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Review of New Zealand's scallop fishery stock assessment data and methods

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Preface

The Ministry for Primary Industries and its predecessor, the Ministry of Fisheries, have conducted fully-independent expert reviews of stock assessments, research methodologies and research programmes since 1998. We also run specialist technical review workshops to further advance fisheries and other marine science methodologies and techniques. These fully-independent reviews and technical workshops are separate from, but complementary to, the annual Science Working Group processes that are used to ensure the objectivity and reliability of most of our scientific research and analyses.

A new publication series, Fisheries Science Reviews, was initiated in 2015 to ensure that reports from these reviews are readily accessible. The series will include all recent and new fully-independent reviews and technical workshop reports, and will also incorporate as many historical reports as possible, as time allows. In order to avoid confusion about when the reviews were actually conducted, all titles will include the year of the review. They may also include appendices containing the Terms of Reference, a list of participants, and a bibliography of supporting documents, where these have not previously been incorporated. Other than this, there will be no changes made to the original reports composed by the independent experts or workshop participants.

Fisheries Science Reviews (FSRs) contain a wealth of information that demonstrate the utility of the processes the Ministry uses to continually improve the scientific basis for managing New Zealand's fisheries.

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Review of New Zealand's scallop fishery stock assessment data and methods

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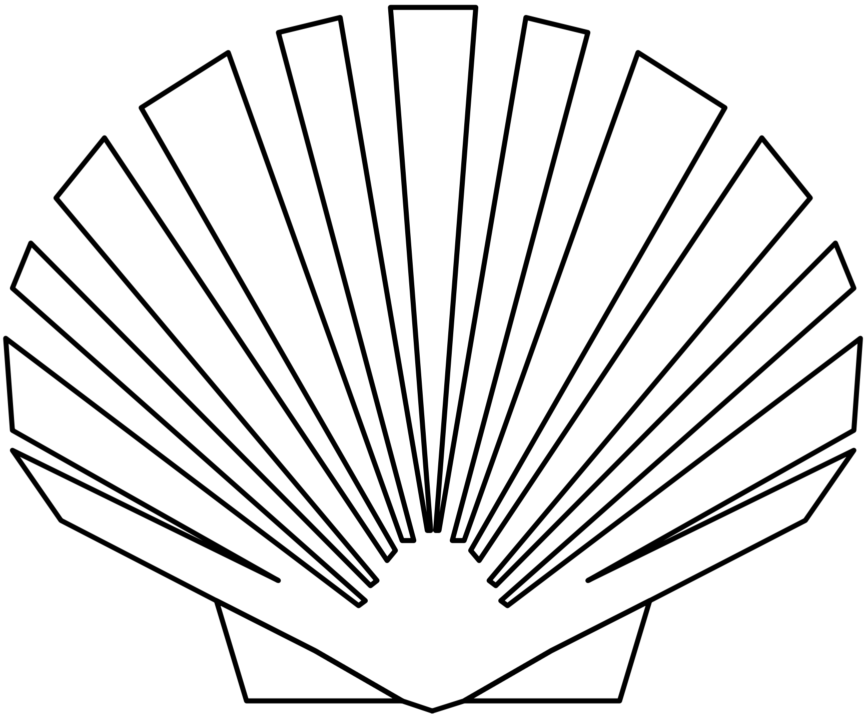

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1. Introduction

Presently the three main New Zealand scallop (*Pecten novaezelandiae*) fisheries have very different histories with separate assessment and management programmes. In addition, the documentation of methods and findings in each of the fisheries was found across numerous inter-connected documents, sometimes with important details difficult to find. While the differences between fisheries make this review more difficult, there remain, however, many commonalities. So it is recommended that 1) a synthesis of scallop documentation detailing what was done where would help ensure that standard methods are used. In addition, 2) improved documentation across all methods used would make all assessments and analyses more defensible.

Landings from the three main fisheries have always been variable (**Figure 1**) with higher catches in SCA 7 than in SCA CS and SCA 1, although catches in SCA 1 and SCA CS have been of similar scales. Importantly all three fisheries, especially SCA 1 and SCA 7, are currently at a relatively low level.

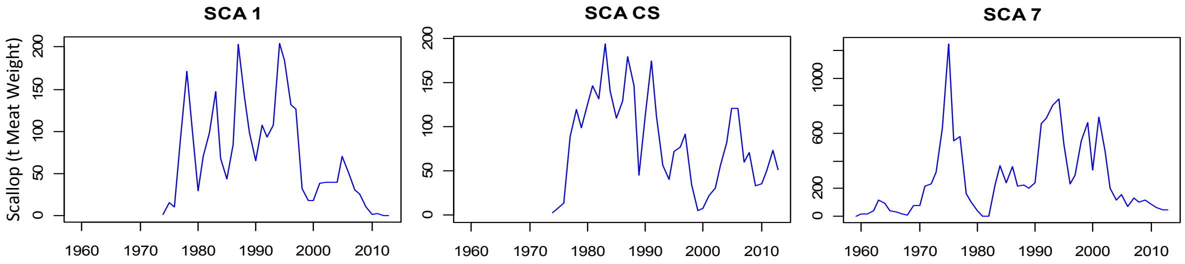

Figure 1. Reported landings of scallop meat weight from the three main scallop fisheries in New Zealand: SCA 1 (Northland), SCA CS (Coromandel), or SCA 7 (Challenger). Diagrams modified from Williams (2016).

The recent relatively poor performance of the three fisheries was part of the inspiration for this current review. There are currently no formally accepted target or limit reference points for any of the SCA 1 (Northland), SCA CS (Coromandel), or SCA 7 (Challenger) scallop fisheries. They all have had surveys that attempt to estimate current exploitable biomass and gain other information (scallop density, size-composition, and spatial distribution) although these have not occurred in SCA 1 since 2007, and, because the survey boundaries have been altered over time, the annual biomass estimates are not strictly comparable.

The highly stochastic recruitment dynamics of scallops leads naturally to highly variable population size even in the absence of fishing. This means that developing target and limit reference points that facilitate the determination of a stock status for scallops is not simple.

Currently it is possible to estimate suitable catch limits but without explicit or recognised target and limit reference points it is not possible to produce a defensible stock status.

The objectives of this review were thus aimed at recent scientific issues that have arisen in the three scallop fisheries.

Review objectives

Review the historical data collected from SCA 7, SCA CS and SCA 1 (including survey data, biological data, dredge efficiency data and catch data) and provide recommendations on:

a. How effectively the data are currently being used to assess the status of the stocks in SCA 7, SCA CS and SCA 1, additional data that could be collected and improvements to data collection methods

b. The most appropriate target and limit reference points to use in the SCA 7, SCA CS and SCA 1 fisheries in order to achieve sustainable utilisation on these fisheries.

c. The most appropriate method to use to assess the status of the stocks in SCA 7, SCA CS and SCA 1. For example: developing a delay difference stock assessment model, or a length based stock assessment model.

d. The CPUE Limit Rule model as a reliable monitoring tool for sustainable commercial fishing in New Zealand's scallop fisheries.

2. Effectiveness of Data for Stock Status Assessment

Weight-at-length

Currently, LW (length to green-weight) data for scallops from the Coromandel stock (parameter estimates, a = 0.00037 and b = 2.69) are used to predict biomass in SCA 7. Some routine collection of length-weight data is recommended in each area to obtain relationships from areas other than Coromandel and to monitor long-term or inter-annual changes.

Currently, the LW relationship is obtained by back-transforming a log-log regression. This method is biased, although the bias may be small. Additionally, the precision of this relationship may be overestimated because multiple observations from the same tow are considered independent, even though they tend to be correlated. Mixed-effects generalised linear models (GLMM) can avoid both these issues (Hennen and Hart 2012). For bootstrapping purposes, because of within-tow similarities, individuals within a tow should either be considered to be a single observation, or better, treat within-tow observations as random effects.

Meat weight

Data for recovery of meat-weight from green-weight exists for some years, areas and times (e.g., for SCA 7, fishing seasons 1996–97 to 2008–09). "Mean" (12.5%) recovery rates are often used to convert green-weight to meat-weight, although there was no documentation of exactly where this figure came from. Because the recovery fraction varies seasonally, its "mean" value can be affected by the timing of the fishery. However, this effect is probably fairly small because the fisheries are typically conducted at about the same time each year.

Similar to the length to green-weight relationship, it would be useful to monitor the meat-weight recovery rate periodically in each region. It would be preferable to collect length to meat-weight data and then meat-weight could be estimated directly from length, but the panel was told that this is impractical.

Growth

Tuck & Williams (2009) modelled shell growth using tag-return data, primarily using the inverse logistic model of Haddon et al. (2008). The inverse logistic appears to be a viable approach, but it is a purely empirical description of the growth process. As the variability of growth increment at length is modelled separately, this provides all the information required to generate the needed growth transition matrices. Residuals could be plotted as a function of time at large to explore tagging artefacts as well as seasonal effects; a residual pattern for short times at large could indicate that the tagging process affects growth for a period immediately after tagging.

An alternative is to use a von Bertalanffy model with random effects on one of the parameters, most likely on L_∞. One advantage of this approach is that it also gives an estimate of the variability of that parameter among the populations as well as estimates of the population parameter means, thus allowing for growth transition matrices to be constructed.

Growth appears to have decreased in most recent years, but in some cases growth data from the 1990s are still being used instead of more recent data. If growth has changed, the most recent growth data needs to be used for projections and yield per recruit (YPR) analysis.

The panel had some concerns about anomalous data, including negative increments as well as some apparently very fast growth. Negative increments could be due to measurement error, clerical error, or because of chipping of the shell during recovery. If chipping is occurring, it could be biasing the growth increments small, so it is important to ascertain why there are negative increments. A simple measurement – re-measurement experiment could be used to estimate the extent of measurement error.

When annualizing the increments, it is inappropriate to extrapolate the negative increments, since the processes that create negative increments are independent of the time at large. Potentially, anomalous data could be removed from the analysis, but this should be done with caution; at very least an analysis with and without such apparently outlying data should be conducted to improve confidence in the parameter estimates eventually used.

The panel had some concern regarding bootstrapping the growth data to account for uncertainty. Given the limited amount of recent data and the anomalous increments, some bootstraps could give unrealistic estimates or not converge at all.

Other methods of estimating growth could be explored. This includes tracking of year classes from modes in successive surveys and stable isotope analysis. Year class tracking can be especially helpful for smaller lengths that are under-represented in the tagging set, and may help inform whether tagging affects growth. Tracking of large year classes, potentially in closed areas, can be especially useful since there would be minimal confounding from other year classes. Stable isotope analysis is expensive, but analysis of even a few shells may give insights regarding seasonality of growth and whether any feature on the shells could potentially be used to estimate growth.

Natural mortality

Current modelling assumes natural mortality $M = 0.5$ and imposes limited normally distributed uncertainty around this when uncertainty in this parameter is included (standard deviation of 0.0325).

One method that has been used to estimate natural mortality (M) is based on the ratio of “cluckers” (“clappers”) to live scallops, which, under some assumptions (including an equilibrium assumption) should be proportional to M, with a constant of proportionality of $1/S$, where S is the time it takes the cluckers to separate. Separation times may be length-dependent, and the separation times for smaller scallops may be too small for the clucker ratio approach to be useful. Some types of natural mortality, such as predation by decapods and fish, may not produce cluckers, so that natural mortality from the clucker ratio may be biased low. Additionally, uncertainty in the separation time could bias the point estimate of the clucker ratio low, since it is a ratio estimator.

Tagging can provide an alternative method for the estimation of natural mortality, but it is important to assess the extent of tagging-induced mortality since that can be confounded with natural mortality. Discard mortality estimates might help inform this to some extent.

The ratio M/K is considered a life-history invariant for related species (Beverton and Holt 1959), where K is the Brody growth coefficient, one of the parameters of the von Bertalanffy growth model. The estimated value of M/K for Atlantic sea scallops is about 0.4. Thus, M could be estimated from a value of K if that was available; although it is recognised that such estimates may only be applicable to larger scallops.

Confidence intervals for M are more likely to be skewed to the right (positively skewed) than to be symmetrical. For example, if the point estimate of M is 0.4, 0.6 is a more probable value of M than 0.2. One possibility is to model the distribution of M as $1/G$, where G is a gamma random variable

(Hart 2013). This guarantees that the value of M will be positive as well as skewing the distribution to the right (see **Figure 2)**.

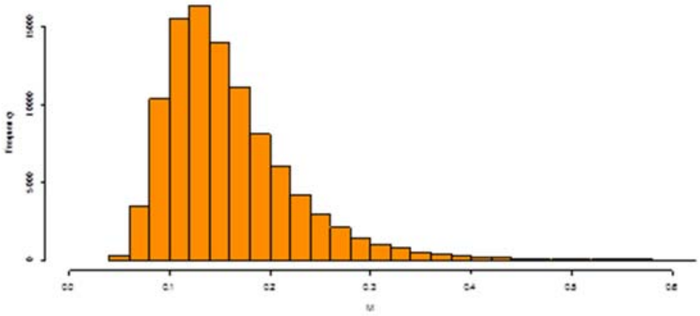

Figure 2. An illustration of the estimated probability density of natural mortality (*M*) for the U.S. Georges Bank sea scallops (*Placopecten magellanicus*).

Incidental fishing mortality

Based on limited experimental evidence, incidental fishing mortality from box dredges is high, and strongly affects reference point and yield per recruit calculations. For that reason, more experimental data on incidental fishing mortality as it occurs on different substrates and also using the southern bag-type dredge would improve estimates of available biomass and YPR.

Commercial fisheries data

The spatial distribution of recent commercial fisheries continues to hold value when used to aid survey designs. This would especially be the case if they are to be used to focus surveys on the core fishery areas. The identification of those core reliable fishing areas, within particular reporting areas, needs to be agreed to by all parties so that survey designs can become more repeatable and focused.

If the *limitCPUE* harvest control rule is to be used then an agreed method of estimating mean CPUE is needed. At very least it would need to be region or sub-region specific, it would require catch-weighting (although if it is based on individual records, as it should be, then weighting is automatic with reference to the number of records). Finally, standardisation of the CPUE must include vessel and/or skipper effects as a minimum.

3. The Most Appropriate Target and Limit Reference Points

The survey method for setting catch limits

The current survey based harvest control rule (HCR) entails obtaining an estimate of exploitable biomass expected at the start of the following season (B_{beg}). This is then multiplied by a given harvest rate (fishing mortality rate; F_{ref}) to provide a predicted catch for the season using (M is natural mortality rate):

$$CAY = \frac{F_{ref}}{F_{ref} + M}\left(1 - e^{-\left(F_{ref} + M\right)}\right)B_{beg} \qquad (1)$$

This is the standard equation defined in the annual plenary stock assessment document as part of Method 2 (MPI 2014), although three other equations are also suggested in the same document for use if most of the fishing mortality occurs over a short period each year:

$$CAY = \left(1 - e^{-F_{ref}}\right) B_{beg} \tag{2}$$

$$CAY = \left(1 - e^{-F_{ref}}\right) e^{-\frac{M}{2}} B_{beg} \tag{3}$$

$$CAY = \left(1 - e^{-F_{ref}}\right) e^{-M} B_{beg} \tag{4}$$

These equations are applicable to situations where most of the fishing mortality is at the start of year, in the middle of a year, or the end of a year, respectively.

Use of the catch equation

The Baranov catch equation was derived to calculate catch in numbers from a cohort, and takes into account simultaneous fishing and natural mortality. It is not correct to use the catch equation to estimate catch or CAY in biomass, since growth of individual animals and recruitment to the exploited size classes is not taken into account. For biomass, the catch equation is:

$$C = \int_0^T FB(t)\,dt = \int_0^T FB_0 \exp(-Zt) g(t)\,dt \tag{5}$$

where g(t) represents the growth (in biomass) of the population as a fraction of the starting biomass in the absence of mortality. If g(t) = 1 for all t, then the above equation is equivalent to the Baranov catch equation. However, typically g(t) > 1, and so using the Baranov formula will underestimate catch. The integral in the above equation can easily be estimated numerically.

The CPUE limit rule model

The CPUE rule does not translate into a fixed fishing mortality. In productive areas, where CPUE can exceed 200 kg/hr, a reduction to a limit of 75 kg/hr translates into an instantaneous fishing mortality rate of almost 1 (~63% per annum), so that growth overfishing can occur in these areas. Additionally, these higher density areas may contribute most of the larvae because scallops likely need to be in close proximity for eggs to be successfully fertilised. Thus, removing a high proportion of scallops in these areas may remove important spawning aggregations and reduce fertilised egg production, which may have negative implications for sustainability.

In less productive areas, for example where the CPUE starts at 100 kg/hr, a reduction to 75 kg/hr corresponds to an annual fishing mortality of less than 0.3, so that these areas will be underfished from a yield per recruit perspective. This means that the CPUE rule will underfish some areas and overfish others, and hence the realised YPR will be far less than optimal, regardless of the overall mean fishing mortality (Hart 2001, Truesdell et al. 2016). Rotational fishing of the most productive areas (e.g., where each area is alternately closed for one or more years, depending on circumstances within each bed, and then opened for one year) would prevent growth overfishing of the productive areas, increase larval production, and possibly enhance recruitment.

One strategy that might ameliorate some of these potentially negative effects would be to include an extra input control such that there was an inter-annual rotation of fishing between high density areas. Such rotational fishing obviously alleviates fishing pressure in the no-fishing years, which allows for undisturbed reproduction and on-going growth. Thus the advantages of a rotational strategy include ensuring at least some egg production but also reducing the likelihood of growth overfishing. Rotational fishing was an important component of the Golden Bay and Tasman Bay re-seeding program while it was operating successfully.

Coromandel scallop fishermen reported in the workshop that they currently apply informal at-sea cooperation that limits the pressure applied to dense concentrations of scallops. In addition they also concentrate on larger animals and back off if they discover large concentrations of small animals.

Yield per recruit analysis

It is important to include incidental fishing mortality in yield per recruit analysis. However, it is not necessary to use an IBM-type simulation to do so (Cryer *et al*., 2009). Instead, the incidental and discard mortalities can be included in the total mortality terms (see the Appendix in Hart 2003). This would produce a more well-defined and simpler curve. Code can be supplied to assist this work.

The panel was struck by the mismatch between the gear selectivity and the minimum legal size. Increases in the ring size would likely improve yield per recruit, and should be investigated both experimentally and theoretically; it may also reduce mortality from discarding of undersized scallops. A simple starting point would be to simply shift an existing selectivity curve to the right, and rerun the YPR analysis. If that indicates that such a shift would improve YPR, experiments could be done to compare existing gear to ones with larger ring or mesh size. In SCA 7, the existing and proposed gear could be towed side by side.

4. Stock Status Determination

Stock status

Determination of the 'status' of a stock can only be done quantitatively by comparing the performance of a stock relative to its objectives, which are described in terms of limit and target reference points. A target reference point of 40% of unfished spawning biomass literally implies that the objective of a fishery is to conduct itself in such a way that the stock will approach and remain around the level of 40% B_0. The expectation when setting a target reference point is thus that it is possible to manage a fishery towards the target and act in a such a way to keep it close to that target (hence it is at least one objective for that fishery). In contrast, the expectation for a limit reference point is that it is possible to manage a fishery away from the limit back up towards the target.

Given the stochastic recruitment dynamics and spatial heterogeneity in density, growth, and natural mortality of scallops, it may be the case that classical target and limit reference points, which tend to be based on setting specific spawning biomass or fishing mortality levels, are not suitable for scallop fisheries. Scallop stocks may not be capable of being managed in such a way as to be able to meet the expectations of standard reference points in a predictable way. Even completely stopping fishing will not necessarily lead to stock recovery in a predictable time. In a review of many different scallop beds in the Tasmanian and Australian Commonwealth fisheries (Haddon 2011) some beds were found to have a delay of five or fewer years between significant settlements but others could have delays between 7 – 10 years and one occurred at a 15 year interval and another at a 20 year interval. It is possible that settlements occurred without being detected but generally, when the fishery was open each bed was visited by more than one fisher when exploring.

It remains possible to set biomass and fishing mortality reference points and, given the natural variability in the stock size of scallops, achieving chosen fishing mortality levels may be more feasible than meeting biomass targets and avoiding biomass limits (see Prager *et al,* 2003, Hart, 2013, and associated literature). Nevertheless, other definitions or proxies for success in a scallop fishery may be required (for example, having a scallop fishery in at least 9 years out of ten might define success reasonably well in an episodic fishery). More discussion of realistic aspirations for each scallop fishery, that recognises how the inherent variability of the species may make the imposition of biomass targets and limits impractical, is required in at least the shellfish working group.

It needs to be noted that a Yield-per-Recruit analysis may be able to provide a sensible fishing mortality target or limit (perhaps $F_{0.1}$), although the implied equilibrium biomass target may not be considered realistic because it requires equilibrium conditions, which does not appear to apply well to many scallop fisheries. While candidate fishing mortality limits from YPR do not give any direct measure of preserving spawning biomass, it is more important for scallops to preserve spawning aggregations to ensure spawning success. On the other hand, note that the preservation of spawning aggregations is not guaranteed by the usual application of $F_{0.1}$ or F_{MSY} as a limit. election of a specific upper limit to fishing mortality would assist management meeting its need for a limit reference point and perhaps even formalise what appears to be the practice of industry in the Coromandel fishery of ceasing fishing on highly dense beds before the limit CPUE level is reached (as reported to the review workshop).

The survey based catch-rule which uses a biomass estimate to generate a catch limit also does not give a stock's 'status' although, empirically it should be able to provide an estimate of a safe level of catch. But neither the survey-rule or the limitCPUE rule effectively determine stock status in an absolute sense. Fortunately, such empirical harvest control rules can maintain sustainability; they just don't explain the dynamics that are observed within a fishery.

Use of a model

Attempts to understand the underlying dynamics of any fishery usually involve the use of a model of those dynamics. Model use generally requires more data than simple empirical catch control rules but they can usually generate estimates of biomass and fishing mortality in such a manner that the more classical reference points can be used. However, for *Pecten* fisheries, the uncertainty around model estimates has invariably been high so that in many cases the fishery can appear to have minimal effect on the dynamics.

Scallop fisheries can certainly be overfished, which implies that the fishery can certainly affect the dynamics, but even if catches are controlled, the signal of the fishery on the stock can disappear within variations due to other causes. For this reason it is not recommended that a model be applied to New Zealand stocks. The most stable fishery appears to be the fishery in the Coromandel. But even there, a whole new bed was only discovered recently. The data requirements and coverage of data collection required for most models are such that fisheries need to be large and valuable to justify the expense of the on-going gathering of such representative data.

Models provide population estimates by scaling relative abundance data (survey estimates or possibly CPUE) by catch and other aspects of the assumed population dynamics. The current approach for the New Zealand fisheries scales abundance data to the population using dredge efficiency to provide estimates of exploitable biomass. If an improved understanding of the dynamics and sustainability of the scallop fisheries is wanted (rather than just using empirical catch rules), then it seems likely to be more effective to improve the estimates of what is being currently estimated. If the surveys were to become more repeatable then an index of relative abundance through time would become available.

Determination of the best way to move forward remains a policy decision. But the review panel felt that from a scientific point of view it would be better to obtain improved estimates of dredge efficiency and exploitable biomass than to attempt to apply a model to the stock dynamics at this time. If the survey-rule or limitCPUE-rule were used with a regime of rotational fishing of identifiable scallop beds, this may be more achievable than using a stock assessment model.

There may also be room to explore other options that may assist the scallop fisheries; for example, the current practice is to remove the whole animal from the sea and shuck them on land. This has compliance advantages but it has the secondary effect of removing an enormous amount of shell from the scallop beds and sea-bed. If dead scallop shells aid the settlement of post-larval scallops, as has been observed for some other scallop species, then this is a poor practice and it would be worth experimenting on whether returning scallop shells to the vicinity of scallop beds would improve their viability.

Estimation of surveyed biomass

Information was provided to the panel on the stratified random surveys that have been used in SCA 1, SCA 7 and SCA CS. Annual surveys were undertaken in SCA 1 and SCA CS for a number of years. Coverage and strata definition have varied over time and in many of the areas, the surveys concentrated on the beds of commercial interest for the upcoming season. Survey catches were corrected by swept-area; however, gear mensuration instruments that could minimise the corrections needed are not currently used in these surveys.

Dredge stations were allocated to strata using the Allocate.r function which determines the allocation of stations to obtain a target coefficient of variation (CV) or to minimise the CV of the overall estimate based on a fixed sample size and previous year's survey catch densities (Francis 2006). However, the allocations are often modified in the field to accommodate time and area considerations. There does not seem to be a post-survey evaluation of the performance of the realised allocation in terms of the CV of the estimate (see Smith and Gavaris 1993, Francis 2006).

In all areas there is a 10 step process that is used to convert survey catches at length to population estimates, green-weight biomass at the time survey and at the time of fishery, and expected meat weight. Steps 3 to 10 are applied to each bootstrap sample of survey catches and final estimates are functions of these bootstrap replicates. The major steps are as follows:

Step 3: Correct survey LF to population: Dredge efficiency (SCA CS or SCA 7)

Step 4: L/W regression: log-log regression without bias correction (SCA CS data)

Step 5: Stratified mean/total: standard formula for stratified random

Step 7: Growth model: Based on data from 129 recaptures, 1990–1997, Mostly from SCA CS, log regression

Step 8: Natural mortality: sample from Normal distribution (mean = 0.5, standard deviation = 0.0325)

Step 10: Predict meat weight: based on fishery recovery rates (SCA CS or SCA 7)

For steps 3, 4, and 7 random samples of stored parameter estimates for the associated model are obtained by bootstrapping (3 for SCA 7, 4 and 7) or from the MCMC posteriors (step 3 for SCA CS) and are used to convert the data. Natural mortality is generated using a random number generator and a recovery rate is randomly sampled from the actual annual rates from the associated fishery.

The population biomass estimates were obtained as the medians of the bootstrap replicates and the associated variances and CVs are also obtained from the replicates. The 95% confidence intervals were calculated as the 0.025 and 0.975 percentiles of the replicates.

The bootstrap methodology was designed to estimate the variability of particular estimates but not to replace the original estimate. Calculating the mean or median of the bootstrap replicates is usually done to determine if there is any bias in the estimate. The bootstrap survey means and medians of green-weight biomass at the time of the survey for SCA CS (2007–2012) and SCA 7 (2009–2015) were compared with the stratified estimate of the same calculated using posterior mean parameter estimates for dredge efficiency and the log-log regression estimates based on all of the data. The differences between the stratified and bootstrap mean were around±1% and 1–3% between the stratified mean and the bootstrap median indicating that the (non-bootstrapped) stratified mean estimates appear to have little bias associated with them.

In the usual application of the bootstrap to survey means unencumbered by all of the processing in steps 3-10 above, the straightforward bootstrap sample of tows with replacement from each stratum will result in an underestimation of the actual stratified variance and as a consequence confidence intervals will have less than the expected coverage (e.g., 95%; see Smith 1997 and references therein). For the survey bootstrap replicates investigated above, the bootstrap standard error was 0.99 – 2.8 times higher than the stratified standard error. However, the generally higher bootstrap estimates reflect the additional variation contributed by steps 3 and 4 and may actually be underestimates due to the simple resampling within stratum. Alternative resampling schemes that provide more accurate estimates standard errors and confidence intervals are discussed in Smith (1997).

There are number of ways of calculating bootstrap confidence intervals from bootstrap replicates (e.g., percentile, bias-corrected, bias-corrected accelerated) and while these should be investigated, percentile based limits are often suitable.

Estimation of dredge efficiency

Dredge efficiency models have been developed for the New Zealand scallop fisheries to estimate the population numbers at length from the survey catches. The first to be used was a non-parametric model (Cryer & Parkinson 2006) based on diver and dredge data from 32 dredge efficiency experiments that have been carried out in Coromandel and East Northland in the 1980s and 1990s. Some concerns were raised during this meeting and in Bian et al. (2012) regarding this method, which the panel shares. Recently, this model was replaced by a more complex model applied to the same data sets that uses a six parameter dredge efficiency model (Bian et al. 2012). The panel identified a number of issues with this more complex model.

The diver and dredge counts of scallops were both modelled using negative binomial distributions. The mean for the diver counts was defined as $a_{si}\delta_{slt}$, where a_{si} is the area searched at site *s* during dive *i* and δ_{slt} is the density for site *s*, length bin *l* in year *t*. The mean for the dredge counts similarly included the density term δ_{slt}, area dredged A_{sj} in the *jth* dredge sample and the efficiency function $\mathrm{E}(l, \alpha)$ where α refers to covariate terms affecting efficiency. In this configuration density on the bottom is jointly estimated by both the dredge and dive data in combination with the efficiency terms. Tremblay et al. (2006) conducted a similar type of experiment with divers and traps to determine the catchability of lobsters and used a similar configuration for the model but found that the catchability could only be estimated when the trap data was modelled conditional on the density from the diver data. It is possible that the joint estimation appears to be working here because of the hyper prior set on the density and the restrictive priors set for the efficiency parameters.

The efficiency model has six parameters to describe the shape of the efficiency curve with shell length but three of the parameters are associated with very small or large length classes for which there are little data available. "Informative" priors based on literature sources were used in Bian et al. (2012) to deal with this issue. The panel felt that some of these priors were too confining. In particular, the priors suggested almost near-certainty that selectivity is domed at large sizes. While the possibility of doming cannot be ruled out and there is some existing literature to support this, the panel does not agree that this evidence is strong enough to eliminate at a high level of certainty the possibility of flat-topped selectivity *a priori*.

These priors were all defined as lognormal random variates with specified means (μ) and coefficients of variation (CV). The contribution to the log-likelihood for each prior was defined as:

$$\ell^p(\alpha) = ln(\alpha) + 0.5\left(\frac{ln(\alpha) - ln(\mu)}{s} + \frac{s}{2}\right)^2 \tag{6}$$

where α represents any one of the efficiency parameters and $ln(\mu)$ and s, represent the mean and standard deviation in the log scale. While the standard deviation was properly calculated from the original scale of measurement, $s = \sqrt{\ln(1 + CV^2)}$, $ln(\mu)$ is incorrect and the mean should have been calculated as

$$\mu_{ln} = \ln\left(\frac{\mu}{\sqrt{1+CV^2}}\right) \tag{7}$$

Use of this formula would make a small change to the means of the priors (e.g., μ=0.55 for ε; $ln(\mu)$ = -0.598 and μ_{ln} = -0.641).

A more serious point of concern is the penalty term *s/2* in the log-likelihood which will result in restricting the range of the posterior for each parameter. The comparison presented in Bain et al. (2012) of the prior and posterior distributions for each of the efficiency parameters shows that means of each parameter line up closely while the ranges of the posteriors were quite reduced compared to the priors (**Figure 3**, see below). This is primarily due to the penalty term, rather than influences from the data, as evidenced by the fact that the relationships between the prior and posterior appear almost the same for all six parameters, and that the posteriors appear to be very precise even for the parameters which are informed by few data. While these figures were interpreted in Bain et al. (2012) as indicating that the means set for the priors were appropriate for dredge efficiency parameters, the usual interpretation is that the lack of differences between the prior and posterior means indicates that there is little if any information in the data to estimate the efficiency parameters. That is, the priors appear to be the main source of the parameter estimates not the data. Thus, the differences in the prior and posterior variances were primarily due to the penalty term. The impact of using these priors could be evaluated by moving the means away from where they were set and re-running the model to see what happens to the posteriors and whether they line up again. In addition, runs with the penalty term removed should be conducted to evaluate its impact on the posteriors.

A number of covariates expected to have effects on the dredge efficiency parameters were included in the model. Five of the covariates were categorical (substrate, depth, early tow termination indicator, mesh size and dredge width (box dredge)) while the length of the tow was included as a continuous covariate. It is unclear why depth was set as a categorical variable other than to possibly deal with uncertainties with respect to the range of depths swept by the gear. It was also unclear why dredge width was set as a categorical variable. Recall that swept area was included in the definition of the mean and so length of tow is already included in the model while swept area also includes the actual width of gear and therefore there is overlap with the width categorical variable.

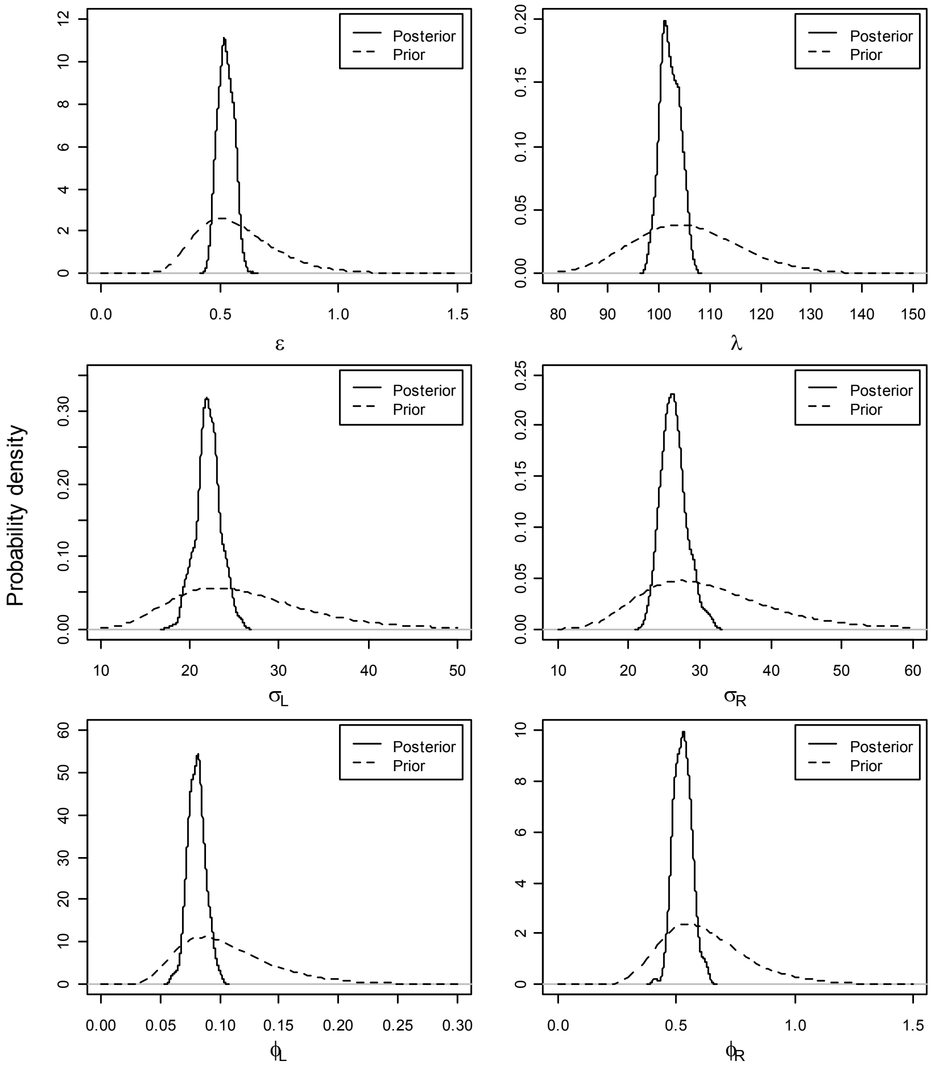

Figure 3 from Bain et al. (2012): Dredge efficiency parameters' posteriors verse their priors for model SDT (S=CM, D<25m, and T=True). Panel top left for ε, panel top right for λ, panel left for σ_L, panel right for σ_R, panel bottom left for ϕ_L, panel bottom right for ϕ_R.

Modelling the categorical variables as 5-way factorial main effects model would result in a large number of empty cells because not all combinations of levels will exist. In addition, many of the combinations that do exist may do so for only one level of most of the factors (e.g., early tow termination only occurred in depths < 15 m on mud bottoms in Coromandel, Williams et al. 2013). The method described in Bian et al. (2012) and also discussed in Williams et al. (2013) to account for this lack of balance in the covariate combinations is very confusing. Apparently, those combinations of different categorical covariates that did exist were defined as unique sets (Williams et al. 2013).

α_0 = (CM, D=<15m, Term=F) + (W, M+L)
α_1 = (CS, D=<15m, Term=F) + (W, M+L)
α_2 = (NS, D=<15m, Term=F) + (W, M+L)
α_3 = (CM, D=15-20m, Term=F) + (W, M+L)
α_4 = (CM, D=20-25m, Term=F) + (W, M+L)
α_5 = (CM, D=>25m, Term=F) + (W, M+L)
α_6 = (CM, D=<15m, Term=T) + (W, M+L)

where C=Coromandel, N=Northland, M=mud, S=sand, D=depth, Term=early termination indicator, W=gear width, M=mesh, L=length of tow, T=true and F=false. Note that α performs double duty by representing both the vector of efficiency parameters and the vector of covariates used to model these efficiency parameters. Williams et al. (2013) claim there are six parameters associated with each of the seven combinations given above. However, as main effects many of the terms in these combinations are shared between combinations and therefore are not unique to them. Each of these combinations appear to represent interaction terms and therefore may only contain one parameter or two, maybe three each depending upon the combinations of W and M; or possibly an additional parameter if tow length is simply an additive continuous covariate. As an unbalanced factorial design, the main effects for each of the categorical covariates may not be estimable. More clarity is required with respect to exactly how the covariates were included in the model.

The DIC criterion was used to evaluate whether any of the covariates needed to be included in the model. Note that this kind of model could be interpreted as a mixture model of a sort since both dive and dredge data are being modelled at the same time with at least one shared parameter. The analysis in Tremblay et al. (2006) was conducted with WinBUGS and that software identified the joint use of the two models for the dive and trap data as a mixture model resulting in the usual measure of complexity for DIC being invalid. This may also be true for the use of DIC for the dive and dredge model discussed here.

The panel noted that the dredge efficiency model estimates from Bian et al. (2012) were only used for the Coromandel survey and a different approach was used for the SCA 7 survey. In SCA 7, efficiency was estimated from data obtained from two studies in the area. Estimates consisted of weighted averages of the average density of scallops taken by dredge, divided by the average density of scallops taken by divers grouped by broad length bins over four separate area-study combinations (Tuck and Brown 2008). The estimates for scallops sizes used for projecting the commercial biomass to the beginning of the fishery were similar to those in the other SCA areas but with the broad length bins and the conversion to the population estimate was mostly around twice the survey estimate.

Determination of the dredge efficiency is key to using scallop surveys to estimate absolute population abundance and biomass. As such, it is important to continue experimental and modelling work to develop estimates of efficiency. However, the general approach used in New Zealand to date was to estimate both selectivity and efficiency simultaneously, which led to problems estimating the numerous parameters. It may be more fruitful to consider modelling these two processes separately, since after selectivity has been estimated, efficiency can be obtained by estimating a single parameter only. The former could be modelled based on data from paired tows such as lined and unlined gear or different ring sizes (e.g., Yochum and DuPaul 2008).

5. Management Strategy Evaluation Testing of Control Rules

Introduction

The management strategy evaluation (MSE) testing and comparison of the performance of the *limitCPUE* harvest control rule relative to the performance of the survey based harvest control rule was conducted based on the Coromandel scallop fishery. Nevertheless, the review objective is explicit that the examination should relate to the potential use of the “CPUE Limit Rule model as a reliable monitoring tool for sustainable commercial fishing in New Zealand’s scallop fisheries.” The characteristics of the three main fisheries (SCA 1 – Northland, SCA CS – Coromandel, and SCA 7 – Challenger) are rather different, with SCA 7 currently restricted to the Marlborough Sounds, SCA CS relatively continuous from 1995-96 to 2015-16, albeit with the halving and doubling of catches over short periods of years (MPI 2014). The recent inclusion of previously unfished scallop beds inside of the Hauraki Gulf led to an increase in TACC although not to a great increase in landings (MPI 2014). Finally, SCA 1 differs by having a minimum legal size of 100 mm instead of 90 mm as elsewhere. It appears to occur more episodically than the other fisheries and since 2010 has also been at a relatively low level. Applying a *limitCPUE* harvest control rule anywhere other than Coromandel might therefore involve difficulties not experienced in the Coromandel (the requirement for full cooperation from all fishers, an effective CPUE monitoring system independent of the statutory data collection, calibrating particular soft and hard limit reference points, and, in SCA 7, determining whether the use of a *limitCPUE* is likely to be useful in a fishery that was originally designed to be enhanced by re-seeding).

Evaluation

The comparison of the CPUE limit harvest control rule (CPUE-rule) and the survey harvest control rule (survey-rule) was strongly influenced by the inclusion in the survey-rule of an inflexible *baseTACC*. The use of fixed *baseTACC* levels of 22 t and 52 t is unrealistic because if the survey indicated that the stock were at very low levels then fishing would stop as, for example, has happened in Golden Bay and Tasman Bay. To imply that the fleet would continue to try to take the *baseTACC* distorts the dynamics of the survey harvest strategy and it is not surprising that this led to failures to avoid the stock falling below $0.2B_0$. Importantly, in the additional analyses, when the survey-rule was run with a zero *baseTACC* the probability of falling below $0.2B_0$ was greatly reduced (Haist and Middleton, 2014). The important difference is that for the two lower mean catch levels in the 0 t *baseTACC*, the survey-rule leads to risk levels either below 10% or very close to it and appear to be similar to that of the CPUE-rule for similar mean catch levels (**Figure 4**).

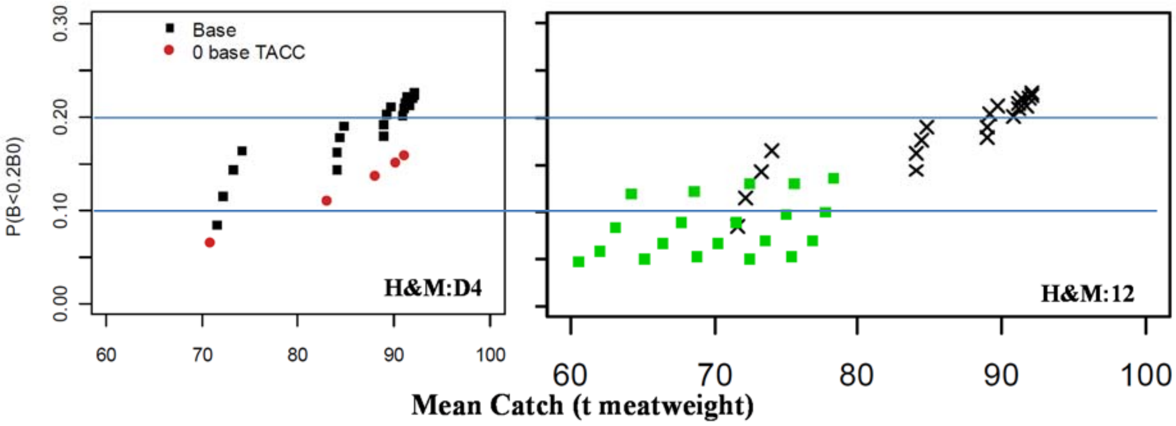

Figure 4. Two figures extracted from Haist and Middleton (2014) comparing the probability of the biomass falling below $0.2B_0$. The left panel is Fig. B4 (page 53) and the right panel is Fig. 12 (page 33), where the green boxes represent results for the *limit*CPUE rule and the black x's are for the survey limit rule. The left hand panel depicts an additional analysis whereby the *baseTACC* was set to 0 t rather than either 22 t or 52 t.

The negative effect of the *baseTACC* can also be seen clearly in Haist and Middleton (2014, Figures 7 and D4; **Figure 5** in this document). In the CPUE-rule the catches can be seen to vary each year with minimal catches during low biomass periods and upper limits set by the TACC. However, the lower plateau on catches, reflecting the constant *baseTACC*, identify periods when the survey harvest strategy would recommend taking lower catches and yet the *limitTACC* is still assumed to be taken. That is, the survey method as implemented in the MSE becomes a "constant catch" strategy at low biomass, which, as is well known, can cause sustainability risks.

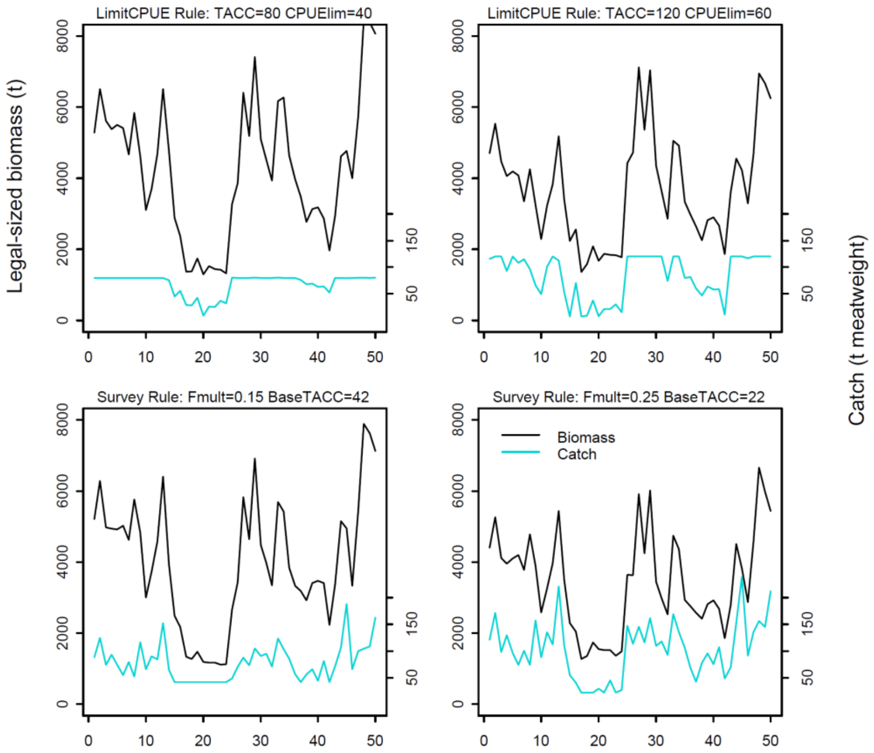

Figure 5. Part of Figure 7 from Haist and Middleton (2014) illustrating the influence of the unrealistic fixed *baseTACC* on the outcome of the simulations.

In management strategy evaluations (MSE), the use of 3010 year projections, and only one projection per scenario, instead of multiple shorter projections from known starting states, is unusual. While it is a strategy aimed at answering the long-term expectations of such fisheries it makes it difficult to understand the drivers behind any properties exhibited by the model. If this MSE is to continue to use this long-term projection approach then the inclusion of diagnostics for each scenario, such as the annual harvest rate implied by the simulation, might improve confidence in the realism of those projections.

It seems likely that initiating the simulated scallop stock at different pre-set levels of depletion and projecting these for fewer years (20 – 50) over many replications would permit the relative effectiveness of each management strategy (survey-rule vs *limit*CPUE-rule; plus others) under different conditions to be made clear. That is, the results for each management approach can be evaluated when the stocks are at a low level, at a middling level or at a high level.

In the MSE the fact that the fishery only has a minor effect on the stock dynamics is seen even more clearly when more years of each 3010 year projection are plotted (**Figure 6**).

One projection for each scenario (even for 3000 years) is not enough to adequately characterise the properties of each management strategy. Multiple projections, using a different random seed, are needed to canvas the possibilities implied by the model and its assumptions. Many of the assumptions would ideally need to be tested as well; for example, the appropriateness of the survey catchability estimates and the form of the relationship between CPUE and abundance.

MSEs often record and plot rates of meeting or violating constraints or conditions within the management strategies (e.g., How often does the *baseTACC* limit catches or how often was the *limitCPUE* reached?). This would also improve the presentation of the MSE outcomes and make them more persuasive (see Punt et al, 2014).

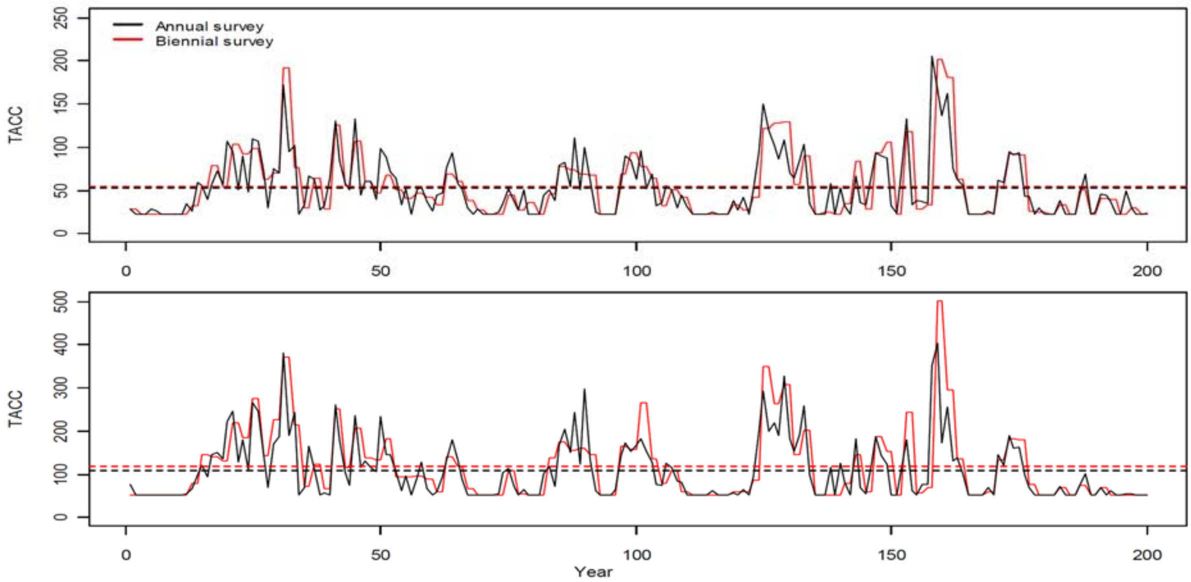

Figure 6. The first 200 years of the projections in two scenarios of the MSE. The first ten years used to burn in the model can be seen on the left-hand side. The same random number seed must be used as reflected in the dynamics in the two scenarios being extremely similar. This is Figure D2 from Haist and Middleton (2014).

6. Recommendations

1. A synthesis of scallop documentation detailing what was done where would help ensure that standard methods are used.

2. Improved documentation across all methods used would make all assessments and analyses more defensible.

3. Add performance sensors to the survey dredge gear to minimise the amount of correction required to convert to density for a standard swept area.

4. Evaluate the degree of bias in the estimate of the standard error and confidence intervals for the simple bootstrap for the stratified mean (mean parameter values in steps 3-5) compared to the bootstrap sampling schemes given in Smith (1997) or others that may have appeared more recently in the literature. Investigate the applicability of one or more of these bootstrap sampling schemes through simulation to evaluate the actual coverage probabilities for the confidence intervals.

5. The variable coverage of the survey areas over time and the different strata definitions used have resulted in a series of annual estimates that are not always comparable with respect to changes in abundance or biomass over time. A focus on consistent coverage of “core” areas that reflect long term high productivity for scallops could help meet budget restrictions and provide a consistent time series. New areas that exhibit high productivity could be added either temporarily or permanently, but should not replace the core area coverage.

6. Fine scale habitat maps have proven useful for understanding scallop distribution and productivity and any opportunities to obtain bottom type information via multi-beam, or other

similar technology through joint studies with other interests or stakeholders in the area should be explored.

7. Underwater camera surveys should be considered as an option as they can provide information on scallops densities, habitat structure and other associated species at a finer scale and with more accuracy and higher coverage than dredge gear. Such surveys are currently routine for the weathervane scallops in the Gulf of Alaska and sea scallops on the US east coast. Note that physical samples of scallops will still be required for length frequencies and other biological measurements.

8. Continue work on estimating dredge efficiency. The current data sets used for both the SCA CS and SCA 7 areas have their challenges with respect to modelling and application and attention should be directed to new studies that can address some of these challenges to simplify the modelling. Also these studies should be directed towards estimating selectivity separately from efficiency.

9. The extent of incidental mortality has an important impact on the calculation of YPR, TACCs, and long-term yield. It would therefore be worthwhile to conduct further studies regarding this process in different substrates and gear configurations.

10. Shell material provides either directly or indirectly important habitat for settling spat and post-settlement juveniles in many pectinid species and related taxa (e.g., oysters). It is possible that the declining landings and abundance observed in New Zealand scallops is due at least in part to the practice of shucking on land and not returning the shells to the scallop grounds. Experiments could be conducted where shells are returned to experimental plots to determine whether this hypothesis is valid for New Zealand scallops.

7. Appendices

Terms of reference

An independent, external, expert panel comprising Stephen Smith (DFO Emeritus, Canada), Dr Malcolm Haddon (CSIRO, Australia) and Dr Dvora Hart (NOAA, USA) will be convened. Collectively, the panel has scientific expertise in scallop fishery stock assessments, target and limit reference points for scallop fisheries and scallop biology.

Panel members must have no connection with the original work and must declare any actual or potential conflicts of interest that might affect their ability to come to an objective view of the data, the model structure and results and any alternative approaches.

Several people will assist with the review:

1. James Williams (NIWA): the lead researcher for recent scallop stock assessments
2. Vivian Haist: Vivian developed the Management Strategy Evaluation model for Coromandel scallop fishery.
3. Julie Hills (the Chair of the Shellfish Technical Working Group) and Pamela Mace (Workshop Chair).
4. Erin Breen and Allen Frazer: MPI Fisheries Managers

The primary objectives for the expert panel are to:

- Review the historical data collected from SCA 7, SCA CS and SCA 1 (including survey data, biological data, dredge efficiency data and catch data) and provide recommendations on:
  a. How effectively the data are currently being used to assess the status of the stocks in SCA7, SCA CS and SCA 1, additional data that could be collected and improvements to data collection methods
  b. The most appropriate target and limit reference points to use in the SCA 7, SCA CS and SCA 1 fisheries in order to achieve sustainable utilisation on these fisheries.
  c. The most appropriate method to use to assess the status of the stocks in SCA 7, SCA CS and SCA 1. For example: developing a delay difference stock assessment model, or a length based stock assessment model.
  d. The CPUE Limit Rule model as a reliable monitoring tool for sustainable commercial fishing in New Zealand's scallop fisheries.

The expert panel will summarise their findings and any recommendations in a report to the Principal Advisor Fisheries Science, Ministry for Primary Industries. Where consensus cannot be reached by the external reviewers, any differences of opinion should be recorded in their report.

Background documents

All documents are stored in the Fisheries Science Working Group Website:
http://cs.fish.govt.nz/

Out of scope

It must be noted that the scope of this review and workshop is to conduct a technical review of the data and research associated with New Zealand's scallop fisheries, how effective the current research programme is in informing management decisions and improvements that could be made to the research programme. Aside from an initial background presentation from MPI providing the management context for scallop data collection and research programmes, management issues will not be discussed in this workshop. In particular, the following are out-of-scope for this review:

- the efficacy of past scallop fisheries management measures themselves, or the QMS in general
- the efficacy of local area scallop management actions
- the efficacy of any possible future management actions
- other issues not directly related to the Terms of Reference

Please give this careful consideration before deciding if you wish to attend the workshop.

Format for review

The format for the review will be a workshop involving the independent external expert reviewers (“the Panel”), key players and other interested parties in Wellington, New Zealand to discuss the data, analyses and results in detail over a period of five days. The review will start with a number of presentations to ensure a common understanding of the work (about 1.5-2 days), and will be followed by a period of contemplation by the Panel, focused discussions with lead researchers or other parties (at the Panel’s discretion), and drafting of a report containing the Panel’s conclusions and recommendations (2–3 days). The Panel will present a draft version of their findings to interested parties on the morning of the final day to receive feedback and suggest corrections on matters of fact. The Panel may, at their discretion, reflect such feedback in their report. The aim is to have a near-final version of the Panel’s report by the end of the week, although it could take an additional 1-3 weeks or so until the final version becomes available. The final version will be made publically available once completed to the satisfaction of the review panel, but drafts will not be circulated.

Timetable

The workshop is set down for 29 February to 4 March 2016 and will be held in the Allen Boardroom, National Institute for Water and Atmospheric Research (NIWA), Greta Point, Wellington, New Zealand. Dr Pamela Mace, Principal Advisor Fisheries Science, MPI, will chair the open sessions.

Monday 29th February	Presentations to panel	Open session
Tuesday 1st March a.m.	Presentations conclude	Open session
Tuesday 1st March p.m.	Panel confers with individual researchers or works alone	Panel’s discretion
Wednesday 2nd March	Panel confers with individuals or works alone	Panel’s discretion
Thursday 3rd March	Panel continues working on their review and report	Closed session
Friday 4th March a.m.	Panel presents draft findings	Open session
Friday 4th March p.m.	Panel concludes review and report	Closed session

It is anticipated that the review can be largely concluded by 5 pm on Friday 4th March, although final drafting of the report may take place over subsequent days.

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Agenda
for independent review of New Zealand's scallop fishery stock assessment data and methods (revised)

29 February to 4 March
9:30 am to 4:30 pm
Allen Boardroom, Allen Building
NIWA
Greta Point, Wellington

Chair: Dr Pamela Mace, DDI 027 240 8262

Monday 29 February

9:30 am

- Introductions, housekeeping and general business
- Pamela will inform the working group of the Terms of Reference for the workshop and outline the itinerary for the week.

- Overview of the management of the three main scallop fisheries SCA 7, SCA CS and SCA 1 (Allen Frazer, Erin Breen, Tony Brett, MPI fisheries managers)

10:30-10:50 am

- Morning tea

10:50 am

- Research overview - scallop biology, survey based approach (James Williams, NIWA)

12:00 to 12:30 pm

- Lunch

12:30 pm

- SCA 7 fisheries and survey data (James Williams, NIWA)

- SCA 7 effects of enhancement (Ian Tuck, NIWA)

3:00 to 3:20 pm

- Afternoon tea

3:20 pm

- Growth data (Ian Tuck, NIWA)

- Dredge efficiency data (Ian Tuck, NIWA)

5:30 pm

- End of Day 1

Tuesday 1 March

9:30 am

- SCA CS fisheries and survey data (James Williams, NIWA)

10:30-10:50 am

- Morning tea

10:50 am

- Management Strategy Evaluation model in the Coromandel scallop fishery (Vivian Haist, Canada; independent advisor for MPI)

12:00 to 12:30 pm

- Lunch

12:30 pm

- CPUE Limit Rule implementation in the Coromandel scallop fishery (David Middleton, Trident Systems)

- YPR model including incidental effects (Martin Cryer, MPI)

- Projection approach (Ian Tuck, NIWA)

3:00 to 3:20 pm

- Afternoon tea

3:20 pm

- SCA 1 fisheries and survey data (James Williams, NIWA)

5:00 pm

- End of Day 2

Wednesday 2 and Thursday 3 March

Expert panel works alone or confers with particular working group participants

Friday 4 March

9:30 am

- Scallop stock assessment review panel will provide their draft conclusions and recommendations

11:30 am

- Morning tea

11:50 am

- Continue discussions with review panel

1:30 pm

- Working group meeting closes
- Review panel work alone

List of Attendees

Pamela Mace (MPI), Chair

Julie Hills (MPI), Shellfish Science Working Group Convener

Malcom Haddon (CSIRO, Australia), Dvora Hart (NOAA-Fisheries, USA), Stephen Smith (Canada): Independent Expert Panel Members

Paul Breen (Breen Consulting New Zealand; independent advisor for MPI); Vivian Haist (Canada; independent advisor for MPI)

Erin Breen, Tony Brett, Martin Cryer, Allen Fraser, Adam Watson (MPI)

Alistair Dunn, Rosemary Hurst, Keith Michael, Ian Tuck, James Williams (NIWA)

Michael Arbuckle, John Reid (Challenger Scallop Enhancement Company), Joshua Barclay ((NZSFC), Tom Clark (FINZ), Hilton Leigh (Coromandel Scallops), David Middleton, Oliver Wilson (Trident Systems), Philipp Neubauer (Dragonfly Science), Peter Sopp (Coromandel Scallops), Elisha Yahel (AFL)