

# Dredge survey and stock assessment for the Coromandel scallop fishery, 2004

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## Final Research Report for Ministry of Fisheries Research Project SCA2003/01 Objectives 1 & 2

National Institute of Water and Atmospheric Research

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#### FINAL RESEARCH REPORT

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#### 7. Executive Summary:

A dredge survey for scallops were carried out in the Coromandel fishery in May 2004. Many strata and parts of strata thought by fishers unlikely to support commercial fishing in 2004 and all areas closed to commercial fishing were excluded. Biomass estimates at the time of the survey were made using the area swept method "corrected" using length-specific dredge efficiency scalars. Start-of-season biomass was predicted using a stochastic growth transition matrix, and stochastic scalars for average survival and average recovery of meatweight from greenweight. These biomass estimates are treated as estimates of absolute biomass.

Absolute start-of-season biomass over 100 mm shell length (for comparison with previous surveys) was predicted to be 2767 t greenweight (345 t meatweight) with CVs of 21 and 26%, respectively. This is a substantial improvement on all surveys since 1999, especially for the Mercury Island beds, and exceeds all survey estimates since "complete" surveys began in 1995 (using comparable analyses). Including all scallops likely to be over the commercial minimum legal size of 90 mm by the start of the season increased this estimate to 9024 t greenweight (1131 t meatweight) with CVs of 20 and 25%, respectively. The most pessimistic yield estimate (as CAY) for the 2004 season is about 50 t meatweight (using a slightly revised estimate of  $F_{0.1}$  an assumed 100% dredge efficiency, no growth between survey and season, and the lowest recorded annual average recovery of meatweight from greenweight). These are very conservative assumptions, however, and CAY estimates made using average values for dredge efficiency, growth, and meatweight recovery were 300–350 t (200–250 t if incidental effects of the dredge method on habitat heterogeneity and juvenile mortality and a 0.04 m<sup>-2</sup> minimum acceptable density for fishing are included).

## 8. Objectives:

## **Overall Objectives:**

1. To carry out a stock assessment of scallops (*Pecten novaezelandiae*) in the Coromandel fishery, including estimating abundance and sustainable yields.

## Specific Objectives:

- 1. To estimate the absolute abundance and population size frequency of scallops in the main scallop beds. The target coefficient of variation (c.v.) of the estimate of absolute recruited abundance is 20 %.
- 2. To estimate yield following the completion of the survey described in Objective 1.

## 9. Methods:

## 9.1 General

This report describes dredge surveys for scallops (*Pecten novaezelandiae*) carried out in the Coromandel fishery under contract to the Ministry of Fisheries (project SCA2003/01). Surveys were conducted almost annually between 1978 and 1999, with coverage of the fishery increasing with time. Survey designs incorporating both dive and dredge components were refined over the years using historical survey data, catch-effort information, a review of optimisation procedures, and discussions with managers and fishers. There was no survey in 2000, but dredge surveys were conducted in 2001, 2002, and 2003.

Between 1992 and 1994, survey results were used to estimate Provisional Yield (PY, after a method by Cryer, 1994) and set catch limits for the Coromandel fishery. The minimum legal size (MLS) for scallops taken commercially in the Coromandel fishery was reduced from 100 to 90 mm at the start of the 1995 season, but remained at 100 mm for amateur fishers. The management plan (for the 1995 to 1997 seasons) adopted an assessment regime whereby the available yield was calculated using Provisional Yield (PY) based on the abundance and biomass of scallops greater than 100 mm shell length, pending research into more appropriate methods. This research was completed in late 1997 when the management plan expired. A more typical "Current Annual Yield" (CAY) yield estimator was adopted in 1998, based on predicted start-of-season recruited biomass and estimates of reference fishing mortality incorporating incidental effects of the dredge method on scallop growth and mortality (Cryer & Morrison 1997).

For poorly understood reasons, biomass and catches from the fishery were very low between about 1998 and 2002 (e.g., Cryer 2001a, Cryer & Parkinson 2002). Especially in the late 1990s, low biomass coincided with poor condition, "black gill" syndrome (Diggles et al. 2000), and a peak (following a rapid increase since 1996) of the filter-feeding tubeworm *Chaetopterus* sp. on many of the beds. *Chaetopterus* sp. builds large clumps of parchment-like tubes and renders dredging for scallops impossible (because the dredge fills with tubes and therefore cannot catch scallops). There was probably catastrophic mortality of scallops on some beds (Cryer 2001b), but the role played by the expansion of *Chaetopterus* sp. is not known, despite some (not all)

anecdotal reports that scallops do not survive where *Chaetopterus* sp. is abundant. The causes of these apparently major changes to the ecology of the Coromandel fishery remain unclear, but the 2003 survey suggested that scallop biomass was increasing and *Chaetopterus* was becoming rare.

The principal aim of the 2004 survey work was to estimate the absolute abundance of scallops by size class on those beds of the Coromandel fishery likely to support successful commercial fishing in the 2004 season. This information can be used to estimate recruited biomass at the start of the forthcoming season (15 July of each year) using information on growth, mortality, and likely condition (of scallops) during the season. Estimates of yield are derived using the projected start-of-season biomass estimates and reference rates of fishing mortality from stochastic yield-per-recruit and egg-per-recruit models (Cryer & Morrison 1997).

#### 9.2 Survey methods

The survey was conducted between 6 and 13 May 2004. The choice of an appropriate time for surveys entails balancing the conflicting pressures of operational ease and uncertainty in the results. Early surveys benefit from long daylight hours and settled weather, but the long lag between survey completion and season opening render biomass estimates sensitive to the assumed values for growth and mortality. In addition, scallops are susceptible to periodic catastrophic declines in abundance, and a longer lag between survey and season increases the probability of such an occurrence. Surveys undertaken later in the year can be hampered by short working days and less favourable conditions, and they risk being seriously delayed by inclement .weather increases. However, the effect on biomass estimates of poor assumptions about growth and mortality is smaller, and the chance of catastrophic declines in abundance following the survey is reduced.

All sampling for the 2004 surveys was undertaken by dredge and no diving to estimate dredge efficiency was done. We used the same vessel and skipper as used in most historical estimates of dredge efficiency. Single phase stratified random sampling was undertaken in 14 strata: Waiheke Island (one stratum), Colville (one stratum), Little Barrier Island (two strata), Mercury Islands (seven strata), and the western Bay of Plenty (three strata at Motiti Island, Papamoa Beach, and off the Katikati Entrance). After excluding strata thought unlikely to be productive in 2004 and areas closed to commercial fishing, the total area of strata sampled in 2004 was 149 km<sup>2</sup>, compared with 130 km<sup>2</sup> in 2003, 119 km<sup>2</sup> in 2002, 125 km<sup>2</sup> in 2001, and 253–341 km<sup>2</sup> between 1996 and 1999, Figure 1; Appendix 1).

The survey was not formally optimised to minimise the predicted CV of the estimate of recruited biomass for two main reasons. First, some strata were redesigned for the 2004 survey in response to changes in the fishery performance. Second, time constraints on the surveys limited the number of ways in which stations could realistically be allocated to strata. These constraints necessitated a more pragmatic approach than was used in the 1990s (e.g., Cryer & Parkinson 1999a). Strata that were sufficiently close together to tackle in a single day (e.g., those at the Mercury Islands) were grouped. Up to about 25 shots can be completed in a problem-free day with little steaming, so stations were allocated to strata within groups according to their relative stratum sizes and a qualitative understanding of historical performance until the total for the group was 20–25.

The positions of stations within strata were randomised using the Random Stations package (RAND\_STN v 1.7 for PCs; MAF Fisheries 1990) constrained to keep all stations at least 500 m apart. This package estimates the area of each stratum, and gives the latitude and longitude of each random station.



Figure 1: Location of strata for the survey of the Coromandel scallop fishery in 2004. Groups of strata are labelled with geographic descriptions used in the text (see Appendix 1 for details and stratum areas).

Dredging was undertaken from the chartered commercial dredge vessel *Kataraina* (dredge width 2.0 m). The skipper's brief was to tune his gear (select course, speed, warp length, etc.) to maximise his total catch at each station. Tows were nominally 0.3–0.5 nautical miles long (556–926 m, assessed using non-differential GPS), depending on the expected average size of the catch. However, the dredge occasionally lost contact with the bottom or "flew" (because of hard or uneven substrates, an increase in depth, a dredge full of detritus or scallops, etc.) and, on these occasions, the tow was terminated and the actual distance travelled along the ground was estimated using GPS. At the end of each tow, the dredge was retrieved and emptied onto the sorting tray on the boat. All live scallops were separated from the detritus and bycatch and their maximum lengths measured to the nearest millimetre rounded down. Occasionally, large catches

were randomly subsampled for length. All unmeasured scallops were counted. No facilities for weighing the catch at each station were available to estimate the fraction sampled by weight.

## 9.3 Estimating and correcting for dredge efficiency

Work on estimating the efficiency of toothed (or "tined") scallop dredges started in Europe in the 1950s (Baird 1955, 1959, Baird & Gibson 1956) and in Canada in the 1960s (Caddy 1968, 1971). The efficiency of dredges in common use was found to be very low, leading to the development of more efficient gear such as the "Baird" dredge whose efficiency for scallops close to the minimum legal size (MLS) was found to be 24-33% (Rolfe 1969). Zacharin (1988), however, found the efficiency of Australian variants of the Baird dredge to have an efficiency of less than 10%, and McLoughlin et al. (1993) described attempts by the Australian industry and researchers to find more efficient and less damaging alternatives. Chapman *et al.* (1977) studied spring-loaded dredges used in Scottish fisheries and found their efficiency to be about 13% for scallops close to MLS, whereas "standard" dredges with a fixed tooth-bar were about 20% efficient. They found small scallops to be caught with very low efficiency (<5%) by all dredge designs, and efficiency for very large scallops sometimes to be low compared with scallops close to MLS. Chapman *et al.* (*op.cit.*) also reported that dredge efficiency varied with substrate type.

Dredges have been used to survey a wide variety of relatively sessile, soft-sediment invertebrates including oysters (Allen & Cranfield 1979, Chai et al. 1992, Doonan et al. 1994), crabs (Stagg & Abbe 1996, Voelstad et al. 2000), whelks (Stoner et al. 1998), and surf clams and other burrowing bivalves (Michael et al. 1990, Thorarinsdottir & Einarsson 1996, Gaspar et al. 1999), as well as scallops (e.g., Rudders et al. 2000, Weinberg et al. 2000). Dare et al. (1993, 1994) examined the efficiency of research and commercial dredges used in biomass surveys for stock assessments in the English Channel and found the two to be comparable. Average efficiency for scallops close to MLS was about 30%, being lower for very large scallops and much lower for very small scallops. Mason et al. (1979) found efficiency in Scottish surveys to be about 20% and attributed the low dredge efficiency to a mound of sediment collecting in the mouth of the dredge during towing. This sediment "block" resulted in further sediment and scallops being pushed out to the side of the dredge. Giguere & Brulotte (1994) found French survey dredges to be 8-19% efficient for scallops close to MLS based on comparisons with parallel video transects. In northern New Zealand, commercial box dredges have been used in a stock assessments since the 1980s and estimates of efficiency close to the MLS have varied between ~10% and ~100% (e.g., Cryer & Morrison 1997; Cryer & Parkinson 1999b).

All strata in the 2004 survey were sampled by dredge and therefore required correction for sampling efficiency. Separate estimates of dredge efficiency have historically been used for sandy and silty or muddy substrates (Cryer & Parkinson 1999b), although relatively fewer data have been collected on silty substrates. Most previous estimates of dredge efficiency and selectivity have been made using the two very similar vessels *L'Aries* and *Kataraina* (the latter was used for the 2004 survey) and the same skipper (Karl Aislabie). Data were collated from 24 experiments in the Coromandel fishery (19 on sand, 5 on silt or mud, Table 1) where scallop density (by 5 mm size classes) was estimated by dredging and diving. Most of these experiments were conducted as adjuncts to surveys designed to estimate absolute biomass as part of annual stock assessments, but some data collected in 1995 were part of a project to assess the implications of incidental effects of dredging for fishery performance (Cryer & Morrison 1997).

Table 1: Summary statistics (means with ranges) of dredge efficiency experiments used in this study.

	Sandy substrates	Silty substrates
No. of experiments	19	5
No. of vessels	2	3
Mean depth	22.3	19.2
	12-30	15-23
Mean number of dredge tows	2.9	11.0
	2–4	4–20
Mean area swept by dredge	4 048	4 322
intean area shope by areage	1 148-7 445	3 593-5 371
Mean no. scallops caught by dredge	415	1 797
	83-938	377-4 129
Mean number of dive sites	4.8	10.6
	2–9	3–16
Mean area swept by divers	602	552
1 - 5	245-1 731	67-1 431
Mean no. scallops caught by divers	180	335
1 0 7	12-435	147-547
Mean density of scallops >85 mm	0.264	0.874
	0.011-1.162	0.082-1.543

Each experiment consisted of 2–20 dredge tows (mean 5.0) and 2–16 dive sites (mean 7.1). Basic units for re-sampling were these tows and dive sites, not individual scallops within them. Because area swept was variable for both dredge tows and dive sites within most experiments, and length was recorded with differing precision, all data were first standardised to estimated density per square metre in 5 mm size classes (10–14.9 mm, 15–19.9 mm, etc.). This approach entails an implicit assumption that all samples taken by the same method within an experiment are equivalent. Other approaches could be applied, giving more weight to samples in proportion to area swept or animals caught.

Within an experiment, the mean estimated density was calculated for each size class using each sampling method, and dredge efficiency for that size class in that experiment was estimated by dividing the density estimated by dredge by the density estimated by dive (assumed to be 100% and without error). Size classes where divers caught no scallops were excluded, as were size classes where fewer than a pre-set number of scallops were caught by either dredging or diving (all samples combined) in that experiment. This "minimum acceptable number" of scallops in a size class was varied from 1 (i.e., no filter) through 4 and 9 to 15 to explore the effects of excluding estimates of dredge efficiency that are likely to be highly imprecise (confidence limits for binomial proportions expand markedly at small samples sizes).

Average efficiency for a size class was estimated by calculating the reciprocal of each relevant estimate (resulting in "biomass scalars"), calculating the mean of these reciprocals, and converting back to efficiency (i.e., by averaging the biomass scalars rather than the efficiency estimates). Estimates of zero dredge efficiency for a size class (divers found scallops, but dredges did not) confound this approach because  $1/0 = \infty$  and any average biomass scalar containing  $\infty$  must also be  $\infty$  (a dredge efficiency of zero). Biomass scalars for size classes where dredge

efficiency was estimated to be zero were therefore set to 100, roughly the lowest estimable dredge efficiency, given our experimental approaches (Figure 2).



Figure 2: Cumulative frequency distribution of the estimated biomass scalar for the base non-parametric analysis. This distribution was used to establish 100 as the default biomass scalar when estimated dredge efficiency for a size class in an experiment was zero.

Biomass scalars for sandy and silty substrates were averaged separately. Each mean scalar was calculated un-weighted (i.e., assuming each estimate to be equivalent) then using weights to give more credence to estimates that are likely to be more reliable. Weights for a 5 mm size class were proportional to the smaller of the number of scallops caught by dredge and dive sampling (divers usually caught fewer scallops), or by the logarithm of this number plus one. Over the range of counts for the binomial numerator and denominator encountered, the confidence interval (relative to the mean) decreased roughly exponentially with increasing number of animals in the smaller sample so, *a priori*, the log weighting is probably to be preferred.

Because efficiency estimated by 5 mm size classes was found to be poorly estimated for "rare" size classes, average efficiency was estimated for three wider size classes of interest in stock assessment surveys. These were 85–99 mm (immediate pre-recruits and some of the recruited biomass in the Coromandel fishery), 100–114 mm (the bulk of the recruited biomass), and  $\geq 115$  mm (large scallops that preliminary analyses and many anecdotes suggest are caught with low efficiency). Efficiency for each experiment and for each combination of experiments was estimated as for the narrower size classes.

Confidence limits for the size-class-based approach were estimated by non-parametric bootstrapping, re-sampling from experiments and dredge tows and dive sites within experiments. Within an experiment, the dredge tows and dive sites were re-sampled, with replacement, taking the same number of tows and sites as in the original experiment. The mean estimated density was calculated for each size class using each sampling method, and dredge efficiency for that size class in that bootstrap run was estimated by dividing the density estimated by dredge by the density estimated by dive (assumed to be 100% efficient). For each bootstrap sample, average efficiency for a size class was calculated exactly as for the base estimate (weighting according to



the log of the number of animals in the smaller sample) for sandy and silty substrates separately. The bootstrap estimates were stored and their statistical distributions used to estimate confidence intervals around the base estimates.

*Figure 3*: Mean dredge selectivity (efficiency) curves by 5 mm size classes for sandy and silty substrates in the Coromandel fishery estimated using the log (minimum N) weighting procedure and constraints on the minimum acceptable count by method increasing from 1 to 15 scallops in a size class. In each plot, the middle line is the mean of 1000 bootstraps and the outer lines delimit the 5<sup>th</sup> and 95<sup>th</sup> percentiles of the bootstraps. The larger dots are estimates for wider size classes.

The results of this analysis were sensitive to the method of averaging the results of individual experiments (averaging efficiencies or their reciprocals), to the weighting procedure, and to constraints on the minimum acceptable number of scallops in a size class. However, the overall trends appear clear; dredge efficiency (relative to divers) increases with increasing scallop size up to about 100 mm, beyond which it declines (Figure 3). This pattern is consistent among substrates and is not very sensitive to the weighting procedure (results not shown) or the minimum acceptable number on a size class.

These results suggest that it is inappropriate to apply dredge efficiency corrections as a simple scalar for all size classes likely to be part of the recruited biomass (as in scallop stock assessments between 1997 and 2002). That approach is likely to underestimate the density of large, heavy scallops and, consequently, probably underestimates biomass.

## 9.4 Analytical approach

The method of estimating start-of season recruited biomass for scallops was modified during the 2002 and 2003 stock assessments (Cryer & Parkinson 2004) and contains the following steps:

- 1. The length frequency distribution for each sample is scaled according to the sampling fraction (if any).
- 2. The length frequency distribution for each sample is converted to "uncorrected" density per unit area of seabed i.e., assuming the dredge to be 100% efficient for all size classes).
- 3. The length frequency distribution for each sample is "corrected" for dredge efficiency to estimated "real" density per unit area of seabed. Dredge efficiency corrections for sandy or silty substrate are used as appropriate. These are combined to estimate the population length frequency distribution.
- 4. The weight (per unit area) of scallops at or above the minimum legal size (or other length of interest) is estimated using a length-weight regression. Variance associated with the regression is included by bootstrapping from the raw length-weight data.
- 5. The mean recruited biomass (per unit area) for each stratum and for the whole population (or any subset of strata), together with the sampling variance are estimated using bootstraps from the sampling data.
- 6. The absolute recruited biomass at the time of the survey is estimated by scaling the estimate of the mean biomass by the combined area of all pertinent strata. The stratum areas are considered to be known without error.
- 7. The corrected population length frequency distribution (from step 3) is projected to the start of the forthcoming season using a growth transition matrix based on tag return data. Uncertainty about the expected average growth between survey and season is incorporated by bootstrapping, generating a new growth model for each iteration by bootstrapping from the original tag return data.

- 8. Mortality between survey and season is incorporated as an instantaneous rate of  $M = 0.5 \text{ y}^{-1}$ , bootstrapping (parametrically) from an estimated statistical distribution of M.
- 9. The absolute recruited biomass at the start of the season is estimated by repeating steps 4–6, again assuming the stratum areas to be known without error.

Sampling variance can be estimated for the simple case of no dredge efficiency scaling using standard parametric methods (e.g., Snedecor & Cochran 1989) but is estimated for all more complex approaches by re-sampling (with replacement) the individual tows within each stratum. For each iteration, the tows are re-sampled, the average density and standing biomass by size class are estimated, and scaled by stratum size. Variance associated with a slightly revised length-weight regression ( $W = 0.00037 L^{2.690}$ ) is included (optionally) by bootstrapping from the raw length-weight data and generating a new length-weight relationship for each iteration. The absolute recruited biomass at the time of the survey is then estimated by summing all the pertinent stratum estimates (the stratum areas are considered to be known without error).

To estimate start-of-season biomass, the corrected population length frequency distribution is projected forward using a length-based growth transition matrix based on tag return data. Growth increments for 129 scallops tagged during pre-season surveys and recaptured in the first half of the following fishing season (after 47-247 days at liberty, mean 113 days) were used to develop a growth model. This growth model was not designed to represent all life stages and seasons, but rather to represent likely growth between survey and season. Linear and logarithmic regressions of expected increment on initial length were tested. The two models fitted the available data similarly and predicted similar increments for scallops of 70-120 mm, but the logarithmic fit was selected because it predicted the average growth of smaller scallops (not included in the model) much better (Figure 4). Negative expected average growth was disallowed, but negative growth of some animals is possible given some combinations of expected increment and variability. Variability about the expected increment in the growth model was estimated by regressing the standard deviation of observed increments against initial length, including only size classes with three or more observations. The standard deviation of the expected increment in the growth model was constrained to be at least 0.05 mm. Uncertainty about the expected average growth between survey and season is incorporated by re-sampling (with replacement) from the 129 tag returns, refitting the regressions for the expected increment and its standard deviation and generating a new growth model for each iteration (e.g., Figure 5). For a given size class, the expected weekly increment is multiplied by the number of weeks between survey and season. This assumes that growth is essentially linear between survey and season, and mimics the approach used to calculate the observed weekly increments from the tag return data.

Within each bootstrap iteration, the growth model is applied to each 1 mm size class of the length frequency distribution to generate an expected distribution of sizes at the start of the season. Mortality between survey and season is incorporated at this stage by applying an instantaneous rate of  $M = 0.5 \text{ y}^{-1}$  over the time between survey and season, bootstrapping (parametrically) from a normal distribution with a standard deviation of 6.5% of the expected absolute mortality. The growth transition matrix then sums the expected densities by size class to estimate the start of season length frequency distribution which is, in turn, used to estimate recruited biomass (in greenweight). Confidence limits are estimated using the statistical distribution of 1000 or more bootstrap iterations. Correction for average dredge efficiency, and the variances associated with dredge efficiency, the length-weight regression, and the growth transition matrix are all optional.

![](_page_11_Figure_0.jpeg)

Figure 4: Relation between initial length and subsequent weekly increment for scallops tagged and released April–June (top panel, open circles). The solid and dotted lines are, respectively, linear and logarithmic regressions through these data. Crosses indicate data from scallops with a release length of less than 80 mm. These illustrate the choice between growth models; none was tagged April–June, and they were not used to fit the regressions. The relation between initial length and the standard deviation of the weekly increment is shown in the bottom panel with a fitted linear regression line (length classes with 3 or more observations).

![](_page_11_Figure_2.jpeg)

Figure 5: Development of a growth model (right, lines indicate mean plus and minus 1 and 2 standard deviations) from regressions predicting mean expected increment (left) and the standard deviation of the mean increment (middle). Data from one bootstrap iteration.

The final step in the analysis is the prediction of meatweight from expected start-of-season greenweight. An analysis of recovery of meatweight from greenweight in the Coromandel fishery suggests that average recovery over a season varies widely from about 10% to about 15% (Table 2). This is important because the TACC and ACE are allocated in meatweight. Average recovery is typically about 11% when the season opens in July and typically increases to 15% or more through the first 12–16 weeks of the season (to about late October, Figure 6). In some years the recovery rate in the middle of the season is very high, but in others it remains low. Towards the end of the season, the average recovery rate usually declines again.

Table 2: Estimated average recovery for Coromandel scallop seasons 1995–2002, based on the ratio of actual measured meatweight (reported on the bottom half of CELRs) to estimated greenweight (often reported on the top half of CELRs).

Year	Mean recovery (%)
1995	13.65
1996	13.71
1997	12.88
1999	10.37
2000	9.93
2001	12.54
2002	15.61

![](_page_12_Figure_3.jpeg)

Figure 6: Seasonal progress of weekly average recovery of meatweight from greenweight in the Coromandel fishery based on CELR records. The left panel shows weekly averages and the right panel shows fourth order polynomial regressions to simplify trends (indicative only). The bold line shows the average across all these years.

It is, therefore, difficult to predict the average recovery of meatweight from greenweight for the coming season and the average actually attained will depend on the behaviour of fishers as well as the biological state of the animals. This uncertainty was incorporated by selecting one of the seasonal averages from Table 2 for each bootstrap estimate of start-of-season recruited biomass (in greenweight). This approach assumes that growth, mortality, and recovery of meatweight from greenweight (essentially "condition") are independent. In reality, this is unlikely; poor conditions for growth between survey and season are likely to result in little or no growth, higher than expected mortality, and poor condition. In a "bad" year, therefore, predictions and confidence limits based on averages are likely to be highly optimistic; the "real" biomass available to the fishery may fall outside the confidence bounds of the stock assessment.

#### 10. Results

#### 10.1 Scallop population at the time of the survey (May 2004)

During the survey, 11 267 of 30 822 scallops caught in 100 tows (sweeping  $0.120 \text{ km}^2$ ) were measured. Approximate pooled length frequency distributions corrected for dredge efficiency and scaled to estimated population size (assuming historical average dredge efficiency for each substrate type) are shown for the four major areas in Figure 7. The beds at the Mercury Islands (off Whitianga) and in the Bay of Plenty (off Papamoa and Waihi) had higher proportions of large scallops than those in the Hauraki Gulf and at Little Barrier Island. Only at the Mercury Islands were there reasonable proportions of very small scallops (less than 60 mm shell length).

![](_page_13_Figure_4.jpeg)

Figure 7: Length frequency distributions for major areas (corrected for historical average dredge efficiency), May 2004.

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Using a simple parametric approach to estimation (including simple size-dependent scalars to correct for dredge efficiency), the biomass of scallop of 90 mm shell length or more at the time of the survey was about 1850 t (no correction for dredge efficiency, Table 3a) or 7000 t (corrected using historical average efficiency, Table 3b) with CVs of about 9%. These biomass estimates are probably reasonably reliable, but their uncertainty is grossly underestimated because this simple approach cannot incorporate additional variability associated with dredge efficiency. By far the largest recruited population (and biomass) was found around the Mercury Islands, where 70–73% of the total biomass larger than 90 mm was found (depending on whether a correction for dredge efficiency) are the most conservative interpretation of the survey data possible and might be interpreted as the minimum absolute biomass at the time of the survey.

Table 3a: Estimated density and biomass of scallops 90 mm shell length or more at the time of the survey, assuming 100% dredge efficiency. SEM, standard errors of estimated quantities immediately to the left.

	Area (km <sup>2</sup> )	Sites	Density (m <sup>-2</sup> )	SEM	CV	Millions	Mean wt. (g)	Biomass (kg.m <sup>-2</sup> )	SEM	CV	Biomass (t green)
Mercury	63.2	37	0.2359	0.0234	0.10	14.914	86.5	20.40	1.99	0.10	1 290
Plenty	28.1	18	0.0719	0.0162	0.23	2.024	85.1	6.12	1.39	0.23	172
Barrier	8.6	18	0.1192	0.0506	0.42	1.021	79.0	9.42	4.00	0.42	81
Waiheke	33.2	21	0.1221	0.0329	0.27	4.058	74.6	9.11	2.45	0.27	303
Colville	15.8	6	0.0057	0.0032	0.56	0.090	86.2	0.49	0.29	0.60	8
Fishery	148.9	100	0.1484	0.0130	0.09	22.107	83.8	12.44	1.07	0.09	1 853

Table 3b: Estimated density and biomass of scallops 90 mm shell length or more at the time of the survey, assuming historical average dredge efficiency (but not including variance associated with dredge efficiency). SEM, standard errors of estimated quantities immediately to the left.

	Area (km <sup>2</sup> )	Sites	Density (m <sup>-2</sup> )	SEM	CV	Millions	Mean wt. (g)	Biomass (kg.m <sup>-2</sup> )	SEM	CV	Biomass (t green)
Mercury	63.2	37	0.9698	0.1000	0.10	61.316	82.9	80.42	8.06	0.10	5 085
Plenty	28.1	18	0.2995	0.0675	0.23	8.429	81.8	24.49	5.52	0.23	689
Barrier	8.6	18	0.5602	0.2383	0.43	4.799	76.6	42.94	18.25	0.43	368
Waiheke	33.2	21	0.3296	0.0893	0.27	10.950	74.8	24.64	6.67	0.27	819
Colville	15.8	6	0.0166	0.0099	0.60	0.261	88.1	1.46	0.93	0.64	23
Fishery	148.9	100	0.5758	0.0505	0.09	85.756	81.4	46.89	4.01	0.09	6 983

Historically, the fishery operated to a Minimum Legal Size (MLS) of 100 mm and yield was similarly estimated, so biomass was also estimated using 100 mm as the size at recruitment (Tables 4a and 4b). At the time of the survey, the estimated biomass above 100 mm was 770–2400 t, only about 34–42% of the biomass above 90 mm (in both cases the range covers the alternative treatments of dredge efficiency). About 82% of these larger scallops were estimated to

be within the Mercury Island strata. It is normal for the Mercury Island strata to be an important part of the Coromandel fishery, especially for these relatively large scallops, but this level of dominance is atypical.

	Area (km <sup>2</sup> )	Sites	Density (m <sup>-2</sup> )	SEM	CV	Millions	Mean wt. (g)	Biomass (kg.m <sup>-2</sup> )	SEM	CV	Biomass (t green)
Mercury	63.2	37	0.0992	0.0106	0.11	6.269	101.4	10.06	1.08	0.11	636
Plenty	28.1	18	0.0277	0.0072	0.26	0.781	99.7	2.77	0.73	0.27	78
Barrier	8.6	18	0.0232	0.0119	0.51	0.199	98.4	2.29	1.16	0.51	20
Waiheke	33.2	21	0.0114	0.0042	0.37	0.377	93.1	1.06	0.40	0.37	35
Colville	15.8	6	0.0022	0.0020	0.91	0.035	101.2	0.23	0.21	0.91	4
Fishery	148.9	100	0.0514	0.0048	0.09	7.662	100.8	5.18	0.49	0.10	772

Table 4a: Estimated density and biomass of scallops 100 mm shell length or more at the time of the survey, assuming 100% dredge efficiency. SEM, standard errors of estimated quantities immediately to the left.

Table 4b: Estimated density and biomass of scallops 100 mm shell length or more at the time of the survey, assuming historical average dredge efficiency (but not including variance associated with dredge efficiency). SEM, standard errors of the estimated quantities immediately to the left.

	Area (km <sup>2</sup> )	Sites	Density (m <sup>-2</sup> )	SEM	CV	Millions	Mean wt. (g)	Biomass (kg.m <sup>-2</sup> )	SEM	CV	Biomass (t green)
Mercury	63.2	37	0.3084	0.0328	0.09	19.499	100.6	31.03	3.30	0.09	1 962
Plenty	28.1	18	0.0863	0.0220	0.26	2.429	98.6	8.51	2.22	0.26	240
Barrier	8.6	18	0.0743	0.0383	0.52	0.637	97.5	7.24	3.70	0.51	62
Waiheke	33.2	21	0.0367	0.0137	0.37	1.220	93.3	3.43	1.28	0.37	114
Colville	15.8	6	0.0078	0.0071	0.91	0.123	101.9	0.79	0.72	0.91	12
Fishery	148.9	100	0.1605	0.0150	0.09	23.907	100.0	16.04	1.51	0.09	2 390

A more sophisticated "re-sampling" approach to variance estimation, including variability associated with the dredge efficiency scalars and length-weight regressions, reproduced the point estimates to within 2% but increased the CVs on the fishery-wide density estimates to about 20% and on the fishery-wide biomass estimates to about 19% (for a size at recruitment of 90 mm). The statistical distribution of the biomass estimate is slightly skewed to the right (Figure 8), and estimates of the CV are slightly broader (1 percentage point) when calculated from a fitted normal distribution than when estimated by dividing the 95% confidence range of the bootstraps by 3.92 (1.96 \* 2).

Converting this estimate of greenweight biomass (6 983 t) to meatweight at the time of the survey is not simple because few data are available on the average conversion factor outside the commercial season. However, conversion rates of 12.5% (the official gazetted rate) or 13.5% (historical research data) could be used, suggesting a meatweight biomass of about 870–940 t (with a CV of about 20%) at the time of the survey.

![](_page_16_Figure_0.jpeg)

Figure 8: Frequency distribution of estimated recruited biomass ( $\geq$  90 mm) at the time of the survey. The results of a completely non-parametric re-sampling approach are shown as solid circles with a fitted Gaussian (normal) approximation as a solid line. The dotted line shows the results of the basic analysis (including only sampling variance) from Table 3b. All curves scaled to similar heights for clarity.

#### 10.2 Trends in scallop population since 1992

Discerning trends in the biomass of recruited scallops is complicated by changes to survey coverage, the establishment of closed areas, and uncertainty about dredge efficiency in any particular year. However, some changes have been so large as to transcend this combined uncertainty, and there seems to have been a substantial improvement since the turn of the century (Table 5, although this table may be slightly misleading because it is based on a size at recruitment of 95 mm). The "mainstay" of the fishery at the Mercury Islands has apparently recovered enormously from the lowest recorded abundance of scallops in 2001 (25.1 vs. 1.5 million scallops 95 mm or longer). Chaetopterus tubeworms have become quite rare, and were never a hindrance to surveying (filling the dredge and causing it to "fly"). The biomass at Little Barrier Island has remained similar since 2001 (1-2 million scallops), and the Motiti-Papamoa bed has continued the increase noted in 2003 (2.4 million vs. 0.7 million scallops 95 mm or longer in 2001). Scallops were again very numerous at Waiheke Island in 2004 and, as in 2003, were significantly larger than they were around the turn of the century. The 2004 survey at Colville was quite limited but gave a substantially lower biomass estimate than in 2003 (which was the highest estimated biomass since surveys began there in 1993). The beds at Waihi, off the Katikati Entrance, had an estimated 1 million scallops of 95 mm or more at time of the 2004 survey, more than most recent surveys, but substantially less than the very high biomass estimates in the mid-1990's. Beds at Great Barrier, Otama, and at Shoe and Slipper Islands were not surveyed because they were not expected to contain many scallops. Overall, the 2004 survey gives a considerably higher estimate of total biomass of scallops (95 mm or greater length) than any survey since 1995 (by which time surveys covered most of the fishery). The improvement is greatest, however, at the Mercury Islands.

Table 5: Number of scallops at the time of survey in constituent areas of the Coromandel fishery since 1990 (millions 95 mm or greater shell length assuming historical average dredge efficiency for all years, including 2001–03 when different vessels were used). Totals include data from all surveyed beds and are not directly comparable among years. Dashes (-) indicate no survey in an area or year.

	Mercury		Motiti,	Little	0-1-111-	Waiheke	<b>T</b> 1
	Islands	Waihi	Papamoa	Barrier	Colville	Island	l'otal
1990	7.4	_	_	_	_	6.4	13.8
1991	11.1	_	_	_	_	2.8	13.9
1992	10.7	_			-	0.7	11.4
1993	6.6	7.1	_	_	0.3	0.4	14.4
1994	4.8	1.5	_	_	_	0.0	6.3
1995	4.4	0.6	4.5	2.5	0.1	0.3	12.5
1996	6.1	0.2	2.2	3.3	0.1	0.3	12.6
1997	6.1	0.7	1.9	4.0	0.3	5.4	18.4
1998	6.4	0.1	1.2	1.0	0.2	5.3	14.2
1999	1.8	0.2	0.9	0.2	0.0	0.2	3.3
2000	_	_	_	-	-	_	-
2001	1.5	_	0.7	1.6	_	0.2	4.2
2002	2.7	_	0.7	0.8	-	1.0	5.3
2003	4.2	_	2.1	1.4	3.5	1.7 .	12.9
2004	25.1	1.0	2.4	1.2	0.3	4.5	34.5

## 10.3 Projected scallop population at the start of the season (July 2004)

The simplest (and most precautionary) approach to estimating start-of-season biomass (in greenweight) would be to accept estimates of minimum absolute biomass from Table 3a or 4a (depending on MLS) and allow for about 9% mortality over the 9.4 weeks between survey and season. This approach assumes no growth, natural mortality of M = 0.5 spread evenly throughout the year, and 100% dredge efficiency. Projecting meatweight biomass at the start of the season is complicated by the unpredictability of the average recovery fraction (which depends on fisher behaviour as well as biology), but the most conservative approach would be to adopt the lowest annual average recorded historically (9.93% in 2000). All steps in this calculation (other than the assumed even seasonal spread of M with no possibility of catastrophic mortality) are conservative, so the overall result is probably very conservative. However, it might be considered a "benchmark" against which other approaches can be compared. Based on these assumptions, therefore, the minimum expected biomass at the start of the 2004 season would be about 1700 t greenweight, or about 170 t meatweight (Table 6).

It is almost certainly unrealistic (pessimistic) when estimating start-of-season greenweight to assume 100% dredge efficiency and no growth between survey and season (although it is certainly feasible that natural mortality could exceed the 9% suggested by spreading M = 0.5 evenly throughout the year). Assuming historical average values for dredge efficiency and growth greatly increases the estimate to about 9000 t greenweight (median projected value) with a CV of

about 21% (Table 7, Figure 9), mostly as a result of applying the correction for dredge efficiency. Further, assuming historical average recovery of meatweight from greenweight leads to an estimate of 1130 t meatweight (median projected value) with a CV of about 25% (Table 7, Figure 10). The average weight of a recruited scallop was projected to decline from 83.5 to 80.7 g by the start of the season, largely because of the growth of relatively numerous scallops just under the MLS into the recruited biomass.

Table 6: Projected biomass of scallops 90 mm shell length or more at the start of the season assuming 100% dredge efficiency, no growth, M = 0.5 spread evenly through the year, and lowest historical recovery of meatweight from greenweight.

	Area (km <sup>2</sup> )	Millions	Mean wt. (g)	Biomass (kg.m <sup>-2</sup> )	SEM	CV	Biomass (t green)	Meat-weight (t)
Mercury	63.2	13.624	86.5	18.64	1.82	0.10	1 1 7 8	117
Plenty	28.1	1.842	85.1	5.57	1.26	0.23	157	16
Barrier	8.6	0.929	79.0	8.57	3.64	0.42	74	7
Waiheke	33.2	3.707	74.6	8.32	2.24	0.27	277	28
Colville	15.8	0.082	86.2	0.45	0.26	0.60	7	1
Fishery	148.9	20.195	83.8	11.36	0.98	0.09	1 693	168

![](_page_18_Figure_3.jpeg)

Figure 9: Frequency distribution of projected recruited biomass ( $\geq$  90 mm) at the assumed start of the season in mid-July. The results of a non-parametric re-sampling and projection approach are shown as solid circles with a fitted Gaussian approximation as a grey line. The dotted line shows the results of the basic analysis (including only sampling variance) from Table 3b and the solid line repeats the normal approximation to the biomass estimate at the time of the survey. All curves scaled to similar heights for clarity.

Table 7: Projected biomass of scallops 90 mm shell length or more at the start of the season assuming historical average dredge efficiency at length, average growth (from previous tagging studies), M = 0.5 spread evenly through the year, and average recovery of meatweight from greenweight.

	Area (km <sup>2</sup> )	Millions	Mean wt. (g)	Biomass (t green)	CV	Meat- weight (t)	CV
Mercury	63.2	76.882	82.1	6 308	0.23	791	0.27
Plenty	28.1	9.414	82.4	778	0.29	97	0.34
Barrier	8.6	7.143	76.4	550	0.50	67	0.50
Waiheke	33.2	15.958	75.4	1211	0.48	150	0.51
Colville	15.8	0.376	87.1	33	0.53	4	0.55
Fishery	148.9	111.498	80.7	9 024	0.21	1 131	0.25

![](_page_19_Figure_2.jpeg)

Figure 10: Frequency distribution of projected recruited meatweight ( $\geq 90$  mm) at the assumed start of the season in mid-July. The results of a non-parametric re-sampling and projection approach are shown as solid circles with a fitted Gaussian approximation as a grey line. The solid line repeats the normal approximation to the biomass estimate at the time of the survey, converted from greenweight to meatweight using the gazetted factor of 12.5%. The dotted line shows a very pessimistic approach assuming 100% dredge efficiency, no growth between survey and season, M = 0.5 spread evenly through the year, and lowest historical recovery of meatweight from greenweight. All curves scaled to similar heights for clarity.

For comparison with earlier work (including the calculation of Provisional Yield), the same assessment and projection model was applied using a MLS of 100 mm shell length, assuming historical average dredge efficiency at length, growth, and recovery of meatweight from

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greenweight. The start-of-season biomass at 100 mm was predicted to be 2767 t greenweight with a CV of 21% (29% of the biomass above 90 mm shell length) and the start-of-season meatweight at 100 mm was predicted to be 345 t with a CV of 26%.

#### 10.4 Sensitivity of biomass estimates to exclusion of areas of low scallop density

Most recent assessments have explored the sensitivity of biomass estimates to exclusion of areas of low scallop density, and it has generally been accepted that  $0.04 \text{ m}^{-2}$  (one recruited scallop for each 25 m<sup>2</sup> of seabed) is a reasonable working definition for the limit of acceptable fishing. Working at a station level, therefore, the biomass estimates at the time of the survey were recalculated assuming that all stations where scallops were scarcer than  $0.04 \text{ m}^{-2}$  had zero density, and stations where scallops were denser than  $0.04 \text{ m}^{-2}$  had a density of the actual density minus  $0.04 \text{ m}^{-2}$ . These corrections were applied before any scaling for dredge efficiency, so they are conservative.

Excluding areas of low density (<  $0.04 \text{ m}^{-2}$ ) reduced the biomass estimates at the time of the survey by about 20% (Figure 11); excluding areas where the density was less than  $0.08 \text{ m}^{-2}$  reduced the biomass estimates by almost 40% (although  $0.08 \text{ m}^{-2}$  is a high density by historical standards).

![](_page_20_Figure_4.jpeg)

Figure 11: Effect of excluding areas of low scallop density on biomass estimates. For increasing minimum acceptable densities (un-scaled for dredge efficiency) the median biomass estimates (solid line, closed circles) are given together with their quartiles (dashed lines) and 95% confidence range (dotted lines).

#### 10.5 Yield estimates

#### 10.5.1 Reference rates of fishing mortality

Yield estimates are generally calculated using reference rates of fishing mortality applied in some way to an estimate of current or reference biomass. However, the choice among reference rates is not simple. It is probably useful to use Caddy's (1998) notation of target reference points (TRP)

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and limit reference points (LRP) where reference points can be measures of fishing mortality (F) or biomass (B).  $F_{max}$  (the zenith of a yield-per-recruit curve) was the classical TRP (Caddy 1998), but may be too high as a target (Annala et al. 2002) and has more recently been generally regarded as an LRP (a "threshold" beyond which fishing mortality should not be increased) and  $F_{0.1}$  has been substituted as a "target" (Caddy 1998). Mace (1994) recommended  $F_{40\%}$  as a "default" TRP in the "common situation where there is adequate information to place bounds on all relevant life history parameters except those characterising the stock-recruit relationship" (although Collie & Gislason, 2001, list  $F_{40\%}$  as a LRP). Mace's (1994) simulations showed that, in her model,  $F_{40\%}$  was similar to  $F_{0.1}$  when recruitment and maturity schedules coincided, and approximated  $F_{MSY}$  for fisheries of average to high resilience to fishing. She also affirmed Mace & Sissenwine's (1993) proposition that the default overfishing threshold should be 20% B<sub>0</sub> for stocks of at least average resilience to fishing. However, Myers et al. (1994) considered this threshold (20% of virgin stock size) the least desirable and the least precautionary of those they examined.

Cryer & Morrison's (1997) study of the incidental effects of scallop dredges in the Coromandel fishery allowed the estimation of  $F_{max}$  as a LRP, and  $F_{0.1}$  and  $F_{40\%}$  as TRPs. They estimated  $F_{40\%}$  (0.514 y<sup>-1</sup>),  $F_{0.1}$  (0.508 y<sup>-1</sup>), and  $F_{max}$  (0.650 y<sup>-1</sup>) (all assuming M = 0.50 y<sup>-1</sup>). Because of the derivation of these estimates, they should be applied to the modified version of the Baranov equation given by Cryer & Morrison:

$$CAY = \frac{F_{ref}}{F_{ref} + \frac{5M}{12}} * \left[ 1 - e^{-(F_{ref} + 5M_{12})} \right] * B_{jul}$$

where  $B_{jul}$  is the estimate of recruited biomass in July. In this formulation of the Baranov equation, natural mortality is assumed to act in tandem with fishing mortality for the first 5 months of the year, the length of the current Coromandel commercial scallop season. Cryer & Morrison (1997) derived their estimates of F<sub>0.1</sub> by selecting the better fitting of related domed and asymptotic two-parameter curves:

$$YPR = \alpha . F_{apparent} . e^{-\beta . F_{apparent}}$$

and

$$YPR = \frac{\alpha . F_{apparent}}{\beta + F_{apparent}}$$

where YPR is the yield (catch) for each recruiting scallop of 1 mm shell length, and  $\alpha$  and  $\beta$  are constants defining the shape of the curve of YPR against F<sub>apparent</sub>. This first formulation generates a domed curve and is based on Ricker's (1954) stock-recruit curve. The second formulation generates an asymptotic curve and is based on the Beverton & Holt's (1957) stock-recruit curve. Recent work (Cryer et al. 2003a) has indicated that a more flexible three parameter formulation based on Deriso's (1980) generalized stock-recruitment curve (see also Schnute 1985) provides much better fits than either of the two parameter curves, so these curves were fitted to Cryer & Morrison's (1997) model outputs and the YPR curves and reference rates of fishing mortality were recalculated:

$$YPR = \alpha F_{apparent} \left( 1 - \beta \gamma F_{apparent} \right)^{1/\gamma}$$

where  $\gamma$  is a parameter that affects the degree of doming of the curve (allowing infinite variation between the Ricker and Beverton-Holt curves, both of which are special cases of the Deriso curve). Cryer & Morrison's (1997) approach of weighting each data point by the inverse of  $F_{apparent}$  was repeated to improve the fit to the ascending limb and, hence, the reference fishing mortality rate  $F_{0.1}$ . Refitting the model results in this way suggested a slightly higher  $F_{max}$  (0.66 *vs.* 0.65) and a lower  $F_{0.1}$  (0.43 *vs.* 0.51) than reported by Cryer & Morrison (1997) (Table 8, Figure 12). There was no change to  $F_{40\%}$  Because the form of the fitted curve was not changed.

Table 8: Estimates of reference rates of fishing mortality, yield per recruit at  $F_{max}$  and  $F_{0.1}$ , and predicted egg production (as a percentage of unfished egg production) for models including no incidental effects, incidental effects on growth and mortality of adults, and a link between fishing effort and juvenile mortality acting over the specified time. MLS = Minimum legal size, h/g = high-grading.

MLS	F <sub>max</sub>	F <sub>0.1</sub>	F <sub>40%</sub>	F <sub>25%</sub>	YPR <sub>max</sub>	YPR <sub>0.1</sub>	E <sub>Fmax</sub>	$E_{F0.1}$
No incidental off	oto							
100 Incluentar ene	4 270	1.071			2405	2001	54 (0/	66 30/
100	4.372	1.071	-	-	2405	2001	54.6%	66.2%
95	3.311	1.065		_	2818	2423	45.9%	57.6%
90	3.536	0.993	1.775	_	3161	2672	36.7%	49.3%
90 h/g	1.867	0.916	1.788	_	2941	2676	39.6%	50.3%
85	2.134	0.933	1.099	_	3285	2941	31.7%	43.4%
80	1.665	0.847	0.801	2.210	3382	3094	27.0%	38.7%
Effects on adults								
100	0.683	0.434	0.845	1.646	1024	967	46.2%	59.4%
95	0.705	0.437	0.650	1.188	1258	1184	37.6%	51.9%
90	0.659	0.431	0.514	0.875	1459	1383	32.5%	45.6%
90 h/g	0.425	0.309	0.380	0.596	1180	1136	36.2%	47.2%
85	0.555	0.385	0.420	0.658	1562	1494	30.6%	43.1%
80	0.469	0.352	0.347	0.517	1597	1544	28.7%	39.5%
Effects on adults	and juvenile	es						
None (90)	0.659	0.431	0.514	0.875	1459	1383	32.5%	45.6%
l wk	0.516	0.391	0.509	0.850	1276	1236	32.4%	46.2%
2 wk	0.471	0.368	0.497	0.766	970	943	32.7%	45.8%
3 wk	0.441	0.348	0.471	0.719	746	727	33.0%	45.9%
4 wk	0.431	0.340	0.454	0.692	583	567	32.6%	45.5%
6 wk	0.360	0.283	0.380	0.579	338	329	32.6%	45.7%
8 wk	0.349	0.274	0.356	0.543	212	207	31.7%	44.6%

The modelling conducted by Cryer & Morrison (1997) considered only direct effects (on mortality, growth, and condition) on essentially adult scallops. However, Talman et al. (2004) investigated indirect effects on juvenile scallops *via* the relationships among fishing effort, habitat "heterogeneity" and juvenile mortality. They estimated juvenile mortality at four sites of widely differing scallop dredging history using tethering experiments. At two sites that had never been dredged commercially, they estimated mortality rates of 15 and 24% per week and found high habitat heterogeneity. At two sites that are dredged routinely for scallops, they estimated mortality rates of 39 and 59% per week and found low habitat heterogeneity. The relationship

between fishing and habitat heterogeneity is not new (see Thrush et al. 1998, 2001 for two New Zealand studies in soft-sediment systems), but this is the first study of the implications for scallops. There may be problems with mortality rates estimated using tethered animals but, taking these results at face value (although applying them relatively conservatively), Cryer et al. (2003a) modelled the consequences for their yield-per-recruit curves of additional juvenile mortality that increased with increasing fishing effort. They found that the YPR curves became progressively more dome-shaped and both  $F_{max}$  and  $F_{0.1}$  decreased as the period over which higher juvenile mortality was assumed to act was increased (Figure 13).

![](_page_23_Figure_1.jpeg)

Figure 12 (after Cryer et al. 2003a): Yield per recruit curves based on Cryer & Morrison's Individual-Based Model outputs and Deriso's three-parameter curve. Incidental effects of the dredge method on adult scallops are included as indicated.

Our knowledge of the life history and mortality schedules of juvenile scallops is still rudimentary, but the modeling conducted by Cryer et al. (2003) indicates that reference rates of fishing mortality are sensitive to the indirect effects of fishing as well as to the more frequently studied direct effects on growth and incidental mortality. Both  $F_{max}$  and  $F_{0.1}$  declined when indirect effects were included in a model, and the consequence of fishing beyond  $F_{max}$  became progressively worse as the influence of the indirect effects were increased in the model. Disturbing though these results might be, they still exclude indirect effects of fishing on recruitment. Scallop spat are known to require some sort of structure to settle (hence the success of the artificial enhancement programme in the Nelson-Marlborough fishery), but the nature of the relationship between habitat structure (heterogeneity) and spatfall remains poorly understood.

Recruitment of scallops could be affected in two ways: through sub-lethal effects of disturbance on adults as they develop gonads prior to spawning (e.g., Chícharo et al. 2003 for *Spisula solida*),

or through the destruction and removal of foliose material necessary as spat settlement surfaces (e.g., Harvey and Bourget 1995, Harvey et al. 1995). These latter changes could be incremental, as fishing activity gradually decreases the amount of foliose or highly structured material in the environment and, thereby, gradually depresses recruitment. There are many examples of declining scallop stocks in the literature (Dredge 1989, Pezzuto and Borzone 1997, Wilbur et al. 1999, Dichmont et al. 2000) and landings in the Coromandél fishery declined from >1000 t in the 1980s and early 1990s to <200 t in the early years of this century. Simplistic stock assessment models including no incidental effects of fishing could not have predicted this from landings over this time.

![](_page_24_Figure_1.jpeg)

Figure 13 (after Cryer et al. 2003a): Yield per recruit curves based on Cryer & Morrison's Individual-Based Model outputs and Deriso's three-parameter curve. The top curve includes no incidental effects, the second curve is a repeat of the bottom curve in Figure 11, and the remaining curve come from models including a link between average fishing effort and juvenile mortality that acts for the indicated time.

Walters & Martell (2002) reviewed some of the dramatic changes that occurred in fisheries stock assessment during the 1990s. They discussed how  $F_{0.1}$  from a basic YPR analysis excluding any incidental effects was generally considered a conservative target in the early 1990s, and often led to rates of fishing mortality of roughly F = M. However, inclusion of mean recruitment effects (e.g., Walters & Martell 2002), Patterson's (1992) broad analysis of pelagic fish stocks, and Myers & Mertz's (1998) and Myers et al.'s (1999) work on stock-recruit relationships, all contradicted this supposition. Walters & Martell (2002) recommend that any stock assessment that suggests an optimum rate of fishing mortality much greater than half of the rate of natural mortality (M = 0.5 yr<sup>-1</sup> in our case) would need to be very carefully justified. Our lowest estimates of  $F_{max}$  (and, indeed,  $F_{0.1}$ ) including the incidental effects of dredging on adults, juveniles, and habitat, are still higher than 0.5M, even though we have not (and cannot) account for recruitment effects using the current "per-recruit" model structure. (Similarly, the agestructured models incorporating mean recruitment effects discussed by Walters & Martell (2002) do not explicitly include the incidental effects of their fishing methods, and they too may be optimistic.)

#### 10.5.2 Estimation of Maximum Constant Yield (MCY)

MCY is not normally estimated for scallops and, given the highly variable nature of most wild scallop fisheries, is likely to be close to zero. Cryer et al. (2003b) showed that constant catch strategies for scallops produced lower yield at much higher biological risk than strategies wherein catch was varied as biomass varied.

#### 10.5.3 Estimation of Current Annual Yield (CAY)

#### **Pessimistic outlook**

The most pessimistic outlook from the 2004 survey (assuming no growth, natural mortality of M = 0.5 spread evenly throughout the year, 100% dredge efficiency, and the lowest recorded annual average meatweight recovery of 9.93%) leads to a start of season biomass of 1693 t greenweight or 168 t meatweight (see Table 6). Based on these calculations, CAY was calculated using each of the reference fishing mortality rates  $F_{0.1}$ ,  $F_{max}$ , and  $F_{40\%}$  as follows:

For F <sub>40%</sub> (0.514),	CAY = 0.7116 * 0.5144 * 1693 (168) = 620 t (green) or 61 t (meat)
For F <sub>0.1</sub> (0.431),	CAY = 0.6741 * 0.4724 * 1693 (168) = 539 t (green) or 53 t (meat)
For F <sub>max</sub> (0.683),	CAY = 0.7666 * 0.5898 * 1693 (168) = 765 t (green) or 76 t (meat)

These estimates of CAY would have CVs at least as large as that of the estimate of start-ofseason recruited biomass (20–25%) but are based on very conservative assumptions about dredge efficiency, growth, and expected recovery of meatweight from greenweight, and relate to the surveyed beds only. Excluding areas of low density would reduce these estimates by no more than 20% for a criterion of 0.04 m<sup>-2</sup>. The level of risk to the putative Coromandel scallop stock of fishing at the estimated CAY level cannot be determined but would be low.

#### Average outlook, excluding habitat effects

The recruited biomass of scallops 90 mm in shell length or greater in the Coromandel fishery was predicted to be 9024 t (greenweight) and 1131 t (meatweight) in 2004 (see Table 7). CAY was calculated using these biomass estimates and each of the reference fishing mortality rates  $F_{0.1}$ ,  $F_{max}$ , and  $F_{40\%}$  as follows:

For $F_{40\%}$ (0.514),	CAY = 0.7116 * 0.5144 * 9024 (1131) = 3303 t (green) or 414 t (meat)
For $F_{0.1}$ (0.431),	CAY = 0.6741 * 0.4724 * 9024 (1131) = 2874 t (green) or 360 t (meat)
For F <sub>max</sub> (0.683),	CAY = 0.7666 * 0.5898 * 9024 (1131) = 4080 t (green) or 511 t (meat)

These estimates of CAY would have a CVs at least as large as that of the estimate of start-ofseason recruited biomass (20–25%), are sensitive to assumptions about dredge efficiency, growth, and expected recovery of meatweight from greenweight, and relate to the surveyed beds only. Excluding areas of low density would reduce these estimates by no more than 20% for a criterion of 0.04 m<sup>-2</sup>. The level of risk to the putative Coromandel scallop stock of fishing at the estimated CAY level cannot be determined.

#### Average outlook, including putative habitat effects

Cryer et al. (2003a) modelled the "feedback" effects of habitat modification by the dredge method on juvenile mortality in scallops. They developed estimates of  $F_{0.1}$ , and  $F_{max}$  that incorporated such effects, but had to make assumptions about the duration of what they called the "critical phase" of juvenile growth during which scallops were susceptible to increased mortality. To give some guidance on the possible impact of including "recruitment" effects on yield estimates, Cryer et al.'s (2003a) estimates of  $F_{0.1} = 0.274$ ,  $F_{max} = 0.349$ , and  $F_{40\%} = 0.356$  are applied here:

For F <sub>40%</sub> (0.356),	CAY = 0.6308 * 0.4313 * 9024 (1131) = 2455 t (green) or 308 t (meat)
For $F_{0.1}$ (0.274),	CAY = 0.5681 * 0.3827 * 9024 (1131) = 1962 t (green) or 246 t (meat)
For F <sub>max</sub> (0.349),	CAY = 0.6262 * 0.4272 * 9024 (1131) = 2414 t (green) or 303 t (meat)

These estimates of CAY would have CVs at least as large as that of the estimate of start-ofseason recruited biomass (20–25%), are sensitive to assumptions about dredge efficiency, growth, and expected recovery of meatweight from greenweight, to the duration of any habitatmediated increase in juvenile mortality, and relate to the surveyed beds only. Excluding areas of low density would reduce these estimates by no more than 20% for a criterion of 0.04 m<sup>-2</sup>. The level of risk to the putative Coromandel scallop stock of fishing at the estimated CAY level cannot be determined.

#### **10.5.4 Estimation of Provisional Yield**

Provisional Yield (PY) (Cryer 1994) is estimated as the lower limit of a 95% confidence distribution for the estimate of start-of-season recruited biomass, plus an amount to account for beds not surveyed before the season. The amount added for unsurveyed beds is estimated as the product of the variability factor (Annala et al. 2002; M > 0.35, c = 0.6 for scallops) and the historical average landings from the unsurveyed beds. PY is estimated only for comparison with historical estimates of yield.

For Coromandel scallops in 2004, start-of-season recruited biomass (100 mm or greater) was estimated at 2767 t (greenweight, CV = 21%) or 345 t (meat weight, CV = 26%), giving lower tails to the respective 95% confidence distributions of 1605 t (greenweight) and 166 t (meatweight). Beds not included in the surveys were excluded specifically because they were

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thought unlikely to support commercial fishing in 2004, so no addition for un-surveyed beds was made:

$$PY = 1605 (166) t + 0.6 * 0 t = 1605 t (greenweight) or 166 t (meat)$$

These estimates of PY would have CVs at least as large as that of the estimates of start-of-season recruited biomass at 100 mm shell length (21–26%), and are sensitive to assumptions about dredge efficiency and growth. The sensitivity to excluding areas of low density has not been formally estimated, but the reduction could be as high as 40% for a criterion of 0.04 m<sup>-2</sup> (100 mm scallops are considerably rarer than 90 mm scallops so a minimum density of 0.04 m<sup>-2</sup> is much less frequently exceeded). The level of risk to the putative Coromandel scallop stock of fishing at the estimated PY level cannot be determined.

#### 11. Conclusions:

- 1. A survey of 100 valid dredge stations in the Coromandel scallop fishery was completed in mid-May 2004.
- 2. Absolute start-of-season biomass over 100 mm shell length (for comparison with previous surveys) was predicted to be 2767 t greenweight (345 t meatweight) with CVs of about 21% and 26%, respectively. These estimates are sensitive to assumptions about dredge efficiency, growth between survey and season, and exclusion of areas of low scallop density.
- 3. Absolute start-of-season biomass over 90 mm shell length (the commercial minimum legal size) was predicted to be 9024 t greenweight (1131 t meatweight) with CVs of about 21% and 25%, respectively. These estimates are sensitive to assumptions about dredge efficiency, growth between survey and season, and exclusion of areas of low scallop density.
- 4. These results suggest a substantial improvement, especially in the Mercury Islands beds, since 2003 (which was itself an improvement on the very poor years between 1999 and 2002).
- 5. Yield estimates (as CAY) for the 2004 season vary widely depending on assumptions about dredge efficiency, growth between survey and season, exclusion of areas of low scallop density, and, for estimates in meatweight, recovery of meat from greenweight. The most pessimistic estimates suggest a meatweight yield of about 50 t, but assuming average values for important assumed variables increases this to about 300–350 t. Incorporating habitat effects on juvenile mortality reduces the CAY estimates to about 200–250 t meatweight.

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#### 12. Publications:

There are no other publications except a Research Progress Report (essentially an earlier version of this report with the same authors that was circulated to the Shellfish Fishery Assessment Working Group for comments in June 2004), and a Voyage Programme and Voyage Report summarising the fieldwork.

## 13. Data Storage:

Data have been transferred to the appropriate Empress database, *scallop*. Various analytical files in MS-Excel are held on a secure, backed-up server at NIWA Auckland and copied to the Ministry's data manager.

Stratum	Location / description	Area (m <sup>2</sup> )	Method	2004 Shots
1.2	Mercury (Black Jack)	12 609 192	Dredge (sand)	5
2	Mercury (Three Mile Bank)	16 745 117	Dredge (sand)	7
3	Mercury (Opito Bay)	5 393 677	Dredge (sand)	4
3.5	Mercury (Opito Bay deeps)	13 923 828	Dredge (sand)	4
4	Mercury (Mercury Cove)	1 713 959	Dredge (silt)	5
7	Mercury (Bumper Cove)	8 970 459	Dredge (sand)	8
8	Mercury (3MB deeps)	3 869 141	Dredge (sand)	4
11	Waihi (Katikati Entrance)	15 767 090	Dredge (sand)	4
13	Motiti South	7 808 594	Dredge (sand)	12
14	Papamoa Beach	4 572 998	Dredge (sand)	*2
18	Little Barrier West	3 771 729	Dredge (sand)	8
19	Little Barrier South	4 795 044	Dredge (sand)	10
20	Waiheke Island	33 222 534	Dredge (silt)	21
31	Colville South (shallow)	15 770 264	Dredge (silt)	6
Total		148 933 626	_	100

Appendix	1: Stratum	definitions	and	station	allocations,	Coromandel	scallop	survey	2004.	*,	data	for	five
additional	tows lost.												

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Appendix 2: Summary of dredge efficiency experiments used in the historical analysis and revised analytical approach. L'Aries and Kataraina were of similar size and displacement, had identical motors, reduction gear, and propellers, and deployed the same dredge between 1992 and 1999.

						Dredge	Depth			Dredge			Dive	Density
Code	Year	Vessel	Site	Fishery	Substrate	width (m)	(m)	Tows	Area	Scallops	Sites	Area	Scallops	>=85 mm
1984CP	1984	L'Aries	Pakatoa	Coro	Silt/mud	1.55	23	11	3 950	1 037	3	150	238	0.6867
1985CT	1985	L'Aries	Tarahiki	Coro	Silt/mud	1.90	15	16	4 607	2 615	7	67	147	1.5528
1986CT	1986	Elco	Tarahiki	Coro	Silt/mud	1.30	15	20	4 090	4 129	12	108	238	1.3609
1992C3	1992	L'Aries	3 Mile Bank	Coro	Sand/shell	2.00	17	4	7 445	938	15	628	350	0.4711
1993CA	1993	L'Aries	Opito A	Coro	Sand/shell	2.00	16	4	2 963	90	16	314	80	0.1560
1993CB	1993	L'Aries	Opito B	Coro	Sand/shell	2.00	27	4	3 148	920	8	314	435	1.1618
1993CC	1993	L'Aries	Opito C	Coro	Sand/shell	2.00	30	2	1 148	224	4	314	206	0.4806
1994CA	1994	L'Aries	Opito A	Coro	Sand/shell	2.00	16	2	3 704	193	4	314	47	0.1273
1994CB	1994	L'Aries	Opito B	Coro	Sand/shell	2.00	27	2	3 704	125	4	314	12	0.0222
1994CC	1994	L'Aries	Opito C	Coro	Sand/shell	2.00	30	2	1 926	208	4	314	97	0.1432
1995CC	1995	Kataraina	Opito C	Coro	Sand/shell	2.00	27	2	5 556	446	4	647	127	0.0971
1995CA	1995	Kataraina	Opito A	Coro	Sand/shell	2.00	17	2	5 556	217	4	647	210	0.2702
1995CB	1995	Kataraina	Opito B	Coro	Sand/shell	2.00	26	3	3 395	423	4	245	154	0.1959
1995C3	1995	Kataraina	3 Mile Bank	Coro	Sand/shell	2.00	17	3	3 297	83	4	1 181	206	0.1019
1995CN	1995	Kataraina	Opito N	Coro	Sand/shell	2.00	20	3	2 963	189	2	1 329	304	0.1398
1995CS	1995	Kataraina	Opito S	Coro	Sand/shell	2.00	17	3	2 926	93	9	1 731	286	0.1251
1996CB	1996	Kataraina	Opito B	Coro	Sand/shell	2.00	25	3	4 889	870	8	481	193	0.2609
1996CA	1996	Kataraina	Opito A	Coro	Sand/shell	2.00	12	3	5 556	499	9	569	232	0.4010
1996CC	1996	Kataraina	Opito C	Coro	Sand/shell	2.00	28	3	2 290	610	4	358	170	0.3763
1997CB	1997	Kataraina	Opito B	Coro	Sand/shell	2.00	27	3	6116	752	· 4	525	120	0.1960
1997CA	1997	Kataraina	Opito A	Coro	Sand/shell	2.00	15	3	7 408	186	4	804	12	0.0112
1997CC	1997	Kataraina	Opito C	Coro	Sand/shell	2.00	29	4	2 929	822	4	402	175	0.2857
1998CH	1 <b>998</b>	Kataraina	Hooks Bay	Çoro	Silt/mud	2.00	20	4	3 593	825	4	1 002	506	0.6888
1999CT	1999	Kataraina	Tarahiki	Coro	Silt/mud	2.00	23	4	5 371	377	4	1 431	547	0.0820

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