

Impacts and efficiency of scallop dredging on different soft substrates

David R. Currie and Gregory D. Parry

Abstract: Impacts of scallop dredges and their efficiency were examined experimentally in three areas with different soft substrates in Port Phillip Bay, southeastern Australia. Physical and biological changes were measured on large (600 × 600 m) experimental plots that were dredged with an intensity and duration similar to normal fishing operations. Dredges were most efficient on soft, flat, muddy sediments (51–56% of commercial-sized scallops caught) and least efficient on firm, sandy sediments with more topographic variation (38–44%). Dredging flattened all plots, but changes to topography were most apparent on plots dominated initially by callianassid mounds. Dredges caught predominantly the scallop *Pecten fumatus*, and damage to bycatch species was slight, except for high mortality rates (>50%) of spider crabs and the probable mortality of many discarded ascidians. Changes to benthic community structure caused by scallop dredging were small compared with differences between study areas, and even marked reductions in the size and longevity of scallops over the last two decades may not be due entirely to dredging. The recent cancellation of all scallop dredging licences offers a unique opportunity to determine the contribution of scallop dredging to ecological changes in the bay over the past 30 years.

Résumé : Les impacts des dragues à pétoncles et leur efficacité ont été étudiés expérimentalement dans trois secteurs présentant des substrats mous différents dans la baie de Port Phillip, dans le sud-est de l'Australie. On a mesuré les changements physiques et biologiques dans de grandes parcelles expérimentales (600 × 600 m) qui ont été draguées avec une intensité et pour une durée semblables à celles des opérations normales de pêche. Les dragues étaient plus efficaces sur les sédiments mous, plats et vaseux (51 à 56% des pétoncles de taille commerciale capturés) et moins efficaces sur les sédiments fermes sableux présentant plus de variations topographiques (38–44%). Le dragage a aplati toutes les parcelles, mais les changements topographiques étaient plus apparents dans les parcelles dominées initialement par les monticules des callianassidés. Les dragues ont surtout capturé le pétoncle *Pecten fumatus*, et les dommages aux espèces des prises accessoires étaient peu importants, sauf dans le cas des araignées de mer (taux de mortalité élevés, >50%) et de nombreuses ascidies rejetées qui sont probablement mortes. Les changements dans la structure des communautés benthiques causés par le dragage des pétoncles étaient peu importants en comparaison des différences entre les secteurs de l'étude, et même les réductions marquées de la taille et de la longévité des pétoncles au cours des deux dernières décennies ne peuvent être entièrement attribuées au dragage. L'annulation récente de tous les permis de dragage des pétoncles offre une occasion unique pour déterminer la contribution de cette pêche aux changements écologiques survenus dans la baie au cours des 30 dernières années.

[Traduit par la Rédaction]

Introduction

Despite the long history of concern about the effects of dragging fishing gear along the seafloor (De Groot 1984), few impacts of dredging and trawling have been well documented. Previous studies of these fishing impacts indicate that physical effects depend on the softness of sediments and the weight of gear (Caddy 1973), while biological effects depend on the vulnerability of the communities present and, particularly, on the amount of epibiota in the area being fished (Bradstock and Gordon 1983; De Groot 1984; Peterson et al. 1987; Van Dolah et al. 1987; Sainsbury 1988; Collie et al. 1997). Increased environmental awareness has focused

attention on a range of potential impacts of shellfish dredging (reviewed by Messieh et al. 1991; Jones 1992; Currie and Parry 1996) but particularly on the need to more adequately assess biological impacts. These impacts are difficult to investigate because of the complexity of biological communities and our limited knowledge of their natural variability (Messieh et al. 1991). Biological impact studies are also usually costly (Gislason 1994), and because they require closure of commercial fishing grounds to create unfished areas, they are frequently opposed by the fishing industry. These difficulties have caused compromises in the design of most studies. Comparisons of areas with historically different intensities of fishing have been made (Graham 1955; BEON 1992; Collie et al. 1997), but in these studies the effects of fishing cannot be distinguished from spatial differences unrelated to fishing. In other studies, experimental fishing has involved disturbances of such small areas (McShane 1981; Eleftheriou and Robertson 1992; Thrush et al. 1995) that the relevance of these studies to large-scale commercial operations is uncertain.

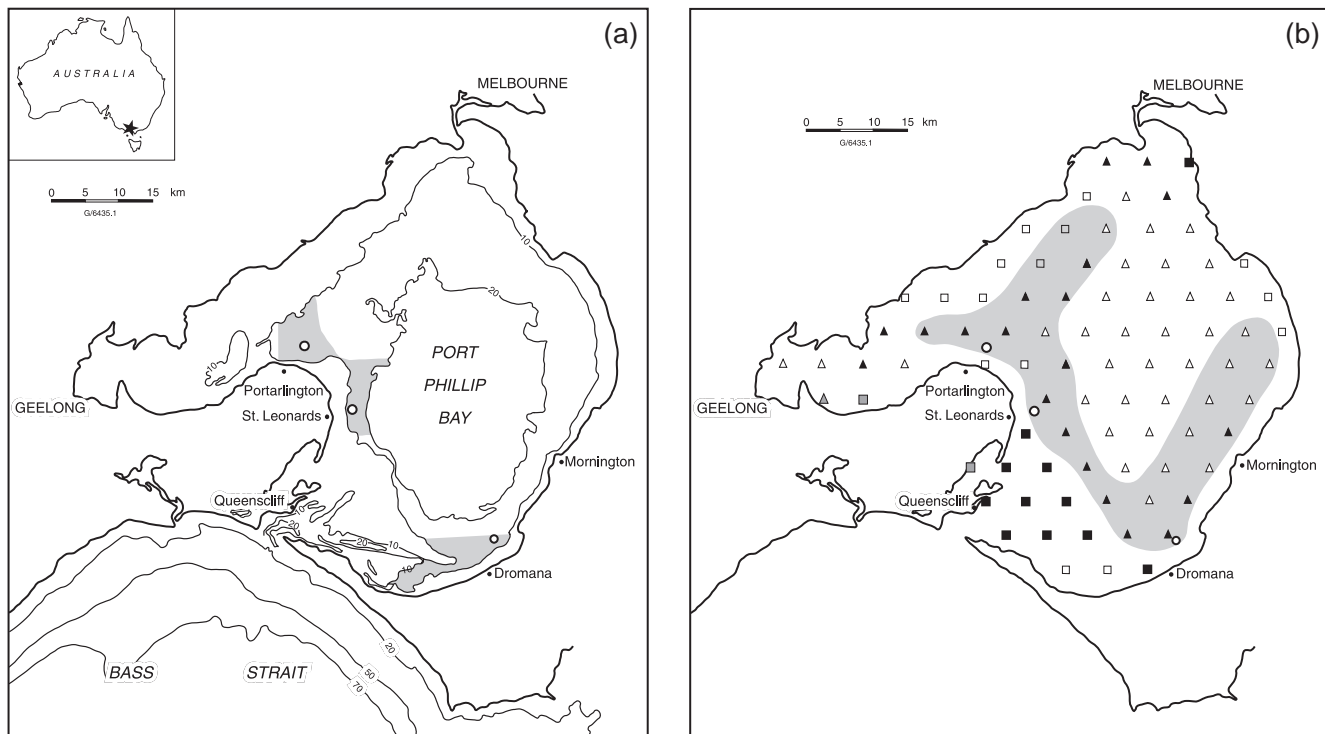
The Port Phillip Bay, southeastern Australia, scallop fish-

Received April 4, 1998. Accepted November 13, 1998.
J14554

D.R. Currie and G.D. Parry.¹ Marine and Freshwater Resources Institute, P.O. Box 114, Queenscliff, Victoria 3225, Australia.

¹Author to whom all correspondence should be addressed.
e-mail: greg.parry@nre.vic.gov.au

Fig. 1. (a) Location of the study area showing dredge plots (○) and areas closed to scallop dredging during 1991 (shaded). (b) Infaunal communities in Port Phillip Bay (from Currie and Parry 1999). ■, Low-diversity sands; □, high-diversity sands; ▲, intermediate sediments; △, central muds. Stippled symbols are seagrass communities. The location of dredge plots (○) at each study area and the main scallop grounds (shaded) are shown.



ery commenced in 1963 and catches and dredging intensity increased dramatically until the fishery collapsed in 1968 (Gwyther and McShane 1988). The intensity of dredging in the 1990's was lower than in the early years of the fishery, but 84 vessels were still licensed to dredge for scallops in Port Phillip Bay in 1996. All but the shallowest regions of the bay have been dredged, but with varying intensities. Scallop dredging occurred mostly in depths between 10 and 20 m (Fig. 1a) and was illegal in areas shallower than 10 m in the east and 5 m in the west of the bay after 1985. In 1997, after this study was completed, all scallop licences were cancelled because of public concern about the environmental effects of scallop dredging.

This study was designed to assess short-term impacts of dredging in different areas, rather than the cumulative ecological effects of 30 years of scallop dredging. Three large areas of Port Phillip Bay were closed to all scallop dredging throughout 1991 (Fig. 1a). In each closed area, a large experimental plot was dredged by commercial fishers with an intensity that was typical of either heavy or very heavy dredging, based on historical levels of dredging in the bay. Dredging-related changes to epifauna in all three areas and infauna in one area were examined using a BACI (before, after, control, impact) experimental design (Stewart-Oaten et al. 1986). The expense of this large-scale experiment meant that in the other two areas, dredging-related changes to infauna were assessed using only uncontrolled BAI comparisons. Related studies assessed the duration of impacts on infauna (Currie and Parry 1996), effects on fish communities (G.D. Parry and D.R. Currie, in preparation), and the extent of

sediment disturbance by dredging (Black and Parry 1994, 1999).

In addition to investigating environmental impacts, the efficiency of dredges used to collect scallops in Port Phillip Bay and the damage they caused to the catch were also measured in each study area. These studies were considered important because inefficient dredges may cause more damage, as they must be dragged over larger areas. Previous studies had suggested that the dredges used in Port Phillip Bay were very inefficient (12%) and could cause massive damage to nonharvested scallops (McLoughlin et al. 1991).

Methods

Study design

Throughout 1991, three large areas (30–40 km²) in Port Phillip Bay near Dromana, Portarlington, and St. Leonards were closed to all scallop dredging (Fig. 1a). Closed areas were policed effectively by fisheries enforcement officers, visited frequently in the course of this research, and strongly supported by the scallop industry. We are confident that no illegal scallop dredging occurred in any closed area. Areas were chosen to represent a wide range of sediment types in the 10–20 m depth zone where most commercial scallop fishing occurred (Fig. 1b). The sediments varied from medium sand (*sensu* Friedman and Sanders 1978, $\phi = 2$) at Dromana to fine sand ($\phi = 3-4$) at St. Leonards and coarse silt ($\phi = 5-6$, excluding the considerable amount of coarse shell material present) at Portarlington (Rosenbaum et al. 1992; Black and Parry 1994; Greilach et al. 1995).

Two 600 × 600 m experimental plots were located in each closed area. At St. Leonards, plots were adjacent in a depth of 12–

Table 1. Mean catch rates of scallops and bycatch species (mean no./drag) and number of drags during each days dredging on experimental plots at Dromana, Portarlington, and St. Leonards (all dredging was undertaken during 1991).

Species	Dromana		Portarlington				St. Leonards		
	26 Aug.	28 Aug.	20 Nov.	21 Nov.	10 Dec.	11 Dec.	16 July	17 July	18 July
<i>Pecten fumatus</i> ≥70 mm	264	186	760	466	286	207	93	72	59
<i>P. fumatus</i> <70 mm	397	328	1206	908	627	653	92	82	30
<i>P. fumatus</i> damaged	5.0	5.1	2.5	5.3	1.0	1.9	0.4	0.7	0.3
<i>Ostrea angasi</i> clumps	208	204	1.0	0.8	0.9	0.7	10.3	9.8	11.8
<i>Pyura stolonifera</i>	2.1	2.5	13.2	13.0	11.2	15.4	1.3	1.3	1.7
<i>Herdmania momus</i>	2.3	2.2	—	—	—	—	0.4	0.5	0.6
<i>Cnemidocarpa etheridgii</i>	0.4	0.6	3.0	3.3	2.3	3.7	0.4	0.7	0.3
<i>Leptomithrax gaimardii</i>	2.9	4.4	4.8	1.4	1.9	1.0	0.5	0.4	0.3
<i>Stichopus mollis</i>	0.01	—	3.3	2.4	1.1	1.3	0.06	0.08	0.01
<i>Tosia magnifica</i>	0.05	0.02	0.5	0.4	0.3	0.4	0.02	0.03	0.02
<i>Costinasterias calamari</i>	—	—	0.01	0.02	0.01	0.01	—	0.01	—
<i>Octopus australis</i> and <i>O. pallidus</i>	0.08	0.06	0.2	0.4	0.2	0.1	0.01	0.02	0.01
<i>Austrosipho grandis</i>	0.06	0.05	—	—	—	—	—	—	—
Hermit crabs (mostly <i>Trizopagurus strigimanus</i>)	0.08	0.06	—	—	—	—	0.02	0.03	0.04
Total nonbivalve epibiota (TNBE)	7.98	9.89	25.0	20.9	17.0	21.9	2.71	3.07	2.98
TNBE/ <i>P. fumatus</i> ≥70 mm (%)	3.0	5.3	3.3	4.5	5.9	10.6	2.9	4.3	5.0
No. of drags	209	173	160	195	178	183	147	191	121

15 m, while at Dromana and Portarlington, they were 600 m apart in 14–16 and 13–15 m of water, respectively. In each closed area, one of the experimental plots was left undredged (“control” plot), whereas the remaining plot was dredged (“dredge” plot). The large size of the plots enabled changes to fish populations within the plots to be monitored using a large trawl net (G.D. Parry and D.R. Currie, unpublished). However, the large size of the plots made replication of control plots (Hulbert 1987; Underwood 1991) prohibitively expensive in all experiments and limited the use of control plots to studies of changes in the abundance of large epifauna and of infauna within the St. Leonards area (Currie and Parry 1996).

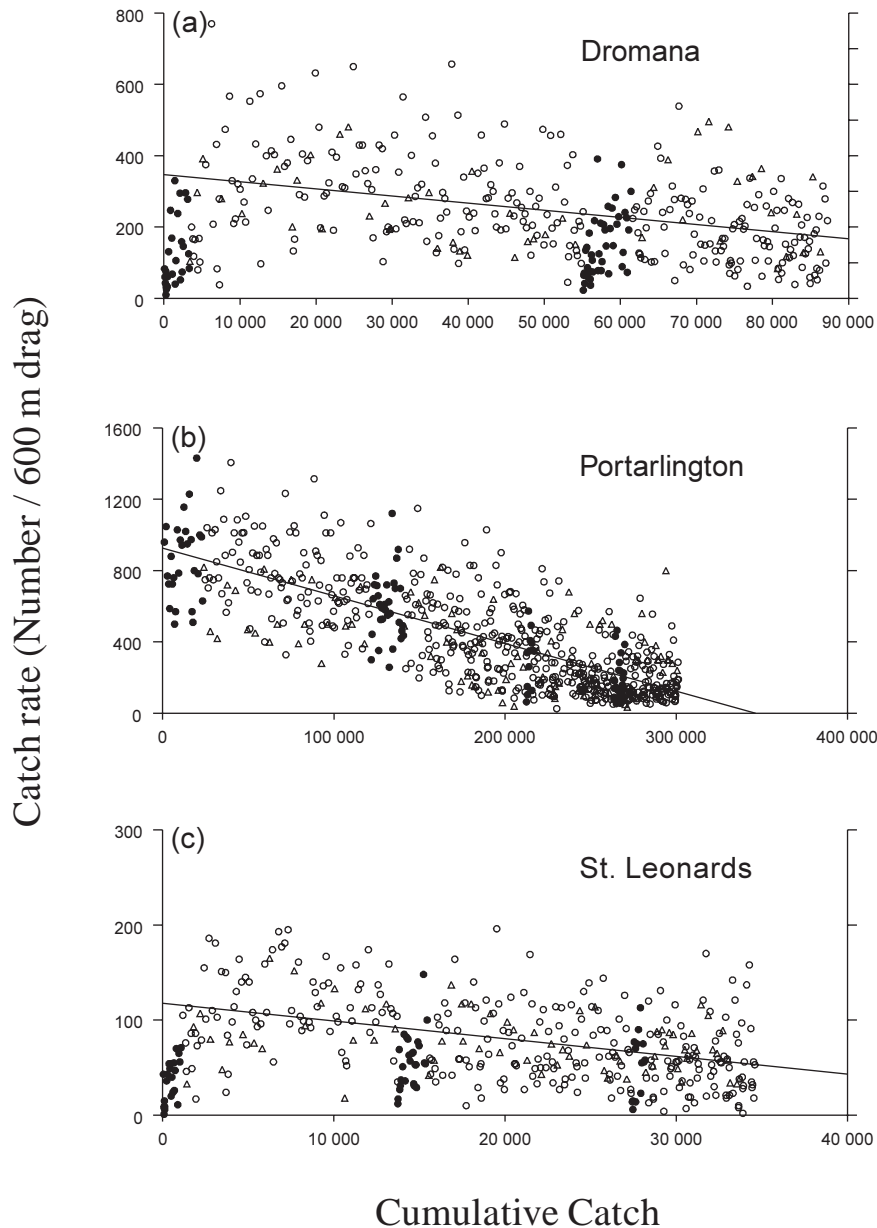
Dredge plots were dredged to closely duplicate normal fishing practices, as described in Currie and Parry (1996). Plots were dredged over 2–4 days (Table 1) by a fleet of five to seven commercial scallop vessels using 3-m-wide “Peninsula” dredges (see diagram in Hughes 1973) fitted with scraper or cutter bars that did not extend below the level of the skids. Dredging was conducted for a maximum of 3 h/day and coincided with periods in which tidal currents carried dredging-related sediment away from the control plots. Sediment traps (Black and Parry 1994) placed down-current and in the centre and on the southern edge of the control plot at St. Leonards confirmed that negligible sedimentation occurred on the control plot, while sedimentation was greatly elevated down-current. At St. Leonards and Dromana, dredge plots were dredged until they had, on average, been passed over twice by a scallop dredge (2× intensity). The dredge plot at Portarlington was dredged twice with an interval of 3 weeks between dredgings of equal (2×) intensity. A 2× intensity of dredging was consistent with a moderately high fishing intensity, based on historical levels in Port Phillip Bay, and a 4× intensity corresponded to very intense dredging (Currie and Parry 1996). A lower level of dredging would also have made detection of impacts more difficult, as much of the dredged plot would have remained undredged. Even at the 2× dredging intensity used in this experiment, an estimated 13% of the plot remained undredged (Currie and Parry 1996).

On each 600-m drag, observers on each vessel recorded the catch of scallops and all bycatch species large enough to be retained by the 70 × 45 mm dredge mesh. Scallops >70 mm were harvested from the catch, and bycatch was discarded on the plot. The numbers of commercial-sized scallops (>70 mm), small scallops (<70 mm), and broken scallops were counted separately. When large numbers of small scallops were caught, only a proportion (0.25 or 0.5) were counted, and when large numbers of oysters were caught the number of fish bins that they filled was recorded. The number of oysters per fish bin was determined subsequently. For oysters and ascidians the number of clumps of live organisms was counted rather than the number of individuals. At Portarlington the number of damaged ascidians and “severely damaged” spider crabs was recorded. Severely damaged spider crabs had their carapace damaged or were missing more than two legs and were considered unlikely to survive. Kaiser and Spencer (1995) found that all crabs taken in trawl nets with cracked carapaces or missing >50% of their limbs died within 48 h.

To assess visually apparent changes caused by dredging, diver-operated video recordings were taken of the seabed on both plots at St. Leonards 3 months before dredging, 8 days after dredging, 6 months after dredging, and 11 months after dredging. At Dromana, video recordings were taken on the dredge plot 1 day before dredging and 5 days after dredging. At Portarlington, poor visibility made it impossible to take video recordings before dredging, but all plots were recorded 43 days, 50 days, 6 months, and 7 months after dredging.

The depth of bed sediment disturbed by dredges was determined using 32–38 sets of colour-coded steel rings that were inserted 2, 4, 6, and 8 cm into the sediment within each dredge plot before dredging commenced. Rings were also placed on the sediment surface, except at St. Leonards. Rings in each set were located within a circle ~50 cm in diameter and sets were placed ~4 m apart (i.e., slightly greater than the width of a dredge) on a line at ~45° to the direction of dredging. During the experimental dredging, observers on each vessel identified each ring and were therefore able to de-

Fig. 2. Leslie–DeLury plots of catch rate against cumulative catch at Dromana, Portarlinton, and St. Leonards. The regression lines shown do not include data collected during the first 30 min of dredging each day (●). ○, Catch rates of all vessels other than those (△) for which regressions had a significantly different intercept on the x -axis.



termine the depth from which it was recovered (Black and Parry 1994).

Dredge efficiency

A Leslie–DeLury depletion experiment (DeLury 1947; Joll and Penn 1990) was used on each plot to estimate the catch efficiency of dredges for scallops >70 mm. In this method, catch efficiency is estimated from the rate of decline of catch rate compared with the cumulative catch of all vessels (Fig. 2). If the capture efficiency of a fishing gear remains constant, then a fixed fraction of a closed population will be removed with each use of the gear. For example, if a pond is fished four times and the resulting catches are 135, 90, 60, and 40, then the efficiency of the fishing gear will be $1/3$ and the initial population size 405. As a scallop dredge only collects scallops in a path the width of the dredge (~ 3 m) and the plot was 600 m wide, the efficiency of the dredge will be 200 (i.e., $600/3$) times the rate of the decline of the population measured in

Fig. 2. As the dredge efficiency was constant throughout the dredging on all plots, except during the first 30 min of each day's dredging at Dromana and St. Leonards, catch rates during the first 30 min of each day's dredging were excluded from all estimates of catch efficiency.

The total scallop population on each plot was estimated from the intercept of each regression in Fig. 2 with the x -axis. At this intercept the catch rate of all vessels is expected to be zero, so for each vessel the regression of catch rate against cumulative catch should pass through this point. For each vessel the significance of the reduction in sums of squares due to forcing the regression through this point was tested (Steel and Torrie 1960). On each study plot, there was one vessel (shown in Fig. 2) where fitting the regression through the estimated origin significantly increased the sum of squares. Each of these vessels had unexpectedly high catch rates towards the end of each trial and were probably targeting unfished areas near the plot boundaries. Consequently, catch rates for these

three vessels were then excluded from a recalculated regression of catch rate versus cumulative catch from which another estimate of dredge efficiency was obtained. Differences between the slopes of these recalculated regressions at Dromana, Portarlington, and St. Leonards were tested using analysis of covariance (Snedecor and Cochran 1967).

Mortality of ascidians

To estimate the mortality of ascidians discarded from a scallop dredge, 60 clumps of *Pyura stolonifera* and 60 individuals of *Herdmania momus* were collected from a scallop dredge on 28 October 1992, maintained in flowing seawater for 6 h, and then placed by divers in three marked rows on the seabed near the Dromana plot at a depth of 13–14 m. These ascidians were observed by divers after 7 days and again after 5 weeks and the number of live, dead, and anchored ascidians recorded. Anchored ascidians were those that could not be moved when touched gently by a diver.

Changes in the abundance of large epifauna

Large epifauna was sampled with a 1-m-wide Ockelmann sled towed into the prevailing current for 5 min at 2.2–3.1 kn, except during the preimpact sampling at Portarlington when the sled was inadvertently towed at 3.6–3.8 kn. The duration and the location of the start and finish of each drag were recorded and used to calculate the towing speed and distance of each tow.

In each area, changes in epifauna were assessed using a BACI design where both the control and dredge plots were sampled before and after the experimental dredging. At St. Leonards, epifauna was sampled on three dates before (8 April, 16 May, and 5 July 1991) and on five dates after the experimental dredging (25 July, 9 August, 31 October, and 18 December 1991 and 23 March 1992); on each sampling date, three sled hauls were located randomly on each plot. At Dromana, epifauna was sampled 1 week before (18 August 1991) and twice after the experimental dredging (2 September 1991 and 1 April 1992); nine sled hauls were taken at random within each plot, except on 1 April 1992 when only six hauls per plot were taken. At Portarlington, epifauna was sampled 1 week before the first dredging (14 November 1991), once in the 3 weeks between dredgings (26 November 1991), and twice in the 4 months after the second dredging (12 December 1991 and 2 April 1992); on each sampling date, six sled hauls were located randomly on each plot.

The numbers of large scallops (>70 mm), small scallops (<70 mm), and heart urchins (*Echinocardium cordatum*) caught in each 5-min sled haul were used to estimate the catch of each species per 500 m². Differences between the number of individuals per 500 m² found on each plot before and after dredging were examined using analysis of variance (ANOVA), as outlined in Underwood (1991, table 1c). In these nested ANOVAs, plot × time interactions were tested against the mean square for the plot × date (time) term. Homogeneity of variance was examined using Cochran's test and heterogeneity was removed by a log₁₀(N + 1) transformation.

Infauna

The distribution and abundance of infauna on each plot were determined from replicate 0.1-m² Smith–McIntyre grab samples. All animals retained on a 1.0-mm sieve were sorted to the lowest practical taxonomic level (generally species) under a dissecting microscope before being counted.

At St. Leonards, 15 grabs were taken on both the control and dredge plot on three occasions in the 3 months before and on three occasions in the 3.5 months after the experimental dredging, as described in Currie and Parry (1996). At Dromana, 15 grabs were taken randomly on the dredge plot 8 days before and 2 days after the experimental dredging. At Portarlington, 20 grabs were taken

randomly 10 days before the first dredging and 1 day after the second dredging.

Differences in community structure between areas and the effect of dredging on communities in each area were examined using Bray–Curtis (B-C) dissimilarities (Bray and Curtis 1957) and multi-dimensional scaling (MDS). The PATN computer package (Belbin 1990) was used for the nonmetric MDS ordinations in this study.

There was a pronounced gradient in community structure across both the control and dredge plots at St. Leonards (Currie and Parry 1996). Consequently the eastern third and the western third of these plots were considered as separate “sites”. On each sampling date, only the five most easterly and five most westerly grabs taken on the St. Leonards plots were analysed. On each sampling date, plots at Dromana and Portarlington were also subdivided into three and four sites, respectively. The number of individuals of each species on each site was calculated from the total number of individuals found in the five adjacent grabs that formed the site. Before calculating B-C dissimilarities, double square root transformations were applied to the number of individuals of each species. B-C dissimilarity measures calculated for all site × date combinations resulted in a triangular matrix of dissimilarities that was used to map site × date interrelationships in two dimensions.

At Dromana and Portarlington, changes in density on the dredge plots following dredging were examined for the 10 most abundant species using one-way ANOVA's. A log₁₀(N + 1) transformation removed significant heterogeneity. At Portarlington, many species increased in abundance in the 3-week period between the two experimental dredgings, probably due to recruitment of juveniles. As recruitment obscured dredging impacts on nonrecruits, the size of all animals collected on these plots was measured and size-frequency distributions constructed for periods before and after dredging. Total length was measured for all species except for worms, where head width was used. Measurements accurate to 0.1 mm were made using a camera lucida image projected onto a digitising pad. For each species the largest size class potentially increased by recruitment was estimated and only animals larger than this size (Fig. 3) were included in the ANOVA's. Whilst recruitment was identified in six species, it could only be recognised in species abundant enough for a recruitment mode to be distinguished from size-frequency distributions.

Statistical significance

Results were considered significant if $p < 0.10$. This convention was adopted to reduce the likelihood of making a type II error. Failure to detect a real change due to dredging (type II error) was considered as serious a problem as identifying a change due to dredging when none had occurred (type I error) (Cohen 1988; Peterman 1990; Peterson 1993).

Results

Physical changes to seafloor

At St. Leonards the bedforms were dominated by volcano-like mounds and depressions formed by burrowing callianassid ghost shrimps. Mounds were up to 10 cm high and had an average density of 1.2/m² (Currie and Parry 1996), and depressions frequently contained large scallops. Eight days after dredging, ~90% of the seafloor within the dredged plot was very flat, while the remainder appeared undredged. Dredges had a graderlike impact; callianassid mounds were removed and depressions filled, but most callianassids appeared to have survived (Currie and Parry 1996; cf. Peterson 1977) and mounds were being rebuilt. Parallel tracks from the dredge skids (up to 25 mm deep) were obvious. Small scallops remained plentiful and divers estimated that <5% of

Fig. 3. Size–frequency distributions for six species that recruited between samplings at Portarlinton before (open bars) and after dredging (solid bars). The arrowhead indicates the size below which individuals were excluded from ANOVA’s for each species.

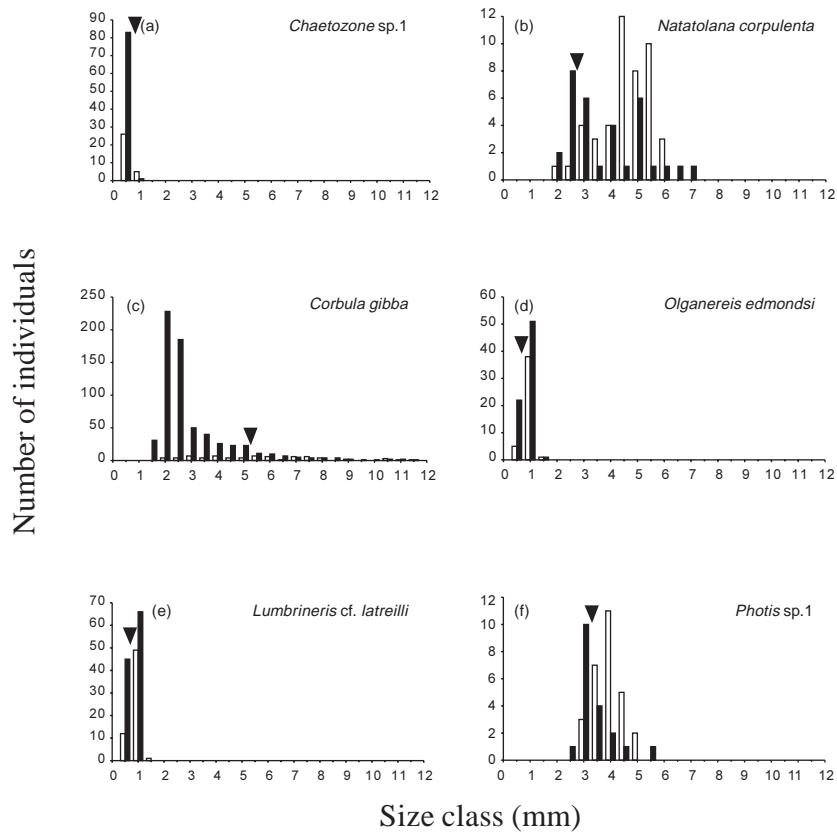


Table 2. Total number of steel rings deployed and recovered from each depth and the number recovered during each day’s dredging of experimental plots.

	Depth inserted (cm)														
	Dromana					Portarlinton					St. Leonards				
	0	2	4	6	8	0	2	4	6	8	0	2	4	6	8
Total no. of rings deployed	33	33	33	33	33	38	38	38	38	38	0	32	32	32	32
No. of rings recovered on day 1	7	1	1	0	0	0	0	0	0	0	—	0	0	0	0
No. of rings recovered on day 2	1	0	0	0	0	1	0	0	0	0	—	1	0	1	0
No. of rings recovered on day 3	—	—	—	—	—	0	0	0	0	0	—	5	2	0	0
No. of rings recovered on day 4	—	—	—	—	—	1	1	1	0	0	—	—	—	—	—
Total no. of rings recovered	8	1	1	0	0	2	1	1	0	0	—	6	2	1	0

the remaining scallops were dead. One month after dredging, the seabed remained flat and dredge tracks could still be distinguished. Six months after dredging, tracks could no longer be seen, and while mounds and depressions had re-established, the regions between them remained unusually flat. After 11 months the appearance of the dredged plot could not be distinguished from that of the undredged plot.

At Dromana the seabed consisted of sandy sediments containing some coarse shell. Small parallel sand ripples were evident in parts of the plot, but throughout the plots the bottom relief was mostly the result of callianassid mounds, but these were smaller than and less abundant than in the plots off St. Leonards. Five days after the experimental dredging, and immediately following a period of storm activity, the

seafloor was dominated by small parallel sand ripples. Storms had apparently reworked the sediments into sand ripples, obliterating the predredging bedforms and any signs of tracks from dredge skids. Divers estimated that 20% of the remaining scallops were dead.

At Portarlinton the dredge plot was initially very flat, except for small numbers of callianassid mounds. Mounds were flattened by dredging, and although they became reestablished, they were still smaller than in adjacent undredged areas 7 months later.

The recovery rate of rings was low on all areas (Table 2), but particularly at Portarlinton. The maximum depth of disturbance was 4 cm at Dromana and Portarlinton and 6 cm at St. Leonards (Black and Parry 1994).

Table 3. Regression equations of catch rate against cumulative catch from Leslie–DeLury depletion experiments (Fig. 2) and the initial scallop density and dredge efficiency a , derived from these equations, and dredge efficiency b , estimated after excluding vessels where the intercept on the x -axis for that vessel differed significantly from the overall intercept.

Plot	Gradient	\pm SE	Intercept	Initial scallop density (no./m ²)	Mean dredge width (m)	Dredge efficiency a	\pm SE	Dredge efficiency b
Dromana	-0.00200	0.00026	347.2	0.48	3.17	37.9	4.9	43.9
Portarlington	-0.00268	0.00010	927.1	0.96	3.14	51.2	1.9	55.6
St. Leonards	-0.00187	0.00020	117.7	0.18	3.12	45.0	4.8	49.5

Table 4. Percentage of severely damaged spider crabs (*Leptomithrax gaimardii*) and damaged ascidians (*Pyura stolonifera* and *Cnemidocarpa etheridgii*) found in scallop dredges during each day of experimental dredging at Portarlington.

Date	<i>Leptomithrax gaimardii</i>	<i>Pyura stolonifera</i>	<i>Cnemidocarpa etheridgii</i>
20 Nov. 1991	36	1.4	2.7
21 Nov. 1991	54	0.6	0.5
10 Dec. 1991	22	0.6	0.2
11 Dec. 1991	55	0.9	0.3

Dredge efficiency

Dredge efficiencies differed between experimental plots. These differences were of marginal significance: $p = 0.07$ for analysis of heterogeneity of slopes of regression lines in Fig. 2. Dredges were most efficient on the soft, muddy, and flat Portarlington plot (51–56%, Table 3) and least efficient on the firm, sandy, and moderately flat Dromana plot (38–44%). However, dredge efficiencies during the first 30 min of dredging differed markedly between areas (Fig. 2). At St. Leonards and Dromana the efficiency of dredging increased during the first 30 min of fishing on each day, and this was particularly marked on the first day (Fig. 2). In contrast, at Portarlington, dredges always operated at their maximum efficiency from the commencement of each day's fishing.

Effect on larger epifauna

Dredge catches

There were clear differences in the bycatch between areas (Table 1). Oysters (*Ostrea angasi*), sea squirts (*Herdmania momus*), giant spider crabs (*Leptomithrax gaimardi*), whelks (*Austrosipho grandis*), and hermit crabs were collected most frequently in the sandy sediments at Dromana. In the muddy sediments at Portarlington, sea squirts (*Pyura stolonifera* and *Cnemidocarpa etheridgii*), sea cucumbers (*Stichopus mollis*), biscuit stars (*Tosia magnifica*), and octopus (*Octopus australis* and *Octopus pallidus*) were the dominant species of bycatch. At St. Leonards, bycatch included species taken at both Dromana and Portarlington.

Dredges caught mostly scallops. The total number of non-bivalve epibiota (TNBE) caught was typically <5% of the number of commercial-sized scallops caught (Table 1). The highest percentage of TNBE/scallops >70 mm was 10.6% at Portarlington but occurred only after 4 days of dredging, when most of the scallops had been caught. Catches of scallops <70 mm were high at all areas, and many oysters were caught at Dromana (Table 1).

Damage and mortality caused by dredging

Less than 1% of scallops taken in dredges were damaged (Table 1). The percentage of damaged scallops was higher at Dromana (0.75–1.0%) than at Portarlington (0.11–0.38%) and St. Leonards (0.22–0.45%), but it is likely that some damaged scallops were overlooked by observers.

Damaged oysters were not counted, but they were observed only rarely and their shells are more robust than those of scallops. At Portarlington, slightly more than half the spider crabs (up to 55%, Table 4) were probably killed by dredges, and a similar percentage of the many spider crabs caught at Dromana (Table 1) was also damaged (personal observations). The percentage of ascidians damaged at Portarlington (0.2–2.7%, Table 4) was low, and while the proportion damaged was probably higher in firmer sediments at Dromana, the number of conspicuously damaged ascidians was always low.

Although most oysters and ascidians were not physically damaged by dredges, their survival after they are discarded also depends on their orientation on the seafloor and their ability to reanchor themselves in the sediment. Trials to measure survival of ascidians caught by scallop dredges and returned to the sea by divers showed nearly 100% survival of *Herdmania momus* and *Pyura stolonifera* in the week following their return, but storms during the following 5 weeks moved most of the experimental animals out of the study area (Table 5) and they could not be relocated. Only 12% of *Pyura stolonifera* and 2% of *Herdmania momus* successfully reanchored on our experimental area after 6 weeks. *Herdmania momus* is more globular in shape than *Pyura stolonifera* and seems less able to reanchor itself. Fewer than 7% of the experimental ascidians were known to have died, and the proportion that reanchored elsewhere was unknown, but most of them probably died.

Changes to epifauna

The abundance of small (<70 mm) scallops decreased significantly ($p < 0.10$) by 57% in the 7 months following dredging at Dromana and by 26% in the 5 months following the first dredging at Portarlington, but increased significantly by 19% in the 8 months following dredging at St. Leonards (Table 6). At Dromana and Portarlington, both large scallops (>70 mm) and heart urchins decreased in abundance, but these changes were not significant, while at St. Leonards, their abundance increased, but not significantly (Table 6).

Infauna

Differences between infaunal communities

An MDS plot (Fig. 4) indicated that all sites sampled at Dromana and Portarlington and the eastern and western sides

Table 5. Percentage of live, anchored, and dead ascidians on an experimental plot 1 and 5 weeks after 60 clumps of two species of ascidians were caught in a scallop dredge and placed on the seabed.

No. of weeks after dredging	<i>Pyura stolonifera</i>			<i>Herdmania momus</i>		
	Live	Anchored	Dead	Live	Anchored	Dead
1	100.0	38.3	0	66.7	20.0	3.3
5	41.7	11.7	6.7	1.7	1.7	0

Table 6. Mean densities on control plots, mean difference between dredge and control plots, and percentage change following dredging in the abundance of common fauna collected by Ockelmann sled at Dromana, Portarlinton, and St. Leonards.

Site	Taxa	Mean densities (SE) on control plot/500 m ²		Mean % difference dredge vs. control		
		Before dredging	After dredging	Before dredging	After dredging	% change
Dromana	Scallops ≥70 mm	4.1 (0.7)	16.0 (3.6)	-13.4	-79.4	-66.0
	Scallops <70 mm	63.3 (7.9)	125.1 (29.4)	+13.8	-43.1	-56.9*
	Heart urchins	2.0 (0.8)	32.4 (9.9)	+338.8	+45.2	-293.6
Portarlinton	Scallops ≥70 mm	21.9 (3.6)	33.3 (5.0)	+33.2	-28.6	-61.8
	Scallops <70 mm	81.8 (19.1)	179.8 (29.8)	+3.9	-22.3	-26.2*
	Heart urchins	0.6 (0.3)	7.2 (2.5)	-1.8	-54.6	-52.8
St. Leonards	Scallops ≥70 mm	2.4 (1.0)	4.6 (1.8)	-76.7	-49.4	+27.3
	Scallops <70 mm	50.6 (12.1)	57.8 (9.4)	-23.6	-5.1	+18.5*
	Heart urchins	36.9 (13.2)	46.8 (15.6)	+18.2	+37.1	+18.9

Note: * $p < 0.1$.

of the St. Leonards plot formed separate clusters. The stress value of 0.127 indicates that the pattern is unlikely to mislead (Clarke 1993). The most similar plots were west St. Leonards and Dromana, both of which occurred in sandy sediments (1–24% sediment <63 µm) and both of which had a high diversity of species. The east St. Leonards plot had a lower diversity of species and lower numbers of individuals and was muddier than the west (Currie and Parry 1996). The Portarlinton plot was the muddiest (15–55% sediment <63 µm) and had the lowest diversity of species and the lowest number of individuals. Sites sampled before and after dredging within the same plot did not form separate clusters (Fig. 4), indicating that changes to community structure caused by dredging were smaller than differences between plots.

Changes to infauna

Three of the 10 most abundant species at Dromana (*Bullowanthura pambula*, *Chaetozone* sp.1, and *Nephtys inornata*) decreased significantly in abundance by 20–40% (Table 7). The abundance of *Bullowanthura pambula* decreased at all areas following dredging, although the significant “decrease” at St. Leonards appeared to be caused by aggregation of this motile paranthurid in undredged regions immediately following dredging (Currie and Parry 1996). *Chaetozone* sp.1 was rare at Portarlinton but appeared to decrease by 30–40% at both Dromana and St. Leonards (Table 7). The decrease in abundance of *Nephtys inornata* may be greater at Dromana than at St. Leonards (Table 7).

Two of the 10 most abundant species at Portarlinton (*Theora lubrica* and *Natolana corpulenta*) decreased significantly in numerical abundance by 30–70%, although these changes should be interpreted cautiously, as densities of all species at Portarlinton were low (Table 7). Changes to the

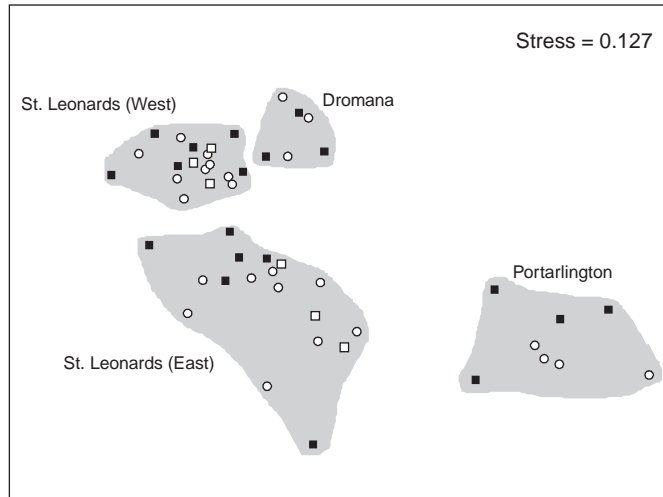
abundance of *Theora lubrica* are not considered reliable, as this species has a very patchy distribution (Currie and Parry 1996). The decrease in the abundance of *Natolana corpulenta* is similar to its sharp decrease immediately following dredging on the St. Leonards plot but contrasts with its apparent increase on the St. Leonards plot in the months following dredging (Currie and Parry 1996).

Discussion

Physical differences between the experimental plots at Dromana, St. Leonards, and Portarlinton affect the impact of dredges and their efficiency. Dredging removed most topographic variation from all plots and caused the most noticeable changes at St. Leonards where callianassid mounds were a prominent predredging feature. Recovery of rings and measurements of sediment concentrations behind dredges (Black and Parry 1994) suggest that dredges usually disturb less than the top 1–2 cm of sediment, although they can clearly disturb up to at least 6 cm. Plume modelling suggests that on average, ~0.5 cm of sediment was disturbed in all areas (Black and Parry 1999).

Our estimates of the efficiency of Peninsula dredges in Port Phillip Bay (38–56%, Table 3) were similar to other estimates obtained in Port Phillip Bay (42–66%, McShane and Gwyther 1984; 37–50%, Gwyther and Burgess 1986) but were markedly higher than the 12% measured experimentally in Bass Strait (McLoughlin et al. 1991). We observed the lowest efficiency on the hardest substrate (Dromana, Table 3) most like the sediments in Bass Strait, but the very low efficiency found by McLoughlin et al. (1991) was probably due to the small number of drags (~20) undertaken in their study. We found that dredge efficiencies were markedly lower during the first 30 min (~20 drags) of dredging on

Fig. 4. MDS ordination of benthic community data based on five pooled grab samples taken from study plots on sampling dates before and after dredging. ○, Dredge plot before dredging and (St. Leonards only) control plot before and after dredging; ■, dredge plot after dredging; □, dredge plot before dredging (St. Leonards only).



both plots with sandy substrates. Higher vessel speeds used in our study (5.5–6.5 versus 2–3 kn) and the use of toothbars only in Bass Strait may also have contributed to differences between the studies.

Increases in dredge efficiency observed during the first 30 min of dredging on two of our three experimental plots appear similar to increases noticed by commercial scallop fishers as they “worked up” scallops. Differences in dredge efficiency between areas and the observation that sediment plumes usually cause scallops to swim suggest an explanation for this phenomenon. On soft flat substrates (e.g., Portarlinton) all scallops are similarly likely to be caught by a passing dredge. In contrast, on substrates with greater topographic relief (e.g., Dromana and particularly St. Leonards), most large scallops occur in depressions where they may be passed over by a dredge. However, the sediment plume thrown up by a passing dredge stimulates many scallops to swim and they become more randomly distributed and more vulnerable to dredges that pass subsequently. As dredging continues the substrate is progressively flattened, so fewer scallops remain in depressions deep enough to be passed over by a dredge. The increasing flatness of the plots at Dromana and St. Leonards on the second and third days of dredging probably caused the progressively higher efficiencies observed at the commencement of dredging on successive days (Fig. 2). Levelling of the previously uneven substrate on the St. Leonards plot was also consistent with postdredging increases in the efficiency of the Ockelmann sled (Table 6) and a fish trawl net (G.D. Parry and D.R. Currie, in preparation) that occurred only on this plot. The increase in dredge efficiency from a more rapid increase in turbidity and a more rapid levelling of the substrate when several vessels operate in the same area probably explains why scallop vessels usually operate in groups.

The proportion of scallops damaged by dredging was consistently low but was highest at Dromana (Table 1), where

the relatively firm sediments probably increased the forces on scallops trapped beneath the passing dredge skids. These firmer sediments may also cause dredges to bounce more, further increasing the likelihood of damage to scallops. Divers also observed the highest percentage (20%) of dead scallops on the Dromana plot after it was dredged, and mortality estimates of small scallops, based on catches in an Ockelmann sled, were also highest at Dromana (57% in 7 months, Table 6). However, estimates of scallop mortality at Portarlinton and St. Leonards may be unreliable. At Portarlinton, all epifaunal abundances were underestimated predredging due to excessive towing speed, whereas at St. Leonards the increased catches of all species after dredging (Table 6) appear to be an artefact caused by an increase in the efficiency of the Ockelmann sled after dredging flattened the plot. The proportion of damaged scallops even at Dromana (~5%, Table 1) was much lower than was observed on an intensively fished, very dense bed of scallops in Bass Strait where over 40% of scallops taken in dredges were damaged (McLoughlin et al. 1991).

Scallop dredges caught mostly scallops (Table 1) because scallops are the dominant epifaunal species in the main scallop grounds in Port Phillip Bay. The main bycatch species taken in dredges were oysters, ascidians, and, periodically, spider crabs. Survival of discarded epifauna requires further investigation, but mortality rates of oysters were probably low, while those of ascidians were probably high (Table 5) and those of spider crabs were high (>50%, Table 4). The ecological consequences of the removal of spider crabs are uncertain, but as their populations remain large, despite being dredged for 30 years, their survival cannot be threatened seriously by scallop dredging.

At St. Leonards, infauna was typically reduced in abundance by 20–30% in the 3.5 months following dredging (Table 7; Currie and Parry 1996). Changes caused by dredging at Dromana and Portarlinton were similar to those at St. Leonards despite the much higher intensity of dredging at Portarlinton. However, few changes in abundance measured at Dromana and Portarlinton were significant because sampling was inadequate to detect changes comparable with those observed at St. Leonards (power of test, Table 7). Kaiser and Spencer (1996) found that trawling reduced infauna by up to 58% but that effects were undetectable in more mobile sediments. The BACI design used at St. Leonards (Currie and Parry 1996) provides a robust measure of fishing impacts on infauna. The absence of spatial control plots at Dromana and Portarlinton in this study and of before/after temporal controls in Kaiser and Spencer’s (1996) studies means that unmeasured temporal or spatial variation may have influenced the results of these studies.

Infaunal communities on our study plots were similar to the “intermediate sediment” community found in the main scallop grounds in Port Phillip Bay (Fig. 1b; Currie and Parry 1999). Scallop dredging also occurred on other infaunal communities in coarser and finer sediments. Dredging impacts are only likely to be greater than measured on our plots in the finer sediments in the centre of the bay. These sediments are seldom disturbed by natural storms (De Groot 1984; Theil and Schriever 1990; Jones 1992; Kaiser and Spencer 1996), and their infaunal communities may be more sensitive to the short-lived but very large increases in sedi-

Table 7. Percentage change following dredging, predredging abundance, and power of ANOVA's to detect 30 and 50% change following dredging for the 10 most abundant species at Dromana, Portarlington, and St. Leonards.

Species	Percentage change following dredging			Predredging abundance(no./0.1 m ²)		Power of test ($p = 0.05$)			
	Dromana	Portarlington	St. Leonards	Dromana	Portarlington	Dromana		Portarlington	
						$\Delta 30\%$	$\Delta 50\%$	$\Delta 30\%$	$\Delta 50\%$
<i>Aricidea</i> sp. 1	152		-26	3.5		0.15	0.43		
<i>Artacamella</i> <i>dibranchiata</i>	3	-23	-28**	21.9	3.1	0.12	0.34	0.24	0.69
<i>Bullozanthura</i> <i>pambula</i>	-28**	-16	-0.4**	17.9	2.3			0.35	0.87
<i>Byblis mildura</i>	-28		13	5.0		0.26	0.73		
<i>Corbula gibba</i>	14		-66**	21.3		0.16	0.46		
<i>Dimorphostylis</i> <i>cottoni</i>	28	-49	119	13.9	6.8	0.47	0.96	0.17	0.49
<i>Lumbrineris</i> cf. <i>latreilli</i>	8		-18	19.5		0.30	0.80		
<i>Nephtys inornata</i>	-21*		-3	20.3					
<i>Photis</i> sp. 1	14		-79**	10.5		0.17	0.48		
<i>Prionospio</i> <i>coorilla</i>	3		-60	46.5		0.16	0.46		
<i>Ampharete</i> sp. 1	-35			8.7		0.28	0.76		
<i>Chaetozone</i> sp.1	-38**		-32	11.0					
<i>Asychis</i> sp. 1		11			7.7			0.72	0.99
<i>Marphysa</i> sp. 1		-12			4.9			0.62	0.99
<i>Natatolana</i> <i>corpulenta</i>	-39*	-66**	23	3.1	2.1				
<i>Theora lubrica</i>		-29*			3.6				

Note: At St. Leonards, percentage change is that during 3.5 months following dredging (Currie and Parry 1996). Data are shown only for species with densities >2.0/0.1 m². * $p < 0.1$, ** $p < 0.05$.

mentation rates that occur behind scallop dredges (Black and Parry 1994).

Adequate data on the abundance of epifauna in Port Phillip Bay before dredging commenced do not exist. Anecdotal information suggests that dredging may have levelled low-relief inshore reefs on the east of the bay and severely damaged some seagrass communities on the west of the bay before dredging in these shallow regions was prohibited in 1985. But in the main scallop grounds, where epibiota other than scallops is uncommon, changes to infauna are small enough to be difficult to detect without a large and expensive sampling program. Results of this study and Currie and Parry (1996) are consistent with an emerging generalisation that the effects of fishing gear dragged along the seafloor depend largely on the amount of epibiota in the area being trawled or dredged. Impacts are large where epifauna and epiflora (e.g., bryozoans, tube-building polychaetes, sponges, and seagrass) form a habitat that is itself vulnerable to fishing gear because it is fragile, inflexible, easily uprooted, or large enough to be retained by the gear (Bradstock and Gordon 1983; Peterson et al. 1987; Sainsbury 1988; Collie et al. 1997). The extent of impacts will also depend on the frequency of disturbance and the recovery rate of the biota (Van Dolah et al. 1987; Hall 1994).

Changes to infauna caused by scallop dredging are smaller than the cumulative changes to community structure of infauna in Port Phillip Bay over the past 20 years (Currie and Parry 1999). The contribution of 30 years of scallop dredg-

ing to these ecological changes is unclear. Like many bays with urbanised catchments, there are many possible causes of long-term change. Over 20 years, there has been an increase in nutrient loads, increasing algal blooms (Parry et al. 1989), the introduction of several new exotic species (Currie and Parry 1999), decreased discharge of most toxicants (Fabris et al. 1992; Phillips et al. 1992) but an increase in tributyl tin (Foale 1993), and increasing fishing pressure (Hobday et al. 1999). In addition, natural temporal changes in community structure may be contributing, although they remain unmeasured.

Scallop dredging may be the main cause of the marked decrease in size and longevity of scallops in Port Phillip Bay over this same period. Few scallops now live beyond 3 years or 75 mm, when they once lived to an age of 7 years and the minimum legal size during the early years of the fishery was 80 mm (Gwyther and McShane 1988). Scallops of this size have not been seen for many years, even in lightly fished areas, suggesting that factors other than fishing mortality may be important. A large bloom of the toxic diatom *Rhizosolenia chunii* caused widespread mortality in 1988 (Parry et al. 1989), resulting in the closure of the fishery in 1989 and 1990. Blooms of *Rhizosolenia chunii* have been recorded in 1975, 1987 (Parry et al. 1989), 1993, 1994, and 1997 (G. Arnot, Marine and Freshwater Resources Institute, Queenscliff, personal communication), and their frequency appears to be increasing. In 1992 and 1993, very large recruitments (1+ age class) made only minor contributions to increased

fishing yields, as mortality rates were very high and many died while the fishery was closed over summer (N. Coleman, Marine and Freshwater Resources Institute, Queenscliff, personal communication). It is also of concern that there have been no commercial quantities of scallops in central Port Phillip Bay since the virgin stock was eliminated in the 1960's. However, the failure of scallop populations to reestablish in the centre of the bay may be unrelated to dredging. Since the 1970's the European bivalve *Corbula gibba* has been introduced into Port Phillip Bay where it now occurs in densities of 1000/m² at St. Leonards (= *Corbula cf. coxi*, Currie and Parry 1996), and competition for food between scallops and *Corbula gibba* is likely, particularly in muddier regions in the centre of the bay where *Corbula gibba* is abundant (Parry et al. 1995).

Most community concern with scallop dredging and other potential impacts is with their possible long-term effects. The recent, and apparently permanent, closure of the scallop fishery in Port Phillip Bay provides an opportunity to determine more confidently the contribution of scallop dredging to long-term changes. Studies are now in progress to monitor the recovery of Port Phillip Bay, but as dredging is only one of many potential impacts causing ecological changes on time scales of decades, it is important that dredging impacts are maintained in small areas within the "recovering" community so that the role of dredging itself can be identified. Ironically, two of the species most impacted by scallop dredging are recently introduced species, the bivalve *Corbula gibba* (Currie and Parry 1996) and the large epifaunal sabellid polychaete *Sabella spallanzanii* (G.D. Parry, personal observations), and cessation of dredging may increase their ecological effects.

Acknowledgements

We thank the Victorian scallop industry whose assistance was vital to the success of this study. Field support was provided by Ross Haughton, Dave Byer, Bob Metcalfe, and John Barry. Laboratory and field studies were undertaken by Arnold Jahnecke, Rhonda Flint, Anna Bos, Matt Hoskins, and Mike Forsyth, with further assistance from Melinda Miller, Steve Frlan, Bruce Waters, Di Crookes, and the many staff at the Marine and Freshwater Resources Institute directly involved in the dredging experiments. Statistical assistance was provided by Anne Gason, Nik Dow, and Mick Keough. Gary Poore, Robin Wilson, and Sue Boyd (Museum of Victoria) checked identifications of all specimens in our reference collection.

References

- Belbin, L. 1990. PATN technical reference manual. CSIRO Division of Wildlife and Ecology, Canberra, Australia.
- BEON. 1992. The effects of beamtrawl fishery on the bottom fauna in the North Sea III — the 1991 studies. Beleidsgericht Ecologisch Onderzoek Noordzee en Waddenzee Rep. 16. pp. 1–27.
- Black, K.P., and Parry, G.D. 1994. Sediment transport rates and sediment disturbance due to scallop dredging in Port Phillip Bay. Mem. Queensl. Mus. **36**: 327–341.
- Black, K.P., and Parry, G.D. 1999. Entrainment, dispersal and settlement of scallop dredge sediment plumes: field measurements and numerical modelling. Can. J. Fish. Aquat. Sci. **56**. In press.
- Bradstock, M., and Gordon, D.P. 1983. Coral-like bryozoan growths in Tasman Bay, and their protection to conserve commercial fish stocks. N.Z. J. Mar. Freshwater Res. **17**: 159–163.
- Bray, J.R., and Curtis, J.T. 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. **27**: 325–349.
- Caddy, J.F. 1973. Underwater observations on tracks of dredges and trawls and some effects of dredging on a scallop ground. J. Fish. Res. Board Can. **30**: 173–180.
- Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. Aust. J. Ecol. **18**: 117–143.
- Cohen, J. 1988. Statistical power analysis for the behavioural sciences. 2nd ed. L. Erlbaum Associates, Hillsdale, N.J.
- Collie, J.S., Escanero, G.A., and Valentine, P.C. 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. Mar. Ecol. Prog. Ser. **155**: 159–172.
- Currie, D.R., and Parry, G.D. 1996. The effect of scallop dredging on a soft sediment community: a large scale experimental study. Mar. Ecol. Prog. Ser. **134**: 131–150.
- Currie, D.R., and Parry, G.D. 1999. Changes to benthic communities over 20 years in Port Phillip Bay, Victoria, Australia. Mar. Pollut. Bull. **38**: 36–43.
- De Groot. 1984. The impact of bottom trawling on benthic fauna of the North Sea. Ocean Manage. **9**: 177–190.
- DeLury, D.B. 1947. On the estimation of biological populations. Biometrics, **3**: 145–167.
- Eleftheriou, A., and Robertson, M.R. 1992. The effects of experimental scallop dredging on the fauna and physical environment of a shallow sandy community. Neth. J. Sea Res. **30**: 289–299.
- Fabris, J.G., Monahan, C., Nicholson, G., and Walker, T.I. 1992. Total mercury concentrations in sand flathead *Platycephalus bassensis* Cuvier and Valenciennes, from Port Phillip Bay, Victoria. Aust. J. Mar. Freshwater Res. **43**: 1393–1402.
- Foale, S. 1993. An evaluation of the potential of gastropod imposex as a bioindicator of tributyltin pollution in Port Phillip Bay, Victoria. Mar. Pollut. Bull. **26**: 546–552.
- Friedman, G.M., and Sanders, J.E. 1978. Principles of sedimentology. John Wiley & Sons, New York.
- Gislason, H. 1994. Ecosystem effects of fishing activities in the North Sea. Mar. Pollut. Bull. **29**: 520–527.
- Graham, M. 1955. Effect of trawling on animals of the sea bed. Deep-Sea Res. **3**: 1–6.
- Greilach, P.R., Black, K.P., Parry, G.D., and Forsyth, M. 1995. Scallop dredging and sedimentation in Port Phillip Bay. Vol. 2. Sediment analyses. Working Pap. No. 29, Victorian Institute of Marine Sciences, Victoria, Australia.
- Gwyther, D., and Burgess, D.C. 1986. Abundance of scallops in Port Phillip bay and predictions of yields for the 1986 season. Mar. Sci. Lab. Tech. Rep. No. 59, Queenscliff, Victoria, Australia.
- Gwyther, D., and McShane, P.E. 1988. Growth rate and natural mortality of the scallop *Pecten alba* Tate in Port Phillip Bay, Australia, and evidence for changes in growth rate after a 20-year period. Fish. Res. **6**: 347–361.
- Hall, S.J. 1994. Physical disturbance and marine benthic communities: Life in unconsolidated sediments. Oceanogr. Mar. Biol. Annu. Rev. **32**: 179–239.
- Hobday, D.K., Officer, R.A., and Parry, G.D. 1999. Changes to fish communities in Port Phillip Bay over 20 years. Mar. Freshwater Res. In press.
- Hughes, W.D. 1973. Operational tests of Victorian scallop boats. Aust. Fish. **33**: 14–16.
- Hulbert, S.H. 1987. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. **54**: 187–211.

- Joll, L.M., and Penn, J.W. 1990. The application of high-resolution navigation systems to Leslie-DeLury depletion experiments for the measurement of trawl efficiency under open-sea conditions. *Fish. Res.* **9**: 41–55.
- Jones, J.B. 1992. Environmental impact of trawling on the seabed: a review. *N.Z. J. Mar. Freshwater Res.* **26**: 59–67.
- Kaiser, M.J., and Spencer, B.E. 1995. Survival of by-catch from a beam trawl. *Mar. Ecol. Prog. Ser.* **126**: 31–38.
- Kaiser, M.J., and Spencer, B.E. 1996. The effects of beam-trawl disturbance on infaunal communities in different habitats. *J. Anim. Ecol.* **65**: 348–358.
- McLoughlin, R., Young, P.C., Martin, R.B., and Parslow, J. 1991. The Australian scallop dredge: estimates of catching efficiency and associated indirect fishing mortality. *Fish. Res.* **11**: 1–24.
- McShane, P.E. 1981. The effect of scallop dredging on the macrobenthos of a muddy environment in Port Phillip Bay. *Mar. Sci. Lab. Tech. Rep. No. 4*, Queenscliff, Victoria, Australia.
- McShane, P.E., and Gwyther, D. 1984. Port Phillip Bay scallop population assessment and fishery catch forecast — January 1984. *Mar. Sci. Lab. Tech. Rep. No. 35*, Queenscliff, Victoria, Australia.
- Messieh, S.N., Rowell, T.W., Peer, D.L., and Cranford, P.J. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. *Continental Shelf Res.* **11**: 1237–1263.
- Parry, G.D., Langdon, J.S., and Huisman, J.M. 1989. Toxic effects of a bloom of the diatom *Rhizosolenia chunii* on shellfish in Port Phillip Bay, southeastern Australia. *Mar. Biol. (Berl.)*, **102**: 25–41.
- Parry, G.D., Hobday, D.K., Currie, D.R., Officer, R.A., and Gason, A.S. 1995. The distribution, abundance and diets of demersal fish in Port Phillip Bay. *CSIRO Port Phillip Bay Environ. Stud. Tech. Rep. No. 21*, CSIRO, Melbourne, Australia.
- Peterman, R.M. 1990. Statistical power analysis can improve fisheries research and management. *Can. J. Fish. Aquat. Sci.* **47**: 2–15.
- Peterson, C.H. 1977. Competitive organisation of the soft-bottom macrobenthic communities of southern California lagoons. *Mar. Biol. (Berl.)*, **43**: 343–359.
- Peterson, C.H. 1993. Improvement of environmental impact analysis by application of principles derived from manipulative ecology: lessons from coastal marine case histories. *Aust. J. Ecol.* **18**: 21–52.
- Peterson, C.H., Summerson, H.C., and Fegley, S.R. 1987. Ecological consequences of mechanical harvesting of clams. *Fish. Bull. U.S.* **85**: 281–298.
- Phillips, D.J.H., Richardson, B.J., Murray, A.P., and Fabris, J.G. 1992. Trace metals, organochlorines and hydrocarbons in Port Phillip Bay, Victoria: a historical review. *Mar. Pol. Bull.* **25**: 200–217.
- Rosenbaum, M.A., Black, K.P., and Parry, G.D. 1992. Scallop dredging and sedimentation in Port Phillip Bay. Vol. 1. Sediment and hydrodynamic measurements: field data collection and analysis. Working Pap. No. 24, Victorian Institute of Marine Sciences, Victoria, Australia.
- Sainsbury, K.J. 1988. The ecological basis of multispecies fisheries, and management of a demersal fishery in tropical Australia. *In* Fish population dynamics. *Edited by* J.A. Gulland. John Wiley & Sons, London, U.K. pp. 349–382.
- Snedecor, G.W., and Cochran, W.G. 1967. *Statistical methods*. 6th ed. Iowa State University Press, Ames, Iowa.
- Steel, R.G.D., and Torrie, J.H. 1960. *Principles and procedures of statistics*. McGraw-Hill, New York.
- Stewart-Oaten, A., Murdoch, W.M., and Parker, K.R. 1986. Environmental impact assessment: “pseudoreplication” in time? *Ecology*, **67**: 929–940.
- Theil, H., and Schriever, G. 1990. Deep-sea mining, environmental impact and the DISCOL project. *Ambio*, **19**: 245–250.
- Thrush, S.F., Hewitt, J.E., Cummings, V.J., and Dayton, P.K. 1995. The impact of habitat disturbance by scallop dredging on marine benthic communities: what can be predicted from the results of experiments? *Mar. Ecol. Prog. Ser.* **129**: 141–150.
- Underwood, A.J. 1991. Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Aust. J. Mar. Freshwater Res.* **42**: 569–587.
- Van Dolah, R.F., Wendt, P.H., and Nicholson, N. 1987. Effects of a research trawl on a hard-bottom assemblage of sponges and corals. *Fish. Res.* **5**: 39–54.