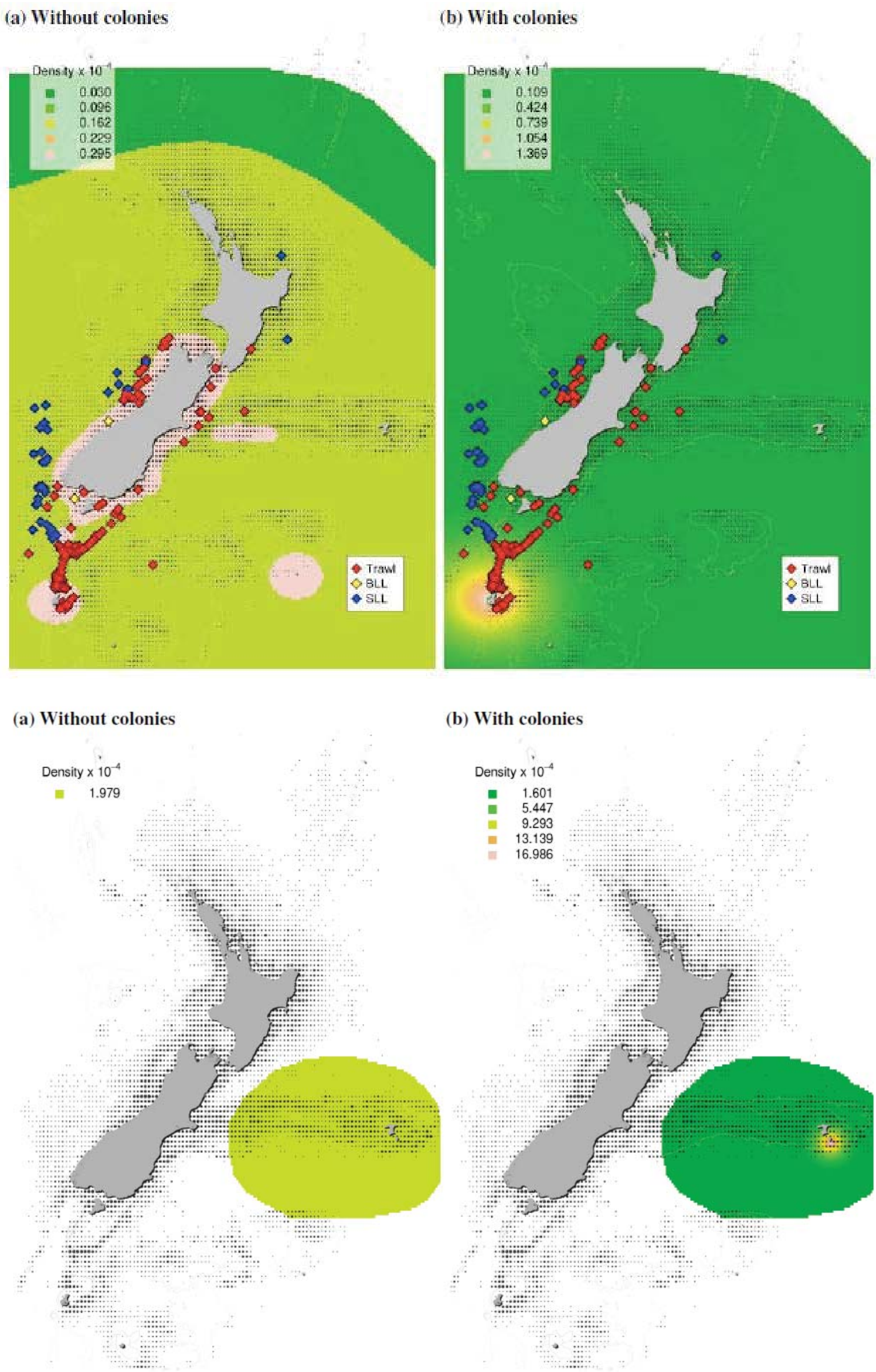


Figure 5.12 (reproduced from Richard *et al.* 2011 supplementary material): Captures and relative density of White-capped albatross (top) and Chatham petrel (bottom) showing large differences in the extent of distributions and overlap with fishing effort (in grey), and in the number of observed captures. The distribution base maps were obtained from NABIS (white-capped albatross) and the BirdLife single-layer range maps (Chatham petrel).

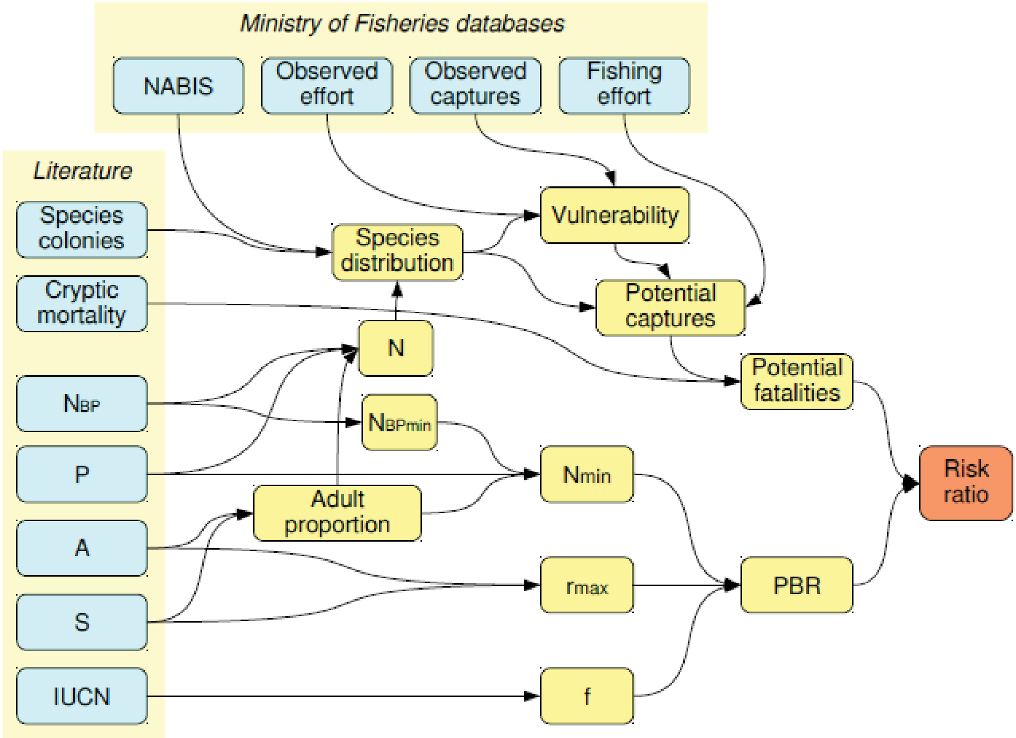


Figure 5.13 (reproduced from Richard *et al.* 2011): Diagram of the modelling approach to calculate the risk index for each taxon. N_{BP}, number of annual breeding pairs; *N*, total number of birds over one year old; N_{BPmin}, lower 25% of the distribution of N_{BP}; *Nmin*, lower 25% of the distribution of the total number of birds over one year old; r_{max}, maximum population growth rate; *f*, recovery factor; *PBR*, Potential Biological Removal; *P*, proportion of adults breeding in a given year; *A*, age at first reproduction; *S*, annual adult survival rate.

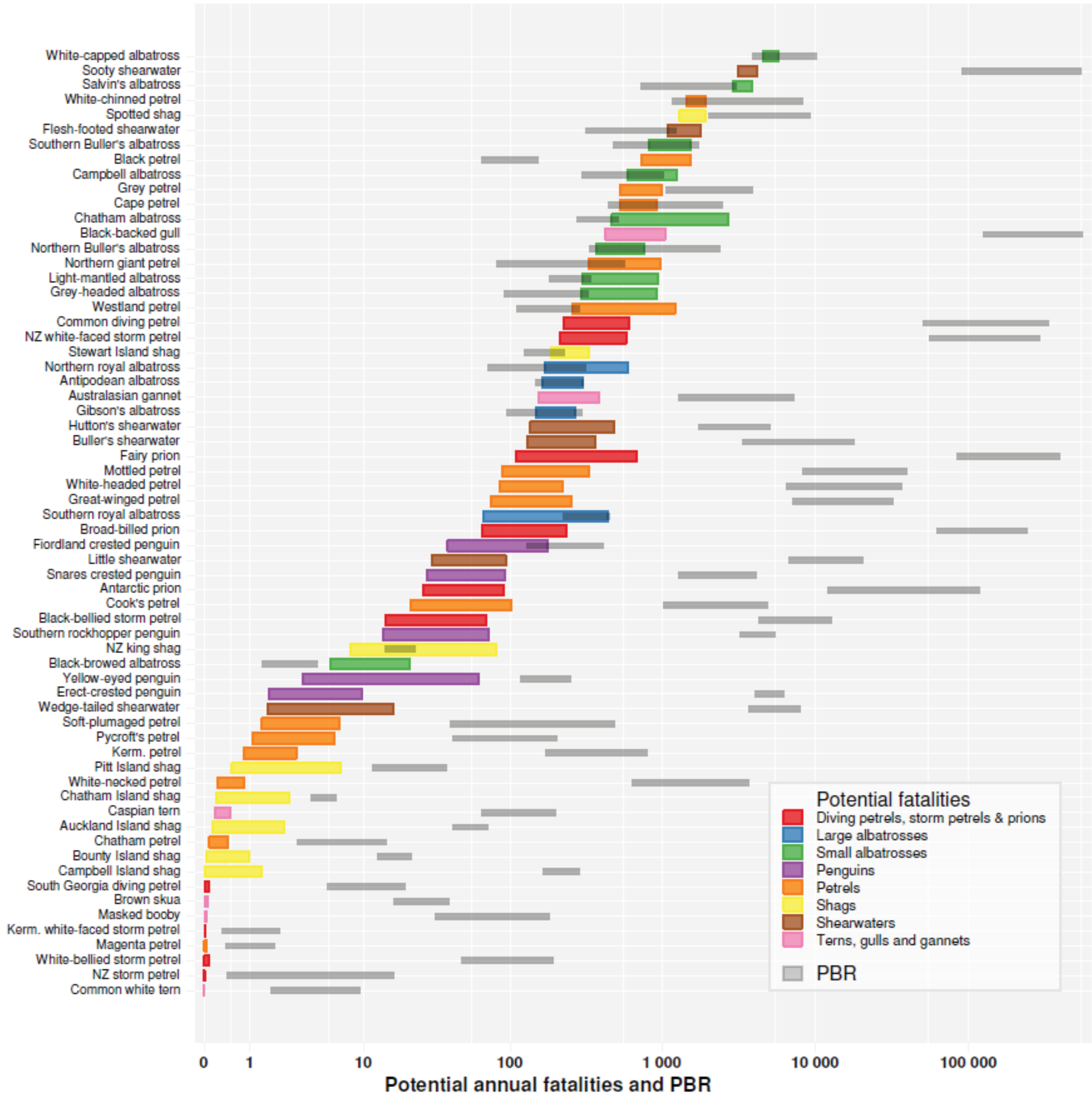


Figure 5.14 (reproduced from Richard *et al.* 2011): Mean annual potential seabird fatalities in the assessed fishery groups (colour bars) and the PBR (grey bars), for each of the 64 studied taxa. The bars indicate the 95% confidence intervals of the distributions. Taxa are sorted in decreasing order of the lower confidence level of the number of fatalities.

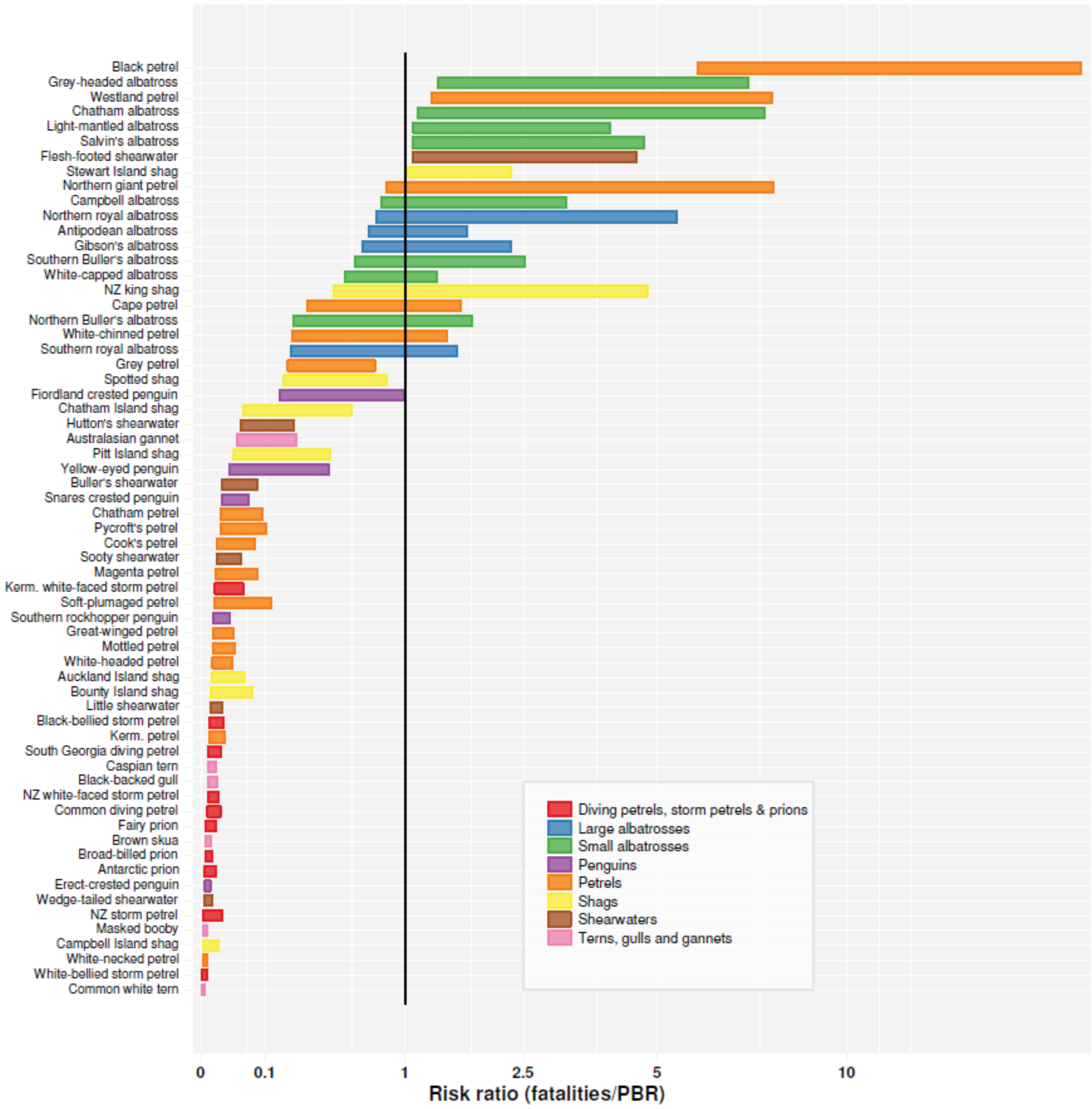


Figure 5.15 (reproduced from Richard *et al.* 2011): Risk ratio (total annual potential fatalities / PBR) for each of the studied taxa except black-browed albatross. The risk ratio is displayed on a logarithmic scale. The threshold where the number of potential bird fatalities equals the PBR is presented by the vertical black line. The bars indicate the 95% confidence intervals of the distributions. Taxa are sorted in decreasing order of the lower confidence level of the risk ratio.

Many limitations were identified in the risk assessment. These may result in biased estimates (either too high or too low) of the risk of fishing to some seabirds. Moreover, some fisheries were not included in our analysis, and other sources of human-induced mortality were ignored. The conclusions of our results should therefore be interpreted with caution, as some taxa might be at risk, even if their risk ratio was estimated to be lower than one. Conversely, the fisheries-related fatalities may be overestimated in poorly observed fisheries because the method is designed to answer the question “how bad could it be?” (e.g., Figure 5.16 which shows the results of different estimation approaches and questions). The method assumed a high number of captures in the absence of data to the contrary, so the estimated potential fatalities in poorly-observed fisheries may be higher than the actual fatalities.

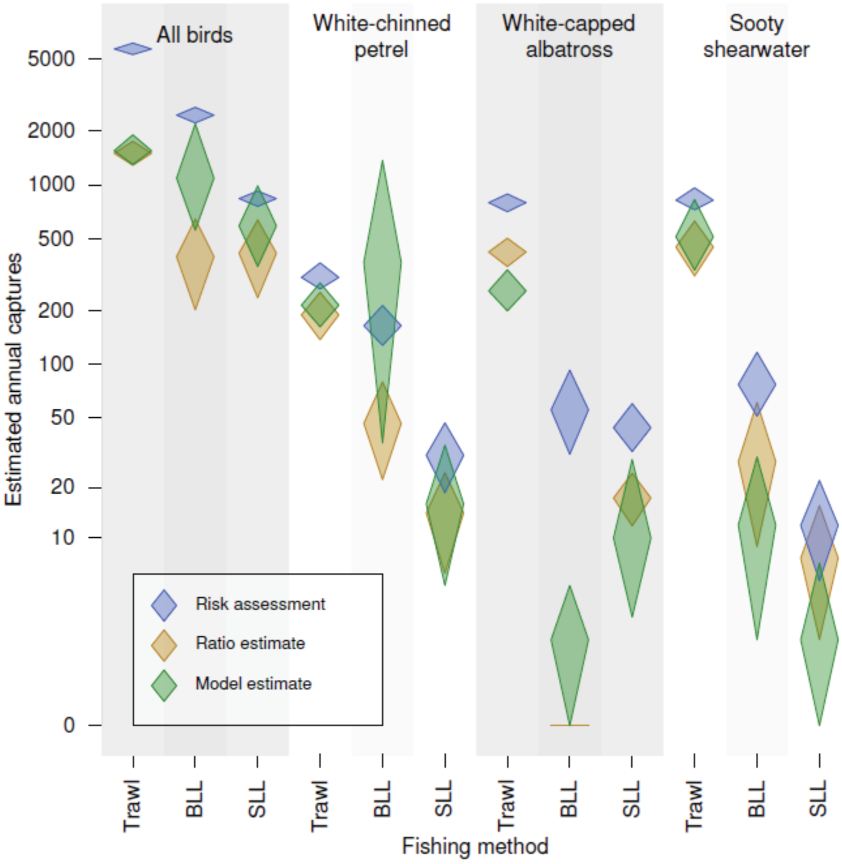


Figure 5.16 (reproduced from Richard *et al.* 2011): Comparison of the number of potential annual captures (without cryptic mortality) estimated using the risk assessment method used by Richard et al (2011), a simple ratio scalar, and statistical modelling, for white-chinned petrel, white-capped albatross, sooty shearwater, and all birds combined, in trawl, bottom longline, and surface longline fisheries. Each symbol represents the mean and the 95% confidence interval of an estimate.

The method described by Richard *et al.* (2011) offers the following advantages that make it particularly suitable for assessing risk to multiple seabird populations from multiple fisheries:

- risk is assessed separately for each seabird taxon; fisheries managers must assess risk to seabirds with reference to units that are biologically meaningful;
- the method does not rely on the existence of universal or representative fisheries observer data to estimate seabird mortality (fisheries observer coverage is generally too low and/or too spatially unrepresentative to allow direct impact estimation at the species or subspecies level); the method can be applied to any fishery for which at least some observer data exists;
- the method does not rely on detailed population models (the necessary data for which are unavailable for the great majority of taxa) because risk is estimated as a function of population-level potential fatalities and biological parameters that are generally available from published sources;
- the method assigns risk to each taxon in an absolute sense, i.e. taxa are not merely ranked relative to one another; this allows the definition of biologically meaningful performance standards and ability to track changes in performance over time and in relation to risk management interventions;
- risk scores are quantitative and objectively scalable between fisheries or areas, so that risk at a population level can be disaggregated and assigned to different fisheries or areas based on their proportional contribution to total impact to inform risk management prioritisation;

- the method allows explicit statistical treatment of uncertainty, and does not conflate uncertainty with risk; numerical inputs include error distributions and it is possible to track the propagation of uncertainty from inputs to estimates of risk; and
- the method readily incorporates new information; assumptions in the assessment are transparent and testable and, as new data becomes available, the consequences for the subsequent impact and risk calculations arise logically without the need to revisit other assumptions or repeat the entire risk assessment process.

The key disadvantages of the method are that:

- fisheries for which no observer information on seabird interactions is available cannot be included in the analysis
- the assumption that the vulnerabilities of particular seabirds to capture in different fisheries are independent does not allow "sharing" of scarce observer information between fisheries within the risk assessment
- the spatial overlap method relies on appropriate spatial and temporal scales for the distributions of birds and fishing effort being used; use of inappropriate scales can lead to misleading results
- strong assumptions have to be made about the distribution and productivity of some taxa, the relative vulnerability of different taxa to capture by particular fisheries, cryptic mortality associated with different fishing methods, and the applicability of the allometric method of estimating Potential Biological Removals.

Most of these limitations are a result of the scarcity of relevant data on seabird populations and fisheries impacts and can be addressed only through the collection of more information or, in some cases, sensitivity testing. In particular, it was not possible to include some fishery groups identified by Rowe's (2010b) level 1 analysis as posing substantial risk to seabirds. Notable among these fisheries was the commercial setnet fishery group. In the absence of quantitative information for these fishery groups, the Ministry of Fisheries combined the level 1 and level 2 results to generate a comprehensive assessment of seabird risk across all New Zealand seabirds and fisheries (Table 5.14). Apart from filling some important information gaps in the assessment, the level 1 results were also useful as a cross-check on the level 2 outputs. A number of likely misleading results were identified in this way, including those from poor input data (e.g. spatial distribution layers) or faulty structural assumptions for particular seabird taxa, and these were noted so that inappropriate conclusions were not made and to provide for better treatment in subsequent iterations of the level 2 analysis.

At the time of going to press, a major update and revision of the level 2 risk assessment published by Richard *et al.* (2011) was undergoing final review. This revision includes several substantial improvements on the 2011 version including:

- fisheries and observer data from 2006/07 onwards (i.e., post-mitigation only, allowing a better estimate of current risk)
- inclusion of set net fisheries (obviating the need to combine level 1 and level 2 analyses)
- revised bird distributions
- inclusion of seasonal stratification of bird distribution and overlap with fisheries
- an integrated approach to estimating species-specific vulnerability to particular fisheries
- correction of a bias in the estimation of productivity from age at first reproduction and annual adult survival rate
- inclusion of uncertainty in estimates of cryptic mortality

This revised risk assessment is expected to be available early in 2013.

Table 5.14: Combined level 2 and level 1 risk assessments for seabird taxa with a risk ratio of 0.3 or greater (i.e., mean potential fatalities 30% of the estimated PBR or greater). INS, inshore trawl fisheries including for flatfish; SQU, squid trawl fisheries; SCI, scampi trawl fisheries; OFF, other offshore trawl fisheries; BLL, bottom longline fisheries; SLL, surface longline fisheries; SN, setnet (from level 1); Other, all other fisheries considered in the level 1 risk assessment. * indicates an unreliable assessment.

Taxon	INS	SQU	SCI	OFF	BLL	SLL	SN	Other	Risk ratio
Black (Parkinson's) petrel	4.37	0.12	0.17	0.37	5.56	0.41	0.00	0.45	11.45
Black-browed albatross *	1.07	0.02	0.04	0.64	2.46	1.37	0.00	0.00	* 5.59
New Zealand king shag	1.84	0.00	0.00	0.05	0.15	0.00	1.51	0.91	4.46
Grey-headed albatross *	2.38	0.03	0.09	0.37	0.52	0.07	0.00	0.00	* 3.46
Westland petrel	1.99	0.12	0.05	0.36	0.59	0.15	0.00	0.00	3.26
Chatham albatross	1.81	0.04	0.08	0.19	0.55	0.03	0.00	0.00	2.70
Stewart Island shag	1.59	0.00	0.00	0.00	0.01	0.00	1.01	0.00	2.62
Northern giant-petrel	1.65	0.06	0.12	0.32	0.34	0.05	0.00	0.00	2.55
Pitt Island shag	0.00	0.00	0.00	0.01	0.12	0.00	1.51	0.80	2.45
Flesh-footed shearwater	0.72	0.01	0.31	0.08	1.12	0.19	0.00	0.00	2.42
Chatham Island shag	0.00	0.00	0.00	0.02	0.17	0.00	1.51	0.60	2.31
Salvin's albatross	1.43	0.05	0.22	0.22	0.34	0.03	0.00	0.00	2.29
Light-mantled albatross *	1.46	0.02	0.05	0.22	0.34	0.04	0.00	0.00	* 2.14
Northern royal albatross	0.96	0.05	0.03	0.21	0.31	0.54	0.00	0.00	2.09
Campbell albatross	0.61	0.06	0.06	0.24	0.51	0.33	0.00	0.00	1.81
New Zealand storm-petrel	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.51	1.51
Yellow-eyed penguin	0.08	0.00	0.00	0.01	0.02	0.00	1.26	0.00	1.38
Spotted shag	0.47	0.00	0.00	0.00	0.03	0.01	0.75	0.00	1.27
Fiordland crested penguin	0.07	0.00	0.00	0.01	0.33	0.04	0.80	0.00	1.25
Wandering albatross	0.00	0.00	0.00	0.00	0.00	1.21	0.00	0.00	1.21
Southern Buller's albatross	0.36	0.17	0.03	0.38	0.05	0.20	0.00	0.00	1.19
Gibson's albatross	0.23	0.02	0.01	0.07	0.15	0.67	0.00	0.00	1.16
Antipodean albatross	0.22	0.02	0.01	0.06	0.14	0.64	0.00	0.00	1.10
Hutton's shearwaters	0.05	0.00	0.00	0.01	0.04	0.00	1.01	0.00	1.10
Pied shag	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.30	1.06
South Georgia diving-petrel	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.30	0.91
Indian yellow-nosed albatross	0.00	0.00	0.00	0.00	0.00	0.60	0.00	0.00	0.60
White-capped albatross	0.32	0.34	0.02	0.09	0.02	0.01	0.00	0.00	0.79
Sooty shearwater *	0.01	0.01	0.00	0.00	0.00	0.00	0.75	0.00	* 0.77
White-chinned petrel	0.11	0.37	0.01	0.09	0.15	0.03	0.00	0.00	0.77
Fluttering shearwater	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.75
Little black shag	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.75
Northern blue penguin	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.75
White-flippered blue penguin	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.75
Northern Buller's albatross	0.19	0.00	0.05	0.14	0.19	0.17	0.00	0.00	0.75
Southern royal albatross	0.31	0.05	0.02	0.07	0.08	0.21	0.00	0.00	0.74
Cape petrel *	0.27	0.00	0.03	0.14	0.21	0.03	0.00	0.00	* 0.69
Southern giant petrel	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.00	0.15
Chatham Island blue penguin	0.00	0.00	0.00	0.00	0.00	0.00	0.45	0.00	0.45
Grey petrel	0.13	0.00	0.00	0.03	0.08	0.12	0.00	0.00	0.37
Magenta petrel	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.30	0.33

5.4.3.4. Fully quantitative modelling

Fully quantitative population modelling has been conducted only for southern Buller's albatross, black (Parkinson's) petrel, white capped albatross (mollymawk), and Gibson's (wandering) albatross. Data of similar quality and quantity are available for Antipodean (wandering) albatross, and this work should be commissioned soon, but data for other species or populations appear unlikely to be adequate for comprehensive population modelling. The poor estimates of observable and cryptic fishing-related mortality have restricted such work to comprehensive population modelling rather than formal assessment of risk.

5.4.3.4.1. Quantitative models for southern Buller's albatross

Francis *et al.* (2008, see also Francis and Sagar 2012) assessed the status of the Snares Islands population of southern Buller's albatross (*Thalassarche bulleri bulleri*). They estimated (see also Sagar and Stahl 2005) that the adult population had increased about 5-fold since about 1950 (Figure 5.17) at a rate of about 2% per year, and concluded from this that the risk to the viability of this population posed by fisheries had been small. This conclusion depends critically on the reliability of the first census of nesting birds conducted in 1969, but the authors give compelling reasons to trust that information. They noted, however, that population growth had slowed by about 2005 (and perhaps reversed) and adult survival rates were falling, but could discern neither the cause nor significance of these changes because they had included survival data only up to 2007. An additional 5 years of survival and other demographic data have since been recorded (Sagar *et al.* 2010) and all monitored sites at the Snares Islands show substantial declines in the number of breeding pairs since 2006. The modelling has not yet been repeated.

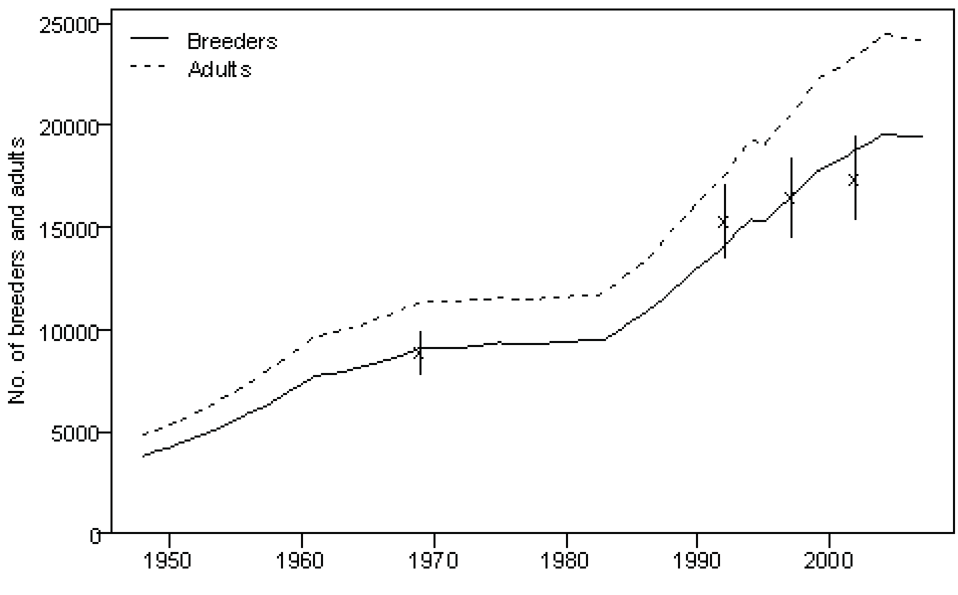


Figure 5.17 (reproduced from Francis *et al.* 2008): Estimates from model SBA21 of numbers of breeders (solid line) and adults (broken line) in each year. Also shown are the census observations (after (Sagar and Stahl 2005) of numbers of breeders (crosses), with assumed 95% confidence intervals (vertical lines).

Fishery discards are an important component of the diet of chicks, but Francis *et al.* (2008) were not able to assess whether the associated positive effect on population growth (e.g., from increased breeding success) is greater or less than the negative effect of fishing-related mortality.

5.4.3.4.2. Quantitative models for black petrel

Francis and Bell (2010) analysed data from the main population of black (Parkinson's) petrel (*Procellaria parkinsoni*), which breeds on Great Barrier Island. Abundance data from transect surveys were used to infer that the population was probably increasing at a rate between 1.2% and 3.1% per year. Mark-recapture data were useful in estimating demographic parameters, like survival and breeding success, but contained little information on population growth rates. Fishery bycatch data from observers were too sparse and imprecise to be useful in assessing the contribution of fishing-related mortality. Francis and Bell (2010) suggested that, because the population was probably increasing, there was no evidence that fisheries posed a risk to the population at that time. They cautioned that this did not imply that there was clear evidence that fisheries do not pose a risk.

Subsequent analysis (Bell *et al.* 2012) included an additional line transect survey in 2009/10 in which the breeding population was estimated to be ~22% lower than in 2004/05 (the latest available to Francis and Bell, 2010). Updating the model of Francis and Bell (2010) made little difference to estimates of demographic parameters such as adult survival, age at first breeding, and juvenile survival (which had 95% confidence limits of 0.67 and 0.91). The uncertainty in juvenile survival gave rise to uncertainty in the estimated population trend, with a mean rate of population growth over the modelling period ranging from -2.5% per year (if juvenile survival = 0.67) to +1.6% per year (if juvenile survival = 0.91, close to the average annual survival rate for older birds) (Figure 5.18). Bell *et al.* (2012) concluded that the mean rate of change of the population over the study period had not exceeded 2% per year, though the direction of change was uncertain.

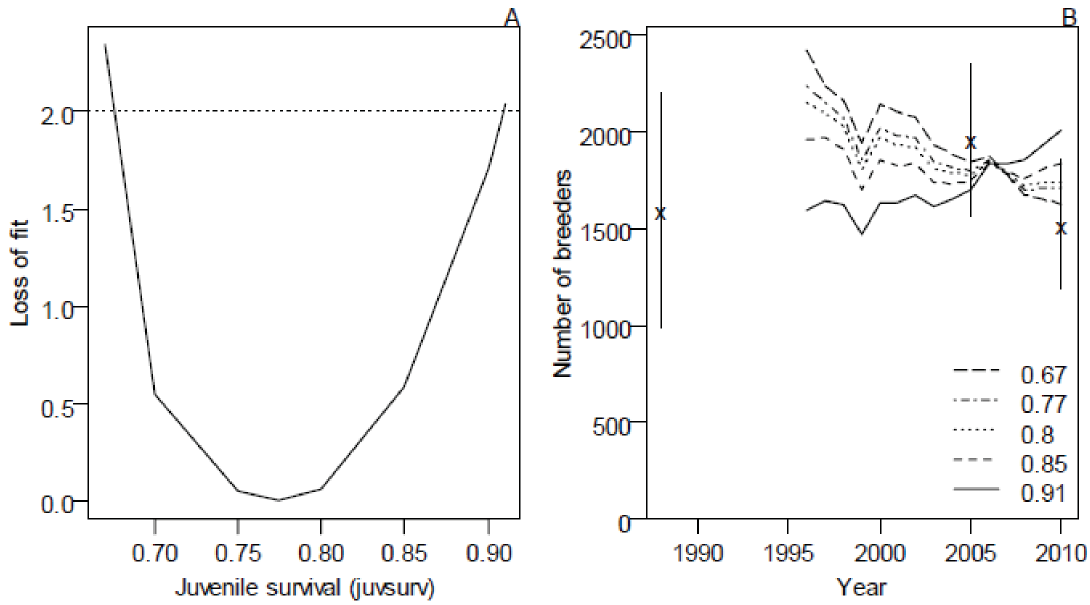


Figure 5.18 (reproduced from Bell *et al.* 2012): Likelihood profile for annual probability of juvenile survival showing: A, the loss of fit (the horizontal dotted line shows a 95% confidence interval for this parameter); and B, population trajectories corresponding to different values of juvenile survival, together with population estimates from transect counts (crosses with vertical lines indicating 95% confidence intervals. Note that the 1988 population estimate was not used in the model.

5.4.3.4.3. Quantitative models for white-capped albatross

Francis (2012) described quantitative models for white-capped albatross (*Thalassarche steadi*), New Zealand's most numerous breeding albatross, and the most frequently captured, focussing on the population breeding at the Auckland Islands. After a correction for a probable bias introduced by sampling at different times of day in one of the surveys, aerial photographic counts by Baker *et al.* (2007, 2008, 2009, and 2010, see also Table 5.15) suggest that the adult population declined at about 9.8% per year between 2006 and 2009. However, this estimate is imprecise and is not easily reconciled with the high adult survival rate (0.96) estimated from mark-recapture data. Francis (2012) also compared the trend with his estimate of the global fishing-related fatalities of white-capped albatross (slightly over 17 000 birds per year, about 30% of which is taken in New Zealand fisheries) and found that fishing-related fatalities were insufficient to account for the number of deaths implied by a decline of 9.8% per year (roughly 22 000 birds per year over the study period). The scarcity of information on cryptic mortality makes these estimates and conclusions uncertain, however.

Table 5.15 (data from Baker *et al.* 2007, 2008, 2009, 2010): Aerial-photographic counts of breeding pairs of white-capped albatrosses on three islands in the Auckland Islands group in December 2006–2009. Confidence limits for these counts published by Baker (*op. cit.*) were based on a Poisson model and were very narrow (although uncertainty introduced by the proportion of non-nesting birds at the colonies during photography was not included).

Year	Disappointment	SW Cape	Adams	Total
2006	110 649	6 548	–	117 197
2007	86 080	4 786	79	90 945
2008	91 694	5 264	131	97 089
2009	70 569	4 161	132	74 862

5.4.3.4.4. Quantitative models for Gibson's albatross

Francis *et al.* (in press) concluded there is cause for concern about status of the population of Gibson's wandering albatross (*Diomedea gibsoni*) on the Auckland Islands. Since 2005, the adult population has been declining at 5.7%/yr (95% c.i. 4.5–6.9%) because of sudden and substantial reductions in adult survival, the proportion of adults breeding, and the proportion of breeding attempts that are successful (Figure 5.19). Forward projections showed that the most important of these to the future status of this population is adult survival (Figure 5.20).

The population in 2011 was 64% (58–73%) of its estimated size in 1991. The breeding population dropped sharply in 2005, to 59% of its 1991 level, but has been increasing since 2005 at 4.2% per year (2.3–6.1%). The 2011 breeding population is estimated to be only 54% of the average of 5831 pairs estimated by Walker & Elliott (1999) for 1991–97.

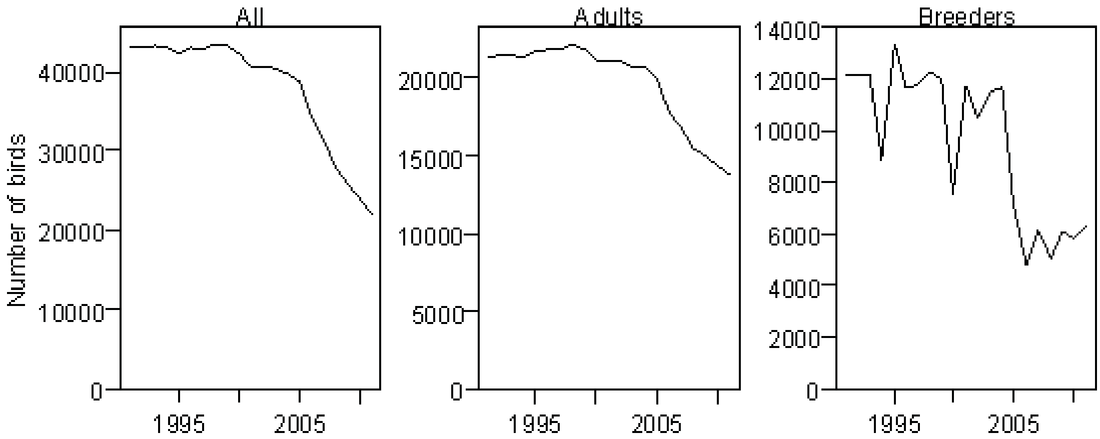


Figure 5.19: Estimated population trajectories for the whole Auckland Islands population of Gibson's wandering albatross. These were calculated by scaling up Francis *et al.*'s (in press) GIB5 trajectories to match the Walker & Elliott (1999) estimate for the whole population.

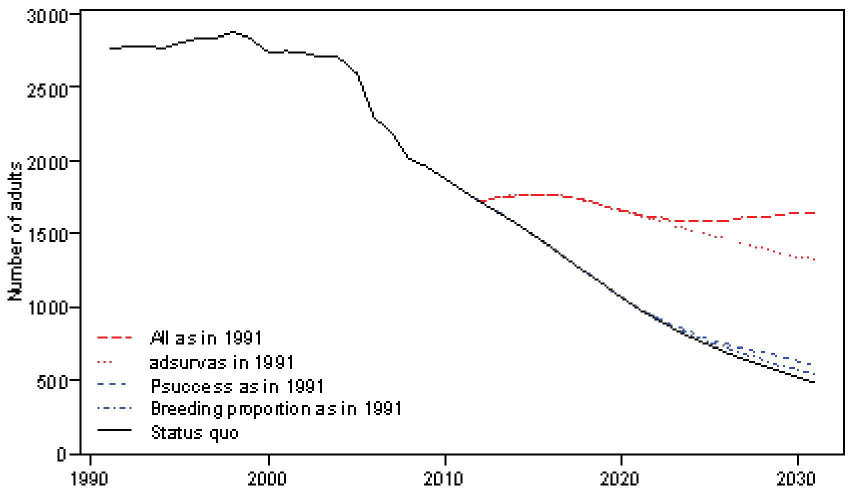


Figure 5.20: Estimated population trajectory for adults from Francis *et al.*'s (in press) model GIB5 with 20-year projections under five alternative scenarios about three demographic parameters: adult survival (adsurv); breeding success (Psuccess); and proportion of adults breeding. These scenarios differ according to whether each parameter remains at its status quo (=2011) level or recovers immediately to its 1991 level.

Francis *et al.* (in press) found it difficult to assess the effect of fisheries mortality on the viability of this population because, although some information exists about captures in New Zealand and Australian waters, the effect of fisheries in international waters is unknown. Three conclusions are possible from the available data: most fisheries mortality of Gibson's is caused by surface longlines; mortality from fishing within the New Zealand EEZ is now probably lower than it was; and there is no indication that the sudden and substantial drops in adult survival, the proportion breeding, and breeding success were caused primarily by fishing.

5.4.3.4.5. Other quantitative models

This section is not intended to cover all quantitative modelling of seabird populations, rather to focus on recent studies that sought to assess the impact of fishing-related mortality.

Maunder *et al.* (2007) sought to assess the impact of commercial fisheries on the Otago Peninsula yellow-eyed penguins using mark-recapture data within a population dynamics model. They found the data available at that time inadequate to assess fisheries impacts, but evaluated the likely utility of additional information on annual survival or an estimate of bycatch for a single year. Including auxiliary information on average survival in the absence of fishing allowed estimation of the fishery impact, but with poor precision. Including an estimate of fishery-related mortality for a single year improved the precision in the estimated fishery impact. The authors concluded that there was insufficient information to determine the impact of fisheries on yellow-eyed penguins and that quantifying fishing-related mortality over several years was required to undertake such an assessment using population a modelling approach.

Fletcher *et al.* (2008) sought to assess the potential impact of fisheries on Antipodean and Gibson's wandering albatrosses (*Diomedea antipodensis antipodensis* and *D. a. gibsoni*); black petrel (*Procellaria parkinsoni*) and southern royal albatross (*Diomedea epomophora*). Because of problems with the available fisheries and biological data, they were unable to use their models to predict the impact of a change in fishing effort on the population growth rate of a given species. Instead, they used the models to estimate the impact that changes in demographic parameters like annual survival are likely to have on population growth rate. They found that: reducing breeder survival rate by k percentage points will lead to a reduction in the population growth rate of about 0.3k percentage points (0.4 for black petrel); and a reduction of k percentage points in the survival rate for <u>each</u> stage in the life cycle (juvenile, pre-breeder, non-breeder and breeder) will lead to a reduction in the population growth rate of approximately k percentage points. Fletcher *et al.* (2008) also made estimates of PBR for 23 New Zealand seabird taxa and summarise and tabulated non-fishing-related threats for 38 taxa.

Newman *et al.* (2009) combined survey data with demographic population models to estimate the total population of sooty shearwaters within New Zealand. They estimated the total New Zealand population between 1994 and 2005 to have been 21.3 (95% c.i. 19.0–23.6) million birds. The harvest of "muttonbirds" was estimated to be 360 000 (320 000–400 000) birds per year, equivalent to 18% of the chicks produced in the harvested areas and 13% of chicks in the New Zealand region. This directed harvest is much larger than estimates of captures in key fisheries (Table 5.4) or potential fatalities in the level 2 risk assessment (Figure 5.16). Newman *et al.* (2009) did not assess the likely impact of fishing-related mortality but concluded that the much larger directed harvest was not an adequate explanation for the observed declines in the past three decades.

5.4.3.4.6. General conclusions from quantitative modelling

Fully quantitative modelling has now been conducted for four of the five seabird populations for which apparently suitable data are available. That modelling suggests very strongly that one population had been increasing steadily (southern Buller's albatross, but note this trend may have reversed) and another is declining quite rapidly (Gibson's albatross). White-capped albatross and black petrel both more likely to be declining than not but, even for these relatively data rich populations, the conclusions are uncertain. General conclusions from the modelling conducted to date, therefore, can be summarised as:

- Very few seabird populations have sufficient data for modelling
- Except for the two most complete data sets (southern Buller's and Gibson's albatross) it has been difficult to draw firm conclusions about trends in population size.

- Information from surveys or census counts is much more powerful for detecting trends in population size than data from the tagging programmes and plot monitoring implemented for New Zealand seabirds to date.
- The available information on incidental captures in fisheries have not allowed rigorous tests of the role of fishing-related mortality in driving population trends
- Although comprehensive modelling provides additional information to allow interpretation, we will have to rely on level 2 risk assessment approaches for much of our understanding of the relative risks faced by different seabird taxa and posed by different fisheries.

5.4.3.5. Sources of uncertainty in risk assessments

There are several outstanding sources of uncertainty in modelling the effects of fisheries interactions on sea birds, especially for the complete assessment of risk to individual seabird populations.

5.4.3.5.1. Scarcity of information on captures and biological characteristics of affected populations

These sources of uncertainty can be explored within the analytical framework of the level 2 risk assessment (Richard *et al.* 2011), noting that the results of that exploration are constrained by the structure of that analysis. Richard *et al.* (2011) provided plots of such an exploration for four example taxa (Figure 5.21). It can be concluded from this analysis that substantially more precise estimates of risk would be available for black petrel and Stewart Island shag if better estimates of potential captures were available. Conversely, substantially more precise estimates of risk would be available for Salvin's albatross and flesh-footed shearwater if better estimates of average adult survival were available. This analysis is a powerful way of assessing the priorities for collection of new information, including research.

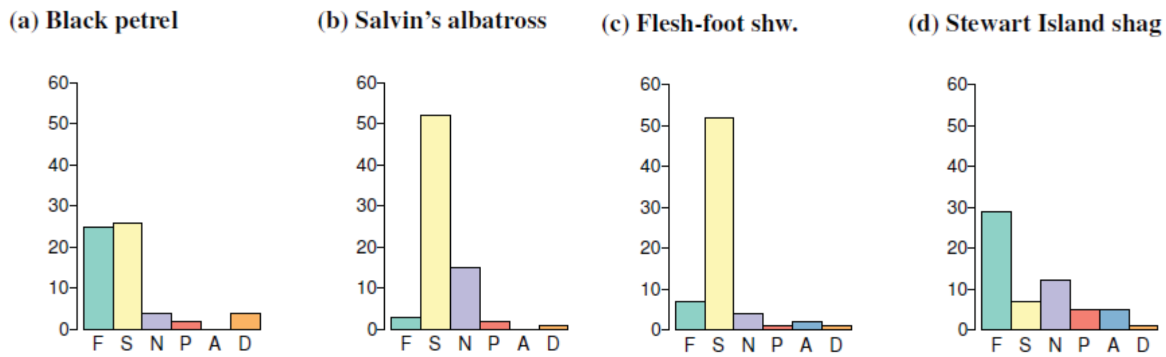


Figure 5.21 (reproduced from Richard *et al.* 2011): Sensitivity of the risk ratio to the uncertainty in the mean number of annual potential fatalities (F, reflecting the uncertainty in vulnerability), the adult annual survival rate (S), the number of annual breeding pairs (N), the proportion of adults breeding in a given year (P), the age at first reproduction (A), and to the distribution map (D), for the taxa most at risk (lower bound of the 95% c.i. above 1). This sensitivity is expressed as the percentage reduction in the 95% confidence interval of the risk ratio when each parameter is fixed to its mean.

5.4.3.5.2. Scarcity of information on cryptic mortality

Cryptic mortality is particularly poorly understood but has substantial influence on the results of the risk assessment. Richard *et al.* (2011) provided a description of the method used to incorporate cryptic mortality into their estimates of potential fatalities in the level-2 risk assessment (their Appendix B authored by B. Sharp, MPI). This method builds on the published information from Brothers *et al.* (2010) for longline fisheries and Watkins *et al.* (2008) and Abraham (2010) for trawl fisheries. Brothers *et al.* (2010) observed almost 6 000 seabirds attempting to take longline baits during line setting, of which 176 (3% of attempts) were seen to be caught. Of these, only 85 (48%) were retrieved during line hauling. They concluded that using only observed captures to estimate seabird fatalities grossly underestimates actual levels in pelagic longline fishing. Similarly, Watkins *et al.* (2008) observed 2454 interactions between seabirds and trawl warps in the South African hake fishery over 189.8 hours of observation. About 11% of those interactions (263) involved birds, mostly albatrosses, being dragged under the water by the warps, and 30 of those submersions were observed to be fatal. Of the 30 birds observed killed on the warps, only two (both albatrosses) were hauled aboard and would have been counted as captures by an observer in New Zealand. Aerial collisions with the warps were about 8 times more common but appeared mostly to have little effect (although one white-chinned petrel suffered a broken wing which would almost certainly have fatal consequences).

Given the relatively small sample sizes in both of these trials, there is substantial (estimatable) uncertainty in the estimates from the trials themselves and additional (non-estimatable) uncertainty related to the extent to which these trials are representative of all fishing of a given type, particularly as both trials were undertaken overseas. The binomial 95% confidence range (calculated using the Clopper-Pearson “exact” method) for the ratio of total fatalities to observed captures in Brothers *et al.*’s (2010) longline trial is 1.8–2.5 (mean 2.1), and that for Watkins *et al.*’s trawl warp trial is 5–122 (mean 15.0 fatalities per observed capture). Abraham (2010) estimated that there were 244 (95% c.i. 190–330) warp strikes by large birds for every one observed captured, and 6 440 (3400–20 000) warp strikes by small birds for every one observed captured (although small birds tend to be caught in the net rather than by warps). There is also uncertainty in the relative frequencies and consequences of different types of encounters with trawl warps in New Zealand fisheries (Abraham 2010, Richard *et al.* 2011 Appendix B).

5.4.3.5.3. Mortalities in non-commercial fisheries.

Little is known about the nature and extent of incidental captures of seabirds in non-commercial fisheries, either in New Zealand or globally (Abraham *et al.* 2010). In New Zealand, participation in recreational fishing is high and 2.5% of the adult population are likely to be fishing in a given week (mostly using rod and line). Because of this high participation rate, even a low rate of interactions between individual fishers and seabirds could have population-level impacts. A boat ramp survey of 765 interviews at two locations during the summer of 2007–08 revealed that 47% of fishers recalled witnessing a bird being caught some time in the past. Twenty-one birds were reported caught on the day of the interview at a capture rate of 0.22 (95% c.i.: 0.13–0.34) birds per 100 hours of fishing. Observers on 57 charter trips recorded seabird captures at rate of 0.36 (0.09–0.66) birds per 100 fisher hours. The most frequently reported type of bird caught in rod and line fisheries were petrels and gulls. Captures of albatrosses, shags, gannets, penguins, and terns were also recalled.

The ramp surveys reported by Abraham *et al.* (2010) were limited and covered only two widely-separated parts of the New Zealand coastline. However, they also report two other pieces of information that suggest non-commercial captures are likely to be very widespread. First, the Ornithological Society of New Zealand’s beach patrol scheme records seabird hookings and entanglements as a common occurrence throughout New Zealand. Second, returns of banded birds caught in fisheries (separating commercial and non-commercial fisheries is very difficult) are very widely distributed around the coast (Figure 5.22).

Noting that our understanding of seabird capture rates in amateur fisheries is very sketchy, it is possible to make first-order estimates of total captures using information on fishing effort. For example, in the north-eastern region where most of Abraham *et al.*’s (2010) interviews were conducted, there were an estimated 4.8 (4.4–5.2) million fisher hours rod and line fishing from trailer boats in 2004–05 (Hartill *et al.* 2007). Applying Abraham *et al.*'s (2010) capture rate leads to an estimate of 11 500 (6600–17 200) captures per year in this area. Based on estimates of nationwide recreational fishing effort, this could increase to as many as 40 000 bird captures annually. Most birds captured by amateur fishers were reported to have been released unharmed (77% of the incidents recalled) and only three people reported incidents where the bird died. Because of likely recall biases and the qualitative nature of the survey, the fate of birds that are captured by amateur fishers remains unclear.

Non-commercial fishers are allowed to use setnets in New Zealand and two studies suggest that these have an appreciable bycatch of seabirds. A study of captures in non-commercial setnets in Portobello Bay, Otago Harbour, between 1977 and 1985 (Lalas 1991) suggested spotted shags were the most frequently caught taxa (82 recorded, compared with 14 Stewart Island shags and two little shags). Lalas (1991) suggested that up to 800 spotted shags (20% of the local population) may have been caught in the summer of 1981/82. A broader-scale study of yellow-eyed penguin mortality in setnets in southern New Zealand (Darby and Dawson 2000) suggested non-negligible captures of this species by non-commercial fishers, also reporting other seabirds like spotted shags and little blue penguin.

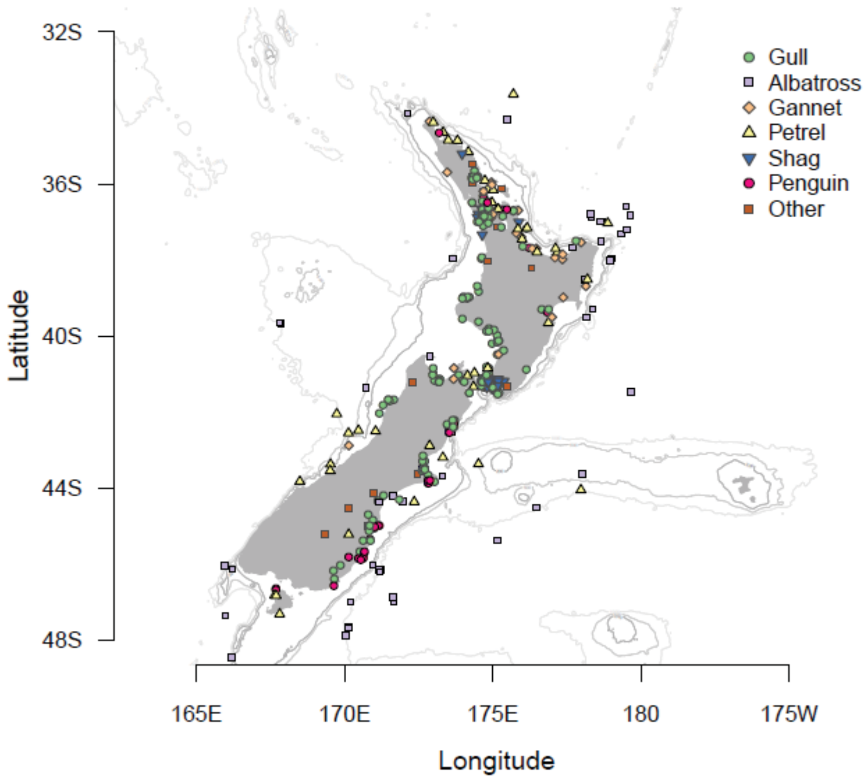


Figure 5.22 (reproduced from Abraham *et al.* 2010): Distribution of the reported capture locations for banded seabirds reported as being captured in fishing gear, 1952–2007. Note, band recovery locations are reported with low spatial precision and some of the inland locations may be correct.

5.4.3.5.4. Out of zone mortality.

Robertson *et al.* (2003) mapped the distribution of the 25 breeding (mainly endemic) New Zealand seabird taxa they considered most at risk outside New Zealand waters. These ranged widely: 4 used the South Atlantic; 4 the Indian Ocean; 22 Australian waters and the Tasman Sea; 15 used the South Pacific Ocean as far afield as Chile and Peru; and 6 used the North Pacific Ocean as far north as the Bering Sea. These taxa therefore use the national waters of at least 18 countries. For example, the level-2 risk assessment described by Richard *et al.* (2011) includes only that part of the range of each taxon contained within New Zealand waters, but many including commonly-caught seabirds like white-capped albatross and white-chinned petrel range much further and are vulnerable to fisheries in other parts of the world. For instance, fatalities of white-capped albatross outside the New Zealand EEZ greatly exceed fatalities within the zone (Baker 2007, Francis 2012, Table 5.16), and more than 10 000 white-chinned petrel are killed off South America each year (Phillips *et al.* 2006), noting that reliable records are not available for most of the fisheries involved. Based on similar analyses, Moore and Zydelis (2008) concluded that a population-based, multi-gear and multi-national framework is required to identify the most significant threats to wide-ranging seabird populations and to prioritize mitigation efforts in the most problematic areas. To that end, the Agreement for the Conservation of Albatrosses and Petrels (ACAP) adopted a global prioritisation framework at the Fourth Session of the Meeting of the Parties (MoP4) in April 2012 (ACAP 2012).

Table 5.16 (after Francis 2012): Estimates of the number of white-capped albatrosses killed annually, by fishery. The first two columns are from Baker *et al.* (2007) (mid-point where a range was presented), including their assessment of reliability (L = low, M-H = medium-high, H = high). Updated estimates are from Watkins *et al.* (2008, *) and Petersen *et al.* (2009, **). Estimates not already corrected for cryptic mortality are either doubled to allow for this (***) or replaced by estimates of potential fatalities from Richard *et al.* (2011, *), noting that potential fatalities may considerably overestimate actual fatalities.**

Fishery	From Baker *et al.* 2007		Updated	Incl. Cryptic mortality
South African demersal trawl	4 750	(L)	* 6650	6 650
Asian distant-water longline	1 255	(L)	–	*** 2 510
Namibian demersal trawl	910	(L)	* 1270	1 270
Namibian pelagic longline	180	(L)	** 195	*** 390
NZ hoki and squid trawl	513	(MH)	–	**** 4 920
NZ longline	60	(MH)	–	**** 199
Australian (line fisheries)	15	(MH)	–	*** 30
South African pelagic longline	570	(H)	** 570	*** 1 140
Total	8 210	–	–	17 110

5.4.3.5.5. Other sources of anthropogenic mortality.

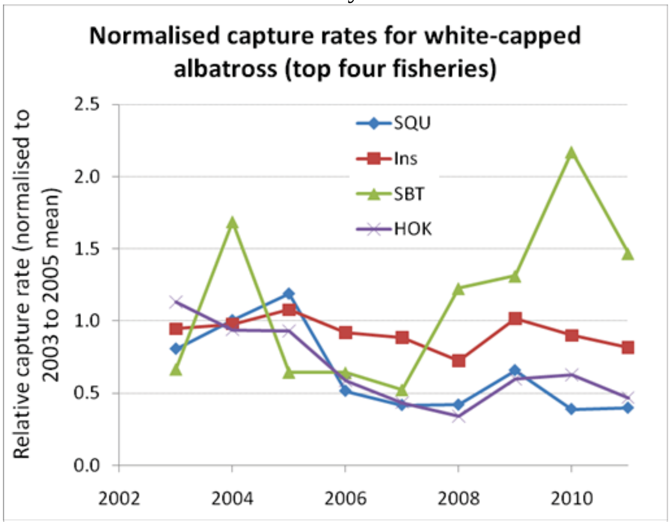
Taylor (2000) listed a wide range of threats to New Zealand seabirds including introduced mammals, avian predators (weka), disease, fire, weeds, loss of nesting habitat, competition for nest sites, coastal development, human disturbance, commercial and cultural harvesting, volcanic eruptions, pollution, plastics and marine debris, oil spills and exploration, heavy metals or chemical contaminants, global sea temperature changes, marine biotoxins, and fisheries interactions. Relatively little is known about most of these factors, but the parties to ACAP have agreed a formal prioritisation process to address and prioritise major threats (ACAP 2012). Croxall *et al.* (2012) identified the main priorities as: protection of Important Bird Area (IBA) breeding, feeding, and aggregation sites; removal of invasive, especially predatory, alien species as part of habitat and species recovery initiatives. Lewison *et al.* (2012) identified similar research priorities (in addition to direct fishing-related mortality), including: understanding spatial ecology; tropho-dynamics; response to global change; and management of anthropogenic impacts such as invasive species, contaminants, and protected areas. Non fishing-related threats to seabirds in New Zealand are largely the mandate of the Department of Conservation and a detailed description is beyond the scope of this document (although causes of mortality other than fishing are clearly relevant to the interpretation of risk assessment restricted to the direct effects of fishing). These threats are identified in DOC’s Action Plan for Seabird Conservation in New Zealand (Taylor 2000).

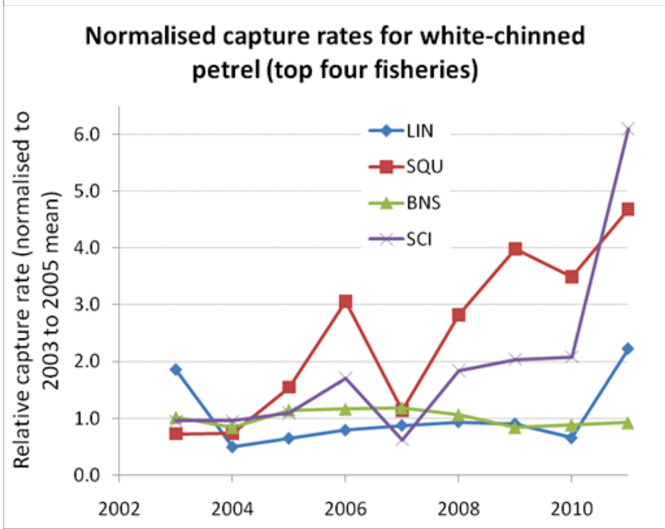
5.5. *Indicators and trends*

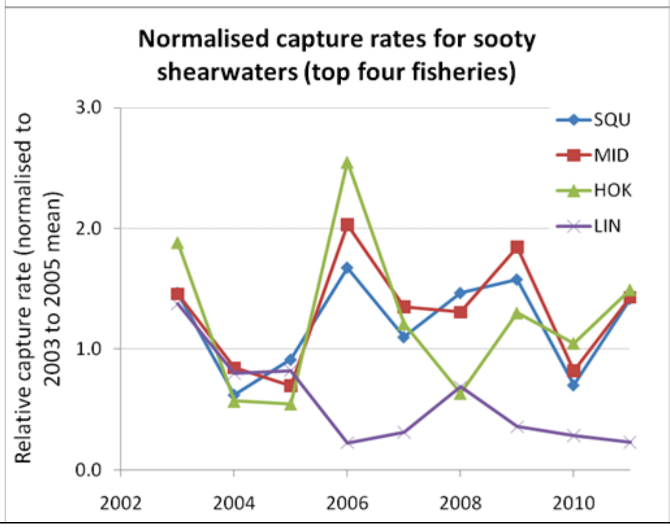
<table>
<tr><td>Population size</td><td>Multiple species and populations: see Taylor (2000)</td></tr>
<tr><td>Population trend</td><td>Multiple species and populations: see Taylor (2000)</td></tr>
<tr><td>Threat status</td><td>Multiple species and populations: see Miskelly et al. (2008) and updates</td></tr>
<tr><td>Number of interactions</td><td>In the 2010/11 October fishing year, there were an estimated 4931 seabird captures (excluding cryptic mortalities) across all trawl and longline fisheries (excluding about 14% of bottom longline effort that could not be included in the models) (Data version v20121101). About 57% of the captures were in trawl fisheries, 15% in surface longline fisheries, and 28% in bottom longline fisheries:
<table>
<tr><th>Bird group</th><th>Trawl</th><th>Surface longline</th><th>Bottom longline</th><th>All these methods</th></tr>
<tr><td>White-capped albatross</td><td>356</td><td>84</td><td>2</td><td>442</td></tr>
<tr><td>Other albatrosses</td><td>808</td><td>287</td><td>257</td><td>1 352</td></tr>
<tr><td>White-chinned petrel</td><td>540</td><td>34</td><td>422</td><td>996</td></tr>
<tr><td>Sooty shearwater</td><td>488</td><td>2</td><td>69</td><td>559</td></tr>
<tr><td>Other birds</td><td>596</td><td>333</td><td>652</td><td>1 581</td></tr>
<tr><td>All birds combined</td><td>2 788</td><td>740</td><td>1 403</td><td>4 931</td></tr>
</table>
</td></tr>
<tr><td>Trend in interactions</td><td>Captures of all birds combined show a decreasing trend between 2002/03 and 2010/11 (Data version v20121101) but there are substantial differences in trends between species and fisheries. Captures of white-capped albatross have decreased, especially in offshore trawl fisheries, whereas captures of white-chinned petrel have increased:

Capture rate trends (excluding cryptic mortalities) are described for the four fisheries estimated to account for most of captures of a species (accounting for 80% or more of the total). Capture rates of white-capped albatross have fallen in trawl fisheries for</td></tr>
</table>

Trend in interactions contd.	hoki and squid but have remained steady in inshore trawl fisheries and increased in the southern bluefin tuna fishery. Capture rates for white-chinned petrel have increased in trawl fisheries for squid and scampi but have remained steady in longline fisheries. Capture rates of sooty shearwater have declined in the ling longline fishery but have fluctuated without trend in other key fisheries.







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THEME 2: NON-PROTECTED BYCATCH

6. Non-protected species (fish and invertebrates) bycatch

Scope of chapter	This chapter outlines the main non-protected bycatch species (fish and invertebrates) and annual levels and trends in bycatch and discards in New Zealand's major offshore fisheries. Research in this field is conducted fishery by fishery and this first summary of current knowledge reflects that strategy. It is expected that future summaries will be aligned to the general format used in other sections of this report, and be based on fishing method, habitat type, region, or a combination of these. The fisheries summarised are as follows: *Trawl fisheries*: Arrow squid; Hoki/hake/ling; Jack mackerel; Southern blue whiting; Orange roughy; Oreo; Scampi *Longline fisheries*: Ling; Tuna *Other fisheries*: Albacore troll; Skipjack purse seine
Area	All areas and fisheries
Focal localities	*Arrow squid*: Auckland Islands and Stewart/Snares Shelf (80–300 m). *Hoki/hake/ling*: Chatham Rise, West Coast South Island, Campbell Plateau, Puysegur Bank, and Cook Strait (200–800 m). *Jack mackerel*: West Coast of the North and South Islands, Chatham Rise, and Stewart-Snares Shelf (0–300 m). *Southern blue whiting*: Campbell Plateau and Bounty Plateau (250–600 m). *Orange roughy*: The entire New Zealand region (700–1200 m). *Oreos*: South Chatham Rise, Pukaki Rise, Bounty Plateau, and Southland (700–1200 m). *Scampi*: East coasts of the North and South Islands, Chatham Rise, and Auckland Islands (300–450 m). *Ling longline*: Chatham Rise, Bounty Plateau, and Campbell Plateau (150–600 m). *Tuna longline*: Surface waters off the east coast of the North Island and west coast of the South Island. *Albacore troll fishery:* Surface waters off the west coasts of the North and South Islands. *Skipjack purse seine fishery:* Northern North Island

Key issues	Under-utilisation (including shark finning) of high volume, low value bycatch species, especially rattails, spiny dogfish, deepsea sharks, blue sharks, porbeagle sharks, and swimming crabs. Potential for considerable reduction of discards by discretionary fishing practices such as the use of mid-water nets, where practicable, and meal plants. Unseen mortality in longline fisheries due to predation by large fish and sharks, marine mammals, seabirds, and sea lice.
Emerging issues	Trends of increasing rates and levels of bycatch and discarding in several categories of catch, especially non-QMS fish species and invertebrates. The effect on bycatch rates in the ling longline fishery of a change to heavier fishing gear (including integrated weights) as used in the Antarctic toothfish fishery. Increasing trawl lengths in the squid, scampi, and orange roughy fisheries due to changes in fishing gear or reduction of target species catch rates—leading to greater bycatch levels in some categories.
MPI Research (current)	DAE201002 (bycatch and discards in deepwater fisheries) DEE201004 (ecological risk assessment in deepwater fisheries) DEE201005A (environmental indicators in deepwater fisheries) HMS200901 (bycatch in tuna longline fisheries)
Other Govt Research (current)	None
Links to 2030 objectives	Objective 6: Manage impacts of fishing and aquaculture.
Related chapters/issues	NPOA sharks

6.1. *Context*

Management of non-protected species bycatch aligns with Fisheries 2030 Objective 6: *Manage impacts of fishing and aquaculture*.

The management of non-protected species bycatch in the deepwater and middle-depth fisheries is described in the National Fisheries Plan for Deepwater and Middle-depth Fisheries (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of non-protected species bycatch is Management Objective 2.4: *Identify and avoid or minimise adverse effects of deepwater and middle-depth fisheries on incidental bycatch species.* Specific objectives for the management of non-protected species bycatch will be outlined in the fishery-specific chapters of the National Deepwater Plan. Estimation of non-protected species bycatch is carried out for each of the Tier-1 deepwater fisheries on an annual rotational basis, with each of the following fisheries updated about every 4–5 years:

- Arrow squid
- ling bottom longline
- hoki/hake/ling trawl
- Jack mackerel trawl
- southern blue whiting trawl
- orange roughy/oreo trawl
- scampi trawl

Non-protected fish species bycatch in the Highly Migratory Species (HMS) is addressed in the HMS fish plan. Tuna fisheries incidental bycatch has been regularly examined, with updates every 2–3 years. Some data on bycatch in the Albacore troll fishery and the skipjack tuna purse seine fishery are also available.

The three National Fisheries Plans for Inshore species (finfish, shellfish and freshwater fisheries) also include objectives which address non-protected species bycatch, but research on these objectives has yet to be conducted. However, summaries of the main bycatch species are occasionally included in reports from fisheries characterisation projects, for example school shark, red gurnard, and elephantfish (Starr In Prep; Starr *et al.* 2010a, b, c, Starr & Kendrick 2012).

6.2. *Global understanding*

Bycatch of unwanted, low value species and discarding of these and of target species that are damaged or too small to process are significant issues in many fisheries worldwide. Few, if any, fisheries are completely without bycatch and this issue has been the subject of innumerable studies and international meetings. Saila (1983) made the first comprehensive global assessment and estimated, albeit with very poor information, that at least 6.7 million tonnes was discarded each year. Alverson *et al.* (1994) extended that work and estimated the global bycatch at 27.0 (range 17.9–39.5) million tonnes each year. An update by Kelleher (2005) suggested global bycatch of about 8% of the global catch, or 7.3 million tonnes, in 1999–2001.

Tropical shrimp trawl fisheries typically have the highest levels of unwanted bycatch, with an average discard rate of 62% (Kelleher 2005), accounting for about one-quarter to one-third of global bycatch. Discard rates in demersal trawl fisheries targeting finfish are typically much lower but, because they are so widespread, their contribution to global discards is considerable. Tuna longline fisheries have the next largest contribution and tend to have greater unwanted bycatch than other line fisheries (Kelleher 2005).

The estimated global level of discards has reduced considerably since the first estimates were made, but differences in the methodology and definition of bycatch used (Kelleher 2005, Davies *et al.* 2009) make it difficult to quantify the decline. The main reasons for the decline in bycatch are thought to have been a combination of higher retention rates, better fisheries management, and improved fishing methods.

Bycatch and discard estimation is frequently very coarse, and estimates of rates based on occasional surveys are often scaled up to represent entire fisheries and applied across years, or even to other fisheries (e.g., Bellido *et al.* 2011). Data from dedicated fisheries observers are also frequently used for individual fisheries, and these are considered to provide the most accurate results, providing that discarding is not illegal (leading to bias due to “observer effects”, Fernandes 2011). Ratio estimators similar to those applied in New Zealand fisheries are frequently used to raise observed bycatch and discard rates to the wider fishery, and the methods used in New Zealand fisheries are broadly similar to those used elsewhere (e.g., Fernandes 2011, Borges *et al.* 2005).

Discard data are increasingly incorporated into fisheries stock assessments and management decision-making, especially with the move towards an Ecosystem Approach to Fisheries (EAF) (Bellido *et al.* 2011), and as third party fishery certification schemes examine more closely the effects of fishing on the ecosystem. They can also be used to assess impacts on non-target species (e.g., Pope *et al.* 2000, Casini *et al.* 2003, Piet *et al.* 2009).

6.3. *State of knowledge in New Zealand*

Estimation of annual bycatch and discard levels of non-protected species in selected New Zealand fisheries have been undertaken at regular intervals since 1998 (Table 6.1).

Table 6.1: Summary of research into bycatch in New Zealand fisheries

Fishery	Report
Arrow squid	Anderson *et al.* (2000) Anderson (2004b) Ballara and Anderson (2009) Anderson (In Press)
Ling bottom longline	Anderson *et al.* (2000) Anderson (2008)
Hoki trawl	Clark *et al.* (2000) Anderson *et al.* (2001) Anderson and Smith (2005) Ballara *et al.* (2010)
Hake trawl	Ballara *et al.* (2010)
Ling trawl	Ballara *et al.* (2010)
Jack mackerel trawl	Anderson *et al.* (2000) Anderson (2004b) Anderson (2007)
Southern blue whiting trawl	Clark *et al.* (2000) Anderson (2004a) Anderson (2009b)
Orange roughy	Clark *et al.* (2000) Anderson *et al.* (2001) Anderson (2009a) Anderson (2011)
Oreo trawl	Clark *et al.* (2000) Anderson (2004a) Anderson (2011)
Scampi trawl	Anderson (2004b) Ballara and Anderson (2009)
Tuna longline	Francis *et al.* (1999a, 1999b) Ayers *et al.* (2004) Francis *et al.* (2004) Griggs *et al.* (2007) Griggs *et al.* (2008) Griggs & Baird (In Press)
Albacore troll fishery	Griggs *et al.* (In Press)
Skipjack purse seine fishery	Griggs (unpublished data)

The estimation process uses rates of bycatch and discards in various categories (in most cases “all QMS species combined”, “all non-QMS species combined”, “all invertebrate species combined”) and fishery strata in the observed fraction of the fishery, and effort statistics from the wider fishery, to calculate annual bycatch and discard levels. This ratio-based approach calculates precision by incorporating a multi-step bootstrap algorithm which takes into account the effect of correlation between trawls in the same observed trip and stratum. Estimates of the annual bycatch of a wide range of individual species were also made in the most recent analysis of the arrow squid fishery (Anderson In Press).

The approach used in these analyses relies heavily on an appropriate level and spread of observer effort being achieved, and this is examined in detail in each report. Although details of bycatch and discards are also recorded directly by vessel skippers for the entire fishery, through catch effort forms, these data are often incomplete as the forms list only the top 5 catch species, discards are not well recorded, and they generally lack the accuracy and precision of observer data. Nevertheless, annual bycatch totals are also derived from these data, but only as secondary estimates.

6.3.1. Arrow squid trawl fishery

Since 1990–91 the level of observer coverage in this fishery has ranged from 6% to 53% of the total annual catch, and has been higher in more recent years due to the management measures imposed for the protection of New Zealand sealions (*Phocarctos hookeri*). This coverage has been spread across the fleet and annually 10–68% of all vessels targeting arrow squid have been observed, with this fraction increasing over time. Observers have covered the full size range of vessels operating in the fishery, although the smallest vessels have been slightly undersampled and the largest oversampled.

The observer effort was mostly focussed on the main arrow squid fisheries around the Auckland Islands and Stewart-Snares Shelf, but the smaller fisheries on the Puysegur Bank and off Banks Peninsula were also covered, although less consistently. Observer coverage was more focussed on the central period of the arrow squid season, February to April, than the fleet was in general – with fishing in January and May slightly undersampled.

Appropriate stratification for the analyses was determined using linear mixed-effect models (LMEs) to identify key factors influencing variability in the observed rates of bycatch and discarding. This approach addresses the significant vessel-to-vessel and trip-to-trip differences in bycatch and discard rates in this fishery by treating the trip variable as a random effect (whereby the trip associated with each record is assumed to be randomly selected from a population of trips) and treating other variables as fixed effects. This process consistently identified the separate fishery areas (Auckland Islands, Stewart-Snares Shelf, Puysegur Bank, Banks Peninsula) as having the greatest influence on bycatch and discard rates (with trawl duration of secondary importance) and so area was used in all cases to stratify the calculation of annual levels.

Since 1990–91, over 470 bycatch species or species groups have been identified by observers in this fishery, most being non-commercial species (including invertebrate species) caught in low numbers. Arrow squid have accounted for about 80% of the total estimated catch recorded by observers. The main bycatch species or species groups were the QMS species barracouta (8.5%), silver warehou (2.5%), spiny dogfish (1.7%), and jack mackerel (1.1%); of these only spiny dogfish were mostly discarded (Figure 6.1).

Of the other invertebrate groups crabs (0.8%), in particular smooth red swimming crabs (*Nectocarcinus bennetti*) (0.5%), were caught in the greatest amounts and these were mostly discarded. Smaller amounts of octopus and squid, sponges, cnidarians, and echinoderms were also often caught and discarded.

When combined into broader taxonomic groups, bony fish (excluding rattails, tuna, flatfish, and eels) contributed the most bycatch (16.5% of the total catch), followed by sharks and dogfish (1.9%), crustaceans (0.8%), and rattails (0.2%). The combined bycatch of all other fish (tuna, rays & skates, chimaeras, flatfish, and eels) accounted for a further 0.5% of the total catch.

More than 75% of the sharks & dogfish, rattails, and eels were discarded, whereas about half the flatfish were retained, as were most of the tuna, rays & skates, chimaeras, and other fish not in any of these groups. The fish species discarded in the greatest amounts were spiny dogfish, redbait, rattails, and silver dory. Of the invertebrates, virtually all the echinoderms, other squid, sponges, cnidarians, and polychaetes were discarded, but crustaceans, octopuses, and other molluscs were often retained.

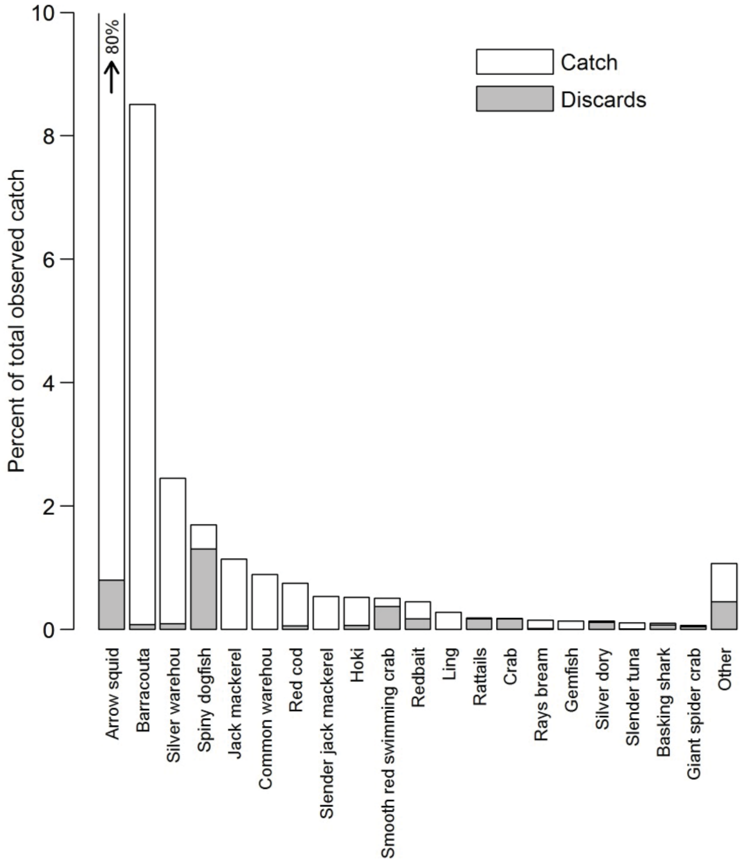

Figure 6.1: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch) in the observed portion of the arrow squid fishery, and the percentage discarded. The "Other" category is the sum of all bycatch species representing less than 0.05% of the total catch.

Total annual bycatch in the arrow squid fishery ranged from about 4500 t to 25 000 t, with low levels in the early 1990s and after 2007–08, and a peak in the early 2000s (Figure 6.2). The large majority of the bycatch comprised QMS species, with less than 1000 t of non-QMS species and invertebrate species bycatch in most years.

Estimated total annual discards ranged from just over 200 t in 1995–96 to about 5500 in 2001–02 and, like bycatch, peaked in the early 1990s and were at relatively low levels after 2006–07 (Figure 6.3). The majority of discards were QMS species (about 62% over all years), followed by non-QMS species (19%), invertebrate species (11%), and arrow squid (7%).

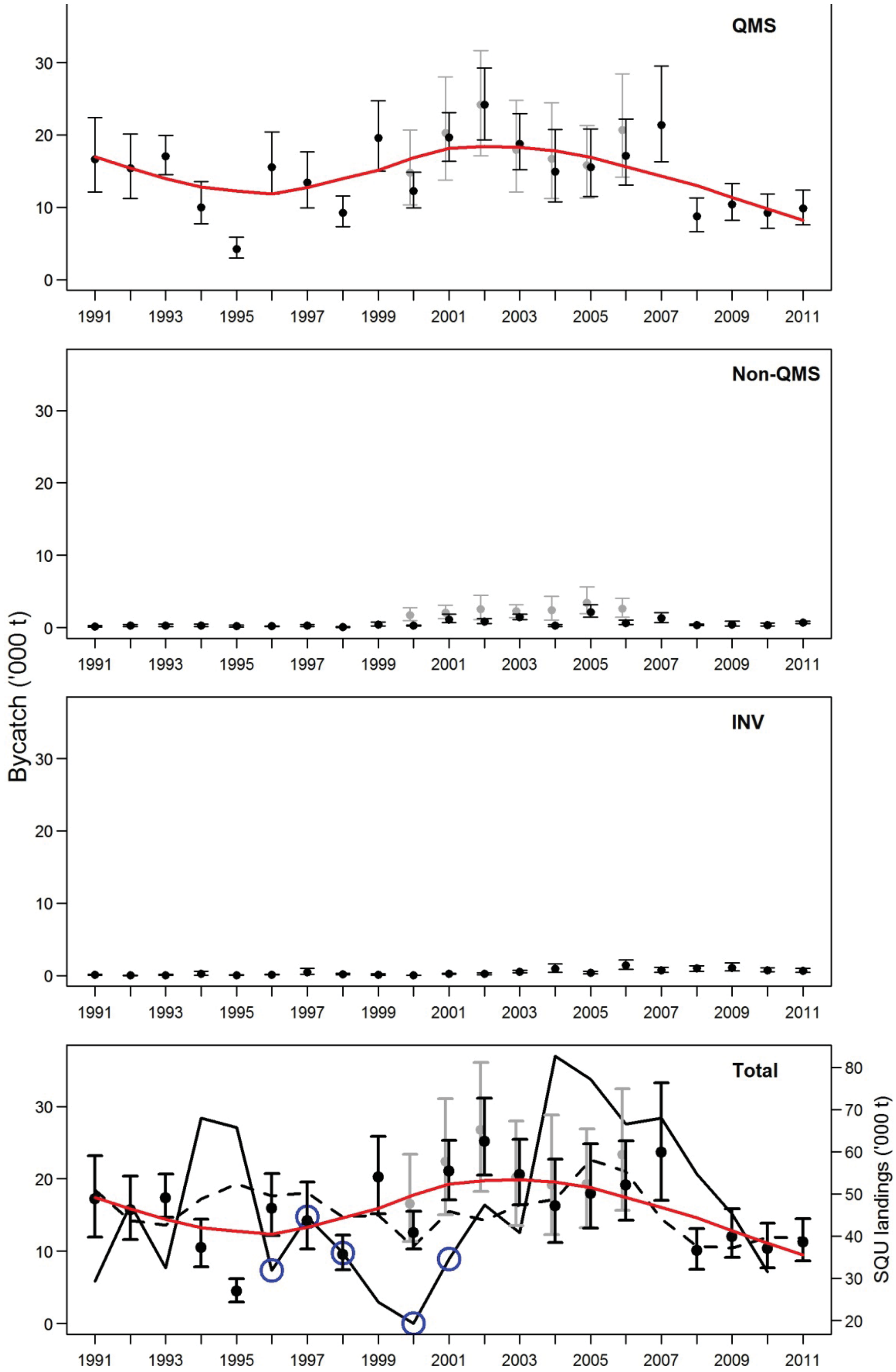

Figure 6.2: Annual estimates of bycatch in the arrow squid trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 1990–91 to 2010–11. Also shown (in grey) are estimates of bycatch in each category (excluding INV) calculated for 1999–2000 to 2005–06 (Ballara & Anderson 2009). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally-weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported trawl-caught landings of arrow squid (Ministry of Fisheries 2011), with circles indicating years in which the fishery closed early after reaching the sea lion FRML; and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

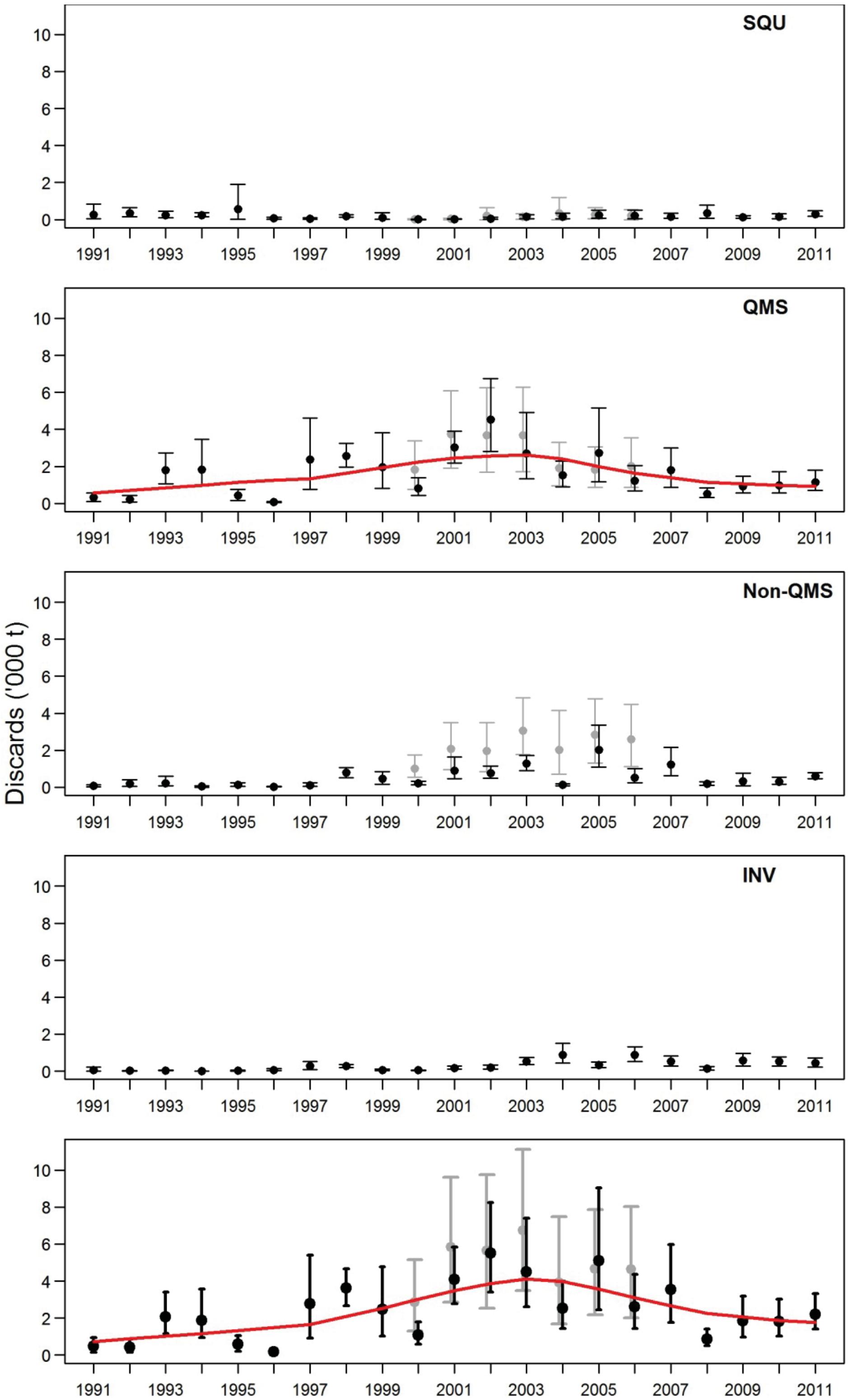

Figure 6.3: Annual estimates of discards in the arrow squid trawl fishery, for arrow squid (SQU), QMS species, non-QMS species, invertebrates (INV), and overall for 1990–91 to 2010–11. Also shown (in grey) are estimates of discards in each category (excluding INV) calculated for 1999–2000 to 2005–06 (Ballara & Anderson 2009). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally-weighted polynomial regression to annual discards.

6.3.2. Ling longline fishery

The first analysis of bycatch and discards in this fishery covered the period from 1990–91 to 1997–98, and the second (and latest) analysis covered the following years up to 2005–06. To enable a comparison of estimates between studies, which used slightly different methodologies, the 1994–95 fishing year was re-assessed in the recent analysis. In addition to estimating the bycatch of all quota species combined, and all non-quota species combined, in the recent analysis annual bycatch was estimated separately for three commonly caught individual species, spiny dogfish, red cod, and ribaldo. Comparative estimates of only total annual bycatch are available from the first analysis for 1990–91 to 1997–98.

The ratio estimator used in these analyses to calculate bycatch and discard rates was based on the number of hooks set. The ratios were applied to hook number totals calculated from commercial catch-effort data to make annual estimates for the target fishery as a whole.

Regression tree methods were used to minimise the number of levels of season and area variables used to stratify data for the calculation of annual discard bycatch totals in all categories with minimal loss of explanatory power. This reduced the number of areas in each category from eight down to between two and four, and split the year into three or four periods. The area variables created in this way tended to have more explanatory power. .

Between 1998–99 and 2005–06 only 9% of the vessels operating in this fishery were observed (14 vessels in all) but these tended to be the main operators (including most of the larger autoliners) and accounted for between 7.7% and 52.5% of the annual target ling catch and 7.8% to 61% of the annual number of longlines set during these years. The annual number of observed sets ranged from 324 to 1605 compared with the total target fishery effort of about 2500 to 4150 sets. Observer coverage before 1998–99 was very low, exceeding 5% of the annual target ling catch only in 1994–95 and 1996–97.

Ling accounted for 68% of the total estimated catch from all observed sets targeting ling between 1998–99 and 2005–06, and spiny dogfish accounted for about a further 14%. About half of the remaining 18% of the catch comprised other commercial species; especially red cod (*Pseudophycis bachus*), (2.3%), ribaldo (*Mora moro*) (2.2%), rough skates (*Zearaja nasuta*, 1.9%), smooth skates (*Dipturus innominatus*) (1.8%), and sea perch (*Helicolenus* spp.) (1.2%). Altogether, 93% of the observed catch was comprised of QMS species, representing 40 of the 96 species in the QMS prior to 1 October 2007. Over 130 species or species groups were identified by observers, the majority being non-commercial species caught in low numbers, especially black cod (*Paranotothenia magellanicus*) and Chondrichthyans, often unspecified but including shovelnose spiny dogfish (*Deania calcea*), *Etmopterus* species, and seal sharks (*Dalatias licha*). A surprising number of echinoderms, especially starfish (of which almost 200 000 were observed caught during the period), anemones, crustaceans, and other invertebrates were also recorded by observers.

Total annual bycatch estimates for 1998–99 to 2005–06 ranged from about 2200 t to 3700 t, compared with approximate target species catches in the same period of between about 3500 and 8700 t. A large part of this bycatch (40–50%) comprised a single species, spiny dogfish, and 80% of the bycatch were quota species (Figures 6.4 & 6.5). Bycatch levels decreased during the period, in line with decreasing effort in the fishery. Total bycatch estimates for the years before 1998–99 ranged from about 880 t to 3900 t. Differences in methodology between the two studies, coupled with generally low observer coverage, resulted in significantly different estimates of total bycatch for 1994–95.

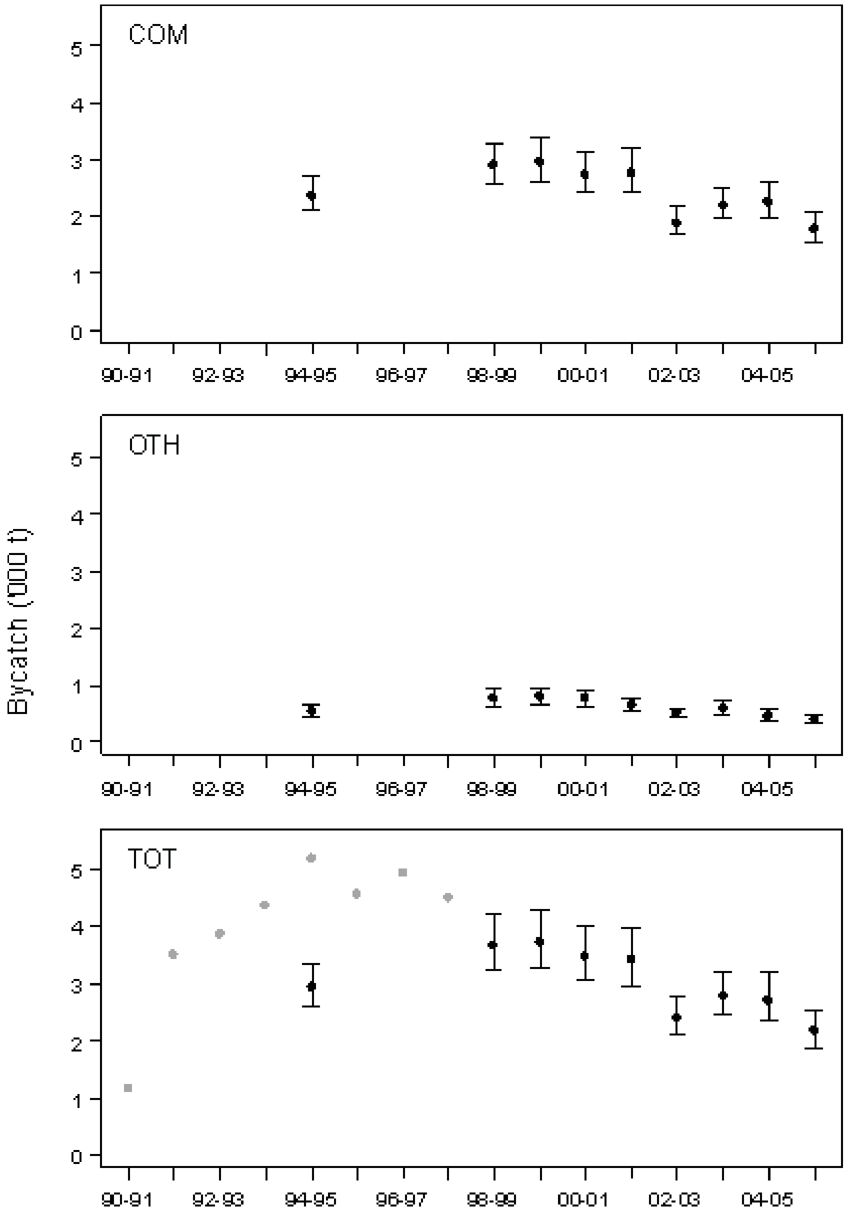

Figure 6.4: Annual estimates of fish bycatch in the target ling longline fishery, calculated for commercial (QMS) species (COM), non-commercial (non-QMS) species (OTH), and overall (TOT) for the years 1994–95 and 1998–99 to 2005–06 (in black). Also shown (in grey) are estimates of total bycatch calculated for the period 1990–91 to 1997–98 by Anderson *et al.* (2000). Error bars show the 95% confidence intervals.

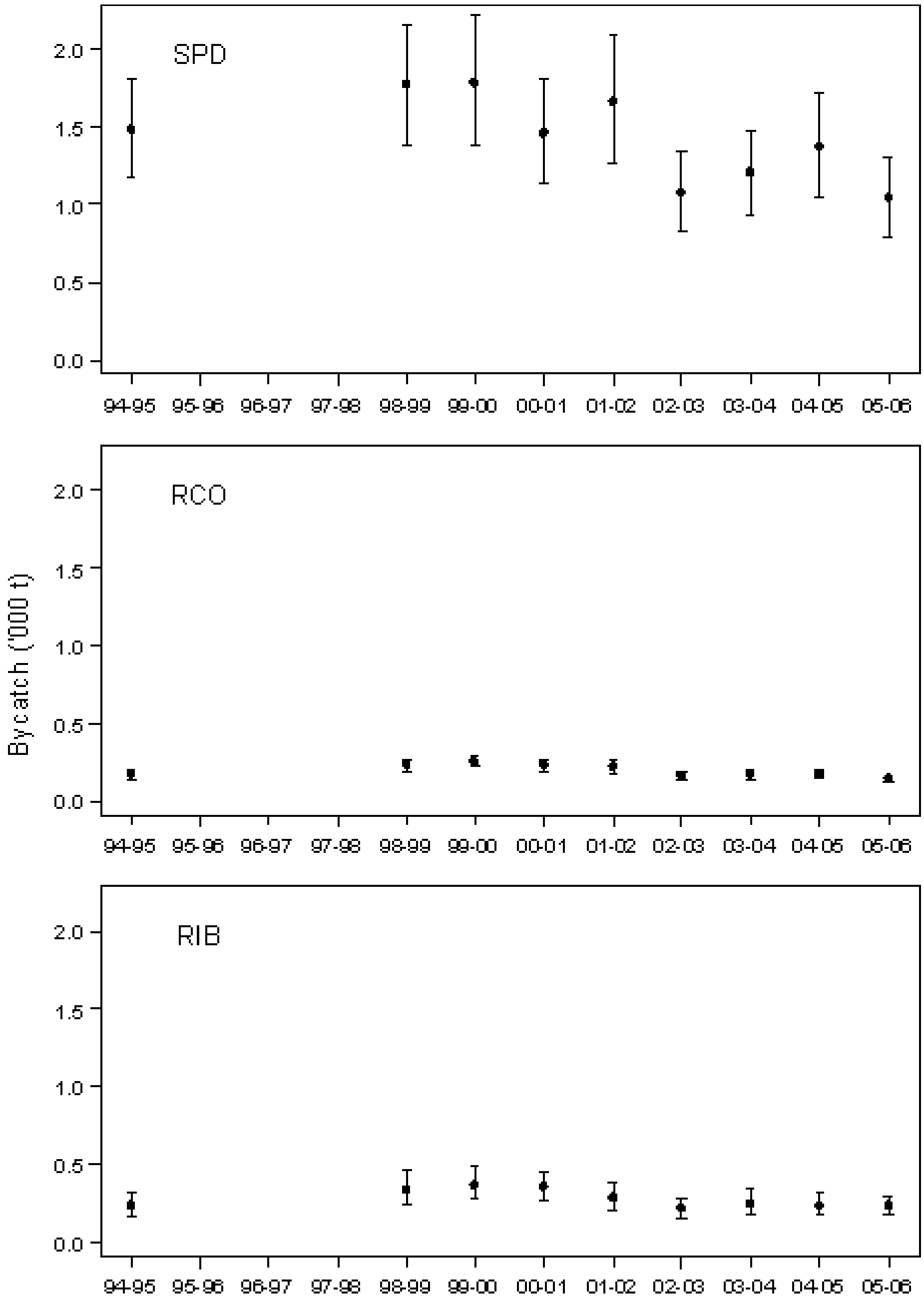

Figure 6.5: Annual estimates of the bycatch of spiny dogfish (SPD), red cod (RCO), and ribaldo (RIB) in the target ling longline fishery for the years 1994–95 and 1998–99 to 2005–06. Error bars show the 95% confidence intervals.

Total annual discard estimates for 1998–99 to 2005–06 ranged from about 1400 t to 2400 t, and generally decreased during the period (Figure 6.6). About 70–75% of these discarded fish were quota species, and 60–70% spiny dogfish, the remainder being non-quota, generally non-commercial, species. Ling were discarded in small amounts (40–90 t per year), these discards generally being attributable to fish being lost on retrieval or predated by marine mammals and birds. Estimated annual discards were generally lower for the earlier period (1990–91 to 1997–98) and between about 350 t and 1600 t. Total discard estimates for 1994–95 were similar for the two studies.

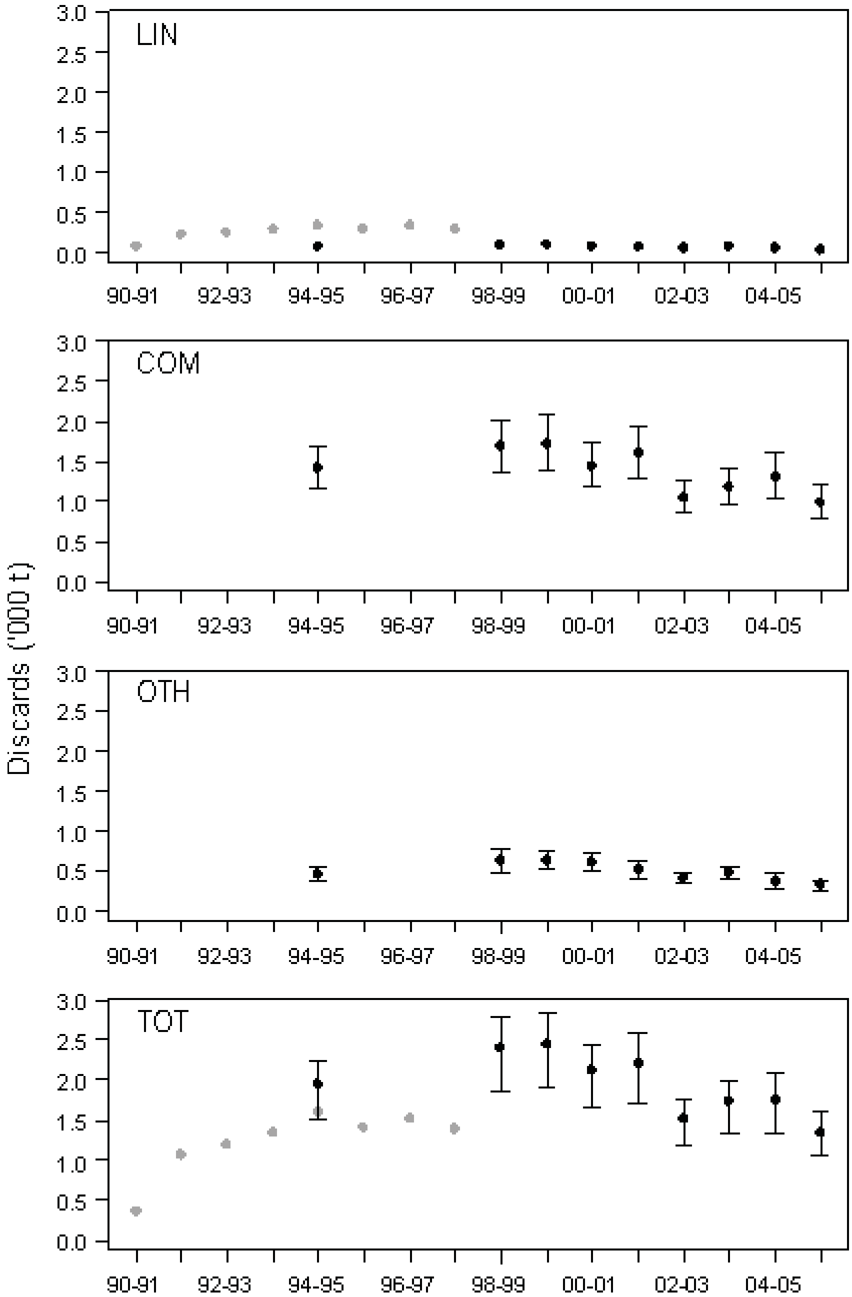

Figure 6.6: Annual estimates of fish discards in the target ling longline fishery, calculated for ling (LIN), commercial (QMS) species (COM), non-commercial (non-QMS) species (OTH), and overall (TOT) for the years 1994–95 and 1998–99 to 2005–06 (in black). Also shown (in grey) are estimates of the ling and total discards calculated for 1990–91 to 1997–98 by Anderson *et al.* (2000). Error bars show the 95% confidence intervals.

6.3.3. Hoki/hake/ling trawl fishery

Earlier reports were limited to the hoki target fishery and only the most recent report considers bycatch and discards for the fishery as defined by the three target species combined—but hoki is dominant in this fishery, accounting for over 90% of the catch.

Observer coverage in the hoki, hake, and ling trawl fishery between 2000–01 and 2006–07 ranged from 11% to 21% of the annual target fishery catch, and 78 separate vessels were observed, covering the full range of vessel sizes. The annual number of observed tows decreased from 3580 in 2000–01 to 1999 in 2006–07. Coverage has been spread over the geographical range of this fishery, with high sampling throughout the west coast South Island (WCSI) and Chatham Rise fishing grounds and, less frequently, in the Sub-Antarctic. Lower levels of sampling have been achieved in the Cook Strait and Puysegur fisheries, and coverage was lower still around the North Island although this area accounts for very little of the overall catch. Good observer coverage was achieved during the hoki spawning season (July to early September), but coverage outside of this period was variable and under-representative in some months in some years, especially in the Sub-Antarctic, Chatham Rise and Puysegur fisheries.

Hoki, hake, and ling accounted for 87% (77%, 6%, and 4% respectively) of the total observed catch from trawls targeting hoki, hake, and ling between 2000–01 and 2006–07. The remaining 13% comprised a large range of species, especially javelinfish (2.1%), silver warehou (1.7%), rattails (1.4%), and spiny dogfish (1.1%). In total, over 470 species or species groups have been identified by observers, the majority of which are non-commercial species caught in low numbers. Chondrichthyans in general, often unspecified but including spiny dogfish and basking shark, have accounted for much of the non-commercial catch. Echinoderms, squids, crustaceans, and other unidentified invertebrates were also well represented in the bycatch of this fishery.

Total bycatch in the hoki, hake, and ling fishery between 2000–01 and 2006–07 ranged from about 36 000 to 58 000 t per year (compared to the combined total landed catch of hoki, hake, and ling of 130 000 to 238 000 t). Estimates of total bycatch for 1990–91 to 1998–99 from earlier projects (for the hoki target fishery alone), ranged from about 15 000 t to 60 000 t (Figure 6.7). Overall, total bycatch increased during the 1990s to a peak in the early 2000s, and has since declined slowly. Annual bycatch for the 1990–01 to 2006–07 period was also estimated for commercial species (QMS species and species which were generally retained (>75%) and comprised 0.1% or more of the total observed catch) and non-commercial species, rather than QMS and non-QMS species. Roughly similar amounts of these two categories were caught overall, and each showed a similar pattern over time to total bycatch.

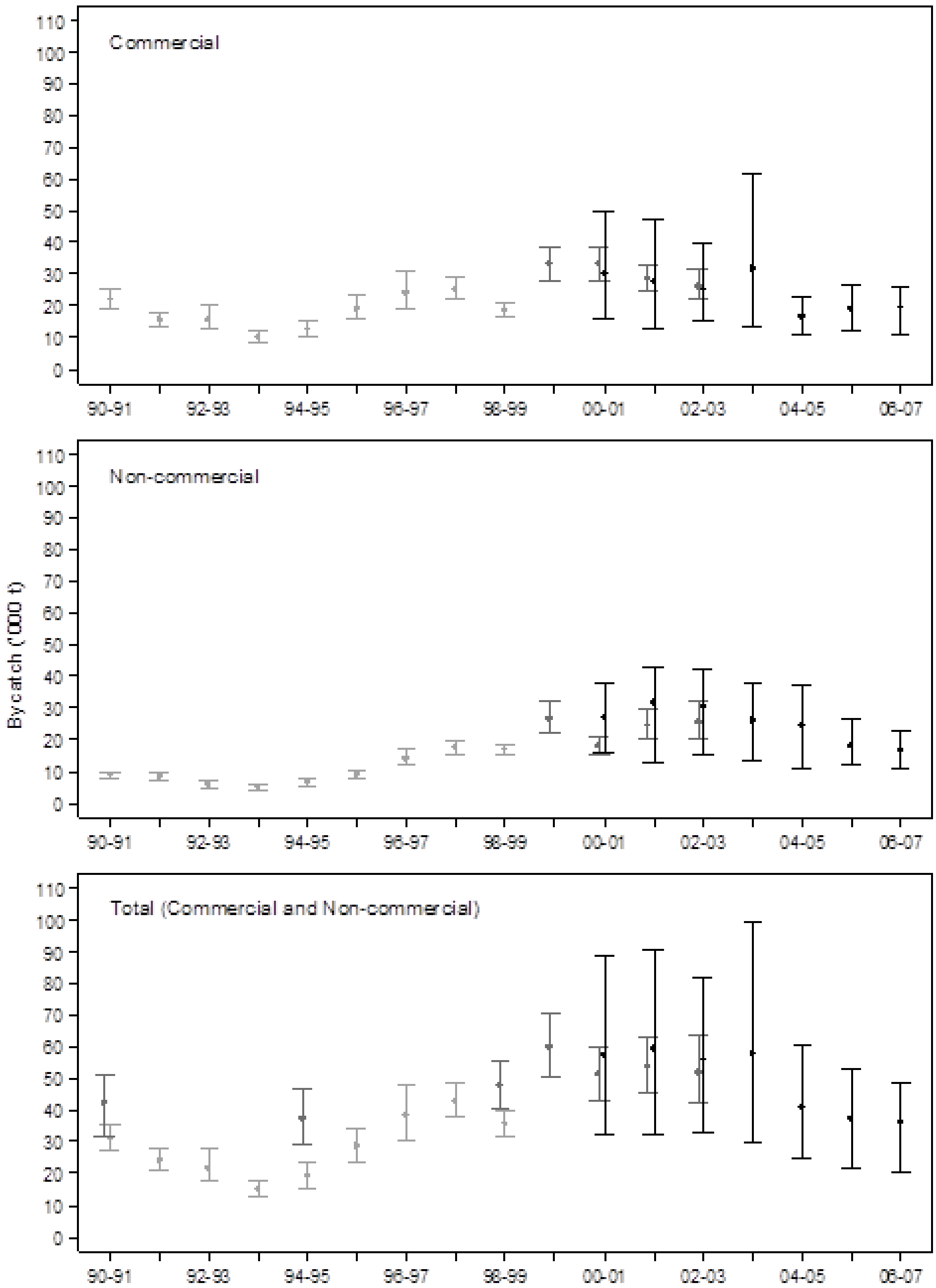

Figure 6.7: Annual estimates of fish bycatch in the target hoki, hake and ling trawl fishery, calculated for commercial species, non-commercial species, and overall for 2000–01 to 2006–07 (black). Also shown (in light grey) are the equivalent bycatch estimates calculated for 1990–91 to 1998–99 by Anderson *et al.* (2001), and for the years 1990–91, 1994–95, 1998–99 and 1999–2000 to 2002–03 by Anderson and Smith (2004), (in dark grey). Error bars show the 95% confidence intervals.

Total annual discard estimates for 2000–01 to 2006–07 ranged from about 5500 to 29 000 t per year with the main species being discarded including spiny dogfish, rattails, javelinfish, hoki, and shovelnose dogfish. Total annual discards for 1990–91 to 1998–99 were between 6600 t and 17 900 t, and overall there has been no obvious trend in total discards (Figure 6.8). The target species (hoki, hake, and ling) made up 9.7% of total observed discards. Discard rates were strongly influenced by the use of meal plants on fishing vessels; discards of non-commercial species on factory vessels without meal plants was up to twice the level of discards for vessels with meal plants. The use of meal plants, especially for species such as javelinfish and other rattails, has become more prevalent in recent years.

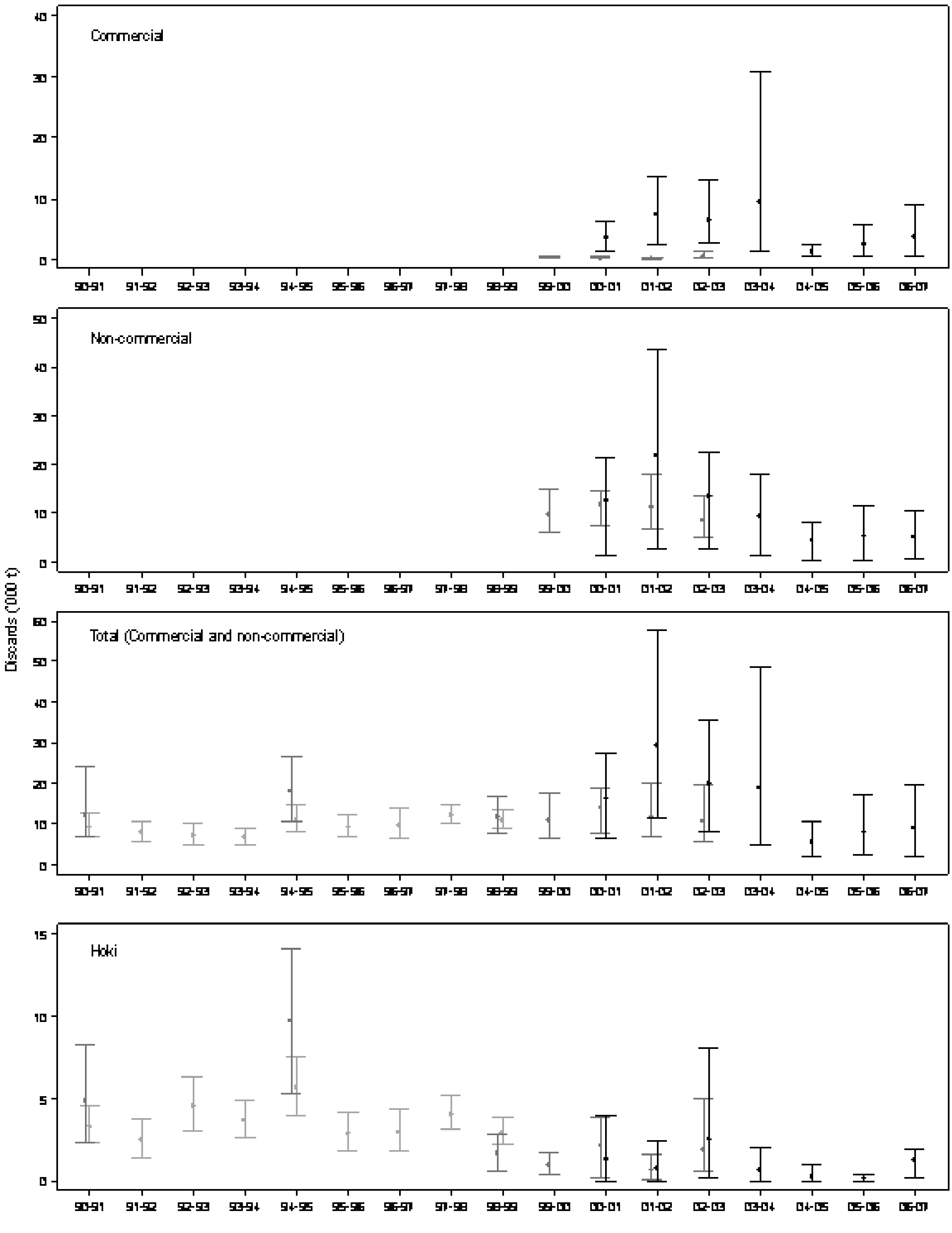

Figure 6.8: Annual estimates of fish discards in the target hoki, hake, and ling trawl fishery, calculated for commercial species, non-commercial species, hoki, and overall for the period 2000–01 to 2006–07 (black). Also shown (in light grey) are the equivalent discard estimates calculated for the period 1990–91 to 1998–99 by Anderson *et al.* (2001), and for 1990–91, 1994–95, 1998–99 and 1999–2000 to 2002–03 by Anderson and Smith (2004), (in dark grey). Error bars show the 95% confidence intervals.

6.3.4. Jack mackerel trawl fishery

Estimates of annual bycatch in this fishery are available for 1990–91 to 2004–05, with this fishery due to for reassessment in 2013. The annual level of observer coverage in this fishery has varied between 8% and 27% of the target fishery catch but was usually between 15% and 20%. For the most recent period examined, 2001–02 to 2004–05, the majority of the observer effort has focussed on the main fishery, off the west coasts of the North and South Islands, with some additional coverage on the Stewart/Snares Shelf and Chatham Rise fisheries. However, in 2003–04 and 2004–05, there was a total of only 12 trawls observed outside of the western fishery. During this time the fishery was dominated by seven large trawlers and observers were able to complete a trip on each vessel in most years. The fishery runs year round, and although there were significant periods in each year when commercial fishing effort was not observed, coverage encompassed all seasons for the four years combined.

Jack mackerel species accounted for 70% of the total estimated catch from all trawls targeting jack mackerel between 2001–02 and 2004–05. The remaining 30% mostly comprised other commercial species; especially barracouta (15.6%), blue mackerel (4.8%), frostfish (3.1%), and redbait (2.7%). Overall about 130 species or species groups were identified by observers, and about half of these were non-commercial, non-QMS species caught in low numbers. The species most discarded was the spiny dogfish, which comprised about 0.5% of the total catch. The bycatch of non-QMS invertebrate species has yet to be closely studied in this fishery, but species of squid, salps, jellyfish were the most commonly recorded by observers during this period.

Total bycatch in the jack mackerel trawl fishery between 2001–02 and 2004–05 ranged from about 7700 t to 11 900 t. Estimates of total bycatch for 1990–91 to 2003–04 from earlier projects ranged from about 5400 t to 15 500 t (Figure 6.9). After an abrupt increase in the late 1990s, annual bycatch steadily decreased to a level comparable to that of the 1990–91 to 1996–97 period. This bycatch almost entirely comprised commercial (mainly QMS) species.

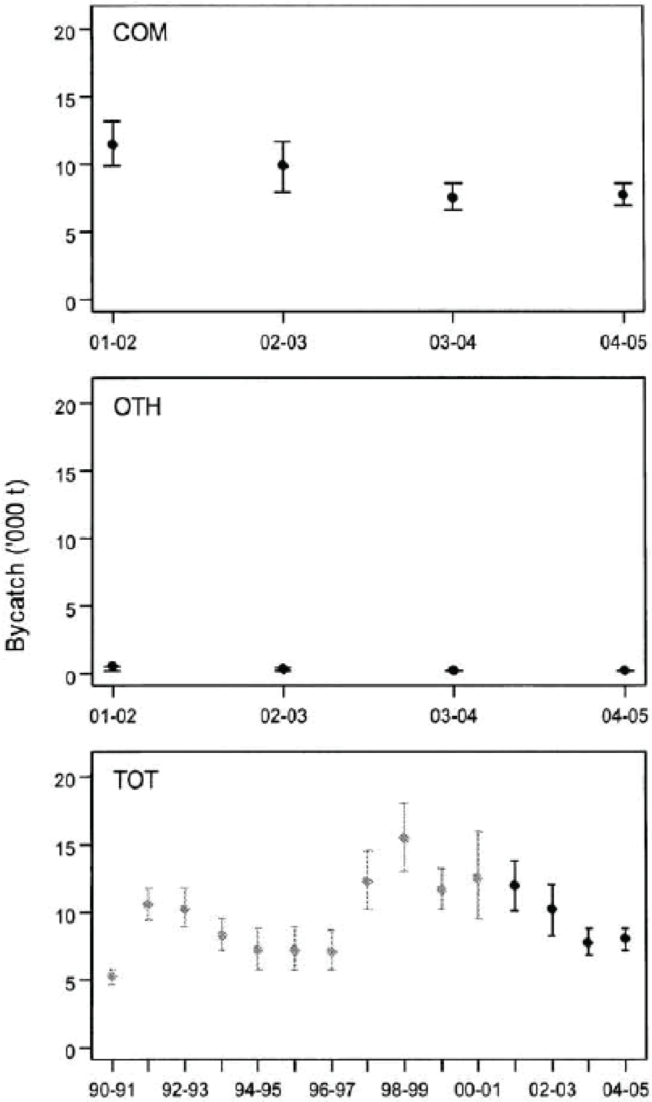

Figure 6.9: Annual estimates of fish bycatch in the target jack mackerel trawl fishery for the 2001-02 to 2004-05 fishing years (in black), calculated for commercial species (COM), non-commercial species (OTH), and overall (TOT). Also shown (in grey) are estimates of overall bycatch calculated for 1990–91 to 2000–01 by Anderson *et al.* (2000) and Anderson (2004a). Error bars show the 95% confidence intervals.

Total annual discards decreased between 2001–02 and 2004–05, continuing a trend that began in 1998–99, to a level of only 90–100 t per year. This is about 5% of the level of 1997–98 (1850 t), when annual discards were at their greatest, and is lower than in any year since 1990–91 (Figure 4.10). Discards of the target species were about 200–400 t per year prior to 1998–99 but thereafter decreased to only about 10 t per year, mainly due to the absence of recorded losses of large quantities of fish through rips in the net or intentional releases of fish during landing. Discards comprised a roughly equal amount of commercial and non-commercial species in the recent study, although commercial species discards were substantially greater in 2001–02.

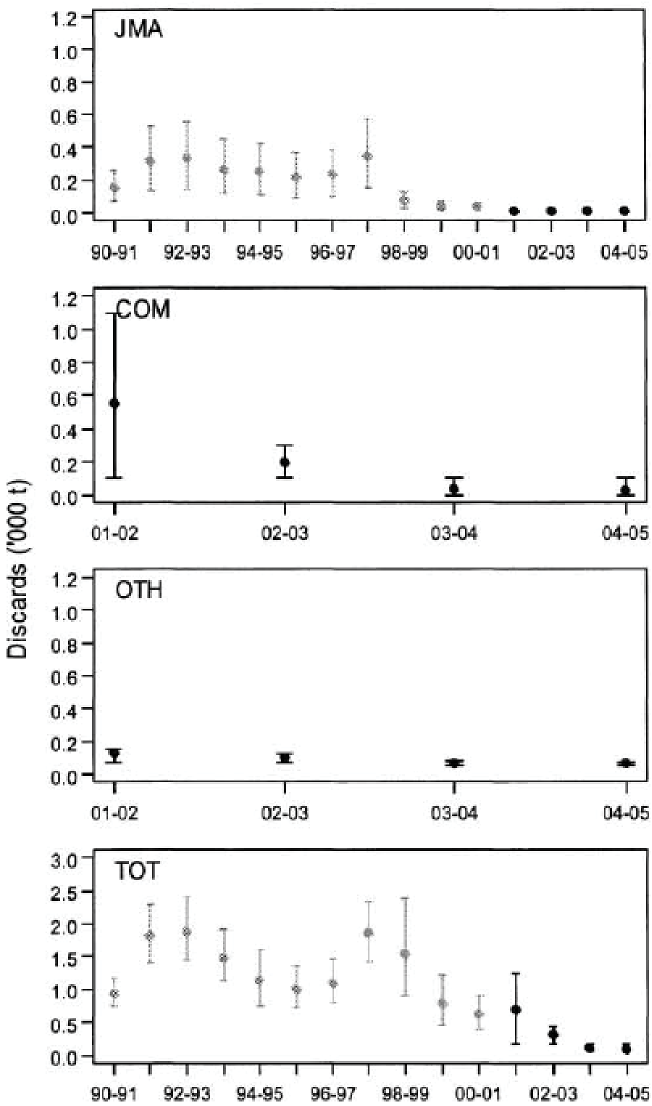

Figure 6.10: Annual estimates of fish discards in the target jack mackerel trawl fishery for the 2001-02 to 2004–05 fishing years (in black), calculated for jack mackerel (JMA), commercial species (COM), non-commercial species (OTH), and overall (TOT). Also shown (in grey) are estimates of jack mackerel and overall discards calculated for 1990–91 to 2000–01 by Anderson *et al.* (2000) and Anderson (2004a). Error bars show the 95% confidence intervals.

6.3.5. Southern blue whiting trawl fishery

In the most recent study, covering the period 2002–03 to 2006–07, the ratio estimator used to calculate bycatch and discard rates in this fishery was based on trawl duration. Linear mixed-effect models (LMEs) identified fishing depth as the key variable influencing bycatch rates and discard rates in this fishery, and regression tree methods were used to optimise the number of levels of this variable in order to stratify the calculation of annual bycatch and discard totals in each catch category.

The key categories of catch/discards examined were; southern blue whiting, other QMS species combined, commercial species combined (as defined above for hoki/hake/ling), non-commercial species combined, and three commonly caught individual species, hake, hoki, and ling.

The level of observer coverage represented between about 22% and 53% of the target fishery catch between 2002–03 and 2006–07 and similar levels were reported from earlier reports, for 1990–91 to 2001–02. The spread of observer data, across a range of variables, has shown no significant shortcomings, due to a combination of the highly restricted distribution of the southern blue whiting fishery over space and time of year, a stable and uniform fleet composition, and a high level of observer effort.

Southern blue whiting accounted for more than 99% of the total estimated catch from all observed trawls targeting southern blue whiting between 2002–03 and 2006–07. About half the remaining total catch was made up of ling (0.2%), hake (0.1%), and hoki (0.1%). These three species, along with other QMS species, comprised over 80% of the total bycatch. In all, over 120 species or species groups were identified by observers, most being non-commercial species caught in low numbers. Porbeagle sharks (introduced into the QMS in 2004), javelinfish and other rattails, and silverside, accounted for much of remaining bycatch. Invertebrate species (mainly sponges, crabs, and echinoderms) were also recorded by observers, but no taxon accounted for more than 0.01% of the total observed catch.

Total annual bycatch estimates for 2002–03 and 2006–07 ranged from about 40 t to 390 t, compared with approximate target species catches in the same period of about 22 000 to 42 000 t. This bycatch was fairly evenly split between commercial species (55%) and non-commercial species (45%), although QMS species accounted for about 80% of the total bycatch during this period. Total annual bycatch decreased during the period, to an all-time low of 40 t in 2006–07. Total annual bycatch estimates for 1990–91 to 2001–02, from earlier reports, were mostly between about 60 t and 500 t but reached nearly 1500 t in 1991–92 (Figure 6.11). This year immediately preceded the introduction of southern blue whiting into the QMS, and effort and catch were exceptionally high.

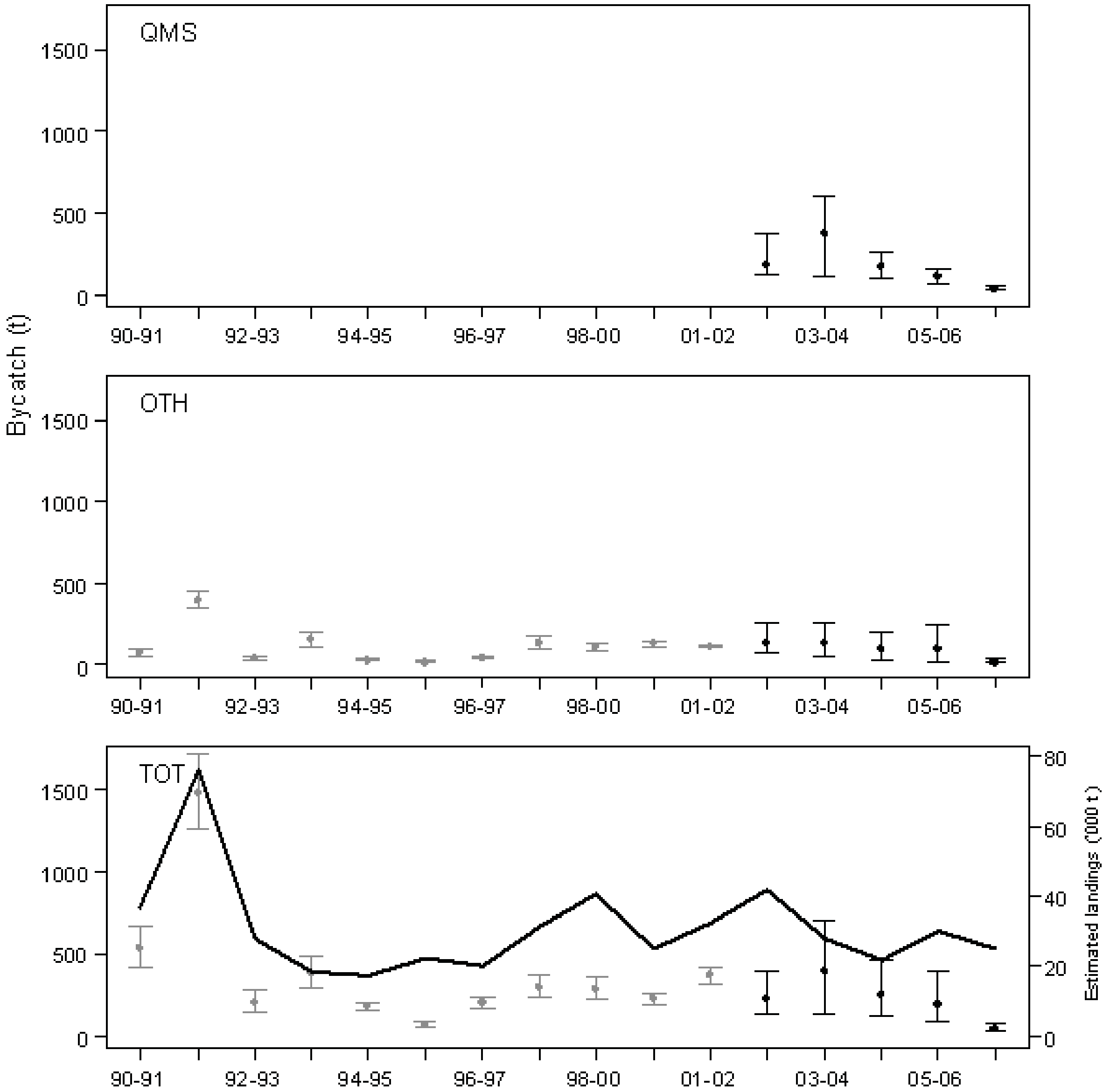

Figure 6.11: Annual estimates of fish bycatch in the southern blue whiting trawl fishery, calculated for QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black). Also shown (in grey) are estimates of bycatch in each category (excluding QMS) for 1990–91 to 2001–02 (Anderson 2004a). Error bars show the 95% confidence intervals. Note: the 98–00 fishing year encompasses the 18 months between September 1998 and March 2000, the transitional period between a change from an Oct–Sep to Apr–Mar fishing year. The dark line in the bottom panel shows the total annual estimated landings of SBW (Ministry of Fisheries 2009).

Total annual discard estimates between 2002–03 and 2006–07 ranged from about 90 t to 250 t per year. Discard amounts sometimes exceeded bycatch due to the large contribution of the target species (50–230 t per year) to total discards – the result usually of fish losses during recovery of the trawl. Discarding of commercial species was virtually non-existent in most years and discards of non-commercial species amounted to only 10–50 t per year. The main species discarded were southern blue whiting, rattails and porbeagle sharks. Total annual discard estimates for 1990–91 to 2001–02, from earlier reports, were mostly between about 140 t and 750 t but were about 1200 t in 1991–92 (Figure 6.12). Discards of southern blue whiting (and therefore total discards) decreased substantially at the end of the 1990s and have remained at low levels, below 250 t per year, at least up until 2006–0

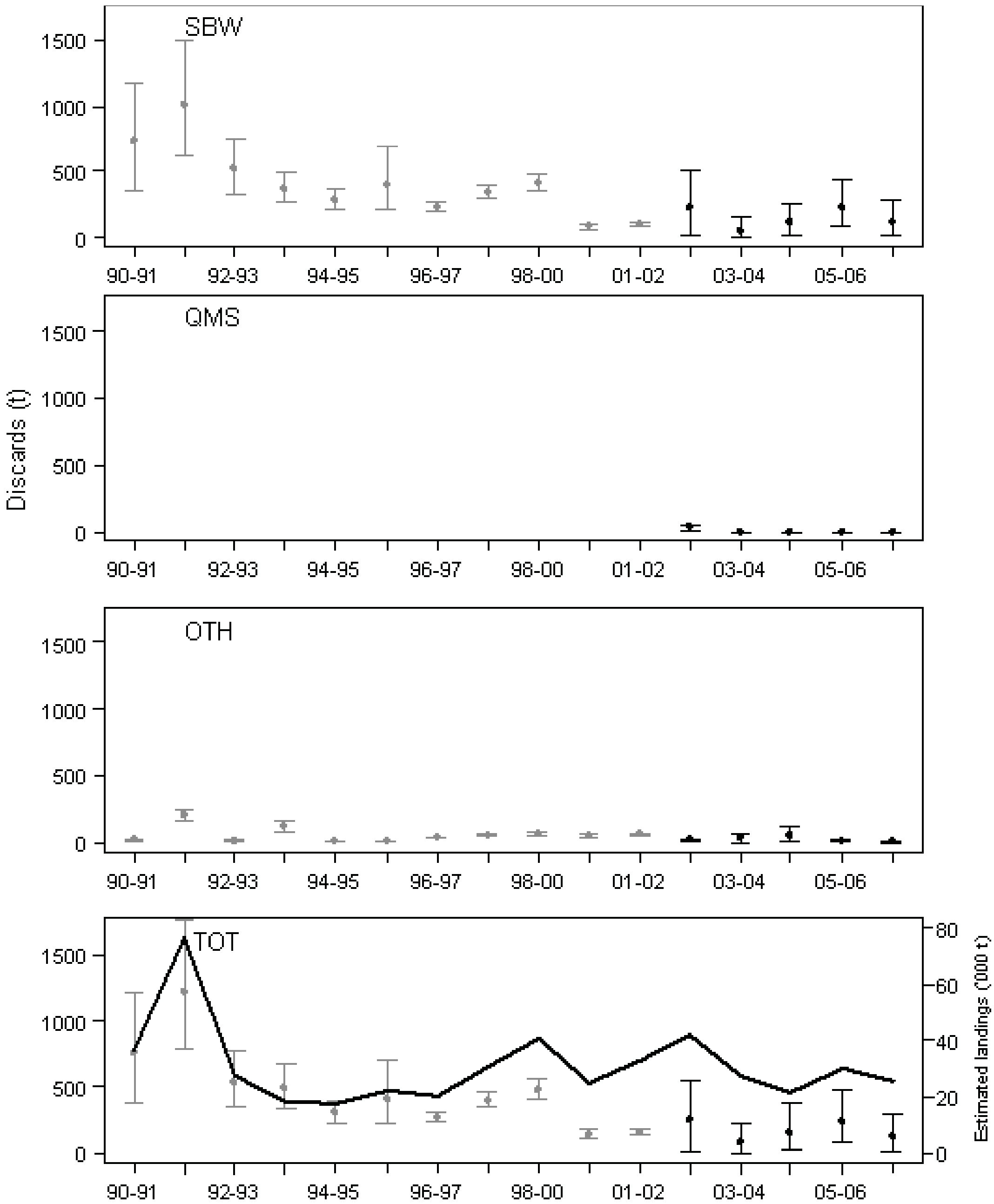

Figure 6.12: Annual estimates of fish discards in the southern blue whiting trawl fishery, calculated for the target species (SBW), QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black). Also shown (in grey) are estimates of discards in each category (excluding QMS) calculated for 1990–91 to 2001–02 by Anderson (2004a). Error bars show the 95% confidence intervals. The dark line shows the total annual estimated landings of SBW (Ministry of Fisheries 2009).

6.3.6. Orange roughy trawl fishery

In the most recent study, covering the period 1990–91 to 2008–09, the ratio estimator used to calculate bycatch and discard rates in the orange roughy fishery was based on the number of trawls. Linear mixed-effect models (LMEs) identified trawl duration as the key variable influencing bycatch rates and discard rates in this fishery, and regression tree methods were used to optimise the number of levels of this variable in order to stratify the calculation of annual bycatch and discard totals in each catch category.

The key categories of catch/discards examined were; orange roughy, other QMS species (excluding oreos) combined, commercial species combined (as defined above for hoki/hake/ling), and non-commercial species combined.

The level of observer coverage in this fishery has been relatively high over the entire period of the fishery—more than 10% (in terms of the total fishery catch) in all but one year, and over 50% in some years. Observer coverage was not evenly spread across all parameters of the orange roughy fishery, the most widespread of any New Zealand fishery, with notable undersampling of smaller vessels, the east coast fisheries in QMAs ORH 2A, ORH 2B, and ORH 3A, and some of the earlier years of the period.

For the recent orange roughy fishery (since 2005–06), orange roughy accounted for about 84% of the total observed catch. Much of the remainder of the total catch (about 10%) comprised oreo species: mainly smooth oreo (8%), and black oreo (2.1%). Rattails (various species, 0.8%) and shovelnose spiny dogfish (*Deania calcea*, 0.6%) were the species most adversely affected by this fishery, with over 90% discarded. Other fish species frequently caught and usually discarded included deepwater dogfishes (family Squalidae), especially *Etmopterus* species, the most common of which is likely to have been Baxter's dogfish (*E. baxteri*), slickheads, and morid cods, especially Johnson's cod (*Halargyreus johnsonii*) and ribaldo. In total, over 250 bycatch species or species groups were observed, most being non-commercial species, including invertebrate species, caught in low numbers. Squid (mostly warty squid, *Onykia* spp.) were the largest component of invertebrate catch, followed by various groups of coral, echinoderms (mainly starfish), and crustaceans (mainly king crabs, family Lithodidae).

Total annual bycatch in the orange roughy fishery since 1990–91 ranged from about 2300 t to 27 000 t, and declined over time alongside the decline in catch and effort in this fishery to be less than 4000 t in each of the last four years estimated (Figure 6.13). Bycatch mostly comprised commercial species, with non-commercial species accounting for only 5–10% of the total bycatch in the recent period.

Estimated total annual discards also decreased over time, from about 3400 t in 1990–91 to about 300 t in 2007–08 (Figure 6.14), and since about 2000 were almost entirely non-commercial, non-QMS species. Large discards of orange roughy and other commercial species, more prevalent early in the fishery, were often due to fish lost from torn nets during hauling.

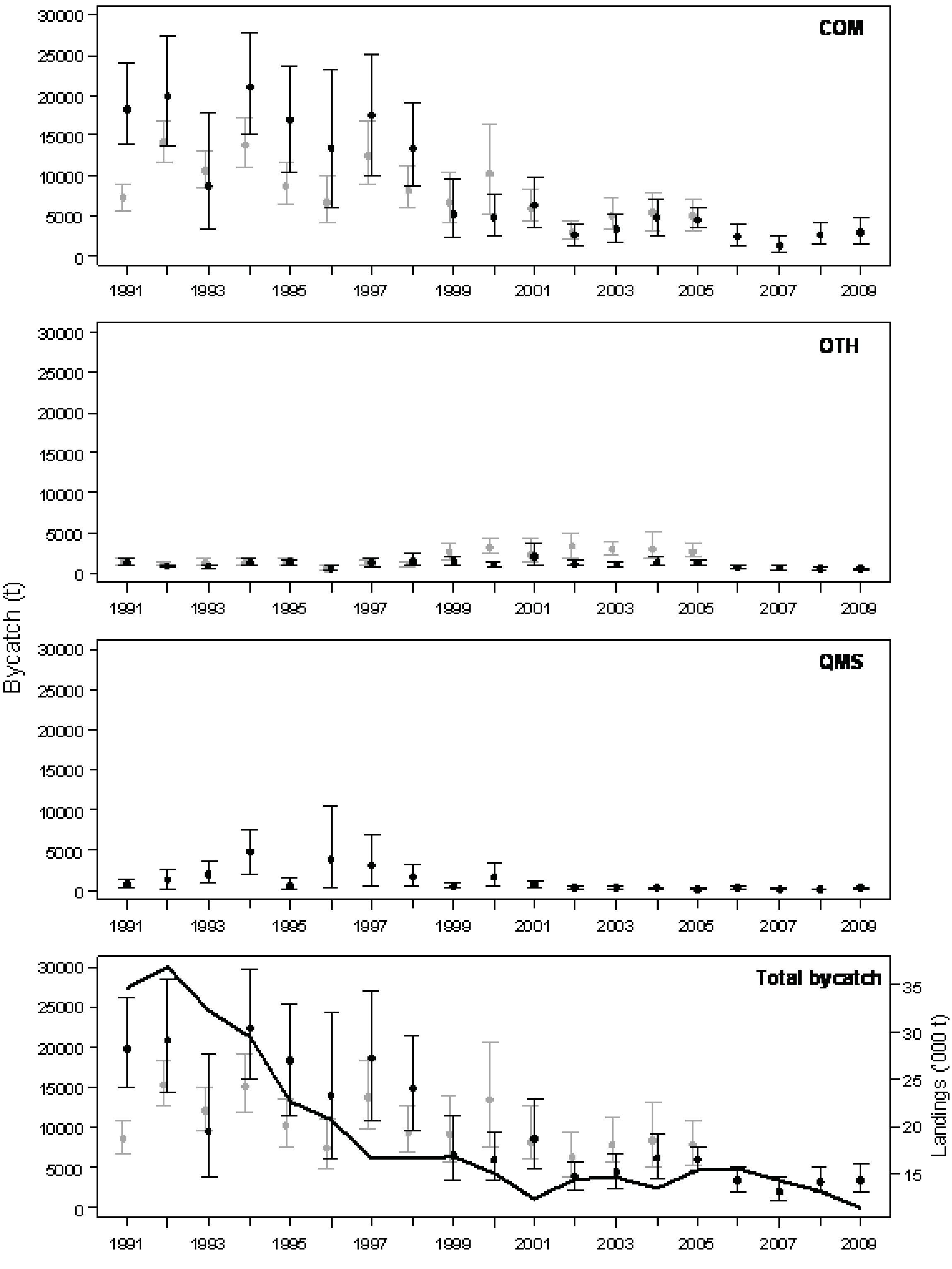

Figure 6.13: Annual estimates of fish bycatch in the orange roughy trawl fishery, calculated for commercial species (COM), non-commercial species (OTH), QMS species, and overall for 1990–91 to 2008–09 (black points). Also shown (grey points) are earlier estimates of bycatch in each category (excluding QMS) calculated for 1990–91 to 2004–05 (Anderson *et al.* 2001, Anderson 2009a). Error bars show the 95% confidence intervals. The black line in the bottom panel shows the total annual estimated landings of orange roughy (O. Anderson & M. Dunn (NIWA), unpublished data).

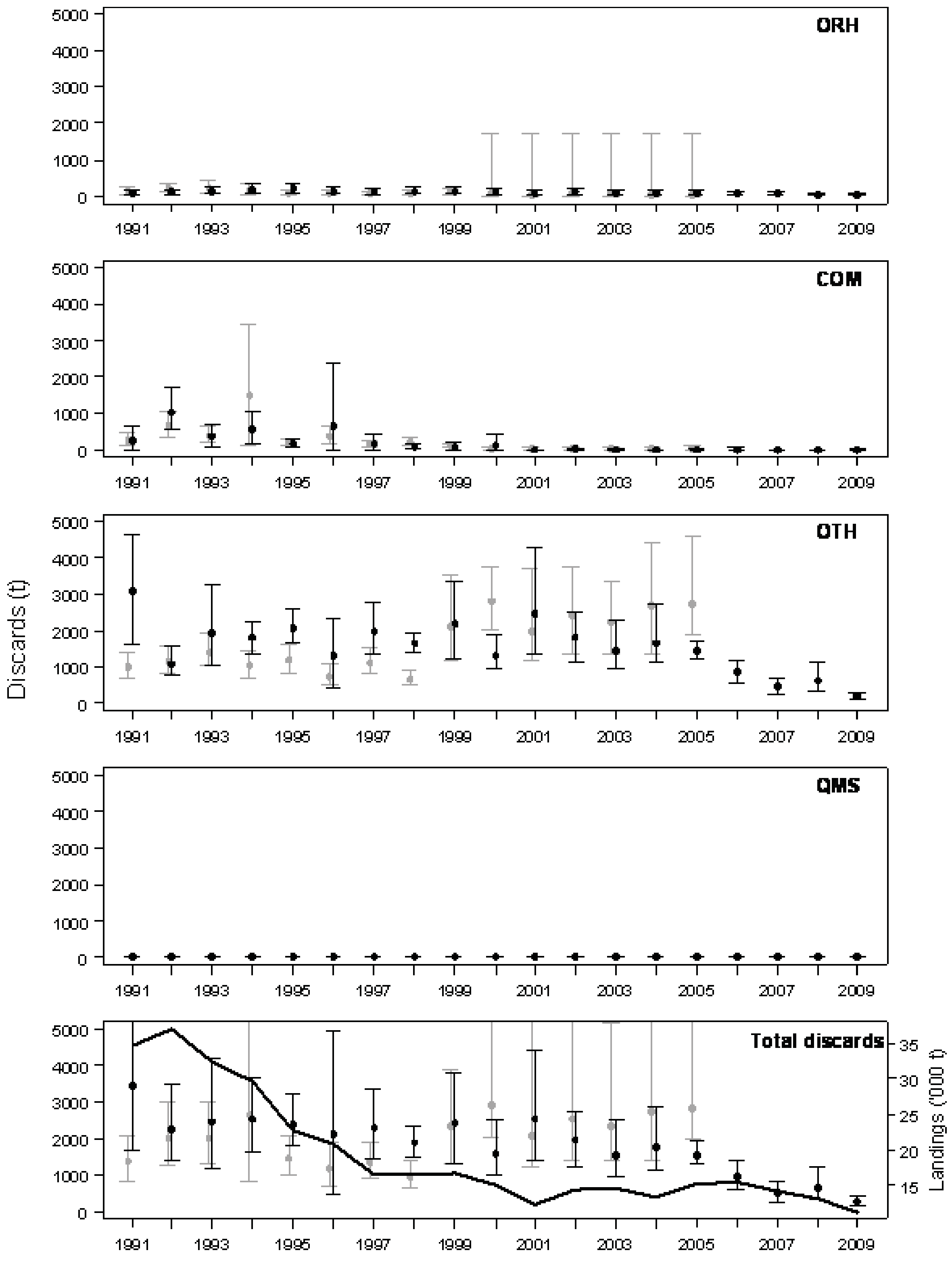

Figure 6.14: Annual estimates of fish discards in the orange roughy trawl fishery, calculated for the target species (ORH), commercial species (COM), non-commercial species (OTH), QMS species, and overall for 1990–91 to 2008–09 (black points). Also shown (grey points) are estimates of discards in each category (excluding QMS) calculated for 1990–91 to 2004–05 (Anderson *et al.* 2001, Anderson 2009a). Error bars show the 95% confidence intervals. The black line in the bottom panel shows the total annual estimated landings of orange roughy (O. Anderson & M. Dunn (NIWA), unpublished data).