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Biomass surveys and stock assessments for the
Coromandel and Northland scallop fisheries, 2005

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EXECUTIVE SUMMARY

Cryer, M.; Parkinson, D.M. (2006). Biomass surveys and stock assessments for the Coromandel and Northland scallop fisheries, 2005. *New Zealand Fisheries Assessment Report 2006/34*. 53 p.

A survey of 105 valid dredge stations in the Coromandel scallop fishery was completed in mid May 2005. Absolute start-of-season biomass over 100 mm shell length (for comparison with previous surveys) was predicted to be 5080 t greenweight (630 t meatweight) with c.v.s of about 22% 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. Absolute start-of-season biomass over 90 mm shell length (the commercial minimum legal size) was predicted to be 14 370 t greenweight (1800 t meatweight) (median projected values) with c.v.s of about 23% and 27%, respectively. These estimates are sensitive to assumptions about dredge efficiency, growth between survey and season, and exclusion of areas of low scallop density. These results suggest a continuation of the substantial improvement, especially in the Mercury Islands beds, observed since 2003 (at which time some improvement on the very poor years between 1999 and 2002 had already been observed). Yield estimates (as CAY) for the 2005 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 90–110 t, but assuming average values for important assumed variables increases this to about 570–660 t. Incorporating habitat effects on juvenile mortality reduces the CAY estimates to about 390–490 t meatweight.

A survey of 140 valid dredge stations in the Northland scallop fishery was completed in May-June 2005. Absolute start-of-season biomass over 100 mm shell length (the commercial minimum legal size) was predicted to be 5565 t greenweight (753 t meatweight) (median projected values) with poorly defined and highly skewed statistical distributions. These estimates are sensitive to assumptions about dredge efficiency, growth between survey and season, and exclusion of areas of low scallop density. These results suggest a very substantial improvement in Bream Bay (as predicted by fishers), a biomass close to average in Rangaunu Bay, and relatively low recruited biomass in Spirits Bay, Tom Bowling Bay, at Mangawhai, and along Pakiri Beach. Yield estimates (as CAY) for the 2005 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 118 t, but assuming average values for important assumed variables increases this to about 280 t. The incidental effects of habitat modification on juvenile mortality could not be incorporated in the Northland stock assessment. Substantial uncertainty stemming from assumptions about dredge efficiency during the survey and predicting the average recovery of meatweight from greenweight during the 2005 season remain in this stock assessment. Future research should include some means of minimising the effects of this uncertainty.

1. INTRODUCTION

1.1 Overview

This report summarises research and fishery information for scallops (*Pecten novaezelandiae*) in the Coromandel and Northland fisheries. Biomass surveys undertaken in May-June 2005 are summarised and yield estimates for 2005 are derived using methods after Annala et al. (2004). In each section of this document, information for the Coromandel fishery will be presented first, followed by information for the Northland fishery. This is because the Coromandel fishery developed first and most survey and stock assessment developments have started there first. This work was funded by the Ministry of Fisheries under projects SCA2004/01 and SCA2004/02.

1.2 Description of the fisheries

Scallops support regionally important commercial fisheries and an intense non-commercial interest off the northeast coast of the North Island. Both the Northland (SCA 1) and Coromandel (SCA CS) commercial fisheries are managed under the Quota Management System (QMS); the two are divided by a line from Cape Rodney to the northernmost tip of Great Barrier Island (Figure 1). A wide variety of effort controls and daily catch limits have been imposed in the past, but both fisheries have been limited by explicit seasonal catch limits specified in meatweight (processed weight, being the adductor muscle plus attached roe) since the early to late 1990s. Some additional controls remain on dredge size, fishing hours, and (in the Coromandel fishery) non-fishing days. Catch and catch rates from both fisheries are variable both within and among years (Tables 1 and 2), a characteristic of scallop fisheries worldwide (Shumway & Sandifer 1991).

All commercial fishing is by dredge and self-tipping “box” dredges are preferred to the ring bag designs in common use in southern fisheries. Fishing in the Northland fishery is within discrete beds in Spirits Bay, Tom Bowling Bay, Great Exhibition Bay, Rangaunu Bay, Doubtless Bay, Stephenson Island, the Cavalli Passage, Bream Bay, the coast between Mangawhai and Pakiri Beach, and to the north of Little Barrier Island. In the Coromandel fishery, the main beds are found north of Whitianga (at the Mercury Islands), east of Waiheke Island, to the west and south of Little Barrier Island, to the west of Great Barrier Island, and in the Bay of Plenty (principally off Waihi, and around Motiti and Slipper Islands). Recreational and Maori customary fishing is undertaken in suitable areas throughout both fisheries, more especially in enclosed bays and harbours, many of which are closed to commercial fishing.

The minimum legal size (MLS) for scallops for commercial and amateur fishers throughout both fisheries was 100 mm (shell length) until 1995. Starting with the 1995 commercial season in July 1995, the MLS for scallops taken commercially from the Coromandel fishery was reduced to 90 mm as part of a package of measures which also included further voluntary closed areas (VCAs) and reduced commercial catch limits. This package was introduced to address concerns expressed by all user groups over the impact of scallop dredging on juvenile scallops. The MLS has remained at 100 mm for all non-commercial fishers and for commercial fishers in the Northland fishery.

1.3 Literature review

General descriptions of the biology of the New Zealand scallop, *Pecten novaezelandiae*, were given by Bull (1988) and Cryer (1994), and little new information on the biology has become available subsequently other than an unpublished PhD thesis by Morrison (1998) and a thesis (Williams 2005) and some papers on spawning ecology by Williams & Babcock (2004a, 2004, 2005). The New Zealand scallop is one of several species of “fan shell” bivalve molluscs found in New Zealand

waters. They have a characteristic round shell with a flat upper valve and a deeply concave lower valve. Scallops inhabit waters to about 60 m deep (to 85 m in the Chatham Islands), but are more common in the Coromandel fishery in depths of 10 to 30 m and in the Northland fishery in depths of 20 to 50 m. Growth rates are spatially and temporally variable; growth to 100 mm takes between 1.5 and 3.5 years. The maximum age of scallops in unexploited populations is thought to be about 6 or 7 years.

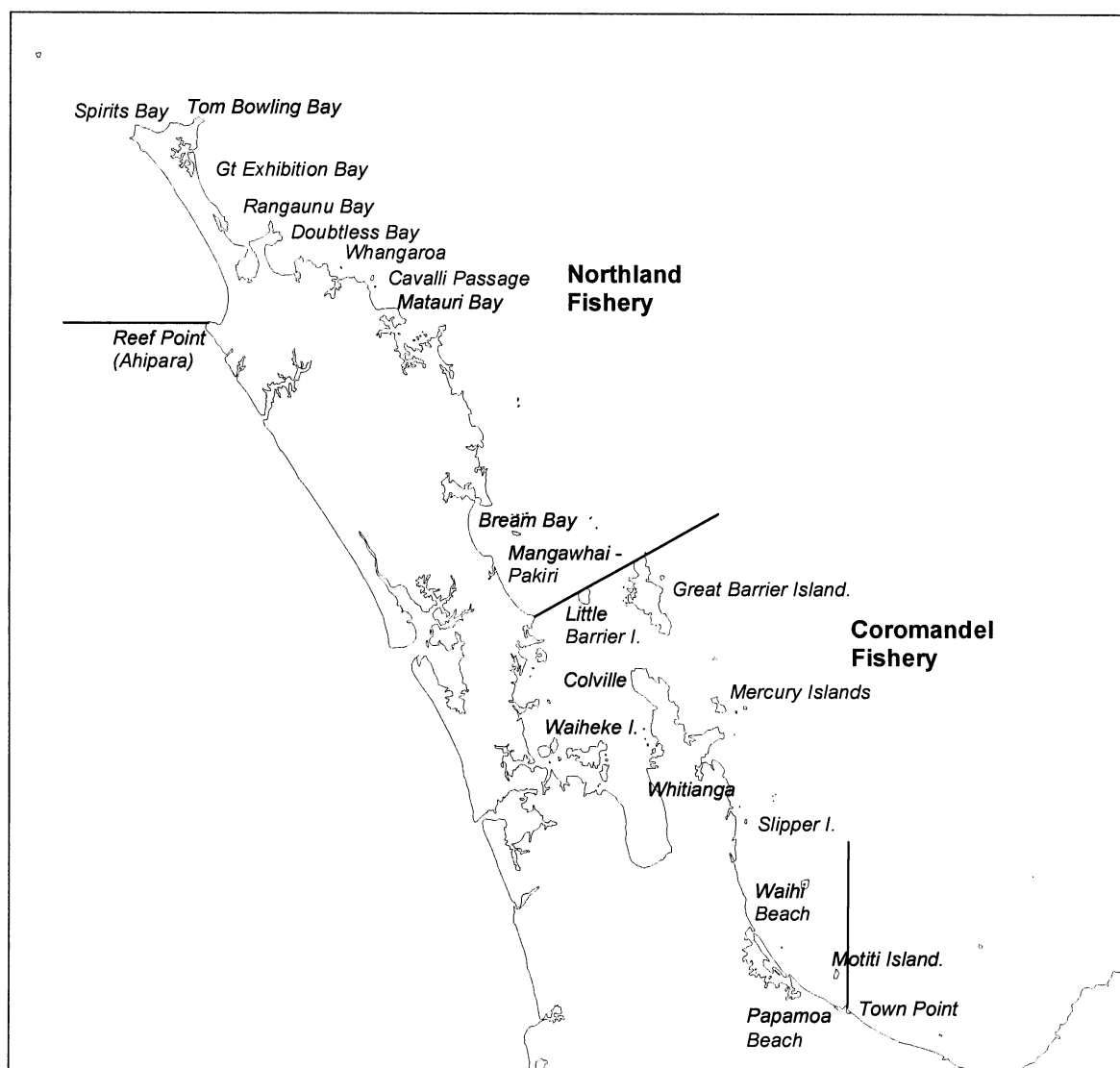


Figure 1: Geographic distribution of the two northern scallop fisheries and locations mentioned in the text.

P. novaezelandiae is hermaphroditic; each individual carries both male and female gonads at the same time. Most individuals are sexually mature at about 60–70 mm shell length (see Williams (2005) and Williams & Babcock (2005) for a comprehensive treatment in the Hauraki Gulf), although larger individuals have disproportionately larger gonads. They are extremely fecund and can spawn several times each year (Williams & Babcock 2004, 2004b), although not all spawning events lead to successful spat settlement. Larval development lasts for about 3 weeks, depending on water temperature. Initial spat settlement is by byssus thread attachment to some surface free of sediment (shell hashes, hydroids, spat bags, etc.). The characteristic scallop shell does not develop until a few days after the spat loses the byssus thread and settles to the seabed.

Scallops grow rapidly (albeit with considerable variation), have high natural mortality ($M \sim 0.50 \text{ y}^{-1}$), and exhibit variable recruitment. Such a life history results in fluctuating biomass, catch, and CPUE in most fisheries for scallops, and reliance on relatively few year-classes (Caddy & Gulland 1983, Orensanz et al. 1991, Shumway & Sandifer 1991). New Zealand stocks are not extreme examples, but Cryer (1994) examined data from 1978 to 1992 and found that recruited biomass in the Coromandel fishery could not be predicted from historical biomass estimates, nor even from the biomass in the previous year together with estimates of intervening removals by commercial fishing.

Simulation modelling by Cryer et al. (2003a) suggested that strategies that vary catch in proportion to biomass (constant-F strategies) should outperform constant catch strategies. This is not surprising, but constant-F strategies provide about 30% more catch at higher catch rates with lower biological risk. “Tuning” the exploitation rate (especially to conservative levels) and setting it to zero at low biomass both decreased biological risk. Conversely, maintaining a “base TACC” (the current management strategy) increased biological risk. Full cost-benefit analysis was not undertaken but, over the long run, the additional model catch available from a constant-F strategy had a much higher value than the cost of the necessary surveys, and there were additional benefits in terms of higher average catch rates and lower biological risk. Rotational fishing provided good levels of catch at relatively low biological risk, but needed high rates of exploitation in the open areas. These high rates of extraction might not be economically sensible (because of low catch rates as biomass declines during a year), or environmentally sustainable (because of reduced habitat structure).

2. REVIEW OF THE FISHERIES

2.1 TACCs, catch, landings, and effort data

The Coromandel and Northland scallop fisheries are managed under the QMS using individual transferable quotas (ITQ) that are proportions of the Total Allowable Commercial Catch (TACC). Landings are shown in Tables 1 and 2. Both fisheries have been gazetted on the Second Schedule of the Fisheries Act 1996 which specifies that, for certain “highly variable” stocks, the Annual Catch Entitlement (ACE) can be increased within a season. The TACC is not changed by this process and the ACE reverts to the level of the TACC at the end of each season. Seasons run from 15 July to 14 February (Northland fishery) or 21 December (Coromandel fishery). Catch rates are variable both within and among seasons, but the relationship between biomass and CPUE is complex and (declines in) CPUE cannot be used to estimate biomass within a season (Cryer 2001b). Effort data are, therefore, not presented.

2.2 Other information

Incidental mortality caused by commercial scallop dredges was estimated in 1996–97 (Cryer & Morrison 1997). Individual-based modelling and stochastic yield-per-recruit (YPR) analysis suggest that neither the 100 mm MLS in force in Northland nor the Provisional Yield (PY) method of estimating yield were optimal (for maximising long-term average landings).

2.3 Recreational and Maori customary fisheries

There is an intense amateur interest in scallops throughout the Coromandel and Northland fisheries. Amateurs usually dive or use small dredges, but, in some circumstances, they collect scallops by hand from the intertidal zone. To some extent, management of northern scallop fisheries has concentrated on spatial separation of commercial and amateur fisheries through the closure of harbours and

enclosed waters to commercial dredging. There remain, however, areas of contention and conflict, some of which have been addressed using additional voluntary or regulated closures.

Recreational catch in 1993–94 from the area shared with the Northland commercial fishery (Bradford 1997) was 40–60 t (green weight). Commercial landings from the Northland fishery in the most comparable period (July 1994 to February 1995 scallop season) were about 1300 t, suggesting that, in that year, the recreational catch of scallops was probably less than 5% of total removals. Estimates of catch by recreational fishers in 1993–94 (Bradford 1997) are 60–70 t from the area shared with the Coromandel commercial fishery. Commercial landings from the Coromandel controlled fishery in the most comparable period (July to December 1994 scallop season) were about 300 t, suggesting that, in that year, the recreational catch of scallops was about 20% of total removals.

Table 1: Catch limits and landings (t greenweight or meatweight) from the Coromandel fishery since 1974. Data before 1986 from Fisheries Statistics Unit (FSU) forms. Landed figures come from the landed section of the Catch Effort and Landing Returns (CELR) form and from Licensed Fish Receiver Returns (LFRR), whereas “Estimated” figures come from the CELR effort section and are pro-rated to sum to the CELR landed greenweight. “Hauraki” = 2X and 2W, “Mercury” = 2L and 2K, “Barrier” = 2R, 2S, and 2Q, “Plenty” = 2A–2I. Seasonal catch limits (since 1992) are specified in meatweight (“Green” assumes the gazetted conversion factor of 12.5% and probably overestimates the actual greenweight taken in most years). * 1991 landings include about 400 t from Colville.

Season	Catch limits		Landings (t)			Estimated catch (t green)			
	Meat	“Green”	LFRR Meat	CELR Meat	CELR Green	Hauraki	Mercury	Barrier	Plenty
1974	—	—	—	—	26	0	26	0	0
1975	—	—	—	—	76	0	76	0	0
1976	—	—	—	—	112	0	98	0	14
1977	—	—	—	—	710	0	574	0	136
1978	—	—	—	—	961	164	729	3	65
1979	—	—	—	—	790	282	362	51	91
1980	—	—	—	—	1 005	249	690	23	77
1981	—	—	—	—	1 170	332	743	41	72
1982	—	—	—	—	1 050	687	385	49	80
1983	—	—	—	—	1 553	687	715	120	31
1984	—	—	—	—	1 123	524	525	62	12
1985	—	—	—	—	877	518	277	82	0
1986	—	—	162	—	1 035	135	576	305	19
1987	—	—	384	—	1 431	676	556	136	62
1988	—	—	182	—	1 167	19	911	234	3
1989	—	—	104	—	360	24	253	95	1
1990	—	—	153	—	903	98	691	114	0
1991	—	—	203	—	1 392	*472	822	98	0
1992	154	1 232	147	—	901	67	686	68	76
1993	132	1 056	62	—	455	11	229	60	149
1994	66	528	49	—	323	17	139	48	119
1995	86	686	88	79	574	25	323	176	50
1996	88	704	81	80	594	25	359	193	18
1997	105	840	94	89	679	26	473	165	15
1998	110	880	37	19	204	1	199	2	1
1999	31	248	8	7	47	0	12	17	18
2000	15	123	7	10	70	0	24	2	44
2001	22	176	22	20	161	1	63	85	12
2002	35	280	32	31	204	0	79	12	112
2003	58	464	58	56	451	63	153	13	223
2004	78	624	78	78	624	27	333	27	237

2.4 Other sources of fishing mortality

Quantitative information is available on the incidental impacts on scallop growth and mortality of encounters with commercial dredges of several designs (Cryer & Morrison 1997). Individual-based population modelling and yield-per-recruit analysis strongly suggest that incidental effects, especially on mortality rates, greatly affect yield from scallop dredge fisheries (Cryer & Morrison 1997). Despite the high incidental mortality rates associated with the current box dredge, this design was found to be the best of the three tested (for a MLS of 85 mm or more) in terms of yield-per-recruit, largely as a result of its higher catching efficiency compared with the ring-bag and Japanese “Keta Ami” designs. This work suggested that the current MLS of 90 mm was close to optimal (for maximising long-term average landings) for the Coromandel fishery (Cryer & Morrison 1997).

Table 2: Catch limits and landings (t greenweight or meatweight) from the Northland fishery since 1980. Data before 1986 from Fisheries Statistics Unit (FSU) forms. Landed figures come from the landed section of the Catch Effort and Landing (CELR) form and from Licensed Fish Receiver Returns (LFRR), whereas “Estimated” figures come from the CELR effort section and are pro-rated to sum to the CELR landed greenweight. “Whangarei” includes beds south of Cape Brett, “Far North” includes beds from Cape Brett to North Cape, and “Spirits Bay” includes beds to the west of North Cape. Seasonal catch limits (since 1997) are specified in meatweight (“Green” assumes the gazetted conversion factor of 12.5% and probably overestimates the real greenweight taken in most years). *, split by area not available; a, b, voluntary catch limits in seasons starting in 1999 and 2000 were 40 and 30 t, respectively.

Season	Catch limits		Landings (t)			Estimated catch (t green)		
			LFRR	CELR				
	Meat	“Green”	Meat	Meat	Green	Whangarei	Far North	Spirits
1980	—	—	—	—	238	*	*	*
1981	—	—	—	—	560	*	*	*
1982	—	—	—	—	790	*	*	*
1983	—	—	—	—	1 171	78	1 093	—
1984	—	—	—	—	541	183	358	—
1985	—	—	—	—	343	214	129	—
1986	—	—	114	—	675	583	92	—
1987	—	—	183	—	1 625	985	640	—
1988	—	—	171	—	1 121	1 071	50	—
1989	—	—	164	—	781	131	650	—
1990	—	—	115	—	519	341	178	—
1991	—	—	158	—	854	599	255	—
1992	—	—	135	—	741	447	294	—
1993	—	—	114	—	862	75	787	1
1994	—	—	205	—	1 634	429	1 064	142
1995	—	—	208	214	1 469	160	810	499
1996	188	1 504	129	132	954	55	387	512
1997	188	1 504	136	126	877	22	378	477
1998	106	848	31	32	233	0	102	130
1999	106	785	18	18	132	0	109	23
2000	60	444	17	17	128	0	88	40
2001	40	320	38	38	291	14	143	134
2002	40	320	40	40	296	42	145	109
2003	40	320	39	39	309	11	228	70
2004	40	320	40	40	319	206	77	37

3. RESEARCH

3.1 Stock structure

Little is known of the stock structure of New Zealand scallops. It is currently assumed for management that the Coromandel fishery is separate from the adjacent Northland fishery and from the various west coast harbours, Golden Bay, Tasman Bay, Marlborough Sounds, Stewart Island, and Chatham Island fisheries.

3.2 Resource surveys

3.2.1 Survey design and field methods

The choice of an appropriate time for surveys entails balancing the conflicting pressures of operational ease and uncertainty in the results. Early surveys (March–April) 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 (June–July) can be hampered by short working days and less favourable conditions, and the danger of surveys 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.

The Coromandel survey was conducted between 12 and 19 May 2005 (Cryer & Parkinson 2005a). All sampling 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 11 strata: Waiheke Island (one stratum), Colville (one stratum), Little Barrier Island (two strata), Mercury Islands (four strata, with the western boundary extended substantially towards Kennedy Bay), 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 2005 and areas closed to commercial fishing, the total sampled area in 2005 was 174 km², compared with 149 km² in 2004, 130 km² in 2003, 119 km² in 2002, 125 km² in 2001, and 253–341 km² between 1996 and 1999 (Figure 2, Appendix 1).

The Northland survey was conducted between 20 May and 11 June 2005 (Cryer & Parkinson 2005b). All sampling was undertaken by dredge and no diving to estimate dredge efficiency was done. We used the same vessel and skipper as used in most recent surveys, but we have no historical estimates of dredge efficiency for this vessel. Single phase stratified random sampling was undertaken in nine strata: Spirits Bay (one stratum), Tom Bowling Bay (one stratum), Rangaunu Bay (two strata), Bream Bay (two strata), and the coast between Bream Tail and Cape Rodney (Mangawhai-Pakiri, three strata). After excluding strata thought unlikely to be productive in 2005 and areas closed to commercial fishing, the total sampled area in 2005 was 379 km², compared with 490 km² in 2003, 369 km² in 2002, 403 km² in 2001 and 553–714 km² between 1996 and 98 (Figure 3, Appendix 2).

Neither survey was formally optimised to minimise the predicted c.v. of the estimate of recruited biomass for two main reasons. First, some strata were added or redesigned for the 2005 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 in Bream Bay or around 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.

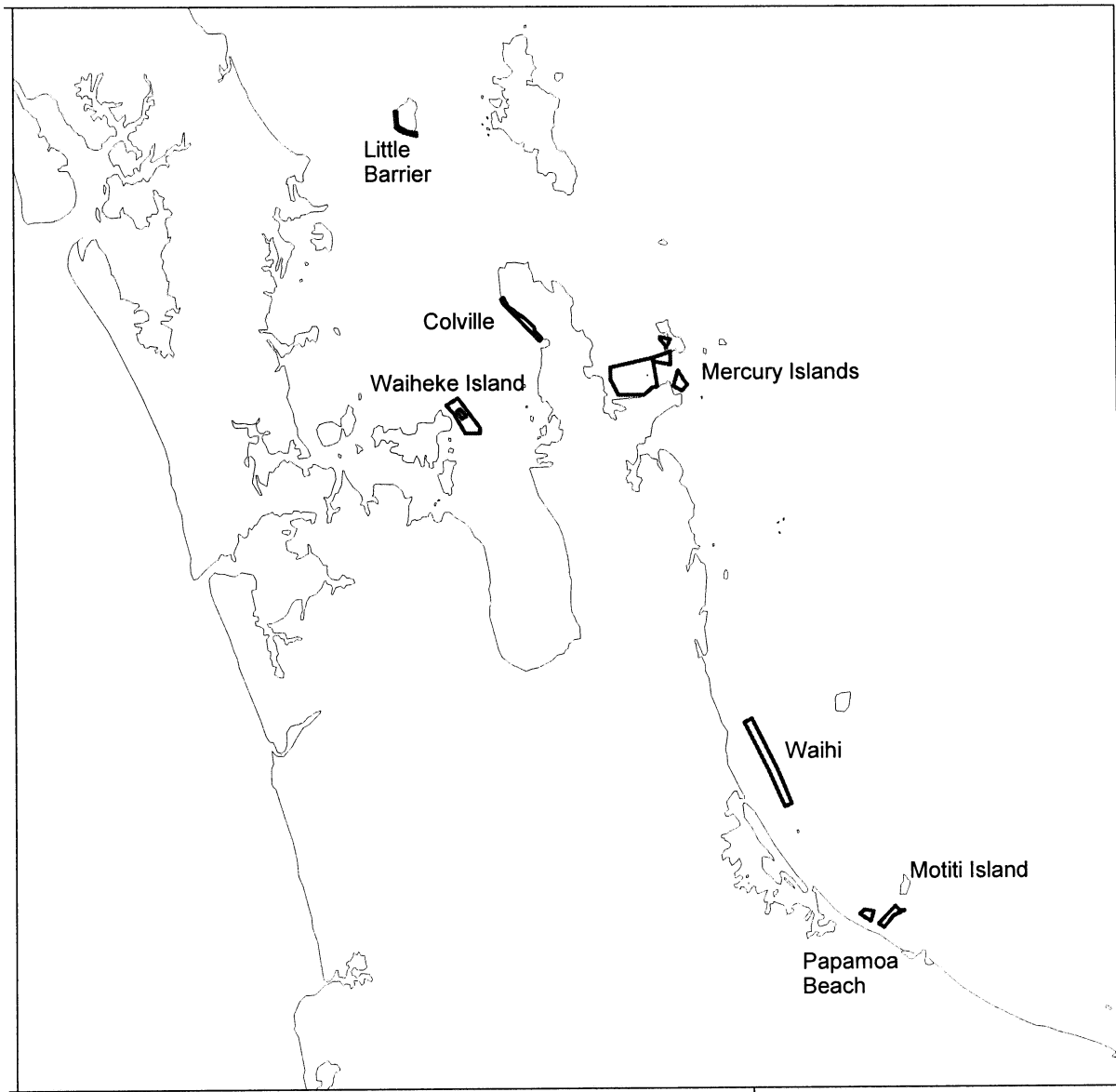


Figure 2: Location of strata for the survey of the Coromandel scallop fishery in 2005. Groups of strata are labelled with geographic descriptions used in the text (see Appendix 1 for details and stratum areas).

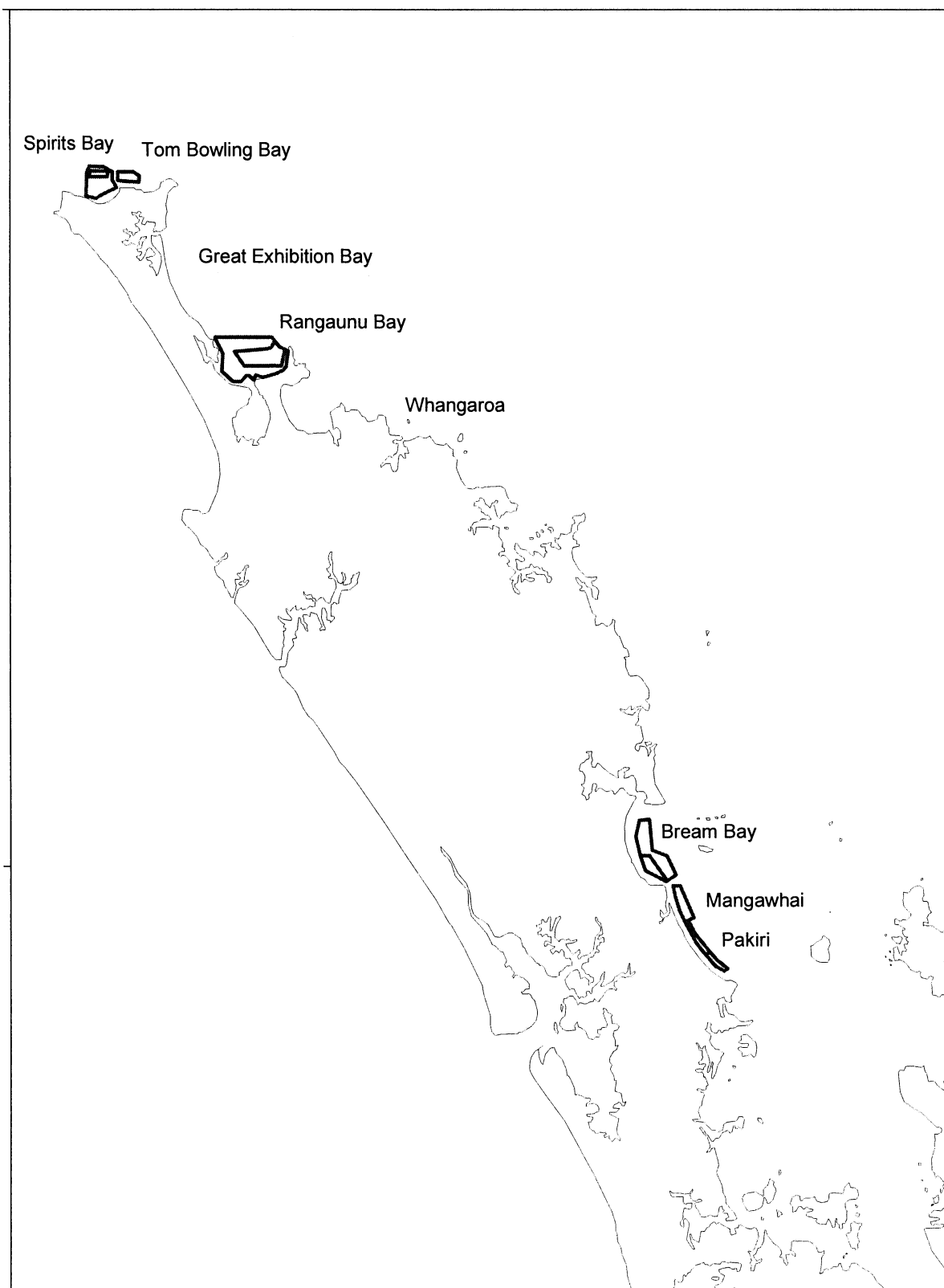


Figure 3: Location of strata for the survey of the Northland scallop fishery in 2005. Strata or groups of strata are labelled with geographic descriptions used in the text (see Appendix 2 for details and stratum areas).

Dredging in the Coromandel survey was undertaken from the chartered commercial dredge vessel *Kataraina* (whose dredge has a width of 2.0 m for which considerable historical information on dredge efficiency exists). Dredging in the Northland survey was undertaken from the chartered commercial dredge vessel *Kamahl* (whose dredge has a width of 8ft (2.44 m)). Unfortunately, although this vessel has been used several times for recent surveys, no information on dredge efficiency exists. The skippers were instructed to tune their gear (select course, speed, warp length, etc.) so as to maximise the total catch at that station. Tows were nominally 0.3–0.5 nautical miles (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.

3.2.2 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 (less than 5%) by all dredge designs, and efficiency for very large scallops sometimes to be low compared with scallops close to MLS. Chapman et al. (1977) 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) and Dare & Palmer (1993) 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 stock assessments since the 1980s and estimates of efficiency close to the MLS have varied between about 10% and about 100% (e.g., Cryer & Morrison 1997; Cryer & Parkinson 1999b).

All strata in both 2005 surveys were sampled by dredge and, therefore, required correction for sampling efficiency. In the Northland fishery, few data are available from a variety of different survey vessels, and most of the beds are found on relatively coarse substrates, so a single pooled estimate of

dredge efficiency has been used. Conversely, in the Coromandel fishery, separate estimates of dredge efficiency have historically been used for sandy and silty or muddy substrates, although relatively fewer data have been collected on silty substrates. Most previous estimates of dredge efficiency and selectivity in the Coromandel fishery have been made using the two very similar vessels, *L'Aries* and *Kataraina*, and the same skipper. Because information for the Coromandel fishery was so much better, the analysis was first done for that fishery then extended to Northland. Data were collated from 32 experiments (19 on sand, 5 on silt or mud, 8 in Northland, Table 3) 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 3: Summary statistics (means with ranges) of dredge efficiency experiments used to describe scallop dredge efficiency by size class.

	Northland	Coromandel sand	Coromandel silt
No. of experiments	8	19	5
No. of vessels	4	2	3
Mean depth	20.6 17–23	22.3 12–30	19.2 15–23
Mean number of dredge tows	6.1 3–10	2.9 2–4	11.0 4–20
Mean area swept by dredge	11 424 3 467–21 957	4 048 1 148–7 445	4 322 3 593–5 371
Mean no. scallops caught by dredge	664 146–1 193	415 83–938	1 797 377–4 129
Mean number of dive sites	10.3 7–16	4.8 2–9	10.6 3–16
Mean area swept by divers	1 977 1 128–3 217	602 245–1 731	552 67–1 431
Mean no. scallops caught by divers	218 27–477	180 12–435	335 147–547
Mean density of scallops >85 mm	0.083 0.014–0.183	0.264 0.011–1.162	0.874 0.082–1.543

Each experiment in the Coromandel fishery consisted of 2–20 dredge tows (mean 5.0) and 2–16 dive sites (mean 7.1). For Northland, each experiment consisted of 3–10 dredge tows (mean 6.1) and 7–16 dive sites (mean 10.3). 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 different 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 ∞ must also be ∞ (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 4).

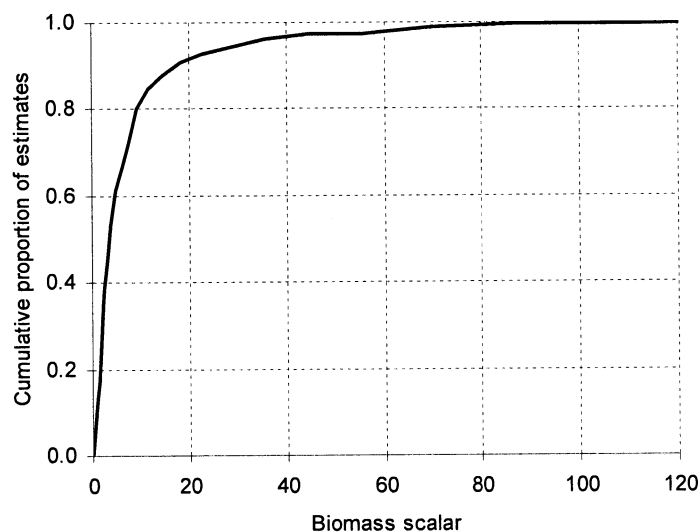


Figure 4: Cumulative frequency distribution of the estimated biomass scalar for the base non-parametric analysis. This distribution was used to establish 100 as the default biomass scalar when estimated dredge efficiency for a size class in an experiment was zero.

Averaging of biomass scalars was done for sandy and silty substrates separately. Each mean scalar was calculated unweighted (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 in both fisheries and some of the recruited biomass in the Coromandel fishery), 100–114 mm (the bulk of the recruited biomass), and 115 mm and over (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.

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 5). 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). This approach is likely to underestimate the density of large, heavy scallops and, consequently, probably underestimates biomass slightly.

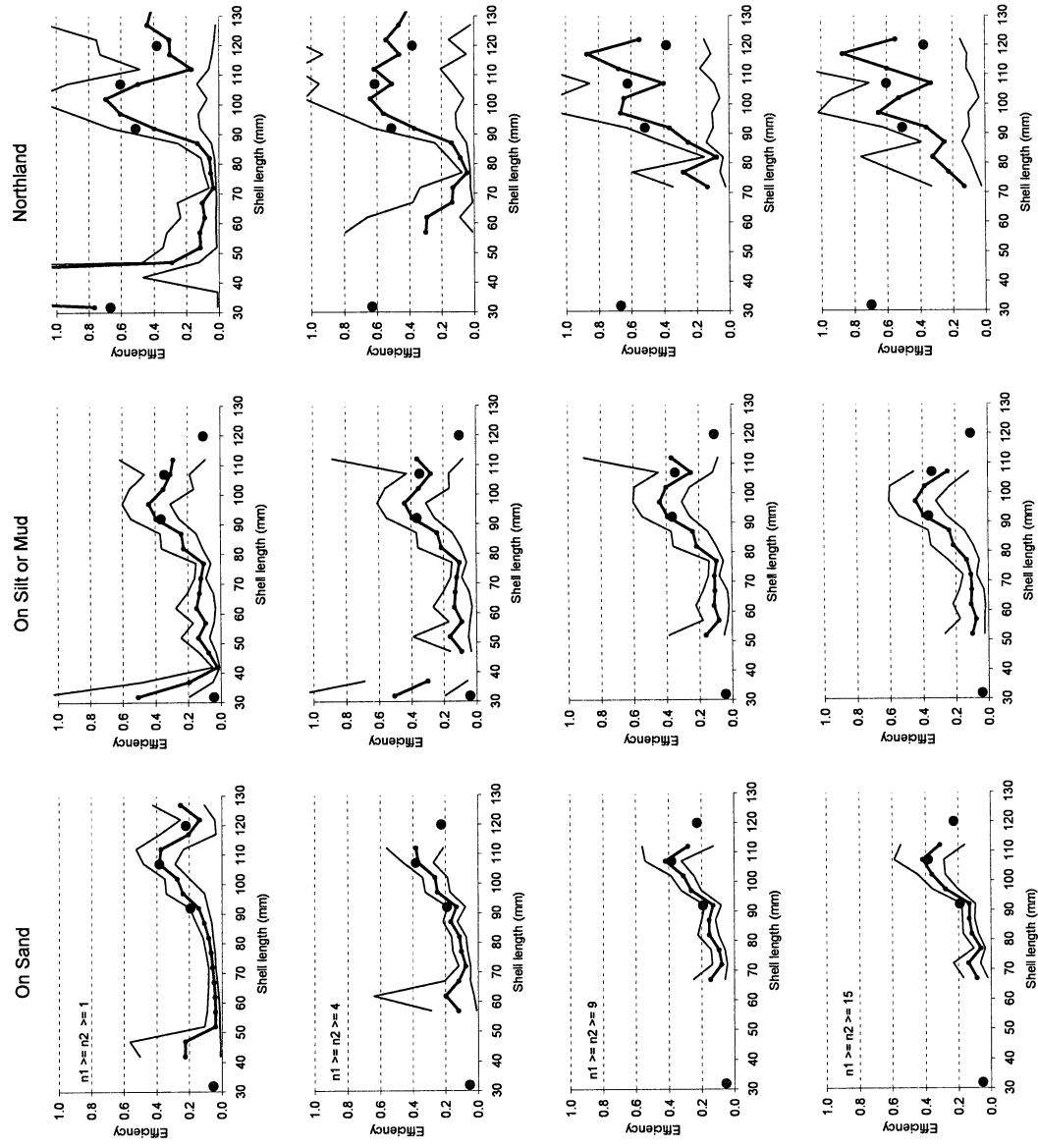


Figure 5: Mean dredge efficiency curves by 5 mm size classes in the Coromandel fishery (left two) and Northland fishery (right) estimated using the log (minimum N) weighting procedure and constraints on the minimum acceptable count by method increasing from 1 to 15 scallops in a size class. In each plot,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

3.2.2 Estimating biomass at the time of survey and the start of the season

The analytical approach to estimating start-of-season recruited biomass for scallops was developed during the 2002 and 2003 stock assessments (e.g., 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. 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 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 by applying 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 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 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 various pre-season surveys (mostly in the Coromandel fishery) between 1990 and 1997 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. It is very important to note that 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 well 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 6). 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 7).

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 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, total weight of the animal including shell). 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.

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 from about 10% to about 15% (Table 4). 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 mid-late October, Figure 8). 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.

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 4 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.

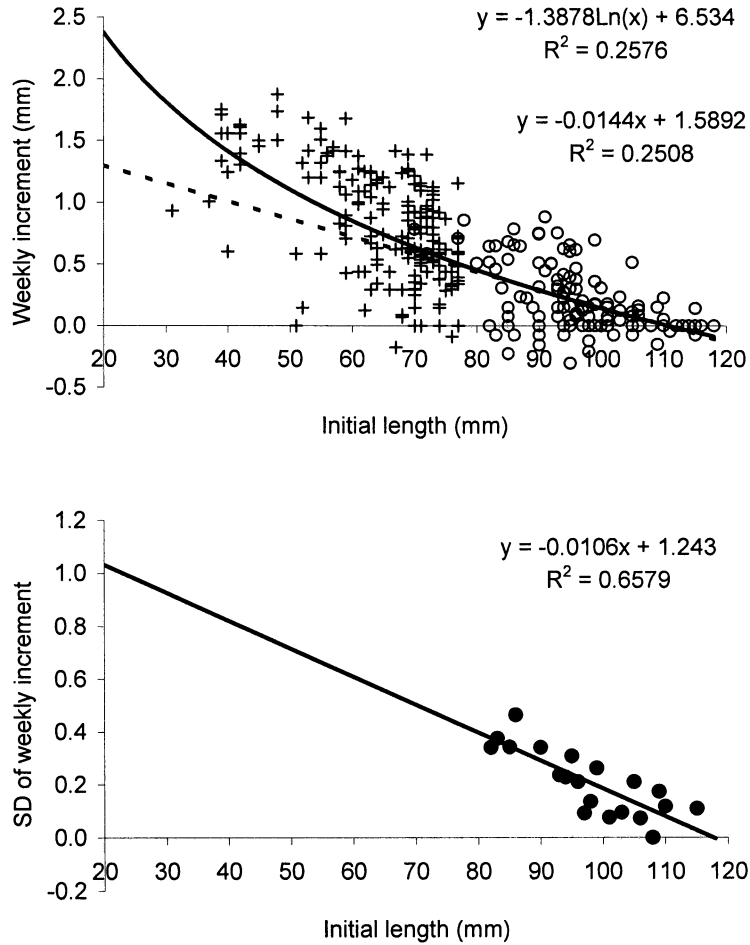


Figure 6: Relationship between initial length and subsequent weekly increment in length for scallops tagged during April, May, or June (top panel, open circles). The dotted and solid lines are, respectively, linear and logarithmic regressions through these data. Crosses indicate data from scallops with a release length of less than 80 mm. These are shown only to illustrate the choice between growth models; none was tagged during in April, May, or June, and none was used to fit the regression lines. The relationship between initial length and the estimated standard deviation of the weekly increment is shown in the bottom panel with a linear least squares regression line (fitted to length classes with 3 or more observations).

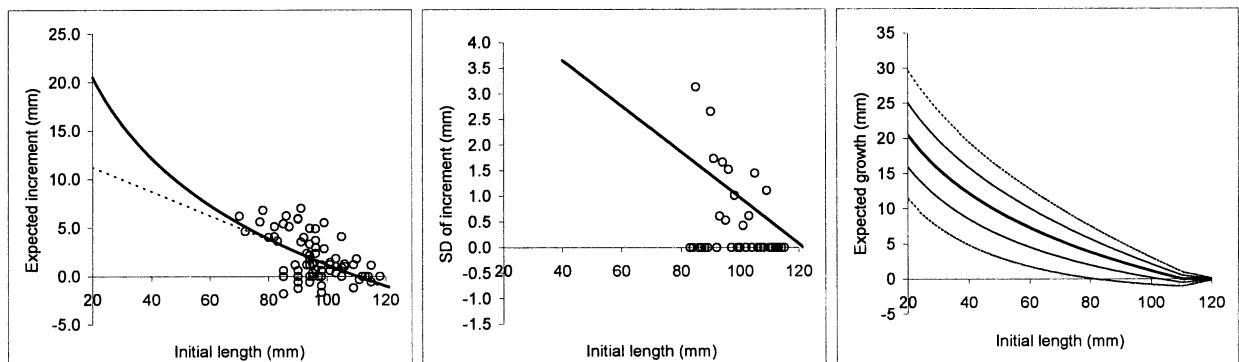


Figure 7: Development of a growth model (right) from regressions predicting mean expected increment (left) and the standard deviation of the mean increment (middle). Data from one bootstrap iteration.

Table 4: Estimated average recovery for Coromandel and Northland 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), screened to remove extreme outliers (recovery <5% or >30%).

Year	Coromandel (%)	Northland (%)
1995	13.7	14.6
1996	13.7	13.9
1997	12.9	14.5
1999	10.4	11.1
2000	9.9	12.5
2001	12.5	12.9
2002	15.6	11.3

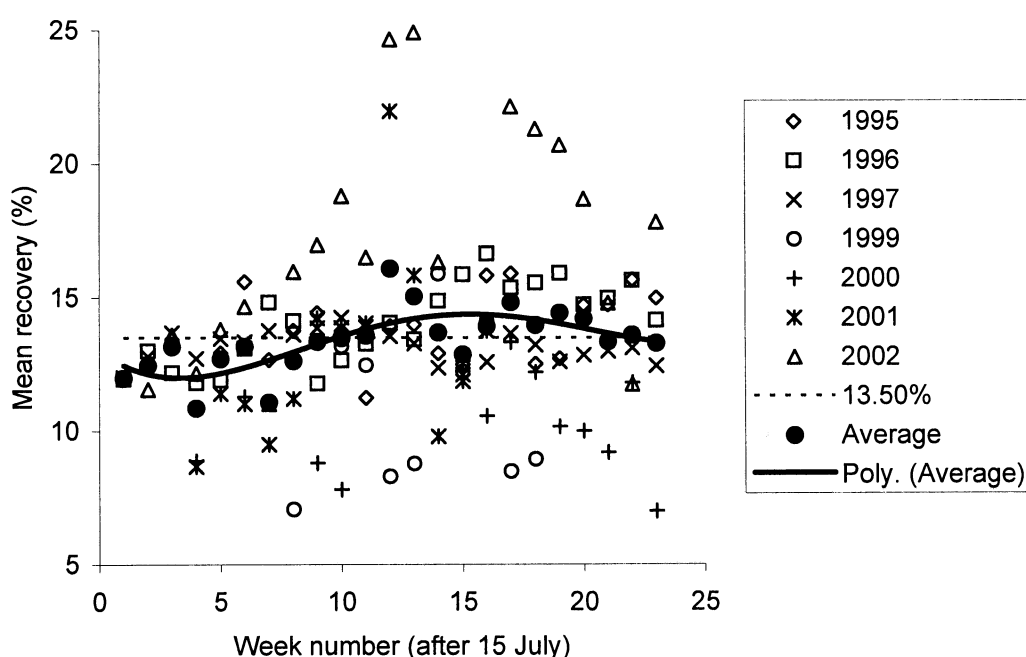


Figure 8: Seasonal progress of weekly average recovery of meatweight from greenweight in the Coromandel fishery based on CELR records when estimated greenweight and actual meatweight were recorded. The solid dots show the average across all these years and the bold line shows an indicative fourth order polynomial regression.

3.2.4 2005 survey results

3.2.4.1 Coromandel fishery at the time of the survey

During the Coromandel survey, 17 156 of 41 291 scallops caught in 105 tows (sweeping 0.131 km²) 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 9. The beds at the Mercury Islands (to the north of Whitianga), in the Bay of Plenty (off Motiti Island, Papamoa Beach, and Waihi), and at Little Barrier Island had relatively higher proportions of large scallops than those in the Hauraki Gulf.

However, there were reasonable proportions of small scallops (less than 70 mm shell length) in all beds, suggesting good recruitment for 2006.

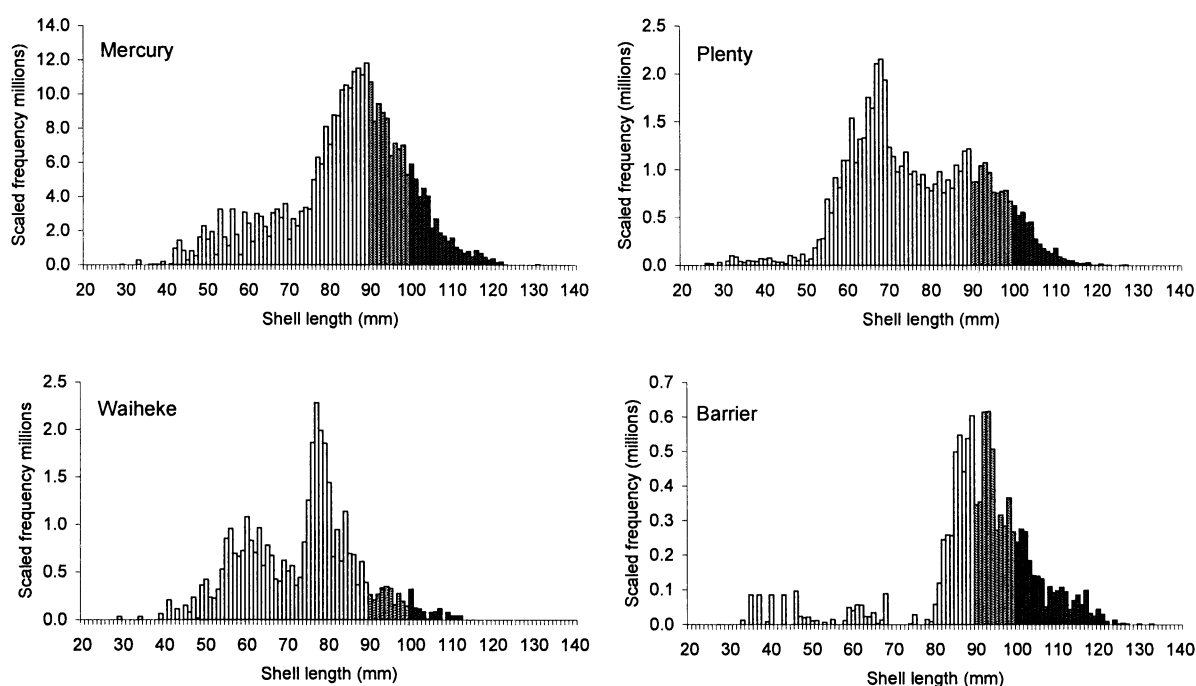


Figure 9: Length frequency distributions for major areas in the Coromandel fishery (corrected for historical average dredge efficiency), May 2005. Shaded bars show scallops larger than 90 mm and dark bars show scallops larger than 100 mm.

Using a simple parametric approach to estimation (including simple size-dependent scalars to correct for dredge efficiency), the biomass of scallops of 90 mm shell length or more at the time of the survey was about 3240 t (no correction for dredge efficiency, Table 5a) or 12 470 t (corrected using historical average efficiency, Table 5b) with c.v.s of 14–15%. These biomass estimates are 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 81–82% of the total biomass larger than 90 mm was found (depending on whether or not a correction for dredge efficiency was applied). The biomass estimates in Table 5a (i.e., those with no 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.

Historically, the fishery operated to a MLS of 100 mm and yield was similarly estimated, so biomass was also estimated using 100 mm as the size at recruitment (Tables 6a and 6b). At the time of the survey, the estimated biomass above 100 mm was 1690–5380 t, 52 or 43% of the biomass above 90 mm depending on the treatment of dredge efficiency). About 80% 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 the level of dominance seen in 2004 and 2005 is atypical.

Table 5a: Estimated density and biomass of scallops 90 mm shell length or more at the time of the Coromandel survey, assuming 100% dredge efficiency.

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Mercury	84.2	36	0.3537	0.0619	0.18	29.782	88.05	31.14	5.24	0.17	2 622
Plenty	49.8	32	0.0629	0.0077	0.12	3.135	86.91	5.47	0.69	0.13	272
Barrier	4.2	14	0.3835	0.0801	0.22	1.597	91.54	35.10	7.53	0.22	146
Waiheke	24.6	13	0.0535	0.0363	0.70	1.314	81.27	4.35	3.05	0.70	107
Colville	11.0	10	0.0891	0.0448	0.50	0.977	96.80	8.62	4.57	0.53	95
Fishery	173.7	105	0.2119	0.0307	0.15	36.805	88.10	18.67	2.61	0.14	3 242

Table 5b: Estimated density and biomass of scallops 90 mm shell length or more at the time of the Coromandel survey, assuming historical average dredge efficiency (but not including variance associated with dredge efficiency).

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Mercury	84.2	36	1.4299	0.2640	0.18	120.402	84.58	120.9	21.5	0.18	10 184
Plenty	49.8	32	0.2559	0.0305	0.12	12.746	83.49	21.4	2.6	0.12	1 064
Barrier	4.2	14	1.5365	0.3183	0.21	6.398	87.57	134.5	28.2	0.21	560
Waiheke	24.6	13	0.1490	0.1029	0.69	3.659	82.49	12.3	8.8	0.72	302
Colville	11.0	10	0.3215	0.1737	0.54	3.527	103.14	33.2	18.9	0.57	364
Fishery	173.7	105	0.8447	0.1298	0.15	146.732	85.01	71.8	10.6	0.15	12 474

Table 6a: Estimated density and biomass of scallops 100 mm shell length or more at the time of the Coromandel survey, assuming 100% dredge efficiency.

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Mercury	84.2	36	0.1576	0.0253	0.16	13.270	103.03	16.2	2.6	0.16	1 367
Plenty	49.8	32	0.0268	0.0043	0.16	1.333	101.04	2.7	0.4	0.16	135
Barrier	4.2	14	0.1889	0.0464	0.25	0.787	107.43	20.3	5.1	0.25	85
Waiheke	24.6	13	0.0136	0.0124	0.92	0.333	97.62	1.3	1.2	0.92	33
Colville	11.0	10	0.0612	0.0375	0.61	0.671	106.35	6.5	4.0	0.62	71
Fishery	173.7	105	0.0944	0.0127	0.13	16.393	103.11	9.7	1.3	0.13	1 690

Table 6b: Estimated density and biomass of scallops 100 mm shell length or more at the time of the Coromandel survey, assuming historical average dredge efficiency (but not including variance associated with dredge efficiency).

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Mercury	84.2	36	0.4957	0.0816	0.16	41.741	102.60	50.9	8.3	0.16	4 283
Plenty	49.8	32	0.0834	0.0129	0.15	4.154	100.06	8.3	1.3	0.16	416
Barrier	4.2	14	0.5894	0.1433	0.24	2.454	107.51	63.4	15.7	0.25	264
Waiheke	24.6	13	0.0461	0.0425	0.92	1.132	98.39	4.5	4.2	0.93	111
Colville	11.0	10	0.2497	0.1560	0.62	2.739	111.10	27.7	17.5	0.63	304
Fishery	173.7	105	0.3006	0.0415	0.14	52.220	102.98	31.0	4.3	0.14	5 378

A more sophisticated “re-sampling” approach to variance estimation, including variability associated with the dredge efficiency scalars and length-weight regressions, increased the c.v.s on the fishery-wide density estimates to about 23% and on the fishery-wide biomass estimates to about 22%. The statistical distribution of the biomass estimate is slightly skewed to the right (Figure 10), but not markedly so, and estimates of the c.v. are very similar whether calculated from a fitted normal distribution or by dividing the 95% confidence range of the bootstraps by 3.92 (1.96×2). Converting this estimate of greenweight biomass to meatweight would not be 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 1550–1670 t (with a c.v. of about 22%) at the time of the survey.

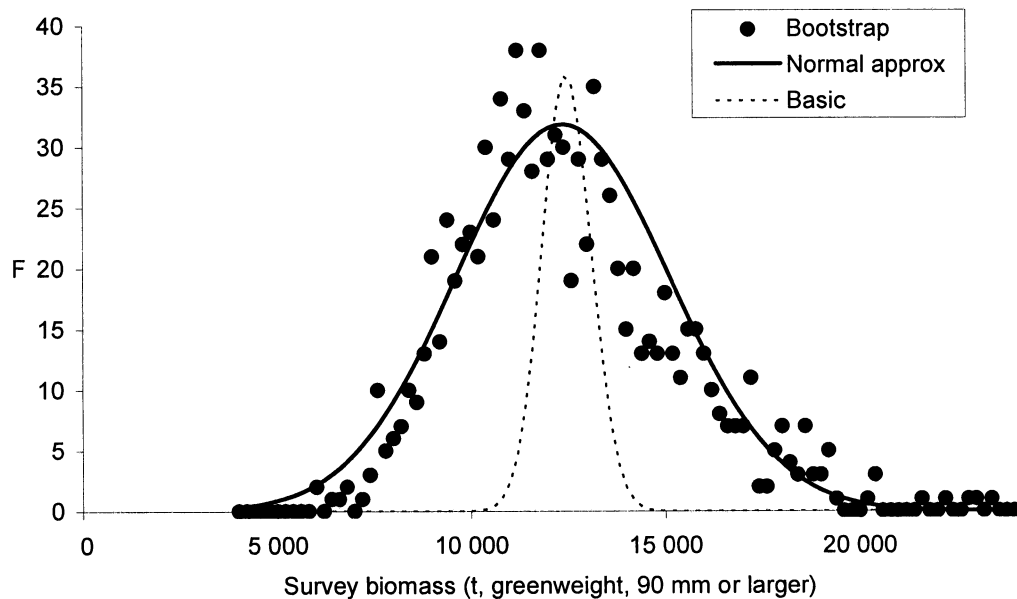


Figure 10: Frequency distribution of estimated recruited biomass (90 mm or larger) in the Coromandel fishery at the time of the survey. The results of a completely non-parametric re-sampling approach are shown as solid dots with a fitted Gaussian (normal) approximation as a solid line. The dotted line shows the results of the basic analysis (including only sampling variance). All curves scaled to similar heights for clarity.

3.2.4.2 Northland fishery at the time of the survey

During the Northland survey, 18 056 of 43 847 scallops caught in 140 tows (sweeping 0.239 km²) were measured. Approximate pooled length frequency distributions corrected for dredge efficiency and scaled to estimated population size (assuming historical average dredge efficiency) are shown for the four major areas in Figure 11. As is typical for this fishery, the beds at Spirits Bay and Rangaunu Bay had relatively higher proportions of large scallops than those in Bream Bay and south. However, there were reasonable proportions of small scallops (less than 80 mm shell length) in both areas, suggesting good recruitment for 2006.

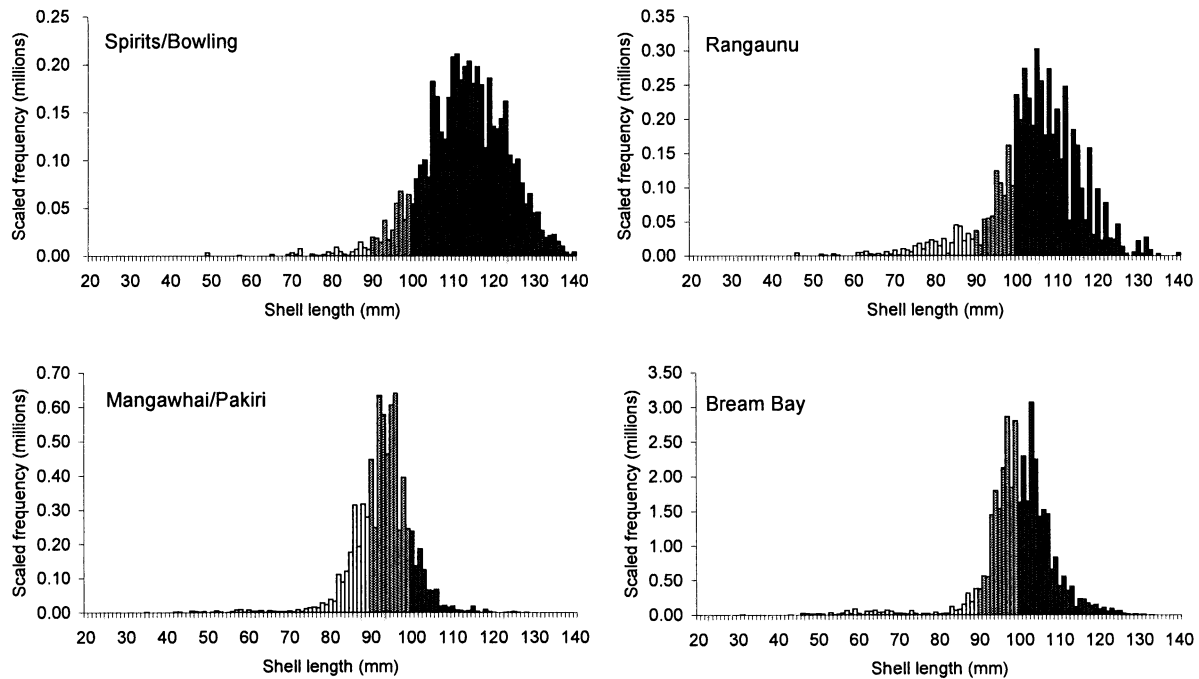


Figure 11: Length frequency distributions for major areas in the Northland fishery (corrected for historical average dredge efficiency), May 2005. Shaded bars show scallops larger than 90 mm and dark bars show scallops larger than 100 mm.

Using a simple parametric approach to estimation (including simple size-dependent scalars to correct for dredge efficiency), the biomass of scallop of 100 mm shell length or more at the time of the survey was about 2983 t with a c.v. of 13% (with no correction for dredge efficiency, Table 7). About 62% of the recruited biomass was found in Bream Bay with the remainder being mostly in Spirits Bay (19%) and Rangaunu Bay (15%). The biomass estimates in Table 7 (i.e., those with no 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. Converting this estimate of greenweight biomass to meatweight is not simple because no 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 or catch and effort data) could be used, suggesting a meatweight biomass of about 370–400 t (with a c.v. of about 13%) at the time of the survey.

Including dredge efficiency as a simple scalar (independent of length for all legal sized scallops) increases the biomass estimate to 4680 t with a c.v. of 17% (Table 8). However, analysis of the real pattern of dredge efficiency with length (see Figure 5) suggests that this approach is inappropriate and likely to be biased, especially where the average size of scallops is large. Applying average dredge efficiency at length increases the mean biomass estimate to 8668 t (Table 9) but the c.v. of 13% grossly underestimates the real uncertainty (because this approach does not incorporate the variability around observed average dredge efficiency). This estimate is reliable only if it can be assumed that the 2005 survey vessel had a similar average dredge efficiency during the survey as that of the four vessels tested previously.

Table 7: Estimated density and biomass of scallops 100 mm shell length or more at the time of the Northland survey, assuming 100% dredge efficiency.

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Rangaunu	186.8	39	0.0216	0.0041	0.19	4.040	113.6	2.46	0.48	0.20	459
Bream	82.8	37	0.2230	0.0442	0.20	18.473	101.5	22.65	4.50	0.20	1 876
Pakiri	47.0	24	0.0218	0.0045	0.21	1.026	97.3	2.12	0.44	0.21	100
Spirits	62.1	40	0.0689	0.0087	0.13	4.282	128.1	8.83	1.04	0.12	549
Fishery	378.8	140	0.0735	0.0100	0.14	27.822	107.2	7.88	1.03	0.13	2 983

Table 8: Estimated density and biomass of scallops 100 mm shell length or more at the time of the Northland survey, assuming a single dredge efficiency scalar for all legal-sized scallops (and including variance associated with that scalar).

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Rangaunu	186.8	39	0.0340	0.0072	0.21	6.343	113.4	3.85	0.82	0.21	719
Bream	82.8	37	0.3502	0.0774	0.22	29.003	101.5	35.57	7.86	0.22	2 945
Pakiri	47.0	24	0.0343	0.0078	0.23	1.61	97.4	3.34	0.76	0.23	157
Spirits	62.1	40	0.1082	0.0175	0.16	6.723	127.8	13.86	2.25	0.16	861
Fishery	378.8	140	0.1153	0.0193	0.17	43.682	107.1	12.34	2.06	0.17	4 676

Table 9: Estimated density and biomass of scallops 100 mm shell length or more at the time of the Northland survey, assuming historical average dredge efficiency (but not including variance associated with dredge efficiency).

	Area (km ²)	Sites	Density (m ⁻²)	SEM	c.v.	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)
Rangaunu	186.8	39	0.0623	0.0116	0.19	11.628	114.9	7.15	1.38	0.19	1 336
Bream	82.8	37	0.6373	0.1255	0.20	52.787	101.5	64.67	12.72	0.20	5 356
Pakiri	47.0	24	0.0633	0.0135	0.21	2.978	97.8	6.19	1.33	0.22	291
Spirits	62.1	40	0.2059	0.0247	0.12	12.789	131.7	27.12	3.00	0.11	1 685
Fishery	378.8	140	0.2117	0.0284	0.13	80.182	108.1	22.89	2.91	0.13	8 668

A more sophisticated “re-sampling” approach to variance estimation, including all variability associated with the dredge efficiency scalars and length-weight regressions, produced biomass estimates with statistical distributions with extreme right-hand skews and c.v.s (estimated assuming normality) in excess of 100% (Figure 12). As an indication of the extreme skew, the mean of the bootstrap biomass estimates was 8870 t but the median was only 5790 t. Sensitivity analysis (not shown) indicated that almost all of the additional variance came from the dredge efficiency information, and especially a modest number of rather extreme outliers in the biomass scalars. Some biomass scalars were of the order of 50 (which would indicate a dredge efficiency of 2%) and others were less than 1 (indicating dredges are more efficient than divers for large scallops). We cannot accept either proposition and infer that the dredge efficiency data are insufficient to describe the variance structure of average dredge efficiency in this fishery, let alone dredge efficiency for this vessel in this survey. It is hard to know how to deal with this level of uncertainty. Short of developing

an entirely new “risk analysis” approach, the options we have considered are retaining the old method until better information on dredge efficiency is available (survey biomass about 4700 t), constraining the dredge efficiency scalars to exclude outliers (about 6000–8000 t, depending on the constraints), or adopting different measures of central tendency (such as the median, about 5800 t, or mode, about 5200 t). We have used median values for yield calculations.

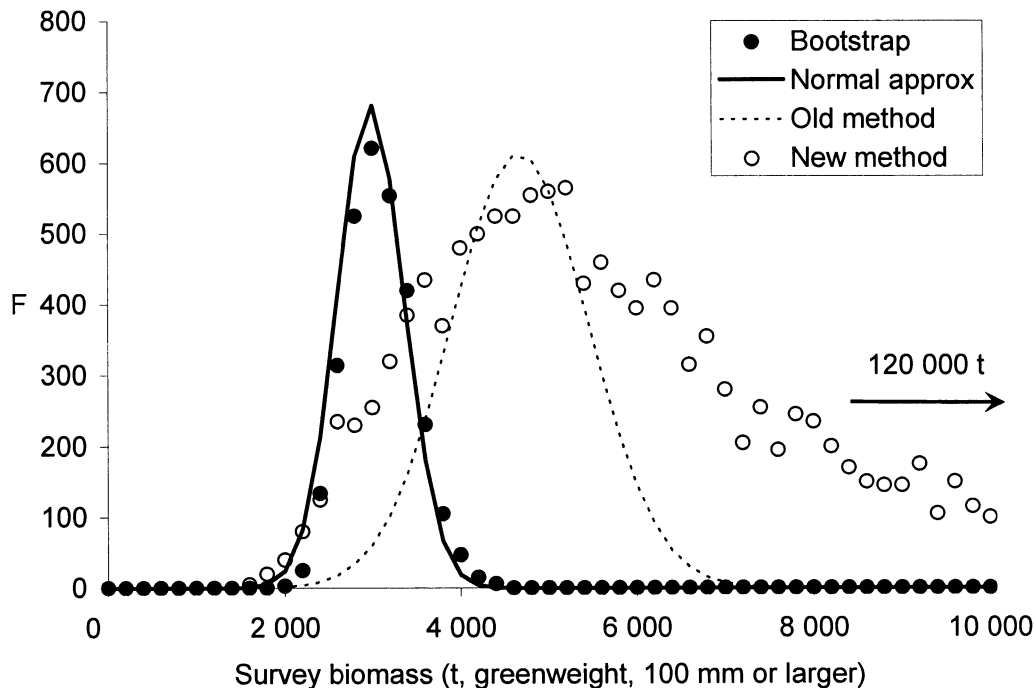


Figure 12: Frequency distribution of estimated recruited biomass (100 mm or larger) in the Northland fishery at the time of the survey. The results of a completely non-parametric re-sampling approach assuming 100% dredge efficiency is shown as solid circles with a fitted Gaussian (normal) approximation as a solid line. The dotted line shows the results of the “old” analysis (including dredge efficiency as a scalar independent of length but with an associated variance) and the open circles show the new method using length-dependent scalars incorporated by bootstrapping both survey and dredge efficiency data. All curves scaled to similar heights for clarity.

3.2.4.3 Trends in the Coromandel fishery since 1990

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 the substantial improvement observed since the turn of the century has continued, especially at the Mercury Islands (Table 10, 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 recovered enormously from the lowest ever recorded abundance of scallops in 2001 (53.2 vs. 1.5 million scallops 95 mm or longer, Table 10). *Chaetopterus* tubeworms have become quite rare, and were never a hindrance to surveying (filling the dredge and causing it to “fly” in 2005). The biomass at Little Barrier Island has increased to levels recorded in the first two surveys in the mid 1980s (about 3 million scallops), and the beds at Waihi, off the Katikati Entrance, had an estimated 3.7 million scallops of 95 mm or more at time of the 2005 survey, much more than most recent surveys, but still less than the very high biomass estimate in 1993. The Motiti–Papamoa bed has had a similar biomass

for the past 3 years (about 2 million scallops 95 mm or longer). Scallops were again numerous at Waiheke Island in 2005, but only a small proportion were of legal size. The 2005 survey at Colville was again quite limited, but gave a similar biomass estimate to that of 2003 (which was the highest estimated biomass since surveys began there in 1993). Beds at Great Barrier, Otama, and at Shoe and Slipper Islands were not surveyed because they were not expected to support commercial fishing in 2005. Overall, the 2005 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 at the Mercury Islands, but the large numbers of small scallops in the Bay of Plenty beds (especially) suggest that recruitment to the fishery there in 2006 is likely to be good.

Table 10: 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 applied as simple length-independent scalars 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 Islands	Waihi	Motiti, Papamoa	Little Barrier	Colville	Waiheke Island	Total
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	23.5	1.0	2.4	1.2	0.3	4.7	33.2
2005	53.2	3.7	1.8	2.8	2.5	2.4	66.6

3.2.4.4 Trends in the Northland fishery 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 a substantial improvement is clear in Bream Bay (Table 11, although this table could be slightly misleading because it is based on a size at recruitment of 95 mm). There were good numbers of scallops in both Rangaunu Bay and south of Bream Bay, but the estimates for Spirits Bay suggest a rather poor year there in comparison with most previous estimates. The improvement for 2005 is greatest in Bream Bay, but the large numbers of small scallops there and in Rangaunu Bay (see Figure 11) suggest that recruitment to the fishery in 2006 is likely to be reasonable.

Table 11: Number of scallops at the time of survey in constituent areas of the Northland fishery since 1992 (millions 95 mm or greater shell length assuming historical average dredge efficiency of 64% for all scallops 95 mm or larger in all years). Total includes data from beds not mentioned specifically. Asterisks (*) indicate unreliable results, dashes (–) indicate no survey. Surveys in 2002 and 2003 were completed very close to the start of the season and entries in this table are, therefore, positively biased with respect to other years.

	Spirits	Rangaunu	Doubtless	Whangaroa	Cavalli	Bream	Pakiri	Total
1992	–	7.0	0.7	–	0.4	16.8	4.0	28.9
1993	–	*1.5	0.7	1.7	0.4	5.5	–	*9.8
1994	–	8.5	1.3	0.6	–	4.2	0.2	14.8
1995	–	9.0	1.0	2.3	1.2	3.5	0.1	18.2
1996	24.4	7.7	0.3	1.2	0.9	2.2	–	37.6
1997	15.8	9.9	0.7	1.1	0.7	*5.7	0.4	35.3
1998	4.7	6.0	0.3	0.5	0.9	0.2	<0.1	14.0
1999	–	–	–	–	–	–	–	–
2000	–	–	–	–	–	–	–	–
2001	5.4	6.6	0.0	0.1	–	1.1	–	13.2
2002	10.5	9.3	–	0.1	–	5.4	–	26.6
2003	8.3	7.5	0.1	0.4	0.0	1.6	–	18.0
2004	–	–	–	–	–	–	–	–
2005	4.9	7.2	–	–	–	46.7	4.9	66.1

3.2.4.5 Predicting start-of-season biomass in the Coromandel fishery

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 5a (a total of 3240 t) and allow for about 8% mortality over the 8.6 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). Almost all steps in this calculation are conservative, so the overall result is, therefore, very conservative. However, it might be considered a “benchmark” against which other approaches can be compared. Based on these assumptions, the minimum expected biomass at the start of the season would be about 2980 t greenweight, or about 300 t meatweight (Table 12).

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 8% suggested by spreading $M = 0.5$ evenly throughout the year). Assuming historical average values for dredge efficiency and growth greatly increases the estimate to 14 370 t greenweight (median projected value) with a c.v. of 23% (Table 13, Figure 13), 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 1800 t meatweight (median projected value) with a c.v. of 27% (Table 13, Figure 14). In contrast with the substantial decline predicted in 2004, the average weight of a recruited scallop in 2005 was projected to decline only modestly from 85.0 to 84.2 g by the start of the season.

Table 12: Mean projected biomass of scallops 90 mm shell length or more in the Coromandel fishery at the start of the season assuming 100% dredge efficiency, no growth, $M = 0.5$ spread evenly through the year, and lowest ever recovery of meatweight from greenweight. These are conservative assumptions.

	Area (km ²)	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)	Meatweight (t)
Mercury	84.2	29.8	88.05	28.65	4.82	0.17	2 412	240
Plenty	49.8	3.1	86.91	5.03	0.63	0.13	250	25
Barrier	4.2	1.6	91.54	32.29	6.93	0.22	134	13
Waiheke	24.6	1.3	81.27	4.00	2.81	0.70	98	10
Colville	11.0	1.0	96.80	7.93	4.20	0.53	87	9
Fishery	173.7	36.8	88.10	17.18	2.40	0.14	2 983	296

Table 13: Mean projected biomass of scallops 90 mm shell length or more in the Coromandel fishery 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.

	Area (km ²)	Millions	Mean wt. (g)	Biomass (t green)	c.v.	Meatweight (t)	c.v.
Mercury	84.2	142.8	83.9	11 979	0.26	1 501	0.29
Plenty	49.8	14.8	83.5	1 231	0.23	154	0.27
Barrier	4.2	7.2	86.8	628	0.30	79	0.33
Waiheke	24.6	5.1	80.5	409	0.71	51	0.74
Colville	11.0	3.7	100.2	370	0.61	46	0.63
Fishery	173.7	173.6	84.2	14 617	0.23	1 832	0.27

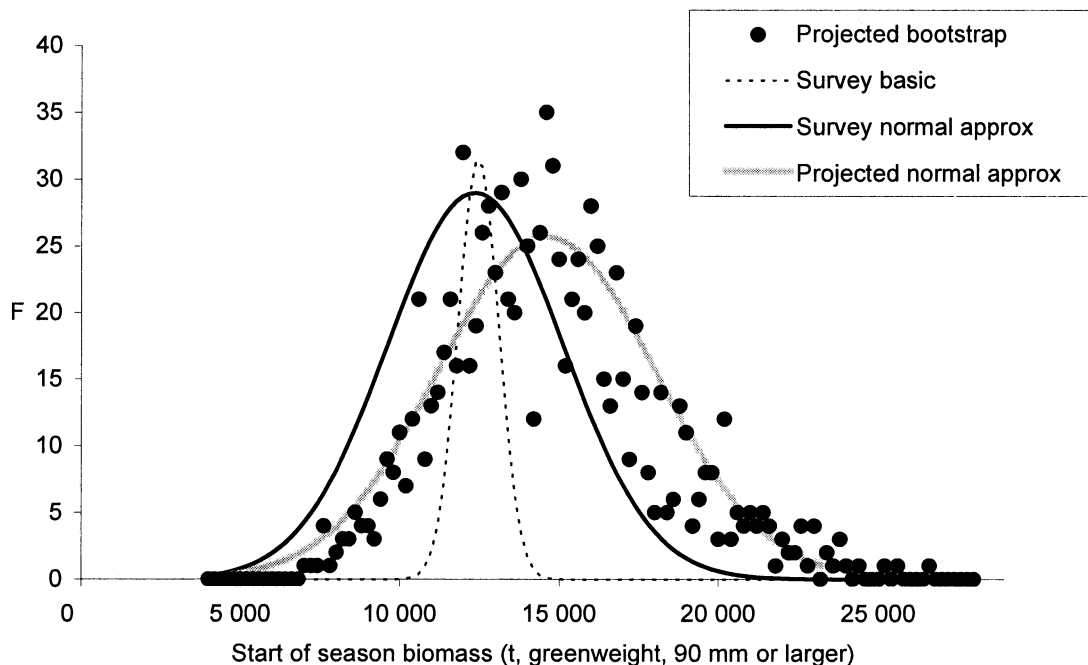


Figure 13: Frequency distribution of projected recruited biomass (90 mm or larger) in the Coromandel fishery at the assumed start of the season in mid-July. The results of a non-parametric re-sampling and projection approach are shown as solid circles with a fitted Gaussian approximation as a grey line. The dotted line shows the results of the basic analysis (including only sampling variance) at the time of the survey and the solid line repeats the normal approximation to the biomass estimate including dredge efficiency variability at the time of the survey. All curves scaled to similar heights for clarity.

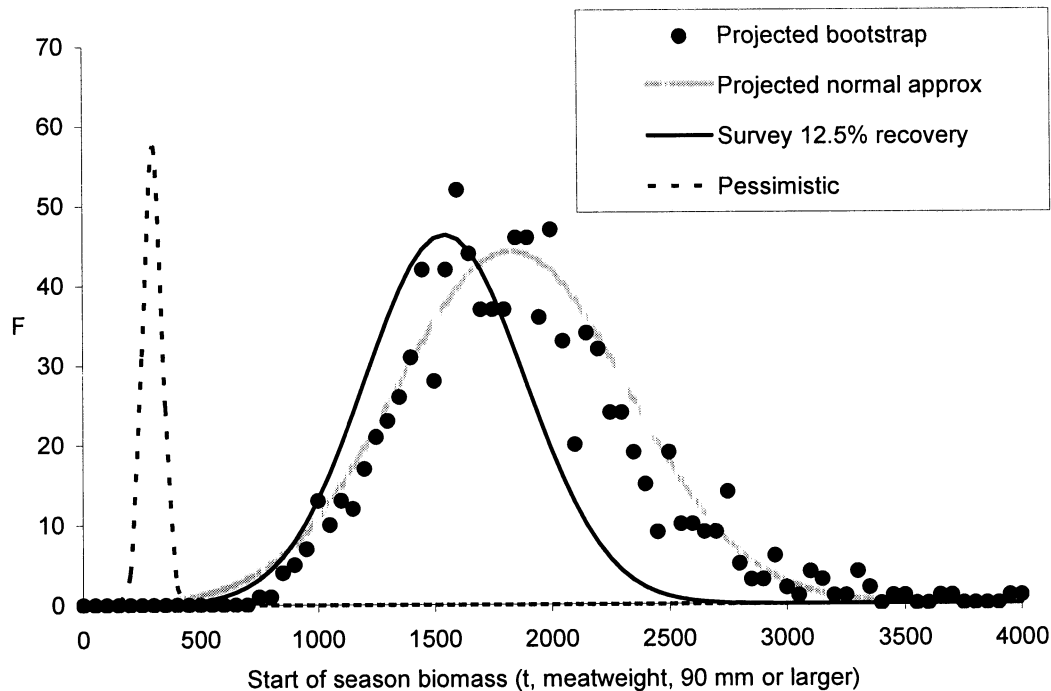


Figure 14: Frequency distribution of projected recruited meatweight (90 mm or larger) in the Coromandel fishery at the assumed start of the season in mid-July. The results of a non-parametric re-sampling and projection approach are shown as solid circles with a fitted Gaussian approximation as a grey 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 greenweight. The start-of-season biomass at 100 mm was predicted to be 5080 t greenweight (median projected value) with a c.v. of 22.0% (35% of the biomass above 90 mm shell length) and the start-of-season meatweight at 100 mm was predicted to be 630 t (median projected value) with a c.v. of 26.3%.

3.2.4.6 Predicting start-of-season biomass in the Northland fishery

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 7 (a total of 2983 t) and allow for about 5% mortality over the 5.1 weeks between the survey midpoint and the 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 in the Northland fishery (11.13% in 1999). Almost all steps in this calculation are conservative, so the overall result is, therefore, very conservative. However, it might be considered a “benchmark” against which other approaches can be compared. Based on these assumptions, the minimum expected biomass at the start of the season would be about 2840 t greenweight, or about 316 t meatweight (Table 14).

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 5% suggested by spreading $M = 0.5$ evenly throughout the year). Retaining the 100% dredge efficiency but allowing for average growth increases the estimate to 3025 t greenweight (median projected value) with a c.v. of 13%, and a start-of-season meatweight of 410 t with a c.v. of 17%. Including average historical dredge efficiency further increases the estimate to 5570 t greenweight (median projected value) and a start-of-season meatweight of 750 t (Table 15), but c.v.s for both of these estimates are very broad as a result of the extreme right-hand skew of their distributions (Figure 15).

Table 14: Mean projected biomass of scallops 100 mm shell length or more in the Northland fishery 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 in Northland. These are very conservative assumptions. Median values are almost identical.

	Area (km ²)	Millions	Mean wt. (g)	Biomass (kg.m ⁻²)	SEM	c.v.	Biomass (t green)	Meatweight (t)
Rangaunu	186.8	3.846	113.6	2.34	0.46	0.20	437	49
Bream	82.8	17.586	101.5	21.56	4.28	0.20	1786	199
Pakiri	47	0.977	97.3	2.02	0.42	0.21	95	11
Spirits	62.1	4.076	128.1	8.41	0.99	0.12	523	58
Fishery	378.8	26.487	107.2	7.50	0.98	0.13	2840	316

Table 15: Mean and median projected biomass of scallops 100 mm shell length or more in the Northland fishery at the start of the season assuming historical average dredge efficiency at length (with its multiplier constrained to between 1 and 5 for each size class over 80 mm in each bootstrap), average growth (from previous tagging studies, mostly in the Coromandel fishery), $M = 0.5$ spread evenly through the year, and average recovery of meatweight from greenweight. Mean weights are calculated using mean values, and c.v.s are calculated assuming normality (and are, therefore, very approximate).

	Millions		Mean wt. (g)	Green (t)			Meat (t)		
	Mean	Median		Mean	Median	c.v.	Mean	Median	c.v.
Rangaunu	9.171	7.348	115.2	1 057	849	0.63	143	115	0.65
Bream	43.555	32.946	103.2	4 494	3 367	0.82	610	459	0.83
Pakiri	2.811	2.099	98.3	276	206	0.89	38	28	0.90
Spirits	9.723	8.325	129.6	1 260	1 092	0.48	171	148	0.49
Fishery	65.260	51.318	108.6	7 087	5 565	0.70	962	753	0.71

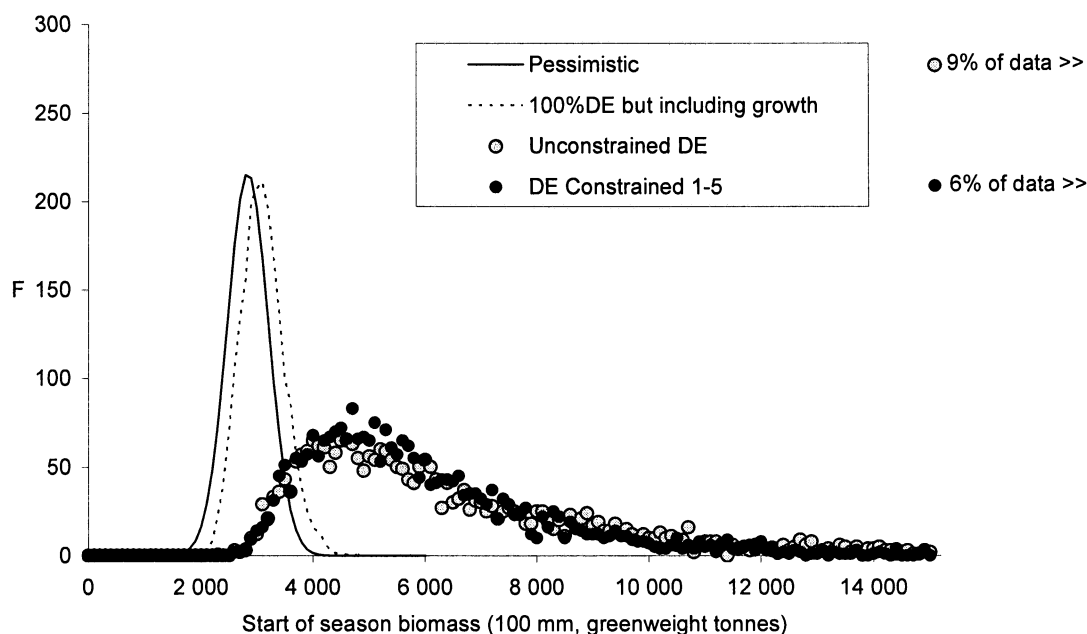


Figure 15: Frequency distribution of projected recruited biomass (100 mm or larger) in the Northland fishery at the assumed start of the season in mid July. The solid line shows the results assuming 100% dredge efficiency and no growth (pessimistic), the dotted line shows 100% dredge efficiency and average growth, the open dots show unconstrained historical average dredge efficiency, and the closed dots show average dredge efficiency with its multiplier constrained to be between 1 and 5 for a size class. Both analyses incorporating historical average dredge efficiency have extreme right hand tails, to about 120 000 t or 30 000 t, respectively.

3.2.4.7 Sensitivity of Coromandel biomass to exclusion of areas of low scallop density

Most recent stock 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 m^{-2} (one recruited scallop for each 25 m^2 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 m^{-2} had zero density, and stations where scallops were denser than 0.04 m^{-2} had a density of the actual density minus 0.04 m^{-2} . These corrections were applied before any scaling for dredge efficiency, so they are conservative.

Excluding areas of low density (under 0.04 m^{-2}) reduced the fishery-wide biomass estimates at the time of the survey by about 15% (Figure 16); excluding areas where the density was less than 0.08 m^{-2} reduced the fishery-wide biomass estimates by about 25% (although 0.08 m^{-2} is quite a high density by historical standards).

The effect of excluding areas of low density from biomass estimates varied markedly among the main beds (Figure 17). Where the mean density of scallops was high at the Mercury Islands (where most of the biomass was concentrated) and at Little Barrier Island, the biomass estimate declined only about 10% when areas where scallops were less dense than 0.04 m^{-2} were excluded. Conversely, biomass estimates at Colville and Waiheke Island declined by 20–25% when areas where scallops were less dense than 0.04 m^{-2} were excluded, and that for the Bay of Plenty declined by more than 50%. Thus, a reasonable fraction of the biomass outside the Mercury Island and Little Barrier Island beds is contained in areas of relatively low density where catch rates may not be very good. These beds contain only an estimated 15% of the total biomass, however.

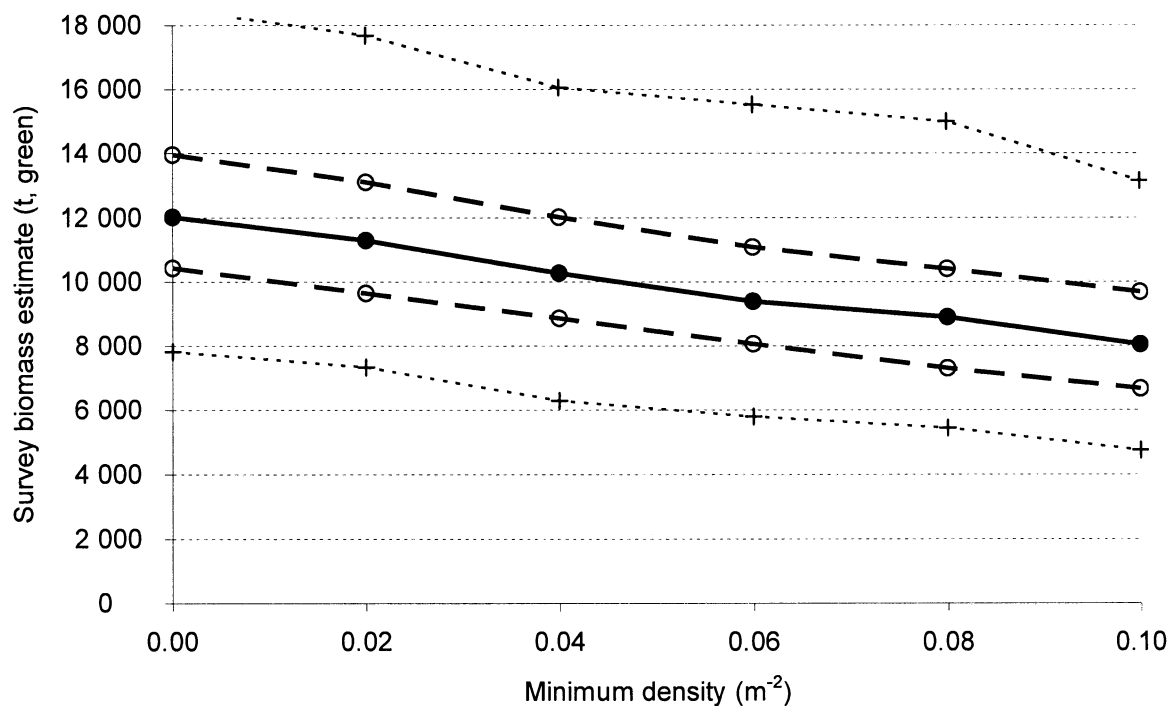


Figure 16: Effect of excluding areas of low scallop density on the fishery-wide biomass estimate in the Coromandel fishery at the time of the survey. For increasing minimum acceptable densities (un-scaled for dredge efficiency) the median biomass estimates (solid line, closed circles) are given together with their quartiles (dashed lines) and 95% confidence range (dotted lines).

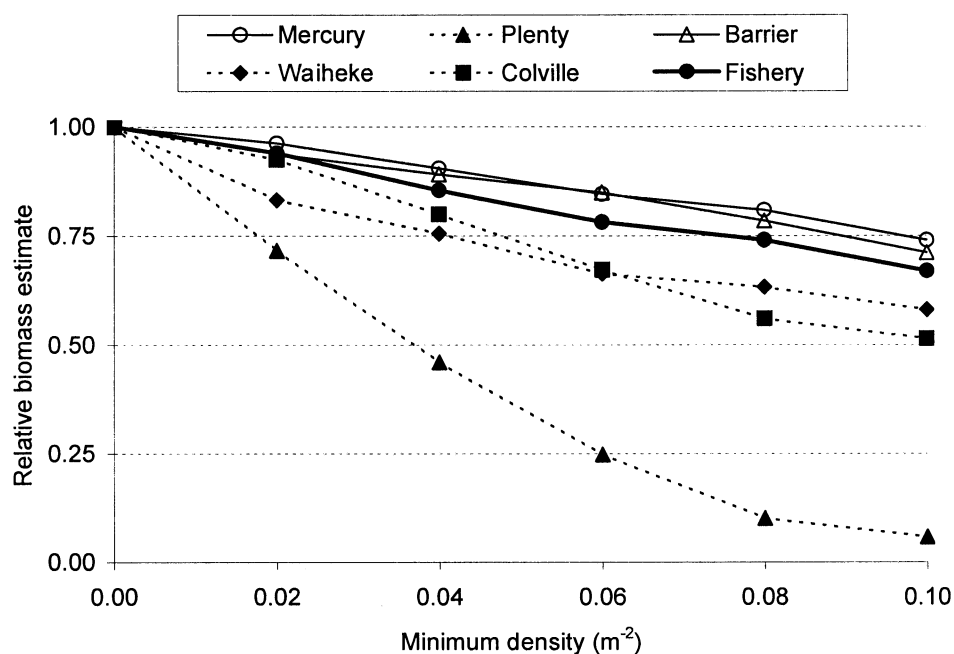


Figure 17: Effect of excluding areas of low scallop density on biomass estimate by bed in the Coromandel fishery at the time of the survey. For increasing minimum acceptable densities (un-scaled for dredge efficiency) the median biomass estimates for each area are given.

3.2.4.7 Sensitivity of Northland biomass to exclusion of areas of low scallop density

Excluding areas of low density (under 0.04 m^{-2}) reduced the fishery-wide biomass estimates at the time of the survey by about 25% (Figure 18); excluding areas where the density was less than 0.08 m^{-2} reduced the fishery-wide biomass estimates by about 45% (although 0.08 m^{-2} is a very high criterion by historical standards).

The effect of excluding areas of low density from biomass estimates varied markedly among the main beds (Figure 19). Where the mean density of scallops was high in Bream Bay (where most of the biomass was concentrated), the biomass estimate declined only about 13% when areas where scallops were less dense than 0.04 m^{-2} were excluded. Conversely, biomass estimates at Spirits Bay and Rangaunu Bay declined by 50% when areas where scallops were less dense than 0.04 m^{-2} were excluded, and that for Mangawhai and Pakiri Beach declined by more than 80%. Thus, a reasonable fraction of the biomass outside Bream Bay is contained in areas of relatively low density where catch rates may not be very good. These beds contain only about one-third of the total biomass, however.

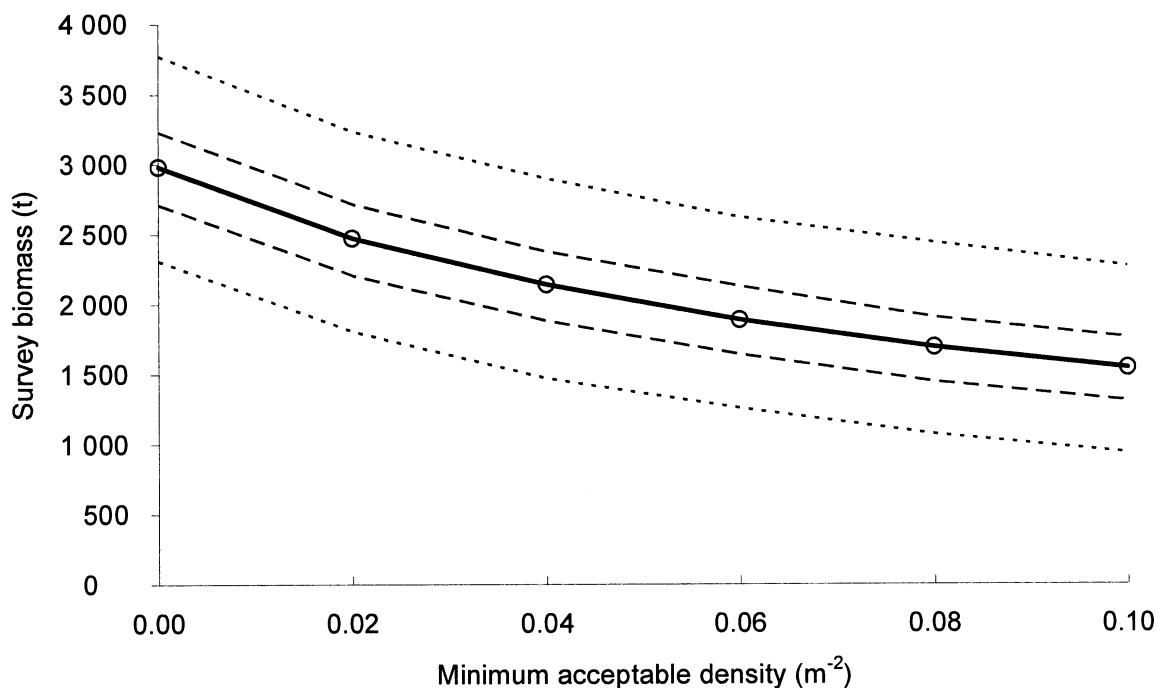


Figure 18: Effect of excluding areas of low scallop density on the fishery-wide biomass estimate in the Northland fishery at the time of the survey. For increasing minimum acceptable densities (un-scaled for dredge efficiency) the median biomass estimates (solid line, closed circles) are given together with their quartiles (dashed lines) and 95% confidence range (dotted lines).

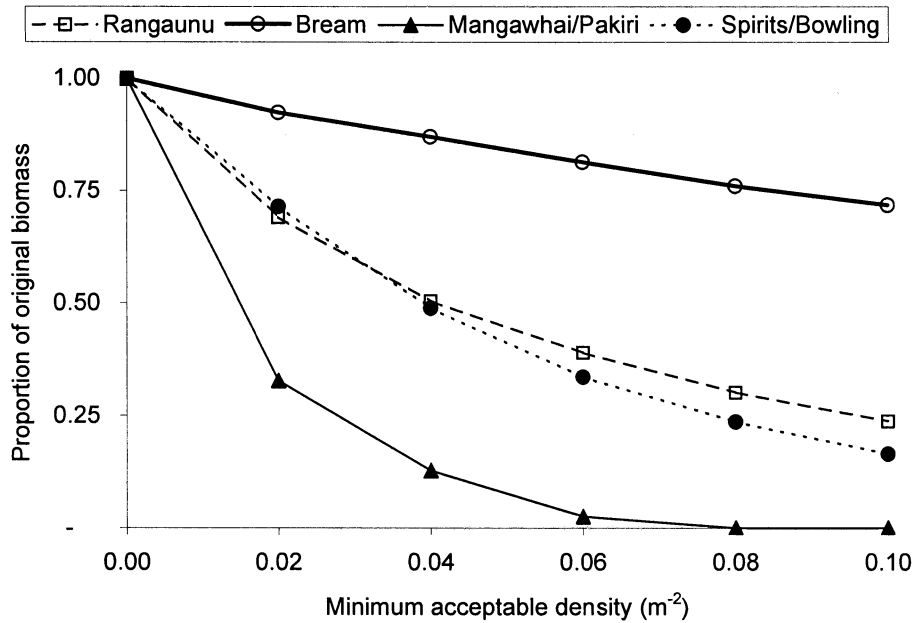


Figure 19: Effect of excluding areas of low scallop density on biomass estimate by bed in the Northland fishery at the time of the survey. For increasing minimum acceptable densities (un-scaled for dredge efficiency) the median biomass estimates for each area are given.

3.3 Biomass estimates

Estimates of current biomass are described above. Given the highly variable nature of northern scallop populations, the concept of virgin biomass is probably not meaningful, but average (recruited) biomass could be estimated from the data presented in the tables.

3.4 Yield estimates

3.4.1 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) 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. 2004) 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 an 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_0 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 an LRP, and $F_{0.1}$ and $F_{40\%}$ as TRPs. They estimated $F_{40\%}$ (0.514 y^{-1}), $F_{0.1}$ (0.508 y^{-1}), and F_{\max} (0.650 y^{-1}) (all assuming $M = 0.50 \text{ y}^{-1}$). Because of the derivation of these estimates, they should be applied to the modified version of the Baranov equation given by Cryer & Morrison (1997):

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

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 $F_{0.1}$ by selecting the better fitting of related domed and asymptotic two-parameter curves:

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

and

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

where YPR is the yield (catch) for each recruiting scallop of 1 mm shell length, and α and β are constants defining the shape of the curve of YPR against $F_{apparent}$. 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 Beverton & Holt's (1957) stock-recruit curve. Recent work (Cryer et al. 2004) has indicated that a more flexible three parameter formulation based on Deriso's (1980) generalised 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 γ 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} (0.68 vs. 0.65) and a lower $F_{0.1}$ (0.43 vs. 0.51) than reported by Cryer & Morrison (1997) (Table 16, Figure 20). There was no change to $F_{40\%}$ because the form of the fitted curve was not changed.

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. (2004) 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 21).

Table 16: 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_{\max}	$F_{0.1}$	$F_{40\%}$	$F_{25\%}$	YPR_{\max}	$YPR_{0.1}$	$E_{F_{\max}}(\%)$	$E_{F_{0.1}}(\%)$
No incidental effects								
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 juveniles								
None (90)	0.659	0.431	0.514	0.875	1459	1383	32.5	45.6
1 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

Our knowledge of the life history and mortality schedules of juvenile scallops is still rudimentary, but the modelling conducted by Cryer et al. (2004) 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 direct 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 unstudied.

Recruitment of scallops could be affected by fishing in at least three 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*); through disruption or removal of high density patches of adults leading to lower fertilisation success (Williams & Babcock 2004a, 2004b); or through the destruction or removal of foliose material necessary as spat settlement surfaces (e.g., Harvey and Bourget 1995, Harvey et al. 1995). The last change 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 & Borzone 1997, Wilbur et al. 1999, Dichmont et al. 2000) and landings in the Coromandel fishery declined (albeit with considerable fluctuation) from over 1000 t in the 1980s and early 1990s to less than 200 t in the early years of this century during a period of relatively heavy fishing pressure. Simplistic stock assessment models including no incidental effects of fishing could not have predicted this from landings over this time.

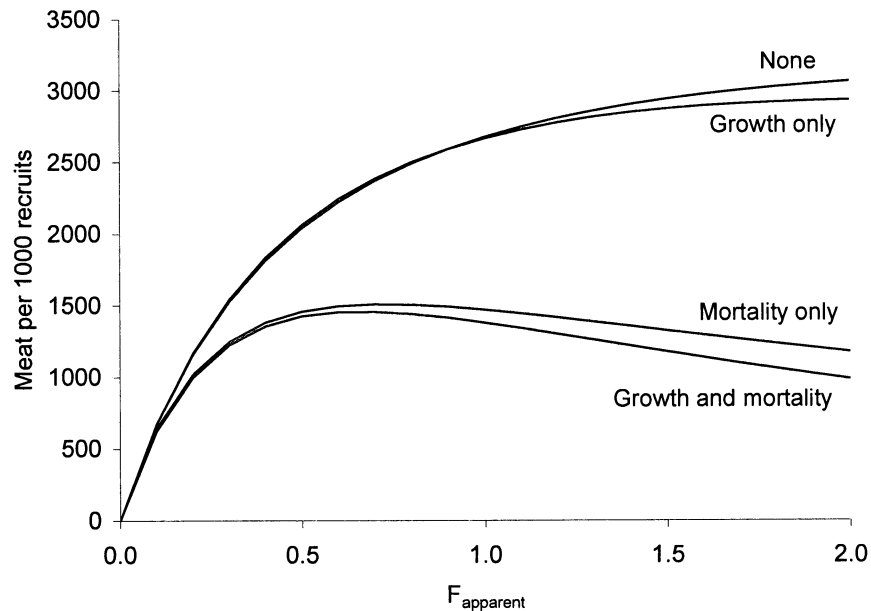


Figure 20: Yield per recruit curves based on Cryer & Morrison's individual-based model outputs and Deriso's three-parameter curve. Incidental effects of the dredge method on adult scallops are included as indicated (after Cryer et al. 2004).

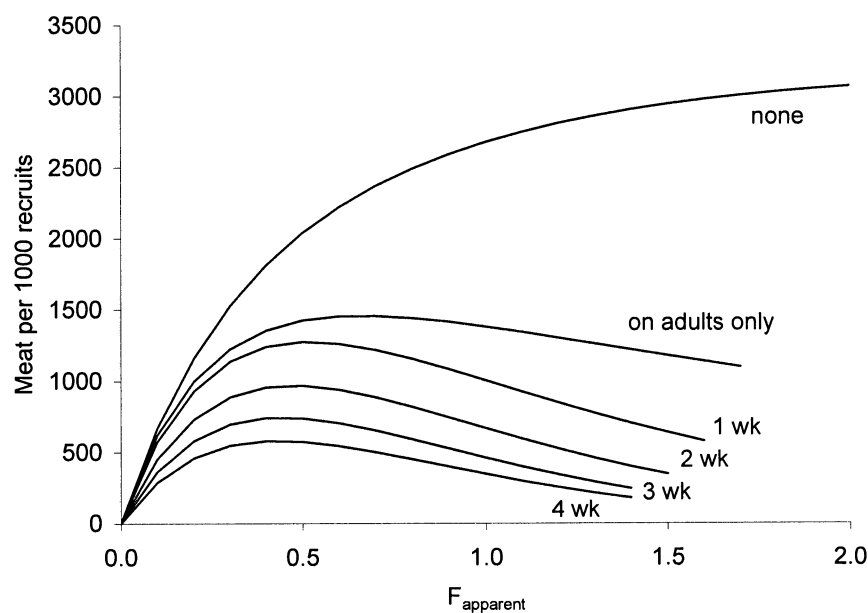


Figure 21: Yield per recruit curves based on Cryer & Morrison's individual-based model outputs and Deriso's three-parameter curve. The top curve includes no incidental effects, the second curve is a repeat of the bottom curve in Figure 20, and the remaining curve come from models including a link between average fishing effort and juvenile mortality that acts for the indicated time (after Cryer et al. 2004).

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 (cited by Walters & Martell (2002)), Patterson's (1992) broad analysis of pelagic fish stocks, and Myers et al.'s (1999) work on stock-recruit relationships, contradicted this supposition. Walters & Martell (2002) suggested that any stock assessment that suggests an optimum rate of fishing mortality much greater than half of the rate of natural mortality ($M = 0.5 \text{ yr}^{-1}$ 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 age-structured models incorporating mean recruitment effects discussed by Walters & Martell (2002) do not explicitly include the incidental effects of their fishing methods; they too may be optimistic.)

3.4.2 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. (2003a) showed that constant catch strategies for scallops produced lower yield at much higher biological risk than strategies wherein catch was varied as biomass varied.

3.4.3 Estimation of Current Annual Yield (CAY)

3.4.3.1 CAY for the Coromandel fishery

Pessimistic outlook

The most pessimistic outlook from the 2005 Coromandel 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%) led to a start of season biomass of 2983 t greenweight or 296 t meatweight (see Table 12). 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 $F_{40\%}$ (0.514), $\text{CAY} = 0.7116 * 0.5144 * 2983 \text{ (296)} = 1092 \text{ t (green) or } 108 \text{ t (meat)}$

For $F_{0.1}$ (0.431), $\text{CAY} = 0.6741 * 0.4724 * 2983 \text{ (296)} = 950 \text{ t (green) or } 94 \text{ t (meat)}$

For F_{\max} (0.683), $\text{CAY} = 0.7666 * 0.5898 * 2983 \text{ (296)} = 1349 \text{ t (green) or } 134 \text{ t (meat)}$

These estimates of CAY would have a c.v. at least as large as that of the estimates of start-of-season 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. The sensitivity of excluding areas of low density has not been calculated, but the reduction would be no more than 20% for a criterion of 0.04 m^{-2} . 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 at the start of the 2005 season in the Coromandel fishery was predicted to be 14 370 t (greenweight) and 1800 t (meatweight) (see

Table 13). 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), $CAY = 0.7116 * 0.5144 * 14\ 370\ (1800) = 5260\ t\ (green)\ or\ 659\ t\ (meat)$

For $F_{0.1}$ (0.431), $CAY = 0.6741 * 0.4724 * 14\ 370\ (1800) = 4576\ t\ (green)\ or\ 573\ t\ (meat)$

For F_{max} (0.683), $CAY = 0.7666 * 0.5898 * 14\ 370\ (1800) = 6497\ t\ (green)\ or\ 814\ t\ (meat)$

These estimates of CAY would have a c.v. at least as large as that of the estimate of start-of-season recruited biomass (20–27%), are sensitive to assumptions about dredge efficiency, growth, and expected recovery of meatweight from greenweight, and relate to the surveyed beds only. The sensitivity of excluding areas of low density has not been calculated, but the reduction would be up to about 15% for a criterion of $0.04\ m^{-2}$ (depending on which beds were fished). 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. (2004) 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 effect of including “recruitment” effects on yield estimates, Cryer et al.’s (2004) estimates of $F_{0.1} = 0.274$, $F_{max} = 0.349$, and $F_{40\%} = 0.356$ are applied here:

For $F_{40\%}$ (0.356), $CAY = 0.6308 * 0.4313 * 14\ 370\ (1800) = 3910\ t\ (green)\ or\ 490\ t\ (meat)$

For $F_{0.1}$ (0.274), $CAY = 0.5681 * 0.3827 * 14\ 370\ (1800) = 3124\ t\ (green)\ or\ 391\ t\ (meat)$

For F_{max} (0.349), $CAY = 0.6262 * 0.4272 * 14\ 370\ (1800) = 3844\ t\ (green)\ or\ 482\ t\ (meat)$

These estimates of CAY would have a c.v. at least as large as that of the estimate of start-of-season recruited biomass (20–27%), are sensitive to assumptions about dredge efficiency, growth, and expected recovery of meatweight from greenweight, to the duration of any habitat-mediated increase in juvenile mortality, and relate to the surveyed beds only. The sensitivity of excluding areas of low density has not been calculated, but the reduction would be up to about 15% for a criterion of $0.04\ m^{-2}$ (depending on which beds were fished). The level of risk to the putative Coromandel scallop stock of fishing at the estimated CAY level cannot be determined.

3.4.3.2 CAY for the Northland fishery

Pessimistic outlook

The most pessimistic outlook from the 2005 Northland 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 11.13%) led to a start of season biomass of 2840 t greenweight or 316 t meatweight (median values, see Table 14). Based on these calculations, CAY was calculated using the two reference fishing mortality rates $F_{0.1}$ and F_{max} as follows:

For $F_{0.1}$ (0.550), $CAY = 0.6535 * 0.5690 * 2840 (316) = 1056 \text{ t (green) or } 118 \text{ t (meat)}$

For F_{max} (0.700), $CAY = 0.7059 * 0.6290 * 2840 (316) = 1261 \text{ t (green) or } 140 \text{ t (meat)}$

These estimates of CAY would have a c.v. at least as large as that of the estimates of start-of-season recruited biomass (13–17%) 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. The sensitivity of excluding areas of low density has not been calculated, but the reduction would be no more than 25% for a criterion of 0.04 m^{-2} . The level of risk to the putative Northland scallop stock of fishing at the estimated CAY level cannot be determined but would probably be low.

Average outlook, including incidental effects on adults

The recruited biomass of scallops 100 mm in shell length or greater at the start of the 2005 season in the Northland fishery was predicted to be 5565 t (greenweight, median projected value using constrained historical dredge efficiency) and 753 t (meatweight, median project value) (see Table 15). CAY was calculated using the two reference fishing mortality rates $F_{0.1}$ and F_{max} as follows:

For $F_{0.1}$ (0.550), $CAY = 0.6535 * 0.5690 * 5565 (753) = 2070 \text{ t (green) or } 280 \text{ t (meat)}$

For F_{max} (0.700), $CAY = 0.7059 * 0.6290 * 5565 (753) = 2470 \text{ t (green) or } 334 \text{ t (meat)}$

These estimates of CAY would have a c.v. at least as large as that of the estimate of start-of-season recruited biomass (about 70%, if calculated using the highly suspect assumption of normality), are sensitive to assumptions about dredge efficiency, growth, and expected recovery of meatweight from greenweight, and relate to the surveyed beds only. Sensitivity to the exclusion of areas of low density (modelled assuming 100% dredge efficiency) suggests a reduction of up to about 25% for a criterion of 0.04 m^{-2} (depending on which beds were fished, much less of a reduction for Bream Bay). The level of risk to the putative Northland scallop stock of fishing at the estimated CAY level cannot be determined.

3.5.4 Estimation of Provisional Yield

3.4.4.1 Definition 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. 2004; $M > 0.35$, $c = 0.6$ for scallops) and the historical average landings from the unsurveyed beds. PY is estimated only for comparison with historical estimates of yield.

3.4.4.2 PY for the Coromandel fishery

For Coromandel scallops in 2005, start-of-season recruited biomass (100 mm or greater) was estimated at 5080 t (greenweight, c.v. = 22.0%) or 630 t (meat weight, c.v. = 26.3%), giving lower tails to the respective 95% confidence distributions of 3352 t (greenweight) and 299 t (meatweight). Beds not included in the surveys were excluded specifically because they were thought unlikely to support commercial fishing in 2005, so no addition for unsurveyed beds was made:

$$PY = 3352 (299) \text{ t} + 0.6 * 0 \text{ t} = 3352 \text{ t (greenweight) or } 299 \text{ t (meat)}$$

This estimate of PY would have a c.v. at least as large as that of the estimates of start-of-season recruited biomass at 100 mm shell length (22–26%), and is sensitive to assumptions about dredge efficiency and growth. The sensitivity of excluding areas of low density has not been calculated, but the reduction would probably be about 15% or more for a criterion of 0.04 m^{-2} (depending on which beds were fished). The sensitivity is higher for 100 mm scallops than for 90 mm scallops because the former are considerably rarer so a minimum density of 0.04 m^{-2} is less frequently exceeded. The level of risk to the putative Coromandel scallop stock of fishing at the estimated PY level cannot be determined.

3.4.4.3 PY for the Northland fishery

For Northland scallops in 2005, the statistical distribution of the start-of-season recruited biomass (100 mm or greater) was approximately normal if dredge efficiency is assumed to be 100%, but is extremely skewed to the right if a dredge efficiency correction is applied. However, the 2.5 percentiles for both types of distributions can be estimated from our non-parametric analyses. For the pessimistic analysis, the 2.5 percentile was 2197 t (greenweight) and 271 t (meatweight), for the projection using constrained historical dredge efficiency, the 2.5 percentile was 3213 t (greenweight) and 410 t (meatweight). Beds not included in the surveys were excluded specifically because they were thought unlikely to support commercial fishing in 2005, so no addition for unsurveyed beds was made:

$$\text{PY} = 2197 (271) \text{ t} + 0.6 * 0 \text{ t} = 2197 \text{ t (greenweight) or } 271 \text{ t (meat), or}$$

$$\text{PY} = 3213 (410) \text{ t} + 0.6 * 0 \text{ t} = 3213 \text{ t (greenweight) or } 410 \text{ t (meat)}$$

These estimates of PY are sensitive to assumptions about dredge efficiency and growth. The sensitivity of excluding areas of low density has not been calculated, but the reduction would probably be about 25% or more for a criterion of 0.04 m^{-2} (depending on which beds were fished). The level of risk to the putative Northland scallop stock of fishing at the estimated PY level cannot be determined.

The yield estimates for Northland presented here are based on modelling that is not as comprehensive as that used for the Coromandel stock assessment. In particular, YPR curves from the individual-based model of Cryer & Morrison (1997) have not been refitted with flexible three-parameter curves and habitat effects (and consequent effects on juvenile mortality) have not been incorporated. In the Coromandel fishery, both changes reduced the estimate of $F_{0.1}$, and similar decreases might be expected for the Northland fishery. This means that not all components of the “pessimistic” yield estimates are necessarily conservative (though many are) and the yield estimates based on the average outlook may not be conservative. The highly skewed nature of biomass estimates corrected for dredge efficiency mean that any yield estimates based on the mean projected biomass are likely not to be conservative. We have, therefore, used median projected biomass, but even the median tends to be somewhat higher than the modal classes for these distributions.

4. MANAGEMENT IMPLICATIONS

Estimates of current biomass for the Coromandel fishery are available (from a 2005 survey), but the only reference biomass that might be calculated is average recruited biomass. Scallop biomass can be expected to vary from one year to the next, so the long-run average is difficult to estimate and not necessarily a good indicator. However, biomass estimates around the turn of the century were consistently at or near the lowest on record, and it seems reasonable to conclude that the population was, for unknown reasons, then at a very low ebb. In contrast, following reasonable increases in

biomass, catch rate, and condition of scallops in 2003 and, especially, 2004, the biomass in 2005 (almost regardless of what is assumed about dredge efficiency) was the highest on record, and probably higher than in the mid 1980s when not all the beds were surveyed. This remarkable resurgence was strongest at the Mercury Islands bed, but most beds show some increase in density (Figures 22 and 23).

Similarly, estimates of current biomass for the Northland fishery are available (from a 2005 survey), but the only reference biomass that might be calculated is average recruited biomass (and the time-series of surveys in Northland is much shorter than that in the Coromandel fishery). Estimates of biomass in 2005 were much higher than those in 2003 for some beds (there was no survey in 2004), but similar or lower in others. There seems to have been a “shift” in biomass away from the Far North and towards Bream Bay and Pakiri Beach (Figures 24 and 25). This is the “reverse” of the shift towards the Far North that occurred in the early 1990s.

Given the dramatic changes observed in both fisheries, and the natural variability and unpredictability of scallop stocks, it is not known if landings at the level of CAY are sustainable or whether they will allow the stock to move towards a size that will support the MSY. Such indicators probably have little meaning for scallops.

5. ACKNOWLEDGMENTS

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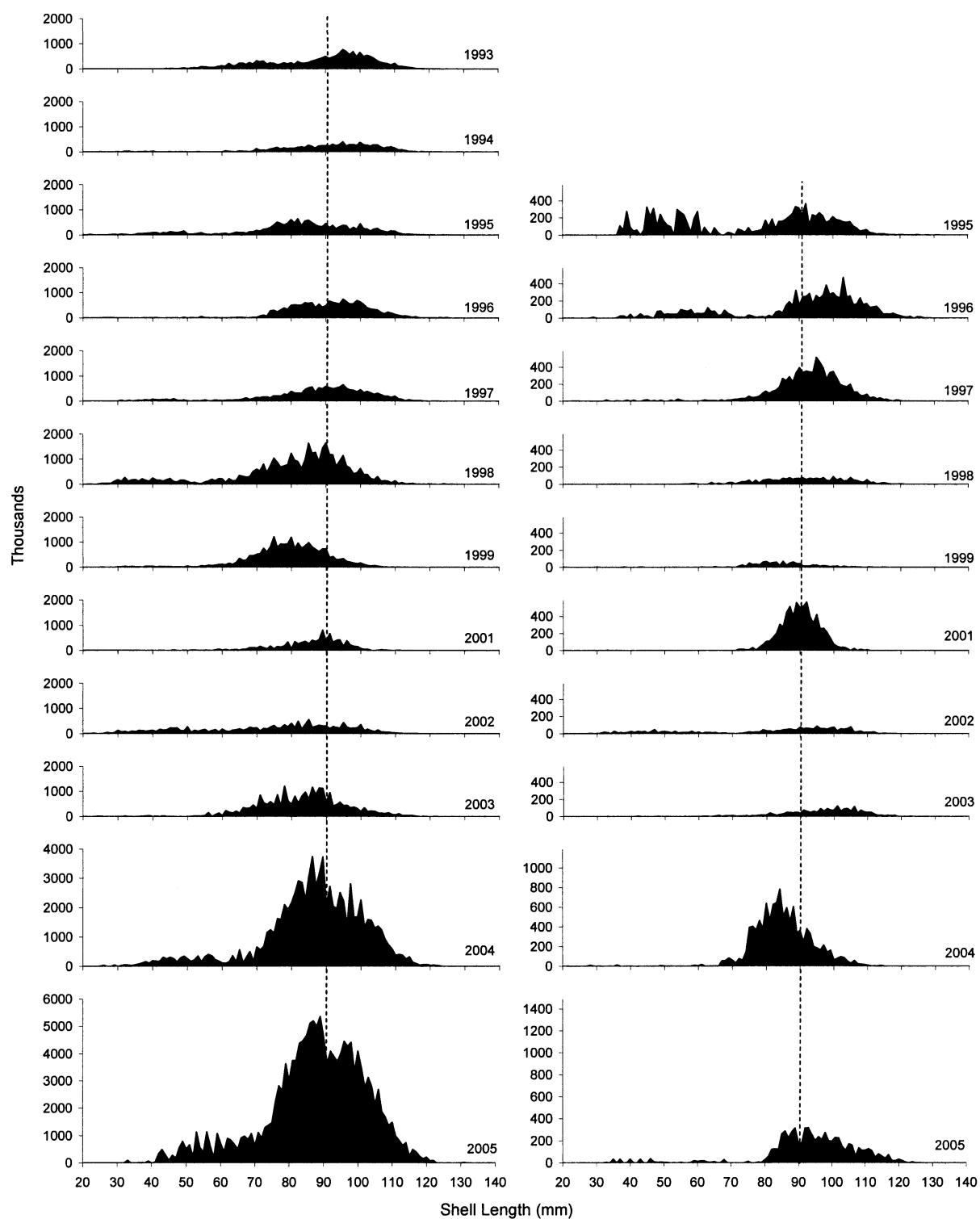


Figure 22: Scaled length frequency distributions at the time of surveys estimated using historical average dredge efficiency for scallops in the Coromandel fishery at the Mercury Islands (left) and Little Barrier Island (right) since 1993. Vertical dotted lines indicate the minimum legal size of 90 mm shell length.

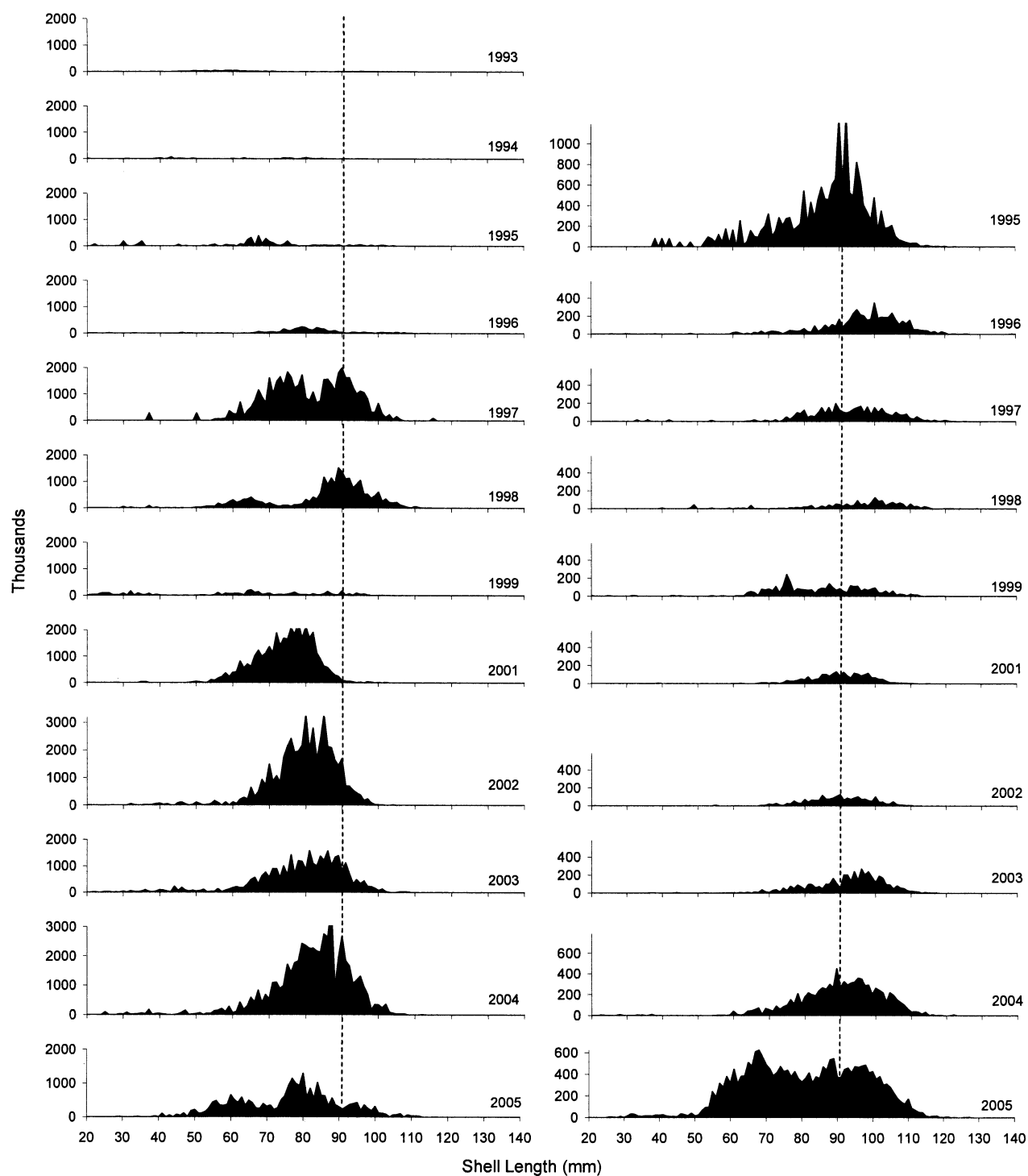


Figure 23: Scaled length frequency distributions at the time of surveys estimated using historical average dredge efficiency for scallops in the Coromandel fishery at Waiheke Island (left) and Motiti-Papamoa (right) since 1993. Vertical dotted lines indicate the minimum legal size of 90 mm shell length.

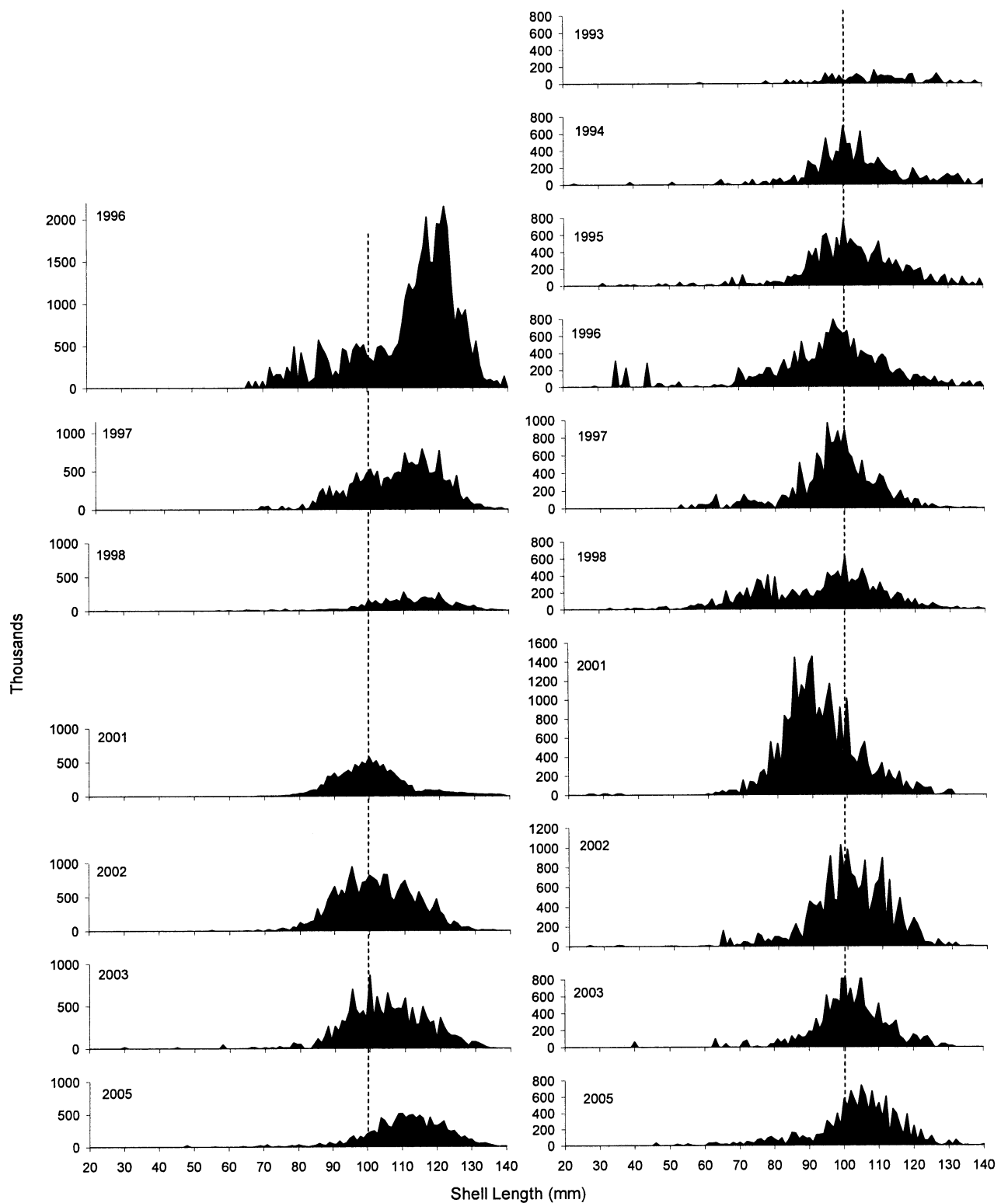


Figure 24: Scaled length frequency distributions at the time of surveys estimated using historical average dredge efficiency for scallops in the Northland fishery in Spirits Bay (left) and Rangaunu Bay (right) since 1993. Vertical dotted lines indicate the minimum legal size of 100 mm shell length.

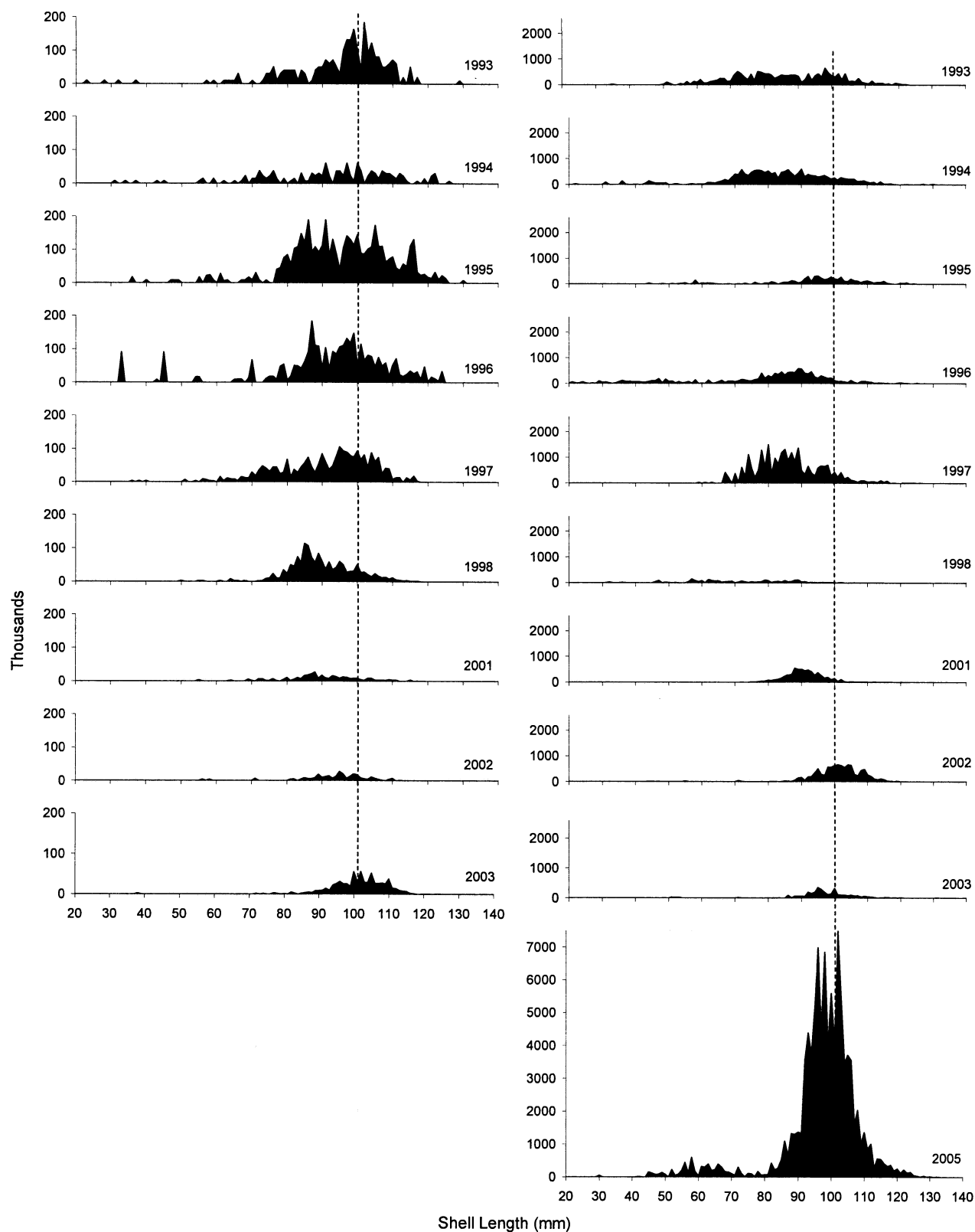


Figure 25: Scaled length frequency distributions at the time of surveys estimated using historical average dredge efficiency for scallops in the Northland fishery at Whangaroa (left) and Bream Bay (right) since 1993. Vertical dotted lines indicate the minimum legal size of 100 mm shell length.

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Appendix 1: Stratum definitions and station (tow) allocations, Coromandel scallop survey 2005.

Stratum	Location / description	Area (m ²)	Method	2005 tows
1	Mercury (Black Jack)	66 853 516	Dredge (sand)	15
2	Mercury (Three Mile Bank)	7 311 208	Dredge (sand)	8
3	Mercury (Opito Bay, extended)	7 384 277	Dredge (sand)	8
4	Mercury (Mercury Cove)	1 361 542	Dredge (silt)	5
11	Waihi (Katikati Entrance)	37 730 469	Dredge (sand)	10
13	Motiti South	7 580 078	Dredge (sand)	14
14	Papamoa Beach	4 505 615	Dredge (sand)	8
18	Little Barrier West	1 361 542	Dredge (sand)	6
19	Little Barrier South	2 802 673	Dredge (sand)	8
20	Waiheke Island	24 559 814	Dredge (silt)	13
31	Colville South (shallow)	10 970 947	Dredge (silt)	10
Total	—	173 715 489	—	105

Appendix 2: Stratum definitions and station (tow) allocations for the Northland scallop survey, 2005.

Stratum	Location / description	Area (m ²)	Method	2005 tows
2.0	Rangaunu medium	128 955 170	Dredge (sand)	14
2.1	Rangaunu high	57 828 445	Dredge (sand)	25
8.0	Bream Bay medium	62 935 806	Dredge (sand)	21
8.1	Bream Bay high	19 892 811	Dredge (sand)	16
9	Mangawhai	25 542 969	Dredge (sand)	8
10	Pakiri North	12 376 465	Dredge (sand)	8
11	Pakiri South	9 116 943	Dredge (sand)	8
91	Tom Bowling Bay	14 528 320	Dredge (sand)	9
92	Spirits Bay	47 587 177	Dredge (sand)	31
Total	—	378 764 106	—	140

Appendix 3: Summary of dredge efficiency experiments for which data were available for analysis.

Code	Year	Vessel	Site	Fishery	Substrate	Dredge		Depth (m)	Dredge		Tows	Dredge		Dive		Density >=85 mm
						width (m)	Area		width (m)	Area		Scallops	Area	Scallops	Sites	
1984CP	1984	L'Aries	Pakatoa	Coro	Silt/mud	1.55	3 950	23	11	3 950	1 037	3	150	238	0.6867	
1985CT	1985	L'Aries	Tarahiki	Coro	Silt/mud	1.90	4 607	15	16	4 607	2 615	7	67	147	1.5528	
1986CT	1986	Elco	Tarahiki	Coro	Silt/mud	1.30	4 090	15	20	4 090	4 129	12	108	238	1.3609	
1992C3	1992	L'Aries	3 Mile Bank	Coro	Sand/shell	2.00	7 445	17	4	7 445	938	15	628	350	0.4711	
1993CA	1993	L'Aries	Opito A	Coro	Sand/shell	2.00	2 963	16	4	2 963	90	16	314	80	0.1560	
1993CB	1993	L'Aries	Opito B	Coro	Sand/shell	2.00	3 148	27	4	3 148	920	8	314	435	1.1618	
1993CC	1993	L'Aries	Opito C	Coro	Sand/shell	2.00	1 148	30	2	1 148	224	4	314	206	0.4806	
1994CA	1994	L'Aries	Opito A	Coro	Sand/shell	2.00	3 704	16	2	3 704	193	4	314	47	0.1273	
1994CB	1994	L'Aries	Opito B	Coro	Sand/shell	2.00	3 704	27	2	3 704	125	4	314	12	0.0222	
1994CC	1994	L'Aries	Opito C	Coro	Sand/shell	2.00	1 926	30	2	1 926	208	4	314	97	0.1432	
1995CC	1995	Kataraina	Opito C	Coro	Sand/shell	2.00	5 556	27	2	5 556	446	4	647	127	0.0971	
1995CA	1995	Kataraina	Opito A	Coro	Sand/shell	2.00	5 556	17	2	5 556	217	4	647	210	0.2702	
1995CB	1995	Kataraina	Opito B	Coro	Sand/shell	2.00	3 395	26	3	3 395	423	4	245	154	0.1959	
1995C3	1995	Kataraina	3 Mile Bank	Coro	Sand/shell	2.00	3 297	17	3	3 297	83	4	1 181	206	0.1019	
1995CN	1995	Kataraina	Opito N	Coro	Sand/shell	2.00	2 963	20	3	2 963	189	2	1 329	304	0.1398	
1995CS	1995	Kataraina	Opito S	Coro	Sand/shell	2.00	2 926	17	3	2 926	93	9	1 731	286	0.1251	
1996CB	1996	Kataraina	Opito B	Coro	Sand/shell	2.00	4 889	25	3	4 889	870	8	481	193	0.2609	
1996CA	1996	Kataraina	Opito A	Coro	Sand/shell	2.00	5 556	12	3	5 556	499	9	569	232	0.4010	
1996CC	1996	Kataraina	Opito C	Coro	Sand/shell	2.00	2 290	28	3	2 290	610	4	358	170	0.3763	
1997CB	1997	Kataraina	Opito B	Coro	Sand/shell	2.00	6 116	27	3	6 116	752	4	525	120	0.1960	
1997CA	1997	Kataraina	Opito A	Coro	Sand/shell	2.00	7 408	15	3	7 408	186	4	804	12	0.0112	
1997CC	1997	Kataraina	Opito C	Coro	Sand/shell	2.00	2 929	29	4	2 929	822	4	402	175	0.2857	
1998CH	1998	Kataraina	Hooks Bay	Coro	Silt/mud	2.00	3 593	20	4	3 593	825	4	1 002	506	0.6888	
1999CT	1999	Kataraina	Tarahiki	Coro	Silt/mud	2.00	5 371	23	4	5 371	377	4	1 431	547	0.0820	
1992NB	1992	Avalon	Bream Bay	North	Sand/shell	2.40	8 190	17	4	8 190	938	8	1 608	349	0.1834	
1994NB	1994	Wyzanne	Bream Bay	North	Sand/shell	2.10	9 450	22	6	9 450	505	9	1 810	477	0.1149	
1995NR	1995	BenGunn	Rangauu	North	Sand/shell	2.40	10 326	18	7	10 326	711	16	3 217	233	0.0619	
1995NB	1995	BenGunn	Bream Bay	North	Sand/shell	2.40	11 779	21	6	11 779	429	10	2 011	119	0.0522	
1996NR	1996	BenGunn	Rangauu	North	Sand/shell	2.40	3 467	22	3	3 467	146	7	1 128	59	0.0549	
1997NW	1997	BenGunn	Whangaroa	North	Sand/shell	2.40	9 512	23	5	9 512	643	14	2 579	400	0.1389	
1998NW	1998	Marewa	Whangaroa	North	Sand/shell	2.40	21 957	22	10	21 957	1 193	10	1 854	81	0.0423	
1998NM	1998	Marewa	Matauri Bay	North	Sand/shell	2.40	16 712	20	8	16 712	747	8	1 608	27	0.0143	