for new bellies the most important aperture for selection is through the rings. However, it was difficult to model the between the rings selectivity. Effectively in the system modelled the rings and washers were held static and the scallops moved relative to them. Potentially, because the components can move relative to each other the aperture for the scallops to pass through could be quite large. Further investigation would require field trials with the ring selection eliminated by welding a bar across each of the rings.

Population bellies

The results show that catches of the 60mm rings were probably an adequate description of the population of scallops for the purpose of this study. There were very few scallops captured in any of the experimental dredges of less than 70mm length. The results from the geometric study suggest that 72mm is the longest scallop which, could pass through the 60mm rings. Thus effectively the 60mm rings were catching the full selection range of all these dredges.

Effect of wear

The two new 12mm wire thickness rings at 80 and 88mm had the largest differences between the modelled and observed L_{50s} . This difference could partly be related to 3D effects which mean that the shape of the wire would have an effect on the selection characteristics, but it is more likely to be related to the size of the washers which were larger on the thicker rings. For the two 85mm ring sizes (10 and 12mm wire) the observed and modelled L_{50s} were very close together. However the 85mm ring size (12mm wire) was worn suggesting that wear had some effect.

These models suggest that washer dimensions play an important part in selectivity. Washer grazing effects were modelled in most ring sizes on large scallops for their length. The internal dimensions of the washer dimensions changed considerably as the bellies wore ([Section 3.2.2.5](#)). This would not only increase the aperture between the rings but reduce the washer grazing effects in through the rings selectivity. Comparison between the changes in the selectivity of 80mm (12mm wire) rings and the 75mm (10mm wire) rings show very similar characteristics for the first part of their working lives. Then in the final observation made the 80mm (12mm wire) becomes more selective without substantial elongation of the rings.

Running the geometrical model with successively reduced diameter washers showed that a reduction of 2mm in the washer diameter would eliminate washer grazing during the passage of the largest scallop for its length passing through in the 80mm rings. Thus the results of the field trials and the geometrical modelling suggests that increased selectivity observed in the field observations is as a result of the changes in wear on the washers. Passage of the scallops through the rings and between the rings selectivity would be augmented by increased wear on the washers. If the through the rings selection model is correct then these it would be a valuable aid in examining the effect of different belly component shapes and wear patterns.

3.2.3. Summary statement

This study was able to demonstrate species selectivity in scallop dredges targeting queen scallops (*Aequipecten opercularis*). The use of skids and tickler chains rather than teeth resulted in a reduction almost to zero of by catches of king scallop (*Pecten maximus*).

Selectivity mechanisms were studied in sprung toothed king scallop dredges. It was found that belly ring size and tooth spacing were the most important components affecting selectivity. In Portuguese clam dredges the most important size selectivity mechanism was the mesh size tooth spacing did not play a part in selectivity, but species selectivity was considered to be affected by tooth length. Size selectivity was quantified for Portuguese clam dredges (species *Spisula solida*, *Callista chione* and *Donax trunculus*) and sprung toothed king scallop dredges (*Pecten maximus*). It was shown that both Portuguese clam dredges and scallop dredges could be highly selective with the correct mesh (clam dredges) or belly ring (scallop dredges).

However, for king scallop dredges the selectivity of the bellies changed through their commercial life, losing small scallops as they became worn. Some sizes were optimal when new but not robust enough for the fishery using current materials. It seems possible that improving hardening techniques improve this situation, but smaller rings of the same material would be stronger by virtue of their geometry. It was also found that the washers were severely worn in the older dredge bellies and that this had an effect on selectivity. Improving longevity of bellies with larger rings would require improvements in the hardness of both rings and washers and to investigate the effect of changing the geometry of these components, particularly the washers, on their ability to resist wear.

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3.3 Dredge design and environmental effects

This Section examines the effect of dredge design features on the environmental effects of dredging. A number of features designed to replace or modify teeth as a mechanism for lifting scallops out of the seabed (*Pecten maximus*) were examined in prototype form, by models in the flume tank, or by observation of other fisheries. The information gained was used to design a prototype hydraulic scallop dredge. The environmental effects of this dredge were then tested in comparison with the classical dredges used in the fishery.

For Portuguese clam dredges, modifications to dredges already in use in parts of the fisheries for *Callista chione* and *Spisula solida* were compared with traditional dredges for their relative environmental effects.

The environmental effects examined included observations of damage rates and benthic impacts and also stress indices (AEC levels) in scallop fisheries.

3.3.1. Investigation of novel dredge design features

This study investigates possible modifications of the means for lifting scallops out of the seabed. The intention was to substitute or modify the teeth, which are considered to be the dredge component, which has the most effect on the environment. Four innovative techniques were examined: –

- The Magnus effect, which is the effect of water pressure differences around an immersed rotating cylinder. Incorporated into a dredge design it has the potential for removing the scallops from the seabed.
- Hydrodynamic foils are used to improve stability and seabed contact in certain dredge fisheries. This investigation examined whether the foil could be used to generate sufficient lift to remove scallops from the seabed.
- Modification of the tooth design based on designs for seabed ploughs used in the offshore cable industry.
- Hydraulic or water-jet dredges ([Review Section 3.2.2](#)) are used in clam and cockle fisheries, and this study considered their use in scallop fisheries.

The objective of this study was to investigate all four of these options and to choose one of the techniques and develop a dredge prototype to be tested at sea.

3.3.1.1 THE MAGNUS EFFECT

Introduction

A rotating cylinder with its axis perpendicular to a fluid flow is subject to a lift force: this is the Magnus effect. This force is due to the asymmetry of the flow created by the rotation (Figure 1).

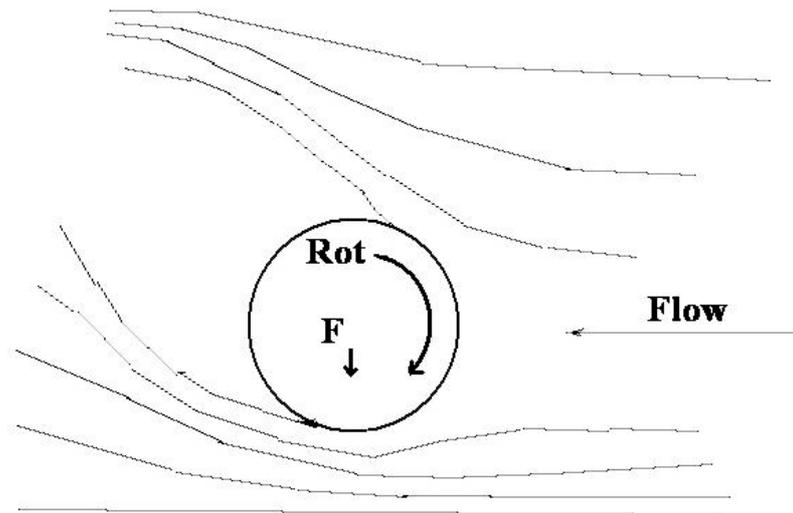


Figure 1 The Magnus effect representation: The rotation (Rot) in the flow creates a lift (F) on the cylinder.

The commonly accepted explanation of the Magnus effect is that a rotating cylinder creates a whirlpool of rotating fluid about itself. On the side where the motion of the whirlpool is in the same direction as that of the stream to which the cylinder is exposed, the velocity will be enhanced. On the opposite side, where the motions are opposed, the velocity will be decreased. According to Bernoulli's principle the pressure is lower on the side where the velocity is greater, and consequently there is an unbalanced force at right angles to the flow. This is the Magnus force.

The Bernoulli's principle is that along a current line the following equation can be written:

$$p + \frac{1}{2} \rho V^2 = \text{constant}$$

V: flow speed of the fluid (m/s),
 p: pressure in the fluid (Pa),
 ρ: density of the fluid (kg/m³).

This principle rests in that if the speed increases the pressure decreases.

The most recent studies agree that the Magnus force results from the asymmetric distortion of the boundary layer displacement thickness caused by the combined

rotating and flow past the cylinder. In the case of a cylinder, the so-called whirlpool, or more accurately the circulation, is not produced by fluid set into rotation by the friction with a spinning cylinder. Actually a cylinder can only impart a rotating motion to only a very thin layer next to the surface. The motion imparted to this layer affects the manner in which the flow separates from the surface in the rear. The boundary layer separation is delayed on the side of the rotating cylinder that is moving in the same direction as the free stream flow, while the separation occurs prematurely on the side moving against the free stream flow. The wake then shifts toward the side moving against the free stream flow. As a result, flow past the object is deflected, and the resulting change in momentum flux causes a force on the cylinder in the opposite direction (down-wards in the case shown in Figure 1).

In fact, in the case of dredges, it is not intended to the transverse force but the deflected wake behind the cylinder is expected to be strong enough to lift the scallops from the seabed.

The lifting coefficient could be used to quantify the lifting effect. Figure 2 shows the values of the lifting coefficient versus the ratio of rotating velocity and current velocity.

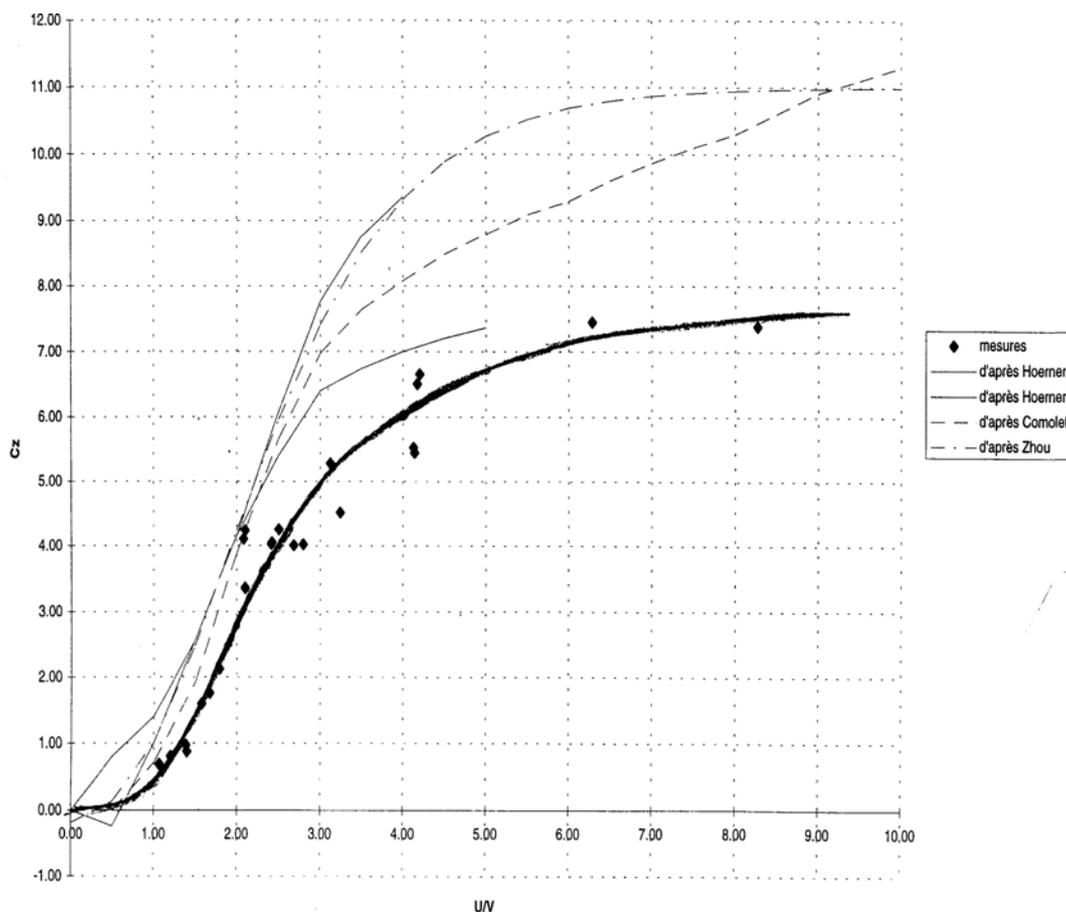


Figure 2 Lifting coefficient of rotating cylinder. U: periphery velocity of the cylinder. V: current velocity. Cz: lifting coefficient.

Materials and methods

To investigate this technique, the dredge prototypes based on “Magnus effect” and built in France are studied.

The first prototype of dredge for laid shellfishes

From the Magnus effect principle, two inventors, Messrs Le Béon and Jégo, from Lorient, France, have taken out a patent for a process named:

“Procédé d’affouillement d’un sol immergé et création d’un courant d’eau ascendant, et dispositif mettant en œuvre un tel procédé”

(Process of undermining of an immersed ground (sea bed) and creation of an upwelling, and device implementing such a process)

The invention consists in setting two parallel cylinders rotating in opposite directions, and to make a translation of these cylinders near the sea bottom perpendicularly to their axis. This induces an ascending vortex able to lift elements laid on the sea floor. Then, these elements follow a trajectory that moves away from the cylinder turbulence. A dredge scheme is proposed in the patent (Figure 3).

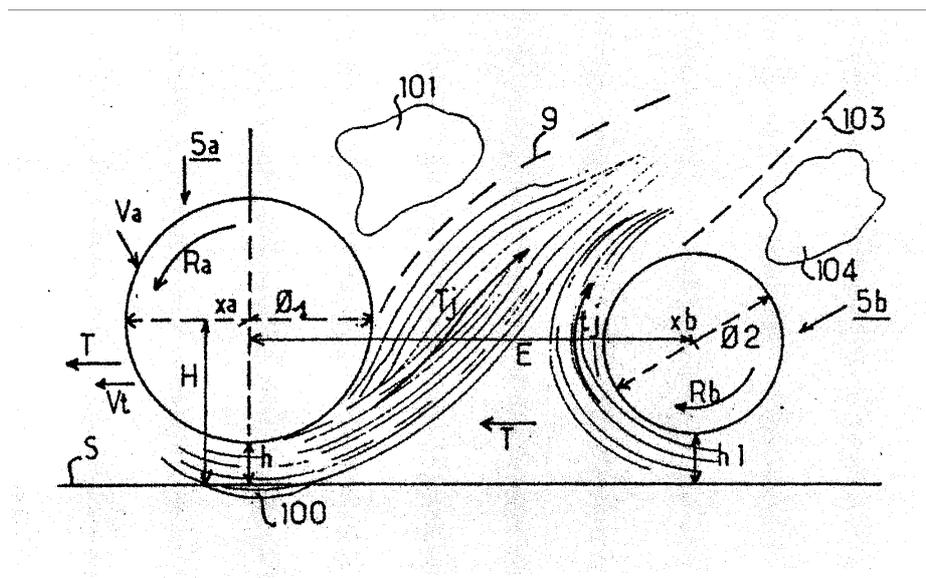


Figure 3 Functioning principle of the Magnus effect with 2 cylinders (from the patent). Ra and Rb: Rotating direction, T: Translating movement relatively to the sea floor, Vt: Towing speed, Va: Peripheral speed, Tj: Trajectory of water particles, h and h1: Ground clearance

A first prototype was built, which consists of a frame fixed on wheels, which turn with the towing motion. This drives the cylinder rotation by mean of a belt. The dredge was supported by skis to prevent it from sinking into the sediment and to facilitate its sliding. A grating or netting basket collects the shellfishes (Figure 4).

This prototype was tested at sea; unfortunately the results did not come up to the expectations.

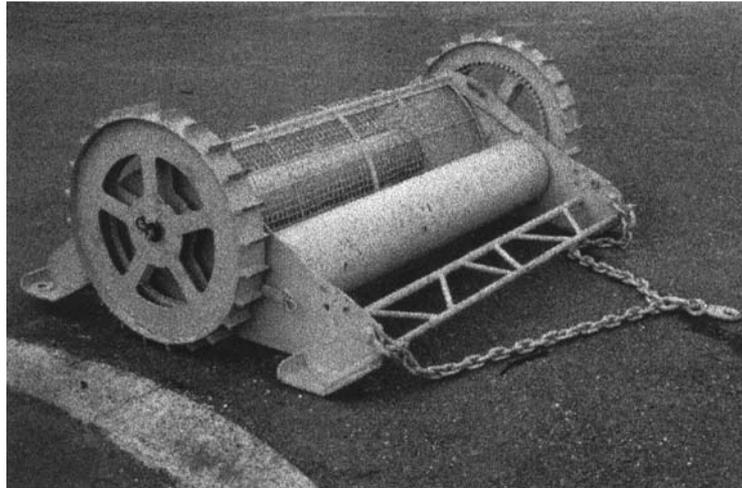


Figure 4 First prototype of dredge, built by Mr Le Béon.

The second prototype, built by IFREMER

The second prototype was designed and built at IFREMER (Brest France, Figure 5). It consists of:

- A metallic frame resting on skis.
- Two cylinders fitted with flanges on their extremities. The cylinders turn in opposite directions. Hydraulic engines (Figure 5) ensure the rotation. The low pressure and the water is intended flow, due to the Magnus effect, induces suction of the shellfish from the seabed.
- A netting basket to store the collected shellfish.

The main difference between this and the first prototype consisted of the hydraulic engines which ensured the cylinders' rotation.

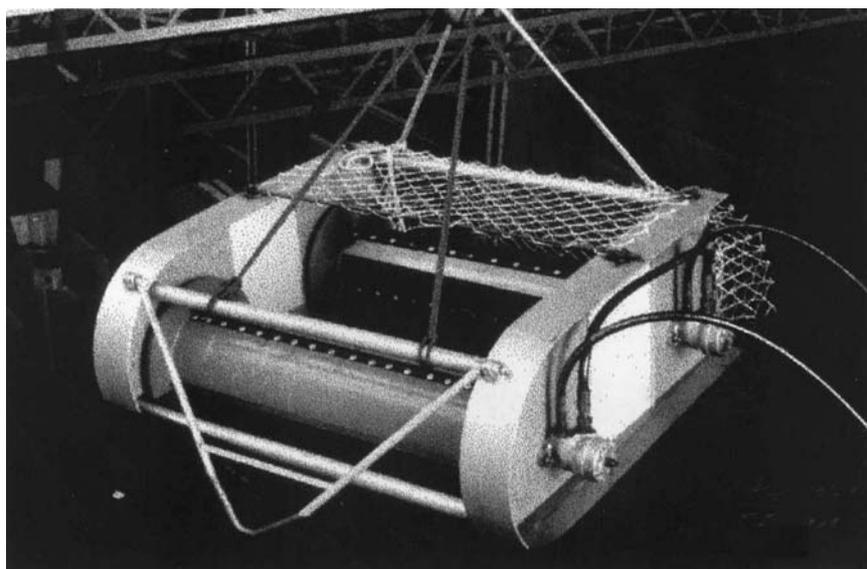


Figure 5 The second prototype with the hydraulic engines

Two sets of sea trials demonstrated that the critical points were, on the one hand the boundary effect at the extremities of the cylinders, and, on the other hand the inefficacy of the flanges set on the extremities. The video shows a limited aspiration at the middle of each cylinder along with a perturbed flow on each side. By adding small horizontal wings to the first cylinder it seems that this cylinder shows a propensity to lift the scallops from the seabed.

The dredge, 1.4 m wide, heavy and not very efficient, showed too large a vulnerability of the flanges on account of the fact that they were not protected. It was decided to build a smaller prototype (1m wide).

The third prototype, built by IFREMER

This third prototype was smaller than the one previously built. It was designed with the same components, keeping the same diameter for the cylinders, the same ground clearance, and the same spacing, but with a modification of the skis width so that the flanges, now being inside of the frame, be protected (Figure 6).



Figure 6 Third prototype during sea trials

Trials were carried out, at first in a hydrodynamic tank to investigate the effects of several configurations of cylinder diameter, axes spacing, movement speed and rotation speed.

Results

Trials in hydrodynamic tank

The third dredge prototype was towed on the bottom of the hydrodynamic tank. Scallops models were obtained by gluing and weighting empty shells. These models were laid on the bottom.

The first series of trials was dedicated to the Magnus effect alone (smooth cylinders). The diameter of the rear cylinder was larger than that of the fore one. The trial conditions are described in Table 1 and the results are summarised as follows: During trial 1, there was less turbulence in the middle, less drag of the fore cylinder, and quite good overall results. On trial 2, the results were less good. Trial 3 showed very good results (similar to those of trial 1). During trial 4, very few scallops were lifted.

Overall these results indicate that there was deterioration in the catching power if the distance between the cylinders was increased. The results improve as the ground clearance decreases. Also there were better results with a small fore cylinder and a large rear cylinder.

Table 1 Dredge configuration in the hydrodynamics tank with smooth cylinders. Gr. Dist: ground clearance, rpm: rotating speed, Towing speed in knots.

Trial n°	Fore cylinder φ mm	Gr. dist	rpm	Cyl. Dist mm	Rear cylinder φ mm	Gr. dist	rpm	Tow. speed
1	200	100	231	450	250	100	163	2.2
2	200	100	231	480	250	100	163	2.2
3	200	70	280	350	250	70	235	2.2
4	200	70	280	420	250	70	235	2.2

The second series was dedicated to investigating the effect of small wings (2 cm high) on the rear cylinder which diameter was half that of the fore one. The trial conditions are presented in Table 2 and the results summarised as follows: During the trial 1 turbulence appeared, and the results were poor. Trial 2 showed better results. Trial 3 gave the same results as trial 1. Good results were obtained on trial 4. Trial 5 showed better results. During trial 6 the results were similar to those of trial 3.

The overall results showed an improvement as the towing speed increases. The results improve as the fore cylinder rotation speed increases. Also an increase of the cylinder diameter did not alter the results.

Table 2 Dredge configuration in the hydrodynamics tank with 4 small wings fitted on the rear cylinder. Gr. Dist: ground clearance, rpm: rotating speed, Towing speed in knots.

Trial n°	Fore cylinder			Cyl. Dist mm	Rear cylinder			Tow. speed
	ϕ mm	Gr. dist	rpm		ϕ mm	Gr. dist	rpm	
1	200	100	231	420	110	100	163	1
2	200	100	231	420	110	100	163	2.5
3	200	100	231	420	110	100	163	2.5
4	200	100	231	420	110	100	249	2
5	200	100	231	420	110	100	249	2.5
6	200	100	231	450	200	100	163	2.5

Considering the results obtained in the hydrodynamic tank, the features of the dredge tested at sea were the following:

Fore cylinder: 200 mm diameter, 280 rpm
 Rear cylinder: 250 mm diameter, 235 rpm
 Distance between cylinders: 350 mm
 Ground clearance: 70 mm
 Towing speed: around 2.2 knot.

Sea trials

The first series of sea trials was conducted with the previous features. The two following trials were completed with successive modifications. The second series of sea trials was stopped on account of breakage of the dredge skis.

Sea trial series on Maerl

The results were pretty bad: hauls duration were about 10min and only 4 scallops were collected. Some other configurations (Table 3) were then tested which ended in the same poor results.

Table 3 Configuration of the first sea trials series. Gr. Dist: ground clearance, rpm: rotating speed, Towing speed in knot.

Trial n°	Fore cylinder			Cyl. Dist mm	Rear cylinder			Tow. speed
	ϕ mm	Gr. dist	rpm		ϕ mm	Gr. dist	rpm	
1	200	90	270	340	250	70	250	1.8
2	200	90	336	340	250	70	340	1.8
3	250	85	340	420	250	70	340	1.5 to 1.8
4	250	85	340	420	100	70	680	1.8 to 2.5

The divers noted that 50% of the shellfishes were extracted from the Maerl, 10% were turned over but were not collected by the dredge.

The sediment, a few centimetres thick, seems to have been an obstacle to the scallops lifting.

Sea trial series on sand and fine gravel

The trials configuration are provided in Table 4. The rear cylinder was equipped with 8 wings 20 mm wide. The duration of hauls was about 7minutes. Around nineteen shellfishes were collected per haul. During trial 2, the divers had counted 46 scallops left on the seabed on some 100m (approximately half of the towing length of a haul). This poor efficiency was considered to be due to the poor functioning of the hydraulic motors driving the cylinders.

Furthermore, the dredge collected only the mean size scallops, the biggest ones remaining buried. The rotation speed was probably too low relatively to the previous trials (the hydraulic motor was not powerful enough to reach higher speed).

Table 4: Configuration of the third sea trial series. Gr. Dist: ground clearance, rpm: rotating speed, Towing speed in knot.

Trial n°	Fore cylinder			Cyl. Dist mm	Rear cylinder			Tow. speed
	ϕ mm	Gr. dist	rpm		ϕ mm	Gr. dist	rpm	
1	250	70	145	450	200	70	145	1
2	250	70	145	450	200	70	145	2
3	250	70	185	450	200	70	185	1.5 to 2
4	250	70	185	450	200	70	185	2.5

Discussion

The prototype designed, built, and tested at sea, demonstrated that the fishing of scallops with two cylinders turning in opposite directions is possible. Nevertheless, the results were not as expected with such a process. This process presents both advantages and drawbacks:

Advantages:

- Better security due to the decrease of risk of obstruction on account to the dredge shape and the rotating cylinders.
- Decrease of boat power required to tow the dredge.
- Decrease in the damage on animal life and environment due to the absence of teeth.

Drawbacks:

- Definitely more expensive than a classical dredge.
- Notably more complicated than a classical dredge on account of the hydraulic engines.
- Notably less efficient than a classical dredge.

Sea trials were also carried out on a seabed overcrowded with starfish. Within a few minutes, the dredge was full of a mixture (around 100 kg) of starfish, mussels and gravel. Thus this dredge could turn out to be useful for the eradication of starfish from oyster beds.

The process of the Magnus effect dredge was also applied by IFREMER to two other applications:

- Clam dredge on beds. This application had no output as clam farming is experiencing a significant decrease.
- Algae collection, which was tested on *Laminaria hyperborea*, with a possible development on account of the industrial exploitation of this algae.

3.3.1.2 HYDRODYNAMIC LIFTING PERFORMANCE OF THE DREDGE FOIL

Introduction

Principle of the foil

The lifting effect results from the shape of the foil (Figure 7). The top of the foil being longer than the bottom the current line, which follows the top, is faster than the bottom line. Following Bernoulli's principle the pressure on the top is lower, that creates a force on the foil toward the top. Thus, there is a reaction on the flow, which is deflected to the bottom (Figure 8).

Figure 9 shows values of lifting coefficient of foils.

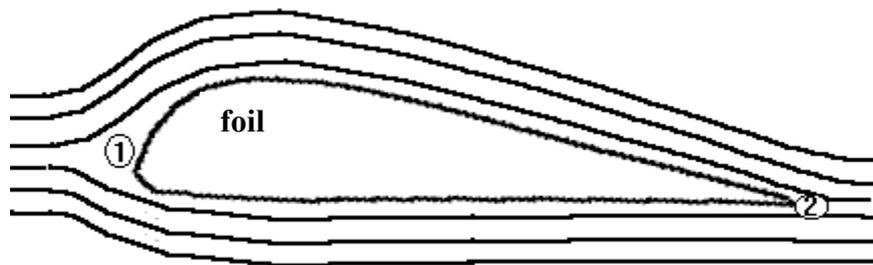


Figure 7 The path (from 1 to 2) over the foil is longer than under, so the fluid speed is higher and the pressure lower.

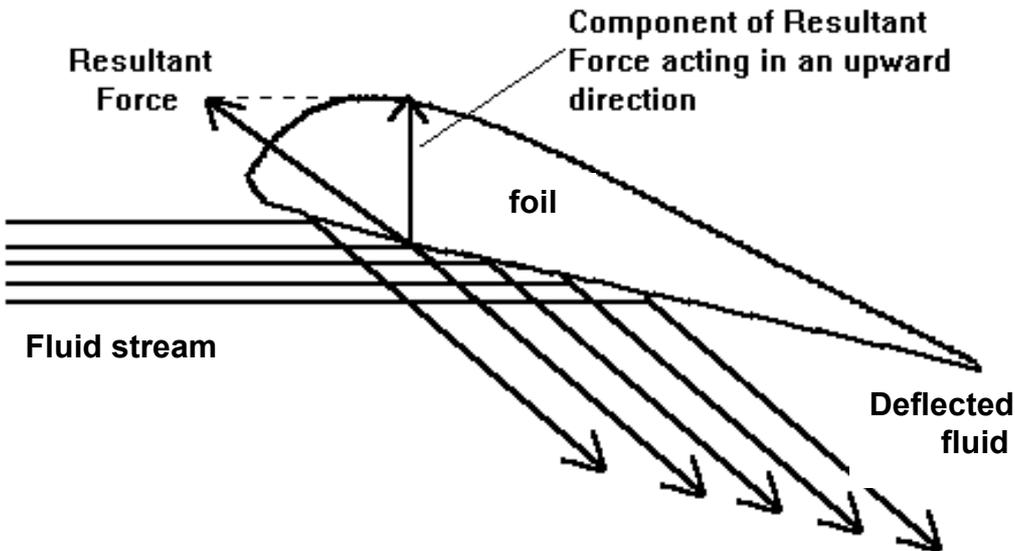


Figure 8 Foil, the flow comes from left. The lifting effect is due to the fluid stream, which is deflected, and the low pressure on the top of the wing.

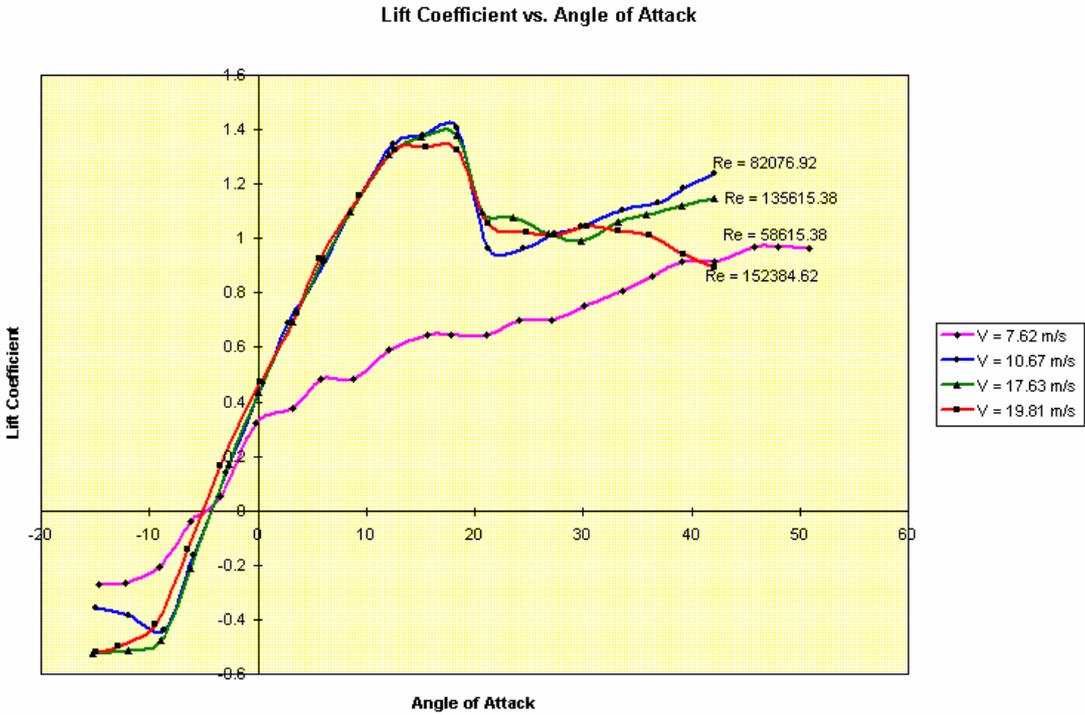


Figure 9 Lifting coefficient of foil. Re: Reynolds number.

Foil dredge principle

The foil dredge is made of a foil (Figure 10) and a basket supported by two skis sliding on the seabed. The movement of the foil creates a lack of pressure at the rear part of the foil (area A) and consequently a suction effect under the leading edge of the foil (area B). This suction is expected to generate sufficient a current to remove the scallops from the seabed. The flow must also take-off from the seabed (arrow C) so that most of the scallops are collected in the basket.

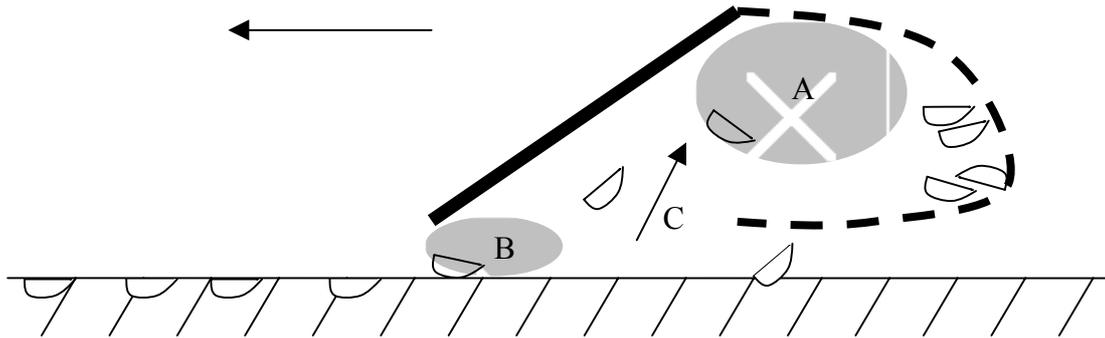


Figure 10 Principle scheme of the dredge: the lack of pressure in A creates a suction in B which removes the scallops. The scallops are lifted by the foil (bold line) and collected in the basket (discontinued line).

The foil must reach two objectives: (i) create a water speed near the bottom to remove the scallops from the seabed; (ii) create a lifting effect to collect the scallops in the bag.

In a first approximation to meet this final objective it is required to define a foil configuration which would give the largest vortex behind the foil. These vortices would be able to lift the scallops from the seabed. Generally, foil designs are such that they avoid vortices. Especially foils on boats, or wings on planes because in these cases the foils are used to create the larger lifting effect on boat or plane, whereas in the dredge case the larger lifting effect must be created on the scallops (or on the water) not necessary on the foil. This means that this is not a normal use of the foil, and consequently that the foil is not necessarily the best system for lifting the scallops from the seabed. Some other system may turn out to be more efficient: a horizontal cylinder or a vertical plate could create wakes sufficient to lift the scallops. Anyway, taking into account this paradox, the foil was chosen because dredge or hydrodynamics specialists generally believe that such a system could be adapted to lifting scallops.

Previous works

The foil dredge principle has been partly studied by Cover (1996). In this study the foil was an added appendix to traditional dredge. The foil was used to increase the catching efficiency by increasing the lifting effect on scallops. The lifting effect is measured in term of pressure drop around the foil. The foils involved had incidences between 0° to 90°. This incidence range is smaller than the incidences studied in the

Ecodredge project (up to 150°). These studies did not tackle the requirement to remove the scallops from the seabed into the basket (B on fig. 4).

Critical water speed

The critical water speed is the water speed necessary to remove the commercial size scallop from the seabed.

The critical speed U_{cr} required to move a grain of diameter d on a flat horizontal unrippled bed of water of depth h could be predicted by Soulsby formula. This is valid for any non-cohesive sediment and water conditions for which $D_* > 0.1$.

$$U_{cr} = 7 \left(\frac{h}{d_{50}} \right)^{1/7} \sqrt{g(s-1)d_{50}f(D_*)}$$

$$f(D_*) = \frac{0.30}{1 + 1.2D_*} + 0.055[1 - \exp(-0.020D_*)]$$

$$D_* = \left[\frac{g(s-1)}{\nu^2} \right]^{1/3} d_{50}$$

With

U_{cr} : critical water speed (m/s),
 s : ratio of densities of grain and water,
 ν : kinematic viscosity of water,

For scallop dredging with the previous foil:

$h = 0.1$ m (height under the leading edge)
 $d_{50} = 0.1$ m (size of scallops)
 $g = 9.81$ m/s² (gravity)
 $s = 1.2$
 $\nu = 0.00000135$ m²/s

which gives $U_{cr} = 0.73$ m/s

This water speed is required to remove the scallop from the seabed. This speed seems to be reasonable and reachable with such a process.

Materials and methods

Flume tank and numerical model were used to investigate this system. The main objective of the flume tank test was to verify the water speed under the foil and whether it was of the same order as the critical water speed previously calculated. The objective of the numerical model was to verify the global flow around the foil.

Flume tank tests

The system used for these flume tank tests was made of two vertical Plexiglas flanges on a supporting frame (Figure 11). The flanges were chamfered with a 7° angle on the front part to avoid the development of the boundary layer. Holes in the flanges allow operator to chose various incidences of the foil. The vertical position of the flanges can be adjusted to choose the gap between the foil and the bottom.

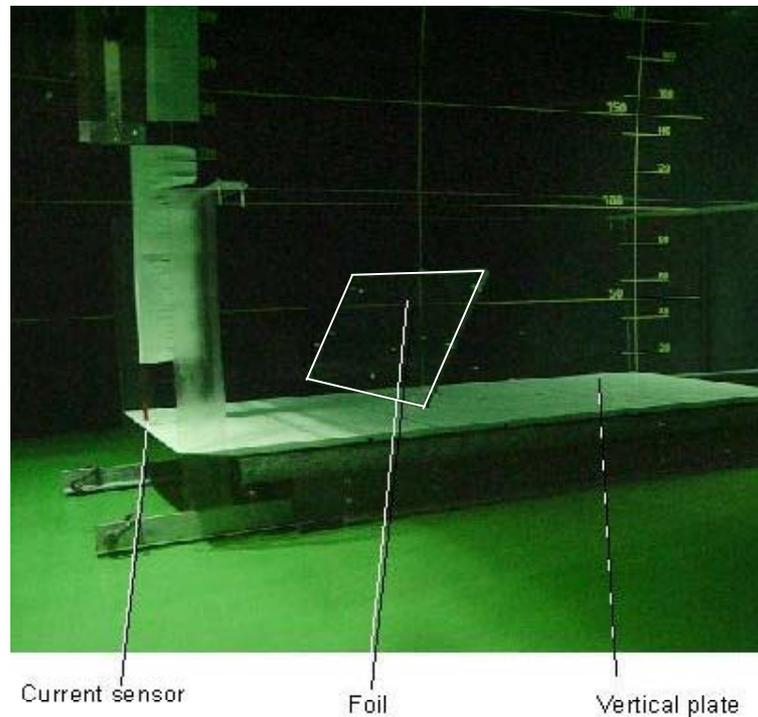


Figure 11 **System used for tests in flume tank**

The reference plate, which is horizontal, was also chamfered with 7° angle on the front part. The system was set in front of the third window of the flume tank in IFREMER Boulogne sur Mer, to be able to film the flow around the foil.

A second flume tank test was carried out with another system which tended to increase the lifting effect of the foil (Figure 12). This new system was equipped with two laterally convergent plates under the foil.

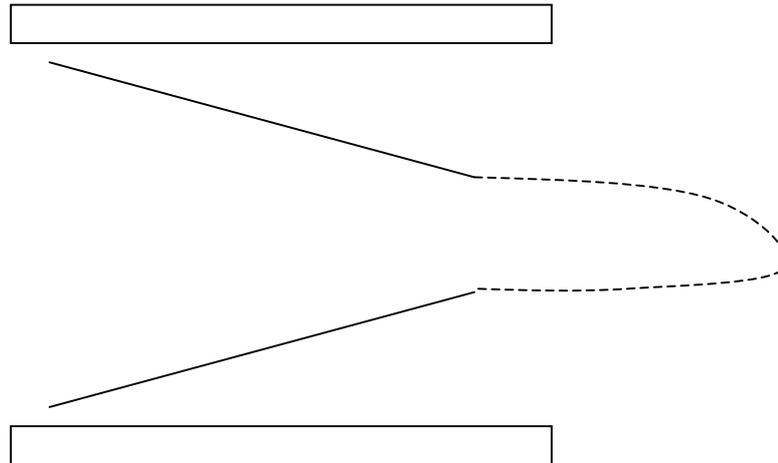


Figure 12 Top view of the dredge with convergent lateral plates, basket and skis.

Configurations

Several configurations were investigated:

- Two foil lengths: 200 mm and 400 mm
- Two gaps between the lowest edge and the tank bottom: 50 mm and 150 mm
- One water speed: 1 m/s
- 3 incidences: 15°, 30° & 150°.

Two current meters were used:

- Propeller current meter
- Laser Doppler velocimetry

The configuration summary is in the Table 5:

Table 5: Configurations tested in the flume tank.

Foil length [mm]	Gap [mm]	Incidence [deg.]	Convergent	Water speed Measurement	Case Flume tank	Case model
200	50	30	[-]	Propeller	1	1
200	50	150	[-]	Propeller	2	2
400	50	15	[-]	Propeller	3	nm
400	50	150	[-]	Propeller	4	6
400	150	15	[-]	Propeller	5	nm
400	50	150	[-]	Laser	6	6
400	50	15	[-]	Laser	7	5
400	50	30	[-]	Laser	8	7
400	150	30	[-]	Laser	9	nm
400	150	15	[-]	Laser	10	nm
400	150	30	[20°]	Laser	11	nm

Note; nm= not modelled

Procedures

In order to obtain water speed profiles were measured during the flume tank trials the trial procedure was as follows:

1. Positioning in the flume tank
2. Water flow start.
3. Stabilisation of the flow.
4. Start of the Laser Doppler velocimetry.
5. Water speed measurement
6. Water flow stopped.

Numerical model

During this study of the flow around a foil with the numerical model, the first objective was to verify that the result obtained with this model was close to those obtained in the flume tank in the same configuration. This first part was to verify the quality of the numerical model results. The second and final objective was to verify the global flow around the foil and to propose a foil configuration, which gives the best effect in term of lifting scallops.

The configuration studied is described in Figure 13. It consisted of a moving plane near the seabed.

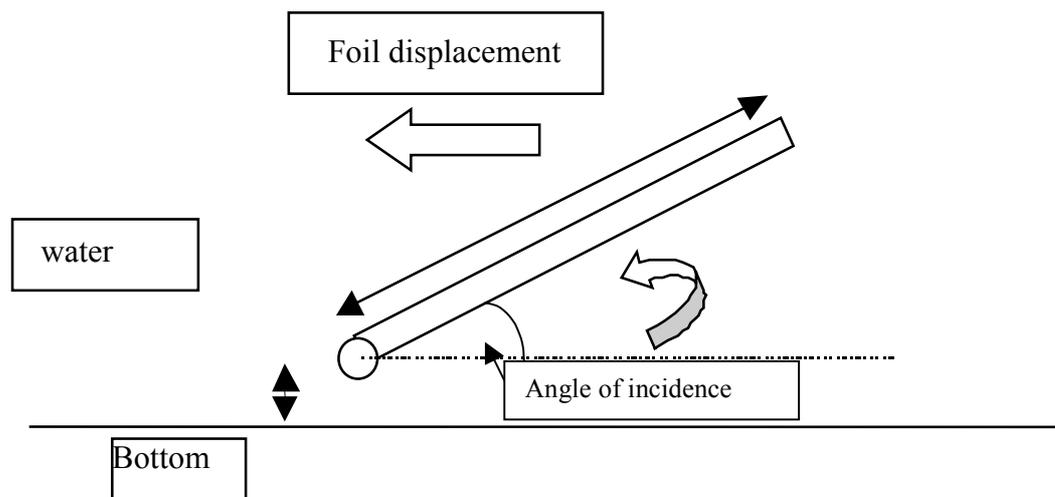


Figure 13 Foil configuration studied with the numerical model.

Foil configurations

The foil configurations tested corresponded to those tested during the flume tank tests (Table 6) together with some more variants.

Table 6 Foil configurations studied with the numerical model

Case	Foil length (mm)	Distance foil/bottom (mm)	Incidence (°)
1	200	50	150
2	200	50	30
3	400	150	15
4	400	150	30
5	400	50	15
6	400	50	30
7	400	50	150

The leading edge of the foil was modelled as a cylinder, and the geometry of the system was the same as those used in the flume tank:

Water speed: 1.01m/s

Foil thickness: 15mm,

Foil width: 1000mm.

Initial hypothesis

The Reynolds number was calculated to estimate the flow type. In the foil configuration chosen this number equals 400,000 which is close to the critical value (500,000) between the laminar and the turbulent flow, though more laminar than turbulent.

The flow seemed unsteady around the foil. However, the calculation of such a flow is time consuming, so the study was carried out in the first step in steady mode so as to reduce the calculation time and to have an idea of the behaviour of the flow around the foil.

The geometry of the system used during the flume tank trials with the two lateral parallel plates was adapted to the calculation in two dimensions. This calculation in 2D is less time consuming than in 3D.

The best solution for the calculation of such problem would be to calculate the flow in unsteady turbulent flow in 3D. Due to time consuming calculation and computer ability, the calculations are carried out to compare laminar and turbulent flows, next steady and unsteady flows, and finally 2D and 3D configurations.

FLUENT software

The FLUENT software was used for the numerical simulations. FLUENT provides a wide array of advanced physical models for turbulence, combustion, and multiphase applications. The fully unstructured, mesh-based technology in FLUENT is the industry standard, established over time and multiple validations. FLUENT's technology ensures a fast and robust solution process. The parallel computing capabilities give an added performance boost.

Problem set-up, monitoring, and post-processing are easy with FLUENT's common-sense interface and switch-on-the fly interactivity. Solution-based mesh adaptation is just one of many productivity-enhancing features.

Results

Flume tank tests

Several configurations of the foil were investigated, with the parameter defined in Table 5. The results of case 9 are given in the Table 7.

Four speed profiles were measured for each configuration. These profiles were defined by A, B, C and D. The best effect was obtained with a foil length of 400mm, a gap of 150mm and an incidence of 30° (Figure 15).

Table 7 Water speed measurement. Foil length = 400 mm, Gap = 150 mm, Incidence = 30°, No convergent, Laser current meter

Z [mm]	Profile A V [m/s]	Profile B V [m/s]	Profile C V [m/s]	Profile D V [m/s]
560	0,97	1,2	1,2	1,08
510	0,96	1,19	1,2	1,02
460	0,93	1,2	1,28	1,08
410	0,89	1,25	1,16	0,86
360	0,83	0,87	0,72	0,76
310	0,81	-0,3	-0,12	0,62
260	0,74	-0,29	-0,31	0,5
210	0,64	-0,07	-0,14	0,63
200	0,59	-0,05	-0,06	0,65
190	0,56	-0,02	0,12	0,64
180	0,53	0,03	0,19	0,73
170	0,49	0,1	0,44	0,69
160	0,46	0,29	0,4	0,66
150	0,47	0,34	0,62	0,74
140	0,68	0,49	0,67	0,79
130	0,86	0,68	0,75	0,81
120	0,95	0,87	0,91	0,76
110	1,03	0,95	1,01	0,71
100	1,03	1,15	1,02	0,82
90	1,05	1,3	1,09	0,8
80	1,05	1,35	1,19	0,79
70	1,07	1,44	1,24	0,73
60	1,09	1,43	1,29	0,78
50	1,07	1,47	1,24	0,76
40	1,09	1,46	1,22	0,65
30	0,65	1,03	1,09	0,62
20	0,19	0,31	0,32	0,4
10	0,04	0,06	0,09	0,11
0	0	0	0,01	0

Figure 14 shows the effect of the convergent lateral plates, more precisely it shows a comparison between water speed profile of a foil with and without the convergent plates.

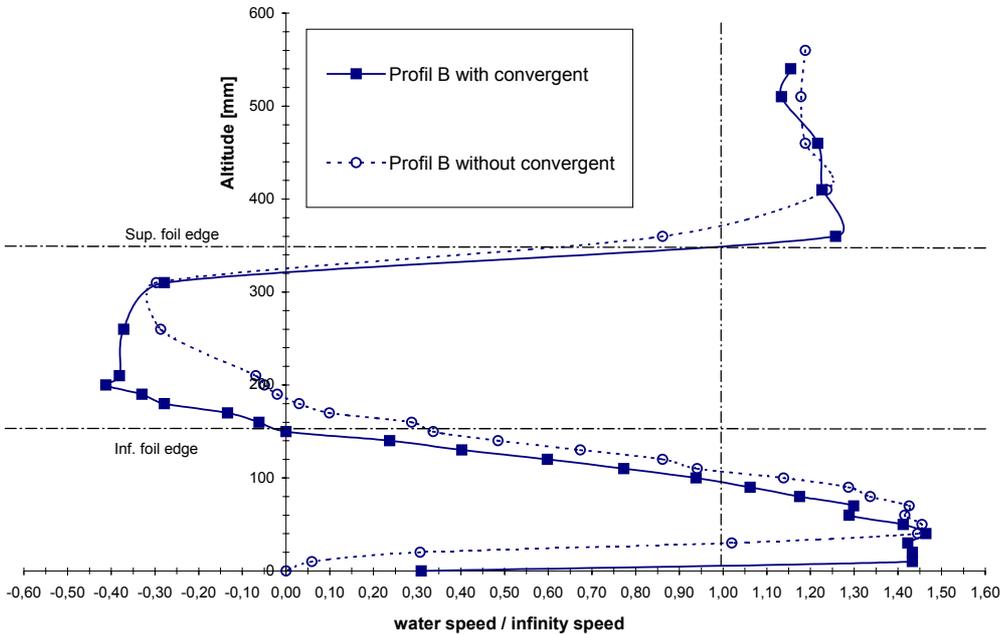


Figure 14 Foil system; Comparison between water speed profile with and without convergent plates see Figure 6 for design of convergent plates

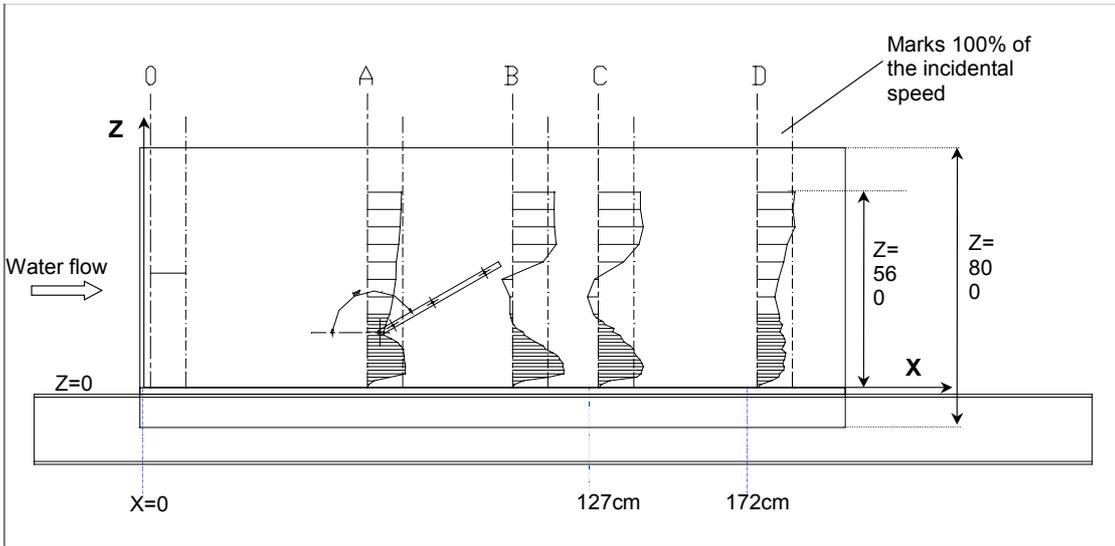


Figure 15 Speed profile positions. Foil length: 400mm, gap: 150mm, incidence: 30° measured behind the foil as in Figure 13

Numerical model; 2D study

Size of the calculation box

The calculation was done in a rectangular box around the foil. This box has to be large enough to have no influence on the calculated flow. The sides of the box were parallel or perpendicular to the flow at infinity and such the speed was constant on the box sides (Figure 16). The distance between the front part of the box and the foil was 0.8m. The length of the box was 4m. The height of the box is 1.25m.

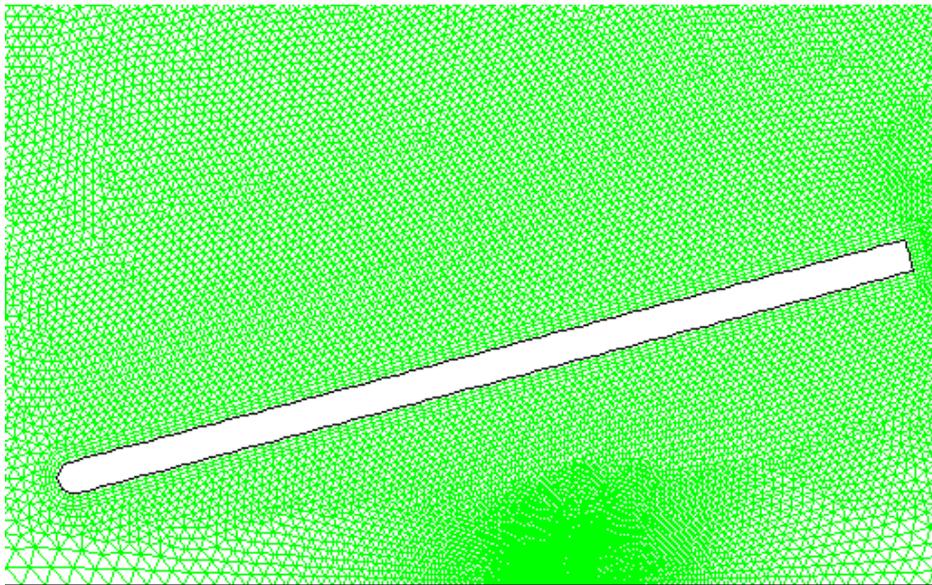


Figure 16 **Meshing just around the foil**

Meshing of the box

Because the flow was between laminar and turbulent, a structured meshing was carried out around the foil to follow the flow. For the remaining parts of the box an unstructured meshing with triangles was used (Figure 16).

Calculation cases

The conditions of calculation are defined in Table 8.

Table 8 Numerical conditions of cases calculated with the numerical model.

case	Reynolds	geometry	Cells nb.	Turbulence model	Convergence criteria
1	20000	2D	95000	Laminar	Cl, Cd
2	20000	2D	95000	Laminar	Cl, Cd
3	40000	2D	76000	Laminar	Cl, Cd
4	40000	2D	80000	Laminar	Cl, Cd
5	40000	2D	63000	Laminar	Cl, Cd
6	40000	2D	68000	Laminar	Cl, Cd
7	40000	2D	73000	Laminar	Cl, Cd
8	40000	2D	68000	k-ε	10 ⁻³
9	40000	3D	40000	Laminar	10 ⁻⁴

Steady laminar flow

Because the flow is unsteady due to the detachment, the usual convergence criteria are not reached especially for the continuity equation. In fact a classical well-adapted NACA wing shows a large detachment at 16° incidence (Abbott and von Doenhoff 1980). The convergence criterion was applied to drag (Cd) and lift (Cl) coefficients. Nevertheless, these two parameters are subjected to oscillating variations to the unsteady phenomenon.

The calculation has been made for the cases of Table 9. The results for the pressure are given in the Figure 17 for the case 4 of Table 9.

Table 9: Physical conditions of cases of calculation.

Case n°	Length (mm)	Distance (mm)	Angle (°)
1	200	50	150
2	200	50	30
3	400	150	15
4	400	150	30
5	400	50	15
6	400	50	30
7	400	50	150

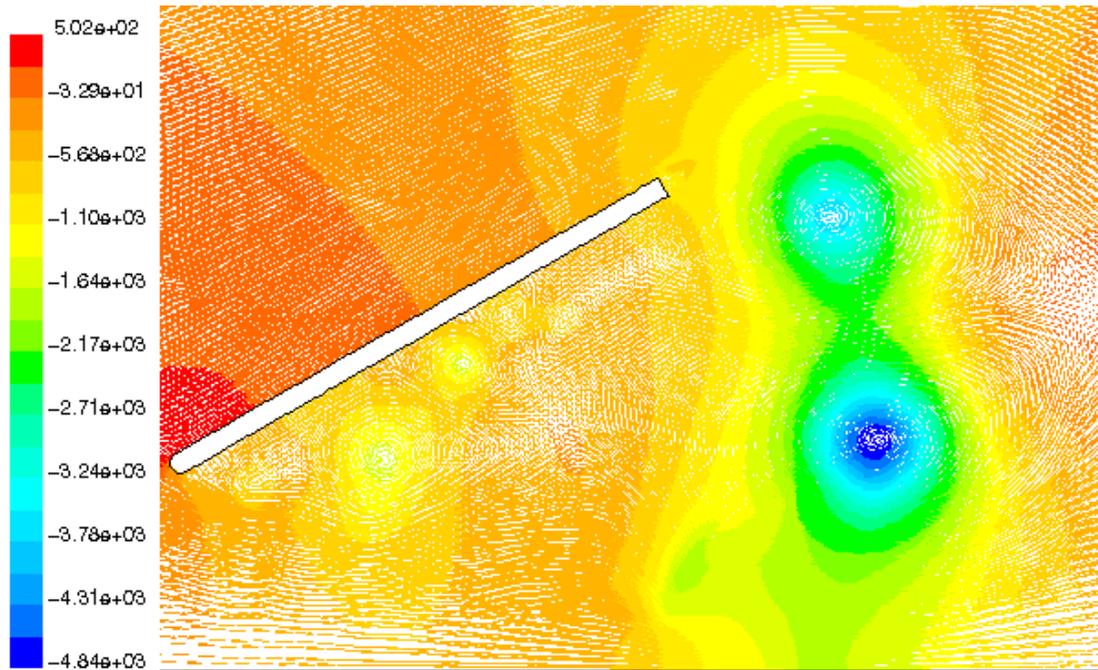


Figure 17 Case 4, velocity vectors coloured by static pressure (Pa). 2 vortexes are in the wake of the foil.

Case number 5 was judged the most efficient. Case 6 shows also a positive situation. It seems that the longer the foil and the closer to the bottom, the more efficient is the configuration.

A rough classification of the cases can be made according to the efficiency (Table 10).

Table 10 Classification of efficiency of cases.

Classification	Case
1	5
2	6
3	3
4	4
5	2
6	7
7	1

Steady turbulent flow

A calculation in turbulent mode was achieved to validate (qualitatively) the results obtained with the laminar mode for case number 6. The results are shown on Figure 18.

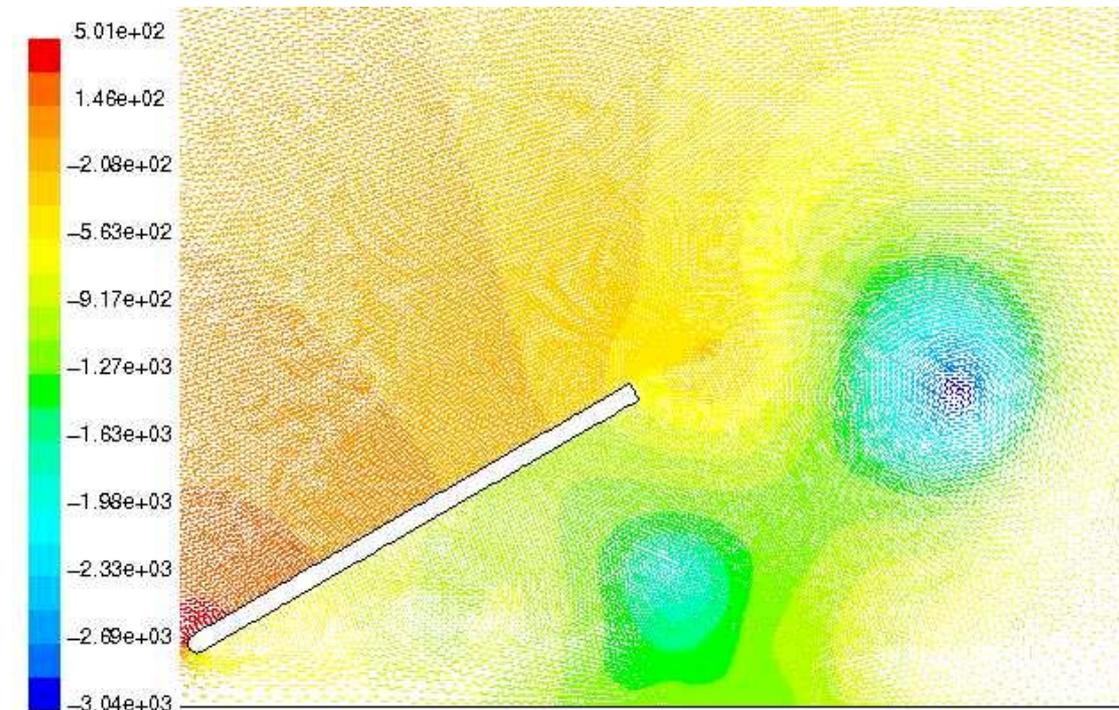


Figure 18 Case 6 steady turbulent mode, velocity vectors coloured by static pressure (Pa). 2 vortexes are in the wake of the foil.

The comparison with the results, calculated with the steady laminar mode shows that the stopping pressure is far better in turbulent mode: 501Pa against 496Pa in laminar mode. The objective stopping pressure is 500Pa ($=\frac{1}{2} \rho V^2$, with $\rho = 1000\text{kg/m}^3$, $V = 1\text{m/s}$). Actually, the turbulent calculation converged even though the convergence in the laminar mode is not obvious.

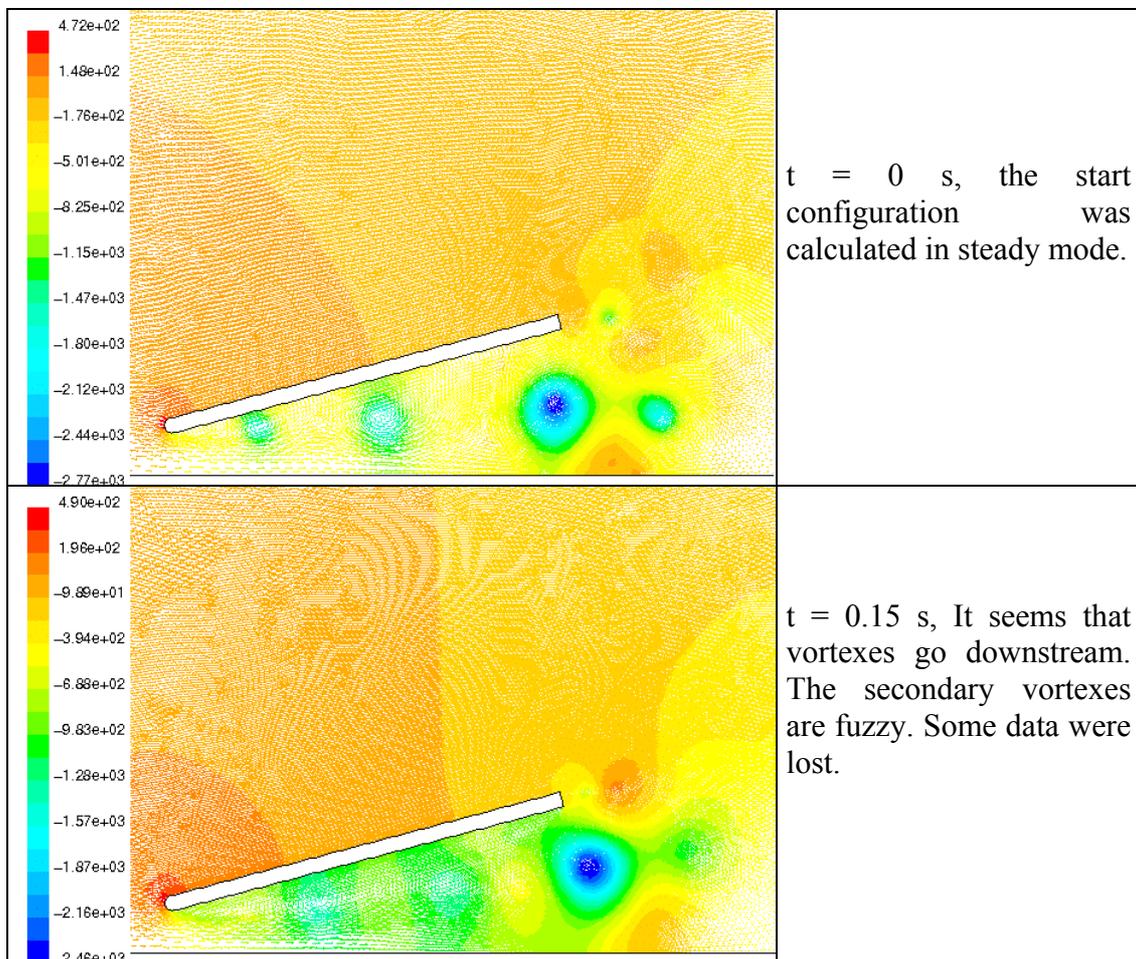
The vertical water speed reaches different maximum: 0.93m/s in turbulent mode and 1.78m/s in laminar mode. This means that the two modes are quantitatively different. Nevertheless, the results of the two modes are qualitatively the same.

Unsteady laminar flow

This calculation requires a time step. Unfortunately the characteristic time step is not known, which is necessary to model the physical phenomenon.

Nevertheless, the results obtained with a time step of 0.05s give an idea of the behaviour of the flow. The vortexes had a rotation speed around 30rad/s, which means that the time step used is probably too large. A study with a time step of 0.01 s would be better but much more time consuming.

The results are presented with several sequences on Figure 19 and are as follows:



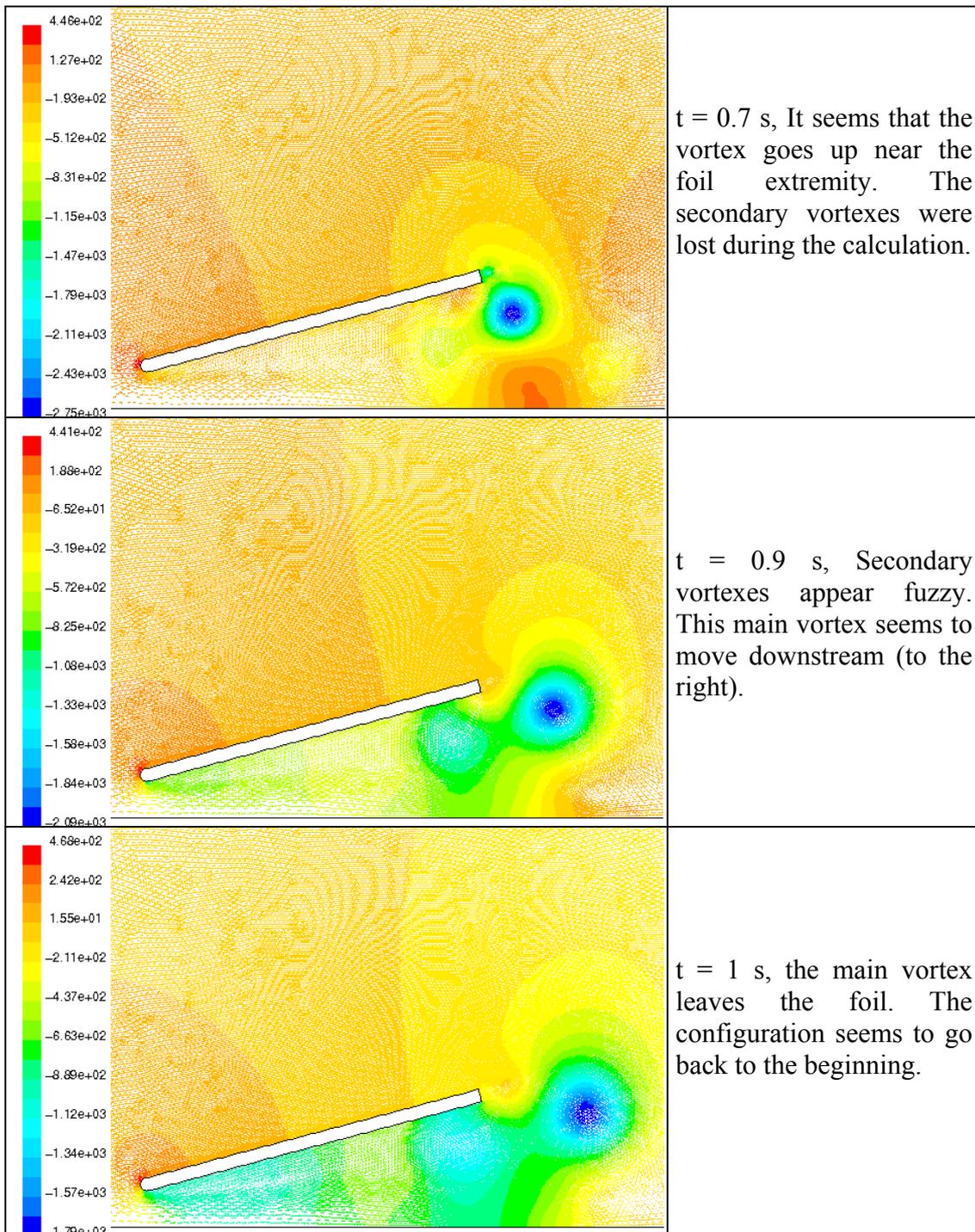


Figure 19 Case 5 Unsteady laminar mode, velocity vectors coloured by static pressure (Pa). The vortex leaves the foil.

Numerical model; 3D study

The 2D study seems to be adapted to the cases studied in the flume tank, a 3D study in steady mode of case 5 was carried out.

Unfortunately the results (Figure 20) were quite different of those obtained in 2D. In this model the detachment and the vortices were reduced. This arises from the meshing which is courser in 3D than in 2D.

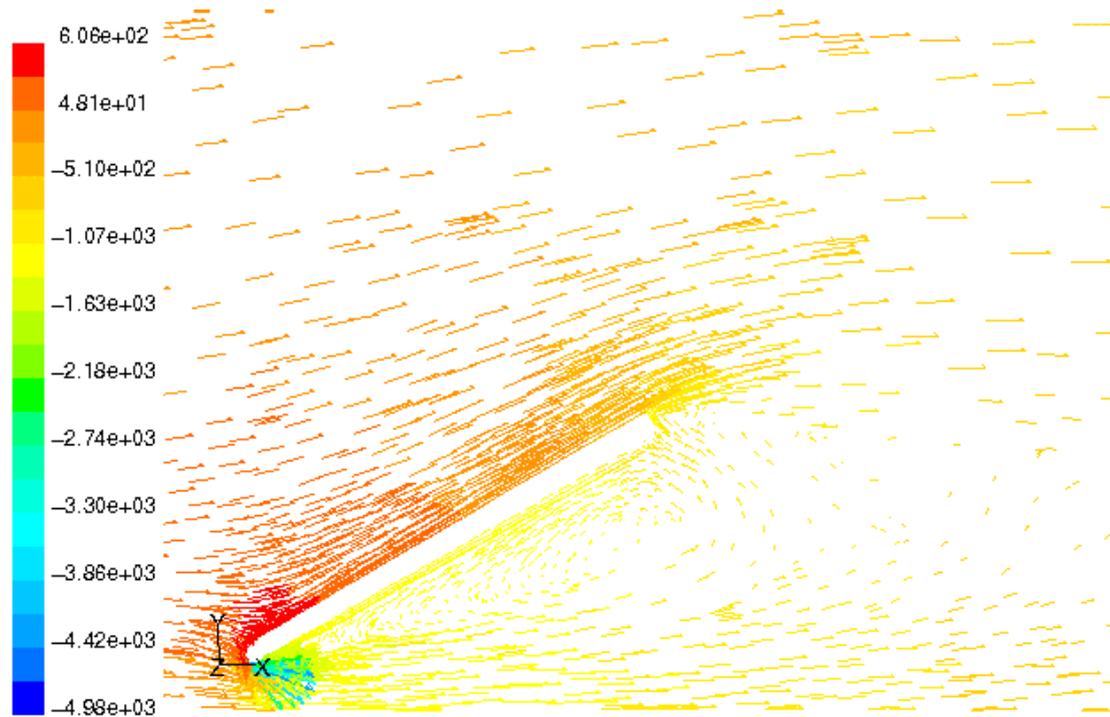


Figure 20 Case 5 Steady laminar mode in 3D, velocity vectors coloured by static pressure (Pa).

Discussion

Flume tank trials

Some observations arise from these flume tank tests:

1. The boundary layer is clearly visible on profiles made with the laser sensor. This boundary layer (zone near the bottom where the water speed is close to zero: speed of the bottom) is a few centimetres high. Obviously, in these tests both the foil and the bottom are immobile, so the speed limit on the bottom is zero. In real fields, the dredge moves along the seabed, so the speed limit will be 1. An improved simulation would include a moving seabed, which would give a better indication of the boundary layer effects.
2. The speed profiles show that the water speed can be superior to 1 (foil speed in actual field). This excess of speed can reach 46% for a foil length equal to 400mm, an incidence angle equal to 30° and a gap of 150mm (profile B, Figure 14). This water speed seems sufficient to remove the scallops from the seabed. The water speed could be obtained with a towing speed of 3 knots, which is reasonable.
3. The water flow which passes under the foil remains along the bottom. This means that if the scallops are carried away by the flow they will remain close to the bottom and may well pass under the basket and thus not be collected.

In summary, it seems that the excess of water speed will be sufficient to remove the scallops from the seabed, but the lifting effect seems insufficient for them to be collected even with convergent lateral plates (Figure 12).

The two main drawbacks of these trials are that it is difficult to “visualise” the flow around the foil, and more precisely whether vortex are created behind the foil, and that a great deal of repetitive measurements will be necessary to get a “view” of the phenomenon.

The two drawbacks of the measurement with the propeller current meter are:

1. The propeller measures an average of the speed due to the size of the propeller and the time required to measure.
2. The propeller reduces the passing section of the water due to the size of the propeller, which means that the speed increases owing to the constant debit.

Numerical model

Due to the difficulty in changing easily the configuration of the foil when conducting flume tank tests, a numerical model was used in the same configuration as that used for the flume tank tests. The aim was to find the best configuration in terms of scallops lifting. The configuration with the two lateral parallel plates, calculation in 2D was adapted.

The results turn out to be different quantitatively between laminar, turbulent steady and unsteady modes. Nevertheless, the laminar steady calculation of the cases seems to show that a foil of 400mm, positioned at 50mm from the bottom with an angle of 15° and a cylindrical leading edge presents the best qualitative efficiency (case 5). It seems clear that the longer the foil and the closer to the bottom the larger are the vortexes. For an incidence of 150° (case 7) the vortex is quite large but too far from the bottom.

To summarise, this investigation showed that the water speed under the foil is probably sufficient to remove the scallops from the seabed but insufficient to lift the scallops off the seabed.

Application to dredge design

Although these results suggested that these configurations were not adequate to move the scallops into the collecting basket they do not rule out a role for the use of hydrodynamic effects to move the sediment from around the scallop to expose them for collection. It is possible that this effect may already be happening in some foil dredges.

Experiments, using flume tank trials and numerical simulations on different shaped foils or cylinders could be an approach to be considered. Other factors of dredge design would be considered such as the dredges ability to adapt to seabed features and also the means to collect the scallops from the seabed.

3.3.1.3 TOOTH DESIGN

A modified tooth design was tested on the beach on the Holderness coastline. The design was based on principles used for submarine ploughs and incorporated a backward raked share designed to give a vertically downward component on the leading edge of the tooth, pulling it into the ground. Depth can be controlled by raising or lowering the front of the tooth, as the length will always try to return the tooth to an equilibrium horizontal position. The configuration of this tooth design is shown in Figure 21

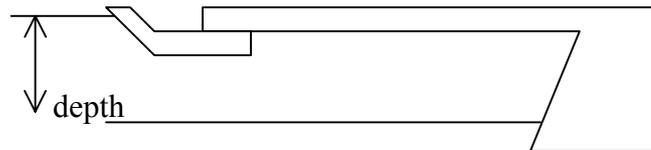


Figure 21 Drawing and photograph of trial dredge teeth

The trials indicated that the design reduced the apparent disturbance to the seabed, but they also indicated that stones could become caught, despite the ability of the teeth to ride up, or spread out laterally when a stone was encountered. No offshore trials were attempted with this configuration.

3.3.1.4 Water-Jet (Hydraulic) dredges

Introduction

In February 2000, a French delegation paid a visit to IRPRM-CNR plant in Ancona to investigate the use of low-pressure water-jets (for scallop dredges) as used on the Italian turbo soffianti dredges used along the Adriatic Sea. These dredges implement water-jetting for clam dredging. The two French attendees were André Le Gall (scallops dredging professional) and Christian Danioux (Engineer at IFREMER). The objective was to study the hydraulic dredges used for *Venus gallina* dredging in order to adapt the same technique to scallop dredging in the bay of Brest (France).

Instead of the usual teeth, water jets are used on the clam dredges so as to dig the seabed (Figure 22) and thus lift the clams (see [Section 3.1.1.4](#)). The use of these water jets could be adapted to scallop dredges. The jet would drive the water from around the shell (Figure 23), and it is assumed that this current would lift the scallops from the seabed. This use is quite different from the clam dredges.

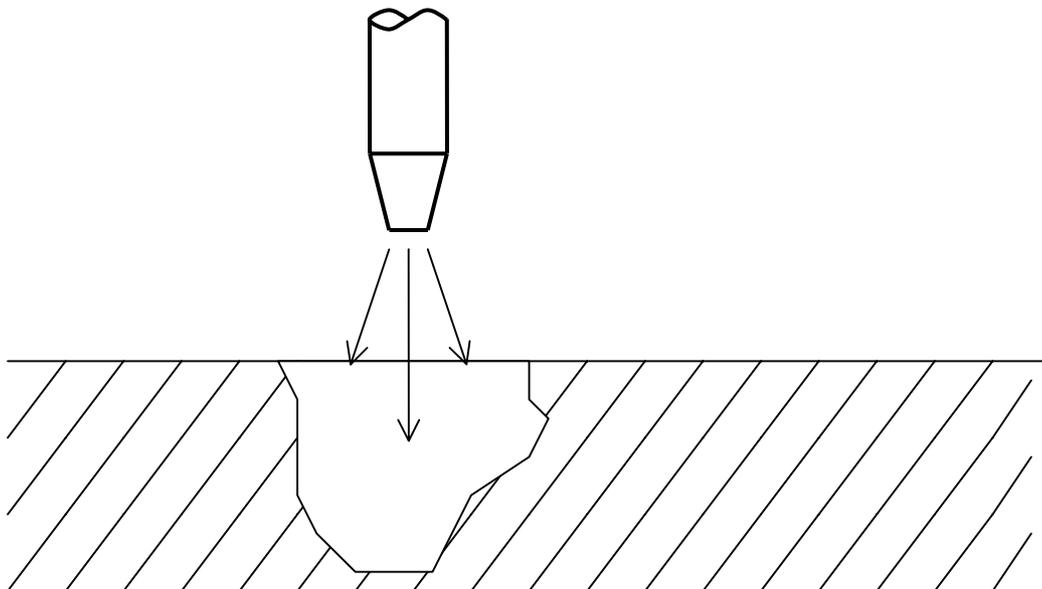


Figure 22 The water jet used on clams dredge digs out in the seabed.

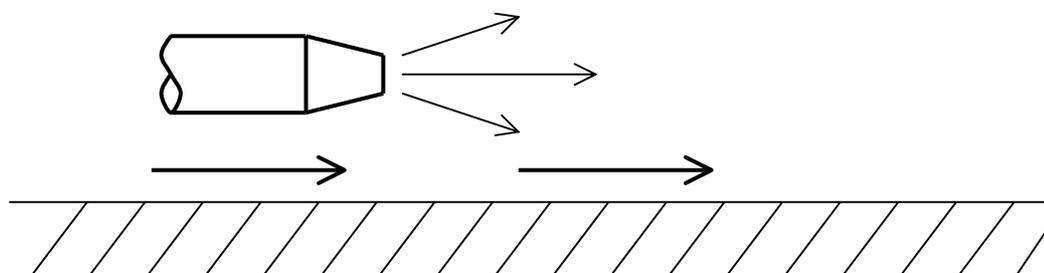


Figure 23 The water jet (slender arrows) on scallops dredge is expected to drive the surrounding water (bold arrows) and lift the scallops.

Materials and method

During a visit to IRPEM-CNR in Ancona, a discussion with Carlo Frogliani (biologist) co-ordinator of the visit was undertaken and also a literature search and Video viewing were undertaken. Meeting with professionals and expected boarding of a dredge boat. This boarding was not possible because of strikes, bad weather and effort controls. An opportunity was taken to view the dredging system and to visit a blacksmith involved in hydraulic dredge fabrication (Campanelli firm at Marotta di Mondolfo). There are some 8 blacksmiths along the Adriatic coast involved in this business.

Results

Chronological account of clam dredging in the Adriatic

In the past *Venus gallina* was dredged from rowing boats with 2 or 3 persons. These boats used anchor with 100m cable and crane. The manual process was carried with the vessel travelling astern: the anchor was dropped, the cable was uncoiled, the 1.5 m wide dredge was laid on the bottom and the dredge was pulled by coiling the cable with the crane. The cleaning and sieving of clams were manual. The catch was around 200 kg/boat/day.

Hydraulic dredges have equipped dedicated boats since the early 1970s, taking after the system in use in the United States of America. These boats are operated as the rowing boats; travelling astern: dropping the anchor, uncoiling the anchor cable, lying the dredge and coiling the cable. This gives a particular feature to these boats with the dredge on the front and supported by a metallic frame (Figure 24).



Figure 24 Classical Italian fishing wooden boat. The metallic frame on the front supports the dredge.

Present practice

At the present time, the limitations are:

- Boats: - length 10 m,
 - power 118 kw (150 hp).
- Dredge: - width between 2 and 3 m depends on the boat length,
 - weight 800 kg.

In Ancona, on account of the boat configuration motorised dredging astern is permitted. Many boats do not use their anchor, which remains onboard without being linked to traction cable.

The actual fishing conditions are the following (See also [Review Section 4.6.1](#)):

- quota: 600kg/boat/day,
- duration: 3h/day and forbidden on Saturday and Sunday,
- fishing period: 4days/week in summer, 5days/week in winter, closed season during June and another month between May and September.

Technical description

In Ancona the dredging is carried out between 3 and 11 m deep. The dredge is lifted on the metallic frame and empties the catch into the tank onboard. The clams are collected by an endless screw and fall on the vibrating sieve. The sieve is equipped with lead seal to avoid fraud on clam size. The endless screw and the sieve are water fed to facilitate the sorting and to ensure that the clams clear the machine. The dredge makes a first sieving at sea: the bars of the catch bag are spaced out 11mm apart. The sorting is assisted by 2 or 3 manifolds located inside the dredge. These manifolds come in addition to the water-jet manifold used for digging the clams. The water-jet manifolds inside the dredge push the clams to the dredge back and clear them (Figure 25).

In most dredges a pump, connected to the boat engine, supplies the water, the intake being under the boat. Only few dredges are equipped with a centrifugal pump directly fixed on the dredge; in this case the water intake is forward of the dredge. The pump is actuated by a hydraulic motor of 30kw (40 hp) power. The water pressure ranges from 2.5 to 3.5 bars for a towing speed around 2.5 knot. When dredging with the only engine of the boat, this pressure is reduced to 1.8 bar for a towing speed between 2 and 3knot. Whatever the type of dredge, each of them is equipped with a lead seal valve which limits the pressure.

Discussion

The equipment used for fishing *Chamelea (Venus) gallina* cannot directly be adapted for scallops dredging. Only few elements can be used: the skis which are adjustable in height, and the cheap water injectors. On another hand, the use of an exhaust pipe, not cumbersome for shallow water (<10 m), can pose problem in deeper sea. Should it be the case the use of a pump directly fixed onto the dredge would be more adapted.



Figure 25 Water-jet manifolds on the top of the dredge

3.3.2. Implementation of novel dredge design features

Three dredge principles were investigated as part of this project (foil, Magnus effect and hydraulic). The analysis of the investigations shows varied results.

The **Magnus effect** principle was investigated by the evaluation of three prototypes tested at sea. This system did not reach sufficient efficiency. The investigations of **tooth design** did not proceed further than a prototype; it was recognised that there was a requirement to investigate the mechanism of capture by spring teeth ([Section 3.1.1.1](#)) before the tooth design could be improved.

The flume tank tests and numerical simulations have largely investigated the **foil system**. The flume tank and numerical model results are not exactly comparable. Furthermore, the optimal configuration was not found. This principle seems to be very promising, because the system does not require energy except the towing speed of the boat and because the first value of water speed created by the foil seems sufficient to remove the scallops from the bottom. Nevertheless, there was insufficient lifting effect on the scallops. This principle requires further investigation, first of all with a numerical model able to provide results for different configurations; this approach being easier than flume tank tests, even though it is computer time consuming.

The **hydraulic or water jets** solution was investigated by a visit to the Italian dredges used along the Adriatic Sea which implement this system. The system seems suited to reduce the effect of the dredge on the bivalves; thus, this principle was chosen for the prototype dredge. The fishery in which the test was carried out was in the scallop fishery in Bay of Brest (Western France).

3.3.2.1 FRENCH HYDRAULIC (WATER-JET) DREDGE for Scallops

Introduction

Once the principle of the prototype dredges was chosen (hydraulic or water-jets), the prototype was designed based on a classical dredge used for scallops in France. The blade of the classical dredge was replaced by a water jet system. This dredge used the same components as the hydraulic dredges used in other fisheries, particularly in Italian fishery for clams in the Adriatic Sea (see above). To build the prototype the blacksmith used a classical dredge identical to those used for the scallop fishery in the Bay of Brest.

Components

The prototype was made of several parts:

- the dredge with the water-jets,
- motor-driven pump for water injection,
- suction pipe and exhaust pipe linked to the dredge,
- flow and pressure gauges.

This dredge is made of a metallic frame, supported by 3 skis, on which are fixed the various accessories (Figure 26).

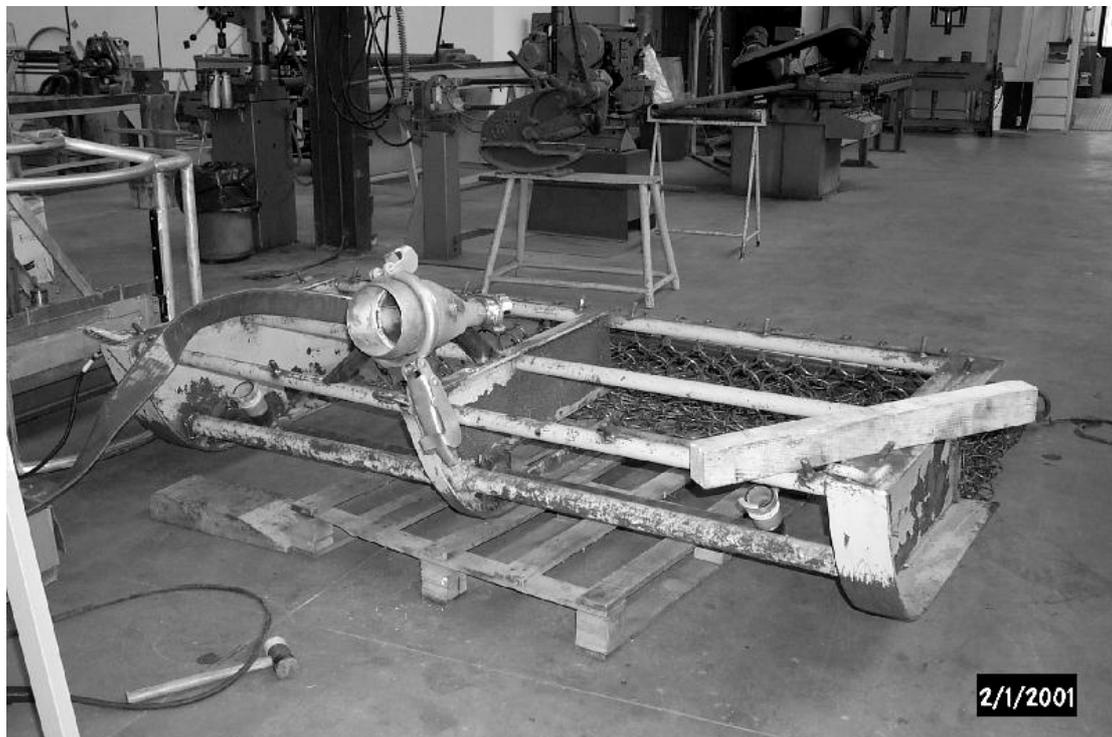


Figure 26 General view of the prototype dredge

The frame width was 2.24m for an effective working width of 1.80m. Its height was 0.50m. A dredge belly (Figure 28), made of rings of 92mm internal diameter was used for collecting the scallops, which is the standard for the Bay of Brest. This bag

includes a quick opening system to pour the scallops on to the deck. An injector manifold which was sliding and adjustable is used to remove the scallops from the sea-bed (Figure 27). During the tests two diameters of injectors were used: 36 injectors with a 5mm diameter and then 37 with a 7mm diameter. These injectors were made of bronze and were supplied from Italy (Campanelli Arideo et Giannino – Via 25 Aprile, 1 – 61035 Marotta di Mondolfo PS).

Two flexible pipes of diameter 50mm are connected to the injector manifold and to a join which is connected to the exhaust pipe. Irrigation connectors make all the connections between the pipes. A piece of trawl net covers the top of the metallic frame, to prevent the scallops escape (Figure 28).

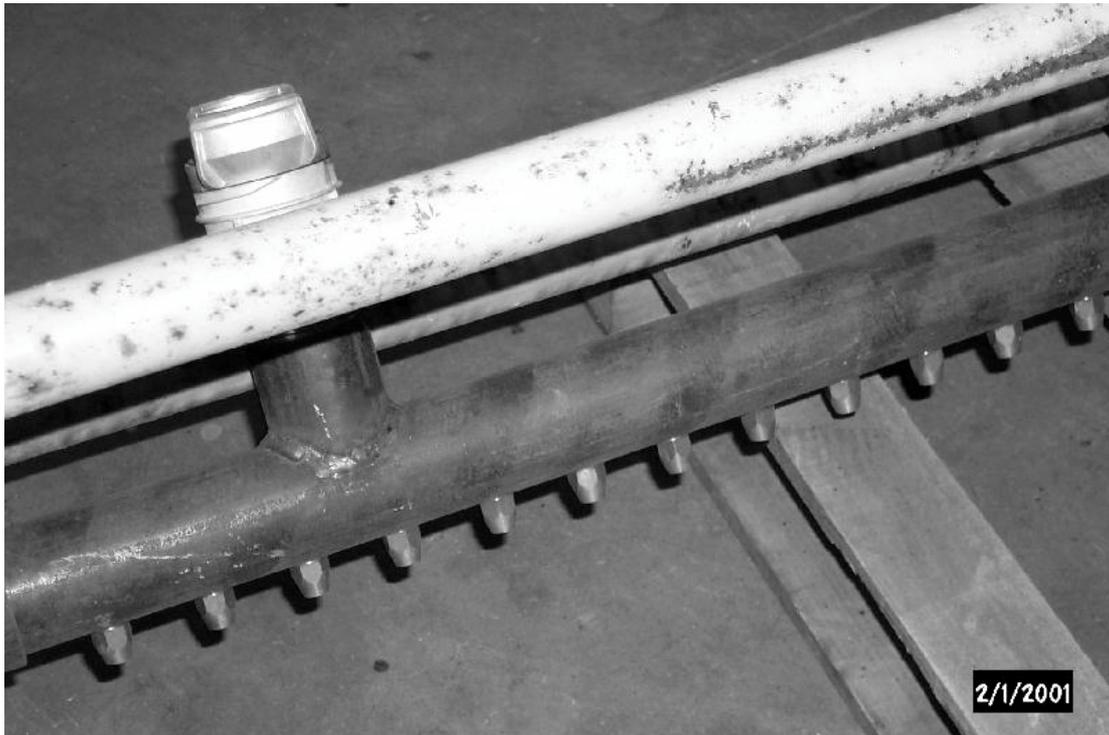


Figure 27 Injectors manifold of the prototype dredge.



Figure 28 The piece of net on the top of the dredge prevents the scallops escape.

The motor-driven pump for water injection

A diesel engine, a pump and accessories were assembled on a frame, which also housed a fuel tank (Figure 30). The pump was an AZCUE AN 100/200 with a cast steel barrel with a stainless steel shaft. It ensures a flow of 250 m³/h with a pressure of 40m of water (Figure 29).

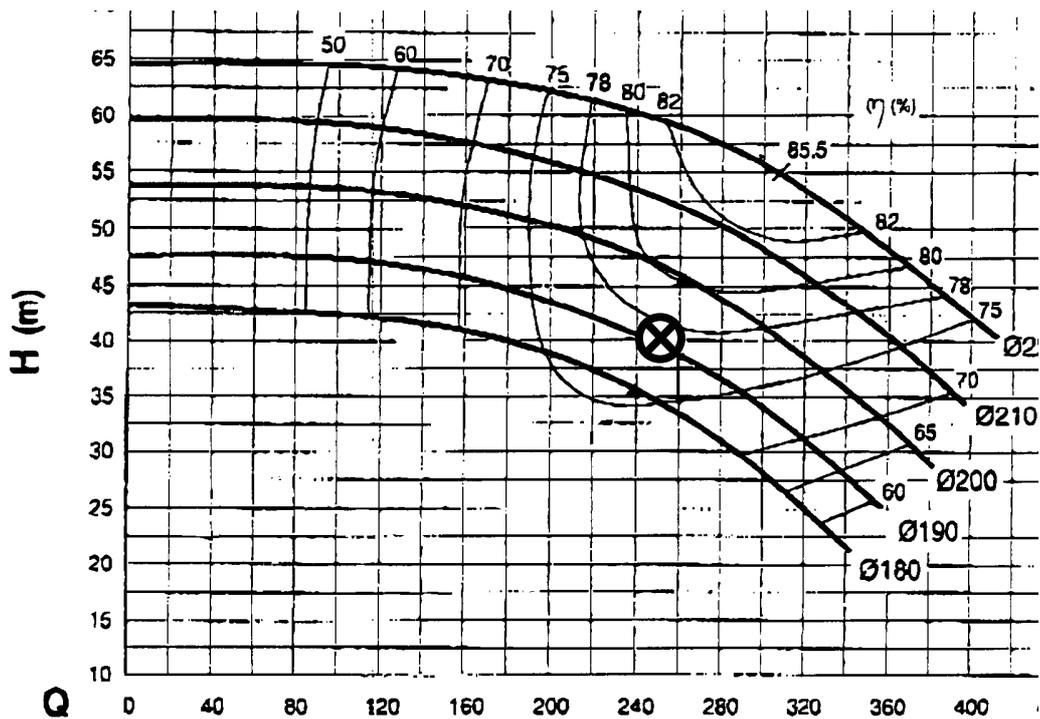


Figure 29 The operation point of the pump is 40 m and 250 m³.

The pump was driven by an IVECO diesel engine of 60 hp and with water cooling.

The suction pipe and exhaust pipe were linked to the dredge. The suction pipe was of the semi-rigid ringed type, 150mm diameter. Its length has been tuned to the boat (5m, Figure 30). The pipe is equipped with a filter to avoid large particles entering the pump, and of a inlet valve to avoid loss of priming. An irrigation connector maded the connection of the pipe to the pump. The exhaust pipe was a flexible made of PVC reinforced with polyester weft. Its diameter is 150mm and its length depends on the dredging depth. Lengths of 10m, 20m and 30m were provided, which could be connected with irrigation connectors.



Figure 30 The pump and the pipes on board.

The flow and pressure gauges

The pump is equipped with systems for measuring parameters such as pressure and flow and for tuning them. A valve is used to vary the pressure and consequently the flow. A pressure gauge on the exit of the pump measures the pressure. A water meter on the exit of the pump is used to measure the flow rate.

3.3.3. Comparative environmental effects of dredge designs

This section compares the environmental effects of the hydraulic dredge developed for scallops in the French fishery with the classical dredge. The dredges were compared in terms of efficiency, damage rates, both in the catch and on the seabed post dredging and stress levels (AEC level see [Section 3.1.3.2-4](#)) in the scallops.

Two designs of Portuguese dredge used for targeting *Callista chione* were compared, one a traditional type with a net bag and the other a novel design using a metallic grid structure for retaining the catch. Comparison was made in terms of relative efficiency and selectivity, damage rates both of animals in the catch and those captured in the cover. These parameters were examined in relation to tow duration.

Finally, a comparison between 3 dredge types was carried out in the Portugues *Spisula solida* fishery. In this case two traditional dredge types were compared with a novel design using a cage instead of the traditional bag. Comparisons were made in terms of efficiency, effects on benthic fauna, and fishing yields of the three dredge types.

3.3.3.1 COMPARISON BETWEEN HYDRAULIC AND CLASSICAL FRENCH SCALLOP (*PECTEN MAXIMUS*) DREDGES

Introduction

Once the prototype dredge has been built, the tests on this prototype were carried out. Due to the complexity of this dredge the sea trials were done on a research vessel and not on a commercial vessel as foreseen. For the comparison with a classical dredge trial have been carried out on commercial vessel.

Adenylic energetic charge (AEC) ([Section 3.1.3.2-4](#)) was used to measure the stress effect of scallops caught by the prototype hydraulic dredge.

Materials and method

Prototype dredge

The sea trials on the prototype dredge have been carried out on the GWEN-DREZ a Research Vessel belonging to IFREMER (Jean-Michel Delbarre Skipper, Figure 31). The trials began the 12th February 2001 and finished the 23rd of the same month. That means 10 days of boat time, but just 5 days of effective trials due to the complexity of the dredge and the tuning of the system.

The ship specifications:

Length overall: 24.50 m

Overall breadth: 7.40 m

Draught: 3.50 m

Loaded displacement 249 Tonnes

Gross tonnage 106.31

The tuning of the prototype dredge

The pipe, which drives the water to the dredge, had a length (50 m) just longer to the warp (40 m), to avoid the boat towing the dredge by the pipe if it is too short, or the pipe tangling if it is too long. Once the pipe length is tuned, that means that the warp length is still the same, and consequently that the depth must be more or less the same; from 13 m to 21.5 m. This depth involves a constraint on the choice of the scallop grounds used during these trials.



Figure 31 Water jet prototype dredge on board.

The hauls

A total of 35 hauls were carried out. Of these 25 produced valid data, 10 were non valid (4 hauls to ensure the functioning of the dredge, 2 cancelled, 2 for assessment of the seabed, 2 cancelled because of engine over heating). Divers followed Severn hauls; five of these produced valid efficiency data, one was undertaken for the observation of the dredge, one cancelled because of poor following of the track.

The physical characteristics of the hauls were noted. The duration of the hauls was around 3 to 5 minutes, the boat speed was around 1 to 2 knot, which is much less than a commercial boat speed with a classical dredge. For each haul, the date, the number of the haul, the duration, the volume of water passing in the pipe, the pressure of the pump, the speed over the seabed, the depth and some observations (Table 11).

Table 11 Prototype dredge. Physical condition of hauls.

Haul	Pressure (Bar)	Speed (knot)	Depth (m)	ϕ (mm)	Slope Injec.	Observation
1	-	-	-	-	-	Test of the system
2	4.2	1-1.5	17	5	45°	
3	3.5	1.2	17	5	45°	
4	4.1	1.2	17	5	45°	
5	4.2	1.2	17	5	45°	12 injectors are plugged
6	4.2	1.2	17	7	45°	
7	4.2	1.2	18	7	45°	
8	4.2	1.3	15	7	45°	
9	4.2	1.4	14.5	7	45°	
10	4.2			7	45°	Warp traction problem
11	4.2	1.15	14.5	7	45°	
12	4.25	1.6	17	7	45°	
13	4.3	1.1	17	7	45°	Boulders > 30cm
14	4.3	0.6	17	7	45°	Boulders
15	3.6	1	17	7	45°	Divers
16	4.2	1	17	7	45°	Divers
17	4.2	2.2	16.5	7	45°	
18	4.2	1.9	18	7	45°	
19	4.2	2.1	17	7	45°	
20	0.8	1	13.5	7	45°	Engine heating
21	4.1	2	21.5	7	45°	Engine heating
22	4.1	1	20	7	45°	
23	3	2	13.5	7	45°	Engine heating
24	4	1	13.5	7	45°	Divers
25	4.1	2	17	7	0°	Camera jammed on warp
26	-	1.2	16.5	7	0°	
27	4.1	2	17	7	0°	
28	-	2	17	7	70°	
29	-	2	19.5	7	70°	
30	-	2	19.5	7	70°	
31	3	2	17	7	70°	Skis did not touch the bottom
32	3.6	1.4	15.5	7	70°	Drilled exhaust pipe
33	4.1	1	13	7	70°	Divers followed 79 m track
34	4.1	1.5	15	7	70°	
35	4.1	1.5	15	7	70°	

General conditions of Classical dredge trial

This took place over 2 days (3 & 4 April 2001) on the vessel JEANCANI, Jean-Yves Le Goff skipper (Figure 32). 23 hauls with 7 hauls followed by divers: 5 of these are exploitable, 2 not exploitable due to a being at too fast speed poor track. The area was the same as the previously used for the prototype dredge.



Figure 32 Classical dredge, similar to the one used during sea trials.

Conditions of trials

A grapnel was attached on the rear part of the dredge with a thin rope. A buoy was linked to the grapnel by a long rope. The dredge was lowered down to the seabed. Once the grapnel was anchored the fine rope breaks which marks the beginning of the track for the divers. At the end of the haul the dredge was hauled on board. Once the dredge was on board, the divers swam down the track to the grapnel. They collected the scallops along the track.

Measure of stress

The field trial was carried out in the Rade de Brest in February 2001 using the prototype dredge. Twelve separate dredge events (tow length 5 minutes) were carried

out and 80 scallops were collected from the dredge bag in total. Thirty scallops were collected by divers from the dredge track. These animals had come in contact with the dredge but remained uncaught. Twenty scallops were collected by divers outside the track. These animals were the control group. Immediately after collection the animals were rapidly dissected and a small portion (0.2g) of the striated adductor muscle was removed and placed in liquid nitrogen. There it was stored until analysis using the method of Moal et al. (1989a); See also [Section 3.1.3.2-4](#)).

Results

Prototype dredge

The results of the hauls in term of scallop and other species catches are shown in Table 12. The length frequency histogram of captured scallops is presented on Figure 33.

Damage rates

Only 7 broken scallops were recorded in the dredge or on the track. This is a small number relatively to the total number of scallops (835) caught by dredge or divers. The ratio of damage scallops was less than 1% by number. This small number does not permit analysis of the effect of the dredge configuration (injector diameter, injector slope...) on the damage.

Efficiency and density

The efficiency of the dredge is the ratio between the number of scallops caught by the dredge to the number on the seabed before the haul. If the efficiency equals 1, the dredge catches all the scallops on the field, if it equals 0 the dredge does not catch any scallop. This efficiency can be calculated by the hauls treated by divers, from the number of scallops in the dredge and the number collected from the track (Table 13). This efficiency varies from 0.09 to 0.29 with the mean at 0.13.

The density of scallops the sea bed can be also estimated on the hauls observed by divers (Table 13). This density varies largely from 0.03 to 0.32 scallop/m² with the mean at 0.17scallop/m². Unfortunately the density was not constant enough to try to assess the relation between the efficiency and the tuning parameters of the dredge (injector angle, injector diameter, depth, haul duration, boat speed).

Table 12 **Prototype dredge. Number of scallops caught in the dredge or by the divers on the track, broken or intact.**

haul	dredge intact	dredge broken	track intact	track broken
1	9	0	0	0
2	54	0	0	0
3	4	0	0	0
4	32	0	0	0
5	11	0	0	0
6	51	0	0	0
7	38	0	0	0
8	30	1	0	0
9	52	0	0	0
10	0	0	0	0
11	15	0	0	0
12	21	0	0	0
13	0	0	0	0
14	28	0	0	0
15	15	0	37	0
16	16	2	0	0
17	39	0	0	0
18	71	0	0	0
19	38	1	0	0
20	3	0	29	0
21	0	0	0	0
22	21	0	0	0
23	15	0	0	0
24	3	0	33	0
25	1	0	0	0
26	2	0	50	1
27	9	0	0	0
28	9	0	0	0
29	6	1	0	0
30	2	0	0	0
31	0	0	0	0
32	10	0	0	0
33	10	0	49,1	1
34	26	0	0	0
35	28	0	0	0

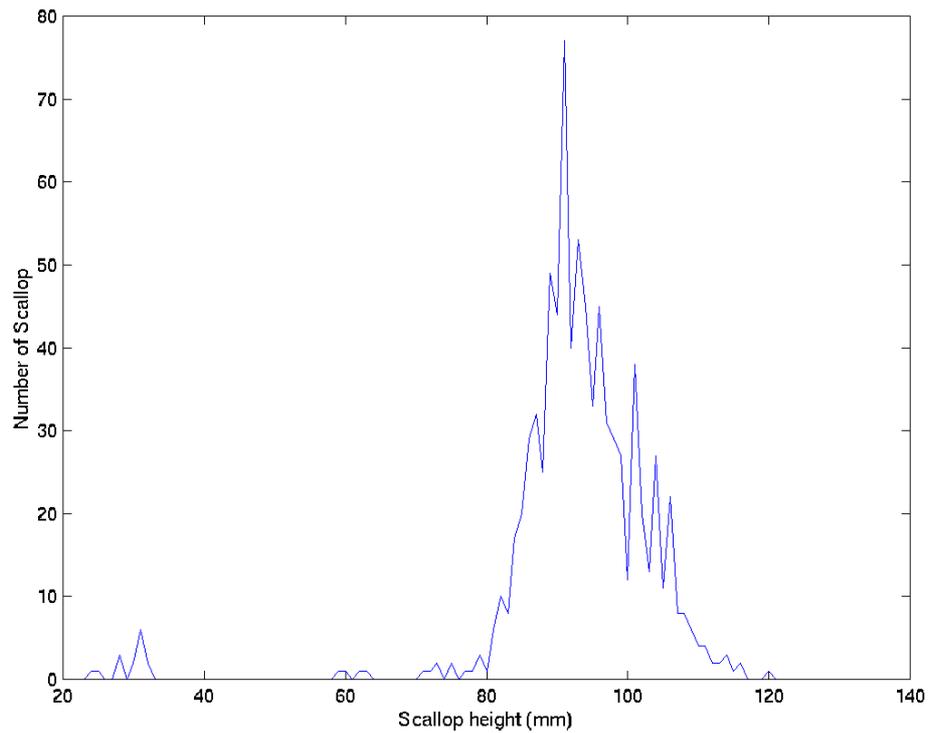


Figure 33 Prototype dredge. Histogram of whole scallops caught by dredge or divers

Table 13 Density of scallops on the sea bed and efficiency of the prototype calculated from the hauls treated by divers.

haul	scallops in dredge	scallops on track	surface m ²	scallops total	density sc/m ²	efficiency
15	15	37	187	52	0.28	0.29
20	3	29	250	32	0.13	0.09
24	3	33	1248	36	0.03	0.08
26	2	51	624	53	0.08	0.04
33	10	50.11	187	60.11	0.32	0.17
mean					0.17	0.13

Classical dredge

The conditions of the trials are shown in the Table 14. The results of the hauls in term of scallops were also noted (Table 14). The length frequency histogram of captured scallops is presented on Figure 34. This histogram is quite similar of the one obtained during the prototype trials (Figure 33).

Table 14 Classical dredge. Physical condition of hauls. Number of scallops caught in the dredge or by the divers on the track, broken or intact.

Haul	Speed (knot)	Durat. (min)	Depth (m)	dredge intact	dredge broken	track intact	track broken	Observation
1	1.2 to 1.7	3	18	28	1	59	0	Divers
2	1 to 2	3	20	35	0	0	0	
3	1 to 1.7	2	19	37	0	21	0	Divers
4	1.2 to 1.7	3	21	35	2	0	0	
5	3 to 3.5	1	28	17	2	0	0	Div., no track
6	3.5	1	23	29	0	0	0	
7	3	1	22	14	0	0	0	
8	2.7 to 3.7	1	18	23	0	0	0	Div., no track
9	3	1	19	21	1	0	0	
10	1 to 1.5	2	20	22	1	0	0	
11	0.8 to 2.6	2	15	10	0	0	0	
12	1 to 1.2	2	17	13	1	0	0	
13	1.1	2	17	14	0	17	0	Divers
14	3	1	16	18	2	0	0	
15	3 to 3.5	1	16	8	1	0	0	
16	2.5 to 3.8	1	17	45	4	53	4	Divers
17	1 to 1.6	2	16	17	0	0	0	
18	1	2	16	5	0	0	0	
19	3	1	21	43	2	48	2	Divers
20	1 to 1.8	2	17	15	1	0	0	
21	2.7 to 3.5	1	17	14	0	0	0	
22	3 to 4	10	17	89	7	0	0	
23	2 to 4	10	17 to 25	88	7	0	0	

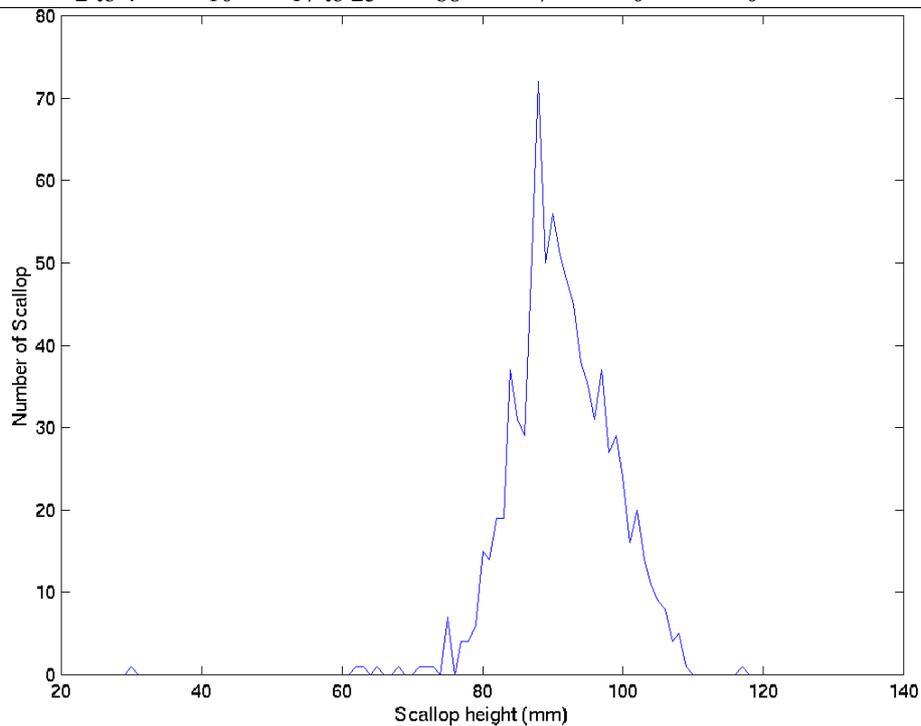


Figure 34: Classical dredge. Histogram of whole scallops caught by dredge or divers

Damage ratio

There were 38 broken scallops in the dredge or on the track. This number must be compared to the whole number of scallops caught in the dredge or by divers (876). The ratio of damage in the classical dredge reaches 4%. This value is four times the value obtained by the prototype dredge.

Efficiency and density

The efficiency of the dredge can be calculated by the hauls treated by divers, from the number of scallops in the dredge and the number on the track (Table 15). This efficiency varies largely from 0.33 to 0.64 with the mean at 0.47

The density of scallops on the sea bed can be also estimated on the hauls treated by divers (Table 15). This density varies largely from 0.25 to 1.00 scallop/m² with the mean at 0.52 scallop/m².

Table 15 Density of scallops on the bottom and efficiency of the classical dredge calculated from the hauls treated by divers.

haul	dredge scallops	track scallops	surface m ²	whole scallops	Density sc/m ²	efficiency
1	29	59	242	88	0.36	0.33
3	37	21	150	58	0.39	0.64
13	14	17	122	31	0.25	0.45
16	49	57	106	106	1.00	0.46
19	45	50	167	95	0.57	0.47
mean					0.52	0.47

Measures of stress

The results on the stress measurement are presented in Figure 35.

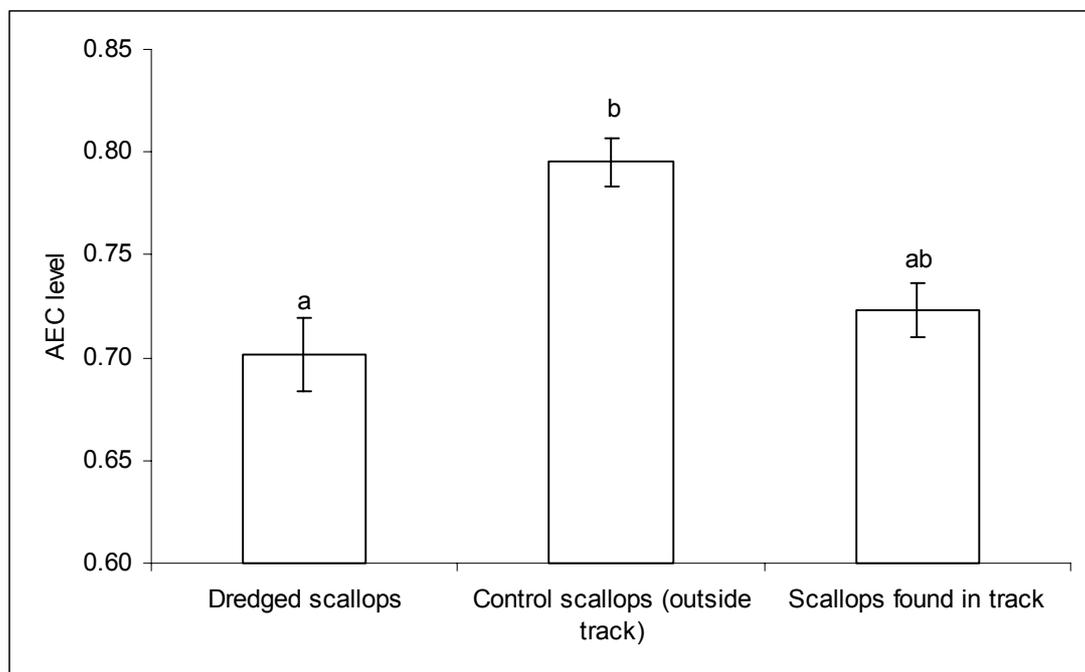


Figure 35 The effects of water jet dredging on the AEC level (mean \pm s.e.) of scallops from the Rade de Brest in February 2001. Any two means sharing a common letter within each column were not significantly different at $p < 0.05$ (Tukey test).

Ivanovici (1980) reported that high AEC levels (>0.8) were found in organisms living in optimal conditions where the animals were growing and reproducing. Levels between 0.5 and 0.7 were found in organisms whose environment was limiting in some way such animals had reduced growth rates and did not reproduce, but recovered when returned to optimal conditions. Organisms whose AEC levels were less than 0.55 had a negative scope for growth and did not recover. The results indicate that the mean AEC level for all treatments in this study was >0.7 therefore the animals had an increased chance of survival when returned to the sea post-dredging.

The AEC level of dredged scallops decreased significantly from the control scallops ($F_{127,2} = 3.31, p < 0.05$). The AEC reduction was calculated as 12.5%. Maguire et al. ([Section 3.1.3.3](#) and in press b) studied the effect of spring teeth dredging on the AEC level of scallops on a seasonal basis in the Isle of Man and found the greatest reduction took place in February (44.8%). The AEC level of dredged scallops from the Isle of Man in February was 0.43. In this study at the same time of the year the level in dredged scallops was 0.70. However, the tow length was much longer in the Isle of Man study (45 minutes) compared to just 5 minutes for the study in Brest. There was also a difference in the sediment type of the two sites, the Isle of Man ground included hard rocky substrate compared to the sandy substrate of the Brest

site. Short periods of exposure to simulated dredging (up to 8-16minutes) produced similar reductions in AEC to those observed above ([Section 3.1.3.2](#)).

Scallops that came in contact with the dredge but remained uncaught and were collected on the dredge track had a lower AEC level than the control scallops and a higher AEC level than the dredged scallops but this result was not significant. Although the AEC level in these scallops was not as low as in the captured scallops it is important to demonstrate that the impact of dredging is not solely confined to those scallops landed on deck. In the study by Maguire et al. (in press) and [Section 3.1.3.2](#) divers also collected scallops from the dredge track of a spring toothed dredge and similar AEC results were found. The AEC level in the Isle of Man scallops from the dredge track was 0.75 compared to 0.72 in the scallops from the dredge track in this study. Owing to the relatively low efficiency of scallop dredges (Dare et al. 1993) far more scallops encounter dredges and remain on the seabed than are captured. These non-captured animals may suffer enhanced mortality due to dredge induced stress in the same way as undersized discards.

Discussion

It seems clear that the prototype dredge creates less damage on scallops than classical dredge. This reduction reaches around a 4 fold reduction in the number of broken scallops (cf. damage ratio). This reduction on the number of broken scallops could be extrapolated to the environmental damage to benthos and other organisms.

This conclusion must be tempered by the small efficiency of the prototype relatively to the classical dredge (around 3 times less) though the efficiency of the prototype varies widely which means that it can be improved if the parameters which influenced this efficiency could be assessed.

The reasons of this reduction of damage are not clear, or more exactly the parameter which lead to this reduction are not well defined. Possible factors are described below:

1. The absence of teeth on the prototype should reduce the impact on the sea bed, even on though very small organisms this reduction is not clear.
2. The towing speed could also be also a reason of this reduction, actually the towing speed on the commercial vessel for the trials on classical dredge could not be so low than on the research vessel during the trials on the prototype dredge. This reduction of speed must reduce the damage on environment.

The density of scallops on the bottom (Table 13 and Table 15) gives strange results: the density calculated during the prototype trials is 3 times less than during the classical dredge trials, although the trials have been done in more or less the same area. The periods of trials (end of February and beginning of April) cannot explain this large difference. Probably the areas are not exactly the same or that the speed log was not precise enough.

The good result on the damage ratio of the prototype dredge is unfortunately compensated by the complexity of the dredge and of its use. This dredge requires a pump engine onboard with a suction pipe and an output pipe quite difficult to use. The

tuning of the length of the output pipe leads to tune the warp length just superior which forces to dredge at more or less the same depth. This complexity can be reduced by industrial development.

3.3.3.2 THE INFLUENCE OF PORTUGUESE DREDGE DESIGN ON THE CATCH OF *CALLISTA CHIONE* (LINNAEUS, 1758)

Introduction

An important bivalve fishery takes place along the Southwest coast of Portugal. At present, in this part of the Portuguese coast, the most important commercial species is the clam *Callista chione* (Linnaeus, 1758). This species has been recorded from Great Britain, to south to the Iberian Peninsula, into the Mediterranean, along the Atlantic coast of Morocco and to the Canary Islands and the Azores (Tebble 1966). On the Southwest coast of Portugal, this species is preferentially distributed on clean sandy bottoms from 10 to 20m depths (Gaspar et al. 1998).

C. chione constitutes the target of a specific fishing activity carried out by an artisanal fleet. Dredges are the gears operated by the fishery. The shape and structure of the Portuguese dredge has remained unchanged throughout time and consists of a small, heavy semicircular iron structure, with a net bag and a toothed lower bar at the mouth. Welded to this iron structure are three metal shafts forming a kind of hen's foot where the towing cable is attached (Figure 36 A and B).

Recently, some fishermen from the Setúbal region have been trying to introduce a new dredge typology in the *C. chione* fishery arguing that it is a more efficient fishing gear. The basic difference between the two dredges resides in the retention structure for the bivalves. In fact, in the new dredge the net bag is replaced by a rectangular metallic grid (Figure 36 A and C).

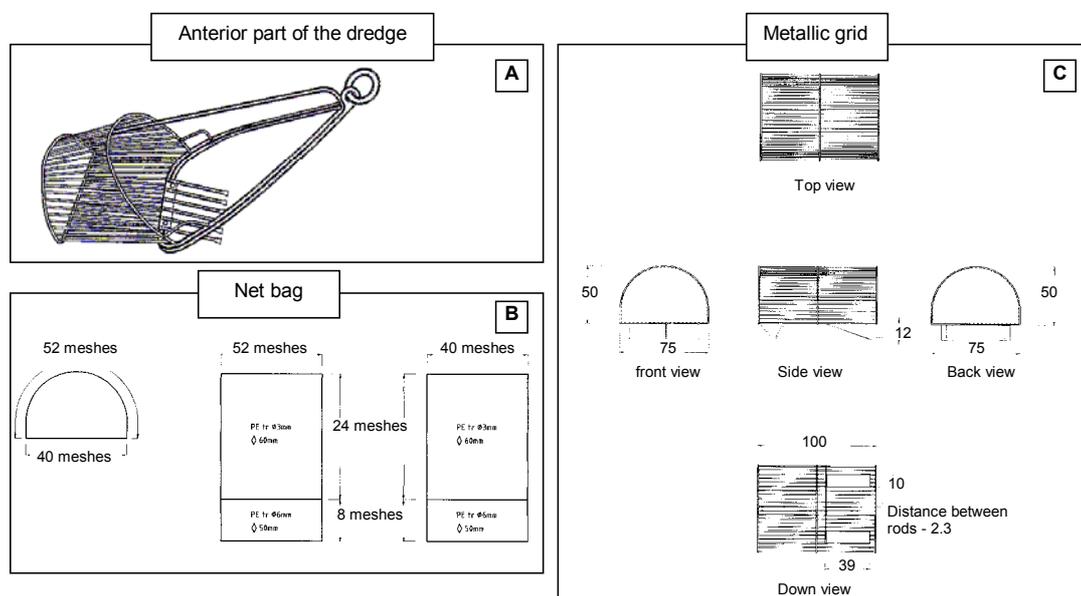


Figure 36. Diagram of the *Callista chione* dredge. Technical specifications of the net bag and the metallic grid (measures in cm). 1B is Traditional Dredge (TD); 1C is New Dredge Design (NDD).

The introduction of a new dredge design in any fishery should take into consideration two aspects: first it should reduce the impact on the macrobenthic community and second it should bring something to the fishermen such as higher incomes. Thus, in the comparative study of the two dredge typologies, besides the efficiency comparison of the gears, the ecological impact should also be assessed. The fact that the dredge is a fishing gear that is towed over the bottom causes necessarily an impact on the sediment as well as on the benthic community. However, although there may be a similar impact on the sediment in both gears, the same should not happen in the benthic community even though using the same capture methods.

In this context a comparative study was conducted with the aim of evaluating a possible introduction of the new dredge design in the fishery of *C. chione*.

Methodologies

Sampling design

The comparative study was conducted during March 1999 in the Setúbal region on the western coast of Portugal, in a site off Comporta, the most important fishing ground for *C. chione*. The study was carried out onboard the “Surpresa” (SB-867-C), a vessel belonging to the commercial dredging fleet. The samples were collected on sandy bottoms between 8 and 10m depth. The specifications of the anterior part of the gears used in the experiment are summarised in Table 16.

Table 16 Gear specifications of the anterior part of the dredges used in the present study.

	New dredge design	Traditional dredge
Length of the mouth	43 cm	60 cm
Space between rods	2.4 cm	2.4 cm
Number of teeth	6	6
Space between teeth	4.3 cm	4.0 cm
Tooth length	30 cm	34 cm
Tooth angle	20°	20°

In order to estimate the effect of tow duration on the catch of *C. chione* and on the number of damaged individuals, three-tow durations were investigated (5, 10 and 20 minutes). A total of twenty-four tows were accomplished, twelve for each type of dredge and with consistent durations for both dredges. The duration of dredge hauls was measured from the time the winch stopped paying out the towing cable to the time when the winch was restarted. The dredges were towed at a commercial speed of 2-3 Knots.

In order to determine the proportion of clams under the size of 50mm retained by both dredges and also to determine the respective impact, by means of the number of damaged individuals, the cover method was employed. This technique consists in the covering of the net bag, in the case of the TD, or the grid, in the case of the NDD, with a second net bag with a small mesh size. This bag has a 20mm mesh and was stretched loosely over the entire back of the dredges, enough slack being left in the

cover to reduce any masking effect on selection and to allow the natural flow of water through the net (Gaspar et al. 1999).

The captures of each dredge, coming from the bag or the grid and from the cover, were processed separately in all tows. All catches were identified, weighted and measured (in the case of the bivalves). Standard shell length measurements were made along the longest linear dimension to the millimetre below. The species identification was made according to Tebble (1966), FAO (1987) and Poppe (1993). The nomenclature adopted was that of FAO (1987).

To determine direct mortality each organism caught was attributed a score, which evaluated the degree of damage suffered (Table 17). The scale in the present work used four scores, where score 1 corresponds to the individuals in perfect condition while score 4 corresponds to the dead individuals. Ten *taxa* were considered: Bivalvia, Gastropoda, Asteroidea Ophiuroidea, Echinoidea, Anomura, Brachyura, Natantia, Osteichthyes and Polychaeta. For all *taxa* considered, except for Bivalvia, the attribution of the respective score for each individual caught was done visually during the sampling operation on board. For Bivalvia the attribution was done in the laboratory, during the processing of the collected samples. Underwater images were collected by divers during the tows. In the present work the dredge usually used to harvest bivalves was called the traditional dredge (TD), while the new dredge was denominated by new dredge design (NDD)

Table 17 Criteria used in the attribution of a score to each *taxon*.

	1	Score	1	2	3	4
Crustacea	2	Bivalvia	In good condition	Edge of shell chipped	Hinge broken	Crushed / dead
		Gastropoda	In good condition	Edge of shell chipped	Shell cracked or punctured	Crushed / dead
		Asteroidea	In good condition	Arms missing	Worn and arms missing	dead
		Ophiuroidea	In good condition	Arms missing	Worn and arms missing / minor disc damage	Major disc damaged / dead
		Echinoidea	In good condition	<50% spine loss	>50% spine loss / minor cracks	Crushed / dead
	3	Anomura	In good condition	Out of shell and intact	Out of shell and damaged	Crushed / dead
		Brachyura	In good condition	Legs missing / small carapace cracks	Major carapace cracks	Crushed / dead
		Natantia	In good condition			Dead
		Osteichthyes	In good condition	Small amount of scales missing/ small cuts or wound	Large amount of scales missing/ severe wounds	Dead
		Polychaeta	In good condition			Sectioned

Data analysis

Statistical procedures followed Zar (1996) and Sokal and Rohlf (1981). Size frequency distributions were constructed from raw data and compared using a Kolmogorov-Smirnov test. To investigate the differences between the fishing yields obtained for both dredges and for the three-tow duration, a two sample *t*-test was

used. Analyses of variance ANOVA or Kruskal-Wallis ANOVA for comparison of overall proportions was used to test the effect of tow duration within each dredge on the proportion of by-catch and on the proportion of damaged individuals. Multiple comparisons were performed using the Student-Newman-Keuls test. If no significant statistical differences were observed on the effect of tow duration on the proportion of both by-catch and damaged individuals, the data was pooled and *t*-test or Mann-Whitney test were used to investigate the effect of dredge design on those proportions. Prior to the application of ANOVA, Kruskal-Wallis ANOVA, *t*-test or Mann-Whitney test, data was standardised and, when expressed as a percentage, was converted to arcsine square root values. All statistical analyses but Kolmogorov-Smirnov test were performed using the SigmaStat statistical software.

Results

Overall catches

Table 18 shows the species that entered the dredges for each *taxon* considered. A total of 36 species entered the dredges. Bivalvia was the most common *taxon* with 22 species, followed by Gastropoda and Brachyura with 3 species each.

Table 18 Total number of individuals caught during the experimental catches.

	NDD		TD	
	Number	Weight (g)	Number	Weight (g)
Bivalvia				
<i>Acanthocardia aculeata</i>	53	7421	12	1720
<i>Acanthocardia spinosa</i>	4	629	1	321
<i>Acanthocardia tuberculata</i>	198	20356	114	9663
<i>Callista chione</i>	2288	166615	838	69311
<i>Dosinia exoleta</i>	79	2228	32	1067
<i>Dosinia lupinus</i>	61	293	5	47
<i>Ensis ensis</i>	23	130	2	11
<i>Ensis siliqua</i>	5	260		
<i>Gari fervensis</i>	2	7		
<i>Laevicardium crassum</i>	575	19676	621	21533
<i>Lutraria angustior</i>	27	161	3	6
<i>Lutraria lutraria</i>	2	141	2	176
<i>Maetra coralina</i>	10	226	17	223
<i>Maetra glauca</i>	86	4622	127	7439
<i>Pandora inaequalis</i>	1	2	1	2
<i>Pharus legumen</i>	3	14	2	18
<i>Sollen marginatus</i>	1	25		
<i>Spisula solida</i>	2	7	1	3
<i>Tellina incarnata</i>	76	210	4	8
<i>Thracia papyracea</i>	4	1		
<i>Venus fasciata</i>	3	8	2	6
<i>Venus striatula</i>	2014	7632	552	1783
Gastropoda				
<i>Cymbium olla</i>	3	167		
<i>Gibbula sp.</i>	10	33	4	4
<i>Nassarius reticulatus</i>	3	12	1	5
Asteroidea				
<i>Astropecten aranciacus</i>	31	2622	19	1921
Ophiuroidea				
<i>Ophiocomina nigra</i>	7	19	11	10
Echinoidea				
<i>Echinocardium cordatum</i>	51	1340	27	354
<i>Sphaerechinus granularis</i>	1	5		
Anomura				
<i>Pagurus sp.</i>	14	41	6	25
Brachyura				
<i>Calappa granulata</i>	21	605	21	704
<i>Corystes cassivelaunus</i>	1	8		
<i>Polybius henslowi</i>	66	768	71	936
Natantia				
<i>Palaemon serratus</i>			1	52
Osteichthyes				
<i>Dicologlossa cuneata</i>			2	39
Polychaeta	97	1557		274
Total	5822	237841	2499	117661

The analysis of Table 18 show that a higher number of species entered the NDD than in the TD (34 and 29 species respectively). In terms of weight, *C. chione* largely dominated the total catches for the two dredges (70.05% and 58.91% respectively for NDD and TD). In terms of numbers, the most abundant species was *C. chione*, *Venus striatula* (da Costa, 1778) and *Laevicardium crassum* (Linnaeus, 1758) (39.3%, 34.6% and 10% for the NDD and 33.3%, 21.9% and 24.7% for the TD, respectively).

Callista chione catches

Histograms of size frequency distributions for tow duration of 5, 10 and 20 min. and for the two dredges are shown in Figure 37. Results from the Kolmogorov-Smirnov test indicate that all size distributions were significantly different ($P < 0.05$).

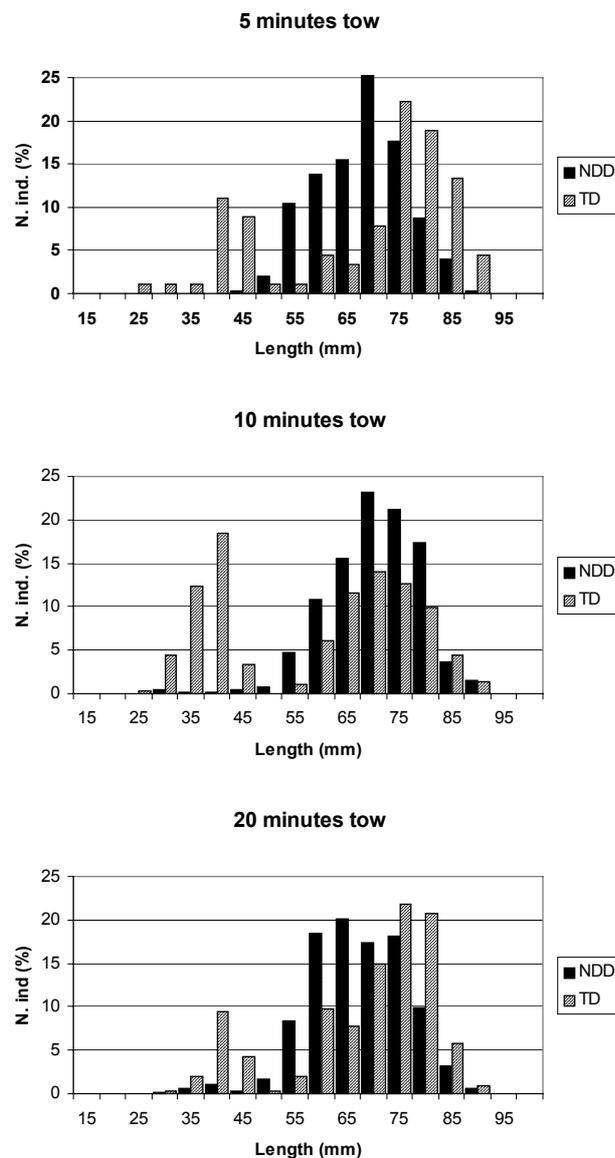


Figure 37. *Callista chione* length frequency distributions for the NDD and TD and for the 3 tow duration investigated.

From Figure 37 it can be seen that the catches from TD, for the 3-tow duration investigated, are made up of a significant fraction of sub-dimensioned individuals, or else, with a size inferior to 50mm. This fraction corresponds to 23.35%, 37.8% and 16.1%, respectively for 5, 10 and 20 minutes tow duration. In the NDD the captures are almost totally made up of individuals with a size superior to the legal minimum (99.6%, 98.8% and 97.8% for tow duration of 5, 10 and 20 min. respectively).

The retention analysis of *C. chione* by both dredges was accomplished by isolating the fraction of the retained individuals below the minimum legal size (MLS - 50 mm) and the fraction of those with size equal of superior to the MLS. It was hoped, in this manner, to enhance the differences between the efficiency of both fishing gears in view of both groups.

Figure 38 represents the average percentage of individuals retained with a size inferior to the MLS, for each of the tow durations. The NDD only retains a small percentage of undersized individuals (< 7%). In the TD the opposite was observed. In fact the TD retained a higher percentage of undersized individuals (97.6%, 87.6% and 99.1%, respectively for 5, 10 and 20 minutes of tow duration).

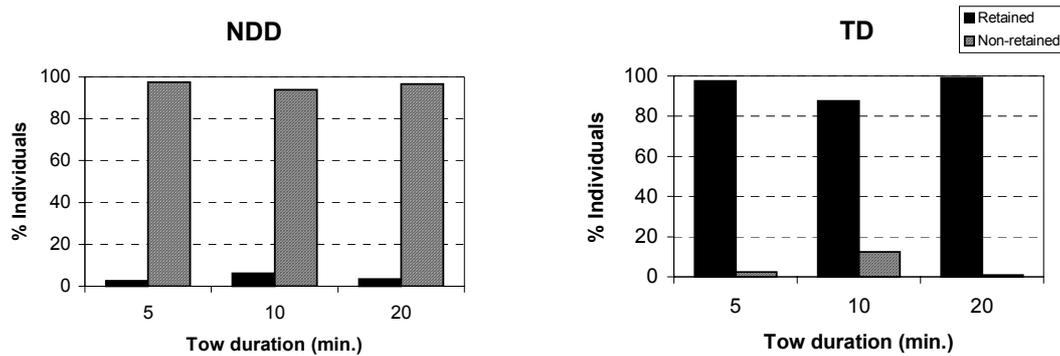


Figure 38 Proportion of individuals retained with length < 50mm (MLS)

The average retention percentage of both dredges with regard to the accessible legal population (Figure 39) was always high, exceeding 93% retention in the case of the NDD and reaching 100% retention in the case of TD.

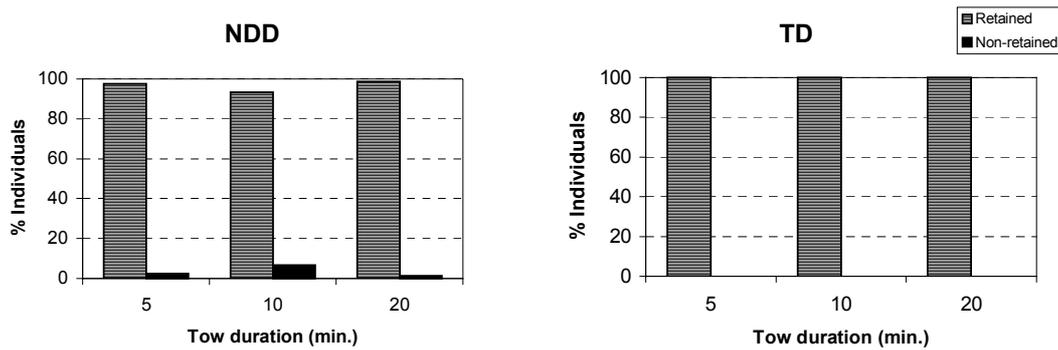


Figure 39 Proportion of individuals retained with length $\geq 50\text{mm}$ (MLS)

The mean fishing yield obtained for both dredges and for 5, 10 and 20 minutes tow duration is shown in Figure 40. From its analysis it can be seen that for the same tow duration the mean fishing yield obtained for the NDD is always substantially superior to that obtained by the TD. Results of the *t*-test showed significant differences between the mean fishing yield obtained for the 3 tow duration (*t*-test. 5min tow: $t=3.241$, d.f.=6, $P=0.018$; 10min tow: $t=3.125$, d.f.= 6, $P=0.017$; 20min tow: $t=2.871$; d.f.=6, $P=0.035$)

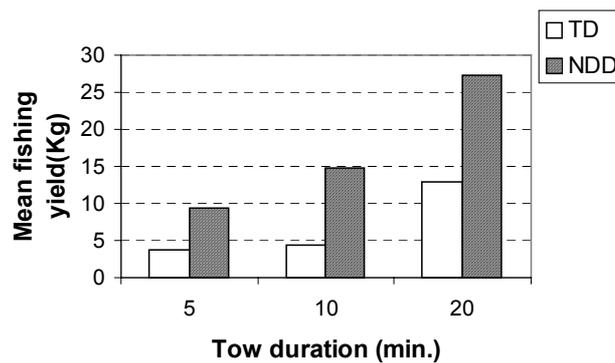


Figure 40 Comparisons of the mean fishing yield (Kg / tow duration) from NDD and TD.

By-catch

In this fishery, only the target species is retained on board because legislation allows landing of one species per boat each day. Therefore all the other species caught are discarded. The comparison of the average percentage of by-catch, in numbers and

weight, obtained for each dredge and for the 3-tow duration investigated is shown in

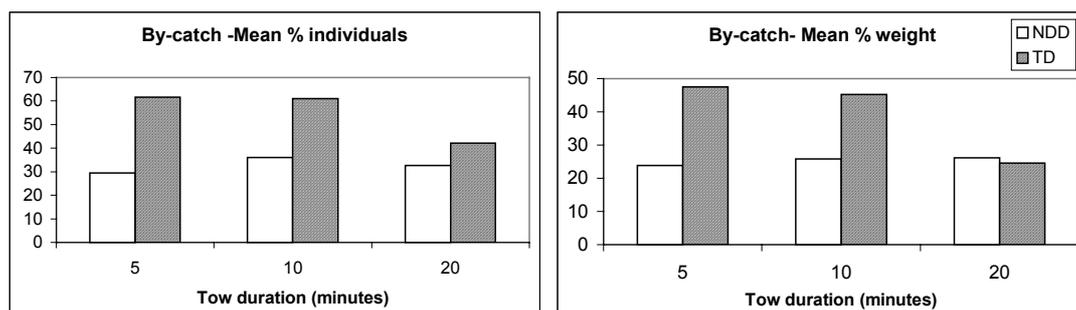


Figure 41.

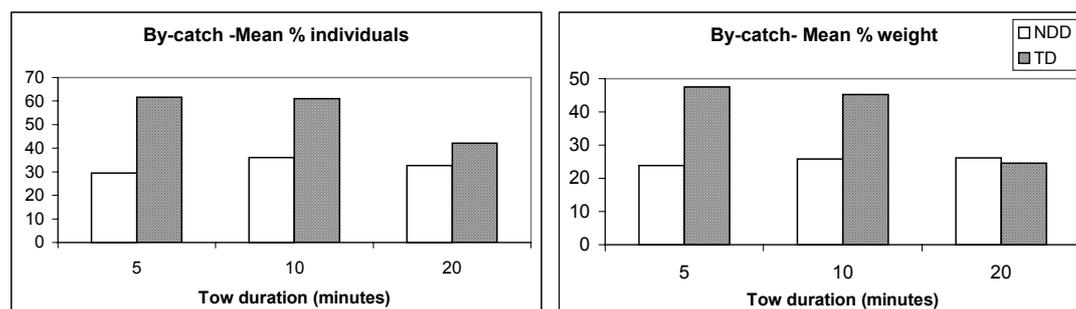


Figure 41 Comparisons of the mean proportion of by-catch in number and weight from NDD and TD, for the 3 tow duration investigated.

For 5 and 10 minutes tow duration the mean by-catch from the NDD is substantially lower than those observed for the TD, both in terms of number and weight. Surprisingly, for 20 minutes tow duration the mean by-catch obtained for the TD both in number and weight was similar to that obtained for the NDD. It is important to emphasise that within the beds of *C. chione* there exist extended patches of *V. striatula* that are characterised by a low bio-diversity. We believe that this feature explains the reduction of the by-catch observed in the TD in the 20 minutes tow. In fact, for a single tow, the dredge was towed over one of these patches and caught a high number of juveniles of *V. striatula* clams that passed through the primary bag. As a consequence the percentage of by-catch obtained for this tow was very low. If we exclude this single tow, it can be observed that the mean percentage of the by-catch remains almost constant regardless of tow duration. Thus, when using the NDD a percentage of by-catch of 31% and 25% in terms of number and weight can be expected and 61% and 48%, respectively in number and weight, when the TD is used.

The statistical analysis carried out showed that, within each dredge, the tow duration has no effect on the percentage of the by-catch (K-W, number $H=0.269$, d.f.=2, $P_{\text{est}}=0.874$, $P_{\text{exact}}=0.913$; ANOVA, weight $P>0.05$). Results of the *t*-test showed significant differences between the mean percentage of by-catch obtained for the 2 dredges, both in number and in weight (*t*-test, number $t=-3.411$, $df=22$, $P=0.003$; weight $t=-2.653$, $df=22$, $P=0.015$).

Direct mortality

Figure 42, graphically shows the mean percentage of damaged and undamaged *C. chione* clams obtained during the fishing experiments for 5, 10 and 20 minutes tow duration and for the dredges assayed. For both dredges the percentage of clams damaged was very low, never exceeding 9% in the case of the NDD and 6% in the case of TD. For this species and for both dredges the percentage of damaged clams did not vary with tow duration (ANOVA, $P > 0.05$). Similarly, there was no significant difference between of dredge designs in the percentage of *Callista chione* specimens damaged (t -test, $t = 0.971$, $d.f. = 22$, $P = 0.342$).

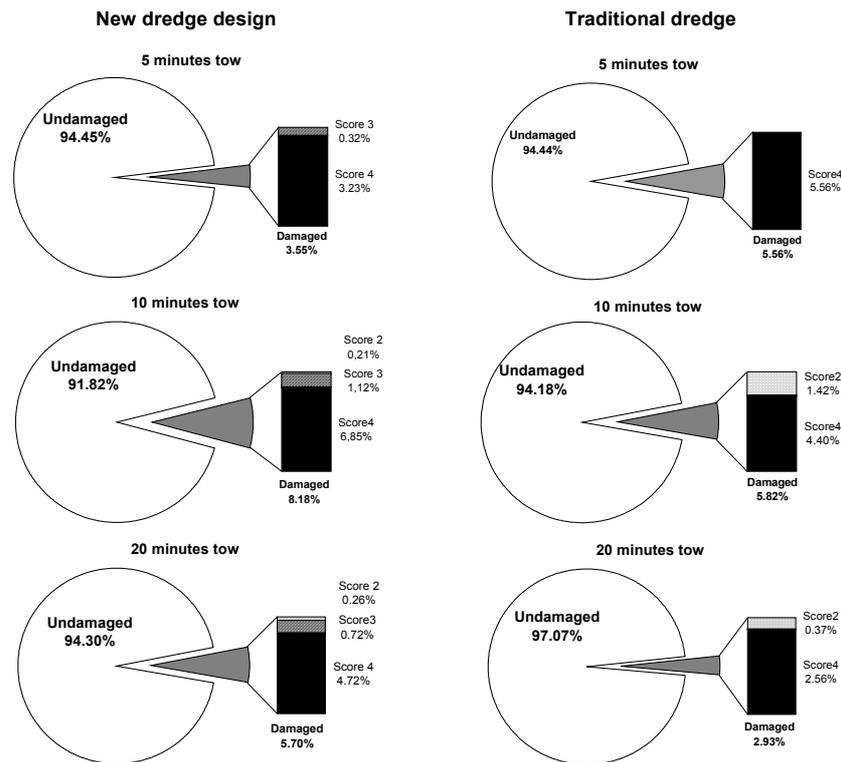


Figure 42 Relative proportion of damaged individuals of *Callista chione* for the two dredges and for the 3 tow duration investigated.

Regarding the impact of this type of fishing upon the macrobenthic community (Table 19), in terms of percentage of damaged individuals (scores 2-4), it was observed that the most affected *taxa* in the area where the study was conducted was, in decreasing order of importance, the echinoidea, asteroidea and brachyura.

Table 19. Comparison of the mean number of individuals damaged (scores 2 to 4) and dead (scores 3 and 4) for each *taxon*.

		New dredge design			Traditional dredge		
		Total	Damaged	Dead	Total	Damaged	Dead
			N (%)	N (%)		N (%)	N (%)
5 minutes	Bivalvia	317.0	15.8 (5.0)	15.3 (4.8)	162.0	11.3 (6.9)	10.5 (6.5)
	Gastropoda	1.3	0.0 (0.0)	0.0 (0.0)	0.3	0.0 (0.0)	0.0 (0.0)
	Asteroidea	1.3	0.3 (20.0)	0.0 (0.0)	0.8	0.5 (66.7)	0.5 (66.7)
	Ophiuroidea						
	Echinoidea	5.0	4.3 (85.0)	4.0 (80.0)	1.3	1.3 (100.0)	1.3 (100.0)
	Brachyura	5.3	1.0 (19.0)	1.0 (19.0)	5.0	1.8 (35.0)	1.3 (25.0)
	Anomura	0.3	0.0 (0.0)	0.0 (0.0)	0.3	0.0 (0.0)	0.0 (0.0)
	Natantia						
	Osteichthyes				0.5	0.0 (0.0)	0.0 (0.0)
	Polychaeta	8.8	0.0 (0.0)	0.0 (0.0)	0.8	0.0 (0.0)	0.0 (0.0)
	Total	338.8	21.3 (6.3)	20.3 (6.0)	170.8	14.8 (8.6)	13.5 (7.9)
10 minutes	Bivalvia	399.8	33.0 (8.3)	30.0 (7.5)	242.6	12.4 (5.1)	11.2 (4.6)
	Gastropoda	1.5	0.0 (0.0)	0.0 (0.0)	0.2	0.0 (0.0)	0.0 (0.0)
	Asteroidea	3.3	1.3 (38.5)	1.0 (30.8)	1.6	0.8 (50.0)	0.4 (25.0)
	Ophiuroidea	0.3	0.0 (0.0)	0.0 (0.0)	2.2	2.2 (100.0)	2.2 (100.0)
	Echinoidea	4.0	2.5 (62.5)	2.5 (62.5)	4.0	3.0 (75.0)	3.0 (75.0)
	Brachyura	7.5	1.5 (20.0)	1.5 (20.0)	6.6	0.2 (3.0)	0.0 (0.0)
	Anomura	1.5	0.3 (16.7)	0.0 (0.0)	1.0	0.2 (20.0)	0.0 (0.0)
	Natantia				0.2	0.0 (0.0)	0.0 (0.0)
	Osteichthyes						
	Polychaeta	5.8	0.0 (0.0)	0.0 (0.0)	0.4	0.0 (0.0)	0.0 (0.0)
	Total	423.5	38.5 (9.1)	35.0 (8.3)	258.8	18.8 (7.3)	16.8 (6.5)
20 minutes	Bivalvia	662.5	40.3 (6.1)	36.8 (5.5)	158.0	19.3 (12.2)	16.3 (10.3)
	Gastropoda	1.3	0.0 (0.0)	0.0 (0.0)	1.0	0.0 (0.0)	0.0 (0.0)
	Asteroidea	3.3	2.3 (69.2)	0.3 (7.7)	2.7	2.3 (87.5)	0.0 (0.0)
	Ophiuroidea	1.5	0.0 (0.0)	0.0 (0.0)			
	Echinoidea	4.0	3.0 (75.0)	3.0 (75.0)	0.7	0.7 (100.0)	0.7 (100.0)
	Brachyura	9.3	2.0 (21.6)	1.8 (18.9)	13.0	3.0 (23.1)	2.7 (20.5)
	Anomura	1.8	0.3 (14.3)	0.0 (0.0)			
	Natantia						
	Osteichthyes						
	Polychaeta	9.8	0.0 (0.0)	0.0 (0.0)	4.3	0.0 (0.0)	0.0 (0.0)
	Total	693.3	47.8 (6.9)	41.8 (6.0)	179.7	25.3 (14.1)	19.7 (10.9)

In a comparative analysis of the influence of both dredges in the percentage of individuals damaged, not many differences were found. It is worth to underline that for all *taxa*, with the exception for Bivalvia, the mean number of individuals caught was consistently low. Nevertheless the TD damaged a greater percentage of sea urchins and sea stars than the NDD (Table 19). The results showed that for the NDD the average percentage of damaged individuals was 6.3%, 9.1% and 6.9% for tow duration of 5, 10 and 20 minutes respectively, whilst for the TD the mean percentage of damaged individual observed was 8.6%, 7.3% and 14%, respectively for 5, 10 and 20 minutes tow duration.

Regarding the influence of tow duration on the percentage of overall damaged individuals, the results of the statistical tests do not reveal significant differences in both dredges (ANOVA, $P > 0.05$). The dredge design also does not have any effect on the percentage of damaged individuals (t -test, $t = 1.929$, $d.f. = 22$, $P = 0.067$)

In the present work a survival study was not carried out. Thus it was assumed that all individuals severely damaged (score 3) died and that all individuals lightly damaged (score 2) survived. Therefore, the impact produced by both dredges on the macrobenthic community was assessed considering only the individuals scored as 3 or 4. The analysis of Table 19, showed that for the NDD the average percentage of dead individuals was 6.0%, 8.3% and 6.0% for tow duration of 5, 10 and 20 minutes respectively. For the TD the mean percentage of damaged individual observed were 7.9%, 6.5% and 10.9%, respectively for 5, 10 and 20 minutes tow duration. The comparison of mortality produced by both dredges revealed that the dredge design does not influence the number of dead individuals (Mann-Whitney, $T=169$, $n_{\text{small}}=12$, $n_{\text{big}}=12$, $P=0.285$).

For each dredge, data from all tows was pooled in order to evaluate the relative vulnerability of the species that comprises the by-catch to dredging (Table 20). As it was expected, the most sensitive species to this kind of fishery are the thin shelled bivalves such as *Pharus legumen* (Linnaeus, 1758), *Ensis ensis* (Linnaeus, 1758), *Solen marginatus* (Pennant, 1777), *Ensis siliqua* (Linnaeus, 1758), *Mactra glauca* Born, 1778 and *Lutraria anguistor* Phillipi, 1844, and the sea urchin *Echinocardium cordatum* (Pennant, 1777). Little or no damage occurs to the solid shelled bivalve such as *C. chione*, *L. crassum*, *Spisula solida* (Linnaeus, 1758), *V. striatula*, *Dosinia exoleta* (Linnaeus, 1758), *Dosinia lupinus* (Linnaeus, 1758), and *Acanthocardia tuberculata* (Linnaeus, 1758). The mean proportion expected to die is also very low for the gastropods *Cymbium olla* (Linnaeus, 1758), *Gibbula sp.* and *Nassarius reticulatus* (Linnaeus, 1758), for the large sea star *Astropecten aranciacus* (Linnaeus, 1758), for hermit crabs *Pagurus sp.* and for the crabs *Polybius henslowi* Leach, 1820 and *Calappa granulata* (Linnaeus, 1758).

Discussion

The catch composition of a species is conditioned by the structure of its population in the location where fishing is exercised and by the selectivity of the fishing gear. By this means, the comparative analysis of the clam composition coming from both dredges must be done, on its first approach, by its length frequency distributions retained in the net bag. In regard to the retention capacity of gears, one must particularly consider the percentages of individuals retained with size inferior to the minimum legal and, on the other hand, the percentages of individuals retained with size superior to the minimum legal.

Table 20 Relative vulnerability of the species to dredging.

	New dredge design				Mean proportion expected to die	Traditional dredge				Mean proportion expected to die	
	Score1	Score2	Score3	Score4		Score1	Score2	Score3	Score4		
Bivalvia											
<i>Pharus legumen</i>	0	0	0	3	100.0	<i>Pharus legumen</i>	0	0	0	2	100.0
<i>Solen marginatus</i>	0	0	0	1	100.0	<i>Lutraria anguistor</i>	0	0	0	3	100.0
<i>Thracia papyracea</i>	0	0	0	4	100.0	<i>Mactra glauca</i>	20	6	3	98	79.5
<i>Lutraria anguistor</i>	1	0	2	24	96.3	<i>Acanthocardia aculeata</i>	6	0	0	6	50.0
<i>Mactra glauca</i>	7	6	2	71	84.9	<i>Ensis ensis</i>	1	0	0	1	50.0
<i>Ensis siliqua</i>	1	0	0	4	80.0	<i>Lutraria lutraria</i>	1	0	0	1	50.0
<i>Acanthocardia aculeata</i>	26	0	4	23	50.9	<i>Mactra coralina stultorum</i>	13	1	0	3	18.8
<i>Mactra coralina</i>	5	0	0	5	50.0	<i>Acanthocardia tuberculata</i>	102	0	0	12	10.5
<i>Acanthocardia spinosa</i>	2	0	0	2	50.0	<i>Dosinia exoleta</i>	26	4	0	2	6.3
<i>Ensis ensis</i>	12	0	1	10	47.8	<i>Callista chione</i>	804	2	0	31	3.7
<i>Tellina incarnata</i>	70	0	0	6	7.9	<i>Laevicardium crassum</i>	606	5	4	6	1.6
<i>Acanthocardia tuberculata</i>	183	0	5	10	7.6	<i>Tellina incarnata</i>	4	0	0	0	0.0
<i>Callista chione</i>	2156	4	18	110	5.6	<i>Pandora inaequivalvis</i>	1	0	0	0	0.0
<i>Dosinia lupinus</i>	51	7	0	3	4.9	<i>Dosinia lupinus</i>	5	0	0	0	0.0
<i>Laevicardium crassum</i>	557	3	1	14	2.6	<i>Venus fasciata</i>	2	0	0	0	0.0
<i>Dosinia exoleta</i>	69	8	0	2	2.5	<i>Spisula solida</i>	1	0	0	0	0.0
<i>Venus striatula</i>	2011	0	0	3	0.1	<i>Acanthocardia spinosa</i>	1	0	0	0	0.0
<i>Pandora inaequivalvis</i>	1	0	0	0	0.0	<i>Venus striatula</i>	552	0	0	0	0.0
<i>Venus fasciata</i>	3	0	0	0	0.0	Gastropoda					0.0
<i>Spisula solida</i>	2	0	0	0	0.0	<i>Gibbula sp.</i>	4	0	0	0	0.0
<i>Lutraria lutraria</i>	2	0	0	0	0.0	<i>Nassarius reticulatus</i>	1	0	0	0	0.0
<i>Gari fervensis</i>	2	0	0	0	0.0	Asteroidea					0.0
Gastropoda						<i>Astropecten aranciacus</i>	6	9	2	2	21.1
<i>Cymbium olla</i>	3	0	0	0	0.0	Ophiuroidea					0.0
<i>Gibbula sp.</i>	10	0	0	0	0.0	<i>Ophiocomina nigra</i>	0	0	0	11	100.0
<i>Nassarius reticulatus</i>	3	0	0	0	0.0	Echinoidea					0.0
Asteroidea						<i>Echinocardium cordatum</i>	5	0	0	22	81.5
<i>Astropecten aranciacus</i>	16	13	0	2	6.5	Anomura					0.0
Ophiuroidea						<i>Pagurus sp.</i>	5	1	0	0	0.0
<i>Ophiocomina nigra</i>	7	0	0	0	0.0	Brachyura					0.0
Echinoidea						<i>Polybius henslowi</i>	57	3	2	9	15.5
<i>Echinocardium cordatum</i>	12	1	0	38	74.5	<i>Calappa granulata</i>	17	2	0	2	9.5
<i>Sphaerechinus granularis</i>	1	0	0	0	0.0	Natantia					0.0
Anomura						<i>Palaemon serratus</i>	1	0	0	0	0.0
<i>Pagurus sp.</i>	12	2	0	0	0.0	Osteichthyes					0.0
Brachyura						<i>Dicologlossa cuneata</i>	2	0	0	0	0.0
<i>Corystes cassivelaunus</i>	0	0	0	1	100.0	Polychaeta	18	0	0	0	0.0
<i>Polybius henslowi</i>	49	0	0	17	25.8						0.0
<i>Calappa granulata</i>	21	0	0	0	0.0						
Polychaeta	97	0	0	0	0.0						

The comparison of the length frequency distribution of *C. chione*, from the TD and the NDD, showed that the captures from the former dredge were composed by a great fraction of juveniles, while the captures from the NDD were composed, almost in its whole, by individuals with size superior to the minimum legal. Considering the clams captured exclusively with size inferior to the minimum legal (50mm), it was observed that the NDD retained a small percentage of these individuals, whilst the TD retained a higher percentage. The difference obtained in percentage of undersized individuals retained by both dredges is obviously related with its selectivity. Therefore, from the results obtained for the TD it can be concluded that the mesh size that has been used in the *C. chione* fishery, 50mm, is not the most adequate for the exploitation of this resource. In fact, the selectivity studies conducted by IPIMAR concluded that the mesh size used in the exploitation of this species must not be inferior to 70mm.

Although the selectivity of the TD may be improved, becoming identical to the NDD, by increasing the mesh size of the bag, the way that the dredge operates during the tow must be taken in consideration. The underwater observations carried out, allowed to verify that, during the tow, the individuals that are not damaged and that pass through the parallel rods of the NDD grid are buried immediately (in the case of the infauna) or recover their activity (in the case of epifauna). In the case of the TD, during the tow, the mesh of the net bag closes preventing that most of the individuals with smaller dimension escape. In this way, the gear only turns truly selective during the operation of hauling the gear, namely, when the net bag is being washed. In the case of bivalves, the fact that individuals are retained for a long time in the bag makes them rotate a lot, increasing the stress in which they are submitted. As a consequence, the bivalve that escape through the net bag do not bury themselves immediately, being subject to predation. The predation of exposed invertebrates was reported by several authors (e.g. Caddy 1968; Arntz and Weber 1970; Michael et al. 1990; Meyer et al. 1981; Kaiser and Spencer 1994; Gaspar 1996a). Meyer et al. (1981) found that 80% of the *Spisula solidissima* dislodged by hydraulic dredges reburied within 2 hours. Michael et al. (1990) have shown that almost all of the clams *Paphies donacina*, *Spisula aequilateralis*, *Dosinia anus*, *Mactra discors* and *Mactra murchiosini* were reburied after 20 minutes. Meyer et al., *op. cit.* stated that 3% of the undamaged clams were eaten by predators. Ramsay and Kaiser (1998) reported a high rate of predation of undamaged whelks by starfish in an area disturbed by scallop dredging. Their laboratory studies demonstrated that whelks that have been in contact in bottom gears and which have been rolled, took longer to straighten themselves and were less likely to have an escape response than whelks that had not been under the influence of the gear.

The difference of the mean fishing yield obtained in this experiment reflects a higher efficiency of capture of the NDD. This greater efficiency is a result of the fact that the sand that enters the dredge is rapidly filtered by both grids (Figure 43). Therefore, all the clams that are found on the dredge path enter the dredge. In the case of the TD the closing of mesh of the net during the tow prevents the sand from getting out through the mesh, blocking rapidly the dredge mouth (Figure 44). As a consequence, part of the clams that are found on the dredge path are pulled out of the dredge's mouth decreasing, therefore, its efficiency of capture.

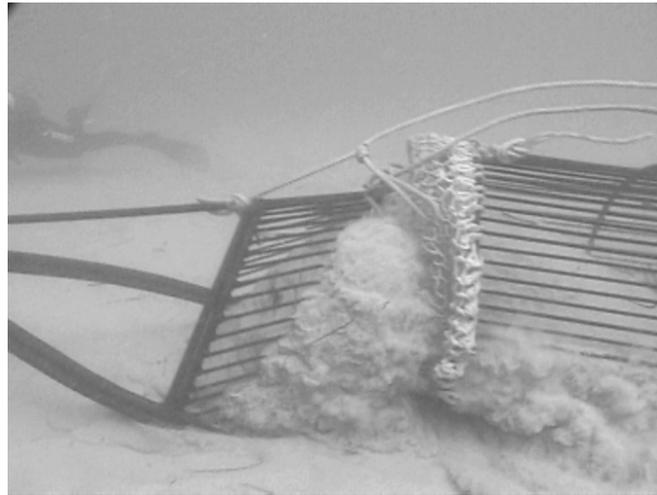


Figure 43 Underwater photograph of the NDD during the tow.



Figure 44 Underwater photograph of the TD during the tow.

Despite of the higher efficiency of capture of the NDD in comparison to the TD, the proportion of by-catch was significantly reduced when the NDD was used in the fishery. This is an important result since damage may occur during sorting on board of the vessels, before the animals are returned to the sea. Furthermore, if sorting times are long or conditions on deck are unfavourable, the survival of by-catch decreases (Medcof and Bourne 1964b). According to Fonds (1994.) the survival chances of flatfish and roundfish discarded are almost nil. For non-commercial invertebrates, however, part of the discarded by-catch may survive. Both starfish and brittlestars autotomise their arms as an escape response to predators or as a result of gear damage, subsequently regenerating their arms. Hence the survival of both orders are expected to be high. According to Kaiser and Spencer (1995) nearly 100% of the common starfish, *Asterais rubens* and nearly 80% of brittlestars, *Ophiura ophiura*, survive to captures in the nets of beam trawls. Similarly, Bergman et al. (1990) refer that common starfish have a high chance of survival after returning to the sea. Survival among undamaged juveniles may also be low if the time spent out of water on the deck is long, especially when air temperatures are high (Gaspar and Monteiro 1999).

Other important source of fishing mortality is damage caused to uncaught animals on the bottom. In clams, shell damage can result from animals passing under the dredge. These clams may be forced into the bottom and the mantle cavity packed with sediment which may damage the soft tissues (Caddy 1968). Damage also occurs on the benthic organisms that are caught by the dredge, due to the mass of debris or other fauna in the net or during their escape through the meshes of the net (Bergman and Hup 1992; Gaspar 1996a).

The study of direct mortality carried out in the present work aimed to evaluate the influence of tow duration and the type of dredge on the number of individuals damaged both on the target species and on the macrobenthic community. Our results showed that for both dredges, tow duration did not have any effect on the proportion of damaged individuals, neither on the target species nor on the macrobenthic community in general. Nevertheless, the influence of tow duration on the percentage of damaged individuals may be significant if the community is mainly comprised by vulnerable species. Gaspar et al. (1998) reported that for *E. siliqua* fishery the number of damaged individuals increases dramatically with tow duration.

The results of Gaspar et al. (1998) on the effect of tooth length on the number of damaged razor clams (*E. siliqua*) showed that damage to these stocks is inversely proportional to catching efficiency (defined as the percentage of clams on the dredge path that entered the dredge). Although it has observed that the efficiency of capture of the NDD is higher than the TD, the proportion of damaged individuals both of the target species and other macrobenthic species was similar. This result indicates that the way that individuals are retained by the dredge does not influence the number of damaged individuals. It is important to point out that we have no information on the proportion of damaged individuals that have come within the influence of the dredge but that were not caught. Nevertheless, we believe that one of the most important features of dredges is tooth length. The effect on bottom species depends on the action of fishing gear in relation to their vertical distribution. In this experiment the tooth length used on both types of dredges was the same and, therefore, the penetration on the sediment was presumably the same, resulting on the same proportion of damaged individuals.

Our results showed that mortality induced by both dredges on the macrobenthic community was very low. Direct fishing mortality has been found to be important in some dredge fisheries. Medcof and Caddy (1971) reported that 80% of the uncaught clams were damaged while only 20% of the individuals caught had the shells broken. Gruffydd (1972) estimated that the Manx scallop dredge efficiency was about 15% and that death due to natural mortality and indirect fishing mortality could reach 56.5%. He also reported that from the damaged undersized scallops caught and thrown back into the sea, between 50 to 75%, would die. Meyer et al. (1981) conducted a study in order to estimate the efficiency of the hydraulic dredge used in the fishery of *S. solidissima*. These authors observed that when the dredge performance was low about 92% of the larger clams were damaged. On the other hand, when the efficiency was high the mortality decreased to 30%. Naidu (1988) estimated, on the basis of dead shell: live shell ratio, that mortality on the fishery of *Chlamys islandica* was 4-8 times higher as a result of the encounter with the fishing gear than mortality from natural causes. McLoughlin et al. (1991) found that the catching efficiency of the Australian scallop dredge was very low. They estimated

that only 12% of the scallops in the tow path were caught. These authors reported that 12-22% of the initial stock in Banks Strait was landed as catch being the rest of the stock loss through direct and indirect mortality resulting from dredging. Michael et al. (1990) compared two hydraulic dredges to catch subtidal clams. Despite the high efficiency, they found that nearly 38% of the clams caught, although they hadn't shell damage, had severely or partly damaged feet. Lambert and Goudreau (1996) studied the harvesting efficiency of the New England hydraulic dredge used to catch Stimpson's surf clams (*Mactromeris polynyma*). They estimated a dredge efficiency of 90% for large clams. These authors observed that two-thirds of the clams that remained on the bottom were damaged which represents less than 10% of capturable clams. A small percentage of other species was also damaged. From the clams harvested nearly 20% were damaged. Despite of the high efficiency of the hydraulic dredges, the percentage of damaged individuals was higher than the observed in the present study. The mortality associated to hydraulic dredges may be related to water pressure, blade height or tow speed (Kauwling and Bakus 1979; Lambert and Goudreau 1996).

The ecological impact of the dredge fishery on shallow sandy bottom is not as deleterious as on hard bottoms or on deeper water. In fact on shallow waters, with a highly dynamic seabed environment resulting from natural physical disturbances, the infaunal communities are dominated by short-lived species adapted to frequent physical disturbances such as wave and tidal currents. Hence the impact of dredging on this areas may be limited. Nevertheless, it can be expected the loss of some species in the community, usually the larger-bodied species living buried in the sand. The effects on biodiversity are most severe where natural physical disturbance is least prevalent, where storm-wave damage is negligible and biological processes tend to be slower (Watling and Norse 1998; De Alteris et al. 1999). Bergman and Hup (1992) studied the direct effects of beamtrawling on macrofauna in a sandy sediment in the North Sea. These authors observed that the density of a number of species of echinoderms, polychaetas and molluscs decreased 10-65% due to trawling activity. Similarly, Kaiser and Spencer (1996a) concluded that beam-trawling caused significant short-term changes in the diversity and abundance of infauna species. Eleftheriou and Robertson (1992), concluded that dredging operations on sandy bottoms have a limited effect both on the physical environmental and on the smaller infauna but a high impact on some epifauna organisms and larger infauna. From these studies and from our results it can be concluded that the impact produced by both dredges in the area where this study was conducted is minimal. Although the direct mortality induced by the dredges assayed was similar, from a management and ecological point of view there are advantages in using the NDD in the *C. chione* fishery. Along the Portuguese coast this fishery is managed by daily quotas per boat, and fishermen are allowed to land only one species per day. Therefore, if we take in consideration the mean fishing yield obtained for both dredges, it can be concluded that the impact on the macrobenthic community is reduced in about 50% when using the NDD because to reach the daily quota the tow area is reduced to 50%.

As far as what has been exhibited, we consider that the New Dredge Design can be used in the exploitation of the bivalve molluscs. However, being a gear that represents a higher capture efficiency relatively to the Traditional Dredge, we underline the importance of accomplishing the maximum limits of daily capture per vessel

stipulated by the current legislation. Otherwise it can be assisted to the collapse of the stocks.

3.3.3.3 COMPARISON OF DIRECT MORTALITY INFLICTED ON MACROFAUNAL ORGANISMS BY THREE TYPES OF DREDGES USED IN THE PORTUGUESE CLAM FISHERY

Introduction

The exploitation of subtidal bivalve beds along the Portuguese coast is relatively recent, and was initiated in the late sixties. Although several species of commercial importance are harvested, only the white clam *Spisula solida* is caught by the whole dredge fleet, since it is the only species that occurs along the entire Portuguese continental coast. For management purposes the Portuguese coast was divided into three main fishing areas, the northwest, the southwest and the south area, which were defined taking into consideration the distribution of clam beds and fishing ports, the coastal topography and environmental conditions. Although the majority of technical measures used in the bivalve fishery management are similar in those fishing areas, differences are related to the number of fishing licenses, boat engine power and daily quotas per boat and species. In the Portuguese bivalve fishery only mechanical dredges are allowed, being constituted by a rigid iron structure with a toothed lower bar, and a collecting system. The main differences between the dredges used in the *Spisula solida* Portuguese fishery are related to shape and length of the dredge mouth and the collecting system. Figure 45 shows photographs of the three types of dredges used in this fishery. Until 1999, the dredge fleet only operated with dredge type 1 (northwest dredge fleet) and type 2 (southwest and south dredge fleets). Recently, a new dredge design (type 3) was introduced in the fishery and since then the majority of the fleet operating along the southwest and south coasts of Portugal started using this new gear. This dredge instead of using a net bag to retain the catch employs a metallic grid. Due to the weight of this new dredge, only small boats remained using the dredge type 2.

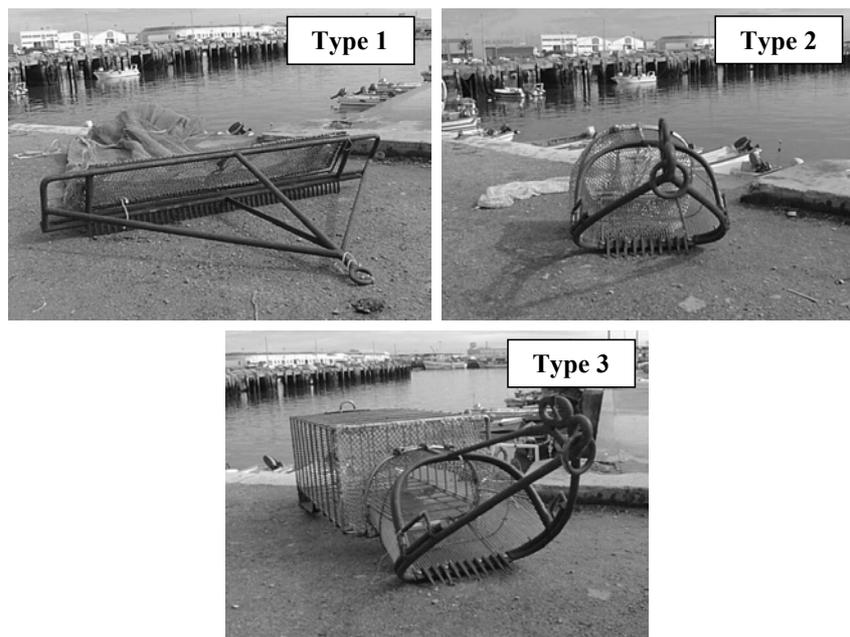


Figure 45 Photographs of the three dredge types used in the *Spisula solida* fishery. Type 1- Dredge used by the northwest fleet; Type 2- Dredge used by small boats; and Type 3- New dredge.

These dredges were designed to dig clams out of the sediment, causing impacts on benthic habitat both on physical structure and biological communities. Direct impacts include scraping and ploughing of the substrate, sediment re-suspension, destruction of the benthos and loss of biodiversity (e.g. van Dolah et al. 1987; Eleftheriou and Robertson 1992; Jones 1992; Currie and Parry 1996; Kaiser et al. 1996; Collie et al. 1997). Although the impact on the sediment provoked by the three dredges types used in the *S. solida* fishery is expected to be similar since the capture method is identical, the impact on the macrofauna may be different. Therefore it is important to estimate the direct mortality induced by each dredge type on the benthic macrofauna in order to introduce modifications on the dredges to reduce the mortality or even to ban the dredge types that cause higher impacts. During this study the direct effects of 3 different dredge types on macrobenthic mortality were compared. The relationship between this mortality and catching efficiency for each type of dredge was also assessed.

Methodology

Experimental design

The comparative study was conducted during June 2001 in the Sines region on the southwestern coast of Portugal, in a site off Lagoa de Santo André (38°02'99N; 008°49'78''W), which is one of the most important fishing ground for *Spisula solida*. The samples were collected on sandy bottoms between 8 and 10m depth. The study was carried out onboard the research vessel DONAX, which is of similar size and power of the commercial fishing boats. The dredges assayed were identical to those used by the commercial dredge fleet. In the present work the dredge usually employed in the northwest coast of Portugal was designated north dredge (ND), the dredge used

by small boats was denominated traditional dredge (TD), while the dredge fitted with a grid collecting system was named grid dredge (GD). Table 21 summarises the gear specifications of the dredges used in this study.

Table 21 Gear specifications of the dredges used in this study.

	Northwest dredge	Traditional dredge	New dredge design
Anterior part			
Length of the mouth (cm)	150	64	64
Space between rods (cm)	---	0.8	0.8
Number of teeth	49	10	10
Space between teeth (cm)	2	2.2	2.2
Tooth length (cm)	12	15	15
Tooth angle	20°	20°	20°
Net bag			
Length (cm)	450	250	---
Mesh size (mm)	25	25	---
Grid			
Space between rods (cm)	---	---	1.2

A total of twelve tows were accomplished, four for each dredge type. Dredges were towed for 5 minutes at a mean speed of 1.5 knots. Both tow duration and fishing speed used in this experiment were similar to those used by the commercial vessels operating with these types of dredges. The duration of dredge hauls was measured from the time the winch stopped paying out the towing cable to the time when the winch was restarted.

In order to determine the proportion of damaged individuals that entered the dredge but passed through the net bag during the fishing operation and the dredge retrieval the cover method was adopted, which consists in covering the net bag or the grid, with a second net bag with a smaller mesh size (20mm mesh). This cover bag was stretched loosely over the entire back of the dredges, enough slack being left in the cover to reduce any masking effect on selection and to allow the natural flow of water through the net as suggested by Gaspar et al. (1999). The catches of each dredge, coming from the bag or the grid and from the cover, were always processed separately. In order to estimate the macrofauna mortality inflicted by each dredge, for each organism caught the extent of any damage sustained was recorded using a four score scale (Table 22). This was visually assessed by the same individual on all occasions, on board during the sampling operations.

Table 22 Criteria used in the attribution of a damage score to each *taxon*.

Score	1	2	3	4
Bivalvia	In good condition	Edge of shell chipped	Hinge broken	Crushed / dead
Gastropoda	In good condition	Edge of shell chipped	Shell cracked or punctured	Crushed / dead
Cephalopoda	In good condition			Dead
Ophiuroidea	In good condition	Arms missing	Worn and arms missing / minor disc damage	Major disc damaged / dead
Crustacea	Anomura	In good condition	Out of shell and intact	Out of shell and damaged / Crushed / dead
	Brachyura	In good condition	Legs missing / small carapace cracks	Major carapace cracks / Crushed / dead
	Natantia	In good condition		Dead
	Osteichthyes	In good condition	Small amount of scales missing/ small cuts or wound	Large amount of scales missing/ severe wounds / Dead

Diving surveys were also conducted in order to estimate the percentage of damaged macrofaunal individuals left on dredge tracks, to determine the length of the dredge path and to estimate the dredges efficiency of capture. The efficiency of capture is defined as the proportion of the number of target species clams in the dredge path that enters the dredge (Caddy 1971). For each haul, divers randomly collected 54 sediment samples using quadrats (area=0.0625m²) within the dredge path: 27 corers in the furrow and 27 in the ridge. Samples were sieved *in situ* through a 5-mm mesh bag and on board the individuals collected were preserved in 70% ethanol. In the laboratory, the organisms were identified, counted and a damage score was attributed to each specimen caught using the damage scale described above. The species identification was made according to Tebble 1966), FAO (1987) and Poppe (1993). The nomenclature adopted was that of FAO (1987).

Data analysis

The PRIMER software package (Clarke and Warwick 1994) was used to investigate the influence of the capture method (grid *vs* mesh) on the number of individuals per species that escape through the meshes of the bag (ND and TD) or through the bars of the grid (GD). Abundance data from the cover was square-root-transformed prior to cluster analysis with Bray-Curtis to produce a similarity matrix. The relationships between samples were examined by nonmetric multidimensional ordination plots (MDS) and the analysis of similarities (ANOSIM) routine (Clarke and Warwick, 1994) was used to detect any strong difference on the dredges selectivity.

Analyses of variance ANOVA or Kruskal-Wallis ANOVA were used to investigate differences between the fishing yields obtained from each dredge and to test the effect of dredge design on the proportion of damaged and dead individuals. The damage inflicted by dredges on macrofauna was analysed separately for the individuals that entered the dredge and for the organisms left on the dredge path. Multiple comparisons were performed using the Student-Newman-Keuls test. Prior to the application of ANOVA or Kruskal-Wallis ANOVA, data was standardised and transformed to arcsine square root values when expressed as percentage. Statistical analyses were undertaken using the SigmaStat statistical software.

Results

A total of 29119 individuals belonging to 8 *taxa* were caught during the fishing experiments (

Table 23). The catches from the GD, TD and ND composed 52.9%, 37.4% and 9.7% of the total number of individuals caught, respectively. Bivalvia was the most represented *taxon* with 8 species, followed by Osteichthyes and Brachyura with 4 and 3 species respectively. Besides the target species *Spisula solida*, the most abundant species were the bivalves *Donax vitattus*, *Tellina tenuis* and *Ensis siliqua*, the crabs *Atelecyclus undecimdentatus* and *Leucarcinus depurator*, and the heart urchin *Echinocardium cordatum*.

From

Table 23 it can be observed that ND and TD retained almost all individuals that entered the dredge (93.9% and 97.1%, respectively), whilst when GD was used in the fishery, a smaller proportion of individuals was retained (76.1%). Cluster analysis and subsequent multidimensional scaling (MDS) of all samples collected from the cover using abundance data for each species or group revealed two main groupings of points (Figure 46). One group corresponded to GD and the other group contained ND and TD. The ANOSIM test that accounted for retention type effects (grid vs mesh bag) showed significant differences between the GD and both TD and ND ($R=0.969$, $p<0.001$), reflecting differences on the selectivity of these fishing gears.

Table 23 Total number of individuals that entered the dredges and retained in the cover.

	Species	North Dredge		Grid Dredge		Traditional Dredge	
		Total	Cover	Total	Cover	Total	Cover
Polychaeta	Polychaeta			12	12	5	0
Bivalvia	<i>Donax trunculus</i>	1	1				
	<i>Donax vitattus</i>	309	89	1392	1385	1441	152
	<i>Dosinia exoleta</i>			2	0		
	<i>Ensis siliqua</i>	2	1	117	110	45	0
	<i>Maetra corallina stultorum</i>	1	0	33	9	19	0
	<i>Spisula solida</i>	2347	31	12211	1343	8484	91
	<i>Tellina tenuis</i>	11	10	640	638	87	47
	<i>Venus striatula</i>			5	5	2	0
Cephalopoda	<i>Sepia officinalis</i>					2	0
Anomura	<i>Pagurus</i> spp.	4	4	5	2		
Brachyura	<i>Atelecyclus undecimdentatus</i>	61	10	705	47	478	2
	<i>Leucarcinus depurator</i>	39	20	150	129	295	28
	<i>Polybius heslowi</i>	4	1	21	2	14	0
Echinoidea	<i>Echinocardium cordatum</i>	25	4	120	7	12	0
Ophiuroidea	<i>Ophiura texturata</i>	1	1				
Osteichthyes	<i>Citharus linguatula</i>	1	0			2	0
	<i>Dicologlossa cuneata</i>					5	0
	<i>Trachinus draco</i>	1	1				
	<i>Trachinus vipera</i>	7	1				
TOTAL		2815	174	15413	3689	10891	321

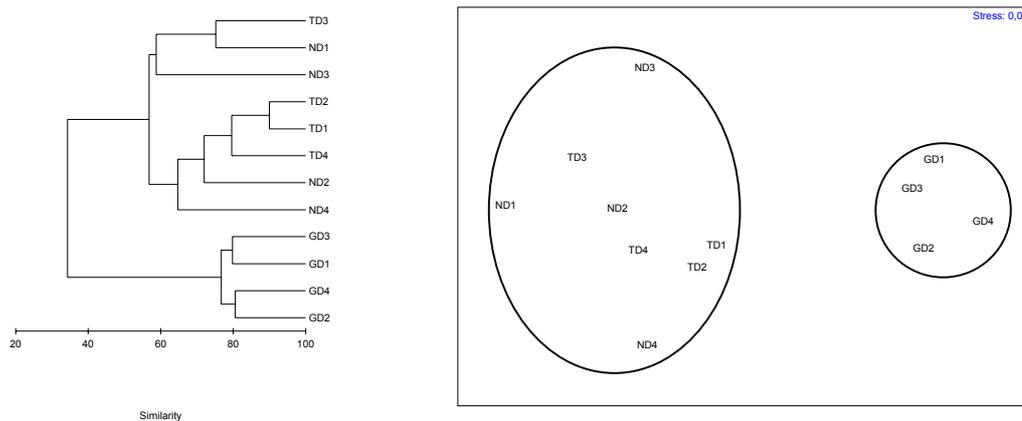


Figure 46 Bray-Curtis cluster analysis and Multidimensional Scaling Ordination (MDS) plot from cover data.

Table 24 resumes the data concerning the mean percentage of damaged (scores 2 to 4) and dead individuals (scores 3 and 4) that entered the dredges. Although the mean percentage of both damaged and dead individuals of the overall catch is very low, it was observed that ND damages and kills a slightly lower proportion of individuals (mean damaged = 3.3%; mean mortality = 2.5%) than GD (mean damaged = 5.0%; mean mortality = 4.8%) and TD (mean damaged = 7.4%; mean mortality = 5.9%). However, the statistical analysis carried out showed that gear type had no effect either on the percentage of damaged (ANOVA, $F = 1.48$; $P = 0.240$) or dead individuals (K-W, $H = 5.538$; $df = 2$; $P = 0.057$). Within the most abundant species, it was observed that the most affected by this kind of fishery were the thin-shelled bivalves *Ensis siliqua* and *Tellina tenuis*, the heart urchin *Echinocardium cordatum* and the crab *Atelecyclus undecimdentatus*.

Table 24 Mean number and mean proportion of individuals damaged and killed that entered the dredges, for each *taxon* and gear type.

	North Dredge					Grid Dredge					Traditional Dredge				
	TOTAL	Damaged		Mortality		TOTAL	Damaged		Mortality		TOTAL	Damaged		Mortality	
	N	(%)	N	(%)	N	(%)	N	(%)	N	(%)	N	(%)	N	(%)	
Polychaeta															
Polychaeta						2,9	0,6	20,0	0,6	20,0	1,2	0,6	50,0	0,6	50,0
Bivalvia															
<i>Donax trunculus</i>	0,3	0,0	0,0	0,0	0,0										
<i>Donax vitattus</i>	77,2	3,4	4,4	3,4	4,4	348,0	25,8	7,4	25,2	7,2	360,4	28,1	7,8	27,0	7,5
<i>Dosinia exoleta</i>						0,6	0,0	0,0	0,0	0,0					
<i>Ensis siliqua</i>	0,6	0,6	100,0	0,6	100,0	29,3	22,3	76,0	22,3	76,0	11,1	9,4	84,2	9,4	84,2
<i>Mactra corallina stultorum</i>	0,3	0,0	0,0	0,0	0,0	8,2	4,7	57,1	4,1	50,0	4,7	2,3	50,0	2,3	50,0
<i>Spisula solida</i>	586,8	11,0	1,9	5,8	1,0	3052,7	41,6	1,4	35,7	1,2	2121,1	87,3	4,1	55,7	2,6
<i>Tellina tenuis</i>	2,8	0,3	11,1	0,3	11,1	160,0	10,5	6,6	10,5	6,6	21,7	4,7	21,6	2,9	13,5
<i>Venus striatula</i>						1,2	0,0	0,0	0,0	0,0	0,6	0,0	0,0	0,0	0,0
Cephalopoda															
<i>Sepia officinalis</i>											0,6	0,6	100,0	0,6	100,0
Anomura															
<i>Pagurus spp.</i>	0,9	0,0	0,0	0,0	0,0	1,2	0,0	0,0	0,0	0,0					
Brachyura															
<i>Atelecyclus undecimdentatus</i>	15,3	3,7	24,0	3,7	24,0	176,4	63,9	36,2	63,3	35,9	119,5	37,5	31,4	35,2	29,4
<i>Leucarcinus depurator</i>	9,8	0,9	9,4	0,6	6,3	37,5	7,6	20,3	7,6	20,3	73,8	27,0	36,5	22,9	31,0
<i>Polybius heslowi</i>	0,9	0,6	66,7	0,6	66,7	5,3	2,9	55,6	2,9	55,6	3,5	2,3	66,7	2,3	66,7
Echinoidea															
<i>Echinocardium cordatum</i>	6,1	2,1	35,0	2,1	35,0	29,9	12,9	43,1	12,9	43,1	2,9	1,8	60,0	1,8	60,0
Ophiuroidea															
<i>Ophiura texturata</i>	0,3	0,3	100,0	0,3	100,0										
Osteichthyes															
<i>Citharus linguatula</i>	0,3	0,3	100,0	0,3	100,0						0,6	0,0	0,0	0,0	0,0
<i>Dicologlossa cuneata</i>											1,2	0,0	0,0	0,0	0,0
<i>Trachinus draco</i>	0,3	0,0	0,0	0,0	0,0										
<i>Trachinus vipera</i>	1,8	0,0	0,0	0,0	0,0										
TOTAL	703,8	23,3	3,3	17,8	2,5	3853,1	192,8	5,0	185,2	4,8	2722,9	201,6	7,4	160,5	5,9

The scuba-diving surveys carried out during this study allowed us to estimate the dredges efficiency of capture. For ND it was estimated an efficiency of capture of 64% and the mean incidental mortality on uncaught white clams was in the range of 5-20%. Higher efficiencies of capture were estimated for both GD (98%) and TD (90%) and for both dredges no damage on the uncaught white clams were observed. It is interesting to emphasise that higher efficiencies result in lower proportions of damaged individuals left on the dredge path. From Table 25, it can be observed that ND damages and kills an higher proportion of uncaught individuals than GD and TD.

Table 25. Mean number and mean proportion of damaged and dead individuals left on the dredge path, for each *taxon* and gear type.

	North dredge					Grid dredge					Traditional dredge				
	TOTAL	Damaged		Mortality		TOTAL	Damaged		Mortality		TOTAL	Damaged		Mortality	
	N	%	N	%	N	%	N	%	N	%	N	%	N	%	
Bivalvia															
<i>Donax vitattus</i>	147,0	0,0	0,0	0,0	0,0	17,3	0,0	0,0	0,0	0,0	75,3	58,0	77,0	58,0	77,0
<i>Spisula solida</i>	326,0	146,7	45,0	33,3	10,2	58,0	0,0	0,0	0,0	0,0	233,3	0,0	0,0	0,0	0,0
<i>Tellina tenuis</i>	402,0	217,7	54,1	217,7	54,1	1170,7	208,7	17,8	208,7	17,8	1060,7	300,7	28,3	185,3	17,5
Anomura															
<i>Pagurus spp.</i>											4,0	0,0	0,0	0,0	0,0
Brachyura															
<i>Atelecyclus undecimdentatus</i>						2,0	2,0	100,0	2,0	100,0					
Echinoidea															
<i>Echinocardium cordatum</i>	6,0	6,0	100,0	6,0	100,0						1,3	0,0	0,0	0,0	0,0
TOTAL	881,0	370,3	42,0	257,0	29,2	1248,0	210,7	16,9	210,7	16,9	1374,7	358,7	26,1	243,3	17,7

Similarly, the mean percentage of both damaged and dead uncaught individuals from TD is higher than those obtained from GD. The results of one-way ANOVA showed that gear type has an effect either on the percentage of damaged ($F = 10.114$; $P = 0.005$) or dead ($F = 4.341$; $P = 0.048$) individuals left on the dredge track. Student-Newman-Keuls multiple pairwise comparison showed significant differences between the N and both GD and TD both in terms of damaged and dead individuals. Within the dredge tracks, bivalve species were the most abundant group composing nearly

100% of the total number of macrofaunal individuals collected. Among these, *Tellina tenuis* was the most affected species followed by *Donax vittatus*.

Fishing yield is directly related to gear efficiency of capture and therefore differences in the mean fishing yield (Kg/ 5 min. tow) obtained for each dredge were observed. From Figure 47 it can be seen that mean fishing yield registered both for the GD and TD was substantially superior to that observed for the ND. The one-way ANOVA analysis performed revealed significant differences ($F = 16.486$; $P = 0.004$) in the mean fishing yield obtained for the dredges assayed. The application of the Student-Newman-Keuls test showed the existence of significant differences in the mean fishing yield, between GD and ND (S-N-K, $P < 0.05$) and between TD and ND (S-N-K, $P < 0.05$).

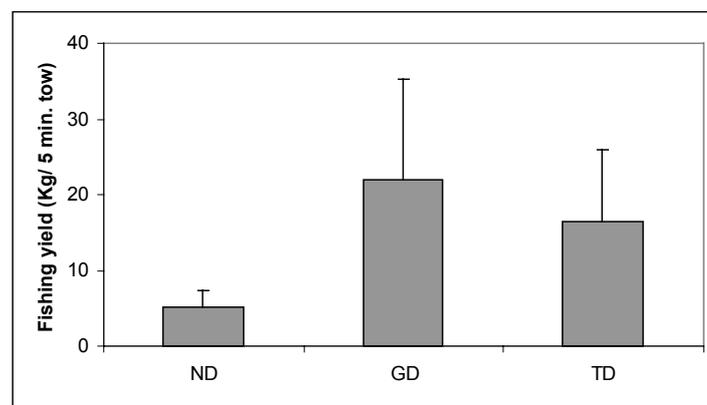


Figure 47 Standardised mean fishing yields (Kg/ 5 min. tow) obtained for the three dredges assayed.

Finally, for each dredge and tow, data from the bag, cover and dredge path were pooled in order to estimate and compare total mortality. Table 26 resumes the data obtained for each dredge and tow in terms of damaged and dead individuals percentage. Its analysis shows that for the overall community ND damages and kills a higher proportion of macrofaunal individuals than GD and TD. The Kruskal-Wallis One Way Analysis of Variance on Ranks revealed the existence of significant differences both in terms of damaged (K-W, $H = 8.769$; $df = 2$; $P = 0.001$) and dead individuals percentage (K-W, $H = 6.615$; $df = 2$; $P = 0.024$). The Student-Newman-Keuls Pairwise Multiple Comparison showed significant differences ($P < 0.05$) between the ND and both GD and TD for damaged and dead individuals.

Table 26 Percentage of damaged and dead individuals obtained per tow and dredge type.

	Tow									
	#1		#2		#3		#4		Mean	
	Dam (%)	Mort (%)								
North Dredge	36,35	26,79	19,51	11,64	18,30	13,26	30,44	21,47	26,15	18,29
Grid Dredge	11,92	11,40	9,82	9,57	6,03	6,00	5,64	5,50	8,35	8,12
Traditional Dredge	11,27	8,44	16,27	12,75	10,78	7,12	17,26	11,73	13,89	10,01

Discussion

In this study, it was determined and compared the direct mortality on the macrobenthic community inflicted by three types of clam dredges used in the *Spisula solida* fishery. Total direct mortality was assessed taking into consideration the degree of damage sustained by individuals that entered the dredges plus those individuals damaged in the dredge path. Our results showed significant differences on total direct mortality between the north dredge and both the grid and traditional dredges. These differences were largely attributed to the mortality of animals that died in the dredge track as a direct result of physical damage inflicted by the dredge passage. In this study it was found that damage on uncaught individuals is directly related with gear efficiency. The lower catching efficiency of the north dredge led to an higher proportion of damaged individuals left on the dredge path, when compared with the more efficient grid and traditional dredges. This relation between catching efficiency and damage was also observed by other authors. Meyer et al. (1981) reported that when the efficiency of dredges was low, larger clams, which burrowed deeper into the sediment, suffered mortalities as high as 92%, while when high, mortalities decreased to 30%. Caddy (1973) referred that the low efficiency of the Alberton dredge was responsible for causing lethal and sublethal damage to scallops left in the track, which were greater on rough bottom. McLoughlin et al. (1991) concluded that in addition to its low catching efficiency, the Australian mud dredge damages many more scallops than it catches producing a post-fishing mortality rate seven times the estimated natural mortality rate for *Pecten fumatus*. However, it should also be emphasised that the maximum dredging impact may not occur immediately after dredging, suggesting that some indirect ecological changes, such as exposed organisms becoming vulnerable to predation, may be taking place (Currie and Parry 1996). The attraction of epifaunal scavengers and predators to fished areas has been recorded in other studies (Meyer et al. 1981; Kaiser and Spencer 1994; Lambert and Goudreau 1996; Ramsay et al. 1996). Analysis of diet composition of scavengers collected from trawled areas indicated that they feed primarily on animals that were damaged or disturbed by the trawl (Kaiser and Spencer 1996b).

Besides direct mortality from being caught and indirect mortality due to predation on uncaught clams, there might be further mortality on discarded individuals, especially if sorting times are long and conditions on deck are unfavourable (Medcof and Bourne 1964b). Furthermore, re-location into unsuitable habitat and predation while returning to seafloor may also contribute to increase mortality. This type of mortality also depends on many conditions, such as depth, type of species, individual's size, degree of damage and predator concentration. Gaspar and Monteiro (1999) reported that length of exposure was directly related to juvenile *Spisula solida* mortality. Robinson and Richardson (1998) found that undersized *Ensis arcuatus* individuals

returned to the seabed were slow to re-bury becoming highly vulnerable to attack from predatory crabs. These two examples illustrate the importance of designing high selective dredges. Our results showed that GD retained a significant smaller proportion of captured individuals than TD and ND, reflecting differences on the collecting system used in the dredges (metallic grid vs net bag). When a net bag is used to retain the individuals, mesh stretches while the dredge is being towed preventing the escapement of organisms through the meshes. Therefore, the dredge only becomes selective during the hauling process. On the contrary, when the metallic grid is used, selection of the captured individuals occurs during the tow. Gaspar et al. (In press) and [Section 3.3.3.2](#) reported that undamaged individuals that pass through the parallel rods of the GD grid burrow immediately (in the case of the infauna) or recover their activity (in the case of epifauna). This rapid reburying response decrease the probability of dislodged organisms being predated.

From our results it can be concluded that there are considerable direct effects on some benthic species as certain groups of animals suffer heavy damage although others are less affected. Consistently, studies have demonstrated that there is an immediate effect in the density of non-target organisms after mobile fishing impacts. The short-term environmental effects of dredging on the sea-bottom have received increased attention in the last decade, and several studies detected changes in the ecosystem due to dredging (e.g. Hall et al. 1990; Eleftheriou and Robertson 1992; Kaiser and Spencer 1996b; Lambert and Goudreau 1996). Short-term effects are also expected in Portuguese bivalve dredge fishery, but the question is whether this type of fishing causes long-term effects in the benthic community structure.

The biological communities that utilise a particular habitat have adapted to that environment through natural selection, and, therefore, the impact of mobile fishing gears on the habitat structure and biological community must be scaled against the magnitude and frequency of seabed disturbance due to natural causes (De Alteris et al. 1999). Although for various species mortality due to dredging appears to be fairly high, recolonization can occur over a relatively short time period. Currie and Parry (1996) using a before-after-control-impact design experiment reported the size and duration of scallop dredging impacts on soft sediment communities. These authors stated that reductions in density caused by dredging were usually small compared to annual changes in population density, that is, seasonal and particularly inter-annual changes were greater than those caused by dredging. Kaiser et al. (1998) found that immediately after fishing the composition of the community in stable sediments was significantly altered, while in mobile sediments the effects of fishing were not detectable. Nevertheless, after six months, seasonal changes had occurred in both communities and the effects of trawling disturbance were no longer evident. Similarly, Hall et al. (1990) found that despite suction dredging for *Ensis* sp. had profound immediate effects on benthic community structure with consistent reduction in many macrofaunal species, after 40 days the abundance of species returned to pre-impact levels. On contrary, Pranovi and Giovanardi (1994) found that hydraulic dredging produces considerable negative effects (long term) on the bottom environment of the Venetian lagoon. These authors hypothesized that the slow recovery of the infaunal community was related to the medium/low energy conditions of the lagoon environment. Benthic communities inhabiting deeper waters may be less capable of sustaining disturbance than benthic populations that occur on shallow

waters in more dynamic coarser sediments and accordingly have much longer recovery times (Jones, 1992).

Besides sediment type and conditions at the site, the severity of accumulated fishing effects also depend on the scale and intensity of the activity. If a major proportion of a fishing area is affected then it is quite conceivable that the scope for movement by the associated benthos would reduce and the recovery takes longer (Hall 1994; Thrush et al. 1995). Moreover, although the effects of a single passage of a gear may be relatively limited, chronic fishing disturbance may produce long-term changes in benthic communities (Sainsbury 1988; Collie et al. 1997; Jennings and Kaiser 1998; Bradshaw et al. 2000). Nevertheless, evidence suggests that long-term changes in mobile sediments are probably restricted to long-lived fragile species (Eleftheriou and Robertson 1992). Therefore, population reductions may only persist if the sediments in which they live are immobile (eg. Kaiser et al. 1998; Ball et al. 2000) or the affected area is large relative to the remainder of the habitat so a dilution effect cannot occur (Kaiser et al. 1998).

Thus, given the depth (< 35m) and the type of sediment (sandy bottoms) at which fishing is practised along the Portuguese coast, as well as the relatively high natural disturbance found all year round, clam dredging is unlikely to have persistent effects on most infaunal communities. The effects on long-lived bivalve species could, however, be more serious. From this study, it was found that ND damages and kills a higher proportion of macrofaunal individuals than GD and TD. It was also found that for the same tow duration the ND mean fishing yield is significantly lower than those obtained with GD and TD. Finally, our data showed that GD is more selective than the other two dredges assayed. These results clearly show that there are advantages in using GD in the white clam fishery, which are more evident when the results obtained with GD are compared with those registered with ND. Furthermore, since this fishery is managed by daily quotas per boat it can be concluded that the use of GD instead of ND leads to a reduction of approximately 40% in the impact over the macrobenthic community, because to attain the daily quota the towed area is reduced in the same proportion. Therefore, and as a conclusion, from a management and ecological point of view there are obvious advantages in using GD in the *Spisula solida* Portuguese fishery.

3.3.4. Summary Statement

The effect of dredge design on environment has been studied on two ways during this project:

- Investigation of innovative features for scallops dredging.
- Improvement of clam dredge design.

Investigation on innovative features for scallops dredging

Three innovative features have been investigated: the “Magnus effect”, the foil and the hydraulic or water-jet system. They have been chosen to avoid the use of teeth since dredge teeth are considered to be the component with the most impact on the environment.

These features are used to create an upward flow of water in order to transform the scallop dredge into a kind of “vacuum cleaner” to avoid, as far as possible, the contact of the dredge with the sea bed. The results of the investigation of the three new features are summarised:

Magnus effect

The Magnus effect is the transverse force acting on a rotating cylinder perpendicular to a fluid flow. This effect is used on the dredge to deflect the flow behind the cylinder. This deflected flow near the seabed is aimed at lifting the scallops from the seabed. Unfortunately the efficiency of such dredge was found to be very low, even after several configurations of cylinder diameter and rotating speed were tested. Another drawback is that such system is active; it needs energy, not only the towing force, but also energy for the rotation of the cylinder. This makes it more complicated than a classical dredge which is a passive system.

Foil

A foil can deflect water flow. This principle is expected to be used on a dredge to lift scallops from the sea bed. In fact two effects are necessary for scallops dredging. The first is a horizontal water flow just near the bottom to remove scallops from their position; that is to unstick from the sediment. The second is to lift scallops upward to be collected in a the dredge bag.

The investigation of such system has been made through flume tank tests and numerical simulations on the simplest foil: i.e. plate. Several configurations of length foil, inclination and distance to the bottom have been studied. If for several configurations the horizontal water flow near the bottom seems sufficient, the lifting effect was not strong enough. In fact due to vortices the effect of the foil is not constant all the time.

Even so, this system is very promising, because it is passive which means that needs only the energy of the towing speed, it needs more investigation to define the best shape of foil to reach a strong and constant effect especially on the lifting part. This investigation could be done firstly by means of numerical model and further examined by flume tank tests.

Water-jet

A water-jet in the sea, drags the surrounding water. If this water jet, on a dredge, is near the seabed, the scallops can be lifted by this current up to the bag of the dredge.

A visit of Italian clam dredges using this system permitted to define the main components of a prototype, the operating use of it and finally to design the prototype.

Sea trials on the water jet dredge prototype and on a classical dredge showed that the damage level to the scallops with the water jet dredge lower than with the classical dredge (4 times less). Unfortunately the efficiency also decreases (3 times less). Stress on the scallops was low in the hydraulic dredge catch and also those which had been left on the seabed after dredging. However, this could have been the result of the very short tow length; short simulations of dredging gave similar results.

With development these results could perhaps be improved on. However, due to the complexity of the water jet dredge (water pump, pipes) this principle seems not adapted to scallops dredging.

Improvement of Portuguese clam dredges

Along the Portuguese coast exists an important bivalve fishery, which is undertaken using mechanical dredges. These dredges were designed to dig clams and razor clams out of the sediment causing deleterious effects on the benthic habitat, both in terms of the sediments physical structure and benthic communities. In order to minimise the impacts it is important to carry out studies for determining the direct mortality inflicted by dredges, as well as the indirect mortality resulting from dredging. Direct mortality corresponds to those individuals (caught and uncaught) that will die in consequence of the damage degree suffered, while indirect mortality corresponds to those undamaged individuals that will die in consequence of the fishing process. This includes those individuals that are not caught but that will come within the influence of the gear being dislodged and exposed, which will die due to predation, and those individuals discarded that will die due to the high level of stress induced by prolonged deck exposure, re-location into unsuitable habitat or predation. Hence, fishing gear impacts on benthic communities may be minimised by improving gears through changes in their specifications or by developing new gears. In the present work it was observed that mortality is correlated with catching efficiency. Therefore, in order to decrease mortality, it is important to improve or develop fishing gears that are more selective and simultaneously more efficient. With these aims, new dredges have been developed that differs from the traditional ones in the way that the catch is retained. This new dredge, instead of using a net bag to retain the catch, employs a metallic grid that is 10 cm apart from the sediment by using a kind of skis.

In order to evaluate the potential introduction of this new dredge in the clam fishery the direct mortality inflicted on the macrobenthic community by this dredge was determined. This was compared with the direct mortality induced by the other dredge types (northwest and traditional dredges) used in the Portuguese clam fishery. Total direct mortality was assessed, taking into consideration the degree of damage sustained by individuals that entered the dredges plus those individuals damaged in the dredge path.

Our results showed significant differences on total direct mortality between the Northwest dredge and both the new and traditional dredges in the North west *Spisula solida* fishery. No significant differences on the proportion of dead individuals retained were observed between dredges, the differences on direct mortality were largely attributed to the mortality of animals that died in the dredge track as a direct result of physical damage inflicted by the dredge passage. The lower catching efficiency of the Northwest dredge led to a higher proportion of damaged individuals left on the dredge path, when compared with the more efficient new and traditional dredges.

The underwater observations carried out, allowed verification that, during the tow, the individuals that are not damaged and that pass through the parallel rods of the new dredge grid are immediately buried (in the case of the infauna) or recover their activity (in the case of epifauna). In the case of both Northwest and traditional dredges, the mesh of the net bag closes during the tow preventing the escapement of smaller individuals. In this way, the gear only turns truly selective during the operation of hauling the gear, namely when the net bag is being washed. In the case of bivalves, the fact that individuals are retained for a long time in the bag makes them rotate a lot, increasing the stress to which they are submitted. As a consequence, the bivalves that escape through the net bag do not bury immediately, being subjected to predation. Another advantage of using the new dredge is that the proportion of bycatch is significantly reduced when compared with the proportion of bycatch retained by the other two types of dredges.

Nevertheless, the development of more efficient dredges also results in higher fishing yields for fishermen, increasing the fishing pressure over bivalve beds that may lead to the overexploitation of the commercial resources. However, since the Portuguese bivalve fishery is managed by daily quotas per boat and species, the introduction of the new dredge in the fishery will reduce the daily fishing time and consequently the dredged area, with all inherent ecological benefits.

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4. DISCUSSION

This study systematically examined the physical, chemical, biological and ecological effects of the dredging process. The aim was to describe these effects in relation to dredge design and selectivity with a view to reducing undesirable effects of dredging.

The means used for measuring the environmental and selectivity effects of dredging are reviewed below with a discussion of the interactions where appropriate. This is followed by discussion of the main findings of the report relating environmental effects to dredge design, selectivity and fishing effort. Finally, there is a consideration of the ways in which results from these and similar studies could be used in the management of dredge fisheries.

4.1 Measurement of Environmental Effects

[Review Section 5](#) examines the literature on environmental effects up until spring of 2000. The discussion below considers the results of this project in the context of literature published since this time. Factors affecting the execution of the studies and validity of the results are described. The studies of the environmental effects can be divided into four main themes, physical, chemical, biological and ecological.

4.1.1 Physical effects

Amongst the most predominant environmental effects were the physical, mostly mechanical, effects both on the organisms encountered and on the substrate. This study was able to develop methods for monitoring physical ([Sections 3.1.1.1-2](#)) and simulating biological effects on the organisms ([Section 3.1.1.3](#)), and also monitoring and modelling the physical effects on the substrate ([Section 3.1.1.1](#) and [3.1.1.4](#)). These methods enabled the study of critical aspects of dredge design and operation, and could be used as a basis for research into future dredge designs.

The physical effects of scallop dredging on both the seabed and organisms were studied using instrumentation ([Section 3.1.1.1](#)), both mounted on the dredge and contained within scallop analogues (instrumented scallops), damage indices ([Sections 3.1.1.2](#) and [3.1.4](#)), video ([Section 3.1.1.1](#)) and diver observations ([Section 3.1.4.1](#)). The effects of scallop dredging were simulated in the laboratory and related to behaviour and physiological stress ([Sections 3.1.1.3](#), [3.1.3.1-4](#)). Hydraulic clam dredging (turbo soffianti in the N. Adriatic) was studied using instrumentation and modelling ([Section 3.1.1.4](#)) and towed dredge clam dredging (Portugal) was studied using video observations ([Section 3.1.5.2](#)).

Scallop dredging

[Review Section 5.1.1](#) discusses the effects of scallop dredging on rocky substrates, and mearl beds. This study concentrates on the physical effects of scallop dredging on what should be considered as 'normal' scallop dredging areas.

Observations of the action of spring-toothed scallop dredges, using both the instrumented scallop ([Section 3.1.1.2](#)) and by video observations, indicate that most of the physical effects of scallop dredging occurred at the teeth which was usually the first point of contact with the dredge. For all species the efficiency of the dredge was low (<25% by number: [Section 3.1.4.1](#) and Beukers-Stewart et al. 2001), and it is

clear that the majority of the damaged megafauna are left behind in the dredge tracks after encountering the gear. Thus descriptions of the effects of dredging cannot be confined to those organisms in the discarded by-catch.

The possibility of modelling the interaction between the dredge teeth and the sediment using models designed for seabed ploughing was investigated ([Section 3.1.1.1](#)). However, because it was found that the teeth were at an angle of greater than 90° to the seabed, it was not possible to use these models. What the models did indicate was that raking the teeth at less than 90° to the seabed could have resulted in increased instability. The mechanism by which dredges caught scallops was examined by video and a working hypothesis described. Scallops were observed to be rotated out of the seabed by the downward force of the teeth. It is suggested that the damage to scallops could be related to the dredge bouncing on the seabed when its equilibrium is perturbed. This would explain why damage to the scallops occurred predominantly on the flat valve which is level with the seabed when scallops are recessed.

Percentage damage levels in the catch were consistent from year to year on particular fishing grounds ([Section 3.1.1.2](#)), suggesting that damage was related to some feature of the substrate. Scallop dredging grounds are very heterogeneous (Review [Section 2.1.1](#) and [Section 3.1.4.1](#).) so dredges may be constantly changing their equilibrium state as they move from substrate to substrate. This would be expected to result in the bouncing behaviour observed. Therefore the largest physical impact may be a result of movement from one substrate to another. This suggests that some form of damping system on the dredge springs might be effective in reducing damage levels. However, more research is needed into what factors affect the equilibrium of the dredges, possibly using data from accelerometers *in situ* on the dredge frame compared with seabed discrimination data and data on damage levels.

Hydraulic dredging

[Review Section 5.1.2](#) categorised the physical effects of hydraulic dredging on the seabed as compression due to the blade runners, cutting action due to the thin steel blade behind the jets, jetting action and sediment re-suspension (see [Review Section 3.2.2](#) for dredge design). These studies concentrated on modelling the jetting action.

The effects of hydraulic dredging (*turbo soffianti*) was more easily related to models of sediment behaviour than scallop dredging and an empirical relationship was found between the depth of fluidisation, nozzle pressure and flow rate, mean particle size, and speed of the dredge over the seabed ([Section 3.1.1.4](#)). There appeared to be some variability between sites, which was also observed in the damage rate and physiological effects (see below).

Portuguese clam dredging

[Review Section 3.2.1](#) describes basic Portuguese clam dredge design and [Sections 3.3.3.2-3](#) describe and discuss the effects of innovative dredge designs.

Physical effects of Portuguese clam dredging were investigated by divers taking videos of the action of the dredge on the seabed ([Section 3.1.5.2](#)). It was observed that the dredge built up a sand buffer inside and in front of the dredge as it was towed along the seabed. The dynamics of the sand buffer affected the distribution of the

organisms found after dredging, and hence the sampling strategies required to adequately describe the effects on the benthos. Altering the design of the collecting bag by making it into a cage reduced the sand buffer effect and improved the flow of catch into the dredge and the passage of selected organisms through the bars of the cage. This was clearly visible on the video taken by divers.

4.1.2 Chemical effects

The Review ([Review Section 5.2](#)) found few references to the likely chemical effects of dredging except that the action of the dredge might alter the flux of nutrients between the surface layer of the sediments and the benthos.

Chemical effects were studied in the Portuguese clam dredge fishery and in the turbo soffianti hydraulic dredging in the N. Adriatic ([Section 3.1.2](#)). The parameters studied included nutrient fluxes and chlorophyll *a* distribution. The sediments can be a source of nutrients due to recycling occurring within interstices. Phytobenthos contain chlorophyll *a* and are sediment surface dwelling microscopic plants. The ability of divers to locate and sample the small-scale features of the dredge tracks in these fisheries substantially assisted the understanding of these processes. The main features were short-term perturbations in nutrient flux and changes in chlorophyll *a* distribution. Laboratory experiments, which investigated the fate of phosphates after dredging, indicated that they were likely to be re-adsorbed rapidly onto the surface of suspended particles after dredging ([Section 3.1.2.1](#)). Thus, since nitrogen compounds would continue to be released, there may be a change in the Nitrogen: Phosphorus ratio if large areas were frequently disturbed. Similar effects were observed in both fisheries studied and it was considered that, unless effort was very intense, chemical effects were likely to be minimal, particularly in areas where there was high-energy input into the seabed from other sources such as tidal currents and wave energy.

4.1.3 Biological effects

[Review Section 5.3](#) examines the biological effects of dredging on the target species and on other organisms encountered. Time series data on the growth of scallops on heavily exploited grounds in the bay of St Brienc and comparison with less heavily exploited grounds off west Brittany suggests that heavy exploitation of scallops stocks over the period since 1979 has affected their growth. Although these data are open to other interpretations, they do suggest that prolonged heavy fishing places a load on the metabolism of the scallops or affects the environment in ways that affect their growth.

Also discussed in the review ([Review Section 5.3](#)) are observations of efficiency and non-catch mortality of both target and non-target organisms. The studies reviewed suggest that there is a need to investigate the fate of both damaged and undamaged organisms encountering the dredges.

This study used a range of methods to characterise the biological effects of dredging on the individual organisms, particularly the target species encountered by the dredges ([Section 3.1.3](#)). In order to survive dredging, bivalves must be able to defend themselves against predation as soon as possible after contact with the dredge.

Scallops do this by swimming ([Section 3.1.3.1](#)) and recessing (Minchin 1992; Fleury et al. 1996; Minchin et al. 2000) into the substrate, while clams burrow into the substrate to avoid predation. This study investigated the behavioural response of scallops encountering a predator and the effect of stress due to dredging on this response ([Section 3.1.3.1](#)). The acute biological stress and behavioural indices described the short-term physiological effects of dredging ([Sections 3.1.3.1-5, 7,8](#), the predation and scavenger studies ([Sections 3.1.4.2, 3.1.5.1](#); scallops, [3.1.5.3](#) and [3.3.3.2-3](#) for Portuguese clams) were intended to examine the consequences of these effects on the ability of the affected organisms to survive dredging.

The chronic biological stress ([Section 3.1.3.6](#)) indices were intended to examine longer-term effects of dredging on the affected species. There were also studies of the effect of dredging at the community level ([Section 3.1.5](#)).

Acute stress, damage and predation

These studies are summarised in Tables 1, and 2 where Section references are given. For the acute biological stress indices it was possible to set up a framework to examine the effects. In most cases it was possible to simulate dredging and demonstrate changes in the indices that were similar to those found in the field studies, and to obtain adequate controls. Lethal levels were obtained from literature sources, but in most cases bivalves proved more resilient than reported, surviving levels of stress that might be considered lethal.

Table 1 Acute biological stress indices

Index & Section	Species	Method	Results
Behaviour in response to stimulation with a predator (<i>Asterias rubens</i>) 3.1.3.1	<i>Pecten maximus</i>	Dredging was simulated and animals were then immobilised in tanks and aspects of their behaviour observed when stimulated with a starfish arm.	Demonstrated a reduction in the swimming ability of captured undersized scallops. Effects persisted for at least 24 hours, suggesting that dredged scallops remain vulnerable to predation for at least that period.
Adenylic Energy Charge (AEC) Righting and reccessing ability 3.1.3.2-4	<i>Pecten maximus</i>	Comparison of different sizes (60, 70 and 80 mm), different lengths of tow (0,0.5, 1, 2, 8, 15, 16, 30, 45 mins), effect of emersion, clamping scallops, diurnal variation, speed, cumulative effects of dredging at 2 speeds (simulated), season and site (ground effects)	Dredged scallops and simulated dredged scallops show similar AEC levels of 0.5. Size was important; smaller scallops had a higher AEC level and behavioural score than medium and larger animals. Simulated dredging showed increased stress up to 15- 30 minutes of dredging but no significant increase beyond this. Dredging followed by emersion had an added stress effect. AEC levels of clamped scallops were not a low as simulated dredged scallops for the same time period. There was no diurnal variation in AEC level but animals maintained in the dark were more active. The speed of agitation had an important impact on both indices but no cumulative effect of stress was shown at the slower speed (faster speed not tested). No differences between site (grounds), but differences between seasons; lowest levels in February, coinciding with the period of lowest reserves of glycogen.
Adenylic Energy Charge (AEC) 3.1.3.5	<i>Chamelea gallina</i>	Sample collected manually (control) Sample collected by dredging at high pressure and using a sorter (undersized and commercial size);	Seasonal difference, lowest values in July. Difference between high and low pressure but no difference between sorted and unsorted clams in Febuary. Generally no difference in AEC between sizes, however, in Lido in October, commercial sized clams had significantly higher AEC levels than undersized clams. A significant difference in cumulative effects was observed at 24 hours but not at 48 hours.
Re-burrowing 3.1.3.5.	<i>Chamelea gallina</i>	Sample collected by dredging at high pressure and no sorter;	Results similar to AEC but reburrowing rates were faster in July. Difference between high and low pressure treatments but no difference between sorted and unsorted animals. Generally no difference in behaviour between sizes. Cumulative trend shown (control different from treatment at 48 hours) but this result was not significant.
Damage 3.1.4.4	<i>Chamelea gallina</i>	Sample collected by dredging at low pressure and no sorter All samples collected seasonally	Damage increased with increasing size of clams. Also difference in damage between different methods of collection (i.e. higher damage in high pressure treatments). A difference was observed between animals that were sorted and high pressure, and those not sorted
Immunological (Haematocrit and Phagocytic Index) 3.1.3.5.	<i>Chamelea gallina</i>		Both indices decreased with increasing stress i.e. between low and high pressure but the results were not clear between the treatments high pressure sorted and not sorted.

Adenylic Energy Charge (AEC) ATP/mg dry weight Reburrowing time 3.1.3.7-8	<i>Spisula solida</i> <i>Donax trunculus</i>	Comparison of AEC using two techniques, HPLC and luminometry. Cumulative effects experiment Seasonal study (before, during and after dredging). Aerial exposure experiment of <i>Donax trunculus</i> , before and after dredging at 0, 30, 60, 120 minutes.	No significant differences between HPLC and luminometry techniques. A correlation between length and AEC was observed. Residual values were used to eliminate the effect of size. Changes in AEC, ATP ug dry wt and reburrowing time can be related with increased aerial exposure. $\mu\text{gATP/mg dry wt}$ was considered a better index than AEC.
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Table 2 Predation and scavenger studies

Index	Species	Method	Results
Field survival of dredged scallops 3.1.4.2	<i>Pecten maximus</i>	Scallops were subjected to treatments: Control, no dredging or emersion; Stressed (by simulated dredging and emersion) but not damaged; and stressed (as above) and damaged. Scallops than tethered on longlines or in crab pots to expose, or protect them from predation respectively for a period of 20 days.	Almost all of the scallops in the crab pots survived, indicating that most of the 48% tethered scallops which died had been attacked by predators. However, no significant differences were observed in the mortality of the three treatments. The caveat which has to be added to these results is that there were very few <i>Asterias rubens</i> present on the grounds which are important predators of scallops.
Predator and scavenger activity 3.1.5.1	<i>Pecten maximus</i>	Scallops were tethered adjacent to a camera at three levels of treatment: Stressed (by simulated dredging and emersion); Stressed plus slight damage; and Stressed plus high levels of damage. They were then observed using an underwater video camera for 96 hours during which time the behaviour of animals attracted to the scallops was observed.	Over the periods monitored a number of species were observed to be attracted to the scallops, and the composition of the scavenger community is likely to have an effect on the fate of the scallops; for example crabs facilitated the feeding of fish on the scallops. There was no aggregation of predators on the undamaged scallops but a marked increase of scavengers on the damaged scallops. As with the above experiment there was an absence of <i>Asterias rubens</i> , which may have limited the chance of the stressed scallops being preyed upon.
Aggregation of predators on dredge tracks 3.1.5.3.	Clam species Sea urchins	Diver observation after dredging of predator aggregation on the dredge tracks. Dredge type was the new Portuguese 'cage' type dredge.	Divers observed rapid aggregation of scavenging species <i>Ophiura albida</i> onto dredge tracks, which then dispersed rapidly within 20 minutes. All undamaged bivalves had already buried immediately after being released from the dredge. There were other damaged organisms; thin-shelled bivalves and sea urchins, which were clearly affected by the scavengers.

The effects of stress and damage on vulnerability to predators and scavengers depended on the presence of a relevant community of predators in the environment to which the samples were exposed. It appears that there are certain key predators that tend to make the bivalves vulnerable to other scavenging animals; if these are not present then there is less vulnerability to predation ([Section 3.1.5.1](#)). In these examples ([Section 3.1.4.2](#) and [3.1.5.1](#)) only damage, not stress, increased the bivalves' vulnerability to predation. However, an important predator of scallops, the starfish (*Asterias rubens*), was not common during these experiments on scallops, though it had been common at this site four years earlier (Veale et al. 2000a). It is likely that, if *Asterias* had been present, mortality of the stressed but undamaged scallops would have ensued.

The results obtained from observations of scavengers aggregating on Portuguese dredge tracks illustrate how rapidly scavengers can arrive and disperse from the dredge track; this can occur within 20 minutes ([Section 3.1.5.3](#)). At the same time small bivalves buried very rapidly after passing through these highly selective dredges; only damaged animals, the majority of which were thin-shelled species, were vulnerable to the scavengers. The Portuguese dredges used in this study were of a novel design that was shown to be highly selective, and with higher efficiency, than the traditional dredge ([Section 3.3.3.2](#)).

Chronic stress indices

Chronic stress indices (See Table 3 for those used in this study and Section references) were originally designed to evaluate the levels of stress imposed on populations of bivalves by pollutants in the marine environment. Gradients of stress would be expected in these circumstances which would enable the definition of control sites against which stress levels can be monitored. An important difficulty with the application of these indices to these fisheries was the definition of appropriate control sites. Clam fishing in the areas of the Northern Adriatic, where the project performed investigations on *Chamaelea gallina*, is a widespread activity, and it proved difficult to find adequate control sites.

Thus the study examined these indices on clams from various parts of the fishing operation: controls collected by divers, clams collected with low and high pressure dredging, and clams collected before and after mechanical sorting. This approach provided information on the response of clams to mechanical stress on a seasonal basis. Differences between locations were observed, both for acute and chronic indices, and damage levels ([Sections 3.1.3.5, 3.1.3.6 & 3.1.4.4](#)), suggesting that stress level was related to environmental conditions such as sediment types. For the Portuguese clam fishery the indices of chronic stress used (nucleic-acid based indices and neutral/polar lipids) suggested a difference in response after spawning but this was not significant (Table 3).

Table 3 Chronic stress indices

Index & Section	Species	Method	Results
Survival parameters 3.1.3.6	<i>Chamelea gallina</i>	Sample collected manually (control)	Survival in air showed significantly different LT_{50} values between low water pressure sample and high water pressure with sorted sample; this parameter seldom reflects the effects of sorter. Manually collected clams not always have the expected behaviour. Undersized clams result less stressed than commercial sized ones.
Physiological parameters 3.1.3.6	<i>Chamelea gallina</i>	Sample collected by dredging at high pressure and using a sorter; Sample collected by dredging at high pressure and no sorter;	A general worsening of physiological clam conditions, expressed by SFG value, is observed with increasing stress due to different fishing systems; this correlation seems to be mostly influenced by filtration rate. SFG value exhibits seasonal variations.
Biochemical parameter: catalase activity 3.1.3.6	<i>Chamelea gallina</i>	Sample collected by dredging at low pressure and no sorter.	Presence of seasonal trend with values that are lower in winter. There were no significant differences among the three different fishing systems and the control.
Histochemical parameter: Latency 3.1.3.6	<i>Chamelea gallina</i>	All samples collected seasonally	No presence of seasonal trend. There were no significant differences among the three different fishing systems and the control.
Nucleic acid based indices: RNA/DNA DNA/DW, residuals 3.1.3.7 , 3.1.3.6	<i>i su la solida</i> <i>Donax trunculus</i> <i>Chamelea gallina</i>	Samples collected seasonally Samples collected before dredging, during dredging and after dredging Control samples collected manually before dredging by divers	Presence of seasonal trend with lower condition in winter and after spawning. There was a decline of condition after dredging, more intense after spawning, but not significantly.
Neutral/polar lipids 3.1.3.7	<i>Spisula solida</i> <i>Donax trunculus</i>		

4.1.4 Community effects

In Review Section 5.4 studies on the ecological effects of dredging are discussed. These included short-term and long-term studies, studies of the fate of species encountering the dredges and the aggregation of predators and scavengers in the dredge tracks.

The investigation of community effects ([3.1.5](#)) was approached in several ways:

1. Studies of predation and scavenging on organisms affected by the dredge (Sections [3.1.4.2](#), [3.1.5.1](#) and [3.1.5.3](#)).
2. Comparisons between benthic communities as found by benthic sampling ([Section 3.1.5.6](#)), and observations of damage levels in the catches and behind the dredge on the seabed using divers ([Section 3.1.4.1](#)).
3. Examining spatial differences in macrofauna in the dredge track and relating these to dredge behaviour ([Section 3.1.5.2](#)).
4. Examination of the effects of dredging and recovery over periods of up to 2 months ([Section 3.1.5.4](#)).
5. Comparison between dredged and un-dredged areas; the un-dredged areas had been left un-fished for 4 years ([Section 3.1.5.5](#)).
6. Comparison with samples taken in previous decades before fisheries were introduced ([Section 3.1.5.6](#)).

These approaches provide differing sources of information on the ecological effects on the communities affected. The studies of predation (1) and comparisons with other sampling methods (2) indicate which groups of organisms are affected by dredging and their relative chances of survival. Sampling regimes in studies comparing the distribution of benthic organisms before and after dredging should be related to gear behaviour (3), since this can affect the spatial distribution of the organisms on the seabed after dredging. It is important to establish whether there is sufficient statistical power in the sampling strategy to distinguish changes (Bradshaw et al. 2000). Short-term recovery studies (4) could be used to examine the effects of dredging in relation to other variables such as seasonal changes in sea conditions and tidal scour.

Comparisons between the dredged and control areas would clearly be useful since ecosystems are likely to undergo long-term changes and hence this is one way of studying the effects of dredging on whole systems (5). However, practical difficulties occur because fishing is often so widespread that finding comparable un-fished areas to act as controls is difficult or even impossible. Two areas off southern Portugal, one fished intensively and the other un-fished for the past four years, were compared in this study. Differences between the two areas in meiofaunal and macrofaunal diversity, abundance, biomass, relative abundance of feeding groups, and occurrence of species with different body-forms and life-spans were largely in accord with those expected to result from differences in fishing (Collie et al. 2000; Jennings et al. 2001; Bradshaw et al. 2002). However, contrary to theoretical expectations, macrofaunal

biomass was higher in the fished area, largely due to the presence of the relatively large, long-lived *Spisula solida*, the main target species of the fishery. This suggests that the un-fished area was not sufficiently similar to be a good control site and that a substantially longer period of time may be necessary for the *Spisula solida* population to re-establish in the un-fished area.

Comparison with past samples was investigated (6) but it was clear that differences in sampling and identification between the studies undertaken in this project and previous studies precluded adequate interpretation of the differences observed. However, with a conservative approach to data analysis and interpretation, valid comparisons can be made over long time periods (Reise and Schubert 1987; Buhs and Reise 1997; Frid et al. 1999). Indeed, in the absence of good quality data series and experiments, comparisons with poorer quality historical data is often the only possible way to determine long-term change and can yield valuable results (Bradshaw et al. 2002).

4.2 Measurement of Dredge selectivity

Dredge selectivity is discussed in [Review Section 6](#). This review compares the habitats and behaviour of the two scallop species *Pecten maximus* and *Aequipecten opercularis* and describes the biological basis for the current Minimum Legal Landing Sizes (MLLS) for *P. maximus*. Studies of the relative efficiency and selectivity of scallop and Portuguese clam dredges are discussed.

This section discusses our studies of selectivity and relates the results to bivalve behaviour. The results are divided into species ([Section 4.2.1](#)) and size selectivity ([Section 4.2.2](#))

4.2.1 Species selectivity

Aspects of the bivalves' behaviour could be related to species selectivity in the gear. Seasonal variations in catchability in queen scallops (*Aequipecten opercularis*) were related to seasonal differences in behaviour ([Section 3.2.1.1](#)). Over the course of 13 months observations were conducted using a 'top net' set to catch queen scallops swimming over the top of the normal toothed dredges. During the period from June to mid or late autumn queen scallops were found in the 'top net', indicating that they were swimming in advance of the arrival of the gear and so avoiding capture. These results indicated clear changes in the seasonal behaviour of this species of scallop. This avoidance behaviour was related to the reaction time of the queen scallops under laboratory conditions; during these months the queen scallops reacted more quickly and so avoided the dredge, but there was no difference in the scallops' ability to perform adductions.

Direct comparison between skid and toothed dredges targeting queen scallops showed that the catch per unit effort of skid dredges was equal to or exceeded that of toothed dredges ([Section 3.2.1.1](#)). King scallops (*Pecten maximus*) were caught as a by-catch in toothed queenie dredges but in much smaller quantities in skid dredges. Comparison between the habitats of the two species may explain these results. Queen scallops live on the surface of the sediment, compared with the king scallops that recess into the surface layer. Thus teeth are required to dig king scallops out of the

sediment, whilst queen scallops can be scraped off the sediment surface by tickler chains.

However, there may be some element of active swimming that makes the queen scallops more vulnerable to tickler chains. The field and laboratory experiments indicate that queen scallops are more likely to avoid the dredges by swimming over them during the summer, which is when the comparison between the skid and toothed dredges was carried out. It is not known whether skid dredges would be equally efficient during the winter and early summer when the queen scallops' reaction time was shorter. Further work is required to investigate this aspect.

In Portuguese clam dredge fisheries it is considered ([Section 3.2.2.2](#)) that species selectivity was affected by tooth length, related to the normal burrowing depth of the clam species, and hence tooth length is the important parameter in species selectivity of these dredges.

For non-target species ([Section 3.1.4.1](#)) there are clear differences in the catchability and damage rates between species encountering scallop gear, and thus the catch is not a representative sample of the animals encountering the gear. For scallop dredging the numbers of animals affected and left on the seabed mostly exceeded those retained in the gear.

4.2.2 Size selectivity

Clam dredges

Experiments were performed using three levels of tooth spacing and three levels of mesh size in the *Spisula solida*, and for *Callista chione* and *Donax trunculus* fishery off NW and southern Portuguese coasts respectively ([Sections 3.2.2.1-2](#)). Cover nets were used to examine the population of bivalves available to the mesh and comparative catches to investigate selectivity due to tooth spacing. Three species of bivalves, *Spisula solida*, *Callista chione* and *Donax trunculus*, were investigated and it was found possible to obtain good selectivity data for all these species. Although tooth spacing was not found to affect selectivity, it is clear that there is maximum tooth spacing and a minimum tooth length to allow efficient operation of the dredge on a given species (see above). Mesh size was found to have a clear effect for all these species and the results from this work allowed a mesh size to be defined for this fishery.

Spring-toothed scallop (*Pecten maximus*) dredges

This was studied by taking a systematic approach to comparing catches from dredges using varied sizes of the different components of the gear within a balanced experimental design ([Section 3.2.2.3](#) and Lart et al. 1997). The selectivity of the gear was investigated by comparing the relative efficiency by length class of bivalve. Fivas et al.(in prep) found that absolute efficiency (as obtained by comparison between catch and diver observations) varies by scallop length, both for small and large sizes of scallops, with efficiency peaking at a particular length. These studies describe relative efficiencies between gear components of differing dimensions. It was also possible to investigate interactions between the effects of these factors because, within the designs used, different levels of the same factor were used in combination. For example, large mesh backs were used with large and small ring size bellies.

The results for scallop dredges indicated that only the belly ring size and tooth spacing were important in selectivity. The mesh size of dredge backs and the ring size of chain mail backs on the dredges did not affect selectivity, and there was no interaction between the tooth spacing and ring size.

Tooth spacing affected selectivity, but the effects were not consistent (Sections [3.2.2.3](#) and [3.2.2.4](#)); it appeared to depend on the design of the teeth and tension in the springs. The instrumentation studies ([Section 3.1.1.1](#)) indicated mean tooth penetration of 25-35mm into the seabed. With a tooth length during the experiments of 85-90mm this indicates a space of 50-65mm between the tooth bar and the seabed. This suggests that there is a space available between the teeth and the seabed to act as a selectivity aperture. This would vary with tooth penetration into the seabed, which could be variable with substrate type. When the teeth were shortened they became less selective, which is what would be expected, as the tooth bar would be closer to the seabed and the effective aperture smaller.

Although it was possible to show that tooth spacing affected selectivity (Section [3.2.2.4](#)), definitive values of L_{50} and other selectivity parameters were difficult to obtain. This is because of the difficulty in obtaining a tooth spacing that would catch a representative population sample, and the L_{50} would probably also vary between substrates because of variation in tooth penetration. The smallest tooth spacing used was clearly catching higher proportions of small scallops but it was possibly less efficient at catching larger scallops.

The ring size and dimensions of the wire making up the rings of the bellies had a clear effect on selectivity ([Section 3.2.2.4](#)). These results indicated that the selectivity of the rings on spring-toothed scallop dredges, as indicated by the L_{50} and other parameters, was similar to that found for French diving plate dredges. The models of ring selectivity ([Section 3.2.2.6](#)) suggest that the most important aperture is the ring itself and that changes in the geometry of the rings or washers would have an effect on ring selectivity.

The selectivity characteristics of the bellies changes due to wear and distortion of the rings with use ([Sections 3.2.2.3-5](#)). The ascending limb of the length frequency curve moves to the right as the bellies wear ([Sections 3.2.2.3-4](#)). The result is that the rings become more selective with time, but the larger rings also begin to lose commercial sized larger scallops earlier than the smaller rings. Changes in the design of the bellies; thicker wire used for the rings and variations in the thickness of the washers can have an effect of the pattern of change in selectivity with wear ([Section 3.2.2.4](#)).

4.3 Dredge Design and Environmental Effects

The main dredge designs, both those used on the target species studied and other species in the participating nations' fisheries, are reviewed in Review Section 3. This study investigated several elements of dredge design that potentially or actually had an effect on dredge impacts.

Novel capture mechanisms

Novel mechanisms to lift scallops (*Pecten maximus*) into the dredge bag were investigated as prototypes or models. The intention was to substitute or modify the teeth, which are considered to be the dredge component that has the most effect on the environment. These included:

- 'Magnus effect' systems which utilise pressure differences created by an immersed rotating cylinder to lift scallops off the seabed ([Section 3.3.1.1](#)).
- The performance of hydrodynamic foils was examined to establish whether the foil could be used to generate sufficient lift to remove scallops from the seabed ([Section 3.3.1.2](#)).
- Modification of the tooth design based on designs for seabed ploughs used in the offshore cable industry ([Section 3.3.1.3](#)).
- Hydraulic or water-jet dredges ([Section 3.3.1.4](#)).

Of these mechanisms hydraulic or water-jet systems were considered the most likely to be successful as replacement for the fixed teeth used on French scallop dredges. The hydrodynamic foils were considered to have potential but further work is required on foil design is required before this can be realised. Accordingly a hydraulic dredge, based on the classical french scallop dredge ([Section 3.3.2.1](#)) was designed and constructed and compared with the classical counterpart in use in the fishery.

Environmental impact comparisons

Over the course of the study the environmental effects of a number of dredge designs were compared. These were:

1. Hydraulic scallop (*P. maximus*) dredges were compared with the classical French dredges (with fixed teeth and no diving plate) ([Section 3.3.3.1](#)).
2. The skid dredges were compared with toothed dredges targeting queen scallop (*Aequipecten opercularis*). The use of various configurations of net set above the skid dredge to catch swimming queen scallops was also investigated ([Section 3.2.1.1](#)).
3. Two Portuguese clam dredge designs, one targeting *Callista chione* ([Section 3.3.3.2](#)) and the other *Spisula solida* ([Section 3.3.3.3](#)), were also compared with traditional dredge designs. In these, the bag for collecting the catch was radically

altered making it a rigid cage with fixed apertures for selection. This not only improved selection but improved the flow of sediment through the dredges by eliminating the sand buffer effects observed in the operation of the traditional dredge designs ([Section 3.1.5.2](#)).

Comparisons were made using a selection of the techniques described in this project; selectivity comparisons, stress levels, damage levels of catch and by-catch, impact on macrofauna and relative and absolute efficiencies. In all cases there was some improvement in at least one aspect of the dredges' environmental impact.

Hydraulic dredges

The hydraulic scallop dredge ([Section 3.3.2.1](#)) showed reduced damage rates and stress levels compared with the catches of the classical French dredge ([Section 3.3.3.1](#)). However, the observed reduction in scallop stress level was probably mostly related to the very short haul duration, and the efficiency rates were lower than for the classical dredge. This type of dredge could be suitable in an aquaculture system where there are strong incentives to avoid damaging the scallops, and the dredge can be purpose designed for use on a particular substrate. Further developments, perhaps using models of hydraulic dredge effects as discussed in this project ([Section 3.1.1.4](#)), could improve the performance of the hydraulic scallop dredge.

The results of the study of the sediment mechanics of turbo soffianti hydraulic dredging suggest that the models described and tested ([Section 3.1.1.4](#)) would have some application in the design of these dredges, perhaps leading to minimal disturbance of the sediments for a given catch of clams. Levels of many of the acute (Table 1) and chronic biomarkers (Table 3) and damage indices (Table 2) were related to the pressure used in the dredge; more pressure resulted in more stress or damage. Thus, use of these models to optimise the pressure used could help to reduce these aspects of the impact of the dredge.

Queen Scallop dredge

In the catches of the skid dredges designed for queen scallop there was a reduction in the by-catch of king scallops relative to the toothed dredges. Also, queen scallop dredges have much smaller ring sizes than the king scallop dredges and so a reduction in the by catch of king scallops in a targeted queen scallop fishery would be beneficial in reducing stress on undersized king scallops. Catches and damage rates of other benthic megafauna were similar in both the skid and toothed dredges, but no investigations were made of scallops and other macrofauna left on the seabed. There was some reduction in the relative proportions of rocks (predominantly cobble sized) caught in the skid dredges.

The experiments using various configurations of net set above the skid dredges showed that the use of a cone shaped 'top net' was more efficient at catching swimming queenies than a cover net which directed the queenies downwards and into the dredge.

Portuguese clam dredges

The Portuguese dredge design changes are likely to result in reductions in the stress levels in the species selected because selection happens more rapidly in these dredges.

The damage rates of catch and by-catch species were similar in *Callista chione* dredges, but the rate at which the clams were selected was increased and the overall efficiency was improved relative to the traditional design. The novel *Spisula solida* dredge, which has been in use by larger vessels in the fleet, was compared with two traditional designs (one from the NW fishery and the other used by small boats). Differences were found in the catch composition, damage rates, selectivity and efficiency of the three dredges. The novel design showed the lowest percentage of damaged individuals in the catch, taking into account both those animals caught in the dredge and cover, and those left behind on the seabed. An important benefit of this novel dredge design would be a reduction in the amount of impact per clam landed because of the increased efficiency of the dredges. However, the impact reduction and stock management objectives can only be realised if a suitable management scheme is in place to avoid the increased efficiency resulting in increased catches.

These results show that it is possible to alter the impact of dredges, and define and monitor those impacts using the types of indices developed in this project. In the most successful cases these improvements have been achieved through the modification of existing designs. Of the dredges studied, the skid dredges for catching queen scallops (*Aequipecten opercularis*) and the two innovative dredges for Portuguese clams (*Callista chione* and *Spisula solida*) are in commercial use in the fisheries.

4.4 Selectivity and Environmental Effects

4.4.1 Scallops

Experiments on behavioural changes and AEC⁷ level, and response to predation, showed that it was possible to examine the effects of stress on scallops during dredging, and to examine the effects of selection at the seabed as opposed to sorting the catch on deck ([Section 3.1.3.2-4](#)). Figure 1 illustrates the AEC levels in scallops at various stages in the dredging process. These data were collated from different experiments ([Sections 3.1.3.2-3](#)) during the course of the study and are intended to illustrate the amount of stress reduction that can be achieved by selection at the dredge, compared with on deck.

⁷ Adenyle Energy Charge; an indication of the level of immediately available energy to the organism see [Section 3.1.3](#)

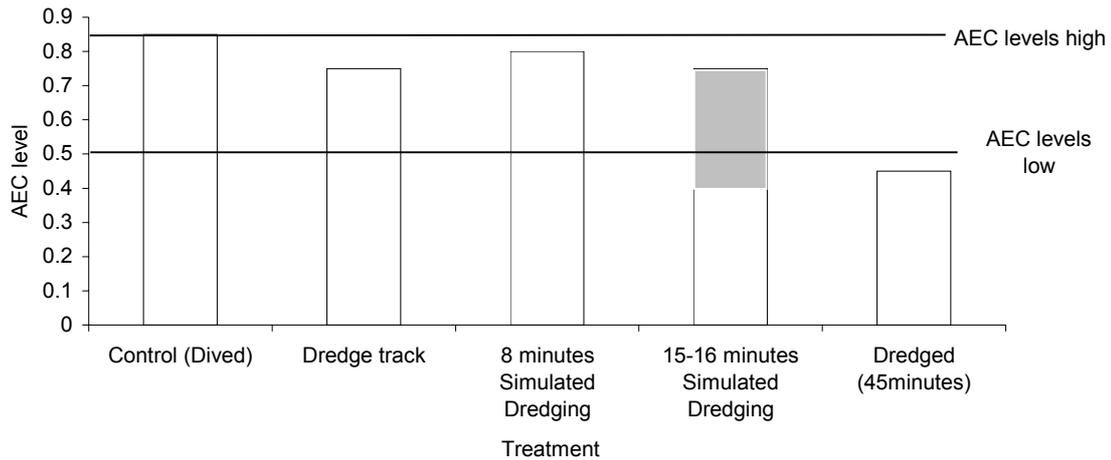


Figure 1 Adenylic Energy Charge in scallops (*Pecten maximus*) after various treatments. The grey area on the 15-16 minutes of dredging bar indicates the range of values obtained for this period of simulation.

Scallops are only mildly stressed by short periods of dredging of 8 minutes, with stress being more variable at 15-16 minutes, but they are consistently more stressed by dredging followed by sorting on deck. Those scallops collected from the dredge tracks had stress levels similar to those which had undergone 8-16 minutes of dredging.

Thus the benefit of selection at the seabed is a reduction in AEC depletion; instead of falling from about 0.85 to 0.45 it falls to only 0.75, a 75% reduction in stress. Recovery from stress in terms of AEC level takes about 6 hours from simulated dredging (Section 3.1.3.4). The behavioural studies (Section 3.1.3.1) indicate that normal behaviour does not resume for at least 24 hours. Thus stress levels are affected for approximately one tide and behaviour for at least 24 hours. Section 3.1.5.1 shows that this is adequate time for predator to aggregate, if they are present, as demonstrated also in other studies (Kaiser and Spencer 1996; Ramsay et al. 1997; Veale et al. 2000a). Further studies (Sections 3.1.4.2 and 3.1.5.1) to examine whether increased predation occurred in stressed as compared to damaged or unstressed (control) scallops indicated that predation was more related to damage than to stress. However, a caveat must be added to these field experiments because there were very few starfish (*Asterias rubens*), a major predator of scallops, present in the area in which they were carried out, though they were common at this site four years earlier (Veale et al. 2000a).

Of the two mechanisms of selection described, tooth selection and ring selection, it would be advantageous to select the scallops at the teeth, because this would result in the minimum contact time with the dredge. Video observation of a limited number of scallops interacting with the teeth found the maximum contact time is around 30 seconds and most contact times were much shorter at 1-3 seconds (Section 3.1.1.1). However, damage rates of scallops retained in the dredge bag and those found behind the dredge by divers (Section 3.1.4.1) were not significantly different, indicating that not much extra damage is done during selection at the rings. Also ring selection

proved very much more reliable ([Section 3.2.2.4](#)) than tooth selection which did not occur at all in some circumstances ([Section 3.2.2.3](#)).

This indicates that both tooth spacing and ring size should be optimised to minimise stress on scallops during the dredging process. However, this approach would not be expected to make a substantial alteration to damage rates since most of the damage appears to occur at first point of contact with the teeth ([Section 3.1.1.2](#)).

4.4.2 Portuguese Clam dredging

The results of the experiments on stress in *Spisula solida* indicate that the cumulative stress due to the mechanical action of the dredge, and to temporary aerial exposure, affected the stress levels of undersized animals in ways that potentially reduced their survival ([Sections 3.1.3.7](#) and [3.1.3.8](#)). Damage rates were low but not significantly different for both the different tooth spacing and different mesh sizes ([Section 3.2.2.2](#)). However, tooth spacing had no effect on size selectivity, and selection in traditional clam dredges occurs at the end of the tow, thus prolonging the mechanical stress. Improvements in dredge design ([Section 3.3.3.2-3](#)) have resulted in reduced damage rates (both for target and non-target species), a more rapid rate of selection and improved efficiency. Observations of predator aggregation on the dredge tracks ([Section 3.1.5.3](#)) indicated that, with clams burying rapidly after selection, predation occurred on the damaged, mostly non-target by-catch. Thus these improvements should result in not only reduced stress in the undersized animals but a reduction in the overall non-catch mortality per unit catch of clams.

4.5 Fishing effort and environmental effects

European dredge fisheries in the participating nations are described in [Review Section 4](#). These descriptions include target species, resource characteristics, commercial by-catch, fishing strategy, markets, interactions with other fisheries, and trends in vessels and management of the fisheries. Also included are tabulations of regulations in terms of input and output controls.

The fishing strategies and local management legislation are the two main factors controlling disposition of effort spatially and temporally. Spatial distribution of effort is controlled by the density of available bivalves ([Section 3.1.4.3](#) and [Review Section 4.1.1](#)) and closures for management and public health (harmful algal blooms and sewage contamination [Section 4.2](#)) reasons. Temporal distribution of effort is controlled by seasonal reproductive cycles that cause changes in the marketability of the bivalve, and seasonal management measures. Overlying all these factors are likely to be seasonal and spatial trends in the market for the target species.

Strategies adopted by scallop fishers in the UK Channel fisheries are described in [Review Section 4.1.6](#), and for Portuguese clam fisheries in [Section 3.1.4.3](#). For both these fisheries the patterns of effort are similar: bivalve beds are exploited until the catches reach uneconomic levels; they are then left for a period, perhaps years, until the densities reach levels that will once again allow economic catches. In the English Channel and Western Approaches the UK fleet counters the patchiness of scallop distribution in two ways. The larger vessels (>20m) deploy large numbers of dredges over considerable distances and will probably therefore exploit a number of patches during the course of a single tow. Smaller vessels (<15m), having identified a patch

producing high catch rates, will concentrate on it with shorter, circular tows until catch rates decline. They then move off in search of a new high-density patch. Overall effort levels tend to correspond to the amount of stock available at economic densities (Palmer in prep see also [Review Section 4.1.6](#)).

In French scallop fisheries, effort on listed beds is heavily controlled by management measures ([Review Section 4.5.2](#)). There is also the effect of improving technology, such as more powerful vessels and seabed discrimination technology, resulting in better targeting and hence increased catchability; this is particularly evident in the French fisheries (discussed in [Review Section 4.5.2](#)) but would be expected to happen in other fisheries. Effort in the turbo soffianti clam fisheries is considered to be high in relation to the available area for fishing and is restricted to given days of the week for this reason ([Review Section 4.6.4](#)).

These factors, and the accuracy or otherwise of the available statistics, make experimentation on the environmental effects of different effort levels very difficult, as was found when control areas were sought in the N. Adriatic. However, some of the results of this project have been orientated towards the description of the temporal and spatial effects of fishing effort.

4.5.1 Seasonal effects

Seasonal changes were observed in both the acute (scallops; [3.1.3.3](#), clams; [Section 3.1.3.5](#)) and chronic stress responses in clams ([Section 3.1.3.6](#)). These could be related to other aspects of the animal's reproductive and growth physiology, and to seasonal conditions such as temperature.

For scallops the stress effects could be related to the glycogen content of the muscle and gonad. The animals had the lowest levels of glycogen in the muscles in February and the highest in October, and these corresponded to periods of lowest and highest AEC in the adductor muscles in both the dredged and control scallops. Glycogen was lowest in the gonad during October but by February it had increased, suggesting a mobilisation of reserves for gonadal development and a corresponding reduction in the animals ability to respond to stress, hence the lower AEC level. In the behavioural studies ([Section 3.1.3.1](#)), only one parameter was observed to vary between seasons, the number of adductions which the animals made on first stimulation with the starfish arms, which was lower in the spring than in the autumn; this also possibly reflects the immediate energy available to the adductor muscles.

Clams (*Chamaelea gallina*) in the N. Adriatic exhibit seasonal changes in several biomarkers such as scope for growth and survival in air ([Section 3.1.3.6](#)). However, because of the natural endogenous stress levels due to breeding condition, and the natural variation in environmental conditions, it would be difficult to separate out the period when the highest change in these parameters may be produced by fishing. An adequate control population would improve the resolution of this problem.

4.5.2 Recovery time and cumulative effects

Acute stress and recovery in scallops was investigated in relation to speed and duration of tow and the cumulative effect of repeated daily stress ([Section 3.1.3.4](#)). It was found that recovery rates were related to the speed of dredging (as defined by the

speed of the simulator). At levels of acceleration approximately corresponding to normal dredging, scallops will recover to normal levels of AEC within 6 hours of dredging. Lower levels of acceleration (corresponding to approximately half normal dredging speed) results in a more rapid recovery with normal levels of AEC reached in 2 hours.

Simulated tow lengths of 15 and 30 minutes did not result in significant differences in AEC level but tow durations of less than this resulted in reduced stress (Section [3.1.3.2](#)). Repeated dredging on a daily basis for 3 days at both high intensity of simulation and low intensity did not affect the level of stress found the following day (Section [3.1.3.4](#)). For scallops exposed to low intensity simulation on a daily basis for three days, the reduction in the AEC level was less on the first day of simulated dredging than on subsequent days. However, recovery time was similar (2 hours) for each of the three days tested. At a higher intensity of simulation the AEC levels were reduced to the same level each day for three days of repeated simulation. The rate of recovery was not tested on a daily basis, in this experiment but all AEC levels had fully recovered after a further 3 days recovery.

This suggests that undamaged scallops can recover from repeated dredging. However, in order to recover their AEC levels there would be a load on the metabolism and each time the animals are stressed they may be vulnerable to predation. The behavioural studies (Section [3.1.3.1](#)) suggested that vulnerability persisted for up to at least 24 hours.

Investigation of recovery times for ecosystems have been carried out in a number of studies. Chemical effects were studied (Section [3.1.2](#)) in the Portuguese and Italian fisheries in which recovery took place rapidly (within hours). The only risk of chemical effects occurring would be under conditions of frequent effort when there would be changes in the nitrogen: phosphorus ratio in the surface layers. Both nutrients are released from the surface layers but phosphorus is adsorbed rapidly onto the surfaces of the clouds of suspended particles produced during dredging and so tends to return rapidly to the sediments (Section [3.1.2.1](#)).

Investigations of short-term benthic community effects (Section [3.1.5.4](#)) showed that recovery took place in meiofauna more rapidly (8 days) than in the macrofauna (2 months). However, the rate of recovery is likely to be affected by season (Jennings and Kaiser 1998; Kaiser et al. 1998; Veale et al. 2000b). When larval settlement is occurring, impact is likely to be greater, and when storms occur post-dredging changes are likely to last a shorter period of time.

Comparison between an area off the south coast of Portugal that had not been dredged for 4 years with one that had been dredged continuously, suggest that changes in the benthic communities (both in the macrofauna and mieofauna) occur when an area is left for this period of time (Section [3.1.5.5](#)). A higher proportion of faster growing species were and scavengers were found in the fished area, whilst slower growing species and deposit feeders were found in the un-fished area. A caveat has to be added to these results, which are discussed in Section [4.1.4](#). It seems possible that these sites were not wellmatched as a control and experimental comparison.

4.5.3 Spatial distribution of effort

Studies in Canada (Anon 1992; Burns et al. 1995), and in Northern Ireland (Magorrigan 1995) have shown that it is possible, using ground discrimination systems to target bivalve dredging effort on specific habitats thereby reducing undesirable environmental effects and improving efficiency. The video (Section [3.1.1.1](#)) observations indicated periods of reduced stability in the behaviour of the dredge, although it was not always possible to establish the cause.

Data from the instrumented scallop (measuring 3-dimensional acceleration) clamped to the dredge frame as in [Section 3.1.1.1](#), as distinct from in the dredge bag described in [Section 3.1.1.2](#), suggest that it would be possible to use this instrument, or an improved version of it, to investigate the stability of the dredges. Further work could investigate the relationship between the output from the seabed discrimination systems, damage rates and the stability of the dredges as inferred from accelerations on the dredge frames. This could lead to better targeting of effort onto substrates where the dredge was most stable and caused least damage to the scallops and other fauna. Concomitantly, it could also improve efficiency so management measures would have to be considered.

4.6 Consideration of management measures

Some ecological issues relevant to management are discussed in [Review Section 2.1](#) for *Pecten maximus* and [2.2](#) for *Aequipecten opercularis*. For these, and other bivalves better understanding of larval ecology and recruitment mechanisms would aid stock delineation and management.

This section discusses the possible implications of the results of these investigations for management of dredge fisheries, both in terms of Government regulations and the management of individual vessels. It is not intended to be prescriptive, only to illustrate some of the possible uses of these results in dredge fishery management.

4.6.1 Dredge design, efficiency and fishing effort

It is clear that dredge design affects both environmental effects and efficiency, and that it is possible to assess these effects using the techniques described in this study. Four dredge designs (the prototype hydraulic scallop dredge (for *Pecten maximus*), the skid dredge for queen scallops (*Aequipecten opercularis*) and two Portuguese clam dredges (for *Callista chione* and *Spisula solida*) were compared with traditional or classical designs in terms of environmental effects. Of these the skid dredges and the Portuguese clam dredges showed equal or improved efficiency and reduced environmental effects when compared with the traditional counterparts. Both the skid and Portuguese dredges are in commercial use by at least some vessels in the fisheries. The prototype hydraulic scallop dredge that had reduced environmental effects in terms of stress and damage on the scallops but reduced efficiency when compared with the classical dredge. Thus the impact per catch of scallops may not have been an improvement; with development this aspect could perhaps be improved.

Stock management considerations may require the avoidance of increased catch in fully exploited bivalve fisheries. The evolution of dredge design to minimise environmental impact must therefore take into account changes that improve efficiency. The management of the Portuguese clam fisheries, where reduced impact

and improved efficiency has been observed, have taken steps to avoid increased catches through the introduction of these selective, reduced impact dredges, by limiting catches to a daily quota. This way the impact per catch of clams is reduced and the overall impact of the fishery is reduced. Similar assessments should be made when considering means for improved targeting of fishing effort; there may be improved efficiency that should be taken into account when implementing such schemes.

There may also be a need to define the operational characteristics of a given dredge type in terms of its environmental effects. One example from the review is the Thames cockle (*Cerastoderma edule*) fishery ([Review Section 4.1.3](#)) which operates a system of type approval for the dredges used which stipulates a maximum rate of damage to the target species achieved by the dredge.

4.6.2 Selectivity

Species Selectivity

The ability to design fishing gear, which specifically targets a given species, is clearly a benefit for fisheries management since it enables the avoidance of by-catch. Two dredge features investigated in this study were found to influence species selectivity: the use of skids and tickler chains instead of teeth in queen scallop dredges ([Section 3.2.1.1](#)) and tooth length in Portuguese clam dredges ([Section 3.2.2.2](#)).

Queen scallop dredges are used in the Irish Sea (ICES division VIIa) where there is a closed season for landing king scallops during the summer months (June-October). Thus avoiding the use of toothed dredges at this time of year may be considered as a management option. There would also be a reduction in the by-catch of king scallops at other times of year if these dredges were used. Since smaller ring sizes are used on queen scallop dredges than on king scallop dredges this would result in a reduction in the stress levels in small king scallops. Another feature of the skid dredge catches was the reduced proportion of stones in the catch which might be advantageous in that wear and tear on the bellies would be reduced and there is a potential fuel saving from a reduction in the weight of the catch.

However, it must be recognised that the vessels using the toothed gear to target queen scallops are designed to use this type of dredge. The teeth are used as a pivot point against the gunwale of the vessel about which the dredges are tipped when the catch is discharged onto the deck. The spring teeth would also be expected to act as shock absorbers on areas of stony substrate; vessels using skid dredges and tickler chains overcome this by adding a weak link to the tickler chains. Although this study compared skid dredges with toothed dredges on several grounds the full spatial extent of the fishery was not covered and comparison was only made during the summer months.

Thus further work on the engineering and operational aspects as well as the investigation of the spatial and temporal viability would be required if the skid dredges were to be considered a viable option for the queen scallop fishery as a whole.

Scallop size selectivity

As discussed in [Section 4.4](#), size selectivity in dredges is a measure that would reduce stress in bivalves. The evidence from this study is that it would not reduce damage levels since most damage appears to occur at the point of contact with the teeth ([Section 3.1.1.2](#)). Investigations suggest that predation is more likely to occur on scallops which are damaged rather than those which were only stressed ([Section 3.1.4.2](#) and [3.1.5.1](#)). Thus the main reason for implementing such a measure would be to reduce the metabolic stress on the bivalves of being subjected to being dredged, sorted on deck and returned to the sea.

Currently there are no universal regulations governing tooth spacing and ring sizes used in dredge bellies in the UK king scallop fisheries. In most areas vessels use tooth spacing and ring sizes that suit local conditions but 9teeth/700mm bar and 75mm internal diameter ring size has been adopted in local regulations in some areas. In the French fisheries ([Review Section 4.5](#)) minimum ring sizes vary between fisheries, the smallest being 72mm and largest 92mm, with the majority at 85mm. These sizes are based on variations in the growth of scallops; the minimum legal landing size also varies between fisheries.

The results on tooth spacing suggest that selection at the teeth does not always occur and varies with tooth design, length and conditions ([Section 3.2.2.3](#) and [3.2.2.4](#)). Where teeth do function as selective devices 9 teeth/700mm bar and 8 teeth/700mm bar correspond to the optimum for 100 and 110mm MLLS ⁸ respectively (10 mm wide teeth).

Belly ring sizes vary, with a number of vessels using 75mm internal diameter rings but other vessels using larger 85mm rings. This study ([Section 3.2.2.4](#)) found that, when new, belly ring sizes of 85mm internal diameter would be optimal for areas where the MLLS is 100mm. Larger ring sizes (88 and 92mm) were investigated for the MLLS of 110mm but it was difficult to achieve full efficiency on the marketable scallops for these ring sizes. However, the characteristics of the ring selectivity changed with time and they started to lose small scallops including marketable scallops as the dredge bellies wore. In some circumstances the rings proved not to be robust enough for the fishery. It might be possible to develop sufficiently hard enough rigs to counter this tendency but the smaller rings would still have improved tensile strength by virtue of their shape if made out of the same material. For whole rings the tensile strength of all sizes tested was probably adequate when new but the effect of wear is likely to be the concentration of stress at the locations where wear occurs. Therefore the effect of wear on the ring strength has to be taken into account when deciding a suitable ring size for selection of scallops. The results in this study show that it may be possible to develop more selective, larger, harder rings but that development is required to fulfil this requirement.

Two lines of development would be pursued:

1. Improving the hardness of the rings and washers making up the dredge bellies. There are at least two dredge belly manufacturers who are developing specialised

⁸ Minimum Legal Landing Size

steels and hardening processes that could perhaps be used to achieve more selective gear.

2. Altering the geometry of the rings and washers so as to reduce the effects of wear on the strength of the belly rings.

This would require research and commercial testing of the belly designs. There is clearly scope for improving the selectivity of bellies by using bellies of greater than 75mm rings. However this depends on commercially viable ring sizes of greater than 75mm internal diameter and this would require further development.

Clam size selectivity

The study of Portuguese clam dredge size selectivity ([Section 3.2.2.2](#)) in the NW *Spisula solida* have resulted in legislation in this fishery raising the minimum mesh size from 25 to 40mm.

4.6.3 Seasonal and rotational closures

Many of the fisheries in this study have seasonal closures as management measures and the studies of fishing strategy suggest that vessels move from one area to another in response to bivalve abundance. The stress response of bivalves to fishing varies on a seasonal basis and the ecosystems recover on the scale of days, weeks or years after fishing.

This suggests that studies of stress responses, both at individual and ecosystem level might help to elucidate optimum harvesting strategies. For example, it might be considered important to conserve the growth of the stock and this would indicate closure during periods of most growth. Alternatively, it might be considered necessary to conserve the animals at times of year when the short-term stress response is greatest. However, an important difficulty would be obtaining adequate controls, particularly for the chronic and ecosystem responses ([Section 4.1.3-4](#)).

Many areas have short-term closures for public health reasons because of harmful algal blooms. Currently this is a major factor affecting short-term patterns of fishing that leads, in effect, to spatial and temporal closure, although these closures are not necessarily connected to the condition or stress response of the scallops.

Rotational closures designed to manage the effects of dredging on ecosystems could take two forms. For short-term effects the systems appear to recover in time periods of days or weeks. If left longer, for periods of the order of years, then these results indicate that succession into a different state would be expected with longer-lived species becoming more important. Closures of scallop fishing grounds for periods of several years can have dramatic effects on the exploitable biomass of scallops (Murawski et al. 2000; Brand and Beukers-Stewart 2002) and rotational closures are now being considered as management tools in several scallop fisheries.

5. CONCLUSIONS

- 1) **To review dredge fisheries and their operating conditions in all the nations considered.**

Bivalve biology, dredge technology, fisheries and their operating conditions were reviewed in all the participating nations. The majority of the resources exploited by dredges are managed on a regional or local basis. The knowledge of selectivity and environmental effects was reviewed to set to provide background information for the experimental work.

- 2) **To develop the techniques appropriate to each fishery to study the selectivity and environmental effects of dredging.**

Techniques were developed to study selectivity mechanisms and quantify the selectivity of spring-toothed scallop dredges and Portuguese clam dredges, and the physical, chemical, behavioural, physiological and ecological effects of dredge fisheries. The main difficulties were in finding suitable models for the physical effects of scallop dredging, and in finding suitable controls for the chronic physiological effects. Not all effects were studied in all fisheries but in most cases it was possible to develop techniques which were suitable for the conditions.

- 3) **To examine and quantify the role of dredge components in the selectivity of *Aequipecten opercularis* and *Pecten maximus* and to consider possible technical measures to improve selectivity.**

The selectivity mechanisms of both queen scallop (*Aequipecten opercularis*) and king or great scallop (*Pecten maximus*) dredges were studied. It was found that the capture of king scallop was dependent on the presence of teeth; with dredges using skids and tickler chains catches of queen scallop were maintained but king scallop were much reduced. This study showed that in certain seasons (June-Autumn) the behaviour of queen scallops means that they can be captured in a 'top net' set above the dredge.

The studies of spring tooth bar king scallop dredge selectivity showed that both tooth spacing and belly ring size had an effect on selectivity, but that selectivity at the rings was more reliable than at the teeth. Back mesh size and ring size had no effect on selectivity. Optimum ring size and tooth spacing were described but it was also found that selectivity varied with wear. Rings become more selective with wear and teeth become less selective. Ring sizes, which were optimal when new, began to lose marketable scallops as they wear. Improvements in belly ring materials and design are required to optimize selectivity. Models of belly rings and scallops and are available from work done in this study. There have been improvements in materials which should help in this process.

- 4) **To study the selectivity of dredges used on Portuguese clams.**

Portuguese clam dredge selectivity was studied in two fisheries. Selectivity of *Spisula solida* in the NW fishery and selectivity of *Callista chione*, and *Donax*

trunculus was studied in the South coast fishery for these species. It was found that size selectivity was related to mesh size and tooth spacing had no effect on selectivity. Selectivity curves could be obtained for all species studied, leading to a recommendation that the minimum mesh size in the NW clam fishery should be 40 mm, an increase of 15 mm on the previous minimum mesh size of 25 mm. This has been included in regulations for this fishery from 2000.

5) **To examine the mode of action of dredging on affected species and the seabed and to develop means to reduce physical impacts.**

The effects of scallop dredging on both the seabed and organisms were studied using instrumentation, both mounted on the dredge and contained within scallop analogues, damage indices, video and diver observations. Simulation studies were carried out in the laboratory and related to behaviour, physiological stress and predation. Hydraulic clam dredging (turbo soffianti in the N. Adriatic) was studied using instrumentation and modeling, and towed dredge clam dredging (Portugal) was studied using video observations.

The results of these investigations enabled an improved understanding of the effects of bivalve dredging on the environment and species encountered. It was found that for all the species encountered by scallop dredging, more individuals were left on the seabed than were retained by the gear. Other gears, particularly hydraulic dredges for clams (*Chamelea gallina*), were more efficient but in all cases there was some by-catch.

For the species studied (predominantly target species) there was an increased vulnerability to predation after encountering the gear. This increased vulnerability was related more to damage than physiological stress, although it is likely that stress does affect vulnerability to predation. For scallop dredges, most damage occurs at the point of first contact with the gear, that is with the teeth, and most of the physiological stress occurred within the dredge bag or during sorting on deck. For the Italian turbo soffianti dredge used for the clam *Chamelea gallina*, damage was related to two factors: high water pressure used in the dredge and mechanical sorting on the boat deck, of which the latter is the most important.

Recovery from physiological stress (in *Pecten maximus*) was within 2-6 hours (dependent on the intensity of dredging) but behaviour was affected for longer periods (<24 hours). There was no apparent cumulative effect on physiological stress; scallops that had been stressed one day could withstand further dredging the next day without increased physiological stress, but repeated stress would be expected to place a load on the animal's metabolism.

Instrumentation of scallop dredges successfully enabled the description of a number of parameters of physical impact, including tooth penetration, angle of the tooth and dredge to the seabed, and acceleration (both on the dredge and the scallop analogues). Video observations of the mode of capture of the scallops, and the behaviour of the dredge, suggest that an important parameter affecting damage by the teeth is the stability of the dredge; dredges were observed to bounce on the seabed apparently in response to changes operating conditions.

This suggests that impacts of dredges might be reduced if stability could be improved through damping mechanisms and/or targeting the dredge on substrates on which it was most stable.

In studies of the effects of hydraulic (turbo soffianti) dredging on *Chamela gallina* these could be related, both for damage and acute physiological stress on the operating parameters of the dredge (high or low pressure) pressure, plus mechanical sorting operations. There were also differences between locations, possibly related to the different seabed features. Physical impacts on the seabed were successfully monitored, and modeled in hydraulic (turbo soffianti dredging) for *Chamela gallina* using models developed for seabed engineering. It is possible that these methods could be further developed to optimize hydraulic dredge design both for clams, and in some circumstances scallops, to minimize disturbance.

Observations of the mode of action of traditional Portuguese clam dredges on the seabed showed that there was a 'sand buffer' effect that built up inside the dredge. This affected the distribution of the macro fauna after dredging which would influence the sampling strategies required to describe the effects of dredging. Selectivity occurred when the traditional dredge was being hauled. For dredges with an improved design of bag which took the form of a rigid cage through which the smaller clams were selected through the bars, selection occurred more rapidly and therefore with less stress on the by-catch.

6) To study the incidental mortality, biological stress, ecological, physical and chemical, environmental effects of dredging on a seasonal basis.

Physiological impacts of dredging were related to seasonal changes in the environment and condition of the target species encountering the dredges. For acute effects, where a suitable control could be obtained, it was possible to demonstrate changes in the effect of dredging on a seasonal basis. Indices of chronic effects were studied in relation to turbo soffianti dredging of *Chamelea gallina* and there are clear seasonal changes in these indices. However, without an adequate control population, it is difficult to separate the endogenous stress from that induced by dredging.

Impacts on ecological communities were studied by observations of predation and scavenging on organisms affected by dredging, and short-term impacts on the meio- and macrofauna. The studies of predation and scavenging enabled an understanding to be gained of the implications of damage and stress on the survival of bivalves encountering the dredges. Short-term recovery of dredged areas occurs within 2 weeks for meiofauna and longer for macrofauna (8 weeks).

7) To study the role of selectivity components and dredge design on the environmental effects of dredging.

Selectivity Components

The effect of selection of bivalves at the seabed is largely to reduce stress on undersized animals. In this study there was a reduction of approximately 75% in the level of stress in scallops (*Pecten maximus*) and a similar effect is described

in Portuguese clams (*Spisula solida*). Although selection at the teeth in scallop dredges would probably be optimal because of the very short contact time, this study shows that this mechanism of selection does not occur on a consistent basis and that selection at the rings is more consistent. To minimise stress in undersized scallops it would be best to optimise both tooth spacing and belly ring size. For Portuguese clam dredges selectivity was optimised by mesh size, but it was also found that modification of the dredge bag, making it into a rigid cage design produced, very rapid selection with minimum impact on the bivalves.

Dredge design

Four methods of lifting king scallops (*Pecten maximus*) off the seabed into the dredges were examined: the Magnus effect, foil effects, redesigned teeth and water jets (hydraulic system). Of these the foil and the hydraulic system appeared to have the best potential. Although the results suggested that there could be enough water speed under the foil to remove scallops from the seabed, there was not enough to lift them into the dredge bag. The hydraulic dredge designed for catching scallops was constructed and tested in comparison with the classical dredge. Damage rates and stress levels of the scallops were very low for this dredge but it was not as efficient as the traditional toothed dredge.

The changes in the Portuguese dredges (for *Spisula solida* and *Callista chione*) and revolved around redesign of the collecting bag. Comparison was made between traditional spring toothed dredges for queen scallop (*Aequipecten opercularis*) and skid dredges with tickler chains substituted for teeth. These design changes have been successful in commercial fisheries and are in use in some but not all boats in the fleets pursuing these species. They showed some reduction in their environmental effects per haul, compared with the traditional dredges and the Portuguese designs showed an increase in efficiency. The Portuguese fishery is managed on a daily quota basis; thus improved efficiency reduces the environmental effect per weight of clams landed.

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