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Evaluation of fisheries data used to determine the status of Queensland snapper (*Pagrus auratus*)

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The Australian Bureau of Agricultural Resource Economics – Bureau of Rural Sciences (ABARE–BRS) was engaged by Fisheries Queensland in September 2010 to undertake a evaluation of data used to determine the status of Queensland snapper according to terms of reference. This is the draft report of that review.

Summary

This report reviews the 2009 assessment of the Queensland snapper fishery at the request of Fisheries Queensland. It follows an earlier review by Dr Chris Francis.

The terms of reference for this ABARE–BRS review were narrowly targeted around some of the data inputs and assumptions within the assessment.

We are generally of the opinion that Campbell *et al.* (2009a, 2009b) have employed standard stock assessment methods and explored uncertainty in a manner that is consistent with accepted practice. Although there is considerable uncertainty, this does not imply that the trends and results presented in the report are wrong. Estimated trends in relative exploitable biomass appear robust under a number of sensitivity scenarios considered.

A number of issues highlighted in this review, particularly concerning data, suggest that the degree of uncertainty in the 2009 stock assessment of Queensland snapper is substantial. We cannot know whether the net result of alternative assumptions would increase or reduce the predicted depletion. The extensive set of sensitivities explored by the authors (Campbell *et al.* 2009a, 2009b) tend to provide consistent conclusions about stock status. From the assessment and its subsequent scientific review it is plausible that, for snapper, exploitable biomass in 2007 (B_{2007}) was less than 40 percent of virgin exploitable biomass.

There is substantial uncertainty in the catch history data. Major sources of uncertainty in the catch data are around numbers of fish caught by the recreational sector and the average weight of fish caught by the recreational sector over time. A modified catch history scenario was run, which considered historic catches 100 per cent above estimates with 20 per cent standard deviation. We feel that this run goes some way to addressing the uncertainty in the catch history data. However, we feel that there were problems with the way that average annual snapper weights were estimated in this assessment that are unlikely to be fully addressed by additional sensitivity runs.

There is very little information on the age structure of the stock over its period of exploitation. The assessment used length and age frequency data from 1994, 1995 and 2007 and exploitation was assumed to have begun in 1945. When tasked with fitting an age structured model for this period of time it would be preferred that similar data were available from many more than three years. This issue contributed to key uncertainties around potential changes in the age structure of the stock over time, average weight and hence biomass of the recreational catch, and the vulnerability schedules assumed.

It would have been extremely difficult to adequately address the uncertainty in the age structure of the stock. We feel that disproportionate weight was assigned, in general, to the length and age frequency data used in the estimation of the unfished recruitment parameter, R_0. This aspect of the

model might have limited the extent to which sensitivity runs were able to realistically explore the uncertainty related to other assumptions, although it is impossible to quantify.

Variability in the spatial distribution of recreational fishing effort, coupled with expected incomplete mixing of adult snapper between local populations, is an additional source of uncertainty for the assessment of Queensland snapper. The particular spatial characteristics of this fishery might produce levels of depletion that vary by region. If levels of depletion and exploitation rates vary across the fishery, the reliability of fishery dependent catch per unit effort (CPUE) from discrete parts of the fishery to reflect total abundance is questionable. We consider that the range of sensitivity runs that incorporated the commercial and recreational CPUE series adequately addresses uncertainty in the CPUE series.

The vulnerability schedules are also uncertain because of the lack of length and age frequency data over the period of snapper exploitation. The authors used the available length and age frequency data to estimate the vulnerability schedules outside of the model and consulted expert opinion. Given the paucity of data on catch at age, we feel that it was not possible for the full uncertainty around this aspect of the assessment to be addressed.

Discard mortality is uncertain. Sensitivity runs that considered different rates of discard mortality resulted in model outputs that were only marginally changed. The vulnerability of undersized fish to discard mortality was based on assumptions that were criticised by Francis (2009). We suggest the authors consider the estimates of the number of fish released by the recreational sector published in the RFISH survey reports (McInnes, 2008) in the estimation of the number of fish lost through discard mortality.

The above concerns may have affected model outputs, but any changes are impossible to quantify without more data and further sensitivity testing.

Given the nature of the assessment, and the many sensitivity trials carried out, the overfished status finding suggested in the original report remains plausible.

Background and Terms of Reference (from Fisheries Queensland)

In 2009 Fisheries Queensland released the stock assessment for snapper (*Pagrus auratus*), following on from assessments made in 2005 and again in 2006 using more limited data. The report 'Stock assessment of the Queensland snapper fishery (Australia) and management strategies for improving sustainability' by Campbell *et al.* contains the outcomes of the modelling using the best available fisheries data to 2007 and the outcomes of an independent review of the assessment. Based on the weight of evidence from this series of assessments, Fisheries Queensland has taken a precautionary approach and determined that the snapper stock in Queensland waters is overfished. Management options to achieve the rebuilding of the stock have been developed and will be released in the near future for public comment.

The stock assessment report has been made publically available and forums have been run to explain the outcomes of the assessment. Some stakeholders have expressed strong opposition to the use of outputs from the assessment to inform future management of snapper. The basis of this opposition appears to primarily relate to the estimates of standardised CPUE for recreational, charter and commercial sectors, the estimated total catch for the recreational sector and the snapper age/length data. The catch rates are important data inputs for the model together with total annual snapper landings, length and age structures and vulnerability schedules.

Sunfish, a peak body representing recreational fishing interests, have questioned the stock status assessment of snapper. Sunfish's view is that stakeholders cannot understand that snapper is severely depleted when data from the past decade or so shows that the total snapper catch continues to rise, total fishing effort is relatively stable, catch rates from commercial and recreational sectors (RFISH surveys which include charter components) are relatively stable - taking into account the impacts of changes to minimum sizes and bag limits; and the size frequency of snapper shows very strong presence of large/older fish in recent catches compared with those in the 1990s.

Snapper is an iconic species in the multi-species rocky reef fin fish fishery. Approximately 2/3 of the estimated total catch is landed by recreational fishers. Outputs of several state-wide surveys, targeted research surveys and boat ramp surveys have been used to produce estimates of total catch and catch rates for the recreational sector. Commercial logbooks and charter fishing logbooks have been used to provide estimates of total catch and catch rates for these sectors. In 2006 Fisheries Queensland initiated a fishery dependent Long Term Monitoring Program to derive estimates of age and length of catch from all three sectors.

Given this background, Fisheries Queensland decided to undertake a further review of the snapper stock assessment with a particular focus on some of the data inputs.

Terms of reference for review of the 2009 snapper stock assessment

1. **Total Catch**
   a) Comment on whether the sensitivity analyses in the stock assessment adequately deal with the level of uncertainty in the accuracy and precision of total recreational catch estimates. Stakeholders are concerned about: the potential recall bias from the recreational estimates of numbers of fish caught from the surveys prior to 2001 NRIFS; the potential for the catch of recreational fishers operating from charter vessels being included in the recreational catch estimates as well as in the charter logbook derived estimates (Charter logbooks were

compulsory from July 1994); the conversion of numbers of recreational fish landed to biomass and the use of biomass rather than fish numbers in the stock assessment models.

2. **Catch Rates (Catch per Unit Effort – CPUE)**
   a) Comment on the standardization process used to determine the CPUE for all sectors and implications for using these data (i.e. uncertainties around estimates) in the snapper stock assessment. Include the method used, underlying assumptions, variances around estimates, the use of CPUE estimates from both the Gold Coast charter industry independent data and the compulsory Charter logbook data, and the implications of snapper being part of a multi-species fishery.

   b) Comment on the appropriateness of the decision to not use the commercial CPUE data in the stock assessment as the base case due to potential hyperstability. Comment on and rank the likelihood for hyperstability existing in the commercial, recreational and charter CPUE data respectively.

3. **Length-Age Frequency**
   a) Comment on the methods used, underlying assumptions, and variances around the estimates of total mortality and implications for using this data (i.e. uncertainties around estimates) in the snapper stock assessment.

   b) What do the length-age frequencies and estimated mortalities from the 1994/95 and 2007 data infer about stock status.

4. **Vulnerability Schedules**
   a) Comment on the appropriateness of the approach taken and underlying assumptions used to determine the vulnerability schedules, noting the outcomes of the Francis review and the response of the stock assessment authors.

5. **Discard mortality estimates**
   a) Comment on the appropriateness of using the range of discard mortality estimates in the stock assessment model. Consideration should be given to the differential survival of fish caught and released in shallow areas such as Moreton Bay.

6. **Stock Assessment Model**
   a) Given the range of predicted exploitable biomass ratios derived from the stock assessment comment on the likelihood of the snapper stock being overfished (base assessment on the Fisheries Queensland definition of overfished as ‘stock levels less than 40% virgin biomass’).

1 Total Catch

1.1 Sensitivities considered in estimated catch history

A reliable time series of total catch is a fundamental component of fisheries stock assessment . The requirement for a substantial proportion of the catch history of Queensland snapper to be estimated introduces considerable uncertainty in the estimates of the unfished recruitment parameter, R_0, upon which the SALSA model (Campbell *et al.* 2009a, Equation 8) outputs are dependent.

There is a large degree of uncertainty in the total catch of Queensland snapper, particularly that part of the catch taken by the recreational sector. The standard errors quoted in the latest RFISH survey report (McInnes, 2008) suggest that the estimates of the number of snapper caught by the recreational sector for 1997, 1999, 2002 and 2005 are relatively precise. However, the survey estimates differ greatly between years. For instance, approximately 527 thousand fish were estimated to have been caught in 1999 and just 232 thousand in 2000. This would suggest that there is likely to be considerable uncertainty in the catch estimates calculated using Equation 2 of the original report (Campbell, *et al.* 2009a, p. 3).

The harvest estimates derived from the RFISH survey as shown in Figure 3.1 (Campbell *et al.* 2009a) would seem in some cases not to be in close agreement with those that would have been obtained from Equation 2. The recreational harvests for 2003, 2004, 2006 and 2007 were all estimated to be amongst the highest on record. Given survey estimated harvest of 209 tonnes in 2000, 267 tonnes in 2002 and a ballpark estimate of 150 – 200 tonnes for 2008 (Campbell et al, 2009b, p. 3), it is conceivable that the total recreational catch between 2003 and 2007 (from the RFISH surveys) might have been much less than estimated. It is likely in our view that the steep decline in relative biomass suggested by the biomass trajectories shown in Figure 3.12 (Campbell *et al.* 2009a) is driven to a large extent by the size of the catch estimates in recent years.

It might be reasonable to suggest that certain aspects of the reconstructed base case recreational catch history are likely to differ substantially from the actual recreational catch. However, based on the increase in vessel registrations illustrated in Figure 2.1 of the report, we argue that it would be difficult to suggest that there has not been a marked increase in the recreational snapper catch since the late 1980s.

The modified catch history assumed in scenario 13 (Campbell *et al.* 2009a, Fig. 3.12) goes some way to addressing the uncertainty in the catch history of the snapper stock. However, we disagree with the assumptions made about the average annual weight of snapper caught by the recreational sector. In light of this problem which is explained in greater detail in Section 1.4, we feel that the full extent of uncertainty in catch history is not addressed in the assessment.

1.2 Potential for recall bias from early estimates of recreational fish caught.

It would appear from the documentation that the survey methodology has been fairly consistent throughout RFISH and NRIFS and so any recall bias should also have been consistent throughout the surveys. If there were specific issues about methodology prior to 2001 then these should be clarified.

1.3 Potential for the catch of recreational fishers operating from charter vessels being included in the recreational catch estimates as well as in the charter logbook derived estimates

From our understanding of the various data sources, the correct way to estimate combined recreational and charter catch would be as follows:

RFISH survey estimate (includes Qld residents only, recreational and charter)
+ Charter log estimate for non-Qld residents only
+ Estimate of non-Qld recreational (available for 2001 NRIFS only)

An alternative method would be:

RFISH survey estimate, excluding charter
+ Charter log estimate
+ Estimate of non-Qld recreational (available for 2001 NRIFS only)

It is not clear whether either of the above methods were used in the snapper assessment. In any case, the size of the charter catch relative to the overall estimated recreational catch as shown in Figure 3.1 would suggest that any double counting is unlikely to be very influential on model outputs.

1.4 Conversion of numbers of fish to weight

This really depends on whether the estimated mean weights are reasonable. Ultimately the assessment requires that estimates be made of relative exploitable biomass which requires that the total biomass of fish removals be estimated.

Ideally, an accurate time series of total catch and length frequency data would be available upon which to base estimates of annual catch at age numbers. The absence of length frequency data from before 1994 meant that this option was not available and average weight was assumed to have been 0.9 kg for all years up to and including 2002 and 1.685 kg all subsequent years.

It would be expected that for a given vulnerability schedule, everything else being equal, the mean weight of fish caught would tend to be lower following periods of high fishing mortality due to lower proportions of older fish in the catch. Therefore the assumption of constant mean weight for a given vulnerability is likely to introduce biased estimates of the total biomass caught.

Model estimated weights calculated following the first reviewer's recommendation (Francis, 2009) produced results that were substantially higher than the values assumed (Campbell *et al.* 2009b, Figure 2). Ferrell and Sumpton (1997, p. 8) note that most pre 1960 historical reports of recreational snapper catch suggest average individual catch weights between 4 and 6 pounds (1.8 – 2.7 kg). It would seem likely that the assumption of average catch weights of 0.9 kg are too low for the early fishery. Model outputs suggest that the stock was lightly fished up until 1970. It would be expected in this case that the population would include a reasonable proportion of fish over 10 years of age. Furthermore the vulnerability schedule assumed would suggest that if older fish were present in the population they would also be expected to be well represented in the catch.

Exploitation rates are calculated as the ratio of estimated total catch weight in year y, $C(y)$, to the estimated total exploitable biomass in year y, $B(y)$, using Equation 11 (Campbell *et al.* 2009a, p.9).

The authors advise (Campbell et al, 2009a, p.2) that the number of fish estimated to have been caught by the recreational sector was obtained by extrapolating estimated recreational catch numbers from the RFISH surveys (McInnes, 2008) back in time according to a procedure outlined in Allen (2006, p.8). The total catch weight for recreationally caught fish was then obtained by multiplying the number estimated to have been caught by 0.9 kg for all years before 2003, and 1.685 kg for years 2003 to 2007. The total exploitable biomass, $B(y)$, is calculated using weights that depend on the estimated length distribution of the exploitable biomass as given by Equation 12.

If average fish weight in the exploitable population as suggested by the length distribution of the stock is different from the assumed average weight of the catch, then the number of fish removed from the population according to Equation 9 will be different to the number of fish estimated to have been caught by extrapolating the RFISH survey estimates. It would appear from Figure 2 (Campbell *et al.* 2009b) that model estimated average weights are typically between 2 kg and 2.7 kg.

The apparent anomaly between the number of fish estimated to have been caught by the recreational sector and the corresponding number removed by the model potentially has implications for the outputs of the model.

2.1 Catch Rates

2.1.1 Standardisation Process

The methods used to standardise catch rates were typical of normal practice. The models fitted were limited in some cases by the information available. The commercial logbooks did not include effort where no fish were caught. It is unclear how frequently zero catches occur in the commercial sector, but the authors suggested that this might have concealed some reduction in effective commercial catch rates (Michael O'Neill, pers. comm.).

Catch rates at offshore locations are likely to differ from inshore locations. Offshore factor data were only available to be included in recreational CPUE standardisation model 4 (Campbell *et al.* 2009a, Table 2.1) and ideally would have been included in all models.

The recreational CPUE index produced by combining the estimates from the four individual recreational and charter CPUE indexes (Campbell *et al.* 2009a, Table 2.1, models 2 -5) would appear to be a reasonable representation of these indexes.

2.1.2 Implications for using standardised CPUE in the snapper stock assessment method

It has been pointed out by Francis (2009) that the inclusion of CPUE data has only a minor influence on depletion level as suggested by the SALSA model outputs. This can be assessed by comparison of the biomass trajectories from scenarios 4 and 5 with the base case in Figure 3.12.

The standardised catch rate series plot in Figure 3.3 might be used to gauge the trend in relative abundance between 1988 and 2006 from the commercial data and between 1993 and 2006 from the recreational data. This might be of interest if the results from the SALSA model are considered to be unreliable. As we have said, we do not have issue with how CPUE was standardised, however, since the series suggest markedly different trends in abundance, at least one of them must give a misleading impression of the trend in the abundance of the snapper stocks in recent years. Findings presented in Ferrell & Sumpton (1997) would tend to indicate that CPUE might not be a reliable indicator of the status of the snapper stock. Ferrell & Sumpton (1997, p. 120) found that whilst the snapper population of eastern Australia should be regarded as a single genetic stock, few snapper released in tagging studies were recaptured more than 100 km from their release point. Regional differences in size and age composition were also observed (p. 126). These observations suggest limited adult mixing between local populations in which case local trends in CPUE might not be representative of the stock overall. The standardisation process is likely to address this effect to some extent, but is limited by the spatial resolution of the data which is too coarse to address the problem fully. We revisit this issue in our response to the question of hyperstability in the commercial CPUE index.

2.1.3 Underlying assumptions, variance of estimates, use of Gold Coast Charter industry data, compulsory charter logbook data, multispecies fishery.

The key assumption when considering a standardised CPUE series is that the index derived exhibits some expected relationship (usually proportional) with the abundance of the stock. For this to be plausible it needs to be, on average, that each individual in the vulnerable population is equally likely to be captured for a given unit of effort. Furthermore, all factors that influence catch rates for a given abundance need to be roughly equal in all time periods or otherwise able to be corrected in the standardisation procedure.

Standardising CPUE using only *kept* fish for the recreational catch (as opposed to total catch, kept and released) is not ideal. If the proportion of caught fish released changes over time, the kept fish index would be expected to be biased. Figures D.6 and D.7 suggest that standardised CPUE using *kept plus released* fish for the recreational sector is likely to have declined less in the same time. An increase in the proportion of released fish is likely to have occurred following the increase in MLS that occurred in December 2002. The RFISH data (McInnes, 2008) suggest that the proportion of snapper released has increased since the surveys began, from around 60% released up until 2000 and then closer to 80% thereafter. The decision to base indices of abundance on the catch rates of kept fish might have been made because the status of the fishery is based on the relative biomass of the vulnerable portion of the population. We agree with Francis (2009) that the proportion of spawning stock biomass would be a better basis for deciding stock status.

The authors provided additional plots of standardised CPUE indices for our review that included standard error estimates. The standard errors suggested that the sampling variance of these estimates were small compared with the uncertainty in annual relative abundance suggested by the different CPUE models as shown in Figure D.4 (Campbell *et al.* 2009a, p. 55). Small standard errors for the estimated year effects suggest that different catch rates were able to be reliably predicted between years in each of the CPUE standardisation models given the effects of the other factors.

As we have pointed out previously, evidence exists to suggest that the level of adult mixing between local populations is limited for this stock. In this case the relative abundance in the areas fished by the Gold Coast charter fishery might have followed a different trajectory over time compared with the fishery overall. The original report cites anecdotal information indicating overfishing is likely to be greater in the southern part of the fishery, particularly in the Gold Coast (Campbell, *et al.* 2009a, p. l). It might be expected therefore that declines in CPUE derived from Gold Coast charter data would overestimate the decline in the fishery as a whole. Walters (2003) suggests that failing to consider the abundance of unfished (or lightly fished) strata is liable to produce exaggerated trend indices.

The degree to which trends in abundance from the compulsory charter fishery data are likely to be representative of the stock overall depends on the degree to which the charter fleet spatially encompasses the stock in its operations. Figure D.4 suggests that there has been little trend in the standardised charter boat catch rates since data collection began in 1996.

Standardisation of catch rates is more difficult and may be less reliable in multispecies fisheries. In models 2 – 4 (Campbell *et al.* 2009a, Table 2.1) used to standardise the snapper catch rates, the authors have included an "other fish" term that would be expected to account for most of the variation in snapper catch rates due to targeting other species. We further note that snapper has actually been a primary target species for the recreational sector since the RFISH surveys commenced.

2.2.1 Appropriateness of decision not to use commercial CPUE in the stock assessment as the base case due to potential hyperstability.

Allen *et al.* (2006) expressed concern that the relationship between the abundance of the snapper stock and catch rates calculated for the commercial sector might be hyperstable. Hilborn & Walters (1992, p. 175) suggest that a hyperstable relationship between CPUE and abundance can be expected in almost any fishery where search is highly efficient allowing fishers to concentrate effort in regions where fish abundance remains high as the overall population declines. The authors expressed the view that a lack of information on fishing effort returning zero fish and lack of spatial resolution in the commercial logbook data might have obscured an effective reduction in snapper catch rates within the commercial sector (Michael O'Neill, pers. comm.). The potential for localised depletion and spatial heterogeneity in the relative abundance of the snapper stock means that a hyperstable relationship between the commercial CPUE and abundance is plausible. The first reviewer was also of the view that a hyperstable relationship between commercial CPUE and abundance was more likely than a hyperdepletion relationship (Francis, 2009, p.1). It would appear that the authors were of the opinion that the recreational catch rates were more likely to be representative of the relative abundance of the snapper stock and hence their decision to use the recreational CPUE index for the base case in the assessment model. We agree with the authors' choice in this regard. In any case, we note that the differences between the base case and scenario 4 (Figure 3.12) are slight and would seem unlikely to have led to a different status for the stock.

2.2.2 Likelihood for hyperstability in the commercial, recreational and charter CPUE data.

The ability for commercial fishers using modern gear to efficiently find and exploit areas of high relative abundance means it is quite likely that the overall relative vulnerable abundance of the snapper stock has declined more than would be suggested by the standardised commercial catch rates. The extent to which commercial fishers can travel greater distances to maintain catch rates is limited by their need to operate at a profit on average. Total commercial catch has been maintained in recent years suggesting that profitability has not constrained catch to a greater extent in this period than was previously the case.

Charter and recreational catch rates can also be subject to hyperstability. Ferrell & Sumpton (2007) suggest that the introduction of GPS technology has allowed a higher proportion of recreational fishers to fish offshore. Charter and recreational fishers are not constrained by economics to the same extent as commercial operators. Recreational and charter fishers are likely to be more restricted by time. Recreational fishers are also likely to be restricted by vessel size and gear.

Overall we suggest that commercial catch rates are likely to be the most hyperstable, charter catch rates next most likely and recreational catch data least likely.

3 Length-Age Frequency

3.1.1 Methods used to estimate total mortality

The available length and age frequency data were considered in the assessment in two ways. Length frequency data were converted to age frequency data using separate age length keys for different regions and different years. The need to use separate keys for separate regions to prevent biases, that might otherwise be introduced due to regional and temporal differences in growth rates, is stressed by Ricker (1975, p. 205) .

Most simply, the age frequency data from the individual years sampled were used for cross sectional catch curve analysis. The estimates of total mortality (Z) from this analysis are displayed in Figure 3.7.

Secondly, the length and age frequency data were key inputs in the SALSA model likelihood functions that informed the estimation of the unfished recruitment parameter, R_0 upon which all SALSA model output was dependent. Exploitation rates each year are dependent upon the estimated value of this recruitment parameter and the estimated catch history for all years up to and including the year of interest through equation (11) on page 9 of the report.

We would add that the assumed values of natural mortality are important to the interpretation of total mortality and feel that the values assumed in the assessment are well founded and reasonable.

3.1.2 Underlying assumptions

Estimation of total mortality through analysis of catch curves relies upon a number of assumptions. As mentioned in the report, recruitment is assumed constant (for cross sectional catch curve analysis) as well as natural mortality. Strictly speaking catch curve analysis assumes that total mortality is constant across age classes and years. A closed population must also be assumed, as well as constant vulnerability across age classes and years. Other assumptions include unbiased sampling of lengths and accurate ageing.

In practice, the assumption of constant recruitment is always violated and this is really an inherent weakness in cross sectional catch curve analysis as a method of estimating mortality that cannot be avoided without sufficient annual sampling of length frequency data to allow at least informal assessment of year class strength. The authors express the view that large variation in recruitment is not suspected for the Queensland snapper stock in contrast with snapper stocks elsewhere (Campbell *et al.* 2009a, p.34) and we suggest that the authors are in a better position than ourselves to make this judgement. Constant natural mortality is almost always assumed, not only in catch curve analysis, but in fishery stock assessment more generally, at least amongst recruited age classes. Similarly the assumption of a closed population is standard practice. The assumption of constant fishing mortality is also standard, although this is more reasonable in some situations than in others. The assumptions of constant vulnerability and unbiased sampling are questionable in this case. For instance the authors estimate that vulnerability decreases between age classes 4 and 9. If this is accurate then conventional, cross sectional catch curve analysis would be expected to overestimate total mortality because the proportion of older age classes in the sample will be lower, on average, than would be expected given constant vulnerability across age classes. The increase in the minimum legal size from 30 cm to 35 cm in 2003 also suggests that the vulnerability of young fish was not constant leading up to the collection of the 2007 length frequency data. The effect of this change would also tend to be an overestimation of fishing mortality due to older age classes experiencing some additional vulnerability to fishing mortality beyond that assumed. Sampling from pooled length frequency data each year, weighting by region and sector, is a reasonable method for estimating the proportion of catch at length. Given a suspicion of limited adult mixing within the stock, it is less clear if this pooling method is appropriate for the estimation of total mortality rates for the fishery as a whole. Reliable estimation of total mortality would require that fishing effort by region be proportional to snapper abundance and this would seem unlikely. It should be pointed out that the authors do not place a great deal of emphasis on the results of the catch curve analysis.

Accurate estimation of exploitation rates within the SALSA model depends upon reliable catch history data and a reliable estimate of unfished recruitment and is limited by the deterministic nature of recruitment within the SALSA model. Model estimated exploitation rates are not presented in the report.

3.1.3 Variances around mortality estimates

The original report illustrates 95 per cent confidence intervals around catch curve derived estimates of total mortality (Campbell *et al.* 2009a, Figure 3.7). The confidence intervals are relatively 'tight' which suggests that the proportion of age classes observed in years where catch at age are estimated are in relatively good agreement with the linear regression equation used to estimate total mortality. The reliability of the confidence intervals are subject to the same strong assumptions as the point estimates and so should be considered with caution.

3.1.4 Implications of using length-age frequency data in the snapper stock assessment

Given the shortage of data available to inform the assessment of the snapper stock it would seem desirable to make use of all data possible unless there was good reason for some data to be excluded. Length and age data are key inputs for most age structured models and, when representative of the stock, have the advantage of providing an estimate of total mortality that does not depend on knowledge of stock size. The authors seem to feel that sampling programs used to collect length frequency data in 1994-95 are inconsistent with those used in 2007. This view was expressed by the first author (Alex Campbell, pers. comm.) and is borne out by the statement:

> The new analysis mainly serves to highlight issues that have already been raised in the assessment: that 1990s total mortality rates (M+F) were possibly unsustainably very high, that pre-2000 vulnerabilities were strongly domed, that sampling was biased in either 1994/1995 or 2006/97 age length data and/or that recreational catch was over estimated by the telephone/diary survey technique.
>
> (Campbell *et al.* 2009b, p. 4)

The plots illustrating goodness of fit to length data presented in the authors response to the Francis review (Campbell *et al.* 2009b, Figure 17, Figure 20, Figure 22, Figure 25 and Figure 32) suggests the model does not fit the 1994 and 1995 length frequency data well. This would suggest that either these data are a poor representation of the stock overall or that the models do not reliably predict catch at length over time.

On this issue we are in agreement with comments made by Francis (2009, p.2). The general poor fit of the model to the 1994-95 length data means that considering these data in the base case with the assumed vulnerability schedules was not ideal. We would have advocated using scenario 7 (Campbell et al, 2009a, Table 3.1) as the base case, with possibly the current base case as a sensitivity. An indication of the effect of this might be gained by comparing, scenario 4 with the base case and also comparing scenario 7 with scenario 8 (Campbell *et al.* 2009a, Figure 3.12).

The effect of assuming the commercial CPUE in place of the recreational CPUE under the base case as suggested by comparing the biomass trajectory of scenario 4 with the base case is relatively minor. We would argue that given the very different trends in relative abundance suggested by the CPUE indices, a greater difference would have been expected.

The effect of assuming the commercial CPUE in place of the recreational CPUE when the 1994/95 data are excluded can be assessed by comparing relative biomass trajectories 7 and 8 (Figure 3.12). In this case, the differences in these estimated trajectories are considerable and more in line with what would be expected.

We add that selection of base case scenarios is a subjective exercise and often involves input from a group of scientists and industry experts larger than the group that ultimately conducts the stock assessment. We also point out that the status that would be suggested by our recommended base case is the same as that suggested by the base case used in the original assessment (i.e. below B_{40}).

An alternative to removing the 1994/95 length frequency data would be to reduce the weighting on these data in parameter estimation. We are of the view that overall the likelihood function used in this assessment given by the sum of the right hand sides of Equation 17, 18 and 19 (Campbell *et al.* 2009a, p.10) possibly assigns too much weight to the length and age data (which is largely dependent on the length data) relative to the CPUE. We agree with the comments of Francis (2009, Sections 1.5, 1.6) on the weighting of data.

3.2 Implications of length-age frequencies and estimated mortalities from the 1994/95 and 2007 data for status.

We presume the length-age frequencies and estimated mortalities from 1994 and 1995 are only important for current status to the extent that they influence model outputs. Similarly the 2007 length-age frequency data and mortality estimates are presumed to not be directly relevant to the determination of status. Reference points for fishing mortality in Queensland fisheries have now been defined by the Department of Employment, Economic Development and Innovation (DEEDI, 2010, Table 1). Using natural mortality, $M = 0.19$ year^{-1} as assumed in the base case scenario in the report, the threshold reference point would be fishing mortality $F = 0.19$ year^{-1}. Therefore under the $M = 0.19$ year^{-1} assumption, threshold fishing mortality would be suggested by a total mortality rate, $Z = 0.38$ year^{-1}. Estimated total mortality rates for the Queensland snapper stock based on age and length frequency data are displayed graphically in the report (Campbell *et al.* 2009a, Figures 3.7, 3.8).

It is worthwhile noting that the length frequency data displayed by sector and region in Figure C.3 and Figure C.4 (Campbell, 2009a) suggest that exploitation rates might have been higher in some regions than others.

4 Vulnerability Schedules

We consider it unlikely that reliable estimates for a full vulnerability schedule could have been obtained within the model due to the paucity of data. This being the case, the decision to estimate the schedule outside of the model seems sensible.

The vulnerability schedules used in the assessment were estimated using the limited length frequency data. A lack of information for older fish meant that the vulnerability for fish older than nine years was extrapolated. We believe this to have been a reasonable assumption *a priori*. Vulnerability schedules were later debated by stakeholders, but consensus was not reached.

It would appear that the authors felt that the available data could support the estimation of only a single parameter. This is understandable given the data, but permits only a model that lacks any flexibility to accommodate contradictory data.

In response to comments made by Dr Francis (2009) about vulnerability, the authors write:

> A key issue in the assessment is the question of whether the increase of older/larger fish in the 2006/07 length-age frequency data is the result of a real increase in abundance (possibly due to prior MLS changes kicking in) or merely a change in targeting.

(Campbell *et al.* 2009b, p.8)

With R_0 the only parameter estimated and the vulnerability of older fish assumed the same in 1994/95 as in 2007, the model has to assume that there are fewer older fish in 2007 than in 1994/95, due to the estimated exploitation rates in the intervening years (moderated by the change in the vulnerability of young fish in 2003) and that there was no change in targeting between 1995 and 2007 other than that which is consistent with the assumed decrease in vulnerability of young fish. That is, the SALSA model scenario runs are unable to attain a state consistent with either explanation offered by the authors for a higher proportion of older fish observed in the catch in 2007.

The paucity and contradictory nature of the data are the key problem for this assessment. It might be argued that a single vulnerability parameter for fish older than 9 years for all years prior to 2003 and another for all fish older than 9 years for all years beginning in 2003 should be included. However, the paucity of data available to determine such parameters would still lead to uncertainty in their reliability. Furthermore, uncertainty about the vulnerability of undersized fish before and after changes in MLS would still be an issue.

5 Discard mortality

An attempt to consider the effect of discard mortality is commendable. The rates of discard mortality suggested for snapper would indicate that, for some stocks at least, losses due to death of discarded fish are likely to be important. It would have been preferable as Francis (2009, p.2) pointed out if an estimate for rates of discard mortality specific to the fishery were available, but this would appear not to be the case. The findings of the Stewart (2008) study would seem to be a reasonable substitute for fishery specific rates. Ferrell & Sumpton (1997, p. 10) suggest that the major snapper fishing grounds (New South Wales and Queensland combined) are at depths between 20 m and 100 m. Catch rates of undersized fish are likely to be higher in shallower areas where discard mortality rates would be expected to be lower. In any case, the more important issue for the assessment is the total numbers of fish lost due to discard mortality, which is the product of the overall discard mortality rate and the total number of fish released. Estimates of the number of fish released by the recreational sector from the RFISH surveys would suggest that total model estimated discard mortality is possibly an underestimate.

Stewart (2008) suggested short term mortality rates for snapper caught in depths less than 30 m of around 2 per cent, around 39 per cent for depths 30 m - 45 m and around 55 per cent for depths between 45 m and 59 m. Walters (2006) considered 30 per cent to be a realistic estimate of overall discard mortality.

For the 2009 snapper assessment the assumed discard mortality rate seems to have little effect on the predicted biomass ratio. Relative biomass trajectories for scenarios 9 and 10 that test sensitivity to discard mortality are indistinguishable from the base case (Campbell *et al.* 2009a, Fig. 3.12).

6 Stock Assessment Model

A number of issues highlighted in this document suggest that the degree of uncertainty in the 2009 stock assessment of Queensland snapper is likely to be substantial. We cannot know whether the net result of these would increase or reduce the predicted depletion. The sensitivities explored by the authors (Campbell *et al.* 2009a, 2009b) tend to provide consistent conclusions around relative biomass, with most estimates for 2007 below the B_{40} reference point.

From the assessment and scientific review it is plausible that, for snapper, exploitable biomass in 2007, B_{2007}, was less than 40 percent of virgin exploitable biomass.

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